# FLATHEAD LAKE AND RIVER SYSTEM FISHERIES STATUS REPORT

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# ACKNOWLEDGMENTS

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The framework for the long-term tributary monitoring program developed during the EPA-funded Flathead River Basin Study which ran from 1978 through 1983. Flathead National Forest provided funding beginning in 1982 allowing continuation of standardized data collection at a core group of index sites up through the present time. Additional funding from Bonneville Power Administration (1986-ongoing) and Montana Department of Natural Resources and Conservation (1992-ongoing) has enabled us to expand our data collections basin-wide. These activities now provide one of the longest running data sets on a large lake and river system and, specifically to bull trout and their habitat, one of the most complete available anywhere. These data are now included as an integral component of the Flathead Basin Commission’s Master Monitoring Program to track overall water quality and aquatic health basin-wide.

# INTRODUCTION

Fisheries management plans incorporate biological and social issues to create an acceptable and realistic approach to resource conservation. The following report compiles available biological fisheries information for the Flathead Lake and River system. It will provide the public and decision makers with the best available science needed to discuss management issues.

This report contains recent research and long-term monitoring results of fisheries field surveys. Much of the data have not been reported in the last decade. This report consolidates summaries from various surveys on Flathead Lake, the Flathead River, and tributaries in an effort to describe changes in and present status of fish populations and habitat quality.

The report follows a standard format, beginning with a background section containing a study area description and a discussion of changes in the lake foodweb and aquatic community that have occurred in response to introductions of exotic fish species and the establishment of *Mysis relicta* (*Mysis*). Following this section, there are 20 sections which present summaries of recent research and monitoring results. Each of these sections contain separate introductions, methods, and results and discussions to allow each to be considered separately from the main body of the report. These individual studies are separated into four groups, work conducted on Flathead Lake, Hungry Horse Reservoir, the Flathead River (main stem, North, Middle, and South forks), and tributary streams to the North, Middle, and South forks.

This report emphasizes how important the inter-connected lake, river, and tributary system is to fisheries of the Flathead drainage, especially to native fish species. Our monitoring strategies and conclusions reflect the comprehensive approach needed to evaluate this system. The monitoring strategy is not new. It was initiated in 1978 to collect baseline biological resource information for the Flathead River Basin Environmental Impact Study (Graham et al. 1980, Shepard and Graham 1983). Montana Fish, Wildlife & Parks (MFWP) has successfully conducted some of these monitoring activities annually or at least intermittently throughout the last two decades. Other monitoring activities have been reinstigated only in recent years.

Fieldwork conducted within the last two decades encompasses the time period in which *Mysis* entered the Flathead Lake and River system and radically changed foodweb interactions. Surveys spanning the late 1970s and into the mid-1980s characterize the pre-*Mysis* conditions. More recent surveys (mid-1980s to present) portray resulting changes to and status of the fish community following *Mysis* establishment.

Montana Fish, Wildlife & Parks is not alone in monitoring the aquatic resources of Flathead Lake. The Confederated Salish and Kootenai Tribes (CSKT) co-manage the fisheries of Flathead Lake and conduct monitoring and research studies on Flathead Lake, some of which are included in this report. Since the early 1990s, MFWP and CSKT have conducted research activities, habitat enhancements, and experimental fish stocking through mitigation programs associated with Hungry Horse and Kerr dams. The U.S. Fish and Wildlife Service contributed to fish stocking efforts. Programs have been funded by Bonneville Power Administration. In addition, the University of Montana, through the Flathead Lake Biological Station, has conducted numerous surveys of water quality parameters and described characteristics of lower trophic levels.

Recent monitoring efforts are combined and summarized in this report in order to comprehensively describe the known characteristics, changes, and trends in the status of fisheries resources in the Flathead Lake and River system. It has been roughly 15 years since *Mysis* became established in Flathead Lake, but the resulting changes to the aquatic community are still incomplete. It appears that *Mysis* will persist and the densities of large zooplankton will remain much lower than their levels prior to *Mysis* establishment. Remaining questions include: What will be the resulting composition of the fish community?; Will the native bull trout (*Salvelinus confluentus*) and westslope cutthroat trout (*Oncorhynchus clarki lewisi*) persist?, and; What will be the future recreational fisheries? In 1998, the U.S. Fish and Wildlife Service listed the bull trout as threatened under the Endangered Species Act and the westslope cutthroat trout has been petitioned for listing. Due to the large size of the Flathead Lake drainage, Flathead Lake native fish populations have historically been important to the overall status and persistence of these species in Montana. MFWP has monitored bull trout spawner escapement in the Flathead drainage for 20 years. In addition to this database, stream electrofishing, stream substrate assessments, and lake gill-netting surveys track current and changing trends in status of fish populations and habitat quality. Future surveys will provide the information needed to formulate viable management alternatives to preserve these important native fish species. CSKT and MFWP maintain responsibility for fisheries management, and over the next two years, will combine biological information with social concerns and public opinion to help define the direction of future fisheries management in Flathead Lake.

## BACKGROUND

### Description of Study Area

The Flathead Lake and River system located in northwest Montana consists of Flathead Lake, the main stem Flathead River above Kerr Dam, and major tributaries including the Swan River, Whitefish River, and Stillwater River drainages, and the North, Middle, and South forks of the Flathead River and their major tributaries. The Flathead Basin drains an area of roughly 18,400 $km^2$, which is underlain by nutrient-poor Precambrian sedimentary rock. The drainage is known for its high water quality (Zackheim 1983). The system is managed as one ecosystem due to the migratory nature and complex life-histories of many species in the system. Adfluvial fish interact with lake and river stocks, emphasizing the interdependency and connectivity of the lake and river fisheries.

Flathead Lake is oligomesotrophic with a surface area of roughly 510 $km^2$ (125,250 acres), a mean depth of 50.2 m, and a maximum depth of 113.0 m (Zackheim 1983). The southern half of the lake lies within the Flathead Indian Reservation. Kerr Dam was built in 1938 and is located on the southern end of Flathead Lake, seven km downstream of the natural lake outlet. Kerr Dam regulates the top three meters of water and is operated to provide flood control and power production. Presently, flood control and recreation require the lake level to be dropped to the low pool elevation 879.3 m above sea level (2,883 feet) by April 15, refilled to 881.5 m (2,890 feet) by May 30, raised to full pool elevation of 882.4 m (2,893 feet) by June 15, and held at full pool through Labor Day.

Two major tributaries to Flathead Lake are the Swan and Flathead rivers. The Swan River drains the Swan Valley and Swan Lake. Fish movement upstream from Flathead Lake into the Swan River is blocked by Bigfork Dam, located less than two kilometers above Flathead Lake. The dam was built in 1902 for electrical power production. The three forks of the Flathead River supply roughly 80 percent of the annual discharge (9 million acre-feet) in the Flathead system (Zackheim 1983). The North Fork flows out of British Columbia, defines the western border of Glacier National Park (GNP), and primarily drains forested lands of GNP, the Flathead National Forest, and other managed forest lands. The Middle Fork flows out of the Great Bear Wilderness Area, defines the southern boundary of GNP and drains forested lands of GNP and the Flathead National Forest. The South Fork flows for over 95 km in the Bob Marshall Wilderness Area before impoundment in Hungry Horse Reservoir (56 km in length) located in the Flathead National Forest. Hungry Horse dam was completed in 1953, located 8.5 km upstream from the confluence of the South Fork and the main stem of the Flathead River. Hungry Horse Dam blocks upstream fish migrations and effectively isolates the South Fork drainage from fish of Flathead Lake. Hungry Horse Dam provides flood control, electrical power production, and water storage capability for the Columbia River system.

The major sport fish species in Flathead Lake include westslope cutthroat trout, bull trout, lake trout (*S. namaycush*), lake whitefish (*Coregonus clupeaformis*), and yellow perch (*Perca flavenscens*). The major sportfish in the river are westslope cutthroat trout, bull trout, rainbow trout (*O. mykiss*), mountain whitefish (*Prosopium williamsoni*). Scattered populations of largemouth bass (*Micropterus salmoides*), yellow perch, and northern pike (*Esox lucius*) occur in and old oxbows of the river. Other native fish in the Flathead system include longnose sucker (*Catostomus catostomus*), largescale sucker (*C. macrocheilus*), northern pikeminnow (*Ptychocheilus oregonensis*), peamouth (*Mylocheilus caurinus*), pygmy whitefish (*P. coulteri*), and reside shiner (*Richardsonius balteatus*) (Table 1).

The native trout and char, westslope cutthroat trout and bull trout, have evolved varied life histories to be successful in the Flathead drainage. There are three life history forms: (1) adfluvial stocks which spawn and rear in river tributaries and move downstream to mature and reside in Flathead Lake; (2) fluvial stocks which spawn and rear in river tributaries then move downstream to mature and reside in the Flathead River, and; (3) tributary or “resident” stocks which spawn, rear, and reside for their entire life cycle in a tributary stream (Shepard et al. 1984, Fraley and Shepard 1989, Liknes and Graham 1988). Westslope cutthroat trout employ all three of these strategies in the Flathead system, although it appears bull trout are primarily adfluvial.

Individual fish may combine the first two strategies. Juveniles reside in tributaries for 1-3 years before migrating downstream into river or lake habitats (Shepard et al. 1984). Adfluvial fish take advantage of improved forage and growth rates during lake residence and thus reach larger sizes than either fluvial or tributary residents. Tributary fish mature at relatively smaller sizes (≈200 mm) and don't grow as large (>400 mm) as fish using the other strategies (Shepard et al. 1984, Liknes and Graham 1988).

These three life history forms inhabit three general types of habitat: tributary streams, main stem river and forks, and lake. In order for fish populations in the basin to be successful, all habitats must present adequate conditions for fish survival at related life history stages. Degraded conditions in one of these habitat types may limit the population, stressing the importance of habitat quality and connectivity within the lake-river-tributary system.

## The Changing Fish Community of Flathead Lake

From a fish community perspective, Flathead Lake has supported three very different species assemblages. Prior to settlement by European man, the fish community was solely comprised of the native species which colonized the waters following the last glacial period, roughly 10,000 years ago. Bull trout, westslope cutthroat trout, and mountain and pygmy whitefish were the only salmonids. Bull trout and northern pikeminnow were the dominant piscivorous fishes. Most likely, the minnows (n. pikeminnow and peamouth) dominated in fish abundance and biomass (Elrod 1929). Accurate depiction of relative species abundance is difficult due to lack of recorded and quantified surveys or fishery encounters.

In the mid 1880s, Europeans arrived and beginning in the early 1900s, introduced a number of other fish species (Table 1)(Hanzel 1969, Alvord 1991). Federal and state government agencies aggressively introduced gamefish, both native and exotic species, into Montana waters (Alvord 1991). They constructed fish hatcheries and developed fish transport systems incorporating railroads. In addition to fish introductions, managers tried other means to modify the fish community. For example, in 1913, a few thousand pounds of bull trout were reportedly seined from Flathead Lake during a period of legalized netting. This was an effort to reduce predation on more desireable fish species. Following this large harvest, bull trout were restored to the gamefish category making them illegal to harvest by nets (Alvord 1991). By the 1920s, a new fish community was established with abundant kokanee, lake trout, lake whitefish, and yellow perch in addition to the native species. Kokanee and yellow perch dominated the recreational fishery. By the early 1930s, anglers were annually harvesting an estimated 100 tons of kokanee from Flathead Lake (Alvord 1991). Angler creel surveys in 1962, 1981, and 1985 show kokanee provided the majority of the sport fishery, from 77 to 97 percent of harvested fish numbers (Evarts 1998). This new fishery composition was relatively stable until the mid 1980s.

Table 1. List of native and non-native fish species currently found in Flathead Lake, and the dates non-native fish were introduced (Hanzel 1969, Alvord 1991).

| Native | Non-Native | Date Introduced |
|---|---|---|
| Bull Trout | Lake Trout | 1905 |
| Westslope Cutthroat Trout | Lake Whitefish | 1890 |
| Mountain Whitefish | Kokanee | 1916 |
| Pygmy Whitefish | Yellow Perch | 1910 |
| Longnose Sucker | Northern Pike | 1960's (Illegally) |
| Largescale Sucker | Rainbow | 1914 |
| Northern Pikeminnow | Brook Trout | 1913 |
| Peamouth Chub | Largemouth Bass | 1898 |
| Redside Shiner | Pumpkinseed Sunfish | 1910 |
| Sculpins | Black Bullhead | 1910 |

In the 1960s, fisheries management agencies across the western United Sates and Canada introduced the opossum shrimp, *Mysis relicta* into hundreds of lakes where they did not naturally occur. The impetus for this action was apparent increased growth rates for kokanee salmon following the establishment of *Mysis* in Kootenay Lake, B.C. In 1968, 1975, and 1976 MFWP introduced *Mysis* into four lakes (Ashley, Swan, Tally, and Whitefish) in the Flathead Lake drainage. Although no *Mysis* were stocked directly into Flathead Lake, *Mysis* moved out of these lakes and downstream into Flathead Lake where they were first collected in 1981. By the mid-1980s, *Mysis* established an abundant population and caused the third shift in the fish assemblage in Flathead Lake.

Following their first collection in Flathead Lake in 1981, the *Mysis* population increased exponentially from under three *Mysis*/m$^2$ in 1984 to a peak of 130 *Mysis*/m$^2$ in 1986 (Beattie and Clancey 1991, Spencer et al. 1991). *Mysis* density then dropped below 60/m$^2$ by 1988 and has since varied between 16 and 68/m$^2$ (Spencer et al. 1991, Beattie and Clancey 1991, Flathead Basin Commission 1993, Stanford et al. 1997). A similar temporal pattern of *Mysis* densities, peaking and then declining to a lower level, has been observed in other lakes and reservoirs throughout the western United States (Nesler and Bergersen 1991).

*Mysis* created unforeseen and far-reaching changes to the Flathead Lake system due to their unique feeding behavior. *Mysis* avoid light. During the day they primarily rest on the lake bottom in water over 100 feet deep. After dark they move up into the water column and feed, again descending by first light, at which time pelagic species such as kokanee begin to feed. *Mysis* eat larger zooplankton, the same forage preferred by fish species including kokanee, and are able to severely deplete zooplankton populations (Morgan et al. 1978, Rieman and Bowler 1980, Bowles et al. 1991, Martinez and Bergersen 1991). Thus, *Mysis* become a competitor with fish species dependent on the zooplankton forage base and not forage as managers desired. *Mysis* did provide an abundant food source for benthic fishes, such as lake trout and lake whitefish, and substantially increased survival, recruitment, and abundance of these species.

The introduction and establishment of *Mysis* has considerably altered the zooplankton community in Flathead Lake. Principally, there has been a dramatic decrease in the abundance of larger zooplankton, cladocerans, and copepods. The larger zooplanktors, *Daphnia thorata*, *Epischura nevadensis*, *Leptodora kindtii*, were the principle food for kokanee and were seasonally important to other fish species including westslope cutthroat trout. Before *Mysis*, D. *thorata* comprised 72 percent of the total food biomass eaten by older kokanee, age 3+ and older (Leathe and Graham 1982). When *Mysis* densisities peaked, cladoceran densities severely declined. Two of four principle cladocerans, *D. longiremis* and *L. Kindtii*, disappeared from lake samples, while the other two, *D. thorata* and *Bosmina longirostris*, persisted but at greatly reduced densities (Spencer et al. 1991). Mean annual abundances for cladocerans dropped from 2.8 to 0.35 organisms per liter following *Mysis* establishment (Spencer et al. 1991, Beattie and Clancey 1991). Similarly, copepods significantly declined (Beattie and Clancey 1991). In years following the decline from peak *Mysis* densities, *D. longiremis* and *L. kindtii* have reappeared in samples but at very low levels (Spencer et al. 1991). Presently, the zooplankton community has stablized with a shift from dominance by large cladocerans to small cladocerans, copepods, and rotifers (Stanford et al. 1997).

Not only has the abundance of larger zooplanktors declined, but the summer blooms or peaks in abundance are reduced and delayed, by roughly one month. In 1986 and 1987, as *Mysis* densities peaked, the spring population bloom of *D. thorata* was delayed from June into July and the maximum summer abundance was less than one-third of 1980-1982 levels (Beattie and Clancey 1991). The bloom appears to be delayed until the lake surface waters thermally stratify, possibly providing zooplankton some thermal refuge from *Mysis* predation, since *Mysis* tend to avoid warmer water temperatures.

The declines and delays in zooplankton abundance in Flathead Lake have been attributed to grazing pressure of *Mysis* (Beattie and Clancey 1991, Spencer et al. 1991, Stanford et al. 1997). Similar declines in cladoceran abundance are well documented in numerous lakes in the western United States and Canada (Morgan et al. 1978, Reiman and Falter 1981, Lasenby et al. 1986, Bowles et al. 1991, Martinez and Bergersen 1991). Declines in large zooplankton appear to be persistent and represent an interspecific competive element important when comparing conditions and species composition in Flathead Lake prior to and following *Mysis* establishment.

It has been 12 years since *Mysis* densities peaked in Flathead Lake and the fish community has changed. In the following sections, we compare sampling results of the 1980s with those of recent surveys, we evaluate these changes and assess the current status of fish populations.

# FLATHEAD LAKE MONITORING SURVEYS

## ANNUAL SPRING GILL-NET MONITORING SURVEYS

### Introduction

The Confederated Salish & Kootenai Tribe (CSKT) and Montana Fish, Wildlife & Parks (MFWP) annually conduct a relative fish abundance survey in Flathead Lake. This survey allows managers to track changes and trends in fish populations over the long term. Nets fish designated areas and depths to provide comparable trend data between years (Shepard and Graham 1983).

In the late 1970s, concerns of potential adverse changes to the Flathead River drainage associated with coal mining, timber harvest, and other human development established the need for a series of studies to acquire baseline fisheries information. These data are used to assess changes in resource condition (Leathe and Graham 1982). A portion of this effort was focused on Flathead Lake, including seasonal gill-net surveys. From 1980 through 1983, MFWP conducted netting surveys in each of the four seasons. Following this collection period, investigators created a protocol for a standardized spring monitoring program to assess relative fish abundance in five areas of Flathead Lake (Shepard and Graham 1983). In 1981 and 1983, this spring survey was completed and provides a baseline of fisheries information prior to establishment of *Mysis relicta* (*Mysis*). Unfortunately, the spring monitoring program was discontinued until the early 1990s. From 1990 through 1995, MFWP and CSKT conducted only partial sinking net surveys and did not complete the standard monitoring protocol until 1996. However, for the floating net portion of the series, MFWP and CSKT have completed the lake-wide surveys since 1992 (only 1990 and 1991 surveys were incomplete). Complete surveys in 1996, 1997, and 1998 represent the current status and allow valid comparison with 1981 and 1983 surveys.

### Methods

Agency personnel followed methodology established by previous investigators in the early 1980s (Shepard and Graham 1983). Netting occurred in spring (late April/early May) before spring runoff when the lake temperatures were isothermal. Gillnetting was completed in five areas of the lake (Figure 1). In each area we fished three sets of floating nets and three sets of sinking nets. At sampling sites, we set both sinking and floating multi-strand nylon gill nets, 38.1 m long by 1.8 m deep, consisting of five panels of bar mesh sizes, 19, 25, 32, 38, and 51 mm. Each set consisted of two ganged nets, one sinking net tied end to end to another sinking net, and likewise for floating nets. We set nets perpendicular to the shoreline. Floaters were set with one end close to shore in roughly 2 meters of water, stretching the net out over deeper water. Sinking nets were set at depths greater than 10 meters. Previous years’ netting records were consulted to determine depths fished in each area. We fished sets overnight by setting nets in late afternoon and retrieving nets in mid-morning hours.

To calculate catch-per-unit-effort (CPUE), we recorded the number of each species captured in each sinking or floating set and divided by two, in order to report catch per single standard net type. Sinking and floating net catches were reported separately. Percent composition of catch by species was also reported separately by net type. We enumerated, measured total length and weight, and collected age, growth, sexual maturity, and food habits data from captured fish.

## Results And Discussion

From 1996 through 1998, we successfully fished all five areas of the lake, for a total of 30 sinking nets and 30 floating nets per year. Catch in sinking nets best describes fish species with benthic orientation, such as lake trout and bull trout, suckers, and whitefish. Catch in floating nets best describes the changes in westslope cutthroat trout and minnow populations, species that are more surface or shallow water oriented.

Sinking gill net catch was similar in 1996, 1997, and 1998. Sinking nets caught seven fish species for a total of 286, 524, and 633 fish in 1996, 1997, and 1998 respectively. Lake whitefish dominated percent composition, ranging from 74.7 to 74.9 percent of the total number of captured fish (Table 2). Lake trout and northern pikeminnow made up the majority of remaining catch. Bull trout comprised less than one percent of catch.

Total combined catch of all species in floating nets has varied widely in the last three years, while the number of species caught remained more consistent. Floating nets captured nine fish species for a total of 134 fish in 1997 and 608 fish in 1998. In 1996, they caught seven species for a total of 41 fish. In the 1997 and 1998 floating nets, northern pikeminnow (37.3 and 37.7 percent) and peamouth (23.9 and 46.7 percent) dominated the catch composition, followed by westslope cutthroat trout, bull trout, largescale sucker, and lake whitefish (4.1 percent) (Table 2). Similarly, in the 1996 floating nets, peamouth (26.8 percent) and northern pikeminnow (19.5 percent) dominated catch, followed by westslope cutthroat trout, bull trout, and lake trout. Kokanee abundance is not adequately portrayed in our netting series, due to the pelagic nature of kokanee and littoral distribution of our nets. We have other indices which accurately show the abundance trends of kokanee during the sampling period (Beattie and Clancey 1987). However, kokanee have essentially disappeared from our catch in recent years. This is not surprising since the population crashed in the late 1980s.

Percent species composition of our catch has changed dramatically since *Mysis* became established in the lake. *Mysis* densities began to increase in 1985 and peaked in 1986. For gill-net surveys, sample years 1981 and 1983 describe the pre-*Mysis* fish community and provide baseline fishery information for comparison to current populations. In the sinking nets, there was a shift in species composition from numerical dominance by peamouth (pre-*Mysis*) to lake whitefish (post-*Mysis*) (Table 2, Figure 2). From 1996 through 1998, the catch composition has been relatively stable (Table 2). In 1981 and 1983, peamouth comprised 41.1 and 39 percent of catch composition, while lake whitefish comprised only 16.2 and 13.7 percent, respectively. In

recent catches, lake whitefish comprised roughly 75 percent of the catch.
One of the more dramatic transformations was the relative abundance of bull trout and lake trout. In 1981 and 1983, bull trout numbers comprised 10 and 13 percent of fish caught in sinking nets, while lake trout numbers comprised only 0.2 and 0.9 percent, respectively. Since 1996, bull trout comprised roughly 1 percent, while lake trout comprised 6 to 14 percent of gill-net catch.

We have observed similar declines in mountain whitefish in sinking net catch (Table 2). Mountain whitefish comprised roughly four percent of catch composition in the early 1980s and now have a very low incidence (<1 percent).

Species composition of the floating net catch has not varied as widely as that of the sinking net catch. Westslope cutthroat trout showed the greatest declines. In the early 1980s, westslope cutthroat trout made up 20 to 40 percent of catch while in recent years less than 20 percent. With the exception of lake trout and northern pikeminnow, the other species have not shown obvious changes in percent composition. Declines in peamouth relative abundance observed in sinking net catch were not evident in floating nets. Peamouth values remained strong and steady comprising a large percentage of catch, ranging from 24 to 47 percent in recent years (Table 2). The apparent discrepancy between sinking and floating net catch may be explained by the difference between lake whitefish catch in sinking versus floating nets. We did not see an increase in lake whitefish catch in the floating nets as we did in the sinking net catch, most likely due to lake whitefish behavior and benthic nature. Northern pikeminnow, another native minnow, has also comprised a large percentage of floating net catch. The 1997 and 1998 percentages (37 percent) were greater than those of the early 1980s, 12 and 15 percent (Table 2, Figure 3). In recent years, peamouth and northern pikeminnow dominated catch composition in floating nets. Lake trout increased representation in floating net catch. In the early 1980s surveys, lake trout were not captured in floating nets, whereas, in recent years they have comprised 2 to 12 percent of species composition (Table 2).

We observed similar changes in catch-per-unit-effort (CPUE) for individual fish species in the spring gill-net survey as we observed in the percent species composition. Time series of CPUE showed the same general trends (Table 3). In sinking net sets, bull trout and lake trout showed opposite trends. The number of bull trout has dropped from 2.6 and 1.6 fish per net in 1981 and 1983 to 0.1, 0.2, and 0.1 in 1996, 1997, and 1998, respectively. Conversely, lake trout catch has increased from 0.0 and 0.1 fish per net in 1981 and 1983 to 1.3, 1.7, and 1.3 fish per net in 1996, 1997, and 1998 respectively. Lake whitefish catch has also increased. Lake whitefish catch increased from 3.2 and 2.1 fish per sinking net in 1981 and 1983 to 7.1, 12.3, and 15.8 fish per net in 1996, 1997, and 1998 respectively. Peamouth CPUE was much lower in the mid 1990s than in the early 1980s, while northern pikeminnow CPUE appears unchanged (Table 3). In 1998, the floating sets have the highest CPUE for peamouth and northern pikeminnow observed in the study period while the 1998 CPUEs for other species was similar to 1996 and 1997 values. Floating net catch best depicts changes in westslope cutthroat trout abundance. A decreasing trend similar to bull trout has been evident. In the early 1980s, catch of cutthroat trout was two to three fish per net. In the late 1990s catch has dropped to less than one fish per net.

In an effort to summarize and compare CPUE between pre- and post-*Mysis* establishment, we calculated means for the number of fish per net, combining 1981 and 1983 for pre-*Mysis* values and 1996 through 1998 for post-*Mysis* values (Figure 4). There has been over a ten-fold increase in lake trout CPUE, conversely there has been a ten-fold decrease in bull trout CPUE. Lake whitefish CPUE has increased, while westslope cutthroat trout CPUE has decreased.

Until recent years, the sampling protocol established in the early 1980s was not adhered to and gillnetting surveys were either not conducted or incomplete. For example, spring lake wide gill-net surveys were not conducted from 1984 through 1989. Lake wide spring gillnetting with floating nets has been conducted since 1992. From 1990 to 1994, spring netting with sinking nets using established protocol was only repeated at the northern sampling sites. Therefore, the lake wide sinking series conducted since 1995 are most comparable to the early 1980s. Caution should be applied when reviewing species composition and catch per net values from sinking nets for 1990 through 1994 and in comparing these values with results from earlier surveys. In an effort to reduce bias associated with incomplete surveys and still use 1990 to 1994 data from sinking gill-net surveys, we removed catch from southern areas in all complete surveys and then compared netting in only the northern areas over the sample years. This removed 40 percent of sets, reducing the sample size of sinking nets fished from 30 (15 ganged sets) to 18 (9 ganged sets) per year. Table 4 and Figures 5 and 6 portray the percent species composition and catch per net in the northern nets only for bull trout, lake trout, and lake whitefish. For both indices, trends were similar to those observed in these indices when all netted sites were included; lake whitefish dominated recent catch, and bull trout CPUE and percent composition declined, while lake trout CPUE and percent composition increased. Lake trout have replaced bull trout as the dominant salmonid piscivore in Flathead Lake.

As described previously, the bull trout catch was lower in recent surveys than in surveys conducted in the early 1980s. In addition, the length frequency of bull trout catch also changed. Nets caught bull trout in a wide range of length groups (Figures 7, 8, and 9). In 1996, lengths (n=9) ranged from 207 to 724 mm in total length. In 1997, total lengths of captured bull trout (n=18) ranged from 244 to 584 mm. In 1998, we caught 17 bull trout, ranging from 258 to 745 mm in total length. Although the smaller size groups of fish were fairly well represented, there were missing size groups, most prominently the subadult 4+ and 5+ year olds (375 to 475 mm) and adult fish in the largest sizes (>600 mm). In 1996, we did not catch fish in the lengths ranging from 376 to 700 mm. In 1997, the length groups 376 to 475 mm are not well represented. In 1998, a gap appeared between 451 to 525 mm, while capturing fish in the 376 to 450 range (Figure 9). Catch in spring, 1981 was not only greater in number, but these size groups were well represented, especially the 375 to 475 m range (Leathe and Graham 1982). If these fish emigrated as two and three year olds to Flathead Lake, as did most juvenile bull trout in Flathead River tributaries (Shepard et al. 1984), then they most likely resided in the lake for one to two years prior to capture.

In 1996 and 1997, we caught few bull trout suspected of residing two years or more in the lake.

In 1998, there was an increase in catch of age 3 and 4 year old fish. Although the catch of smaller fish is encouraging for future persistence of bull trout, the low numbers continue to raise concern. In recent surveys, we caught more bull trout in floating nets than in sinking nets. This was not the case for Leathe and Graham (1982) who found the opposite. Although gill net mesh size biased catch for specific sizes of fish, the selectivity was consistent among years, since the same equipment was employed.

With the exception of smaller sample sizes of cutthroat trout captured in 1996, 1997, and 1998, recent size range and length frequencies (Figure 10) were similar to those of 1981 (Leathe and Graham 1982). In 1996, there were missing length groups in the catch. In 1997, there were fewer gaps and in 1998 there were no gaps. Recent size ranges were wider than 1981, ranging 177 to 437 mm, 220 to 458 mm, and 242 to 422 mm in 1996, 1997, and 1998 respectively. In spring 1981, investigators caught 99 cutthroat trout but did not catch trout in lengths less than 220 mm or over 400 mm (Leathe and Graham 1982). Most of the catch was between 225 and 350 mm (mostly subadult fish). The spring netting surveys occurred when many adult cutthroat trout were migrating up the Flathead River toward spawning tributaries and thus fewer were vulnerable to capture. However, since the timing of surveys was consistent between 1980s and recent years, this does not explain the difference in the range of lengths.

Lake trout length frequency histograms have also changed since the early 1980s and during the 1990s. In the 1990s, lake trout lengths range from less than 300 to over 900 mm. The length groups with the highest incidence were generally between 376 and 600 mm (Figures 11 through 18). Since 1996, we have caught few fish less than 376 mm in total length. In the early 1990s this was not the case, even though fewer nets were fished. Figures 15 through 18 depict 1991 through 1994 length frequencies for lake trout captured in nets set only in the north half of Flathead Lake. These charts show a higher incidence of small lake trout (<376 mm) than observed in more recent surveys. Thus, there appeared to be more small lake trout in the early 1990s than in recent years. In 1981, sample size was too small in the spring surveys to create a length frequency chart.

Length frequency charts for lake whitefish showed two peaks in 1996 and 1997 (Figures 19 and 20). In 1996, the first peak centered on the 226 to 275 mm length groups and the second on the 401 to 475 mm groups. In 1997, the first peak was wider than the 1996 peak encompassing the 226 to 350 mm groups, while the second was similar to the 1996 peak centering on the 401 to 475 mm groups. With the exception of one larger fish (586 mm) captured in 1997, the range of sizes were similar between 1996 and 1997. The length frequency chart for lake whitefish caught in 1998 depicts a similar size range to the two previous years. However, in 1998 the two distinct modes are missing (Figure 21). In 1998, we caught numerous fish with lengths in the 300 to 400 mm size groups. The 1981 length frequency distribution for spring captured lake whitefish did not show the two distinct peaks. There was a wide peak which encompassed size groups from 340 to 440 mm. Another observed difference between the 1980s and 1990s was the number of small fish captured. In the 1980s, few fish were captured in lengths less than 260 mm (Leathe and Graham 1982). In the 1990s, small fish made up a large proportion of the catch.

## LAKE TROUT OTOLITH ANALYSIS

### Introduction and Methods

This study was initiated to determine the general growth rates and age structure of lake trout in Flathead Lake. Because age determination from scales is difficult in long-lived fish species, we collected otoliths. Otoliths are small bones found in the head of fish that are associated with the auditory system. Our sampling spanned the period from 1986 to 1994. We collected otoliths from lake trout captured in gill-netting surveys. We mailed 143 lake trout otoliths from fish in a wide size range to EFS Consultants (Dr. Edward Brothers, 3 Sunset West, Ithaca, NY 14850). For each sample, Dr. Brothers estimated age and measured annual growth increments. We determined fish length at annuli by modifying the Lee back-calculation procedure (Carlander 1981) using a biological intercept of a fish-otolith trajectory (Campana 1990). We used 16 mm as the fish length corresponding to the initiation of otolith development (Balon 1980). We combined fish aged at five years old and less to estimate the mean total length at annuli I, II, and III. All samples were combined to determine a grand mean length for the remaining annuli. Samples were also partitioned by sex to determine a mean length at annuli. We fitted a Von Bertalanffy growth curve ($L\infty$=903 mm, K=0.119234 and $t_o$=-1.055129) to the back-calculated lengths and ages for all fish less than 21 years old.

### Results And Discussion

Unfortunately, the first two or three annuli were very indistinct and the microstructure (daily increments) of these otoliths did not help in interpreting growth in the first years (personal communication, Dr. Brothers). Dr. Brothers felt that interpretation of ages should be validated by another method and that ages and growth estimates may change slightly with further examination. This analysis will be adjusted and fine tuned with further study, but at this time provides a starting point for estimating lake trout age and growth. Presently, we are collecting otoliths to refine these data. However, conclusive determination of early growth rates is not possible.

Lake trout are relatively slow growing and long-lived fish (Figure 22). Of the 143 samples, the oldest was 38 years old and 865 mm (34 inches) in length. Other fish were younger, yet reached lengths over 956 mm (38 inches). Males and females had similar growth rates, reached lengths over 914 mm (36 inches) and lived to be greater than 30 years old. Fish grew more rapidly in the first 10 years of life, into the 600+ mm length categories. It appears that growth slowed after fish reached sexual maturity. Table 5 shows the mean back-calculated lengths at annuli formation. On average, fish entered the lower boundary of the slot limit (762 mm or 30 inches) at 16 years old but this ranged from 9 to 20 years old (Figure 22). It appeared that some individuals may not grow larger than the upper boundary of the slot limit (914 mm or 36 inches).

The Von Bertalanffy curve did not appear to fit the data very well (Figure 22). We used the mean back-calculated lengths at annuli for all lake trout under 21 years of age. The majority (n=131) of the sample was used. The fish aged older than 20 years were not the longest fish in the sample. Our sample was biased in that the oldest fish were not well represented due to small sample size or possibly that faster-growing fish were not captured or inadvertently excluded from the analysis. The theoretical maximum length ($L\infty$) based on these data was 903 mm (35.5 inches), which is smaller than fish in our sample and a large component of the population. Payne et al. (1990) describe a valid method for improving the fit of a Von Bertalanffy equation by constraining the model and setting the values of $L\infty$ and the theoretical age at zero length ($t_o$). The $L\infty$ value is based on the sizes of observed large fish and $t_o$ is set to zero. This was done to reduce error in estimating $L\infty$ and bias in youngest age class due to sampling selectivity. The constraints allowed investigators to improve estimates for parameters to be used in other models from data believed to be biased. They called the new value $L\infty'$ and estimated a $K'$ (growth coefficient) using the constrained equation and the age classes of four to seven years. In effort to compensate for small sample size, we chose 1118 mm (44 inches) to be $L\infty'$. The largest lake trout observed in Flathead Lake in recent years (1992) was a 1121 mm (44 _ inch) fish, which tied the lake record of 42 lbs. Following their methodology, we estimated $K'$ to be 0.100 for lake trout from Flathead Lake with $L\infty'=111.8$ and $t_o=0$. Because we based age determination on otolith analysis and to increase our sample size, we recalculated values using the age classes 4 to 11, which produced $K'=0.092$ with $L\infty'=111.8$ and $t_o=0$. The line using the values $K'=0.092$, $L\infty'=111.8$ and $t_o=0$ appears to most accurately describe the data and would thus be the most realistic estimates for parameters needed in modeling (Figure 22).

The first year's growth, on average 173 mm (6.8 inches), is longer than that observed in a number of other investigations. This was possibly due to uncertainties in accurate detection of annuli. In Priest Lake, Idaho, lake trout averaged 3.5 to 4 inches (TL) at the first annulus (Rieman et al. 1979). Similarly, first year growth for lake trout from 4 of 6 Canadian Shield Lakes were less than those we observed (Scott and Crossman 1973). Likewise, lake trout averaged 4.6 inches (fork length) from Lake Tahoe during the 1938 to 1964 period (Hanson and Cordone 1967). For Flathead Lake, lake trout growth across all ages were near the maximums observed for lake trout across their range. Healey (1978) compared lake trout growth in populations under various circumstances, throughout their normal range in lakes north of $60^o$ N and under heavy exploitation or predation. The growth rates we observed were similar to lake trout populations with the highest growth rates in their normal range (Wollaston Lake and Lac La Ronge), greater than growth rates observed in lakes north of $60^o$ N, and again similar to populations experiencing heavy exploitation or predation (Huron, Michigan, and Superior). It has been shown that growth was slower at higher latitudes and faster if fish forage was available (Martin 1951). Both of these characteristics may partially explain the rapid growth rate in Flathead Lake. Another explanation for rapid growth rate could be high exploitation. An increased growth rate presumably would be a compensatory mechanism for heavy mortality. Lake trout populations under high exploitation or predation showed increased growth rates when compared to those of the same populations prior to the increased mortality (Healey 1978). In recent years, the Flathead Lake population has been heavily exploited (see theoretical yield and creel survey sections in this report). CSKT and MFWP began an intensive lake sampling program and lakewide creel survey in summer of 1998. The results of these activities will provide additional detailed information to further address this analysis.

## THEORETICAL LAKE TROUT YIELD INDICES

### Introduction

There are a number of theoretical yield indices available to estimate the annual production of lake trout for a lake. In northern British Columbia, fisheries managers have used these indices to help determine management strategies for lakes dominated by lake trout (deLeeuw et al. 1991). Yields are estimated using mathematical relationships between morphologic or chemical lake characteristics and measured fish production. These values should be applied with caution since they are estimates based on relationships developed from other lakes and not based on empirical data from Flathead Lake. Managers can use production estimates to determine fishery characteristics and for comparisons with estimates of harvest. For example, lakes with relatively high production would more likely support intense fisheries with high harvests than lakes with relatively low production (deLeeuw et al. 1991).

### Methods

To estimate annual and sustainable yield for Flathead Lake we referenced available literature and case studies. Equations were constructed using a relationship of sustained yield with various lake characteristics. We used four equations to estimate lake trout yield. The first was constructed from 19 moderate to heavily exploited Canadian lakes. The average annual lake trout yield (y) was 0.225 kg/ha/yr, with 95 percent confidence limits of 0.165 to 0.365 kg/ha (deLeeuw et al. 1991). This lead to the following equation:

(1) $y\ (\text{kg}) = \text{lake area (ha)} \times .225$.

A second equation drawn from the same database produced a “weak positive correlation” between yield and mean depth (Z) (deLeeuw et al. 1991):

(2) $y\ (\text{kg/ha}) = 0.094 + 0.085\ \text{Log}_e\ (Z)$.

A third method was the thermal habitat volume (THV) estimate developed by comparing sustained yield (SY) of lake trout in 15 lakes (located across Canada and the north central United States) with the THV. Thermal habitat volume is the volume of water available in the optimal temperature range for lake trout during the summer months (Christie and Regier 1988). The relationship for lake trout is described by the following equation:

(3) $\text{Log SY} = 0.81\ \text{Log THV} + 0.94;\ n=15\ r^2=0.86$.

A fourth method we employed was also derived from the 15 lakes used in equation (3). This equation related the total surface area (A) of the lake with estimates of sustained yield (Christie and Regier 1988):

(4) $\text{Log SY}=0.933 \text{ Log (A)} - 0.111$; $n=15$ $r^2=0.706$.

## Results and Discussion

Using Geographic Information System (GIS) technology, we estimated the surface area of Flathead Lake excluding islands to be 50,053 ha (123,374 acres). We used estimated mean depth and volume values of 50.2 m and 23.2 $km^3$, respectively (Zackheim 1983).

Using equation (1), the estimated annual yield in catchable lake trout for Flathead Lake was 11,405 Kg (25,148 pounds). Using 50.3 m to be the mean depth for Flathead Lake in equation 2, the annual lake trout yield (kg) per hectare for Flathead Lake would be 0.239 kg/ha/yr. Using this value in place of 0.225 in equation (1), the annual yield for lake trout in Flathead Lake would be 11,963 kg/yr or 26,378 lb/yr.

The previous models correlated morphologic characteristics with sustainable yields. Water temperature is an important physical characteristic that influences many biological and ecological functions. The following model incorporates the volume of water in a lake within a preferred temperature range for a fish species (Christie and Regier 1988). Investigators measured the thermal habitat space over the summer season by integrating the pelagic volume with water temperatures within species' optimal thermal niches. The amount of water during the summer months within a temperature range that is physiologically optimal for lake trout relates strongly to the productive capacity of a given lake (Christie and Regier 1988). Thermal habitat volume, THV (cubic hectometers per 10d), was used as a predictor variable in a regression equation estimating total sustained yield (SY, kilograms per year) of lake trout being commercially fished. THV was strongly correlated with lake trout yield (Equation 3)(Christie and Regier 1988). We combined this equation with 1990 water temperature profile data for Flathead Lake to estimate the annual sustained yield in lake trout for Flathead Lake. The optimal temperature range for lake trout was determined to be 8 to 12$^o$ C (Christie and Regier 1988). The first step in the estimation process was to create a hypsographic curve for Flathead Lake (Figure 23). A hypsographic curve relates water depth to lake surface area, allowing investigators to estimate pelagic volumes (Hakanson 1977). The second step involved creating an isotherm diagram for the 1990 water temperature data on Flathead Lake (Figure 24). From this curve, we could estimate the depth range encompassing the optimal temperatures throughout the summer season (June 5 through September 4). Using these two curves and a total surface area of 50,053 ha (123374. 4 acres) we were able to calculate the pelagic volume of water within the preferred temperature range for lake trout for nine 10-day intervals. Summing these volumes we estimated the total THV for the summer season to be 47382 ($hm^3$). This value in the previously mentioned equation estimated SY to be 8265.1 kg/yr (18224.5 lb/yr) for lake trout in Flathead Lake. Finally, dividing this value by the surface area (50053 ha) produced an estimate for annual sustained yield of 0. 17 kg/ha (0.15 lb/acre).

The fourth equation produced a sustained yield estimate of 18,776 kg/yr (41,401 lb/yr) or 0.38 kg/ha/yr. This was the highest estimate of the four methods we employed.

Discussing available data at the time, Healey (1978) concluded lake trout populations are sparse and have low productivity, especially among the reproductive year classes, and that sustainable yields from lake trout are unlikely to exceed 0.5 kg/ha. He also predicted that if yield was above 0.5 kg/ha the trout population was likely to be overfished. Depending on relative growth rates and standing stock, sustainable yield would likely range from less than 0.2 kg/ha in low potential lakes to up to a maximum of 0.5 kg/ha in high potential lakes (Healey 1978). The above estimates for the annual lake trout yield in Flathead Lake ranged from 0.17 to 0.38 kg/ha or 8,265 kg/yr (18,225 lb/yr) to 18,776 kg/yr (41,401 lb/yr).

## LAKE TROUT TAGGING PROJECTS

### Introduction

We are using a number of different surveys to estimate lake trout population parameters such as abundance and mortality and growth rates. In 1997, we began an extensive lake trout tagging program in Flathead Lake. The goal of this project is to tag, release, and recapture as many lake trout as possible in all size classes. We hope to tag, release, and recapture enough fish to produce estimates of abundance, population size structure, mortality and growth rates, and biomass. This information is important to the development of successful management and mitigation alternatives.

Previous mark-recapture studies have been conducted on Flathead Lake (1992-1996) and other waters such as Lake Tahoe and Flaming Gorge Reservoir. Such projects were completely dependent on the volunteer participation of lake anglers. This approach included marking lake trout with Floy anchor tags, accurately recording biological and catch data, releasing tagged fish, and then later recapturing tagged fish.

### Methods

A lake trout tagging program was conducted from 1992 through 1996 on Flathead Lake. Anglers tagged lake trout on both the north and south halves of the lake using a variety of angling techniques. Anglers recorded fish length, weight, and location of capture and inserted a numbered Floy tag. Defining each year as a sample period, we used a modified Schnabel estimate to calculated the abundance of catchable (>400 mm) lake trout (Ricker 1975). This sampling methodology was again followed for the tagging study initiated in 1997.

### Results

Over the first five-year period (1992-1996), volunteer anglers tagged 1,376 lake trout, caught 11,572 fish and recaptured 11 tagged fish. We estimated abundance at 353,732 catchable lake trout (>400 mm) with a 95 percent confidence interval of 215,472 to 786,071. We were concerned with possible loss of tagged fish during the five-year interval. So at the end of each year, we applied a mortality rate of 20 percent per year to the number of tagged fish at large. Annual mortality of 20 percent was a conservative value which most likely underestimates the true mortality rate (Beauchamp 1996, Walters et al. 1980, Healey 1978, Payne et al. 1990). Assuming 20 percent is a minimum value, the estimated number of tagged fish at large is a maximum value. Recalculating the modified Schnabel estimate, the number of catchable lake trout dropped to 237,026 with a 95 percent confidence interval of 144,381 to 526,725. These estimates did not address the incidence of tag loss from marked fish, which is another source of error in population estimates (Ricker 1975).

Recent studies using the FD-97 Floy tag or similar Floy anchor tags describe the incidence of tag shedding in lake and bull trout. Baxter and Westover (1999) double-tagged adult bull trout in the Wigwam River, British Columbia. They assessed tag loss using the FD-97 style tag and found a ten percent annual loss of this tag in 188 returning double-tagged fish. The ten percent value may slightly underestimate loss, since double-tagged fish which lost both types of tags would not be included. However, since the other type of tag (P1I) also showed good retention (89 percent), the proportion of tagged fish losing both tags was small. This study portrayed high retention over a time period of one year. During this period, tagged fish migrated out of the river downstream to a larger reservoir, where they remained until the following year's spawning migration. Fabrizio et al. (1996) assessed tag retention over a longer time period (9 to 18 years). They used Floy anchor tags similar to the FD-97 style, which we and the previously mentioned study employed. Fabrizio et al. (1996) constructed and compared models to estimate tag shedding rates in Lake Superior lake trout. These investigators observed overall higher rates of annual loss and variation between tag types, although this was not statistically significant. Models estimated the rate of annual tag loss for tag styles FD-67, FD-67C, and FD-68BC (all Floy anchor tags) to be 25.9 percent, 35.7 percent, and 48.1 percent, respectively. These two studies portrayed annual tag loss ranging from 10 to 48 percent. Taking a mean of these four values (10.0, 25.9, 35.7, and 48.1 percent) results in 29.9 percent. Although this value was not determined empirically, we can apply this 30 percent annual tag loss to our rough calculations for estimating lake trout abundance in Flathead Lake.

We can incorporate annual tag loss into the calculations used to estimate lake trout abundance in Flathead Lake. By applying an annual tag loss of 30 percent and a mortality of 20 percent, we reduce annual tag retention to 56 percent. We applied this value to the number of marks at large at the end of each year (1992-1996) in the abundance estimate derived from tagging data. The corresponding abundance estimate was 134,249 lake trout greater than 400 mm in length with a 95 percent confidence interval ranging from 85,665 to 310,146 fish. Thus, the abundance estimate was reduced by 43 percent following the inclusion of the estimated tag loss.

While this tagging program was in progress, two of the anglers were removing adipose fins from lake trout they captured and recording the incidence of recaptures and the total number of lake trout caught. These anglers fished only the north half of Flathead Lake. During 1993 through 1995, they caught 5,676 lake trout, clipped 4,729, and recaptured 38. Using a modified Schnabel estimate and monthly sample periods, we calculated 283,609 catchable lake trout (>400 mm) with a 95 percent confidence interval of 214,151 to 419,753. The opportunity for loss of tagged fish increases with sampling periods extending over numerous years. Using only the 1993 and 1994 data (3,599 caught, 2,667 marked, and 15 recaptured), the estimate was 229,185 trout, 95 percent confidence interval of 142,130 to 390,102. Again, we applied a 20 percent annual mortality to the number of tagged fish at large at the end of each year. This reduced the estimate to 240,217 catchable lake trout (95 percent confidence interval of 181,386 to 355,531) for the 1993 to 1995 period and 209,898 lake trout (95 percent CI of 130,169 to 357,273) for 1993 and 1994 data. Both of the mark-recapture projects estimated similar numbers of catchable lake trout. We hope to refine these estimates with the ongoing tagging project.

The ongoing program started in May, 1997. As of November 1998, 12 volunteer anglers have caught 7,008 lake trout of which 3,581 were tagged and released. These same anglers have recaptured 31 tagged lake trout. Catch was not distributed evenly among the 12 anglers. To date, one angler has caught roughly half of all caught lake trout. The mean and median total lengths of lake trout caught were 526 mm (20.7 inches) and 508 mm (20 inches), respectively (Figure 25). The program will continue through May 1999, at which time anglers will cease tagging fish but will continue to record catch data for an additional year.

## LAKE TROUT FOOD HABITS

### Introduction and Methods

We collected data on lake trout food habits to calibrate a bioenergetics model (following section page 23) for Flathead Lake and improve the model's predictive potential. We collected stomach samples in four seasons using lakewide sampling techniques including gill-net surveys, fishing derbies, and volunteer anglers. In April and May 1996, we took samples from the lakewide gill-net catch and from over 30 anglers in the south half of the lake. In June 1996, we collected samples at the MacMania fishing derby conducted on the north half of the lake. In both August and December 1996, we took samples from lakewide gill-net surveys. For all samples, fish stomachs were removed, prey items were enumerated and their wet weights recorded. We separated prey items into eight general categories, kokanee, lake whitefish, other fish, unidentified fish, aquatic insects, terrestrial insects, *Mysis*, and other. We combined wet weights of each prey type in five size groups of lake trout to determine which segment of the lake trout population imposed the greatest predation pressure on recently stocked kokanee.

### Results and Discussion

In 1996, we examined 449 lake trout stomachs (Table 6). There were seasonal differences in the proportional wet weight and total weight of each prey item, frequency of prey occurrence and in the proportion of empty stomachs. December samples had the greatest percentage of empty stomachs (56 percent), while August samples had the lowest percentage (9 percent).

Samples collected in April and May had the highest incidence of kokanee (Table 6). Lake trout in the 376 to 500 mm and the 501 to 625 mm length groups contained the majority of observed kokanee biomass, 23 and 43 g, respectively. Lake trout less than 375 mm TL or greater than 626 mm (TL) contained no or few kokanee. In April 1996, as part of the Hungry Horse mitigation kokanee reintroduction test, 939,000 yearlings were stocked into South Bay (Carty et al. 1997). It appeared that a large number of these fish moved north into the main body of Flathead Lake and became available to lake trout. By June 1996, the incidence of kokanee in lake trout decreased. Apparently, kokanee were less abundant, less available to lake trout, and thus less frequently observed in samples.

Lake whitefish comprised a large proportion of total prey biomass in all seasons, with the exception of the December samples (Table 6). June samples showed the highest values for lake whitefish total biomass (3363 g), percent of total prey biomass (82 percent), and percent frequency of occurrence (46 percent). The August samples contained the second highest values for lake whitefish in total prey biomass (605 g) and percent of total biomass (75 percent). The total prey biomass values in the two summer samples were greater than those in the other sampling periods and thus lake whitefish dominated total and percent biomass of prey when all seasons were combined. We did not observe lake whitefish or kokanee in December samples.

The “other” fish category became important in the April/May and December samples, making up 30 to 40 percent of prey biomass. The “other” fish category included numerous fish other than kokanee or lake whitefish including suckers, minnows, trout and char, and yellow perch. We observed aquatic insects in lake trout diet in each of the four sample periods. Although the percent frequency of occurrence was high in most seasons and when all seasons were combined, the total biomass of aquatic insects and the percent of total prey biomass were very low. Similarly, we observed *Mysis* in samples from all seasons and *Mysis* comprised only a small percentage of the combined total prey biomass. However, *Mysis* made up the highest percentage of total prey biomass in the December sample. Lake trout under 500 mm (TL) contained the majority of observed *Mysis*. When all seasons were combined, fish biomass greatly outweighed the biomass of other prey items for all length groups of lake trout. *Mysis* and insect biomass were higher in the smaller lake trout length groups and lower in the larger length groups.

Lake trout food habits were examined to identify the types and relative proportion of different prey items for this dominant predator in Flathead Lake. Similar predator food habits information has been collected for lake trout and northern pikeminnow in the Flathead River (Zollweg 1998). All of these studies indicate a low incidence of trout and char in predator diets. However, due to their high abundance, predator populations likely impose a significant source of mortality for species such as bull trout and westslope cutthroat trout. Estimates of these losses will be more feasible as we gain a better understanding of population sizes, and the spatial/temporal overlap of predator and prey populations.

## MERCURY AND POLYCHORINATED BIPHENYLS LEVELS IN FISHES

Chemical contaminants in the environment accumulate in fish tissues. To assess the level of health risk for anglers and fish consumers in Montana, MFWP tested fish from selected waters across the state. Flathead Lake and Whitefish Lake were included in this test. The survey looked at levels of polychlorinated biphenyls (PCBs) and methylmercury (Hg) in lake trout and lake whitefish (Phillips and Bahls 1994).

We collected fish with gill nets and preserved fillets for laboratory analysis (Phillips and Bahls 1994). Table 7 contains the results of the testing. The fish's age and position in the food chain influence toxin accumulation. A species at or near the top of the food chain bioaccumulates toxins by consuming species which have previous accumulated toxins. The longer a fish lives, the more contaminants it accumulates. Therefore, large piscivores have the highest concentrations of contaminants. Lake trout fit these criteria. Lake trout from Flathead Lake have moderate to high levels of Hg and PCBs, levels high enough to warrant public advisory warnings on consumption of larger fish. Lake trout from Whitefish Lake showed similar levels of Hg and PCBs. The sample of larger lake trout for Whitefish Lake was small and did not include the largest sizes, which contained the highest levels of PCBs in Flathead Lake. Lake whitefish from Flathead Lake had low to moderate levels of Hg and PCBs were not detected. Table 8 depicts the meal guidelines for consumption of fish with these containment levels. Generally, anglers need to be cautious with regular consumption of lake trout, particularly the large fish. The Montana Department of Public Health and Human Services presented these cautions in a Montana Fish Consumption Advisory.

## ANGLER CREEL SURVEYS AND LAKE TROUT EXPLOITATION

Angler creel surveys provide valuable information, including estimates of angler use, catch, harvest, and availability of fish species. A number of creel surveys and survey techniques have been employed on Flathead Lake in the last 40 years. For example, since 1969, MFWP has conducted a mail-in creel survey to estimate angler pressure on state waters, including Flathead Lake. Presently, this survey is conducted every other year; the most recent survey was completed in 1997. In addition to the mail-in survey, roving creel surveys were conducted. The most recent lakewide roving creel survey was completed in 1992 and one is in progress in 1998/1999, both were part of the Hungry Horse Dam Fisheries Mitigation Program (Evarts et al. 1994, MFWP and CSKT 1993).

Similar to other indices presented in this report, creel surveys highlight dramatic changes in the Flathead Lake fishery. For example, angling pressure recently decreased on Flathead Lake (Figure 26) (MFWP 1998, Evarts 1998). There appeared to be roughly a 50 percent drop in angler pressure from the 1980s to the early 1990s. This drop in pressure is believed to be a response by anglers to changes in fish species composition, specifically the collapse of the kokanee fishery (Evarts et al. 1994).

Prior to the late 1980s, kokanee and yellow perch provided most of the fish harvest on Flathead Lake, and in the early 1980s, kokanee represented over 90 percent of harvest (Robbins 1966, Graham and Fredenberg 1983). Following *Mysis* establishment, the fish community changed dramatically and kokanee disappeared. Lake trout now provide most of the harvest in Flathead Lake. In 1992, no kokanee were harvested and lake trout represented roughly 55 percent of harvest (Figure 27) (Evarts et al. 1994). In the 1980s, lake trout made up a very small percentage (less than 2 percent) of harvest. In all years, native bull and westslope cutthroat trout comprised a relatively small proportion of total fish harvest. In the 1960s, 1980s, and 1990s, they combined provided less than five percent of harvest (Evarts 1998).

In the 1992 survey, investigators estimated that anglers harvested 23,605 lake trout (Evarts et al. 1994). Approximately 98 percent of harvest consisted of fish less than 660 mm (26 inches) and the average length of harvested lake trout under 660 mm was 521 mm (20.5 inches) (Evarts et al. 1994). In 1992, a slot limit of 660 to 915 mm (26 to 36 inches) was in effect, which prohibited harvest of lake trout within the slot limit (Appendix A). For the reported catch, the majority of fish were less than 660 mm (86 percent), just over 2 percent were greater than 915 mm, and about 12 percent were within the slot. A length-weight regression ($r^2 = 0.987$, $n = 136$) for lake trout was developed from lake trout captured by gillnetting in Flathead Lake ($W_{(g)} = 0.00000584 * TL\ (mm)^{3.05}$, where W equals weight in grams and TL equals total length in mm), the 521 mm average fish weighed 1129 grams (2.5 lbs.).

By standardizing pressure estimates for earlier surveys, using the statewide mail-in survey, and recalculating lake trout harvests, investigators compared harvests reported in earlier surveys with the 1992 survey (Evarts 1998). There was a progressive increase in lake trout harvest over the last four decades. In 1962, lake trout harvest was estimated at 1,248 fish, while in 1981 it rose 55 percent to 3,600 lake trout with only an estimated 17 percent increase in angler pressure. In 1992 it rapidly increased to 21,656 lake trout (Figure 28), a 500 percent increase with a 50 percent drop in total angler pressure (Evarts 1998). This increasing trend in the lake trout harvest is due to increased lake trout abundance (reflected in gill-net monitoring surveys), and re-directed angler pressure (resulting from the loss of the kokanee fishery). In 1992, approximately 80 percent of angler pressure was directed at lake trout (Evarts et al. 1994) while prior to the kokanee population crash, they received less than 15 percent of the total angler pressure.

At this time, we have two ways to evaluate lake trout exploitation by comparing lake trout harvest in 1992 with estimates of lake trout abundance. One analysis compares the estimated number of harvested lake trout (23,605) and the estimated abundance of catchable lake trout (estimates ranged from 134,000 to 354,000 catchable lake trout) from the limited mark/recapture estimates (as described in a previous section). If harvest was apportioned to catchable lake trout, it represents a fishing harvest of roughly 7 to 18 percent per year.

A second approach is to multiply the weight of the average lake trout harvested (1129 g) by the estimated number of lake trout harvested (23,605) to produce a rough estimate of the harvested lake trout biomass (26,650 kg) in 1992. Harvested biomass in 1992 may then be compared to our theoretical annual yield estimates for lake trout which ranged from 8,843 kg/yr to 18,775 kg/yr (as discussed in a previous section).

Estimated annual harvest was greater than the theoretical annual yield estimates. The 1992 harvest represents 0.53 kg/ha yield. Evans et al. (1991) reported 0.20-0.75 $kgha^{-1}yr^{-1}$ as the observed range of long-term sustainable lake trout yields. Healey (1978) concluded that sustainable yields from lake trout are unlikely to exceed 0.5 kg/ha and predicted that if harvest was above this value then the lake trout population was being over-exploited. These data indicate that Flathead Lake lake trout are being heavily exploited. However, we must be cautious when applying the theoretical yield estimates since these are not based on actual empirical data for Flathead Lake.

Lake trout populations respond to high exploitation in predictable ways. In general, high mortality rates or exploitation results in specific changes in population characteristics including reductions in average age, length, weight, and number of age-classes, and increases in growth rate, fecundity, and biomass of younger age-classes (Johnson 1976, Healey 1978, Evans et al. 1991, deLeeuw et al. 1991). As mortality rates increase, the number of older fish decreases leading to a population dominated by smaller fish. In fisheries having management goals which include a trophy component or a natural length distribution a high level of harvest is generally not an option. At present, this appears to be the condition of the Flathead Lake lake trout population, although a fishery for larger fish still exists. As creel and gill-netting results indicate, the smaller lake trout (<660 mm) dominate the population with relatively fewer large (>660 mm) lake trout. Recent creel data show decreasing CPUE for large (>915 mm) lake trout suggesting a decrease in abundance of the larger fish (Evarts 1998). There have been a number of changes to Flathead Lake in recent years. These include dramatic changes in the aquatic community and trophic dynamics. *Mysis relicta* became established in Flathead Lake in the mid-1980s and reduced the abundance of large zooplanktors (Beattie and Clancey 1991, Spencer et al. 1991). The kokanee salmon population collapsed in the late-1980s and lake trout and lake whitefish have become the dominant gamefish. It is unclear which specific mechanisms or combination have changed the lake trout population, but possibilities include improvements to juvenile lake trout forage as *Mysis* became established leading to increased survival and abundance of small lake trout and/or a decrease in the abundance of older, larger lake trout due to disappearance of a preferred prey fish (kokanee) and/or high exploitation rates by anglers. One point is clear, the fishery has not yet stabilized since the perturbations associated with *Mysis* changed the foodweb and, likewise, the lake trout fishery is still developing as pressure and harvest continue to increase.

## FISHING LOG PROGRAM

Since 1951, MFWP has compiled fishing logs from anglers across the state. These anglers volunteer to record fishing activities and have provided a long-term record of species distribution, angler effort, and catch. Once a year data are summarized for each waterbody. For Flathead Lake there were numerous log entries over the 45-year period. These logs also reflect the major changes in the lake fishery.

Summer logs from 1965 to 1994 provided insight into fishery changes in Flathead Lake. Percent species composition in catch showed many of the same trends as did other monitoring indices presented in previous sections of this report. For example, from 1965 to 1983, with the exception of 1970, kokanee dominated the catch (Table 9). By 1987, kokanee had completely disappeared from the catch, corresponding with the documented crash in the kokanee population (Beattie and Clancey 1991, Spencer et al. 1991). Conversely, lake trout numbers increased in angler catch following the establishment of *Mysis*. The log showed that in the mid to late 1980s, lake trout began to increase in the proportion of catch and, since 1992, dominated catch (Table 9 and Figure 29). In all years, non-native fish provided the majority of harvest and fishing opportunity. These logs indicate that anglers witnessed the same changes in fisheries we observed in our monitoring indices.

## KOKANEE REINTRODUCTION TEST

As part of the Hungry Horse Dam mitigation program, fisheries biologists from the CSKT, U.S. Fish and Wildlife Service (USFWS), and MFWP have for five years cooperatively monitored and reported the outcomes of the “kokanee test,” an experimental effort to re-introduce kokanee salmon into Flathead Lake. Findings were documented and published in reports prepared for the Bonneville Power Administration (see Deleray et al. 1995, Hansen et al. 1996, Carty et al. 1997, Carty et al. 1998, and Fredenberg et al. 1999).

From 1993 through 1997, about 3.2 million kokanee yearlings and 2.6 million young-of-year kokanee were stocked into the Flathead Lake and River System. Survival of stocked kokanee was monitored to develop and adjust management strategies designed to maximize survival of stocked fish. In 1998, monitoring results were used to reach a decision to stop the five-year “kokanee test” due to the inability of the test to meet established success criteria. The three success criteria were: (1) 30 percent survival of kokanee one year after stocking; (2) yearling survival to adulthood of 10 percent; and (3) annual angler harvest of 50,000 kokanee (≥11 inches) and fishing effort ≥100,000 angler hours. Kokanee stocking was discontinued following the 1997 plants. Monitoring continued through 1998. The Hungry Horse Fisheries Technical Team summarized the important findings for each year of the program and, based on that summary, agreed on the following general conclusions about the kokanee mitigation program in Flathead Lake.

### Summary of Kokanee Stocking and Monitoring

**1993**

1. Lake trout predation was a major source of kokanee mortality.

2. Monitoring efforts must be increased to adequately evaluate kokanee survival.

**1994**

1. Lake trout predation on kokanee was very high.
2. In the absence of predation, hatchery-reared kokanee could adapt and grow in the lake, based on the summer net-pen experiment and fall captures.
3. Kokanee broodstock held at Creston Hatchery could contribute substantially to egg supplies.

**1995**

1. Lake trout predation was the primary factor limiting hatchery kokanee survival.
2. Short-term survival could be increased by stocking kokanee in a thermal refuge, (ie. South Bay) an area from which lake trout are excluded for at least part of the year.
3. Hatchery-reared kokanee released as yearlings grew to similar size at maturity as wild kokanee did historically in Flathead Lake. However, densities of salmon were currently much lower than historic levels.

**1996**

1. Downstream movement of kokanee over Kerr Dam and out of Flathead Lake was a considerable source of short-term loss when kokanee were stocked into South Bay in early spring (ie. April).
2. The thermal refuge in South Bay did not develop until late June.
3. Hatchery-reared kokanee matured in the lake at ages 1 through 4.
4. Most mature kokanee observed homed to their stocking location.
5. Even with the kokanee season open (Appendix A), a fishery did not develop.
6. Bioenergetics modeling showed that, at current stocking levels, lake trout predation accounted for nearly all yearling kokanee stocked during the first 12 months post-stocking.

**1997**

1. Kokanee stocking from 1993 to 1997 did not meet any of the three predetermined success criteria: (1) 30 percent survival of kokanee one year after stocking; (2) yearling survival to adulthood of 10 percent; and (3) annual angler harvest of 50,000 kokanee ($\geq$11 inches)

and fishing effort ≥100,000 angler hours.

2. The stocking strategy using South Bay did not successfully protect kokanee from predation.

3. The kokanee stocking effort was terminated.

**1998**

1. The abundance of mature one-year-old males six months after stocking was not a reliable indicator of adult abundance one year later.

2. A kokanee fishery did not develop and previous year's stocking efforts did not meet success criteria.

### General Conclusions Based on Stocking and Monitoring 1993-1998

1. The three success criteria were not met with current stocking levels in the present lake environment, based on data from monitoring and predictions of bioenergetic models.

2. When using yearling kokanee, lake trout predation was the primary obstacle to possibly achieving the three success criteria.

3. Monitoring efforts were sufficient to evaluate whether the kokanee test met the three success criteria.

## FLATHEAD LAKE BIOENERGETICS MODELING

### Introduction

Monitoring and research efforts suggested that lake trout predation was the primary factor limiting the success of kokanee restoration in Flathead Lake (Deleray et al. 1995, Hansen et al. 1996, Carty et al. 1997). Lake trout populations increased dramatically since the establishment of *Mysis* in the early 1980s and now impose a huge predatory demand on kokanee and other forage. Kokanee monitoring results indicated high post-stocking losses and low adult spawner returns, but have not allowed us to quantify first-year survival or to extrapolate data to predict outcomes of alternative stocking and management strategies. We employed a bioenergetic modeling to examine the predator/prey relationship between lake trout and kokanee in Flathead Lake. By quantifying the temporal, spatial, and size related processes involved in kokanee predation, we hoped to identify which segments of the lake trout population imposed the greatest impact on kokanee. Model simulations were completed by Dr. David Beauchamp (Utah State Cooperative Fisheries and Wildlife Research Unit, USU) using existing empirical data on diet, distribution, growth, abundance, and survival of lake trout in Flathead Lake (Beauchamp 1996). Using the model to define the dynamics of predation over time, space, and body size, different management scenarios were evaluated to determine the number of kokanee required to satisfy piscivore demand, supply a satisfactory fishery, and to meet spawning or egg take goals. The simulations were designed to evaluate predation under: (1) existing kokanee stocking scenarios; (2) other stocking scenarios; and (3) changes in the lake trout abundance and size structure.

## Methods

Lake trout consumption demand on kokanee and alternative prey in Flathead Lake was estimated by applying a bioenergetics model (Hewett and Johnson 1992) parameterized for lake trout (Stewart et al. 1983). Methods used to estimate growth, survival, size- and season-specific diets, thermal experience, and lake trout population parameters are described in detail by Beauchamp (1996). Lake trout diet patterns employed in the model were based on data collected from Flathead Lake in 1994.

Data needs for the model were provided by MFWP, CSKT, USFWS, and the University of Montana's Flathead Lake Biological Station. Input data were based primarily on field data collected in 1994. Model simulations and sensitivity analyses were completed by Dr. Beauchamp. After initial data preparation and preliminary model runs, Dr. Beauchamp and biologists representing the cooperating agencies collaborated in a workshop where alternative management and stocking scenarios were examined. Results of nominal model runs and simulations based on alternative scenarios are included in the final report (Beauchamp 1996).

## Results and Discussion

Model simulations suggested that lake trout predation imposed serious losses on the kokanee population in Flathead Lake (Beauchamp 1996). The heaviest predation in 1994 occurred during the first month after the June stocking. Kokanee losses during this first month exceeded total predation losses accrued during July through September. Lake trout in the 501-625 mm and 626-750 mm length groups were responsible for more than 64 percent of the estimated predation, and lake trout 376-500 mm consumed another 21 percent. Due to the relatively low numbers of lake trout greater than 626 mm, larger lake trout were responsible for the smallest percentage of kokanee predation. Lake trout abundance was likely underestimated in model simulations, because size and abundance was based on hydroacoustic and gill-net surveys conducted in August 1995. Since standard hydroacoustic methods cannot detect fish $\leq 1$ m from the bottom, some unknown fraction (possibly 10-50 percent) of the predator population was probably not detected. When larger lake trout populations were modeled, a 10 percent increase in lake trout abundance resulted in kokanee survival one year after stocking dropping from 13.2 percent to 4.2 percent. If the lake trout population was 50 percent larger than the acoustic-based estimate, no kokanee survival was predicted after one year.

Model simulations suggested that the kokanee mitigation program could not meet its goals under the current stocking regime of releasing 800,000-1,000,000 yearling kokanee in late spring. The simulations of lake trout predation indicate that predation losses alone could account for nearly all of the kokanee stocked. In addition to lake trout predation, there were other sources of mortality and emigration from the system which further reduced recruitment of adult kokanee. The primary areas of uncertainty in our model application included lake trout abundance and size structure, the spatial distribution of predation throughout the lake, and seasonal diet composition. These research needs have been or are currently being addressed through research projects on Flathead Lake.

# HUNGRY HORSE RESERVOIR GILL NET SURVEYS

## Introduction

Hungry Horse Dam impounds the South Fork of the Flathead River approximately 8 km from its confluence with the main stem Flathead River. The dam isolates a native species assemblage in the reservoir by preventing upstream migration of fishes from the lower Flathead system. The reservoir is a stronghold for bull trout and westslope cutthroat trout with restrictive fishing regulations.

MFWP has used gill netting to monitor fish population abundance, size- and age-structure, and community composition in Hungry Horse Reservoir (HHR) since 1958. Consistent sampling during this period provided data on long-term population trends and served as a baseline for current population assessments. Gillnetting was one of two indices used to monitor bull trout in the South Fork Drainage and one of three indices used to monitor westslope cutthroat trout populations. In this section we summarize historical netting information, but focus on fall gill-netting results.

## Methods

Field crews used standard, experimental floating and sinking nets to sample fish in near-shore areas. Nets were 38.1 m long and 1.8 m deep and consisted of five equal length panels of 19, 25, 32, 38, and 51 mm (bar) square mesh. Floating nets sampled fish from the surface down 1. 8 m and sinking nets sample from the bottom up 1.8 m. A floating net set consisted of 2 nets tied end-to-end and is fished perpendicular to shore. A sinking net set is a single net fished perpendicular to shore. All nets were set directly from shore.

In 1988-1989, we continued with established seasonal netting protocol (as described in May et al. 1988). Gill nets were set during May, August, and October in three reservoir areas (Figure 30): Emery (northern 1/3 of reservoir), Murray (middle 1/3), and Sullivan (southern 1/3). Seven floating net sets and five sinking nets were set overnight during each sampling period in each area. In 1990-1995, the number of nets set per night was reduced to four floating and three sinking sets in each area. Seasonal netting was discontinued in 1992. Only the fall (October) series has continued for annual monitoring.

Summer was the least effective season to catch trout, whitefish, and other species due to warm surface temperatures and was discontinued after 1992. Spring netting was discontinued in 1992 because of large catches of mature cutthroat trout migrating to spawning streams. Therefore, we have narrowed recent and future population monitoring in the reservoir to fall gill-netting.
Gill net catch consisted almost exclusively of native fish species since monitoring began in 1958. These species include westslope cutthroat trout, bull trout, mountain whitefish, northern pikeminnow, largescale sucker, longnose sucker, and pygmy whitefish. Floating nets were used to target westslope cutthroat trout because they generally inhabit the upper water column.

Sinking net catch was more representative of species composition in the reservoir. Neither net type was effective for capturing pygmy whitefish because they generally do not inhabit shallow, near-shore areas and are rarely captured in the mesh sizes of our nets.

Fish caught in nets were identified to species, weighed (g), and total length measured (mm). For gamefish, we determined sex and state of maturity (immature, mature, ripe, spent). Scales and otoliths were also removed for age and growth information. Data not summarized in this section have been archived by MFWP.

## Results and Discussion

Long-term gill net data exhibit the stability of the HHR fish community. Species composition and relative abundance appear to be consistent based on seasonal sampling (1988-1992, Tables 10, 11, and 12) and long-term trends, based exclusively on fall gill net catch (1958-1998) (Table 13).

Westslope cutthroat trout catch rates in fall floating gill nets were variable (mean=2.2, sd=0.8), but no significant trend was detected over time ($r_s$=-0.04, p>0.9, n=15 yr, see Figure 31). Despite moderate annual variability, long-term catch rates were also consistent in sinking nets for most species. Tables 13 includes MFWP data for fall sinking gill nets for 1958-1998. Trend analyses (rank correlation) using these data indicate that bull trout abundance is stable, with evidence of increase over time ($r_s$=0.60, p<0.05; see Figure 31). Mountain whitefish catch was more variable than other gamefish, but does not indicate any dramatic population changes in the long-term ($r_s$=-0.04, p>0.9). Relative abundance of mountain whitefish and several non-game fishes is shown in Figure 32.

No significant changes were detected in the size distribution of bull trout and cutthroat trout caught in gill nets. Length-frequency histograms for these species are displayed at 5-year intervals in Figures 33 and 34. Comprehensive age and growth information was also calculated for game fishes in 1983-87 by May et al. (1988). Recent age structure data has not been analyzed since there is little indication of change in these populations.

The size distribution of mountain whitefish appears to have changed in recent years (Figure 35). The population mode has traditionally been in the 300-324 mm size range. In 1997 and 1998, the modal and mean size decreased. This trend may warrant further investigation if it persists.

We assume that gill net catch is an accurate index for most fish population characteristics. Shoreline net catches were influenced by differences in species abundance and vulnerability to nets, as well as seasonal variation in water temperatures, fish migration, and habitat use. However, annual variation attributed to these factors should have little effect on our interpretation of long-term population trends given the number of years in these data sets.

# FLATHEAD RIVER: MAIN STEM AND SOUTH, MIDDLE, AND NORTH FORKS MONITORING SURVEYS

## WATER TEMPERATURE MONITORING AND ASSESSMENT OF SELECTIVE WITHDRAWAL

### Background

Hungry Horse Dam impounds the South Fork of the Flathead River approximately 8 km upstream from its confluence with the main stem Flathead River. The North and Middle forks are unregulated and retain natural flow and temperatures regimes throughout the year. The influence from Hungry Horse Dam effects discharge and water temperature in the South Fork below the dam and throughout the main stem Flathead River from the South Fork confluence approximately 64 km downstream to Flathead Lake.

Hungry Horse Dam was originally designed with 4 turbine penstocks located 73 meters (241 feet) below full pool. Water discharge from this depth into the South Fork Flathead River remained about 4-6 $^{o}$C (39-43 $^{o}$F) year round. Occasionally, surface water as warm as 20 $^{o}$C was also released as spill. Thermal effects included short term fluctuations of up to 8.3 $^{o}$C and a gross reduction in annual accumulation of degree days. Rapid thermal spikes corresponded with sudden changes in discharge volume. Seasonal perturbations were typified by summer cooling and winter warming. These unnatural thermal conditions affected invertebrate (Hauer et al. 1994) and fish communities in the 72 km (45 miles) of the South Fork and main stem Flathead River downstream of Hungry Horse Dam.

In August 1995, selective withdrawal structures became operational on Hungry Horse Dam (Christenson et al. 1996). These structures were designed to allow thermally selective release of reservoir water and restore a more natural temperature regime to the Flathead River downstream.

The United States Geological Survey (USGS) has monitored water temperatures at consistent stations in the Flathead Drainage for decades. In 1994, thermal monitoring was expanded by MFWP, primarily to track the effects of selective withdrawal structures installed at Hungry Horse Dam. Monitoring of river temperatures was expanded to gain base line data prior to installation and to track temperatures as the system is operated. This information was one basis for operational recommendations at Hungry Horse Dam (Cavigli et al. 1998) and is critical for several ongoing fisheries studies involving predator distribution, radio telemetry of bull trout and westslope cutthroat trout, and fish growth.

### Methods

Ryan Instruments temperature recorders were installed at 5 locations in the Flathead River system (Figure 36). These locations, combined with established USGS stations, provided a thorough coverage of the river system. Thermographs operated upstream of the South Fork confluence served as controls, unaffected by Hungry Horse Dam releases. The Stillwater River site tracks inflows that moderate downstream reaches of the lower Flathead River. Other stations were positioned to track temperatures as dam releases progressed downstream to Flathead Lake. Thermographs installed by MFWP record temperatures every 30 min and are downloaded monthly. Thermographs maintained by the USGS have a similar recording interval. For the purposes of data management and analysis, temperature measurements are converted to daily maximums, minimums, and averages for each site.

## Results and Discussion

Operation of selective withdrawal returned a more normative thermal regime to the Flathead River upstream of Flathead Lake. Temperatures at Columbia Falls now closely parallel natural temperatures measured in the unregulated reach just upstream of the South Fork confluence (Figure 37). One noticeable exception is evident in late fall and winter, when the selective withdrawal operation ceased and hypolimnetic water was again released from the reservoir via penstocks near the base of the dam. This water remains at 4-6 $^{o}$C and actually warms the main stem when combined with natural flows from the North and Middle forks (typically 0-3 $^{o}$C November through February).

Benefits of selective withdrawal were apparent during its period of operation from June-October. The selective withdrawal apparatus has been operated each year since installation in 1995. Relatively isothermal dam discharge was replaced by warmer water that met or approached normative targets established for the South Fork (Figure 38). Target ranges were developed from historical temperature data from the North and Middle forks.

Limited stratification in the reservoir can make it difficult for dam operators to meet temperature targets early in the summer. For example, in 1996 the minimum target temperature could not be met until July 21 despite operation of selective withdrawal beginning June 1 (Figure 38). High spring runoff and a cool spring in 1996 delayed the establishment of warm surface layers for correct temperature moderation. However, even with limited stratification, South Fork temperatures were increased from 4-6 $^{o}$C to 10-13 $^{o}$C. This was likely a worst case scenario as reservoir models predict stronger thermal stratification in most years (Marotz et al. 1994). In reality, meeting targets is less critical in May and June because South Fork flows are diluted by high spring runoff when combined at the main stem; flows from the South Fork comprise a smaller percentage of the total discharge in the main stem in early summer.

Downstream effects of selective withdrawal in the main stem Flathead River are illustrated in Figure 39. Differences in South Fork and main stem thermographs between 1992 (pre-selective withdrawal) and 1996 (post-selective withdrawal) are dramatic. Main stem temperature spikes shown for 1992 resulted from a combination of hydropower generation or peaking operations and cold water releases at the dam. This inverse relationship between dam releases and temperature in the main stem is highlighted in Figure 40. In 1996, drastic changes in dam outflow still occurred, but did not result in temperature spikes because of selective withdrawal.

## Ongoing and Future Investigations

Continued monitoring of selective withdrawal will include assessment of effects on the Flathead River food web. Return of normative river temperatures should increase diversity and abundance of certain groups of macroinvertebrates. Prior to selective withdrawal, Hauer et al. (1994) designed and completed a study of macrozoobenthos in the Flathead River system. The study quantified seston drift and macroinvertebrate density and diversity at five stations throughout the year (monthly). In an ongoing study, we are repeating these methods to directly compare pre- and post-treatment data.

We also predict that warmer river temperatures will increase (or alter) the availability of macroinvertebrate forage for fish. Prior to operation of selective withdrawal, we collected scale samples (in winter) from rainbow trout and mountain whitefish from several sites in the lower Flathead River. These species were chosen because of their fluvial life histories. Annual growth increments will be back-calculated for specific age classes (ages 2-4). At these ages, fish should be immature and living in the main river. In 1999-2000, we will repeat electrofishing procedures to collect our post-treatment sample.

In 1999, the Flathead River Instream Flow project will be initiated. This study will incorporate a modified Instream Flow Incremental Methodology (IFIM) application to evaluate alternatives for Hungry Horse Dam operation, particularly seasonal flow windows and ramping rates. Thermal monitoring information will be a key component of physical models. Locations of Ryan thermographs will likely also be modified to accommodate specific data needs for the study.

# WESTSLOPE CUTTHROAT TROUT ABUNDANCE ESTIMATES

## Introduction

Managers assess westslope cutthroat trout abundance through population estimates in the upper Flathead River drainage. Investigators had limited success assessing population status with standard electrofishing techniques due to high spring flows, access limitations, and wilderness restrictions. Consequently, MFWP created a population monitoring strategy for sections of the South, Middle, and North forks of the Flathead River. This strategy relies on multiple-day, hook-and-line marking runs followed by a snorkel recapture run.

## Description of the Drainage and Fishery Characteristics

### South Fork Flathead River

Zubik and Fraley (1987) described the South Fork Flathead River Drainage. The upper South Fork originates within the Bob Marshall Wilderness at the junction of Danaher and Young's Creeks and flows in a northerly direction for nearly 95 km before entering Hungry Horse Reservoir (HHR)(Figure 41). The upper 84 km of the South Fork from the headwaters to the

Spotted Bear River is classified as a wild river under the National Wild and Scenic River's Act of 1976 and downstream to HHR the South Fork is classified as a recreational river. The average annual discharge into HHR was 2301 cubic feet per second (cfs) with a maximum of 30,200 cfs and a minimum of 127 cfs (1964-1980). Hungry Horse Dam impounds the 4,403 $km^2$ South Fork drainage basin. No fish passage structures were installed in the dam which became operational in 1953. The South Fork flows for a distance of 8 km below the dam to it's confluence with the Flathead River.

Zubik and Fraley(1987) distinguished three primary fish habitat types in the South Fork Flathead River. The upper area began at the confluence of Young's and Danaher Creeks and extended downstream to Independence Park and was typified by the 2.2 km long Gordon sampling section (Figure 42) which extends from the mouth of Gordon Creek downstream to Brownstone Creek. The middle section of the South Fork begins below Independence Park and ends at Meadow Creek Gorge just north and outside of the wilderness boundary. This area is represented by the 4. 4 km Black Bear sampling section bounded by the Black Bear footbridge upstream and Black Bear Creek below (Figure 42). The downstream sampling reach begins immediately below Meadow Creek Gorge and runs downstream to the Spotted Bear River mouth. The 2.2 km Harrison sampling section typifies this area and begins at Harrison Creek and extends downstream to Cedar Creek (Figure 42).

Nutrient-poor, transparent water are characteristic of the South Fork drainage because the area is underlain by Precambrian sedimentary rock which is frequently deficient in carbonates and nutrients. The geomorphic processes that shaped the area include alpine and continental glaciation as well as fluvial and gravitational processes associated with stream dissection and structural faulting. Elevation ranges from 1085 meters at HHR during full pool, to mountain peaks exceeding 3000 meters in the wilderness. Precipitation ranges from about 75 centimeters annually near HHR to more than 230 cm on the higher mountain ridgetops. The wider valleys of the upper South Fork and the "rain shadow effect" of the Mission Mountain range result in progressively drier climates moving upriver from the reservoir.

## The Middle Fork of the Flathead River

Zubik and Fraley (1987) described the Middle Fork Drainage. The Middle Fork of the Flathead River originates at the confluence of Strawberry and Bowl Creeks at the northern end of the Bob Marshall Wilderness along the Continental Divide. From this point it flows in a northwesterly direction through the Great Bear Wilderness approximately 146 km to meet the North Fork of the Flathead River below West Glacier (Figure 43). The drainage area of the Middle Fork encompasses 2922 $km^2$ with an average annual discharge of 2956 cfs.

MFWP selected three sections of the Middle Fork within the Wilderness area to collect fisheries information. The uppermost section begins at the Gooseberry Park USFS cabin and extends downstream for 3 km to the mouth of Clack Creek (Figure 44). This section contains similar habitat and fish densities from the river’s headwaters downstream to Calbick Creek. The Schafer section of the river extends downstream from the Schafer-Dolly Varden trail ford for a distance of 3 km to a floater put-in site (Figure 44). The Schafer section represents similar fishery and habitat qualities that extend from Calbick Creek downstream to the section end. The lowest section on the upper Middle Fork is located adjacent to the USFS Spruce Park cabin and begins at the mouth of Vinegar Creek and continues down river for 3.6 km to the Spruce Park Cabin trail (Figure 44). The Spruce Park section typifies similar habitat from below the Schafer section down to Bear Creek.

From Bear Creek to where it meets the North Fork, the river flows for 70 km mainly through a steep canyon, except for the Nyack Flats area where the floodplain is up to 3 km wide. This lower portion of the Middle Fork is classified as a recreational river and is outside Wilderness boundaries. The Middle Fork drops an average of 0.31 percent along this lower portion.

We selected one section outside the Wilderness area to evaluate the fishery. The Paola section extends from the USFS boat access at Paola Creek downstream for 3.2 km to the mouth of Muir Creek (Figure 44). This section represents similar habitats that extend from Bear Creek to the upper end of Nyack Flats near the mouth of Nyack Creek.

## North Fork Flathead River

The North Fork drainage was described by Graham et al. (1980). The North Fork of the Flathead River originates in the Rocky Mountains of British Columbia, Canada and flows south across the U.S. and Canadian border into Montana. The North Fork crosses the boundary at an elevation of 1201 m and flows approximately 92 km south to it's confluence with the Middle Fork immediately above Blankenship Bridge located between the towns of West Glacier and Coram, Montana (Figure 45). The upper portion of the river flows through a broad, glaciated valley approximately 12.9 km wide and was classified in 1976 as a Scenic River under the National Wild and Scenic River's Act.

The only cutthroat trout monitoring section for the North Fork is located 22 km south of the border and is designated the Ford section (Figure 46). The section begins at the USFS floater access at Ford and extends downstream for 6.4 km to immediately above the mouth of Whale Creek.

## Flathead River Forks Fishery Characteristics

Westslope cutthroat trout, bull trout, and mountain whitefish are the native gamefish species found in the South, Middle, and North Forks of the Flathead River and their tributaries. Three distinct life history forms of westslope cutthroat trout commonly occur within the forks of the Flathead River. Adfluvial cutthroat trout spend one to three years in tributaries before emigrating as juveniles to a lake or reservoir. They generally reside in a lake or reservoir system for one to three years, mature and return to their natal stream for spawning. Cutthroat trout exhibiting this life history form generally occur in the lower South Fork up to Meadow Creek Gorge, and in the

Middle and North forks. Fluvial westslope cutthroat trout are found primarily in the main stem of the South Fork above Meadow Creek Gorge, and portions of the Middle Fork. These fish have a similar life cycle except they grow and mature in a river rather than a lake or reservoir prior to spawning in their natal stream. The resident form of westslope cutthroat trout completes it's entire life cycle solely in headwater tributaries to all three Flathead River forks. Resident cutthroat trout seldom reach lengths greater than 200 mm, whereas fluvial and adfluvial fish may attain lengths up to and exceeding 450 mm.

Bull trout appear to be primarily of the adfluvial life history in the Flathead River forks. At this time we have not observed evidence of fish residence in tributaries for complete life cycles. We have observed all age classes during summer river surveys, which may be evidence of a fluvial component.

## Methods

To allow comparisons between forks, we developed a single method for use in all population estimates. We conducted surveys during similar time periods in July or August, recognizing similar flow conditions and the return of adult westslope cutthroat trout to the river from tributaries after spawning. We used a mark and recapture sample design to assess fish abundance and size distribution. To conduct the estimates, we captured and released cutthroat trout by angling with dry flies. Small cutthroat trout less than 254 mm in length (TL) were marked with a blue Floy crustacea tag; fish measuring 254 to 305 mm received a numbered and addressed red Floy or red crustacea tag; fish greater than 305 mm received a numbered yellow Floy or yellow crustacea tag. Generally, in the river reaches where we lacked fish movement information, we utilized the marked Floy anchor tags on fish greater than 254 mm. If movement information was no longer required in a particular section, we only used crustacea tags which have a shorter retention time and are less obtrusive. Crustacea tags were needle inserted under the flesh in the anterior rays of the dorsal fin. Floy anchor tags were placed at the posterior attachment of the dorsal fin, on a longitudnal axis with the fish. After measuring and marking, fish were released within the stream feature where they were captured. Angling times were recorded to develop catch-per-effort. We marked cutthroat trout for two to three days until previously caught and marked fish comprised a portion of the total daily catch.

In the afternoon of the third or fourth day we conducted the recapture run by downstream snorkeling. To estimate the population size by snorkeling, we used the total number of angler caught fish as the number of marked fish at large (M) and then snorkel observations to estimate the ratio of tagged (R) to untagged (C) cutthroat trout for each size class. The number of experienced snorkelers was dependent on water clarity, underwater visual distance, and river width. The visual distance was the length at which the size-class and species could no longer be determined. Snorkel counts were conducted mid-day during optimal light condition. Snorkelers recorded the number and size-class of marked and unmarked cutthroat trout on diving slates. Divers floated in designated lanes to survey all available habitats. Generally, there was a diver near each bank and two to three divers spread across the remaining channel width. Frequent stops at riffle breaks were necessary to maintain a relatively even line of snorkelers throughout the section length. Other fish species observed were also recorded.

To estimate the total population for the section, we added all snorkel lane counts and utilized the Adjusted Petersen Estimate technique (Ricker 1975). In addition, we calculated mean length, length range, percent size composition, and catch rate for all fish handled during the marking runs.

Age and growth rates of westslope cutthroat trout were calculated from scales collected in 1985, 1986, 1987, and 1988. Scales were taken from an area just above the lateral line posterior to the insertion of the dorsal fin and anterior to the insertion of the anal fin. Cellulose acetate impressions of scales were examined on a microfiche reader. Distances from focus to annuli were measured to the nearest millimeter and recorded. Age and growth information was analyzed using the FIRE 1 computer program described by Hesse (1977) and the AGEMAT program designed by MFWP personnel. Body length-scale radius relationships were most accurately described using log-log plots constructed from pooled samples of South Fork cutthroat trout.

## Results and Discussion

### South Fork Flathead River

Beginning in the uppermost (Gordon) section of the South Fork, we conducted estimates in 1984 and 1987 (Table 14). In 1984, techniques were still being developed and the population estimate combined all size groups of westslope cutthroat trout. In 1987, cutthroat trout were divided into two groups, trout less than 254 mm and those greater than 254 mm. Estimates combining all fish were quite similar between the two years with 206 (±62) and 183 (±37) cutthroat trout, respectively. Catch data from cutthroat trout in the Gordon and Youngs/Danaher confluence area indicate that a higher proportion of large fish inhabit the upper river during July and August, with over 50 percent of the cutthroat trout surveyed larger than 254 mm (Table 15). Large cutthroat trout tend to reside in this portion of the South Fork at least until fall and then seek preferred habitat for overwintering. Mean lengths and catch rates were consistantly the highest in the Youngs and Danaher Creeks confluence area and in the Gordon section when compared to other South Fork sections and streams (Table 15). From 1960-1996 the mean catch rate was 7.2 cutthroat trout per hour. Mean lengths of angler caught fish ranged from 243 to 289 mm during the 1985 to 1996 period.

After 1987, estimates were discontinued in the Gordon section and the Black Bear section was selected as the long-term monitoring section for the upper and middle portions of the South Fork. This limits our capability to compare the Gordon section estimates with other sections because estimates were only conducted in 1984 and 1987.

Population estimates in the Black Bear section began in 1983 and were conducted at least once every four years through 1998 (Table 14). Over the period, the Black Bear Section consistently contained the highest estimated number of cutthroat trout per kilometer of the South Fork sections. When combining all sizes of cutthroat trout, the Black Bear estimates ranged between 346 and 641 fish per kilometer. The mean number for the period was 473 fish per kilometer.

Cutthroat trout less than 254 mm made up 75 percent of fish numbers in the section, followed by 17 percent 254-305 mm fish and eight percent fish greater than 305 mm based on combined estimates for all years (Figure 47). Combining all sampling dates, the mean number of cutthroat trout less than 254 mm was 353 per kilometer. The number of small cutthroat trout was highest during 1983 and 1985 (494 and 419 per kilometer, respectively). Since then numbers have decreased to the 1998 density of 232 per kilometer. It appears that the Black Bear section is more conducive to rearing small fish than the Gordon section and consequently their numbers were higher. Numbers of mid-sized cutthroat trout (254-305 mm) also showed variation and peaked in abundance in 1992 at 151 per kilometer (Table 14, Figure 47). The mean number of 254-305 mm cutthroat trout was 83 per kilometer for the six year period. Numbers of large cutthroat trout (>305 mm) were quite low but remained stable with a mean of 38 per kilometer for the period. The lowest number ocurred in 1989 with 31 per kilometer, and the highest in 1992 with 51 per kilometer (Table 14, Figure 47).

Mean lengths for cutthroat trout in the Black Bear Section have ranged between 213 and 274 mm for the years sampled (Table 15). Catch rates in the Black Bear Section were variable, ranging from 1.7 to 6.3 fish per hour. Catch rates overall average 4.3 fish per hour for the entire period which ranked it second to catch rates in the Gordon Section.

Estimated cutthroat trout numbers in the Harrison Section were generally lower than in the Black Bear Section (Table 14). For the sampling period, 86 percent of the estimated population were less than 254 mm in length. The proportion of small cutthroat trout in the population fluctuated from a low of 186 in 1985 to a high of 443 in 1996 (Table 14, Figure 48). For the five years sampled, small cutthroat trout abundance averaged 258 per kilometer.

Mid-sized (254-305 mm) cutthroat trout in the Harrison Section comprised roughly 10 percent of estimated fish abundance when averaged over all years. Their numbers have remained relatively stable but have ranged from 15 to 62 fish per kilometer, averaging 31 per kilometer (Table 14, Figure 48).

For all sampled years, large cutthroat trout (>305 mm) averaged only four percent of the estimated population in the Harrison Section. Their estimated numbers were very low in 1984 (four per kilometer) and since then they have increased to more constant levels averaging 13 per kilometer over the period (Table 14, Figure 48).

The mean lengths of cutthroat trout in the Harrison Section varied considerably over the years, however the fish were consistently smaller than those in other surveyed sections (Table 15). Catch rates in this section average 3.5 fish per hour over the period, which were the lowest of the

three South Fork sections.
During recent estimates on the South Fork (since 1989), we recorded the incidence of hooking scars on all fish handled during marking runs. In the Black Bear Section scars were first detected on one percent of the small cutthroat trout in 1995. In 1998 this value increased to three percent. For mid-sized cutthroat trout in the same years, only two percent had detected scars in 1989. In 1998 the rate increased substantially to 11 percent. For large cutthroat trout, two percent of the total number handled had scars in 1989, while no scars were noted in 1992, 11 percent had scars in 1995, and 19 percent had scars in 1998.

Hooking scars in the Harrison Section were recorded for 1990, 1993, and 1996. Four percent of the small cutthroat trout had scars in 1990, none in 1993, and four percent in 1996. For mid-sized fish, we found seven percent with scars in 1990, three percent in 1993, and a large increase to 21 percent in 1996. Similar percentages were observed in 1990, 1993, and 1996 for the cutthroat trout larger than 305 mm with seven percent, zero percent, and 21 percent having scars, respectively.

Westslope cutthroat trout were quite vunerable to angling and we see signs of increased angler use on this fishery. In the South Fork, angler use is directly related to the ease of access. The Gordon and Black Bear sections are about 12 and 25 miles, respectively, within the Bob Marshall Wilderness. Private and outfitted floater use has steadily increased. The Harrison section is outside the wilderness adjacent to a forest road and is more accessable. Fishing regulations have progressively become more restrictive. In 1984, special regulations were enacted that allowed anglers to harvest only three cutthroat trout under 12 inches in length per day from streams above Hungry Horse Reservoir and in the Bob Marshall Wilderness Complex. From Meadow Creek Bridge to the Spotted Bear footbridge (encompasses the Harrison section) fishing is restricted to catch and release with artificial lures.

## South Fork Westslope Cutthroat Trout Age and Growth

Pooled scale samples taken from the upper, middle, and lower areas of the river expressed the mean growth rates for cutthroat trout in the South Fork. From the 251 samples analyized, cutthroat trout exhibited the following mean lengths when back calculated to annulus formation: Age I-54 mm; Age II-109 mm; Age III-171 mm; Age IV-251 mm; Age V-321 mm; Age VI-344 mm. We did not observe fish older than six years in the sample.

## South Fork Westslope Cutthroat Trout Movement

May (1988) found that cutthroat trout tagged in the South Fork above Meadow Creek Gorge exibited little movement during summer months, 1985 to 1987. Approximately 76 percent of 81 adult fish moved less than two kilometers between the initial marking location and recapture site. Five fish were recaptured more than one kilometer upstream from where they were tagged with the maximum distance moved about 35 kilometers. The remainder of the fish (16) moved downstream. One cutthroat trout tagged at the confluence of Youngs and Danaher Creeks in July of 1986, was recaptured at Gorge Creek in May 1987; a downstream movement of 66 kilometers. A total of three fish tagged in the upper South Fork were later recaptured downstream in the Meadow Creek Gorge area. May (1988) also noted that only 18 percent of the tags returned from adult cutthroat trout indicated a movement of more than 10 kilometers. Seventy-two percent of the tags from juvenile fish exhibited less than one kilometer of movement. May (1988) thus concluded that most of the cutthroat trout tagged above Meadow Creek Gorge were fluvial fish moving short distances in the South Fork and did not migrate from Hungry Horse Reservoir. The three adult fish recaptured in the Gorge area indicate that there was some limited downstream movement between the upper and lower South Fork.

In recent population estimates in the South Fork, we have also seen limited movement based on tag returns. Occasionally we capture a fish that was Floy tagged in a previous year in the same area. We therefore assume that cutthroat trout above Meadow Creek Gorge are generally a separate population with a fluvial life history, while cutthroat trout below the Gorge are both fluvial and adfluvial fish, some utilizing Hungry Horse Reservoir.

## Middle Fork Flathead River

Estimates conducted in the Middle Fork Flathead River are summarized in Table 16. In the uppermost (Gooseberry) section, we observed an increasing trend in total cutthroat trout abundance when comparing 1988 (77/km), 1991 (102/km), and 1994 (127/km)(Figure 49). This trend primarily reflected the number of small (<254mm) cutthroat trout, which represented 96 percent of the sample in this section. The number of mid-sized and large cutthroat trout has remained stable but abundance was extremely low over the same period (Table 16, Figure 49). The mean length of cutthroat trout ranged from 174 mm to 191 mm for these years (Table 17). Catch rates have fluctuated between 2.0 and 3.7 fish per hour, and averaged 3.1 fish per hour. We believe that cutthroat trout in the upper reaches, including the Gooseberry section, are primarily resident fish, spending their entire life in or near the survey section.

Two estimates were conducted in the Schafer Section (1988 and 1994)(Table 16, Figure 50). The estimated number of small cutthroat trout increased dramatically from 37 per kilometer in 1988 to 148 per kilometer in 1994. Larger cutthroat trout were present in extremely low numbers during 1994. Small cutthroat trout made up 98 percent of the total population for those years. The catch rate for 1994 was 1.4 fish per hour. Limited data suggest that fish in the Schafer section were primarily resident and fluvial stocks.

Estimates have been conducted for two years (1997 and 1998) in the Spruce Park section (Table 16, Figure 51). Field crews partially completed a survey during 1980. A higher proportion of larger fish were present in this section than in upstream sections with 67 percent less than 254 mm, 25 percent between 254-305 mm, and 8 percent greater than 305 mm in length. Catch data from 1980, 1997, and 1998 indicate similar size and composition for all years (Table 17).

Estimates in the Paola Section were conducted anually from 1995 through 1997 to establish a baseline data set. Abundance of small cutthroat trout in the Paola section appeared to increase steadily over the three years (Table 16, Figure 52). Both mid sized and larger cutthroat trout abundances were low in all three years. Small, mid-size, and large cutthroat trout comprised 72 percent, 20 percent, and eight percent of fish numbers, respectively, for the three years. The average catch rate of 1.2 for the period was lower than all other Middle Fork sections (Table 17).

During estimates in the Gooseberry and Schafer sections, little information was kept regarding hook scars and we presume very few if any were observed. During 1997 in the Spruce Park section, we found four percent of cutthroat trout larger than 305mm contained hook scars and no incidence of scars in the other size categories. During 1998, eight percent of the small fish (<254 mm), 12 percent of the mid-sized (254-305 mm), and nine percent of the large size group (>305 mm) cutthroat trout had hook scars. In the Paola section we found six percent of the mid sized fish and 11 percent of the larger fish had scars during the 1995 survey. In 1996, seven percent of the mid-sized fish had hook scars with no observed scars in the other size groups. In 1997 scars were only apparent on four percent of the cutthroat trout <254 mm and no incidence of scars in other size groups.

As in the South Fork, there are restrictive regulations (daily harvest limit of three cutthroat trout under 305 mm) applying to the rivers and streams in the wilderness portion of the Middle Fork. In 1998, cutthroat trout limits for North and Middle fork waters outside of wilderness boundaries, main stem Flathead River, and Flathead Lake were restricted to catch and release only (Appendix A). Glacier National Park regulations are the same for the North and Middle forks, however two cutthroat trout may be harvested daily from all other waters within the Park, including Lake McDonald.

## Middle Fork Westslope Cutthroat Trout Movement

We compiled movement information from 16 tag returns of cutthroat trout tagged in recent abundance estimates in the Spruce Park and Paola sections of the Middle Fork (Table 18). The majority of tag returns (63 percent) were recaptured within the same area where fish were initially marked. Four fish from the Paola section were caught in the same area within a month of marking. The remaining six fish were all recaptured nearly one year later in the same area where they were marked. Where these fish resided during the time period between being tagged and recaptured is not known. We can only conclude that they prefer these respective areas during summer months.

Recapture locations for the other six marked cutthroat trout showed all exhibited downstream movement. Two fish marked in the Spruce Park section moved downstream 66.8 km and 69.5 km between August and October of 1997. The other fish marked in the Spruce Park section in August of 1997 was captured in September of 1998, 83 km downstream at the confluence of the North and Middle Forks of the Flathead River. Two fish from the Paola section moved down the Middle Fork 42 km, up McDonald Creek in Glacier National Park another 3.7 km, and into Lake McDonald between August and September of the same year. The greatest movement was exibited by fish marked in the Paola section that moved down the Middle Fork 49 km, down the main stem Flathead River 52 km, and then 2 miles up into Brenneman Slough. This fish was marked in August and was caught by an angler in March of the following year.

Cutthroat trout below the Schafer section appear more migratory in nature than those in above sections, suggesting the presence of all three life history forms within the Middle Fork. Our tag returns documented that a significant proportion of cutthroat trout migrate downstream for winter and returned to the same areas for summer months. Lake McDonald appears to be utilized by some Middle Fork cutthroat trout. Graham (1980) documented cutthroat trout migrating upstream from Flathead Lake into the Middle Fork as well. Future radio telemetry surveys using cutthroat trout will provide additional movement information.

## North Fork Flathead River

Results from three years of population estimates for the Ford section are shown in Table 19 and Figure 53. From 1990 to 1996, overall cutthroat trout numbers dropped dramatically from 282 to 96 per kilometer. Small (<254 mm) cutthroat trout comprised 94 percent of total cutthroat trout abundance with mid-size representing five percent and large cutthroat trout only one percent. The majority of the decline occurred in the small cutthroat trout with mid and large size fish maintaining low numbers in all three years. Catch data for the Ford section demonstrated an increase in the average size (from 192mm to 214mm) and a steady decrease in catch rates (6.0 to 4.0 fish per hour) (Table 20).

During the 1996 estimate, incidence of hook scars were recorded for all captured fish. We observed scars on two percent of the small cutthroat trout, 18 percent of the mid-size fish, and 25 percent of the large cutthroat trout. This was the highest incidence of hook scars in any of the surveyed sections in the Flathead River drainage.

There were no movement data obtained during estimates on the Ford section. However, Graham (1980) documented considerable cutthroat trout migration to and from Flathead Lake. From this and other work, all three life history forms (resident, fluvial, and adfluvial) of cutthroat trout most likely exist in the North Fork and its tributaries.

In 1990, MFWP developed special cutthroat trout regulations between the Canadian border and Polebridge to determine if harvest was impacting the number of large fish in this section. For a period of four years the regulation was a daily harvest of five cutthroat trout <12 inches, or four <12 inches, and one >20 inches, using artificial lures only (Appendix A). Cutthroat trout are rarely observed in lengths greater than 20 inches in the Flathead River drainage. This regulation was not popular with the public and did not drastically increase the number of large fish over the sample period. However, mean length, size range, and percentages of fish ≥254 mm increased from 1990 to 1996 estimates. However, lower fish abundance may have influenced values. The regulation was dropped in 1994. In 1998, MFWP placed catch-and-release regulations on cutthroat trout in the North Fork, as well as the Middle Fork, main stem Flathead River and Flathead Lake.

## FLATHEAD RIVER WINTER TROUT ABUNDANCE

### Introduction

Salmonids using the Flathead River have diverse life history strategies, making it difficult to assess the status of populations. Mountain whitefish, westslope cutthroat trout, and bull trout have both fluvial and adfluvial life histories, while rainbow trout appear to be primarily fluvial. Within a species, individual fish of one life history are generally not visually distinguishable from those of another life history. Determining population status for these species is difficult due to the timing of seasonal migrations and overlapping habitat use by the different life histories. Adfluvial westslope cutthroat trout use the main stem river and North and Middle forks as a migratory corridor. Adults migrate to and from spawning tributaries from early winter through summer, while juveniles migrate from rearing streams toward the lake from early summer through winter (Shepard et al. 1984, Likness and Graham 1988). Similarly, juvenile bull trout emigrate from tributaries to the Flathead River and Lake system from early summer through winter. In early summer (April-July), adult adfluvial bull trout migrate from the lake into the river and move toward staging areas. They then move into spawning tributaries generally in August and following spawning in September, they move rapidly back downstream to Flathead Lake (Shepard et al. 1984). Adult mountain whitefish also make spawning migrations as the fall spawning period approaches and rainbow trout adults move in response to spring spawning. Thus, at any time of the year, different salmonids, life histories, and age groups are migrating throughout the river system. These migrations compromise general assumptions of mark-recapture methodologies and complicate standardizing the timing of annual monitoring surveys. This is especially true for the native westslope cutthroat trout and bull trout.

From 1979-1981, catch per unit effort (CPUE) electrofishing surveys were conducted in three sections of the Flathead River (McMullin and Graham 1981). In an effort to assess fish populations and avoid the above constraints, monitoring efforts were spread out over an extended time period (months) to encapsulate the migration periods. Past methods attempted to describe the relative abundance of these fishes and specific size groups at a number of different times throughout the year. It was believed that repeated sampling (biweekly) would account for annual variation in the timing of seasonal migrations. In winters of 1997 and 1998, we followed past methods and conducted CPUE surveys to assess changes over the last two decades in westslope cutthroat trout, rainbow trout, and bull trout abundances. We then compared these results to those observed in other monitoring indices to help determine the efficiency of river electrofishing. We chose February and March to best describe adult adfluvial cutthroat trout and rainbow trout abundance based on results from previous surveys. In addition, we felt bull trout and juvenile cutthroat trout catch may help describe fluvial components of the populations.

### Methods

We followed the methodology of previous surveys (McMullin and Graham 1981). We sampled two sections of the Flathead River; the Kalispell section (2.95 km) near U.S. Highway 2 Bridge and the Columbia Falls section (2.0 km) near the Montana Highway 2 Bridge. Surveys were conducted at two-week intervals. We began sampling after sunset and continued until we completed two passes on each bank (four passes in total) in the section per night. We electrofished from a jet boat rigged with fixed-boom anodes. The Coffelt M22 produced straight DC at 3 to 5 amperes. McMullin and Graham (1981) did not specify the wave form or type and power levels used during electrofishing sampling. Most likely, a pulsed DC waveform (60 Hz per second) was used. In recent years, MFWP has established electrofishing policy which dictates use of straight DC or pulse rates $\leq$ 30 Hz per second when sampling waters with native fishes. This variance in methodology could affect CPUE comparisons between the two sampling periods. Passes began at the upstream boundary of each section and progressed down one of the banks. We netted all trout, measured total length and weight, and collected scales and genetic samples from cutthroat trout and rainbow trout. In 1997, river flows were regulated at 9,500 cfs for the first of three surveys and at 12,400 cfs for the fourth. In 1998, river flows were lower at 3,500 to 4,500 cfs. Marking rainbow trout allowed us to complete a mark-recapture abundance estimate (Schnabel multiple census) in the Columbia Falls section (Ricker 1975).

We calculated CPUE in two ways. In the first, used by McMullin and Graham (1981), CPUE was calculated as the number of a fish species or size group captured divided by the time (hr) spent electrofishing and the length of the sample section (km). McMullin and Graham (1981) graphically displayed CPUE values. We estimated values from figures and, therefore, these values are the best available to compare with the 1997 and 1998 calculated values. The second method used to calculate CPUE was to divide the number of a fish species or size group captured by the time (hr) spent electrofishing. Catch per hour was reported only for rainbow trout in the 1980s report.

We collected genetic samples to assess the level of hybridization between rainbow and westslope cutthroat trout. At both the Columbia Falls and Kalispell sections, we partially clipped fins from 25 trout, which were randomly picked from our collection of rainbow trout and cutthroat trout. From the Columbia Falls section, we also selected 10 samples from fish which appeared to be hybrids. The Wild Trout and Salmon Genetics Lab (University of Montana) analyzed samples using vertical polyacrylamide gel electrophoresis of nuclear DNA fragments.

## Results and Discussion

### Comparisons of Rivers Sections in 1997 and 1998 Surveys

In both 1997 and 1998, mountain whitefish was the most numerous species (hundreds per night) in both river sections. We also observed but did not enumerate largescale suckers, which were relatively lower in abundance. In 1997, we captured four species of trout and char: rainbow, westslope cutthroat trout, bull trout, and brook trout. Three brook trout (all <200 mm) were captured in the Columbia Falls section. In 1998, we captured the four trout and char species mentioned above and one lake trout (450 mm), which came from the Kalispell section. We captured five brook trout (152-250 mm) in the Columbia Falls section and one brook trout (211 mm) in the Kalispell section. In 1998, we also captured five lake whitefish (368-460 mm) in the two sections and in the Kalispell section, we also caught one redside shiner.

Rainbow, westslope cutthroat trout and bull trout dominated trout and char catch. In 1997, we caught 315 individuals of these species. The Columbia Falls section accounted for 74 percent of the catch. In 1998, we caught a total of 714 rainbow, westslope cutthroat trout, and bull trout, of which 53 percent came from the Kalispell section. We are uncertain why catch was over two times greater in 1998 than 1997, but lower river discharge in 1998 leading to increased efficiency in electrofishing or differences in the timing of spawning migrations may be responsible. Comparing 1997 and 1988 CPUE (#/hr) for each of the three species by river section, at the 95 percent confidence level, there were significant increases in catch rates for westslope cutthroat trout and bull trout in the Kalispell section. There were no significant differences in CPUE between years in the Columbia Falls section or for rainbow trout in the Kalispell section. Westslope cutthroat trout mean CPUE (#/hr) was over seven times greater in 1998 than in 1997 for the Kalispell section (Table 21). In both years, there was a high percentage of recaptured (marked) rainbow trout in the Columbia Falls section. For example, on the final survey nights in 1997 and 1998, 32 and 40 percent of captured rainbow trout (>200 mm) had fin clips from earlier surveys, respectively (Table 22). Future surveys would be needed to assess trends in population abundances or relate variation in catch to river discharge or other factors.

In 1997, mean CPUE for rainbow trout was higher in the Columbia Falls section than in the Kalispell section (Table 21). In both river sections, rainbow trout dominated catch followed by westslope cutthroat trout and then bull trout in 1997. In 1998, this pattern partially changed. Bull trout remained the least abundant of the three species in both sections and in the Columbia Falls section, mean CPUE for rainbow trout was again the highest (Table 21). However, in the Kalipsell section mean CPUE for westslope cutthroat trout was greater than rainbow trout CPUE values and also greater than mean CPUE for westslope cutthroat trout in the Columbia Falls section.

## Westslope Cutthroat Trout

Abundance of adult (>300 mm) westslope cutthroat trout in the main stem Flathead River was greatest in late winter in 1979-1980 (McMullin and Graham 1981). In 1997, there was not a significant difference, at the 95 percent confidence level, in cutthroat trout CPUE (#/hr) between the two river sections (Table 21), although juveniles (<300 mm) appeared more abundant in the Columbia Falls section. In 1998, CPUE (#/hr) was significantly greater in the Kalispell section for both juveniles and adults (Table 21).

In the Columbia Falls section, CPUE for adult cutthroat trout was relatively consistent over the four sampling dates in both years, ranging from 1.20 to 1.41 fish/km/hr in 1997 and from 0.00 to 0.66 fish/km/hr in 1998 (Figure 54, Tables 23 and 24). Between sampling dates, juvenile cutthroat trout CPUE varied widely in this section in both years ranging from 0.26 to 1.99 fish/km/hr and from 0.47 to 1.68 fish/km/hr in 1997 and 1998, respectively (Tables 23 and 24).

In the Kalispell section, CPUE for adult cutthroat trout was also relatively consistent in both years with the exception of the last sampling date in 1997 (Table 23, Figure 55). Adult CPUE ranged from 0.38 to 1.33 fish/km/hr and from 1.69 to 3.80 fish/km/hr in 1997 and 1998, respectively (Tables 23 and 24). Juvenile CPUE in the Kalispell section varied little in 1997, ranging 0.00 to 0.45 fish/km/hr, and more widely in 1998, ranging from 1.18 to 5.89 fish/km/hr.

Comparisons of CPUE (#/km/hr) for adult cutthroat trout between the early 1980s surveys and the late 1990s did not exhibit an obvious change in abundance. For the Columbia Falls section, CPUE appeared to be lower in 1997 and 1998 than 1981; however, the March 1980 values are similar to the 1990s values (Figure 54). Similarly, for the Kalispell section, CPUE (#/km/hr) for adults in 1997 appeared to be lower than most previous surveys; however, the 1998 survey had higher CPUE than observed in any of the 1980s surveys (Figure 55). Comparing mean CPUE (#/km/hr) for all sizes of cutthroat trout between 1980s and 1990s surveys did not show changing trends in either of the sampled river sections (Figures 56 and 57).

We caught cutthroat trout in a wide range of sizes, ranging from 150 to 480 mm and from 75 to 548 mm (TL) in 1997 and 1998, respectively. Length frequency charts in both years showed two peaks (Figures 58 and 59); one from 175 to 275 mm and another centered around 400 mm. We captured few fish less than 200 mm. The smaller sizes captured were juvenile fish either migrating through the river system toward the lake or residing in the river. The larger size peak was associated with the spawning migration of adfluvial adults from Flathead Lake.

In 1998, we examined every captured trout for the incidence of hooking scars and external deformations of mouth parts. Cutthroat trout showed a high incidence of scars. For all sizes, 21 percent of cutthroat trout had deformities while for adults (>300 mm) the incidence was 26 percent. These percentages were similar for fish visually identified as hybrids of cutthroat trout and rainbow trout (genetic analysis determined that field identification was correct). Westslope cutthroat trout were highly vulnerable to angling pressure as indicated by the high proportion of scars.

Hybridization between rainbow and westslope cutthroat trout was prevalent in the Flathead River. For the Columbia Falls section, 10 of 22 samples visually identified as either rainbow trout or hybrid were rainbow trout x westslope cutthroat trout hybrids and the remaining 12 were rainbow trout. The remaining three trout in the 25 fish sample were visually identified as westslope cutthroat trout, one was genetically determined to be a rainbow trout x westslope cutthroat trout hybrid and the other two were westslope cutthroat trout. Thus, 44 percent of the sample were hybrid trout.

In the Kalispell section, 19 of 25 samples were westslope cutthroat trout, five were hybrid rainbow x westslope cutthroat trout, and the remaining sample was rainbow trout. Thus, 20 percent of the sample consisted of hybridized trout.

Although field work was conducted in the middle of the night by artificial light, workers readily identified hybrid trout. Of the 10 visually identified hybrid samples, nine were rainbow x westslope cutthroat trout hybrids and one was a rainbow trout. With the exception of one misidentified westslope cutthroat trout, rainbow and hybrid trout were correctly separated in the field from westslope cutthroat trout.

The concentration of hybrid trout appears higher in the Columbia Falls section than in the Kalispell section. There were more rainbow trout in the Columbia Falls section and more westslope cutthroat trout in the Kalispell section. Upcoming surveys will attempt to further identify rainbow trout distribution and locate the spawning streams where hybridization is occurring.

## **Rainbow Trout**

Rainbow trout were the most numerous of all the trout and char species we captured, comprising 65 percent and 48 percent of trout and char captured in 1997 and 1998, respectively. Rainbow trout CPUE (#/hr) was significantly greater in the Columbia Falls section than the Kalispell section ($p = 0.01$ and $p = 0.02$ in 1997 and 1998, respectively) (Table 21). This was also observed in the 1980s surveys (McMullin and Graham 1981). Rainbow trout were more abundant in the Columbia Falls section than in the Kalispell section during the winter months. In the Columbia Falls section, mean CPUE (fish/km/hr) for rainbow trout in February and March was greater in the late 1990s (9 and 11 fish/km/hr) than in the early 1980s (1 and 2 fish/km/hr) (Table 21 and Figure 56). Increases in CPUE in the Kalispell section over this time period were smaller (Figure 57). Mean CPUE was less than one fish/km/hr in 1997 and was roughly one and two fish/km/hr in 1998. CPUE was relatively consistent over the four sampling dates in both 1997 and 1998 for the Columbia Falls section (Figure 60), while CPUE increased in the later surveys in the Kalispell section (Figure 61, Tables 23 and 24).

In the 2.0 km Columbia Falls section, we estimated that there were 191 (95 percent confidence interval of 145 to 285) and 194 (125 to 401) rainbow trout greater than 200 mm long in 1997 and 1998, respectively (Table 22) . We considered 95 fish/km (154 fish/mile) to be low density. For rainbow trout greater than 400 mm, we estimated 12 (95 percent confidence interval of 9 to 18) fish/km (19 fish/mile) in 1997. Caution should be used when interpreting these results since spawning migrations may have influenced rainbow distribution. These estimates may not be representative of other river reaches. We handled males which were ripe and others which were developing spawning colors. Although mean CPUE for rainbow trout in the 1990s appears to have increased in the Columbia Falls section from those of the 1980s, rainbow trout abundance remains low.

We caught rainbow trout in a wide range of sizes with good representation in many size groups (Figures 62 and 63). Rainbow trout ranged from 75 to 541 mm and 61 to 472 mm in 1997 and 1998, respectively. Rainbow trout are established and self-sustaining. Fish appeared in good physical condition and reached large sizes. Numerous larger rainbows had obvious hooking scars. For example, on the final night of 1997 sampling (Columbia Falls section), one-third of the rainbows (>300 mm) handled (8 of 24) had highly deformed mouths. In 1998, when combining all nights over five percent of captured rainbows had hooking scars with just under four percent incidence of hook scars in rainbow trout over 300 mm in length.

**Bull Trout**

In both 1997 and 1998, bull trout comprised 10 to 12 percent of total trout and char catch for both sections combined. In the Columbia Falls section, CPUE for juvenile (<400 mm) bull trout was similar for both years and ranged from 0.2 to 1.6 fish/km/hr (Figure 64). In 1981, no juvenile bull trout were captured in February or March in the Columbia Falls section; however, they were captured in the other months (McMullin and Graham 1981). In 1998, CPUE (fish/km/hr) for bull trout was significantly greater in the Kalispell section than in the Columbia Falls section (p = 0.01) and significantly greater than the 1997 CPUE for the Kalispell section (p = 0.01) (Figure 65, Table 21). In both years and sections, February and March catch rates of juvenile (<400 mm) bull trout was greater than catch rates for adults (>400 mm). Compared with CPUE of juveniles in 1981 for the Kalispell section, the 1997 juvenile values were similar while the 1998 values were higher (Figure 65).

We caught bull trout in a wide range of sizes (Figures 66 and 67). McMullin and Graham (1981) reported similar findings. Many sub-adult fish (<400 mm) migrated from spawning and rearing tributaries more than one year before capture. These fish resided year round in the river or moved between the river and lake. McMullin and Graham (1981) found juvenile bull trout (<400 mm) in the Flathead River throughout the year. This provides some evidence that a certain proportion of the bull trout population may reside for extended periods if not entirely in the river system. If so, this behavior may be very important to sustaining the bull trout population into the future in the face of recent changes to the Flathead Lake food web.

## ANGLER CUTTHROAT TROUT TAGGING PROJECT

### Introduction and Methods

In 1985, MFWP solicited anglers to participate in fisheries tagging surveys in the Flathead River Drainage (Hanzel and Weaver 1991). This project lasted two years. One angler actively tagged westslope cutthroat trout in the Flathead River. MFWP issued tagging guns and Floy anchor tags, anglers kept catch and tagging records. Fish were captured by hook and line, tagged, measured, and released. The river angler continued tagging cutthroat trout and recording catch data after the MFWP-sponsored project ended and is presently active and continues to record data and tag fish using his own personal equipment and tags. He fishes the main stem or valley section of the Flathead River and has tagged westslope cutthroat trout, rainbow trout, and bull trout. Generally, only cutthroat trout over 12 inches in length were tagged. Most cutthroat trout over 12 inches are adfluvial fish from Flathead Lake.

### Results and Discussion

From July 1985 through March 1997, the river angler caught and tagged 868 previously untagged westslope cutthroat trout of 305 mm or greater in length and fished approximately 1,531 hours

(Table 25). Throughout this time period, the angler's average CPUE were greatest from December through June (Table 25) as migratory fish move into and through the river prior to spring spawning in tributary streams.

There was a decrease in catch rate over the sampling period (Figure 68). In the years 1985 through 1991, monthly winter CPUE ranged from 0.38 to 2. 00 fish per hour. The catch rates dropped off in 1992. From 1992 to 1997, monthly winter CPUE ranged from 0.00 to 0.65 (Table 26). By comparing (t-test) the mean monthly winter (January through April) CPUE for the 1985 to 1991 period (1.07 fish/hour) with that of the 1992 to 1997 period (0.26 fish/hour), we found a significant difference in CPUE ($p < 0.0001$).

Table 25. Total number of untagged westslope cutthroat trout ($\geq$305 mm in length) caught, hours fished, and average catch rate for an angler on the Flathead River, 1985 through 1997.

| Month | Total Number Caught | Total Hours Fished | Fish Per Hour |
|---|---|---|---|
| January | 216 | 233 | 0.93 |
| February | 141 | 222 | 0.64 |
| March | 238 | 325 | 0.73 |
| April | 100 | 145 | 0.69 |
| May | 31 | 57 | 0.54 |
| June | 23 | 58 | 0.40 |
| July | 9 | 70 | 0.13 |
| August | 6 | 43 | 0.14 |
| September | 7 | 36 | 0.19 |
| October | 20 | 130 | 0.15 |
| November | 26 | 108 | 0.24 |
| December | 51 | 104 | 0.49 |
| Total | 868 | 1,531 | 0.57 |

The CPUE values for the sample period represent an index for the relative abundance of adult adfluvial westslope cutthroat trout migrating from Flathead Lake toward spawning tributaries.

The observed decreasing trend corroborates similar decreasing trends observed in other monitoring indexes noted in previous sections of this report (Flathead Lake gill-net survey) leading us to the conclusion that the adfluvial component of westslope cutthroat trout in the Flathead Lake and River System has decreased in abundance during the 1990s from higher levels in the 1980s.

Fish tagged in the main stem river were recaptured in the main stem reaches, the North Fork of the Flathead River, and in Flathead Lake.  Cutthroat trout were recaptured upstream as far as the British Columbia reaches of the North Fork and as far downstream as the south end of Flathead Lake.  No tagged cutthroat trout were recaptured in the Middle Fork of the Flathead River.  In the early 1980s, investigators documented similar migration patterns (Shepard et al. 1982). Investigators found adfluvial westslope cutthroat trout from Flathead Lake migrated into North Fork tributaries, three in British Columbia and numerous in the United States.  In the Middle Fork, adfluvial cutthroat trout were found in only Ole and McDonald creeks, although they felt further study was needed to conclusively determine adfluvial use of the Middle Fork tributaries.

# TRIBUTARY STREAM MONITORING

## STREAMBED CORING

### Introduction

Successful egg incubation and fry emergence are dependent on gravel composition, gravel permeability, water temperature, and surface flow conditions. The female bull trout begins redd construction by digging an initial pit or depression in the streambed gravel with her tail. After the spawning pair deposits eggs and sperm into this area, the female moves upstream a short distance and continues the excavation, covering the deposited eggs. The process is then repeated several more times, resulting in a series of egg pockets formed by the upstream progression of excavations. The displaced gravel mounds up, covering egg pockets already in place. After egg laying is complete the female creates a large depression at the upstream edge of the redd, which enhances intragravel flow and displaces more gravel back over the entire spawning area. Excavation of the redd causes fine sediments and organic particles to be washed downstream, leaving the redd environment with less fine material than the surrounding substrate. Weather, streamflow, and transport of fine sediment and organic material in the stream can change conditions in redds during the incubation period. Redds can be disturbed by other spawning fish, animals, human activities, or by high flows which displace streambed materials (Chapman 1988).

Redd construction by migratory bull trout in the Flathead drainage disturbs the streambed to a depth of at least 18.0 to 25.0 cm (Weaver and Fraley 1991). Egg pockets of smaller fish tend to be shallower. The maximum depth of gravel displacement is indicative of egg deposition depth (Everest et al. 1987). Freeze coring documented larger substrate particles (up to 15.2 cm) at the base of egg pockets than in overlying substrates (Weaver and Fraley 1991). These particles are likely too large for the female to dislodge during redd construction. Eggs are deposited and settle around these larger particles (Chapman 1988). Continued displacement of streambed materials by the female then covers the eggs.

Redds become less suitable for incubating embryos if fine sediments and organic materials are deposited in interstitial spaces of the gravel during the incubation period. Fine particles impede movement of water through the gravel, thereby reducing delivery of dissolved oxygen to, and flushing of metabolic wastes away from incubating embryos. This results in lower survival (Wickett 1958; McNeil and Ahnell 1964; Reiser and Wesche 1979). For successful emergence to occur fry need to be able to move within the redd, but high levels of fine sediment can restrict their movements (Koski 1966; Bjornn 1969; Phillips et al. 1975). In some instances, embryos that incubate and develop successfully can become entombed (trapped by fine sediments). Sediment levels can alter timing of emergence (Alderdice et al. 1958; Shumway et al. 1964) and affect fry condition at emergence (Silver et al. 1963; Koski 1975).

Measurements of the size range of materials in the streambed are indicative of spawning and incubation habitat quality. In general, research has shown negative relationships between fine sediment and incubation success of redd constructing salmonids (Chapman 1988). A significant inverse relationship existed between the percentage of fine sediment in substrates and survival to emergence of westslope cutthroat trout and bull trout embryos in incubation tests (Weaver and White 1985; Weaver and Fraley 1991, 1993). Mean adjusted emergence success ranged from about 80 percent when no fine material was present, to less than 5 percent when half of the incubation gravel was smaller than 6.35 mm; about 30 percent survival occurs at 35 percent fines. Entombment was the major mortality factor. Median percentages of streambed materials smaller than 6.35 mm at fry emergence ranged from 24.8 to 50.3 percent in 29 separate bull trout spawning areas sampled during the Flathead Basin Forest Practice Water Quality and Fisheries Study (Weaver and Fraley 1991). Linear regression of results against output from models assessing ground disturbing activity and water yield increases in these 29 Flathead Basin tributary drainages showed significant positive relationships (Weaver and Fraley 1991). These results demonstrate a linkage between on-the-ground activity and spawning habitat quality. This testing allowed development of models which predict embryo survival to emergence, given the percentage of material smaller than 6.35 mm in the incubation environment. We monitor bull trout spawning and incubation habitat quality by determining the percent fines in a given spawning area through hollow core sampling.

## Methods

Field crews used a standard 15.2 cm hollow core sampler (McNeil and Ahnell 1964) to collect four samples across each of three transects at each study area. We located actual coring sites on the transects using a stratified random selection process. The total width of stream having suitable depth, velocity, and substrate for spawning was visually divided into four equal cells. We randomly took one core sample in each cell. In some study areas we deviated from this procedure due to limited or discontinuous areas of suitable spawning habitat. We selected study areas based on observations of natural spawning. We only sampled in spawning areas used by adfluvial westslope cutthroat trout and bull trout. During the period of study, these fish spawned in the same general areas, so sampling locations remained similar.

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

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

We oven dried the bagged samples and sieve separated them into 13 size classes ranging from >76.1 mm to <0.063 mm in diameter (Table 27).  We weighed the material retained on each sieve and calculated the percent dry weight in each size class.  The estimated dry weight of the suspended fine material (Imhoff cone results) was added to the weight observed in the pan, to determine the percentage of material <0.063 mm.  We summed these percentages, obtaining a cumulative particle size distribution for each sample (Tappel and Bjornn 1983).

Table 27. Mesh size of sieves used to gravimetrically analyze hollow core (McNeil and Ahnell 1964) streambed substrate samples collected from the Flathead River Basin tributaries.

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

We refer to each set of samples by using the median percentage <6.35 mm in diameter.  This size class is commonly used to describe spawning gravel quality, and it includes the size range typically generated during land management activities.  We examined the range of median values for this size class observed throughout the basin.

## Results and Discussion

Field crews began core sampling some spawning areas utilized by Flathead Lake's migratory fish stocks in 1981 (Table 28). Initially, we sampled the main bull trout spawning areas in four North Fork tributaries; Big, Coal, Whale, and Trail creeks. We subsequently expanded our program to include Granite Creek, an important bull trout spawning stream in the Middle Fork Drainage and two additional spawning areas in the Coal Creek Drainage; North Coal and South Coal (Table 28). These seven spawning areas comprise our long-term data set for monitoring bull trout spawning habitat quality relative to Flathead Lake. Additional spawning areas have been sampled periodically throughout the basin but are not included in this analysis.

Recommendations resulting from the Flathead Basin Cooperative Forest Practice Study identified that fine sediment (<6.35 mm) levels exceeding 35 percent "threaten" embryo survival to emergence (FBC 1991). At 35 percent fines, survival to emergence is approximately one-third (Weaver and Fraley 1991). At 40 percent fines, survival drops to approximately one quarter and at this level, survival to emergence is considered "impaired" (FBC 1991).

When examining the streambed coring data set by individual spawning area it is obvious that all sites have had periods of high fine sediment levels (Table 28, Appendix B). Big Creek exceeded the threshold for impaired status (40 percent) during three consecutive years beginning in 1988 (Table 28). When sampling showed fine sediment levels in Big Creek's bull trout spawning area peaked at over 50 percent in 1990, survival to emergence was predicted to be less than 5 percent (Weaver and Fraley 1991). Although some recovery was suggested in 1991, this spawning area again exceeded threatened status (35 percent) in 1992 and 1993 (Table 28). The main bull trout spawning area in Coal Creek near Dead Horse Bridge has chronically had fine sediment problems. Its status has been in the impaired category three years (1982, 1987, and 1990) and threatened for ten of the past 17 years (Table 28). Although peak level samples from the Coal Creek spawning area were not as high as sampling in Big Creek indicated, the chronic presence of high levels of fine sediment may be having serious impact on the fish stocks in Coal Creek. Sampling in both North and South Coal creeks as well as Whale Creek showed high levels of fine sediment during the late 1980s (Table 28). Sampling in Trail Creek has shown fine sediment levels in this spawning area have remained more stable over time. Results exceeded threatened status only once in 1982 and approached 35 percent in 1990 and again during 1996 (Table 28). Granite Creek in the Middle Fork Drainage has shown a similar pattern of change over time exceeding impaired status during six years and threatened during two years (Table 28). This portion of the Middle Fork Drainage was strongly influenced by the 1964 flood event. Unstable soils and high precipitation zones also predominate in the upper Granite Creek watershed. This combination of geology and precipitation typically result in reduced spawning habitat quality. Figures illustrating results of annual hollow core sampling for each individual spawning area are provided in Appendix B.

Previous studies in the Flathead Basin have shown significant positive relationships between ground disturbing activity and results from hollow core sampling in spawning areas (Weaver and

Fraley 1991, FBC 1991). This means that as the amount of disturbed ground in a drainage increases, the amount of fine sediment in spawning gravel also increases. At this point in time we do not have the site specific information on land management activities necessary to assess cause and effect relationships at individual stream locations and it is not our intent to do so as this type of study was recently completed as part of the Cooperative Forest Practice Study (Potts 1991, FBC 1991). Our sampling results show that sediment sources and water yield problems have and will likely continue to cause fluctuations in fine sediment levels in streams, which strongly effect both embryo survival to emergence and juvenile rearing capacity.

Our indices of habitat quality appear to be very sensitive to flushing flows. To illustrate this sensitivity while providing an overall description of bull trout spawning habitat quality we calculated and plotted composite fine sediment levels (Figure 69). The composite percent fines is simply the average of all hollow coring results during any given year. An increasing trend in composite fine sediment level began in 1986. Fine sediment levels peaked during 1989 and 1990. This increase corresponds to the extended period of drought which spanned the late 1980s. Streamflows during this period were extremely low through fall and winter. Field crews observed dewatered bull trout spawning sites during winter surveys in 1986 (Weaver 1988). Limited snowpack resulted in only low to moderate runoff during the spring melt periods. Spring runoff in 1991 was the first normal “flushing flow” which occurred during the several preceding years. Our sampling results show a corresponding reduction in the level of fine sediment present in the main bull trout spawning areas (Figure 69). We have had good flushing flows during most spring runoffs since 1991. The improving trend in spawning habitat quality, although not continuous, is evident up through the 1997 sampling. Current conditions, as indicated by composite percent fines, are approaching the best observed during the 17 year period of record. However, bull trout embryo survival to emergence is still problematic in Coal Creek at Dead Horse Bridge.

## SUBSTRATE SCORING

### Introduction

Environmental factors influence distribution and abundance of juvenile bull trout within drainages throughout the range of the species, as well as within specific stream segments (Oliver 1979, Allan 1980, Leathe and Enk 1985, Pratt 1985, Fraley and Shepard 1989, Ziller 1992). Temperature, cover, and water quality regulate general distributions and abundances of juvenile salmonids within drainages, and juvenile presence at specific locations in a stream is affected by depth, velocity, substrate, cover, predators, and competitors. Although spawning occurs in limited portions of a drainage, juvenile salmonids disperse to occupy most of the areas within the drainage that are suitable and accessible (Everest 1973; Leider et al. 1986).

Juvenile bull trout rear for up to four years in Flathead Basin tributaries. Snorkel and electrofishing observations during past studies indicate juvenile bull trout are extremely substrate-oriented and can be territorial (Fraley and Shepard 1989). This combination of traits results in partitioning of suitable rearing habitat and a carrying capacity for each stream. We monitor substrate-related habitat potential by calculating substrate scores (Crouse et al. 1981, Leathe and Enk 1985).

Substrate composition influences distribution of juvenile bull trout and rearing capacities of nursery streams. Sediment accumulations reduce pool depth, cause channel braiding or dewatering, and reduce interstitial spaces among larger streambed particles (Megahan et al. 1980, Shepard et al. 1984, Everest et al. 1987). Juvenile bull trout are almost always found in close association with the substrate (McPhail and Murray 1979, Shepard et al. 1984, Weaver and Fraley 1991). A significant positive relationship existed between substrate score and juvenile bull trout densities in Swan River tributaries (Leathe and Enk 1985) and Flathead River tributaries (Weaver and Fraley 1991), where a high substrate score was indicative of large particle sizes and low score of embeddedness (Crouse et al. 1981). This relationship is thought to reflect substrate types favoring overwinter survival (Pratt 1984, Weaver and Fraley 1991).

A substrate score is an overall assessment of streambed particle size and embeddedness. Large particles which are not embedded in finer materials provide more interstitial space that juvenile bull trout favor. This situation generates a higher substrate score. Low substrate scores occur when smaller streambed particles and greater embeddedness limit the interstices within the streambed materials.

Linear regression of substrate scores against output from a model assessing ground disturbing activity in 28 Flathead Basin tributary drainages showed a significant negative relationship. Researchers also obtained a significant negative relationship between substrate scores and output from a model predicting increases in water yields (Weaver and Fraley 1991). These results demonstrate a linkage between ground disturbance and increased water yield and streambed conditions. Linear regression of juvenile bull trout density against substrate scores in 15 Flathead Basin streams showed a significant positive relationship (Weaver and Fraley 1991). This showed a strong linkage between streambed condition as measured by substrate scoring and actual juvenile bull trout abundance.

## Methods

Substrate scoring involves visually assessing the dominant and subdominant streambed substrate particles, along with embeddedness in a series of cells across transects. Surveyors assign a rank to both the dominant and subdominant particle size classes in each cell (Table 29). They also rank the degree to which the dominant particle size is embedded (Table 29). The three ranks are summed, obtaining a single variable for each cell. All cells across each transect are averaged and a mean of all transects in a section results in the substrate score.

Table 29. Characteristics and associated ranks for computing substrate score (modified by Leathe and Enk 1985 from Crouse et al. 1981).

| Rank | Characteristic |
|---|---|
| | Particle Size Class[1] |
| 1 | Silt and/or detritus |
| 2 | Sand (<2.0 mm) |
| 3 | Small gravel (2.0-6.4 mm) |
| 4 | Large gravel (6.5-64.0 mm) |
| 5 | Cobble (64.1-256.0 mm) |
| 6 | Boulder and/or bedrock (>256.0 mm) |
| | Embeddedness |
| 1 | Completely embedded or nearly so |
| 2 | ¾ embedded |
| 3 | ½ embedded |
| 4 | ¼ embedded |
| 5 | Unembedded |

[1] Used for both dominant and subdominant particle ranking

We scored 150 m sections using equally spaced transects. Cell width varied depending on wetted width, allowing a minimum of five evaluations for any transect. Maximum cell width was 1.0 m. Again, lower scores indicate poorer quality rearing habitat; higher values indicate good conditions.

## Results and Discussion

Field crews began collecting substrate scores in Flathead Lake rearing streams in 1984 (Table 30). Our initial efforts during 1984 and 1985 included only the Coal Creek Drainage in the North Fork of the Flathead River. Due to this limited sampling, assessment of basinwide conditions is not possible. However, by 1986 we were sampling at least six rearing streams annually which are tributaries to the North and Middle forks of the Flathead River. From 1986 on, the data set provides a better index of juvenile bull trout rearing habitat quality throughout the basin.

Recommendations resulting from the Flathead Basin Cooperative Forest Practice Study identified that substrate scores of 10. 0 or less “threatened” juvenile bull trout rearing capacity; at scores less than 9. 0, rearing capacity was considered “impaired” (FBC 1991). When examining the substrate scoring data set by individual site, the section of Coal Creek near Dead Horse Bridge fell into the threatened category between 1987 and 1991 (Table 30). Although substrate scores at this location have improved since 1991, the index section in Coal Creek remains close to the level where rearing capacity is threatened. Individually, all other sites scored higher than 10. 0 annually over our period of record. The highest substrate scores have been recorded in the North Coal and Morrison creek sections (Table 30). Figures illustrating results of annual substrate scoring for each individual section are provided in Appendix C.

Although previous studies in the Flathead Basin have shown significant negative relationships between ground disturbance and substrate score we do not have the site specific information on land management activities to assess cause/effect at individual stream locations. Our intent here is to provide an overall description of juvenile bull trout rearing habitat quality and how it has changed over the period of record. To best describe basinwide rearing habitat quality we averaged all substrate scores available during each year and plotted these composite scores (Figure 70).

As previously stated, 1984 and 1985 are not representative due to limited sampling. From 1986 through 1990 composite substrate score decline sharply. This corresponds to an extended period of drought which spanned the late 1980s. During 1988, a section of Coal Creek upstream from Dead Horse Bridge dewatered except for standing isolated pools from mid August through early September. A rain-on-snow event in the fall of 1989 was the first “flushing flow” in several years. Spring runoff in 1991 provided flushing as have several more recent spring runoffs. An improving trend in composite substrate score began in 1991 and although not continuous, this trend is evident through our most recent sampling. Current conditions as indexed by composite substrate score are approaching the highest observed to date. Juvenile bull trout rearing habitat in Flathead Lake nursery streams is presently in good condition.

## STREAM ELECTROFISHING/ JUVENILE SALMONID ABUNDANCE ESTIMATES

### Introduction

Estimation of fish population abundance is necessary for understanding basic changes in numbers, species composition and year class strength. Direct enumeration is the most accurate technique, but in most situations indirect methods must be employed. We generally use a combination of techniques in order to minimize errors. Fish populations are dynamic and may fluctuate considerably, even over relatively short periods of time, regardless of human influence.

Consequently, managers seeking to assess the effects of various activities on fish populations must understand the nature and causes of such fluctuations as fully as possible.

We developed a protocol to assess fish abundance in the Flathead Basin using electrofishing techniques (Shepard and Graham 1983). Monitoring focuses on quantifying yearly variation of fish abundance in stream sections sampled consistently year after year. We recommend using electrofishing techniques to assess fish abundance in accessible streams because:

1. The precision of electrofishing estimates can be estimated and reported, providing a measure of reliability;

2. There is less bias associated with changes in field personnel; and

3. Estimates derived using electrofishing techniques are presently better accepted by fisheries professionals.

## Methods

Through analysis of fish abundance estimation data collected during development of the above protocol and review of pertinent literature, we developed the following fish abundance monitoring guidelines:

1. In streams less than 10 cfs, use a two-pass electrofishing estimation technique. In these small streams adequate numbers of fish can be captured using a back-pack mounted generator-Variable Voltage Pulsator combination. Probability of capture (p) should be higher than 0.6 to obtain reliable results.

2. In streams 10 to 20 cfs, two-pass electrofishing estimation can be used; however, p values must be higher than 0.6. Bank shocking techniques must be used. If the p value falls below 0. 6 for a sample site, more effort (third pass) should be made instead of simply reporting the two-catch estimate.

3. In streams larger than 20 cfs, two-pass electrofishing estimation technique can be used; however p value must be higher than 0.6. Electrofish the sample section using both bank shocking equipment and backpack mounted equipment simultaneously.

Equipment needed to electrofish sample sections includes gear to block off the section, capture fish, collect information from fish and record data.

Two-pass Assumptions (Seber and LeCren 1967):

1. Probability of capture (p) is large enough to have a significant effect upon population total (N).

This assumption can be tested by computing p after two passes are complete. If p is less than 0. 5, assumption 1 probably has been violated (Junge and Libovarsky 1965) and more effort is required. We recommend p should be 0.6 or larger.

2. Probability of capture is constant. Fishing effort is the same for both catches and fish remaining after the first fishing are as vulnerable to capture as were those that were caught in the first fishing.

Assumption 2 has frequently been found to be faulty when electrofishing (Lelek 1965, Gooch 1967, Cross and Stott 1975, Mahon 1980). White et al. (1982) found if p was 0.8 or larger, two-catch estimates were reliable because failure of constant probability of capture (assumption 2) did not matter. We found that as long as p was 0. 6 or larger and stream discharge was less than 20 cfs, estimates computed using two-catch estimators were similar to mark-recapture estimates. Zippin (1958) determined that if the probability of capture (p) decreases with subsequent fishings, the estimate was an underestimate of the true population size. These estimates may still be reported, but should be used cautiously. They can be used to compare trends in population abundance, provided the same techniques are used throughout the monitoring program.

3. There is no recruitment, mortality, immigration or emigration between the times of the two fishings.

Assumption 3 can be easily met, since both electrofishing fishings take place within a single day and the section is isolated using block nets.

4. The first catch is removed from the population or, if returned alive, the individuals are marked so they can be ignored when counting the second catch.

This assumption can be met by removing the first catch from the population.

Two-pass Procedure:

We placed a braided nylon block net (12.7 mm mesh) at the lower boundary of the shocking section. When using a block net, we placed the net in the stream with the bottom edge facing upstream and place rocks on the weighted (bottom) edge of the net to hold it in position. We tied the ropes along the top edge of the net to a tree (or any available stable item) on each bank to stretching the net tight and holding it perpendicular to the flow. Rocks placed along the entire bottom edge of the net ensure no fish move past the net. Willow or alder branches cut into 1.0 to 1.5 m length on-site supported the net upright.

In streams less than 10 cfs, a backpack mounted generator - Variable Voltage Pulsator combination was used to electrofish the stream. In streams larger or equal to 10 cfs, we used the bank shocking technique. The bank shocking method was more efficient for capturing fish and should be used where possible.

We electrofished the section working from the upstream boundary down to the lower block net. We found that downstream electrofishing was more efficient than upstream electrofishing, and if two passes were needed for each catch (to provide a reliable estimate), both passes should be downstream. It is important to extend equal efforts during each pass, so that if two passes were used for the first catch, two passes must also be completed for the second catch. Mahon (1980) believed longer time periods between catches improved the accuracy of catch per unit effort estimators. For this reason, we recommend waiting a minimum of 90 minutes between fishings. During this time, work all fish captured on the first pass.

Two-Pass Estimators:

We used the following formula to estimate population number (Seber and LeCren 1967):

$$N = \frac{C_1^{\ 2}}{C_1 - C_2}$$

Where N = population size at the time of first pass

$C_1$ = number of fish $\geq$75 mm captured during first pass (by species)

$C_2$ = number of fish $\geq$75 mm captured during second pass (by species)

Variance of the estimate:

$$V(N) = \frac{C_1^{\ 2} C_2^{\ 2} (C_1 + C_2)}{(C_1 - C_2)^4}$$

Probability of capture (p):

$$p = \frac{C_1 - C_2}{C_1}$$

As stated previously, p must be $\geq$0.6 for a reliable, two-pass estimate to be made. If p $\leq$0.6, the estimate can be reported, but must be viewed with caution. If p $\geq$0.6 we completed the estimate; otherwise, more fishing effort was expended. This effort can be expended for computing a multiple estimate (by completing additional electrofishings and computing a multi-catch estimate using formulas presented in Zippin 1958).

When reporting the estimates of fish numbers computed by electrofishing, we reported the estimate, the 95 confidence interval in parentheses, the area of the section surveyed, the date, and the density and number of mortalities. When reporting two-pass estimates, report the probability of capture (p) with the estimate.

We compared these estimates by section with population estimates calculated from electrofishing during previous years to assess trends in fish abundance. The technique described by Platts and Nelson (1988) was used to assess population fluctuation. The maximum relative fluctuation ($M_s$) was defined as the percentage difference between the highest and lowest value of each population statistic relative to the lowest value:

$$M_s = \frac{X_{max} - X_{min}}{X_{min}} \times 100;$$

$X_{max}$ = largest annual value and $X_{min}$ = smallest annual value.

This statistic relates the largest observed change to the smallest observed value during the study period, and gives and indication of the magnitude of potential for change for each population statistic evaluated.

Average relative fluctuation ($A_s$) was used to describe the magnitude of change in each population statistic with respect to the mean value of that statistic over the source of the study:

$$A_s = \frac{X_{max} - X_{min}}{X_{avg}} \times 100;$$

$X_{max}$ and $X_{min}$ are as above and $X_{avg}$ = average value over the entire study period.

Total biomass ($B_t$), the estimated total trout weight, and areal biomass ($B_a$), the estimated trout weight per unit surface area, were computed as:

$$B_t = NW \quad \text{and} \quad B_a = \frac{B_t}{lw};$$

N = estimated trout population size. W = mean trout weight, l = length of the stream section, and w = mean width of the study section.

## Results and Discussion

### Big Creek

The Big Creek fish abundance section is located just upstream from the bridge crossing of Forest Road 316E, locally known as Skookoleel Bridge. Field crews have electrofished this section annually since 1986. Throughout this area the channel is unconfined and stream gradient is less than two percent. The substrate is dominantly cobble and large gravel. The habitat type here is generally riffle/run with occasional pools formed by large woody debris. The channel is highly unstable and major changes have occurred during recent high flow events. This section is in the lower end of the bull trout spawning reach; we usually observe redds in or near this section during annual index counts.

Over the past 13 years, estimates of Age I and older bull trout abundance in the Big Creek section have ranged from a high of 83±22 during 1989 to a low of 21±2 during 1997 (Table 31). During the three-year period from 1994 through 1996, the electrofishing crew did not capture enough juvenile bull trout to calculate valid estimates. The values reported for $\hat{N}$ in Table 31 during those years are the total numbers of juvenile bull trout captured during the first electrofishing pass. During the years when estimates could be calculated the average estimated abundance is 49. 4 Age I and older bull trout. Juvenile bull trout density during this period of record has ranged from 4.90 to 1.15 Age I and older bull trout per 100 $m^2$ of stream surface area (Table 31). During the ten years when estimates could be calculated juvenile bull trout density in the Big Creek section has averaged 3.02 per 100 $m^2$. Densities reported in Table 31 for 1994, 1995, and 1996 are expansions from the numbers captured during first pass electrofishing and are underestimates of actual densities.

This section is one of the largest of our index areas. Wetted width can be up to 12 m and discharge can be as high as 50 cfs. The electrofishing crew failed to obtain first pass capture efficiencies of 0.6 or greater during six of the ten years when actual estimates could be calculated (Table 31). Multiple pass estimators requiring additional electrofishing effort were employed during these years. This section is most difficult to work during high flow years due to depth in several areas with substantial cover, undercut banks, and backwater areas.

Estimated abundance and density increased from our initial year of sampling in 1986 peaking in 1989 (Table 31, Appendix D). We observed a declining trend over the next several years until in 1994 the electrofishing crew captured only four juvenile bull trout during the first pass. No additional fish were observed avoiding capture so the effort was aborted after completion of pass one. We obtained similar results during 1995 and 1996. No estimates were possible during this three-year period (1994-1996). We again captured estimatable numbers of juvenile bull trout during the 1997 effort (Table 31). During the most recent sampling, abundance appeared to be back within the range observed prior to 1994.

## Coal Creek

The Coal Creek fish abundance section is located just downstream from the crossing of Forest Road 1693, locally known as Dead Horse Bridge. Field crews have electrofished this section annually since 1982. Throughout this area the channel is occasionally confined and stream gradient is approximately 1.0 percent. The substrate is dominantly cobble and large gravel. The habitat type here is generally riffle/run with occasional pools formed by large woody debris. The channel is relatively stable; no major changes have occurred during the period of record. This section is midway in the bull trout spawning reach. We have observed redds in or near this section.

Over the past 17 years estimates of Age I and older bull trout abundance in the Dead Horse section has ranged from a high of 179±55 during 1987 to a low of 39±8 in 1995 (Table 32). During the past three years (1996-1998) the electrofishing crew did not capture enough juvenile bull trout to calculate valid estimates. The values reported for $\hat{N}$ in Table 32 during these years are the total numbers of juvenile bull trout captured during the first electrofishing pass. During the years when estimates could be calculated, the average estimated abundance is 97. 7 Age I and older bull trout. Juvenile bull trout density during this period has ranged from 11.93 to 2.60 Age I and older bull trout per 100 $m^2$ of stream surface area (Table 32). During the 14 years when estimates could be calculated, juvenile bull trout density in the Dead Horse section has averaged 6.62 per 100 $m^2$. Densities reported in Table 32 for 1996-1998 are expansions from the numbers captured during first pass electrofishing and are underestimates of actual densities.

This section is moderate in size with average wetted widths of approximately 8.0 m and discharges of 25-35 cfs during low summer flows. From 1982-1988 we employed mark-recapture estimators so no values of $\hat{p}$ are reported in Table 32. During these years we were able to determine that the two-pass estimator averaged 68 percent of the mark-recapture technique. From 1989 on, we only used two-pass techniques and all values of $\hat{N}$ reported have been standardized for comparison (Table 32). In Table 32, the 1982-1988 mark-recapture estimates were standardized by multiplying values by 68 percent. Due to the low p value in the 1991 survey, a third pass was required to produce a reliable estimate.

Estimated abundance and densities remained stable during the initial three years of monitoring then increased in 1985 (Table 32, Appendix D). Numbers and densities peaked during 1987 then we observed a gradual declining trend which has continued through the most recent sampling in 1998. No estimates were possible during the past three years (1996-1998) due to limited numbers of juvenile bull trout captured. As previously mentioned, fine sediment levels in the spawning and incubation environment have chronically been above the recommended threshold (Appendix B). The current level of juvenile abundance, combined with habitat conditions and low redd numbers, creates a major concern over the future of the bull trout stock inhabiting Coal Creek.

## North Fork of Coal Creek

The North Coal electrofishing section is located just upstream from the upper bridge crossing of Forest Road 317. Field crews have electrofished this section annually since 1982. Throughout this area the channel is stable and confined by high banks. Stream gradient is slightly over four percent and the substrate is dominated by large particle sizes. Boulders larger than 1.0 m are common. The most abundant habitat type is pocketwater with little woody debris present. No bull trout spawning occurs within this general area but redds have been documented both up and downstream from here.

Over the past 17 years, estimates of Age I and older bull trout abundance in the North Coal section have ranged from a high of 48±12 during 1984 to a low of 6±2 during 1993 (Table 33). During the past five years (1994-1998) the electrofishing crew did not capture enough juvenile bull trout to calculate valid estimates. The values reported for $\hat{N}$ in Table 33 during these years are the total numbers of juvenile bull trout captured during the first electrofishing pass. During years when estimates could be calculated, the average estimated abundance is 29.0 Age I and older bull trout. Juvenile bull trout density during this period has ranged from 4.89 to 0.63 Age I and older bull trout per 100 $m^2$ of stream surface area (Table 33). During the 12 years when estimates could be calculated, juvenile bull trout density in the North Coal section has averaged 2.69 per 100 $m^2$. Densities reported in Table 32 for 1994-1998 are expansions from the numbers captured during first-pass electrofishing and are underestimates of actual densities.

This section is moderate in size with wetted widths typically from 6.0-8.0 m and discharge of approximately 25 cfs during low summer flows. The higher gradient and large substrate size create some difficulty but in general electrofishng is relative efficient. Once fish are stunned it is easy to keep them downstream from the positive electrode. Quite a few fish are captured off the block net in this section.

Estimated abundance and densities increased during 1984 and remained relatively stable throughout the following six years (Table 33, Appendix D). A sharp decline occurred in the early 1990s and since 1994, the field crew could not capture enough juvenile bull trout in the North Coal section to calculate valid estimates. Habitat indices show that fine sediment in the spawning/ incubation environment exceeded the recommended threshold level during 1988 and 1989 (Appendix B). This spawning area is several kilometers upstream from the North Coal fish abundance section and it is difficult to tie the decline in juvenile bull trout to conditions there. Substrate scores in North Coal Creek have remained in good to excellent condition since we began monitoring them in 1984 (Appendix C).

## South Fork of Coal Creek

The South Coal fish abundance section is located approximately 2.0 km upstream from the gate on Forest Road 317. With the exception of 1986, field crews have sampled this section annually since 1985. Throughout this area the channel is unconfined and stream gradient is less than three percent. The substrate is dominated by cobble-sized material. The habitat type here is generally riffle/run with low to moderate amounts of woody debris. This area was clear-cut during the late 1970s and in several locations the channel was artificially straightened with heavy equipment. This area is highly unstable and extensive bedload movement occurs during high flows. The bull trout spawning area in South Coal Creek is several kilometers in length and is located just upstream from this section.

Over the past 14 years, estimates of Age I and older bull trout abundance in the South Coal section have ranged from a high of 62±8 during 1985 to a low of 9±2 during 1994 (Table 34). No estimates were possible in 1996 and again in 1998 due to the low number of juvenile bull trout captured. The values reported for $\hat{N}$ in Table 34 during these years are the total numbers of juvenile bull trout captured during the first electrofishing pass. During the years when estimates could be calculated, the average estimated abundance is 33.9 Age I and older bull trout. Juvenile bull trout density during this period of record has ranged from 5.91 to 0.75 Age I and older bull trout per 100 $m^2$ of stream surface area (Table 34). During the 12 years when estimates could be calculated, juvenile bull trout density in the South Coal Creek section has averaged 3.03 per 100 $m^2$. Densities reported in Table 34 for 1996 and 1998 are expansions from the numbers captured during the first pass electrofishing and are underestimates of actual densities.

This section is moderate in size with wetted widths from 5.0-7.0 m and discharge of approximately 15-20 cfs during low summer flows. Electrofishing is generally efficient; only one pool with substantial cover creates some difficulty during high flow years. Probability of first-pass capture have generally equaled or exceeded the recommended level of 0.6 assuring valid estimates (Table 34).

Estimated abundance and densities have fluctuated more in the South Coal section than in the other sections in the Coal Creek Drainage (Appendix D). This may be due to the unstable nature of the channel throughout this area. This instability results from past land management activities in the drainage. Despite this instability our habitat indices have remained at levels suggesting adequate conditions, especially in recent years. Both spawning and rearing habitat indices show that since 1994 conditions have been as good as we have observed since we began monitoring in 1985 (Appendix B and C). The current low level of juvenile bull trout abundance in the Coal Creek Drainage as a whole creates a major concern over the future of this bull trout stock.

## Red Meadow Creek

The Red Meadow Creek fish abundance section is located at the first crossing of Forest Road 115. The bridge is the center of the section which extends 75 m up and downstream. Field crews have electrofished this section during 10 of the past 16 years. Our initial survey was in 1983. Throughout this area the channel is occasionally confined by steep banks and stream gradient is approximately 2.0 percent. The substrate is dominantly cobble and large gravel. The habitat type is a combination of riffle/run and pocketwater. The channel is relatively stable with moderate amounts of large woody debris. The Red Bench fire burned over this section in 1988 and we saw a substantial increase in woody debris following the fire. This section is located at the downstream end of the bull trout spawning area in Red Meadow Creek.

During the years when we surveyed Red Meadow Creek estimates of Age I and older bull trout abundance have ranged from a high of 75±11 during 1983 to a low of 14±5 during 1998 (Table 35). During the three year period between 1994 and 1996 the electrofishing crew did not capture enough juvenile bull trout to calculate valid estimates. The values reported for $\hat{N}$ in Table 35 during these years are the total numbers of juvenile bull trout captured during the first electrofishing pass. The average estimated number of Age I and older bull trout in this section is 45.6. Juvenile bull trout density during the period of record has ranged from 7.50 to 1.04 Age I and older bull trout per 100 $m^2$ of stream surface area (Table 35). During the seven years when estimates could be calculated, juvenile bull trout density in the Red Meadow section has averaged 3.21 per 100 $m^2$. Densities reported in Table 35 for 1994, 1995, and 1996 are expansions from the numbers captured during the first electrofishing pass and are underestimates of total density.

This section is moderate in size with wetted widths of approximately 6.0-8.0 m and discharges of 15-20 cfs during low summer flows. The electrofishing crew failed to obtain first pass capture efficiencies of 0.6 or greater during the three year period between 1988 and 1990. Multiple pass techniques requiring additional electrofishing effort were employed during these years (Table 35). This was largely due to the increase in woody debris following the Red Bench fire. We did not conduct electrofishing surveys here in 1991, 1992, or 1993 and by 1994 most of the new woody debris was gone. We did not capture enough juvenile bull trout to calculate valid estimates in 1994, 1995, or 1996. We did not survey this section again in 1997, but the 1998 effort showed that juvenile bull trout abundance had rebounded slightly (Table 35).

## Whale Creek

The Whale Creek fish abundance section is located just downstream from the confluence with Shorty Creek. Field crews have electrofished this section annually since 1981 with the exceptions of 1982, 1984, 1985, 1988, and 1991, or 13 of the past 18 years. The channel in this area is occasionally confined and stream gradient is approximately 1.0 percent. The streambed substrate is dominantly cobble and large gravel. The habitat type is generally riffle/run with occasional pools formed by large woody debris. Following the spring runoff of 1997 the lower half of this section changed from a pool and tailout with large wood to a run. High flows moved most of the wood and the pool filled in with cobble/gravel. Overall this area is relatively stable and is located at the upstream end of the bull trout spawning reach. Whale Creek falls is located 1.0 km upstream and blocks upstream fish migration.

Over the past 18 years estimates of Age I and older bull trout abundance in the Whale Creek section have ranged from a high of 134±7 during 1998 to 32±10 during 1986 (Table 36). During 1997, the electrofishing crew did not capture enough juvenile bull trout to calculate valid estimates. The value reported for ℕ in Table 36 during 1997 is the total number of juvenile bull trout captured during the first electrofishing pass. Average estimated abundance over the period of record is 63.2 Age I and older bull trout (n=12 years). Juvenile bull trout density has ranged from 8.51 to 2.13 Age I and older bull trout per 100 $m^2$ of stream surface area (Table 36). Over the 12 years when estimates were completed juvenile bull trout density averaged 3.94 Age I and older fish per 100 $m^2$. The density reported in Table 36 for 1997 is an expansion from the number captured during first pass electrofishing and is an underestimate of actual density.

This section is one of the largest of our index areas. Wetted widths can be up to 13.0 m and discharge can be as high as 40 cfs. The electrofishing crew had trouble meeting the first pass capture efficiency of 0.6 during several years. Multiple pass techniques requiring additional electrofishing effort were employed during those years (Table 36). The large pool which formed the downstream portion of this section was extremely difficult to work during high flow years. However, spring flows in 1997 washed out most of the large woody debris and filled in cobble and gravel making it easier to capture fish during the past two years (1997 and 1998).

Estimated abundance and densities have fluctuated since we began monitoring here in 1981 (Table 36). A decline occurred in 1997 which may have resulted from the channel change in our section. However, the 1998 estimates are the highest on record to date and are encouraging.

Habitat quality indices show that fine sediment levels in the spawning/incubation environment reached or exceeded recommended thresholds during 1988 and 1989 but have improved since then (Appendix B). The juvenile rearing habitat index has remained in good condition throughout the period of record (Appendix C).

## Morrison Creek

The Morrison Creek fish abundance section is located approximately 1.5 km upstream from the gate on Forest Road 569 below Puzzle Creek. With the exception of 1981 and 1984, field crews have sampled this area annually over a 19-year period between 1980 and 1998. The channel meanders through alluvial material deposited during the 1964 flood. Gradient in this portion of Morrison Creek is approximately five percent and the streambed and channel area are comprised mostly of boulder/cobble substrate. Pocketwater habitat is predominant with riffle/run type scattered through the section. Active channel braiding is occurring and in recent years low summer flows have been split into several channels. Prior to 1990, there was only one area where the channel split. Bull trout spawning has been documented in the general vicinity of this section.

Over the past 19 years, estimates of Age I and older bull trout abundance in the Morrison Creek section ranged from a high of 138±9 during 1987 to a low of 16±3 during 1994 (Table 37). Field crews have captured estimatable numbers each year since our efforts began. Annual estimates average 75.4 Age I and older bull trout (n=17). Densities have ranged from 17.47 to 1.46 Age I and older bull trout per 100 $m^2$ of stream surface area (Table 37). The average density during the period of record is 8.77 Age I and older bull trout per 100 $m^2$ surface area.

This section is one of the smaller index areas with wetted widths less than 5.0 m and discharge of less than 10 cfs during low summer flows. This section is easily shocked with a single backpack electrofishing unit and we have typically obtained adequate first pass capture efficiencies. Although the braided sections take longer to work through, we generally have few problems getting valid estimates in this section.

In the past, we observed high estimated numbers and densities in the Morrison Creek section. Strongest populations occurred during the 10-year period between 1980 and 1989 (Table 37). During the spawning runs in 1987 and 1988 an upstream migration barrier occurred at stream km 5. 5. Progeny from these years would have been Age I and II fish during the 1990 estimate. The estimated number and density of juvenile bull trout in our electrofishing section at stream km 18.5 declined to extremely low levels in 1990 (Table 37). Estimated abundance rebounded in 1991 then returned to extremely low levels again in 1992 (Table 37). This pattern of high-low-high-low continued through 1996. Estimates during the past two years showed more stability but remain low. However, 1997 and 1998 estimates are higher than the four lowest years following 1990 and the barrier-related decline. The barrier was removed by USFS personnel in 1992.

Our habitat index of juvenile bull trout rearing shows that in general this portion of Morrison Creek has remained in good to excellent condition over the period of record (Appendix C). We do not index spawning and incubation habitat quality in Morrison Creek.

To assess overal juvenile bull trout abundance in tributaries to Flathead Lake we developed annual composite densities (Figure 71). This composite is simply the average of all estimates of Age I and older bull trout in the sections electrofished during any given year. As previously discussed, juvenile bull trout densities are strongly correlated with substrate scores (Weaver and Fraley 1991, FBC 1991). Densities may also be influenced by fine sediment levels in the spawning/incubation environment. Composite density began to decline during the late 1980s (Figure 71). This trend coincides with the extended drought period when both spawning/incubation and juvenile rearing habitat quality indices showed declining trends. Our indices suggest that habitat responded positively to flushing flows in the early 1990s, however composite juvenile bull trout density continued to decline through 1996 (Figure 71). It is likely that changes in the trophic dynamics of Flathead Lake began to influence bull trout abundance during the early to mid-1990s. Bull trout spawner escapement declined precipitously between 1991 and 1992 then remained stable but low for six years (see next section). During the past two years, composite density has increased even though spawner escapement was extremely low during 1992-1997 (Figure 71). This suggests better survival of these year classes due to improving habitat conditions.

## BULL TROUT REDD COUNTS

### Introduction

A reliable census 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. They also provide an index of success in regulating the fishery. Observations during past studies indicate that migratory fish populations in the Flathead System consistently use the same stream sections for spawning. Flathead Lake bull trout spawned in 28 percent of the 750 km of available stream habitat surveyed in 1978-1982 (Fraley and Shepard 1989). In the Swan River drainage, 75 percent of all bull trout spawning during 1983 and 1984 took place in 8.5 percent of the available habitat (Leathe and Enk 1985). About 70 percent of spawning in the Swan drainage during 1995, 1996, and 1997 occurred in portions of four streams, which amounted to less than 10 percent of available stream habitat (Montana Fish, Wildlife & Parks, Kalispell, unpublished file data). Bull trout spawned in 13 of 37 streams surveyed in the South Fork of the Flathead River drainage upstream from Hungry Horse Dam during 1993. Portions of eight of these, totaling less than 10 percent of the total habitat, supported 80 percent of the spawning (MBTSG 1995a, 1995b). Similar findings resulted from spawning site surveys in the Kootenai and Clark Fork River drainages (Montana Fish, Wildlife & Parks, Kalispell, unpublished file data; MBTSG 1996b, 1996c). As a result of specific spawning habitat requirements, the majority of bull trout spawning is clustered in a small portion of the available habitat, making these areas critical to bull trout production.

Field crews annually monitor the number of spawning sites (redds). These counts provided information on trends in escapement into upper basin tributaries and allowed us to choose sampling locations for other monitoring activities. Timing of salmonid spawning has likely evolved in response to seasonal changes in water temperature (Bjornn and Reiser 1991). Initiation of spawning by bull trout in the Flathead drainage appeared to be strongly related to water temperature, although photoperiod and streamflow may also have been factors (Shepard et al. 1984). Most bull trout spawn between late August and early November (McPhail and Murray 1979; Oliver 1979; Shepard et al. 1984; Pratt 1985; Brown 1992; Ratliff 1992). Spawning in the Flathead drainage (Fraley and Shepard 1989) and in Mackenzie Creek, British Columbia (McPhail and Murray 1979), began when daily maximum water temperatures declined to 9-10$^o$ C. Spawning takes place primarily at night (Heimer 1965; Weaver and White 1985), but has been observed during daylight hours (Needham and Vaughan 1952; T. Weaver, Montana Fish, Wildlife & Parks, personal communication; Russ Thurow, USFS Intermountain Research Station, personal communication).

Bull trout spawning typically occurs in areas influenced by groundwater (Allan 1980; Shepard et al. 1984; Ratliff 1992; Fraley and Shepard 1989). Such areas tend to remain open in the Flathead drainage during harsh winter conditions, while adjacent stream sections ice over or contain extensive accumulations of anchor ice. Recent investigations in the Swan River drainage found that bull trout spawning site selection occurred primarily in stream reaches that were gaining water from the subsurface, or in reaches immediately downstream of upwelling reaches (Baxter 1997).

Reaches used by spawning adults typically have gradients less than 2 percent (Fraley and Shepard 1989). Water depths at the upstream edges of 80 redds of migratory bull trout in the Flathead drainage ranged from 0.1 to 0.6 m and averaged 0.3 m; water velocities (at 0.6 of the depth below the surface) ranged from 0.09 to 0.61 m/s and averaged 0.29 m/s (Fraley et al. 1981). Similar mean depths (0.3 m) and water velocities (0.31 m/s) at migratory bull trout redds were documented in the Swan River drainage (Kitano et al. 1994).

The large sizes of migratory bull trout redds can restrict spawning potential in specific locations. Migratory bull trout redds ranged from 1.0 to 3.1 m in length (mean 2.1 m) in tributaries of the North and Middle forks of the Flathead River (n=465); width of these redds ranged from 0.8 to 1.5 m and averaged 1.1 m (Fraley et al. 1981). The largest redd observed in the Swan drainage was about 5.1 m long and 3.3 m wide (T. Weaver, Montana Fish, Wildlife & Parks, personal observation).

Areas in which redds are counted on a routine basis are called “index” areas. In some cases these index surveys continue to an upstream barrier. It is important to establish upper and lower limits of index areas. Through repeated annual index surveys we obtain valuable trend information to use in monitoring bull trout populations. Detection of trends will often require at least 10 years of monitoring index areas (Rieman and Meyers 1997).

## Methods

We conduct preliminary surveys to determine appropriate timing for final counts. Final inventories begin after we observed numerous completed redds, few adult fish, and little

evidence of active spawning during the preliminary surveys. Timing of final counts is critical, because as redds age, they lose the characteristic "cleaned" or "bright" appearance becoming more difficult to identify.
Experienced field crews conduct surveys by walking the channel within these known spawning areas. They visually identify redds by the presence of a pit or depression and associated tail area of disturbed gravel. If timing is proper, identification of redds presents little problem. We classify redds based on the following criteria:

1. Definite - no doubt. The area is definitely "cleaned" and or pit and tail area are recognizable. Not in an area typically cleaned by stream hydraulics.

2. Probable - an area cleaned that may be due to stream hydraulics but a pit and tail are recognizable, or an area that does not appear clean but has a definite pit and tail.

We call the upper boundary of the survey section pace zero and keep track of paces while walking downstream through the section. When the surveyors encounter a redd, they record its certainty class along with its location in paces from the start of the survey. Surveyors record distinct landmarks by noting the pace number at the location of each landmark. We include both classes of redds in final totals, which we compare annually as an index of spawner escapement.

During a basin-wide count all habitat which appears suitable for bull trout spawning (as described above) is surveyed. From this basin-wide survey, index areas can be identified for annual surveys. Basin-wide counts were done every 5-7 years.

## Results And Discussion

### Flathead Lake Population

Each fall field crews monitor the number of bull trout spawning sites (redds) in specific stream sections. These counts provide information on the number of adult bull trout successfully spawning in upper basin tributaries. Over the past 19 years, we have monitored high density spawning areas in four tributaries to both the North and Middle forks of the Flathead River. Fish spawning in these eight index streams have migrated upstream from Flathead Lake, where they spend their adult lives. In addition to our work in these annual index sections, we have periodically surveyed all known bull trout spawning areas presently available to Flathead Lake bull trout. Over the 19 years on record we have completed basin-wide counts during seven years. We believe that only a small percentage (<10 percent) of all bull trout spawning is unaccounted for during years when field crews complete basin-wide counts.

Historically, bull trout were one of four native salmonid species distributed throughout the Flathead Drainage. The other native salmonids are westslope cutthroat trout, mountain whitefish, and pygmy whitefish. The Flathead Lake bull trout population had access to all three forks of the Flathead as well as the other interconnected streams and rivers both above and below the lake. The downstream extent of this range was likely Metaline Falls below Lake Pend Oreille. Although bull trout had access to all of this area, their preference for colder water temperatures likely restricted their distribution and movement. For example, in larger lakes where there is surface outflow, summer/fall temperatures downstream are higher than bull trout prefer so little movement occurs. This suggests that migration of spawning bull trout from Flathead Lake up into the Swan River's warmer water below Swan Lake was minimal even prior to Bigfork Dam. Similar conditions occur below Flathead Lake, Stillwater Lake, Whitefish Lake, Big Salmon Lake, and many of the lakes in Glacier National Park. Recent genetic testing has shown the fish in Swan River tributaries are indeed distinct from those in the Flathead. It is likely that fish in Stillwater, Whitefish, Big Salmon, and Glacier Park lakes are also genetically distinct although little testing has been completed to date in the Glacier Park lakes. These populations are considered to be disjunct and are monitored separately.

Construction of Hungry Horse Dam on the South Fork of the Flathead River in 1953 blocked off an estimated 38 percent of the historic bull trout spawning and rearing areas available to Flathead Lake fish (Zubik and Fraley 1987). Bull trout presently occupying the reservoir as adults utilize tributaries to the reservoir and the South Fork upstream as spawning and rearing areas. No exchange is possible with the Flathead Lake population.

There are limited data on the bull trout spawning run out of Flathead Lake prior to the current monitoring scheme. The earliest and only comparable data on the number of spawning bull trout are from a study in the North Fork during the early 1950s. Personnel from the MFWP operated a two-way weir in Trail Creek during 1954. In addition to stream trapping activities they also conducted a complete redd count survey. Results from this work yielded an estimate of the total number of adult bull trout spawning in Trail Creek during 1954 of 160 fish (Block 1955).

During our initial years of redd counts in 1979 and 1980 field crews attempted to set up standard sections for annual counts. Our intent was to identify high density spawning areas with distinct upper and lower boundaries. Counts in these sections could be duplicated each year, allowing development of an index for comparison over time. We selected sections of four North Fork and four Middle Fork tributary streams for our annual index surveys (Table 38, Figure 72). Counts from 1979 are not directly comparable to subsequent years because of differences in the stream sections surveyed, only portions of the Trail and Morrison creeks index areas were counted and Ole Creek was not surveyed at all. The total number of redds reported in Table 38 for 1979 is lower than the true number, since the entire lengths of present index areas were not surveyed in 1979.

Redd numbers reported from 1980 and beyond are directly comparable. During the 12-year period from 1980 through 1991 the Flathead Lake index count averaged 372 redds with a range from 243 in 1991 to 600 in 1982. In comparing the number of spawners in Trail Creek during this 12-year period to the 1954 estimate for Trail Creek, we see similar numbers. As previously mentioned the 1954 estimate of total adult bull trout in Trail Creek was 160 fish. The estimated 12-year average for Trail Creek between 1980 and 1991 is 174 fish. To convert our redd numbers to total adult fish we multiplied the number of redds observed by a factor of 3.2 (Fraley and Shepard 1989). This coefficient was developed from trapping the spawning run in several Flathead Basin streams over several years and passing a known number of adults upstream. Then annual redd counts were completed upstream of each trap site and we calculated an average of

3.2 fish per redd.

A large decline in bull trout redd numbers occurred between 1991 and 1992 (Table 38, Figure 72). Indices show this change resulted from alterations in the trophic dynamics in Flathead Lake following establishment of *Mysis relicta*. Department personnel first detected *Mysis* in Flathead Lake in 1981. *Mysis* densities increased exponentially through 1985 peaking in 1986. It appears that the presence of *Mysis* enhanced Lake Superior whitefish and lake trout survival and growth. The fish community composition and species abundance changed dramatically from dominance by kokanee, bull trout, and westslope cutthroat trout to dominance by these introduced gamefish (see Flathead Lake gill-net section of this report). Since 1992, the Flathead Lake index count has averaged 129 redds ranging from a low of 83 in 1996 to a high of 187 in 1998. This represents a reduction by approximately 65 percent from the 12-year period from 1980-1991 (Table 38 and Figure 72). The North Fork index counts appear to have declined to a greater degree than Middle Fork streams (Table 38). During the 12 pre-*Mysis* years, North Fork index streams averaged 231 redds or 62 percent of the total Flathead Lake index count. Post-*Mysis* counts show closer to a 50:50 split between North and Middle fork index tributaries (Table 38).

We completed the 1997 bull trout redd counts in North and Middle fork index areas between September 25 and November 1, under optimal conditions. Based on the number of redds observed, the 1997 spawning run out of Flathead Lake again appeared below the numbers observed in the 1980s (Table 38 and Figure 72). This was the sixth consecutive year field crews reported low but relatively stable redd numbers. Despite the apparent stability during the past six years, the low number of redds observed created concern over persistence of the Flathead Lake bull trout population.

Crews completed the 1998 bull trout redd count surveys between September 17 and October 12, under optimal conditions. In the four North Fork index areas, we counted 101 redds, the highest count since the 1991 survey (Table 38). Similarly, in the four Middle Fork index areas, we counted 86 redds, the highest since 1991 (Table 38). Thus, the combined count of 187 redds in the eight index areas was the highest in the last seven years. Although the increased count appeared encouraging for bull trout persistence in Flathead Lake, the combined count is 50 percent of the 12-year (1980-1991) average (372).

Surveyors have documented bull trout spawning in 30 tributaries in the Flathead basin (Table 39). During the seven years when we completed basin-wide counts an average of 52 percent of all spawning occurred in 14 Middle Fork tributaries (annual range: 42 percent - 67 percent) while 16 North Fork streams supported an average of 48 percent of the total Flathead Lake spawning run (annual range: 33 percent - 61 percent). The Canadian portion of the North Fork on average supports 17 percent of the Flathead run (annual range: 8 percent - 24 percent) in seven streams. Observed redd numbers have ranged from a high of 1,156 in 1982 to a low of 236 in 1997 (Table 39).

When comparing our annual index counts with the basin-wide counts during the seven years on record we see that our annual index has ranged from 39 to 52 percent of the basin-wide number (Table 40). These data show an average of 45 percent of all Flathead Lake bull trout spawn in the eight stream sections in which we conduct our annual redd count surveys. It appears that the annual index counts accurately reflect basin-wide trends. However, basin-wide counts should be completed at least once every five years to assure that the index counts remain adequate.

Table 40. Basin-wide bull trout redd numbers compared with the number of redds observed in the stream sections (North and Middle fork tributaries) where annual monitoring occurs (index areas).

| | 1980 | 1981 | 1982 | 1986 | 1991 | 1992 | 1997 |
|---|---|---|---|---|---|---|---|
| Basin-wide Redd Numbers | 564 | 705 | 1,156 | 850 | 624 | 291 | 236 |
| Redd Numbers in Index Areas | 272 | 300 | 600 | 351 | 243 | 123 | 114 |
| % of Redds in Index Areas | 48.2 | 42.6 | 51.9 | 41.3 | 38.9 | 42.3 | 48.3 |

_ = 45% of all redds were in index areas
Range: 39% - 52% (n = 7 years)

The actual proportion of the adult bull trout population in Flathead Lake which spawns in any given year is unknown. This number is likely variable over time. The question is further complicated by the fact that we know some mature fish spawn every year while others spawn every other year. We also have evidence of fish which may only spawn one out of every three years. Redd count surveys provides a relative abundance index for spawner escapement and over an extended timeframe allows management agencies to assess trends and changes in the status of populations.

In summarizing the information available it appears that between 1980 and 1991 total estimated bull trout spawner escapement fluctuated between 2,000 and 4,000 fish. Limited information from the early 1950s suggests similar numbers of spawners at that time. We do not know whether the population was depressed prior to the early 1950s. Perturbations likely occurred as the spawning and rearing areas in the upper basin were developed and became more accessible. Both legal and illegal harvest influenced the number of spawning fish. In 1981, a Flathead River creel survey estimated that 41 percent of the adult bull trout in the spawning run were harvested by anglers (Fredenberg and Graham 1983). Creel limits were reduced in response (Appendix A). Construction of Hungry Horse Dam on the South Fork blocked 38 percent of the population's historic habitat (Zubik and Fraley 1987). Human population growth continues in the basin with associated pressures on the bull trout population and its habitat. A significant decline in redd numbers occurred during the early 1990s due to alteration of the trophic dynamics in Flathead Lake. From 1992 to 1997, the number of bull trout redds remained relatively stable (six years), but this level was approximately 70 percent below the average during the preceding 12-year period (1980-1991). Our 1998 count showed an encouraging increase over the previous six years but was still 50 percent below its pre-*Mysis* levels. The mechanisms causing the decline are not completely clear and there remains considerable uncertainty about bull trout ecology and trophic interactions in Flathead Lake.

There are separate bull trout populations occupying the Swan and South Fork Flathead drainages which are presently stable or increasing. There are also 27 disjunct bull trout populations in the Flathead Basin. Little is known about some of these populations. We recommend continuing the monitoring program. It provides the longest term data set on bull trout population status available anywhere. Annual index counts adequately reflect basin-wide trends in bull trout redd numbers, but basin-wide counts should be completed every five years. Future efforts should focus on the inter-specific interactions and overall ecology of Flathead Lake and the lower main stem Flathead River. Determination of population genetic structure and status of the numerous disjunct bull trout assemblages in the Flathead Basin should be a high priority in future work.

## South Fork of Flathead River Populations (Hungry Horse Reservoir):

From 1993 to 1998, we have monitored high density spawning areas in four Hungry Horse Reservoir tributaries. In addition, from 1993-97 we monitored four tributaries to the upper South Fork of the Flathead River upstream. Fish spawning in these eight streams have migrated from Hungry Horse Reservoir, where they spend their adult lives. In addition to our work in these annual index sections, we surveyed all spawning habitat available to Hungry Horse Reservoir bull trout during 1993.

Bull trout in the South Fork Flathead Drainage were part of the Flathead Lake population prior to construction of Hungry Horse Dam in 1953. This population had access to all three forks of the Flathead as well as the other interconnected streams and rivers both above and below Flathead Lake. Construction of Hungry Horse Dam blocked off an estimated 38 percent of the historic bull trout spawning and rearing areas available to Flathead Lake fish (Zubic and Fraley 1987). Bull trout which were trapped upstream from the dam have developed into the existing population. Subadults reach sexual maturity and live their adult lives in the reservoir. Spawning takes place in tributaries to the reservoir and to the river upstream from the reservoir. Juvenile bull trout rear one to four years in natal tributaries prior to moving downstream into the reservoir becoming subadults.

Within the South Fork basin there are two lakes, Big Salmon and Doctor lakes, which support bull trout populations that appear to be self-reproducing and functionally isolated from the reservoir population. These populations are considered to be disjunct. Little is known about the Doctor Lake population. In Big Salmon, fish could pass downstream into the South Fork and the reservoir, but water temperatures below the outlet of Big Salmon Lake during late summer are much warmer than preferred by bull trout and likely discourage upstream movement of spawners from the reservoir during this period.

During 1993, field crews conducted spawning site inventories in the South Fork Drainage for the first time. This initial effort was a basin-wide count where we surveyed all tributaries where bull trout spawning was suspected based on past agency reports and review of information obtained from the public. In total, we surveyed six reservoir tributaries and 28 streams in the upper South Fork Drainage. Our main goal was to obtain baseline information and identify key spawning areas for annual monitoring in future years.

Field crews counted 64 redds in the tributaries draining directly into Hungry Horse Reservoir (Table 41). Wounded Buck, Wheeler, Sullivan, and Quintonkin creeks were identified as our four annual index streams. We observed 274 redds in tributaries to the South Fork of the Flathead River upstream from the reservoir. Little Salmon, Gordon, and Youngs creeks along with the White River were identified as the annual monitoring streams in the upper basin. Field crews documented bull trout spawning in 13 streams; we observed no bull trout spawning in 21 of the 34 tributaries surveyed during 1993 (Table 41).

Based on our limited South Fork data set the annual index sections supported 85 percent of all bull trout spawning during the single year when basin-wide counts were completed. As more information becomes available we may choose to reassess our annual index area selection in order to obtain the most information for our efforts. The 1997 counts are likely to be the last year for continuous annual survey of the four upper basin index streams (Table 42). This is due to the time required and logistical problems which accompany survey work in a remote backcountry setting. Backcountry surveys will most likely occur on a three to five year basis. This should not be a problem since most of the South Fork drainage is protected in a wilderness area.

Table 42. Summary of South Fork Flathead bull trout spawning site inventories from

1993-1998 in the annual index sections.

| Reservoir Tributaries | | | | | | | Upper River Tributaries | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | | 1993 | 1994 | 1995 | 1996 | 1997 |
| Wounded Buck | 22 | 29 | 34 | 41 | 14 | 5 | Youngs | 40 | 24 | 34 | 74 | 43 |
| Wheeler | 12 | 10 | 1 | 3 | 1 | 4 | Gordon | 35 | 44 | 46 | 58 | 30 |
| Sullivan | 25 | 8 | -- | 52 | 50 | 54 | White River | 39 | 60 | 45 | 86 | 31 |
| Quintonkin | 5 | 3 | 7 | 4 | 0 | 11 | Little Salmon | 56 | 47 | 43 | 134 | 100 |
| Totals | 64 | 50 | 42 | 100 | 65 | 74 | Totals | 170 | 175 | 168 | 353 | 204 |

We completed the 1998 bull trout redd counts in the South Fork Drainage between September 15 and October 10, under optimal conditions. Based on the number of redds observed in the annual index sections, the 1998 spawning run out of Hungry Horse Reservoir was above average (Table 43 and Figure 73).

Spawning seemed to be more concentrated in 1997 and 1998 than during past years. For example, Sullivan Creek supported 70 to 80 percent of all bull trout spawning in the reservoir index tributaries. In 1997, the other three reservoir index areas equaled or set new record low counts ranging from 56 to 100 percent below the average number of redds observed over the preceding four years (Table 42). The number of redds observed in Sullivan Creek during 1997 and 1998 was much greater than in previous years. In 1997, counts in two of the upper basin index areas, Gordon Creek and White River, were record lows. Conversely, counts in Little Salmon Creek was 43 percent above the previous four years' average, while counts in Youngs Creek were average.

Data are only available from five years, making further interpretations impossible. However, it appeared that redd numbers in the reservoir index sections fluctuated to a greater degree than they did in upper basin index streams. Over the initial four years of redd counts, field crews observed an average of 280 bull trout redds in our annual monitoring sections. The 1997 total of 269 is 4 percent below this average figure (Table 43).

In light of the U. S. Fish and Wildlife Service listing of bull trout under the Endangered Species Act, it becomes necessary to expand our redd count data set to estimate the size of the adult population in Hungry Horse Reservoir. The following calculations are provided to illustrate the average number of adult bull trout present in the reservoir during the time period for which redd count data are available. The numbers generated are not to be considered as statistically valid population estimates; no confidence intervals are provided. We make a number of assumptions during calculations based on survey data from the Flathead System.

During our five year period of record we have observed an average of 278 redds in the annual index sections. We will use 280 as our starting point for the following exercise. The first assumption required to extend redd count data to total adult numbers is to adjust for the difference between redd numbers in the index sections versus redd numbers basin-wide. From the 1993 basin-wide count we estimated that 85 percent of all spawning occurred in the index areas so we increase our average index count of 280 by 15 percent obtaining 322. We then assume that 10 percent of all spawning remains unquantified during basin-wide counts so an additional 32 redds are included bringing the total to 354. To convert estimated redd numbers to total adult fish we multiply the number of redds by a factor of 3.2. This coefficient was developed from trapping spawning runs in several Flathead Basin streams over several years and passing a known number of adults upstream. Then redd counts were completed upstream from each trap site and we calculated an average of 3.2 fish per redd. This conversion results in an estimate of 1,133 bull trout in the average spawning run during the past five years.

Next, to address the question of what proportion of the adult population spawns during any given year, we made another assumption. We assumed that half of the adult bull trout spawn in any given year. To obtain an estimate of the average number of adult bull trout in Hungry Horse Reservoir during each of the last five years we simply double our estimate of potential spawner escapement which produces a value of 2,266 fish.

In summarizing the available information, it appears that between 1993 and 1997, estimated spawner escapement ranged from about 1,000 to 1,700 adult bull trout. The total adult population in Hungry Horse Reservoir was likely double this number. We do not know whether this was typical during the years from impoundment through 1993. Catch in sinking gill nets set during fall in similar locations have ranged between two and six fish per net. Netting has been conducted since 1958 and catch during recent years appears to be some of the highest recorded during this 38-year period (see Hungry Horse Gill Net Surveys in this report). This suggests a relatively stable population similar to current estimated levels.

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# Appendix B

## Substrate Scoring

**Results of annual substrate scoring for individual stream sections providing juvenile bull trout rearing for the Flathead Lake population. The bold line at the score of 10. 0 indicates the level below which rearing capacity becomes threatened (FBC 1991). At scores less than 9. 0 rearing capacity is impaired.**

# Appendix C

## Streambed Coring

**Results of annual hollow core sampling in individual bull trout spawning areas for the Flathead Lake population from 1981-1997. The bold line at 35 percent less than 6. 35 mm indicates the level above which embryo survival to emergence is threatened (FBC 1991). At over 40 percent less than 6. 35 mm, survival is impaired.**

# Appendix D

## Juvenile Bull Trout Density Estimates

### Densities of Age I and older bull trout calculated from annual electrofishing in rearing areas for the Flathead Lake population from 1980-1998.

# Appendix A

## Changes in fishing regulations for selected fish species in Flathead Lake.

Table A1. Changes in fishing regulations for daily bag limits for lake trout in Flathead Lake and River system.

| Year | Lake | River | Comment |
|---|---|---|---|
| Pre 1959 | 15 fish, not to exceed 10 lbs. and fish | Same | |
| 1959 | 10 fish, not to exceed 10 lbs. and 1 fish | Same | 1962 harvest: 1,243 |
| 1982 | 1 | 5 | *Mysis* appear 1981; lake trout harvest: 3,600 |
| 1983 | 1 | 5 | |
| 1984 | 2 (or 1 lake trout and 1 bull trout) | 5, only 1 >14" | |
| 1985 | 2 (or 1 lake trout and 1 bull trout) | 5, only 1 >14" | *Mysis* peak |
| 1986 | 5 lake trout, only 1 >28" | 5, only 1 >14" | Kokanee crashing |
| 1988 | 5 lake trout, only 1 >28" | 5, only 1 >14" | |
| 1990 | 7, only 1 >26" | 5, only 1 >14" | Lake trout show up in River |
| 1992 | 10 <26" or 9 <26" and 1 >36" | Same as Lake | Lake trout harvest: 21,656 |
| 1994 | 10 <30" or 9 <30" and 1 >36" | Same as Lake | |
| 1996 | 15 <30" and 1 >36" | Same as Lake | |
| 1998 | 15 <30" and 1 >36" | Same as Lake | |

Table A2. Changes in fishing regulations for daily bag limits for bull trout in Flathead Lake and River system.

| Year | Lake | River | Comment |
|---|---|---|---|
| 1953 | | | First spawning tributaries closed (Big, Coal, Whale, Trail creeks) |
| Pre 1959 | 15 fish, not to exceed 10 lbs. and 1 fish | Same as Lake | 18" minimum length |
| 1959 | 10 fish, not to exceed 10 lbs. and 1 fish | Same as Lake | |
| 1962 | | | More spawning tributaries closed (Granite, Morrison, Lodgepole, Long creeks) |
| 1972 | | | More spawning tributaries closed (Ole, Park, Nyack, Muir creeks) |
| 1982 | 1; 18" minimum length | Same as Lake | |
| 1985 | 1 | Same as Lake | No minimum size |
| 1988 | 1 | Same as Lake | |
| 1990 | 1 | Same as lake | Bull trout given separate limit from general trout; illegal to possess a live bull trout (high grade) |
| 1992 | 1 | Closed | Emergency closures on river system |
| 1993 | Closed | Closed | All bull trout fishing closed except Hungry Horse Reservoir and Swan Lake |
| 1996 | Closed | Closed | Bull trout fishing in Hungry Horse Reservoir closed |

Table A3. Changes in fishing regulations for daily bag limits for westslope cutthroat trout in the Flathead Lake and River system.

| Year | Lake | River | Comment |
|---|---|---|---|
| Pre 1982 | 10 fish, not to exceed 10 lbs. and 1 fish | Same | |
| 1982 | 5 | 5 | |
| 1984 | 5 | 5, only 1 >14" | |
| 1990 | 2 | 5, only 1 >14" | North Fork 5 <12" or 4 <12" and 1 >20" |
| 1994 | 2, only 1 >14" | 2, only 1 >14" | North Fork same as River |
| 1998 | Catch and release | Catch and release | |

Table A4. Changes in fishing regulations for daily bag limits for kokanee in the Flathead Lake and River system.

| Year | Lake | River | Comment |
|---|---|---|---|
| Pre 1982 | 35 | 35 | Number that would fit in a smoker |
| 1982 | 20 | 20 | |
| 1983 | 10 | 10 | |
| 1985 | 10 | 5 | Snagging closed |
| 1986 | 10 | 5 | River lure fishery develops |
| 1988 | 10 (5/1 - 11/30) | Closed | |
| 1994 | Closed | Closed | |
| 1996 | 5 (3rd Sat. in May - Sept. 15) | Closed | |
| 1998 | Standard limit (20) | Standard limit (20) | Salmon recovery halted, special regulations dropped, snagging still closed |