Mitigation for the Construction and Operation of Libby Dam

Libby Mitigation

Annual Report
2001 - 2002

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MITIGATION FOR THE CONSTRUCTION AND OPERATION OF LIBBY DAM

ANNUAL REPORT
2001-2002

By:

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EXECUTIVE SUMMARY

"Mitigation for the Construction and Operation of Libby Dam" is part of the Northwest Power Planning Council's 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.

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 during 2001 and 2002.

- 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. However, collectively bull trout redd numbers have significantly increased over the past 7-8 year period.

- We surgically implanted radio tags into 65 adult bull trout from late January 1998 to early December 2000 to assess the movement, behavior, and spawning distribution of bull trout in the Kootenai River. We had a relatively high success rate of tracking radio tagged bull trout after release, accounting for approximately 88% of all tagged fish at least once after initial release, and were observed an average of 30.7 observations per fish. We identified several common locations that adult radio tagged bull trout frequented in the Kootenai River above Kootenai Falls including the Libby Dam tailrace area that extends from Libby Dam to approximately 2 miles downstream to confluence of Alexander Creek.

- Montana FWP has monitored the relative abundance of burbot in the stilling basin below Libby Dam since 1994 using baited hoop traps. Burbot catch in during the 01-02 and 02-03 trapping seasons represented the lowest catch rates (fish per trap day) since trapping at this location was initiated in the 94-95 season.

- We sampled macro-invertebrates at the Libby Creek Demonstration Project and the Grave Creek Phase I Project in order to assess the benthic community response to these two restoration projects. Results from the Libby Creek site include pre and post implementation results, with 4 of the 6 indices of diversity increasing after project implementation.

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

- Montana FWP has documented the changes in species composition, and species size and abundance within Koocanusa 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.

- 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 and Park, 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 (Montana Fish, Wildlife and Parks et al. 1998). A mix of mitigation techniques is necessary to offset losses caused by dam construction and operation. During the past two years, Montana Fish, Wildlife and Parks 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 city of Troy, Montana and Lincoln County to construct community-fishing ponds at the Troy Recreation Park and Lincoln County Fairgrounds. These projects 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 Libby Creek and Grave Creek as high priority streams for restoration activities based on habitat quality, fish community composition, and native fish abundance, Montana FWP adopted a phased restoration approach for both streams, with the initial phases of restoration on both streams targeting the elimination of some of the largest supplies of bedload sediment. Restoration activities on both streams were first implemented in the fall of 2001.

- Montana FWP continued at the watershed level during 2002 with the implementation of a phased restoration approach with the construction of the Libby Creek Cleveland Project and the Grave Creek Phase I projects during the fall of 2002. These projects effectively changed the stream channel pattern profile and dimension. These changes resulted in a narrower, deeper stream channels that are likely to improve the quantity and quality of rearing habitat for native salmonids.

- A rigorous monitoring program for all stream restoration projects includes pre- and post-construction monitoring that allows comparisons to describe changes in the physical environment as a result of these restoration projects.

Young Creek is one of the most important westslope cutthroat trout spawning tributaries to Koocanusa Reservoir because it represents one of the last know genetically pure populations of westslope cutthroat trout in the region and it is also one of the most potentially productive tributary streams to Koocanusa Reservoir. Westslope cutthroat thrived in Koocanusa Reservoir from the early 1970s through the early 1980s, adfluvial runs of cutthroat in Young Creek were abundant during this period. However since then the abundance of adfluvial cutthroat trout in the reservoir and Young Creek has declined due likely to a combination of factors. Montana FWP conducted a pilot study from 1996-2000 that utilized remote site incubators (RSIs) in Young Creek in an effort to increase the abundance of adfluvial and resident westslope cutthroat in Young Creek. Westslope cutthroat trout eggs were obtained from Washoe Park State Fish Hatchery in Anaconda, Montana. The following points summarize the results of this study to date.

- The RSIs produced approximately 57,000 – 89,700 cutthroat trout fry from 1997-2000, with egg-to-fry survival ranging from 53-75%.

- Montana FWP operated the Young Creek fish trap in 1998 to monitor juvenile recruitment and adult escapement from Young Creek and Koocanusa Reservoir respectively, in order to assess the success of the RSI project. We randomly collected otolith samples from juvenile and adult cutthroat trout from 1998-2002 at the Young Creek fish trap,

- Attempts to thermally mark the otoliths of embryo cutthroat trout at the Anaconda Hatchery in order to differentiate between hatchery and natural origin fish were only successful during the 1999 brood year.

- This project will investigate an alternative method that uses trace elemental differences between the Anaconda Hatchery and Young Creek to differentiate between hatchery and natural origin fish collected at the trap in an attempt differentiate between of hatchery and natural origin cutthroat trout collected at the Young Creek trap, and ultimate evaluate the effectiveness of the RSIs.

Spill at Libby Dam has been an infrequent event since the fourth turbine unit went online in 1976. As a result of infrequent spill, subsequent information regarding the gas exchange processes, particularly dissolved gas production from spill releases and dissolved gas dissipation downstream from the project are limited. Additional knowledge related to gas production dynamics in the Kootenai River below Libby Dam could help water managers make critical decisions during events that require spill. Therefore the U.S. Army Corps of Engineers proposed to conduct a comprehensive test of total dissolved gas resulting from a range of releases at Libby Dam during June 2002 that were designed to systematically vary the spillway flow over time while monitoring downstream water quality and fish. However, warm weather and high inflows into a nearly full reservoir required forced spill at Libby Dam beginning on June 25 and lasting 13 days until July 7, and then commencing again for another 7 days from July 11 to July 17. Fish monitoring during the spill activities at Libby Dam in the summer of 2002 used three general approaches including the examination of captive fish and fish captured via electrofishing for signs of gas bubble disease, and radio telemetry to assess fish displacement and behavior changes. The following points summarize the results associated with the fish monitoring activities during the spill event at Libby Dam during the summer of 2002.

- Signs of gas bubble disease developed rapidly in the captive fish, and quickly escalated to 100% incidence, relative to fish captured via nighttime electrofishing.

- Approximately 86% of the rainbow trout 80% of the bull trout *Salvelinus confluentus* and 31% of the mountain whitefish collected via electrofishing during the peak total discharge and spill at Libby Dam exhibited signs of gas bubble disease.

- Results from the radio telemetry work suggests that most radio tagged rainbow trout (n= 7; 100%), bull trout (n = 3; 75%) and mountain whitefish (n = 2; 67%) did not move substantially during the spill activities at Libby Dam, and remained within the general vicinity of Libby Dam (RM 221.7) downstream to Dunn Creek (RM 219.8), with the center of gravity more near Libby Dam.

- Spill activities at Libby Dam during the summer of 2002 created relatively rapid response of total dissolved gas concentrations with relatively small amounts of spill water, and impacted resident fish of the Kootenai River below the dam. Therefore, the use of spill as a regular management activity at Libby Dam appears to have limited practical application under the current dam configuration.

ACKNOWLEDGEMENTS

We are thankful to the many people that substantially contributed to this project. Neil Benson, Monty Benner, Kimber Hrach, and Josh Smith helped with field data collection, data entry and proofing, and lab processing. Brian Marotz was the project leader and provided thoughtful review of this report. Hatchery staff from Washoe Park State Fish Hatchery in Anaconda, Montana provided westslope cutthroat trout eggs for the remote site incubator studies in Young Creek, which made that research possible. Many of the aquatic habitat restoration projects that were accomplishments and summarized in this report were cooperative efforts between Montana Fish, Wildlife and Parks and other agencies and organizations. The Glen Lake Irrigation Diversion Project was accomplished by funding from several cooperating entities including the USFWS "Partners for Wildlife", Glen Lake Irrigation District, and the USFS. Lincoln County assisted with the design and construction of the Eureka Community Fishing pond. The City of Troy contributed to the Troy Water Works Project and the Troy Community fishing pond. This work was funded by the Bonneville Power Administration, and 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 redevelop fisheries and fisheries habitat in basin streams and lakes.

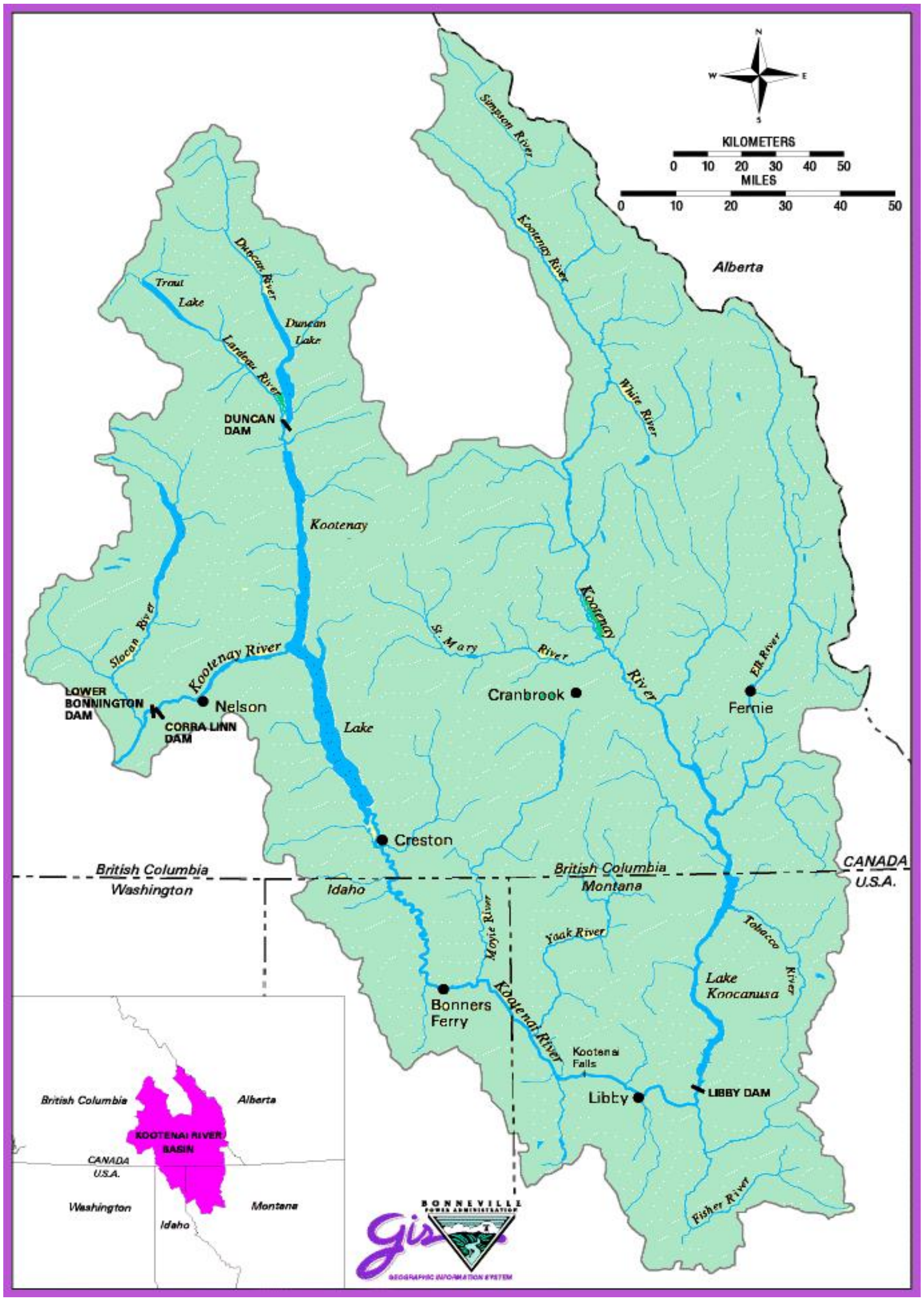

Figure 1. Kootenai River Basin (Montana, Idaho and British Columbia, Canada).

PROJECT HISTORY

Work on Libby Reservoir to assess the effects of operation on fish populations and lower trophic levels began in 1982. This project established relationship between reservoir operation and biological productivity, and incorporated the results in the computer 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 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 salmon/steelhead biological opinion, the strategy was adopted by the White Sturgeon Recovery Team in their Kootenai white sturgeon recovery plan (USFWS 1999).

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 similarly 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 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 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.

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

1997 - We 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 - We 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 - We developed an isolation facility for the conservation of redband rainbow 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 - We 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 - We 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 - We 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 - We were 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 - We 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 - We 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.

ASSOCIATIONS

The primary goals of our 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). 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 can be utilized by 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 Koocanusa 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 Location

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. In terms of runoff volume, the Kootenai is the second largest Columbia River tributary. In terms of watershed area (36,000 km^2 or 8.96 million acres), it ranks third (Knudson 1994).

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.

Climate

The subbasin has a relatively moist climate, with annual precipitation even at low elevations generally exceeding 20 inches. Warm, wet air masses from the Pacific bring abundant rain and 1,000 to 7,500 mm (40 to 300 inches) of snowfall each year. In winter, Pacific air masses dominate and produce inland mountain climates that are not extremely cold, although subzero continental-polar air occasionally settles over the mountains of northern Idaho and vicinity.

The Continental Divide Range, with crest elevations of 10,000 to 11,500 feet along nearly 250 km (155 miles) of ridgeline, is a major water source for the river. The range receives 2,000 to 3,000 mm (80 to 120 inches) of precipitation annually (Bonde 1987). Some of the high elevation country in the Purcell Range around Mt. Findlay receives 2,000 mm (80 inches) of precipitation a year; but most of the range, and most of the Selkirk and Cabinets, get only 1,000 to 1,500 mm (40 to 60 inches) annually (Daley et al. 1981). In the inhabited valley bottoms, annual precipitation varies from just under 500 mm (20 inches) at Rexford, Montana (USACE 1974) and Creston, British Columbia (Daley et al. 1981) to just over 1,000 mm (40 inches) at Fernie, British Columbia (Oliver 1979).

Topography

The drainage basin is located within the Northern Rocky Mountain physiographic province, which is characterized by north to northwest trending mountain ranges separated by straight valleys that run parallel to the ranges.

The topography of the Kootenai River subbasin is dominated by steep, heavily forested mountain canyons and valleys. Consequently, nearly all of the major tributaries to the river, including the Elk, Bull, White, Lussier, and Verrnillion Rivers have a very high channel gradient, particularly in their headwaters. In contrast, the mainstem of the Kootenai has a fairly low channel gradient after entering the Rocky Mountain Trench near Canal Flats. The river drops less than 1,000 feet (305 meters) in elevation from Canal Flats to Kootenay Lake, a distance of over 300 miles (480 km). However, even along the river's slow meandering course, valley-bottom widths are generally less than two miles and are characterized by tree-covered rolling hills with few grassland openings. The only exceptions to this topography are the slightly wider valley bottoms in the Bonners Ferry-to-Creston area and the Tobacco Plains, located between Eureka, Montana and Grasmere, British Columbia.

Synder and Minshall (l996) identified three different geomorphic reaches of the Kootenai River between Libby Dam and Kootenay Lake. The first reach (Canyon) extends from Libby Dam to the Moyie River (92 km). It flows through a canyon in places, but otherwise has a limited flood plain due to the closeness of the mountains. The substrate consists of large cobble and gravel. The second reach (Braided) extends from the Moyie River to the town of Bonners Ferry (7.5 km). It is extensively braided with depths that are typically less than 9 m, and substrates that consist mostly of gravels. The river has an average gradient of 0.6 m/km, and velocities higher than 0.8 m/s. The third reach (Meander) extends from just below the town of Bonners Ferry to the confluence of the Kootenay Lake (82.5 km). Here, the river slows to an average gradient of 0.02 m/km, deepens, and meanders through the Kootenai Valley back into British Columbia and into the southern arm of Kootenay Lake. The meandering section through the Kootenai Valley is characterized by water depths of up to 12 meters in runs and up to 30 meters in pools (Snyder and Minshall 1994). This reach has been extensively diked and channelized, which has had profound effects on ecosystem processes.

Geology

Mountains in the subbasin are composed of folded, faulted, and metamorphosed blocks of Precambrian sedimentary rocks of the Belt Series and minor basaltic intrusions (Ferreira et al. 1992). Primary rock types are meta-sedimentary argillites, sitlites, and quartzites, which are hard and resistant to erosion. Where exposed, they form steep canyon walls and confined stream reaches. The porous nature of the rock and glaciation and have profoundly influenced basin and channel morphology (Hauer et al. 1997).

The river character changes dramatically from a bedrock-controlled regime in Montana to a silt/clay regime near the town of Bonners Ferry, Idaho. During the Pleistocene, continental glaciation overrode most of the Purcell Range north of the river, leaving a mosaic of glacially scoured mountainsides, glacial till, and lake deposits. Late in the glacial period, an ice dam blocked the outlet at West Arm of Kootenay Lake. The dam formed glacial Kootenay Lake, the waters of which backed all the way to present-day Libby, Montana. Glacial Kootenay Lake filled the valley with lacustrine sediments, which included fine silts and glacial gravels and boulders. The Kootenai River and lower tributary reaches in Idaho are actively reworking these lacustrine sediments today. A terrace of lacustrine sediments on the east side of the valley is approximately 150 feet above the current floodplain and is a remnant of the ancestral valley floor. Tributary streams working through remnant deposits to meet the present base level of the mainstem and from the mainstem reworking existing floodplain and stream bank deposits continue to be a source of fine sediments. An extensive network of marshes, tributary side channels, and sloughs were formed by lowering of the lake level, flooding, and the river reworking its floodplain. Some of these wetlands continued to be supported by groundwater recharge, springtime flooding, and channel meandering. Much of this riverine topography however, has been eliminated by diking and agricultural development, especially in the reach downstream of Bonners Ferry, Idaho.

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).

Soils

Soils formed from residual and colluvial materials eroded from Belt rocks or in materials deposited by glaciers, lakes, streams, and wind. Wind deposits include volcanic ash from Cascade Range volcanoes in Washington and Oregon. In many areas, soils formed in glacial till and are generally loamy and with moderate to high quantities of boulders, cobbles, and gravels. Although soils within the mountainous regions vary widely in character, most mountain and foothill soils are on steep slopes and well drained, with large amounts of broken rock. Rock outcrops are common.

Soils deposited by glaciers or flowing water are, for the most part, deep, well-drained, and productive soils. Most of forest soils in the subbasin are somewhat resistant to erosion by water. In most of the valleys, soils are deep, relatively productive, and gently sloping.

Ustolls, Ochrepts, and Ustalfs are the dominant soils in valleys and on lower mountain slopes. Ochrepts, Borolls, and Orthents are dominant on upper mountain slopes and crests. Orthents and areas of rock outcrop are extensive on steep mountain slopes, and Fluvents and Aquolls are in valleys (NRCS 2000).

Land Use

The Kootenay Basin remains relatively remote and sparsely populated. Fewer than 100,000 people live within the basin upstream from Kootenay Lake, an area larger than the states of Maryland and Delaware combined. The largest municipal center is Cranbrook/Kimberley, which has a population of about 25,000. Only a handful of other communities have populations larger than 2,000. They include Libby, Montana, Bonners Ferry, Idaho, and Fernie, Sparwood, Elkford, and Creston, British Columbia.

The forest products industry remains the most dominant employment and most extensive development activity in the subbasin. Roughly 90 percent of the drainage is forested. Logging and associated road building has occurred in nearly all of the lower elevation valleys and on many higher elevation ridges. Roadless areas larger than 5,000 acres are uncommon. Nine roadless areas totaling 139,600 acres exist in the Idaho portion of the subbasin (IPNF 1991). In the Montana portion, nine roadless areas totaling 241,500 acres are present, including approximately 60,000 acres of upper Libby and Lake creeks within the Cabinet Mountains Wilderness Area (USDA 1987). The largest contiguous block of land without logging roads in the British Columbia portion of the Kootenay Basin is the 390,000-acre Kootenay/Mt. Assiniboine National and Provincial Parks (Rocchini 1981). Approximately 150,000 acres of the headwaters of the St. Mary River and Findlay Creek northwest of Cranbrook/Kimberley are within the Purcell Wilderness Conservancy. The total surface area of undeveloped areas amounts to about 10 percent of the Kootenai Subbasin above Kootenay Lake.

Coal and hard rock mining are prominent activities in the subbasin, particularly along the Elk and St. Mary rivers and in the northern Cabinet Mountains. Large-scale, open-pit coal mining began in the Elk River watershed in the early 1970s. Since the late 1930s, the Sullivan Mine at Kimberley, B.C. has been the largest metal producer in the basin. In 1981 it was one of the two largest lead-zinc mines in the world (Daley et al. 1981). From 1981 to the present, a large copper and silver mine and chemical floatation mill has operated in the Lake Creek watershed south of Troy, MT.

About two percent of the subbasin is agricultural land, much of it used for pasture and forage production (Bonde and Bush 1982). Agricultural development is confined primarily to narrow valley bottoms. Though it utilizes a relatively small area, it has had a large impact on habitats of the mainstem river and tributary mouths because most of the activity occurs in the floodplain. The largest contiguous block of agricultural land is within the Purcell Trench, which extends roughly from Bonners Ferry, Idaho to the river's entry into Kootenay Lake. Production of oats, wheat and barley account for 62 percent of the agricultural output in the Bonners Ferry/Creston area, with livestock production accounting for 20 percent. Hay and grass seed production and livestock grazing are the most common agricultural activities in the rest of the subbasin.

The two largest industrial operations and point-source discharges to the Kootenay River are the Crestbrook Forest Industries' pulp mill in Skookumchuck, B.C. and the Cominco mining, milling, and fertilizer plant in Kimberley, B.C. (Daley et al. 1981).

Another industrial operation in the basin was the mining and processing of vermiculite by the W.R. Grace Company northeast of Libby, Montana on Rainy Creek.

Fish Species

Eighteen species of fish are present in Koocanusa 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” rainbow trout fishery, and the state record fish was harvested there in 1997 (over 38 pounds). Bull trout *Salvelinus confluentus* are captured “incidentally”, and 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
White sturgeon	*Acipenser transmontanus*	R	D[Î]	Y[Ï]
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

[Î] Five white sturgeon were relocated from below Libby Dam to the reservoir. At least one of these fish moved upriver out of the reservoir while two have been accounted for from angler reports and one verified mortality.

[Ï] An abundance of anecdotal reports exist of white sturgeon above Kootenai Falls although research to date has failed to validate any reports.

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 at Libby Dam to allow for withdrawal of water from the reservoir's upper stratum.

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 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. A barren varial zone has been created by daily changes in water-flow and stage. Power operations cause rapid fluctuations in dam discharges (as great as 400 percent change in daily discharge), which are inconsistent with the normative river concept (ISAB 1997; ISAB 1997b). Flow fluctuations widen the riverine varial zone, which becomes 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 impacts due to erosion and destabilizing 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 subterranean because of the 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.

Bull Trout Habitat

Forestry practices are the dominant land use in all bull trout core areas and represent the highest risk to bull trout in the middle Kootenai (Libby Dam to Kootenai Falls). This risk to the bull trout population in the middle Kootenai is elevated due to the low number of spawning streams (Quartz, Pipe, O'Brien and Libby Creek drainages) available; a direct result of habitat fragmentation caused by Libby Dam. The Fisher River drainage is also being considered for designation as a core area. The middle Kootenai is a nodal habitat containing critical over-wintering areas, migratory corridors, and habitat required for reproduction and early rearing.

Dam operations are considered a very high risk to the continued existence of the Kootenai drainage population of bull trout (Montana Bull Trout Scientific Group 1996a). Dam operations represent a direct threat to bull trout in the middle Kootenai because of the biological affects associated with unnatural flow fluctuations and potential gas supersaturation problems arising from spilling water. The dam is a fish barrier, restricting this migratory population to 29 miles of river. Habitat fragmentation caused by Libby Dam increases the likelihood that localized effects become a higher risk to the confined population.

In the upper Kootenai (above Libby Dam), the threats to bull trout habitat include illegal fish introduction, introduced fish species, rural residential development, and forestry. Additional risks come from mining, agriculture, water diversions, and illegal harvest (Montana Bull Trout Scientific Group 1996b). Critical spawning streams include the Grave Creek drainage in the U.S.

and the Wigwam drainage in British Columbia. Transboundary research is ongoing in Canadian tributaries known to be used by spawning bull trout: Elk River, St. Mary River, Skookumchuck Creek, White River, Palliser River, and the Kootenay River upstream (Baxter and Oliver 1997). Nodal habitats for this population are provided by Libby Reservoir, Tobacco River, and the Kootenay River in Canada.

Bull trout are found below Kootenai Falls in O'Brien Creek and in Bull Lake, the latter a disjunct population. Montana Fish, Wildlife & Parks (MFWP), in cooperation with Idaho Department of Fish and Game, are monitoring movement patterns of fish tagged after spawning in O'Brien Creek. These fish inhabit areas in the lower Kootenai River and Kootenay Lake during most of the year.

White Sturgeon Habitat

Alteration of the annual hydrograph in the Kootenai River caused by the operation of Libby Dam is considered a primary reason for declines in the Kootenai River white sturgeon population (USFWS 1999 and 2000). Very few young sturgeon have recruited to the population since Libby Dam began impounding the river. Research suggests that the spring freshet is required by white sturgeon for reproduction and early life survival. Historically, white sturgeon spawning corresponded with the May to July runoff period when suitable temperature, water velocity, and photoperiod conditions would normally exist. Prior to the initiation of experimental flow augmentation to restore normative conditions in 1992, Libby Dam had effectively eliminated the naturally high spring runoff event. In addition, cessation of periodic channel maintenance or "flushing" flows has allowed fine sediments to build up in Kootenai River bottom substrates. This sediment fills the spaces between riverbed cobbles, reducing fish egg survival, larval and juvenile fish security cover, and insect production. Biological production was diminished as a result.

Since 1992, experimental flow augmentation during the spawning period appears to have improved conditions for spawning, as evidenced by the collection of more sturgeon eggs (Paragamian et al. 2001). Although spawning has been documented during each year of the flow augmentation tests, few wild juvenile white sturgeon have been captured. Recruitment of juveniles to the Kootenai River white sturgeon population has been insufficient to recover the population and remains a serious concern.

Kootenai River white sturgeon spawn within an 18-km river reach downstream of Bonners Ferry, Idaho (river kilometers (rkm) 228-246). Acoustic Doppler profiles of the Kootenai River bottom have revealed large sand dunes located in the spawning reaches (IDFG/USGS unpublished data). The shifting sand substrate may contribute to egg suffocation and/or prolonged contact with contaminated sediments, contributing to the declining recruitment of young white sturgeon. Sand substrate is thought to be poor habitat for survival of eggs and larva when compared to spawning habitat in unimbedded cobble in the Columbia River (Parsley and Beckman 1994; Paragamian et al. 2001). More suitable substrates of cobble and gravel occur upstream of Bonners Ferry (Apperson 1992, Paragamian et al. 2001).

Researchers have postulated that it may be possible to entice sturgeon to spawn further upstream over unembedded cobble substrates. It is possible that the decline of white sturgeon recruitment may be related to changes in the operation of Kootenay Lake in British Columbia. Concomitant to Libby Dam construction, the springtime maximum surface elevation of Kootenay Lake was lowered 2 m. Higher lake elevations create a backwater effect in the spawning reach. Evidence suggests that as the lake elevation rose during any given spawning season, sturgeon spawned progressively further upstream (Paragamian et al. 2001). Fifty-nine percent of the variation in spawning location was attributable to Kootenay Lake elevation. A linear regression model indicated higher lake elevations might promote spawning further upstream over cobble substrate.

As a consequence of altered flow patterns, average water temperatures in the Kootenai River are typically warmer (by 3 degrees Celsius) during the winter and colder (by 1 - 2 degrees Celsius) during the summer than prior to impoundment at Libby Dam (Partridge 1983). However, during large water releases at Libby Dam in the spring, water temperatures in the Kootenai River may be colder than under normal spring flow conditions.

Much of the Kootenai River has been channelized, diked and stabilized from Bonners Ferry downstream to Kootenay Lake, resulting in reduced aquatic habitat diversity, altered flow conditions at potential spawning and nursery areas, and altered substrates in incubation and rearing habitats necessary for survival (Partridge 1983, Apperson and Anders 1991). Side-channel slough habitats in the Kootenai River flood plain were eliminated by diking and bank stabilization in the Creston Valley Wildlife Management Area in British Columbia and Kootenai National Wildlife Refuge in Idaho.

The overall biological productivity of the Kootenai River downstream of Libby Dam has also been altered. Libby Dam blocks the open exchange of water, organisms, nutrients, and coarser organic matter between the upper and lower Kootenai River. Snyder and Minshall (1996) stated that a significant decrease in concentration of all nutrients examined was apparent in the downstream reaches of the Kootenai River after Libby Dam became operational in 1972. Libby Dam and the impounded Lake Koocanusa reduced downstream transport of phosphorus and nitrogen by up to 63 and 25 percent respectively (Woods 1982), with sediment-trapping efficiencies exceeding 95 percent (Snyder and Minshall 1996). The Kootenai River, like other large river-floodplain ecosystems, was historically characterized by seasonal flooding that promoted the exchange of nutrients and organisms among a mosaic of habitats (Junk et al. 1989; Bayley 1995). As a result of channel alterations, the Kootenai River has a lowered nutrient and carbon-retention capacity. Wetland drainage, diking and subsequent flood control has eliminated the “flood pulse” of the river and retention and inflow of nutrients. Removal of riparian and floodplain forests has eliminated sources of wood to the channel and potential retention structures.

In relation to reduced productivity, potential threats to Kootenai River white sturgeon include decreased prey availability for some life stages, and a possible reduction in the carrying capacity in the Kootenai River and Kootenay Lake to sustain populations of white sturgeon and other native fishes. A limited food supply for young of the year could contribute to increased mortality rates, either through starvation or through increased predation mortality, because young of the year would spend more time feeding, thereby exposing themselves to higher predation risk. The reduction in native kokanee in the South Arm of Kootenay Lake may have also reduced nutrient contributions (deteriorating carcasses from spawners) from tributaries in Northern Idaho and British Columbia flowing into the Kootenai River. Kokanee were also considered an important food source for adult sturgeon to build reserves for the winter and help in final gonad maturation. Growth rates of sturgeon have declined and relative weights in the Kootenai River/Lake population are the lowest in reported sturgeon populations in the Northwest.

Releases from Libby Dam effect water retention time, and thus biological productivity in Kootenay Lake, British Columbia (USFWS 1999). The warm, sunlit epilimnion contains the highest density of photosynthetic phytoplankton, as well as zooplankton. As inflow to the lake increases, more water must flow through the outlet or be stored in the pool. If the pool elevation is stable or declining, inflowing waters displace a commensurate volume that passes through the outlet. The physical configuration of Kootenay Lake, including a shallow sill at the outlet to the West Arm and a downstream control called Grohman Narrows at the outlet to Corra Linn Dam, result in an epilimnetic release of water from the lake. Decreased water retention in the lake's epilimnion results in greater downstream loss (entrainment) of organisms through the turbines. This effect, caused by high summer discharges from Libby Dam is exacerbated during the summer when thermal stratification in Kootenay Lake is well established. Downstream loss of free nutrients and biomass reduces food availability within Kootenay Lake which is inhabited by white sturgeon. Concerns over nutrient levels in the lake are evident by past investigations of nutrient loading (Daley et al. 1981) and ongoing lake fertilization experiments being conducted by Ashley and Thompson (1996).

The Adaptive Environmental Assessment modeling performed for the Kootenai River system in 1997 identified predation on eggs and larvae as a potential threat to successful white sturgeon recruitment. For broadcast spawners like white sturgeon, the mortality rate on eggs and larvae will increase with: 1) an increase in the number of predators; 2) an increase in the vulnerability of eggs or larvae to predation associated with changes in habitat or foraging behavior; and 3) a decrease in the volume or area of water that the eggs/larvae are dispersing into or over (as volume or area decreases, prey concentration to predators in increases). In post-impoundment years, Kootenai River springtime flows have been reduced substantially and vulnerability has increased due to an increase in water clarity and reduced food supply, as well as loss of unimbedded habitat in the spawning reach (Korman and Walters 1999).

Georgi (1993) noted that the chronic effects on wild sturgeon spawning in "chemically polluted" water and rearing over contaminated sediments, in combination with bioaccumulation of contaminants in the food chain, is possibly reducing the successful reproduction and early-age recruitment to the Kootenai River white sturgeon population. Results from a contaminant study performed in 1998 and 1999 showed that water concentrations of total iron, zinc, and manganese, and the PCB Arochlor 1260 exceeded suggested environmental background levels (Kruse 2000). Zinc and PCB levels exceeded EPA freshwater quality criteria. Several metals, organochlorine pesticides, and the PCB Arochlor 1260 were found above laboratory detection limits in ova from adult female white sturgeon in the Kootenai River. Plasma steroid levels in adult female sturgeon showed a significant positive correlation with ovarian tissue concentrations of the PCB Arochlor 1260, zinc, DDT, and all organochlorine compounds combined, suggesting potential disruption of reproductive processes. In an experiment designed to assess the effects of aquatic contaminants on sturgeon embryos, results suggest that contact with river-bottom sediment increases the exposure of incubating embryos to metal and organochlorine compounds (Kruse 2000). Increased exposure to copper and Arochlor 1260 significantly decreased survival and incubation time of white sturgeon embryos and could be a potentially significant additional stressor to the white sturgeon population.

Burbot Habitat

The timing of the collapse of the burbot fisheries in Idaho and British Columbia coincide with the operation of Libby Dam and associated changes in discharge volumes and water temperature. McPhail (1995) stated, "although burbot populations often increase after impoundment, the downstream effects of impoundment can be detrimental." Burbot are plentiful in Lake Koocanusa, Montana (Skaar, D. MFWP, pers. com. 2000) and make up a portion of the fish entrained through Libby Dam (Skaar et al. 1996). The population downstream of Libby Dam has declined, however.

Winter hydropower operations produce higher flows and wider flow fluctuations than occurred naturally prior to Libby Dam. Burbot are winter spawners, known to spawn at temperatures from 1 to 4 ^{o}C (McPhail and Paragamian 2000). The Kootenai River is now 4^{o}C warmer during winter than prior to impoundment. Unnaturally high flows or altered temperatures during winter may have altered the spawning behavior of fluvial and adfluvial burbot in the Kootenai River, disrupted their spawning synchrony [burbot are considered highly ordered in their spawning (Becker 1983)], or affected their physiological fitness or spawning readiness. Burbot can move extensive distances during the winter to spawn. Burbot are weak swimmers and have a low endurance for extended periods of increased flow (critical velocity of about 24 cm/s) (Jones et al. 1974). In the Kootenai River, traditional spawning tributaries in Idaho are 50 to 120 km upstream from Kootenay Lake. Current velocities in the lower Kootenai River are subject to change daily due to operations at Libby Dam, and water velocity is a function of river discharge and Kootenay Lake surface elevation. Flows in the Kootenai River at Copeland, Idaho greater than 255 m^3/s produce average current velocities higher than the critical velocity (>24cm/s) for burbot (Paragamian 2000). Flow near the Idaho/B.C. border can often be as high as 510 m^3/s during normal winter dam operations. Tagging and telemetry studies in the river have shown that burbot move freely between the lake and the river in Idaho, providing flow velocities are low. Paragamian (2000) provided telemetry data that indicated high flows during the winter inhibit spawning migrations of burbot in the Kootenai River. In addition, biopsies of post-spawn female and male burbot indicated that some burbot do not spawn and are reabsorbing gonadal products (Paragamian 1994; Paragamian and Whitman 1996).

Westslope Cutthroat Trout and Interior Redband Trout Habitat

Libby Dam has affected westslope cutthroat trout and interior redband trout in many of the same ways as it has affected bull trout. Alterations of the hydrograph have resulted in a loss of mainstem salmonid spawning and rearing habitat. Fluctuating discharges from Libby Dam force juvenile salmonids to frequently seek new habitat, increasing the risk of predation. In addition, the widely fluctuating flows prevent colonization of the varial zone by periphyton and macroinvertebrates, reducing the efficiency with which energy is transferred from one trophic level to another. Abundance and diversity of important aquatic invertebrates has declined since construction of Libby Dam (Hauer et al. 1997), further reducing food abundance for trout. All of these factors combined have likely resulted in reduced trout abundance in the Kootenai River.

Kokanee Habitat

Kootenai River kokanee are spawning populations from Kootenay Lake and the numbers of spawners in the river within Idaho and Montana are affected by habitat conditions altered by lake and river regulation. The construction of Duncan Dam on the Duncan River in 1967 and Libby Dam on the Kootenai River in 1972 resulted in reduced nutrient loading (primarily nitrogen and phosphorus) to Kootenay Lake followed by a decline in phytoplankton, zooplankton, and ultimately kokanee abundance (Ashley and Thompson 1993 and 1996). Kokanee populations continued to decline throughout the 1980s, and by 1990 the South Arm stocks of kokanee had become virtually extinct (Richards 1996). The presence of Mysis shrimp

(*Mysis relicta)* in Kootenay Lake and their potential to compete with juvenile kokanee for zooplankton makes it difficult to quantify the affect of the reduced phosphorus loading on kokanee numbers. Dike construction and channelization in the lower river and grazing activity in key spawning tributaries in Idaho may also have influenced the decline of kokanee.

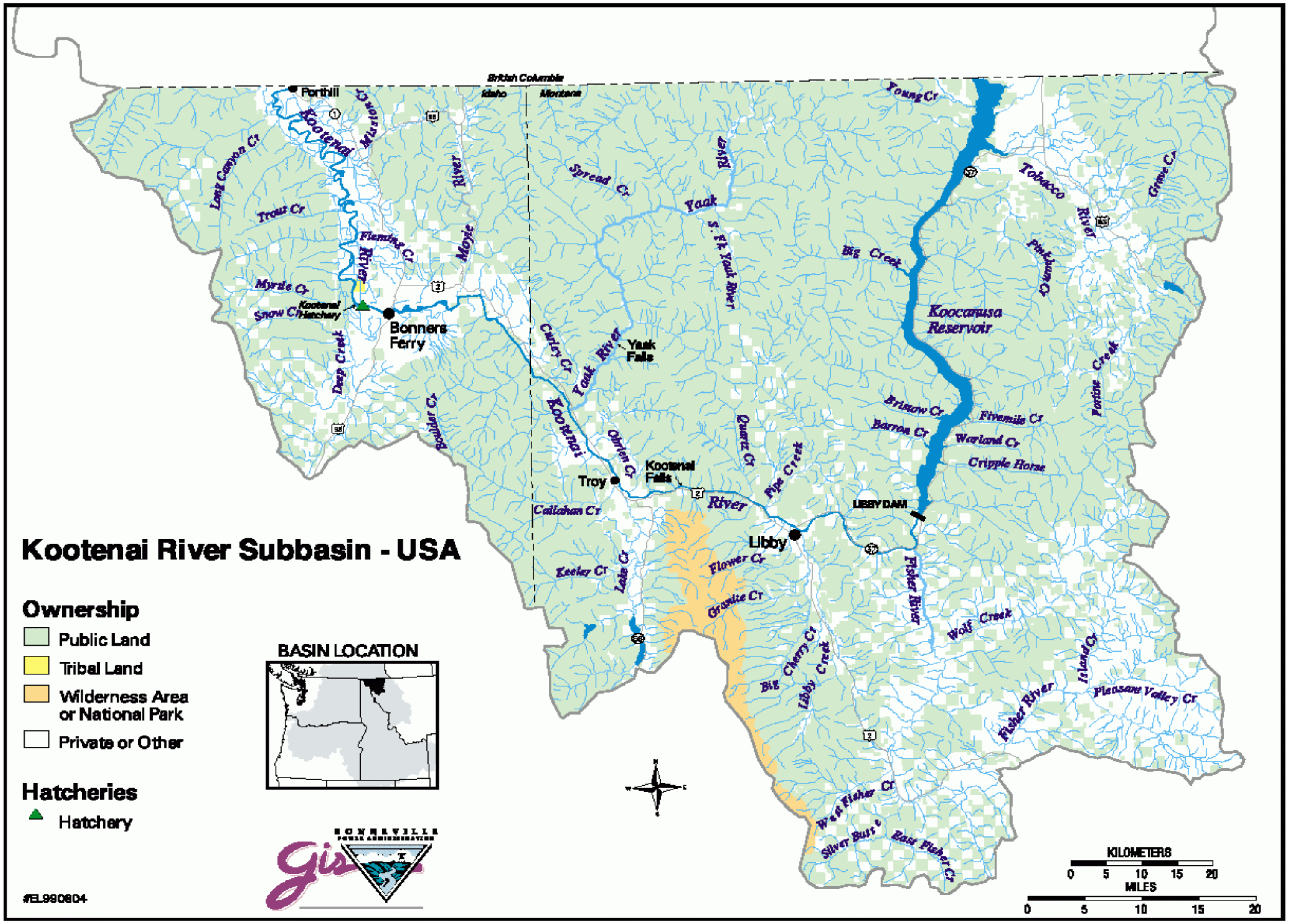

Figure 2. Kootenai River Basin, Montana.

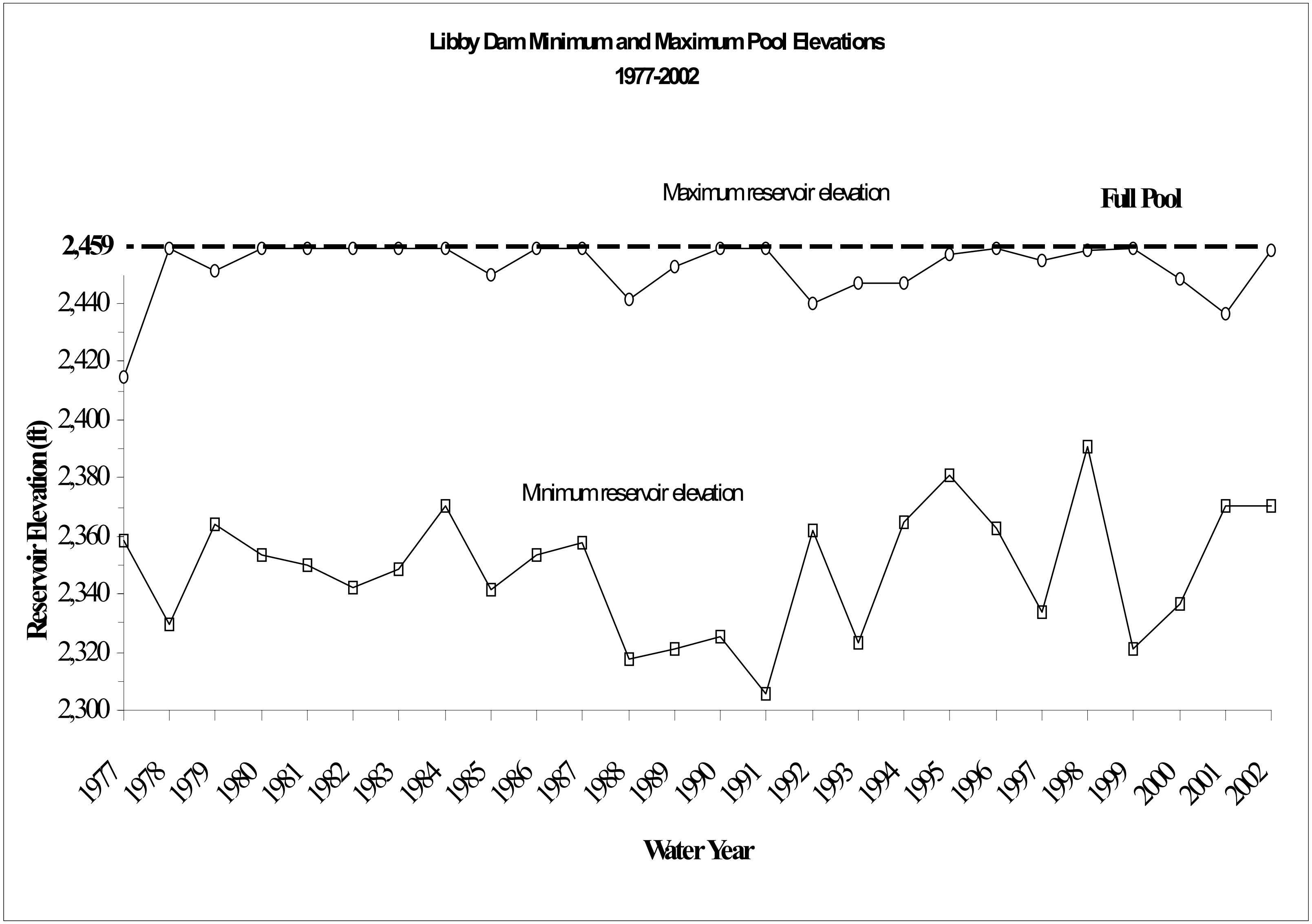

Figure 3. Libby Reservoir elevations (minimum, maximum), water years 1976 through 2000.

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Chapter 1

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

Abstract

Montana 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 Grave Creek and the Wigwam River. 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), 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. However, collectively bull trout redd numbers have significantly increased over the past 7-8 year period. We used radio telemetry to assess the movement, behavior, and spawning distribution of bull trout in the Kootenai River from 1998 to 2002. We surgically implanted radio tags into 65 bull trout from late January 1998 to early December 2000. Most (51; 78.5%) of the tagged fish were captured in the Kootenai River below Libby Dam (between Libby Dam and Alexander Creek; river mile [RM] 220.5 - 221.7). We had a relatively high success rate of tracking radio tagged bull trout after release, accounting for approximately 88% of all tagged fish at least once after initial release, and were observed an average of 30.7 observations per fish. We identified several common locations that adult radio tagged bull trout frequented in the Kootenai River above Kootenai Falls including the Libby Dam tailrace area that extends from Libby Dam to approximately 2 miles downstream to confluence of Alexander Creek. A substantial proportion of the radio tagged bull trout (34%) migrated downstream of Kootenai Falls. One of these fish returned upstream over the falls during a period when flows in the Kootenai River were approximately 8,000 cfs. We were only able to document 4 out of the 65 (6.2%) radio tagged bull trout in tributaries to the Kootenai River during spawning season. These 4 bull trout included 2 fish entering Quartz Creek, one entered the Fisher River and one entered O'Brien Creek. We have monitoring the relative abundance of burbot in the stilling basin below Libby Dam using hoop traps since 1994. We captured a total of 7 and 5 burbot were captured during the 01-02 and 02-03 trapping seasons, respectively. These total catches each year translated into the lowest total catch and catch per effort (burbot per trap day) on record since trapping began in the 94-95 trapping season. We sampled macro-invertebrates at the Libby Creek Demonstration Project and the Grave Creek Phase I Project in order to assess the benthic community response to these two restoration projects. Results from the Libby Creek site include pre and post

implementation results, with 4 of the 6 indices of diversity increasing after project implementation. The data collected on Grave Creek preceded restoration work, and will serve a baseline for comparison after restoration activities are completed. We conducted juvenile salmonid population estimates within reference reaches on Sinclair, Therriault, Grave, Young, Libby, Parmenter, Pipe, and Barron creeks. Trend analyses relevant to stream restoration projects are presented for Sinclair, Grave, Libby, and Parmenter creeks. Montana FWP has documented the changes in species composition, and species size and abundance within Koocanusa 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. Likewise, 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. 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 (Marotz et al. 1988).

Biological monitoring data was proven to be critical during the development of models used in management of water resources and operation of Libby Dam. These models include Integrated rule curves (IRC's), the Libby Reservoir model (LRMOD) and an alternate water management plan called VARQ, which stands for variable flow (Q). In several of these instances the models have been empirically calibrated using field data from an extensive sampling program. For example, the LRMOD was empirically 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). The ultimate result in many of these cases 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 manipulation measures may be the most promising method of 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 interior redband rainbow trout in the Kootenai Basin (see chapter 2). Biological and physical monitoring is critical to assess effectiveness of restoration or management activities intended to restore native fishes and their habitats. Evaluation of restoration activities and pilot projects will continue in order to determine the most cost-effective methods of 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 (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 to assess population trends.

Adult Bull Trout Radio Telemetry

We used radio telemetry to assess the movement, behavior, and spawning distribution of bull trout in the Kootenai River from 1998 to 2002. We surgically implanted radio tags into 65 bull trout from late January 1998 to early December 2000. Most (51; 78.5%) of the tagged fish were captured in the Kootenai River below Libby Dam (between Libby Dam and Alexander Creek; river mile [RM] 220.5 - 221.7). Fish captured at this location were tagged over the same general time period, and were captured via nighttime jetsled electrofishing using a Coffelt model Mark 22 electrofishing unit, operating with an electrical output ranging from 200-350 volts at 5-8 amps. Nine out of the 65 radio tagged bull trout (13.8%) were captured in a downstream weir in lower Quartz Creek (above Kootenai Falls; RM 199.1) between 9/28/99 and 10/7/99, after the fish had likely spawned in Quartz Creek. We also captured one bull trout (1.5%) in a downstream weir in lower Callahan Creek (below Kootenai Falls; RM 186.4) in October 1998. Three additional bull trout (4.6%) were also captured from Koocanusa Reservoir and radio tagged. All but 4 of the radio tagged bull trout were released in the general vicinity of capture. Release locations and study objectives of these four bull trout are described below.

Skaar et al. (1996) documented bull trout entrainment at Libby Dam. However, the proportion of bull trout in the Kootenai River downstream of Libby Dam and the ultimate fate bull trout that survive entrainment are unknown. Therefore in order to assess whether bull trout that survive entrainment will spawn in tributaries downstream of Libby Dam, we captured 4 bull trout in Koocanusa Reservoir and released these fish below Libby Dam. These fish were to serve as surrogates for bull trout that survived entrainment. Likewise, one of the bull trout captured below Libby Dam was released in the reservoir, and was intended to serve as a surrogate for an entrained bull trout. This bull trout was captured below Libby Dam in March 1999 and released in Koocanusa Reservoir approximately 1 mile above Libby Dam. The objective of releasing this fish in the reservoir was to assess whether or not this fish would spawn in a tributary above the dam.

Upon capture of all bull trout, we examined each fish for marks, tags, and injuries, and then we anesthetized each fish using an aqueous non-buffered solution of MS-222, measured them, and surgically implanted the radio tag. We used two sizes of radio tags for this study, in an attempt to balance battery life versus tag size. We reserved our 12 smaller tags for smaller bull trout. These tags weighed 9.5 g, had a minimum life span of 180 days, a burst rate of 56 pulses per minute, a 25 cm flexible external whip antenna attached to one end, and transmitted on frequencies ranging from 48.021 to 48.251 MHz. The remaining 53 (larger) radio tags weighed 25.6 g had a minimum life span of 750 days, a burst rate of 55 pulses per minute, a 35.6 cm flexible external whip antenna, and transmitted on frequencies ranging from 49.014 to 49.800 MHz. All tags transmitted on a unique frequency that allowed individual fish identification. Tags and were manufactured by Advanced Telemetry Systems, Inc. and were powered by a single 3.6 V lithium battery. We used telemetry receivers manufactured by Lotek Engineering (Model SRX-400) for mobile monitoring activities. We determined the location of tagged fish using mobile tracking that consisted of a combined effort of fixed wing aircraft and jetsled observations. Each mobile monitoring unit consisted of a radio receiver, data processor, internal clock, and either a single (jetboat) or double (fixed wing aircraft) tuned loop antenna. Fish movement and visual observations were used as the primary as indicators of live fish. The fish were generally tracked weekly through the spring and summer months and monthly during the winter due to a lack of fish migration.

Burbot Monitoring

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.

Stream Macroinvertebrate Monitoring

We collected macroinvertebrates during September 2002 using Surber samplers and kick-nets within and below the Libby Creek and Grave Creek stream restoration project areas. The sampling effort at these locations was intended to serve as an indicator of aquatic health and to provide a comparison through time. The following section summarizes sample collection protocols. Bukantis (1998) provides additional details related to sample collection methodologies.

We sampled three consecutive riffles representative of the available microhabitats at each project area that contained substrates at least 1 inch in diameter, sample depths less than one foot, and stream velocities of at least 1 foot per second (fps), but not exceeding 3 fps. Each riffle sample consisted of 3 Surber samples pooled together. For the first riffle, a diagonal transect was measured from the top right corner of the riffle to the bottom left corner (looking downstream); 1 Surber sample was taken at the top right stream margin, 1 midway between the top right and the center of the diagonal (at 1/4 the length of the diagonal), and 1 midway between the center of the diagonal and the bottom left corner of the riffle (at 3/4 the length of the diagonal). For the second riffle, a diagonal was measured from the bottom right corner of the riffle to the top left corner (looking downstream) and 1 Surber sample was taken at the bottom right stream margin, 1 midway between the bottom right and the center of the diagonal (at 1/4 the length of the diagonal), and 1 midway between the center of the diagonal and the top left corner of the riffle (at 3/4 the length of the diagonal). In the third riffle, a transect perpendicular to flow was measured; a single Surber sample was taken at the left margin, mid-stream, and midway between the center of the transect and the right margin.

We sampled microhabitats between the first and second riffles with a kick-net utilizing the "20 jab" method. The approximate proportions of productive macroinvertebrate habitats in the chosen reach were recorded using the following habitat types: riffles, snags, aquatic vegetation, and bank margins. The 20 jabs were collected proportionally among the habitats. A 1 m traveling kick, or a 1 m sweep if the current was too swift, was used to sample riffle habitats. We sampled with a 1 m sweep through and around snags. We scrubbed macroinvertebrates from coarser snags by hand. We sampled aquatic vegetation using a 1meter sweep, and bank margins with a combination of the techniques described above.

We stored samples in 95% ethanol; those with excessive organic detritus were decanted and refreshed with preservative in the lab. We sorted all samples as soon as possible to minimize decomposition. An independent contractor identified all aquatic invertebrates to the level of genus, and calculated several indices and metrics for comparison. However, for this report, we chose only to present species richness, Ephemeroptera, Plecoptera and Trichoptera (EPT) richness, and the Montana Biotic Index developed by the Montana Department of Environmental Quality (Bukantis 1998). We chose these particular measures due to their sensitivity to detect change due to perturbation, and for consistency with other similar efforts in the region and the state of Montana. Measures for the three riffles sampled at each site were pooled using the arithmetic mean.

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

Sinclair Creek

We established three sections on Sinclair Creek to perform population estimates. Sections one and two were located within a reach of lower Sinclair Creek where stream restoration activities have occurred and were established to monitor the response of salmonid abundance to restoration activities at this location (Hoffman et al. 2002). Section three represented our hydrological relic reach, due to the stable channel and comparable channel type at this site. We sampled Section 2 in 2001 and all three sections in 2002. These five sections include the following.

- Section 1: is located 442 m upstream of the Highway 93 culvert (T36N,R27W, Sec24).
- Section 2: is located 209 m upstream of the upper boundary of Section 1.
- Section 3: is located in the Willow Fire Ranch property, approximately 4.8 stream kilometers upstream from the Purdy project site (NE1/4, Sec. 18 T36N,R26W).

Therriault Creek

We established three monitoring sections in Therriault Creek to be used for juvenile salmonid trend analyses (Hoffman et al. 2002). Only section 2 was sampled during the 2001 field season. No sampling occurred in Therriault Creek in 2002. The upper boundary for Section 2 is located at the first culvert above highway 93 and proceeds downstream 120 m. This section is located on private property, and can be characterized as an unstable and entrenched stream channel with unstable banks.

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 salmonids 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 in 2001 and 2002.

Barron Creek

Montana FWP randomly established 4 electrofishing sections on Barron Creek in 2002 ranging throughout the lower 5 miles within the watershed. The location of these sections is as follows.

Section 1: is located directly upstream of the culvert on the FDR highway.
Section 2: is located at approximately 1.3 miles up forest road 615.
Section 3: the lower boundary of Section 3 is located at the upper road sign for forest road 4803.
Section 4: begins at the 5-mile marker on forest road 615.

Koocanusa 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^{o}C.

2) Fall (September - October): Drafting of the reservoir began, surface water temperature decreased to 13 - 17^{o}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.

Koocanusa Reservoir Zooplankton Monitoring

Montana FWP has collected zooplankton from Koocanusa 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 2002. 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 a total of 173 and 199 bull trout redds in Grave Creek in 2001 and 2002, respectively (Table 1). Bull trout have exhibited a positive trend in spawning abundance in Grave Creek since 1993 (Figure 2; $r^2 = 0.733$; $p = 0.0016$).

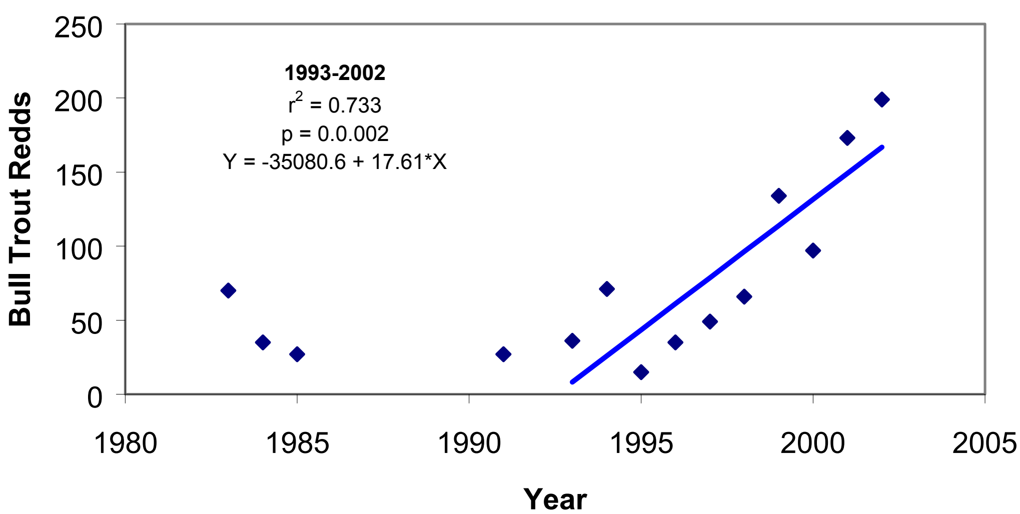

Figure 2. Bull trout redd counts, and trend analysis in Grave Creek, 1993 through 2002.

Wigwam Drainage

Bull trout redd counts for the Wigwam River includes the tributary streams of Bighorn, Desolation, and Lodgepole creeks. A total of 1496 and 1916 redds were observed in the Wigwam Drainage in 2001 and 2002, respectively (Table 1). Bull trout redds in the Wigwam River have consistently increased each year since 1995 (Figure 3; $r^2 = 0.946$; $p = 4.9*10^{-5}$).

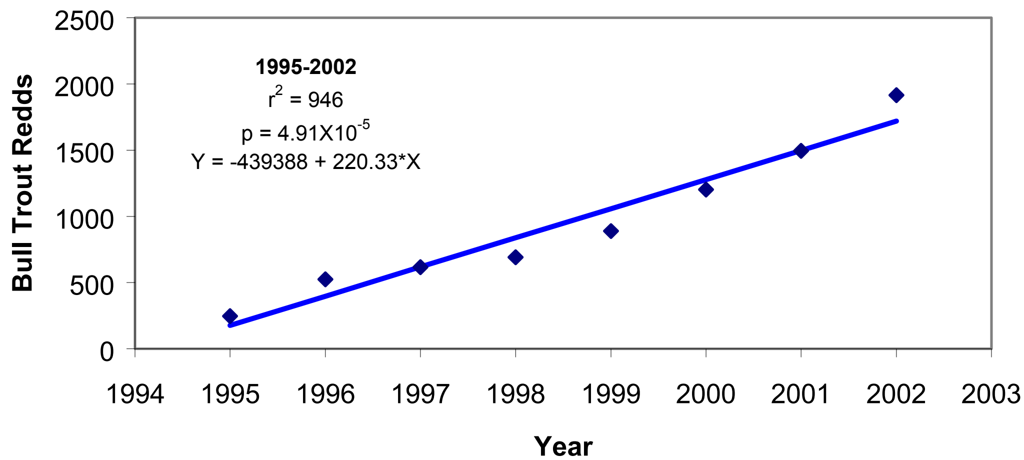

Figure 3. Bull trout redd counts and trend analysis for the Wigwam River (including Bighorn, Desolation, and Lodgepole creeks) 1995-2002.

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
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
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
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	
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
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
	2002	102	
West Fisher River	1995	3	10
	1996	4	6
	1997	0	6
	1998	8	6
	1999	18	10

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

Stream	Year Surveyed	Number of Redds	Miles Surveyed
	2000	23	10
	2001	1	10
	2002	1	6
Wigwam (B.C and U.S.)	1995	247	22
Includes Bighorn, Desolation, Lodgepole Creeks	1996	524	22
	1997	615	22
	1998	691	22
	1999	889	22
	2000	1204	22
	2001	1496	22
	2002	1916	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
White River (B.C.)	2001	166	7.8
	2002	153	7.8

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 = 0.224). Although overall trend is positive, annual variation limits our ability to statistically distinguish this relationship from a stable (zero slope) population (Figure 4; p = 0.102). We observed a total of 154 and 62 redds in Quartz and West Fork Quartz creeks in 2001 and 2002, respectively (Table 1). The average number of redds of the period of record was 78.4 redds. The 2001 observation represented a record number of bull trout redds in Quartz Creek, and a 96.5% increase over the average. However, the 2002 observation of 62 redds was 20.9% 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 2002 may have limited bull trout spawner escapement in 2002. If we remove the 2002 bull trout redd counts from the dataset, and repeat the regression analysis, the variation between years decreases slightly (r^2 = 0.385), and the positive trend is significant (p = 0.031).

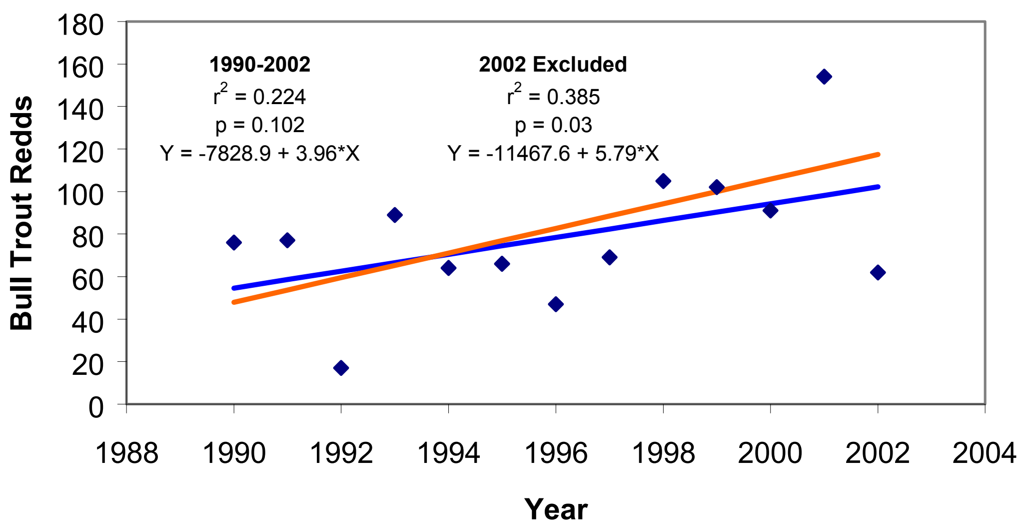

Figure 4. Bull trout redd counts and trend analysis (blue line) for Quartz Creek (including West Fork Quartz) 1990-2002. The 2002 observation was removed and the regression analysis repeated due to the presence of a log jam in the West Fork Quartz Creek in 2002 (orange line).

Pipe Creek

Bull trout redd counts in Pipe Creek peaked in 1999 with 36 redds, with redd numbers decreasing since that peak. Despite the decreasing trend of bull trout redds during the last three years, the overall general trend during the time period 1995-2002 has been variable, but a slightly increasing trend (Figure 5; $r^2 = 0.2478$; $p = 0.0834$). We observed a recent low number of 6 redds observed in 2001 which may be partially explained by the presence of a man made obstruction (swimming hole) on lower Pipe Creek. If we remove the 2001 bull trout redd counts from the dataset, and repeat the regression analysis, the variation between years decreases slightly ($r^2 = 0.433$), and the positive trend is significant ($p = 0.0201$).

Bear Creek

Bear Creek bull trout redd counts have been variable during the period 1995-2002 (Figure 6; $r^2 = 0.06$). Although the overall general trend has increased since 1995, the relationship is not statistically different than a stable population (Figure 6; $p = 0.5465$). A log jam was also located on lower Bear Creek in the fall of 2001, that may have limited bull trout spawner escapement during that year. If we remove the 2001 bull trout redd counts from the dataset, and repeat the regression analysis, the variation between years decreases slightly ($r^2 = 0.313$), but the overall trend remains non-significant ($p = 0.191$), suggesting that the population is stable. The average number of bull trout redds since 1995 in Bear Creek has been 16.4 redds. The number of redds observed in 2001 was 75.6% lower than the annual average since 1995. In 2002, we observed an increase of 3.8% more bull trout redds than average in Bear Creek.

O'Brien Creek

The general trend of bull trout redds in O'Brien Creek is generally increasing since 1995 (Figure 7; $r^2 = 0.547$; $p = 0.006$). We observed a total of 47 and 45 bull trout redds in O'Brien Creek in 2001 and 2002, respectively (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 (1993-2002). From the period 1993-2000, the general trend was one of increasing abundance. However, we observed only 1 bull trout redd in each 2001 and 2002 (Figure 8). The overall trend was not significantly different than a stable (zero slope) population ($r2 = 0.113$; $p = 0.343$). Given the amount of variation present within this dataset, the overall mean number of redds in the West Fisher (mean = 6.0 redds) does an equally well job at predicting redd numbers.

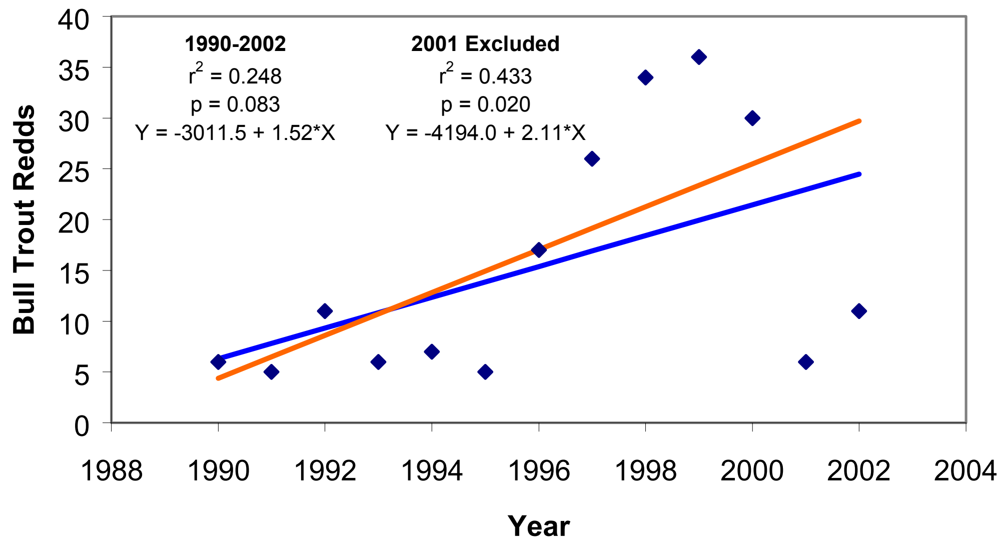

Figure 5. Bull trout redd counts and trend analysis (blue line) for Pipe Creek 1990-2002. A manmade dam was present in lower Pipe Creek in the fall of 2001 that likely impeded bull trout migration. Therefore the 2001 observation was removed and the regression analysis was repeated (orange line).

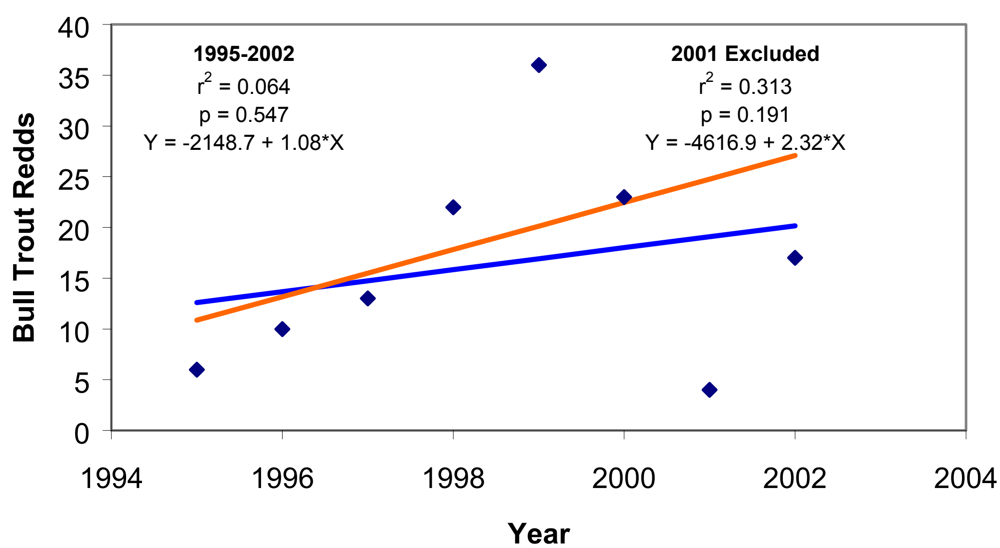

Figure 6. Bull trout redd counts and trend analysis (blue line) in Bear Creek, a tributary to Libby Creek, 1995-2002. A log and debris jam was present in lower Bear Creek in the fall of 2001 that likely impeded bull trout migration. Therefore the 2001 observation was removed and the regression analysis was repeated (orange line).

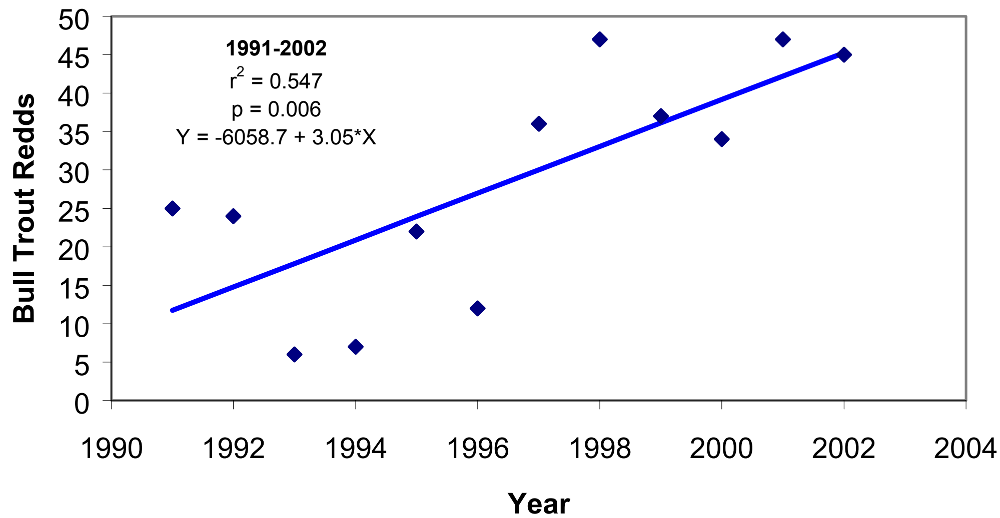

Figure 7. Bull trout redd counts and trend line (blue line) in O'Brien Creek 1991-2002.

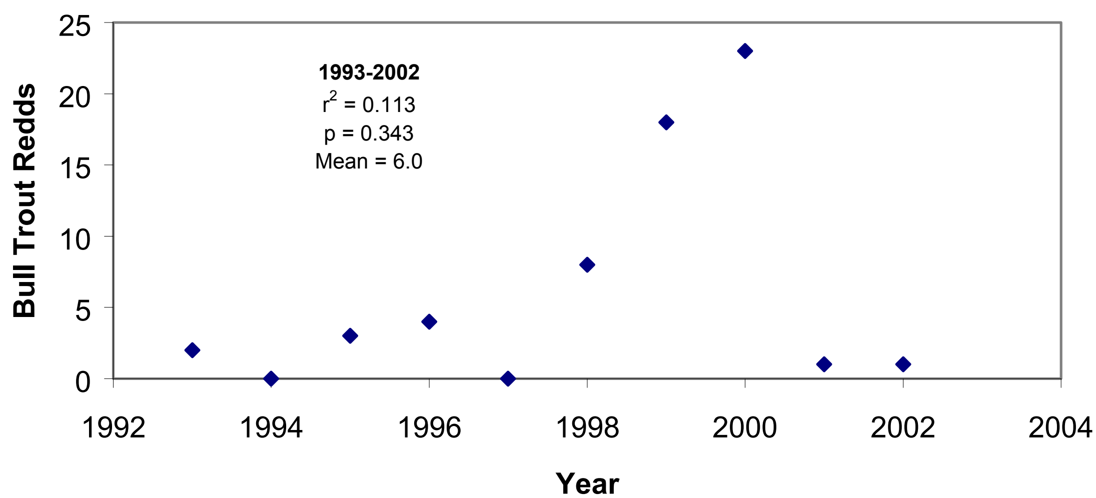

Figure 8. Bull trout redd counts in the West Fisher River, a tributary to the Fisher River, 1993-2002.

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 Restoration Team 1996a). 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 13 and 102 bull trout redds in Keeler Creek and associated tributaries in 2001 and 2002, respectively (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 = 0.001; Figure 9), and the trend is a decreasing population, although the relationship is not significantly different from a stable population (Figure 9; p = 0.958). Given this relationship, the annual mean (75.6 redds) does an equally well job of prediction. The 2002 observation represents a 35% increase over the annual mean, and the 2001 observation represents an 82.8% reduction from 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 significant increasing trend since 1996 (Figure 9; r^2 = 0.587; p = 0.076).

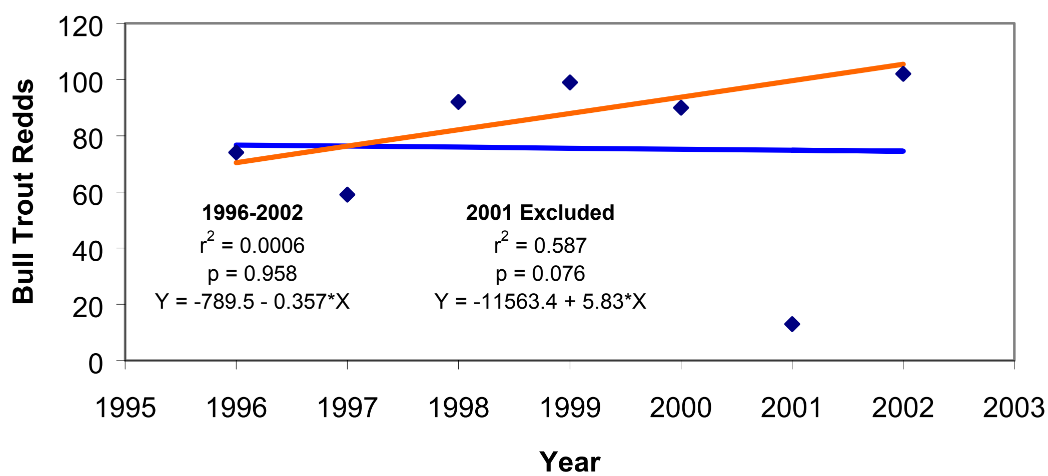

Figure 9. Bull trout redd counts and trend line (blue line) in Keeler Creek, a tributary to Lake Creek, 1996-2002. A beaver dam was present in lower Keeler Creek in the fall of 2001 that likely impeded bull trout migration. Therefore the 2001 observation was removed and the regression analysis was repeated (orange line).

Adult Bull Trout Radio Telemetry

Montana FWP radio tagged a total of 65 bull trout ranging from 362-823 mm total length (Figure 10). The length frequency distribution of radio tagged bull trout tagged throughout the duration of this study was bimodal (Figure 10). We attribute this bimodal distribution to our intentional selection smaller fish in 1998 and 1999. This was especially true in 1998, where the 12 bull trout tagged with the 48 MHz (smaller) radio tags were significantly smaller sized fish (mean total length = 459 mm; $p = 5.02*10^{-6}$) when compared to all bull trout tagged with the 49 MHz (larger) tags. The overall mean total length of all bull trout radio tagged in 1998 was 552.9 mm (Table 2), and was significantly smaller than those fish tagged in 1999 and 2000 ($p < 0.05$; Table 2). However, when the fish tagged with the 48 MHz (smaller) tags were removed from this analysis, and the analysis was repeated with only those fish tagged with the 49 MHz (larger) tags, the mean total length of fish tagged each year (Table 2) was not significantly different between years ($p = 0.499$).

Table 2. Total number, frequency, and mean total length of bull trout radio tagged in the Kootenai River from 1998-2000.

	1998 All Tags	1998 48 MHz Only	1998 49 MHz Only	1999	2000
Number Tags	32	12	20	22	10
Mean Total Length (mm)	552.9	458.8	609.4	632.8	660
Range Total Length (mm)	362-818	362-666	482-818	430-823	565-800

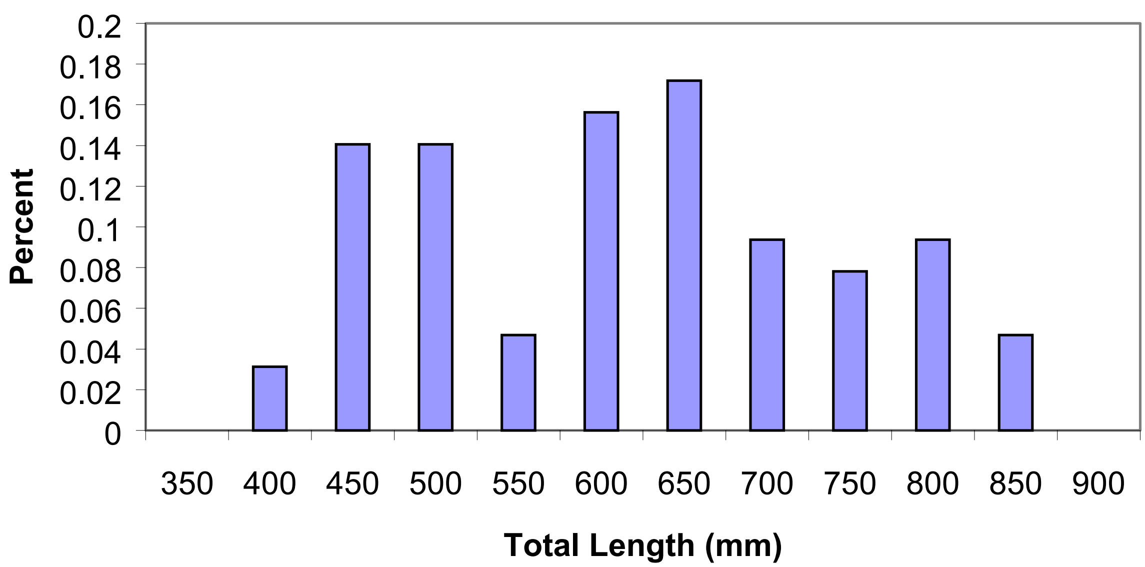

Figure 10. Length frequency distribution of the total length (mm) of all bull trout radio tagged in the Kootenai River from 1998-2000.

We had a relatively high success rate of tracking radio tagged bull trout after release. We tracked the radio tagged bull trout throughout the battery's lifespan for each particular tag. The smaller (48 MHz) tags were all placed in bull trout in 1998 and were tracked through 1998 and most of 1999. The remaining larger (49 MHz) radio tags were placed in fish from 1998 to 2000 and tracked from 1998 to 2002. However, eight (12.3%) of our tagged fish were never located after the first month of survey. These included the single bull trout captured and tagged as an outmigrant from Callahan Creek, one of the three captured in Koocanusa Reservoir and placed below Libby Dam, one of the nine bull trout captured in the Quartz Creek weir, and five bull trout captured in the Kootenai River below Libby Dam. In particular, the bull trout that was captured in the Kootenai River below Libby Dam and placed in the reservoir was entrained through the dam a few days after tagging and an angler recovered the tag. The fate of the other 7 bull trout not located is unknown.

The three bull trout that were captured in Koocanusa Reservoir and placed in the Kootenai River directly below Libby Dam collectively provided little information as to the behavior or movement patterns due to a limited number of observations on these fish. One of these fish was never observed after release in the Kootenai River. A second bull trout (tag number 49.055) was tagged on May 7, 1999, and subsequently observed twice 3.6 miles downstream of the release point within two weeks of release. The only other observations of this bull trout (n =21) were near the top of Jennings Rapids (RM 217.5) between June 22, 1999 and January 2, 2001, when the tag was ultimately recovered. Given the extended period of time this fish was observed at this location and the recovery of the tag, it is likely this fish died or shed the tag within a couple of weeks after tagging within 3.8 miles of the release location. The third bull trout that was captured in Koocanusa Reservoir and released below Libby Dam migrated below Kootenai Falls. This bull trout was tagged and released on May 18, 1998 and first detected after release approximately 1 mile upstream of the Leonia gauging station (RM 167.9) 203 days after release. This fish was detected an additional 15 times within 2.3 miles of this location. The tag was last detected on September 24, 2001 at RM 165.6.

The remaining 58 radio tagged bull trout (89.2%) that were observed after tagging were all released above Kootenai Falls, and were observed an average of 30.7 observations per fish. The average number of days between observations was 22.6 and the average date of encounter was 8/14/99. We estimated that 36 of the 58 radio tagged bull trout (62%) remained above Kootenai Falls for the duration of our mobile tracking efforts. The remaining 22 of the 58 radio tagged bull trout (38%) migrated downstream of Kootenai Falls. All the radio tagged bull trout that migrated below Kootenai Falls were either originally captured in the downstream weir in Quartz Creek or captured in the Kootenai River directly below Libby Dam. The total proportion of the fish that migrated below Kootenai Falls was similar for Quartz Creek fish (n=4; 44.4%) and the Kootenai River below Libby Dam (n = 18; 36.0%). The maximum distance a radio tagged bull trout was observed below Kootenai Falls had traveled a maximum distance of 84 miles from the original release location. The mean distance traveled between observations for all radio tagged bull trout was 1.2 miles (standard deviation = 6.2 miles). The mean total length (at time of tagging) of those bull trout that migrated downstream of Kootenai Falls was 622 mm, and was not significantly different ($p = 0.189$) than those bull trout that remained above Kootenai Falls (mean total length = 576). We were not able to assess differences in either the sex or age of bull trout that migrated below Kootenai Falls because we didn't collect age or sex information at the time of tagging.

In order to assess whether our handling at time of tagging influenced the occurrence of migration of tagged fish below Kootenai Falls, we attempted to bracket the true number of days after tagging that it took for each fish to migrate below Kootenai Falls. For each fish that migrated below Kootenai Falls, we calculated a minimum number of days by subtracting the last observation date prior to being detected below Kootenai Falls from the tagging date. Likewise, we calculated the maximum number of days by subtracting the first date each fish was detected below Kootenai Falls from the tag date. The mean number of days between the minimum number of days before migration over Kootenai Falls was 158.3 days (Figure 11; standard deviation = 187.9 days and median = 68.5 days), and the mean number of days between the maximum number of days before migration below Kootenai Falls was 231 days (standard deviation = 223.7; median = 180.5 days). However, the distribution of both the minimum and maximum are skewed (Figure 11). For example, the minimum number of days before migration over Kootenai Falls was 25 days or less for 27.3% of the fish, and 25 to 50 days for 22.7% of the fish. In comparison, the maximum number of days before migration over Kootenai Falls was 25 days or less for none of the tagged bull trout and 25 to 50 days for 9.1% of the fish.

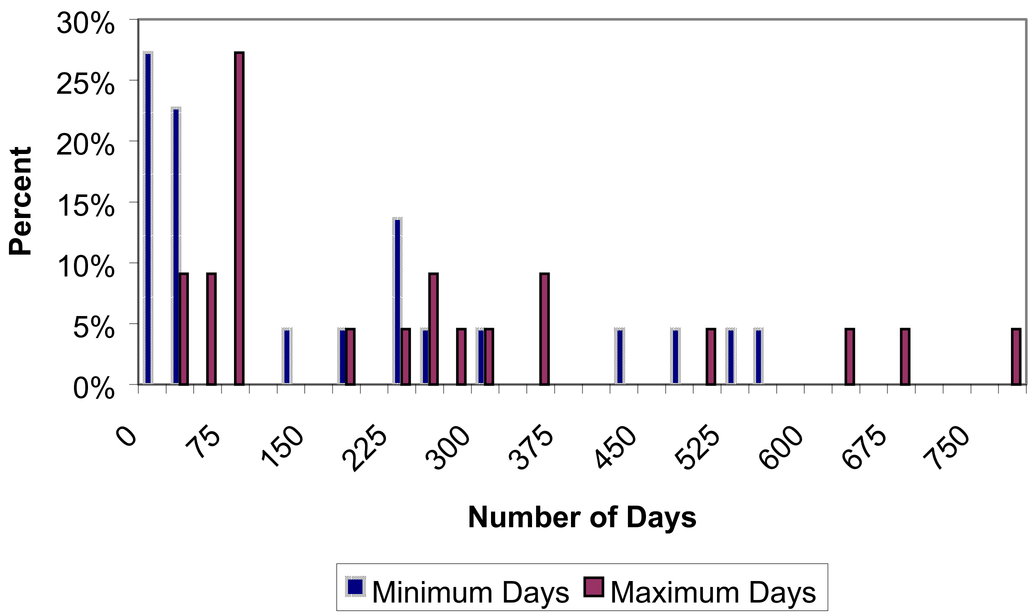

Figure 11. A histogram of the minimum (last observation date prior to being detected below Kootenai Falls minus tagging date) and maximum number of days (first date each fish was detected below Kootenai Falls minus the tag date) for all radio tagged bull trout that migrated below Kootenai Falls from 1998-2001.

Out of the 22 radio tagged bull trout that migrated over Kootenai Falls, we only documented one fish (tag number 49.221) that returned upstream over the falls. This female bull trout was originally radio tagged as an outmigrant from Quartz Creek on 10/7/99, and observed below Kootenai Falls on 11/4/99. The fish ascended Kootenai Falls between 7/18/00 and 9/18/00. The Kootenai River mean discharge including discharge at Libby Dam and the Fisher River during this period was 8090 cfs (standard deviation = 47.4; range 7956 – 8275 cfs). The fish was observed on a probable spawning migration in Quartz Creek on 9/18/00 through 9/27/00. This female bull trout remained above the falls for the remainder of 2000 and throughout 2001 and entered Quartz Creek again in 2001 for a third year of probable spawning. Although this is the first documented instance of fish migrating over Kootenai falls, in 2001 a second bull trout (tag number 49.650) was observed in upper Kootenai Falls on 7/18/01, but this was our last observation for this fish. We suspect that the transmitter battery failed shortly after this observation, and although we suspected that this fish might have also migrated over the falls, we could not confirm this suspicion. The Kootenai River discharge during the period July 18, 2001 to September 30, 2001 was 6278 cfs (standard deviation = 476.6; range 5826 – 9170 cfs). The upstream migration pattern we observed for these two bull trout shortly before spawning season was not a common movement pattern for all bull trout that migrated below Kootenai Falls. Although some of the fish below the falls did exhibit similar movements in the fall, others moved only slightly or not at all.

We were only able to document 4 out of the 65 (6.2%) radio tagged bull trout in tributaries to the Kootenai River during spawning season. These 4 bull trout included 2 fish entering Quartz Creek (including tag number 49.221 described above), one entered the Fisher River and one entered O'Brien Creek. We averaged 8 observations days per year during the spawning seasons (August to mid November) 1998 through 2001. Our strongest effort occurred during the 1998 spawning season where we searched on 14 separate occasions (days). However, our effort in 1999 and 2000 was approximately half of what it was during the 1998 spawning season where the number of observation days dropped to 7 days each year. Due to the higher occurrence of battery failure we observed during 2001 and 2002, we reduced our effort even further, with the number of observation days during the spawning season dropping to 4 and 1 day, respectively. Of the fish that were observed in tributaries two returned to Quartz Creek.

One female bull trout spawned in Quartz Creek in 2000 and 2001 (described above), but the other fish (tag number 49.210) was observed in Quartz Creek only in 2000. In 2000 both fish entered Quartz Creek between 7/20/00 and 9/15/00, one traveled in excess of five miles and the other traveled over eight miles. Both fish migrated out of Quartz Creek between 9/27/00 and 11/6/00. The fish that returned to spawn in Quartz Creek in 2001 began staging near the mouth of Quartz Creek around 7/18/01 and then entered the creek between 8/28/01 and 9/24/01. We were unable to document an out migration time for this fish. However, it is unknown whether the other fish that spawned in Quartz Creek the previous year migrated up Quartz Creek in 2001. The radio signal from tag number 49.210 was detected on 11 different occasions between 11/6/00 and 9/24/01 at the same location approximately 1.7 miles downstream from the confluence of Quartz Creek. Tag number 49.210 was last detected on 8/28/01 and 9/24/01, at which time the batter had been active for

727 days, which was approaching the life span for the tag. However, we made no attempts to locate this radio tag between 8/28/01 and 9/24/01. In 2000, this bull trout entered Quartz Creek before 9/15, and exited between 9/27 and 11/6/00. We are therefore unable to determine the location of this fish between 8/28/01 and 9/24/01.

We also observed a bull trout that was originally tagged in the Kootenai River directly below Libby Dam on 2/22/98; migrate into the Fisher River on a probable spawning run in 1998 and 1999. In 1998 the fish migrated into the Fisher River between 9/24/98 and 10/19/98, traveling over 30 miles upstream. The fish then started its out migration between 10/19/98 and 10/26/98 and didn't reach the Kootenai River until 1/3/99. The following year this fish migrated in to the Fisher River sometime after 8/25/99. We located the shed tag on a gravel bar approximately 26 miles up the Fisher River on 7/24/00, directly downstream of the confluence of the West Fisher River. It is likely that this fish spawned in the West Fisher River.

The only other bull trout that may have entered a tributary did so entering O'Brien Creek in 1999. This fish was originally captured and subsequently tagged and released in the Kootenai River directly below Libby Dam, and migrated below Kootenai Falls less than a month after being tagged. An aerial survey using a fixed wing aircraft on 8/25/99 estimated the location of this tag approximately 200 meters upstream from the O'Brien Creek confluence. However, we did not confirm this observation with an independent field reconnaissance effort. Our next attempt to locate this fish on 10/22/99 within the same general vicinity was unsuccessful. This fish was next detected in the Kootenai River between 12/10/99 and 1/10/00 approximately 2 miles upstream of the O'Brien Creek confluence.

Twelve other bull trout were detected near the confluence of the Fisher and Yaak rivers, O'Brien, Lake, and Quartz creeks during the late summer/fall of this study. However, we never observed these fish entering the tributaries during any of our aerial surveys. We observed five radio tagged bull trout in the vicinity of the Fisher River confluence. However, our search patterns can neither confirm nor deny that these fish entered the Fisher River. For example, 4 of the 5 fish that were located near the vicinity of the Fisher River confluence were unaccounted for when searches of the mainstem Kootenai River were conducted over a period ranging from 12 to 26 or more days during the spawning season. Searches within the Fisher River Basin were not conducted for these four fish. The fifth bull trout that was located near the Fisher River confluence during the spawning season was unaccounted for during the period October 13 to December 4, 1998. An aerial search of the mainstem Kootenai River and Fisher River Basin was conducted, but failed to locate that fish. Two bull trout were observed around the mouth of the Yaak River on September 24, 2001 and one bull trout was observed at this location on October 19, 1998. However, no searches were conducted for the two tagged bull trout located in 2001 until June 2002. The bull trout observed near the Yaak River confluence in 1998 was located 7 days later directly below Kootenai Falls, and remained there the rest of the year. Previous bull trout redd surveys and juvenile electrofishing surveys conducted by Montana FWP within the Yaak Basin, suggest that bull trout do not spawn in the Yaak Basin. Two were located near the O'Brien Creek confluence near the bull trout spawning period. One radio tagged bull trout was located near Kootenai Falls on 6/1/99 and unaccounted for until 10/22/99 when it was detected near the O'Brien Creek Confluence, and then unaccounted for until 12/10/99 when it was detected again at this location. We conducted aerial searches within the O'Brien Creek watershed on 7/8 and 8/25/99, but did not locate this fish. The other radio tagged bull trout that was located near the O'Brien Creek confluence was observed at this location in 1998 and 1999. This fish was detected 10/19/98 near the confluence, and then again on 10/26/98 near Kootenai Falls, leaving 6 days unaccounted for. The same fish was present approximately 2 miles downstream of Kootenai Falls in June 1999 and then unaccounted for the entire 1999 spawning season. We did not search for this fish in the O'Brien Creek drainage in 1998, however, aerial searches were conducted in the O'Brien Creek drainage twice in 1999 and once in the O'Brien, Lake, and Callahan creek drainages in 2000. These aerial searches did not locate either of these fish. The same bull trout that was present near the O'Brien Creek confluence in 1998 and 1999 was observed around the mouth of Lake Creek in 2000. This fish was also unaccounted for a portion of the 2000 spawning season for a period of 60 days (9/27-11/6). No surveys were conducted during this period.

The final bull trout observed near a tributary confluence was observed near Quartz Creek in 2000. This fish was present near the Quartz Creek confluence on 7/20/00 and then unaccounted for until 9/22/00 (64 days), when it was located approximately 3 miles downstream of the Quartz Creek confluence. We conducted 2 aerial searches within the Quartz Creek drainage during this period but failed to located this fish.

We identified several common locations that adult radio tagged bull trout frequented in the Kootenai River above Kootenai Falls. Common locations for above the falls vary slightly from season to season with some of the more popular year round areas being the Libby Dam tailrace area that extends from Libby Dam to approximately 2 miles downstream to confluence of Alexander Creek. We defined seasons of the years as follows; spring (April – June), summer (July – August), fall (September – November), and winter (December – March). During the spring and winter seasons radio tagged bull trout were frequently located in the tailrace area in addition to area between Dunn Creek confluence (RM 219.8) downstream to the Fisher River (RM 219.2) and the Jennings Rapids area (RM 217.3). Common year round holding areas in the lower Kootenai River below the falls included the area from Flemming Creek, Idaho (RM 137.6) to approximately five miles down river. However, during fall and winter seasons, tagged bull trout were frequently located between Throops Lake (RM 190.4) and the Sturgeon Hole (RM 191.4).

We attempted to assess seasonal bull trout movement by stratifying our radio tag observations based on season and location (above or below Kootenai Falls). We compared the mean number of days between detections, mean miles traveled between detections, and mean number of miles traveled between detections (miles traveled between detections divided by number of days between detections) between season and location using ANOVA and subsequent multiple comparisons. Significant differences existed between the mean number of days between detections ($p = 4.91*10^{-13}$; Table 3). The subsequent multiple comparisons revealed that 19 out of 26 possible comparisons were significantly different ($p = 0.05$; Table 3). However, we found no significant difference between the mean number of miles or the mean number of miles traveled per day each radio tagged bull trout traveled between detections when stratifying by season or location (above and below Kootenai Falls) ($p = 0.242$ and 0.144, respectively; Table 3). Since neither of these analyses suggested that seasonal bull trout movement differed by the location of the fish above or below Kootenai Falls, we pooled those observations from above and below the falls within a season and repeated the ANOVA (Table 4). In each instance, both ANOVA suggested that at least one pair wise comparison differed significantly ($p = 0.10$). Subsequent multiple comparisons indicated that both the mean miles traveled and the number of miles traveled per day between detections differed during the fall season from all other seasons. Bull trout during the fall season moved and average of 1.35 miles between detections, and an average of 0.10 miles per day (Table 4). We also pooled the seasonal movement information in an attempt to determine if movement differed between fish locations above or below Kootenai Falls. Although the mean distance traveled and the mean distance traveled per day between detections was higher for radio tagged bull trout below Kootenai Falls (seasons pooled), these differences were not significantly different than fish located above Kootenai Falls ($p = 0.727$ and 0.663, respectively; Table 4).

Table 3. The sample size, mean days between detection, mean distance traveled between detections, and mean distance traveled per day between detections for radio tagged bull trout in the Kootenai River. The analyses were stratified based on season and fish above and below Kootenai Falls. The p-value from the ANOVA testing for differences between seasons and above/below Kootenai Fall, and those pair wise comparisons that were significantly different are also given.					
Season	**Above or Below Kootenai Falls**	**Sample Size**	**Mean Days Between Detection**	**Mean Distance (miles) Traveled Between Detections**	**Mean Distance (miles) Traveled Per Day**
Spring	Above	47	16.9	0.58	0.049
Spring	Below	17	33.2	1.00	0.036
Winter	Above	53	16.5	0.50	0.038
Winter	Below	17	23.8	0.34	0.013
Fall	Above	36	13.7	1.35	0.073
Fall	Below	16	25.2	1.37	0.172
Summer	Above	45	12.8	0.28	0.037
Summer	Below	13	23.9	0.16	0.008
Overall Mean			18.1	0.66	0.051
ANOVA p-value			$4.91*10^{-13}$	0.242	0.144
<u>Non-</u> Significant (p > 0.05) pair wise comparisons				Spring Above/Winter Above Spring Above/Fall Above Winter Above/Fall Above Winter Above/Summer Above Winter Below/Fall Below Winter Below/Summer Below Fall Above/Summer Above	

Table 4. The sample size, mean distance traveled between detections, and mean distance traveled per day between detections for radio tagged bull trout in the Kootenai River. The analyses were stratified based on season and above and below Kootenai Falls. Fish from above and below Kootenai Falls were pooled from these analyses due to a lack of significant differences (above and below the falls), and by season (see Table 3). The p-value from the ANOVA testing for differences between seasons and by location above or below the falls is stated. Significantly different pair wise comparisons are also given.

Season	**Above/Below Kootenai Falls**	**Sample Size**	**Mean Distance (miles) Traveled Between Detections**	**Mean Distance (miles) Traveled Per Day**
Spring	Pooled	63	0.69	0.046
Winter	Pooled	70	0.46	0.032
Fall	Pooled	52	1.35	0.103
Summer	Pooled	58	0.25	0.031
Overall Mean			0.66	0.051
ANOVA p-value			0.035	0.087
Significant pair wise (p =0.10) Comparisons			Spring/Fall Winter/Fall Fall/Summer	Spring/Fall Winter/Fall Fall/Summer
Pooled	Above	181	0.63	0.048
Pooled	Below	62	0.74	0.059
ANOVA p-value			0.727	0.663

Burbot Monitoring

The burbot catch in our hoop traps below Libby Dam has declined precipitously since 1996–97 (Figure 12). A total of 7 and 5 burbot were captured during the 01-02 and 02-03 trapping seasons, respectively. These total catches each year translated into the lowest total catch and catch per effort (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, which may correlate with lower water temperatures. Since the 1995/1996 trapping season catch has significantly decreased ($r^2 = 0.779$; $p = 0.004$; Figure 12).

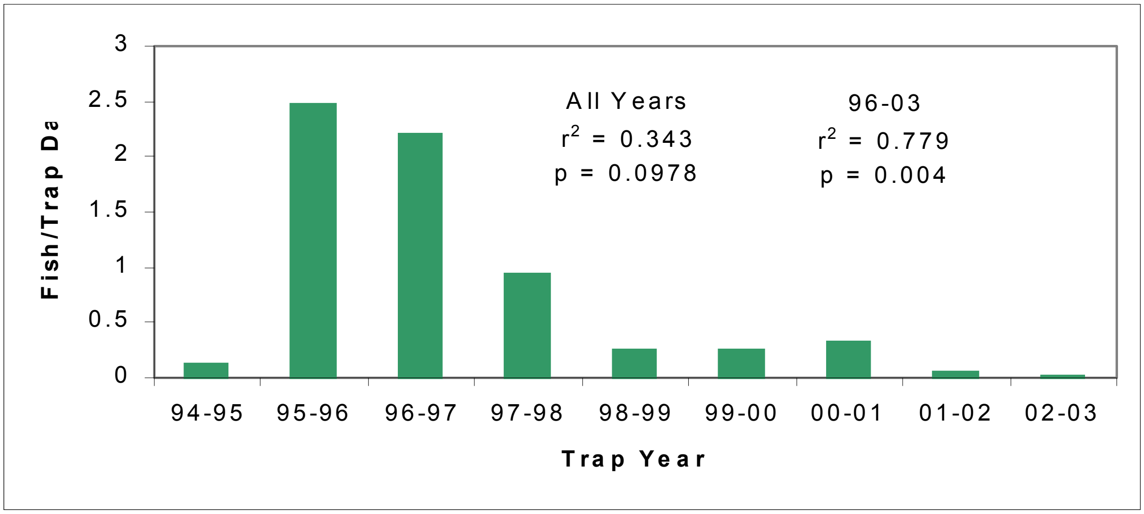

Figure 12. Total catch per effort (burbot per trap day) of baited hoop traps in the stilling basin downstream of Libby Dam 1994/1995 through 2002/2003. The traps are baited with kokanee salmon and fished during December and February.

Stream Macroinvertebrate Monitoring

Results were similar for sampling that occurred in the riffles at the Libby Creek and Grave Creek Restoration Projects, with the Species Richness, Ephemeroptera, Plecoptera, and Tricoptera (EPT) richness, and the Montana Biotic index highest for riffle samples collected at the Libby Creek Restoration Project, although differences were not significant for any of the three indices between streams ($p = 0.266$; 0.088, and 0.323, respectively; Table 5). Conversely, the Species Richness, EPT richness, and the Montana Biotic index were highest at the Grave Creek site for the 20 Jab Survey (Table 5). A statistical comparison between sites could not be performed for the 20 Jab surveys because of a lack of replication within streams for this sample method.

Macroinvertebrates were first surveyed in the Libby Creek Demonstration Project in 2000 using a similar methodology (Hoffman et al. 2002). The general trend for riffle samples from 2000 to 2002 was an increase of diversity for all species, EPT diversity and the Montana Biotic Index (Table 5). The overall species diversity estimates obtained from the riffle samples in 2002 (34.67) increased from samples collected in 2000 (28.67), although the differences were not significant ($p = 0.084$). However, the EPT richness estimates for riffle samples significantly increased in from 2000 to 2002 ($p = 0.038$; 18.33 and 24.0, respectively). The Montana Biotic Index from riffle surveys increased slightly between 2000 and 2002 (2.50 and 2.54, respectively), although the difference was not significant ($p = 0.887$).

Differences in the overall species diversity, EPT diversity and the Montana Biotic Index were not consistent between years for the 20 Jab samples at the Libby Creek Demonstration Project (Table 5). Overall species diversity and the Montana Biotic Index decreased from 2000 to 2002, but the EPT richness increased through time (Table 5). Statistical comparisons between years were not performed because the 20 Jab surveys lacked replication within a year.

Table 5. Measures of species richness, Ephemeroptera, Plecoptera, and Tricoptera (EPT) richness, and the Montana Biotic Index for the Libby Creek and Grave Creek Demonstration Restoration Projects sampled in 2002. Results from sampling that occurred in 2000 at the Libby Creek Demonstration Project are also presented.

Site and Sample Method	Species Richness	EPT Richness	MT Biotic Index
Libby Creek Demonstration Project: Riffle Surveys, 2002	34.67	24.00	2.54
Libby Creek Demonstration Project: 20 Jab Survey, 2002	29.00	23.00	2.07
Libby Creek Demonstration Project: Riffle Surveys, 2000	28.67	18.33	2.50
Libby Creek Demonstration Project: 20 Jab Survey, 2000	42.00	18.00	3.82
Grave Creek Demonstration Project: Riffle Surveys,2002	32.67	21.00	2.43
Grave Creek Demonstration Project: 20 Jab Survey, 2002	48.00	25.00	2.37

Juvenile Salmonid Population Estimates

Sinclair Creek

Westslope cutthroat trout abundance in Section 1 of Sinclair Creek was 73.2% higher after the implementation of the habitat restoration project in 2002 (97 fish per 1000 feet) compared to the mean prior to project implementation for the period 1997-1999 (56.7 fish per 1000 feet). However, a statistical comparison between pre- and post-project fish abundance estimates was not possible due to lack of annual replication following project implementation. Additionally, the annual variation associated with the pre-project fish data was high, which resulted in a 95% confidence interval that exceeded the 2002 point estimate (Figure 13; Table A1). Although cutthroat trout abundance was positively correlated with time (year), the relationship was not significant ($p = 0.331$; $r^2 = 0.448$). Conversely, mean brook trout abundance in Section 1 of Sinclair Creek were 23.7% lower after the implementation of the restoration project than compared to pre-project brook trout abundance (57 and 74.7 fish per 1000 feet, respectively, Figure 13). Brook trout abundance from 1997-2002 was highly variable, with no significant trend ($r^2 = 0.0005$; $p = 0.976$).

Westslope cutthroat trout and brook trout in Section 2 of Sinclair Creek increased 25.2 and 14.3%, respectively after the implementation of the stream restoration project on lower Sinclair Creek, however differences between treatments were not significant ($p =$ 0.710 and 0.560, respectively; Figure 14). Although both cutthroat trout and brook trout show a weak general trend of increasing abundance through time ($r^2 = 0.326$ and 0.421, respectively), neither relationship was significantly different than a stable population ($p =$ 0.237 and 0.163, respectively; Table A1).

Cutthroat trout abundance in Section 3 of Sinclair Creek was significantly higher in Section 3 than either Section 1 or 2 ($p < 0.05$), but cutthroat trout abundance between Sections 1 and 2 was similar ($p > 0.05$; Table A1). Brook trout abundance between all three sections was also similar ($p = 0.389$). Cutthroat trout abundance in Section 3 of Sinclair Creek has varied from 1985-2002, with no apparent trend through time ($r^2 = 0.326$; $p =$ 0.429), although this section was not sampled in 2000 or 2001 (Figure 15). Brook trout abundance in Section three was less variable than cutthroat trout abundance, and exhibited a general trend of increasing abundance through time at this section ($r^2 = 0.699$; $p = 0.078$; Figure 15).

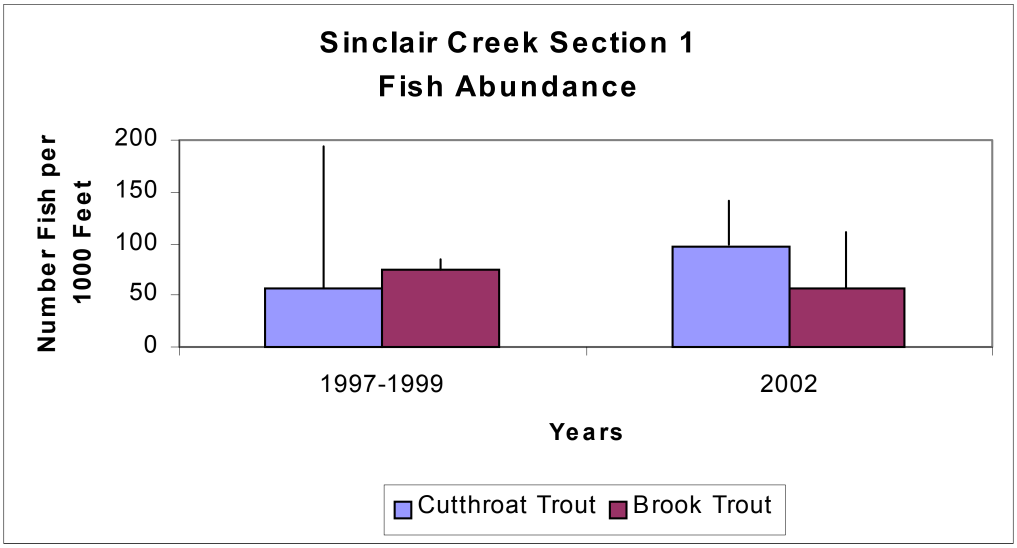

Figure 13. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Sinclair Creek Section 1 monitoring site located within the Sinclair Creek restoration project area using a backpack electrofisher. Data from 1997-1999 represent mean pre-project trends in fish abundance, and 2002 represents fish abundance estimates collected after project implementation. Upper 95% confidence intervals are represented by the whisker bars.

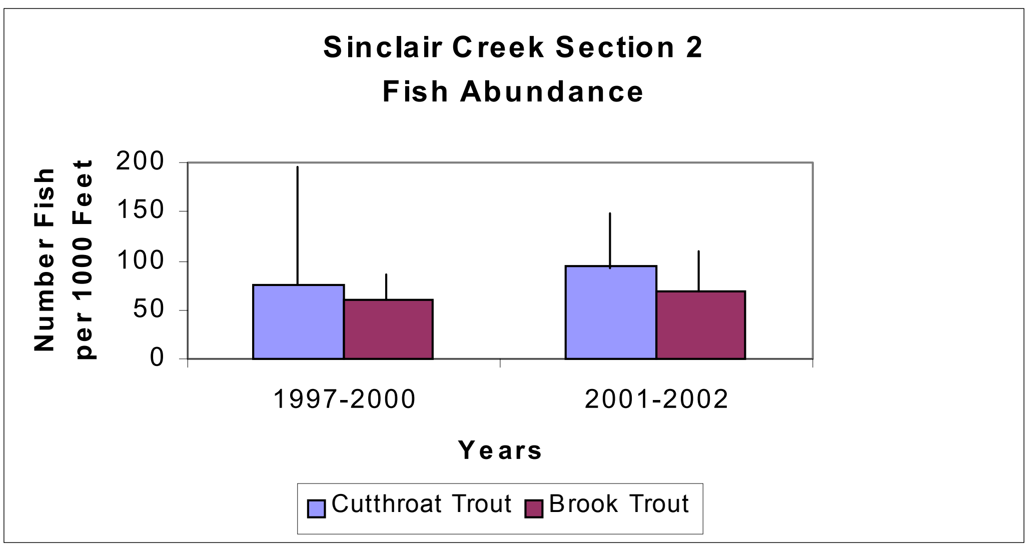

Figure 14. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Sinclair Creek Section 2 monitoring site located within the Sinclair Creek restoration project area collected using a backpack electrofisher. Data from 1997-2000 represent pre-project trends in fish abundance, and 2002 and 2001 represents fish abundance estimates collected after project implementation. Upper 95% confidence intervals are represented by the whisker bars.

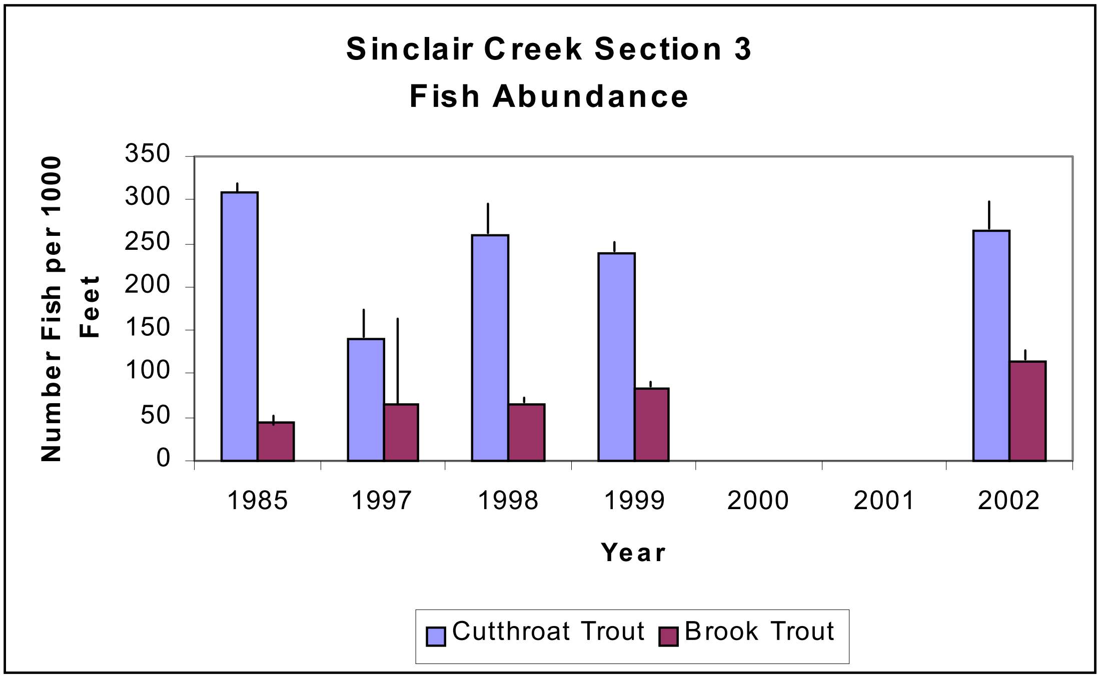

Figure 15. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Sinclair Creek Section 3 monitoring site from 1985-2002 collected using a backpack electrofisher. Upper 95% confidence intervals are represented by the whisker bars.

Therriault Creek

Rainbow trout abundance in Section 2 of Therriault Creek has generally increased from 1997-2001 by an average of 12.1 fish per 1000 feet per year ($r^2 = 0.714$; $p = 0.155$; Table A2), ranging from 36 to 93 fish per 1000 feet in 1997 and 2001, respectively (Figure 16). Estimated brook trout abundance in this section peaked in 1998 at 125 fish per 1000 feet, but the overall trend for the period was not significantly different than a stable population ($r^2 = 0.005$; $p = 0.929$; Figure 16; Table A2). The overall average abundance of brook trout from 1997 to 2001 was 83.75 fish per 1000 feet. Our estimates of bull trout abundance have decreased each year since 1997 by an average of 10.1 fish per 1000 feet per year ($r^2 = 0.672$; $p = 0.180$; Figure 16; Table A2).

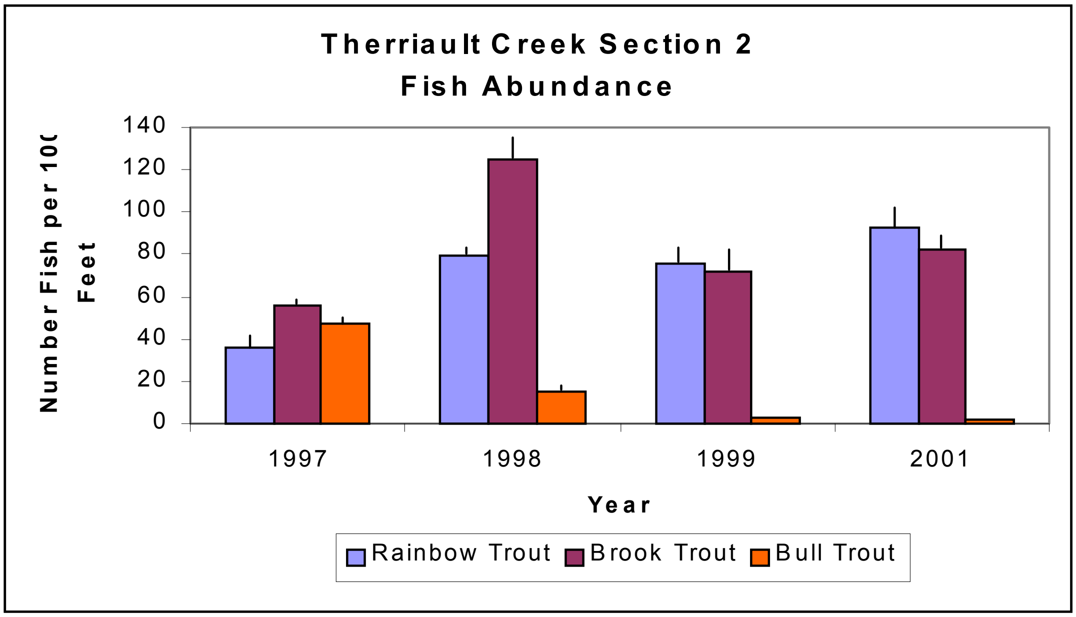

Figure 16. Cutthroat trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 2 monitoring site from 1997-2001 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

Grave Creek

Juvenile bull trout were most abundant in the lower Grave Creek Section in 2001 (Table A3). Cutthroat trout were the most abundant salmonid species sampled in 2000, and rainbow trout was the most abundant salmonid species observed in 2002 (Table A3). We were unable to make statistical comparisons between pre-project and post-project fish abundance estimates collected within the Grave Creek Demonstration Project due to the lack of replication of post-project data, and the inconsistent sampling methodology between years. However, the variability in pre-project fish abundance estimates is high (Figure 17) and therefore will likely reduce our ability to distinguish statistical differences in abundance between years in future comparisons.

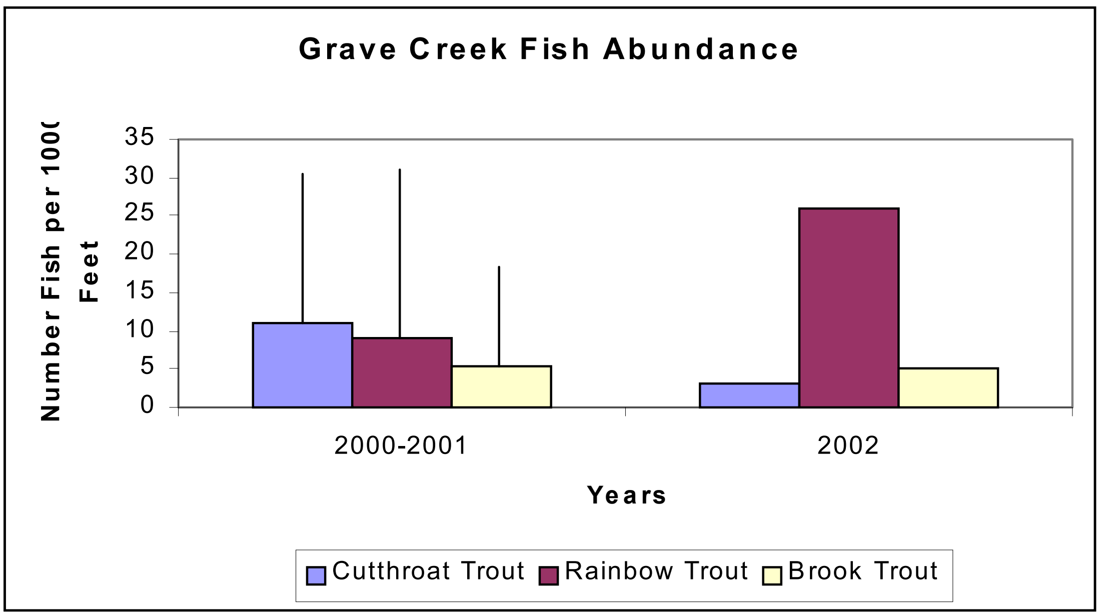

Figure 17. Cutthroat trout, rainbow trout and brook trout densities (fish per 1000 feet) within the Grave Creek Demonstration Project area. Data collected during 2000 and 2001 represent pre-project implementation fish abundances and were collected using single pass electrofishing. Fish abundance data 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.

Young Creek

Fish abundance in Section 1 of Young Creek has demonstrated consistent and similar trends for cutthroat, rainbow and brook trout since 1997 (Figure 18; Table A4). Rainbow and brook trout abundance peaked in 1998, and have declined since. Mean annual abundance for rainbow trout was 20 fish per 1000 feet. Brook trout abundance was higher, with a mean abundance of 59.8 fish per 1000 feet. Cutthroat trout abundance peaked in 1999 at 139 fish per 1000 feet (Figure 18). Trends for all three species were not significantly different than a stable population (zero slope; $p > 0.4$).

Cutthroat trout were the most abundant fish species in Section 4 of Young Creek (Table A4), and exhibited a similar trend as cutthroat trout abundance in Section 1 (Figure 18 and 19, respectively). Cutthroat trout abundance peaked in 1998 at 439 fish per 1000 feet. Overall the population was not significantly different than a stable population ($r^2 = 0.001$; $p = 0.947$). The mean annual cutthroat trout abundance was 233 fish per 1000 feet. Brook trout were observed at this site in 1999 and 2001-2002 at a substantially lower abundance than cutthroat trout (Figure 19). Mean annual brook trout abundance was 2.2 fish per 1000 feet, and peaked in 2001 at 6 fish per 1000 feet.

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 203.8 cutthroat trout per 1000 feet and 40.8 brook trout per 1000 feet (Figure 20; Table A4). 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 20).

Cutthroat trout and brook trout abundance differed between sampling sections within Young Creek. Mean annual cutthroat trout abundance estimates were highest for Sections 4 and 5, and were both significantly higher than cutthroat trout estimates in Section 1 ($p < 0.05$). However, cutthroat trout estimates were not significantly different between Sections 4 and 5. Mean annual brook trout abundance estimates in Sections 1 and 5 were significantly higher than brook trout estimates in Section 4 ($p < 0.05$), and although mean annual brook trout abundance in Section 1 was 47% higher than Section 5, the difference was not significant ($p>0.05$).

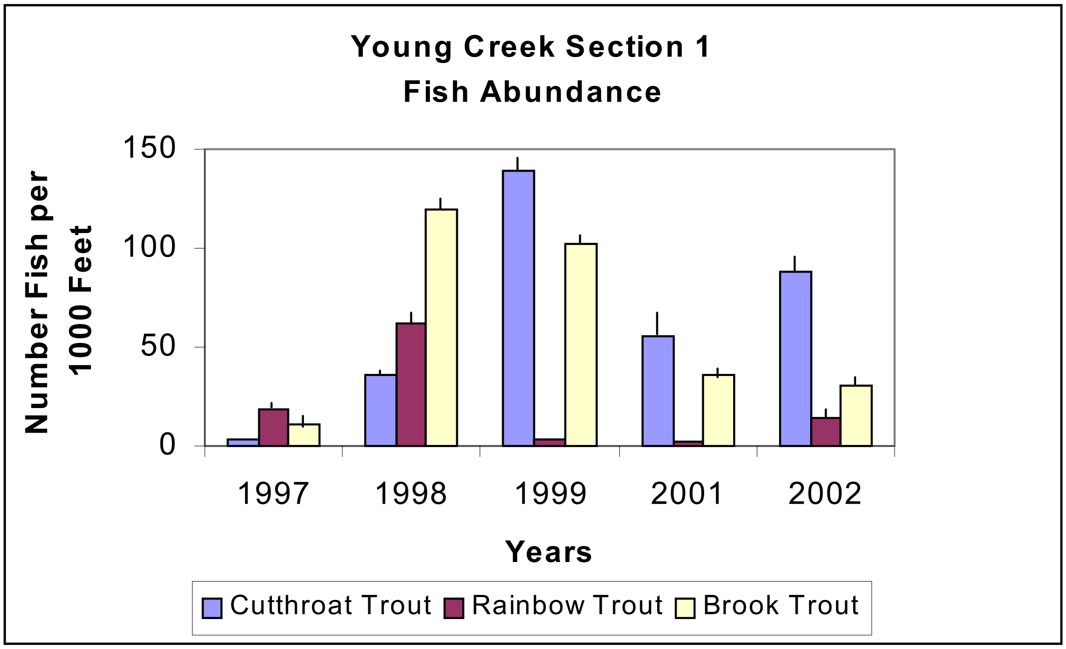

Figure 18. Cutthroat trout, rainbow trout and brook trout densities (fish per 1000 feet) within the Young Creek Section 1 monitoring site from 1997-2002 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

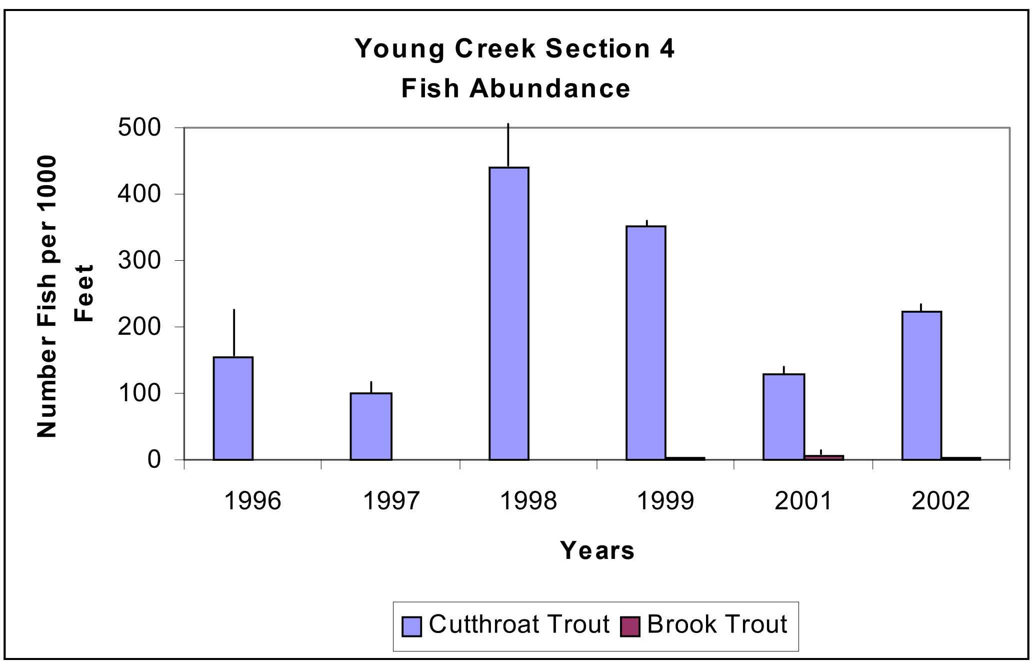

Figure 19. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Young Creek Section 4 monitoring site from 1997-2002 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

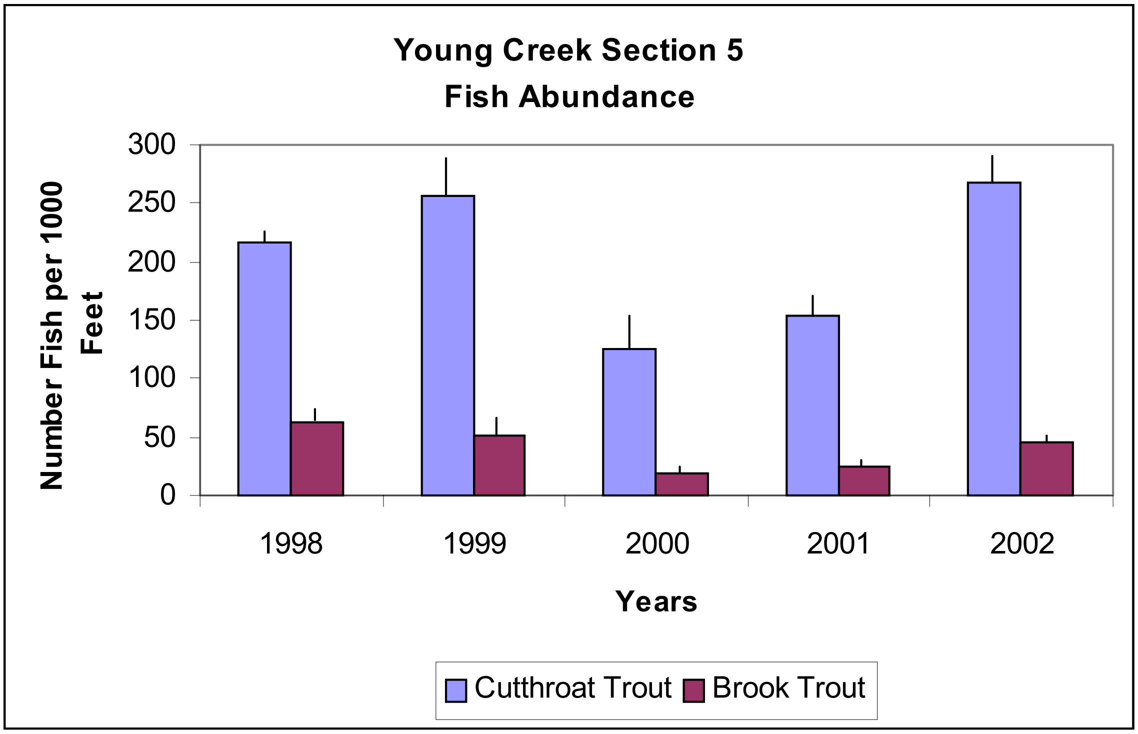

Figure 20. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Young Creek Section 5 monitoring site from 1997-2002 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

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 in 2001 was collected prior to project implementation. Therefore, only a single year (2002) of post project fish monitoring data exists. Nevertheless, rainbow trout and brook trout abundance was 68.3 and 82.9% higher, respectively in 2002 than compared to the mean prior to project implementation (Figure 21; Table A5). A statistical comparison between pre- and post-project fish abundance estimates was not possible due a single year of data following project implementation. There is no apparent temporal trend in rainbow trout abundance within this section ($r^2 = 0.113$; $p = 0.579$), although brook trout are generally becoming slightly more abundant at this site ($r^2 = 0.750$; $p = 0.058$). Juvenile bull trout were first observed in this section in 2002, with an estimated abundance of 3 fish per 1000 feet.

Section 2 of Libby Creek was sampled in 1998 and 2001 (Table A5). Rainbow trout were substantially more abundant at this section than brook trout during both years (Figure 22). We estimated 203 and 148 fish per 1000 feet in 1998 and 2001, respectively. Bull trout were observed in this section only in 1998, with an estimated abundance of 5 fish per 1000 feet.

Our estimates of rainbow trout abundance in Section 3 of Libby Creek were similar between 2000 and 2002 (Figure 23; Table A5), with no evidence that the population differed from a stable population ($p = 0.469$; $r^2 = 0.548$). No brook trout were observed at this site. Estimates of juvenile bull trout abundance at this site ranged from 3 to 8 fish per 1000 feet over the three years (Figure 23). Rainbow trout were significantly more abundant at Sections 2 and 3 than Section 1 ($p < 0.05$), but not significantly different between Sections 2 and 3.

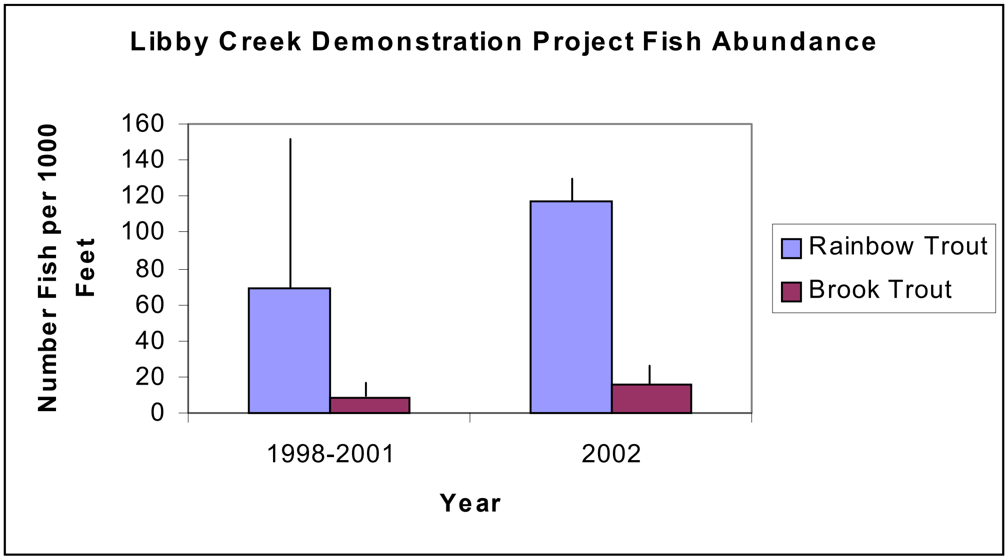

Figure 21. Rainbow trout and brook trout densities (fish per 1000 feet) within the Libby Creek Demonstration Project area, comparing annual mean pre-project (1998-2001) data and post-project (2002) using mobile electrofishing gear. Upper 95% confidence intervals are represented by the whisker bars.

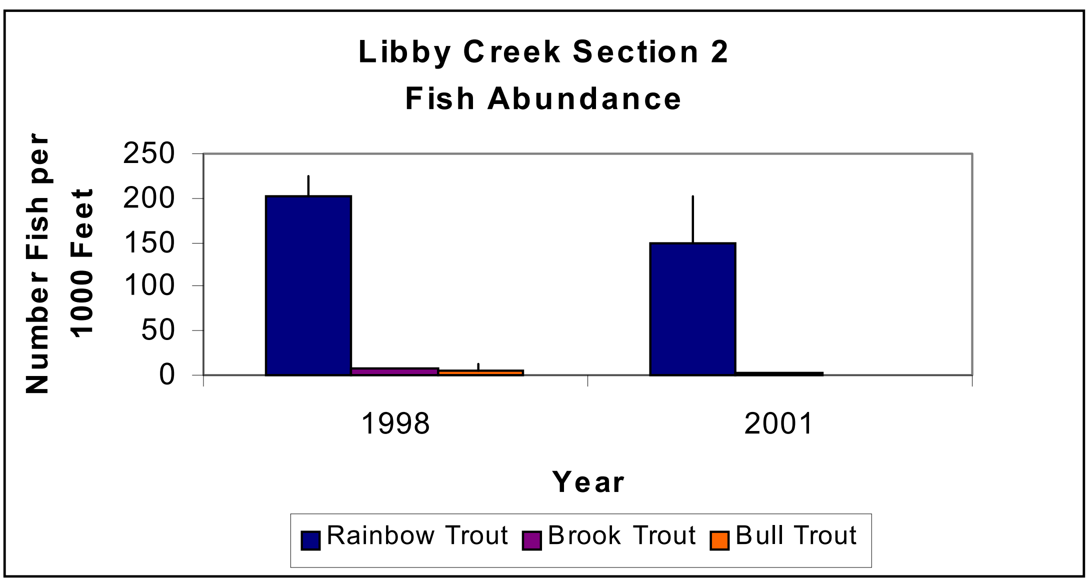

Figure 22. Rainbow trout, bull trout, and brook trout densities (fish per 1000 feet) within the Libby Creek Section 2 monitoring site in 1998 and 2001 using a backpack electrofisher. Upper 95% confidence intervals are represented by the whisker bars. The site was not sampled in 2002.

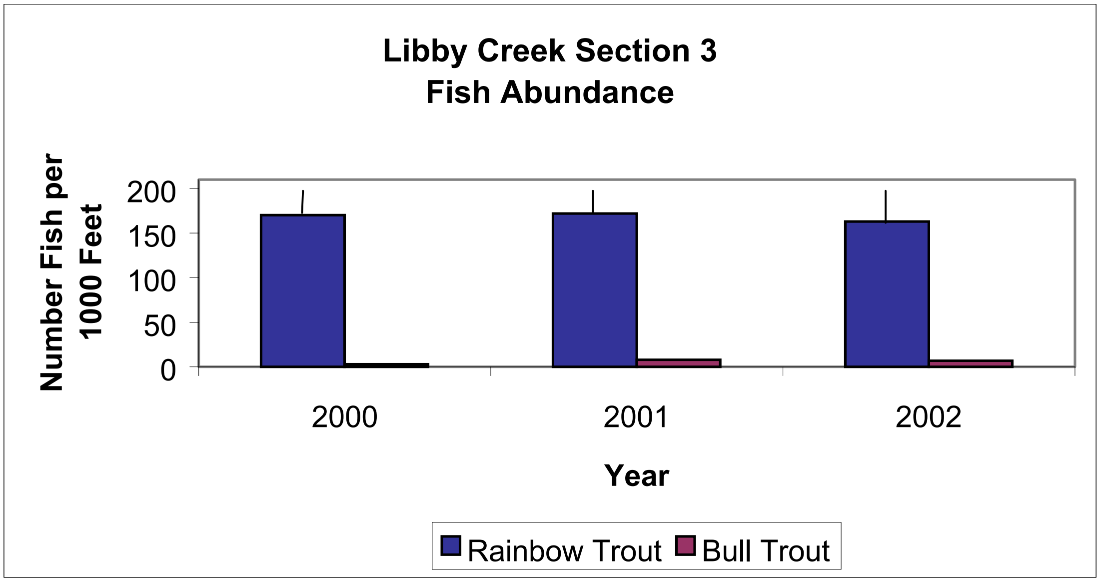

Figure 23. Rainbow trout and bull trout densities (fish per 1000 feet) within the Libby Creek Section 3 monitoring site in 2000-2002 using a backpack electrofisher. Upper 95% confidence intervals are represented by the whisker bars. This site is located within the upper Libby Creek restoration project area. These data represent pre-project trends in fish abundance.

Parmenter Creek

Rainbow trout were the most abundant fish species observed in Parmenter Creek in 2000 and 2001 (Figure 24), with estimates of 92 and 79 fish per 1000 feet, respectively (Table A6). 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 24). 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 A6).

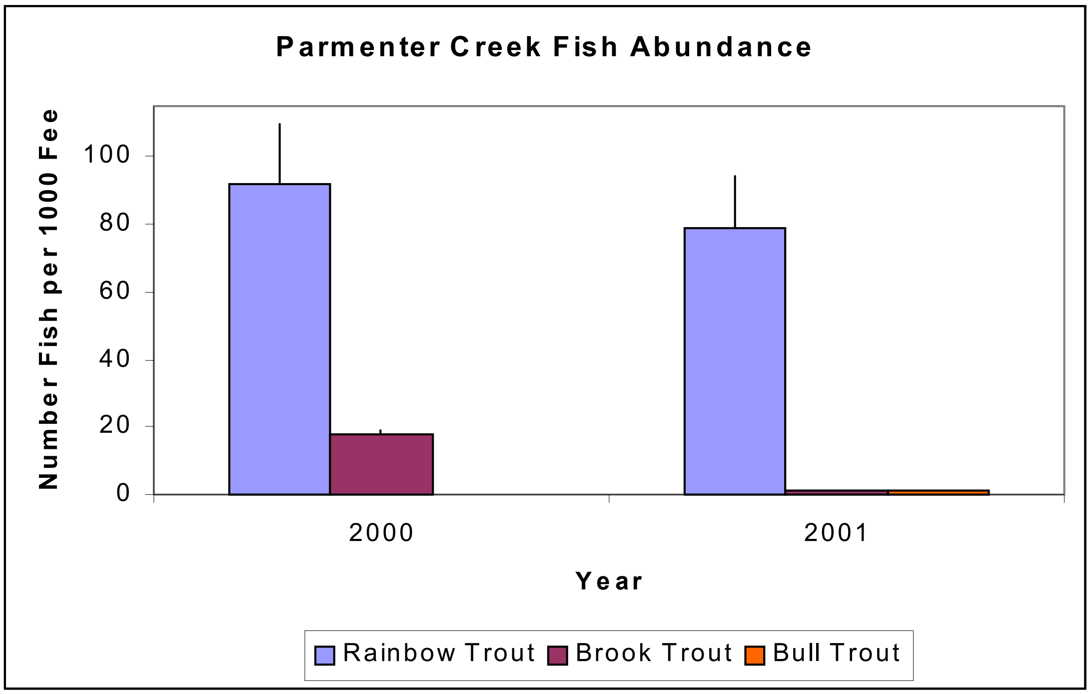

Figure 24. Cutthroat trout, bull trout and brook trout densities (fish per 1000 feet) within the Parmenter Creek monitoring site collected by performing backpack electrofishing. Fish abundance estimates from 2000 represent pre-project information, and surveys conducted in 2001 represent post-project 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 (Table A7), with estimates of 73 and 42 fish per 1000 feet, respectively (Figure 25). We also estimated 3 brook trout per 1000 feet in 2002, but did not observe any brook trout in 2001 (Table A7). We also captured were 43 mountain whitefish ranging from 51 to 105 mm and one pumpkinseed.

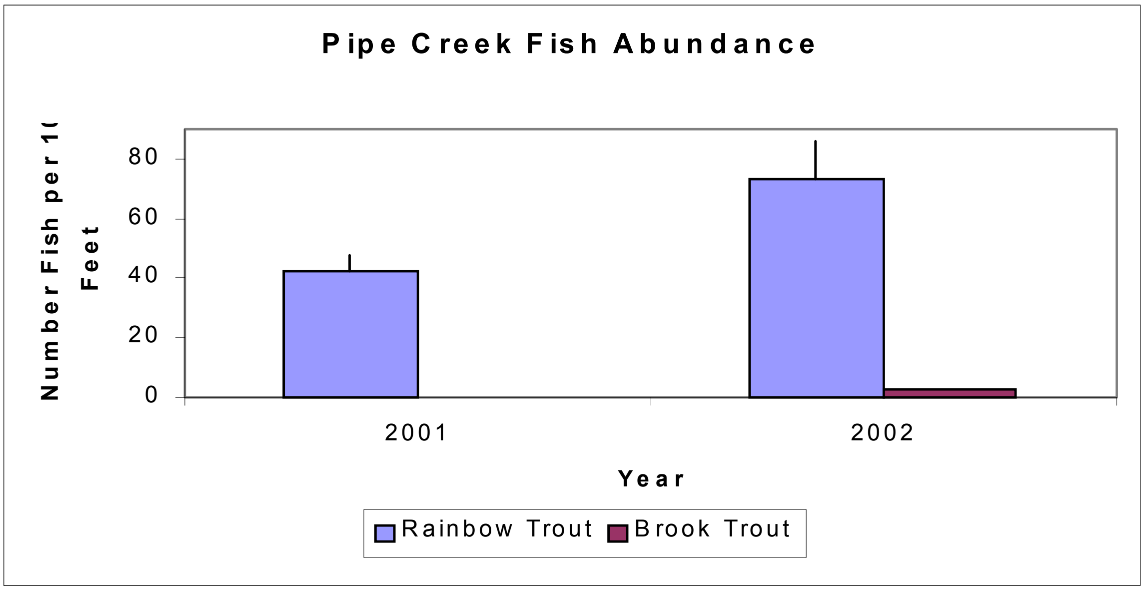

Figure 25. Cutthroat trout and brook trout densities (fish per 1000 feet) within the Pipe Creek monitoring site from 2000 and 2001 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars. The site was not sampled in 2002.

Barron Creek

Brook trout was the most abundant salmonid species observed in Barron Creek at all sampling sections, especially at Section 3, where we estimated 301 brook trout per 1000 feet (Figure 26; Table A8). Hybridization between rainbow trout and cutthroat trout was prolific at Sections 1-3; we therefore combined abundance estimates for cutthroat trout, rainbow trout and hybrids at these sections. Cutthroat, rainbow and hybridized trout were most abundant at Section 1, where we estimated 30 trout per 1000 feet (Figure 26; Table A8). We did not observe any *Oncorhynchus* species at Section 4.

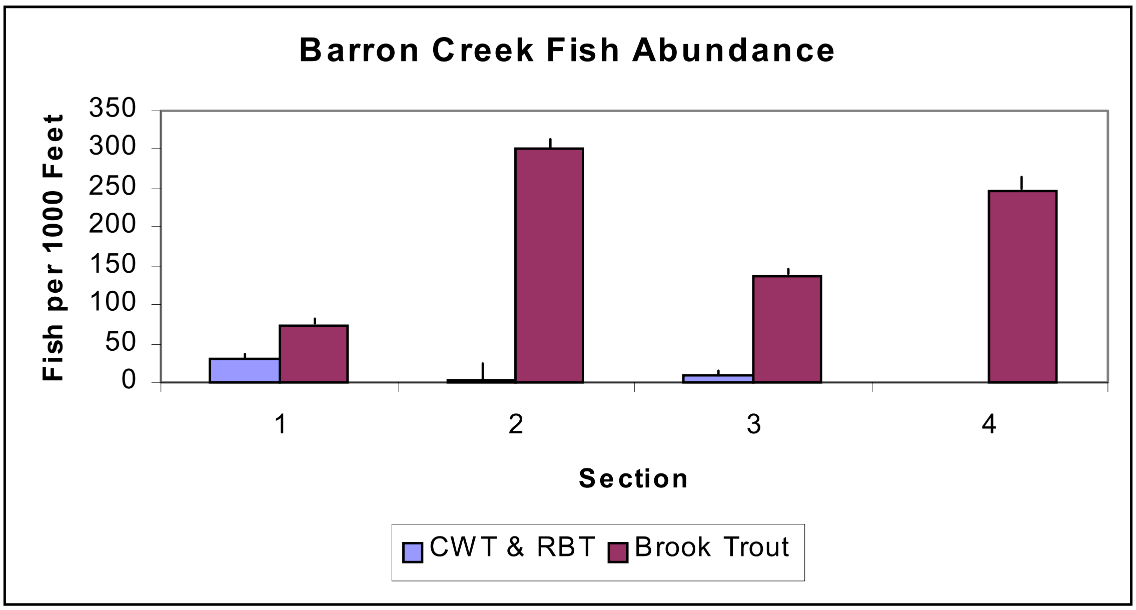

Figure 26. Oncorhynchus species (cutthroat trout and rainbow trout combined) and brook trout densities (fish per 1000 feet) within 4 sections of Barron Creek in 2002 collected by performing backpack electrofishing. Upper 95% confidence intervals are represented by the whisker bars.

Koocanusa 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). Kamloops rainbow trout were first introduced in 1985 by British Columbia Ministry of Environment (BCMOE). 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 trend in abundance (Figure 27). Average length of kokanee varied among years. Average length and weight between 1988 and 2002 was 292.0 mm and 239.2 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).

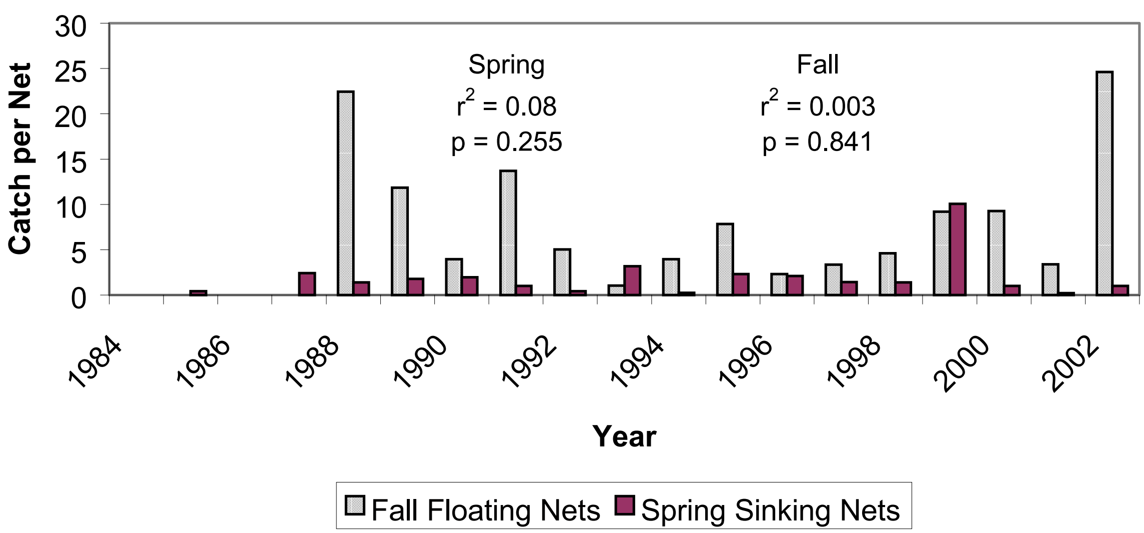

Figure 27. Average catch per net of kokanee for fall floating (1988-2002) and spring sinking (1984-2002) gill nets in Koocanusa Reservoir.

Table 6. Average length and weight of kokanee salmon captured in fall floating gillnets (Tenmile and Rexford) in Libby Reservoir, 1988 through 2002.

YEAR	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	AVG.
Sample size (n)	2150	1259	517	624	250	111	291	380	132	88	76	200	342	120	357	
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	292.0
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	239.2

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). Mountain whitefish catch rates since 1989 (mean catch = 0.8 fish per net) have significantly decreased ($p = 0.0003$) from the initial years following impoundment (1975-1988; mean catch = 3.5 fish per net). Catch rates since 1988 remained low; with mountain whitefish comprising an average of 1.1% of the spring catch during 1988 through 2002. Reasons for whitefish decline in Libby Reservoir may include conversion from lotic to lentic environment, barriers to spawning habitat and poor quality of that habitat, and loss of spawning substrate in the old Kootenai River channel.

Rainbow and Westslope Cutthroat Trout

Rainbow trout and westslope cutthroat trout catch have both significantly declined since the impoundment of Koocanusa Reservoir (Figure 28). Rainbow trout catch per net since 1975 has declined more precipitously than cutthroat trout catch per net. However, catch statistics for both species exhibit similar trends. Rainbow trout have exhibited two general trends since impoundment. The first trend was the initial decline in abundance from 1975 to 1989, which is characterized by significant decline (Figure 28), followed by a period of relative stability from 1990 to 2002, where the average catch per net during this period (mean fish per net = 0.313) was not significantly different than a stable population (zero slope; Figure 28). 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. The first is a significant and precipitous decline during the early years of impoundment from 1975 to 1986 (Figure 28), 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 2002,and is characterized by a significantly lower level of abundance (0.13 fish per net; $p = 0.001$), and a somewhat stable level ($r^2 = 0.013$; $p = 0.768$). The general trend of cutthroat trout relative abundance observed between 1987 and 1993 may be an artifact of the presence of hatchery cutthroat trout stocked in the reservoir during this period (Table 7). Hatchery cutthroat trout were last stocked in the reservoir in 1994.

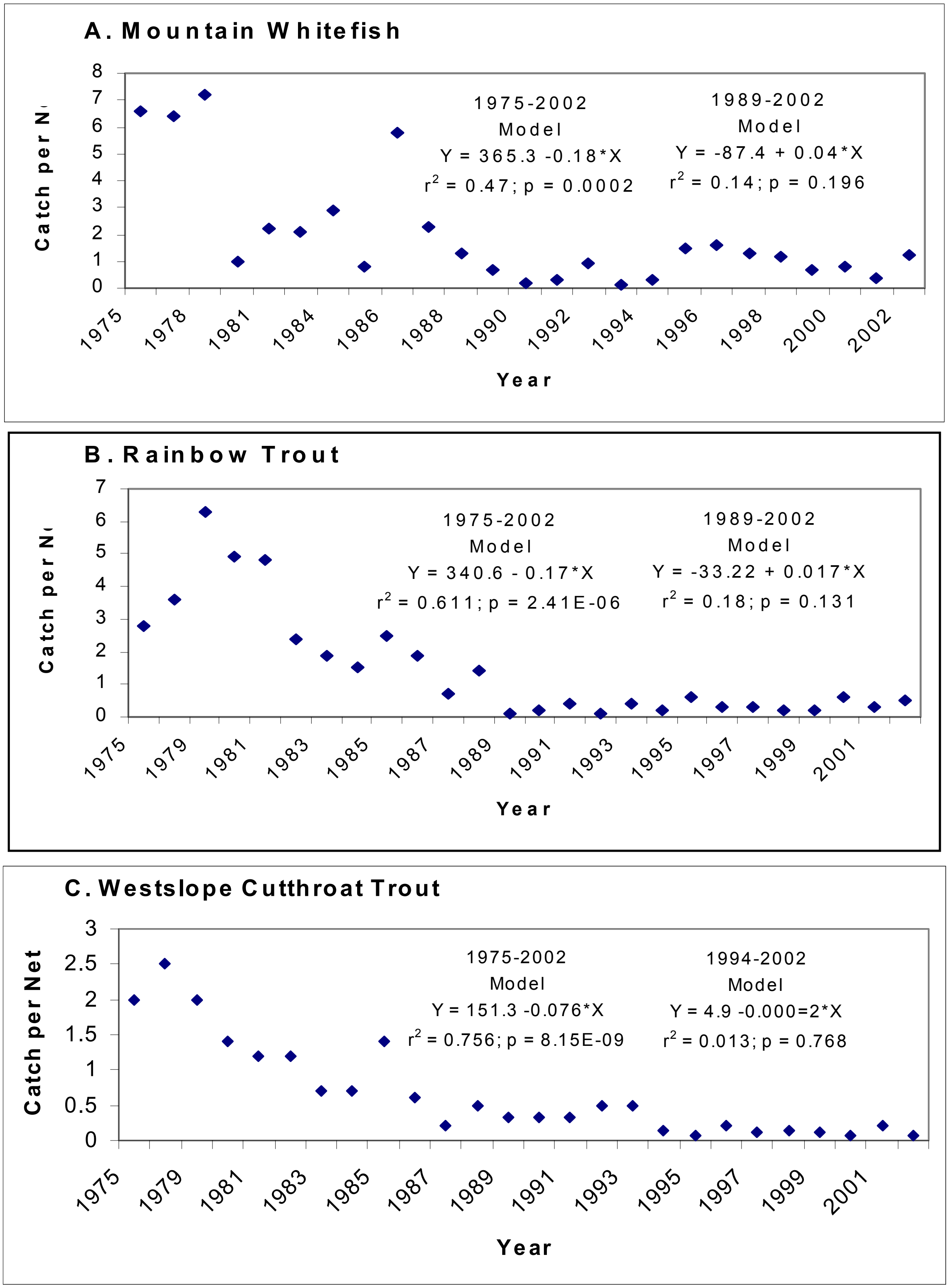

Figure 28. Mean catch rates (fish per net) of three native species (mountain whitefish (a) in spring sinking gillnets in the Rexford area, rainbow and westslope cutthroat trout (b) and (c) in floating gillnets from Tenmile and Rexford areas in Libby Reservoir, 1975 through 2002. The Tenmile area was not sampled during the fall in 2001 or 2002.

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 2002. The Tenmile location was not sampled in 2001 and 2002.

	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
No. Caught	0.11	0.14	0.11	0.07	0.21	0.07
Avg. Length (mm)	225	267	305	302	259	305
Avg. Weight (gm)	128	228	296	271	175	256
No. Stocked	none	none	none	none	none	none
Length (mm)	n/a	n/a	n/a	n/a	n/a	n/a

Kamloops Rainbow Trout (Duncan Strain)

Kamloops rainbow trout were first introduced to Koocanusa 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 ($P=0.002$; $r^2 = 0.63$; Table 8) for 1988 through 1999. However, the catch rate of Kamloops rainbow trout in fall floating gillnets shows no significant trend (Figure 29; $r^2 = 0.136$; $p = 0.177$). Catch rates for Kamloops rainbow trout in fall gillnets has been low since 1996.

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	
No. Caught	2	1	2	3	3	0	0	
Avg. Length (mm)	460	395	376	378	395	N/A	N/A	
Avg. Weight (gm)	1192	518	450	504	555	N/A	N/A	
No. Stocked	12,561	22,610	16,368	13,123	none	none	29,546	
Length (mm)	170.5	152-178	127-152	255-280	N/A	N/A	80.3	

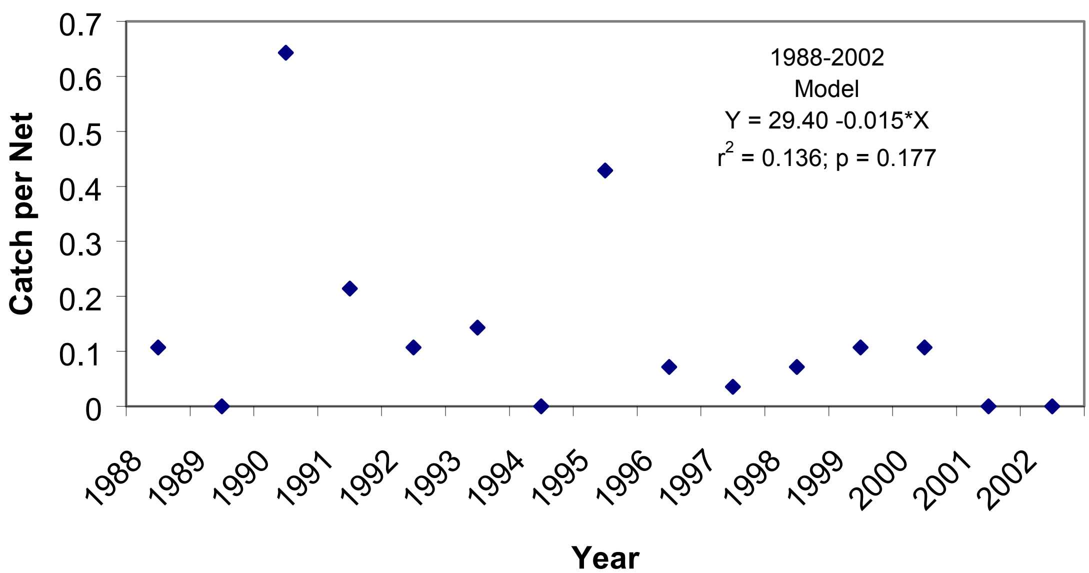

Figure 29. Average catch (fish per net) of Kamloops rainbow trout (Duncan strain) in fall floating gill nets in Koocanusa Reservoir at the Rexford and Tenmile sites 1988-2002. The Tenmile site was not sampled in 2001 or 2002.

Bull Trout

Spring gill net catch of bull trout has shown a significant increasing trend over the past 13 years (Figure 30). Bull trout redd counts (see above) in both the Wigwam River and Grave Creek are both significantly and positively correlated ($r^2 = 0.562$; $p = 0.03$ and $r^2 = 0.485$; $p = 0.02$, respectively) to spring gill net catch rates for bull trout. However, our fall gill netting series typically captures few bull trout. The primary reasons are that sampling dates purposely coincided with the period in which adults were in spawning tributaries, and that bull trout are not traditionally captured in floating gillnets.

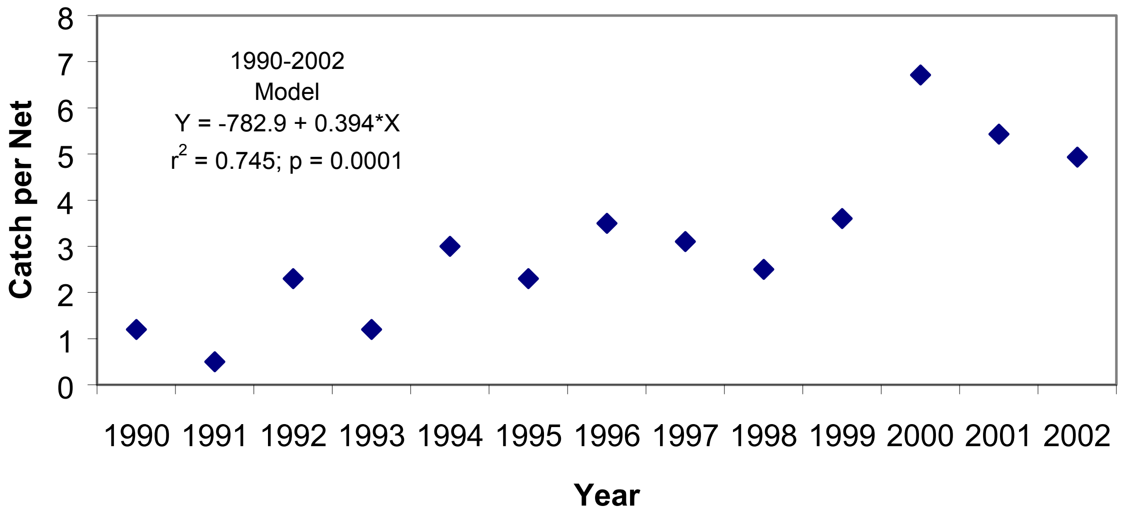

Figure 30. Average catch per net of bull trout in spring gill nets at the Rexford site on Koocanusa Reservoir.

Burbot

Burbot catch rates in spring sinking gillnets since 1990 show no clear trend in abundance (Figure 31; r2 = 0.09; p = 0.325). Burbot catch per net for spring sinking nets has averaged 0.3 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 (r2 = 0.52; p = 0.04; Figure 32) to daily catch of burbot in baited hoop traps in the stilling basin below Libby Dam (see above), suggesting that burbot abundance in Koocanusa Reservoir may be influencing burbot abundance in the Kootenai River below Libby through entrainment.

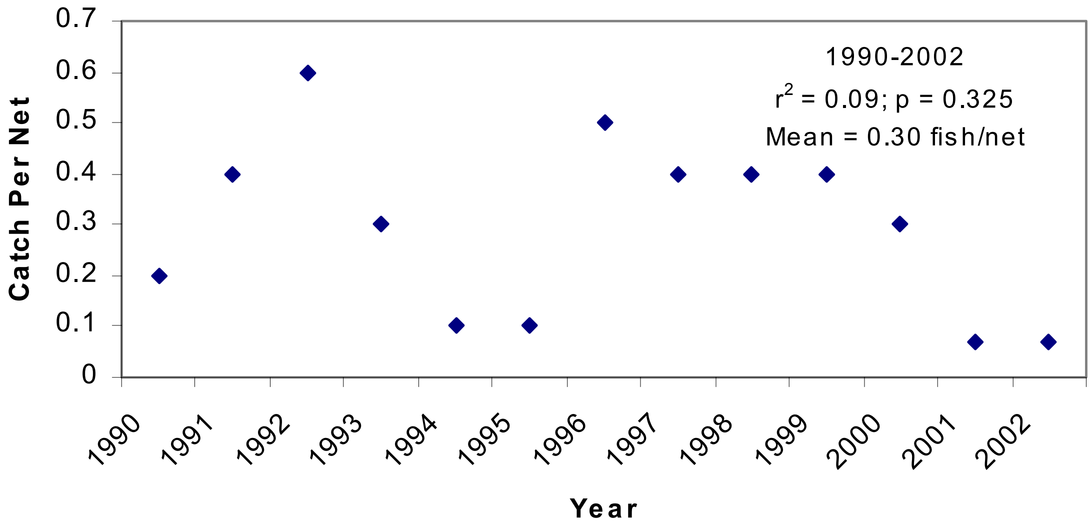

Figure 31. Mean catch per net of burbot in sinking gillnets during spring gillnetting activities at the Rexford site on Koocanusa Reservoir, 1990-2002. The mean catch per net during the period was 0.30 fish per net.

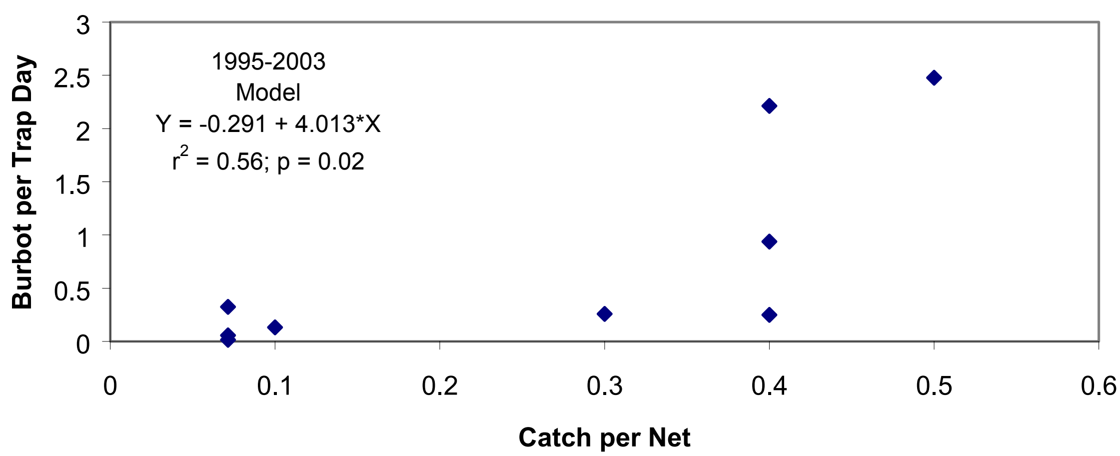

Figure 32. The relationship between mean burbot catch per net for spring sinking gillnets on Koocanusa Reservoir and burbot catch rates (fish/trap day) of baited hoop traps in the stilling basin below Libby Dam.

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 33; $r^2 = 0.31$; $p = 0.003$; 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 33; $r^2 = 0.07$; $p = 0.202$; Table 10). Species composition for the catch of fall and spring gillnets has remained relatively stable since 1988 (Table 10).

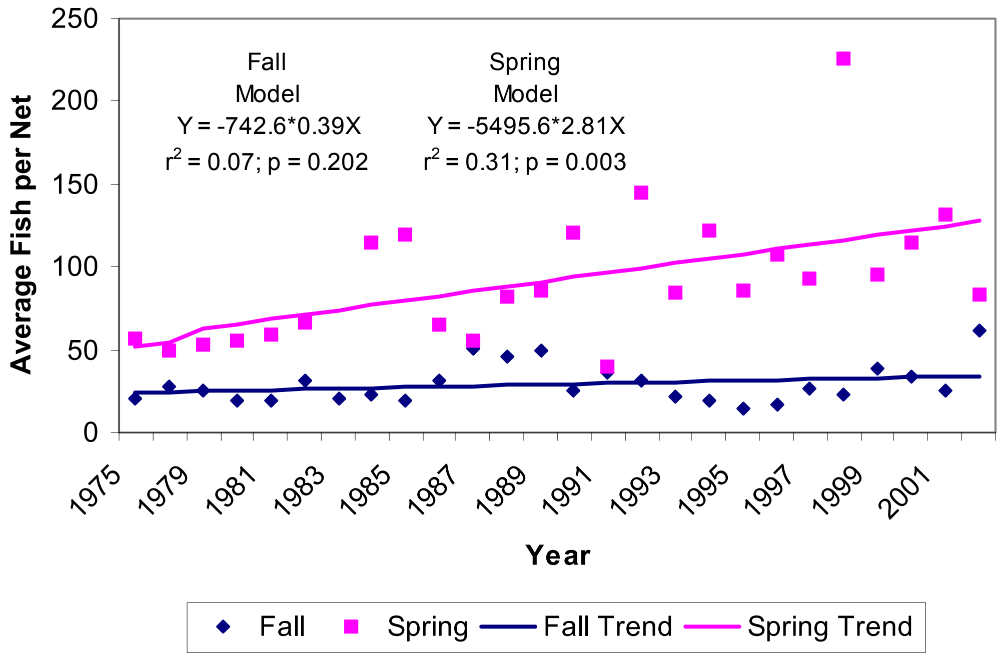

Figure 33. Catch per net (all species combined) in fall floating and spring sinking gillnets and associated trend lines in Libby Reservoir, 1975 through 2002.

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 2002.

	YEAR												
	1990	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
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
Number of Floating Nets	54	28	28	28	28	28	28	28	28	28	28	14	14
Reservoir Elevation	2456	2448	2421	2441	2446	2454	2450	2448	2439	2453	2434	2433	2441
	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
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
RB X WCTĐ	0.3	0.2	0.2	0	0	0	0	0	<0.1	0	0	0	0
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
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
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
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
RSS	0	0.1	0	0	0.3	0.2	0.1	1.0	0.3	0.3	<0.1	0	0.3
BT	0	0	0.1	0.3	0	1.2	<0.1	0	<0.1	<0.1	0.2	0	0.1
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
KOK	3.9	13.7	5	1	4	7.9	2.3	3.1	2.7	7.3	8.0	2.1	14.2
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

*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 2002.

	YEAR												
	1990	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**
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
Number of Sinking Nets	27	28	28	28	28	28	28	28	27	28	14	14	14
Reservoir Elevation	2358	2330	2333	2352	2405	2386	2365	2350	2417	2352	2371	2392	2384
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
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
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
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
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
CRC	104.8	31	119	63.3	94.2	54.1	60.9	51.1	171.7	54.4	76.4	25	24.1
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
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
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
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
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
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
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
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
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

*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 2002. 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		Average	
	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	1.4	0.4
WCT	0.6	0.4	0.5	0.1	0.3	0.2	0.8	0.1	1.7	0	0.1	0	1.0	0.1
HB	0	0	2.3	0	0	0	0	0	0	0	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	2.8	0.5
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	1.2	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	52.9	58.6
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	13.3	11.3
RSS	3.5	0.2	0.8	1.4	0.7	0.1	0.1	0.4	1.4	0	0.9	0	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	0.0	4.3
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	1.6	13.7
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	25.4	2.0
YP	0	7.4	0	3.2	0.1	2.8	0.3	1.1	0	2.7	0.1	0.8	0.1	3.6
BT	0.1	5.1	0.3	3.3	0.1	2.6	0	5.8	0.3	9.2	0	5.9	0.1	3.6

*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.

Koocanusa Reservoir Zooplankton Monitoring

Zooplankton species composition and abundance within Koocanusa Reservoir has remained relatively stable during the past several years (Appendix Tables A9-A15). Since 1997, *Cyclops* and *Daphnia* have been the first and second most abundant genera of zooplankton present in the reservoir (Figure 34). Other lesser abundant genera in decreasing order of abundance include *Diaptomus, Bosmina, Diaphanosoma, Epichura and Leptodora* (Figure 34). Zooplankton abundance within the reservoir varies by season (Table 12; Figure 35). The results from 14 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 2001 and 2002 (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 36) have remained relatively consistent across years. For example, *Daphnia* abundance has peaked during July each year since 1997, *Diaphanosoma* abundance has peaked in September during 5 of the last 6 years, and *Diaptomus* has peaked during October during 4 of the last 6 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 little evidence that zooplankton abundance differed within the three sampling areas (Tenmile, Rexford, and Canada) in 2001 or 2002 (Table 12). However, when significant differences did occur, in each case, zooplankton abundance was always highest for most downstream site. During 2001, abundance estimates of *Diaptomas*, *Cyclops*, and *Epischura* differed between sampling areas. Subsequent multiple comparisons indicated that *Diaptomas* densities were significantly higher at the Tenmile site than the Canada site, *Cyclops* abundance at Rexford was significantly higher than Canada, and *Epischura* densities were significantly higher at the Tenmile site than the Rexford and Canada sites. During 2002, only 2 genera of zooplankton exhibited differences between sampling area. *Cyclops* densities at the Tenmile and Rexford sites were significantly higher than the Canada site, and *Diaphanosoma* densities at the Tenmile site were significantly higher than the Rexford and Canada sites. The month and area interaction term was significant for 5 and 4 of the zooplankton genera in 2001 and 2001, respectively.

Zooplankton of the genus *Daphnia* have remained particularly stable in terms of abundance (Figure 34) and size (Figures 36 and 37) during the past several years. Mean annual Daphnia densities in Koocanusa Reservoir from 1997 through 2002 have averaged 1.94 *Daphnia* /liter (standard deviation = 0.59/liter; Figure 34). Mean *Daphnia* length has also varied relatively little since 1991, averaging 0.90 mm (standard deviation = 0.05; Figure 37). 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 = 0.60, standard deviation = 0.049; Figure 36), with the majority of the remainder in the size class 1.0 – 1.499 (mean annual proportion = 0.339, and standard deviation = 0.036). *Daphnia* larger than 1.5 mm have on average comprised less than 5% of the total since 1993 (Figure 36).

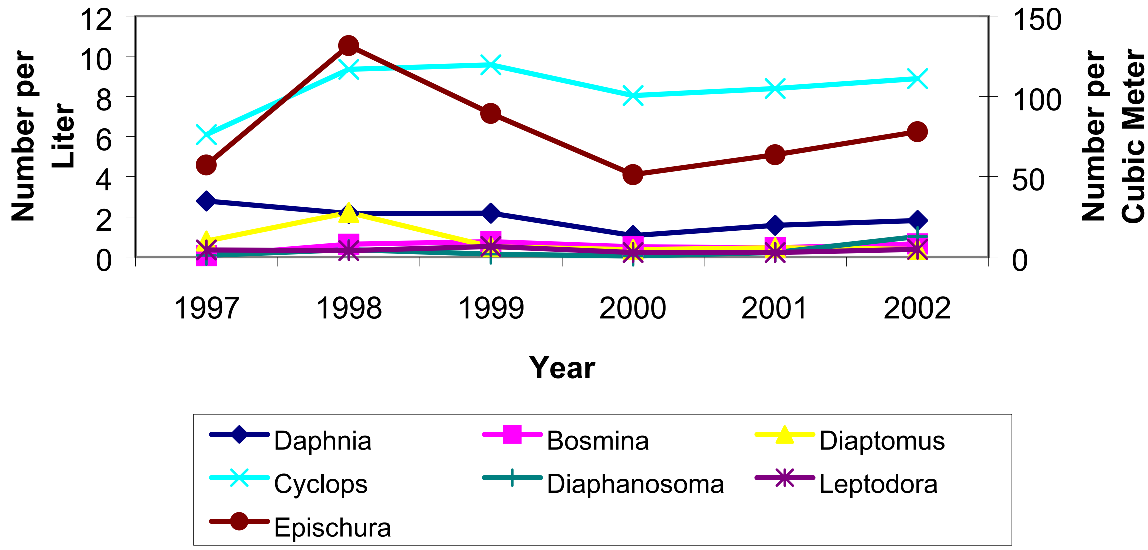

Figure 34. Annual zooplankton abundance estimates for 7 genera observed in Koocanusa Reservoir from 1997-2002. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter. The data utilized for this figure are presented in Appendix Table A15.

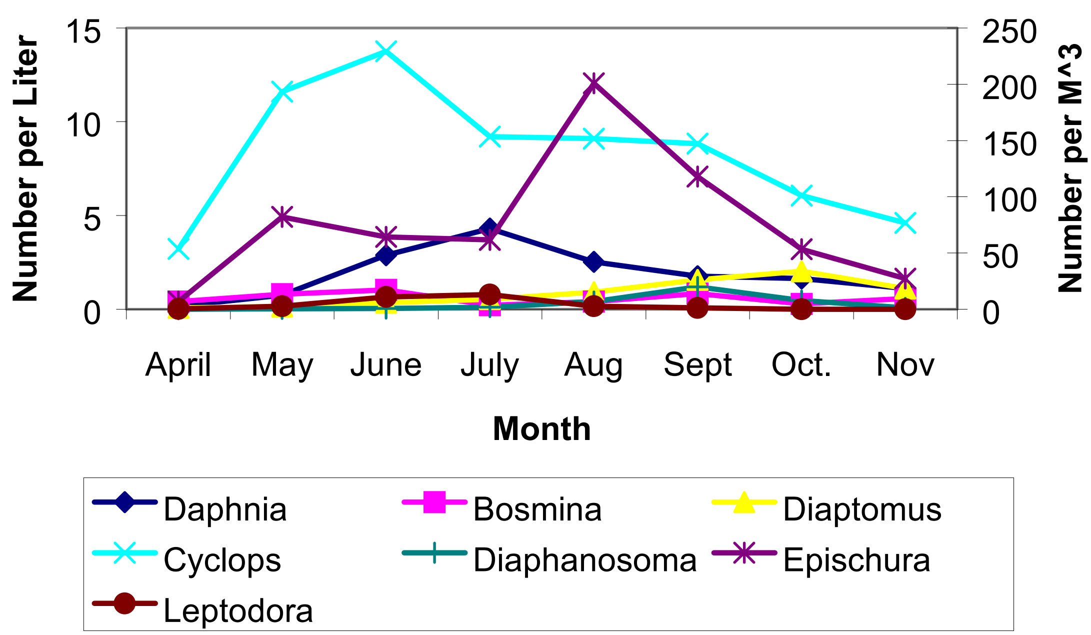

Figure 35. Mean monthly zooplankton abundance estimates for 7 genera observed in Koocanusa Reservoir from 1997-2002. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter.

Table 12. Individual probability values (p values) resulting from analysis of variance procedures that tested for differences in zooplankton densities by month, area (Tenmile, Rexford and Canada) and a month by area interaction in 2001 and 2002.

Year	Factor	Daphnia	Bosnian	Diaptomas	Cyclops	Leptodora	Epischura	Diaphanosoma
2001	Month	2.70 E -8	0.0026	4.55 E-13	0.00011	1.584 E-9	1.287 E-6	3.596 E-11
2001	Area	0.131	0.847	0.095	0.0158	0.338	0.0016	0.281
2001	Month X Area Interaction	0.006	0.0748	0.004	0.0062	0.402	2.691 E -6	0.269
2002	Month	3.00 E -9	5.436 E – 7	0.0014	3.44 E -6	2.083 E -12	0.0009	9.34 E -16
2002	Area	0.388	0.166	0.735	0.0022	0.121	0.716	0.006
2002	Month X Area Interaction	0.002	0.818	0.365	0.09	0.007	0.127	5.779 E -5

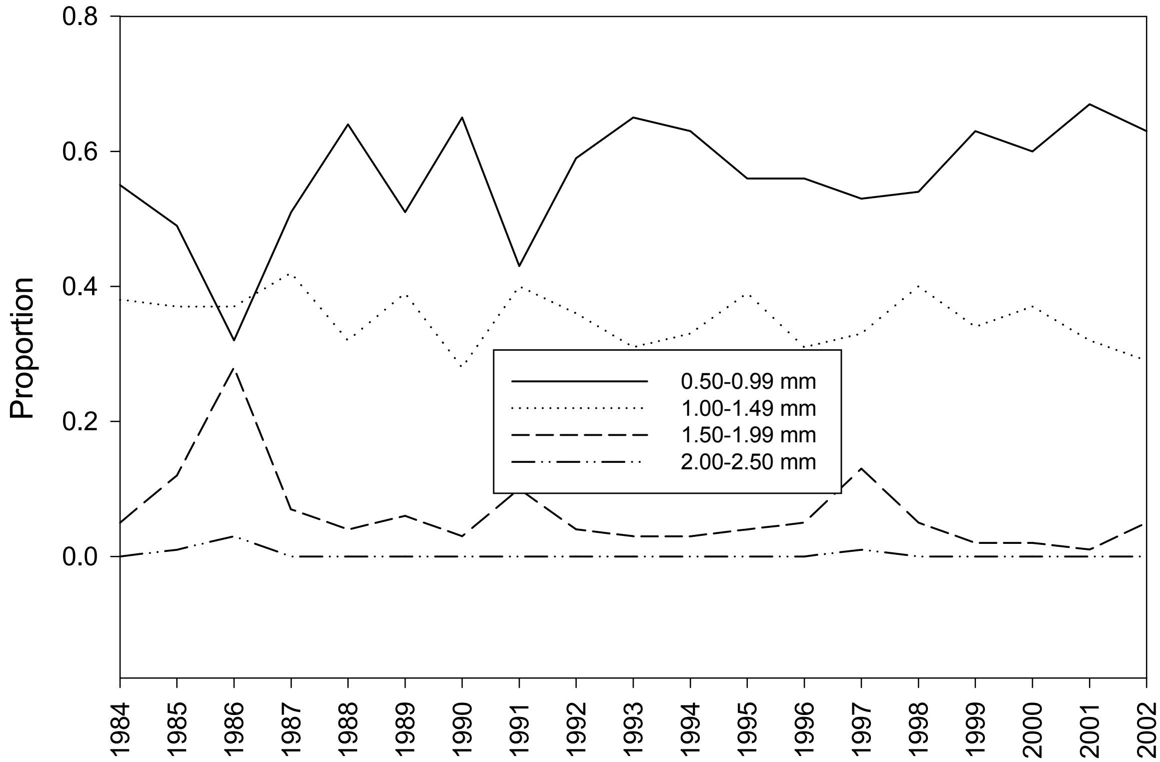

Figure 36. *Daphnia* species size composition in Libby Reservoir, 1984 through 2002.

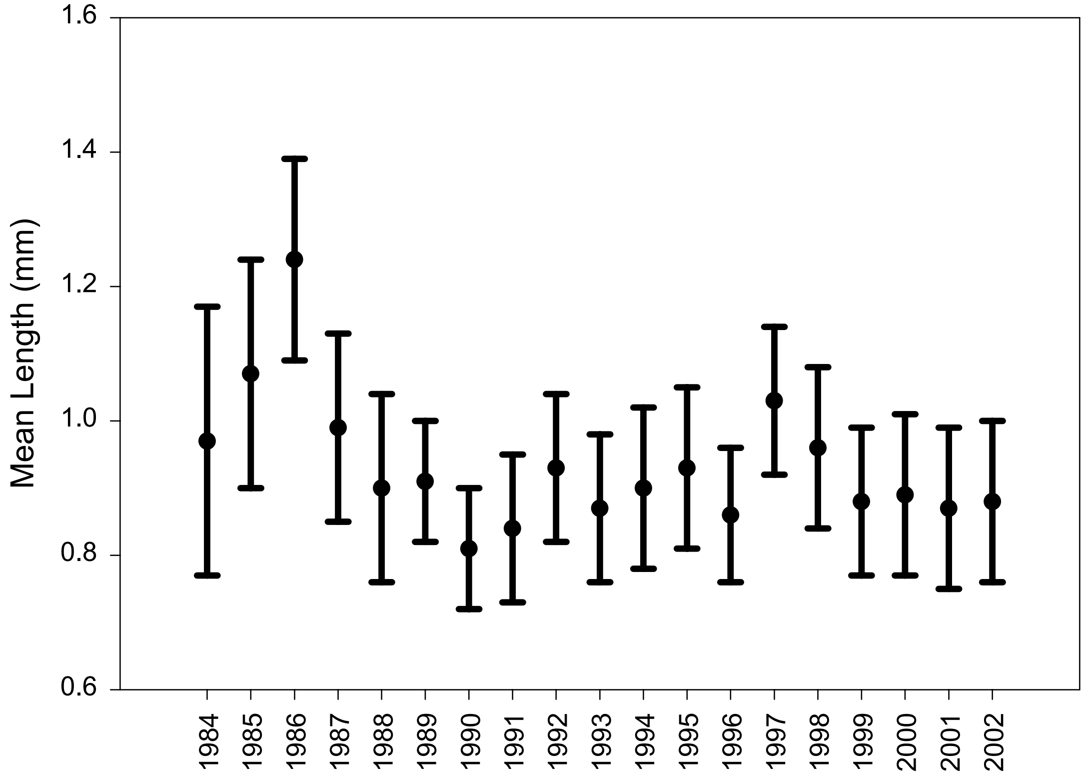

Figure 37. Mean length of *Daphnia* species in Libby Reservoir, 1984 through 2002, with 95% confidence intervals.

Discussion

The U.S. Fish and Wildlife Service listed the Columbia River population of bull trout as a threatened species on June 10, 1998 (63 FR 31647), and have subsequently determined that the Kootenai River Recovery Unit forms part of the range within the Columbia River Population segment (USFWS 2002). The USFWS recovery plan makes the distinction between primary and secondary core bull trout areas based mostly on size, connectedness, and complexity of the watershed and the degree of natural population isolation. The two primary core areas in the Kootenai River Recovery Unit include Lake Koocanusa and the Kootenai River/Kootenay Lake complex that begins downstream of Libby Dam to Kootenay Lake. The two secondary core areas are Bull Lake and Sophie Lake. The recovery plan has set four general recovery criteria. 1. Distribution criteria will be met when the total number of identified local populations (currently number 10 in the United States waters) has been maintained or increased and local populations remain broadly distributed in all 4 existing core areas. 2. Abundance criteria will be met when the primary Lake Koocanusa and Kootenai River/Kootenay Lake core areas are each documented to host at least 5 local populations (including British Columbia tributaries) with 100 adults in each and each of these primary core areas contains at least 1000 adult bull trout. The abundance criteria for the Bull Lake and Sophie Lake secondary core areas will be met when each area supports at least 1 local population of bull trout containing 100 or more adult fish. 3. Trend criteria will be met when the overall bull trout population in the Kootenai River Recovery Unit is accepted, under contemporary standards of the time, as stable or increasing, based on at least 10 years of monitoring data, and 4. Connectivity criteria will be met when dam operational issues are satisfactorily addressed at Libby Dam (as identified through U.S. Fish and Wildlife Service Biological Opinions) and when over half of the existing passage barriers identified as inhibiting bull trout migration on smaller streams within the Kootenai River Recovery Unit have been remedied.

Long-term monitoring of bull trout redd numbers can be an important and useful tool to assess bull trout population trends (Rieman & McIntyre 1993). Based on annual redd counts in the Wigwam River and Grave Creek, and spring gill net catch rates in Koocanusa Reservoir, adult bull trout abundance within Koocanusa Reservoir has increased over the past 8-10 years. Baxter and Baxter (2002) reported 132 and 143 bull trout redds in 2001 and 2002, respectively in Skookumchuck Creek, a tributary to the upper Kootenai River. Therefore, we combined bull trout redd counts for Grave and Skookumchuck creeks and the Wigwam River in an attempt to estimate the total number of upper Kootenai River redds. We estimated there were 2221 and 2250 redds in 2001 and 2002, respectively. 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 Wigwam River and Grave Creek observed in 2001 and 2002, there may have been 3443 and 3500 spawning bull trout within the reservoir in 2001 and 2002, respectively (ranges = 2665-4664 and 2710-4742, respectively). However, these estimates are certainly an underestimate of the total reservoir bull trout population because it fails to take into account alternate year spawning individuals. Nevertheless, these data indicate that most of the criteria established by the USFWS (2002) for the Lake Koocanusa core area are currently being met. It is likely that the only criterion that is not currently fully realized is the portion of the abundance criterion (number 2) that states that each core area must contain at least 5 local populations of 100 adult bull trout. Four local populations in the Lake Koocanusa population contained at least 100 bull trout the previous 2 spawning periods. The tributaries include the Wigwam and White rivers, and Grave and Skookumchuck creeks (Table 1).

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 the Lake Koocanusa primary core area over the past 8 years. We are however, confident that we have identified the important core spawning tributaries below Libby Dam. We identified bull trout redds in Quartz, Pipe, O'Brien, and Bear Creeks. O'Brien Creek was the only bull trout spawning tributary in the Kootenai River Basin located downstream of Libby Dam that has exhibited a significant ($p > 0.05$) positive population trend analyses over the past 7-8 year period. However, draught conditions in 2001 and 2002 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.

Continuous bull trout redd count data from Quartz, O'Brien, Pipe creeks exists for the past 12 years. Redd counts for Bear Creek extend back to 1995. The lack of information from Bear Creek makes long-term trend (>8 years) not possible at this time. However, it is likely that the bull trout population trend for the core Kootenai River mainstem tributaries downstream of Libby Dam have collectively and significantly increased in adult abundance based on redd counts over the past 10-12 years.

We completed 3 separate regression analyses to assess bull trout trends below Libby Dam, including all bull trout redd counts during1993-2002 including Bear Creek data from 1995-2002, all bull trout redd counts during 1993-2002 excluding Bear Creek data, and all bull trout redd counts during 1995-2002. Bull trout redds exhibited a significant and positive trend in each of the first two instances ($r^2 = 0.488$; $p = 0.025$, and $r^2 = 0.458$; $p = 0.031$, respectively). However, when the time frame was limited to 1995-2002, the trend was no longer significant ($r^2 = 0.301$; $p = 0.159$). The total estimated number of bull trout redds in spawning tributaries below Libby Dam were 212 and 136 redds in 2001 and 2002, respectively. Therefore, if we apply the mean number of fish per redd Baxter and Westover (2000) observed in the Wigwam River to the total number of redd we counted below Libby Dam in 2001 and 2002, there may have been 329 and 211 spawning bull trout in 2001 and 2002, respectively (ranges = 254 – 445 and 163 - 286, respectively). Therefore, it seems likely that bull trout in the Kootenai River/Kootenay Lake primary core area are meeting the distribution and trend recovery criteria (criteria numbers 1 and 3) set fourth by the USFWS (2002). However, bull trout abundance criteria (criteria number 2) within this primary core area are likely falling short of recovery goals. Our monitoring data indicate that it is likely that less than 1000 individuals exist within this area, and that only the Quartz Creek local population consistently contains greater than 100 spawning individuals within a year.
Bull trout redd counts in Keeler Creek are the primary index of abundance for the Bull Lake secondary core area, and were initiated in 1996. Therefore, the data collected over the

previous seven years do not meet the ten-year minimum requirement set by the USFWS for the trend criteria (criteria number 3; USFWS 2002). Nevertheless, it seems likely that this population is stable or increasing over the past 7 years. It is also likely that the Bull Lake secondary core are has met the abundance criteria of at least 100 adults during a minimum of 5 of the last 7 years.

The USFWS also identified Sophie Lake as a secondary core area for bull trout. Little information exists for this area. Gill net surveys were conducted in Sophie Lake in 2001 an 1993. Gill netting conducted in Sophie Lake in October 11, 2001 caught 1 adult bull trout (MT FWP, unpublished data) using 6 sinking gill nets set overnight. The surveys conducted on 10/27/93 captured 4 adult bull trout using 4 sinking nets in Sophie Lake. Bull trout redd surveys are not currently conducted in the British Columbia portion of Phillips Creek, the only tributary to Sophie Lake.

The bull trout redd counts and juvenile 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 are met within this basin. Therefore, collection of this information will remain a high priority for long-term monitoring conducted by Montana FWP.

We believe that our bull trout radio telemetry study provided us with an accurate assessment of seasonal movement patterns, overall spatial distribution, and areas of congregation for bull trout within the Kootenai River below Libby Dam. We base this assessment on relatively high proportion of tagged fish that we maintained locations on (89%), the relatively high number of mean observations per tagged fish (30.7 observations per fish) and the relatively short mean period between observations for tagged fish (22.6 days) throughout the duration of the 3-year study. However we acknowledge that the estimates of the proportion of radio tagged fish that migrated over Kootenai Falls and the proportion of radio tagged bull trout that ascended tributaries during the spawning season may not accurately represent the behaviors of non-tagged bull trout in the Kootenai River below Libby Dam. For example, up to 50% of the radio tagged bull trout that migrated below Kootenai Falls did so within a minimum of 50 days after being tagged, suggesting the possibility that our handling and tagging the fish may have influenced their behavior, and contributed to the fallback of 11 of the radio tagged bull trout. The remaining 50% of the radio tagged bull trout that migrated over Kootenai Falls did so a relatively long time after being handled and tagged. We are not certain whether these observations are an accurate indicator of the prevalence of bull trout migration over the falls. If we assume that the effects of tagging and handling did not contribute to this latter group of fish that migrated over Kootenai Falls, then approximately 19% of the bull trout in the Kootenai River may be migrating over the falls. Although we did document a single bull trout ascend Kootenai Falls proving that the falls are not a complete fish barrier, Kootenai Falls is likely a substantial obstacle to upstream migration, especially during period of extremely high and low flows. Given the low rate of ascending upstream of the falls and the relatively high proportion of bull trout that may be migrating below Kootenai Falls, this situation may be constitute a source/sink population which may influence the probability of the long-term persistence of this population (Harrison 1991; Gilpin 1987). The effects of this situation may be exacerbated by the presence of Libby Dam. Prior to the construction of Libby Dam, bull trout above Kootenai Falls had access to the entire Kootenai River into British Columbia, but are presently restricted to 28.7 miles of the mainstem Kootenai River between the dam and the falls.

We observed 4 (6.2%) of the radio tagged bull trout throughout the duration of the three-year study that ascended tributaries during the fall. Although we did not observe any of these fish spawning, the timing and behavior suggested that these fish did likely spawn. Two of these four bull trout entered (and presumably spawned) in consecutive years in Quartz Creek and the Fisher River, respectively. Two other radio tagged bull trout may have also spawned in consecutive years. However, mobile tracking information was insufficient to confirm this assumption. Given the broad geographical distanced required to effectively cover all spawning tributaries in the lower Kootenai River with mobile tracking gear, it is likely that we may have not observed an additional 12 bull trout that ascended the Fisher and Yaak river, O'Brien, Lake and Quartz creeks. Therefore, given the relatively low number of fish that we observed ascending tributaries during the spawning season, we an not confident making refined descriptions of Kootenai River bull trout life history, such as the proportion of bull that that are repeat and alternate year spawning fish. However, Baxter and Westover (1999 and 2000) found that during a four-year study that was conducted between 1996-1999 on the Wigwam River, an average of 29.4% (37.7, 23.1, and 27.5%, respectively) of the spawning bull trout population within a year was comprised of fish that had spawned the previous year (repeat spawners). This is higher than Baxter and Baxter (2002a and 2002b) observed for a similar 3 year trapping study conducted on Skookumchuck Creek, where they estimated that 13.9% and 11.5% of fish spawning in 2001 and 2002 had also spawned the previous year. They also estimated that approximately 1.9% of the bull trout that spawned in 2002 also spawned both of the previous two years. Baxter and Westover (2000) estimated that an average of 6.5% (8.4 and 4.6%, respectively) of the bull trout that spawned in the Wigwam River study in 1998 and 1999 were alternate year spawners. However, this was lower than the estimated proportion of alternate year spawners in Skookumchuck Creek, where Baxter and Baxter (2002b) estimated that 8.1% of the bull trout that spawned in 2000 also spawned in 2002.

Our radio telemetry study confirmed our suspicions that bull trout seasonally congregate in several locations below Libby Dam to the Fisher River confluence. Angling is very common in many of these areas, and therefore has the potential to create a public impression that bull trout may be much more abundant than they actually are within the Kootenai River below Libby Dam. The congregation of bull trout within common locations within the Kootenai River also has the potential to create a mixed stock (population) bull trout fishery that could potentially impact the weakest population present within this mixed group either through non-compliance, hooking mortality, or the establishment of an angling season for bull trout. Potential for this situation would be highest especially during the spring and winter. Fish movement was lowest during the spring and winter seasons. The bull trout we tagged during this study moved nearly twice as much (based on mean distance between observations) during the fall season. We assumed that increase in distanced moved during the fall season was due to spawning movements.

We found no evidence of a significant trend of burbot abundance in Koocanusa 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). Catch rates of burbot in our baited hoopnets in the Kootenai River directly below Libby Dam have precipitously and significantly decreased in recent years. The degree to which the population of burbot below Libby Dam and the reservoir population are correlated is not know, but evidence suggests the downstream population may be influenced by entrainment through Libby Dam. These uncertainties identify the need to address these issues during future project activities.

We believe that Koocanusa 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.

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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 and Park, 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 (Montana Fish, Wildlife and Parks et al. 1998). A mix of mitigation techniques is necessary to offset losses caused by dam construction and operation. During the past two years, Montana Fish, Wildlife and Parks has implemented several project to mitigate for a portion of the losses attributable to the construction and operation of Libby Dam. We worked cooperatively with the city of Troy, Montana to construct a community fishing pond at the Troy Recreation Park. A similar project was constructed at the Lincoln County Fairgrounds, near Eureka, MT. These projects will enhance fishing and educational opportunities for young anglers. We identified Libby Creek and Grave Creek 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, and Grave Creek has been identified as a core area for bull trout. We adopted a phased restoration approach for both streams, with the initial phases of restoration on both streams targeting the elimination of some of the largest supplies of bedload sediment. Restoration activities on both streams were first implemented in the fall of 2001. A rigorous 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 projects. The changes that occurred after implementation of these first two initial projects included a decrease in bankfull width and bank erosion and an increase in stream depth, stream substrate mean particle size, and quality and quantity of salmonid rearing habitat. Restoration of both Grave and Libby creeks continued at the watershed level during 2002 with the implementation of a phased restoration approach. The Libby Creek Cleveland Project and the Grave Creek Phase I projects were constructed during the fall of 2002 and effectively changed the stream channel pattern profile and dimension. These changes resulted in a narrower, deeper channel that are likely to improve the quantity and quality of rearing habitat for native salmonids. A rigorous monitoring program also accompanies these projects and the information gathered will allow us to assess whether we continue to meet project 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. Impoundment of the Kootenai River blocked the migrations of fish populations that once migrated freely between Kootenai Falls (29 miles below Libby Dam) and the headwaters in Canada. Historically, the fish residing downstream of Libby Dam could access quality spawning habitat upstream of Libby Dam in the United States and 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 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 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 (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 et al. 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 and Park, 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 (Montana Fish, Wildlife and Parks et al. 1998). This plan identifies several non-operational actions that 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 and 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 having 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 (Marotz et al. 1998).

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

Troy Fishing Pond

In an effort to help mitigate for a loss of fisheries habitat and recreation opportunity in the Montana portion of the Kootenai River sub-basin, we constructed the Troy fishing pond as a cooperative effort between Montana FWP and the city of Troy, MT. This project will enhance fishing and educational opportunities for young anglers on land located at the Troy Recreational Park. The pond was an existing saw millpond that was excavated to a depth of 17 feet maximum depth, and lined with a mixture of silt and granular bentonite to minimize leaking. The pond has an area of approximately 2 acres (Figure 1). The water source for the Troy fishing pond is the city of Troy's old water supply system from O'Brien Creek that currently serves as a backup water system for the city. Water exits the pond via an outlet pipe into the Kootenai River.

The fishing pond is occasionally stocked by MFWP with rainbow trout grown at Murray Springs Hatchery in Eureka, MT. Remote site incubators could be used to stock the pond and provide an educational opportunity in future years.

Figure 1. A photograph of the Troy fishing pond shortly after construction during the 2002 summer. The maximum depth is 17 feet and total area is approximately 2 acres. Landscaping will be completed by the City of Troy. The pond has been successfully stocked with rainbow trout produced by Montana FWP at the Murray Spring Fish Hatchery in Eureka, MT.

Eureka Pond

Also in an effort to help mitigate for a loss of fisheries habitat and recreation opportunity in the Montana portion of the Kootenai River sub-basin, 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. 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 (Figure 2). 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. The gravel/cobble substrate beneath the pond has proven difficult to seal. Leaks in the pond bottom have prevented complete filling. Additional attempts to seal the pond will be made during the summer of 2003. The pond has not been stocked with fish yet.

Figure 2. A photograph of the Eureka fishing pond taken shortly after construction. The pond has a maximum depth is 8 feet and a toal surface area of 0.4 acres. Fish stocking has been delayed until the pond completely seals.

Libby Creek Demonstration Project

Restoration of Libby Creek to a properly functioning stream will be approached at the watershed level, with implementation occurring in several phases. The initial phases will target the elimination of some of the largest supplies of bedload sediment into the stream. The Libby Creek Demonstration Project was the first project undertaken by Montana FWP in Libby Creek. The main objective of this project was to reduce sediment loads into Libby Creek and educate local private landowners and agency administrators about the benefits of constructing a properly functioning stream channel. The specific objectives of this project were to 1) Decrease coarse and fine sediment sources, 2) Decrease the stream's width depth ratio, and 3) Return the stream channel to a properly functioning configuration able to efficiently transport bed load sediment during high discharge events; and 4) Increase the quality and quantity of fisheries habitat within this reach of Libby Creek.

Field reconnaissance and monitoring identified one of the largest point source of sediment in the Libby Creek watershed above the confluence of Elliot Creek (RM 12.0). This project area was located on land owned by Plum Creek Timber Company, which substantially contributed to the project by donating large rock materials and large woody debris. Two unstable and eroding banks in this area were contributing substantial amounts of course and fine sediment to Libby Creek each year. The largest eroding bank within the project site was over 700 feet long, averaged 80 feet high and was contributing an estimated 5,900 cubic yards of sediment annually to Libby Creek (Figure 3). A second large unstable bank in the lower section of the project area was also contributing substantial amounts of sediment to Libby Creek. This bank was approximately 340 feet long and averaged 12 feet high. The sediment resulting from these two banks increased sediment deposition; accelerated bank erosion; increased width/depth ratio and decreased meander width ratio in Libby Creek both within and downstream of the Demonstration Project area.

The Libby Creek Demonstration Project was begun in September 5, 2001. When the project was completed in September 29, 2001, it had restored one meander length the Libby Creek stream channel (approximately 1,700 feet). All channel and structure construction was completed in the dry. The project restored this section of Libby Creek to a properly functioning stable stream channel capable of maintaining its course through the valley through the construction of 1,700 feet of stream channel, 7 rock J-hook vanes, 7 rootwad and log complexes, and numerous channel plugs to fill the old stream channel. The channel was designed to mobilize a 185mm particle size during a bankfull discharge of 1,200 cfs. Reference reach, substrate data, longitudinal and cross sectional profiles where collected by FWP and used to develop channel dimensions, Regional Curves, developed by the Kootenai National Forest and Dave Rosgen (Wildland Hydrology), were used to reference bankfull channel dimension calculations. USFS stage/discharge data were useful for developing the proposed channel at Elliot Creek.

Figure 3. Top photograph shows the largest of the two eroding banks within the Libby Demonstration Project prior to project implementation. The lower photograph was taken after project construction. Note the position of the stream in the upper photograph against the eroding hillside that was over 700 feet long and averaged 80 feet high.

FWP designed a long term monitoring program to evaluate the effectiveness of the stream restoration project. Both biological (see Chapter 1) and stream geomorphological data were collected beginning in 1998 to properly assess both short and long term effects of restoration efforts. Our monitoring program for this project included 4 permanent cross sections within the project area (Figures 4-7), Wolman pebble counts (Wolman 1954) at each cross section, a longitudinal profile survey (Figure 8), and permanent photographic points.

The stream channel within the project area prior to project implementation was over widened, shallow, and braided throughout much of the length (Figures 4-7). The project strategically installed the designed stream structures throughout the project reach (Figure 8) that significantly ($p < 0.0008$) decreased mean bankfull width at the 4 permanent cross sections from an average of 119.5 to 66.3 feet (Table 1). Likewise, both maximum and mean bankfull depth at the 4 cross sections nearly doubled, which resulted in a significant ($p = 0.003$ and 0.0008, respectively) increase of mean depth as a result of the restoration work (Table 1). The ultimate result was a deeper and narrower channel, which translates into a significantly ($p = 0.0002$) lower width/depth ratio after project implementation (Table 1). The stream channel sinuosity, length, and gradient remained similar before and after project implementation (1.35, 1,950 feet and 0.7%, respectively). The narrower and deeper stream channel effectively increased shear stress at high flows, which resulted in the stream channel's ability to mobilize larger substrate particle size.

Wolman pebble counts conducted before (1998) and after (2002) project construction indicated that substrate size (median diameter) significantly increased within the project area by 38% after project construction (Table 2). The increase in particle size also resulted in significant increase in the D15th , D35th , and D50th percentile particle sizes (Table 2). The mean D84th and D90th percentile particle sizes at the 4 cross sections increased after project implementation, and although the increases were substantial (at least 46%), differences were not significant (Table 2). However, the power of these two tests was low (43% and <25%, respectively).

The stream restoration work on lower Libby Creek also increased the quantity and quality of rearing habitat for native salmonids within the project reach. The total number of pools within the project reach increased by 25%, and maximum pool depth measured during summer base flow increased by 45.7% (Table 3). Statistical comparisons of mean pool length and maximum depth pre and post project implementation were not performed because the pool inventory was a complete census of pools present within this reach of Libby Creek. Therefore, the pool characteristics are parameters, not estimates.

This project also reduced bank erosion within the project reach by limiting creek access to the two large eroding banks located within the project reach (Figure 4 and 6). We surveyed the toe of the largest eroding slope in the winter of 2002/2003 and estimated that approximately 10,500 cubic yards of course and fine sediment had recruited to the toe of the slope. This sediment would have entered Libby Creek if the project had not been implemented. We also predicted the erosion rates (feet of stream bank per year) that may have occurred if this restoration project were not implement from predictive curves developed by Rosgen (2001). The predictive curves we used utilized the bank erodibility hazard index (BEHI; Rosgen 2001) that takes into account bank height, root depth, root density, bank angle, and total bank surface protection. The BEHI index ranges from very low (5-9.5) to extreme (46-50). We also used survey data from cross sectional surveys to estimate near bank stress indices (Table 4), which range from very low (< 0.8) to extreme (>1.6). We estimated the BEHI and near bank stress indices for the upper larger eroding bank (represented by cross sections 1-3), and for the lower eroding bank (represented by cross section 4) before and after project implementation. In each case, the BEHI and Near Bank Stress Indices, predicted erosion rates and measured erosion rates decreased as a result of the restoration project (Table 4). Although each of the indices are estimates, it seems likely that the overall result was a reduction in the amount of sediment that entered Libby Creek since the project was implemented.

In addition to the stream channel work, substantial effort has been expended to re-establish a healthy riparian vegetation community. In October and November of 2001 we seeded the riparian area with a native grass and annual forb mixture and planted 1200 rooted sandbar willow and 500 rooted cottonwood cuttings in the riparian area. However due to draught conditions during the fall of 2001, the willow and cottonwood shrubs experienced low survival.

Due to poor plant survival on the well-drained alluvial soils in the project site we contracted additional planting with a hydraulic stinger. During November 2002 1,100 containerized stock and 2,000 sprigs were mechanically planted 3-4 feet deep in the project banks and floodplain. Springs included willow and cottonwood poles cut by Montana FWP near the project site. The containerized native plants were provided by the contractor, and were inoculated with a full spectrum of beneficial soil microbes to stimulate plant vigor. Containerized plant species included alder, coyote (sandbar willow), Bebbs willow, dogwood, cottonwood, water birch, with rose, serviceberry, hawthorn and aspen on the drier sites. The spacing and density of plants and sprigs varied in the project area. The highest planting densities are along the outside bank of the meanders. Two rows of cuttings and containerized plants would be planted immediately adjacent to the stream at bankfull elevation. The cuttings are ½ to ¾ inch diameter on the planted end and 3 ½ to four feet long. Fifty to 100 cottonwood poles were randomly planted along the channel above the bankfull elevation. The poles are two to four inches in diameter and eight feet in length.

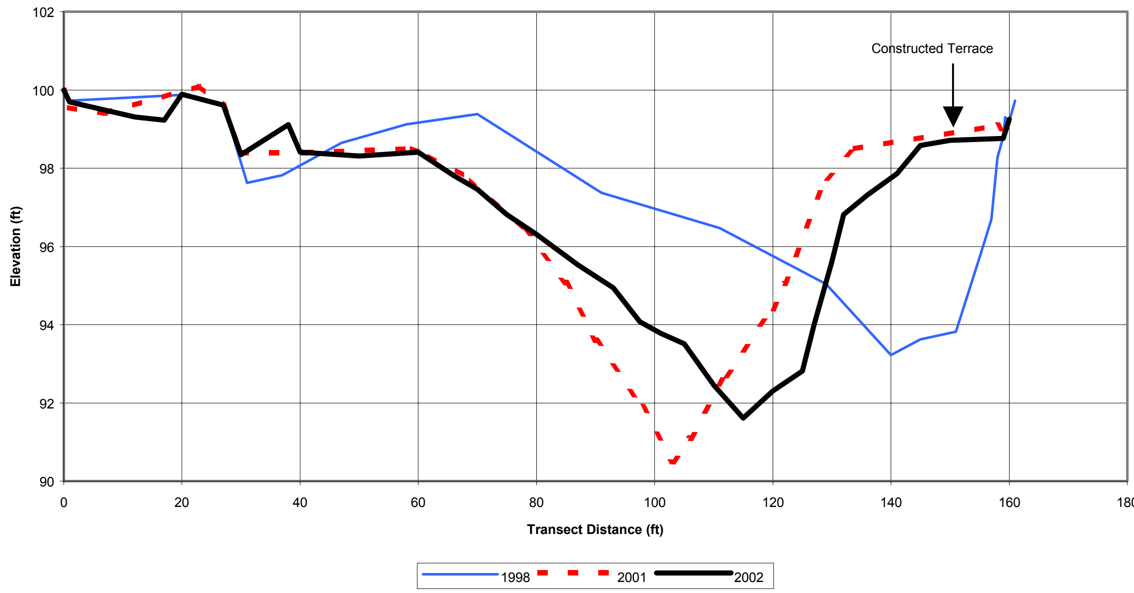


Figure 4. Cross section #1 within the Libby Creek Demonstration Project, surveyed before (1998) and after (2001 & 2002) project construction. This site is located 75 feet below the upper boundary of the project area. Note the constructed terrace that relocated Libby Creek away from an 80 feet high eroding bank that was contributing course and fine sediment to Libby Creek during each high flow event.

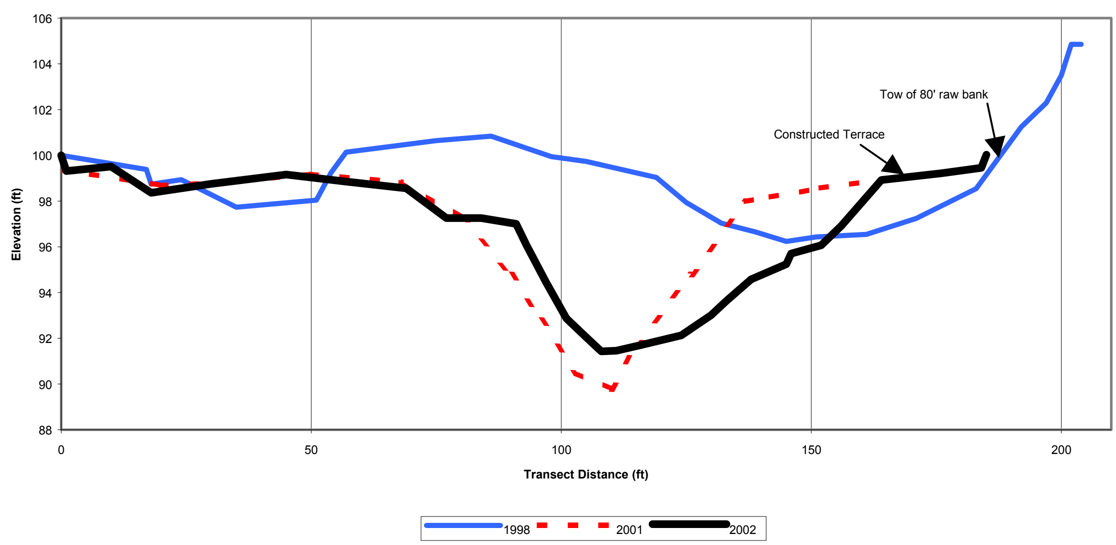


Figure 5. Cross section #1 within the Libby Creek Demonstration Project, surveyed before (1998) and after (2001 & 2002) project construction. This site is located 277 feet below the upper boundary of the project area. Note the constructed terrace that relocated Libby Creek away from an 80 feet high eroding bank that was contributing course and fine sediment to Libby Creek during each high flow event.

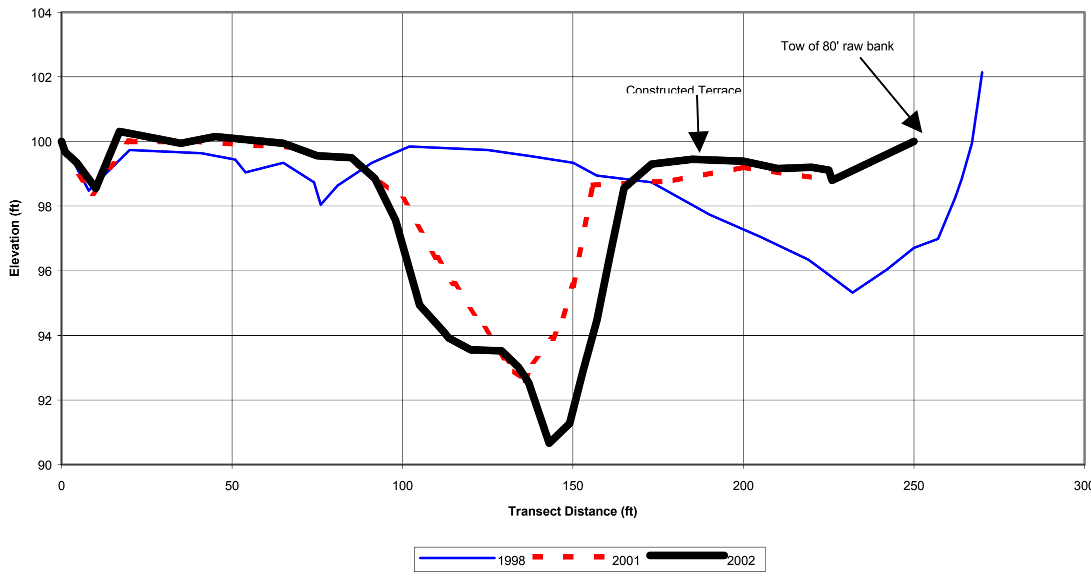


Figure 6. Cross section #3 within the Libby Creek Demonstration Project, surveyed before (1998) and after (2001 & 2002) project construction. This site is located 467 feet below the upper boundary of the project area. Note the constructed terrace that relocated Libby Creek away from an 80 feet high eroding bank that was contributing course and fine sediment to Libby Creek during each high flow event.

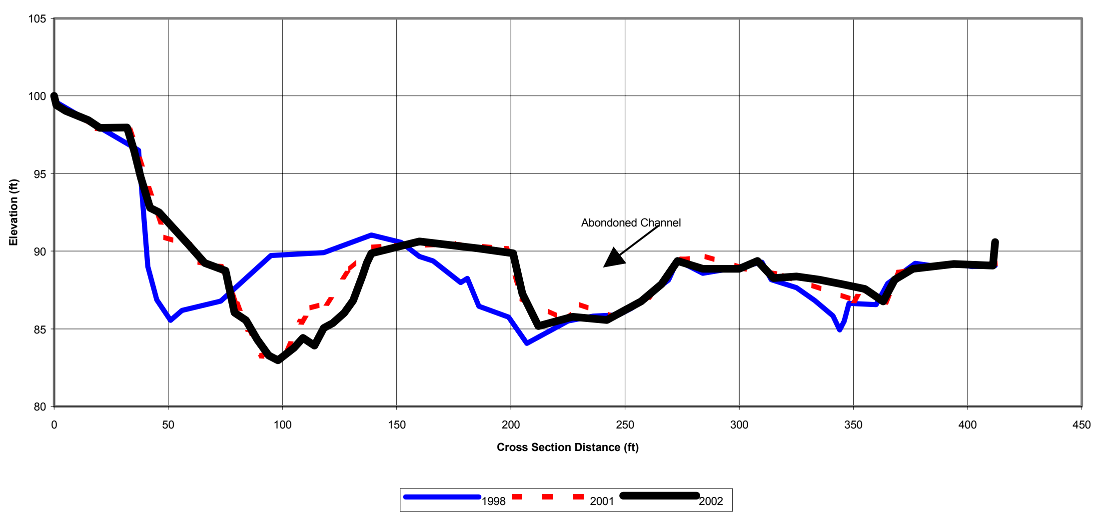


Figure 7. Cross section #4 within the Libby Creek Demonstration Project, surveyed before (1998) and after (2001 & 2002) project construction. This site is located 1,212 feet below the upper boundary of the project area. Note the abandoned channel on the right side of the figure. The stream channel at this location was braided (multiple channel) prior to project construction. Several channel plugs were installed in order to prevent the stream from gaining access to these channels. The stream is currently single thread throughout the project area.

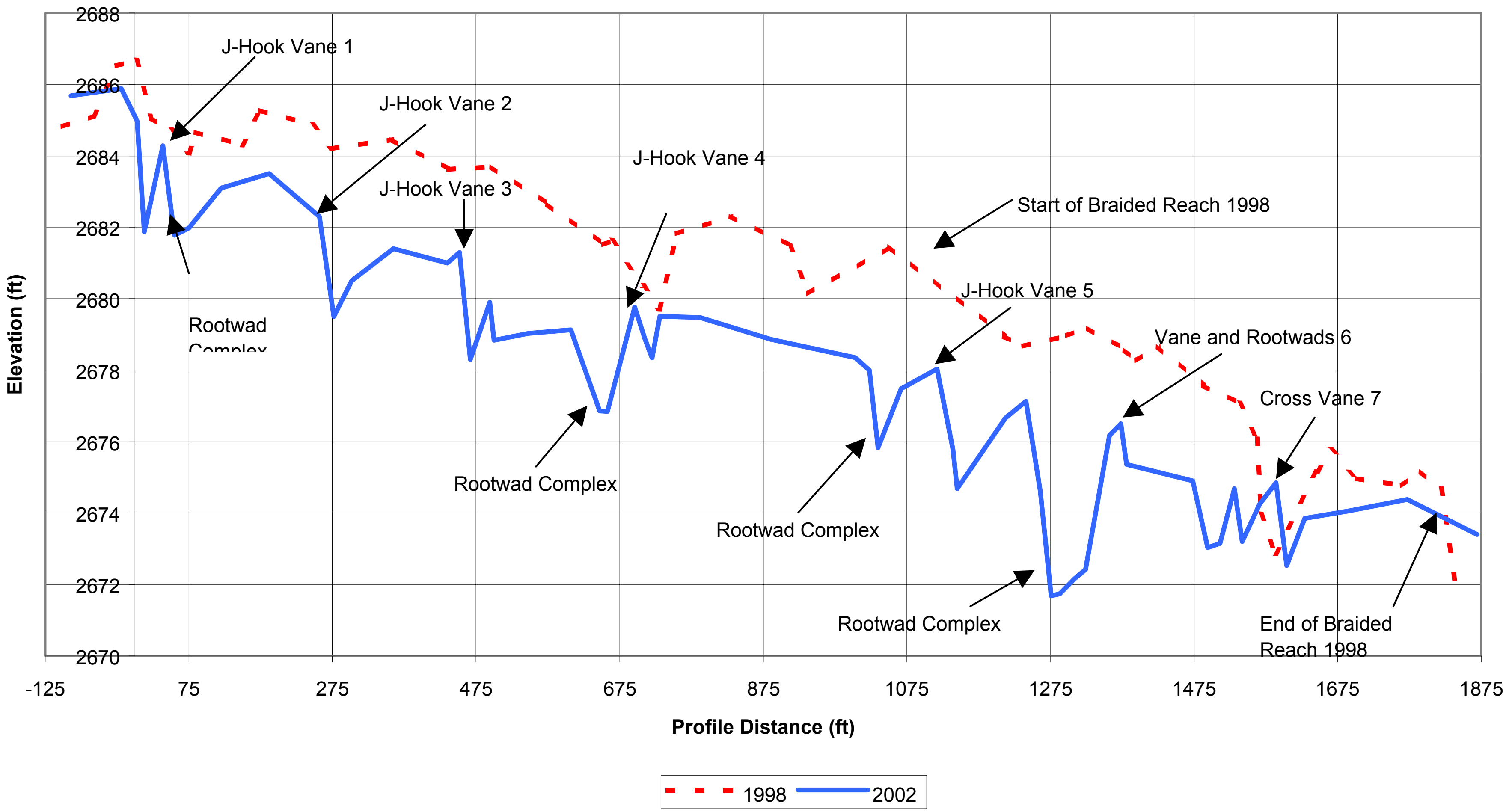


Figure 8. The longitudinal profile survey of the Libby Creek Demonstration Project before (1998) and after (2002) project construction in the fall of 2001.

Table 1. Summary data for 4 permanent cross sections within the Libby Creek Demonstration Project. Data was collected prior to project implementation in 1998 and after project implementation in 2002.

	Pre-Project					**Post-Project**				
Cross Section Number	4	3	2	1	**Mean**	4	3	2	1	**Mean**
Year	1998	1998	1998	1998	1998	2002	2002	2002	2002	2002
Distance (ft) from upper project boundary	1212	467	277	75		1212	467	277	75	
Bankfull Width (ft)	158	85	110	125	119.5	71	63	65	66	66.3
Mean Bankfull Depth (ft)	1.94	1.46	2.07	1.9	1.84	3.6	4.1	3.2	2.9	3.45
Maximum Bankfull Depth (ft)	4.95	3	3.5	4.1	3.89	6.3	6.9	5.5	5.8	6.13
Width to Depth Ratio	81	58	53	66	64.6	20	15	20	23	19.5

Table 2. Mean particle size, D15, D35, D50, D84, and D90 mean particle size for 4 permanent cross sections located in the Libby Creek Demonstration Project. Data was collected prior to project implementation (1998) and after project construction (2002). The p-value that resulted from the pairwise statistical comparison between pre and post project comparisons is also stated.			
	Pre Project (1998)	Post Project (2002)	P-value
Mean Particle Size (mm)	55.8	77.2	0.001
D15 Particle Size (mm)	6.42	97.6	0.108
D35 Particle Size (mm)	25.5	42.8	0.04
D50 Particle Size (mm)	41.5	64.5	0.05
D85 Particle Size (mm)	100.7	147.6	0.132
D90Particle Size (mm)	165.1	326.7	0.303

Table 3. Pool characteristics based on longitudinal channel surveys through the Libby Creek Demonstration Project reach. Maximum pool depth was based on water depth of pool during base-flow for Libby Creek (10-20 cfs).				
	Pre Project (1998)		**Post Project (2002)**	
Pool #	**Pool Length (ft)**	**Max Depth (ft)**	**Pool Length (ft)**	**Max Depth (ft)**
1	42	2.3	26	3.2
2	100	1.9	73	2.8
3	162	1.2	103	3.0
4	255	3.4	153	3.2
5	108	2.3	50	3.8
6	30	0.8	44	2.7
7	177	3.3	73	3.1
8	19	3.3	96	5.5
9	N/A	N/A	58	3.1
10	N/A	N/A	40	2.5
Mean	**111.6**	**2.31**	**77.3**	**3.41**
Total	**893**		**618**	

Table 4. Estimates of the bank erodibility hazard index (BEHI), near bank stress index, and predicted and measured erosion rates for the upper and lower eroding stream banks within the Libby Creek Demonstration Project.

	Lower Bank (Cross Section 4)		**Upper Bank (Cross Sections 1-3)**	
Year	1998	2001	1998	2001
Total BEHI Index	47.2	35.3	56	36.5
BEHI Rating	Extreme	High	Extreme	High
Near Bank Stress Index	0.92	0.95	0.84	0.77
Near Bank Stress Rating	Low/Mod	Low/Mod	Low/Mod	Low
Bank Composition (%Cobble; %Gravel; %Sand)	20%; 30%; 50%	60%; 30%; 10%	20%; 40%; 40%	60%; 30%; 10%
Bank Height (ft)	12	6	80	7.2
Bank Length	340	340	700	700
Predicted Erosion Rate (ft/year)	0.85	0.4	1.2	0.4
Predicted Erosion (cubic yards/year)	128	30.2	2,133	64
Measured Erosion (cubic yards/year)	N/A	0	N/A	1,422

Libby Creek Cleveland Project

Montana FWP continued our watershed approach to the restoration of Libby Creek in 2002, with the implementation of the upper Libby Creek Cleveland Project (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 9 and 10). 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 the implementation of this project was over-widened with frequent lateral migration of the active stream channel, these conditions resulted in frequent multiple channels existing within the project reach (Figure 11). Width depth ratios were high (ranging from 15-35 feet) and shallow bankfull channel depths ranging from 0.58 to 1.79 feet in depth (Table 5). We established design criteria for the channel dimensions according to criteria established by Rosgen (1996). Table 5 provides the design criteria and summary of existing conditions for several stream channel parameters. Future monitoring activities will evaluate whether these criteria are maintained through time.

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 5) including an average design bankfull width and depth of 32 feet and 3 to 7 feet, respectively. We designed the channel pattern (Appendix Figure A16) 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. The stream channel profile prior to project construction contained few pools (Figure 12), 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 (Figure 13). The designed channel profile required excavation at numerous depositional areas throughout the project reach (Figure 13) and resulted in an increased quantity of pool habitat within the project area. 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 project completion.

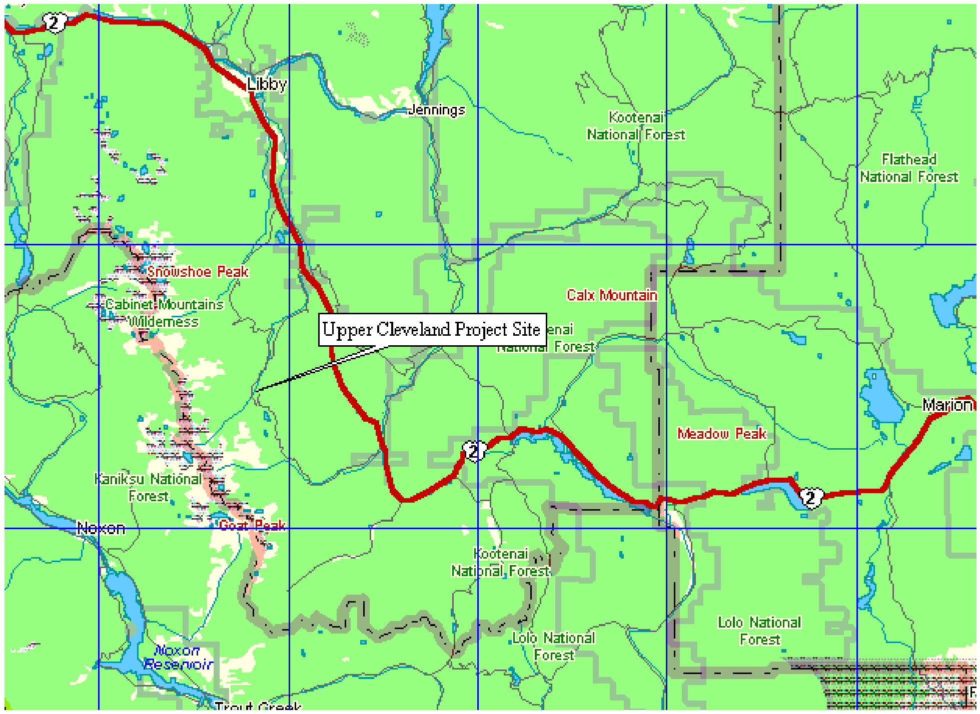


Figure 9. Vicinity Map for Upper Libby Creek Cleveland Project.

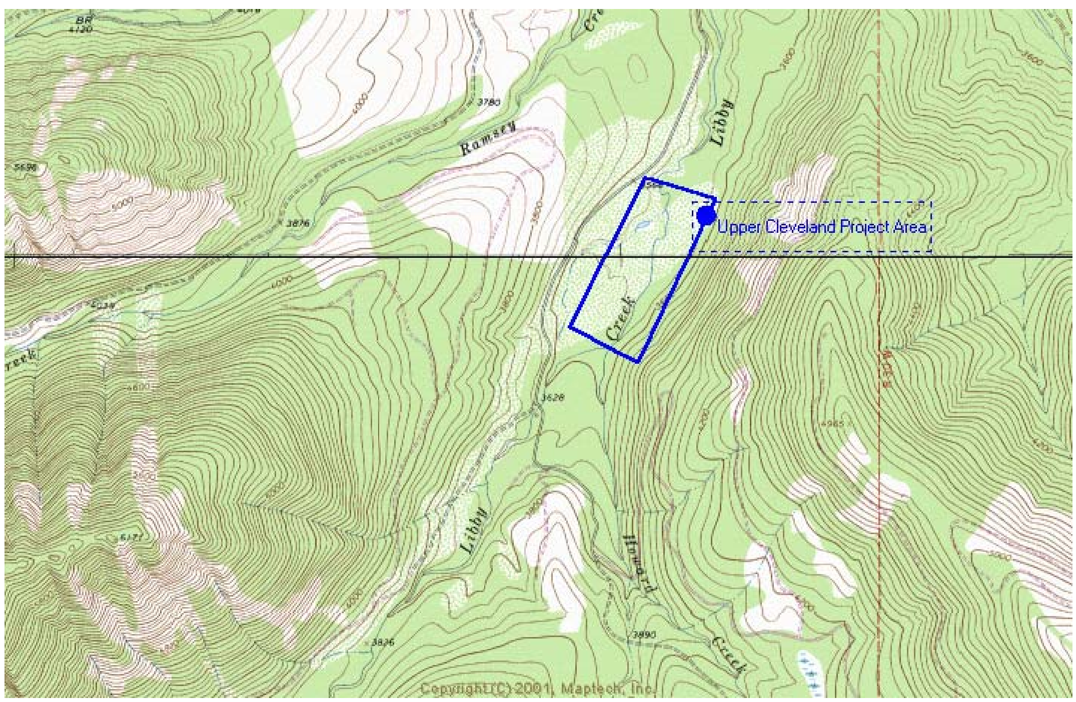
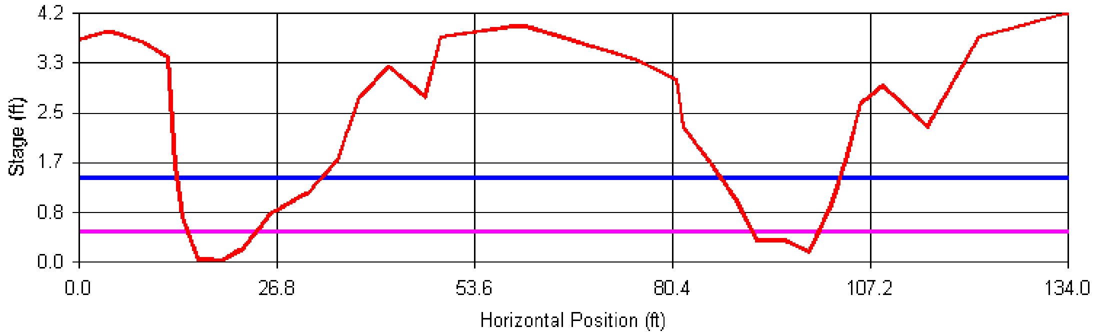


Figure 10. Detailed site location map of the upper Libby Creek Cleveland Project Area.



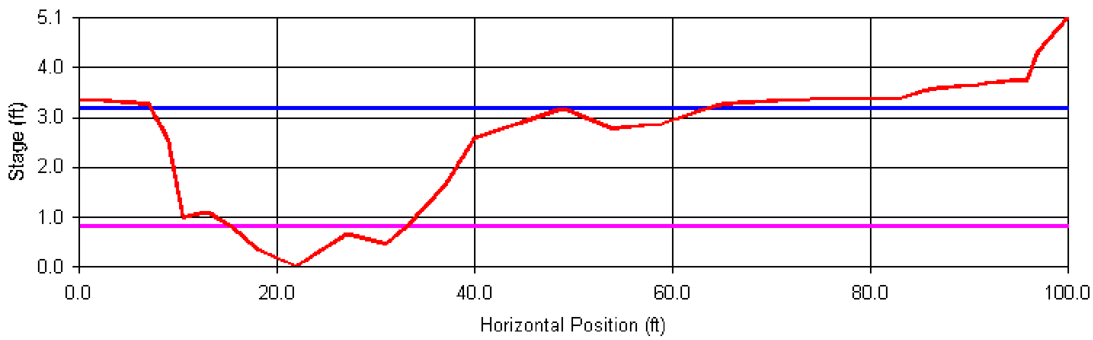


Figure 11. The top cross sectional survey of Libby Creek (#12C) was surveyed by Montana FWP in 1999, is typical representation of the braided channel and large amounts of deposition within the floodplain of the upper Libby Creek Cleveland Project site. The lower figure characterizes the design criteria used to implement project construction activities at this site.

Table 5. Design specifications for the upper Libby Creek (Cleveland's) channel restoration project.			
Channel Design	Range	Mean	Existing (Mean)
Design Channel Type (C4) Drainage Area = 12 sq. miles			
Total Length		3,200	2,700 (braided throughout)
Bankfull Width (ft)	28-35	33	27-63
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	1.58 – 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.

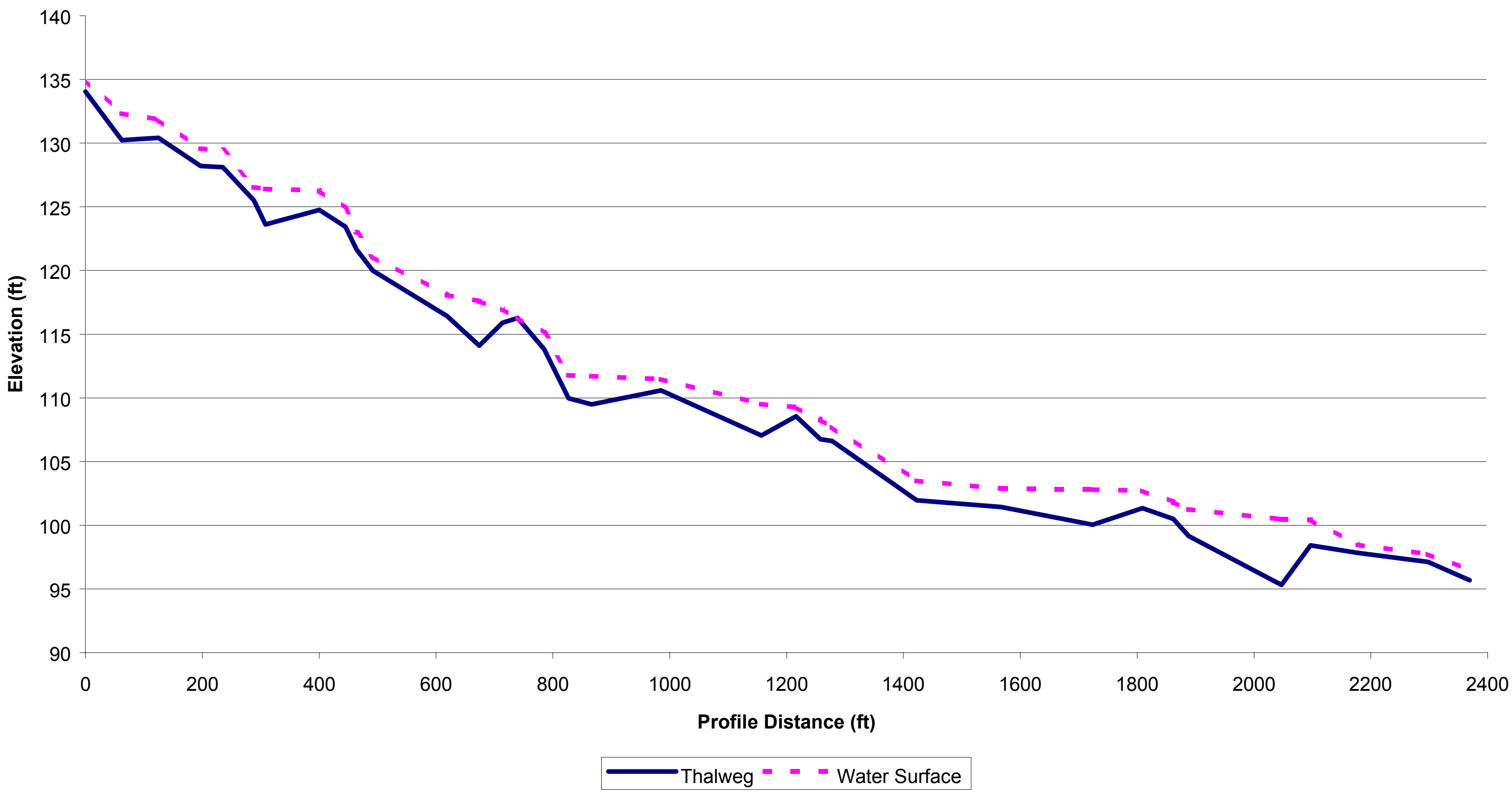


Figure 12. The longitudinal profile of the existing stream channel prior to the implementation of the restoration project. The survey was conducted beginning at station 0 (upper project boundary) to approximately 2350 feet prior to the implementation of the restoration project in the fall of 2002. The survey was not completed for the lower approximate third of the project area. Note the lack of pool habitat within the project area.

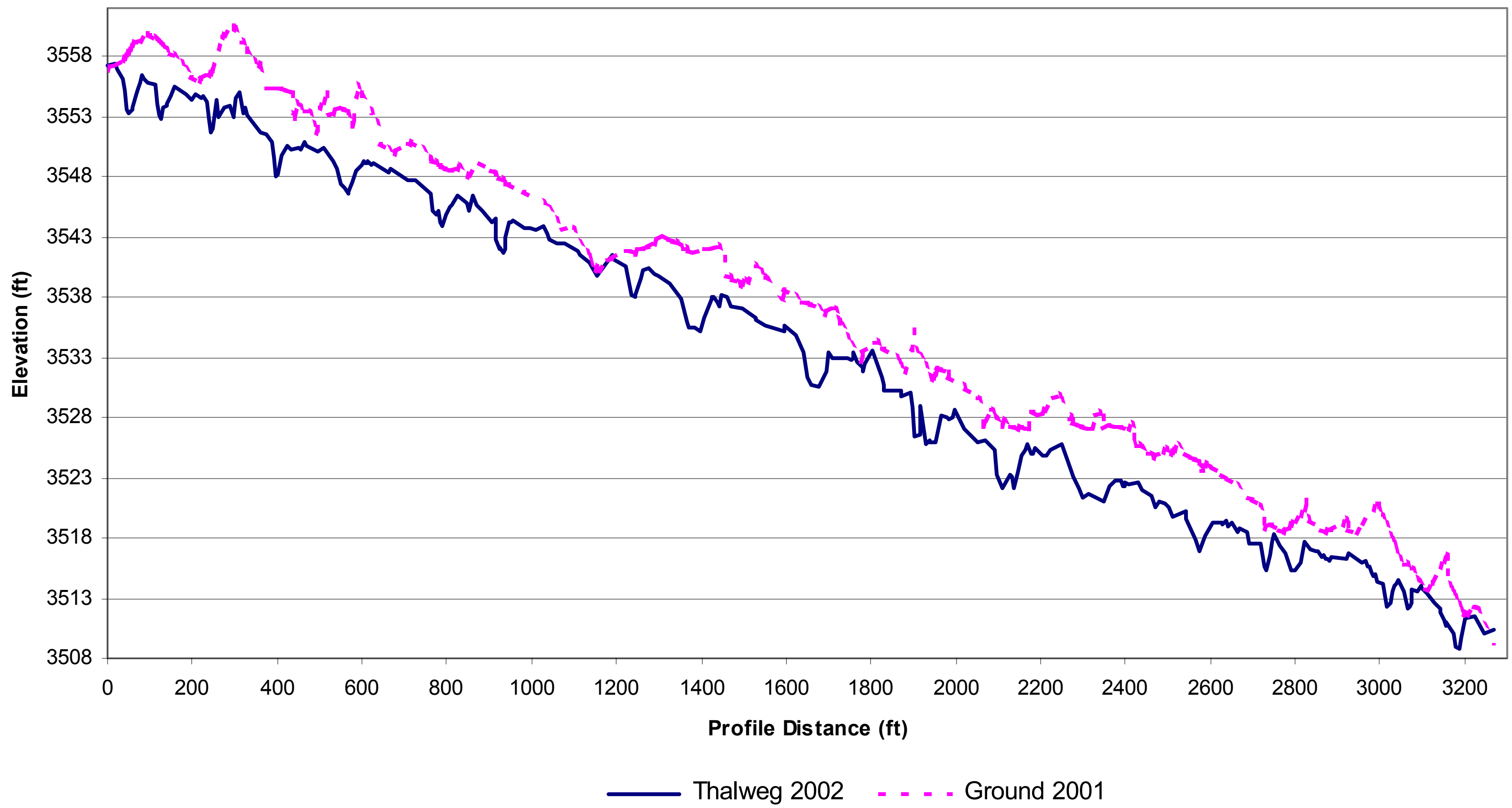


Figure 13. The longitudinal profile of the constructed stream channel thalweg (Thalweg 2002). The survey begins at the upper project boundary (station 0) and proceeds downstream to the lower project boundary (approximate station 3200). The elevation of the riparian area prior to channel construction is represented by the Ground 2001 line. The stream channel prior to channel construction was not located within the same general plan view as the newly constructed stream channel (see Appendix Figure 1A). Therefore, the existing stream channel longitudinal profile (Figure 12) 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.

Grave Creek Demonstration 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 the restoration of approximately 840 feet of channel on lower Grave Creek beginning at the Vukonich Bridge and proceeding downstream. Additional contributors included U.S Fish and Wildlife Service (Partners for Wildlife Program), the U.S. Forest Service, the Natural Resource Conservation Service, the Kootenai River Network, Water Consulting Incorporated, Kirby Excavating, and local landowners Pat and Blanch Flanagan. This project was completed during November 2001, and was termed the Grave Creek Demonstration Project because in addition to returning a relatively short reach of lower Grave Creek into a properly functioning stream, it was intended to serve in a working example of the practical solutions possible with natural stream restoration techniques.

The Grave Creek Demonstration Project reconstructed approximately 840 feet of stream channel, with approximately half the length of the project consisting of a 20 feet high eroding bank. This bank contributed substantial amounts of sediment annually to Grave Creek, and was treated by contouring the eroding bank and constructing a 15 feet wide armored bank terrace to prevent the stream from regaining access to the toe of the slope (Figure 14). The project also planted the bank with grass, and installed 2 J-hook vanes and a rock vane that were designed to center flow toward the channel thalweg. Throughout the remainder of the project area, we installed an additional cross vane, 4 rootwad complexes, and transplanted approximately 6,300 square feet of sod mats and numerous shrub clumps to center stream flow, increase fisheries habitat pool habitat and complexity, and stabilize stream banks. The project also accommodated an existing water right within the project area by installing a flashboard headgate at the point of diversion and a McKay flat panel fish screen to eliminate juvenile fish entrainment in the irrigation ditch.

Monitoring efforts associated with the Grave Creek Demonstration Project were designed to assess how the project changed this section of Grave Creek relative to before project implementation and to assess how the stream channel responds after the restoration work. The primary tools for monitoring these assessments include longitudinal gradient profiles (Figure 15), several cross sectional surveys (Figures 16-20) and photo points.

Figure 14. The top photograph shows the lower portion of the Grave Creek Demonstration Project prior to project initiation. The lower photograph shows the lower portion of the restoration project after project implementation. Note the constructed terrace and stream structured in the lower photograph designed to direct stream flow away from the large eroding bank. The project also decreased the slope of this bank and seeded it with grass.

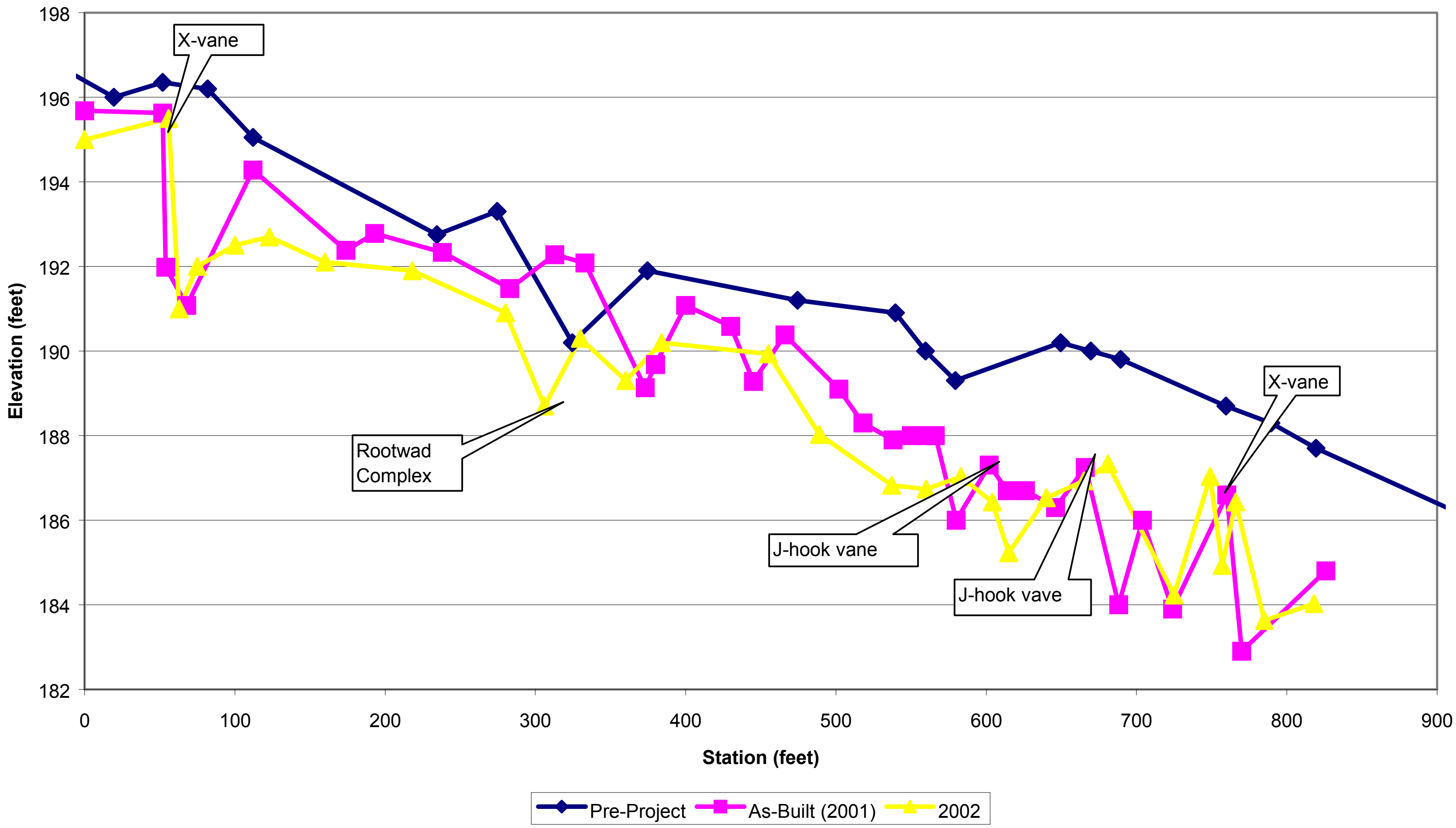


Figure 15. The longitudinal profile of the reach of Grave Creek located within the Grave Creek Demonstration Project. The survey was completed before (April 2001) and after (2001 and 2002) project completion. The station (longitudinal location measured at the channel thalweg) begins at the upstream boundary of the project. Locations of project structures are also noted on the figure.

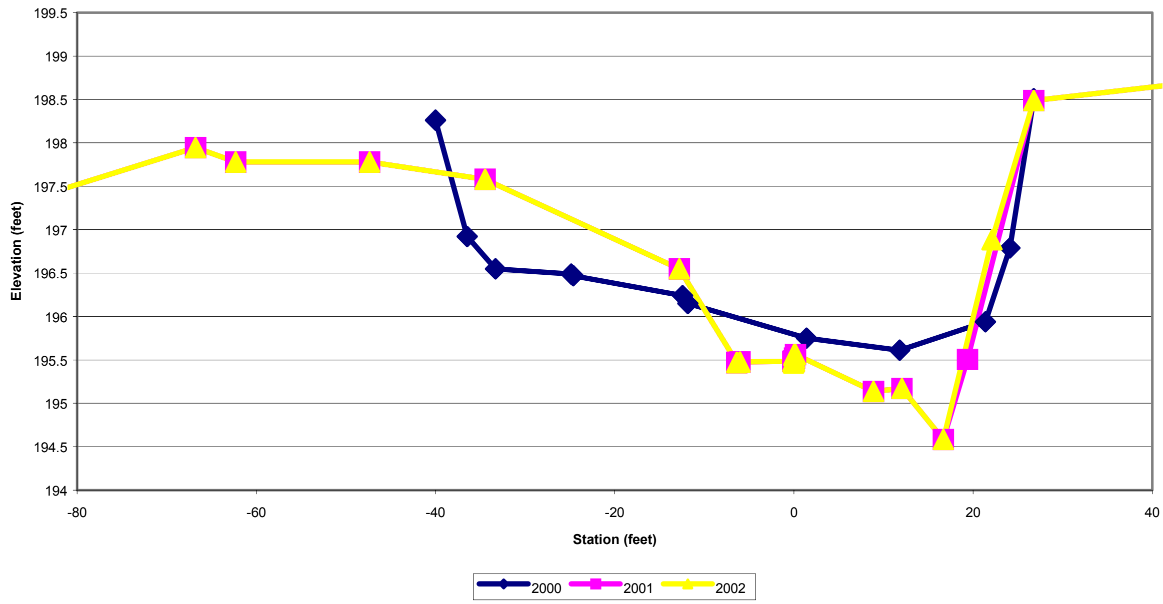


Figure 16. Cross Section 1, located at station 55 within the Grave Creek Demonstration Project. Surveys were conducted before (April 2001) and after (2001 and 2002) project construction. Surveys in 2001 and 2002 were conducted each year after peak spring flows. This cross section was classified as riffle habitat in all three years.

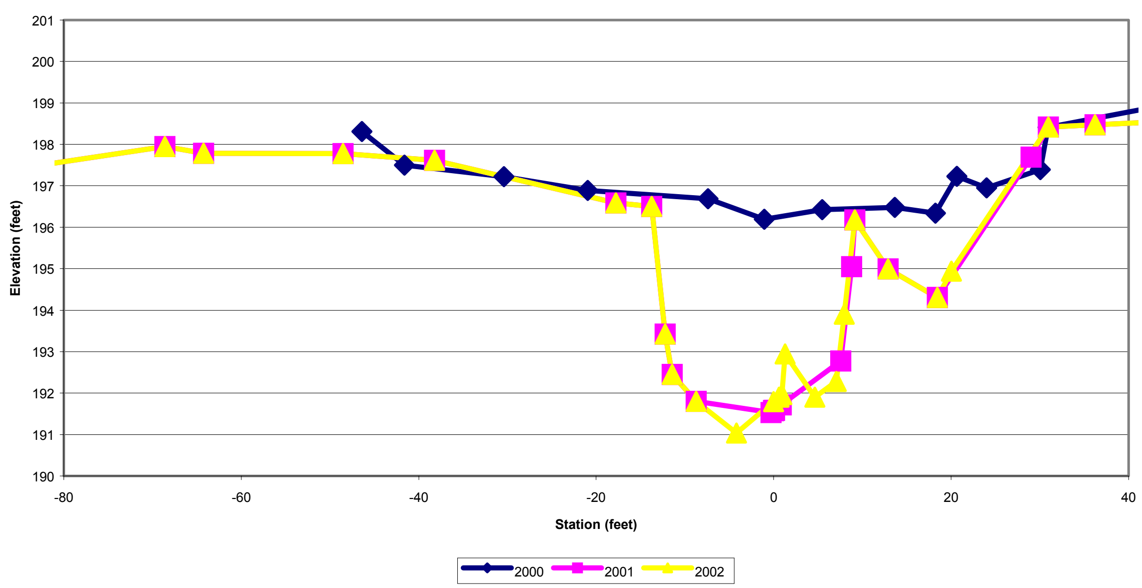


Figure 17. Cross Section 2, located at station 70 within the Grave Creek Demonstration Project. Surveys were conducted before (April 2001) and after (2001 and 2002) project construction. Surveys in 2001 and 2002 were conducted each year after peak spring flows. This cross section was classified as pool habitat in all three years.

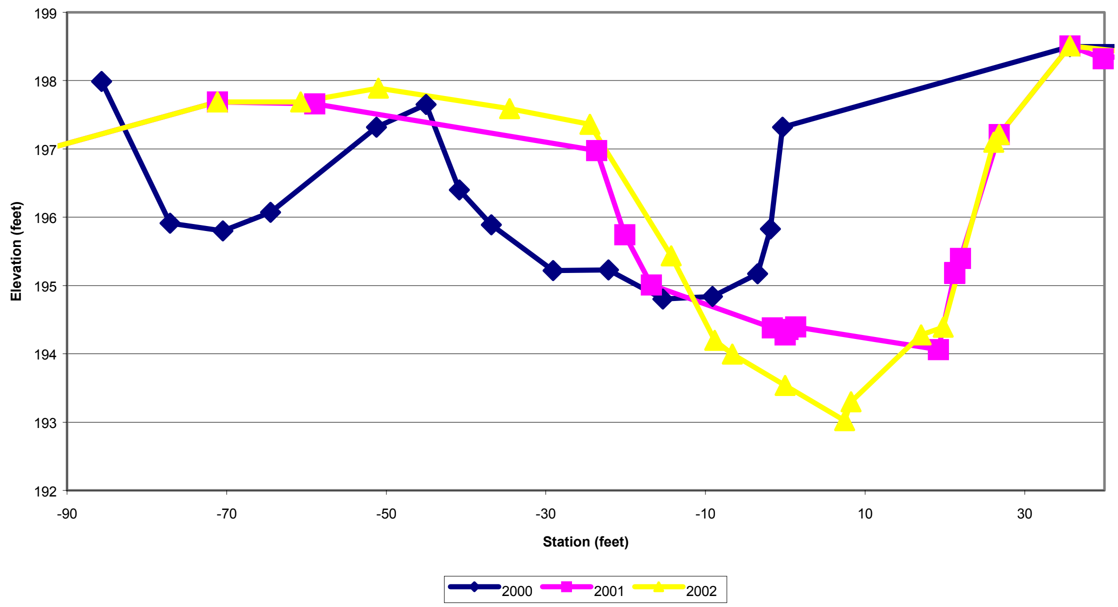


Figure 18. Cross Section 3, located at station 122 within the Grave Creek Demonstration Project. Surveys were conducted before (April 2001) and after (2001 and 2002) project construction. Surveys in 2001 and 2002 were conducted each year after peak spring flows. This cross section was classified as riffle habitat in all three years.

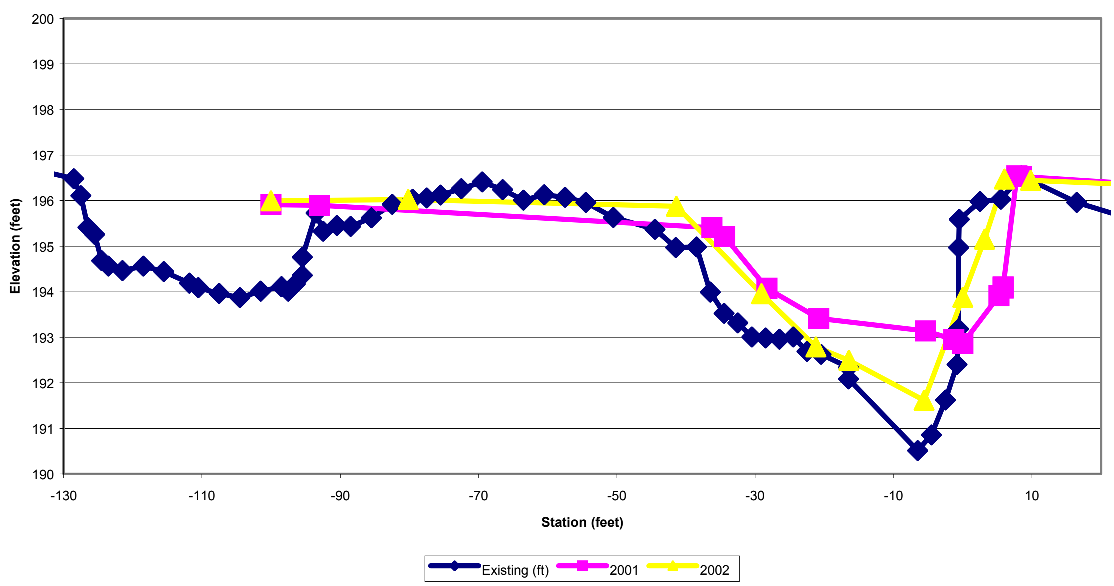


Figure 19. Cross Section 4, located at station 245 within the Grave Creek Demonstration Project. Surveys were conducted before (April 2001) and after (2001 and 2002) project construction. Surveys in 2001 and 2002 were conducted each year after peak spring flows. This cross section was classified as riffle habitat in all three years.

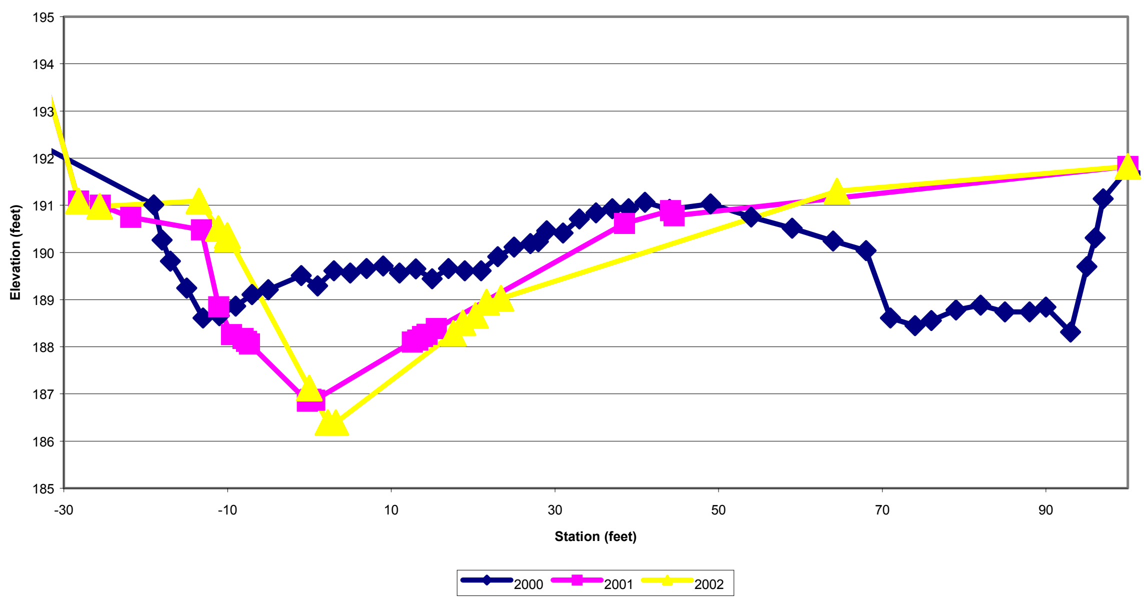


Figure 20. Cross Section 5, located at station 652 within the Grave Creek Demonstration Project. Surveys were conducted before (April 2001) and after (2001 and 2002) project construction. Surveys in 2001 and 2002 were conducted each year after peak spring flows. This cross section was classified as riffle habitat in all three years.

The Grave Creek Demonstration Project changed the pattern, profile, and dimension of this section of Grave Creek. Prior to the implementation of the Grave Creek Demonstration Project, this section of Grave Creek was homogeneous, consisting of long riffles (Figure 15). Although the overall stream gradient remained unchanged as a result of this restoration work (1.3%), this project increased the overall diversity of the longitudinal profile within this reach (Figure 15). We collected information needed to produce 5 cross sectional surveys of the stream channel within the restoration reach prior (April 2001) to project implementation (Table 6, and Figure 16-20). Four of these surveys were performed within riffles and one was within a pool. We repeated surveys at these 5 locations and 5 additional locations in 2001 and 2002 after the annual peak flows had occurred. We used analyses of variance (ANOVA) and subsequent multiple comparisons (pending significance [p =0.10] of the ANOVA) to determine if cross sectional area, bankfull width, mean and maximum bankfull depth, and width to depth ratios differed between years. Although we observed substantial changes in the five parameters we examined at each cross sectional survey, only cross sectional area, bankfull width, and maximum depths changed significantly between years (Table 7). In all cases survey parameters did not differ significantly between 2001 and 2002 surveys. The largest differences in all parameters except cross sectional area were observed between 2001 and 2002. These results support the conclusion that this restoration project ultimately decreased channel width, cross sectional area, width to depth ratio, and increased channel depth.

Bankfull width and width to depth ratio decreased and cross sectional area, mean and maximum depth increased from 2001 to 2002. Although these comparisons were not significantly different between years, these observations are consistent with the hypothesis that the channel slightly incised between years. This was the case at 3 of the 5 existing cross sectional surveys conducted in 2001, 2001 and 2002 (Figures 18, 19 and 20).

The Grave Creek Demonstration Project increased the quantity and quality of pool habitat within this section of Grave Creek. Based on the 2002 survey, the total number of pool remained similar before and after project construction, but total pool length increased slightly (16.6%), and both mean and maximum pool depths increased substantially (36.8 and 53.5%, respectively). The total number of pools from 2001 to 2002 decreased as a result of channel adjustments at bankfull discharge. However, the total length of pools and therefore percent pool habitat composition remained similar between the two years (Table 7). The loss of pools between 2001 and 2002 may partially be explained by the loss of some of the pool tail crests. For example, this is illustrated at station 120 on the longitudinal profile (Figure 15). Statistical comparisons of pool characteristics for pre and post project implementation were not performed because the pool inventory was a complete census of pools present within this reach of Grave Creek. Therefore, the pool characteristics are parameters, not estimates. 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 Grave Creek.

Prior to the implementation of this project, a substantial proportion of the Grave Creek stream channel within the project area consisted of a multiple thread channel (Figures 18,19 and 20). However, since the completion of the project, the restoration work has been exposed to 2 spring runoff events, and the channel has continued to maintain itself as a single thread channel. The ability of this channel to maintain itself as a single thread and the effective decrease in width, cross sectional area, and width to depth ratio (see above), and increased depth should increase shear stress and the channel's ability to transport course sediment through this reach of Grave Creek.

Table 6. Comparisons of mean cross sectional area, bankfull width, mean and maximum bankfull depth, and width to depth ratio for cross section surveys within the Grave Creek Demonstration Project Area, before (April 2001) and after (2001 and 2002) project construction. Analyses of variance and subsequent multiple comparison tests were performed to determine if cross sectional area, bankfull width, mean and maximum bankfull depth, and width/depth ratio differed by year.						
Year	**Number of Cross Sections**	**Cross Sectional Area (sq. ft.)**	**Bankfull Width (ft)**	**Mean Depth (ft)**	**Maximum Depth (ft)**	**Width/Depth Ratio**
April 2001	5	139.2	87.2	1.60	3.07	57.2
2001	10	108.6	71.9	1.57	3.80	51.9
2002	10	115.8	65.2	1.77	4.30	39.2
Significant Comparisons		April 2001/2001 April 2001/2002	April 2001/2002	P = 0.43	April 2001/2002	P = 0.195

Table 7. Pool characteristics based on longitudinal channel surveys through the Grave Creek Demonstration Project reach before (April 2001) and after (2001 and 2002) project completion. The percent pool habitat (based on total length) is also presented.									
	199?			**2001**			**2002**		
Pool #	**Pool Length (ft)**	**Max Depth (ft)**	**Mean Depth (ft)**	**Pool Length (ft)**	**Max Depth (ft)**	**Mean Depth (ft)**	**Pool Length (ft)**	**Max Depth (ft)**	**Mean Depth (ft)**
1	77.5	2.82	2.54	60	6.37	4.16	67	6.51	4.63
2	82	3.76	3.31	75	3.33	2.84	104	5.82	4.33
3	100	5.31	3.70	67	4.57	3.22	192	5.5	4.17
4	110	3.08	2.43	36	3.54	2.71	68	5.15	3.27
5				36	5.16	4.03			
6				160	5.93	4.21			
Mean	**92.4**	**3.74**	**3.00**	**72.3**	**4.82**	**3.53**	**107.8**	**5.75**	**4.10**
Total	**369.5**			**434**			**431**		
% Pool Hab.	**53.0**				**53.0**		**44.0**		

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 above). Project construction work began during the fall of 2002. The objectives of the project are 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 majority of the revegetation work was not completed in the late fall of 2002 due to unfavorable weather conditions that prohibited planting. The revegetation work is scheduled to be completed during the spring of 2003, and is expected to serve as the primary stabilization mechanism in the long-term.

This section of Grave Creek that the Phase I Restoration Project encompasses 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. These changes have resulted in a much less stable stream than would have likely occurred naturally (WCI 2002). An example of the stream condition prior to the restoration project is shown in Figures 21 and 22.

Figure 21. Existing condition of Grave Creek Phase I Project Area. Note the extremely high channel width/depth ratios, excessive sediment supply, multiple channel development, and poor riparian conditions on foreground streambank.

Figure 22. Lower Grave Creek within the Phase I Restroation Project Area prior to restoration work (top photograph). Upon completion of the Grave Creek restoration work (bottome photograph) stream channel width was decreased and the amount and complexity of rearing habitat for juvenile salmonids was increased.

This restoration project changed the dimension, pattern and longitudinal profile of this section of Grave Creek. The intended result was to produce changes achieved the objectives stated above and were sustainable in the long-term. Table 8 presents the existing and design criteria for some important geomorphical stream characteristics. We surveyed 27 cross sectional surveys within this section of Grave Creek prior to project construction in order to describe existing conditions. Figures 23 – 28 represent 6 cross sections located throughout the project area that typify the conditions that existed within this section of Grave Creek prior to the restoration project, and the changes that will occur at each location as a result of the restoration project. The stream consisted of multiple channels throughout much of the project reach with lateral channel migration common between and within years. These conditions resulted in an over widened and shallow channel with a mean bankfull width ranging from 45-240 feet, a mean bankfull depth of 1.24 feet, and a mean width to depth ratio of 93.5. The designed channel will reduce the mean width and width to depth ratio to 52 feet and 10, respectively. The designed mean bankfull depth will range from 2.2-2.4 feet.

The 41 stream restoration structures described above, increased channel diversity within the project area along the longitudinal profile (Figure 29). The existing stream channel prior to the implementation of this project contained long riffle sections and relatively low sinuosity (Table 8). 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 8).

The Grave Creek Phase I 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 (2000) and the as built channel (2002) using the longitudinal profiles from each respective year. We realized an almost nine fold increase in the total number of pools present in this section of Grave Creek as a result of this restoration project. The total number of pool increased from 3 pools to 26 pools. However, due to a decrease in channel width the total area of pools remained relatively constant (approximately 90,000 square feet) as a result of the in stream channel work, but the total volume of maximum pool habitat increased by 230% (173,000 and 570,260 ft^3, respectively), due primarily to increased pool depth. We were not able to calculate mean pool volume from the longitudinal profiles. 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 Grave Creek.

Table 8. 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	10	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

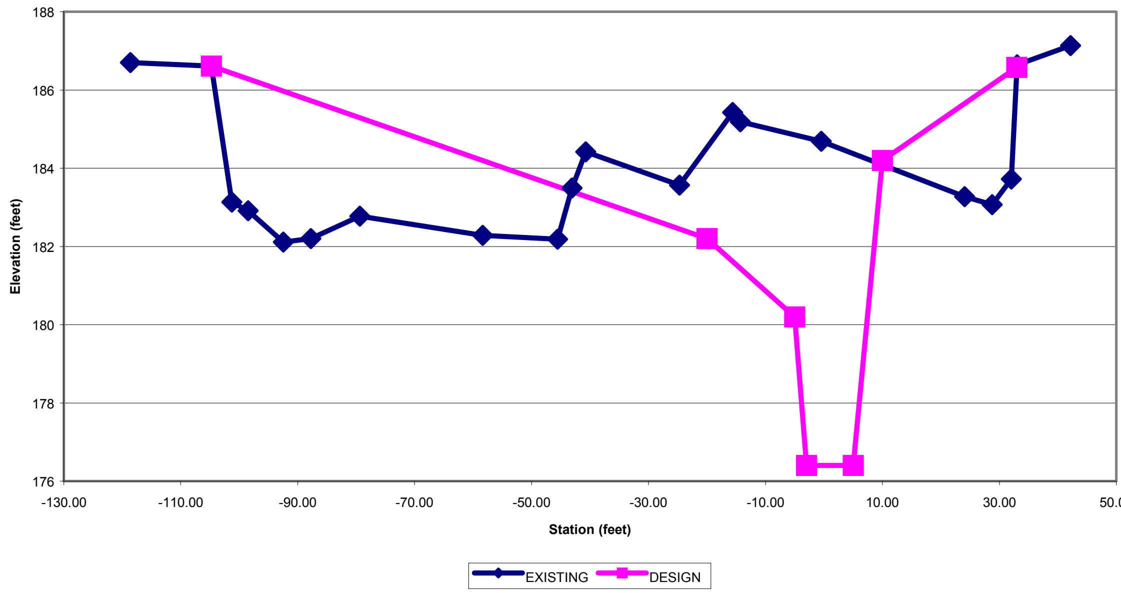


Figure 23. Cross Section 1, located at station 448 within the Grave Creek Phase I Project. This figure graphically displays the results of surveys were conducted before (existing) and the designed specifications at this site.

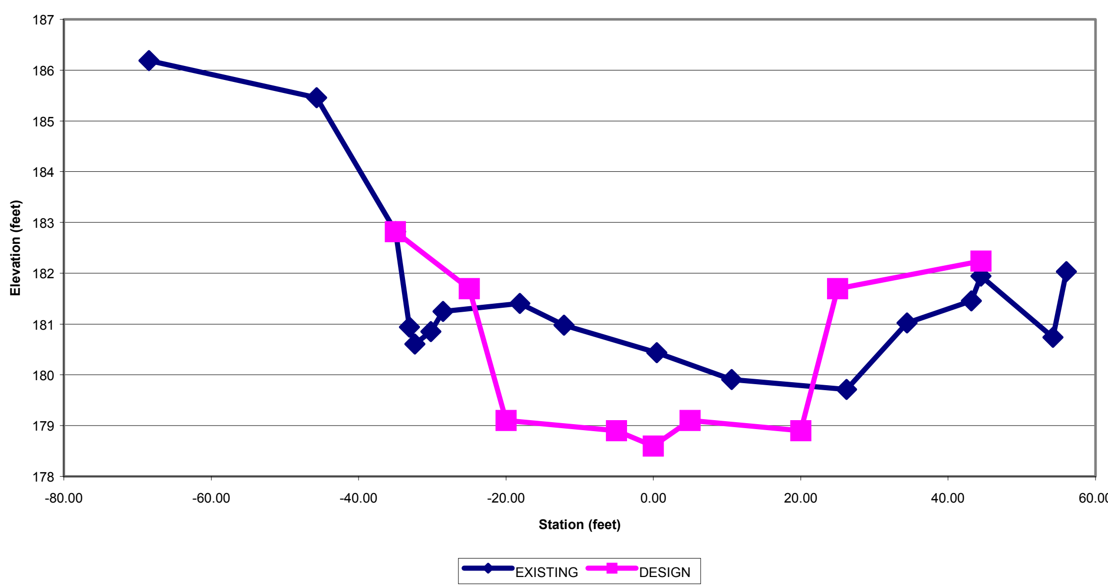


Figure 24. Cross Section 2, located at station 751 within the Grave Creek Phase I Project. This figure graphically displays the results of surveys were conducted before (existing) and the designed specifications at this site.

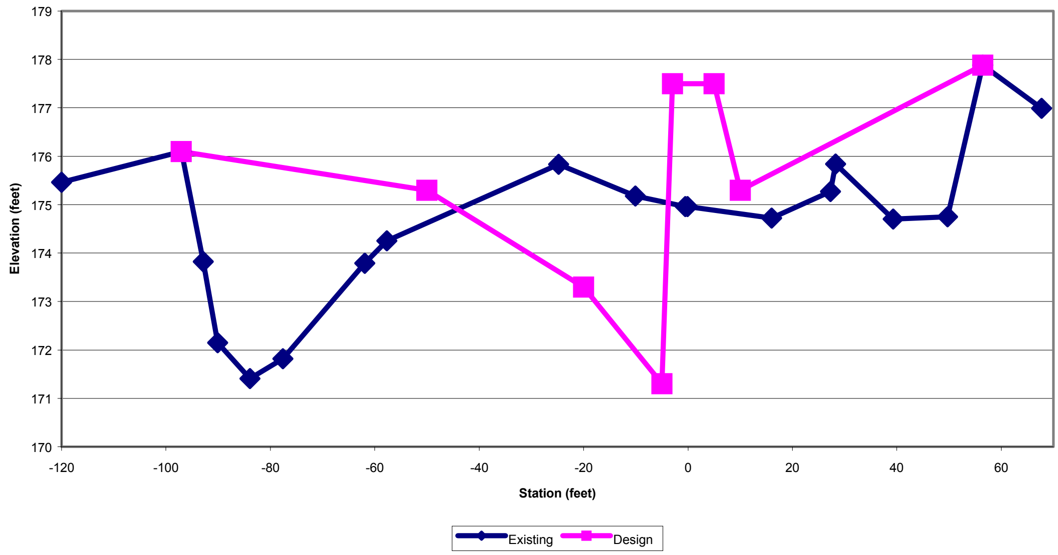


Figure 25. Cross Section 3, located at station 1379 within the Grave Creek Phase I Project. This figure graphically displays the results of surveys were conducted before (existing) and the designed specifications at this site.

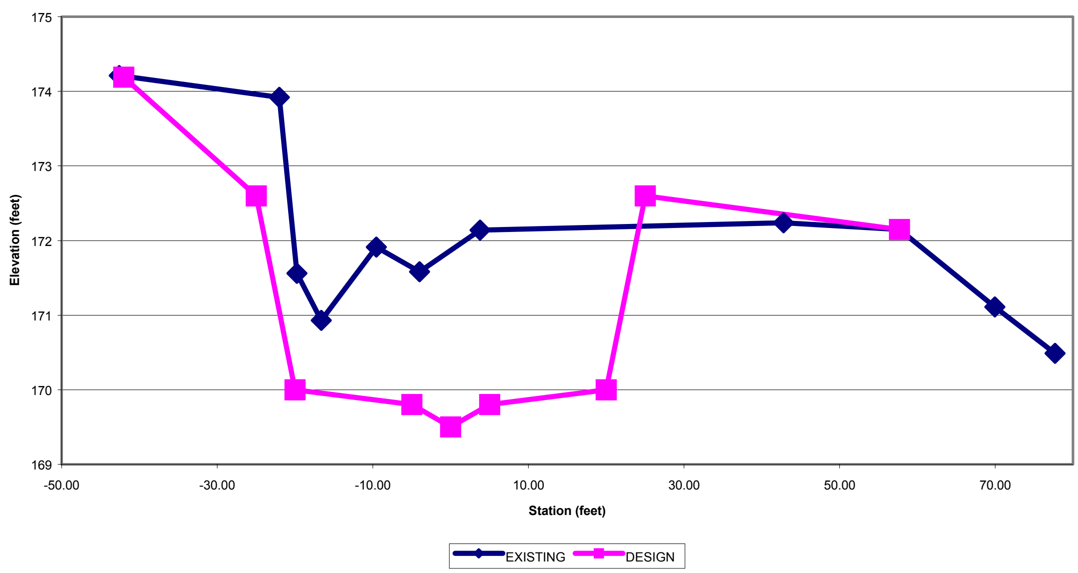


Figure 26. Cross Section 4, located at station 1790 within the Grave Creek Phase I Project. This figure graphically displays the results of surveys were conducted before (existing) and the designed specifications at this site.

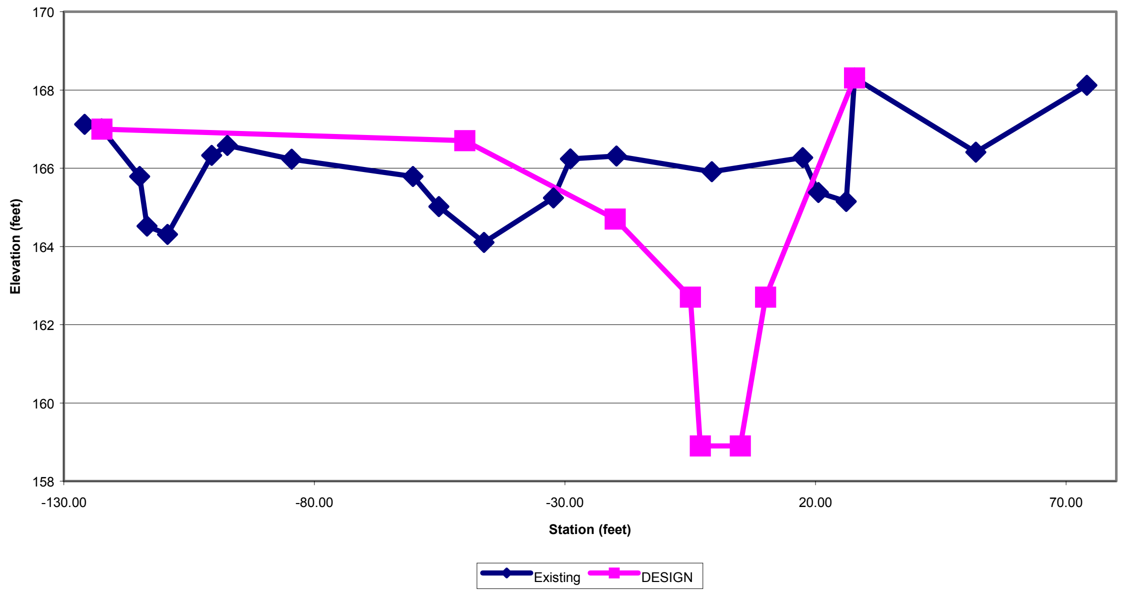


Figure 27. Cross Section 5, located at station 2387 within the Grave Creek Phase I Project. This figure graphically displays the results of surveys were conducted before (existing) and the designed specifications at this site.

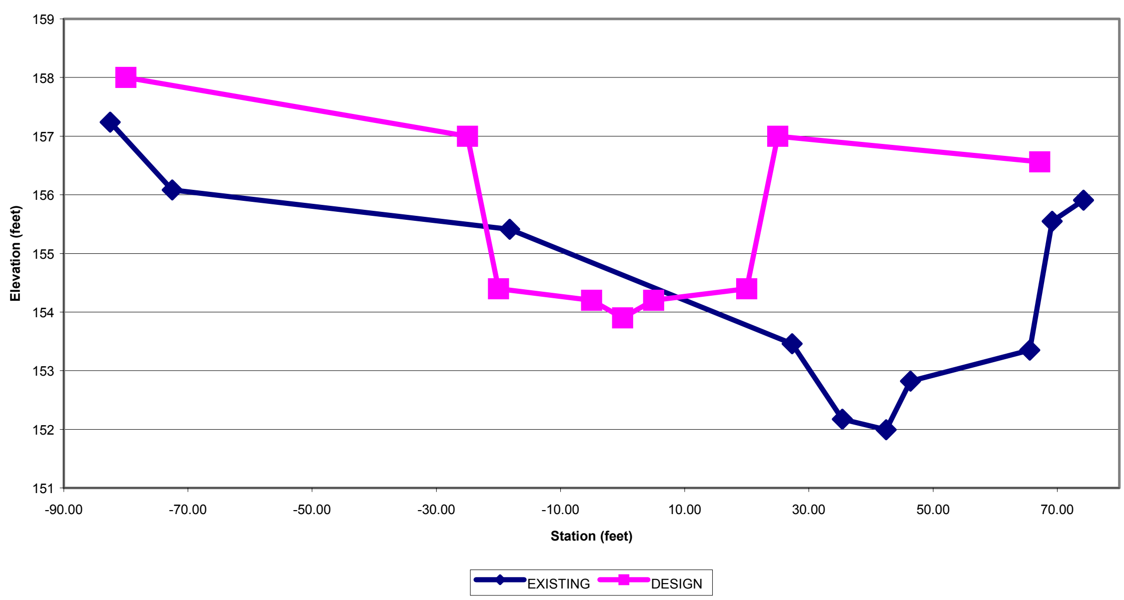


Figure 28. Cross Section 6, located at station 3700 within the Grave Creek Phase I Project. This figure graphically displays the results of surveys were conducted before (existing) and the designed specifications at this site.

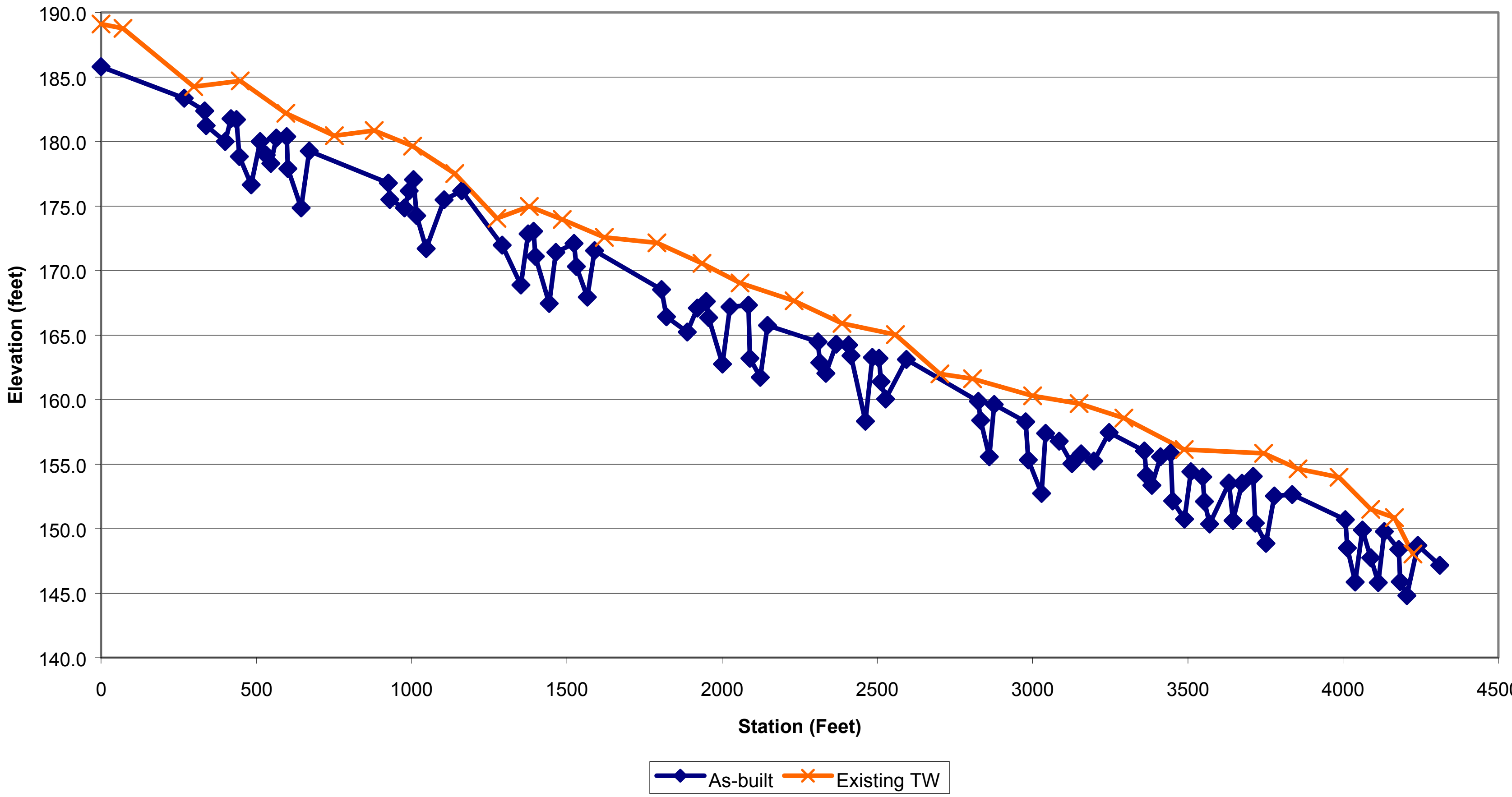


Figure 29. The longitudinal profile of the reach of Grave Creek located within the Grave Creek Phase I Restoration Project. The survey was completed before (2001) and after (Winter 2002) project completion. The station (longitudinal location measured at the channel thalweg) begins at the upstream boundary of the project.

Discussion

Channel dimensions have maintained the designed channel parameters of width, depth and the designed stable channel types on both the Grave and Libby Creek Demonstration Projects. Streams with C3 channel types should have width/depth ratios >12 and typically average approximately 29.3 (Rosgen 1996). Currently, both the Libby Creek and Grave Creek Demonstration Project areas are within this criterion. The Libby Creek Demonstration Project had three flow events at or above bankfull discharge during the winter/spring immediately following construction (USFS discharge data), and has continued to maintain a stable channel form. Both restoration projects were designed to address the cumulative effects of stream channel disequilibria that had resulted in bedload aggredation, lateral accretion, loss of fish habitat and increased land loss downstream. Increases in width/depth ratio are often associated with accelerated stream bank erosion, excessive sediment deposition, stream flow changes, or stream channel widening due to evolutionary shifts from one stream type to another (Rosgen, 2001*a*). Past and present land management practices and channel alterations had changed both streams from stable C3 channel types to over-widened C3 and D3 channel types. Transformations of these types typically result in short and long-term loss of physical and biological function and produce an increase is total sediment load from lateral migration (Rosgen 2001*b*).

We documented an increase in the substrate particle size distribution at the Libby Creek Demonstration Project (Table 2) that we believe resulted from decreasing the width depth ratio and increasing the quantity of pool habitat. These changes created a hydraulically efficient channel that shifted the substrate particle size to larger sized substrate. Walters (1995) discussed the abundance of stream benthic invertebrates correlating positively with particle substrate size. These findings are consistent with our observations also (see Chapter 1). We observed an overall increase in macro-invertebrate species richness relative to pre-project construction in four of our six indices of species diversity at the Libby Creek Demonstration Project. We will continue to monitor macro-invertebrate production at this site in order to continue to assess changes in the benthic community that may result from the restoration work.

We increased pool habitat quality and quantity by using native materials to create structures that prevent bank erosion and create scour by creating secondary velocity cells in the stream channel away from the near bank region of the channel. We used J-hook vanes and rootwad complexes to create pool habitat a stabilize banks. Rosgen (2001*c*) showed J-hook vanes decreased velocity, shear stress, stream power and velocity gradients in the near bank region and increased these conditions in the center of the channel. Although the J-hook vanes in our projects did protect the banks, the rootwad structures created deeper pool habitat, especially in the Libby Creek Demonstration project (Figure 8). The rootwads provide additional cover all life stages of aquatic vertebrates. Rosgen (2001*c*) found that vortex rock weirs and root wads installed in previous years produced bank-eddy erosion during major floods resulting in stream bank erosion and loss of some structures. These findings indicate the science of natural channel design is still evolving, with improvements in structure design occurring in this relatively new science. We have found some problems with a portion of our structures discussed in the 'maintenance' portion of this report. In future projects we will attempt to integrate the best pool habitat forming structures with the most efficient grade control and bank protecting structures. Nevertheless, we observed an overall increase in salmonid abundance at both the Grave and Libby Creek Demonstration Projects after project completion (see Chapter 1). Rainbow trout and brook trout abundance at the Libby Creek Demonstration Project increased 68.3 and 82.9% respectively, compared to pre-project monitoring data. Juvenile bull trout were also observed at the Libby Creek site for the first time in several years. Total salmonid abundance (rainbow trout, cutthroat trout and brook trout) at the Grave Creek site also increased approximately 33% relative to pre-construction monitoring.

Although our monitoring efforts did show that the Grave Creek and Libby Projects did not prevent all bank erosion through out the project reaches, both projects did substantially reduce erosion at several sites within each project where high bank erosion rates were occurring prior to project implementation, which substantially reduced overall sedimentation rates from within the project sites. Rosgen (2001*b*) addressed two factors influencing bank erosion rates: near-bank shear and bank conditions. Bank conditions were broken down into bank height/bankfull ratio, rooting depth and density, surface area protection, bank slope and material composition of the bank. Using natural channel design structures we were able to decrease near-bank shear and move shear away from the bank and into the middle of the channel, which resulted in an increase if bank roughness. We increased bedload conveyance by creating a channel with proper dimension pattern and profile. We decreased bank height ratio indices from 10 to 1.5-1.9 by developing floodplains at the tow of large unstable banks with high bank slopes. We expect that rooting depth and density will increase as planted vegetation becomes established, and thus further stabilize these sites.

Leopold, et al, (1964) found that river form is associated with the eight variables of slope, width, depth, velocity discharge, boundary roughness, size of sediment transported, and concentration of sediment. If any one variable is changed, it sets up mutual, concurrent adjustments of the other variables. All these variables must be considered while installing structures in a stream channel.

We have performed some maintenance on the Libby Creek Demonstration project following high flow events on the stream. One reason for maintenance in the project was the aggredation of bedload material in pool tail outs or glide areas that was deposited during high flows. Degraded glides decreased pool depth, which increased shear stress and hydraulic jump over structures such as J-hook and cross vanes. This in turn increased scour below the structure. This increased scour also undermined some of the footer rocks at some structures, which further reduced the integrity of these structures. To address this problem during the repair/maintenance activities, we created hardened tail outs using material larger than the D95 (approximately 210 mm) of the channel and used logs for the vane portion of the J-hook instead of rock. We used geotextile material to seal the log vanes and prevent undermining.

In future projects we will build cobble grade control and log vane structures in glides below pool forming structures, as we did for the Libby Creek Cleveland Project constructed in the fall of 2002. We expect that the log vanes in the upper portion of pools will maintain gradient within the lower end of the riffles. When a new channel is constructed, gradient control in lower and upper portions of riffles is needed to maintain the channel stage. 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). Therefore, increasing the amount of gradient control should help to stabilize the channel as rapidly as possible and decrease maintenance in future natural channel designed projects.

We will continue our rigorous monitoring efforts of stream naturalization projects in future years in an effort to determine whether these projects are meeting the objectives identified during project planning. Our monitoring program will continue to assess the effectiveness to restoring stream stability and reducing impacts to stream biota. Bank erodibility assessments will also be continued along with photo documentation of vegetated condition of stream channel banks.

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Chapter 3

Young Creek Natural and Hatchery Origin Juvenile and Adult Cutthroat Trout Production Estimates

Abstract

Young Creek is one of the most important westslope cutthroat trout (*Oncorhynchus clarki lewisi*) spawning tributaries to Koocanusa Reservoir because it represents one of the last know genetically pure populations of westslope cutthroat trout in the region and it is also one of the most potentially productive tributary streams to Koocanusa Reservoir. Westslope cutthroat thrived in Koocanusa Reservoir from the early 1970s through the early 1980s, adfluvial runs of cutthroat in Young Creek were abundant during this period. However since then the abundance of adfluvial cutthroat trout in the reservoir and Young Creek has declined due likely to a combination of factors including the natural aging of the reservoir, increased predation and food competition, cutthroat trout entrainment through Libby Dam (especially during periods of excessive drawdown), and loss of habitat within Young Creek. Montana FWP conducted a pilot study from 1996-2000 that utilized remote site incubators (RSIs) in Young Creek in an effort to increase the abundance of adfluvial and resident westslope cutthroat in Young Creek. Westslope cutthroat trout eggs were obtained from Washoe Park State Fish Hatchery in Anaconda, Montana. We estimated that the RSIs produced approximately 57,000 – 89,700 cutthroat trout fry from 1997-2000, with egg-to-fry survival ranging from 53-75%. This is an experimental pilot project to assess the effectiveness of RSIs for re-establishing cutthroat in Libby Reservoir tributaries and possibly the reservoir itself. We attempted to thermally mark the otoliths of embryo cutthroat trout at the Anaconda Hatchery in order to differentiate between hatchery and natural origin fish. Montana FWP operated the Young Creek fish trap in 1998 to monitor juvenile recruitment and adult escapement from Young Creek and Koocanusa Reservoir respectively, in order to assess the success of the RSI project. We randomly collected otolith samples from juvenile and adult cutthroat trout from 1998-2002 at the Young Creek fish trap, but the efforts to thermally mark the hatchery cutthroat trout were successful only in 1999. This project will investigate an alternative method to differentiate between hatchery and natural origin fish collected at the trap in an attempt to produce the most meaningful results from this study. This alternative technique will assess the feasibility of using trace elemental differences between the Anaconda Hatchery and Young Creek, and ultimately differences in trace elemental ratios within the otoliths of hatchery and natural origin cutthroat trout collected at the Young Creek trap.

Introduction

Young Creek is a 17 km long tributary to Libby Reservoir, 5 km south of the Montana-British Columbia border that drains a 119 km^2 basin of the Purcell Mountains. Median annual low and high flows range from 5 to 100 cfs, respectively.

Fish population data collected by Montana Fish, Wildlife and Parks (FWP), prior to the construction of Libby Dam, indicates that Young Creek contained a fish species assemblage consisting mainly of brook trout *Salvelinus fontinalis* and westslope cutthroat trout *Oncorhynchus clarki lewisi*. Prior to reservoir impoundment, the lower seven miles of the stream supported a fluvial run of cutthroat trout from the Kootenai River. When population estimates were first conducted in Young Creek 1967 to 1969, the headwaters of Young Creek supported mostly resident cutthroat trout. Brook trout were a large percentage of the fish population in the lower seven miles of Young Creek (Huston 1972). In 1969 Young Creek was chosen as a study stream by Montana FWP and the Army Corps of Engineers (ACOE) to determine the feasibility of converting other Kootenai River tributaries to spawning areas for Libby Reservoir. Montana FWP and ACOE designed and constructed a permanent cement fish weir in Young Creek near the confluence. The weir is capable of capturing adult fish migrating upstream and juvenile fish emigrating downstream.

In August of 1970, Montana FWP chemically treated a seven mile section starting four miles from the confluence (T37N, R28W, Sec.5), removing all fish. The lower four miles of stream were not treated. In July, 1975, Montana FWP removed over 500 brook trout from the meadow section located in the lower four miles of Young Creek, using electrofishing gear.

Starting in 1970, Montana FWP began stocking westslope cutthroat fry into Young Creek in consecutive years through 1974, averaging about 50,000 fish per year (Table 1). The goal of the first four years of stocking was to create an adfluvial population of westslope cutthroat in the reservoir that would return to Young Creek as adults to spawn (May and Huston 1975). In addition, Montana FWP stocked about three million cutthroat trout directly into the reservoir from 1972 through 1975. Stocking of cutthroat trout in Young Creek was continued in 1985 with the stocking of 9,840 fingerlings and approximately 20,000 fry. In 1992, 7,000 westslope cutthroat fry were stocked into Young Creek. Stocking of hatchery cutthroat trout in Young Creek continued through 1995.

Table 1. Stocking summary of westslope cutthroat trout in Young Creek by the state of Montana.		
Date	**Number Fish**	**Size (mm)[a]**
9/8/70	50,706	25
8/24/71	25,344	25
9/8/71	25,156	25
7/12/72	32,375	25
8/28/72	21,840	25
6/20/73	31,873	25
7/23/74	30,636	25
8/14/74	14,052	25
8/5/75	59,536	25
7/18/85	9,840	178
9/10/85	19,950	25
8/3/92	7,000	23
9/15/93	7,126	46
7/27/94	6,554	36
7/27/94	3,606	33
8/23/94	8,009	36
8/11/95	4,191	33
8/22/95	10,100	36
a) Average total length at time of stocking		

Westslope cutthroat thrived in Koocanusa Reservoir from the early 1970s through the early 1980s, adfluvial runs of cutthroat in Young Creek were abundant during this period. After 1984 there was a sharp decrease in adult cutthroat trout migrating into Young Creek (Table 2). Three years after the adults decreased, juvenile cutthroat trout emigrating from Young Creek into Libby Reservoir decreased dramatically as indicated by the number of fish captured by the fish trap in Young Creek (see below). Westslope cutthroat captured in fall gillnets also declined significantly (Kruskall-Wallis ($P<0.01$) between 1978 and 1982 (Huston et al. 1984). Gill net catches of westslope cutthroat remained relatively stable between 1988 and 1996 (Dalbey et. al. 1997). Declines in the reservoir cutthroat population may be attributed to natural aging of the reservoir, increased predation and food competition. Cutthroat are also lost to the reservoir by entrainment through Libby Dam, approximately 60 percent of tags returned by anglers from fish tagged in 1973 and 1974 were captured below the dam (May and Huston 1975). However the reservoir experienced excessive drawdown levels during this period.

Table 2. Summary of westslope cutthroat trout eggs stocked in Young Creek, Montana using remote site incubators (RSIs) 1997-2000.					
Date	**Egg Lot #**	**No. Incubators**	**No. Eggs per Incubator**	**Total Eggs**	**Estimated Fry**
6/5/97	1	6	8,303	49,818	20,453
6/18/97	2	5	17,600	88,000	17,192
7/8/97	3	5	8,000	40,000	14,348[a]
7/9/97	4	2	8,500	17,000	5,000[b]
1997 Total		**18**		**194,818**	**56,993**
6/6/98	1	6	5,000	30,000[c]	19,500[c]
6/13/98	2	6	5,000	29,574[c]	17,449
6/20/98	3	6	4,250	25,000[c]	13,250[c]
6/29/98	4	6	8,500	30,000	19,500[c]
1998 Total		**24**		**114,574**	**69,699**
6/7/99	1[d]	5	10,000	50,000	46,070
6/17/99	2	5	6,000	30,000	25,455
6/27/99	3	5	5,000	25,000	6,800
7/4/99	4	5	5,500	27,500	11,350
1999 Total		**20**		**132,000**	**89,675**
6/12/00	C3[d]	6	8,310	56,000[e]	19,314[e]
6/19/00	D4	5	11,200	29,574[e]	22,450[e]
7/5/00	G4	6	8,310	56,000[e]	37,860[e]
2000 Total		**17**		**141,574[e]**	**79,624[e]**

a) Most eggs died in two incubators from loss of water into the incubators.
b) Most eggs were killed due to vandalism of incubators
c) Emerged fry numbers were estimated by calculating egg to fry survival in the monitored incubator from each egg lot. Initial egg mortality averaged 5% for each incubator.
d) Placed near the channelized section of Young Creek.
e) Initial egg mortality averaged 20-30% because eggs were not picked at the hatchery, number of eggs reported here does not take into account this mortality.

The decrease in the adfluvial population in Young Creek is likely linked to the reservoir population, but other factors such as habitat degradation and competition with brook trout in Young Creek may have also contributed to the Young Creek population decline. During the 1970s timber harvest and road construction increased sediment into Young Creek. From 1975 to 1987, 50 kilometers of new roads were constructed to access timber sales on USFS land in the Young Creek Drainage (USFS Eureka District, unpublished data). Residential land development has increased, affecting the lower four miles of Young Creek. Livestock grazing has also adversely effected habitat quality of Young Creek. Effects of roads and low bank cover ratings negatively correlate to densities of westslope cutthroat trout (Shepard et. al. 1998). In addition, ACOE channelized a 1200-foot section of Young Creek in the lower four miles for flood prevention. The lower four miles of the stream was historically very important to westslope cutthroat trout, as most redds were observed in this reach of the stream during the 1970s and 1980s (Montana FWP files).

Methods

Remote Site Incubators

From 1996-2000 we utilized remote site incubators (RSIs) in Young Creek in an effort to increase the abundance of adfluvial and resident westslope cutthroat in Young Creek. Westslope cutthroat trout eggs were obtained from Washoe Park State Fish Hatchery in Anaconda, Montana. This is an experimental pilot project to assess the effectiveness of RSIs for re-establishing cutthroat in Libby Reservoir tributaries and possibly the reservoir itself. RSIs have been used successfully in the state of Washington using green salmon eggs to reestablish runs in costal streams (Manuel 1992).

Otoliths of incubating trout from brood years 1997-2000 were thermally marked at the Washoe State Park Fish Hatchery using the methodology described in Snelson et al. (2000). Thermal marking of otoliths has been used successfully on early life stages of Pacific salmon *Oncorhynchus spp*., in Washington and Alaska for identification purposes. (Schroder et al. 1996 ; Munk et al.1993; Hagen et al. 1995). Thermal marking has also been used successfully on lake trout *Salvelinus namaycush* in Minnesota (Negus 1997). This marking method utilizes temperature changes to vary growth ring densities in otoliths of fish during the early developmental stages. These bands are retained throughout the fishes' life.

Observing thermal marks requires a microscope of 400X power. This scope, fitted with computerized image scanning capability will allow us to detect thermal marks in otolith samples (OPTIMAS Corp. Software). Otolith thermal marking technology will allow us to sample juvenile and adult westslope cutthroat trout in Young Creek and estimate the proportion of juvenile and adult of hatchery and wild origin, and ultimately allow us to assess the effectiveness of the RSI pilot study.

We placed eyed westslope cutthroat trout eggs in RSIs from 1996 to 2000. During 1997 - 1998 all RSI's were placed within one mile above and below the Forest Service Road

303 bridge (approximately river mile 5.5 on Young Creek). We deployed five incubators near the channelized section in 1999 and 2000 (approximately river mile 4.0 on Young Creek). The remaining RSI's were deployed near the road 303 bridge in 1999 and 2000.

In 1996, approximately 50,000 westslope cutthroat trout fry emerged from RSIs placed in Young Creek, and some westslope cutthroat eyed eggs were placed in artificially constructed redds. However, from 1997 to 2000 we used only RSIs, purchased from the USFWS in Washington, and stocked them with 114,600 to 195,000 eggs annually (Table 2). A total of 194,818 west slope cutthroat eyed eggs were placed in 18 RSIs, ranging from 8,000 to 17,600 eggs in each incubator in 1997 (Table 2). We reduced stocking densities within the RSIs in 1998 to 5,000 per RSI (Table 2) in an attempt to increase egg to fry survival. We monitored one incubator from each lot in order to estimate egg to fry survival. We used water displacement ratios to estimate number of emerging fry during period of high emergence in 1997 and 1998. We counted the number of fry per displacement of one ml of water in a 100 ml graduated cylinder, and then estimated total displacement of all emerging fry to estimate total volume.

We collected up to 20 emerging fry from the RSIs on Young Creek during 1997, 1998 and 1999 to be used as reference samples. Samples were sent the Washington Department of Fish and Wildlife otolith lab for preparation and interpretation of the thermal marks. Reference samples were not collected in 2000.

Young Creek Fish Trap

Montana FWP operated the Young Creek fish trap in 1998, the first time since 1991 to monitor juvenile recruitment and adult escapement from Young Creek and Libby Reservoir respectively, in order to assess the success of the RSI project. Historically most cutthroat remained in the reservoir for at least two years before returning to Young Creek as spawning adults (May and Huston 1975). Because most cutthroat emigrate as two to three year old and spend one to two years rearing in the reservoir, the first significant number of adult cutthroat trout originating from the RSI project would be in 2000 and 2001.

In 1998, Montana FWP personnel began operating the trap in April, similar to past years of operation. We modified the downstream weir so it could be run with less maintenance, and operated the trap from mid-April through July each year. We recorded species and length of all fish captured in both upstream and downstream weirs. In order to estimate downstream trap efficiency each year, we marked a portion of the cutthroat trout captured in the trap with a fin clip and them released above the trap. All fish captured in the trap were examined for the presence of external marks. We estimated overall seasonal trap efficiency by dividing the total number of recaptured cutthroat trout by the total number of unmarked fish captured in the trap.

In 1998 we began collecting samples of juvenile cutthroat trout at the Young Creek trap in order to estimate the total number of emigrating juveniles produced from the RSI's we began using in 1996. Juveniles were collected through 2002. During the period 1998-2001, we attempted to collect at least 20 juvenile westslope cutthroat trout within the following 50 mm size classes: 48-110, 111-160, 161-210, and 2110 to 260 mm total length. Samples were only collected only during days when at least 6 juvenile cutthroat trout each day were captured in the trap. However in 2002, we changed our collection protocols and began systematically collecting samples from every fifth juvenile cutthroat trout. The fish was euphemized and the otoliths and scales were collected on site or the entire fish was frozen and the scale and otoliths samples were later collected in the laboratory. The first potential adult cutthroat trout may have returned from Libby Reservoir to Young Creek in 2000, but we did not capture any adult cutthroat trout in 2000 for otolith analysis. In 2001 and 2002 we collected both juvenile and adult cutthroat trout for otolith analysis.

The sagittal otoliths of individual cutthroat trout collected at the Young Creek trap were dissected and oriented sulcus-side down on a glass plate. The glass plate was then labeled to associate the otoliths with their respective sample. Under a fume hood, the otoliths on the glass plate were surrounded with a preformed rubber mold which was filled with a clear fiberglass resin and placed in an oven for approximately one hour for curing. The resulting blocks of resin containing the otoliths were sectioned and polished. The blocks were lapped on a rotating disc of 400-600 grit Carborundum paper until the core of each otolith was clearly visible under a dissecting microscope. The otoliths were then polished using a rotating polishing cloth saturated with a one micron deagglomerated alpha alumina and water slurry. Upon completion of lapping and polishing, the otoliths were examined with a compound microscope at 100X and/or 400X magnification.

Results

Remote Site Incubators

We estimated that approximately 60,000 and 69,700 cutthroat trout fry emerged from the RSIs in 1997 and 1998 respectively, with egg-to-fry survival ranged from 53 percent to 65 percent from 1996-1998 (Table 2). However, egg-to-fry survival rates in 1999 and 2000 increased to 70 and 75%, respectively, which lead to an increased number of emergent fry during both years compared to the first two years of the study. We estimated that approximately 89,700 and 79,600 cutthroat trout fry emerged from the RSIs in 1999 and 2000, respectively. We believed that two factors influenced egg to fry survival in the RSI's between years; the number of eggs per incubator and the time to emergence. Both factors varied between years and made determining the optimal number of eggs per incubator difficult. However, we recommend keeping densities relatively low (approximately 5,000 – 6,000 per incubator) in order to reduce egg fungus mortality.

Washington Department of Fish and Wildlife (WDFW) mounted otoliths from cutthroat fry that were sampled RSIs and interpolated the otolith thermal marks. WDFW concluded that the marking techniques we used during 1997 and 1998 did not leave an identifiable mark on the otoliths, but the embryos marked in 1999 did produce visible marks (J. Grimm, WDFW, personal communication). We don't know if marking efforts in 2000 were successful or not since reference samples were not collected. Therefore, we would be able to identify only those cutthroat trout captured at the Young Creek fish trap that produced in the RSIs in brood year 1999 using the thermal otolith technology.

Young Creek Fish Trap

We began capturing emigrating juvenile cutthroat in April with the peak number of cutthroat captured during June and July 1998 through 2002 (Figure 1). The mean capture date of all westslope cutthroat trout captured in the Young Creek trap from 1998 to 2002 was June 16. The mean cumulative passage dates for 25, 50 and 75% passage during the period 1998 – 2002 were June 6, June 20, and July 2, respectively. The estimated juvenile trap efficiency from 1998 to 2002 averaged 48.0% (range 25.5 – 74.8%; Table 3).

The historic mean catch of emigrating juvenile cutthroat trout < 250 mm at the Young Creek trap is 837 fish (Table 3). The average catch of cutthroat trout during the period that includes most emigrating RSI juvenile cutthroat trout (1998-2002; 358 fish) is approximately 43% of the historic average. Adult cutthroat trout escapement into Young Creek reached a peak within a decade after reservoir construction, and has averaged 194 adult cutthroat trout since 1970 (Table 3). Adult cutthroat trout escapement since 1998 has averaged only 13.4 fish per year.

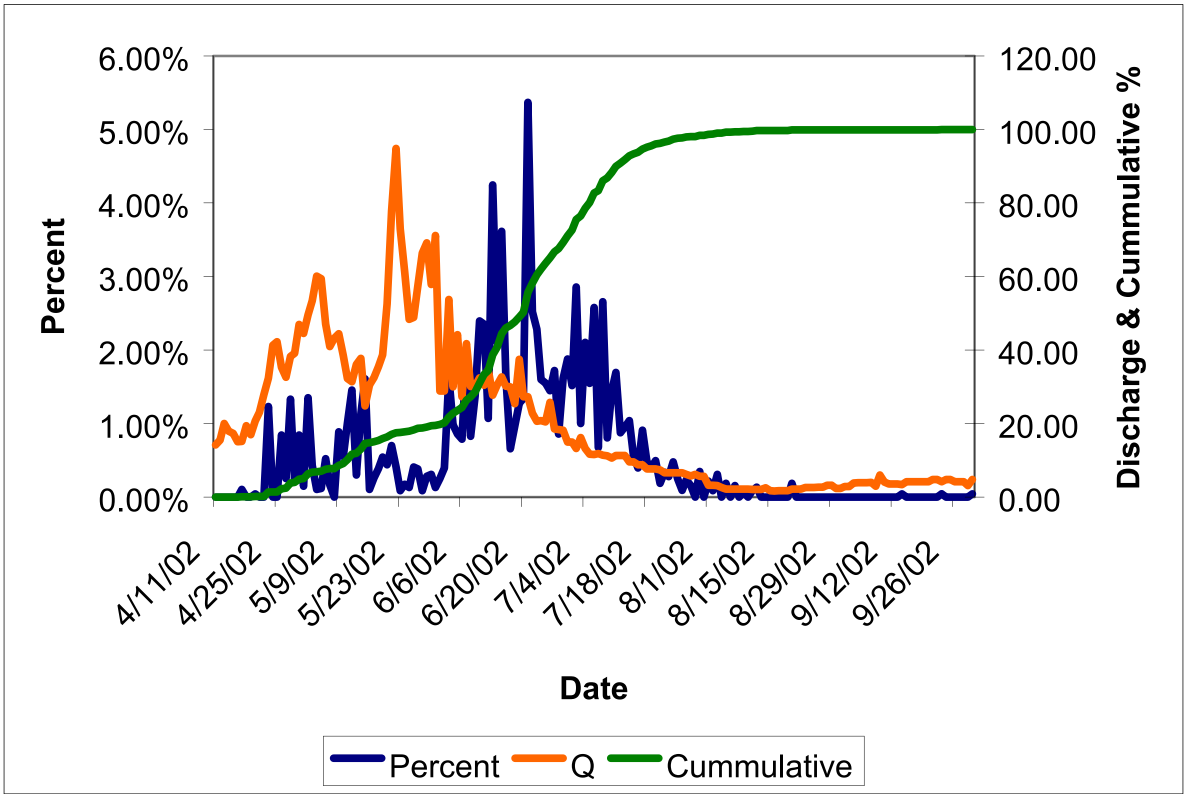

Figure 1. Mean daily and cumulative percent (primary Y axis) catch of westslope cutthroat trout < 250 mm total length at the Young Creek juvenile trap 1998-2002. Mean daily Young Creek flow (Q) for years 1998, 2000, and 2001 (the secondary Y axis). Daily flow records for 2002 were not available.

Table 3. Total catch of adult (> 250 mm total length [TL]) and juvenile (< 250 mm TL) westslope cutthroat trout (WCT) captured in the Young Creek trap from 1970 to 2002. Also presented are the total number of adult and juvenile westslope cutthroat trout otoliths collected in order to determine hatchery (RSI) and natural origin recruits. Catch numbers have not been adjusted for trap efficiency. Years that are not listed represent years in which the trap was not operated.					
Year	WCT Adults (> 250 mm TL)	WCT Juveniles (< 250 mm TL)	Trap Efficiency (%)	WCT Adult Otoliths Collected (>250 mm TL)	WCT Juvenile Otoliths Collected (<250 mm TL)
1970	21	498			
1971	54	161			
1972	8	352			
1973	115	1408			
1974	305	1558			
1975	390	1341			
1976	750	1850			
1977	750	N/A			
1979	345	N/A			
1980	380	1850			
1983	260	1321			
1984	318	962			
1985	82	1274			
1986	83	1629			
1987	55	451			
1988	14	118			
1991	17	176			
1998	4	457	39.0	0	22
1999	6	639	55.2	0	24
2000	0	191	25.5	0	41
2001	44	454	74.8	10	50
2002	5	48	45.7	3	13
Mean	**194**	**897**	**48.0**	**2.6**	**30**

Peak discharge usually occurs in Young Creek during mid/late May to early June (Figure 1). Although catch in the Young Creek juvenile trap is usually low during peak flow periods (Figure 1), we can not determine if passage is truly low during these periods or if our data merely reflect low trap efficiency during periods of excessive discharge. We cannot operate the trap during stream discharges of in excess of 80 cfs. Future trapping efforts at the Young Creek trap will evaluate the efficacy of estimating trap efficiency during varying flow conditions within a year.

The length frequency distribution of all emigrating cutthroat trout at the Young Creek downstream trap for years 1998 – 2002 has been very similar (Figure 2). Although age analyses based on scale and otolith samples for this period are still ongoing, the length frequency information (Figure 2) and age data prior to 1998 (Montana FWP, unpublished data) suggests that most migrants are two and three year old juveniles. The mean total lengths of all cutthroat trout emigrants captured in the Young Creek trap in 1998 – 2002 were 164.9, 160.9, 168.3, 160.3, and 142.1 mm, respectively. Although the length frequency distributions and the mean lengths between years were similar, an analysis of variance and subsequent Fisher's Least Significant Difference (Zar 1996) test did determine significant differences in the mean total length between years. All potential comparisons between years were significantly different except the 1998/2000 and 1999/2001 comparisons (p=0.05).

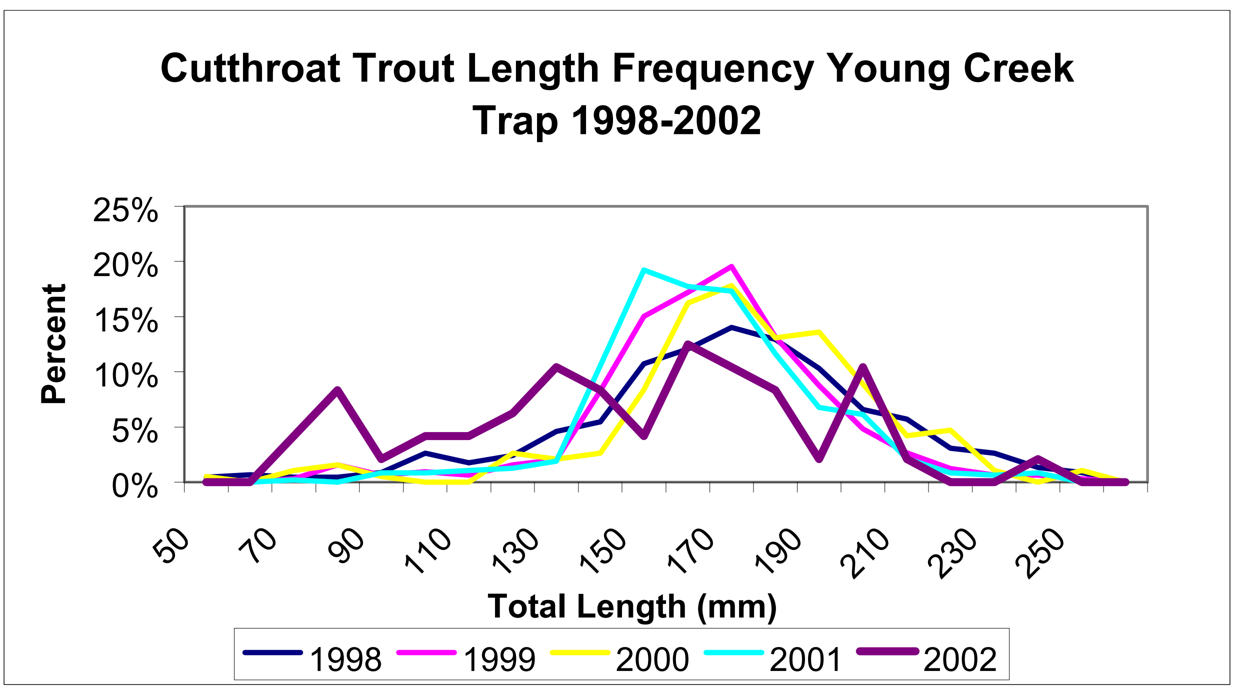

Figure 2. Length frequency distributions for juvenile westslope cutthroat trout < 250 mm total length at the Young Creek trap 1998-2002.

Discussion

Attempts to thermally mark otoliths of the cutthroat trout produced in the RSIs from 1997-2000 were largely unsuccessful, except for the 1999 brood year. Given the relatively poor results from the thermal marking attempts, would likely only be able to identify approximately one quarter of the juvenile and adult cutthroat trout produced from the RSIs using these techniques, and would likely not produce very meaningful results to this study using the thermal marking technology. We attribute the low success rate of the thermal marks mostly to inconsistent methodology used to apply the thermal marks, especially for the 1997 and 1998 brood years. The lack of reference collections from the 2000 brood year limits our ability to differentiate between hatchery and wild origin fish within this cohort.

Recently developed technologies that rely on differences in the ratios of trace elements between natal rearing waters may provide an opportunity to discriminate between hatchery and natural origin juvenile and adult cutthroat trout collected at the Young Creek fish trap. In 2001, the Hungry Horse Mitigation Program (BPA Project Number 199101903) began to develop and test a non-lethal technique to determine stock origin and life history of native migratory bull trout and westslope cutthroat trout populations inhabiting the Flathead River drainage upstream of Flathead Lake. Trace elements in scales from juvenile WCT rearing in natal tributary streams were quantified and correlated with each stream. This non-lethal technique examines specific parts of individual scales within limits of detection less than 100 mg/g and requires a suite of elemental analyses (i.e. Sr, Mg, Ca, Ba, Mn and specific isotopes of Sr) to establish baseline signatures for different streams. This technique may be the most effective method to differentiate trace element signatures in stream-dwelling salmonid populations due to the relatively large differences in geomorphology and lack of mixing between stream systems. While no studies have focused on resident salmonid populations in the Pacific Northwest, this technique has been successful in identifying trace element signatures in scales from juvenile weakfish in estuarine locations along the Atlantic coast (Wells et al. 2000). The study in the Flathead Basin used laser ablation inductively coupled mass spectrometry to quantify Mg:Ca, Mn:Ca, Sr:Ca, and Ba:Ca levels in scales from juvenile westslope cutthroat trout collected from streams of the North Fork, Middle Fork, and South Fork Flathead River during the summers of 2001 and 2002. The study also determined Mg:Ca, Mn:Ca, Sr:Ca, and Ba:Ca levels in the water throughout the study streams, and also found that the chemical compositions of trout scales were strongly correlated with Sr:Ca and Ba:Ca levels in the water. Statistical analyses were able to correctly classify individual fish back to their natal stream with an overall classification accuracy of 91% in the North Fork, 98% in the Middle Fork, and 78% in the South Fork.

Similar techniques may be feasible using the otolith samples collected from juvenile and adult cutthroat trout collected at the Young Creek fish trap, and could differentiate between hatchery and natural origin cutthroat trout. The otoliths collected at the Young Creek trap were preserved, cataloged and stored at the Libby Field Station, and are available for analyses. However, the first step required to investigate the feasibility of using these techniques is to collect and analyze water samples from the Anaconda Hatchery and Young

Creek to test for differences in trace element composition between sites. It seems reasonable to suspect that trace elemental ratios differ between the two locals given geographic distances and parent geology types that differ between the two sites. If water chemistry differences exist between the two sites, the results could be applied to known samples of hatchery and natural origin cutthroat trout otoliths to determine if the technique could correctly classify the samples. Future annual reports will provide updates on the progress of this work.

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Chapter 4

Kootenai River Fisheries Monitoring Results From the Spill Events at Libby Dam, June-July 2002

Abstract

Spill at Libby Dam has been an infrequent event since the fourth turbine unit went online in 1976. As a result of infrequent spill, subsequent information regarding the gas exchange processes, particularly dissolved gas production from spill releases and dissolved gas dissipation downstream from the project are limited. Additional knowledge related to gas production dynamics in the Kootenai River below Libby Dam could help water managers make critical decisions during events that require spill. Therefore the U.S. Army Corps of Engineers proposed to conduct a comprehensive test of total dissolved gas resulting from a range of releases at Libby Dam during June 2002 that were designed to systematically vary the spillway flow over time while monitoring downstream water quality and fish health and behavioral response. However, warm weather and high inflows into a nearly full reservoir required forced spill at Libby Dam beginning on June 25 and lasting 13 days until July 7, and then commencing again for another 7 days from July 11 to July 17. Fish monitoring during the spill activities at Libby Dam in the summer of 2002 used three general approaches including the examination of captive fish and fish captured via electrofishing for signs of gas bubble disease, and radio telemetry to assess fish displacement and behavior changes. Signs of gas bubble disease developed rapidly in the captive fish, and quickly escalated to 100% incidence, relative to fish captured via nighttime electrofishing. Approximately 86% of the rainbow trout *Oncorhynchus mykiss*, 80% of the bull trout *Salvelinus confluentus* and 31% of the mountain whitefish *Prosopium williamsoni* collected during the peak total discharge and spill at Libby Dam exhibited signs of gas bubble disease. We developed 2 indices of exposure to saturated water that used total volume of spill water and the proportion of spill water to correlate with observations of gas bubble disease. Results from the radio telemetry work suggests that most radio tagged rainbow trout (n= 7; 100%), bull trout (n = 3; 75%) and mountain whitefish (n = 2; 67%) did not move substantially during the spill activities at Libby Dam, and remained within the general vicinity of Libby Dam (RM 221.7) downstream to Dunn Creek (RM 219.8), with the center of gravity more near Libby Dam. Spill activities at Libby Dam during the summer of 2002 created relatively rapid response of total dissolved gas concentrations with relatively small amounts of spill water, and impacted resident fish of the Kootenai River below the dam. Therefore, the use of spill as a regular management activity at Libby Dam appears to have limited practical application under the current dam configuration.

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Introduction

Spilling water at hydroelectric projects can cause supersaturated gas conditions in waters downstream. Water and air become mixed when water passes over the spillway, and can be carried to substantial depths in the plunge basin where hydrostatic pressure increases the solubility of the atmospheric gases. The air can then pass into solution in sufficient quantities to promote supersaturated conditions with respect to surface or atmospheric pressure. These conditions can cause gas bubble disease in aquatic organisms. Bouck (1980) defines gas bubble disease as "a noninfectious, physically induced process caused by uncompensated, hyperbaric total dissolved gas pressure, which produces primary lesions in blood (emboli) and in tissues (emphysema) and subsequent physiological dysfunctions. Emboli and gas bubbles can form only when the sum of the dissolved gas pressures or cavitation pressure exceeds the sum of the hydrostatic and other compensating pressures." Workers first reported supersaturation associated with hydroelectric projects in Sweden in the 1940s and 1950s (Jarnefelt 1948 and Lindroth 1957, respectively). The problem was also well documented on the Columbia and Snake rivers during the 1960's (Ebel 1969).

During the construction phase at Libby Dam, operators exclusively used the sluiceways and/or spillway to pass water beginning in March 1972 until August 1975 when the first turbine unit went online. The fourth turbine unit at Libby Dam went online in March 1976. River operations during this period caused supersaturated conditions in the Kootenai River below the dam that adversely impacted mountain whitefish (*Prosopium williamsoni*) and westslope cutthroat trout (*Oncorhynchus clarkii*), the two dominant game fish species in the Kootenai River at that time (May and Huston 1976; May and Huston 1973; May 1973). Since then, sluiceways and spillway have been infrequent methods of passing water at Libby Dam, and subsequent information regarding the gas exchange processes, particularly dissolved gas production from spill releases and dissolved gas dissipation downstream from the project are limited. Additional knowledge related to gas production dynamics in the Kootenai River below Libby Dam could help water managers make critical decisions during 3 potential future events. The 2000 U.S. Fish and Wildlife Service (USFWS) Biological Opinion on the operation of the Federal Columbia River Power System, which includes Libby Dam, calls for spill at Libby Dam to augment powerhouse discharges to benefit sturgeon in the Kootenai River (USFWS 2000). An alternative flood control operation called VARQ, could slightly increase the probability of involuntary spill at Libby Dam (ACOE 2002). Extraordinarily rare flow conditions or discharge forecasting errors may require spill. Therefore the U.S. Army Corps of Engineers proposed to conduct a comprehensive test of total dissolved gas (TDG) resulting from a range of releases at Libby Dam during June 2002. The spill test schedule was designed to systematically vary the spillway flow over time while monitoring downstream water quality and fish. The study was a cooperative effort with the U.S. Army Corps of Engineers as the lead agency responsible for operations and gas monitoring within the river and Montana Fish, Wildlife and Parks as the lead agency responsible for fish monitoring.

May and Huston (1976) concluded that during the construction period of Libby Dam, game fish populations 17 or more miles below Libby Dam were not substantially impacted by supersaturated conditions in the Kootenai River. In fact, it seems logical to conclude that the greatest potential for supersaturated waters from Libby Dam to impact aquatic life would occur directly below the dam. This is especially true given that the spillway is located on the left (east) bank and the turbines are located on the right (west) bank and that an unknown time interval and travel distance would be required for spill and turbine waters to be fully mixed. Montana Fish, Wildlife and Parks was especially concerned about the tailrace area because of the unique fishery that exists there. The lower 3 miles of river directly downstream of Libby Dam supports a unique abundance of world-class rainbow trout (*Oncorhynchus mykiss*) and bull trout (*Salvelinus confluentus*), that likely congregate in this location due to the abundant rich food source of kokanee salmon (*Oncorhynchus nerka*) that are entrained through Libby Dam.

Gas bubble disease (GBD) can cause a variety of signs and lesions, and identification of the disease requires familiarity with the symptoms and careful examination of fish. Exopthalmia or pop eye is a common outward symptom of GBD. However, the absence or presence of exophthalmia alone cannot be considered conclusive evidence of either the presence or absence of the disease, respectively since not all fish ailing from GBD exhibit this symptom and exophthalmia can result from other diseases or infections (Weitkamp and Katz 1980). A much more common symptom of GBD is the presence of bubbles or blisters under the skin, frequently present between fin rays, on the head and in the lining of the mouth (Marsh and Gorham 1905; Weitkamp 1974; 1976; Dawley et al. 1976). Bubbles are most frequent on the unpaired fins, but may occur on the paired fins, head, jaws, and mouth, generally after the appearance of bubbles in the unpaired fins first (Weitkamp 1976). The appearance of gas emboli along the lateral line is generally the first external symptom of the disease to appear in juvenile salmonids, but these bubbles are usually small and difficult to observe, which accounts for the absence in most descriptions of the disease (Schiewe and Weber 1976; Weber and Schiewe 1976). Scattered bubbles (covering less than 15%) along the lateral line have also been observed in fish not exposed to supersaturated water (Dawley et al. 1976). Therefore, this symptom should not be considered an indication of GBD unless extensive portions of the lateral line contain gas emboli (Weitkamp and Katz 1980). Hemorrhages at the base of the paired fins are a frequent sign of the disease in cases of chronic exposures (Meekin and Turner 1974). Fish with gas bubble disease have also been shown to exhibit abnormal behavior including loss of equilibrium (Marsh and Gorham 1905; Wyatt and Beiningen 1971), inability to maintain position in current and avoid obstacles (Wyatt and Beiningen 1971), and reduced growth with chronic exposure (Dawley and Ebel 1975; Meekin and Turner 1974). The most pertinent and conclusive physical external sign of GBD is probably the presence of gas emboli in the gill blood vessels (Dawley et al. 1976; Wyatt and Beiningen 1971; Weitkamp and Katz 1980). Fish mortality due to gas bubble disease is generally attributed to anoxia resulting from stasis in the blood. Sufficient quantities of gas in the circulatory system can lead to accumulations of gas in the heart, preventing blood movement through the vascular system (Marsh and Gorham 1905). Lesser quantities of gas in the circulatory system can result in emboli only in the gills, that can cause blood stasis in the gill arterioles, causing death (Woodbury 1941; Renfro 1963; Dawley et al. 1976). Stroud et al. (1975) also noted that other sublethal effects such as blindness, stress, and diminished lateral line sensitivity can lead to death through secondary causes such as increased vulnerability to predation. Gas bubble disease can also increase the susceptibility to other diseases, such as secondary fungal, and bacterial infections (Weitkamp 1976).

The tolerance to supersaturated water varies by fish life stage. Fish eggs are perhaps the most tolerant life stage to supersaturated water (Rucker and Kangas 1974; Meekin and Turner 1974). A review of supersaturation tolerance by life stage by Weitkamp and Katz (1980) concluded that tolerance by life stage follows two consecutive trends. Tolerance to supersaturation in the early life stages decreases from high tolerance in the egg stage to very low tolerance in the early juvenile life stages. The tolerance of post juvenile life stages tends to increase, with adults being generally most tolerant of supersaturation.

Fish can recover from gas bubble disease. Several authors have found that after a recovery period of 2 weeks in water at equilibrium, fish no longer exhibited external signs of the disease (Dawley and Ebel 1975; Dawley et al. 1976; Meekin and Turner 1974). Recovery can be promoted using equilibrated water, hydrostatic pressure, or artificially produced pressure. Wietkamp (1976) used increased depth to recover juvenile chinook salmon (*Oncorhynchus tshawytscha*) exposed to 118-126% total gas pressure for 10 to 20 days. He observed about 10% mortality, and concluded that most fish that died had developed secondary fungal infections. Meekin and Turner (1974) reported a similar mortality rate of juvenile chinook salmon during recovery. Seven of 67 (10.5%) fish died within the first 24 hours after being placed in equilibrated water. Secondary infections were not identified as an issue related to mortality in this study.

Fish can escape the effects of supersaturated water by either avoiding it, if the choice exists, or by sounding to compensate for supersaturated conditions at surface pressures. However, Weitkamp and Katz (1980) report that it is generally accepted that fish are not able to detect supersaturated conditions and avoid them. A study by Ebel (1971) supports this statement. He found that juvenile chinook salmon held in volitional 0-4.5 m deep cages suffered higher mortality than fish forced to remain in deep (3-4 m) cages. Ebel (1971) concluded that these fish were unable to detect or not willing to avoid saturated water. However, several studies contradict this generalization and suggest that the ability to detect and avoid saturated water may be species specific. Blahm et al. (1976) found that juvenile chinook salmon were able to detect and avoid supersaturated water when given a choice, but that juvenile steelhead (*Oncorhynchus mykiss*) were not able to detect supersaturation. Dawley et al. (1976) concluded that both juvenile steelhead and chinook salmon were able to detect and avoid supersaturated water by sounding. Meekin and Turner (1974) found that juvenile chinook salmon were able to detect and avoid supersaturated water when given the choice, but that steelhead were not. However, temperature differences during this study limit its inferential power. Bentley et al. (1976) also demonstrated that northern pikeminnow (*Ptychocheilus oregonensis*) may be able to detect and avoid supersaturated conditions given the opportunity.

Nevertheless, Weitkamp and Katz (1980) concluded that insufficient information exists to conclude whether or not fish are able to detect and avoid supersaturated water.

Methods

The primary objective for conducting the spill test in June 2002 was to increase the understanding of gas exchange processes within the Kootenai River during spill operations, particularly dissolved gas production, mixing and dissolved gas dissipation downstream from Libby Dam. However, another important objective was to ensure that aquatic life was not harmed during the collection of these data. Montana Fish, Wildlife and Parks was the lead agency in a cooperative effort to monitor fish in the Kootenai River for signs of gas bubble disease during the scheduled spill test. Fish monitoring during the spill activities at Libby Dam in the summer of 2002 used three general approaches. Sentry fish were held in cages and checked for signs of gas bubble disease, fish were captured using electrofishing gear and examined for signs of gas bubble disease, and finally radio telemetry was used to investigate fish movement or displacement during spill activities. In order to ensure that supersaturation did not impact fish in the Kootenai River three threshold criteria were established that would stop the spill test at Libby Dam. Spill was to be suspended if any of the three criteria were realized. A real time TDG monitoring station was established approximately one mile downstream of Libby Dam. Spill was to be terminated under two criteria established for this monitoring station. The criteria were exceeding either a three-hour average of TDG saturation of 120%, or an hourly average of TDG saturation exceeded 125%. The final criterion was the identification of signs of gas bubble disease in either the captive fish or fish captured via electrofishing. Estimating fish mortality was not an intended objective because the criteria for stopping spill activities during the test period were considered conservative to the point that spill would be stopped before mortality was prevalent.

Captive Fish

Captive fish were held in hoop traps at three locations on the left bank approximately 0.4, 0.8 and 1.7 miles below Libby Dam (sites 1-3, respectively; Figure 1) throughout the spill duration. A total of 9 spill events were scheduled for the 3-day spill test (Table 1). However, warm weather and high inflows into a nearly full reservoir required forced spill at Libby Dam beginning on June 25 and lasting 13 days until July 7, and then commencing again for another 7 days from July 11 to July 17.

Three hoop traps measuring 2 foot diameter, approximately 6-8 feet in length with ¾ inch net mesh (Figure 2) were located at each of the three sites (9 total hoop nets) in 3-6 feet of water. Large weights attached to each hoop trap prevented downstream movement. The protocol had intended to stock each hoop trap with 5 mountain whitefish and 5 rainbow trout. However, species composition within individual hoop nets varied depending upon the electrofishing catch. Fish were captured using nighttime electrofishing by jetsled using a Coffelt model Mark 22 electrofishing unit, operating with an electrical output ranging from 200-350 volts at 5-8 amps. Captured fish were examined for external signs of gas bubble disease prior to being placed in hoop nets. Captive fish were stocked in the hoop nets on three separate occasions during the spill event (Table 2), because investigators were concerned that handling mortality may be substantial and could potentially confound the results. Captive fish from one hoop net at each site were examined between each spill event (Table 2) up until forced spill occurred. Monitoring intensity of captive fish was reduced once spill operations shifted from test conditions to forced spill (Table 2). During each examination period, fish were removed from the hoop nets, anesthetized using an aqueous non-buffered solution of MS-222, and then externally examined for signs of gas bubble disease. Investigators examined the fins, eyes, and head using either an ophthalmoscope manufactured by Welch Allyn or a hand held loup 5X magnifying glass for the presence of gas emboli, and then recorded the total proportion of each fin or anatomical feature that contained emboli.

Because we replaced the captive fish in the hoop nets throughout the duration of the spill activities, it was necessary to attempt to quantify the amount (dose) of potentially saturated water fish were exposed to, and relate that exposure to the presence of signs of gas bubble disease observed. We derived two indices of exposure time to correlate to signs of gas bubble disease. The first index was cumulative hourly spill discharge (CSpill) a particular group of fish was exposed to, and was calculated using the following equation.

$$CSpill_j = \sum_i HSD$$

Where $Cspill_j$ = The cumulative hourly spill discharge for fish group j at time of examination, and HSD (Hourly Spill Discharge) = the sum of i hourly spill discharge measurements (kcfs) that fish group j was exposed to until examination. For example, if a fish were exposed to 5 kcfs spill for 10 hours, the cumulative hourly spill discharge would be 50. The second index of exposure (CSpWtd) was similar to the previous index, but differed in that it utilized a weighting factor based on the proportion of the spill discharge relative to total discharge. We calculated cumulative spill weighted discharge ($CSpWtd_j$) for fish group j using the following equation.

$$CSpWtd_j = \sum_i (HSD) * (\frac{HSD}{TD})$$

Where HSD (Hourly Spill Discharge) is the hourly spill (kcfs), and TD is the total discharge at Libby Dam (kcfs) for the i^{th} hourly periods. For example, if a fish were exposed to 5,000 cfs spill with at a total discharge of 10,000 cfs for 10 hours, the cumulative spill weighted discharge would be 25. We used nonlinear regression to correlate our two indices of captive fish saturated water exposure to fish response. We used the proportion (percent) of fish in an individual hoop net that were identified with signs of gas bubble disease as the response variable in the nonlinear regression. Individual regression analyses were completed using the proportion of mountain whitefish, rainbow trout and all fish species pooled, exhibiting signs of gas bubble disease as the response variable. We used r^2 values from competing models to determine which model provided the best fit to the data.

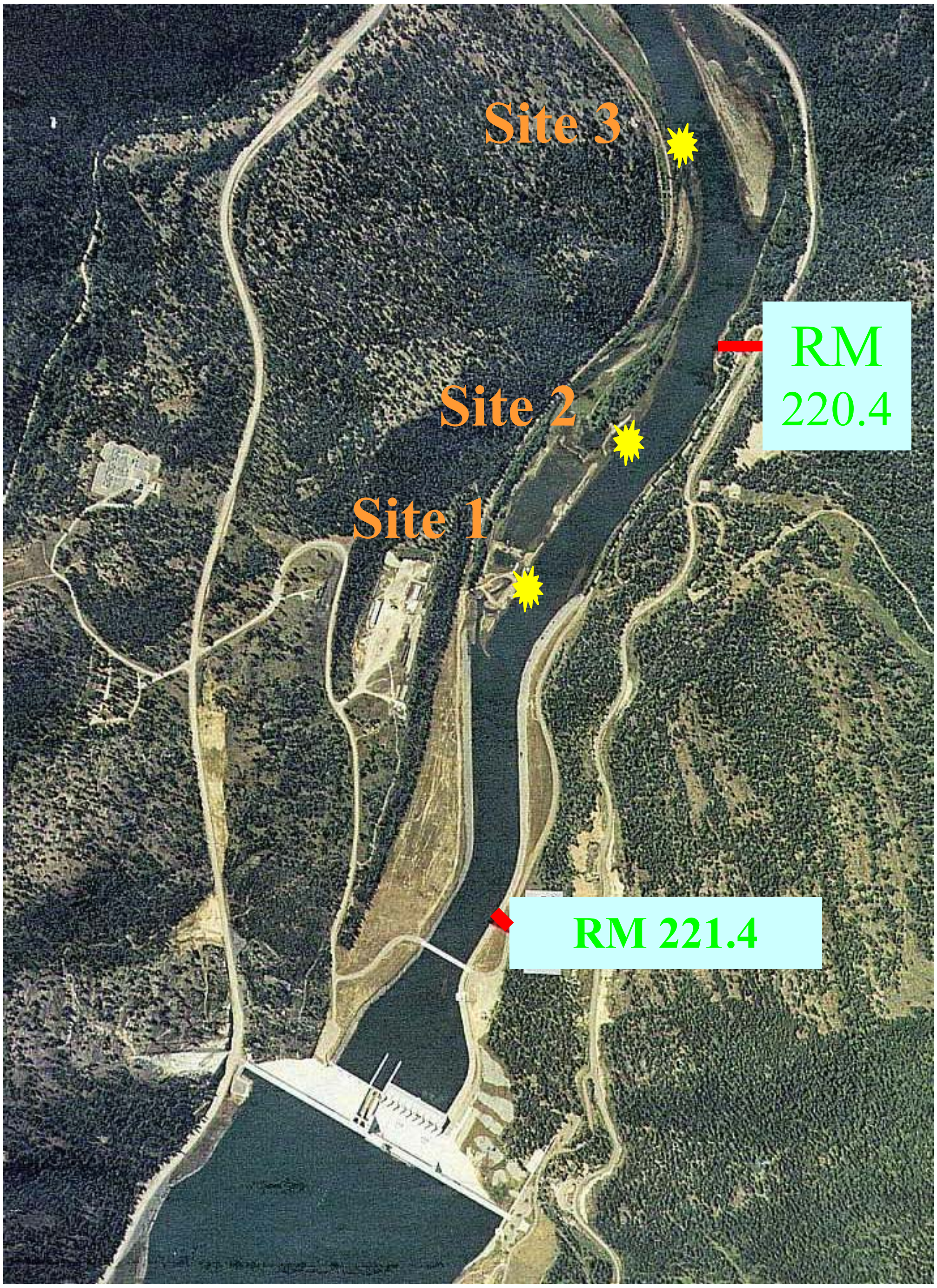


Figure 1. Aerial photograph of Libby Dam, looking downstream. The three locations marked with yellow symbols on the photograph represent the approximate location of three hoop traps used to hold captive fish during the spill activities. River mile (RM) locations are shown for reference. Thee hoop traps were located at each location at depths ranging from 3-6 feet.

Figure 2. Investigators, Brian Marotz, Monty Benner (Montana FWP), Pat Dwyer (consultant), and Evan Lewis (USACOE) checking mountain whitefish and rainbow trout held in a hoop trap during spill activities at Libby Dam.

Table 1. Scheduled spill events, duration, and powerhouse, spill and total flows at Libby Dam.						
Event	**Date**	**Time**	**Number Hours**	**Powerhouse Flow (Kcfs)**	**Spill Flow (Kcfs)**	**Total Flow (Kcfs)**
1	6/25	0700-1030	3.5	23	2	25
2	6/25	1100-1430	3.5	22	3	25
3	6/25	1500-1830	3.5	21	4	25
	6/25-26	1830-0700		25	0	25
4	6/26	0900-1230	3.5	20	5	25
5	6/26	1300-1630	3.5	19	6	25
6	6/26	1700-2030	3.5	18	7	25
	6/26-27	2030-0900		25	0	25
7	6/27	0900-1230	3.5	17	8	25
8	6/27	1300-1630	3.5	16	9	25
9	6/27	1700-2030	3.5	15	10	25

Table 2. Date and times that hoop traps were stocked with fish (S), fish examined for signs of gas bubble disease (E), and examined for signs of gas bubble disease and released (ER) during the spill event at Libby Dam in June and July 2002. Fish were held in three hoop traps on the left bank at three sites approximately 0.4, 0.8 and 1.7 miles below Libby Dam (sites 1-3 respectively).

	Site 1			***Site 2***			**Site 3**		
Date	**Trap 1**	**Trap 2**	**Trap 3**	**Trap 1**	**Trap 2**	**Trap 3**	**Trap 1**	**Trap 2**	**Trap 3**
6/24	S (23:00)	S (23:00)	S (23:00)	S (23:00)	S (23:00)	S (23:00)	S (23:00)	S (23:00)	S (23:00)
6/25	E (9:35) E (17:45) ER (23:25)	E (13:45) ER (22:30)		E (9:53) E (18:00) ER (22:40)	E (14:00) ER (22:45)		E (10:15) E (18:15) ER (22:55)	E (14:15) ER (23:00)	
6/26	S (00:30) E (10:05) E (18:35)	S (00:30) E (15:35)		S (00:30) E (10:22) E (19:00)	S (00:30) E (15:45)		S (00:30) E (10:41) E (19:15)	S (00:30) E (15:55)	
6/27	ER (23:21)	ER (23:40)	ER (24:00)	ER (22:26)	ER (22:43)	ER (23:00)	ER (21:12)	ER (21:29)	ER (21:58)
6/28	S (00:20)	S (00:22)	S (00:25)				S (00:50)	S (00:55)	S (01:00)
7/1	ER (23:50)						ER (23:00)		
7/3		ER (15:20)						ER (14:54)	
7/8			ER (14:00)						ER (14:20)

Electrofishing

We used electrofishing to capture free-swimming fish below Libby Dam for examination of signs of gas bubble disease in the Kootenai River. Fish were captured using nighttime electrofishing by jetsled using a Coffelt model Mark 22 electrofishing unit, operating with an electrical output ranging from 200-350 volts at 5-8 amps. Sampling occurred on evenings of June 25, July 1, July 8, and July 24 from directly below Libby Dam (river mile; RM 221.7) downstream to the confluence of Alexander Creek (RM 220.5), and was generally restricted to the left bank, with the exception of sampling on July 1, when both left and right banks were sampled and reported separately. Electrofishing was generally confined to the left bank because the spill water had not thoroughly mixed across the river channel at this location, in order to maximize our ability to detect symptoms of GBD. We attempted to net all salmonids encountered during electrofishing. We anesthetized captured fish using an aqueous non-buffered solution of MS-222, and then externally examined each fish for signs of gas bubble disease. Investigators examined the fins, eyes, and head using either an ophthalmoscope manufactured by Welch Allyn or a hand held loup 5X magnifying glass for the presence of gas emboli, and then recorded the total proportion of each fin or anatomical feature that contained emboli. Fish were held in vessels circulating with fresh river water and then released on the left bank at RM 221.0 once fully recovered.

We used nonlinear regression to quantify the amount (dose) of potentially saturated water fish were exposed to, and related that exposure to the presence of signs of gas bubble disease observed in the electrofishing catch. We used the same two indices of exposure time used for captive fish, cumulative hourly spill discharge (CSpill), and cumulative spill weighted discharge ($CSpWtd_j$). We used the proportion (percent) of fish captured on a particular evening that were identified with signs of gas bubble disease as the response variable in the nonlinear regression. Individual regression analyses were completed using the proportion of rainbow trout, bull trout, mountain whitefish and all fish species pooled exhibiting signs of gas bubble disease as the response variable. We used r^2 values from competing models to determine which model provided the best fit to the data.

Radio Telemetry

We used radio telemetry to assess the movement and behavior of fish below Libby Dam during the spill activities. We used electrofishing to capture and then surgically implant radio tags into 5 bull trout, 8 rainbow trout, and 3 mountain whitefish. Fish were captured via nighttime jetsled electrofishing using a Coffelt model Mark 22 electrofishing unit, operating with an electrical output ranging from 200-350 volts at 5-8 amps on the evening of June 18 (one week prior to spill activities). Collection occurred from directly below Libby Dam (RM 221.7) downstream to the confluence of Alexander Creek (RM 220.5). We examined fish for marks, tags, and injuries, and then we anesthetized captured fish using an aqueous non-buffered solution of MS-222, measured them, and surgically implanted the radio tag. Tagged fish were released in the general vicinity of capture. We used 9.5 g tags manufactured by Advanced Telemetry Systems, Inc. that were powered by a single 3.6 V lithium battery and had a minimum life span of 80 days and a burst rate of 45 pulses per minute. Each transmitter had a 29 cm flexible external whip antenna attached to one end. Each tag transmitted on a unique frequency ranging from 49.105 to 49.811 kHz, allowing individual fish identification. We used telemetry receivers manufactured by Lotek Engineering (Model SRX-400) for mobile monitoring activities. Each mobile monitoring unit consisted of a radio receiver, data processor, internal clock, and a tuned loop antenna. We determined the location of tagged fish using mobile tracking that consisted of a combined effort of fixed wing aircraft and jetsled observations. Field crews conducted observations 3 times per day during the first 2 days of spill activities below Libby Dam and then approximately 2-4 days per week until July 7. Field crews manually recorded the location description of each fish identified. Fish movement and visual observations were used as the primary as indicators of live fish.

Results

As a result of warm weather and high inflows (with a peak in excess of 70,000 cfs) into a nearly full reservoir, the spill test was superseded after the first day as dam operations shifted to flood control. Forced spill continued until July 16 and at substantially higher levels than those that had been intended for the spill test. The planned spill activities are presented in Table 1, and the actual spill events that occurred at Libby Dam are presented in Table 3. If forced spill had not occurred at Libby Dam, the spill test would have been stopped when one of the three criteria established to protect aquatic life in the Kootenai River were met. The 125% one-hour average numeric criterion was exceed for 3 hours on July 1 13:30-16:30 while the 120% three-hour criterion was exceeded from June 26 10:45 to July 6 13:00 as measured near the left bank (spillway side of channel) at the USGS gauging station (Schneider and Carroll 2002). The criterion for observations of gas bubble disease in fish is discussed below.

Captive Fish

Signs of gas bubble disease manifested, as emboli, were first identified in the eyes of rainbow trout and mountain whitefish in the late evening of June 25. Two of the three hoop traps at each of the three sites were examined and released at approximately June 25 22:00 (Table 4). Approximately 5% of the mountain whitefish examined at this time had gas emboli in at least one eye, and 10% of the captive rainbow trout had gas emboli in at least one eye (Table 4). Each of the 6 hoop traps was restocked with fresh fish collected from downriver. These observations on June 25 would have warranted stopping the spill test, but forced spill for flood control continued. The size of the rainbow trout and mountain whitefish used for the captive fish studies were similar (rainbow trout mean total length 268 mm; range 152-406 mm, and the mean total length of mountain whitefish 290.4 mm; range 152-406 mm). The severity of the symptoms of gas bubble disease observed in both rainbow trout and mountain whitefish increased with the duration of exposure to saturated water (Figure 3). Mortality of the captive fish was generally low (less than 10% overall) during the spill activities. Mortalities were only included in estimates of the incidence of signs of gas bubble disease if they had recently died, as evidenced by red gills.

Signs of gas bubble disease developed rapidly in the captive fish, and quickly escalated to 100% incidence, relative to the duration of spill activities at Libby Dam in the summer of 2002 (Figures 4 and 5; Table 4). The nonlinear regression model that provided the best fit utilized cumulative hourly spill weighted discharge as the independent variable and proportion of all fish (mountain whitefish and rainbow trout combined) that exhibited signs of gas bubble disease as the response variable ($r^2 = 0.875$; Figure 4). The nonlinear regression model that used cumulative hourly spill discharge and the proportion of all fish that exhibited signs of gas bubble disease, produced similar results as the previous model, but was the was the third best model ($r^2 = 0.870$; Figure 5). Competing models using cumulative hourly spill weighted discharge and proportion of mountain whitefish that exhibited signs of gas bubble disease, cumulative hourly spill discharge and proportion of mountain whitefish that exhibited signs of gas bubble disease, cumulative hourly spill weighted discharge and proportion of rainbow trout that exhibited signs of gas bubble disease, and cumulative hourly spill discharge and proportion of rainbow trout that exhibited signs of gas bubble disease, all yielded similar results ($r^2 = 0.870$, 0.845, 0.797, and 0.766, respectively).

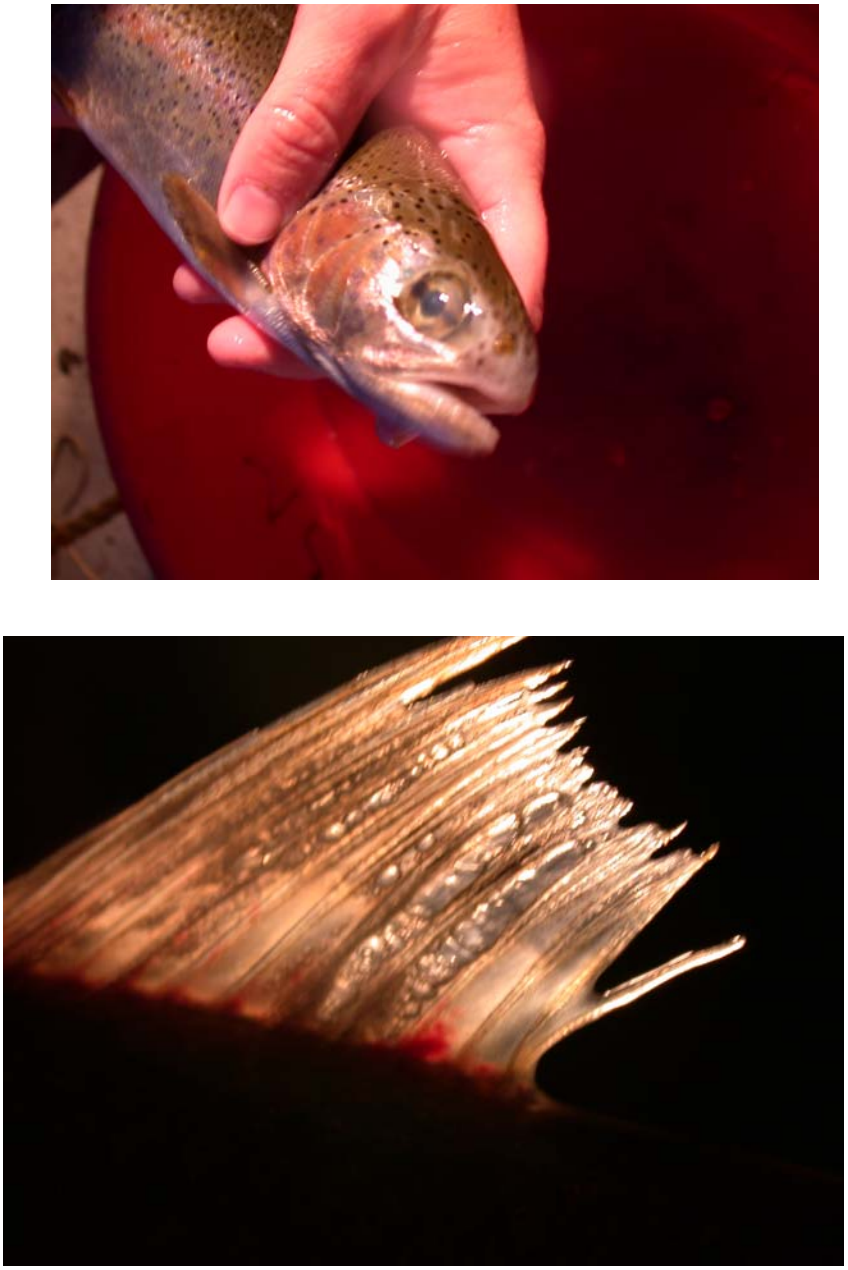


Figure 3. Examples of severe signs of gas bubble disease observed in the eyes and head of a rainbow trout (top photograph) and the dorsal fin of a mountain whitefish (bottom photograph) at the peak of spill activities at Libby Dam on July 1, 2002. Fish in these photographs were captive fish held in hoop traps.

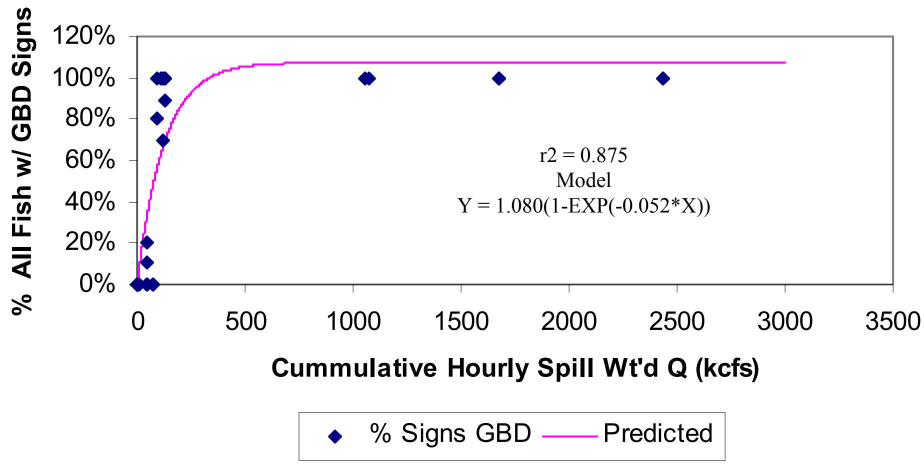


Figure 4. The relation between the cumulative hourly spill weighted flow (Q; kcfs) and the proportion of all captive fish (rainbow trout and mountain whitefish combined) observed exhibiting signs of gas bubble disease (GBD) during spill activities in the Kootenai River below Libby Dam in the summer of 2002.

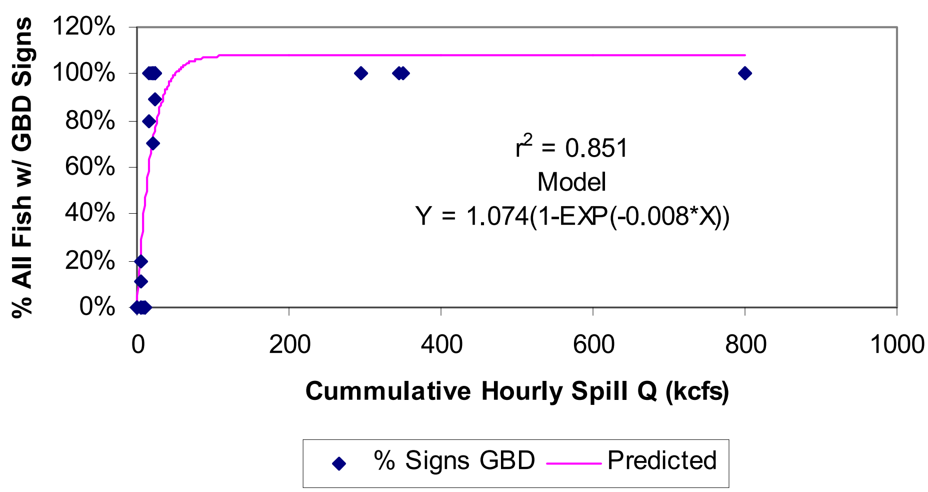


Figure 5. The relation between the cumulative hourly spill flow (Q; kcfs) and the proportion of all captive fish (rainbow trout and mountain whitefish combined) observed exhibiting signs of gas bubble disease (GBD) during spill activities in the Kootenai River below Libby Dam in the summer of 2002.

Table 3. Summary of the spill events at Libby Dam during June and July, 2002 including start and stop date and time, duration (hours), total discharge (thousand cubic feet per second; kcfs), spill discharge, and turbine discharge.						
Event Number	**Starting Date and Time**	**Ending Date and Time**	**Duration (hr:min)**	**Total Discharge (kcfs)**	**Spill Discharge (kcfs)**	**Turbine Discharge (kcfs)**
1	6/25 7:00	6/25 9:45	2:45	23.5	0.7	22.8
2	6/25 11:45	6/25 13:45	2:00	23.8	3.0	20.8
3	6/25 15:00	6/25 17:45	2:45	29.0	6.0	23.0
4	6/25 18:00	6/26 8:45	14:45	29.0	4.0	25.0
5	6/26 9:00	6/26 15:45	6:45	30.0	5.0	25.0
6	6/26 16:00	6/28 7:45	39:45	32.0	7.0	25.0
7	6/28 10:00	6/28 13:45	3:45	32.0	7.4	24.6
8	6/28 14:00	6/28 15:45	1:45	33.0	8.4	24.6
9	6/28 16:00	6/30 10:45	42:45	35.0	10.6	24.4
10	6/30 11:00	6/30 12:45	1:45	36.0	11.6	24.4
11	6/30 13:00	6/30 14:45	1:45	37.0	12.6	24.4
12	6/30 15:00	7/1 11:45	20:45	38.0	13.6	24.4
13	7/1 12:00	7/1 13:45	1:45	39.0	14.6	24.4
14	7/1 14:00	7/3 9:45	43:45	40.0	15.6	24.4
15	7/3 10:00	7/3 12:45	2:45	39.0	14.6	24.4
16	7/3 13:00	7/4 9:45	20:45	38.0	13.6	24.4
17	7/4 10:00	7/4 12:45	2:45	37.0	12.6	24.4
18	7/4 13:00	7/4 15:45	2:45	36.0	11.6	24.4
19	7/4 16:00	7/5 10:45	18:45	35.0	10.6	24.4
20	7/5 11:00	7/5 13:45	2:45	32.5	8.1	24.4
21	7/5 14:00	7/6 11:45	21:45	30.0	5.6	24.4
22	7/6 12:00	7/7 9:45	21:45	28.0	3.6	24.4
23	7/7 10:00	7/7 12:45	2:45	26.0	2.0	24.4
24	7/11 10:00	7/12 11:00	25:00	25.8	2.0	23.8
25	7/12 11:00	7/12 12:00	1:00	24.5	0.7	23.8
26	7/12 12:00	7/14 22:00	58:00	27.0	3.2	23.8
27	7/14 22:00	7/15 17:00	19:00	28.0	4.2	23.8
28	7/15 17:00	7/15 18:00	1:00	29.5	5.7	23.8
29	7/15 18:00	7/16 13:00	19:00	30.0	6.2	23.8
30	7/16 13:00	7/17 11:00	22:00	27.0	3.2	23.8

Table 4. A summary of the results of the examination of captive fish held in hoop traps along the left bank below Libby Dam during spill activities. The first number represents the sample size followed by the percent exhibiting signs of gas bubble disease in parentheses. The locations of hoop trap sites 1-3 are shown in Figure 1.

Date and Time	Site #	Net #	Rainbow Trout	Mountain Whitefish
6/25 9:35	1	1	0	12 (0)
6/25 9:53	2	1	5 (0)	5 (0)
6/25 10:15	3	1	3 (0)	5 (0)
6/25 13:45	1	2	4 (0)	5 (0)
6/25 14:00	2	2	4 (0)	5 (0)
6/25 14:15	3	2	3 (0)	7 (0)
6/25 22:00	2	1	5 (0)	5 (0)
6/25 22:10	1	1	0	12 (0)
6/25 22:20	1	2	3 (0)	5 (0)
6/25 22:30	3	1	5 (10%)	4 (0)
6/25 22:40	3	1	3 (33%)	7 (14%)
6/26 10:00	1	1	2 (0)	7 (0)
6/26 10:22	2	1	3 (0)	6 (0)
6/26 15:30	2	2	1 (0)	7 (0)
6/26 15:35	1	2	2 (0)	5 (0)
6/26 16:00	3	2	3 (0)	5 (0)
6/26 18:35	1	1	2 (100%)	3 (67%)
6/27 21:12	3	1	0 (0)	3 (100%)
6/27 21:12	3	2	3 (100%)	5 (100%)
6/27 22:26	2	1	3 (100%)	4 (100%)
6/27 22:43	2	2	1 (100%)	7 (100%)
6/27 23:00	2	3	4 (25%)	6 (100%)
6/27 23:00	1	1	2 (100%)	7 (100%)
6/27 23:00	1	2	2 (100%)	5 (100%)
6/27 23:50	3	3	2 (100%)	7 (86%)
6/27 23:56	1	3	5 (100%)	5 (100%)
7/1 23:00	3	1	3 (100%)	5 (100%)
7/1 23:00	1	1	0	10 (100%)
7/3 14:54	1	2	0	10 (100%)
7/3 15:15	3	2	0	8 (100%)
7/8 12:00	1	3	0	4 (100%)
7/8 12:30	3	3	0	8 (100%)

Electrofishing

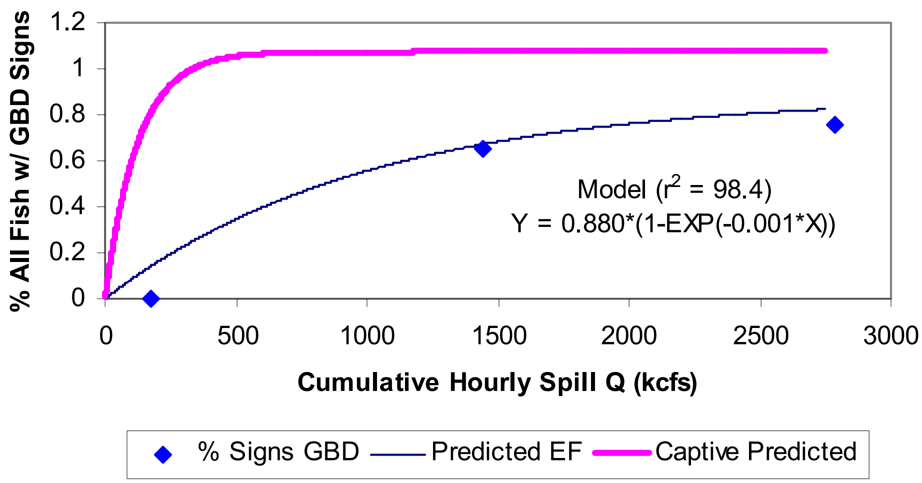
Our electrofishing activities to capture fish for examination of signs of gas bubble disease occurred less frequently than examination of captive fish. The lower frequency limited our ability to estimate the precise time when signs of gas bubble disease first appeared in free-swimming fish in the Kootenai River below Libby Dam. The first electrofishing survey in the Kootenai River below Libby Dam was conducted on June 26. No signs of gas bubble disease were observed in any fish captured via electrofishing on June 26 (Table 5). Signs of gas bubble disease were first identified in the electrofishing catch on the evening of July 1, 2002. At that time signs of gas bubble disease were common in all salmonid species examined (Table 5). Approximately 86% of the rainbow trout, 80% of the bull trout and 31% of the mountain whitefish collected from the electrofishing along the left bank and examined on July 1 exhibited signs of gas bubble disease. Gas emboli in the eyes were the most common sign of gas bubble disease identified by observers. Spill at Libby Dam peaked in terms of volume (kcfs) and the proportion of spill relative to total discharge peaked during this period (Table 3). Spill discharge peaked during spill event number 14 (July 1 15:00 – July 3 9:45) at 15.6 kcfs, and represented approximately 39% of the total discharge passing Libby Dam. We also collected fish from the right bank on the evening of July 1. Although the proportion of rainbow trout, bull trout and mountain whitefish exhibiting signs of gas bubble disease differed between left and right bank (Table 5), these differences were not significant ($P > 0.05$), but the power of the three tests was low (0.46, 0.35, and 0.21 respectively, for $\alpha = 0.05$).

The only survey conducted prior to the July 1st survey was conducted 4 days prior on the evening of June 26. We intentionally maintained a low frequency of electrofishing within the area directly below Libby Dam to minimize impacts to the large salmonids inhabiting this section of the river. The next collection of fish via electrofishing below Libby Dam occurred on July 8. Signs of gas bubble disease were prevalent during the July 8 sampling period also with approximately 67% of the rainbow trout, 71% of the bull trout, and 83% of the mountain whitefish examined exhibiting signs of gas bubble disease (Table 5). Spill activities at the time of collection and examination on July 8 had been suspended for approximately 22 hours after continuous spill activities at Libby Dam lasting 12.2 days ranging from 0.7 – 15.6 kcfs. Our last electrofishing survey was conducted on July 24, approximately 7.5 days after the final spill event. We did not observe gas emboli in any of the 4 salmonid species examined at that time (Table 5). However, a substantial proportion of these fish had split fins. Field crews estimated that approximately 56% of the rainbow trout, 50% of the bull trout and 88% of the mountain whitefish below Libby Dam had at least one split fin on the evening of July 24. We did not estimate the proportion of fish that had split fins on any of the other observation dates, but we recall that this was not a noticeable infliction on prior sampling dates.

Table 5. A summary of the results of nighttime electrofishing surveys below Libby Dam to examine fish species for signs of gas bubble disease. The first number represents the sample size followed by the percent exhibiting signs of gas bubble disease in parentheses.					
Species	**6/26 Left Bank**	**7/1 Left Bank**	**7/1 Right Bank**	**7/8 Left Bank**	**7/24 Left Bank**
Rainbow Trout	6 (0)	14 (86%)	5 (80%)	12 (67%)	16 (0)
Bull Trout	11 (0)	10 (80%)	9 (44%)	7 (71%)	8 (0)
Mountain Whitefish	4 (0)	13 (31%)	3 (67%)	18 (83%)	8 (0)
Suckers (all spp.)	0 (0)	7 (0)	1 (0)	5 (40%)	0 (0)
Kokanee	0 (0)	1 (0)	3 (100%)	10 (100%)	3 (0)
Columbia River Chub (*Mylocheilus caurinus*)	0 (0)	1 (0)	0 (0)	0 (0)	0 (0)
Northern Pikeminnow	0 (0)	0 (0)	3 (0)	0 (0)	0 (0)
Redsided Shiner (*Richardsonius balteatus*)	0 (0)	2 (100%)	0 (0)	0 (0)	0 (0)
Burbot (*Lota lota*)	0 (0)	0 (0)	0 (0)	1 (100%)	0 (0)

Cumulative hourly spill weighted discharge compared to cumulative hourly spill discharge consistently provided a better regression fit to the data sets of the proportion of rainbow trout, bull trout, mountain whitefish, and all species combined exhibiting signs of gas bubble disease. The nonlinear model using cumulative hourly spill weighted discharge as the independent variable and the proportion of all species combined exhibiting signs of gas bubble disease provided the best fit for the fish captured via electrofishing ($r^2 = 0.993$; Figure 6). The nonlinear models for rainbow trout and bull trout were similar to the model using all species pooled when either cumulative hourly spill weighted discharge or cumulative hourly spill discharge was used as the independent variable (Figures 6-8). Linear regression provided a better fit to the mountain whitefish data set than did nonlinear regression (Figure 9).

The rate of fish response to supersaturated water in terms of the proportion of captive fish and fish captured via electrofishing appeared to be substantially different (Figure 6). Differences appeared to include both the rate in which signs of gas bubble disease affected each group and the maximum proportion observed exhibiting symptoms. Fish captured via electrofishing seldom exhibited incidence rates of 100%, even though the condition was common for captive fish (Figure 6). However, statistical analyses were not performed to evaluate whether nonlinear regressions significantly differed between captive fish and fish captured via electrofishing.



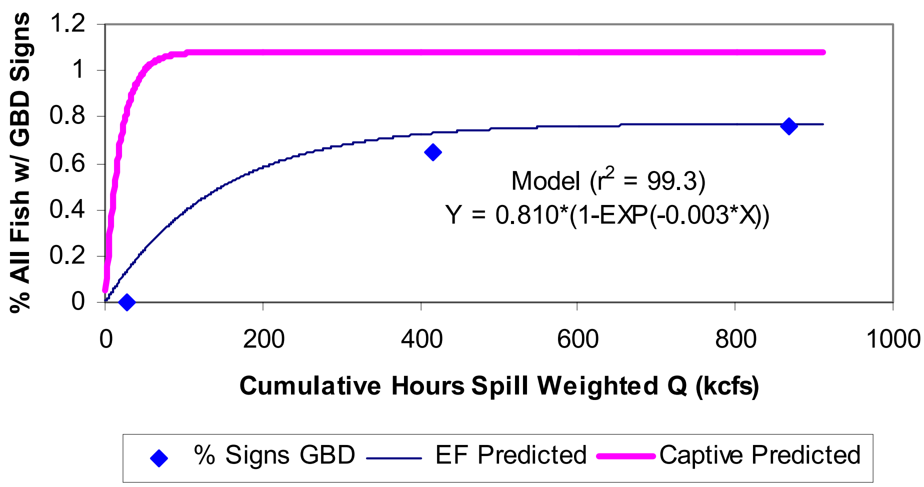
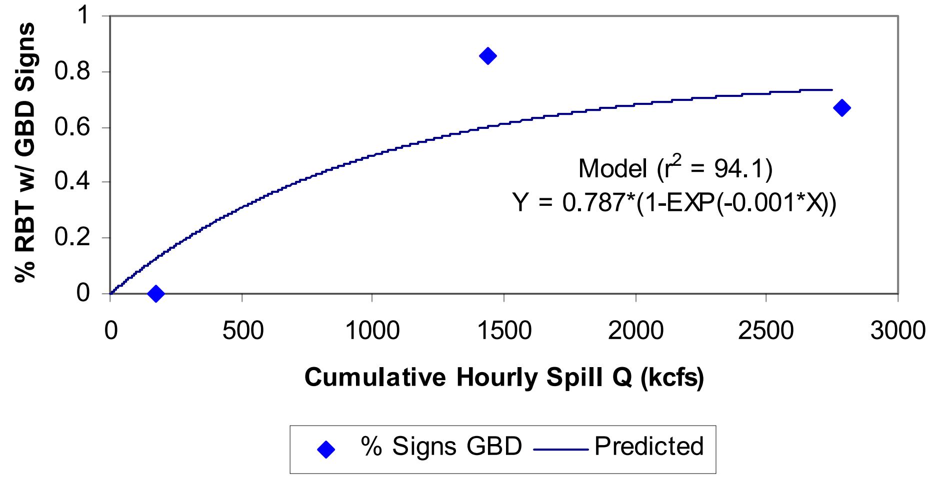


Figure 6. The relation between the cumulative hourly spill flow (top graph), cumulative hourly spill weighted flow (bottom graph), and the proportion of all fish (rainbow trout bull trout, and mountain whitefish combined) captured via electrofishing observed exhibiting signs of gas bubble disease (GBD) during spill activities in the Kootenai River below Libby Dam in the summer of 2002. The blue solid line represents the predicted relationship for fish captured via electrofishing and the pink solid line represents the predicted relationship for captive fish (all species pooled) for comparison. The model and r^2 value describe the relationship for fish captured via electrofishing.



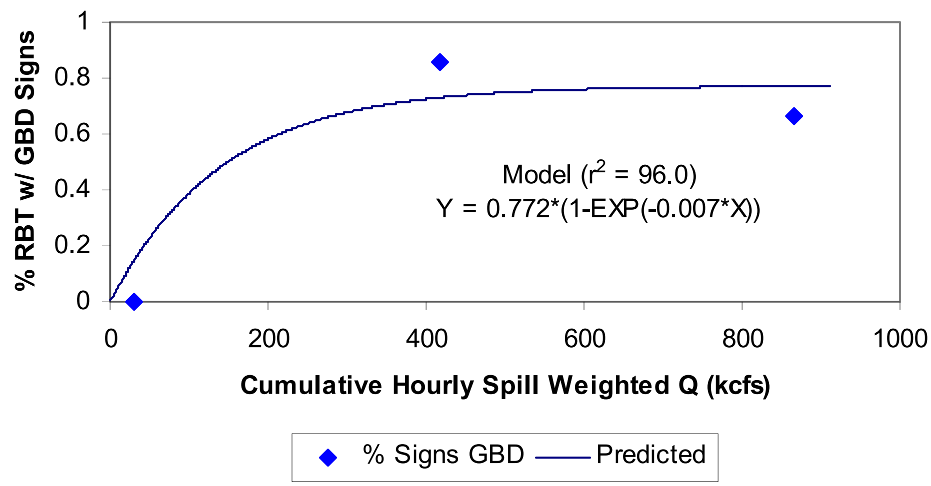
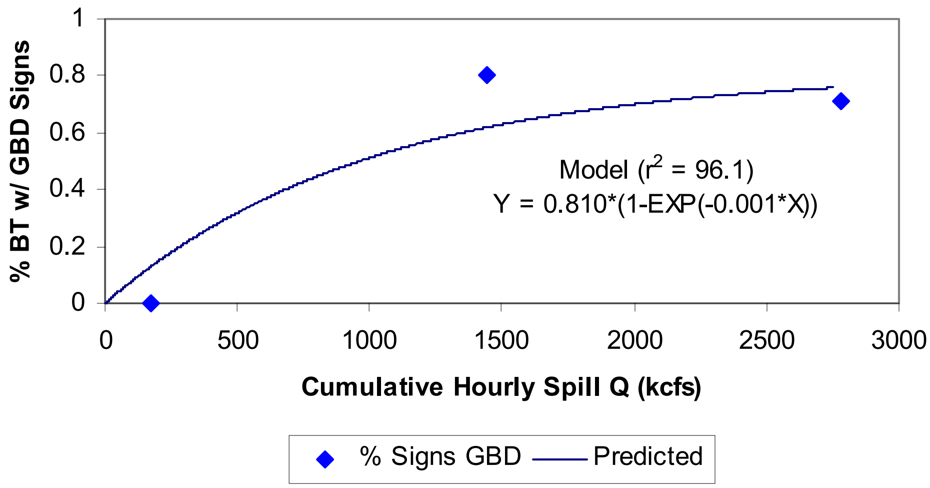


Figure 7. The relation between the cumulative hourly spill flow (top graph), cumulative hourly spill weighted flow (bottom graph), and the proportion of rainbow trout (RBT) captured via electrofishing observed exhibiting signs of gas bubble disease (GBD) during spill activities in the Kootenai River below Libby Dam in the summer of 2002.



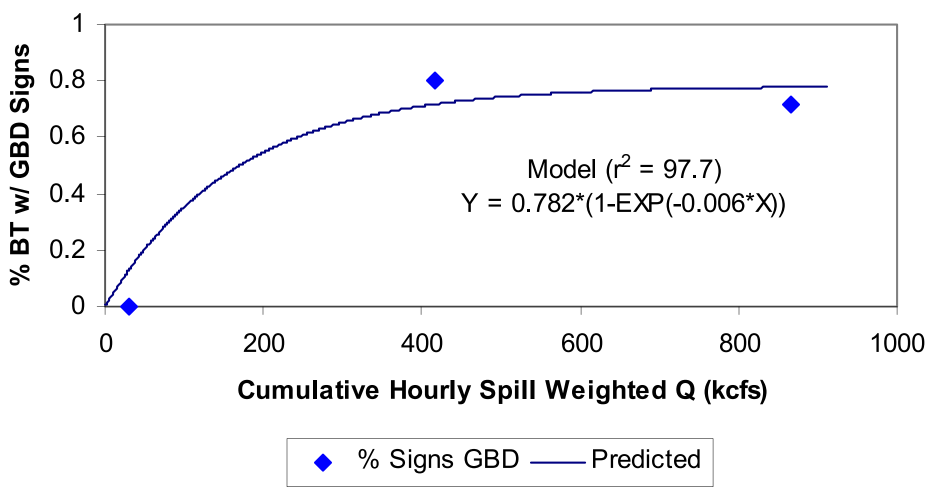
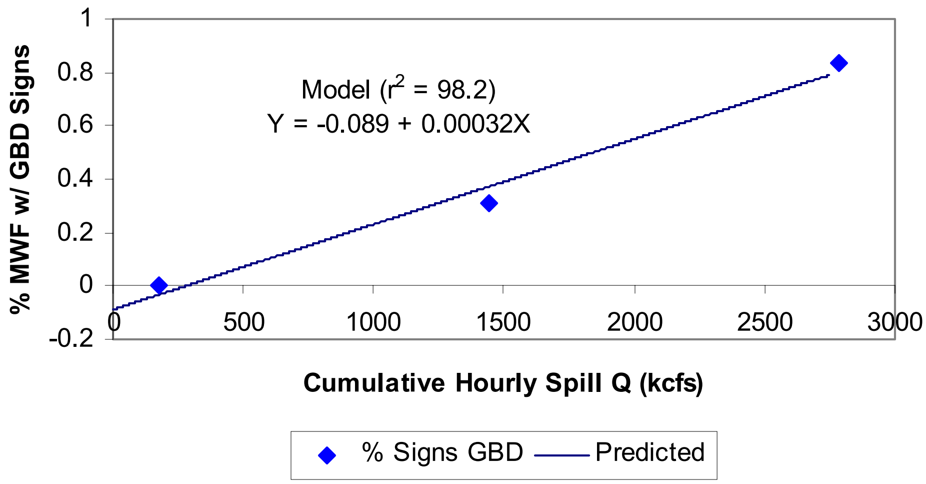


Figure 8. The relation between the cumulative hourly spill flow (top graph), cumulative hourly spill weighted flow (bottom graph), and the proportion of bull trout (BT) captured via electrofishing observed exhibiting signs of gas bubble disease (GBD) during spill activities in the Kootenai River below Libby Dam in the summer of 2002.



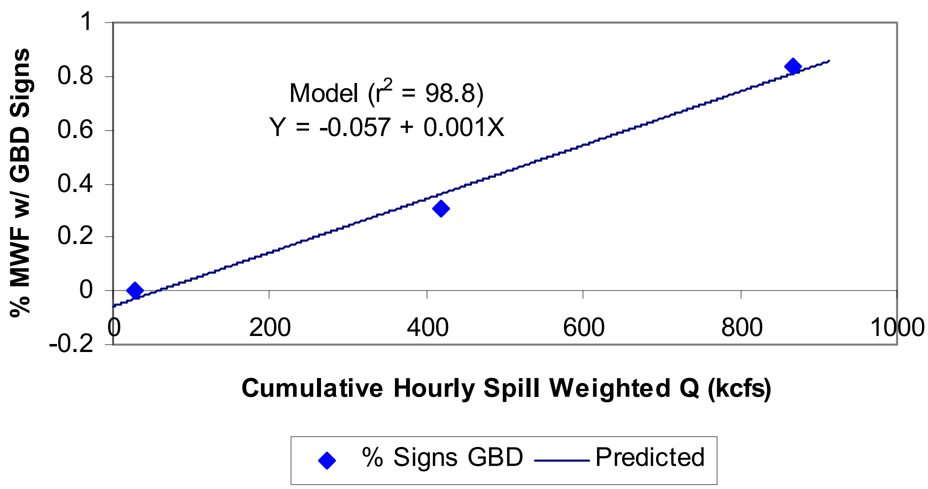


Figure 9. The relation between the cumulative hourly spill flow (top graph), cumulative hourly spill weighted flow (bottom graph), and the proportion of mountain whitefish (MWF) captured via electrofishing observed exhibiting signs of gas bubble disease (GBD) during spill activities in the Kootenai River below Libby Dam in the summer of 2002.

Radio Telemetry

We were able to locate all the radio tagged fish at least once with the exception of 1 rainbow trout and 1 bull trout (Table 6). Radio tagged fish were located an average of 7.9 times during our mobile tracking occurring 1 day prior to spill to 33 days after the last spill event at Libby Dam. The 3 radio tagged mountain whitefish were located an average of 9.7 times per fish, and bull trout and rainbow trout were located an average of 7.0 and 7.8 times per fish, respectively, but were not significantly different ($P = 0.717$).

Table 6. Summary of radio tag frequencies, total length and total number of observations (detections) for 8 rainbow trout, 5 bull trout and 3 mountain whitefish radio tagged and mobile tracked during the spill activities at Libby Dam.

Species	Radio Frequency (kHz)	Total Length (mm)	Total Observations
Rainbow Trout	49.105	510	11
Rainbow Trout	49.341	368	13
Rainbow Trout	49.541	406	2
Rainbow Trout	49.591	384	11
Rainbow Trout	49.711	375	6
Rainbow Trout	49.751	457	10
Rainbow Trout	49.801	394	0
Rainbow Trout	49.811	435	9
Bull Trout	49.531	711	0
Bull Trout	49.551	710	7
Bull Trout	49.571	813	9
Bull Trout	49.651	686	12
Bull Trout	49.741	610	7
Mountain Whitefish	49.601	440	11
Mountain Whitefish	49.611	406	13
Mountain Whitefish	49.771	403	5
Average			**7.9**

We believe that all the radio tagged fish that were detected at least once during the spill activities at Libby Dam were alive, based on repeated upstream movements during mobile tracking activities. Of those radio tagged fish that were detected at least once, most rainbow trout (n= 7; 100%), bull trout (n = 3; 75%) and mountain whitefish (n = 2; 67%) did not move substantially during the spill activities at Libby Dam, and remained within the general vicinity of Libby Dam (RM 221.7) downstream to Dunn Creek (RM 219.8), with the center of gravity more near Libby Dam. This information suggests that spill activities at Libby Dam did not cause substantial geographic fish displacement. We detected substantial movement of 1 mountain whitefish and 1 bull trout. We observed mountain whitefish, tag number 49.611 in the vicinity of the Libby Dam afterbay from

6/25 – 6/27 (9 observations total). The next detection was near the confluence of Rainy Creek (RM 209.9) on 7/2. The final detection dates for this fish were 7/8 and 8/19 in the vicinity of the Libby Dam afterbay. Bull trout number 49.741 was detected near Canoe Gulch (RM 216.7) on 8/19, which was the last observation date for this fish. However, the second to last observation was in the lower Alexander Creek side channel (RM 220.3) on 6/26. Since we don't have any location information between 6/26 and 8/19, we don't know if the downstream movement occurred during the spill activities.

A gradient of supersaturated water occurred across the river channel in the tailrace of Libby Dam during the spill activities (Schneider and Carroll 2002; Figure 10). This mixing zone and associated cross sectional gradient of supersaturated water extends downstream from Libby Dam at least 6.1 miles (Schneider and Carroll 2002). We examined the radio telemetry data collected prior to and during the spill activities to attempt to determine if the tagged fish detected and avoided the supersaturated water along the left bank below Libby Dam. used a paired t-test to assess whether the number of detections for radio tagged fish were consistently higher along the left or right banks below Libby Dam, on a fish by fish basis during spill activities. We detected over twice as many radio tagged fish along the left bank below Libby Dam as those detected along the right bank (mean number of detections per fish = 3.4 and 1.6, respectively). However, differences were not significantly different either for a single-tailed or two-tailed test (P = 0.064 and 0.128, respectively). The power of these two tests were 0.560 and 0.709, respectively for alpha = 0.05. Radio tagged fish also moved across the river channel during the spill activities. We detected at total of 6 fish that changed bank orientation at least once, these included 3 rainbow trout, 2 mountain whitefish, and 1 bull trout. Radio tagged fish were also infrequently observed inhabiting the mid-river channel. We observed 4 different radio tagged fish in the mid-river channel, including 2 rainbow trout, 1 bull trout and 1 mountain whitefish. Each of these fish was only observed in the mid-river channel once.

Figure 10. This photograph was taken from the top of Libby Dam looking downstream during the first spill event at Libby on June 25, 2002. Spill discharge was approximately 700 cfs, total dam discharge was 22.8 kcfs. The lack of mixing of spill and turbine water across the channel created a gradient of total dissolved gas concentrations across the river channel for several miles downstream.

Discussion

Spill activities at Libby Dam during June and July, 2002 impacted resident fish in the Kootenai River, including bull trout, a species listed as threatened under the Endangered Species Act. This field studies indicated that signs of gas bubble disease were common for fish held in cages and fish collected from the river, and that symptoms generally developed rapidly (within 2-3 days) relative to the duration of spill activities. Observations for these field studies were limited to an area within 1.7 miles downstream of Libby Dam, and likely represent the worst-case scenario of potential impacts to fish in the lower Kootenai River because the highest TDG concentrations in the Kootenai River during the spill activities were observed in close proximity to Libby Dam (Schneider and Carroll 2002).

Our field study did not estimate mortality caused by gas bubble disease or other factors associated with the spill activities at Libby Dam. This was in part because the original intent of the study was to provide real time monitoring information that would identify early signs of gas bubble disease in fish and stop spill activities. Although we did observe mortality of captive fish, we were not able to extrapolate mortality rates of captive fish to the population at large in the river due to several confounding factors. The stress associated with electrofishing and handling likely contributed to mortality of captive fish held in the hoop traps. Captive fish also likely experienced additional stress due to confinement at densities that were many times higher than those existing in the Kootenai River. Due to the narrow range of depths available within the hoop traps (2 feet diameter) and relatively shallow positioning of the hoop traps (3-6 feet), captive fish had little opportunity for hydrostatic compensation. Many studies have demonstrated that hydrostatic pressure increases with water depth and can compensate for the effects of supersaturation exerted on a fish (Marsh and Gorham 1905; Blahm et al. 1973; Blahm 1974; Blahm et al. 1976; Ebel 1969; Ebel 1971; Meekin and Turner 1974; Weitkamp 1976). Hoffman et al. (2002) found that adult rainbow trout preferred habitat on the Kootenai River was characterized by water depths of 3-5 feet, mean velocities (mean velocities of 20% and 80% depth) of 1-3 feet/second, and large cobble and larger substrates. In the same study, mountain whitefish tended to prefer slightly deeper water, with suitability values peaking at depths of 4-7.5 feet, similar focal velocities ranging from 1.5-3.5 feet/second, and slightly smaller substrate ranging from large gravel to large cobble. We were not able to locate similar information for bull trout. However, given the habitat preferences for rainbow trout and mountain whitefish and the velocity and depths conditions that actually occurred over the range of observed discharges during the spill activities (23.5 – 38 kcfs), rainbow trout and mountain whitefish might have inhabited deeper water than that which the captive fish were held. If this were the case, hydrostatic compensation may help explain why we observed different responses to supersaturated water exposure time for captive and fish captured via electrofishing (Figure 6). We acknowledge that the response curves for fish captured via electrofishing (Figures 6-9) were developed with three data points each, and that the relatively low number of observations contributed to the relatively high regression r^2 values. However, we believe these data accurately represent conditions that existed in the Kootenai River directly below Libby Dam. We base our conclusions on the consistency of similar results across the three fish species for the three sampling periods, and the similarity in the general shape of the response curves between the captive fish and the fish captured via electrofishing.

The majority of the bull trout, rainbow trout and mountain whitefish captured via electrofishing below Libby Dam on July 24, one week after the final spill event at Libby Dam, exhibited fin damage (Table 5). This damage was presumably caused by necrosis of the fin tissue between the fin rays that was ultimately caused by gas emboli between within the fins. This observation suggests that many fish survived the initial spill period and were beginning to heal. However, the injuries we observed might have resulted in an increased susceptibility to fungal and bacterial infections, that could have resulted in delayed mortality. Weitkamp (1976) found fungal infections were responsible for delayed mortality of juvenile chinook exposed to supersaturated water that caused lesions and hemorrhages near the base of the caudal fin. We did not however observe any fungal infections associated with the damaged fins observed on the evening of July 24.

This field study provided insufficient evidence to conclusively decide whether or not bull trout, rainbow trout or mountain whitefish were able to detect the elevated TDG concentrations below Libby Dam during the spill activities. In a review by Weitkamp and Katz (1980), they concluded that the ability to detect and avoid supersaturated may be species specific, but that in general too little information exists to draw definitive conclusions on the subject. Although mobile radio telemetry tracking efforts suggested that radio tagged fish preferred the left bank, the differences were not statistically significant. These results for rainbow trout are consistent with those of Blahm et al. (1976), who found that juvenile steelhead did not avoid supersaturated water. We were unable to find published information regarding whether or not bull trout or mountain whitefish were able to detect supersaturated water.

The Montana Clean Water Act is the foundation that the Montana Department of Environmental Quality used to establish surface water quality standards, including limits on total dissolved gas concentrations. This standard is 110% (MT DEQ 2001), and was established to protect aquatic gill breathing organisms from the harmful effects of gas supersaturation. The federal standard set by the Environmental Protection Agency is also 110%. The Montana Department of Environmental Quality granted a short-term exemption from the 110% TDG standard during the spill test at Libby Dam. The standard was exceeded below Libby Dam early during the spill activities at Libby Dam, where TDG exceeded 120% approximately 250 feet below the stilling basin during the second spill event (Schneider and Carroll 2002). Approximately 3.0 kcfs passed over the spillway at Libby Dam during the second spill event, comprising approximately 12.6% of the total discharge passing Libby Dam at that time. The 110% standard was exceeded for the remainder of the spill activities at Libby Dam at this sampling location (Schneider and Carroll 2002). TDG concentrations peaked in the stilling basin below Libby Dam during spill event 16, exceeding 134% (Schneider and Carroll 2002). Therefore, given the relatively rapid response of TDG concentrations to relatively small amounts of spill water and the findings of this field study, the use of spill as a regular management activity at Libby Dam appears to have limited practical application. Libby Dam managers may seek to explore the feasibility and cost efficiency of making structural modifications at Libby Dam that could potentially reduce spill water plunge into the stilling basin, and reduce TDG concentrations or increase powerhouse capacity to allow higher dam discharges without spilling water.

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Appendix

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

Year	1985	1997	1998	1999	2000	2001	2002
Section 1							
Westslope Cutthroat	-----	6 (6.16)	35 (42.27)	129 (134.22)	-----	-----	97 (105.63)
Brook Trout	-----	40 (41.73)	95 (110.30)	89 (90.16)	-----	-----	57 (60.24)
Bull Trout	-----	-----	-----	-----	-----	-----	2
Mountain Whitefish	-----	-----	-----	-----	-----	-----	2
Total Population [A]	-----	47 (48.60)	132 (149.68)	221 (225.64)	-----	-----	155 (162.95)
Section 2							
Westslope Cutthroat	-----	8 (9.79)	52 (69.33)	153 (158.31)	89 (122.65)	75 (76.78)	114 (129.66)
Rainbow Trout	-----	8 (8.88)	-----	4	-----	1	-----
Brook Trout	-----	43 (63.56)	64 (71.33)	63 (66.61)	68 (71.32)	51 (51.99)	85 (94.82)
Bull Trout	-----	-----	-----	7 (10.08)	1	-----	2
Total Population [A]	-----	56 (70.43)	116 (131.81)	226 (233.79)	149 (164.97)	127 (129.60)	202 (221.97)
Section 3							
Westslope Cutthroat	308 (314.85)	139 (172.33)	258 (292.35)	239 (253.37)	-----	-----	264 (292.90)
Brook Trout	43 (49.98)	66 (162.11)	64 (67.14)	82 (85.77)	-----	-----	114 (122.21)
Rainbow Trout	26 (27.75)	2	-----	32 (33.79)	-----	-----	2
Bull Trout	-----	-----	1	-----	-----	-----	2
Total Population [A]	378 (388.63)	232 (320.50)	320 (348.97)	354 (369.01)	-----	-----	385 (412.98)

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

Table A2. 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
Section 1						
Rainbow Trout	123 (260.84)	130 (150.91)	82 (89.15)	-----	-----	-----
Brook Trout	41 (46.52)	49 (56.27)	60 (63.67)	-----	-----	-----
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)	-----
Brook Trout	56 (57.53)	125 (136.96)	72 (80.47)	-----	82 (87.34)	-----
Bull Trout	47 (48.87)	15 (16.42)	3	-----	2	-----
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)	-----		-----
Brook Trout	74 (76.7)	82 (87.79)	110 (116.71)	-----		-----
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 A3. Lower Grave Creek demonstration project area electrofishing. Numbers are total catch within the 1,000 foot section.

Year	2000[A]	2001[B]	2002[C]
Westslope Cutthroat	4	18	3
Rainbow Trout	1	17	26
Brook Trout	1	10	5
Bull Trout	9	33	5
Mountain Whitefish	54	3	33
Long Nose Dace	6	-----	-----
Water Temp. °C	-----	17	-----
Effort (minutes)	44	56.9	NA

A) Four bull trout ≥ 490 mm were likely lacustrine - adfluvial fish from Koocanusa 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 Koocanusa 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 Koocanusa Reservoir moving into Grave Creek to spawn. Long nose dace were observed but not counted.

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

Year	1996	1997	1998	1999	2000	2001	2002
Section 1 (Tooley)							
Westslope Cutthroat[B]	-----	3	36 (37.05)	139 (147.55)	-----	55 (64.28)	88 (95.53)
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)	-----	-----	-----
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)
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)
Rainbow Trout	-----	-----	-----	-----	-----	-----	-----
Brook Trout	-----	-----	62 (70.63)	52 (65.33)	19 (21.86)	25 (27.08)	46 (48.81)
Bull Trout	-----	-----	-----	-----	-----	-----	2
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 A5. 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
Section 1 – below Hwy 2					
Rainbow Trout	81 (126.80)	26	125	46 (51.09)	117 (129.56)
Brook Trout	6 (8.27)	6	13	10 (12.33)	16 (24.29)
Bull Trout	-----	-----	-----	-----	3
Mountain Whitefish	-----	-----	-----	-----	3
Total Population [B]	90 (115.89)	32	138	57 (63.79)	138 (152.67)
Water Temp. °C	9	-----	16	15	14
Discharge (cfs)	6.9	-----	-----	-----	-----
Section 2 -above Hwy 2					
Rainbow Trout	203 (225.20)	-----	-----	148 (192.77)	-----
Brook Trout	7	-----	-----	2	-----
Bull Trout	5 (6.26)	-----	-----	-----	-----
Total Population [B]	208 (228.39)	-----	-----	160 (213.40)	-----
Water Temp. °C	5	-----	-----	20	-----
Discharge (cfs)	6.9	-----	-----	-----	-----
Section 3 - upper Cleveland					
Rainbow Trout	-----	-----	170 (193.73)	172 (182.26)	163 (183.16)
Brook Trout	-----	-----	-----	-----	-----
Bull Trout	-----	-----	3	8 (11.15)	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 A6. 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
	Pre-reconstruction	Post-reconstruction
Rainbow Trout	92 (110.65)	79 (95.9)
Brook Trout	18 (19.20)	1
Bull Trout	-----	1
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 A7. 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]
Rainbow Trout	42 (46.42)	73 (84.97)
Brook Trout	-----	3
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 A8. 2002 Barron Creek depletion population estimate for fish ≥ 75 mm per 1,000 feet using 95 % confidence. Upper confidence intervals are in parenthesis.

Section	1	2	3	4
Oncorhynchus spp.[A]	30 (32)	3.6 (27.03)	9.9 (11.9)	0
Brook Trout	73 (78)	301 (315.3)	136.6 (148.5)	246.2 (267.7)
Total Population[B]	104 (109)	304.5 (318.9)	146.5 (158.4)	246.2 (267.7)

A). Hybridization between rainbow and cutthroat trout was prolific, making identification difficult. Estimates were therefore combined for rainbow, cutthroat and rainbow x cutthroat trout hybrids.

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

Table A9. 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 A10. 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 A11. 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 A12. 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 A13. 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 A14. 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 A15. 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**

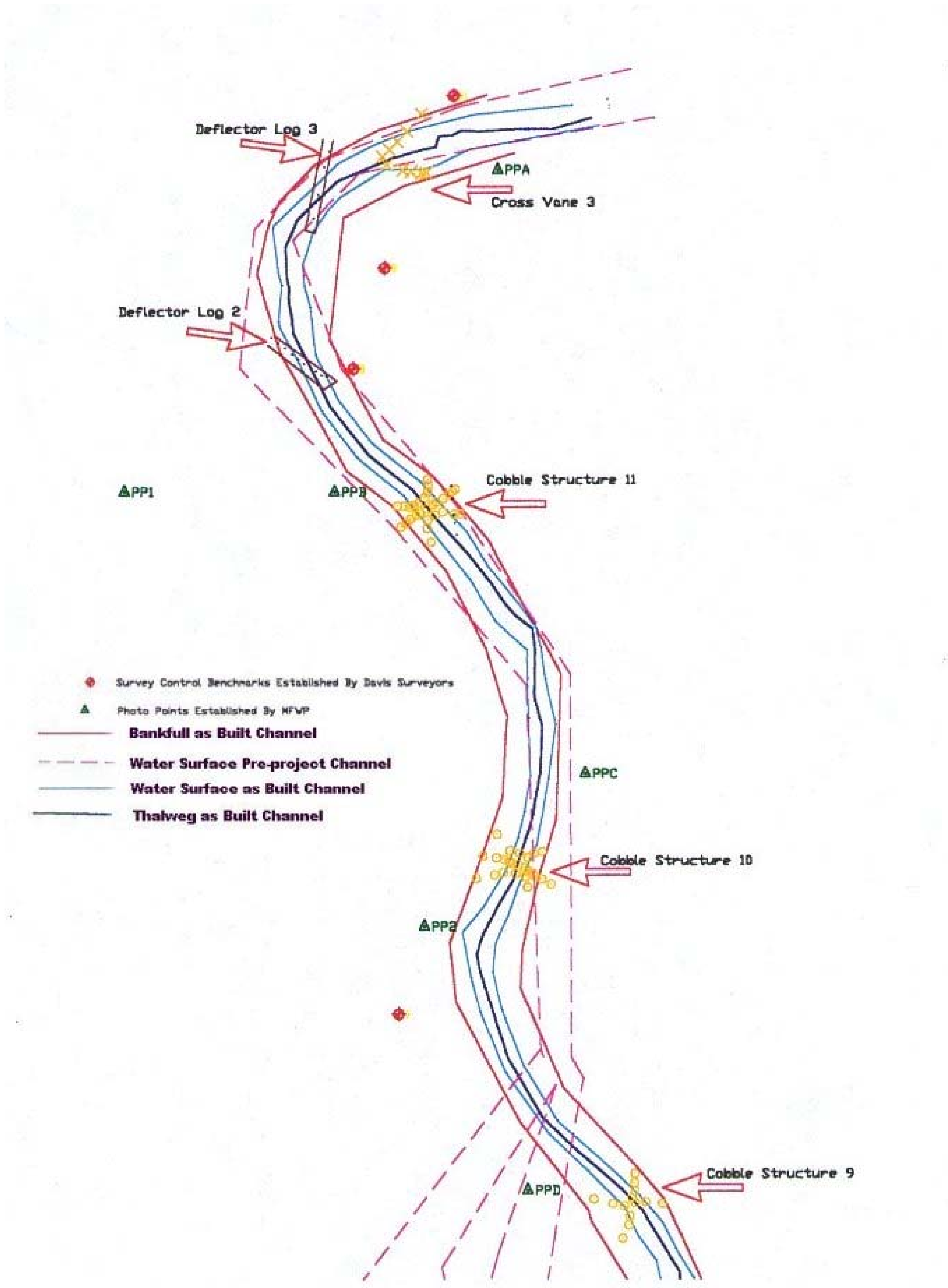

Figure A1. Plan view of the existing stream channel and constructed stream channel for the Libby Creek Cleveland Restoration Project beginning at the lower project site (top of figure) to upstream to approximately station 800 (lower portion of figure).

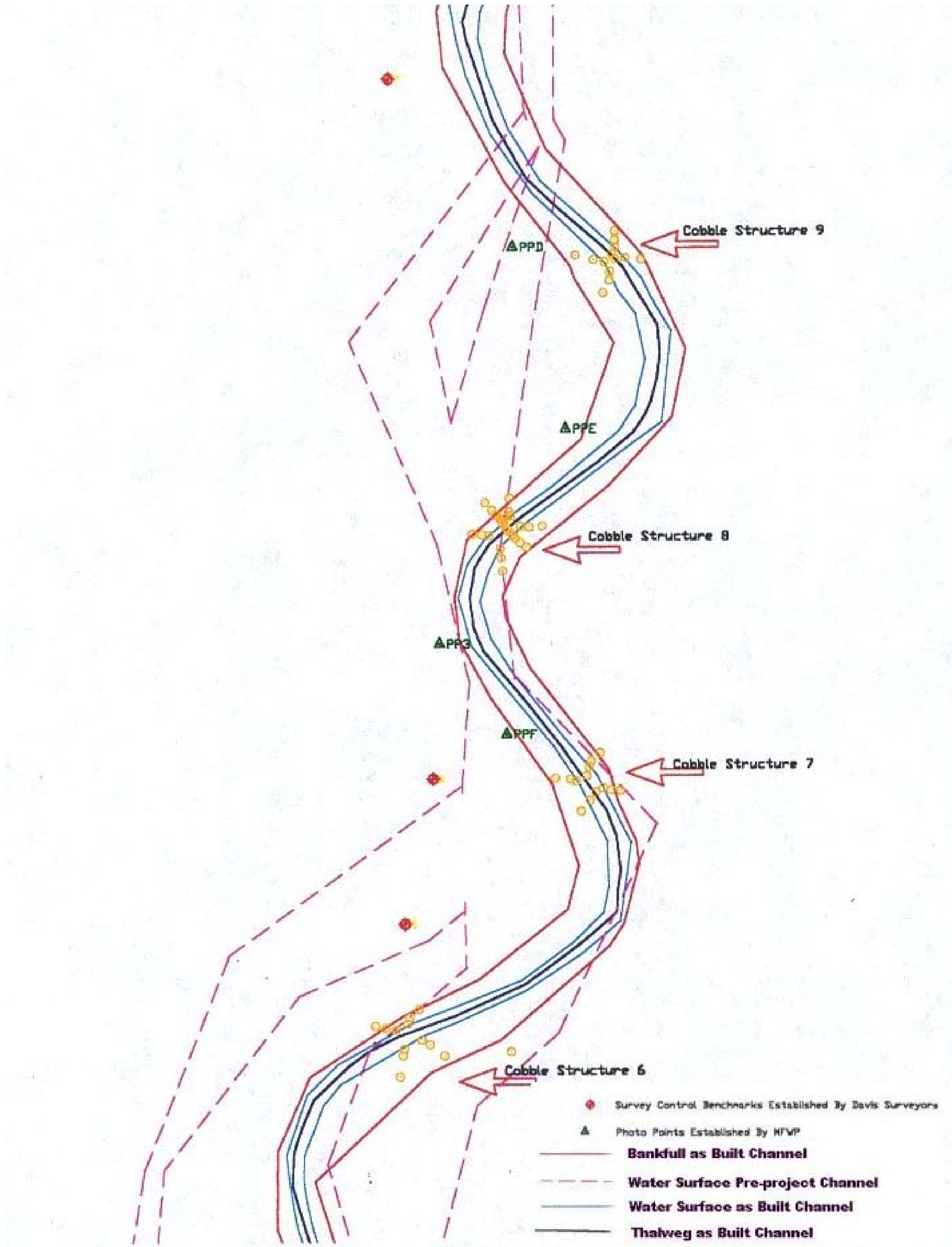

Figure A1 (continued). Plan view of the existing stream channel and constructed stream channel for the Libby Creek Cleveland Restoration Project beginning at approximately station 800 (upper portion of figure) proceeding upstream approximately to station 1600 (lower portion of figure).

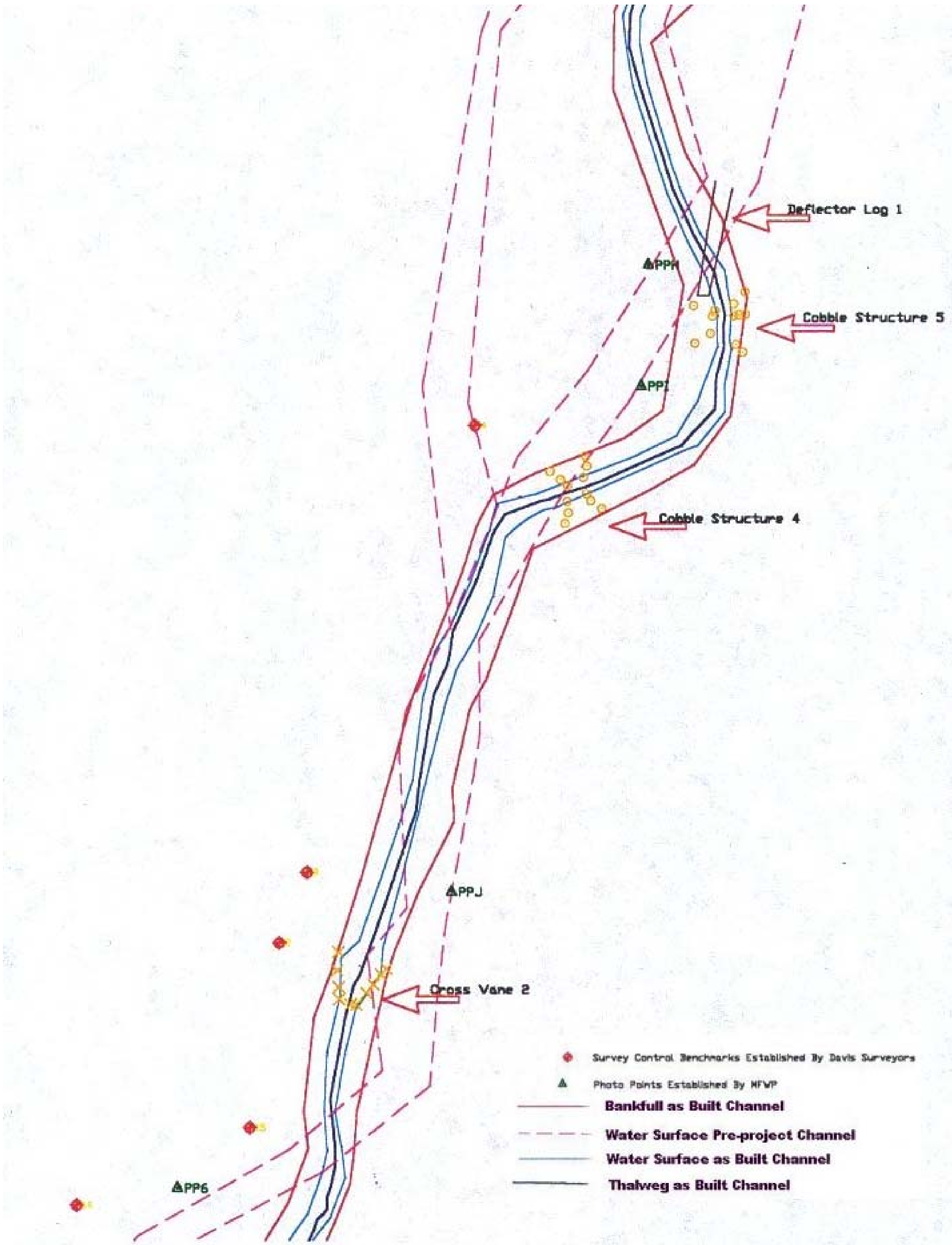

Figure A1 (continued). Plan view of the existing stream channel and constructed stream channel for the Libby Creek Cleveland Restoration Project beginning at approximately station 1600(upper portion of figure) proceeding upstream approximately to station 2400 (lower portion of figure).

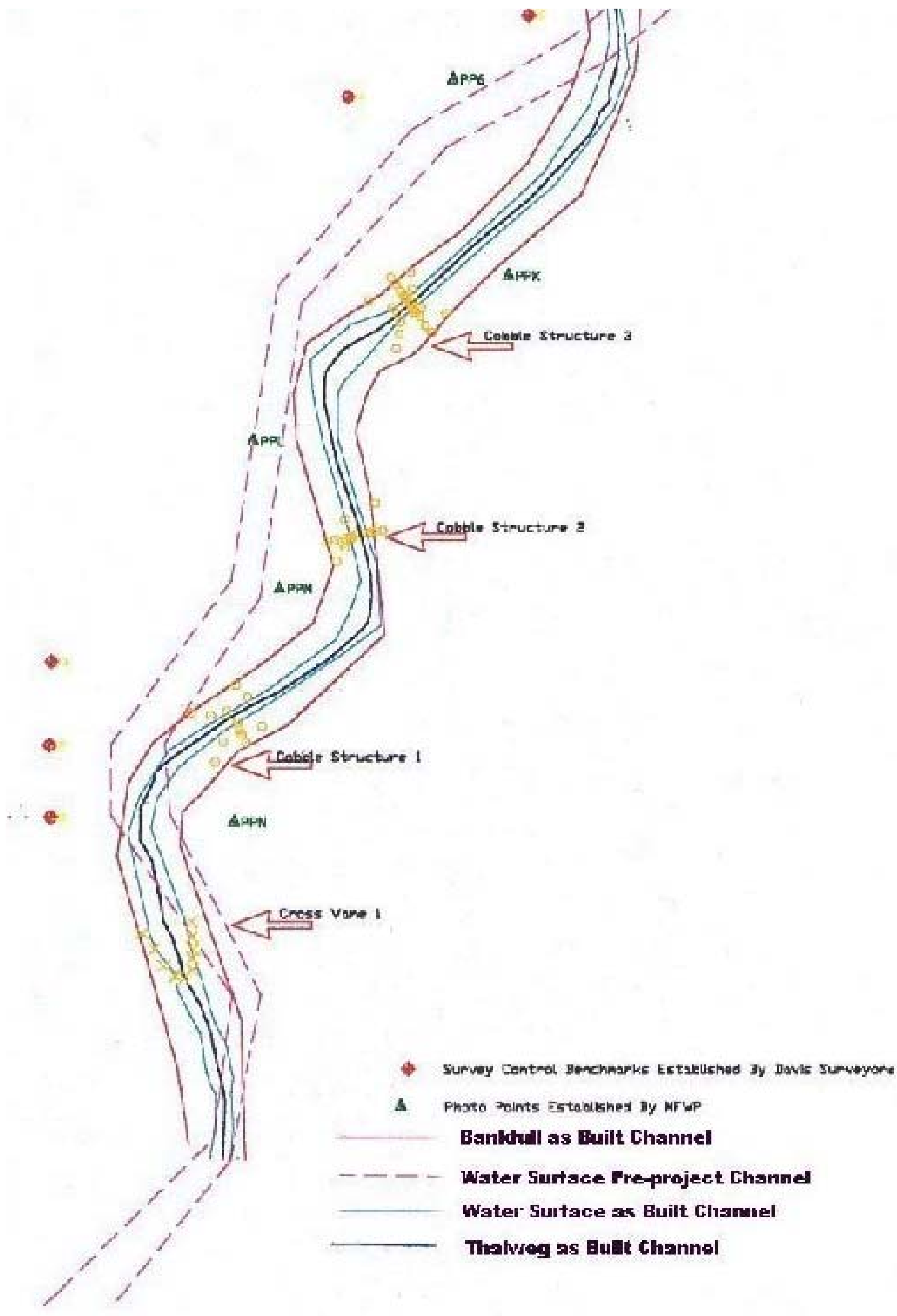

Figure A1 (continued). Plan view of the existing stream channel and constructed stream channel for the Libby Creek Cleveland Restoration Project beginning at approximately station 2400 (upper portion of figure) proceeding upstream to the upper project boundary located at approximately to station 3200 (lower portion of figure).