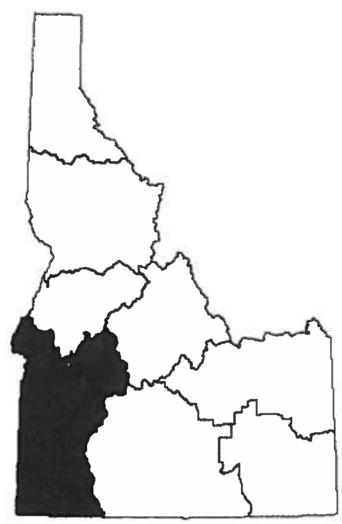
FISHERY MANAGEMENT INVESTIGATIONS

IDAHO DEPARTMENT OF FISH AND GAME

FISHERIES MANAGEMENT ANNUAL REPORT

Virgil Moore, Director



SOUTHWEST REGION

2010

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2010 Southwest Region (Nampa) Fisheries Management Report

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2010 Southwest Region (Nampa) Annual Fisheries Management Report

Lowland Lake Surveys

Brownlee, C.J. Strike, And (Lake) Lowell Reservoirs - Assessments Of Larval Fish Production

ABSTRACT

Regional staff conducted larval trawl surveys in Brownlee, C.J. Strike, and (Lake) Lowell reservoirs during 2010 to gain a better understanding of the reproductive patterns of recreationally-important warm water fish, primarily black and white crappie *Pomoxis nigromaculatus* and *P. annularis*, factors that may affect reproductive success, and to monitor trends in the distribution and abundance of larval fish over time. Larval fish density was monitored by horizontally trawling a neuston net at six to 11 sites within each reservoir at times when larval fish are most abundant and susceptible to this gear (mid-June to mid-July). In addition, we transferred approximately 3,000 pre-spawn adult-sized crappie from C.J. Strike Reservoir to Lake Lowell to bolster production potential.

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INTRODUCTION

Fisheries for black and white crappie and, bluegill *Lepomis macrochirus*, and yellow perch *Perca flavescens* are popular among anglers in southwest Idaho, when abundant. However, reproductive success or earlier survival is often variable creating strong and weak year classes and eventually inconsistent fisheries. Fisheries personnel are interested in quantifying year-class strength before fish become vulnerable to anglers, so that anglers may be informed of potential fisheries quality. Monitoring larval fish densities with neuston nets is one way to provide information on reproductive success and eventual year-class strength as long as strength isn't affected substantially by population bottlenecks later in life (e.g. survival during winter). At a minimum, determination of years with low larval production will identify potentially poor fishing years two to three years later. Monitoring of year-class strength in Brownlee and C.J. Strike reservoirs has been conducted by Idaho Department of Fish and Game (IDFG) fisheries research personnel since 2005 as part of a statewide research project. Regional staff plans to continue these efforts utilizing the same sites and time periods when larval fish were consistently most abundant (mid-June through mid-July). Also, regional personnel have been monitoring larval fish production in Lake Lowell since 2006.

OBJECTIVES

1. Assess reproductive success of recreationally important warm-water fishes.

2. Translocate adult, pre-spawn crappie to bolster production potential in Lake Lowell and monitor whether efforts led to increased production.

METHODS

Horizontal surface trawls were used to index the abundance of larval fish in Brownlee, C.J. Strike, and Lake Lowell reservoirs. Trawls were made with a 1-m high x 2-m wide x 4-m long neuston net, with a 1.3 mm mesh size. Trawls were made at 6 to 11 sites spread throughout each of the reservoirs (Figures 1, 2, and 3). Trawls were begun at dusk and all sites were completed within three or four hours. The net was fit with a flow meter to estimate the volume of water sampled. Trawl duration was 5 min and an average of 559 m^3/trawl was sampled. Trawls were made on an approximately bi-weekly basis beginning June 22 and ending July 22, 2010, which overlapped peaks of crappie production in previous years. Specimens were stored in 10% formalin and viewed under a dissecting microscope. Sampled fish were identified to species, when possible, and measured for length, unless the total number of larval fish exceeded 50 individuals. For large samples, we randomly selected 50 individuals, identified and measured those, and counted the remainder.

During April and May 2010, we captured black and white crappie in C.J. Strike Reservoir and transferred them to Lake Lowell to bolster depressed adult populations. Pre-spawn adult-sized fish were captured in C.J. Strike Reservoir using trap nets and electrofishing gear. Additionally, volunteer anglers caught and donated live crappie for transfer. Fish were transferred to live cars and held until sufficient numbers were captured to fill a transport truck or trailer. Once loaded, fish were supplied with supplemental oxygen at the rate of 2 L/min. All translocations occurred on the day of capture.

RESULTS

Brownlee Reservoir

A total of 32 trawls were conducted on three sampling dates. Four species or groups of species were sampled including crappie spp., channel catfish *Ictalurus punctatus*, smallmouth bass *Micropterus dolomieu*, and cyprinids. Crappie were by far the most abundant group sampled comprising 90% of the fish collected on June 28th, 91% on July 7th, and 98% on July 15th. Density of crappie increased at our later sampling dates with average density equaling 29 crappie/100 m^3 on June 28th, 162 crappie/100 m^3 on July 7th, and 264 crappie/100 m^3 on July 15th. The highest density of 1,493 crappie/100 m^3 occurred at site 10 (near Woodhead Park) on July 15th. On July 15, 2010, high larval abundances were well distributed throughout the middle and lower reservoir with densities exceeding 127 larval crappie /100 m^3 from site 4 (8 km upstream of Sturgill Creek) downstream to site 11 (Brownlee Dam). Larval densities in Brownlee Reservoir during 2010 were the highest measured since IDFG began monitoring during 2005. Mean (264 crappie/100 m^3) and maximum densities (1,493 crappie/100 m^3) for 2010 exceeded that of all other years (Figure 4).

C.J. Strike Reservoir

A total of 30 trawls were conducted on 3 sampling dates. Three species or groups of species were collected including crappie spp., smallmouth bass, and cyprinids. Crappies were by far the most abundant group sampled comprised 99% of the collected fish on June 23th, 90% on July 6th, and 81% on July 14th. Mean density of crappie averaged among all sites was highest at our initial sampling date (7.4 crappie/100 m^3 on June 23th) and decreased successively on the next two occasions (1.8 crappie/100 m^3 on July 6th, and 0.2 crappie/100 m^3 on July 14th). The highest density of 20 crappie/100 m^3 occurred at site 4 (west side of Bruneau Pool) on June 23th. Mean density of larval crappie during the week of peak abundance was much lower than measured during 2008 and 2009, but was similar to 2005 and 2006 (Figure 5). Mean larval density during 2008 (36 crappie/ 100 m^3) was near five-fold higher than measured during 2010. Similarly, maximum larval density during 2008 (240 crappie/100 m^3) was 12-fold higher than measured during 2010.

Lake Lowell

Approximately 3,000 pre-spawn adult crappie were transferred to Lake Lowell during late April and May 2010. We caught a total of 278 larval fish with the neuston net during 18 separate tows (six fixed sites on three sampling dates). Fish species sampled included bluegill, black crappie, channel catfish, largemouth bass, white crappie, yellow perch, cyprinids, and some that we were unable to identify. Bluegill were by far the most numerous species (53%) captured, followed by crappie (16%), and unknown (16%). Most of the larval fish (67%) were caught at the two sites located in the upper reservoir (sites 5, & 6). Mean density of crappie averaged among all sites was highest at our middle sampling date (1.1 crappie/100 m^3 on July 8th). The highest density, 4.7 crappie/100 m^3, occurred at site 6 (east side of reservoir near the inlet) on July 8th. Mean density of larval bluegill averaged among all sites was highest at our last sampling date (5.8 bluegill/100 m^3 on July 22th). The highest density of 25.2 bluegill/100 m^3 occurred at site 6 (east side of reservoir near the inlet) on July 22th. Mean and maximum larval crappie densities measured during 2010 were at or near the lowest levels recorded since monitoring began during 2006 (Figure 6). Similarly, mean and maximum bluegill densities were very low and only exceeded levels measured during 2008. Maximum larval bluegill densities measured during 2010 were approximately one-quarter of densities measured during 2006 (Figure 7).

DISCUSSION

Production of larval crappie in large reservoirs in the Southwest Region shows high spatial and temporal variation and was asynchronous among reservoirs during 2010. For instance larval production in Brownlee Reservoir was higher than documented in the previous five years. Production during 2010 was different spatially from another high production year, 2006, during which most production originated in the upper reservoir. During 2010, spawning events in the middle and lower reservoir produced nearly all larvae, whether spatial differences affect eventual year class strength is unknown. Unlike Brownlee Reservoir, larval production in C.J. Strike Reservoir was relatively low compared to previous years, but consistent among sites located throughout the reservoir. Larval production in C.J. Strike during 2010 on average was similar to 2006, a year class that created excellent fisheries; however, 2006 and 2010 densities were much lower than 2008 and 2009. Lake Lowell is a stark contrast to these systems. Larval production in all years has been the lowest of the three systems. For instance, from 2006-2010 average larval densities in Lake Lowell are about 15% of densities in C.J. Strike and about 2% of densities in Brownlee Reservoir. This disparity was less only twice: (1) during 2007, when larval production was low in all systems, and (2) during 2009, when adult crappie were translocated to Lake Lowell and supposed to boost larval production, though seemingly not recruitment. The translocation of 3,000 adult crappie during 2010 failed to increase larval crappie density in Lake Lowell, despite high reservoir levels, unlike 2009 efforts (Butts et al. 2011). Furthermore, neither the 2009 or 2010 translocation efforts resulted in any substantial increase in the abundance of advanced age-0 crappie or age-1 crappie, which would have been caught during netting operations for other projects.

MANAGEMENT RECOMMENDATIONS

1. Monitor age structure in the harvest for fisheries on Brownlee and C.J. Strike reservoirs.

2. Attempt to capture younger age classes with otter trawls to document relative abundance of advanced age-0 and age-1 crappie

3. Seek methods to increase larval production and recruitment of panfish in Lake Lowell.

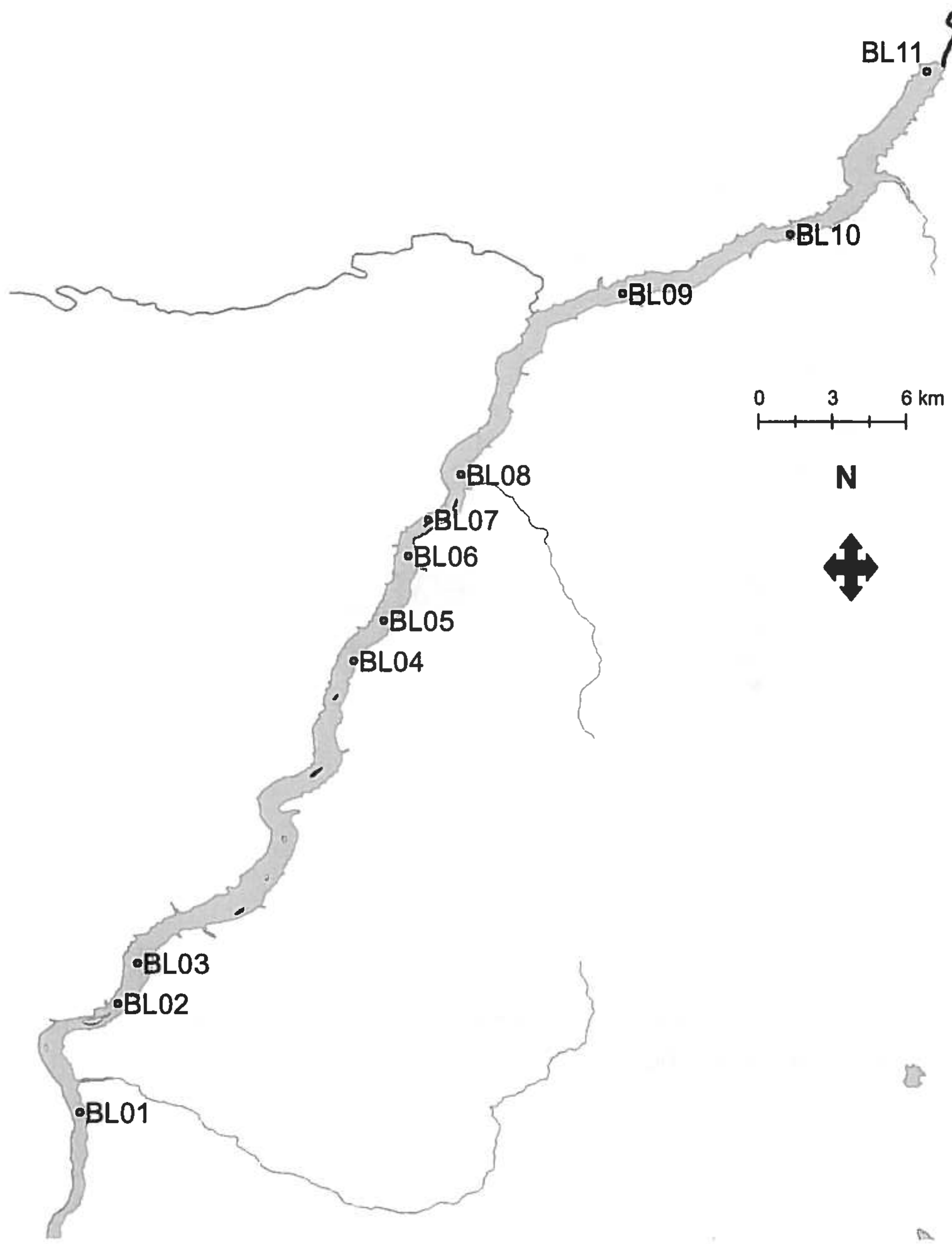

Figure 1. Location of eleven trawl sites used to index the abundance of larval fish in Brownlee Reservoir from 2005 - 2010.

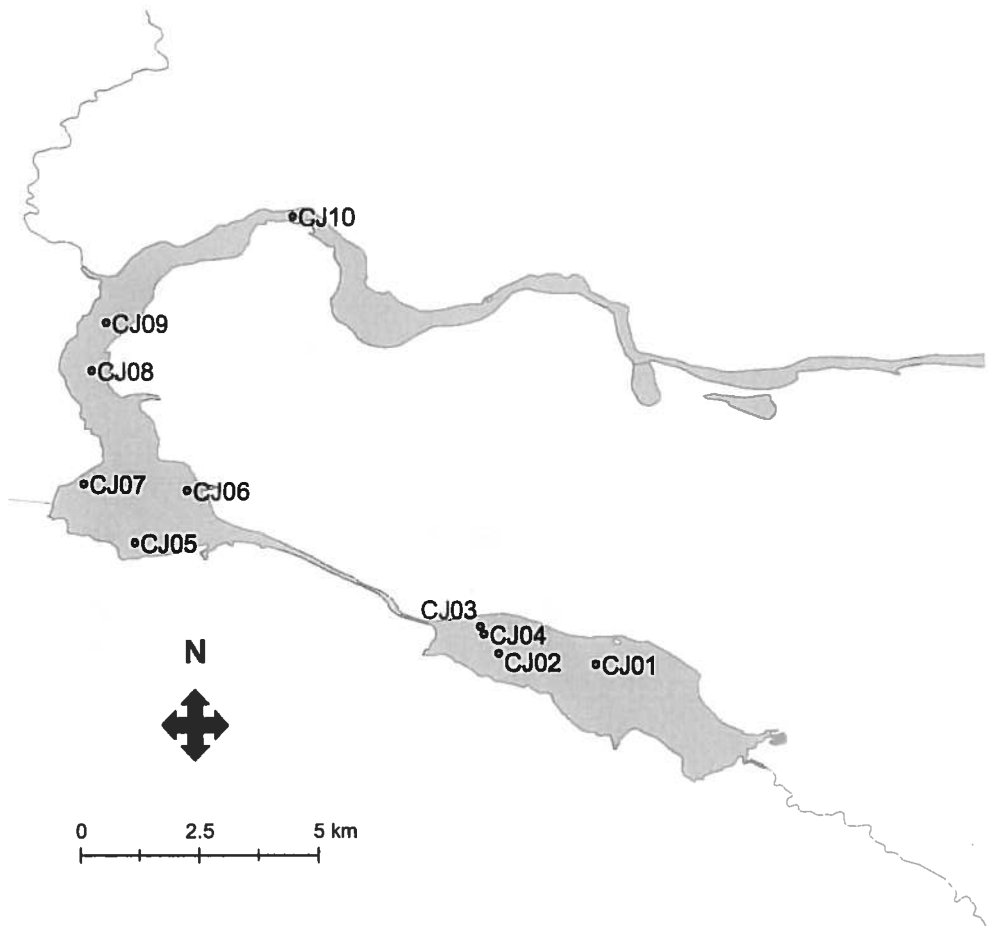

Figure 2. Location of ten trawl sites used to index the abundance of larval fish in C.J. Strike Reservoir from 2005 - 2010.

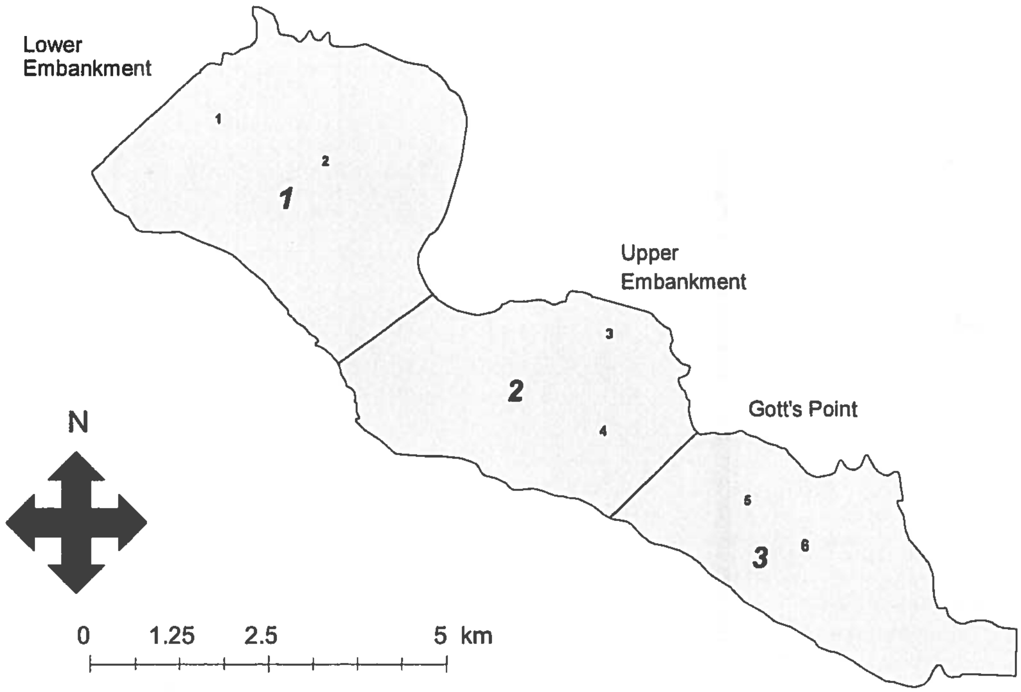

Figure 3. Location of six trawl sites used to index the abundance of larval fish in Lake Lowell from 2006 - 2010.

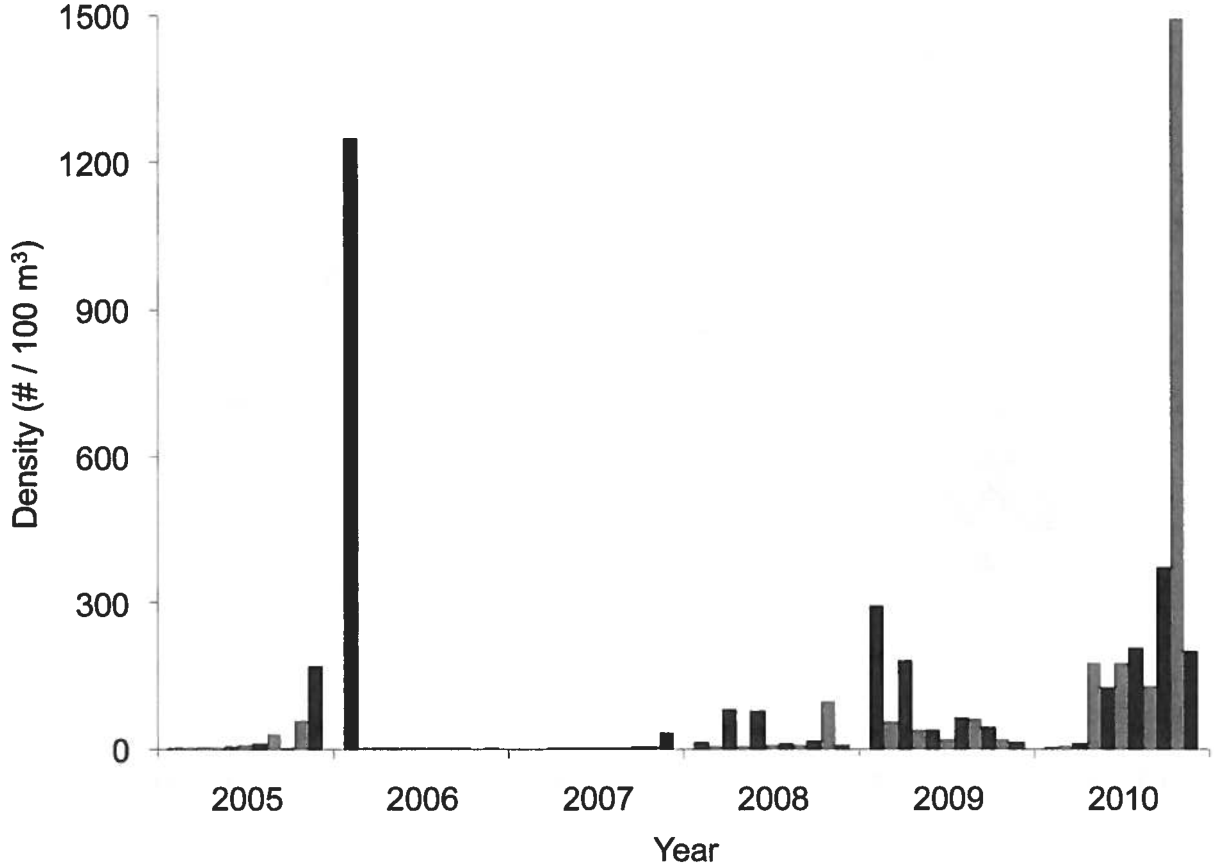

Figure 4. Densities of larval crappie (#/100 m^3) in Brownlee Reservoir during 2005 through 2010. Bars within each year represent eleven individual sites. Site 1 (upstream) through site 11 (near Brownlee Dam) are displayed from left to right within X-axis categories. Displayed densities reflect the annual peak in abundance sampled between June 15 and July 15.

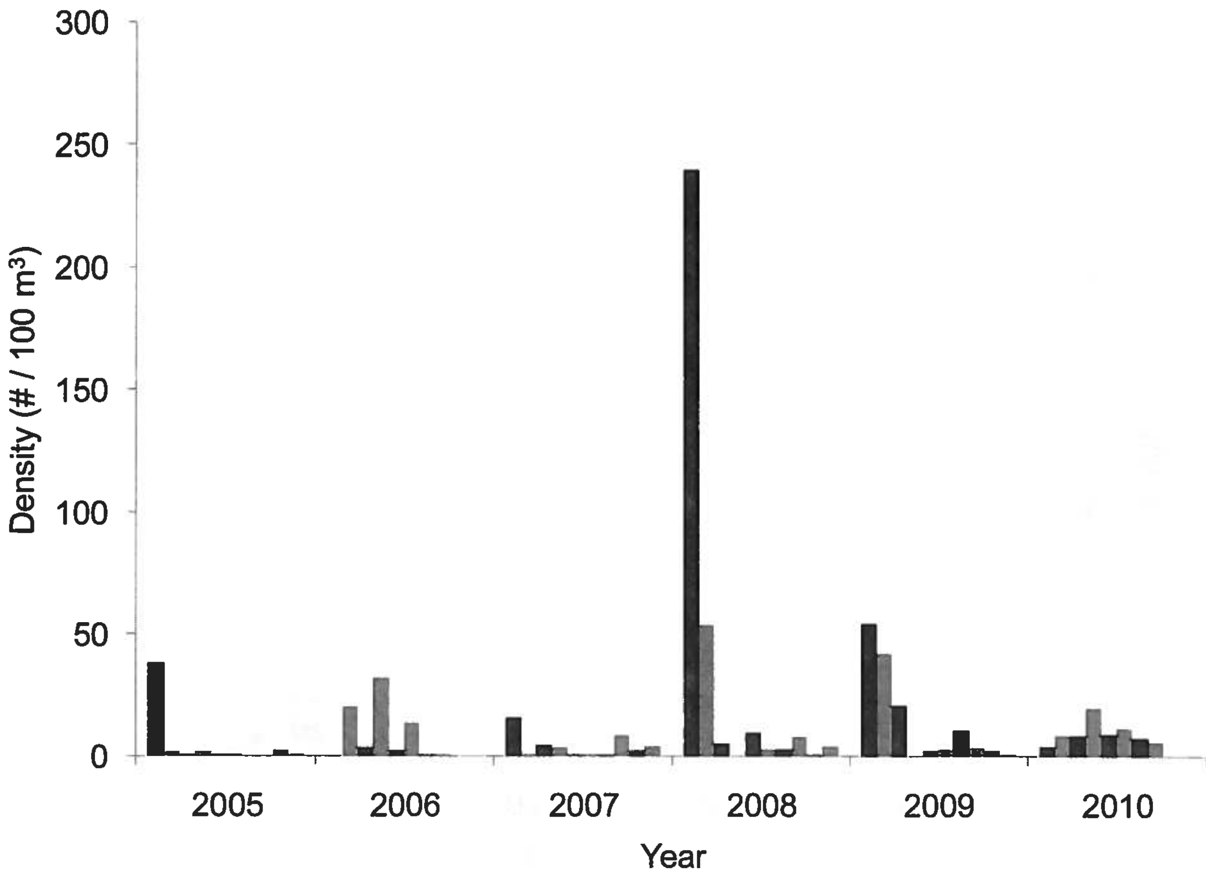

Figure 5. Densities of larval crappie (#/100 m^3) measured in C.J. Strike Reservoir during 2005 through 2010. Bars within each year represent ten individual sites. Sites 1 through 10 are displayed from left to right within X-axis categories. Displayed densities reflect the annual peak in abundance sampled between June 15 and July 15.

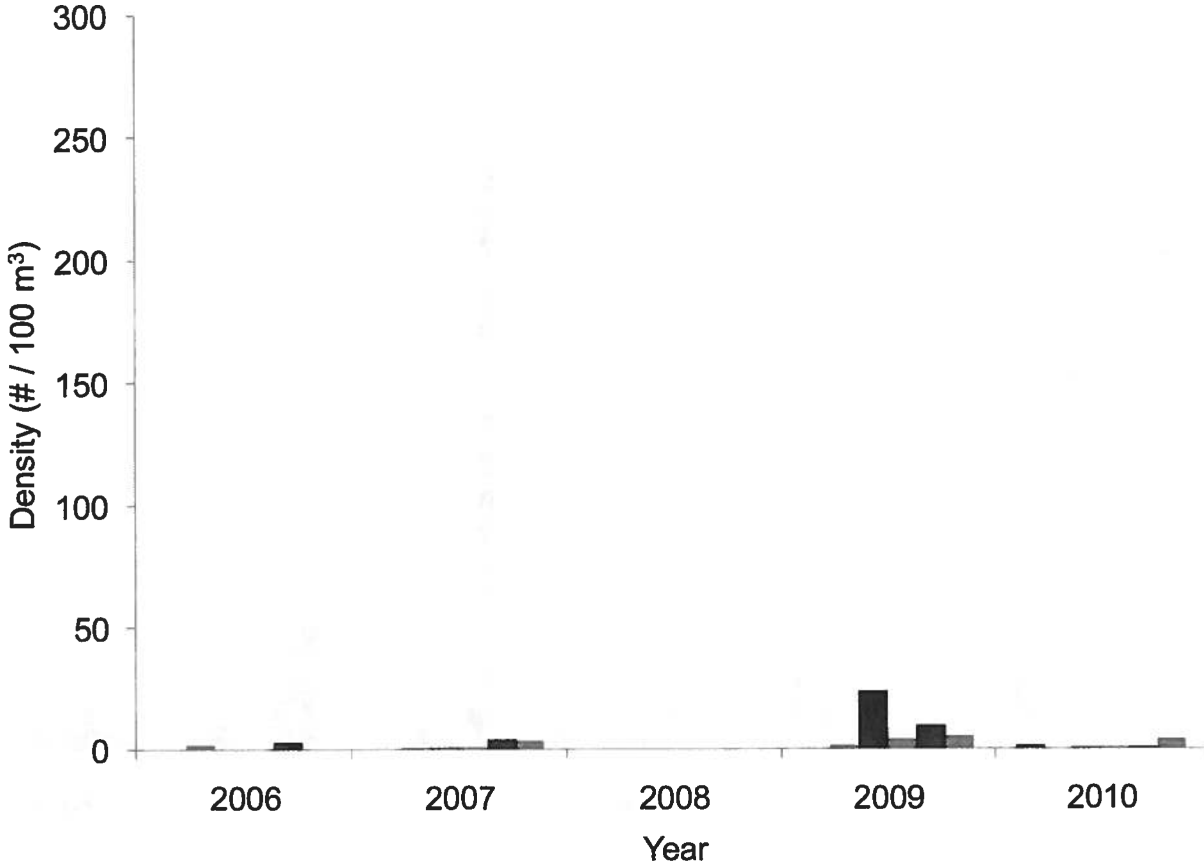

Figure 6. Densities of larval crappie (#/100 m^3) measured in Lake Lowell during 2006 through 2010. Bars within each year represent six individual sites. Site 1 (western end) through site 6 (eastern end) are displayed from left to right within X-axis categories. Displayed densities reflect the annual peak in abundance sampled between June 15 and July 15.

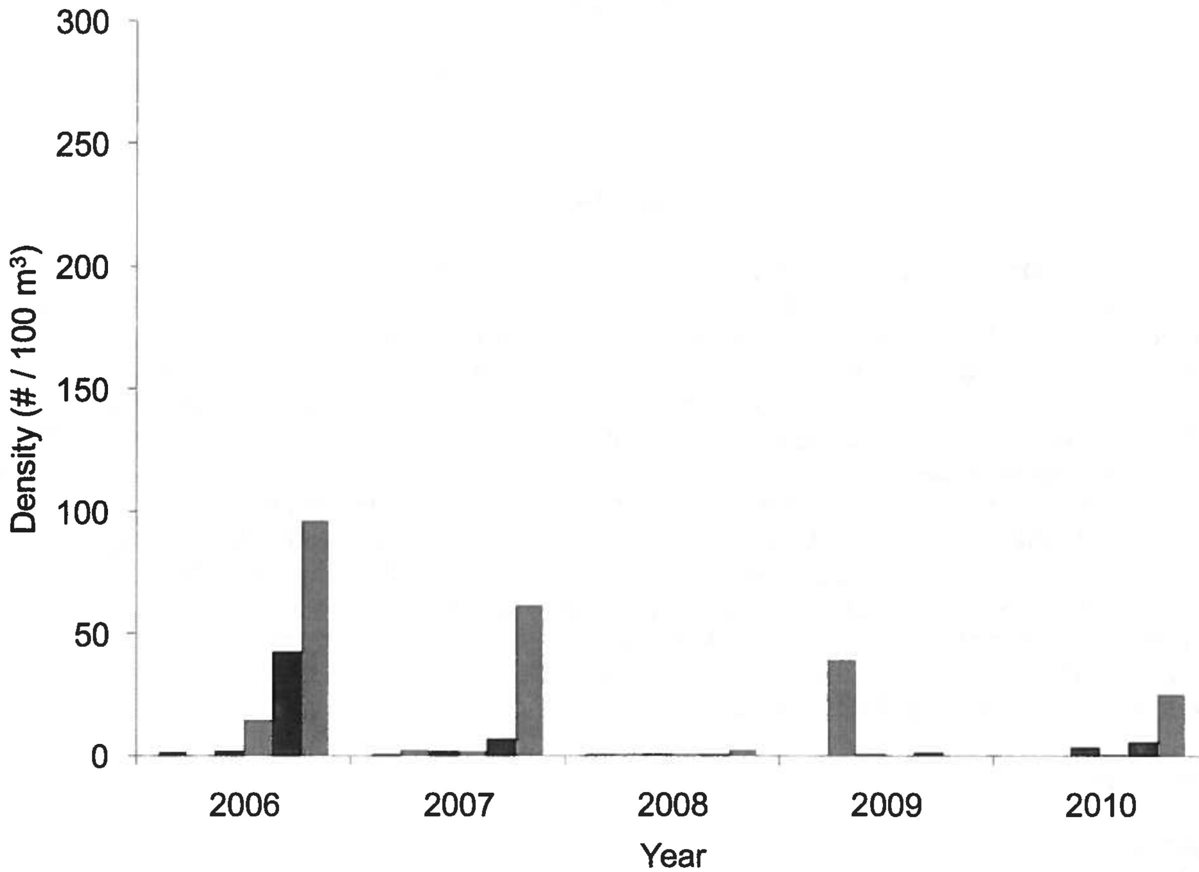

Figure 7. Densities of larval bluegill (#/100 m^3) measured in Lake Lowell during 2006 through 2010. Bars within each year represent six individual sites. Site 1 (western end) through site 6 (eastern end) are displayed from left to right within X-axis categories. Displayed densities reflect the annual peak in abundance sampled between June 15 and July 15.

Lowland Lake Surveys

Bull Trout and Martin Lakes

ABSTRACT

Fish populations in Bull Trout and Martin lakes were sampled with paired standard IDFG lowland lake gill nets during June 27 - 28, 2010. A total of 146 fish were captured in Bull Trout Lake (143 brook trout *Salvelinus fontinalis* and 3 kokanee *Oncorhynchus nerka*) while 54 fish were caught in Martin Lake (51 brook trout and 3 hatchery rainbow trout *O. mykiss*). Brook trout catch per unit effort (CPUE) in Bull Trout Lake was 48 fish while kokanee CPUE was 1 fish. In Martin Lake, brook trout CPUE was 26 fish and hatchery rainbow trout CPUE was 1.5 fish. No other fish species were captured in the lakes. Brook trout have dominated historic standard gill net surveys at Bull Trout Lake and continue to do so in 2010. The mean length of these fish has remained relatively constant throughout this period as well, with brook trout averaging 199 ± 8 mm (mean ± 90% CI) in 1991, 190 ± 6 mm in 1994, and 192 ± 5 mm in 2010. No brook trout ≥300 mm have been captured in any of the previous standard gill net surveys. Brook trout reduction efforts such as the introduction piscivorous fish such as sterile tiger muskellunge *Esox lucius X masquinongy* or intensive gill net sets are low-cost methods that should improve the quality of the fishery at Bull Trout Lake.

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INTRODUCTION

Bull Trout Lake and Martin Lake are located in the Payette River drainage in the Lowman Ranger District, Boise National Forest. Both lakes are surrounded by developed campgrounds and are popular areas for anglers and families. Bull Trout Lake is a 30 ha (surface area) lake at 2,119 m elevation and Martin Lake is a 2 ha lake at 2107 m elevation. The lakes are stocked with catchable-sized hatchery rainbow trout by Idaho Department of Fish and Game (IDFG) periodically through the summer, beginning in June and ending in mid-August. Bull Trout and Martin lakes are annually stocked with approximately 5,500 and 3,500 catchable-sized rainbow trout, respectively. In addition to hatchery rainbow trout, both lakes have abundant naturally-reproducing populations of brook trout *Salvelinus fontinalis*. Historical stocking of Bull Trout Lake included Atlantic salmon *Salmo salar*, Chinook salmon *Oncorhynchus tshawytscha*, Arctic grayling *Thymallus arcticus*, and bull trout *Salvelinus confluentus*.

Bull Trout Lake campground is considered high use by the United States Forest Service (USFS), which operate and maintain the campground. In 2010, USFS recreation personnel estimated that the campground received 10,330 visitors, with fishing being the main attraction to the area (D. Erwin, U.S. Forest Service, personal communication). In 1998, a creel survey was conducted on both lakes by IDFG in cooperation with Lowman Ranger District personnel. It was estimated that anglers spent 3,020 h fishing Bull Trout Lake and 1,276 h at Martin Lake (Allen et al. 2001). Return rates of catchable-sized hatchery trout were estimated to be 50% for Bull Trout Lake and 87% for Martin Lake in 1998. The average catch rates were estimated to be 0.3 rainbow trout / h and 0.8 brook trout / h in Bull Trout Lake. In Martin Lake, catch rates were 1 rainbow trout / h and 0.6 brook trout / h.

IDFG has not surveyed the fish populations in either lake since fall 1994; and therefore we planned gill net surveys in 2010. We are particularly interested in hatchery rainbow trout survival and carry over. The 2010 survey was conducted prior to either lake receiving catchable plants for the season.

METHODS

Fish populations in Bull Trout and Martin lakes were sampled with paired standard IDFG lowland lake gill nets during June 27 - 28, 2010. Paired gill net sets included floating and sinking monofilament nets, 46 m x 2 m, with six panels composed of 19, 25, 32, 38, 51, and 64-mm bar mesh. One floating and one sinking net, fished for one night, equaled one unit of gill net effort.

Captured fish were identified to species, measured for total length (± 1 mm), and weighed (± 1g for fish under 5,000 g or ± 10 g for fish greater than 5,000 g) with a digital scale. Relative weight, W_r, was calculated as an index of general body condition for selected species, where a value of 100 is considered average (Anderson and Neumann 1996). Values greater than 100 describe robust body condition, whereas values less that 80 indicate suboptimal body condition and suggest less than ideal foraging conditions. Catch data were summarized as the number of fish CPUE and the weight in kg caught per unit effort (WPUE).

RESULTS

Three gill net pairs were set overnight in Bull Trout Lake and two gill net pairs were set in Martin Lake on June 27, 2010 (Figure 8). A total of 146 fish were captured in Bull Trout Lake (143 brook trout and three kokanee), while 54 fish were caught in Martin Lake (51 brook trout and three hatchery rainbow trout); (Table 1). Brook trout CPUE in Bull Trout Lake was 48 fish, while kokanee CPUE was one fish. In Martin Lake, brook trout CPUE was 26 fish and hatchery rainbow trout CPUE was two fish. No other fish species were captured in the lakes.

The ranges of brook trout lengths were similar for both lakes. Lengths ranged from 142-286 mm in Bull Trout Lake and 168 - 282 mm in Martin Lake (Figure 9). However, the proportion of fish >200 mm was much higher in Martin Lake (72%) than in Bull Trout Lake (26%). Therefore, despite the higher brook trout CPUE in Bull Trout Lake, WPUE was similar between the lakes, with 3 kg in Bull Trout Lake and 3 kg in Martin Lake. Mean W_r for brook trout was 90 and 96 for Bull Trout and Martin lakes, respectively.

Three kokanee, all approximately 170 mm, were also captured in Bull Trout Lake. Although stocking records from 1967 to present do not indicate kokanee were stocked, they had previously been observed in standard gill net surveys in 1991. No hatchery rainbow trout were captured in Bull Trout Lake suggesting carry-over for hatchery fish was negligible.

Three hatchery rainbow trout were captured in Martin Lake, but the fish appeared to be in poor condition. Average W_r of the three fish was 71, indicating poor body condition. In addition, a number of dead hatchery rainbow trout were visible on the lake bottom throughout the lake. Based on decomposition, it appeared the fish had died within the previous month.

DISCUSSION

Historically, brook trout have dominated the catch in standard gill net surveys at Bull Trout Lake, and this trend continued in 2010. Previously, CPUE for brook trout was 54 in 1991 and 86 in 1994 (Holubetz et al. 1994; Allen et al. 2000). The mean length of these fish have remained relatively constant throughout this period as well, with brook trout averaging 199 $\pm$ 8 mm (mean $\pm$ 90% CI) in 1991, 190 $\pm$ 6 mm in 1994, and 192 $\pm$ 5 mm in 2010. No brook trout $\geq$300 mm have been captured in any of the previous surveys. Therefore, there is strong evidence that the brook trout population in Bull Trout Lake is stunted. Bull Trout Lake is low in productivity, possesses ample spawning habitat (shoreline and inlet/outlet streams), and has no predators capable of consuming fish $\geq$ 200 mm, which all contribute to stunting (Donald and Alger 1999). Parker et al. (2001) demonstrated that anglers lose interest in stunted brook trout populations. Successful density reduction efforts have been shown to shift the length distributions of brook trout toward larger fish, thus improving the fishery quality (Koenig 2010). As there currently are no species conservation reasons to eradicate brook trout in Bull Trout Lake, a reduction in density and increased growth of brook trout would benefit anglers and perhaps the survival and growth of hatchery rainbow trout as well. Reduction efforts such as the introduction piscivorous fish such as the sterile Tiger muskellunge *Esox lucius X masquinongy* or overwinter gill net sets are attractive low-cost methods that should improve the fishery quality at Bull Trout Lake.

Martin Lake has not been previously sampled with standard gill nets; and therefore historic data on species composition and size distribution is not available. Based on length distribution of brook trout collected in 2010 along with creel results in 1998, brook trout in Martin

Lake are also at high density. However, given the much smaller size of Martin Lake and a length distribution of slightly larger fish, angler harvest may be having more of an effect on the brook trout population. In addition, the 2010 gill net survey and the 1998 creel survey showed that some hatchery rainbow trout are surviving through winter and contributing to the fishery the following year. Therefore, the brook trout fishery and carry-over of rainbow trout in Martin Lake may also be improved by brook trout reduction efforts.

MANAGEMENT RECOMMENDATIONS

1. Investigate the use of tiger muskellunge or intensive gill netting to reduce brook trout populations in both Bull Trout and Martin lakes to increase the size and quality of brook trout and improve survival and carry-over of hatchery rainbow trout.

2. Survey anglers at Bull Trout and Martin lakes to determine if they would support reduced catch rates in exchange for catching larger brook trout.

Table 1. Catch and catch per unit effort (CPUE), biomass (kg), weight per unit effort (WPUE) statistics by species for the standardized gillnet surveys at Bull Trout and Martin lakes in June 2010.

Water	Species	Gill Net Catch	Gill Net CPUE	Gill Net Weight (kg)	Gill Net WPUE (kg)	Average W_r
Bull Trout Lake	Brook trout	143	47.7	9.6	3.2	90.1
	Kokanee	3	1.0	0.1	0.0	86.1
Martin Lake	Brook trout	51	25.5	5.8	2.9	95.5
	Rainbow trout	3	1.5	0.6	0.3	71.1

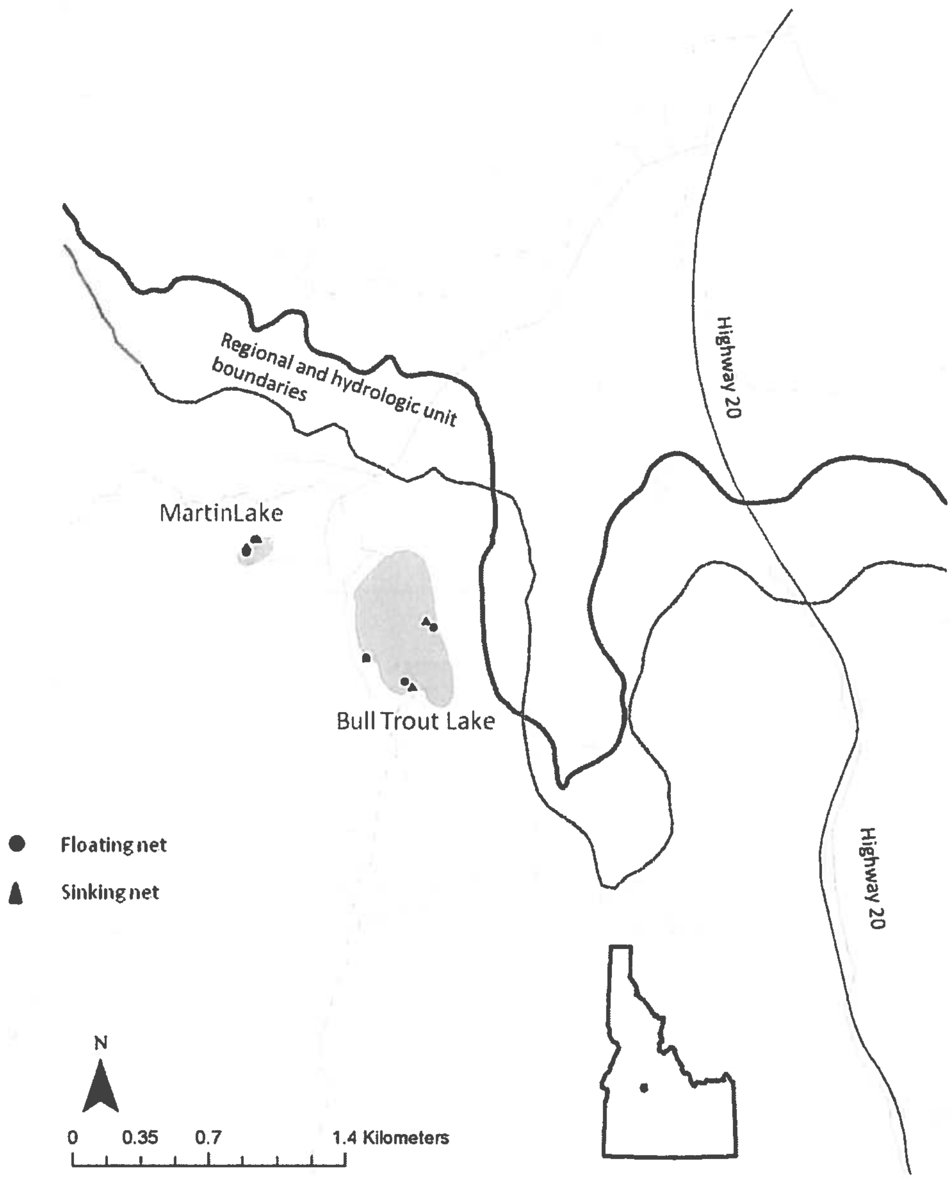

Figure 8. Map of Bull Trout and Martin lakes, Idaho, showing gill net locations during the June 2010 standardized gill net survey.

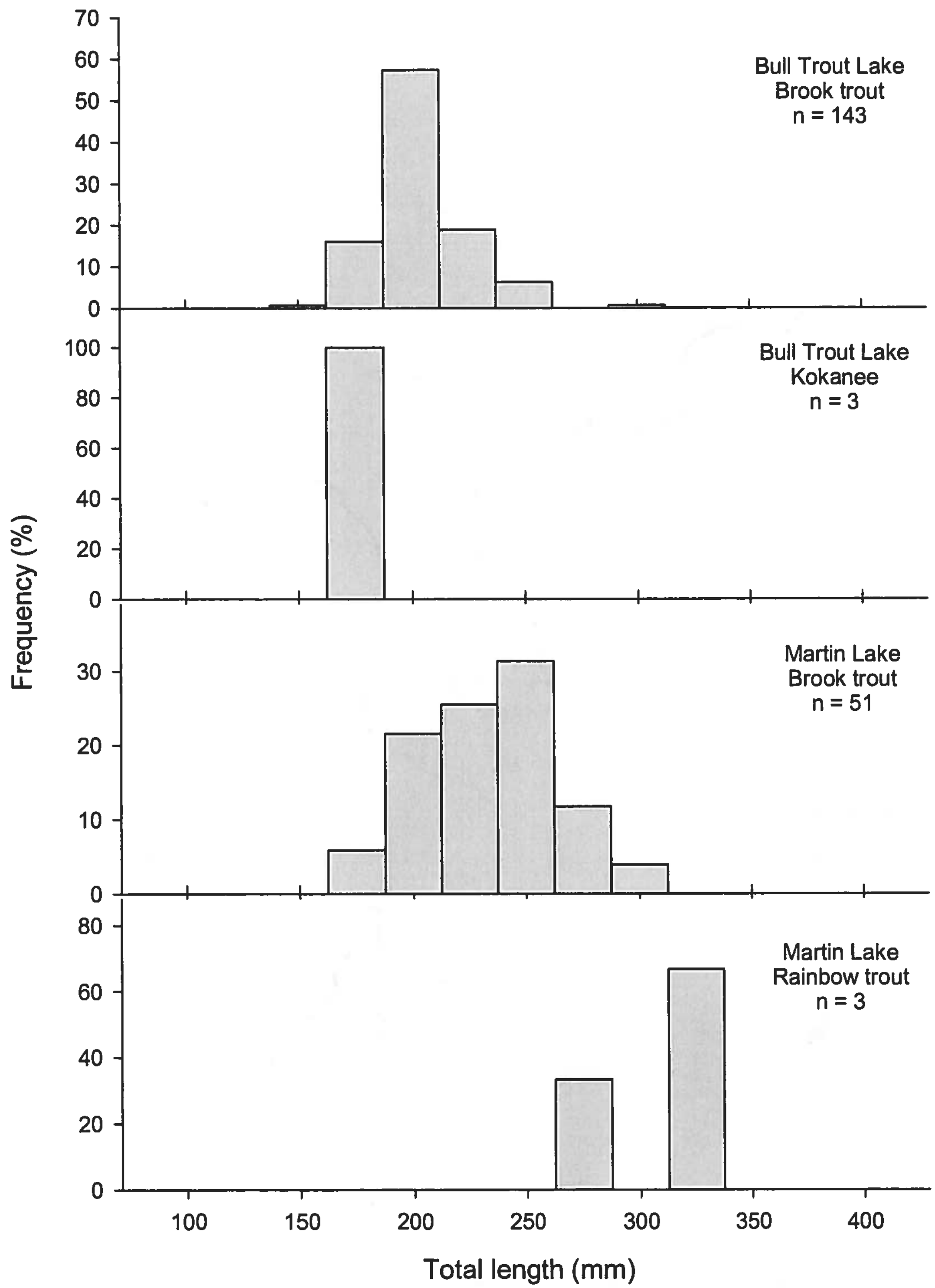

Figure 9. Length distributions of brook trout, kokanee, and rainbow trout collected during the standardized gill net surveys at Bull Trout and Martin lakes in June 2010.

Lowland Lakes Surveys

Community Fishing Ponds - Chemical Treatment of Nuisance Aquatic Plants in Duff Lane and Lowman Ponds

ABSTRACT

Excessive aquatic plant growth in Lowman Ponds and Duff Lane Pond was hampering fishing opportunities especially for shore-bound anglers. In order to maintain fisheries quality, we chemically treated these waters with Navigate®, a granular 2, 4 D, at 100 - 150 lbs/acre. Submerged aquatic plant abundance was reduced by greater than 95% in these ponds. Effective weed management in the coming years will require vigilance and finding a balance between weed eradication and maintaining aquatic plants communities for invertebrates communities and as well as juvenile fish cover.

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INTRODUCTION

IDFG Southwest Region manages more than 25 publically-accessible small ponds and reservoirs. Ponds receive significant fishing pressure and are an important resource for providing family-friendly fishing opportunities; and therefore, are thought to be important for angler recruitment and retention efforts. Excessive plant growth, especially during the summer months, in some ponds may limit access or in extreme cases may totally preclude fishing. Furthermore, excessive plant growth may create biological problems such as excessive oxygen consumption during decomposition or may provide too much cover for juvenile fish leading to high abundances and small average sizes. By early spring 2010, excessive plant growth had covered much of the surface area of Lowman Ponds and Duff Lane Pond. Plants in Lowman Ponds were identified as northern milfoil *Myriophyllum sibericum.* Plants identified in Duff Lane were predominantly curly-leaf pondweed *Potamogeton crispus*, small amounts of Eurasian milfoil *Myriophyllum spicatum*, as well as cattails *Typha spp.* surrounding the perimeter. Regional personnel using partial financial assistance from the Idaho Department of Agriculture treated these waters with granular herbicides to reduce plant abundance.

METHODS

We selected Navigate®, a granular 2, 4 d, to treat these waters, based on past efficacy in nearby waters. Prior to treatment, we closed these ponds to prevent human contact. Additionally, we blocked the diversion inflow structure at Lowman Ponds to maximize herbicide contact time and eliminate downstream transport of herbicide over the spillway. Recommended treatment levels were 112 – 170 kg per ha for identified plant species. We used IDFG Graphic Information System (GIS) information to estimate surface ha for Lowman Ponds (0.7 ha), and Duff Lane Pond (2.3 ha). We applied 90 kg of Navigate®, a granular 2, 4 d, in Lowman ponds on May 25, 2010 using a hand-held fertilizer spreader while standing on the deck of a small boat. After 14 days, we returned to Lowman Ponds to manually remove dead plants with rakes. Several thousand pounds of plant material was removed over 4 days. On June 3, 2010, we applied 363 kg of Navigate® to Duff Lane Pond in the same manner, except we mounted the spreader to the gunwale of a small boat and no subsequent raking occurred. Ponds were visually inspected several times before fall.

RESULTS AND DISCUSSION

Application of herbicide was effective in Lowman Ponds. Maximum depth was approximately 3 m; however, many of the dead plants exceeded 3.5 m in length. Nearly all plants in the upper pond were killed, whereas the majority (>90%) of plants in the lower pond were killed. After plants died and raking efforts were completed, Lowman ponds were cleared sufficiently to allow unhampered fishing opportunities. No significant plant re-growth occurred prior to fall.

Similarly, herbicide treatments were effective in Duff Lane Pond. Over 95% of rooted submerged vegetation was killed. Emergent vegetation (i.e. cattails) was only impacted where directly contacted by the herbicide, near the open water perimeter. No significant plant re-growth occurred prior to fall. Also, no dead fish were observed during or after either treatment. Effective weed management in the coming years will require vigilance and finding a balance between weed eradication and maintaining aquatic plants communities for invertebrates and as juvenile fish cover.

MANAGEMENT RECOMMENDATIONS

1. Monitor plant re-growth in Lowman Ponds. Re-apply herbicide on a semi-annual basis as needed.

2. Utilize grass carp in Duff Lane Pond as a proactive measure to keep aquatic plant abundance at relatively low levels.

3. Periodically monitor plant growth in other community fishing waters and treat as necessary.

Lowland Lakes Surveys

Community Fishing Ponds - Use of Transplanted Catfish to Provide Enhanced Summer Fishing Opportunities

ABSTRACT

Capturing wild adult channel catfish and translocating them to high-use ponds may be a cost effective alternative to stocking commercially produced fingerlings for creating fisheries during summer months, if translocated fish are caught readily by anglers. During summer 2009 and 2010, we captured and translocated 1,311 catfish to eight ponds in southwest Idaho. Carlin-Dangler tags were affixed to 438 of these fish prior to release. Mean length and weight was 556 mm (± 4; 90% CI) and 1,885 g (± 37). Voluntary tag returns were monitored through IDFG's tag reporting hotline and entered into a database. We queried the database on January 1, 2011, and thus estimates for fish translocated during 2009 are for one full year, plus a second partial year, whereas estimates for fish translocated during 2010 are partial, first year only. Return rates were corrected to account for non-reporting. For 2009 translocations, total corrected harvest rate averaged 27% (± 11), whereas total corrected release rate equaled 9% (± 6). There was no difference in harvest rate among the three translocation periods (June, July, and September). Mean time to harvest for 2009 translocations was 220 d (± 32) with a maximum of 500 days. There was no difference in time to harvest between translocation periods, despite initial evidence to the contrary. All catfish were harvested or released from March through October. For 2010 translocations, total corrected harvest rate averaged 13% (± 10), whereas total corrected release rate equaled 2% (± 2). Capture and translocating channel catfish has shown to be a useful tool for increasing summer fishing opportunities in some community fishing ponds. Continued monitoring of tag returns will allow us to fine tune stocking locations, determine persistence of translocated fish, as well as assess inter-annual variation in performance.

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INTRODUCTION

IDFG's Southwest Region manages more than 25 small ponds and reservoirs, hereafter referred to as ponds. The majority are located within heavily-populated areas, and receive significant fishing pressure. Also, they seem to be an important resource for providing easily-accessible, family-friendly fishing opportunities, and are thought to be vital in angler recruitment and retention efforts. Most ponds have self-sustaining largemouth bass *Micropterus salmoides* and bluegill populations. Natural production in these ponds is unable to meet angling demands; and therefore most ponds are stocked seasonally with catchable-sized rainbow trout. Catchable-sized rainbow trout are usually stocked on a bi-weekly or monthly basis from September through June. Summer water temperatures in southwestern Idaho ponds usually exceed thermal limits for rainbow trout, requiring a stocking cessation during July and August, occasionally stretching into June and September during warm years (Hebdon et al. 2008). Unfortunately, stocking cessations coincide with peak fishing-effort periods.

Regional staff is interested in improving fisheries quality in the Southwest Region's urban ponds during summer peak effort periods through a channel catfish stocking program. A cost-based assessments of our management options led us to switch to an adult channel catfish capture and translocation program rather than the purchasing and stocking of commercially produced channel catfish fingerlings (Kozfkay et al. 2010). To further gauge the cost effectiveness of this new program, we needed to gain a better understanding of how anglers were utilizing translocated channel catfish. Specifically, we sought to estimate relative harvest and release rates, total harvest and the number of fish released, times to encounters (harvest or release), as well as persistence.

METHODS

Catfish capture efforts were conducted on the Snake River from Walters Ferry to Nyssa, Oregon. An aluminum jet sled equipped with two boom-mounted anodes and a 5,000 watt generator was used. Output was controlled by a Smith Root VVP-15. Frequency was set at 80-120 pulses/ sec and a pulse width of 15, which yielded an output of 5-6 amps.

Captured catfish were held in a 280 L livewell equipped with a re-circulating pump and supplemental oxygen. After approximately 75 - 100 catfish had been captured, they were placed in a 1,100 L fish transport trailer at a boat ramp. After several runs and the capture of 225 - 300 fish, channel catfish were translocated to local ponds. Our translocation targets were approximately 25 - 75 catfish once-per-month in June (early), July (middle), and August or September (late), though not all ponds were stocked at this level in all months. Channel catfish were the primary target species; however, we did capture and translocate, flathead catfish *Pylodictis olivaris* occasionally, but only in small numbers (approximately 2% of the total) and none were tagged. Capture and transport efforts required 2- ½ ton trucks, 1 jet boat electrofishing unit, 1 fish transport trailer, and 4 IDFG employees. Usually, it required about 3 d of effort to complete the capture and transport efforts for eight ponds. During 2009, eight ponds were used including Beachs, Caldwell #2, Ed's, Horseshoe Bend Mill, McDevitt (aka Norms), Park Center, Quinns, and Sawyers ponds (Figure 10). During 2010, Caldwell #2 and Quinns ponds were replaced by Caldwell Rotary and Riverside ponds. Ponds ranged from 0.4 to 4 ha (surface).

Tagged channel catfish were released during all stocking events to estimate angler utilization. Prior to release, we affixed Carlin dangler tags (Wydoski and Emery 1983) to approximately one-third of translocated fish. Each tag was threaded to the mid-point of a 200 mm piece of stainless steel wire. After the tag was positioned at the mid-point of the wire, we twisted the wire five times to lock it in place. Then, the tagging apparatus, a pair of hypodermic needles affixed to a wooden dowel, was inserted into each tagged fish's body below and slightly posterior to the dorsal spine. The wire's tag ends were slid through the hypodermic needles, the needles were then removed, and the tag ends were twisted about five times on the opposite side of the fish and trimmed. Each tag possessed a unique identification number, the abbreviation IDFG, and a tag reporting hotline phone number (1-866-258-0338) to facilitate the reporting of caught fish. Furthermore, a tag reporting portal was available on IDFG's website (http://fishandgame.idaho.gov/apps/fishtag/). Length and weight of each tagged fish was recorded prior to release.

Catch, harvest, and release information were recorded in IDFG's fish tagging database and we queried available information January 1, 2011. Thus, estimates reported in this document for fish stocked during 2009 are for 478 to 572 days at large, whereas estimates for fish stocked during 2010 are partial first year only (135 to 194 days at large). Voluntary return of tags by anglers were adjusted for non-reporting by dividing tag returns by the mean tag reporting rate (53%) estimated for other Idaho fisheries (Meyer et al. 2009). No further corrections were made as tagging mortality and tag loss for channel catfish with this tagging method equaled zero in Missouri impoundments (Michaletz et al. 2008).

RESULTS

2009 Capture and Translocation Efforts

During June, July, and September 2009, we captured 1,296 channel catfish and translocated them to eight ponds. Mean length and weight was 562 mm (± 4; Figure 11) and 1,879 g (± 45). We affixed tags to 438 of these fish (33%). Tags were voluntarily reported by anglers from 74 catfish. Anglers reported harvesting 60 and releasing 14 catfish (Table 2 and 3) during the 478 to 572 d these fish were at large. Additionally, one tagged fish in McDevitt Pond was reported to have been released twice.

For channel catfish captured and translocated during 2009, un-corrected tag return rates indicating harvest showed high variation across ponds. Translocation period had little effect on first-year return rate. For all of the 2009 stocking periods and ponds combined, the un-corrected tag return rate indicating harvest within one year of stocking was 11%, whereas partial, second-year return rates indicating harvest were 3% (Table 4). Un-corrected, first-year tag return rates indicating harvest for the early (11%), middle (12.7%), and late (10%) translocation periods were similar. Second-year return rates showed a translocation period effect with earlier translocations returning at higher rates than later translocations, though time at large, especially during optimal fishing periods, confounded this observation. High tag return rates indicating harvest (first and partial second-year combined) were noted from McDevitt (33%), Parkcenter (23%), Caldwell #2 (17%), and Eds (17%) ponds (Table 4). Harvest rates for the other four ponds were 5 - 7%. Using tag reporting rates of 53%, total exploitation (i.e. harvest) for fish captured and translocated during 2009, ranged from 12% (Beachs Pond) to 63% (McDevitt Pond) with a mean of 26%. Mean length and weight of catfish harvested was 566 mm (±12; *n* = 59) and 1,881 g (± 122).

Reported release rates for channel catfish captured and translocated during 2009 were approximately one-quarter of the harvest rate. For all the 2009 ponds and translocated efforts combined, the un-corrected tag return rate indicating release within one year of stocking was 3.2%, whereas partial, second-year return rate indicating release was 1.8% (Table 5). High release rates (first and second year partial combined) were noted from McDevitt (15%) and Parkcenter (11%) ponds. Release rates for the other six ponds ranged between 0.5 to 4%. Using the same tag reporting rate, adjusted release rates averaged 9.5%. Mean length and weight of catfish released was 582 mm (± 29; 90% CI; n = 14) and 2,029 g (± 357).

Analysis of time to harvest and time to release indicated that channel catfish captured and translocated during 2009 returned to anglers over a relatively long time period. Anglers reported harvesting channel catfish on the day of translocation and for up to 500 days with bi-modal peaks in the distribution (Figure 12), with the trough coinciding with late fall through early spring. Mean time to harvest (± 90% CI) was 220 d (± 32;). Mean time to harvest was not statistically different among early ($\bar{x}$ = 191 d ± 60, n = 25), middle ($\bar{x}$ = 237 d ± 56, n = 20), and late ($\bar{x}$ = 244 d ± 46, n = 15) translocation periods. Similarly, anglers reported releasing channel catfish on the day of translocation and for up to 397 days afterwards. Mean time to release was 201 d (± 72). Sample sizes were insufficient to allow meaningful comparisons of time to release by translocation period. All fish reported harvested or released were caught during the months of March through October (Figure 13).

2010 Capture and Translocation Efforts

During 2010, we captured 1,239 channel catfish and translocated them to eight ponds. Two of the poorer performing ponds based on 2009 returns (Caldwell #2 & Quinns ponds) were replaced with alternatives (Caldwell Rotary and Riverside ponds) with the intention of increasing overall utilization of these fish. Mean length and weight was 551 mm (± 5; n = 448) and 1,890 g (± 60) and was not statistically different from 2009, nor was any size difference detected among translocation periods in either year. We affixed tags to 448 channel catfish (36%). Through December 31, 2010, tags were voluntarily reported by anglers from 44 channel catfish. Anglers reported harvesting 36 and releasing 8 catfish (Table 6 and 7).

Partial, first-year, unadjusted tag return rates indicating harvest differed widely among ponds, and showed a translocation period effect. Overall, partial-year, tag return rates indicating harvest were 8%, and ranged from zero (Beachs, Eds, and Horseshoe Bend ponds) to 27% (McDevitt Pond; Table 8). Translocation period seemed to affect these rates with the June translocation (14.3%) returning at higher rates than the July (2.5%), or August translocations (7.3%), though time at large likely influenced these results. Adjusted partial first-year harvest rates averaged 3.4%. Mean length and weight of catfish harvested was 550 mm (± 24; n = 36) and 1,839 g (± 213).

Partial, first-year, unadjusted tag return rates indicating release occurred for only three of the eight ponds that received translocated catfish. Overall, eight (1.8%) of the 448 channel catfish caught, tagged, and translocated during 2010 were reported as being caught and released (Table 9). The release of tagged fish was only reported from McDevitt, Parkcenter, and Sawyers ponds. Adjusted partial first-year release rates averaged 3.4%. Mean length and weight of catfish released was 586 mm (±59; n = 8) and 2,246 g (± 687).

DISCUSSION

Wide variation in encounters (harvest or release) of translocated channel catfish among ponds was evident. For instance, approximately 90% of channel catfish translocated to McDevitt Pond during 2009 have been encountered within approximately 1.5 years, when harvest and release are corrected for and combined. Similarly, encounter of tagged fish in Parkcenter Pond was also high (65%). In contrast, Beachs (12%), Horseshoe Bend (11%), and Quinns (13%) ponds performed poorly. Encounter rates in all other ponds were intermediate to these values, but had sufficient utilization to justify continued capture and translocation efforts. Replacement of poor performing ponds with alternatives may be necessary if few additional tags are reported as this study continues. Alternatively, we may need to further publicize stocking locations to increase angling effort or investigate whether fish are leaving these waters through outlet structures.

Tag returns indicated that anglers tended to favor harvesting captured fish. For the 2009 efforts, returns indicated that three out of four captured fish were harvested, with the fourth being released. We also saw no evidence of size selection by anglers. Fish were harvested and released throughout the range of available sizes. Translocated catfish showed strong seasonal performance differences with no tag returns reported during late fall, winter, and early spring. Translocated catfish seemed to be remarkably persistent with little evidence of mortality. Return of tags after fish had spent one winter at large was high; evidenced by average times to harvest exceeding 200 d. Quite remarkably, McDevitt Pond was drawn down to very low levels to control Eurasian milfoil levels during winter 2009-2010 and many translocated channel catfish survived this effort and were caught after the pond refilled. Based on the frequency of second year returns, we fully expect to receive additional returns as tag return monitoring continues.

The utilization of captured and translocated channel catfish has shown this to a be an effective, low-cost method for creating fisheries in some high-use ponds. This seems to be especially true during the summer months when the typical stocking of catchable-sized rainbow trout is precluded due to warm water temperatures. Our previous preliminary observations from the 2009 capture and translocation efforts indicated that translocation period had an effect on eventual return rates; however, after including over a full year's worth of return data, this was not the case. We saw no difference in eventual return rate among translocation periods. Apparently, channel catfish translocated later in the year that initially showed poor return rates were able to survive the winter. As water temperatures warmed and fishing effort increased the following spring, the return rates of subsequent translocated groups of fish increased and was similar to that of earlier groups that returned more quickly.

MANAGEMENT RECOMMENDATIONS

1. Continue to monitor the return of tagged catfish until fish from the 2010 capture and translocation efforts have been at large for a full two years (September 2012). Additional tag return information should allow further evaluation of pond and seasonal differences in performance as well as whether publicity efforts have increased utilization.

2. Continue to publicize this program through media outlets to gain attention, especially for underperforming waters.

3. Add Caldwell Pond #2 back to the list of ponds for this program, inclusion of additional data revealed that captured and translocated catfish were utilized at a high rate.

4. Beaches, Horseshoe Bend, and Sawyers ponds have performed poorly. Increase publicity of the program for these ponds. If poor performance persists, remove from translocation list and replace with alternatives.

Table 2. Number of channel catfish tagged and translocated by period to eight Southwestern Idaho Urban ponds during 2009 as well as the number of tags reported by anglers indicating harvest within the first and second years after translocation.

	Early				Middle				Late				Total			
Pond Name	# Tagged	First	Second	Total	# Tagged	First	Second	Total	# Tagged	First	Second	Total	# Tagged	First	Second	Total
Beachs	25	2		2					6				31	2		2
Caldwell #2	25	3	3	6	25	2	1	3	25	4		4	75	9	4	13
Eds	7	2		2	6				10	2		2	23	4		4
Horseshoe Bend	29	1	1	2	33				24	2		2	86	3	1	4
McDevitt	17	4	1	5	11	5	1	6	12	2		2	40	11	2	13
Park Center	26	4	1	5	19	7	1	8	25	3		3	70	14	2	16
Quinns					20	2		2	22	1		1	42	3		3
Sawyers	25	1	2	3	20	1		1	26	1		1	71	3	2	5
Total	154	17	8	25	134	17	3	20	150	15		15	438	49	11	60

Table 3. Number of channel catfish tagged and translocated by period to eight Southwestern Idaho Urban ponds during 2009 as well as the number of tags reported by anglers indicating release within the first and second years after translocation.

	Early				Middle				Late				Total			
Pond Name	# Tagged	First	Second	Total	# Tagged	First	Second	Total	# Tagged	First	Second	Total	# Tagged	First	Second	Total
Beachs	25							0	6			0	31			
Caldwell #2	25				25	2		2	25	1		1	75	3		3
Eds	7				6			0	10	1		1	23	1		1
Horseshoe Bend	29				33			0	24	1		1	86	1		1
McDevitt	17	3	1	4	11	1	1	2	12			0	40	4	2	6
Park Center	26	1	4	5	19	2		2	25		1	1	70	3	5	8
Quinns					20			0	22			0	42			0
Sawyers	25				20	1		1	26	1	1	2	71	2	1	3
Total	154	4	5	9	134	6	1	7	150	4	2	6	438	14	8	22

Table 4. Tag return rate indicating harvest (top) and exploitation rate (bottom) estimates for channel catfish tagged during 2009. Rates were calculated from tags reported by anglers and were corrected to account for non-reporting.

Tag Return Rate

	Early			Middle			Late			Total		
Pond Name	First	Second	Total	First	Second	Total	First	Second	Total	First	Second	Total
Beachs	0.08		0.08						0.00	0.06		0.06
Caldwell #2	0.12	0.12	0.24	0.08	0.04	0.12	0.16		0.16	0.12	0.05	0.17
Eds	0.29		0.29			0.00	0.20		0.20	0.17		0.17
Horseshoe Bend	0.03	0.03	0.07			0.00	0.08		0.08	0.03	0.01	0.05
McDevitt	0.24	0.06	0.29	0.45	0.09	0.55	0.17		0.17	0.28	0.05	0.33
Park Center	0.15	0.04	0.19	0.37	0.05	0.42	0.12		0.12	0.20	0.03	0.23
Quinns				0.10		0.10	0.05		0.05	0.07		0.07
Sawyers	0.04	0.08	0.12	0.05		0.05	0.04		0.04	0.04	0.03	0.07
Total	0.11	0.05	0.16	0.13	0.02	0.15	0.10		0.10	0.11	0.03	0.14

Exploitation Estimates

	Early			Middle			Late			Total		
Pond Name	First	Second	Total	First	Second	Total	First	Second	Total	First	Second	Total
Beachs	0.15		0.15						0.00	0.12		0.12
Caldwell #2	0.23	0.23	0.45	0.15	0.08	0.23	0.30		0.30	0.23	0.10	0.33
Eds	0.54		0.54				0.38		0.38	0.33		0.33
Horseshoe Bend	0.07	0.07	0.13				0.16		0.16	0.07	0.02	0.09
McDevitt	0.44	0.11	0.55	0.86	0.17	1.03	0.31		0.31	0.52	0.09	0.61
Park Center	0.29	0.07	0.36	0.70	0.10	0.79	0.23		0.23	0.38	0.05	0.43
Quinns				0.19		0.19	0.09		0.09	0.13		0.13
Sawyers	0.08	0.15	0.23	0.09		0.09	0.07		0.07	0.08	0.05	0.13
Total	0.21	0.10	0.31	0.24	0.04	0.28	0.19		0.19	0.21	0.05	0.26

Table 5. Un-corrected (top) and corrected release rate (bottom) estimates for channel tagged during 2009. Rates were calculated from tags reported by anglers (that indicated release) and were corrected to account for non-reporting.

Un-corrected release rates

	Early			Middle			Late			Total		
Pond Name	First	Second	Total	First	Second	Total	First	Second	Total	First	Second	Total
Beachs												
Caldwell #2				0.08		0.08	0.04		0.04	0.04		0.04
Eds							0.10		0.10	0.04		0.04
Horseshoe Bend							0.04		0.04	0.01		0.01
McDevitt	0.18	0.06	0.24	0.09	0.09	0.18				0.10	0.05	0.15
Park Center	0.04	0.15	0.19	0.11		0.11		0.04	0.04	0.04	0.07	0.11
Quinns												
Sawyers				0.05		0.05	0.04	0.04	0.08	0.03	0.01	0.04
Total	0.03	0.03	0.06	0.04	0.01	0.05	0.03	0.01	0.04	0.03	0.02	0.05

Corrected release rates

	Early			Middle			Late			Total		
Pond Name	First	Second	Total	First	Second	Total	First	Second	Total	First	Second	Total
Beachs												
Caldwell #2				0.15		0.15	0.08		0.08	0.08		0.08
Eds							0.19		0.19	0.08		0.08
Horseshoe Bend							0.08		0.08	0.02		0.02
McDevitt	0.33	0.11	0.44	0.17	0.17	0.34				0.19	0.09	0.28
Park Center	0.07	0.29	0.36	0.20		0.20		0.08	0.08	0.08	0.13	0.22
Quinns												
Sawyers				0.09		0.09	0.07	0.07	0.15	0.05	0.03	0.08
Total	0.05	0.06	0.11	0.08	0.01	0.10	0.05	0.03	0.08	0.06	0.03	0.09

Table 6. Number of channel catfish tagged and translocated by period to eight Southwestern Idaho urban ponds during 2010 as well as the number of tags reported by anglers indicating harvest.

Pond Name	Early		Middle		Late		Total	
	# Tagged	First	# Tagged	First	# Tagged	First	# Tagged	First
Beachs	15		11		8		34	
Caldwell Rotary	15	2	20	1	16		51	3
Eds	9		7		8		24	
Horseshoe Bend	19		23		12		54	
McDevitt	21	10	22	1	17	5	60	16
Park Center	31	2	43	1	29	5	103	8
Riverside	23	7	15	1	14		52	8
Sawyers	21	1	16		33		70	1
Total	154	22	157	4	137	10	448	36

Table 7. Number of channel catfish tagged and translocated by period to eight Southwestern Idaho Urban ponds during 2010 as well as the number of tags reported by anglers indicating release.

	Early		Middle		Late		Total	
Pond Name	# Stock	First	# Stock	First	# Stock	First	# Stock	First
Beachs	15		11		8		34	
Caldwell Rotary	15		20		16		51	
Eds	9		7		8		24	
Horseshoe Bend	19		23		12		54	
McDevitt	21	1	22	1	17		60	2
Park Center	31	4	43		29	1	103	5
Riverside	23		15		14		52	
Sawyers	21		16		33	1	70	1
Total	154	5	157	1	137	2	448	8

Table 8. Tag return rate (top) and exploitation rate (bottom) estimates for channel catfish tagged during 2010. Rates were calculated from tags reported by anglers (that indicated harvest) and were corrected to account for non-reporting.

Tag return rate (harvest)

Pond Name	Early	Middle	Late	Total
Beachs	0	0	0	0
Caldwell Rotary	0.13	0.05	0	0.06
Eds	0	0	0	0
Horseshoe Bend	0	0	0	0
McDevitt	0.48	0.05	0.29	0.27
Park Center	0.06	0.02	0.17	0.08
Riverside	0.30	0.07	0	0.15
Sawyers	0.05	0	0	0.01
Total	0.14	0.03	0.07	0.08

Exploitation rate

Pond Name	Early	Middle	Late	Total
Beachs	0	0	0	0
Caldwell Rotary	0.25	0.09	0	0.11
Eds	0	0	0	0
Horseshoe Bend	0	0	0	0
McDevitt	0.90	0.09	0.55	0.50
Park Center	0.12	0.04	0.33	0.15
Riverside	0.57	0.13	0	0.29
Sawyers	0.09	0	0	0.03
Total	0.27	0.05	0.14	0.15

Table 9. Un-corrected (top) and corrected release rate (bottom) estimates for channel tagged during 2010. Rates were calculated from tags reported by anglers (that indicated release) and were corrected to account for non-reporting.

Un-corrected Release Rates

Pond Name	Early	Middle	Late	Total
Beachs	0	0	0	0
Caldwell Rotary	0	0	0	0
Eds	0	0	0	0
Horseshoe Bend	0	0	0	0
McDevitt	0.05	0.05	0	0.03
Park Center	0.13	0	0.03	0.05
Riverside	0	0	0	0
Sawyers	0	0	0.03	0.01
Total	0.03	0.01	0.01	0.02

Corrected Release Rates

Pond Name	Early	Middle	Late	Total
Beachs	0	0	0	0
Caldwell Rotary	0	0	0	0
Eds	0	0	0	0
Horseshoe Bend	0	0	0	0
McDevitt	0.09	0.09	0	0.06
Park Center	0.24	0	0.07	0.09
Riverside	0	0	0	0
Sawyers	0	0	0.06	0.03
Total	0.06	0.01	0.03	0.03

Figure 10. Location of ponds that received captured and translocated catfish during 2009 or 2010.

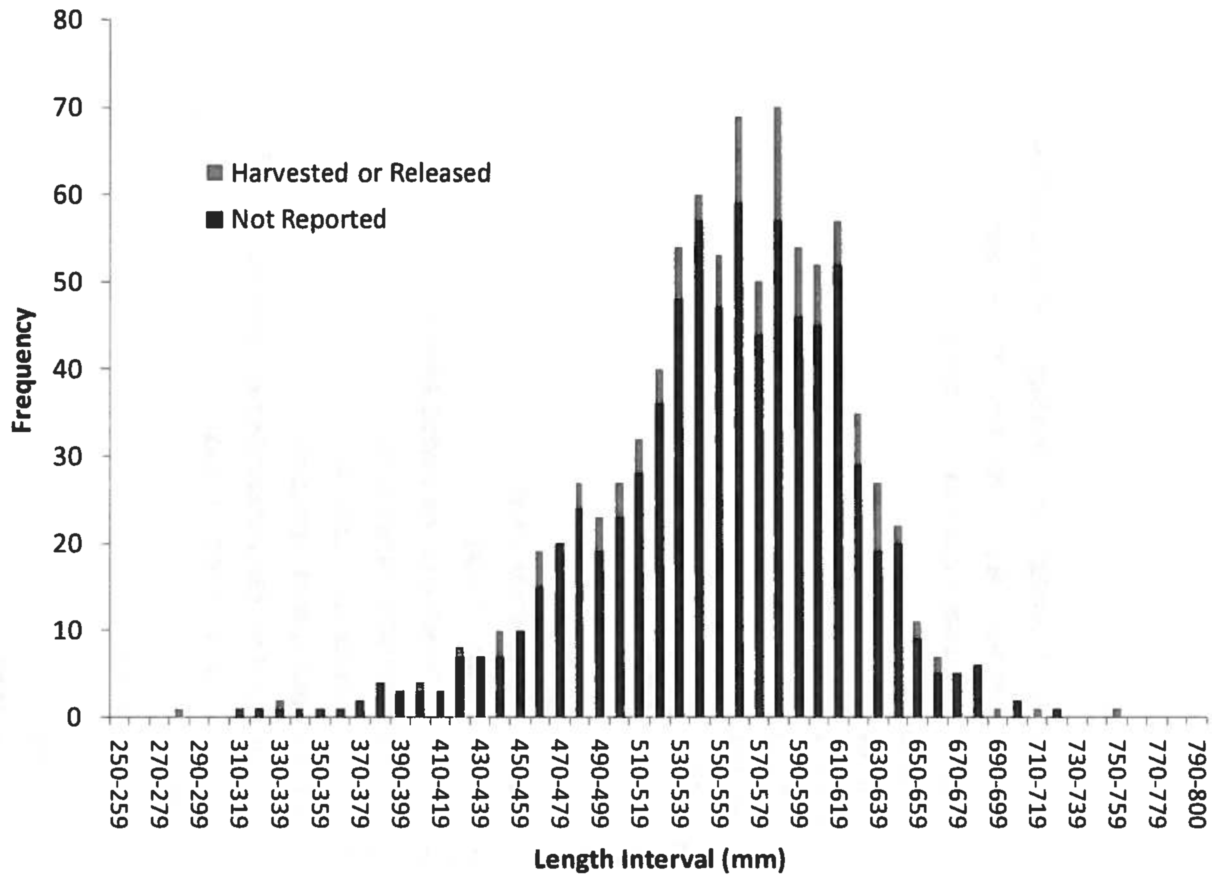

Figure 11. Length frequency of channel catfish (n = 885) captured, tagged, and translocated during 2009 and 2010.

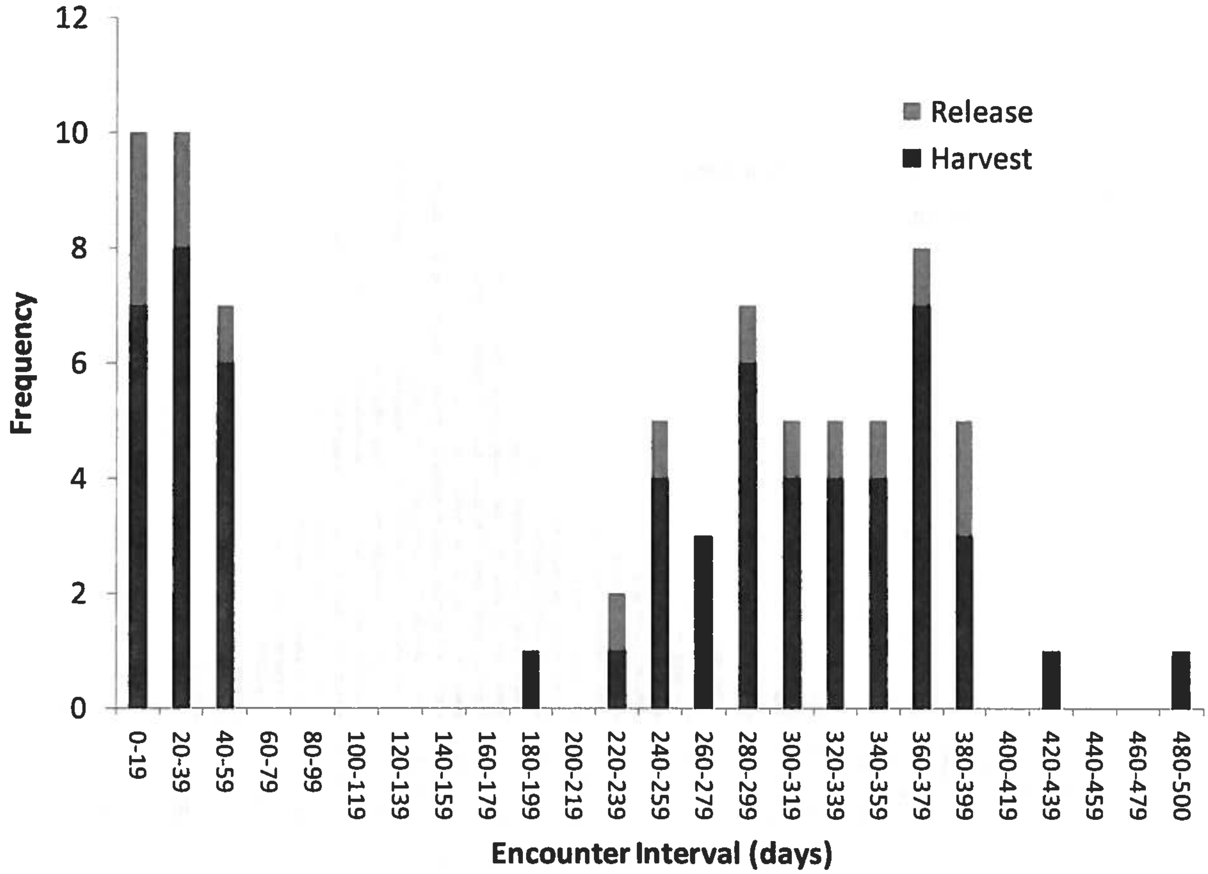

Figure 12. Time to encounter (harvest or release) for channel catfish captured and translocated during 2009.

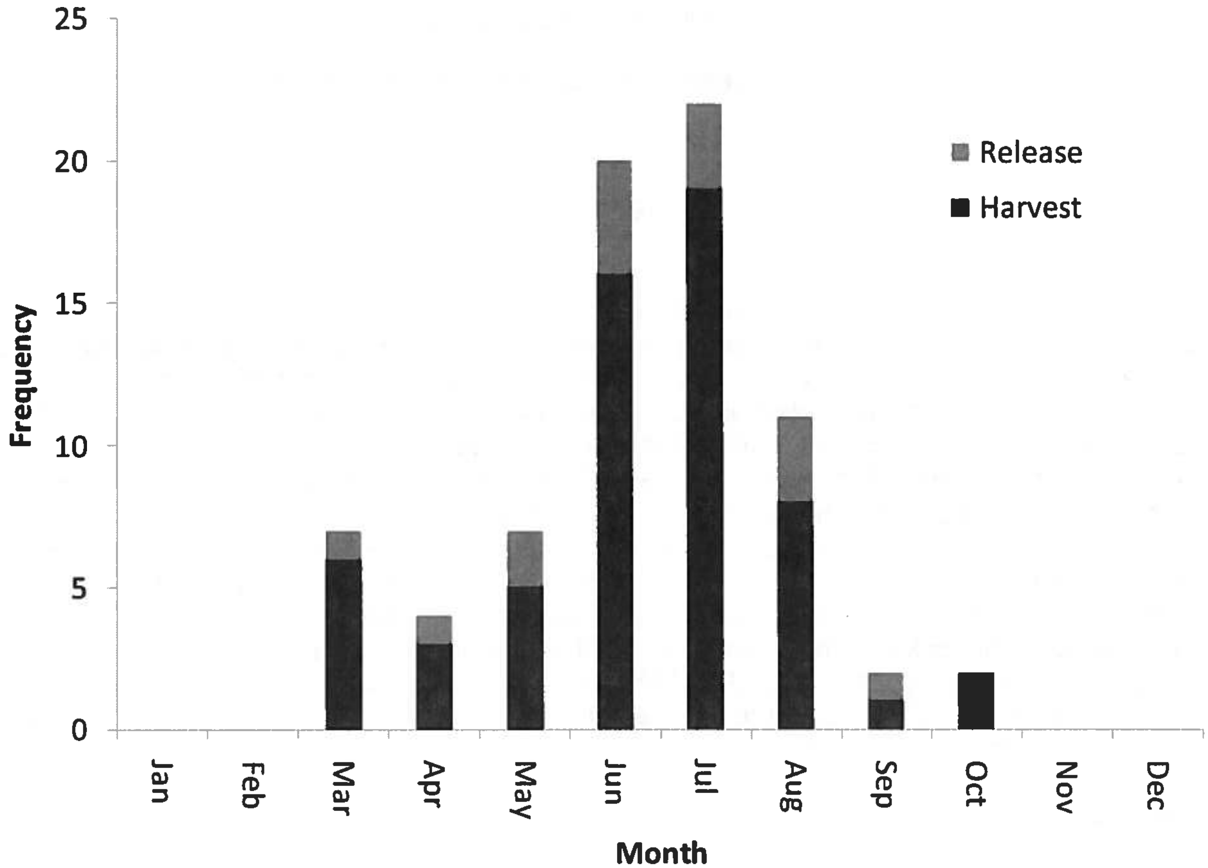

Figure 13. Month of encounter for channel catfish (harvest or release; n = 74) captured and translocated during 2009.

Lowland Lakes Surveys

Deadwood Reservoir - 2010 Kokanee Monitoring

ABSTRACT

Fourteen hydroacoustic transects were conducted at Deadwood Reservoir on July 14, 2010 to estimate kokanee abundance. Converted target strengths suggested that kokanee ranged between 30 and 380 mm, and the length frequency from converted target strength corresponded well with fish collected during mid-water trawling. Fish densities among transects ranged from 1,101 fish/ha to 3,461 fish/ha with the highest densities (1,455 fish/ha) of fish corresponding to age-0 fish. Age-3 kokanee displayed the lowest densities (43 fish/ha) among age classes. Overall, total mean kokanee density was 1,801 (1,599 to 2,093) fish/ha. When expanded to a population estimate using the reservoir surface area (1,212 ha) on the survey date, a total of 2,183,301 (1,938,090 to 2,459,519) kokanee were estimated. Hydroacoustic evaluations of the Deadwood Reservoir kokanee population suggest that that the population is responding to the lack of control efforts in 2009. Assuming the 2010 age-0 year class results in a large spawning escapement in 2013, IDFG may wish to operate the Deadwood River weir for the duration of the spawning run to limit recruitment. In addition it may be necessary to restrict spawning fish from Trail Creek.

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INTRODUCTION

Deadwood Reservoir is a 1,260-ha impoundment located on the Deadwood River in Valley County, approximately 40 km southeast of Cascade and 85 km northeast of Boise, Idaho (Figure 14). Deadwood Reservoir provides sport fishing opportunity for kokanee, rainbow trout, and westslope cutthroat trout *O. clarkii lewisii.* Bull trout are present in Deadwood Reservoir at very low numbers. In addition, resident fall Chinook salmon, have been stocked at low densities (5 fish/ha) beginning in 2009.

Over the last 10 years, the kokanee population in Deadwood Reservoir has cycled drastically. Because kokanee exhibit density-dependent growth, increases in population result in decreases in adult fish length. Historically, this relationship has been especially evident at Deadwood Reservoir as the kokanee population experiences relatively low angler pressure and has access to five tributaries with excellent spawning habitat. In addition, Deadwood Reservoir contains very small populations of piscivorous predators that are not capable of exerting an impact upon the kokanee population.

Mean female kokanee length observed at the kokanee spawning trap on the Deadwood River has varied from a low of 208 mm in 1992 to a high of 421 mm in 2003 with mean size decreasing since 2003. The management goal for adult kokanee at Deadwood Reservoir is an average size of 325 mm. Deadwood Reservoir also functions as one of the state's primary egg sources in Idaho, providing early spawning kokanee for stocking throughout the state. However, the egg take operation at Deadwood Reservoir was discontinued for one year in 2009 because a permanent weir was constructed on the South Fork Boise River (SFBR). Egg take operations at Deadwood Reservoir resumed in 2010 and are expected to remain there for the foreseeable future.

METHODS

Mid-Water Trawling

Mid-water trawling was conducted at night during the dark (new) moon on July 13, 2010 for hydroacoustic target and age verification. Trawling methodology and analysis is described in detail in Butts et al. (2011).

Hydroacoustics

Hydroacoustic estimates of fish densities, lengths, and vertical depth distributions were obtained with a Hydroacoustic Technology, Inc. (HTI) Model 241-2 split-beam digital echosounder on July 14, 2010. Hydroacoustic methodology and analysis is described in detail in Butts et al. (2011).

RESULTS

Mid-Water Trawling

Mid-water trawling captured 116 kokanee, ranging in size from 29 - 232 mm, on July 13, 2010 (Figure 15). Length frequency was used along with counting annuli on whole otoliths to construct size ranges of three age classes. These analyses suggested that age-0 fish were fish <100mm, age-1 fish were between 100 - 200 mm, age-2 fish between 200 - 300 mm, and age-3 fish were >300 mm.

Hydroacoustics

Fourteen hydroacoustic transects were run at Deadwood Reservoir on July 14, 2010. Converted target strengths suggested that kokanee ranged between 30 and 380 mm and the length frequency from converted target strength corresponded well with fish collected during mid-water trawling (Figure 15). Therefore, length-age relationships estimated from mid-water trawling were used to partition hydroacoustic estimates into estimates for individual age classes.

Fish densities among transects ranged from 1,101 fish/ha to 3,461 fish/ha with the highest densities (1,454.9 fish/ha) of fish corresponding to age-0 fish (Table 10). Age-3 kokanee displayed the lowest densities (42.7 fish/ha) among age classes. Overall, total mean kokanee density was 1,801 (1,599 to 2,093) fish/ha. When expanded to a population estimate using the reservoir surface area (1,212 ha) on the survey date, a total of 2,183,301 (1,938,090 to 2,459,519) kokanee were estimated. Age-0 kokanee made up 81% of this total or 1,763,366 (1,526,106 to 2,037,482) fish. Population estimates for remaining age classes are reported in Table 10.

Total kokanee abundance in 2010 has increased 300% since 2009, mostly due to the abundant year class of age-0 fish (Figure 16). Hydroacoustic abundance trend information from 2000 - 2010 shows that age-0 kokanee numbers are at their highest numbers since hydroacoustic surveys began. In contrast, abundance of age-1+ fish increased only slightly, particularly fish >200 mm. Lower numbers of these fish were expected in 2010 as a result of escapement control efforts conducted by IDFG in 2006 - 2008.

DISCUSSION

Hydroacoustic evaluations of the Deadwood Reservoir kokanee population suggest that that the population is responding to the lack of spawning escapement control in 2009. In 2009, egg take weir operations were suspended at Deadwood Reservoir, which generally also operates to control the number of spawners in the mainstem Deadwood River. In addition, control efforts to remove spawning fish from the Deadwood River and other tributaries were in operation from 2006 - 2008. Thus, 2010 age-0 kokanee are the first year class in some time to be produced from a totally unregulated spawning run. Older age classes displayed densities that were likely influenced by previous control measurements which resulted in an overall mean female length of 339 mm, exceeding the management objective minimum length of 325 mm (Figure 17).

In 2009, IDFG stocked approximately 5,000 Chinook salmon fingerlings in hopes that the fish would grow and feed on kokanee. The program continued in 2010, when approximately 7,000 Chinook salmon were again stocked in June. IDFG biologists plan to continue the program as a management tool for kokanee in addition to providing a sport fishery. However, the low densities at which Chinook have been stocked are unlikely to influence the 2010 kokanee year class, given the extremely large estimated abundance. Assuming the 2010 year class results in a large spawning escapement in 2013, IDFG may wish to operate the Deadwood River weir for the duration of the spawning run to limit numbers. In addition it may be necessary to limit or block spawning fish from Trail Creek.

MANAGEMENT RECOMMENDATIONS

1. Continue monitoring the kokanee population in Deadwood Reservoir with mid-water trawling and hydroacoustics and sample spawning fish to estimate mean length in 2011.

2. Stock additional 5,000 fall Chinook fingerling in spring or early summer 2011. Evaluate survival and growth of stocked Chinook salmon in summer 2012, after three full years of stocking.

Table 10. Kokanee fish densities (number/ha) per transect and total abundance estimates calculated by arithmetic and geometric mean densities at Deadwood Reservoir, Idaho on July 14, 2010.

		Fish densities (number / ha)				
Transect	**Transect length (m)**	**Age-0**	**Age-1**	**Age-2**	**Age-3**	**Total**
1	536	1,096	193	115	66	1,470
2	451	2,212	362	159	38	2,772
3	461	1,256	180	120	67	1,622
4	453	1,824	72	77	43	2,016
5	299	811	214	15	62	1,102
6	532	862	244	123	65	1,294
7	402	1,220	359	151	115	1,844
8	442	1,419	140	77	24	1,660
9	1443	1,447	244	117	105	1,912
10	133	2,708	47	132	0	2,887
11	578	1,091	128	53	48	1,320
12	791	3,194	151	44	72	3,462
13	510	1,763	95	43	47	1,948
14	837	1,176	44	52	54	1,326
	Arithmetic Mean (AM)	**1,577**	**177**	**91**	**58**	**1,902**
	90% CI (AM)	**226**	**33**	**15**	**10**	**223**
	Abundance (AM)	**1,911,456 ± 274,326**	**214,155 + 39,832**	**110,484 ± 17,699**	**69,644 ± 11,469**	**2,305,739 ± 269,751**
	Geometric Mean (GM)	**1,455**	**147**	**78**	**43**	**1,801**
	90% CI (GM)	**1,259 to 1,681**	**116 to 187**	**62 to 98**	**28 to 65**	**1,599 to 2,029**
	Abundance (GM)	**1,763,366 1,526,106 to 2,037,482**	**178,519 140,651 to 226,496**	**94,254 74,575 to 119,043**	**51,761 33,925 to 78,651**	**2,183,301 1,938,090 to 2,459,519**

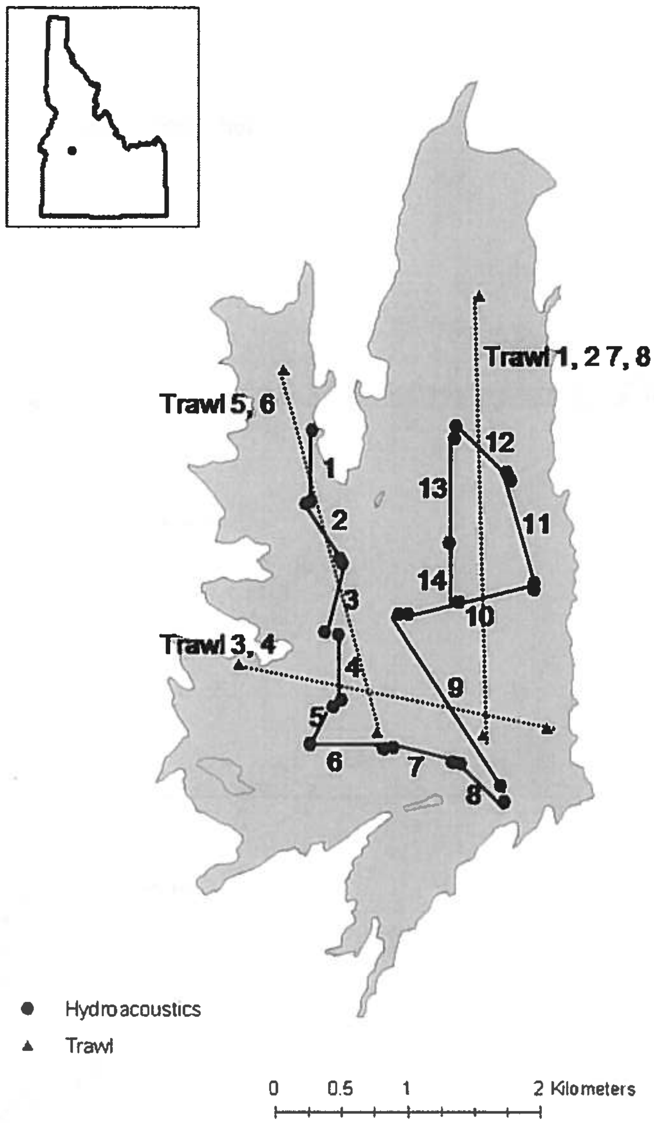

Figure 14. Map of Deadwood Reservoir, Idaho showing mid-water trawling and hydroacoustic transect locations during the 2010 survey.

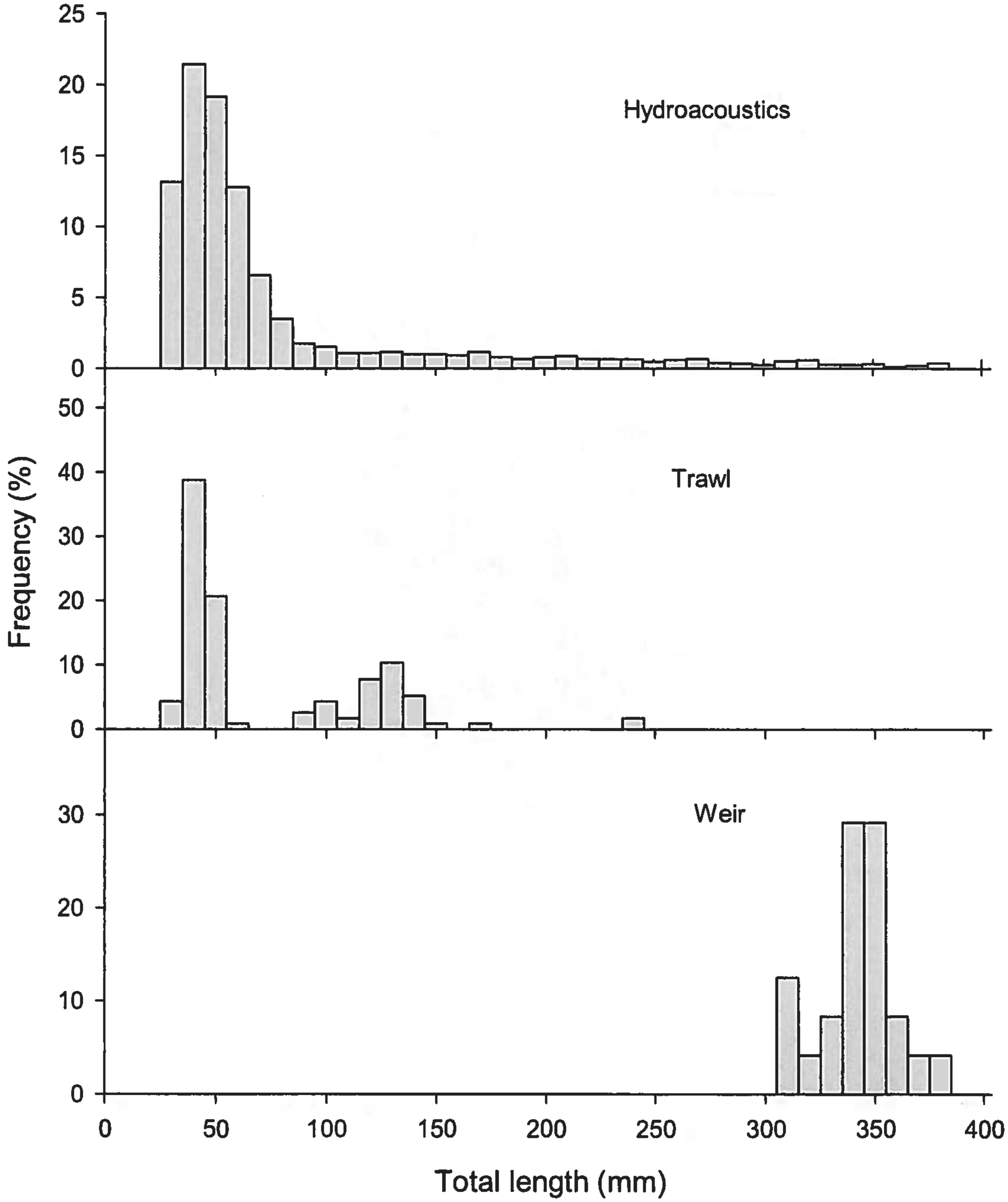

Figure 15. Length frequency of kokanee as estimated by converted hydroacoustic target strengths and length frequency of kokanee captured during mid-water trawling during the survey at Deadwood Reservoir, Idaho on July 13 - 14, 2010.

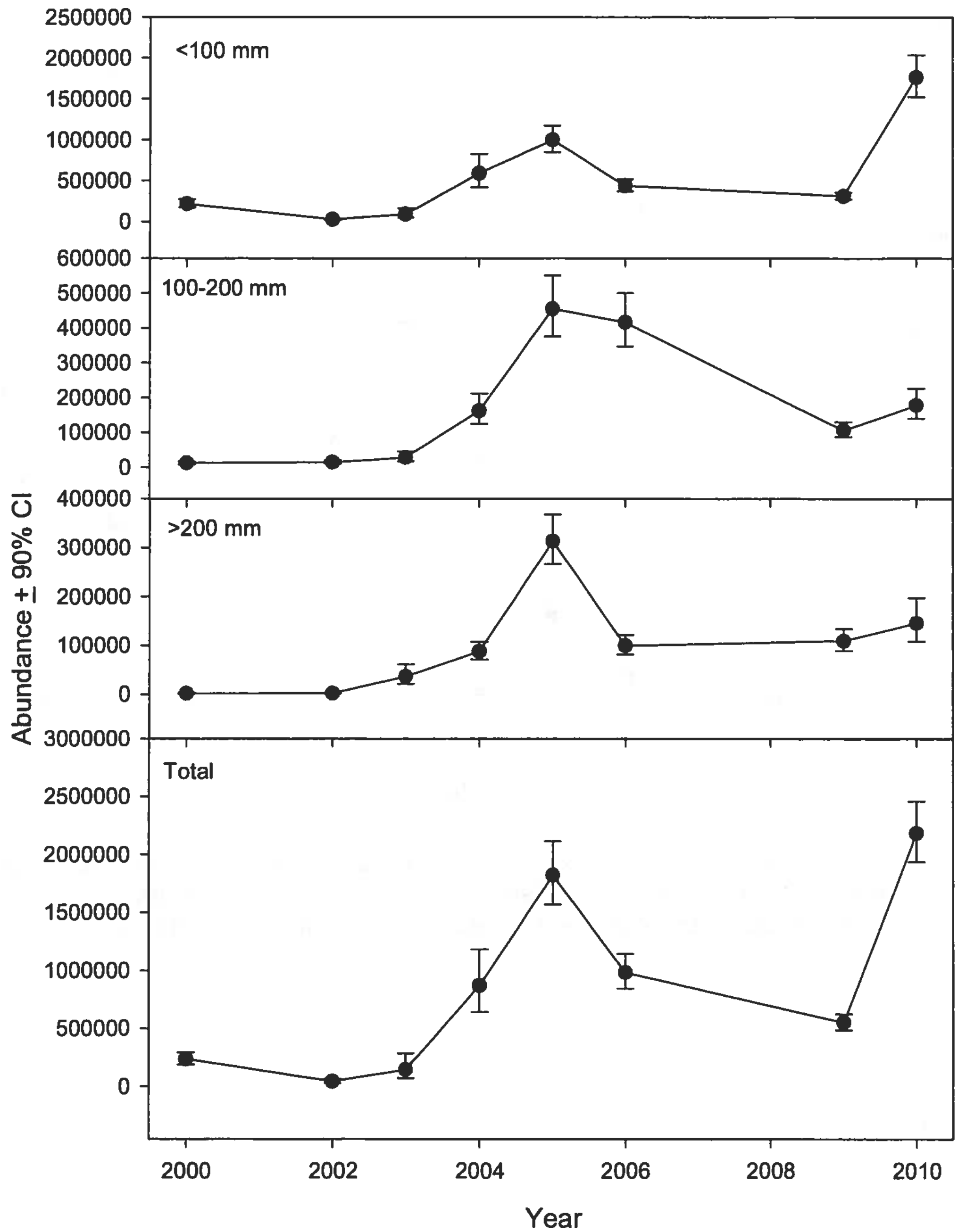

Figure 16. Comparison of kokanee abundance estimates ± 90% CI for fish <100 mm (age-0), 100-200 mm (age-1), >200 mm (age 2+), and total fish as estimated from annual hydroacoustic surveys in 2000 - 2010 at Deadwood Reservoir, Idaho.

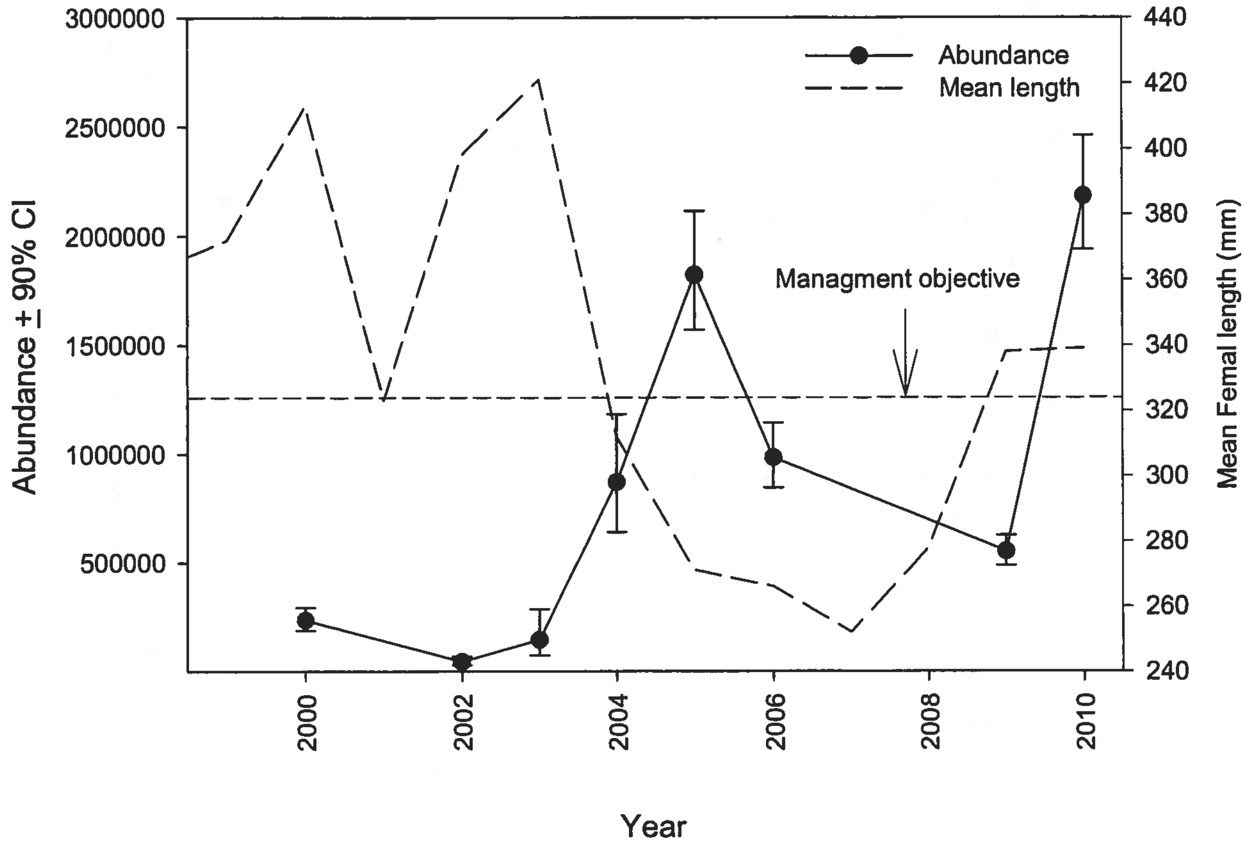

Figure 17. Trend data for 2000 - 2010 hydroacoustic abundance estimates and mean female total length (mm) collected at the Deadwood River trap from 1998 - 2010. The management objective for mean adult length is also shown.

Lowland Lakes Surveys

Lake Lowell - An Assessment of Carp Populations And Control Options For Improving Sportfish Populations, Water Quality, And Aquatic Habitats

ABSTRACT

Recent lowland lake surveys have indicated that the fish community in Lake Lowell has shifted to a nongame fish dominated state, composed primarily of common carp *Cyprinus carpio* and largescale sucker *Catostomus macrochelius*. This shift has negatively affected sportfish populations, water quality, and aquatic habitats. During 2010, we initiated efforts to gain a better understanding of the carp population with the intention of determining effective control options. A mark-recapture estimate indicated that the lake contained 1.2 million carp or about 5 million pounds of carp. These numbers and biomass place Lake Lowell at the high end of the spectrum compared to estimates for other waters. Assessment of ages, age-frequency, and a catch curve indicate that carp in Lake Lowell are relatively young, exhibit frequent successful reproduction (annual), and have high natural mortality rates. Because of these characteristics, which are unlike most other carp populations in North America, and the size of Lake Lowell, manual control is not a realistic option. Instead, future control efforts should focus on biological options such as the introduction of carp diseases or chemical control timed to coincide with an extremely low-water year to minimize cost.

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INTRODUCTION

Lake Lowell is a 3,640 ha irrigation reservoir located 10 km southwest of Nampa, Idaho. The reservoir was built from 1906 to 1909 by forming four embankments around a naturally-occurring low-lying area. Shortly thereafter, the lands surrounding the reservoir were incorporated into the National Wildlife Refuge system. Refuge lands, including public access to Lake Lowell, continue to be managed by the U. S. Fish and Wildlife Service. Fisheries management activities, such as rules setting, fish population monitoring, and fish stocking, are directed by and fall under the statutory authority of IDFG. Uniquely, no streams or rivers flow into the reservoir; instead, water is supplied by the New York Canal which diverts water from the Boise River. The reservoir is fairly shallow with a maximum depth of 12 m. Much of the littoral zone is occupied by extensive beds of smartweed (*Polygonum* spp.).

Lake Lowell is managed with IDFG general fishing regulations, except for largemouth bass which are managed under a no harvest regulation from January 1st through June 30th and a 2 fish, 305 - 406 mm protected slot limit thereafter. Furthermore, the Deer Flat National Wildlife Refuge restricts motorized-boating access from October 1st through April 14th. Approximately 17,000 fishing trips were taken on Lake Lowell during 2006 and anglers generated $457,000 in economic activity (Willard et al. 2007), relatively low amounts of pressure and economic activity for a reservoir of this size, especially considering its proximity to Idaho's population centers. For instance, C.J. Strike Reservoir (3,035 ha) generated four-times more trips and nearly 20-times more economic activity during 2006. In Lake Lowell, largemouth bass receive the majority of the attention from recreational and tournament anglers. High catch rates for largemouth bass are possible at times, but condition has been poor, growth rates have been slow, and large fish have appeared to decline in abundance recently (Kozfkay et al. 2009). Channel catfish are one of the few other sportfish in the reservoir. Recent plants of 6,000 to 9,000 fingerling channel catfish annually have created a healthy, yet underutilized, population. Natural recruitment of channel catfish is very limited. Panfish fisheries (black crappie, bluegill, and yellow perch) are also popular despite widely fluctuating population abundances that have led to inconsistent use. Prior to winter 1990-1991, panfish populations were more robust (Pollard 1974; Grunder et al. 1993), but after a series of very low-water years populations declined possibly due to partial winter kills (Allen et al. 1998). We speculate that common carp and largescale sucker populations (hereafter referred to as carp and sucker) increased in abundance after these events (Bajer and Sorenson 2009), due to declines in the number of egg, larvae, and juvenile predators, and carp have since depressed other more desirable fish populations.

Recent fish population surveys have indicated that the Lake Lowell fish community has become dominated by rough fish. Weight per unit effort indices indicated that carp (49%) and sucker (27%) compose the majority of the fish biomass (Kozfkay et al. 2009). In other systems, highly-abundant rough fish populations, especially carp, have degraded water quality, altered food webs, and negatively impacted native or recreationally important fish populations (Zambrano et al. 2001; Jackson et al. 2010). Carp are benthic omnivores and feed primarily on aquatic invertebrates by rooting in sediments (Panek 1987). This feeding behavior increases turbidity by re-suspending sediments leading to lower light penetration. Additionally, nitrogen and phosphorus are re-distributed in the water column which may facilitate nuisance algae blooms further reducing light penetration (Moss et al. 2002). Increased turbidity at high magnitudes decreases foraging efficiency for sight-oriented piscine predators, reduces beneficial primary productivity, and aesthetic values of lakes and reservoirs. Furthermore, the feeding behavior of carp destroys the root system of aquatic plants, which decreases food or cover for invertebrates, juvenile fish, and waterfowl (Crivelli 1983; Bajer et al. 2009).

To compound these problems, carp populations are able to reach high abundance and biomass compared to most other species. Adult females are highly fecund and are capable of producing well over a million eggs (Crivelli 1981). Eggs are broadcast over submerged vegetation while being fertilized by groups of males. In north temperate zones, spawning occurs during May typically at temperatures between 17 to 24 °C (Richards 1958; Phelps 2006). Despite high fecundity, successful carp reproduction in some systems is sporadic and strong year classes may only be created once every several years, possibly due to egg predation by other fishes or unfavorable climatic conditions (Bajer and Sorenson 2009; Phelps 2006). However, once a strong year class becomes established, carp are long lived and a year class may dominate a system for decades. Carp grow quickly and are capable of reaching 15 cm or larger by the end of their first year. Maximum size of carp in Idaho exceeds 80 cm and 20 kg. Because of their high fecundity and growth potential, carp populations can achieve some of the highest biomasses in freshwater. Densities of carp average 120 kg/ha, but may exceed 480 kg/ha in productive systems (Panek 1987). At densities of these magnitudes, carp populations can cause extensive damage to aquatic resources.

Control efforts designed to reduce or eliminate populations of undesirable fish species such as carp and suckers are commonly initiated by fisheries managers to improve populations of recreationally important species. Long-term assessment of sportfish responses to control efforts are rare. Successful control efforts have led to reduced turbidity, lower nitrogen and phosphorus levels, as well as recovery of native aquatic plant communities leading to better fish and waterfowl habitats (Moss et al. 2002). Control efforts can be categorized as biological (predator stocking or disease), chemical (piscicides), and physical (netting, seining, movement barriers; Wydoski and Wiley 1999). Control efforts that incorporate multiple techniques and function as part of a broader pest management program are often more effective than simple efforts (Meronek et al. 1996).

A series of carp studies and control efforts were conducted on Lake Lowell during the 1950s (Culpin 1956; Richards 1956; Richards 1958). A large commercial harvest of carp (214,000 kg) occurred during fall-winter 1952 - 53. Studies began during 1955 and a mark-recapture population estimate indicated that the population was composed of about 1.24 million individuals (> 36 cm) that averaged 1.5 kg. In the 1955 population estimate, a total of 4,342 carp were marked and during recapture efforts 11,458 carp were examined of which 39 possessed marks (Richards 1956). Arbitrarily, an annual removal goal of 180,000 kg was set (~9.5% of estimated population biomass), based on doubling of the average annual commercial harvest of the preceding five years. Estimated removals for 1955-56, 1956-57, and partial season 1957-58 were 12,400 kg, 31,100 kg, and 45,500 kg, respectively. The target removal was not achieved during these efforts due to high water levels and poor market conditions. Sportfish and water quality responses were not documented as target removal levels were not reached and resources were shifted to other projects.

MANAGEMENT GOAL

To increase the abundance, recruitment, and growth rates of recreationally-important sportfish populations in Lake Lowell by reducing carp and sucker abundance with a science-based control program.

OBJECTIVES

1. Describe life history characteristics, seasonal movement patterns, and abundance of carp in Lake Lowell through research efforts.

2. Determine techniques that would be optimal for removing high abundances of carp and sucker. Optimal techniques would be cost effective, efficient from a manpower perspective, and could be performed with limited disturbance to other species.

METHODS

Successful carp control requires an understanding of population size and dynamics. In order to estimate population size, we initiated a mark-recapture study during fall 2010. We set mark and recapture target goals of 5,000 carp each. We based these targets on an assumed population of approximately one million individuals (based on the 1955 population estimate and extrapolated carp biomass estimates for other waters) as well as our desire to have an estimate with a 50% error bound (Robson and Regier 1964). For our marking efforts, we captured fish in Merwin traps (Figure 18) baited with corn. Two to four traps were fished continuously from September 8 through November 5, 2010. Traps possessed 10 - mm mesh and were 3.5 or 5 m in depth (two of each). Traps were attached to floating pontoons and anchored offshore. Depending on site depths, 30 - 100-m long leads were attached and oriented perpendicular to shore, stretched, and secured to shore or the lake bottom with a fence post. All traps were checked on a two- or three-day rotation, occasionally stretching to four days during inclement conditions. We attempted to disperse trap locations throughout the reservoir; however, adequate trapping locations (i.e. deep water near shore) were limited to only a few areas along the northern shore (Figure 18). All newly-captured carp and sucker were marked by excision of the left pelvic fin and released 2-3 km from the capture site. We measured and weighed a subset of newly captured carp and sucker and all recaptures, until sample sizes were sufficient to allow counting.

Recapture efforts utilized two additional gear types and were conducted shortly after marking efforts were completed. A 9.1 m deep purse seine was used to sample open water areas of the reservoir. This net was constructed with square 10 mm mesh and encircled an area of approximately 0.27 ha, assuming the 184 m net was set in an approximately circular fashion. Crews of 6 - 7 people using two boats set, pursed, and hauled the net in 50 minutes. Although the net was 9.1 m deep, we fished the net in waters as shallow as 7.5 m. Over an approximately two-week period during December 2010, 21 hauls were made. Attempts were made to quasi-randomly sample throughout portions of the reservoir exceeding 7.5 m in depth. The second recapture method involved examining carp caught by local commercial fishermen. Commercial fishermen utilized a 184-m drag seine, along with two; 50 m tow ropes and two boats. The net was set near the middle of the reservoir and stretched straight parallel to shore. Two tow boats then pulled the net into a U-shape and towed it towards the north shore in a perpendicular direction. After reaching shore, the ropes and net were pulled in manually. Captured fish were loaded onto one of the boats and sorted into live crates. All carp and sucker were counted and examined for marks, and a subset was measured and weighed (along with all recaptures). A total of 14 trawls were made during December 2010.

We used the Chapman's modification of the Peterson model for two-event mark-recapture experiments to estimate population size. Population size ($\hat{N}$) was estimated as:

$$\hat{N} = \frac{(M+1)(C+1)}{(R+1)} - 1$$

where M = number of fish caught, marked, and released in first event, C = number of fish captured during second event, and R = number of marked fish captured during the second event.

We collected common carp for determining age structure of the population with Merwin traps after target marking goals had been reached. Dorsal spines were collected from 187 carp during November (Jackson et al. 2007). We collected up to 10 individuals per 10 mm length interval. In addition, we included four small carp caught during commercial seining. Spines were air dried, and then sectioned using a low-speed rotary saw. Dorsal spine sections were placed on a slide, immersed in oil, and imaged at a magnification of 20X. Ages were assigned independently by two readers by counting annuli. Subsequently, images of dorsal spine sections for which a consensus age could not be reached independently, were re-aged jointly to resolve disagreements. A winter 2010 - 2011 annulus was not apparent yet, and was not counted. Age-length keys were constructed to develop an age frequency from which we estimated mortality rates.

Determining seasonal movement patterns of carp may be an important aspect of control efforts. Carp may become vulnerable seasonally as they congregate in spawning, feeding, or wintering areas (Johnsen and Hasler 1977; Penne and Pierce 2010). Carp were collected from as wide of a geographical area of the reservoir as possible using boat electrofishing equipment and Merwin traps. Captured fish were weighed and measured. We surgically implanted acoustic tags in 31 adult carp during early October 2010. All transmitters (Model CT-82-2-I) were manufactured by Sonotronics, Inc. (Tucson, AZ) and expected to emit for 14 months. Surgical methods were similar to Penne and Pierce (2008), except we used MS-222 for anesthesia. Carp were echo-located along 12 transects on a bi-weekly basis. After each fish's location had been determined using triangulation, we recorded location coordinates with a hand-held global positioning system unit, and lake depth, as well as surface water temperature. Maps of carp distribution were created on a bi-weekly basis using GIS software to visually assess the location where carp tended to congregate.

RESULTS

During marking efforts, we captured 6,289 common carp utilizing 206 Merwin-trap days and clipped their left ventral fin. CPUE for individual traps ranged from 0.25 unmarked carp/net day to 159 unmarked carp/net day and averaged 33 unmarked carp/net day. CPUE was very low during the first week of trapping; however, after trap locations were improved and bait was utilized, CPUE exceeded 10 unmarked carp/net day for subsequent sets. Mean length and weight (± 90% CI) were 517 mm (±3; n = 947) and 1,699 g (±31), respectively (Figure 19). Capture of fin-clipped carp in Merwin traps (not part of recapture effort) increased with the duration of a set at a particular site and was surprisingly high considering release locations greater than 2 km from trap sites. By the last week of Merwin trapping, 31% (mean of four traps) of carp examined had been marked previously, indicating strong site fidelity or trap happiness. We also marked 977 largescale suckers. Mean length and weight were 512 mm (± 3; n = 270) and 1248 g (± 27).

Recapture efforts were initiated approximately one month after marking efforts concluded and utilized two gear types. Utilizing a purse seine, we captured 1,400 carp, including seven recaptures with 21 hauls. CPUE ranged from one to 521 carp/purse seine haul and averaged 50 carp/purse seine haul. Mean length of captured carp with the purse seine was 505 mm (± 5; n = 265). In addition, the commercial fishermen captured 4,674 carp during 14 drag seine hauls, including 24 recaptures. CPUE ranged from 27 to 1,340 carp/drag seine haul and averaged 359 carp/drag seine haul. Mean length of carp captured with the drag seine was 498 mm (± 3; n = 425). Combining these gears, we examined 6,074 carp for marks during recapture efforts and noted 31 marked carp. Mean length of recaptured carp was 496 mm (±18; n = 25). From these data, we calculated population estimates using the Chapman estimator. Assuming no mortality of marked fish, carp population abundance equaled 1,194,118 (± 340,174; 90% CI). Population density equaled 557 kg/ha (497 lbs/acre), assuming full pool surface area. It is likely that some mortality of marked carp occurred (due to confinement, holding, and marking stresses), although we did not specifically estimate this parameter. To understand how mortality of marked carp could have affected abundance and density, we also estimated abundance assuming 10% mortality of marked fish. Under this scenario, population abundance and density decreased only slightly to 1,074,725 (± 306,075) and 501 kg/ha (447 lbs/acre), respectively. Catch of largescale sucker during recapture efforts was poor (n = 199). Only one sucker was a recapture, which prevented us from estimating population size.

Carp ranged in age from 1 to 15 years. Carp grew very quickly through age-3 ($\bar{x}$ = 402 mm), but slowed afterwards. Average length of age-4 carp ($\bar{x}$ = 476 mm) was only 90 mm less than the average length of age-9 carp ($\bar{x}$ = 566 mm; Figure 20). Analysis of age-frequency plots indicated that carp were not fully recruited to our gears until age-6 (Figure 21). Regression analysis of frequency (natural log) versus age produced a line with the slope of -0.56 (i.e. Z, the instantaneous annual mortality rate; Figure 22). Mortality rate (A) equaled 0.43 (± 0.08; 95% CI).

We recorded 128 locations of carp during October, November, and December 2010. Individual fish were located from zero to six times with a mean of 3.9 times per tagged carp. Average lake depth for located fish increased in each of three months of monitoring and was 3.3 m in October, 4.8 m in November, 5.5 m in December. Three of the tagged carp were never located after release and an additional fourth fish is assumed to be a mortality at this time due to no movement. Of the remaining 124 locations, 57 locations occurred in the eastern third of the reservoir, 20 were from the middle of the reservoir, and 47 were from the western third (Figure 23). Movement of carp between sections (western, middle, eastern) of the reservoir was relatively rare during this time frame. Twenty-two tagged carp were located in only one section of the reservoir. Seven tagged carp were located in two sections of the reservoir, and only one tagged carp was located in all three sections. Two distinct movement patterns were evident. Seven carp were located frequently near the Caldwell boat ramp/lower embankment, and as water cooled most moved to deeper portions of the reservoir west of the equalizer. A second cluster (n = 11) was located only east of Gotts Point, and as water cooled carp moved northwest to where the no-wake buoy lines intersects the north shore. No discernible movement pattern could be described for 12 other tagged carp, although most moved to deeper areas of the reservoir as water temperature decreased.

DISCUSSION

Population abundance and density estimates indicated that Lake Lowell supports a abundant population of common carp. Carp density in other waters has ranged from 120 to 480 kg/ha (Panek 1987). Our density estimate (557 kg/ha) exceeded the high of this range by 16%. Only one other population and density estimate is available for carp in Lake Lowell (Richards 1956). During the 1950s, the carp population was very similar in abundance (1.24 million), average size (1.5 kg), and density (518 kg/ha; Richards 1956) to our estimates.

Mark-recapture abundance and biomass estimates for closed populations are based on six assumptions. It is unlikely that our estimate was biased significantly by violation of any of the first four assumptions, (1) demographically or (2) geographically closed population, (3) no marks lost or missed, or (4) marking (pelvic fin excision) does not change behavior or vulnerability to capture. However, it is possible that violation of the random mixing assumption (5) created bias. We were only able to set Merwin traps in six locations along the north shore though due to inadequate trapping depths along the south shore. Therefore, it is possible that we missed groups of carp that utilized south shore areas predominantly. Fortunately, our tracking data indicated that nearly all carp left shallow water areas of the reservoir by December 1, which would have allowed marked and unmarked carp to mix prior to the initiation of our recapture efforts. Violation of the sixth assumption, equal probability of capture, was also possible and was likely the greatest concern for creating bias during this study. Although we were able to mark and release relatively equal numbers of fish in all sections of the reservoir, recapture efforts were focused disproportionately in the middle section of the reservoir, due to inadequate depths (for purse seining), and lack of clean beaches and smooth bottoms (for drag seining) in both the eastern and western sections of the reservoir. Once again, mixing of fish during movement to wintering areas, likely reduced this bias, but it still existed and would have positively biased our estimates. Therefore, our estimates should be considered as maximum values.

Many of the studies examining common carp life-history strategies and population dynamics suggest that common carp grow to large sizes, are relatively long lived, have low natural mortality rates, and reproduce successfully at infrequent intervals. The carp population in Lake Lowell does not seem to fit this mold well. Maximum size of carp in Lake Lowell during this study was 750 mm and 4.3 kg, and relative weights were very low. Carp in nearby waters of similar lengths frequently exceed 10 - 15 kg, under more ideal feeding conditions and lower carp densities. The maximum age documented was 15, and very few fish over age-12 occurred in our samples. Common carp have been reported to exceed age-20 in other waters (Panek 1987). Because of this longevity, natural mortality rates in most systems are assumed to be low, but this parameter is rarely estimated or reported. Carp mortality during this study was higher, 0.42, than we expected and suggests that this population replaces itself at a high rate. Young carp are difficult to sample and therefore, recruitment is rarely assessed directly. So as a surrogate, we plotted age-frequencies. This analysis method suggested that carp recruitment was frequent and consistent based on a flat, declining-right limb and the presence of all age classes under 13. Because of this high natural mortality rate and annual recruitment, suppression of spawning could be considered a more viable option than it is for most other water bodies and populations, though this option would have other drawbacks.

During the relatively short time we monitored movement, carp showed a tendency to spend most of their time in relatively small areas. Well over half of our tagged fish spent all of their time in one section of the reservoir. Within that section, movement was generally restricted to a small area (200 - 400 ha) until water temperatures dropped to less than 2°C. This tendency for small home range was also supported by the high occurrence of previously marked carp in our Merwin traps. Capture of previously marked carp in subsequent trapping days was high even with relatively short soak times and distant release locations, suggesting that carp had preferred territories and returned to them quickly after being moved. By December when surface water temperatures dropped below 2°C, a distinct movement pattern was detected for most carp residing in the eastern and western sections. Most of these fish moved from mid-depth flats, presumably preferred feeding areas to adjacent deeper water areas, forming loose aggregations with other tagged carp.

Carp population density in Lake Lowell exceeded thresholds reported to cause ecological problems in other systems. Jackson et al. (2010) estimated that an ecological threshold was reached in Iowa lakes when catch rates of common carp exceeded 2 kg of carp/ fyke-net night. At higher densities, lakes shifted to a turbid water state, with unbalanced fish populations, and losses of aquatic plant communities. During 2006, the biomass of common carp in trap nets (WPUE 26 kg/net night), a similar gear, were approximately 13 fold higher than this threshold. Bajer et al. (2009) documented that carp populations had limited impacts on aquatic plants and water quality at densities of 30 kg/ha, but when populations increased to 100 kg/ha, negative impacts were noted. Clearly, the density in Lake Lowell far exceeds the range of densities reported as thresholds for detrimental effects. Improvement of fish populations, water quality, and aquatic plant communities will require substantial reduction in carp density.

Chemical, biological, and physical control methods have been used to reduce density of carp and other nongame fish populations. Several physical control methods may be used to control nongame fish populations. Large drag seines (up to 300 m in length) have been effective in other systems (Fritz 1987), and pre-baiting areas with grain further concentrates fish leading to increased effectiveness. On Lake Mattamuskeet, NC, pre-baited areas were drag seined and from 900 to 6,300 kg of carp were captured and removed per haul (Cahoon 1953). Over a four-year period 726,000 kg were removed culminating in improved water quality as well as recovery of sportfish populations and aquatic plants. Utilizing the same gear on Lake Lowell in recent years, one commercial fisher has removed up to 22,700 kg of carp annually and much larger catches were reported from the 1950s. These removals had little or no affect on carp populations. A large manual removal effort is ongoing at Utah Lake with the intent of recovering an endangered sucker population. Target removal goals are 2,268,000 kg annually for six consecutive years with costs exceeding $1,000,000 annually.

Biological control of rough fish is another option. Koi-herpes virus, a pathogen that occurs naturally within the native range of carp, has been imported accidently into the United States. On several occasions, outbreaks have caused substantial die offs of common carp and their close relatives. Purposeful release of this pathogen has yet to be attempted, as no testing has been conducted to determine if non-target organisms are susceptible. If testing is conducted, release of this pathogen may allow cost-effective treatment of large water bodies. Stocking potential predators is another biological control method that could reduce carp and other rough fish populations. Panfish are known predators of carp eggs and larvae. Rebuild spawning stocks of these fishes will be important to maintain predation pressure on carp eggs and young, but may be difficult until carp density is reduced.

Chemical control methods require application of a fish toxicant, usually rotenone, at a concentration (0.2 ppm active rotenone) sufficient to cause mortality. Timing treatments to coincide with low, reservoir-storage periods would be necessary to minimize cost. Low, reservoir-storage periods have occurred recently during a sustained drought period (the early- to mid-1990s) and in the absence of drought conditions would require permission and cooperation of managing entities and irrigators to purposefully reduce reservoir storage. Since 1991, storage in September (typically the lowest volume month of the year) has averaged approximately 25,090 ha; however, during 1992 and 1994, average September storage was approximately 4,451 ha. At these volumes and the recommended treatment concentrations

(~11.25 kg/ha), cost to purchase Prentox (rotenone fish toxicant powder) would be $1,300,000 for 25,090 ha or $231,000 for 4,451 ha, for the chemical alone. Total cost would be substantially higher as these calculations do not include cost of liquid rotenone for treating the perimeter, additional equipment, or man-power costs for planning and application.

MANAGEMENT RECOMMENDATIONS

1. Continue to explore cost-effective alternatives to reducing carp biomass in Lake Lowell.
2. Encourage commercial removal of carp and other nongame fish from Lake Lowell.
3. Continue working with the Irrigation District, U.S. Fish and Wildlife Refuge staff, and local public to build support for restoration of the gamefish populations at Lake Lowell.

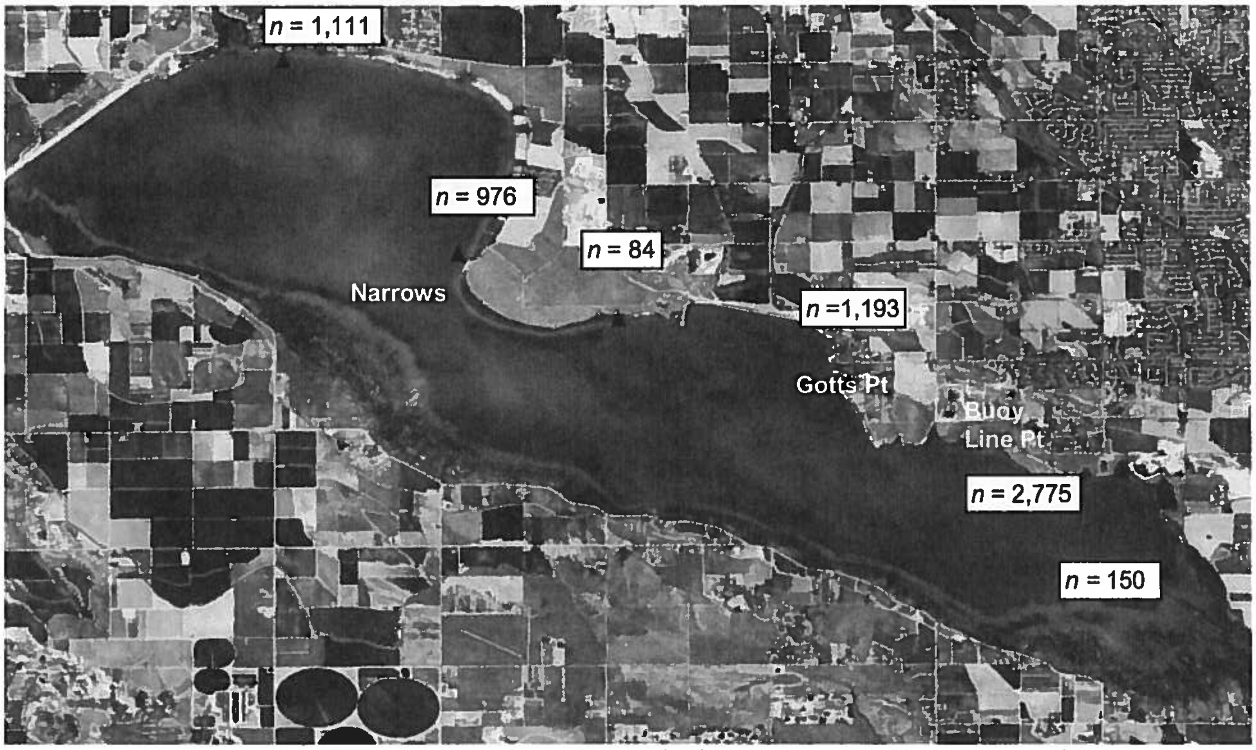

Figure 18. Location of Merwin traps (black triangles) used to capture and mark common carp in Lake Lowell during 2010. Number of fish marked at each trap are displayed near trap markers.

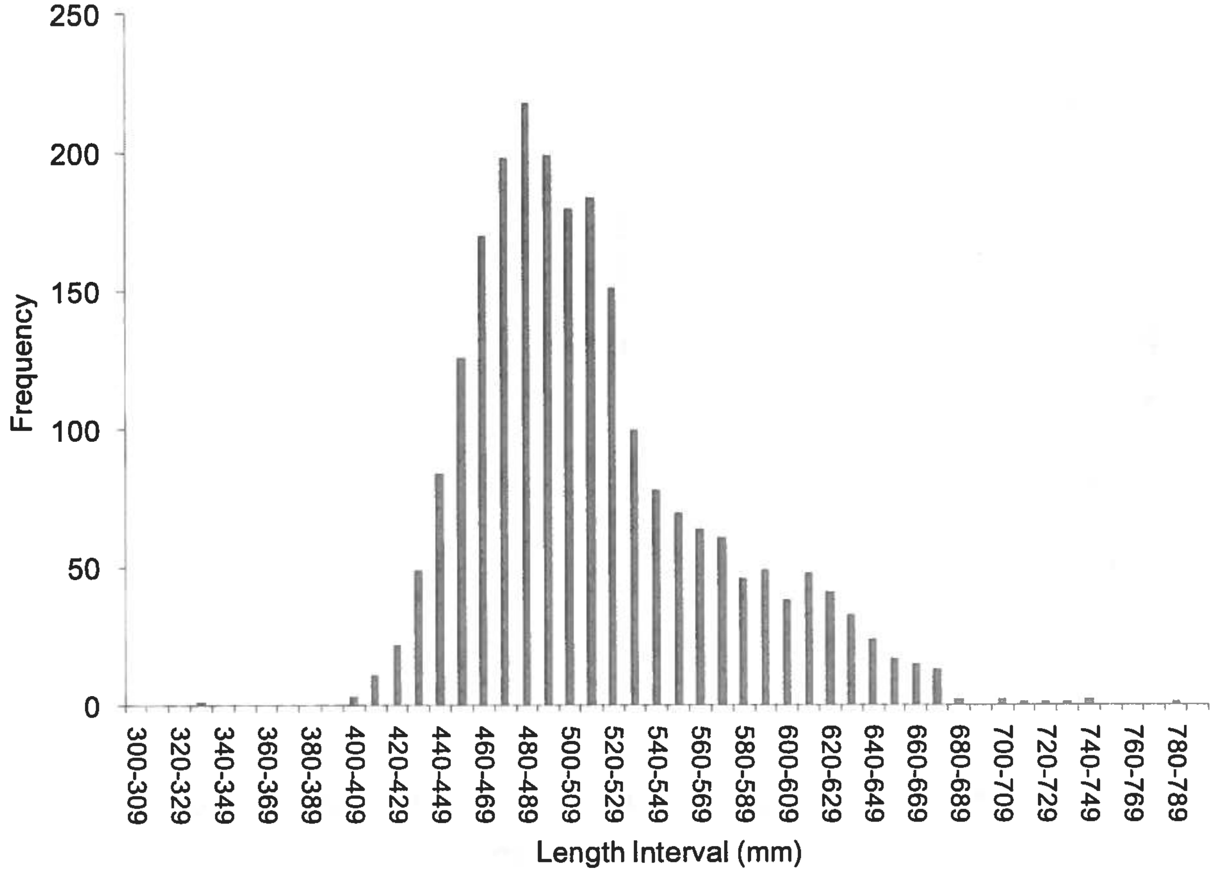

Figure 19. Length frequency (n = 2,303) of common carp sampled from Lake Lowell during fall 2010.

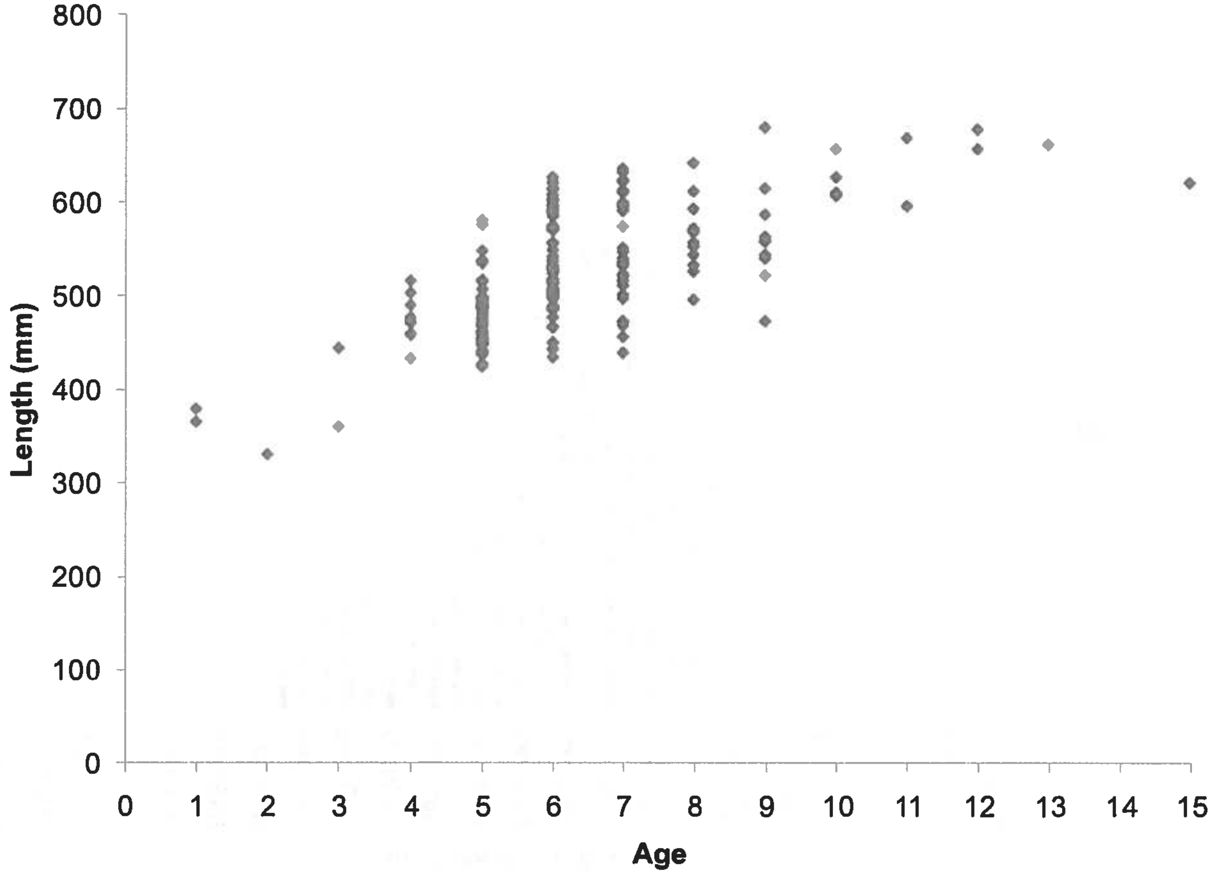

Figure 20. Length at age for common carp (*n* = 182) from Lake Lowell sampled during fall 2010.

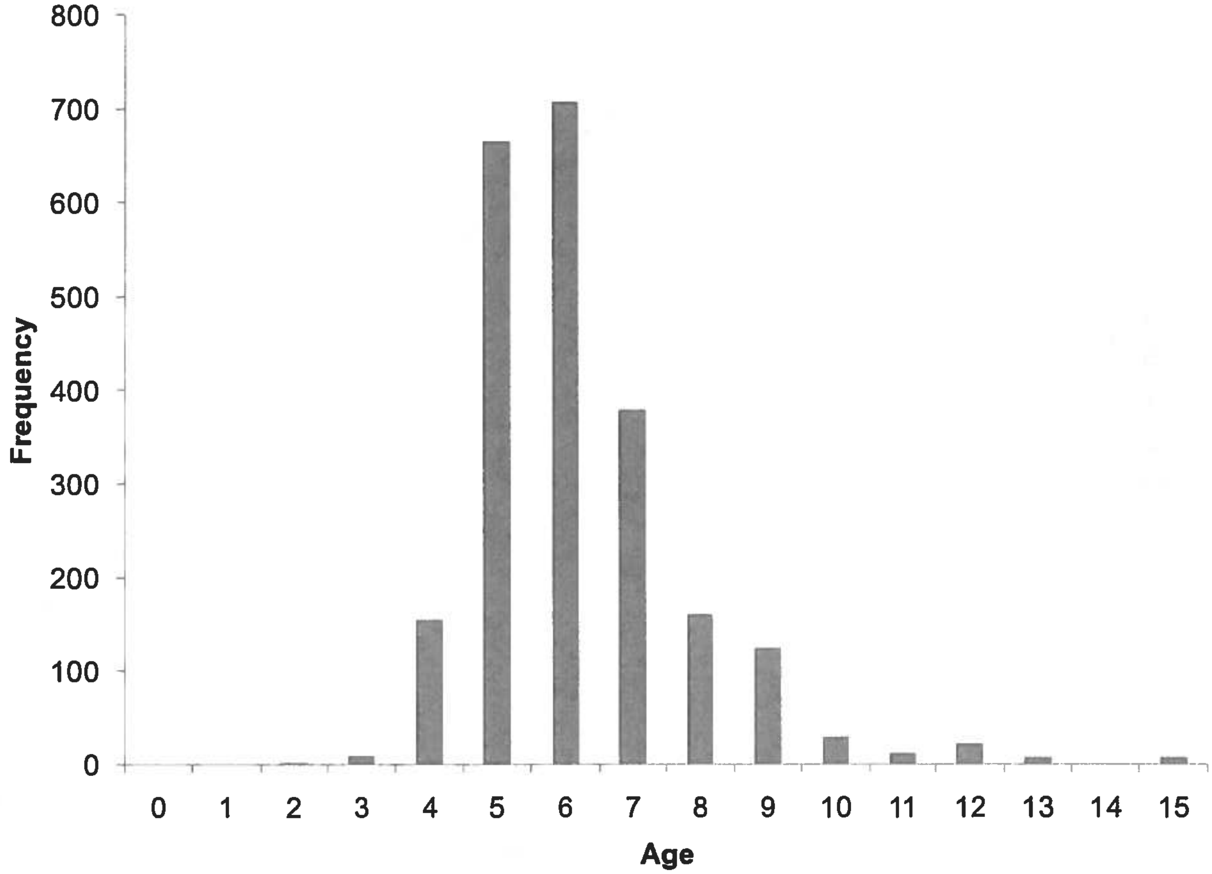

Figure 21. Age-frequency of common carp (n = 2,295) sampled from Lake Lowell during fall 2010.

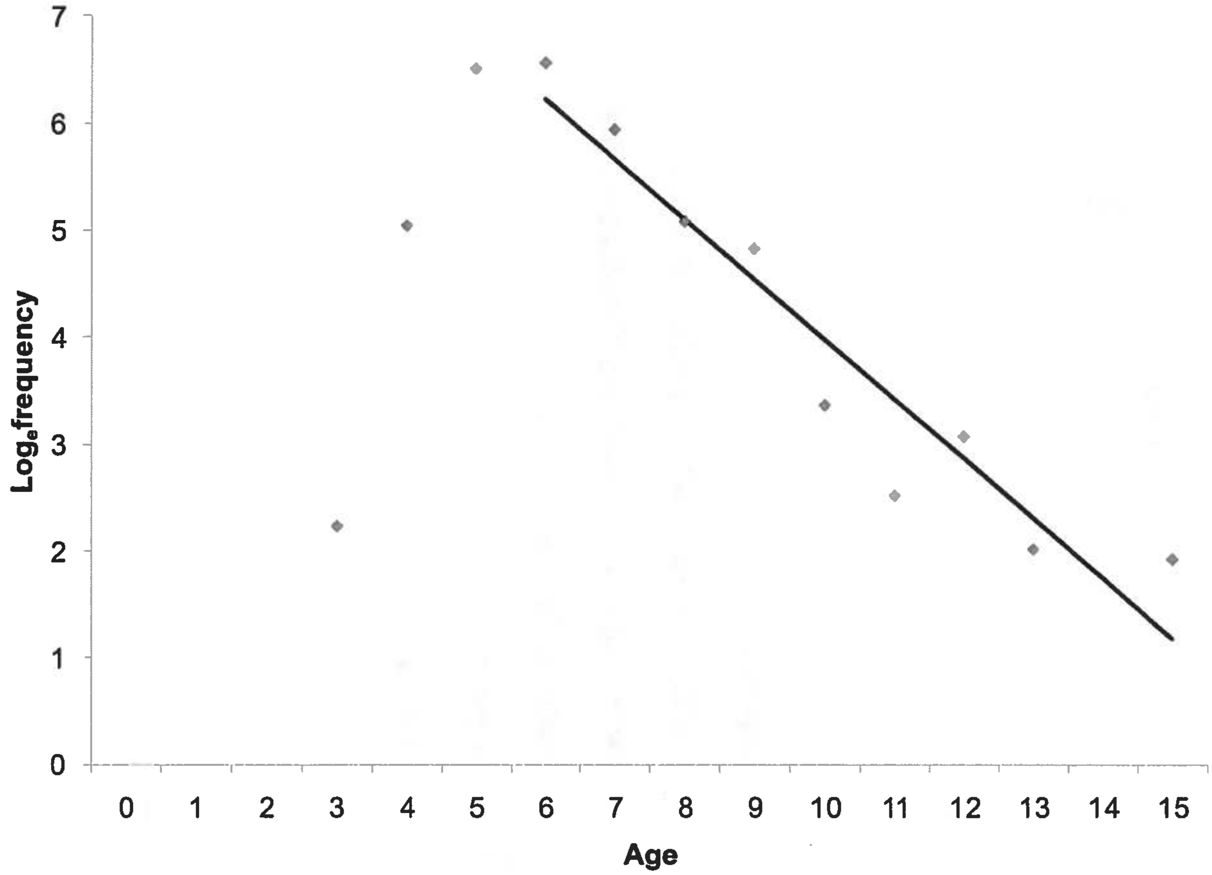

Figure 22. Catch curve for common carp sampled from Lake Lowell during fall 2010.

Figure 23. Locations of carp during October (n = 45), November (n = 54), and December 2010 (n = 29) determined with telemetry.

High Mountain Lakes Surveys

ABSTRACT

IDFG personnel conducted surveys at 99 mountain lakes in the Southwest Region in 2010. The lakes were located in the headwaters of the South Fork Payette River, Queens River, and Upper Middle Fork Boise River drainages. Most of the lakes surveyed were located in remote, steep country and appear to receive little use by recreationalists. Fish, amphibian, and habitat surveys were conducted at each lake. Fish were present in 23 lakes and amphibians were observed in 39 lakes. Westslope cutthroat trout and rainbow x cutthroat hybrids were the most abundant fish species in the lakes, and brook trout were also present in some lakes. The mean condition factor of fish captured in all lakes was 0.99, suggesting fish are in relatively good condition. It is recommended that all lakes within these drainages that currently receive westslope cutthroat trout be changed to triploid rainbow trout for native redband trout conservation reasons.

Amphibians encountered in the surveys included Columbia spotted frogs *Rana pretiosa*, Western toads *Bufo boreas*, and long-toed salamanders *Ambystoma macrodactylum*. Most of the lakes where amphibians were observed do not support fish, and it is recommended that they not be stocked with hatchery produced fish.

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OBJECTIVES

1. Describe the distribution, relative abundance, and species composition of fish and amphibian populations in high mountain lakes of the Southwest Region.

2. Assess factors affecting the distribution, relative abundance, and species composition of fish and amphibian populations in high mountain lakes including stocking strategies, habitat characteristics, and human use.

3. Adjust stocking where appropriate to more efficiently use hatchery resources and to conserve native fauna.

METHODS

We conducted surveys on 99 mountain lakes in the Southwest Region between August 3 and August 25, 2010. Sixty-six lakes were located in the headwaters of the South Fork Payette River drainage (Figure 24), 13 lakes were in the Queens River drainage (Figure 25), and 20 lakes were located in the Upper Middle Fork Boise River drainage (Figure 26). The lakes either had not been surveyed in recent years or had never been surveyed according to IDFG records. At each lake, we assessed fish and amphibian presence/absence, human use, and habitat characteristics. In some lakes that were capable of supporting fish or had previous stocking history, we set Swedish type gill nets that measured 46 m long by 1.5 m deep, with 19, 25, 30, 33, 38, and 48 mm bar mesh panels. One unit of sampling effort was defined as one gill net fished overnight. In other lakes containing fish, hook and line sampling was used to obtain species, length, and weight information. All fish captured were identified to species, measured for total length (mm), and weighed (g). Fulton's condition factor was calculated for each captured fish according to the formula (Anderson and Neumann 1996):

$$K = (100{,}000 \times W)/L^3$$

where W is weight (g) and L is length (mm). For instances when time constraints prevented fish sampling, lakes were visually surveyed during amphibian surveys for the presence of fish.

Habitat surveys consisted of collecting limnological and morphological data in individual lake basins. Lake length was measured across the long axis of each lake using a laser rangefinder (Bushnell Yardage-Pro), and width measurements were recorded at ¼, ½, and ¾ distances along the length axis. Average depth was determined by taking cross-sectional measurements at three points along each width measurement transect using a hand-held sonar device (Strikemaster Polar Vision). Maximum depth was estimated as the greatest depth observed during the cross-sectional measurements. Surface water temperatures were recorded along the lake shore at one point. A visual assessment of spawning habitat availability in each lake and the inlets and outlets was determined based on substrate quality, flow, and gradient.

Amphibian surveys were conducted by walking the perimeter of each lake and noting the abundance and life stages of individual species. Life stages were classified as adult, sub-adult, or larvae. Shoreline habitat adjacent to lakes including areas under logs and rocks were also inspected to detect hidden amphibians.

Human use of mountain lakes was evaluated based on general appearance of use, number and condition of campsites, number of fire rings, access trail condition and difficulty, and presence of litter. General levels of human use were categorized by IDFG personnel as rare, low, moderate, and high based on an overall visual assessment of the factors described above.

RESULTS AND DISCUSSION

IDFG stocking records show 17 of the 99 lakes sampled had been stocked in the past 20 years and surveys confirmed fish presence in 12 of these lakes (Table 11). An additional 76 (77%) lakes were determined to be fishless. We encountered westslope cutthroat trout ,rainbow trout *O. mykiss*, rainbow x cutthroat trout hybrids, brook trout, and Arctic grayling in our surveys (Table 11).

Amphibians were observed in 39 (39%) of the surveyed lakes. Of the 39 lakes with amphibians, 25 (64%) were fishless and 14 (36%) contained fish (Tables 12, 13, and 14). Amphibian species observed included Columbia spotted frog, long-toed salamander, and Western toad (Table 15). We observed neither fish nor amphibians in 51 lakes (52%) surveyed in 2010.

Human use in the area was generally low, but a few lakes which contained fishable populations of trout received moderate use. The long hiking distance required to reach the lakes coupled with steep, rugged terrain likely contributes to the lakes' infrequent use. Results of human use and most habitat characteristics are not reported here, but have been recorded in the statewide IDFG lake survey database.

Survey results from individual and groups of lakes, organized by drainage (HUC 5), are summarized below.

South Fork Payette River Drainage

We surveyed 66 lakes in the headwaters of the South Fork Payette River drainage during August 23 - 27, 2010 (Table 12). Fish were observed in 15 lakes (23%), and 12 of these also contained amphibians. We surveyed 18 lakes (27%) which contained only amphibians and 33 lakes (50%) contained neither fish nor amphibians. Seventeen of the lakes sampled in this drainage were reported as dry or wet meadow bowls. For the lakes surveyed in 2010, Ardeth, Benedict, Vernon, and Virginia lakes appear to be the most frequented lakes by anglers, all of which have been stocked previously by IDFG.

Ardeth Lake was surveyed on August 23 -24, 2010. Ardeth Lake was last stocked by IDFG with westslope cutthroat trout in 2004 and has since been removed from the stocking rotation. Two overnight gill net sets caught 50 brook trout ranging from 135 - 355 mm in total length, a mean length $\pm$ 90% CI of 257 $\pm$ 10.5 mm (90%CI), resulting in a CPUE of 25 fish. The mean condition factor for fish captured in Ardeth Lake was 1.1 $\pm$ 0.03 (90% CI; Figure 27). Brook trout are naturally reproducing in the lake and in some of the tributaries and appear to be in good condition although the mean size of fish is somewhat small. No westslope cutthroat trout were observed in 2010. No amphibians were observed in Ardeth Lake. It is uncertain if this is an effect of an established fish population, lack of suitable amphibian habitat, or a combination of factors. The lake contains shallow habitat with vegetation and logs in a number of areas that should provide refuge for amphibians from fish predators.

Benedict Lake was surveyed on August 24, 2010. Benedict Lake has a stocking history of cutthroat trout and rainbow x cutthroat hybrids in 1989 and 1990, respectively. The lake was last stocked in 2009 with westslope cutthroat trout. Hook-and-line sampling resulted in 10 rainbow x cutthroat hybrids ranging from 155 - 270 mm, a mean length $\pm$ 90% CI of 217 $\pm$ 23 mm, and a mean condition factor of 1.02 $\pm$ 0.13 (Figure 27). A single adult and 20 juvenile Columbia spotted frogs were observed during a survey around the lake perimeter. Sampling suggests that natural recruitment is occurring in the lake and additional stocking of Benedict Lake is not needed.

Vernon and Virginia lakes were surveyed on August 24, 2010. Vernon Lake was most recently stocked with triploid rainbow trout fry in 2010 prior to the survey, whereas Virginia Lake was stocked in 2009 with westslope cutthroat trout. Hook-and-line sampling was unsuccessful at Vernon Lake, but produced three westslope cutthroat trout ranging from 220 - 300 mm, with a mean condition factor of 1.12 $\pm$ 0.01 at Virginia Lake (Figure 27). Despite having not been stocked since 1994, westslope cutthroat trout were observed at Vernon Lake swimming near shore during a visual survey around the lake perimeter. Columbia spotted frogs were observed in moderate abundance at both lakes (Table 15). Natural recruitment appears to be occurring at both lakes and stocked fry are likely not contributing to fish populations. Therefore further stocking should be discontinued.

Fish were not observed in 5 of the 15 lakes that have been previously stocked by IDFG but two of the apparent fishless lakes, Ten Lake Creek lakes #5 and #7 were stocked with triploid rainbow trout fry in 2010 and fish may have been missed due to their small size (Table 11). Benedict Lake #2, and Ten Lake Creek lakes #3 and #6 did not appear to contain any fish. Ten Lake Creek lakes #3 and #5 contained depths no greater than 3 m and winter kill is likely at those depths. Amphibians were observed in all of these lakes except Benedict #2 and Ten Lake Creek Lake #3 (Table 15).

<u>Queens River-Middle Fork Boise River</u>

We surveyed 13 lakes located in the Queens River drainage in 2010 during August 2 - 6, 2010 (Table 13). Fish were observed in three lakes and amphibians in one lake. No lakes contained both fish and amphibians, while nine lakes (69%) contained neither fish nor amphibians (Table 13). One lake in this drainage, LLID# 1151355438792, was observed as dry in 2010.

Two lakes, Flat Top Lake and Tripod Creek Lake, are currently on the IDFG stocking rotation. Fish were observed in both of these lakes, as well as Tripod Creek Lake #2, which is adjacent and connected to Tripod Creek Lake by a short stream. A single gill net was set overnight in Flat Top Lake, capturing 18 westslope cutthroat trout, ranging between 170 - 365 mm. Mean length $\pm$ 90% CI was 309 $\pm$ 15.3 mm and mean condition factor (K) was 1.01 $\pm$ 0.06 (Figure 27). Neither of the Tripod Creek lakes was sampled for fish due to time constraints, but rainbow trout were visibly abundant along the shorelines, ranging between 120 - 350 mm. Amphibians were not observed in Flat Top, Tripod Creek, or Tripod Creek lakes during visual surveys around the entire perimeter of these lakes (Table 15).

A single juvenile long-toed salamander was observed at an unnamed lake within the Queens River drainage (LLID# 1151239439157; Table 15).

Upper Middle Fork Boise River

We surveyed 20 lakes in the Upper Middle Fork Boise River drainage during August 23-27, 2010 (Table 14). Fish were observed in five lakes (25%), two of which also contained amphibians. Six lakes (30%) contained only amphibians and nine lakes (45%) contained neither fish nor amphibians. Nine of the lakes sampled (45%) in this drainage were reported as dry or wet meadow bowls.

None of the lakes in the Upper Middle Fork Boise River drainage that were sampled in 2010 had been stocked since 1967 (Table 11). However, fish were observed in LLID# 1150254439428, Little Spangle Lake, Queens River lakes #10 and #11, and Spangle Lake.

Of the lakes surveyed in 2010, Spangle Lake and Little Spangle Lake appear to be the most frequented lakes by anglers. A single gill net was set overnight at Spangle Lake on August 23, 2010 which captured 24 brook trout ranging between 165 - 324 mm total length. Mean length ± 90% CI were 256 ± 16 mm and mean condition factor (K) was 0.93 ± 0.04 (Figure 27). Brook trout are reproducing naturally in the lake and appear to be well established. A single juvenile long-toed salamander was observed during the amphibian survey at Spangle Lake (Table 15).

Hook-and-line sampling captured 14 brook trout in Little Spangle Lake, ranging between 152 - 297 mm. Mean length ± 90% CI was 223 ± 20.5 mm, but equipment malfunction prevented the recording of weights (Figure 27). Both Columbia spotted frogs and long-toed salamanders were observed during the amphibian survey conducted around the entire perimeter of Little Spangle Lake (Table 15). As Spangle and Little Spangle lakes are not currently stocked, no changes to current management are needed.

Fish were also observed in Queen River lakes #10 and #11, and LLID# 1150254439428. In Queen River Lake #10, 14 rainbow x cutthroat hybrids were captured with hook and line with a mean length of 242 ± 12 mm and a mean condition factor (K) of 0.76 ± 0.1 (Figure 27). Fish have not been stocked by IDFG after 1967 and fish appear to be reproducing naturally. In Queen River Lake #11, a single rainbow x cutthroat hybrid trout approximately 600 mm in length was observed near shore. No other fish were observed and it may be possible that fish can migrate between lakes. Finally, unnamed lake (LLID# 1150254439428) has decreased to a wet meadow with meandering stream channels that were abundant with fingerling brook trout. It is likely under wetter conditions that a lake will form again. We recommend that these lakes remain off the stocking rotation as they are rarely visited and already have self sustaining populations.

MANAGEMENT RECOMMENDATIONS

1. Shift from stocking westslope cutthroat trout to triploid rainbow trout in Flat Top Lake, Ten Lake Creek lakes #6, #8 and #10.
2. Discontinue stocking trout in Benedict and Virginia lakes as self-sustaining trout populations are present. Remove Ten Creek Lake Creek Lake #5 from stocking rotation because of lack of depth.
3. Maintain all lakes surveyed in 2010 where fish were not present as fishless amphibian habitat.

Table 11. Fish stocking data from lakes where fish were observed during 2010 surveys. The last year each species was stocked in a given lake is listed, and superscript letters denote previous ten-year stocking history. Fish observed refers to species which were encountered by IDFG during surveys conducted from August 2-6, and 23-27, 2010.

		Trout species					
Lake	Map #	Westslope cutthroat	Cutthroat (unspecified)	Rainbow (triploid)	Rainbow/Cutthroat hybrid	Grayling	Fish observed
Headwaters South Fork Payette River drainage							
Ardeth Lake	34	2004					BKT
Benedict Lake	38	2009[1]	1989		1990		RXC
Benedict Lake #2	36		1989				None
Rockslide Lake	41	1994					RXC
Ten Lake Creek Lake #03	45			2002		2002	None
Ten Lake Creek Lake #05	54	2008[2]		2010[*]			None
Ten Lake Creek Lake #06	6	2008				1992	None
Ten Lake Creek Lake #07	58	2008[2]		2010[*]			None
Ten Lake Creek Lake #08	59	2008				1993	WCT
Ten Lake Creek Lake #09	60	2008[2]		2010[*]			WCT
Ten Lake Creek Lake #10	42	2008[2]					WCT, RXC
Ten Lake Creek Lake #11	43					2002	WCT, ARG
Three Island Lake	61	2008[2]		2010[*]			WCT, RXC
Vernon Lake	63	1994		2010[*]			WCT
Virginia Lake	64	2009[1]					WCT
Queens River drainage							
Flat Top Lake	72	2008[2]					WCT
Tripod Creek Lake	77	2008[2]					HRBT

[1]Fry stocked every 2 years since 1999

[2]Fry stocked every 3 years since 1999

[*]Programmatic change to triploid rainbow trout

Table 12. Map number, elevation, area, maximum depth, and fish and amphibian presence/absence results for lakes surveyed in the South Fork Payette River drainage during August 23-27, 2010. The LLID # is used to identify unnamed lakes.

Lake name / LLID#	Map #	Elevation (m)	Area (ha)	Maximum depth (m)	Fish observed	Amphibians observed
1149891439668	1	2884	0.17	Dry	No	No
1149923439672	2	2882	0.09	1	No	Yes
1149926439666	3	2885	0.19	1	No	Yes
1150009439541	4	3031	0.98	1.5	No	No
1150025439547	5	3031	0.22	Dry	No	No
1150072439623	99	2856	0.15	Dry	No	No
1150089439616	98	2830	0.15	Dry	No	No
1150127439591	6	3013	0.05	0.3	No	Yes
1150128439586	7	3013	0.10	1	No	No
1150132439582	8	3013	0.06	0.4	No	No
1150164439377	9	2774	1.42	Dry	No	Yes
1150186439621	10	2842	0.38	0.5	No	No
1150219439649	11	2901	0.06	1	No	Yes
1150238439589	12	2901	0.27	0.5	No	No
1150249439562	13	2927	0.81	Dry	No	No
1150253439586	14	2901	0.09	Dry	No	No
1150476439602	15	3062	0.19	Dry	No	No
1150493439610	16	3037	0.17	Dry	No	No
1150565439594	17	2844	0.27	Dry	No	No
1150604439575	18	2851	0.04	0.5	Yes	Yes
1150606439661	19	2808	0.26	Dry	No	No
1150614439553	20	2862	0.04	0.3	No	Yes
1150635439730	21	2793	0.71	0.67	No	Yes
1150642439588	22	2889	0.27	0.3	No	Yes
1150642439604	23	2899	0.95	2	No	Yes
1150647439575	24	2901	0.07	1.1	No	Yes
1150694439738	25	2659	0.31	0.33	No	Yes
09-U139.00	26	3038	0.37	Dry	No	No
09-U150.00	27	2904	0.12	3	No	Yes
09-U153.00	28	2903	0.20	Dry	No	No
09-U154.00	29	2900	0.32	0.3	No	No
09-U158.00	30	3034	0.19	Dry	No	No
10-U143.00	31	3103	0.11	Dry	No	No
10-U144.00	32	3077	0.09	Dry	No	No
10-U152.00	33	2904	0.08	3	No	Yes

Table 12. Continued.

Lake name / LLID#	Map #	Elevation (m)	Area (ha)	Maximum depth (m)	Fish observed	Amphibians observed
Ardeth Lake	34	2805	31.75	17	Yes	No
Arlin Lake	35	3008	1.60	5.6	Yes	Yes
Benedict Creek Lake #2	36	2872	2.04	10	No	No
Benedict Creek Lake #3	37	2857	1.70	8	Yes	Yes
Benedict Lake	38	2814	4.20	6	Yes	Yes
Heather Lake	39	2800	0.46	1.5	Yes	Yes
Little Vernon Lake	40	3005	1.27	6	No	Yes
Rock Slide Lake	41	2952	3.11	8	Yes	Yes
Ten Lake Creek Lake #10	42	3034	2.19	7	Yes	Yes
Ten Lake Creek Lake #11	43	2900	4.02	5	Yes	Yes
Ten Lake Creek Lake #2	44	2899	0.17	2	No	Yes
Ten Lake Creek Lake #3	45	2900	1.39	3	No	No
Ten Lake Creek Lake #4	46	2995	0.80	4	No	No
Ten Lake Creek Lake #41	47	3021	0.14	0.4	No	No
Ten Lake Creek Lake #42	48	3019	0.16	1	No	No
Ten Lake Creek Lake #43	49	3023	0.09	0.4	No	No
Ten Lake Creek Lake #43A	50	3012	0.09	0.5	No	No
Ten Lake Creek Lake #43B	51	3020	0.09	0.8	No	No
Ten Lake Creek Lake #44	52	3010	0.11	0.3	No	No
Ten Lake Creek Lake #45	53	3010	0.07	1	No	No
Ten Lake Creek Lake #5	54	3007	0.57	3	No	Yes
Ten Lake Creek Lake #53	55	3020	0.07	Dry	No	No
Ten Lake Creek Lake #6	56	3011	1.12	7	No	Yes
Ten Lake Creek Lake #6A	57	3012	0.01	1	No	No
Ten Lake Creek Lake #7	58	3012	1.60	11	No	Yes
Ten Lake Creek Lake #8	59	3023	0.97	10	Yes	No
Ten Lake Creek Lake #9	60	3022	3.05	19	Yes	No
Three Island Lake	61	2932	4.57	9	Yes	Yes
Upper Vernon Lake	62	2888	2.39	5	Yes	Yes
Vernon Lake	63	2885	12.71	6	Yes	Yes
Virginia Lake	64	2814	4.76	9	Yes	Yes

Table 13. Map number, elevation, area, maximum depth, and fish and amphibian presence/absence results for lakes surveyed in the Queens River drainage during August 2-6, 2010. The LLID # is used to identify unnamed lakes.

Lake name / LLID#	Map #	Elevation (m)	Area (ha)	Maximum depth (m)	Fish observed	Amphibians observed
1151239439157	65	2833	0.1	1	No	Yes
1151355438792	66	2885	0.16	Dry	No	No
1151447439187	67	2964	1.86	1.75	No	No
1151470439167	68	2977	0.05	0.75	No	No
1151486439194	69	2939	0.79	1.25	No	No
1151516439146	70	2815	5.66	0.5	No	No
Fezip Creek Lake	71	2748	2.52	2	No	No
Flattop Lake	72	2816	2.18	6	Yes	No
Flattop Lake #2	73	2658	0.1	6.3	No	No
King Lake	74	2382	0.26	3	No	No
No Name Lake	75	3020	1.14	6	No	No
Tripod Creek Lake	77	2882	1.33	10	Yes	No
Tripod Creek Lake #2	76	2841	2.96	3	Yes	No

Table 14. Map number, elevation, area, maximum depth, and fish and amphibian presence/absence results for lakes surveyed in the Upper Middle Fork Boise River drainage during August 2 - 6, 2010. The LLID # is used to identify unnamed lakes.

Lake name / LLID#	Map #	Elevation (m)	Area (ha)	Maximum depth (m)	Fish observed	Amphibians observed
1150199439395	78	2800	0.14	2.5	No	Yes
1150205439397	79	2804	0.04	Dry	No	No
1150228439434	80	2912	0.21	Dry	No	No
1150254439428	81	2859	1.72	1.7	Yes	No
1150267439442	82	2870	0.22	Dry	No	No
1150344439369	83	3009	0.35	Dry	No	No
1150351439416	84	2943	0.57	3	No	Yes
1150361439385	85	3007	0.34	Dry	No	No
1150403439446	86	3010	0.31	0.75	No	Yes
1150416439449	87	3020	0.12	Dry	No	No
1150591439434	88	3063	2.92	Dry	No	No
10-U141.00	89	3078	0.16	Dry	No	No
10-U145.00	90	3061	0.09	Dry	No	No
Little Spangle Lake	91	2921	5.8	10.7	Yes	Yes
Queens River Lake #10	92	2857	0.14	22.9	Yes	No
Queens River Lake #11	93	2820	2.29	8.03	Yes	No
Queens River Lake #12	94	2860	1.1	7.8	No	Yes
Spangle Lake	95	2927	15.63	21	Yes	Yes
Unnamed Lake	96	2960	0.02	1	No	Yes
Upper Spangle Lake	97	2997	1.42	5	No	Yes

Table 15. Results from amphibian surveys conducted by IDFG personnel from August 2-27, 2010. LLID # is used to identify unnamed lakes. Juveniles include all sub-adult and larval stages. Fish status for the lakes has been included.

	Columbia spotted frog		Long-toed salamander		Western toad		Fish
Lake name / LLID#	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	present
South Fork Payette River drainage							
1149923439672	9	-	-	4	-	-	No
1149926439666	1	-	-	9	-	-	No
1150127439591	-	58	-	-	-	-	No
1150164439377	-	1	-	-	-	-	No
1150219439649	-	88	-	-	-	-	No
1150604439575	16	7	-	-	-	-	Yes
1150614439553	1	62	-	-	-	-	No
1150635439730	-	-	-	42	-	-	No
1150642439588	1	-	-	-	-	-	No
1150642439604	-	350	-	-	-	-	No
1150647439575	-	-	1	-	-	-	No
1150694439738	-	33	-	-	-	-	No
09-U150.00	2	127	-	-	-	-	No
10-U152.00	5	135	-	-	-	-	No
Arlin Lake	1	-	-	-	-	-	Yes
Benedict Creek Lake #3	-	-	1	-	-	-	Yes
Benedict Lake	1	20	-	-	-	-	Yes
Heather Lake	-	-	1	-	-	-	Yes
Little Vernon Lake	-	-	-	-	12	630	No

Table 15. Continued.

Lake name / LLID#	Columbia spotted frog		Long-toed salamander		Western toad		Fish present
	Adult	Juvenile	Adult	Juvenile	Adult	Juvenile	
Rock Slide Lake	1	-	-	-	-	-	Yes
Ten Lake Creek Lake #10	3	21	-	-	-	-	Yes
Ten Lake Creek Lake #11	4	68	-	-	-	-	Yes
Ten Lake Creek Lake #2	2	33	-	-	-	-	No
Ten Lake Creek Lake #5	12	-	-	-	-	-	No
Ten Lake Creek Lake #6	15	220	-	-	-	-	No
Ten Lake Creek Lake #7	2	50	-	-	-	-	No
Three Island Lake	6	100	-	-	-	-	Yes
Upper Vernon Lake	14	75	-	-	-	-	Yes
Vernon Lake	1	75	-	-	-	-	Yes
Virginia Lake	17	78	-	-	-	-	Yes
Queens River drainage							
1151239439157	-	-	-	1	-	-	No
Upper Middle Fork Boise River drainage							
1150199439395	-	-	-	35	-	-	No
1150351439416	-	-	10	4	-	-	No
1150403439446	-	69	-	-	-	-	No
Little Spangle Lake	3	24	3	-	-	-	Yes
Queens River Lake #12	-	1	-	-	-	-	No
Spangle Lake	-	-	3	-	-	-	Yes
Unnamed Lake	-	-	-	38	-	-	No
Upper Spangle Lake	-	-	11	1	-	-	No

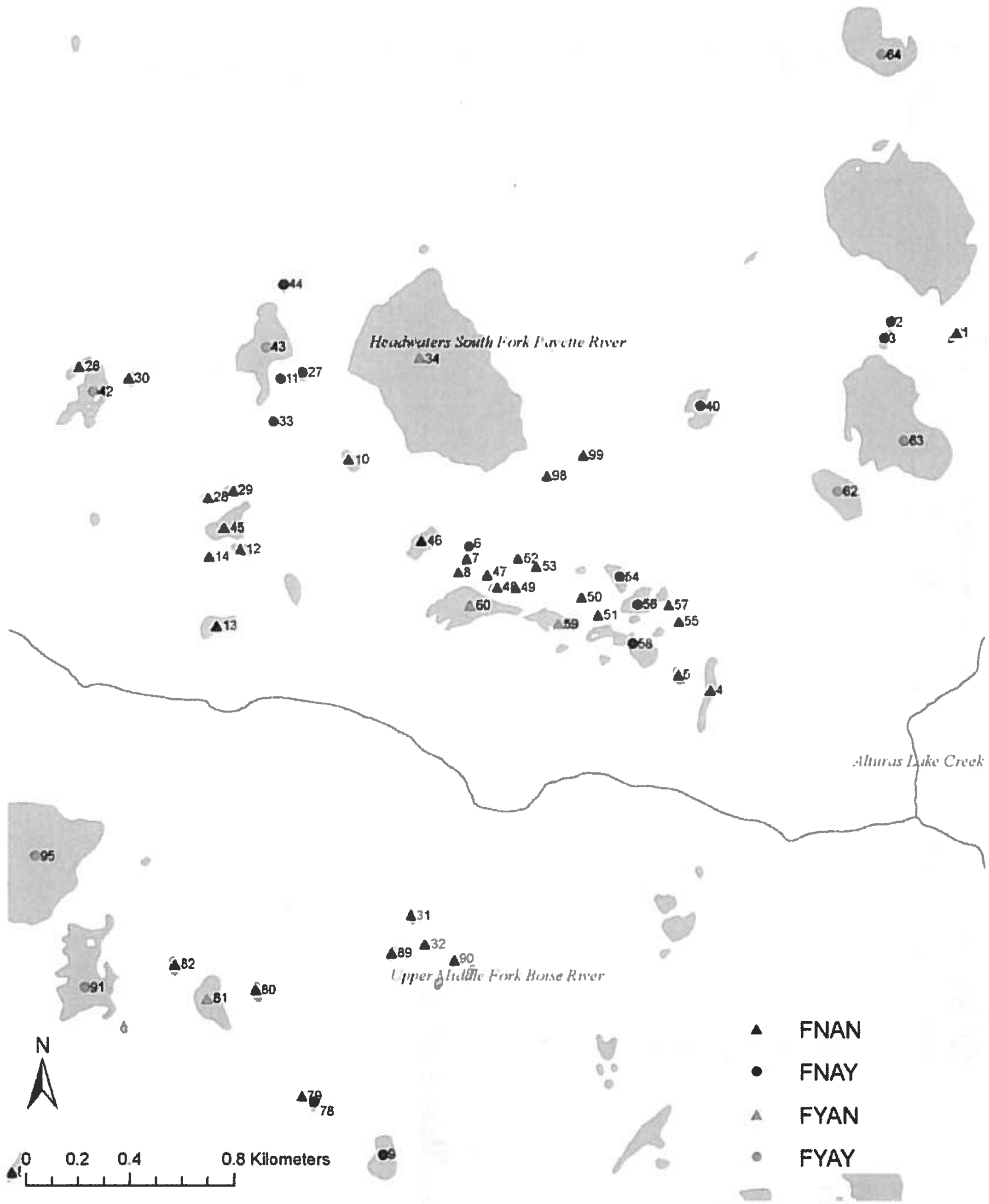

Figure 24. Lakes surveyed by IDFG personnel during August 23 - 27, 2010 in the headwaters of the South Fork Payette River drainage (HUC 5). Lake names can be found in table X and are referenced by the numbers displayed above. Legend denotes survey results for presence/absence of fish and amphibians: FNAN is no fish, no amphibians, FNAY is no fish, yes amphibians, FYAN is fish yes, amphibians no, and FYAY is fish yes, amphibians yes.

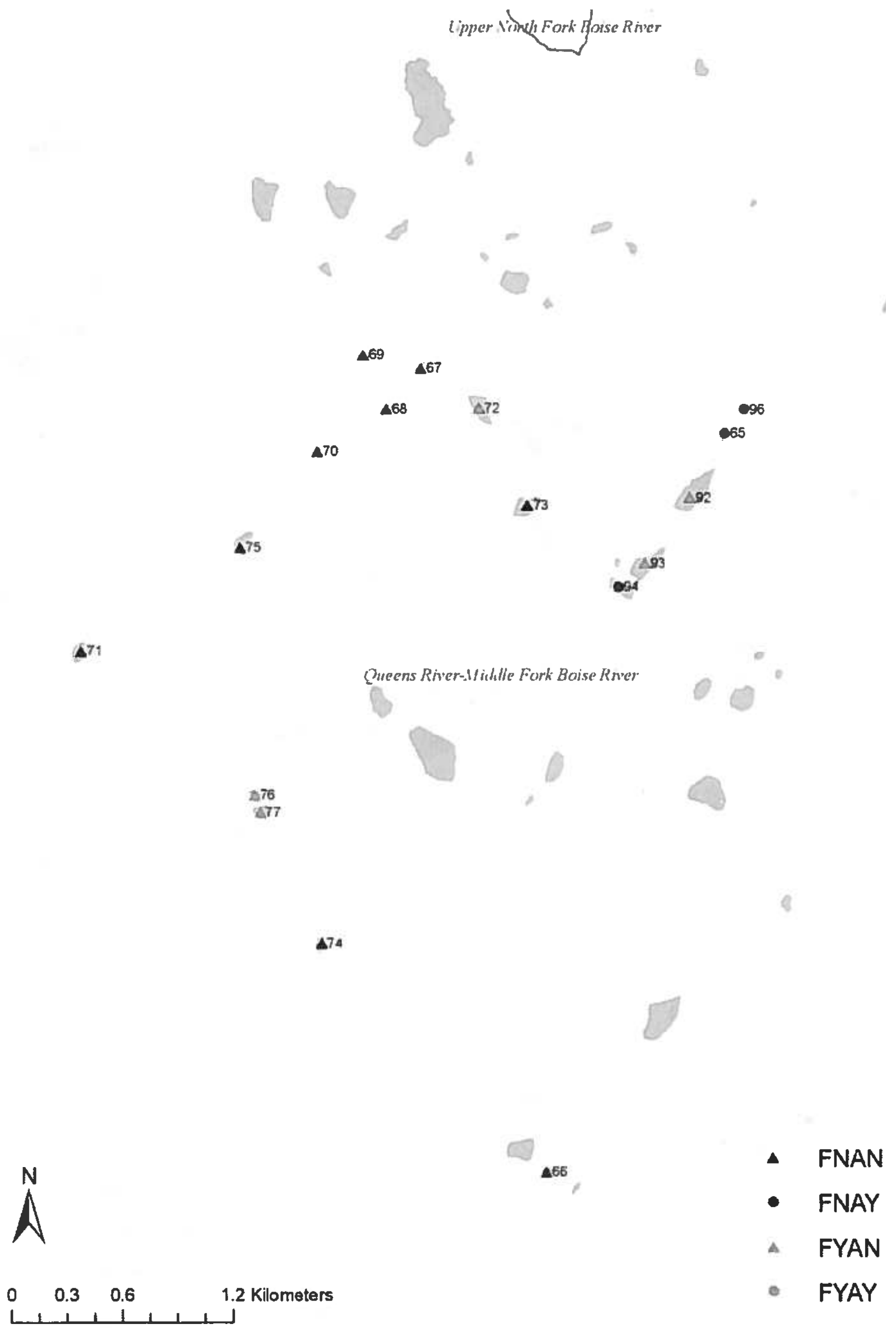

Figure 25. Lakes surveyed by IDFG personnel during August 2 - 6, 2010 in the Queens River drainage (HUC 5). Lake names can be found in table X and are referenced by the numbers displayed above. Legend denotes survey results for presence/absence of fish and amphibians: FNAN is no fish, no amphibians, FNAY is no fish, yes amphibians, FYAN is fish yes, amphibians no, and FYAY is fish yes, amphibians yes.

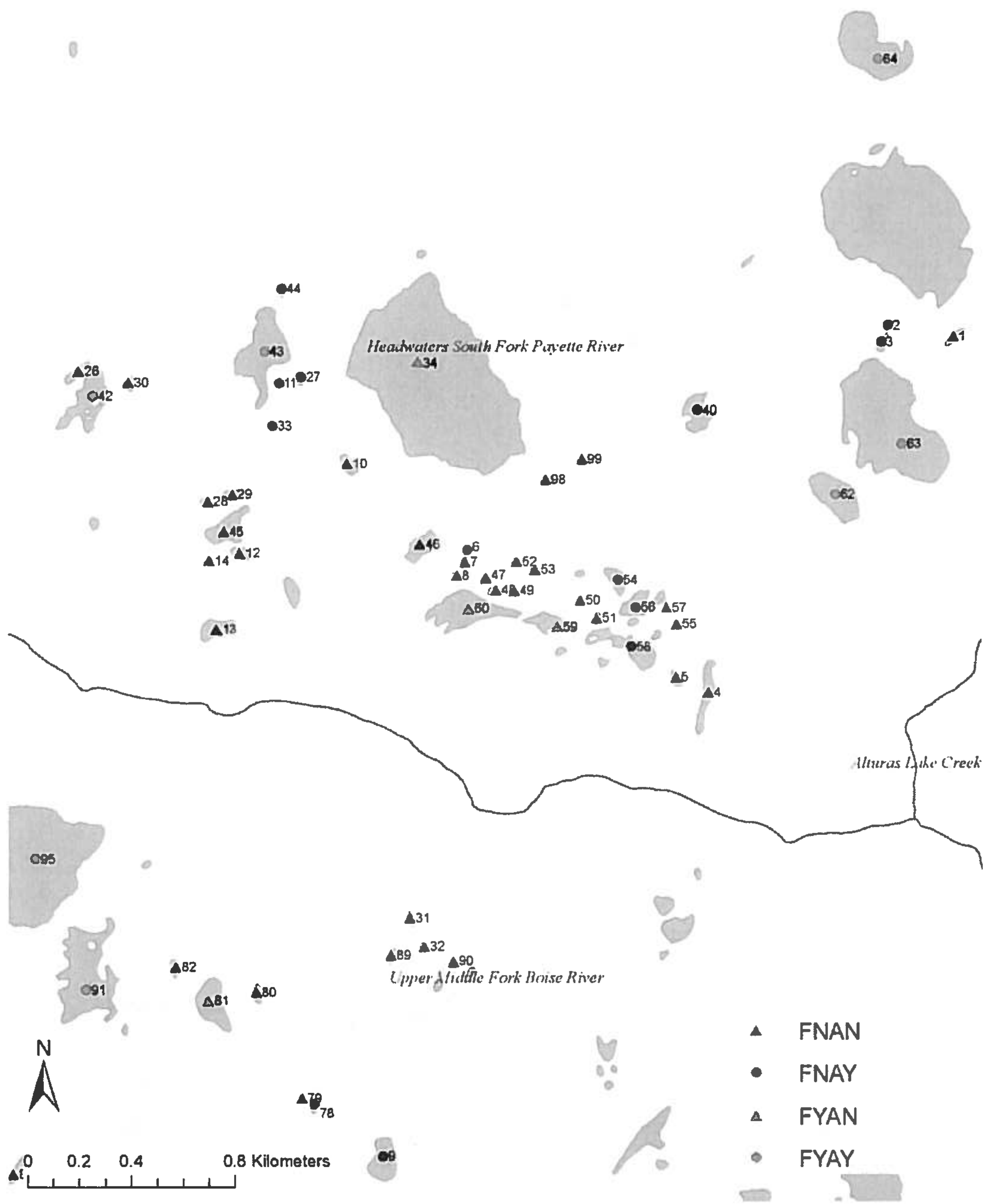

Figure 26. Lakes surveyed by IDFG personnel during August 23 - 27, 2010 in the Upper Middle Fork Boise River drainage (HUC 5). Lake names can be found in table X and are referenced by the numbers displayed above. Legend denotes survey results for presence/absence of fish and amphibians: FNAN is no fish, no amphibians, FNAY is no fish, yes amphibians, FYAN is fish yes, amphibians no, and FYAY is fish yes, amphibians yes.

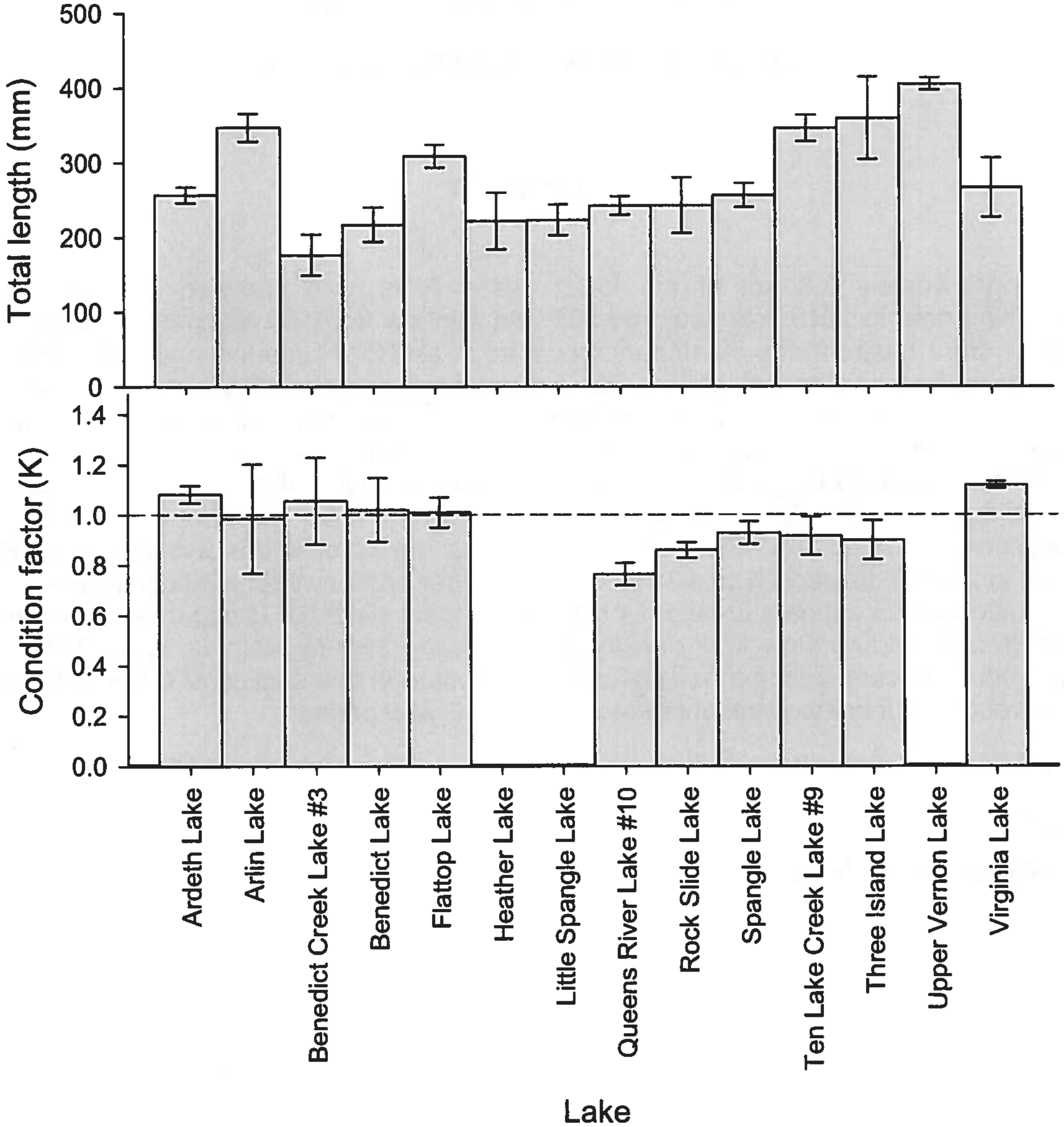

Figure 27. Mean values for fish length and Fulton's condition factor (K) calculated for lakes where fish were sampled during 2010 surveys. Error bars represent the 90% CI. A condition factor value of 1.0 represents the standard value for healthy fish.

River and Stream Investigations

Lower Boise River Electrofishing Survey

ABSTRACT

Approximately 3.8 km of the lower Boise River was sampled in four sites with electrofishing gear in 2010. We captured 567 wild rainbow trout, 65 hatchery rainbow trout, 73 wild brown trout *Salmo trutta*, 5 hatchery brown trout, and 2,174 mountain whitefish *Prosopium williamsoni* at four sections during the 2010 electrofishing survey. Population estimates were calculated for wild rainbow trout, wild brown trout, and mountain whitefish in the upper and middle sites. Rainbow trout lengths ranged from 74 to 522 mm, with most fish $\leq$ 300 mm at all sites. We estimated 3,210 $\pm$ 2,093 (90% CI) wild rainbow trout in the middle section and 544 $\pm$ 266 in the upper section. Brown trout lengths ranged from 114 - 740 mm, with the majority of fish measuring between 150 - 300 mm. Population estimates for wild brown trout were 80 $\pm$ 39 (90% CI) in the middle section and 31 $\pm$ 16 in the upper section. Wild rainbow trout abundance in the middle section appears to be at its highest numbers since IDFG began routine monitoring in 2004. In fact, wild rainbow trout density has increased over two-fold since 2007 from 3.3 to 8.3 fish/100m^2. In comparison to a 1994 density estimate in this section of 0.5 fish/100m^2, wild rainbow trout have increased seventeen-fold over a 16 year period.

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INTRODUCTION

The lower Boise River is a heavily used fishery that flows through the center of the Boise metropolitan area. Prior to 2004, sampling efforts captured few wild trout and anecdotal information suggests that the number of wild rainbow trout and brown trout in the river has improved over the last 20 years. Standardized population monitoring sections were established in 2004 to estimate populations of wild rainbow trout, brown trout, and mountain whitefish in the lower Boise River between Barber Park and the East Parkcenter Bridge. The sections are scheduled to be sampled every three years.

METHODS

Trout and mountain whitefish populations in the lower Boise River have been monitored every three years since 2004 in three sections of river between Barber Park and the East Parkcenter Boulevard Bridge (Hebdon et al. 2009; Flatter et al. 2011). The upper section begins at the first diversion below Barber Park and continues down to the Loggers Creek diversion, less than 50 m upstream from the West Parkcenter Boulevard Bridge. The middle section starts at the Canal diversion and stops downstream at the first riffle downstream of the confluence of Heron Creek. This section is within the reach managed with quality trout regulations of (two trout limit, none under 360 mm). The lower section begins at the first riffle below the first water treatment plant diversion (East River footbridge) and continues to the first riffle above the West Parkcenter Bridge.

Because the middle and lower sections are close in proximity (0.7 km), the lower section was dropped in 2010 in favor of adding two single-pass sections further downstream. The Americana section starts at the riffle just upstream of Americana Blvd. Bridge and continues to the first riffle after passing under the Fairview Ave Bridge. The Glenwood section starts in the north channel of the island just downstream of the Idaho Fairgrounds footbridge and stops at the first riffle above Glenwood Blvd. Bridge. Section length was determined from 1:24,000 km topographic maps. Wetted widths were measured with a hand-held laser range finder (Leupold RX series). Section area was estimated by multiplying mean widths and section length. For braided channels mean width was measured across the river excluding any distances across islands.

We used mark-recapture techniques to estimate abundance of trout and mountain whitefish in the upper and middle trend sections while the Americana and Glenwood sections were single passes for species composition and size distributions. Fish were collected with a canoe electrofishing unit consisting of a 5.2 m Grumman aluminum canoe fitted with two mobile anodes connected to 15.2 m cables. The canoe served as the cathode and carried the generator, Coffelt VVP-15, and a livewell for holding fish. Oxygen was introduced to the live well (2 L/min) through an air-stone. Pulsed direct current was produced by a 5,000 watt generator (Honda EG500X). Frequency was set at 60 pulses / sec and a pulse width of 60-80, with an output of 4-5 amperes. Crews consisted of six to seven people. Two operators managed the mobile anodes, one person guided the canoe and operated the safety switch controlling the output, the remaining crew of four or five people were equipped with dip nets to capture stunned fish. Only trout and whitefish were placed in the livewell.

Marking and recapture runs were conducted with a single pass from upstream to downstream. The canoe was held upstream of the anode operators. Anodes were swept through the water or thrown across the stream and retrieved. Crews with dip nets walked backward facing upstream, while staying downstream of the anodes and capturing stunned fish. Fish were placed in the live well and when the live well was judged to be at capacity the crew stopped at the nearest riffle to process fish.

The lower Boise River was sampled between October 26 and November 2, 2010. Fish were identified, enumerated and measured for total length (mm) and weighed (g). Rainbow trout, mountain whitefish and brown trout $\geq$100 mm were marked in the upper and middle sections on October 26. Fish were marked with a 7 mm diameter hole from a standard paper punch on the upper and lower section of the caudal fin and anal fin, corresponding to their capture reach. Fish were released 50 to 100 m upstream from the processing site to prevent them from drifting downstream into the next section of water to be sampled. Recapture sampling was completed on November 1. During the recapture effort all whitefish and trout greater than 100 mm were captured and placed in the live well. Fish were examined for marks on the caudal fin. All recaptured fish were measured for length (mm). The Americana and Glenwood sections were sampled on October 27 and November 2, respectively.

To account for selectivity of electrofishing gear population estimates (N) were calculated using a maximum likelihood estimation to fit the recapture data. A capture probability function of the form

$$Eff = (exp(-5+\beta_1 L+ \beta_2 L^2)) /(1+ exp(-5+\beta_1 L+ \beta_2 L^2))$$

where *Eff* is the probability of capturing a fish of length L, and β_1 and β_2 are estimated parameters (MFWP 2004). Then N is estimated by length group where M is the number of fish marked by length group.

$$N = M / Eff$$

Population estimates were calculated for each reach and pooled for a comprehensive estimate expressed as # fish/100 m^2 for comparison to previous surveys.

Trout population estimates ($\check{N}$) for surveys from which the number of mark-recapture numbers were not adequate to use log-likelihood, were estimated using the modified Petersen equation for fish $\geq$100 mm.

$$\check{N} = [((M+1)*(C+1)) / (R+1)] - 1$$

Where M is the number of fish marked, C is the number of fish captured and R is the number of fish recaptured. Population estimates, length frequencies, and species composition, were compared to results reported from prior surveys (Hebdon et al 2009; Flatter et al. 2011; Allen et al. 2000).

RESULTS

Approximately 3.8 km of the lower Boise River was sampled in four sites with electrofishing gear in 2010 (Figure 28). We captured 567 wild rainbow trout, 65 hatchery rainbow trout, 73 brown trout, 5 hatchery brown trout, and 2,174 mountain whitefish at four sites during the 2010 electrofishing survey. Population estimates were calculated for wild rainbow trout, wild brown trout, and mountain whitefish in the upper and middle sites. However, due to inexperience of volunteer netters used during the marking run, we captured more fish during the recapture run which resulted in wide confidence intervals around population estimates. Species composition and length frequency analyses were conducted for the Americana and Glenwood sites.

Wild rainbow trout made up 6.3% to 37% of the catch in the four sites, with the middle section having the highest densities, and Americana the lowest (Table 16). Rainbow trout lengths ranged from 74 - 522 mm, with most fish exceeding 300 mm at all sites (Figure 29). We estimated 3,210 ± 2,093 (90% CI) wild rainbow trout in the middle site and 544 ± 266 in the upper site. Hatchery rainbow trout comprised just a small proportion of fish handled (0.5% to 5.4%) at all sites and population estimates were not calculated (Table 17).

Brown trout ranged between 0.4% and 6.6% of the fish captured in 2010, with the middle section holding the highest densities of the four sites (Table 16). Brown trout lengths ranged from 114 - 740 mm, with the majority of fish measuring between 150 - 300 mm (Figure 30). We estimated 80 ± 39 (90% CI) brown trout in the middle site and 31 ± 16 in the upper site. Hatchery brown trout were an extremely small percentage of fish captured in 2010, ranging between 0% at the Glenwood and middle sites and 0.8% at the upper site (Table 16).

Mountain whitefish were the predominant fish collected in terms of numbers caught, making up between 54% of total catch at the middle site and 93% at the Americana site (Table 16). Total lengths for mountain whitefish ranged between 63-478 mm, with the majority of fish between 150 - 300 mm at all sites (Figure 31). We estimated 2,857 ± 936 mountain whitefish in the middle site and 2,417 ± 696 in the upper site.

DISCUSSION

Wild rainbow trout abundance in the middle site is at its highest numbers since IDFG began routine monitoring in 2004 (Figure 32). In fact, wild rainbow trout density has increased over two-fold since 2007 from 3.3 to 8.3 fish/100 m^2 (Table 17). In comparison to a 1994 density estimate in this section of 0.5 fish/100 m^2, wild rainbow trout have increased seventeen-fold over a 16 year period. Rainbow trout length distributions have varied over time and location. While the number of fish ≤ 150 mm has decreased in the middle site between 2007 and 2010, fish between 150 - 300 mm have increased from 24 to 65% (Hebdon et al. 2009). The upper site also suggested a similar change over the same period and we noted an increase in the proportion of fish ≥ 300 mm. The Americana section had the lowest catch rate and total catch for wild rainbow trout, due likely to the difficulty of shocking the section because of the deep pools and runs. Rainbow trout in the Glenwood section were mostly wild origin (87%) compared to hatchery (13%), despite receiving 200-500 hatchery fish on a monthly basis.

Wild brown trout abundance appears to be quite variable over time. Brown trout numbers were higher in 1994 and 2007, but much lower than 2004 (Figure 32). Overall wild brown trout density has increased from 0.09 to 0.21 fish/100 m^2 since 1994 (Table 17).

Approximately 15,000 adipose-clipped hatchery brown trout have been stocked into the Boise above Glenwood Blvd. in both 2009 and 2010. However, only five hatchery brown trout were captured during the 2010 survey. The proportion of wild brown trout $\geq$ 300 mm in the middle site increased from 2007, with most of the fish increasing in the 300 - 450 mm range (Hebdon et al. 2009). Additional time is needed to determine if stocking hatchery brown trout will improve the fishery in the lower Boise River.

Mountain whitefish abundance has remained relatively stable since 2004 in the middle site, ranging from a high of 3,352 $\pm$ 655 (90% CI) in 2007 to a low of 2,273 $\pm$ 172 in 2004 (Figure 32). However, the increase in mountain whitefish density since 1994 has been dramatic, going from 0.8 to 7.4 fish/100 m^2 in 2010 (Table 17). Length distribution of mountain whitefish has been comparably stable compared to the other surveyed species. Though there was a smaller proportion of fish $\leq$ 150 mm in 2010, the proportion of fish between 150 - 450 mm has been similar between years and sites (Hebdon et al. 2009).

The wild rainbow trout population still appears to be increasing in abundance in much of the lower Boise River. This is most likely related to the establishment of a minimum winter flow in the late 1990's. Low winter flows have been shown to inhibit survival of juvenile trout in numerous systems (Hurst 2007; Mitro 2002). The proportion of fish $\leq$150 mm captured in 2010 was somewhat lower than in previous years, suggesting that recruitment may have been limited in 2009. Further examination of flows, as well as spawning and rearing habitat would be useful for a better understanding of the trout population dynamics in the lower Boise River.

MANAGEMENT RECOMMENDATIONS

1. Continue population monitoring in the upper and middle sections every three years.
2. Identify important rearing areas for wild trout in the urban reach of the lower Boise River by shoreline electrofishing to establish trend sites for age-0 trout.

Table 16. Species composition (%) of surveyed fish for four electrofishing sites in the lower Boise River in 2010.

Site	Percent species composition (%)				
	Rainbow trout (wild)	Rainbow trout (hatchery)	Brown trout (wild)	Brown trout (hatchery)	Mountain whitefish
Upper	28	5.4	2.3	0.8	63.6
Middle	36.9	2.5	6.6	0	54
Americana	6.3	0.5	0.4	0.1	92.7
Glenwood	15.7	2.4	1.7	0	80.1

Table 17. Density estimates (fish/100m^2) for wild rainbow trout, wild brown trout, and mountain whitefish in the middle section of the lower Boise River, 1994 - 2010.

	Density (fish/100m^2)		
Year	Rainbow trout (wild)	Brown trout (wild)	Mountain whitefish
1994	0.47	0.09	0.80
2004	3.55	1.23	5.53
2007	3.29	0.57	8.15
2010	8.33	0.21	7.41

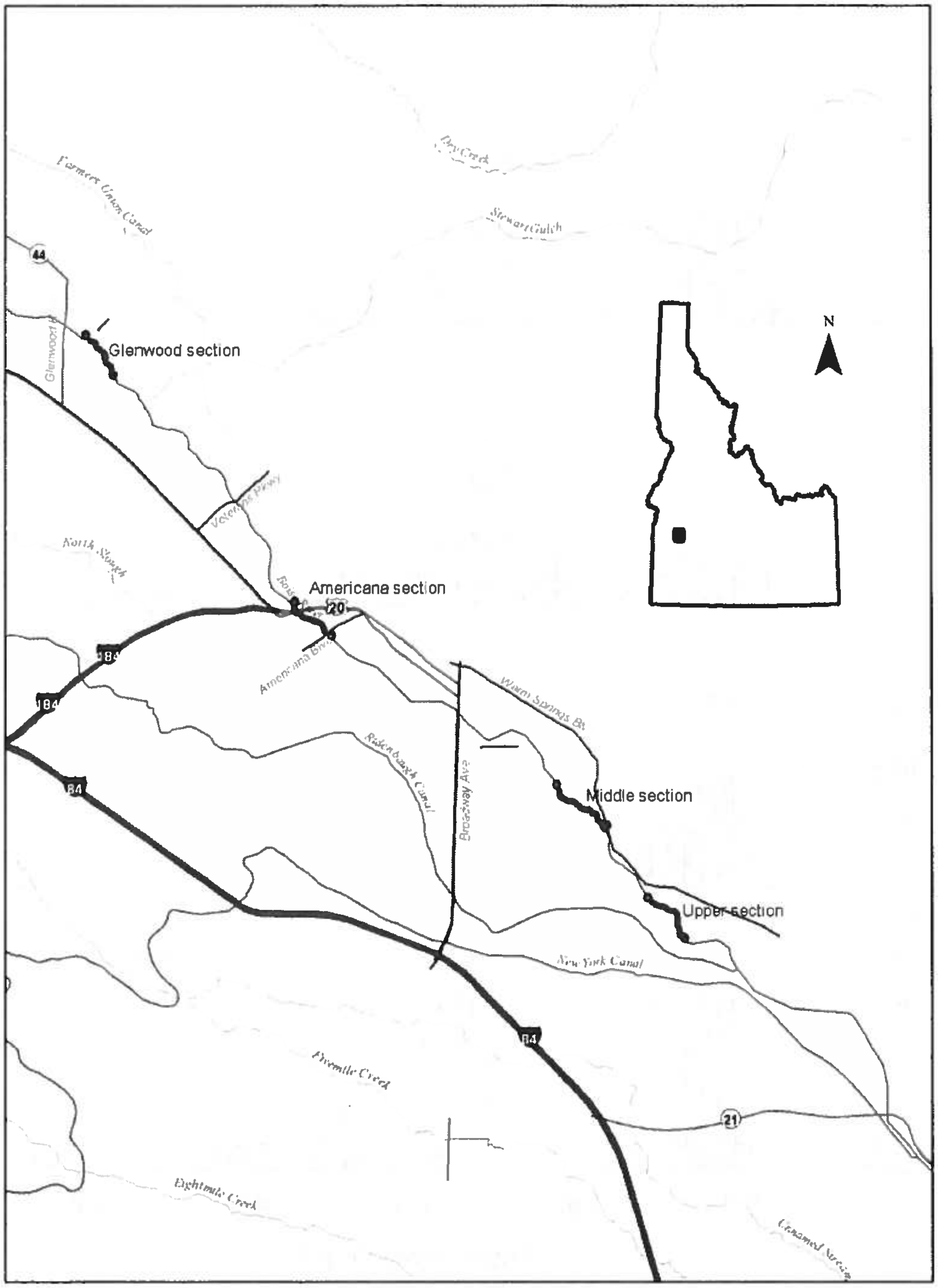

Figure 28. Map of the lower Boise River, Idaho sampling sites showing boundary sections for the 2010 upper, middle, Americana, and Glenwood sites.

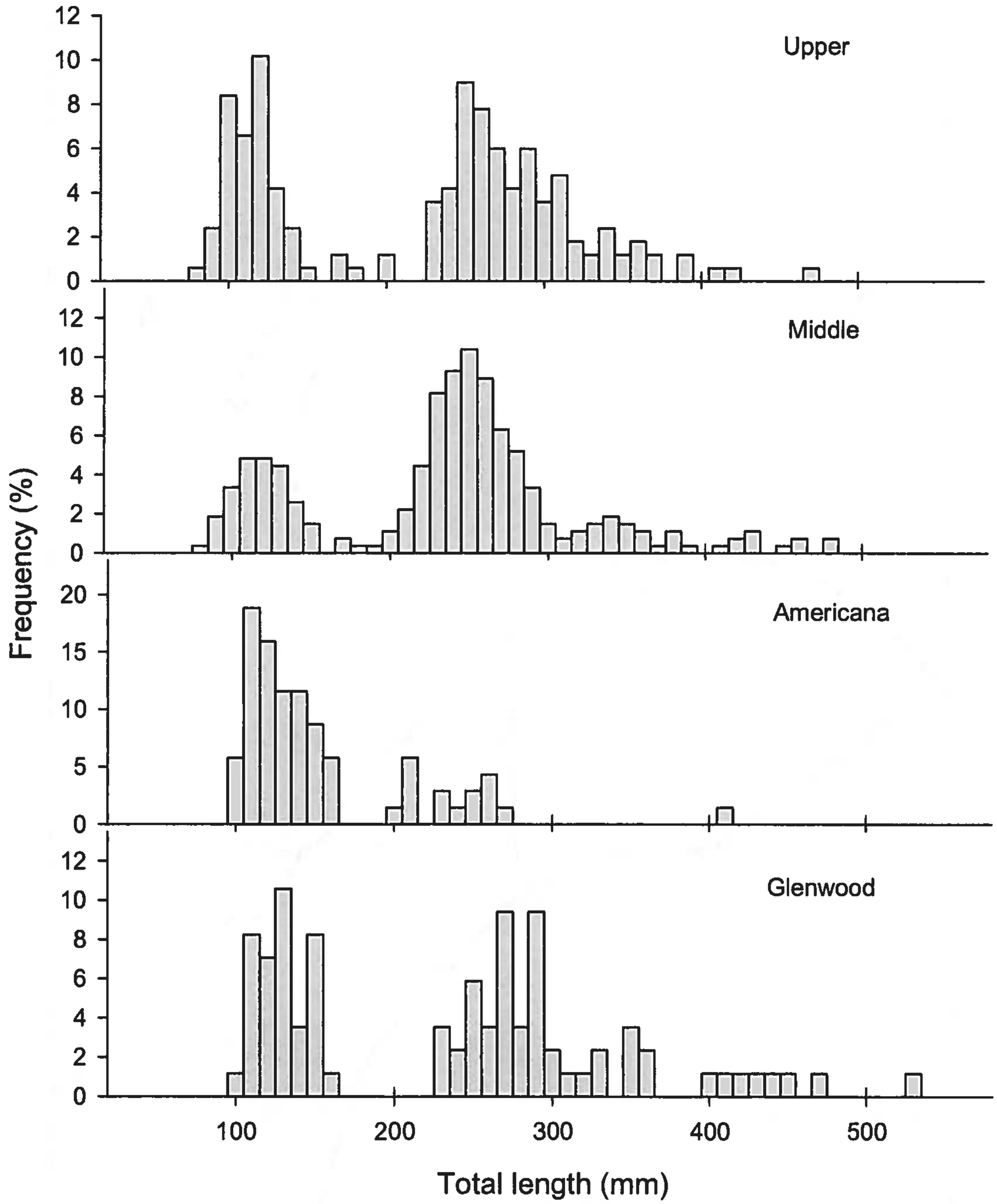

Figure 29. Length distribution of wild rainbow trout collected during the 2010 lower Boise River electrofishing survey at four sites.

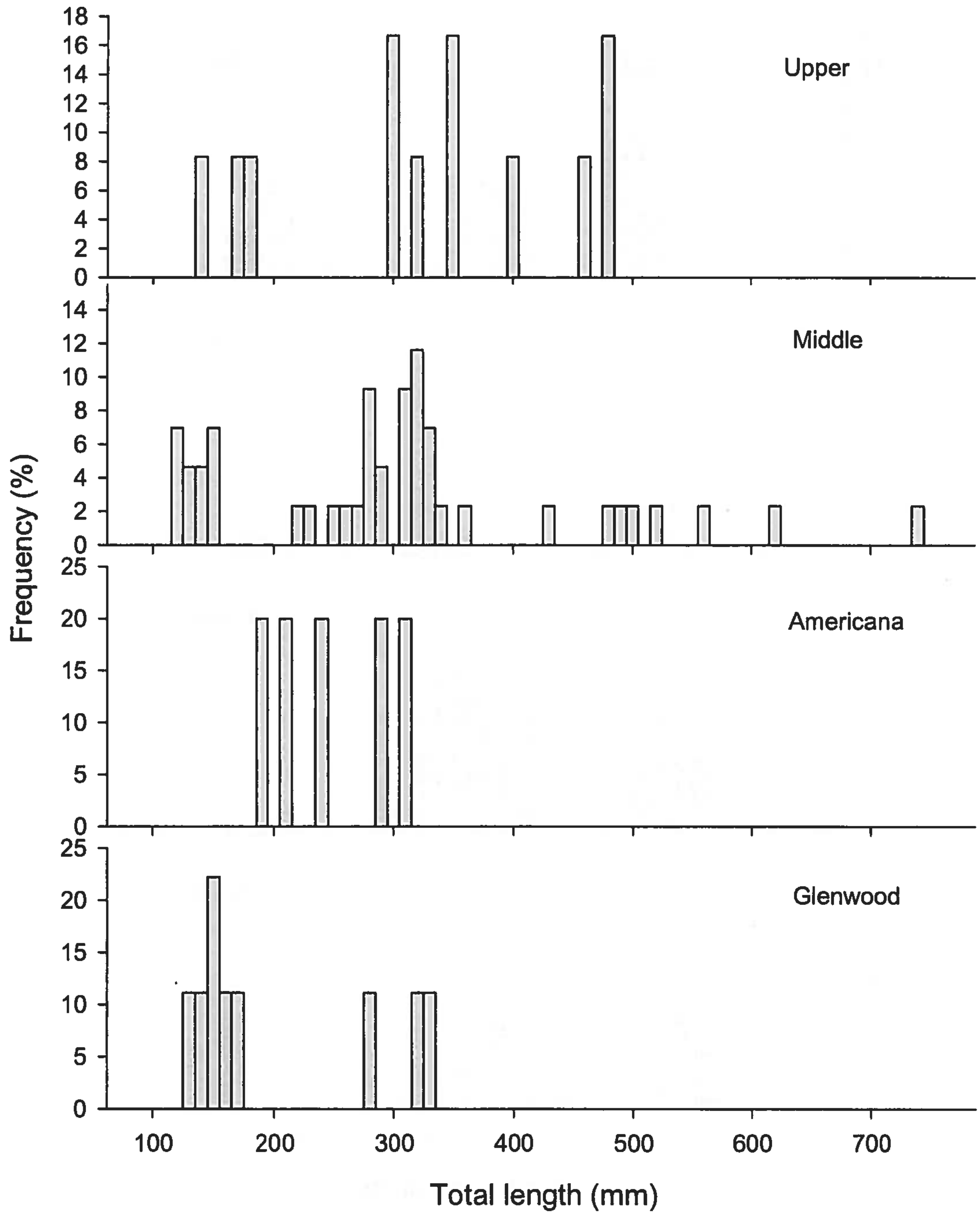

Figure 30. Length distribution of wild brown trout collected during the 2010 lower Boise River electrofishing survey at four sites.

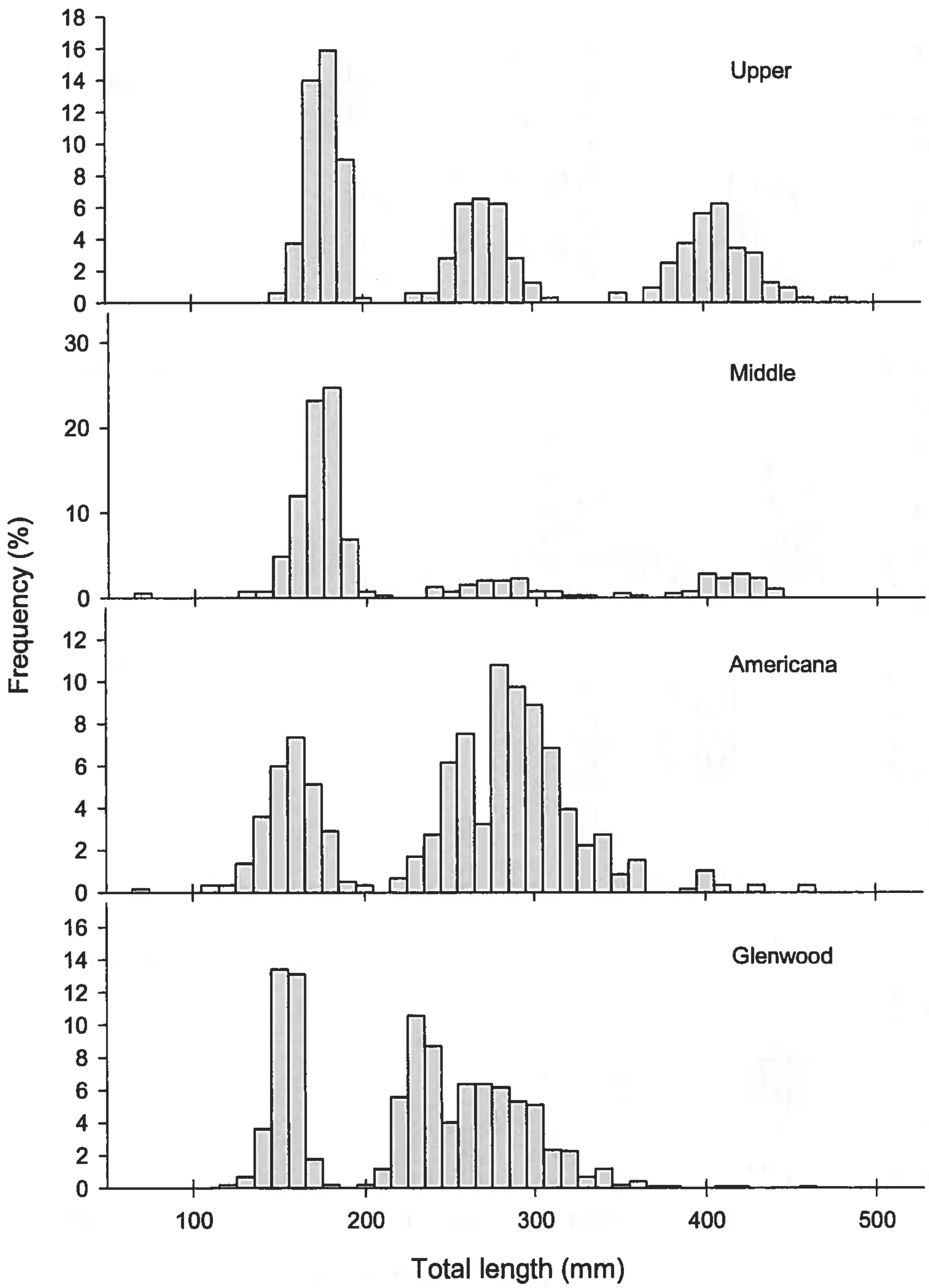

Figure 31. Length distribution of mountain whitefish collected during the 2010 lower Boise River electrofishing survey at four sites.

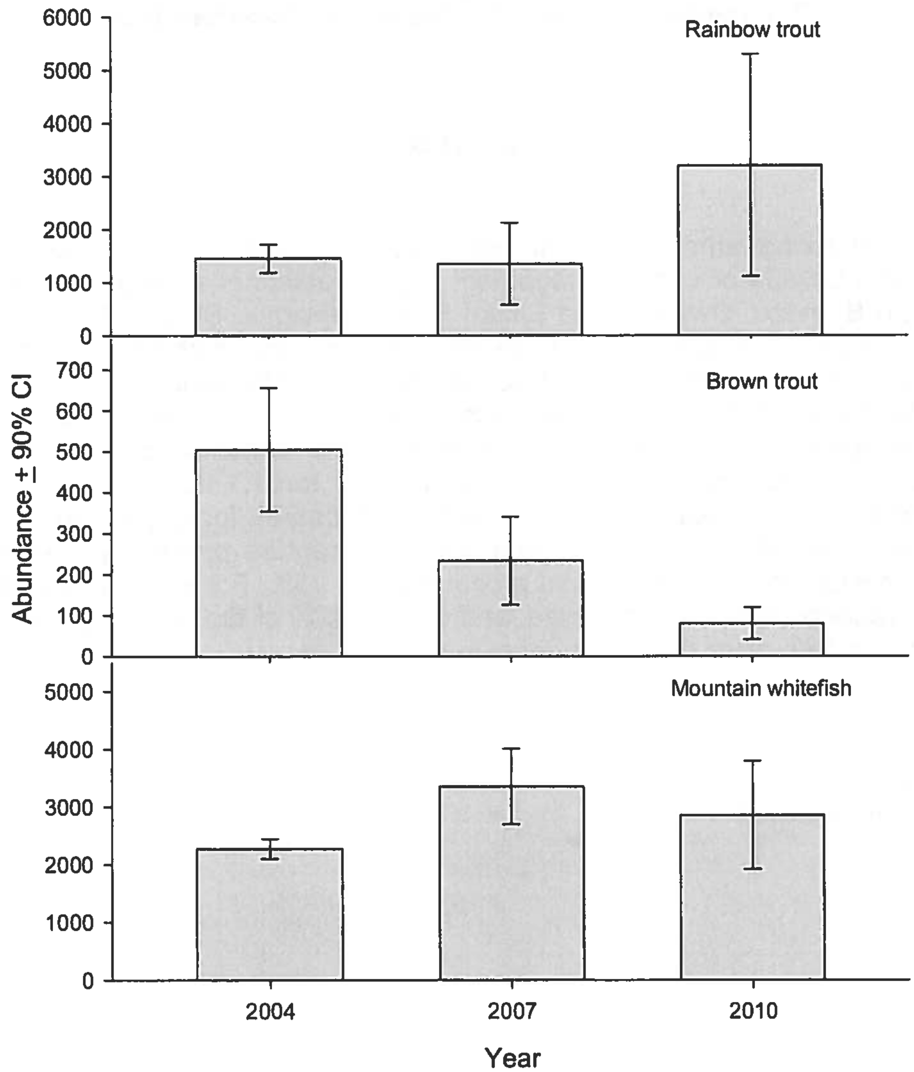

Figure 32. Abundance estimates for wild rainbow trout, brown trout, and mountain whitefish in the middle section of the lower Boise River 2004 - 2010. Error bars represent 90% CI for the population estimates.

<u>**River and Stream Investigations**</u>

Bruneau, Owyhee, and Snake River Drainages - Long-Term Monitoring of Redband Trout Populations in Desert Basins of Southwestern Idaho

ABSTRACT

As part of a long-term redband rainbow trout *Oncorhynchus mykiss gairdneri* monitoring effort, IDFG and Bureau of Land Management (BLM) personnel agreed to sample 63 stream sites within the Bruneau, Owyhee, and Snake River drainages. During 2010, the third year of sampling for this effort, we completed 15 standard stream surveys primarily on the north side of the Owyhee Mountains and south side of Bennett Mountain. Redband trout were captured at 14 of the 15 sites sampled during 2010. No redband trout were sampled at the Pole Creek site, where only smallmouth bass were present. Among the streams containing redband trout, redband trout abundance ranged from 0.1 trout/100 m^2 to 51.1 trout/100 m^2, with a mean of 15.8 ± 7.1 trout/100 m^2 (mean ± 90% CI). Capture probability for larger and smaller redband trout were nearly equal. For fish less than 100 mm, capture probability was 82%. For fish greater than or equal to 100 mm, capture probability was 72%. For all the 2010 sites combined, a total of 567 redband trout were sampled, and overall, 42% of the fish sampled were less than 100 mm, whereas 58% were greater or equal to 100 mm.

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INTRODUCTION

Redband trout are native to all major river drainages in Southwestern Idaho. Within this large and diverse geographical area, redband trout have adapted to a variety of stream habitats including those of montane and desert areas. Some controversy has existed regarding whether adaptation to these disparate habitats has led to speciation at some level. During 1995, those redband trout that reside in desert locales were petitioned for listing under the federal Endangered Species Act (ESA), under the assumption that they could be considered a separate sub species. The petition was denied. Since that time, additional research has indicated that only one species of resident stream dwelling redband trout may exist in Southwestern Idaho (Cassinelli 2008). Regardless of species designations, it is important to monitor redband trout population status and describe abundance trends across their full distribution. Population status of the redband trout from montane habitats has been extensively studied in Southwestern Idaho. However, due to remoteness and little angling interest (Schill et al. 2007), the redband trout from desert habitats has received less attention. These habitats include tributaries of the Bruneau, Owyhee, and Snake River drainages most often in headwater areas. As these populations are near the southern extent of their range and water temperatures are projected to increase, it has become more important to monitor these populations closely.

A long-term assessment of redband trout distribution, population abundances, and size structures was completed by Zoellick et al. (2005). This assessment compared redband population characteristics at 43 sites within the Bruneau, Owyhee, and Snake River drainages, from 1993 – 2003, to data collected at the same sites during 1977 – 1982. Site numbers referred to in this report correspond to the site numbers in Zoellick et al. (2005). As a continuation of this effort, IDFG and BLM personnel agreed to resample these 43 sites over a three to four year period beginning in 2008. Also, an additional 20 sites were added to more fully encompass the redband trout's distribution in the high desert environs of Southwest Idaho.

METHODS

Multiple-pass depletion methods were used to estimate fish population characteristics at all sites. Previously-sampled sites were located using descriptions, photographs, or coordinates. Block nets were installed at the upstream and downstream end of each transect. Fish capture efforts utilized a Smith Root backpack electrofisher (Model 15-B) and a two or three-person crew equipped with dip nets. Captured fish were held in small buckets and transferred to a livewell placed downstream of the site. Capture efforts focused on redband trout, but non-game species were also captured, identified, and visually categorized as sparse (1 – 10), many (10 – 50), or abundant (>50; Meyer 2009). The number of passes completed depended on numbers of fish caught during the first pass. If redband trout catch in the first pass was less than five, sampling was terminated. If more than five redband trout were sampled, a second pass was completed. If catch remained relatively high in subsequent passes (> 25% of the previous pass) additional passes were completed. Also, herpetofauna were identified visually to species and recorded as eggs, larval form, juvenile, or adult. We sampled 15 sites during 2010 primarily in tributaries and the main-stems of the Reynolds, Deep, and Cold Springs creeks as well as streams that flow directly to the Snake River. These sites are south of Marsing, ID and north-east of Mountain Home, ID (Figure 33). Fourteen of the sites had been sampled previously (Zoellick et al. 2005), whereas one site was new. Population estimates were calculated using MicroFish 3.0 (Van Deventer 2006). Due to the potential for size-related catchability differences, population abundance estimates were calculated for two strata: (1) trout less than 100 mm, and

(2) trout greater than or equal to 100 mm, then summed. Confidence intervals for mean abundance and the difference between abundance for a particular site across time (d) were calculated using an $\alpha = 0.1$.

RESULTS

Redband trout were captured at 14 of the 15 sites sampled during 2010. No redband trout were sampled at the Pole Creek site (# 26). Total catch at the remaining sites ranged from one to 145 redband trout. No non-native trout were sampled in this subset of sites, and smallmouth bass were the only other gamefish sampled. Smallmouth bass were present at the Pole Creek site and at the lower Deep Creek site. In addition, three native species were sampled during these stream surveys including bridgelip sucker *Catostomus columbianus*, dace *Rhinichthys* spp., and redside shiner *Richardsonius balteatus*. Also, fathead minnow *Pimephales promelas* was sampled in lower Reynolds Creek. To our knowledge, this is a range expansion for this non-native nongame fish, which had only been documented previously in the Snake River.

Redband trout abundance for the 14 sites from which redband trout were sampled averaged 15.8 ± 7.1 trout/100 m^2 of stream (mean ± 90% CI) and was highly variable among sites (Table 18). The lowest abundance of 0.7 trout/100 m^2 occurred at the lower Reynolds Creek site (#53), whereas the highest abundance of 51.4 trout/100 m^2 occurred at the East Fork Reynolds Creek site (#49). We saw little difference in capture probability among the two size classes used for this analysis. For fish less than 100 mm, capture probability was 82%. For fish greater than or equal to 100 mm, capture probability was 72%. For all the 2010 sites combined, a total of 567 redband trout were sampled and overall 239 (42%) of the trout sampled were less than 100 mm, whereas 328 (58%) were greater or equal to 100 mm. Smallmouth bass abundances at the Deep and Pole creeks sites were similar (~10-15 smallmouth/100 m^2). Redband trout co-occurred with smallmouth bass at the Deep Creek site only, where only large redband trout (> 225 mm) were sampled.

Many of the sites sampled during 2010 had been sampled previously during 1977 – 1983 or during 1997 – 2003, which allowed us to compare population trends through time. Eleven of the sites sampled during 2010 were sampled during both the 1977 – 1983 and 1997 – 2003 survey efforts. For this set of sites, no difference in abundance could be detected ($\bar{d}$ = 0.1 fish/100 m^2 ± 18.7) for the 1977 – 1983 ($\bar{x}$ = 14.9 fish/100 m^2) and 1997 – 2003 ($\bar{x}$ = 14.8 fish/100 m^2) survey efforts (Table 20). Presence of redband trout was also similar during these efforts, and redband trout were sampled at 10 of the 11 sites. Comparing historical surveys to the current year's surveys, little change in distribution was noted, but a small decline in mean abundance, though this decline was not statistically significant. Ten sites were sampled during both the 1977 – 1983 and 2010 time periods. Mean abundance for the 1977 – 1983 period was 19.5 fish/100 m^2, whereas mean abundance for 2010 was 14.0 fish/100 m^2. Despite this 28% decline in mean abundance between these two periods, no statistical difference was apparent ($\bar{d}$ = 6 fish/100 m^2 ± 7.3), based on a confidence interval that included zero. Seven of ten sites sampled during 1977 – 1983 had redband trout present, whereas nine of ten of the same sites had redband trout present during 2010.

DISCUSSION

For the relatively small number of older trend sites re-sampled during 2010, presence of redband trout was similar to past surveys; in fact, redband trout were seen at more sites than previously. However, there was a small decline in redband trout abundance compared to historical surveys, though this tendency was not statistically significant. It is important to note that this is just a small sub-set of the sites that will be sampled for this effort. After completion of the remaining trend monitoring sites during 2011, a more thorough analysis of redband trout distribution and abundance will be completed as well as assessment of correlated factors.

MANAGEMENT RECOMMENDATIONS

1. Continue surveying historical sample sites to develop trend information on redband trout density and life stages

2. Use repeat sites to evaluate fish/amphibian community structure trends and changes.

Table 18. Abundance estimates (#/100 m^2) by length group and total for redband trout sampled at 15 monitoring sites sampled during 2010. Lower 95%confidence and upper 95% confidence limits abbreviated as LCL and UCL, respectively.

		< than 100 mm			≥ 100 mm			Total		
Site #	Site Name	Abundance	LCL	UCL	Abundance	LCL	UCL	Abundance	LCL	UCL
27	Deep	0.0	0.0	0.0	0.4	0.4	0.4	0.4	0.4	0.4
28	Salmon	12.4	12.4	12.4	2.1	2.1	2.1	14.5	14.5	14.5
29	Reynolds	3.5	2.9	4.1	14.1	12.9	15.3	17.9	16.5	19.4
30	Sinker	3.8	2.7	4.8	3.2	1.6	4.8	7.5	4.8	10.2
31	Sinker	0.3	0.3	0.3	3.9	1.0	6.9	3.9	2.0	5.9
40	EF Cold Springs	1.3	1.3	1.3	22.2	19.6	24.8	24.8	22.2	27.5
42	WF Cold Springs	0.0	0.0	0.0	0.9	0.9	0.9	0.9	0.9	0.9
43	Little Canyon	6.0	5.5	6.5	21.7	17.7	25.6	27.6	24.1	31.1
48	Deep	0.0	0.0	0.0	1.1	-62.1	69.6	1.1	-62.1	69.6
49	EF Reynolds	28.3	15.4	41.1	26.7	25.7	27.8	51.4	46.8	56.1
50	Reynolds	0.0	0.0	0.0	0.7	0.0	1.5	0.7	0.0	1.5
52	Jump	7.3	6.9	7.6	7.6	6.2	9.0	14.9	13.9	15.9
53	Jump	18.4	17.4	19.3	11.4	10.8	12.0	29.9	28.5	31.3
54	Squaw	11.5	10.7	12.3	14.3	11.1	17.5	25.8	23.4	28.2

Table 19. Abundance estimates (#/100 m^2) by length group and total for smallmouth bass sampled at 2 of 15 monitoring sites sampled during 2010. Lower 95%confidence and upper 95% confidence limits abbreviated as LCL and UCL, respectively.

		< than 100 mm			≥ 100 mm			Total		
Site #	Site Name	Abundance	LCL	UCL	Abundance	LCL	UCL	Abundance	LCL	UCL
26	Pole	0.7	-1.2	2.6	9.7	9.4	9.9	10.4	9.9	10.9
48	Deep	3.0	2.1	3.9	7.9	3.9	11.8	11.0	7.3	14.8

Table 20. Comparison of redband trout Abundance estimates (#/100 m^2) and 95% confidence intervals (abbreviated as LCL and UCL) over the last thirty-plus years for trend monitoring sites near the north side of the Owyhee Mountains and south side of Bennett Mountain.

		1977-1982				1993-2003				2010			
Site #	Creek Name	Abundance	LCL	UCL	YEAR	Abundance	LCL	UCL	YEAR	Abundance	LCL	UCL	YEAR
26	Pole	0			1979	0			2000	0			2010
27	Deep	12.5	11.8	15.1	1977	0			1997	0.4	0.4	0.4	2010
28	Salmon	0			1977	95.5	89.4	107.7	2001	14.5	14.5	14.5	2010
29	Reynolds	23.1	18.3	31.8	1977	5.8			2002	17.9	16.5	19.4	2010
30	Sinker	34.3	21.6	66.4	1977	3.8	3.8	4.6	2001	7.5	4.8	10.2	2010
31	Sinker	3.8	3.8	4.4	1977	2.7	2.7	3.6	2003	3.9	2.0	5.9	2010
40	EF Cold Springs	21.9	21.9	23.7	1981	2.4	2.4	4.8	2001	24.8	22.2	27.5	2010
41	EF Cold Springs	5.5	5.5	12.8	1981	12.7	12.7	14.3	2001				
42	WF Cold Springs	24.5	24.5	61	1982	29.3	29.3	30.8	1993	0.9	0.9	0.9	2010
43	Little Canyon	38.5	36.7	43	1981	10.7	10.1	13	2003	27.6	24.1	31.1	2010
48	Deep					0			1997	1.1	-62.1	69.6	2010
49	EF Reynolds					13.1			2002	51.4	46.8	56.1	2010
50	Reynolds	0	0	0	1994	0.5			2002	0.7	0.0	1.5	2010
52	Jump					39.2			2002	14.9	13.9	15.9	2010
53	Jump					56.2			2002	29.9	28.5	31.3	2010
54	Upper Squaw									25.8	23.4	28.2	2010

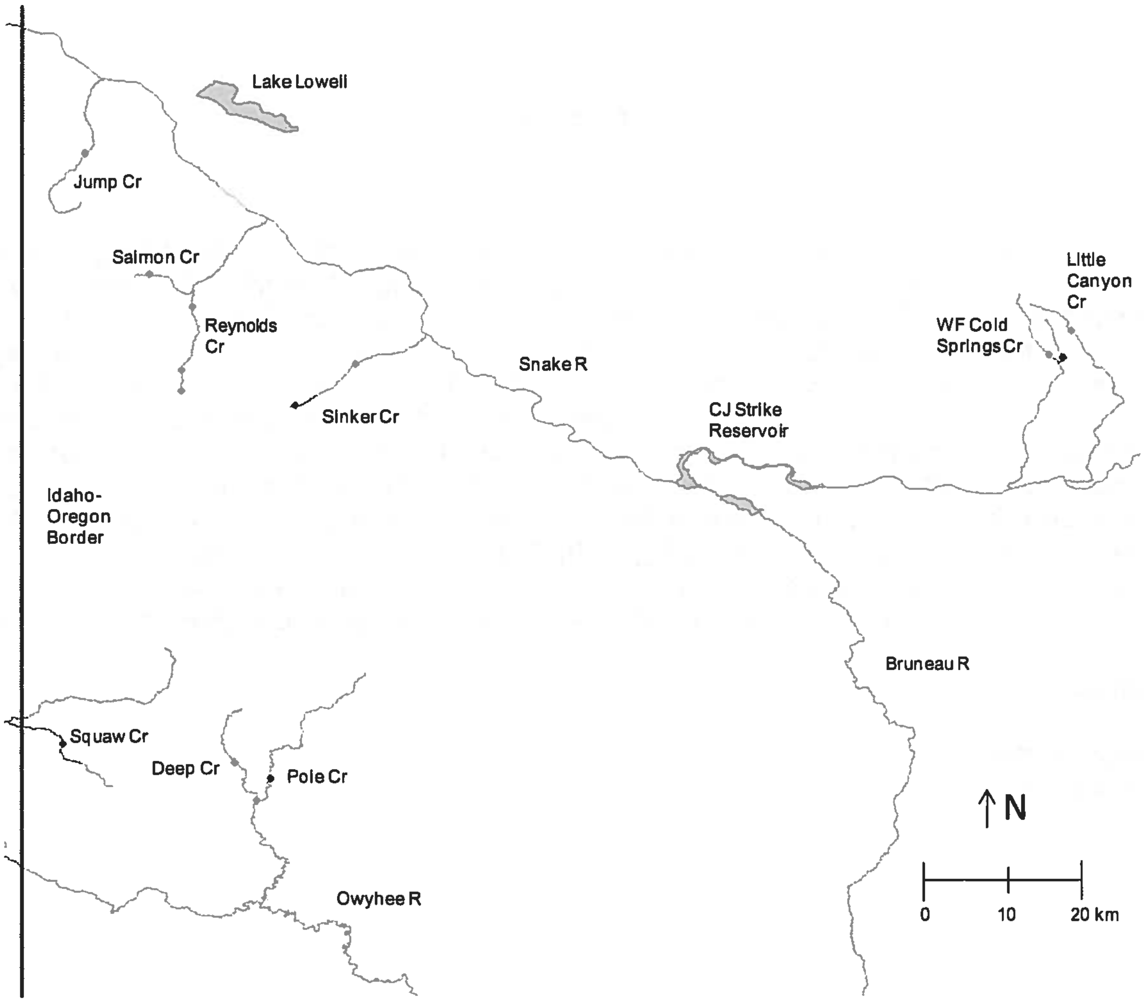

Figure 33. Location of 15 trend redband trout monitoring sites within the Snake and Owyhee river drainages. Sites were sampled during 2010. Black triangles denote sampling sites.

River and Stream Investigations

Middle Fork Salmon River Chinook Salmon Redd Counts

ABSTRACT

Spawning ground surveys were conducted at 11 historical trend monitoring transects in Bear Valley, Elk, and Sulphur creeks from August 30 through September 2, 2010 to index the abundance of wild Chinook salmon. In Bear Valley Creek, a total of 178 redds were counted along six transects. Overall, this represents a 24% increase from 2009 (143 redds), but represents a 51% decline when compared to more the recent high of 2003 (364 redds) and a 74% decline from the highest counts ever noted during 1961 (675 redds). In Elk Creek, a total of 240 redds were counted along three transects. Overall, the 2010 counts represent a 98% increase from 2009 (121 redds), a 36% decline from the recent high of 2002 (377 redds), and a 63% decline from the historical high of 1961 (654 redds). In Sulphur Creek, a total of 52 redds were counted along two transects during 2010. For Sulphur Creek transects, the 2010 counts represented a 126% increase from 2009 (23 redds), and represented a 44% decline from the recent high of 2002 (93 redds), and an 86% decline from the historical high of 1957 (381 redds).

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INTRODUCTION

Tributaries of the upper Middle Fork Salmon River, including Bear Valley, Elk, and Sulphur creeks possess some of the best remaining spring/summer Chinook salmon spawning habitat in the Snake River basin. IDFG has conducted annual spawning ground surveys on these streams since 1957 to enumerate the number of Chinook salmon redds, primarily, as an index of adult population abundance. Initially, surveys were conducted along fairly long transects (6 - 8 km) using aerial counts or, less often, on foot; however, beginning in about 1989, transects were split into shorter segments (3 - 4 km) and have been surveyed on foot annually during the last week of August (Hassemer 1993).

Despite the abundance of high quality spawning and juvenile rearing habitat, overall numbers of wild Chinook salmon have declined precipitously from highs observed during the late 1950 and 1960's. All Snake River Chinook salmon stocks were listed as threatened under the Endangered Species Act during 1992. Since then, returning adult abundances have remained critically low, except for a three-year period from 2001 - 2003, when adult numbers rebounded temporarily. During 2004 - 05, this trend reversed, and adult abundances have returned to near historical low levels of the late 1990s.

OBJECTIVES

1. To index the abundance of returning wild adult Chinook salmon by counting redds within historical trend monitoring transects in Bear Valley, Elk, and Sulphur creeks during 2008.
2. To compare current redd count information to historical data.

METHODS

Spawning ground surveys were conducted at 11 historical trend monitoring transects in Bear Valley, Elk, and Sulphur creeks (Figure 34) from August 30 through September 2, 2010. The timing of initial surveys conducted along Bear Valley and Elk creeks occurred within the interval of past sampling dates, at a time when nearly all adult Chinook salmon had recently spawned.

All surveying techniques followed the protocol outlined by Hassemer (1992). Prior to conducting surveys, surveyors were required to attend an IDFG sponsored training session taught by experienced biologists. Afterwards, pairs of surveyors walked upstream through each transect. After locating a prospective redd site, surveyors determined and recorded whether they observed a single redd, multiple redds, or a test dig. Redd locations were recorded with hand-held global positioning system units. For each site, surveyors also recorded the number of live and dead adult Chinook salmon observed, as well as their estimated age and sex. Biological samples were collected from salmon carcasses and provided to the Idaho Natural Production Monitoring and Evaluation Project. All survey data were entered and archived in the Spawning Ground Survey (version 2.3.11.0) database.

RESULTS AND DISCUSSION

In Bear Valley Creek, a total of 178 redds were counted along six transects during 2010. Overall, this represents a 24% increase from 2009 (143 redds), but was a 51% decline when compared to the recent high of 2003 (364 redds) and a 74% decline from the highest count ever noted during 1961 (675 redds; Figure 35, 36, and 37). Despite this marked difference between present and historical counts, trend counts in these six transects have increased in each of the last four years. In Bear Valley Creek, redds were concentrated (114 of the 178 redds) in the two transects bracketing the mouth of Elk Creek (WS-10a and WS-9d). The number of redds counted in the four remaining Bear Valley Creek sites was less than 27 each. A total of 44 live adult Chinook salmon and 227 carcasses were observed.

In Elk Creek, a total of 240 redds were counted along three transects during 2010 surveys. Overall, the 2010 count represents a 98% increase from 2009 (121 redds), a 36% decline from the recent high of 2002 (377 redds), and a 63% decline from the historical high of 1961 (654 redds; Figure 38). The majority of redds in Elk Creek (n = 130) were concentrated in the most upstream monitoring sites, WS-11a. Whereas, 77 and 33 redds were counted in the middle (WS-11b) and lower (WS-11c) Elk Creek transects, respectively. A total of 23 live adult Chinook salmon and 146 carcasses were observed.

In Sulphur Creek, a total of 52 redds were counted along two transects during 2010 surveys. This total is over double the count in any of the previous six years. However, 2010 redd counts were still much lower than recent and historical highs (Figure 39). Overall for Sulphur Creek transects, the 2010 count (52 redds) represented a 126% increase from 2009 (23 redds), but represents a 44% decline from the recent high of 2002 (93 redds), and an 86% decline from the historical high of 1957 (381 redds). A total of five live adult Chinook salmon and 33 carcasses were observed.

Over the three monitoring streams and 11 trend monitoring transects combined, a total of 470 redds were counted in 2010. This total is a 64% increase compared to the 2009 count (n = 287), and is the highest count since 2003 (n = 783; Figure 40). Despite a general increasing trend since 2004, total redd counts in this area are still much lower than the high of 1,440 redds counted within these streams during 1957 and the consistently high counts document during the 1960s (only 10 transects were surveyed until 1988). During this decade, cumulative counts in this area exceeded 770 redds in all years except 1965 when 536 redds were counted. Furthermore, total redd counts during 2010 were still 27% less than recent highs documented during 2001 - 2003, when cumulative counts averaged 643 redds for this period.

MANAGEMENT RECOMMENDATIONS

1. Continue to index the abundance of wild adult Chinook salmon by counting redds in Bear Valley, Elk, and Sulphur creeks.
2. Continue to pursue strategies that improve down river and ocean survival of these stocks.

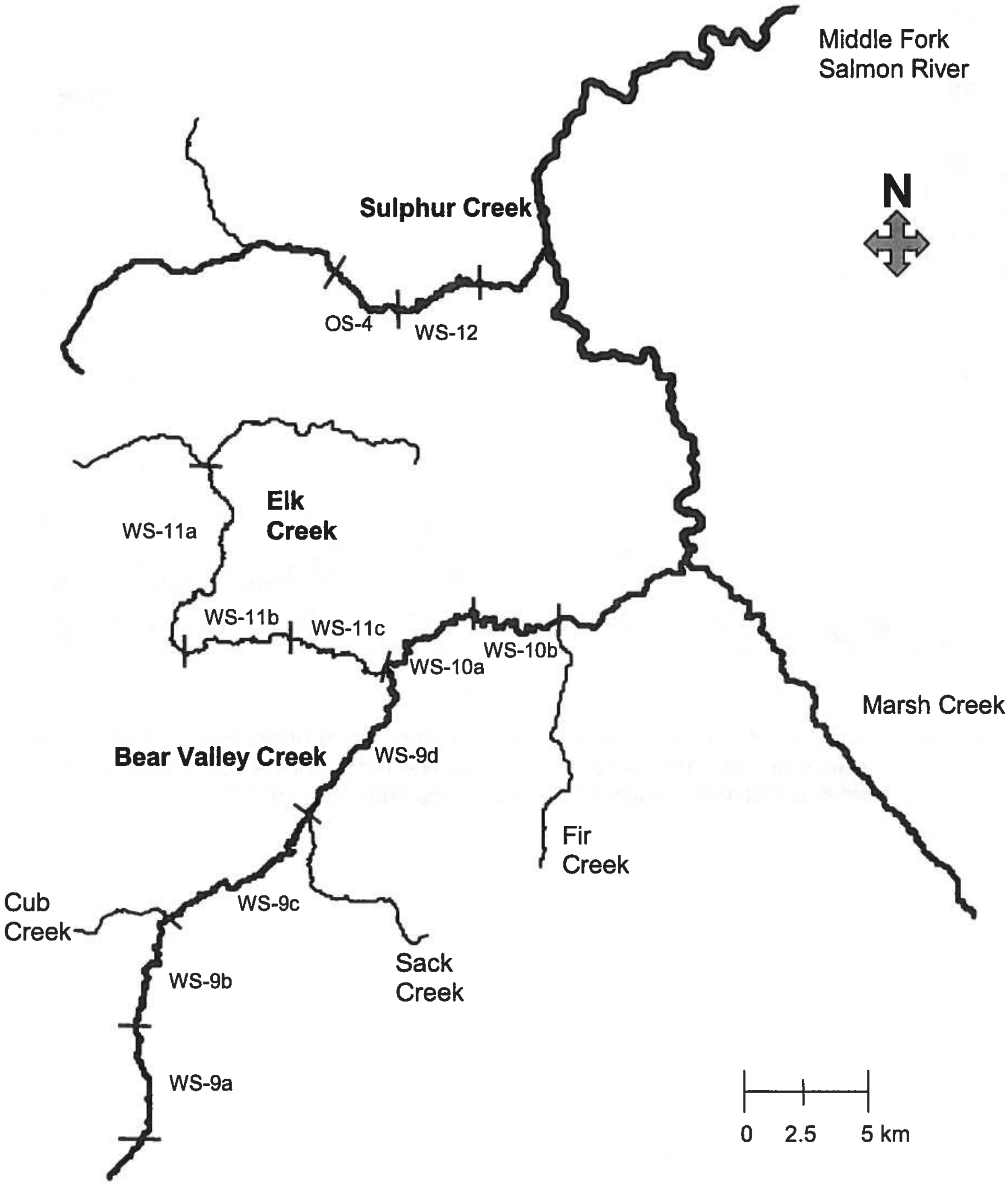

Figure 34. Location of 11 trend monitoring transects on Bear Valley, Elk, and, Sulphur creeks used to index the abundance of wild spring/summer-run Chinook Salmon in the upper Middle Fork Salmon River Drainage, ID. Red lines denote transect boundaries.

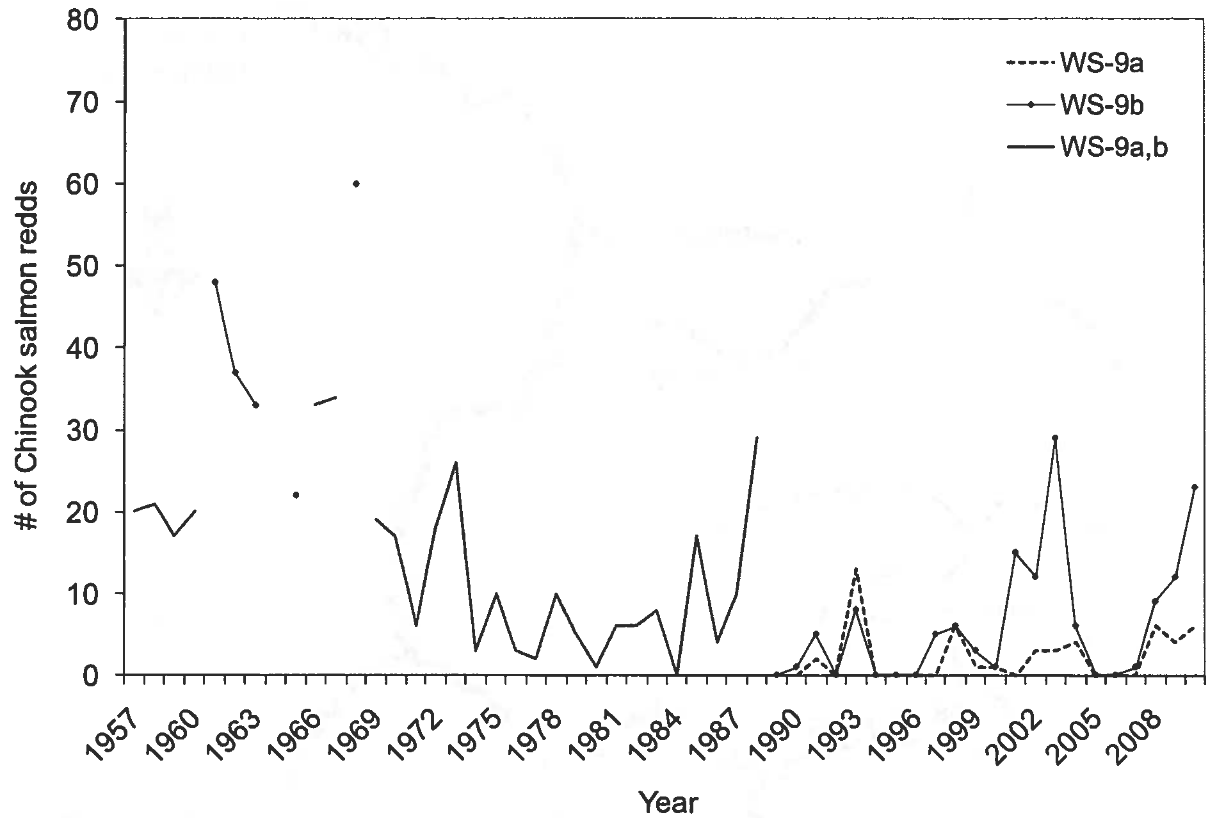

Figure 35. Number of Chinook salmon redds counted along upper Bear Valley Creek index transects from 1957 - 2010. The solid line represents a cumulative count for WS-9a & b that was monitored in most years from 1957 to 1989.

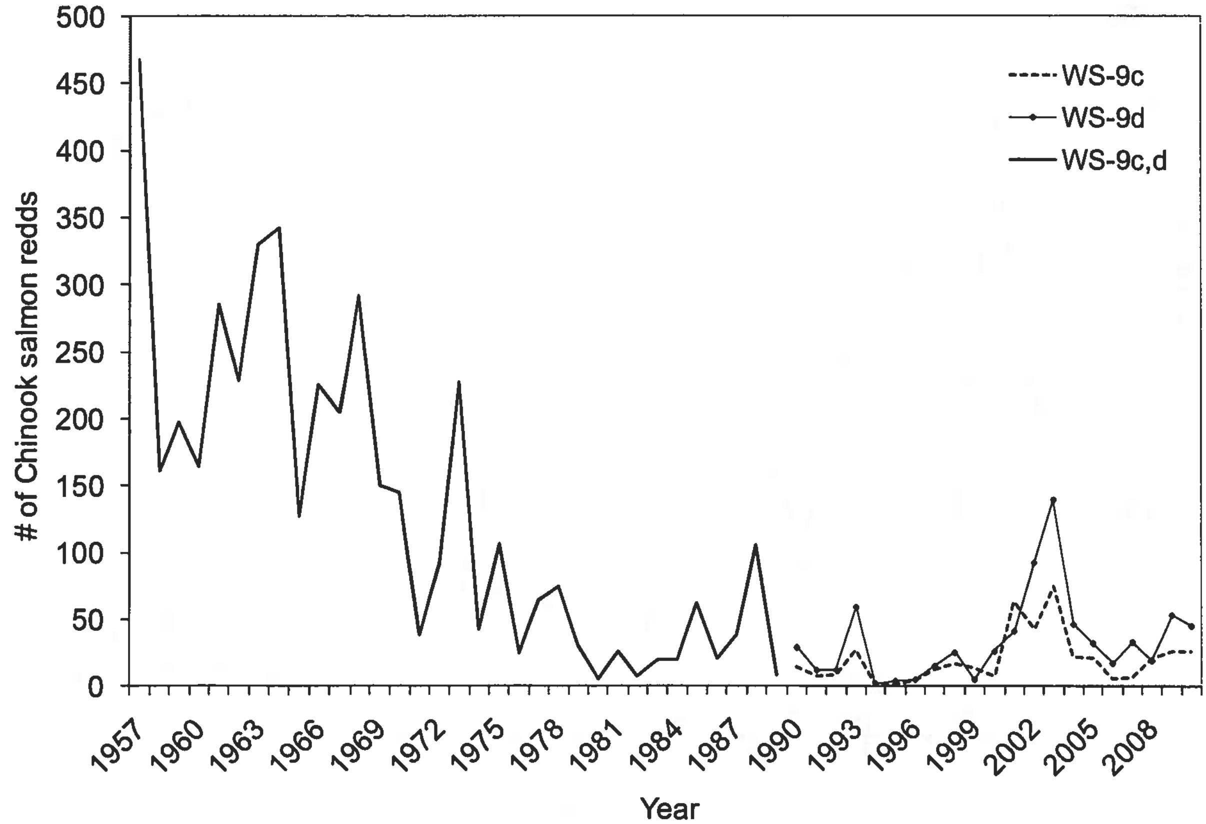

Figure 36. Number of Chinook salmon redds counted along middle Bear Valley Cr. index transects from 1957 - 2010. The solid line represents cumulative counts for WS-9c & d.

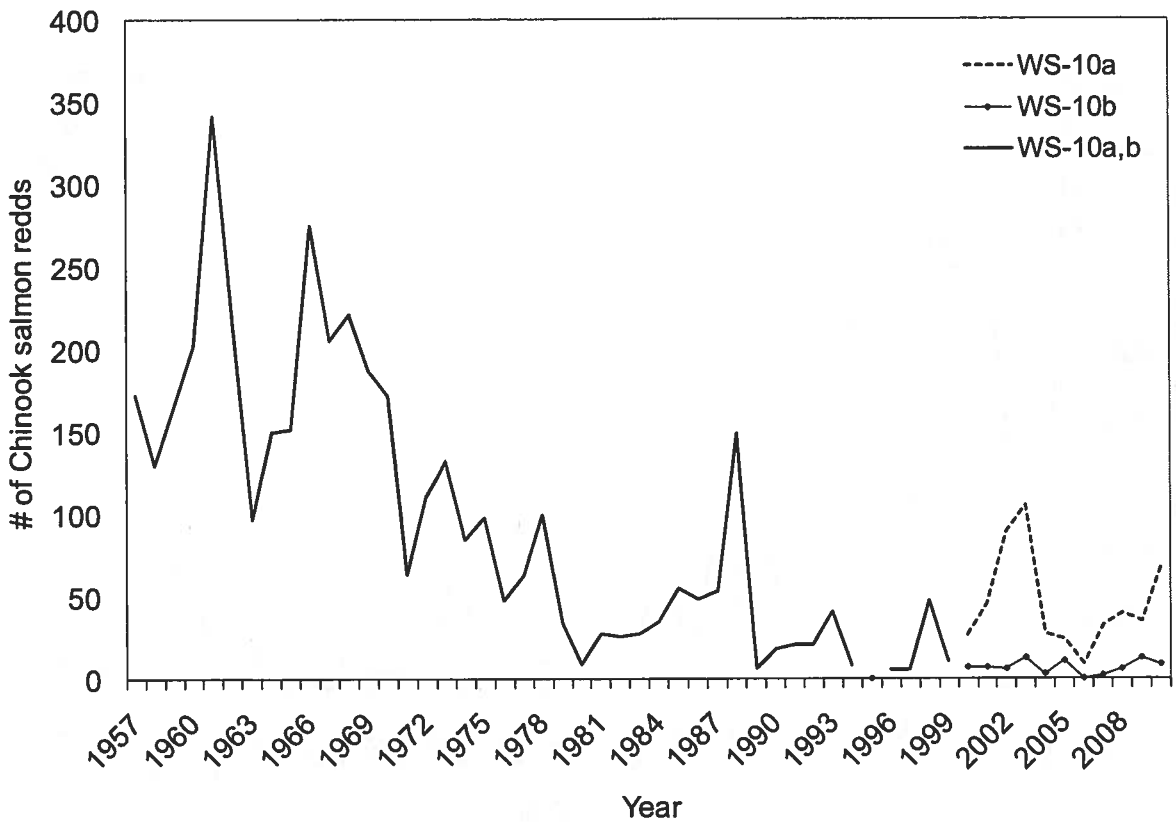

Figure 37. Number of Chinook salmon redds counted along lower Bear Valley Cr. index transects from 1957 - 2010. The solid line represents cumulative counts for WS-10 a & b.

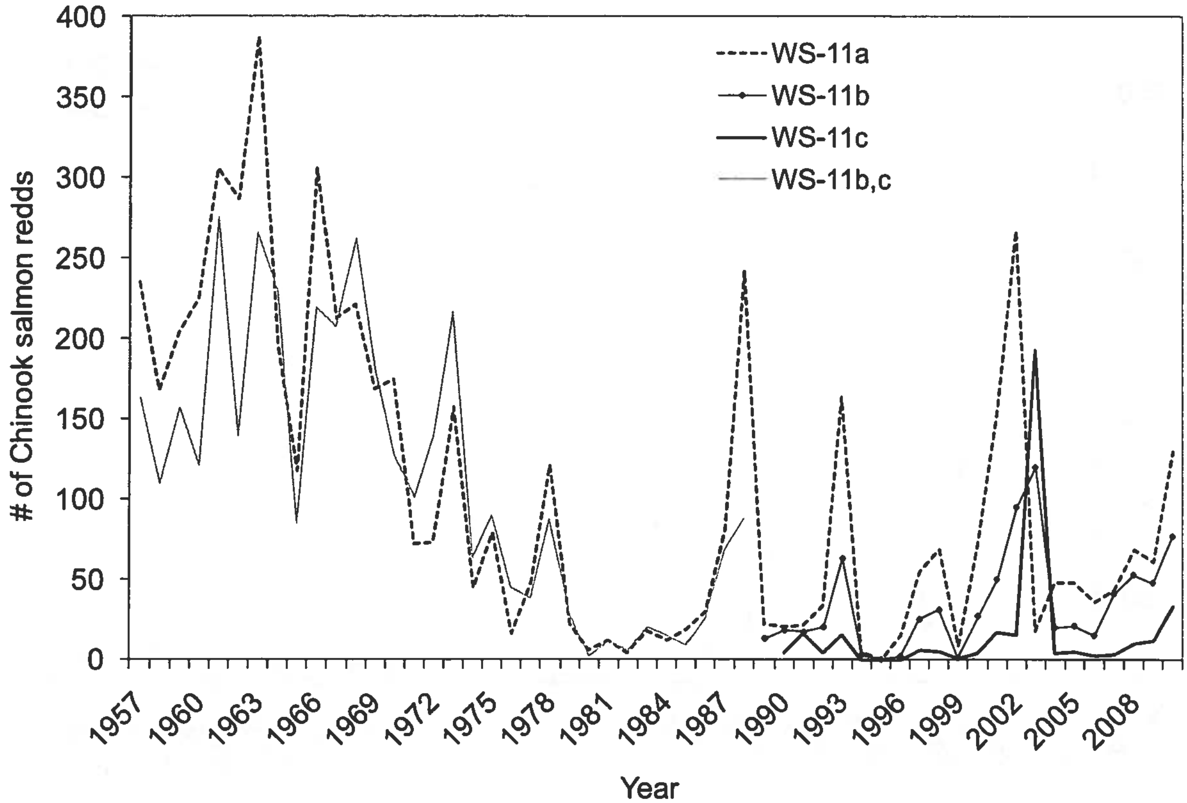

Figure 38. Number of Chinook salmon redds counted along Elk Creek index transects from 1957 - 2010. The solid line represents a cumulative count for WS-11b and WS-11c, whereas all other lines represent individual transects.

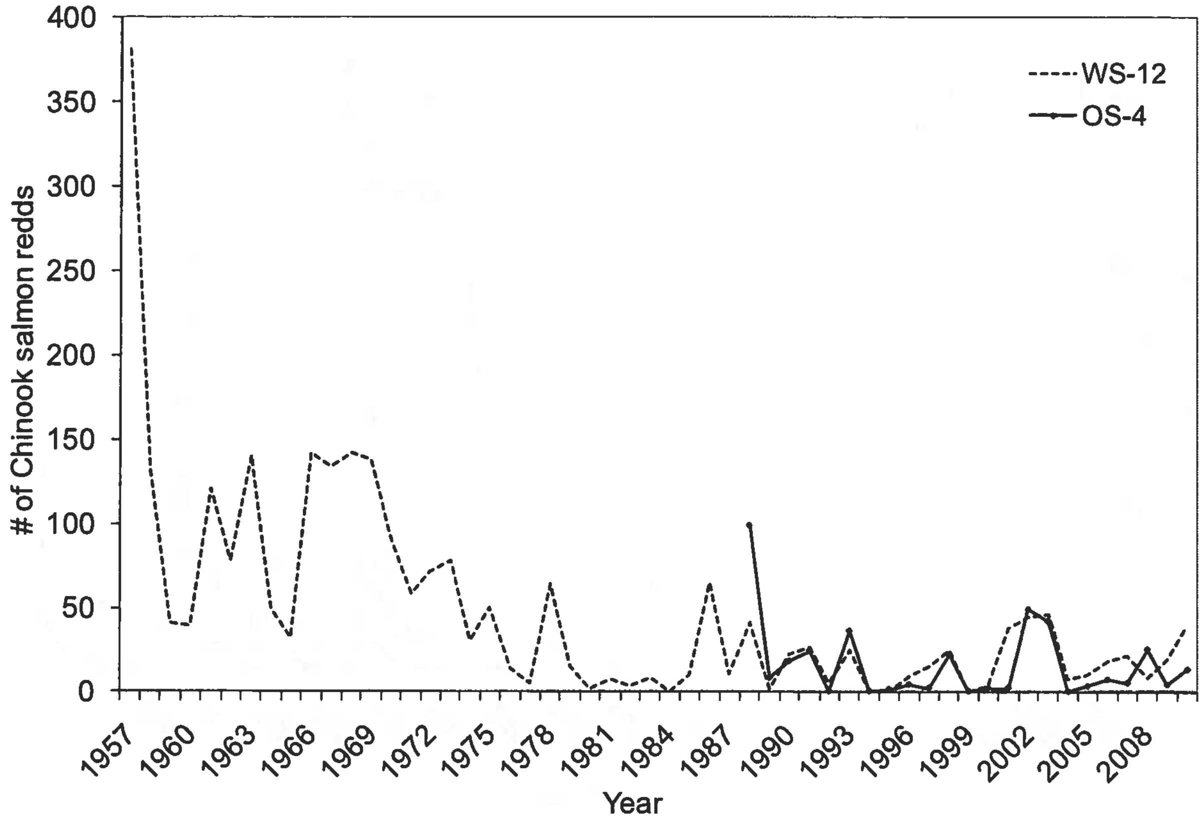

Figure 39. Number of Chinook salmon redds counted along Sulphur Creek index transects from 1957 - 2010.

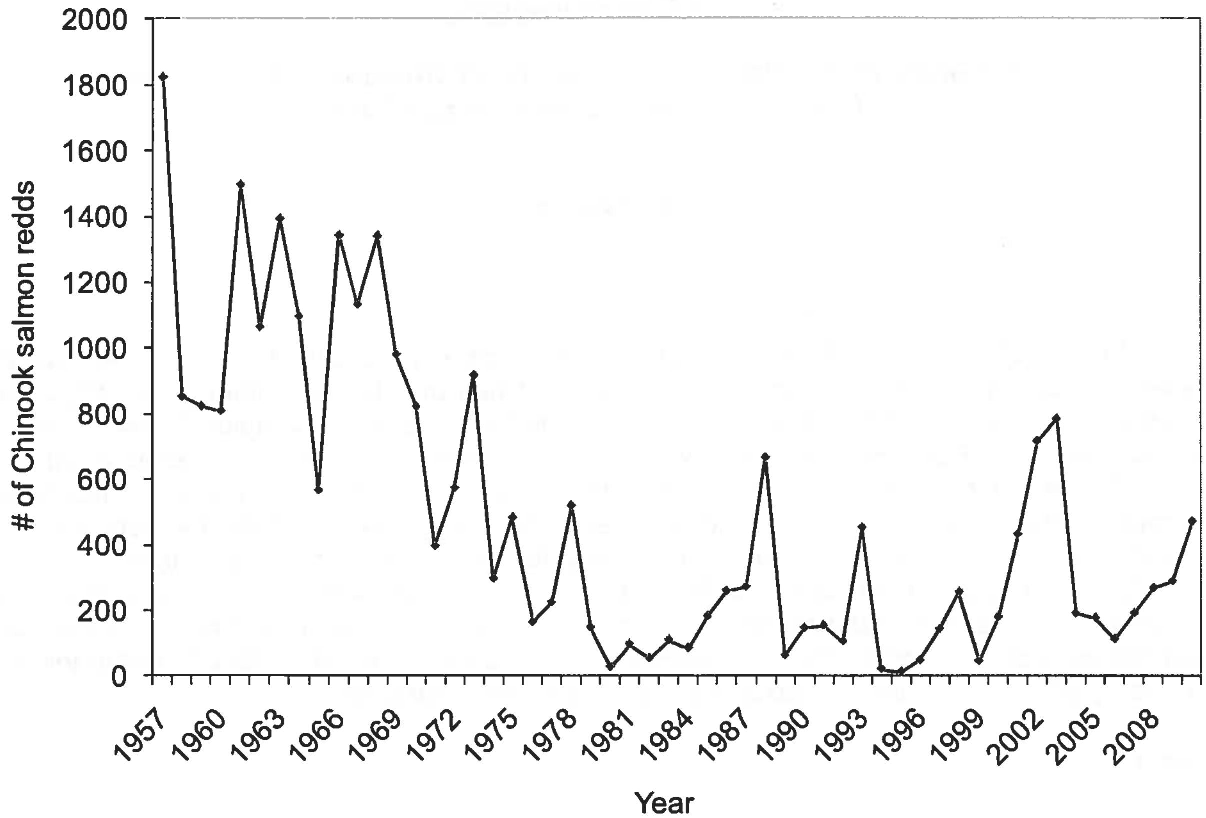

Fig 40. Cumulative number of Chinook Salmon redds counted along 11 trend monitoring sites in Bear Valley, Elk, and Sulphur creeks from 1957 through 2010. Counts for the upper Sulphur Creek transect (OS-4) from 1957 - 1987 were estimated with linear regression techniques and data collected from 1988 - 2010.

River and Stream Investigations

South Fork Boise River Tributaries Evaluation Of Rainbow Trout Populations Downstream Of Anderson Ranch Dam

ABSTRACT

Five tributaries to the South Fork Boise River (SFBR) were sampled in 2010 to evaluate presence, population density, and size distribution of fish populations within these tributaries. Seven sites in Dixie, Granite, Pierce, Rock, and Rough creeks were sampled between June 8 and July 26, 2010. Rainbow trout were collected in two of the five streams sampled in 2010. A total of 48 fish were collected at three sites in Pierce and Rock creeks. No redband trout were sampled in Dixie, Granite, and Rough creeks, and only Dixie Creek contained enough water to support a fish population. Rainbow trout density for the three sites ranged from 6 to 12.1 fish/100 m^2 in Pierce Creek, and 0 to 8.7 fish/100 m^2 in Rock Creek. Nearly 70% of the fish captured were less than 100 mm, and length frequency distributions show that all fish captured were between 50 - 180 mm. The 2010 stream surveys provided an important first step towards prioritizing SFBR tributaries for habitat work such as barrier removal.

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INTRODUCTION

The South Fork Boise River (SFBR) downstream of Anderson Ranch Dam is a nationally renowned tail-water trout fishery and was the first river section in Southwest Idaho to be managed under "Trophy Trout" regulations. The total reach is 43 km, with road access along the upper 16 km, and the remaining section accessible only by non-motorized boat. Regulations prohibit the use of bait and barbed hooks from Neal Bridge (Forest Road 189) upstream to Anderson Ranch Dam. Rainbow trout harvest is restricted to 2 fish, none under 20 inches. The fishery is supported by a population of wild rainbow trout and mountain whitefish. Migratory bull trout are present at very low densities, and native nongame fish include largescale suckers, northern pikeminnow and sculpin. Approximately 15,000 rainbow trout were stocked annually in SFBR between Anderson Ranch Dam and Cow Creek Bridge until 1976, when management emphasis shifted towards wild trout (Beach 1975; Moore et al. 1979).

Rainbow trout populations in the SFBR have been monitored in a 9.6 km section upstream from Danskin Bridge every three years since 1994 (Butts et al. 2011). Mark-recapture techniques are used to estimate abundance of trout and mountain whitefish in three sections of the SFBR. Results have suggested that rainbow trout populations in the SFBR have been relatively stable, but the relative absence of trout in the 200 to 400 mm length range upstream of Danskin Bridge is puzzling. The numbers of trout greater than 400 mm are currently providing an excellent fishery despite the relative lack of smaller trout in the roaded survey sections. A population survey in the canyon section downstream of Danskin Bridge in 2008 showed that rainbow trout between 250 - 400 mm were present in higher proportions than observed in the tail-water section (Kozfkay et al. 2010). The SFBR wild trout population is thought to be supported primarily through main-stem spawning of fish with little input from tributaries, as migration barriers are known to be present on most tributaries with spawning habitat (Moore et al. 1979).

Recently, interest has increased in restoring connectivity to tributaries to the SFBR below Anderson Ranch Dam. Specifically, biologists wish to determine whether or not the tributaries currently have fish populations, contain spawning habitat, if barriers to fish migration exist, and if the potential exists to provide spawning opportunities if barriers were removed. Surprisingly, there is little information on fish populations within these tributaries. A number of tributaries were sampled in 2008 by United States Forest Service (USFS) biologists to obtain samples for a genetic study on rainbow and redband trout within the SFBR drainage. However, little or no population information was collected during these surveys. Prior to this, Moore et al. (1979) characterized the majority of the SFBR tributaries below Anderson Ranch and evaluated streams both for the presence of spawners and spawning habitat. However, changes in land use practices, roads, and climate may have altered habitat and fish communities over the past 30 years. To properly describe current habitat conditions and prioritize barrier removal projects, new surveys were initiated in 2010.

METHODS

Five tributaries to the SFBR were sampled in 2010 to evaluate fish spp. presence, population density, and size distribution. Seven sites in Dixie, Granite, Pierce, Rock, and Rough creeks were sampled between June 8 and July 26, 2010 (Figure 41). Sample sites were selected from a 1:100,000 hydrography layer through the Environmental Protection Agency's Environmental Monitoring and Assessment Program (see Stevens and Olsen 2004). Sampling occurred during base flow conditions so that streams could be evaluated as to whether they contained enough water to support fish and so that migrations barriers could be better assessed.

At each site that contained enough water to support fish, we used depletion electrofishing to determine the abundance of salmonids, using a backpack electrofisher (Smith-Root Model 15-D) with pulsed DC. Block nets were installed at the upper and lower ends of the sites to prevent fish from leaving or entering a study site during the survey. Study sites were generally 100 m in length of shockable stream; sections of stream where vegetation was too thick to sample effectively, were not included in the sample site. Fish were identified, enumerated, measured to the nearest mm (total length, TL) and g, and released downstream of the study sites. Nongame fish and amphibian species were also recorded if observed. Maximum-likelihood abundance and variance estimates were calculated with the MicroFish software package (Van Deventer and Platts 1989). When no trout were captured on the final pass, we estimated abundance to be the total catch. Because electrofishing is characteristically size selective (Sullivan 1956; Reynolds 1996), trout were separated into two length groups (<100 mm TL and ≥100 mm TL) and abundance estimates were calculated individually for each size group. Depletions were attempted only for salmonids, whereas relative abundance was recorded for all nongame fish and amphibian species.

Various habitat measurements were recorded at ten equally spaced transects within the sample site. Stream width was measured at each transect and depth (m) was measured at ¼, ½, and ¾ distance across the channel. The sum of these depth measurements was divided by four to account for zero depths at the stream margins for trapezoidal channels (Platts et al. 1983; Arend 1999). Wetted stream width (m) was calculated from the average of all transect measurements. In most cases, stream temperature (°C) and conductivity (µS/cm) were measured at the bottom of a site with a calibrated hand-held meter accurate to ± 2%. Various other habitat measurements such as percent substrate composition, percent shading, and bank stability were measured, but the results are not reported here and are instead stored in the IDFG Standard Stream Survey database.

RESULTS

Rainbow trout were collected in two of the five streams sampled in 2010. A total of 48 fish were collected at three sites in Pierce and Rock creeks. No redband trout were sampled in Dixie, Granite, and Rough creeks, and only Dixie Creek contained enough water support a fish population. Sculpin and tailed frogs *Ascaphus truei* (adults and tadpoles) were also collected in Pierce and Rock creeks.

Rainbow trout density for the three sites ranged from 6 to 12.1 fish/100 m^2 in Pierce Creek, and 0 to 8.7 fish/100 m^2 in Rock Creek (Table 21). Nearly 70% of the fish captured were less than 100 mm and capture probability for that size ranged from 50 - 83%. The lower capture probability of Pierce Creek is likely a result of the electrofisher being unable to reach all areas of the stream within each section because of the thick vegetation surrounding the stream. Capture probability for fish ≥ 100 mm ranged from 75 - 100% in both streams. Length frequency distributions show that all fish captured were between 50 - 180 mm (Figure 42).

DISCUSSION

These stream surveys provided important first steps towards prioritizing SFBR tributaries for habitat work such as barrier removal. Granite and Rough creeks did not contain adequate water to sustain fish populations in June. However, 100 spawning female rainbow trout were observed in Rough Creek and 30 in Granite Creek in 1978 (Moore et al. 1979). The previous year, which was considered a drought year, no spawning fish were observed in either stream. Therefore it is possible that both streams have historically contained large, fluvial rainbow trout under higher stream flows. Additionally, both streams contain culverts at the FS113 road crossing, which appears to limit upstream migration from the SFBR main stem, at least during low flows.

Pierce Creek sites 1 and 5 contained rainbow trout, sculpin, and tailed frogs and site 9 was high gradient and low flow, preventing fish from residing in the upper drainage. Pierce Creek does have a formidable culvert at the FS113 crossing and the 2010 sampling likely occurred too late to observe any spawners using the tributary. In 1977 and 1978, 100 and 200 female spawners were estimated to have used Pierce Creek, respectively. Pierce Creek also receives a great deal of sediment and silt from erosion below a poorly installed culvert on the Smith Prairie Grade (FS113). Given that Pierce Creek contains fair densities of smaller redband trout and good spawning habitat; it should be considered a high priority for future habitat improvements.

Rock Creek was the largest stream sampled in 2010 and appears to offer good trout habitat throughout much of its drainage. Fish were found higher up in the drainage at Rock Creek site 9, approximately 7.5 km upstream from the confluence of SFBR. However, at site 5, 3.7 km above the confluence, no fish were collected and water temperatures reached 31 °C. The warm temperatures may be a result of irrigation withdrawal and returns along the Smith Prairie. In addition, approximately 3 km above the confluence with SFBR, FS113 crosses Rock Creek, with a culvert that appears to be a substantial barrier to upstream migration. Improving this culvert has the potential to open up an addition 5 km of spawning habitat above the culvert. However, irrigation diversion and practices on Rock Creek need to be further investigated to understand the source for the high stream temperatures at site 5. Finally, Neville and Dunham (In Press) found that 32 fish collected from Rock Creek in 2008 were rainbow x cutthroat trout hybrids. Although there appears to be no barriers to downstream movement, further discussion will be needed regarding the implications of reconnecting Rock Creek to SFBR in terms of upstream migration. Sites below the culvert will need to be sampled in 2011 to assess the fish community below the barrier and whether summer stream temperature may limit trout residence downstream to the confluence.

More tributary sampling will be conducted in 2011, with most in the roadless section downstream of Danskin Bridge. In addition, the presence of an irrigation return or intermittent dewatering on Rock Creek will be further investigated.

MANAGEMENT RECOMMENDATIONS

1. Install temperature a logger in Rock Creek above culvert crossing to assess seasonal stream temperatures.

2. Continue SFBR tributary inventories downstream of Danskin Bridge.

Table 21. Rainbow trout population and density (fish/100 m^2) estimates by length group, and stream temperatures (°C) at 7 monitoring sites during June-July 2010 in the Dixie, Granite, Pierce, Rock, and Rough Creek drainages.

				< 100 mm		≥ 100 mm		Total	
Stream	Site	Temp (°C)	Passes	Estimate	95% CI	Estimate	95% CI	Estimate	fish/100m^2
Dixie Creek	DX1	11.1	1	-	-	-	-	-	-
Granite Creek	GC1	18.4	1	-	-	-	-	-	-
Pierce Creek	PC1	15.2	3	19	12-26	3	2-4	22	12.1
Pierce Creek	PC2	13.1	3	6	2-10	3	3-3	9	6
Pierce Creek	PC5	-	1	-	-	-	-	-	-
Rock Creek	RK5	31.2	1	-	-	-	-	-	-
Rock Creek	RK9	14.8	2	10	8-12	9	9-9	19	8.7
Rough Creek	RG1	18.1	1	-	-	-	-	-	-

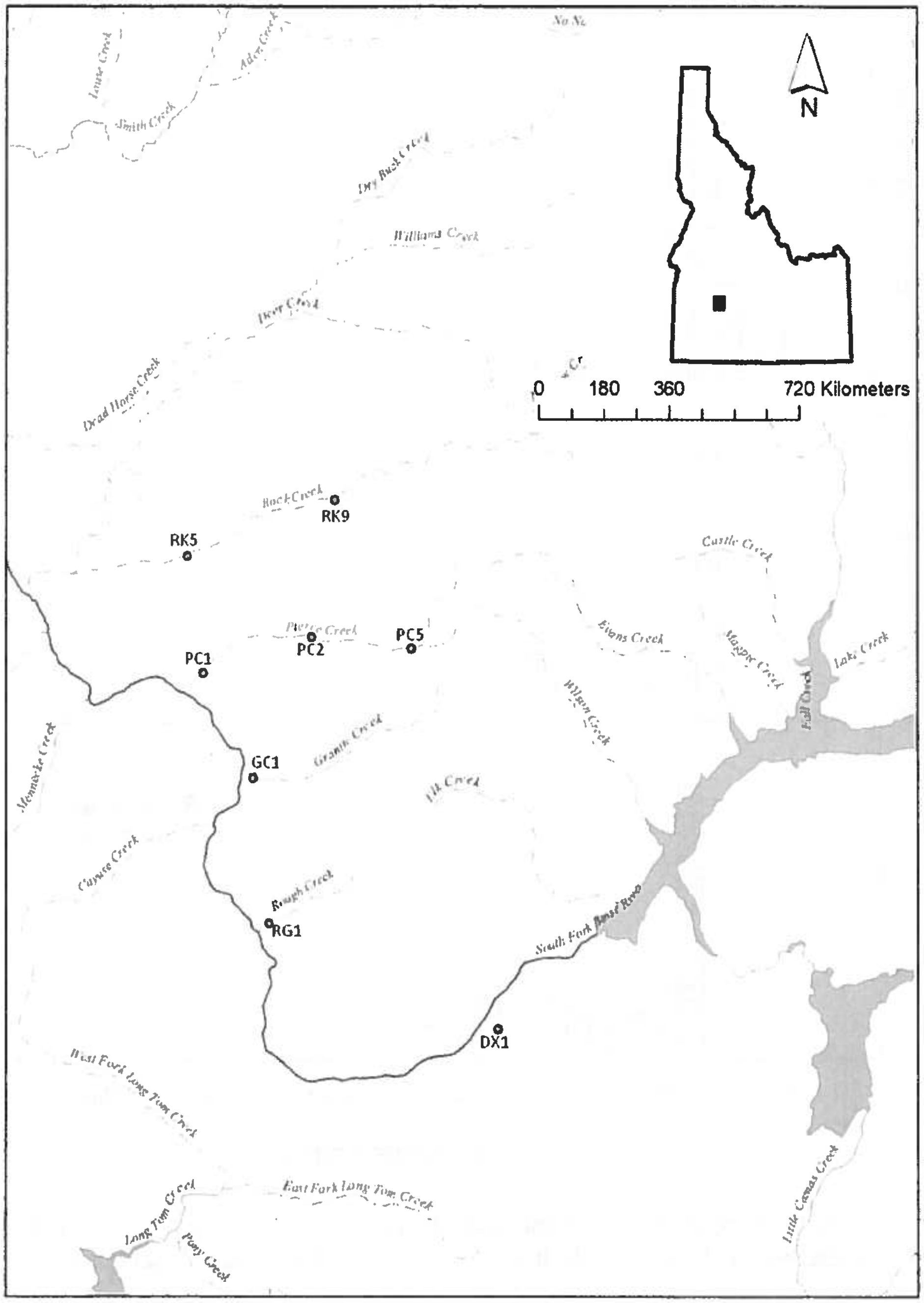

Figure 41. Map of the South Fork Boise River drainage, Idaho and the seven sites sampled to assess fish populations within the drainage during June - July 2010.

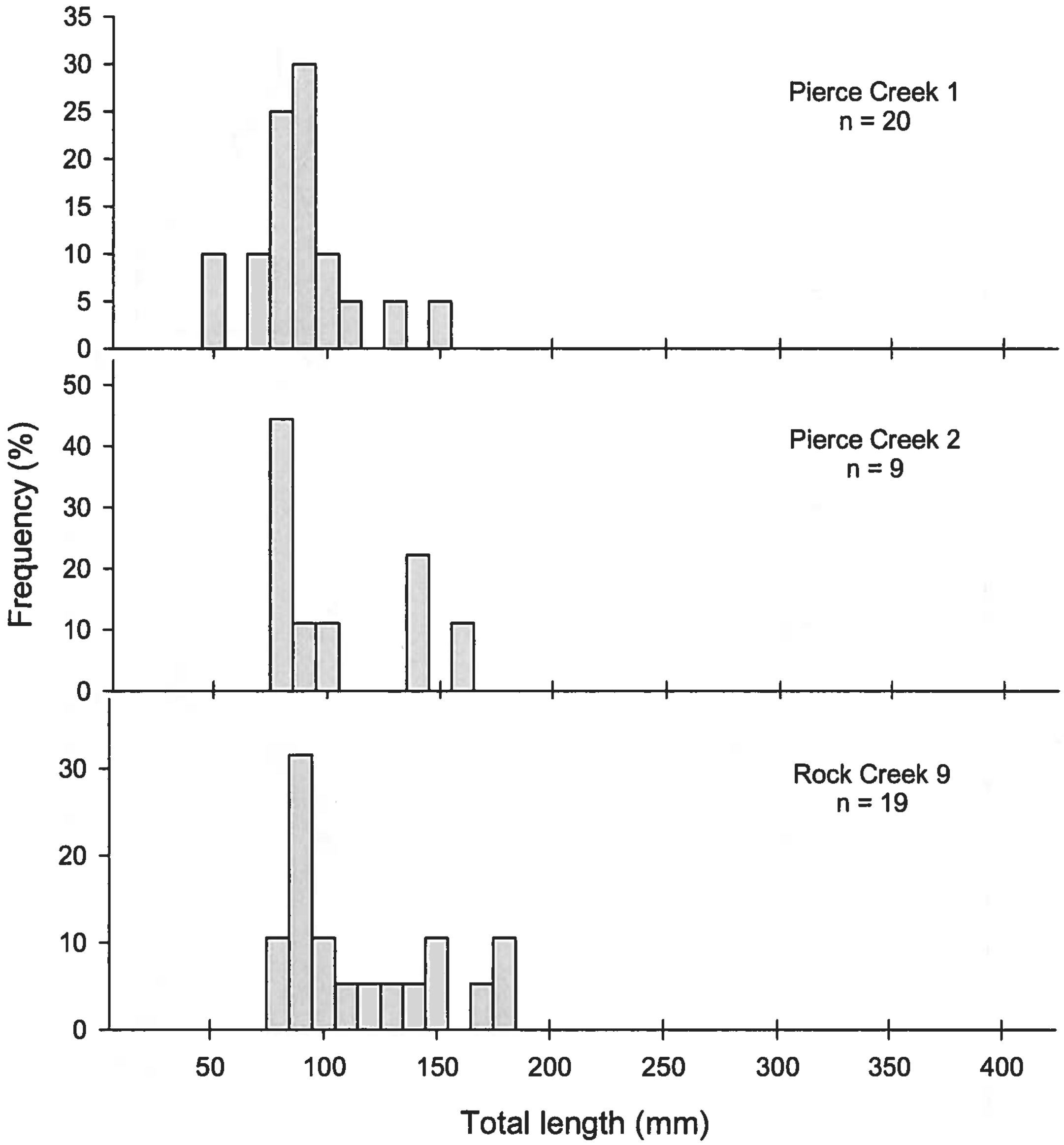

Figure 42. Rainbow trout length distribution (%) of fish captured in June - July 2010 at 3 sites with fish present in the Pierce and Rock Creek drainages.

River and Stream Investigations

Bull trout and Redband Trout Population Monitoring in Headwater Tributaries of The Middle Fork Boise River

ABSTRACT

The Yuba River and three surrounding tributaries were sampled during 2010 to evaluate bull trout presence, trout population densities, and size distributions of fish populations within these tributaries. A total of 325 redband trout were captured in 2010 and all 11 sites contained fish. A single juvenile bull trout was collected in both Decker Creek (D04) and the Yuba River (Y0.2) and sculpin and tailed frogs were observed in all streams. No migratory bull trout were captured during the 2010 sampling. Redband trout density for the 11 sites averaged 12.4 ± 4 trout/100 m^2 of stream (mean ± 90% CI) and individual estimates ranged from 2.5 to 20.8 fish/100 m^2. The lowest densities of redband trout occurred in the three sites sampled in the Yuba River, while Decker, Grouse, and James creeks contained similar densities. Redband trout ≥ 100 mm comprised 80% of the fish captured suggesting that rearing areas exist higher in the drainage. For sites with previous data, average fish density increased 111%, from 5.9 fish/100m^2 in 2000 to 12.4 fish/100m^2 fish in 2010. Only 2 of the 11 sites sampled in 2010 suggested a decrease in fish density over the ten-year period (Yuba River: YUB-2 and Y0.2). Resident bull trout density within these streams is low, but appears to be stable.

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INTRODUCTION

The Yuba River is a 4th order tributary to the Middle Fork Boise River (MFBR) located near Atlanta, in Elmore County, Idaho. Yuba River enters the MFBR approximately 60 km upstream of Arrowrock Reservoir and 0.3 km upstream of Kirby Dam, which prior to the construction of a fish ladder in July 1999, had blocked upstream fish passage for approximately 90 years. The drainage has recently been designated as critical habitat for bull trout *Salvelinus confluentus* recovery by the U.S. Fish and Wildlife Service during September 2010. In addition to bull trout, the drainage also contains native redband trout *Oncorhynchus mykiss gairdneri* and sculpin *Cottus spp.* Fish bearing tributaries to the Yuba River include Decker, Grouse, Trail, and Sawmill creeks. James Creek is adjacent to the Yuba River, entering the MFBR just downstream of Kirby Dam, and is also known to contain redband and bull trout.

Both resident and migratory bull trout utilize the MFBR and its tributaries. Migratory bull trout (adfluvial and fluvial) winter in Arrowrock Reservoir or the South Fork Boise River (SFBR) and typically enter the lower MFBR in mid-May through early June and proceed to migrate upstream towards a number of higher elevation spawning tributaries in the North Fork Boise River (NFBR) and MFBR drainages (Flatter 2000). Spawning generally occurs in August and September, after which fish move back downstream to wintering areas.

Kirby Dam was built in the early 1900s to provide electricity and water for nearby mining activities. The structure had to be rebuilt in 1992 after the original earthen material failed. The dam is owned by the U.S. Forest Service (USFS) and operated by the Atlanta Hydropower Corporation to provide electricity to the town of Atlanta. After functioning as a barrier to upstream fish migration for approximately 90 years, a fish ladder was built by Idaho Department of Fish and Game (IDFG) and USFS in 1999. The fish ladder, when functioning properly, reconnects the upper headwaters of the MFBR to the mainstem and opens access to potential spawning tributaries such as the Yuba River drainage to migratory bull trout.

METHODS

The Yuba River and three surrounding tributaries were sampled at 11 sites in 2010 to evaluate bull trout presence, trout population density, and size distribution of fish populations (Figure 43). Previously sampled sites were revisited in Decker, Grouse, and James creeks and the Yuba River. Sites were located using descriptions and coordinates listed in Flatter et al. (2003). Sampling occurred in August to maximize the chance of encountering migratory bull trout from Arrowrock Reservoir. All streams were sampled at base flow conditions to allow for maximum electrofishing efficiency.

At each site, we used depletion electrofishing to determine the abundance of salmonids, using 1-2 backpack electrofishers (Smith-Root Model 15-D) with pulsed DC. Block nets were installed at the upper and lower ends of the sites to prevent fish from leaving or entering a study site during the survey. Study sites were variable in length and in most cases we used the site lengths that were sampled in Flatter et al. 2003. Fish were identified, enumerated, measured to the nearest millimeter (total length, TL) and gram, and released downstream of the study sites. Nongame fish and amphibian species were also recorded if observed. Maximum-likelihood abundance and variance estimates were calculated with the MicroFish software package (Van Deventer and Platts 1989). When no trout were captured on the final pass, we estimated abundance to be the total catch. Because electrofishing is characteristically size selective (Sullivan 1956; Reynolds 1996), trout were separated into two length groups (<100 mm TL and

≥100 mm TL) and abundance estimates were calculated individually for each size group. Depletions were attempted only for salmonids, whereas relative abundance was recorded for all nongame fish and amphibian species.

Various habitat measurements were recorded at ten equally spaced transects within the sample site. Stream width was measured at each transect and depth (m) was measured at ¼, ½, and ¾ distance across the channel. The sum of these depth measurements was divided by four to account for zero depths at the stream margins for trapezoidal channels (Platts et al. 1983; Arend 1999). Wetted stream width (m) was calculated from the average of all transect measurements. In most cases stream temperature (°C) and conductivity (µS/cm) were measured at the bottom of a site with a calibrated hand-held meter accurate to ± 2%. Various other habitat measurements such as percent substrate composition, percent shading, and bank stability were measured. Habitat data are not presented in this report, but were archived in the IDFG stream survey database.

RESULTS

A total of 325 redband trout were captured in 2010 and all 11 sites contained fish. Total catch of redband trout ranged from 9 at James Creek (J02) to 90 fish at Decker Creek (D02). A single juvenile bull trout was collected in both Decker Creek (D04) and the Yuba River (Y0.2) and sculpin *sp.* and tailed frogs were observed in all streams. No migratory bull trout were captured during the 2010 sampling.

Redband trout density for the 11 sites averaged 12.4 ± 4 trout/100 m^2 of stream (mean ± 90% CI) where individual estimates ranged from 2.5 to 20.8 fish/100 m^2 (Table 22). The lowest densities of redband trout occurred in the 3 sites sampled in the Yuba River while Decker, Grouse, and James creeks contained similar densities. Redband trout ≥ 100 mm comprised 80% of the fish captured suggesting that rearing areas for smaller fish exist higher in the drainage. Capture probabilities for the two size classes of trout at 68% for fish <100 mm and 71% for fish ≥100 mm. Length distributions for redband trout in each stream were quite similar (Figure 44).

The sites at Yuba River, Decker, Grouse, and James creeks were previously sampled in 2000 (Flatter et al. 2003). Two sites on the Yuba River (YU-1 and YUB-2) were sampled via snorkeling, and 2-pass electrofishing depletion estimates were conducted at the remaining sites. Average fish density increased 111%, from 5.9 fish/100 m^2 in 2000 to 12.4 fish/100 m^2 fish in 2010 (Table 23). Only 2 of the 11 sites sampled in 2010 suggested a decrease in fish density over the ten year period (Yuba River: YUB-2 and Y0.2). However fish density in one of these, Yuba River YUB-2, was originally estimated with a snorkeling survey, so differences in methods make comparisons difficult.

DISCUSSION

Redband trout abundance appears to have increased in the Yuba River drainage during the past 10 years, at least at the sites that were re-sampled in 2010. Angling regulations during this period have not changed, but land use or forest management practices may have changed in the drainage during this period. Much of the lower drainage was affected by the Hot Creek Fire in 2003, and the Yuba River in particular has very little canopy cover and many sediment sources given the loose granitic nature of the surrounding soils.

Resident bull trout density within these streams is low but appears to be stable. In 2000, a single bull trout was collected in Decker Creek (D0.4) and Yuba River (Y0.2; Flatter et al. 2003). Two bull trout were also collected in Grouse Creek in 2000. Migratory bull trout were not observed in 2000 or 2010, therefore it is difficult to assess whether or not the Kirby Dam fish ladder has allowed for migratory fish to access the drainage. However, sampling in 2000 occurred in July, which may not have allowed enough time for migratory fish to reach the drainage. Additionally, sampling a small number of stream sections does not provide a good indication as to whether or not migratory fish are using the Yuba drainage. A better approach may be to disregard obtaining population estimates and electroshock long stream sections or seek additional reaches.

MANAGEMENT RECOMMENDATIONS

1. Complete sampling at trend sites in the upper Yuba River, and Sawmill and Trail creeks.

2. Conduct qualitative electrofishing in upper reaches of Yuba River to confirm presence of adfluvial bull trout.

Table 22. Redband trout population and density (fish/100 m^2) estimates by length group at 11 monitoring sites during August 2010 in the Yuba River and James Creek drainages.

			< 100 mm		≥ 100 mm		Total	
Stream	Site (length-m)	Passes	Estimate	95% CI	Estimate	95% CI	Estimate	fish/100m^2
Decker Creek	D02 (110)	2	19	18-20	76	68-84	95	12.6
Decker Creek	D04 (53)	2	7	6-8	55	46-64	62	19.3
Grouse Creek	G00 (23)	2	4	2-6	13	11-15	17	10.8
Grouse Creek	G01 (28)	2	1	1-1	15	12-18	17	13.2
Grouse Creek	G02 (26)	2	9	6-6	17	15-19	28	20.1
Grouse Creek	G03 (48)	3	5	5-5	12	3-21	16	7.3
Yuba River	Y0.2 (80)	2	7	7-7	12	6-18	18	2.5
Yuba River	YUB-2 (66)	2	6	6-6	26	22-30	36	4.8
Yuba River	YU-1 (100)	2	1	1-1	34	24-44	35	3.9
James Creek	J02 (21)	2	4	1-9	5	4-6	9	20.7
James Creek	J1.2 (35)	3	8	7-9	12	10-14	20	20.8

Table 23. Comparison of redband trout population and density estimates (fish/100 m^2) over the last ten years (2000-2010) at monitoring sites in the Yuba River and James Creek drainages.

		2000			2010		
Stream	Site	Estimate	95% CI	fish/100m^2	Estimate	95% CI	fish/100m^2
Decker Creek	D02	19	15-34	5.9	95	87-103	12.6
Decker Creek	D04	11	11-14	5.4	62	53-71	19.3
Grouse Creek	G00	6	6-7	3.1	17	15-19	10.8
Grouse Creek	G01	7	7-8	8.3	17	12-22	13.2
Grouse Creek	G02	9	9-11	12.3	28	14-22	20.1
Grouse Creek	G03	5	5-7	2.8	16	11-21	7.3
Yuba River	Y0.2	12	12-13	4.2	18	16-20	2.5
Yuba River	YUB-2	-	-	8.76	36	20-52	4.8
Yuba River	YU-1	-	-	0.6	35	25-45	3.9
James Creek	J02	8	8-10	7.1	9	6-12	20.7
James Creek	J1.2	6	6-7	5.9	20	8-22	20.8

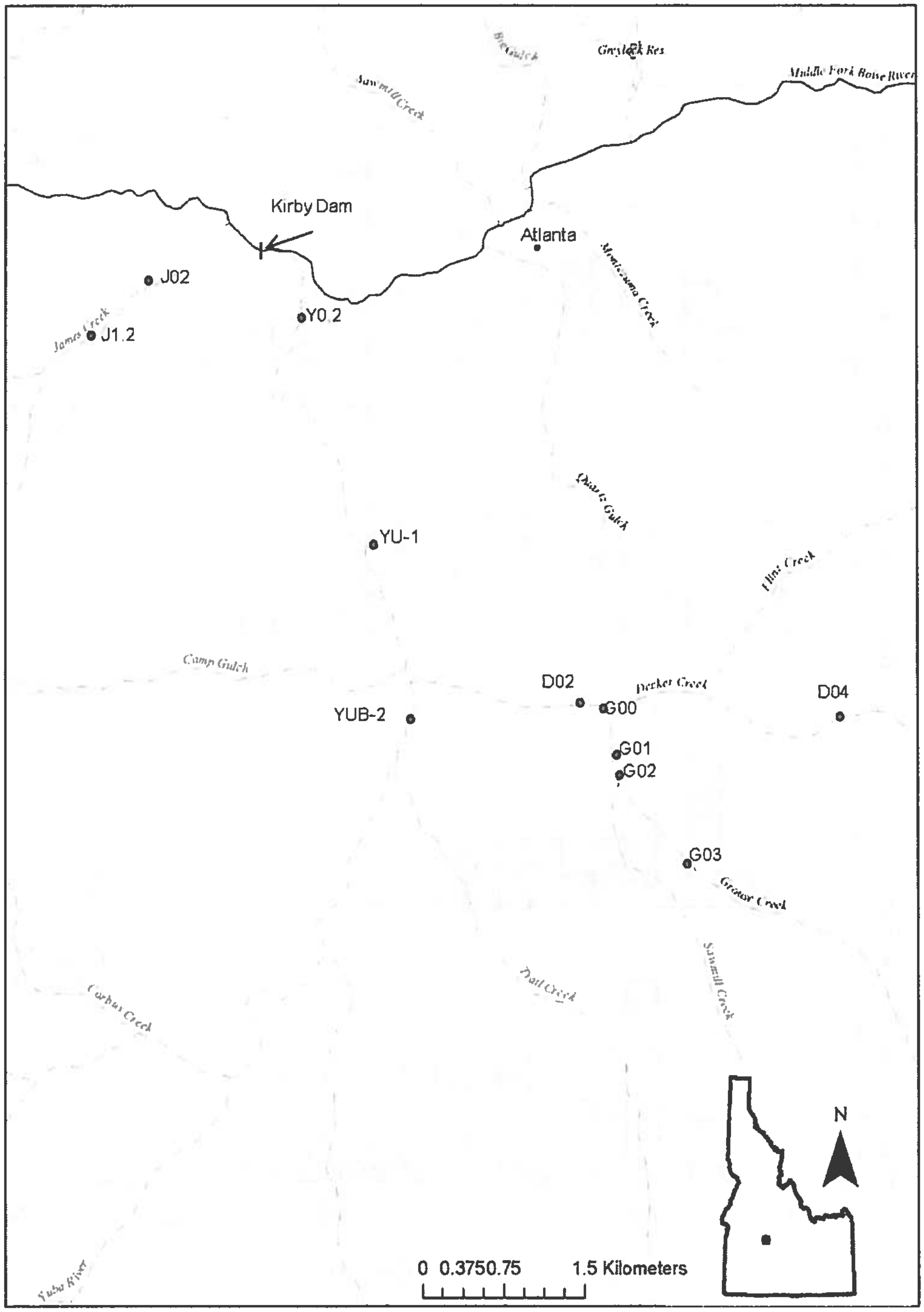

Figure 43. Map of the Yuba River and James Creek drainages, Idaho and the 11 sites sampled to assess fish populations during August 2010.

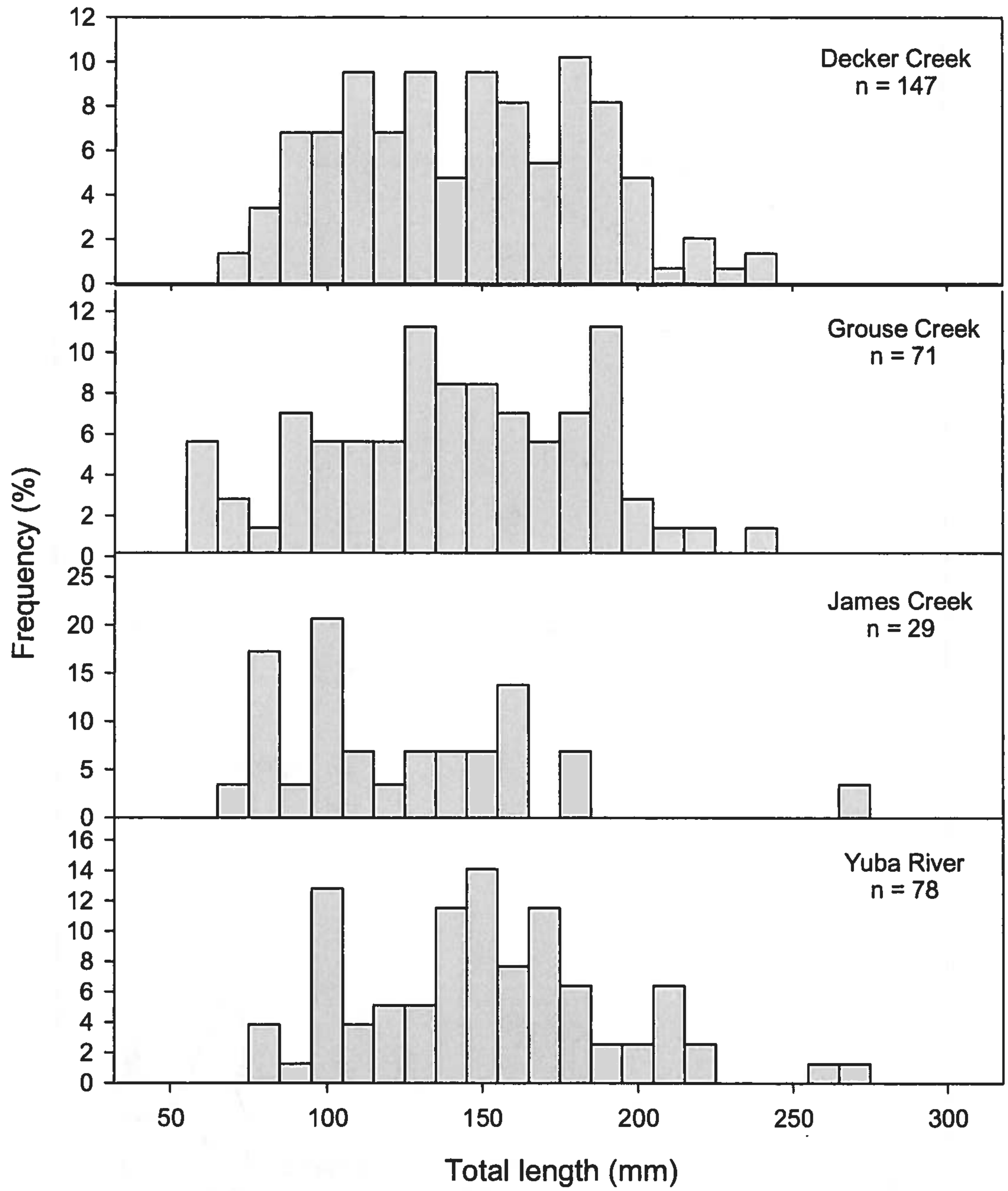

Figure 44. Redband trout length distribution (%) of fish captured in August 2010 at 11 monitoring sites in the Yuba River and James Creek drainages.

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