Libby Mitigation Program

Mitigation for the Construction and Operation of Libby Dam

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MITIGATION FOR THE CONSTRUCTION AND OPERATION OF LIBBY DAM

ANNUAL REPORT
2007
(Work Activities July 1, 2007 – June 30, 2008)

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EXECUTIVE SUMMARY

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

Montana Fish, Wildlife & Parks (MFWP) uses a combination of techniques to collect 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 and threatened species, and the assessment of restoration or management activities designed to restore native fishes and their habitats. The following points summarize the biological monitoring accomplished from July 2006 to June 2007.

- MFWP monitored the relative abundance of burbot in the stilling basin below Libby Dam using hoop traps since 1994. During the 2007/2008 trapping season we captured only two burbot below Libby Dam, which represents the third lowest catch rate during the period of record. The catch of burbot at this location has exhibited a significant decline since 1994.
- We conducted juvenile salmonid population estimates within reference reaches on Therriault, Grave, Young, Libby, and Pipe creeks in order to evaluate fish population response to habitat work. Trend analyses and before/after/control analyses related to stream restoration projects are presented for Therriault, Young, Grave and Libby creeks.
- MFWP has documented the changes in fish species composition, and species size and abundance within Libby Reservoir since the construction of Libby Dam. We continued monitoring fish populations within the reservoir using spring and fall gill netting and present the results and trend analyses for 11 fish species. The average length and weight of kokanee was the highest it has been in past eight years, while the average catch rate was also the lowest since 1992. The spring gill net catch of bull trout has significantly increased since 1990. The spring gill net catch of bull trout is significantly and positively correlated to the bull trout redd counts in the Wigwam River and Grave. We were able to

improve trend analyses using an adjusted catch of bull trout that accounted for differing reservoir levels at the time of netting. The catch of Kamloops rainbow trout in fall floating gillnets was significantly and positively correlated with the number of age one hatchery Kamloops rainbow trout stocked in the reservoir the previous year, but was not correlated with the number of age 0 hatchery trout released in the reservoir the previous year.

- MFWP 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* has been the most abundant genera of zooplankton present in the reservoir since 1997, and *Daphnia* was the second most abundant genera of zooplankton within the reservoir in 2007. Zooplankton abundance within the reservoir varies by month, with the monthly abundance peaks over the past ten years remaining relatively consistent.
- Bull trout redd counts in Grave Creek and the Wigwam River have both exhibited significant positive trends since the mid 1990s. There were a total of 1,883 bull trout redds within the index portion of the Wigwam River, which was approximately an order of magnitude higher than any other tributary within the Kootenai Basin. Bull trout core areas in the Kootenai River downstream of Libby Dam include Quartz, Pipe, Bear (Libby Creek drainage), O'Brien creeks and the West Fisher Creek. 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. Three of the four populations between Libby Dam and Kootenai Falls were below average over the period of record. West Fisher Creek was the only stream in this group that was slightly above average. Record low bull trout redd counts were observed in Pipe Creek, and a second highest counts was observed in O'Brien Creek. The only bull trout population that exhibited a significant positive trend, was O'Brien Creek. The adjunct Bull Lake population, which spawns in Keeler Creek, was below the long-term average and the lowest since 2001.
- MFWP conducted a population estimate for adult bull trout below Libby Dam during April 2008, and estimated 381 (95% Confidence Interval = 158 – 605) adult bull trout were present within this 3.5-mile section of the Kootenai River, which was similar to the previous year. We recaptured 16 bull trout that were previously marked in 2004, 2005, 2006, or 2007 below Libby Dam ranging between 313 to 1,117 days prior. The recaptured bull trout grew an average of 96.3 mm (0.21 mm per day), and gained an average of 1,649.9 g (3.07 g per day).
- MFWP collected DNA samples from *Salvelinus spp.* from Keeler, Pipe, West Fisher, and O'Brien creeks in order to examine hybridization rates between brook and bull trout. All four streams contained hybridized fish, with hybridization especially problematic in O'Brien and Keeler creeks. However, bull trout were rare in Pipe Creek. Overall, we were able to correctly visually identify 91% of the fish collected for analysis. The most

common visual misidentification involved calling later generation hybrids as being either brook trout or bull trout.

- MFWP monitored fine sediment (<6.35 mm) levels in eight bull trout spawning tributaries within the Montana portion of the Kootenai Basin using core sampling. Keeler Creek had the lowest mean levels of fine sediment amongst the eight streams we monitored over the period of record, and O'Brien Creek had the highest mean levels of fine sediment. Of the four core bull trout spawning tributaries located on the Kootenai River between Libby Dam and Kootenai Falls, West Fork Quartz Creek had the lowest annual mean fine sediment levels. Mean annual fine sediment levels on Pipe Creek and Bear Creek were both relatively high, but similar. The two bull trout spawning tributaries located in Montana upstream of Libby Dam (Grave Creek and MT Wigwam) had relatively low fine sediment levels.

A cooperative mitigation and implementation plan developed by MFWP, the Kootenai Tribe of Idaho and the Confederated Salish and Kootenai Tribes documents hydropower-related losses and mitigation actions attributable to the construction and operation of Libby Dam, as called for by the Northwest Power and Conservation Council's Fish and Wildlife Program (MFWP et al. 1998). A mix of mitigation techniques is necessary to offset losses caused by dam construction and operation. During the 2006 contract period, MFWP implemented a small lake rehabilitation project, a stream restoration project, and monitoring activities to evaluate stream channel response to six previously completed stream restoration projects. The following points summarize these activities.

- MFWP treated Loon Lake (Pipe Creek Watershed) and its tributaries with rotenone in September 2007 to remove non-native brook trout and black bullheads. We added potassium permanganate to the stream near an existing fish barrier approximately miles downstream of the lake. Gillnetting and backpack electrofishing efforts conducted in the spring of 2008 confirmed that we had successfully all fish from the lake and outlet stream, respectively. MFWP restocked the lake in the spring of 2008 with age 0 westslope cutthroat trout.
- The Grave Creek Phase I Restoration Project was completed in the fall of 2002, and changed the dimension, pattern and profile of this section of Grave Creek, which increased the overall stream length and created a deeper and narrow stream channel with increased pool habitat. In 2007 we observed a slight increase in the total number of pools within the project area, but all other stream channel dimensions remained similar between 2006 and 2007. Stream channel dimensions within riffle habitats and riffle slopes also changed little between 2006 and 2007, with changes generally < 5%. The substantial changes in planform, stream channel dimensions and longitudinal profile that resulted from this project have been sustained through time.
- MFWP continued our partnership with entities of the Kootenai River Network and completed the Grave Creek Phase II Restoration Project in the fall of 2004. We collected stream channel dimensions annually before and after project construction in order to

evaluate project performance through time. Substantial maintenance and revegetation efforts were completed in the fall of 2006 which was likely attributable to the increase in total pool habitat from 2006. Stream channel dimensions in riffle habitats remained similar between 2006 and 2007.

- The Libby Creek Upper Cleveland Restoration Project was completed in the fall of 2002, and restored 3,200 feet of stream channel. The Libby Creek Lower Cleveland Phase Restoration Project Area is located approximately one mile downstream of the Upper Cleveland Project. The Lower Project was conceived as a three-phase effort, with phases I and II completed in the fall of 2005 and 2006, respectively. However, during the first week of November 2006, the Libby Creek watershed experienced a rain on snow weather event with an occurrence interval of approximately a 20-year event. The most conspicuous changes within the three restoration project areas on Libby Creek as a result of the November 2006 flow event were changes in the stream planform, with substantial stream straightening and loss of meander pattern throughout, and a loss of the total number of pools. Stream width increased, especially within riffle habitats after the flood event. These patterns of disturbance were exacerbated along a downstream gradient, with changes most substantial within the Lower Cleveland Phase II project area. Despite the substantial changes resulted from the flood, the quantity of pool habitat in the Upper and Lower Phase I Cleveland Project areas has remained higher than existed prior to the restoration.
- MFWP completed the Young Creek State Lands Restoration Project in the fall of 2003, which changed the stream channel dimensions within this area. The monitoring results presented in this document evaluated whether these physical changes were maintained since construction. The stream channel dimensions within the riffles of this section of Young Creek changed only slightly, with an overall increase in cross sectional area and bankfull width, and maximum depth, and a decrease in overall bankfull depth within riffle habitats. Pool dimensions and numbers changed little since construction (generally < 10%) within the project area.
- MFWP partnered with The Kootenai River Network (KRN), the USFWS Partners for Wildlife and the local landowner in 2004 and 2005 to complete the Therriault Creek Restoration Project, which reconstructed a total of 9,100 feet of entirely new stream channel, which represented a two-fold increase in stream length. This project also created approximately 55 acres of prior converted wetland. The largest fundamental change that resulted from this project was that of stream channel type and planform. The existing stream channel was an entrenched G4 and F4 Rosgen stream type, and restoration work converted it to an E4 channel type, with low entrenchment and width to depth ratios. Mean bankfull width increased as a result of the project. Mean and maximum bankfull depth, cross sectional area and width within the riffles/runs increased in both reach also as a result of this project. Stream channel dimensions have changed little since 2004.

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INTRODUCTION

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

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

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

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

PROJECT HISTORY

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

A long-term database was established for monitoring populations of kokanee, bull trout, westslope cutthroat trout, rainbow trout and burbot and other native fish species. Long-term monitoring of zooplankton and trophic relationships was also established. A model was calibrated to estimate the entrainment of fish and zooplankton through Libby Dam as related to hydro-operations and use of the selective withdrawal, thermal control structure. Research on the entrainment of fish through the Libby Dam penstocks began in 1990, and results were published in 1996 (Skaar et al. 1996). The effects of dam operation on benthic macroinvertebrates in the Kootenai River was also assessed (Hauer et al. 1997) for comparison with conditions measured in the past (Perry and Huston 1983). This study was replicated in 2005 with the addition of examining the effect of a nuisance diatom (*Didymosphenia geminata*) on the benthic community (Marshall 2007). 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. This project developed non-lethal genetic methodologies to differentiate between native redband trout and non-native rainbow trout (Brunelli et al. 2008), and a non-lethal genetic methodology to identify natal tributary origin for bull trout in the upper Kootenai Watershed and quantify bull trout entrainment at Libby Dam (Ardren et al. 2007). Research on the effects of operations on the river fishery using Instream Flow Incremental Methodology (IFIM) techniques was initiated in 1992. Assessment of the effects of river fluctuations on Kootenai River burbot fishery was examined in 1994 and 1995. IFIM studies were also completed in Kootenai River below Bonners Ferry, Idaho, to determine spawning area available to sturgeon at various river flows. Microhabitat data collection specific to species and life-stage of rainbow trout and mountain whitefish has been incorporated into suitability curves. River cross-sectional profiles, velocity patterns and other fisheries habitat attributes were completed in 1997. Hydraulic model calibrations and incorporation of suitability curves and modification of the model code were completed in 1999, and updated by Miller Ecological Consultants, Inc in 2003 (Miller and Geise 2004).

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

1997 – MFWP 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 - MFWP 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 - MFWP developed an isolation facility for the conservation of native redband trout at the Libby Field Station. Existing ponds were restored and the inlet stream was enhanced for natural outdoor rearing. Natural reproduction may be possible. Activities included chemically rehabilitating the system and constructing a fish migration barrier to prevent fish movement into the reclaimed habitat.

1998 - MFWP 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 - MFWP 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 - MFWP 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 - MFWP was a major contributor (financial and in-kind services; primarily surveying) towards completion of Parmenter Creek re-channelization/rehabilitation work (Project Impact). Parmenter Creek has the potential to provide additional spawning and rearing habitat for Kootenai River fish, most likely westslope cutthroat trout.

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

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

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

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

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

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

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

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

2004 – MFWP collaborated with the Kootenai River Network to reconstruct approximately 3,100 feet of stream channel on Grave Creek (Phase II Restoration Project) in order to stabilize stream banks, reduce sediment, and improve rearing habitat for salmonids.

2005 - MFWP excavated approximately 2,950 feet of new stream channel during fall 2005 to complete the Libby Creek Lower Cleveland Phase I Restoration Project. The resulting stream pattern design increased sinuosity and subsequently increased total stream length from approximately 2,700 to 3,200 feet. This project represented the second phase of restoration activities in the upper Libby Creek Watershed.

2005 – MFWP collaborated with the Kootenai River Network to restore the ecological function to Theriault Creek, a tributary of the Tobacco River by restoring the meander pattern and profile of a 9,300 feet section of stream that had been straightened. This project approximately doubled the stream length within this section of creek.

2006 – MFWP completed the The Libby Creek Lower Cleveland Phase II Project, which started at the downstream boundary of the Phase I project area and restored 3,175 feet of stream to a sustainable planform, profile and channel dimension.

2006 - MFWP chemically rehabilitated Kilbrennan Lake to remove illegally nonnative brook trout, rainbow trout, yellow perch and black bullheads and reestablished redband trout in the lake. We also installed a fish barrier on Kilbrennan Creek, downstream of the lake in order to prevent nonnative fishes from recolonizing the lake.

2006 – MFWP collaborated with the Kootenai River Network to perform maintenance and revegetation efforts on the Grave Creek Phase I and II Restoration Projects.

2006 – MFWP installed a fish screen on an irrigation diversion on lower Libby Creek.

ASSOCIATIONS

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

This project uses radio-telemetry to identify migration habits, habitat preferences and spatial distribution of species in the Kootenai system. Information on species habitat selection is shared with the IFIM project in the Flathead Watershed (Project 199101903).

Project personnel are completing activities in the lower Kootenai River in Montana to provide 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 (20000004 – British Columbia Ministry of Environment) for several years to monitor the status of bull trout in the upper Kootenai River, its tributaries, and Libby Reservoir. Our cooperative activities have included radio tagging and tracking of adult bull trout, redd counts, sediment and temperature monitoring, and migrant fish trip operations.

MFWP 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. MFWP serves on the KRN Executive Board. Formal participation in the KRN helps MFWP achieve our goals and objectives toward watershed restoration activities in the Kootenai Basin.

DESCRIPTION OF STUDY AREA

Subbasin Description

The Kootenai River Subbasin is an international watershed that encompasses parts of British Columbia (B.C.), Montana, and Idaho (Figure 1). The headwaters of the Kootenai River originate in Kootenay National Park, B.C. The river flows south within the Rocky Mountain Trench into the reservoir created by Libby Dam, which is located near Libby, Montana. From the reservoir, the river turns west, passes through a gap between the Purcell and Cabinet Mountains, enters Idaho, and then loops north where it flows into Kootenay Lake, B.C. The waters leave the lake's West Arm and flow south to join the Columbia River at Castlegar, B.C. The annual runoff volume makes the Kootenai the second largest Columbia River tributary. The Kootenai ranks third in watershed area (36,000 km² or 8.96 million acres)(Knudson 1994). The climate, topography, geology, soils and land use characteristics of the Kootenai Basin were previously described in Dunnigan et al. (2003).

Drainage Area

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

Hydrology

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

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

Major tributaries to the Kootenai River below Libby Dam include the Fisher River (838 sq. mi.; 485 average cfs), the Yaak River (766 sq. mi. and 888 average cfs) and the Moyie River (755 sq. mi.; 698 average cfs). Kootenai River tributaries are characteristically high-gradient mountain streams with bed material consisting of various mixtures of sand, gravel, rubble, boulders, and drifting amounts of clay and silt, predominantly of glacio-lacustrine origin. Fine materials, due to their instability during periods of high stream discharge, are continually abraded and redeposited as gravel bars, forming braided channels with alternating riffles and pools. Stream flow in unregulated tributaries generally peaks in late-May or early 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 impediment to fish migrations, 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 Kootenay River has an average annual discharge of 868 m³/s (30,650 cfs).

Fish Species

Eighteen species of fish are present in Libby Reservoir and the Kootenai River (Table 1). The reservoir currently supports an important fishery for kokanee *Oncorhynchus nerka* and rainbow trout *Oncorhynchus mykiss*, with annual fishing pressure over 500,000 hours (Chisholm and Hamlin 1987). Burbot *Lota lota* are also important game fish, providing a popular fishery during winter and spring. The Kootenai River below Libby Dam is a “blue ribbon” trout fishery, and the state record rainbow trout was harvested there in 1997 (over 33 pounds). Although bull trout *Salvelinus confluentus* fishing was banned in the Kootenai River, “incidental captures” provide a unique seasonal fishery.

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

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

* Native species are designated Y, and nonnatives N

Reservoir Operation

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

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

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

Table 2. Morphometric data for Libby Reservoir.

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

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

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

Figure 2. Kootenai River Basin, Montana.

Figure 3. Libby Reservoir elevations (minimum, maximum), water years (October 1 – Sept. 30), 1976 through 2007.

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Chapter 1

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

By:

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Abstract

Montana Fish, Wildlife & Parks (MFWP) uses a combination of techniques to collect 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 and threatened species, and the assessment of restoration or management activities designed to restore native fishes and their habitats.

We monitored the relative abundance of burbot in the stilling basin below Libby Dam using hoop traps since 1994. The burbot catch in our hoop traps below Libby Dam has declined precipitously since 1994, exhibiting a significant negative trend. During the 2007/2008 trapping season we captured two burbot below Libby Dam, which represents the third lowest catch rate during the period of record.

We investigated the relative abundance of burbot in Libby Reservoir during the winter of 2007/2008 using baited hoop traps. We fished the traps at five different reaches throughout the US portion of the reservoir at three sites per reach and 3 traps per site for a total of 702 trap-days, and captured 39 burbot. Catch rates decreased over the trapping season (December-April), which allowed us to adjust catch rates for trap date in order to compare across sites over the season. Adjusted catch rates were lowest for the two lowest reaches (closest to Libby Dam), intermediate for the middle reach and highest for the two most northern reaches within the reservoir.

We conducted juvenile salmonid population estimates within reference reaches on Therriault, Grave, Young, Libby, and Pipe creeks in order to evaluate fish population response to habitat work. Trend analyses related to stream restoration projects are presented for Therriault, Young, Grave and Libby creeks.

MFWP has documented the changes in fish species composition, and species size and abundance within Libby Reservoir since the construction of Libby Dam. We continued

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monitoring fish populations within the reservoir using spring and fall gill netting and present the results and trend analyses for 11 fish species. The average length and weight of kokanee was the highest it has been in past eight years, while the average catch rate was also the lowest since 1992. The spring gill net catch of bull trout has significantly increased since 1990. The spring gill net catch of bull trout is significantly and positively correlated to the bull trout redd counts in the Wigwam River and Grave. We were able to improve trend analyses using an adjusted catch of bull trout that accounted for differing reservoir levels at the time of netting. The catch of Kamloops rainbow trout in fall floating gillnets was significantly and positively correlated with the number of age one hatchery Kamloops rainbow trout stocked in the reservoir the previous year, but when we included the number of age 0 hatchery trout released in the reservoir, it did little to improve the fit.

MFWP 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* has been the most abundant genera of zooplankton present in the reservoir since 1997, and *Daphnia* was the second most abundant genera of zooplankton within the reservoir in 2007. Zooplankton abundance within the reservoir varies by month, with the monthly abundance peaks over the past ten years remaining relatively consistent.

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 MFWP's quantitative reservoir model (LRMOD), Integrated Rule Curves (IRC), Instream Flow Incremental Methodology (IFIM) and Libby Dam fish entrainment model (ENTRAIN), to improve biological production by modifying dam operation, and 3) Recover native fish species including the endangered Kootenai River white sturgeon, threatened bull trout, westslope cutthroat trout, interior redband rainbow trout, and burbot. A loss statement, site-specific mitigation actions and monitoring strategies were documented in the Libby Mitigation and Implementation Plan (MFWP et al. 1988) and Kootenai Subbasin Plan (KTOI and MFWP 2004).

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

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

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

Methods

Burbot Monitoring Below Libby Dam

MFWP 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 flood control strategy (VARQ) was implemented beginning in January 2003. Two hoop traps measuring 2-foot diameter, approximately 6-8 feet in length with $\frac{3}{4}$ 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 among years.

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

Burbot Monitoring Libby Reservoir

MFWP began a detailed field study on Libby Reservoir during mid-November 2007 to investigate relative abundance and distribution of burbot in the reservoir. Little information exists regarding the life history, spawning distribution, abundance or status of burbot populations upstream of Libby Dam. MFWP last investigated burbot distribution and relative abundance in the reservoir during the winter of 2005/2006 (Dunnigan et al. 2007). We used baited hoop traps to capture burbot in the reservoir during the winter months (November – May). Hoop traps measured 2-feet in diameter and were approximately 6-8 feet in length with $\frac{3}{4}$ inch net mesh baited with cut bait (usually kokanee, depending upon availability). Sash weights attached to the cod end of each hoop trap securely positioned the trap on the bottom. Traps were fished for two days at depths ranging from 7 – 28.3 m (mean = 16.7 m) from November 28, 2007 to May 21, 2008. We divided the Montana portion of the reservoir into 5 reaches that each included 3 tributaries (Table 1). During each two-day trapping session, we fished three traps within the general vicinity of the 3 tributary confluences within a given reach. We randomly determined which reaches of the reservoir were trapped using a random number table. Captured burbot were enumerated, examined for a PIT (passive integrated transponder) tag, measured, PIT tagged with a 134 (ISO) KHz PIT tag if not previously tagged, and released. PIT tags were inserted with an 8-gauge hypodermic needle into the musculature of the left operculum.

We used general linear regression of effort-adjusted catch on indicators for reach, indicators for sites within reach, and date on which traps were set. Effort was measured in days (the difference between date and time that the trap was set and the time the trap was checked). The analysis procedures are described in the Appendix A.

Table 1. Reaches, tributaries ,and river mile locations on Libby Reservoir trapped for burbot during the 2007/2008 season.

Reach	Tributary	River Mile	Side of Reservoir
1	Canyon Creek	222.5	East
1	Cripple Horse Creek	226.8	East
1	Warland Creek	227.8	East
2	Barron Creek	228.7	West
2	Bristow Creek	231.5	West
2	Five Mile Creek	233	East
3	Ten Mile Creek	235.8	East
3	Parsnip Creek	243.8	West
3	McGuire Creek	246.8	East
4	Big Creek	248.7	West
4	Sutton Creek	250.3	East
4	Pinkham Creek	257	East
5	Tobacco River	264	East
5	Dodge Creek	264.1	West
5	Young Creek	268.4	West

Juvenile Salmonid Population Estimates

MFWP conducted juvenile salmonid population estimates on Sinclair, Therriault, Young, Libby, Grave, and Pipe creeks in 2006, as part of an effort to monitor long-term trends in juvenile salmonid abundance, size distribution and species composition associated with past or future stream restoration efforts. 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.

Based on captures made during the first two passes, if P was ≥ 0.7 , a third pass was conducted. Population estimates were performed for fish ≥ 75 mm, consistency 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 2005 follows for each stream.

Therriault Creek

We established three monitoring sections in Therriault Creek for juvenile salmonid trend analyses (Hoffman et al. 2002). Section one began at the Highway 93 culvert and extended 82 m upstream. Section 2 began at the first culvert above highway 93 and extended 120 m downstream. The property is privately owned and is within the Therriault Creek Restoration Project that was finalized in the spring of 2005. Section 3 extends from the second culvert above highway 93 downstream for 131 m. This section is moderately stable and is 400 m upstream from the highly entrenched reach of Therriault Creek, and is located upstream of the stream restoration project. Sections one and three are intended as control sites.

Grave Creek

We established a representative sampling reach on Grave Creek to perform population estimates. The shocking section begins at the Vukonich Bridge and extends downstream 1,000 feet to the lower boundary of the Demonstration Project. 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 canoe 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. This section of Grave Creek was sampled via electrofishing in 2003-2006. However, sampling in 2002 was limited to snorkel observations due to the presence of >2,000 adult kokanee salmon in the monitoring section. Two observers moved slowly upstream enumerating trout estimated to be ≥ 75 mm total length.

Young Creek

MFWP previously established five monitoring sections in Young Creek to assess trends in juvenile salmonid abundance. These five sections include the following:

- Section 1: Tooley Lake Section (Sec.23 T37N,R28W).
- Section 4: Dodge Creek Road #303, upstream from bridge (Sec. 18 T37N,R28W).
- Section 5: State Lands Restoration Project (NE $\frac{1}{4}$ of Section 16, T37 N, R28W).

We conducted population estimates on Sections 1, 4 and 5 in 2006.

Libby Creek

MFWP personnel collected fish population information in six sites on Libby Creek in 2006. We sampled Section 1 using a Coleman canoe outfitted with a mobile electrode. The system consisted of a Cofelt model VVP-15 rectifier powered by a 4000 watt generator. The other sections were sampled with a two Smith Root backpack electrofishers. The sections sampled in 2006 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 Restoration Project.
- Section 4: is a 201 m long reach located downstream of the lower Cleveland property, and is intended to serve as a control site for the lower Cleveland Stream Restoration Project.
- Section 5: is a 143 m long reach located upstream of the lower Cleveland property upstream of the bridge on Forest Rd. number 231, and is intended to serve as a control site for the lower Cleveland Stream Restoration Project.
- Section 6: is a 172 m long reach near the confluence of Midas Creek located within the lower Cleveland Phase II Stream Restoration Project.

Pipe Creek

MFWP 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 with the Kootenai River. This section was established to collect baseline biological data prior to a scheduled stream restoration project on lower Pipe Creek. This section has been sampled annually since 2001.

Libby Reservoir Gillnet Monitoring

MFWP 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). This report focuses on the period 1988 through 2006, but the entire database (1975 through 2006) was occasionally used 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 to 14 ganged nets from 2000 to present. 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 as either age 0+ or age 1+. These fish were also marked with adipose fin clips in recent years prior to release to facilitate 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), redbreast shiner (RSS), largescale sucker (CSU), longnose sucker (FSU), and yellow perch (YP).

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

- 1) Spring (April - June): The reservoir was being refilled, surface water temperatures increased to 9 - 13°C.
- 2) Fall (September - October): Drafting of the reservoir began, surface water temperature decreased to 13 - 17°C.

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

Fourteen to twenty-eight floating (ganged) and one or two single, sinking nets were set in the fall in the 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 were not included in this report because net placement was not standardized. Nets were set perpendicular from the shoreline in the afternoon and were retrieved before noon the following day. All fish were removed from the nets and identified, followed by collection of length, weight, sex and maturity data. Scales and a limited number of otoliths were collected for age and growth analysis. When large gamefish (Kamloops rainbow, cutthroat, bull trout or burbot) were captured alive, only a length was recorded prior to release.

Libby Reservoir Zooplankton Monitoring

MFWP has collected zooplankton from Libby Reservoir since 1983 in an attempt to relate changes in density and structure of the community to parameters of other aquatic communities, and 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, 153 μ 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 (Kruskal-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 results 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 in 1990, 1) We only collected zooplankton samples when depth was at least 10 m, and 2) 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 2007.

Results

Burbot Monitoring Below Libby Dam

The burbot catch in our hoop traps below Libby Dam has declined precipitously since 1996/1997 (Figure 12). During the 2007/2008 trapping season we caught two burbot below Libby Dam after fishing a total of 188 trap days. This catch represents the third lowest catch rate during the period of record. The highest catch rates occurred in 1995-96 and 1996-97; these years correspond with higher than normal snow-pack, and perhaps greater reservoir drafting, and higher burbot entrainment through Libby Dam. The mean annual catch rate since the 1995/1996 trapping season was 0.480 burbot per trap day. However, the catch rates since then have exhibited a significant negative trend ($r^2 = 0.601$; $p = 0.002$; Figure 2). This relationship was not improved using an exponential fit ($r^2 = 0.343$; $p < 0.05$; Figure 2), which was contrary to the previous two of three years (Dunnigan et al. 2007 and 2008).

Figure 2. Total catch per effort (burbot per trap day) of baited hoop traps in the stilling basin downstream of Libby Dam 1994/1995 through 2007/2008. The data were fit with linear regression for all years and with an exponential model for 1995/1996 – 2007/2008. The traps were baited with kokanee salmon and fished during December and February.

Burbot Monitoring in Libby Reservoir

We fished a total of 39 sessions (single randomized reach with 3 sites per reach and 3 traps per site) for a total of approximately 702 trap-days. We captured a total of 39 burbot during the trapping season, having a mean total length of 573.2 mm (range 415-803 mm). Burbot mean total length did not differ significantly by reach or site ($p = 0.09$ and 0.43 ; respectively). However, the power of these tests was relatively low (0.58 and 0.44 ; respectively).

The analysis of variation of the catch-per-unit-effort is summarized in Table 2., adjusted reach means are presented in Table 3., and reach-difference t-tests and associated p values are presented in Table 4. The overall mean catch-per-unit-effort for the season was 0.057 burbot/trap/day (Table 3).

The effort-adjusted catch per trap decreased over date of trap setting and this decrease did not significantly differ among reaches. While the 5% significance level for this negative trend was not quite attained ($p = 0.0580$, Table 2), we still adjusted the effort-adjusted catch for date of trap setting because not all reaches were equally distributed over the trapping date domain (see Appendix B.). The effort-adjusted mean catch and the associated mean-catch/unit-effort were calibrated to the mean date³ of trap setting over all sites using the common negative trend coefficient⁴ between effort-adjusted catch and date of trap setting.

As can be seen from Table 4., Reach 3 mean catch/trap is significantly lower than those of Reaches 4 and 5. Since significant levels for Reach 4 compared to both Reaches 1 and 2 and for Reach 3 compared to Reach 1 reached the 10% level, and since Reaches 1 and 2 were the lowest reaches in the reservoir and since Reaches 4 and 5 were highest reaches in the reservoir, the estimates from Reaches 1 and 2 were pooled into a lower-reach grouping, and Reaches 4 and 5 were pooled into an upper-reach grouping. Reach 3 is between these two groupings and is referred to as a middle-reach. Table 5 presents the means of these grouping, and Table 6 presents the t-values and p-values associated with the differences between these groupings. As can be seen from Table 6, all of the three groups' means significantly differ at the 5% level, with the upper-reach mean being the largest, the lower-reach mean being the second largest, and the middle-reach mean grouping being the lowest

³ Feb 2, 2008, Julian Date = 52, a scaled Julian Date actually used in the regression which included a fraction of a day since time that the trap was set was also included in the estimate. The scaling was done to avoid negative numbers (there was trapping in the latter months of 2007) since a curvilinear fit was performed. The curvilinear test is not included since higher order terms of the scaled Julian Date did not improve the fit.

⁴ Separate trend fits were made for each reach separately but trend coefficients were not significantly different from the common trend fit ($p = 0.29$, Table 2); therefore the calibration based on the common trend was justified.

(Table 5). This is consistent with the means of the five individual-reach means (Table 3), Reaches 4 and 5 having the highest means and Reaches 1 and 2 the next highest and Reach 3 having a substantially lower mean than any of the other 4.

Table 2. Analysis of variation* of burbot catch/trap adjusted for effort (days trapped).

Source	Deviance**	Degrees of Freedom	Mean Deviance	F-Ratio	Type 1 Error (P)
Among Sites within Reach	11.24	10	1.12	1.43	0.1669
Response to Date to Trap Setting	2.85	1	2.85	3.62	0.0580
Among Reach's Date Responses	3.9	4	0.98	1.24	0.2947
Error	260.78	331	0.79		

* General linear regression of $\ln(\text{catch}/\text{trap})$ based on Poisson Distribution (refer to Appendix A).

** Analogous to sums of squares in analysis of variance (least squares regression).

Table 3. Burbot mean catch/trap and Catch-per-Unit-Effort (CPUE = Catch/Trap/Effort) for five reaches on Libby Reservoir.

Burbot Catch [retransform from Poisson linear fit of $\ln(\text{Catch})$ adjusted for date/time set as covariate and $\ln(\text{effort as offset})$]			Burbot Catch per Unit Effort (CPUE)		Effective Effort/Trap
Reach	Catch/Trap	SE	CPUE	SE	
1	0.0737	0.0239	0.0386	0.0125	1.91
2	0.1407	0.0423	0.0709	0.0213	1.98
3	0.0615	0.0229	0.0324	0.0121	1.90
4	0.1647	0.0485	0.0840	0.0247	1.96
5	0.1171	0.389	0.0597	0.0199	1.96
Mean*	0.1111		0.0569		1.94

*Weighted by number of trapping events: 81 each for reaches 1 and 2, and 63 each for reaches 3, 4, and 5.

Table 4. Statistical test (t-test) for differences in burbot catch/trap among five reaches in Libby Reservoir. P-values are listed in parentheses.

	Reach 1	Reach 2	Reach 3	Reach 4
Reach 2	<u>-1.91 (0.0568)</u>			
Reach 3	0.45 (0.6545)	2.30 (0.0218)		
Reach 4	-2.32 (0.0212)	-0.56 (0.5725)	-2.65 (0.0085)	
Reach 5	-1.28 (0.1999)	0.62 (0.5330)	-1.71 (0.0887)	1.10 (0.2702)

*Values in bold face type are significant at the 1% level and underlined values are significant at the 10% level.

Table 5. Burbot mean catch/trap and Catch-per-Unit-Effort (CPUE = Catch/Trap/Effort) for the lower, middle and upper reaches on Libby Reservoir.

	Burbot Catch [retransform from Poisson linear fit of ln(Catch) adjusted for date/time set as covariate and ln(effort as offset)]		Burbot Catch per Unit Effort (CPUE)		Effective Effort/Trap
Reach	Catch/Trap	SE	CPUE	SE	
Lower (1&2)	0.1072	0.0284	0.0275	0.0073	3.89
Middle (3)	0.0615	0.0229	0.0324	0.0121	1.90
Upper (4&5)	0.1409	0.0382	0.0359	0.0097	3.92
Mean*	0.3248				9.71

*Weighted by number of trapping events: 162 each for lower, 63 for middle, and 126 for upper reaches.

Table 6. Statistical test (t-test) for differences in burbot catch/trap among the lower, middle and upper reaches in Libby Reservoir. P-values are listed in parentheses.

	Lower Reaches (1&2)	Midde Reach (3)
Middle Reach (3)	1.44 (0.1520)	
Upper Reaches (4&5)	-1.38 (0.1691)	-2.36 (0.0187)

*Values in bold face type are significant at the 5% level.

Juvenile Salmonid Population Estimates

Therriault Creek

Section 1 on Therriault Creek is located downstream of the Therriault Creek Restoration Project Area, and is used as a control site to compare pre- and post-restoration fish populations. MFWP has sampled this site annually since 1997, with the exception that this site was not sampled in 2000-2002. Rainbow and brook trout have been observed at this site every year it has been sampled. Rainbow trout abundance in Section 1 of Therriault Creek has decreased from 1997-2006, although this trend has not differed significantly from a stable population ($r^2 = 0.17$; $p = 0.310$; Figure 3; Table A1). The mean abundance of rainbow trout during the period of record was 97.3 fish per 1,000 feet, with the observed abundance in 2007 (53.5 fish per 1,000 feet) the lowest on record, and 45.0% lower than the annual mean. The trend in brook trout abundance for this section has also been variable during the past several years, and has not differed significantly from a stable population ($r^2 = 0.283$; $p = 0.18$; Figure 3; Table A1), and has averaged 62.6 brook trout per 1,000 feet, with the observed abundance of brook trout in 2007 (100.6 brook trout per 1,000 feet) approximately 1.6 fold higher than the average of record. Juvenile bull trout were only detected at this site during the past 5 years, with abundance being highest in 2004 (92.1 bull trout per 1,000 feet). Bull trout abundance in 2007 was substantially lower with an estimated 28.3 bull trout per 1,000 feet, but higher than the mean over the observation period (23.6 bull trout per 1,000 feet). Despite the recent increase in bull trout abundance at this site over the past 4 years, the trend does not differ significantly from a stable population ($r^2 = 0.31$; $p = 0.15$; Figure 3).

Section 2 on Therriault Creek lies within the Therriault Creek Restoration Project area and was sampled in 1997-1999, 2001, and 2003-2007. The data we collected in 2007 represented the third year after project completion, and was used to compare to data collected prior to project implementation (1997-2004). We observed rainbow, brook and bull trout at this site every year data was collected (Table A1). We used linear regression to evaluate population trends for each of these three species, but did not detect significant trends (Figure 4; $p > 0.20$ for all species). We observed approximately 10.9, 40.1 and 1.8 rainbow, brook, and bull trout, respectively per 1,000 feet within the project area in 2007. The mean abundance of rainbow trout we observed within this section after implementation (2005-2007) was 28.3 rainbow trout per 1,000 feet, which was 63.9% lower than the mean abundance prior to project completion (Figure 5; pre-project mean = 78.4 fish per 1,000 feet). Brook trout abundance at the restoration site also decreased after project implementation by an average of 10.0% (Figure 6). Brook trout abundance slightly decreased within the project area (Section 2) after project completion decreasing from 69.5 to 62.5 fish per 1,000 feet after the project completion. The variation in bull trout abundance over time was higher than the variation in rainbow or brook trout abundance at all three sections (Figure 7). Bull trout abundance within the project reach after implementation decreased compared to the pre-project levels by a factor of 7.8 fold. Given the variability in bull trout abundance in the pre-implementation years (1997-2004) detecting a significant difference as a result of the restoration project is difficult (see below).

Figure 3. Rainbow trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 1 monitoring site from 1997-1999 and 2003-2007 collected by backpack electrofishing. The error bars represent 95% confidence intervals.

Figure 4. Rainbow trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 2 monitoring site from 1997-1999, 2001 and 2003-2007 collected by backpack electrofishing. The error bars represent 95% confidence intervals.

Section 3 on Therriault Creek is located upstream of the Therriault Creek Restoration Project area and was sampled in 1997-1999, and 2003-2007 (Table A1), and is used as a control site to compare pre- and post-restoration fish populations. We observed rainbow and brook trout at this site each year, but bull trout were only observed in 2003-2007. The trends of rainbow, brook, and bull trout abundance have not differed significantly from a stable population since 1997 ($p > 0.05$; Figure 5; Table A1). Section 3 had the highest abundance of bull trout of any of the three sections sampled in 2006 on Therriault Creek, where we observed an estimated 46.3 bull trout per 1,000 feet.

We compared the abundance of rainbow, brook and bull trout within the restoration project area (Section 2) to control sites located below and above the restoration project (Sections 1 and 3, respectively), using the Before/After/Control/Impact (BACI) statistical design. This design uses a paired t-test to assess differences between the Control and Treatment (impact) sites before and after project implementation. Because the test is only capable of using a single control site, we conducted the test for each of the control sites (Sites 1 and 3) and for each species of fish (rainbow, brook and bull trout), for a total of six individual tests (Tables 7 and 8). The mean difference (Control minus Treatment) in rainbow, brook and bull trout abundance between Sections 1 and 2 decreased in each comparison, suggesting that the abundance of each species within the project area decreased after project implementation (Table 7). However, none of comparisons were significant using the more conservative two-tailed test ($p > 0.05$; Table 7). This trend was nearly the opposite scenario when we used Section 3 as a control and compared it to Section 2. Rainbow and brook trout abundance in Section 2 increased relative to Section 3 before and after the project was implemented (Table 8). However, these comparisons were not significant ($p > 0.05$; Table 8). Bull trout abundance in Section 2 relative to Section 3 significantly decreased the three years following project completion ($p = 0.034$; Table 8). Given these results, it seems as though additional data is needed to determine if fish abundance has significantly changed within the project area.

Figure 5. Rainbow trout, bull trout and brook trout densities (fish per 1,000 feet) within the Therriault Creek Section 3 monitoring site from 1997-1999 and 2003-2007 collected by backpack electrofishing. The error bars represent 95% confidence intervals.

Figure 6. Rainbow trout densities (fish per 1,000 feet) within the Therriault Creek Sections 1 and 3 represent control sites located downstream and upstream, respectively of the treatment section (Section 2). Data that was collected from 1997-2004 represents pre-project, and data collected in 2005-2007 represents post-project results. Depletion estimates were calculated from backpack electrofishing. The error bars represent 95% confidence intervals.

Figure 7. Brook trout densities (fish per 1,000 feet) within the Therriault Creek Sections 1 and 3 represent control sites located downstream and upstream, respectively of the treatment section (Section 2). Data that was collected from 1997-2004 represents pre-project, and data collected in 2005-2007 represents post-project results. Depletion estimates were calculated from backpack electrofishing. The error bars represent 95% confidence intervals.

Figure 8. Bull trout densities (fish per 1,000 feet) within the Therriault Creek Sections 1 and 3 represent control sites located downstream and upstream, respectively of the treatment section (Section 2). Data that was collected from 1997-2004 represents pre-project, and data collected in 2005-2007 represents post-project results. Depletion estimates were calculated from backpack electrofishing. The error bars represent 95% confidence intervals.

Table 7. Results from a paired t-test (BACI) of the differences between control (Section 1) and treatment (Section 2) on Therriault Creek before and after the restoration project was completed in 2005.

	Rainbow Trout		Brook Trout		Bull Trout	
	Before	After	Before	After	Before	After
Mean Difference (Control-Treatment)	24.2	64.9	-23.1	31.0	-10.1	25.3
Variance	2038.8	384.8	918.9	937.9	1157.9	354.4
Sample size (n)	5	3	5	3	5	3
P-value (1-tailed)	0.064		0.036		0.054	
P-value (2-tailed)	0.129		0.72		0.107	

Table 8. Results from a paired t-test (BACI) of the differences between control (Section 3) and treatment (Section 2) on Therriault Creek before and after the restoration project was completed in 2005.

	Rainbow Trout		Brook Trout		Bull Trout	
	Before	After	Before	After	Before	After
Mean Difference (Control-Treatment)	45.5	18.6	7.8	-1.1	-28.1	18.6
Variance	1916.9	1789.3	920.3	859.9	365.9	435.6
Sample size (n)	5	3	5	3	5	3
P-value (1-tailed)	0.219		0.35		0.017	
P-value (2-tailed)	0.438		0.70		0.034	

Grave Creek

Juvenile salmonid monitoring within the Grave Creek Demonstration Project had two primary objectives, to determine fish population trends through time and to evaluate the fish community response to the restoration activities completed during the fall of 2001 (Grave Creek Demonstration Project). Bull trout were the most abundant fish species present at this site for the fifth consecutive year (Table A2). We compared mean fish abundance (by species) for pre (2000-2001) and post (2002-2007) restoration projects using t-tests (two-tailed tests; Figure 19). However, the variability in pre- and post-project fish abundance estimates is high (Figure 9 and 10), and sampling methodology differed between years. These factors reduced our ability to distinguish statistical differences in abundance before and after project completion. In 2007, we were unable to determine if any of the four trout species present at this site had significantly ($p < 0.05$; two-tailed test) increased after project completion. Rainbow trout abundance increased substantially from 9.0 to 31.3 rainbow trout per 1,000 feet after project construction, and the difference was nearly significant (Figure 9; $p = 0.165$; two-tailed test). Bull trout abundance after project completion also increased over three fold from 17.0 to 60.8 bull trout per 1,000 feet after project completion, but the difference was insufficient to declare the difference significant ($p = 0.122$; two-tailed test). However brook trout and westslope cutthroat trout abundance were nearly identical before and after project completion, with the mean differences less than 1.2 fish per 1,000 feet ($p > 0.89$; Figure 9). We used linear regression to assess whether there was a temporal trend in abundance for the four fish species at this site (Figure 10). Rainbow, cutthroat, and brook trout population trends did not differ significantly from a stable population through time (Figure 10; $p > 0.05$). Bull trout abundance at this site exhibited a positive and significant trend during the period of record (Figure 20; $r^2 = 0.646$; $p = 0.016$). We were unable to use the more powerful Before/After/Control/Impact (BACI) statistical design to compare fish abundances at this site due the lack of an adequate control site nearby in Grave Creek.

Figure 9. Mean cutthroat, rainbow, brook, and bull trout densities (fish per 1,000 feet) within the Grave Creek Demonstration Project area prior to (2002-2001) and after (2002-2007) the completion of the Grave Creek Demonstration Restoration Project. Data collected during 2000 and 2001 represent pre-project implementation fish abundances and were collected using single pass electrofishing. Fish abundance data collected in 2002 represents post-project implementation fish abundances and was collected via snorkel counts. Error bars represent 95% confidence intervals.

Figure 10. Cutthroat, rainbow, brook, and bull trout abundance estimates (fish per 1,000 feet) and linear regression trend analyses within the Grave Creek Demonstration Project monitoring site from 2000-2006 collected by backpack electrofishing. The 2000 and 2001 data were collected using single pass electrofishing, the data collected in 2002 was collected via snorkel counts, and the 2003- 2007 data was collected using multiple pass electrofishing. Error bars represent 95% confidence intervals.

Young Creek

The Young Creek Section 1 juvenile monitoring site was sampled consecutively from 1997-2007, with the exception of 2000 and 2003 (Table A3), and is intended to serve as a control section relative to the restoration project area (Section 5). There was no evidence of linear trends in abundance for cutthroat, rainbow or brook trout from 1997-2007 ($p > 0.1$; Figure 11). Cutthroat trout have been the most abundant species of fish at this site since 1997, with the exception of 2005, when brook trout were slightly more abundant (Figure 11). Cutthroat trout abundance peaked at this site in 1999 when we observed 139 cutthroat trout per 1,000 feet. In 2007, we observed approximately 46.6 cutthroat trout per 1,000 feet, which was approximately 25% lower than the average over the observation period of 62.5 fish per 1,000 feet (Figure 11). In 2007, brook trout were less than half as abundant as cutthroat trout at this site, with an estimated abundance of 20.2 brook trout per 1,000 feet (Figure 21), which was 58.7% lower than the long-term average since 1996. Bull trout were first observed at Section 1 in 2004-2006, but not observed in 2007. Densities been relatively low, and ranged from 2 to 10 fish per 1,000 feet (Figure 11). Both cutthroat and brook trout abundances decreased in 2007 compared to those observed in 2006.

The Young Creek Section 4 juvenile monitoring site was sampled consecutively from 1996-2007, with the exception of 2000 and 2003 (Table A3). Westslope cutthroat trout dominated the fish community at this sampling location during all years, with cutthroat trout densities averaging approximately 37 fold higher than brook trout densities. Over the period of record for this site, cutthroat trout densities averaged 243.2 fish per 1,000 feet, and brook trout densities averaged 6.6 fish per 1,000 feet. We were unable to distinguish the trend in westslope cutthroat trout abundance from a stable population ($r^2 = 0.01$; $p = 0.79$; Figure 12). However, brook trout abundance has significantly increased at this site over time ($r^2 = 0.535$; $p = 0.016$; Figure 12). Brook trout have increased an average of 1.5 fish per 1,000 feet at this site since 1996. In 2007, we observed a single bull trout at this location, which represented the first ever in this section of Young Creek. Both cutthroat and brook trout abundances decreased in 2007 compared to those observed in 2006.

The Young Creek Section 5 lies entirely within the stream restoration project completed on State land in the fall of 2003. Therefore, all data collected through 2003 represents data gathered prior to the restoration project completion. Like the other two sections we sampled on Young Creek in 2007, cutthroat and brook trout abundances decreased in 2007 compared to those observed in 2006. We observed an estimated equal number of cutthroat trout and brook trout at this site in 2007 (56.9 fish per 1,000 feet; Figure 13). Bull trout have exhibited a significantly increasing trend in abundance in Section 5 of Young Creek since 1998 ($r^2 = 0.419$, $p = 0.043$; Figure 13). However, the magnitude of the average annual change has been relatively small, averaging only 0.26 bull trout per 1,000 feet per year. Brook trout abundance has also increased at this site since 1998. However, this trend was not significantly different than a stable population ($r^2 = 0.23$; $p = 0.15$; Figure 13). Cutthroat trout abundance at this site has significantly declined over time ($r^2 = 0.434$; $p = 0.038$; Figure 13) at an average rate of 14.2 fish per 1,000 feet per year. Annual mean abundance estimates for cutthroat, brook and bull trout have averaged 165.6, 59.1 and 1.0

fish per 1,000 feet for each species respectively (Table A3). We compared mean fish abundance (by species) for pre (1998-2003) and post (2004-2007) restoration projects using t-tests (two-tailed tests; Figure 14). Abundance estimates for cutthroat trout significantly ($p = 0.034$) decreased after project completion from an average of 199.5 fish per 1,000 feet prior to restoration to 114.6 fish per 1,000 feet in 2004-2007 (Figure 14). Brook trout and bull trout both exhibited opposite trends. The abundance of brook trout significantly increased after the restoration project was completed from a mean of 39.8 fish per 1,000 feet before the project to 88.0 fish per 1,000 feet after the project ($p = 0.027$, for a 2-tailed test; Figure 14). Bull trout abundance significantly increased after the project, from a mean of 0.3 to 2.1 bull trout per 1,000 feet after the project ($p = 0.012$, for a 2-tailed test; Figure 14).

We compared the abundance of cutthroat and brook trout within the restoration project area (Section 5) to control sites located below (Section 1) and above (Section 4) the restoration project, using the Before/After/Control/Impact (BACI) statistical design. This design uses a paired t-test to assess differences between the Control and Treatment (impact) sites before and after project implementation. Because the test is only capable of using a single control site, we conducted the test for each of the control sites (Sections 1 and 4) and for each species of fish (cutthroat, brook and bull trout), for a total of six individual tests (Tables 9-11). The mean difference (Control minus Treatment) in cutthroat trout abundance between Sections 1 and 5 significantly ($p = 0.016$) decreased by approximately 62% four years after project completion (Table 6). An opposite trend was observed for brook trout at these two sites. The mean difference in brook trout abundance between Sections 1 and 5 significantly ($p = 0.015$) increased by approximately three fold before and after project completion (Table 9). The mean difference in bull trout abundance between Sections 1 and 5 decreased after the restoration work in Section 5, but this difference was not significant ($p = 0.483$; Table 9). When we repeated the tests using Section 4 as the control section, the trends were similar, with the difference in brook trout abundance increasing by 73%, and the difference in cutthroat trout abundance increasing by approximately 1.3 fold within the restoration reach. However, we weren't able to declare these differences significant ($p > 0.05$; Table 10). The mean difference in bull trout abundance between Sections 4 and 5 increased by a factor of 3.5 after the restoration work in Section 5, but the difference was not significant ($p = 0.169$; Table 10).

We also used a factorial analysis of variance to assess differences between sections and treatments within Young Creek. In this analysis we used three sections (Sections 1, 4 and 5) and four treatments. The treatments included; pre-restoration within the two control sections (Sections 1 and 4 combined), post-restoration within the two control sections combined, pre-restoration within the treatment section, and post-restoration within the treatment section. We conducted a separate analysis for cutthroat, brook and bull trout. The results were similar to the BACI design reported above. Brook trout abundance in Section 5 after the restoration was significantly higher than Section 5 before restoration and Sections 1 and 4 (combined) before or after restoration ($p < 0.05$; Table 11), but brook trout abundance within Section 5 before restoration was not significantly higher than brook trout abundance in Sections 1 and 4 (combined) before or after restoration (Table 11). At the overall Section level, brook trout abundance was significantly higher in Sections 1 and 5 than Section 4 ($p <$

0.05; Table 11). The interaction term for brook trout abundance between section and treatment was nearly significant ($p=0.082$; Table 11). Cutthroat trout abundance differed significantly between sections ($p < 0.001$; Table 12). Cutthroat trout abundance in Sections 4 and 5 were significantly higher than those observed in Section 1 ($p < 0.05$; Table 12), and although the mean cutthroat trout in Sections 4 was higher than in Section 5, the difference was not significant ($p=0.092$; Table 12). We were unable to detect any significant differences in cutthroat trout abundance between any of the four treatments. However, the power of this post hoc test was only 0.276. The interaction term for cutthroat trout abundance between section and treatment was not significant ($p=0.692$; Table 12), but the power of this test was very low (0.067). Bull trout abundance differed significantly at the Section and Treatment level in Young Creek (Table 13). Bull trout abundance significantly increased in the project reach after restoration by a mean difference of 1.7 bull trout per 1,000 feet. Bull trout abundance within the project section after restoration was significantly higher than Sections 1 and 4 before restoration ($p = 0.001$; Table 13), but not significantly higher than Sections 1 and 4 after restoration ($p = 0.639$; Table 13). The interaction term for bull trout abundance between section and treatment was significant ($p = 0.002$; Table 13).

Figure 11. Cutthroat, rainbow, brook and bull trout densities (fish per 1,000 feet) within the Young Creek Section 1 monitoring site from 1997-2007, with the exception of 2003. Data was collected by backpack electrofishing. Error bars represent 95% confidence intervals.

Figure 12. Cutthroat trout and brook trout densities (fish per 1,000 feet) within the Young Creek Section 4 monitoring site from 1996-2007, with the exception of 2000 and 2003. Data was collected by backpack electrofishing. Error bars represent 95% confidence intervals.

Figure 13. Cutthroat, brook and bull trout densities (fish per 1,000 feet) within the Young Creek Section 5 monitoring site from 1997-2007 collected by backpack electrofishing. The data presented for 2004-2007 represent post restoration data. The error bars represent 95% confidence intervals.

Figure 14. Cutthroat, brook and bull trout densities (fish per 1,000 feet) within the Young Creek Section 5 (State Lands Restoration Project Area), comparing annual mean pre-project (1998-2003) data and post-project (2004-2007) using mobile electrofishing gear. Comparisons were made using a 2-tailed t-test. Error bars represent 95% confidence intervals.

Table 9. Results from a paired t-test (BACI) of the differences between control (Section 1) and treatment (Section 5) on Young Creek before and after the restoration project was completed in 2003.

	Cutthroat Trout		Brook Trout		Bull Trout	
	Before	After	Before	After	Before	After
Mean Difference (Control-Treatment)	-143.8	-54.3	25.8	-52.3	-0.5	1.4
Variance	1812.3	1124.3	1196.3	275.8	1.0	22.2
Sample size (n)	4	4	4	4	4	4
P-value (1-tailed)	0.008		0.007		0.241	
P-value (2-tailed)	0.016		0.015		0.483	

Table 10. Results from a paired t-test (BACI) of the differences between control (Section 4) and treatment (Section 5) on Young Creek before and after the restoration project was completed in 2003.

	Cutthroat Trout		Brook Trout		Bull Trout	
	Before	After	Before	After	Before	After
Mean Difference (Control-Treatment)	62.5	143.9	-43	-74.6	-0.5	-1.76
Variance	15321.7	1929.4	324.7	1393.1	1.0	1.59
Sample size (n)	4	4	4	4	4	4
P-value (1-tailed)	0.141		0.101		0.084	
P-value (2-tailed)	0.283		0.201		0.169	

Table 11. Results from a factorial analysis of variance of brook trout abundance in Young Creek for the two control sites (Sections 1 and 4) and the treatment site (Section 5) before and after the stream restoration project was completed in 2004. Multiple comparisons for each Section and Treatment comparison are also presented. The overall R^2 for this model was 0.596.

Source	Comparison	Mean Difference (First – Second)	Significance
Overall Model			0.001
Intercept			<0.001
Section			0.016
	Section 1 and 4	39.6	0.013
	Section 1 and 5	-12.9	0.574
	Section 4 and 5	52.4	0.001
Treatment			0.025
	Section 5 Before/Sections 1 and 4 Before	11.4	0.832
	Section 5 Before/After	-48.1	0.048
	Section 5 Before/Sections 1 and 4 After	22.1	0.460
	Section 5 After/Sections 1 and 4 After	70.3	0.002
	Section 5 After/Sections 1 and 4 Before	59.6	0.005
	Sections 1 and 4 Before/Sections 1 and 4 After	10.7	0.841
Section*Treatment Interaction			0.082

Table 12. Results from a factorial analysis of variance of cutthroat trout abundance in Young Creek for the two control sites (Sections 1 and 4) and the treatment site (Section 5) before and after the stream restoration project was completed in 2004. Multiple comparisons for each Section and Treatment comparison are also presented. The overall R^2 for this model was 0.549.

Source	Comparison	Mean Difference (First – Second)	Significance
Overall Model			0.002
Intercept			<0.001
Section			<0.001
	Section 1 and 4	-180.7	<0.001
	Section 1 and 5	-103.1	0.024
	Section 4 and 5	77.7	0.092
Treatment			0.258
	Section 5 Before/Sections 1 and 4 Before	43.2	0.704
	Section 5 Before/After	84.8	0.362
	Section 5 Before/Sections 1 and 4 After	40.1	0.783
	Section 5 After/Sections 1 and 4 After	-44.8	0.790
	Section 5 After/Sections 1 and 4 Before	-41.6	0.802
	Sections 1 and 4 Before/Sections 1 and 4 After	-3.2	1.0
Section*Treatment Interaction			0.692

Table 13. Results from a factorial analysis of variance of bull trout abundance in Young Creek for the two control sites (Sections 1 and 4) and the treatment site (Section 5) before and after the stream restoration project was completed in 2004. Multiple comparisons for each Section and Treatment comparison are also presented. The overall R^2 for this model was 0.694.

Source	Comparison	Mean Difference (First – Second)	Significance
Overall Model			<0.001
Intercept			<0.001
Section			0.002
	Section 1 and 4	1.01	0.032
	Section 1 and 5	0.11	0.953
	Section 4 and 5	-0.90	0.045
Treatment			<0.001
	Section 5 Before/Sections 1 and 4 Before	0.33	0.835
	Section 5 Before/After	-1.74	0.012
	Section 5 Before/Sections 1 and 4 After	-1.15	0.065
	Section 5 After/Sections 1 and 4 After	0.58	0.639
	Section 5 After/Sections 1 and 4 Before	2.07	0.001
	Sections 1 and 4 Before/Sections 1 and 4 After	-1.49	0.004
Section*Treatment Interaction			0.002

Libby Creek

Section 1 of Libby Creek lies within the Libby Creek Demonstration Restoration Project Area, which was completed in the fall of 2001. This site has been sampled each consecutive year since 1998. Fish monitoring data collected from 1998 to 2001 represents the fish community prior to project implementation. Electrofishing conducted in 1999 and 2000 were limited to single pass catch estimates. We used three statistical methods to evaluate the fish community response to the restoration activities at this site. We used linear regression to investigate temporal trends for each species at this section. We also compared abundances before and after restoration at this site using a student's t-test, and finally, we compared species abundance within the restoration project area to a control site located upstream of the restoration project (Sections 2, see below), using the Before/After/Control/Impact (BACI) statistical design. This design uses a paired t-test to assess differences between the Control and Treatment (impact) sites before and after project implementation and has greater statistical power to detect changes over time than the student's t-test. We electrofished Section 1 of Libby Creek on August 1, 2007, and after completing two removal passes with the mobile electrofishing gear, we discovered that one of the electrode wires was maintaining intermittent contact which presumably lowered capture efficiency. The situation required repair, and dry forest conditions prompted a mandatory noon work window. Therefore, we placed block nets at the upper and lower ends of the section and returned the next day to complete the estimate. The resulting depletion estimates for rainbow and brook trout in 2007 are likely under estimates of the true population, and the confidence intervals are extraordinarily wide, due to the presumed low capture efficiency during the first two passes and the length of time between the second and third passes. Nevertheless, rainbow trout are the most abundant fish species within this section, with an estimated abundance of 179.0 fish per 1,000 feet. The estimate of brook trout abundance at this site proved to be especially problematic since the removal pattern for the three passes resulted in seven fish captured on each pass. Therefore, depletion estimate is was not possible, and the catch estimate should be viewed as a minimum number present. Brook trout were also observed within this section every year, but at much lower abundance than rainbow trout. Brook trout abundance has ranged from 5 fish per 1,000 feet in 2003 to 19.1 fish per 1,000 feet in 2007. Juvenile bull trout were only observed in this section in 2002 and 2005, with an estimated abundance of 3 and 0.9 fish per 1,000 feet, respectively. Rainbow trout exhibited significantly ($p < 0.05$) positive trends ($r^2 = 0.58$) in abundance at this site from 1998 to 2007 despite the technical difficulties (Figure 15). The minimum estimate of 19.1 brook trout per 1,000 feet at this section in 2007 resulted in a nearly significant positive trend for the species ($r^2 = 0.36$; $p = 0.067$). Despite the positive trend in abundance for both rainbow and brook trout at this site through time since 1998, we were not able to detect significant changes for either of these species before and after restoration using the two-tailed student's t-test. The abundance of rainbow trout at this section was more than twice as high after the restoration than before (Figure 16), but the difference was not significant (154.5 and 69.5 fish per 1,000 feet, respectively) ($p = 0.056$). Similarly, mean brook trout abundance at this site increased by approximately three fold after project completion (8.8 and 24.5 fish per 1,000 feet, respectively; Figure 16), but was not significant ($p = 0.12$). Bull trout were only observed at this section after the project was completed

during two of the five years, for a mean bull trout abundance of 0.65 fish per 1,000 feet. The pre and post difference in bull trout abundance was not significant ($p = 0.32$; two tailed test). Using the BACI statistical design, we were able to detect a significant increase in rainbow trout abundance since the project was completed (Table 14). The difference in rainbow trout abundance between the control (Section 2) and the project area approximately doubled after the project was completed (Table 14). However, even though the mean difference in brook trout and bull trout abundance also each approximately doubled between the control and treatment sections after the restoration activities, we were not able to declare either set of differences significant (Table 14). Our lack of power with these tests was likely limited by the fact that the control section was not sampled consecutively since 1998, as was the case with the Demonstration Section which resulted in eliminating corresponding years from the dataset from the Demonstration Section. The difficulties with the sampling gear experienced in 2007 also likely contributed to this situation by producing a low population estimate for brook trout in 2007.

Section 2 of Libby Creek was established and sampled primarily as a control site for the Libby Creek Demonstration Project. This site was sampled in 1998, 2001, and 2003-2007 (Table A4). Rainbow trout were substantially more abundant at this section than brook trout and bull trout during all years (Figure 17; Table A4). Rainbow trout abundance at this site has ranged from 203 fish per 1,000 feet in 1998 to a low of 76 fish per 1,000 feet in 2005. We observed an estimated 98.4 fish per 1,000 feet in 2007. There was a significant negative trend in rainbow trout abundance through time at this site ($r^2 = 0.74$; $p = 0.01$), for an average decline of approximately 11.5 fish per 1,000 feet per year. Rainbow trout abundance significantly decreased when compared over the pre (1998 and 2001) and post (2003-2007) implementation period for the Libby Creek Demonstration Project. Brook trout abundance at Section 2 exhibited a positive trend over the period of record. However, this trend did not differ significantly from a stable population ($r^2 = 0.45$; $p = 0.101$; Figure 17). Brook trout abundance increased when compared over the pre and post implementation period for the Libby Creek Demonstration Project, but not significantly so ($p = 0.12$; two-tailed test). Bull trout were observed in this section in four of the seven years sampled (1998, 2003, 2005, and 2006) (Figure 17; Table A4), and have exhibited a negative trend through the period of record, although the relationship was not significant ($r^2 = 0.36$; $p = 0.15$). Bull trout abundance decreased when compared over the pre and post implementation period for the Libby Creek Demonstration Project, although the difference was not significant ($p = 0.43$; two-tailed test).

Our estimates of rainbow trout abundance in Section 3 of Libby Creek were similar between 2000 and 2002 (Figure 18; Table A4), with no evidence that the population differed from a stable population ($p = 0.469$; $r^2 = 0.548$) during this period. These data represent conditions prior to completion of the upper Cleveland's Stream Restoration Project. However, our monitoring data after project construction showed a significantly lower number of rainbow trout at this site than previous years (mean abundance 79.8 and 168.3 fish per 1,000 feet, respectively; $p = 0.005$; two-tailed test; Figure 19). Rainbow trout abundance through the period of record has exhibited a significant negative trend ($r^2 = 0.83$; $p = 0.002$; Figure 18), with an average decrease of 19.5 fish per 1,000 feet per year. No brook trout

were observed at this site during the past seven years. We did not observe any juvenile bull in this section in 2007 (Figure 18). However, estimates of juvenile bull trout abundance before and after project implementation were similar (means = 6.0 and 3.4 fish per 1000 feet, respectively), and did not differ significantly ($p = 0.39$; two-tailed test; Figure 19). Similarly, we found no evidence of a trend in bull trout abundance through time at this site for the period of record ($r^2 = 0.32$; $p = 0.14$; Figure 18).

We established juvenile monitoring Sections 4, 5, and 6 on upper Libby Creek in 2004 to monitor the fish community response to the lower Cleveland Stream Restoration Project that we began implementing in the fall of 2005. Sections 4 and 5 serve as control sites and are located downstream and upstream of the proposed restoration project area, respectively. Section 6 is located within the Phase II project area implemented in the fall of 2006. Fish population data collected from 2004 through 2006 provide the baseline data for comparison after project implementation, and data collected in 2007 represent the first year of post-implementation data collection. Rainbow trout (presumed to be redband trout) dominated the fish community at all three sampling locations during all years (Table A4). However, rainbow trout abundance at each of the sections in 2007 decreased by 54.6% in Section 4 (Below), 44.0% in Section 6 (Within), and 24.8% in Section 5 (Above; Figure 20) compared to the three years previous. The decrease in abundance is presumable due to the rain-on-snow event that occurred in November 2006 within the upper Libby Creek watershed. Statistical comparisons between pre and post restoration were not possible due to a single year of post restoration data. Brook trout have not been observed in Section 5 during the any of the four years. However, brook trout abundance at the other two sections showed mixed results in 2007. Brook trout abundance within Section 6 (Within) in 2007 increased by approximately an order of magnitude increase when compared to the average abundance the previous three years (Figure 21). Even though overall abundance remained relatively low in 2007 (6.4 fish per 1,000 feet). Brook trout abundance within Section 4 (Below) showed a decrease of 39.0% in 2007 compared to the three previous years (Figure 21). Bull trout were encountered at relatively low abundance at all three sections during all four years of samplings (Figure 22), ranging from 0-7 fish per 1,000 feet within the three sections across years. Bull trout abundance decreased at each of the four sites in 2007, with the sharpest decrease occurring at Section 6 (Within; 100% decrease), followed by Section 4 (Below; 37.7%), and finally by 0.4% at Section 5 (Above; Figure 22).

Figure 15. Rainbow trout, brook trout, and bull trout densities (fish per 1,000 feet) within the Libby Creek Section 1 monitoring site 1998 through 2007 using mobile electrofishing gear. Upper 95% confidence intervals are represented by the error bars. The site was sampled using single pass electrofishing in 1999 and 2000.

Figure 16. Rainbow trout and brook trout densities (fish per 1,000 feet) within the Libby Creek Demonstration Project area, comparing annual mean pre-project (1998-2001) data and post-project (2002-2007) using mobile electrofishing gear. The error bars represent 95% confidence intervals.

Table 14. Results from a paired t-test (BACI) of the differences between control (Section 2) and treatment (Section 1) on Libby Creek before and after the Libby Creek Demonstration Restoration Project was completed in 2001.

	Rainbow Trout		Brook Trout		Bull Trout	
	Before	After	Before	After	Before	After
Mean Difference (Control-Treatment)	112.0	59.8	-3.5	-6.5	2.5	1.0
Variance	200.0	5125.3	40.5	243.3	12.5	1.0
Sample size (n)	2	5	2	5	2	5
P-value (1-tailed)	0.002		0.37		0.33	
P-value (2-tailed)	0.004		0.74		0.66

Figure 17. Rainbow trout, brook trout, and bull trout densities (fish per 1000 feet) within the Libby Creek Section 2 monitoring site sampled in 1998, 2001, 2003-2007 using a backpack electrofisher. The error bars represent 95% confidence intervals.

Figure 18. Rainbow trout and bull trout densities (fish per 1,000 feet) within the Libby Creek Section 3 monitoring site in 2000-2007 using a backpack electrofisher. The error bars represent 95% confidence intervals. This site is located within the upper Libby Creek restoration project area. The data from 2000-2002 represent pre-project trends of fish abundance, and the 2003-2007 data represent data after project completion.

Figure 19. Rainbow trout and bull trout densities (fish per 1,000 feet) within the Libby Creek Upper Cleveland's Stream Restoration Project area (Section 3), comparing annual mean pre-project (2000-2002) data and post-project (2003-2007) using mobile electrofishing gear. The error bars represent 95% confidence intervals.

Figure 20. Rainbow trout densities (fish per 1,000 feet) within the Libby Creek Sections 4-6 monitoring sites in 2004 – 2006 (pre-project), and 2007 (post project). These sites were sampled using a backpack electrofisher. The Error bars represent the 95% confidence intervals.

Figure 21. Brook trout densities (fish per 1,000 feet) within the Libby Creek Sections 4-6 monitoring sites in 2004 – 2006 (pre-project), and 2007 (post project). These sites were sampled using a backpack electrofisher. The Error bars represent the 95% confidence intervals.

Figure 22. Bull trout densities (fish per 1,000 feet) within the Libby Creek Sections 4-6 monitoring sites in 2004 – 2006 (pre-project), and 2007 (post project). These sites were sampled using a backpack electrofisher. The Error bars represent the 95% confidence intervals.

Pipe Creek

Rainbow trout were the most abundant fish species at the lower Pipe Creek Section during all years sampled (Table A5), with estimates ranging from a high of 73 fish per 1,000 feet in 2002 to our lowest estimate of 20.8 fish per 1,000 feet in 2005 (Figure 23; mean for all years = 46.1 fish per 1,000 feet). We observed an estimated 54.1 rainbow trout per 1,000 feet in 2006, which was 17.3% higher than the long term average. The trend in rainbow trout abundance has not differed significantly from a stable population since 2001 ($r^2 = 0.0011$; $p = 0.94$). Brook trout abundance has increased through time ($r^2 = 0.40$), but the increase has not been significant ($p = 0.13$). However brook trout abundance is generally about an order of magnitude less abundant at this site than rainbow trout, with estimates ranging from a high of 14.6 brook trout per 1,000 feet in 2006 to a low of 0 brook trout in 2001, and averaging 5.6 fish per 1,000 feet since 2001. Brook trout abundance at this site in 2007 was slightly less than the long term average (4.7 brook trout per 1,000 feet). Bull trout were first observed at this site in 2006, at an estimated 2.1 fish per 1,000 feet, and not observed in 2007.

Figure 23. Rainbow, brook and bull trout densities (fish per 1,000 feet) within the Pipe Creek monitoring site from 2001-2007. Fish were collected using a backpack electrofishing. The error bars represent 95% confidence intervals.

Libby Reservoir Gillnet Monitoring

We documented changes in the assemblage of fish species sampled in Libby Reservoir since impoundment. Kokanee salmon, Kamloops rainbow trout and yellow perch did not occur in the Kootenai River prior to impoundment but are now present. Kokanee were released into the reservoir from the Kootenay Trout Hatchery in British Columbia (Huston et al. 1984). Yellow perch may have dispersed into the reservoir from Murphy Lake (Huston et al. 1984). The British Columbia Ministry of Environment (BCMOE) first introduced Kamloops rainbow trout in 1985, and since 1988, MFWP annually stocked between 11,000 to 73,000 Duncan strain Kamloops rainbow trout directly into the reservoir (see below). 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 since the reservoir filled. Mountain whitefish, rainbow trout, westslope cutthroat trout and redbreasted sunfish were common in the Kootenai River before impoundment, but have decreased in abundance since impoundment.

Kokanee

Since the unintended introduction of fry from the Kootenay Trout Hatchery in British Columbia into Libby Reservoir in the early 1980s, kokanee have become the second or third most abundant fish captured during fall gillnetting. Fluctuations in catch have corresponded to the strength of various year classes (Hoffman et al. 2002), and have varied by year, with no apparent continuous trend in abundance (Figure 24). However, kokanee catch in the fall net series follows a general trend of decreasing abundance from 1988-1995 and an increasing trend in abundance from 1996-2007 (Figure 24), but specifically in 2007, the catch rate of kokanee in the fall Rexford nets series (0.8 fish per net) was the lowest since 1992. Average length of kokanee has varied among years. Average length and weight between 1988 and 2006 was 285.0 mm and 223.3 g respectively (Table 15). The maximum average size occurred in 1992 (350 mm, 411 g) when numbers were low and the minimum mean length was observed in 2005 when catch was relatively high (Table 15). This may be attributable to density dependant growth exhibited by kokanee (Skaar et al. 1996). We used multiple linear regression to determine which factors since 1997 correlate to the length of kokanee in the reservoir. Hoffman et al. (2002) observed that kokanee mean length and mean length of *Daphnia*, especially during August are correlated. We re-examined this relationship with the current data and added mean catch per net of kokanee during the fall netting series and found and found both covariates were significantly and positively correlated with kokanee mean length ($r^2 = 0.627$; $p < 0.02$).

Figure 24. Average catch per net of kokanee for fall floating (1988-2007) and spring sinking (1984-2008) gill nets in Libby Reservoir.

Table 15. Average length and weight of kokanee salmon captured in fall floating gillnets (Rexford and Canada Sites) in Libby Reservoir, 1988 through 2007.

YEAR	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Sample size (n)	2150	1259	517	624	250	111	291	380	132	88	76
Length (mm)	315.5	275	257.3	315.8	350	262.7	270.2	300.2	293.7	329.6	333.9
Weight (g)	289.1	137.2	158.4	327.3	411.3	162.3	191.7	261.6	234.5	363.2	322.0

Table 15. Continued

YEAR	1999	2000	2001	2002	2003	2004	2005	2006	2007	AVG.
Sample size (n)	200	342	120	357	263	194	320	163	118	398
Length (mm)	291.6	271.3	261.6	251.3	264.9	261.0	232.2	276.3	290.2	285.2
Weight (g)	229.6	185.6	161.6	152.2	175.5	159.2	117.4	202.5	237	224.0

Mountain Whitefish

Mountain whitefish are one of three native species that have exhibited the most significant decline in abundance since impoundment of the Kootenai River (Huston et al. 1984; Figure 25). A natural logarithm transformation provided the best fit to the sinking gillnet catch data for the period 1975 to 2007 (Figure 25; $r^2 = 0.451$, $p < 0.05$). The trend in catch data for mountain whitefish during the first 13 years after reservoir impoundment (1975-1988; mean catch = 3.5 fish per net) decreased annually, until it reached a significantly lower ($p < 0.001$; two-tailed test) equilibrium. However, since 1989 mountain whitefish catch rates have averaged 0.77 fish per net, with no evidence of an apparent trend ($r^2 = 0.02$; $p = 0.58$). We attribute the initial (1975-1988) mountain whitefish decline in Libby Reservoir to the loss of spawning habitat and rearing habitat that resulted from a conversion from a lotic to lentic environment through reservoir construction.

Rainbow and Westslope Cutthroat Trout

Rainbow trout and westslope cutthroat trout catch have both significantly declined since the impoundment of Libby Reservoir (Figure 25). Similarly to mountain whitefish, rainbow and westslope cutthroat trout gillnet catch data was best fit with linear regression using a natural logarithm transformation (Figure 25). Although both species exhibit similar declining trends in catch since 1975, rainbow trout catch per net since 1975 has declined more precipitously than cutthroat trout catch per net. Rainbow trout have exhibited two general trends since impoundment. The first trend showed a significant decline in abundance from 1975 to 1988 (Figure 25), followed by a period of relative stability from 1989 to 2006, where the average catch per net during this period (mean fish per net = 0.36), and did exhibit a slightly increasing population trend ($r^2 = 0.30$; $p = 0.02$; Figure 25). Gill net catch of cutthroat trout in Libby Reservoir exhibit a similar pattern, with the exception that that cutthroat trout catch rates exhibit 3 general trends through the same period. The first is a significant and precipitous decline during the early years of impoundment from 1975 to 1986 (Figure 25), where mean catch rates averaged 1.37 fish per net. The second trend showed 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 trend occurred from 1994 to 2007, characterized by a significantly lower level of abundance (0.143 fish per net; $p < 0.001$), at a somewhat stable level ($r^2 = 0.02$; $p = 0.62$). We believe that the period of general equilibrium during the period 1987-1993 may have been artificially elevated by the presence of hatchery cutthroat trout that were extensively stocked in the reservoir during this period (Table 16). Hatchery cutthroat trout were last stocked in the reservoir in 1994.

Figure 25. Mean catch rates (fish per net) of three native species (mountain whitefish (a) in spring sinking gillnets in the Rexford area (1975-2008), rainbow (b) and westslope cutthroat trout (c) in fall floating gillnets (1975-2007) from Tenmile and Rexford areas in Libby Reservoir. The Tenmile area was not sampled from 2001-2007.

Table 16. Average catch rate (fish per net) 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 2007. The Tenmile location was not sampled in 2000-2007.

	Year													
	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
Catch Rate	0.50	0.32	0.32	0.32	0.50	0.50	0.14	0.07	0.21	0.11	0.14	0.11	0.07	0.21
Avg. Length (mm)	295	264	238	261	275	260	251	314	252	225	267	305	302	259
Avg. Weight (gm)	249	196	146	191	211	191	156	316	161	128	228	296	271	175
No. Stocked	none	5,779	40,376	67,387	72,376	72,367	1,360	none	none	none	none	none	none	none
Length (mm)	n/a		33	104	216	190	287	n/a	n/a	n/a	n/a	n/a	n/a	n/a

	Year						
	2001	2002	2003	2004	2005	2006	2007
Catch Rate	0.21	0.07	0.21	0.07	0.36	0.14	0.07
Avg. Length (mm)	259	305	270	196	215	286	205
Avg. Weight (gm)	175	256	206	76	132	243	91
No. Stocked	none	none	none	none	none	none	none
Length (mm)	n/a	n/a	n/a	n/a	n/a	n/a	n/a

Kamloops Rainbow Trout (Duncan Strain)

Kamloops rainbow trout were first introduced to Libby Reservoir in 1985 by BCMOE. The BCMOE continued stocking approximately 5,000 fingerling Kamloops (Gerrard strain) annually into Kikomun Creek (a tributary to the Kootenai River) from 1988-1998 (L. Siemens, BCMOE, personal communication). MFWP has stocked approximately 11,000 to 73,000 Duncan strain Kamloops rainbow trout since 1988 directly into the reservoir (Table 17). Catch rates for Kamloops rainbow trout in fall gillnets has been low since 1996, averaging only 0.06 fish per gillnet. The catch rate in 2007 at the Rexford site was nearly identical to the mean, capturing a single Kamloop trout per 14 nets (0.07 fish/net). The catch rate 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 ($p = 0.067$; $r^2 = 0.18$; Table 17) for 1989 through 2007. This relationship is even further strengthened when we regressed the number of age 1 Kamloops released the previous year to gillnet catch rates ($p = 0.0005$; $r^2 = 0.52$; Table 17). However, there was no suggestion of a relationship between number of age 0 hatchery Kamloops released the year before and gillnet catch rates ($r^2 = 0.09$; $p = 0.22$). Age 0 Kamloops trout were first stocked in 1996 and then annually since 2002. These data suggest that age 0 releases of Kamloops trout in Libby Reservoir have very low survival. The catch rate of Kamloops rainbow trout in fall floating gillnets shows a slight negative trend that is nearly significant (Figure 26; $r^2 = 0.15$; $p = 0.09$).

Table 17. Kamloops rainbow trout captured in fall floating gillnets in the Rexford and Tenmile areas of Libby Reservoir, 1988 through 2007. The Tenmile site was not sampled in 2001-2007.

	Year							
	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
Age 0 Stocked	0	0	0	0	0	0	0	0
Age 1 Stocked	26,756	73,386	39,683	15,004	12,918	10,831	16,364	15,844
Total Stocked	26,756	73,386	39,683	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

	Year							
	1996	1997	1998	1999	2000	2001	2002	2003
No. Caught	2	1	2	3	3	0	0	5
Avg. Length (mm)	460	395	376	378	395	N/A	N/A	260.8
Avg. Weight (gm)	1192	518	450	504	555	N/A	N/A	159.2
Age 0 Stocked	3,165	0	0	0	0	0	29,564	31,039
Age 1 Stocked	9,396	22,610	16,368	13,123	0	0	0	13,721
Total Stocked	12,561	22,610	16,368	13,123	0	0	29,564	44,760
Length (mm)	170.5	152-178	127-152	255-280	N/A	N/A	80.3	81-206

	Year			
	2004	2005	2006	2007
No. Caught	0	0	1	1
Avg. Length (mm)	N/A	N/A	256	277
Avg. Weight (gm)	N/A	N/A	174	220
Age 0 Stocked	46,944	33,265	28,578	32,240
Age 1 Stocked	16,110	14,933	22,638	16,091
Total Stocked	63,054	48,198	51,216	48,331
Length (mm)	76 - 178	106-190	107-191	107-218

Figure 26. Average catch (fish per net) of Kamloops rainbow trout (Duncan strain) in fall floating gill nets in Libby Reservoir at the Rexford and Tenmile sites 1988-2007. The Tenmile site was not sampled in 2001-2007.

Bull Trout

Spring gill net catch of bull trout during the period 1975-1989 appeared to exist at an equilibrium with a slope (0.0091) that was not significantly different than zero ($r^2 = 0.011$; $p = 0.751$). However, beginning in approximately 1990, bull trout catch per net in Libby Reservoir began significantly increasing through 2008 (Figure 27; $r^2 = 0.685$; $p = 1.2 \times 10^{-5}$). The mean catch rate we observed in 2008 was 5.4 bull trout per net, which was slightly higher than the previous year (2007 mean catch rate = 4.5 bull trout per net; Dunnigan et al. 2008). Bull trout catch rates on Libby Reservoir peaked in 2000 at 6.71 bull trout per net. We attempted to account for differing reservoir levels during the gillnetting activities between years by multiplying the mean bull trout catch per net by reservoir volume at the time the nets were fished each year. This adjustment substantially improved the regression model's fit for both the overall (all years) and 1990-2008 data sets (Figure 28). However, the largest improvement was observed when the adjustment was applied over the time period 1990-2008 (Figures 27 and 28, respectively), where we observed an improvement in the model's r^2 by 0.101, versus an improvement of only 0.013 when the adjustment was applied to overall dataset. Bull trout redd counts in both the Wigwam River and Grave Creek (see above) are both significantly and positively correlated to the spring gill net catch rates for bull trout (Figure 29; $r^2 = 0.436$; $p = 0.007$). The adjustment we made using reservoir volume was also applied to the previous model, which improved the overall model fit by increasing the r^2 by 0.077 (Figure 30).

Figure 27. Average catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir 1975-2008.

Figure 28. Average adjusted catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir. Average annual bull trout catch per net was adjusted by multiplying catch by reservoir volume at the time of gillnetting.

Figure 29. Average catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir related to total annual bull trout redd counts for the Wigwam River and Grave Creek during the period 1994-2008.

Figure 30. Average adjusted catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir related to total annual bull trout redd counts for the Wigwam River and Grave Creek during the period 1994-2008. Average annual bull trout catch per net was adjusted by multiplying catch by reservoir volume at the time of gillnetting.

Burbot

Burbot catch rates in spring sinking gillnets since 1990 show a nearly significant negative trend in abundance (Figure 31; $r^2 = 0.199$; $p = 0.063$). Burbot catch per net for spring sinking nets has averaged 0.26 fish per net, and ranged from a low of 0 in four of the last seven years to a high of 0.6 fish per net in 1992. Burbot catch rates in spring gillnets is however significantly and positively correlated ($r^2 = 0.493$; $p = 0.016$; 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 Libby Reservoir may be influencing burbot abundance in the Kootenai River below Libby Dam through entrainment.

Figure 31. Mean catch per net of burbot in sinking gillnets during spring gillnetting at the Rexford site on Libby Reservoir, 1990-2008.

Figure 32. The relationship between mean burbot catch per net for spring sinking gillnets on Libby Reservoir and burbot catch rates (Burbot/trap day) of baited hoop traps in the stilling basin below Libby Dam 1995-2007.

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 increased since impoundment, but the trend was not significant (Figure 33; $r^2 = 0.04$; $p = 0.32$; Table 18), and is indicative of an increase in the biomass of species that prefer reservoir habitats: Columbia River chub, suckers, northern pikeminnow, etc. However, there is no significant trend in total catch (fish per net) for fall gillnets (Figure 33; $r^2 < 0.0001$; $p = 0.90$; Table 19). Species composition for the catch of fall and spring gillnets has remained relatively stable since 1992 (Table 20).

Figure 33. Catch per net (all species combined) in fall floating (1975-2007) and spring sinking gillnets (1975-2008) and associated trend lines in Libby Reservoir.

Table 18. 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, 1992 through 2007. The Tenmile area was not sampled from 2001-2007.

	YEAR															
	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007
Date	9/25	10/5	9/27	10/10	9/23	9/22	9/21	9/14	9/12	9/20	9/10	9/16	9/14	9/21	9/13	9/11
Number Nets	28	28	28	28	28	28	28	28	28	14	14	14	14	14	14	14
Res. Elevation	2421	2441	2446	2454	2450	2448	2439	2453	2434	2433	2441	2435	2445	2437	2441	2437
Average number of fish caught per net for individual fish species																
RBT	0.1	0.4	0.2	0.6	0.3	0.3	0.2	0.2	0.6	0.3	0.5	0.5	0.4	0.4	0.6	0.4
WCT	0.5	0.9	0.1	0.1	0.2	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.4	0.1	0.1
RB X WCT	0.2	0	0	0	0	0	<0.1	0	0	0	0	<0.1	0	0	0	0
SUB-TOTAL	0.8	1.3	0.3	0.7	0.5	0.4	0.3	0.3	0.7	0.4	0.6	0.7	0.4	0.8	0.7	0.5
MWF	0.2	0.3	0.4	0.3	0.3	0.5	0.4	0.1	0.1	0.2	0.4	0.4	0.6	0.1	0.6	0.9
CRC	23.3	17.1	10.4	1.2	11.7	17.8	14.4	24.3	12.9	5.6	21.4	5.0	1.6	11.2	9.9	11.4
NPM	1.8	2.2	3.4	2.7	1.8	4.0	4.9	6.4	3.9	3.9	8.1	3.36	3.3	7.3	5.6	11.7
RSS	0	0	0.3	0.2	0.1	1.0	0.3	0.3	<0.1	0	0.3	<0.1	0	0.1	0.4	0.6
BT	0.1	0.3	0	1.2	<0.1	0	<0.1	<0.1	0.2	0	0.1	0	0.21	0.04	0	0
CSU	0	0.1	0.1	0	0.4	0.1	0.1	0.1	0.1	0.3	0.1	0.2	0.2	0.2	0.3	2.4
KOK	5	1	4	7.9	2.3	3.1	2.7	7.3	8.0	2.1	14.2	7.4	3.5	11.4	5.4	0.8
TOTAL	31.2	22.3	18.9	14.2	17.1	26.9	23.1	38.8	25.9	12.5	45.1	17.1	9.8	31.2	23.1	28.5

*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 = redbside shiner, BT = bull trout, CSU = coarse scale sucker, and KOK = kokanee.

Table 19. Average catch per net for 12 different fish species* captured in sinking gillnets set during spring in the Rexford area of Libby Reservoir, 1992 through 2008.

	YEAR																
	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Date	5/5	5/17	5/16	5/8	5/12	5/12	5/11	5/17	5/14	5/15	5/13	5/13	5/11	5/10	5/10	5/21	5/13
Number of Nets	28	28	28	28	28	28	27	28	14	14	14	14	14	14	14	14	14
Pool Elevation	2333	2352	2405	2386	2365	2350	2417	2352	2371	2392	2384	2417	2419	2425	2424	2408	2397
Average number of fish caught per net for individual fish species																	
RBT	0.1	0.3	0.2	0.2	0.7	0.1	<0.1	1.1	0.3	0.2	0.4	0.7	0.6	0.4	0.3	1.5	0.4
WCT	0.1	0.0	<0.1	0.1	0.1	0.2	0.0	0.3	0.1	0	0	0.2	0.2	0.1	0.0	0.1	0.1
RB x WCT	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0	0.2	0.0	0	0	0.0	0	0
SUB-TOTAL	0.2	0.3	0.2	0.3	0.9	0.3	0.0	1.4	0.4	0.2	0.6	0.9	0.8	0.5	0.3	1.6	0.5
MWF	0.9	0.1	0.3	1.5	1.6	1.3	1.2	0.7	0.8	0.4	1.2	1.2	0.5	0.2	0.8	0.9	0.6
CRC	119	63.3	94.2	54.1	60.9	51.1	171.7	54.4	76.4	25	24.1	42.1	44.4	23.1	63.9	26.1	45.2
NPM	4.2	3.8	7.6	8.0	10.0	13.1	15.1	14	12.6	11	9.9	13.0	11.9	9.7	10.9	20.3	17.9
RSS	0.5	0.0	0.0	0.0	0.0	0.1	1.0	0.1	0.4	0	0	0.1	0	0	0.0	0.4	0
BT	2.3	1.2	3.0	2.3	3.5	3.1	2.5	3.6	6.7	5.4	4.9	5.4	6.4	5.9	4.4	4.5	5.4
LING	0.6	0.3	0.1	0.1	0.5	0.4	0.4	0.4	0.3	0.1	0.1	0.3	0.1	0.2	0.2	0.1	0
CSU	12.9	9.8	9.0	12.0	19.9	14.3	21.1	8.3	10.6	14.2	9.9	10.2	5.2	11.8	8.6	9.4	9.6
FSU	2.9	4.1	6.5	3.0	4.8	4.7	9.5	5.9	5.1	1.1	2.9	2.3	0.3	1.1	0.9	2.1	1.0
YP	1.8	1.1	0.7	2.5	3.7	4.75	2.4	1.8	1.3	1.6	0.6	0.1	0.5	0.4	0.4	1.6	0.4
KOK	0.4	3.5	0.3	2.1	2.0	1.4	1.3	5.3	1.0	0.2	1.0	1.2	0.9	3.4	0.6	2.1	0.8
TOTAL	145.3	84.3	121.9	86.3	107.1	93.25	226.2	95.9	115.1	59.2	55.2	76.8	70.9	53.4	0.8	69.4	81.6

*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 = redbside shiner, BT = bull trout, LING = burbot, CSU = coarse scale sucker, FSU = fine scale sucker, YP = yellow perch, and KOK = kokanee.

Table 20. Percent composition of major fish species* caught in fall floating and spring sinking gillnets in Libby Reservoir, 1992 through 2008. Blank entries in table indicate either no fish were captured or that they occurred in very small proportions.

Species	YEAR																	
	1992		1993		1994		1995		1996		1997		1998		1999		2000	
	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Fall	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.
RB	0.3		1.8		0.9		4.4		1.1	1.1	1.8		0.9		4.4		1.1	0.2
WCT	1.7		3.8		0.7		0.8		0.8	0.8	3.8		0.7		0.8		0.8	0.1
HB	0.7		0.2		0.0		0.3		0	0	0.2		0.0		0.3		0	0
ONC	2.7	0.1	5.8	0.3	1.7	0.2	5.5	0.4	1.9	1.9	5.8	0.3	1.7	0.2	5.5	0.4	1.9	0.3
MWF	0.7	0.6	1.4	0.2	2.2	0.3	2.1	1.7	0.5	0.5	1.4	0.2	2.2	0.3	2.1	1.7	0.5	0.7
CRC	72.6	81.7	72.8	73.9	54.3	77.0	8.6	62.9	46.4	46.4	72.8	73.9	54.3	77.0	8.6	62.9	46.4	66.0
NPM	5.6	2.9	9.3	5.0	17.5	6.2	19.6	9.3	18.1	18.1	9.3	5.0	17.5	6.2	19.6	9.3	18.1	10.8
RSS	0.0	0.3	0.0	0.0	1.5	0.0	1.3	0.0	0.1	0.1	0.0	0.0	1.5	0.0	1.3	0.0	0.1	0.4
FSU	0.1	2.0	0.0	5.2	0.0	5.3	0.0	3.5	0.1	0.1	0.0	5.2	0.0	5.3	0.0	3.5	0.1	4.0
CSU	0.0	8.8	0.6	9.7	0.6	7.3	0.0	13.9	4.0	4.0	0.6	9.7	0.6	7.3	0.0	13.9	4.0	9.1
KOK	15.7	0.3	4.4	3.4	20.6	0.2%	57.4	2.4	28.6	28.6	4.4	3.4	20.6	0.2%	57.4	2.4	28.6	0.9
YP		1.2		1.1		0.9		2.9	0.3	0.3		1.1		0.9		2.9	0.3	1.1
BT		1.7		1.1		2.5		2.8	0	0		1.1		2.5		2.8	0	5.8

Table 20. (Continued) Percent composition of major fish species* caught in fall floating and spring sinking gillnets in Libby Reservoir, 1992 through 2008. Blank entries in table indicate either no fish were captured or that they occurred in very small proportions.

Species	YEAR																	
	2001		2002		2003		2004		2005		2006		2007		2008		Average	
	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall
RB	1.4	0.4	0.4	0.4	2.8	0.9	3.7	0.8	1.2	0.8	0.3	2.8	2.2	1.5	0.5	N/A	0.5	1.7
WCT	1.7	0	0.1	0	0.8	0.3	0.7	0.3	1.1	0.1	0	0.6	0.1	0.3	0.2	N/A	0.1	1.0
HB	0	0	0	0	0.4	0.0	0.0	0.0	0.0	0.0	0	0	0	0	0.0	N/A	0.0	0.3
ONC	3.1	0.4	0.5	0.4	4.0	1.2	4.5	1.1	2.3	0.9	0.3	3.4	2.3	1.8	0.7	N/A	0.7	3.0
MWF	2.5	0.6	0.3	1.5	2.0	1.6	6.0	0.7	0.5	0.4	0.9	2.8	1.3	3.3	0.8	N/A	1.1	1.9
CRC	49.3	42.2	41.5	62.4	27.7	54.9	16.4	62.6	35.2	41.1	70.1	42.9	37.7	39.9	55.4	N/A	54.8	45.3
NPM	22.5	18.6	14.4	11.8	18.6	16.9	34.3	16.8	22.8	17.2	12.0	24.1	29.2	41.1	21.9	N/A	14.8	20.0
RSS	1.4	0	0.9	0	0.4	0.1	0.0	0.0	0.5	0.0	0	1.9	0.5	2.0	0.0	N/A	0.2	1.0
FSU	0	1.9	0	3.4	0.4	3.0	0.0	0.4	0.0	1.9	1.0	0	3.1	0	1.2	N/A	4.1	0.1
CSU	3.4	24.0	0.6	12.3	1.2	13.3	2.2	7.4	0.7	20.9	9.5	1.2	13.6	8.5	11.7	N/A	14.4	2.3
KOK	17.5	0.4	41.6	1.2	41.1	1.6	36.6	1.2	36.0	6.1	0.7	23.5	3.1	3.0	1.0	N/A	2.2	24.9
YP	0	2.7	0.1	0.8	5.1	0.1	0.0	0.7	1.8	0.6	0.4	0.3	2.3	0	0.4	N/A	1.7	0.7
BT	0.3	9.2	0	5.9	0	7.0	2.2	9.0	0.1	10.5	4.8	0	6.5	0	6.6	N/A	5.3	0.3

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

Libby Reservoir Zooplankton Monitoring

Zooplankton species composition and abundance within Libby Reservoir has remained relatively stable during the past several years (Appendix Tables A6-A9). During the period 1997 through 2007, *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*, *Epischura* and *Leptodora* (Figure 34). The results from the monitoring in 2007 were similar with the exception that in 2007, *Bosmina* were slightly more abundant than *Diaptomas*. However, the fact that we were unable to sample zooplankton in the reservoir in September or October 2007, may have been partially responsible, since *Diaptomas* peaks seasonally in either September or October (see below). Zooplankton abundance within the reservoir varies by month (Table 21; Figure 35). The results from 7 analysis of variance procedures that tested for differences in monthly zooplankton abundance (by genera) indicated that at least one month was significantly different from other months in 2007 for all 7 of the most common genera of zooplankton. Although we did not perform multiple comparisons required to determine pairwise comparisons, zooplankton abundance varies within a season, with seasonal peaks in abundance over the past ten years (Figure 35) remaining relatively consistent across years. For example, *Daphnia* abundance has peaked during July each year except 2003 and 2006 (June peak) since 1997, *Diaphanosoma* abundance has peaked in late August or September 9 the previous 10 years, excluding 2007 since we did not sample zooplankton during the months of September or October in 2007 due to malfunctions to our lake boat. *Diaptomus* has peaked in either September or October during 8 of the previous 10 years, also excluding 2007 since we did not sample in September or October in 2007. *Cyclops* has peaked in either May or June during 10 of the last 11 years. In most cases when an individual annual peak differed from the mean peak, the difference was not more than several weeks.

Our sampling design stratified the reservoir into thirds, and although each stratum was long (> 58 km), we found only weak evidence that zooplankton abundance differed between the three sampling areas (Tenmile, Rexford, and Canada) in 2007 (Table 21). For the 7 most abundant genera of zooplankton in the reservoir at the three sites *Epischura* and *Cyclops*, were the only groups of zooplankton that differed significantly between areas ($P < 0.001$; Table 21), and *Daphnia* was also close to being significantly different ($P = 0.065$; Table 21). *Epischura* and *Cyclops* densities differed on a longitudinal gradient, with the highest abundances generally observed in the lower two strata within the reservoir. Multiple comparisons indicated that *Epischura* densities within the Tenmile stratum were significantly higher than the Rexford and Canada strata, but that the Rexford and Canada strata were not significantly different from each other. Although the abundance of *Epischura* in the Rexford stratum was slightly lower than in the Canada stratum. *Cyclops* abundance exhibited a similar trend in 2007. *Cyclops* densities in the Tenmile and Rexford strata were significantly higher than the Canada stratum, but were not different than each other. The month and area interaction term was significant for all the most abundant genera of zooplankton except *Bosmina*, *Leptodora*, *Epischura*, and *Diaphanosoma* in 2007 (Table 21).

The trends in *Daphnia* abundance (Figure 34) and size (Figures 36 and 37) in Libby Reservoir have remained particularly stable during the past several years. Mean annual *Daphnia* densities in Libby Reservoir from 1997 through 2007 have averaged 1.98 *Daphnia* /liter (standard deviation = 0.16/liter; Figure 36). Although, the mean abundance of *Daphnia* we observed in 2007 (1.48 *Daphnia*/l) was the second lowest on record since 1997. Mean *Daphnia* length has also varied relatively little since 1991, averaging 0.90 mm (standard deviation = 0.045; Figure 47). Most *Daphnia* since 1993 are between 0.5 – 1.5 mm, with majority of *Daphnia* being represented in the smaller size class 0.5 – 0.99 mm (mean annual proportion = 0.61, standard deviation = 0.049; Figure 46), with the majority of the remainder in the size class 1.0 – 1.499 (mean annual proportion = 0.34, and standard deviation = 0.033). *Daphnia* larger than 1.5 mm have on average comprised less than 5% of the total since 1993 (Figure 46).

Figure 34. Annual zooplankton abundance estimates for seven genera observed in Libby Reservoir from 1997-2007. 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 A9.

Figure 35. Mean monthly zooplankton abundance estimates for seven genera observed in Libby Reservoir from 1997-2007. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter.

Table 21. Individual probability values (p values) resulting from analysis of variance procedures that tested for differences in zooplankton densities by month (April – November), area (Tenmile, Rexford and Canada) and a month by area interaction in 2007.

Genus	Month	Area	Month*Area Interaction
<i>Daphnia</i>	<0.001	0.065	0.001
<i>Bosmina</i>	<0.001	0.285	0.404
<i>Diaptomas</i>	<0.001	0.306	0.002
<i>Cyclops</i>	0.002	<0.001	<0.001
<i>Leptodora</i>	<0.001	0.148	0.091
<i>Epischura</i>	<0.001	<0.001	<0.001
<i>Diaphanosoma</i>	<0.001	0.368	0.520

Figure 36. *Daphnia* species size composition in Libby Reservoir, 1984 through 2007.

Figure 37. Mean length of *Daphnia* species in Libby Reservoir, 1984 through 2007, with error bars representing plus and minus one standard deviation from the mean.

Discussion

The CPUE of burbot from baited hoop traps in Libby Reservoir in 2007/2008 are low compared to other studies that used similar gear. Breeser et al. (1988) found catches of 2-11 burbot per hoop trap when fishing overnight sets in the Chisana and Tanana rivers, Alaska. Bernard et al. (1993) reports mean burbot CPUE for five moderately sized (538 – 6,519 hectare) Alaskan lakes ranging from 0.12 to 2.24 adult burbot (> 449 mm total length) per 48 hour set. Burbot CPUE on the upper Missouri River, Montana ranged from 0.17 to 2.94 fish per hoop trap per 48 hour set (T. Horton, Montana Fish, Wildlife & Parks, personal communication). Burbot CPUE in Libby Reservoir were similar to those reported by Krueger and Hubert (1997) in 7 lakes and reservoirs in the Big Horn / Wind River drainage in Wyoming where mean catch rates ranged from 0.02-0.45 burbot per 24 hour net set. Krueger and Hubert (1997) concluded that exploitation was reducing burbot abundance in the three natural lakes and sediment and water level fluctuations were likely limiting burbot abundance in the four reservoirs. Although relative burbot abundance in Libby Reservoir is low compared to other studies, it is approximately an order of magnitude higher than the mean CPUE Paragamian and Hoyle (2003) reported for the lower Kootenai River.

Burbot spawning is synchronized with water temperature during the winter months from January to March (McCrimmon 1959; Scott and Crossman 1973). Spawning usually occurs at night in 0.3 – 1.2 m of water over sand or gravel in shallow bays or in 1.5 – 3.0 m of water over gravel shoals (Scott and Crossman 1973). We surmise that burbot in Libby Reservoir spawn between late January and mid February based on observations of gravid females captured in the hoop traps, and water temperatures. However, reservoir operations during some years may be limiting burbot recruitment during the period January through March. The U.S. Army Corps of Engineers operates Libby Dam primarily for flood control and power production, with maximum pool draw downs typically occurring during March and April. Draw downs during March and April during the period 1990-2008 have averaged 89.8 and 88.7 feet, respectively below full pool. Libby Reservoir becomes isothermic beginning in approximately December and maintains this condition typically through April – May (U.S. Army Corps of Engineers, unpublished data). Maximum mean daily water temperatures during January – March in Libby Reservoir during the period 1999-2007 averaged between 4.0 and 4.5° C (U.S. Army Corps of Engineers, unpublished data). Scott and Crossman (1973) report that burbot eggs hatch in 30 days when held at 6° C. Therefore, it is likely that burbot eggs in Libby typically require more than 30 days to hatch, and given the fact that minimum reservoir elevations occur shortly after the suspected period of burbot spawning and that water level fluctuations are greater than the reported depth of burbot spawning, seasonal water level fluctuations in Libby Reservoir may be exposing burbot spawning beds prior to the hatch. Bergersen et al. (1993) observed a similar situation in a much smaller Wyoming irrigation reservoir.

Our juvenile fish monitoring associated with the stream restoration efforts is further discussed in Chapter 2 (see below). However we acknowledge that the lack of at least one control site associated with the Grave Creek Demonstration Project is a limitation on our

ability to detect change over time. Annual variability in fish populations associated with our monitoring is also relatively large from year to year, and likely limits our ability to detect changes that result from restoration efforts. Additional years of fish population monitoring prior to restoration may have improved the statistical power of most of our tests.

Our results indicate that secondary and tertiary production in Libby Reservoir has stabilized during the previous 12-17 years. Total fish abundance, as indexed by trends in gill net catch rates, has 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 29) following the first ten years after reservoir filling, but have stabilized at much lower levels of abundance than the pre-dam period. Fish species composition also shifted during the first 10 years after reservoir construction, but has also stabilized, with the exception of bull trout, which has increased in abundance during the past several years, but appear to perhaps to be beginning to stabilize. 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 due to the aging process of the reservoir (Kimmel and Groeger 1986) and the operational history of Libby Dam during the past 17 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 & Parks, the Kootenai Tribe of Idaho and the Confederated Salish and Kootenai Tribes documents the hydropower related losses and mitigation actions attributable to the construction and operation of Libby Dam, as called for by the Northwest Power and Conservation Council's Fish and Wildlife Program (MFWP et al. 1998). The actions and projects MFWP prioritizes and implements are also identified in the Kootenai Subbasin Plan (KTOI and MFWP 2004). A mix of mitigation techniques is necessary to offset losses caused by dam construction and operation. This report describes eight stream restoration projects and associated physical monitoring that has occurred from one to five years since project completion.

MFWP treated Loon Lake (Pipe Creek Watershed) with CFT-Legumine rotenone formulation on September 17, 2007 to remove non-native brook trout and black bullheads. We used a combination of powdered rotenone and CFT-Legumine to remove fish from the lake tributaries and the unnamed outlet stream downstream to an existing barrier located approximately 1 mile upstream from the confluence with Pipe Creek. In order to prevent the rotenone laden water flowing out of Loon Lake from reaching Pipe Creek, we added potassium permanganate to the stream near the barrier. The rotenone in the lake and outlet stream was neutralized by November 30, 2007. We used gillnetting and backpack electrofishing in the spring of 2008 to ensure that all fish had been removed from the lake and outlet stream, respectively. MFWP restocked the lake in the spring of 2008 with age 0 westslope cutthroat trout.

MFWP collaborated on two restoration projects on lower Grave Creek, near Eureka. The Grave Creek Phase I Restoration Project was completed in the fall of 2002, and the Phase II Restoration Project was completed in the fall of 2004. MFWP also collaborated with a local landowner on upper Libby Creek to complete three restoration projects, the Cleveland Upper and Lower (Phases I and II) Restoration Projects completed in 2002, 2005, and 2006, respectively. MFWP also completed restoration projects on Young and Therriault creeks in 2003 and 2005, respectively. The physical monitoring efforts associated with each of these restoration project focuses on describing the initial physical changes that occurred to the stream as a result of project construction, and an evaluation of whether physical changes are sustained through time. All the restoration projects changed the dimension, pattern and profile of the stream within the project reaches relative to preexisting conditions. We observed an overall increase of stream length and the creation of a deeper and narrow stream channel with increased pool habitat. Each restoration project substantially increased the frequency, and depth of pool-type habitat within the project areas, with total pool number and volume showing the largest increases relative to existing conditions. We generally found no evidence that stream channel dimensions significantly changed within the riffle habitat, with

annual changes generally <10-15% for each project from one to five years after completion with the exception of the Libby Creek projects. Changes in pool habitat within the projects after construction have been more variable and generally less sustained through time when compared to riffle dimensions. When pool characteristics changed over time after construction, it was generally a slight decrease in pool mean bankfull depth and total number of pools. Nevertheless, the quantity and quality of pool habitat within each project area still exceeded preexisting conditions by a factor of several fold.

A rain on snow event occurred in November 2006 within the Libby Creek watershed that substantially damaged much of the restoration work in upper Libby Creek. The most conspicuous changes within the three restoration project areas on Libby Creek as a result of the November 2006 flow event were changes in the stream planform, with substantial stream straightening and loss of meander pattern throughout, and a loss of the total number of pools. Stream width increased, especially within riffle habitats after the flood event. These patterns of disturbance were exacerbated along a downstream gradient, with changes most substantial within the Lower Cleveland Phase II project area.

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 downstream of Libby Dam) and the headwaters in Canada.

Operations of Libby Dam cause large fluctuations in reservoir levels and rapid daily fluctuations in the 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, limits production of fall-hatching aquatic insects, and also reduces the deposition of terrestrial insects from the surrounding landscape. Surface elevations continue to decline during winter, arriving at the lowest point in the annual cycle during April. Deep drafts

reduce food production and concentrate young trout with predators. Of greatest concern is the dewatering and desiccation of aquatic dipteran larvae in the bottom sediments. These insects are the primary spring food supply for westslope cutthroat, a species of special concern in Montana, and other important game and forage species. Deep drawdowns also increase the probability that the reservoirs will fail to refill. Refill failure negatively effects recreation and reduces biological production, which decreases fish survival and growth in the reservoir (Marotz et al. 1996, Chisholm et al. 1989). Investigations by Daley et al. (1981), Snyder and Minshall (1996), and Woods and Falter (1982) have documented the declining productivity of the Kootenai System and, specifically, reduced downstream transport of phosphorous and nitrogen by 63 percent and 25 percent, respectively.

Large daily fluctuations in river discharge and stage (4-6 feet per day) strand large numbers of sessile aquatic insects in the varial zone (Hauer and Stanford 1997). 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 macroinvertebrate species diversity and standing crop (Hauer and Stanford 1997). Aquatic insects are affected by the reduction of microhabitat and food sources, as evidenced by the loss of species and total numbers since impoundment (Voelz and Ward 1991). Hauer and Stanford (1997) 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 macroinvertebrate densities are one of the most important factors influencing growth and density of trout in the Kootenai River (May and Huston 1983).

The mitigation and implementation plan developed by MFWP, the Kootenai Tribe of Idaho and the Confederated Salish and Kootenai Tribes documents the hydropower related losses and mitigation actions as called for by the Northwest Power Planning Council's Fish and Wildlife Program (MFWP et al. 1998). This plan identifies several mitigation actions capable of partially mitigating impacts to Montana's aquatic resources associated with the construction and operation of Libby Dam. These include aquatic habitat improvement, fish passage improvements, off-site mitigation, fisheries easements, and conservation aquaculture and hatchery products.

Loon Lake is located approximately 15 miles northwest of Libby, Montana, and has a surface area of 33 acres and a maximum depth of 25 feet. Loon Lake is fed by ground water and three seasonally intermittent streams from the northeast. The unnamed outlet stream that leaves the lake is the only outlet stream for the lake and flows from the lake through public land (USFS) for about 3 miles to Pipe Creek. Westslope cutthroat trout were likely the dominant fish species historically present in Loon Lake. However, black bullheads were illegally planted in Loon Lake sometime after the late 1960s or early 1970s. MFWP also stocked rainbow trout in the 1930s, 1980s, and 1990s, arctic grayling in the 1930s and 1940s, and brook trout from 1948-1961 on seven occasions. Currently, the only two species remaining in the lake are black bullheads and brook trout at relatively low levels in Loon Lake, but at sufficiently high abundance to be self-sustaining. Angler use of the lake has also

declined within recent years. In 1991, MFWP estimated that Loon Lake received 490 angler-days per year, but in 2003, angling pressure on Loon Lake dropped to 82 angler-days per year.

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. Restoration activities on Grave and Libby creeks are consistent with those strategies identified in the Fisheries Mitigation and Implementation Plan for the Losses attributable to the Construction and Operation of Libby Dam (MFWP et al. 1998) and the Kootenai Subbasin Plan (KTOI and MFWP 2004).

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 disrupted the natural equilibrium within Libby Creek resulting in accelerated bank erosion along a number of meander bends, causing channel degradation. This resulted in impaired fish habitat that likely reduced the productivity and carrying capacity for resident salmonids within Libby Creek. Prior to restoration the stream channel is over-widened and shallow with limited pool habitat (Sato 2000). Many of the problems related to 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).

Young Creek is one of the most important westslope cutthroat trout spawning tributaries to Libby Reservoir, containing one of the last known genetically pure populations of westslope cutthroat trout in the region. We identified and prioritized a restoration project on Young Creek because it is one of the most potentially productive tributaries to Libby Reservoir, and the degraded habitat on the state owned section of the creek. During the

1950's, approximately 1,200 feet of the channel located on the state owned section (DNRC School Trust Land) was straightened, diked, and moved near the toe of the hill slope. This channelization compromised the stream's ability to effectively transport sediment through the channelized area, causing the channel to aggrade (deposit bedload materials) and exacerbating flood conditions. Sediment aggradation caused numerous problems with the stream, including poor aquatic habitat, increased flood potential, lateral bank scour and increased sediment supply. Additionally, livestock grazing and timber management in the upper reaches of Young Creek likely contributed to channel instability.

Therriault Creek is a tributary to the Tobacco River and is located approximately 6 miles southeast of the town of Eureka in Lincoln County, Montana. MFWP identified a 4,500 foot long section of Therriault Creek as a priority restoration project because it contained a native fish assemblage, was substantially degraded, but has a high potential for restoration, and the lower section was owned by a single landowner that was willing to partner to improve conditions on Therriault Creek.

Stream restoration efforts when applied appropriately can be successful at restoring streams to a state of equilibrium. 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 related to 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 describe the restoration projects completed and discuss monitoring results to date.

Methods and Results

Loon Lake Rehabilitation Project

MFWP applied 285 gallons of the CFT-Legumine rotenone formulation to Loon lake on September 19, 2007 using venture and trash pumps on a small motorboat and backpack sprayers around the lake margins to achieve an overall lake concentration of 5.1 ppm rotenone to remove the brook trout and black bullheads inhabiting the lake. We treated the tributaries of the lake and outlet stream with powdered rotenone to remove fish from these waters and ensure fish could not use these areas as refugia. In addition, we applied CFT-Legumine the outlet stream proper using a combination of drip stations and backpack sprayers to achieve an overall concentration of 2 ppm rotenone. Rotenone treatments occurred downstream to an existing barrier on the outlet stream approximately 1 mile upstream from the confluence with Pipe Creek. In order to prevent the rotenone laden water flowing out of Loon Lake from reaching Pipe Creek, we operated a detoxification station located near an existing fish barrier continuously from September 19 to October 19, 2007. During this period we added 272.1 kg of dry potassium permanganate directly to the outlet stream to neutralize the rotenone using an auger powered by a 4000-watt generator. Potassium permanganate concentrations of ranged from 4 – 6 ppm depending on stream stage with a target concentration of 1 –2 ppm residual potassium permanganate achieved at 30 minutes travel time below the detoxification station. We used bioassays to determine when the lake effluent contained non-lethal levels of rotenone, and determined the lake was capable of supporting trout on November 30, 2007. We used gillnetting and backpack electrofishing in the spring of 2008 to ensure that all fish had been removed from the lake and outlet stream, respectively. These surveys found no living fish in either waters. MFWP restocked the lake with age 0 westslope cutthroat trout later during the spring of 2008, and will continue stocking the lake on regular intervals.

Grave Creek Phase I Restoration Project

MFWP partnered with the Kootenai River Network to restore approximately 4,300 feet of channel within the lower three miles of Grave Creek, named the Grave Creek Phase I Restoration Project, which begins at the downstream end of the Grave Creek Demonstration Project (see Dunnigan et al. 2005). Project construction was completed during fall 2002. The objectives of the project were to: 1) Reduce the sediment sources and bank erosion throughout the project area by incorporating stabilization techniques that function naturally with the stream and which decrease the amount of stress on the stream banks, 2) Convert the channelized portions of stream into a channel type that is self maintaining and will accommodate floods without major changes in channel pattern or profile, 3) Use natural stream stabilization techniques that will allow the stream to adjust slowly over time and be representative of a dynamic 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.

The Grave Creek Phase I Restoration Project changed the dimension, pattern and longitudinal profile within the project area. These changes were designed to achieve the long-term project objectives and are described in detail in Dunnigan et al. (2005). The 41 stream restoration structures that the restoration project constructed increased channel diversity within the project area along the length of the project area and are described in Dunnigan et al. (2005). The existing stream channel prior to implementing this project contained long riffle sections and relatively low diversity and complexity (Figure 1). This project constructed a stream pattern that decreased the overall stream gradient by increasing stream length (increased sinuosity). As a result of the restoration work conducted in 2002, bankfull width and width to depth ratio significantly decreased and maximum and mean bankfull depth increased throughout the project area in 2002 and 2003 compared to pre-existing conditions (Dunnigan et al. 2005). However, severe icing conditions in lower Grave Creek in the winter of 2005/2006 and high spring flows in 2006 damaged some of the previously completed work, and as a result, additional maintenance and revegetation work to this section of Grave Creek was completed in the fall of 2006. This work description and locations within the project area are described in Table 1.

Table 1. Structures types, meander and station of the maintenance work performed on the Grave Creek Phase 2 Project in the fall of 2006.

Structure Type	Meander	Station (feet)
Straight log vane with cobble throat	D	0-50
Vegetated Soil Lift	D	50-90
Engineered Debris Jam	D	90-115
Vegetated Soil Lift	D	115-155
Add Wood to existing Engineered Debris Jam	D	155-170
Lower terrace to 0.5 ft. above bankfull elevation	D	0-90

We continued to monitor the physical stream dimensions, pattern and profile annually to determine if the reconstructed stream channel changes through time. We re-photographed the 25 photo points that were originally established shortly after project construction, documenting as built conditions. We currently have photo documentation for the post construction from 2002 through 2005. Photo points were not replicated in 2006 or 2007. Initially, we established six permanent cross-sections that were annually surveyed in 1999 (pre project) through 2002-2005 (post-project). However, the benchmark locations for these permanent cross sections could not be relocated in 2006, so these points have not been surveyed since. We surveyed the longitudinal profile of this section of Grave Creek in 2007, and found that stream channel diversity and complexity remains substantially higher than existed prior to initiation of this restoration project (Figure 1).

The Grave Creek Phase I Restoration Project increased the quality and quantity of rearing habitat for native salmonids by increasing the total number and depth of pools compared to conditions that existed prior to restoration (Dunnigan et al. 2004 and 2005). Due to the importance of pool habitat to rearing native salmonids within lower Grave Creek, we continued to monitor pool habitat after project construction to evaluate whether the pools maintained depth, width and length through time. We measured the mean width, length and maximum bankfull depth, total length and total surface area of all pools within the project area annually since 2002 (Table 2). There were a total of 26 pools in 2002, 27 pools in 2003 and 2004, but the total number of pools decreased to 23 in 2005, 21 in 2006, and then increased to 22 in 2007. We did not perform a statistical comparison for these data because the pool measurements represented all pools within the project area (i.e. complete census), making statistical comparisons unnecessary. Of the five parameters we measured, all of them increased or remained equal from 2006 to 2007 (Table 2). Of the pool dimensions we measured, total pool area showed the highest relative change from 2006 to 2007, increasing by 12.6% (Table 2). Total pool length also increased slightly (8.7%) from 2006 to 2007, with the mean pool length in 2007 of 77.5 feet being the largest measured before or after project construction (Table 2). However, other mean pool dimensions (width, and maximum depth) all remained relatively stable (< 5% change; Table 2) between 2006 and 2007. Therefore the increase in total pool area and length within the project area is mainly attributable to the formation of an additional pool between 2006 and 2007 and the overall increase in mean length between pools in 2007. The overall total number of pools within this section of Grave Creek remains higher than the six that were present prior to the restoration work (Dunnigan et al. 2004), and given the disparity between the total number of pools prior to the restoration work and after (Figure 1), total area and volume also likely remains higher as a result of this project. These data suggest that pool quantity and quality (depth) continued to be sustained within this section of Grave Creek as a result of the restoration work implemented five years earlier.

In addition to a complete census of all pools within the project area, we also completed a survey of all riffles within the project area in all years since 2003 in order to evaluate changes in riffle dimensions through time. Cross sectional surveys were performed at the longitudinal mid-point of each riffle, where we measured the bankfull width, maximum and mean depths, cross sectional area, and slope. Surveys were conducted in each respective year after the conclusion of the spring freshet. Our cross-sectional surveys indicated that riffle dimensions changed little between years since the project was completed. Stream channel dimensions within riffle habitats changed little between 2006 and 2007, with changes generally < 5% (Table 2). Although riffle slope increased substantially between 2004/2006 and 2005/2006, with increases of 46.5% and 43.2%, respectively, riffle slope between 2006/2007 only decreased by 1.2% (Table 2). Similar to stream channel dimensions in pool type habitats, dimensions in riffle-type habitats have generally been sustained in this section of Grave Creek since the restoration work was implemented, and remain substantially different than conditions that existed prior to the work.

Figure 1. The longitudinal profile for the Grave Creek Phase I Restoration Project. Data are presented for pre-project (2001), as-built conditions (2002), and 2007. The station (longitudinal location measured at the channel thalweg) begins at the upstream boundary of the project.

Table 2. Mean bankfull width, maximum bankfull depth, and mean length, total length and surface area measured from all pool-type habitats in 2003-2007 located in the Grave Creek Phase I Project. Variance estimates for annual mean values are presented in parentheses. A statistical comparison of annual mean values was not performed because all pools within the project area were surveyed, and therefore represents a complete census. The percent change for each parameter year to year is also presented.

	Number of Pools	Mean Bankfull Width (ft)	Maximum Bankfull Depth (ft)	Mean Length (ft.)	Total Length (ft.)	Total Area (ft²)
2003	27	54.0 (46.4)	5.6 (1.9)	74.8 (842.3)	1,944	109,058
2004	27	49.5 (63.6)	4.9 (1.0)	66.9 (341.6)	1,739	89,412
2005	23	51.1 (56.1)	5.1 (0.7)	61.2 (278.8)	1,407	71,892
2006	21	53.6 (58.2)	5.1 (0.9)	74.7 (814.1)	1,569	84,054
2007	22	55.9 (107.8)	5.2 (1.0)	77.5 (1,529.8)	1,705	94,668
Percent Change						
2003/2004	0%	-8.3%	-12.5%	-10.6%	-10.5%	-18.0%
2003/2005	0%	-5.4%	-8.9%	-18.2%	-27.6%	-34.1%
2003/2006	-22.2%	-0.7%	-8.9%	-0.1%	-19.3%	-22.9%
2003/2007	-18.5%	3.6%	-7.1%	3.6%	-12.3%	-13.2%
2004/2005	-14.8%	3.2%	4.1%	-8.5%	-19.1%	-19.6%
2004/2006	-22.2%	8.3%	4.1%	11.7%	-10.0%	-6.0%
2004/2007	-18.5%	13.0%	6.1%	15.8%	-2.0%	5.9%
2005/2006	-8.7%	4.9%	0%	22.1%	11.5%	16.9%
2005/2007	-4.3%	9.5%	2.0%	26.6%	21.2%	31.7%
2006/2007	4.8%	4.4%	2.0%	3.7%	8.7%	12.6%

Table 3. Mean bankfull width, depth, maximum bankfull depth, cross sectional area, width to depth ratio, and slope of riffles located in the Grave Creek Phase I Restoration Project in 2003 - 2007. Variance estimates for annual mean values are presented in parentheses. The percent change for each parameter year to year is also presented.

	Number of Riffles	Mean Bankfull Width (ft)	Maximum Bankfull Depth (ft)	Mean Bankfull Depth (ft.)	Cross Sectional Area (sq. ft.)	Width to Depth Ratio	Riffle Slope (%)
2003	7	49.4 (31.0)	3.3 (0.12)	2.16 (0.03)	106.0 (61.3)	23.2 (18.3)	1.06 (2.65*10 ⁻⁵)
2004	7	51.7 (36.0)	3.5 (0.05)	2.22 (0.01)	114.7 (132.2)	23.3 (11.1)	0.86 (9.87*10 ⁻⁶)
2005	10	52.3 (64.2)	3.5 (0.31)	2.18 (0.16)	111.5 (274.3)	25.2 (76.1)	0.88 (1.42*10 ⁻⁵)
2006	11	54.2 (44.6)	3.1 (0.29)	1.90 (0.15)	102.3 (428.8)	29.9 (69.9)	1.26 (3.45*10 ⁻⁵)
2007	9	55.7 (79.4)	3.0 (0.06)	1.94 (0.08)	107.8 (555.6)	29.5 (57.8)	1.24 (2.80*10 ⁻⁵)
Percent Change							
2003/2004	0%	4.6%	6.5%	3.0%	8.2%	0.7%	-18.3%
2003/2005	42.9%	5.7%	5.8%	1.1%	5.2%	8.9%	-16.6%
2003/2006	57.1%	9.7%	-6.1%	-12.0%	-3.5%	28.9%	18.9%
2003/2007	28.6%	12.8%	-9.8%	-10.4%	1.7%	27.2%	17.4%
2004/2005	42.9%	1.0%	0%	-1.8%	-2.8%	8.1%	2.0%
2004/2006	57.1%	4.8%	-11.4%	-14.4%	-10.8%	28.3%	46.5%
2004/2007	28.6%	7.8%	-14.9%	-12.8%	-6.0%	26.7%	44.7%
2005/2006	10.0%	3.6%	-11.4%	-12.8%	-8.3%	18.7%	43.2%
2005/2007	-10.0%	6.6%	-14.9%	-11.2%	-3.3%	17.1%	41.4%
2006/2007	-18.2%	2.8%	-3.9%	1.9%	5.3%	1.3%	-1.2%

Grave Creek Phase II Restoration Project

MFWP partnered with the U.S. Fish and Wildlife Service Private Stewardship Grant Program, the U.S. Forest Service Resource Advisory Committee, the U.S. Environmental Protection Agency/MT Department of Environmental Quality (319 Program), the U.S. Fish and Wildlife Service Partners for Wildlife Program, the Lincoln County Conservation District and the Flanagan Family (landowners) to restore approximately 3,050 feet of channel within the lower three miles of Grave Creek, named the Grave Creek Phase II Restoration Project. This project was administered by the Kootenai River Network, and begins at the downstream end of the Grave Creek Phase I Restoration Project (see above). The project was originally proposed to encompass 4,875 feet of lower Grave Creek. However, the lower most landowner on this section of the creek declined to participate in the project. Therefore, the project was shortened to the upper 3,050 feet beginning at the lower end of the Phase I Project. Project construction was completed during fall 2004. The objectives of the project were to: 1) Reduce both instream and floodplain derived sediment sources by incorporating stabilization techniques that function naturally with the stream and decrease the amount of stress on streambanks and the channel perimeter; 2) demonstrate the use of natural stream stabilization techniques that will allow the stream to adjust slowly over time and be representative of a naturally dynamic stream system; 3) improve native fish habitat, particularly overwintering and migratory habitat for threatened bull trout, by improving the form and function of the river and adjacent riparian habitats and; 4) apply knowledge learned from monitoring of the Grave Creek Demonstration and Phase I Restoration projects to further advance and encourage techniques that function naturally with the stream system and minimize the introduction of large rock and foreign material (RDG 2003).

Stream restoration work began in September 2004 and proceeded through November. During this period River Design Group excavated approximately 3,050 feet of new channel including an average design bankfull width and depth of 50-76 and 2-2.8 feet, respectively. The resulting stream pattern design increased sinuosity (stream length divided by valley length) from 1.06 to 1.35, and subsequently increased total stream length from approximately 2,790 to 3,050 feet. During construction phase of this project, numerous structures were installed including 5 engineered log jams, 3 straight log vanes, 5 log J-hook vanes, 2 rootwad composites, 3 cobble grade control structures, and 8 deflector log composites to provide bank stabilization, gradient control and pool habitat. However, severe icing conditions in lower Grave Creek in the winter of 2005/2006 and high spring flows in 2006 damaged some of the previously completed work, and as a result, substantial maintenance and revegetation work to this section of Grave Creek was completed in the fall of 2006. This work, like the original project, was a cooperative effort, with this project funding a portion of the work described below.

The purpose and need for maintenance and revegetation activities conducted in the fall of 2006 was to establish bank and floodplain vegetation, increase streambank resistance to erosion, and lower the risk of Grave Creek capturing the historical (pre restoration) avulsion channels. Overall, the maintenance and revegetation work involved minimal

channel reshaping compared to original project construction. Stream channel reshaping occurred over approximately 725 feet in 2006 (Table 4). The majority of the work involved supplementing existing structures and adding vegetated soil lifts to promote willow establishment and bank stability. The maintenance work added a total of seven engineered debris jams over within the Phase II Project Area, and eight vegetated soil lifts over a total of 530 feet of stream bank (Table 4). A straight log vane was added that was not part of the original as built design at station 1500 feet, and a single log j-hook vane was reset after flows the previous winter scoured it. This log vane was located in Meander F at station 2525.

Table 4. Structures types, meander and station of the maintenance work performed on the Grave Creek Phase 2 Project in the fall of 2006.

Structure Type	Meander	Station (feet)
Engineered Debris Jam	A	800-820
Vegetated Soil Lift	A	820-880
Engineered Debris Jam	A	880-905
Vegetated Soil Lift	A	905-1000
Add Wood to existing Engineered Debris Jam	A	1000-1020
Add floodplain grade control and microtopography	A	Right Floodplain
Re-shape channel (w/d ratio of 18)	C-D	1400-1725
Regrade floodplain, construct sills, add roughness	C-D	1400-1725
Straight log vane w/ engineered debris jam	C-D	1500
Vegetated Soil Lift	C-D	1500-1550
Engineered Debris Jam	C-D	1550-1575
Vegetated Soil Lift	C-D	1575-1675
Engineered debris jam	F	2375-2400
Re-shape channel (w/d ratio of 18)	F	2600-3000
Fill / reconstruct left bank line to original design	F	2800-3000
Re-set log j-hook vane	F	2525
Vegetated soil lift (left bank)	F	2475-2550
Engineered debris jam	F	2725-2750
Vegetated soil lift (left bank)	F	2750-2800
Engineered debris jam	F	2800-2825
Vegetated soil lift (left bank)	F	2850-2900
Engineered debris jam	F	2900-2950
Vegetated soil lift	F	2950-3000

MFWP has annually monitored this section of Grave Creek since 2004 with the intent of quantifying the physical changes to the stream channel as a result of original restoration work, maintenance activities and the overall sustainability of those changes through time. The monitoring results from 2006 were completed after the maintenance activities were completed, but prior to the spring freshet. The stream channel profile prior to project construction contained was over widened, often braided and contained only three pools (Dunnigan et al. 2005). The designed channel profile required excavation at numerous depositional areas throughout the project reach (Figure 2) and resulted in an increased quantity of pool habitat within the project area. Prior to project construction, the mean pool-to-pool distance was 930 feet. Mean pool spacing in 2006 was 246 feet, which resulted in a 73.5% reduction from the existing conditions. Pool spacing in 2007 decreased to 111 feet, which represented the lowest level since monitoring began. This was primarily due to the overall increase in the total number of pools present within the project reach. Pool spacing measurements during both years represented a complete sampling of all pools present. A longitudinal profile was surveyed for this section of Grave Creek annually since 2004 with the exception of 2006 (Figure 2).

Due to the importance of pool habitat to salmonids inhabiting Grave Creek, we devoted a substantial effort to monitor pool habitat before and after project construction to evaluate whether the restoration and maintenance efforts increased the quantity and quality of pool habitat within the project area. Prior to the initiation of this project in the summer of 2004, we measured bankfull depth, width, length, maximum bankfull depth, total area and total volume of the 3 pools within the project area. We repeated these measurements on all existing pools each year thereafter following the spring freshets. We did not perform a statistical comparison for these data because the pool measurements represented all pools within the project area (i.e. complete census), making statistical comparisons unnecessary. The initial restoration effort overwhelmingly increased the total number of pools present in this section of Grave Creek (Table 5), and even though we observed a slight decrease the following two years after the project was initially constructed, the number of pools was remained stable through 2006. This included a total count of pools after the winter of 2005/2006 that damaged some of the structures within the project area. However, after the spring freshet of 2007, we observed an increase in the total number of pools from 10 in 2005 and 2006 to 16 (60%; Table 5). This increase is partially attributed to the installation of seven engineered debris jams that were installed during the maintenance activities in the fall of 2006, which scoured 2 new pools during the 2007 spring freshet.

We observed changes in channel dimensions within pool habitats through time. Pools that resulted from this construction project through 2006 were slightly shorter than existing prior to the project. However in 2007, mean pool length increased to 79.2 feet, which was slightly longer than existed prior any restoration activity (Table 5). Mean width decreased slightly in 2007 from the previous two years (19.5 and 13.3%, respectively; Table 5). Despite the slight decrease in mean width in 2007, the increase in the total number of pools and pool length resulted in an overwhelming increase in total pool area and volume in 2007, which even exceeded as built conditions in 2004 by approximately 36% and 18%, respectively (Table 5). The increases in total pool area and volume were even greater in 2007 relative to 2005 and 2006

conditions. Total existing pool area in 2007 increased by approximately 45 and 86% relative to 2005 and 2006, respectively. Total pool volume in 2007 increased slightly more than did pool area, relative to 2005 and 2006 (Table 5). Both mean and maximum pool depths decreased slightly in 2007 from 2006 (Table 5). However the slight decrease in pool depth had little impact on pool volume. Our monitoring efforts associated with restoration project demonstrate that the quantity of salmonid rearing habitat was increased over existing conditions, and that these changes were self-sustaining even up until the time that maintenance activities were performed in the fall of 2006. The maintenance activities installed several features that further enhanced the quantity of pool habitat within this section of Grave Creek. Furthermore, although we did not attempt to quantify pool cover complexity for salmonids, our field observations strongly suggested that this project also increased the quality of rearing habitat for salmonids within this section of Grave Creek.

Figure 2. The longitudinal profile for the Grave Creek Phase II Restoration Project. Data are presented for pre-project (2004), as-built conditions (2004), 2005 and 2007. The station (longitudinal location measured at the channel thalweg) begins at the upstream boundary of the project.

We also monitored the stream channel dimensions within riffle habitats before and after restoration efforts. During the summer of 2004, we measured stream channel morphology at 3 cross-sectional survey locations in riffle habitat within the project area in order to characterize the stream channel dimensions in the riffle type habitat prior to project construction (existing). After project construction during the fall of 2004 we surveyed every riffle within the newly completed project area (as built). We surveyed every riffle in 2005 -2007, after the project had experienced the first spring freshet each year. At each transect we measured mean bankfull width, depth, width to depth ratio, and cross sectional area. We used analysis of variance (ANOVA) and a subsequent multiple comparison test (Tukey Test; Zar 1996) to test for significant differences between existing and all subsequent years after construction ($\alpha = 0.05$; Table 6). Statistical comparisons between 2004 (as built) and all subsequent years riffle dimensions were unnecessary since these measurements were performed on all riffle habitats within the project area (i.e. complete census). Mean bankfull width and width to depth ratio were significantly reduced from existing conditions when compared to 2004-2007 conditions, with decreases ranging from 39.7 to 43% for width and 52.5% to 61.8% for width to depth ratio (Table 6). Mean bankfull width increased slightly in 2005, from as built conditions, and then remained similar in 2006, but then decreased to within 2.4% of the as built conditions (Table 6). Maximum and mean bankfull depth increased as a result of the restoration activities, and then decreased annually from 2004. We were unable to declare any of the post restoration annual differences in mean or maximum depth significantly different from mean depths prior to restoration even though increases in mean depth increased by over 27% (Table 6). Our lack of power in this test was most likely due to the small number of riffles sampled (3) to characterize the existing conditions prior to restoration activities. We observed a similar trend for cross sectional area after project completion, with mean cross sectional area decreasing each year after project completion. However, the only two comparisons that differed significantly between existing dataset were for 2004/2006 and 2004/2007 (Table 6). The Grave Creek Phase II Restoration Project created a significantly deeper and narrower stream channel within the riffle habitats, with these changes being self-sustaining after project completion. The maintenance activities conducted in the fall of 2006 had little effect on stream channel dimensions. Stream channel cross sectional area exhibited the highest relative change from 2006 to 2007, decreasing by approximately 15%, but remaining lower than existed before restoration by approximately 32% (Table 6). The decrease in cross sectional area between 2006 and 2007 was most attributable to a decrease in mean bankfull depth between years by approximately 13% (Table 6). All other riffle dimensions changes only slightly between 2006 and 2007, with changes generally less than 5% (Table 4). Riffle slopes were not measured in 2004, but have remained relatively stable since the project was completed, especially between 2006 and 2007, in which case decreased from 1.2% to 1.2% (6%; Table 6).

Table 5. Mean bankfull width, maximum bankfull depth, and mean length, total length and surface area measured from pools located in the Grave Creek Phase II Project. The project area was surveyed in the summer of 2004, prior to project implementation (existing), the fall of 2004 after the project was completed (as built), and in 2005 -2007 after the spring freshets. Variance estimates for annual mean values are presented in parentheses. A statistical comparison of annual mean values was not performed because all pools within the project area were surveyed, and therefore represents a complete census. The percent change for each parameter year to year is also presented.

	Number of Pools	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Mean Length (ft.)	Total Area (ft²)	Total Volume (ft³)
2004 (Existing)	3	77.0 (48.0)	2.0 (0.1)	4.4 (1.7)	78.7 (646.3)	18,236	37,570
2004 (As Built)	14	59.0 (344.0)	2.9 (0.7)	5.6 (1.5)	57.1 (421.9)	46,252	141,092
2005	10	62.3 (72.6)	2.5 (0.1)	5.2 (1.1)	70.9 (452.3)	43,629	108,993
2006	10	57.8 (238.3)	2.6 (0.4)	5.4 (1.5)	59.1 (74.1)	33,884	87,768
2007	16	50.1 (46.9)	2.5 (0.5)	4.9 (1.3)	79.2 (1,036.8)	63,050	166,316
Percent Change							
Exist/2004	366.7%	-23.4%	43.9%	28.6%	-27.5%	153.6%	275.6%
Exist/2005	233.3%	-19.1%	23.5%	18.2%	-9.9%	139.2%	190.1%
Exist/2006	233.3%	-24.9%	30.0%	22.7%	-24.9%	85.8%	133.6%
Exist/2007	433.3%	-34.9%	25.6%	11.8%	0.6%	245.7%	342.7%
2004/2005	-28.6%	5.6%	-14.2%	-8.1%	24.2%	-5.7%	-22.8%
2004/2006	-28.6%	-2.0%	-10.3%	-3.6%	3.5%	-26.7%	-37.8%
2004/2007	14.3%	-15.0%	-13.4%	-12.2%	38.7%	36.3%	17.9%
2005/2006	0%	-7.2%	4.0%	3.8%	-16.6%	-22.3%	-19.5%
2005/2007	60.0%	-19.5%	0.5%	-5.4%	11.7%	44.5%	52.6%
2006/2007	60.0%	-13.3%	-3.4%	-8.9%	34.0%	86.1%	89.5%

Table 6. Mean bankfull width, depth, maximum bankfull depth, cross sectional area, width to depth ratio, and slope of riffles located in the Grave Creek Phase II Restoration Project. All riffles were surveyed in 2004-2007, but only sub-sampled in 2004 (existing). Variance estimates for annual mean values are presented in parentheses. An analysis of variance was performed for each parameter, subsequent multiple comparisons were performed using the Tukey Test. Significant comparisons are indicated via * ($\alpha < 0.05$).

	Sample Size	Mean Bankfull Width (ft)	Maximum Bankfull Depth (ft)	Mean Bankfull Depth (ft.)	Cross Sectional Area (sq. ft.)	Width to Depth Ratio	Riffle Slope
2004 (Existing)	3	100 (1657)	2.83 (0.04)	1.44 (0.11)	135.3 (382.3)	77.1 (2726.0)	Not collected
2004 (As Built)	7	58.2 (36.7)	3.17 (0.29)	2.03 (0.12)	118.3 (495.9)	29.4 (42.2)	0.0108 (6.3×10^{-6})
2005	8	60.3 (52.6)	3.01 (0.30)	1.91 (0.16)	113.0 (182.6)	33.8 (168.1)	0.009 (2.08×10^{-5})
2006	9	60.1 (152.7)	2.95 (0.21)	1.84 (0.16)	106.9 (182.7)	35.9 (327.0)	0.012 (7.9×10^{-6})
2007	9	56.8 (48.6)	2.81 (0.30)	1.61 (0.11)	91.5 ((505.9)	36.6 (76.4)	0.011 (1.1×10^{-5})
P-value		0.001	0.692	0.091	0.008	0.009	0.372
Percent Change							
exist/2004	133%	-41.8% *	11.9%	40.9%	-12.6%	-61.8% *	N/A
exist/2005	166.7%	-39.7% *	6.3%	32.5%	-16.5%	-56.1% *	N/A
exist/2006	200%	-39.9% *	4.2%	27.8%	-21.0% *	-53.4% *	N/A
exist/2007	200%	-43.0%	-0.8%	11.5%	-32.4% *	-52.5% *	N/A
2004/2005	14.3%	3.6%	5.0%	-6.0%	-4.5%	14.8%	-17.0%
2004/2006	28.6%	3.3%	-6.9%	-9.5%	-9.6%	21.8%	9.7%
2004/2007	28.6%	-2.4%	-11.4%	-20.9%	-22.7%	24.4%	3.2%
2005/2006	12.5%	-0.3%	-2.0%	-3.7%	-5.4%	6.0%	32.1%
2005/2007	12.5%	-5.8%	-6.7%	-15.9%	-19.1%	8.3%	24.3%
2006/2007	0%	-5.6%	-4.7%	-12.6%	-14.5%	2.1%	-5.9%

Libby Creek Upper Cleveland Project

MFWP completed the Libby Creek Upper Cleveland Stream Restoration Project in the fall of 2002 (approximate river mile 22), which restored approximately 3,200 feet of stream channel to the proper dimension, pattern and profile (Dunnigan et al. 2005). Past land management activities including logging, mining, riparian road construction, and stream channel manipulation have resulted in accelerated bank erosion along a number of meander bends, resulting in an over widened, unstable, and shallow channel (Sato 2000), which has resulted in low quality habitat for native salmonids including bull trout and redband trout. The existing channel prior to this restoration project was over-widened with frequent lateral migration of the active stream channel. These conditions resulted in frequent multiple channels within the project reach (Dunnigan et al. 2004). Width depth ratios were high and bankfull channel depths were shallow.

Dunnigan et al. (2004; 2005; 2007; 2008) demonstrated that this restoration project decreased the bankfull width and bank erosion and increased stream depth, overall length, substrate mean particle size, and the quality and quantity of salmonid rearing habitat through 2006. However, during the first week of November 2006, the Libby Creek watershed experienced a rain on snow weather event that created higher than average runoff conditions throughout the entire watershed including the headwater regions. The US Forest Service gauged the peak flows during this event at 3,093 cubic feet per second, which translated to a 19-year return interval using the Log-Pearson type III Flood Frequency Analysis (J. Boyd, US Forest Service, personal communication). Therefore, this document evaluates changes in the physical habitat within this section of Libby Creek after this event by comparing conditions in 2007 to those existing before restoration (1999) and after restoration (2002-2006).

We surveyed the stream restoration project area before (1999) and after (2002-2007) using a Nikon Model DTM-420 Total Station Survey Instrument, which records the geo-referenced location of stream channel features. The most conspicuous changes within this section of Libby Creek as a result of the November 2006 flow event were changes in the stream plan form (Figures 3-5). The restoration work increased the stream length within the project area by approximately 900 feet by increasing sinuosity. Total stream length in 1999 was 2,369 feet, and increased to 3,270 as a result of restoring meander frequency (Figures 3-6). The rain on snow event of November 2006 created two chute cut offs within the project area at stations 12+50-14+50 and 21+50-24+00 feet (Figures 3-5). These chute cut offs reduced total stream length within the project area in 2007 to 3,181 feet. The reduction represented a reduction of 90 feet (approximately 3%) relative to the conditions that existed in 2006. However, the total stream length within the project area after the November 2006 flow event remained 812 feet longer (34%) than existed prior to the restoration work in 1999 (Figures 3-6). The stream channel in 1999 existed in a braided configuration at two locations, which represented approximately 30% of the stream length within this section (Figures 3-5). We constructed a single thread stream channel that remained single thread even after the flow event of November 2006 (Figures 3-5).

We performed total station surveys by surveying the stream channels prior to and after restoration efforts, and based on substantial changes in the planform or longitudinal profile after the 2006 flood event, we identified five stations (based on the 2007 channel profile) within the project area where substantial deposition or erosion occurred (Table 7). At each of these stations, we performed cross sectional surveys in the middle of pools, riffles between pools, and at the top and bottom of pool tail out areas. We also surveyed the upper and lower portions of all visually apparent depositional or erosional features. These surveys allowed us to compare changes in bed material before and after the flood event at each depositional or erosional feature. We used Auto Cad software to measure surface areas of depositional or erosional features, and WinX Pro software to measure the cross sectional area of these features. We tabulated the volume of bed material deposited and eroded for each of the five stations (Table 7). We estimated a total of 29,023 ft³ of bed material was eroded within the project area, whereas, a total of 26,827 ft³ of bed material was deposited, for a net difference of 2196 ft³ (Table 7). Not surprisingly, the majority of the net erosion within the project area occurred within the three reaches where chute cut offs and lateral stream channel migration occurred (Table 7). Typically these areas of major channel erosion were in relative close proximity to large areas of deposition in the 2006 channel. In many instances where the 2006 stream channel was either abandoned via chute cut off, or the channel laterally migrated, deposition in the 2006 stream channel occurred, to a level that approximated bankfull elevation. This situation contributed to the relative balance between deposition and erosion within the project area (Table 7), and based on this observation, it seems unlikely that erosion or deposition within the project area substantially contributed to the downstream instability.

We surveyed riffles within the project area to evaluate changes in stream channel dimensions from 1999-2007. Cross sectional surveys were performed at the longitudinal mid-point of each riffle, where we measured mean bankfull width, depth, width to depth ratio, and cross sectional area. We also measured riffle slope of all riffles present within the project area in 2002-2007. We used analysis of variance (ANOVA) and a subsequent multiple comparison test (Tukey Test; Zar 1996) to test for significant differences between years ($\alpha = 0.05$; Table 8). The stream channel dimensions changed most within the riffle-type habitats relative to the pool habitats (see below) after the November 2006 flow event. The most apparent change within the riffle habitats was mean bankfull width. The restoration work significantly decreased mean riffle width (Table 8), but after the November 2006 flow event, mean riffle width did not significantly differ from conditions existing in 1999 (Table 8). The fact that riffle cross sections were not surveyed in the braided sections in 1999 may have under estimated riffle width in 1999, since most braided sections were riffle habitat. Despite the increase in bankfull width, mean riffle depth remained similar after the major flow event. Mean bankfull width all years after project construction (2002-2007) remained significantly higher than existed in 1999 (Table 8), with little difference existing between years after the restoration work through 2007. The increased riffle width and maintenance of riffle depths observed in 2007 resulted in a slight increase in width to depth ratios that were intermediate between those that existed prior the project and those that existed after the project up until the large flow event (Table 8). The resulting statistical comparisons indicated that despite a slight increase in 2007, width to depth ratios remained significantly higher than those existing in 1999 (Table 8). The increased

bankfull width observed in 2007 had a similar effect on riffle cross sectional area. Cross sectional area within riffles significantly increased in 2007 over conditions existing prior to the project in 1999 (Table 8). Riffle mean cross sectional area also increased by 13.7–48.3% when compared to values observed in 2003–2007, respectively (Table 8). During these years a statistical comparison wasn't necessary due to a complete sample design of all riffles from 2003-2007. Riffle slope wasn't measured prior to project implementation in 1999, but remained similar from 2002-2007, ranging from 1.3-2.0% (Table 8). Riffle slopes measured in 2007 were intermediate (1.64%) compared to those measured in 2002-2006 (Table 8).

Due to the importance of pool habitat to rearing redband and bull trout within the project area, we also devoted substantial effort to monitor pool habitat within the project area in order to evaluate changes in pool spacing, numbers, depth (mean and maximum), width and length after project construction. We only measured pool spacing in 1999 prior to the restoration work. The restoration work completed in 2002 installed numerous structures that increased the quantity of pool habitat within the project area (Dunnigan et al. 2004), which is evident on the longitudinal profiles (Figure 6). The increased quantity of pools resulted in a decrease in mean pool-to-pool spacing after project construction. Mean pool spacing ranged between 2002-2006 ranged from 152 feet in 2002 to 191 feet in 2005 (Table 9). However, mean pool spacing increased to 223 feet in 2007, which represented an overall increase over the previous 5 year period, but remained 31.5% lower than existed prior to restoration work (Table 9). The primary reason pool spacing changed between years was the change in the number of pools between years. Only five pools existed in this section of Libby Creek prior to the restoration work, but increased approximately 4 fold as a result of the project in 2002 and 2003, but decreased slightly through 2006 (Table 9). The flow event that occurred in November 2006 resulted in the loss of 5 from the previous year, which represented a 29.4% decrease (Table 9). Despite this substantial decrease from the previous year, the total number of pools remained 140% higher than existed prior to the restoration work. The total number of pools in 2007 decreased as a result of two phenomena, the complete or partial filling of pool habitat with bed material and channel meander abandonment that resulted from the two chute cut offs that occurred as a result of the flood event (see above; Figures 3-6). We did not measure stream channel dimensions within pool habitat in 1999, but did measure them in 2002-2007. We established cross sectional surveys at the point within each pool where we measured maximum depth, mean bankfull width, depth, and cross sectional area. We calculated total pool surface area by multiplying mean length by mean bankfull width by the total number of pools present. We calculated total pool volume by multiplying total pool surface area by mean bankfull depth. Mean bankfull width, depth and maximum depth each decreased from existing conditions after the first project experienced the first channel forming bankfull flow (2003), but remained relatively stable between 2003-2006 (Table 9). However, after the flood event in November 2006, mean pool widths and depths increased to slightly less than existed immediately after project construction (as built conditions in 2002; Table 9). However, in 2007 mean pool length increased to the highest level since the project was constructed in 2002 (Table 9). Pool total surface area and volume decreased slightly after 2002, but increased in 2007 to levels that were nearly equivalent to those existing in 2002 (as built; Table 9). We didn't measure stream channel dimensions in pool habitats in 1999, but regardless we are confident that both total pool

area and volume within the project area in 2007 are higher than existed prior to restoration due to the increased number of pools that were maintained through 2007.

Figure 3. Plan view of the lower section (station 21+00 to 31+40) of Libby Creek Upper Cleveland Restoration Project showing the existing (1999), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 4. Plan view of the middle section (station 9+00 to 21+00) of Libby Creek Upper Cleveland Restoration Project showing the existing (1999), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 5. Plan view of the top section (station 0+00 to 9+00) of Libby Creek Upper Cleveland Restoration Project showing the existing (1999), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 6. The longitudinal profiles of the Libby Creek Upper Cleveland Restoration Project area for the pre-construction (upper; 1999) and after (lower) construction (2003 and 2005-2007). The survey begins at the upper project boundary (station 0) and proceeds downstream to the lower project boundary. Due to differences in stream length and pattern, the before and after figures could not be displayed on the same graphic.

Table 7. Estimated bed material volume (ft³) eroded and deposited within the Libby Creek Upper Cleveland Restoration Project area after the November 2006 flood event. Station locations were based on the 2007 channel profile.

Station	Erosion (ft³)	Deposition (ft³)	Net Eroded/Deposited (ft³)
4+30 - 6+60	682	3,599	2,917 Deposited
13+00 - 15+00	10,130	5,438	4,692 Eroded
15+50 - 17+00	8,441	7,709	732 Eroded
20+00 - 21+00	4,932	3,182	1,750 Eroded
22+00 - 24+00	4,838	6,899	2,061 Deposited
Total	29,023	26,827	2,196 Eroded

Table 8. Mean bankfull width, depth, maximum bankfull depth, cross sectional area, width to depth ratio, and slope of riffles located in the Libby Creek Upper Cleveland Restoration Project. Variance estimates for annual mean values are presented in parentheses. An analysis of variance was preformed for each parameter, and the P value is presented. Multiple comparisons were performed using the Tukey Test. Significant comparisons are indicated via * (alpha < 0.05). The riffle slope was measured for every riffle in the project area in 2002-2007.

	Sample Size	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Width to Depth Ratio	Cross Sectional Area (Square ft.)	Slope (%)
1999 (Pre)	5	41.5 (35.2)	0.94 (0.07)	47.6 (359.8)	39.6 (211.3)	Not Measured
2002 (As Built)	9	34.3 (30.5)	1.33 (0.09)	26.7 (59.0)	46.0 (114.3)	1.91 (7.499*10 ⁻⁵)
2003	9	31.5 (18.5)	1.48 (0.04)	21.8 (25.4)	47.9 (62.5)	1.28 (1.941*10 ⁻⁵)
2005	15	31.9 (32.8)	1.36 (0.05)	24.3 (65.0)	43.0 (26.7)	1.46 (2.548*10 ⁻⁵)
2006	15	28.3 (11.8)	1.31 (0.02)	22.1 (18.6)	36.7 (30.0)	1.96 (2.57610 ⁻⁵)
2007	10	40.9 (37.8)	1.34 (0.11)	33.0 (96.0)	54.4 (188.9)	1.64 (2.67610 ⁻⁵)
P-Value		>0.001	0.009	>0.001	0.001	N/A
Percent Change						
1999/2002		-17.4%	41.5%*	-44.0%*	16.0%	N/A
1999/2003		-24.1%*	59.6%*	-54.2%*	20.9%	N/A
1999/2005		-23.1%*	44.4%*	-48.9%*	8.5%	N/A
1999/2006		-31.8%*	39.4%*	-53.6%*	-7.3%	N/A
1999/2007		-1.4%	42.6%*	-30.7%*	37.5%*	N/A
2002/2003		-7.9%	12.8%	-18.3%	4.3%	-33.0%
2002/2005		-7.0%	2.1%	-8.8%	-6.5%	-23.2%
2002/2006		-17.5%	-1.5%	-17.2%	-20.2%	2.6%
2002/2007		19.6%	0.8%	23.6%	18.3%	-14.1%
2003/2005		1.2%	-9.5%	11.7%	-10.3%	14.5%
2003/2006		-10.2%	-12.7%	1.4%	-23.4%	53.1%
2003/2007		29.9%	-10.7%	51.4%	13.7%	28.1%
2005/2006		-11.3%	-3.5%	-9.1%	-14.7%	34.2%
2005/2007		28.4%	-1.3%	35.6%	26.6%	12.3%
2006/2007		44.6%	2.3%	49.3%	48.3%	-16.3%

Table 9. Pool dimensions within the Libby Creek Upper Cleveland Restoration Project including mean bankfull width, depth, maximum bankfull depth, length and total volume from 1999-2007. Variance estimates for annual mean values are presented in parentheses. A statistical comparison of annual mean values was not performed because all pools within the project area were measured, and therefore represents a complete census. The percent change for each parameter from year to year is also presented. Mean bankfull depth was used to calculate total volume.

	# Pools	Pool Spacing (ft)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Length (ft.)	Total Area (ft ²)	Total Volume (ft ³)
1999 (Existing)	5	325	N/A	N/A	N/A	N/A	N/A	N/A
2002 (As Built)	20	N/A	38.0 (23.8)	2.6 (0.8)	4.3 (1.1)	36.7 (205.2)	27,892	72,519
2003	20	173	34.5 (16.1)	2.2 (0.3)	3.8 (0.7)	30.2 (130.8)	20,838	45,843
2005	18	191	28.8 (31.7)	1.8 (0.2)	3.9 (0.9)	36.9 (75.69)	19,129	34,432
2006	17	152	28.1 (40.1)	1.9 (0.2)	3.8 (0.8)	46.6 (109.5)	22,261	42,296
2007	12	223	36 (144.5)	2.1 (0.3)	4.0 (0.4)	55.4 (361.2)	24,531	50,534
Percent Change								
1999/2007	140.0%	-31.5%	N/A	N/A	N/A	N/A	N/A	N/A
2002/2003	0.0%	N/A	-9.2%	-18.01%	-12.03%	-17.67%	-25.3%	-36.8%
2002/2005	-10.0%	N/A	-24.1%	-27.96%	-9.3%	0.46%	-31.4%	-52.5%
2002/2006	-15.0%	N/A	-26.0%	-26.9%	-11.6%	27.0%	-20.2%	-41.7%
2002/2007	-40.0%	N/A	-2.8%	-21.9%	-7.7%	50.8%	-12.1%	-31.3%
2003/2005	-10.0%	10.4%	-16.5%	-12.14%	2.63%	22.02%	-8.2%	-24.9%
2003/2006	-15.0%	12.1%	-18.6%	-13.6%	0%	54.3%	6.8%	-7.7%
2003/2007	-40.0%	28.9%	7.0%	-4.7%	4.5%	83.2%	17.7%	12.1%
2005/2006	5.6%	-20.4%	-2.4%	5.5%	-2.6%	26.3%	16.4%	22.8%
2005/2007	-33.3%	16.8%	28.1%	8.4%	1.8%	50.1%	28.2%	39.0%
2006/2007	-29.4%	46.7%	31.3%	8.4%	-4.5%	18.9%	10.2%	19.5%

Libby Creek Lower Cleveland Phase I Project

The lower Cleveland property on Libby Creek is located approximately 1 mile downstream of the upper Cleveland Property, near the original Libby town site, and has been identified by MFWP as a high priority site for stream restoration. 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, which has resulted in low quality habitat for native salmonids including bull trout and redband trout. The present length of Libby Creek through the entire lower Cleveland property is approximately 9,100 feet. MFWP proposes to implement the restoration of this large site in 3 phases. The Libby Creek Lower Cleveland Phase I Project (approximate river mile 21-20), completed a stream restoration project in this area in October 2005 during which we excavated approximately 2,950 feet of new channel according to the design criteria including an average design bankfull width and depth of 32 feet and 3 to 7 feet, respectively. Dunnigan et al. (2007) presents a complete description of the materials and structures installed in this section of Libby Creek.

During the first week of November 2006, only a few weeks after the construction of this phase of the restoration work was completed, the Libby Creek watershed experienced a rain on snow weather event that created higher than average runoff conditions throughout the entire watershed including the headwater regions. This storm event caused substantial damage to the restoration work accomplished on this project phase as well as work previously completed within the watershed (see above). US Forest Service gauged the peak flows during this event at a minimum flow of 3,093 cubic feet per second, which translated to a 19-year return interval using the Log-Pearson type III Flood Frequency Analysis (J. Boyd, US Forest Service, personal communication). This report summarizes monitoring completed during the summer of 2007 and compares those results to conditions that existed prior to the restoration work (2004) and the as built conditions at the completion of the restoration activities in 2005 and after the first spring freshet in 2006.

We surveyed the stream restoration project area before (2004), October 2005 (as built), 2006, and 2007 using a Nikon Model DTM-420 Total Station Survey Instrument, which records the geo-referenced location of stream channel features including channel dimensions, profile and planform. The most conspicuous changes within this section of Libby Creek as a result of the November 2006 flow event were changes in the stream plan form (Figures 7 and 8). We identified six stations (based on the 2007 channel profile) within the project area where substantial deposition or erosion occurred (Table 10). At each of these stations, we performed cross sectional surveys in the middle of pools, riffles between pools, and at the top and bottom of pool tail out areas. We also surveyed the upper and lower portions of all visually apparent depositional or erosional features. These surveys allowed us to compare changes in bed material before and after the flood event at each depositional or erosional feature. We used Auto Cad software to measure surface areas of depositional or erosional features, and WinX Pro software to measure the cross sectional area of these features. We tabulated the volume of bed material deposited and eroded for each of the six stations (Table 10). We estimated a total of 154,261 ft³ of bed material was

eroded within the project area, where as, a total of 88,060 ft³ of bed material was deposited, for a net difference of 66,202 ft³ (Table 10). Not surprisingly, the majority of the net erosion within the project area occurred within the three reaches where chute cut offs and lateral stream channel migration occurred (Table 10; Figures 7 and 8). In many instances where the 2006 stream channel was either abandoned via chute cut off, or the channel laterally migrated, deposition in the 2006 stream channel occurred, to a level that approximated riffle elevation. This situation differed from the Upper Cleveland Restoration Project area where filling occurred to the approximate bankfull elevation of the abandoned channel (see above). Based on these calculations, seems unlikely that the bed material eroded from this section of Libby Creek had the potential to contribute to instability downstream sections of Libby Creek.

Table 10. Estimated bed material volume (ft³) eroded and deposited within the Libby Creek Lower Cleveland Phase I Restoration Project area after the November 2006 flood event. Station locations were based on the 2007 channel profile.

Station	Erosion (ft ³)	Deposition (ft ³)	Net Eroded/Deposited (ft ³)
6+00 - 8+50	24,226	14,860	9,366 Eroded
9+00 - 11+00	13,534	15,617	-2,083 Deposited
13+00 - 14+50	19,556	21,630	-2,074 Deposited
15+50 - 18+25	25,174	15,247	9,927 Eroded
20+00 - 22+00	34,695	7,934	26,761 Eroded
22+50 - 25+85	37,076	12,772	24,304 Eroded
Total	29,023	26,827	66,201 Eroded

We measured total stream length during all years along the channel thalweg. The stream channel length in 2004 was 2,695 feet (Figure 6), and the restoration work increased stream length to 2,793 feet, representing a 3.6% increase due to increased sinuosity (1.24 and 1.30, respectively) due to increased meander frequency (Figures 7 and 8). However, as a result of several chute cutoffs that occurred throughout the project area during the November 2006 flood event, the stream channel lost 274 feet (2,519 feet total length), representing a loss of 9.8% relative to as-built conditions. Stream length after the flood event (2007) was 176 feet shorter (6.5%) than existed prior to the project. Sinuosity decreased to 1.16 in 2007.

The designed channel profile required excavation at numerous depositional areas throughout the project reach, and resulted in an increased quantity of pool habitat within the project area, which is clearly present on the 2005 longitudinal profile survey (Figure 9). Prior to project construction, the mean pool-to-pool distance was 811 feet. The newly constructed channel (2005) mean pool-to-pool spacing was 152 feet, representing an 81.3% reduction from existing conditions. We found that mean pool-to-pool spacing increased in 2007 to 259 feet, representing an increase of 70.4% compared to as built (2005) conditions,

but remained 68.1% lower than conditions existing prior to restoration work in 2004. The increased frequency of pool type habitat within the project area had an overwhelming effect on the total number of pools and the total surface area and volume of pool habitat within the project area. The total number of pools present increased by 3.5 fold compared to existing conditions. However, all but six pools were lost as a result of the November 2006 flood event, representing an overall reduction of 66.7% from as built conditions (Table 11), but remained 50% higher than existed prior to restoration. The increased number of pools that resulted from the restoration work also translated into sharp increases in total pool area and volume, increasing by approximately 2.9 and 4.2 fold, respectively relative to existing conditions (Table 11). Although the flood event of fall 2006 diminished total pool area and volume within this section of Libby Creek relative to both 2005 and 2006 conditions, both parameters remained substantially higher than existed prior to the restoration work (1.4 and 2.0 fold, respectively; Table 11). Mean bankfull width, depth and maximum depth in 2007 were similar to as built conditions in 2005, exhibiting less than 5% change. Mean pool length increased as a result of the flood event, increasing by an average almost double to as built conditions and approximately 78% relative to existing conditions.

In addition to a complete census of all pools within the project area, we also surveyed riffles habitats within the project area to evaluate changes in riffle dimensions as a result of the restoration effort and the November 2006 flood. We measured mean bankfull width, depth, and width to depth ratio, of riffles throughout the project area in 2004-2007, and we used an analysis of variance and subsequent multiple comparison (Tukey Test) to test for significant differences between years ($\alpha = 0.05$; Table 12). The restoration work we performed created a significantly narrower stream channel within the riffle habitats in 2005. Mean bankfull width within the riffles decreased from 69.8 feet to 34.1 feet (51.1%) from 2004 to 2005 (Table 12), and remained similar between 2005 and 2006. However, as a result of the 2006 flood event, riffle widths in 2007 increased to an average of 45.2 feet, but remained significantly and substantially lower than existed prior to the restoration work (Table 12). Mean riffle depth in 2007 was nearly the same as it was in 2005, but not significantly lower than we observed in 2005 or 2006. The overall effect of decreased riffle width during all years following the restoration resulted in significant differences between 2005 and all following years (Table 12). Mean width to depth ratio did not significantly differ between 2005, 2006, or 2007, even though it increased by approximately half from 2006 to 2007 (Table 12). We did not survey riffle slopes in 2004. However, mean riffle slope did not change significantly between 2005 and 2007, ranging from 2.1 to 2.4% (Table 12).

Figure 7. Plan view of the lower section (station 11+00 to 25+00) of Libby Creek Lower Cleveland Phase 1 Restoration Project showing the existing (2001), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 8. Plan view of the upper section (station 0+00 to 11+00) of Libby Creek Lower Cleveland Phase 1 Restoration Project showing the existing (2001), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 9. The longitudinal profile of the Libby Creek thalweg within the Lower Cleveland Phase I Restoration Project surveyed in 2001 (existing), 2005 (as built), 2006, and 2007. The survey was conducted beginning at station 0 (upper project boundary) to the downstream project boundary. Stream channel length varied between years due to differences in channel sinuosity.

Table 11. Pool dimensions including mean bankfull width, depth, maximum bankfull depth, length and total volume in 2004 (pre-existing), 2005 (as built), 2006 and 2007 for the Libby Creek Lower Cleveland Phase I Restoration Project. Variance estimates for annual mean values are presented in parentheses. A statistical comparison of annual mean values was not performed because all pools within the project area were measured, and therefore represents a complete census. The percent annual change is also presented.

	Number	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Length (ft.)	Total Area (ft²)	Total Volume (ft³)
2004 (existing)	4	42.2 (44.2)	2.33 (0.34)	4.2 (1.05)	42.8 (131.6)	7,260	16,186
2005 (As Built)	18	39.6 (63.9)	2.84 (0.34)	5.32 (0.85)	38.8 (95.1)	28,249	84,023
2006	13	39.9 (721.6)	2.64 (0.13)	4.98 (0.79)	57.7 (591.2)	30,534	80,477
2007	6	38.4 (35.5)	2.83 (0.59)	5.20 (1.70)	76.1 (769.4)	17,733	48,344
Percent Change							
2004/2005	350%	-6.2%	22.29%	26.7%	9.3%	289.1%	419.1%
2004/2006	225%	-5.5%	13.3%	18.6%	35.0%	320.6%	409.6%
2004/2007	50%	-9.0%	21.6%	23.8%	78.0%	144.2%	198.7%
2005/2006	-27.8%	0.8%	-7.3%	-6.5%	48.8%	8.1%	-1.8%
2005/2007	-66.7%	-3.0%	-0.6%	-2.3%	96.2%	-37.2%	-42.5%
2006/2007	-53.9%	-3.8%	7.2%	4.5%	31.9%	-41.9%	-41.4%

Table 12. Riffle dimensions including mean bankfull width, depth, maximum bankfull depth and width to depth ratio in 2004 (pre-existing) and (as built) for the Libby Creek Lower Cleveland Phase I Restoration Project. Variance estimates for annual mean values are presented in parentheses. Analysis of variance was performed for each parameter, the P value is presented. Multiple comparisons were performed using Tukey's Test. Significant comparisons are indicated via * ($\alpha < 0.05$). The riffle slope was not measured in 2004.

	Sample Size	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Width to Depth Ratio	Slope
2004 (existing)	7	69.8 (695.9)	1.94 (0.20)	3.16 (0.07)	41.2 (305.3)	N/A
2005 (As Built)	9	34.1 (13.9)	2.21 (1.7)	3.39 (0.39)	15.9 (12.4)	0.024 (1.9×10^{-5})
2006	9	34.5 (22.1)	2.14 (0.06)	3.08 (0.11)	16.4 (15.0)	0.023 (5.85×10^{-5})
2007	7	45.2 (106.9)	1.95 (0.13)	2.97 (0.30)	24.6 (68.3)	0.021 (4.72×10^{-5})
P-value		<0.001	0.363	0.330	<0.001	0.571
Percent Change						
2004/2005	28.6%	-51.1% *	14.1%	7.3%	-62.2% *	N/A
2004/2006	28.6%	-50.6% *	10.3%	-2.5%	-60.3% *	N/A
2004/2007	0%	-35.2% *	-4.0%	-6.1%	-40.7% *	N/A
2005/2006	0%	1.1%	-3.2%	-9.2%	5.0%	17.2%
2005/2007	-22.2%	32.6%	-11.9%	-12.5%	56.8%	3.6%
2006/2007	-22.2%	31.2%	-8.8%	-3.6%	49.4%	-11.7%

Libby Creek Lower Cleveland Phase II Project

The lower Cleveland property on Libby Creek is located approximately 1 mile downstream of the upper Cleveland Property, and has been identified by MFWP as a high priority site for stream restoration, and consists of approximately 9,100 feet of stream channel. MFWP planned to implement the restoration of this large site in 3 phases. Phase I of this project was completed in the fall of 2005 (see above), and Phase II was completed in October 2006. 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, which has resulted in low quality habitat for native salmonids including bull trout and redband trout. The Libby Creek Lower Cleveland Phase II Project started at the downstream boundary of the Phase I project area and continued 3,175 feet downstream (see above). This project constructed a variety of structures intended to improve fish habitat and increase bank stability including 8 rootwad complexes, 22 J-Hook vanes, 3 log/rock cross vanes, 4 cobble gradient control structures, 5 vegetated soil lifts, and transplanted 700 native trees and shrubs.

During the first week of November 2006, only a few weeks after the construction of this phase of the restoration work was completed, the Libby Creek watershed experienced a rain on snow weather event that created higher than average runoff conditions throughout the entire watershed including the headwater regions. This storm event caused substantial damage to the restoration work accomplished on this project phase as well as work previously completed within the watershed (see above). The US Forest Service gauged the peak flows during this event at a minimum flow of 3,093 cubic feet per second, which translated to a 19-year return interval using the Log-Pearson type III Flood Frequency Analysis (J. Boyd, US Forest Service, personal communication). This report summarizes monitoring completed during the summer of 2007 and compares those results to conditions that existed prior to the restoration work (2004) and the as-built conditions at the completion of the restoration activities.

We surveyed the stream restoration project area before (2004), October 2006 (as built), and 2007 using a Nikon Model DTM-420 Total Station Survey Instrument, which records the geo-referenced location of stream channel features including channel dimensions, profile and planform. The most conspicuous changes within this section of Libby Creek as a result of the November 2006 flow event were changes in the stream plan form (Figures 10-12). Changes in Phase II more severe than either the Lower Cleveland Phase I or the Upper Cleveland Restoration Project areas. We identified seven stations (based on the 2007 channel profile) within the project area where substantial deposition or erosion occurred (Table 13). At each of these stations, we performed cross sectional surveys in the middle of pools, riffles between pools, and at the top and bottom of pool tail out areas. We also surveyed the upper and lower portions of all visually apparent depositional or erosional features. These surveys allowed us to compare changes in bed material before and after the flood event at each depositional or erosional feature. We used Auto Cad software to measure surface areas of depositional or erosional features, and WinX

Pro software to measure the cross sectional area of these features. We tabulated the volume of bed material deposited and eroded for each of the five stations (Table 13). We estimated a total of 182,157 ft³ of bed material was eroded within the project area, where as, a total of 171,609 ft³ of bed material was deposited, for a net difference of 10,548 ft³ (Table 13).

Table 13. Estimated bed material volume (ft³) eroded and deposited within the Libby Creek Lower Cleveland Phase II Restoration Project area after the November 2006 flood event. Station locations were based on the 2007 channel profile.

Station	Erosion (ft ³)	Deposition (ft ³)	Net Eroded/Deposited (ft ³)
27+00 - 29+00	18,880	10,372	8,508 Eroded
29+25 - 33+50	46,619	37,204	9,415 Eroded
33+50 - 40+00	28,399	30,510	2,111 Deposited
40+00 - 42+25	11,168	22,909	11,741 Deposited
44+00 - 45+75	19,846	28,407	8,561 Deposited
46+00 - 48+50	22,006	25,320	3,314 Deposited
49+00 - 53+00	35,239	16,887	18,352 Eroded
Total	182,157	171,609	10,548 Eroded

The stream channel profile prior to project construction consisted of a total of 2,632 feet of stream channel (Figures 10-12), and the restoration work increased stream length to 3,273 feet, representing a 24.4% increase due to increased sinuosity due to increased meander frequency (Figures 10-12). However, as a result of several chute cutoffs that occurred throughout the project area, the stream channel lost 423 feet (2,850 feet total length) after the flood event, representing a loss of 12.9% relative to as-built conditions. However, after the flood event, stream length remained 218 feet longer (8.3%) than existed prior to the project.

The designed channel profile required excavation at numerous depositional areas throughout the project reach, and resulted in an increased quantity of pool habitat within the project area, which is clearly present on the 2006 longitudinal profile survey (Figure 13). Prior to project construction, the mean pool-to-pool distance was 690 feet. The newly constructed channel mean pool-to-pool spacing was 123 feet, representing an 82.2% reduction from existing conditions. We found that mean pool-to-pool spacing increased in 2007 to 257 feet, representing an approximate doubling compared to as built (2006) conditions, but remained 62.7% lower than conditions existing prior to restoration work in 2004. The increased frequency of pool type habitat within the project area had an overwhelming effect on the total number of pools and the total surface area and volume of pool habitat within the project area. The total number of pools present increased by 5.8 fold compared to existing conditions. However, all but five pools were lost as a result of the November 2006 flood event, representing an overall reduction of 81.5% from as built conditions (Table 14), but remained slightly higher (25%) than existed prior to restoration. The increased number of pools that resulted from the restoration work also translated into sharp increases in total pool area and volume, increasing by approximately 2.1 and 3.5 fold,

respectively relative to existing conditions (Table 14). However, in addition to a decrease in the total number of pools, mean width, depth, and maximum depth also decreased as a result of the November 2006 flood (Table 14). The impacts of the flood event also deteriorated most pool habitat channel dimensions relative to conditions that existed prior to restoration. For example, mean pool width and length in 2007 were 13.8 and 29.2% lower than 2004 (Table 14), and although total number of pools, mean and maximum pool depth remained higher in 2007 than 2004 condition, the overall effect of the November 2006 flood event was a reduction in total pool area and volume within the project area.

In addition to a complete census of all pools within the project area, we also surveyed riffles habitats within the project area to evaluate changes in riffle dimensions as a result of the restoration efforts and the November 2006 flood. In 2004 (existing), 2006 (as built), and 2007, we measured mean bankfull width, depth, width to depth ratio, and cross sectional area of riffles throughout the project area. We used an analysis of variance and subsequent multiple comparison (Tukey Test) to test for significant differences between years ($\alpha = 0.05$; Table 15). The restoration work we performed created a significantly narrower and deeper stream channel within the riffle habitats in 2006. Mean bankfull width within the riffles decreased from 70.4 feet to 36.9 feet (47.6%), and mean bankfull depth decreased from 1.4 feet to just over 2 feet (45%) from 2004 to 2006 (Table 15). However, mean bankfull width, cross sectional area and width to depth ratio increased after the 2006 flow event, and even though these relative changes ranged from 27.6 to 31.3%, the only significant change was for cross sectional area within riffles between 2006 and 2007 (Table 15). Mean bankfull depth between 2006 and 2007 was similar (Table 15). Despite the changes in riffle dimensions that resulted from the flood event, riffles within this section of Libby Creek remained substantially and significantly deeper (51.9%) and narrower (32.3%) than existed prior to the restoration work (Table 15). These factors combined to have an overwhelming effect on width to depth ratio, which was 56.7% lower in 2007 than 2004. Mean riffle slope significantly increased from 1.5% in 2004 to 2.8% in 2006 (Table 15), but significantly decreased from 2006 to 2007 (1.9%). Although riffle slope in 2007 was 26.7% higher than existed prior to the restoration work, the difference was not significant.

Figure 10. Plan view of the lower section (station 45+00 to 53+00) of Libby Creek Lower Cleveland Phase 2 Restoration Project showing the existing (2001), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 11. Plan view of the middle section (station 36+00 to 45+00) of Libby Creek Lower Cleveland Phase 2 Restoration Project showing the existing (2001), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 12. Plan view of the lower section (station 25+50 to 36+00) of Libby Creek Lower Cleveland Phase 2 Restoration Project showing the existing (2001), 2006 and 2007 bankfull channels. The stations listed on the figure are positioned along the 2007 channel thalweg (not pictured).

Figure 13. The longitudinal profile of the Libby Creek thalweg within the Lower Cleveland Phase I Restoration Project surveyed in 2001 (existing), 2006 (as built), and 2007. The survey was conducted beginning at station 0 (upper project boundary) to the downstream project boundary. Stream channel length varied between years due to differences in channel sinuosity.

Table 14. Pool dimensions including mean bankfull width, depth, maximum bankfull depth, length and total volume in 2004 (pre-existing), and 2006 (as built), for the Libby Creek Lower Cleveland Phase II Restoration Project. Variance estimates for annual mean values are presented in parentheses. The percent annual change is also presented.

	Number	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Length (ft.)	Total Area (ft²)	Total Volume (ft³)
2004 (existing)	4	42.9 (75.1)	2.24 (0.21)	4.18 (0.70)	87.3 (1546.9)	15,600	35,035
2006 (As Built)	27	41.6 (26.3)	3.24 (0.38)	5.50 (1.11)	43.3 (77.3)	48,635	157,576
2007	5	37.0 (417.8)	2.61 (0.53)	4.92 (0.28)	61.8 (474.2)	10,595	27,496
Percent Change							
2004/2006	575%	-3.0%	44.6%	31.6%	-50.4%	211.5%	352.8%
2004/2007	25%	-13.8%	16.2%	17.8%	-29.2%	-32.1%	-21.5%
2006/2007	-81.5	-11.2%	-19.7%	-10.5%	42.7%	-78.2%	-82.7%

Table 15. Riffle dimensions including mean bankfull width, depth, maximum bankfull depth and width to depth ratio in 2004 (pre-existing), 2006 (as built), and 2007 for the Libby Creek Lower Cleveland Phase II Restoration Project. Variance estimates for annual mean values are presented in parentheses. Analysis of variance was preformed for each parameter and the P value is presented. Multiple comparisons were performed using Tukey's Test. Significant comparisons are indicated via * (alpha < 0.05).

	Sample Size	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Cross Sectional Area (ft²)	Width to Depth Ratio	Slope
2004 (existing)	4	70.4 (417.0)	1.4 (0.14)	95.3 (458.1)	54.3 (622.3)	1.5% (6.67*10 ⁻⁶)
2006 (As Built)	12	36.9 (26.8)	2.03 (0.05)	75.3 (217.5)	18.4 (6.6)	2.8% (0.007)
2007	6	47.6 (62.1)	2.1 (0.23)	98.9 (64.9)	23.5 (67.9)	1.9% (1.76*10 ⁻⁵)
P-value		<0.001	0.007	0.008	<0.001	0.001
Percent Change						
2004/2006		-47.6% *	44.4% *	-21.0%	-66.1% *	83.9% *
2004/2007		-32.3% *	51.9% *	3.8%	-56.7% *	26.7%
2006/2007		29.0%	5.2%	31.3% *	27.6%	-31.3% *

Young Creek State Lands Restoration Project

MFWP reconstructed 1,200 feet of the Young Creek stream channel in the fall of 2003 (Dunnigan et al. 2005). The Young Creek State Lands Restoration Project significantly changed the dimension, pattern and longitudinal profile of this section of Young Creek (see Dunnigan et al. 2005). The stream restoration project significantly ($p < 0.05$) reduced the mean width and width to depth ratio, and significantly increased the cross sectional area, maximum depth, and mean bankfull depth for both riffles and pools within the project area. The monitoring activities we conducted on this section of Young Creek since the initial project construction have been directed at determining if the stream channel maintained the pattern and dimensions relative to as built conditions in 2003.

The changes that occurred in the stream channel dimensions within the Young Creek State Lands Restoration Project area between 2004 and 2007 were relatively small. We measured the cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio within each riffle that existed within the project area before (2002), during (2003; as built), and after (2004-2007) project construction (Table 12). We established the transect location at each riffle at the longitudinal mid-point of each riffle. The total number of riffles within this section of Young Creek has remained relatively similar since the project was constructed, varying by no more than a single riffle between years. Mean cross sectional area, mean bankfull width, maximum bankfull depth, and width to depth ratio have also remained relatively similar to the constructed stream channel dimensions, within changes less than 10% between years (Table 16). We did not perform any statistical tests on these data due to the fact that these surveys were a complete census of all riffles within the project area. Therefore, given the data collected since project completion, it appears that the channel dimensions in the riffles of this project are continuing to maintain through time.

The Young Creek State Lands Restoration Project also increased the quality and quantity of pool habitat for resident salmonids, and these changes are being sustained three years after the project was completed. The total number of pools, total pool area and total pool volume, remain 750, 404.2, and 957.3% higher than existed in this section of Young Creek prior to the restoration work (Table 17). The large woody debris stems and root wads used during project construction also likely increased cover available to rearing and migrating salmonids within this reach of Young Creek. We measured the same 5 parameters that we measured at each riffle transect in addition to pool length. We established the transect location within each pool at the location of maximum depth. The results from our pool monitoring were similar to the results we observed in riffles. The total number of pool increased from 8 in 2003 to 14 in 2004 to 15 in 2005, and 17 in 2006 and 2007 (Table 17), primarily as a result of the formation of several new pools that formed within several of the meanders. However, the pool dimensions changed relatively little between years after the project construction, especially during the past three years. Stream channel dimensions with the pool changes within the past three years were generally $<5\%$ (Table 17). Despite the slight changes in pool dimensions since the project was completed, the constructed pool habitat continues to provide an improvement

in the amount of depth and cover that existed prior to the project (Table 17). As was the case with the riffle surveys, we did not perform any statistical tests on these data due to the fact that these surveys were a complete census of all riffles within the project area.

The stream restoration techniques we employed on this section of Young Creek increased channel diversity and stability, stream length, and sinuosity within the project area. Although we did not present a figure that displays the stream plan for this section of Young Creek, it has changed little since the project was completed. This project continues to meet the original objectives (Dunnigan et al. 2005) set forth for this project.

Table 16. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio measured for the total number of riffles 2002-2007 for the Young Creek State Lands Stream Restoration Project. The project was constructed in the fall of 2003. Variance estimates for annual mean values are presented in parentheses. The percent change between years is also presented.

	Number Of Riffles	Cross Sectional Area (ft²)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Width to Depth Ratio
2002 (Existing)	4	16.75 (1.58)	27.88 (22.73)	0.60 (0.008)	1.05 (0.017)	48.3 (239.6)
2003 (As Built)	10	21.99 (10.07)	16.3 (9.18)	1.24 (0.05)	1.99 (0.09)	13.7 (21.2)
2004	11	18.71 (6.25)	14.83 (3.63)	1.28 (0.07)	1.85 (0.13)	12.29 (17.30)
2005	11	21.93 (22.04)	16.14 (4.38)	1.37 (0.08)	1.79 (0.09)	12.34 (11.40)
2006	10	19.70 (14.06)	15.59 (4.65)	1.29 (0.12)	1.89 (0.14)	13.04 (22.47)
2007	10	(19.08 (15.51)	14.84 (4.04)	1.32 (0.14)	1.72 (0.12)	14.41 (25/10)
Percent Change						
2002/2005	175%	30.9%	-42.1%	128.3%	129.4%	-74.5%
2002/2006	150%	17.6%	-44.1%	115.0%	80.0%	-73.0%
2002/2007	150%	13.9%	-46.8%	121.6%	63.8%	-74.3%
2003/2005	10%	0.3%	-1.0%	-9.8%	10.6%	9.9%
2003/2006	0%	-10.4%	-4.4%	4.0%	-5.0%	-4.8%
2003/2007	0%	-13.2%	-8.9%	6.8%	-13.4%	-9.4%
2004/2005	0%	17.2%	8.8%	-3.4%	6.8%	0.4%
2004/2006	-9.0%	5.4%	5.1%	0.1%	2.2%	6.1%
2004/2007	-9.0%	-1.1%	-2.6%	3.0%	-7.9%	-0.9%
2005/2006	-9.0%	-10.2%	-3.4%	-5.8%	5.6%	5.7%
2005/2007	-9.0%	-13.0%	-8.0%	-3.4%	-4.0%	0.5%
2006/2007	0%	-3.2%	-4.8%	2.3%	-8.8%	-4.9%

Table 17. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, width to depth ratio, and total length measured for the total number of pools (n) 2002-2007 for the Young Creek State Lands Stream Restoration Project. The project was constructed in the fall of 2003. Variance estimates for annual mean values are presented in parentheses. The percent change between years is also presented.

	Number of Pools	Cross Sectional Area (ft²)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Total Area (ft²)	Total Volume (ft³)
2002 (Existing)	2	19.3 (3.1)	23.5 (24.5)	0.79 (0.005)	2.35 (0.13)	1,998	1,578
2003 (As Built)	8	37.7 (65.1)	21.8 (18.0)	1.73 (0.084)	3.23 (0.42)	8,480	14,671
2004	14	31.8 (37.0)	19.2 (24.7)	1.73 (0.23)	3.63 (0.53)	8,602	14,881
2005	15	29.1 (48.6)	17.8 (12.8)	1.71 (0.28)	3.08 (0.67)	8,218	14,053
2006	17	30.1 (135.6)	17.4 (9.8)	1.74 (0.32)	3.12 (0.40)	10,667	17,923
2007	17	28.7 (54.4)	16.8 (10.0)	1.75 (0.26)	3.04 (0.22)	10,090	16,544
Percent Change							
2002/2005	650.0%	51.0%	-24.5%	116.5%	31.1%	311.4%	790.6%
2002/2006	750.0%	56.1%	-25.4%	120.3%	32.7%	433.9%	1035.8%
2002/2007	750.0%	48.9%	-28.6%	121.7%	29.2%	404.2%	957.3%
2003/2005	87.5%	-22.9%	-18.6%	-1.0%	-4.6%	-3.1%	-4.2%
2003/2006	112.5%	20.2%	-20.1%	0.6%	-3.7%	25.8%	22.2%
2003/2007	112.5%	-23.9%	-23.1%	1.3%	6.3%	21.8%	19.1%
2004/2005	7.1%	-11.6%	-7.4%	-3.1%	-15.2%	-4.5%	-5.6%
2004/2006	21.4%	8.5%	-9.5%	-1.6%	-13.1%	24.0%	20.4%
2004/2007	21.4%	-12.8%	-12.8%	0.9%	-15.4%	2.6%	-1.9%
2005/2006	13.3%	3.4%	-1.9%	1.6%	1.2%	29.8%	27.5%
2005/2007	13.3%	-1.4%	-5.5%	2.2%	-1.5%	20.2%	21.5%
2006/2007	0%	-4.6%	-3.7%	0.6%	-2.6%	-5.4%	-7.7%

Therriault Creek Restoration Project

MFWP partnered with The Kootenai River Network (KRN), the USFWS Partners for Wildlife and the local landowner to complete the Therriault Creek Restoration Project during the summer of 2005. Prior to the restoration work, the lower section of Therriault Creek was extensively modified through land cover disturbance, riparian vegetation clearing, and physical stream straightening prior to the mid-1900s. These past activities resulted in an incised stream channel, accelerated bank erosion, channel degradation, and poor fish habitat. This project reconstructed a total of 9,100 feet of entirely new stream channel that will restore the proper dimension, pattern, and profile of the channel, which approximately doubled the stream length by increasing meander frequency. Cooperators for this project initiated restoration work in 2004 and completed the stream channel restoration work during the summer 2005. The goals for this restoration project were to 1. To reduce nonpoint source pollution to Therriault Creek and the Tobacco River through mitigation of chronic instream sources of sediment, 2. Eliminate an existing partial fish barrier (perched culvert), 3. Restore and create approximately 55 acres of prior converted wetland, and 4. Improve and increase fish habitat for resident fish species.

The stability of the channel is tied to the structure and composition of riparian vegetation, which provides rooting structure to maintain lateral channel stability by preventing accelerated lateral erosion. Initial revegetation efforts associated with the restoration work in 2005 included the installation of 5,000 riparian shrubs, 10,000 dormant willow cuttings and seeding of disturbed areas. However, poor survival of the plants installed in the initial phase of the restoration project prompted further work. MFWP believes that the completed stream channel restoration work created the hydrologic and geomorphic conditions necessary to sustain ecological processes that will support long term establishment and maintenance of a mosaic of riparian plant communities throughout the project area, but additional steps were needed to promote site conditions that favor native a plant community at this site. Therefore, MFWP accomplished several actions intended to improve containerized plant survival and also implemented additional revegetation treatments aimed at restoring site conditions capable of supporting native riparian woody vegetation along the restored Therriault Creek channel. The specific actions implemented in the fall of 2007 included those actions listed below, and are described in greater detail in Geum Environmental Consulting (2007).

1. Residual Shrub Protection:

This treatment included protecting approximately 500 surviving shrubs planted during initial (2004 and 2005) restoration activities at the site. Many of these surviving shrubs had been heavily browsed by ungulates or girdling by voles. In addition, most shrubs were being subjected to high levels of competition from aggressive grasses or other invasive species. Approximately 500 residual shrubs were protected from browse by securing rigid plastic mesh browse protectors and vole protectors around each plant. In addition, plants were protected from competition by installing 2' x 2' brush blankets around the base of each plant.

2. Containerized Planting:

This treatment included planting a total of 1,000, one-gallon shrubs and 50, five-gallon shrubs or trees containerized native shrubs in select locations along the channel.

Containerized plant materials included primarily one-gallon native shrubs and a lesser number of five gallon native plants (Table 18). Planting was concentrated in 'islands' along outer meander bends where woody vegetation is not establishing naturally. Islands of plants were installed throughout the project reach, primarily within the 'Pasture Grass' and 'Quackgrass' plant community types. Planting included a mix of early-successional species, such as willows, alder, snowberry and rose, that may be better suited for the high light environment. The species mix also included later-successional species, such as spruce, in anticipation of the time when site conditions will favor these species.

Each one-gallon shrub will also received a rigid plastic mesh browse protector and Tubex® vole guard to protect the shrub from browse and girdling. Browse protectors were at least 4 feet in height and constructed from rigid plastic netting which was secured in place with 2-inch square by 4-foot tall wooden posts and UV stabilized zip ties. Tubex® vole was at least 8 inches high with an adjustable diameter, and will be pushed or trenched into the ground to ensure that rodents cannot burrow under them. Vole guards were placed inside of each larger browse protector. Due to the aggressive grasses present throughout the project reach, approximately 3 inches of bark mulch was placed within each vole guard to provide protection from weeds at the base of the plant. In addition to browse protection, a two-foot by two-foot plastic mulch mat was also installed around the base of each one gallon shrub to reduce competition from aggressive grasses and other invasive species. The five gallon plants did not require protection.

Table 18. Containerized plant species, size and number planted in the Therriault Creek Restoration Project area in fall 2007.

Genus	Species	Common Name	Size	Percent	Total Each
<i>Alnus</i>	<i>incana</i>	Thinleaf alder	10x10x36 cm	5%	50
<i>Betula</i>	<i>occidentalis</i>	Water birch	10x10x36 cm	5%	50
<i>Cornus</i>	<i>stolonifera</i>	Red-osier dogwood	10x10x36 cm	10%	100
<i>Crataegus</i>	<i>douglasii</i>	Douglas hawthorne	10x10x36 cm	15%	150
<i>Prunus</i>	<i>virginiana</i>	Common chokecherry	10x10x36 cm	5%	50
<i>Rosa</i>	<i>woodsii</i>	Wood's rose	10x10x36 cm	5%	50
<i>Salix</i>	<i>bebbiana</i>	Bebb's willow	10x10x36 cm	10%	100
<i>Salix</i>	<i>boothii</i>	Booth's willow	10x10x36 cm	5%	50
<i>Salix</i>	<i>drummondiana</i>	Drummond willow	10x10x36 cm	5%	50
<i>Salix</i>	<i>exigua</i>	Sandbar willow	10x10x36 cm	15%	150
<i>Salix</i>	<i>geyeriana</i>	Geyer's willow	10x10x36 cm	10%	100
<i>Symphoricarpos</i>	<i>occidentalis</i>	Common snowberry	10x10x36 cm	10%	100
TOTAL ONE GALLON CONTAINERIZED PLANTS					1000
<i>Alnus</i>	<i>incana</i>	Thinleaf alder	5 gallon grow bag	N/A	20
<i>Cornus</i>	<i>stolonifera</i>	Red-osier dogwood	5 gallon grow bag	N/A	10
<i>Salix</i>	<i>bebbiana</i>	Bebb's willow	5 gallon grow bag	N/A	10
<i>Salix</i>	<i>geyeriana</i>	Geyer willow	5 gallon grow bag	N/A	10
TOTAL FIVE GALLON GROW BAG PLANTS					50

3. Solarization Treatments:

The purpose of this treatment was to suppress invasive plant species, particularly grasses, and create site conditions more suitable for supporting desired plant species. This treatment was used in floodplain areas dominated by exotic grasses, specifically targeting reed canary grass. The seed mix used to supplement the seed bank in conjunction with this treatment is provided in Table 19.

A total of 0.5 acres (0.4 acres temporary and 0.1 acres long-term) within the project reach was treated with solarization. Long-term solarization planting sites, functioning as a weed barrier treatment, were located on outer meanders and planted through using one-gallon shrubs. Within long-term solarization areas, shade cloth will remain for up to three growing seasons or until shrubs have become established. Five temporary and two long-term solarization areas were installed as a pre-planting treatment or as a weed barrier treatment to completely eradicate small areas of invasive species. Only temporary solarization sites were seeded with native wetland forbs and grasses prior to placement of shade cloth.

Treatment sites were covered with two side-by-side strips of 15-foot wide fabric laid out on the floodplain, with three feet of the fabric overlapping. Each strip was approximately 50 feet long. Fabric was secured using eight inch by two inch metal staples. For long-term, planted, sites, the outside edges of the fabric weretrenched into the ground with at least one foot of the fabric buried. For temporary sites, the fabric was not trenched.

Table 19. Grass and forb seed mix used to supplement the existing seed bank in conjunction with the solorization treatments in the Therriault Creek Restoration Project area in 2007.

Genus	Species	Common Name	Percent of Mix
<i>Agropyron</i>	<i>riparium</i>	Streambank wheatgrass	15%
<i>Calamagrostis</i>	<i>canadensis</i>	Bluejoint reedgrass	15%
<i>Carex</i>	<i>nebrascensis</i>	Nebraska sedge	10%
<i>Carex</i>	<i>stipata</i>	Sawbeak sedge	5%
<i>Deschampsia</i>	<i>cespitosa</i>	Tufted hairgrass	20%
<i>Elymus</i>	<i>trachycaulus</i>	Slender wheatgrass	10%
<i>Juncus</i>	<i>balticus</i>	Baltic rush	11%
<i>Juncus</i>	<i>tenuis</i>	Poverty rush	9%
<i>Achillea</i>	<i>millefolium</i>	Common yarrow	5

4. Vegetated Soil Lifts:

Vegetated soil lifts were installed at two sites along outer meander bends near the upstream end of the project where there was risk of channel avulsion into the abandoned Therriault Creek channel. The total length of the two soil lifts were approximately 50 feet each (100 feet total). Vegetated soil lifts are a revegetation and bank construction technique that combines layers of dormant willow cuttings with fabric-wrapped soil to revegetate and stabilize stream banks. Soil is wrapped within two layers of biodegradable coconut fiber (coir) fabric, to hold the soil in place while vegetation becomes established in the relatively high stress land/water interface. Soil lifts, combined with a bankfull bench, will result in near bank areas where native woody vegetation can become established. To increase success, the face of the bottom soil lift was reinforced with an aspen excelsior log to help maintain the lift shape, keep fine soil particles from filtering out through the lift face, and maintain surface tension. A minimum of 300-500 live willow cuttings were placed within the soil lifts. The majority of the willow species used

were sandbar willow (*Salix exigua*), Drummond Willow (*Salix drummondiana*), Booth willow (*Salix boothii*), and yellow willow (*Salix lutea*).

5. Live Willow Fascines:

Live willow fascines are willow cuttings tied together to form a linear bundle and anchored. This treatment was installed at recent depositional areas within the channel for a total of 800 feet within the project area. The willow species composition for the fascines was similar to the vegetated soil lifts (see above).

6. Large Woody Debris Structures:

This technique includee placing large woody debris throughout the lower one third of the project area at select locations of the channel to enhance habitat and provide roughness features to encourage over-bank flooding and retention of flood waters in adjacent floodplain and riparian areas. Materials for this aspect of the project included whole logs and trees with limbs, branches and root wads attached. Only portions of the whole trees were placed in the channel, but were grouped together to form large woody debris interlocking complexes designed to trap and retain organic matter transported through the system. Approximately 20 whole trees were used for this treatment.

7. Herbicide Application:

We used herbicide as the primary method of weed control in the project reach over a combined total area of 5 acres. We targeted three species including: Canada thistle, Yellow toadflax and reed canarygrass. Canada thistle and Yellow toadflax were treated using a selective herbicide at similar rates. Reed canarygrass requires a non-selective herbicide which was applied carefully to spot treat canarygrass and not negatively impact desirable species. Herbicide was applied twice, once in the summer and fall of 2008.

Monitoring

The existing stream channel prior to restoration consisted of an entrenched F4 /G4 Rosgen channel type (Rosgen 1996), and the restoration work converted the stream back an E4 channel type that has access to the historic floodplain. This restoration project approximately doubled the stream length within the project area due to the increased meander frequency resulting from project construction (Figure 14). The stream pattern and total length have not changed substantially since it was constructed. This project also created approximately 55 acres of prior converted wetland. A detailed and rigorous evaluation of changes to the stream channel dimensions, pattern, and profile that resulted from this restoration effort are presented in Dunnigan et al. (2008). The results of monitoring reported within this document therefore focus on evaluating whether these changes have been maintained through time.

We stratified Therriault Creek within the project area into two reaches based on changes in valley slope. Reach 1 included the upper 3,750 feet of constructed stream channel, where

valley slope measured 1.44%. The valley slope of Reach 2 measured 0.75%, and included the lower 5,350 feet of constructed stream channel. In order to evaluate the immediate physical changes to the stream channel as a result of the restoration work and the long-term sustainability of these changes, we completed cross sectional surveys before and after project implementation.

Prior to project construction, we surveyed 10 riffles and 10 pools within the existing stream channel. Within the riffle habitats, we established each transect at the longitudinal mid-point of the first 10 riffles downstream of the upper project boundary. Within the pool habitats, we established the cross section transects within each pool where the maximum depth occurred. We also selected the first ten pools downstream of the upper project boundary. Upon completion of the project, we established permanent cross sections within 10 pool and 10 riffle habitats that were distributed throughout the entire project area. We surveyed the 20 permanent cross sections annually since project completion, measuring mean bankfull width, depth, cross sectional area, maximum bankfull depth and width to depth ratio at each. We used an analysis of variance to test each of these parameters for significant differences between years. Multiple comparisons were performed using the Tukey Test.

This restoration project changed the stream channel type, pattern, profile and channel dimensions of Therriault Creek within the project area (Dunnigan et al. 2008), and these changes appear to be self-sustaining since the project was completed. Stream channel type, length, and pattern have not changed since the project was completed. Stream channel dimensions also changed as a result of the project, but we were unable to detect significant changes in the stream channel dimensions at either riffle or pool type habitats within the two reaches since 2004 (Tables 20-23). Of the parameters we measured within Reach 1 for riffles and pools, maximum bankfull depth showed the highest relative change between 2006 and 2007. However, this change was not consistent between habitat types. For example, mean maximum bankfull riffle depth decreased by 0.3 feet (-16.9%; Table 20), within Reach 1, but mean maximum pool depth increased by 0.1 feet (6.3%) within Reach 1. Within Reach 2, mean depth of both riffles and pools showed the highest relative change of any of the parameters we measured, with a general trend of increased depth across both habitat types (Table 21 and 23, respectively). The mean depth of riffles increased by 0.3 feet (14.2%; Table 21) from 2006 to 2007, while the mean depth of pools increased by 0.5 feet (31.8%; Table 23). The relatively small changes in channel dimensions between reaches and habitat types we measured are relatively small, suggest the stream channel is maintaining in a state of relative dynamic equilibrium in which case depositional and erosion are occurring at approximately equal rates within the project area.

Figure 14. The plan view and typical cross-section views for the Theriault Creek Restoration Project. The creek flows from upstream project boundary (upper right corner of photograph) to the lower project boundary (lower left portion of the photograph).

Table 20. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio for riffle-type habitats in Reach 1 of the Therriault Creek Restoration Project area. The project was constructed in the fall of 2004-2005. The range for annual mean values is presented in parentheses. The percent change between years is also presented. Cross sectional surveys from 2003 were not stratified by reach. Analysis of variance was preformed for each parameter, and the P value is presented. Multiple comparisons were performed using Tukey's Test. Significant comparisons are indicated via * ($\alpha < 0.05$).

	# Runs	Cross Sectional Area (ft²)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	W/D Ratio
2003 (Existing)	10	13.9 (8.2-20.5)	12.6 (8.3-17.2)	1.1 (1.0-1.2)	1.5 (1.1-1.8)	11.5 (7.4- 15.0)
2004 (As Built)	4	11.2 (8.6-12.1)	8.3 (7.9-9.0)	1.3 (1.0-1.5)	1.8 (1.2-2.0)	6.4 (5.2-8.3)
2005	4	13.8 (11.9-16.7)	9.6 (8.0-13.0)	1.5 (1.3-1.6)	1.9 (1.7-2.3)	6.7 (5.4- 10.0)
2006	4	12.8 (9.3-15.5)	10.0 (8.0-12.7)	1.3 (1.0-1.7)	1.9 (1.4-2.2)	8.2 (4.7- 12.3)
2007	4	13.2 (9.1-20.4)	10.5 (8.0-15.1)	1.2 (1.0-1.6)	1.6 (1.1-2.0)	8.7 (5.0-11.2)
P-Value		0.723	0.068	0.055	0.088	0.009
Percent Change						
2003/2004		-19.7%	-34.0%	22.2% *	17.8%	-44.5% *
2003/2005		-1.3%	-23.9%	32.1% *	27.9%	-41.4% *
2003/2006		-8.1%	-20.6%	18.5%	29.6%	-28.7%
2003/2007		-5.6%	-16.4%	13.1%	7.7%	-24.1%
2004/2005		3.8%	10.0%	-6.7%	4.5%	5.7%
2004/2006		-8.3%	11.1%	20.0%	-9.1%	28.7%
2004/2007		17.6%	26.7%	-7.4%	-8.6%	36.9%
2005/2006		-11.7%	10.1%	14.3%	-13.4%	21.8%
2005/2007		-4.3%	9.8%	-14.4%	-15.8%	29.5%
2006/2007		2.8%	5.3%	-4.6%	-16.9%	6.4%

Table 21. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio for riffle-type habitats in Reach 2 of the Therriault Creek Restoration Project area. The project was constructed in the fall of 2004-2005. The range for annual mean values is presented in parentheses. The percent change between years is also presented. Cross sectional surveys from 2003 were not stratified by reach. Analysis of variance was preformed for each parameter, and the P value is presented. Multiple comparisons were performed using Tukey's Test. Significant comparisons are indicated via * (alpha < 0.05).

	# Runs	Cross Sectional Area (ft ²)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	W/D Ratio
2003 (Existing)	10	13.9 (8.2-20.5)	12.6 (8.3-17.2)	1.1 (1.0-1.2)	1.5 (1.1-1.8)	11.5 (7.4-15.0)
2004 (As Built)	6	14.5 (10.2-16.6)	8.3 (7.9-9.0)	1.7 (1.3-2.1)	2.2 (1.7-2.5)	4.9 (4.3-6.1)
2005	6	15.1 (11.7-18.9)	8.3 (7.5-9.0)	1.8 (1.6-2.1)	2.2 (1.8-2.4)	4.6 (4.3-5.0)
2006	6	14.7 (10.5-17.0)	8.4 (7.9-8.9)	1.7 (1.3-2.0)	2.2 (1.6-2.6)	4.9 (4.2-5.9)
2007	6	16.3 (13.3-19.2)	8.2 (7.7-8.8)	2.0 (1.7-2.3)	2.3 (1.9-2.8)	4.1 (3.8-4.5)
P-Value		0.630	<0.001	<0.001	<0.001	<0.001
Percent Change						
2003/2004		4.4%	-33.7%*	57.0%*	45.9%*	-57.3%*
2003/2005		8.6%	-33.9%*	63.5%*	45.9%*	-59.5%*
2003/2006		5.7%	-33.0%*	57.9%*	48.1%*	-57.0%*
2003/2007		17.3%	-35.0%*	80.3%*	57.1%*	-63.8%*
2004/2005		-3.0%	1.1%	-5.3%	-16.1%	-5.2%
2004/2006		-11.8%	9.8%	-21.1%	-19.4%	0.8%
2004/2007		12.4%	-1.7%	14.8%	7.7%	-15.2%
2005/2006		-9.1%	-8.6%	-16.7%	-3.8%	6.3%
2005/2007		8.6%	-1.6%	10.3%	7.7%	10.6%
2006/2007		11.0%	-2.9%	14.2%	6.1%	-15.9%

Table 22. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio for pool-type habitats in Reach 1 of the Therriault Creek Restoration Project area. The project was constructed in the fall of 2004-2005. The range for annual mean values is presented in parentheses. The percent change between years is also presented. Cross sectional surveys from 2003 were not stratified by reach. Analysis of variance was performed for each parameter, and the P value is presented. Multiple comparisons were performed using a Tukey Test. Significant comparisons are indicated via * (alpha < 0.05).

	Number Of Pools	Cross Sectional Area (ft²)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)
2003 (Existing)	10	18.9 (13.8-27.2)	13.4 (7.8-19.5)	1.5 (0.9-1.8)	2.5 (1.9-3.0)
2004 (As Built)	4	13.2 (9.3-16.6)	9.0 (8.0-10.0)	1.5 (1.2-1.7)	2.2 (1.7-2.8)
2005	4	13.7 (10.7-17.6)	9.9 (8.9-11.9)	1.4 (1.2-1.9)	2.3 (1.9-2.6)
2006	4	12.1 (8.7-16.1)	10.0 (9.7-10.5)	1.2 (0.8-1.7)	2.0 (1.4-2.5)
2007	4	12.1 (8.5-14.1)	10.0 (9.6-10.5)	1.2 (0.9-1.4)	2.1 (1.7-2.5)
P-Value		0.007	0.046	0.471	0.229
Percent Change					
2003/2004		-30.3%	-33.0%	-1.0%	-11.3%
2003/2005		-27.5%	-26.5%	-5.6%	-8.8%
2003/2006		-35.8% *	-25.4%	-17.2%	-19.4%
2003/2007		-35.8% *	-25.4%	-19.9%	-14.3%
2004/2005		23.2%	15.7%	15.4%	5.6%
2004/2006		14.3%	20.5%	0%	5.6%
2004/2007		-7.9%	11.4%	-17.0%	-3.4%
2005/2006		-7.2%	4.2%	-13.3%	0%
2005/2007		-11.3%	1.5%	-13.0%	-6.1%
2006/2007		0.1%	0%	0.8%	6.3

Table 23. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio for pool-type habitats in Reach 2 of the Therriault Creek Restoration Project area. The project was constructed in the fall of 2004-2005. The range for annual mean values is presented in parentheses. The percent change between years is also presented. Cross sectional surveys from 2003 were not stratified by reach. Analysis of variance was performed for each parameter, and the P value is presented. Multiple comparisons were performed using a Tukey Test. Significant comparisons are indicated via * (alpha < 0.05).

	Number Of Pools	Cross Sectional Area (ft²)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)
2003 (Existing)	10	18.9 (13.8-27.2)	13.4 (7.8-19.5)	1.5 (0.9-1.8)	2.5 (1.9-3.0)
2004 (As Built)	6	16.9 (12.6-21.5)	9.2 (8.7-10.3)	1.9 (1.2-2.4)	3.1 (2.3-3.7)
2005	6	16.4 (11.2-20.7)	9.3 (8.2-10.4)	1.8 (1.3-2.3)	2.6 (1.9-3.2)
2006	6	14.9 (9.8-19.4)	10.1 (8.4-12.8)	1.5 (0.9-1.8)	2.5 (1.8-3.0)
2007	6	18.1 (14.6-23.1)	9.2 (8.2-10.9)	2.0 (1.6-2.2)	2.7 (2.2-3.1)
P-Value		0.240	0.005	0.034	0.157
Percent Change					
2003/2004		-10.4%	-31.4%*	26.1%	23.0%
2003/2005		-14.4%	-30.9%*	19.3%	6.2%
2003/2006		-21.1%	-24.8%	1.7%	2.2
2003/2007		-3.9%	-31.3%*	34.1%*	8.2%
2004/2005		4.1%	0%	5.9%	0%
2004/2006		1.4%	1.2%	0.0%	0%
2004/2007		7.0%	0.2%	6.3%	-12.0%
2005/2006		-2.6%	1.2%	-5.6%	0%
2005/2007		12.2%	0.5%	12.4%	11.9%
2006/2007		21.8%	-8.6%	31.8%	5.9%

Discussion

Within this report, we presented physical monitoring from seven stream restoration projects on four separate streams ranging from one to five years after completion. Restoration techniques were generally similar between projects, consisting primarily of stream channel reconstruction with the use of large rock, woody debris and bioengineered structures to stabilize previously unstable stream banks and create pool-type habitats. Each of the three streams had generally similar in stream channel type (Rosgen 1996), with the exception of Therriault Creek. These streams did however differ in discharge capacity.

These restoration projects unequivocally changed the pattern, profile and dimension of the stream within the project area. Within the riffle habitats several conditions were generally evident for each of the seven restoration projects. We were generally able to show a significant increase in mean bankfull depth and a decrease in stream bankfull width, and change in channel dimensions were generally less than 10% annually. Pool-type habitats generally changed more so than riffle habitats after construction. All the restoration projects presented within this document demonstrated substantial increases in the quantity, depth and spacing of pools within the project areas. Total pool numbers and total pool area and volume increased by several fold for all five projects after construction. However, we have observed a slight annual loss of the total number of pools, and mean pool depth through time up to three years after construction, but despite these reductions, pool depth, quantity and quality still exceeded conditions that existed prior to project construction.

The stream restoration activities that we have undertaken as part of the Libby Mitigation Project differ fundamentally from those typically reported in the literature. Most of the stream restoration activities that others report either the successes (Binns 1994; Binns and Remmick 1994; Burgess and Bider 1980; Hunt 1976; House and Boehne 1985 and 1986) or failures (Frissell and Nawa 1992; Pattenden et al. 1998; Hamilton 1989) typically implemented what we would consider habitat enhancement activities rather than stream channel reconstruction as was the case with each of the projects we completed. Frissell and Nawa (1992) and Pattenden et al. (1998) agreed that the risk of failure of stream restoration activities is highest in streams with recent watershed disturbance, high instream sediment budgets, and unstable stream channels. It seems ironic however, that many of the stream systems that fit within these characterizations are those most important to fisheries populations and in the most need of restoration. Grave, Libby, Young and Therriault creeks are good examples of streams that fit both sets of circumstances. Frissell and Nawa (1992) and Pattenden et al. (1998) also noted that when failure or impairment occurred in stream restoration projects, it generally was a result of watershed driven aspects of stream channel dynamics rather than internal structural failures, and that rain on snow events produced some of the highest incidences of structure failure. This was certainly the case with the three restoration projects on Libby Creek in November 2006, in which case a rain on snow event occurred throughout the watershed.

Our monitoring indicates the 2006 rain on snow event in the Libby Creek watershed substantially damaged much of the restoration work in upper Libby Creek. The most substantial and conspicuous changes within the three restoration project areas were changes in the stream planform, with substantial stream straightening and loss of meander pattern throughout, and a loss of the total number of pools. These patterns of disturbance were exacerbated along a downstream gradient, with changes most substantial within the Lower Cleveland Phase II project area. However, stream power, as indexed by total discharge may have influenced these observations, given that Libby Creek discharge increases substantially between the Upper Cleveland and lower Cleveland project areas due to the influx of three sizeable tributaries (Ramsey, Poorman and Midas creeks). Future monitoring in the Lower Cleveland Phase III is warranted and may serve as a control that could be used to compare the flood response to treated and untreated sections of Libby Creek with increased rigor.

Our fish monitoring data at six locations throughout the watershed (see Chapter 1) suggest that this flood event negatively impacted rainbow trout abundance systemwide, with observed decreases in abundance observed at all six monitoring sites in 2007 relative to 2006. We observed the largest annual declines within our restoration areas, specifically the Upper Cleveland and Lower Phase II project areas. Several authors have noted substantial negative impacts associated with stream dwelling salmonids due to flood events (Seegrist and Gard 1972; Elwood and Waters 1969; Hoopes 1975; and Pearsons and Wi 1992). Seegrist and Gard (1972) found that floods lowered egg to fry survival of rainbow and brook trout when flood events coincided with incubation periods. Impacts to adult fish increased with increasing flood intensity. Hoopes (1975) also found that a large flood was especially lethal for young of the year brook trout in a Pennsylvania stream, eliminating approximately 96% of this cohort. Older age classes were also impacted, but less substantially. Elwood and Waters (1969) documented almost complete elimination of two year classes of brook trout, and that standing crop was reduced due to a decrease in the stream's carrying capacity due to sediment deposition and pool filling. Pearsons and Wi (1992) used hydraulic retention to index hydraulic complexity of high desert streams in north central Oregon and found that following the floods, hydraulically complex stream reaches lost proportionately fewer fish, had generally higher fish diversities, and had higher fish assemblage similarity than hydraulically simple stream reaches.

-Need for revegetation efforts due to stability of monitoring data.

The number of cases reported in the literature where stream restoration work has increased fish abundance at the population level is relatively small and somewhat dated relative to the overall effort expended to improve fisheries habitat (Frissell and Nawa 1992; Roper et al. 1997). Habitat enhancement has been shown to increase the abundance of resident salmonids in streams (Binns 1994; Binns and Remmick 1994; Hunt 1976; Saunders and Smith 1962; House and Boehne 1986). However, in nearly every study the physical changes to the habitat were sustained with relatively low structure failure rates. Therefore, in order for the stream restoration projects we

completed to be successful over the long term, the changes to the quantity and quality of the habitat will need to be sustained through time. Most of the restoration projects discussed in this report have sustained the changes through time, with the exception of parts of upper Libby Creek. However, the work we performed relies on the physical structures to maintain streambank stability through time, but we acknowledge these structures have a limited life expectancy, and that riparian vegetation will be the glue that holds these projects together in the long-term. Therefore, many of our recent efforts have been to promote recovery of healthy riparian areas associated with our restoration projects, as is case with the Therriault and Grave Creek projects.

Many investigators (Hunt 1976; Binns 1994; Binns and Remmick 1994) argue that several years are needed for fish populations to fully respond to habitat enhancement. Binns and Remmick (1994) lobby for a minimum of several years of pretreatment data be collected and at least 4-8 years post treatment data collection is necessary for a valid evaluation of fish populations to habitat restoration work. The life histories of the fish species inhabiting these streams dictates that they will not sexually mature until age 3-5, and in the case of bull trout, the age at maturity is as long as 5-8 years.

Many restoration efforts do not monitor the fish population response to the habitat manipulations. This project does monitor fish populations (see Chapter 1), but monitoring efforts have not yet indicated that fish populations have increased significantly since our first projects were completed, despite the sustained increases in the quantity and quality of pool habitat and cover within the restored reaches. We attribute the lack of fish response to the limited time since the restoration work was completed. Two of the first stream restoration projects the Libby Mitigation Project completed were the Libby and Grave Creek Demonstration Projects in the fall of 2001. In the case of these two projects, we observed an increase in rainbow and bull trout abundance, but the results were not significant (see Chapter 1). However, in the case of the Young Creek and Libby Creek Upper and Lower Cleveland (Phase I and II) Projects, we observed a non-significant decrease in our primary focal native species immediately following the restoration activities, and in the case of the Young Creek project, we observed an increase in the number of non-native brook trout. We attribute the decrease in fish abundance to the initial impact associated with project construction and interactions with other species.

We are confident that the physical changes to the habitat will translate into real and substantial increases at the local population level, but that these changes may take many years to realize. We feel our monitoring components associated with the Libby Mitigation stream restoration projects will be adequate to detect these changes through time.

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Chapter 3

Bull Trout Monitoring and Research in the Kootenai River Basin

By:

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Abstract

Bull trout core areas upstream of Libby Dam include Grave and Skookumchuck creeks and the Wigwam and White rivers, with the majority of the spawning located in the tributaries located in British Columbia. Bull trout redd counts in Grave Creek and the Wigwam River have both exhibited significant positive trends since the mid 1990s. There were a total of 1,883 bull trout redds within the index portion of the Wigwam River, which was approximately an order of magnitude higher than any other tributary within the Kootenai Basin. Bull trout core areas in the Kootenai River downstream of Libby Dam include Quartz, Pipe, Bear (Libby Creek drainage), O'Brien creeks and the West Fisher Creek. 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. Three of the four populations between Libby Dam and Kootenai Falls were below average over the period of record. West Fisher Creek was the only stream in this group that was slightly above average. Record low bull trout redd counts were observed in Pipe Creek, and a second highest counts was observed in O'Brien Creek. The only bull trout population that exhibited a significant positive trend, was O'Brien Creek. The adjunct Bull Lake population, which spawns in Keeler Creek, was below the long-term average and the lowest since 2001.

MFWP conducted a population estimate for adult bull trout below Libby Dam during April 2008, and estimated 381 (95% Confidence Interval = 158 – 605) adult bull trout were present within this 3.5-mile section of the Kootenai River, which was similar to the previous year. We recaptured 16 bull trout that were previously marked in 2004, 2005, 2006, or 2007 below Libby Dam ranging between 313 to 1,117 days prior. The recaptured bull trout grew an average of 96.3 mm (0.21 mm per day), and gained an average of 1,649.9 g (3.07 g per day).

MFWP randomly collected fin clips from approximately 50 *Salvelinus spp.* from Keeler, Pipe, West Fisher, and O'Brien creeks for genetic analyses in order to collect DNA to examine hybridization rates between brook and bull trout. We determined individual fish genotypes using markers at seven microsatellite loci. All four streams contained hybridized fish, with hybridization especially problematic in O'Brien and Keeler creeks. However, bull trout were

⁵ Principal author for the monitoring aspects of this report.

⁶ Principal author for the genetics aspects of this report.

rare in Pipe Creek. Overall, we were able to correctly visually identify 91% of the fish collected for analysis. The most common visual misidentification involved calling later generation hybrids as being either brook trout or bull trout.

MFWP monitored fine sediment (<6.35 mm) levels in eight bull trout spawning tributaries within the Montana portion of the Kootenai Basin using core sampling in close proximity to bull trout redds near the time of emergence. Keeler Creek had the lowest mean levels of fine sediment amongst the eight streams we monitored, averaging 30.2% over the period of record. O'Brien Creek had the highest mean levels of fine sediment at 40.6%. Of the four core bull trout spawning tributaries located on the Kootenai River between Libby Dam and Kootenai Falls, West Fork Quartz Creek had the lowest annual mean fine sediment levels (33.4%). Mean annual fine sediment levels on Pipe Creek and Bear Creek were both relatively high, but similar (38.4% and 38.6%, respectively). The two bull trout spawning tributaries located in Montana upstream of Libby Dam (Grave Creek and MT Wigwam) had relatively low fine sediment levels of 32.2% and 34.1% respectively.

Introduction

The bull trout that inhabit Libby Reservoir and Kootenai River represent geographically and genetically distinct and important populations within their range (USFWS 1999; Ardren et al. 2007). MFWP list bull trout as a species of special concern and in 1996 the United States Fish and The US Wildlife Service (USFWS), through the Endangered Species Act, listed bull trout as threatened throughout their range in 1998 (USFWS 1999).

Libby Dam, constructed on the mainstem Kootenai River in 1972, represents a major limiting factor affecting bull trout in the Kootenai River (USFWS 2002; Montana Bull Trout Scientific Group 1996a). Presently no fish passage facilities exist at Libby Dam and migration only occurs downstream through the dam. Previous studies have documented the passage of bull trout (Dunnigan et al. 2005; Skaar et al. 1996) downstream through Libby Dam, and a recent study funded by this project has indicated that at least half of the bull trout in the three mile section of river downstream of the dam between 2004 and 2007 were entrained (DeHaan et al. 2008). Dam operations represent a direct threat to bull trout in the middle Kootenai because of the biological affects associated with unnatural flow fluctuations and real potential gas supersaturation problems arising from spilling water. The dam is a fish barrier, generally restricting a portion of this migratory population to 29 miles of river between Libby Dam and Kootenai Falls. Although MFWP has documented upstream bull trout passage at Kootenai Falls, the falls represent a substantial fish barrier at most current flow regimes. The Kootenai River is a nodal habitat containing critical over-wintering areas, migratory corridors, and habitat required for reproduction and early rearing. Land use practices also constitute a high risk to bull trout in the middle Kootenai (Libby Dam to Kootenai Falls) due impacts on spawning and rearing habitat. These risks are accentuated due to the low number of spawning streams (Quartz, Pipe, O'Brien, Callahan and Libby Creek drainages) available; a direct result of habitat fragmentation caused by Libby Dam.

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 practices. 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. Beginning in 2004, MFWP opened a recreational bull trout fishery on Libby Reservoir for the first time since 1993. The fishery was established as an experimental exception to the Federally Listed threatened status of bull trout within the Columbia River Subbasin through negotiations with the US Fish and Wildlife Service. This fishery was established due to the relatively high abundance of bull trout in Libby Reservoir.

Bull trout are found below Kootenai Falls in O'Brien Creek, Callahan Creek and in Bull Lake. The latter is a disjunct population that migrates out of Bull Lake, downstream to Lake Creek then upstream in Keeler Creek. These fish inhabit areas in the lower Kootenai River and Kootenay Lake during most of the year.

MFWP conducts annual monitoring to assess bull trout trends in abundance and critical spawning and rearing habitat. We monitor annual escapement in eight critical tributaries used for spawning by conducting redd counts within index reaches of each stream, and within these stream we monitor fin sediment levels in order to evaluate the potential impact of sediment on egg survival. We also monitor bull trout abundance within the Libby Dam tailrace and conduct genetic assessments to estimate annual entrainment (DeHann et al. 2008). In 2007, we also assessed the impact of non-native brook trout on hybridization with bull trout in four tributaries in the Kootenai Basin.

Methods

Bull Trout Redd Counts

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

Kootenai River Adult Bull Trout Population Estimate

We collected adult bull trout using nighttime electrofishing by jet boat to perform a mark-recapture population estimate of bull trout in the Kootenai River from Libby Dam (River mile [RM] 221.7) downstream to the confluence of the Fisher River (RM 218.2). We marked bull trout on the evenings of April 9 and 10, 2008, and performed recapture sessions on April 16 and 17, 2008. We operated two jet boat electrofishing crews during each of the other two sampling events. Each boat contained a driver and two netters. Our electrofishing unit on each boat consisted of a Coffelt model Mark 22 electrofishing unit operating with an electrical output ranging from 200-350 volts at 5-8 amps powered by a 5,000 watt gasoline powered generator. In order to thoroughly electrofish the entire 3.5 miles of Kootenai River, we divided the sample area into 2 sections, and conducted electrofishing on each section on a single evening. Section 1 was from Libby Dam downstream to the Alexander Creek confluence (RM 220.5), and was 1.2 miles long, and was sampled on April 9 and 16, respectively. Section 2 was from the Alexander Creek confluence downstream to the Fisher River Confluence, and was 2.3 miles long, and was sampled on April 10 and 17.

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

We estimated bull trout abundance using a mark-recapture population estimation technique which assumes the population of bull trout is “closed”, suggesting no births, deaths or migrations occurred during sampling periods (Ricker 1958). Additional assumptions were that marked and unmarked fish have equal mortality rates, marked fish were randomly distributed throughout the study area, marks were not lost, and all marked fish captured were recognized and counted (Lagler 1956). We used the Petersen Estimator as modified by Chapman (Ricker 1958) to estimate absolute abundance of bull trout where:

$$N = \frac{(M + 1) \bullet (C + 1)}{R + 1} - 1 + Morts \quad [1]$$

Where: N = population estimate,
C = total fish captured in the recapture sample(s),
M = number of marked fish at the start of recapture sample period and
R = number of marked fish in the recapture sample(s)
Morts = number unmarked mortalities captured during the marking sessions.

We used the following formula to calculate bounds (B) for 95% confidence intervals for N:

$$B = 1.96 \times \sqrt{\frac{N^2 \bullet (C - R)}{(C + 1) \bullet (R + 2)}} \quad [2]$$

We compared the mean length of bull trout captured during our 2008 sampling to the mean length of bull trout captured during similar sampling conducted in 2004, 2005, 2006 and 2007 using ANOVA and subsequent Tukey multiple comparison. For all PIT tagged bull trout that were captured from previous marking sessions, we calculated the average total length and weight gain since time of original capture.

Bull/Brook Trout Genetic Analyses in Bull Trout Natal Streams

In order to determine if there is evidence of hybridization between bull trout, *Salvelinus confluentus*, and brook trout, *S. fontinalis*, in the four core bull trout streams in the Montana portion of the Kootenai Subbasin that also contained brook trout, we randomly collected fin clips from approximately 50 *Salvelinus* spp. from the bull trout juvenile monitoring sections (see Hensler and Benson 2006) from Keeler, Pipe, West Fisher, and O'Brien creeks for genetic analyses using a backpack electrofisher. The remaining bull trout core streams in the Montana portion of the Kootenai Subbasin that do not contain brook trout (Grave, Quartz, Libby, and Bear creeks) were previously genetically tested and found no evidence of hybridization with brook trout (Ardren et al. 2007). DNA was extracted from fin clips, and using the polymerase chain reaction (PCR), each fish's genotype was determined at seven microsatellite loci that distinguish bull trout from brook trout (Table 1). Such loci are commonly termed diagnostic loci because the alleles (length of the DNA fragment copied during PCR) detected at them can be used to determine whether an individual was a nonhybridized bull or brook trout or was of hybrid origin between these fishes.

For each fish, we calculated a hybrid index as follows. At each locus, an allele characteristic of bull trout was given a value of zero and an allele characteristic of brook trout was given a value of one. Thus, at a locus an individual could have a value of zero, one, or two. These values summed over all loci yields an individual's hybrid index. A nonhybridized bull trout, therefore, would have a hybrid index of zero and a nonhybridized brook trout would have a hybrid index of 14. A first generation hybrid (F_1) would have a hybrid index of seven. First generation backcrosses to bull trout (F_1 X bull trout) would have hybrid indices ranging from zero to seven with a mean of 3.5. First generation backcrosses to brook trout (F_1 X brook trout) would have hybrid indices ranging from seven to 14 with a mean of 10.5. Finally, second (F_2 , F_1 X F_1) or later generation hybrids would have hybrid indices ranging from zero to 14 with a mean of seven.

Table 1. Alleles at the diagnostic microsatellite loci between bull trout and brook trout.

Locus	Bull trout alleles	Brook trout alleles
<i>Omm1128*</i>	282	213
	283	217
	287	220
	291	221
	295	224
	299	225
	303	228
	307	229
	311	232
	315	233
	319	237
	327	253
	331	263
	335	271
<i>Sco102*</i>	164	126
	168	130
		134
<i>Sco107*</i>	273	192
	285	196
	289	
	293	
	297	
	301	
	305	
	309	
	325	
<i>Sco215*</i>	285	327
	289	328
	293	329
		333
		335
		337
		341
		343
		345
		351
<i>Sco216*</i>	225	132
	241	136
	245	140
	249	144
	253	148
	261	152
	265	160
		164

Table 1 (Continued). Alleles at the diagnostic microsatellite loci between bull trout and brook trout.

Locus	Bull trout alleles	Brook trout alleles
<i>Sco216*</i> (continued)		168
		172
		176
		180
		185
		187
		191
<i>Sco218*</i>	202	146
	210	150
	214	154
	218	158
	222	162
	226	166
	254	170
	266	174
	270	178
	274	182
	278	190
	282	194
	290	198
	294	
	306	
<i>Sfo18*</i>	152	174
	158	176
		178
		180
		182

Bull Trout Spawning Substrate Surveys

Sample Collection and Processing

We used a standard 15.2 cm hollow core sampler (McNeil and Ahnell 1964) to collect four samples across each of three transects from each fluvial or adfluvial bull trout tributary in the Montana portion of the Kootenai watershed in order to assess potential trends in bull trout emergence success. We located coring sites within each stream using a stratified random selection process during the winter months (generally February through March). The total width of stream having suitable depth, velocity, and substrate for spawning was visually divided into four equal cells. We randomly took one core sample in each cell. In some study areas we deviated from this procedure due to limited or discontinuous areas of suitable spawning habitat. We selected study areas based on observations of natural spawning. We only sampled in

spawning areas used by adfluvial and fluvial bull trout. During the period of study, these fish spawned in the same general areas, so sampling locations remained similar.

Sampling involved working the corer into the streambed to a depth of 15.2 cm. We removed all material inside the sampler and placed it in a heavy duty plastic bags. We labeled the bags and transported them to the Kootenai National Forest Soils Laboratory in Libby, Montana, for gravimetric analysis. We sampled the material suspended in water inside the corer using an Imhoff settling cone (Shepard and Graham 1982). We allowed the cone to settle for 20 minutes before recording the amount of sediment per liter of water. After taking the Imhoff cone sample, we determined total volume of the turbid water inside the corer by measuring the depth and referring to a depth to volume conversion table (Shepard and Graham 1982).

The product of the cone reading (ml of sediment per liter) and the total volume of turbid water inside the corer (liters) yields an approximation of the amount of fine sediment suspended inside the corer after sample removal. We then applied a wet to dry conversion factor developed for Flathead tributaries by Shepard and Graham (1982), yielding an estimated dry weight (g) for the suspended material.

We oven dried the bagged samples and sieve separated them into 13 size classes ranging from >76.1 mm to <0.063 mm in diameter (Table 12). We weighed the material retained on each sieve and calculated the percent dry weight in each size class. The estimated dry weight of the suspended fine material (Imhoff cone results) was added to the weight observed in the pan, to determine the percentage of material <0.063 mm. We refer to each set of samples by using the mean percentage <6.35 mm in diameter. This size class is commonly used to describe spawning gravel quality, and it includes the size range typically generated during land management activities. We examined the range of median values for this size class observed throughout the survey area.

Analyses

We pooled up to 12 substrate samples from each stream within a given year and performed an arcsine transformation on the percentage on the <6.35 mm in diameter within each sample. We re-transformed all data back for graphical display purposes. We performed an analysis of variance to test for differences between years within a given stream. Post hoc multiple comparisons (Tukey's HSD) were performed if yearly differences existed ($p < 0.05$). All statistical analyses were performed using SPSS 7.5 Software.

Table 2. Mesh size of sieves used to gravimetrically analyze hollow core streambed substrate samples collected from Kootenai River basin tributaries.

Mesh Size (mm)	Mesh Size (inches)
76.1 mm	(3.00 inch)
50.8 mm	(2.00 inch)
25.4 mm	(1.00 inch)
18.8 mm	(0.74 inch)
12.7 mm	(0.50 inch)
9.52 mm	(0.38 inch)
6.35 mm	(0.25 inch)
4.76 mm	(0.19 inch)
2.00 mm	(0.08 inch)
0.85 mm	(0.03 inch)
0.42 mm	(0.016 inch)
0.063 mm	(0.002 inch)
Pan	(<0.002 inch)

Results

Bull Trout Redd Counts

Grave Creek

MFWP counted redds in Grave Creek (including Blue Sky, Clarence, Williams and Lewis Creeks) for the first time in 1983, as well as in 1984, 1985, and 1993 through 2006. Grave Creek was surveyed from the confluence of Cat Creek River upstream to near the mouth of Lewis Creek (approximately 4.9 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. MFWP 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 208 bull trout redds in Grave Creek in 2007, which was the second highest redd count observed over the past 15 years. The highest redd count was 245 redds and was observed in 2003 (Table 3). Nevertheless, bull trout have exhibited a significant positive trend in spawning abundance in Grave Creek since 1993 (Figure 1; $r^2 = 0.711$; $p < 0.0001$).

Figure 1. Bull trout redd counts and trend analysis in Grave Creek, 1993 through 2007.

Wigwam Drainage

Bull trout redd counts for the Wigwam River includes the tributary streams of Bighorn, Desolation, and Lodgepole creeks, and the portion of the Wigwam River within Montana. In 2007, a total of 1,883 bull trout redds were observed in the ten index reaches typically surveyed in the Wigwam Drainage (Figure 2). The trend in bull trout redd abundance since 1995 continues to represent a significant positive relationship. However, the relationship appears to be approaching an equilibrium over the past 5-6 years. We observed a total of 33 bull trout redds in the Montana portion of the Wigwam River, which represents the upper most reach. This was approximately 10 fish more than the previous five year average.

Figure 2. Bull trout redd counts and trend analysis for the Wigwam River (including Bighorn, Desolation, and Lodgepole creeks) 1995-2007.

Table 3. Bull trout redd survey summary for all index tributaries in the Kootenai River Basin.

Stream	Year Surveyed	Number of Redds	Miles Surveyed
Grave Creek Includes Clarence and Blue Sky Creeks	1995	15	9
	1996	35	17
	1997	49	9
	1998	66	9
	1999	134	9
	2000	97	9
	2001	173	9
	2002	199	9
	2003	245	9
	2004	141	9
	2005	194	9
	2006	148	9
	2007	208	9
Quartz Creek Includes West Fork and Mainstem	1995	66	12.5
	1996	47	12.0
	1997	69	12.0
	1998	105	8.5
	1999	102	8.5
	2000	91	8.5
	2001	154	8.5
	2002	62 ^e	8.5
	2003	55	8.5
	2004	49	10.0
	2005	71	8.5
	2006	51	8.5
	2007	35	8.5
O'Brien Creek	1995	22	4.5
	1996	12	4.0
	1997	36	4.3
	1998	47	4.3
	1999	37	4.3
	2000	34	4.3
	2001	47	4.3
	2002	45	4.3
	2003	46	4.3
	2004	51	4.3
	2005	81	4.3
	2006	65	4.3
	2007	77	4.3
Pipe Creek	1995	5	10
	1996	17	12.0
	1997	26	8.0
	1998	34	8.0
	1999	36	8.0
	2000	30	8.0
	2001	6 ^a	8.0
	2002	11	8.0
	2003	10	8.0
	2004	8	8.0
	2005	2	8.0
	2006	6	8.0
	2007	0	8.0

Table 3 (Continued). Bull trout redd survey summary for all index tributaries in the Kootenai River Basin.

Stream	Year Surveyed	Number of Redds	Miles Surveyed
Bear Creek	1996	10	4.5
	1997	13	4.25
	1998	22	4.25
	1999 ^b	36	4.25
	2000	23	4.25
	2001	4 ^e	4.25
	2002	17	4.25
	2003	14	4.25
	2004	6	4.25
	2005	3	4.25
	2006	14	4.25
	2007	9	4.25
Keeler Includes South and North Forks	1996	74	9.3
	1997	59	8.9
	1998	92	8.9
	1999	99	8.9
	2000	90	8.9
	2001	13 ^d	8.9
	2002	102	8.9
	2003	87	8.9
	2004	126	8.9
	2005	186	8.9
	2006	142	8.9
	2007	84	8.9
West Fisher Creek	1995	3	10
	1996	4	6
	1997	0	6
	1998	8	6
	1999	18	10
	2000	23	10
	2001	1	10
	2002	1	6
	2003	1	6
	2004	21	10
	2005	27	10
	2006	4	10
	2007	18	10
Wigwam (B.C and U.S.) Includes Bighorn, Desolation, and Lodgepole Creeks	1995	247	22
	1996	512	22
	1997	598	22
	1998	679	22
	1999	849	22
	2000	1195	22
	2001	1496	22
	2002	1892	22
	2003	2053	22
	2004	2133	22
	2005	642	22
	2006	2298	22
	2007	1883	22

Table 3 (Continued). Bull trout redd survey summary for all index tributaries in the Kootenai River Basin.

Stream	Year Surveyed	Number of Redds	Miles Surveyed
Skookumchuck Creek (B.C.)	1997	66	1.9
	1998	105	1.9
	1999	161	1.9
	2000	189	1.9
	2001	132	1.9
	2002	143	1.9
	2003	134	15
	2004	140	1.9
	2005	111	
	2006	163	
	2007	144	
White River (B.C.)	2001	166	7.8
Includes Blackfoot Creek in 2002, 2003, and 2005-2007	2002	261	7.8
	2003	249	
	2004	190	8.1
	2005	243	
	2006	311	
	2007	266	

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 3; $r^2 = 0.004$). Although the overall trend is positive, annual variation limits our ability to statistically distinguish this relationship from a stable (zero slope) population (Figure 3; $p = 0.811$). We observed a total of 35 redds in Quartz and West Fork Quartz creeks in 2007 (Table 3). The average number of redds of the period of record was 71.1 redds. The 2007 observation of 51 redds was 50.8% lower than the mean over the period of record. Despite the lack of a significant trend in bull trout redds within Quartz Creek, it still represents the strongest bull trout populations residing in the Kootenai Basin located downstream of Libby Dam. Keeler Creek is also a relatively strong population, but is a disjunct population residing in Lake Creek and Bull Lake.

Figure 3. Bull trout redd counts and trend analysis (blue line) for Quartz Creek (including West Fork Quartz) 1990-2007.

Pipe Creek

Bull trout redd counts in Pipe Creek peaked in 1999 at 36 redds. Redd numbers have generally decreased annually since 1996, and dropped to zero in 2007. Despite the recent decreases, there is no apparent overall general trend of bull trout redds in Pipe Creek during the period of record (1990-2007; Figure 4). The mean number of bull trout redds since 1990 has been 12.6 redds.

Bear Creek

Bear Creek bull trout redd counts have been variable during the period of record (1995-2007; Figure 5; $r^2 = 0.053$). Although the overall general trend has been a decreasing one since 1995, the relationship is not statistically different than a stable population (Figure 5; $p = 0.48$). We observed 9 redds in Bear Creek in 2007, which represents a 34% decrease over the mean number observed since surveys began in 1995.

O'Brien Creek

The general trend of bull trout redds in O'Brien Creek has generally increased since 1991 (Figure 6; $r^2 = 0.778$; $p < 0.0001$). We observed a total of 77 bull trout redds in O'Brien Creek in 2007, which represented the second highest redd count since we began surveying O'Brien Creek in 1991 (Table 3).

West Fisher Creek

The trend in bull trout redd abundance in the West Fisher Creek over the period of record (1993-2007) was nearly significant with the addition of the 2007 redd count (Figure 7; $r^2 = 0.238$; $p = 0.065$). We observed a total of 18 redds in the West Fisher Creek in 2007, which represented an approximate two fold increase over the mean number of the period of record (8.7 redds).

Figure 4. Bull trout redd counts and trend analysis (blue line) for Pipe Creek 1990-2007.

Figure 5. Bull trout redd counts and trend analysis (blue line) in Bear Creek, a tributary to Libby Creek, 1995-2007. The mean number of bull trout redds since 1995 is 13.6.

Figure 6. Bull trout redd counts and trend line (blue line) in O'Brien Creek 1991-2007.

Figure 7. Bull trout redd counts in the West Fisher Creek, a tributary to the Fisher River, 1993-2007.

Keeler Creek

Bull trout that spawn in Keeler Creek (including the North, South and West Forks) are an adfluvial stock that migrates downstream out of Bull Lake into Lake Creek, then up Keeler Creek. This downstream spawning migration is somewhat unique when compared to other bull trout populations (Montana Bull Trout Scientific Group 1996b). 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 84 bull trout redds in Keeler Creek and associated tributaries in 2007, which represented the lowest count since 2001 (Table 3), and the fourth lowest observation since 1995. This relatively low count we observed in 2007, nullified the significant trend in bull trout redds in Keeler Creek observed over the past several years (Figure 8; $r^2 = 0.259$; $p = 0.091$).

Figure 8. Bull trout redd counts and trend line (blue line) in Keeler Creek, a tributary to Lake Creek, 1996-2007.

Kootenai River Tributaries from Kootenai Falls to Libby Dam

The Montana bull trout spawning tributaries that have direct access to the Kootenai River between Kootenai Falls and Libby Dam include the West Fisher Creek, Bear, Pipe, and Quartz, and O'Brien creeks. These tributaries are somewhat geographically isolated due to complete and partial upstream passage barriers at Libby Dam and Kootenai Falls, respectively. MFWP has documented two bull trout that have ascended Kootenai Falls. One fish was a radio tagged bull trout in 2000 (Dunnigan et al. 2003), and a PIT tagged bull trout in 2008 (Sylvester et al. 2008). Therefore, it has been documented that Kootenai Falls is not a complete upstream barrier to bull trout migration at some discharges, but it remains likely that the falls represent a substantial barrier at least during certain periods of varying discharge. Therefore, we present two separate analyzes of bull trout redd abundance in the Kootenai River below Libby Dam. One analysis included all tributaries connected to the Kootenai River below Libby Dam including O'Brien Creek (below Kootenai Falls), and the other analysis omits O'Brien Creek. Bull trout redd counts first began in Bear Creek in 1995, and in the West Fisher in 1993, redd counts in the other tributaries began in 1990 (Table 3). Therefore in order to present an accurate assessment, the data presented in both analyzes was limited to the time period 1995-2007.

We observed a total of 53 bull trout redds in the four streams between Kootenai Falls and Libby Dam in 2007. Redd counts during this period peaked in 2001 at 161 redds. The overall mean bull trout redd count for this period equaled 98.2 redds, with no apparent trend over this period (Figure 9; $r^2 = 0.105$; $p = 0.28$). The 53 redds we observed in 2007 represented a 46.0% reduction from the overall average for the period.

The overall trend in total bull trout redds below Libby Dam is very similar when we include counts from O'Brien Creek (Figure 10). We observed a total of 35 bull trout redds in O'Brien Creek in 2007, bringing the total to 139 redds for all five streams in 2007. The overall mean bull trout redd count for this period equaled 158.0 redds, with no apparent trend over this period (Figure 10; $r^2 = 0.03$; $p = 0.86$). Similar to the trend of total redds for all streams located upstream of Kootenai Falls, the total observed in 2007, was lower than the overall average for the period of record by 8.1%.

Figure 9. Bull trout redd counts in the tributaries to the Kootenai River between Kootenai Falls and Libby Dam, including West Fisher Creek, Bear, Pipe, and Quartz creeks, 1995-2007.

Figure 10. Bull trout redd counts in the tributaries that have direct access to the Kootenai River below Libby Dam, including West Fisher Creek, Bear, Pipe, Quartz, and O'Brien creeks, 1995-2007.

Kootenai River Adult Bull Trout Population Estimate

We captured and marked 45 and 28 (73 total) bull trout on April 9 and 10, respectively. Three mortalities occurred during the two marking sessions. During the recapture sampling events on April 16 and 17, we captured 24 and 16 (16 total) bull trout, respectively. We recaptured a total of 7 bull trout originally marked on April 9 and 10, and estimated that 381 adult bull trout were present below Libby Dam during this period (Table 4). We also standardized each population estimate and 95% confidence interval into fish per mile, with a mean of 109 bull trout per mile (95% confidence interval = 43–175 fish per mile). The average bull trout total length was 603 mm (range = 237 – 900 mm; Figure 11). The mean length of bull trout in 2004, 2005, 2006, and 2007 captured below Libby Dam was 649, 677, 692, and 655 mm, respectively. The mean length of bull trout captured in 2008 (Figure 11) was significantly less than all previous years ($p < 0.05$). These were the only comparisons between years that differed.

We recaptured 16 bull trout during our sampling period April 9 –17, 2008 that were previously marked in 2004, 2005, 2006, or 2007 below Libby Dam ranging between 313 to 1,117 days prior. The recaptured bull trout grew an average of 96.3 mm (0.21 mm per day; Table 5), and gained an average of 1,649.9 g (3.07 g per day; Table 5). Two fish exhibited weight loss since the first capture. One of these fish lost 592 g 722 days after being initially tagged (0.82g/day), and the other lost 128 g in 344 days (3.07 g/day; Table 5). Both of these fish grew slightly longer (Table 5). All other recaptured fish gained length and weight between captures (Table 5).

Table 4. The sampling dates for the number of adult bull trout marked, recaptured, and the estimated total population and number of fish per mile in the Kootenai River from Libby Dam downstream to the Fisher River confluence. The 95 percent confidence intervals (CI) are presented in parentheses.

Dates	Number Marked	Number Recaptured	Total Population Estimate (95 % CI)	Fish per Mile (95 % CI)
April 9 and 10, 2008	73	N/A		
April 16 and 17, 2008	40	7	381 (158 –605)	109 (43 – 175)

Table 5. Recapture summary information for bull trout recaptured below Libby Dam between April 9 and 17, 2008. Information includes the date each fish was originally captured, recaptured, and length and weight for each encounter. Fish were captured via nighttime electrofishing. Mean daily growth rates are presented in parentheses.

Original Tag Date	Recapture Date	PIT tag Number	Length at Capture (mm)	Weight at Capture (g)	Length at Recapture (mm)	Weight at Recapture (g)	Length Increase (mm)	Weight Increase (g)
4/19/2006	4/10/2008	3D9.1BF1C41539	770	5445	797	4853	27	-592
4/8/2004	4/16/2008	3D9.1BF1C67300	550	1468	803	5350	253	3882
5/8/2007	4/16/2008	3D9.1BF1C679B7	758	5200	797	5072	39	-128
4/21/2004	4/16/2008	3D9.1BF1C682B5	543	1582	724	4111	181	2529
4/19/2006	4/10/2008	3D9.1BF1C68AB5	622	2695	788	5799	166	3104
3/31/2007	4/10/2008	3D9.257C59FA51	508	N/A	626	2744	118	
5/8/2007	4/16/2008	3D9.257C6C096E	658	2953	758	4957	100	2004
3/23/2007	4/9/2008	3D9.257C6C1197	787	N/A	806	5788	19	
3/28/2007	4/9/2008	3D9.257C6C1874	749	N/A	816	5657	67	
5/8/2007	4/16/2008	3D9.257C6C42DD	654	2815	687	3219	33	404
5/8/2007	4/9/2008	3D9.257C6C4815	756	4877	802	5802	46	925
2/9/2007	4/17/2008	3D9.257C6C4BFC	508	N/A	634	2685	126	
5/8/2007	4/17/2008	3D9.257C6C4CC8	577	2097	708	4382	131	2285
5/8/2007	4/9/2008	3D9.257C6C4F83	786	6877	796	7159	10	282
5/9/2007	4/9/2008	3D9.257C6C512D	746	5057	806	8030	60	2973
5/8/2007	4/10/2008	3D9.257C6C53AB	437	830	604	2760	167	1930
Mean			650.6	3491.3	747	4898	96.3 (0.21)	1,649.9 (3.07)

Figure 11. Length frequency distribution of bull trout captured via jet boat electrofishing on April 9 to 17, 2008 in the Kootenai River below Libby Dam. The mean total length for all fish captured was 603 mm.

Bull/Brook Trout Genetic Analyzes in Bull Trout Natal Streams

Keeler Creek

The sample from Keeler Creek contained 37 bull trout, eight brook trout, and five F₁ hybrids (Figure 12; Table 6). Bull trout, therefore, appear to be the predominant char in the section of Keeler Creek that was sampled. Brook trout and hybrids, however, are not uncommon.

Pipe Creek

The sample from Pipe Creek contained 48 brook trout and two later generation hybrids with brook trout (Figure 13; Table 6). Thus, bull trout appear to be either absent from the portion of the drainage that was sampled or are not very common.

West Fisher Creek

The sample from West Fisher Creek contained 36 bull trout, 12 brook trout, and one later generation hybrid with brook trout (Figure 14; Table 6). Thus, bull trout appear to be the predominant char in the reach of West Fisher Creek that was sampled. Brook trout, however, are not uncommon. In contrast, hybrids appear to constitute only a small proportion of the fish.

O'Brien Creek

The sample from O'Brien Creek contained two bull trout, 24 brook trout, 17 F₁ hybrids, four later generation hybrids with brook trout, and two later generation hybrids with bull trout (Figure 15; Table 6). Thus, brook trout and hybrids appear to be the predominant char in the section of O'Brien Creek that was sampled and bull trout constitute only a minor proportion of the fish.

Combined sample analysis

Among the four samples, seven fish were later generation hybrids with brook trout. In order to determine if these fish appeared to be first generation backcrosses, we compared the observed distribution of hybrid indices among them to the expected distribution among first generation backcrosses based on Mendelian inheritance. The observed distribution statistically conformed ($X^2_5=9.378$, $P>0.05$) to the expected suggesting these fish were first generation backcrosses with brook trout (Figure 16). A similar meaningful comparison was precluded for later generation hybrids with bull trout because overall only two such fish were sampled.

All of the fish collected were visually identified as being bull trout, brook trout, or hybrids. Compared to genetic identification, overall 91% of the fish were visually correctly identified (Tables 7 and 8). The most common visual misidentification involved calling later generation hybrids as being either brook trout or bull trout. Only 11% of these fish were visually identified as being hybrids. Not surprisingly, later generation hybrids with bull trout tended to be visually identified as bull trout and later generation hybrids with brook trout tended to be visually identified as brook trout. The other error was visually identifying F₁ hybrids as being either bull or brook trout. Only 55% of the F₁ hybrids were visually identified as being hybrids.

Table 6. Sample numbers and collection information for 198 *Salvelinus* spp. collected from 4 streams in the Kootenai Subbasin, and the genetic results.

Sample #	Stream Name /Collection		N	Taxa ID	Number of Individuals
	Date/Collector				
3723	Keeler Creek		50	BU	37
	7/24/2007			BU X BR	5
	Jim Dunnigan			BR	8
3724	Pipe Creek		50	F ₁ X BR	2
	7/3/2007			BR	48
	Jim Dunnigan				
3725	West Fisher Creek		49	BU	36
	7/9/2007			F ₁ X BR	1
	Jim Dunnigan			BR	12
3726	O'Brien Creek		49	BU	2
	7/16/2007			F ₁ X BU	2
	Jim Dunnigan			BU X BR	17
				F ₁ X BR	4
				BR	24

N=Sample size. BU=bull trout. BR=brook trout. BU X BR=first generation hybrid (F₁). F₁ X BU=backcross to bull trout. F₁ X BR=backcross to brook trout. F₂=second or later generation hybrid.

Table 7. Comparison of visually identified bull trout, brook trout and hybrids to genetic identifications. F1= First generation hybrid, F1XBU = backcross to a bull trout, F1XBR = backcross to a brook trout. Numbers in bold represent visual misidentifications.

Genetic Identification	Visual Identification		
	Bull	Hybrid	Brook
Bull	75	0	0
F ₁ X BU	2	0	0
F ₁	3	12	7
F ₁ X BR	0	1	6
Brook	0	0	92

Table 8. Char collected from three streams in the Kootenai Subbasin that were visually misidentified. F1= First generation hybrid, F1XBU = backcross to a bull trout, F1XBR = backcross to a brook trout.

Stream	ID #	Visual ID	Genetic ID
Pipe Creek	114	Brook	F ₁ X BR
Pipe Creek	497	Brook	F ₁ X BR
West Fisher Creek	468	Brook	F ₁ X BR
O'Brien Creek	83	Bull	F ₁
O'Brien Creek	312	Brook	F ₁ X BR
O'Brien Creek	325	Brook	F ₁ X BR
O'Brien Creek	330	Brook	F ₁
O'Brien Creek	332	Brook	F ₁
O'Brien Creek	333	Brook	F ₁
O'Brien Creek	334	Bull	F ₁ X BU
O'Brien Creek	336	Brook	F ₁
O'Brien Creek	344	Brook	F ₁
O'Brien Creek	362	Brook	F ₁
O'Brien Creek	364	Brook	F ₁ X BR
O'Brien Creek	365	Bull	F ₁ X BU
O'Brien Creek	381	Brook	F ₁
O'Brien Creek	491	Bull	F ₁
O'Brien Creek	502	Bull	F ₁

Figure 12. Distribution of hybrid indices among char collected from Keeler Creek.

Figure 13. Distribution of hybrid indices among char collected from Pipe Creek.

Figure 14. Distribution of hybrid indices among char collected from West Fisher Creek.

Figure 15. Distribution of hybrid indices among char collected from O'Brien Creek.

Figure 16. Observed and expected distribution of hybrid indices among hybrids appearing to be first generation backcrosses to brook trout collected from three streams in the Kootenai River drainage. Note the observed distribution statistically conforms ($P > 0.05$) to the expected distribution.

Bull Spawning Habitat Surveys

The adjunct bull trout population that resides in Bull Lake and spawns in Keeler Creek had the lowest mean levels of fine sediment (< 6.35 mm) amongst the eight streams we monitored. Fine sediment levels in the monitoring section of North Fork Keeler Creek averaged 30.2% from 1998-2007 (Figure 17; Appendix Table 10). Levels of fine sediment were nearly significant between years ($p = 0.052$; power = 0.80), and ranged between 22.8% in 2000 to 36.7% in 2003.

O'Brien Creek had the highest mean levels of fine sediment amongst the eight stream we monitored. Mean annual fine sediment levels in O'Brien Creek ranged from a low of 34.0% in 2001 to a high of 46.1% in 2005, and had an overall annual mean fine sediment level of 40.6% (Appendix Table 10). Mean annual fine sediment levels in O'Brien Creek did differ between years ($p = 0.002$; Figure 18). Multiple comparisons between years indicated that fine sediment levels observed in 2001 were significantly

lower than those observed in 2003, 2004 and 2005, and that 2005 was significantly higher than 2007. All other annual comparisons were not significantly different ($p > 0.05$).

Of the four core bull trout spawning tributaries located on the Kootenai River between Libby Dam and Kootenai Falls, West Fork Quartz Creek had the lowest annual mean fine sediment levels. Sediment levels averaged 33.4% from 1998 to 2006 and ranged from 24.6% in 2002 to 37.5 in 1998 (Figure 19; Appendix Table 10). Fine sediment levels on West Fork Quartz Creek did not differ between years ($p = 0.331$). However, the power of this test was relatively low (0.429). Mean annual fine sediment levels on Pipe Creek and Bear Creek were both relatively high, but similar. We first began monitoring sediment levels in Bear Creek in 2002, and mean annual sediment levels in Bear Creek averaged 38.6% (range = 28.3-42.5; Figure 20). The only two years that differed significantly were 2003 and 2004 ($p=0.009$). All other annual comparisons were statistically similar. Mean annual fine sediment levels on Pipe Creek averaged 38.4% from 1998-2006 (range = 33.3-47; Figure 21; Appendix Table 10). Fine sediment levels on Pipe Creek were not significantly different between years ($p = 0.16$; power = 0.616). Fine sediment levels on the West Fisher Creek were first monitored in 2006 and averaged 32.3% (Figure 22).

The two bull trout spawning tributaries located in Montana upstream of Libby Dam (Grave Creek and MT Wigwam) had relatively low fine sediment levels. We monitored fine sediment levels in Grave Creek annually from 1998 to 2006, with the exception of 1999. During this time period, fine sediment levels in Grave Creek averaged 32.2%, ranging from 28.8% to 36.8% (Figure 23; Appendix Table 10). ANOVA suggested that yearly differences in fine sediment levels were nearly significant ($p = 0.056$). The power of this test was 0.763. We monitored fine sediment in the Montana portion of the Wigwam River annually from 1998 to 2007, with the exception of 2001, 2002 and 2003 when conditions prevented us from accessing the stream coring sites. Mean annual fine sediment levels in the Montana portion of the Wigwam River averaged 34.1%, and ranged from a low value of 30.6% in 2007 to 38.3% in 2004 (Figure 24; Appendix Table 10). Fine sediment levels in the Montana portion of the Wigwam River did not differ significantly between years ($p = 0.495$). However the power of this test was low (0.327).

Figure 17. Fine sediment levels (<6.35 mm) in North Fork Keeler Creek. The annual mean values appear above each year, and the whisker bars represent 95% confidence intervals. The p-value from an analysis of variance that tested for annual differences is also presented.

Figure 18. Fine sediment levels (<6.35 mm) in O'Brien Creek. The annual mean values appear above each year, and the whisker bars represent 95% confidence intervals. The p-value from an analysis of variance that tested for annual differences is also presented. Significant annual differences are listed.

Figure 19. Fine sediment levels (<6.35 mm) in West Fork Quartz Creek. The annual mean values appear above each year, and the whisker bars represent 95% confidence intervals. The p-value from an analysis of variance that tested for annual differences is also presented.

Figure 20. Fine sediment levels (<6.35 mm) in Bear Creek. The annual mean values appear above each year, and the whisker bars represent 95% confidence intervals. The p-value from an analysis of variance that tested for annual differences is also presented. Significant annual differences are listed.

Figure 21. Fine sediment levels (<6.35 mm) in Pipe Creek. The annual mean values appear above each year, and the whisker bars represent 95% confidence intervals. The p-value from an analysis of variance that tested for annual differences is also presented.

Figure 22. Fine sediment levels (<6.35 mm) in West Fisher Creek. The annual mean values appear above each year, and the whisker bars represent the 95% confidence interval.

Figure 23. Fine sediment levels (<6.35 mm) in Grave Creek. The annual mean values appear above each year, and the whisker bars represent 95% confidence intervals. The p-value from an analysis of variance that tested for annual differences is also presented.

Figure 24. Fine sediment levels (<6.35 mm) in the Montana portion of the Wigwam River. The annual mean values appear above each year, and the whisker bars represent 95% confidence intervals. The p-value from an analysis of variance that tested for annual differences is also presented.

Discussion

Long-term monitoring of bull trout redd numbers can be an important tool to assess bull trout population trends (Rieman & McIntyre 1993). Bull trout redd counts in the tributaries between Kootenai Falls and Libby Dam were 60% of the overall mean from 1995-2006, with only 61 redds being observed in all four tributaries. 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 their ratios to bull trout redd counts in the 4 spawning tributaries below Libby Dam (53 redds) and extrapolate, there may have been 82 (range 64 – 111) spawning bull trout in the Kootenai River below Libby Dam in 2007. However, we estimated a total of 381 bull trout (95% confidence interval = 158 – 605) were present in the three miles of the Kootenai River directly below Libby Dam in early spring 2008 (see above). Based on the total length of the bull trout captured during this estimate, we believe most were adult fish. Baxter and Baxter (2002) operated a fish trap on Skookumchuck Creek in 2001 and estimated the mean size of spawning adults to be 640 mm (range 400-920 mm), which was similar to the size of fish we found below Libby Dam during our spring 2008 population estimate (mean total length 603 mm; range 237-900 mm). Although we acknowledge that the 2007 bull trout redd counts within the tributaries below Libby Dam may have underestimated of the total Kootenai River bull trout population because they did not take into account alternate year spawning individuals or immature fish that may not have participated in spawning that year, the disparity between our adult population estimate and the redd count data is nevertheless large.

Bull trout entrainment at Libby Dam is a major factor that partially explains the large differences between cumulative redd counts downstream of Libby Dam and our adult bull trout population estimate below Libby Dam. Populations of bull trout upstream of Libby Dam have exhibited significant positive increases in abundance over the past several years, and likely occur at a level of abundance at least an order of magnitude higher than populations in the Kootenai River between Kootenai Falls and Libby Dam. Skaar et al. (1996) estimated that bull trout were the tenth most common fish entrained through Libby Dam during the period January 1992 to June 1994, comprising 1.8% of the catch of nets fished within the draft tubes at Libby Dam. However, bull trout abundance in Libby Reservoir during the past few years is probably at least 5-6 fold higher than it was during the period Skaar et al. (1996) conducted the entrainment study (see bull trout redd count section above).

A recent study funded by this project investigated the genetic structure of bull trout in the Kootenai Basin, and found significant differences between populations residing in all of the known spawning tributaries in Montana and British Columbia (Ardren et al. 2007). Genetic differences were sufficient between tributaries to provide a means to correctly assign unknown fish captured in the Kootenai River as originating either upstream or downstream of Libby Dam with a high degree of power (greater than 95%). Ardren et al. (2007) conducted population assignments for adult bull trout collected from the Kootenai River below Libby Dam in the spring of 2006, and DeHaan

et al. (2008) performed a similar analysis for fish collected in 2004, 2005 and 2007. From these studies, we estimated that the proportion of bull trout residing below Libby Dam between 2004 and 2007 that were entrained ranged from 49.1% to 62.7% (Table 9). Ardren et al. (2007) and DeHaan et al. (2008) estimated that most of the fish identified as entrained fish originated from the Wigwam River, and that most of the fish below Libby Dam identified as originating downstream of the dam originated in the West Fisher Creek. However, these results contradict annual bull trout redd counts and juvenile abundance (MFWP, unpublished data) monitoring datasets collected in the West Fisher Creek over the past decade, where adult escapement and juvenile abundance estimates are typically low compared to other bull trout streams below Libby Dam. Given these results it seems likely that an unknown portion of the entrained bull trout may be spawning in this tributary presumably due to its close proximity to Libby Dam.

Brook and bull trout hybridization was higher than expected in the four Kootenai bull trout spawning tributaries with sympatric distribution. We were only able to visually identify slightly more than half (55%) of the F1 hybrids, indicating the need for periodic genetic testing of these stream through time to determine introgression trends through time. Introgression was highest for O'Brien and Pipe creeks and coincided with the highest levels of fine sediment (see above) and brook trout abundance (MFWP, unpublished data). In these instances it seems likely that habitat degradation and introgression are acting synergistically to reduce bull trout abundance within these two tributaries. Indeed in each case, juvenile abundance has decreased (MFWP, unpublished data), but in the case of O'Brien Creek, redd counts have not exhibited a corresponding decrease. In fact, redd counts in O'Brien have exhibited a significant positive trend and the 2007 count was the second highest on record. This situation warrants further investigation.

Table 9. Extrapolations from Ardren et al. (2007) and DeHaan et al. 2008 to estimate the percent of bull trout residing below Libby Dam that were entrained in 2004-2007.

Year	Percent Entrained	95% Confidence Interval
2004	62.7%	51.1% – 74.2%
2005	50.0%	31.7% - 68.3%
2006	49.1%	38.5% - 59.9%
2007	56.6%	47.1% - 66.2%

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Appendix

Appendix A.

The catch per trap is expected to have a Poisson-like distribution; therefore, the usual least-squares analysis of variance was deemed inappropriate. A general linear regression procedure was adopted where the log of the catch-per-trap was regressed on indicators for reach and site and on date that the trap was set as a covariate. Since effort varied slightly over samples, an adjustment for effort was made based on the natural log (ln) transformation

$$\ln\left[\frac{\text{catch}}{\text{effort}}\right] = \ln(\text{catch}) - \ln(\text{effort})$$

$y = \ln(\text{catch})$ served as variable analyzed and $\ln(\text{effort})$ was an offset variable.

For each reach, means were adjusted using the exponential (exp) retransform

$$\text{Catch}'(\text{site}) = \exp\{[(\text{site's regression estimate mean } y) - \text{site's mean } \ln(\text{effort})] + [\text{predicted coefficient}(\text{date trap set})][\text{mean}(\text{trap-set date})]\}$$

The Catch per site was then adjusted as follows

$$\text{Adjusted Catch (Site)} = \text{Catch}'(\text{site}) - \text{Mean-over-sites}[\text{Catch}'(\text{site})] + \text{Mean-over-sites of actual data}$$

This latter adjustment was performed because exponential retransforms of natural-log fits on covariates are biased.

The reach's adjusted catch estimates were then divided by the reach's effective effort, the effective effort being the reach's actual mean untransformed catch divided the reach's actual mean catch per unit effort.

All statistical comparison are based on differences in adjusted log transforms and their associated standard errors. The standard errors of the retransformed catch means were approximated using the delta method.

Table A1. Therriault Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis. If the upper confidence interval is not presented, it was not able to be calculated because all fish were captured on the first pass of the depletion. Therriault Creek was not sampled during the 2000 or 2002 field seasons, and only Section 2 was sampled in 2001.

Year	1997	1998	1999	2001	2003	2004	2005	2006	2007
Section 1									
Rainbow Trout	123 (261)	130 (151)	82 (89)	0	56 (57)	108 (111)	106 (119)	121 (124)	53 (n/a)
Brook Trout	41 (47)	49 (56)	60 (64)	0	59 (66)	11 (13)	66 (73)	114 (120)	101 (104)
Bull Trout	0	0	0	0	0	92 (95)	10 (n/a)	48 (54)	28 (31)
Total Population ^A	149 (214)	182 (207)	141 (149)	0	115 (122)	200 (203)	175 (201)	235 (241)	154 (157)
Section 2									
Rainbow Trout	36 (41)	79 (82)	76 (83)	93 (102)	84 (n/a)	102 (107)	32 (34)	42 (43)	11 (n/a)
Brook Trout	56 (58)	125 (137)	72 (80)	82 (87)	58 (61)	24 (27)	67 (91)	46 (48)	40 (42)
Bull Trout	47 (49)	15 (16)	3 (n/a)	2 (n/a)	40 (42)	49 (53)	4 (n/a)	4 (n/a)	2 (n/a)
Total Population ^A	92 (96)	205 (217)	149 (163)	180 (193)	144 (151)	153 (160)	95 (107)	123 (125)	53 (55)
Section 3									
Rainbow Trout	54 (58)	164 (170)	177 (205)	0	99 (104)	112 (117)	99 (109)	28 (29)	15 (n/a)
Brook Trout	74 (77)	82 (88)	110 (117)	0	67 (72)	41 (45)	82 (90)	46 (48)	57 (59)
Bull Trout	0	0	0	0	10 (n/a)	3 (n/a)	15 (17)	2 (n/a)	4 (n/a)
Total Population ^A	66 (93)	248 (257)	284 (308)	0	170 (180)	118 (124)	183 (201)	74 (76)	72 (74)

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

Table A2. Grave Creek Demonstration Project population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis.

Year	2000 ^A	2001 ^B	2002 ^C	2003	2004	2005	2006	2007
Westslope Cutthroat	4	18	3	13 (n/a)	4 (n/a)	14 (15)	16 (17)	12 (13)
Rainbow Trout	1	17	26	25 (29)	41 (45)	63 (66)	25 (27)	10 (12)
Brook Trout	1	10	5	9 (18)	1(n/a)	3 (7)	4 (n/a)	3 (n/a)
Bull Trout	9	33	5	41 (144)	63 (67)	63 (66)	84 (87)	72 (84)
Mountain Whitefish	54	3	33	21 (22)	70 (73)	60 (62)	47 (48)	51 (52)

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

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

Year	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007
Section 1 (Tooley)											
Westslope Cutthroat ^B	3	36 (37)	139 (148)	0	55 (64)	88 (96)	Not sampled	68 (70)	66 (72)	61 (63)	47 (51)
Rainbow Trout ^B	19 (23)	62 (70)	3 (n/a)	0	2 (n/a)	14 (19)		8 (n/a)	2 (n/a)	2 (n/a)	2 (n/a)
Brook Trout	11 (17)	120 (124)	102 (105)	0	36 (39)	30 (31)		20 (n/a)	72 (80)	30 (36)	20 (24)
Mountain Whitefish	0	0	0	0	0	2 (n/a)		2 (n/a)	4 (n/a)	2 (n/a)	0
Total Population ^A	36 (40)	220 (228)	248 (258)	0	96 (107)	148 (158)		96 (98)	86 (96)	95 (101)	67 (71)
Section 4 (303 Rd.)											
Westslope Cutthroat	100 (114)	439 (500)	352 (367)	0	130 (142)	222 (237)	Not sampled	218 (228)	327 (351)	323 (337)	165 (170)
Rainbow Trout	0	0	0	0	0	0		0	0	2 (n/a)	0
Brook Trout	0	0	3 (n/a)	0	6 (12)	4 (n/a)		10 (12)	12 (17)	26 (30)	5 (11)
Bull Trout	0	0	0	0	0	0		0	0	0	1 (n/a)
Total Population ^A	100 (114)	439 (500)	358 (373)	0	136 (148)	232 (249)		230 (241)	338 (364)	351 (366)	169 (174)
Section 5 (State)											
Westslope Cutthroat	0	216 (227)	256 (290)	126 (153)	153 (174)	268 (290)	178 (183)	115 (118)	151 (164)	137 (143)	57 (60)
Rainbow Trout	0	0	0	0	0	0	0	0	0	0	0
Brook Trout	0	62 (71)	52 (65)	19 (22)	25 (27)	46 (49)	35 (n/a)	60 (63)	142 (147)	93 (96)	57 (60)
Bull Trout	0	0	0	0	0	2 (n/a)	0	3 (n/a)	2 (n/a)	3 (5)	2 (n/a)
Total Population ^A	0	280 (294)	314 (353)	113 (119)	176 (195)	315 (335)	213 (183)	230 (241)	296 (309)	115 (122)	115 (122)

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

B) Sampling crew did not distinguish between westslope cutthroat trout and rainbow trout.

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

Year	1998	1999 ^A	2000 ^A	2001	2002	2003	2004	2005	2006	2007
Section 1 – below Hwy 2										
Rainbow Trout	81 (127)	26	125	46 (51.09)	117 (130)	84 (96)	113 (118)	169 (191)	271 (293)	174 (422)
Brook Trout	6 (8)	6	13	10 (12.33)	16 (24)	5	9 (15)	57 (64)	26 (27)	19 (n/a)
Bull Trout	0	0	0	0	3	0	1 (n/a)	1 (n/a)	0	0
Mountain Whitefish	0	0	0	0	3	1	0	0	0	1 (n/a)
Total Population ^B	90 (116)	32	138	57 (63.79)	138 (153)		138 (144)	227 (256)	296 (317)	261 (615)
Section 2 –above Hwy 2										
Rainbow Trout	203 (225)	0	0	148 (193)	0	100 (108)	120 (128)	76 (92)	117 (122)	98 (120)
Brook Trout	7	0	0	2	0	2	30 (34)	25 (28)	19 (20)	23 (25)
Bull Trout	5 (6)	0	0	0	0	2.08	0	2 (n/a)	2 (n/a)	0
Total Population ^B	208 (228)	0	0	160 (213)	0		150 (160)	105 (116)	135 (141)	123 (139)
Section 3 – upper Cleveland										
Rainbow Trout	0	0	170 (194)	172 (182)	163 (183)	112.3 (127)	88 (104)	63 (75)	105 (110)	30 (34)
Brook Trout	0	0	0	0	0	0	0	0	0	0
Bull Trout	0	0	3	8 (11)	7	11 (14)	2 (n/a)	2 (n/a)	3 (n/a)	0
Mountain Whitefish	0	0	0	0	1	0	0	0	0	0
Total Population ^B	0	0	170 (194)	172 (182)	163 (183)		88 (104)	63 (75)	105 (110)	30 (34)

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 A4 (Continued). 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	2000	2001	2002	2003	2004	2005	2006	2007
Section 4 – below lower Cleveland										
Rainbow Trout							352 (365)	273 (283)	314 (324)	141 (148)
Brook Trout							0	2 (n/a)	2 (n/a)	1 (n/a)
Bull Trout							5 (n/a)	0	0	1 (n/a)
Total Population ^B							355 (368)	276 (286)	316 (326)	143 (150)
Section 5 –above lower Cleveland										
Rainbow Trout							172 (185)	173 (183)	170 (177)	129 (144)
Brook Trout							0	0	0	0
Bull Trout							6 (n/a)	0	0	2 (n/a)
Total Population ^B							172 (185)	173 (183)	170 (177)	129 (144)
Section 6 – lower Cleveland										
Rainbow Trout							218 (234)	221 (250)	273 (298)	133 (158)
Brook Trout							1 (n/a)	0	0	6 (9)
Bull Trout							0	4 (n/a)	0	0
Total Population ^B							219 (235)	221 (250)	273 (298)	141 (169)

Table A5. Pipe Creek depletion population estimate for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals surveyed directly downstream of the Bothman Road Bridge. Upper confidence intervals are in parenthesis.

Year	2001	2002 ^B	2003	2004	2005	2006	2007
Rainbow Trout	42 (46)	73 (85)	39 (43)	25 (27)	21 (25)	69 (73)	54 (59)
Brook Trout	0	3	7 (8)	4 (n/a)	6 (10)	15 (n/a)	5 (n/a)
Bull Trout	0	0	0	0	0	2 (n/a)	0
Total Population ^A	42 (46)	73 (85)	0	27 (29)	27 (31)	83 (85)	59 (64)

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 A6. 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 2007. *Epischura* and *Leptodora* were measured as number per m³. Sampling was not conducted in September or October 2007.

Month	(N)	<i>Daphnia</i>	<i>Bosmina</i>	<i>Diaptomus</i>	<i>Cyclops</i>	<i>Leptodora</i>	<i>Epishura</i>	<i>Diaphanosoma</i>
April	(3)	0.03	0.40	0.03	1.53	0.00	0.00	0.00
		0.00	0.14	0.00	0.53	0.00	0.00	0.00
May	(3)	1.41	2.92	0.74	39.57	1.18	704.48	0.00
		0.56	3.15	0.00	14.33	1.16	36,076.62	0.00
June	(3)	0.74	1.10	0.12	6.76	0.71	53.76	0.01
		0.02	0.41	0.01	1.39	1.50	1,257.03	0.00
July	(3)	3.76	0.01	0.80	10.09	1.88	211.72	0.22
		2.32	0.00	0.08	1.25	0.17	1,685.47	0.01
August	(3)	2.67	0.07	2.96	9.01	1.18	589.43	0.12
		0.24	0.00	0.56	0.24	0.66	87,967.20	0.01
November	(3)	0.80	2.71	0.27	4.29	0.00	42.44	0.00
		0.12	3.59	0.01	3.52	0.00	2,409.18	0.00

Table A7. 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 2007. *Epischura* and *Leptodora* were measured as number per m³. Sampling was not conducted in September or October 2007.

Month	(N)	<i>Daphnia</i>	<i>Bosmina</i>	<i>Diaptomus</i>	<i>Cyclops</i>	<i>Leptodora</i>	<i>Epishura</i>	<i>Diaphanosoma</i>
April	(3)	0.68	0.45	0.05	20.48	0.00	0.47	0.00
		0.24	0.10	0.01	305.51	0.00	0.66	0.00
May	(3)	0.43	0.50	0.10	8.90	0.47	26.41	0.03
		0.06	0.13	0.00	34.16	0.66	87.21	0.00
June	(3)	3.50	0.27	0.22	19.26	8.02	0.00	0.00
		0.78	0.02	0.02	251.92	28.72	0.00	0.00
July	(3)	4.12	0.01	0.30	10.49	5.66	37.72	0.42
		0.88	0.00	0.04	4.41	5.99	4,269.15	0.07
August	(3)	0.60	0.04	2.48	3.85	0.00	364.27	0.04
		0.02	0.00	0.57	2.16	0.00	1,042.32	0.00
November	(3)	1.36	4.07	0.58	6.30	0.94	16.03	0.00
		1.15	12.16	0.08	12.63	2.67	210.74	0.00

Table A8. 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 2007. *Epischura* and *Leptodora* were measured as number per m³. Sampling was not conducted in September or October 2007.

Month	(N)	<i>Daphnia</i>	<i>Bosmina</i>	<i>Diaptomus</i>	<i>Cyclops</i>	<i>Leptodora</i>	<i>Epishura</i>	<i>Diaphanosoma</i>
April	(3)	0.02	0.03	0.03	1.19	1.53	34.85	0.01
		0.00	0.00	0.00	1.02	0.04	310.89	0.00
May	(3)	0.14	0.31	0.02	2.23	0.00	13.46	0.02
		0.03	0.19	0.00	8.17	0.00	325.22	0.00
June	(3)	2.35	0.06	0.07	6.71	8.96	9.90	0.14
		4.10	0.00	0.01	34.12	84.71	254.14	0.04
July	(3)	1.86	0.51	0.26	1.53	2.48	46.04	0.71
		2.21	0.49	0.04	2.12	1.65	1,938.52	0.56
August	(3)	0.34	0.06	1.39	1.38	0.00	338.38	0.01
		0.01	0.00	1.23	0.49	0.00	4,704.88	0.00
November	(3)	1.84	2.20	1.73	5.54	0.00	19.45	0.00
		0.92	7.93	1.60	13.45	0.00	1,135.30	0.00

Table A9. 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 m³.

Year	(N)	<i>Daphnia</i>	<i>Bosmina</i>	<i>Diaptomus</i>	<i>Cyclops</i>	<i>Leptodora</i>	<i>Epishura</i>	<i>Diaphanosoma</i>
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
2003	72	3.42	0.83	1.79	11.34	2.24	98.02	0.90
		20.29	1.93	4.46	64.61	19.74	19,825.83	1.68
2004	72	2.10	1.63	1.38	10.26	3.39	95.06	0.53
		6.70	8.72	3.21	169.71	29.53	37,077.33	0.88
2005	72	1.50	2.62	0.51	7.74	2.43	91.36	0.30
		4.05	37.88	0.59	80.18	26.13	15,412.56	0.19
2006	63	1.81	1.09	1.37	9.10	2.78	121.03	0.23
		2.65	3.42	2.24	69.20	16.67	28,439.64	0.16
2007	54	1.48	0.87	0.68	8.84	1.83	139.38	0.10
		2.19	2.53	0.92	112.66	12.33	50,542.01	0.06

Table A10. Mean annual fine sediment (<6.35 mm) in eight Montana bull trout spawning tributaries of the Kootenai River. 95% confidence intervals are presented in parenthesis.

Stream	Year	Sample Size	Mean % Fine Sediment
N. Fork Keeler Crk.	1998	12	32.7 (23.6 - 42.4)
N. Fork Keeler Crk.	1999	0	N/A
N. Fork Keeler Crk.	2000	11	22.8 (17.5 – 28.5)
N. Fork Keeler Crk.	2001	12	29.5 (25.9 – 33.3)
N. Fork Keeler Crk.	2002	12	31 (26.4 – 35.8)
N. Fork Keeler Crk.	2003	12	36.7 (28.1 – 45.7)
N. Fork Keeler Crk.	2004	12	36.2 (30.1 – 42.6)
N. Fork Keeler Crk.	2005	12	30.3 (25 – 36)
N. Fork Keeler Crk.	2006	12	23.9 (14.4 – 34.9)
N. Fork Keeler Crk.	2007	12	28.4 (20.2 – 37.3)
O'Brien Crk.	1998	12	42.1 (38.5 – 45.7)
O'Brien Crk.	1999	0	N/A
O'Brien Crk.	2000	0	N/A
O'Brien Crk.	2001	12	34 (30.4 – 37.8)
O'Brien Crk.	2002	12	39.8 (35.3-44.4)
O'Brien Crk.	2003	12	43 (37.9 – 48.2)
O'Brien Crk.	2004	12	43.3 (40.6 – 45.9)
O'Brien Crk.	2005	12	46.1 (41.9 – 50.3)
O'Brien Crk.	2006	12	40 (35.4-44.9)
O'Brien Crk.	2007	12	36.8 (30.7-43.2)
W. Fork Quartz Crk.	1998	8	37.5 (30.7 – 44.6)
W. Fork Quartz Crk.	1999	0	N/A
W. Fork Quartz Crk.	2000	0	N/A
W. Fork Quartz Crk.	2001	11	37.3 (32.8 – 42)
W. Fork Quartz Crk.	2002	12	24.6 (18.3 – 31.5)
W. Fork Quartz Crk.	2003	12	34.1 (31.7 – 36.6)
W. Fork Quartz Crk.	2004	8	36.7 (31.2 – 44.3)
W. Fork Quartz Crk.	2005	8	30.6 (23.3 – 38.4)
W. Fork Quartz Crk.	2006	12	32.3 (27.9 – 36.8)
W. Fork Quartz Crk.	2007	0	N/A
Bear Crk.	1998	0	N/A
Bear Crk.	1999	0	N/A
Bear Crk.	2000	0	N/A
Bear Crk.	2001	0	N/A
Bear Crk.	2002	8	34.5 (30.5 – 38.6)
Bear Crk.	2003	8	28.3 (19.3 – 38.2)
Bear Crk.	2004	12	42.5 (38 – 47.4)
Bear Crk.	2005	12	39 (33.9 – 44.2)
Bear Crk.	2006	8	38.7 (28.7 – 49.2)
Bear Crk.	2007	0	N/A

Table A10 (continued). Mean annual fine sediment (<6.35 mm) in eight Montana bull trout spawning tributaries of the Kootenai River. 95% confidence intervals are presented in parenthesis.

Stream	Year	Sample Size	Mean % Fine Sediment
Pipe Crk.	1998	10	47 (41.4 – 52.6)
Pipe Crk.	1999	10	35.3 (26.5 – 44.7)
Pipe Crk.	2000	0	N/A
Pipe Crk.	2001	12	33.3 (25.9 – 41.2)
Pipe Crk.	2002	12	38.5 (33.2 – 45.7)
Pipe Crk.	2003	12	39.8 (31.1 – 43.4)
Pipe Crk.	2004	12	37.1 (31 – 43.4)
Pipe Crk.	2005	12	38.1 (33.6 – 42.6)
Pipe Crk.	2006	12	38 (32.8 – 43.2)
Pipe Crk.	2007	0	N/A
West Fisher Crk.	1998	0	N/A
West Fisher Crk.	1999	0	N/A
West Fisher Crk.	2000	0	N/A
West Fisher Crk.	2001	0	N/A
West Fisher Crk.	2002	0	N/A
West Fisher Crk.	2003	0	N/A
West Fisher Crk.	2004	0	N/A
West Fisher Crk.	2005	0	N/A
West Fisher Crk.	2006	12	32.3 (25.9 – 39.1)
West Fisher Crk.	2007	0	N/A
Grave Crk.	1998	12	28.8 (23.3 – 34.5)
Grave Crk.	1999	0	N/A
Grave Crk.	2000	11	29.8 (19.4 – 41.5)
Grave Crk.	2001	7	32.7 (25.6 – 40.3)
Grave Crk.	2002	10	33.6 (27.1 – 40.6)
Grave Crk.	2003	12	36.4 (33.2 – 39.6)
Grave Crk.	2004	12	36.8 (31.7 – 42)
Grave Crk.	2005	12	30.7 (25.4 – 36.3)
Grave Crk.	2006	12	29.1 (23.9 – 34.6)
Grave Crk.	2007	0	N/A
Wigwam River (MT)	1998	6	34.2 (29 – 39.7)
Wigwam River (MT)	1999	3	36.6 (25 – 49)
Wigwam River (MT)	2000	6	32.6 (27 – 38.6)
Wigwam River (MT)	2001	0	N/A
Wigwam River (MT)	2002	0	N/A
Wigwam River (MT)	2003	0	N/A
Wigwam River (MT)	2004	12	38.3 (34.1 – 43.5)
Wigwam River (MT)	2005	8	33.9 (26 – 42.1)
Wigwam River (MT)	2006	12	32 (22.8 – 42.1)
Wigwam River (MT)	2007	12	30.6 (26.9 – 34.3)