MITIGATION FOR THE CONSTRUCTION AND OPERATION OF LIBBY DAM

ANNUAL REPORT
2003
(Work Activities July 1, 2003 – June 30, 2004)

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EXECUTIVE SUMMARY

"Mitigation for the Construction and Operation of Libby Dam" is part of the Northwest Power and Conservation Council's (NPCC) resident fish and wildlife program. The program was mandated by the Northwest Planning Act of 1980, and is responsible for mitigating for damages to fish and wildlife caused by hydroelectric development in the Columbia River Basin. The objective of Phase I of the project (1983 through 1987) was to maintain or enhance the Libby Reservoir fishery by quantifying seasonal water levels and developing ecologically sound operational guidelines. The objective of Phase II of the project (1988 through 1996) was to determine the biological effects of reservoir operations combined with biotic changes associated with an aging reservoir. The objectives of Phase III of the project (1996 through present) are to implement habitat enhancement measures to mitigate for dam effects, to provide data for implementation of operational strategies that benefit resident fish, monitor reservoir and river conditions, and monitor mitigation projects for effectiveness. This project completes urgent and high priority mitigation actions as directed by the Kootenai Subbasin Plan.

Montana FWP uses a combination of diverse techniques to collect a variety of physical and biological data within the Kootenai River Basin. These data serve several purposes including: the development and refinement of models used in management of water resources and operation of Libby Dam; investigations into the limiting factors of native fish populations, gathering basic life history information, tracking trends in endangered, threatened species, and the assessment of restoration or management activities intended to restore native fishes and their habitats. The following points summarize the biological monitoring accomplished from July 2003 to June 2004.

- Bull trout redd counts in Grave Creek and the Wigwam River have significantly increased since 1995. However, bull trout redd counts in tributaries downstream of Libby Dam including Quartz, Pipe, Bear, and O'Brien creeks, and the West Fisher River have been variable over the past several years, and have not increased in proportion to bull trout redd counts upstream of Libby Dam.

- Montana FWP conducted an adult bull trout population estimate below Libby Dam in the spring of 2004 using mark recapture techniques and estimated a total of 920 adult bull trout, which represented 263 bull trout per mile.

- Montana FWP monitored the relative abundance of burbot in the stilling basin below Libby Dam using hoop traps since 1994. We captured a total of 3 burbot during the 03-04 trapping seasons which represented the lowest total catch and catch per effort (burbot per trap day) on record since trapping began in the 94-95 trapping season.
- Montana FWP began a detailed field study on Libby Reservoir during mid-November 2003 to investigate the life history of burbot in Libby Reservoir. During the period from November 14 to April 26, 2004 we expended a total effort of 1887 trap-days, and caught a total of 127 burbot at 10 trapping locations throughout the reservoir. Burbot catch at all trapping locations averaged 0.064 fish per trap-day. Mean burbot catch was highest near the mouth of Cripple Horse Creek and lowest near the mouth of Dodge Creek.

- We surgically implanted 28 coded acoustic and 12 combined radio/acoustic tags in burbot at 8 trapping sites. We estimated that the mean home range was 6524 m (range 166 – 27470 m) for the 30 tagged burbot that relocated at least once. The mean estimated depth of burbot that were relocated during daylight searches using acoustic gear was 35.6 m, which was significantly deeper than the mean depth which we operated traps (14.8 m). We were not able to discern any clear movement patterns for the tagged fish in terms of either upstream or downstream movement. Of the 40 tagged burbot. However, we were able to determine that many of the tagged burbot extensively utilized the submerged Kootenai River channel and floodplain during daylight hours.

- We sampled macroinvertebrates at the Libby Creek Upper Cleveland Project and the Grave Creek Phase I Restoration Projects to assess the macroinvertebrate community response to these two stream restoration projects. We used the following metrics for comparison before and after restoration activities: taxa richness, EPT [Ephemeroptera, Plecoptera and Tricoptera] index (e.g., contribution of EPT-taxa), Simpsons Diversity (D), the ratio of Baetidae: Ephemeroptera, the contribution of Diptera, the proportional abundance of burrowers and sprawlers (sediment tolerant), and the contribution of collector-gatherers. At the Libby Creek Upper Cleveland Project site, 3 of the 7 metrics had values in 2003 that were significantly different from the values attained from the pre-restoration values. However, they were all in the opposite direction expected. We specifically expected the relative abundance of sensitive organisms (% EPT) to increase and the dominance of tolerant organisms (% Diptera and % Collector-gatherers) to decrease as conditions become more natural. We observed the opposite and this usually indicates an ecosystem is that has been recently disturbed. None of the Grave Creek metrics tested showed a statistically significant difference between the pre- and post-restoration samples. Results show that sampling should continue for a longer period as the streams recover.

- We conducted juvenile salmonid population estimates within reference reaches on Sinclair, Therriault, Grave, Young, Libby, Parmenter, and Pipe creeks. Trend analyses relevant to stream restoration projects are presented for Grave and Libby creeks.

- Montana FWP has documented the changes in species composition, and species size and abundance within Libby Reservoir since the construction of Libby Dam. We continued monitoring fish populations within the reservoir using spring and fall gill netting and present the results and trend analyses for 11 fish species. The spring gill net catch of bull trout has significantly increased since 1990. We were able to improve the linear regression model for bull trout gillnet catch between years by adjusting the mean bull trout catch per net by reservoir volume at the time the nets were fished each year. Bull trout redd counts in both the Wigwam River and Grave Creek are both significantly and positively correlated to the spring gill net catch rates for bull trout adjusted for reservoir elevation.

- Montana FWP has monitored zooplankton species composition, abundance and size of zooplankton within the reservoir since the construction and filling of Libby Dam. Zooplankton abundance, species composition, and size distribution have also all been similar during the second half of the reservoir's history. *Cyclops* and *Daphnia* have been the first and second most abundant genera of zooplankton present in the reservoir since 1997.

A cooperative mitigation and implementation plan developed by Montana Fish, Wildlife & Parks, the Kootenai Tribe of Idaho and the Confederated Salish and Kootenai Tribes documents the hydropower related losses and mitigation actions attributable to the construction and operation of Libby Dam, as called for by the Northwest Power Planning Council's Fish and Wildlife Program (MFWP, CSKT and KTOI. 1998). A mix of mitigation techniques is necessary to offset losses caused by dam construction and operation. During the past two years, Montana FWP has implemented several project to mitigate for a portion of the losses attributable to the construction and operation of Libby Dam. The following points summarize these projects.

- Montana FWP worked cooperatively with the Lincoln County Fair Board to construct community-fishing pond at the Lincoln County Fairgrounds. This recently completed project will enhance fishing and educational opportunities for young anglers, and help partially mitigate for losses attributable to the construction and operation of Libby Dam.

- After identifying Young Creek as a high priority stream for restoration activities based on habitat quality, fish community composition, and native fish abundance, Montana FWP planned, designed and implemented the Young Creek State Lands Restoration Project effectively changed the stream channel pattern profile and dimension of approximately 1,200 feet of stream channel. These changes resulted in a narrower, deeper channel designed to improve the long-term quantity and quality of rearing habitat for native salmonids. A site-specific monitoring program will assess the success of project accomplishments through time.

- Montana FWP continued monitoring the Libby Creek Cleveland Project and the Grave Creek Phase I stream restoration projects that were completed during the fall of 2002. These projects effectively changed the stream channel pattern profile and dimension. Post-treatment monitoring results presented in this document demonstrated that physical changes were minimal after the first spring freshet.

ACKNOWLEDGEMENTS

We are thankful to the many people that substantially contributed to this project. Neil Benson, Monty Benner, and Cameron Sinclair helped with field data collection, data entry and proofing, and lab processing. We would like to thank EcoAnalysts, Inc. (Moscow, Idaho), and especially Bret Marshall for their assistance with the taxonomy and analysis of the aquatic macroinvertebrates data associated with the Libby Creek and Grave Creek Restoration monitoring. Many of the aquatic habitat restoration projects that were accomplishments and summarized in this report were cooperative efforts between Montana Fish, Wildlife & Parks and other agencies and organizations. Lincoln County assisted with the design and construction of the Eureka Community Fishing pond. This work was funded by the Bonneville Power Administration, and the contract was administered by Ron Morinaka.

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INTRODUCTION

Libby Reservoir was created under an International Columbia River Treaty between the United States and Canada for cooperative water development of the Columbia River Basin (Columbia River Treaty 1964). Libby Reservoir inundated 109 stream miles of the mainstem Kootenai River in the United States and Canada, and 40 miles of tributary streams in the U.S. that provided habitat for spawning, juvenile rearing, and migratory passage (Figure 1). The authorized purpose of the dam is to provide power (91.5%), flood control (8.3%), and navigation and other benefits (0.2%; Storm et al. 1982).

The Pacific Northwest Power Act of 1980 recognized possible conflicts stemming from hydroelectric projects in the northwest and directed Bonneville Power Administration to "protect, mitigate, and enhance fish and wildlife to the extent affected by the development and operation of any hydroelectric project of the Columbia River and its tributaries..." (4(h)(10)(A)). Under the Act, the Northwest Power Planning Council was created and recommendations for a comprehensive fish and wildlife program were solicited from the region's federal, state, and tribal fish and wildlife agencies. Among Montana's recommendations was the proposal that research be initiated to quantify acceptable seasonal minimum pool elevations to maintain or enhance the existing fisheries (Graham et al. 1982).

Research to determine how operations of Libby Dam affect the reservoir and river fishery and to suggest ways to lessen these effects began in May, 1983. The framework for the Libby Reservoir Model (LRMOD) was completed in 1989. Development of Integrated Rule Curves (IRCs) for Libby Dam operation was completed in 1996 (Marotz et al. 1996). The Libby Reservoir Model and the IRCs continue to be refined (Marotz et al 1999). Initiation of mitigation projects such as lake rehabilitation and stream restoration began in 1996. The primary focus of the Libby Mitigation project now is to restore the fisheries and fisheries habitat in basin streams and lakes.

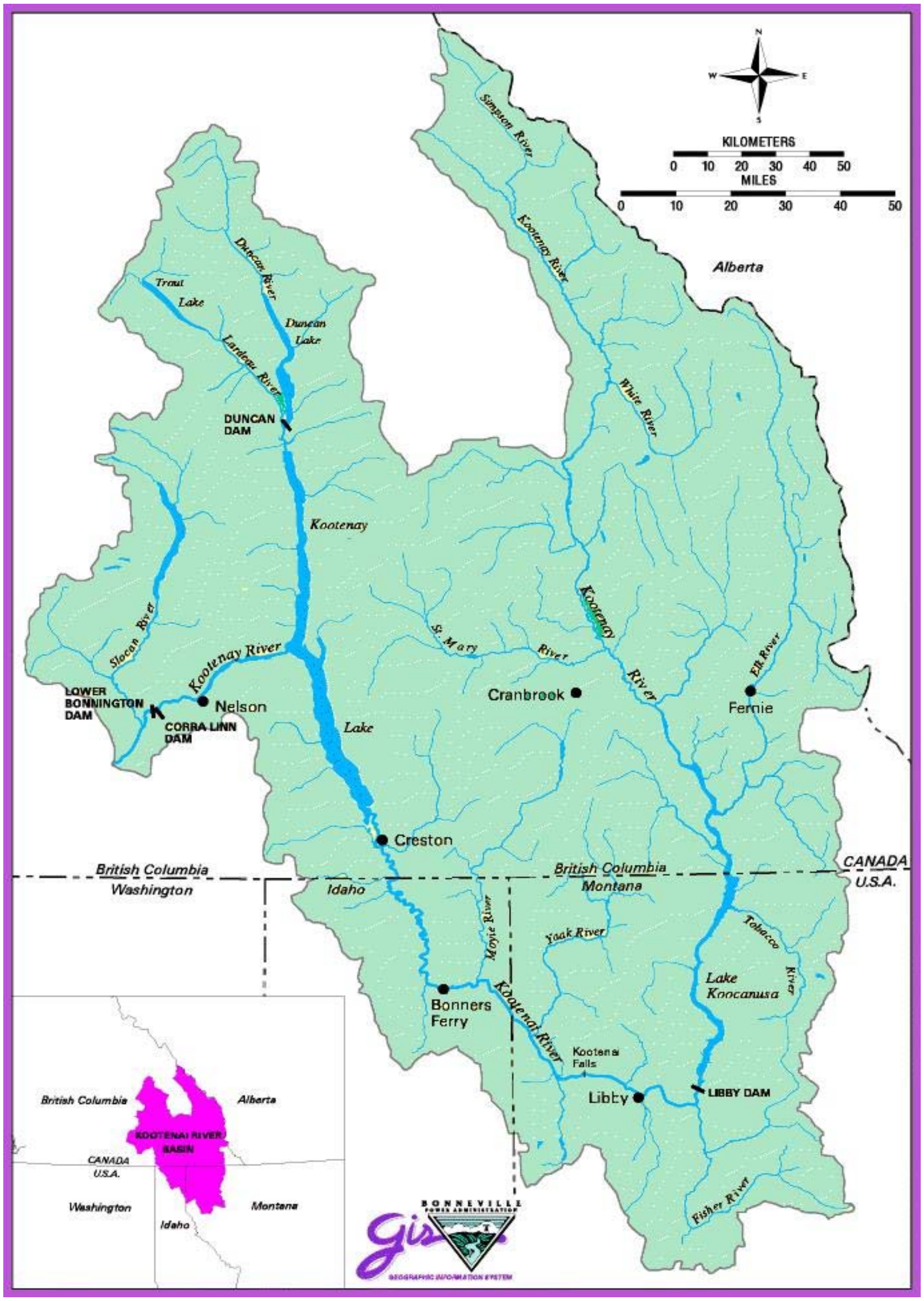

Figure 1. Kootenai River Basin (Montana, Idaho and British Columbia, Canada).

PROJECT HISTORY

Montana Fish, Wildlife and Park's (FWP) began to assess the effects of Libby Dam operation on fish populations and lower trophic levels in 1982. This project established relationship between reservoir operation and biological productivity, and incorporated the results in the quantitative biological model LRMOD. The models and preliminary IRC's (called Biological Rule Curves) were first published in 1989 (Fraley et al. 1989), then refined in 1996 (Marotz et al. 1996). Integrated Rule Curves (IRC's) were adopted by NPPC in 1994, and have recently been implemented, to a large degree, in the federal Biological Opinion (BiOp) for white sturgeon and bull trout (USFWS 2000). This project developed a tiered approach for white sturgeon spawning flows balanced with reservoir IRC's and the NOAA-Fisheries BiOp for salmon and steelhead.The sturgeon tiered flow strategy was adopted by the White Sturgeon Recovery Team in their Kootenai white sturgeon recovery plan (USFWS 1999) and later refined in the USFWS 2000 BiOp.

A long-term database was established for monitoring populations of kokanee, bull trout, westslope cutthroat trout, rainbow trout and burbot and other native fish species. Long-term monitoring of zooplankton and trophic relationships was also established. A model was calibrated to estimate the entrainment of fish and zooplankton through Libby Dam as related to hydro-operations and use of the selective withdrawal, thermal control structure. Research on the entrainment of fish through the Libby Dam penstocks began in 1990, and results were published in 1996 (Skaar et al. 1996). The effects of dam operation on benthic macroinvertebrates in the Kootenai River was also assessed (Hauer et al. 1997) for comparison with conditions measured in the past (Perry and Huston 1983). The project identified important spawning and rearing tributaries in the U.S. portion of the reservoir and began genetic inventories of species of special concern. Research on the effects of operations on the river fishery using Instream Flow Incremental Methodology (IFIM) techniques was initiated in 1992. Assessment of the effects of river fluctuations on Kootenai River burbot fishery was examined in 1994 and 1995. IFIM studies were also completed in Kootenai River below Bonners Ferry, Idaho, to determine spawning area available to sturgeon at various river flows. Microhabitat data collection specific to species and life-stage of rainbow trout and mountain whitefish has been incorporated into suitability curves. River cross-sectional profiles, velocity patterns and other fisheries habitat attributes were completed in 1997. Hydraulic model calibrations and incorporation of suitability curves and modification of the model code were completed in 1999. The IFIM model is currently being updated and frefined by Miller Ecological Consultants, Inc.

Montana FWP has completed several on-the-ground projects since beginning mitigation activities since 1997. Highlights of these accomplishments are listed below for each year.

1997 – Montana FWP chemically rehabilitated Bootjack, Topless and Cibid Lakes (closed-basin lakes) in eastern Lincoln County to remove illegally introduced pumpkinseeds and yellow perch and re-establish rainbow trout and westslope cutthroat trout.

1998 - Montana FWP rehabilitated 200' of Pipe Creek stream bank in cooperation with a private landowner to prevent further loss of habitat for bull trout and westslope cutthroat trout. Pipe Creek is a primary spawning tributary to the Kootenai River.

1998 through 2000 - Montana FWP developed an isolation facility for the conservation of native redband trout at the Libby Field Station. Existing ponds were restored and the inlet stream was enhanced for natural outdoor rearing. Natural reproduction may be possible. Activities included chemically rehabilitating the system and constructing a fish migration barrier to prevent fish movement into the reclaimed habitat.

1998 - Montana FWP chemically rehabilitated Carpenter Lake to remove illegally introduced pike, largemouth bass and bluegills and reestablish westslope cutthroat trout and rainbow trout. Natural reproduction is not expected in this closed basin lake.

1999 - Montana FWP rehabilitated ~400' of Sinclair Creek to reduce erosion, stabilize highway crossing, and install fisheries habitat for westslope cutthroat trout. Sinclair Creek is a tributary to Libby Reservoir.

2000 - Montana FWP completed additional work on Sinclair Creek to stabilize a bank slough for westslope cutthroat habitat improvement. Sinclair Creek is now accessible to adfluvial spawners from Libby Reservoir.

2000 - Montana FWP was a major contributor (financial and in-kind services; primarily surveying) towards completion of Parmenter Creek re-channelization/rehabilitation work (Project Impact). Parmenter Creek has the potential to provide additional spawning and rearing habitat for Kootenai River fish, most likely westslope cutthroat trout.

2000 - Montana FWP completed stream stabilization and re-channelization project at the mouth of O'Brien Creek to mitigate for delta formation and resulting stream instability, and to ensure bull trout passage in the future. The work was completed in cooperation with private landowners and Plum Creek Timber Company.

2000 - Montana FWP completed stream stabilization and a water diversion project in cooperation with the city of Troy on O'Brien Creek to ensure bull trout passage in the future. The project removed a head cut and stabilized a section of stream. O'Brien Creek is a core bull trout recovery stream, and this project helped ensure access to spawning areas.

2001 – Montana FWP designed and reconstructed approximately 1,200 feet of stream channel on Libby Creek to stabilize stream banks, reduce sediment, and improve rearing habitat for salmonids. This project eliminated a mass wasting hill slope that was contributing an estimated 4,560 cubic yards of sediment per year.

2001 – Montana FWP collaborated with the Kootenai River Network to reconstruct approximately 1,200 feet of stream channel on Grave Creek in order to stabilize stream banks, reduce sediment, and improve rearing habitat for salmonids.

2001 – Montana FWP chemically rehabilitated Banana Lake in order to remove exotic fish species from this closed basin lake. Banana Lake will be restocked with native fish species for recreational fishing opportunities.

2001 – Montana FWP worked cooperatively with the city of Troy, MT to construct a community fishing pond in Troy. The pond was completed in 2002 and stocked with fish from Murray Spring Fish Hatchery.

2002 – Montana FWP collaborated with the Kootenai River Network and 7 other contributors to reconstruct approximately 4,300 feet of stream channel on Grave Creek in order to stabilize stream banks, reduce sediment, improve rearing habitat for salmonids, and restore riparian vegetation. A long-term monitoring plan was also implemented in conjunction with this project to evaluate whether project objectives are maintained through time.

2002 – Montana FWP collaborated with the landowner on upper Libby Creek to reconstruct approximately 4,300 feet of stream channel that was previously impacted by mining activities. The project objectives were to stabilize stream banks, reduce sediment, improve rearing habitat for salmonids, and restore riparian vegetation. Similar to the Grave Creek restoration activities, we also implemented a long-term monitoring plan with this project to evaluate whether project objectives are maintained through time. This restoration project was intended to benefit native redband rainbow trout and bull trout.

2003 – Libby Fisheries Mitigation coordinated with the Wildlife Mitigation Trust to complete a conservation easement in the Fisher River corridor. Fisheries mitigation dollars were used to secure riparian habitat along 8.3 km of the Fisher River and important tributaries.

ASSOCIATIONS

The primary goals of the Libby Mitigation project are to implement operational mitigation (Integrated Rule Curve refinement and assessment: measure 10.3B of the Northwest Power Planning Council's Fish and Wildlife Program) and non-operational mitigation (habitat and passage improvements) in the Kootenai drainage. Results complement and extend the Kootenai Focus Watershed Program (Project 199608720) and the draft Kootenai Subbasin Plan (KTOI and MFWP 2004, see NPCC web page). This project creates new trout habitat by restoring degraded habitat to functional condition through stream rehabilitation and fish passage repairs. The projects compliment each other in the restoration and maintenance of native trout populations in the Kootenai River System.

This project has direct effects on the activities of Idaho Department of Fish and Game (IDFG)-Kootenai River Fisheries Investigations (198806500 – IDFG) and White Sturgeon Experimental Aquaculture (198806400 – Kootenai Tribe of Idaho). The project manager, Brian Marotz, is on the Kootenai white sturgeon recovery team and works closely with project sponsors from IDFG and KTOI. Results and implementation of recommendations derived from the IRCs, sturgeon tiered flow strategy and IFIM models affect white sturgeon recovery activities.

The radio-telemetry work of this project will identify migration habits, habitat preferences and spatial distribution of species in the Kootenai system. Much of this information is shared with the IFIM project in the Flathead Watershed (Project 199101903).

Project personnel are completing activities in the lower Kootenai River in Montana that will gather data to serve as baseline, control information for Kootenai River Ecosystem Improvement Study (19940490 – Kootenai Tribe of Idaho). The intent of their study is to determine if fertilization of the Kootenai River is a viable alternative for increasing primary productivity in the Idaho portion of the river.

We have been cooperating with the efforts of the bull trout recovery project in Canada (2000004 – British Columbia Ministry of Environment) for several years to monitor the status of bull trout in the upper Kootenai River, it's tributaries, and Libby Reservoir. Our cooperative activities have included radio-tagging and tracking of adult bull trout, redd counts, sediment and temperature monitoring, and migrant fish trip operations.

Montana FWP is an active partner with the Kootenai River Network (KRN). KRN is a non-profit organization created to foster communication and implement collaborative processes among private and public interests in the watershed. These cooperative programs improve resource management practices and the restoration of water quality and aquatic resources in the Kootenai basin. KRN is an alliance of diverse citizen's groups, individuals, business and industry, and tribal and government water resource management agencies in Montana, Idaho, and British Columbia. KRN enables all interested parties to collaborate in natural resource management in the basin. Montana FWP serves on the KRN Executive Board. Formal participation in the KRN helps Montana FWP achieve our goals and objectives toward watershed restoration activities in the Kootenai Basin.

DESCRIPTION OF STUDY AREA

Subbasin Description

The Kootenai River Subbasin is an international watershed that encompasses parts of British Columbia (B.C.), Montana, and Idaho (Figure 1). The headwaters of the Kootenai River originate in Kootenay National Park, B.C. The river flows south within the Rocky Mountain Trench into the reservoir created by Libby Dam, which is located near Libby, Montana. From the reservoir, the river turns west, passes through a gap between the Purcell and Cabinet Mountains, enters Idaho, and then loops north where it flows into Kootenay Lake, B.C. The waters leave the lake's West Arm and flow south to join the Columbia River at Castlegar, B.C. The annual runoff volume makes the Kootenai the second largest Columbia River tributary. The Kootenai ranks third in watershed area (36,000 km^2 or 8.96 million acres)(Knudson 1994). The climate, topography, geology, soils and land use characteristics of the Kootenai Basin were previously described in Dunnigan et al. (2003).

Drainage Area

Nearly two-thirds of the river's 485-mile-long channel, and almost three-fourths of its watershed area, is located within the province of British Columbia. Roughly twenty-one percent of the watershed lies within the state of Montana (Figure 2), and six percent falls within Idaho (Knudson 1994). The Continental Divide forms much of the eastern boundary, the Selkirk Mountains the western boundary, and the Cabinet Range the southern. The Purcell Mountains fill the center of the river's J-shaped course to Kootenay Lake. Throughout, the subbasin is mountainous and heavily forested.

Hydrology

The headwaters of the Kootenay River in British Columbia consist primarily of the main fork of the Kootenay River and Elk River. High channel gradients are present throughout headwater reaches and tributaries.

Libby Reservoir (Lake Koocanusa) and its tributaries receive runoff from 47 percent of the Kootenai River drainage basin. The reservoir has an annual average inflow of 10,615 cfs. Three Canadian rivers, the Kootenay, Elk, and Bull, supply 87 percent of the inflow (Chisholm et al. 1989). The Tobacco River and numerous small tributaries flow into the reservoir south of the International Border.

Major tributaries to the Kootenai River below Libby Dam include the Fisher River (838 sq. mi.; 485 average cfs), the Yaak River (766 sq. mi. and 888 average cfs) and the Moyie River (755 sq. mi.; 698 average cfs). Kootenai River tributaries are characteristically high-gradient mountain streams with bed material consisting of various mixtures of sand, gravel, rubble, boulders, and drifting amounts of clay and silt, predominantly of glacio-lacustrine origin. Fine materials, due to their instability during periods of high stream discharge, are continually abraded and redeposited as gravel bars, forming braided channels with alternating riffles and pools. Stream flow in unregulated tributaries generally peaks in May and June after the onset of snow melt, then declines to low flows from November through March. Flows also peak with rain-on-snow events. Kootenai Falls, a 200-foot-high waterfall and a natural fish-migration barrier, is located eleven miles downstream of Libby, Montana.

The river drops in elevation from 3618 m at the headwaters to 532 m at the confluence of Kootenay Lake. It leaves the Kootenay Lake through the western arm to a confluence with the Columbia River at Castlegar. A natural barrier at Bonnington Falls, and now a series of four dams isolate fish from other populations in the Columbia River basin. The natural barrier has isolated sturgeon for approximately 10,000 years (Northcote 1973). At its mouth, the Kootenai River has an average annual discharge of 868 m^3/s (30,650 cfs).

Fish Species

Eighteen species of fish are present in Libby Reservoir and the Kootenai River (Table 1). The reservoir currently supports an important fishery for kokanee *Oncorhynchus nerka* and rainbow trout *Oncorhynchus mykiss*, with annual fishing pressure over 500,000 hours (Chisholm and Hamlin 1987). Burbot *Lota lota* are also important game fish, providing a popular fishery during winter and spring. The Kootenai River below Libby Dam is a "blue ribbon" trout fishery, and the state record rainbow trout was harvested there in 1997 (over 38 pounds). Although bull trout *Salvelinus confluentus* fishing was banned in the Kootenai River, are "incidental captures" provide a unique seasonal fishery.

Table 1. Current relative abundance (A=abundant, C=common, R=rare) and abundance trend from 1975 to 2000 (I=increasing, S = stable , D = decreasing, U = unknown) of fish species present in Libby Reservoir.

Common Name	Scientific name	Relative abundance	Abundance trend	Native
Game fish species				
Westslope cutthroat trout	*Oncorhynchus clarki lewisi*	C	D	Y
Rainbow trout	*Oncorhynchus mykiss*	C	D	Y
Bull trout	*Salvelinus confluentus*	C	I	Y
Brook trout	*Salvelinus fontinalis*	R	U	N
Lake trout	*Salvelinus namaycush*	R	U	N
Kokanee salmon	*Oncorhynchus nerka*	A	U	N
Mountain whitefish	*Prosopium williamsoni*	R	D	Y
Burbot	*Lota lota*	C	D	Y
Largemouth bass	*Micropterus salmoides*	R	U	N
Northern pike	*Esox lucius*	R	U	N
Nongame fish species				
Pumpkinseed	*Lepomis gibbosus*	R	U	N
Yellow perch	*Perca flavescens*	C	I	N
Redside shiner	*Richardsonius balteatus*	R	D	Y
Peamouth	*Mylocheilus caurinus*	A	I	Y
Northern pikeminnow	*Ptychocheilus oregonensis*	A	I	Y
Largescale sucker	*Catostomus macrocheilus*	A	S	Y
Longnose sucker	*Catostomus catostomus*	C	D	Y

Reservoir Operation

Libby Dam is a 113-m (370-ft) high concrete gravity structure with three types of outlets: sluiceways (3), operational penstock intakes (5, 8 possible), and a gated spillway. The dam crest is 931 m long (3,055 ft), and the widths at the crest and base are 16 m (54 ft) and 94 m (310 ft), respectively. A selective withdrawal system was installed on Libby Dam in 1972 to control water temperatures in the dam discharge by selecting of water various strata in the reservoir forebay.

Completion of Libby Dam in 1972 created the 109-mile Libby Reservoir. Specific morphometric data for Libby Reservoir are presented in Table 2. Filling Libby Reservoir inundated and eliminated 109 miles of the mainstem Kootenai River and 40 miles of critical, low-gradient tributary habitat. This conversion of a large segment of the Kootenai River from a lotic to lentic environment changed the aquatic community (Paragamian 1994). Replacement of the inundated habitat and the community of life it supported are not possible. However, mitigation efforts are underway to protect, reopen, or reconstruct the remaining tributary habitat to partially offset the loss. Fortunately, in the highlands of the Kootenai Basin, tributary habitat quality is high. The headwaters are relatively undeveloped and retain a high percentage of their original wild attributes and native species complexes. Protection of these remaining pristine areas and reconnection of fragmented habitats are high priorities.

Between 1977 and 2000, reservoir drawdowns averaged 111 feet, but were as extreme as 154 feet (Figure 3). Drawdown affects all biological trophic levels and influences the probability of subsequent refill during spring runoff. Refill failures are especially harmful to biological production during warm months. Annual drawdowns impede revegetation of the reservoir varial zone and result in a littoral zone of nondescript cobble/mud/sand bottom with limited habitat structure.

Table 2. Morphometric data for Libby Reservoir.

Surface elevation	
maximum pool	749.5 m (2,459 ft)
minimum operational pool	697.1 m (2,287 ft)
minimum pool (dead storage)	671.2 m (2,222 ft)
Area	
maximum pool	188 sq. km (46,500 acres)
minimum operational pool	58.6 sq. km (14,487 acres)
Volume	
maximum pool	7.24 km^3 (5,869,400 acre-ft)
minimum operational pool	1.10 km^3 (890,000 acre-ft)
Maximum length	145 km (90 mi)
Maximum depth	107 m (350 ft)
Mean depth	38 m (126 ft)
Shoreline length	360 km (224 mi)
Shoreline development	7.4 km (4.6 mi)
Storage ratio	0.68 yr
Drainage area	23,271 sq. km (8,985 sq. mi)
Drainage area:surface area	124:1
Average daily discharge	
pre-dam (1911-1972)	11,774 cfs
post-dam (1974-2000)	10,991 cfs

Similar impacts have been observed in the tailwater below Libby Dam. The zone of water fluctuation or *varial zone* has been enlarged by daily changes in water-flow and stage caused by power operations. The resulting rapid fluctuations in dam discharges (as great as 400 percent) are inconsistent with the normative river concept (ISAB 1997). The varial zone is neither a terrestrial nor aquatic environment, so is biologically unproductive. Daily and weekly differences in discharge from Libby Dam have an enormous impact on the stability of the riverbanks. Water logged banks are heavy and unstable; when the flow drops in magnitude, banks calve off, causing serious erosion in the riparian zone. These impacts are common during winter but go unnoticed until spring. In addition, widely fluctuating flows can give false migration cues to burbot and white sturgeon spawners (Paragamian 2000 and Paragamian and Kruse 2001).

Also, barriers have been deposited in critical spawning tributaries to the Kootenai River through the annual deposition of bedload materials (sand, gravel, and boulders) at their confluence with the river (Marotz et al. 1988). During periods of low stream flow, the enlarged deltas and excessive deposition of bedload substrate in the low gradient reaches of tributaries impedes or blocks fall-spawning migrations. During late spring and summer, when redband and cutthroat trout are out-migrating from nursery streams, the streams may flow subsurface through the porous deltas (Paragamian V., IDFG, personal communication 2000). As a result, many potential recruits are stranded. Prior to impoundment, the Kootenai River contained sufficient hydraulic energy to annually remove these deltas, but since the dam was installed, peak flows have been limited to maximum turbine capacity (roughly 27 kcfs). Hydraulic energy is now insufficient to remove deltaic deposits. Changing and regulating the Kootenai River annual hydrograph for power and flood control and altering the annual temperature regime have caused impacts typical of dam tailwaters.

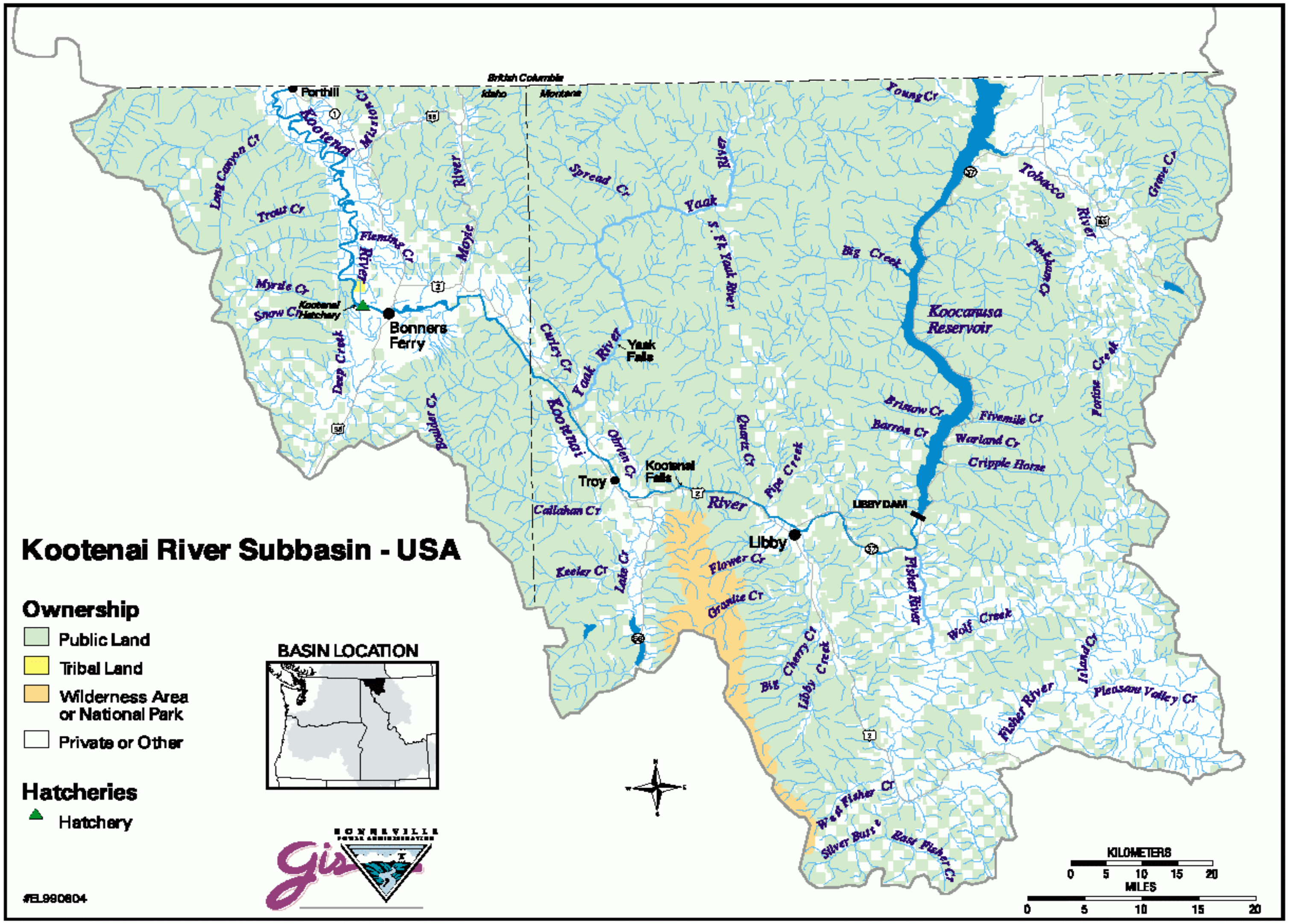

Figure 2. Kootenai River Basin, Montana.

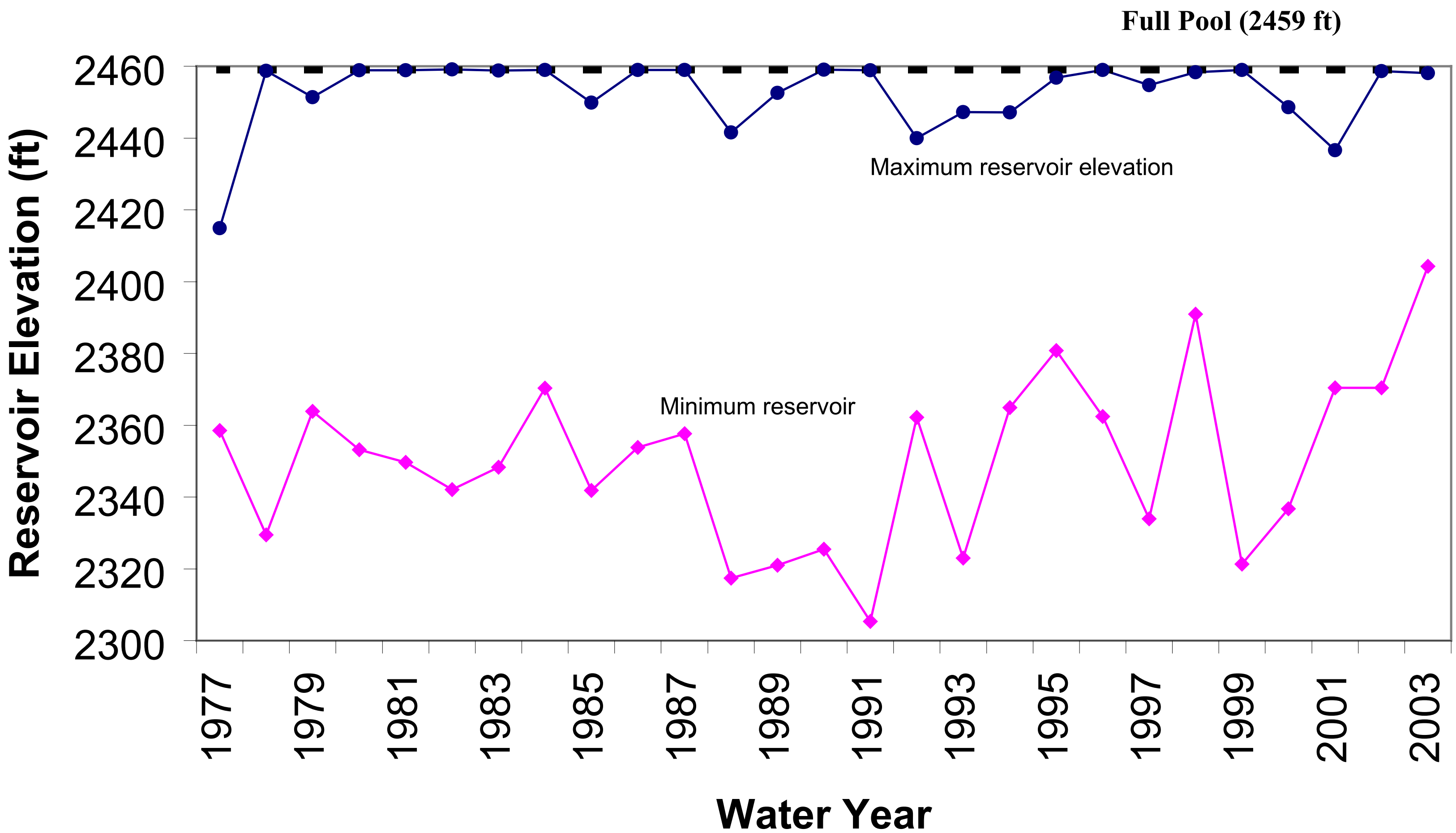

Figure 3. Libby Reservoir elevations (minimum, maximum), water years (October 1 – Sept. 30), 1976 through 2003.

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Chapter 1

Physical and Biological Monitoring in the Montana Portion of the Kootenai River Basin

Abstract

Montana Fish, Wildlife & Parks (FWP) uses a combination of diverse techniques to collect a variety of physical and biological data within the Kootenai River Subbasin. These data serve several purposes including: the development and refinement of models used in management of water resources and operation of Libby Dam; investigations into the limiting factors of native fish populations, gathering basic life history information, tracking trends in endangered, threatened species, and the assessment of restoration or management activities intended to restore native fishes and their habitats. Bull trout core areas for the Koocanusa population include Grave and Skookumchuck creeks and the Wigwam and White rivers, with the majority of the spawning located in the tributaries located in British Columbia. Bull trout redd counts in Grave Creek and the Wigwam River have significantly increased since 1995. Bull trout core areas in the Kootenai River downstream of Libby Dam include Quartz, Pipe, Bear (Libby Creek drainage), O'Brien creeks and the West Fisher River. Bull trout redd counts within these individual core streams have been variable over the past several years, and have not increased in proportion to bull trout redd counts upstream of Libby Dam. Montana FWP conducted an adult bull trout population estimate below Libby Dam in the spring of 2004 using mark recapture techniques and estimated a total of 920 adult bull trout, which represented 263 bull trout per mile. We have monitored the relative abundance of burbot in the stilling basin below Libby Dam using hoop traps since 1994. We captured a total of 3 burbot during the 03-04 trapping seasons which represented the lowest total catch and catch per effort (burbot per trap day) since trapping began in the 1994-95 trapping season. During mid-November 2003, Montana FWP began a detailed field study to investigate the life history of burbot in Libby Reservoir (named Lake Koocanusa). During the period from November 14 to April 26, 2004 we completed 1887 trap-days, and caught a total of 127 burbot at 10 locations throughout the reservoir. Burbot catch at all trapping locations averaged 0.064 fish per trap-day. Mean burbot catch was highest near the mouth of Cripple Horse Creek and lowest near the mouth of Dodge Creek. We surgically implanted 28 coded acoustic and 12 combined radio/acoustic tags in burbot at 8 trapping sites. Each tag had a unique frequency or code to allow for individual fish identification. We estimated that the mean home range was 6524 m (range 166 – 27470 m) for the 30 tagged burbot that were relocated after release. The mean estimated depth of burbot that were relocated during daylight searches using acoustic gear was 35.6 m, which was significantly deeper than the mean depth at which we operated traps (14.8 m). We did not detect clear upstream or downstream movement patterns in the tagged fish relocations. Of the 40 tagged burbot, however, many were observed extensively utilizing the submerged Kootenai River channel and floodplain during daylight hours. We sampled macro-invertebrates at the Libby Creek Upper Cleveland Project and the Grave Creek Phase I Restoration Projects to assess the macroinvertebrate community response to these two restoration projects. We used the following metrics for comparison before and after restoration activities: taxa richness, EPT

index (e.g., contribution of EPT-taxa), Simpsons Diversity (D), the ratio of Baetidae: Ephemeroptera, the contribution of Diptera, the proportional abundance of burrowers and sprawlers (sediment tolerant), and the contribution of collector-gatherers. At the Libby Creek Upper Cleveland Project site, 3 of the 7 metrics had values in 2003 that were significantly different from the values attained from the pre-restoration values. However, they were all in the opposite direction expected. We specifically expected the relative abundance of sensitive organisms (% EPT) to increase and the dominance of tolerant organisms (% Diptera and % Collector-gatherers) to decrease as conditions become more natural. We observed the opposite and this usually indicates an ecosystem is that has been recently disturbed. None of the Grave Creek metrics tested showed a statistically significant difference between the pre- and post-restoration samples. Results indicate that a longer-term monitoring program will be necessary to document changes as the streams recover. We conducted juvenile salmonid population estimates within reference reaches on Sinclair, Therriault, Grave, Young, Libby, Parmenter, and Pipe creeks. Trend analyses relevant to stream restoration projects are presented for Grave, Libby, and Parmenter creeks. Montana FWP has documented the changes in species composition, and species size and abundance within Libby Reservoir since the construction of Libby Dam. We continued monitoring fish populations within the reservoir using spring and fall gill netting and present the results and trend analyses for 11 fish species. Our spring gill net catch of bull trout has especially increased since 1990. We were able to improve the linear regression model for bull trout gillnet catch between years by adjusting the mean bull trout catch per net by reservoir volume at the time the nets were fished each year. This adjustment substantially improved the regression model's fit to the data. Bull trout redd counts in both the Wigwam River and Grave Creek are both significantly and positively correlated to the spring gill net catch rates for bull trout adjusted for reservoir elevation. Montana FWP has also monitored zooplankton species composition, abundance and size of zooplankton within the reservoir since the construction and filling of Libby Dam. Zooplankton abundance, species composition, and size distribution have also all been similar during the second half of the reservoir's history. Since 1997, *Cyclops* and *Daphnia* have been the first and second most abundant genera of zooplankton present in the reservoir.

Introduction

The primary objectives of the Libby Mitigation Project are to 1) Correct deleterious effects caused by hydropower operations and mitigate for fisheries losses attributed to the construction and operation of Libby Dam using watershed-based, habitat enhancement, fish passage improvements, and offsite fish recovery actions, 2) Integrate computer models into a watershed framework using Montana FWP's quantitative reservoir model (LRMOD), Integrated Rule Curves (IRC), Instream Flow Incremental Methodology (IFIM) and Libby Dam fish entrainment model (ENTRAIN), to improve biological production by modifying dam operation, and 3) Recover native fish species including the endangered Kootenai River white sturgeon, threatened bull trout, westslope cutthroat trout, interior redband rainbow trout, and burbot. A loss statement, site-specific mitigation actions and monitoring strategies were documented in the Libby Mitigation and Implementation Plan (MFWP, CSKT and KTOI. 1988).

Biological monitoring data was proven to be critical for empirically calibrating computer models used in management of water resources and operation of Libby Dam. The quantitative biological model LRMOD was calibrated using field data collected by project personnel from 1983 through 1990. Field data from 1991 through 1995 were used to refine and correct uncertainties in the model and add a white sturgeon component (Marotz et al. 1996 and 1999). These models include Integrated rule curves (IRC's), the Libby Reservoir model (LRMOD) and an alternate flood control strategy called VARQ, which stands for variable flow (Q). The ultimate result has been the integration of fisheries operations with power production and flood control to reduce the economic impact of basin-wide fisheries recovery actions.

Investigations into the factors limiting native fish populations require a combination of diverse field evaluation techniques. Characteristics evaluated include population densities, species assemblages and composition, fish length-at-age (otolith and scale aging), growth, condition factors, indices of abundance and biomass estimates. In this chapter we describe the results of the field activities required to gather this information.

In addition, habitat enhancement and fish passage improvement measures may be the most promising methods for recovering native resident stocks. This project has embraced this approach and implemented several restoration projects on a basin wide priority basis using a step-wise, adaptive management approach to correct limiting factors for bull trout, burbot, white sturgeon, and redband trout in the Kootenai Basin (see chapter 2). Biological and physical monitoring is critical to assess the effectiveness of restoration or management actions designed to restore native fishes and their habitats. Evaluation of restoration activities and pilot projects will continue to determine the most cost-effective methods for enhancing these diverse populations. This chapter describes the physical and biological monitoring activities necessary to achieve the activities described above.

Methods

Bull Trout Redd Counts

Redd surveys were conducted in October after bull trout spawned in the Wigwam and West Fisher rivers, Grave, Quartz, Bear (a tributary to Libby Creek), Keeler, Pipe, and O'Brien creeks. MFWP and U.S. Forest Service (USFS) personnel walked streams in the United States and personnel from the British Columbia Ministry of Water, Land, and Air Protection walked the Wigwam River and associated tributaries. Observers enumerated "positive" and "possible" redds. "Possible" redds were those that did not have fully developed pits and gravel berms. Since 1993, only "positive" redds have been counted, and are included in tables and figures for this report. In addition to counting redds, size and location of redds were also noted. Surveyors recorded suitable habitat and barriers to spawning bull trout when a stream was surveyed for the first time. We used linear regression of redd counts to assess population trends.

Kootenai River Adult Bull Trout Population Estimate

We collected adult bull trout using nighttime electrofishing by jet boat to perform a mark-recapture population estimate of bull trout in the Kootenai River from Libby Dam (River mile [RM] 221.7) downstream to the confluence of the Fisher River (RM 218.2). We operated two jet boat electrofishing crews during each sampling event. Each boat contained a driver and two netters. Our electrofishing unit on each boat consisted of a Coffelt model Mark 22 electrofishing unit operating with an electrical output ranging from 200-350 volts at 5-8 amps powered by a 5,000 watt gasoline powered generator. In order to thoroughly electrofish the entire 3.5 miles of Kootenai River, we divided the sample area into 2 sections, and conducted electrofishing on each section on a single evening. Section 1 was from Libby Dam downstream to the Alexander Creek confluence (RM 220.5), and was 1.2 miles long. Section 2 was from the Alexander Creek confluence downstream to the Fisher River Confluence, and was 2.3 miles long. We marked bull trout on the evenings of April 8 and 15, and performed recapture efforts on April 21 and 22 in sections 1 and 2, respectively.

We recorded the total time (minutes) electrical current was generated in the water as a measure of effort. We measured total length (mm), weighed (g), examined all fish for marks, collected scale samples, and released all bull trout captured near their capture location. All bull trout were marked with individually numbered 134 (ISO) KHz passive integrated transponder (PIT) tags and an adipose fin clip in order to evaluate PIT tag retention. PIT tags were inserted with an 8-gauge hypodermic needle into the musculature behind the dorsal fin.

We estimated bull trout abundance using a mark-recapture population estimation technique which assumes the population of bull trout is “closed”, suggesting no births, deaths or migrations occurred during sampling periods (Ricker 1958). Additional assumptions were that marked and unmarked fish have equal mortality rates, marked fish were randomly distributed throughout the study area, marks were not lost, and all marked fish captured were recognized and counted (Lagler 1956). We used a computer software program called Mark/Recapture (version 7.0) that uses a log-likelihood estimator to estimate the absolute abundance of adult bull trout within the study reach.

Burbot Monitoring Below Libby Dam

Montana FWP has monitored burbot densities directly below Libby Dam since 1994, using baited hoop traps during December and February to capture burbot in or near spawning condition. The trapping effort in 2003 was expanded to include the month of January because a modified operational plan (VARQ) was implemented beginning in January 2003. Two hoop traps measuring 2-feet diameter, approximately 6-8 feet in length with ¾ inch net mesh were baited with cut bait (usually kokanee, depending upon availability) and lowered in the stilling basin below Libby Dam at depths ranging from 20-55 feet (Figure 1). Sash weights attached to the cod end of each hoop trap securely positioned the trap on the bottom. Traps were generally checked twice per week unless catches substantially increased between periods. Captured burbot were enumerated, examined for a PIT (passive integrated transponder) tag, measured, PIT tagged with a 125 KHz PIT tag if not previously tagged, and released. Fish less than approximately 350 mm total length were not tagged. PIT tags were inserted with an 8-gauge hypodermic needle into the musculature of the left operculum. We standardized the catch in terms of the average catch per trap day, in order to compare burbot catch rates across years.

Figure 1. An aerial photograph of Libby Dam, looking downstream. The red symbols represent typical locations that hoop traps are positioned below Libby Dam for burbot monitoring.

Burbot Monitoring Libby Reservoir

Montana FWP began a detailed field study on Libby Reservoir during mid-November to investigate the life history of burbot in the reservoir. Little information exists regarding the life history, spawning distribution, abundance or status of burbot populations upstream of Libby Dam. Little information also exists regarding the degree to which burbot above and below Libby Dam are connected through entrainment at Libby Dam. The field investigations used a combination of trapping, using baited hoop traps and sonic and radio telemetry techniques. We captured burbot using baited hoop traps measuring 2-feet diameter, approximately 6-8 feet in length with ¾ inch net mesh baited with cut bait (usually kokanee, depending upon availability). Sash weights attached to the cod end of each hoop trap securely positioned the trap on the bottom. Traps were fished at depths ranging from 2 – 33 m (mean = 18 m) from November 2003 to April 26, 2004, and were checked an average of every 4.8 days. Traps were fished within the general vicinity of 10 tributary confluences throughout the reservoir ranging from river mile (RM) 222.5 – 268.4 (Table 1). Throughout the trapping season, we fished up to 4 locations at any given time using 3 traps per location. Traps were fished in the vicinity of Canyon, Cripple Horse, Barron, Bristow, Ten Mile, Big, Sutton, Dodge, and Young creeks and the Tobacco River. Captured burbot were enumerated, examined for a PIT (passive integrated transponder) tag, measured, PIT tagged with a 134 (ISO) KHz PIT tag if not previously tagged, and released. PIT tags were inserted with an 8-gauge hypodermic needle into the musculature of the left operculum. We standardized the catch for each trapping location to average catch per trap day, in order to compare burbot catch rates across locations and time. We used multiple linear regression to evaluate trends in burbot catch versus trapping date and hoop trap depth.

In addition to PIT tagging, 28 and 12 burbot were tagged with coded acoustic and combined acoustic and radio telemetry tags, respectively. The coded acoustic tags weighed 30 g and were manufactured by Lotek Inc. (model CAFT16-1). Tags were powered by a single 3.6 V lithium battery and had a minimum life span of 717 days. The tags transmitted at a frequency of 76.8 KHz, and had a burst rate of 5.0 seconds. Each tag had a unique code that allowed for individual identification. The combined coded acoustic and radio tags weighed 25.3 g and were also manufactured by Lotek Inc. (model CART16-2S) and were powered by a 3.6 V lithium battery. Each transmitter had a 29 cm flexible external whip antenna attached to one end. The coded acoustic portion of this tag also transmitted at a frequency of 76.8 KHz, and the coded radio frequency portion of this tag transmitted at a frequency of 150.077 MHz. These tags operated by alternating between radio and acoustic signals. Each particular tag had the same code for both the radio and acoustic components that allowed for individual fish identification.

Burbot that were selected for acoustic or telemetry tagging were held in a hoop trap at 3-5 m depths for a period of 24-48 hours to allow the fish to equilibrate for pressure differences. Burbot were placed into an anesthesia tank containing a solution of tricaine methanesulfonate (MS-222) until the fish lost equilibrium, and then placed in a surgical trough on their back and their gills were irrigated with freshwater for the duration of the surgical procedure. We used a scalpel to make an incision approximately one third of the distance between the vent and pectoral fins approximately 10-20 mm off the ventral mid-line in the abdominal wall just large enough to fit the tag into. For the combined acoustic and radio tags we used a 13 cm long ten gauge hypodermic need to make a small hole for the antenna located approximately 5 cm toward the posterior of the incision. The antenna was threaded through the needle, the tag inserted into the abdominal cavity and the needle removed, leaving the antenna trailing along the burbot's body. The coded acoustic tags did not require a hole for the antenna because they did not have antennae. The incision for both types of tags were closed with 4-0 silk using 3-5 stitches. An additional stitch was usually placed near the antenna puncture wound for those fish tagged with the combined acoustic and radio tags. The entire surgical procedure was usually completed within 3-7 minutes. After each surgery, the fish was allowed to recover in a tank containing fresh water for 10-30 minutes and then released near the site of capture.

Tracking efforts were conducted approximately weekly on Libby Reservoir using a 23 foot long Woolridge outboard motorized boat during the period of mid December 2003 to May 2004, and were limited to daylight hours. We used telemetry receivers manufactured by Lotek Inc. (Model SRX-400; W7 Firmware) for locating both coded acoustic and combined acoustic and radio tags. Each receiver unit consisted of a radio receiver, data processor, internal clock, and data logger. We used a tuned loop antenna for locating the radio component of the combined acoustic and radio tags. For locating the coded acoustic tags, we used an ultrasonic upconverter (model UUCN-150) and a hydrophone (model LHP-1) manufactured by Lotek Inc. that were connected to the SRX-400 telemetry receiver. The ultrasonic upconverter modified the 76.8 KHz signal transmitted by the acoustic tags and converted it 150.077 KHz, which was capable of being decoded by the telemetry receiver. We used triangulation methodology to estimate the position of each tagged fish. We recorded the date, time, approximate depth (m), general location, and used a Global Positioning System (GPS) to identify and geo-reference the location of each tagged fish. We estimated the home range for those burbot for which we had relocated at least once after release by measuring the furthest distance (along the reservoir mid-channel) between upstream and downstream observations.

Stream Macroinvertebrate Monitoring

Montana Fish, Wildlife & Parks used macroinvertebrates to assess the effectiveness of two ongoing restoration projects. One project was on Libby Creek and was completed in 2002 (See Chapter 2). This study compares samples collected from the site in 2000 with samples collected in 2003—about 1-year after restoration. The other project was completed on Grave Creek in 2002 and compared samples collected in 2002 (prior to restoration) with samples from 2003. The projects used the same field- and laboratory-methods as well as a similar experimental design.

Since the principle reason for most restoration projects—including these projects—is to improve ecosystem function, it is sensible that the success of restoration efforts be evaluated using a measure of ecological function. Macroinvertebrates are a useful for this type of assessment because they are diverse, abundant and respond to changes in their environment relatively quickly. Moreover, they are critical for proper ecosystem function.

Libby Creek Cleveland Project

Montana FWP has applied a watershed approach to the restoration of Libby Creek. In September, 2002 Montana FWP restored about 3,200 linear feet of stream channel at the Upper Libby Creek Cleveland (ULCC) Project (located at river mile 22). The goal was to restore proper dimension, pattern and profile to the stream channel. This was required because previous activities in the area (i.e., logging, mining, riparian road construction, stream channel manipulation) have accelerated bank erosion; causing reduced habitat quality for salmonid fishes; including bull trout and redband trout (Dunnigan et al. 2003).

In 2003, we began to evaluate the effectiveness of the restoration project using benthic macroinvertebrates. Macroinvertebrates assemblages will often respond to stream restoration more rapidly than fish. This is partially due to their relatively short lifecycles and high reproductive capacity and adult's air-borne mobility. Moreover macroinvertebrates are ideal indicators of community function and are important forage for fish.

There are many ways to use macroinvertebrates for biological assessment and biological monitoring. We used a combination of methods based upon the Regional bioassement protocols, qualitative interpretation, and statistical analysis. This report's purpose is to describe the changes in the benthic community related to the ULCC restoration through several objectives. First, we statistically compare biological metrics and the contribution of different functional feeding groups by sampling before and after the restoration activities. Second, we evaluate the "quality" of Upper Libby Creek by comparing it to regional reference criteria developed for Montana DEQ. Third, we qualitatively describe some of the reasons for changes in metrics as related to community response to changes in the watershed. An ideal assessment might include secondary production estimates for macroinvertebrates and fish, however we believe these analyses provide a cost-effective assessment of changes in the ecology of Upper Libby Creek, at the Cleveland restoration site.

We compared samples collected in September 2000 (before the ULCC restoration) with samples collected in August 2003. Three samples were collected from separate riffles with a Surber Sampler (0.098 m^2, 500-µm mesh), preserved in the field with ethanol and sent to EcoAnalysts, Inc. (Moscow ID) for analysis.

We used the standard protocols for macroinvertebrate sample analysis, as outlined by Bukantis (1998) for the Montana DEQ. These protocols use a 300-organism subsample and genus- or species-level taxonomic determinations for all organisms. Occasionally specimen condition or maturity prohibited taxonomy to the species level, and taxa were identified to family or genus only.

Rapid bioassessment protocols (RBP) for Montana compare biological metrics from study site to a population of regional reference sites located throughout the state. The original framework was developed by Bahls et al. (1992) and modified by Bukantis (1998), Bollman (1998), and Marshall and Kerans (2003). We used the metrics recommended by Marshall and Kerans (2003) for the assessment of mountain streams in Montana because statistical power analysis indicated these metrics were sensitive to environmental changes.

It is important to note that RBPs and their regional reference criteria are not scaled to assess local-scale or reach-scale ecological changes. Furthermore, the field collection methods used by an RBP design use different techniques and are not replicated. Therefore we used a simple replicated statistical design to test for differences before and after restoration. To avoid ethical problems with data-snooping, we selected the metrics prior to analysis. We included the metrics recommended by Marshall and Kerans (2003) because they have been successfully used in Montana mountain streams. We also used several others that have historically been used in Montana that summarize different kinds of information. We used these metrics because they summarize portions of the community and provide indicators of community function.

The metrics were included Taxa Richness, EPT index (e.g., , Contribution of Ephemeroptera, Plecoptera, and Tricoptera-taxa), Simpsons Diversity (D), the Ratio of Baetidae: Ephemeroptera, the Contribution of Diptera, the proportional abundance of burrowers and sprawlers (sediment tolerant), and the contribution of collector-gatherers. The metrics were compared using two-sample t-tests. The critical level of alpha was kept at 0.05, following the conventions of modern ecology.

The Grave Creek Phase I Restoration Project

Montana FWP entered into a cooperative agreement that was coordinated through the Kootenai River Network to retain a consultant to develop and implement a restoration plan for approximately 4,300 feet of channel within the lower three miles of Grave Creek (WCI 2002). Additional contributors to the project included Montana Department of Environmental Quality, the National Fish and Wildlife Foundation, the Steele-Reese Foundation, the U.S. Fish and Wildlife Service (Partners for Wildlife Program), the Montana Community Foundation, the Montana Trout Foundation, and the Cadeau Foundation. The project is termed the Grave Creek Phase I Restoration Project, and begins at the downstream end of the Grave Creek Demonstration Project (see Dunnigan et al. 2003). Stream restoration work began and was completed during the fall of 2002. Numerous structures were installed including 12 rootwad composites, 11 debris jams, 8 log J-hook vanes, 4 cobble patches, 3 log cross vanes, 1 rock cross vane, 1 rock J-hook vane, 1 straight log vane, and 2.4 acres of sod transplants. The restoration project changed the physical habitat of this section of Grave Creek, which generally resulted in a narrower, deeper stream channel with improved habitat for native salmonids (see Chapter 2 and Dunnigan et al. 2003). Initial monitoring indicated that the restoration successfully reinforced the eroding banks. Additionally, the alterations significantly improved the profile, pattern and dimension of the streambed in the study site (Dunnigan et al. 2003).

We used the same methods and experimental design as for the Libby Creek Cleveland Project (above). This study compares data (3 replicates) from 2002 (pre-restoration) to data collected from 2003 (approximately 1 year after restoration). We compared differences in benthic assemblages between years using two-sample t-tests of biological metrics. Variances were not equal in both years, so the p-values used assume non-equal variances between the two treatments.

Juvenile Salmonid Population Estimates

Montana FWP conducted juvenile salmonid population estimates on Sinclair, Therriault, Young, Libby, Grave, Parmenter, Pipe, and Barron creeks in 2001 and 2002, as part of an effort to monitor long-term trends in juvenile salmonid abundance, size distribution and species composition. We conducted estimates on each stream with mobile electrofishing gear using DC current for multiple pass depletions similar to Shepard and Graham (1983). We placed a block net at the lower end of each section and electrofished from the upper end of the section towards the lower end. After two such passes were completed, we estimated the probability of capture (P) using the following formula.

$$P = C1 - C2 / C1$$

Where: C1 = number of fish >75 mm total length captured during first catch and
C2 = number of fish > 75 mm total length captured during second catch.

Generally, if, based on captures made during the first two passes, P was ≥ 0.6, a third pass was conducted. Population estimates were performed for fish ≥ 75 mm, in order to make estimates consistent with historic data collected prior to 1997. Population estimates and associated 95% confidence intervals were estimated using *Microfish 2.2* (Van Deventer and Platts 1983). A description of reach sampled in 2001 and 2002 follows for each stream

Therriault Creek

We established three monitoring sections in Therriault Creek for juvenile salmonid trend analyses (Hoffman et al. 2002). Section one starts at the Highway 93 culvert and proceeds 82 m upstream. Section 2 starts at the first culvert above highway 93 and proceeds 120 m downstream. The property is privately owned and the stream channel is highly entrenched with unstable banks and is within the restoration project that is scheduled to begin in the later spring 2004. Section 3 starts at the second culvert above highway 93 and extends downstream 131 m. This section is moderately stable and is 400 m upstream from the highly entrenched reach of Therriault Creek.

Grave Creek

We established a representative sampling reach on Grave Creek to perform population estimates. The shocking section begins at the Vukonich property bridge and extends downstream 1,000 feet to the beginning of the demonstration project area. Baseline fish population data for Grave Creek prior to the completion of the demonstration project were collected in 2000 and 2001.

Due to the high volume of water in lower Grave Creek, a CPUE was conducted rather than the usual depletion population estimate in 2000 and 2001. We used a Coleman Crawdad electrofishing boat with a mobile electrode to sample this section. The system consisted of a Cofelt model VVP-15 rectifier powered by a 4000 watt generator. Our estimates are for fish $\geq$ 75 mm long (total length, TL) for consistency with data previously collected on other Kootenai River tributaries. Sampling in 2002 was limited to snorkel observations due to the presence of >2,000 adult kokanee salmon in the monitoring section. Two observers moved slowly upstream enumerating trout estimated to be $\geq$ 75 mm total length.

Young Creek

Montana FWP previously established five monitoring sections on Young Creek for use as trend indicators of juvenile salmonid abundance. These five sections include the following.

- Section 1: Tooley Lake Section (Sec.23 T37N,R28W).
- Section 2: Meadow Section, near confluence with Spring Creek (Sec.15,T37N,R29W).
- Section 3: Dodge Creek Spur Road #303A (Sec.17 T37N,R28W).
- Section 4: Dodge Creek Road #303, upstream from bridge (Sec. 18 T37N,R28W).
- Section 5: North Fork 92 meters from confluence of North and South Forks (Sec. 5,T37N,R29W).

We conducted population estimates on Sections 1, 3,4 and 5 in 2001 and 2002.

Libby Creek

MFWP personnel collected fish population information in three reference reaches on Libby Creek from 1998 through 2002. We sampled Section 1 using a Coleman Crawdad electrofishing boat with a mobile electrode. The other sections were sampled with a Smith Root backpack electrofisher. The system consisted of a Cofelt model VVP-15 rectifier powered by a 4000 watt generator. The three sections sampled in 2001 and 2002 include the following.

- Section 1: is a 274 m long reach located approximately 2.4 km below the Highway 2 bridge.
- Section 2: is a 171 m long reach located ~100 m upstream of the Highway 2 bridge.
- Section 3: is a 171 m long reach located on the upper Cleveland property.

The Cleveland property has had a lengthy history of site disturbance dating back over a century of mineral exploration (Sato 2000). Stream restoration activities were initiated on Libby Creek at Sections 1 and 2 in 2001 and 2002, respectively (See Chapter 2). Fisheries population work at these two sites was intended to assess fish population response to restoration activities.

Parmenter Creek

The Parmenter Creek drainage has a lengthy history of repetitive flooding. Parmenter Creek is generally stable until it exits a confined valley approximately 2.5 miles above the confluence with the Kootenai River. Flood plain encroachment and channel manipulation have substantially reduced stream stability. The valley mouth is an alluvial fan, which is a natural sediment depositional area. In attempts to control flooding, the stream was channelized and confined to the highest point on the alluvial fan. This left many houses at lower elevations than the streambed that substantially exacerbated the effects of flooding. Lincoln County was the lead entity responsible for overseeing the implementation of a stream restoration project on lower Parmenter Creek in 2000 to help alleviate many of the problems occurring on lower Parmenter Creek (Hoffman et al. 2002). Montana FWP established a fisheries monitoring section within the restoration area in 2000, and sampled that reach in 2000 (Hoffman et al. 2002) and 2001.

Pipe Creek

Montana FWP personnel established a single monitoring section on lower Pipe Creek in 2001 below the Bothman Road Bridge at approximately 0.25 miles upstream of the confluence. This section was established in order to collect biological information in anticipation of a stream restoration project on lower Pipe Creek. This section was sampled during the 2003 field season.

Libby Reservoir Gillnet Monitoring

Montana FWP has used gillnets since 1975 to assess annual trends in fish populations and species composition. These yearly sampling series were accomplished using criteria established by Huston et al. (1984). Data presented in this report focus on the period 1988 through 2002, but in several instances the entire database (1975 through 2002) is presented to show long-term catch trends.

Netting methods remained similar to those reported in Chisholm et al. (1989). Netting effort has continually been reduced since it was first initiated in 1975. During the period 1975-1987 a total of 128 ganged (coupled) nets were fished. This was reduced to 56 in 1988-1990, and reduced again to 28 ganged floating and 28 single sinking nets in 1991-1999. Effort was further reduced from 2000 to present to 14 ganged nets. Furthermore, netting effort occurred in the spring and fall, rather than the year round effort prior to 1988. Only fish exhibiting morphometric characteristics of pure cutthroat (scale size, presence of basibranchial teeth, spotting pattern and presence of a red slash on each side of the jaw along the dentary) were identified as westslope cutthroat trout; all others were identified as rainbow trout (Leary et al. 1983). Kamloops (Gerrard and Duncan strain) rainbow trout were distinguished from wild rainbow trout by eroded fins (pectoral, dorsal and caudal); these fish are held in the hatchery until release into the reservoir at age 1+. These fish are also marked (tetracycline) prior to release into the reservoir that allows post-mortem age and origin determination.

Species abbreviations used throughout this report are: rainbow trout (RB), Kamloops rainbow trout (KAM), westslope cutthroat trout (WCT), rainbow X cutthroat hybrids (HB), bull trout (BT), kokanee salmon (KOK), mountain whitefish (MWF), burbot (LING), peamouth chub (CRC), northern pikeminnow (NPM), redside shiner (RSS), largescale sucker (CSU), longnose sucker (FSU), and yellow perch (YP).

The year was stratified into two gillnetting seasons based on reservoir operation and surface water temperature criteria:

1) Spring (April - June): The reservoir was being refilled, surface water temperatures increased to 9 - 13°C.

2) Fall (September - October): Drafting of the reservoir began, surface water temperature decreased to 13 - 17°C.

Seasonal and annual changes in fish abundance within the nearshore zone were assessed using floating and sinking horizontal gillnets. These nets were 38.1 m long and 1.8 m deep and consisted of five equal panels of 19-, 25-, 32-, 38-, and 51-mm mesh.

Fourteen to twenty-eight floating (ganged) and one or two single, sinking nets were set in the fall in the Tenmile, Rexford and Canada portions of the reservoir. Spring netting series consisted of 20 to 111 (standardized to 28 in 1991) sinking nets and an occasional floating net set only in the Rexford area. Spring floating and fall sinking net data are not included in this report due to a lack of standardization in net placement. Nets were set perpendicular from the shoreline in the afternoon and were retrieved before noon the following day. All fish were removed from the nets and identified, followed by collection of length, weight, sex and maturity data. Scales and a limited number of otoliths were collected for age and growth analysis. When large gamefish (Kamloops rainbow, cutthroat, bull trout or burbot) were captured alive, only a length was recorded prior to release.

Libby Reservoir Zooplankton Monitoring

Montana FWP has collected zooplankton from Libby Reservoir since 1983 in an attempt to relate changes in density and structure of the community to parameters of other aquatic communities, as well as to collect data indicative of reservoir processes, including aging and the effects of reservoir operation. We performed monthly vertical zooplankton tows using a 0.3 m, 153µ Wisconsin net in each of three reservoir areas (Tenmile, Rexford and Canada) from 1983 to 1996. However, beginning in 1997, we reduced sampling effort to the period April through November, after a rigorous analysis indicated we would not compromise our ability to identify trends (Hoffman et al. 2002). In an effort to further standardize sampling methodologies, we experimented with the effects of sample depth on the resulting analyses. When we excluded samples of greater than 20 m, the results were statistically similar (Kruska-Wallis $p = 0.05$; Hoffman et al. 2002) relative to analyses including depths of 30 m with regards to total zooplankton abundance. These results corroborate previous from Schindler trap sampling that found that approximately 90% of all zooplankton captured were from depths of 20 m or less (Skaar et al. 1996). Therefore, beginning in 1997, we conducted 20 m sampling tows when depth permitted, and when depth was between 10 and 20 m we sampled the entire water column. We did not collect samples when depth was less than 10 m. This differed from sampling protocols used from 1983 through 1989, where one sample was taken from a permanent station and two samples were taken randomly in each area, regardless of water depth. However, we made two sampling protocol changes that were implemented in 1990 that included the following. We only collected zooplankton samples when depth was at least 10 m, and all sampling locations (reservoir mile) and bank (east, west or middle) were randomly selected. All samples were pulled at a rate of 1 m/second to minimize backwash (Leathe and Graham 1982).

Zooplankton samples were preserved in a water / methyl alcohol / formalin / acetic acid solution from September 1986 to November 1986. After December 1986, all samples were preserved in 95% ethyl alcohol to enhance egg retention in Cladocerans.

Low density samples (<500 organisms total) were counted in their entirety. High-density samples were diluted to a density of 80 to 100 organisms in each of five, five ml aliquots. The average of the five aliquots was used to determine density. We randomly subsampled and measured the length of 33-34 *Daphnia*, *Diaptomus*, *Epischura* and *Diaphanosoma*. We used analysis of variance, and subsequent multiple comparisons to assess whether zooplankton abundance differed by month and sampling area in 2001 and 2002.

Results

Bull Trout Redd Counts

Grave Creek

MFWP counted redds in the Grave Creek Basin (including Blue Sky, Clarence, Williams and Lewis Creeks) for the first time in 1983, as well as in 1984, 1985, and 1993 through 2003. Grave Creek was surveyed from its confluence with the Tobacco River upstream to near the mouth of Lewis Creek (approximately13 miles), where it becomes intermittent. Most redds in Grave Creek were located upstream from the mouth of Clarence Creek to the confluence with Lewis Creek. Surveyors found 10 redds between the confluence with the Tobacco River and one mile below Clarence Creek in 1983 However, we did not find redds in this reach during surveys conducted in 1993 and 2000. The distribution of bull trout redds in Blue Sky, Clarence, Williams and Lewis creeks was similar to observations in previous years (Hoffman et al. 2002).

We observed 245 bull trout redds in Grave Creek in 2003, which was a record number for the period of record (Table 1). Bull trout have exhibited a significant positive trend in spawning abundance in Grave Creek since 1993 (Figure 2; $r^2 = 0.794$; $p = 0.0002$).

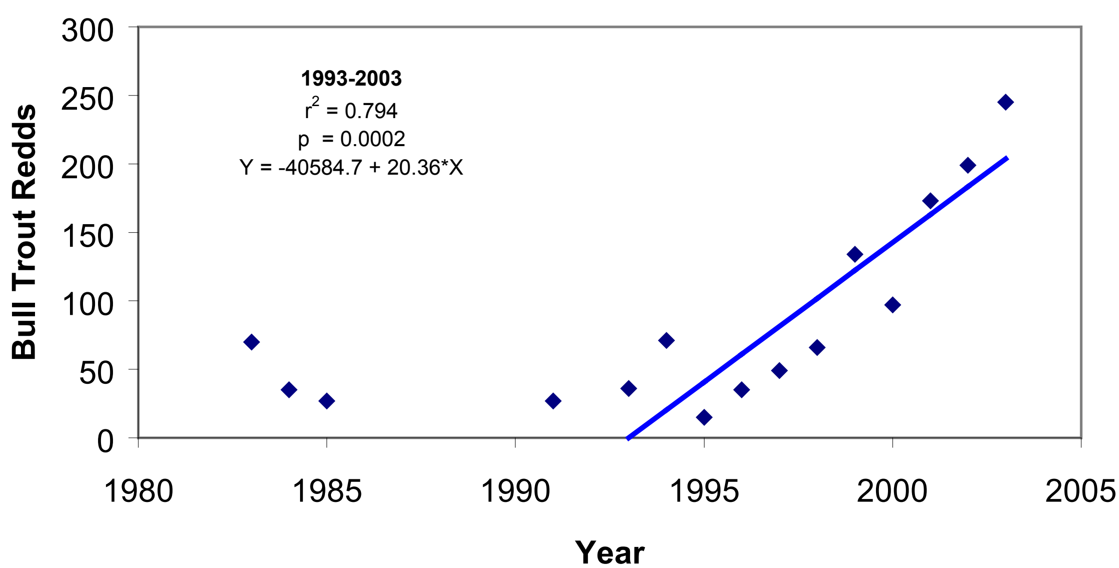

Figure 2. Bull trout redd counts, and trend analysis in Grave Creek, 1993 through 2003.

Wigwam Drainage

Bull trout redd counts for the Wigwam River includes the tributary streams of Bighorn, Desolation, and Lodgepole creeks, and the portion of the Wigwam River within Montana. A total of 2053 bull trout redds were observed in the Wigwam Drainage in 2003, which was a record high since counts began (Table 1). This Bull trout redds in the Wigwam River have consistently increased each year since 1995 (Figure 3; $r^2 = 0.959$; $p = 4.18*10^{-6}$).

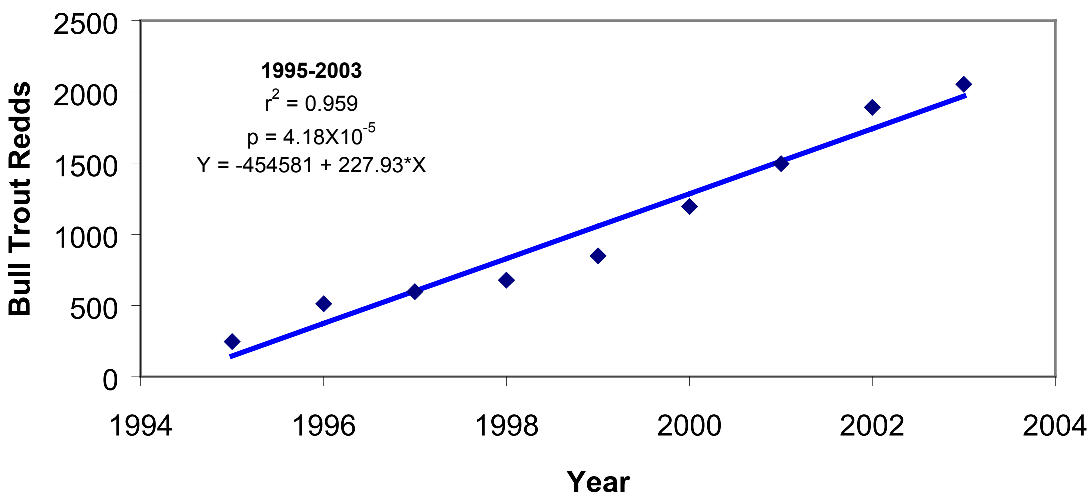

Figure 3. Bull trout redd counts and trend analysis for the Wigwam River (including Bighorn, Desolation, and Lodgepole creeks) 1995-2003.

Table 1. Bull trout redd survey summary for all index tributaries in the Kootenai River Basin.

Stream	Year Surveyed	Number of Redds	Miles Surveyed
Grave Creek	1995	15	9
Includes Clarence and Blue Sky Creeks	1996	35	17
	1997	49	9
	1998	66	9
	1999	134	9
	2000	97	9
	2001	173	9
	2002	199	9
	2003	245	9
Quartz Creek	1995	66	12.5
Includes West Fork and Mainstem	1996	47	12.0
	1997	69	12.0
	1998	105	8.5
	1999	102	8.5
	2000	91	8.5
	2001	154	8.5
	2002	62[e]	8.5
	2003	55	8.5
O'Brien Creek	1995	22	4.5
	1996	12	4.0
	1997	36	4.3
	1998	47	4.3
	1999	37	4.3
	2000	34	4.3
	2001	47	4.3
	2002	45	4.3
	2003	46	4.3
Pipe Creek	1995	5	10
	1996	17	12.0
	1997	26	8.0
	1998	34	8.0
	1999	36	8.0
	2000	30	8.0
	2001	6[a]	8.0
	2002	11	8.0
	2003	10	8.0
Bear	1995	6	3.0
	1996	10	4.5
	1997	13	4.25
	1998	22	4.25
	1999[b]	36	4.25
	2000	23	4.25
	2001	4[e]	4.25
	2002	17	4.25
	2003	14	4.25
Keeler	1996	74	9.3
Includes South and North Forks	1997	59	8.9
	1998	92	8.9
	1999	99	8.9
	2000	90	8.9
	2001	13[d]	8.9
Keeler Creek (Continued)	2002	102	8.9

Table 1. Bull trout redd survey summary for all index tributaries in the Kootenai River Basin.

Stream	Year Surveyed	Number of Redds	Miles Surveyed
	2003	87	8.9
West Fisher River	1995	3	10
	1996	4	6
	1997	0	6
	1998	8	6
	1999	18	10
	2000	23	10
	2001	1	10
	2002	1	6
	2003	1	6
Wigwam (B.C and U.S.)	1995	247	22
Includes Bighorn, Desolation, Lodgepole Creeks	1996	512	22
	1997	598	22
	1998	679	22
	1999	849	22
	2000	1195	22
	2001	1496	22
	2002	1892	22
	2003	2053	22
Skookumchuck Creek (B.C.)	1997	66	1.9
	1998	105	1.9
	1999	161	1.9
	2000	189	1.9
	2001	132	1.9
	2002	143	1.9
	2003	134	
White River (B.C.)	2001	166	7.8
Includes Blackfoot Creek in 2002 and 2003	2002	261	7.8
	2003	249	

a: Human built dam below traditional spawning area
b: Included resident and migratory redds
c: Libby Creek dewatered at Highway 2 bridge below spawning sites during spawning run
d: Beavers dammed lower portion during low flows, dam was removed but high water made accurate redd counts impossible
e: Log jam may have been a partial barrier

Note that during low water years, beavers in some streams (Keeler, Pipe, Quartz) have an opportunity to build dams across entire stream rather than just in side channels. Some bull trout migrate upstream before dam construction is complete, most either try to build redds below the dams or appear to leave the streams entirely. This happened in Keeler Creek and Pipe Creek in 2001.

Quartz Creek

Bull trout redd counts in Quartz Creek since 1995 have been variable (Figure 4; r^2 = 0.107). Although overall trend is positive, annual variation limits our ability to statistically distinguish this relationship from a stable (zero slope) population (Figure 4; p = 0.252). We observed a total of 55 redds in Quartz and West Fork Quartz creeks in 2003 (Table 1). The average number of redds of the period of record was 76.7 redds. The 2003 observation of 55 redds was 28.3% lower than the average over the period of record. A log jam located approximately 0.25 miles upstream of the confluence of West Fork Quartz Creek in 2003 may have limited bull trout spawner escapement in 2002 and 2003.

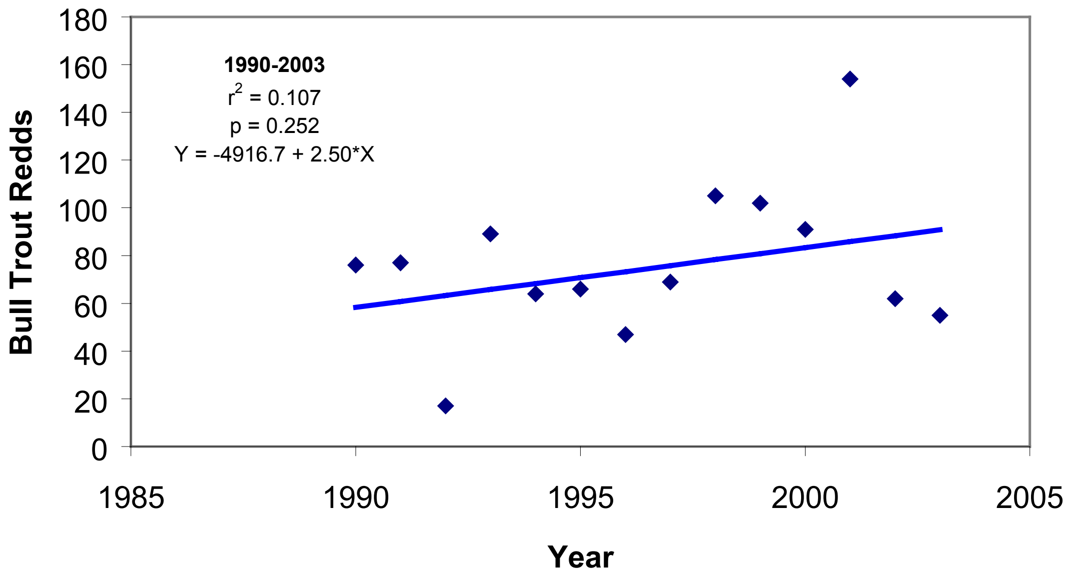

Figure 4. Bull trout redd counts and trend analysis (blue line) for Quartz Creek (including West Fork Quartz) 1990-2003.

Pipe Creek

Bull trout redd counts in Pipe Creek peaked in 1999 with 36 redds, with redd numbers and have decreased since that peak. Despite the decreasing trend of bull trout redds during the last four years, the overall general trend during the time period 1995-2003 has been variable, with a slope that is not significantly different than a stable population (Figure 5; $r^2 = 0.149$; $p = 0.173$). The mean number of bull trout redds since 1990 has been 15 redds. The 10 redds we observed in Pipe Creek in 2003 was 33.3% lower than the 13 year average. Low water conditions during the fall spawning season during the past three years may partially explain the low spawner escapement into Pipe Creek.

Bear Creek

Bear Creek bull trout redd counts have been variable during the period 1995-2003 (Figure 6; $r^2 = 0.03$). Although the overall general trend has increased since 1995, the relationship is not statistically different than a stable population (Figure 6; $p = 0.668$). Low water conditions in Bear Creek during the past three years also partially explain the low spawner escapement in Bear Creek. The average number of bull trout redds since 1995 in Bear Creek has been 16.1 redds. The 14 redds we observed in Bear Creek in 2003 was 13.1% less than the 8 year average.

O'Brien Creek

The general trend of bull trout redds in O'Brien Creek is generally increasing since 1995 (Figure 7; $r^2 = 0.592$; $p = 0.002$). We observed a total of 46 bull trout redds in O'Brien Creek in 2003 (Table 1).

West Fisher River

We were unable to determine a significant trend in bull trout redds in the West Fisher River over the period of record for this stream (1993-2003). From the period 1993-2000, the general trend was one of increasing abundance. However, we observed only 1 bull trout redd in each of the previous three years (Figure 8). The overall trend was not significantly different than a stable (zero slope) population (r2 = 0.036; $p = 0.578$). Given the amount of variation present within this dataset, the overall mean number of redds in the West Fisher (mean = 5.6 redds) does an equally good job at predicting redd numbers. Drought conditions during the previous 3 summers/late fall periods may have contributed to the lower bull trout spawner escapement into the West Fisher River.

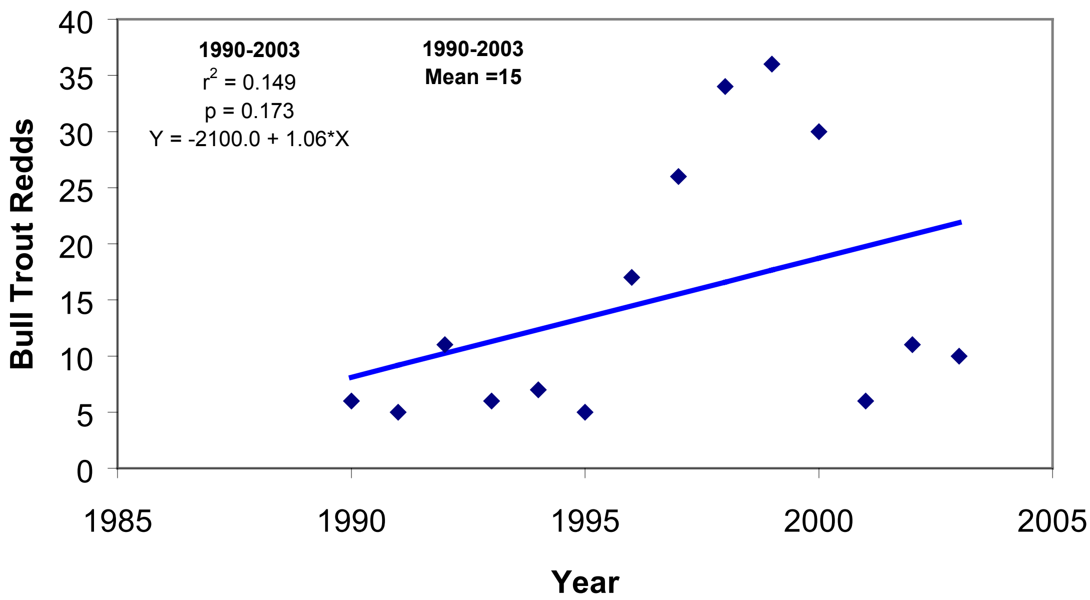

Figure 5. Bull trout redd counts and trend analysis (blue line) for Pipe Creek 1990-2003.

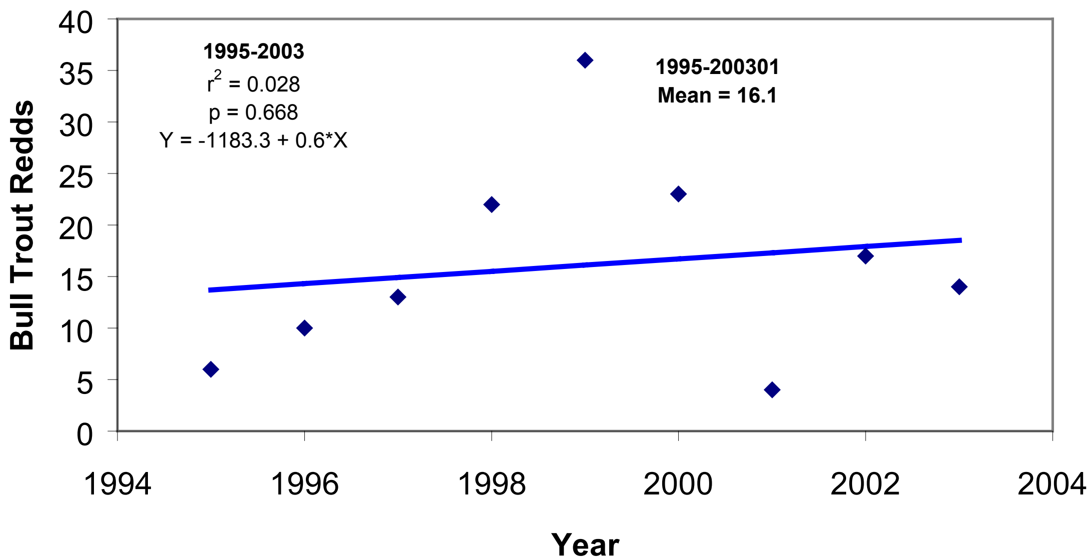

Figure 6. Bull trout redd counts and trend analysis (blue line) in Bear Creek, a tributary to Libby Creek, 1995-2003. The mean number of bull trout redds was 16.1.

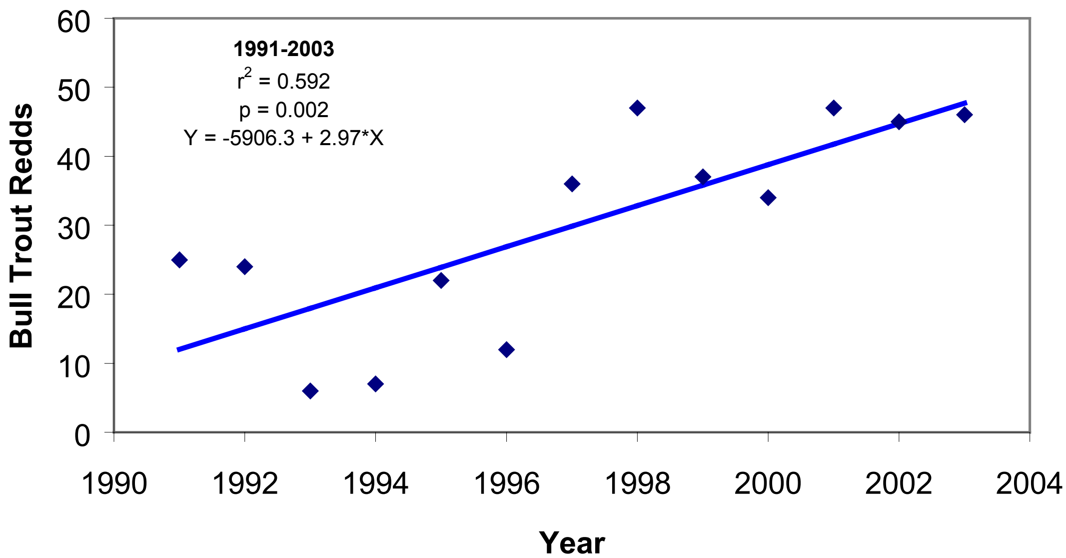

Figure 7. Bull trout redd counts and trend line (blue line) in O'Brien Creek 1991-2003.

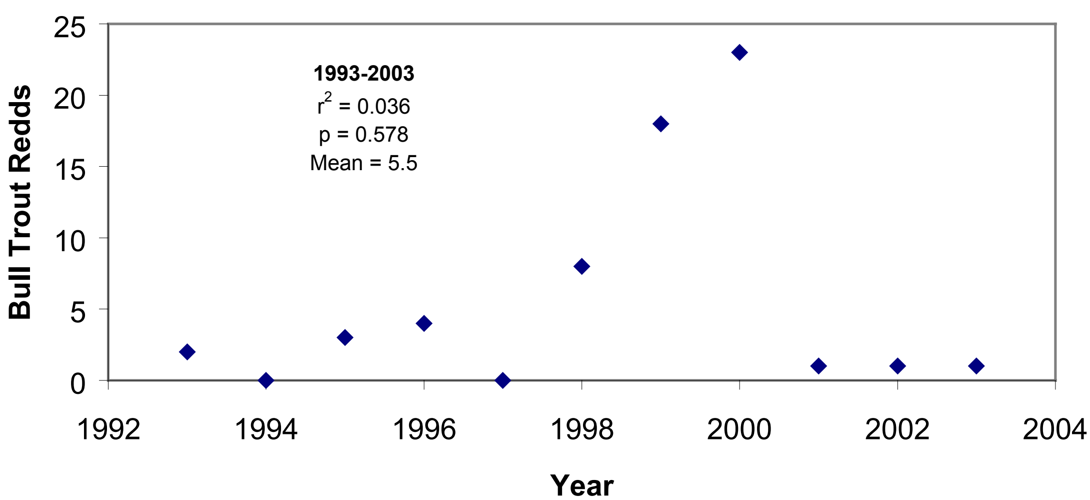

Figure 8. Bull trout redd counts in the West Fisher River, a tributary to the Fisher River, 1993-2003.

Keeler Creek

Bull trout that spawn in Keeler Creek (including the North, South and West Forks) are an adfluvial stock, that migrate downstream out of Bull Lake into Lake Creek, then up Keeler Creek. This downstream spawning migration is somewhat unique when compared to other bull trout populations (Montana Bull Trout Scientific Group 1996). Lake Creek, a tributary of the Kootenai River, has an upstream waterfall barrier isolating this population from the mainstem Kootenai River population. A micro-hydropower dam constructed in 1916 covered the upper portion of the waterfall. A series of high gradient waterfalls are still present below the dam, and are barriers to all upstream fish passage. Keeler Creek may supply some recruitment to the Kootenai River through downstream migration. We observed a total of 87 bull trout redds in Keeler Creek and associated tributaries in 2003 (Table 1). A beaver dam located in lower Keeler Creek during late summer/early fall 2001 likely impeded upstream bull trout migration. The dam was removed, but stream flow increased substantially after the dam was removed and prevented counts from being made after removal of the dam. Therefore, the 13 redds observed in 2001 is an underestimate of the true number of redds in Keeler Creek in 2001. With the 2001 observation included, annual variation is high ($r^2 = 0.004$; Figure 9), although the trend is an increasing population trend, although the relationship is not significantly different from a stable population (Figure 9; $p = 0.889$). Given this relationship, the annual mean (77 redds) does an equally good job of prediction. The 2003 observation represents a 13% increase relative to the annual mean. However, if we remove the 2001 observation from the dataset and repeat the regression trend analysis, bull trout redds in Keeler Creek show a nearly significant increasing trend since 1996 (Figure 9; $r^2 = 0.366$; $p = 0.150$).

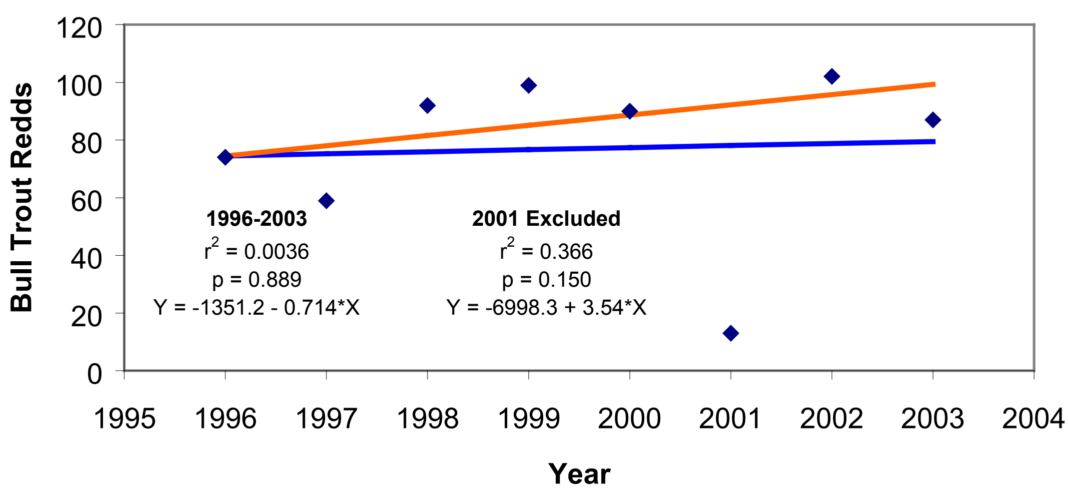

Figure 9. Bull trout redd counts and trend line (blue line) in Keeler Creek, a tributary to Lake Creek, 1996-2003. A beaver dam was present in lower Keeler Creek in the fall of 2001 that likely impeded bull trout migration. Therefore the 2001 observation was removed and the regression analysis was repeated (orange line).

Kootenai River Adult Bull Trout Population Estimate

We marked a total of 109 bull trout on April 8 and 15, 2004 below Libby Dam, and captured a total of 116 bull trout during the recapture runs, of which 13 bull trout were marked. We estimated an overall capture efficiency of 12%. The average bull trout total length was 649 mm (range = 343 – 861 mm; Figure 10). We estimated a total of 920 bull trout in the Kootenai Dam from Libby Dam downstream to the Fisher River confluence. The 95% confidence interval ranged from 698 – 1142 bull trout. We standardized the population estimate and 95% confidence interval into fish per mile, for a total of 263 bull trout per mile (95% confidence interval = 199-326).

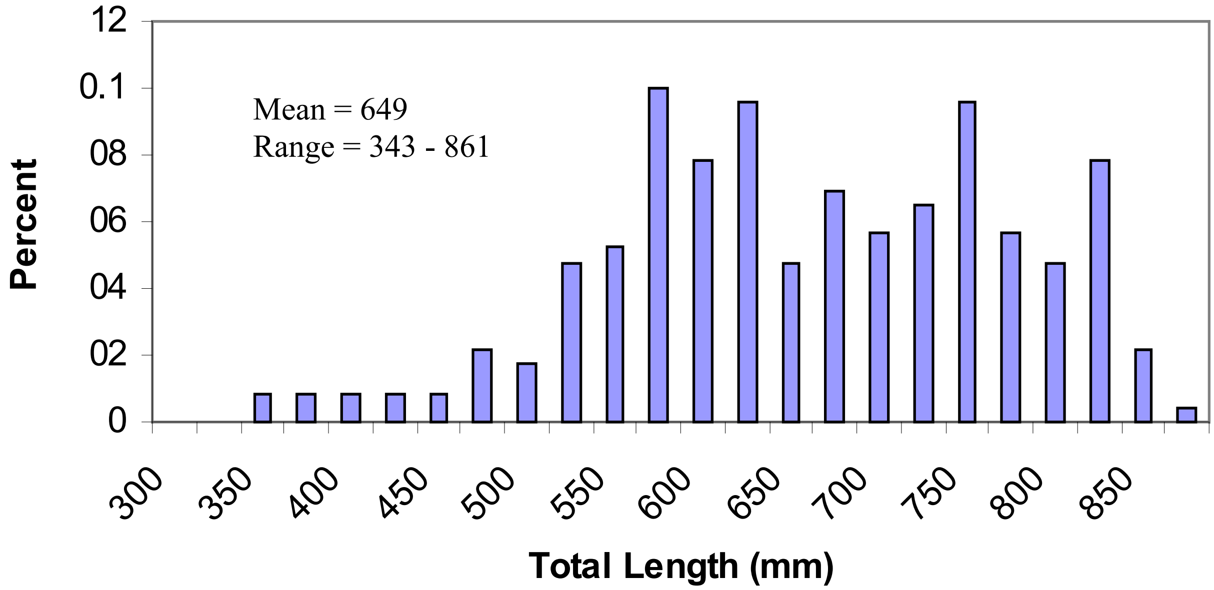

Figure 10. Length frequency distribution for bull trout captured via jet boat electrofishing on April 8 – 22, 2004 below Libby Dam.

Burbot Monitoring Below Libby Dam

The burbot catch in our hoop traps below Libby Dam has declined precipitously since 1996/1997 (Figure 12). A total of 3 burbot were captured during the 03/04 trapping season, which represented the lowest total catch and catch per effort (0.031 burbot per trap day) on record since trapping began in the 94-95 trapping season. The most numerous captures occurred in 1995-96 and 1996-97; these years correspond with higher than normal snow-pack, and perhaps greater reservoir drafting. The mean annual catch rate since the 1995/1996 trapping season was 0.668 burbot per trap day. However, the catch rates since then have significantly decreased (r^2 = 0.742; p = 0.003; Figure 12). This relationship was further improved using an exponential fit (r^2 = 0.930; p < 0.001; Figure 12).

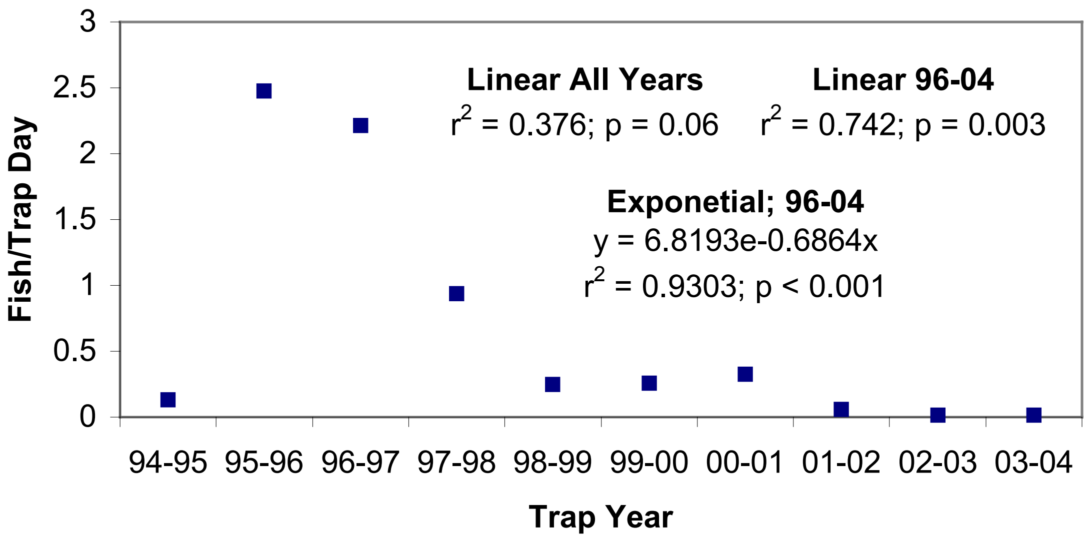

Figure 12. Total catch per effort (burbot per trap day) of baited hoop traps in the stilling basin downstream of Libby Dam 1994/1995 through 2003/2004. The data were fit with linear regression for all years (1994/1995 – 2003/2004), 1995/1996 – 2003/2004 and with an exponential model for 1995/1996 – 2003/2004. The traps are baited with kokanee salmon and fished during December and February.

Burbot Monitoring Libby Reservoir

During the period from November 14 to April 26, 2004 we expended a total effort of 1887 trap-days, and caught a total of 127 burbot at 10 trapping locations throughout the reservoir (Table 1). We were unable to trap the Tobacco River, Dodge and Young creek sites from January 3rd to March 25th, 2004 due to surface ice formation. Burbot catch at all trapping locations averaged 0.064 fish per trap-day. Mean burbot catch was highest near the mouth of Cripple Horse Creek and lowest near the mouth of Dodge Creek (Table 2). Although catch rates differed between sites, the observed differences were not significant when compared using an analysis of variance procedure ($p = 0.316$). The mean total length of burbot captured in the hoop traps was 576.1 mm (range 350-880 mm; Figure 13). Burbot mean length did not significantly differ between the nine trapping locations where we captured burbot ($p = 0.439$). The mean condition factor (K; Carlander 1969) for all burbot captured was 0.615. The length-weight relationship for burbot captured in Libby Reservoir is presented in Figure 14. We recaptured 3 burbot in the hoop traps that were previously captured and PIT tagged. One burbot was recaptured twice near Big Creek. This fish was originally captured and PIT tagged on 11/17/03, and subsequently recaptured on 11/20 and 12/5/03. Another burbot was originally PIT tagged near Cripple Horse Creek on 12/8/03 and recaptured near Barron Creek on 3/1/04. The third burbot recaptured was marked in the Tobacco Bay area on 11/25 and recaptured near the capture release site on 4/8/04.

We used multiple linear regression to evaluate trends in burbot catch versus trapping date and hoop trap depth. We found no evidence to suggest that catch rates differed by trapping date (Figure 15; $r^2 = 0.007$; $p = 0.794$). We did however, find a significant relationship between trap depths and catch rates when we grouped trap depths into categories of 3 m intervals and averaged the proportion of those traps containing burbot for those depth categories (Figure 16; $r^2 = 0.620$; $p = 0.012$). This relationship suggested that catch rates increased up to approximately 11 m and then remained relatively constant up to the 33 m, which was the maximum depth we trapped.

Table 2. Summary information for the baited hoop trapping effort for burbot on Libby Reservoir. Montana FWP used 2 types of tags to investigate the life history of burbot for this study; acoustic (A) and combined radio/acoustic tags (C).

Sample Site	**River Mile**	**Dates Trapped**	**Total Trap-Days**	**Total Burbot Catch**	**Catch per Trap-Day[1]**	**Catch per Trap-Day[2]**	**Tags Released**
Canyon Crk.	222.5	1/12 – 3/25	219	16	0.073	0.063	2-C
Cripple Horse Crk.	226.8	11/14 – 1/12	177	17	0.096	0.102	11-A
Barron Crk.	228.7	2/27 – 4/26	177	11	0.062	0.076	0
Bristow Crk.	231.5	1/12 - 2/27	138	7	0.051	0.054	1-A & 2-C
Ten Mile Crk.	235.8	1/16 – 3/25	195	10	0.051	0.047	1-A & 2-C
Big Crk.	248.7	11/14 – 1/16	193	12	0.062	0.076	7-A
Sutton Crk.	250.3	1/16 – 4/26	291	24	0.082	0.084	2-C
Dodge Crk.	264.1	11/14 – 12/16	96	0	0	0	0
Tobacco River	264.0	11/14 – 1/13 & 3/25/04 – 4/26/04	254	20	0.079	0.067	4-A & 2-C
Young Crk.	268.4	12/16 – 1/2/04 & 3/25/04 – 4/26/04	147	10	0.068	0.072	4-A & 2-C
Total / Mean			**1887**	**127**	**0.062**	**0.064**	**28-A & 12-C**

[1] Average burbot catch per trap-day calculated as the average catch for each trap set at each sample site.
[2] Average burbot catch per trap-day calculated by dividing the total burbot catch by total trap-days.

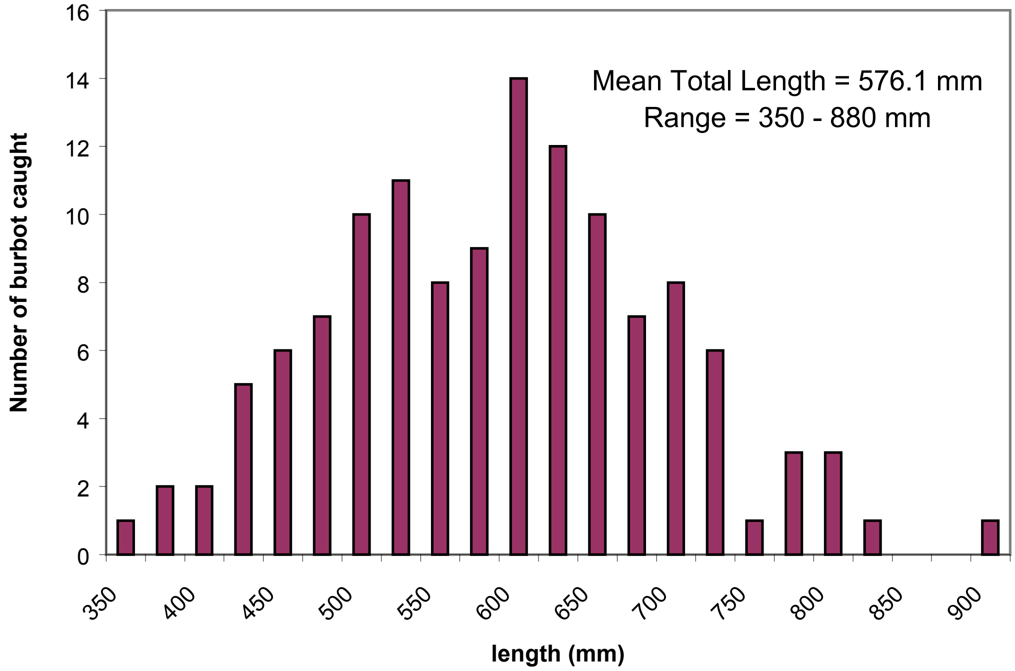

Figure 13. Length frequency distribution for burbot captured in baited hoop traps in Libby Reservoir, 2003-2004.

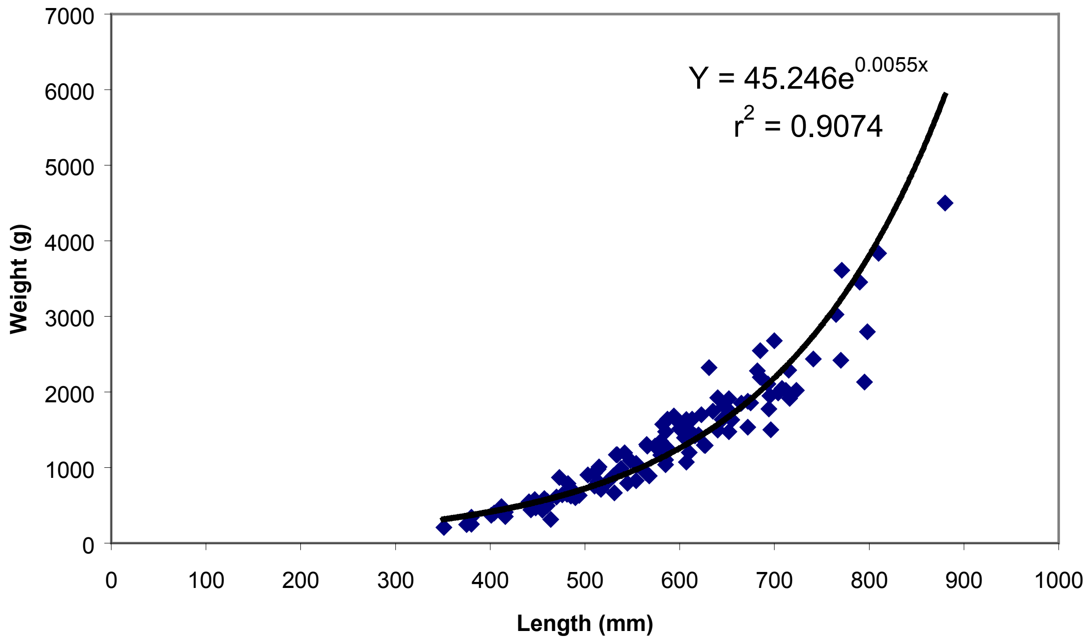

Figure 14. Length weight relationship for burbot captured in baited hoop traps in Libby Reservoir 2003-2004.

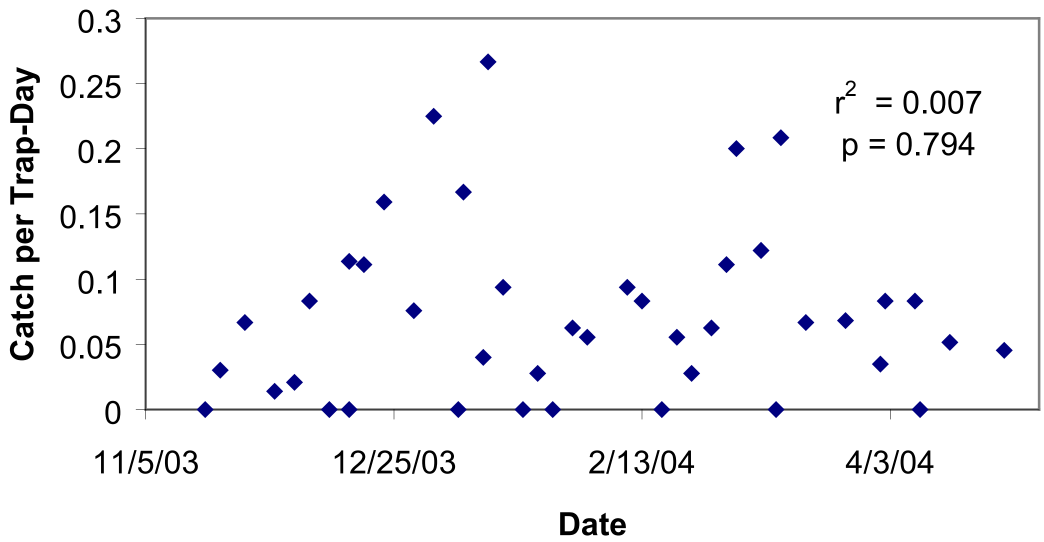

Figure 15. Scatter plot of mean burbot catch rates (catch per trap-day) versus trap date.

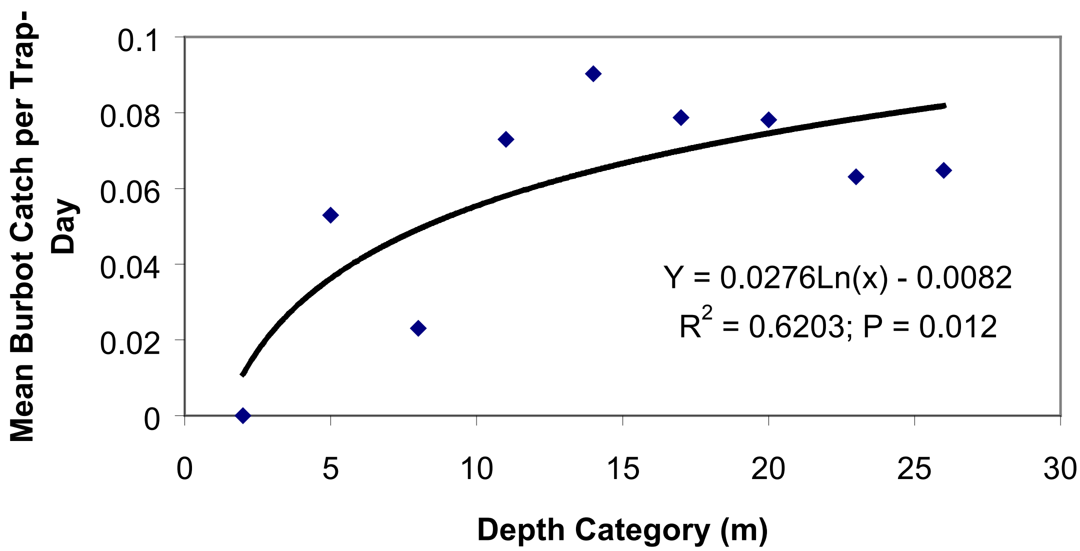

Figure 16. Scatter plot and regression of mean burbot catch rates (catch per trap-day) for 3 meter depth categories ranging from 2 to 33 meters.

The Libby FWP Mitigation Staff surgically implanted 28 coded acoustic and 12 combined radio/acoustic tags in burbot 6 trapping sites, respectively (Table 3). Each tag had a unique frequency or code to allow for individual fish identification. The first burbot was tagged on November 20, 2003 and the last burbot was tagged on April 26, 2004. The Libby FWP Mitigation staff has expended approximately 1-2 days per week since trapping efforts began in November to search for tagged fish using primarily the acoustic gear. Currently we have relocated the 40 burbot tagged with a coded acoustic and combined radio/acoustic transmitters an average of 1.8 times per fish (Figure 17). However, there were 3 combined radio/acoustic and and 7 acoustic tagged burbot, that were not observed since release. We estimated that the mean home range was 6524 m (range 166 – 27470 m; Table 3) for the 30 tagged burbot that relocated at least once. The mean estimated depth of burbot that were relocated during daylight searches using acoustic gear was 35.6 m, which was significantly deeper ($p = 2.47*10^{-67}$) than the mean depth which we operated traps (14.8 m). We did not perform any nighttime acoustic observations. We were not able to discern any clear movement patterns for the tagged fish in terms of either upstream or downstream movement. Of the 40 tagged burbot, 14 of these fish had 3 or more relocation observations, six of these fish moving upstream, 7 moved downstream and one fish showed relatively little movement. However, we were able to determine that many of the tagged burbot extensively utilized the old Kootenai River channel and floodplain during daylight hours (Figure 18). For example, of the 14 tagged burbot for which we had at least 3 relocation observations, 11 of those fish were consistently and repeatedly using either the old Kootenai River channel or floodplain area. On average, the acoustic and acoustic/radio tags will be active until early 2006, which will allow us to collect additional observations on the tagged fish, which in turn will provide additional insight into the spawning distribution and habitat preferences of burbot within Libby Reservoir.

Table 3. Capture location and summary observation information for 28 acoustic and 12 combined radio/acoustic tagged burbot in Libby Reservoir. The range of observed depths for each fish is in parentheses.

Capture Location	Tag Code	Tag Type	River Mile	Date released	Number of Observations	Home range (m)	Observation Average Depth (m)
Big Creek	30	S	248.7	12/23/03	3	27470	47.7 (20.0 – 65.2)
Big Creek	169	S	248.7	12/29/03	3	27400	39.2 (20.0 – 60.1)
Big Creek	9	S	248.7	1/14/04	3	16920	48.7 (20.0 – 60.9)
Big Creek	156	S	248.7	1/2/04	3	6890	21.0 (15.2 – 24.7)
Big Creek	22	S	248.7	12/23/03	1	4860	32.6 (20.0 – 45.1)
Big Creek	7	S	248.7	1/14/04	5	1010	46.8 (20.0 – 60.0)
Big Creek	14	S	248.7	12/16/03	0		20.0
Bristow Creek	114	C	231.5	2/19/04	1	9880	30.4 (20.0 – 36.3)
Bristow Creek	44	C	231.5	2/25/04	1	3950	42.6 20.0 – 60.0)
Bristow Creek	5	S	231.5	1/20/04	2	3350	30.3 (13.1 – 57.9)
Canyon Creek	100	C	222.5	3/1/04	1	13170	37.1 20.0 – 71.3)
Canyon Creek	2	C	222.5	2/2/04	1	410	16.3 12.5 – 20.0)
Cripple Horse	143	S	226.8	12/29/03	3	14350	46.3 (20.0 – 66.7)
Cripple Horse	183	S	226.8	1/8/04	4	8070	49.9 (20.0 – 71.3)
Cripple Horse	16	S	226.8	12/23/03	2	5610	36.5 (18.3 – 71.3)
Cripple Horse	138	S	226.8	12/29/03	4	4750	44.6 (12.2 – 66.8)
Cripple Horse	178	S	226.8	1/8/04	3	3060	45.0 (20.0 – 61.9)
Cripple Horse	111	S	226.8	1/8/04	5	2120	35.4 (20.0 – 61,9)
Cripple Horse	45	S	226.8	12/29/03	5	1470	45.3 (20.0 – 61.9)
Cripple Horse	193	S	226.8	1/8/04	4	1370	43.2 (20.0 – 55.8)
Cripple Horse	204	S	226.8	1/8/04	4	470	25.7 (3.0 – 49.7)
Cripple Horse	10	S	226.8	12/2/03	2	200	38.4 (20.0 – 69.5)
Cripple Horse	12	S	226.8	11/20/03	0		20.0
Sutton Creek	109	C	250.3	2/2/04	3	3370	35.5 (20.0 – 48.7)

Table 2. (Continued) Capture location and summary observation information for 28 acoustic and 12 combined radio/acoustic tagged burbot in Libby Reservoir. The range of observed depths for each fish is in parentheses.

Capture Location	Tag Code	Tag Type	River Mile	Date released	Number of Observations	Home range (m)	Observation Average Depth (m)
Sutton Creek	15	C	250.3	2/2/04	2	1530	37.1 (20.0 – 46.0)
Ten Mile Creek	34	C	235.8	2/25/04	1	8270	51.7 (20.0 – 80.5)
Ten Mile Creek	8	S	235.8	2/13/04	2	710	18.8 (12.2 – 24.1)
Ten Mile Creek	29	C	235.8	3/12/04	0		20.0
Tobacco Bay	101	S	264.0	12/29/03	1	7040	41.3 (20.0 – 62.5)
Tobacco Bay	18	C	264.0	4/8/04	1	1470	20.0
Tobacco Bay	1	C	264.0	4/8/04	1	1230	20.0
Tobacco Bay	6	S	264.0	1/13/04	0		20.0
Tobacco Bay	11	S	264.0	12/23/03	0		20.0
Tobacco Bay	203	S	264.0	1/13/04	0		20.0
Young Creek	13	S	268.4	12/23/03	2	15150	31.4 (20 – 42.7)
Young Creek	152	S	268.4	1/2/04	1	160	23.0 (20 – 25.9)
Young Creek	4	S	268.4	1/13/04	0		5.0
Young Creek	40	S	268.4	12/29/03	0		20.0
Young Creek	129	C	268.4	4/8/04	0		20.0
Young Creek	121	C	268.4	4/26/04	0		20.0
Mean					**1.8 (all tags)** **2.2 (acoustic)** **1.0 (combined)**	**6524**	**36 (18.0 – 52.0)**

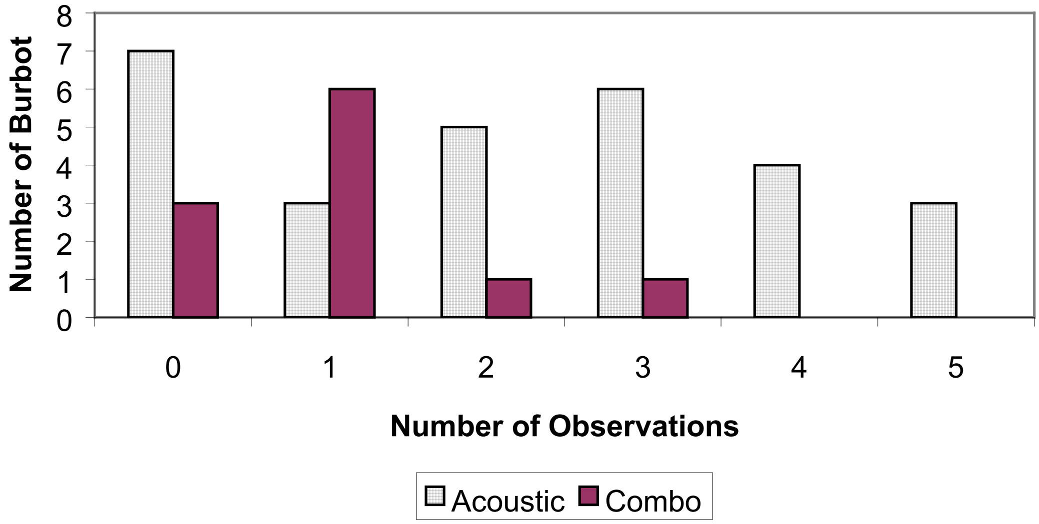

Figure 17. Histogram of the number of acoustic observations for burbot tagged with acoustic and combined radio/acoustic tags in Libby Reservoir 2003-2004.

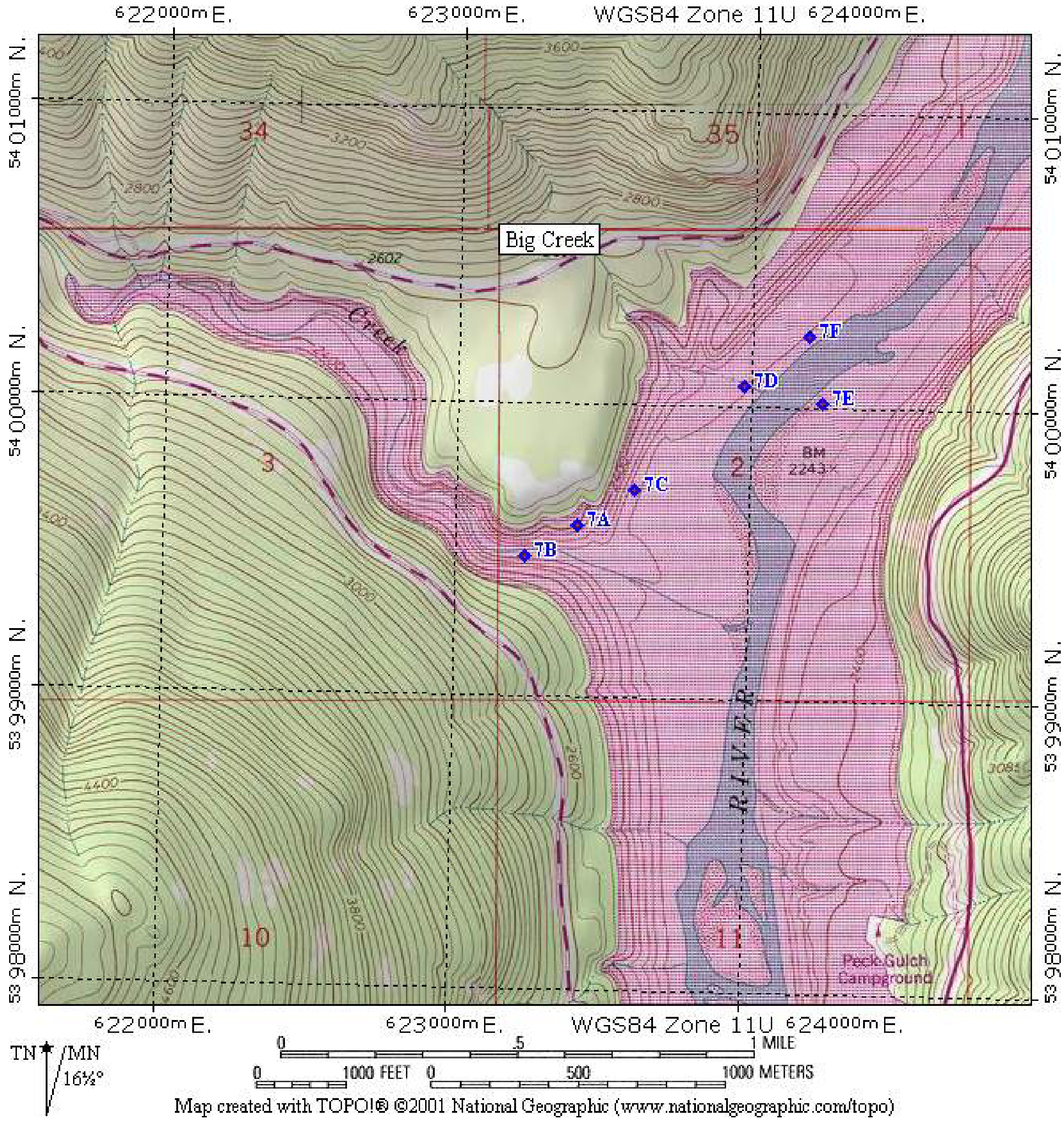

Figure 18. Capture locations for a typical burbot tagged with an acoustic tag. This particular burbot was captured on January 14, 2004 (Point A) in a baited hoop trap fished near the confluence of Big Creek (river mile 248.7). This fish was relocated on 1/21/04, 2/11/04, 2/20/04, 2/25/04, and 3/3/04, Points B thru E, respectively.

Stream Macroinvertebrate Monitoring

Libby Creek Cleveland Project

Of the seven metrics selected before the study, 3 had values in 2003 that were significantly different from the values attained from the 2000 sampling (Table 4. However, they were all in the opposite direction expected. That is, we expected the samples to reflect an improvement after restoration, but the directions of change for the three significantly different metrics were consistent with disturbance. We specifically expected the relative abundance of sensitive organisms (% EPT) to increase and the dominance of tolerant organisms (% Diptera and % Collector-gatherers) to decrease as conditions become more natural. We observed the opposite and this usually indicates an ecosystem is that has been recently disturbed.

The change in the dominance of EPT organisms (Ephemeroptera, Plecoptera, Trichoptera) was mainly due to a significant reduction in the abundance of Plecoptera (Table 4, but was also accompanied by a slight (not statistically significant) reduction in the abundance of caddisflies. The dominant Plecoptera taxon was *Sweltsa* sp., which is small predatory stonefly. *Sweltsa* was present before and after the restoration, but dropped from an average of 10.8% (in 2000) to 3.8% in 2003. There may be several reasons for this shift, including climatic factors, or increased predation (by fish or larger macroinvertebrates). *Sweltsa* is somewhat intolerant to sedimentation.

The change in the percent Diptera and collector-gatherers was due to an increase in the abundance of chironomid midges. Specifically two midge taxa increased and raised the average: *Micropsectra* (a collector-filterer from the tribe tanytarsini) and *Tvetenia* (a collector-gatherer from the subfamily Orthocladiinae). Generally an increase in the abundance of midges is consistent with disturbance. The mean abundance of midges increased from an average of 2.7% (in 2000) to 17.5% in 2003. Marshall and Kerans (2003) identified thresholds for deviation from MT DEQ reference conditions at between 13% (moderate) to 18% (extreme). However, the Montana DEQ samples use a courser mesh-size (1000 µm) and should sample fewer small organisms—like midges. Considering this a value of 17% does not sound extreme, even though there was a large increase among years. In fact, 2.7% chironomidae with a 500 µm -mesh Surber seems exceptionally low. Both *Micropsectra* and *Tvetenia* are somewhat tolerant to sedimentation.

The increase in collector-gatherers was also accompanied by a significant increase in collector filterers and together the total increase in collectors was significant. Collectors are generalists and increase when specialists (such as grazers, shredders, and predators) decline. The use of functional feeding groups as biological metrics has been criticized (Karr and Chu 1999) because they are often highly variable; making it difficult to detect significant effects, even with large changes. However, when

Table 4. T-test results. This table shows the result of t-tests for the Libby Creek upper Cleveland restoration study. Hypothesized response assumes describes if the metric should increase or decrease if conditions in the stream improve. For example, Taxa Richness should increase if the restoration improved the quality of the stream for macroinvertebrates. Similarly, as more taxa inhabit a healthy stream we expect the dominance of sediment tolerant groups (such as Collectors, Sprawlers, or burrowers) should decrease.

A priori / post hoc **Analysis**	**Metrics**	**Hypothesized Response to Restoration**	**Pre-Restoration Mean**	**Post-Restoration Mean**	**P**
a priori	Taxa Richness	Increase	33	35	0.515
a priori	EPT Richness	Increase	25	28	0.347
a priori	% EPT	Increase	95.0	81.4	0.019*
a priori	Baetidae: Ephemeroptera	Decrease	0.068	0.078	0.702
a priori	% Diptera	Decrease	3.46	18.25	0.021*
a priori	% Burrowers & Sprawlers	Decrease	22.3	18.5	0.849
a priori	% Collector Gatherers	Decrease	8.28	23.27	0.004*
post hoc	% Collector Filterers[1]	Decrease	2.75	6.16	0.008*
post hoc	% Collectors[2]	Decrease	11.03	29.43	0.003*
post hoc	% Predators[3]	?	16.6	11.4	0.040*
post hoc	% Shredders[4]	Increase	13.58	3.40	0.015*
post hoc	% Ephemeroptera[5]	Increase	61.2	60.0	0.674
post hoc	% Plecoptera[6]	Increase	26.3	11.7	<0.001*
post hoc	% Trichoptera[7]	Increase	7.5	9.7	0.210
post hoc	% Dominant taxa (3)	Decrease	55.76	48.38	0.276

[1] % Collector-filterers was tested post hoc to explain the examine changes in community function.

[2] % Collectors total was tested post hoc to explain the examine changes in community function.

[3] % Predators was tested post hoc to explain the examine changes in community function.

[4] % Shredders was tested post hoc to explain the examine changes in community function.

[5] % Ephemeroptera was tested post hoc to explain the cause of differences in the % EPT metric.

[6] % Plecoptera was tested post hoc to explain the cause of differences in the % EPT metric.

[7] % Plecoptera was tested post hoc to explain the cause of differences in the % EPT metric.

differences occur they usually indicate a shift in the trophic composition of the community. Thus these changes may indicate changes in community function as well as community structure. In addition, collectors often benefit from the addition of fine particulate organic material; which may come from animal grazing, sewage particulates or some forms of sedimentation. The increase in collectors was also accompanied by a decrease in the Shredders and Predators (Table 4 and a slight (not significant) decrease in scrapers.

Shredders are the specialists that typify mountain streams. That is, they represent a functional feeding group that should be abundant at reaches higher in the river continuum (e.g., Vannote et al. 1980). Their decline may be due to the sites failing to retain coarse particulate organic material, such as leaves, bark, pinecones, smaller limbs, or other coarse detritus. These items make up the principle forage for shredders and the shredder-aided breakdown of course detritus is one of the key functional characteristics of mountain streams.

Predators are common in many stream types and are not more important in mountain streams than in any other stream system. The combination of declining shredders and predators in a mountain streams can be disconcerting for resource managers because many of taxa in these groups are large stoneflies; which are excellent forage for fish. In addition, many are cool stenotherms; requiring consistently cool temperatures and high oxygen concentrations for success. Thus declines in these groups can also suggest situations that may become stressful for salmonids.

However, a closer look at these data indicates that these changes in function are not outside of range of normal levels for Mountain streams. Shredders are usually aggregated on deposits of organic detritus, and therefore have a wide range of mean abundances usually from about 3-25%. Predators usually comprise about 10 – 30. So the values are within the expected range. More importantly the changes are largely due to the increase in the number of collectors—which were probably under represented in the year 2000 sampling. That is, the combined abundance of collector-filterers and collector gatherers is usually much higher than the 11% reported for the year 2000. The collectors were higher in 2003 (23%), which caused a significant decrease in other functional feeding groups because of the fixed-count subsampling procedure used.

For comparison with regional streams, we ran the metrics through the Montana Mountains metric battery proposed by Marshall and Kerans (2003). The evaluation compares 6 metrics to two regional deviation thresholds (1 moderate, 1 extreme). If the metric is beyond the extreme deviation threshold, there is a 1% chance of type-1 and type-2 error and the metric scores 2 deviation points. If a metric is between the two thresholds it is moderately deviant from the reference condition, there is a 15% chance of error, and the metric scores 1 deviation point. If the metric is not beyond the moderate deviation threshold it is not different from the values attained from the states reference sites and scores zero deviation points. The deviation points for all 6 metrics are summed and the stream is classified according to the total sum. The classification scores are 0-3- Normal; 4-7 moderate deviant; 8-11 extreme deviant; 12 non-functioning. In both 2000 and 2003, the stream scored 3; normal (Figure 19). This needs to be cautiously interpreted, because the mesh size and sampling methods were different from the criteria used to develop the MT DEQ reference criteria.

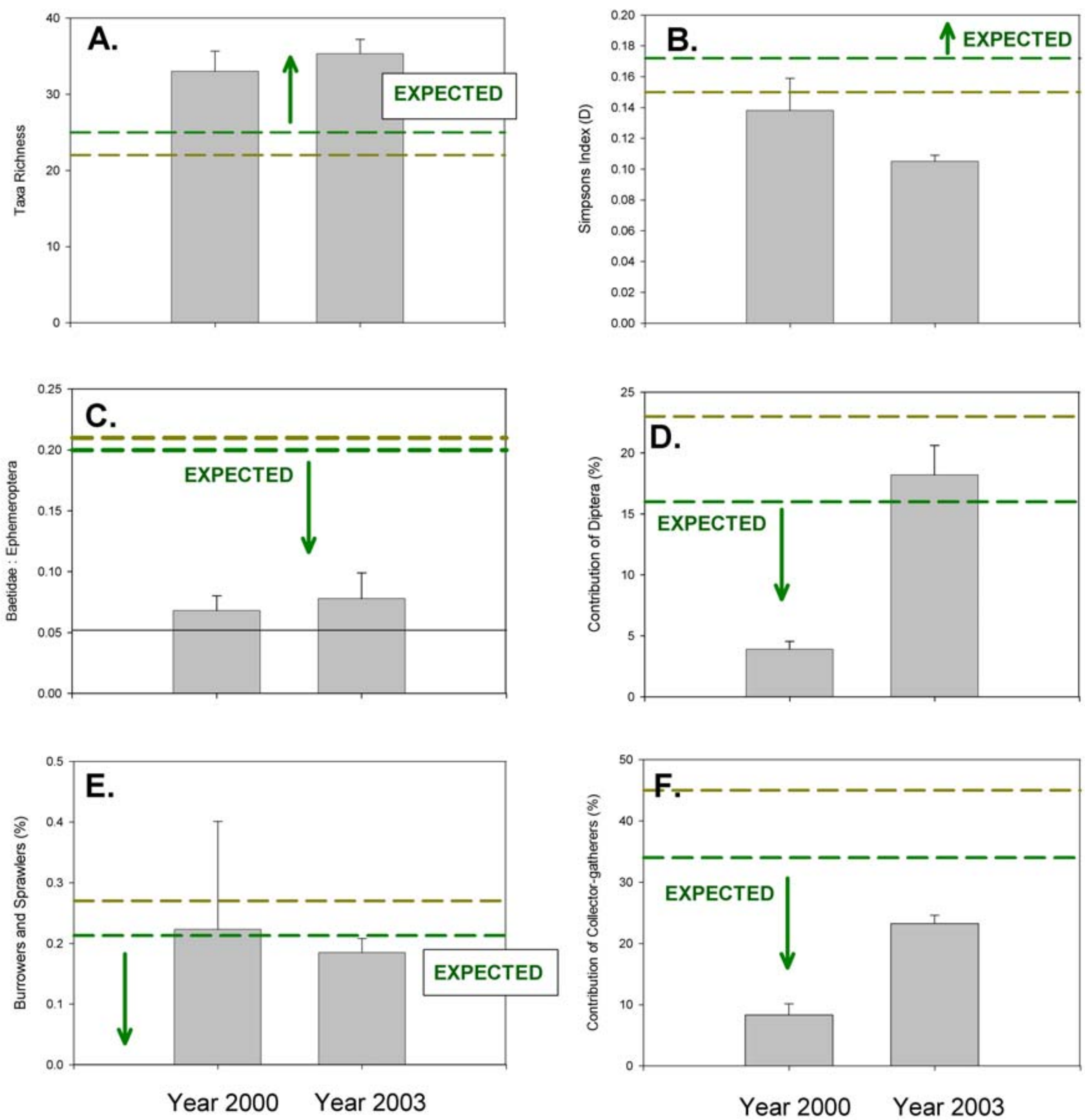

Figure 19. Montana Mountain Metrics. These were the metrics that provided the best 6-metric suite for the mountain regions of Montana (Marshall and Kerans 2003). Each metric has two thresholds. If the mean was within the expected range of variation, it scored zero-deviation points. If it was beyond one threshold, it scores 1 deviation point for moderate deviation from reference conditions ($0.01 < P < 0.15$; $0.01 < \beta < 0.15$). If the metric extends past the extreme threshold, it scores 2 deviation points ($P < 0.01$; $\beta < 0.01$). The sum of the deviation points provides an Community Deviance Index (CDI) that can be used as an estimate of ecological disturbance. CDI values < 3 are very similar to reference conditions for all metrics. Samples where, $4 \leq CDI \leq 7$, are moderately deviant from reference conditions and CDI > 7 are very deviant from reference conditions. Both before and after restoration, the sites scored 3 but there were significant differences among the metrics %

Diptera and the contribution of collector-gatherers were both significantly greater after the restoration.

Grave Creek Phase I Restoration Project

None of the metrics tested showed a statistically significant difference between the 2002 samples and the 2003 samples (Table 5). This was true for both the *a priori* and the *post-hoc* metrics. Most of the metric means suggested that the differences between the years were small. A notable exception was an increase in the percent shredders in 2003 (P=0.054); which increased ~4x (from 1.9% in 2002, to 7.7% in 2003) but the variance increased in the 2003 sampling (2002 SD= 0.3; 2003 SD= 2.4). The increased variance in observed in 2003 reduced the statistical power of the test. Sample sizes of 3 (both years) achieved 60% power to detect the difference in the % shredders metric (estimated group standard deviations of 0.3 and 2.5 and with a significance level (alpha) of 0.05` using a two-sided two-sample t-test). Thus, there is a high (40%) probability of type-2 statistical error. This is important because shedders represent an important link between riparian and aquatic food webs. In addition, many shredders are large and excellent forage for fish. Resource managers may want to quantify the retention of course particulate organic matter in future assessments to document the success of this aspect of the restoration.

For comparison with regional streams, we ran the metrics through the Montana Mountains metric battery proposed by Marshall and Kerans (2003). The evaluation compares 6 metrics to two regional deviation thresholds (1 moderate, 1 extreme). If the metric is beyond the extreme deviation threshold, there is a 1% chance of type-1 and type-2 error and the metric scores 2 deviation points. If a metric is between the two thresholds it is moderately deviant from the reference condition, there is a 15% chance of error, and the metric scores 1 deviation point. If the metric is not beyond the moderate deviation threshold it not different from the values attained from the State's reference sites and scores zero deviation points. The deviation points for all 6 metrics are summed and the stream is classified according to the total sum.

Table 5. T-test results. This table shows the result of t-tests for the Grave Creek Phase I Restoration Project evaluation. Hypothesized response assumes describes if the metric should increase or decrease if conditions in the stream improve. For example, Taxa Richness should increase if the restoration improved the quality of the stream for macroinvertebrates. Similarly, as more taxa inhabit a healthy stream we expect the dominance of sediment tolerant groups (such as Collectors, Sprawlers, or burrowers) should decrease.

A priori / post hoc ANALYSIS	METRICS	HYPOTHESIZED RESPONSE TO RESTORATION	RESTORATION MEAN - 2002	RESTORATION MEAN - 2003	P
a priori	Taxa Richness	Increase	32.7	31	0.842
a priori	EPT Richness	Increase	21	18	0.584
a priori	% EPT	Increase	8.67	10.33	0.446
a priori	Baetidae: Ephemeroptera	Decrease	0.360	0.118	0.0850
a priori	% Diptera	Decrease	38.1	29.3	0.719
a priori	% Burrowers & Sprawlers	Decrease	18.5	22.3	0.849
a priori	% Collector Gatherers	Decrease	49.2	50.5	0.877
post hoc	% Collector Filterers[8]	Decrease	1.55	0.41	0.207
post hoc	% Collectors[9]	Decrease	50.7	50.7	> 0.999
post hoc	% Predators[10]	?	8.70	7.16	0.496
post hoc	% Shredders[11]	Increase	1.91	7.77	0.054
post hoc	% Ephemeroptera[12]	Increase	63.8	53.2	0.433
post hoc	% Plecoptera[13]	Increase	8.46	10.1	0.594
post hoc	% Trichoptera[14]	Increase	5.37	4.08	0.658
post hoc	% Dominant taxa (3)	Decrease	54.25	48.32	0.390

8 % Collector-filterers was tested post hoc to explain the examine changes in community function.
9 % Collectors total was tested post hoc to explain the examine changes in community function.
10 % Predators was tested post hoc to explain the examine changes in community function.
11 % Shredders was tested post hoc to explain the examine changes in community function.
12 % Ephemeroptera was tested post hoc to explain the cause of differences in the % EPT metric.
13 % Plecoptera was tested post hoc to explain the cause of differences in the % EPT metric.
14 % Plecoptera was tested post hoc to explain the cause of differences in the % EPT metric.

The classification scores are 0-3- Normal; 4-7 moderate deviant; 8-11 extreme deviant; 12 non-functioning. In 2002, Grave Creek scored 6—moderately deviant from mountain reference conditions. In 2003, the site scored 5—also moderately deviant from mountain reference conditions. Both years had about 50% of the community composed of collector-gatherers whereas reference conditions anticipate about < 33%. Similarly, both years (2002-38%; 2003-29%)had a greater proportion of the community composed of Dipera than anticipated for mountain reference sites (<16%). It is noteworthy that although no significant differences were observed in these metrics, that 2003 demonstrated a 10% improvement (reduction) in Diptera abundance. Statistical power to detect this difference was very low (5%) because of the extremely high variance among the three samples collected in 2002 (range 4-77%). The samples collected in 2003 had much lower variance (range 24-34%) and should be considered an improvement over previous years.

The five dominant taxa among all samples were *Baetis tricaudatus*, *Rhithrogena*, Heptageniidae, *Pericoma*, and *Drunella doddsi*. A couple of these are particularly noteworthy. Specifically, *D. doddsi* is sensitive to many stressors—especially sedimentation—and showed a dramatic increase in abundance (from 1.3 to 9%). Similarly, the abundance of the sensitive mayfly, *Rhithorgena*, increased from 8.1% to 17.8% of the sampled assemblage. The increases in these taxa corroborate findings that suggest that conditions were slightly better in 2003 than in 2002. More importantly, they suggest that conditions are suitable for invertebrates that are sensitive to disturbance.

Juvenile Salmonid Population Estimates

Therriault Creek

Rainbow trout abundance in Section 1 of Therriault Creek has significantly decreased from 1997-2003 ($r^2 = 0.822$; $p = 0.093$; Figure 20; Table A1). This site was not sampled in 2000-2002. The trend in brook trout abundance for this section has not differed significantly from a stable population ($r^2 = 0.093$; $p = 0.249$; Figure 20; Table A1), and has averaged 52 brook trout per 1,000 feet. We first observed bull trout at this site in 2003, with an estimated 11 bull trout per 1,000 feet (Figure 16).

Section 2 on Therriault Creek lies within the Therriault Creek Restoration Project area and was sampled in 1997-1999, 2001 and 2003. We observed rainbow, brook and bull trout at this site. We used linear regression to evaluate population trends for each of these three species, but weren't able to detect significant trends (Figure 21). Rainbow, brook and bull trout abundances at this site has averaged 74, 79 and 21 fish per 1,000 feet, respectively.

Section 3 on Therriault Creek is located upstream of the Therriault Creek Restoration Project area and was sampled in 1997-1999 and 2003. We observed rainbow and brook trout at this site each year, but bull trout were only observed in 2003 with an estimated abundance of 9.9 bull trout per 1,000 feet (Figure 18; Table A1). The trends of rainbow and brook trout abundance did not differ significantly from a population with zero slope ($p > 0.10$; Figure 22).

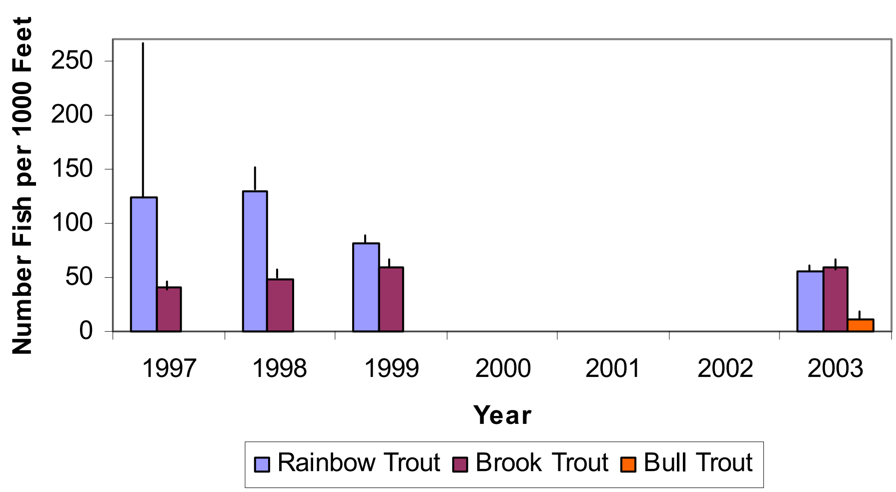

Figure 20. Cutthroat trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 1 monitoring site from 1997-1999 and 2003 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

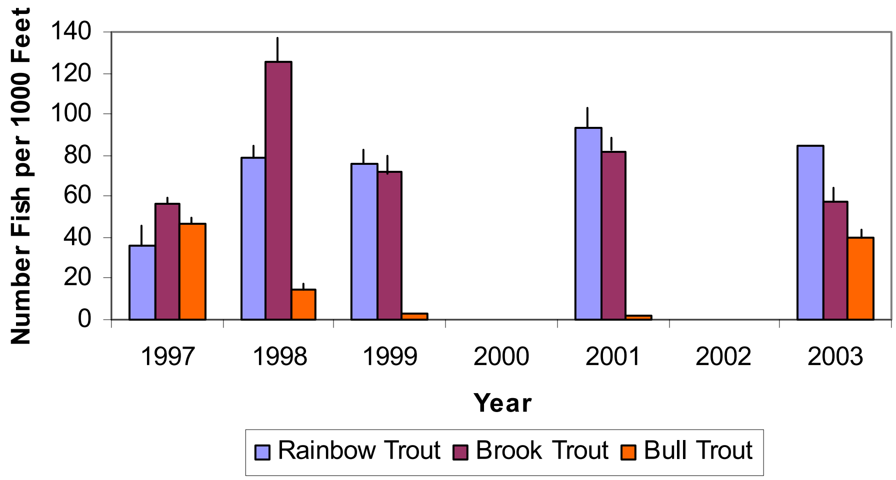

Figure 21. Cutthroat trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 2 monitoring site from 1997-1999, 2001 and 2003 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

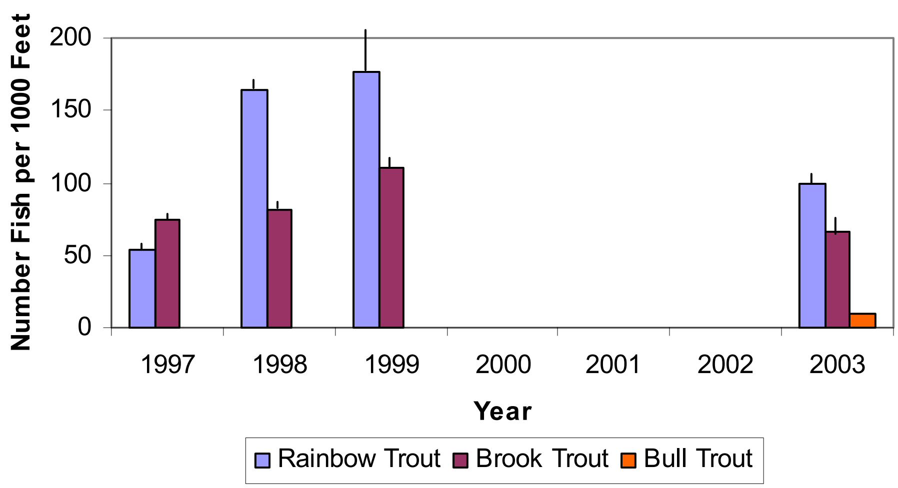

Figure 22. Cutthroat trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 3 monitoring site from 1997-1999 and 2003 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

Grave Creek

Juvenile salmonid monitoring within the Grave Creek Demonstration Project had two primary objectives, to determine fish population trends through time and to evaluate the fish community response to the restoration activities completed during the fall of 2001 (Grave Creek Demonstration Project). Cutthroat and Rainbow trout were the two combined most abundant fish species present at this site in all years except 2003, when juvenile bull trout were the most abundant species present (Table A2). We compared mean fish abundance (by species) for pre (2000-2001) and post (2002 and 2003) restoration projects using t-tests (Figure 23). However, the variability in pre and post project fish abundance estimates is high (Figure 23 and 24), and sampling methodology differed between years. These factors reduced our ability to distinguish statistical differences in abundance before and after project completion. Brook trout and bull trout abundance increased after at this site after project completion, although the differences were not significant (Figure 23). Rainbow trout abundance at this site did significantly increase from 9.0 to 24.5 fish per 1,000 feet after project completion ($p = 0.099$). Mean westslope cutthroat trout abundance decreased slightly, although not significantly ($p = 0.38$; Figure 23). We used linear regression to assess whether there was a temporal trend in abundance for the four fish species at this site (Figure 24). Although the r^2 values for the regression analyses for rainbow, brook and bull trout all exceeded 0.50, none of the trends differed significantly from a zero slope ($p > 0.1$; Figure 24). There was no apparent trend in westslope cutthroat trout abundance over the period 2000-2003 (Figure 24).

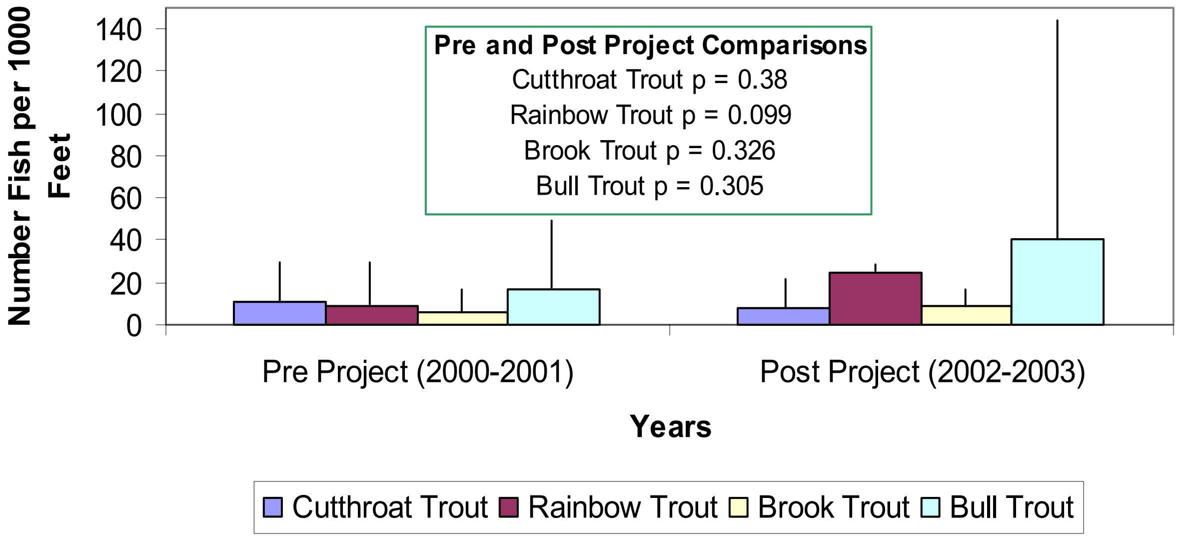

Figure 23. Mean cutthroat, rainbow, brook, and bull trout densities (fish per 1000 feet) within the Grave Creek Demonstration Project area prior to (2002-2001) and after (2002-2003) the completion of the Grave Creek Demonstration Restoration Project. Data collected during 2000 and 2001 represent pre-project implementation fish abundances and were collected using single pass electrofishing. Fish abundance data collected in 2002 represents post-project implementation fish abundances and was collected via snorkel counts. Upper 95% confidence intervals are represented by the whisker bars.

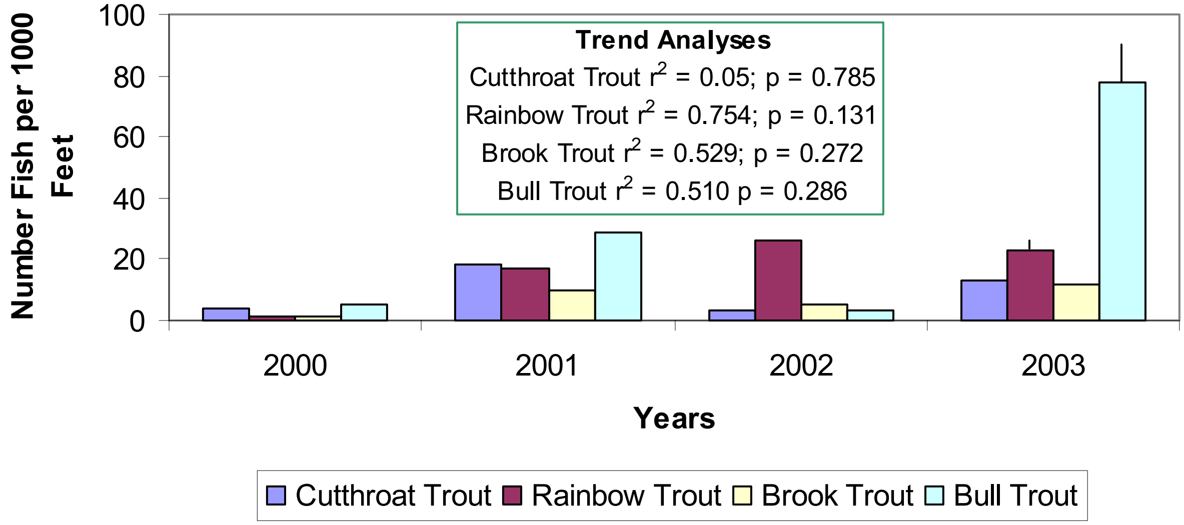

Figure 24. Cutthroat, rainbow, brook, and bull trout abundance estimates (fish per 1000 feet) and linear regression trend analyses within the Grave Creek Demonstration Project monitoring site from 2000-2003 collected by performing backpack electrofishing. The 2000 and 2001 data were collected using single pass electrofishing, the data collected in 2002 was collected via snorkel counts, and the 2003 data was collected using multiple pass electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

Young Creek

Section 5 was the only section of Young Creek sampled in 2003, and lies entirely within the stream restoration project completed on State land in the fall of 2003. Therefore, all data collected through 2003 represents data gathered prior to the restoration project completion. Cutthroat trout and brook trout have exhibited a stable population trends in Section 5 of Young Creek since 1998, with annual mean abundance estimates of 199.5 cutthroat trout per 1000 feet and 39.8 brook trout per 1000 feet (Figure 25; Table A3). Abundance estimates for cutthroat trout have ranged from 126 fish per 1000 feet in 2000 to 268 fish per 1000 feet in 2002. Brook trout abundance was also variable, ranging from 19 to 62 fish per 1000 feet (Figure 25). These data presented here will be used in future years to evaluate the fish community response to the restoration project during the fall of 2003.

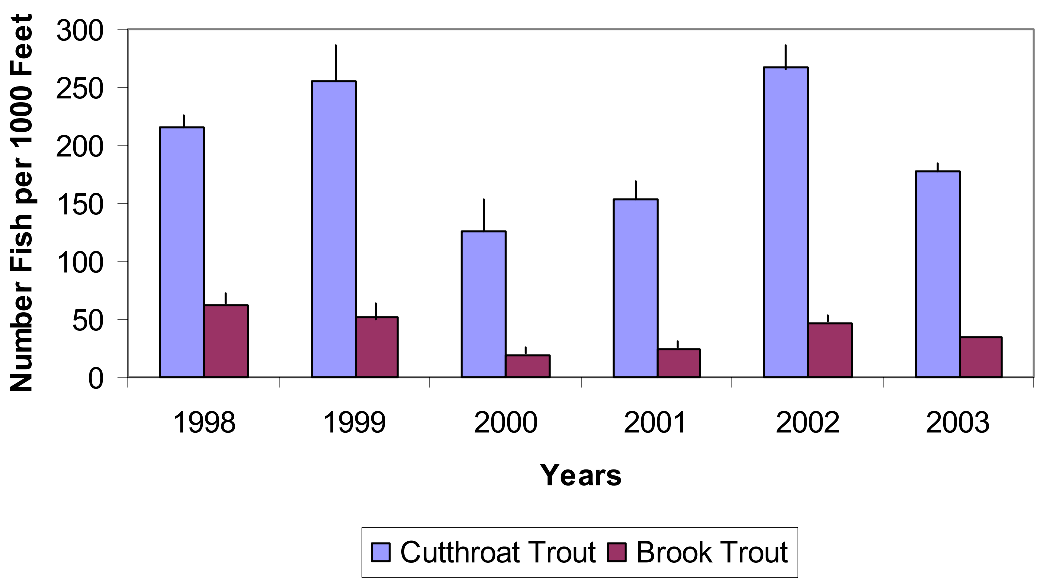

Figure 25. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Young Creek Section 5 monitoring site from 1997-2003 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

Libby Creek

Section 1 of Libby Creek has been sampled each consecutive year since 1998, and although the Libby Creek Demonstration Restoration Project was completed in the fall of 2001. Fish monitoring data collected from 1998 to 2001 represents the fish community prior to project implementation. Electrofishing conducted in 1999 and 2000 were limited to single pass catch estimates. Although mean rainbow trout densities at this site were higher for the two years following the restoration project implementation (100.5 fish per 1,000 feet) compared to the four years prior to implementation (69.5 fish per 1,000 feet), the differences were not significant ($p = 0.207$). Similarly, mean brook trout abundance at this site before and after project completion were slightly higher after project completion (8.8 and 10.5 fish per 1,000 feet, respectively; Figure 26), but the differences were not significant ($p = 0.350$). Juvenile bull trout were only observed in this section in 2002, with an estimated abundance of 3 fish per 1000 feet. There is no apparent temporal trend in rainbow trout ($r^2 = 0.08$; $p = 0.58$) or brook trout abundance ($r^2 = 0.07$; $p = 0.61$) within this section (Figure 27; Table A4).

Section 2 of Libby Creek was sampled in 1998, 2001 and 2003 (Table A4). Rainbow trout were substantially more abundant at this section than brook trout and bull trout during all years (Figure 28). We estimated 203, 148 and 100 rainbow trout per 1000 feet in 1998 through 2003, respectively. There was a significant negative trend in rainbow trout abundance through time at this site ($r^2 = 0.994$; $p = 0.048$). Bull trout were observed in this section in 1998 and 2003 (Figure 28; Table A4).

Our estimates of rainbow trout abundance in Section 3 of Libby Creek were similar between 2000 and 2002 (Figure 29; Table A4), with no evidence that the population differed from a stable population ($p = 0.469$; $r^2 = 0.548$) during this period, which represents conditions at this site prior to the upper Cleveland's Stream Restoration Project was completed. However, the rainbow trout estimate we conducted in 2003 was substantially lower than previous years (mean abundance 112.3 and 168.3 fish per 1,000 feet, respectively). We did not perform any statistical analyses to evaluate fish response before and after project completion due to lack of replication after completion. No brook trout were observed at this site. Estimates of juvenile bull trout abundance at this site ranged from 3 to 10.8 fish per 1000 feet over the four years (Figure 29).

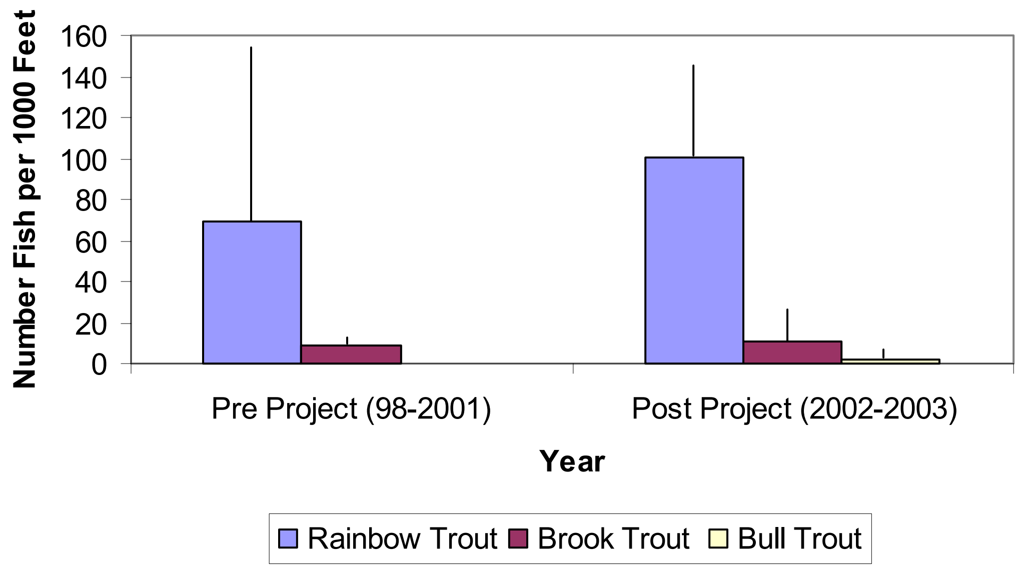

Figure 26. Rainbow trout and brook trout densities (fish per 1000 feet) within the Libby Creek Demonstration Project area, comparing annual mean pre-project (1998-2001) data and post-project (2002) using mobile electrofishing gear. Upper 95% confidence intervals are represented by the whisker bars.

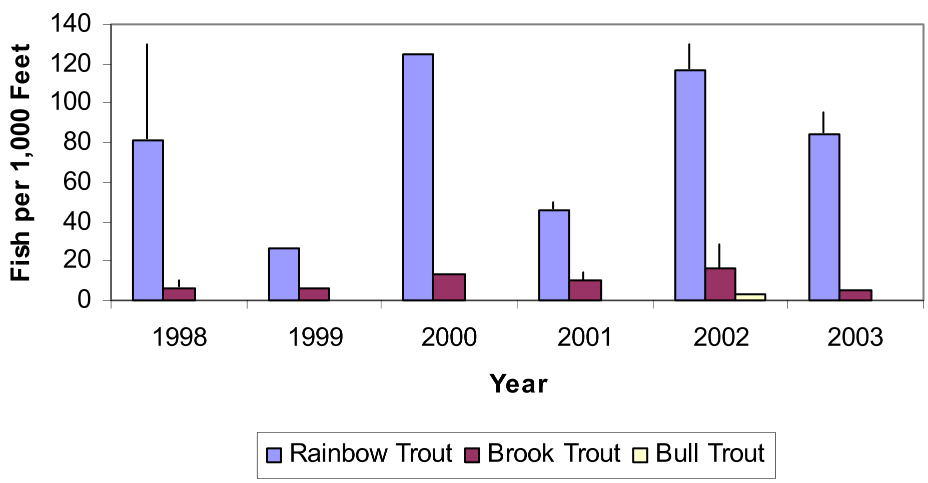

Figure 27. Rainbow trout, brook trout, and bull trout densities (fish per 1000 feet) within the Libby Creek Section 1 monitoring site 1998 through 2003 using a backpack electrofisher. Upper 95% confidence intervals are represented by the whisker bars. The site was sampled using single pass electrofishing in 1999 and 2000.

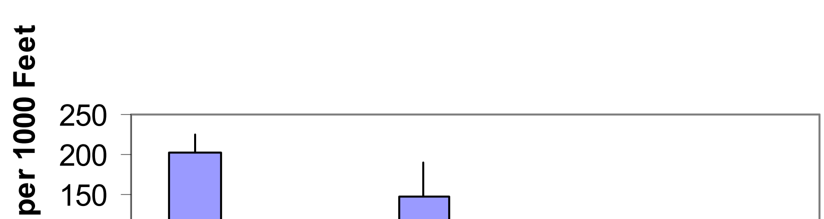

Figure 28. Rainbow trout, brook trout, and bull trout densities (fish per 1000 feet) within the Libby Creek Section 2 monitoring site sampled in 1998, 2001 and 2003 using a backpack electrofisher. Upper 95% confidence intervals are represented by the whisker bars.

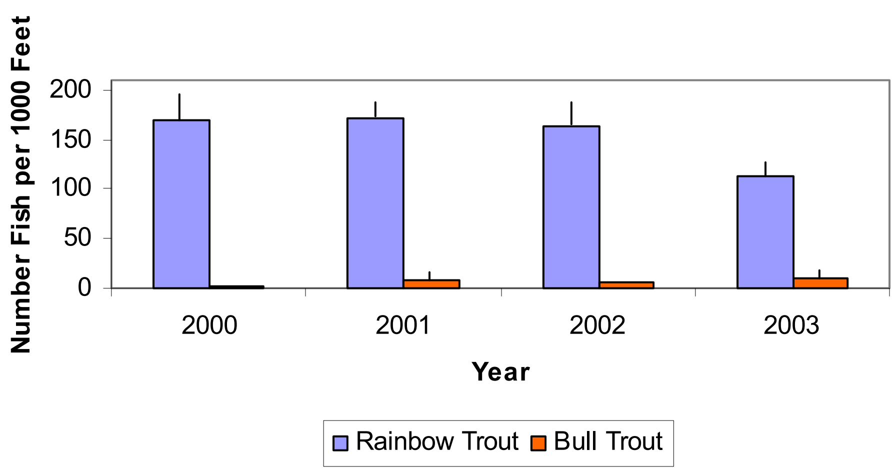

Figure 29. Rainbow trout and bull trout densities (fish per 1000 feet) within the Libby Creek Section 3 monitoring site in 2000-2003 using a backpack electrofisher. Upper 95% confidence intervals are represented by the whisker bars. This site is located within the upper Libby Creek restoration project area. The data from 2000-2002 represent pre-project trends of fish abundance, and the 2003 data represent data after project completion.

Parmenter Creek

Juvenile salmonid abundance estimates conducted in 2000 represent pre-project estimates, while the surveys conducted in 2001 and 2003 represent estimates after the restoration efforts on lower Parmenter Creek were completed. We did not survey Parmenter Creek in 2002. Rainbow trout were the most abundant fish species observed in Parmenter Creek during all years of sampling (2000, 2001 and 2003; Figure 30), with estimates of 92 and 79 fish per 1000 feet, respectively (Table A5). Statistical analysis was not performed to compare abundances before and after the stream channel restoration work due to lack of replication. However, the overlapping 95% confidence intervals for rainbow trout estimates between years suggests differences were not likely significant. Brook trout were more abundant in 2000 than 2001 at this site (Figure 30). We did not observe any juvenile bull trout at this site in 2000, but did observe bull trout at this location in 2001, with an estimated abundance of 1 fish per 1000 feet (Table A5).

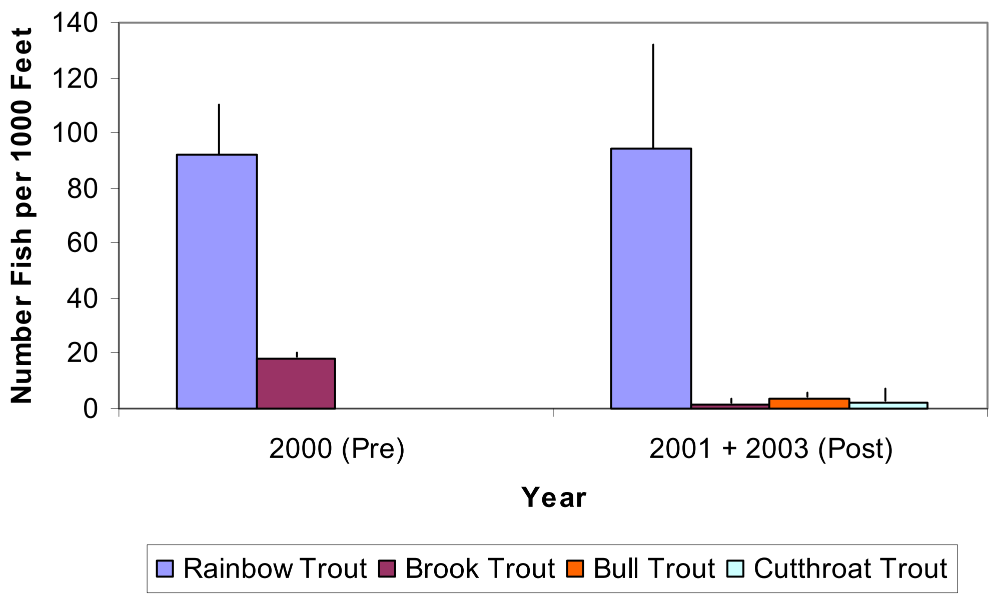

Figure 30. Cutthroat trout, bull trout and brook trout densities (fish per 1000 feet) within the Parmenter Creek monitoring site collected by performing backpack electrofishing. Fish abundance estimates from 2000 represent pre-project information, and surveys conducted in 2001 represent post-project data. Upper 95% confidence intervals are represented by the whisker bars. The site was not sampled in 2002.

Pipe Creek

Juvenile rainbow trout were more abundant at the lower Pipe Creek Section in 2002 than 2001 and 2003 (Table A6), with estimates of 73 fish per 1000 feet in 2002 and 46 and 43 fish per 1000 feet in 2001 and 2003, respectively (Figure 31). Brook trout abundance increased slightly through time, with 0, 3, and 6.5 brook trout per 1,000 feet estimated from 2001 through 2003. We did not capture any mountain whitefish in either 2001 or 2002, but did capture sufficient numbers in 2003 to produce a population estimate for this site of 8.7 mountain whitefish per 1,000 feet.

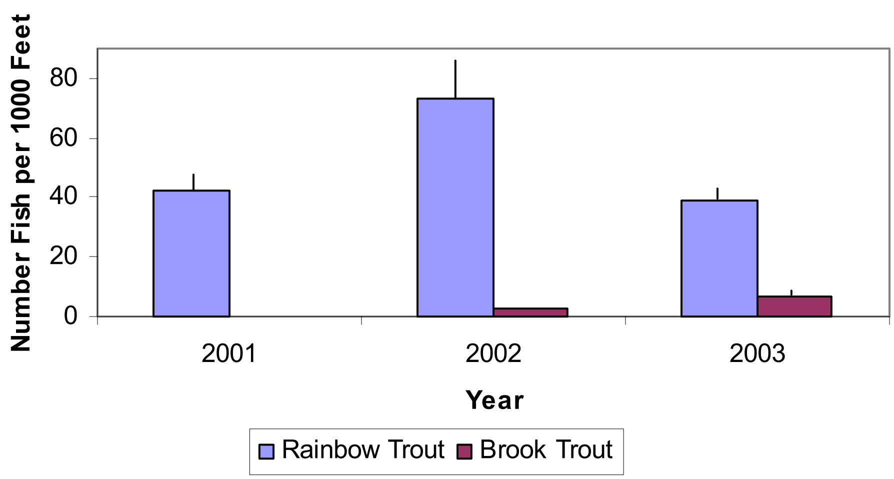

Figure 31. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Pipe Creek monitoring site during 2000, 2001 and 2003 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars. The site was not sampled in 2002.

Libby Reservoir Gillnet Monitoring

We documented changes in the assemblage of fish species sampled in Libby Reservoir since impoundment. Kokanee salmon, Kamloops rainbow trout and yellow perch did not occur in the Kootenai River prior to impoundment but are now present. Kokanee were released into the reservoir from the Kootenay Trout Hatchery in British Columbia (Huston et al. 1984). Yellow perch may have dispersed into the reservoir from Murphy Lake (Huston et al. 1984). The British Columbia Ministry of Environment (BCMOE) first introduced Kamloops rainbow trout in 1985. Eastern brook trout are not native to the Kootenai Drainage, but were present in the river before impoundment and continue to be rarely captured in gillnets within the reservoir. Peamouth and northern pikeminnow were rare in the Kootenai River before impoundment, but have increased in abundance in the reservoir. Mountain whitefish, rainbow trout, westslope cutthroat trout and redside shiner were common in the Kootenai River before impoundment, but have decreased in abundance since impoundment.

Kokanee

Since the accidental introduction of 250,000 fry from the Kootenay Trout Hatchery in British Columbia into Libby Reservoir in 1980, kokanee have become the second most abundant fish captured during fall gillnetting. Fluctuations in catch have corresponded to the strength of various year classes (Hoffman et al. 2002), and have varied by year, with no apparent continuous trend in abundance (Figure 32). However, kokanee catch in the fall net series follows a general trend of decreasing abundance from 1988-1995 and an increasing trend in abundance from 1996-2003 (Figure 32). Average length of kokanee has varied among years. Average length and weight between 1988 and 2003 was 290.3 mm and 235.2 g respectively (Table 6), while maximum average size occurred in 1992 (350 mm, 411 g). However, the minimum mean length was observed in 2002 (Table 6).

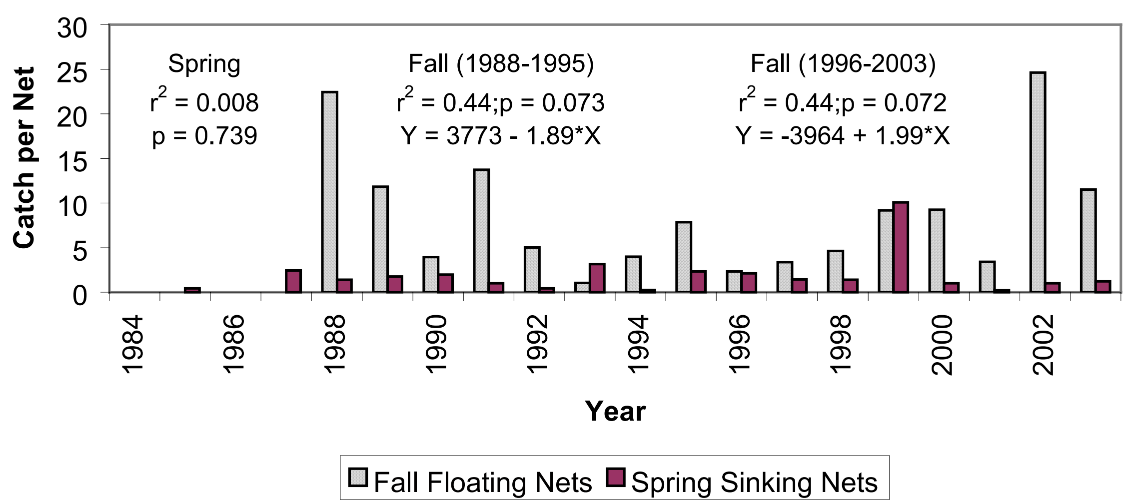

Figure 32. Average catch per net of kokanee for fall floating (1988-2003) and spring sinking (1984-2003) gill nets in Libby Reservoir.

Table 6. Average length and weight of kokanee salmon captured in fall floating gillnets (Tenmile and Rexford) in Libby Reservoir, 1988 through 2003.

YEAR	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	AVG.
Sample size (n)	2150	1259	517	624	250	111	291	380	132	88	76	200	342	120	357	263	447.5
Length (mm)	315.5	275	257.3	315.8	350	262.7	270.2	300.2	293.7	329.6	333.9	291.6	271.3	261.6	251.3	264.9	290.3
Weight (gm)	289.1	137.2	158.4	327.3	411.3	162.3	191.7	261.6	234.5	363.2	322.0	229.6	185.6	161.6	152.2	175.5	235.2

Mountain Whitefish

Mountain whitefish are one of three native species that have declined in abundance since impoundment of the Kootenai River (Huston et al. 1984, Figure 28). A natural logarithm transformation provided the best fit to the sinking gillnet catch data (Figure 33; $r^2 = 0.598$, $p < 0.05$). The trend in catch data for mountain whitefish during the first 13 years (1975-1988; mean catch = 3.5 fish per net) after reservoir impoundment decreased annually, until it reached a significantly ($p = 0.0003$) lower equilibrium with mountain whitefish catch rates since 1989 averaging 0.81 fish per net ($r^2 = 0.14$; $p = 0.20$). Catch rates since 1988 remained low; with mountain whitefish comprising an average of 1.1% of the spring catch during 1988 through 2003. We attribute the initial (1975-1988) mountain whitefish decline in Koocanusa Reservoir to the loss of spawning habitat and rearing habitat that resulted from a conversion of lotic to lentic habitat through reservoir construction.

Rainbow and Westslope Cutthroat Trout

Rainbow trout and westslope cutthroat trout catch have both significantly declined since the impoundment of Koocanusa Reservoir (Figure 33). Similarly to mountain whitefish gillnet catch data, rainbow and westslope cutthroat trout gillnet catch data was best with linear regression after performing a natural logarithm transformation (Figure 33). Although both species exhibit similar declining trends in catch since 1975, rainbow trout catch per net since 1975 has declined more precipitously than cutthroat trout catch per net. Rainbow trout have exhibited two general trends since impoundment. The first trend was the initial decline in abundance from 1975 to 1988, which is characterized by significant decline (Figure 33), followed by a period of relative stability from 1989 to 2003, where the average catch per net during this period (mean fish per net = 0.344) was not significantly different than a stable population (zero slope; Figure 33). Gill net catch of cutthroat trout in Koocanusa Reservoir exhibit a similar pattern, with the exception that that cutthroat trout catch rates exhibit 3 general trends through the same period. The first is a significant and precipitous decline during the early years of impoundment from 1975 to 1986 (Figure 33), where mean catch rates averaged 1.37 fish per net. The second general trend reduced abundance (0.38 fish per net), but at a level of stability from 1987 to 1993 ($r^2 = 0.337$; $p = 0.172$). The third general trend occurs from 1994 to 2003,and is characterized by a significantly lower level of abundance (0.136 fish per net; $p = 0.001$), and a somewhat stable level ($r^2 = 0.023$; $p = 0.674$). We believe that the period of general equilibrium during the period 1987-1993 may have been artificially elevated by the presence of hatchery cutthroat trout that were extensively stocked in the reservoir during this period (Table 7). Hatchery cutthroat trout were last stocked in the reservoir in 1994.

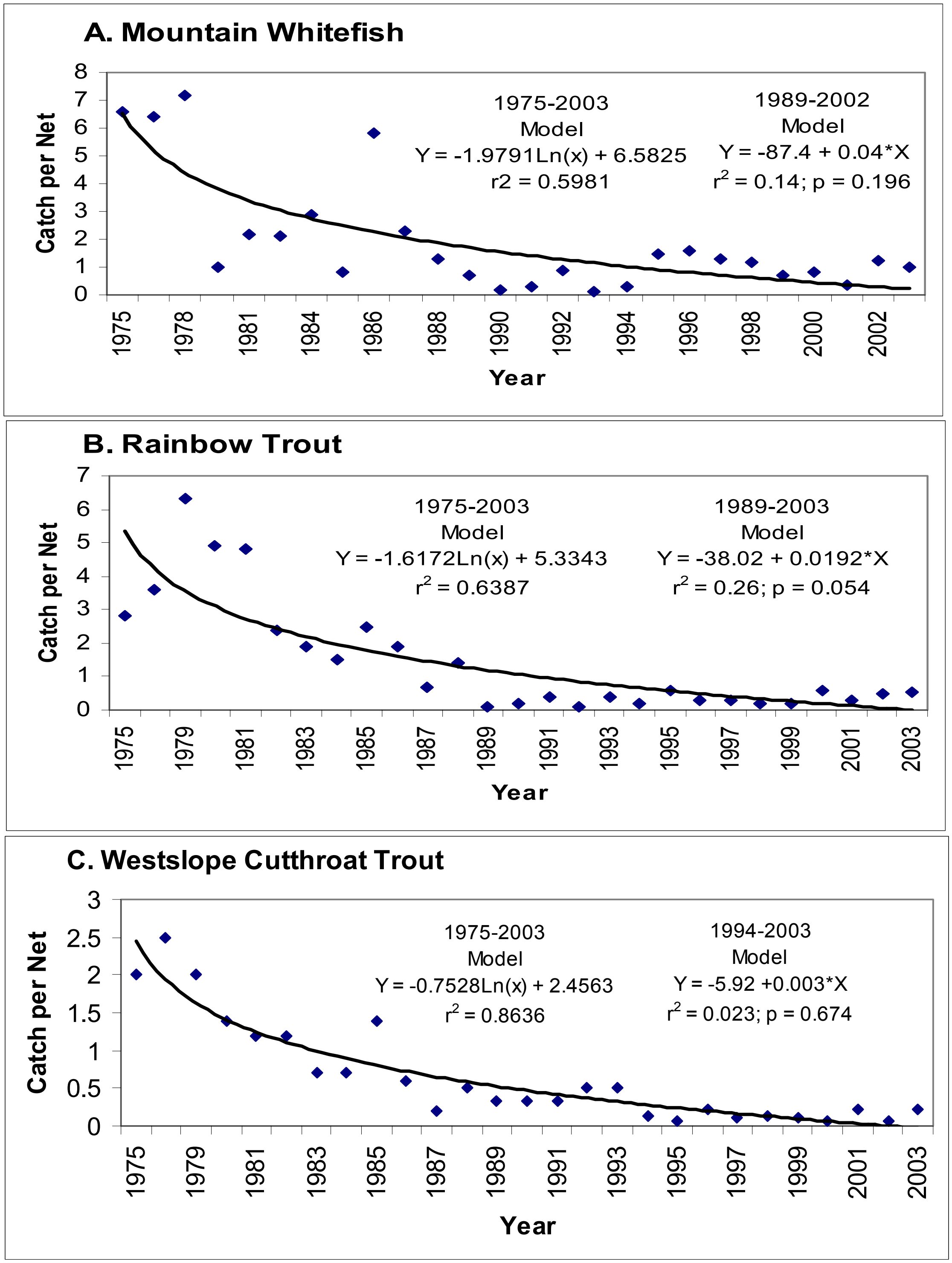

Figure 33. Mean catch rates (fish per net) of three native species (mountain whitefish (a) in spring sinking gillnets in the Rexford area, rainbow (b) and westslope cutthroat trout (c) in floating gillnets from Tenmile and Rexford areas in Libby Reservoir, 1975 through 2003. The Tenmile area was not sampled during the fall from 2001-2003.

Table 7. Average catch of westslope cutthroat trout per floating gill net caught in the Rexford and Tenmile areas during the fall, average length, average weight, number stocked directly into Libby Reservoir, and corresponding size of stocked fish between 1988 and 2003. The Tenmile location was not sampled in 200-2003.

	1988	1989	1990	1991	1992	1993	1994	1995	1996
No. Caught	0.50	0.32	0.32	0.32	0.50	0.50	0.14	0.07	0.21
Avg. Length (mm)	295	264	238	261	275	260	251	314	252
Avg. Weight (gm)	249	196	146	191	211	191	156	316	161
No. Stocked	none	5,779	40,376	67,387	72,376	72,367	1,360	none	none
Length (mm)	n/a		33	104	216	190	287	n/a	n/a

	1997	1998	1999	2000	2001	2002	2003
No. Caught	0.11	0.14	0.11	0.07	0.21	0.07	0.21
Avg. Length (mm)	225	267	305	302	259	305	270
Avg. Weight (gm)	128	228	296	271	175	256	206
No. Stocked	none	none	none	none	none	none	none
Length (mm)	n/a	n/a	n/a	n/a	n/a	n/a	N/a

Kamloops Rainbow Trout (Duncan Strain)

Kamloops rainbow trout were first introduced to Libby Reservoir in 1985 by BCMOE. The BCMOE continued stocking approximately 5,000 fingerling Kamloops (Gerrard strain) annually into Kikomun Creek (a tributary to the Kootenai River) from 1988-1998 (L. Siemens, BCMOE, personal communication). Montana FWP has stocked approximately 11,000 to 73,000 Duncan strain Kamloops rainbow trout since 1988 (Table 8). The catch of Kamloops rainbow trout in fall floating gillnets (fish per net) was significantly and positively correlated with the number of hatchery Kamloops rainbow trout stocked in the reservoir the previous year ($P=0.0003$; $r^2 = 0.66$; Table 8) for 1989 through 2003. However, the catch rate of Kamloops rainbow trout in fall floating gillnets shows no significant trend (Figure 34; $r^2 = 0.09$; $p = 0.247$). Catch rates for Kamloops rainbow trout in fall gillnets has been low since 1996, averaging only 0.07 fish per gillnet.

Table 8. Kamloops rainbow trout captured in fall floating gillnets in the Rexford and Tenmile areas of Libby Reservoir, 1988 through 2002. The Tenmile site was not sampled in 2001 or 2002.

	1988	1989	1990	1991	1992	1993	1994	1995
No. Caught	3	0	18	6	3	4	0	12
Avg. Length mm)	289	n/a	301	383	313	460	N/A	313
Avg. Weight (gm)	216	n/a	243	589	289	373	N/A	311
No. Stocked	20,546	73,386	36,983	15,004	12,918	10,831	16,364	15,844
Length (mm)	208-327	175-198	175-215	180-190	198-208	165-183	168-185	165-178
	1996	1997	1998	1999	2000	2001	2002	2003
No. Caught	2	1	2	3	3	0	0	5
Avg. Length (mm)	460	395	376	378	395	N/A	N/A	260.8
Avg. Weight (gm)	1192	518	450	504	555	N/A	N/A	159.2
No. Stocked	12,561	22,610	16,368	13,123	none	none	29,546	44,769
Length (mm)	170.5	152-178	127-152	255-280	N/A	N/A	80.3	81-206

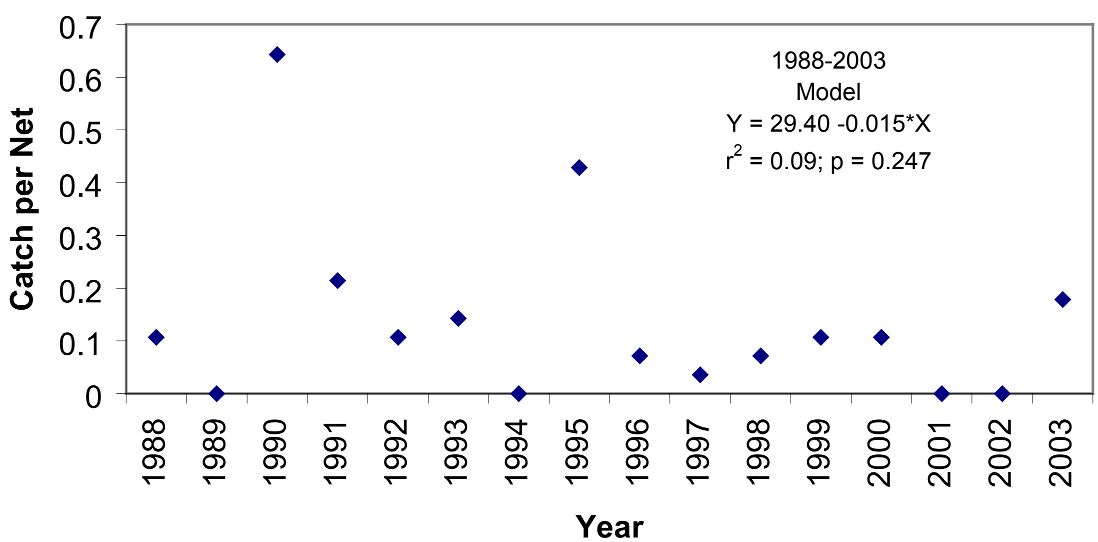

Figure 34. Average catch (fish per net) of Kamloops rainbow trout (Duncan strain) in fall floating gill nets in Libby Reservoir at the Rexford and Tenmile sites 1988-2003. The Tenmile site was not sampled in 2001-2003.

Bull Trout

Spring gill net catch of bull trout during the period 1975-1989 appeared to exist at an equilibrium with a slope (0.0091) that was not significantly different than zero ($r^2 = 0.011$; $p = 0.751$). However, beginning in approximately 1990, bull trout catch per net in Libby Reservoir began significantly increasing through 2003 (Figure 35; $r^2 = 0.769$; $p = 3.80*10^{-5}$). We attempted to account for differing reservoir levels during the gillnetting activities between years by multiplying the mean bull trout catch per net by reservoir volume at the time the nets were fished each year. This adjustment substantially improved the regression model's fit to the data (Figure 36; r2 = 0.777; $p = 3.07*10^{-5}$). Bull trout redd counts (see above) in both the Wigwam River and Grave Creek are both significantly and positively correlated to the spring gill net catch rates for bull trout adjusted for reservoir elevation ($r^2 = 0.690$; $p = 0.003$).

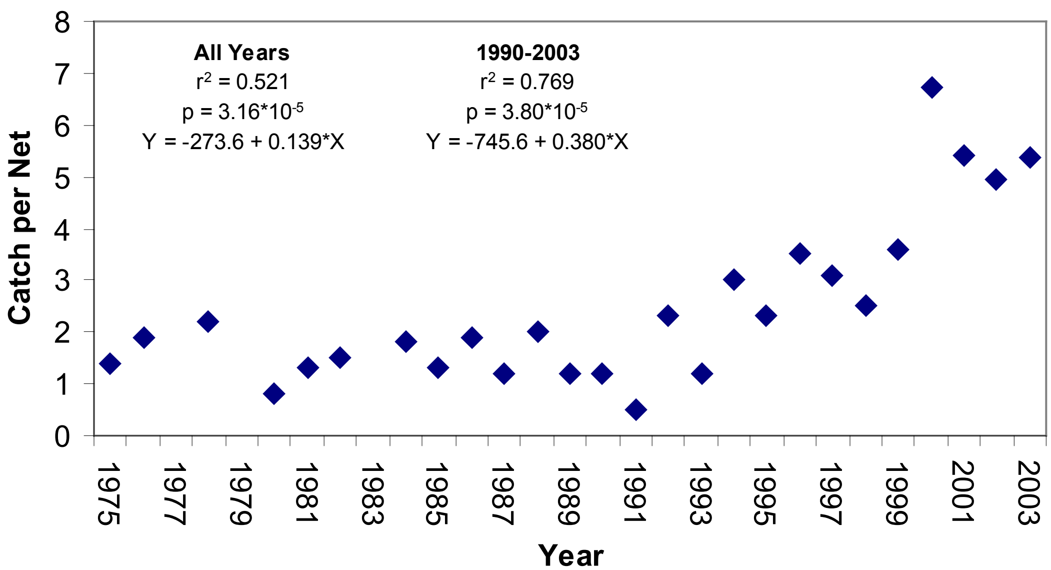

Figure 35. Average catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir 1975-2003.

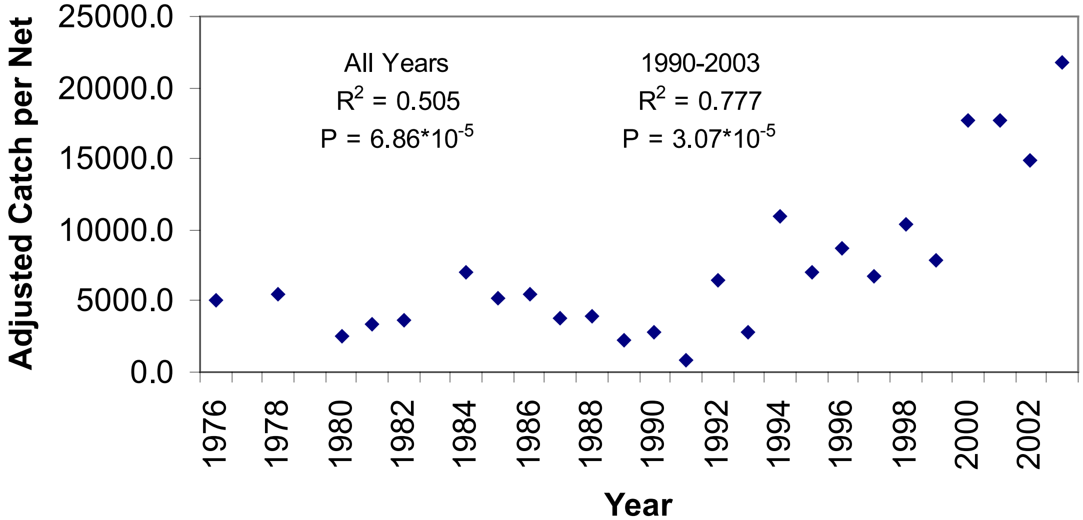

Figure 36. Average adjusted catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir. Average annual bull trout catch per net was adjusted by multiplying catch by reservoir volume at the time of gillnetting.

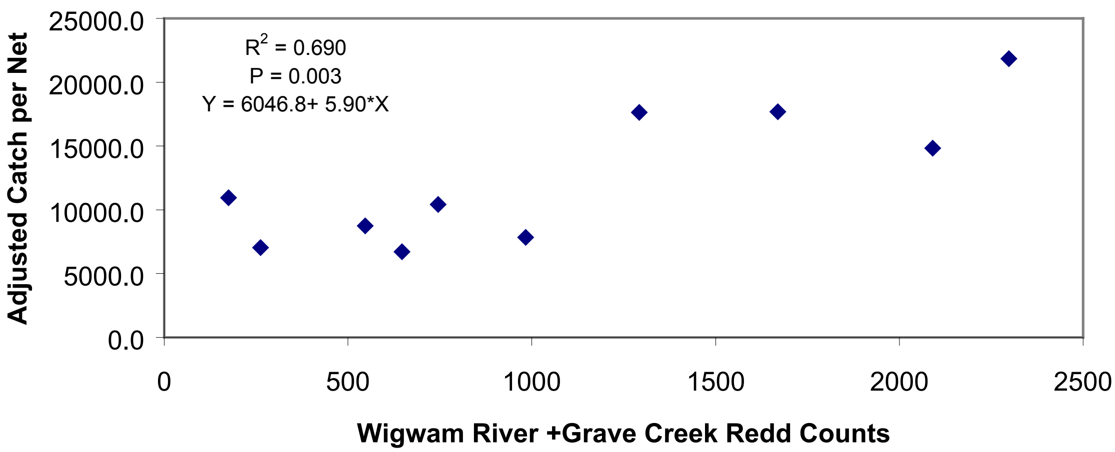

Figure 37. Average adjusted catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir related to total annual bull trout redd counts for the Wigwam River and Grave Creek during the period 1994-2003. Average annual bull trout catch per net was adjusted by multiplying catch by reservoir volume at the time of gillnetting.

Burbot

Burbot catch rates in spring sinking gillnets since 1990 show no clear trend in abundance (Figure 38; $r^2 = 0.07$; $p = 0.346$). Burbot catch per net for spring sinking nets has averaged 0.29 fish per net, and ranged from 0.07 to 0.5 fish per net. Burbot are not readily captured in floating gill nets. Burbot catch rates in spring gillnets is however significantly and positively correlated ($r^2 = 0.47$; $P = 0.04$; Figure 39) to daily catch of burbot in baited hoop traps in the stilling basin below Libby Dam (see above), suggesting that burbot abundance in Libby Reservoir may be influencing burbot abundance in the Kootenai River below Libby Dam through entrainment.

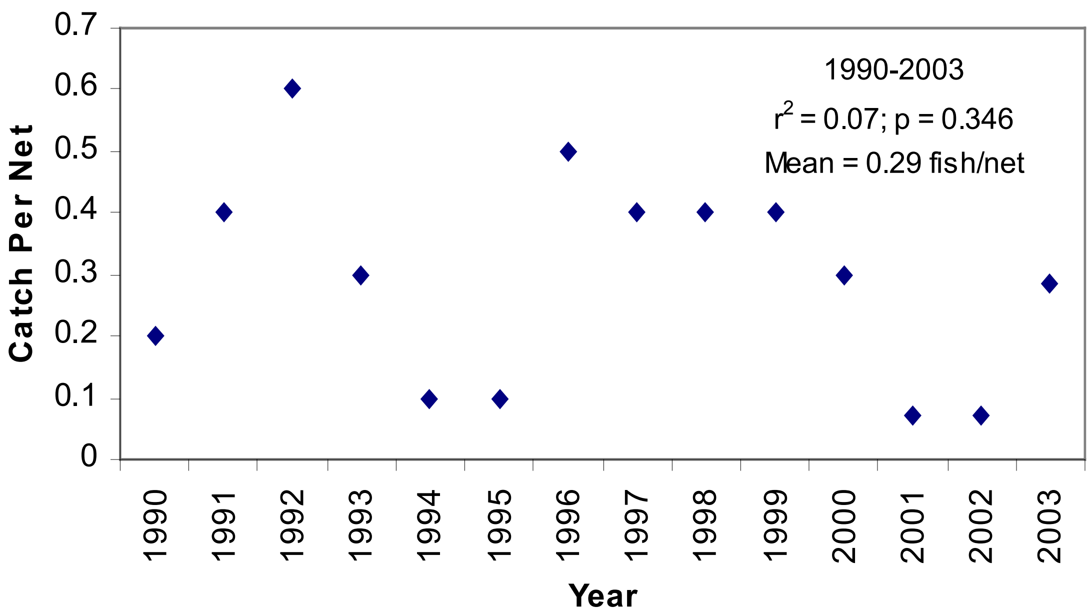

Figure 38. Mean catch per net of burbot in sinking gillnets during spring gillnetting activities at the Rexford site on Libby Reservoir, 1990-2003. The mean catch per net during the period was 0.29 fish per net.

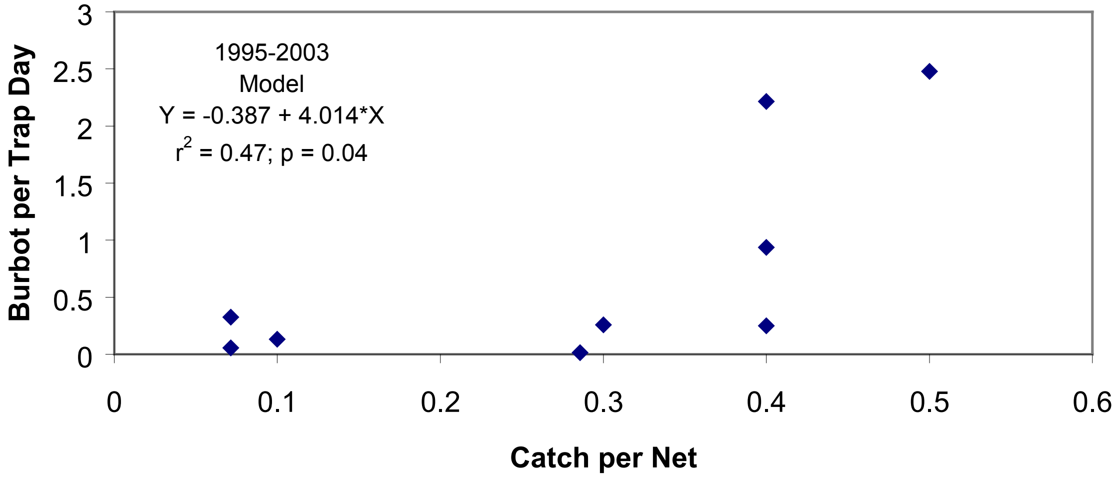

Figure 39. The relationship between mean burbot catch per net for spring sinking gillnets on Libby Reservoir and burbot catch rates (fish/trap day) of baited hoop traps in the stilling basin below Libby Dam 1995-2003.

Total Fish Abundance

The long-term trends in total fish abundance in the reservoir reflect the changes that have occurred in the reservoir since impoundment. Total catch (fish per net) for spring gillnets has significantly increased since impoundment (Figure 40; $r^2 = 0.104$; $p = 0.09$; Table 9) is indicative of an increase in the biomass of species that prefer reservoir habitats: peamouth chub, suckers, northern pikeminnow, etc. However, there is no significant trend in total catch (fish per net) for fall gillnets (Figure 40; $r^2 = 0.003$; $p = 0.76$; Table 10). Species composition for the catch of fall and spring gillnets has remained relatively stable since 1988 (Table 11).

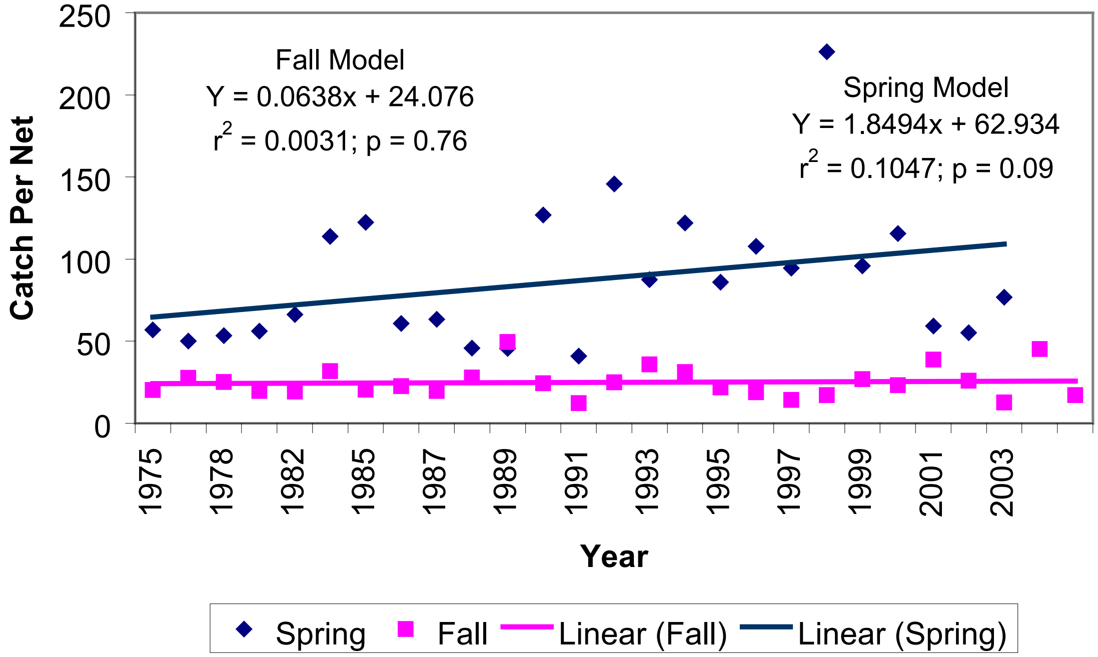

Figure 40. Catch per net (all species combined) in fall floating and spring sinking gillnets and associated trend lines in Libby Reservoir, 1975 through 2003.

Table 9. Average catch per net for nine different fish species* captured in floating gillnets set during the fall in the Tenmile and Rexford areas of Libby Reservoir, 1990 through 2003.

	YEAR													
	1990	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**
Surface Temperature	16	15	13.8	13.8	16.6	15.8	15.5	17.2				19		
Date	9/25	10/2	9/25	10/5	9/27	10/10	9/23	9/22	9/21	9/14	9/12	9/20	9/10	9/16
Number of Floating Nets	54	28	28	28	28	28	28	28	28	28	28	14	14	14
Reservoir Elevation	2456	2448	2421	2441	2446	2454	2450	2448	2439	2453	2434	2433	2441	2435
	Average number of fish caught per net for individual fish species													
RBT	0.2	0.4	0.1	0.4	0.2	0.6	0.3	0.3	0.2	0.2	0.6	0.3	0.5	0.5
WCT	0.2	0.4	0.5	0.9	0.1	0.1	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1
RB X WCT∠	0.3	0.2	0.2	0	0	0	0	0	<0.1	0	0	0	0	<0.1
SUB-TOTAL	0.7	1	0.8	1.3	0.3	0.7	0.5	0.4	0.3	0.3	0.7	0.4	0.6	0.7
MWF	0.2	0.5	0.2	0.3	0.4	0.3	0.3	0.5	0.4	0.1	0.1	0.2	0.4	0.4
CRC	18.2	18.4	23.3	17.1	10.4	1.2	11.7	17.8	14.4	24.3	12.9	5.6	21.4	5.0
NPM	1.8	2.1	1.8	2.2	3.4	2.7	1.8	4.0	4.9	6.4	3.9	3.9	8.1	3.36
RSS	0	0.1	0	0	0.3	0.2	0.1	1.0	0.3	0.3	<0.1	0	0.3	<0.1
BT	0	0	0.1	0.3	0	1.2	<0.1	0	<0.1	<0.1	0.2	0	0.1	0
CSU	0.1	0.1	0	0.1	0.1	0	0.4	0.1	0.1	0.1	0.1	0.3	0.1	0.2
KOK	3.9	13.7	5	1	4	7.9	2.3	3.1	2.7	7.3	8.0	2.1	14.2	7.4
TOTAL	24.9	35.9	31.2	22.3	18.9	14.2	17.1	26.9	23.1	38.8	25.9	12.5	45.1	17.1

*Species Codes (RBT = rainbow trout, WCT = westslope cutthroat trout, RBXWCT = rainbow and cutthroat trout hybrid, MWF = mountain whitefish, CRC = Columbia River chub, NPM = northern pikeminnow, RSS = redside shiner, BT = bull trout, CSU = coarse scale sucker, and KOK = kokanee.

Table 10. Average catch per net for 12 different fish species* captured in sinking gillnets set during spring in the Rexford area of Libby Reservoir, 1990 through 2003.

	YEAR													
	1990	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**
Surface Temperature	11.7	9.8	16.7	14.4	13.3	13.5	8.9							
Date	5/10	5/16	5/5	5/17	5/16	5/8	5/12	5/12	5/11	5/17	5/14	5/15	5/13	5/13
Number of Sinking Nets	27	28	28	28	28	28	28	28	27	28	14	14	14	14
Reservoir Elevation	2358	2330	2333	2352	2405	2386	2365	2350	2417	2352	2371	2392	2384	2417
	Average number of fish caught per net for individual fish species													
RBT	0.1	0.1	0.1	0.3	0.2	0.2	0.7	0.1	<0.1	1.1	0.3	0.2	0.4	0.7
WCT	<0.1	0.0	0.1	0.0	<0.1	0.1	0.1	0.2	0.0	0.3	0.1	0	0	0.2
RB x WCT	0.0	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0	0.2	0.0
SUB-TOTAL	0.1	0.2	0.2	0.3	0.2	0.3	0.9	0.3	0.0	1.4	0.4	0.2	0.6	0.9
MWF	0.2	0.3	0.9	0.1	0.3	1.5	1.6	1.3	1.2	0.7	0.8	0.4	1.2	1.2
CRC	104.8	31	119	63.3	94.2	54.1	60.9	51.1	171.7	54.4	76.4	25	24.1	42.1
NPM	6.0	2.0	4.2	3.8	7.6	8.0	10.0	13.1	15.1	14	12.6	11	9.9	13.0
RSS	<0.1	0.0	0.5	0.0	0.0	0.0	0.0	0.1	1.0	0.1	0.4	0	0	0.1
BT	1.2	0.5	2.3	1.2	3.0	2.3	3.5	3.1	2.5	3.6	6.7	5.4	4.9	5.4
LING	0.2	0.4	0.6	0.3	0.1	0.1	0.5	0.4	0.4	0.4	0.3	0.1	0.1	0.3
CSU	5.8	2.4	12.9	9.8	9.0	12.0	19.9	14.3	21.1	8.3	10.6	14.2	9.9	10.2
FSU	1.8	1.1	2.9	4.1	6.5	3.0	4.8	4.7	9.5	5.9	5.1	1.1	2.9	2.3
YP	4.7	2.1	1.8	1.1	0.7	2.5	3.7	4.75	2.4	1.8	1.3	1.6	0.6	0.1
KOK	2.0	1.0	0.4	3.5	0.3	2.1	2.0	1.4	1.3	5.3	1.0	0.2	1.0	1.2
TOTAL	120.7	40.0	145.3	84.3	121.9	86.3	107.1	93.25	226.2	95.9	115.1	59.2	55.2	76.8

*Species Codes (RBT = rainbow trout, WCT = westslope cutthroat trout, RBXWCT = rainbow and cutthroat trout hybrid, MWF = mountain whitefish, CRC = Columbia River chub, NPM = northern pikeminnow, RSS = redside shiner, BT = bull trout, LING = burbot, CSU = coarse scale sucker, FSU = fine scale sucker, YP = yellow perch, and KOK = kokanee.

Table 11. Percent composition of major fish species* caught in fall floating and spring sinking gillnets in Libby Reservoir, 1988 through 2003. Blank entries in table indicate either no fish were captured or that they occurred in very small proportions.

	1988		1989		1990		1991		1992		1993		1994		1995		1996	
	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.
RB	3.0		0.1		0.7		1.0		0.3		1.8		0.9		4.4		1.4	
WCT	0.5		0.3		0.7		1.0		1.7		3.8		0.7		0.8		1.2	
HB	1.0		0.3		1.1		0.5		0.7		0.2		0.0		0.3		0.2	
ONC	4.5	0.7	0.7	0.4	2.4	0.1	2.4	0.4	2.7	0.1	5.8	0.3	1.7	0.2	5.5	0.4	2.8	1.0
MWF	0.5	1.6	0.2	0.8	0.9	0.2	1.4	0.7	0.7	0.6	1.4	0.2	2.2	0.3	2.1	1.7	1.4	1.5
CRC	39.4	63.8	70.5	66.0	71.4	82.6	50.0	76.5	72.6	81.7	72.8	73.9	54.3	77.0	8.6	62.9	66.5	56.9
NPM	2.9	7.7	4.1	7.4	7.2	4.8	5.8	5.0	5.6	2.9	9.3	5.0	17.5	6.2	19.6	9.3	10.2	8.7
RSS	0.8	0.2	0.2	0.1	0.0	0.0	0.3	0.0	0.0	0.3	0.0	0.0	1.5	0.0	1.3	0.0	0.6	0.0
FSU	0.0	2.3	0.0	1.6	0.0	1.5	0.0	2.6	0.1	2.0	0.0	5.2	0.0	5.3	0.0	3.5	0.0	4.4
CSU	0.0	12.7	0.2	10.3	0.2	4.5	0.3	5.9	0.0	8.8	0.6	9.7	0.6	7.3	0.0	13.9	2.4	18.6
KOK	47.3	1.7	23.4	2.1	15.5	1.5	37.3	1.6	15.7	0.3	4.4	3.4	20.6	0.2%	57.4	2.4	13.2	1.8
YP		5.5		9.4		3.7		5.2		1.2		1.1		0.9		2.9		3.4
BT		2.4		1.4		1.0		1.1		1.7		1.1		2.5		2.8		3.3

	1997		1998		1999		2000		2001		2002		2003		Average	
	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.
RB	1.7	0.2	1.5	0.1	0.6	0.9	1.1	0.2	1.4	0.4	0.4	0.4	2.8	0.9	1.3	0.3
WCT	0.6	0.4	0.5	0.1	0.3	0.2	0.8	0.1	1.7	0	0.1	0	0.8	0.3	0.9	0.1
HB	0	0	2.3	0	0	0	0	0	0	0	0	0	0.4	0.0	0.4	0.0
ONC	2.3	0	4.2	0.4	0.9	1.3	1.9	0.3	3.1	0.4	0.5	0.4	4.0	1.2	2.6	0.4
MWF	2.4	1.9	1.2	2.5	0.6	1.1	0.5	0.7	2.5	0.6	0.3	1.5	2.0	1.6	1.1	1.1
CRC	56.0	33.8	50.2	33.0	44.6	38.3	46.4	66.0	49.3	42.2	41.5	62.4	27.7	54.9	49.6	54.1
NPM	18.0	20.0	21.1	17.6	22.5	20.8	18.1	10.8	22.5	18.6	14.4	11.8	18.6	16.9	12.4	10.5
RSS	3.5	0.2	0.8	1.4	0.7	0.1	0.1	0.4	1.4	0	0.9	0	0.4	0.1	0.8	0.2
FSU	0	7.2	0.3	12.1	0.1	8.7	0.1	4.0	0	1.9	0	3.4	0.4	3.0	0.0	4.0
CSU	3.38	20.8	4.6	24.1	3.3	13.7	4.0	9.1	3.4	24.0	0.6	12.3	1.2	13.3	1.5	12.7
KOK	14.4	2.2	17.3	1.8	27.1	8.1	28.6	0.9	17.5	0.4	41.6	1.2	41.1	1.6	23.9	1.8
YP	0	7.4	0	3.2	0.1	2.8	0.3	1.1	0	2.7	0.1	0.8	5.1	0.1	0.1	3.3
BT	0.1	5.1	0.3	3.3	0.1	2.6	0	5.8	0.3	9.2	0	5.9	0	7.0	0.1	3.3

*Species Codes = RB = Rainbow trout, WCT = westslope cutthroat trout, HB = hybrid rainbow trout X cutthroat trout, ONC= Combined Rainbow, westslope cutthroat and hybrid trout, MWF = mountain whitefish, CRC = Columbia River chub (peamouth), NPM = northern pikeminnow, RSS = red side shiner, FSU = fine scale sucker, CSU = course scale sucker, KOK = kokanee, YP = yellow perch, BT = bull trout.

Libby Reservoir Zooplankton Monitoring

Zooplankton species composition and abundance within Libby Reservoir has remained relatively stable during the past several years (Appendix Tables A7-A13). Since 1997, *Cyclops* and *Daphnia* have been the first and second most abundant genera of zooplankton present in the reservoir (Figure 41). Other lesser abundant genera in decreasing order of abundance include *Diaptomus, Bosmina, Diaphanosoma, Epichura and Leptodora* (Figure 41). Zooplankton abundance within the reservoir varies by season (Table 12; Figure 42). The results from 7 analysis of variance procedures that tested for differences in monthly zooplankton abundance (by species) indicated that at least one month was significantly different from other months in 2003 for the most abundant 7 species of zooplankton (Table 12). We did not perform multiple comparisons required to determine pairwise comparisons. Although zooplankton abundance varies within a season, seasonal peaks in abundance over the past six years (Figure 42) have remained relatively consistent across years. For example, *Daphnia* abundance has peaked during July each year except 2003 (June peak) since 1997, *Diaphanosoma* abundance has peaked in September during 6 of the last 7 years, *Diaptomus* has peaked during October during 4 of the last 7 years, and Cyclops has peaked in June during 4 of the last 7 years. In most cases when the annual peak differed from the mean peak, the difference was not more than several weeks.

Our sampling design stratified the reservoir into thirds, and although each stratum was long (> 58 km), we found only weak evidence that zooplankton abundance differed between the three sampling areas (Tenmile, Rexford, and Canada) in 2003 (Table 12). For the 7 most abundant species of zooplankton in the reservoir at the three sites, we only found significant differences (by species) for 8 out of the possible 21 comparisons. When significant differences did occur between sampling location, there was no clear trend in terms of whether zooplankton abundance was always highest for most downstream site. During 2003, abundance estimates of *Daphnia, Diaptomas*, *Cyclops*, *Leptodora*, and *Epischura* differed between at least one of possible three comparisons between sampling areas. Subsequent multiple comparisons indicated that *Daphnia* densities were significantly higher at the Rexford and Canada sites than the Tenmile site and *Diaptomas* densities were significantly higher at the Rexford site than the Tenmile site, *Cyclops* abundance at Rexford was significantly higher than the Canada and Tenmile sites, *Leptodora* abundance was significantly higher at the Canada site than Tenmile and Rexford sites, and *Epischura* densities were significantly higher at the Rexford site than the Canada site. The month and area interaction term was significant for *Bosmina*, *Cyclops*, and *Leptodora* in 2003 (Table 12).

Although the abundance of zooplankton of the genus *Daphnia* in 2003 was the highest during the previous 8 years, the trend has remained particularly stable in terms of abundance (Figure 41) and size (Figures 43 and 44) during the past several years. Mean annual *Daphnia* densities in Libby Reservoir from 1997 through 2003 have averaged 1.94 *Daphnia* /liter (standard deviation = 0.59/liter; Figure 43). Mean *Daphnia* length has also varied relatively little since 1991, averaging 0.90 mm (standard deviation = 0.05; Figure 44). Most *Daphnia* since 1993 are between 0.5 – 1.5 mm, with majority of *Daphnia* being represented in the smaller size class 0.5 – 0.99 mm (mean annual proportion = 0.61, standard deviation = 0.052; Figure 43), with the majority of the remainder in the size class 1.0 – 1.499 (mean annual proportion = 0.336, and standard deviation = 0.036). *Daphnia* larger than 1.5 mm have on average comprised less than 5% of the total since 1993 (Figure 43).

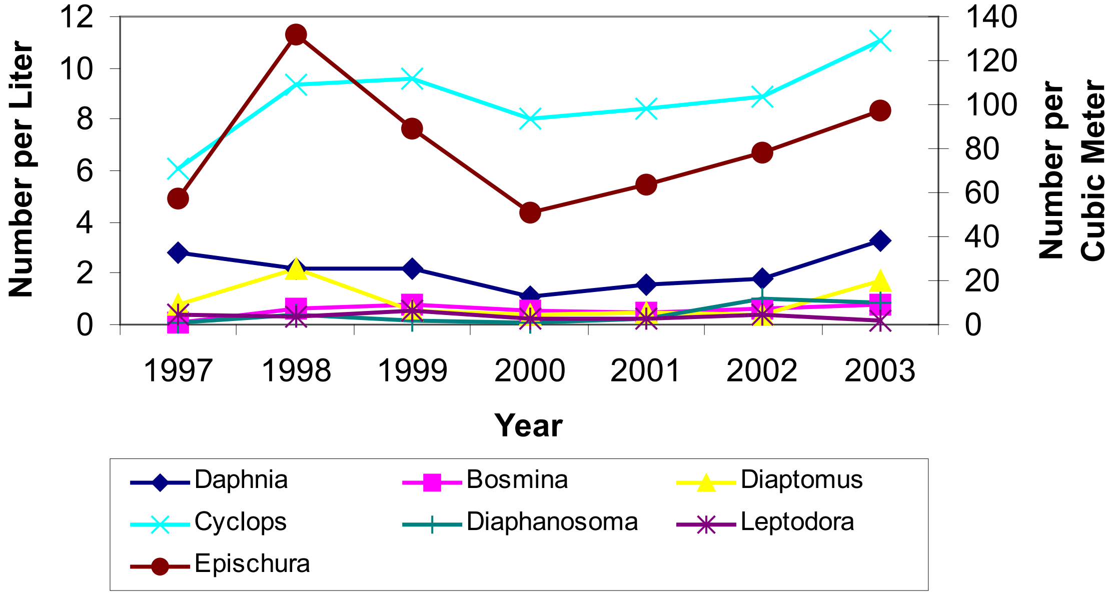

Figure 41. Annual zooplankton abundance estimates for 7 genera observed in Libby Reservoir from 1997-2003. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter. The data utilized for this figure are presented in Appendix Table A13.

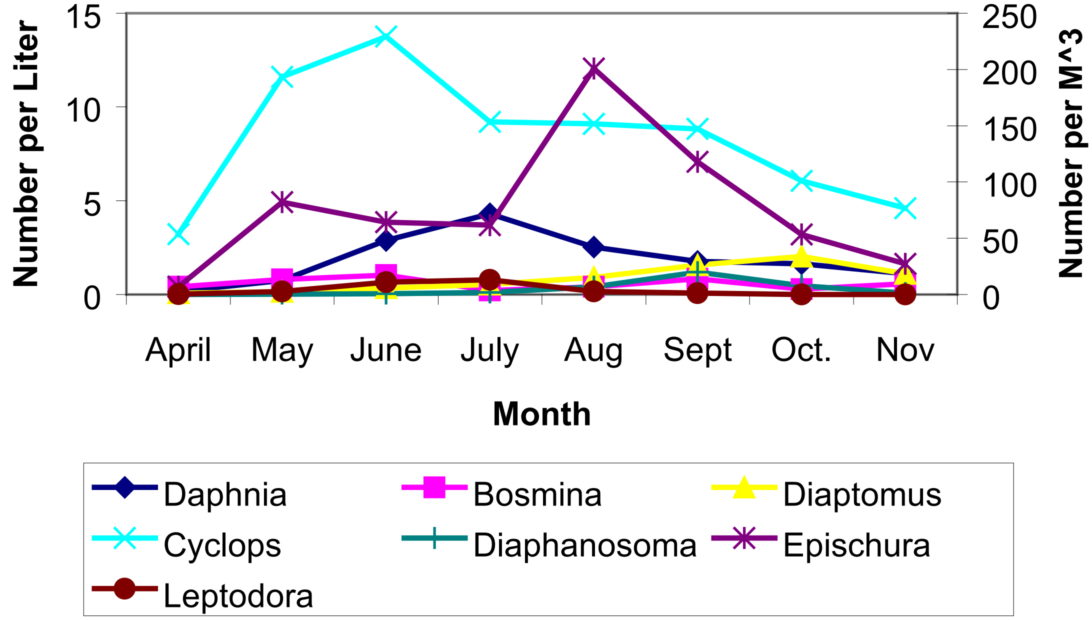

Figure 42. Mean monthly zooplankton abundance estimates for 7 genera observed in Libby Reservoir from 1997-2003. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter.

Table 12. Individual probability values (p values) resulting from analysis of variance procedures that tested for differences in zooplankton densities by month (April – November), area (Tenmile, Rexford and Canada) and a month by area interaction in 2003.			
Genus	**Month**	**Area**	**Month X Area Interaction**
Daphnia	$4.01*10^{-7}$	0.0093	0.1052
Bosmina	$3.25*10^{-6}$	0.2553	$1.56*10^{-10}$
Diaptomas	$1.02*10^{-7}$	0.0398	0.7331
Cyclops	0.0333	0.0050	0.0500
Leptodora	$3.24*10^{-10}$	0.0003	0.0082
Epischura	$1.56*10^{-5}$	0.0118	0.2073
Diaphanosoma	$1.16*10^{-6}$	0.2447	0.2555

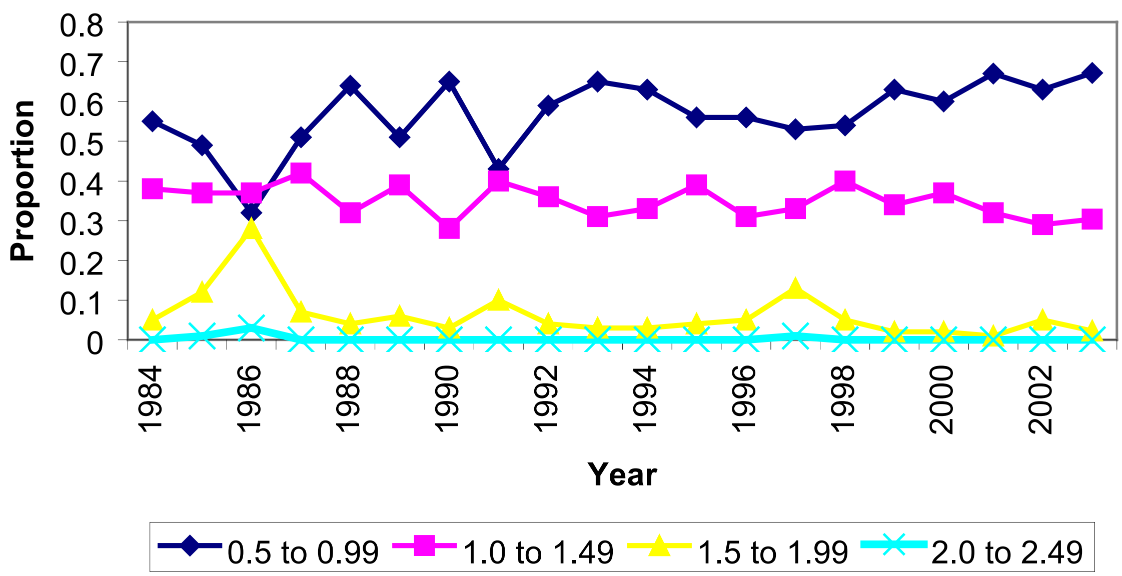

Figure 43. *Daphnia* species size composition in Libby Reservoir, 1984 through 2003.

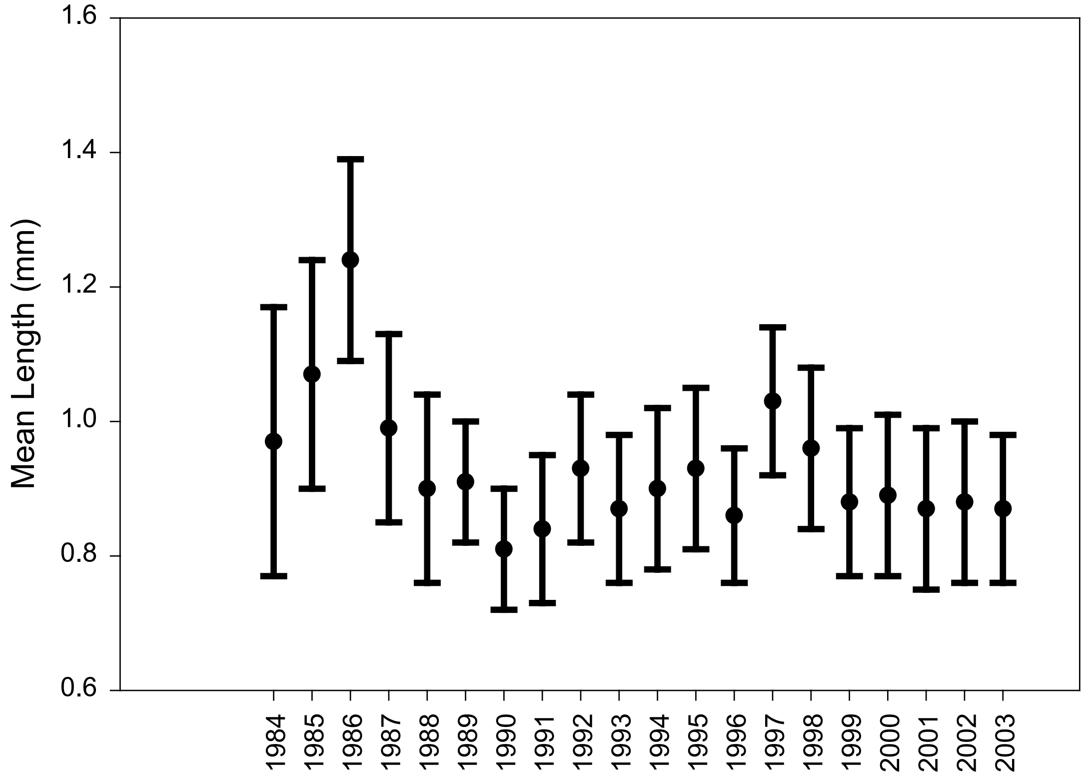

Figure 44. Mean length of *Daphnia* species in Libby Reservoir, 1984 through 2003, with 95% confidence intervals.

Discussion

Long-term monitoring of bull trout redd numbers can be an important tool to assess bull trout population trends (Rieman & McIntyre 1993). Bull trout redd counts in the tributaries that we monitor below Libby Dam have not increased in proportion to the increases we have observed in redd counts in bull trout spawning tributaries located upstream of Libby Dam over the past 9 years. We are however, confident that we have identified the important core spawning tributaries below Libby Dam within the Montana portion of the basin. Although drought conditions in 2001 through 2003 likely exacerbated the effects of debris jams in some of these streams by forming impassible and substantial barriers in some of the tributaries may have reduced redd counts, the bull trout redd count information does not correlate well with our adult bull trout population estimate conducted in the spring of 2004 below Libby Dam. We observed a total of 126 bull trout redds in Quartz, Pipe, O'Brien, and Bear Creeks in the fall of 2003. Baxter and Westover (2000) estimated an average of 1.55 fish per redd (range = 1.2-2.1 fish per redd) for spawning bull trout in the Wigwam River in 1996-1999. If we apply these ratios to bull trout redd counts in the 4 spawning tributaries below Libby Dam 2003 (Table 1; 126 total redds), there may have been 195 (range 150 – 265) spawning bull trout in the Kootenai River below Libby Dam in 2003. We estimated a total of 920 bull trout (95% confidence interval 698 – 1142). We believe most of these fish were likely adult fish. Baxter and Baxter (2002) operated a fish trap on Skookumchuck Creek in 2001 and estimated the mean size of spawning adults to be 640 mm (range 400-920 mm), which was similar to the size of fish we found below Libby Dam during our spring 2004 population estimate (mean total length 649 mm; range 343 – 861 mm). Although we acknowledge that the 2003 bull trout redd counts within the tributaries below Libby Dam may have underestimated of the total Kootenai River bull trout population because they did not take into account alternate year spawning individuals, the disparity between our adult population estimate and the redd count data is nevertheless large. One possible explanation for these large differences may bull trout entrainment through Libby Dam from within Libby Reservoir. The bull trout redd counts, juvenile estimates and adult bull trout estimates collected by Montana FWP provide critical information required to assess the status and trends of bull trout within the Kootenai River Basin. This information will be essential to determine whether recovery criteria (USFWS 2002) are met within this basin. Therefore, collection of these data will remain a high priority for long-term monitoring conducted by Montana FWP. This project will also investigate the use of other methodologies, such as scale and otolith micro-chemistry (Wells et al. 2000; Wells et al. 2003; Kennedy et al. 2000) to determine the origin of adult bull trout below Libby Dam.

Catch rates of burbot in our baited hoopnets in the Kootenai River directly below Libby Dam have precipitously and significantly decreased in recent years. Our burbot catch rates using baited hoop traps in the stilling basin below Libby Dam were the lowest since we began monitoring this site in 1994/1995, and averaged only 0.0139 burbot per trap day, or one burbot every 72 trap-days during the 2003/2004 trapping season. In comparison, our catch rates for burbot using similar gear and techniques in Libby Reservoir during the same period were approximately 5 times higher. However, since this was the first year of trapping burbot in Libby Reservoir, using techniques similar to those we use in the stilling basin, we are not able to determine if burbot abundance in the reservoir has declined in recent years. We found no evidence of a significant trend of burbot abundance in Libby Reservoir from our spring gill netting data since 1990. However, gillnets might not provide an accurate indication of burbot population trends due to seasonal differences in movement and activity, and variable catch rates. Some investigators suggest that baited hoopnets are a more efficient capture method (Jensen 1986; Bernard et al. 1991). Paragamian and Hoyle (2003) found that burbot abundance in the lower Kootenai River in Idaho and British Columbia has also declined since 1995. However, by 1995, the abundance of burbot in the lower Kootenai River was substantially reduced from historic levels (Paragamian and Hoyle 2003). Paragamian and Hoyle (2003) reported burbot catch rates using baited hoop traps in the lower Kootenai River ranging from 0.055 burbot per trap day in 1995/1996 to a low of 0.006 burbot per trap day in 2002/2003. The decline of burbot abundance in the stilling basin below Libby Dam correlates to burbot catch rates that Paragamian and Hoyle (2003) reported in the lower Kootenai during the same time period (Figure 45; $r^2 = 0.615$; $p < 0.05$). Paragamian et al. (1999) concluded that two genetically dissimilar burbot populations existed in the Montana and Idaho/British Columbia portions of the Kootenai Basin, which suggests that the correlation we observed between our catch rates in the stilling basin and Paragamian and Hoyle's catch rates in the lower Kootenai was not likely due to fish migration, but rather possible similar environmental conditions that may influence population dynamics of both stocks.

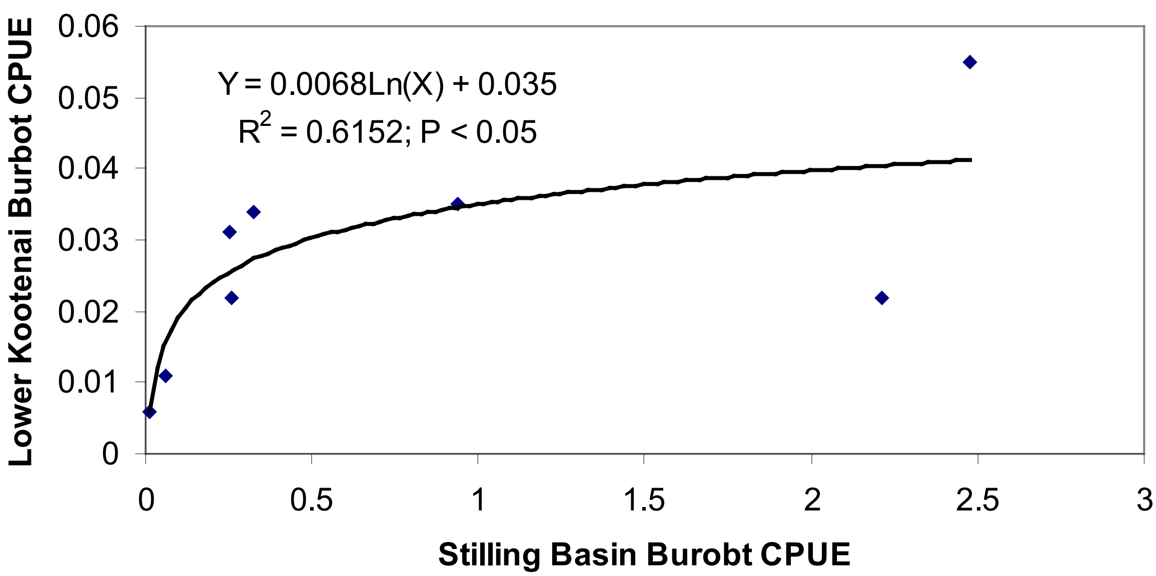

Figure 45. A regression analysis of mean burbot catch rates (burbot per trap day) for baited hoop traps in the stilling basin in the Kootenai River below Libby Dam and the lower Kootenai River (Paragamian and Hoyle 2003) between 1995/1996 and 2003/2003.

We believe that Libby Reservoir during the previous 12-15 years has stabilized in terms of biological production. Total fish abundance, as indexed by trends in gill net catch rates have stabilized since 1988. Fish and zooplankton species composition and abundance have also experienced similar trends. Mountain whitefish, rainbow trout and westslope cutthroat trout abundance all exhibited dramatic decreases in abundance (Figure 28) following the first ten years after reservoir filling, but have stabilized at much lower abundances than the pre-dam period. Fish species composition also shifted during the first 10 years after reservoir construction, but has also stabilized. Zooplankton abundance, species composition, and size distribution have also all been similar during the second half of the reservoir's history. We attribute these trends toward trophic equilibrium to the aging process of the reservoir (Kimmel and Groeger 1986) and the operational history of Libby Dam during the past 15 years.

Although all of the statistically significant trends observed for our macroinvertebrate monitoring on the Libby Creek Upper Cleveland Project are consistent with declining ecological integrity, the declines were relatively minor. The direction of change and taxa involved were consistent with changes caused by mild sedimentation, but were probably due to adjustments from an exceptional year at the beginning of the study and a return to more natural conditions. That is, the exceptionally low abundance of collectors in 2000, may have be responsible for some of the observed changes. The site will be monitored to ensure that sedimentation is not an issue. However, before we disregard the potential impairments, we should note that the restoration was relatively recent and there may be some associated ecosystem disturbance. Nonetheless, the evidence in this study suggests that these effects, if present are relatively minor.

Two of the most tangible macroinvertebrate metrics that can be used to assess ecological integrity are Taxa Richness and EPT Richness because they summarize how many taxa (species, genera, families) inhabit the site. Both of these measures showed subtle improvements that were not statistically significant. We ran a post hoc power analysis and found that we had 81.2% power to detect a difference between the taxa richness of the 2 years at the Libby Creek site. Similarly, we found that the study had 27% Power to detect differences in EPT Richness. For future reference, we ran an analysis to show how replication could have improved the ability to detect differences in these two metrics (Figures 46 through 48). Based on these results, we will further evaluate the utility of modifying future sampling to improve our ability to detect differences between years.

The rapid bioassessment classification we used for the Libby and Grave Creek classified both sites as moderately deviant from mountain reference conditions. It is important that these results are interpreted conservatively because they were not collected using the same procedures used to calibrate Marshall and Kerans (2003) biocriteria. In particular, the method used by FWP may increase the collection of midges (Diptera, Collector-gatherers), and be partially responsible for moderately deviant classification rating. However, it is also likely that the results could be indicative of sediment problems upstream in the watershed.

We know that biological significance is not always the same as statistical significance. Increases in the abundance of several sensitive taxa indicate that there may have been subtle improvements in the biotic condition during 2003. Moreover, the occurrence of these taxa as some of the most abundant (overall) taxa is important because it underscores the presence of sensitive taxa in the stream. That is, to colonize the restoration site, they must be able to live in Grave Creek in the first place. The causes of these subtle improvements may—or may not—be related to the restoration. These small improvements may be due to environmental differences between the two years—at the basin or regional scale. The best way to assess the benefits of the restoration is to compare the restored site to sites that have not been restored. This will allow changes associated with the restoration to be distinguished from broader environmental variation in the region.

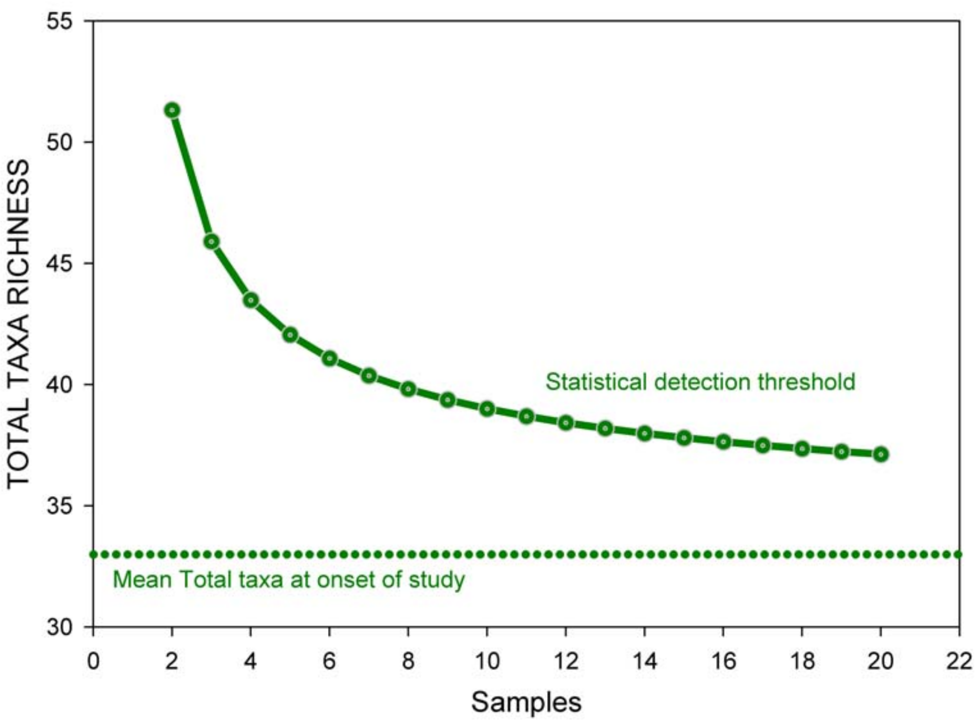

Figure 46. Minimal detectible difference for Taxa Richness for Libby Creek. This graph assumes a= 0.05, and b=0.1. With three replicates, the taxa richness must increase from 33 to > 46 for the change to be statistically significant.

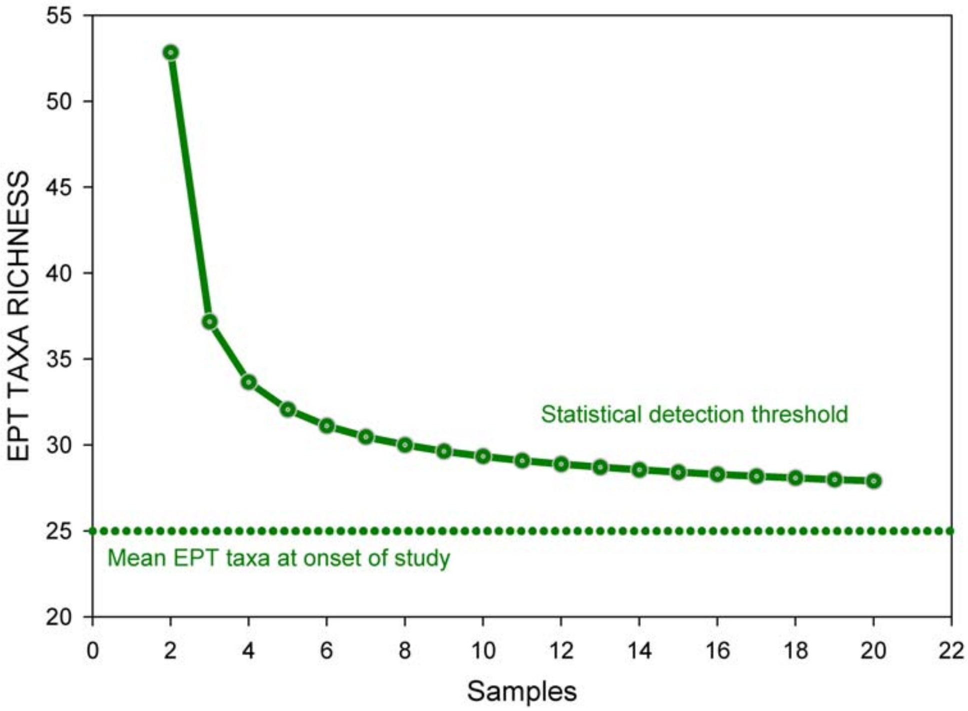

Figure 47. Minimal detectible difference for EPT Taxa Richness for Libby Creek. This graph assumes a= 0.05, and b=0.1. With three replicates, the EPT richness must increase from 25 to > 37 for the change to be statistically significant.

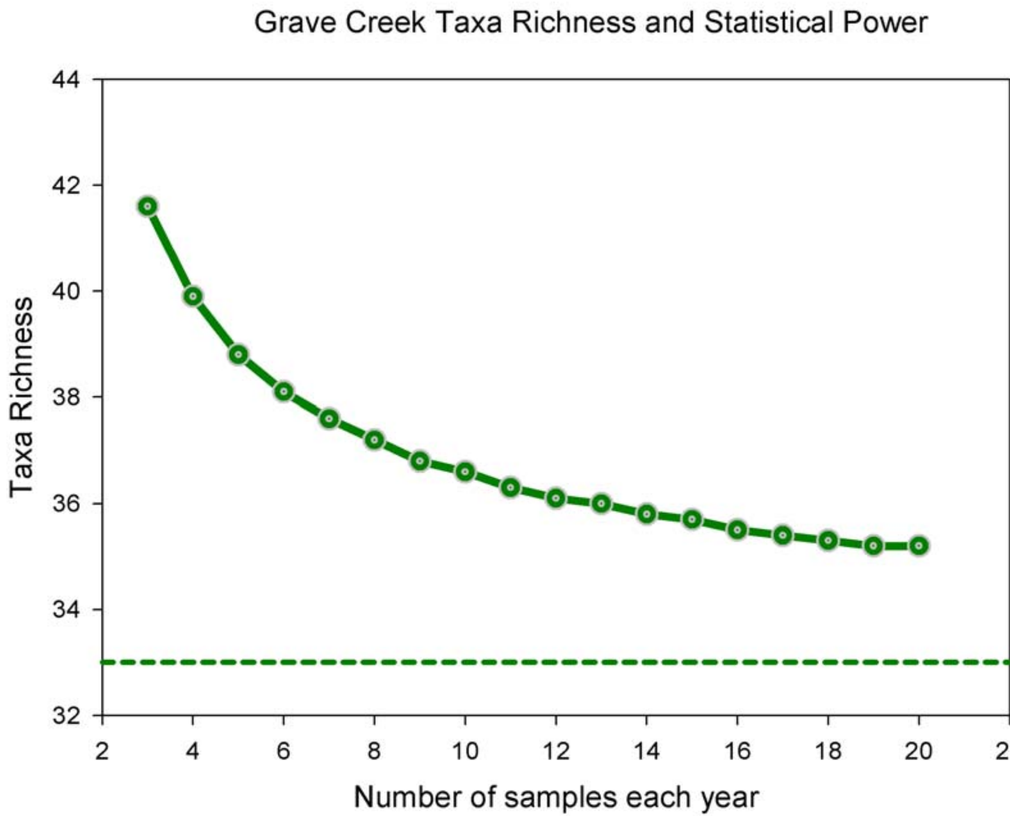

Figure 48. Minimal detectible difference for Taxa Richness for the Grave Creek Phase I Restoration Project. This graph assumes a= 0.05, and b=0.1. With three replicates, the taxa richness must increase from 33 to >41 for the change to be statistically significant. Similarly, if 20 samples are collected each year, a change from 33 to 36 taxa becomes statistically significant.

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Chapter 2

Stream Restoration and Mitigation Projects in the Montana Portion of the Kootenai River Basin

Abstract

A cooperative mitigation and implementation plan developed by Montana Fish, Wildlife & Parks, the Kootenai Tribe of Idaho and the Confederated Salish and Kootenai Tribes documents the hydropower related losses and mitigation actions attributable to the construction and operation of Libby Dam, as called for by the Northwest Power Planning Council's Fish and Wildlife Program (MFWP,CSKT and KTOI 1998). A mix of mitigation techniques is necessary to offset losses caused by dam construction and operation. In 2003, Montana FWP implemented several projects to mitigate for a portion of the losses attributable to the construction and operation of Libby Dam. In 2002, we began working cooperatively with the Lincoln County Fair board to construct a community-fishing pond at the Lincoln County Fair Grounds that will enhance angling and education opportunities for young angler. This project was completed in 2003. We identified Libby, Grave and Young creeks as high priority streams for restoration activities based on habitat quality, fish community composition, and native fish abundance. Libby Creek has been identified as a core area for native redband trout and bull trout, while Grave Creek has been identified as a core area for bull trout, and Young Creek is one of the most important westslope cutthroat trout spawning tributaries to Koocanusa Reservoir because it represents one of the last known genetically pure populations of westslope cutthroat trout in the US portion of the subbasin. We adopted a phased approach for restoring Libby and Grave creeks. The Grave Creek Phase I Restoration Project represents the third phase of restoration efforts on lower Grave Creek, and the upper Libby Creek Restoration Project was the second restoration project Montana FWP has implemented on Libby Creek. Project objectives for the Libby, Grave and Young Creek restoration projects focused on restoring channel stability and increasing the quality and quantity of trout rearing habitat. Restoration activities on the Libby Creek Cleveland Project and the Grave Creek Phase I Project were completed in the fall of 2002, and Young Creek State Lands Project was completed in the fall of 2003. The monitoring program for each of these projects includes pre- and post-construction monitoring that allows comparisons to describe changes in the physical environment as a result of these restoration activities. Dunnigan et al. (2003) demonstrated that the Libby and Grave Creek projects decreased the bankfull width and bank erosion and increased stream depth, substrate mean particle size, and the quality and quantity of salmonid rearing habitat. The monitoring results presented in this document evaluate whether these physical changes were maintained after the first spring freshet. The Young Creek State Lands Project effectively changed the stream channel pattern profile and dimension. These changes resulted in a narrower, deeper channel that are likely to improve the long-term quantity and quality of rearing habitat for native salmonids. The monitoring program for this project will allow us to assess whether the project continues to meet our objectives through time.

Introduction

Libby Dam, on the Kootenai River, near Libby, Montana, was completed in 1972, and filled for the first time in 1974. The dam was built for hydroelectric power production, flood control, and recreation. However, the socio-economic benefits of the construction and operation of Libby Dam have come at the cost to the productivity and carrying capacity of many of the native fish species of the Kootenai River Sub-basin. Libby Reservoir inundated 109 stream miles of the mainstem Kootenai River in the United States and Canada, and 40 miles of tributary streams in the U.S. that provided some of the most productive habitat for spawning, juvenile rearing, and migratory passage. Historically, the fish residing downstream of Libby Dam could access quality spawning habitat upstream of Libby Dam in the United States and Canada. Impoundment of the Kootenai River blocked the migrations of fish populations that once migrated freely between Kootenai Falls (29 miles downstream of Libby Dam) and the headwaters in Canada.

Operations of Libby Dam cause large fluctuations in reservoir levels and rapid daily fluctuations in volume of water discharged to the Kootenai River. Seasonal flow patterns in the Kootenai River have changed dramatically, with higher flows during fall and winter, and lower flows during spring and early summer. Reservoir operations that cause excessive drawdowns and refill failure are harmful to aquatic life in the reservoir. Jenkins (1967) found a negative correlation between standing crop of fish and yearly vertical water fluctuations in 70 reservoirs.

Problems occur for resident fish when Libby Reservoir is drawn down during late summer and fall, the most productive time of year. The reduced volume and surface area reduces the potential for providing thermally optimal water volume during the high growth period, and limits production of fall-hatching aquatic insects. Surface elevations continue to decline during winter, arriving at the lowest point in the annual cycle during April. Deep drafts reduce food production and concentrate young trout with predators. Of greatest concern is the dewatering and desiccation of aquatic dipteran larvae in the bottom sediments. These insects are the primary spring food supply for westslope cutthroat, a species of special concern in Montana, and other important game and forage species. Deep drawdowns also increase the probability that the reservoirs will fail to refill. Refill failure negatively effects recreation and reduces biological production, which decreases fish survival and growth in the reservoir (Marotz et al. 1996, Chisholm et al. 1989). Investigations by Daley et al. (1981), Snyder and Minshall (1996), and Woods and Falter (1982) have documented the declining productivity of the Kootenai System and, specifically, reduced downstream transport of phosphorous and nitrogen by 63 percent and 25 percent, respectively.

Large daily fluctuations in river discharge and stage (4-6 feet per day) strand large numbers of sessile aquatic insects in the varial zone (Hauer and Stanford 1996). The reduction in magnitude of spring flows has caused increased embeddedness of substrates, resulting in loss of interstitial spaces in cobble and gravel substrates, and in turn, loss of habitat for algal colonization and an overall reduction in species diversity and standing crop (Hauer and Stanford 1996). Aquatic insects are affected by the reduction of microhabitat and food sources, as evidenced by the loss of species and total numbers since impoundment (Voelz and Ward 1991). Hauer and Stanford (1996) found a significant reduction in insect production for nearly every species of insect during a 13-14 year interval in the Kootenai River. These losses can be directly attributed to hydropower operations. Benthic macro-invertebrate densities are one of the most important factors influencing growth and density of trout in the Kootenai River (May and Huston 1983).

Large gravel deltas have formed at the mouths of several tributaries of the Kootenai River (Quartz, O'Brien and Pipe Creeks) due to the loss of high spring flows. These deltas have reached proportions that are potential barriers to migrating fish such as bull trout, westslope cutthroat trout, burbot, and mountain whitefish at low river levels below Libby Dam (Graham 1979; Marotz et al. 1988).

A mix of mitigation techniques is necessary to offset losses caused by dam construction and operation. A cooperative mitigation and implementation plan developed by Montana Fish, Wildlife & Parks, the Kootenai Tribe of Idaho and the Confederated Salish and Kootenai Tribes documents the hydropower related losses and mitigation actions as called for by the Northwest Power Planning Council's Fish and Wildlife Program (MFWP, CSKT and KTOI 1998). This plan identifies several actions that do not require modification of Dam operation to be successful. These include aquatic habitat improvement, fish passage improvements, off-site mitigation, fisheries easements, and conservation aquaculture and hatchery products.

The Libby Creek watershed is the second largest tributary between Kootenai Falls and Libby Dam, and has an area of 234 square miles. Libby Creek provides critical spawning and rearing habitat and a migratory corridor for the threatened bull trout, and resident redband trout. The U.S. Fish and Wildlife Service's Bull Trout Recovery Plan designates Libby Creek as part of the Kootenai River and Bull Lake Critical Habitat Sub-Unit (USFWS 2002). Libby Creek has been degraded by past management practices, including road building, hydraulic and dredge mining, and riparian logging. These past activities likely disrupted the natural equilibrium within Libby Creek that resulted in accelerated bank erosion along a number of meander bends causing channel degradation. This resulted in poor fish habitat that likely reduced the productivity and carrying capacity for resident salmonids within Libby Creek. Currently the stream channel is over-widened and shallow with limited pool habitat (Sato 2000). Many of the problems related with the unstable conditions within the Libby Creek watershed are a result of land management activities that occurred in the upper watershed, and therefore restoration activities should first focus on the upper watershed (Sato 2000).

Grave Creek is a fourth order tributary to the Tobacco River, with a watershed area of approximately 55 square miles. Grave Creek is one of the most important bull trout spawning streams in the Montana portion of the Kootenai River (see Chapter 1), and has been designated as critical habitat within the U.S. Fish and Wildlife Service's Bull Trout Recovery Plan (USFWS 2002). Grave Creek is also currently on the Montana Water Quality Limited Segment List as an impaired stream. The State of Montana has proposed that Grave Creek be a high priority for Total Mean Daily Load allocation (TMDL). Grave Creek also provides water for westslope cutthroat trout habitat, agriculture and other riparian dependent resources. Timber harvest and road construction in the headwaters and agriculture, grazing, riparian vegetation losses, channel manipulation, and residential and industrial encroachment in lower reaches have impacted the lower three miles of Grave Creek by reducing stream stability, the quality and quantity of available fish habitat, and the composition of the riparian community. Therefore, lower Grave Creek is much less stable than it was historically, which has likely resulted in a reduction of salmonid productivity and carrying capacity from historic conditions. Restoration activities on Grave and Libby creeks are consistent with those strategies identified in the Fisheries Mitigation and Implementation Plan for the Losses attributable to the Construction and Operation of Libby Dam (MFWP, CSKT and KTOI 1998) and the Kootenai Subbasin Plan (KTOI and MFWP 2004).

Stream restoration efforts when applied appropriately can be successful at restoring streams to an equilibrium state. However, there are several critical fundamental issues that must be resolved prior to the design and implementation of any restoration project (Rosgen 1996). These include a clear definition and causes of the problems, an understanding of the future potential of the stream type as conditioned by the watershed and valley features, and an understanding of the probable stable form of the stream under the current hydrology and sediment regime (Rosgen 1996). The restoration projects described below were designed and implemented after considering these issues and other recommendations found in Rosgen (1996). The following sections discuss the results of the restoration activities and monitoring results.

Methods and Results

Eureka Pond

The MFWP staff began working with the Lincoln County Fairgrounds board of directors to construct a fishing pond on the fairgrounds property in Eureka in 2000, in an effort to help mitigate lost fisheries habitat and recreation opportunity in the Montana portion of the Kootenai River subbasin. Design work and discussions about liability issues delayed construction until the summer of 2002. The pond was excavated and lined with a mixture of silt and granular bentonite to minimize leaking. However, the course material beneath the pond was difficult to seal, and as a result, in the fall of 2003, the Libby Mitigation staff installed a polypropylene liner to seal the pond. The maximum depth of the pond is 8 feet and has a surface area of 0.4 acres. The water source for the Eureka pond is a 50 gallons per minute water right out of Mill Spring held by the Lincoln County Fair. Currently the only consumptive use in water for the pond is evaporation. The pond will be stocked with fish in the summer of 2004.

Figure 1. A photograph of the Eureka fishing pond taken in the spring of 2003, with the polypropylene pond liner installed. The pond has a maximum depth is 8 feet and a toal surface area of 0.4 acres. The pond will be stocked for recreational family fishing in the summer of 2004.

Libby Creek Upper Cleveland Project

Montana FWP completed the Libby Creek Upper Cleveland Stream Restoration Project in 2002 (approximate river mile 22), which restored approximately 3,200 feet of stream channel to the proper dimension, pattern and profile. This was conducted on Libby Creek located approximately 18 miles southwest of the town of Libby, Montana within Township 27 North, Range 31 West, Section 1 in Lincoln County (Figures 2 and 3). Past land management activities including logging, mining, riparian road construction, and stream channel manipulation have resulted in accelerated bank erosion along a number of meander bends, resulting in an over widened, unstable, and shallow channel (Sato 2000), which has resulted in low quality habitat for native salmonids including bull trout and redband trout.

The existing channel prior to this restoration project was over-widened with frequent lateral migration of the active stream channel. These conditions resulted in frequent multiple channels within the project reach (Figure 4). Width depth ratios were high (ranging from 28-43 feet) and shallow mean bankfull channel depths ranging from 0.58 to 1.79 feet in depth (Table 1). We established design criteria for the channel dimensions according to reference reach criteria established by Rosgen (1996). Table 1 provides the design criteria and summary of existing conditions for several stream channel parameters.

Stream restoration work began in September 2002 and proceeded through November 2002. During this period Montana FWP excavated approximately 3,200 feet of new channel according to the design criteria (Table 1) including an average design bankfull width and depth of 32 feet and 3 to 7 feet, respectively. We designed the channel pattern (Dunnigan et. al 2003) to utilize existing riparian vegetation in project reach wherever possible, in an attempt to maximize channel stability, and promote recovery of the riparian area. The resulting stream pattern design increased sinuosity (stream length divided by valley length) from 1.1 to 1.6, and subsequently increased total stream length from approximately 2,700 to 3,200 feet. During construction phase of this project, numerous structures were installed including 11 Cobble grade control structures for grade control and bank protection in pool tail-outs created by outside bends and rootwad complexes, 19 rootwad complexes for bank stabilization on outside bends of the newly constructed stream channel, 3 rock vanes to provide gradient control and pool habitat. Substantial effort was also expended to restore a healthy riparian vegetative community. These efforts included transplanting approximately 500 shrubs during construction and planting approximately 2,000 willow cuttings, 75 cottonwood poles, and 1,600 containerized native shrubs after stream channel construction.

The stream channel profile prior to project construction contained few pools (Figure 5), and due to the limited geographical overlap with the newly designed channel thalweg could not accurately be displayed on the same figure as the new channel profile surveyed in 2002 and 2003 (Figure 6). The designed channel profile required excavation at numerous depositional areas throughout the project reach (Figure 6) and resulted in an increased quantity of pool habitat within the project area. Prior to project construction, the mean pool-to-pool distance was 325.4 feet. We resurveyed the project area in the summer of 2003 after the restoration work had been subjected to the first spring freshet after construction, and the mean pool spacing was 172.8 feet, which represented a 46.9% reduction in the distance between pools. Pool spacing measurements in 2003 represented a complete sampling of all pools present during both years. However, the measurements collected in 1999 represented a sampling of the pools. We conducted a t-test and concluded that the distance between pools decreased significantly as a result of project construction ($p < 0.05$).

In 1999, prior to project construction, we measured stream channel morphology at 5 cross-sectional survey locations in riffle habitat within the project area. After project construction in 2002 we established 9 permanent transects in riffle habitats. These same locations were surveyed again in the summer of 2003, after the project had experienced the first spring freshet. At each transect we measured mean bankfull width, depth, width to depth ratio, and cross sectional area. We used analysis of variance (ANOVA) and a subsequent multiple comparison test (Fisher's Least Significant Difference; Zar 1996) to test for significant differences between years (alpha = 0.05; Table 2). Mean bankfull width, depth, and width to depth ratio were significantly reduced from 1999 to 2002 and 1999 to 2003. Comparisons for the three parameters between 2002 and 2003 were not significantly different ($p > 0.05$; Table 2). However, the associated variance for each of the 4 parameters decreased each year (Table 2).

Due to the importance of pool habitat to rearing redband and bull trout within the project area, we devoted a substantial effort to monitor pool habitat after project construction to evaluate whether the pools maintained depth, width and length through time. After project construction in the fall of 2002, we measured mean bankfull depth, width, length and maximum bankfull depth of the 20 pools constructed in the project area. We repeated these measurements in the summer of 2003 after the project had experienced the first spring freshet. We did not perform a statistical comparison for these data because the pool measurements represented all pools within the project area (i.e. complete census), making statistical comparisons unnecessary. We observed a decrease in the mean values of each of the 4 parameters from 2002 to 2003 (Table 3). Mean bankfull depth showed the sharpest decline (18%) from 2002 to 2003, followed by pool length, which decreased by 17.7% (Table 3). The variance of each of the 4 parameters also decreased from 2002 to 2003 (Table 3). The mean annual variance for mean bankfull depth had the highest decrease (18%) between 2002 and 2003, followed by maximum bankfull depth variance, which decreased by 40.4% (Table 3). Although mean pool length decreased by 17.7%, the total proportion of pool habitat (sum of all pool lengths divided by total project area stream length) within the project area decreased by 4%, from 22% in 2002 to 18% in 2003.

In addition to a complete census of all pools within the project area, we also surveyed all 13 riffles within the project area to evaluate changes in riffle slope through time. The post-construction mean riffle slope in 2002 was 1.95% (variance = $7.79*10^{-5}$), which decreased by 33.3% in 2003 to a mean riffle slope of 1.30% (variance = $2.033*10^{-5}$). We attribute the overall flattening of the riffles within this project to 2 factors. Further examination of our survey information revealed that the top end of our riffles generally incised within the channel (degraded) due in part to the scour achieved below many of the gradient control cobble structures installed at the tailout area of many of the pool structures. The lower portion of many of these same riffles aggraded with bed materials, which had the overall result of reducing the overall riffle slope by one third.

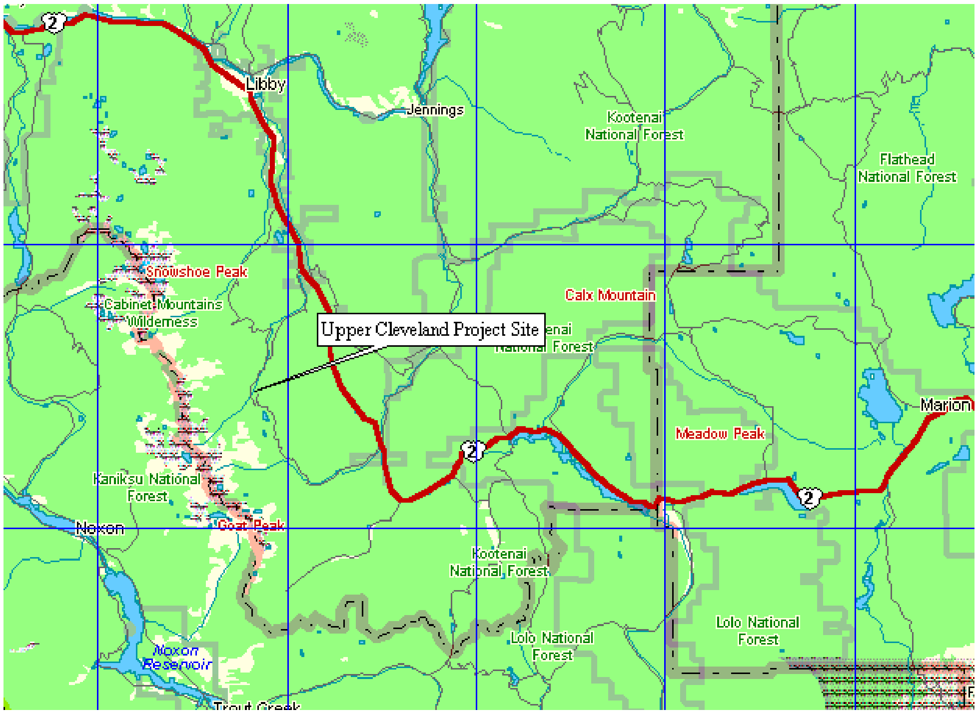


Figure 2. Vicinity Map for Upper Libby Creek Cleveland Project.

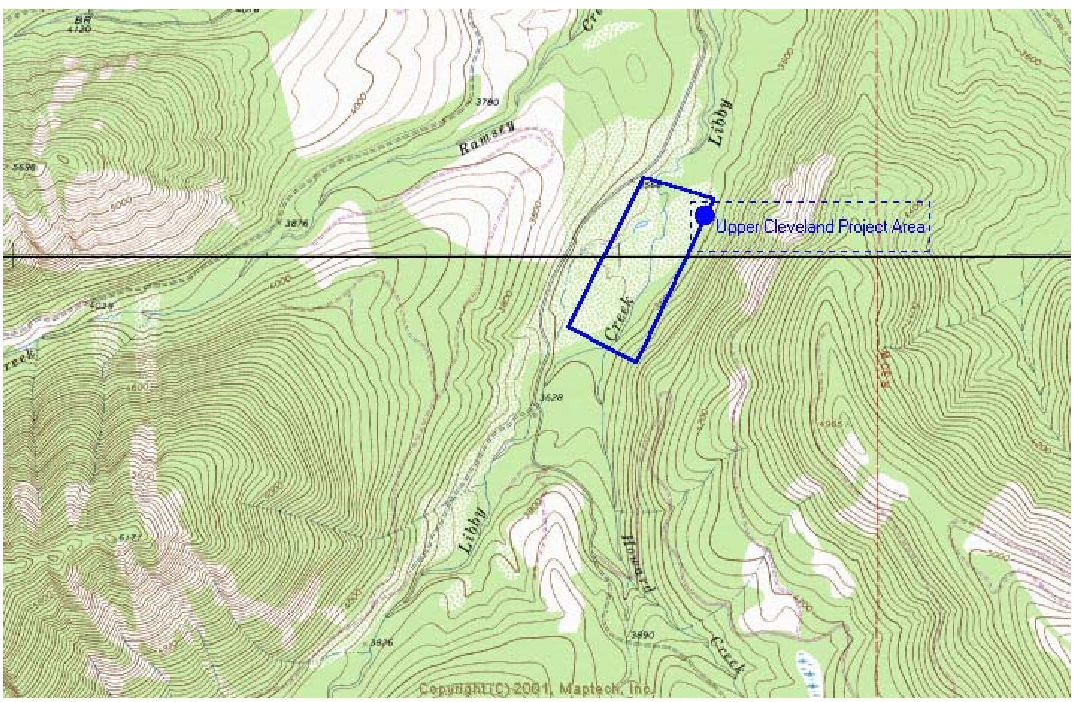
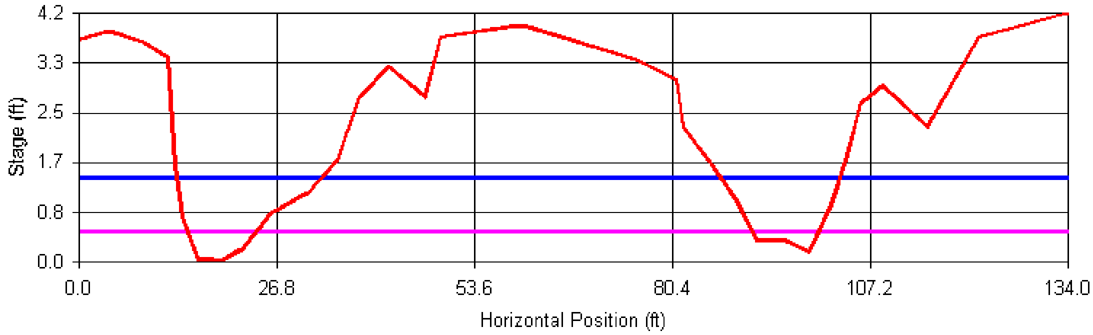


Figure 3. Detailed site location map of the upper Libby Creek Cleveland Project Area.



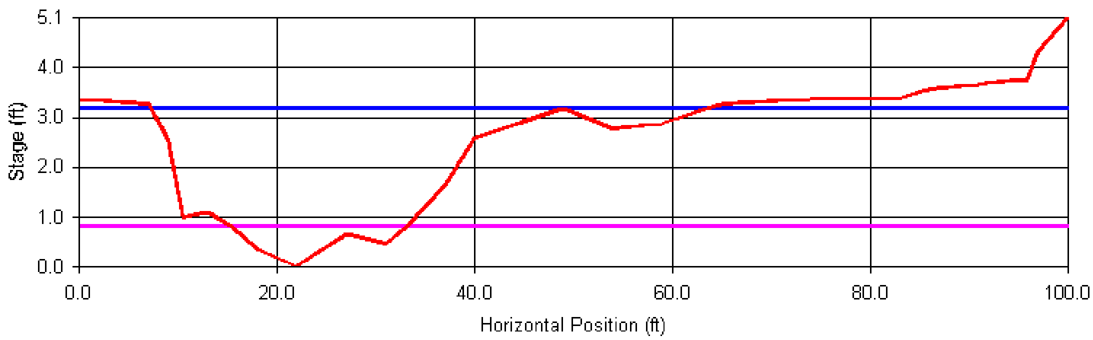


Figure 4. The top cross sectional survey of Libby Creek (#12C) was surveyed by Montana FWP in 1999, and is typical representation of the braided channel and large amounts of deposition within the floodplain of the upper Libby Creek Cleveland Project site. The lower figure characterizes the design criteria used to implement project construction activities at this site.

Table 1. Design specifications for the upper Libby Creek (Cleveland's) channel restoration project.			
Channel Design	Range	Mean	Existing – 1999 (Mean)
Design Channel Type (C4) Drainage Area = 12 sq. miles			
Total Length		3,200	2,700 (braided throughout)
Bankfull Width (ft)	28-35	33	28-43 (37)
Bankfull Area (ft^2)	40-60	47	47-123
Width/Depth Ratio	20-23	21.5	15-35
Sinuosity	1.3 – 1.7	1.6	1.1 – 1.06 (1.1)
Band Width (ft)	100 -140	135	135
Radius of Curvature (Rc) (ft)	88-135	106	108-204 (143)
Rc/Wbf Ratio*	2.75 – 4.2	3.3	3.3 – 6.3 (4.4)
Mean Bankfull Depth (ft)	2.0-2.2	2.1	0.58 – 1.79 (1.18)
Max Bankfull Depth (ft)	2.5-3.2	3.0	1.6-3.1 (2.15)
Max Scour Depth (ft)	7.0		7.0
Riffle Mean Velocity (fps)	5.0-6.0	5.5	6.5 – 7.0 (6.8)
Meander Length (ft)	290-485	369	184 – 900 (481)
Pool Spacing (ft)	75 –100'	80'	127 –500 (247)
Riffle Slope ft/ft (Base Flow)	0.018 – 0.020	0.019	0.011-0.033 (0.019)
Pool Slope ft/ft (Base Flow)	0.002 – 0.003	0.0025	0.007-0.003 (0.005)

* Rc/Wbf ratio for higher bedload C4 stream types should average 3.0 – 3.5 to effectively transport sediment.

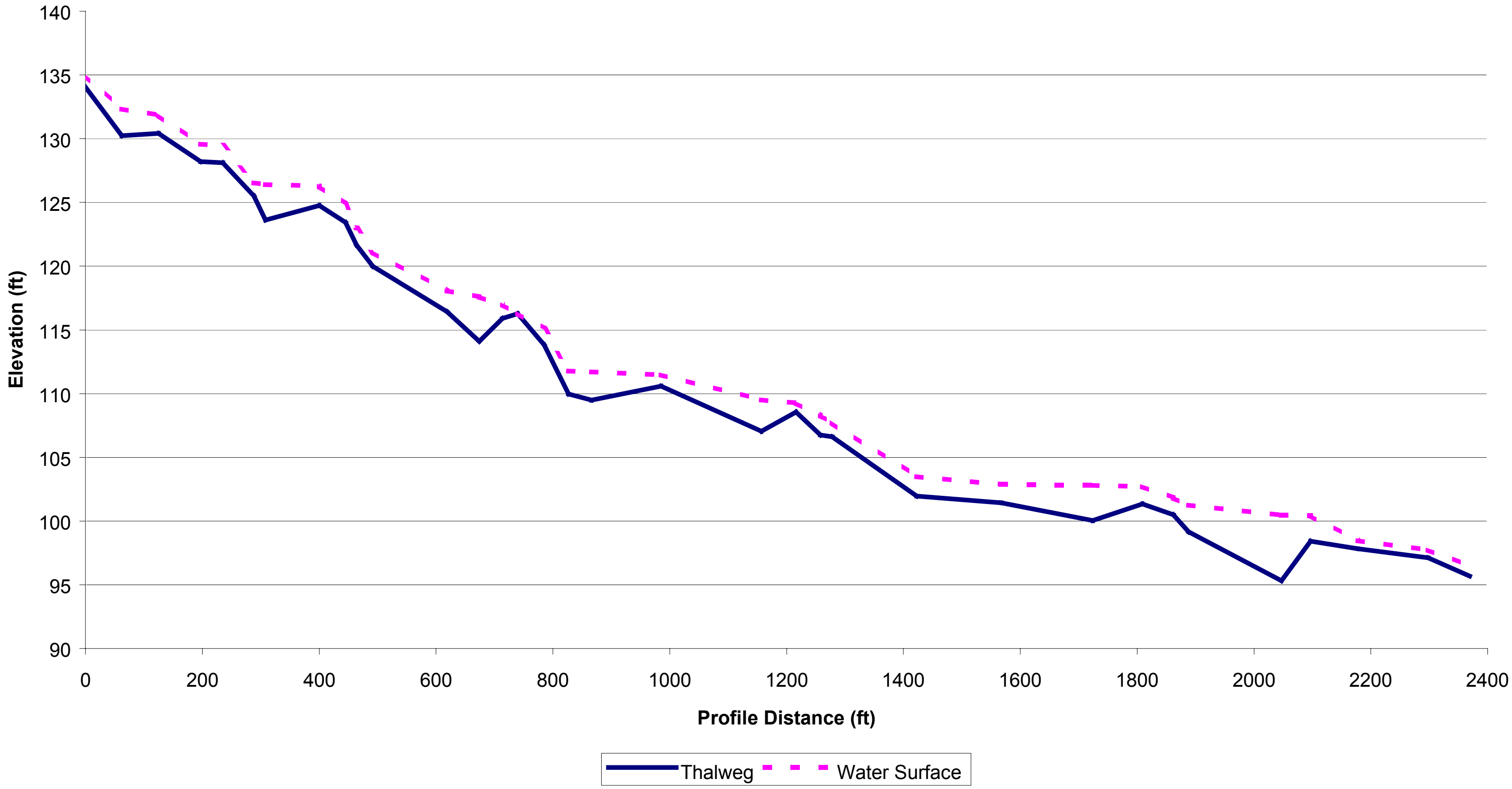


Figure 5. The longitudinal profile of the existing stream channel prior to the implementation of the restoration project. The survey was conducted beginning at station 0 (upper project boundary) to approximately 2350 feet prior to the implementation of the restoration project in the fall of 2002. The survey was not completed for the lower approximate third of the project area. Note the lack of pool habitat within the project area.

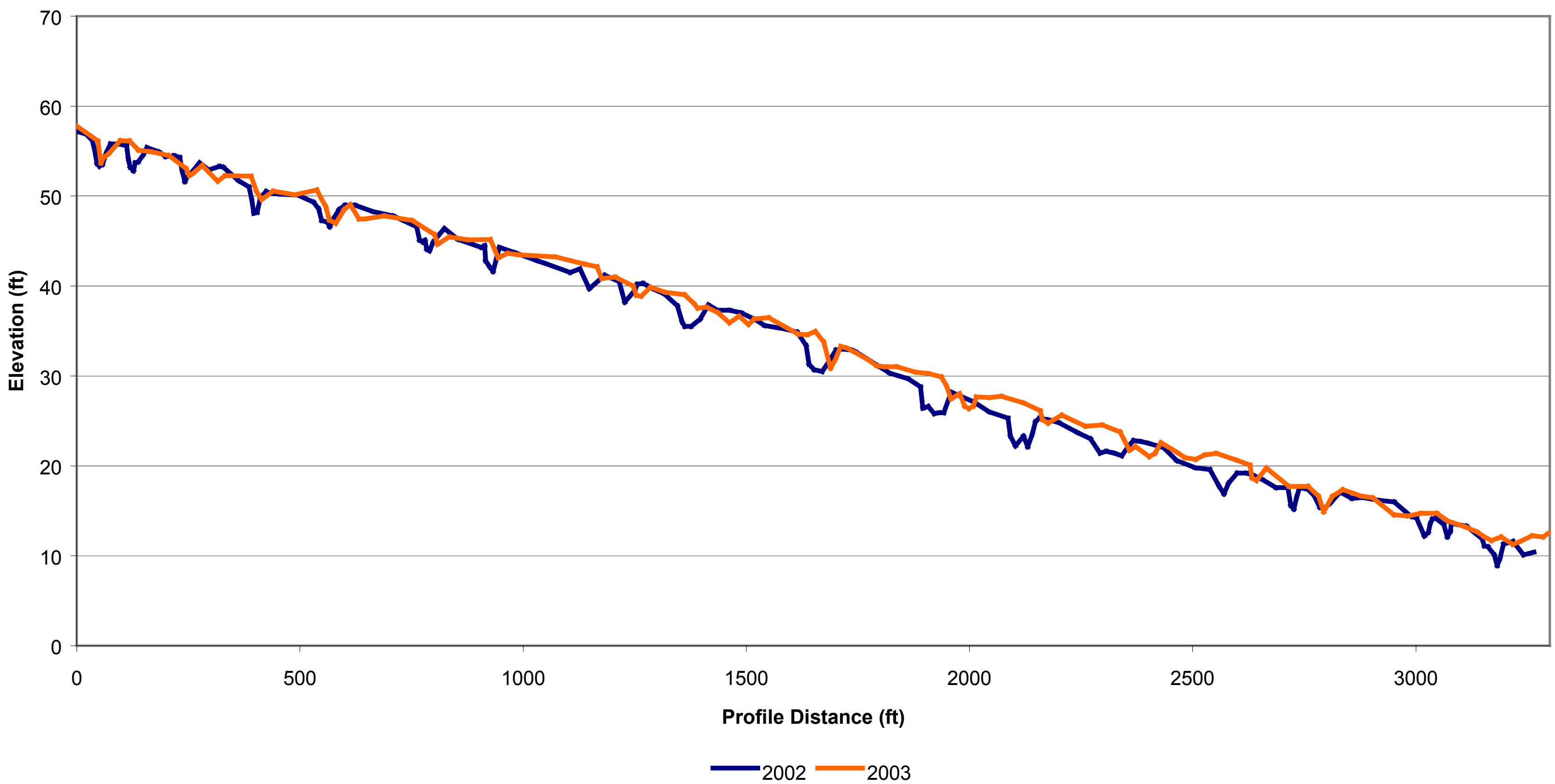


Figure 6. The longitudinal profile of the constructed stream channel thalweg in 2002 (as built) and 2003 after the project experienced the first spring freshet. The survey begins at the upper project boundary (station 0) and proceeds downstream to the lower project boundary (approximate station 3200). The stream channel prior to channel construction was not located within the same general plan view as the newly constructed stream channel. Therefore, the existing stream channel longitudinal profile (Figure 5) could not be superimposed on this figure due to differences in stream channel length that resulted from an overall increase in overall channel sinuosity and length after project construction.

Table 2. Mean bankfull width, depth, width to depth ratio and cross sectional area for 5 cross sectional surveys in 1999 and 9 different cross sectional surveys in 2002 and 2003. Variance estimates for annual mean values are presented in parentheses. An analysis of variance (ANOVA) was preformed for each parameter, and the P value is presented for each ANOVA. The results of subsequent multiple comparisons (Fisher's Least Significant Difference) are also presented (alpha = 0.05).

	Mean Bankfull Width (ft)	**Mean Bankfull Depth (ft)**	**Width to Depth Ratio**	**Cross Sectional Area (Square ft.)**
1999 (Pre Restoration)	41.5 (35.2)	0.94 (0.07)	47.6 (359.8)	39.6 (211.3)
2002 (As Built)	34.3 (30.5)	1.33 (0.09)	26.7 (59.0)	46.0 (114.3)
2003	31.5 (18.5)	1.48 (0.04)	21.8 (25.4)	47.9 (62.5)
ANOVA and Multiple Comparison Results	P = 0.008 1999/2002 Significant 1999/2003 Significant 2002/2003 Non-Signif.	P = 0.004 1999/2002 Significant 1999/2003 Significant 2002/2003 Non-Signif.	P = 0.0007 1999/2002 Significant 1999/2003 Significant 2002/2003 Non-Signif.	P = 0.385 1999/2002 Non-Signif. 1999/2003 Non-Signif. 2002/2003 Non-Signif.

Table 3. Mean bankfull width, depth, maximum bankfull depth, and length measured from 20 pools in 2002 and 2003. Variance estimates for annual mean values are presented in parentheses. A statistical comparison of annual mean values was not performed because the 20 pools represented all pools within the project area, and therefore represents a complete census. The percent change for each parameter from 2002 to 2003 is also presented.

	Mean Bankfull Width (ft)	**Mean Bankfull Depth (ft)**	**Maximum Bankfull Depth (ft)**	**Length (ft.)**
2002 (As Built)	37.95 (23.75)	2.64 (0.763)	4.32 (1.148)	36.7 (205.2)
2003	34.5 (16.07)	2.16 (0.30)	3.80 (0.684)	30.2 (130.8)
Percent Change	-9.16% (-32.3%)	-18.01% (60.7%)	-12.03% (-40.4%)	-17.67% (-36.2%)

Grave Creek Phase I Restoration Project

Montana FWP entered into a cooperative agreement that was coordinated through the Kootenai River Network to retain a consultant to develop and implement a restoration plan for approximately 4,300 feet of channel within the lower three miles of Grave Creek (WCI 2002). Additional contributors to the project included Montana Department of Environmental Quality, the National Fish and Wildlife Foundation, the Steele-Reese Foundation, the U.S. Fish and Wildlife Service (Partners for Wildlife Program), the Montana Community Foundation, the Montana Trout Foundation, and the Cadeau Foundation. The project is termed the Grave Creek Phase I Restoration Project, and begins at the downstream end of the Grave Creek Demonstration Project (see Dunnigan et al. 2003). Project construction work began during the fall of 2002. The objectives of the project were to: 1) Reduce the sediment sources and bank erosion throughout the project area by incorporating stabilization techniques that function naturally with the stream and which decrease the amount of stress on the stream banks, 2) Convert the channelized portions of stream into a channel type that is self maintaining and will accommodate floods without major changes in channel pattern or profile, 3) Use natural stream stabilization techniques that will allow the stream to adjust slowly over time and be representative of a natural stream system, 4) Improve fish habitat, particularly for bull trout, and improve the function and aesthetics of the river and adjacent riparian ecosystem, and 5) Reduce the effects of flooding on adjacent landowners.

Stream restoration work began in September 2002 and proceeded through December 2002. During that period numerous structures were installed to accomplish the above stated objectives. These structures included 12 rootwad composites, 11 debris jams, 8 log J-hook vanes, 4 cobble patches, 3 log cross vanes, 1 rock cross vane, 1 rock J-hook vane, 1 straight log vane, and 2.4 acres of sod transplants. The revegetation work was started in the fall of 2002, but due to unfavorable weather conditions, it was not completed until the spring of 2003, and is expected to serve as the primary stabilization mechanism in the long-term.

Grave Creek Phase I Restoration Project area has been subjected to long-term urban encroachment, removal of riparian vegetation, and extensive channel manipulation. These activities have resulted in the substantial reduction in floodplain and streamside vegetation, and the alteration of lower Grave Creek's natural dimension and meander pattern.

The Grave Creek Phase I Restoration Project changed the dimension, pattern and longitudinal profile within the project area, which were designed to achieve the long-term project objectives. Table 4 presents the existing and design criteria for some important geomorphical stream characteristics. We surveyed 6 permanent cross-sections located throughout the project area in 1999 (pre project), 2002 (post-construction) and 2003 (after the first spring freshet), and measured mean bankfull width, depth, cross sectional area, maximum depth, and width to depth ratio at each transect. We used a repeated measures analysis of variance and subsequent multiple comparisons to test for significant differences between years. Of the 5 parameters that we measured at each of the 6 transects, width to depth ratio changed the most as a result of project implementation compared to existing conditions (Table 5). Other significant results of project construction occurred in the mean bankfull depth, maximum depth, and bankfull width, respectively, with trends toward a narrower and deeper stream channel. Cross-sectional area did not differ significantly between years ($p = 0.166$; Table 5). None of the 6 channel dimension parameters differed significantly ($p < 0.05$) between 2002 and 2003. However, width to depth ratio and mean bankfull depth were close to being significant (Table 5), with trends toward a slightly deeper channel.

The 41 stream restoration structures described above, increased channel diversity within the project area along the longitudinal profile (Figure 7). The existing stream channel prior to the implementation of this project contained long riffle sections and relatively low sinuosity (Table 4). This project constructed a stream pattern within this reach of Grave Creek that decreased the overall stream gradient by increasing stream length (increased sinuosity; Table 4).

The Grave Creek Phase I Restoration Project also increased the quality and quantity of rearing habitat for native salmonids by increasing the total number and depth of pools compared to conditions that existed prior to restoration (Dunnigan et al. 2003). Due to the importance of pool habitat to rearing native salmonids within lower Grave Creek, we devoted a substantial effort to monitor pool habitat after project construction to evaluate whether the pools maintained depth, width and length through time. After project construction in the fall of 2002, we measured mean width, length and maximum bankfull depth of the 27 pools constructed in the project area. We repeated these measurements in the summer of 2003 after the project had experienced the first spring freshet (Table 6). We did not perform a statistical comparison for these data because the pool measurements represented all pools within the project area (i.e. complete census), making statistical comparisons not necessary. Pool length had the highest relative change between years. Both total length and mean pool length increased by 16.2% from 2002 to 2003 (Table 6). Mean maximum bankfull depth decreased by 14.8% from 2002 to 2003, and although we did not measure mean bankfull depth of the 27 pools, it is likely that it also decreased between years. Total pool surface area within the project area increased by 26.2% from 3,206 ft^2 in 2002 to 4,046 ft^2 in 2003. Therefore, even if mean bankfull pool depth decreased from 2002 to 2003, it is likely that total pool volume remained similar between years since pool area increased.

In addition to a complete census of all pools within the project area, we also surveyed 7 riffles within the project area in order to evaluate changes in riffle slope through time. The post-construction mean riffle slope in 2002 was 1.46% (variance = $6.55*10^{-5}$), which decreased by 39.2% in 2003 to a mean riffle slope of 0.89% (variance = $1.79*10^{-5}$). This trend was similar to the one we observed for riffles within the Libby Creek Upper Cleveland Project (see above). Furthermore, we attribute the overall flattening of the riffles within this project to the same 2 factors that caused the Libby Creek project riffles to flatten. The top end of the Grave Creek riffles also incised within the channel (degraded) due in part to the scour achieved below many of the gradient control cobble structures installed at the tailouts of many of the pool structures, and the lower portion of many of these same riffles aggraded with bed materials, which had the overall result of reducing the overall riffle slope.

Table 4. Design specifications for the Grave Creek Phase I Restoration Project.			
Channel Design	**Range**	**Mean**	**Existing (Mean)**
Design Channel Type (C3) Drainage Area = 74.2 sq. miles			
Total Length of Project = 4,300 feet.			
Bankfull Width (ft)	50-54	52	45-240
Bankfull Area (ft^2)	108-132	120	143
Width/Depth Ratio	18-22	20	93.5
Sinuosity	N/A	1.4	1.15
Band Width (ft)	270-495	392	330
Radius of Curvature (Rc) (ft)	180-234	208	105-180
Rc/Wbf Ratio	3.5-4.5	4.0	1.3-2.3
Mean Bankfull Depth (ft)	2.2-2.4	2.3	1.24
Max Bankfull Depth (ft)	2.8-3.4	3.0	2.56
Max Scour (Pool) Depth (ft)	7.0-8.0	7.0	4.2-4.4
Meaner Length (ft)	720-1000	860	625
Pool Spacing (ft)	360-500	430	670
Riffle Slope ft/ft (Base Flow)	0.013-0.018	0.015	0.013-0.016 (0.0145)
Pool Slope ft/ft (Base Flow)	0.0018-0.0027	0.0025	0.004-0.005

Table 5. Mean bankfull width, depth, width to depth ratio, cross sectional area, and maximum depth for 6 permanent cross sectional surveys in 1999, 2002 and 2003. Variance estimates for annual mean values are presented in parentheses. A repeated measures analysis of variance (RPANOVA) was preformed for each parameter, and the P value is presented for each RPANOVA. The results of subsequent multiple comparisons are also presented.

	Mean Bankfull Width (ft)	**Mean Bankfull Depth (ft)**	**Width to Depth Ratio**	**Cross Sectional Area (Square ft.)**	**Maximum Bankfull Depth (ft)**
1999 (Pre Restoration)	110.7 (1135.1)	1.26 (0.1)	96.1 (2461.2)	136 (1322)	2.85 (0.8)
2002 (As Built)	53.7 (51.5)	2.06 (0.2)	27.0 (39.8)	114.7 (885.5)	4.67 (2.5)
2003	51.8 (21.0)	2.32 (0.05)	22.5 (8.3)	125 (342.4)	4.73 (1.7)
RPANOVA and Multiple Comparison Results	P < 0.00001 1999/2002 P = 0.005 1999/2003 P = 0.006 2002/2003 P = 0.36	P < 0.00001 1999/2002 P = 0.01 1999/2003 P = 0.001 2002/2003 P = 0.093	P = 0.001 1999/2002 P = 0.016 1999/2003 P = 0.014 2002/2003 P = 0.08	P = 0.166 1999/2002 P = 0.127 1999/2003 P = 0.430 2002/2003 P = 0.127	P = 0.003 1999/2002 P = 0.028 1999/2003 P = 0.007 2002/2003 P = 0.799

Table 6. Mean bankfull width, maximum bankfull depth, and length measured from 27 pools in 2002 and 2003. Variance estimates for annual mean values are presented in parentheses. A statistical comparison of annual mean values was not performed because the 27 pools represented all pools within the project area, and therefore represents a complete census. The percent change for each parameter from 2002 to 2003 is also presented.

	Mean Bankfull Width (ft)	**Maximum Bankfull Depth (ft)**	**Mean Length (ft.)**	**Total Length (ft.)**
2002 (As Built)	49.5 (18.3)	6.5 (1.1)	64.7 (359.0)	1748.0
2003	53.8 (45.9)	5.6 (1.9)	75.2 (815.4)	2031.2
Percent Change	8.6%	-14.8%	16.2%	16.2%

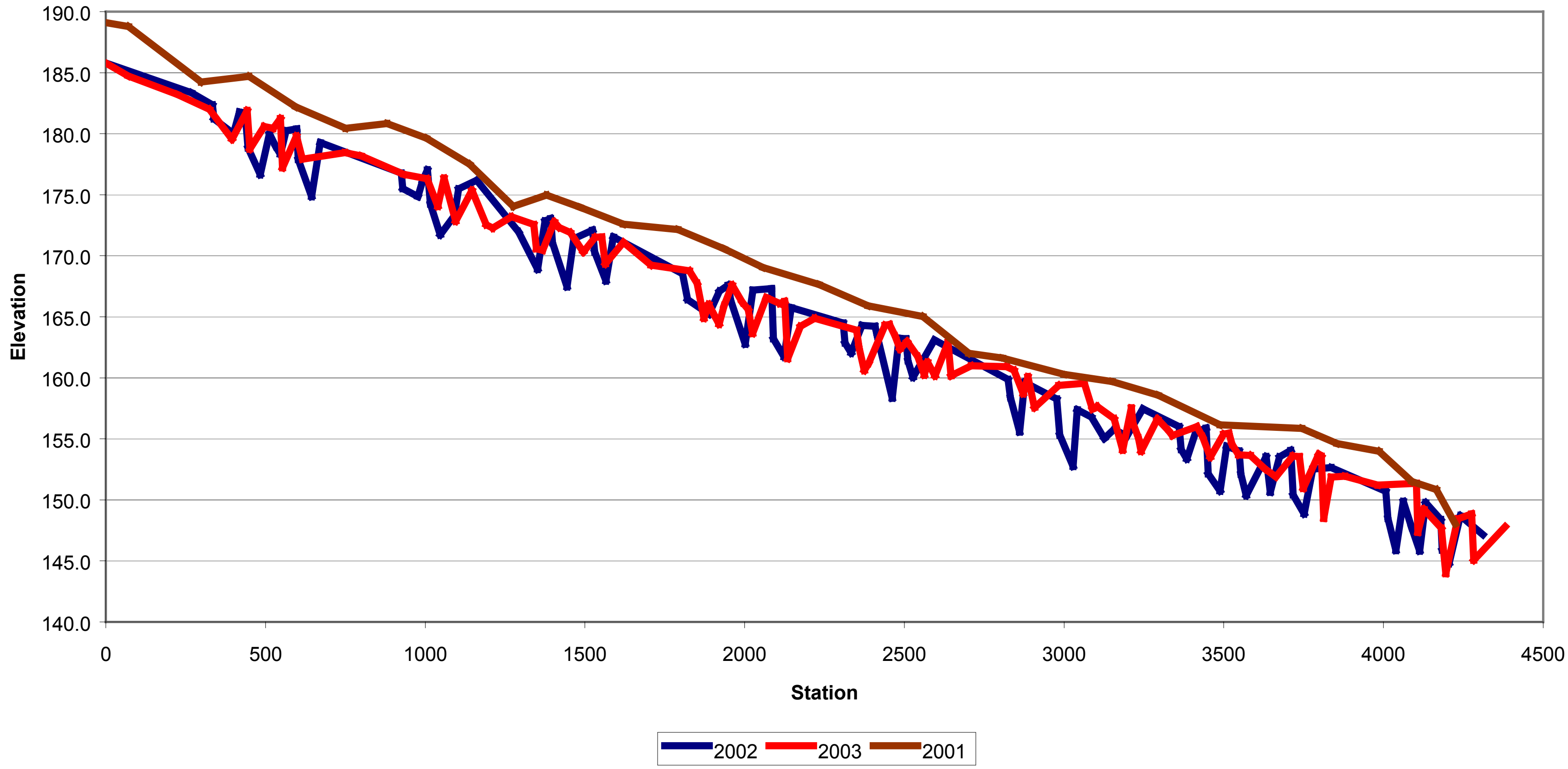


Figure 7. The longitudinal profile for the Grave Creek Phase I Restoration Project. The survey was completed before (2001) and after (2002; as built) project completion and after the first spring freshet (2003). The station (longitudinal location measured at the channel thalweg) begins at the upstream boundary of the project.

Young Creek State Lands Restoration Project

Young Creek is one of the most important westslope cutthroat trout spawning tributaries to Libby Reservoir because it represents one of the last known genetically pure populations of westslope cutthroat trout in the region and it is also one of the most potentially productive tributary streams to Libby Reservoir. Although bull trout (*Salvelinus confluentus*) do not routinely spawn in Young Creek, juvenile bull trout commonly enter Young Creek from the reservoir and rear for extended periods. This stream also provides water for agriculture, and other riparian-dependent resources. During the 1950's, approximately 1,200 feet of the channel located on the state owned section (DNRC School Trust Land; Figures 8 and 9) on Young Creek was straightened, diked, and the stream channel moved near the toe of the hill slope. This channelization compromised the stream's ability to effectively transport sediment through the channelized area, which caused the channel to aggrade (deposit bedload materials) and exacerbate flood conditions. Ironically, these were presumably the conditions that the original channel modification aimed to alleviate. Consequently, the aggradation caused numerous problems with the stream, such as; poor aquatic habitat, increased flood potential, lateral bank scour and increased sediment supply. Additionally, livestock grazing and timber management in the upper reaches of Young Creek likely contributed to the instability of the channel. The degraded condition of this section of Young Creek has contributed to the stream's inability to adequately transport stream flow and bedload supply and still maintain a stable channel. The project site is a 1,200-foot, over-widened reach of Young Creek containing several mid-channel gravel bars and eroding stream banks. In order to improve the function and stability of this section of Young Creek, Montana FWP reconstructed the stream channel in the fall of 2003.

The intent of the project is to: 1) reduce the sediment sources and bank erosion throughout the project area by incorporating stabilization techniques that function naturally with the stream and which decrease the amount of stress on the stream banks; 2) convert the channelized portions of stream into a channel type that is self-maintaining and will accommodate floods without major changes in channel pattern or profile; 3) use natural stream stabilization techniques that will allow the stream to adjust slowly over time and be representative of a natural stream system; and 4) improve fish habitat, particularly for westslope cutthroat trout, and improve the function and aesthetics of the stream and adjacent riparian ecosystem.

The Young Creek State Lands Restoration Project changed the dimension, pattern and longitudinal profile of this section of Young Creek. Prior to project implementation, the stream consisted of multiple channels throughout much of the project reach with lateral channel migration common between and within years. We completed cross-sectional surveys in 4 riffles prior to project construction and 10 after the project was completed. We measured cross sectional area, mean bankfull width, depth, maximum depth, and width to depth ratio at each transect, and compared mean values for each parameter using a t-test. The existing conditions were typified by an over widened and shallow channel with a mean bankfull width of approximately 28 feet, a mean bankfull depth of 0.6 feet, and a mean width to depth ratio of approximately 48 (Table 7). The designed channel significantly ($p < 0.05$)

reduced the mean width and width to depth ratio to approximately 16 feet and 14, respectively (Table 7). Cross sectional area, maximum depth, and mean bankfull depth all significantly ($p < 0.05$) increased as a result of project construction (Table 7). We also conducted cross sectional surveys in 2 pools prior to beginning the project and 8 pools after the project was completed (as built). We measured the same parameters at the pool transects that we measured at the riffle sites, and compared mean values between years using a t-test (Table 8). The results for the pool transects were similar to the trends we observed for the riffle habitats. Mean pool cross sectional area, maximum depth and mean bankfull depth all significantly ($p < 0.05$) increased after project construction. The width to depth ratio of pools significantly decreased as a result of the restoration project. Mean bankfull width also decreased after project construction, but the difference was not significant (Table 8).

The stream restoration techniques we employed increased channel diversity, stream length, and sinuosity within the project area (Figure 10). The existing stream channel prior to the implementation of this project was 908 feet long with an average gradient of 1.3% that consisted of relative long riffles and a fairly low sinuosity of 1.1. This project constructed a stream pattern within this reach of Young Creek that decreased the overall stream gradient to 1.0%, and increased the sinuosity to 1.41by increasing stream length to 1158 feet (Figure 10).

The Young Creek State Lands Restoration Project also increased the quality and quantity of rearing habitat for native salmonids. We compared the number and dimensions of pools from the existing channel (2002) and the as built channel (2003). We measured mean bankfull depth, width length of each pool in order to estimate total pool area and volume before and after project construction. We realized a 500% increase in the total number of pools present in this section of Young Creek as a result of this restoration project. The total number of pool increased from 2 pools to 12 pools. Total pool area and volume also increased. We increased total pool area by 537% and pool volume increased by 1295%, due to increased pool depth. The large woody debris stems and root wads used during project construction also likely increased cover available to rearing and migrating salmonids within this reach of Young Creek.

We will continue to monitor this project in order to determine if project objectives are maintained through time. Pre- and post-construction monitoring to date within the project area include permanent stream channel cross-sections, a longitudinal profile, and numerous photo points. These monitoring activities will allow us to determine how channel morphology and dimension change in time. Fisheries population estimates and aquatic insect surveys have been conducted three years prior to work within the project area in order to assess the aquatic community response to the restoration work.

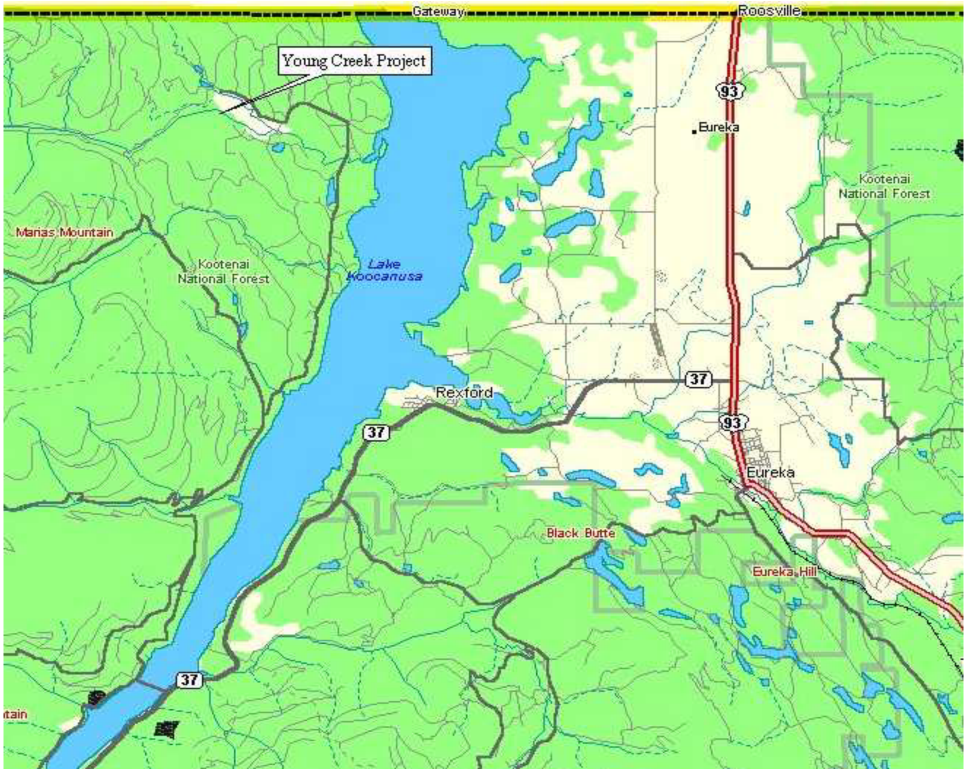


Figure 8. General location map of the Young Creek State Lands Restoration Project area.

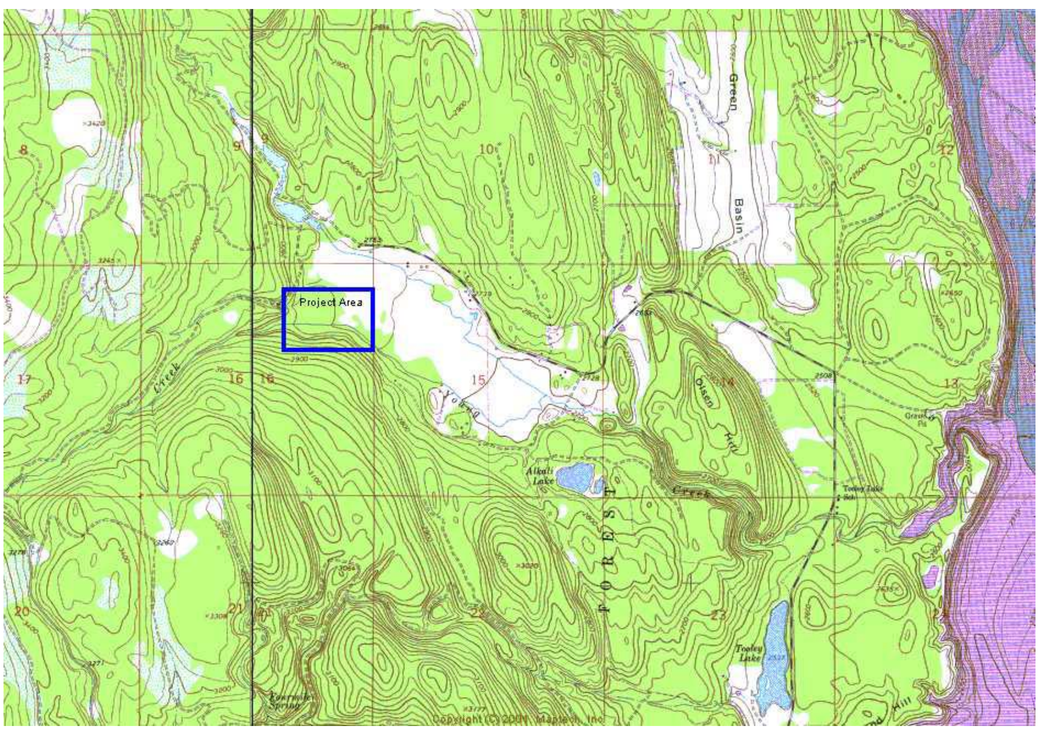


Figure 9. Detailed location map of the Young Creek State Lands Restoration Project area.

Table 7. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio measured for 4 riffles in the existing stream channel and 10 riffles in the stream channel shortly after construction in the fall of 2003 (as built) for the Young Creek State Lands Stream Restoration Project. Variance estimates for annual mean values are presented in parentheses. Mean values between treatments were compared using a t-test, and the one tailed p-value for each comparison is presented. The percent change for each parameter between treatments is also presented.

Riffle Cross Sections	**Cross Sectional Area (ft^2)**	**Mean Bankfull Width (ft)**	**Mean Bankfull Depth (ft)**	**Maximum Bankfull Depth (ft)**	**Width to Depth Ratio**
Existing	16.75 (1.58)	27.88 (22.73)	0.60 (0.008)	1.05 (0.017)	48.3 (239.6)
As Built (2003)	21.99 (10.07)	16.3 (9.18)	1.24 (0.05)	1.985 (0.09)	13.7 (21.2)
Percent Change	31.3%	-41.5%	107.5%	89.0%	-71.6%
t-test P Value	0.004	$6.6*10^{-5}$	$7.5*10^{-5}$	$3.6*10^{-5}$	$1.1*10^{-5}$

Table 8. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio measured for 2 pools in the existing stream channel and 8 pools in the stream channel shortly after construction in the fall of 2003 (as built) for the Young Creek State Lands Stream Restoration Project. Variance estimates for annual mean values are presented in parentheses. Mean values between treatments were compared using a t-test, and the one tailed p-value for each comparison is presented. The percent change for each parameter between treatments is also presented.

Pool Cross Sections	**Cross Sectional Area (ft^2)**	**Mean Bankfull Width (ft)**	**Mean Bankfull Depth (ft)**	**Maximum Bankfull Depth (ft)**	**Width to Depth Ratio**
Existing	19.25 (3.13)	23.5 (24.5)	0.79 (0.005)	2.35 (0.13)	30.14 (80.15)
As Built (2003)	37.68 (65.09)	21.8 (18.0)	1.73 (0.084)	3.23 (0.42)	12.99 (12.78)
Percent Change	95.7%	-7.18%	119.0%	37.77%	-56.9%
t-test P Value	0.0076	0.318	0.0012	0.053	0.0008

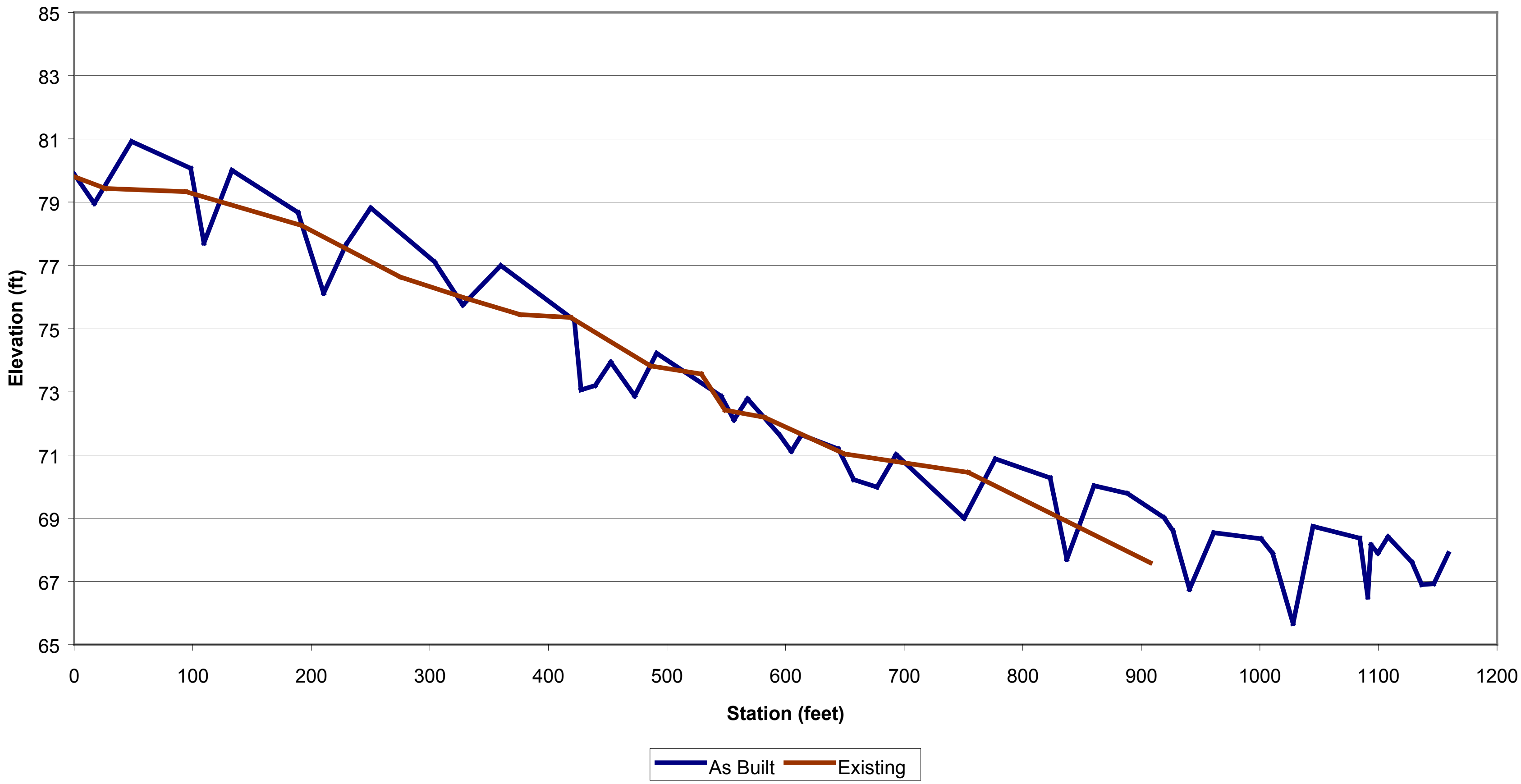


Figure 10. The longitudinal profile of the reach of Young Creek located within the State Lands Restoration Project. The survey was completed before (2002; existing) and after (2003; post-construction) project completion. The station (longitudinal location measured at the channel thalweg) begins at the upstream boundary of the project. The as built stream channel was approximately 250 feet longer due to the increased meander frequency that resulted from constructing the new stream channel.

Discussion

The Grave Creek Phase I and Libby Creek Upper Cleveland Restoration Projects maintained the designed channel dimensions after each stream experienced the spring freshet after project construction. Streams with C3 channel types should have width/depth ratios >12 and typically range from 10-37 (Rosgen 1996). Both the Libby and Grave Creek restoration projects were designed within this criterion and continued to meet it after the channel forming flows that occurred during the first spring freshet following project construction. The stream channel dimensions measured at the cross-section surveys on both the Libby Creek Upper Cleveland and Grave Creek Phase I Restoration Projects did not significantly ($p > 0.05$) change after the first spring freshet. Mean bankfull width, depth, maximum depth, width to depth ratio and cross sectional area all remained similar between years. Pool dimensions within both projects were similar between years. However, mean and maximum depth decreased after each project was subjected to bankfull channel shaping flows during the spring freshets. Although we did observe a decrease in pool depths after project construction, pool depth, quantity and quality still exceeded conditions that existed prior to project construction. We believe the observed changes in the pools that occurred as a result of the spring freshet on both Libby and Grave Creeks had relatively minor effects on the quantity and quality of juvenile salmonid rearing habitat, and that overall the stream channels are existing at a state near dynamic equilibrium. In addition to the minor changes in pool depth, we did observe an overall decrease in riffle slope after the spring freshets that were common for both the Grave and Libby Creek projects. The changes were primarily associated with the cobble gradient control structures that were placed in the pool tail out areas. These were the first two stream restoration projects that the cobble gradient control structures were utilized, and help point out that the science of natural channel design is still evolving, with improvements in structure design occurring in this relatively new science. We believe that these particular structures can serve a useful purpose, but that design modification could improve their function. We currently recommend designing these structures between 1.25 to 1.5 times the mean bankfull riffle width. We further recommend the run width below the cobble gradient control structures be designed at 0.8 to 1.0 times the mean bankfull riffle width. The cobble gradient control structures were constructed with material that was between D90 to D100 sized substrate within each particular stream. We will continue to monitor the performance of the cobble gradient control structures and modify their design criteria accordingly.

We monitored the benthic macroinvertebrate and fish communities at the Libby Creek Upper Cleveland Project in order to evaluate the ecological response the restoration activities (see Chapter 1). Our results were somewhat counter intuitive. Of the seven metrics selected before the study, 3 had values in 2003 that were significantly different from the values attained from the 2000 sampling. However, they were all in the opposite direction expected. That is, we expected the samples to reflect an improvement after restoration, but the directions of change for the three significantly different metrics were consistent with disturbance. Similarly, redband trout abundance decreased by an average of 33.5% the first year after project completion. However, bull trout abundance within the project area increased by 80% compared to data collected three years prior to project initiation. We did not monitor fish abundance within the Grave Creek Phase I Restoration Project area. However, we did monitor the benthic macroinvertebrate community within the project area, but none of the seven metrics selected *a priori* differed significantly approximately 1 year after the project was completed. Although we expected the macroinvertebrates to respond relatively quickly to the changes in the physical environment, our monitoring in Grave and Libby creeks either did not change or shifted in a direction that was more consistent with a disturbed environment. We attribute these changes to the disturbance activities and abundance of fine sediment within the project areas that was present from construction in the fall to the spring freshet. We believe that the first year following the initial construction period is the period of time that channel stability is most vulnerable. Much of the bed material can potentially become mobile until the pavement layer establishes following the initial bankfull discharge in the stream after project construction (D. Rosgen, personal communication). We will continue to monitor macroinvertebrates at both the Grave and Libby creek project to determine if these trends reverse as channel disturbances recover over time. Although we did monitor fish abundance within the Libby Creek project area, we did not expect to be able to detect a response from the fish community within the first year after project construction due to the longer lifecycle of fish compared to macroinvertebrates. We will continue to annually monitor the fish community at this site in order to assess the long-term response to the restoration activities.

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Appendix

Table A1. Therriault Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis.

Year	1997	1998	1999	2000[B]	2001	2002	2003
Section 1							
Rainbow Trout	123 (260.84)	130 (150.91)	82 (89.15)	-----	-----	-----	56 (57)
Brook Trout	41 (46.52)	49 (56.27)	60 (63.67)	-----	-----	-----	59 (66)
Total Population[A]	149 (213.70)	182 (206.89)	141 (149.12)	-----	-----	-----	
Section 2							
Rainbow Trout	36 (41.36)	79 (81.62)	76 (83.34)	-----	93 (101.99)	-----	84 (n/a)
Brook Trout	56 (57.53)	125 (136.96)	72 (80.47)	-----	82 (87.34)	-----	58 (61)
Bull Trout	47 (48.87)	15 (16.42)	3	-----	2	-----	40 (42)
Total Population[A]	92 (95.90)	205 (216.88)	149 (162.50)	-----	180 (192.55)	-----	
Section 3							
Rainbow Trout	54 (58.1)	164 (169.82)	177 (205.30)	-----		-----	99 (104)
Brook Trout	74 (76.7)	82 (87.79)	110 (116.71)	-----		-----	67 (72)
Bull Trout	0	0	0				10 (n/a)
Total Population[A]	66 (92.68)	248 (256.53)	284 (307.71)	-----		-----	

A) Includes rainbow, rainbow x cutthroat hybrids, and brook trout. Bull trout were not included in the total population estimate.
B) Therriault Creek was not sampled during the 2000 or 2002 field seasons.

Table A2. Lower Grave Creek Demonstration Project area electrofishing. Numbers are total catch within the 1,000 foot section.

Year	2000[A]	2001[B]	2002[C]	2003
Westslope Cutthroat	4	18	3	13 (n/a)
Rainbow Trout	1	17	26	24.5 (28.7)
Brook Trout	1	10	5	8.5 (18.2)
Bull Trout	9	33	5	40.5 (144.4)
Mountain Whitefish	54	3	33	21 (21.4)
Long Nose Dace	6	-----	-----	-----
Water Temp. ^{0}C	-----	17	-----	-----
Effort (minutes)	44	56.9	NA	NA

A) Four bull trout ≥ 490 mm were likely lacustrine - adfluvial fish from Libby Reservoir moving into Grave Creek to spawn. Three bull trout < 75 mm were also included in the total.
B) Four bull trout ≥ 470 mm were likely lacustrine - adfluvial fish from Libby Reservoir moving into Grave Creek to spawn. Long nose dace were observed but not counted in 2001.
C) Due to the presence of approximately 2,000 mature kokanee, the section was snorkeled rather than electrofished. Two adult bull trout were observed that were likely lacustrine - adfluvial fish from Libby Reservoir moving into Grave Creek to spawn. Long nose dace were observed but not counted.

Table A3. Young Creek depletion population estimates for fish $\geq$ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis.

Year	1996	1997	1998	1999	2000	2001	2002	2003
Section 1 (Tooley)								
Westslope Cutthroat[B]	-----	3	36 (37.05)	139 (147.55)	-----	55 (64.28)	88 (95.53)	Not sampled
Rainbow Trout[B]	-----	19 (22.37)	62 (69.51)	3	-----	2	14 (18.64)	
Brook Trout	-----	11 (17.18)	120 (124.02)	102 (105.00)	-----	36 (38.75)	30 (31.18)	
Mountain Whitefish	-----	-----	-----	-----	-----	-----	2	
Total Population [A]	12 (13.33)	36 (40.19)	220 (227.99)	248 (257.80)	-----	96 (107.23)	148 (157.82)	
Section 3 (303 A Rd.)								
Westslope Cutthroat	-----	234 (246)	416 (451.97)	314 (336.40)	-----	-----	-----	Not sampled
Rainbow Trout	-----	-----	-----	-----	-----	-----	-----	
Brook Trout	-----	-----	-----	1	-----	-----	-----	
Total Population [A]	-----	234 (246)	416 (451.97)	316 (338.29)	-----	-----	-----	
Section 4 (303 Rd.)								
Westslope Cutthroat	155 (228.67)	100 (113.50)	439 (500.27)	352 (367.35)	-----	130 (141.76)	222 (236.78)	Not sampled
Rainbow Trout	-----	-----	-----	-----	-----	-----	-----	
Brook Trout	-----	-----	-----	3	-----	6 (12.41)	4	
Total Population [A]	155 (228.67)	100 (113.50)	439 (500.27)	358 (373.17)	-----	136 (148.11)	232 (248.77)	
Section 5 (State)								
Westslope Cutthroat	-----	-----	216 (226.81)	256 (290.16)	126 (152.62)	153 (174.11)	268 (289.94)	178.0 (182.9)
Rainbow Trout	-----	-----	-----	-----	-----	-----	-----	-----
Brook Trout	-----	-----	62 (70.63)	52 (65.33)	19 (21.86)	25 (27.08)	46 (48.81)	34.9 (n/a)
Bull Trout	-----	-----	-----	-----	-----	-----	2	0
Total Population [A]	-----	-----	280 (294.47)	314 (352.96)	113 (119.14)	176 (194.79)	315 (335.15)	

A) Includes rainbow, rainbow x cutthroat hybrids, westslope cutthroat, and brook trout. Bull trout were not included in the total population estimate.
B) Sampling crew did not distinguish between westslope cutthroat trout and rainbow trout.

Table A4. Libby Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis.

Year	1998	1999[A]	2000[A]	2001	2002	2003
Section 1 – below Hwy 2						
Rainbow Trout	81 (126.80)	26	125	46 (51.09)	117 (129.56)	84 (95.5)
Brook Trout	6 (8.27)	6	13	10 (12.33)	16 (24.29)	5
Bull Trout	-----	-----	-----	-----	3	0
Mountain Whitefish	-----	-----	-----	-----	3	1
Total Population [B]	90 (115.89)	32	138	57 (63.79)	138 (152.67)	
Water Temp. ^{0}C	9	-----	16	15	14	------
Discharge (cfs)	6.9	-----	-----	-----	-----	------
Section 2 –above Hwy 2						
Rainbow Trout	203 (225.20)	-----	-----	148 (192.77)	-----	100 (107.5)
Brook Trout	7	-----	-----	2	-----	2
Bull Trout	5 (6.26)	-----	-----	-----	-----	2.08
Total Population [B]	208 (228.39)	-----	-----	160 (213.40)	-----	
Water Temp. ^{0}C	5	-----	-----	20	-----	------
Discharge (cfs)	6.9	-----	-----	-----	-----	------
Section 3 – upper Cleveland						
Rainbow Trout	-----	-----	170 (193.73)	172 (182.26)	163 (183.16)	112.3 (126.9)
Brook Trout	-----	-----	-----	-----	-----	------
Bull Trout	-----	-----	3	8 (11.15)	7	10.8 (13.7)
Mountain Whitefish	-----	-----	-----	-----	1	-----
Total Population [B]	-----	-----	170 (193.73)	172 (182.26)	163 (183.16)	

A) Section 1 population estimates in 1999 and 2000 were single pass catch–per-unit-effort estimates due to high escapement rates. Actual population is higher than reported.
B). Includes rainbow, rainbow x cutthroat hybrids, and brook trout. Bull trout were not included in the total population estimate.

Table A5. Parmenter Creek (prior to and following channel reconstruction) depletion population estimate for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals near the Dome Mountain Road Bridge. Upper confidence intervals are in parenthesis.

Year	2000	2001	2003
	Pre-reconstruction	Post-reconstruction	Post-reconstruction
Rainbow Trout	92 (110.65)	79 (95.9)	110 (124.2)
Brook Trout	18 (19.20)	1	2
Bull Trout	-----	1	5.6
Total Population [A]	108 (122.56)	81 (97.73)	
Water Temp. ^{0}C	14.4	-----	-----

A). Includes rainbow, rainbow x cutthroat hybrids, and brook trout. Bull trout were not included in the total population estimate.

Table A6. Pipe Creek depletion population estimate for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals surveyed directly downstream of the Bothman Road Bridge. Upper confidence intervals are in parenthesis.

Year	2001	2002[B]	2003
Rainbow Trout	42 (46.42)	73 (84.97)	39.1 (42.6)
Brook Trout	-----	3	6.5 (7.6)
Bull Trout	-----	-----	------
Total Population [A]	42 (46.42)	73 (84.97)	
Water Temp. ^{0}C	18	17	------

A). Includes rainbow, rainbow x cutthroat hybrids, and brook trout. Bull trout were not included in the total population estimate.

B). Also captured were 43 mountain whitefish ranging from 51 to 105 millimeters and one pumpkinseed sunfish 74 millimeters in length.

Table A7. Mean zooplankton densities (no./l) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Tenmile area of Libby Reservoir during 2002. *Epischura* and *Leptodora* were measured as number per m^3.

Month	**N)**	***Daphnia***	***Bosmina***	***Diaptomus***	***Cyclops***	***Leptodora***	***Epischura***	***Diaphanosoma***
April	(3)	0.07	0.07	0.07	3.54	0.00	0.00	0.00
		0.00	0.01	0.00	0.27	0.00	0.00	0.00
May	(3)	0.29	0.19	0.17	5.63	0.71	9.43	0.01
		0.06	0.00	0.02	1.12	1.50	74.75	0.00
June	(2)	2.97	4.60	0.02	15.91	7.08	55.17	0.02
		0.99	0.99	0.00	0.13	3.98	6,087.46	0.00
July	(3)	9.00	1.12	0.28	21.45	40.08	114.87	0.15
		20.22	1.17	0.00	208.63	316.92	7,558.30	0.02
August	(3)	2.19	0.03	0.61	19.80	0.00	164.10	1.25
		0.38	0.00	0.10	6.23	0.00	26,032.04	1.35
September	(3)	0.51	0.52	0.55	9.16	0.00	203.33	1.90
		0.09	0.09	0.09	0.91	0.00	13,002.34	0.17
November	(3)	0.61	1.96	0.42	4.18	0.00	11.32	0.16
		0.04	2.74	0.01	1.29	0.00	384.20	0.01

Table A8. Mean zooplankton densities (no./l) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Tenmile area of Libby Reservoir during 2001. *Epischura* and *Leptodora* were measured as number per m^3.

Month	(N)	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
April	(3)	0.06	0.06	0.05	1.12	0.00	0.00	0.00
		0.00	0.00	0.00	0.09	0.00	0.00	0.00
May	(3)	0.05	0.03	0.08	2.07	0.71	78.56	0.00
		0.00	0.00	0.00	2.81	1.50	5,577.11	0.00
June	(3)	1.08	0.06	0.25	24.71	9.67	442.31	0.03
		0.11	0.01	0.04	154.97	100.23	44,669.05	0.00
July	(3)	4.24	1.03	0.17	11.75	4.48	36.31	0.06
		5.71	0.16	0.03	26.61	7.19	1,673.50	0.00
August	(3)	1.21	1.01	0.44	9.14	1.18	67.90	0.09
		0.10	0.22	0.04	14.78	1.16	1,152.60	0.01
September	(3)	1.33	1.63	1.44	12.97	1.41	122.51	0.80
		0.05	3.40	0.14	8.61	2.00	3,634.06	0.05
October	(3)	1.19	0.13	1.35	3.75	0.00	11.32	0.15
		0.17	0.00	0.02	1.12	0.00	384.20	0.01
November	(3)	0.99	0.13	0.49	2.78	0.00	63.47	0.15
		0.16	0.01	0.02	1.81	0.00	66.13	0.02

Table A9. Mean zooplankton densities (no./l) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Rexford area of Libby Reservoir during 2002. *Epischura* and *Leptodora* were measured as number per m^3.

Month	(N)	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
April	(3)	0.40	0.10	0.17	8.22	0.00	1.32	0.00
		0.24	0.01	0.03	133.56	0.00	5.23	0.00
May	(3)	0.37	0.05	0.19	3.35	0.79	68.90	0.05
		0.17	0.00	0.06	13.53	0.69	6,207.85	0.00
July	(3)	6.55	0.54	0.21	17.29	28.29	165.92	0.25
		4.05	0.23	0.04	6.68	200.08	11,958.62	0.01
August	(3)	1.58	0.01	0.63	15.86	0.00	155.23	2.38
		0.25	0.00	0.04	15.51	0.00	2,444.41	0.35
September	(3)	0.62	0.57	0.83	11.75	0.00	108.08	5.40
		0.11	0.18	0.02	24.85	0.00	3,231.45	1.79
November	(3)	1.23	2.24	0.50	5.53	0.00	16.98	0.14
		0.06	0.45	0.00	4.62	0.00	864.62	0.01

Table A10. Mean zooplankton densities (no./l) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Rexford area of Libby Reservoir during 2001. *Epischura* and *Leptodora* were measured as number per m^3.

Month	(N)	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
April	(3)	0.20	0.14	0.14	4.25	0.00	0.00	0.00
		0.01	0.01	0.00	2.56	0.00	0.00	0.00
May	(3)	0.43	0.06	0.42	21.98	1.41	107.04	0.02
		0.03	0.00	0.17	169.03	0.50	973.27	0.00
June	(3)	2.39	1.01	0.16	18.46	12.26	52.81	0.00
		0.46	0.42	0.05	153.05	23.16	8,367.74	0.00
July	(3)	2.95	0.22	0.22	10.67	12.02	20.65	0.06
		1.38	0.03	0.00	12.94	18.02	1,279.68	0.00
August	(3)	3.52	0.24	0.50	12.02	1.20	0.00	0.38
		1.02	0.04	0.03	27.39	1.19	0.00	0.04
September	(3)	1.46	1.29	0.89	9.50	0.71	178.06	1.16
		0.18	1.61	0.11	0.62	0.50	2,723.39	0.06
October	(3)	0.75	0.29	0.95	6.40	0.00	0.00	0.32
		0.09	0.03	0.18	0.98	0.00	0.00	0.03
November	(3)	0.70	0.10	0.41	2.77	0.00	63.56	0.11
		0.08	0.01	0.01	0.39	0.00	1,794.23	0.01

Table A11. Mean zooplankton densities (no./l) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Canada area of Libby Reservoir during 2002. *Epischura* and *Leptodora* were measured as number per m^3.

Month	(N)	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
April	(3)	0.01	0.01	0.01	0.10	0.00	1.73	0.01
		0.00	0.00	0.00	0.01	0.00	3.14	0.00
May	(3)	0.22	0.17	0.13	1.53	1.89	67.53	0.08
		0.07	0.02	0.03	2.86	10.68	1,931.37	0.01
July	(3)	2.15	0.02	0.06	2.82	12.73	11.60	0.10
		0.75	0.00	0.00	3.89	54.02	72.30	0.01
August	(3)	2.41	0.02	0.40	10.95	1.90	31.12	0.78
		1.03	0.00	0.00	25.49	7.29	2,905.99	0.22
September	(3)	0.60	0.15	0.77	10.86	0.00	191.76	6.12
		0.10	0.01	0.26	3.35	0.00	17,461.88	5.52
November	(3)	3.15	1.37	1.36	3.32	0.00	95.18	0.09
		11.95	1.43	2.27	4.24	0.00	16,299.22	0.01

Table A12. Mean zooplankton densities (no./l) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Canada area of Libby Reservoir during 2001. *Epischura* and *Leptodora* were measured as number per m^3.

Month	(N)	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
April	(3)	0.11	0.06	0.04	4.02	0.00	0.00	0.00
		0.00	0.00	0.00	4.37	0.00	0.00	0.00
May	(3)	0.23	0.50	0.02	10.02	0.24	62.24	0.01
		0.08	0.17	0.00	88.93	0.17	3,105.76	0.00
June	(3)	1.36	1.41	0.00	7.88	6.05	0.39	0.00
		1.21	1.30	0.00	38.31	22.31	0.46	0.00
July	(3)	3.28	0.05	0.10	4.78	7.55	6.22	0.04
		3.12	0.00	0.00	3.92	33.20	116.19	0.00
August	(3)	2.33	0.23	0.41	2.89	2.93	61.40	0.31
		0.56	0.02	0.09	3.09	8.00	1,610.80	0.09
September	(3)	5.47	0.56	0.50	3.77	3.12	102.47	0.72
		12.65	0.36	0.03	2.51	5.79	16,696.95	0.80
October	(3)	0.91	0.51	0.93	5.36	0.24	0.00	0.72
		0.06	0.02	0.08	3.21	0.17	0.00	0.07
November	(3)	1.59	0.24	1.00	8.19	0.00	52.15	0.12
		0.87	0.03	0.45	42.09	0.00	984.09	0.01

Table A13. Yearly mean total zooplankton densities (no./l) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in Libby Reservoir. *Epischura* and *Leptodora* were measured as number per m^3.

Year	(N)	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
1997	69	2.80	0.07	0.80	6.10	4.34	57.24	0.08
		11.30	0.01	0.88	50.87	108.72	6,013.80	0.02
1998	72	2.17	0.64	2.22	9.35	3.99	131.58	0.36
		4.00	1.80	9.17	64.33	80.92	47,113.37	0.43
1999	57	2.19	0.77	0.51	9.57	6.63	89.41	0.15
		4.53	1.39	2.35	107.88	148.11	14,367.63	0.05
2000	69	1.07	0.51	0.36	8.04	2.72	51.20	0.05
		0.97	1.06	0.20	80.04	14.05	7,153.52	0.01
2001	72	1.58	0.46	0.46	8.39	2.72	63.72	0.22
		2.77	0.46	0.21	59.53	21.18	11,153.71	0.13
2002	56	1.82	0.65	0.39	8.89	4.88	77.96	1.02
		6.85	1.29	0.22	57.44	139.73	9,041.90	3.62
Mean		**1.94**	**0.52**	**0.79**	**8.39**	**4.21**	**78.52**	**0.31**