Mitigation For The Construction and Operation of Libby Dam

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Table of Contents

Table of Figures

CHAPTER 2 FIGURES

CHAPTER 3 FIGURES

CHAPTER 4 FIGURES

CHAPTER 5 FIGURES

CHAPTER 6 FIGURES

CHAPTER 7 FIGURES

CHAPTER 8 FIGURES

CHAPTER 9 FIGURES

CHAPTER 10 FIGURES

CHAPTER 11 FIGURES

List of Tables

CHAPTER 2 TABLES

CHAPTER 3 TABLES

CHAPTER 4 TABLES

CHAPTER 5 TABLES

CHAPTER 7 TABLES

Executive Summary

Project 199500400 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 high priority mitigation actions as directed by the Kootenai Subbasin Plan.

Within this report, we present physical and biological monitoring from five stream restoration projects on three separate streams. 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 restore channel stability, increase pool-type habitats and habitat complexity. Young and Libby creeks are generally similar in stream channel type, except for Therriault Creek. These restoration projects unequivocally changed the pattern, profile and dimension of the streams within the project areas. Within the riffle habitats several conditions were generally evident for each restoration project. We were generally able to demonstrate significant increases in mean bankfull depth and a decrease in stream bankfull width, and annual changes in channel dimensions since project completion were relatively small. Pool-type habitats generally changed more so than riffle habitats after construction. All the restoration projects presented within this report demonstrated substantial increases in the quantity, depth and pool frequency within the project areas. Total pool numbers and total pool area and volume increased by several fold compared to pre-project conditions. However, several of the project areas have been impacted by large flood events, which have resulted in a slight annual loss of the total number of pools, mean pool depth, and increase in riffle width, but despite these changes, the stream channel dimensions within the project areas are still maintained over conditions that existed prior to project construction.

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, and the ecological processes that degraded the habitat must be altered. Our results show that the physical changes have been sustained through time, with almost every metric of habitat quantity and quality remaining substantially higher several years after project completion. Our efforts during the past several years has focused on restoring riparian vegetation at the Therriault Creek Project. These efforts are intended to restore natural processes that create and maintain salmonid habitat, which will increase the long-term sustainability and potential for project success.

This project monitors fish populations in most of the stream restoration projects completed. Efforts summarized in previous reports documented increased salmonid abundance after restoration efforts were complete. However, the results summarized in this report often either failed to demonstrate a positive population response, or lacked power to detect significant changes even when the BACI design was utilized. The clear lack of statistical power in most cases resulted from a limited pre-restoration time series coupled with high annual variability within a particular site. The high annual variability suggests that other physical factors that we did not measure may be important factors that influence population abundance at these sites.

The population level response within the restoration project areas had differing responses, making generalizations between projects difficult. Previous reports summarized the results of fisheries population monitoring associated with projects on Grave Creek and Libby Creek where resident salmonid abundance increased after the restoration projects. The Young Creek Restoration Project increased the abundance of brook trout, but westslope cutthroat trout abundance decreased, which resulted in an overall non-significant increase in total trout abundance. In this situation, it seems likely that ecological interactions between the two species are confounding the results of the improved habitat conditions. In addition, a large wildlife burned much of the Young Creek watershed in 2017, which resulted in a decrease in trout abundance throughout the watershed, with diminishing effects with distance from the burn. The two restoration efforts in upper Libby Creek have yet to result in an increase in redband trout abundance. These results were probably influenced by the large flood event that occurred shortly after the restoration work. Results were similar for the Therriault Creek Restoration Project. However, our monitoring strategy for Therriault Creek may have grossly underestimated total abundance within the project area due to the approximate doubling of stream length after implementation.

The Kootenai River Fertilization project is a collaborative effort between Idaho Fish and Game (Project 198806500) and the Kootenai Tribe of Idaho (Project 199504900*)* and represents the largest river nutrient enhancement effort in the world. The ability to determine the success of this project relies on a rigorous monitoring and evaluation effort that spans all trophic levels*.* The ability to evaluate the fish community response (and other trophic levels as well) is dependent on a carefully and well thought out monitoring program. This project provides a critical component used in that design, and provides valuable information to evaluate that larger project by conducting fish monitoring data at a control site located upstream of the nutrient addition zone. This report summarizes trends in relative abundance, size, and condition of fishes at the Montana control site since 2004. This report includes data collected at an additional control site located in the Montana portion of the river that was established collaboratively by Montana FWP and Idaho Fish and Game in 2015.

Chapter three summarizes work completed in 2018 at the Therriault Creek restoration project site. This work represents MFWP's continued commitment to the long-term success of the Therriault Creek Riparian Revegetation Project. The successful conversion of the riparian vegetation along the riparian zone of the Therriault Creek Project to a mosaic of native riparian shrubs and trees requires a multi-year, phased approach that includes maintenance and monitoring to adaptively manage ongoing restoration efforts. MFWP has completed three phases of riparian vegetation at this site (2007, 2009, and 2010), with nearly annual monitoring, evaluation and maintenance thereafter. Woody vegetation has increased at this site because of these efforts. The Therriault Creek Restoration Project was evaluated by Geum Environmental Consulting in 2018 to document existing site conditions, identify revegetation treatment maintenance needs, and determine weed control needs. These data were used to identify the factors limiting woody vegetation expansion at this site and the result was the development of a five-year riparian vegetation management plan. Three types of maintenance were also completed at this site in 2018 including the repair of the existing riparian protection fence, removal and relocation of a portion of the riparian protection fence and shrub browse protector maintenance.

MFWP conducts annual bull trout redd counts in eight critical bull trout tributaries located in the Montana portion of the Kootenai Basin, including Keeler, O'Brien, Quartz, Pipe, Bear, West Fisher, Grave creeks, and the Wigwam River. We use these data to assess bull trout trends in abundance and critical spawning and rearing habitat. The two adfluvial bull trout populations upstream of Libby Dam include Grave Creek and the Wigwam River and have both exhibited increasing trends over the period of record. The 2018 redd count in Grave Creek was equal to the ten-year average and the count in the Wigwam River was slightly lower than the ten-year average. The four fluvial bull trout populations that reside between Kootenai Falls and Libby Dam include the Quartz, Pipe, Bear and West Fisher creek populations. The Quartz and Bear Creek populations have exhibited a significant negative trend over the period of record. The 2018 redd count in Quartz Creek was 13 redds, and the lowest count on record. The red count in Bear Creek was about half of the ten year average. The Pipe and West Fisher creek counts have been variable with no evidence of a trend. However, the Pipe Creek count was slightly lower than the mean over the period or record, and the West Fisher count was about half of the mean over the period of record. Bull trout redd counts in three of the four tributaries located between Libby Dam and Kootenai Falls have declined to fewer than 10 redds, and collectively the populations within this geographical area have exhibited a declining trend since 1995. If these populations continue their current trajectory the persistence of these populations may be in jeopardy. O'Brien Creek is the only fluvial bull trout population in Montana downstream of Kootenai Falls. Bull trout redds have been variable with no evidence of a significant trend. The 2018 count in O'Brien Creek was 34 redds which is nearly identical to the ten year average. The adjunct bull trout population that spawns in Keeler Creek has significantly declined over the period of record, and the 2018 count was the lowest on record.

Beginning in 2011, MFWP initiated a multi-year study intended to estimate growth and survival of trout in four sections of the Kootenai River downstream of Libby Dam using mark-recapture methodologies. Fish were captured via nighttime electrofishing and marked with PIT tags. Survival estimates have not yet been calculated. Since the initiation of this study, we have measured annual growth of individual recaptured fish in four sections of the Kootenai River over years (2011 to 2018). Growth (length and weight) was relatively high for the three sections sampled during the fall since 2011. Growth rates (length and weight) sharply declined with fish size. Rainbow trout growth (length and weight) differed between several sections (within the same year) and between several years (within the same section) since 2011. A rigorous analysis of these data will allow comparisons of survival and growth between years and sites and lend valuable insight into the population dynamics of trout populations within these four sections of the Kootenai River. Researchers are hopeful that the observed annual variability in survival and growth of trout can be attributed analytically to important biological and environmental conditions that could be managed to improve both survival and growth for Kootenai River trout.

MFWP uses baited hoop traps during the winter months to monitor the trend and status of burbot in the Kootenai River below Libby Dam. During the 2017/2018 season we captured three burbot below Libby Dam after fishing a total of 228 trap days, for and average CPUE of 0.013 burbot per trap-day, which is 66% of the mean catch rate over the past ten years. Burbot catch rates have exhibited a significant negative trend since peak catch rates observed during the early 1990s. The mean catch rate over the past ten years has been low averaging only 0.020 fish per trap day, but has not exhibited a significant trend.

MFWP has used gillnets to monitor the trend and status of fishes in 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 10 fish species. Catch rates of kokanee in the fall gill nets in 2018 averaged 2.7 fish per net, which was lower than the ten-year average. Kokanee catch rates in the fall nets have been variable, with no apparent continuous trend in abundance, but biomass of kokanee per floating gill net has significantly decreased since 1988. The average length and weight of kokanee in 2018 was 288.2 mm and 235.3 g, respectively, which is higher than the overall and ten year averages. We observed a significant negative trend in length and weight of kokanee since their introduction into the reservoir. The catch rates of rainbow trout, cutthroat trout and mountain whitefish exhibited a significant negative trend after reservoir construction. However, catch rates of these three species during the past several years has remained low and not differed significantly from a stable population. Bull trout catch rates in Libby Reservoir generally exhibit three trends through time. From 1975 to 1989, bull trout catch rates were relatively low but stable, averaged 1.5 fish per net. Bull trout catch rates in Libby Reservoir began increasing in approximately 1990 and peaked in 2000. During the period 2005 to 2018, catch rates have been variable, but generally stable, averaging 4.2 fish per net. The mean catch rate we observed in 2018 (3.6 fish per net) was slightly lower than the mean catch rate since 2005. Burbot catch rates in spring sinking gillnets in Libby Reservoir exhibit two general trends since construction of the reservoir. Catch rates during the period 1975-1988 exhibit a significant increasing trend. However, during the period 1989-present, catch rates have exhibited a significant negative trend. We caught one burbot in 2018 for an average catch rate of 0.07 fish per net. Peamouth chub are the most abundant species captured in the sinking gill nets during the spring, with catch rates variable, but peamouth chub biomass has significantly increased since reservoir impoundment. Largescale suckers are generally the third most abundant species captured in the spring sinking nets and catch rates have significantly decreased since reservoir impoundment. However, biomass of largescale suckers has been variable and not exhibited a significant trend through time. During recent years, northern pikeminnow are the second most abundant species captured in the spring sinking nets and catch rates and biomass of northern pikeminnow have significantly increased since reservoir impoundment.

MFWP has monitored zooplankton species composition, abundance and size of zooplankton within the reservoir since the construction and filling of Libby Dam. Zooplankton species composition and abundance within Libby Reservoir has remained relatively stable during the past sixteen years. However, trends in zooplankton abundance in Libby Reservoir have generally been of decreasing abundance of the larger and more numerous genera of zooplankton and an increasing abundance of smaller zooplankton since the late 1970s. Overall total zooplankton densities in 2017 11.5 organisms per liter, which was the 10.3% lower than the average since 1997. *Cyclops* were the most abundant zooplankton in Libby Reservoir, followed by *Daphnia*, averaging 7.4 and 2.4 organisms/liter, respectively over the season. Mean annual densities of *Daphnia* were 22% higher than the mean since 1997. However, *Diaphanosoma, Leptodora*, *Epischura*, *Diaptomus, Bosmina, Clyclops, and Diaptomus* in 2017 were 92.9, 61.2, 45.8, 19.5, 14.1, and 8.5% lower than the respective mean values since 1997. The seasonal peaks in abundance we observed in 2017 were generally consistent with the season peaks observed since 1997. The seasonal peaks in abundance followed a similar trend to the overall seasonal abundance.

The primary objective for investigating primary production in Libby Reservoir is to assess the degree to which primary production may be limiting the growth and abundance of zooplankton and ultimately fish populations in the reservoir. We estimated primary productivity of Libby Reservoir in in three consecutive years (2016-2018) and it was higher at all stations and months compared to the other years for which productivity data exists 1986 and 1972-1980. Most of the primary production in 2016-2018 occurred in the top 15 meters of the water column. The contemporary estimates of primary productivity fall within the range of production that is defined as mesotrophic (250-1000 mg C/m^2/day). In 2018, the phytoplankton community was relatively evenly distributed between the three size classes we examined. Thirty percent of the observed productivity occurred in the size fraction that is most desirable for zooplankton consumption (2.0-20 um). The earlier studies did not size fraction the productivity estimates. The spring of 2018 was cooler than the previous two years of study which resulted in a delay in the runoff from the Canadian Rockies. This resulted in lower productivity in May of 2018 than either 2016 or 2017. The biggest difference in monthly productivity between 2016-2018 occurred in September 2018. The 2018 productivity was the highest measured in the three years of the study. We also observed considerable inter-annual variability between 2016-2018, suggesting that Libby Reservoir may be highly susceptible to changes in timing, and quantity of water entering the system along with the concomitant changes in nutrient loading.

The Ten Lakes Scenic Area is in northwest Montana and lies within the Wigwam River watershed. This mountainous region contains several lakes of which the historic fish distribution is largely unknown. Genetic samples were previously collected from lakes within this area to assess the genetic constituency. Paradise, Upper Wolverine, and No Name Wigwam lakes were determined to contain non-introgressed westslope cutthroat trout. Hybridization with rainbow trout was detected in Bluebird, Lower Wolverine, and Little Therriault lakes at relatively low levels (< 0.3%) indicating that the stocking records are likely incomplete. Rainbow, Weasel, and Big Therriault lakes contained hybridized fish westslope and Yellowstone cutthroat trout. No Name Wolverine lake and samples from the Montana portion of the Wigwam River contained hybridized fish between all three species. MFWP collected additional samples from the three major tributaries downstream of these lakes in 2017 and 2018 to assess if these hybridized populations have influenced the genetic integrity of native westslope cutthroat trout populations located downstream of these lakes. These samples have been sent to the University of Montana genetics laboratory for analyses. These data will be used to develop a management strategy for these waters with the goal of expanding the distribution of westslope cutthroat trout.

The Cabinet Mountains lie south of the Purcell Mountains, between the Kootenai River and Clark Fork River and Idaho's Lake Pend Oreille. Several mountain lakes exist within the wilderness area. It is unknown what species of trout, if any, historically inhabited many of these lakes. Huston et al. (1996) completed genetic surveys of lakes within the Cabinet Mountains Wilderness area to explore potential inland rainbow trout restoration opportunities but did not find any non-introgressed populations of redband trout in any of these waters and concluded that genetic analysis results did little to determine the original range of redband trout in Cabinet Mountain lakes. Therefore, we collected additional genetic sampling of several of the outlet tributaries to many of these lakes to assist with the range assessment. In 2017, MFWP collected genetic samples from nine tributaries to determine if redband trout were historically present in these watersheds. MFWP collaborated with the University of Montana (UM) genetics laboratory to develop genomic resources techniques to assess non-native admixture and population structure in Kootenai drainage redband rainbow trout. The UM laboratory developed a Rapture assay to assess admixture and population structure in Kootenai redband rainbow trout that involved conducting RADSeq on 134 redband rainbow trout from across Kootenai drainage and British Columbia, in addition to 90 westslope cutthroat trout sampled from across their range, and 54 hatchery-origin coastal rainbow trout. Analysis of these data resulted more than 354,000 identified SNPs that were later reduced to the 18,000 most informative SNPs that were most highly differentiated between taxa. The UM laboratory contracted a private lab to quality control test these probes against the rainbow trout reference genome, and to manufacture the sequence capture baits. The final effort identified the 10,000 most reliable and useful capture probes for the Rapture assay. The samples collected in 2017 have been prepared for swift processing that will be conducted at the UM lab. The DNA libraries will be sequenced at a commercial facility and the UM staff will complete the data analysis by the middle of 2019, and the results will be summarized in a subsequent report.

MFWP initiated an applied research project in 2018 to test the efficacy of using strontium isotope analysis of resident fish otoliths collected from Libby Reservoir to identify natal areas of origin. This effort will occur in two-stages. The first stage of this project (2018-2019) collected water samples and young of the year fish from all major Kootenai River tributaries upstream of Libby Dam (Montana and British Columbia) and from within Libby Reservoir to determine if spatial differentiation of water chemistry exists within the study area. The preliminary results of that work are promising and justify moving forward with analysis of the reference fish collected. The results of which are expected to be completed by July. The second phase of the effort which will apply the methodology to adult burbot collected from Libby Reservoir to predict natal origin. The final project is expected to be completed by January 2020.

Acknowledgements

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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 of 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 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 were later refined (Marotz et al 1999). Initiation of mitigation projects such as lake rehabilitation and stream restoration began in 1996. The current primary focus of the Libby Mitigation project is to restore the fisheries and fish habitat in basin streams and lakes, and continue to refine hydro operations to optimize survival and growth of mainstream Kootenai River fishes.

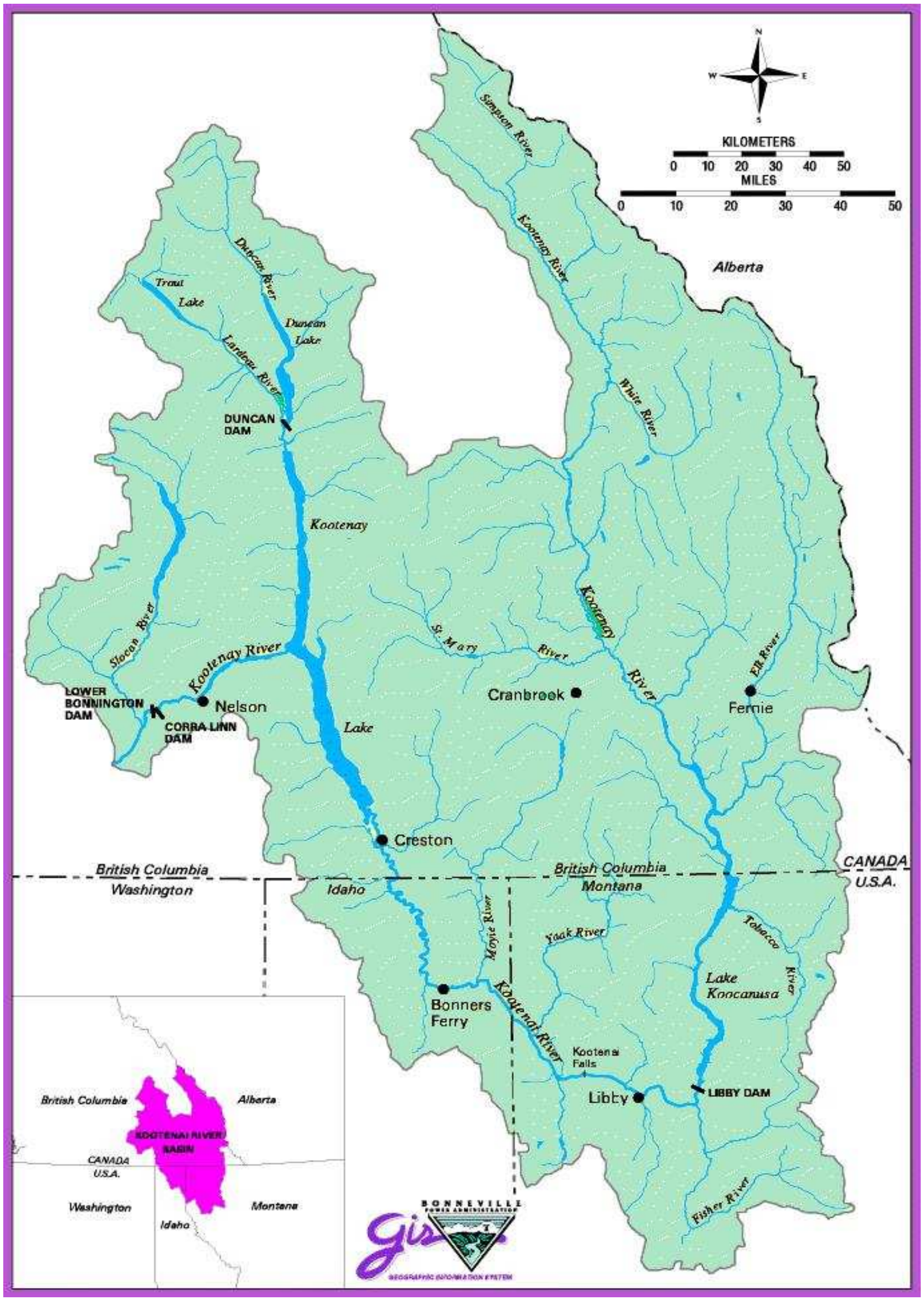


Figure 1. Kootenai River Basin (Montana, Idaho and British Columbia, Canada).

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 1999a) and later refined in the USFWS 2000 biological opinion.

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 and compared to past results in the 1980s Perry and Huston 1983) and 1990s (Hauer et al. 1997). The Hauer et al. (1997) 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 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) 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 Therriault 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 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 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.

2007 - MFWP completed Phase I of the Therriault Creek Project Revegetation effort.

2007 - MFWP chemically rehabilitated Loon Lake to remove nonnative brook trout and black bullheads and reestablished westslope cutthroat trout in the lake.

2008 - MFWP completed Phase II of the Therriault Creek Project Revegetation effort.

2008 - MFWP installed a fish screen on an irrigation diversion on Young Creek.

2008 - MFWP collaborated on the Grave Creek Phase I Project revegetation effort.

2009 – MFWP conducted a creel survey to estimate recreational angler effort, catch and harvest of trout in the Kootenai River downstream of Libby Dam.

2009 – MFWP completed additional revegetation efforts, maintenance and monitoring on the Therriault Creek Restoration Project.

2009 – MFWP chemically removed non-native trout from Boulder Lake and Boulder Creek.

2010 – MFWP collaborated with the single largest water user on Deep Creek to install a fish screen on an irrigation diversion (Phase I).

2010 – MFWP conducted fish monitoring on the Kootenai River immediately downstream of Libby Dam to evaluate the fisheries response to elevated total dissolved gas during a spill operation.

2011 – MFWP completed Phase I of the Pipe Creek Restoration Project.

2011 – MFWP completed monitoring, maintenance and additional revegetation work on the Therriault Creek Restoration Project.

2011 – MFWP collaborated with project partners to complete monitoring, maintenance and revegetation work on 3 phases of stream restoration work previously completed on lower Grave Creek.

2012 – MFWP collaborated with the US Army Corps of Engineers to developing a management technique to limit the distribution and abundance of a nuisance diatom algae (*Didymosphenia geminata*) on the Kootenai River.

2012 – MFWP collaborated with project partners to complete monitoring, maintenance and revegetation work on 3 phases of stream restoration work previously completed on lower Grave Creek.

2012 – MFWP collaborated with the US Forest Service and Plum Creek Timber Company to develop a conceptual plan to restore Dunn Creek which was historically an important rainbow trout spawning tributary to the Kootenai River.

2013 – MFWP collaborated with the single largest water user on Deep Creek to install a fish screen on an irrigation diversion (Phase II).

2014 – MFWP collaborated with the USFWS, and the local landowner to complete maintenance activities to the Grave Creek Demonstration and Phase I project areas.

2015 – MFWP collaborated in the development of a watershed restoration plan for the Kootenai River Basin, which will be useful to guide restoration activities within the Montana portion of the Kootenai Basin for the ultimate improvement of water quality within impairment TMDL listed and non-listed streams alike.

2015 – MFWP collaborated with the University of Idaho to research and develop a control measure to reduce the distribution and abundance of the nuisance diatom *Didymosphenia geminata*.

2016 – MFWP collaborated with the US Forest Service and Weyerhaeuser Company to stabilize 241 feet of eroding stream bank on Dunn Creek.

2017 – MFWP collaborated with the landowner to replace a non-functional fish screen on an irrigation diversion on Grave 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 Subbasin Plan (KTOI and MFWP 2004, see NPCC web page). This project creates and restores degraded trout habitat to functional condition through stream restoration and fish passage repairs. The projects complement 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 Aquaculture Program (198806400 – Kootenai Tribe of Idaho). The project biologist 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.

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.

MFWP has a lengthy history of cooperation with the efforts of the bull trout recovery projects in the Kootenai Watershed where we have monitored 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 the Study Area

Subbasin Description

The Kootenai River Subbasin is an international watershed that encompasses parts of

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

Drainage Area

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

Hydrology

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

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

Major tributaries to the Kootenai River below Libby Dam include the Fisher River (838 sq. mi.; 485 average CFS), the Yaak River (766 sq. mi. and 888 average CFS) and the Moyie River (755 sq. mi.; 698 average CFS). Kootenai River tributaries are characteristically high-gradient mountain streams with bed material consisting of various mixtures of sand, gravel, rubble, boulders, and drifting amounts of clay and silt, predominantly of glacio-lacustrine origin. Fine materials, due to their instability during periods of high stream discharge, are continually abraded and redeposit 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 m^3/s (30,650 CFS).

Fish Species

Eighteen species of fish are present in Libby Reservoir and the Kootenai River (Table 1). The reservoir currently supports an important fishery for kokanee *Oncorhynchus nerka* and rainbow trout *Oncorhynchus mykiss*, with annual fishing pressure over 500,000 hours (Chisholm and Hamlin 1987). Burbot *Lota lota* are also important game fish, providing a popular fishery during winter and spring. The Kootenai River below Libby Dam is a "blue ribbon" trout fishery, and the state record rainbow trout was harvested there in 1997 (over 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 present (I=increasing, S = stable , D = decreasing, U = unknown) of fish species present in the Montana portion of the Kootenai River Basin.

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

* 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.

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 and often go unnoticed until spring. In addition, widely fluctuating flows may provide false migration cues to burbot and white sturgeon spawners (Paragamian 2000; Paragamian and Kruse 2001).

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)	11,055 CFS

Barriers have also 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. 1998). 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. 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 27,000 CFS). 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 tailwater.

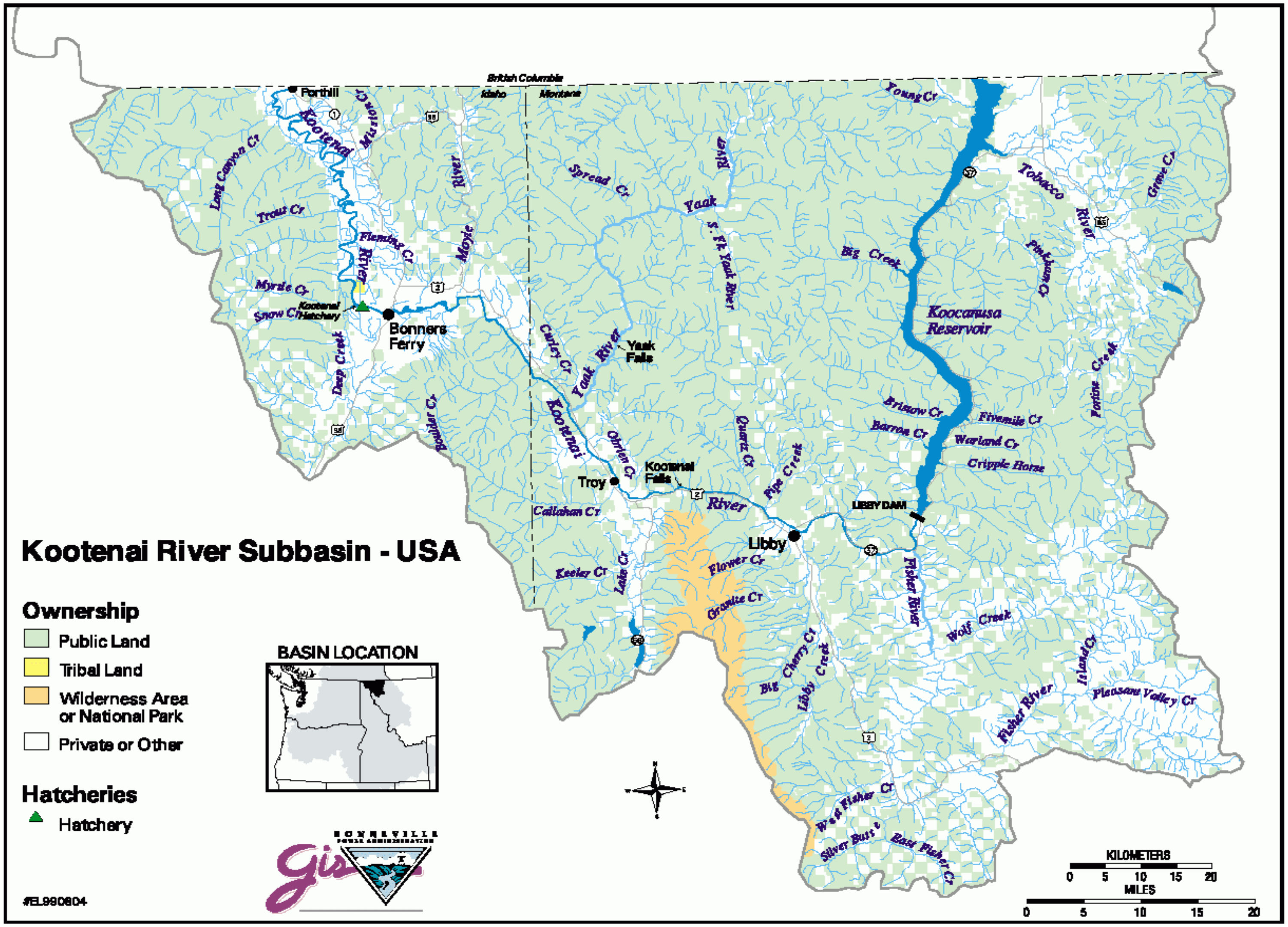


Figure 2. Kootenai River Basin, Montana.

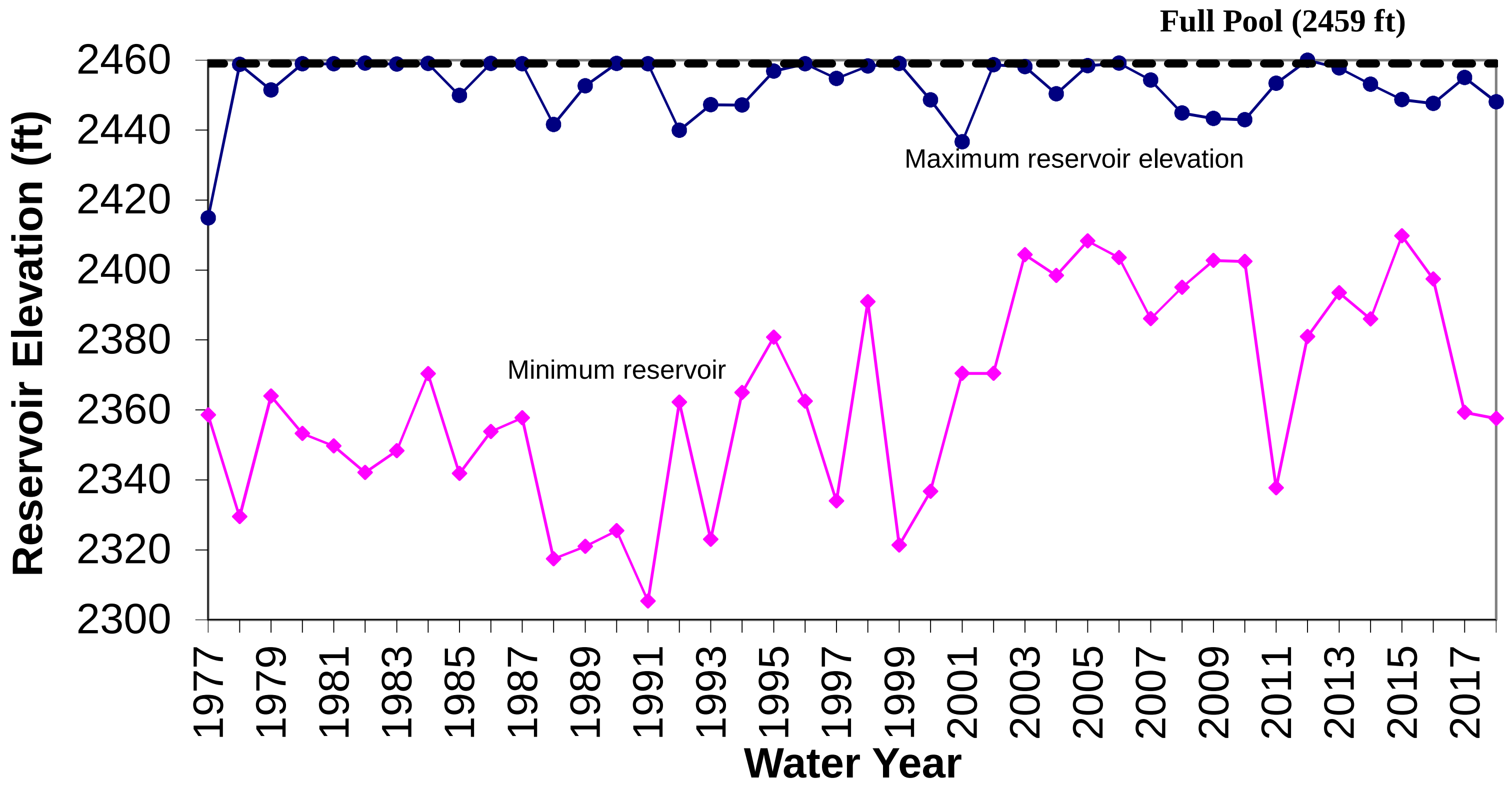


Figure 3. Libby Reservoir elevations (minimum, maximum), water years (October 1 – Sept. 30), 1976 through 2018.

Chapter 1: Mitigation Project Monitoring and Evaluation

This chapter includes the following work elements:

E: Monitor and Evaluate Mitigation Projects for Effectiveness (Contracts 77012 and 76916) and

J: Analyze and interpret Libby Mitigation physical and biologic data (Contracts 77012 and 76916).

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 et al. 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 et al. 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 et al. (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. 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. This chapter describes the physical and biological effectiveness monitoring of several stream restoration projects intended to mitigate for losses attributable to the operation and construction of Libby Dam.

Young Creek

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.

MFWP completed the Young Creek State Lands Restoration Project in 2003 (see Dunnigan et al. 2004). The project significantly changed the dimension, pattern and longitudinal profile of this section of Young Creek (Dunnigan et al. 2004). The stream restoration project significantly 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 (Dunnigan et al. 2004). The annual monitoring activities described within this report are intended to determine if the physical changes made to instream habitat have been maintained since 2003 and evaluate the fishery response to those changes.

Therriault Creek

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 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 restored 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. MFWP completed a riparian vegetation plan in 2007 (Geum Environmental 2007a) and began implementing that plan to restore a functioning riparian community at this site. The results of continuing revegetation and effectiveness monitoring efforts are described in a subsequent chapter of this report (see below). The annual monitoring activities described within this chapter are intended to determine if the physical changes made to instream habitat have been maintained since 2004 and evaluate the fishery response to those changes.

Libby Creek

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 several 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. Much of the watershed MFWP targeted for restoration was previously characterized as 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 focused on the upper watershed (Sato 2000).

Libby Creek Upper Cleveland Project

Past land management activities including logging, mining, riparian road construction, and stream channel manipulation have resulted in accelerated bank erosion along several 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. 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). 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; 2009) 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. 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 at Hammer Cutoff (river mile 8.5) 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 report evaluates changes in the physical habitat within this section of Libby Creek after this event by comparing current conditions to those that existed before restoration (1999) and after the November 2006 flow event, in order to evaluate if changes made during the restoration are sustained after the flow event. Annual fish abundance data was also collected to evaluate the fish community response to both the channel restoration activities and the 2006 flood event. A longitudinal profile survey of the stream channel thalweg was also surveyed during years which data was collected.

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 was previously 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 several 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 total length of Libby Creek through the entire lower Cleveland property prior to restoration efforts was approximately 9,100 feet. MFWP developed a restoration strategy that was intended to be implemented in three phases. The first phase was implemented in October 2005, and is referred to as the Libby Creek Lower Cleveland Phase I Project (approximate river mile 20-21). The restoration work 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, which resulted in a significantly narrower and deeper channel with increased habitat complexity. 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, 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. 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 storm event changed the stream plan form, and channel dimensions (Dunnigan et al. 2009). Therefore, this report compares current habitat conditions to those prior to restoration and after the 2006 flow event and evaluates how the fisheries community has changed in response to these physical changes.

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 within the project area including logging, mining, riparian road construction, and stream channel manipulation have resulted in accelerated bank erosion along several 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 3,273 feet downstream (see above). This project constructed in later summer 2006 and it installed a variety of structures intended to improve fish habitat and increase bank stability (Dunnigan et al. 2007). The overall result of the restoration work was an increase in channel sinuosity, depth and habitat complexity and a reduction in stream width (Dunnigan et al. 2007).

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. 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 storm event changed the stream planform, and channel dimensions (Dunnigan et al. 2009). Therefore, this report compares current habitat conditions to those prior to restoration and after the 2006 flow event. Fisheries abundance data was not collected at this site.

Methods

Protocol Title: MFWP Stream restoration effectiveness monitoring v1.0

Protocol Link: http://www.monitoringmethods.org/Protocol/Details/635

Protocol Summary: This protocol was developed to evaluate the physical and biological responses of stream restoration actions in our project watersheds. Fisheries response variables monitored include: fish species composition, length structure, and abundance. Physical response variables include various stream channel dimensions for both riffle and pool habitats. Vegetative response variables include plant survival, growth metrics, weed abundance/distribution, erosion, and natural vegetation recruitment. Information needed to complete a Rosgen (1996) Channel classification is also included in this protocol.

Estimates of Fish Abundance

MFWP conducted juvenile salmonid population estimates on Grave, Young, Therriault, and Libby, creeks annually, as part of an effort to evaluate fish community response to the restoration efforts on those streams described above. We conducted salmonid population estimates on each stream with mobile electrofishing gear using DC current for multiple pass depletions similar to Shepard et al. (1984). 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 $\geq$ 0.7, a third pass was conducted. Population estimates were performed for fish $\geq$ 75 mm, consistency with historic data collected prior to 1997, which generally represented age 1 and older fish. Population estimates and associated 95% confidence intervals were estimated using multiple pass depletion abundance estimates (Van Deventer and Platts 1983) using *FA plus*, a proprietary software package developed by Montana FWP. We evaluated trends in abundance using multiple regression. We compared fish abundance at sites where we performed stream restoration efforts using a two-sided student's t-test to evaluate differences in abundance before and after restoration was completed. We also evaluated the response of fish abundance and mean length to restoration efforts using the more powerful Before/After/Control (BACI; Underwood 1994) design on Young, Therriault, and Libby creeks, where we had previously established control sections in addition to the treatment sections using a General Linear Mixed Model approach. The response variable was either fish abundance or mean total length. Where site class (control or impact), period (before or after) were fixed effects, site and year were random effects, and the interaction term (site class*period) represents the BACI contrast of interest. All statistical analyzes were performed using R (R Core Team 2018). The BACI design tests the difference of differences using the following formula:

$$BACI = MME_{CA} - MME_{CB} - (MME_{IA} - MME_{IB})$$

Where; MME = estimated marginal mean effect (sometimes also referred to as Least Squares Means); and

CA = control after, CB = control before, IA = impact after, and IB = impact before.

A description of the reaches (sections) sampled within each tributary is presented below.

Young Creek

MFWP previously established five monitoring sections in Young Creek to assess trends in juvenile salmonid abundance within the Young Creek watershed (Huston et al. 1984). However, MFWP has curtailed monitoring to include only three sections; including the following:

Section 1: Tooley Lake Section. This section is located 0.65 miles upstream of Libby Reservoir (at full pool), 2.73 miles downstream of the Young Creek State Lands Restoration Project, and is intended to serve as a control site for the restoration work completed at that project area.

Section 4: Dodge Creek Road #303. This section is located 2.42 miles upstream from Young Creek State Lands Restoration Project, 5.8 miles upstream of Libby Reservoir (at full pool), and is intended to serve as a control site for the restoration work completed at the Young Creek State Lands Restoration Project area.

Section 5: Young Creek State Lands Project. This section is located within the upper portion of Lands Restoration Project, and is 3.38 miles upstream of Libby Reservoir (at full pool).

We evaluated the impact of a large wildfire that burned most of the upper Young Creek watershed in 2017 by calculating mean cutthroat and brook trout abundance for the ten years prior to the burn (2008-2017) for comparison to the first year after the burn (2018) for each of the three sections on Young Creek. We also calculated cutthroat trout abundance estimates by 25 mm size groups for Section 5 during the same periods to attempt to determine which size/age fish were most impacted by the fire.

Therriault Creek

MFWP established three monitoring sections in Therriault Creek, including the following:

Section 1: Highway 93. The lower boundary of this section begins at the Highway 93 culvert and extends 82 m upstream, and is located 0.61 miles downstream of the lower project boundary of the Therriault Creek Restoration Project.
Section 2: Therriault Restoration Project. The upstream boundary of this section begins at the upper end of the Therriault Creek Restoration Project and proceeds downstream into the project area. This section is located approximately 3.4 miles upstream from the Therriault Creek confluence.
Section 3: Therriault Above Project. This section is located 0.23 miles upstream of the upper boundary of the restoration project, and serves as a second control site for the restoration project.

Libby Creek

MFWP collected fish population estimates at six sites on Libby Creek, including the following:

Section 3: Upper Cleveland Project. This section is the upper most section in the Libby Creek watershed sampled, and is located at approximately river mile 22.3.

Section 4: Below Lower Cleveland. This section is a 201-m long reach located downstream of the lower Cleveland Project area, is intended to serve as a control site for the lower Cleveland Stream Restoration Project, and is located at approximately river mile 19.7.

Section 5: Above Lower Cleveland. This section is a 143-m long reach located upstream of the lower Cleveland property. The bridge on Forest Rd. number 231 bisects this section, which is also intended to serve as a control site for the lower Cleveland Stream Restoration Project. This section is located at approximately river mile 20.5.

Section 6: Lower Cleveland Phase II Project. This section is a 172-m long reach near the confluence of Midas Creek located within the lower Cleveland Phase II Stream Restoration Project, and is located at approximately river mile 20.2.

Physical Monitoring

Montana FWP monitors stream channel dimensions within restoration project areas before and after project completion. We use these data to evaluate how the original restoration work physically changed the channel dimensions and determine if those changes are sustained through time. Our methods have been generally refined through time, and therefore differ slightly between specific restoration projects. Due to the importance of pool habitat to rearing native salmonids within each of the project areas, we devoted a substantial effort to monitor pool habitat. We also monitor stream channel dimension within riffle habitats. Below is a description of the methods used within each project area. Habitat data was collected generally annually. The data presented within this report is the most current data available. However, often more recent data has been collected, but due to limited time between data collection and report preparation was not available and will therefore be presented in a subsequent annual report.

Young Creek

Pool data within the Young Creek project area was collected prior to (2002), immediately after (as-built; 2003), and annually since the work was completed. We annually measured the mean width, depth, and maximum depth of all pools within the project area at the longitudinal mid-point of each pool. Maximum pool depth was measured along that transect and did not necessarily correspond to the overall maximum depth for each pool. The preceding measurements were all based on bankfull depth (Rosgen 1996). We also measured the total length of each pool. We calculated total surface area of each pool by multiplying the mean bankfull width by the length. Mean pool volume was calculated by multiplying pool surface area by mean bankfull depth. We did not perform a statistical comparison for these data because the pool data described above represented all pools within the project area (i.e. complete census), making statistical tests unnecessary.

Riffle data within the Young Creek project area was collected on all riffles within the project area prior to (2002), immediately after (as-built; 2003), and annually since the work was completed. 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 width to depth ratio. We did not perform a statistical comparison for these data because the riffle data described above represented all riffles within the project area (i.e. complete census), making statistical tests unnecessary.

Therriault Creek

Prior to project construction (2003), we surveyed 10 riffles and 10 pools within the existing stream channel to characterize stream channel dimensions within these habitats. Within the riffle/run habitats, we established each transect at the longitudinal mid-point of the first 10 riffles/runs 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 (as-built; 2004), 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. We established permanent cross sections in five riffles and five pools throughout each of the two reaches and measured channel dimensions in each of the habitats annually. We established pool cross sections at the location of maximum depth within each pool.

We annually measured the stream channel mean width, depth, and maximum depth, cross sectional area and width to depth ratio of all pools and riffles within the project area. Maximum pool depth was measured along that transect and did not necessarily correspond to the overall maximum depth for each pool. All preceding measurements were all based on bankfull depth (Rosgen 1996). We performed an analysis of variance (ANOVA), and subsequent multiple comparison test (Tukey Test; Zar 1996) for significant differences between years.

Libby Creek Upper Cleveland

Pool data within the Libby Creek Upper Cleveland project area was collected prior to (1999), immediately after (as-built; 2002), and annually since the work was completed except for 2004 and 2010. We annually measured the mean width, depth, and maximum depth of all pools within the project area at the longitudinal mid-point of each pool. However, maximum pool depth was measured at the point of maximum depth within the pool, and therefore did not necessarily correspond with a point on midpoint transect. The preceding measurements were all based on bankfull depth (Rosgen 1996). We also measured the total length of each pool each year and pool spacing (distance between pools) all years except 2002 (as-built). We calculated total surface area of each pool by multiplying the mean bankfull width by the length. Mean pool volume was calculated by multiplying pool surface area by mean bankfull depth. We did not perform a statistical comparison for these data because the pool data described above represented all pools within the project area (i.e. complete census), making statistical tests unnecessary.

We established six cross-sections in 1999 to characterize riffle dimensions prior to the project. However, we were unable to use these after the restoration project was completed because the planform (location) of the stream changed after project construction. Therefore, in 2002 (as-built) we began measuring the dimensions of all riffles within the project area annually (except 2004 and 2010). Beginning in 2002, 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 width to depth ratio. We calculated the riffle slope for each riffle by dividing the horizontal drop of the water surface by the length of each riffle. Riffle slope was not estimated in 1999. Since the data collected in 1999 was a sample of the riffle habitats within the project area, we performed an analysis of variance (ANOVA), and subsequent multiple comparison test (Tukey Test; Zar 1996) for significant differences between mean width, depth, cross sectional area, width to depth ratio, and maximum depth in 1999 and all other years. We also surveyed a longitudinal profile of the stream channel thalweg throughout the project area in all years we completed physical surveys within the project area.

Libby Creek Lower Cleveland Phase I

Pool data within the Libby Creek Lower Cleveland Phases I and II project areas was collected prior to (2004), immediately after (as-built; 2005 and 2006, respectively), and annually since the work was completed except for 2010. We annually measured the mean width, depth, and maximum depth of all pools within the project area at the longitudinal mid-point of each pool. Maximum pool depth was measured at the point of maximum depth within the pool, and therefore did not necessarily correspond with a point on that transect. The preceding measurements were all based on bankfull depth (Rosgen 1996). We also measured the total length of each pool and pool spacing each year surveys were completed. We calculated total surface area of each pool by multiplying the mean bankfull width by the length. Mean pool volume was calculated by multiplying pool surface area by mean bankfull depth. We did not perform a statistical comparison for these data because the pool data described above represented all pools within the project area (i.e. complete census), making statistical tests unnecessary.

Riffle data within the Libby Creek Lower Cleveland Phases I project area was collected on all riffles within the project area prior to (2004), immediately after (as-built; 2005 and 2006, respectively), and annually since the work was completed except for 2010. 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 width to depth ratio. We calculated the riffle slope for each riffle by dividing the horizontal drop of the water surface by the length of each riffle. Riffle slope was not estimated within the Phase I project area in 2004. We did not perform a statistical comparison for these data because the riffle data described above represented all riffles within the project area (i.e. complete census), making statistical tests unnecessary. We also surveyed a longitudinal profile of the stream channel thalweg throughout the two project areas in all years we completed physical surveys.

Results

Estimates of Fish Abundance

Young Creek

The Young Creek Section 1 juvenile monitoring site was sampled consecutively since 1997, except for 2000 and 2003, and is intended to serve as a control section for the restoration project area (Section 5). There was no evidence of linear trends in abundance for westslope cutthroat or brook trout from 1997-2018 ($p = 0.97$ and 0.15, respectively; Figure 1-1). Westslope cutthroat trout were the most abundant fish species at this site in 2018 (50.6 fish per 1,000 feet; Figure 1-1), which was about 17% lower than the mean cutthroat trout abundance in this section since 1997(60.7 fish per 1,000 feet). Brook trout were the second most abundant species observed at this site in 2018, with an estimated 15.6 brook trout per 1,000 feet, which was about 65% lower than the overall mean for the period of record (mean = 44.3 fish per 1,000 feet). Rainbow trout within this section have exhibited a significant decrease in abundance since 1997 ($r^2 = 0.25$; $p = 0.02$), decreasing on average by about 1.1 fish per 1,000 feet per year. We did not observe any rainbow trout at this section in 2018. Bull trout were first observed at Section 1 in 2004, and have been observed annually except 2007 and 2012. We estimated 7.8 bull trout per 1,000 feet within this section of Young Creek in 2018 (Figure 1-1.). Bull trout abundance has exhibited a weak, but significant increasing trend through time ($r^2 = 0.21$; $p = 0.037$; Figure 1-1). We assume that the juvenile bull trout present at this site immigrated from the reservoir since no bull trout spawning is known to occur in Young Creek.

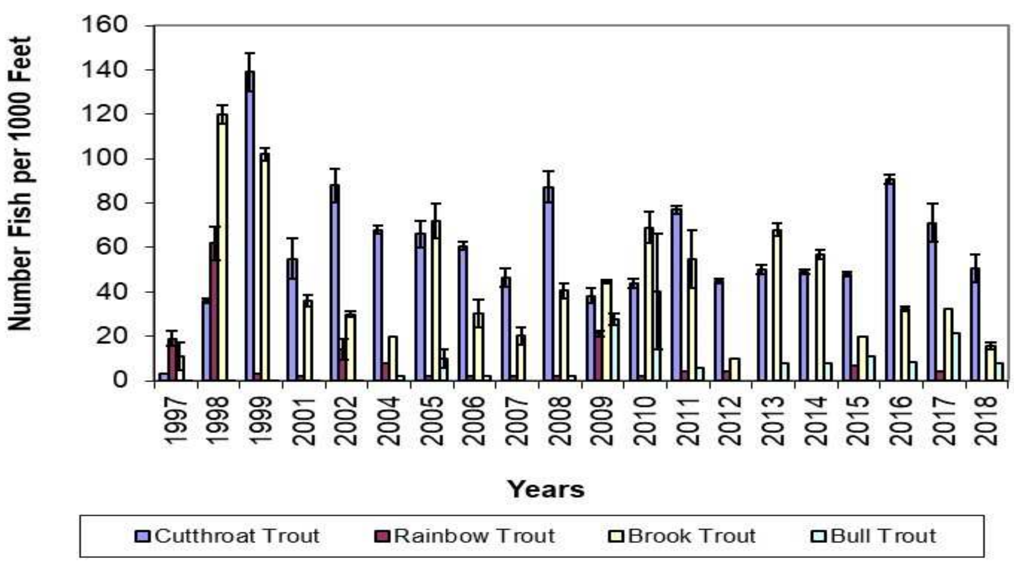


Figure 1-1. Cutthroat, rainbow, brook and bull trout densities (fish per 1,000 feet) within the Young Creek Section 1 monitoring site from 1997-2018, except for 2003. Data was collected by backpack electrofishing. Error bars represent 95% confidence intervals.

A wildfire burned much of the Young Creek watershed in 2017. The USFS estimated that the fire began burning within the upper watershed on August 24, 2017, with fire intensity peaking on September 2-3. The fire burned until about September 20, 2017. The fire burned intensively within the upper watershed, including Section 4 (Figure 1-2), but stopped short of Section 5. The Young Creek Section 4 juvenile monitoring site was sampled consecutively since 1996, except 2000 and 2003 (Figure 1-3; Appendix Table A2). Westslope cutthroat trout dominated the fish community at this sampling location during all years, including 2018, when we observed an estimated 126.1 fish per 1,000 feet, which was 54% lower than the annual average of 275.5 fish per 1,000 feet. We were not able to distinguish a significant trend (r^2 = 0.04; p = 0.38). However, brook trout abundance at this site has significantly increased over time (r^2 = 0.59; p = $5.0*10^{-5}$), despite the decrease after the wildfire that reduced abundance to 15.5 fish per 1,000 feet in 2018. Brook trout are increasing an average of about 2.5 fish per 1,000 feet per year at this site since 1996. We have only observed a single bull trout at this site in 2007 (Figure 1-2).

Figure 1-2. Photograph of Section 4 on Young Creek taken during the spring of 2018 at the downstream end of the section looking upstream. Photograph courtesy Pat Price (USFS).

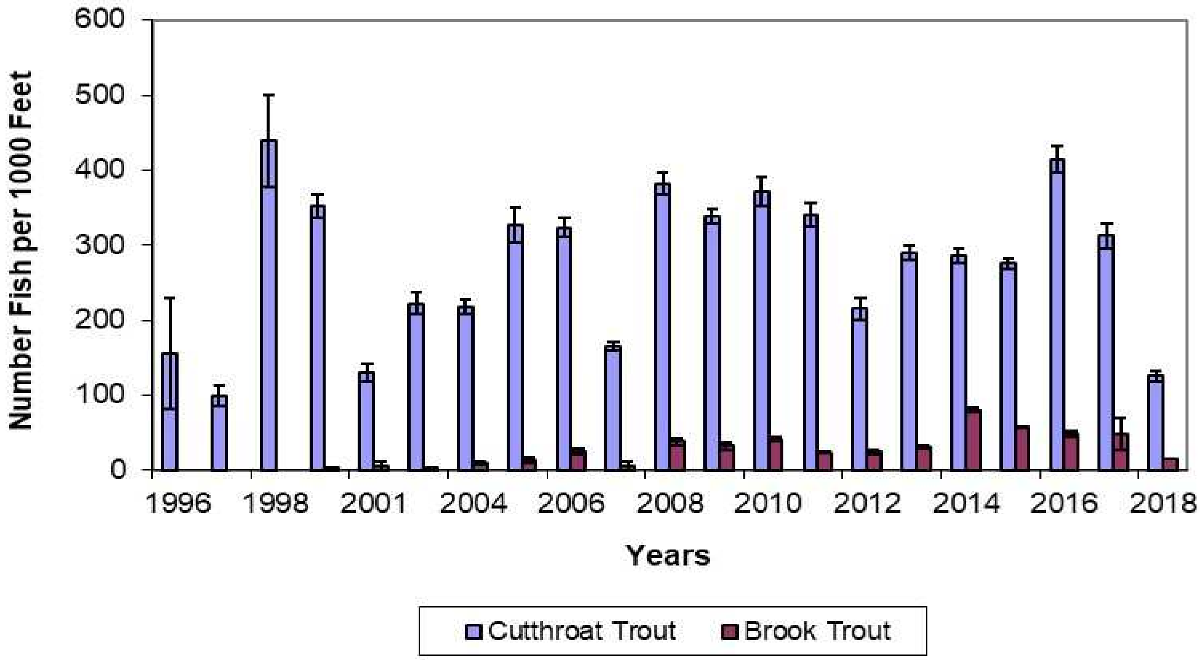


Figure 1-3. Cutthroat trout and brook trout densities (fish per 1,000 feet) within the Young Creek Section 4 monitoring site from 1996-2018, except for 2000 and 2003. Data was collected by backpack electrofishing. Error bars represent 95% confidence intervals.

The stream restoration activities at the Young Creek Section 5 (Project Area) were completed in 2003, and therefore fish abundance estimates up to and including 2003 represents pre-restoration data gathered prior to the restoration project completion. Cutthroat and brook trout have been the first and second most abundant species at this site since we began our annual sampling in 1998, except in 2011 where brook trout were the most abundant species at this site. In 2018, we observed an estimated 135.9 cutthroat trout per 1,000 feet, at this site (Figure 1-4), which was 22% lower than the mean since 1998 (175.2 fish per 1,000 feet). Cutthroat trout abundance at this site has not exhibited a significant trend since 1998 ($r^2 = 0.01$; $p = 0.66$). We observed an estimated 90.6 brook trout per 1,000 feet within this section in 2018, which represented about a 20% decrease from 2017 (figure 1-4). Despite the decrease in 2018, brook trout within this section have demonstrated a significant increasing trend ($r^2 = 0.46$; $p = 0.0007$), increasing on average by 5.1 fish per 1,000 feet per year. We observed and estimated 24.3 bull trout per 1,000 feet within this section in 2018, which was the highest abundance observed. Bull trout abundance has exhibited a weak, but significant increase in abundance since 1998 ($r^2 = 0.16$; $p = 0.044$), increasing on average by 0.4 fish per 1,000 feet per year.

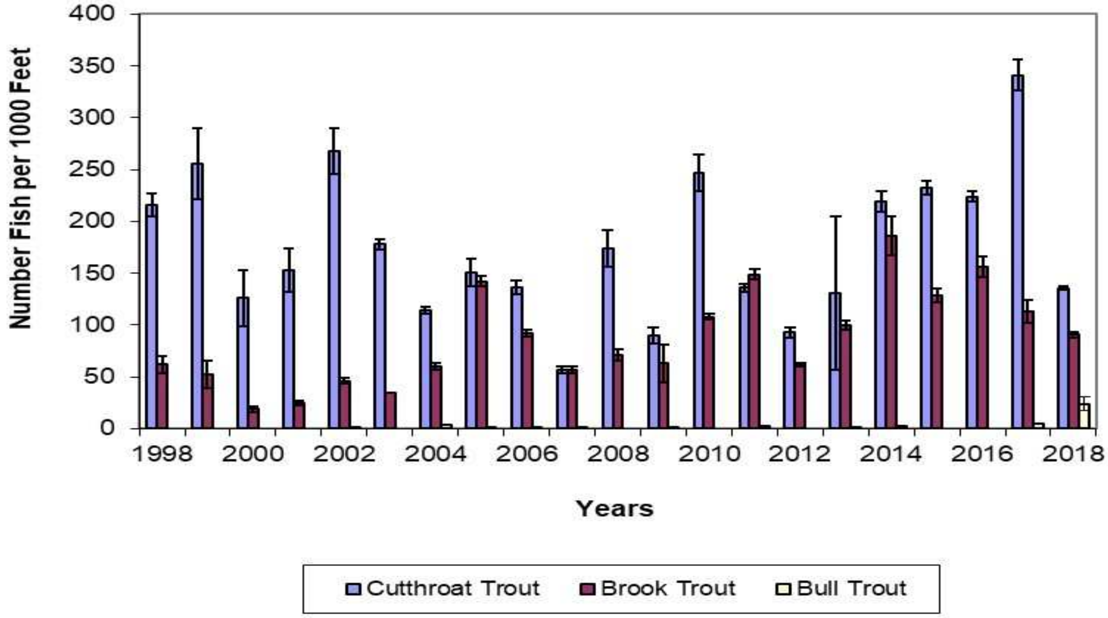


Figure 1-4. Cutthroat, brook and bull trout densities (fish per 1,000 feet) within the Young Creek Section 5 monitoring site from 1997-2018 collected by backpack electrofishing. The data presented for 2004-2016 represent post restoration data. The error bars represent 95% confidence intervals.

Abundance estimates for cutthroat trout within the restoration area (Section 5) have not differed significantly before and after restoration efforts at this site (p = 0.33, for a 2-tailed test; Figure 1-5). However, the abundance of brook trout significantly increased from a mean of 39.8 fish per 1,000 feet before the project to 105.4 fish per 1,000 feet after the project (p = 0.001, for a 2-tailed test; Figure 1-5). Despite a nearly tenfold increase in bull trout abundance after the project, increasing from a mean of 0.3 to 3.1 bull trout per 1,000 feet the increase was not significant (p = 0.28, for a 2-tailed test; Figure 1-5). Total trout abundance (excluding bull trout) after the project was completed, increased from a mean of 234.8 to 275.6 trout per 1,000 feet after the project, which represented about a 17% nonsignificant increase (p = 0.39, for a 2-tailed test; Figure 1-5).

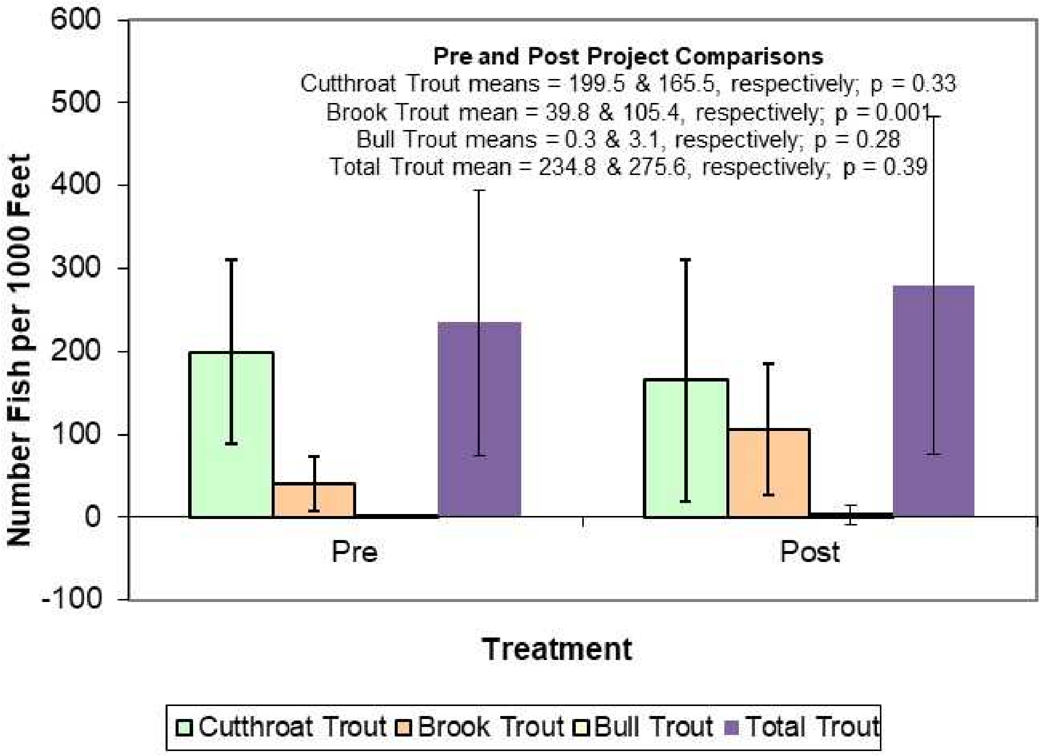


Figure 1-5. Cutthroat, brook, bull and total trout (excluding bull trout) densities (fish per 1,000 feet) within the Young Creek Section 5 (State Lands Restoration Project Area), comparing annual mean pre-project (1998-2003) data and post-project (2004-2018) using mobile electrofishing gear. Comparisons were made using a 2-tailed t-test. Error bars represent 95% confidence intervals.

We compared the abundance of cutthroat, brook, bull and all trout within the restoration project area (Section 5) to control sections located below (Section 1) and above (Section 4) the restoration project, using the Before/After/Control/Impact (BACI) statistical design (Table 1-1). The BACI (differences of the differences) contrast for cutthroat trout showed a significant ($p = 0.015$) decrease in cutthroat trout abundance which was consistent with the before/after (t-test) comparison. The brook trout BACI contrast ($p = 0.001$) was also consistent with the before/after comparison, suggesting that brook trout abundance significantly increased at the restoration site. The BACI contrast for bull trout also suggested that the abundance decreased within the project area after construction, but the test was not statistically significant ($p = 0.598$). This was contrary to the before/after (t-test) comparison, suggesting that the latter was attributable to factors not related to the restoration activities. Lastly, the BACI contrast for all trout species indicated a slight and non-significant increase in total trout abundance at the restoration site after construction ($p = 0.828$; Table 1-1). This result may have been influenced by the fact that cutthroat trout within Section 5 increased sharply in 2017 compared to the two control sections and the increasing trend in brook trout abundance observed within Section 5 over time.

Table 1-1. Results from the before/after/control/impact (BACI) analysis of fish abundance within the Young Creek restoration project area (impact) and two control sections on Young Creek.

Species	**BACI Contrast (fish/1,000 ft)**	**P-Value**
Cutthroat Trout	117.7	0.015
Brook Trout	-58.0	0.001
Bull Trout	2.1	0.598
All Trout (except bull trout)	-9.3	0.828

We compared the mean total length of cutthroat, brook, bull and all trout within the restoration project area (Section 5) to control sections located below (Section 1) and above (Section 4) the restoration project, using the Before/After/Control/Impact (BACI) statistical design (Table 1-2). The BACI (differences of the differences) contrast for cutthroat, brook and all trout showed significant increases in total length within the project area after the project was completed (Table 1-2). Mean bull trout length showed a similar trend, but the contract was not significant ($p = 0.851$).

Table 1-2. Results from the before/after/control/impact (BACI) analysis of fish mean total length within the Young Creek restoration project area (impact) and two control sections on Young Creek.

Species	**BACI Contrast (mm)**	**P-Value**
Cutthroat Trout	-19.0	0.007
Brook Trout	-20.7	0.038
Bull Trout	-12.6	0.851
All Trout (except bull trout)	-22.1	0.001

Cutthroat trout abundance at all three monitoring sections on Young Creek declined in 2018 (one year after the wildfire) compared to mean abundance estimates ten years prior (2008-2017). However, the declines were sharpest within Section 4 (-60.9%), where the fire burned extensively. Cutthroat trout abundance also declined by 28 and 15.8%, respectively at the next two monitoring sections downstream of the wildfire (Figure 1-6). Cutthroat trout abundance did not decrease proportionally for all size classes in 2018 at Section 5. Presumed mortality was skewed heavily to smaller and younger size/aged fish (Figure 1-7). Cutthroat trout within length groups 50-99 mm likely represent age 1 fish and fish within the length groups 100-149 likely represent age 2 fish. Brook trout abundance at all three monitoring sections on Young Creek declined in 2018 compared to mean abundance estimates ten years prior. Brook trout abundance in 2018 at Sections 4 and 1 were both 64% lower than the mean that occurred ten years prior, and brook trout abundance at Section 5 in 2018 was 20% lower than the previous ten years (Figure 1-6). However, the increasing trend in brook trout abundance through time at Sections 4 and 5 (see sections above) make comparison to the ten year means problematic. We attempted to address this by comparing the relative change in brook trout abundance at Sections 4 and 5 between 2017 and 2018. Brook trout abundance at Section 4 exhibited the sharpest decline (68%), which was consistent with the results observed for cutthroat trout. Brook trout abundance at Section 5 declined by 20% from 2017 to 2018 (Figure 1-6). We were unable to compare the change in length groups between periods for brook trout because of the low number of fish captured within years.

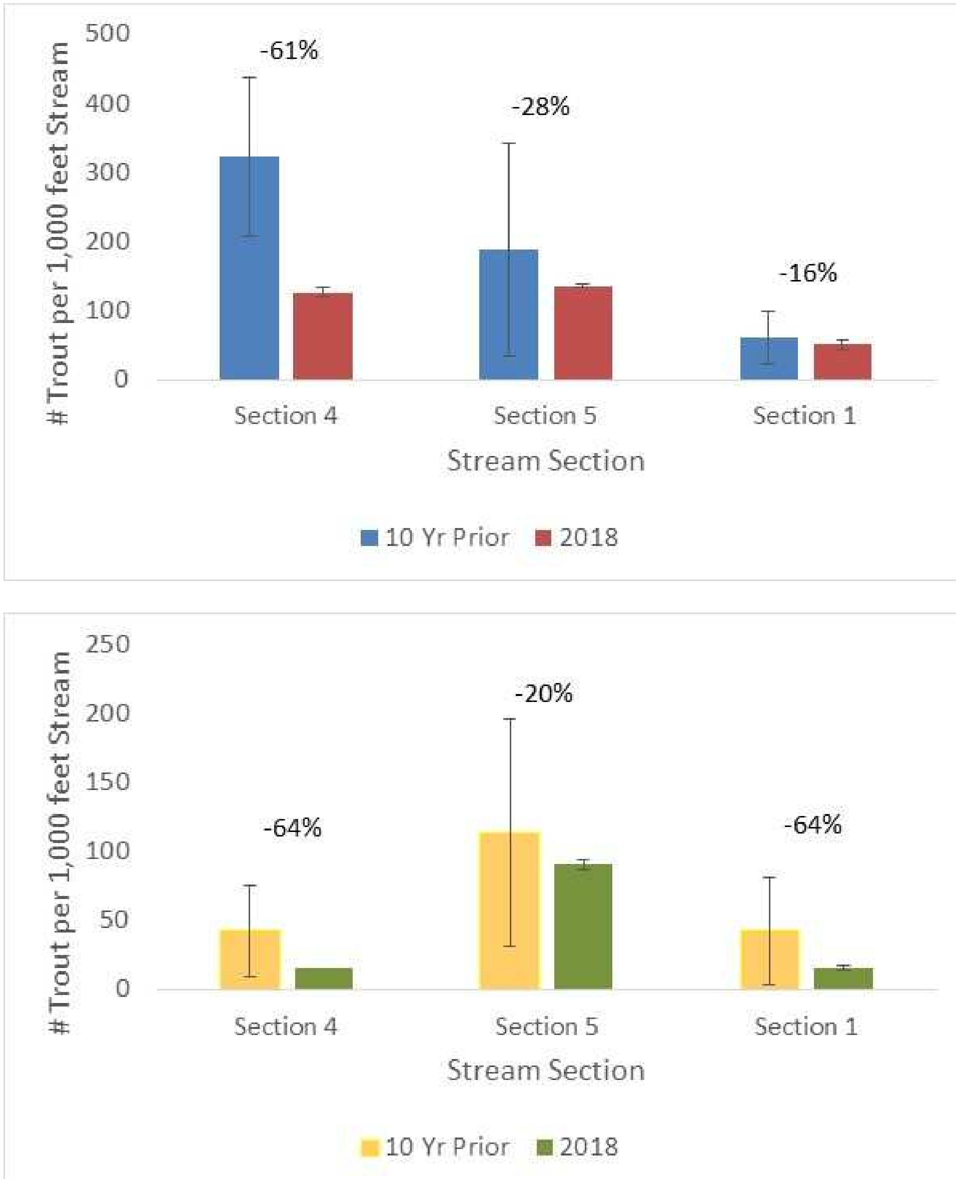


Figure 1-6. Estimated cutthroat trout (upper figure) and brook trout (lower figure) abundance for Sections 4 5 and 1 (upstream to downstream) on Young Creek during ten years prior to a large wildfire (2008-2017) and the first year after the fire (2018). The error bars represent 95% confidence intervals, and the number above each of the paired bars represents the percent change in abundance before and after the wildfire.

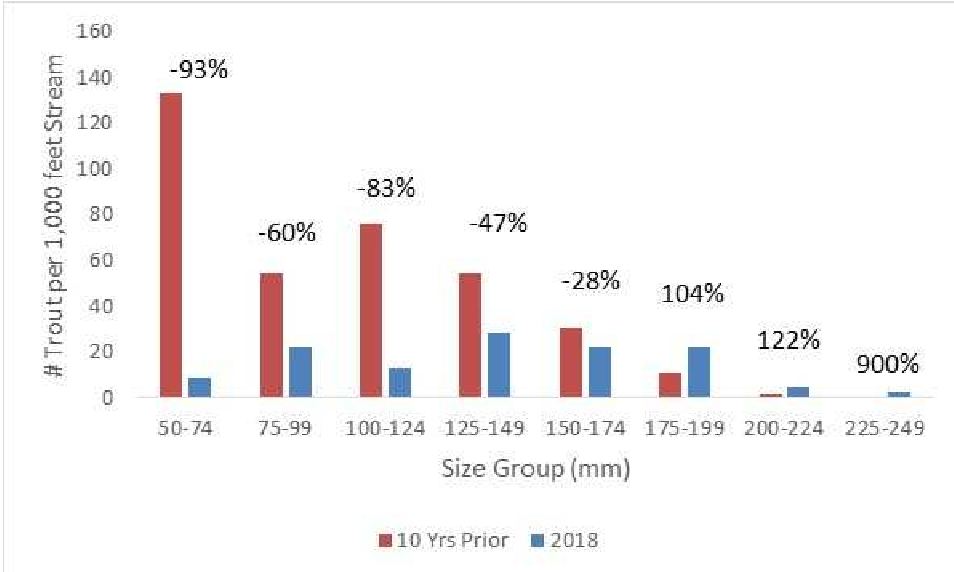


Figure 1-7. Estimated cutthroat trout abundance by 25 mm length groups for Section 4 on Young Creek during ten years prior to a large wildfire (2008-2017) and the first year after the fire (2018). The number above each of the paired bars represents the percent change in abundance before and after the wildfire.

Therriault Creek

Section 1, a downstream control site on Therriault Creek has been sampled annually since 1997, except 2000-2002. We have observed rainbow and brook trout annually at this site. Rainbow trout abundance in Section 1 of Therriault Creek has not differed significantly from a stable population ($r^2 = 0.04$; $p = 0.42$; Figure 1-8; Appendix Table A3). The mean abundance of rainbow trout during the period of record was 99.4 fish per 1,000 feet, with the estimated abundance in 2018 (88.6 fish per 1,000 feet) 11% lower than the annual mean. The trend in brook trout abundance for this section has shown a weak, but significantly increasing trend ($r^2 = 0.25$; $p = 0.02$; Figure 1-8; Appendix Table A3), and has averaged an overall increase of 4.1 brook trout per 1,000 feet each year. The observed abundance of brook trout in 2018 was 94.5 brook trout per 1,000 feet, which was slightly higher than the mean abundance at this site since 1997 (93 fish per 1,000 feet). Juvenile bull trout have been detected annually at this site since 2003, with abundance being highest in 2004 (92.1 bull trout per 1,000 feet) and averaging 15 fish per 1,000 feet over the period of record. However, we didn't observe any bull trout at this site in 2018. The high variability in catch of bull trout at this site precluded detecting a significant trend in abundance ($p = 0.55$; Figure 1-8). Cutthroat trout were only observed at this site in 2008. We also observed a single northern pike at this site in 2015, which was the only such observation.

Section 3 on Therriault Creek, an upstream control site, was sampled in 1997-1999, and annually since 2003 (Appendix Table A3). We observed rainbow and brook trout at this site each year, but bull trout were only observed annually beginning in 2003. The trend of rainbow trout abundance over the period of record has exhibited a weak but significant decline since 1997 ($r^2 = 0.21$; $p = 0.05$; Figure 1-9; Appendix Table A3), decreasing by an average of 3.3 fish per 1,000 feet per year. We estimated 107 rainbow trout per 1,000 feet were present at this site in 2018, which was 54% higher than mean over the period of record and the highest abundance observed since 2004. However, the trend in brook trout abundance has not differed from a stable population ($p = 0.65$). We estimated 123.8 brook trout per 1,000 feet at this site in 2018, which was 57% higher than the overall mean abundance of brook trout at this site since 1997 (79.0 fish per 1,000 feet). Bull trout abundance at this site has shown a weak, and nearly significant increase since 1997 ($r^2 = 0.20$; $p = 0.05$; Figure 1-9), increasing on average by 0.55 fish/1,000 feet per year. We estimated 9.5 bull trout per 1,000 feet at this site in 2018, which was slightly higher than the mean since 1997 (9.0 fish per 1,000 feet).

Section 2, within the restoration area on Therriault Creek, was sampled annually since 1997 except 2000 and 2002. The data we collected in 2018 represented the fourteenth year of post construction data. However, in 2011, we were unable to hold the lower block net during sampling, and therefore unable to produce a reliable abundance estimate at this site. Therefore, subsequent analyses do not include 2011 data. We observed rainbow and brook trout at this site every year we sampled this site, and bull trout were observed every year except 2010 (Figure 1-10; Appendix Table A3). We observed a single cutthroat trout at this site in 2007, 2012, 2015, 2016, and 2018. Rainbow trout abundance at this site has not exhibited a significant trend over the period of record (r^2 =0.08; p = 0.23; Figure 1-10). We observed an estimated 69.3 rainbow trout per 1,000 feet at this site in 2018, which was 38% higher than the mean over the period of record (50.2 fish per 1,000 feet). Bull trout abundance has exhibited a weak but significant negative decline since 1997, (r^2 = 0.21; p = 0.049), declining on average by 1.1 fish per 1,000 feet per year; Figure 1-10). We observed a single bull trout at this site in 2018, which resulted in an estimate of 1.1 fish per 1,000 feet. Brook trout abundance at this site has not exhibited a trend (r^2 = 0.003; p = 0.82). We estimated 73.7 brook trout per 1,000 feet within the project area in 2018 (Figure 1-10).

The mean abundance of rainbow trout we observed within this section after implementation (2005-2018) was 37.2 rainbow trout per 1,000 feet, which was significantly lower (p = 0.002) than the mean abundance prior to project completion (Figure 1-8; pre-project mean = 78.4 fish per 1,000 feet). Brook trout abundance at the restoration site did not differ significantly before (69.5 fish per 1,000 feet) and after (64.6 fish per 1,000 feet) project completion (p = 0.72; for a two-tailed test). Bull trout abundance within the project reach after implementation also significantly decreased (p = 0.004) from an average of 26.0 before the project to 4.6 bull trout per 1,000 feet after the project. Total trout (excluding bull trout) also significantly decreased (p = 0.007) after project completion (means 154.0 and 97.0 fish per 1,000 feet; Figure 1-11).

We compared the abundance of rainbow, brook, bull and all trout within the restoration project area (Section 2) to control sites located below (Section 1) and above (Section 3) the restoration project, using the Before/After/Control/Impact (BACI) statistical design (Table 1-3). The bull trout BACI contrast (difference of the differences) was the contrast that exhibited a significant difference within the restoration section (p = 0.006), which was consistent with bull trout abundance decreasing within the restoration area after the project. The BACI contrasts for rainbow trout indicated abundance decreased within the project area, but the results were not significant (p = 0.895). These results were consistent with the results from the student t-test (above). The BACI results for brook and all trout (excluding bull trout) also indicated that abundances did not differ significantly before and after (p > 0.05) project completion. Even though the results from the before/after student t-test comparisons (above) suggested that rainbow, bull and total trout significantly decreased after project completion, the results from the BACI analysis suggest that any observed differences were not likely attributable to the restoration work.

We compared the mean total length of rainbow, brook, bull and all trout within the restoration project area (Section 2) to control sites located below (Section 1) and above (Section 3) the restoration project, using the Before/After/Control/Impact (BACI) statistical design (Table 1-4). These analyzes found that mean length of brook trout and total trout exhibited marginally significant increases within the project area after the restoration efforts, increasing an average of 16.1 and 10.4 mm, respectively (p = 0.055 and 0.093; Table 1-4). Mean length of rainbow and bull trout after the project completion showed small and non-significant change (Table 1-4).

Our fish abundance monitoring at this site uses fish abundance per unit of stream length, and suggests that abundance may have decreased after implementation. However, our monitoring does not account for the fact that the restoration approximately doubled the length of stream. Therefore, if the single monitoring section within the project area is representative of fish abundance throughout the project area, our abundance estimates grossly under estimate total abundance within the project area. Furthermore, we didn't weigh fish, which would have facilitated estimates of biomass. The results of the BACI analyses (especially for brook trout) indicate that mean total length increased after the restoration work, suggesting that biomass within this section may have also increased after restoration.

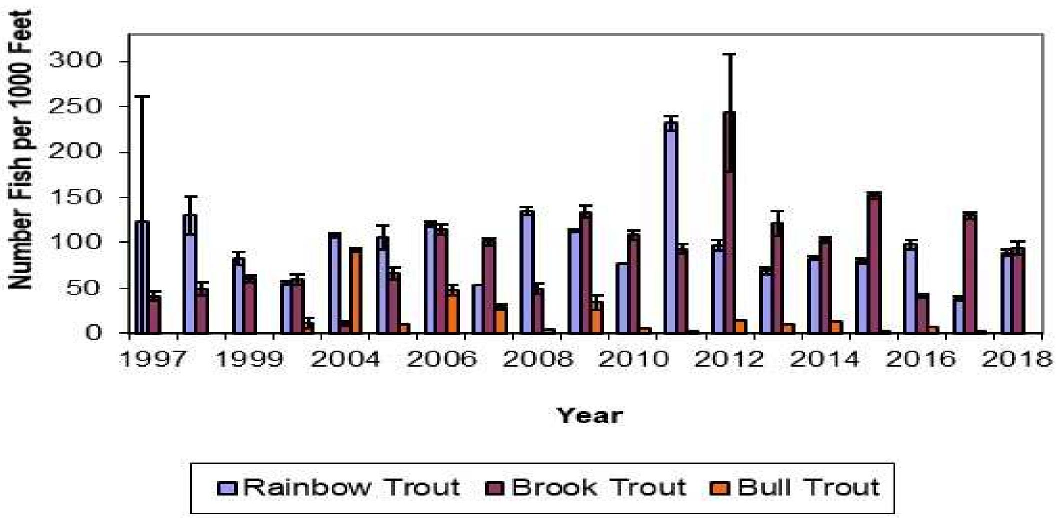


Figure 1-8. Rainbow trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 1 monitoring site from 1997-1999 and 2003-2018 collected by backpack electrofishing. The error bars represent 95% confidence intervals.

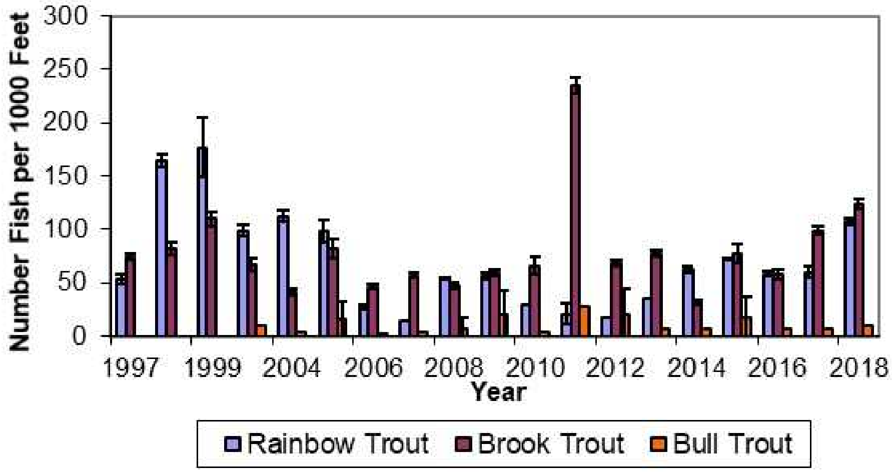


Figure 1-9. Rainbow trout, bull trout and brook trout densities (fish per 1,000 feet) within the Therriault Creek Section 3 monitoring site from 1997-1999 and 2003-2018 collected by backpack electrofishing. The error bars represent 95% confidence intervals.

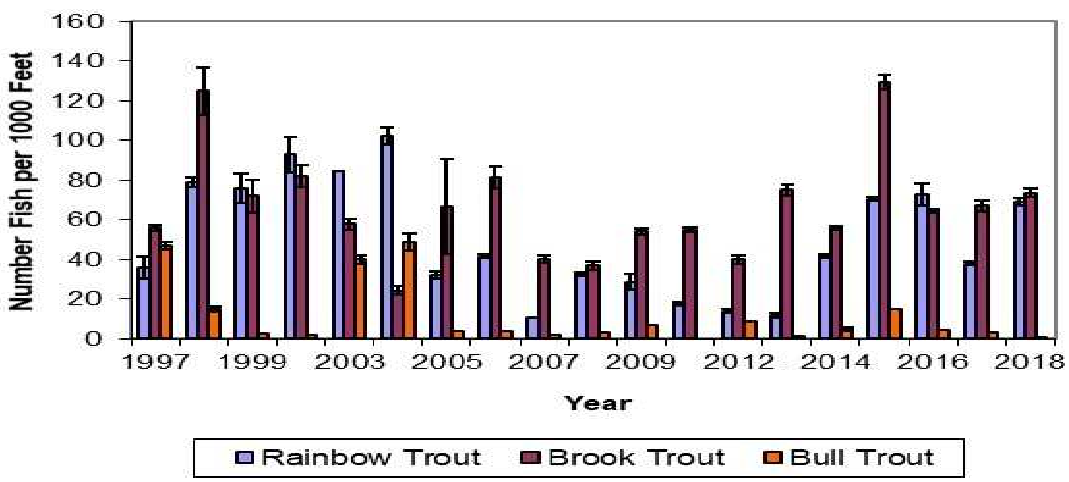


Figure 1-10. Rainbow trout, bull trout and brook trout densities (fish per 1000 feet) within the Therriault Creek Section 2 monitoring site from 1997-1999, 2001 and 2003-2018 collected by backpack electrofishing. The error bars represent 95% confidence intervals.

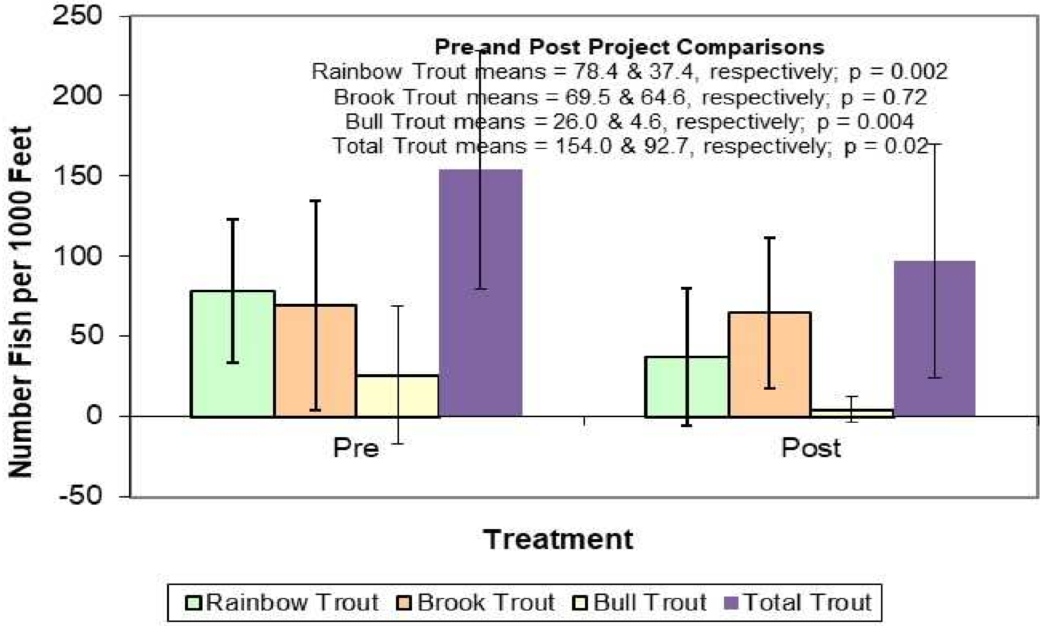


Figure 1-11. Rainbow, brook, bull and total trout densities (fish per 1,000 feet) within the Therriault Creek restoration project area (Sections 2) before (1997-2004) and after project completion (2005-2018). The error bars represent 95% confidence intervals.

Table 1-3. Results from the before/after/control/impact analysis (BACI) for fish abundance within the Therriault Creek restoration project area (impact) and two control sections.

Species	**BACI Contrast (fish/1,000 ft)**	**P-Value**
Rainbow Trout	2.9	0.895
Brook Trout	37.9	0.149
Bull Trout	26.5	0.006
All Trout (except bull trout)	49.2	0.146

Table 1-4. Results from the before/after/control/impact analysis (BACI) of fish mean total length within the Therriault Creek restoration project area (impact) and two control sections.

Species	BACI Contrast (mm)	P-Value
Rainbow Trout	-0.1	0.993
Brook Trout	-16.1	0.055
Bull Trout	4.8	0.871
All Trout (except bull trout)	-10.4	0.093

Libby Creek Upper Cleveland

The Upper Cleveland Project Area (Section 3) was sampled annually since 2000 (Appendix Table A4). Rainbow (redband) trout were the most abundant species within this section of Libby Creek during all years. However, we were unable to determine a trend in redband trout abundance at this site since we began sampling the site in 2000 ($r^2 < 0.03$; $p = 0.37$; Figure 1-12). We estimated 193.5 redband trout per 1,000 feet at this section in 2018. Mean annual redband trout abundance decreased slightly after project implementation, but the difference was not significant (mean abundance 168.3 and 147.9 fish per 1,000 feet, respectively; $p = 0.59$; two-tailed test; Figure 1-13). Brook trout were only observed at this site in 2010 and 2018 when we captured two and one fish, respectively. We observed an estimated 19.5 juvenile bull trout per 1,000 feet at this site in 2018. The mean abundance of juvenile bull trout abundance after project implementation was nearly threefold of that prior to completion of the restoration project, but was not significant (means = 6.0 and 16.8 fish per 1000 feet, respectively; $p = 0.33$ for two-tailed test; Figure 1-13). Bull trout abundance at this site has exhibited a weak and nearly significant increase through time ($r^2 = 0.162$; $p = 0.087$; Figure 1-12), increasing on average by 1.2 bull trout per 1,000 feet per year. We were unable to utilize the more powerful BACI design for this section due to the lack of an adequate control section. The entire Libby Creek watershed experienced a rain on snow event in December 2015 was likely responsible for the decreased abundance estimates we observed at this site in 2016, but likely may not have impacted fish abundance three years later.

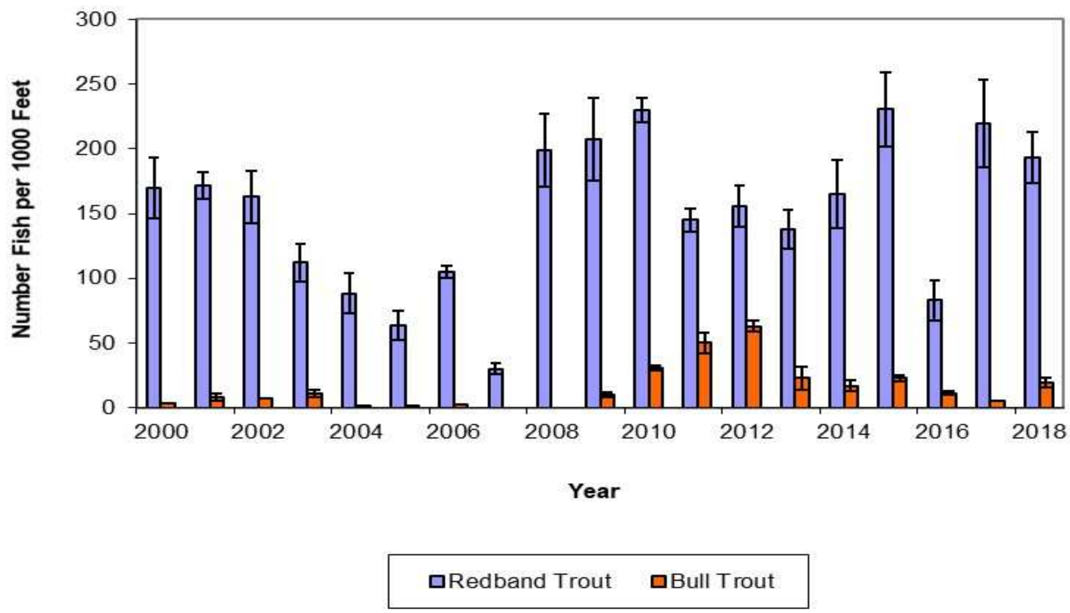


Figure 1-12. Redband trout and bull trout densities (fish per 1,000 feet) within the Libby Creek Upper Cleveland Stream Restoration Project area (Section 3) in 2000-2018 using a backpack electrofisher. The error bars represent 95% confidence intervals. The data from 2000-2002 represent pre-project trends of fish abundance, and the 2003-2018 data represent data after project completion.

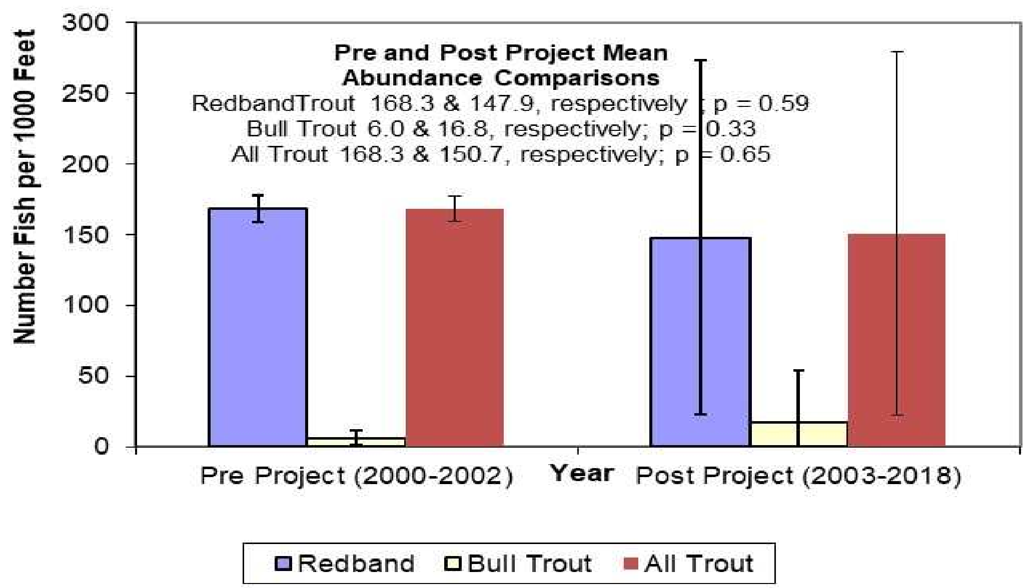


Figure 1-13. Redband trout and bull trout densities (fish per 1,000 feet) within the Libby Creek Upper Cleveland's Stream Restoration Project area (Section 3), comparing annual mean pre-project (2000-2002) data and post-project (2003-2018) using mobile electrofishing gear. The error bars represent 95% confidence intervals.

Libby Creek Lower Cleveland Phase II

Section 4 (downstream control site) was sampled annually since 2004, except for 2010 due to equipment failures and 2018 when we were unable to contact the landowner to obtain access (Appendix Table A4). Dunnigan et al (2018) presents the results of sampling this section through 2017.

We sampled Section 5 (upstream control site) annually since 2004, and rainbow (redband) trout have been the most abundant fish species observed each year. Estimated redband trout abundance has ranged from 129 (2007) to 406 (2008) fish per 1,000 feet at this site (Figure 1-12), and an overall mean abundance of 217.2 fish per 1,000 feet. In 2018 we observed an estimated 296.6 redband trout per 1,000 feet at this site. We were unable to detect a significant trend in redband trout abundance at this site since 2004 ($r^2 = 0.03$; $p = 0.52$; Figure 1-14). Brook trout and bull trout have occurred at a substantially lower abundance than redband trout during all sampling years. Brook trout were first observed at this site in 2009 (2 fish per 1,000), and doubled each subsequent year through 2011, but were not observed again until 2015 (2.2 fish per 1,000 feet) and have not been observed since. We found no evidence of a trend in brook trout abundance ($r^2 < 0.005$; $p = 0.99$). Brook trout have averaged 1.1 fish per 1,000 feet at this site. Bull trout abundance at this site has been relatively low and variable, occurring in ten of the fifteen sampling years. Bull trout abundance at this site peaked in 2012 at 13 fish per 1,000 feet (Figure 1-14). We found no evidence of a significant trend in bull trout abundance at this site since 2004 ($r^2 = 0.14$; $p = 0.16$). We observed an estimated 7.4 bull trout per 1,000 feet at this site in 2018. Bull trout abundance at this site has averaged 4.9 bull trout per 1,000 feet since 2004.

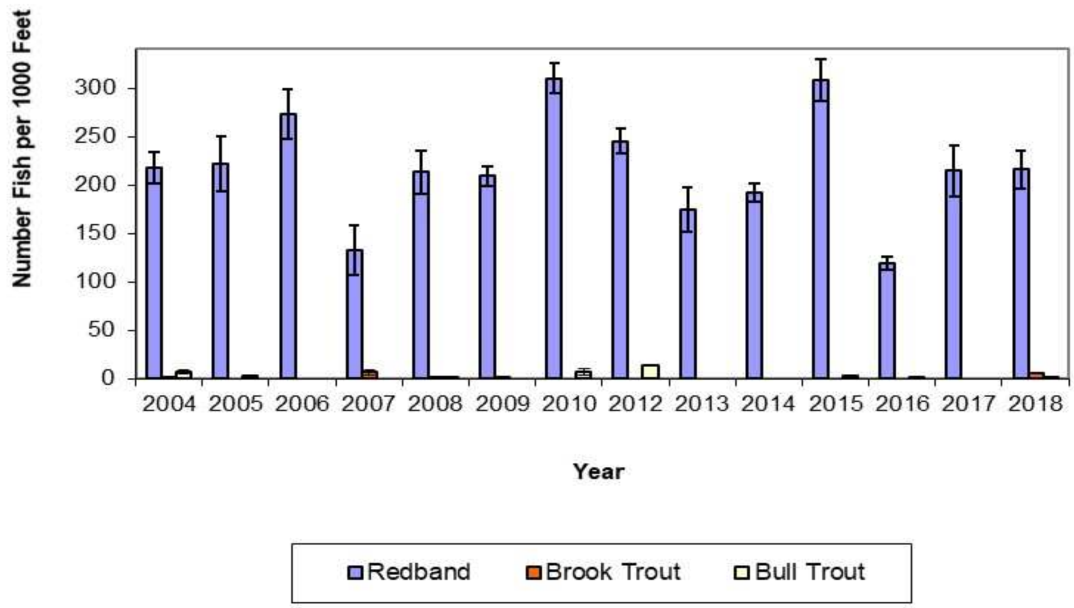


Figure 1-14. Redband, brook and bull trout densities (fish per 1,000 feet) within Section 5 of Libby Creek from 2004 – 2018 (except 2010). This site was sampled using a backpack electrofisher. The Error bars represent the 95% confidence intervals.

Section 6 (Lower Cleveland Phase II site) was sampled annually since 2004, except 2011 when equipment failures prevented us from completing a valid abundance estimate (Appendix Table A4). Therefore, we excluded 2011 from all subsequent analyses. Rainbow (redband) trout were the most abundant species in this section during all years (Figure 1-15). Redband trout abundance at this site has ranged from 310 fish per 1,000 feet in 2010 to a low of 119 fish per 1,000 feet in 2016, and an overall mean since 2004 of 217.5 redband trout per 1,000 feet. We estimated 215.8 redband trout per 1,000 feet at this section in 2018. We were unable to discern a significant trend in redband trout abundance at this site ($r^2 = 0.01$; $p = 0.74$; Figure 1-15). We observed brook trout in this section in six of the previous 15 sampling years (including 2011 and 2015), with no significant trend through time ($r^2 = 0.001$; $p = 0.90$). We observed three brook trout at this site in 2018, for an estimated abundance of 6.2 brook trout per 1,000 feet. The overall mean brook trout abundance at this site since 2004 was 1.3 fish per 1,000 feet. Bull trout have also been observed in nine of the previous 15 years, with a peak abundance (14 fish per 1,000 feet) observed in 2012, and an overall mean abundance of 2.9 fish per 1,000 feet. We observed a single bull trout at this section in 2018 that yielded an abundance estimate of 2.1 fish per 1,000 feet. We were unable to distinguish a significant trend in bull trout abundance through time at this site ($r^2 < 0.02$; $p = 0.62$; Figure 1-15).

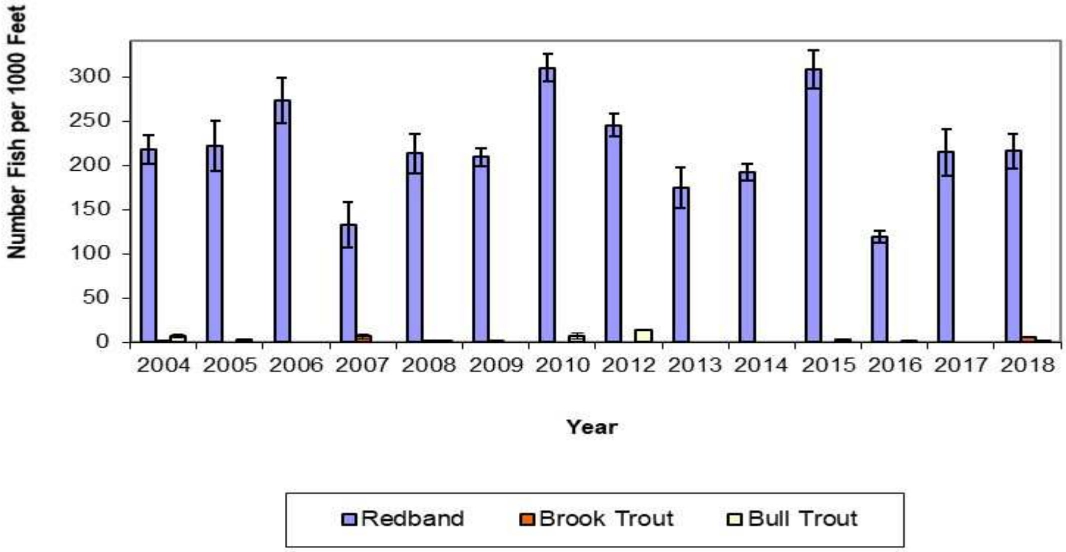


Figure 1-15. Redband, brook and bull trout densities (fish per 1,000 feet) within Section 6 of Libby Creek from 2004 – 2018 (except 2011). This site was sampled using a backpack electrofisher. The Error bars represent the 95% confidence intervals.

Abundance estimates of redband trout within Section 6 (treatment) decreased slightly after project completion from an average of 237.3 fish per 1,000 feet prior to restoration to 212.1 fish per 1,000 feet after (Figure 1-16), but the difference was not significant ($p = 0.51$; two-tailed t-test). Mean brook trout abundance was also not significantly different before and after restoration efforts, despite a two-fold increase from 0.6 prior to restoration to 1.5 fish per 1,000 feet ($p = 0.55$; two-tailed t-test). Bull trout abundance was similar before and after restoration work (3.5 and 2.7 fish per 1,000 feet, respectively), and did not differ significantly ($p = 0.77$; two-tailed t-test; Figure 1-16). Total trout abundance (excluding bull trout) was similar to the results for redband trout because redband trout dominated the catch at this site. Total trout (excluding bull trout) prior to restoration work averaged 238.4 fish per 1,000 feet and did not differ significantly ($p = 0.52$) from the post restoration average (214.1 fish per 1,000 feet; Figure 1-16).

We compared the abundance of redband, brook, bull and all trout within the restoration project area (Section 6) to control sites located below (Section 4) and above (Section 5) the restoration project, using the Before/After/Control/Impact (BACI) statistical design (Table 1-5). The brook trout BACI contrast (difference of the differences) was the only species that showed a small increase (0.4 fish/1,000 feet) within the restoration section, but the difference was not significant ($p = 0.818$; Table 1-5). The BACI contrast for redband trout (BACI = 37.7 fish/1,000 feet; Table 1-5) indicated that redband trout abundance has not differed significantly after the restoration work which was consistent with the before/after (t-test) comparison. The BACI contrast for bull trout (BACI = 3.6 bull trout per 1,000 feet; Table 1-5) also indicated that bull trout abundance did not statistically differ as a result of the restoration work (p = 0.100). These results were also consistent with the before/after (t-test) comparison reported above. Lastly, the BACI contrast for all trout species (except bull trout) showed a similar result to the redband trout comparison since total trout abundance was largely influenced by redband trout that comprised most the catch.

We have little evidence to suggest that the mean total length of redband, brook, bull or all trout within the restoration project area differed significantly after the restoration work using the BACI statistical design (Table 1-6). The brook trout BACI contrast was the only species that showed an increase (23.9 mm) after the restoration, but the difference was not significant (p = 0.382; Table 1-6). The BACI contrast for redband and bull trout mean total length indicated that mean length for both species decreased slightly after restoration, but not significantly so (Table 1-6). Results for the mean total length of total trout were similar to the results for redband trout because redband trout dominated the catch within all three of these sections during all years.

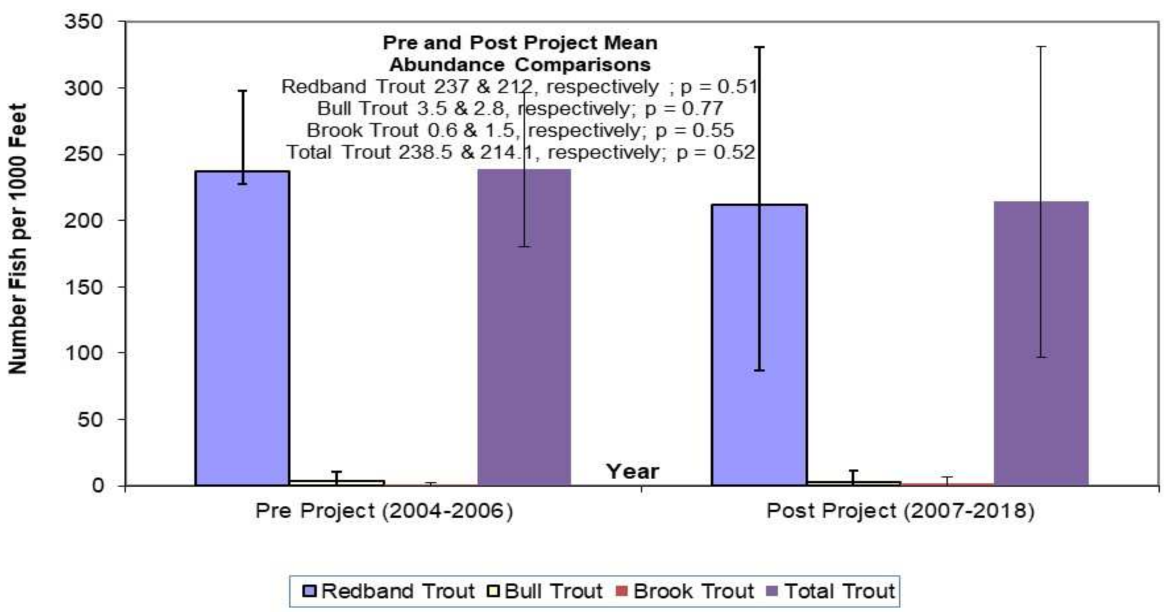


Figure 1-16. Redband trout, bull trout, brook, and total trout densities (fish per 1,000 feet) within the Libby Creek Lower Cleveland's Phase II Stream Restoration Project area (Section 6), comparing annual mean pre-project (2004-2006) data and post-project (2007-2018) using mobile electrofishing gear. The error bars represent 95% confidence intervals.

Table 1-5. Results from the before/after/control/impact analysis(BACI) of fish abundance within the Libby Creek Lower Cleveland's Phase II Restoration Project area (impact) and two control sections (Section 4 and 5).

Species	**BACI Contrast (fish/1,000 ft)**	**P-Value**
Redband Trout	37.7	0.393
Brook Trout	-0.4	0.818
Bull Trout	3.6	0.100
All Trout (except bull trout)	36.8	0.352

Table 1-6. Results from the before/after/control/impact analysis(BACI) of mean total length within the Libby Creek Lower Cleveland's Phase II Restoration Project area (impact) and two control sections (Section 4 and 5).

Species	**BACI Contrast (mm)**	**P-Value**
Redband Trout	6.0	0.158
Brook Trout	-23.9	0.383
Bull Trout	51.0	0.289
All Trout (except bull trout)	5.9	0.160

Physical Monitoring

Young Creek

The Young Creek State Lands Restoration Project substantially reduced the mean width and increased the mean and maximum depth of riffle habitats within this section of Young Creek, and these changes resulted in a compounding effect of reducing width to depth ratio within the project area (Table 1-7). Stream channel dimensions within riffle habitats within this section of Young Creek have changed relatively little between 2004 and 2017, despite the fact that the survey in 2017 was completed after the large wildfire in 2017 burned much of the upper watershed. 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 four riffles annually since 2003 (as-built). We measured eight riffles in 2017, which was a decrease of two riffles compared to the previous year and 2003 (as-built). Mean bankfull width of riffle habitats in 2017 measured 17.5 feet, which was 3.6% higher than the previous year and the highest value since the project was completed. Mean riffle bankfull depth likewise decreased by 7.6% from 2016 to 2017, measuring 1.11 feet. The increased mean width and decrease in mean depth resulted in a subsequent slight decrease in mean riffle cross sectional area and an increase in width to depth ratio (Table 1-7). Mean width to depth ratio in 2017 was 16.0, which was the highest value observed since the project was complete, but nevertheless remained 67% lower than existed prior to project completion. Despite the slight annual changes in the riffle dimensions within this section of Young Creek, the stream channel dimensions remained similar to the constructed stream channel dimensions, with changes generally less than 10-15% between years (Table 1-7). The riffle habitats in 2017 remained 37% narrower and 86% deeper (mean depth) than existed prior to the project (Table 1-7). Therefore, given these data, the channel dimensions within the riffle habitats are being maintained within this section Young Creek since the initial construction in 2003.

The stream channel dimensions of the pool habitats within the project are have also remained relatively stable since project completion. The initial restoration work increased the quality and quantity of pool habitat for resident salmonids, and these changes are being sustained since the project was completed. The total number of pools, total pool area and total pool volume, remain 850, 342, and 837% higher than existed in this section of Young Creek prior to the restoration work (Table 1-8). 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, although we made no attempt to quantify habitat complexity. The total number of pools within this section of Young Creek in 2017 increased by two pools from the year before which was over twice as many pools as were originally constructed as part of the restoration work in 2003. Mean pool dimensions remained similar since the previous year (Table 1-8). Total pool surface area and volume continue to meet or exceed the as-built conditions in 2003. The constructed pool habitat continues to provide an improvement in the amount of depth and cover that existed prior to the project and those changes are overwhelmingly being sustained. (Table 1-8).

Table 1-7. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio measured for the total number of riffles 2002-2017 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 select years is also presented.

Year	Number Of Riffles	Cross Sectional Area (ft^2)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Width to Depth Ratio
2002 (Existing)	4	16.8 (1.6)	27.9 (22.7)	0.60 (0.01)	1.05 (0.02)	48.3 (239.6)
2003 (As Built)	10	22.0 (10.1)	16.3 (9.2)	1.24 (0.05)	1.99 (0.09)	13.7 (21.2)
2004	11	18.7 (6.3)	14.8 (3.6)	1.28 (0.07)	1.85 (0.13)	12.3 (17.3)
2005	11	21.9 (22.0)	16.1 (4.4)	1.37 (0.08)	1.79 (0.09)	12.3 (11.4)
2006	10	19.7 (14.1)	15.6 (4.7)	1.29 (0.12)	1.89 (0.14)	13.0 (22.5)
2007	10	19.1 (15.5)	14.8 (4.0)	1.32 (0.14)	1.72 (0.12)	14.4 (25.1)
2008	10	20.0 (9.4)	16.0 (5.4)	1.25 (0.02)	1.75 (0.07)	13.0 (7.1)
2009	8	18.0 (6.2)	15.2 (5.7)	1.19 (0.02)	1.59 (0.11)	13.1 (9.9)
2010	8	20.0 (5.5)	16.7 (14.9)	1.21 (0.01)	1.72 (0.07)	14.0 (6.8)
2011	9	20.5 (6.8)	16.9 (10.0)	1.24 (0.06)	1.63 (0.09)	14.5 (27.6)
2012	11	22.4 (4.4)	17.4 (10.4)	1.32 (0.07)	1.71 (0.07)	14.0 (25.9)
2013	8	21.1 (4.6)	17.0 (4.3)	1.25 (0.03)	1.71 (0.04)	13.9 (11.4)
2014	7	22.1 (4.9)	17.3 (4.7)	1.28 (0.02)	1.66 (0.04)	13.8 (11.5)
2015	9	18.6 (4.1)	16.9 (3.8)	1.11 (0.02)	1.77 (0.03)	15.7 (17.5)
2016	10	20.2 (11.4)	16.9 (4.9)	1.20 (0.03)	1.70 (0.08)	14.5 (12.8)
2017	8	19.4 (7.9)	17.5 (5.3)	1.11 (0.02)	1.58 (0.04)	16.0 (10.8)
Percent Change						
2002/2003	150.0%	31.3%	-41.5%	107.5%	89.0%	-71.6%
2002/2017	100.0%	15.7%	-37.2%	86.0%	50.0%	-66.9%
2014/2017	14.3%	-12.1%	1.2%	-13.5%	-5.0%	16.4%
2015/2017	-11.1%	4.5%	3.9%	0.1%	7.4%	2.3%
2016/2017	-20.0%	-4.2%	3.6%	-7.6%	-7.1%	10.7%

Table 1-8. Mean bankfull width, maximum bankfull depth, and mean length, total length and surface area measured from pools located in the Young Creek State Lands Stream Restoration Project from 2002-2017. The project was constructed in the fall of 2003. Variance estimates for annual mean values are presented in parentheses. The percent change between select years is also presented.

Year	Number of Pools	Mean Bankfull Width (ft)	Mean Bankfull Depth	Maximum Bankfull Depth (ft)	Mean Length (ft.)	Total Area (ft^2)	Total Volume (ft^3)
2002 (Existing)	2	23.5 (24.5)	0.79	2.35 (0.13)	42.5	1,998	1,578
2003 (As Built)	8	21.8 (18.0)	1.73	3.23 (0.42)	48.6	8,480	14,671
2004	14	19.2 (24.7)	1.73	3.63 (0.53)	32.8	8,602	14,881
2005	15	17.8 (12.8)	1.71	3.08 (0.67)	30.9	8,218	14,053
2006	17	17.4 (9.8)	1.74	3.12 (0.40)	35.5	10,667	17,923
2007	17	16.8 (10.0)	1.75	3.04 (0.22)	34.7	10,090	16,544
2008	17	15.9 (7.8)	1.94	3.12 (0.26)	30.5	8,231	15,751
2009	17	16.6 (9.3)	1.99	2.98 (0.25)	32.5	8,972	17,052
2010	16	15.8 (8.5)	1.99	2.93 (0.11)	34.6	8,682	16,921
2011	16	16.1 (7.3)	1.87	2.90 (0.84)	35.1	9,104	16,027
2012	15	16.8 (7.2)	2.08	3.34 (0.55)	31.9	8,137	16,066
2013	19	16.2 (4.2)	1.73	2.84 (0.28)	32.5	10,051	17,257
2014	18	16.4 (7.3)	1.80	3.02 (0.27)	30.4	8,942	16,009
2015	17	16.2 (6.3)	1.79	2.86 (0.35)	31.3	8,569	14,871
2016	17	16.2 (6.8)	1.86	3.03 (0.33)	30.5	8,470	15,727
2017	19	15.5 (4.3)	1.75	2.77 (0.26)	30.0	8,836	15,345
Percent Change							
2002/2003	300.0%	-7.18%	119.0%	37.8%	14.41%	314.1%	758.7%
2002/2017	850.0%	34.2%	121.1%	18.0%	-29.5%	341.6%	837.1%
2014/2017	5.6%	-5.7%	-3.2%	-8.2%	-1.6%	-1.2%	-4.2%
2015/2017	11.8%	-4.3%	-2.2%	-3.0%	-4.3%	3.1%	3.2%
2016/2017	11.8%	-4.6%	-6.1%	-8.4%	-1.9%	4.3%	-2.5%

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 view for this section of Young Creek, it has changed little since the project was completed in 2003. This project continues to meet the original objectives (Dunnigan et al. 2005) set forth for this project.

Therriault Creek

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 to 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, and although we did not survey the pattern or the stream length in 2017, it remained very similar to the constructed condition.

This restoration project also changed the stream channel dimensions of Therriault Creek within the project area (Dunnigan et al. 2008), and these changes have been self-sustaining since the project was completed (Tables 1-9 to 1-12). Stream channel dimensions for riffle/run habitats within Reach 1 in 2017 were not significantly different ($p > 0.05$) from existing conditions and have remained relatively unchanged since the project was completed a decade earlier (Table 1-9). Within the riffle/run habitats of reach 1, stream channel dimensions changed less than 5% from 2015 to 2016 (Table 1-9). Although none of the stream channel dimensions within Reach 1 in 2017 differed significantly ($p > 0.05$) from existing conditions, the stream channel since the restoration efforts has remained consistently narrower and deeper (Table 1-9). We attribute the general lack of power to detect differences in these attributes to a limited sample size prior to the restoration work. Although we did not measure entrenchment ratio (flood prone width at two times bankfull depth divided by bankfull width), it undoubtedly remains higher than existed prior to the restoration work, and represents perhaps the largest functional change made to section of Therriault Creek after the restoration efforts.

The riffle/run habitat dimensions in Reach 2 changed significantly after the restoration efforts, and those changes have also been sustained since 2004 (Table 1-10). Mean bankfull width and width to depth ratios significantly decreased and mean bankfull depth significantly increased immediately after reconstruction. Mean bankfull width and mean depth averaged 8.9 and 1.58 feet, respectively in 2017, and have changed little since construction efforts (Table 1-10). As a result, mean width to depth ratio also remained significantly lower (48.3%) than existed prior to the stream channel reconstruction. Cross sectional area within the riffle/run habitats in reach 2 have remained similar in all years ($p = 0.413$), changing less than 10% annually (Table 1-10). Stream channel dimensions within the riffle/run habitats in Reach 2 of Therriault Creek changed little (<8%) between 2016 and 2017 (Table 1-10). As was the case with Reach 1, we did not measure entrenchment ratio, but it also undoubtedly remains higher than existed prior to the restoration work and represents a large change in the ecological function within this portion of Therriault Creek.

Pool habitat dimensions within Reaches 1 and 2 were similar before and after the restoration efforts. Stream channel dimensions within pool habitats of Reaches 1 and 2 were slightly wider (~21%) prior to restoration activities. Maximum depth within the pool habitats of both reaches differed little before and after restoration. Mean pool depth within reaches 1 and 2 was slightly (<3%) shallower in 2017 compared to existing conditions. Cross-sectional area in reach 1 decreased by about 30% immediately after construction and remained about 22% lower in 2017 compared to existing conditions. However, none of these differences were significant at the alpha = 0.05 (Table 1-11 and 1-12). Mean pool depth in Reach 1 has been consistently shallower since project completion (Table 1-11), but mean pool depth in Reach 2 has remained consistently deeper after project completion (Table 1-12). We were unable to relocate one of the permanent cross sections in Reach 1 used to survey pool habitats in 2011, which reduced our sample size to three in subsequent years including 2017. Stream channel dimensions within pool habitats have exhibited low annual change within both reaches on this section of Therriault Creek (Tables 1-11 and 1-12). For example, stream channel dimensions in pool habitats in both reaches changed little from 2016 to 2017. Cross sectional area in Reach 2 exhibited the largest relative change from the previous year, decreasing by 5% which also associated with smaller decreases in mean bankfull width (1.4%) and mean depth (3.8%; Table 1-12). The stream channel dimensions of pools in reach 1 changed less than 5% between 2016 and 2017 (Table 1-11).

Although we did not quantify total pool area or volume within the project area, due to the extensive total length of the project area (~9,100 feet), we are confident that the total area and volume increased after the restoration work and that these changes have been sustained through time. The pool habitat dimensions have been relatively stable from since 2003, but the approximate two-fold increase in stream length has undoubtedly had an overwhelming increase on total pool area and volume within the entire project area. This section of Therriault Creek is maintaining in a state of relative dynamic equilibrium, and thus the stream plan form has remained nearly identical since it was originally constructed, sustaining the overall increases of pool habitat within the project reach. These changes in plan form were the largest functional change in this section of Therriault Creek after the restoration efforts, and the stability of which can be attributed to the extensive efforts to restore the riparian vegetative community within the project area (see Chapter 3).

Table 1-9. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio for riffle/run-type habitats in Reach 1 of the Therriault Creek Restoration Project area. The project was constructed in the fall of 2004-2005. The variance for annual mean values is presented in parentheses. The percent change between select 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 Tukey's Test. Significant comparisons are indicated via * (alpha < 0.05).

Year	# Runs	Cross Sectional Area (ft^2)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Width to Depth Ratio
2003 (Existing)	10	13.9 (13.2)	12.6 (8.4)	1.1 (0.01)	1.5 (0.04)	11.5 (6.8)
2004 (As Built)	4	11.2 (2.9)	8.3 (0.30)	1.3 (0.05)	1.8 (0.14)	6.4 (2.1)
2005	4	13.8 (4.3)	9.6 (5.3)	1.5 (0.01)	1.9 (0.08)	6.7 (5.1)
2006	4	12.8 (6.8)	10.0 (4.2)	1.3 (0.11)	1.9 (0.13)	8.2 (10.1)
2007	4	13.2 (25.4)	10.5 (9.8)	1.2 (0.07)	1.6 (0.18)	8.7 (6.8)
2008	4	13.0 (2.9)	10.5 (7.2)	1.3 (0.09)	1.7 (0.15)	8.8 (16.1)
2009	4	14.7 (8.0)	10.9 (7.1)	1.4 (0.05)	1.8 (0.11)	8.3 (8.0)
2010	4	14.0 (10.0)	11.0 (8.4)	1.3 (0.03)	1.9 (0.11)	8.7 (8.0)
2011	3	12.6 (1.2)	10.1 (1.0)	1.3 (0.04)	1.7 (0.04)	8.2 (4.7)
2012	3	11.8 (13.1)	10.6 (1.2)	1.1 (0.19)	1.4 (0.16)	10.9 (41.4)
2013	3	12.5 (7.7)	10.3 (0.2)	1.2 (0.05)	1.7 (0.02)	9.0 (3.3)
2014	3	13.0 (5.1)	11.6 (2.2)	1.2 (0.11)	1.8 (0.02)	11.0 (23.2)
2015	3	13.3 (0.9)	11.6 (2.5)	1.2 (0.05)	1.8 (0.05)	10.4 (12.8)
2016	3	13.6 (1.8)	10.5 (0.1)	1.3 (0.02)	1.7 (0.02)	8.3 (1.7)
2017	3	13.3 (5.4)	10.3 (0.6)	1.3 (0.07)	1.6 (0.04)	8.1 (4.7)
P-Value		0.975	0.412	0.664	0.415	0.345
Percent Change						
2003/2004		-19.7%	-34.0%	22.2%	17.8%	-44.5%
2003/2017		-4.3%	-18.6%	18.4%	7.7%	-29.1%
2004/2005		22.9%	15.3%	8.2%	8.6%	5.7%
2004/2017		19.2%	23.4%	-3.1%	-8.6%	27.9%
2012/2017		12.6%	-3.5%	14.2%	12.7%	-25.5%
2013/2017		6.7%	0.1%	7.6%	-5.9%	-10.0%
2014/2017		2.3%	-11.8%	13.5%	-9.4%	-26.2%
2015/2017		0.2%	-11.6%	11.9%	-8.6%	-21.8%
2016/2017		-1.7%	-2.4%	1.2%	-3.0%	-1.6%

Table 1-10. Mean cross sectional area, bankfull width, depth, maximum bankfull depth, and width to depth ratio for riffle/run-type habitats in Reach 2 of the Therriault Creek Restoration Project area. The project was constructed in the fall of 2004-2005. The variance 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 Tukey's Test. Significant comparisons are indicated via * (alpha < 0.05).

Year	# Runs	Cross Sectional Area (ft^2)	Mean Bankfull Width (ft)	Mean Bankfull Depth (ft)	Maximum Bankfull Depth (ft)	Width to Depth Ratio
2003 (Existing)	10	13.9 (13.2)	12.6 (8.4)	1.1 (0.01)	1.5 (0.04)	11.5 (6.8)
2004 (As Built)	6	14.5 (9.4)	8.3 (0.3)	1.7 (0.08)	2.2 (0.11)	4.9 (0.4)
2005	6	15.1 (6.1)	8.3 (0.3)	1.8 (0.04)	2.2 (0.06)	4.6 (0.1)
2006	6	14.7 (5.6)	8.4 (0.1)	1.7 (0.07)	2.2 (0.13)	4.9 (0.5)
2007	6	16.3 (5.5)	8.2 (0.2)	2.0 (0.04)	2.3 (0.09)	4.1 (0.1)
2008	6	12.8 (3.2)	8.3 (0.2)	1.5 (0.03)	2.0 (0.13)	5.4 (0.3)
2009	4	13.1 (5.0)	8.6 (0.4)	1.5 (0.05)	1.9 (0.09)	5.8 (0.9)
2010	6	13.6 (3.0)	8.4 (0.3)	1.6 (0.07)	2.0 (0.11)	5.3 (1.3)
2011	6	16.2 (3.9)	10.1 (2.6)	1.62 (0.04)	2.3 (0.11)	6.4 (2.6)
2012	6	14.4 (4.2)	8.7 (0.3)	1.66 (0.08)	2.0 (0.19)	5.4 (1.2)
2013	6	14.8 (1.1)	9.1 (0.6)	1.65 (0.06)	2.0 (0.09)	5.7 (1.3)
2014	6	15.0 (4.0)	9.1 (0.4)	1.67 (0.09)	2.1 (0.13)	5.6 (1.6)
2015	6	14.7 (4.0)	9.1 (0.5)	1.63 (0.10)	2.1 (0.18)	5.8 (1.7)
2016	6	15.1 (2.1)	9.1 (0.6)	1.67 (0.07)	2.08 (0.14)	5.7 (1.8)
2017	6	13.9 (3.7)	8.9 (0.5)	1.58 (0.12)	2.01 (0.26)	5.9 (2.5)
P-Value		0.413	<0.001	<0.001	0.004	<0.001
Percent Change						
2003/2004		4.4%	-33.9%*	57.0%*	45.9%*	-57.3%*
2003/2017		-0.2%	-29.3%*	43.3%*	35.2%	-48.3%*
2004/2005		3.5%	0.0%	4.1%	0.0%	-5.2%
2004/2017		-4.4%	7.0%	-8.7%	-7.3%	21.1%
2012/2017		-3.8%	2.1%	-4.8%	0.8%	9.5%
2013/2017		-6.1%	-1.8%	-4.0%	1.3%	4.8%
2014/2017		-7.5%	-2.0%	-5.1%	-3.6%	4.9%
2015/2017		-5.5%	-2.5%	-2.7%	-2.8%	1.4%
2016/2017		-7.7%	-2.2%	-5.1%	-3.6%	4.9%

Table 1-11. Mean cross sectional area, bankfull width, depth, maximum bankfull and depth 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 select 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).

Year	Number of Pools	Cross Sectional Area (ft^2)	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)
2008	4	12.9 (10.3-14.4)	10.3 (9.9-10.8)	1.3 (1.0-1.4)	2.2 (1.8-2.5)
2009	4	13.4 (9.6-18.1)	10.8 (9.9-12.2)	1.2 (0.9-1.5)	2.2 (1.8-2.5)
2010	4	13.4 (9.7-15.6)	10.4 (9.6-11.0)	1.3 (1.0-1.4)	2.3 (1.8-2.5)
2011	3	15.2 (10.3-17.9)	10.7 (10.1-11.8)	1.5 (0.9-1.8)	2.2 (1.4-2.8)
2012	3	14.4 (6.5-20.2)	12.5 (10.4-14.8)	1.2 (0.4-1.9)	2.1 (0.8-2.8)
2013	3	13.9 (10.3-16.1)	11.4 (9.9-12.6)	1.2 (0.9-1.5)	2.4 (1.6-2.8)
2014	3	12.6 (8.9-15.6)	10.8 (10.5-11.0)	1.2 (0.8-1.5)	2.1 (1.4-2.5)
2015	3	13.0 (9.6-14.8)	10.6 (10.3-11.2)	1.2 (0.9-1.4)	2.1 (1.5-2.5)
2016	3	14.2 (10.6-17.6)	10.7 (10.6-10.7)	1.3 (1.0-1.6)	2.3 (1.7 (2.7)
2017	3	17.7 (11.5-17.7)	11.0 (10.5-11.7)	1.3 (1.1-1.6)	2.4 (1.9-2.7)
P-Value		0.126	0.079	0.961	0.976
Percent Change					
2003/2004		-30.3%	-33.0%	-1.0%	-11.3%
2003/2017		-21.9%	-18.1%	-8.7%	-3.2%
2004/2005		3.9%	9.7%	-4.6%	2.8%
2004/2017		12.0%	22.2%	-7.8%	9.1%
2012/2017		2.5%	-12.3%	8.5%	14.3%
2013/2017		6.3%	-3.1%	8.5%	1.4%
2014/2017		16.9%	1.6%	14.4%	16.1%
2015/2017		13.3%	3.4%	9.7%	14.3%
2016/2017		3.8%	3.2%	0.7%	4.3%

Table 1-12. Mean cross sectional area, bankfull width, depth, maximum bankfull and depth 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).

Year	Number of Pools	Cross Sectional Area (ft^2)	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)
2008	6	15.7 (11.9-17.3)	9.9 (8.3-13.0)	1.6 (1.3-1.9)	2.4 (2.0-2.8)
2010	6	16.5 (13.3-18.7)	10.1 (8.4-12.3)	1.7 (1.4-2.0)	2.3 (2.0-2.8)
2011	6	19.3 (14.0-21.9)	10.4 (8.6-12.3)	1.9 (1.6-2.2)	2.6 (2.2-3.2)
2012	6	20.0 (13.3-25.4)	10.3 (7.7-13.4)	1.9 (1.6-2.4)	2.6 (2.1-3.2)
2013	6	19.9 (14.3-26.0)	10.6 (8.2-13.7)	1.9 (1.6-2.2)	2.5 (2.3-2.8)
2014	6	18.8 (11.8-27.3)	10.8 (8.6-13.7)	1.7 (1.0-2.4)	2.4 (2.2-2.7)
2015	6	16.7 (14.7-20.2)	10.2 (8.2-12.8)	1.7 (1.4-1.8)	2.3 (1.9-2.6)
2016	6	20.0 (14.6-22.8)	10.4 (8.4-13.5)	1.9 (1.6-2.3)	2.5 (2.1-2.9)
2017	6	19.0 (14.0-25.8)	10.3 (8.0-12.7)	1.8 (1.5-2.3)	2.5 (2.2-3.0)
P-Value		0.280	0.028	0.029	0.056
Percent Change					
2003/2004		-10.2%	-31.4%*	26.1%	23.0%
2003/2017		0.8%	-23.6%	25.9%	-1.2%
2004/2005		-4.7%	0.7%	-5.4%	-13.7%
2004/2017		12.3%	11.4%	-0.2%	-19.7%
2012/2017		-5.0%	-0.8%	-4.6%	-6.4%
2013/2017		-4.5%	-2.8%	-2.3%	-2.3%
2014/2017		1.3%	-4.8%	6.0%	1.4%
2015/2017		13.7%	1.1%	11.4%	7.7%
2016/2017		-5.0%	-1.4%	-3.8%	-3.0%

Conclusions

Within this report, we presented physical and biological monitoring from four stream restoration projects on three separate streams ranging from two to fifteen 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 stream banks, increase the quantity of pool-type habitats, and increase habitat complexity. Each of the three streams had generally similar stream channel types (Rosgen 1996), except for Therriault Creek. These streams did however differ in discharge capacity. We acknowledge that successful revegetation strategies are vitally important to achieve the long-term objectives of such restoration activities. Therefore, efforts since the initial channel restoration have been focused primarily on restoring a functioning riparian community that will ultimately sustain streambank stability and diverse habitat for fish, especially on Therriault Creek.

These restoration projects unequivocally changed the pattern, profile and dimension of the streams within the project areas. Within the riffle habitats several conditions were generally evident for each restoration project. We were generally able to demonstrate significant increases in mean bankfull depth and a decrease in stream bankfull width. Furthermore, changes in channel dimensions were generally less than 15% annually. Pool-type habitats generally changed more so than riffle habitats after construction. All the restoration projects presented 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 projects after construction. However, several of the project areas have been impacted by large flood events, which have resulted in a slight annual loss of the total number of pools, and mean pool depth, and increase in riffle width, but despite these changes, the stream channel dimensions within the project areas are still maintained over 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 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. Young, Therriault, and Libby 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. In fact, when such failures occur, one may argue that the restoration efforts were likely focused at an inappropriate scale.

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, and the ecological processes that degraded the habitat must be altered. All the restoration projects discussed in this report have sustained the changes through time, with almost every metric of habitat quantity and quality remaining substantially higher several years after project completion. 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 ultimately be the glue that holds these projects together in the long-term. The monitoring data for the revegetation efforts on Therriault Creek (Geum Environmental Consulting 2007b; 2008c; 2009d; 2010; 2011) indicate these efforts are succeeding at reestablishing a healthy riparian community. 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 Creek project. These riparian restoration efforts will restore natural processes that create and maintain salmonid habitat, which will increase the long-term sustainability and potential for project success (Roni et al. 2002).

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 (Roni et al. 2002; 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). Many restoration efforts do not monitor the fish population response to the habitat manipulations. This project does monitor fish populations, but the results often lacked power to detect significant changes even when the BACI design was utilized. The clear lack of statistical power in most cases resulted from a limited pre-restoration time series coupled with high annual variability at a site. The high annual variability suggests that other physical factors that we did no measure may be important factors that influence population abundance at these sites.

The population level response within the restoration project areas had differing responses, making generalizing between projects difficult. The Young Creek Restoration Project increased the abundance of brook trout, but westslope cutthroat trout abundance decreased. In this situation, it seems likely that ecological interactions between the two species are potentially confounding the results of the improved habitat conditions. Within the two restoration efforts in upper Libby Creek we were unable to demonstrate an increase in redband trout abundance after the restoration efforts. These results were probably influenced by the large flood event that occurred shortly after the restoration work. Results were similar for the

Therriault Creek Restoration Project even though this stream has not experienced a substantial flood event since the project was completed.

Due to the large annual variability in salmonid abundance, substantial pre- and post-monitoring is required to thoroughly evaluate the fish community response to habitat manipulation. Ten or more years of monitoring is often required to detect biological responses to restoration efforts (Bisson et al. 1992; Reeves et al. 1997). In many cases, our pre-project (baseline) monitoring is likely limiting our abilities to make rigorous evaluations. Binns and Remmick (1994) also note that a minimum of several years of pretreatment data is also required 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 up to 5-8 years. Given these relatively long life cycles, the high disturbance regimes, the relatively inherently low productivity levels within many of these streams, and the high annual variability in salmonid abundances within many of the restored streams where the work was completed, it seems likely that recovery will be a lengthy process. We are however confident that the physical changes to the habitat, and long-term modifications to the riparian communities that will create and sustain ecological processes will translate into real and substantial increases at the local population levels, but that these changes may take many years to realize, and a continued monitoring program will be required to detect these changes through time.

Chapter 2: Monitoring for the Kootenai River Nutrient Addition Project

This chapter includes the following work elements:

G: Conduct Fish Sampling in the Kootenai River (Yaak Section): Data Collection (Contracts 77012 and 76916),

J: Analyze and interpret Libby Mitigation physical and biologic data (Contracts 77012 and 76916).

Introduction

Libby Dam and the reservoir behind it (Lake Koocanusa), traps and retains approximately 63% of the total phosphorus (P) and 25% of total nitrogen (N) that enters the reservoir (Woods and Falter 1982; Snyder and Minshall 1996). High water retention time within the reservoir is due to low current velocity, which provides the opportunity for nutrients bind to sediments and precipitate out of solution (Snyder and Minshall 1996), making them unavailable to organisms in the river below the dam. In the Idaho portion of the Kootenai River, the diminished nutrients have reduced primary production, which may be a possible contributing factor to poor sport fish production over the past two decades (Ross 2013).

Idaho Fish and Game (IDF&G; Project 198806500) and the Kootenai Tribe of Idaho (KTOI; Project 199504900) collaborate on a joint project that annually adds liquid phosphate fertilizer (10-34-0) to the river since 2005. This project is called the Kootenai River Ecosystem Project. During the first year of the project, the river was dosed to achieve a target concentration of 1.5 µg/l of phosphate. However, in subsequent years the annual target concentration has been 3.0 µg/l (Ross 2013). Nitrogen availability is seldom limiting in the Kootenai River, but when it is limited, it generally occurs during the later summer months. Nitrate fertilizer (32-0-0) has only been added once since the nutrient addition project began.

The ultimate goal of the Kootenai River Ecosystem Project is to restore the fish assemblage in the Idaho reach of the Kootenai River to pre-Libby Dam densities and improve angler success (Ross 2013). However, the Kootenai River Ecosystem Project was designed to take a more ecosystem-based approach to rehabilitating Kootenai River fisheries. IDF&G and the KTOI designed the Kootenai River Ecosystem Project to aid recovery of fish populations at the entire ecosystem level. The addition of nutrients to the Idaho portion of the Kootenai River is intended to stimulate lower trophic production which may ultimately increase the abundance and growth of target fish species including trout, kokanee, mountain whitefish, burbot, and white sturgeon (Ross 2013).

MFWP collaborates and participates in the effort to restore productivity to the Idaho portion of the Kootenai River by participating in planning meetings and conducting field monitoring at two control sites (KR10 and KR10.5) located on the Kootenai River. The KR10 site is located approximately 6 river miles upstream from the fertilization addition site located at the Montana/Idaho border (River mile [RM] 171.4). However, beginning in 2015, the working group collectively decided that an additional control site was needed to improve future statistical power to detect differences in fish abundance, biomass, condition and growth for future experimental treatments. Therefore, in 2015, we added a second control site in the Montana portion of the Kootenai River. We coordinate closely with Idaho Department of Fish and Game and Kootenai Tribe of Idaho project personnel in our annual monitoring, adhering to the sampling protocols established by our three agencies (Ross 2013). This chapter describes the results of the monitoring activities at that the control site (KR10; RM 177) which has been sampled annually since 2002 and an additional control site that was added to the biomonitoring program in 2015 (KR 10.5; RM 186.2). All information collected by MFWP efforts are shared with IDF&G and KTOI.

Methods

Protocol Title: KRRFM- Ecosystem Restoration Fish Biomonitoring (1988-065-00) v1.0

Protocol Link: http://www.monitoringmethods.org/Protocol/Details/554

Protocol Summary: The Kootenai River basin has been impacted by many anthropogenic activities (e.g., agriculture, mining, land use practices, and the construction and operation of Libby Dam), all of which have affected the ecosystem and led to declines in resident fish populations. Libby Dam has significantly altered the flow regimes and channel morphology of the Kootenai River since it was constructed in the early 1970s, and it has depleted nutrients and caused a decline in primary productivity in the Idaho portion of the river (Woods and Falter 1982; Snyder and Minshall 1996). By the 1990s, this reduction in productivity translated to a two- to four-fold decrease in the number of mountain whitefish, compared to numbers present in 1980-81 (Partridge 1983; Paragamian 1990).

Increases in primary production have been successfully facilitated through the addition of inorganic P and N in other aquatic ecosystems (Ashley et al. 1999), which in turn has been successful in recovering wild fish populations. It was proposed that increases in primary production through nutrient restoration could be used to stimulate fish production in the Kootenai River from bottom up trophic cascades (Snyder and Minshall 1996).

The Kootenai River Ecosystem Project was designed to take an ecosystem-based approach to rehabilitating the fish populations in the Kootenai River. Whereas, past fisheries management programs on the Kootenai River have focused on recovering single species, this project was designed to support recovery of fish populations utilizing an ecosystem-based strategy (as opposed to simply treating the symptoms of degrading stocks). The addition of nutrients to this ultraoligotrophic system was hypothesized to stimulate production in the nutrient-depleted food web and reverse the downward trends in populations of trout, kokanee, mountain whitefish, burbot *Lota lota*, white sturgeon *Acipenser transmontanus*, as well as other species. This report summarizes results specific to fish populations. Results relative to changes in primary productivity and macroinvertebrate communities will be reported by the Kootenai Tribe of Idaho.

Electrofishing catch per unit effort (CPUE) at the Yaak reach (KR10) of the Kootenai River has been conducted in September annually since 2002, and sampling of the Troy reach (KR10.5) began in 2015. Both sites are used to index relative abundance of all fish species encountered and to determine growth and age of trout and mountain whitefish. These data are intended to document trends in the fish community over time, and serve as control sites for the Kootenai River that does not receive additional nutrient enhancement. These study reaches were sampled during night time hours using a jet boat equipped with a Coffelt VVP-15 rectifier typically operated with an electrical output ranging from 200-350 volts at 5-8 amps powered by a 5,000-watt gasoline powered generator. The sampling crew consisted of a driver and two netters. All fish species, regardless of size, were netted to get a representative sample of the fish community at each site. We divided the left and right banks at each site into three equal subsections of 333 m with 150 m separating each (six subsections total). We made a single pass through each subsection, starting with lowest subsections first to ensure no fish drifted into areas not yet sampled. After each subsection was electrofished, the elapsed sampling time was recorded and captured fish were processed. We anesthetized, identified to species, measured (total length [TL], mm), enumerated, and weighed (g) all fish captured. Scales from a subsample of mountain whitefish, rainbow, and cutthroat trout were collected (10 fish in each 10-mm class interval) for aging. Scales were pressed onto cellulose acetate slides and sent to Idaho Department of Fish and Game personnel for age determination. We estimated CPUE in each study reach by species for each of the six subsections by dividing the total catch (by species) by the number of minutes of electrofishing effort. We tested for significant differences in CPUE by species between years (KR10 only) using a repeated measures analysis of variance procedure, and we evaluated trends in CPUE by species through time using linear regression. We calculated Fulton's condition factor (K), and mean length by species each year, and compared mean values across years using ANOVA. Fulton's condition factor is calculated using the following formula: $K = (W/L^3) \times 10^5$, where W is the weight of the fish in grams, L is the length in millimeters, and 10^5 is a constant used for scaling purposes. Multiple comparisons were performed using the Tukey test at alpha = 0.05. All statistical analyzes were performed using R (R Core Team 2018).

Results

A summary of mean catch per unit of effort for the six most commonly observed fish species at the KR10 site is presented in Table 2-1. Rainbow trout, northern pikeminnow, and Columbia River chub catch rates at this site have exhibited a significant positive trend since 2002 (Figure 2-1, 2-2, and 2-3 respectively). Catch rates for mountain whitefish, coarse scale suckers and redside shiners have been variable, but not statistically differed from a stable trend ($P > 0.05$).

Table 2-1. The mean catch per unit of effort (fish per minute) for the six most common fish species captured via jetboat electrofishing at KR10 from 2002 to 2018.

Year	Fish Species: Rainbow Trout	Mountain Whitefish	Columbia River Chub	Coarse Scale Sucker	Northern Pikeminnow	Redside Shiner
2002	0.272	1.538	0.047	0.365	0.095	0.316
2003	0.279	2.076	0.032	0.403	0.146	0.254
2004	0.387	1.680	0.131	0.253	0.152	0.027
2005	0.458	3.690	0.050	0.774	0.119	0.027
2006	0.989	4.475	0.000	0.238	0.099	0.035
2007	0.518	2.570	0.019	0.179	0.108	0.126
2008	0.583	3.843	0.055	0.546	0.170	0.083
2009	0.980	2.294	0.231	0.296	0.193	0.149
2010	1.039	4.082	0.224	0.495	0.150	0.298
2011	0.860	2.446	0.072	0.714	0.232	0.434
2012	0.730	3.553	0.034	0.351	0.193	0.439
2013	0.610	4.033	0.169	0.662	0.330	0.322
2014*	0.655	1.687	0.051	0.514	0.246	0.106
2015	1.037	2.132	0.181	0.338	0.174	0.050
2016	1.148	5.164	0.138	0.877	0.115	0.135
2017	1.135	1.948	0.178	0.475	0.476	0.468
2018	0.731	1.255	0.180	0.479	0.397	0.531
Mean	**0.730**	**2.951**	**0.101**	**0.468**	**0.187**	**0.204**

* Fish escaped from one of the six livecars in 2014 prior data collection. Therefore, the mean catch rates in 2014 were calculated for only 5 subsections.

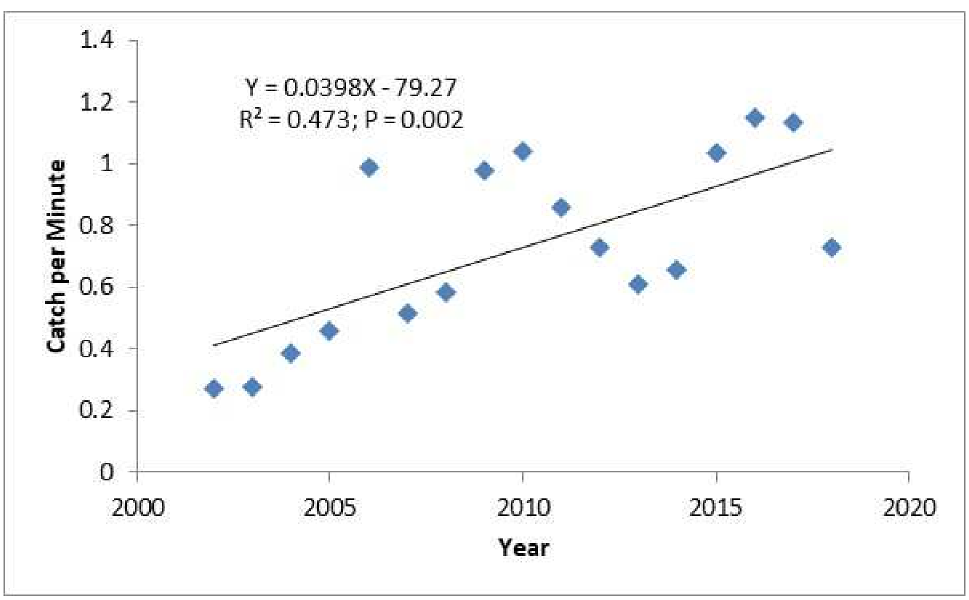


Figure 2-1. Rainbow trout (RBT) catch per unit effort (CPUE) on the Kootenai River at site KR10.

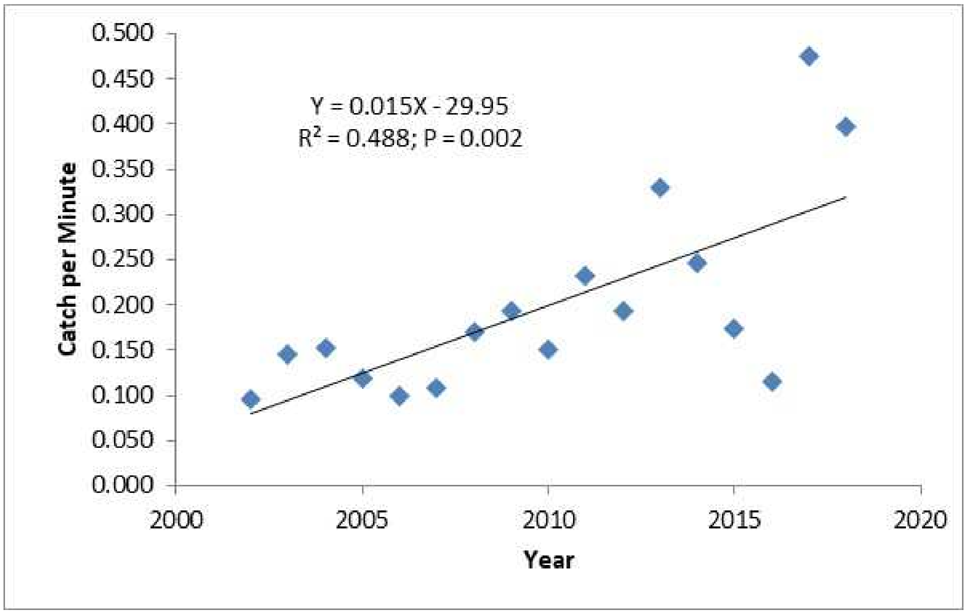


Figure 2-2. Northern pikeminnow (NPM) catch per unit effort (CPUE) on the Kootenai River at site KR10.

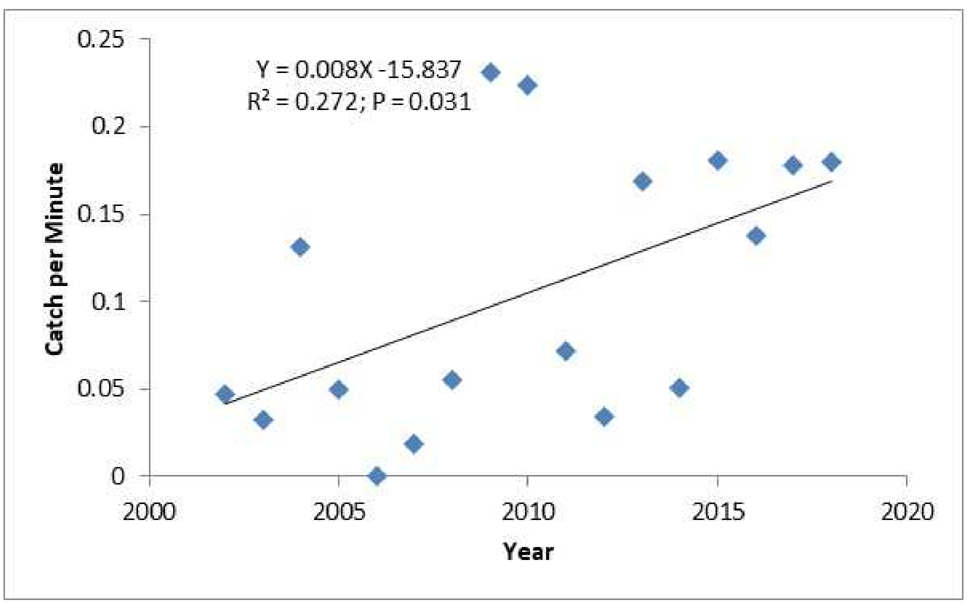


Figure 2-3. Columbia River chub (CRC) catch per unit effort (CPUE) on the Kootenai River at site KR10.

Rainbow trout have consistently been the second most abundant species captured at the KR10 site, ranking the second most abundant species captured during 13 of the past 17 years. We observed an average catch rate of 0.840 rainbow trout per minute at this site in 2018, which represented a slight decrease from the previous year and the overall mean over the period of record. Rainbow trout CPUE at this site differed significantly between years ($p = 0.004$). Multiple comparisons determined that several years differed significantly (Table 2-2). Mean rainbow trout CPUE was lowest in 2002 (0.27 fish/minute) and highest in 2016 (1.50 fish/minute), and averaged 0.730 fish per minute between the sections since 2002. Although, as previously noted, a significant trend exists since 2002. Rainbow trout condition (K) averaged 0.992 since 2002 with condition in 2015 being significantly lower than 2005 and 2008 ($p < 0.05$). All other year-wise comparisons were not significantly different (Table 2-2). We were unable to detect a significant trend in rainbow trout condition ($r^2 = 0.08$; $p = 0.28$), or total length ($r^2 = 0.18$; $p = 0.09$), although mean rainbow trout total length in 2004 was significantly larger than 2008, 2011, 2012, 2015 and 2018 (Table 2-2).

Table 2-2. Mean and standard deviation (S.D.) of rainbow trout catch per unit effort (CPUE), total length and Fulton's condition (K) from 2002-2018 in the KR10 reach. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			Total Length			Fulton's Condition (K)		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2002	0.2724	0.1818	1,2	233.5	71.0		1.001	0.08	
2003	0.2786	0.2667	3,4	264.1	68.2		1.015	0.09	
2004	0.3869	0.3781		288.2	63.2	1,2,3,4	0.929	0.07	
2005	0.4577	0.4595		251.4	57.7		1.053	0.44	1
2006	0.9891	0.3187		246.5	64.3		0.998	0.22	
2007	0.5177	0.2964		250.8	54.1		0.970	0.07	
2008	0.5834	0.3037		218.5	61.8	1	1.043	0.29	2
2009	0.9797	0.7239		242.9	71.3		1.009	0.16	
2010	1.0394	0.5311		250.1	60.8		1.019	0.10	
2011	0.8596	0.4460		237.9	54.6	2	1.014	0.09	
2012	0.7302	0.4531		217.6	67.4	3	0.949	0.11	
2013	0.6097	0.4742		244.6	74.0		0.983	0.10	
2014	0.6547	0.3325		244.1	90.2		1.023	0.10	
2015	1.0371	0.7515		238.8	66.8	4	0.935	0.07	1,2
2016	1.1480	0.6166	1,3	242.3	68.6		1.012	0.10	
2017	1.1350	0.3018	2,4	242.1	58.7		0.962	0.09	
2018	0.7313	0.4887		229.1	70.7	5	0.961	0.11	
Mean	**Trend**			**243.7**	**66.1**		**0.993**	**0.13**	

Mountain whitefish were the most abundant species captured at the KR10 site during all years, averaging 2.95fish/minute since 2002 (Table 2-3). The average catch rate of mountain whitefish at this site in 2018 (1.95 fish/minute) which was 34% lower than the overall average. The observed catch rate of mountain whitefish in 2018 was significantly lower than observed only in 2016. Mean annual mountain whitefish CPUE in 2002 and 2004 were both significantly lower than 2016. All other annual comparisons did not differ significantly (Table 2-3). Mountain whitefish mean length in 2018 was 216.8 mm, which was significantly lower than all years previous. Several other annual pairwise comparisons differed (Table 2-3). Mountain whitefish mean length has exhibited a significant negative trend since 2002 ($r^2 = 0.478$; $p = 0.002$), decreasing on average about 1.5 mm per year. Mountain whitefish condition ranged from a low of 0.827 in 2008 to 1.013 in 2014, and averaged 0.876 in 2018. Mean condition in 2018 was significantly lower than 2002 and 2003, and condition in 2002, 2003 and 2004 was also significantly higher than several other yearly comparisons (Table 2-3). We were unable to distinguish a significant temporal trend in mountain whitefish condition ($r^2 < 0.02$; $p = 0.61$).

On average, Columbia River chub (CRC) are the sixth most abundant species sampled at this site. The repeated measures ANOVA indicated that CRC CPUE did differ significantly between years. ($p = 0.019$). However, the post hoc Tukey test failed to detect any significant pairwise comparisons (Table 2-4). The highest CRC CPUE occurred in 2009 (0.231 fish/minute) and the lowest CPUE occurred in 2006 (0 fish captured). We observed an average CRC CPUE in 2018 of 0.180 fish per minute (Table 2-4), with a weak, but significant positive trend ($r^2 = 0.272$; $p = 0.031$). CRC mean length in 2018 was 249.9 mm which was slightly higher than the mean since 2002 (244.3 mm). We found no evidence that CRC mean length differed between years ($p = 0.201$). There was however a significant difference in condition between years ($p < 0.001$; Table 2-4), with condition in 2018 being significantly lower than 2002, 2008 and 2016. Additionally, other annual pairwise differences also existed (Table 2-4). CRC condition in 2018 averaged 0.790, and the overall mean was 0.866 (Table 2-4).

On average, coarse scale sucker (CSU) have been the third most abundant species sampled at the KR10 site during all years. CSU CPUE in 2018 averaged 0.479 fish/minute, which is nearly equal to the overall mean catch rate since 2002 (0.468 fish/minute; Table 2-1). We found no evidence of trend in CSU CPUE since 2002 ($r^2 = 0.12$; $p = 0.17$). However, CSU CPUE has differed significantly between years ($p = 0.005$; Table 2-5). CSU CPUE was lowest in 2007 (0.18 fish/minute), and highest in 2016 (0.88 fish/minute; Table 2-5). CSU CPUE in 2018 did not differ significantly from other years, although annual differences did exist between other annual pairwise comparisons (Table 2-5). CSU mean length in 2018 was 275.7 mm which was the lowest observed annual mean length and nearly 80 mm shorter than average since 2002. We found no evidence that CSU mean length has exhibited a trend since 2002 ($r^2 = 0.13$; $p = 0.15$), but significant differences existed between years (Table 2-5). CSU mean length in 2018 was significantly lower than 2002, 2004, 1005, 2015 and 2016. Additional annual comparisons also differed significantly (Table 2-5). We found no evidence that CSU mean condition differed by year ($p = 0.149$). CSU condition in 2018 averaged 0.979, which nearly equal to the mean since 2002 (0.992; Table 2-5). However, we observed a weak and marginally significant positive trend in CSU condition since 2002 ($r^2 = 0.17$; $p = 0.10$).

Northern pikeminnow (NPM) on average has been the fifth most abundant species sampled at this site since 2002 (Table 2-1). We observed an average of 0.397 fish/minute at this site in 2018. NPM CPUE has exhibited a significant positive trend since 2002 ($r^2 = 0.488$; $p = 0.002$; Figure 2-2), increasing on average at a rate of about 0.015 fish/minute/year (Figure 2-2). Mean NPM CPUE in 2017 was significantly higher than 2002, 2003, 2005, 2006, 2007, 2010 and 2016, but no other annual comparisons differed (Table 2-6). NPM mean total length in 2018 averaged 202.3 mm which was the shortest mean observed over the period of record and was about 40 mm shorter than the mean since 2002. Despite the relatively small mean length in 2018, it was only significantly different than the mean length measured in 2015 (Table 2-6). However, we found no evidence that mean NPM length or condition have exhibited a significant positive trend since 2002 ($r^2 \sim 0.01$; $p > 0.68$). Mean NPM condition across all years was 0.839, and condition in 2018 was 0.796 (Table 2-6). Mean NPM condition only significantly differed between 2008 and 2011 (Table 2-6).

Table 2-3. Mean and standard deviation (S.D.) of mountain whitefish catch per unit effort, length and Fulton's condition (K) and total length from the KR10 reach 2002-2018. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			Total Length			Fulton's Condition (K)		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2002	1.538	2.56	1	248.8	53.6	1	0.946	0.2877	1-8
2003	2.076	3.96		259.9	51.0	2,17	0.953	0.1178	9-18
2004	1.680	1.85	2	269.2	41.8	3,18-26	0.921	0.0880	19-23
2005	3.690	3.62		259.4	47.6	4,27	0.917	0.0917	
2006	4.475	3.86		259.2	48.7	5,28	0.895	0.0843	1,9
2007	2.570	2.33		256.4	56.0	6,29	0.850	0.0808	2,10,19
2008	3.843	3.03		258.0	42.2	7,30	0.827	0.1502	11,20
2009	2.294	2.71		245.4	44.0	8,18	0.914	0.1897	
2010	4.082	3.36		246.7	37.9	9,19	0.882	0.0777	3,12
2011	2.446	3.06		247.0	39.2	10,20	0.873	0.0728	4,13,21
2012	3.553	3.52		241.9	42.2	11,17,21,27-30	0.814	0.0673	5,14,22
2013	4.033	3.85		244.9	43.3	12,22	0.925	0.0980	
2014	1.607	1.31		254.0	31.9	13,23	1.011	0.0997	6,15,23
2015	2.132	2.03		248.4	47.7	14,24	0.883	0.0884	7,16
2016	5.164	4.51	1,2,3	249.6	43.6	15,25	0.919	0.1003	
2017	1.948	2.35		243.1	41.9	16,26	0.896	0.0792	17
2018	1.255	1.83	3	216.8	67.6	1-16	0.876	0.0792	8,18
Mean	**2.951**	**1.19**		**249.9**	**45.9**		**0.893**	**0.1089**	

Table 2-4. Mean and standard deviation (S.D.) of Columbia River Chub catch per unit effort, length and Fulton's condition (K) and total length from the KR10 reach 2002-2018. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			Total Length			Fulton's Condition (K)		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2002	0.047	0.053	none	251.3	17.8	none	1.127	0.278	1-14
2003	0.032	0.051	none	236.0	14.1	none	0.950	0.099	
2004	0.131	0.320	none	240.2	40.0	none	0.805	0.060	1
2005	0.05	0.092	none	245.0	13.8	none	0.843	0.076	2
2006	0	0		n/a	n/a		n/a	n/a	n/a
2007	0.019	0.030	none	256.0	12.7	none	0.805	0.106	3
2008	0.055	0.135	none	238.0	10.6	none	0.932	0.042	4,15,16
2009	0.231	0.184	none	242.6	21.9	none	0.876	0.056	5,17
2010	0.224	0.303	none	234.4	23.3	none	0.839	0.063	6
2011	0.072	0.072	none	242.6	9.0	none	0.856	0.040	7
2012	0.034	0.056	none	267.0	30.1	none	0.869	0.121	8
2013	0.169	0.137	none	244.9	16.9	none	0.775	0.057	9,15,17,18
2014	0.051	0.075	none	228.3	7.6	none	0.853	0.026	10
2015	0.181	0.279	none	229.8	10.2	none	0.808	0.052	11,19
2016	0.138	0.134	none	255.7	18.7	none	0.927	0.075	12,18,19,20,21
2017	0.178	0.104	none	247.3	19.4	none	0.798	0.082	13,20
2018	0.180	0.078	none	249.1	18.1	none	0.790	0.064	14,16,21
Mean	**0.105**	**0.077**		**244.3**	**18.1**		**0.866**	**0.080**	

Table 2-5. Mean and standard deviation of coarse scale sucker catch per unit effort, length and Fulton's condition (K) and total length from the KR10 site 2002-2018. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			Total Length			Fulton's Condition (K)		
	Mean CPUE	Standard Deviation	Subset(s)	Mean Length (mm)	Standard Deviation	Subset(s)	Mean K	Standard Deviation	Subset(s)
2002	0.365	0.196		397.6	64.5	1	0.924	0.11	none
2003	0.403	0.140		337.4	128.4		0.945	0.13	none
2004	0.253	0.202	1	437.2	74.5	2,3,4	0.926	0.06	none
2005	0.774	0.544		376.9	115.2	5	1.047	0.58	none
2006	0.238	0.184	2	377.0	127.4		0.970	0.08	none
2007	0.179	0.213	3	349.3	146.0		0.978	0.09	none
2008	0.546	0.111		340.6	118.5		0.977	0.15	none
2009	0.296	0.255		311.1	144.5		1.046	0.16	none
2010	0.495	0.502		375.1	107.2		1.023	0.09	none
2011	0.714	0.481		316.9	138.9	2,6	1.009	0.12	none
2012	0.315	0.164		315.8	139.7		0.958	0.08	none
2013	0.662	0.159		325.7	129.9		0.980	0.121	none
2014	0.514	0.320		364.3	109.3		1.084	0.111	none
2015	0.338	0.331		426.1	114.3	7	1.000	0.112	none
2016	0.877	0.466	1,2,3	403.4	123.7	6,8,9	1.023	0.164	none
2017	0.475	0.463		305.0	155.7	3,8	0.983	0.100	none
2018	0.479	0.371		275.7	137.2	1,4,5,7,9	0.979	0.100	none
Mean	**0.468**	**0.198**		**355.0**	**122.1**		**0.991**	**0.139**	

Table 2-6. Mean and standard deviation of northern pikeminnow catch per unit effort, length and Fulton's condition (K) and total length from the KR10 site 2002-2018. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			**Total Length**			**Fulton's Condition (K)**		
	Mean CPUE	**Standard Deviation**	**Subset(s)**	**Mean Length (mm)**	**Standard Deviation**	**Subset(s)**	**Mean K**	**Standard Deviation**	**Subset(s)**
2002	0.095	0.108	1	207.6	36.25		0.810	0.06	
2003	0.146	0.116	2	220.9	59.88		0.857	0.09	
2004	0.152	0.152		244.0	53.46		0.844	0.08	
2005	0.119	0.149	3	290.3	68.86		0.904	0.09	
2006	0.099	0.097	4	239.2	70.72		0.831	0.07	
2007	0.108	0.099	5	236.1	73.30		0.848	0.10	
2008	0.170	0.106		233.3	86.48		0.752	0.22	1
2009	0.193	0.058		233.1	18.11		0.835	0.06	
2010	0.150	0.143	6	259.3	47.24		0.886	0.04	
2011	0.232	0.119		229.3	50.23		0.887	0.06	1
2012	0.193	0.173		244.1	67.0		0.796	0.07	
2013	0.330	0.213		227.9	75.0		0.825	0.06	
2014	0.246	0.211		277.2	40.0		0.872	0.11	
2015	0.174	0.173		290.5	71.2	1,2	0.853	0.232	
2016	0.115	0.085	7	279.3	107.1		0.880	0.062	
2017	0.476	0.237	1-7	204.0	80.0	1	0.795	0.096	
2018	0.397	0.294		202.3	48.5	2	0.796	0.063	
Mean	**0.200**	**0.108**		**242.3**	**62.0**		**0.839**	**0.092**	

The KR10.5 site has been sampled annually from 2015 to 2018 (Table 2-7). Mountain whitefish had the highest relative abundance at this site during all years, averaging 7.2 fish per minute, and has not differed significantly between years ($p = 0.087$). Although, mountain whitefish relative abundance has exhibited a significantly negative trend over the period ($r^2 = 0.93$; $p = 0.03$), decreasing on average by a rate of 3.2 fish per minute per year. Mountain whitefish mean length and condition differed significantly between years ($p < 0.05$). Mountain whitefish mean length averaged 254.1 mm in 2018, which was slightly lower than the overall mean over the period (262.2 mm). Mean length in 2018 was significantly higher than 2015. Other annual pairwise comparisons of mountain whitefish mean length were also significant (Table 2-8). Mountain whitefish mean condition in 2018 was 0.980, which was significantly lower than 2017, but significantly higher than 2015. Mean condition in over the period averaged 0.913 (Table 2-8).

Rainbow trout exhibited the second highest relative abundance during both years (Tables 2-7) and averaged 1.29 fish per minute over the period of record, but did not differ significantly between years ($p = 0.298$). Mean rainbow trout total length in 2018 was 237.5, which was slightly higher than the mean length since 2015 (224.2). Rainbow trout mean length in 2017 was significantly lower than 2016 and 2018 ($p < 0.05$). Mean rainbow trout condition in 2018 was 0.926, which was similar to the overall mean since 2015. The highest average condition of rainbow trout occurred in 2017 and was significantly higher than all other years (Table 2-9).

Mean Coarse scale suckers were the third most abundant species at this site during all years, averaging 0.69 fish per minute, and did not differ significantly between years ($p = 0.784$). Coarse scale sucker relative abundance in 2018 was 0.63 fish per minute. Coarse scale sucker mean length and condition did not differ significantly between years ($p = 0.186$ and 0.049, respectively). Coarse scale sucker mean length and condition in 2018 averaged 413.5 and 0.928, respectively (Table 2-10).

Westslope cutthroat trout relative abundance at this site during the three years has been relatively low, averaging 0.18 fish per minute (Table 2-7), and despite a relatively high abundance in 2016 (0.40 fish per minute) has not differed significantly between years ($p = 0.385$). Mean cutthroat trout relative abundance in 2018 averaged 0.21 fish per minute. Cutthroat trout mean length and condition in 2018 were higher than the average over since 2015, but did not differ significantly between years ($p = 0.963$ and 0.626, respectively; Table 2-11).

Brown trout on average have been the fifth most abundance species at this site since 2015 (Table 2-7). Brown trout relative abundance at this site averaged 0.06 fish per minute and not differing significantly between years ($p = 0.329$). A single brown trout was captured at this site in 2017 and 2018. Brown trout mean length and condition did not differ significantly between years ($p = 0.832$ and 0.326, respectively), averaging 331.0 mm and 0.955, respectively (Table 2-12).

Northern pike minnows have been observed at this site annual except for 2016, for an overall mean abundance of 0.06 fish per minute since 2015 (Table 2-7). We found no evidence that northern pike minnow abundance has differed across years ($p = 0.221$). Northern pike minnow mean length and condition in 2018 averaged 305.8 mm and 0.768, which were both higher than the respective averages since 2015 (Table 2-13). However, we found no evidence to suggest mean length or condition differed between years ($p > 0.05$).

We also captured redside shiners, sculpin, and Columbia River chubs at this site in 2018, with an overall average relative abundance of 0.04, 0.03, and 0.01 fish per minute, respectively.

Table 2-7. The mean catch per unit of effort (fish per minute) for the six most common fish species captured via jetboat electrofishing at KR10.5 from 2015 to 2018.

	Fish Species					
Year	**Rainbow Trout**	**Mountain Whitefish**	**Coarse Scale Sucker**	**Cutthroat Trout**	**Brown Trout**	**Northern Pike Minnow**
2015	1.16	11.66	0.76	0.06	0.08	0.06
2016	1.22	9.90	0.92	0.40	0.09	0
2017	1.10	4.18	0.43	0.06	0.03	0.07
2018	1.68	3.03	0.63	0.21	0.02	0.11
Mean	**1.29**	**7.19**	**0.69**	**0.18**	**0.06**	**0.06**

Table 2-8. Mean and standard deviation (S.D.) of mountain whitefish catch per unit effort (CPUE), total length and Fulton's condition (K) from 2015-2018 in the KR10.5 reach. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			**Total Length**			**Fulton's Condition (K)**		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2015	11.66	5.59	none	215.0	58.9	1,2	0.900	0.069	1,2
2016	9.90	10.74	none	265.0	50.3		0.851	0.078	1,3,4
2017	4.18	2.66	none	254.9	51.3	1	0.991	0.189	2,3,5
2018	3.03	2.44	none	254.1	55.4	2	0.908	0.097	4,5
Mean	**Trend**	**4.23**		**262.2**	**9.7**		**0.913**	**0.058**	

Table 2-9. Mean and standard deviation (S.D.) of rainbow trout catch per unit effort (CPUE), total length and Fulton's condition (K) from 2015-2018 in the KR10.5 reach. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			**Total Length**			**Fulton's Condition (K)**		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2015	1.16	1.04	none	215.0	58.9		0.905	0.062	1
2016	1.22	0.84	none	233.8	75.3	1	0.891	0.083	2
2017	1.10	0.64	none	192.6	69.3	1,2	0.999	0.167	1,2,3
2018	1.68	0.73	none	237.5	70.4	2	0.926	0.065	3
Mean	**1.29**	**0.26**		**224.2**	**31.0**		**0.919**	**0.061**	

Table 2-10. Mean and standard deviation (S.D.) of coarse scale sucker catch per unit effort (CPUE), total length and Fulton's condition (K) from 2015-2018 in the KR10.5 reach. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			**Total Length**			**Fulton's Condition (K)**		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2015	0.76	0.83	none	465.5	49.6	none	0.949	0.081	none
2016	0.92	0.90	none	432.3	81.2	none	0.974	0.097	none
2017	0.43	0.27	none	437.0	89.6	none	1.228	0.910	none
2018	0.63	0.30	none	413.5	121.4	none	0.928	0.090	none
Mean	**0.69**	**0.21**		**437.1**	**21.5**		**1.020**	**0.140**	

Table 2-11. Mean and standard deviation (S.D.) of westslope cutthroat trout catch per unit effort (CPUE), total length and Fulton's condition (K) from 2015-2018 in the KR10.5 reach. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			Total Length			Fulton's Condition (K)		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2015	0.06	0.08	none	251.3	47.9	none	0.925	0.084	none
2016	0.40	0.46	none	267.4	45.8	none	0.952	0.044	none
2017	0.06	0.10	none	262.5	83.2	none	0.986	0.033	none
2018	0.21	0.14	none	270.2	71.2	none	0.965	0.084	none
Mean	**0.18**	**0.16**		**262.5**	**8.3**		**0.957**	**0.025**	

Table 2-12. Mean and standard deviation (S.D.) of brown trout catch per unit effort (CPUE), total length and Fulton's condition (K) from 2015-2018 in the KR10.5 reach. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			Total Length			Fulton's Condition (K)		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2015	0.08	0.16	none	302	183.6	none	0.999	0.027	none
2016	0.09	0.11	none	270.8	102.0	none	0.921	0.092	none
2017	0.03	0.06	none	382	n/a		1.035	n/a	
2018	0.02	0.05	none	369	n/a		0.866	n/a	
Mean	**0.06**	**0.03**		**331.0**	**53.3**		**0.955**	**0.076**	

Table 2-13. Mean and standard deviation (S.D.) of northern pike minnow catch per unit effort (CPUE), total length and Fulton's condition (K) from 2015-2018 in the KR10.5 reach. Years with at least one subset in common were significantly ($p < 0.05$) different.

	CPUE			Total Length			Fulton's Condition (K)		
	Mean CPUE	**S.D.**	**Subsets**	**Mean Length (mm)**	**S.D.**	**Subsets**	**Mean K**	**S.D.**	**Subsets**
2015	0.06	0.09	none	298.7	5.6	none	0.755	0.009	none
2016	0.00	0	none	n/a	n/a		n/a	n/a	
2017	0.07	0.10	none	280.3	58.1	none	0.756	0.198	none
2018	0.11	0.15	none	305.8	39.3	none	0.768	0.040	none
Mean	**0.06**	**0.05**		**294.9**	**13.1**		**0.760**	**0.007**	

Conclusions

The Kootenai River Fertilization project represents the largest river nutrient enhancement effort in the world. The ability to determine the success of this project relies on a rigorous monitoring and evaluation effort that spans all trophic levels (Ross 2013). Additionally, the ability to evaluate the fish community response (and other trophic levels as well) is dependent on a carefully and well thought out monitoring program that utilizes the Before/After/Control/Impact (BACI) design. This monitoring effort at the KR10 site is a critical component used to evaluate the current design, and provides valuable information used to evaluate that larger project. However, the single control site (KR10) used in the BACI design assumes that fish populations at this site are independent of the treatment reaches downstream of the boarder. Unfortunately, the incorporation of an additional control site (KR10.5) will not be helpful for evaluating the current design of the nutrient project, due to the lack of pre-nutrient enhancement data at this site. The advantage of adding an additional control site will be realized if the nutrient enhancement project either stops for a period or modifies the treatment. Nevertheless, continuation of the work reported within this chapter is strongly recommended.

Chapter 3: Therriault Creek Vegetation Restoration

This chapter includes the following work elements:

L and M: Maintain vegetation on Therriault Creek Restoration Project (Contracts 77012 and 76916)

M: Therriault Creek Project supplemental planting (Contract 77012).

Introduction

This chapter summarizes work completed in 2018 at the Therriault Creek restoration project site by MFWP under subcontract with Geum Environmental Consulting. This work represents MFWP's continued commitment to the long-term success of the Therriault Creek Riparian Revegetation Project. As described in previous reports (Geum Environmental 2011; 2010; 2009d; 2008c; 2007b), successfully converting the riparian vegetation along Therriault Creek at the site to a mosaic of native riparian shrubs and trees requires a multi-year, phased approach that includes maintenance and monitoring during the establishment period while vegetation becomes adapted to site conditions. The intent of the initial phase, completed in fall 2007, was to implement a range of treatments based on a detailed evaluation of existing site conditions and ecological processes driving vegetation succession at the site. Effectiveness monitoring of the treatments installed in 2007 was completed in 2008 and 2009. The results were used to determine maintenance needs for 2007 treatments and identify additional revegetation treatments based on how effective the 2007 treatments were at achieving project goals and objectives. A small number of additional revegetation treatments were implemented in September and October 2009 (Phase II). Monitoring continued in 2010 and the results of this and previous monitoring were used to determine treatments for the downstream portion of the project (Phase III). Phase III treatments were implemented during October 2010 and are reported in the Therriault Creek Riparian Revegetation 2010 Implementation and Monitoring Report (Geum Environmental Consulting, Inc. 2010). All treatments were monitored in 2011 and maintenance was completed in 2011 based on the results of 2011 monitoring. Monitoring in 2012 included a 5-year summary assessing the progress of meeting goals and objectives. This summary indicates that the site is trending toward meeting the goals and objectives established for the project and that reduced monitoring could be done in 2013. Work completed at this site in 2013 and 2014 included general observations of all revegetation treatments, photographs of all previous treatments, repeated planting survival estimates, and documentation of all herbicide treatments. A 10-foot tall wildlife exclosure fence was installed in 2013 around the perimeter of all the revegetation planting units in Phase I and most of the units in Phase III. Additional maintenance activities to this fence were completed in 2014. The focus of work completed in 2015 included continuation of effectiveness monitoring to evaluate treatment effectiveness observed since 2008, performing maintenance needs, herbicide application to control noxious weeds, and installation of 100 feet of coir log revegetation treatment at three sites within the Phase I area (Geum Environmental 2015). No site evaluations were completed in 2016 or 2017. However, the following maintenance actions were implemented at the site in 2016 including removal and repair of browse protectors inside and outside the 10-foot wildlife exclosure fence, and repairs to the wildlife exclosure fence. In 2017, Montana FWP completed two maintenance activities at the site including repairs to the wildlife exclosure fence and selective control of yellow toadflax. The purpose of this chapter is to describe the work completed at this site in 2018.

Methods

The Therriault Creek Restoration Project was evaluated by Geum on September 11, 2018. The purpose of the site evaluation was to document existing site conditions, identify revegetation treatment maintenance needs, and determine weed control needs. A detailed description of observations related to the overall condition of the site is provided in a separate document (Geum Environmental 2019). Observations related to revegetation treatment maintenance and weed control are summarized in this chapter. Three types of maintenance were identified during the site evaluation:

- Riparian protection fence removal and relocation
- Riparian protection fence repair
- Browse protector maintenance

A brief description of the observations and maintenance work associated with each of these items is provided below. Geum Environmental Consulting used these observations and their experience and knowledge to develop an updated Riparian Vegetation Management Plan for this site (Geum Environmental 2019a) that MFWP will utilize to identify and prioritize work at this site over the next five years.

Riparian Protection Fence Activities

The riparian protection fence was installed in 2013 and re-built in 2014 and was walked on September 11, 2018. The southern end of the fence had the most damage from wildlife. The fence appeared to block a movement corridor for deer, elk, and black bear as evidenced by a well-defined path immediately north of the livestock fence that was used as the southern boundary of the riparian protection fence. There were tears in the east and west sides of the fence in this location. In addition, a large section of fence was down near the southwest corner near the toe of the hill slope near a dense patch of reed canarygrass. After consultation with Montana FWP, it was decided that relocating the southern end of the fence closer to the Therriault Creek channel (approximately 200 feet north of the current location) would allow animals to move through the area better and relieve pressure on the fence in this area. Geum identified approximately 750 linear feet of fence and reused approximately 350 linear feet of the removed material to be re-used to construct a new section of fence closer to the Therriault Creek channel (Figure 3-1).

The site evaluation conducted in September 2018 identified several locations where the riparian protection fence netting was sagging, ripped, or where deer had created holes underneath the fence. Relocating the fence netting from the outside of the livestock fence to the inside of the livestock fence was a work item in contract between MFWP and Geum Environmental to prevent damage of the fence by livestock in adjacent pastures. Geum did not observe any damage to the fence from cattle rubbing or pushing on it. Along most of the length of the fence the netting is located between the 10-foot t-posts and the livestock fence making removal of the netting difficult. In some locations the netting is already located inside of the livestock fence. Moving the fence inside the livestock fence would also make it more difficult to secure the fence to the ground as it would need to be cut to fit around wooden fence posts. For these reasons, Geum and FWP agreed that moving the netting to inside the livestock fence should not be done. The landowner replaced a section of riparian protection fence netting at the upstream livestock crossing with wire fencing presumably in response to livestock damaging the fence netting. Two solarization plots in the south end of the Site had been planted with dormant willow cuttings in 2014. Recycled fence netting was placed around these plots to protect the cuttings from browse. In 2018, the netting had been damaged and the plots were dominated by reed canarygrass. Geum determined these fences should be removed. Geum recorded locations of fence repair needs and provided the locations of known repairs to the maintenance subcontractor.

Browse Protector Maintenance

Browse protectors were installed on all planted trees and shrubs in 2007 and 2010. Browse protectors were also installed on hundreds of residual shrubs from the original 2005 planting. Browse protectors have been removed, repaired, or expanded every year since 2008 as plants have out grown them or died. After installation of the riparian protection fence in 2013 and 2014, browse protectors were removed from all plants in the 2007 planting area. Some protectors have been removed from the 2010 planting area. All protectors on living shrubs and trees were left on plants outside of the riparian protection fence. Due to the extensive browse observed within the riparian protection fence in 2018, Geum determined that no additional browse protectors should be removed from living plants even within the fence, but these protectors should be repaired and enlarged as needed.

Weed Control

Figure 3-2 shows the distribution of weeds at the site in 2018. A few dense patches of common toadflax were observed near the upstream end of the site. Cover and density of Canada thistle has increased greatly at the site since the last Site-wide herbicide application in 2015 and here are several large, dense infestations of Canada thistle at the site. Within the site itself, cover of Canada thistle was low in 2015, with few to no mature plants remaining. The hayfield south of the Site was treated several times, but also missed in some years allowing the Canada thistle infestations in this area to persist. These infestations have never been controlled completely and were likely a major contributor to reinfestation of the site. Dewatering of this area combined with heavy grazing has created significant disturbance allowing thistle infestations to expand and become denser. Dense patches mostly occur along the channel or expand out from the edges of the channel. Dense patches in the hayfield also expand towards an area south of the channel. Canada thistle is well distributed within planting units, within natural willow expansion areas, and throughout areas consisting of dense sedges and willows.

Cover of reed canarygrass, an aggressive introduced graminoid, has also increased at the site. The density of reed canarygrass in the very downstream portion of the site has always been very high. It appears to have spread to this area through old irrigation ditches and several old swales and channels in this area have allowed it to expand and dominate. In 2018, there were several new patches of reed canarygrass upstream of the large infestation (Figure 3-2). Greater cover on both streambanks and areas further from the channel was observed in 2018 is likely in response to the lack of site-wide herbicide treatment in 2016 and 2017.

Results

Maintenance work was completed between November 13th and 16th, 2018. Table 3-1 summarizes Geum's initial estimate of maintenance work to be completed in 2018 and the actual work completed by Westslope. Figure 3-1 shows the locations of completed maintenance work. A brief description of completed maintenance tasks is included below.

Table 3-1. Summary of estimated and completed maintenance quantities by task.

Task	Unit	Estimated Quantity	Completed Quantity
Riparian protection fence removal	Linear feet	750	800
Riparian protection fence installation	Linear feet	350	350
Riparian protection fence repair	Linear feet	NA[1]	136
Browse protector maintenance	Each	486	302
Trash disposal	Lump sum	1	1

[1] Specific fence repair locations were provided to subcontractor, but total quantity of repairs was not estimated.

Riparian Protection Fence Activities

A total of 800 feet of riparian protection fence was removed and a total of 350 feet was relocated (Table 3-1). Approximately 780 linear feet of existing riparian fence was removed from the south end of the site. The netting and 10-foot t-posts removed from this area were used to reinstall approximately 350 linear feet of new riparian fence in the location shown on Figure 3-1. Newly installed fence was tied into existing riparian fence to re-establish a full exclosure. Riparian fence was also removed from two solarization plots at the downstream end of the project. The exact length of fence is unknown but because removing these areas did not require removal of 10-foot t-posts an additional 20 feet of fence removal was recorded.

A total of 136 linear feet of riparian protection fence was repaired in 2018 (Table 3-1) that included patching small holes and tears in the fence netting, straightening bent or damaged fence posts, re-securing fence netting to existing fence posts, re-securing fence netting to the ground, and patching holes under the fence. Salvaged fence netting and browse protector materials were used to patch holes and rips in and under the fence. Plastic zip ties were used to attach patches and re-secure netting to fence posts. Metal ground staples (18-inch) were used to re-secure fence netting to the ground. The repair locations are identified in Figure 3-1.

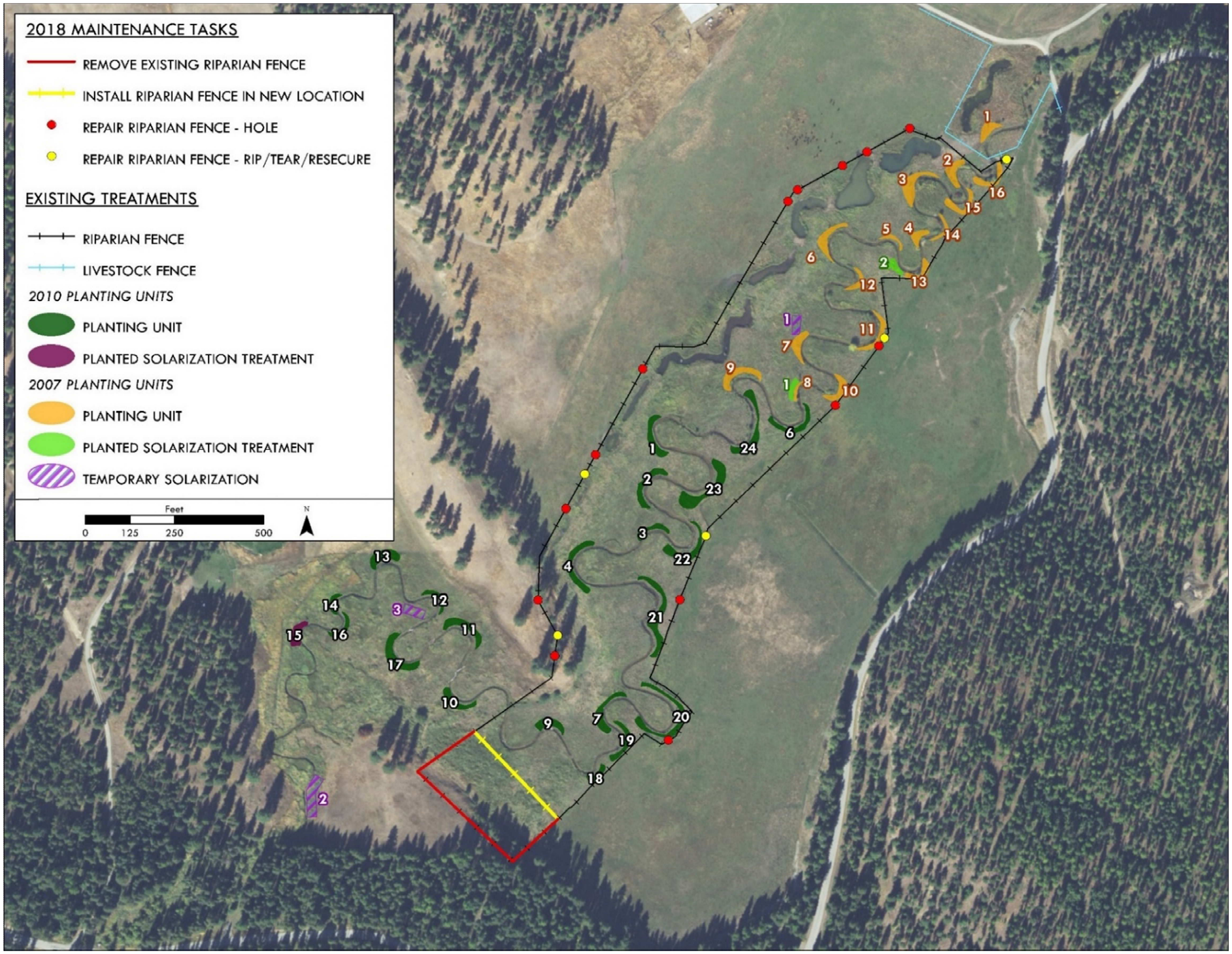


Figure 3-1. Photograph showing the vegetation maintenance locations identified within the Therriault Restoration Project area in 2018.

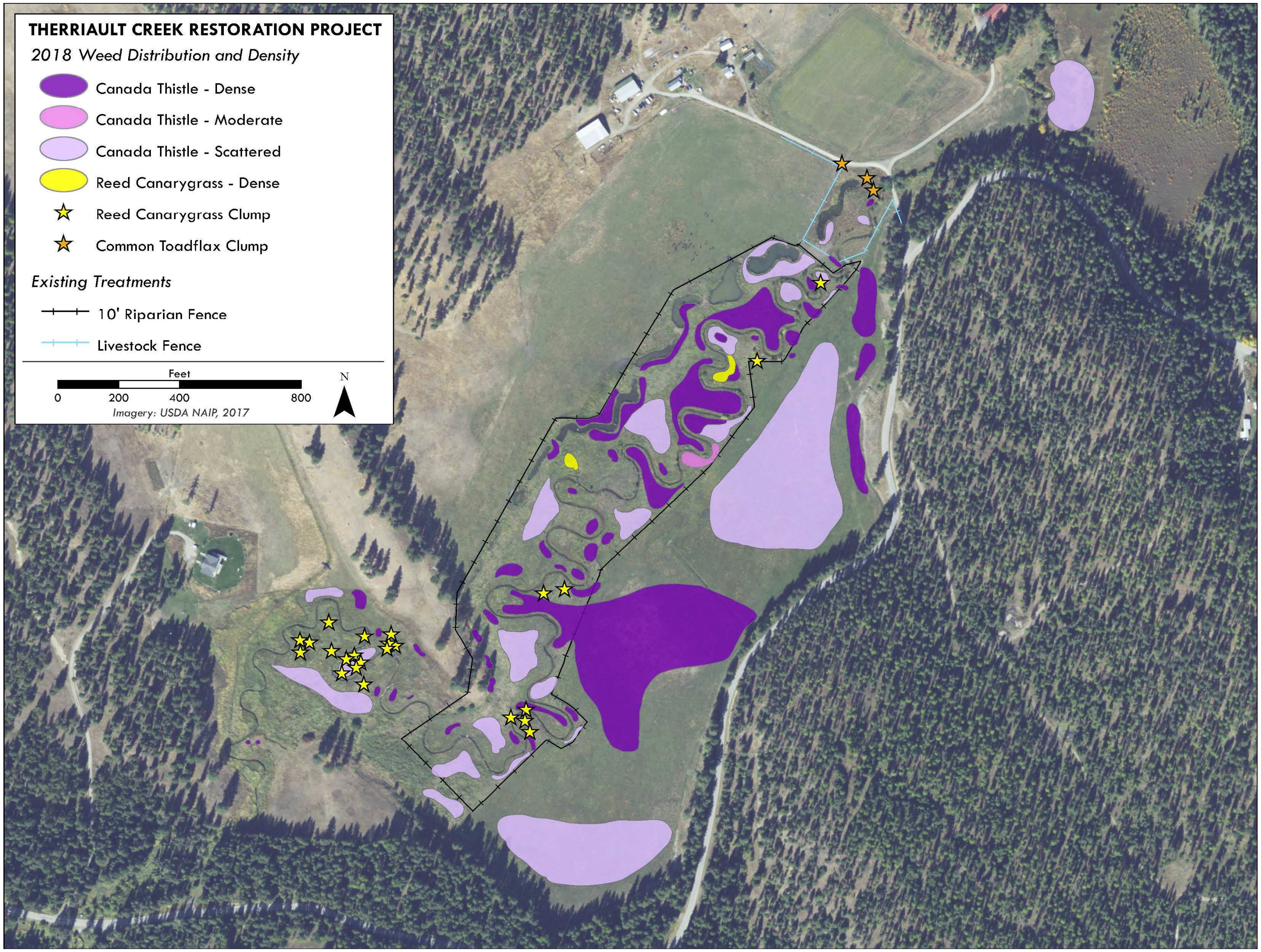


Figure 3-2. Photograph showing the distribution and density of weed species identified within the Therriault Restoration Project area in 2018.

Browse Protector Maintenance

A total of 170 browse protectors were removed and 132 enlarged or installed for a total of 302 browse protectors maintained in 2018 (Table 3-1 and Table 3-2). Browse protector maintenance included removing browse protectors (netting and posts) from dead shrubs, enlarging existing browse protectors around shrubs that had outgrown original browse protectors, and installing new browse protectors on shrubs without browse protection. Removed browse protectors were used to enlarge other protectors or install protectors on plants with no protection. Previous installation and maintenance efforts included the use of wooden stakes to secure browse protectors. However, in 2018 many of the previously used wooden stakes were rotten and required replacement which prevented recycling them. If browse protector repair and maintenance is done in fall 2019, new wooden stakes will be required. FWP has a stockpile of previously removed browse protector nets in Libby, Montana that can be delivered to the site for maintenance work as needed. Table 3-2 summarizes the actual number of browse protectors removed and enlarged/installed within each 2010 planting unit.

Table 3-2. Summary of browse protector maintenance completed by planting unit in 2018.

2010 Planting Unit ID	Browse Protectors Removed	Browse Protectors Enlarged/Installed
1	0	0
2	0	0
3	0	0
4	3	2
6	0	0
7	3	2
9	0	0
10	0	0
11	18	11
12	26	11
13	30	11
14	24	13
15	6	1
16	4	1
17	16	14
18	23	34
19	5	9
20	4	19
21	3	1
22	1	1
23	2	0
24	2	2
Sub-total	**170**	**132**
Total Browse Protectors Maintained		**302**

Weed Control

Figure 3-2 shows weed control needs identified during the site evaluation on September 11, 2018. This guidance was provided to the herbicide applicator, Mountain Valley Plant Management (MVPM). MVPM visited the site in early October 2018. However, there was insufficient fall moisture for Canada thistle to grow so fall treatment would not have been effective. No weed control was completed in 2018. Weed control is planned for spring/summer 2019.

Conclusions

Previous revegetation efforts at this site have increased woody vegetation cover. Woody vegetation is naturally but slowly expanding primarily on low elevation inside meander bends throughout the project area. Previous treatments utilizing dormant willow cuttings have effectively contributed to woody vegetation cover at the site (Geum Environmental Consulting 2019). Overall, the site continues to be dominated by introduced pasture grasses. Lower elevation areas have converted to sedges and native wetland grasses. Dense sedge cover is present in many locations, particularly around the ponds where the abandoned channel was located, on low inside meander bends and streambanks, and low elevation areas of the floodplain. Wildlife browse continues to limit the ability of previously installed vegetation treatments to expand and vigorously grow (Geum Environmental 2019). Therefore, continued maintenance of existing browse protectors and the riparian exclosure fence remain a high priority for this restoration project area. Competition with invasive weeds and aggressive grasses also limits the expansion and growth of woody vegetation within the project area. Additional weed control efforts are therefore a priority for this project.

In general, survival of willow cuttings has been high and the growth of surviving willow cuttings has created dense patches of willow cover along the channel where these treatments were used. However, dense grasses in the adjacent floodplain have limited the expansion of these willows away from the channel. Increasing woody vegetation cover and reducing the risk of bank erosion requires additional active restoration or increased disturbance in the floodplain to open habitats within the pasture grass dominated areas. MFWP has identified three bioengineering treatments in 2019 at this site to support the development of riparian vegetation necessary to restore the ecological function in the riparian and floodplain habitats (Geum Environmental 2019).

Chapter 4: Kootenai Basin Bull Trout Redd Enumeration

This chapter includes the following work elements:

E and F: Monitor trend and status of focal species in MT portion of the Kootenai Basin (Contracts 77012 and 76916).

J: Analyze and interpret Libby Mitigation physical and biologic data (Contracts 77012 and 76916).

Introduction

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

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 via turbine entrainment. 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 estimated that approximately 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; DeHaan and Adams 2011). 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 nodal habitat containing critical over-wintering areas, migratory corridors, and habitat required for reproduction and early rearing.

In the upper Kootenai (above Libby Dam), the threats to bull trout habitat include non-native fish introductions, 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.

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 into Lake Creek then upstream in Keeler Creek. These local populations inhabit areas in the lower Kootenai River and Kootenay Lake during most of the year.

A reliable index of annual spawner escapement is a valuable element of any fisheries monitoring program. These data are frequently used as measures of anticipated production in succeeding generations. Observations during past studies indicate that migratory fish populations in the Kootenai System consistently use the same stream sections for spawning. Similar findings resulted from spawning site surveys in the Flathead and Clark Fork River drainages (Montana Fish, Wildlife & Parks, Kalispell, unpublished file data; MBTSG 1996b). Because of specific spawning habitat requirements, most bull trout spawning is clustered in a small portion of the available habitat, making these areas critical to bull trout production.

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 streams we monitor fine sediment levels in order to evaluate the potential impact of sediment on egg survival (Chapter 5).

Methods

Protocol Title: MFWP Fish Population Monitoring - Reservoirs v1.0

Protocol Link: http://www.monitoringmethods.org/Protocol/Details/511

Protocol Summary: Monitor trends in abundance (i.e., CPUE), species composition, mean lengths, and condition of the fish communities in large reservoirs and lakes within the Flathead and Kootenai River drainages. Monitoring is completed using sinking and floating gill nets depending on season at standardized locations throughout all or sections of each water body.

Field crews annually monitor the number of spawning sites (redds). These counts provide information on trends in escapement into tributaries that support spawning and rearing bull trout populations in the Kootenai Basin. MFWP conducted basin-wide surveys in the 1990s to determine the proper timing and geographical extent of bull trout spawning surveys. We further identified "index" areas within each tributary, which consistently contain most the spawning sites within each tributary, and are counted annually (Hoffman et al. 2002). MFWP conducts bull trout redd surveys in October and November after bull trout have spawned in the Wigwam River and West Fisher, Grave, Quartz, Bear (a tributary to Libby Creek), Keeler, Pipe, and O'Brien creeks. Personnel from the British Columbia Ministry conducted redd counts on the Wigwam River and associated tributaries. Observers visually identify and enumerate redds by the presence of a pit or depression and associated tail area of disturbed gravel. 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.

Results

Grave Creek

MFWP counted redds in Grave Creek (including Blue Sky and Clarence creeks) for the first time in 1983, as well as in 1984, 1985, and annually since 1993. 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. We counted 100 bull trout redds in Grave Creek in 2018, which is nearly identical to the ten-year mean (99.8). However, we were unable to complete bull trout redd counts in Grave Creek in 2016 due to heavy rain and increased stream discharge during most of October. Bull trout redd counts began increasing in about 1998, which coincided with the fishing restrictions associated with Federal listing as an Endangered Species. Counts peaked at 245 redds in 2003, and gradually decreased through 2011 and then either increased slightly or stabilized.

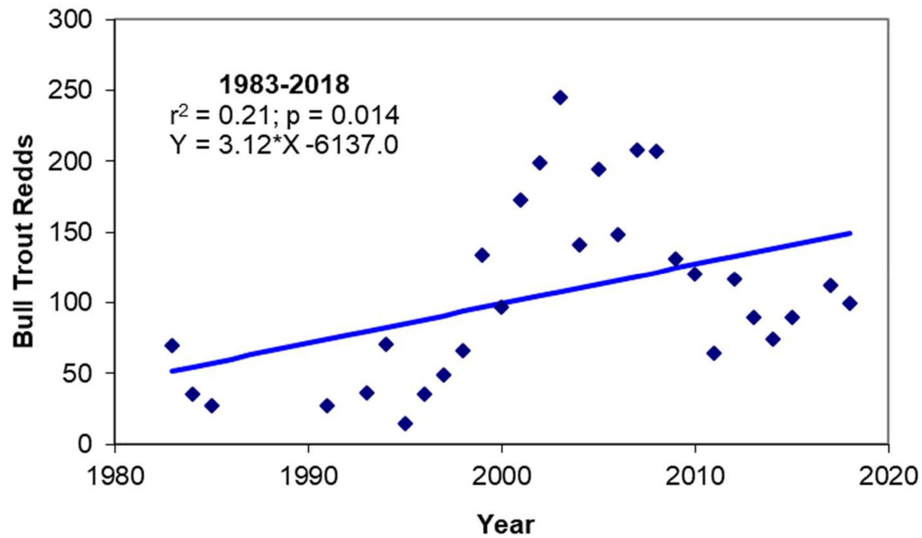


Figure 4-1. Bull trout redd counts and trend analysis for Grave Creek (including Clarence and Blue Sky creeks) 1983-2018.

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 2018, there were a total of 1,410 bull trout redds counted in the Wigwam River, that included two redds within the Montana portion of the watershed (Figure 4-2), which is slightly below the ten-year average (1,433). The trend in bull trout redd counts since 1995 continues to represent a significant positive relationship. Bull trout redd counts have increased on average by about 36 redds per year since 1995. The peak count occurring in 2006 at 2,298 redds, and then decreased for several years, but has been relatively increasing or steady for the past several years (Table 4-1).

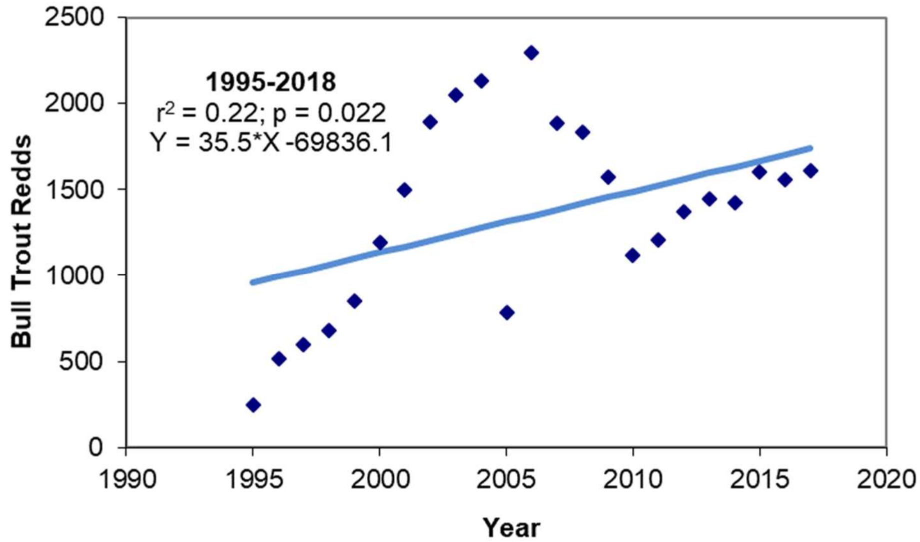


Figure 4-2. Bull trout redd counts and trend analysis for the Wigwam River (including Bighorn, Desolation, and Lodgepole creeks) 1995-2018.

Table 4-1. Bull trout redd counts in eight spawning tributaries of the Kootenai River.

	Stream							
Year	Grave Creek	Wigwam River	Quartz Creek	Pipe Creek	Bear Creek	O'Brien Creek	West Fisher Creek	Keeler Creek
1990			76	6				
1991	27		77	5		25		
1992			17	11		24		
1993	36		89	6		6	2	
1994	71		64	7		7	0	
1995	15	247	66	5	6	22	3	
1996	35	512	47	17	10	12	4	74
1997	49	598	69	26	13	36	0	59
1998	66	679	105	34	22	47	8	92
1999	134	849	102	36	36	37	18	99
2000	97	1195	91	30	23	34	23	90
2001	173	1496	154	6	4	47	1	13
2002	199	1892	62	11	17	45	1	102
2003	245	2053	55	10	14	46	1	87
2004	141	2133	49	8	6	51	21	126
2005	194	785	71	2	3	81	27	186
2006	148	2298	51	6	14	65	4	142
2007	208	1883	35	0	9	77	18	84
2008	207	1833	46	4	14	79	6	62
2009	131	1575	31	9	6	40	8	24
2010	120	1118	39	16	8	27	12	45
2011	64	1206	37	2	3	32	3	29
2012	117	1370	18	12	4	18	5	71
2013	90	1447	14	8	8	35	4	33
2014	74	1427	24	8	11	34	14	34
2015	90	1602	22	0	7	22	4	18
2016		1561	16	0	4	35		
2017	102	1612	27	2	1	35	8	28
2018	100	1410	13	8	3	34	4	18
Mean	**112.8**	**1365.9**	**54.0**	**10.2**	**10.3**	**37.6**	**8.0**	**68.9**
10 Yr. Mean	**98.7**	**1432.8**	**24.1**	**6.5**	**5.5**	**31.2**	**6.9**	**33.3**

Quartz Creek

Bull trout redd counts in Quartz Creek since 1990 have exhibited a significant negative trend (Figure 4-3; $r^2 = 0.38$; $p = 0.0004$), decreasing on average by about 2.4 redds per year since 1990. We observed a total of 13 redds in Quartz and West Fork Quartz creeks in 2018 (Table 4-1) which is the lowest count over the period of record.

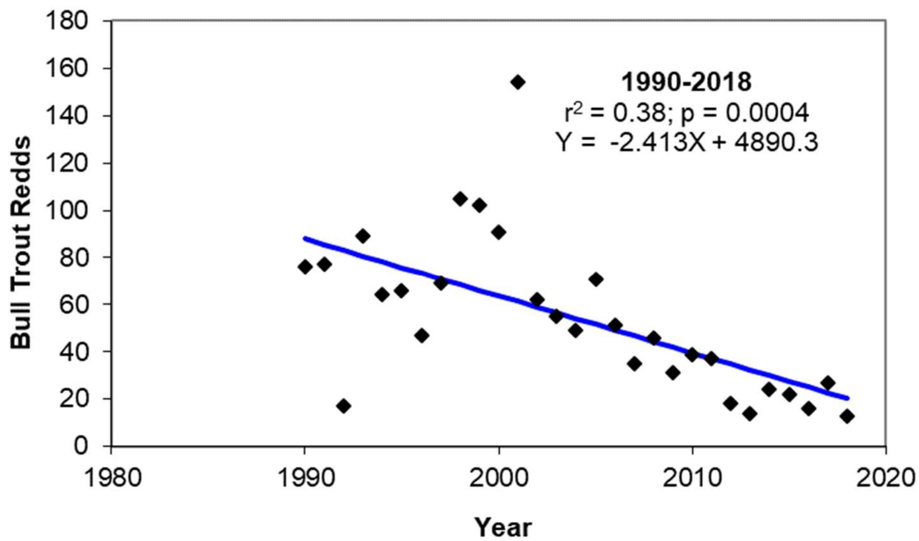


Figure 4-3. Bull trout redd counts, and trend (blue line) for Quartz Creek (including West Fork Quartz) 1990-2018.

Pipe Creek

Bull trout redd counts in Pipe Creek peaked in 1999 at 36 redds. Redd numbers have generally decreased over the next several years, eventually dropping to zero in 2007, and again in 2015 and 2016. We observed eight redds in Pipe Creek in 2018, which was slightly lower than the mean over the period of record (10.2 redds). Despite the relatively low counts observed over the past several years in Pipe Creek (Table 4-1), we were unable to determine a significant overall general trend of bull trout redds in Pipe Creek during the period of record (1990-2018; Figure 4-4).

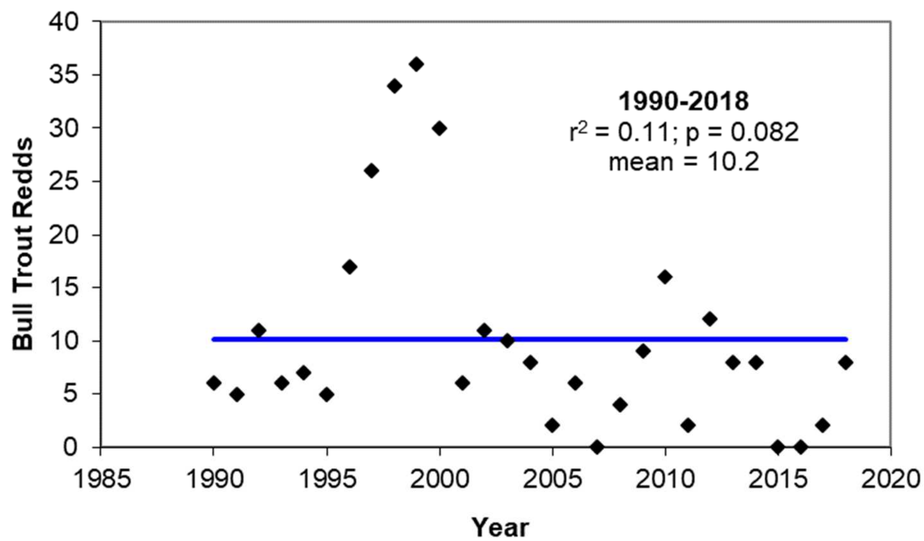


Figure 4-4. Bull trout redd counts, trend analysis, and mean (blue line) for Pipe Creek 1990-2018.

Bear Creek

Bear Creek bull trout redd counts have been variable during the period of record (1995-2018; Figure 4-5), exhibiting a weak, but significant negative trend (r^2 = 0.28; p = 0.007), decreasing on average about 0.6 redds per year. We observed three redds in Bear Creek in 2018, which is slightly more than about half of the ten-year mean (5.5 redds; Table 4-1).

O'Brien Creek

The number of bull trout redds we have observed in O'Brien Creek since 1991 generally increased through 2007 and then exhibited a declining trend since (Figure 4-5). However, over the period of record we were unable to distinguish a significant trend (r^2 = 0.06; p = 0.22). The number of redds in O'Brien Creek peaked in 2005 at 85 redds (Table 4-1). We observed 34 redds in O'Brien Creek in 2018, which similar to the average over the period of record and the ten-year average (37.6 and 31.2 redds respectively; Table 4-1).

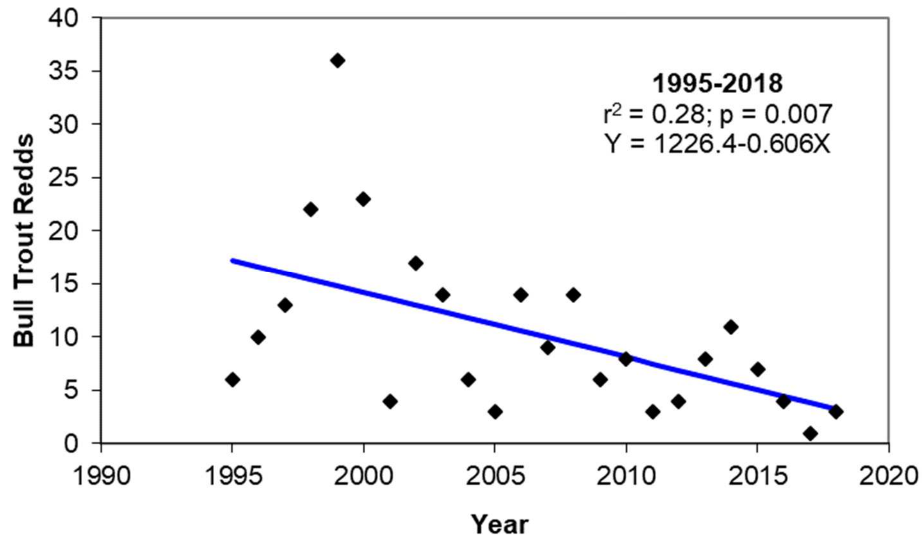


Figure 4-5. Bull trout redd counts, trend analysis, and mean (blue line) in Bear Creek, a tributary to Libby Creek, 1995-2018.

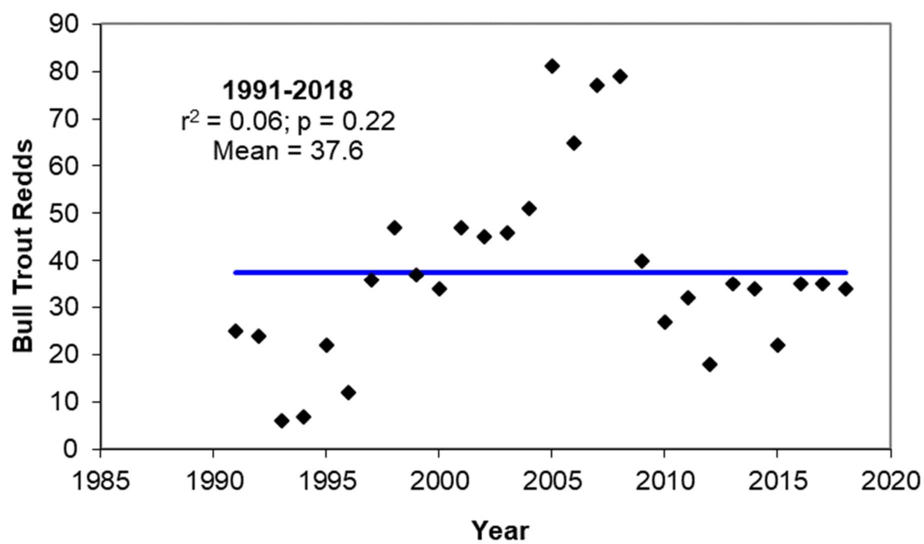


Figure 4-6. Bull trout redd counts and mean (blue line) in O'Brien Creek 1991-2018.

West Fisher Creek

We began counting bull trout redds in West Fisher Creek in 1993 and have attempted to count them annually since. Redd count numbers peaked at 27 redds in 2005, but have generally declined since. We counted a total of 4 redds in West Fisher Creek in 2018, which is half of the average over the period of record (8.0 redds). We were unable to determine a significant trend over the period of record ($r^2 = 0.01$; $p = 0.65$; Figure 4-7).

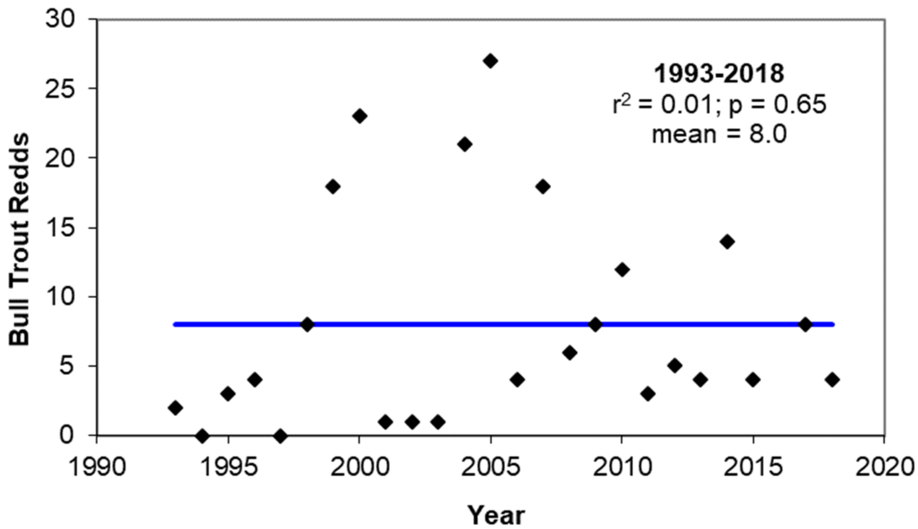


Figure 4-7. Bull trout redd counts and mean (blue line) in West Fisher Creek 1993-2018.

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. Bull trout redd counts in Keeler Creek were started in 1996. Redd counts peaked at 186 redds in 2005 (Figure 4-8). We counted 18 redds in 2015, 2017 and 2018, which was the lowest count over the period of record. We were unable to count redds in Keeler Creek in 2016. Bull trout redds have significantly decreased at an average rate of 3.3 redds per year since 1996 ($r^2 = 0.25$; $p = 0.02$; Figure 4-8).

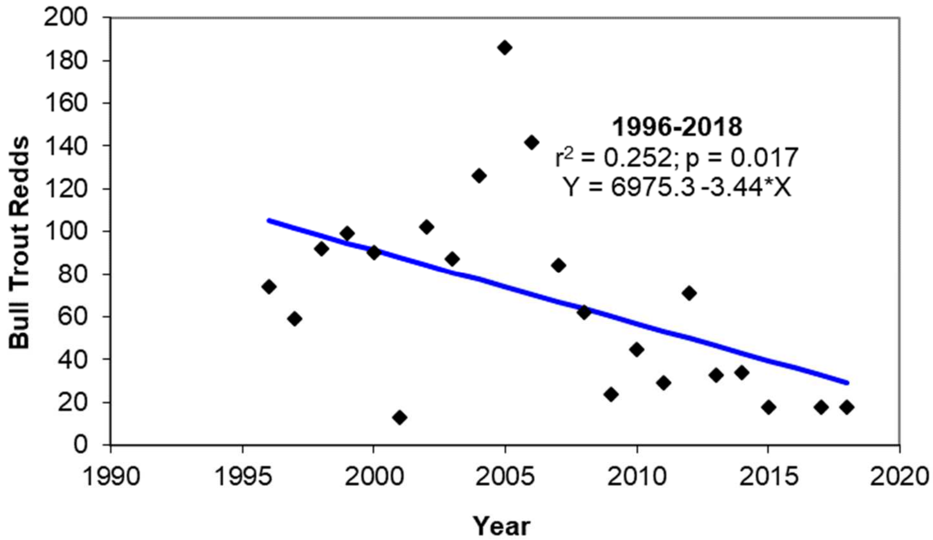


Figure 4-8. Bull trout redd counts and trend (blue line) in Keeler Creek 1996-2018.

Conclusions

The annual bull trout redd counts in the Montana portion of the Kootenai Basin provide reliable indices of annual spawner escapement for each of the core bull trout streams. These data are useful for assessing the trend of these populations and evaluate progress of recovery efforts. Bull trout redd counts in the Wigwam River and Grave Creek located upstream of Libby Dam have decreased from the observed peak counts that occurred during the early/mid 2000s, but remain substantially higher than those core populations residing downstream of Libby Dam, and probably continue to meet recovery goals set forth by the USFWS (2002).

Kootenai River fluvial bull trout populations located downstream of Libby Dam exhibit a high degree of annual variability and are likely influenced by entrainment at Libby Dam (DeHaan and Adams 2011). Bull trout redd counts in three of the four tributaries located between Libby Dam and Kootenai Falls have declined to fewer than 10 redds, and all the populations within this geographical area have collectively exhibited a declining trend since 1995. If these populations continue their current trajectory the persistence of these populations may be in jeopardy.

Chapter 5: Survival and Growth of Kootenai River Trout

This chapter includes the following work elements:

F: Estimate growth and survival of trout in the Kootenai River (Contract 77012).

J: Analyze and interpret Libby Mitigation physical and biologic data, (Contract 77012) and;

K: Mark rainbow trout and burbot in the Kootenai River below Libby Dam (Contract 77012).

Introduction

Libby Dam was constructed 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. In addition to large scale ecological changes upstream of the dam, substantial ecological changes have also occurred in the Kootenai River downstream of Libby Dam.

Research to determine how operations of Libby Dam affect the river fishery and to suggest ways to lessen these effects began in May 1983. 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 were subsequently further refined (Marotz et al 1999). The Northwest Power and Conservation Council (NPCC) implemented the Mainstem Amendment operations at Libby Dam in October 2009. These hydro-operations were designed to provide a better balance for fishes in the upper and lower portions of the Columbia River basin, and included a reduced summer reservoir drawdown (from previous operations), and stabilized water releases into the Kootenai and Flathead Rivers during the productive summer months to protect aquatic resources in both rivers.

Little information exists related to growth and survival of Kootenai River trout populations below Libby Dam. Operations at Libby Dam since construction have generally been a gradual conversion to conditions that more closely simulate normative river conditions below Libby Dam. Our objectives of the work identified within this section is to estimate relative growth and survival of resident trout in four sections of the Kootenai River over a six-year period and assess of how various biological and physical conditions are affecting relative growth and survival in these four sections of the Kootenai River.

Methods

Protocol Title: MFWP Fish Population Monitoring - Large Rivers v1.0

Protocol Link: http://www.monitoringmethods.org/Protocol/Details/509

Protocol Summary: Population estimates (mark-recapture) will be used to monitor the fish populations in the Kootenai and Flathead River systems to monitor annual trends in population numbers, size structure, species composition, and condition of rainbow trout, westslope cutthroat trout, bull trout, and mountain whitefish.

Montana FWP conducts annual mark recapture populations of rainbow trout in the Kootenai River in four sections including the Libby Dam tailrace section (river mile [RM] 218.2-221.7), the Re-Regulation (Re-reg) section (RM 213.2-215.1), the Flower-Pipe section (RM 201.1-204.0), and Troy section (RM 183.8-186.2) (Sylvester and Stephens 2011). The Libby Dam, Flower-Pipe and Troy sections are completed annually during September and the Re-Regulations section is completed in March.

We captured salmonids using nighttime electrofishing by jet boat to estimate growth and apparent survival of fish in the four sections. We conducted two or three sampling events annually within each river section with each sampling sessions per section separated by approximately 7 days (Table 6-1). During each sampling event, we operated two jet boat electrofishing crews, with each boat crew consisted of a driver and two netters. The electrofishing unit on each boat consisted of a Coffelt model Mark 22 electrofishing unit operating with an electrical output ranging from 200-350 volts of continuous direct current (DC) at 5-8 amps powered by a 5,000-watt gasoline powered generator. Netting crews attempted to net all salmonids encountered. 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 marked each fish with individually numbered 134 (ISO) KHz passive integrated transponder (PIT) tags and an adipose fin clip. PIT tags were inserted with an 8-gauge hypodermic needle into the musculature behind the dorsal fin. All fish with an adipose fin clip were assumed to be previously marked and were interrogated for the presence of a PIT tag, and the code was recorded, and the fish noted as a recaptured fish. However, during 2018 at the Libby Dam, Flower Pipe and Troy sections, fish that were not previously marked with a PIT tag were not tagged, but were given a fin clip that was unique to sampling date.

We estimated annual growth for individual fish marked in each of the four sections during the previous year and recaptured during the current year by subtracting the length and weight at the time of marking from the respective length and weight at the time of recapture. We standardized linear growth by dividing length gain by the number of days between capture events, and we standardized weight growth by dividing the weight gained by the weight of the fish at original capture divided by the number of days between capture events. We performed a linear transformation on the relationship between weight at original capture and the standardized weight gain using the negative inverse of weight at original capture (-1/weight). We evaluated trends in rainbow trout growth (length and weight) between sections using linear regression, and compared the slope and intercepts of the regression lines between sections and years using analysis of covariance. All statistical analyzes were performed using R (R Core Team 2018).

Estimates of annual apparent survival for each section will be estimated using either the robust-design model (Kendall 1999; Kendall et al. 1995, 1997), or the combined robust-multistratum (Brownie et al. 1993; Hestbeck et al. 1991) model in Program MARK (White and Burnham 1999). However, these estimates will not be available until study completion. Program Mark allows investigators to correlate environmental covariates (such as discharge, temperature, etc.) to apparent survival.

Results

We conducted three marking sessions in the Re-regulation section in 2018 during which we marked a total of 864 individual salmonids (Table 5-1), including 861 rainbow trout (99.7%), and three westslope cutthroat trout (0.3%). These statistics did not include the 62 rainbow trout marked on March 5, 2018 that were recaptured on March 9, 2018, or the 91 rainbow trout recaptured on March 19, 2018 which had been marked during the two previous marking sessions. The average length and weight of rainbow trout captured in 2018 was 229.6 mm (range 84-450 mm) and 130.6 g (range 6-1,028 g).

We captured 19 rainbow trout in the Re-regulation section in 2018 that were originally marked in that section in 2017, which represented 2.72% of the total from 2017. We also captured six rainbow trout that were originally marked in 2016, and two rainbow trout that was originally marked in 2015. However, the trout that were marked prior to 2017 were not included in any subsequent growth analysis presented in this report. The average total lengths and weights of rainbow trout originally marked in 2017 and recaptured in 2018 were 218.1 mm (range 112 – 340 mm) and 295.4 mm (range 194 – 368 mm) and 122.2 g (range 13 – 375 g) and 270.8 g (range 52 – 508 g), respectively. The average annual growth of these fish was 77.3 mm (0.21 mm/day) with a range of 23 to 126 mm (0.06 to 0.34 mm/day; Figure 6-1). There was a significant negative relationship between length at capture and annual growth (r^2 = 0.564; p = 0.0002; Figure 5-1). The trend of this relationship has been consistent since we began this study in 2011 (Figure 6-1). The average weight gain of all recaptured rainbow trout was 148.6 g (0.006 g/g/day) with a range of 16 to 271 g (range 0.001 – 0.018 g/g/day; Figure 5-2). However, growth rates (g/g/day) are significantly negatively correlated with the transformed weight at capture (-1/weight) (r^2 = 0.664; p = $2.15*10^{-5}$; Figure 5-2).

We conducted three marking sessions in the Libby Dam section in 2018 during which we marked a total of 2,306 individual salmonids (Table 5-1), including 2,263 rainbow trout (99.7%), 14 westslope cutthroat trout (0.3%), and 29 bull trout. These statistics did not include the 32 rainbow trout marked on August 28, 2018 that were recaptured on September 4, 2018, or the 76 rainbow trout captured on September 4, 2018 which had been marked during the two previous marking sessions. The average length and weight of rainbow trout captured in 2018 was 226.6 mm (range 84-640 mm) and 135.8 g (range 6-3,198 g).

We captured 37 rainbow trout in the Libby Dam section in 2018 that were originally marked in that section in 2017, which represented 0.99% of the total from 2017. We also captured six rainbow trout that were originally marked in 2016, one rainbow trout that was originally marked in 2015 and two rainbow trout that were marked in 2014. However, the trout that were marked prior to 2017 were not included in any subsequent growth analysis presented in this report. The average total lengths and weights of rainbow trout originally marked in 2017 and recaptured in 2018 were 210.1 mm (range 163 – 490 mm) and 285.4 mm (range 242 – 535 mm) and 119.3 g (range 47 – 940 g) and 257.9 g (range 148 – 1,601 g), respectively. The average annual growth of these fish was 75.3 mm (0.21 mm/day) with a range of 34 to 116 mm (0.07 to 0.32 mm/day; Figure 6-1). There was a significant negative relationship between length at capture and annual growth ($r^2 = 0.426$; $p = 2.85*10^{-5}$; Figure 5-3). The trend of this relationship has been consistent since we began this study in 2011. However, this fit in 2018 exhibited a substantial non-linear relationship that could have been improved with a data transformation to linearize the relationship, be we chose not to for the sake of consistency between years. The average weight gain of all recaptured rainbow trout was 138.6 g (0.005 g/g/day) with a range of 34 to 661 g (range 0.001 – 0.010 g/g/day; Figure 5-4). However, growth rates (g/g/day) are significantly negatively correlated with the transformed weight at capture (-1/weight) ($r^2 = 0.774$; $p = 7.2*10^{-12}$; Figure 5-4).

We conducted two marking sessions in the Flower Pipe section in 2018 during which we marked a total of 668 individual salmonids (Table 5-1), including 651rainbow trout (97.5%), 17 westslope cutthroat trout (2.5%), and one bull trout (>0.1%). These statistics did not include the 163 rainbow trout or two cutthroat trout marked on August 29, 2018 that were recaptured on September 5, 2018. The average length and weight of rainbow trout captured in 2018 was 218.8 mm (range 108-446 mm) and 125.3 g (range 18-886 g).

We captured 47 rainbow trout in the Flower Pipe section in 2018 that were originally marked in that section in 2017, which represented 1.25% of the total from 2017. We also captured four rainbow trout that were originally marked in 2016, and two rainbow trout that was originally marked in 2015. However, the trout that were marked prior to 2017 were not included in any subsequent growth analysis presented in this report. The average total lengths and weights of rainbow trout originally marked in 2017 and recaptured in 2018 were 190.0mm (range 132 – 353 mm) and 275.9 mm (range 216 – 394 mm) and 95.2 g (range 23 – 616 g) and 234.4 g (range 117 – 562 g), respectively. The average annual growth of these fish was 83.4 mm (0.23 mm/day) with a range of 18 to 122 mm (0.05 to 0.32 mm/day; Figure 6-1). There was a significant negative relationship between length at capture and annual growth ($r^2 = 0.520$; $p = 1.09*10^{-8}$; Figure 5-5). The average weight gain of all recaptured rainbow trout was 143.3 g (0.007 g/g/day) with a range of 62 to 252 g (range 0.001 – 0.015 g/g/day; Figure 5-6). However, growth rates (g/g/day) are significantly negatively correlated with the transformed weight at capture (-1/weight) ($r^2 = 0.774$; $p = 4.12*10^{-16}$; Figure 5-6).

We conducted two marking sessions in the Troy section in 2018 during which we marked a total of 511 individual salmonids (Table 5-1), including 472 rainbow trout (92.4%), 29 westslope cutthroat trout (5.7%), one bull trout (0.2%), and 9 brown trout (1.8%). These statistics did not include the 26 rainbow trout or the four cutthroat trout marked on August 28, 2018 that were recaptured on September 6, 2018. The average length and weight of rainbow trout captured in 2018 was 253.3 mm (range 85-477 mm) and 185.3 g (range 7-1,153 g).

We captured 25 rainbow trout in the Troy section in 2018 that were originally marked in that section in 2017, which represented 5.05% of the total from 2017. We also captured 11 rainbow trout that were originally marked in 2016, four rainbow trout that was originally marked in 2015, and one rainbow trout from 2014. However, the trout that were marked prior to 2017 were not included in any subsequent growth analysis presented in this report. The average total lengths and weights of rainbow trout originally marked in 2017 and recaptured in 2018 were 273.8 mm (range 177 – 376 mm) and 332.8 mm (range 271 – 435 mm) and 229.1g (range 50 – 535 g) and 382.1 g (range 208 – 823 g), respectively. The average annual growth of these fish was 59.0 mm (0.17 mm/day) with a range of 16 to 106 mm (0.04 to 0.31 mm/day; Figure 6-1). There was a significant negative relationship between length at capture and annual growth ($r^2 = 0.618$; $p = 3.19*10^{-6}$; Figure 5-7). The average weight gain of all recaptured rainbow trout was 153.0 g (0.003 g/g/day) with a range of 27 to 288 g (range 0.0002 – 0.009 g/g/day; Figure 5-8). However, growth rates (g/g/day) are significantly negatively correlated with the transformed weight at capture (-1/weight) ($r^2 = 0.875$; $p = 7.18*10^{-12}$; Figure 5-8).

Table 5-1. The sampling dates for the number of salmonids captured and marked in four sections of the Kootenai River from 2011 to 2018. The number in parentheses is the number of fish recaptured from within the same year.

		Fish Species				
Section	Mark Dates	RBT	WCT	Bull	Brown	Total
Re-regulation	**2011 Total**	**281 (15)**	**3**	**2**	**0**	**286 (15)**
Re-regulation	**2012 Total**	**485 (34)**	**2**	**4**	**0**	**491 (34)**
Re-regulation	**2013 Total**	**745 (98)**	**2**	**16 (1)**	**0**	**763 (99)**
Re-regulation	**2014 Total**	**1,126 (114)**	**12**	**9**	**0**	**1,148 (114)**
Re-regulation	**2015 Total**	**1,242 (215)**	**8**	**6**	**0**	**1,256 (215)**
Re-regulation	**2016 Total**	**1,240 (171)**	**13 (2)**	**3**	**0**	**1,256 (173)**
Re-regulation	**2017 Total**	**698 (59)**	**6**	**0**	**0**	**706 (59)**
Re-regulation	3/5/18	460	2	0	0	462
Re-regulation	3/12/18	226 (62)	1 (0)	0	0	227 (62)
Re-regulation	3/19/18	175 (91)	0	0	0	175 (91)
Re-regulation	**2018 Total**	**861 (153)**	**3 (0)**	**0**	**0**	**864 (153)**
Libby Dam	**2011 Total**	**661 (8)**	**5 (1)**	**33**	**0**	**700 (9)**
Libby Dam	**2012 Total**	**1,165 (50)**	**17**	**76 (3)**	**0**	**1,258 (53)**
Libby Dam	**2013 Total**	**1,401 (55)**	**25**	**76 (4)**	**0**	**1,502 (59)**
Libby Dam	**2014 Total**	**1,222 (26)**	**23 (1)**	**55 (3)**	**0**	**1,300 (30)**
Libby Dam	**2015 Total**	**1,728 (107)**	**29 (1)**	**21**	**0**	**1,778 (108)**
Libby Dam	**2016 Total**	**1,396 (99)**	**9 (1)**	**39 (1)**	**0**	**1,444 (101)**
Libby Dam	**2017 Total**	**3,754 (275)**	**26 (1)**	**24 (4)**	**0**	**3,798 (280)**
Libby Dam	8/27/18	864	7	11	0	882
Libby Dam	9/4/18	665 (32)	5 (0)	11 (0)	0	681 (32)
Libby Dam	9/10/18	734 (76)	2 (2)	7 (1)	0	743 (79)
Libby Dam	**2018 Total**	**2,263 (108)**	**14 (2)**	**29 (1)**	**0**	**2,306 (111)**
Flower-Pipe	**2011 Total**	**1,274 (58)**	**23 (1)**	**3**	**0**	**1,360 (59)**
Flower-Pipe	**2012 Total**	**2,031 (168)**	**28 (2)**	**9**	**0**	**2,068 (170)**
Flower-Pipe	**2013 Total**	**2,060 (117)**	**57 (1)**	**8 (1)**	**0**	**2,125 (119)**
Flower-Pipe	**2014 Total**	**1,898 (119)**	**53 (3)**	**0**	**0**	**1,951 (122)**
Flower-Pipe	**2015 Total**	**2,437 (174)**	**125 (18)**	**1**	**0**	**2,563 (192)**
Flower-Pipe	**2016 Total**	**1,399 (120)**	**70 (4)**	**1**[a]	**0**	**1,470 (124)**
Flower-Pipe	**2017 Total**	**3,853 (618)**	**67 (8)**	**1**	**0**	**1,470 (633)**
Flower-Pipe	8/29/18	1,331	31	1	0	1,363
Flower-Pipe	9/5/19	651 (163)	17 (2)	0	0	668 (165)
Flower-Pipe	**2018 Total**	**1,982 (163)**	**48 (2)**	**1**	**0**	**2,031 (165)**
Troy	**2011 Total**	**339 (23)**	**3**	**6**	**33 (2)**	**381 (25)**
Troy	**2012 Total**	**449 (35)**	**11 (1)**	**6**	**49 (2)**	**517 (38)**
Troy	**2013 Total**	**410 (66)**	**11 (1)**	**6**	**64 (13)**	**491 (80)**
Troy	**2014 Total**	**252 (21)**	**6 (1)**	**1**	**25 (2)**	**284 (24)**
Troy	**2015 Total**	**401 (36)**	**40 (2)**	**4 (1)**	**16 (2)**	**461 (41)**
Troy	**2016 Total**	**369 (52)**	**23 (4)**	**3**	**13 (1)**	**408 (57)**
Troy	**2017 Total**	**495 (55)**	**24 (3)**	**2**	**8 (1)**	**533 (59)**
Troy	8/28/18	246	17	1	3	267
Troy	9/6/18	226 (26)	12 (4)	0	6	244 (30)
Troy	**2018 Total**	**472 (26)**	**29 (4)**	**1 (0)**	**9 (0)**	**511 (30)**

[a]Fish identified visually as a bull trout brook trout hybrid

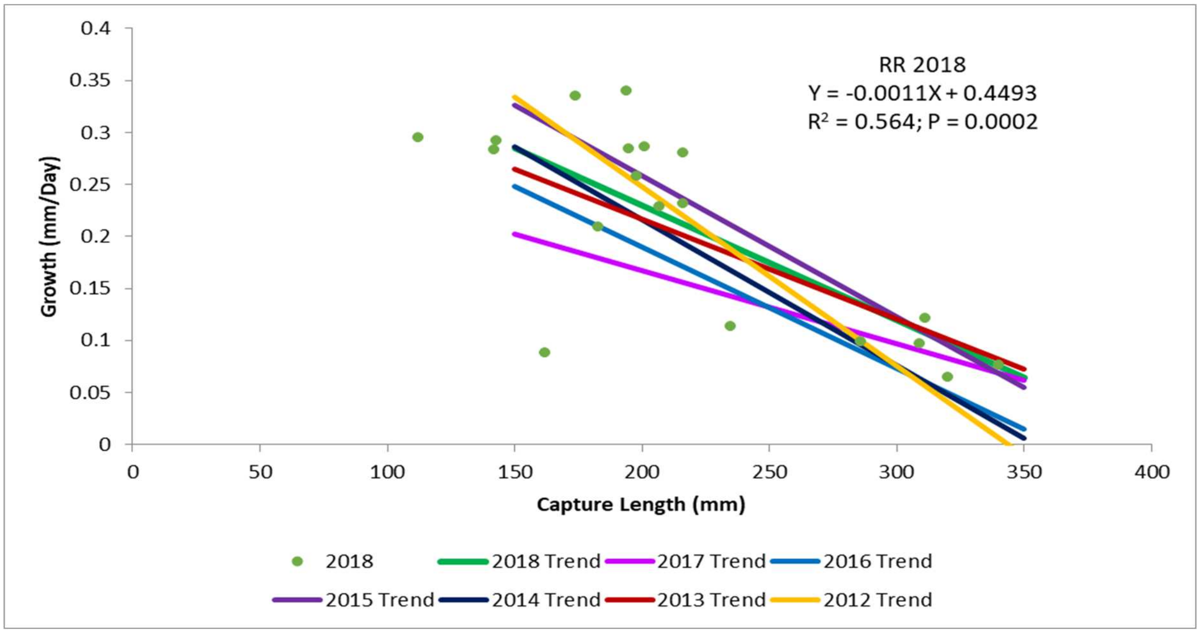


Figure 5-1. Relationship between length at capture and average daily growth (mm per day) of rainbow trout captured in the Re-regulation section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) from 2012-2017 (Dunnigan et al. 2018) are presented for reference purposes.

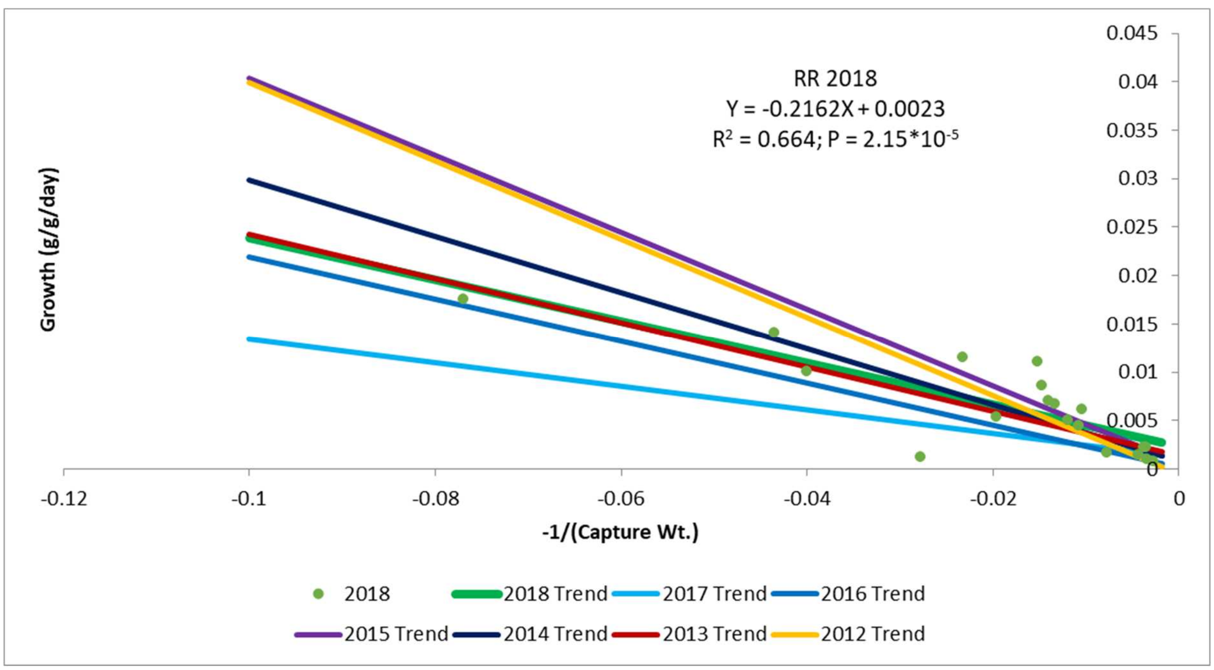


Figure 5-2. Relationship between -1/weight at capture and standardized weight gain of rainbow trout captured in the Re-regulation section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) 2012-2017 are presented for reference purposes.

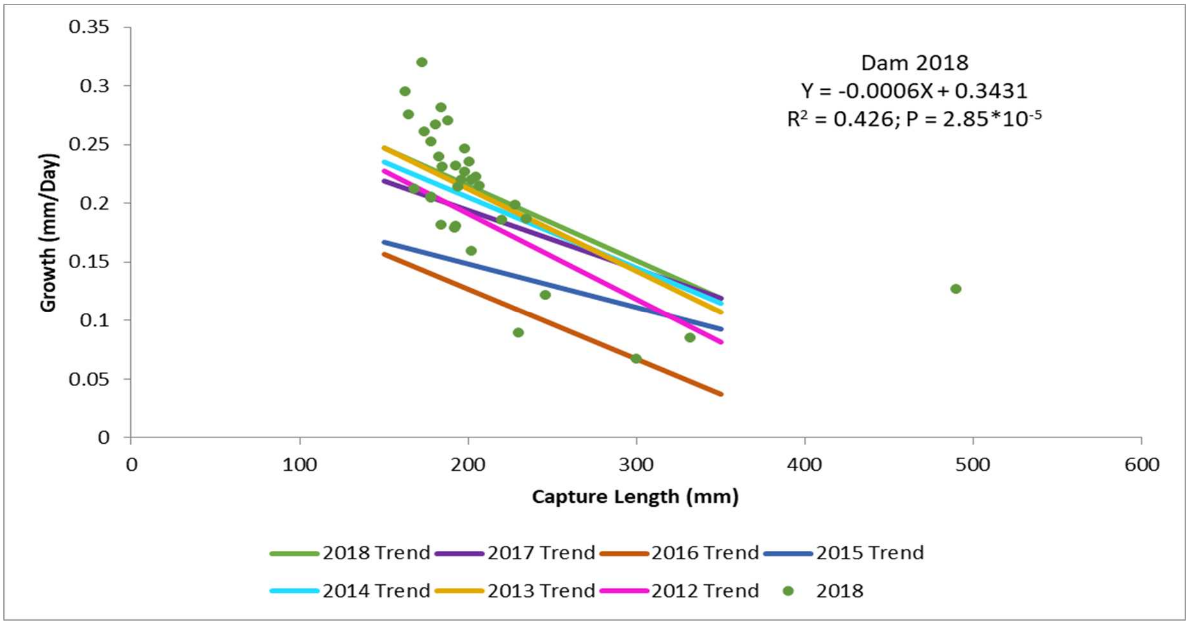


Figure 5-3. Relationship between length at capture and average daily growth (mm per day) of rainbow trout captured in the Libby Dam section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) from 2012-2017 (Dunnigan et al. 2018) are presented for reference purposes.

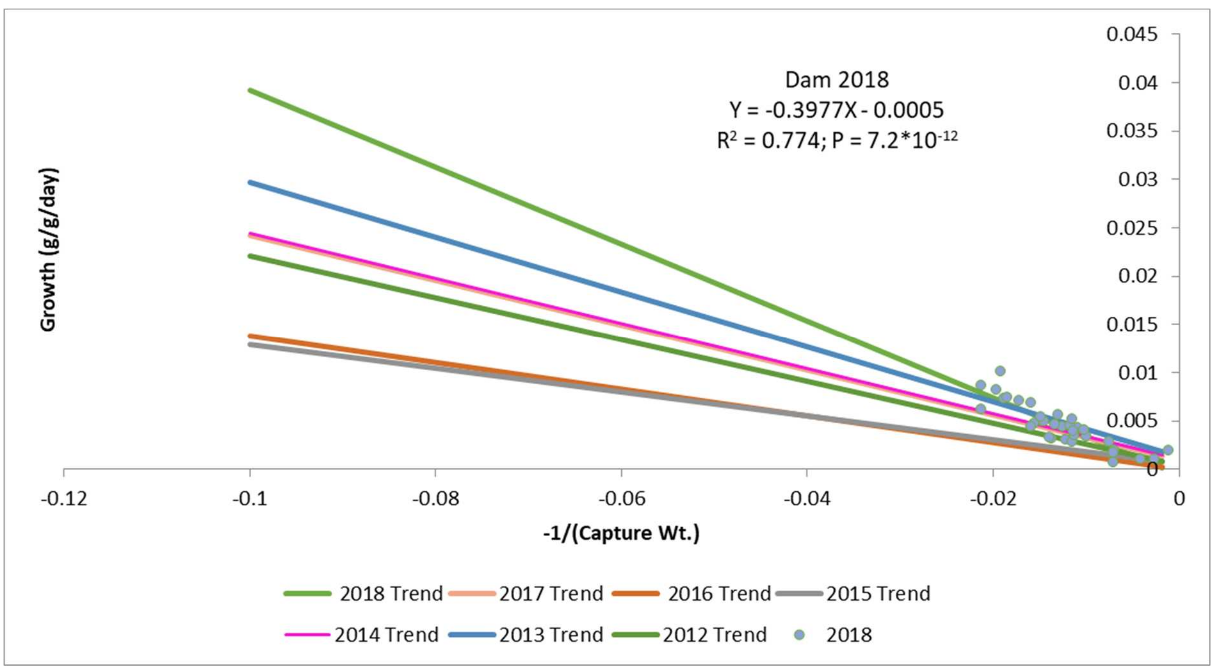


Figure 5-4. Relationship between -1/weight at capture and standardized weight gain of rainbow trout captured in the Libby Dam section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) 2012-2017 are presented for reference purposes.

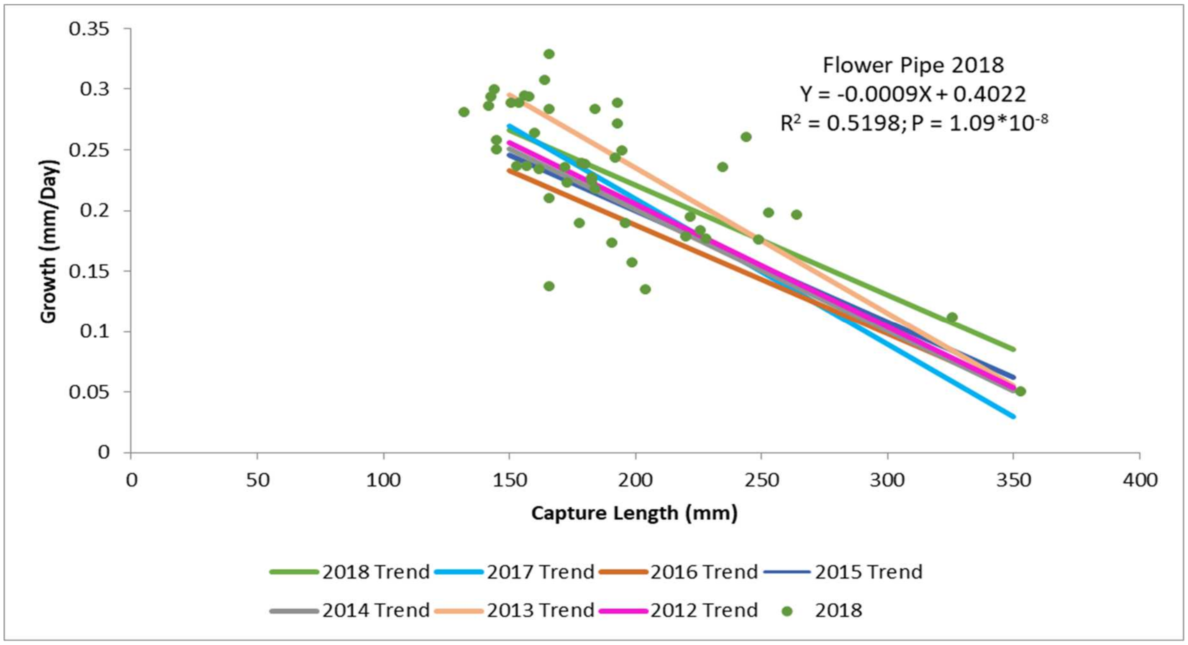


Figure 5-5. Relationship between length at capture and average daily growth (mm per day) of rainbow trout captured in the Flower Pipe section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) from 2012-2017 (Dunnigan et al. 2018) are presented for reference purposes.

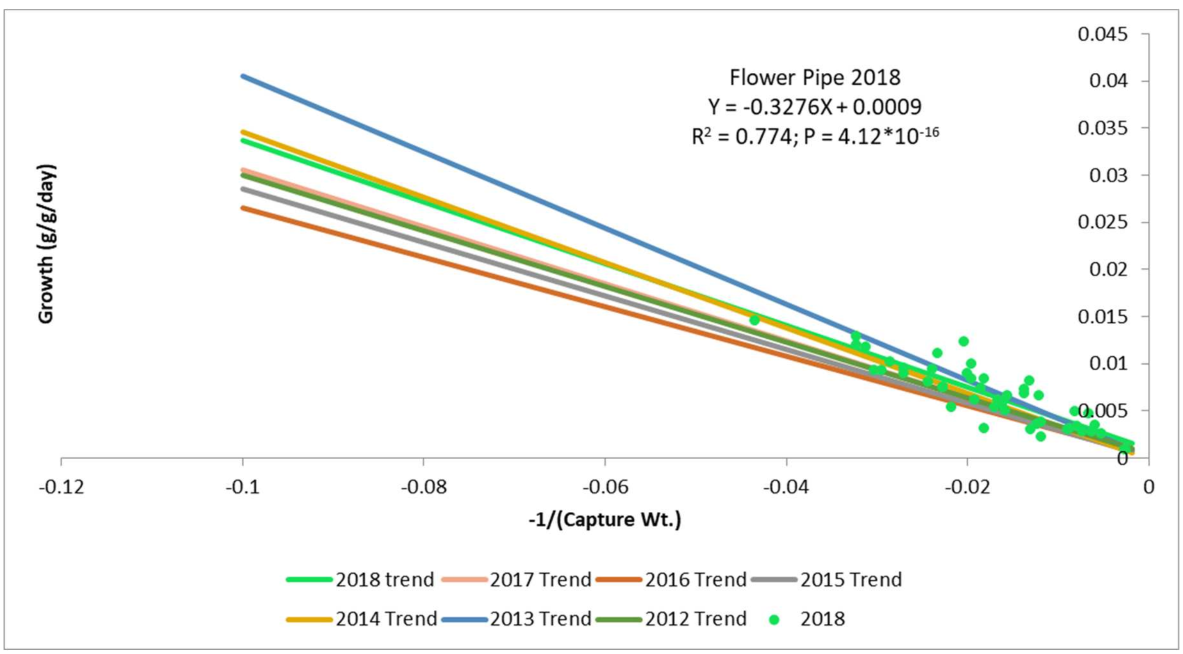


Figure 5-6. Relationship between -1/weight at capture and standardized weight gain of rainbow trout captured in the Flower Pipe section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) 2012-2016 are presented for reference purposes.

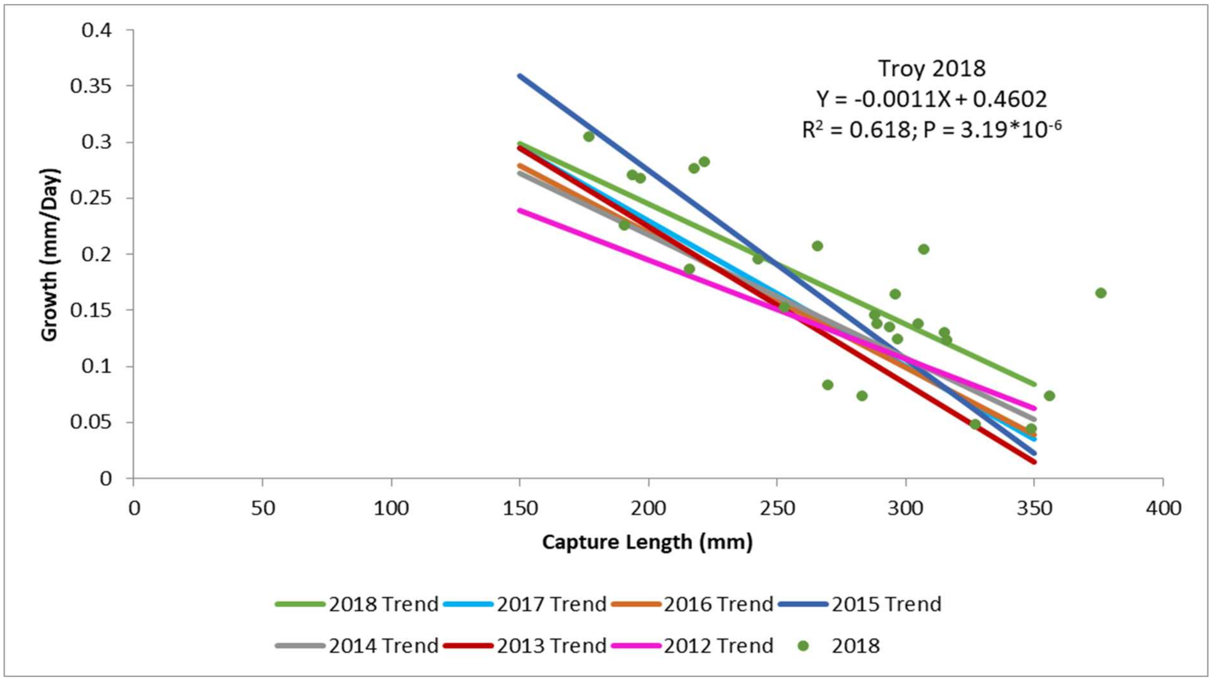


Figure 5-7. Relationship between length at capture and average daily growth (mm per day) of rainbow trout captured in the Troy section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) from 2012-2017 (Dunnigan et al. 2018) are presented for reference purposes.

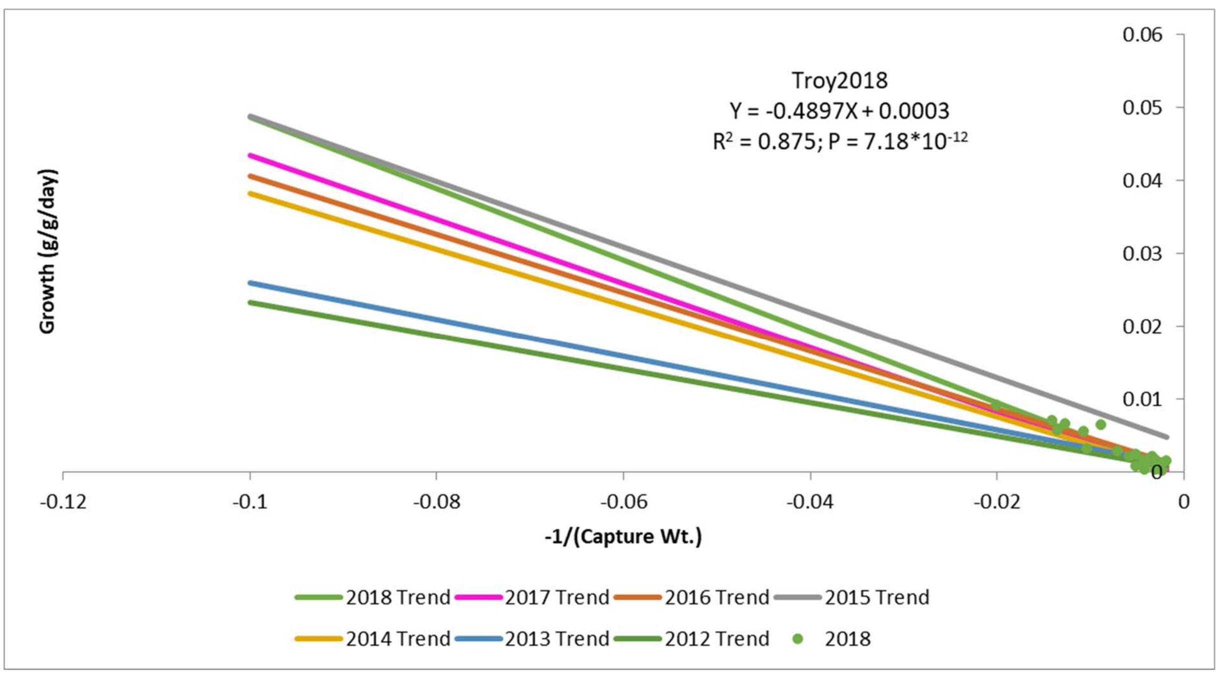


Figure 5-8. Relationship between -1/weight at capture and standardized weight gain of rainbow trout captured in the Flower Pipe section of the Kootenai River in 2018 which were marked in 2017. Linear relationships (line only) 2012-2017 are presented for reference purposes.

Table 5-2. Comparisons of growth rates of recaptured rainbow trout between four sections on the Kootenai River. Annual growth (mm/day) was negatively correlated with length at original capture ($p < 0.05$). Therefore, growth rates (mm/day) were compared by testing for differences in slope and intercept of the regression equations. Only comparisons between years (within the same section) and between sections within the same year were compared. Subsets that differed significantly (alpha = 0.05) share at least one subset number in common.

Section	**Growth Period**	**Number Fish**	**Regression Equation: annual growth (Y) (mm/day) as a function of capture length (mm; X)**	**Annual growth (mm/day) subsets Slope & Intercept**	**Annual growth (mm/day) subsets Intercept Only**
Libby Dam	2011-2012	15	Y = -0.0009X + 0.3641		11,38,49
Libby Dam	2012-2013	27	Y = -0.0007X + 0.3520	1,2,3	12,37,48
Libby Dam	2013-2014	37	Y = -0.0006X + 0.3252	4	13,36,47
Libby Dam	2014-2015	60	Y = -0.0007X + 0.2949	8,11,15	9,14,24,46
Libby Dam	2015-2016	28	Y = -0.0006X + 0.2467	13,14	11,12,13,14,21,23,35
Libby Dam	2016-2017	18	Y = -0.0005X + 0.2940	20,21	21,24,34,45
Libby Dam	2017-2018	34	Y = -0.0006X + 0.3431	28,29,30	34,35,36,37,38,44
Re-Reg	2011-2012	13	Y = -0.0017X + 0.5920	5,15,26	44,45,46,47,48,49
Re-Reg	2012-2013	23	Y = -0.0010X + 0.4085	5	15,26
Re-Reg	2013-2014	55	Y = -0.0014X + 0.4958	2,10,11,25	1,8,10,16
Re-Reg	2014-2015	57	Y = -0.0014X + 0.5302	24	10,17
Re-Reg	2015-2016	93	Y = -0.0012X + 0.4223	15,16,17,23	15,16,17
Re-Reg	2016-2017	39	Y = -0.0007X + 0.3069	23,24,25,26	23,26
Re-Reg	2017-2018	19	Y = -0.0011X + 0.4493	28,32	28
Flower Pipe	2011-2012	50	Y = -0.0010X + 0.4071		3,18,43
Flower Pipe	2012-2013	39	Y = -0.0012X + 0.4750	3,12,18	1,4,25,42
Flower Pipe	2013-2014	90	Y = -0.0010X + 0.4009	4	2,3,4,19,20,41
Flower Pipe	2014-2015	92	Y = -0.0009X + 0.3840	9,10,12,17	9,20
Flower Pipe	2015-2016	74	Y = -0.0009X + 0.3682	14,18,22	18,19,40
Flower Pipe	2016-2017	40	Y = -0.0012 + 0.4498	20,22	22,25,39
Flower Pipe	2017-2018	47	Y = -0.0009X + 0.4022	29,31,32	39,40,41,42,43
Troy	2011-2012	27	Y = -0.0009X + 0.3708	6,7,27	5,33
Troy	2012-2013	30	Y = -0.0014X + 0.5046	1,6	6,32
Troy	2013-2014	22	Y = -0.0011X + 0.4373		2,5,7,31
Troy	2014-2015	21	Y = -0.0015X + 0.5681	7,8,9,19	6,7,8,15,27
Troy	2015-2016	34	Y = -0.0012X + 0.4594	13,19	30
Troy	2016-2017	28	Y = -0.0013X + 0.4897	21,27	22,27,29
Troy	2017-2018	25	Y = -0.0011X + 0.4602	30,31	28,29,30,31,32,33

Table 5-3. Comparisons of growth rates of recaptured rainbow trout between four sections on the Kootenai River. Annual growth (g/g/day) was negatively correlated with -1/weight at original capture ($p < 0.05$). Therefore, growth rates (g/g/day) were compared by testing for differences in slope and intercept of the regression equations. Only comparisons between years (within the same section) and between sections within the same year were compared. Subsets that differed significantly (alpha = 0.05) share at least one subset number in common.

Section	Growth Period	N	Regression Equation: annual growth (Y) (g/g/day) as a function of capture -1/weight (g; X)	Annual growth (g/g/day) subsets Slope & Intercept	Annual growth (g/g/day) subsets Intercept Only
Libby Dam	2011-2012	15	Y = -0.2163X + 0.0004	1,65	1
Libby Dam	2012-2013	27	Y = -0.2838X + 0.0013	2,4,24,64	1,2
Libby Dam	2013-2014	37	Y = -0.2332X + 0.0011	3,5,6,7,25,63	
Libby Dam	2014-2015	60	Y = -0.1231X + 0.0006	1,2,3,8,9,28,46	
Libby Dam	2015-2016	28	$Y = -0.1381X - 1.8*10^{-8}$	24,25,26,27,41	17,22
Libby Dam	2016-2017	17	Y = -0.2333X + 0.0009	41,42,46,62	
Libby Dam	2017-2018	34	Y = -0.3977X -0.0005	54,55,56,62,63,64,65	22
Re-Reg	2011-2012	13	Y = -0.4045X + 0.0005	10,11,29,50,71	
Re-Reg	2012-2013	23	Y = -0.2283X + 0.0004	10,70	4,10,18
Re-Reg	2013-2014	55	Y = -0.2910X + 0.0008	11,12,13,30,49,69	10
Re-Reg	2014-2015	57	Y = -0.3982X + 0.0006	5,12,31,48	3,5
Re-Reg	2015-2016	93	Y = -0.2168X + 0.0002	28,29,30,31,32,33,43	24
Re-Reg	2016-2017	39	Y = -0.1223X + 0.0012	43,44,45,48,49,50	17,18
Re-Reg	2017-2018	19	Y = -0.2161X + 0.0023	54,57,58,69,70,71	24
Flower Pipe	2011-2012	50	Y = -0.2944X + 0.0005	14,15,68	6,11
Flower Pipe	2012-2013	39	Y = -0.4032X + 0.0002	4,13,14,16,17,19,34,47,67	
Flower Pipe	2013-2014	90	Y = -0.3403X - 0.0001	6,15,16,18,37,66,68	3
Flower Pipe	2014-2015	92	Y = -0.2844X + 0.0001	8,17,18,20,32	
Flower Pipe	2015-2016	74	Y = -0.2626X + 0.0003	27,34,37,38,44	16,23
Flower Pipe	2016-2017	40	Y = -0.3029X + 0.0003	47,53	16
Flower Pipe	2017-2018	47	Y = -0.3276X + 0.0009	55,57,66,67	23
Troy	2011-2012	27	Y = -0.2297X + 0.0003	21,22,39,52	4,6,7
Troy	2012-2013	30	Y = -0.2524X + 0.0008	19,23,40,51,60	2,7,8
Troy	2013-2014	22	Y = -0.3837X – 0.0001	7,21,59	5,8,9,13,20
Troy	2014-2015	21	Y = -0.4477X + 0.0004	9,20,22,23,33	9,14,19
Troy	2015-2016	34	Y = -0.4038X - 0.0007	26,38,39,40,45	13,14,15,21
Troy	2016-2017	28	Y = -0.4384X - 0.0004	42,51,52,53	15,19,20
Troy	2017-2018	25	Y = -0.4897 – 0.0003	56,58,59,60,61	21

We observed statistically different linear rainbow trout growth (mm/day) between all four sections in 2018 (Table 5-2). The intercept only for the Troy section compared to the Re-regulation section in 2018 was significantly higher which means that growth for all sizes of rainbow trout in the Troy section in 2018 was significantly higher than the Re-regulations section. However, the slope and intercepts were both significantly different for all other section comparisons in 2018 (Table 5-2). Average growth for a fish 150 mm fish originally marked in 2017 was highest at the Troy section, followed by the Re-regulation, Flower Pipe and lastly the Libby Dam section. However, average growth for a 350 mm fish marked in 2017 was highest at the Libby Dam section followed by the Flower Pipe, Try and Re-regulation sections.

We also observed statistically different weight gain (g/g/day) of rainbow trout between all combinations of sections except for when comparing the growth in 2018 between the Flower Pipe and Troy sections. The slope and intercept differed significantly between all other section comparisons (Table 5-3). The average weight gain of a 10 gram fish during the 2018 growth period was highest at the Troy section followed by the Dam, Flower Pipe and Re-regulations sections. However, the average weight gain of a 550 gram fish was highest at the Re-regulations section followed by the Flower Pipe, Troy and Dam sections.

Linear rainbow trout growth (mm/day) at the Re-regulation section in 2018 differed significantly (intercept only) between all other years except 2015 (Table 5-2). Length growth for all fish in 2018 was significantly higher than 2017, 2016, and 2013, but lower than 2012 and 2014. Average weight gain within this section in 2018 was also significantly between all years except 2017 and 2015. Mean weight gain of all sizes of fish in 2018 was significantly higher than 2016. However, mean weight gain in 2018 differed significantly by size of fish compared to 2014, 2013 and 2012 (slope and intercept; Table 5-3). The average weight gain of a 10 gram fish in 2018 was lower than observed in 2012, 2013 and 2014. However, the average weight gain of a 550 gram fish in 2018 was higher than 2012, 2013 or 2014.

Mean annual rainbow trout linear growth (mm/day) during the 2018 growth season at the Flower Pipe section differed significantly (intercept only) between all other years except 2015 (Table 5-2). Linear growth of rainbow trout during the 2018 growth season was significantly higher for all sizes of fish compared to the 2014, and 2016 growing seasons, but was significantly lower when compared to the 2013 and 2017 growing seasons (Table 5-2). Mean weight gain (g/g/day) in 2018 did not differ significantly between the 2017 or 2015 growing seasons. Growth of all sizes of rainbow trout during the 2018 growing season was significantly higher (intercept only) than the 2016 growth season. However, mean weight gain in 2018 differed significantly by size of fish compared to 2012, 2013 and 2014 (slope and intercept; Table 5-3). The average weight gain of a 10 gram fish in 2018 was higher than observed in 2012, 2013 and 2014. The average weight gain of a 550 g rainbow trout during the 2018 growing season in the Flower Pipe section was higher than observed in 2012, but lower than 2013 or 2014.

Mean annual rainbow trout linear growth (mm/day) during the 2018 growth season at the Libby Dam section differed significantly (intercept only) between all other years except 2015 (Table 5-2). Linear growth of rainbow trout during the 2018 growth season was significantly higher for all sizes of fish compared to the 2014, 2016 and 2017 growing seasons, but was significantly lower when compared to the 2012 and 2013 growing seasons (Table 5-2). We observed a similar trend for mean weight gain (g/g/day) in 2018, with no significant differences between the 2018 and 2015 growth seasons. The mean weight gain of all sizes of fish in 2018 was significantly higher (intercept only) compared to the 2016 growth season (Table 5-3). However, mean weight gain in 2018 differed significantly by size of fish compared to 2012, 2013 and 2014 (slope and intercept; Table 5-3). The average weight gain of a 10 gram fish in 2018 was higher than observed in 2012, 2013 and 2014. However, the opposite was true for the average weight gain of 550 g rainbow trout during the 2018 growth period, where growth was lower than 2012, 2013, and 2014.

Mean annual rainbow trout linear growth (mm/day) during the 2018 growth season at the Troy section differed significantly (intercept only) between all other years except 2015 (Table 5-2). Linear growth of rainbow trout during the 2018 growth season was significantly higher for all sizes of fish compared to the 2012, 2014 and 2016 growing seasons, but was significantly lower when compared to the 2013 and 2017 growing seasons (Table 5-2). Weight gain (g/g/day) of rainbow trout in 2018, did not differ significantly the 2015 or 2017 growth seasons. The mean weight gain of all sizes of fish in 2018 was significantly lower (intercept only) compared to the 2016 growth season (Table 5-3). However, mean weight gain in 2018 differed significantly by size of fish compared to 2012, 2013 and 2014 (slope and intercept; Table 5-3). The average weight gain of a 10 gram fish in 2018 was higher than observed in 2012, 2013 and 2014. However, the opposite was true for the average weight gain of 550 g rainbow trout during the 2018 growth period, where growth was lower than 2012, 2013, and 2014.

Conclusions

The work presented in this chapter represents results from the seventh final year of the multi-year study. The proportion of rainbow trout marked in each of the four sections of the Kootenai River that were recaptured the next year was lowest for the Libby Dam section (range 1.3 to 4.1%) and highest for the Troy section (range 7.2 to 12.8%). However, these rates should not be misinterpreted as estimates of annual survival, nor should they be considered relative to one another due to differences in capture efficiency and the number of marking sessions conducted between river sections across years. More detailed estimates of apparent survival and capture efficiency will be calculated and reported in the next annual report. In addition to quantifying estimates of apparent annual survival between years and sections, we will attempt to correlate important biological and environmental covariates to these vital statistics with the expectation of developing management recommendations to improve either the growth or survival of Kootenai River trout.

Rainbow trout linear growth rates (length and weight) sharply declined with fish size during all years and within all sections. The high degree of variability observed in both cases is mostly attributable to differences in growth rates for different size and age of fish, and makes comparisons between sections and years difficult to interpret. Future attempts to correlate fish growth to important physical and biological covariates will likely involve a general linear mixed modeling approach which will hopefully allow us to identify those environmental factors that influence growth differences between sections and years.

Chapter 6: Burbot Trend and Status Monitoring

This chapter includes the following work elements:

E and F: Monitor trend and status of focal species in MT portion of the Kootenai Basin (Contracts 77012 and 76916).

J: Analyze and interpret Libby Mitigation physical and biologic data, (Contracts 77012 and 76916) and

K and L: Mark rainbow trout and burbot in the Kootenai River below Libby Dam (Contracts 77012 and 76916).

Introduction

Burbot have a wide circumpolar distribution in Eurasia and North America, and southward to about 40°N (Scott and Crossman 1973). Burbot are indigenous to many waterways in Montana, including the Kootenai River. The lower Kootenai River in Idaho and Kootenay Lake in British Columbia, Canada once supported substantial winter recreational and commercial fisheries, but populations have since collapsed after the construction of Libby Dam in 1972 (Paragamian 2000). Concern for the reduction in burbot abundance in the lower Kootenai River in 2001 prompted petitioning for listing under the U.S. Endangered Species Act of 1973, but the U.S. Fish and Wildlife Service ruled that although this population was at low abundance, it was not a distinct population segment, and therefore listing was unwarranted (U.S. Federal Register 2003).

Little information exists regarding the abundance and distribution of burbot in the Montana portion of Kootenai River below Libby Dam or in Libby Reservoir. The trapping efforts in Kootenai River below Libby Dam are the only burbot trend indicator for the Montana portion of the Kootenai River.

Dunnigan and Sinclair (2008) described movements and home ranges of burbot over two spawning seasons in Libby Reservoir. Gill netting efforts conducted by Project 199500400 on Libby Reservoir provide the best long-term burbot abundance trend in the reservoir. These data (see gill netting chapter below) indicate that burbot abundance in Libby Reservoir increased after reservoir construction until the late 1980s and have since declined. This project first initiated a systematic trapping effort directly downstream of Libby Dam that specifically targeted burbot in 1992 and a systematic effort in Libby Reservoir in 2004 to assess the abundance and trend of this focal species in the Montana portion of the watershed. The information presented within this report is a synopsis of those monitoring efforts.

Methods

Protocol Title: MFWP Fish Population Monitoring - Reservoirs v1.0

Protocol Link: http://www.monitoringmethods.org/Protocol/Details/511

Protocol Summary: Monitor trends in abundance (i.e., CPUE), species composition, mean lengths, and condition of the fish communities in large reservoirs and lakes within the Flathead and Kootenai River drainages. Monitoring is completed using sinking and floating gill nets depending on season at standardized locations throughout all or sections of each waterbody.

Burbot Monitoring Below Libby Dam

MFWP has monitored burbot relative abundance directly downstream of Libby Dam since the 1991/1992 season, using baited hoop traps during December and February to capture burbot. For example, if we trapped the stilling basin in during December 2014 and February 2015, we refer to this as the 2014/2015 season. Two to four hoop traps measuring 2-feet diameter, approximately 6-8 feet in length with ¾ inch net mesh were baited with cut bait (usually kokanee, depending upon availability) and lowered in the stilling basin downstream of Libby Dam at depths ranging from 20-55 feet (Figure 6-1). 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 (1994-2001) or 134.2 (2002-present) KHz PIT tag if not previously tagged, and released. Burbot less than approximately 350 mm total length were not tagged. PIT tags were inserted with an 8 or 12-gauge hypodermic needle into the musculature of the left operculum (1994-2005) or the dorsal musculature (2006-present). We standardized the catch of each set by dividing the catch by the number of days each trap soaked (catch per unit effort; CPUE), to compare burbot catch rates among years. We used linear regression to evaluate the trend of CPUE.

Figure 6-1. 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.

Results

Burbot Monitoring Below Libby Dam

The burbot catch in our hoop traps below Libby Dam has declined precipitously since the early 1990s when trapping at this location first began (Figure 7-2). During the 2017/2018 trapping season, we caught three burbot below Libby Dam after fishing a total of 228 trap days, for and average CPUE of 0.013 burbot per trap-day, which is about 66% of the mean catch rate over the past ten years (0.020 burbot per trap-day; including the most recent). Catch rates of burbot in the Kootenai River directly below Libby Dam have exhibited a significant negative trend ($r^2 = 0.349$; $p = 0.001$; Figure 6-2). During this period burbot catch rates have declined an average of 0.075 burbot/trap-day/year. However, over the past ten years (including the most recent), burbot catch rates have not differed significantly from a stable (albeit low) population ($r^2 = 0.06$; $p = 0.46$), exhibiting a mean catch rate of 0.020 burbot per trap-day (Figure 6-3).

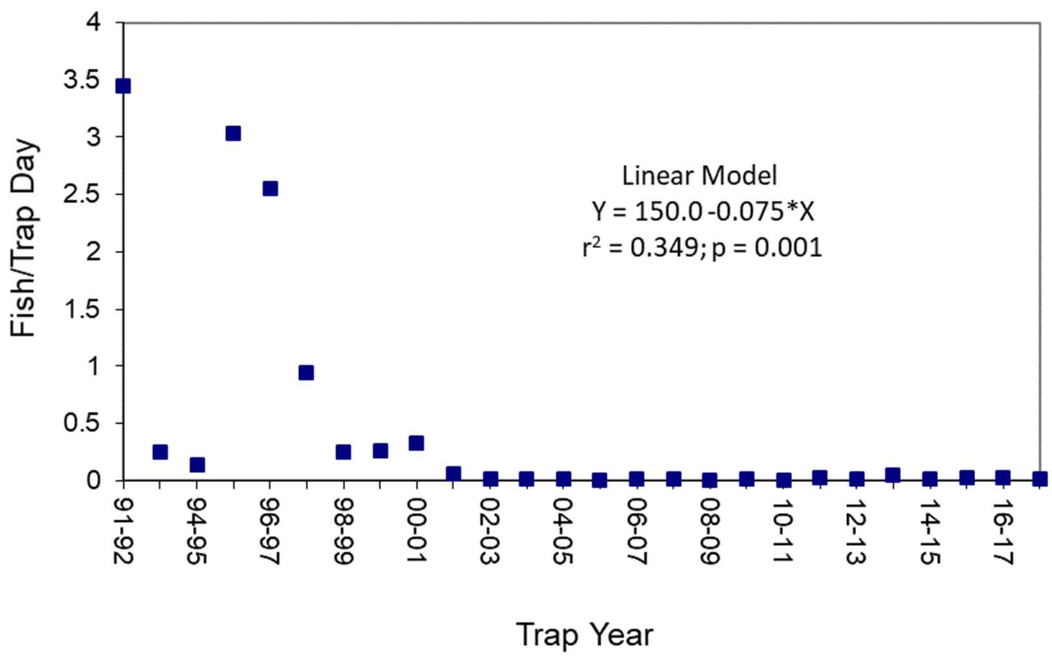


Figure 6-2. Total catch per effort (burbot per trap-day) of baited hoop traps in the stilling basin downstream of Libby Dam 1991/1992 through 2017/2018. The traps were baited with kokanee salmon and fished during December and February.

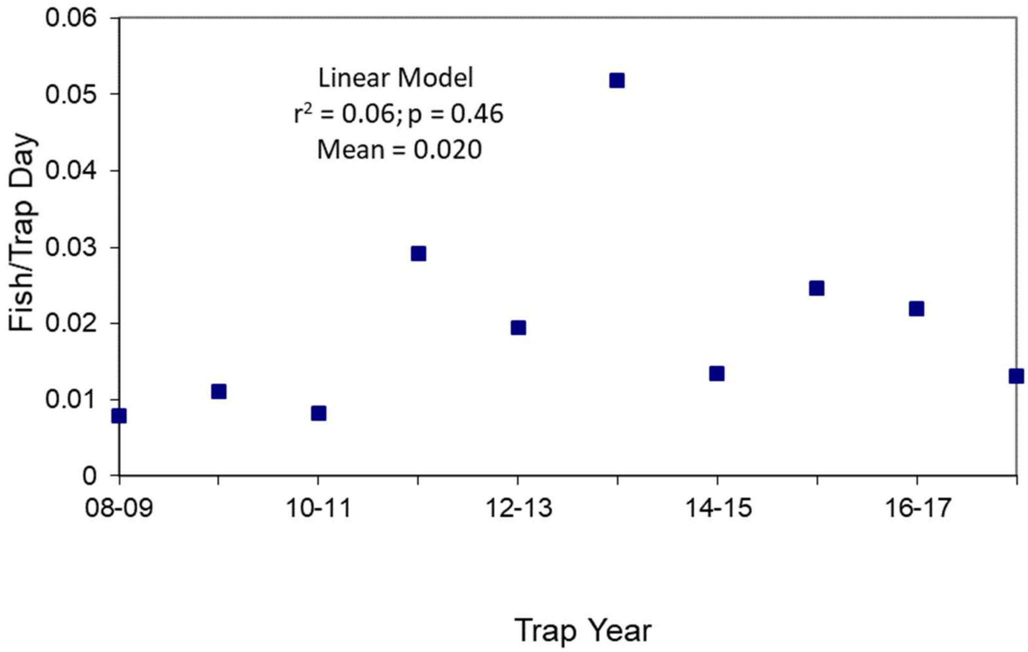


Figure 6-3. Total catch per effort (burbot per trap-day) of baited hoop traps in the stilling basin downstream of Libby Dam 2008/2009 through 2017/2018.

Conclusions

Our estimates of burbot CPUE are low compared to other studies that report similar indices of relative abundance. Breeser et al. (1988) observed catch rates several orders of magnitude higher when fishing overnight hoop trap sets in the Chisana and Tanana rivers, Alaska. Bernard et al. (1991) 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 the Kootenai River below Libby Dam were on the low end of 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. Results from a creel conducted on the Kootenai River immediately downstream of Libby Dam in during the 2014 and 2015 angling season observed no harvest of burbot during the peak season use period (July to September; Sylvester et al. 2015 and 2016). Although MFWP hasn't conducted a yearlong creel specifically to investigate burbot catch and harvest rates, we do not believe that exploitation is an important factor in limiting burbot abundance in the Montana portion of the Kootenai River due primarily to the very low abundance levels that are insufficient to support a viable fishery.

Idaho Fish and Game also monitors burbot relative abundance in the Idaho portion of the Kootenai River. Mean catch rates from 1993-2010 have averaged 0.018 fish/trap/day. Idaho Fish and Game and the Kootenai Tribe of Idaho began stocking hatchery origin burbot in the Idaho portion of the Kootenai River in 2009 (209 individuals released), but the mean number of juvenile hatchery origin burbot released from 2011 to 2016 was 308,554 (Ross et. al 2018). Burbot catch rates since stocking began in the Idaho portion of the Kootenai River increased sharply. Catch rates during the 2009/2010 season averaged 0.005 burbot/trap/day, but increased to mean catch rates of 0.131 burbot/trap/day during the 2016/2017 season (Ross et. al 2018). The increase in catch rates with increased hatchery origin burbot releases suggests that the lower Kootenai River is likely limited by recruitment. It seems likely that a similar situation may exist within the Montana portion of the Kootenai River.

Recent investigations into metal contamination in burbot collected from the reservoir in 2012 and 2013 found that selenium levels were sufficiently high to warrant concern (MFWP, unpublished data). The selenium is likely the result of coal mining activities in the Elk River Valley in British Columbia. When fish bearing waters are contaminated with selenium, the lower trophic levels (fish prey items) accumulate it. Then the selenium bioaccumulates as it is transferred to predators like burbot, which store selenium in their muscles and organs, especially the reproductive organs (Lemly 2002). Symptoms of chronic selenium exposure are multiple, but are especially problematic to the reproductive capacity of ailing fish (Lemly 2002). The coal mining activity in the Elk Valley has been ongoing for about a century, but mining activity has increased in the past several years. It is not known if selenium is the cause of the precipitous decline in the burbot population in Libby Reservoir (Dunnigan et al. 2014), and presumably the Kootenai River downstream of Libby Dam.

Chapter 7: Trend and Status Monitoring of Fishes in Libby Reservoir

This chapter includes the following work elements:

E and F: Monitor trend and status of focal species in MT portion of the Kootenai Basin (Contracts 77012 and 76916).

J: Analyze and interpret Libby Mitigation physical and biologic data, (Contracts 77012 and 76916).

Introduction

Gill nets are passive fishing nets set vertically in the water column so that fish swimming into the net mesh are entangled by the gills, teeth or fins. Fisheries management agencies routinely use gill to evaluate species composition and relative abundance of fish populations in ponds, lakes and reservoirs. Estimates of relative abundance through time are useful for determining trends in population abundance and distribution.

Methods

Protocol Title: MFWP Fish Population Monitoring - Reservoirs v1.0

Protocol Link: http://www.monitoringmethods.org/Protocol/Details/511

Protocol Summary: Monitor trends in abundance (i.e., CPUE), species composition, mean lengths, and condition of the fish communities in large reservoirs and lakes within the Flathead and Kootenai River drainages. Monitoring is completed using sinking and floating gill nets depending on season at standardized locations throughout all or sections of each waterbody.

MFWP has used gill nets to collect fisheries data in the spring and fall since 1975 to assess annual trends in fish populations and species composition in Libby Reservoir (Lake Koocanusa). These annual sampling events were accomplished using criteria established by Huston et al. (1984). This report focuses on the period 1988 through present, but the entire database (1975 through present) was occasionally used to show long-term catch trends. We conducted gill netting during the spring and fall seasons which were characterized based on reservoir operation and surface water temperature criteria. The spring season (April – June) was characterized as reservoir refill period when surface water temperatures ranged between 9 - 13^{o}C, and the fall season (September - October) was characterized as a drafting or stable pool elevation and surface water temperature decreased ranged from 13 - 17^{o}C. Sinking gill nets were used in the spring only in the Rexford transect area, while floater gill nets were used in the fall for the Tenmile, Rexford and Canada sections. MFWP gradually reduced netting effort through time. We fished floating ganged (two nets tied together end to end) nets in the fall in the Tenmile, Rexford and Canada sections of the reservoir. From 1983 to 1990, MFWP fished 28 nets at all three sections, but reduced the effort to

14 nets at all three locations from 1991 to 2000. Netting at the Tenmile section was discontinued in 2000, but continued at the Rexford and Canada sections using 14 nets per section. We fished single sinking nets in the Rexford portion of the reservoir during the spring season. During the period 1983 to 1999, MFWP fished 28 nets at the Rexford section, but reduced that effort to 14 nets beginning in 2000. All the gill nets were multifilament nets that were 38.1 m long and 1.8 m deep and consisted of five equal panels of 19, 25, 32, 38, and 51-mm bar mesh. Set locations were standardized for both fall and spring netting. Nets were set perpendicular to the shoreline in the afternoon by boat and were retrieved before noon the following day. All fish were removed from the nets, identified and counted. We measured the total length (mm) and weighed (g), and calculated Fulton's body condition (K) for the first 25 fish from each net. We also physically examined all game species to identify the gender and state of sexual maturation. When large gamefish (rainbow, cutthroat, bull trout or burbot) were captured alive, only a length was recorded prior to release. We summarized the fish data by season and year calculating the number of nets set, the number of fish caught, and the number of fish caught per net.

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). Stocked inland rainbow trout (Gerrard and Duncan strain) were distinguished from wild rainbow trout by eroded fins (pectoral, dorsal and caudal) and/or presence of hatchery adipose clip. Species abbreviations used throughout this report are: rainbow trout (RBT), inland rainbow trout (IRB), westslope cutthroat trout (WCT), rainbow X cutthroat hybrids (HB), bull trout (BT), kokanee salmon (KOK), mountain whitefish (MWF), burbot (LING), peamouth chub (CRC), northern pikeminnow (NPM), redside shiner (RSS), largescale sucker (CSU), longnose sucker (FSU), and yellow perch (YP).

We calculated catch per unit effort (CPUE) for all species of fish captured during each fall and spring sampling event by dividing the total number of fish captured by the total number of nets fished. We used multiple regression to evaluate trends in catch per unit effort through time. We estimated species composition captured during each sampling period by dividing the number of each species captured by the total number of all fish captured during a given sampling period.

Results

We documented changes in the assemblage of fish species sampled in Libby Reservoir since impoundment, but species composition, and relative abundance has relatively stabilized during the previous 20 years. Kokanee salmon, inland 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 inland rainbow trout in 1985, and since 1988, MFWP annually stocks approximately 30,000-60,000 Duncan or Gerrard strain inland rainbow trout directly into the reservoir (see below). Brook trout are not native to the Kootenai Drainage, but were present in the river before impoundment and continue to be rarely captured in 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 and westslope cutthroat trout abundance all exhibited dramatic decreases in abundance following the first ten years after reservoir filling, but have stabilized at much lower levels of abundance than the pre-dam period (see below). Bull trout abundance in the reservoir has increased since inundation, and is likely a response to reduced angler harvest and an increased forage base (kokanee salmon). Fish species composition also shifted during the first 10 years after reservoir construction, but has generally stabilized in recent years. We attribute many of the changes in species composition and abundance to changes in trophic equilibrium attributable to the aging process of the reservoir (Kimmel and Groeger 1986) and the operational history of Libby Dam during the past 20 years. The following sections present specific trend information for several species of fish currently present in the reservoir.

Kokanee

Since the unintended introduction of fry from the Kootenay Trout Hatchery in British Columbia into Libby Reservoir in the late 1970s, kokanee have become the second or third most abundant fish captured during fall gillnetting. Catch rates of kokanee in the fall gill nets in 2018 averaged 2.7 fish per net, which was lower than the overall mean since 1988, and the mean over the last ten years (6.3 and 5.1 fish/net, respectively). Catch rates in both the spring and fall nets have been variable, with no apparent continuous trend in abundance (Figure 7-1). However, biomass of kokanee per floating gill net has significantly decreased since 1988 ($r^2 = 0.316$; $p = 0.001$; Figure 7-2) an average of 188.5 grams per net per year. The trend in average length and weight of kokanee in the fall nets have exhibited a similar significantly negative trend since 1988 decreasing an average of 1.7 mm and 3.8 g per year (Figure 7-3). The average length and weight of kokanee in 2018 was 288.2 mm and 235.3 g, respectively, which are higher than the ten-year averages (Table 7-1). The data from the fall gill netting efforts provided weak evidence to suggest that kokanee in Libby Reservoir exhibit density dependent growth. When we examined the relationship between catch of kokanee and total length in the fall nets since 1996, a weak and nearly significant negative relationship is evident ($r^2 = 0.13$; $p = 0.095$; Figure 7-4).

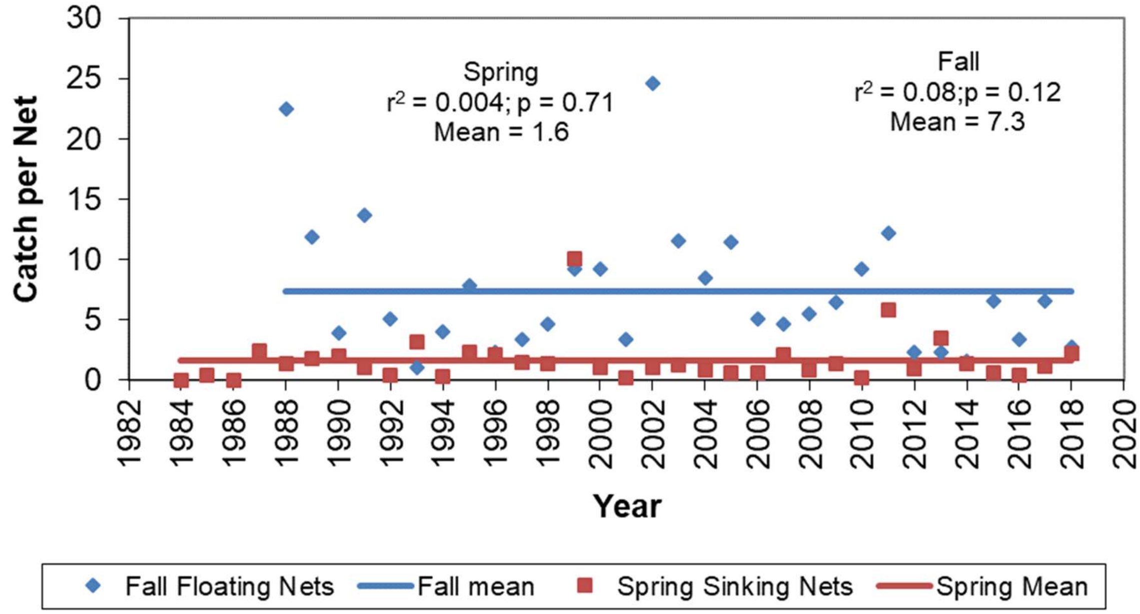


Figure 7-1. Average catch per net of kokanee for fall floating (1988-2018) and spring sinking (1984-2018) gill nets in Libby Reservoir.

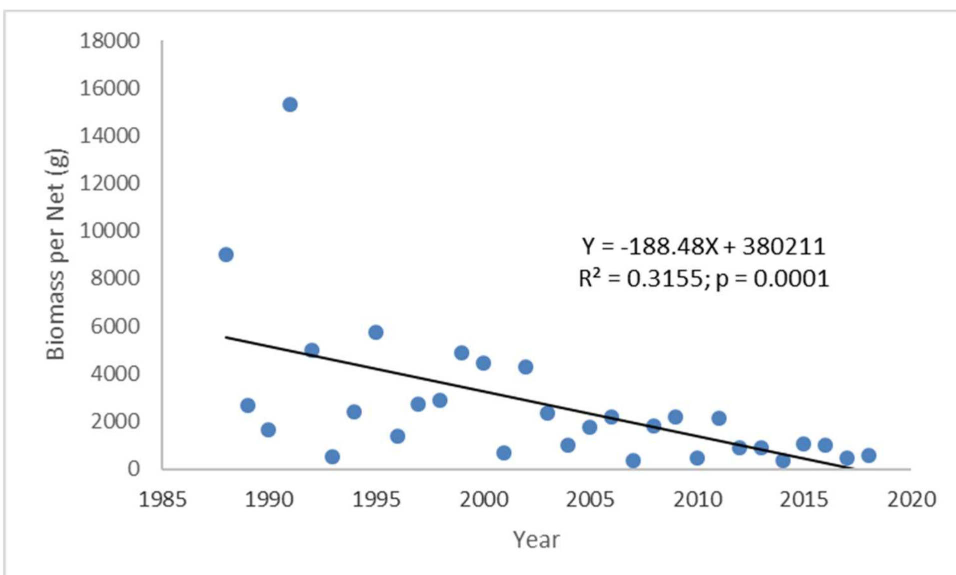


Figure 7-2. Average biomass (grams) per net of kokanee for fall floating (1988-2018) gill nets in Libby Reservoir.

Table 7-1. Average length and weight of kokanee salmon captured in fall floating gillnets (Rexford and Canada Sites) in Libby Reservoir, 1988 through 2017.

Year	Sample Size (n)	Mean Total Length (g)	Mean Weight (g)
1988	2150	315.5	289.1
1989	1259	275	137.2
1990	517	257.3	158.4
1991	624	315.8	327.3
1992	250	350	411.3
1993	111	262.7	162.3
1994	291	270.2	191.7
1995	380	300.2	261.6
1996	132	293.7	234.5
1997	88	329.6	363.2
1998	76	333.9	322
1999	200	291.6	229.6
2000	342	271.3	185.6
2001	120	261.6	161.6
2002	357	251.3	152.2
2003	263	264.9	175.5
2004	194	261	159.2
2005	320	232.2	117.4
2006	163	276.3	202.5
2007	118	290.2	237
2008	206	273.9	187
2009	141	276.8	201.1
2010	171	235.7	122.1
2011	293	239	131
2012	90	297	255.5
2013	33	281.7	191.6
2014	22	233.5	105.4
2015	196	212.7	80.2
2016	99	253.1	154.4
2017	184	278.7	198.5
2018	141	288.2	235.3
Mean	**307.5**	**276.6**	**204.6**
Ten Year Mean	**137**	**259.6**	**167.5**

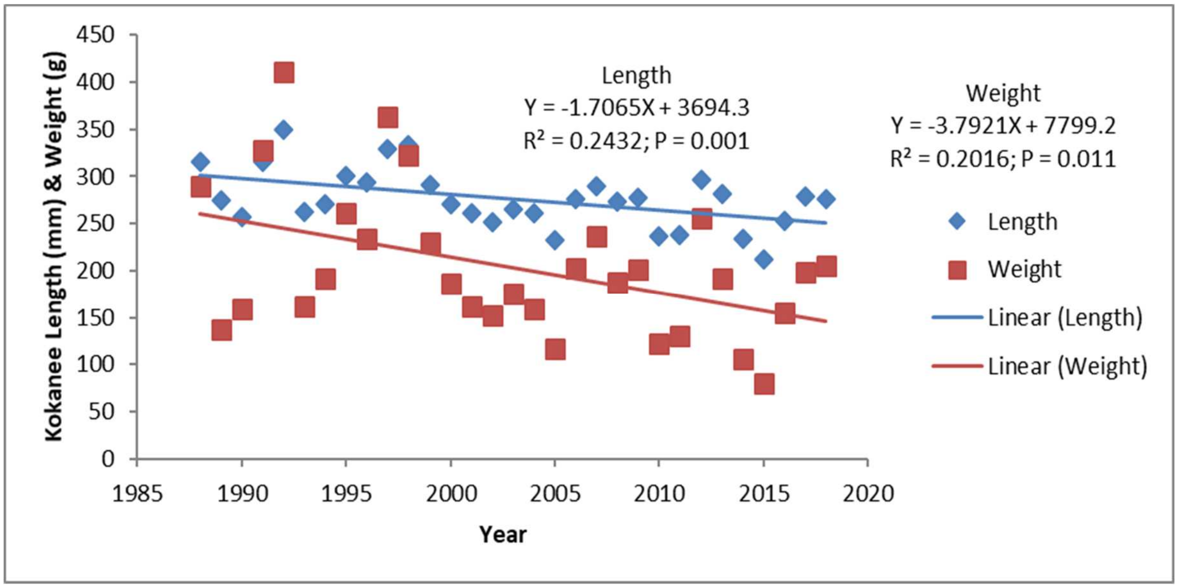


Figure 7-3. Trend in kokanee length and weight in fall gillnets in Libby Reservoir over the period 1988-2018.

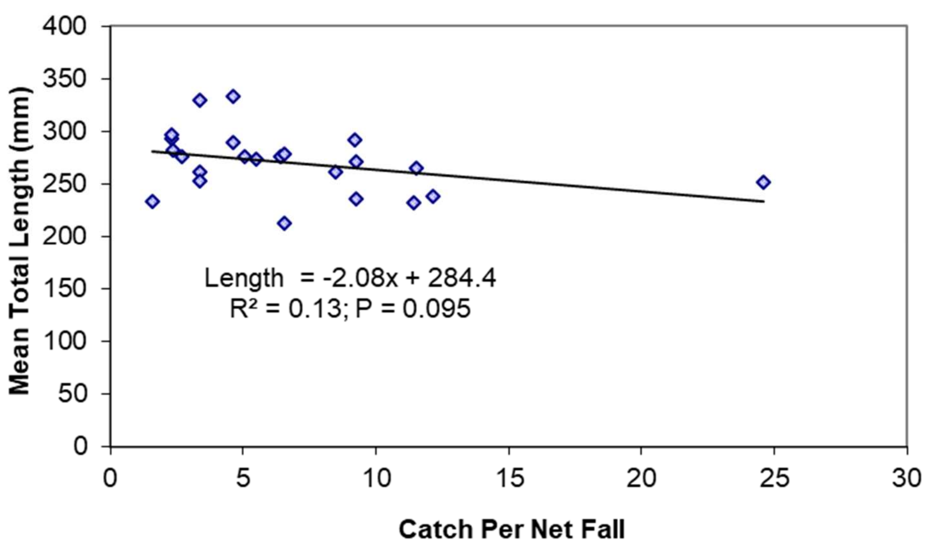


Figure 7-4. Relationship between kokanee length and catch per net in fall gillnets in Libby Reservoir over the period 1996-2018.

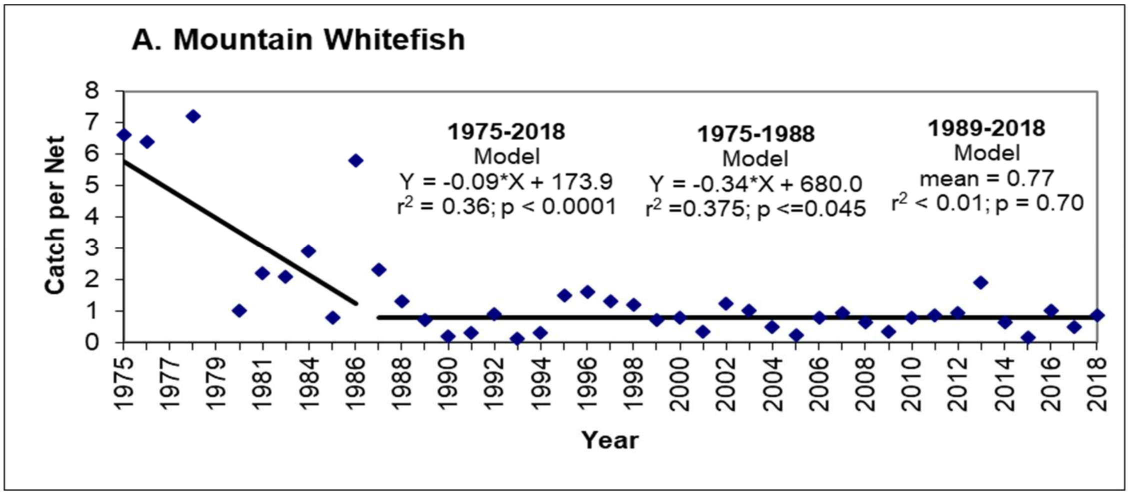
Mountain Whitefish

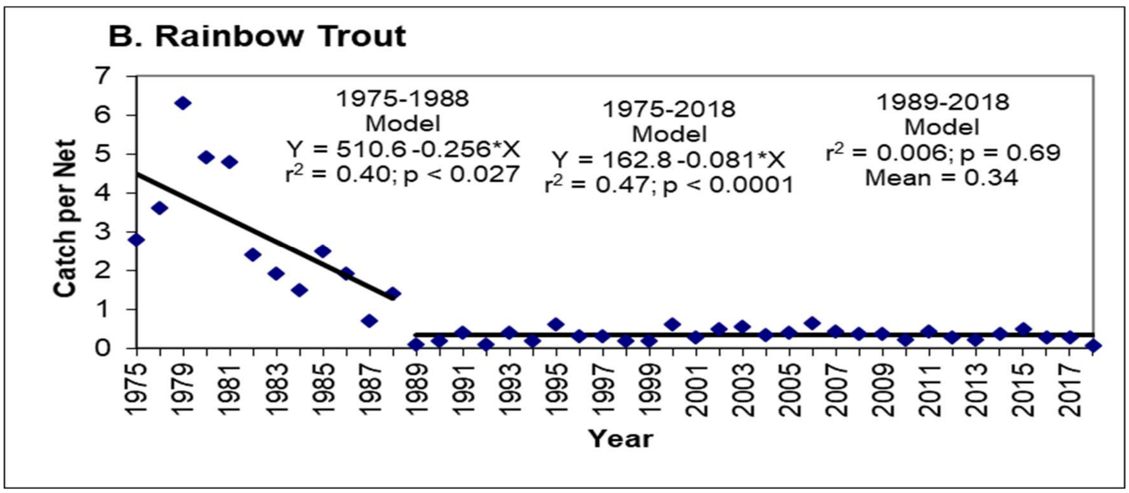
Mountain whitefish are one of three game fish species that have exhibited the most significant decline in abundance since impoundment of the Kootenai River (Huston et al. 1984; Figure 7-5). A linear model provided the best fit to the sinking gillnet mountain whitefish catch data for the period since 1975 (Figure 7-5; $r^2 = 0.36$, $p < 0.0001$). Mountain whitefish catch exhibited a significant negative trend during the first 13 years after reservoir impoundment (1975-1988) decreasing by approximately 0.38 fish/net/year, 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 ($r^2 < 0.01$; $p = 0.77$). Trends in mean biomass per net for mountain whitefish were similar to those for numbers per net (Figure 7-6). Mountain whitefish mean biomass per net since 1988 has been significantly lower but relatively stable ($r^2 = 0.01$; $p = 0.58$; Figure 7-6). We attribute the initial (1975-1988) mountain whitefish decline in Libby Reservoir to the loss of spawning and rearing habitat that resulted from a conversion from a lotic to lentic environment through reservoir construction. Since the initial decline, it appears that mountain whitefish exist at a much lower, but stable equilibrium.

Rainbow and Westslope Cutthroat Trout

Rainbow trout (all strains) and westslope cutthroat trout catch have also both significantly declined since the impoundment of Libby Reservoir (Figure 7-5). Rainbow trout have exhibited two general trends since impoundment. The first trend showed a significant decline in abundance from 1975 to 1988 (Figure 7-5, followed by a period of relatively stable, but reduced abundance from 1989 to 2018 ($r^2 = 0.01$; $p = 0.69$) during which the mean relative abundance was 0.34 fish per net (Figure 8-4). We caught an average of 0.07 rainbow trout per net in 2018. Trends in rainbow trout mean biomass per net were nearly identical to the trend in abundance (Figure 7-6).

Our gill net catch of cutthroat trout in Libby Reservoir exhibits 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 7-5), where mean catch rates decreased on average 0.15 fish per net per year. The second trend from 1987 to 1993 showed reduced but relatively stable abundance (0.41 fish per net; $r^2 = 0.337$; $p = 0.172$). We believe this second period of general equilibrium may have been artificially elevated by the presence of hatchery cutthroat trout that were extensively stocked (but not marked) in the reservoir during this period (Dunnigan et al. 2012). Hatchery cutthroat trout were last stocked in the reservoir in 1994. The third trend occurred from 1994 to 2018, and is characterized by a significantly lower level of abundance (0.11 fish per net; $p < 0.001$), but stable level ($r^2 = 0.12$; $p = 0.10$) compared to the period from 1987 to 1993. We did not catch any westslope cutthroat trout in 2018. Trends in cutthroat trout mean biomass per net were similar to trends in abundance with one notable exception. Mean cutthroat trout biomass per net from 1994 to 2017 has significantly decreased ($r^2 = 0.32$; $p = 0.003$; Figure 7-6) on average by about 2.5 g per net per year.





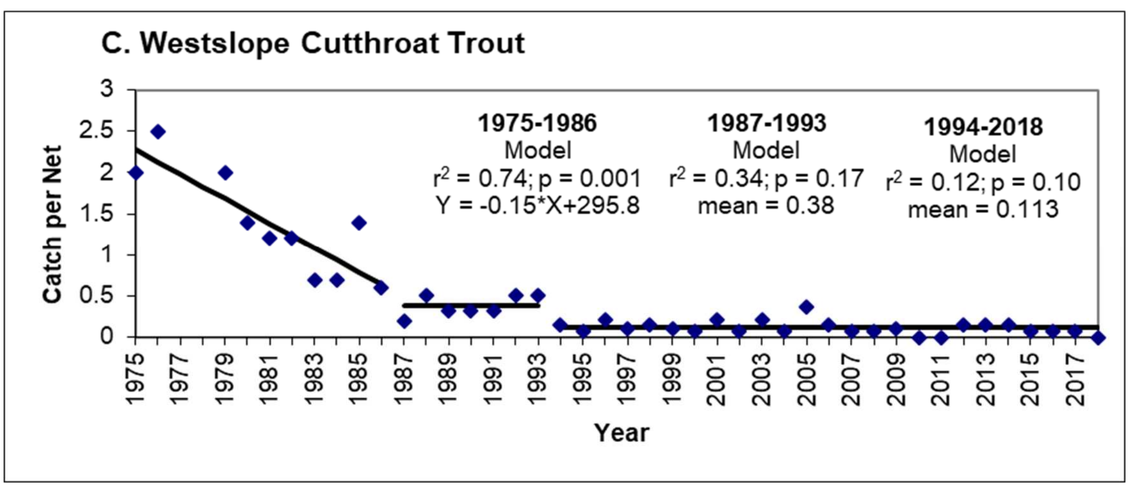
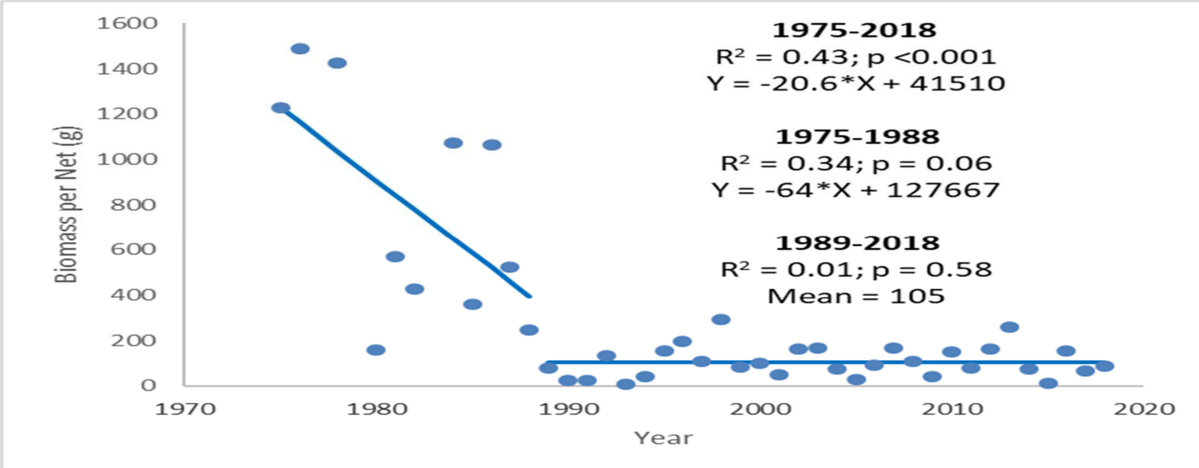
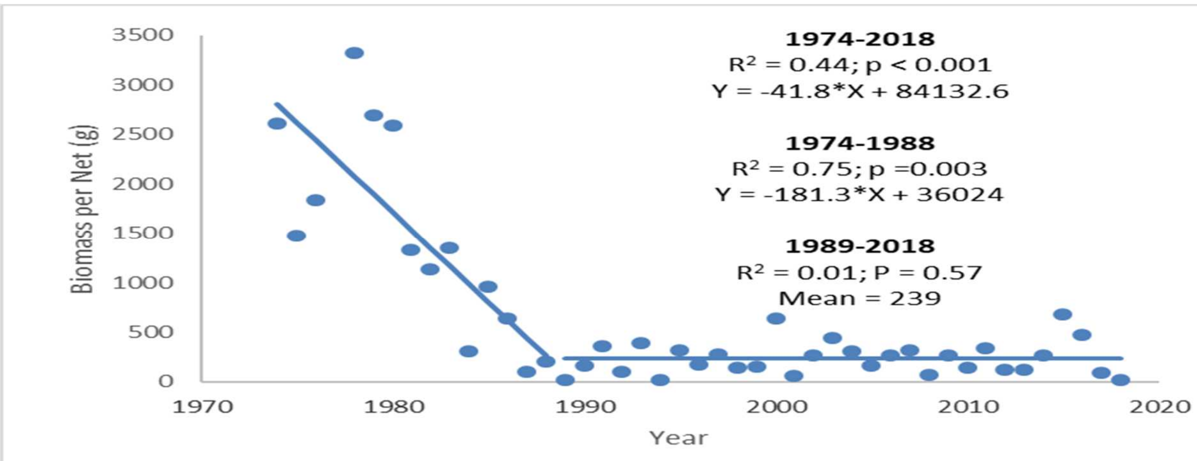


Figure 7-5. Mean catch rates (fish per net) of mountain whitefish (a) in spring sinking gillnets at the Rexford site, rainbow (b) and westslope cutthroat trout (c) in fall floating gillnets from Tenmile and Rexford sites in Libby Reservoir since 1975. The Tenmile site was not sampled since 2000.





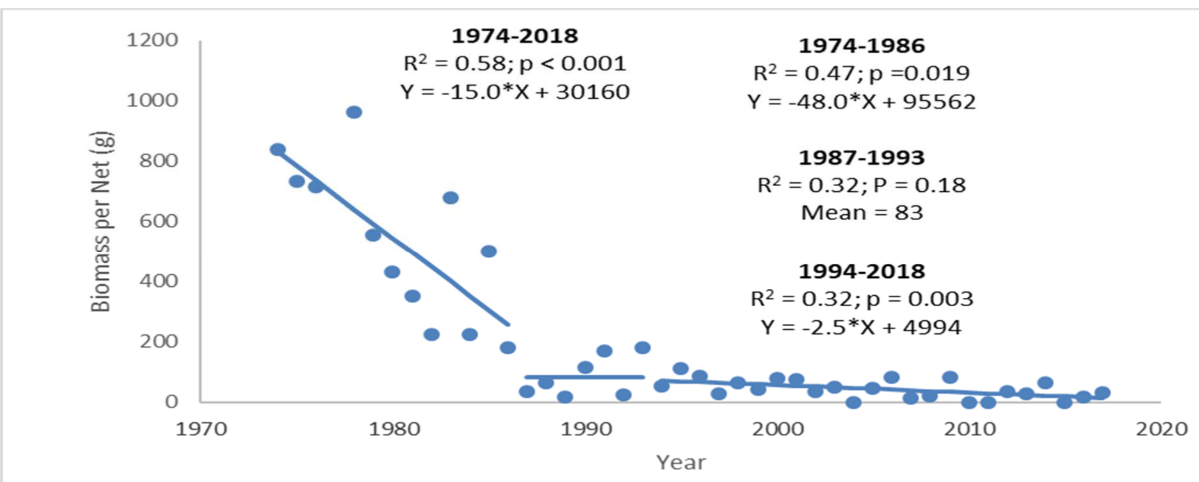


Figure 7-6. Mean biomass (grams per net) of mountain whitefish (upper) in spring sinking gillnets at the Rexford site, rainbow (middle) and westslope cutthroat trout (lower) in fall floating gillnets at the Rexford site in Libby Reservoir since 1975.

Inland Rainbow Trout

Inland rainbow trout were first introduced to Libby Reservoir in 1985 by the British Columbia Ministry of Environment (BCMOE). The BCMOE continued stocking approximately 5,000 fingerling fish (Gerrard strain) annually into Kikomun Creek (a tributary to the Kootenay River) from 1988-1998 (L. Siemens, BCMOE, personal communication). From 1988-1999, MFWP acquired inland rainbow (Duncan Strain) from Ennis National Fish Hatchery, and stocked 10,831-73,386 age one fish into the reservoir annually (Table 8-2). However, at the latter end of this period, Ennis National Fish Hatchery decided to discontinue the broodstock source used to produce these fish. In 1997, MFWP decided to start a broodstock at Murray Springs Fish Hatchery in Eureka, MT using eggs that were collected from Luce Reservoir, Wyoming (J. Lord, M FWP, personal communication). The fish in Luce Reservoir were also originally a result of fish plants from the Ennis NFH broodstock (Duncan Strain). The resulting Murray Springs program first released fish into Libby Reservoir in 2002, with approximately 30,000 age 0 fish released. Releases continued through 2009, with the annual stocking program consisting of approximately 30,000 age 0 fish and 15,000 age one fish (Table 7-2). MFWP evaluated this program through gillnet catch rates, creel surveys on the reservoir, and genetic analysis (Leary 2005), and concluded that the Murray Springs brood fish were not contributing to the trophy fishery in the reservoir. In 2005, MFWP obtained triploid (3N) eggs from Wardner Hatchery in British Columbia (Gerrard strain), and in 2006, stocked these fish into Libby Reservoir as age one fish (Table 7-2). The Murray Springs broodstock (Duncan strain) was discontinued in 2008, with the last release of age one fish occurring in 2009 (Table 7-2). From 2009 to 2013, MFWP acquired infertile (triploid; 3N) Gerrard strain eggs from the Wardner Hatchery with annual stocking of age 1 fish into the reservoir (Table 7-2). However, in 2013, MFWP acquired the entire Gerrard broodstock from the Wardner Hatchery, and transported that broodstock to the Murray Springs facility. MFWP has only released triploid (3N) fish in Libby Reservoir since 2005.

Catch rates for inland rainbow trout in fall gillnets has been low since 1996, averaging only 0.05 fish per gillnet. The catch rate of hatchery rainbow trout has significantly decreased since stocking began in 1988 (Figure 7-7). From 2010 to 2014 and 2017 and 2018, we did not catch any inland rainbow trout in our nets. However, in 2015 and 2016, we captured a single hatchery fish (Table 8-2). The catch rate of inland rainbow trout in fall floating gillnets (fish per net) is not significantly correlated with the number of age one hatchery Inland rainbow trout stocked in the reservoir the previous year ($r^2 = 0.089$; $p = 0.109$) from 1989 through 2018. MFWP discontinued stocking age 0 hatchery inland rainbow trout in the reservoir in 2009, due to the relatively low apparent survival. However, due to excess hatchery production in 2015, MFWP stocked over 56,000 age 0 hatchery fish into the reservoir. Although MFWP's objective for this hatchery program is to produce large (> 5 pounds) fish, our gillnetting generally does not capture large rainbow trout in the reservoir. The average size inland rainbow trout captured in our gillnets has not exceeded 350 mm since 2000. However, the single fish captured in 2016 was 621 mm. We did not capture any hatchery rainbow trout in 2018.

Table 7-2. Inland rainbow trout stocking and capture history in Libby Reservoir, 1988 through 2018. The Tenmile site was not been sampled since 2000.

Year	# Caught	Mean Total Length (mm)	Mean Weight (g)	# Age 0 Stocked	# Age 1 Stocked	Total # Stocked
1988	3	289	216	0	26756[1]	26,756
1989	0	n/a	n/a	0	73,386[1]	73,386
1990	18	301	243	0	396,83[1]	39,683
1991	6	383	589	0	150,04[1]	15,004
1992	3	313	289	0	129,18[1]	12,918
1993	4	460	373	0	108,31[1]	10,831
1994	0	N/A	N/A	0	16364[1]	16,364
1995	12	313	311	0	15844[1]	15,844
1996	2	460	1192	3,165[1]	9,396[1]	12,561
1997	1	395	518	0	22,610[1]	22,610
1998	2	376	450	0	16,368[1]	16,368
1999	3	378	504	0	13,123[1]	13,123
2000	3	395	555	0	0	0
2001	0	N/A	N/A	0	0	0
2002	0	N/A	N/A	29,564[2]	0	29,564
2003	5	260.8	159.2	31,039[2]	13,721[2]	44,760
2004	0	N/A	N/A	46,944[2]	16,110[2]	63,054
2005	0	N/A	N/A	33,265[2,4,5]	14,933[2,4,5]	48,198
2006	1	256	174	28,578[2,4,5]	22,638[3,4,5]	51,216
2007	1	277	220	32,240[2,4]	16,091[2,4]	48,331
2008	1	252	181	38,712[2,4]	18,042[2,4]	56,754
2009	2	283	196	0	16,757[2,4]	16,757
2010	0	n/a	n/a	0	30,709[3,4,5]	30,709
2011	0	n/a	n/a	0	31,046[3,4,5]	31,046
2012	0	n/a	n/a	0	33,571[3,4,5]	33,571
2013	0	n/a	n/a	0	1,498[3,4,5]	1,498
2014	0	n/a	n/a	0	30,007[3,4,5]	39,007
2015	1	337	285	56,355[4,5,6]	33,510[4,5,6]	89,865
2016	1	621	2082	0	68,419[4,5,6]	68,419
2017	0	N/A	N/A	0	72,318[4,5,6]	72,318
2018	0	N/A	N/A	0	69,381[4,5,6]	69,381
Mean	**2.2**	**352.8**	**474.3**	**10,945**	**24,549**	**34,513**
Total	**69**			**328,348**	**761,034**	**1,069,896**

[1]Ennis National Fish Hatchery (Duncan Strain), [2]Murray Springs Hatchery (Duncan Strain), [3]Eggs obtained from Wardner Hatchery B.C (Gerrard Strain) but reared at Murray Springs, [4]Triploid Fish, [5]Adipose Fin Clipped, [6]Progeny of fish spawned and reared at Murray Springs Hatchery (Gerrard Strain).

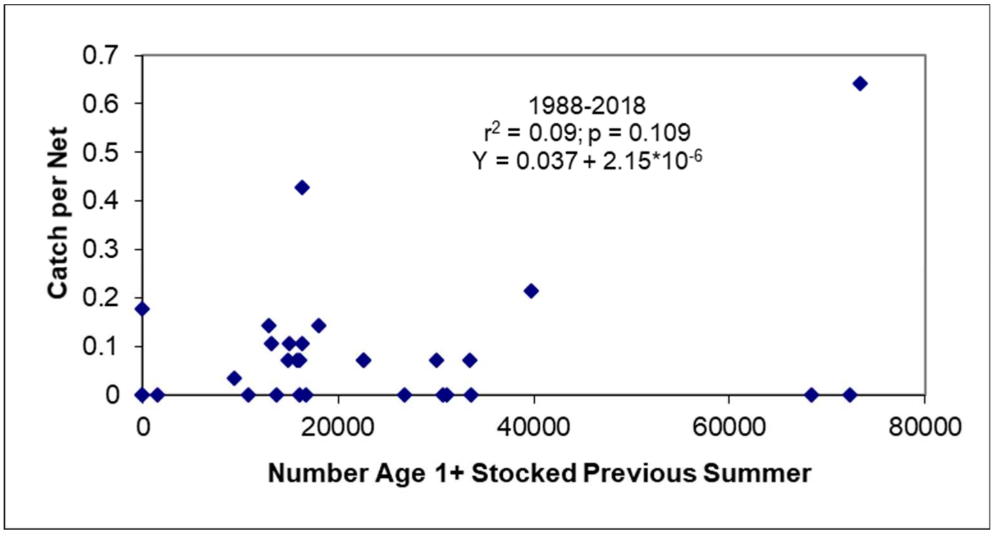


Figure 7-7. Average catch (fish per net) of Inland rainbow trout in fall floating gill nets in Libby Reservoir at the Rexford and Tenmile sites 1988-2018. The Tenmile site was not sampled since 2000.

Bull Trout

The catch of bull trout has significantly increased since reservoir inundation ($r^2 = 0.454$; $p < 0.001$; Figure 7-8). However, there are three apparent trends over this period. Spring gill net catch of bull trout during the period 1975-1989 appeared to exist at an equilibrium with a slope (0.0091 fish per year) that was not significantly different than zero ($r^2 = 0.011$; $p = 0.751$). Bull trout catch rates on Libby Reservoir began increasing during the period 1990-2004 ($r^2 = 0.81$; $p < 0.001$), and peaked in 2000 at 6.71 bull trout per net. During the period 2005 to 2018, catch rates have been variable, without a significant trend ($r^2 = 0.076$; $p = 0.338$), and averaged 4.2 fish per net. The mean catch rate we observed in 2018 (3.57 fish per net) was slightly less than the mean since 2005. Trends in bull trout mean biomass per net have exhibited a trend that was similar to bull trout abundance (Figure 8-9). Bull trout redd counts in both the Wigwam River and Grave Creek (Chapter 4) are significantly and positively correlated to the spring gill net catch rates for bull trout ($r^2 = 0.268$; $p = 0.010$).

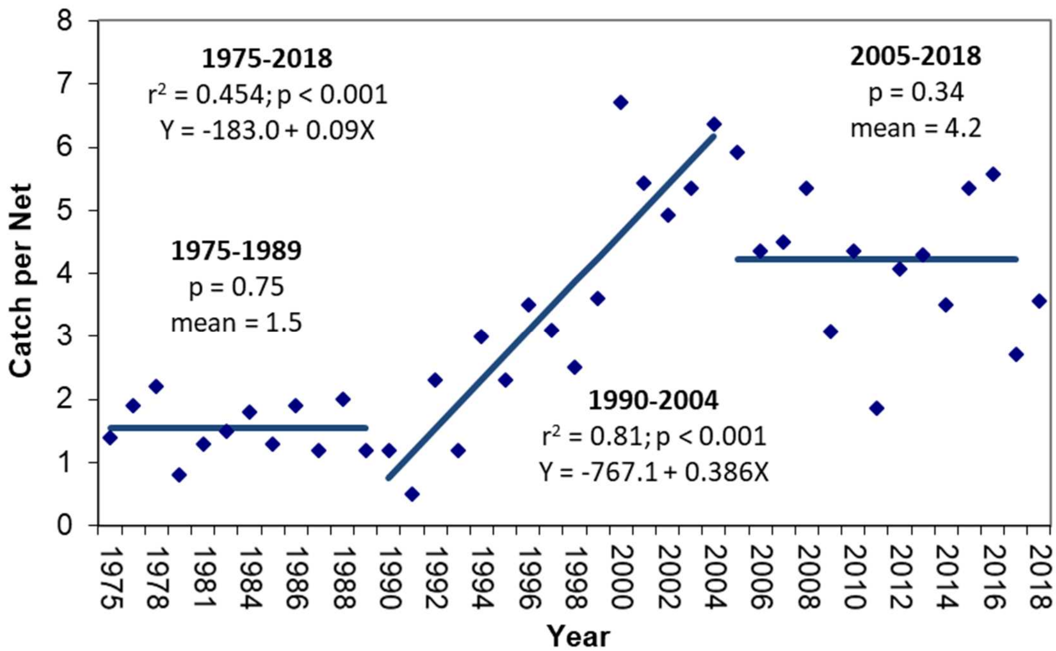


Figure 7-8. Average catch per net of bull trout in spring gill nets at the Rexford site on Libby Reservoir 1975-2018.

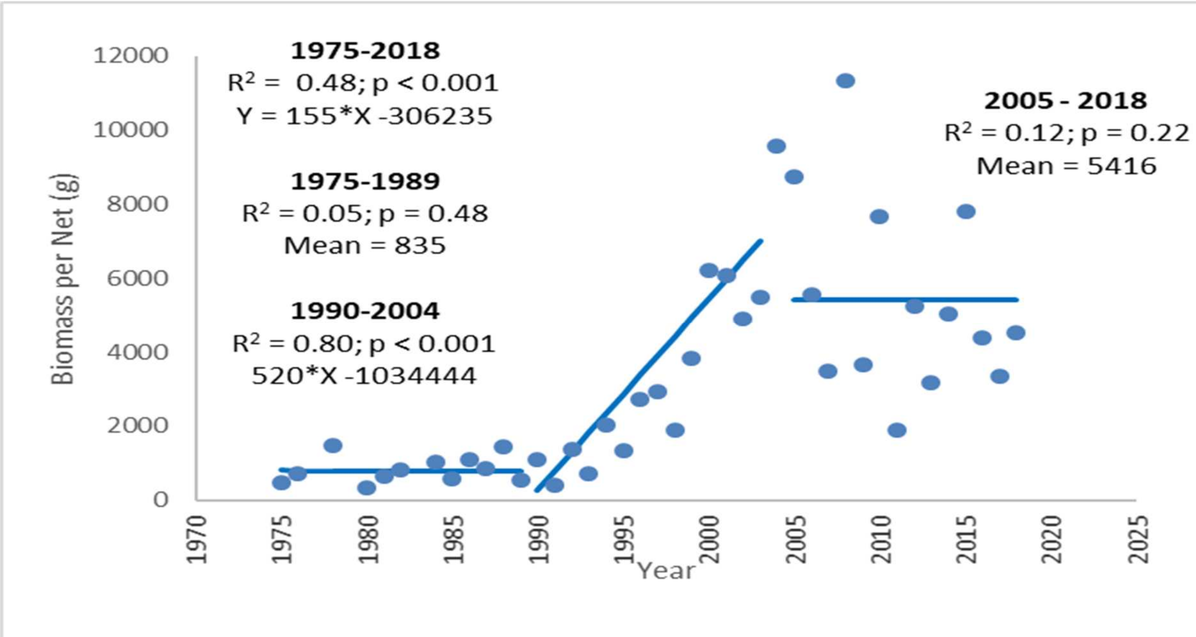


Figure 7-9. Mean biomass (grams per net) of bull trout in spring sinking gillnets at the Rexford site in Libby Reservoir since 1975.

Burbot

Burbot catch rates in spring sinking gillnets in Libby Reservoir exhibit two general trends since construction of the reservoir. Catch rates during the period 1975-1988 exhibit a significant increasing trend ($r^2 = 0.813$; $p < 0.0001$; Figure 7-10). However, during the period 1989-present, catch rates have exhibited a significant negative trend (Figure 8-10; $r^2 = 0.503$; $p < 0.001$). Burbot catch per net for spring sinking nets has declined an average of 0.014 fish per net since 1989. We caught a single burbot in 2018 for an average catch rate of 0.071 burbot per net. Mean burbot biomass per net exhibits a similar trend as burbot abundance per net (Figure 7-11). Burbot catch rates in spring gillnets is however significantly and positively correlated ($r^2 = 0.644$; $p < 0.0001$; Figure 7-12) 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.

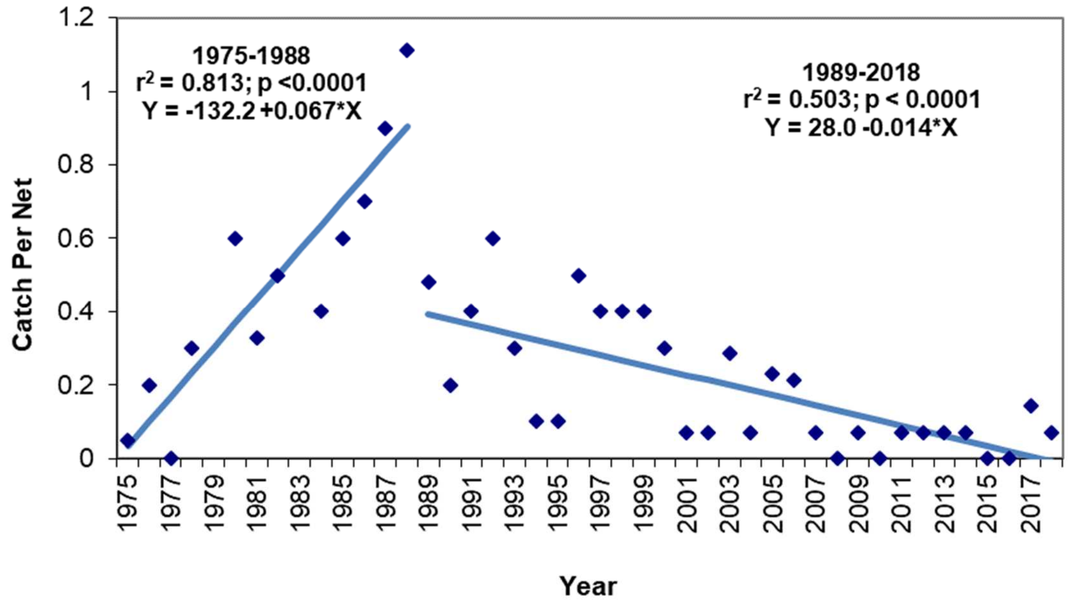


Figure 7-10. Mean catch per net of burbot in sinking gillnets during spring gillnetting at the Rexford site on Libby Reservoir, 1975-2018.

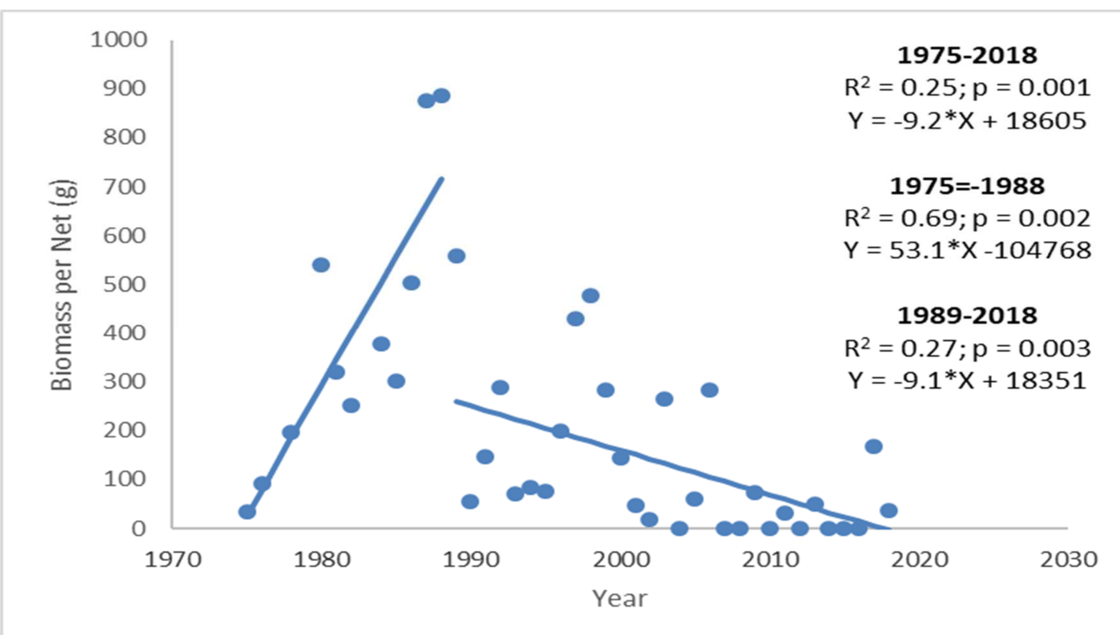


Figure 7-11. Mean biomass (grams per net) of burbot in spring sinking gillnets at the Rexford site in Libby Reservoir since 1975.

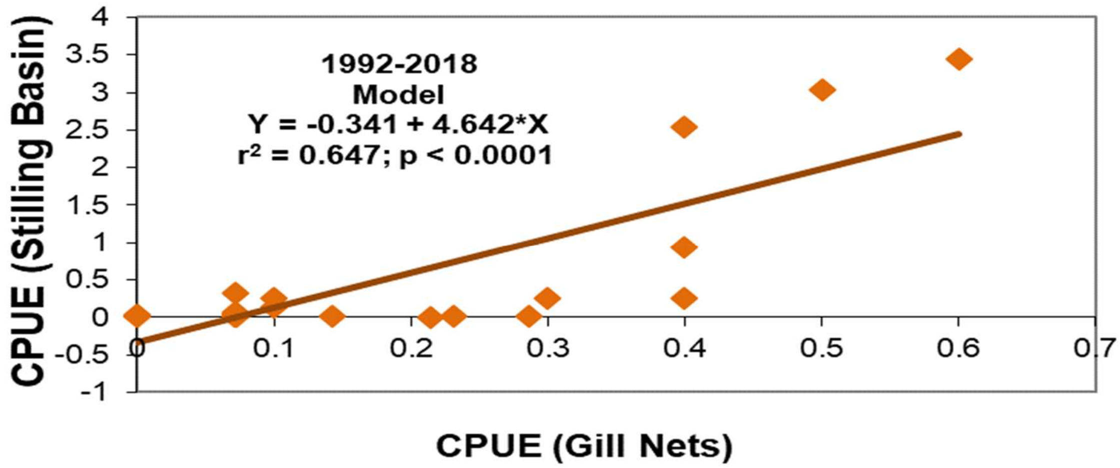


Figure 7-12. The relationship between mean burbot catch per net for spring sinking gillnets on Libby Reservoir and burbot catch rates (Burbot/trap day) of baited hoop traps in the stilling basin below Libby Dam 1995-2018.

Nongame Species

Columbia River Chub (peamouth)

Columbia River Chub are the most abundant fish species captured in the spring sinking gill nets and are either the first or second most abundant fish species captured in the fall floating gill nets. Trends for this species were assessed using the spring sinking nets. Catch rates of Columbia River Chub during the first six years of the reservoir (1975-1980) were less than 10 fish per net, but generally increased since. Catch rates peaked in 1998 at an average of 172 fish per net. The catch rates in 2018 averaged 66.1 fish per net. There has been a weak but significant increase in Columbia River Chub catch rates since reservoir creation ($r^2 = 0.11$; $p = 0.04$) increasing on average of 0.91 fish per net per year since 1975 (Figure 7-13). Mean biomass per net of Columbia River Chubs has also exhibited a significant increasing trend since impoundment ($r^2 = 0.19$; $p = 0.004$; Figure 7-14), increasing on average by almost 128 grams per net per year. Biomass per net of Columbia River Chubs has been relatively stable over the past 15 years (Figure 7-14).

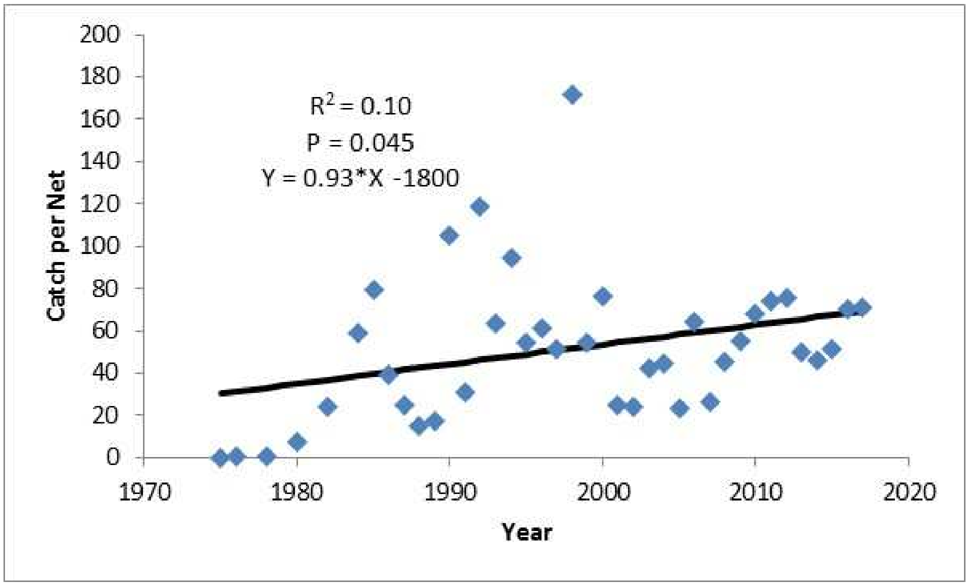


Figure 7-13. Mean catch per net of Columbia River Chub in sinking gillnets during spring gillnetting at the Rexford site on Libby Reservoir, 1975-2016.

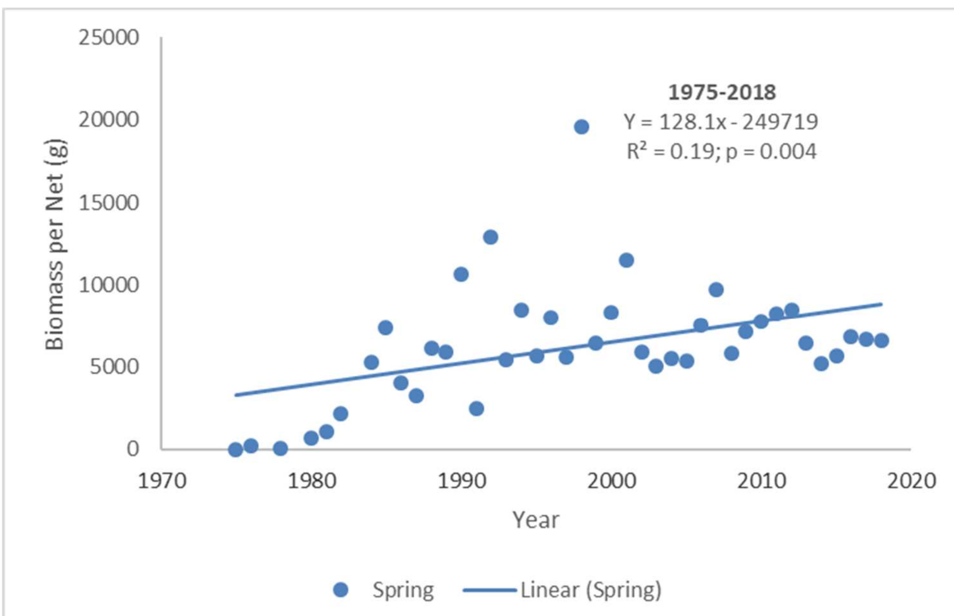


Figure 7-14. Mean biomass (grams per net) of Columbia River Chub in spring sinking gillnets at the Rexford site in Libby Reservoir since 1975.

Largescale Sucker

Largescale sucker are generally the third most abundant species captured in the spring sinking gill nets in Libby Reservoir. However, the abundance of largescale suckers has significantly declined since impoundment ($r^2 = 0.10$; $p = 0.050$; Figure 7-15). The highest catch rate of largescale sucker observed in Libby Reservoir was in 1975 and averaged over 37 suckers per net, and the lowest overall catch rate (2.4 fish per net) was observed in 1991. We captured an average of 22.2 largescale suckers per net in 2018. We were unable to discern a trend in mean biomass per net of largescale suckers ($r^2 = 0.001$; $p = 0.82$; Figure 7-16). The mean biomass per net of largescale suckers has averaged 5,720 grams per net since 1975.

Northern Pikeminnow

Northern pikeminnow in recent years are generally the second most abundant species captured in the spring sinking gill nets in Libby Reservoir. Northern pikeminnow abundance has significantly increased since impoundment at an average rate of 0.478 fish per net per year ($r^2 = 0.689$; $p < 0.001$; Figure 7-17). The highest catch rate observed in Libby Reservoir occurred in 2013 (23.7 fish/net). We captured an average of 22.6 fish/net in 2018. Mean biomass per net of northern pikeminnows has also increased since impoundment ($r^2 = 0.72$; $p < 0.001$; Figure 7-18), increasing on average by 174 grams per net per year.

Longnose Sucker

Longnose sucker abundance in Libby Reservoir has significantly decreased since impoundment ($r^2 = 0.44$; $p < 0.001$), decreasing on average by 0.16 fish per net since 1975 (Figure 7-19). Trends in mean longnose sucker biomass per net have followed a similar trend ($r^2 = 0.32$; $p < 0.001$), decreasing on average by 56 grams per net per year (Figure 7-20).

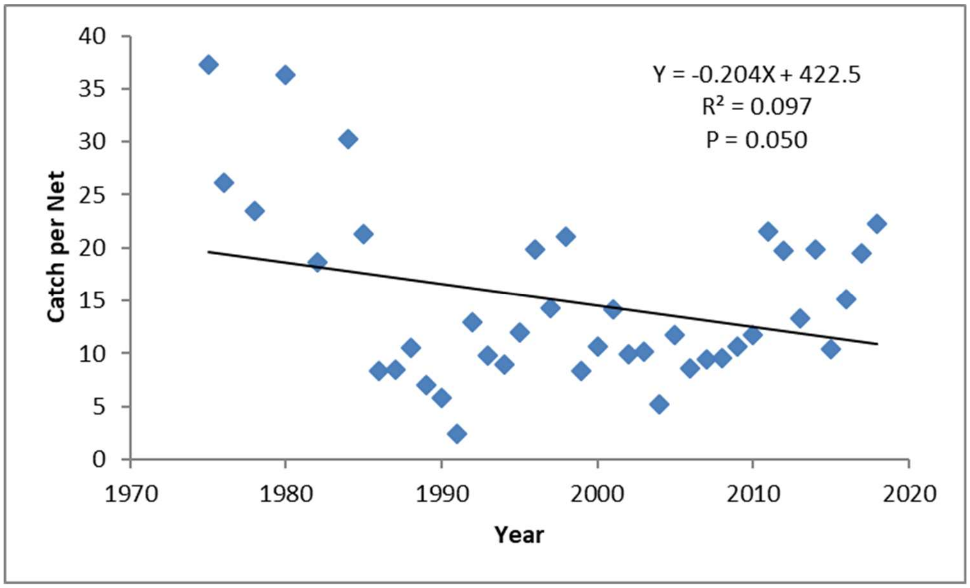


Figure 7-15. Mean catch per net of largescale suckers in sinking gillnets during spring gillnetting at the Rexford site on Libby Reservoir, 1975-2018.

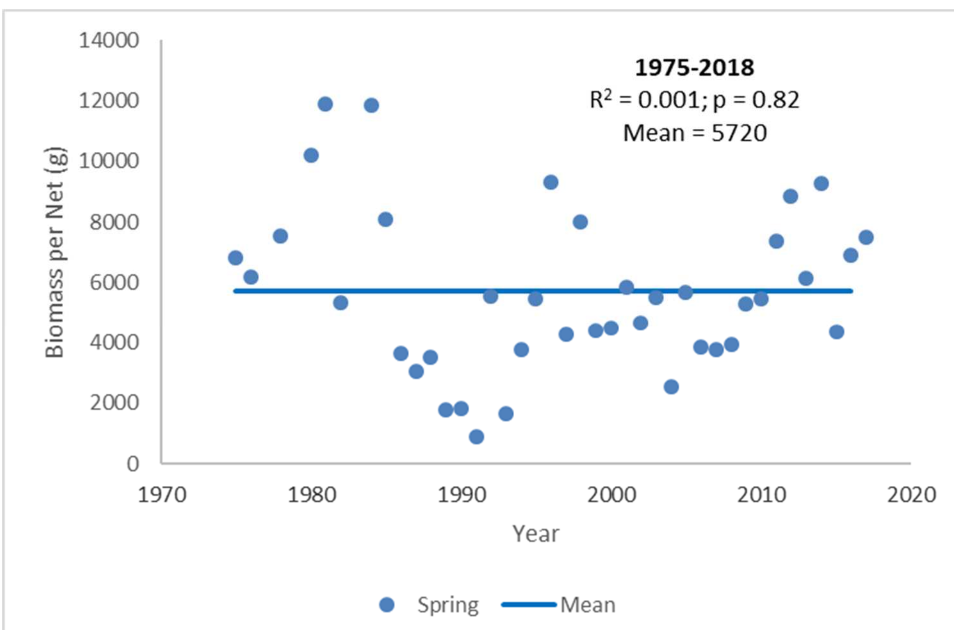


Figure 7-16. Mean biomass (grams per net) of largescale suckers in spring sinking gillnets at the Rexford site in Libby Reservoir since 1975.

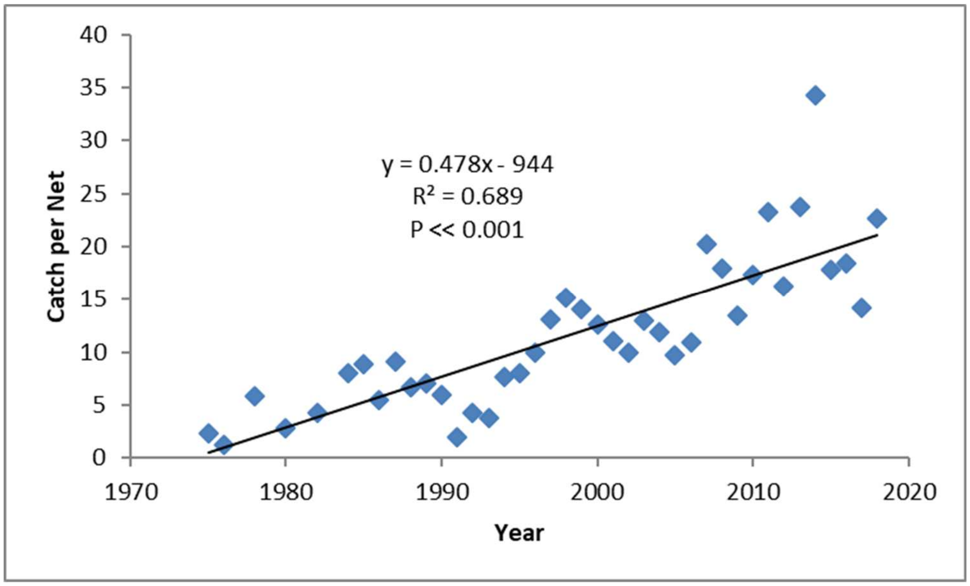


Figure 7-17. Mean catch per net of northern pikeminnow in sinking gillnets during spring gillnetting at the Rexford site on Libby Reservoir, 1975-2018.

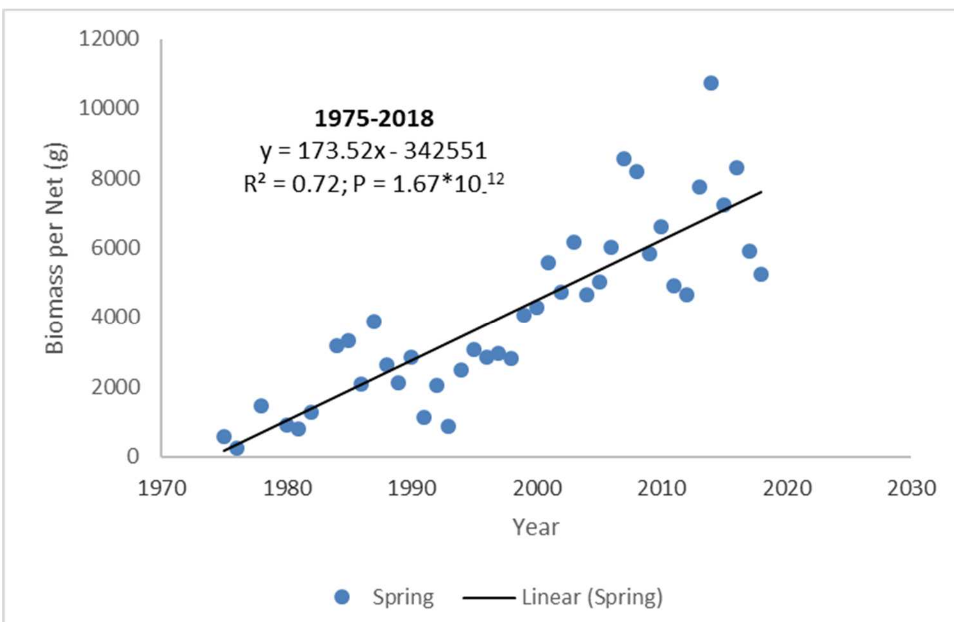


Figure 7-18. Mean biomass (grams per net) of northern pikeminnow in spring sinking gillnets at the Rexford site in Libby Reservoir since 1975.

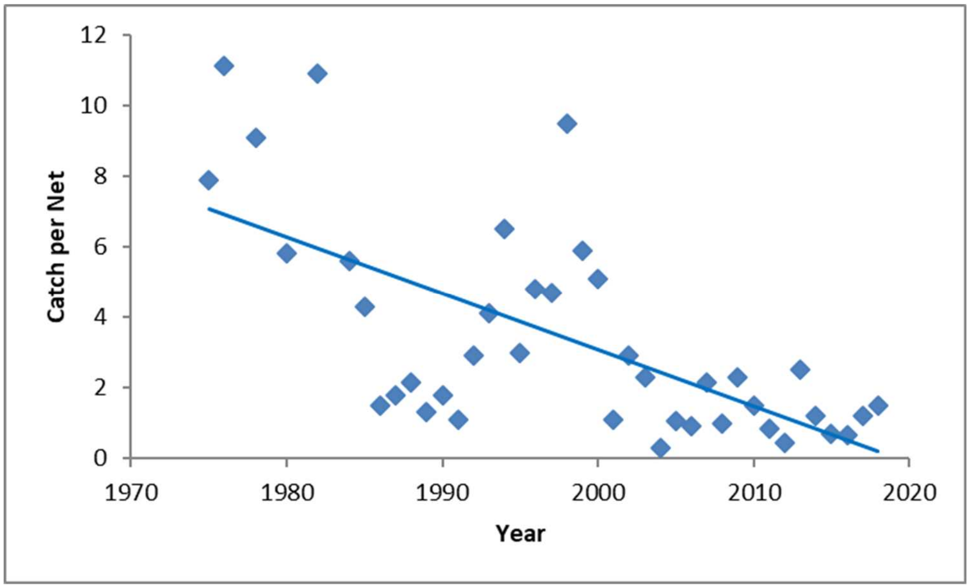


Figure 7-19. Mean catch per net of longnose sucker in sinking gillnets during spring gillnetting at the Rexford site on Libby Reservoir, 1975-2018.

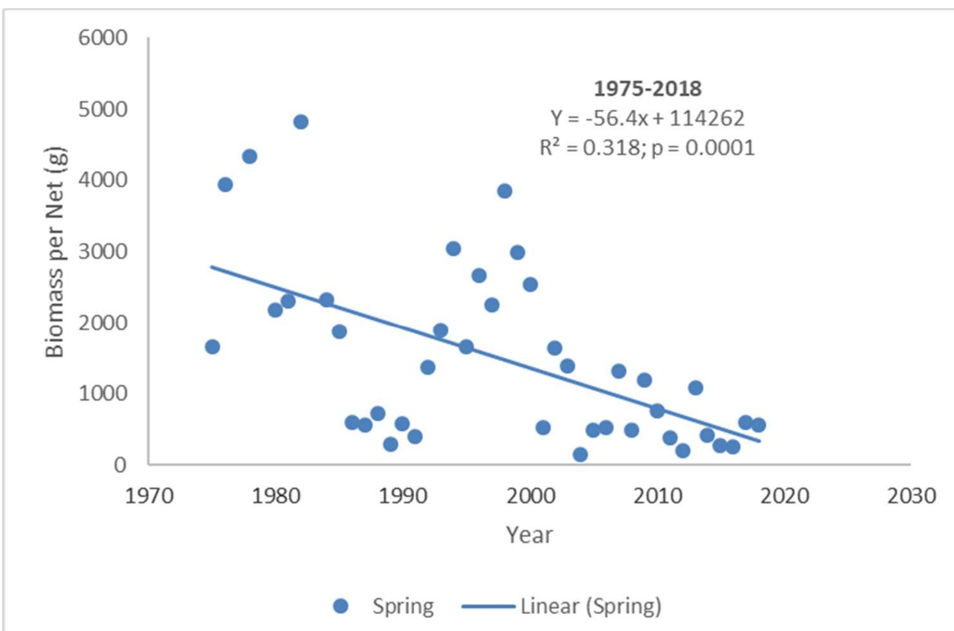


Figure 7-20. Mean biomass (grams per net) of longnose sucker in spring sinking gillnets at the Rexford site in Libby Reservoir since 1975.

Total Fish Abundance and Species Composition

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 exhibited a weak but significant increase since impoundment (Figure 7-21; $r^2 = 0.12$; $p = 0.027$; Table 7-3), increasing on average by 1.03 fish per year per net. We captured an average of 125.4 fish per net in 2018. The mean biomass per net of the spring nets has also significantly increased since reservoir impoundment (Figure 7-22; $r^2 = 0.27$; $p < 0.001$), increasing on average by 387 grams per net per year. The increase in biomass since impoundment is primarily attributable to the shift in species composition that has occurred over the life of the reservoir. Increases of biomass of non-game species such as northern pikeminnow and Columbia River Chubs, and to a lesser extent increases in the abundance and biomass of bull trout coupled with decreases in abundance and biomass of rainbow and cutthroat trout and mountain whitefish are responsible for this shift. Mean relative abundance and biomass of all species during the fall netting efforts have not exhibited a significant trend since impoundment (Figures 7-21 and 7-22, respectively). Total catch (fish per net) for fall gillnets in 2018 averaged 18.1 fish/net and mean biomass per net in 2018 averaged 2,983 grams per net, which was the second lowest on record which represented a 67% reduction from the mean biomass per net of 9,095 since 1975.

Despite the dramatic shifts in species composition that have occurred since impoundment, species composition for the catch of fall and spring gillnets has remained relatively stable over the past 17 years (Table 7-5). Columbia River Chubs, northern pikeminnows, and coarse scale suckers have been the most abundant fish (in descending order) captured in the spring gillnets, comprising 55.2, 18.6, and 14.2% of the catch respectively since 2002. The remaining species make up 12.0% of the catch on average (Table 7-5). The species composition captured in the fall gill netting is similar. Columbia River Chubs and northern pikeminnow are the two most abundant species captured, comprising 38.1 and 31.1% of the total catch. However, kokanee salmon are on average the third most abundant species captured, comprising on average 22.8% of the total catch (Table 7-5). However, kokanee salmon catch in 2017 in the fall nets represented 15.0% of the catch. Since 2002, *Oncorhynchus* species (excluding kokanee) have comprised on average only 0.9 and 2.2% of the total catch on average during the spring and fall gill netting, respectively, but in 2018, the catch averaged 0.4% for the spring and fall nettings efforts.

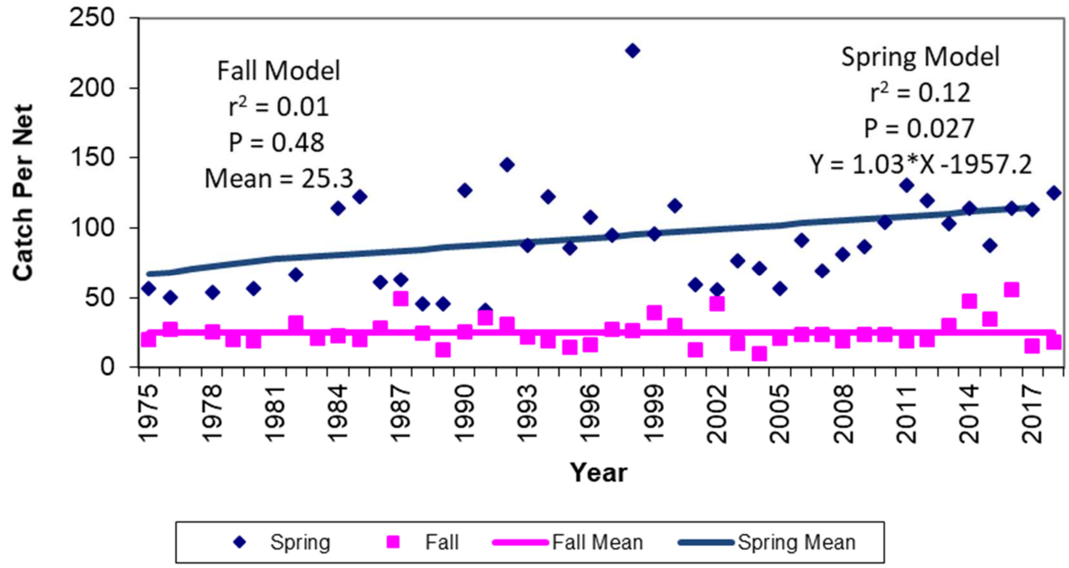


Figure 7-21. Catch per net (all species combined) in fall floating and spring sinking gillnets (1975-2018) and associated trend lines in Libby Reservoir.

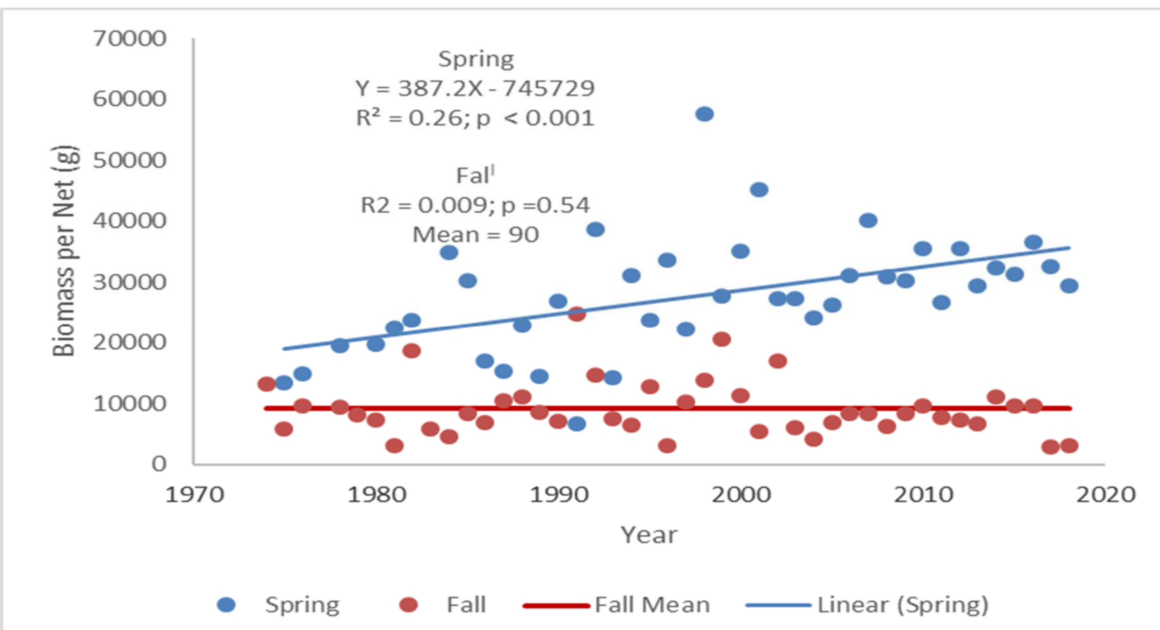


Figure 7-22. Mean biomass per net (all species combined) in fall floating and spring sinking gillnets (1975-2018) and associated trend lines in Libby Reservoir.

Table 7-3. Average catch per net for the most common fish species* captured in the spring sinking gillnets set during spring in the Rexford site of Libby Reservoir, 2002 through 2018.

YEAR																	
	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Date	5/13	5/13	5/11	5/10	5/10	5/21	5/13	5/18	5/17	5/16	5/14	5/13	5/19	5/11	5/16	5/16	5/7
Number Nets	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14
Res. Elev.	2384	2417	2419	2425	2424	2408	2397	2406	2411	2341	2399	2409	2391	2442	2416	2378	2374
	Average number of fish caught per net for individual fish species																
RBT	0.4	0.7	0.6	0.4	0.3	1.5	0.4	0.2	0.2	0.9	0.4	1.4	0.5	0.4	1.4	1.1	0.4
WCT	0.0	0.2	0.2	0.1	0.0	0.1	0.1	0.0	0	0.5	0.1	0.1	0.1	0.3	0.2	0.1	0.1
RBT X WCT	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0	0.0
SUB-TOTAL	0.6	0.9	0.8	0.5	0.3	1.6	0.5	0.2	0.2	1.4	0.5	1.5	0.6	0.7	1.6	1.2	0.5
MWF	1.2	1.2	0.5	0.2	0.8	0.9	0.6	0.4	0.8	0.9	0.9	1.9	0.6	0.1	1.0	0.5	0.9
CRC	24.1	42.1	44.4	23.1	63.9	26.1	45.2	54.8	67.7	74.4	75.3	49.7	46.1	51.4	70.4	71.1	66.1
NPM	9.9	13.0	11.9	9.7	10.9	20.3	17.9	13.5	17.4	23.3	16.3	23.7	34.2	17.9	18.4	14.2	22.6
RSS	0.0	0.1	0.0	0.0	0.0	0.4	0	0.2	0	0.1	0.1	0.4	0.1	0.0	0.0	0.1	0.1
BT	4.9	5.4	6.4	5.9	4.4	4.5	5.4	3.1	4.4	1.1	4.1	4.3	3.6	5.4	5.6	2.7	3.6
Ling	0.1	0.3	0.1	0.2	0.2	0.1	0	0	0	0.1	0.1	0.1	0.1	0.0	0.0	0.1	0.1
CSU	9.9	10.2	5.2	11.8	8.6	9.4	9.6	10.7	11.8	21.6	19.7	13.3	19.9	10.4	15.1	19.4	22.2
FSU	2.9	2.3	0.3	1.1	0.9	2.1	1.0	2.3	1.5	0.9	0.4	2.5	1.2	0.7	0.6	1.2	1.5
YP	0.6	0.1	0.5	0.4	0.4	1.6	0.4	0.2	0.4	0.8	1.2	1.8	1.6	0.1	0.1	1.9	5.6
KOK	1.0	1.2	0.9	3.4	0.6	2.1	0.8	1.4	0.2	5.8	0.9	3.4	1.4	0.6	0.4	1.1	2.2
TOTAL	55.2	76.8	70.9	56.4	91.2	69.4	81.6	87.0	104.3	130.9	119.6	103.1	114.2	87.4	114.0	113.6	125.4

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

Table 7-4. Average catch per net for the most abundant fish species* captured in floating gillnets set during the fall in Rexford site of Libby Reservoir, 2002 through 2018.

YEAR	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Date	9/10	9/16	9/14	9/21	9/13	9/11	9/16	9/12	9/15	9/20	9/20	9/24	9/23	9/16	9/15	9/21	9/19
Number Nets	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14	14
Res. Elev.	2441	2435	2445	2437	2441	2437	2441	2444	2441	2447	2450	2449	2448	2440	2446	2443	2442
	Average number of fish caught per net for individual fish species																
RBT	0.5	0.8	0.4	0.3	0.6	0.5	0.2	0.6	0.2	0.4	0.3	0.2	0.4	0.5	0.3	0.3	0.1
WCT	0.1	0.1	0.0	0.2	0.1	0.1	0.1	0.1	0.0	0.0	0.1	0.1	0.1	0.1	0.1	0.1	0.0
RBT X WCT	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0	0.0
SUB-TOTAL	0.6	1.0	0.4	0.5	0.7	0.6	0.3	0.7	0.2	0.4	0.5	0.3	0.5	0.6	0.4	0.4	0.1
MWF	0.4	0.4	0.6	0.3	0.6	0.7	0.0	0.6	0.3	0.1	0.3	1.3	0.4	0.4	0.2	0.5	0.0
CRC	21.4	5.0	1.6	7.1	9.9	9.6	9.1	9.9	11.1	3.9	10.6	5.4	7.2	2.0	11.5	6.3	9.2
NPM	8.1	3.4	3.3	4.9	5.6	10.0	4.1	5.6	8.2	6.5	6.5	5.0	12.4	6.1	8.1	5.3	5.8
RSS	0.3	0.1	0.0	0.1	0.4	0.6	0.0	0.4	0.0	0.0	0.2	0.1	0.1	0.0	0.1	0.0	0.0
BT	0.1	0.0	0.2	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.2	0.0	0.1	0.1	0.1	0.0
CSU	0.1	0.2	0.2	0.1	0.3	1.0	0.1	0.3	1.4	0.1	0.1	0.4	0.3	1.4	0.1	0.4	0.4
KOK	14.2	7.4	3.5	7.9	5.4	0.8	4.9	5.4	2.1	8.4	1.8	2.4	1.6	6.6	3.4	2.4	2.7
TOTAL	45.2	17.5	9.7	21.3	23.1	23.4	18.7	23.1	23.3	19.4	20.1	15.0	22.4	17.2	23.9	15.3	25.3

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

Table 7-5. Percent composition of major fish species* caught in fall floating and spring sinking gillnets in Libby Reservoir, 2002 through 2018.

	Year																	
Species	2002		2003		2004		2005		2006		2007		2008		2009		2010	
	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall
RBT	0.7	1.1	0.9	4.5	0.8	3.7	0.3	2.2	0.7	1.1	0.9	4.5	0.8	3.7	0.2	2.8	0.2	0.9
WCT	0.0	0.2	0.3	0.8	0.3	0.0	0.0	1.7	0.0	0.2	0.3	0.8	0.3	0.0	0.0	0.6	0.0	0.0
HYB	0.4	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0
ONC	1.1	1.3	1.2	5.7	1.1	3.7	0.3	3.9	1.1	1.3	1.2	5.7	1.1	3.7	0.2	3.4	0.2	0.9
MWF	2.2	0.8	1.6	2.0	0.7	5.9	0.7	1.7	2.2	0.8	1.6	2.0	0.7	5.9	0.4	2.8	0.8	1.2
CRC	43.7	47.4	54.9	28.6	62.6	16.2	42.2	43.6	43.7	47.4	54.9	28.6	62.6	16.2	63.0	42.9	64.9	47.5
NPM	17.9	18.0	16.9	19.2	16.8	33.8	18.6	30.4	17.9	18.0	16.9	19.2	16.8	33.8	15.5	24.1	16.6	35.3
RSS	0.0	0.6	0.1	0.4	0.0	0.0	0.0	0.0	0.0	0.6	0.1	0.4	0.0	0.0	0.2	1.9	0.0	0.0
FSU	5.3	0.0	3.0	0.4	0.4	0.0	1.9	0.0	5.3	0.0	3.0	0.4	0.4	0.0	2.6	0.0	1.4	0.0
CSU	17.9	0.2	13.3	1.2	7.4	2.2	24.0	2.2	17.9	0.2	13.3	1.2	7.4	2.2	12.3	1.2	11.3	6.1
KOK	1.8	31.4	1.6	42.4	1.2	36.0	0.3	18.2	1.8	31.4	1.6	42.4	1.2	36.0	1.6	23.5	0.2	8.9
YP	1.1	0.2	0.1	0.0	0.7	0.0	2.7	0.0	1.1	0.2	0.1	0.0	0.7	0.0	0.2	0.3	0.3	0.0
BT	8.9	0.2	7.0	0.0	9.0	2.2	9.1	0.0	8.9	0.2	7.0	0.0	9.0	2.2	3.6	0.0	4.2	0.0

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

Table 7-5 (Continued). Percent composition of major fish species* caught in fall floating and spring sinking gillnets in Libby Reservoir, 2002 through 2018.

	Year																	
Species	2011		2012		2013		2014		2015		2016		2017		2018		Mean	
	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall	Spr.	Fall
RBT	0.7	2.2	0.4	1.4	1.3	0.7	0.4	0.8	0.5	1.5	1.2	0.5	0.9	1.9	0.3	0.4	0.7	1.7
WCT	0.4	0.0	0.1	0.7	0.1	0.5	0.1	0.3	0.3	0.2	0.2	0.1	0.1	0.5	0.1	0.0	0.1	0.4
HYB	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ONC	1.0	2.2	0.4	2.5	1.4	1.2	0.6	1.1	0.8	1.7	1.4	0.6	1.0	2.3	0.4	0.4	0.9	2.2
MWF	1.1	0.4	0.8	1.4	1.8	4.3	0.6	0.8	0.2	1.2	0.9	0.4	0.4	3.3	0.7	0.0	0.9	1.9
CRC	56.9	20.2	62.9	53.0	48.2	35.7	40.3	31.6	58.8	11.6	61.7	55.5	62.6	41.1	52.7	50.8	55.2	38.1
NPM	17.8	33.5	13.6	32.4	23.0	33.3	30.0	55.7	20.4	35.7	16.1	28.9	12.5	34.6	18.1	31.9	18.6	31.1
RSS	0.1	0.0	0.1	1.1	0.4	0.2	0.1	0.3	0.0	0.0	0.0	0.3	0.1	0.0	0.1	0.0	0.1	0.6
FSU	0.7	0.0	0.4	0.0	2.4	0.0	1.1	0.0	0.8	0.0	0.6	0.0	1.1	0.0	1.2	0.0	1.7	0.0
CSU	16.5	0.4	16.5	0.7	12.9	1.2	17.4	0.6	11.8	4.1	13.3	0.3	17.1	2.3	17.7	2.4	14.2	1.8
KOK	4.4	43.4	0.8	8.9	3.3	15.7	1.2	6.6	0.7	38.2	0.3	12.0	1.0	15.9	1.8	15.0	1.8	22.8
YP	0.6	0.0	1.0	0.4	1.7	0.0	1.4	0.3	0.2	0.0	0.1	0.1	1.6	0.0	4.5	0.0	1.0	0.1
BT	1.4	0.0	3.4	0.0	4.2	0.7	3.1	0.0	6.1	0.2	4.9	0.1	2.4	0.5	2.8	0.0	5.3	0.3

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

Conclusions

The construction of Libby Dam and subsequent inundation of the Kootenai River substantially and irreversibly changed the fisheries community within that ecosystem. The fundamental change has been a shift in species composition from a species community with life history types that thrive in lotic environments to those with life histories which favor lentic environments. Rainbow trout, cutthroat trout, mountain whitefish, and longnose sucker abundance and biomass precipitously declined, with a nearly simultaneous increase of abundance and biomass of many of the minnow species including Columbia River Chubs, and northern pikeminnow. Kokanee salmon were introduced in the British Columbia portion of the watershed, but now constitute approximately one quarter of the catch in the fall netting. Bull trout populations have likely benefited from the introduction of kokanee salmon in the reservoir. The abundant food source the kokanee salmon provide, in conjunction with the harvest restrictions imposed when bull trout were listed as threatened in 1998 (USFWS 1999b) likely resulted in increased abundance of bull trout in the reservoir. The increased catch of bull trout in the spring nets is strongly correlated with redd counts in the Wigwam River and Grave Creek (Dunnigan et al. 2016). Burbot catch in the spring nets exhibited an increase in abundance from 1975-1988, but then declined, and has continued at low catch rates during the past several years. The cause of the decline in burbot abundance is not known, but likely a result of decreased recruitment and early life stage survival.

Libby Reservoir supported an estimated 18,726 angler days during the 2017 angling season (March 1, 2017 to February 28, 2018; Montana FWP 2018), with most the anglers targeting kokanee salmon and inland rainbow trout (MFWP, unpublished data). MFWP's gillnetting efforts on Libby Reservoir provide important monitoring data to evaluate reservoir operations on resident fish relative abundance, condition and length used to evaluate hydro operations, angling regulations, and recovery efforts of listed and imperiled species.

Chapter 8: Zooplankton Trend and Status Monitoring in Libby Reservoir

This chapter includes the following work elements:

E and F: Monitor trend and status of focal species in MT portion of the Kootenai Basin (Contracts 77012 and 76916).

J: Analyze and interpret Libby Mitigation physical and biologic data, (Contracts 77012 and 76916).

Introduction

Early researchers on Libby Reservoir identified the importance of zooplankton communities in supporting viable populations of resident sport fish within the reservoir (Chisholm et al. 1989). The zooplankton community of Libby Reservoir forms a critical link between phytoplankton and secondary consumers. Zooplankton constitutes a substantial portion of the diets of many Libby Reservoir fishes including: kokanee salmon, coarse scale suckers and Columbia River chub, rainbow trout, and mountain whitefish (Chisholm et al. 1989). Many of these species in turn serve as an important food source for piscivorous bull trout and large rainbow trout. MFWP has collected zooplankton from Libby Reservoir since 1983 to relate changes in density and structure of the zooplankton community to parameters of the fish communities, and to collect data indicative of reservoir processes, including aging and the effects of reservoir operation.

Methods

Protocol Title: MFWP Fish Population Monitoring - Reservoirs v1.0

Protocol Link: http://www.monitoringmethods.org/Protocol/Details/511

Protocol Summary: Monitor trends in abundance (i.e., CPUE), species composition, mean lengths, and condition of the fish communities in large reservoirs and lakes within the Flathead and Kootenai River drainages. Monitoring is completed using sinking and floating gill nets depending on season at standardized locations throughout all or sections of each waterbody.

MFWP divided Libby Reservoir into three areas for study purposes (Huston et al. 1984; Chisholm and Fraley 1986) based on reservoir morphometry, effects of reservoir drawdown, and political boundaries. The three strata were Tenmile (RM 221.7 to RM 256.2), Rexford (RM 256.2 to RM 271.0) and Canada (RM 271.0 to RM 291.6). We performed monthly vertical zooplankton tows using a 0.3 m, 153μ Wisconsin net in each of three reservoir areas (Tenmile, Rexford and Canada) from 1983 to 1996. However, from 1997 to present, 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). Furthermore, from 1990 to present, we randomly selected three sampling locations (reservoir mile) and bank (east, west or middle) within each of the three reservoir areas for monthly sampling. All samples were pulled at a rate of 1 m/second to minimize backwash (Leathe and Graham 1982).

Hoffman et al. (2002) further standardized sampling methodologies after analyzing the effects of sample depth had on abundance and species composition. They concluded total zooplankton abundance of samples taken from greater than 20 m were statistically similar to samples taken from 30 m (Kruskal-Wallis $p = 0.05$; Hoffman et al. 2002). 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 was greater than 20 m, 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.

Zooplankton samples were preserved in a water / methyl alcohol / formalin / acetic acid solution from September 1983 to November 1986. However, from December 1986 to present, 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 (>500 organisms total) 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 approximately 30 *Daphnia*, *Diaptomus*, *Epischura* and *Diaphanosoma* to estimate abundance within 0.5 mm length classes, and to estimate mean length of each genera.

We estimated mean monthly zooplankton abundance (by genera) within each of the three strata by averaging each of the three replicates per strata. We estimated mean annual zooplankton abundance within Libby Reservoir by averaging monthly abundance estimates within the three strata. We used analysis of variance, and subsequent Tukey's Honestly Significant Difference (HSD) multiple comparisons to assess whether zooplankton abundance differed by month, sampling area and year. We used multiple regression to evaluate changes Daphnia mean size and size structure through time. All statistical analyzes were performed using R (Core Team 2018).

Results

During the months April through November 2017, we estimated that *Cyclops* were the most abundant zooplankton in Libby Reservoir, followed by *Daphnia*, averaging 7.4 and 2.4 organisms/liter, respectively over the season. We estimated a mean annual density of 1.0 *Bosmina* per liter, which made it the third most abundant zooplankton organism in 2017

(Figure 8-1). Other lesser abundant genera in 2017 in decreasing order of mean annual abundance include *Diaptomus* (0.7 organisms/l)*, Diaphanosoma* (0.01 organisms/l)*, Epischura* (41.1 organisms/m^3)*, and Leptodora* (1.0 organisms/m^3) (Figure 8-1; Appendix Tables A6-A9). Mean annual densities of *Daphnia* in 2017 were 22.7% higher than the respective mean since 1997. However, all other organisms were lower than the respective annual means values since 1997. In decreasing order, Diaphanosoma, Leptodora, *Epischura*, *Diaptomus, Bosmina, Cyclops, and Diaptomus* in 2017 were 92.9, 61.2, 45.8, 19.5, 14.1, and 8.5% lower than the respective mean values since 1997. Overall total mean annual zooplankton density in 2017 were 11.5 organisms per liter, which was 10.3% lower than the average since 1997 (12.8 organisms/l).

Trends in zooplankton abundance in Libby Reservoir have generally been of decreasing abundance of the larger and more numerous genera of zooplankton and an increasing abundance of smaller zooplankton since the late 1970s. For example, *Daphnia* and *Diaptomus* have both significantly and negatively decreased in mean annual abundance since 1977 (Figures 8-2). *Epischura* are relatively large zooplanktons that exhibit opposite trends since the 1970s, significantly increasing over time in the reservoir (Figure 8-3). Even though *Epischura* have increased through time, they exist at relatively low levels compared to *Daphnia* and *Diaptomus*. For example, the mean combined annual density of *Daphnia* and *Diaptomus* (1.66 organisms/l) was about 40X higher than the mean annual abundance of *Epischura*. *Diaphanosoma* were first observed in Libby Reservoir in 1988, but have not exhibited a consistent trend since (Figure 8-3). Density of *Diaphanosoma* over the past decade has averaged 0.04 organisms/l. *Bosmina* and *Cyclops* have both significantly increased in abundance since reservoir construction (Figure 8-4), and remain the most abundant genera during most years. The increase in the smaller zooplankton (especially *Cyclops*) was primarily responsible for the observed overall increase in total zooplankton since 1977 in Libby Reservoir (Figure 8-5). Shifts in the zooplankton community since 1977 may be attributable to the aging process of the reservoir and active selection by planktivorous fish like kokanee salmon that were at low relative abundance in the reservoir during the 1970s and early 1980s.

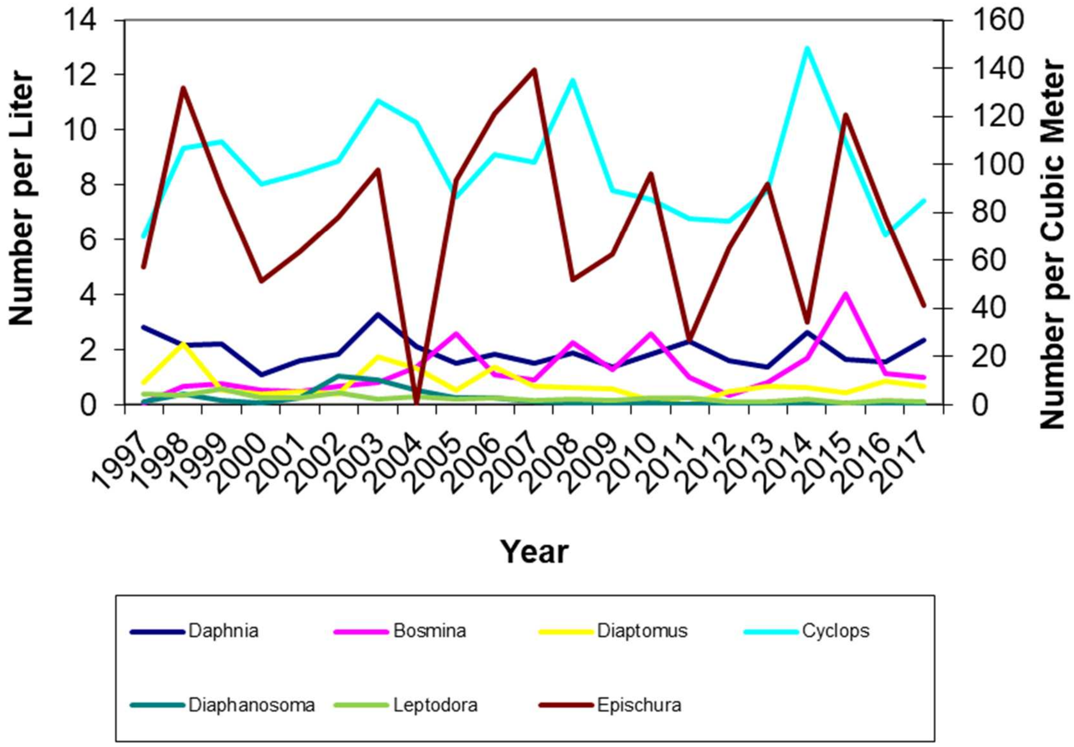
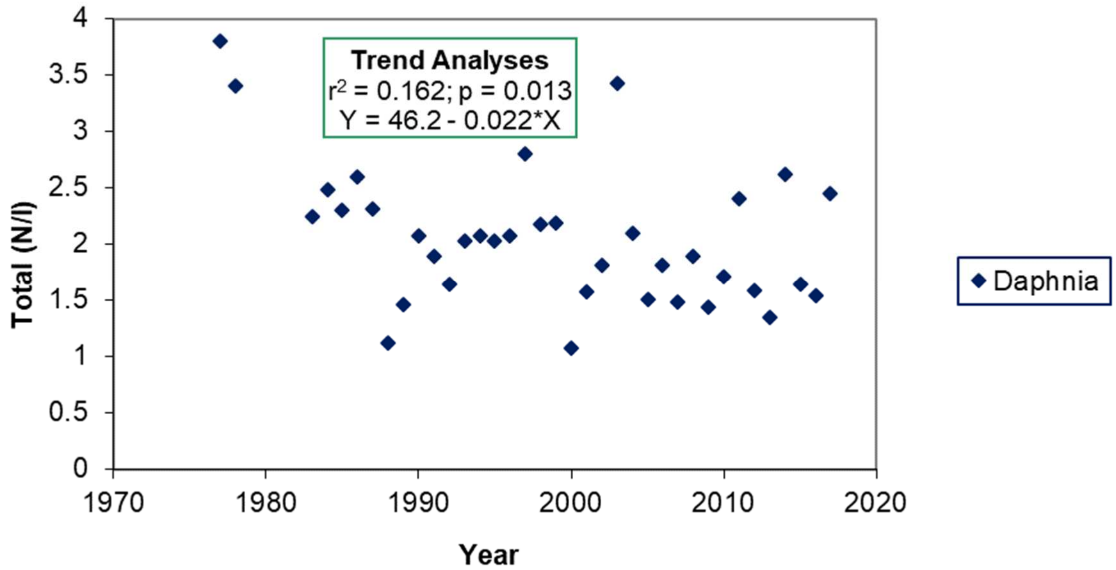


Figure 8-1. Annual zooplankton abundance estimates for seven genera observed in Libby Reservoir from 1997-2017. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter. The data utilized for this figure are presented in Appendix Table A9.



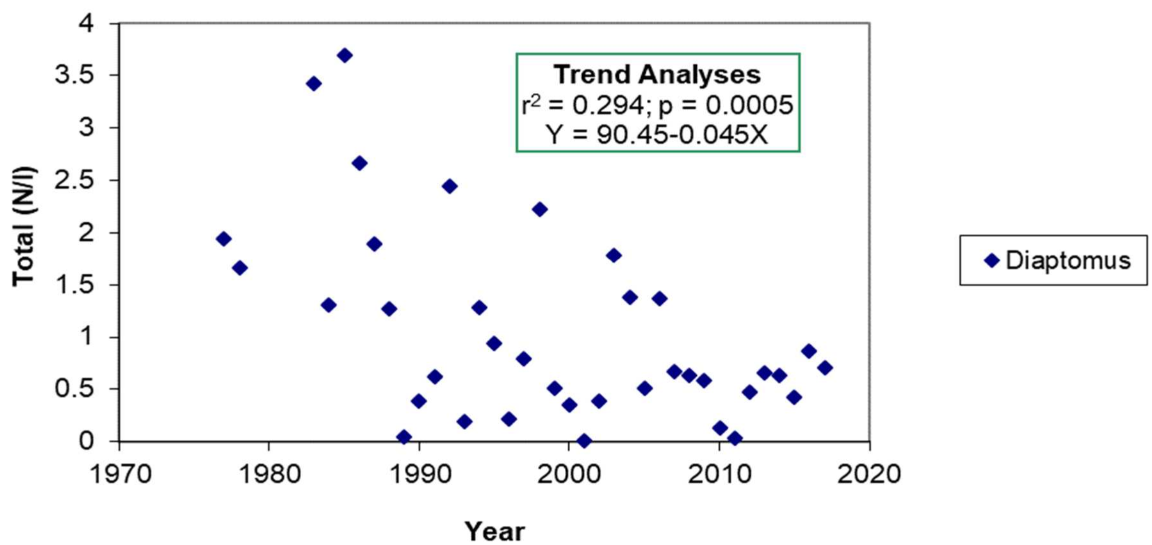
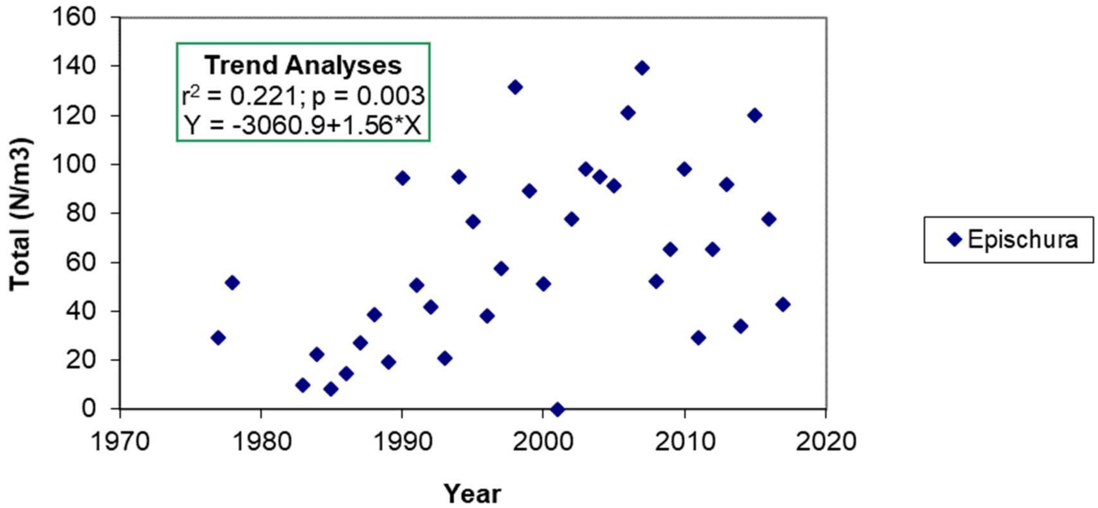


Figure 8-2. Mean annual density (number per liter, N/l) and regression trend analyses of *Daphnia* (top) and *Diaptomus* (bottom) in Libby Reservoir, 1977 through 2017.



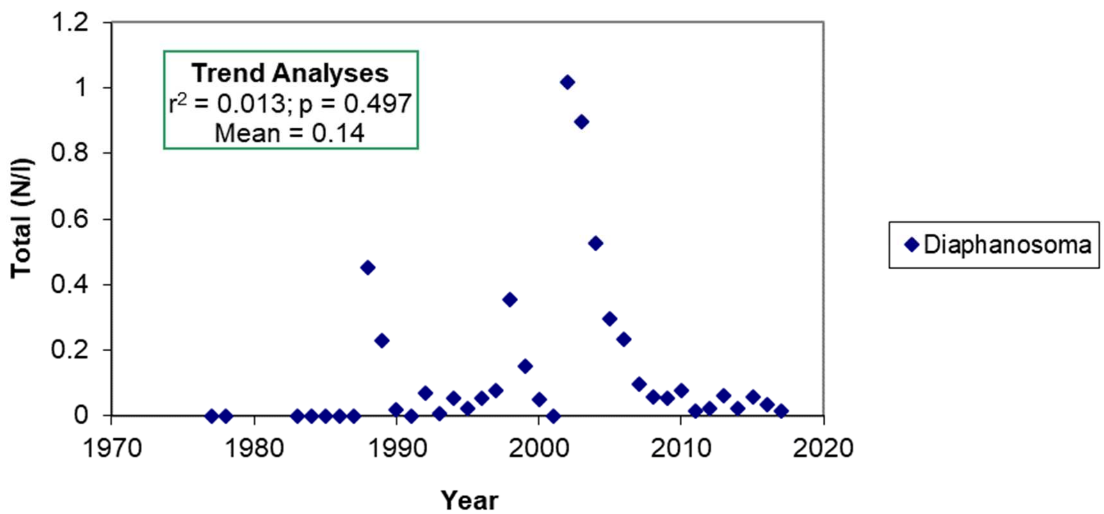


Figure 8-3. Mean annual density and regression trend analyses of *Epischura* (top; number/liter) and *Diaphanosoma* (bottom; number/liter) in Libby Reservoir, 1977 through 2017.

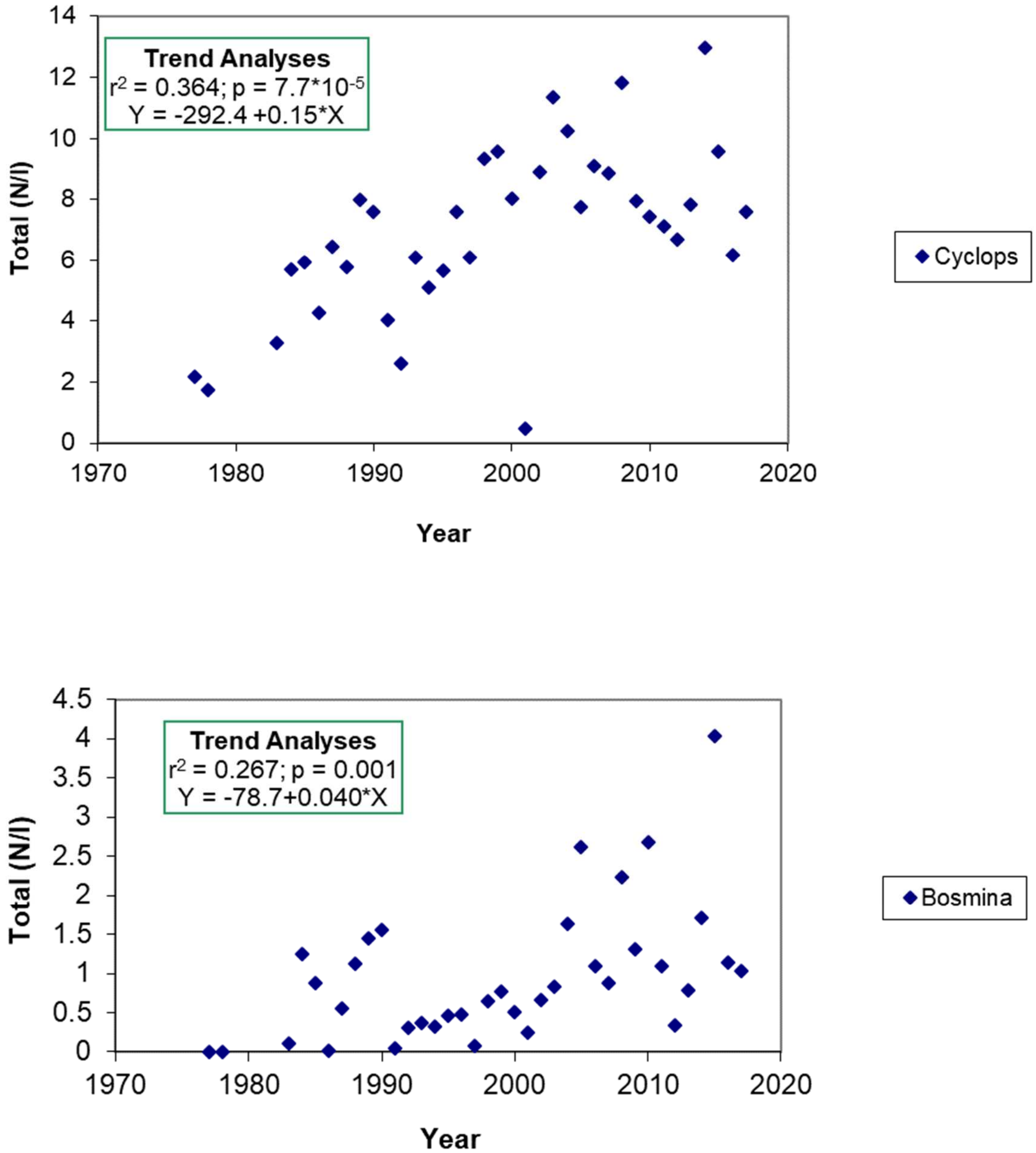


Figure 8-4. Mean annual density (number per liter, N/l) and regression trend analyses of *Cyclops* (top) and *Bosmina* (bottom) in Libby Reservoir, 1977 through 2017.

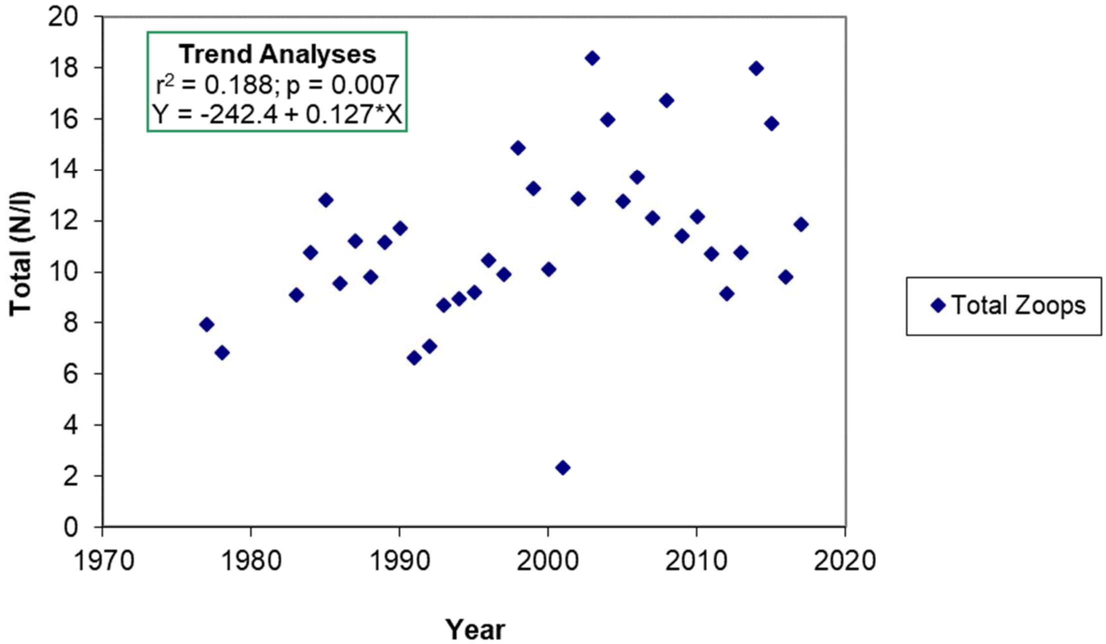


Figure 8-5. Mean annual density (number per liter, N/l) and regression trend analyses of total zooplankton (all species) in Libby Reservoir, 1977 through 2017.

The seasonal peaks in abundance for individual genera of zooplankton we observed in 2017 were generally consistent with the overall seasonal averages and season peaks observed since 1997 (Figure 8-6). For example, *Daphnia* abundance since 1997 usually peaks during June or July. The *Daphnia* abundance in Libby Reservoir in 2017 peaked in June with an estimated 6.8 *Daphnia* per liter (Figure 8-7) which was 86% higher than the mean seasonal peak. *Bosmina* densities have typically peaked twice per year since 1997 which was also the case in 2017 (Figure 8-7). The early peak in 2017 occurred during June (2.4 organisms per liter) and was 19% lower than the mean seasonal peak since 1997 which also typically occurs during May. The second later season peak in *Bosmina* abundance in 2017 occurred in November, which was a month later than it typically peaks, but the second peak in 2017 was 129% higher than second seasonal peak average since 1997. *Cyclops* has peaked in either May or June during 19 of the last 21 years, which was the case in 2017 peaking in May with an average density of 10.2 organisms/l. This peak in 2017 was 32% lower than the seasonal peak since 1997 (Figures 8-2 and 8-3). *Leptodora* abundance in 2017 peaked in July, which was consistent with the pattern of annual peaks since 1997. However, the July peak in 2017 was 57% lower than the typical July peak abundance since 1997. *Diaphanosoma* abundance was low throughout all months in 2017, peaking at 0.04 organisms/l. However, *Diaphanosoma* typically has peaked in late August or September 13 of the previous 21 years. The 2017 monthly peak of *Diaphanosoma* was 94% lower than the mean monthly peak since 1997. *Diaptomus* has peaked in either September or October during 15 of the previous 21 years, but that was not the case in 2017 when it peaked one month earlier in July. However, the July peak in 2017 was 54% higher than the mean seasonal peak since 1997, averaging 2.3 organisms per liter in 2017. *Epischura* typically peaks in abundance during August, as it did in 2017, with a peak that was nearly equal (4% lower) to the mean August peak since 1997. Finally, total zooplankton density in Libby Reservoir since 1997 has peaked in May, but it peaked a month later in June in 2017 (18.1 organisms per liter), with the peak this year being slightly (6%) lower than average. The 2017 peak in May was comprised of primarily *Daphnia* (37.3%) and *Cyclops* (48.5%).

Densities of *Daphnia, Bosmina, Diaptomus, Leptodora, and Epischura* observed in Libby Reservoir in 2017 differed significantly by month ($p < 0.05$; Table 8-1; Figure 8-7). Multiple comparisons for specific monthly differences for each of the seven genera of zooplankton are presented in Table 8-2. We found that abundance of *Diaptomus* also differed significantly by area of the reservoir (Tables 8-1 and 8-3), with mean densities of *Diaptomus* in the Canada area significantly higher than the Tenmile area. Although we could not declare significant differences between the areas for *Leptodora* or *Epischura*, the Month*Area interaction terms were significant ($p < 0.05$; Table 8-1). The significant differences we observed between sampling areas and the interaction terms justify the stratified random sample design we utilized for this study.

The trend in *Daphnia* abundance over the past ten years has not differed from a stable population ($r^2 = 0.05$; $p = 0.450$). *Daphnia* mean length (Figures 8-8 and 8-9) in Libby Reservoir has also remained generally stable during the past several years. Most *Daphnia* since 1997 fall within the size class 0.5 – 0.99 mm (mean annual proportion = 61.8%, standard deviation = 5.5%; Figure 9-8), and most others within the size class 1.0 – 1.499 (mean annual proportion = 33.7%, and standard deviation = 3.7%). *Daphnia* larger than 1.5 mm have on average comprised about 4.3% of the total since 1997 (Figure 8-8). However, in 2017 the size structure of *Daphnia* was slightly skewed toward larger size class, with the proportion of 0.5 - 0.99 mm size class 9.5% lower than average, the 1.0 – 1.499 mm size class 1.9% higher than the average, and the 1.5 – 2.49 mm size class 122.2% higher than average since 1997. The overall mean length of *Daphnia* in 2017 was 0.97 mm, which represented a modest increase (7.1%) from the mean since 1997 (0.906 mm; Figure 8-9). Despite the slight increase in mean size of *Daphnia* observed in 2017, there is evidence that *Daphnia* size structure has shifted toward smaller sizes since 1984. The proportion of *Daphnia* within the 0.5 – 0.99 mm size class has significantly increased on average 0.4% per year since 1984 ($r^2 = 0.212$; $p = 0.006$), while the proportion of *Daphnia* within the 1.0 – 1.49 mm and 1.5 – 1.99 mm size classes decreased, although not significantly ($r^2 = 0.08$; $p = 0.098$ and $r^2 = 0.11$; $p = 0.056$, respectively).

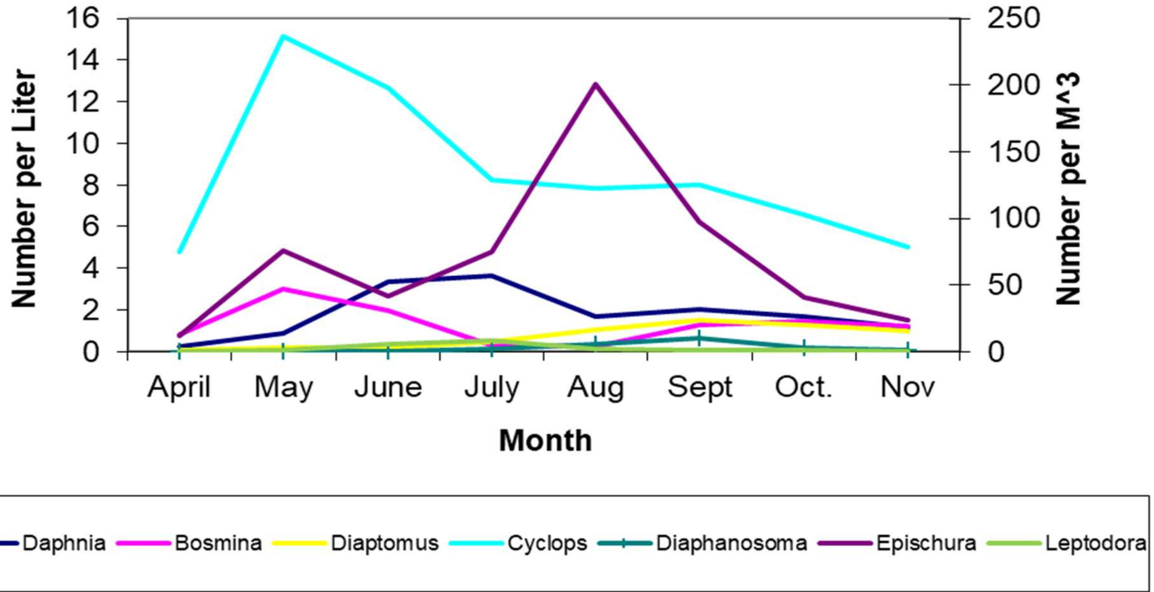


Figure 8-6. Mean monthly zooplankton abundance estimates for seven genera observed in Libby Reservoir from 1997-2017. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter.

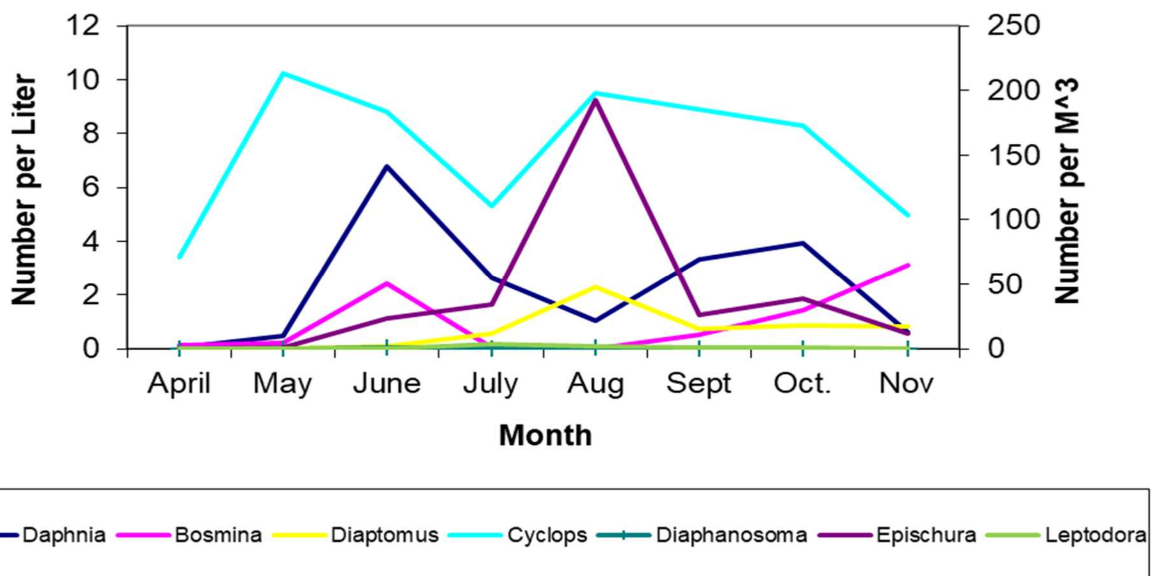


Figure 8-7. Mean monthly zooplankton abundance estimates for seven genera observed in Libby Reservoir in 2017. Abundance for *Epischura* and *Leptodora* are expressed in number per cubic meter. All other densities are expressed as number per liter.

Table 8-1. 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 2017.

Genus	**Month**	**Area**	**Month*Area Interaction**
Daphnia	0.012	0.174	0.627
Bosmina	$1.21*10^{-5}$	0.494	0.173
Diaptomus	$6.25*10^{-9}$	0.002	0.017
Cyclops	0.406	0.770	0.320
Leptodora	$5.30*10^{-7}$	0.270	$5.14*10^{-5}$
Epischura	$6.86*10^{-10}$	0.816	0.006
Diaphanosoma	0.425	0.747	0.767
Total Zooplankton	0.287	0.849	0.302

Table 8-2. Multiple comparisons test results indicating which months differed significantly for each of the seven most abundant zooplankton genera and total zooplankton in Libby Reservoir in 2017. Months that did not differ significantly (alpha = 0.05; Table 9-2) share a common subset.

				Month				
Genus	**April**	**May**	**June**	**July**	**August**	**Sept.**	**Oct.**	**Nov.**
Daphnia	1	2	1,2,3					3
Bosmina	1,2	3,4	1,3,5,6,7	5,8	6,9	7,10		2,4,8,9,10
Diaptomus					All others			
Cyclops								
Leptodora	1	2,3		2,4,5,6	1,3,7	4	5	6,7
Epischura					All others			
Diaphanosoma								
Total Zooplankton								

Table 8-3. Multiple comparisons test results indicating which areas differed significantly for each of the seven most abundant zooplankton genera and total zooplankton in Libby Reservoir in 2017. Areas that differ significantly (alpha = 0.05; Table 9-2) share a common subset.

	Area Subsets		
Genus	**Tenmile**	**Rexford**	**Canada**
Daphnia			
Bosmina			
Diaptomus	1		1
Cyclops			
Leptodora			
Epischura			
Diaphanosoma			
Total Zooplankton			

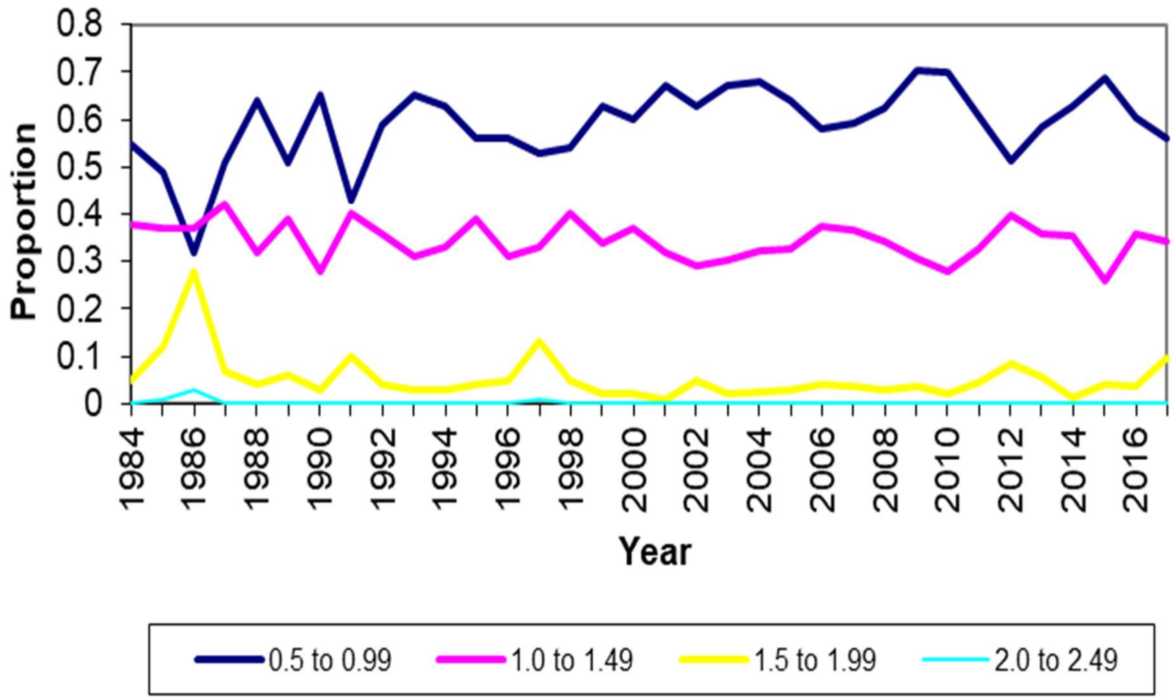


Figure 8-8. *Daphnia* species size composition (mm total length) in Libby Reservoir, 1984 through 2017.

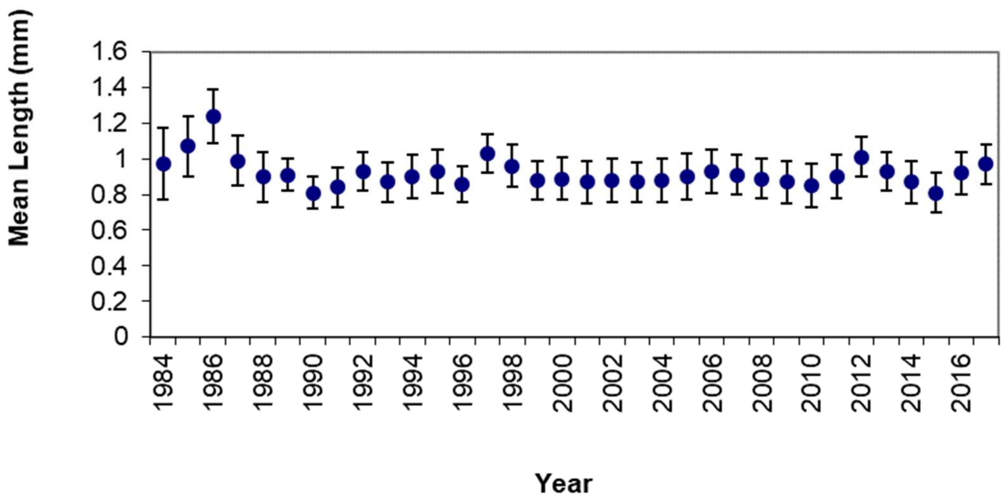


Figure 8-9. Mean length of *Daphnia* species in Libby Reservoir, 1984 through 2017, with error bars representing plus and minus one standard deviation from the mean.

Conclusions

The Zooplankton community in Libby Reservoir has shifted since the 1970s to favor the smaller bodied genera. Mean annual abundance of the two most abundant (and relatively small bodied) genera of zooplankton (*Cyclops* and *Bosmina*) in 2017 in Libby Reservoir were lower than the annual mean since 1997, and the trends of these two genera strongly influence total zooplankton densities. Mean annual *Daphnia* densities in 2017 were slightly higher than average since 1997, but have generally exhibited a significant decreasing trend since the 1970s. However, the timing of the seasonal peaks for the seven most abundant zooplankton genera have remained remarkably consistent over the past several decades. The factors that contribute to these subtle changes within the zooplankton community of Libby Reservoir are likely a complex suite of factors. Nutrient availability and fish predation on zooplankton are thought to be important factors that influence zooplankton population species and size structure. Size and abundance of zooplankton, including those preferentially grazed by planktivorous fish like kokanee salmon, is likely influenced by the production and growth of phytoplankton, which in turn is largely influenced by light and nutrient availability. Changes in the fish community within Libby Reservoir also likely influence the plankton community. For example, Columbia River chub have significantly increased in abundance since the 1970s (see previous chapter), and Dalbey (et al. 1998) found *Daphnia* to be an important food item for Columbia River chub and kokanee salmon. However, the last assessments of primary production on Libby Reservoir occurred about 30 years ago (Chisholm et al 1989). A contemporary assessment of primary production was initiated during the summers of 2016, 2017, and 2018 (see Chapter below). The information from these important research and monitoring efforts will be useful information to fully understand zooplankton abundance and size currently observed in Libby Reservoir. The relative importance of either nutrient availability or selective grazing pressure by planktivorous fish on the plankton community and size structure will require additional investigation.

Zooplankton monitoring conducted by MFWP represents important information used to assess potential changes in the abundance, length and condition of the reservoir fish community, and therefore, we recommend continuing this important baseline monitoring.

Chapter 9: Libby Reservoir Primary Productivity

This chapter includes the following work elements:

I: Estimate primary productivity in Libby Reservoir (Contracts 77012 and 76916)

J: Analyze and interpret Libby Mitigation physical and biological data (Contracts 77012 and 76916)

Introduction

Phytoplankton are photosynthesizing microscopic organisms that inhabit the upper sunlit layer of bodies of fresh water, including Libby Reservoir. Primary production is the creation of organic compounds from inorganic carbon that occurs in the water, a process that sustains the aquatic food web. Phytoplankton obtain energy through the process of photosynthesis and must therefore live in the well-lit surface layer (termed the euphotic zone) of Libby Reservoir. Their cumulative energy fixation in carbon compounds (primary production) is the basis for the food web in Libby Reservoir. The amount of primary production in Libby Reservoir is likely influenced by the availability of nutrients and light, the degree of thermal stratification, algal movements relative to the water, zooplankton grazing, inter-algal species competition, and predation by other organisms.

The primary objective for investigating primary production in Libby Reservoir is to assess the degree to which primary production may be limiting the growth and abundance of zooplankton and ultimately fish populations in the reservoir. The current level of primary production in Libby Reservoir is not known. The most recent assessment of primary production on Libby Reservoir occurred in the mid-1980s (Chisholm et al. 1989), and many of the factors that influence primary production have likely changed since this earlier effort.

In 1986 and 1987 the Montana FWP assessed the limnological conditions within Libby Reservoir (Chisholm et. al. 1989). One of the assessment techniques utilized in that study was to measure the primary production at four locations and 7 depths (1,3,5,10,15,20 and 25m). Chisholm et.al. used the light/dark bottle technique described in Wetzel and Likens (1979). In 2016 the MFW&P decided to replicate the 1986/87 primary productivity study to determine if changes in the reservoir conditions have occurred as the reservoir aged and inputs changed. The assessment of in lake primary productivity was extended to include a total of three consecutive years (2016-2018). The data in this report is from the 2018 assessment.

Although the Wetzel and Likens methodology used in the 1986/87study is a common method, it is possible for differences in how the procedures are implemented to differ between investigators. For instance, the Chisholm et. al. report does not indicate the pore size of the membrane filter used in 1986/87, nor does it address the exact methodology for determining daily productivity using length of incubation versus light intensity and duration. Due to these unknowns, several assumptions have been made based on common methods used in similar assessments.

These assumptions; however, may be in error and could result in differences seen between the 1986/87 assessments and the contemporary assessments. Furthermore, in 2016 - 2018 the samples were filtered through three filter membranes with pore diameters of 0.2um, 2.0um, and 20um. For comparison of the 2016 - 2018 to the 1986/87results, we used only the productivity as determined by radioactivity of the 0.2um filter for comparison to the earlier data.

Methods

Physical measurements were collected concurrently with the PPR assays. A Hydrolab HL4 datasonde with depth, temperature, LDO, pH, specific conductivity, and total dissolved solids (TDS) sensor was lowered from the surface to 30 meters below the surface at a rate of approximately 0.05m/sec. The hydrolab was calibrated following the manufacture's guidelines. D.O. and depth were calibrated in the field immediately prior to the profile at each location. The data was recorded on a Surveyor Hydrolab.

Photosynthetic Active Radiation (PAR) was also collected at each station from June through September. Readings were collected from the surface to the 1% light level in 1m increments.

In addition to physical conditions within the reservoir, water samples were collected for chemical analysis as well. Samples were collected at 3m, 10m, and 20m (when depth allowed) below the water surface during each PPR assessment. Unfiltered sample water was analyzed for nitrite+nitrate, ammonia, total persulfate nitrogen, total phosphorus, and total dissolved solids. Field filtered samples were analyzed for ortho-phosphorus and alkalinity concentrations. All chemical analysis was performed by AmTest Laboratories.

Biological carbon uptake rates (primary productivity) were determined for Libby Reservoir monthly from May through September in 2017. For the months of June, July, August and September the assessment was performed at four locations (Figure 9-1). During the month of May, the assessment was not completed at the Kikomun station due to low reservoir elevation.

Primary productivity was determined using the methods described in Britton and Greeson (1987) and were designed to replicate a study performed on the system in 1986 and 1987. The method described in Britton and Greeson (1987) are based on the same sources as the Wetzel and Likens methodology and will result in comparable results. Water samples were collected from 7 depths in three of the stations and from 5 depths at the Kikomun station due to low water level (Table 9-1).

Figure 9-1. Map of Libby Reservoir showing the approximate locations of the four primary production sampling stations (red dots).

Table 9-1. Primary productivity sample locations, dates, photic zone determination and incubation depths in Libby Reservoir.

Station	Assessment Dates	Assessment Depths
Tenmile	May 22th, 2018 June 20th, 2018 July 17th, 2018 August 22nd, 2018 September 18th, 2018	1,3,5,10,15,20,25m
Stonehill	May 22th, 2018 June 20th 2018 July 17th, 2018 August 22nd, 2017 September 18th, 2018	1,3,5,10,15,20,25m
US/Can Border	May 23rd, 2018 June 19th, 2018 July 16th, 2018 August 21st, 2018 September 17th, 2018	1,3,5,10,15,20,25m
Kikomun	June 19th, 2018 July 16th, 2018 August 21st, 2018 September 17th, 2018	1,3,5,10,15m

Sub-samples of the water taken at depth were placed into three 300mL BOD bottles, two clear bottles (light), and one bottle that was opaque (dark). The bottles were placed into a cooler under low light conditions. After all the samples for the assessment station were collected, 3μCi of ^{14}C was added to each bottle. The bottles were incubated at the depths corresponding to their collection depth for approximately 4 hours. At the end of the incubation period the bottles were retrieved and placed into coolers and kept in the dark. Upon arrival at the filtering station one hundred millimeters of water was filtered through three different pore sized membrane filters (20, 2 and 0.2 μm). The membrane filters were placed in scintillation vials with two drops of 1N HCL acid. The following day the scintillation vials were filled with 5mL of scintillation cocktail.

Each of the filters were taken to the University of British Columbia, Vancouver, BC for analysis using a scintillation counter to determine radioactivity of the filters. The radioactivity of filters is directly related to the amount of carbon uptake that occurred during the incubation period i.e. primary production. Based on the decays per minute, incubation time, sample volume, and the existing pool of dissolved inorganic carbon (DIC) it is possible to calculate the rate of carbon uptake within the system at various depths. The size fractionation also provides information regarding the most productive fraction of the phytoplankton community.

In 2018 an additional analysis was added to the project protocol. In addition to measuring primary production rates a measurement of the chlorophyll *a* concentrations in the upper 10 meters of the reservoir was conducted at each station concurrent with the collection of the water chemistry samples. Water samples were collected at 1, 3, 5, and 10 meters below the water surface. An equal volume of water from each depth was added to a 2 liter Nalgene bottle. The bottle was then placed on ice in a darkened cooler until returning to the filtering station later that day. A known quantity (250 to 500mL) of sample water was filtered through either a 0.2, 2.0 or 20 micron polycarbonate filter. The filters were then folded, placed in labeled aluminum foil envelopes and frozen until analysis. The filters were then analyzed to determine chlorophyll *a* concentrations within the reservoir and the percentage of the chlorophyll *a* that was in each size fraction.

Results

Vertical Profiles

The complete vertical profile data set can be found in Appendix Table A9. The two parameters of most interest for the PPR assessments are temperature and dissolved oxygen. The Kikomun site exhibited little to no change in dissolved oxygen with depth in June, July, or September; however, a distinct reduction in D.O. concentration at depths great than 13 meters was observed in August (Figure 9-2). The temperature profile showed weak stratification in June and September. There was a distinct thermocline present in July at 4 meters and in August at 6 meters.

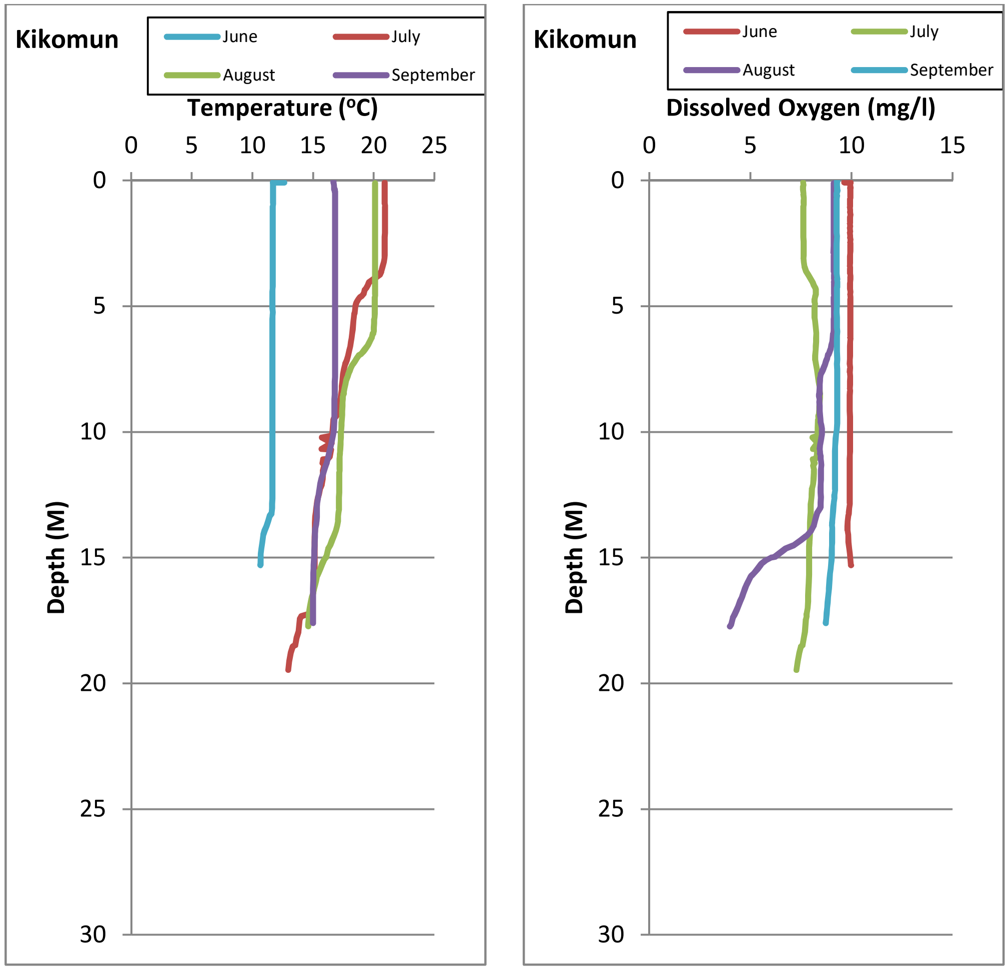


Figure 9-2. Monthly 2018 temperature and dissolved oxygen profiles for the Kikomun station.

At the US/Can border surface warming was observed in May and June in the upper 1 to 2 meters. In July there was a poorly developed thermocline observed around 4 meters. The upper 8 meters were consistent in temperature in August. The water temperature dropped gradually from 8 meters to 30 meters; however, there was not a distinct division between the epilimnion and hypolimnion (Figure 9-3). The surface of the reservoir was cooling by September which resulted in consistent water temperature down to 24 meters. During August and September there was a dissolved oxygen metalimnetic minima at the US/Can border. Metalimnetic minimas are a result of decaying organic matter that accumulates at the density gradients caused by temperature differences within the water column.

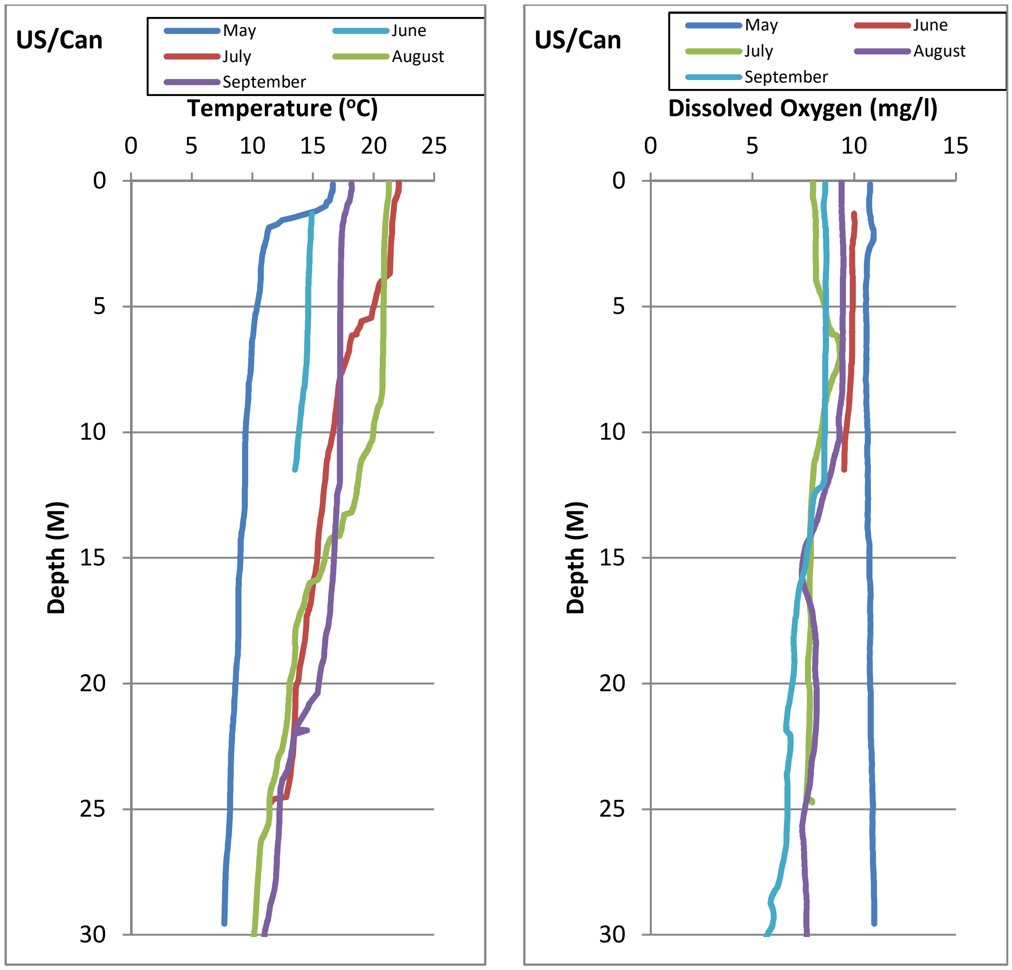


Figure 9-3. Monthly 2018 temperature and dissolved oxygen profiles for the US/Can Border station.

A weak spring thermocline existed in both the Stonehill (Figure 9-4) and Tenmile (Figure 9-5Figure) stations in May but was gone during the June sampling event. The thermocline was well developed at the Stonehill site in July but missing in August before returning in September with a depth of 16 meters. A thermocline was not evident in the Tenmile site until August at a depth of over 18 meters. The transient nature of thermal stratification at the Stonehill and Tenmile stations indicates that the lower reservoir may be influenced by the strong winds that are prevalent in these stations, particularly the Stonehill site. This may have a large impact on the nutrient supply available for primary production in these lower stations, particularly in the late summer.

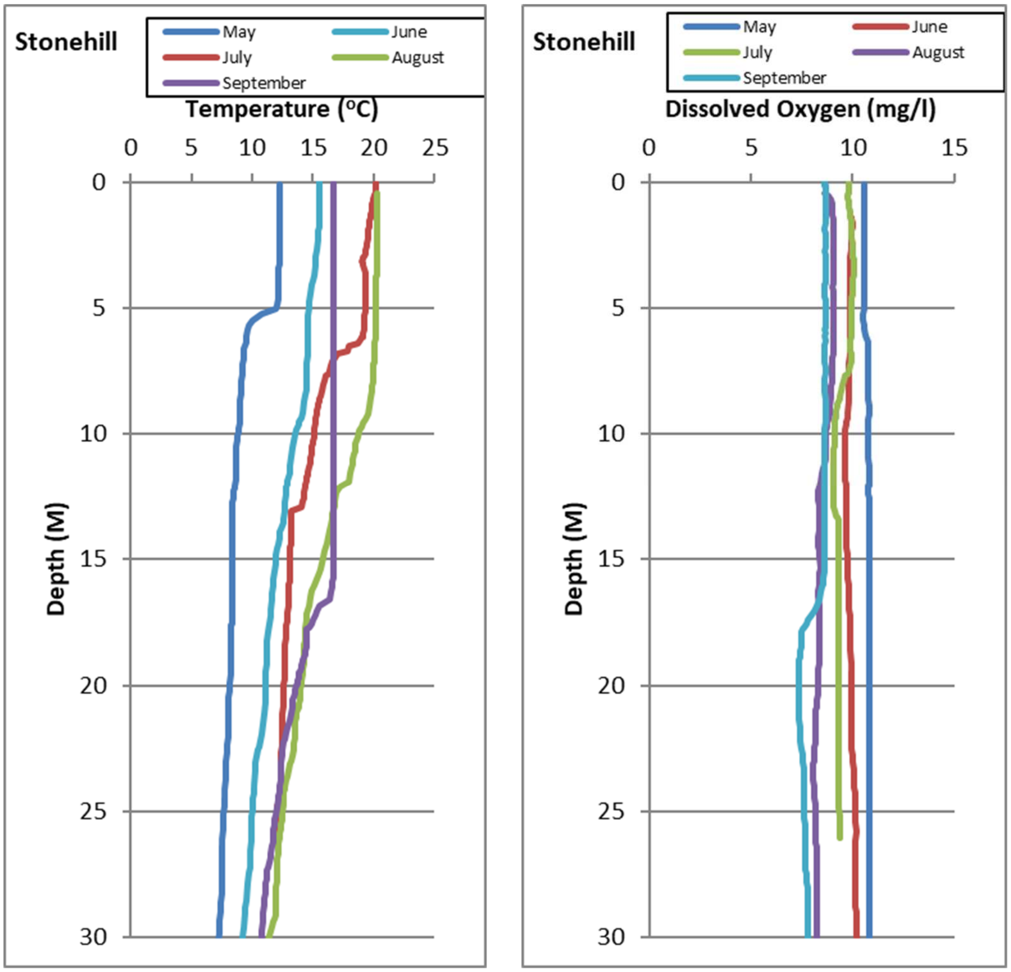


Figure 9-4. Monthly 2018 temperature and dissolved oxygen profiles for the Stonehill station.

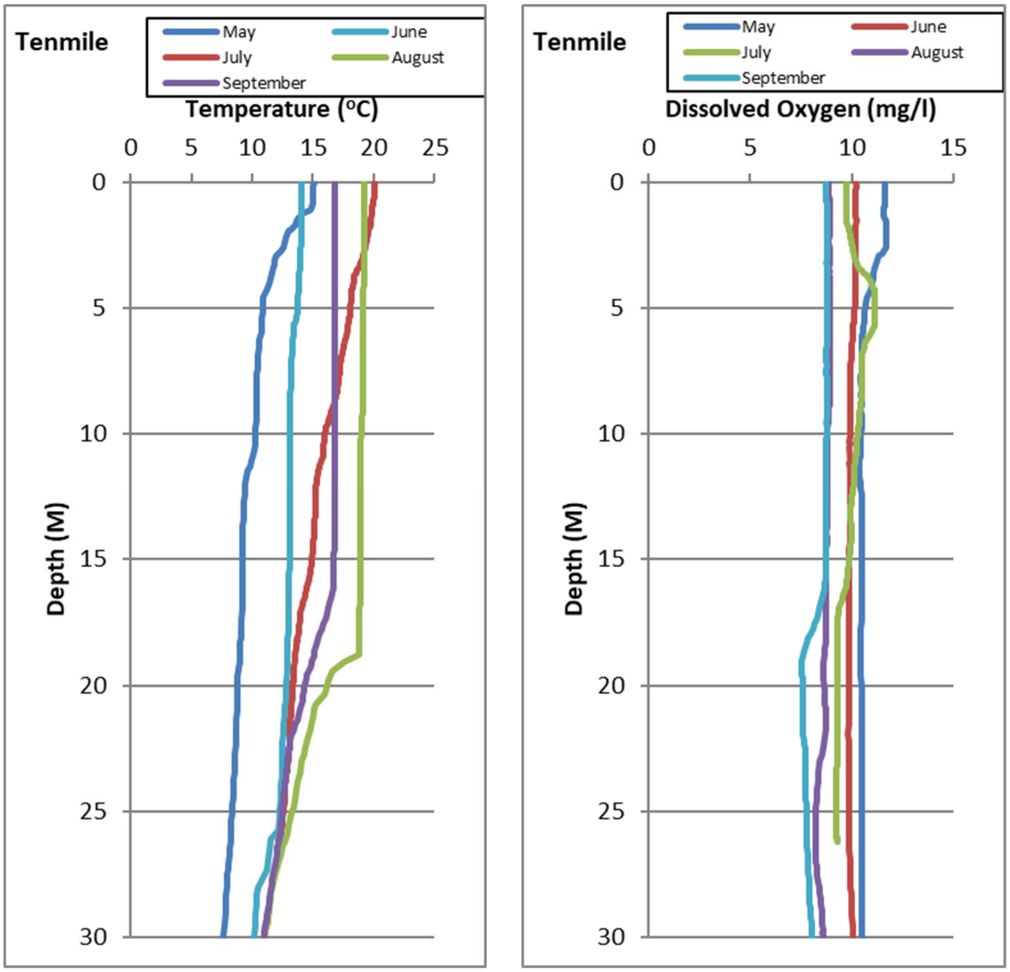


Figure 9-5. Monthly 2018 temperature and dissolved oxygen profiles for the Tenmile station.

Photosynthetic Active Radiation

The 1% photosynthetic active radiation (PAR) depths increased from lows in May of <5.0 meters to over 16 meters in August and September. In general, the Kikomun site had lower light penetration than the other locations sampled (Table 9-2). The complete set of PAR data can be found in Appendix Table A9.

Table 9-2. 1% light depth (m) by station and month.

Station	May	June	July	August	September
Kikomun		6.0	10.1	13.8	16.0
US/Canada Border	4.5	7.9	11.2	16.0	16.1
Stonehill	3.8	9.9	12.0	16.0	17.8
Tenmile	4.8	8.5	13.0	17.0	19.0

Water Chemistry

The analysis results from the water chemistry samples collected from the reservoir can be found in Appendix B. Since the primary purpose of the study was to examine primary productivity of the reservoir only a graphical analysis of the nutrients nitrogen and phosphorus was performed. The phosphorus data suggest that the water entering the reservoir in the spring was quite high (Figure**Error! Reference source not found.** 9-6). There was no May sample taken at Kikomun but the May sample taken at the US/Can border had total phosphorus concentrations considerably higher than the levels found in the lower 2 stations of the same month. After May the total phosphorus in the reservoir dropped to 5 to 10 ug/L.

Ortho-phosphorus concentrations were near the laboratory detection limit for most of the samples. The exception were samples taken in May, which corresponds to the higher total phosphorus concentrations and in the September samples taken at Kikomun and Stonehill.

Total ammonia concentrations were low in May and June (Figure 9-7). The ammonia concentrations increased slightly in July for the 3 southern stations. The August sample at Tenmile collected at 20 meters was considerably higher than the concentrations observed throughout the rest of 2018.

The nitrite+nitrate concentrations observed in Libby Reservoir are very high and increased with depth. This could be an indication that the source water from the Kootenai River, which is very high in Nitrates, could be plunging upon entering the reservoir and therefore not be readily available for algal production (Figure 9-7). The entire 2018 water chemistry data is presented in Appendix Table A10.

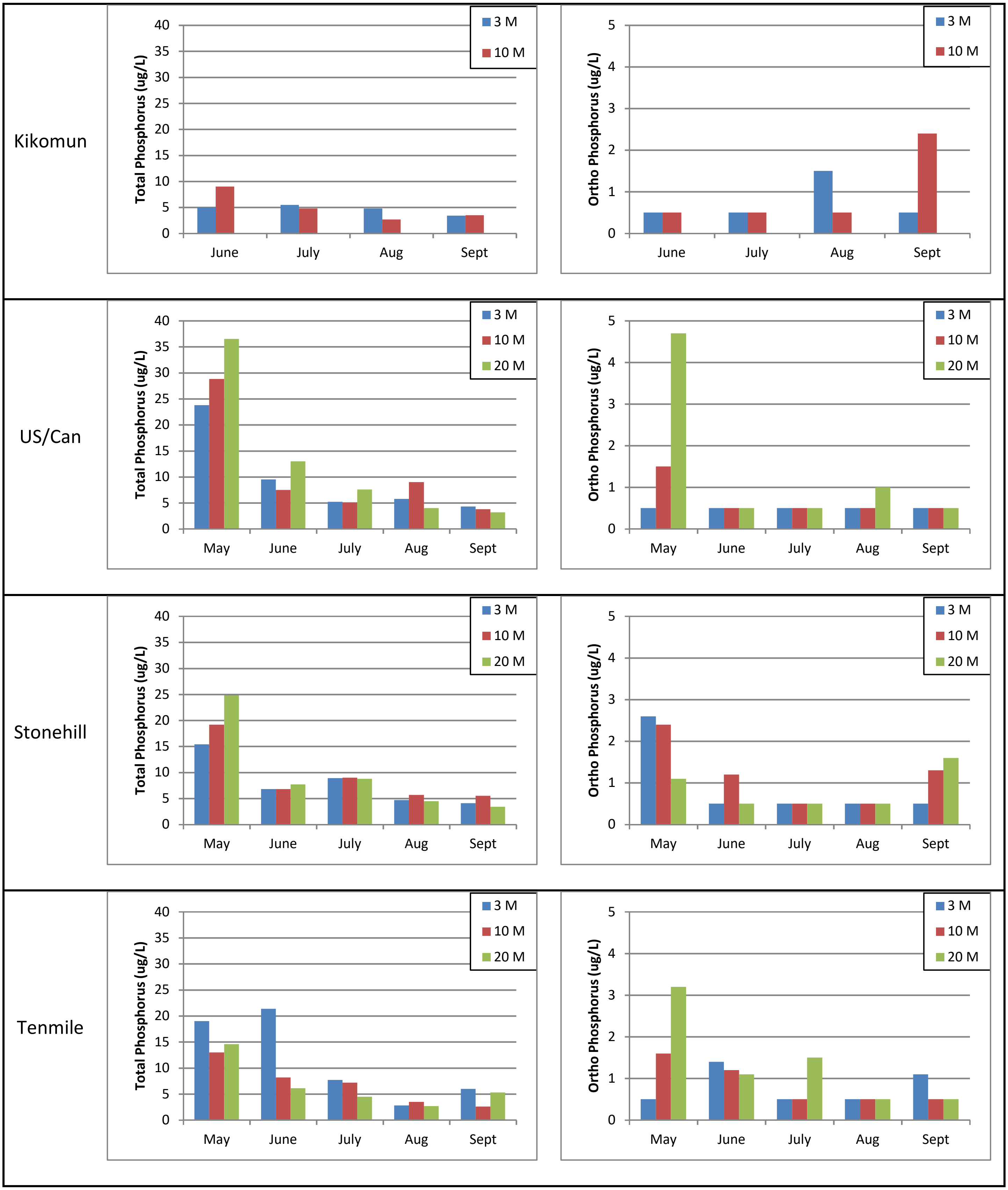


Figure 9-6. Total phosphorus and ortho-phosphorus by station and month.

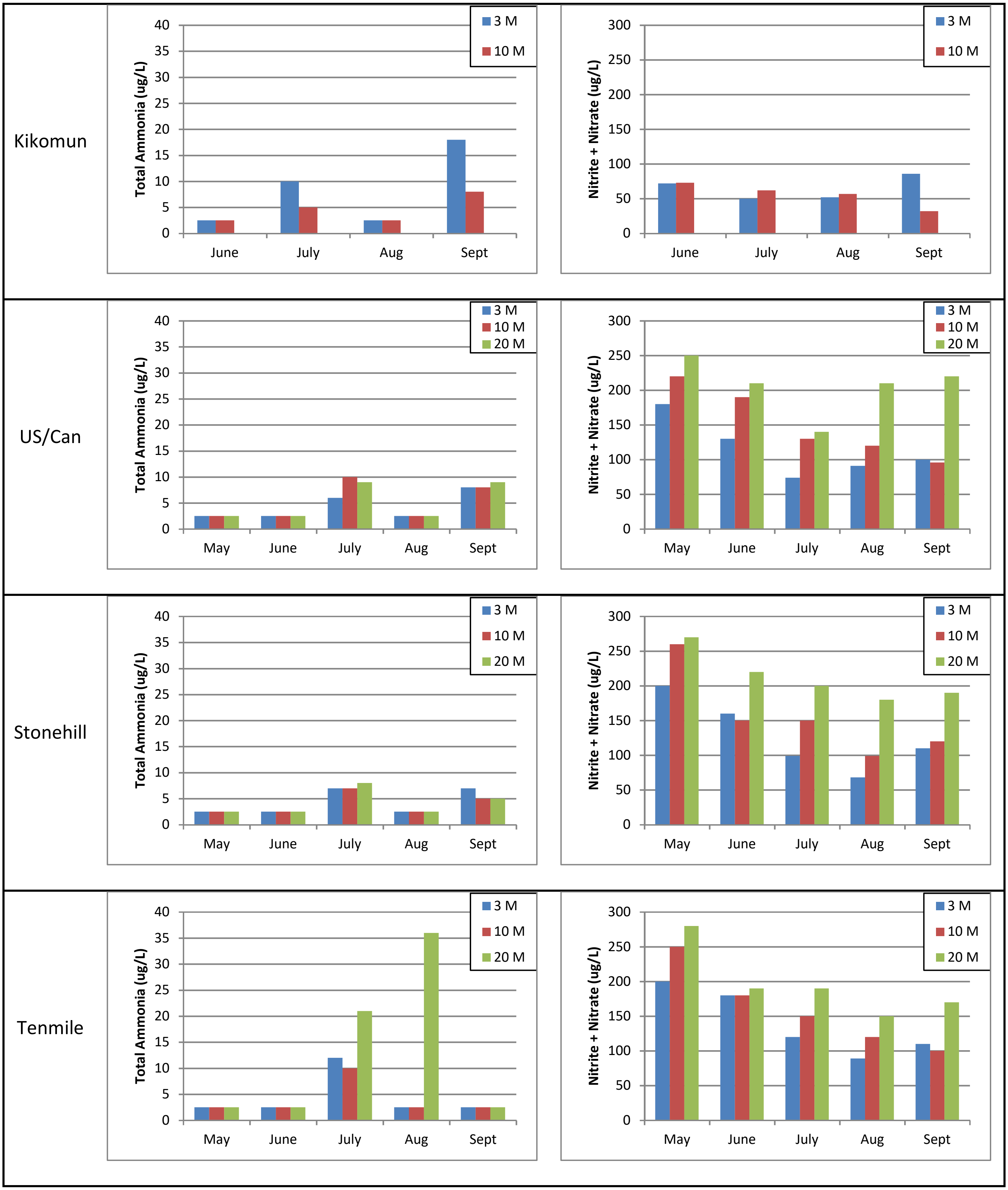


Figure 9-7. Total ammonia and nitrite + nitrate by station and month.

Primary Productivity

The vertical productivity profile is fairly typical with maximum productivity occurring in the upper 3 to 5 meters with a logarithmic decline with increasing depth (Figures 9-8 to 9-11). As was mentioned previously the Kikomun station was not in the lentic section of the reservoir in May and therefore was not surveyed. The water level had increased to result in Kikomun being in the reservoir pool by June; however, the productivity remained low until July.

The productivity at the discrete depths were multiplied by the number of meters between the mid-point of the assessment depth above and below a given assessment depth. This interpolation technique resulted in different weighting based on the distance between sampling depths. The top sample weighting was calculated by determining the distance from the surface to the mid-point of the 1m and 3m sample. This resulted in the upper sample depth representing 2 meters of the water column. The bottom depth weighting was calculated using the mid-point of the depth between the bottom depth and second to bottom depth; in most cases this meant that the sample collected at 25 meters represented 2.5 meters of the water column's productivity. The remaining assessment depths represented the productivity of 5 meters of the water column. The productivity estimates from the discrete depths and their associated weighting were summed to determine the net primary productivity on a per square meter basis. In May of 2018 there was very little total productivity within the reservoir (Table 9-3). This was most likely due to the high turbidity that was observed that resulted in low light penetration into the water column. The productivity in the three lower sites was high in June with the maximum productivity occurring at the US/Can border site. The productivity in Kikomun remained extremely low in June but increased in July and by August it had the greatest productivity of the four stations. The reservoir productivity declined in July followed by increases in August and September. The highest mean reservoir productivity occurred in September of 2018.

In addition to determining gross productivity, productivity by size fraction was performed. Productivity occurring between 0.2um and 2.0um is typically considered too small for zooplankton consumption and the fraction larger than 20um is too large for zooplankton consumption. The ideal size for zooplankton consumption is between 2.0um and 20um (Sieburth et. al. 1978). The primary production of the Libby Reservoir by date, station and size fraction is presented in Table 9-3**Error! Reference source not found.**. The phytoplankton community was relatively evenly distributed between the three size classes. Thirty percent of the observed productivity occurred in the size fraction that is most desirable for zooplankton consumption (2.0-20 um) with 43% of the production occurring in phytoplankton community >20.um and 26.5% occurring in the smallest size fraction. The entire PPR dataset for 2018 is presented in Appendix Table A12

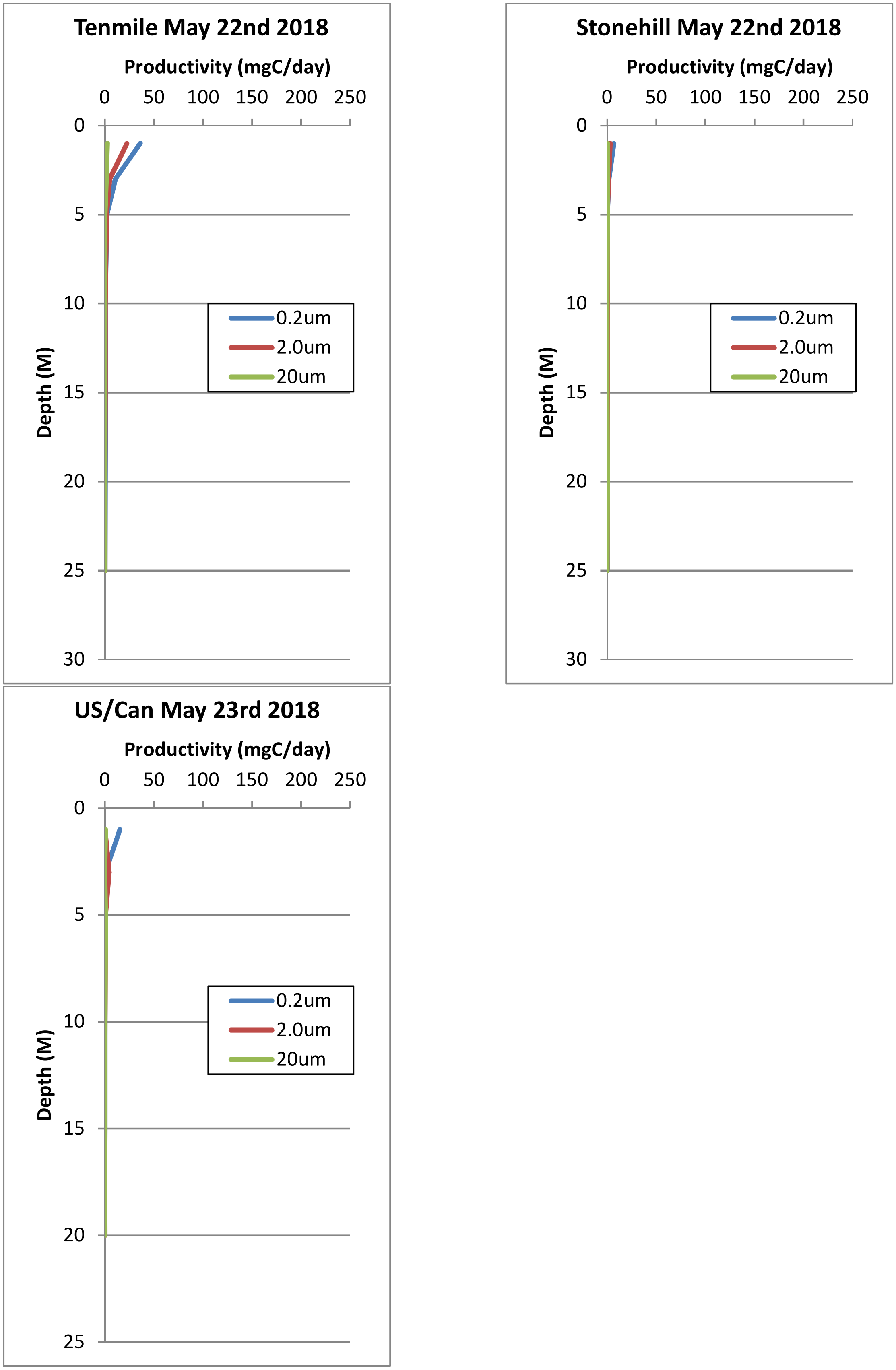


Figure 9-8. May 2018 Productivity by depth and size fraction.

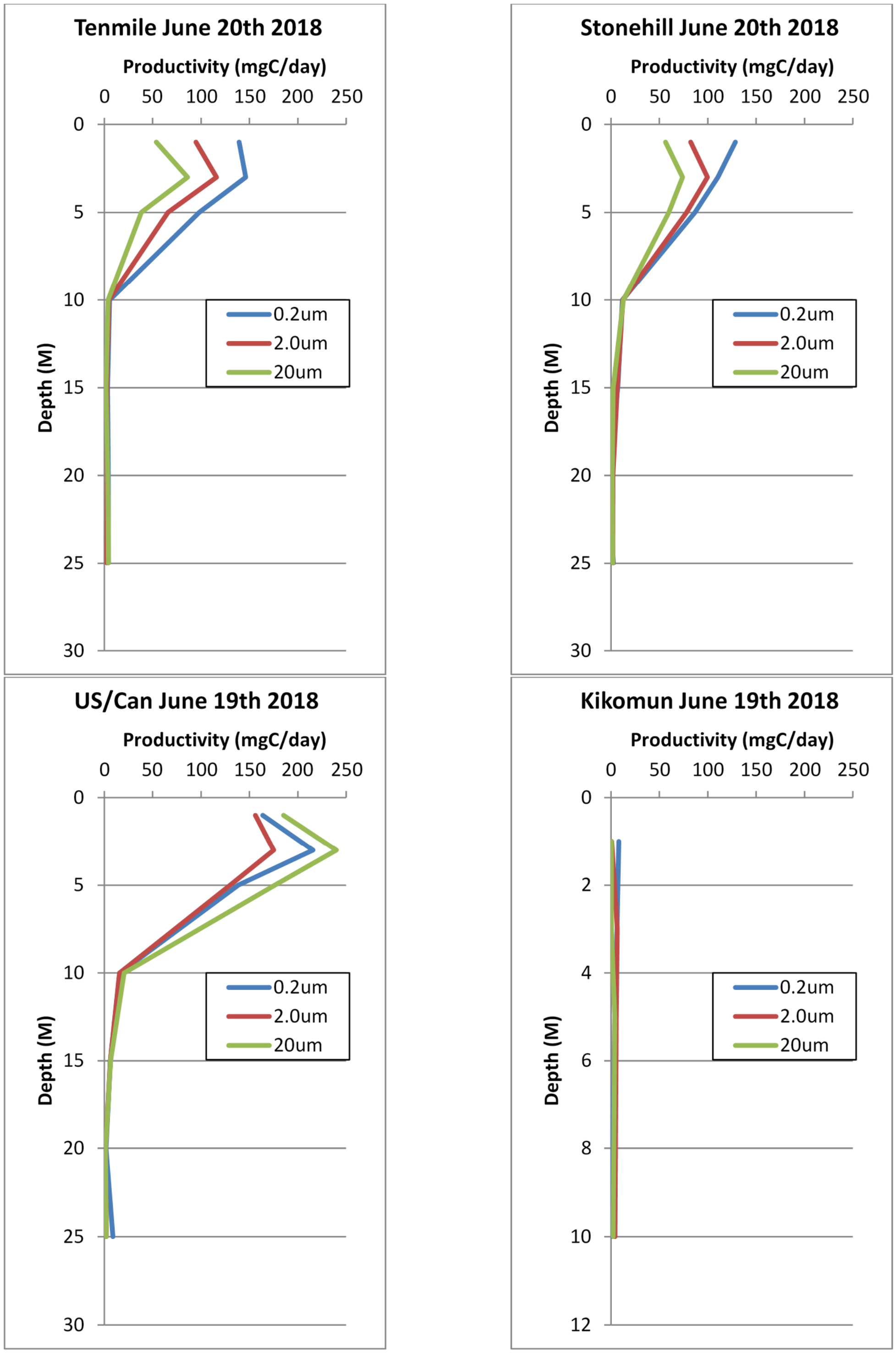


Figure 9-9. June 2018 Productivity by depth and size fraction.

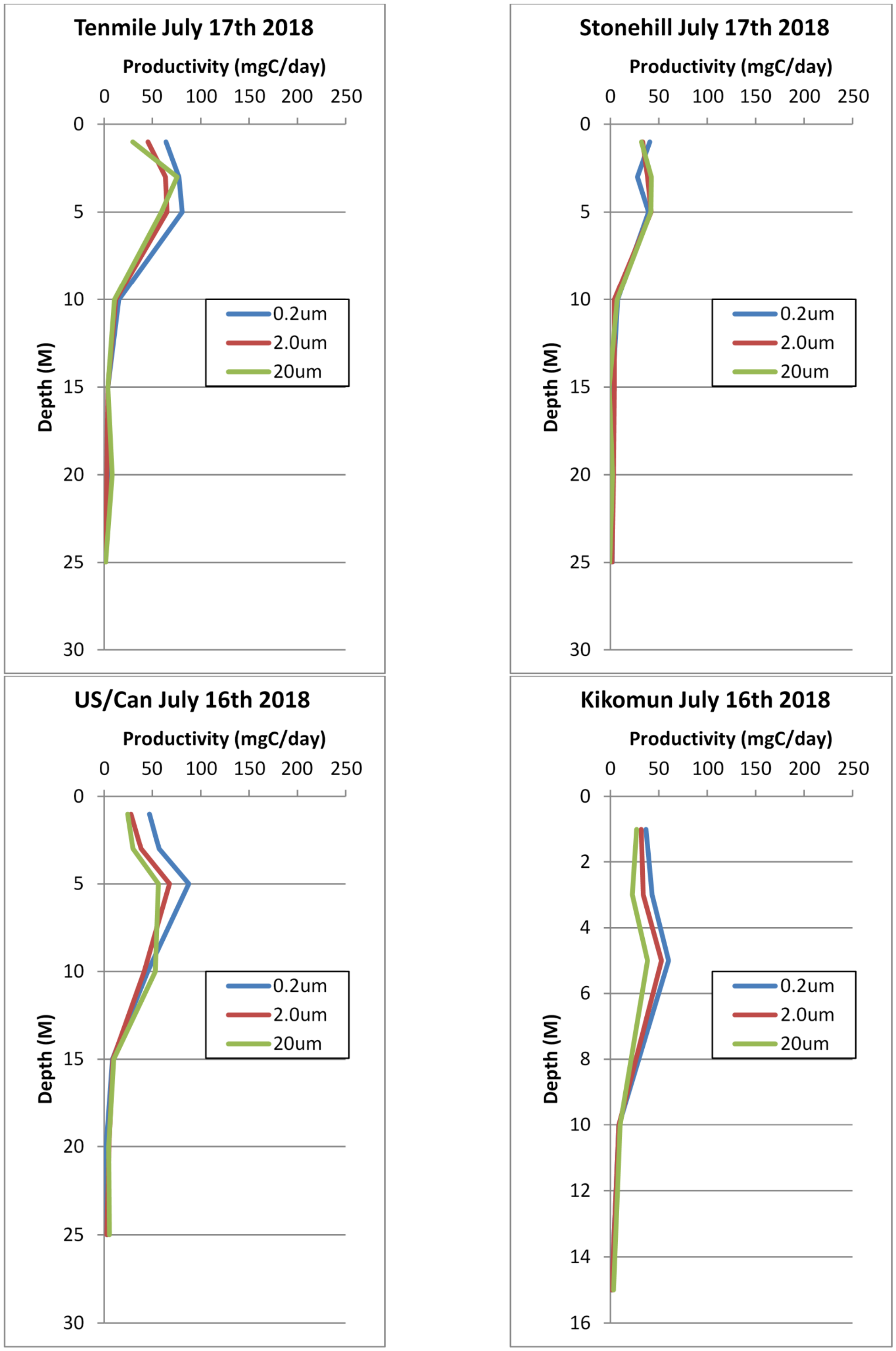


Figure 9-10. July 2018 Productivity by depth and size fraction.

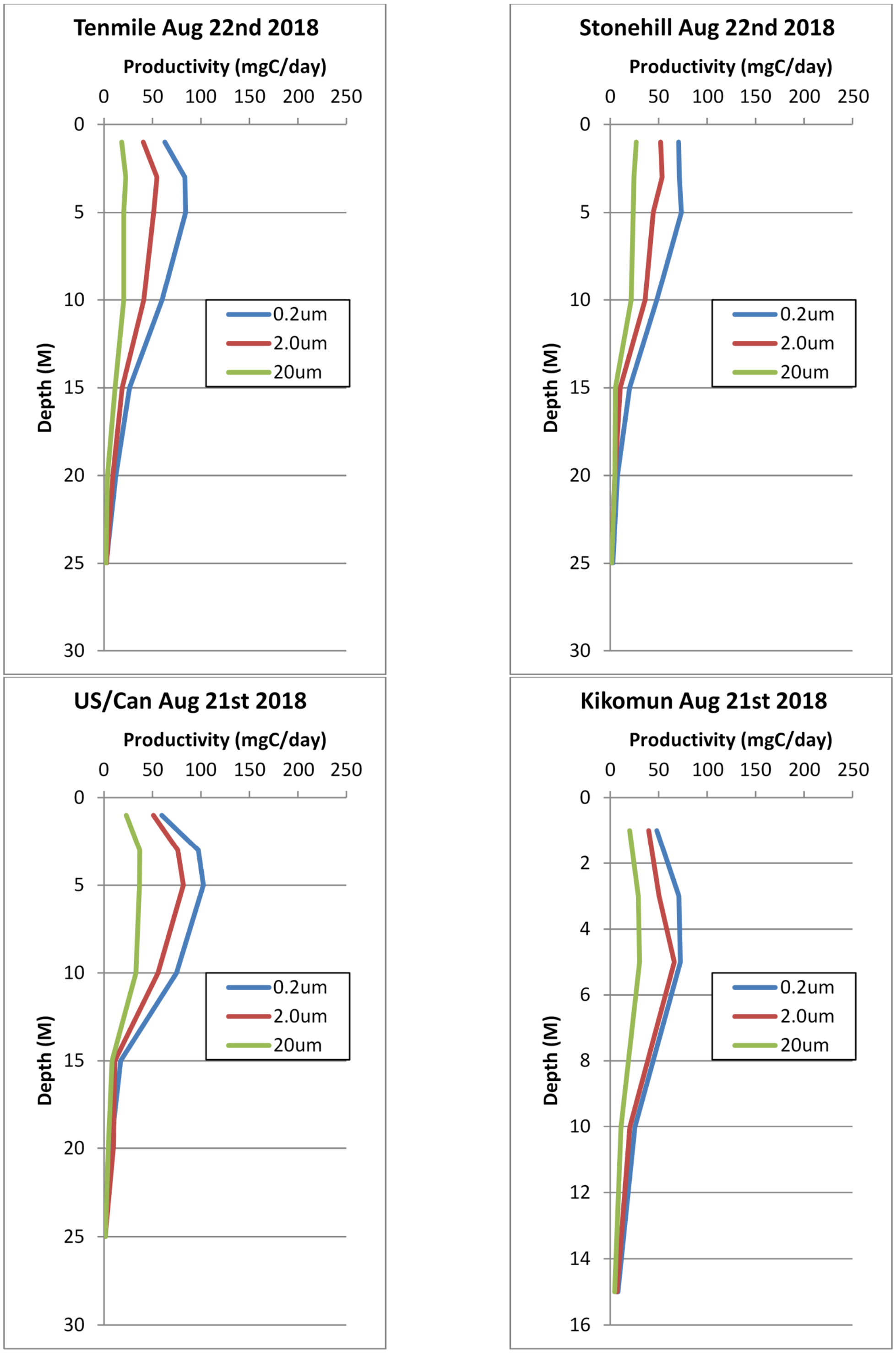


Figure 9-11. August 2018 Productivity by depth and size fraction.

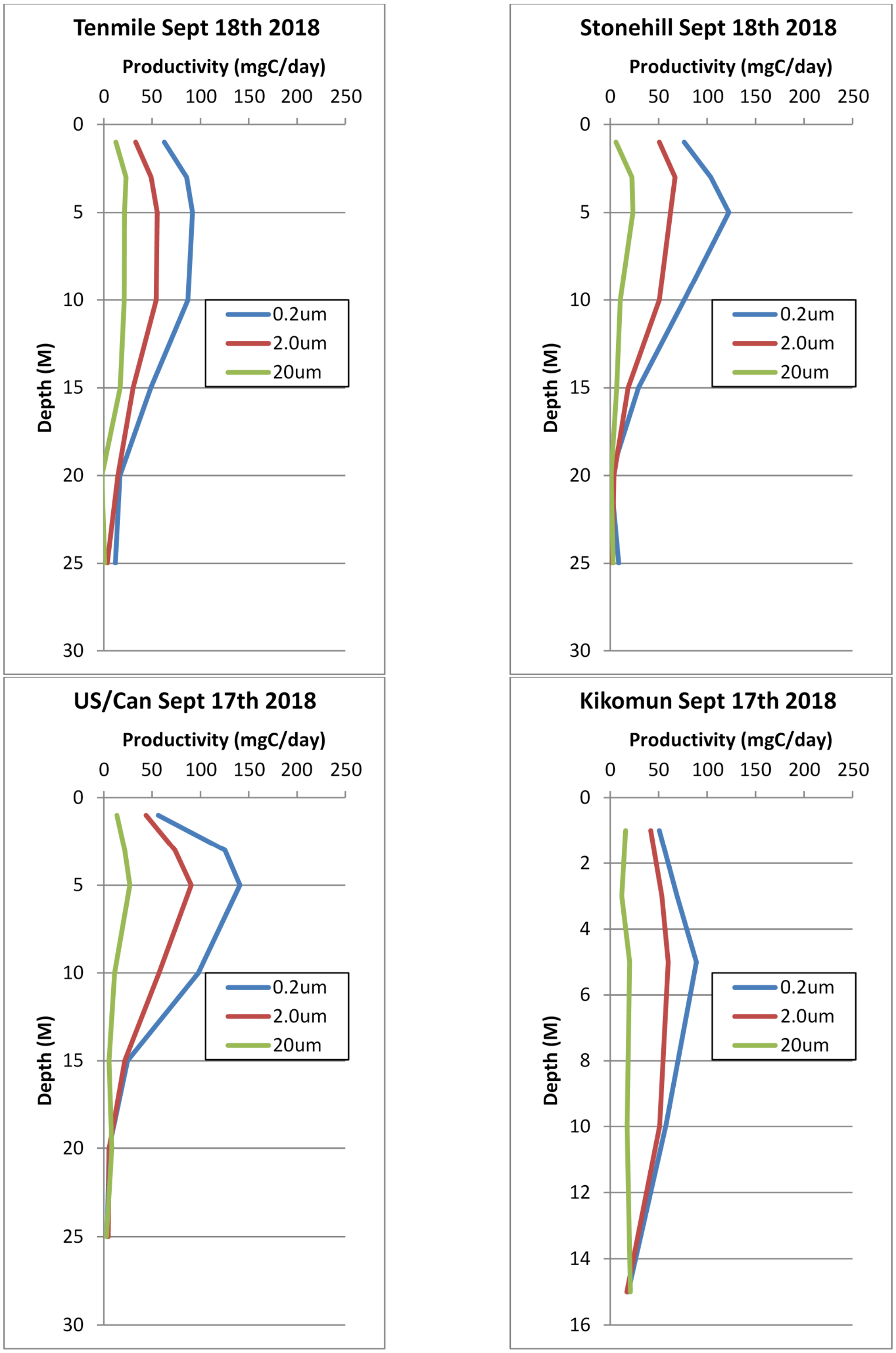


Figure 9-12. September 2018 Productivity by size fraction and depth.

Table 9-3. Productivity values for Libby Reservoir by size fraction for 2018.					
Month	Station	Productivity (mgC/m2/Day)			
		0.2 um – 2.0um	2.0 um – 20um	>20um	Total
May 2018	Tenmile	35.92	60.30	10.96	107.19
	Stone Hill	38.00	9.21	1.40	48.61
	US/Can Border	19.16	9.03	4.71	32.90
	Kikomun	NA	NA	NA	NA
June 2018	Tenmile	279.58	240.14	468.44	988.16
	Stone Hill	119.76	190.81	554.37	864.95
	US/Can Border	148.14	0.00	1612.29	1760.43
	Kikomun	0.00	28.48	1.03	29.51
July 2018	Tenmile	139.64	7.40	533.71	680.75
	Stone Hill	0.00	19.47	333.01	352.48
	US/Can Border	147.08	0.00	655.46	802.54
	Kikomun	101.48	132.99	602.71	837.17
August 2018	Tenmile	365.51	386.40	332.29	1084.20
	Stone Hill	299.36	277.66	347.82	924.85
	US/Can Border	242.20	446.34	477.75	1166.29
	Kikomun	119.79	249.85	381.04	750.67
September 2018	Tenmile	547.35	533.24	329.30	1409.90
	Stone Hill	516.83	586.44	236.54	1339.81
	US/Can Border	517.85	691.30	295.38	1504.53
	Kikomun	171.67	415.77	351.30	938.74

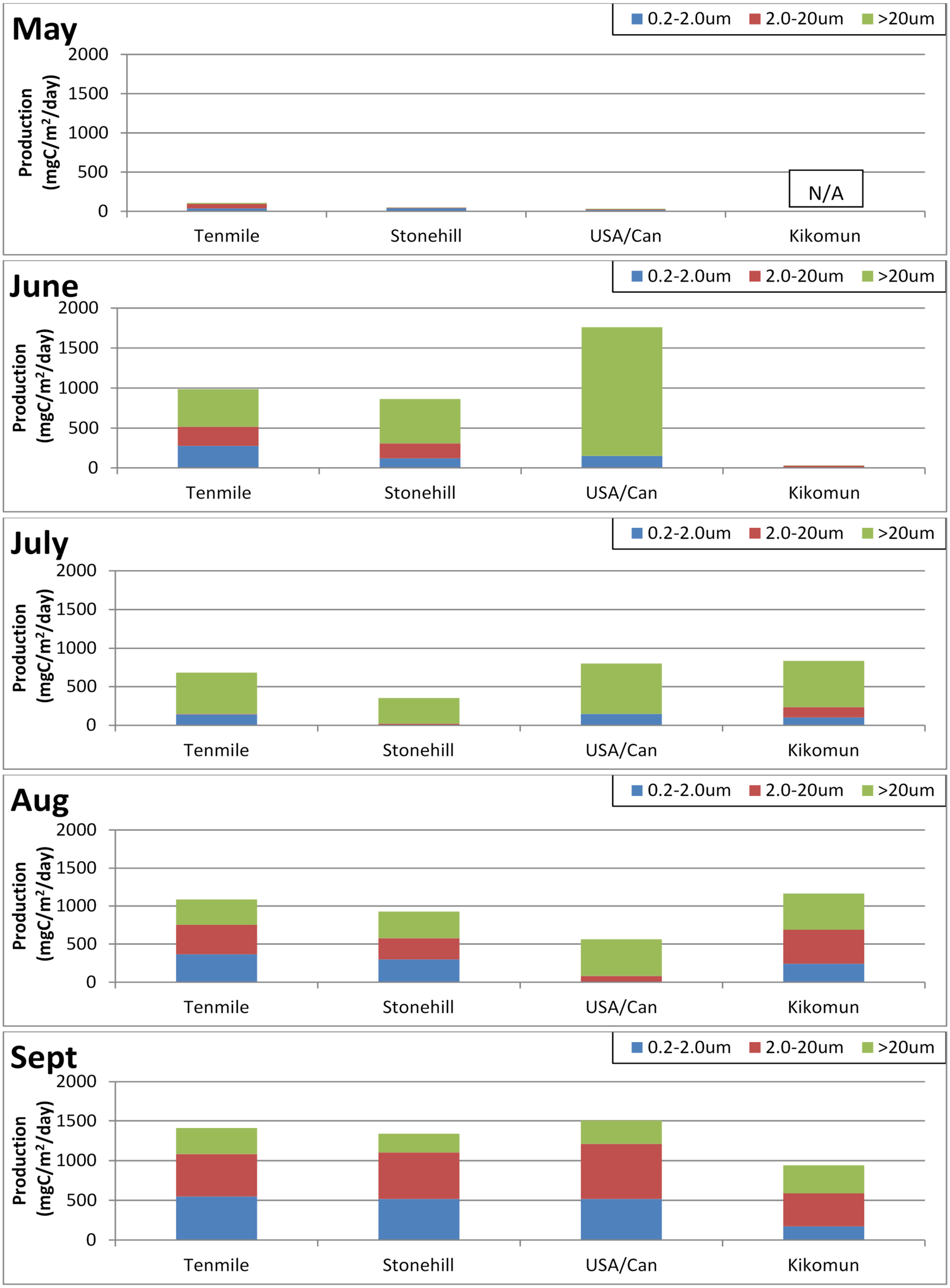


Figure 9-13. Productivity by month, and size class for 2018.

Size Fractionated Chlorophyll a

Chlorophyll *a* is surrogate for the standing crop of the phytoplankton community within a waterbody. It is different from primary production measurements in that it does not provide an estimate of the growth rates of the phytoplankton community. When comparing the chlorophyll *a* concentrations and the primary productivity rates in this study it should be noted that the chlorophyll *a* concentrations were derived from water samples taken in the top 10 meters of the water column and the primary production measurements were integrated over the top 25 meters of the reservoir.

In May and September, the fraction of the chlorophyll *a* in the largest size fraction made up a small percentage of the total chlorophyll *a* observed in the reservoir, whereas in June and July it was the major size fraction present (Figure 9-14). The average percentage of chlorophyll *a* by size class in 2018 was 27% in the 0.2 to 2.0 um size fraction, 41% in the edible 2.0 to 20 um size class and 32% in the inedible size class of greater than 20 um. This differs from the primary productivity measurements where the largest percentage of primary productivity occurred in the >20 um size class. Both chlorophyll *a* concentrations and primary productivity measurements indicated that the smallest size class had the lowest productivity and lowest standing crop in 2018. The entire Chlorophyll a data set is presented in Appendix Table A10.

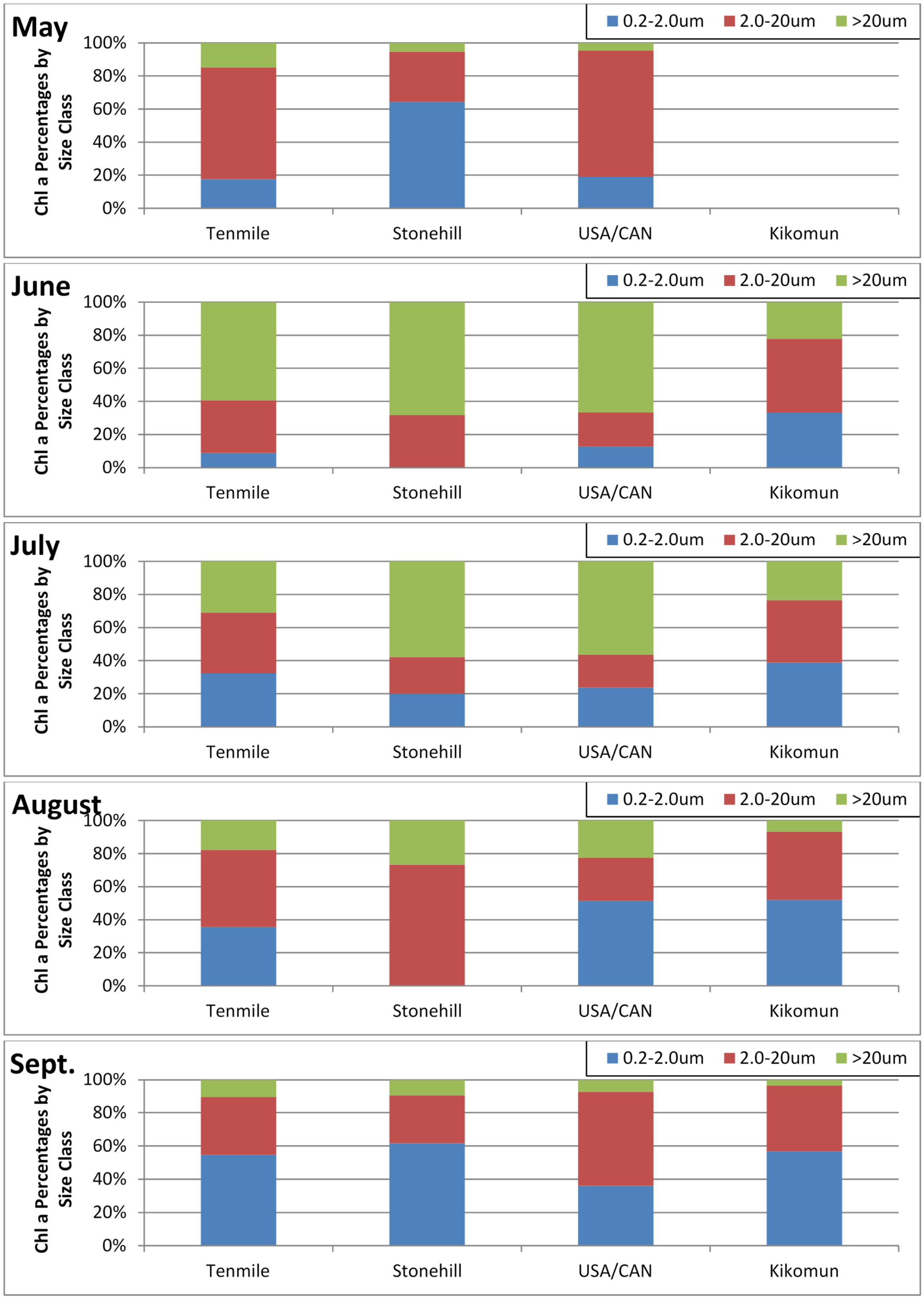


Figure 9-13. Percentages of chlorophyll a by month and station 2018.

Intra-study Comparison

The 1972-1980 study did not have a station at Kikomun. Since Kikomun has the lowest productivity of the stations in the 1986, and the 2016-2018 studies, the Kikomun productivity values were excluded from inter-study comparisons. The productivity at all stations for most months were considerably higher in 2016 through 2018 compared to the other years for which productivity data exists (Figure 14). The Chisholm et.al. (1989) summary report does not go into detail on the calculation methodologies; therefore some of the difference may be due to differences in methods. The mean primary productivity observed in all years where a primary production assessment was conducted fall within the range of production that is defined as mesotrophic (250-1000 mgC/m^2/day)(Wetzel, 1975). However, there were two months (June and September) in 2018 where the productivity was sufficient for the system to be classified as eutrophic. The productivity observed in 1972-80 was considerably lower than the other three years measured. Early predictions of the trophic status of Libby Reservoir indicated that it may become eutrophic based on the Vollenweider model (Vollenweider, 1975). However, Woods (1979, 1981, and 1982) proposed that physical processes within the reservoir such as a weak thermocline and narrow photic zone, would inhibit production within the system. The data from 2016-2018 suggests that the productivity has increased since the 1970's and 1980's. Additional assessment of the system should be undertaken on a regular basis to track changes in productivity of the system.

The spring of 2018 was cooler than the previous two years of study which resulted in a delay in the runoff from the Canadian Rockies. This resulted in lower productivity in May of 2018 than either 2016 or 2017. The biggest difference in productivity, in terms of monthly productivity, between 2016-2017 and 2018 occurred in September. The 2018 productivity was the highest measured in the three years of the study. There was a surprising amount of inter-annual variability within the system between 2016-2018. This suggests that Libby Reservoir is highly susceptible to changes in timing, and quantity of water entering the system along with the concomitant changes in nutrient loading.

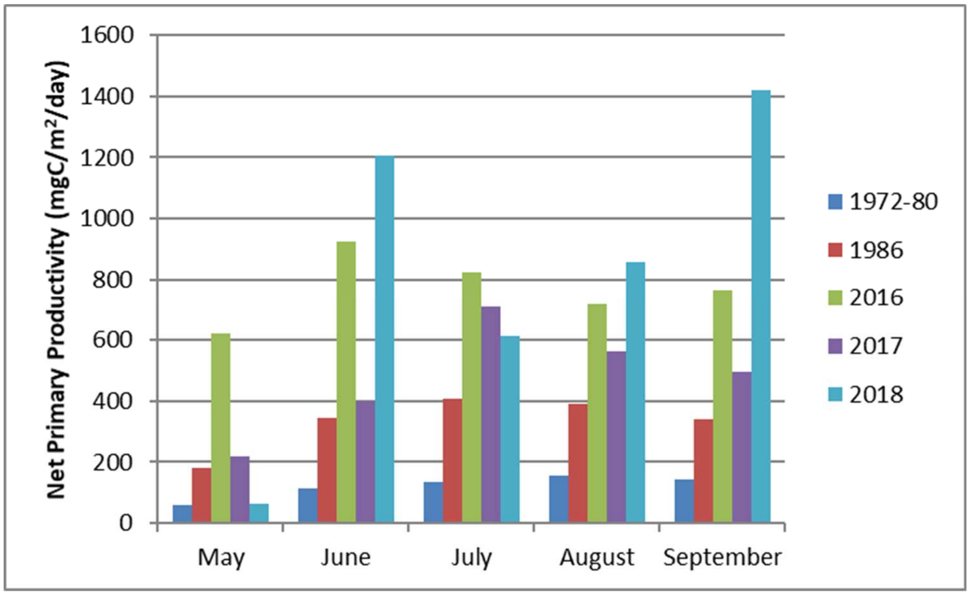


Figure 9-14. Comparison of primary production between 1972-1980, 1986, and 2016 - 2018 in Libby Reservoir.

Conclusions

An extensive body of data exists from 1972 to 1980 detailing nutrient loading, primary productivity, and limnological conditions, with a well-documented and thought out analysis of that data (Woods, 1982). The two years of primary production data collected and analyzed in this study encompassed two very different water years which resulted in different reservoir management of pool elevation and water release. One of the goals of this study was to assist managers in making recommendations to the USACE regarding reservoir operations that would increase fish production. To better accomplish this goal, it may be necessary to conduct a third year of primary production estimates to refine our understanding of reservoir operations on primary productions.

Additionally, to further assess if production has changed within the reservoir, a companion analysis looking at similar sources of data as well as trends since the formation of the reservoir should be examined to determine the conditions that are driving primary production, zooplankton production, fisheries, and nutrient retention and release to downstream waterbodies should occur. This includes the collection and analysis of water samples for phytoplankton and zooplankton community analysis.

Chapter 10: Westslope Cutthroat and Redband Trout Conservation Assessment in the Ten Lakes Scenic Area and Cabinet Mountains

This chapter includes the following work elements:

H: Westslope cutthroat and redband trout conservation assessments (Contracts 77012 and 76916)

J: Analyze and interpret Libby Mitigation physical and biological data (Contracts 77012 and 76916)

Introduction

Ten Lakes Scenic Area

Westslope cutthroat trout (WCT) were likely the most widely distributed subspecies of cutthroat trout (Behnke 1992; 1996). They have a wide general distribution including streams and lakes in the upper Columbia River basin of western Montana, northern and central Idaho, southern British Columbia and Alberta; the upper Missouri River basin of Montana and northwest Wyoming; the upper South Saskatchewan River in Alberta; the Methow River, Pend O'reille River, and Lake Chelan drainages in Washington; and the John Day River drainage in Oregon (Shepard et al. 2003). Most the Montana portion of the Kootenai River is included in this wide area of distribution including the Montana and British Columbia portions of the Kootenai River upstream of Libby Dam.

Range wide, WCT currently occupy an estimated 59% of the species historical habitat. However, WCT with no evidence of genetic introgression currently occupy about 10% of their historically occupied habitats (Shepard et al. 2003). These general trends typify the current WCT distribution within the Montana portion of the Kootenai River, both upstream and downstream of Libby Dam (MFWP unpublished data). Genetic introgression with other species of trout has been a long-recognized cause of the decline in the distribution and abundance of WCT throughout this species historic range (Shepard et al. 2003).

The Ten Lakes Scenic Area is in northwest Montana, and lies within the Wigwam River watershed (Figure 10-1). This mountainous region contains several lakes of which the historic fish distribution was suspected to be either westslope cutthroat trout or fishless. However, prior to the development of an MFWP hatchery stock of WCT in 1970, many of these lakes were stocked with Yellowstone cutthroat trout (YCT). Several of these waterbodies have been subsequently stocked with hatchery WCT (Table 10-1). The lakes within the Ten Lakes Scenic Area have varying degrees of surface water connection with the Wigwam River and its tributaries. Previous genetic sampling within these lakes investigated whether non-native genetic components remain in any of these lakes or if these genes have influenced the genetic integrity of WCT populations located downstream of these lakes (Table 10-1).

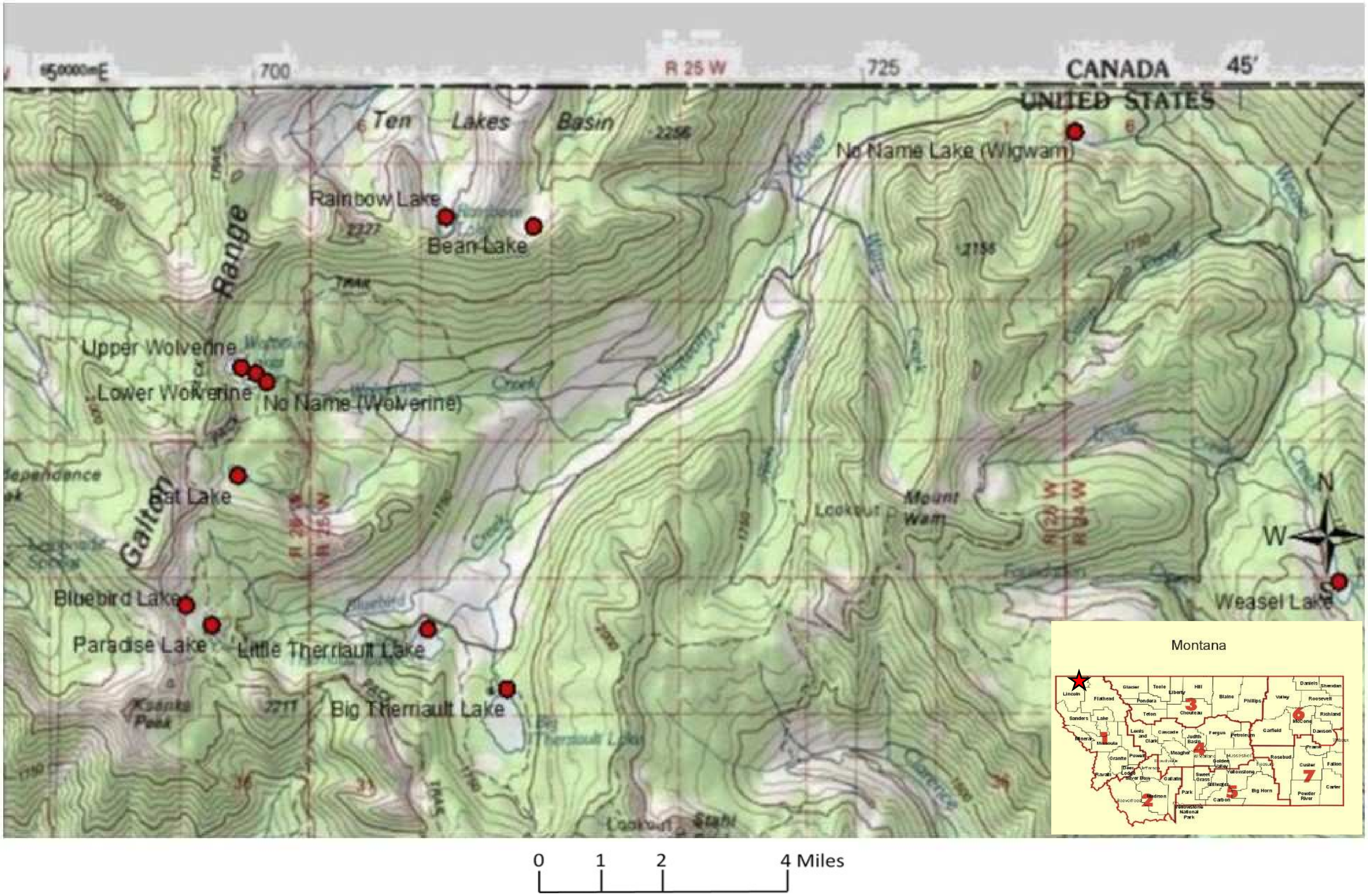


Figure 10-1. Topographic map with locations of the twelve lakes (red dots) in the Ten Lakes Scenic Area that were investigated by FWP.

Table 10-1. Summary of the genetic assessments of fish previously collected from water bodies within the Ten Lakes area. The proportion of westslope cutthroat (WCT), rainbow (RBT), and Yellowstone cutthroat trout is listed. An ND indicates that alleles from that particular species were not detected in the sample.

Water Body	**Currently Stocked Periodically**	**% WCT**	**% RBT**	**% YCT**	**Outlet Stream Name**	**Citation**
Rainbow Lake	No	0.099	ND	0.901	Unnamed trib. to Rabbit Creek	Leary et al. 2015
Big Therriault Lake[1]	Yes	See note	See note	See note	Bluebird Creek	Whitely et al. 2018
Little Therriault Lake[2]	Yes	0.997	0.003	ND	Bluebird Creek	Whitely et al. 2018
Lower Wolverine Lake	Yes	0.998	0.002	ND	Wolverine Creek	Whitely et al. 2018
Upper Wolverine Lake	Yes	1.000	ND	ND	Wolverine Creek	Whitely et al. 2018
No Name Wolverine Lake	No	0.978	0.005	0.017	Wolverine Creek	Whitely et al. 2018
Weasel Lake	No	0.936	ND	0.064	Weasel Creek	Whitely et al. 2018
Bluebird Lake	Yes	0.997	0.003	ND	Bluebird Creek	Whitely et al. 2018
Paradise Lake	No	1.000	ND	ND	Bluebird Creek	Whitely et al. 2018
No Name Wigwam Lake	No	1.000	ND	ND	Unnamed trib. to Wigwam R.	Whitely et al. 2018
Bat Lake	No	fishless	fishless	fishless	Wolverine Creek	Whitely et al. 2018
Bean Lake	Yes	fishless	fishless	fishless	Unnamed trib. to Wigwam R.	Whitely et al. 2018

Note[1]: The YCT alleles detected in the samples were not randomly distributed among all fish, indicating that the sample did not come from a randomly mating population. In addition, many individuals in the sample appeared to be non-introgressed WCT, possibly indicating the fish were hatchery origin.

Note[2]: RBT alleles were not randomly distributed among the samples. One individual appeared to be highly admixed, but after removing this individual from the sample, the RBT alleles were randomly distributed.

Cabinet Mountains

Rainbow Trout *Oncorhynchus mykiss* (RBT) are probably the most popular sport fish in the world which is likely due in part to their wide distribution (both natural and introduced). Native rainbow trout have a wide distribution in western North America (Behnke 1992). Native Rainbow Trout occurring west of the Cascade Range and the Sierra Nevada are currently classified as coastal Rainbow Trout *O. mykiss irideus*, whereas rainbow trout occurring east of these mountain ranges are classified as redband trout (Muhlfeld et al. 2015). Behnke (1992) identified three subspecies of redband trout: (1) Columbia River redband trout *O. mykiss gairdneri*, which occur in the Columbia and Fraser rivers; (2) northern Great Basin and Upper Klamath Lake redband trout *O. mykiss newberrii*; and (3) Sacramento redband trout *O. mykiss stonei*, which occur in the Pit and McCloud rivers. Currently, the distribution of redband trout is estimated to include 42% of its historic range (Muhlfeld et al. 2015). MacCrimmon (1971) and Behnke (1992) proposed that redband trout *O. mykiss gairdneri* were native to the Kootenai River drainage of Montana. Subsequent genetic testing from 1980 to 1995 (Allendorf 1980; Sage et al. 1992; Phelps and Allendorf 1980; and Huston 1995). The range of redband trout in the Kootenai River drainage includes many of the tributaries that begin in the Cabinet Mountains.

The Cabinet Mountains are part of the Rocky Mountains, located in northwest Montana and the Idaho panhandle. The mountains cover an area of 2,134 square miles, of which 147.3 square miles (94,272 acres) were designated as wilderness by the National Wilderness Preservation Act of 1964. The Cabinet Mountains lie south of the Purcell Mountains, between the Kootenai River and Clark Fork River and Idaho's Lake Pend Oreille. Several mountain lakes exist within the wilderness area. It is unknown what species of trout, if any historically inhabited many of these lakes. Huston et al. (1996) completed genetic surveys of lakes within the Cabinet Mountains Wilderness area to explore potential inland rainbow trout restoration opportunities. Huston et al. (1996) determined that several lakes within the Cabinet Mountain Wilderness area were fishless. These lakes included Upper Bramlett, Ozette (West Fisher drainage), Libby lakes, Ramsey (Libby drainage), Big Cherry, Takoka, Martin, Snowshoe lakes (Big Cherry drainage), Vimy, Klatawa (Granite drainage), Osakis, Upper Sky (Flower drainage), and Parmenter (Parmenter drainage). Four of the Cabinet Mountain lakes contain brook trout, which Huston et al. (1996) presumed had likely replaced other trout species present or were the only species ever stocked. These lakes included Leigh Lake in the Big Cherry Creek drainage, and Wishbone and Double Lakes in the Granite Creek drainage. Huston et al. (1996) performed genetic analysis of *Onchorhynchus* collected from eleven lakes in the Cabinet Mountains and found hybridization a common phenomenon (Table 10-2), but did not find any non-introgressed populations of redband trout in any of these waters. However, existing genetic technologies only allow assessment of Kootenai Basin redband introgression at the population level, and not the individual fish. To overcome this shortcoming, we collaborated with the University of Montana (UM) genetics laboratory to develop genomic resources to assess non-native admixture and population structure of Kootenai drainage redband rainbow trout.

Table 10-2. Lakes in the Cabinet Mountain Wilderness area containing *Oncorhynchus* (adopted from Huston et al. (1996).

Drainage	Lake	Genetic Analyses Results[1]
Silver Butte Fisher	Baree	WCT (88) RBT (7) YCT (5)
	Little Bear	WCT (100)
	Big Bear	WCT (100)
West Fork Fisher	Lower Geiger	IRB (NR) CRT (NR)
	Upper Geiger	IRB (NR) CRT (NR)
	Bramlett	IRB (86) YCT (14)
Granite Creek	Granite	WCT (99.2) RBT (0.8)
Granite Creek	Lower Sky	WCT (98) YCT (2)
	Lower Hanging Valley	IRB (NR) CRT (NR)
	Upper Hanging Valley	IRB (NR) CRT (NR)
Cedar Creek	Upper Cedar	CRT (100)
	Lower Cedar	CRT (100)

[1]Species abbreviations are: WCT – westslope cutthroat trout, YCT – Yellowstone cutthroat trout, RBT – non-designated rainbow trout, IRB – Interior redband trout, and CRT – coastal rainbow trout. The number following the species abbreviation is the estimated percent genetic component of the sample for the given species. NR indicates that the percent contribution was not reported.

Huston et al. (1996) concluded that genetic analysis results (Table 10-2) did little to determine the original range of redband trout in Cabinet Mountain waters, and poor stocking records and practices contributed to the issue. They also recommended that additional genetic sampling of several of the outlet tributaries to many of these lakes would be useful information to this end. Many of the streams that Huston et al. (1996) recommended have since been sampled to determine the genetic constituency of those populations. However, genetic analyses have yet to be conducted on fish from nine tributaries including: Baree, Iron Meadow, Porcupine, Bramlett, Greiger, No, Cable, Flower, and Parmenter creeks. The work identified in this report is intended to identify the original range of redband trout in Cabinet Mountain Lakes and associated tributaries.

Methods

We reviewed the results of the genetic analyzes completed within the Ten Lakes Scenic Area (Table 10-1) and identified the need to collect additional samples in the three largest fish baring tributaries downstream of the lakes. We collected genetic samples from fish at three locations on Weasel Creek, two locations on Wolverine Creek and two locations on Bluebird Creek.

For the Cabinet Mountains area assessment, we used the recommendation provided by Huston et al. (1996) to target genetic sampling in nine tributaries to determine if redband trout were historically present in these waters. For all genetic sampling, we identified a total target sample size of 30 fish from each creek for genetic analyses. All samples were collected using backpack electrofishing. We attempted to collect fifteen fish from two locations in the creeks separated by at least a quarter of a mile. All fish collected for genetic analyses were measured and a small (< 0.5 cm^2) portion of a fin was removed and stored in an individually labeled 1.5 ml polypropylene tube that was pre-filled with 95% non-denatured ethanol. Tissue samples from each water body were stored separately in a labeled zip-lock bag and placed in a freezer (-20 F) until samples were delivered to the University of Montana Conservation Genetics Laboratory for genetic analyses. When angling was used to collect the fish, the number of anglers, the total time each angler spent angling, and the individual catch of each angler was recorded to serve as an index of catch rates. When gill nets were used to capture fish, the number of nets set, the time each net fished and the total catch of each net was recorded.

Development of genetic resources for Kootenai Basin redband trout

Redband trout are naturally sympatric with westslope cutthroat trout in the Kootenai drainage, and there appears to be only rare introgression between westslope and redband rainbow trout in this region (Leary et al.). However, introduced coastal rainbow trout readily hybridize with both redband and westslope cutthroat trout which threatens the genomic integrity and potential local adaptations in redband rainbow trout. Hybridization between non-native coastal rainbow trout and westslope cutthroat may also have broken down an historically intact barrier to gene flow between westslope cutthroat trout and redband. The objective of this work is to develop and apply a DNA sequencing assay that includes a large number of ancestry-informative single nucleotide polymorphisms (SNPs) to assess admixture among westslope cutthroat, coastal rainbow, and redband rainbow trout across the Kootenai drainage.

The UM genetics laboratory used a new, reduced-representation sequencing approach (Rapture) (*2*) that includes 10,000 SNPs for this project. Rapture is a combination of restriction site-associated DNA sequencing (RADSeq) (Andrew et al. 2016), and in-solution sequence capture (hence 'Rapture') technologies.

RADSeq is an approach to sequence a small subset of a genome, which allows researchers to obtain enough sequence data to address many research questions while avoiding the huge expense of sequencing entire genomes. The genome is subsampled by sequencing only short stretches of DNA that are adjacent to restriction enzyme cut sites. However,r an inefficiency of RADSeq is that most of the stretches of DNA that are adjacent to these restriction cut sites do not contain any SNPs, and thus represent wasted sequencing expense.

Rapture greatly increases the efficient of RADSeq by selectively sequencing only the most useful RAD sites in the genome. Rapture increases the efficiency of RADSeq by 'capturing' and sequencing only those RAD sites that are deemed the most useful. This is done by hybridizing a set of custom-designed sequence probes to a small subset of RAD sites in the genome; the DNA from the rest of the RAD sites is discarded. The remaining useful RAD sites are sequenced and analyzed at greatly reduced expense compared to traditional RAD sequencing.

The UM genetics laboratory developed a Rapture assay to assess admixture and population structure in Kootenai redband rainbow trout. The laboratory conducted RADSeq on 134 redband rainbow trout from across Kootenai drainage (Wolf Creek, East Fork Yaak, Allahan Creek, Bear Creek) and British Columbia, in addition to 90 westslope cutthroat trout sampled from across their range, and 54 hatchery-origin coastal rainbow trout (278 individuals in total). Analysis of these data resulted in >354,000 identified SNPs after following standard bioinformatics and quality control protocols, which was then reduced to the 18,000 most informative SNPs, with a focus on including the SNPs that were most highly differentiated between taxa (i.e., high F_{ST}, Figure 10-2). The UM genetics laboratory contracted a private lab to conduct quality control test these probes against the rainbow trout reference genome, and to manufacture the sequence capture baits. The effort identified the 10,000 most reliable and useful of the capture probes.

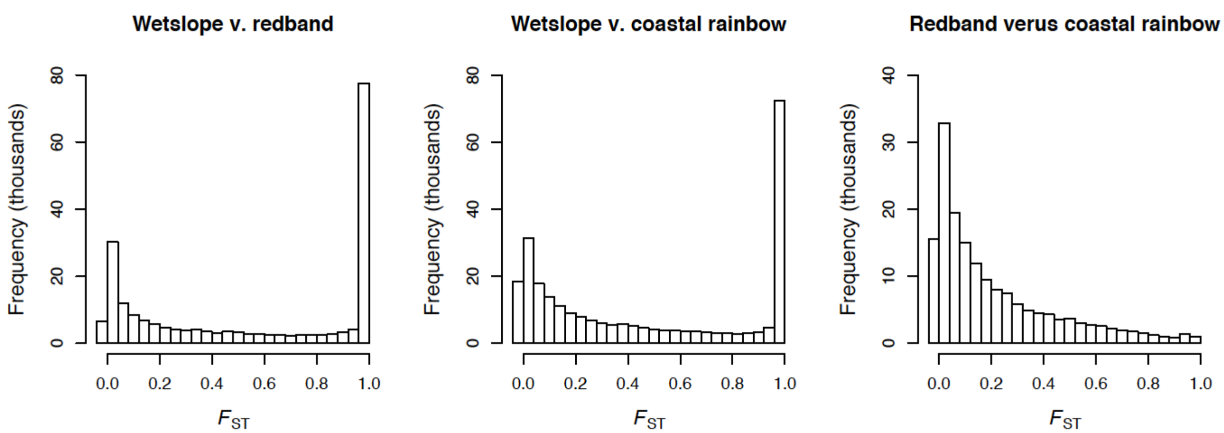


Figure 10-2. Distribution of genetic differentiation between westslope cutthroat trout and redband rainbow trout (left), between westslope cutthroat and coastal (hatchery-origin) rainbow trout (center), and between redband and coastal rainbow (right). Genetic differentiation was measured as F_{ST}, where larger values of indicate greater differentiation. For example, F_{ST} = 1 means each population is fixed for a different allele. $\boldsymbol{F_{ST}}$ = 0 means the two populations/species have identical allele frequencies.

Results

Ten Lakes Scenic Area

We collected tissue samples from 108 fish from Weasel, Wolverine and Bluebird creeks in the summer of 2017 and 2018 (Table 10-2). Genetic analyses of these samples are currently ongoing. The results of which will be reported in a subsequent progress report.

Table 10-2. Summary of sampling locations, size, and other species observed in the three main tributaries within the Ten Lakes Scenic Area in 2017 and 2018.

Stream Name	Location	Sample Size	Effort (seconds)	Other species observed
Weasel Creek	48.97738 N -114.74090 W	15	1815	Bull trout
Weasel Creek	48.97558 N -114.73718 W	13	2114	None
Weasel Creek	48.954899 N -114.738482 W	25	1790	None
Wolverine Creek	48.96735 N -114.88411 W	15	1066	None
Wolverine Creek	48.96490 N -114.85750 W	11	1684	Bull trout
Bluebird Creek	48.94665 N -114.89224 W	10	930	Bull trout
Bluebird Creek	48.95748 N -114.87319 W	19	444	Bull trout

Cabinet Mountains Wilderness Area

We collected a total 270 fish tissue samples from the nine streams (30 fish each) listed in Table 10-3, and delivered those samples to the UM genetics laboratory to assess admixture. The samples have been prepared for swift processing that will occur within the next several months. The DNA libraries will then be sent to a commercial sequencing facility for sequencing, followed by standard data analysis at the UM genetics laboratory. The results and interpretation of the analysis will be reported in a subsequent annual report.

Table 10-3. Streams in the Cabinet Mountains Wilderness Area that were sampled in 2017 for genetic analyses.

Stream	**Location**	**Sample size**	**Effort (seconds)**	**Other Species Observed**
Cable Creek	48.159271 N -115.589912 W	30	1413	Bull trout
Parmenter Creek	48.37863 N -115.638313 W	30	1618	Bull trout; Brook trout
Iron Meadow Creek	47.978518 N -115.484204 W	30	1427	Brook trout
No Creek	48.28567 N -115.57281 W	30	599	None
Baree Creek	47.953882 N – 115.510098 W 47.954788 N – 115.511763 W	15 15	1406	Brook trout
Porcupine Creek	48.00353 N -115.39635 W 48.00902 N -115.40959 W	15 15	435 607	Brook trout
Geiger Creek	48.02530 N -115.50522 W 48.01920 N -115.51235 W	15 15	1083 604	Brook trout and sculpin
Bramlett Creek	48.04087 N -115.50984 W	30	1680	None
Flower Creek	48.34755 N -115.65350 W	30	1959	Bull trout

Conclusions

The historic stocking records for the lakes within the Ten Lakes Scenic Area appear to be incomplete based on the genetic analyses (Dunnigan et al 2018). We only found RBT stocking records for Bean Lake, which is currently fishless. However, we found RBT genes present in Bluebird, Lower Wolverine, No Name Wolverine, Little Therriault lakes and the Wigwam River, albeit at relatively low frequency. YCT genes remain present in Rainbow, Weasel, No Name Wolverine, Big Therriault lakes and the Wigwam River, even though YCT stocking was discontinued in the region by 1970. The results of the genetic assessments of the samples collected from the three main tributaries downstream of these lakes will be collectively interpreted with previous genetic analyzes (Leary et al. 2015 and Whitely et al. 2018) to help determine the risk of introgression in downstream waters. The final results of this collective effort will allow us to formulate management alternatives for lakes that contain a strong non-native genetic component that may include no action, accelerated WCT stocking (swamping) to dilute the non-native genetic component within a lake, or chemical removal of existing fish and restocking with WCT.

The previous genetic assessment of the Cabinet Mountains Lakes conducted by Huston et al. (1996) in combination with the tributary genetics work we collected in 2017 may provide an indication of the historic distribution of westslope cutthroat and redband trout within these watersheds. MFWP will use these data to determine if these watersheds will be managed for the conservation of either westslope cutthroat trout or redband trout, with the eventual goal of also formulating management alternatives for the lakes within the Cabinet Mountains. The development of the genetic assessment methodologies for Kootenai Basin redband trout will be a valuable tool for the conservation and restoration of this species.

Chapter 11: Assignment of Libby Reservoir Fishes to Natal Waters

This chapter includes the following work elements:

K: Collect Upper Kootenai Water Chemistry Samples (Contract 76916)

J: Analyze and interpret Libby Mitigation physical and biological data (Contract 76916)
Introduction

Introduction

Burbot abundance has declined upstream of Libby Dam, and investigators don't understand which areas of the basin were historically important for burbot juvenile production. An understanding of burbot early life history is critical to identifying the factors limiting burbot production. The purpose of this applied research project is to quantify the spatial variation of isotopic and elemental geochemical markers in waters and otoliths of resident fish in the upper Kootenai Basin. This analytical tool is intended to serve as a method of identifying natal tributary of origin and life history reconstruction for burbot that were collected by MFWP upstream of Libby Dam prior to their decline. If successful, the results of this applied research project may lend insight into the potential causes of the burbot decline and identify important future mitigation actions needed to recover this native species.

Methods

Phase I (July 1, 2018 to July 1, 2019)

Water Chemistry and Reference Fish Sample Collection

MFWP collected otoliths from presumed resident fish (sculpin, mountain whitefish, rainbow trout, cutthroat trout, or bull trout in order of decreasing species priority) to investigate the differentiation of isotopic markers of strontium ($^{87}Sr/^{86}Sr$) and oxygen ($\delta^{18}O$) and elemental concentrations of Sr, barium (Ba), selenium (Se) and calcium (Ca) in the lotic waters of the upper Kootenai Basin (Table 11-1). However, in Libby Reservoir, we collected redside shiners to investigate the spatial differentiation of these isotopic and elemental markers. In addition to sampling fish, we also collected water samples from each site that will be used to validate the isotopic and elemental markers observed in the fish otoliths. All water and reference fish samples were collected by MFWP staff during the primary growing season in 2018 (summer months). Sampling locations were determined based on suspected or potential spawning locations and recommendations of local fisheries biologists. Fish were collected via electrofishing, euthanized and the otoliths extracted and preserved in centrifuge vials for future mounting and analysis.

Water samples were collected following methods modified from Shiller (2003) and Muhlfeld et al. (2012). All water samples for $^{87}Sr/^{86}Sr$ and elemental concentrations were collected in perfluoroalkoxy (PFA) bottles (>50 mL) by MFWP staff. The bottles were pre-cleaned by washing with Liquinox soap and rinsed three times with Milli-Q water. Bottles were then heated in 2% HNO_3 for one hour, rinsed again with clean 2% HNO_3, and sealed before use. After collection, water samples were preserved by acidifying to ~2% with 7-M HNO_3 (Optima grade), then transported to the laboratory and filtered through polytetrafluoroethylene (PTFE) membranes prior to analysis. Water samples for $\delta^{18}O$ were collected in 60 mL polyethylene bottles and refrigerated at 4°C until shipment to the laboratory.

Water Analyses

MFWP contracted with Mainstream Fish Research LLC to complete the analyses, interpretation, and reporting for all water chemistry work. Sample preparation, including column chemistry, was conducted in Class 1000 clean room under a Class 100 (or less) positive air flow flume hood. Quantification of water $^{87}Sr/^{86}Sr$ was made by either multi-collector inductively coupled plasma mass spectrometry (MC-CP-MS) or thermal ionization mass spectrometry (TIMS), with a precision of +/- 0.000008 for a standard reference material (e.g. NIST 987). Quantification of water $\delta^{18}O$ was made by laser absorption spectroscopy using a Los Gatos water isotope analyzer or equivalent. Water sample results were analyzed to determine if sufficient spatial differentiation existed among tributaries within the basin to warrant analysis of the young of the year resident fish otoliths.

Young of the Year Reference Fish Otolith Analysis

MFWP contracted with Mainstream Fish Research LLC to complete the analyses, interpretation, and reporting for all reference fish otolith chemistry work. Pending the results of the water chemistry analyses, resident young of the year reference fish otoliths will be analyzed to refine the isotopic and elemental markers of waters in the upper Kootenai Basin as sources of provenance for burbot prior to their decline in Libby Reservoir. Otolith $^{87}Sr/^{86}Sr$ will be assayed by laser ablation using a Nu Plasma 2 (Nu Instruments), Neptune (Thermo Fisher) or equivalent (MC-ICP-MS coupled to a 213-nm (Nd:YAG), 193-nm (eximer) or equivalent laser following the methods reported by Hegg et al. (2013). Otolith elemental markers will be similarly analyzed by ICP-MS using a high-resolution instrument (Nu Plasma Atom, Thermo Fisher Element, or equivalent) to eliminate potential interferences on the elements of interest. The $^{87}Sr/^{86}Sr$ and element/Ca ratios will be quantified with a core to edge transect that represents a chemical profile of the individual's chemical life history. The $\delta^{18}O$ will be measured in otolith samples by micro-milling techniques described by Wurster et al. 1999 and Wurster et al. 2005 and analysis using a gas bench (Thermo Scientific Gas Bench II or equivalent) and isotope ratio mass spectrometer (Thermo Fisher Delta V Plus or equivalent).

Table 11-1. List and location of the waters upstream of Libby Dam targeted for water and reference fish otolith collection to investigate the differentiation of isotopic markers of strontium and oxygen and elemental concentrations of strontium, barium, selenium and calcium. Sample locations (Map Point #) are shown on Figure 11-1.

Map Point #	Water body	Water collected	Species (number collected)	Latitude	Longitude
1	Libby Reservoir Lower	Yes	RSS (5)	48.45950	-115.29484
2	Libby Reservoir Mid	Yes	RSS (5)	48.63769	-115.31142
3	Libby Reservoir Upper	Yes	RSS (5)	48.97042	-115.17847
4	Tobacco River Lower	Yes	Sculpin (5)	48.87712	-115.05362
5	Tobacco River Upper	Yes	Sculpin (5)	48.80001	-114.95391
6	Big Creek	Yes	Sculpin (5)	48.74751	-115.35293
7	Grave Creek	Yes	Sculpin (5)	48.79067	-114.93294
8	Fortine Creek	Yes	Sculpin (5)	48.79438	-114.95427
9	Gold Creek	Yes	Sculpin (5)	49.10355	-115.27419
10	Elk River Lower	Yes	Sculpin (5)	49.17969	-115.16713
11	Elk River Upper	Yes		49.55606	-115.00376
12	Wigwam River Lower	Yes	Sculpin (5)	49.26434	-114.99201
13	Wigwam River Upper	Yes		49.18356	-114.96499
14	Sand Creek	Yes	Sculpin (5)	49.34276	-115.29523
15	Bull River Lower	Yes	Sculpin (5)	49.47321	-115.45068
16	Bull River Upper	Yes		49.53350	-115.32567
17	Norbury Creek	Yes	RBT (5)	49.48949	-115.45297
18	St. Mary River Lower	Yes	Sculpin (5)	49.58824	-115.75750
19	St. Mary River Upper	Yes	Sculpin (5)	49.61486	-116.15328
20	Mathew Creek	Yes	WCT (5)	49.62962	-116.05700
21	Mark Creek	Yes	RBT (5)	49.63689	-115.96056
22	Perry Creek	Yes	Sculpin (5)	49.59526	-115.88251
23	Wild Horse River	Yes	Sculpin (5)	49.62012	-115.61827
	Kootenay River				
24	Above Reservoir	Yes	Sculpin (5)	49.45265	-115.43151
25	near Fort Steele	Yes	Sculpin (5)	49.61629	-115.63726
26	near Skookumchuck Creek	Yes	Sculpin (5)	49.91132	-115.73994
27	near Canal Flats	Yes	Sculpin (5)	50.14637	-115.80321
28	Upstream of White River	Yes	Sculpin (5)	50.36074	-115.62803
29	Skookumchuck Creek	Yes	Sculpin (5)	49.91148	-115.76850
30	Lussier River Lower	Yes	Sculpin (5)	49.90915	-115.72613
31	Lussier River Upper	Yes	Sculpin (5) MWF (2) BT (1)	49.96001	-115.67072
32	Finlay Creek	Yes	Sculpin (5)	50.09612	-115.80608
33	White River	Yes	Sculpin (4) MWF (1)	50.35358	-115.62206
34	North Fork White River	Yes	Sculpin (2) WCT (3)	50.23236	-115.26524
35	Middle Fork White River	Yes	Sculpin (5)	50.21795	-115.23988
36	East Fork White River	Yes	Sculpin (5)	50.18108	-115.27763
37	Blackfoot Creek	Yes	BT (5)	50.12731	-115.36341

Phase II (July 1, 2019 to January 15, 2020)

Adult Burbot Otolith Analysis

MFWP contracted with Mainstream Fish Research LLC to complete the analyses, interpretation, and reporting for all adult burbot chemistry work. Pending successful completion of Phase I of this work, MFWP will initiate the second phase of work. MFWP has conducted annual gill netting surveys on Libby Reservoir since 1974, which includes the period prior to the decline of burbot abundance. In addition to catch data, we also collected burbot otoliths. MFWP will randomly select up to 150 burbot otoliths for isotopic and elemental analyses to assign natal origin and life history reconstruction. Assignments and analyses will require appropriate statistical methods including discriminant function analysis, multiple regression, and/or cluster analysis.

Results

Phase I (July 1, 2018 to July 1, 2019)

Water Chemistry and Reference Fish Sample Collection

MFWP collected 155 presumed resident fish from 34 tributaries upstream of Libby Dam (Figure 11-1). These included 123 sculpin, 10 young of the year rainbow trout (RBT), 13 young of the year westslope cutthroat trout (WCT), 3 young of the year mountain whitefish (MWF), and 6 young of the year bull trout (Table 11-1). We also collected a total of 15 redside shiners from three locations on Libby Reservoir (Figure 11-1; Table 11-1). MFWP successfully extracted the otoliths from all fish and delivered them to the contractor for analysis in October 2018 for future analysis. MFWP also collected water samples from 37 locations (Table 11-1) and delivered them to the contractor in October 2018 for analysis.

Water Analyses

The contractor completed the analysis of the water samples and graphically displayed the results. When the results of $^{87}Sr/^{86}Sr$ in relation to $^{18}O/^{16}O$ ($\delta^{18}O$) are plotted (Figure 11-2), an obvious clustering of the values is apparent. These results are favorable for future discrimination analysis. The highest $^{87}Sr/^{86}Sr$ values are found in the west side tributaries along with a low $\delta^{18}O$ value. The mainstem values for both $^{87}Sr/^{86}Sr$ and $\delta^{18}O$ are low, whereas the east side streams cluster with either the mainstem values, or separately as high values of $\delta^{18}O$ and intermediate $^{87}Sr/^{86}Sr$ values.

The $\delta^{18}O$ in relation to latitude is shown in Figure 11-3, and indicates a trend, likely due to evaporation at lower latitudes preferentially taking up ^{16}O, but when the vapor phase changes to rain at higher latitudes, ^{18}O is preferentially deposited. However, as vapor moves northward, more ^{18}O is deposited as rain, so there is simply less ^{18}O present in the clouds at more northly locations (and higher in latitude).

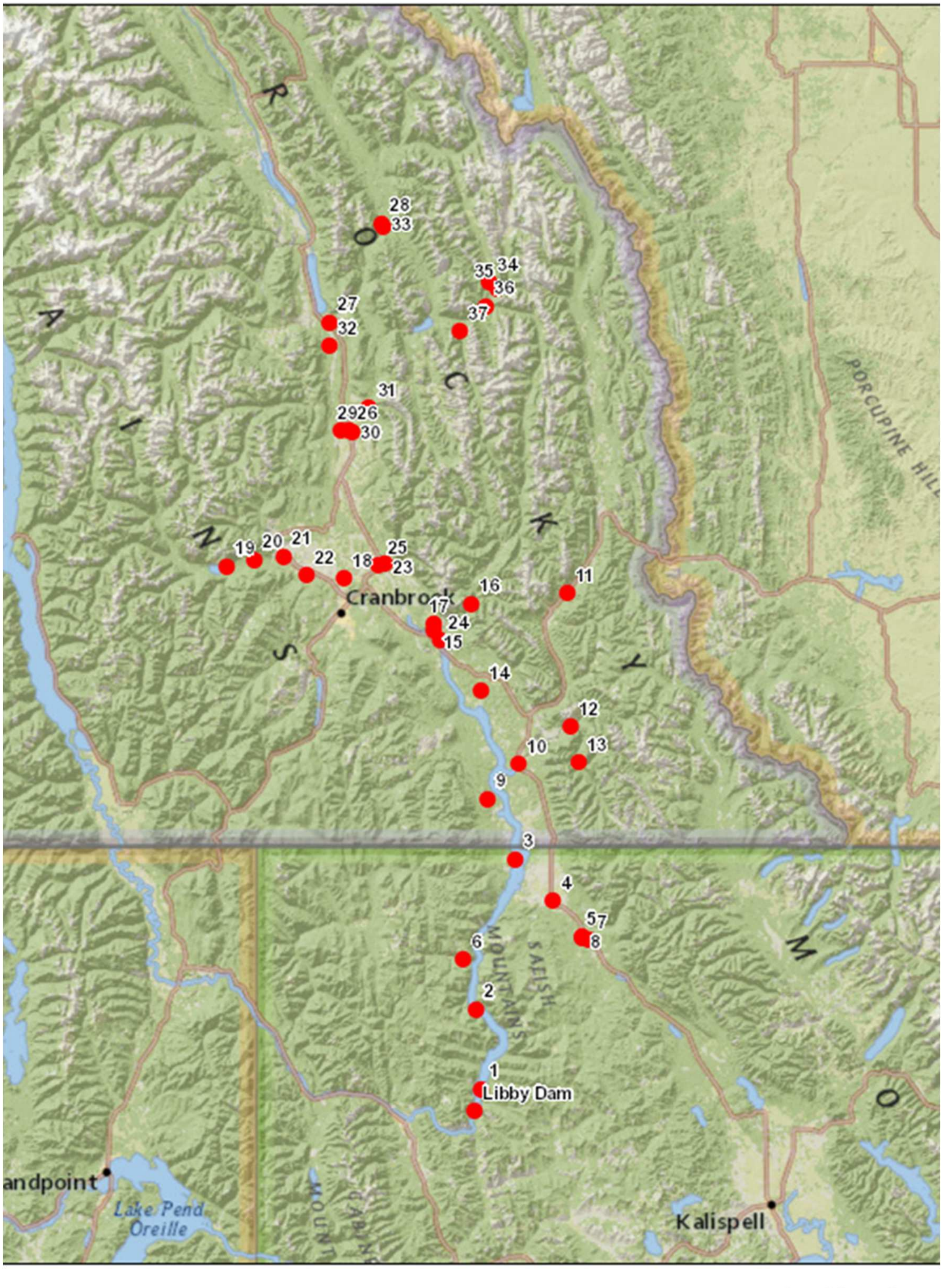


Figure 11-1. Map of the Kootenai Basin upstream of Libby Dam with the sample locations of water and reference fish otolith collection (numbered red dots) to investigate the spatial differentiation of isotopic and elemental markers. Locations of sample numbers are found in Table 11-1.

The results of the relation $^{87}Sr/^{86}Sr$ to Sr/Ca are shown in Figure 11-4. There appears to be wide variation in Sr/Ca across the range of $^{87}Sr/^{86}Sr$ values with no obvious clustering. But this may be advantageous as well for discrimination because it suggests that $^{87}Sr/^{86}Sr$ to Sr/Ca may be orthogonal to each other (uncorrelated), and as such supply independent information. The relationship between Sr/Ca and Ba/Ca is shown in Figure 11-5. Although most of the higher Sr/Ca values are clustered in a narrow band of Ba/Ca, the lowest Sr/Ca values are divided between very high and very low Ba/Ca. This too may yield further discriminating ability.

Young of the Year Reference Fish Otolith Analysis

MFWP conferred with the other scientists performing the laboratory analysis and concluded that a visual inspection of the results of the water chemistry exhibited substantial spatial differentiation to warrant moving forward with the analysis of the reference fish otolith samples. MFWP expects that the results of the resident fish otolith analysis will be completed by March 15 (Table 11-2). Following the laboratory analyses to quantify the chemistry of the resident fish otoliths, additional analysis will evaluate if statistically significant differences among the individual streams or at a coarser scale such as major watersheds exists. If so, combinations of these markers could be used in a linear or quadratic discriminant analysis (or similar classification function) to assign fish of unknown origin to natal streams of origin and potentially characterize their movement between the reservoir and adjoining tributaries.

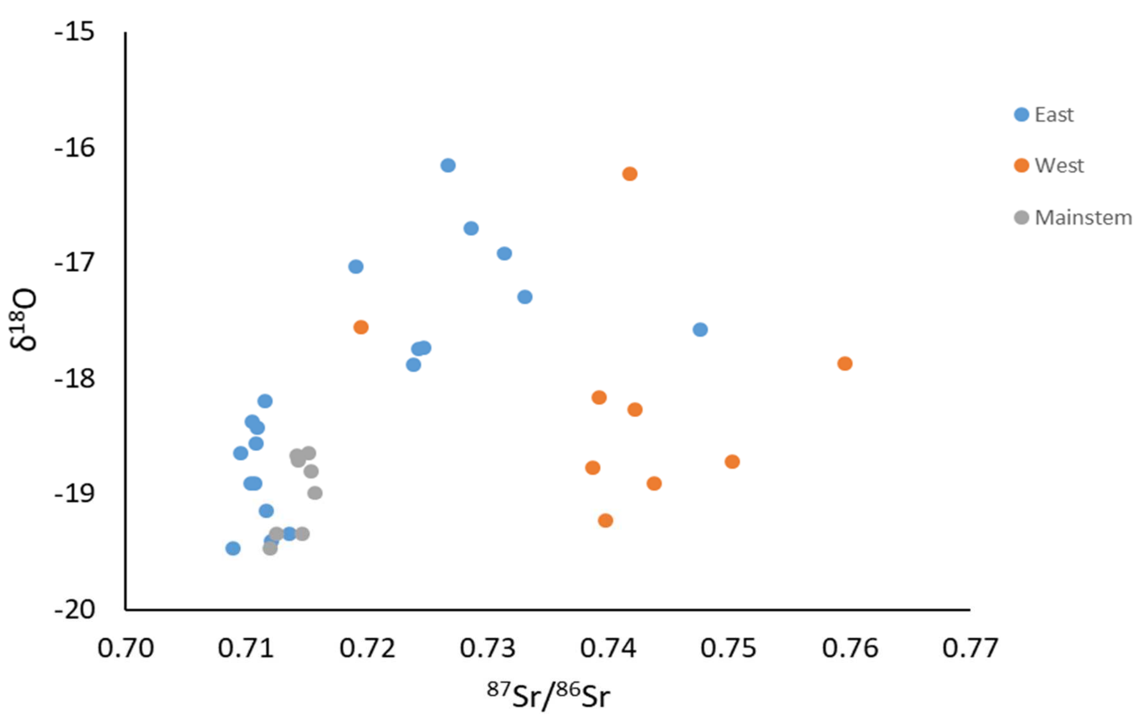


Figure 11-2. Relationship between $^{87}Sr/^{86}Sr$ and $^{18}O/^{16}O$ ($\delta^{18}O$) from 37 water samples collected upstream of Libby Dam. Samples are grouped by mainstem (reservoir), east or west side tributary.

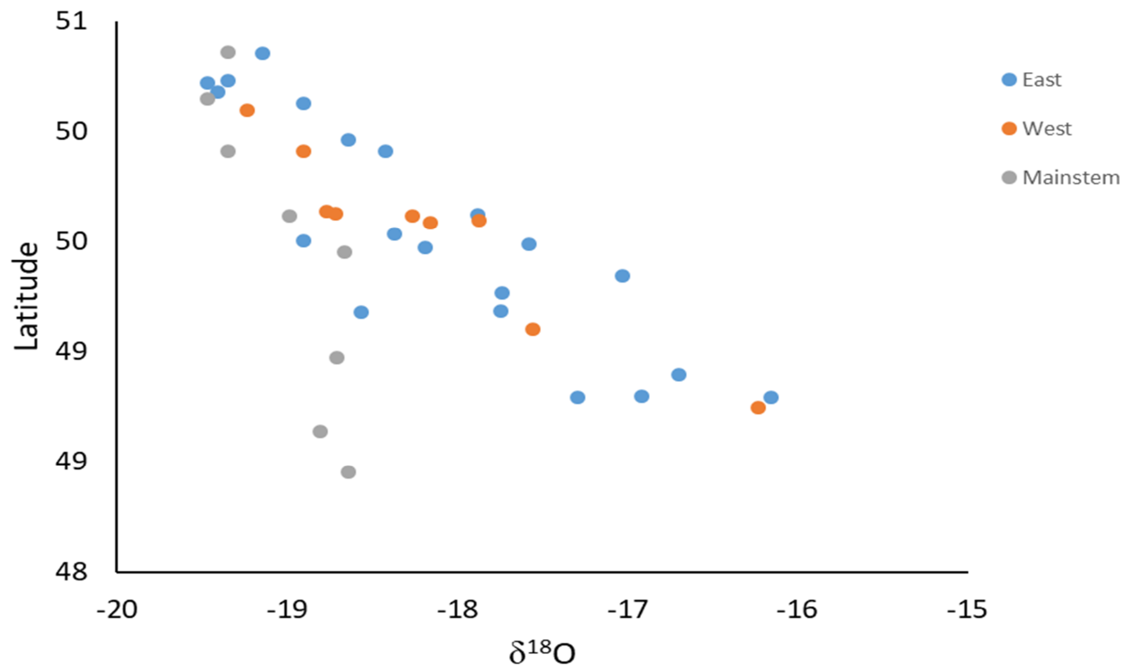


Figure 11-3. Relationship between $^{18}O/^{16}O$ ($\delta^{18}O$) and Latitude from 37 water samples collected upstream of Libby Dam. Samples are grouped by mainstem (reservoir), east or west side tributary.

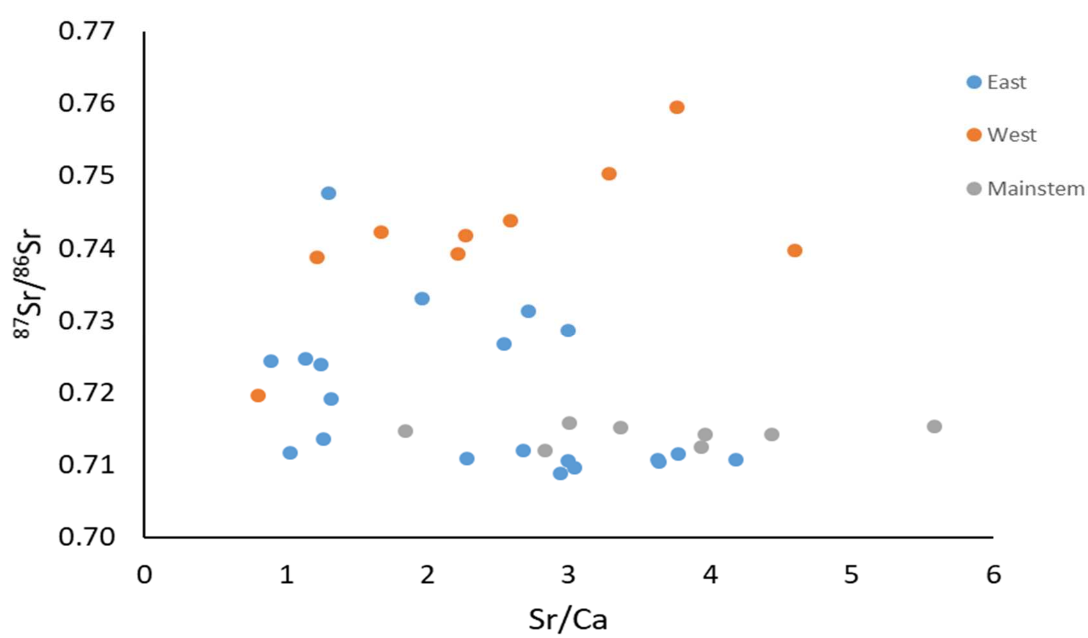


Figure 11-4. Relationship between Sr/Ca and $^{87}Sr/^{86}Sr$ from 37 water samples collected upstream of Libby Dam. Samples are grouped by mainstem (reservoir), east or west side tributary.

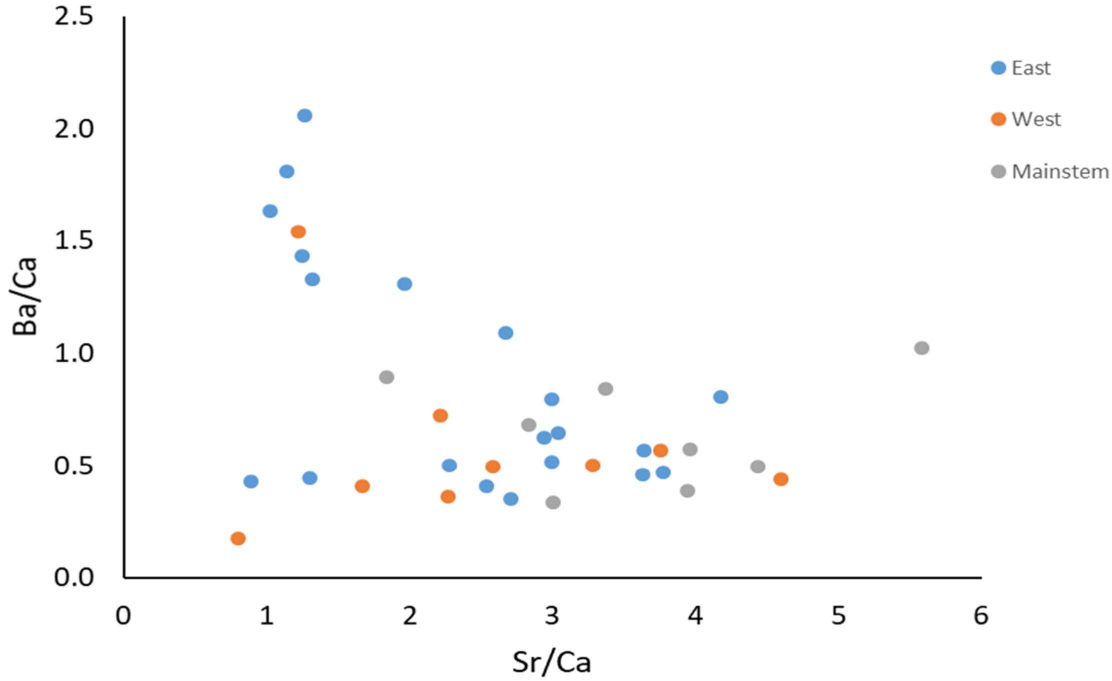


Figure 11-5. Relationship between Sr/Ca and Ba/Ca from 37 water samples collected upstream of Libby Dam. Samples are grouped by mainstem (reservoir), east or west side tributary.

Phase II (July 1, 2019 to January 15, 2020)

Adult Burbot Otolith Analysis

Pending the results of the reference resident fish analyses, MFWP will evaluate if sufficient discriminatory power exists to warrant moving on to the final phase of this applied research project. If successful, MFWP will initiate the second phase of work, MFWP will randomly select up to 150 adult burbot otoliths for isotopic and elemental analyses to assign natal origin and life history reconstruction. This work is expected to be completed by January 15, 2020 (Table 11-2).

Table 11-2. Summary of the milestones and respective completion dates for the remaining work on the upper Kootenai microchemistry applied research project.

Milestone/Deliverable	Due Date
The otoliths of the young of the year resident fish otoliths will be prepared and analyzed to further refine the spatial variation of isotopic and elemental markers of tributaries upstream of Libby Dam.	March 15, 2019
Completion of a draft progress report summarizing the collective results of young of the year resident otoliths and water analyses collected from the upper Kootenai tributaries.	June 1, 2019
Completion of a final progress report summarizing the collective results of young of the year resident otoliths and water analyses collected from the upper Kootenai tributaries.	July1, 2019
Pending the results of successful spatial differentiation of the water and resident reference fish, up to 150 adult burbot otoliths collected from Libby Reservoir will be analyzed to determine isotopic and elemental signatures.	September 30, 2019
Completion of a draft report summarizing the collective results of assignment of adult burbot otoliths collected from Libby Reservoir to natal areas.	December 15, 2019
Completion of a final report summarizing the collective results of assignment of adult burbot otoliths collected from Libby Reservoir to natal areas.	January 15, 2020

Conclusions

Persistence of any population requires long term natural production of adequate frequency and magnitude to prevent extinction. Although not extinct, burbot upstream of Libby Dam have declined over the past 15-20 years (see Chapter 7). Recruitment in fish populations has been generally defined as the number of new fish that enter a population in a given year or that reach a certain size or reproductive stage (Carr and Syms 2006). In populations that provide harvest fisheries, recruitment can also refer to the age at which fish can be caught or captured based on size specific vulnerability to sampling gear (i.e. recruitment to the gear; Jensen 1982). Harvest mortality can influence adult fish abundance, but harvest mortality of burbot on Libby Reservoir is not believed to be a substantial source of mortality to the fishery (MFWP unpublished data).

An understanding of burbot early life history is critical to identifying the factors limiting burbot abundance in the Kootenai Basin upstream of Libby Dam. A critical starting point of understanding the ecology and limiting factors of this species is the identification of important spawning and early rearing habitats. This applied research project may identify these habitats. This analytical tool is intended to serve as a method of identifying natal tributary of origin and life history reconstruction for burbot upstream of Libby Dam that were collected by MFWP prior to their decline. If successful, the results of this applied research project may lend insight into the potential causes of decline and identify important future mitigation actions needed to recover this native species.

References

Ali, O.A. *et al.* 2016. RAD Capture (Rapture): Flexible and Efficient Sequence-Based Genotyping. Genetics 202:389-400.

Allendorf, F.W. 1992. Native trout of western North America. American Fisheries Society, Monograph 6.

Andrews, K.R., J.M. Good, M.R. Miller, G. Luikart, and P.A. Hohenlohe. 2016. Harnessing the power of RADSeq for ecological and evolutionary genomics. Nature Reviews Genetics 17:81-92.

Ardren, W, P. DeHaan, and J. Dunnigan. 2007. Genetic analysis of bull trout in the Kootenai River Basin: Final Report. Bonneville Power Administration. Portland, Oregon. Project Number 1995-004-00.

Ashley, K., L. Thompson, D. Sebastian, D. Lasenby, K. Smokorowski, and H. Andrusak. 1999. Restoration of kokanee salmon in Kootenay Lake, a large intermontane lake, by controlled seasonal addition of limiting nutrients. *In:* Aquatic Restoration in Canada. T. Murphy and M. Munawar, eds. Ecovision World Monograph Series. Backhuys Publishers, Leiden, The Netherlands.

Behnke, R. J. 1992. Native trout of Western North America. American Fisheries Society, Monograph 6, Bethesda, Maryland.

Behnke, R.J. 1996. Conservation assessment for inland cutthroat trout: distribution, status, and habitat management implications. U.S.D.A. Forest Service, Intermountain Region, Ogden, Utah.

Bernard, D. R., G. A. Pearse and R. H. Conrad. 1991. Hoop traps as a means to capture burbot. North American Journal of Fisheries Management 11:91-104.

Binns, N.A. 1994. Long-term responses of trout and macrohabitats to habitat management in a Wyoming headwater stream. North American Journal of Fisheries Management 14:87-90.

Binns, N.A., and R. Remmick. 1994. Response of Bonneville cutthroat trout and their habitat to drainage-wide habitat management at Huff Creek, Wyoming. North American Journal of Fisheries Management 14(4):669-680.

Bisson, P.A., T.P. Quinn, G.H. Reeves, and S.V. Gregory. 1992. Best management practices, cumulative effects, and long-term trends in fish abundance in Pacific Northwest river systems. Pages 189-232 *in* R.J. Naiman, editor. Watershed management. Springer-Verlag, New York.

Breeser, S. W., F. D. Stearns, M. W. Smith, R. L. West, and J. B. Reynolds. 1988. Observations of movements and habitat preferences of burbot in an Alaskan glacial river system. Transactions of the American Fisheries Society 117:506-509.

Britton L.J. and P.E. Greeson, Editors. 1987. Methods for collection and analysis of aquatic biological and microbiological samples. Techniques of Water-Resources Investigations of the United States Geological Survey. Chapter A4. Method #B8020-85.

Brownie, C., J. E. Hines, J. D. Nichols, K. H. Pollock, and J. B. Hestbeck. 1993. Capture–recapture studies for multiple strata including non-Markovian transitions. Biometrics 49:1173–1187.

Brunelli, J.P., G.H. Thorgaard, R.F. Leary, and J.L. Dunnigan. 2008. Single nucleotide polymorphisms associated with allozyme differences between inland and coastal rainbow trout. Transactions of the American Fisheries Society. 137(5):1292-1297.

Burgess, S.A., and J.R. Bider. 1980. Effects of stream habitat improvements on invertebrates, trout populations, and mink activity. Journal of Wildlife Management 44:871-880.

Carr, M., and C. Syms. 2006. Recruitment. Chapter 15, pages 411-427. In: The Ecology of Marine Fishes: California and Adjacent Waters. 2006. L.G. Allen, D.J. Pondella, and M.H. Horn (editors). University of California Press, Berkeley, 670 pp.

Chisholm, I.M. and P.D. Hamlin. 1987. 1985 Libby Reservoir angler census. Prepared for Bonneville Power Administration, Project No. 83-467 by Montana Department of Fish, Wildlife and Parks. Kalispell, Montana.

Chisholm, I.M. and J.J. Fraley. 1986. Quantification of Libby Reservoir levels needed to maintain or enhance reservoir fisheries. Annual report. Prepared for Bonneville Power Administration by Montana Department of Fish, Wildlife and Parks. Kalispell, Montana.

Chisholm, I.M., M.E. Hensler, B. Hansen, D. Skaar. 1989. Quantification of Libby Reservoir levels needed to maintain or enhance reservoir fisheries. Methods and Data Summary 1983-1987. Prepared for Bonneville Power Administration by Montana Department of Fish, Wildlife and Parks. Kalispell, Montana.

Dalbey, S., J. DeShazer, L. Garrow, G. Hoffman, T. Ostrowski. 1989. Quantification of reservoir levels needed maintain or enhance reservoir fisheries: methods and data summary 1988-1996. Bonneville Power Administration. Portland, Oregon. Project Number 1983-467.

Daley, R.J., E.C. Carmack, C.B. Gray, C.H. Pharo, S. Jasper, and R.C. Wiegland. 1981. The effects of upstream impoundments of Kootenay Lake, British Columbia, Canada Inland Waters Directorate, Research Institute, Scientific Series, British Columbia, Canada.

DeHaan, P., and B. Adams. 2011. Genetic analysis of Kootenai River bull trout 2009-2010 report. Prepared for Montana Fish, Wildlife and Parks, Libby.

DeHaan, P., L. Godfrey, and W. Ardren. 2008. Genetic assignments of bull trout collected at Libby Dam 2004 to 2007: Final Report. Montana Fish, Wildlife and Parks, Libby.

Dunnigan, J.L, B. Marotz, J. DeShazer, L. Garrow, and T. Ostrowski. 2003. Mitigation for the construction and operation of Libby Dam: Annual report 2001-2002. Bonneville Power Administration. Portland, Oregon. Project Number 1995-004-00.

Dunnigan, J.L, B. Marotz, J. DeShazer, L. Garrow, and T. Ostrowski. 2004. Mitigation for the construction and operation of Libby Dam: Libby Mitigation Program, 2003 annual progress report. Bonneville Power Administration. Portland, Oregon. Project Number 1995-004-00.

Dunnigan, J.L, J. DeShazer, L. Garrow, B. Marotz, T. Ostrowski, and C. Sinclair. 2005. Mitigation for the construction and operation of Libby Dam: Libby Mitigation Program, 2004 annual progress report. Bonneville Power Administration. Portland, Oregon. Project Number 1995-004-00.

Dunnigan, J.L., J. DeShazer, L. Garrow, T. Ostrowski, and M. Benner, and B. Marotz. 2007. Mitigation for the construction and operation of Libby Dam: Libby Mitigation Program, 2005 annual progress report. Bonneville Power Administration. Portland, Oregon. Project Number 1995-004-00.

Dunnigan, J.L., J. DeShazer, L. Garrow, T. Ostrowski, and M. Benner, and B. Marotz. 2008. Mitigation for the construction and operation of Libby Dam: Libby Mitigation Program, 2006 annual progress report. Bonneville Power Administration. Portland, Oregon. Project Number 1995-004-00.

Dunnigan, J.L., and C.L. Sinclair. 2008. Home range and movement patterns of burbot in Koocanusa Reservoir, Montana, USA. Pages 43-54 *in* V.L. Paragamian and D.H. Bennett, editors, Burbot: Ecology, Management, and Culture: American Fisheries Society Symposium 59, Bethesda, Maryland.

Dunnigan, J.L., J. DeShazer, L. Garrow, T. Ostrowski, and M. Benner, R. Leary, J. Tohtz, and D. Neeley. 2009. Mitigation for the construction and operation of Libby Dam: Libby Mitigation Program, 2007 annual progress report. Bonneville Power Administration. Portland, Oregon. BPA Project Number 1995-004-00.

Dunnigan, J., J. DeShazer, L. Garrow, T. Ostrowski M. Benner, J. Lampton, B. Marotz, and J. Tohtz. 2012. Libby Mitigation Program, 2010 Annual Progress Report: Mitigation for the Construction and Operation of Libby Dam. BPA Project Number 1995-004-00.

Dunnigan, J., J. DeShazer, L. Garrow, T. Ostrowski M. Benner, J. Lampton, B. Marotz, and J. Tohtz. 2014. Libby Mitigation Program, 2012 Annual Progress Report: Mitigation for the Construction and Operation of Libby Dam. BPA Project Number 1995-004-00.

Dunnigan, J., J. DeShazer, L. Garrow, T. Ostrowski M. Benner, J. Lampton, J. Tohtz, and M. Boyer. 2016. Libby Mitigation Program, 2016 Annual Progress Report: Mitigation for the Construction and Operation of Libby Dam. BPA Project Number 1995-004-00.

Dunnigan, J., J. DeShazer, L. Garrow, T. Ostrowski M. Benner, J. Lampton, J. Tohtz, and M. Boyer. 2017. Libby Mitigation Program, 2017 Annual Progress Report: Mitigation for the Construction and Operation of Libby Dam. BPA Project Number 1995-004-00.

Fraley, J., B. Marotz, J. Decker-Hess, W. Beattie, and R. Zubic. 1989. Mitigation, compensation, and future protection for fish populations affected by hydropower development in the Upper Columbia System, Montana. USA Regulated Rivers: Research and Management (3):3-18.

Frissell, C.A., and R.K. Nawa. 1992. Incidence and causes of physical failure of artificial habitat structures in streams of western Oregon and Washington. North American Journal of Fisheries Management 12:182-197.

Geum Environmental Consulting, Inc. 2007a. Therriault Creek riparian revegetation plan. Prepared for: Kootenai River Network. Libby, Montana. Also available at: http://geumconsulting.com/index.php/kootenai-river-network-reports/

Geum Environmental Consulting, Inc. 2007b. Therriault riparian revegetation implementation report (contract number 080067). Prepared for: Montana Fish, Wildlife & Parks. Libby, Montana. Also available at: http://geumconsulting.com/index.php/kootenai-river-network-reports/

Geum Environmental Consulting, Inc. 2008c. Therriault riparian revegetation 2008 monitoring report. Prepared for: Kootenai River Network. Libby, Montana. Also available at: http://www.geumconsulting.com/krn

Geum Environmental Consulting, Inc. 2009d. Therriault Creek riparian revegetation, maintenance and monitoring 2009 report. Prepared for: Kootenai River Network. Libby, Montana. Also available at: http://geumconsulting.com/index.php/kootenai-river-network-reports/

Geum Environmental Consulting, Inc. 2010. Therriault Creek riparian revegetation 2010 monitoring, maintenance and phase III implementation report (contract number 110032). Prepared for: Montana Fish, Wildlife & Parks. Libby, Montana. Also available http://geumconsulting.com/index.php/kootenai-river-network-reports/

Geum Environmental Consulting, Inc. 2011. Grave Creek riparian revegetation 2011 implementation report (task order 1008). Prepared for: Kootenai River Network. Libby, http://geumconsulting.com/index.php/kootenai-river-network-reports/

Geum Environmental Consulting, Inc. 2015. Memorandum: Therriault Creek 2015 Maintenance and Monitoring Summary. Prepared for: Montana Fish, Wildlife and Parks, Libby, Montana.

Geum Environmental Consulting, Inc. 2019. Memorandum: Therriault Creek 2018 Maintenance Summary. Prepared for: Montana Fish, Wildlife and Parks, Libby, Montana.

Geum Environmental Consulting, Inc. 2019a. Therriault Creek Restoration Project Five Year Vegetation Management Plan. Prepared for: Montana Fish, Wildlife and Parks, Libby, Montana.

Graham, P.J., P.W. Fredenberg, and J. Huston. 1982. Supplements to recommendations for a fish and wildlife program. Submitted to: Pacific Northwest Power Planning Council by Montana Department of Fish, Wildlife and Parks. Kalispell, Montana.

Hamilton, J.B. 1989. Response of juvenile steelhead to instream deflectors in a high gradient stream. Pages 149-157 *in* R.E. Gresswell, B.A. Barton, and J.L. Kershner, editors. Practical approaches to riparian resources management. American Fisheries Society, Montana Chapter, Bethesda, Maryland.

Hauer, R., J.T. Gangemi, and J. Stanford. 1997. Long-term influence of Libby Dam Operation on the ecology of macrozoobenthos of the Kootenai River, Montana and Idaho. Flathead Lake Biological Station. Yellow Bay, MT; Prepared for Montana Fish, Wildlife and Parks. Helena, MT. 61 pp.

Hegg, J.C., Kennedy, B.P., and Fremier, A.K. 2013. Predicting strontium isotope variation and fish location with bedrock geology: Understanding the effects of geologic heterogeneity. Chemical Geology 2013: 89-98.

Hestbeck, J. B., J. D. Nichols, and R. A. Malecki. 1991. Estimates of movement and site fidelity using mark–resight data of wintering Canada geese. Ecology 72:523–533.

Hoffman, G. B. Marotz, J. DeShazer, L. Garrow, T. Ostrowski, and J. Dunnigan. 2002. Mitigation for the construction and operation of Libby Dam: Annual report 2000. Bonneville Power Administration. Portland, Oregon. Project Number 199500400.

House, R.A., and P.L. Boehne. 1986. Effects of instream structures on salmonid habitat and populations in Tobe Creek, Oregon. North American Journal of Fisheries Management 6:38-46.

Hunt, R.L. 1976. A long-term evaluation of trout habitat development and its relation to improving management-related research. Transactions of the American Fisheries Society 150(3):361-364.

Huston, J.E., M.E. Hensler, and G. Sage. 1996. A genetic survey of lakes in the Cabinet Wilderness Area and proposed inland rainbow trout recovery, a report to the U.S. Fish and Wildlife Service. Montana Fish, Wildlife and Parks, Helena, Montana.

Huston, J.E. 1995. A report on Kootenai River Drainage native species search. 1994. Montana Fish, Wildlife and Parks, Helena, Montana.

Huston, J.E., P.D. Hamlin, and B. May. 1984. Lake Koocanusa fisheries investigations final report. Prepared for United States Army Corp of Engineers, Seattle District. Seattle, Washington.

ISAB. 1997. The Normative River. Independent Scientific Advisory Board report to the Northwest Power Planning Council and National Marine Fisheries Service. Portland, OR.

Jenkins, R.M. 1967. The influence of some environmental factors on standing crop and harvest of fishes in U.S. reservoirs. Pages 298-321 *in* Reservoir fishery resources symposium. Southern Division American Fisheries Society, Bethesda MD, U.S.A.

Jenson, A.L. 1982. Adjusting catch curves for gill net selection with the logistic distribution. Fisheries Research (1):155-162.

Kendall, W. L. 1999. Robustness of closed capture–recapture methods to violations of the closure assumption. Ecology 80:2517–2525.

Kendall, W. L., J. D. Nichols, and J. E. Hines. 1997. Estimating temporary emigration using capture–recapture data with Pollock's robust design. Ecology 78:563–578.

Kendall, W. L., K. H. Pollock, and C. Brownie. 1995. A likelihood-based approach to capture–recapture estimation of demographic parameters under the robust design. Biometrics 51:293–308.

Kimmel, B.L. and A.W. Groeger. 1986. Limnological and ecological changes associated with reservoir aging. Pages 103-109 *in* G.E. Hall and M.J. Van Den Avyle, editors. Reservoir Fisheries Management: Strategies for the 80's. Reservoir Committee, Southern Division American Fisheries Society. Bethesda, Maryland.

Knudson, K. 1994. Water quality status report: Kootenay (Kootenai) River Basin British Columbia, Montana and Idaho. Kootenai River Network. Libby, MT. and Ecological Resource Consulting, Helena, MT. 57 pp.

Kootenai Tribe of Idaho (KTOI) and Montana Fish, Wildlife & Parks (MFWP). 2004. Kootenai Subbasin Plan. Available on CD from the Kootenai Tribe of Idaho, Bonners Ferry, ID and Montana Fish, Wildlife & Parks, Kalispell. Also available at: http://www.nwcouncil.org/fw/subbasinplanning/kootenai/plan/

Krueger, K. L., and W. A. Hubert. 1997. Assessment of lentic burbot populations in the Big Horn/Wind River Drainage, Wyoming. Journal of Freshwater Ecology 12(3):453-463.

Leary, R.F., F.W. Allendorf, and S.R. Phelps. 1983. Electrophoretic examination of trout from Lake Koocanusa, Montana: Inability of morphological criteria to identify hybrids. Population Genetics Laboratory Report 83/8, MIMCO. University of Montana, Missoula, Montana.

Leary, R. 2005. Letter to Murray Spring Fish Hatchery from Robb Leary. Eureka, Montana.

Leary, R., S. Painter, and A. Lodmell. 2015. Letter to Matt Boyer from Robb Leary, dated December 18, 2015. Kalispell, Montana.

Leary, R.F., G.K. Sage, K. Naohisa, and F.W. Allendorf. Hybridization without introgression between westslope cutthroat and rainbow trout in an area of natural sympatry. (Unpublished).

Leathe, S.A. and P.J. Graham. 1982. Flathead Lake fish food habits study. Final Report. Prepared for the Environmental Protection Agency by Montana Department of Fish, Wildlife and Parks. Kalispell, Montana.

Lemly, D.A. 2002. Symptoms and implications of selenium toxicology in fish: The Belews Lake case example. Aquatic Toxicology 57(1-2):39-49.

Marotz, B.L., D. Gustafson, C. Althen and B. Lonnen. 1996. Model development to establish integrated operational rule curves for Hungry Horse and Libby Reservoirs-Montana. Report to Bonneville Power Administration by Montana Department of Fish, Wildlife, and Parks, Kalispell, Montana. 114pp.

Marotz, B.L., D. Gustafson, C.L. Althen, and W. Lonon. 1999. Integrated operational rule curves for Montana reservoirs and application for other Columbia River storage projects. Pages 329-352 In Ecosystem Approaches for Fisheries Management. Alaska Sea Grant College Program. AK-SG-99-01, 1999.

Marshall, B.D. 2007. Effects of Libby Dam, Habitat, and an invasive diatom, *Didymosphenia geminata*, on the benthic macroinvertebrate assemblages of the Kootenai River, Montana. Prepared for Montana, Fish, Wildlife & Parks, Kalispell, Montana.

May B. and J. Huston 1983. Kootenai river investigations final report: Fisheries investigations. Montana Department of Fish, Wildlife and Parks in cooperation with U.S. Army Corps of Engineers.

MacCrimmon, H.R. 1971. World distribution of rainbow trout (Salmo gairdneri). Journal Fisheries Research Board of Canada 28:663-704.

McNeil, W. J. and W. H. Ahnell. 1964. Success of pink salmon spawning relative to size of spawning bed materials. U. S. Fish and Wildlife Service, Special Scientific Report 169. Washington, DC

Miller W.J., Geise D. 2004. Kootenai River instream flow analysis. Miller Ecological Consultants, Inc., Spatial Sciences Imaging.

Montana Bull Trout Scientific Group (MBTSG). 1996b. Upper Kootenai River drainage Bull Trout status report (including Lake Koocanusa, Upstream of Libby Dam). Prepared for The Montana Bull Trout Restoration Group. Helena, MT.

Montana Fish, Wildlife & Parks, Confederated Salish Kootenai Tribes, and the Kootenai Tribe of Idaho. 1998. Fisheries mitigation and implementation plan for losses attributable to the construction and operation of Libby Dam. Montana Fish, Wildlife and Parks, Kalispell, MT., Confederated Salish and Kootenai Tribes, Pablo, Montana, and Kootenai Tribe of Idaho, Bonners Ferry, Idaho.

Montana Fish, Wildlife & Parks (MFW). 2015. Montana Statewide Angling Pressure Survey 2011. Montana Fish, Wildlife and Parks, Helena, MT., http://fwp.mt.gov/fishing/anglingPressureSurveys/2013.html

Muhlfeld, C.C., Thorrold, S.R., McMahon, T.E., and Marotz, B. 2012. Estimating westslope cutthroat trout (*Oncorhynchus clarkii lewisii*) movements in a river network using strontium isotopes. Canadian Journal of Fisheries and Aquatic Sciences 69:906-916.

Muhlfeld C.C, S.E. Albeke, S.L. Gunckel, B.J. Writer, B.B. Shepard, and B.E. May. 2015. Status and Conservation of Interior Redband Trout in the Western United States, North American Journal of Fisheries Management, 35:1, 31-53, DOI: 10.1080/02755947.2014.951807

Northcote, T. G. 1973. Some impacts of man on Kootenay Lake and its salmonids. Great Lakes Fishery Commission, Technical Report Number 25, Ann Arbor, Michigan.

Paragamian, V. L. 1990. Fish populations of Iowa rivers and streams. Technical Bulletin Number 3. Iowa Department of Natural Resources, Des Moines.

Paragamian, V. L. 1994. Kootenai River fisheries inventory: stock status of burbot and rainbow trout and fisheries inventory. Idaho Department of Fish and Game, Bonneville Power Administration, Project 88-65. Boise, ID.

Paragamian, V. L. 2000. The effect of variable flows on burbot spawning migrations in the Kootenai River, Idaho, USA, and Kootenay Lake, British Columbia, Canada, after construction of the Libby Dam. Pages 111-123 i*n* V. L. Paragamian and D. W. Willis, editors. Burbot: biology, ecology, and management. American Fisheries Society, Fisheries Management Section, Publication Number 1, Bethesda.

Paragamian, V. L., and G. Kruse. 2001. Kootenai River White Sturgeon Spawning Migration Behavior and a Predictive Model North American Journal of Fisheries Management. 21:10-21.

Partridge, F. 1983. Kootenai River Fisheries Investigations. Idaho Department of Fish and Game Completion Report, Boise.

Pattenden, R., M. Miles, L. Fitch, G. Hartman, R. Kellerhals. 1998. Can instream structures effectively restore fisheries habitat? Pages 1-11 *in* M.K. Brewin and D.M.A. Monita, technical coordinators Forest-fish conference: land management practices affecting aquatic ecosystems. Proceedings Forest-Fish Conference, May 1-4, 1996, Calgary Alberta. Natural resources Canada, Canadian Forest Service. North Forest Centre, Edmonton, Alberta. Information report NOR-X-356.

Phelps, S.R. and F.W. Allendorf. 1980. Identification of the source of rainbow trout in Lake Koocanusa: examination of five Canadian hatchery stocks. Zoology Department, University of Montana, Missoula, Montana.

R Core Team (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Reeves, G.H., D.B. Hohler, B.E. Hansen, F.H. Everest. J.R. Sedell, T.L. Hickman, and D. Shively. 1997. Fish habitat restoration in the Pacific Northwest: Fish Creek of Oregon. Pages 335-359 *in* J. E. Williams, C.A. Wood, and M. P. Dombeck, editors. Watershed restoration: principles and practices. American Fisheries Society, Bethesda, Maryland.

Roni, P., T.J. Beechie, R.E. R.E. Bilby, F. E. Loenetti, M.M. Pollock, and G.R. Pess. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. North American Journal of Fisheries Management 22(1):1-20.

Roper, B.B, J.J. Dose, and J.E. Williams. 1997. Stream restoration: Is fisheries biology enough? Fisheries 22(5):6-11.

Rosgen, D.L. 1996. Applied River Morphology. Printed Media Companies, Minneapolis, Minnesota. 343 pp.

Ross, T.J. 2013. Kootenai River Fisheries Investigations: Ecosystem Restoration and Trout Studies. Idaho Department of Fish and Game, 2012 Annual Report to Bonneville Power Administration, Project 88-65, Boise.

Ross, T.J., K. McDonnell, R. Hardy, and S. Stephenson. 2018. Kootenai River Resident Fish Mitigation: White Sturgeon, Burbot, Native Salmonid Monitoring and Evaluation. Annual Progress Report May 1, 2016 to April 30, 2017. Idaho Department of Fish and Game. Report to Bonneville Power Administration, Project 1988-065-00. Boise.

Sage, G.K. et al. 1992. Genetic analysis of 45 trout populations in the Yaak River Drainage, Montana. Wild Trout and Salmon Genetics Laboratory Report 9213. Division of Biological Science, University of Montana, Missoula, Montana.

Sato, S. 2000. Upper Libby Creek stabilization analysis and recommendations. Master of science thesis report, University of Montana, Missoula. 131 pp.

Saunders, J.W., and M.W. Smith. 1962. Physical alteration of stream habitat to improve brook trout production. Transactions of the American Fisheries Society 91:185-188.

Scott, W. B. and E. J. Crossman. 1973. Freshwater fishes of Canada. Bulletin 184, Fisheries Research Board of Canada, Ottawa. 1973.

Shepard, B., B. May, and W. Urie. 2003. Status of westslope cutthroat (*Oncorhynchus clarki, lewisi*) in the United States: 2002. Also available at http://fwp.mt.gov/fwpDoc.html?id=7538

Shepard B.B., Pratt K.L., and P.J. Graham. 1984. Life histories of westslope cutthroat trout and bull trout in the upper Flathead River basin, Montana. Montana Department of Fish, Wildlife & Parks, Helena, MT.

Shiller, A.M. 2003. Syringe filtration methods for examining dissolved and colloidal trace element distributions in remote field locations. Environmental Science Technologies 37:3953-3957.

Sieburth, J.McN., V. Smetacek and J. Lenz. 1978. Pelagic ecosystem structure: heterotrophic compartments of the plankton and their relationship to plankton size fractions. Limnol. Oceanogr., 23 (6): 1256-1263.

Skaar, D., J. DeShazer, L. Garrow, T. Ostrowski and B. Thornberg. 1996. Quantification of Libby Reservoir levels needed to enhance reservoir fisheries. Investigations of fish entrainment through Libby dam, 1990-1994. Final Report. Montana Department of Fish, Wildlife and Parks - Region 1. Prepared for Bonneville Power Administration. Project Number 83-467.

Snyder, E.B. and G.W. Minshall. 1996. Ecosystem metabolism and nutrient dynamics in the Kootenai river in relation to impoundment and flow enhancement for fisheries management. Annual Report. Stream Ecology Center, Idaho State University, Pocatello.

Storm, P.C. et al. 1982. Limnological investigations: Lake Koocanusa, Montana, Part 3 basic data, post impoundment, 1972-1978. U.S. Arm Corps of Engineers, Seattle District, Seattle, Washington.

Sylvester, R., and B. Stephens. 2011. Evaluation of the Biological Effects of the Northwest Power Conservation Council's Mainstem Amendment on the Fisheries Upstream and Downstream of Libby Dam, Montana. Annual Report, July 1, 2009 – June 30, 2010. Prepared by Montana Fish, Wildlife and Parks, Region One, Libby Area Office and Kalispell HQ for Bonneville Power Administration. Bonneville Power Administration Project No. 2006-008-00 Contract No. 43309.

Sylvester, R.M., B.C. Stephens, and J.T. Frye, 2015, Mainstem Columbia Amendments Research at Libby Dam, 1/1/14 to 12/31/14 Annual Report, 2006-008-00.

Sylvester, R.M., B.C. Stephens, and J.T. Frye, 2016, Mainstem Columbia Amendments Research at Libby Dam, 1/1/15 to 12/31/15 Annual Report, 2006-008-00.

Underwood, A.J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. *Ecological Applications* 4: 3–15.

U.S. Federal Register. 2003. Dated March 11, 2003. Volume 68, Number 47, pages 11574-11579.

U.S. Forest Service (USFS). 2013. Dunn Creek watershed restoration conceptual design. Libby Ranger District, Libby, Montana.

U.S. Fish and Wildlife Service. 1999a. Recovery Plan for the white sturgeon (*Acipenser transmontanus*): Kootenai River population. U.S. Fish and Wildlife Service. Portland, OR. 96 pp. Plus appendices.

U.S. Fish and Wildlife Service. 1999b. Endangered and threatened wildlife and plants; determination of threatened status for bull trout in the coterminous United States. Federal Register. 64**:**(1 November 1999), 58910-58933.

U.S. Fish and Wildlife Service. 2000. Biological Opinion on Federal Columbia River Power System Operations. U.S. Fish and Wildlife Service. Portland, OR. 97 pp. Plus appendices

U.S. Fish and Wildlife Service. 2002. Chapter 4, Kootenai River Recovery Unit, Oregon. 89 p. *In*: U.S. Fish and Wildlife Service. Bull Trout (*Salvelinus confluentus*) Draft Recovery Plan. Portland, Oregon.

Van Deventer, J.S., and W.S. Platts. 1983. Sampling and estimating fish populations from streams. Transactions North American Wildlife and Natural Resources Conference 48:349-354.

Voelz, N.J. and J.V. Ward. 1991. Biotic responses along the recovery gradient of a regulated stream. Canadian Journal of Fisheries and Aquatic Sciences. 48: 2477-2490.

Vollenweider, R. A. 1968. Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication: Paris, Organization for Economic Cooperation and Development, Technical report DAS/CSI/68.27, 250 p.Wetzel, R. G., 1975, Limnology: Philadelphia, W. B. Saunders Co.743 p.

Wetzel, R.G., and G.E. Likens. 1979. Limnological analyses. Saunders Company. Philadelphia. 391 pages.

Wetzel, R.G. 1975. Limnology. Saunders Company. Philadelphia. 743 pp.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46(Supplement):120–138.

Whiteley, A., R. Leary, S. Painter, and A. Lodmell. 2018. Letter to Jim Dunnigan from Andrew Whiteley, dated February 17, 2018. Libby, Montana.

Woods, P. F. 1979. Primary productivity in Lake Koocanusa, Montana: Moscow, University of Idaho, Ph.D. dissertation, 112 p.

Woods, P. F. 1981. Physical limnological factors suppressing primary productivity in Lake Koocanusa, Montana, in Stefan, H. G., ed., Proceedings of a symposium on surface water impoundments, v. 2. New York, American Society of Civil Engineers, p. 1368–1377.

Woods, P. F. 1982. Annual nutrient loadings, primary productivity, and trophic state of Lake Koocanusa, Montana and British Columbia, 1972–80. (Geological Survey Professional Paper 1283). 24p.

Woods, P.F., and C.M. Falter. 1982. Annual nutrient loadings, primary productivity, and trophic state of Lake Koocanusa, Montana and British Columbia, 1972-80. Geological Survey Professional Paper 1283, United States Government Printing Office.

Wurster C. M., Patterson W, P., and Cheatham M. M. (1999) Advances in micromilling techniques: A new apparatus for acquiring high- resolution oxygen and carbon stable isotope values and major/minor elemental ratios from accretionary carbonate. *Computers and Geosciences* 25(10):1159-1166.

Wurster C. M., Patterson W. P., Stewart D. J., Stewart T. J., and Bowlby J. N. (2005) Thermal histories, stress, and metabolic rates of chinook salmon in Lake Ontario: evidence from intra-otolith 18O and 13C values and energetics modeling. Canadian Journal of Fisheries and Aquatic Sciences 62:700-713.

Zar, J.H. 1996. Biostatistical analysis. Third edition. Prentice-Hall Inc. Englewood Cliffs, New Jersey.

Appendix

Table A1. Young Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis. If a confidence interval was not possible, it is represented with n/a.

Year	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Section 1 (Tooley)												
Cutthroat Trout[B]	3 (n/a)	36 (37)	139 (148)	Not	55 (64	88 (96)	Not	68 (70)	66 (72)	61 (63)	47 (51)	87 (95)
Rainbow Trout[B]	19 (23)	62 (70)	3 (n/a)	Sampled	2 (n/a)	14 (19)	Sampled	8 (n/a)	2 (n/a)	2 (n/a)	2 (n/a)	2 (n/a)
Brook Trout	11 (17)	120 (124)	102 (105)		36 (39)	30 (31)		20 (n/a)	72 (80)	30 (36)	20 (24)	41 (44)
Bull Trout	0	0	0	0	0	0	0	2 (n/a)	10 (14)	0	0	2 (n/a)
Mountain	0	0	0		0	2 (n/a)		2 (n/a)	4 (n/a)	2 (n/a)	0	0
Total Population [A]	36 (40)	220 (228)	248 (258)		96 (107)	148 (158)		96 (98)	86 (96)	95 (101)	67 (71)	130 (138)
Section 4 (303 Road)												
Westslope	100	439 (500)	352 (367)	Not	130 (142)	222 (237)	Not	218 (228)	327 (351)	323 (337)	165 (170)	382 (398)
Rainbow Trout	0	0	0	Sampled	0	0	Sampled	0	0	2 (n/a)	0	0
Brook Trout	0	0	3 (n/a)		6 (12)	4 (n/a)		10 (12)	12 (17)	26 (30)	5 (11)	38 (43)
Bull Trout	0	0	0		0	0		0	0	0	1 (n/a)	0
Total Population [A]	100	439 (500)	358 (373)		136 (148)	232 (249)		230 (241)	338 (364)	351 (366)	169 (174)	423 (440)
Section 5 (State Project)												
Westslope	Not	216 (227)	256 (290)	126 (153)	153 (174)	268 (290)	178 (183)	115 (118)	151 (164)	137 (143)	57 (60)	174 (191)
Rainbow Trout	Sampled	0	0	0	0	0	0	0	0	0	0	0
Brook Trout		62 (71)	52 (65)	19 (22)	25 (27)	46 (49)	35 (n/a)	60 (63)	142 (147)	93 (96)	57 (60)	71 (77)
Bull Trout		0	0	0	0	2 (n/a)	0	3 (n/a)	2 (n/a)	3 (5)	2 (n/a)	0
Total Population [A]		280 (294)	314 (353)	113 (119)	176 (195)	315 (335)	213 (183)	230 (241)	296 (309)	115 (122)	115 (122)	245 (265)

Table A1 (Continued). Young Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis. If a confidence interval was not possible, it is represented with n/a.

Year	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Section 1 (Tooley)										
Cutthroat Trout[B]	38 (42)	44 (46)	77 (79)	45 (46)	50 (52)	49 (50)	48 (49)	91 (93)	71 (80)	51 (57)
Rainbow Trout[B]	21 (23)	2 (n/a)	4 (n/a)	4 (n/a)	0	0	7 (n/a)	0 (n/a)	4 (n/a)	0
Brook Trout	45 (46)	69 (76)	55 (68)	10 (n/a)	68 (71)	57 (59)	20 (n/a)	32 (34)	32 (n/a)	16 (17)
Mountain	0	0	37 (n/a)	0	10 (n/a)	34 (n/a)	13 (n/a)	2 (n/a)	0	16
Bull Trout	28 (30)	40 (66)	6 (n/a)	0	8 (n/a)	8 (n/a)	11 (n/a)	9 (n/a)	22 (n/a)	8 (n/a)
Total Population [A]	104 (108)	113 (119)	134 (141)	60 (61)	126 (129)	136 (138)	72 (73)	123 (125)	116 (123)	66 (72)
Section 4 (303 Road)										
Westslope	339 (349)	372 (392)	340 (356)	215 (229)	290 (300)	286 (295)	276 (283)	414 (432)	313 (330)	126 (133)
Rainbow Trout	0	0	0	0	0	0	0	0	0	0
Brook Trout	33 (37)	41 (44)	24 (25)	25 (28)	31 (33)	80 (83)	58 (59)	48 (52)	49 (70)	16 (n/a)
Bull Trout	0	0	0	0	0	0	0	0	0	0
Total Population [A]	374 (384)	415 (436)	363 (379)	241 (255)	323 (337)	366 (375)	333 (340)	464 (483)	362 (384)	137 (141)
Section 5 (State Project)										
Westslope	90 (98)	247 (265)	136 (140)	93 (98)	131 (205)	219 (229)	232 (239)	224 (229)	341 (357)	136 (137)
Rainbow Trout	0	0	0	0	0	0	0	0	0	0
Brook Trout	64 (82)	108 (111)	149 (154)	62 (64)	100 (104)	186 (205)	129 (136)	157 (167)	113 (124)	91 (94)
Bull Trout	2 (n/a)	0	3 (n/a)	0	2 (n/a)	3 (n/a)	0	0	5 (n/a)	24 (31)
Total Population [A]	154 (170)	356 (365)	288 (295)	158 (163)	302 (309)	381 (399)	363 (373)	366 (376)	476 (494)	231 (235)

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

[B] In 1996 sampling crew did not distinguish between westslope cutthroat trout and rainbow trout.

Table A2. Therriault Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis. 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. If a confidence interval was not possible, it is represented with n/a.

Year	1997	1998	1999	2001	2003	2004	2005	2006	2007	2008
Section 1- Below Project										
Rainbow Trout	123 (261)	130 (151)	82 (89)		56 (57)	108 (111)	106 (119)	121 (124)	53 (n/a)	135 (139)
Cutthroat Trout	0	0	0	Not	0	0	0	0	0	4 (n/a)
Brook Trout	41 (47)	49 (56)	60 (64)	Sampled	59 (66)	11 (13)	66 (73)	114 (120)	101 (104)	49 (55)
Bull Trout	0	0	0		0	92 (95)	10 (n/a)	48 (54)	28 (31)	4 (n/a)
Total[A]	149 (214)	182 (207)	141 (149)		115 (122)	200 (203)	175 (201)	235 (241)	154 (157)	187 (193)
Section 2 – Therriault Creek Project										
Rainbow Trout	36 (41)	79 (82)	76 (83)	93 (102)	84 (n/a)	102 (107)	32 (34)	42 (43)	11 (n/a)	33 (34)
Cutthroat Trout	0	0	0	0	0	0	0	0	2 (n/a)	0
Brook Trout	56 (58)	125 (137)	72 (80)	82 (87)	58 (61)	24 (27)	67 (91)	46 (48)	40 (42)	37 (39)
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)	4 (n/a)
Total[A]	92 (96)	205 (217)	149 (163)	180 (193)	144 (151)	153 (160)	95 (107)	123 (125)	53 (55)	70 (73)
Section 3 – Above Project										
Rainbow Trout	54 (58)	164 (170)	177 (205)		99 (104)	112 (117)	99 (109)	28 (29)	15 (n/a)	54 (55)
Cutthroat Trout	0	0	0	Not	0	0	0	0	0	0
Brook Trout	74 (77)	82 (88)	110 (117)	Sampled	67 (72)	41 (45)	82 (90)	46 (48)	57 (59)	48 (51)
Bull Trout	0	0	0		10 (n/a)	3 (n/a)	15 (17)	2 (n/a)	4 (n/a)	7 (10)
Total[A]	66 (93)	248 (257)	284 (308)		170 (180)	118 (124)	183 (201)	74 (76)	72 (74)	102 (105)

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

Table A2. (Continued). 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. If a confidence interval was not possible, it is represented with n/a.

Year	2009	2010	2011[B]	2012	2013	2014	2015	2016	2017
Section 1- Below Project									
Rainbow Trout	113 (115)	77 (n/a)	232 (240)	97 (103)	69 (73)	83 (85)	79 (82)	98 (103)	38 (40)
Cutthroat Trout	0	0	0	0	0	0	0	0	0
Brook Trout	134 (140)	108 (113)	93 (98)	243 (308)	121 (135)	103 (106)	152 (156)	41 (43)	130 (134)
Bull Trout	34 (42)	5 (n/a)	3 (n/a)	14 (n/a)	10 (n/a)	13 (n/a)	3 (n/a)	6 (n/a)	3 (n/a)
Total[A]	247 (252)	185 (189)	324 (334)	333 (379)	191 (203)	190 (193)	196 (201)	142 (148)	165 (168)
Section 2 – Therriault Creek Project									
Rainbow Trout	29 (33)	18 (19)	n/a	14 (15)	12 (13)	42 (43)	71 (72)	73 (78)	38 (39)
Cutthroat Trout	0	0	n/a	1 (n/a)	0	0	1 (n/a)	1 (n/a)	0
Brook Trout	54 (55)	55 (56)	n/a	40 (42)	75 (78)	56 (57)	129 (133)	129 (133)	67 (70)
Bull Trout	7 (n/a)	0	n/a	9 (10)	1 (n/a)	5 (6)	7 (n/a)	15 (n/a)	3 (n/a)
Total[A]	83 (86)	73 (74)	n/a	46 (48)	64 (66)	100 (101)	163 (165)	163 (165)	105 (108)
Section 3 – Above Project									
Rainbow Trout	57 (60)	29 (n/a)	21 (31)	18 (n/a)	36 (37)	62 (65)	72 (73)	58 (60)	60 (65)
Cutthroat Trout	0	0	0	2 (n/a)	0	0	0	0	0
Brook Trout	59 (62)	66 (74)	235 (242)	68 (71)	77 (80)	32 (34)	77 (86)	58 (62)	99 (103)
Bull Trout	59 (62)	4 (n/a)	28 (n/a)	20 (25)	7 (n/a)	7 (n/a)	17 (20)	0	0
Total[A]	116 (120)	93 (101)	255 (265)	84 (87)	115 (119)	99 (103)	119 (122)	116 (121)	162 (169)

A) Includes rainbow, rainbow x cutthroat hybrids, and brook trout. Bull trout were not included in the total population estimate.
B) Sampling crew could not hold the block net in 2011 in Section 2. Therefore, a reliable estimate could not be produced.

Table A2. (Continued). Therriault Creek depletion population estimates for fish $\geq$ 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. If a confidence interval was not possible, it is represented with n/a.

Year	2018
Section 1- Below Project	
Rainbow Trout	89 (93)
Cutthroat Trout	0
Brook Trout	95 (102)
Bull Trout	13 (n/a)
Total[A]	187 (196)
Section 2 – Therriault Creek Project	
Rainbow Trout	69 (71)
Cutthroat Trout	0
Brook Trout	74 (76)
Bull Trout	1 (n/a)
Total[A]	149 (152)
Section 3 – Above Project	
Rainbow Trout	107 (110)
Cutthroat Trout	0
Brook Trout	124 (128)
Bull Trout	10 (n/a)
Total[A]	233 (240)

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

Table A3. Libby Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis. If a confidence interval was not possible, it is represented with n/a.

Year	2000[A]	2001	2002	2003	2004	2005	2006	2007	2008	2009
Section 3 – Upper Cleveland Project										
Rainbow Trout	170 (194)	172 (182)	163 (183)	112 (127)	88 (104)	63 (75)	105 (110)	30 (34)	199 (227)	207 (239)
Brook Trout	0	0	0	0	0	0	0	0	0	0
Bull Trout	3	8 (11)	7 (n/a)	11 (14)	2 (n/a)	2 (n/a)	3 (n/a)	0	0	0
Mountain Whitefish	0	0	1 (n/a)	0	0	0	0	0	0	0
Total Population [B]	170 (194)	172 (182)	163 (183)	112 (127)	88 (104)	63 (75)	105 (110)	30 (34)	199 (227)	207 (239)

Table A3 (Continued). Libby Creek depletion population estimates for fish ≥ 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis. If a confidence interval was not possible, it is represented with n/a.

Year	2010	2011	2012	2013	2014	2015	2016	2017	2018
Section 3 – Upper Cleveland Project									
Rainbow Trout	230 (239)	145 (154)	156	138 (153)	165 (191)	231 (259)	82.9	220 (253)	194 (213)
Brook Trout	4 (n/a)	0	0	0	0	0	0	0	2 (n/a)
Bull Trout	31 (32)	50 (58)	63 (67)	23 (32)	17 (21)	22 (25)	11.4	5 (n/a)	20 (23)
Mountain Whitefish	0	0	0	0	0	0	0	0	0
Total Population [B]	235 (246)	145 (154)	165	138 (153)	165 (191)	231 (259)	82.9	220 (253)	195 (215)

[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. Libby Creek depletion population estimates for fish > 75 mm per 1,000 feet using 95 % confidence intervals. Upper confidence intervals are in parenthesis. If a confidence interval was not possible, it is represented with n/a.

Year	2004	2005	2006	2007	2008	2009	2010	2011	2012
Section 4 – below Lower Cleveland Project									
Rainbow Trout	352	273	314	141	289	351	Not Sampled	210 (222)	258 (275)
Brook Trout	0	2 (n/a)	2 (n/a)	1 (n/a)	4 (n/a)	4 (n/a)		0	0
Bull Trout	5 (n/a)	0	0	1 (n/a)	2 (n/a)	2 (n/a)		6 (n/a)	7 (10)
Total Population[A]	355	276	316	143	291	356		210 (222)	258 (275)
Section 5 –above Lower Cleveland Project									
Rainbow Trout	172	173	170	129	406	201	297 (309)	166 (180)	215 (234)
Brook Trout	0	0	0	0	0	2 (n/a)	4 (n/a)	8 (11)	0
Bull Trout	6 (n/a)	0	0	2 (n/a)	0	6 (9)	8 (n/a)	0	13 (15)
Total Population[A]	172	173	170	129	406	203	301 (313)	174 (191)	215 (234)
Section 6 – Lower Cleveland Project									
Rainbow Trout	218	221	273	133	213	209	310 (325)	120 (169)	245 (258)
Brook Trout	1 (n/a)	0	0	6 (9)	2 (n/a)	2 (n/a)	4 (n/a)	2 (n/a)	0
Bull Trout	0	4 (n/a)	0	0	2 (n/a)	0	7 (10)	2 (n/a)	14 (n/a)
Total Population [A]	219	221	273	141	215	213	310 (325)	130 (191)	245 (258)

[A] 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. If a confidence interval was not possible, it is represented with n/a.

Year	2013	2014	2015	2016	2017	2018
Section 4 – below Lower Cleveland Project						
Rainbow Trout	332 (375)	263 (294)	308 (330)	231.1 (257.3)	296 (306)	Not Sampled
Brook Trout	3 (n/a)	0	0	0	5 (7)	
Bull Trout	5 (n/a)	5 (n/a)	5 (n/a)	0	4 (n/a)	
Total Population[A]	333 (375)	263 (294)	308 (330)	231.1 (257.3)	304 (315)	
Section 5 –above Lower Cleveland Project						
Rainbow Trout	190 (214)	209 (219)	293 (303)	139.4 (151.3)	228 (242)	270 (292)
Brook Trout	0	0	2 (n/a)	0	0	0
Bull Trout	11 (13)	6 (n/a)	6 (n/a)	0	7 (n/a)	7 (n/a)
Total Population[A]	190 (214)	209 (219)	295 (305)	139.4 (151.3)	228 (242)	270 (292)
Section 6 – Lower Cleveland Project						
Rainbow Trout	174 (197)	192 (202)	279 (286)	118.9 (126)	214 (241)	216 (235)
Brook Trout	0	0	2 (n/a)	0	0	6 (n/a)
Bull Trout	0	0	2 (n/a)	1.9 (n/a)	0	2 (n/a)
Total Population [A]	174 (197)	192 (202)	281 (288)	118.9 (126)	214 (241)	224 (246)

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

Table A5. Mean zooplankton densities (number per liter) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Tenmile area of Libby Reservoir during 2017. *Epischura* and *Leptodora* were measured as number per m^3.

Month	**Sample Size**	***Daphnia***	***Bosmina***	***Diaptomus***	***Cyclops***	***Leptodora***	***Epischura***	***Diaphanosoma***
April	3	0.03	0.22	0.01	3.56	0.00	0.00	0.00
		0.00	0.05	0.00	7.64	0.00	0.00	0.00
May	3	0.42	0.45	0.06	15.90	0.00	0.00	0.01
		0.00	0.36	0.01	200.48	0.00	0.00	0.00
June	3	4.42	2.09	0.07	6.36	0.00	57.44	0.00
		12.52	3.11	0.00	0.50	0.00	1,789.83	0.00
July	3	2.18	0.18	0.34	7.92	6.60	29.89	0.00
		0.37	0.04	0.01	0.88	2.67	1,126.46	0.00
August	3	0.74	0.04	1.12	11.24	0.94	72.05	0.04
		0.15	0.00	0.13	9.61	1.16	5,882.67	0.00
Sept.	3	3.05	0.87	0.60	12.19	0.47	35.37	0.04
		5.15	0.16	0.08	38.41	0.66	1,050.67	0.00
Oct.	3	0.84	1.13	0.27	4.11	1.42	81.67	0.02
		0.16	0.04	0.00	2.24	1.50	6,702.61	0.00
Nov.	3	0.42	0.90	0.32	2.04	0.00	19.80	0.00
		0.03	0.47	0.02	0.89	0.00	384.63	0.00

Table A6. Mean zooplankton densities (number per liter) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in the Rexford area of Libby Reservoir during 2017. *Epischura* and *Leptodora* were measured as number per m^3.

Month	**Sample Size**	***Daphnia***	***Bosmina***	***Diaptomus***	***Cyclops***	***Leptodora***	***Epischura***	***Diaphanosoma***
April	3	0.05	0.05	0.00	3.29	0.00	0.28	0.00
		0.01	0.00	0.00	26.89	0.00	0.23	0.00
May	3	0.77	0.10	0.01	8.35	0.24	2.55	0.00
		0.30	0.01	0.00	41.32	0.17	19.46	0.00
June	3	8.86	3.88	0.11	12.78	0.24	13.01	0.04
		135.21	0.12	0.01	278.04	0.17	508.04	0.00
July	3	3.04	0.01	0.44	6.08	3.30	17.92	0.02
		0.42	0.00	0.15	4.55	3.18	258.78	0.00
August	3	0.48	0.00	3.48	13.30	0.24	245.92	0.04
		0.02	0.00	1.64	6.22	0.17	25,949.77	0.00
Sept.	3	1.90	0.53	0.42	6.70	0.47	39.23	0.00
		0.03	0.18	0.01	9.66	0.17	52.94	0.00
Oct.	3	0.97	0.79	0.41	5.60	0.86	0.00	0.00
		0.17	0.24	0.01	2.14	0.57	0.00	0.00
Nov.	3	0.63	4.18	0.77	5.54	0.00	6.04	0.00
		0.54	7.44	0.72	35.49	0.00	109.32	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 Canada area of Libby Reservoir during 2017. *Epischura* and *Leptodora* were measured as number per m^3.

Month	Sample Size	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
May	3	0.22	0.15	0.00	6.44	0.00	0.00	0.00
		0.04	0.02	0.00	65.82	0.00	0.00	0.00
June	3	7.03	1.34	0.04	7.27	0.94	0.31	0.08
		135.75	5.14	0.00	144.70	2.67	0.29	0.02
July	3	2.64	0.00	0.95	1.99	1.16	54.97	0.01
		3.87	0.00	0.45	2.04	0.19	1,123.59	0.00
August	3	1.87	0.00	2.26	3.97	5.84	260.23	0.01
		0.51	0.00	1.33	15.16	23.30	5,648.07	0.00
Sept.	3	5.09	0.20	1.21	7.75	1.37	4.28	0.03
		29.30	0.06	1.53	49.16	2.01	54.87	0.00
Oct.	3	10.03	2.34	1.91	15.18	0.29	34.58	0.03
		21.66	9.72	1.26	82.85	0.26	3,587.33	0.00
Nov.	3	0.79	4.22	1.35	7.33	0.00	9.43	0.00
		0.10	14.63	1.35	33.14	0.00	266.77	0.00

Table A8. Yearly mean total zooplankton densities (number per liter) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in Libby Reservoir. *Epischura* and *Leptodora* were measured as number per m^3.

Year	Sample Size	*Daphnia*	*Bosmina*	*Diaptomus*	*Cyclops*	*Leptodora*	*Epischura*	*Diaphanosoma*
1997	69	2.80	0.07	0.80	6.10	4.34	57.24	0.08
		11.30	0.01	0.88	50.87	108.72	6,013.80	0.02
1998	72	2.17	0.64	2.22	9.35	3.99	131.58	0.36
		4.00	1.80	9.17	64.33	80.92	47,113.37	0.43
1999	57	2.19	0.77	0.51	9.57	6.63	89.41	0.15
		4.53	1.39	2.35	107.88	148.11	14,367.63	0.05
2000	69	1.07	0.51	0.36	8.04	2.72	51.20	0.05
		0.97	1.06	0.20	80.04	14.05	7,153.52	0.01
2001	72	1.58	0.46	0.46	8.39	2.72	63.72	0.22
		2.77	0.46	0.21	59.53	21.18	11,153.71	0.13
2002	56	1.82	0.65	0.39	8.89	4.88	77.96	1.02
		6.85	1.29	0.22	57.44	139.73	9,041.90	3.62
2003	72	3.42	0.83	1.79	11.34	2.24	98.02	0.9
		20.29	1.93	4.46	64.61	19.74	19,825.83	1.68
2004	72	2.1	1.63	1.38	10.26	3.39	95.06	0.53
		6.7	8.72	3.21	169.71	29.53	37,077.33	0.88
2005	72	1.5	2.62	0.51	7.74	2.43	91.36	0.3
		4.05	37.88	0.59	80.18	26.13	15,412.56	0.19
2006	63	1.81	1.09	1.37	9.1	2.78	121.03	0.23
		2.65	3.42	2.24	69.2	16.67	28,439.64	0.16
2007	54	1.48	0.87	0.68	8.84	1.83	139.38	0.1
		2.19	2.53	0.92	112.66	12.33	50,542.01	0.06
2008	72	1.9	2.23	0.64	11.83	2.25	52.03	0.06
		6.3	10.1	1.11	124.81	13.14	6,960.08	0.02
2009	69	1.44	1.3	0.59	7.94	1.64	65.03	0.05
		7.02	5.04	0.54	98.31	9.44	14,266.98	0.01
2010	60	1.71	2.69	0.13	7.44	2.3	98.14	0.08
		3.49	18.94	0.02	43.14	7.55	17,989.30	0.03
2011	65	2.41	1.09	0.04	7.12	3.19	29.46	0.01
		19.03	6.33	0	75.64	26.26	1,782.70	0
2012	72	1.59	0.33	0.48	6.68	1.35	65.1	0.02
		3.46	0.52	1.05	125.05	7.37	10,877.11	0
2013	63	1.35	0.79	0.65	7.82	1.07	91.89	0.06
		1.80	1.98	0.91	24.02	5.23	14,906.14	0.01

Table A8 (Continued). Yearly mean total zooplankton densities (number per liter) (top line) and variances (bottom line) estimated from 10-20 m. vertical tows made in Libby Reservoir. *Epischura* and *Leptodora* were measured as number per m^3.

Year	**Sample Size**	***Daphnia***	***Bosmina***	***Diaptomus***	***Cyclops***	***Leptodora***	***Epischura***	***Diaphanosoma***
2014	72	2.62	1.70	0.63	12.97	2.03	34.08	0.02
		12.84	25.56	2.18	129.14	14.89	4931.3	0.00
2015	72	1.64	4.03	0.42	9.56	0.89	129.4	0.06
		3.94	30.84	0.59	17.35	0.92	22,483	0.005
2016	72	1.54	1.14	0.86	6.16	1.49	77.71	0.04
		1.76	4.97	1.65	30.87	4.16	12,715.7	0.02
2017	69	2.45	1.03	0.70	7.60	1.06	42.82	0.02
		18.00	3.07	1.00	46.90	4.28	6,461.30	0.00

Table A9. The 1% photosynthetic active radiation (PAR) profile data collected from Libby Reservoir in 2018.

	Kikomun	US/Can	Stonehill	Tenmile	Kikomun	US/Can	Stonehill	Tenmile	Kikomun	US/Can	Stonehill	Tenmile
Depth (M)	NA	5/23/18	5/23/18	5/23/2018	6/19/18	6/19/18	6/20/18	6/20/18	7/16/18	7/16/18	7/17/18	7/17/18
Air		2331	1948.5	2283	1149.9	2507	2391	2207	2012	2418	2243	2212
0		2107	1137.1	1780.8	737.5	2312	1965.9	1683.2	1348.4	1917.9	2436	1547.1
1		1054	382.9	710.9	401.8	1377	1369.3	1082.1	925.7	1014.6	1449.1	1111.1
2		423	106.7	274	167.03	588.1	778.1	490.8	510.4	680.1	972.9	504.8
3		70.18	26.54	104.78	72	284.7	501.7	323.6	326.2	435.1	641.6	354.1
4		4.39	7.43	43.18	31.65	134.34	306.3	179.55	203.8	278.6	421.1	240.6
5			2.6	15.31	14.24	69.69	184.52	108.49	131.26	168.7	275.3	162.86
6				5.94	6.87	41.94	113.67	65.6	83.26	105.48	183.29	111.01
7						30.51	72.91	37.3	52.16	66.43	122.56	77.56
8						21.46	47.32	22.64	33.8	44.94	84.17	54.52
9						13.29	30.21	13.6	22.28	30.89	60.73	37.54
10							19.17		14.35	22.19	42.66	27.17
11							11.95		9.09	16.34	30.92	19.74
12											22.27	14.22
13											16.93	

Table A9 (Continued). The 1% photosynthetic active radiation (PAR) profile data collected from Libby Reservoir in 2018.

	Kikomun	US/Can	Stonehill	Tenmile	Kikomun	US/Can	Stonehill	Tenmile
Depth (M)	8/21/18	8/21/18	8/22/18	8/22/18	9/17/18	9/17/18	9/18/18	9/18/18
Air	1795.8	2432	2045	1872	973.5	2358	2259	2133
0	948.8	1609	1255.3	1068.1	370.5	1980.5	1685.43	1356.8
1	728.4	1158.7	1029.3	836.1	238.9	1543.2	1314.1	791.7
2	463.5	838.2	719.3	513.2	181.69	1115	969.1	673.8
3	327.8	643	518.8	406	145.05	797.7	696.5	533.5
4	237	486.2	387.4	318.5	120.84	575.8	517.8	396.4
5	173.71	362.7	284.1	264.8	95.58	424.5	368.4	319.2
6	128.36	269.9	203.7	206.2	72.66	307	282.2	217.7
7	91.94	193.3	152.92	157.58	56.29	227.6	220	168.95
8	71.19	142.6	113.84	123.67	42.81	169.58	164.08	147.89
9	52.28	105.38	81.58	94.67	31.95	125.81	129.34	110.41
10	38.55	76.9	61.85	73.47	24.21	96.46	100.55	93.62
11	26.78	57.55	46.77	56.16	17.1	73.08	79.7	76.76
12	19	45.02	35.65	44.31	12.7	55.58	64.06	61.05
13	13.2	34.96	26.91	34.05	9.02	42.89	52.3	49.03
14	8.55	26.64	20.38	27.08	6.08	33.89	40.07	40.98
15		20.94	15.5	21.23		26.78	30.93	32.43
16		15.55	12.1	16.48		20.71	24.64	27.17
17						16.3	19.2	21.72
18							15.43	16.64
19								13.98

Table A10. Water chemistry data collected on Libby Reservoir in 2018 during the primary productivity study.

Station	Depth	Date	Alkalinity (mg/L CaCO3)	Ammonia (ug/L)	Nitrate + Nitrite (ug/L)	Total Nitrogen (ug/L)	Total Phosphorus (ug/L)	O-Phosphate (ug/L)	TDS (mg/L)	DIC (mg/L)
KIKOMUN	10M	6/19/2018	80	2.5	73	100	9	0.5	130	9.8
KIKOMUN	10M	7/16/2018	88	5	62	140	4.8	0.5	100	7.4
KIKOMUN	10M	8/21/2018	76	2.5	57	40	2.7	0.5	160	24
KIKOMUN	10M	9/17/2018	100	8	32	30	3.5	2.4	170	25
KIKOMUN	3M	6/19/2018	82	2.5	72	100	5	0.5	98	19
KIKOMUN	3M	7/16/2018	86	10	50	100	5.5	0.5	130	16
KIKOMUN	3M	8/21/2018	86	2.5	52	110	4.8	1.5	150	23
KIKOMUN	3M	9/17/2018	100	18	86	80	3.4	0.5	180	25
STONEHILL	10M	5/22/2018	68	2.5	260	460	19.2	2.4	92	1
STONEHILL	10M	6/20/2018	80	2.5	150	190	6.8	1.2	110	20
STONEHILL	10M	7/17/2018	78	7	150	140	9	0.5	93	7.3
STONEHILL	10M	8/22/2018	90	2.5	99	170	5.7	0.5	130	20
STONEHILL	10M	9/18/2018	90	5	120	120	5.5	1.3	140	23
STONEHILL	20M	5/22/2018	84	2.5	270	430	24.8	1.1	120	1
STONEHILL	20M	6/20/2018	86	2.5	220	250	7.7	0.5	120	20
STONEHILL	20M	7/17/2018	90	8	200	180	8.8	0.5	120	16
STONEHILL	20M	8/22/2018	84	2.5	180	260	4.5	0.5	140	20
STONEHILL	20M	9/18/2018	88	5	190	190	3.4	1.6	150	22
STONEHILL	3M	5/22/2018	88	2.5	200	400	15.4	2.6	120	1
STONEHILL	3M	6/20/2018	82	2.5	160	160	6.8	0.5	110	19
STONEHILL	3M	7/17/2018	82	7	99	140	8.9	0.5	100	8.1
STONEHILL	3M	8/22/2018	86	2.5	68	160	4.7	0.5	130	22
STONEHILL	3M	9/18/2018	92	7	110	100	4.1	0.5	140	23
TENMILE	10M	5/22/2018	74	2.5	250	420	13	1.6	130	2.1
TENMILE	10M	6/20/2018	84	2.5	180	210	8.2	1.2	110	8.4
TENMILE	10M	7/17/2018	86	10	150	150	7.2	0.5	91	5.9
TENMILE	10M	8/22/2018	90	2.5	120	160	3.5	0.5	120	21
TENMILE	10M	9/18/2018	90	2.5	100	110	2.6	0.5	130	22
TENMILE	20M	5/22/2018	84	2.5	280	460	14.6	3.2	140	1.5
TENMILE	20M	6/20/2018	88	2.5	190	220	6.1	1.1	120	20
TENMILE	20M	7/17/2018	84	12	120	100	7.7	0.5	100	17
TENMILE	20M	8/22/2018	80	36	150	180	2.7	0.5	110	20
TENMILE	20M	9/18/2018	86	2.5	170	170	5.3	0.5	140	21

Station	Depth	Date	Alkalinity (mg/L CaCO3)	Ammonia (ug/L)	Nitrate + Nitrite (ug/L)	Total Nitrogen (ug/L)	Total Phosphorus (ug/L)	O-Phosphate (ug/L)	TDS (mg/L)	DIC (mg/L)
TENMILE	3M	5/22/2018	92	2.5	200	480	19	0.5	130	2.2
TENMILE	3M	6/20/2018	82	2.5	180	250	21.4	1.4	100	19
TENMILE	3M	7/19/2018	82	21	190	160	4.5	1.5	110	11
TENMILE	3M	8/22/2018	92	2.5	89	140	2.8	0.5	130	22
TENMILE	3M	9/18/2018	92	2.5	110	100	6	1.1	130	22
USA/CAN	10M	5/23/2018	72	2.5	220	410	28.8	1.5	110	1.9
USA/CAN	10M	6/19/2018	90	2.5	190	190	7.5	0.5	120	19
USA/CAN	10M	7/16/2018	86	10	130	160	5.1	0.5	99	15
USA/CAN	10M	8/22/2018	96	2.5	120	260	9	0.5	140	22
USA/CAN	10M	9/17/2018	98	8	96	120	3.8	0.5	160	23
USA/CAN	20M	5/23/2018	64	2.5	250	470	36.5	4.7	120	0.5
USA/CAN	20M	6/19/2018	98	2.5	210	220	13	0.5	100	20
USA/CAN	20M	7/16/2018	86	9	140	100	7.6	0.5	120	16
USA/CAN	20M	8/22/2018	86	2.5	210	280	4	1	150	21
USA/CAN	20M	9/17/2018	110	9	220	210	3.2	0.5	180	26
USA/CAN	3M	5/23/2018	68	2.5	180	360	23.8	0.5	84	1.6
USA/CAN	3M	6/19/2018	64	2.5	130	280	9.5	0.5	100	19
USA/CAN	3M	7/16/2018	66	6	74	100	5.2	0.5	110	11
USA/CAN	3M	8/22/2018	92	2.5	91	210	5.8	0.5	110	22
USA/CAN	3M	9/17/2018	96	8	100	120	4.3	0.5	150	18

Table A11. Chlorophyll a data for Libby Reservoir in 2018 collected during the primary productivity study.

Station	Filter (um)	Depth (m)	Date collected	Chla (ug/L)	Phae (ug/L)
Stonehill	0.2	1,3,5,7,10	5/22/2018	1.90	<0.15
Stonehill	2	1,3,5,7,10	5/22/2018	0.68	<0.15
Stonehill	20	1,3,5,7,10	5/22/2018	<0.15	<0.15
Tenmile	0.2	1,3,5,7,10	5/22/2018	2.16	<0.15
Tenmile	2	1,3,5,7,10	5/22/2018	1.78	<0.15
Tenmile	20	1,3,5,7,10	5/22/2018	0.32	<0.15
USA/CAN	0.2	1,3,5,7,10	5/23/2018	1.47	<0.15
USA/CAN	2	1,3,5,7,10	5/23/2018	1.19	<0.15
USA/CAN	20	1,3,5,7,10	5/23/2018	<0.15	<0.15
Kikomun	0.2	1,3,5,7,10	6/19/2018	<0.15	<0.15
Kikomun	2	1,3,5,7,10	6/19/2018	<0.15	<0.15
Kikomun	20	1,3,5,7,10	6/19/2018	<0.15	<0.15
USA/CAN	0.2	1,3,5,7,10	6/19/2018	2.13	0.15
USA/CAN	2	1,3,5,7,10	6/19/2018	1.86	0.47
USA/CAN	20	1,3,5,7,10	6/19/2018	1.42	0.85
Stonehill	0.2	1,3,5,7,10	6/20/2018	1.41	<0.15
Stonehill	2	1,3,5,7,10	6/20/2018	1.87	<0.15
Stonehill	20	1,3,5,7,10	6/20/2018	1.28	<0.15
Tenmile	0.2	1,3,5,7,10	6/20/2018	1.82	<0.15
Tenmile	2	1,3,5,7,10	6/20/2018	1.66	<0.15
Tenmile	20	1,3,5,7,10	6/20/2018	1.08	<0.15
Kikomun	0.2	1,3,5,7,10	7/16/2018	1.22	<0.15
Kikomun	2	1,3,5,7,10	7/16/2018	1.19	<0.15
Kikomun	20	1,3,5,7,10	7/16/2018	0.74	<0.15
USA/CAN	0.2	1,3,5,7,10	7/16/2018	2.75	0.17
USA/CAN	2	1,3,5,7,10	7/16/2018	2.10	0.35
USA/CAN	20	1,3,5,7,10	7/16/2018	1.55	<0.15
Kikomun	0.2	1,3,5,7,10	8/21/2018	1.70	<0.15
Kikomun	2	1,3,5,7,10	8/21/2018	1.35	<0.15
Kikomun	20	1,3,5,7,10	8/21/2018	0.22	<0.15
USA/CAN	0.2	1,3,5,7,10	8/21/2018	2.04	0.19
USA/CAN	2	1,3,5,7,10	8/21/2018	0.99	<0.15
USA/CAN	20	1,3,5,7,10	8/21/2018	0.46	<0.15
Stonehill	0.2	1,3,5,7,10	8/22/2018	1.84	<0.15
Stonehill	2	1,3,5,7,10	8/22/2018	1.91	<0.15
Stonehill	20	1,3,5,7,10	8/22/2018	0.51	<0.15
Tenmile	0.2	1,3,5,7,10	8/22/2018	2.03	<0.15
Tenmile	2	1,3,5,7,10	8/22/2018	1.31	<0.15
Tenmile	20	1,3,5,7,10	8/22/2018	0.36	<0.15
Kikomun	0.2	1,3,5,7,10	9/17/2018	1.34	<0.15
Kikomun	2	1,3,5,7,10	9/17/2018	0.93	<0.15

Station	Filter (um)	Depth (m)	Date collected	Chla (ug/L)	Phae (ug/L)
Kikomun	20	1,3,5,7,10	9/17/2018	0.09	<0.15
USA/CAN	0.2	1,3,5,7,10	9/17/2018	1.75	<0.15
USA/CAN	2	1,3,5,7,10	9/17/2018	1.12	<0.15
USA/CAN	20	1,3,5,7,10	9/17/2018	<0.15	<0.15
Stonehill	0.2	1,3,5,7,10	9/18/2018	1.46	<0.15
Stonehill	2	1,3,5,7,10	9/18/2018	0.56	<0.15
Stonehill	20	1,3,5,7,10	9/18/2018	<0.15	<0.15
Tenmile	0.2	1,3,5,7,10	9/18/2018	1.30	<0.15
Tenmile	2	1,3,5,7,10	9/18/2018	0.59	<0.15
Tenmile	20	1,3,5,7,10	9/18/2018	<0.15	<0.15
Stonehill	0.2	1,3,5,7,10	7/17/2019	2.25	<0.15
Stonehill	2	1,3,5,7,10	7/17/2019	1.80	<0.15
Stonehill	20	1,3,5,7,10	7/17/2019	1.30	<0.15
Tenmile	0.2	1,3,5,7,10	7/17/2019	2.90	<0.15
Tenmile	2	1,3,5,7,10	7/17/2019	1.96	<0.15
Tenmile	20	1,3,5,7,10	7/17/2019	0.90	<0.15

Table A12. PPR data collected in Libby Reservoir in 2018 for the primary productivity study.

Station	Date	Depth (m)	0.2um	2.0um	20um	Station	Date	Depth (m)	0.2um	2.0um	20um
Tenmile	5/22/2018	1	36.12	22.45	2.65	US/Can Border	7/16/2018	5	87.63	67.72	56.17
Tenmile	5/22/2018	3	10.66	4.93	1.58	US/Can Border	7/16/2018	10	44.95	41.34	53.01
Tenmile	5/22/2018	5	1.87	1.72	0.24	US/Can Border	7/16/2018	15	8.51	9.19	9.89
Tenmile	5/22/2018	10	0.29	0.31	0.13	US/Can Border	7/16/2018	20	1.97	4.84	4.56
Tenmile	5/22/2018	15	0.66	0.39	0.01	US/Can Border	7/16/2018	25	3.01	2.71	5.33
Tenmile	5/22/2018	20	0.33	0.14	0.12	Kikomon	7/16/2018	1	36.91	31.80	26.98
Tenmile	5/22/2018	25	0.26	0.13	0.13	Kikomon	7/16/2018	3	43.13	33.79	22.71
Stone Hill	5/22/2018	1	6.70	2.95	0.40	Kikomon	7/16/2018	5	60.05	52.70	38.33
Stone Hill	5/22/2018	3	1.80	1.36	0.17	Kikomon	7/16/2018	10	8.85	8.83	10.18
Stone Hill	5/22/2018	5	0.11	0.27	0.04	Kikomon	7/16/2018	15	1.51	1.47	3.23
Stone Hill	5/22/2018	10	0.02	0.08	0.00	Tenmile	8/22/2018	1	62.80	40.38	18.14
Stone Hill	5/22/2018	15	0.03	0.07	0.01	Tenmile	8/22/2018	3	83.26	54.70	22.37
Stone Hill	5/22/2018	20	0.03	0.06	0.02	Tenmile	8/22/2018	5	84.23	51.07	20.33
Stone Hill	5/22/2018	25	0.04	0.00	0.01	Tenmile	8/22/2018	10	59.94	41.06	20.16
US/Can Border	5/22/2018	1	15.35	0.06	0.62	Tenmile	8/22/2018	15	26.41	18.63	11.28
US/Can Border	5/22/2018	3	0.70	4.52	0.02	Tenmile	8/22/2018	20	12.04	9.02	3.61
US/Can Border	5/22/2018	5	0.09	0.78	0.77	Tenmile	8/22/2018	25	2.11	2.48	1.95
US/Can Border	5/22/2018	10	0.04	0.21	0.09	Stone Hill	8/22/2018	1	70.61	51.89	26.91
US/Can Border	5/22/2018	15	0.04	0.00	0.06	Stone Hill	8/22/2018	3	71.31	53.57	24.36
US/Can Border	5/22/2018	20	0.02	0.16	0.00	Stone Hill	8/22/2018	5	73.36	44.48	23.79
US/Can Border	5/22/2018	25	0.06	0.62	0.08	Stone Hill	8/22/2018	10	48.10	36.08	21.52
Tenmile	6/20/2018	1	139.28	94.72	53.70	Stone Hill	8/22/2018	15	20.06	10.43	5.38

Station	Date	Depth (m)	0.2um	2.0um	20um	Station	Date	Depth (m)	0.2um	2.0um	20um
Tenmile	6/20/2018	3	146.14	115.88	85.92	Stone Hill	8/22/2018	20	7.35	4.78	4.73
Tenmile	6/20/2018	5	98.16	65.71	38.17	Stone Hill	8/22/2018	25	2.69	0.98	1.56
Tenmile	6/20/2018	10	5.42	4.64	3.79	US/Can Border	8/21/2018	1	59.63	50.89	22.89
Tenmile	6/20/2018	15	3.27	2.56	1.78	US/Can Border	8/21/2018	3	97.27	75.93	36.89
Tenmile	6/20/2018	20	4.02	3.00	3.48	US/Can Border	8/21/2018	5	102.47	81.80	36.39
Tenmile	6/20/2018	25	4.10	2.55	4.13	US/Can Border	8/21/2018	10	74.62	55.50	32.75
Stone Hill	6/20/2018	1	128.54	82.33	56.45	US/Can Border	8/21/2018	15	17.24	11.24	8.32
Stone Hill	6/20/2018	3	110.43	99.63	73.85	US/Can Border	8/21/2018	20	6.90	9.50	4.32
Stone Hill	6/20/2018	5	86.75	78.09	59.77	US/Can Border	8/21/2018	25	0.02	1.18	1.53
Stone Hill	6/20/2018	10	11.83	12.31	12.76	Kikomon	8/21/2018	1	47.96	39.69	19.92
Stone Hill	6/20/2018	15	7.05	6.40	2.21	Kikomon	8/21/2018	3	70.86	50.51	29.03
Stone Hill	6/20/2018	20	0.00	1.98	1.13	Kikomon	8/21/2018	5	72.38	66.11	30.25
Stone Hill	6/20/2018	25	2.50	1.81	1.65	Kikomon	8/21/2018	10	25.55	20.18	11.00
US/Can Border	6/19/2018	1	163.79	156.10	185.35	Kikomon	8/21/2018	15	7.96	6.83	4.59
US/Can Border	6/19/2018	3	215.80	175.12	239.79	Tenmile	9/18/2018	1	62.68	32.89	12.48
US/Can Border	6/19/2018	5	138.45	129.97	176.36	Tenmile	9/18/2018	3	85.75	48.90	22.98
US/Can Border	6/19/2018	10	15.89	15.61	20.39	Tenmile	9/18/2018	5	91.87	55.40	21.52
US/Can Border	6/19/2018	15	6.22	6.12	6.47	Tenmile	9/18/2018	10	86.94	54.34	21.06
US/Can Border	6/19/2018	20	1.77	1.22	1.12	Tenmile	9/18/2018	15	48.79	30.39	17.04
US/Can Border	6/19/2018	25	8.74	1.91	1.95	Tenmile	9/18/2018	20	16.57	14.48	0.00
Kikomon	6/19/2018	1	8.13	0.75	0.53	Tenmile	9/18/2018	25	12.01	3.60	0.83
Kikomon	6/19/2018	3	6.17	6.40	0.42	Stone Hill	9/18/2018	1	76.39	50.56	5.91
Kikomon	6/19/2018	5	3.76	5.36	3.81	Stone Hill	9/18/2018	3	103.79	66.79	22.45
Kikomon	6/19/2018	10	0.59	4.07	2.25	Stone Hill	9/18/2018	5	122.39	62.22	23.43

Station	Date	Depth (m)	0.2um	2.0um	20um	Station	Date	Depth (m)	0.2um	2.0um	20um
Tenmile	7/17/2018	1	63.91	45.31	29.46	Stone Hill	9/18/2018	10	76.24	50.63	10.34
Tenmile	7/17/2018	3	77.42	63.42	75.52	Stone Hill	9/18/2018	15	29.12	18.53	6.75
Tenmile	7/17/2018	5	80.75	64.98	59.13	Stone Hill	9/18/2018	20	0.51	3.89	0.95
Tenmile	7/17/2018	10	15.17	12.02	10.60	Stone Hill	9/18/2018	25	8.70	2.12	3.04
Tenmile	7/17/2018	15	3.70	3.87	3.74	US/Can Border	9/17/2018	1	56.39	43.63	13.50
Tenmile	7/17/2018	20	3.74	2.74	8.27	US/Can Border	9/17/2018	3	125.38	73.78	21.65
Tenmile	7/17/2018	25	0.96	1.22	1.50	US/Can Border	9/17/2018	5	140.90	90.52	27.04
Stone Hill	7/17/2018	1	40.82	33.42	32.12	US/Can Border	9/17/2018	10	97.76	57.22	11.16
Stone Hill	7/17/2018	3	28.00	38.22	42.17	US/Can Border	9/17/2018	15	24.67	21.78	5.37
Stone Hill	7/17/2018	5	39.45	41.98	41.71	US/Can Border	9/17/2018	20	5.41	5.72	8.16
Stone Hill	7/17/2018	10	7.41	4.17	7.00	US/Can Border	9/17/2018	25	3.48	4.58	2.78
Stone Hill	7/17/2018	15	2.59	4.18	0.00	Kikomon	9/17/2018	1	50.72	41.84	15.76
Stone Hill	7/17/2018	20	2.31	3.26	2.33	Kikomon	9/17/2018	3	68.99	53.21	11.97
Stone Hill	7/17/2018	25	1.98	1.69	0.00	Kikomon	9/17/2018	5	88.76	59.72	20.04
US/Can Border	7/16/2018	1	46.92	27.60	24.23	Kikomon	9/17/2018	10	57.11	50.81	17.51
US/Can Border	7/16/2018	3	56.93	38.12	29.87	Kikomon	9/17/2018	15	18.45	17.27	20.91

One of the critical pieces of information needed to calculate primary production rates in lakes is the available pool of inorganic carbon for biological uptake. In 2016 this was determined by collecting water samples from multiple depths and then analyzing these samples for Alkalinity. Alkalinity concentrations were then converted to DIC through the use of several regressions that utilized pH and water temperature. The results from these calculations were suspect due to the resultant high DIC concentrations. For the original 2016 report the DIC values used were from values of similar water bodies in the region. In 2017 and 2018 direct measurement of DIC concentrations were determined through laboratory analysis. The data from 2017 and 2018 confirmed the very high DIC concentrations within Koocanusa Reservoir. The following figures and tables are the corrected productivity values from 2016. All multi-year comparisons used in the 2017 and 2018 reports were based off of the corrected productivity measurements of 2016.

Table A13. Revised Productivity values for Koocanusa Reservoir by size fraction for 2016.

Month	Station	Productivity ($mgC/m^2/Day$)			
		0.2 um – 2.0um	2.0 um – 20um	>20um	Total
May 2016	Tenmile	346.31	240.23	0.00	586.55
	Stone Hill	543.07	107.19	89.49	739.76
	US/Can Border	275.63	112.95	144.69	533.28
	Kikomun	0.00	12.82	0.00	12.82
June 2016	Tenmile	240.23	0.00	745.02	985.25
	Stone Hill	107.19	0.00	678.03	785.22
	US/Can Border	112.95	0.00	894.28	1007.24
	Kikomun	12.82	0.00	483.77	496.59
July 2016	Tenmile	0.00	439.71	352.93	792.64
	Stone Hill	89.49	340.96	379.79	810.24
	US/Can Border	144.69	212.21	515.97	872.87
	Kikomun	N/A	N/A	N/A	N/A
August 2016	Tenmile	371.91	137.21	390.90	900.02
	Stone Hill	294.49	175.10	383.37	852.96
	US/Can Border	0.00	267.71	137.82	405.54
	Kikomun	0.00	234.82	227.91	462.73
September 2016	Tenmile	144.55	211.92	267.59	624.06
	Stone Hill	439.97	39.38	248.16	727.52
	US/Can Border	184.32	387.09	370.00	941.41
	Kikomun	0.00	0.00	388.90	388.90

Figure A1 Productivity by month, and size class for 2016.