Glacier National Park Fisheries Inventory and Monitoring

Bi-Annual Report: 2009-2010

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Front cover photo caption: Trend gill net sampling of Lake McDonald.

Inside cover photo captions (top and bottom): Westslope cutthroat trout captured in Midvale Creek and bull trout captured in Kennedy Creek during stream population assessment.

Following photographs (clockwise from top): Burbot captured in St. Mary lake during gill net assessment, Montana Fish, Wildlife and Parks staff assisting with bull trout redd counts on Ole Creek, Blackfeet Nation Fish and Game Department staff and native northern pike collected for contaminants assessment from Sherburne Reservoir, USGS staff assisting with trend gill net sampling on Quartz Lake.

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Native Fish Population Monitoring

ABSTRACT

In 2009, Glacier National Park began development of a monitoring program for native salmonids inhabiting park streams. The intent of the program is to establish baseline abundance levels in key streams that will serve as useful benchmarks for monitoring changes in populations over time. We sampled 19 streams in 2009, and successfully conducted depletion removal estimates on 15 of them. Fish Creek (McDonald Creek drainage) had the highest density of age-1 and older wct, estimated at 12.2/100m^2. Boulder Creek (St. Mary River drainage) had the highest density of age-1 and older bull trout, estimated at 9.0/100m^2. We did not capture any fish in Sprague Creek, sampled in the vicinity of the campground. Many sites also supported tailed frogs *Ascaphus truei* and spotted frogs *Rana pretiosa*. Culverts along the Camas Road may be important isolating mechanisms protecting genetically pure wct from rainbow trout introgression. Similarly, Autumn Creek (tributary to Bear Creek) is protected by a large culvert under the railroad line. Management and conservation of native fish in the North and Middle forks Flathead River remains complicated due to the presence of migratory and resident populations of native salmonids, and expanding distribution of non-native fish species. Midvale Creek (Two-medicine River drainage) has strong densities of genetically pure wct and is seasonally-isolated by a diversion dam, but remains vulnerable to introgression with rainbow trout from downstream. Population and genetic monitoring, as well as rainbow trout removal efforts should continue on Midvale Creek, along with efforts to increase the effectiveness of the existing diversion barrier. Wild Creek (St. Mary tributary) supports one of the only remaining genetically pure wct populations on the east side of the park, yet maintains seasonal connectivity to the mainstem river. Evaluation of seasonal connectivity of Wild Creek to the St. Mary would provide valuable information when considering management and conservation needs of this population. In general, the largest threat currently facing native fish species on the east side of the park is the Milk River Irrigation Project. However, stakeholders are currently working together to address these issues and improve conditions for migrating native fish species.

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INTRODUCTION

Glacier National Park (GNP), located in northwest Montana, represents some of the most pristine and biologically diverse habitat for plants and animals found in the Intermountain West. Sitting at the core of the Crown of the Continent Ecosystem, GNP provides a diversity of stream and lake habitats for aquatic species. GNP covers over 1,000,000 acres, providing high-quality lentic and lotic fish habitat. GNP supports over 700 perennial lakes/ponds, ranging in size from less than an acre, up to Lake McDonald, covering almost 7,000 surface acres. GNP also provides over 2,200 km of high-quality stream habitat for aquatic species. A diversity of native and introduced fish species inhabit park waters (Tables 1 and 2).

Table 1. Native (N) and introduced (I) salmonids in Glacier National Park.

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Arctic grayling *Thymallus arcticus*	--	--	I
Brook trout (ebt) *Salvelinus fontinalis*	I	I	I
Bull trout (blt) *S. confluentus*	N	--	N
Kokanee *Oncorhynchus nerka*	I	--	I
Lake trout *S. namaycush*	I	--	N
Lake whitefish *Coregonus clupeaformis*	I	--	N
Mountain whitefish (mwf) *Prosopium williamsoni*	N	N	N
Pygmy whitefish *P. coulteri*	N	--	N
Rainbow trout (rbt) *O. mykiss*	I	I	I
Westslope cutthroat trout (wct) *O. clarkii lewisi*	N	N	N
Yellowstone cutthroat trout *O. c. bouvieri*	I	I	I

GNP encompasses the headwaters of three major ocean drainages (Figure 1). The western portions of the park drain into the Pacific Ocean via the Columbia River, the southeastern portions of the park drain into the Atlantic Ocean via the Mississippi River, and the northeastern portions of the park drain into the Arctic Ocean via the Hudson Bay Drainage.

Table 2. Native (N) and introduced (I) non-salmonids in Glacier National Park.

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Fathead minnow *Pimephales promelas*	--	--	--
Northern pikeminnow *Ptychocheilus oregonensis*	N	--	--
Peamouth *Mylocheilus caurinus*	N	--	--
Redside shiner *Richardsonius balteatus*	N	--	--
Longnose sucker *Catostomus catostomus*	N	N	N
Largescale sucker *C. macrocheilus*	N	--	--
White sucker *C. commersoni*	--	N	N
Deepwater sculpin *Myoxocephalus thomsoni*	--	--	N
Mottled sculpin *Cottus bairdi*	--	N	N
Slimy sculpin *C. cognatus*	N	--	--
Shorthead sculpin *C. confusus*	N	--	--
Spoonhead sculpin *C. ricei*	--	--	N
Burbot *Lota lota*	--	--	N
Northern pike *Esox lucius*	--	--	N
Trout-perch *Percopsis omiscomaycus*	--	--	N

In order to effectively manage fishery resources and understand how landscape level changes impact these resources, data on species abundance and distribution is needed. Limited historic (Read et al. 1982, Weaver et al. 1983) and contemporary data (Mogen and Kaeding 2004, Dux and Guy 2004, Muhlfeld et al. 2009, D'Angelo and Muhlfeld 2009) exist relative to native fish distribution and abundance in flowing waters across the park. Montana Fish, Wildlife and Parks completes an annual depletion population estimate on Ole Creek, a tributary to the Middle Fork Flathead River to monitor juvenile bull and westslope cutthroat trout abundance over time (Weaver et al. 2006), but such efforts are limited across the park. Much of the data collection effort to date has been focused in the Flathead River drainages in the park, although significant effort has also been focused on describing the distribution of bull trout in the St. Mary River drainage on the east side of the park. Developing a set of "index" population estimate sections spread across the park would be extremely valuable to monitor changes over time in fish community abundance in response to expanding non-native fish species populations and climate change.

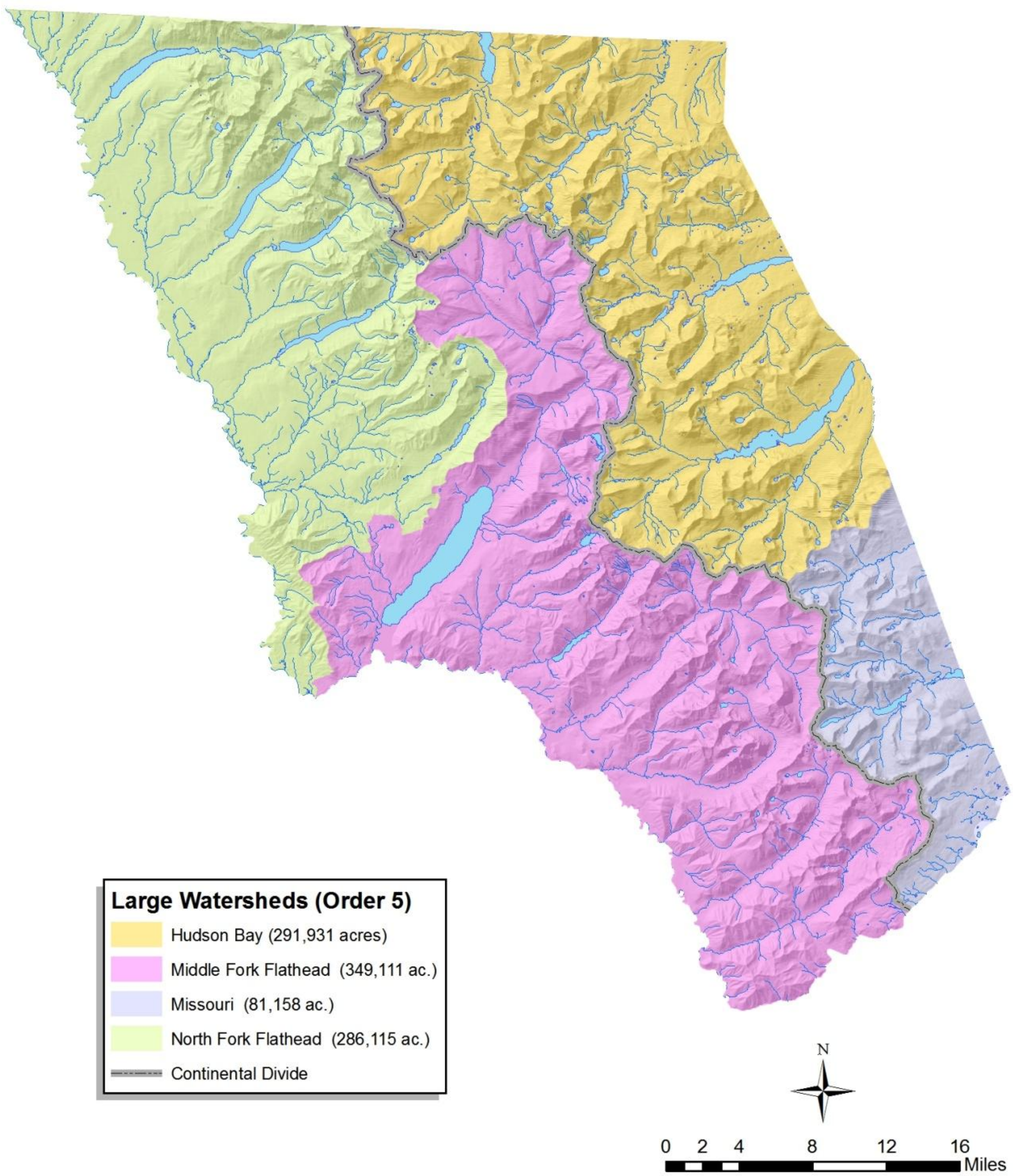

Figure 1. Major watersheds of Glacier National Park, Montana.

METHODS

We used the removal (depletion) method (Zippin 1958) to estimate the abundance and size structure of fish populations in the streams surveyed. We selected 19 streams for initial survey based on their previous sampling history, species composition, and accessibility (Figure 2, Table 3). We desired to sample streams that had been previously sampled, had relatively strong native fish populations, and could be sampled by a two-person crew in one day. The intent was to identify a group of core streams to monitor across the park to characterize native fish population trends into the future.

Removal estimates involved identifying a representative reach of stream approximately 100 m long, and placing a 1.22 m X 9.15 m X 6.35 mm block net (minnow seine) at the downstream end of the section to minimize the likelihood of fish moving into or out of the section. The upstream end of the section was located at a high gradient riffle break or other natural drop in the stream channel bed. We used a Smith-Root model 15-B battery powered electrofisher, using pulsed DC current to capture the fish. Settings were adjusted to use the minimum amount of power required to capture fish while minimizing fish injury. Settings were generally set at 30 htz., 3 ms pulse width, and between 400 and 700 volts, depending on stream temperature and conductivity. A two to three person crew sampled moving downstream, carefully working back and forth across the channel to effectively sample the entire reach. Repeated downstream passes were made through the section until the catch on the most recent pass was reduced to 20% or less of the catch on the first pass for age-1 and older juvenile salmonids. Due to the cold water temperatures generally found in the park, the timing of our sampling, and previous length-frequency data, we assumed bull trout (blt) ≥60 mm and westslope cutthroat trout (wct) ≥55 mm were age-1 and older for estimation of abundance and catch-per-unit effort (CPUE). This was confirmed by examining the length-at-capture data from our 2009 field sampling. In addition, in smaller, streams similar to those of headwater areas of GNP, wct fry may not even emerge from the gravel until mid-August (Scarnecchia and Bergersen 1986, Downs 1995). Scarnecchia and Bergersen (1986) indicated few cutthroat trout from headwater systems in Colorado exceeded lengths of 30-35 mm before the entered their first winter. Therefore using a lower limit (i.e. 55 mm) as a cutoff for inclusion in estimates of age-1 and older wct is more appropriate for most park waters containing rearing wct. Fish of the size can be efficiently sampled with electrofishing gear.

Age-0 bull and westslope cutthroat trout, as well as sculpin were not included in the estimation of abundance or CPUE because sampling efficiency was lower for these fish, and we did not attempt to net all of these fish encountered. Population estimates were calculated in the software program Microfish. We derived density estimates by dividing the population estimate by the product of the mean wetted width and the length of the sampling reach. We also estimated first-pass CPUE as fish captured/100m^2 and fish captured/hr of electrofishing to facilitate comparison with other sampling efforts occurring in the park.

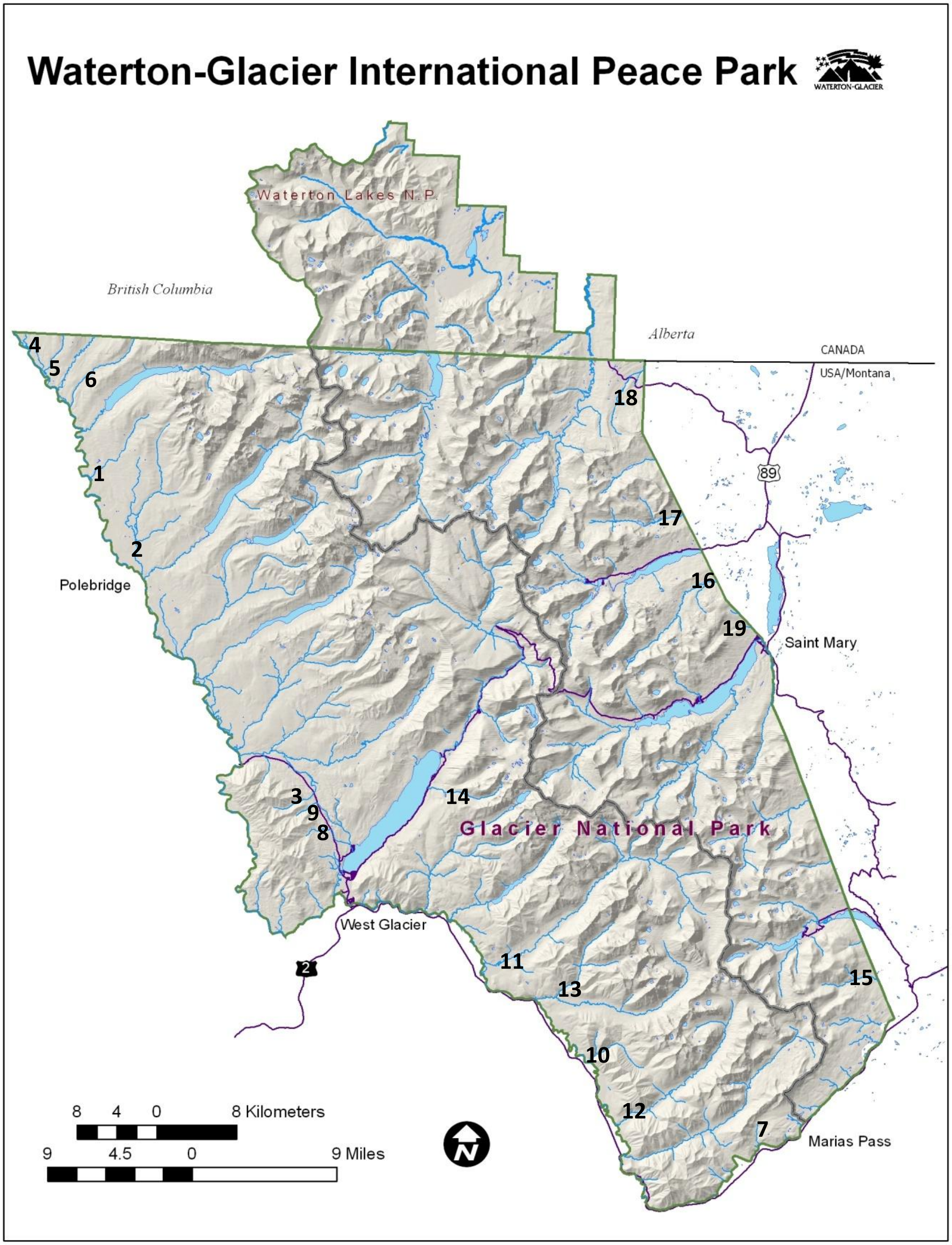

Figure 2. 2009 stream sampling sites for depletion population estimates, Glacier National Park, Montana. Site numbers from Table 3.

GPS coordinates of the upstream and downstream end of the sections were recorded in UTM using the WGS 84 datum, which was later converted to the NAD 83 datum for compatibility with NPS databases. Digital photographs were taken of the upstream and downstream ends of each section, and wetted widths were measured at approximately 20 m intervals to calculate the wetted stream area sampled. Channel gradient was estimated in percent with a clinometer by taking multiple measurements (four or more) of slope in each study reach and averaging them. Stream temperature was recorded in Celsius with a handheld thermometer and electronic temperature recorders. Conductivity, dissolved oxygen, and pH were measured using Extech Exstick II meters. The meters were calibrated according to the manufacturer's specifications periodically, or when field readings indicated it was necessary. Dominant and sub-dominant substrate sizes for the entire reach were estimated visually, using six general size classes of bed material: bedrock, boulder, cobble, gravel, sand, and silt. The presence and relative abundance of amphibians was noted as present-common or present-rare.

Table 3. Stream name, site location, and date sampled for 2009 sampling sites.

Major drainage	Stream name (Site No.)	Location of downstream end of reach (WGS 84, UTM)	Date sampled
N. Fk. Flathead	Akokala (1)	11U 0699251N, 5408441W	8/3/09
	Ford (2)	11U 0694157N, 5417724W	7/20/09
	McGhee (3)	11U 0718256N, 5386976W	7/10/09
	Sage (4)	11U 0685847N, 5430045W	7/28/09
	Spruce (5)	11U 0688239N, 5427453W	7/27/09
	Starvation (6)	11U 0695225N, 5428464W*	7/29/09
M. Fk. Flathead	Autumn (7)	12U 0320621N, 5352007W	7/15/09
	Fern (8)	11U 0720216N, 5383473W	7/9/09
	Fish (9)	11U 0718665N, 5386000W	7/10/09
	Muir (10)	12U 0303790N, 5360495W	7/23/09
	Nyack (11)	12U 0295448N, 5370575W	8/25/09
	Park (12)	12U 0307841N, 5355236W	8/20/09
	Pinchot (13)	12U 0300366N, 5366970W	8/10/09
	Sprague (14)	12U 0287757N, 5387725W	7/16/09
Missouri	Midvale (15)	12U 0332437N, 5368436W	7/22/09
St. Mary (Hudson Bay)	Boulder (16)	12U 0314860N, 5408095W	8/12/09
	Kennedy 1 (17)	12U 0309571N, 5414152W	8/11/09
	Kennedy 2	12U 0310481N, 5414456W	8/17/09
	Lee (18)	12U 0307481N, 5428586W	7/21/09
	Wild (19)	12U 0319740N, 5403793W	7/15/09

* Top of section

Fish were anesthetized, identified to species, measured (total length; mm) and weighed (g). Fulton-type condition factor was calculated for age-1 and older wct and blt (Anderson and Neumann1996). In addition, genetic samples were collected from wct for future analysis. Fish were allowed to recover their equilibrium and were released back into the stream.

RESULTS AND DISCUSSION

North Fork Flathead River Drainage

Akokala Creek

Akokala Creek is a third order tributary to the North Fork of the Flathead River. The lower reaches of Akokala Creek are known to support migratory westslope cutthroat trout spawning (Muhlfeld 2009b), while the upper reaches of the drainage support "disjunct" migratory populations of wct and bull trout using Akokala Lake as adult habitat (Meeuwig et al. 2007, Meeuwig 2008). Resident wct are also likely found in the upper stream portions of the watershed. Migratory wct from Flathead Lake or the mainstem Flathead River reproduce in the lower gradient stream reaches from the time of peak streamflow through the descending limb of the hydrograph (Muhlfeld 2009b). Although hybridization has not been found to date in juvenile westslope cutthroat trout in the creek itself (Muhlfeld et al. 2009), a recent radio-telemetry study documented hybridized westslope cutthroat trout entering Akokala Creek from the North Fork Flathead River in the spring (Muhlfeld et al. 2009b). Longbow Creek, a tributary to Akokala was sampled for westslope cutthroat genetic status in 2008, and did not show any evidence of hybridization (MFWP, unpublished data). We desired to sample Akokala Creek in the migratory spawning reach to develop an index of abundance that would reflect the health of the migratory native fish population, which are of increasing concern in the park due to reductions in abundance and increases in hybridization and competition with, and predation by, non-native species (Deleray 1999, Fredenberg 2000).

We captured wct, mwf, and sculpin in Akokala Creek. We estimated the total abundance of age-1 and older wct in the 148 m-long section at 9 (95% CI: 7-11). The density estimate for these same fish was 0.79 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 9.75 fish/hr and 0.44 fish/100m^2. Average condition factor (K) was estimated at 1.0 for age-1 and older wct (SD = 0.07) (n = 9). Average length of age-1 and older wct was 103 mm (Table 4, Figure 3). We also collected genetic samples from wct for future analysis. We employed 700 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Akokala Creek.

The length-frequency histogram for wct from the reach of Akokala Creek we sampled is truncated at 110 mm (Figure 3). This sampling reach is located approximately 1 km upstream of the confluence with Akokala Creek, in what is presumed to be migratory wct spawning and rearing habitat. Our data is consistent with this presumption. We would expect to find multiple year-classes of wct, including adult fish in the size range 150-200 mm (Downs 1995) if we were sampling a resident wct population, which we did not. Continued annual population estimates in this reach of channel will allow us to monitor juvenile recruitment in one of the remaining Flathead Lake/North Fork Flathead River migratory wct populations in the park.

Tailed frogs *Ascaphus truei* were present, but rare. Gravel was estimated to be the dominant substrate type followed by sand in the reach. Water temperature at the time of sampling was 13C. Dissolved oxygen was measured at 7.72 mg/L, pH at 8.31, and conductivity at 74.4µS. Mean channel slope was 1.7% and mean wetted width was 7.7m.

Physical habitat characteristics were comparable for this reach to those reported by Read et al. (1982), who sampled a much larger section of Akokala Creek in the same general area on 8/19/80. The authors measured conductivity at 65µS, an average channel slope of 1.4%, and bed material composition of 30% rubble, 30% gravel, 30% fines, and 10% boulder. Channel debris was noted as "low" in both studies**.** We found the same species in this reach of Akokala Creek as was found in the 1980 study, however our density estimate was lower for wct. Using snorkeling, the authors estimated 1.8 age-1 and older wct/100m^2, considerably higher than our estimate of 0.79/100 m^2 derived using a depletion population estimate. Muhlfeld et al. (2009) estimated a density of 1.0 age-1 and older wct/100m^2, using multiple pass depletion methods, similar to ours. It should be noted that these estimates were not derived using the same sampling methodology (single pass snorkeling compared to a multiple pass depletion estimate conducted downstream into a blocknets) or in precisely the same locations, but in combination the more recent data suggest lower densities of age-1 and older wct than were previously estimated. Continuation of the depletion population estimate is recommended for long-term monitoring because of the three methods previously used to estimate fish density in Akokala Creek, it is the least sensitive to changes in technique and personnel.

Table 4. Mean length (TL; mm), weight (g), standard deviation (SD), and sample size (n) of age-1 and older wct, as well as mean length and mean weight of all individuals of other species captured in Akokala Creek. Length range represents all fish captured.

Species	Mean length (SD) (n)	Length range (all individuals captured)	Mean weight (SD) (n)
wct	102.7 (10.3) (9)	51-113	11.1 (3..3) (9)
mwf	68.5 (9.2) (2)	62-75	2.5 (0.7) (2)
sculpin	108.3 (13.9) (3)	93-120	14.3 (5.7) (3)

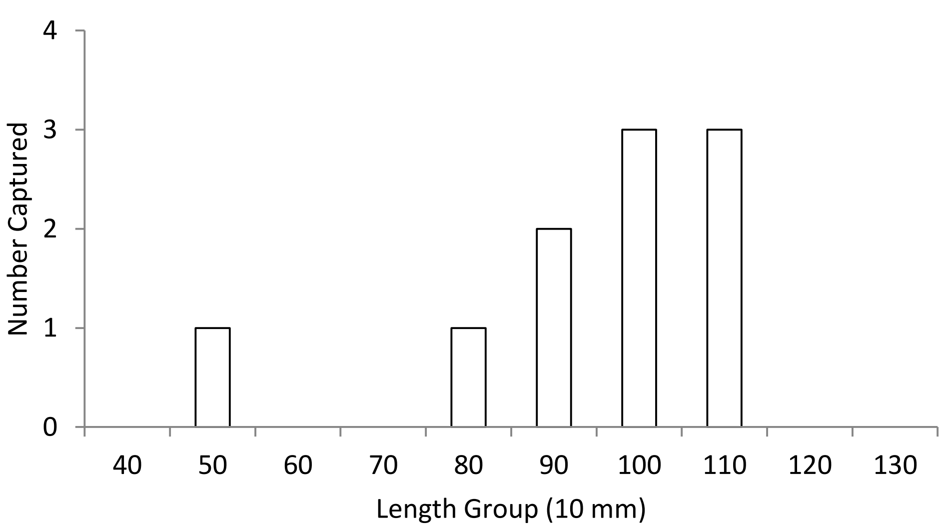

Figure 3. Length-frequency histogram for wct captured in Akokala Creek, Glacier National Park, in 2009.

Ford Creek

Ford Creek is a third order tributary to the N. Fk. Flathead River, and is located in an unroaded drainage lacking maintained trail access. The creek was previously sampled (Read et al. 1982, Muhlfeld et al. 2009), and both westslope cutthroat trout and sculpin were detected. Genetic analysis conducted by Muhlfeld et al. (2009) did not detect hybridization between wct and rainbow trout in Ford Creek.

We captured only wct in Ford Creek, and did not capture any age-0 fish although we did observe westslope cutthroat trout fry in the stream channel, which were estimated to be approximately 20 cm in length. We estimated the total abundance of age-1 and older wct in the 93 m-long section at 17 (95% CI: 12-22). The density estimate for these same fish was 4.8 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 17.2 fish/hr and 3.1 fish/100m^2. Average condition factor (K) was estimated at 1.00 for age-1 and older wct (SD = 0.10) (n = 16). Average length of age-1 and older wct was 82.3 mm (SD = 24.8, n = 16, length range = 58-160 mm) (Figure 4). We also collected genetic samples from wct for future analysis. We employed 700 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Ford Creek.

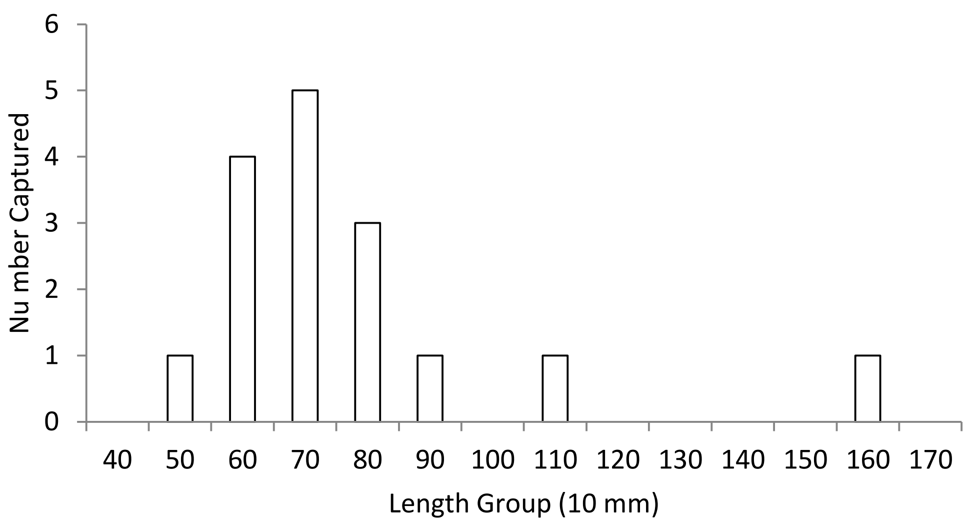

Figure 4. Length-frequency histogram for wct captured in Ford Creek, Glacier National Park, in 2009.

For comparison, using snorkeling Read et al. (1982) estimated a density of 9.8 age-1 and older wct/100 m^2 in the lower reaches of Ford Creek. He also estimated a conductivity of 105µS and a stream gradient of 3.4%. Using electrofishing depletion population estimation, Muhlfeld et al. (2009) estimated a density of 8 wct >75 mm/100m^2 in the lower reaches of Ford Creek. They sampled a reach just upstream of the Kintla Lake Road crossing, in a similar location to the reach we sampled. These estimates are not directly comparable because the precise locations and methods varied, but they do suggest a current density estimate of between 5 and 8 age-1 and older wct/100m^2 in the lower reaches of Ford Creek.

Both adult and juvenile tailed frogs were present in high abundance. Gravel was estimated to be the dominant substrate type followed by cobble in the reach. Water temperature at the time of sampling was 13C. Dissolved oxygen was measured at 8.28 mg/L, pH at 7.58, and conductivity at 81.8µS. Mean channel slope was 4% and average wetted width was 3.8m for the study reach. Muhlfeld et al. (2009) reported a maximum summer water temperature of 15.9C.

McGhee Creek

McGhee Creek is a third order tributary to Camas Creek in the N. Fk. Flathead River drainage. It is historical wct habitat, but is threatened by hybridization with rbt. Muhlfeld et al. (2009) documented hybridization between rbt and wct in Dutch Creek, a large tributary to Camas Creek. Genetic samples collected from lower McGhee Creek in 2008 indicated hybridization was occurring (MFWP, unpublished data). There are no known fish passage barriers in the drainage that would preclude advancing rbt. The culvert under the Camas Road may preclude upstream passage to the headwater reaches of McGhee Creek due to its length and drop out of the pipe, but this remains uncertain.

We captured only putative wct in McGhee Creek, and did not capture any age-0 fish due to the timing of our sampling. We did not capture any fish on our second pass, so we used the total number captured on the first pass as the estimate of population size. We estimated the total abundance of age-1 and older wct in the 76 m-long section at 13. The density estimate for these same fish was 5.3 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 39.1 fish/hr and 5.3 fish/100m^2. Average condition factor (K) was estimated at 1.13 for age-1 and older wct (SD = 0.08) (n = 13). Average length of age-1 and older wct was 121.9 mm (SD = 33.7, n = 13, length range = 80-180 mm) (Figure 5). We also collected genetic samples from wct for future analysis. We employed 500 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in McGhee Creek.

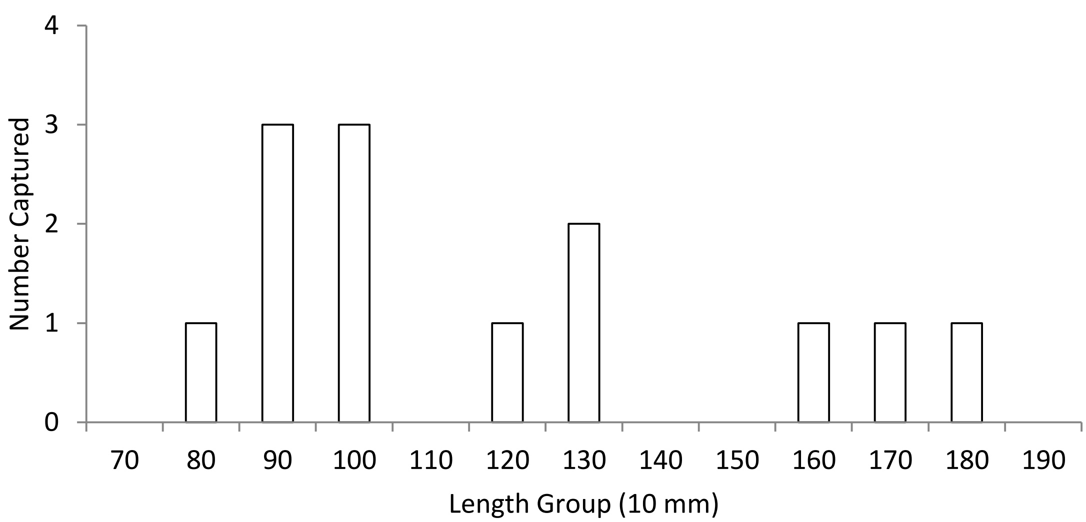

Figure 5. Length-frequency histogram for wct captured in McGhee Creek, Glacier National Park, in 2009.

Both adult and juvenile tailed frogs were present and common. Gravel was estimated to be the dominant substrate type followed by cobble in the reach. Water temperature at the time of sampling was 13C. Dissolved oxygen was measured at 8.28 mg/L, pH at 7.58, and conductivity at 81.8µS. Mean channel slope was 4% and average wetted width was 3.8m for the study reach. Muhlfeld et al. (2009) reported a maximum summer water temperature of 15.9C.

Spruce Creek

Spruce Creek is a third order tributary to the N. Fk. Flathead River. It flows from Canada into the park prior to entering the N. Fk. Flathead River near the Canadian border. Wct genetic status has not been assessed in Spruce Creek. Previous researchers have documented westslope cutthroat trout, mountain whitefish, and sculpin (Read et al. 1982).

We captured only wct in Spruce Creek, and did not capture any age-0 fish. We estimated the total abundance of age-1 and older wct in the 122.5 m-long section at 14 (95% CI: 12-16). The density estimate for these same fish was 3.1 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 16.2 fish/hr and 1.6 fish/100m^2. Average condition factor (K) was estimated at 1.0 for age-1 and older wct (SD = 0.11) (n = 14). Average length of age-1 and older wct was 138.3 mm (SD = 20.1, n = 14, length range = 107-193 mm) (Figure 6). We collected genetic samples from wct for future analysis. We employed 500 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Spruce Creek.

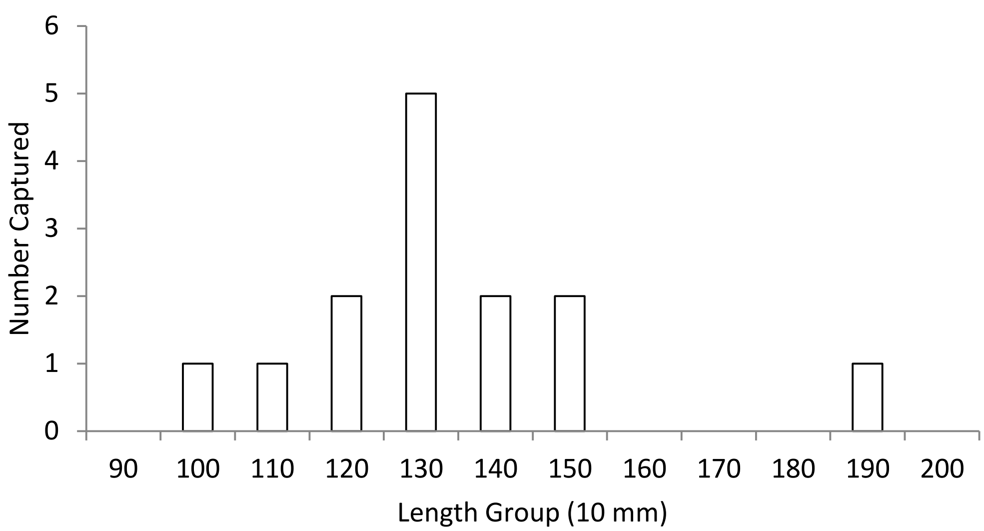

Figure 6. Length-frequency histogram for wct captured in Spruce Creek, Glacier National Park, in 2009.

For comparison Read et al. (1983) estimated 3.6 age-1 and older wct/100m^2 using snorkeling in the lower reaches of Spruce Creek. This estimate is not directly comparable to our sampling as the sites were not duplicated and methods of enumeration differed, but they represent a range of wct densities observed in Spruce Creek.

Cobble was estimated to be the dominant substrate type followed by sand in the reach. Water temperature at the time of sampling was 12C. Dissolved oxygen was measured at 7.78 mg/L, pH at 8.2, and conductivity at 95µS. Mean channel slope was 2.7% and average wetted width was 3.7m for the study reach. No amphibians were observed in Spruce Creek.

Sage Creek

Sage Creek is a fourth order tributary to the N. Fk. Flathead River (Read et al. 1982). It flows from Canada into the park prior to entering the N. Fk. Flathead River near the Canadian border. Previous researchers (Muhlfeld et al. 2009) did not detect hybridization between wct and rbt in Sage Creek. Wct, mwf, and sculpin have been previously documented in Sage Creek (Read et al. 1982).

We captured wct and sculpin in Sage Creek. Sage Creek was too large to install a blocknet and effectively deplete, and as a result we spot-shocked high quality habitat. We did not estimate CPUE as it would not have been repeatable. We captured eight age-1 and older wct and nine sculpin in one pass over approximately 330m of stream. For comparison Read et al. (1983) estimated 11.1 age-1 and older wct/100m^2 using snorkeling in the lower reaches of Sage Creek. We estimated an average condition factor (K) of 1.0 for age-1 and older wct (SD = 0.12) (n = 7). Average length of age-1 and older wct was 105.8 mm (SD = 24.8, n = 8, length range = 69-150.5 mm) (Figure 7). We also collected genetic samples from wct for future analysis. We employed 800 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Sage Creek.

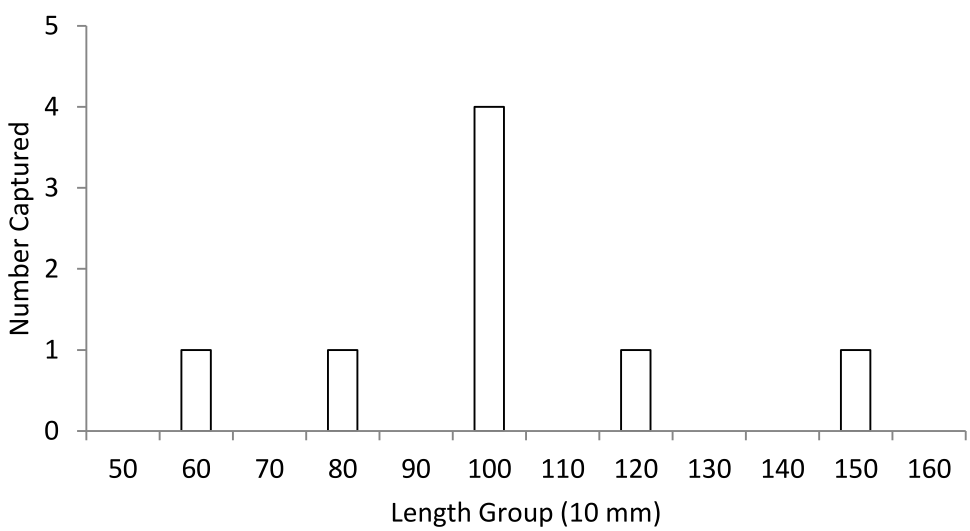

Figure 7. Length-frequency histogram for wct captured in Sage Creek, Glacier National Park, in 2009.

Cobble was estimated to be the dominant substrate type followed by sand in the reach. Water temperature at the time of sampling was 12C. Dissolved oxygen was measured at 8.03 mg/L, pH at 8.2, and conductivity at 146µS. Mean channel slope was 1.7% and average wetted width was 14.7m for the study reach. No amphibians were observed in Sage Creek.

Starvation Creek

Starvation Creek is a third order tributary to the N. Fk. Flathead River (Read et al. 1982). It flows from Canada into the park prior to entering the N. Fk. Flathead River approximately 6 km south of the Canadian border. Wct genetic samples were collected in 2008 and did not show any evidence of hybridization (MFWP, unpublished data). Previous researchers have documented bull trout, westslope cutthroat trout, mountain whitefish, and sculpin in Starvation Creek (Read et al. 1982).

We captured bull trout, westslope cutthroat trout, and mountain whitefish in Starvation Creek. We estimated the total abundance of age-1 and older wct in the 99.2 m-long section at 86 (95% CI: 73-99). The density estimate for these same fish was 11.2 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 74.1 fish/hr and 5.2 fish/100m^2. Average condition factor (K) was estimated at 1.0 for age-1 and older wct (SD = 0.20) (n = 76). Average length of age-1 and older wct was 95.6 mm (SD = 35.5, n = 76, length range = 56-200 mm) (Figure 8). Average weight of wct was 13.2 g (SD = 18.4). We collected genetic samples from wct for future analysis. We captured one adult bull trout (502 mm; 1,275g), and observed a second adult bull trout in the stream. We did not capture any juvenile bull trout. We also captured one mountain whitefish (255 mm; 159g). We employed 550 volts at 30 hertz, and a 4 ms pulse width to capture fish in Starvation Creek.

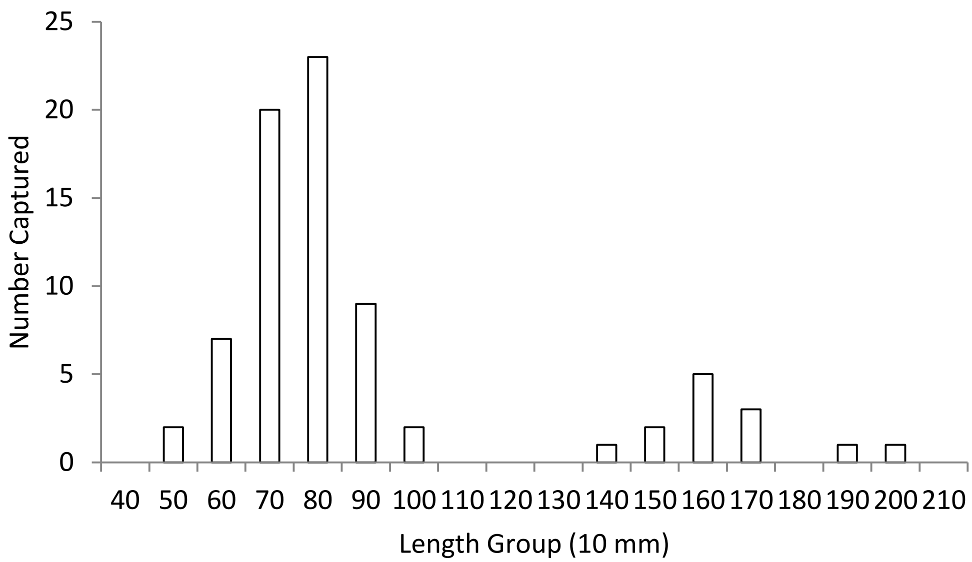

Figure 8. Length-frequency histogram for wct captured in Starvation Creek, Glacier National Park, in 2009.

Our sampling took place approximately 2 km downstream of the lower-most sampling conducted by Read et al. (1983), but was in the middle reaches of the U.S. portion of the stream. Read et al. (1983) estimated much lower densities of juvenile wct (2.0 age-1 and older wct/100m^2 using snorkeling) than we did. They also estimated a density of between 1.3 and 3.5 age-1 and older bull trout /100m^2 in the study reach, whereas we did not capture any juvenile bull trout. These estimates are not directly comparable to our sampling as the sites were not duplicated and methods of enumeration differed, but they do suggest that the wct population remains strong, and that migratory bull trout continue to use the drainage for spawning and rearing. Bull trout are known to be patchy in their distribution in streams, and it is possible that had we sampled higher in the drainage we may have captured juvenile bull trout. Future sampling should be conducted to identify key spawning and rearing reaches in the U.S. portion of the drainage.

Cobble was estimated to be the dominant substrate type followed by boulder in the reach. Water temperature at the time of sampling was 10C. Dissolved oxygen was measured at 7.8 mg/L and conductivity at 120µS. Mean channel slope was 4% and average wetted width was 7.7m for the study reach. No amphibians were observed in Starvation Creek.

Middle Fork Flathead River Drainage

Autumn Creek

Autumn Creek is a third order tributary to Bear Creek in the Middle Fork of the Flathead River Drainage. Autumn Creek is located near Marias Pass, and is isolated from Bear Creek by a perched culvert under the Burlington Northern Railroad tracks that parallel Bear Creek to its confluence with the Middle Fork Flathead River. The culvert is approximately 70 m long, 4.5 m wide, and 5 m tall. Water depth in the culvert at the time of sampling was approximately 2 cm. The culvert was perched approximately 2.2 m above the water surface of Autumn Creek at the outlet, and the drop pool maximum depth was 50 cm. Based on these measurements, we assume the culvert is a year-around fish passage barrier. To our knowledge, Autumn Creek has not been previously sampled.

We captured only wct in Autumn Creek, and did not capture any age-0 fish. We estimated the total abundance of age-1 and older wct in the 110 m-long section at 6 (95% CI: 5-7). The density estimate for these same fish was 1.39 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 15.2 fish/hr and 1.2 fish/100m^2. Average condition factor (K) was estimated at 1.04 for age-1 and older wct (SD = 0.08) (n = 5). Average length of age-1 and older wct was 145.2 mm (SD = 30.4, n = 6, length range = 110-186 mm) (Figure 9). Based on the population size structure and the presence of a physical barrier to upstream movement, we conclude this population is resident. We collected genetic samples from wct for future analysis. We employed 550 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Autumn Creek.

Both adult and juvenile tailed frogs were present and common. Cobble was estimated to be the dominant substrate type followed by boulder in the reach. Water temperature at the time of sampling was 12.5C. Dissolved oxygen was measured at 7.96 mg/L and conductivity at 115.7µS. Mean channel slope was 5% and average wetted width was 3.9m.

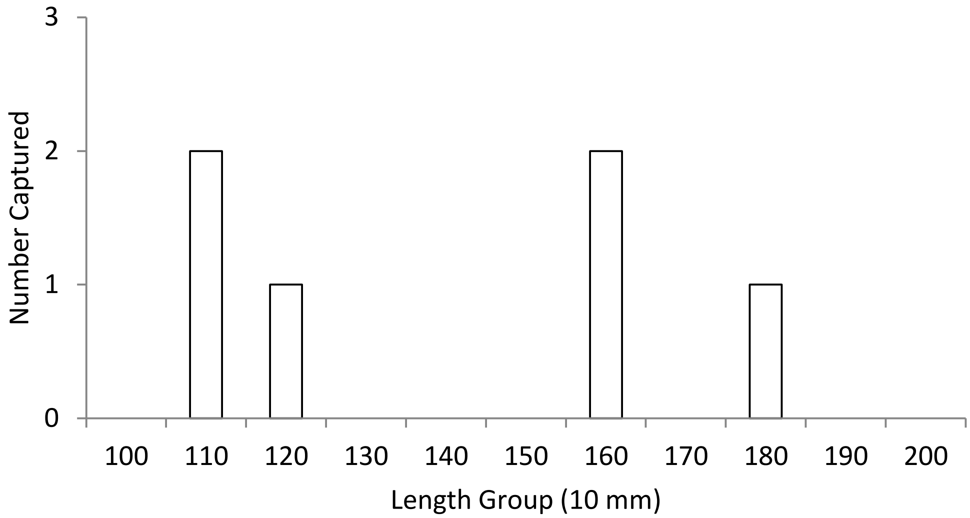

Figure 9. Length-frequency histogram for wct captured in Autumn Creek, Glacier National Park, in 2009.

Fern Creek

Fern Creek is a second order tributary to Fish Creek in the McDonald Creek drainage. The downstream end of the sampling site is located approximately 50 meters upstream of the Camas Road crossing. A perched culvert under the Camas Road may isolate the westslope cutthroat trout upstream of the culvert from downstream fish, protecting the genetic integrity of the upstream fish. Wct hybridization has not been assessed in Fern Creek. The culvert is approximately 60’ long, but with a relatively flat slope, and the drop out of the culvert to the water surface was approximately 2’ at the time of sampling. The area burned in 2003 as part of the Robert Fire, and much of the riparian area is dominated by shrub-type cover as well as standing and downed burned timber.

We captured only wct in Fern Creek, and did not capture any age-0 fish. This is not unexpected as we were sampling prior to wct fry emergence (July 9). We estimated the total abundance of age-1 and older wct in the 90 m-long section at 15 (95% CI: 13-17). The density estimate for these same fish was 4.26 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 29.7 fish/hr and 2.8 fish/100m^2. Average condition factor (K) was estimated at 1.11 for age-1 and older wct (SD = 0.1) (n = 14). Average length of age-1 and older wct was 122.5 mm (SD = 30.7, n = 15, length range = 74-179 mm) (Figure 10). Based on the size structure of the population, the location of the sampling site in the drainage, and the likelihood that the culvert under the Camas Road is at least a partial upstream barrier, it is likely this population is resident. We collected genetic samples from wct for future analysis. We employed 400 volts at 30 hertz and a 5 ms pulse width to capture fish in Fern Creek.

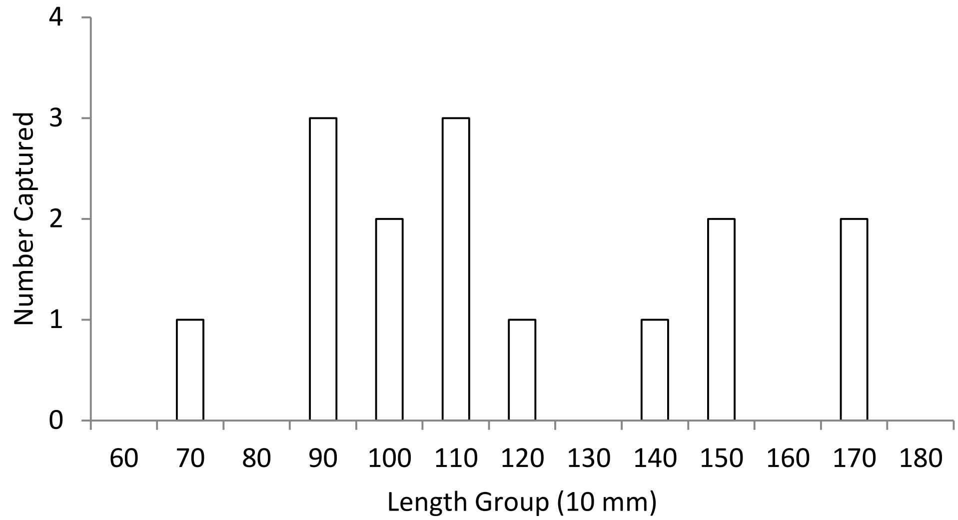

Figure 10. Length-frequency histogram for wct captured in Fern Creek, Glacier National Park, in 2009.

Both adult and juvenile tailed frogs were present and common, while adult spotted frogs *Rana pretiosa* were present and rare. Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Water temperature at the time of sampling was 7.5C. Dissolved oxygen was measured at 8.3 mg/L and conductivity at 101.5µS. Study reach mean channel slope was 5.4% and wetted width was 3.9m.

Fish Creek

Fish Creek is a third order tributary to Lake McDonald. The downstream end of the sampling site is located approximately 50 m upstream of the Camas Road crossing. The channel passes through a culvert approximately 60' long under the Camas Road. The culvert did not appear to be a barrier to upstream fish passage, although we did not obtain detailed measurements to evaluate this. The area burned in 2003 as part of the Robert Fire, and much of the riparian area is dominated by shrub-type vegetation, as well as standing and downed burned timber. Wct genetic status has not been assessed in Fish Creek.

We captured only wct in Fish Creek, and did not capture any age-0 fish. This is not unexpected as we were sampling prior to wct fry emergence (July 10). We estimated the total abundance of age-1 and older wct in the 106 m-long section at 32 (95% CI: 28-36). The density estimate for these same fish was 12.2 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 89.7 fish/hr and 9.2 fish/100m^2. Average condition factor (K) was estimated at 1.07 for age-1 and older wct (SD = 0.13) (n = 31). Average length of age-1 and older wct was 116.9 mm (SD = 35.5, n = 31, length range = 63-206 mm) (Figure 11). Based on the size structure of the population and the location of the sampling site in the drainage it is likely this population is resident. We collected genetic samples from wct for future analysis. We employed 450 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Fish Creek.

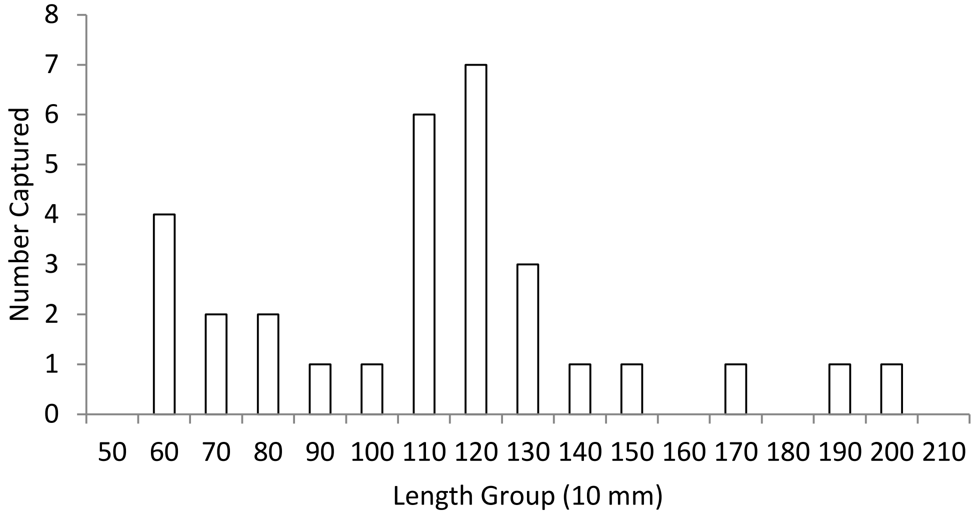

Figure 11. Length-frequency histogram for wct captured in Fish Creek, Glacier National Park, in 2009.

Both adult and juvenile tailed frogs were present and common, while adult spotted frogs were present and rare. Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Water temperature at the time of sampling was 8.0C. Dissolved oxygen was measured at 8.5 mg/L and conductivity at 63.6µS. Mean channel slope was 1.8% and average wetted width was 2.5m for the study reach.

Muir Creek

Muir Creek is a third order tributary to the M. Fk. Flathead River (Weaver et al. 1983). It flows largely southwest and drains approximately 34.8 km^2 of land area (Weaver et al. 1983). Wct genetic status has not been assessed in Muir Creek. The only species of fish documented in Muir Creek by previous researchers are westslope cutthroat trout, as well as an observation of a single adult bull trout. Juvenile bull trout had not been captured in Muir Creek (Weaver et al. 1983).

We captured only westslope cutthroat trout in Muir Creek in 2009. We estimated the total abundance of age-1 and older wct in the 91.4 m-long section at 30 (95% CI: 24-36). The density estimate for these same fish was 7.6 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 60.9 fish/hr and 5.3 fish/100m^2. Average condition factor (K) was estimated at 0.99 for age-1 and older wct (SD = 0.14) (n = 76). Average length of age-1 and older wct was 124.6 mm (SD = 51.9, n = 28, length range = 56-242 mm) (Figure 12). Average weight of wct was 29.6 g (SD = 33.4, n = 28). We also collected genetic samples from wct for future analysis. We employed 500 volts at 30 hertz, 9% duty cycle, and a 3 ms pulse width to capture fish in Muir Creek.

However, NPS sampling in the same reach in 2010 captured 6 age-1 and older juvenile bull trout in Muir Creek (pass-1=5 blt, pass-2=1 blt). They ranged in length from 123mm to 139mm, and appeared to be a single age cohort. It is possible Muir Creek may provide some limited spawning and rearing habitat for migratory bull trout from Flathead Lake.

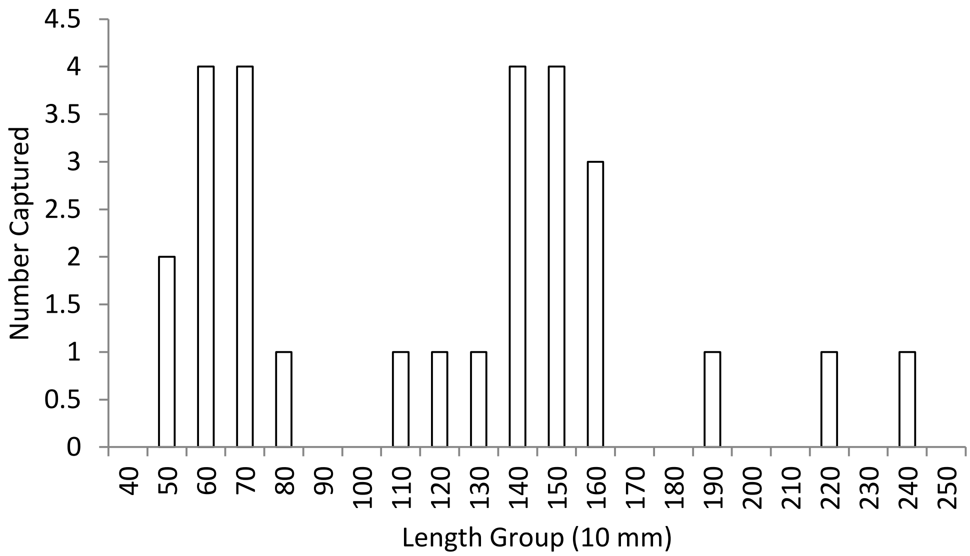

Figure 12. Length-frequency histogram for wct captured in Muir Creek, Glacier National Park, in 2009.

Our sampling took place approximately 1 km downstream of the lower-most snorkel section conducted by Weaver et al. (1983), and was in the lower reaches of the Muir Creek. Using snorkeling, Weaver et al. (1983) estimated densities of 11.6 age-1 and older wct/100m^2 in their lower-most sampling reach, which is considerably higher than our density estimate of 7.6 age-1 and older wct/100m^2 based on a depletion electrofishing population estimate. They also observed one adult bull trout in Muir Creek, whereas we did not observe any. These estimates are not directly comparable to our sampling as the sites were not duplicated and methods of enumeration differed, but they represent a range of potential wct densities in Muir Creek. However, these data also suggest that Muir Creek remains an important conservation area for wct within GNP.

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Water temperature at the time of sampling was 11C. The maximum summer temperature recorded in 2009 for Muir Creek was 15.5C on July 25 (Figure 13). Dissolved oxygen was measured at 8.45 mg/L, pH at 8.3, and conductivity at 86.4µS. Mean channel slope was 6.2% and average wetted width was 4.3m for the study reach. No amphibians were observed in Muir Creek.

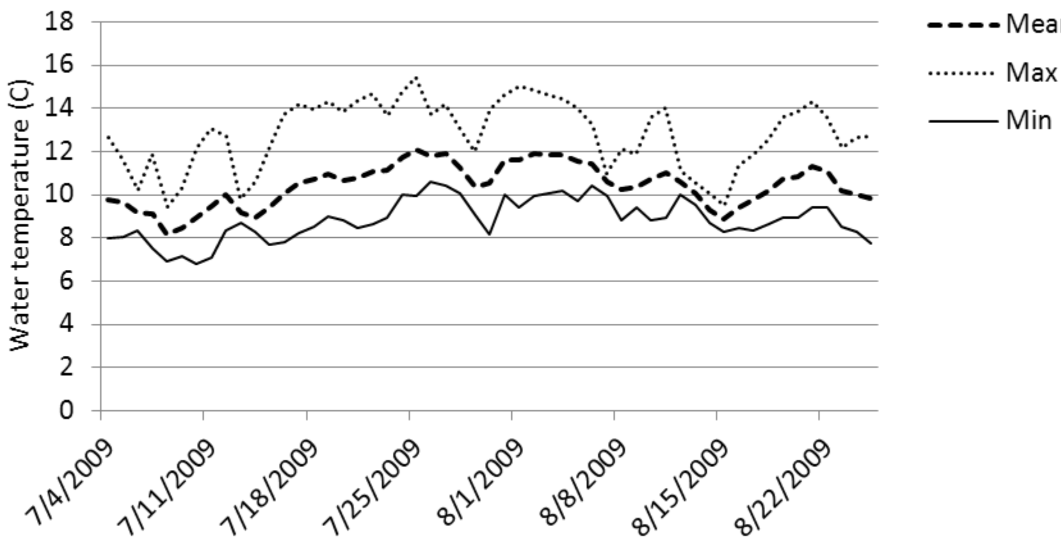

Figure 13. Mean, maximum, and minimum water temperature for Muir Creek, recorded near the confluence with the M. Fk. Flathead River during the summer of 2009.

Nyack Creek

Nyack Creek is a fourth order tributary to the M. Fk. Flathead River (Weaver et al. 1982). It flows largely southwest, draining 214.9 km^2 of the M. Fk. Flathead River drainage within the park (Weaver et al. 1982). Wct genetic status has not been assessed within the Nyack Creek drainage. Previous researchers have documented the presence of wct, blt, and mwf in Nyack Creek.

We captured ebt, sculpin, blt, and mwf in Nyack Creek (Table 5). Sculpin were the most abundant species captured followed by ebt. Nyack Creek was too large to install a blocknet and effectively deplete, and as a result we spot-shocked high quality habitat. We did not estimate CPUE as it would not be repeatable. We captured three age-0 blt, along with 27 sculpin, 23 ebt, and three mwf. Young-of-the-year brook trout dominated the catch of salmonids (Figure 14), likely reflecting use of the area for rearing, but also reflecting our sampling inefficiency with the backpack sampling equipment. We observed one adult migratory bull trout holding in a deep pool downstream of our sampling section. We employed 600 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Nyack Creek.

Table 5. Mean length (TL; mm), weight (g), standard deviation (SD), and sample size (n) of fish species captured in Nyack Creek.

Species	Mean length (SD) (n)	Length range (all individuals of all species)	Mean weight (SD) (n)
Blt	58 (2) (3)	56-60	N/A
Ebt	70.2 (17.9) (23)	47-121	4 (4.0) (20)
Mwf	59.3 (6.4) (3)	52-64	2 (0) (2)
Sculpin	65.2 (23.2) (27)	34-123	6.1 (6.5) (18)

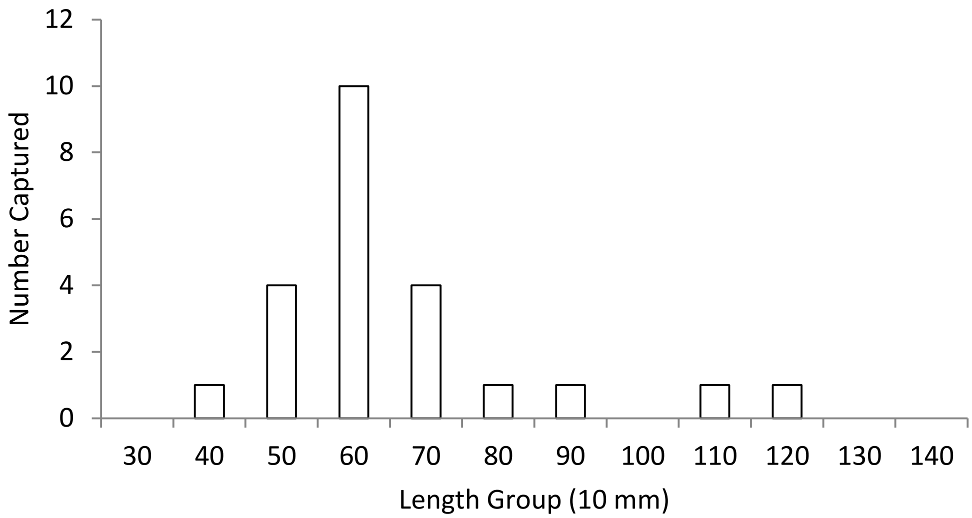

Figure 14. Length-frequency histogram for brook trout captured in Nyack Creek, Glacier National Park, in 2009.

For comparison Weaver et al. (1983) observed very low densities of both bull and westslope cutthroat trout ($\leq$0.2/100 m^2) in Nyack Creek. This is somewhat surprising for bull trout given their upper-most sampling site was located in the middle of the primary bull trout spawning reach. Weaver et al. (1983) did not observe any brook trout in their sampling reaches whereas ebt were the dominant salmonid in our catch. We sampled approximately 1.4 km downstream of earlier sampling efforts, and where they are present, brook trout will often be most abundant in the lower-most stream reaches. It is possible brook trout were present and locally abundant in Nyack Creek in 1983, however it seems more likely that brook trout have continued to expand their distribution and abundance in the M. Fk. Flathead River drainage over the past 25 years. The presence of brook trout in such close proximity to the primary bull trout spawning reach in Nyack Creek is concerning none-the-less.

Gravel was estimated to be the dominant substrate type followed by sand in the reach. Water temperature at the time of sampling was 9C. Conductivity was measured at 134.7µS and pH was measured at 8.2. Mean channel slope was <1.0% and average wetted width was 23.4m for the study reach. Adult spotted frogs were present and common the channel margins in Nyack Creek.

Park Creek

Park Creek is a third order tributary to the M. Fk. Flathead River (Weaver et al. 1982). It flows largely southwest, drains approximately 101.6 km^2 of land area (Weaver et al. 1982), and enters the M. Fk. near the community of Essex. Wct genetic status has not been assessed in Park Creek. Previous researchers have documented westslope cutthroat trout, mountain whitefish, and bull trout in Park Creek (Weaver et al. 1982).

We captured bull trout, westslope cutthroat trout, and sculpin in Park Creek. We did not capture any age-0 salmonids. We estimated the total abundance of age-1 and older bull trout in the 97.3 m-long section at 37 (95% CI: 34-40.1). The density estimate for these same fish was 3.4 blt/100m^2. First pass CPUE for age-1 and older blt was estimated at 34.3 fish/hr and 2.1 fish/100m^2. We estimated the total abundance of age-1 and older wct at 16 (95% CI: 13.7-18.3). The density estimate for these same fish was 3.4 fish/100m^2. First pass CPUE for age-1 and older wct was estimated at 16.4 fish/hr and 1.0 fish/100m^2.

Average length of age-1 and older blt was 105.9 mm while the average length of age-1 and older wct was 110.8 mm (Table 6, Figure 15). Average condition factor (K) was estimated at 0.84 for age-1 and older blt (SD = 0.12) (n = 36) and 0.94 (SD = 0.09) (n = 15) for age-1 and older wct. We employed 600 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Park Creek.

Table 6. Mean length (TL; mm), length range, and mean weight (g) for fish species captured in Park Creek, Glacier National Park, in 2009.

Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)
Blt	105.9 (36.1) (36)	65-160	12.9 (10.7) (36)
Wct	110.8 (38.6) (16)	73-181	18.4 (17.8) (15)
Sculpin	58.6 (15) (23)	35-91	N/A

Cobble was estimated to be the dominant substrate type followed by boulder in the reach. Water temperature at the time of sampling was 9.8C. Conductivity was estimated at 75.1µS and pH at 8.01. Mean channel slope was 2.0% and average wetted width was 11.2 m. No amphibians were observed in Park Creek.

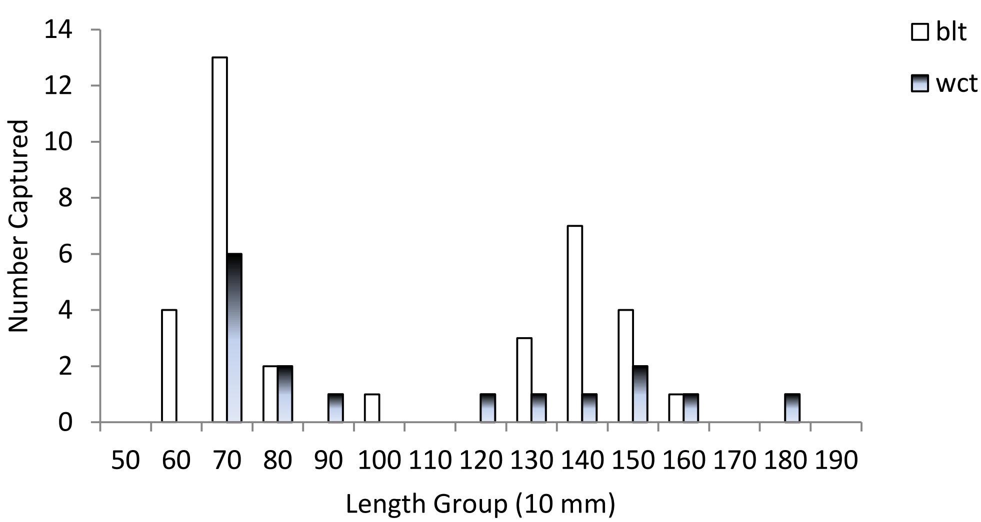

Figure 15. Length frequency histogram for bull and westslope cutthroat trout captured in Park Creek, Glacier National Park, 2009.

Pinchot Creek

Pinchot Creek is a third order tributary to Coal Creek, which flows into the M. Fk. Flathead River (Weaver et al. 1982). It flows largely southwest, draining approximately 49.9 km^2 of land area (Weaver et al. 1982). Wct genetic status has not been assessed in Pinchot Creek. Previous researchers have documented westslope cutthroat trout, mountain whitefish, bull trout, and brook trout in Pinchot Creek (Weaver et al. 1982).

We captured wct, mwf, and ebt in Pinchot Creek. All fish captured were age-1 and older. No bull trout were observed/captured. The total abundance of age-1 and older salmonids in the 111 m-long section ranged from 5 to 87 for mountain whitefish and westslope cutthroat trout, respectively (Table 7). Density for these same fish ranged from 0.7 to 11.8 fish/100m^2. First pass CPUE for age-1 and older fish ranged from 4.1 to 55.8 fish/hr. Average lengths for age-1 and older individuals of these species ranged from 113.3 mm to 200 mm (Table 8, Figure 16). Average condition factor (K) was estimated at 0.95 for age-1 and older wct (SD = 0.18) (n = 83), 0.95 for age-1 and older ebt (SD = 0.10) (n = 9), 0.84 for age-1 and older mwf (SD = 0.04) (n = 5). We employed 600 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Pinchot Creek. Weaver et al. (1982) suggested Pinchot Creek was an important bull trout spawning stream. We did not capture any juvenile bull trout in Pinchot Creek, and it should be re-sampled higher in the watershed to see if it is still used by bull trout for spawning and rearing.

Boulder was estimated to be the dominant substrate type followed by cobble in the reach. Water temperature at the time of sampling was 10.5C. Dissolved oxygen was measured at 8.53 mg/L, pH at 8.3, and conductivity at 93.5µS. Mean channel slope was 4.6% and average wetted width was 6.7m for the study reach. Adult and juvenile tailed frogs were present and common.

Table 7. Population and density estimates, as well as CPUE for age-1 and older westslope cutthroat trout, mountain whitefish, and brook trout captured in Pinchot Creek, Glacier National Park, in 2009.

Species	Population estimate (95% CI)	Density (fish/100m^2)	First pass CPUE (fish/hr)	First pass CPUE (fish/100m^2)
Wct	87 (81-93)	11.8	55.8	7.4
Mwf	5 (4-6)	0.7	4.1	0.54
Ebt	9 (7-11)	1.2	6.1	0.81

Table 8. Mean length (TL; mm), length range, and mean weight for fish species captured in Pinchot Creek, Glacier National Park, in 2009.

Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)
Wct	113.3 (42.2) (83)	59-182	20.3 (19.0) (83)
Mwf	199.8 (52.5) (5)	149-272	80.4 (65.9) (5)
Ebt	112.1 (15.9) (9)	102-147	18.1 (7.7) (9)

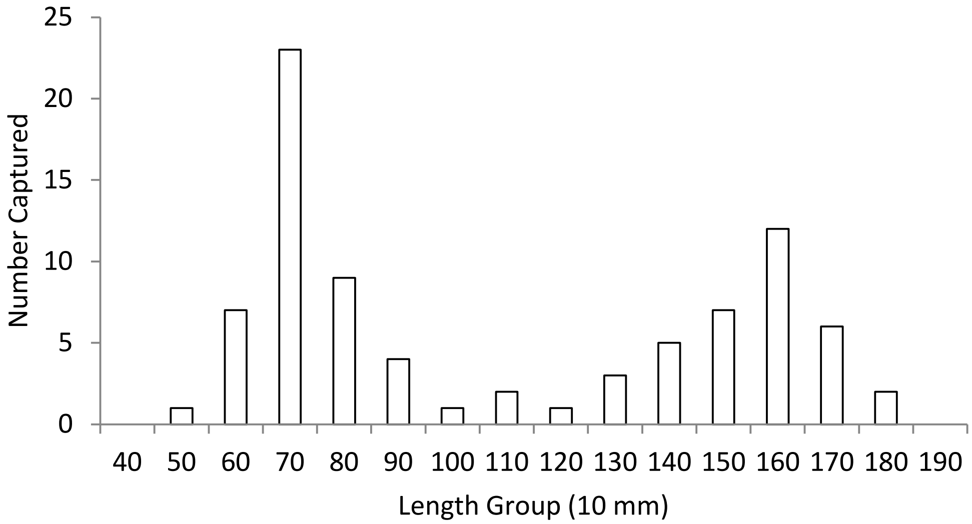

Figure 16. Length frequency histogram for westslope cutthroat trout captured in Pinchot Creek, Glacier National Park, August, 2009.

Missouri River Drainage

Midvale Creek

Midvale Creek is a third order tributary where it leaves the park and flows onto the Blackfeet Indian Reservation. It flows southeast into the Two Medicine River near the town of East Glacier. An upstream fish passage barrier exists near the town of East Glacier, and was created as part of the historic municipal water diversion system for the town of East Glacier. Midvale Creek was also the water source for the Glacier National Park Fish Hatchery located at East Glacier. The hatchery was in operation from 1919 to 1940 (Morton 1968). Stocking records indicate Midvale Creek was stocked with brook trout, rainbow trout, and cutthroat trout between 1933 and 1939. Midvale Creek had not previously been sampled by NPS staff, but is reported to contain genetically pure wct (T.Tabor, Blackfeet Nation Fish and Game Department, personal communication). The original water diversion structure served to function as an upstream fish passage barrier and protect the genetic integrity of wct in Midvale Creek. Fish managers are evaluating options to ensure the structure remains intact to continue to provide protection from non-native species colonization.

We captured wct, suspected wctXrbt hybrids and sculpin in Midvale Creek, and did not capture any age-0 fish. This is not unexpected as we were sampling prior to wct fry emergence (July 22). We implemented our depletion population estimated protocol but were unable to generate and estimate with confidence intervals because we did not capture any fish on our second pass. We captured 14 wct (including suspected hybrids) on the first pass and used this as our estimate of total abundance for age-1 and older wct in the 122 m-long section. The density estimate for these same fish was 1.7 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 24.6 fish/hr and 1.7 fish/100m^2. Average condition factor (K) was estimated at 1.01 for age-1 and older wct (SD = 0.06) (n = 14). Average length of age-1 and older wct was 156.4 mm (SD = 44.0, n = 14, length range = 99-225 mm) (Figure 17). Based on the size structure of the population and the presence of a migration barrier located downstream, is reasonable to conclude this population is resident. We collected genetic samples from wct for future analysis. We employed 400 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Midvale Creek. Results of our sampling suggest the barrier may have been compromised by rainbow trout, and upstream hybridization may be occurring.

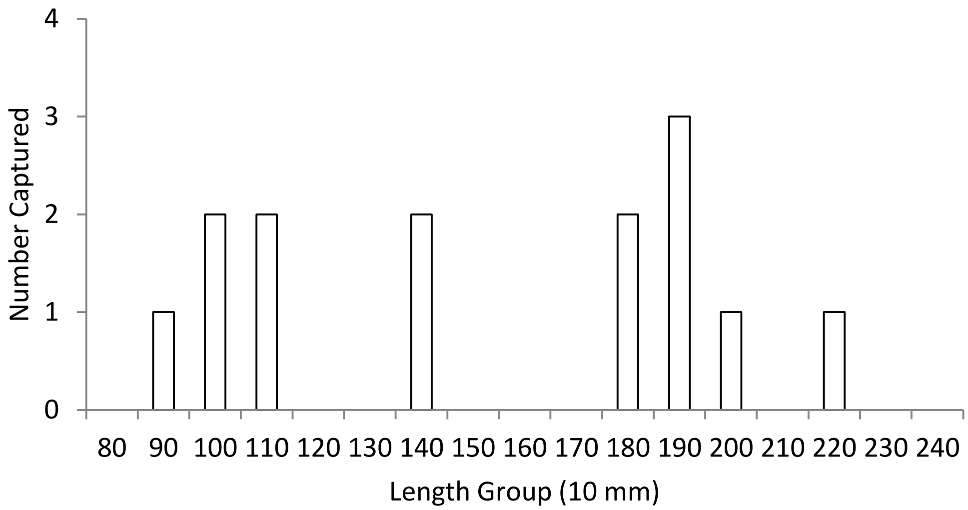

Figure 17. Length-frequency histogram for wct captured in Midvale Creek, Glacier National Park, in 2009.

Boulder was estimated to be the dominant substrate type followed by cobble in the reach. Water temperature at the time of sampling was 9.0C. Dissolved oxygen was measured at 8.87 mg/L and conductivity at 68.9µS. Mean channel slope was 3% and average wetted width was 6.6m for the study reach. We did not observe any amphibians.

St. Mary River Drainage

Boulder Creek

Boulder Creek is a third order tributary to Swiftcurrent Creek in the St. Mary River Drainage. Boulder Creek originates in Boulder Lake, near Siyeh Pass along the Continental Divide in the park. It flows northeast through the park, and onto the Blackfeet Indian Reservation before entering Swiftcurrent Creek. There are no designated trails or roads in the Boulder Creek drainage within the park, and the physical aquatic and terrestrial habitat could be characterized as pristine.

Boulder Creek supports a run of migratory bull trout from the St. Mary River (Mogen and Kaeding 2004). The average number of redds counted in Boulder Creek since counts began in 1997 is 33 redds (Downs and Stafford 2009). Using documented spawner:redd ratios of approximately 3:1 (Fraley and Shepard 1989), Downs and Jakubowski 2005), this would equate to an average annual run size of about 100 individuals. In addition to bull trout, rainbow trout, westslope cutthroat trout, and their hybrids are present. Brook trout are also present at low levels.

We captured bull trout, wct (and apparent hybrids), and one brook trout in Boulder Creek. We estimated the total abundance of age-1 and older bull trout in the 131 m-long section at 102 (95% CI: 90-140). The density estimate for these same fish was 9.0 blt/100m^2. First pass CPUE for age-1 and older blt was estimated at 86.5 fish/hr and 4.9 fish/100m^2. Average length of age-1 and older blt was 112 mm (Table 9, Figure 18). Average condition factor (K) was estimated at 0.89 for age-1 and older blt (SD = 0.09) (n = 55). We observed one adult bull trout approximately 500 mm in length within the sampling section. We employed 600 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Boulder Creek.

Table 9. Mean length (TL; mm), weight (g), standard deviation (SD), and sample size (n) of age-1 and older blt and wct. Length range represents all fish captured for all species.

Species	Mean length (SD) (n)	Length range (all individuals of all species)	Mean weight (SD) (n)
blt	112.2 (18.0) (92)	47-182	13.3 (9.0) (55)
Wct/hybrids	117.3 (49.8) (18)	57-235	15.0 (12.2) (10)

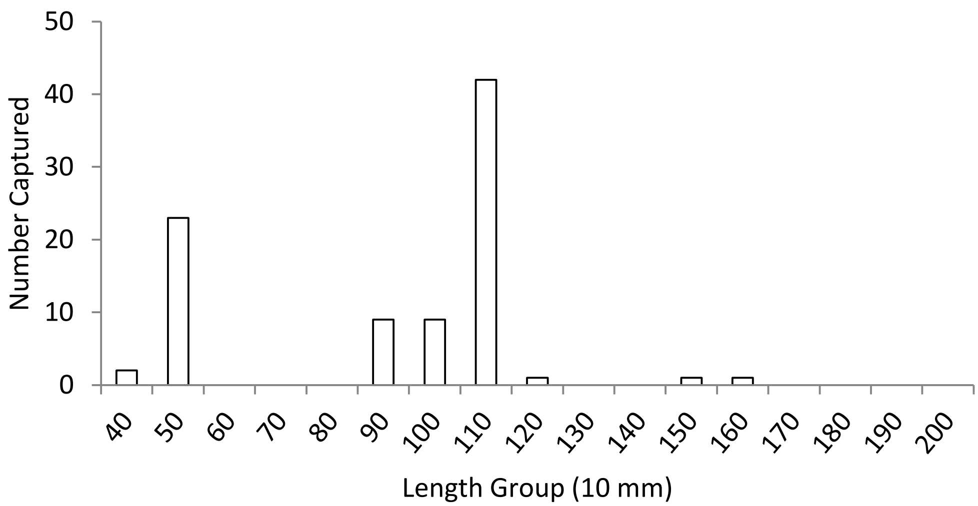

Figure 18. Length-frequency histogram for blt captured in Boulder Creek, Glacier National Park, in 2009.

It appeared as though many of the wct captured were hybridized with rainbow trout. Mogen and Kaeding (2004) reported capturing wct hybrids, but also noted they captured wct which tested at 95% genetic purity from one of the upper tributaries to Boulder Creek. Wct and their hybrids were combined for analysis as we were not confident in our ability to distinguish between them in the field. We classified them together as wct/hybrids. We did not capture any age-0 wct/hybrids. We estimated a total abundance of age-1 and older wct/hybrids at 18 (95% CI: 16-20). The density estimate for these same fish was 1.59 wct/100m^2. First pass CPUE for age-1 and older wct/hybrids was estimated at 15.7 fish/hr and 0.89 fish/100m^2. Average length of age-1 and older wct/hybrids was 117 mm (Table 9, Figure 19**).** Average condition factor (K) was estimated at 1.0 for age-1 and older wct/hybrids (SD = 0.11) (n = 10).

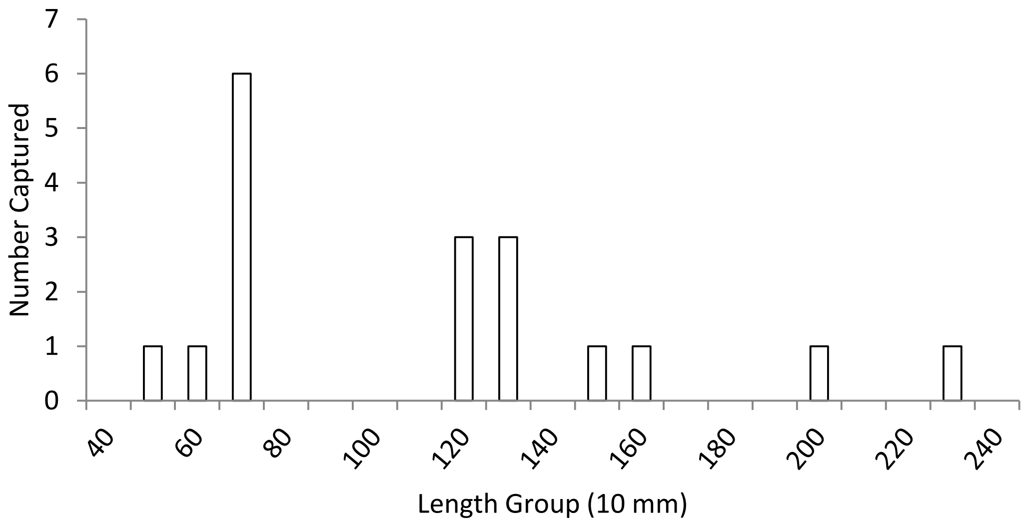

Figure 19. Length frequency histogram for wct/hybrids captured in Boulder Creek, Glacier National Park, August, 2009.

Cobble was estimated to be the dominant substrate type followed by boulder in the study reach. Water temperature at the time of sampling was 8.5C. Peak water temperatures were recorded in late July, with a maximum temperature of 16.2C (Figure 20). Conductivity was measured at 142.2µS. Mean channel slope was 2.2% and the average wetted width was 8.6m for the study reach. No amphibians were observed.

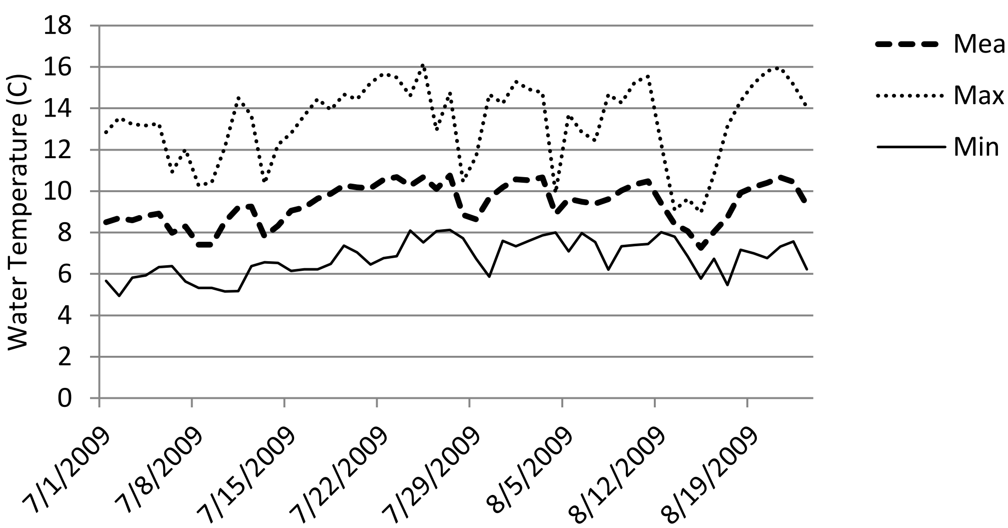

Figure 20. Mean, maximum, and minimum water temperature for Boulder Creek, recorded near the Glacier National Park border during the summer of 2009.

Kennedy Creek

Kennedy Creek is a third order tributary to the St. Mary River located on the east side of GNP. The creek originates in Kennedy Lake and flows through Poia Lake before falling over a 10-m-high waterfall near the outlet of Poia Lake (Mogen and Kaeding 2004). This waterfall represents and upstream fish passage barrier for all species, including migratory bull trout. Both Kennedy and Poia lakes are currently fishless, although Poia Lake was stocked multiple times between 1921-1938 with cutthroat and rainbow trout, as well as grayling. The sample sites were accessed via trail from the Many Glacier entrance station.

Kennedy Creek supports a run of migratory bull trout from the St. Mary River, but it has also been speculated that a resident component may also exist (Mogen and Kaeding 2004). The average number of redds counted in Kennedy Creek from 1997 through 2008 is 21 redds (Downs and Stafford 2009). The 2009 redd count was four. Using documented spawner:redd ratios of approximately 3:1 (Fraley and Shepard 1989), Downs and Jakubowski 2005), this would equate to an average annual run size of about 60 individuals.

We captured blt, wct (all appeared to be significantly hybridized with rainbow trout), ebt, and sculpin in Kennedy Creek. Mogen and Kaeding (2004) also reported capturing wct hybrids as well as mwf. We attempted to conduct a depletion population estimate approximately 1.3 km downstream of Poia Lake, immediately above the braided section near where the trail to Poia Lake encroaches onto the stream channel. The stream was a manageable size from a sampling perspective, but bull trout densities were too low to estimate their abundance. We captured only one 269 mm bull trout, weighing 174g in the 111 m-long stream reach. This could represent either a large "juvenile" or mature "resident" individual. Immediately upon capture, the bull trout attempted to consume one of the juvenile wct/hybrids already in the fish holding bucket. We also captured five age-1 and older wct/hybrids over two electrofishing passes, and estimated their abundance in the study reach at 5 (95% CI: 2-8). The density estimate for the wct/hybrids was 8.9 fish/100m^2. First pass CPUE for age-1 and older wct/hybrids was estimated at 5.3 fish/hr and 0.41 fish/100m^2. First pass CPUE for age-1 and older blt was estimated at 1.8 fish/hr and 0.14 fish/100m^2. We employed 600 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in the upper sampling reach of Kennedy Creek.

We returned one week later and moved approximately 1.6 km further downstream to attempt a second depletion estimate. Our intent was to locate the second section within the known bull trout spawning reach to improve the likelihood that we would capture sufficient numbers of bull trout to compute a depletion population estimate. This approach was unsuccessful as well. Over two passes, we captured only one juvenile bull trout measuring 153 mm long and weighing 31 grams in the 110 m-long section (first pass age-1 and older CPUE = 2.8 blt/hr and 0.13 blt/100m^2). We also captured two apparent wct X rainbow trout hybrids (first pass age-1 and older CPUE = 5.6 wct hybrids/hr and 0.27 wct hybrids/100m^2), two brook trout (first pass age-1 and older CPUE = 2.8 ebt/hr and 0.13 ebt/100m^2), and 11 sculpin. Salmonid fry were observed in channel margins, but were not captured and identified to species. We pooled lengths and weights for analysis as the sample sizes were small and the sites were located within relatively close proximately to one another (Table 10). Average condition factor was 0.88 (SD=0.02, n=2) for bull trout and 0.99 (SD=0.09, n=7) for wct/hybrids.

Table 10. Mean length, length range, and mean weight for fish species captured in Kennedy Creek, Glacier National Park, in 2009.

Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)
Blt	211 (82.0) (2)	153-269	102.5 (101.1) (2)
Ebt	142.5 (64.4) (2)	97-188	38 (42.4) (2)
Wct/hybrids	117.9 (26.0) (5)	93-172	19.0 (16.1) (5)

Cobble was estimated to be the dominant substrate type followed by boulder in both of the reaches. Water temperature at the time of sampling was 12C at the upper site and 11.5C at the lower site. Conductivity was measured at 92.8µS and 120.4µS at the upper and lower sites, respectively. Of note is that the lower site is downstream of a series of beaver ponds and an unnamed tributary flowing into Kennedy Creek from Yellow Mountain, to the North. This may have influenced the variation observed in conductivity measurements between the two sites. The estimated pH was 8.3 and 8.2 for the upper and lower sites, respectively. Average channel wetted width was similar between the two sample sites (6.7m and 6.9m for the upper and lower site, respectively), but gradient was considerably lower for the lower site (3.5% versus 1.5%). No amphibians were observed.

Lee Creek

Lee Creek is a second order tributary (within GNP) to the St. Mary River. Lee Creek flows northeast out of the park and onto the Blackfeet Indian Reservation before crossing the international border with Canada. It subsequently flows into Alberta and enters the St. Mary River near the town of Cardston. There are no designated trails or roads in the Lee Creek drainage within the park, although the Chief Mountain Highway crosses Lee Creek near the Canadian border.

Lee Creek supports a run of migratory bull trout from the St. Mary River (Mogen and Kaeding 2004). Redd counts are not currently conducted in Lee Creek due largely to difficulties with seasonal access, and conflicts with counts on other high priority streams. In addition to blt, wct hybrids and mwf are present in Lee Creek (Mogen and Kaeding 2004).

We captured bull trout and wct/hybrids in Lee Creek. We estimated the total abundance of age-1 and older bull trout in the 125 m-long section at 35 (95% CI: 32-38). The density estimate for these same fish was 7.1 blt/100m^2. First pass CPUE for age-1 and older blt was estimated at 35 fish/hr and 5.0 fish/100m^2. Average length of age-1 and older blt was 101.5 mm (Table 11, Figure 21). Average condition factor (K) was estimated at 0.85 for age-1 and older blt (SD = 0.08) (n = 34). We employed 400 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Lee Creek.

We estimated the total abundance of age-1 and older wct/hybrids ($\geq$55 mm) in the 125 m-long section at 29 (95% CI: 21-37). The density estimate for these same fish was 5.9 fish/100m^2. First pass CPUE for age-1 and older wct/hybrids was estimated at 18 fish/hr and 2.6 fish/100m^2. Average length of age-1 and older wct/hybrids was 121.5 mm (Table 11). Average condition factor (K) was estimated at 0.98 for age-1 and older wct/hybrids (SD = 0.11) (n = 26).

Table 11. Mean length, length range, and mean weight for fish species captured in Lee Creek, Glacier National Park, in 2009.

Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)
Blt	101.5 (24.8) (34)	64-133	10.5 (6.5) (34)
Wct/hybrids	121.5 (36.7) (26)	73-240	22.5 (25) (26)

Cobble was estimated to be the dominant substrate type followed by gravel in the study reach. Water temperature at the time of sampling was 9.5C. Conductivity was measured at 331µS and dissolved oxygen was estimated at 8.08mg/L. Mean channel slope was 3.7% and the average wetted width was 4.0m for the study reach. No amphibians were observed.

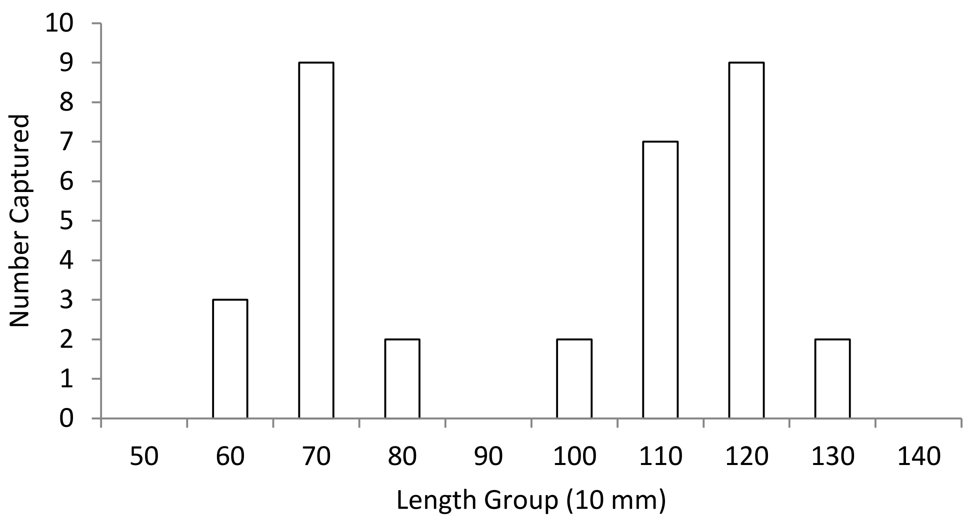

Figure 21. Length frequency histogram for bull trout captured in Lee Creek, Glacier National Park, July, 2009.

Wild Creek

Wild Creek is a first order stream and flows east out of the park and onto the Blackfeet Indian Reservation. Once on the Blackfeet Reservation, Wild Creek enters the St. Mary River between St. Mary and Lower St. Mary lakes. The lower reaches of Wild Creek are intermittent during summer months. Wild Creek is reported to contain genetically pure westslope cutthroat trout (J. Mogen, USFWS, personal communication), despite remaining connected to the St. Mary River system during higher-water periods.

We captured only wct in Wild Creek. We estimated the total abundance of age-1 and older wct in the 128 m-long section at 70 (95% CI: 60-80). Of note however, is that age-1 wct in Wild Creek were the smallest we observed in any system, ranging in length from 45-60mm (Figure 22). We did not observe any wct fry and as a result, we used all fish captured our estimation of age-1 and older wct. We would not expect to see emergent wct fry by the date we sampled (July 15) given the cold thermal regime of Wild Creek (Figure 23). The maximum summer temperature observed was 14.8C on July 25. The density estimate for age-1 and older wct was 11.1 fish/100m^2. First pass CPUE for age-1 and older wct was estimated at 61.1 fish/hr and 7.6 fish/100m^2. Average length of age-1 and older wct was 98.4 mm (SD = 40.7, n = 64). Average condition factor (K) was estimated at 1.1 for age-1 and older wct (SD = 0.08) (n = 37). We employed 500 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Wild Creek.

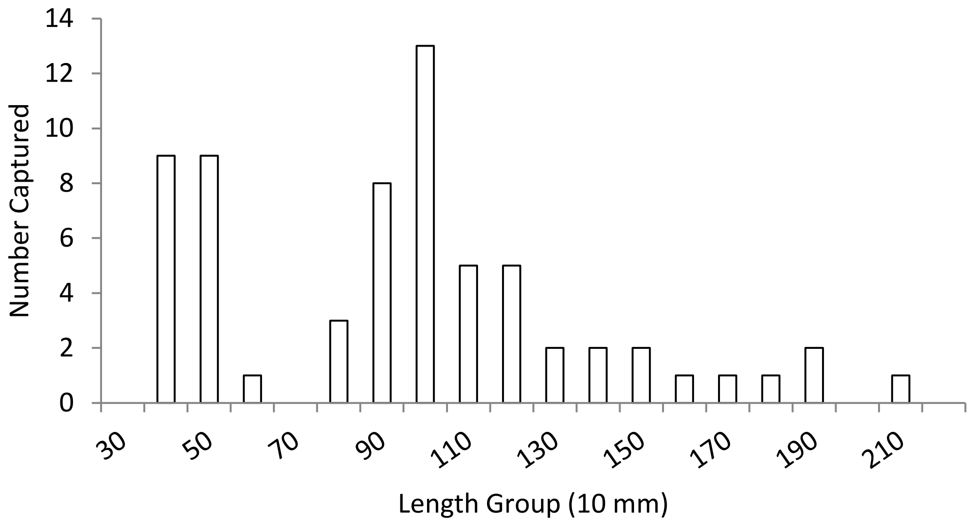

Figure 22. Length frequency histogram for westslope cutthroat trout captured in Wild Creek, Glacier National Park, 2009.

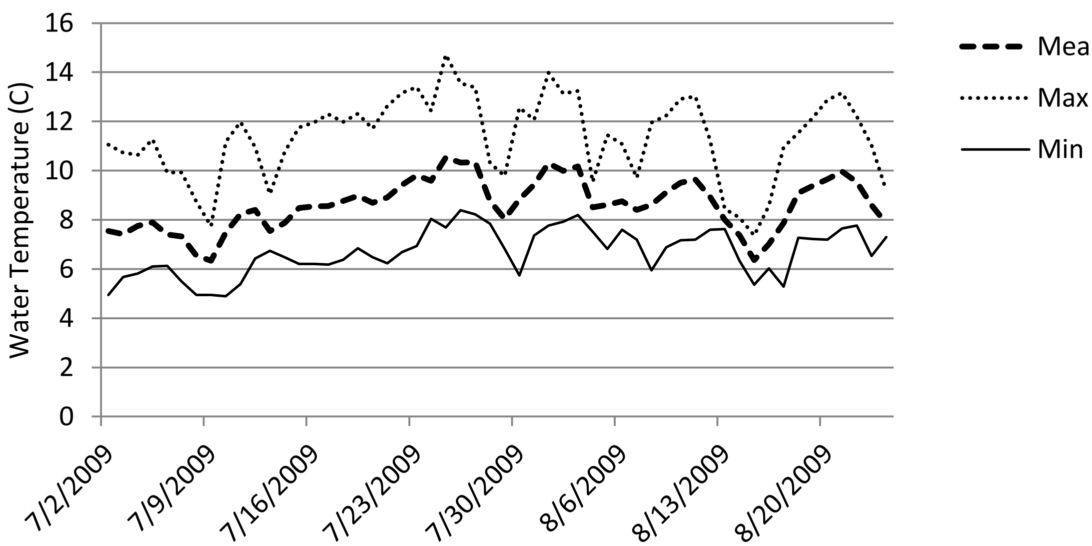

Figure 23. Mean, maximum, and minimum water temperature for Wild Creek, recorded near the Glacier National Park border during the summer of 2009.

Cobble was estimated to be the dominant substrate type followed by boulder in the reach. Water temperature at the time of sampling was 6.9C. Dissolved oxygen was measured at 9.97 mg/L, and conductivity at 180.4µS. Mean channel slope was 6.7%. No amphibians were observed.

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Glacier National Park bull trout redd counts

ABSTRACT

We conducted bull trout *Salvelinus confluentus* redd counts in 13 streams/stream reaches in Glacier National Park in 2010. Seven streams/stream reaches were surveyed in the N. Fk. Flathead River drainage, four were surveyed in the M. Fk. Flathead River drainage, and two were surveyed in the St. Mary River drainage. The total number of bull trout redds counted in these areas in 2010 was 130, down from 147 in 2009. In addition, the outlet of Upper Kintla Lake was surveyed in 2010 and 25 redds were documented. The Quartz/Cerulean lakes complex remains the strongest monitored bull trout population residing wholly within the park, with a total of 33 redds. 2010 redd counts for bull trout populations spawning in the Middle Fork Flathead River tributaries were mixed, with the Ole Creek count being a slightly above average, while Nyack Creek experienced its lowest count on record. Redd counts for bull trout populations spawning in the St. Mary drainage were average to below average in 2010. Monitored bull trout populations on the west side of the park waters continue to show low escapement levels, reflecting the adverse impacts of non-native lake trout on native fish populations. Bull trout populations on the east side of the park continue to be adversely impacted by operational issues associated with Sherburne Dam and the St. Mary Irrigation Canal.

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INTRODUCTION

Bull trout *Salvelinus confluentus* are one of only four native salmonids present in Glacier National Park (GNP) waters located west of the Continental Divide. They are one of six native salmonids present in GNP waters located east of the Continental Divide. GNP and the Blackfeet Nation have the unique distinction of supporting the only bull trout populations located east of the Continental Divide in the U.S. portion of their range. In addition, GNP supports both native (Hudson Bay drainage) and introduced (Columbia River drainage) populations of lake trout found occupying lake habitats along with bull trout, creating unique management challenges.

Bull trout exhibit three distinct general life-history forms – resident, fluvial, and adfluvial. Resident bull trout spend their entire lives in small tributaries, whereas fluvial and adfluvial forms hatch in small tributary streams then migrate into larger rivers (fluvial) or lakes (adfluvial). In the lakes of GNP, bull trout exhibit adfluvial and lacustrine-adfluvial life history strategies. These bull trout grow to maturity in the lakes, and then spawn in tributaries (adfluvial) or lake outlets (lacustrine-adfluvial). Migratory adult bull trout generally move upstream to spawning or staging areas from May through July, although some fish wait until the peak spawning time of September and October before entering spawning streams (Fraley and Shepard 1989; Schill et al. 1994; Downs and Jakubowski 2006). Spawning typically occurs in tributary streams between late August and early November (USFWS 1998), but more commonly in September and October in the Flathead Lake system (Block 1953; Fraley and Shepard 1989; Meeuwig 2008). Eggs over-winter in spawning streams until the following spring, when newly hatched fry emerge from the gravel. Age-0 bull trout can often be found in side-channels and along channel margins following emergence (Fraley and Shepard 1989). Migratory juvenile bull trout have been documented emigrating from natal streams in two pulses, with one pulse occurring in the spring with high water and the other in the fall associated with declining water temperatures and fall precipitation events (Downs et al. 2006). Juveniles may rear from one to five years in natal streams, with most emigrating at age-2 and age-3 (Downs et al. 2006). Age-0 outmigrants have been reported in some adfluvial populations, but these outmigrants did not appear to survive well to adulthood where studied (e.g. Downs et al. 2006). Resident and migratory forms may be found together, and either form can produce resident or migratory offspring.

Bull trout egg incubation success has been inversely correlated to increasing levels of fine sediment (<6.35 mm diameter) in spawning nests (redds) (Montana Bull Trout Scientific Group 1998). Spawning site selection has been related to areas of strong intragravel flow exchange (both upwelling and downwelling) (Baxter and Hauer 2000). Juvenile bull trout abundance has been positively correlated with low summer maximum water temperatures (below 14^0C) and with the number of pocket pools in stream reaches (Saffel and Scarnecchia 1995). Unembedded cobble substrate is an important overwinter habitat type for juvenile bull trout (Thurow 1997; Bonneau and Scarnecchia 1998). Excess fine sediment holds the potential not only to reduce egg and embryo survival, but might also limit juvenile bull trout abundance in streams by reducing the amount of interstitial spaces available for overwinter habitat. Channel stability, habitat complexity, and connectivity are all important components in bull trout population persistence (Rieman and McIntyre 1993).

Bull trout are part of a historic fish assemblage that is fundamental to the biodiversity of GNP, and represent the evolutionary legacy of a top-level aquatic predator in GNP. Protecting native fish resources is a high priority for the park's conservation and management programs (NPS 2006). Ongoing research, monitoring, and management efforts conducted by GNP and its partners remain critical in understanding bull trout population dynamics in the park, and in establishing management programs to benefit native fish.

Redd counts, or spawning nest counts, are used across the range of bull trout to monitor population trends. They are typically used as an index of abundance to gauge the relative strength of adult escapement from year to year. They can also be used to estimate actual adult escapement by expanding the redd counts to fish numbers using various spawner to redd ratios. Redd counts require far less effort to conduct than other traditional monitoring methods such as trapping, and yet provide valuable information on bull trout at the watershed and/or population scale. However, redd counts are not without their limitations, as the technique has been shown to be prone to observer variability and error (Dunham et al. 2001, Muhlfeld et al. 2006), yet they continue to remain an important monitoring tool for bull trout populations.

Redd counts are conducted in Glacier National Park (GNP) annually by the National Park Service (NPS), the U.S. Fish and Wildlife Service (USFWS), the Montana Fish, Wildlife, and Parks (MFWP), and the U.S. Geological Survey (USGS) (Downs and Stafford 2009). The longest redd count dataset on bull trout spawning activity in GNP is from three tributaries (Ole, Park, and Nyack creeks) to the Middle Fork Flathead River, associated with monitoring bull trout populations from Flathead Lake. MFWP biologists have been counting bull trout redds annually in Ole Creek and approximately every five years in Nyack and Park creeks, in GNP since 1980. The USFWS has been conducting bull trout redd counts in the St. Mary drainage on the east side of the park since 1997.

GNP is unique as it and the adjacent Blackfeet Indian Reservation are the only place where bull trout occur east of the Continental Divide in the U.S. portion of their range. GNP supports a diversity of life-history strategies for bull trout, including both resident and migratory forms. Resident bull trout have been documented in the St. Mary River drainage (Mogen and Kaeding 2004), while migratory fish from Flathead Lake use tributaries to the Middle and North forks of the Flathead River for spawning and rearing (Weaver et al. 2006). Other populations on the west side of GNP use the lake systems within the park for subadult rearing and adult residence, while spawning and rearing in upstream reaches of their inflow tributaries (e.g. Quartz Lake) (Meeuwig 2008). Less commonly, other west side populations (e.g. Upper Kintla Lake) appear to use the lake environment for subadult rearing and adult residence, while spawning occurs in the outlet stream.

Bull trout spawning surveys were initiated by USFWS staff between 2002 and 2004 for a number of these "disjunct" west side bull trout populations (Meeuwig et al. 2007). A number of other bull trout populations on the west side of the park have not been monitored beyond recent single year electrofishing and gill net surveys (Meeuwig et al. 2007), and we simply do not know where they spawn or long-term population trends (e.g. Lincoln, Trout, Arrow, Isabel, Upper Isabel lakes). It will be critical to establish index redd count monitoring in these populations on some frequency, as they represent the majority of "secure" populations of bull trout on the west side of GNP (Fredenberg et al. 2007).

METHODS

Experienced fisheries staff from GNP, USGS, MFWP, USFWS, and the Blackfeet Tribe identified and enumerated bull trout redds in 2010. Redd surveys generally occur during the first full three weeks of October. Surveys in 2010 occurred between October 1-22. Early to mid-October is the preferred time for counting bull trout redds as most bull trout spawning has already occurred (peak spawning occurs in September), most redds are still clearly visible, and it is consistent with the timing of earlier counts.

Redds were located visually by walking along annual monitoring sections within each tributary. Redds were defined as areas of clean or “bright” gravels at least 0.3 x 0.6 m in size with gravels of at least 76.2 mm in diameter having been moved by the fish (where other fall spawning species may be present such as brook trout), and with a mound of loose gravel downstream from a depression (Pratt 1984). In areas of superimposition, each distinct depression was counted as one redd. Only disturbed areas of the streambed that observers felt were likely made by fish were classified as bull trout redds and were included in the counts (as opposed to those disturbed areas of the streambed that may have been caused by stream hydraulics). Individual redd locations were located using GPS technology where the spatial distribution of spawning activity was of particular interest.

The draft U.S. Fish and Wildlife Service Bull Trout Recovery Plan (USFWS 2002) suggests using at least 10 years of redd count data for trend analysis. Both Kennedy and Boulder creeks on the east side of the park, as well as Ole, Park, and Nyack creeks on the west side of the park meet the criteria. We used a nonparametric rank-correlation procedure, Kendall’s tau (Daniel 1990), to test for trends in “count year” versus “redd count” in the long-term redd count data set (Rieman and Myers 1997). We used tau-b to compensate for any bias caused by ties in the data, and noted statistical significance at the $\alpha = 0.05$ level (Rieman and Myers 1997).

RESULTS AND DISCUSSION

GNP, USGS, USFWS, and MFWP staff surveyed eight stream reaches in the N. Fk. Flathead River drainage and four in the M. Fk. Flathead River drainage. In addition, two other streams were surveyed in the St. Mary River drainage by the GNP, USFWS, and Blackfeet Tribe personnel. A total of 14 streams/stream reaches in 10 watersheds were monitored in 2010 (Figure 1).

East of the Continental Divide, bull trout redd counts continue to remain relatively strong, although few populations are monitored (Figure 2; Appendix A). Redd counts in 2010 were average for Boulder Creek, and below average for Kennedy Creek. Correlations in “count year” versus “redd count” failed to identify any statistically significant trends, however for Boulder Creek the correlation was positive, but relatively weak (tau-b = 0.37; $p > 0.05$). If strong redd counts continue, it may be possible to detect a significant positive trend in Boulder Creek in the near future. The correlation between “count year” and “redd count” for Kennedy Creek was negative, but not statistically significant (tau-b = -0.34, $p > 0.05$). Since 2004 redd counts have trended downward in Kennedy Creek, which is a concern.

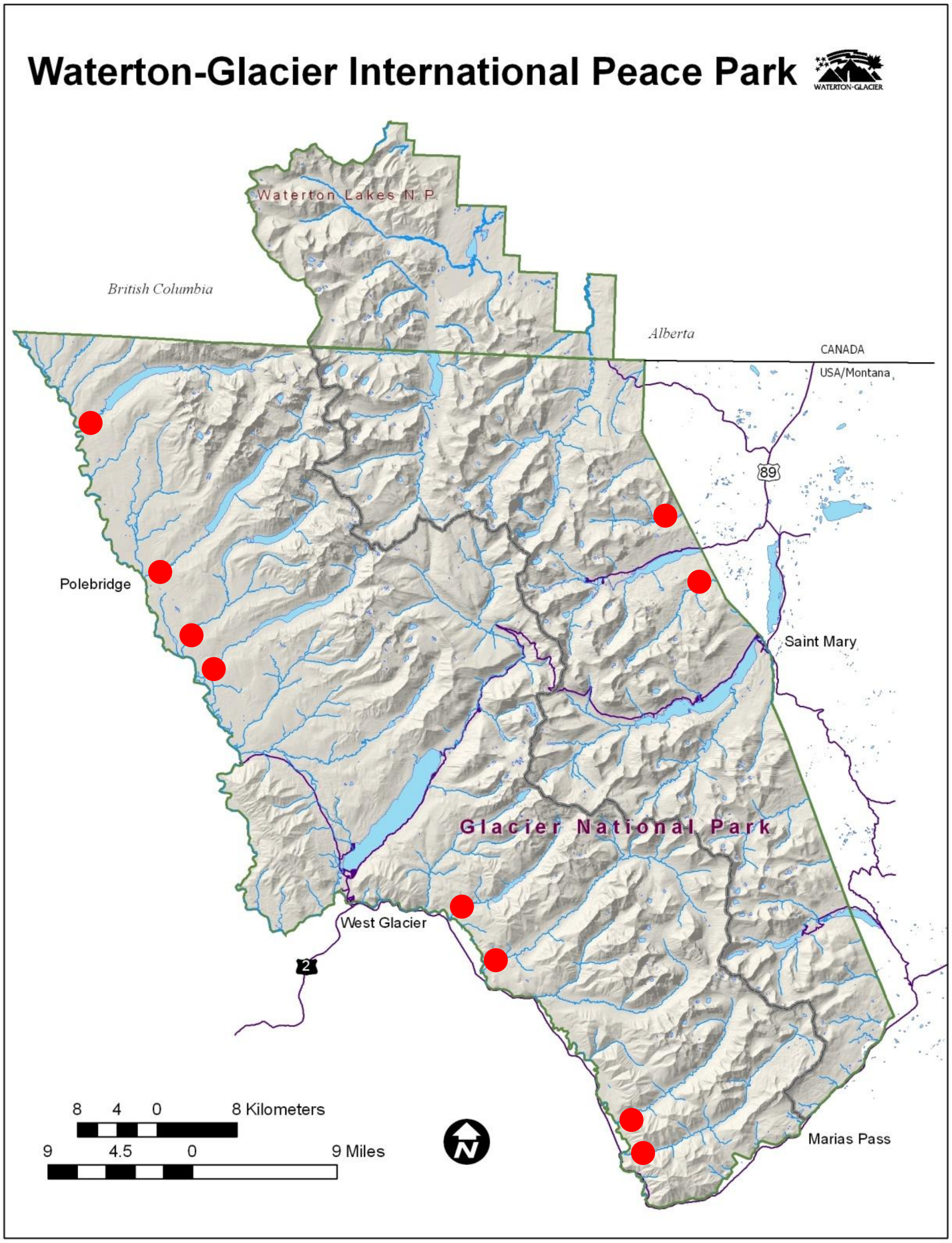

Figure 1. Drainages monitored for bull trout spawning activity (red circles) in Glacier National Park, Montana in 2010.

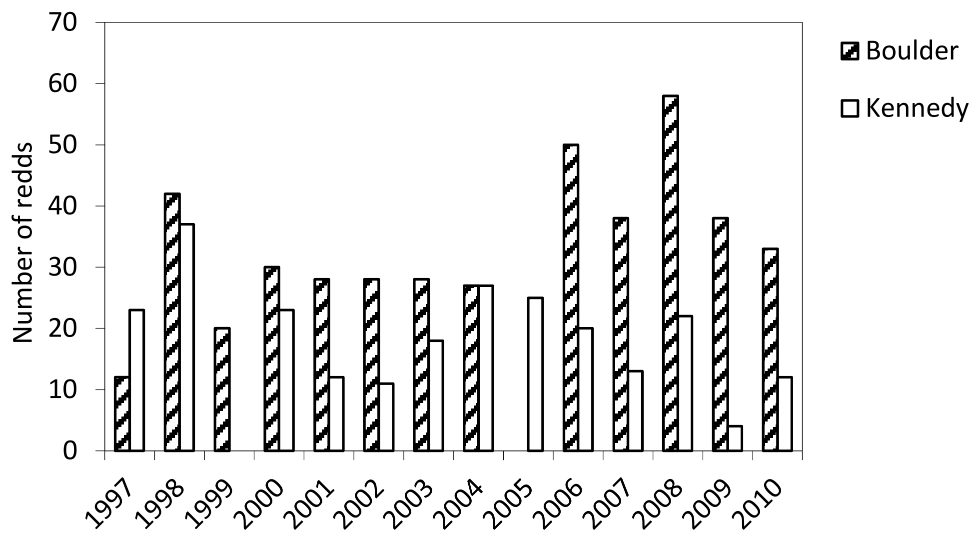

Figure 2. Bull trout redd counts for Boulder and Kennedy creeks, Hudson Bay Drainage, Glacier National Park.

In order to meet bull trout recovery objectives in the St. Mary River drainage established by the USFWS (USFWS 2002), expanded monitoring of bull trout population abundance and trends is needed in GNP. Recovery criteria focus on quantitative measures of adult bull trout abundance and population trends. Recovery Criteria 1 (there are four criteria in total) calls for the presence of nine stable local bull trout populations in the St. Mary-Belly River Recovery Unit, well distributed across the landscape. Recovery Criteria 2 calls for documentation of at least one population in each of the six Core Areas supporting at least 100 adults annually. Recovery Criteria 3 calls for documenting a stable or increasing population of bull trout in the Recovery Unit over time, using at least 10 years of trend data. Recovery Criteria 4 addresses the need for resolution to operational issues associated with Sherburne Dam and the St. Mary Irrigation Canal operated by the U.S. Bureau of Reclamation (BOR). The most cost-effective way to evaluate progress against the first three criteria may be through bull trout redd counts, but existing efforts focusing on monitoring only 2 of the 6 core area populations may fall short in their ability to adequately evaluate local populations against established recovery criteria.

Because the identified spawning habitat for these populations occurs within GNP, it is largely unaffected by threats typically associated with bull trout spawning habitat in other areas of their range (i.e. road building, residential development, timber harvest). Some traditional threats do exist however, largely in the form of trespass cattle grazing in the GNP portion of the Kennedy Creek drainage and the construction and operation of Sherburne Dam and the Milk River Irrigation Project (USFWS 2002). Due to a historic lack of fencing, trespassing cattle have been observed wading in Kennedy Creek in GNP in the primary bull trout spawning area during and after bull trout spawning (J. Mogen, USFWS, personal communication), potentially impacting bull trout incubation and emergence success in the area. Recent studies (Gregory and Gamett 2009) have identified the potential for significant damage to bull trout spawning nests as a result of cattle trampling. Recent efforts by the Blackfeet Nation to fence cattle out of the bull trout spawning area on Kennedy Creek should benefit this bull trout population.

Sherburne Dam and the St. Mary Irrigation Canal impact GNP native fish populations and represent the single largest "connectivity" issue bull trout populations face in the U.S. portion of the Hudson Bay drainage (USFWS 2002). Construction of Sherburne Dam from 1914-1921, located just outside of the GNP boundary, created Sherburne Reservoir which flooded over 8 km of shallow lake and stream habitat in the park within the Swiftcurrent Creek drainage, downstream of Swiftcurrent Falls. Annual operation of the dam dewaters Sherburne Creek downstream of the dam in some months, resulting in the loss of native fish including bull trout (Mogen and Kaeding 2001). The associated St. Mary Irrigation Canal, used to deliver irrigation water to the Milk River, remains unscreened and results in the loss of bull trout and other native fish from the system (Mogen and Kaeding 2001). The St. Mary Diversion Dam, used to provide water into the irrigation canal, creates an approximately 6' high impediment to upstream migration of bull trout during the migration season (Mogen and Kaeding 2005). The BOR has recently initiated formal consultation with the USFWS and has formed stakeholder working groups to identify issues and develop alternatives for consideration in the National Environmental Policy Act (NEPA) process to address fishery issues associated with the Milk River Irrigation Project. Addressing the fishery impacts of this project will significantly improve migration conditions as well as survival of migratory bull trout.

On the west side of GNP, both migratory stocks of bull trout from Flathead Lake as well as populations that reside entirely within the park (known locally as "disjunct" migratory populations) are monitored (Appendix A). Flathead Lake migratory bull trout stocks underwent dramatic declines starting in about 1990, and declines are believed to have been the result of the introduction of mysis shrimp *Mysis relicta* into the system and resulting major alterations in trophic dynamics (i.e. rapidly expanding lake trout population) in the lake, as well as drought conditions (Weaver et al. 2006). As the spawning and stream rearing habitats for the Flathead Lake populations that use GNP are largely located within the park, the traditional land-use threats to habitat quality (i.e. road building, timber harvest, residential development) are not the primary issue of concern for these individual populations. One of the most significant contemporary threats to these populations is predation with and competition by non-native fish species in both the migratory habitats between spawning and rearing areas in GNP and Flathead Lake (i.e. mainstem Flathead River) (Muhlfeld et al. 2008), as well as Flathead Lake itself (Deleray et al. 1999).

The only populations that have been monitored for 10 years or more with redd counts on the west side of GNP are Ole, Nyack, and Park creeks. Bull trout redd counts in Ole Creek have been monitored annually by MFWP since 1980 (Weaver et al. 2006). While Ole Creek is monitored annually, Nyack and Park have generally been counted every five years, as part of a basin-wide effort (Weaver et al. 2006). In 2009, the NPS initiated annual redd counts on these two streams.

The 2010 redd count for Ole Creek of 32 was higher than the long-term average of 26 redds. No statistically significant trends are evident in the long (full data set; tau-b = -0.01, $p > 0.05$) or short-term (most recent 10 years; tau-b = 0.43, $p > 0.05$) data sets for Ole Creek (Figure 3, Appendix A). The 2010 redd count on Nyack Creek was six, which is the lowest redd count on record for this stream (Figure 3, Appendix A). Although the correlation remains negative, no statistically significant trends were detected in the long-term redd count data set (tau-b = -0.42, $p > 0.05$). Sufficient data does not exist to analyze short-term (10 year) trends on Nyack Creek due to the intermittency of the counts. The 2010 redd count on Park Creek was three, which is considerably lower than the long-term average of sixteen for this stream (Figure 3, Appendix A). No statistically significant trends were detected in the long-term redd count data set for Park Creek (tau-b = -0.06, $p > 0.05$). Sufficient data does not exist to analyze short-term (10 year) trends on Park Creek due to the intermittency of the counts.

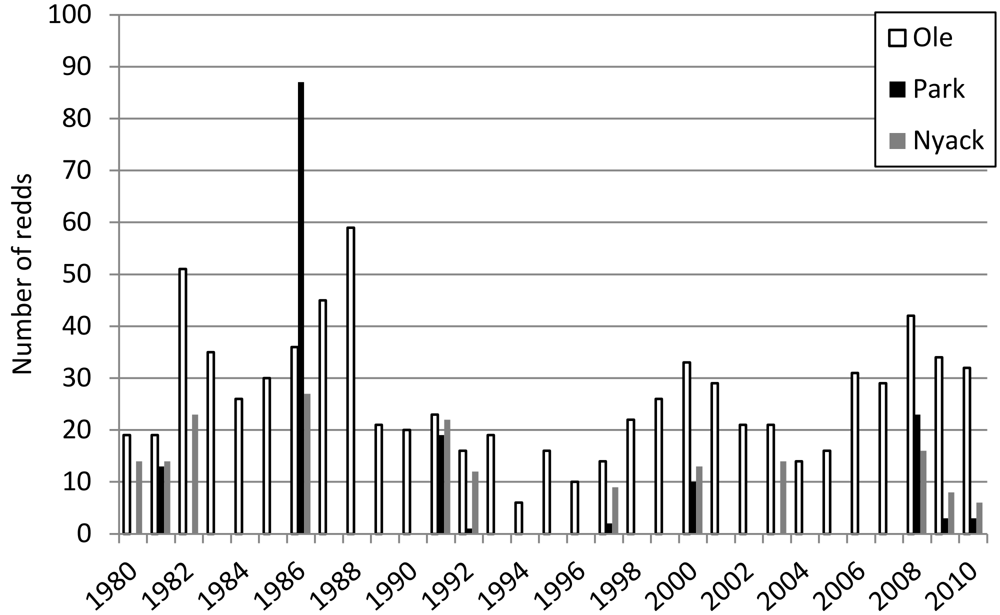

Figure 3. Bull trout redd counts conducted in Ole, Park, and Nyack creeks, Middle Fork Flathead River Drainage, Glacier National Park.

High annual variability in counts can make detecting trends using redd counts difficult and require long data sets. Previous authors using similar data sets predicted it may take over 100 years of continuous redd count data collection before a statistically significant trend can be detected in some systems (Rieman and Myers 1997). However, evaluation of observer error in bull trout redd counts (Dunham et al. 2001, Muhlfeld et al. 2006), as well as documented relationships between redd counts and actual adult spawning escapement (Bonar et al. 1997, Dunham et al. 2001, Downs and Jakubowski 2006) support their continued use as a key monitoring tool for bull trout populations in GNP.

Expanding populations of lake trout from Flathead Lake have colonized almost all of the accessible lake habitats on the west side of GNP, and now threaten the persistence of the majority of the “disjunct” migratory bull trout populations remaining on the west side of GNP. Nine of seventeen lake-dwelling populations of bull trout located on the west side of GNP have been compromised by lake trout (Fredenberg et al. 2007), and lake trout have been documented replacing bull trout as the dominant predator in these waters, where long-term data on fish populations exists (Fredenberg 2002). Some populations appear to be persisting at dangerously low numbers (e.g. Bowman, Logging, and Harrison lakes), and interactions with non-native lake trout are likely the driving force behind the declines and the precarious status of bull trout in these systems (Donald and Alger 1993).

Successful conservation of native fish species in GNP will ultimately require aggressive actions, guided by a multi-year fisheries management plan for GNP that places a high priority on conservation and management of native fish. Such a plan would likely include a strategy of non-native fish removal in some waters, protecting existing natural native fish populations from colonization by non-native fish, as well as potentially establishing new populations of native fish in areas of the park secure from invasion by non-native species. The recently developed Action Plan to Conserve Bull Trout in Glacier National Park (Fredenberg et al. 2007) will serve as a key reference in developing conservation strategies in the future.

In the interim, additional population monitoring and evaluation is appropriate. In addition to periodic gill netting (5 or 10 year frequency)and stream depletion population estimation, redd count index streams/sections should be established for additional bull trout populations to provide a frame of reference to gauge any future changes in population status.

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Appendix A. Bull trout redd counts conducted in Glacier National Park, Montana, 1980 to present.

Table A.1. Bull trout redd counts conducted in Glacier National Park, Montana, 1980 to present.

Stream	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995
Hudson Bay Drainage																
Boulder Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Kennedy Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
N. Fk. Flathead Drainage																
Akokala Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Agassi Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Bowman Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Harrison Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Jefferson Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Logging Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (lower)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (middle)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (upper)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Rainbow Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Upper Kintla outlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	52	--
Upper Kintla inlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
M. Fk. Flathead Drainage																
Ole Cr.	19	19	51	35	26	30	36	45	59	21	20	23	16	19	6	16
Nyack Cr.	14	14	23	--	--	--	27	--	--	--	--	22	12	--	--	--
Park Cr.	--	13	0	--	--	--	87	--	--	--	--	19	1	--	--	--
Starvation Cr.	1	1	--	--	--	--	--	--	--	--	--	--	--	--	--	--

Table A.1. Continued.

Stream	1996	1997	1998	1999	2000	2001	2002	2003	2004f	2005b,f	2006	2007a,c	2008c,d,e	2009g	2010h
Hudson Bay Drainage															
Boulder Cr.	--	12	42	20	30	28	28	28	27	--	50	38	58	38	33
Kennedy Cr.	--	23	37	--	23	12	11	18	27	25	20	13	22	4	12
N. Fk. Flathead Drainage															
Akokala Cr.	--	--	--	--	--	--	--	--	--	--	--	--	11	6	1
Agassi Cr.	--	--	--	--	--	--	--	--	--	--	--	--	0	--	--
Bowman Cr.	--	--	--	--	--	--	0	0	0	0	2	1	0	0	1
Harrison Cr.	--	--	--	--	--	--	--	--	4	0	8	15	14	1	6
Jefferson Cr.	--	--	--	--	--	--	--	--	--	--	--	--	0	0	--
Logging Cr.	--	--	--	--	--	--	--	--	3	20	0	--	5	0	3
Quartz Cr. (lower)	--	--	--	--	--	--	--	--	1	3	2	2	3	2	2
Quartz Cr. (middle)	--	--	--	--	--	--	--	0	0	0	0	0	0	3	0
Quartz Cr. (upper)	--	--	--	--	--	--	--	31	46	4	36	14	51	34	27
Rainbow Cr.	--	--	--	--	--	--	--	--	--	--	--	--	28	12	4
Upper Kintla outlet	--	--	--	--	--	--	--	--	--	--	--	--	0	--	25
Upper Kintla inlet	--	--	--	--	--	--	--	--	--	--	--	--	0	--	--
M. Fk. Flathead Drainage															
Ole Cr.	10	14	22	26	33	29	21	21	14	16	31	29	42	34	32
Nyack Cr.	--	9	--	--	13	--	--	14	--	--	--	--	16	8	6
Park Cr.	--	2	--	--	10	--	--	0	--	--	--	--	23	3	3
Starvation Cr.	--	0	--	--	0	--	--	--	--	--	--	--	--	--	--

a = spawning activity on Upper Quartz likely inhibited by weir at mouth.
b = minimum count due to high flows in Upper Quartz.
c = count accuracy may have been compromised to due to kokanee spawning activity in Harrison.
d = cumulative count based on multiple survey events in Upper Quartz.
e = count conducted by helicopter on Park.
f = minimum count on Ole as high flows may have obliterated some redds.
g = Kennedy count does not include 3 additional redds counted upstream of the index section, or 2 redds counted in unnamed tributary flowing from Yellow Mountain.
h=very poor visibility in Bowman Creek due to glacial runoff from Jefferson Creek.

Quartz Lake Fisheries Monitoring

ABSTRACT

We sampled Quartz Lake with 11 sinking experimental mesh gill nets from September 9-11, 2009 as part of a trend netting effort that occurs every five years. We captured 243 individuals comprised of five species (38 bull trout, 10 lake trout, 68 longnose suckers, 114 mountain whitefish, and 13 westslope cutthroat trout). Quartz Lake has been historically sampled with six shoreline sinking gill net sets. We did not capture any lake trout in these sets in 2009, despite what appears to be a growing population of lake trout in Quartz Lake. Catch composition was relatively similar to earlier sampling suggesting lake trout have not expanded to the point of causing significant adverse impacts to the native fish population. Comparisons of the six long-term net set locations over time suggest these types of sets are not appropriate for monitoring newly established or relatively low density lake trout populations in the larger west-side park lakes. Deeper sets (>100') were more productive at capturing lake trout and reducing bycatch of other species. Future monitoring efforts in Quartz Lake should include approximately equal numbers of shoreline and deep-water net sets, as was implemented in 2009.

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INTRODUCTION

Quartz Lake is a 352 hectare lake located in the N. Fk. Flathead River drainage in Glacier National Park (Figure 1). Quartz Lake has a maximum depth of 83 meters. The lake is approximately 7.2 km long, 0.8 km wide at the widest point, and sits at an elevation of 1,348m. It is the third lake in a four lake chain moving upstream in the Quartz Creek drainage, and is the largest lake in the system. The drainage is roadless, and is accessed by trail from either the Bowman Lake campground or from the inside N. Fk. Road. The physical habitat is pristine. Quartz Lake has one small backcountry campground, and receives little fishing pressure. However, it is known for providing quality fishing opportunities for medium-sized (250-350 mm) westslope cutthroat trout.

Quart Lake supports one of the strongest bull trout populations remaining on the west side of the park. There are 15 lakes on the west side of Glacier National Park that support bull trout, and until recently Quartz Lake was the largest lake remaining on the west side of the park with an intact native fish assemblage not compromised by non-native species. Of the 15 lakes that support bull trout on the west side of the park, 9 have been compromised by lake trout and one has been compromised by brook trout. Only five relatively small lakes remain secure from invasion by non-native species due to natural barriers (Fredenberg et al. 2007). In 2004, National Park Service (NPS) and U.S. Fish and Wildlife Service (USFWS) staff initiated construction of a barrier to prevent lake trout from accessing Quartz Lake. However, during construction lake trout were discovered in the lake upstream of the barrier site and the barrier was left unfinished. In 2009 a cooperative project between the U.S. Geological Survey (USGS) and Glacier National Park was initiated to evaluate the feasibility of lake trout removal/suppression (Muhlfeld and Fredenberg 2010).

Little baseline data exists with regard to native fish population size or fish community structure in Quartz Lake. USFWS staff compiled historical gill netting data where it existed on the west side of the park, including Quartz Lake, and then repeated the survey in 2000. They found that between 1969 and 2000, lake trout had replaced bull trout as the dominant top level aquatic predator in all of the systems where the two species were found in 1969 (Fredenberg 2002). Lake trout were not detected in Quartz Lake in any of the surveys, and it wasn't until 2005 that lake trout were detected using both angling and gill netting on Quartz Lake (Meeuwig 2008). Lake trout were captured at very low levels in 2005, and it was presumed that the Quartz Lake population was newly established. Since 2000, gill net sampling of Quartz Lake has occurred approximately every five years, and we repeated the sampling in 2009 to improve the baseline data set prior to the initiation of experimental gill net removal of lake trout.

METHODS

We used sinking multi-filament experimental mesh gill nets to sample Quartz Lake from September 9-11, 2009. Nets were 38.1m (125') X 1.83m (6') experimental multifilament nylon with five 7.62m (25') panels consisting of 19mm (3/4") (#104 twine), 25mm (1") (#139), 32mm (1-1/4") (#104), 1-1/2" (38mm)(#104), 51mm (2") (#139) bar measured mesh. 30# leadcore and polyfoam line was used to hold the net upright, while maintaining contact with the lake bottom.

On September 9, nets (sets 1-6) were set a standard locations used in previous surveys. We set additional deep-water net sets (sets 7-11) on September 10 to target lake trout (Figure 2, Table 1). Nets were set in late afternoon/early evening and allowed to soak overnight. They were retrieved at sunrise the following day. Nets were set with the smallest mesh towards shore, with the exception of net number four which was set with the largest mesh towards shore.

Figure 1. Location of Quartz Lake, North Fork of the Flathead River drainage, Glacier National Park.

All fish captured alive were removed from the net, counted in the net catch total, and were released. All other fish were measured (TL; mm) and weighed (g). Bull trout were sexed, maturity status was determined, otoliths were removed, and a genetic sample was collected.

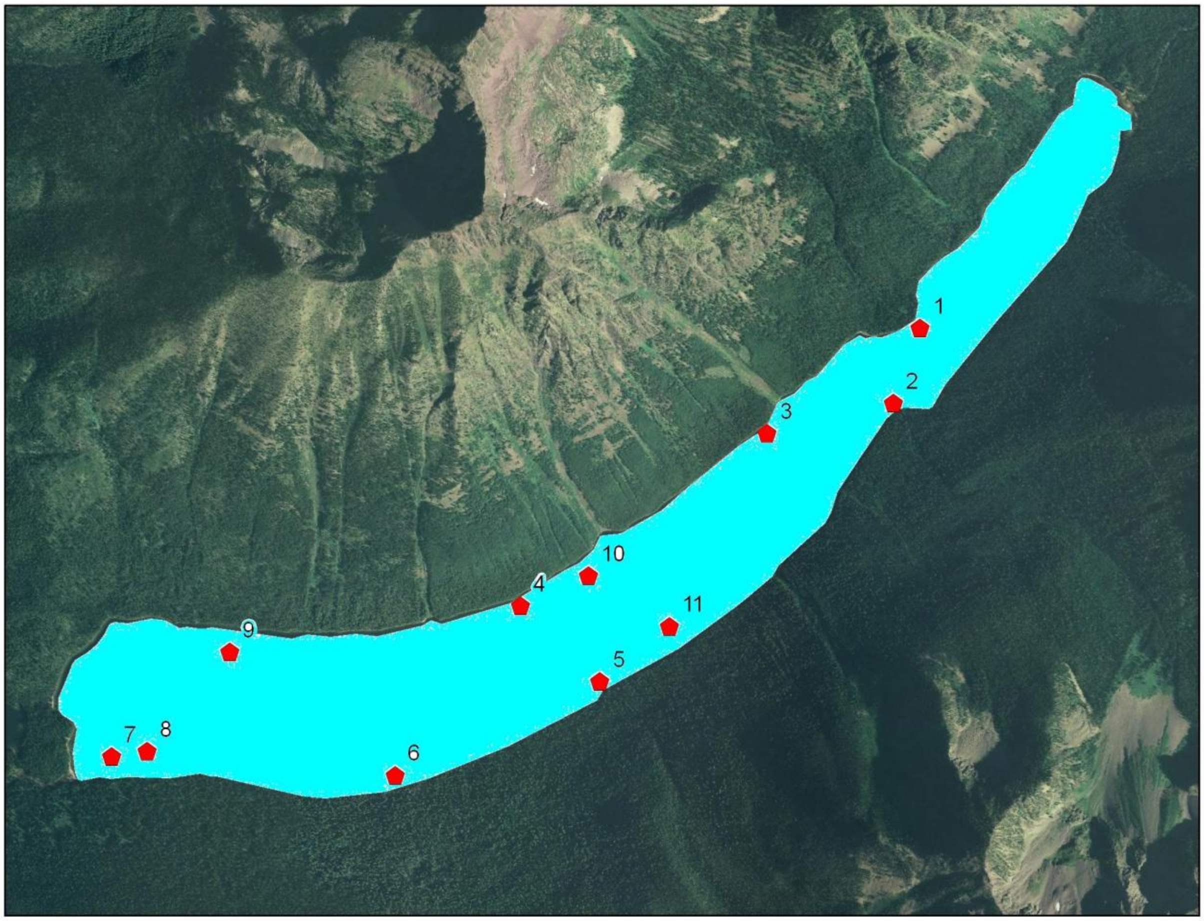

Figure 2. Locations of gill net sets in Quartz Lake, September, 2009.

RESULTS AND DISCUSSION

We captured a total of 217 fish comprising six species during gill netting efforts (Table 2). We captured only native species in net sets 1-6, which represent the historical sets for trend assessment. A general pattern of higher catches of most species occurred in 2005 and 2009 over earlier sampling years (Figure 3, Table 2). We captured 38 bull trout, 25 of which were mortalities (66%). Sampling in 2005 captured considerably more fish than the other years. This was largely driven by large catches of mountain whitefish, but catches were also highest for both bull and westslope cutthroat trout in 2005. Although catches were higher in recent sampling years, the proportion of westslope and bull trout in the total catch appears to have remained fairly constant.

Another measure of relative abundance that can be useful in long-term monitoring is catch-per-unit effort (CPUE), defined in our study as the number of fish captured per net-night. As our comparison across years involves identical numbers of nets set each year, it will mirror trends in time as interpreted from the total number of fish captured for each species. CPUE also suggested consistency over time in the relative abundance of bull and westslope cutthroat trout (Figure 5). In contrast, nearly twice as many suckers were captured in 1969 as any other year, with relatively large numbers of both largescale and longnose suckers in the catch. This may be due to seasonal changes in fish distribution, as the 1969 sampling occurred in June, versus September in all subsequent years. In addition, netting locations were also more consistent in later years due to the use of GPS technology which was not available in 1969. Differences in net set locations can directly influence catch composition due to differences in habitat preferences of each species in the lake. Although the intent was to duplicate the locations of the 1969 net sets, it is apparent when reviewing the data sheets from the 1969 sampling that the more recent net set locations (2000-2009) have been expanded to provide a better representation of the fish community than the earlier sets.

Table 1. Net set locations (NAD 83), depths, and set dates for gill nets set in Quartz Lake, Glacier National Park, in September 2009.

Net	Location description	UTM Zone	Easting	Northing	Date set	Time set	Date pulled	Time pulled	Depth range (ft.)
1	Last point on N. side	11 U	0714356	5413623	9/9	1816	9/10	830	15-67
2	Cr. draining valley south of vulture peak	11U	0714226	5413259	9/9	1840	9/10	845	15-75
3	Point E. of 2nd slide	11U	0713615	5413115	9/9	1809	9/10	815	13-58
4	N. shore - east of first slide	11U	0712421	5412279	9/9	1755	9/10	749	14-53
5	S. shore - 2nd big boulder	11 U	0712805	5411914	9/9	1749	9/10	900	18-79
6	Car sized boulder-south shore	11 U	0711814	5411459	9/9	1724	9/10	915	24-59
7	Patrol cabin	11U	0710442	5411549	9/9	1459	9/10	936	88-93
8	Patrol cabin 2	11U	0710615	5411575	9/10	1845	9/11	732	93-102
9	N. bank near campground	11U	0711015	5412056	9/10	1900	9/11	745	102-128
10	Avalanche 2	11 U	0712751	5412425	9/10	1955	9/11	805	107-139
11	Across from Avalanche 2 on South bank	11 U	0713143	5412181	9/10	2015	9/11	825	133-152

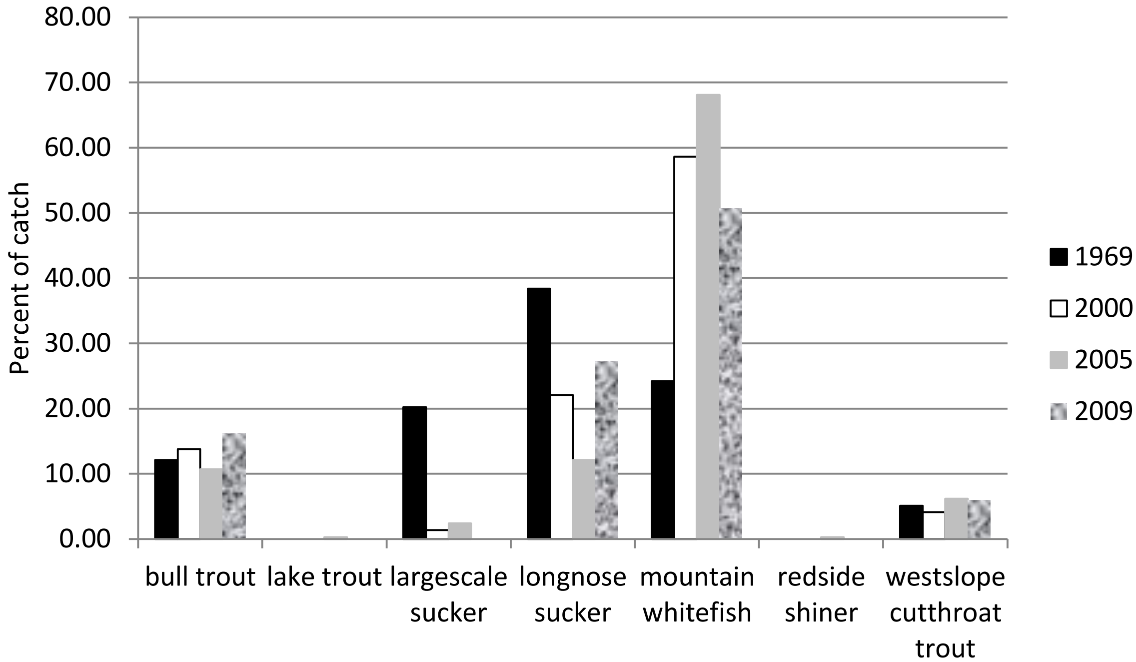

Figure 3. Catch composition of various species in standardized long-term sampling efforts on Quartz Lake, Glacier National Park, 2009.

Table 2. Historical catch composition comparison for six gill net sets in Quartz Lake, Glacier National Park.

Species	**1969**	**2000**	**2005**	**2009 (11 net total)**
bull trout (blt)	24	20	40	35 (38)
lake trout (lkt)	0	0	1	0 (10)
largescale sucker (csu)	40	2	9	0 (0)
longnose sucker (lns)	76	32	45	59 (68)
mountain whitefish (mwf)	48	85	254	110 (114)
redside shiner (rss)	0	0	1	0 (0)
westslope cutthroat trout (wct)	10	6	23	13 (13)
Total	198	145	373	217 (243)

Of note, is the relatively small number of largescale suckers in the catch since 1969. We are unsure if numbers have declined as the data suggests, or if species identification issues are responsible. In the 2009 sampling we likely identified some largescale suckers as longnose suckers. It would be most appropriate to view the 2009 data for the sucker catch as providing a combined catch for both species. Future sampling should more thoroughly evaluate species composition of the sucker catch. Sampling in 2005 caught many more mountain whitefish and westslope cutthroat trout than were captured in other years. This may represent strong year-classes, or more likely slight differences in sampling techniques/locations. In general, nets set in 2005 were set shallower, with the majority of the nets set with their nearshore ends in approximately 6’ of water. As a result, they would have been more likely to capture higher numbers of shoreline/shallow habitat associated species like westslope cutthroat trout and mountain whitefish than would have been captured in either 2000 or 2009.

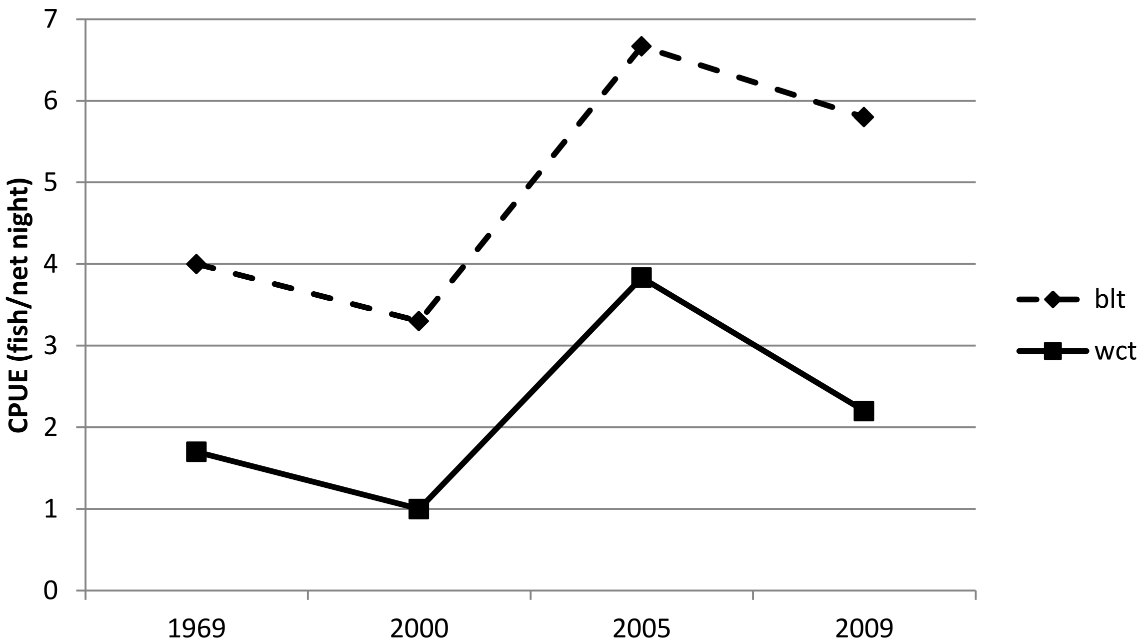

Figure 4. Catch-per-unit-effort for bull and westslope cutthroat trout captured in gill nets in Quartz Lake over time.

Aside from potential changes in sucker relative abundance, the data do not suggest consistent changes in native species relative abundance over time. It is most likely that the year to year variability in catch rates is likely a function of timing and the variability associated with setting a relatively small number of nets in a large lake and slight differences in net locations (depth). This variability captures the range of variability we can expect to see in the future as we continue the monitoring program.

We would not expect to see measurable changes in the fish community due to lake trout at this point, as it appears lake trout abundance remains relatively low. We did not capture any lake trout in the long-term monitoring net sets (nets 1-6), suggesting these shoreline oriented sinking gill nets are not well suited to detect lake trout presence at relatively low abundance in Quartz Lake. This is likely a function of the bathymetry of the lake, fish behavior, and the number of nets fished. All of the long-term sets were set shallower than 25m. Dux (2005) documented lake trout remaining below the thermocline during much of the summer in Lake McDonald, generally at depths greater than 20m. Although the deep ends of some of the earlier established net sets (nets 1-6) did extend deeper than 20m, the deepest of these sets only reached to 25m. We were able to capture lake trout consistently, but at relatively low abundance when we set the entire net deeper than 90'. Depth is a key factor affecting net success for lake trout during periods of lake stratification. Fishing gill nets in deeper water (>25m) has been identified as an effective method to reduce bycatch of non-target fish in lake trout removal efforts (W. Fredenberg, USFWS, personal communication). When set near the shore, our nets didn't reach the depths that lake trout prefer during periods of lake stratification in Glacier National Park. The additional five deep-water sets (nets 7-11) should be continued in future sampling to provide a better representation of changes in the fish community in Quartz Lake. Additional analysis of the shallow set data may identify opportunities to eliminate some redundant sets without effecting repeatability, and reduce mortality of non-target species.

Mean length for bull trout has remained fairly similar, ranging from 453.3 to 488 mm over the three sample years (1969 to present) (Figure 6). The same can also be said of westslope cutthroat trout and mountain whitefish. Average length for westslope cutthroat trout across all three years ranged from 342.1 to 355.4 mm, while it ranged from 205-235 mm for mountain whitefish over the same time period (Figure 6). A comparison of the bull trout length-weight relationship from 1969 to 2009 also suggests consistency over the 40 year period with respect to fish condition (Figure 7). A lack of change in mean length or condition suggests growth conditions and/or mortality rates have remained relatively stable. These will be important metrics to measure and compare over time as the lake trout suppression program progresses. In 2009, we estimated condition factor for bull trout (K) at 0.93 (SD = 0.1, n = 24) (Table 3). This is consistent with K estimated at 0.91 (SD = 0.1, n = 23) for bull trout captured in gill nets in 1969. Although we captured few lake trout in the standardized gill net sets, gill net removal data for lake trout from 2009 (C. Muhlfeld, USGS, personal communication) resulted in a relative weight (Wr) of 89.4 (CI=5.7; n=268). However, these data may be biased high because mature fish were in a pre-spawn condition (many captured in September and October). Fish condition can be biased depending on the reproductive status of individuals. We did not capture any bull trout that had mature gametes. Therefore, we considered our estimate of fish condition to be reasonably unbiased. Sample sizes limited our ability to determine year-class/ages from the length frequency histogram for either bull or westslope cutthroat trout, however it appears as though we sampled at least three different westslope cutthroat trout age-classes and five age-classes of bull trout (Figure 8).

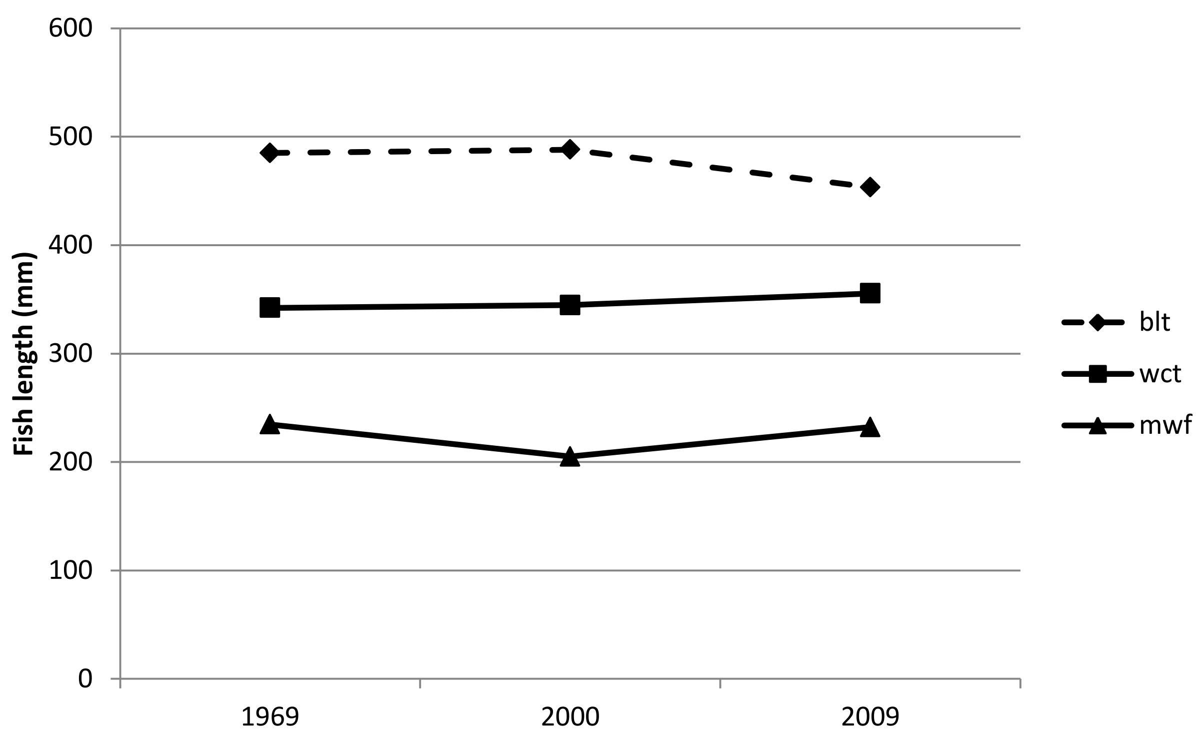

Figure 5. Mean length of bull and westslope cutthroat trout, as well as mountain whitefish over time in Quartz Lake gill net samples.

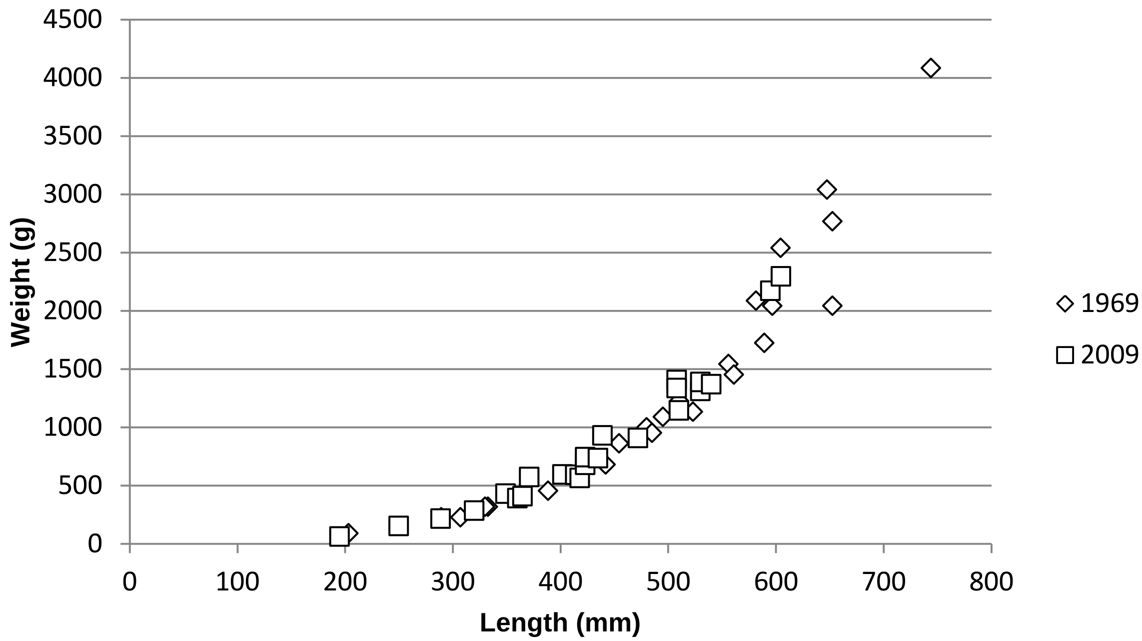

Figure 6. Length-weight relationship for bull trout captured in Quartz Lake in 1969 and 2009.

We did not capture lake trout in sets shallower than 88', and then captured lake trout consistently to a depth of 152'. Our highest catch of lake trout occurred in the deepest set (depth range = 133'-152'), and was comprised entirely of juvenile lake trout (length range = 255-333 mm). Bycatch of native species declined in deeper sets as well, although we did continue to catch low numbers of bull trout, mountain whitefish, and longnose suckers in sets > 100'. It is apparent that the existing six net sampling regime is not sufficient to detect changes in lake trout relative abundance over time at low population levels, and that the incorporation of the deeper sets will improve our long-term monitoring ability.

A four-year cooperative experimental lake trout removal project being led by the USGS was initiated in 2009. This effort resulted in the removal of 140 adult lake trout and 364 juvenile lake trout in 2009 (Muhlfeld and Fredenberg 2010). Based on recapture rates of sonic tagged fish (91%), this may represent a significant proportion of the adult lake trout population in Quartz Lake. However, without a population estimate, quantifying the effectiveness of the program at removing both juvenile and adult lake trout will be difficult. A lake trout population estimate is one component of the research project, and efforts to obtain the estimate will continue. Long-term native fish population monitoring, such as gill netting and bull trout redd counts will be key in determining the overall success of the project.

Table 3. Mean length, length range, mean weight and condition factor (K) for fish species during gill netting on Quartz Lake, Glacier National Park, in 2009.

Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)	K (SD)
blt	453.3 (103.8) (38)	195-603	862.1 (590) (24)	0.93 (0.10)
lkt	355.4 (141.7) (10)	255-680	N/A[1]	N/A
lns	312.7 (110) (68)	115-465	161.3 (129.5)(15)	1.13 (0.09)
mwf	232.2 (57.9) (127)	140-440	152.1 (165.2) (87)	0.93 (0.11)
wct	355.4 (141.7) (10)	261-391	452.1 (52.6) (7)	1.11 (0.06)

[1] Scale malfunction on day 2

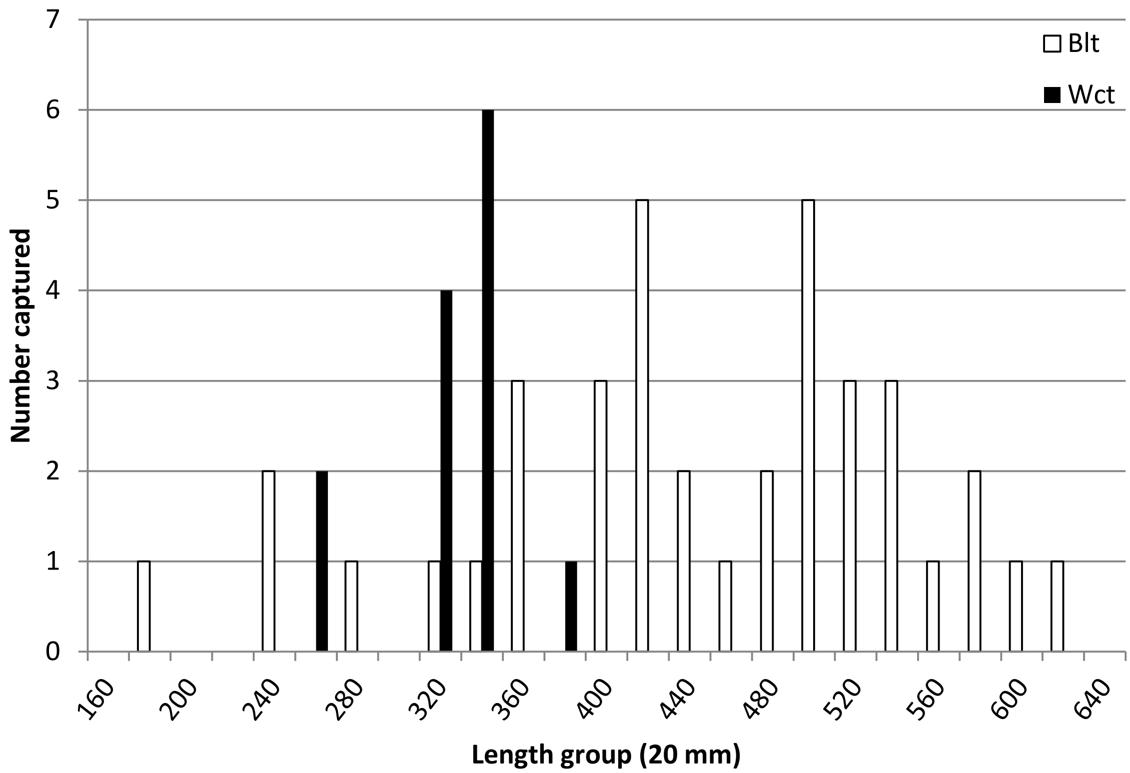

Figure 7. Length frequency for bull and westslope cutthroat trout captured in gill nets in Quartz Lake in 2009.

ACKNOWLEDGEMENTS

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Aquatic Invasive Species Risk Assessment, Monitoring, and Prevention

ABSTRACT

In 2009 we undertook a project to assess Glacier National Park's (GNP) risk of introduction and establishment of aquatic invasive species (AIS), most notably zebra and quagga mussels. In addition, we began a monitoring program for these AIS using artificial substrates on lakes where motorized boat use is permitted. Park staff conducted entrance station interviews conducted at the West Glacier, Polebridge, Two-Medicine, and St. Mary entrances to assess state of origin for a sample of boaters entering the park. In 2009, 436 interviews were conducted and boaters entered the park from 14 states and two Canadian provinces. Eight of these states have established populations of zebra and/or quagga mussels. Although the majority of boaters were from Montana, small numbers of boats entering the park from mussel positive states make GNP vulnerable to accidental introduction of zebra/quagga mussels via transport on trailered watercraft. In response, in 2010 GNP began a program of boater education and boat inspection to reduce this risk. Approximately 800 two-week boat launch permits were issued in 2010. Boats registered in 21 states entered the park in 2010. Thirteen of these states have documented zebra/quagga mussel populations. Several boat owners reported having last launched their boats in mussel positive waters before coming to the park, although the reported dates of last launch prior to visiting the park were three weeks to a month earlier. Review of 760 of these permits indicated 104 non-resident boats entered the park of which 68 were inspected for the presence of AIS. No AIS were found.

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INTRODUCTION

Aquatic Invasive Species (AIS) are non-native species that negatively impact aquatic ecosystems, as well as human services and uses. AIS can impact native species and their habitats through a number of mechanisms including competition, predation, displacement, habitat disruption, or the spread of disease or parasites. Biological invasions by non-native species have become so widespread that they are significant contributors to global environmental change (Vitousek et al. 1996). Non-native fish species have already had significant negative impacts on native fish populations within Glacier National Park (GNP) (Fredenberg 2002).

AIS such as zebra *Dreissena polymorpha* and quagga mussels *D. bugensis* present a growing worldwide problem. Native ecosystems rarely have established control mechanisms for such newcomers, and as such they often establish at the cost of native flora and fauna. Impacts from aquatic invasive species can be extreme and affect ecosystems, recreation, and economics. AIS infestations are generally permanent; prevention is the best strategy to combat them. Public education is critical because many groups of aquatic invasive species need humans to move upstream.

Likely first introduced into the Great Lakes via trans-ocean ballast water transfer in 1986, zebra mussels were subsequently discovered in Lake St. Claire in 1988 (Griffiths et al. 1991). Zebra mussels have had a dramatic impact on aquatic ecosystems as well as public use. Native to southern Russia, zebra mussels are efficient filter feeders and have the potential to reduce productivity of other aquatic species at higher trophic levels through lower trophic level competition for primary production (Ludyanskiy et al. 1993). The Quagga mussel, native to Ukraine, was not discovered in the Great Lakes until 1989 (Mills et al. 1996). Quagga mussels have been found to occupy deeper, colder areas in the Great Lakes than observed in their native range(as deep as 110m), broadening the potential impact area of these species from littoral to profundal areas of lakes (Mills et al. 1996). These mussel species can also adversely impact native bivalve species through competition or by colonizing them as a host substrate and smothering them (Ricciardi et al. 1998). Aside from biological considerations, economic cost associated with management of zebra mussels is significant. It has been estimated that between 1989 and 2004, power generating and water treatment facilities in North America incurred approximately $267 million in total economic costs dealing with zebra mussels (Connelly et al. 2007). Zebra and quagga mussels have continued to move south and west from their initial introductions and threaten to compromise native aquatic ecosystems across the west. Zebra mussels have been found on trailered watercraft in Montana and Washington (http://nas.er.usgs.gov/taxgroup/mollusks/zebramussel/) and recent plankton samples collected from nearby Flathead Lake that contained organisms that resembled exotic mussel veligers resulted in elevated concern over the potential for introduction of zebra and quagga mussels to the Flathead Basin (MFWP 2010, 2011).

Other AIS threaten GNP as well. Plant species such as Eurasian watermilfoil *Myriophyllum spicatum*, purple loostrife *Lythrum salicaria*, and others are present within a three hour drive of the park, and New Zealand mudsnails are present in southwest Montana. Taken together, the potential transport and establishment of additional AIS into park waters is a serious threat. In response, park managers are taking proactive steps to reduce the risk.

METHODS

2009

We began a project to understand GNP's vulnerability to the introduction of additional AIS, and develop basic monitoring capabilities. A simple survey was developed and implemented at various park entrances to determine the registration state for boats entering the park (as well as county if Montana resident registration), as well as the waterbody where the boat was last launched. The survey was implemented during summer months by entrance staff as time permitted. Boaters were asked 2 to 3 questions as they entered the park. The survey design was simple to avoid increasing vehicle entrance times or put significant additional work burden on entrance staff. Entrance stations included in the survey were West Glacier, Polebridge, Two Medicine, and St. Mary (Figure 1). These are the entrances that are most likely to serve the trailered boating public. Upper Waterton Lake is also a primary boat-use area, but the entrance is located in Canada (Waterton Lakes NP), and as such was not included in the survey. Motorized boat use is also permitted on Sherburne Reservoir, but lack of a developed boat launch ramp limits its use.

In addition, we deployed artificial substrates in selected park lakes most heavily used by the boating public. These substrates were generally deployed at the beginning of the boating season and removed at the end of summer. They were inspected for the presence of AIS upon removal for the season.

2010

In 2010, the program was expanded to a pilot AIS prevention program that included boat permitting and inspection. The prevention program was structured such that it required a mandatory boat launch permit for all motorized boats. It was incorporated into the Superintendent's Compendium so that the provisions of the permit requirement could be enforced by NPS law enforcement staff.

The program was structured to use existing staff and the back-country permit office to issue the permits and facilitate the inspection process. We developed a flow chart to facilitate implementation of the permit and inspection process by park staff (Figure 2) and a permit to serve as the inspection and permitting checklist (Figure 3). The permit also provided a template to collect additional survey information on boat transport, as well as provide an educational component requiring boaters to certify that their boat is clean, drained, and dry. GNP staff attended a two-day boat inspection and decontamination certification training hosted by the Colorado Division of Wildlife at Curecanti National Recreation Area in order to return and properly train additional GNP boat inspection staff.

A public education effort regarding the risks of AIS to GNP waters was also initiated in 2010. The Crown of the Continent Learning Center at GNP developed a Resource Bulletin intended for the public addressing AIS threats to GNP (http://www.nps.gov/glac/naturescience/ccrlc.htm). The bulletin is brief (two pages) and is intended to provide key information regarding the status of AIS in GNP, as well as how the public can help protect the park from additional AIS. GNP also added AIS prevention content to its website (http://www.nps.gov/glac/planyourvisit/outdooractivities.htm) so boaters and other visitors would be exposed to the “clean, drain, dry” message before visiting the park. The AIS message was also incorporated into the Waterton-Glacier Guide brochure, available to the public. In addition, GNP is collaborating with the NPS Rocky Mountain Cooperative Ecosystem Studies Unit, the University of Montana and the Crown Managers Partnership to produce a color pocket guide to AIS that are either present or threatening the Crown of the Continent Ecosystem. This will be an excellent public and resource management agency education work product.

We deployed artificial substrates in Bowman Lake, Lake McDonald, Two Medicine Lake, and St. Mary Lake in order to monitor for invasive mussel presence. Electronic temperature recorders were installed in conjunction with the artificial substrates to characterize the summer thermal regime of shallow areas of the lakes (littoral zone). The temperature data, along with available data on calcium levels in park waters was used to characterize the risk of establishment of invasive mussels. An artificial substrate was also deployed by Waterton Lakes National Park staff at the marina boat launch at Upper Waterton Lake.

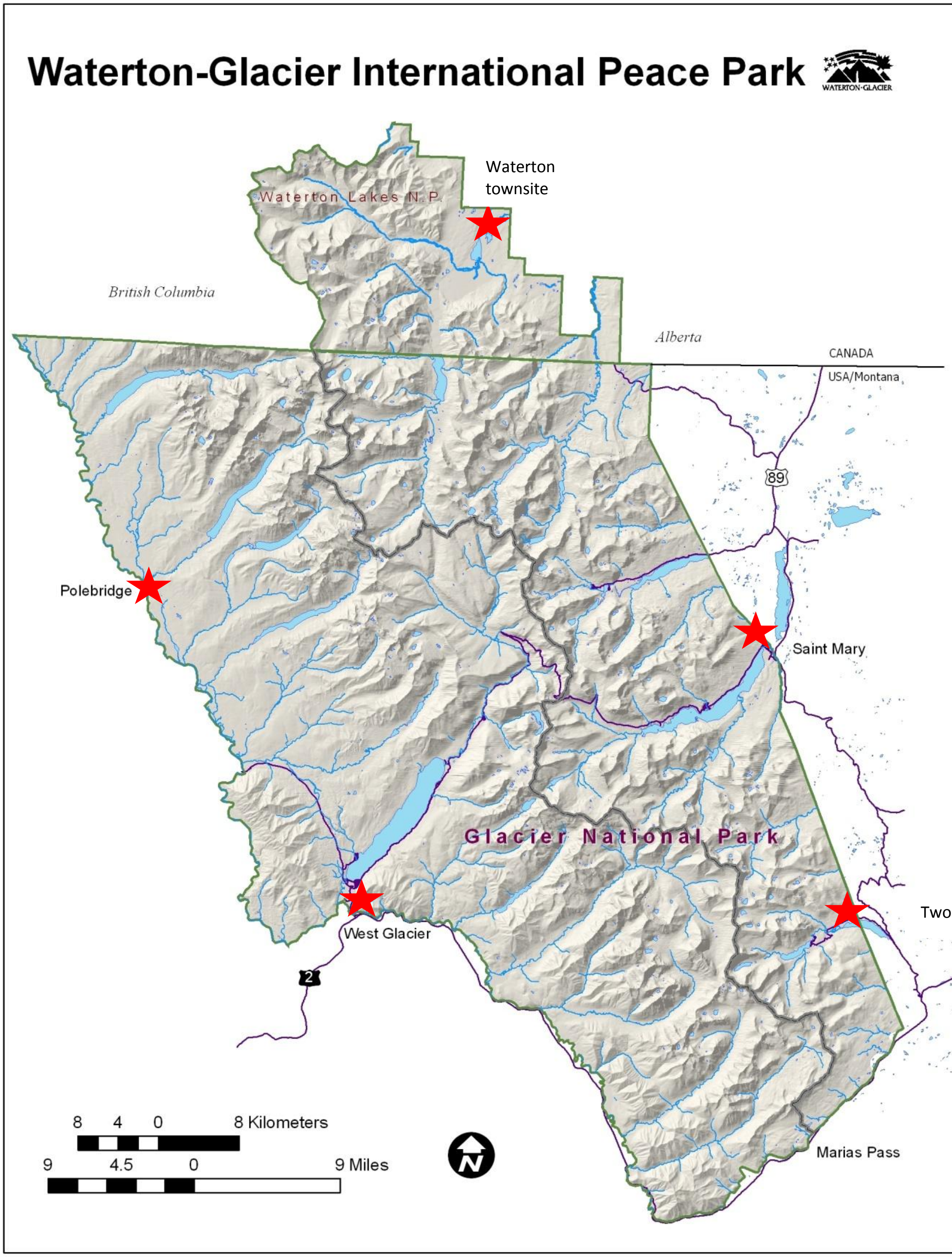

Figure 1. Primary entrance station locations providing access to park waters for motorized boat use.

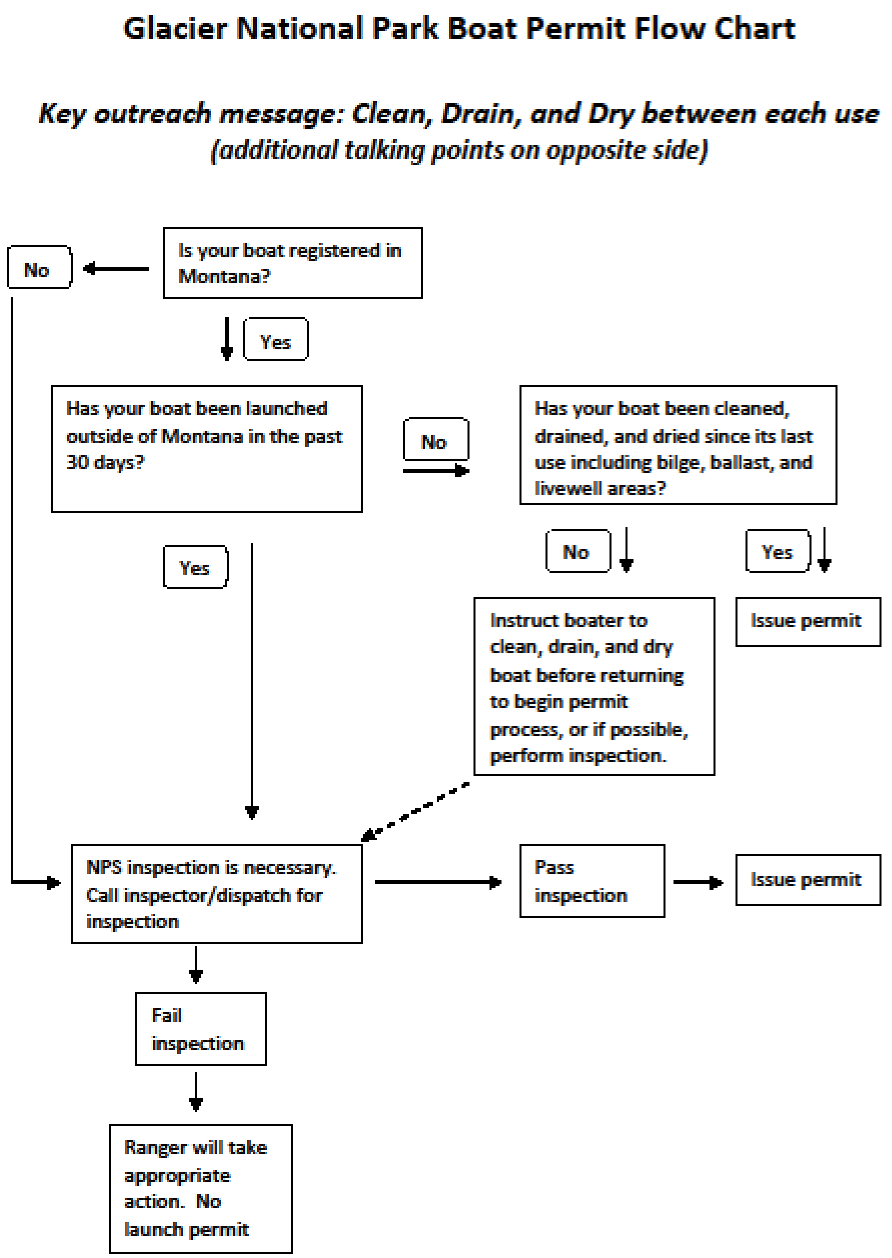

Figure 2. Boat inspection flow chart used in GNP AIS prevention program in 2010.

Operator Name______________________________ **Date/Time:** ______________________________ **Location:** ______________

Vessel Registration # ______________________ **Vehicle Tag #:** ___________ **Trailer Tag#** ______________________

Inspection Required_______ ❑ **Y** _______________ ❑ **N**_______________ **Inspected By/Issued by** ______________________

Reason For Inspection (Check All that apply)

❑ Out of State registered or used out of state within the last 30 days

❑ Been in infested waters within the last 30 days: **Name /State of body of water** ______________________________

❑ Boat has not been cleaned and dried since its last use outside of Glacier National Park

❑ Other ____

Vessel Inspection: Overall Look and feel (check one)		❑ Clean/Smooth	❑ Bumpy/Sandpaper	❑ Other
❑ **Exterior Checked**				
❑ Hull	❑ Trim tabs (Top and bot.)	❑ Through Hull fittings	❑ Water Intakes/outlets	❑ Transom
❑ Transducers	❑ Pilot tubes	❑ Anchors and ropes	❑ Recessed bolts	❑ Water holding pockets ❑
Motor well	❑ Depth Sounders	❑ Lights	❑ Cavitations Plates	
❑ **Motor Checked**				
❑ Exterior Housing	❑ Propeller and assembly	❑ Propeller Shaft	❑ Propeller guard	❑ Rudders
❑ Propeller shaft supports	❑ Propulsion System	❑ Gimbal area	❑ Lower Unit	❑ Water intake
❑ **Trailer Checked**				
❑ License Plate	❑ Rollers, Bunks, Pads	❑ Lights / Wiring	❑ Axels	❑ Trailer Springs
❑ Fenders	❑ Wheels and tires	❑ Hangers		
❑ **Interior/ Equipment Checked**				
❑ Bait and Live wells	❑ Internal Ballast tanks	❑ Float cushions/belts	❑ Ropes/equipment Locke	❑ Anchors
❑ Water Skis or ropes	❑ PFD's	❑ Nets		
❑ **Vessel Thoroughly Drained and Dry**				
❑ Bilge plug or pump	❑ Bait and live wells	❑ Ballast tanks		

Inspection Findings: (Check all that apply)

❑ **Did not find any identified or suspected ANS species** Permit Issued ? ❑ Y ❑ N

❑ **Found:**

❑ Large volume of water (Boat history not high risk) ❑ Vegetation ❑ Caked on Dirt or Mud

(Boat operator should be instructed to clean the vessel and re-apply for permit. Encourage full drying before re-applying)

❑ **Found:**

❑ Large volume of water (recent history in infested waters) ❑ Mussels (Contact Law Enforcement.)

(Consider impoundment for actual mussels. Standing water must be drained, dry, sterilized. Consider decontamination)

Fold Here Fold Here

Glacier

National Park Service
Department of the Interior
Glacier National Park

Boat Launch Permit

Issue Date ______________________ Permit

Display permit this side up, on the dashboard of your tow vehicle.

Pursuant to the United States Code of Federal Regulations, Title 36, Section 1.5, Glacier National Park requires a permit to launch any motorized vessel. Failure to comply with the permit requirement or violating the terms and conditions of a permit is punishable by fines up to $5000, or 6 months in jail.

Knowingly launching or transporting a boat contaminated with an Aquatic Invasive Species is prohibited by Federal and State Law.

By signing below I acknowledge that I understand the risks posed by the introduction of Aquatic Invasive Species to Glacier National Park. I understand how they may be transported on my vessel. I certify that my vessel is free of Aquatic Invasive Species.

Check one:

❑ I certify that I am a Montana Resident and that I have not launched my boat in any waters outside of the State of Montana in the last 30 days. I certify that my vessel is clean and dry, and that I have inspected it for Aquatic Invasive Species.

-OR-

❑ I certify that I am a Montana Resident and that I have launched my boat in waters outside of the State of Montana in the last 30 days. I certify that my vessel is clean and dry, that I have inspected it for Aquatic Invasive Species.

-OR-

❑ I Certify that I am not a resident of Montana, but my boat is clean and dry, has not been in any waters known to have Aquatic Invasive Species in the past 30 days.

State and water last launched in ______________ Date of last launch

Boater Signature ______________ Date

STOP AQUATIC HITCHHIKERS!™

Figure 3. GNP boat launch permit used in GNP AIS prevention program in 2010.

RESULTS AND DISCUSSION

2009

Between June and early October, 2009, a total of 369 interviews were conducted of boaters entering the park. The majority of the entrance interviews were conducted at the park's West Glacier entrance, where most of the boating traffic enters the park. In general, the majority of boaters surveyed were from Montana (Figure 4), and of those, the majority were Flathead County residents (Table 1). Sampling effort was not standardized as some entrance stations may have sampled a higher proportion of boaters entering, but boats were documented entering the park from a large number of Montana counties with a broad geographic footprint.

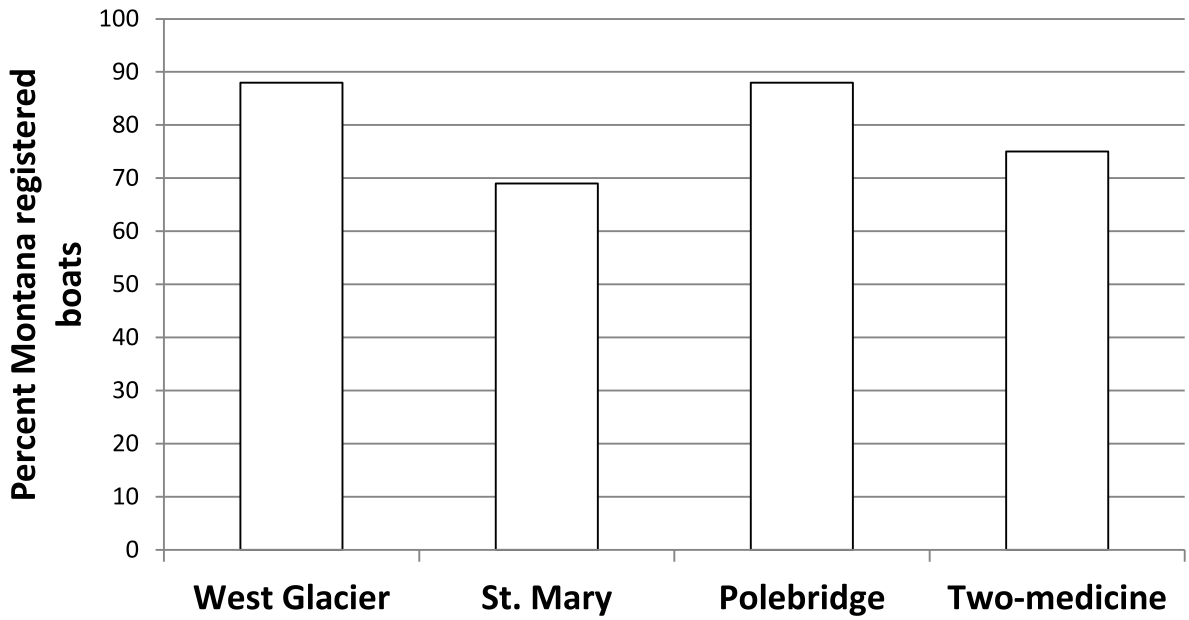

Figure 4. Percent of boats with Montana registration entering Glacier National Park by location in 2009.

Boats entered the park from at least 14 states and two Canadian provinces in 2009. Eight of the fourteen states are known to have zebra and/or quagga mussel populations (Figure 5). This is not intended to suggest that the boaters from these states had recently been in infested waters in their home states, but it does identify the risk of transport of zebra mussels from these states to GNP via trailered watercraft. However, as part of the survey one boater indicated that their boat had been previously launched in a zebra mussel positive water in New York (Cayuga Lake).

Table 1. Number of boats interviewed entering Glacier National Park by county in 2009.

County	Number of entrances
Cascade	3
Flathead	237
Gallatin	3
Glacier	2
Hill	3
Lewis and Clark	1
Liberty	1
Lincoln	1
Meagher	2
Missoula	4
Pondera	2
Powell	1
Ravalli	1
Silverbow	1
Teton	2
Toole	3
Yellowstone	2

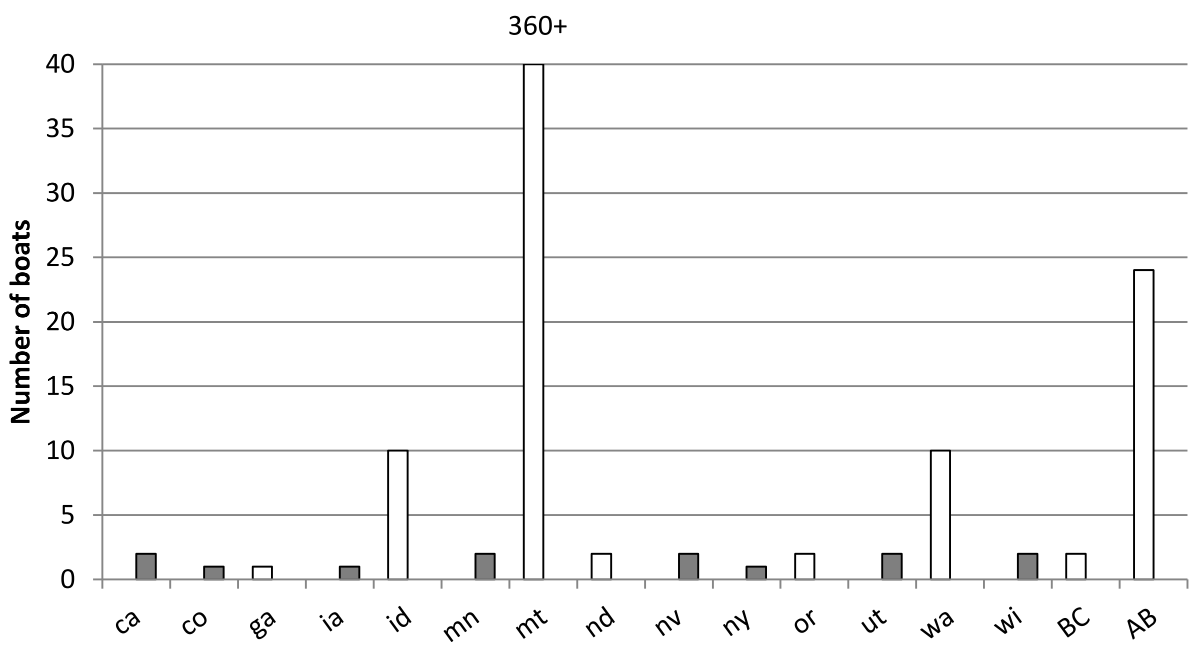

Figure 5. Boat registration origins for boaters surveyed entering the park in 2009. Shaded bars represent States known to have zebra and/or quagga mussel populations.

2010

Following the results of the 2009 AIS Risk Assessment, GNP developed and implemented a pilot boat launch permit system that included both self-certification that boats were clean, drained, dry, and AIS free, as well as boat inspection for the presence of AIS. The boat permit system was adopted into the Superintendent's Compendium so that the regulation could be enforced by NPS staff.

The program was structured such that boat launch permits were required to launch any motorized boat in GNP. This provided an opportunity for face to face communication with every boater entering the park to communicate the issue of the AIS threat to GNP and raise awareness of the importance that boaters have clean, drained, and dry boats, as well as providing the opportunity for targeted boat inspection for high-risk boats. The majority of boating traffic/use is associated with the Lake McDonald area, and the permit system was implemented out of the Backcountry Permit Office in Apgar, using existing staff and volunteers. At the other entrance stations where boat traffic volume is lower (Bowman, Two Medicine, and St. Mary), existing staff were used to issue permits and inspect boats. We collaborated with Waterton Lakes National Park (WLNP) to ensure that boats from mussel-positive states were inspected prior to launch in the Waterton Lakes chain.

Prior to implementing the boat permit/inspection program, it was necessary to adequately train NPS staff and volunteers. Multiple AIS inspection trainings were held on both sides of the park, as well as in WLNP for WLNP staff. Inspection training was initially provided by the Montana Department of Agriculture and Montana Fish, Wildlife, and Parks. GNP assumed the role of primary inspection training for GNP staff and volunteer personnel following certification training in Colorado. A total of 66 NPS employees and eight volunteers were trained in AIS inspection in 2010. Trainings involved both classroom education in AIS identification and inspection methods, as well as "mock" inspections of watercraft. Trainings typically lasted about 2-3 hours.

When a boat entered the park, entrance staff would advise the boater that a free launch permit was required prior to launching a motorized boat in GNP. They would be directed to the appropriate location to obtain the permit. Permit writing staff would then use the flow chart to determine the risk-level of the boat, and then either issue the permit or conduct an inspection of the boat. If the boat was inspected and no AIS were found, a launch permit was issued. If AIS were found, law enforcement staff were to be contacted to address the situation as appropriate. The permit was valid for 14 days from the date of issue to balance the risk of boaters traveling to other waters during the permit period and subsequently returning to the park without a new inspection, convenience for the boating public, as well as GNP's ability to implement the program within existing staffing limits. As part of the permitting process, all boaters were required to certify that there boat was clean, drained, and dry.

A total of 802 permits were issued and entered into a boat permit database. Seven hundred and sixty hard copies of these permits were reviewed for completeness and analyzed for AIS inspection results and boater use patterns. There were cases where Boat Permit forms were incompletely filled out, or boats that should have been inspected according to the flow chart were not. Approximately 104 permits were issued to non-resident registered boats, yet based on the completed forms, 36 of these did not appear to have been inspected prior to issuance of the permit. Some of these 36 permits indicated the boat had been recently inspected for AIS by another state, or the boater certified that their boat was cleaned, drained, and dry, and had not been launched in any AIS positive waters in the past 30-days. Four boats had AIS inspection paperwork from other states. One had been inspected in Colorado and three others in Idaho. None of these four boats was re-inspected. A total of 24 park concessioner boats were also inspected and permitted. No AIS were detected on boats or trailers, however one boat was not granted a permit because it needed to be cleaned before inspection/launch. We do not know if this boat returned at a later date to complete the permitting process.

Some irregularities issues were associated with the first year of implementation of such a large program. In addition, the program was rapidly developed and implemented. Better training and familiarity with the program will lead to improvements in its future implementation. A primary limitation of the program is implementation with existing staffing levels, particularly at the Apgar boat ramp on Lake McDonald. The Backcountry Permit Office, Visitor Center, and boat ramp are all located in a very congested and busy setting. Staff were overwhelmed at times by visitors needing boat permits/inspections on top of their normal work demands for backcountry permits and other visitor needs/services. It is likely this played a role in the quality of implementation of the permit and inspection process. The use of volunteers at the Apgar Boat Ramp helped alleviate some of this problem, but the implementation of the program would clearly benefit from dedicated NPS AIS supervisory, inspection, and permitting staff at Apgar during the boating season.

The majority of boats were permitted (over 500) at the GNP's west entrance boat launch area (Apgar/Lk. McDonald). Of the remaining entrances, Two Medicine issued the most permits (84) followed by St. Mary (60), Polebridge (54), and Many Glacier (6). The majority of the motorized boats using the park were Montana registered (86.5%). However, boats registered from 21 states entered the park in 2010 (Figure 6). Of these states, 13 have zebra and/or quagga mussel populations. Several boats indicated they had last launched in mussel-positive waters (i.e. Fox River in Illinois, Bass Lake in Wisconsin, and Houghton Lake in Michigan). These launches were reported to have occurred three weeks to two months prior to launching in GNP. Data suggests that adult zebra mussels can live out of water for up to 30 days under the right conditions. They live longer out of water during cool, wet weather and die quicker if the weather is hot and dry. Veligers, which are the larval form of the mussels, can live in water in livewells, bilges, and ballast tanks. However, little is known about how long they can live, although it is suspected that they would not survive well under some conditions. The primary boating season in GNP is summer, so weather conditions should be less than ideal for long-term survival of adult zebra mussels on boats/trailers.

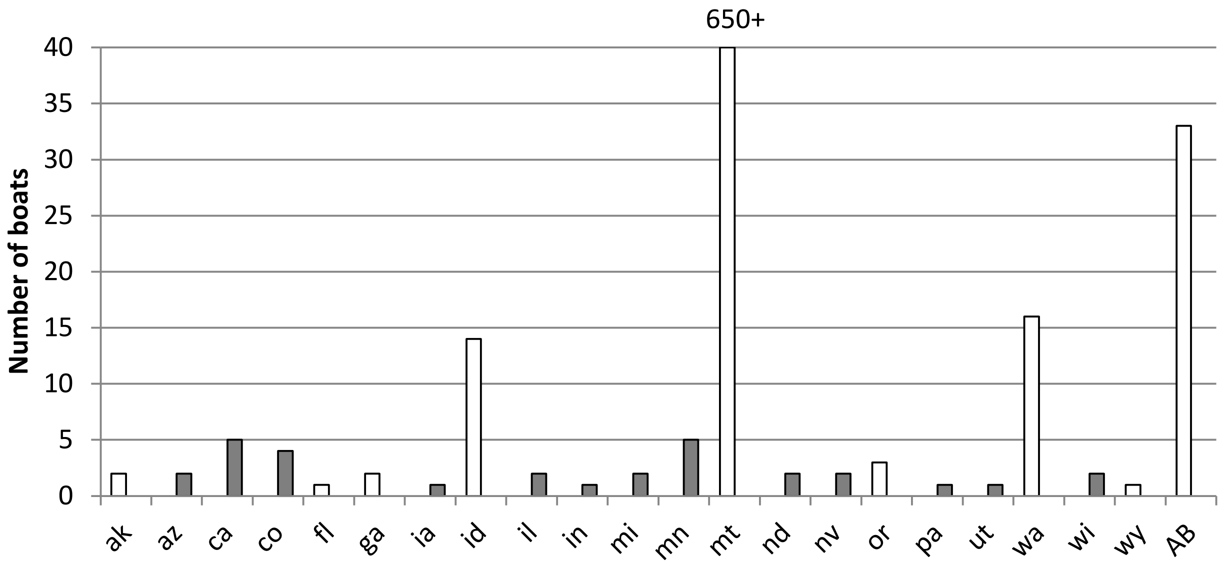

Figure 6. Boat registration origins for boaters entering the park in 2010. Shaded bars represent States known to have zebra and/or quagga mussels.

Some boaters indicated they had last launched in waters (e.g. Lake Pend Oreille) with other AIS, such as Eurasian watermilfoil less than a week before launching in GNP. Nearby Flathead Lake is home to other aquatic/wetland AIS such as flowering rush *Butomus umbellatus*, purple loosestrife *Lythrum salicaria*, and curly-leaf pondweed *Potamogeton crispus*. Resident boaters commonly move between Flathead and Lake McDonald on a daily or weekly basis.

Habitat suitability for invasive mussels appears to be marginal in GNP waters available to trailered watercraft. Key variables believed to effect zebra/quagga mussel colonization and population establishment include water temperature, pH, calcium levels, dissolved oxygen, and salinity. Neither dissolved oxygen nor salinity would limit mussel establishment in park waters. The key remaining variables, temperature, pH, and calcium levels are available on park waters for use in evaluating general habitat suitability. In general calcium levels lower than 9-12 mg/l have been used as a threshold between little to no colonization potential and potentially suitable for colonization for reasons including calcium requirements for shell growth (for review see Cohen and Weinstein 2001). Waters with calcium levels greater than 20 mg/l are generally considered suitable for zebra mussels (Cohen and Weinstein 2001). Less is known about the suitability of waters with calcium levels between 12-20 mg/l and the ability of exotic mussels to establish populations within this range. A pH of between 7.4 and 8.7 appears to be ideal for zebra mussels (Cohen and Weinstein 2001). Spawning, successful veliger development, and adult shell growth has been documented to occur at temperatures at or above 12^{0}C (McMahon 1996). I used these habitat suitability data to develop a table to evaluate mussel colonization risk for each of the study waters (Table 2).

Table 2. Habitat suitability and risk criteria for zebra mussels adapted from Cohen and Weinstein (2001). Water chemistry data taken from Ellis et al. (1992).

Metric	Lower	Moderate	Higher
Calcium (mg/l)	12-15	15-20	>20
pH	<7.3 or > 9	7.3-7.5 or 8.7-9.0	7.5-8.7
Mean summer T (C)	<12	12-15	>15
DO (mg/l)	<4	4-8	>8
Motorized boat use	low	medium	high

The habitat suitability metrics vary across park waters. For McDonald, St. Mary, and Waterton lakes pH ranged from approximately 7.5 to 8.5 throughout the summer months across a range of depths, putting it within the optimal range for zebra mussels (Ellis et al. 1992). In general, pH in Two Medicine Lake is slightly lower than the other lakes evaluated by Ellis et al. (1992), but would still be characterized as being suitable, particularly in the upper 10m of the water column. Similarly, dissolved oxygen levels across the study lakes exceeded criteria for highly suitable habitat (Ellis et al. 1992). Water temperatures appear to be moderately suitable for zebra mussel population establishment as well (Figures 7-10).

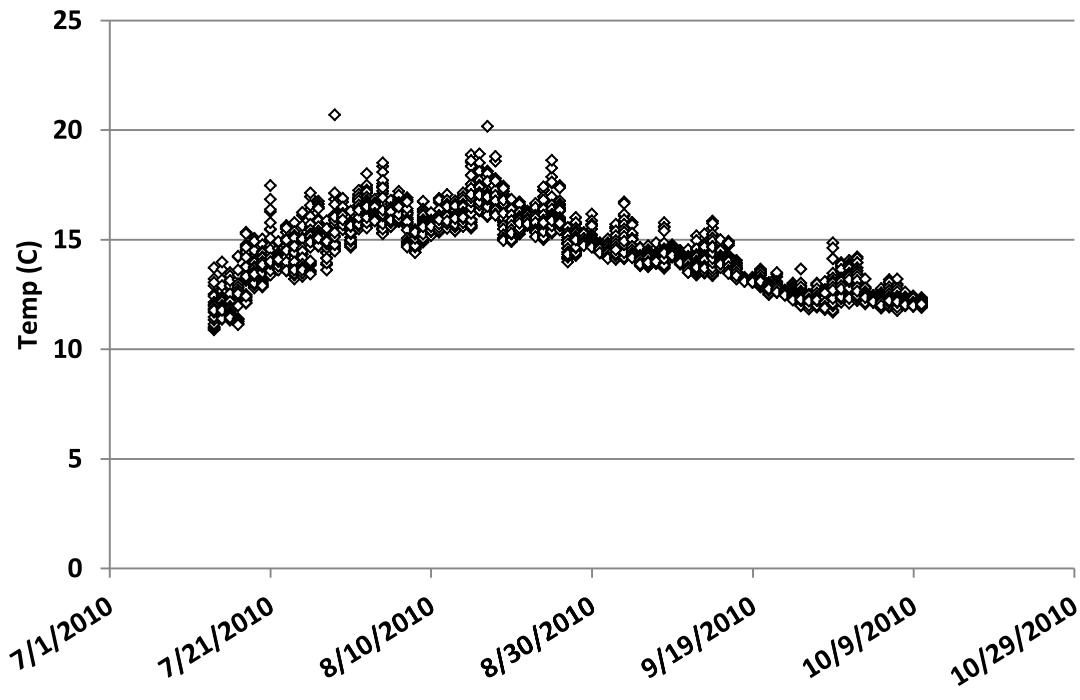

Figure 7. Water temperature measured near the lake bottom in the nearshore zone of Bowman Lake during 2010.

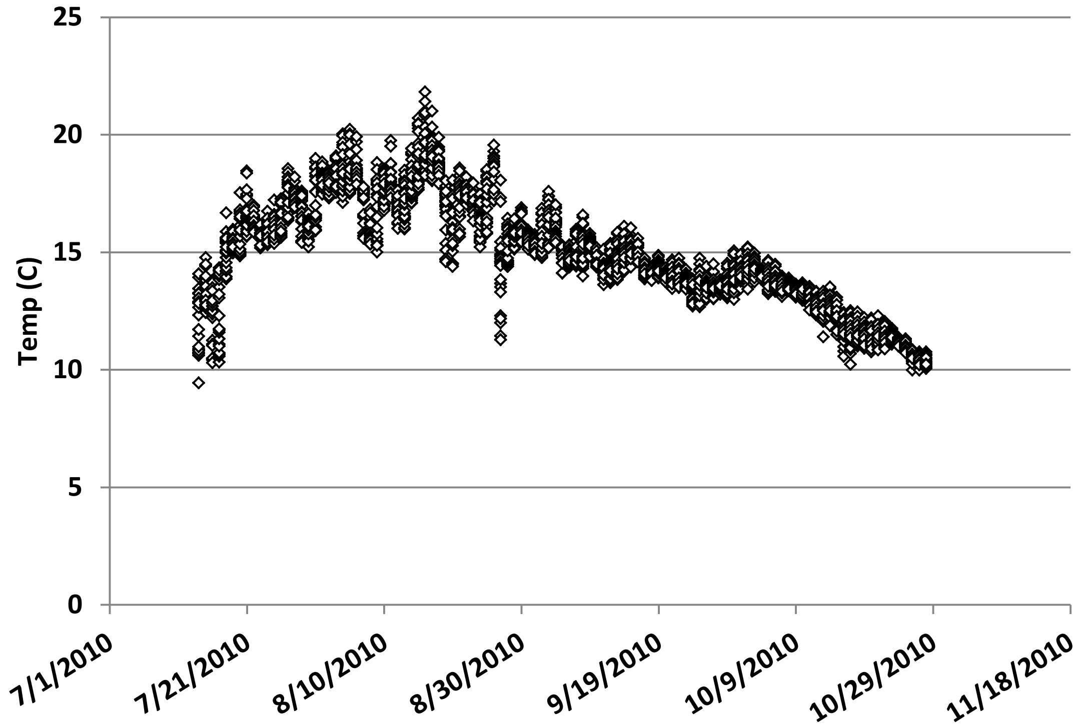

Figure 8. Water temperature measured near the lake bottom in the nearshore zone of Lake McDonald during 2010.

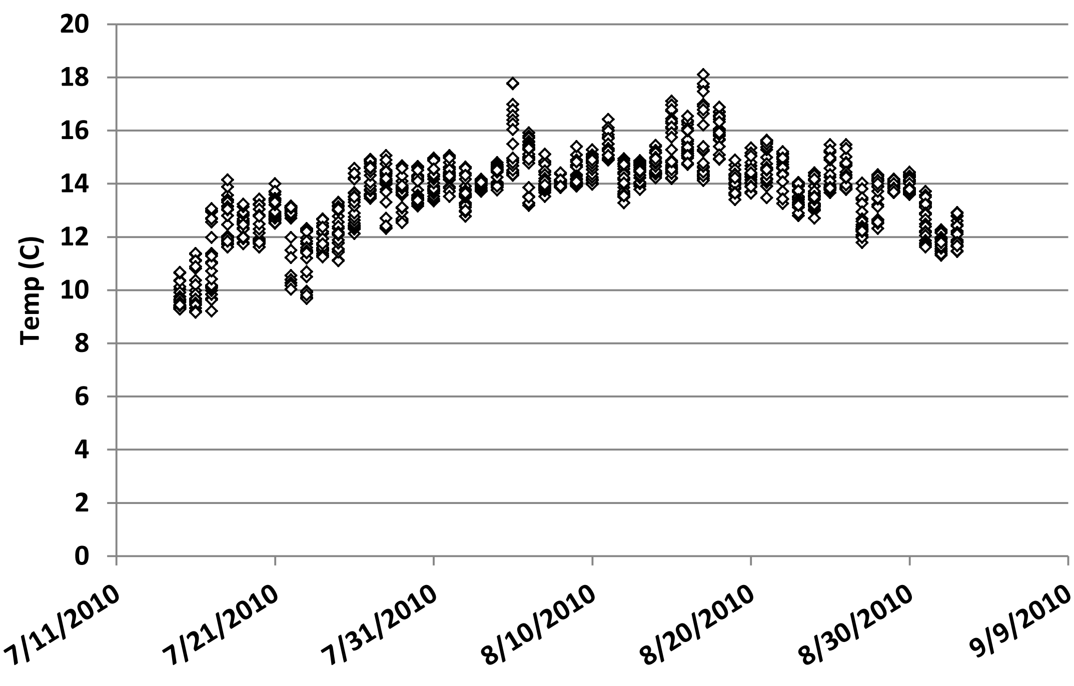

Figure 9. Water temperature measured near the lake bottom in the nearshore zone of St. Mary Lake during 2010.

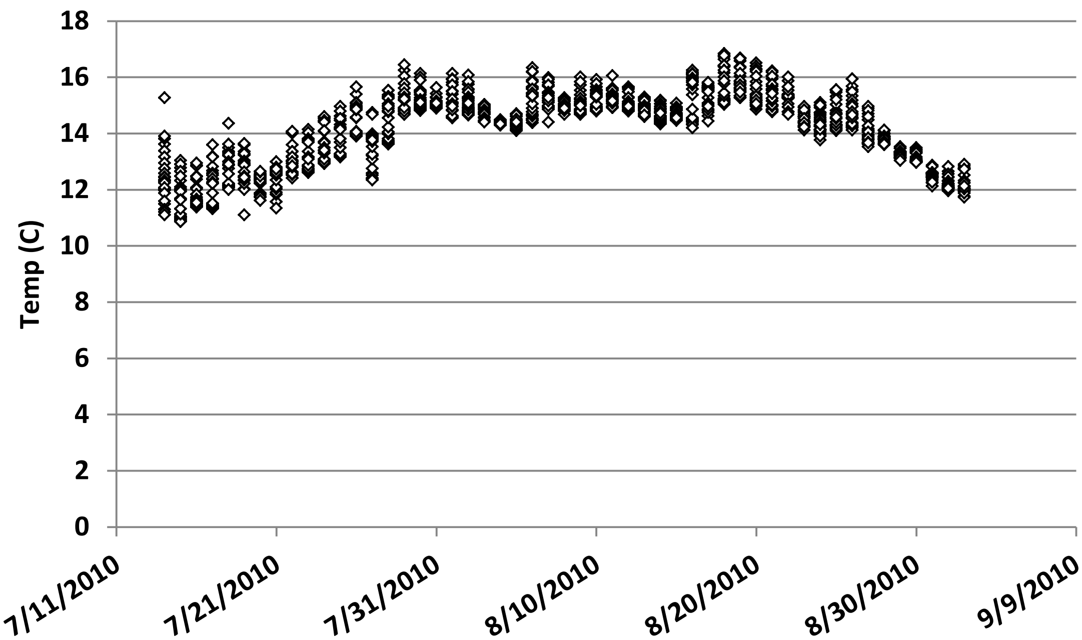

Figure 10. Water temperature measured near the lake bottom in the nearshore zone of Two Medicine Lake during 2010.

We used existing information on habitat parameters and exposure risk to evaluate individual water-body risk for introduction and establishment of zebra mussels. We use the lowest rated risk factor to set the overall risk level for a body of water. When the four study lakes are evaluated using the criteria in Table 2, most fall into the "moderate" category (Tables 3-7). On a relative, within park scale, Lake McDonald and Waterton Lake are the highest risk waters, while Two Medicine and Bowman lakes would be considered the lowest. This is largely due to relatively high boat traffic on Lake McDonald and Waterton Lake, relatively low motorized boater use on Bowman Lake, and low calcium levels in Two Medicine Lake.

Table 3. Zebra mussel introduction and colonization risk assessment for Bowman Lake. Highlighted cells represent values for each metric.

Metric	**Lower**	**Moderate**	**higher**
Calcium (mg/l)	<12-15	15-20	>20
pH	<7.3 or > 9	7.3-7.5 or 8.7-9.0	7.5-8.7
Mean summer T (C)	<12	12-15	>15
DO (mg/l)	<4	4-8	>8
Motorized boat use	low	medium	high

Table 4. Zebra mussel introduction and colonization risk assessment for Lake McDonald. Highlighted cells represent values for each metric.

Metric	Lower	Moderate	higher
Calcium (mg/l)	<12-15	15-20	>20
pH	<7.3 or > 9	7.3-7.5 or 8.7-9.0	7.5-8.7
Mean summer T (C)	<12	12-15	>15
DO (mg/l)	<4	4-8	>8
Motorized boat use	low	medium	high

Table 5. Zebra mussel introduction and colonization risk assessment for St. Mary Lake. Highlighted cells represent values for each metric.

Metric	Lower	Moderate	higher
Calcium (mg/l)	<12-15	15-20	>20
pH	<7.3 or > 9	7.3-7.5 or 8.7-9.0	7.5-8.7
Mean summer T (C)	<12	12-15	>15
DO (mg/l)	<4	4-8	>8
Motorized boat use	low	medium	high

Table 6. Zebra mussel introduction and colonization risk assessment for Two Medicine Lake.

Metric	Lower	Moderate	higher
Calcium (mg/l)	<12-15	15-20	>20
pH	<7.3 or > 9	7.3-7.5 or 8.7-9.0	7.5-8.7
Mean summer T (C)	<12	12-15	>15
DO (mg/l)	<4	4-8	>8
Motorized boat use	low	medium	high

Table 7. Zebra mussel introduction and colonization risk assessment for Waterton lakes. No continuous temperature data exists for Waterton lakes, however we used point water temperature data from Ellis et al. (1992) and Anderson (1976) to assign a water temperature risk category.

Metric	**Lower**	**Moderate**	**higher**
Calcium (mg/l)	<12-15	15-20	>20
pH	<7.3 or > 9	7.3-7.5 or 8.7-9.0	7.5-8.7
Mean summer T (C)	<12	12-15	>15
DO (mg/l)	<4	4-8	>8
Motorized boat use	low	medium	high

2010 saw the beginning of more aggressive efforts by the NPS to prevent additional AIS from colonizing park waters. Initiation of this effort is very timely given the westward movement of AIS such as zebra and quagga mussels, and other AIS. This program should be continued into 2011 and beyond and steps should be taken to improve its effectiveness. Increased frequency of boat inspection, inspecting boats regardless of State of registration, significant program quality control oversight and implementation, and a broadened monitoring effort would likely be reasonably effective at substantially reducing the risk of accidental transport of AIS to GNP waters.

Consideration of type of watercraft permitted for use in the park under any inspection and permit program would also be important. Some boats and trailers are more complicated than others. For example, fairly recent innovations in ski and wakeboard boats have incorporated internal water ballast chambers/tanks to increase the water displacement of a boat for enhanced wake creation. Water is pumped from a waterway into holding areas within the boat to fill the tanks and discharged after use. These ballast areas are reported to be difficult to inspect, clean, fully drain, and dry. Similarly, other types of watercraft likely pose lower risk by virtue of their use and design. Canoes and kayaks for example are typically not moored/kept on the water for extended periods of time, don't require trailers that contact the lake bottom, have fairly simple internal and external designs, and can be readily dried and inspected. As new AIS issues and threats arise, park managers will continue to be challenged in finding a balance between accommodating visitor use of GNP and providing vigilant ecosystem protection and conservation.

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Trend gill-netting and fish tissue contaminants analysis in Glacier National Park

ABSTRACT

We sampled lakes in Glacier National Park in 2010 to characterize fish species composition and simultaneously collected fish tissue for mercury (Hg) and selenium (Se) analysis in subsets of these lakes. Kintla Lake, Bowman Lake, Logging Lake, Hidden Lake, and Lake McDonald were sampled on the west side of the park while east side waters included St. Mary Lake, Cosley Lake, and Sherburne Reservoir. Comparison with historic gill netting records in the west side lakes (except for Hidden Lake which lacks these species) indicates that lake trout are increasing and bull trout are in decline, and bull trout populations in these lakes are increasingly imperiled. One manifestation of these bull trout declines was that for the first time ever no bull trout were captured in gill nets in Logging Lake. Selected fishes from Logging Lake, Hidden Lake, and the three east side water bodies were analyzed for mercury. The species tested for mercury were: lake trout, lake whitefish, mountain whitefish, burbot, northern pike, cutthroat trout, northern pikeminnow, longnose sucker, and largescale sucker. The greatest number species tested for mercury was in Logging Lake which should serve as a particularly useful baseline for mercury contamination in the park. A subset of the fishes collected for mercury assessment in Logging Lake and St Mary Lake were also analyzed for selenium. Mercury testing revealed that mercury generally increased with fish size within a species and varied among species even after considering fish size. Piscivorous fishes, particularly larger individuals, often exceeded current human consumption guidelines. Our findings have resulted in the adaptation of additional human health advisories for anglers consuming Glacier National Park fishes. The wildlife risk assessment (utilizing both the fish mercury and species composition data from St Mary and Logging lakes) suggests that the consumption of relatively small, lightly contaminated fishes generally does not pose a major threat to piscivorous wildlife based on current wildlife thresholds. Northern pikeminnow from Logging Lake represented the worst case scenario in the wildlife risk assessment and often exceeded protective thresholds for piscivorous wildlife. However, the mercury risk to wildlife from northern pikeminnow is presumably reduced by the consumption of a variety of less contaminated fishes, and it should be noted this species has a limited distribution in the park. Further, the results of the wildlife assessment suggest that the mercury consumption risk may be further reduced by the protective effect of selenium. Our selenium results represent the first testing of this element in park fishes, providing baseline data for risk assessment and future studies.

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INTRODUCTION

Glacier National Park (GNP), located in northwest Montana, represents some of the most pristine and biologically diverse habitat for plants and animals found in the Intermountain West. Sitting at the core of the Crown of the Continent Ecosystem, GNP provides a diversity of stream and lake habitats for aquatic species. GNP covers over 1,000,000 acres, providing high-quality lentic and lotic fish habitat. GNP supports over 700 perennial lakes/ponds, ranging in size from less than an acre, up to Lake McDonald, covering almost 7,000 surface acres. GNP also provides over 2,200 km of high-quality stream habitat for aquatic species. A diversity of native and introduced fish species inhabit park waters (Tables 1 and 2).

Table 1. Native (N) and introduced (I) salmonids in Glacier National Park.

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Arctic grayling *Thymallus arcticus*	--	--	I
Brook trout *Salvelinus fontinalis*	I	I	I
Bull trout (BLT) *S. confluentus*	N	--	N
Kokanee (KOK) *Oncorhynchus nerka*	I	--	I
Lake trout (LKT) *S. namaycush*	I	--	N
Lake whitefish (LWF) *Coregonus clupeaformis*	I	--	N
Mountain whitefish (MWF) *Prosopium williamsoni*	N	N	N
Pygmy whitefish *P. coulteri*	N	--	N
Rainbow trout *O. mykiss*	I	I	I
Westslope cutthroat trout (WCT) *O. clarkii lewisi*	N	N	N
Yellowstone cutthroat trout (YCT) *O. c. bouvieri*	I	I	I

Although GNP represents some of the last best wild areas in North America, recent studies have demonstrated that GNP is not immune to anthropogenic impacts such as exotic species and airborne contaminants. Recent fisheries studies (Marnell et al. 1987, Fredenberg 2002, Hitt et al. 2003, Muhlfeld et al. 2009) have demonstrated the ecological damage and genetic consequences associated with non-native fish species in park waters. For example, Fredenberg (2002) documented the replacement of bull trout as the top predator by non-native lake trout in sympatric lakes on the west side of the park during the last several decades, and lake trout are continuing to expand their numbers and range on the west side. In response to the lake trout threat the park has recently initiated collaborative efforts to secure remaining at-risk bull trout populations. Monitoring data from park lakes will be key in evaluating any future shifts in fish community structure that may result from additional impacts of non-native species, climate change, or other habitat perturbations.

In addition to threats posed by exotic species, recent studies have demonstrated the aerial transport and deposition of metals (e.g. mercury), semi-volatile organic compounds (pesticides and herbicides), industrial compounds (e.g. PCB's), and emerging chemicals (e.g. fire retardant PBDE) into the GNP ecosystem (Watras et al. 1995, Landers et al. 2008, Downs and Stafford 2009). Deposition, transport, and bioaccumulation of these substances pose threats to the aquatic ecosystem, piscivorous wildlife, and human consumers of fishes.

Table 2. Native (N) and introduced (I) non-salmonids in Glacier National Park.

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Northern pikeminnow (NPM) *Ptychocheilus oregonensis*	N	--	--
Peamouth (PEA) *Mylocheilus caurinus*	N	--	--
Redside shiner (RSS) *Richardsonius balteatus*	N	--	--
Longnose sucker (LNS) *Catostomus catostomus*	N	N	N
Largescale sucker (CSU) *C. macrocheilus*	N	--	--
White sucker (WSU) *C. commersoni*	--	N	N
Deepwater sculpin *Myoxocephalus thomsoni*	--	--	N
Mottled sculpin *Cottus bairdi*	--	N	N
Slimy sculpin *C. cognatus*	N	--	--
Shorthead sculpin *C. confusus*	N	--	--
Spoonhead sculpin *C. ricei*	--	--	N
Burbot (BUR) *Lota lota*	--	--	N
Northern pike (NPI) *Esox lucius*	--	--	N
Trout-perch *Percopsis omiscomaycus*	--	--	N

Aerial deposition of mercury (Hg) to watersheds and subsequent uptake by aquatic biota is of increasing concern. Atmospheric deposition of Hg is the dominant source of labile Hg to most watersheds, particularly those in remote settings such as GNP. About 2/3 of the Hg in the atmosphere globally is from human activities (Mason et al. 1994), and coal burning is the largest source (Nriagu and Pacyna 1988). Regionally, glacial cores from the Wind River Range, Wyoming show that in the 20th century about 70% of the Hg deposited aerially was from human sources (Schuster et al. 2002). Elemental Hg is transformed into the bio-available form, methyl mercury (MeHg) primarily by sulfate reducing bacteria (Gilmour and Henry 1991). Most MeHg is produced in lakes and their watersheds, although some MeHg is deposited aerially (Fitzgerald et al. 1991, Watras et al. 1995). MeHg biomagnifies as it moves up the food chain, and top level consumers such as humans and piscivorous wildlife are particularly at risk. MeHg can damage developing nervous systems in humans and other animals if ingested in sufficient amounts, and early life stages are particularly at risk. It is estimated that the developing fetus is 5-10 times more sensitive to MeHg than adults (Clarkson 1990). An estimated eight percent of women of childbearing age have Hg levels deemed unsafe for childbearing by the EPA **(**Schober et al. 2003**).** The primary exposure pathway for humans and wildlife to MeHg is through consumption of contaminated fish.

Previous studies have documented mercury contamination in park fishes. These studies include lake trout and lake whitefish from Upper Two-Medicine Lake in GNP [T. Selch, Montana Fish, Wildlife, and Parks (MFWP), personal communication], lake trout and lake whitefish from the Waterton Lakes in Waterton Lakes National Park (Brinkmann 2007), and cutthroat trout from Synder and Oldman lakes (Landers et al. 2008) . A multi lake study focused on lake trout was conducted in 2008 on Lake McDonald, Bowman Lake, Harrison Lake, and St Mary Lake in 2008 (Downs and Stafford 2009). This study also included smaller sample sizes of lake whitefish and bull trout from Lake McDonald and burbot from St Mary Lake. The 2008 testing demonstrated elevated levels of Hg in the tissue samples of larger lake trout and burbot, and resulted in fish consumption advisories.

The existing Hg data, while useful, suffers from several limitations. The testing to date was almost exclusively on salmonids, neglecting a variety of species important to piscivorous wildlife. Several potentially high mercury species remained untested (i.e. northern pike) or under tested (i.e. burbot and bull trout). Even for the most tested species, lake trout, sample sizes within a lake sometimes were too low [i.e. lake trout from St Mary (Downs and Stafford 2009)]. Further, testing to date has largely been conducted on the west side of the park which has a different assemblage of fishes than the east side. To reduce the limitations of the existing mercury data, additional samples were collected in 2010. Our goals of the mercury testing were to sample more species, improve our geographic coverage, and expand sample sizes/size ranges in order to better assess overall Hg contamination levels and ultimately Hg risk to human consumers of fish and piscivorous wildlife. This expanded investigation was also designed to provide a large enough data set to provide a meaningful baseline of mercury contamination in park fishes to evaluate any future changes.

No data exists for selenium (Se) in park fishes, yet this element is important to consider when studying mercury contamination. Selenium is an essential nutrient, but can also be toxic depending on dose (Yang et al. 2008; Peterson et al. 2009). Naturally occurring Se levels in Glacier National Park were not expected to be toxic to fish or their predators, however there is growing evidence that Se has a major effect on the fate and toxicity of Hg in aquatic food webs and in humans (Cuvin-aralar and Furness 1991; Raymond and Ralston 2004; Peterson et al. 2009). Therefore, the combined Se and Hg data sets will improve our ability assess the mercury risk, and likely will be essential for future assessments of mercury toxicity in GNP and other ecosystems as selenium/mercury risk protocols continue to develop through time.

In summary, the purpose of the current study was to repeat long-term gill net trend monitoring in park lakes and provide information on the levels of Hg and Se in fish tissue in selected waters of GNP. This information will be used to assess the status of native fish populations in park lakes, evaluate risks to human and piscivorous wildlife from consuming contaminated fish, and provide a baseline to evaluate any future changes in Hg contamination.

METHODS

Study Sites

Kintla Lake, Bowman Lake, Logging Lake, Hidden Lake, and Lake McDonald were sampled on the west side of the park while east side waters included St. Mary Lake, Cosley Lake, and Sherburne Reservoir (Figure 1). Kintla, Bowman, and Logging lakes, as well as Lake McDonald are part of an historical gill netting monitoring program in the park. The other lakes were sampled to collect tissues for Hg analysis and to begin to establish gill netting monitoring data sets.

Fish Community Assessment

We used sinking multi-filament experimental mesh gill nets to sample seven of the eight park lakes from August 8-September 29, 2010 (Figure 1). All nets were 38.1m (125') X 1.83m (6') experimental multifilament nylon with five 7.62m (25') panels consisting of 19mm (3/4") (#104 twine), 25mm (1") (#139), 32mm (1-1/4") (#104), 1-1/2" (38mm)(#104), 51mm (2") (#139) bar measured mesh. 30# leadcore and polyfoam line was used to hold the net upright, while maintaining contact with the lake bottom. They were either ganged together to form a single 250' net with the large end of the inshore net attached to the small end of the offshore net, or set as a single 125' net. The shallow end was set closest to shore in either configuration. The eighth lake, Cosley Lake was sampled by angling and by using a single light-weight sinking monofilament gill net of similar dimensions to the multi-filament nets, but with panels of 12.5mm (½"), 19mm (¾"), 25mm (1"), 38mm (1 ½"), and 63.5mm (2 ½") inch mesh. In addition, we deployed a single floating net on Hidden Lake in order to sample Yellowstone cutthroat trout more effectively. This net was similar to the standard nets except that it was constructed of monofilament and the smallest mesh was 12.5 mm instead of 19 mm. The remaining mesh sizes were the same as the standard nets. In addition to gill netting, some fish were also collected for contaminants testing using angling techniques. This occurred primarily on Cosley Lake where packing sufficient amounts of gill net into the backcountry was problematic, as well as on Logging Lake where sinking gill nets were not effective at capturing enough westslope cutthroat trout for contaminants analysis.

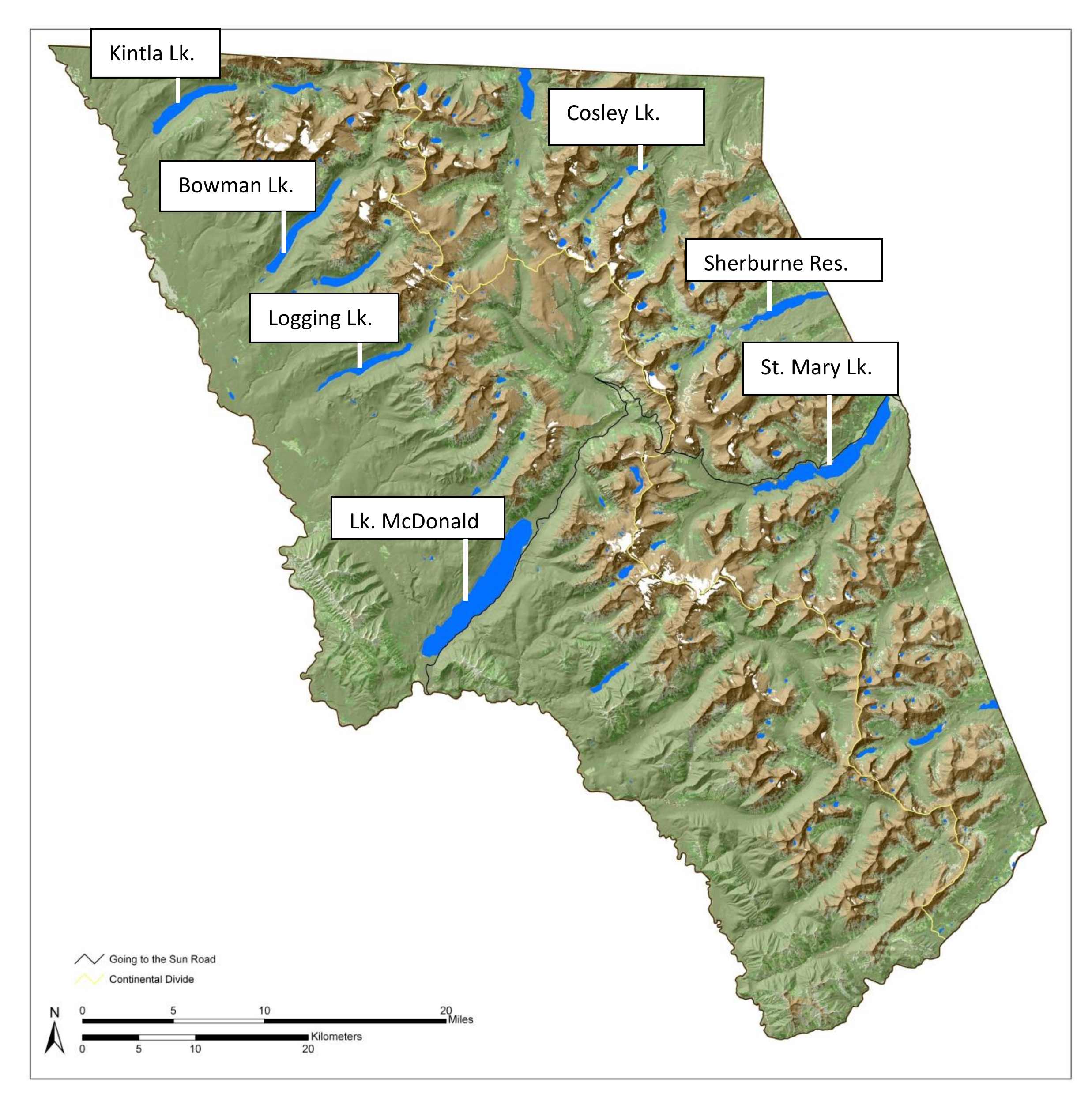

Figure 1. Glacier National Park waters sampled for population assessment and/or fish tissue contaminants assessment.

Nets generally were retrieved in the morning, labeled, and taken to shore to be cleared of fish. Fish were identified to species, weighed (g) and measured (total length (TL); mm). Age structures (scales and otoliths) were collected from some species and locations and archived for possible future use.

We estimated catch-per-unit-effort (CPUE) as the number of fish captured per net-night of sampling, and also as the number of fish captured per hour in over-night net sets. We calculated average length and weight for each species to facilitate comparisons of size structure between populations, and also provide information useful in comparing changes in all of the population and sampling metrics through time. It should be noted that not all captured fish were used for Hg analysis, so these averages do not reflect the average size of fish analyzed for Hg. We estimated relative weight (*Wr*) (Anderson and Neuman 1996) for selected fish species to evaluate growth conditions across the sampled waters. We used standard weight equations for lake trout (Picolo et al. 1993), burbot (Fisher et al. 1996), and lake whitefish (Rennie and Verdon 2008) to estimate *Wr* for each water. Fulton-type (K) condition factors were used for other species (Anderson and Neuman 1996). We compared our current catch composition and CPUE with that of earlier sampling to evaluate trends over time.

Mercury and Selenium Analyses

For the contaminants study we expanded the 2008 mercury testing (Downs and Stafford 2009), and collected data on selenium in park fishes. Selected fishes from Logging Lake, Hidden Lake, and the three east side water bodies were analyzed for Hg. For the mercury component we sampled a broad suite of fish species (lake trout, lake whitefish, mountain whitefish, burbot, northern pike, cutthroat trout, northern pikeminnow, longnose sucker, and largescale sucker), as well as increased sample sizes and size ranges from earlier sampling. Further, we expanded our geographic coverage with the sampling of Cosley Lake, Sherburne Reservoir, and Logging Lake, all of which were previously untested. For the selenium component, the collections from St Mary Lake and Logging Lake represent the first testing of this element in park fishes.

We collected fish muscle tissue for both the mercury and selenium analyses. A long piece of skinless dorsal muscle tissue was removed for contaminants analysis. Muscle tissue was removed with a stainless steel fillet knife on a polyethylene cutting board. During fish processing all materials in contact with the muscle tissue were rinsed in acid (vinegar) then repeatedly rinsed in lake water. Processed samples were placed into a re-sealable plastic bag and the excess air was purged to reduce desiccation. Samples were frozen in the field using dry ice and then stored frozen until contaminants analysis.

Samples were analyzed for total Hg and Se at the University of Montana Geosciences Department's Environmental Biogeochemistry Laboratory (EBL). Hg analysis was conducted using a Milestone Inc. model DMA-80 Direct Hg Analyzer (USEPA method 7473). This method is different from the method used in the 2008 sampling (Downs and Stafford 2009), and we therefore verified its precision and accuracy as an add-on to the fish tissue analyses of 2008. Generally, quality control parameters were similar or improved using the new DMA method. Mercury samples were prepared by removing a slice of muscle tissue from the center of each sample with stainless steel tools on an acid washed plastic cutting board. Sample weights varied between 0.1 and 0.3 g, depending on the anticipated Hg concentration.

For Se analysis, fish tissues were digested using concentrated HNO_3 and H_2O_2 (Tekran Instruments Application Note AN-2600-10). Briefly, 1 g tissue samples were weighed into 50-mL digestion vials, before adding 3.3 mL trace metals grade HNO_3 and 3.3 mL deionized water. After 30 min predigestion at room temperature, samples were refluxed at 80-85°C for 4 h, then allowed to cool before addition of 1 mL 30% ACS grade H_2O_2. After heating for another 30 min at 60-70°C, sample vials were filled to 50 mL and mixed. Se analysis was conducted on these digests using a Perkin-Elmer model Elan DRCII ICP-Mass Spectrometer according to EPA Method 6020. Reported concentrations are for total Se based on results for the ^{82}Se isotope. All Hg and Se tissue concentrations are reported on a wet weight basis and Se:Hg ratios are reported as mol/mol

Contaminants Quality Assurance/Quality Control

Mercury

We quantified the precision and accuracy of our mercury analysis using a suite of quality assurance/quality control samples. The average amount of Hg in blanks (n = 34) was 0.14 ng (± 0.15 std. dev.). In a fish sample weighing 0.23 g, this would result in a blank contribution of 0.0006 mg/kg (±0.0006 std. dev.). Average recovery in dogfish muscle (DORM-3) Standard Reference Material (n=24) was 104% (±3 std. dev.) of the certified value. Duplicate analyses (n = 23) on our fish samples produced an average difference of 3.2% (±4.1 std. dev.). Another set of duplicate samples (n=13) was spiked with a known amount of Hg before analysis, producing an average recovery of 92% (±5 std. dev.).

Selenium

We quantified the precision and accuracy of the selenium analysis using a variety of quality assurance/quality control samples. All analytical blanks (n=11) were below the reported practical quantification limit of 50 µg/kg. Average recovery for DORM-3 samples (Standard Reference Materials, n=4) was 98% (±4 std. dev.). Analysis of separate digestions from the same fish tissue sample (Method Duplicates) using our fish samples (n=4) produced an average difference of 12% (±5 std. dev.). An additional set of method duplicates (n=2) was spiked before digestion (Method Spikes) and average recoveries were 84% (±4 std. dev.). Duplicate analyses on digests (n=9) of our fish samples (Analytical Duplicates) produced an average difference of 6.4% (±5.7 std. dev.). Analysis of spiked digests (n=9) recovered an average of 89% (±4 std. dev). A total of 11 independent standard solutions (Internal Performance Checks were analyzed, with average recoveries of 92% (±2 std. dev.).

Human risk was assessed using muscle tissue (fillet concentration), as recreational fishers generally only consume the fillets. We used these values to further refine consumption guidance for park visitors, based on methods used by Montana Fish, Wildlife, and Parks (T. Selch, MFWP, personal communication). However, because piscivorous wildlife typically consumes the whole fish, we converted fillet concentrations to whole fish concentrations using a regression relationship developed by Peterson et al. (2007):

$$\log(\text{fillet Hg}) = 0.2545 + 1.0623 \log(\text{whole fish Hg})$$

We used literature values for average prey size of piscivorous wildlife (Erlinge 1969, Swenson 1978, Watson et al. 1991, Barr 1996), and developed regression models using our observed data to predict Hg concentrations for various fish species at any length. We then estimated Hg concentrations in each fish species at standard sizes of 150 mm, 200 mm, and 300 mm. These sizes represent intermediate size of fish prey for many piscivorous species in the park. We used our catch composition from gill netting to develop a weighted average Hg concentration for prey species available to piscivorous wildlife. We then compared the weighted average Hg concentrations for the three sizes of fish to various Hg concentration thresholds for piscivorous wildlife to assess risk. We used water quality Se concentrations to assess potential aquatic resource risks based on established literature thresholds. Molar ratios of Se:Hg were developed and used as supplemental information in assessing Hg consumption risk to wildlife.

RESULTS AND DISCUSSION

Fish Community Assessment

Netting data from Bowman Lake and Logging Lake revealed declining abundances of bull trout and increases in lake trout, particularly in Logging Lake, while trends in other species were less clear (Table 3). A comparison of the four west side lakes that contain bull trout reveals that Bowman Lake had the highest CPUE for lake trout, followed by Logging Lake. We caught no bull trout in Logging Lake, while we captured six in Bowman Lake. This is troubling, as Logging and Bowman lakes once supported strong bull trout populations (Fredenberg 2002). Historic assessment of standardized gill net catch data document the continued replacement of bull trout by lake trout as the dominant aquatic predator in the large-interconnected lakes on the west side of the park (Figures 2-5). Bull trout are not yet extirpated from Logging Lake as evidenced by the observation of three redds and two spawning adults in Logging Creek upstream of the lake in October 2010 (this report), but they persist at precariously low levels. Although variable, trends in catch composition of other species in Bowman and Logging lakes do not show a consistent trend (Figures 6 and 7). However, sampling conducted in 1969 captured far more suckers and far fewer mountain whitefish than later years. With limited historical data, it is difficult to speculate as to whether this is an artifact of differences in sampling timing (1969 sampling was consistently 1-2 months earlier than later 2000-2010 sampling), or if it represents a shift in species composition. Sampling in 1969 was conducted in June, likely before the lakes were stratified, as compared with later sampling that likely occurred when the lakes were stratified. Although we captured westslope cutthroat trout in gill net sets, time-trend analysis using these data isn't appropriate as sinking nets do not effectively sample the upper 6'-10' of the water column where westslope cutthroat trout are most active.

Low numbers of bull trout were also captured in both Kintla Lake and Lake McDonald as well, but these lakes do not appear to have the same potential to support a reproducing bull trout population as do Bowman and Logging, due to limited tributary spawning and rearing habitat. Meeuwig (2008) evaluated genetic population structuring in GNP, and determined that Kintla Lake and Lake McDonald were not significantly different from one another, had high genetic diversity, and that they were most similar to a composite genetic sample of bull trout spawners from Flathead Lake (Meeuwig et al. 2007). This suggests that these two "populations" could be comprised of individuals from a number of migratory populations supporting Flathead Lake, rather than their own distinct bull trout populations. Meeuwig (2008) further demonstrated genetic differentiation in GNP bull trout was positively correlated with the stream distance between park lakes and the mainstem North and Middle forks of the Flathead River. Both Kintla and Lake McDonald are located in close proximity to the mainstem of the North and Middle Forks of the Flathead River, making movement between these lakes and the larger Flathead system fairly easy. Expansion patterns of lake trout within the system further support the idea that lakes closer to the mainstem rivers are more readily accessed by migratory fish from Flathead Lake. Gill netting data suggest Lake McDonald and Kintla Lake were the first park lakes to be colonized by lake trout from Flathead Lake (Fredenberg 2002).

Lake trout catch rates, average size, and condition factor showed some geographic variation among the four west side lakes where this species is present. Lake McDonald showed the highest average size while the other three west side lakes were similar in this regard. Lake trout condition was low for all four west side lakes, particularly Logging Lake and Lake McDonald (Table 4). The low condition factor in part may reflect in part that lake trout densities are sufficiently high to reduce their forage base. Among these west side lakes, catch rates were markedly higher in Bowman Lake with declining catch rates from Logging Lake to Lake McDonald to Kintla Lake (Table 5). Catch rates were lowest in St. Mary Lake but given the low net effort (due to inclement winds) no firm conclusions should be drawn on this metric (or the size and condition factor metrics).

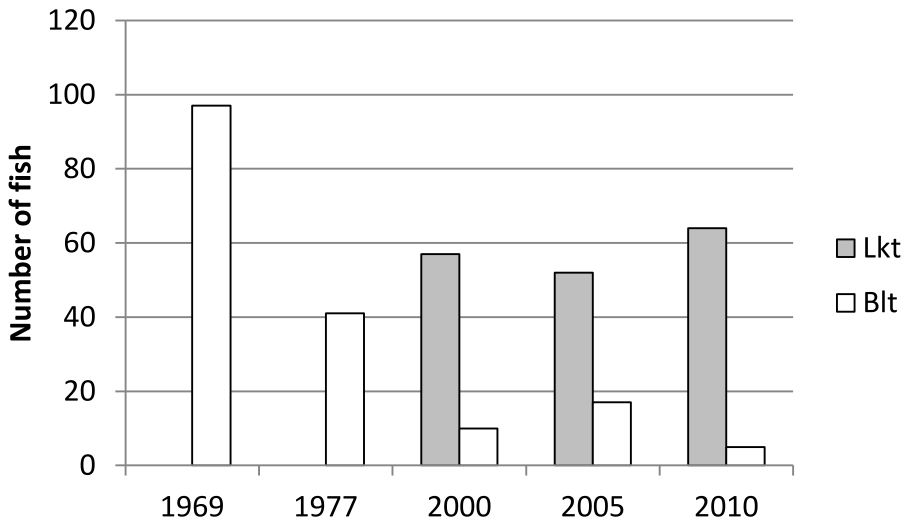

Figure 2. Catch of bull and lake trout in standardized gill nets in Bowman Lake, Glacier National Park.

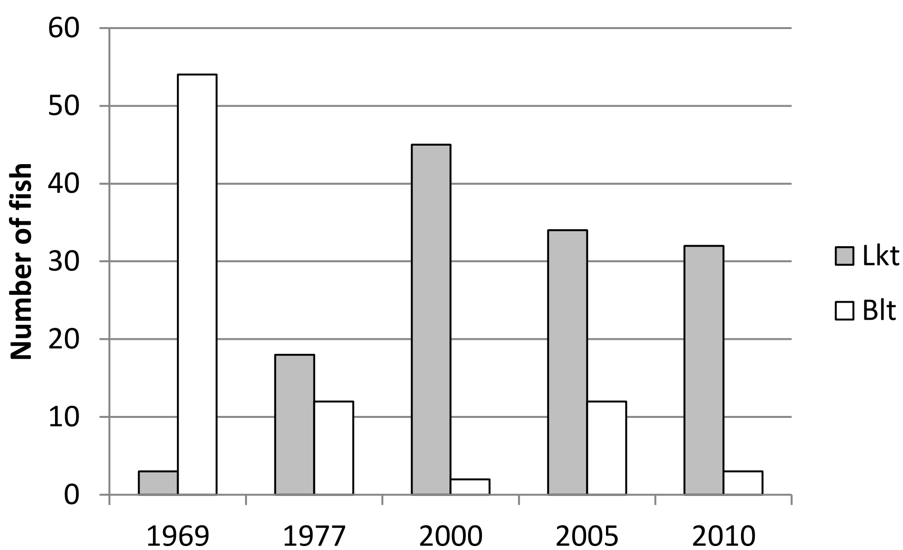

Figure 3. Catch of bull and lake trout in standardized gill nets in Kintla Lake, Glacier National Park.

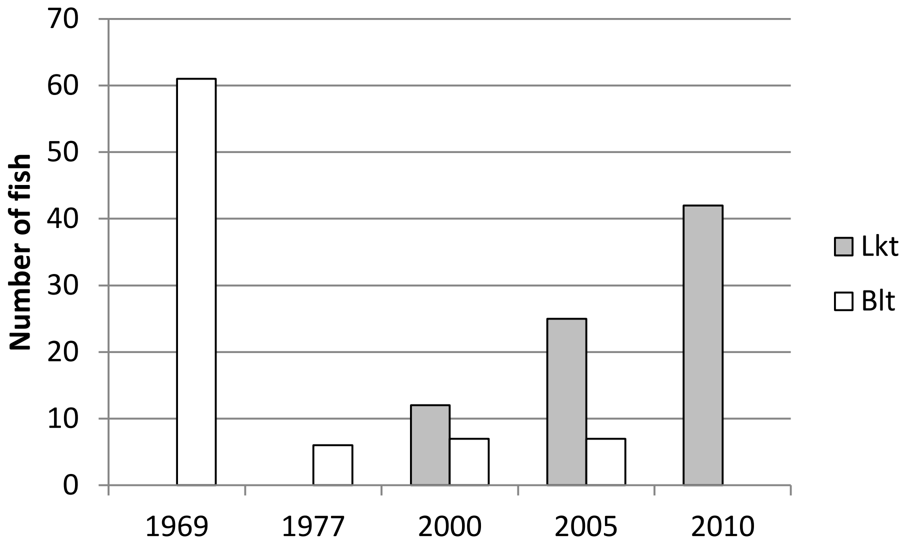

Figure 4. Catch of bull and lake trout in standardized gill nets in Logging Lake, Glacier National Park.

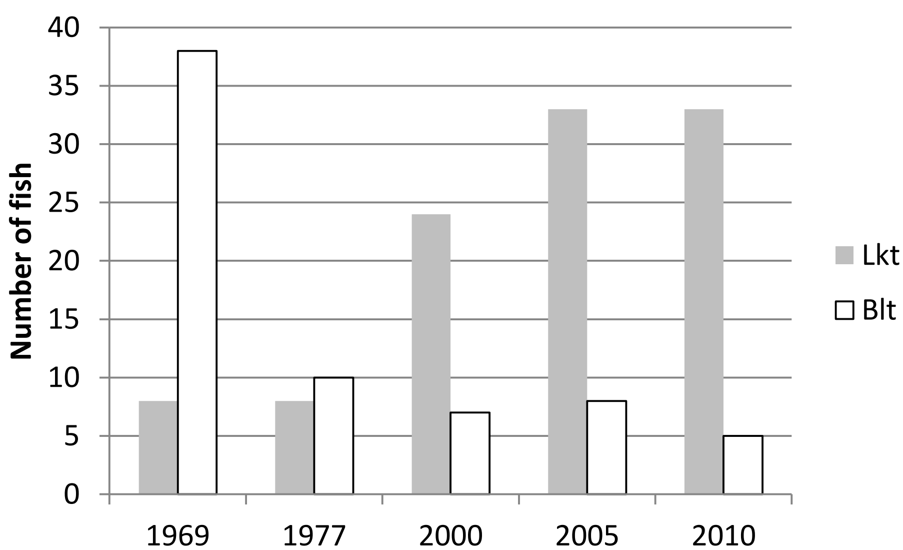

Figure 5. Catch of bull and lake trout in standardized gill nets in Lake McDonald, Glacier National Park.

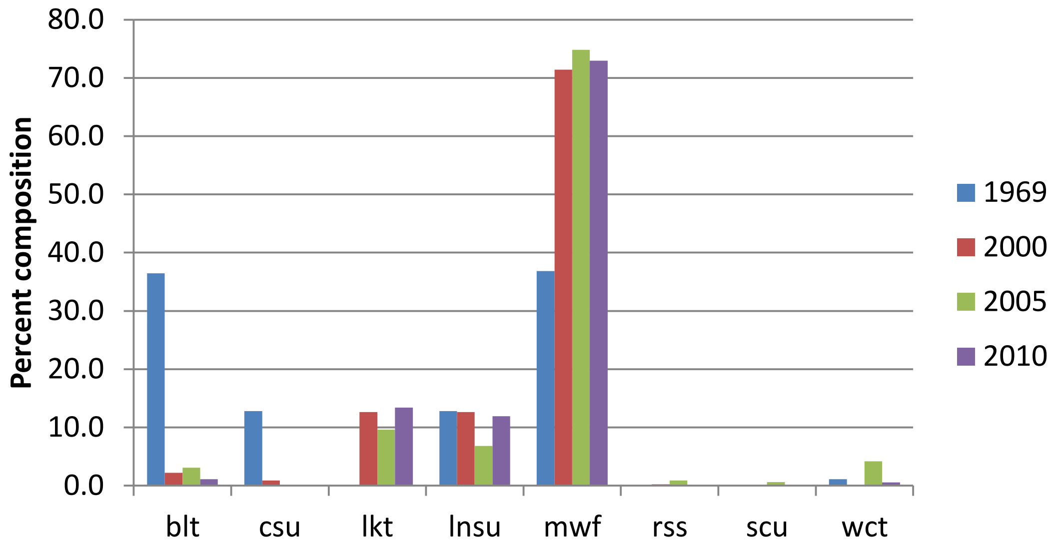

Figure 6. Long-term catch composition in Bowman Lake in standardized gill net sets, Glacier National Park.

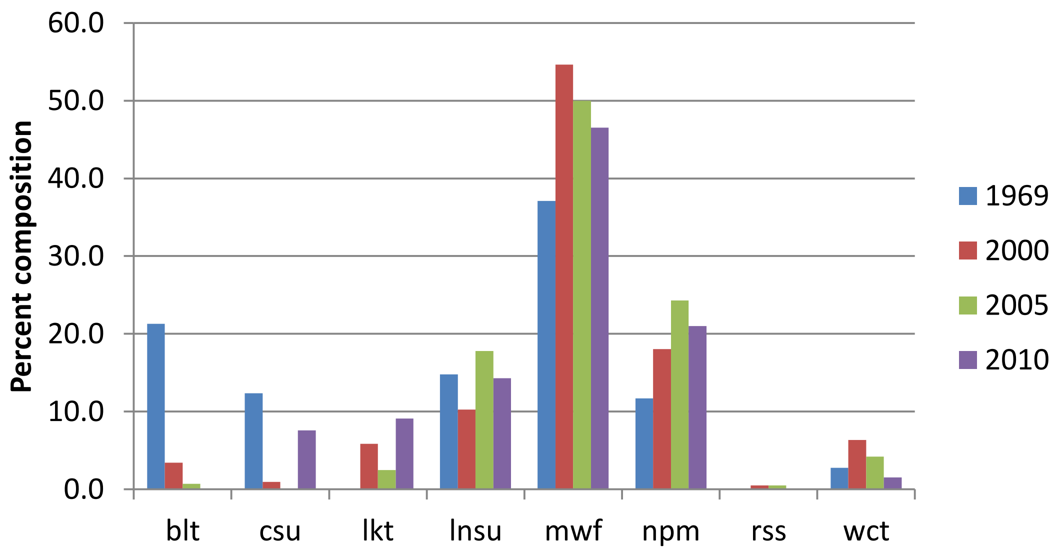

Figure 7. Long-term catch composition in Logging Lake in standardized gill net sets, Glacier National Park.

Table 3. Mean length (TL;mm), length range, and weight (g) for species captured in overnight sinking gill net sets conducted during late summer in Glacier National Park, 2010 (blt = bull trout, bur = burbot, csu = coarsescale sucker, lkt = lake trout, lns = longnose sucker, lwf = lake whitefish, mwf = mountain whitefish, npi = northern pike, npm = northern pikeminnow, pea = peamouth, wct = westslope cutthroat trout, yct = Yellowstone cutthroat trout).

Water (# nets)	Species	Mean length (95% CI) (n)	Length range	Mean weight (95% CI)(n)
Bowman Lk. (10)	blt	290.2 (165.3)(6)	188-600	382.8 (690.2) (6)
	lkt	408.1 (28.3) (71)	192-695	643.8 (135.4) (69)
	lns	202.2 (10.2) (63)	149-338	106.4 (19.0) (61)
	mwf	309.3 (5.5) (386)	105-496	321.3 (26.9) (167)
	wct	173.0 (49.9) (3)	152-192	56.0 (228.7) (2)
Hidden (1 sink,1 float)	yct	341.5 (24.1) (31)	226-443	409.2 (87.8) (25)
Kintla Lk.(10)	blt	489.0 (600.1) (3)	277-752	1,057.7 (2,895.0) (3)
	csu	213.3 (38.4) (19)	150-479	183.1 (178.3) (16)
	lkt	412.0 (66.7) (32)	201-941	825.4 (538.7) (30)
	lns	226.6 (17.4) (33)	152-330	145.2 (31.6) (31)
	mwf	238.0 (5.5) (274)	153-452	122.5 (26.9) (88)
	wct	215.3 (60.9) (7)	164-357	132.2 (147.6) (6)
Logging Lk. (10)	blt	N/A		
	csu	411.6 (33.6) (31)	162-530	809.3 (182.3) (24)
	lkt	398.2 (40.4) (42)	124-748	589.9 (209.1) (40)
	lns	342.5 (27.2) (59)	160-481	554.7 (87.8) (54)
	mwf	225.1 (4.6) (197)	126-387	106.1 (10.6) (160)
	npm	198.0 (6.9) (82)	152-316	77.5 (11.0) (76)
	wct	253.5 (1,289.7) (2)	152-355	N/A
McDonald Lk. (10)	blt	491.6 (86.9) (5)	396-580	1,215.0 (711.7) (4)
	csu	287.8 (98.0) (10)	148-465	401.7 (334.4) (10)
	lkt	443.9 (31.7) (33)	189-637	689.3 (141.4) (33)
	lns	298.1 (26.4) (61)	145-450	398.7 (82.4) (61)
	lwf	471.6 (31.2) (26)	171-556	1,102 (158.4) (26)
	mwf	252.9 (42.0) (12)	171-345	171.9 (84.7) (12)
	pea	179.4 (4.0) (79)	152-246	57.2 (3.9) (78)
	npm	244.1 (13.1) (78)	155-406	159.3 (33.3) (77)
	wct	286.0 (126.5) (5)	202-450	111.7 (78.7) (3)
Sherburne Res. (3)[a]	blt	650 (est.)		N/A
	lns	411.2 (86.7) (13)	180-553	1,044.5 (427.8) (12)
	lwf	351.8 (10.8) (11)	322-377	358.1 (31.0) (11)
	mwf	193.7 (39.9) (3)	177-209	58.0 (35.0) (3)
	npi	798.6 (211) (8)	190-996	3,842.6 (1,516.8) (8)
St. Mary Lk. (2)	bur	419.4 (100.8) (7)	225-572	560.4 (351.5) (7)
	lkt	549.8 (48.8) (5)	507-600	1,388.6 (383.2) (5)
	lns	269.2 (17.6) (45)	157-407	290.6 (55.6) (43)
	lwf	432.6 (22.4) (11)	400-502	791.3 (163.7) (10)
	mwf	249.5 (36.7) (8)	205-329	143.1 (69.7) (8)

Table 4. Fish condition expressed as relative weight (Wr) or Fulton-type condition factor (K) for selected species and waters in Glacier National Park sampled with gill nets during late summer, 2010.

Water	**Species**	**Sample size**	**Mean Wr/K (95% CI)**
Bowman Lk.	blt	6	0.84 (0.08) (K)
	lkt	69	80.2 (2.6)
	lns	60	1.12 (0.04) (K)
	mwf	165	92.0 (2.0)
	wct	2	98.8 (36.1)
Kintla Lk.	blt	3	0.69 (0.29) (K)
	csu	16	1.1 (0.07) (K)
	lkt	30	80.5 (4.3)
	lns	31	1.11 (0.05) (K)
	mwf	88	78.7 (1.8)
	pea	20	0.83 (0.05) (K)
Logging Lk.	csu	24	1.03 (0.02) (K)
	lkt	40	73.8 (3.2)
	lns	54	1.01 (0.02) (K)
	mwf	159	86.1 (1.1)
	npm	76	0.92 (0.02) (K)
	wct	16	87.9 (2.9)
McDonald Lk.	blt	4	85.2 (0.17)
	csu	9	1.09 (0.05) (K)
	lkt	33	75.4 (3.1)
	lns	61	1.1 (0.03) (K)
	lwf	26	92.0 (3.3)
	mwf	12	86.4 (3.6)
	npm	75	0.94 (0.03) (K)
	pea	78	0.98 (0.03) (K)
	wct	3	92.2 (11.4)
Sherburne Res.	lns	11	1.01 (0.07) (K)
	lwf	11	82.8 (3.7)
	mwf	3	79.6 (7.6) (K)
	npi	8	90.3 (9.2)
St. Mary Lk.	bur	7	88.2 (7.8)
	lkt	5	84.2 (13.7)
	lns	43	1.3 (0.03) (K)
	lwf	10	90.4 (3.1)
	mwf	8	84.7 (6.3)

Table 5. Gill netting catch-per-unit-effort (CPUE) for species captured in overnight gill net sets in Glacier National Park lakes during late summer, 2010.

Water (number of overnight net sets)	Species	Number captured	Percent net composition	CPUE (fish/net night)
Bowman Lk. (10)	blt	6	1.13	0.6
	lkt	71	13.42	7.1
	lns	63	11.91	6.3
	mwf	386	72.97	38.6
	wct	3	0.57	0.3
Hidden (1 sinking)[a]	yct	18	100	18
Hidden (1 floating)[a]	yct	13	100	13
Kintla Lk.(10)	blt	3	0.77	0.3
	csu	19	4.85	1.9
	lkt	32	8.16	3.2
	lns	33	8.42	3.3
	mwf	274	69.90	27.4
	pea	24	6.12	2.4
	wct	7	1.79	0.7
Logging Lk. (10)	blt	0	0.00	0
	csu	35	7.80	3.5
	lkt	42	9.35	4.2
	lns	66	14.70	6.6
	mwf	215	47.88	21.5
	npm	97	21.60	9.7
	wct	4	0.87	0.4
McDonald Lk. (10)	blt	5	1.81	0.5
	csu	10	3.61	1
	lkt	33	11.91	3.3
	lns	66	23.83	6.6
	mwf/lwf	79	28.52	7.9
	npm	79	28.52	7.9
	wct	5	1.81	0.5
Sherburne Res. (3)[a]	blt	1	0.03	0.3
	lns	13	0.36	4.3
	lwf	11	0.31	3.7
	mwf	3	0.08	1.0
	npi	8	0.22	2.7
St. Mary Lk. (2)	blt	0	0.00	0
	bur	7	9.21	3.5
	lkt	5	6.58	2.5
	lns	45	59.21	22.5
	lwf	11	14.47	5.5
	mwf	8	10.53	4

[a] 125' nets

[b] floating net was monofilament with smallest mesh 12.5 mm rather than 19 mm.

Based on Wr and K, all species evaluated appeared to have less than optimal body condition (Table 5). Overall, lake trout condition was similar across all sampled waters, suggesting less than optimal feeding and/or temperature conditions may exist. A Wr of 100 represents the 75th percentile of average weight for a given length across a large number of populations for a particular species. In concept, a Wr of 100 is generally representative of good physiological and feeding conditions, and has been shown to be positively correlated with fat content in fish (Anderson and Neuman 1996, Renne and Verdon 2008) and prey availability (Renne and Verdon 2008). Ellis et al. (1992) evaluated trophic status for a number of GNP lakes, including McDonald and St. Mary lakes and concluded all of the waters they sampled were either oligotrophic or ultra-oligotrophic. It would not be unexpected to find lower fish condition in such unproductive waters. This conclusion is supported by the findings of Stafford et al. (2002) who found lake trout from Lake McDonald grew considerably slower than those from the more productive waters of Flathead Lake.

Contaminants Assessment

Of the eight waters sampled for fish community assessment, we collected fish tissue samples for contaminants analysis from two waters west of the Continental Divide and three waters east of the Continental Divide (Table 6). Hg concentration in fish tissue varied by both species and waterbody. In general, top level predators such as northern pike, lake trout, and burbot had the highest concentrations. Elevated concentrations of Hg results from bioaccumulation in long-lived, top level predators. Suckers, westslope cutthroat trout, and Yellowstone cutthroat trout had the lowest concentrations. They feed at lower trophic levels than lake trout and other species evaluated. Power et al. (2002) demonstrated that fish tied to the benthic foodweb had lower Hg levels than those tied to the pelagic foodweb. Interestingly, northern pikeminnow from Logging Lake had the highest concentration of Hg on a size normalized basis of any species/lake tested (Figure 8). This is consistent with recent findings in the Upper Clark Fork River watershed where northern pikeminnow contained two to three times higher Hg levels than other fish species (H. Langner, pers. comm.). In addition, a survey of western stream fishes where northern pikeminnow had the highest mercury burden (Peterson et al. 2009). Both prey selection and growth rate may affect Hg bioaccumulation**,** and a combination of feeding at higher trophic levels at smaller sizes, slower growth (concentration of Hg), and greater longevity may be responsible for the elevated Hg in northern pikeminnow. MeHg has also been shown to biomagnify rapidly as it moves up the foodchain from water concentrations to top level predators, while the fraction of total Hg comprised of MeHg also increases dramatically (Hoffman et al. 2003).

Table 6. Hg and Se concentrations in fish tissue collected from Glacier National Park waters in 2010.

Waterbody	**Sample number**	**Species**	**Fish length (TL;mm)**	**Total Hg (wet weight; mg/kg)**	**Se (wet weight; mg/kg)**
Cosley Lk.	cos10-01	LKT	205	0.054	
Cosley Lk.	cos10-02	LKT	215	0.037	
Cosley Lk.	cos10-03	LKT	230	0.043	
Cosley Lk.	cos10-04	LKT	231	0.052	
Cosley Lk.	cos10-05	LKT	354	0.066	
Cosley Lk.	cos10-06	LKT	367	0.051	
Cosley Lk.	cos10-07	LKT	369	0.092	
Cosley Lk.	cos10-08	LKT	369	0.079	
Cosley Lk.	cos10-09	LKT	379	0.107	

Table 6. Continued.

Waterbody	Sample number	Species	Fish length (TL;mm)	Total Hg (wet weight; mg/kg)	Se (wet weight; mg/kg)
Cosley Lk.	cos10-10	LKT	452	0.253	
Cosley Lk.	cos10-11	LKT	517	0.260	
Cosley Lk.	cos10-12	LKT	569	0.270	
Cosley Lk.	cos10-13	LKT	583	0.144	
Cosley Lk.	cos10-14	LKT	643	0.223	
Cosley Lk.	cos10-15	LKT	777	1.053	
Cosley Lk.	cos10-16	LKT	848	0.878	
Hidden Lk.	hid10-01	YCT	443	0.030	
Hidden Lk.	hid10-02	YCT	426	0.046	
Hidden Lk.	hid10-03	YCT	239	0.034	
Hidden Lk.	hid10-04	YCT	240	0.019	
Hidden Lk.	hid10-05	YCT	362	0.035	
Hidden Lk.	hid10-07	YCT	413	0.025	
Hidden Lk.	hid10-08	YCT	399	0.020	
Hidden Lk.	hid10-10	YCT	346	0.050	
Hidden Lk.	hid10-11	YCT	364	0.030	
Hidden Lk.	hid10-12	YCT	356	0.046	
Hidden Lk.	hid10-13	YCT	278	0.053	
Hidden Lk.	hid10-14	YCT	243	0.018	
Hidden Lk.	hid10-15	YCT	411	0.023	
Hidden Lk.	hid10-16	YCT	391	0.047	
Hidden Lk.	hid10-17	YCT	342	0.052	
Logging Lk.	LSSlog10-1	CSU	458	0.089	
Logging Lk.	LSSlog10-2	CSU	450	0.153	
Logging Lk.	LSSlog10-3	CSU	435	0.127	
Logging Lk.	LSSlog10-4	CSU	468	0.086	
Logging Lk.	LSSlog10-5	CSU	285	0.041	
Logging Lk.	CSUlog10-7	CSU	220	0.040	
Logging Lk.	CSUlog10-8	CSU	414	0.095	
Logging Lk.	CSUlog10-9	CSU	460	0.067	
Logging Lk.	CSUlog10-10	CSU	492	0.106	
Logging Lk.	CSUlog10-11	CSU	388	0.063	
Logging Lk.	CSUlog10-12	CSU	516	0.194	
Logging Lk.	CSUlog10-13	CSU	530	0.141	0.182
Logging Lk.	CSUlog10-14	CSU	226	0.042	
Logging Lk.	CSUlog10-15	CSU	162	0.038	0.119
Logging Lk.	CSUlog10-16	CSU	215	0.041	
Logging Lk.	lktlog10-01	LKT	565	0.292	
Logging Lk.	lktlog10-03	LKT	335	0.113	
Logging Lk.	lktlog10-04	LKT	437	0.198	
Logging Lk.	lktlog10-05	LKT	362	0.162	

Table 6. Continued.

Waterbody	Sample number	Species	Fish length (TL;mm)	Total Hg (wet weight; mg/kg)	Se (wet weight; mg/kg)
Logging Lk.	lktlog10-07	LKT	518	0.272	
Logging Lk.	lktlog10-08	LKT	680	0.407	
Logging Lk.	lktlog10-09	LKT	748	0.617	0.146
Logging Lk.	lktlog10-10	LKT	372	0.125	
Logging Lk.	lktlog10-11	LKT	408	0.202	0.254
Logging Lk.	lktlog10-13	LKT	465	0.392	
Logging Lk.	lktlog10-15	LKT	268	0.148	
Logging Lk.	lktlog10-16	LKT	305	0.115	0.236
Logging Lk.	lktlog10-17	LKT	632	0.392	
Logging Lk.	lktlog10-18	LKT	523	0.247	0.172
Logging Lk.	lktlog10-19	LKT	396	0.144	
Logging Lk.	lktlog10-20	LKT	206	0.078	0.205
Logging Lk.	lktlog10-21	LKT	615	0.635	0.108
Logging Lk.	lktlog10-22	LKT	568	0.295	
Logging Lk.	lktlog10-23	LKT	124	0.027	0.168
Logging Lk.	lktlog10-24	LKT	569	0.171	
Logging Lk.	lnslog10-01	LNS	447	0.107	
Logging Lk.	lnslog10-02	LNS	399	0.093	0.179
Logging Lk.	lnslog10-03	LNS	466	0.111	0.214
Logging Lk.	lnslog10-04	LNS	367	0.075	0.204
Logging Lk.	lnslog10-05	LNS	283	0.057	
Logging Lk.	lnslog10-06	LNS	227	0.029	
Logging Lk.	lnslog10-07	LNS	225	0.039	0.118
Logging Lk.	lnslog10-08	LNS	160	0.037	0.154
Logging Lk.	lnslog10-09	LNS	379	0.063	
Logging Lk.	lnslog10-10	LNS	454	0.085	
Logging Lk.	lnslog10-11	LNS	243	0.041	0.164
Logging Lk.	lnslog10-12	LNS	164	0.030	
Logging Lk.	lnslog10-13	LNS	175	0.043	
Logging Lk.	lnslog10-14	LNS	233	0.062	
Logging Lk.	lnslog10-15	LNS	234	0.047	
Logging Lk.	mwflog10-01	MWF	166	0.025	0.157
Logging Lk.	mwflog10-02	MWF	200	0.030	0.196
Logging Lk.	mwflog10-03	MWF	226	0.077	
Logging Lk.	mwflog10-04	MWF	264	0.081	
Logging Lk.	mwflog10-05	MWF	173	0.033	
Logging Lk.	mwflog10-06	MWF	279	0.067	0.198
Logging Lk.	mwflog10-07	MWF	238	0.084	
Logging Lk.	mwflog10-08	MWF	250	0.104	0.171
Logging Lk.	mwflog10-10	MWF	334	0.182	0.204
Logging Lk.	mwflog10-11	MWF	327	0.093	

Table 6. Continued.

Waterbody	Sample number	Species	Fish length (TL;mm)	Total Hg (wet weight; mg/kg)	Se (wet weight; mg/kg)
Logging Lk.	mwflog10-12	MWF	262	0.079	
Logging Lk.	mwflog10-13	MWF	322	0.056	
Logging Lk.	mwflog10-14	MWF	287	0.075	
Logging Lk.	mwflog10-15	MWF	337	0.137	
Logging Lk.	mwflog10-16	MWF	387	0.162	0.203
Logging Lk.	npmlog10-01	NPM	258	0.212	0.097
Logging Lk.	npmlog10-02	NPM	168	0.080	
Logging Lk.	npmlog10-03	NPM	174	0.052	0.173
Logging Lk.	npmlog10-04	NPM	185	0.124	
Logging Lk.	npmlog10-05	NPM	232	0.292	0.120
Logging Lk.	npmlog10-06	NPM	257	0.211	
Logging Lk.	npmlog10-07	NPM	196	0.064	
Logging Lk.	npmlog10-08	NPM	169	0.150	
Logging Lk.	npmlog10-09	NPM	250	0.411	
Logging Lk.	npmlog10-10	NPM	302	0.268	
Logging Lk.	npmlog10-11	NPM	316	0.203	0.103
Logging Lk.	npmlog10-12	NPM	152	0.071	0.135
Logging Lk.	npmlog10-13	NPM	259	0.141	
Logging Lk.	npmlog10-14	NPM	217	0.222	
Logging Lk.	npmlog10-15	NPM	245	0.188	
Logging Lk.	wctlog10-01	WCT	176	0.017	0.130
Logging Lk.	wctlog10-02	WCT	182	0.024	
Logging Lk.	wctlog10-03	WCT	196	0.023	0.157
Logging Lk.	wctlog10-04	WCT	301	0.075	0.220
Logging Lk.	wctlog10-05	WCT	233	0.022	
Logging Lk.	wctlog10-06	WCT	274	0.041	
Logging Lk.	wctlog10-07	WCT	350	0.060	
Logging Lk.	wctlog10-08	WCT	395	0.123	0.223
Logging Lk.	wctlog10-09	WCT	187	0.016	
Logging Lk.	wctlog10-10	WCT	249	0.030	
Logging Lk.	wctlog10-11	WCT	382	0.100	
Logging Lk.	wctlog10-12	WCT	396	0.100	
Logging Lk.	wctlog10-13	WCT	374	0.085	0.174
Logging Lk.	wctlog10-14	WCT	377	0.085	
Logging Lk.	wctlog10-15	WCT	383	0.125	
Logging Lk.	wctlog10-16	WCT	405	0.100	0.173
Sherburne	LWF-SBR10-01	LWF	372	0.133	
Sherburne	LWF-SBR10-02	LWF	351	0.147	
Sherburne	LWF-SBR10-03	LWF	348	0.110	
Sherburne	LWF-SBR10-04	LWF	353	0.203	
Sherburne	LWF-SBR10-05	LWF	350	0.141	

Table 6. Continued.

Waterbody	Sample number	Species	Fish length (TL;mm)	Total Hg (wet weight; mg/kg)	Se (wet weight; mg/kg)
Sherburne	LWF-SBR10-06	LWF	377	0.179	
Sherburne	LWF-SBR10-07	LWF	355	0.118	
Sherburne	LWF-SBR10-08	LWF	351	0.173	
Sherburne	LWF-SBR10-09	LWF	329	0.148	
Sherburne	LWF-SBR10-10	LWF	322	0.168	
Sherburne	LNS-SBR10-01	LNS	535	0.202	
Sherburne	LNS-SBR10-02	LNS	180	0.044	
Sherburne	LNS-SBR10-03	LNS	183	0.040	
Sherburne	LNS-SBR10-04	LNS	439	0.110	
Sherburne	LNS-SBR10-05	LNS	553	0.392	
Sherburne	MWF-SBR10-01	MWF	195	0.050	
Sherburne	MWF-SBR10-02	MWF	177	0.112	
Sherburne	MWF-SBR10-03	MWF	209	0.031	
Sherburne	NPI-SBR10-01	NPI	190	0.114	
Sherburne	NPI-SBR10-02	NPI	815	0.747	
Sherburne	NPI-SBR10-03	NPI	996	0.677	
Sherburne	NPI-SBR10-04	NPI	894	0.682	
Sherburne	NPI-SBR10-05	NPI	912	0.781	
Sherburne	NPI-SBR10-06	NPI	820	0.663	
Sherburne	NPI-SBR10-07	NPI	886	1.156	
Sherburne	NPI-SBR10-08	NPI	876	0.673	
St. Mary Lk.	BUR-SM10-01	BUR	376	0.090	0.283
St. Mary Lk.	BUR-SM10-02	BUR	448	0.155	0.320
St. Mary Lk.	BUR-SM10-03	BUR	572	0.083	0.235
St. Mary Lk.	BUR-SM10-04	BUR	447	0.195	0.383
St. Mary Lk.	BUR-SM10-05	BUR	378	0.085	0.326
St. Mary Lk.	BUR-SM10-06	BUR	225	0.040	0.460
St. Mary Lk.	BUR-SM10-07	BUR	490	0.185	0.266
St. Mary Lk.	LKT-SM10-01	LKT	550	0.486	0.339
St. Mary Lk.	LKT-SM10-02	LKT	507	0.166	0.368
St. Mary Lk.	LKT-SM10-03	LKT	516	0.341	0.292
St. Mary Lk.	LKT-SM10-04	LKT	600	0.307	0.337
St. Mary Lk.	LKT-SM10-05	LKT	576	0.237	0.309
St. Mary Lk.	LWF-SM10-01	LWF	400	0.180	0.271
St. Mary Lk.	LWF-SM10-02	LWF	502	0.104	0.319
St. Mary Lk.	LWF-SM10-03	LWF	470	0.131	0.246
St. Mary Lk.	LWF-SM10-06	LWF	427	0.151	0.276
St. Mary Lk.	LWF-SM10-09	LWF	447	0.159	0.294
St. Mary Lk.	LWF-SM10-10	LWF	463	0.165	0.231
St. Mary Lk.	LNS-SM10-01	LNS	318	0.042	
St. Mary Lk.	LNS-SM10-02	LNS	207	0.027	0.249

Table 6. Continued.

Waterbody	Sample number	Species	Fish length (TL;mm)	Total Hg (wet weight; mg/kg)	Se (wet weight; mg/kg)
St. Mary Lk.	LNS-SM10-03	LNS	215	0.026	
St. Mary Lk.	LNS-SM10-04	LNS	257	0.039	
St. Mary Lk.	LNS-SM10-05	LNS	304	0.034	
St. Mary Lk.	LNS-SM10-06	LNS	238	0.039	0.295
St. Mary Lk.	LNS-SM10-07	LNS	285	0.050	0.277
St. Mary Lk.	LNS-SM10-08	LNS	330	0.034	
St. Mary Lk.	LNS-SM10-09	LNS	387	0.055	0.278
St. Mary Lk.	LNS-SM10-10	LNS	158	0.022	0.286
St. Mary Lk.	LNS-SM10-11	LNS	310	0.055	0.257
St. Mary Lk.	LNS-SM10-12	LNS	170	0.049	0.263
St. Mary Lk.	LNS-SM10-13	LNS	355	0.043	0.261
St. Mary Lk.	LNS-SM10-14	LNS	382	0.079	
St. Mary Lk.	LNS-SM10-15	LNS	407	0.058	0.241
St. Mary Lk.	MWF-SM10-01	MWF	270	0.060	0.347
St. Mary Lk.	MWF-SM10-02	MWF	329	0.105	0.368
St. Mary Lk.	MWF-SM10-03	MWF	205	0.032	0.434
St. Mary Lk.	MWF-SM10-04	MWF	213	0.027	0.499
St. Mary Lk.	MWF-SM10-05	MWF	212	0.030	0.656
St. Mary Lk.	MWF-SM10-06	MWF	288	0.057	0.402
St. Mary Lk.	MWF-SM10-07	MWF	226	0.054	0.347
St. Mary Lk.	MWF-SM10-08	MWF	253	0.036	0.406

The highest absolute (i.e. non size normalized) mercury levels were observed in large individuals of piscivorous species, and these mercury levels often exceeded various consumption guidelines for human consumers. The highest mercury level (1.16 mg/kg, TL = 886 mm) detected was in a large northern pike from Sherburne Reservoir, and a large lake trout from Cosley Lake was the second highest (1.05 mg/kg, TL = 777 mm). In general, larger individuals of these two species had the highest mercury levels of the fish we tested. Large burbot from St Mary Lake were not captured in 2010, but a large burbot captured in 2008 had mercury levels of concern (0.52 mg/kg, TL = 890 mm).

Conversely Yellowstone cutthroat from Hidden Lake had the lowest Hg concentrations we observed. This may result from the naturally lower propensity for cutthroat trout to bio-accumulate contaminants, as well as a lower supply of MeHg in the food chain. In general, larger individuals of top level predator species such as northern pike, lake trout and burbot had the highest concentrations of Hg. Species of fish at intermediate trophic levels had lower Hg concentrations. Westslope cutthroat from Logging Lake were more contaminated at larger sizes than Yellowstone cutthroat trout from Hidden Lake, suggesting either fewer “links” in the foodweb (i.e. fewer steps to biomagnification), faster fish growth, higher mortality rates, or lower MeHg availability in Hidden Lake (Figure 9). The latter rationale (lower MeHg availability) is the most likely explanation. In addition, Yellowstone cutthroat trout from Hidden Lake failed to show a strong relationship between fish size and Hg concentration. Interestingly, lake trout and lake whitefish from St. Mary Lake also failed to show a clear pattern of size-based bioaccumulation. Broader size ranges of samples from Hidden and St. Mary lakes may help elucidate more subtle patterns, but it does appear as though the fish size-Hg concentration relationship is less pronounced for these species in these waters than it is in other waters.

Lake trout were the most tested species, particularly if the 2008 data are included, and are thus most suitable for examining geographic patterns of mercury contamination across the park. Mercury levels in lake trout from Logging Lake and Cosley Lake were similar with perhaps slightly lower levels in Cosley Lake (Figure 10). The 2008 data from McDonald and Harrison lakes also have similar mercury levels to the current study. St Mary Lake (2008 and 2010) also appears generally to be similar to these five lakes except no obvious mercury versus fish size relationship existed, resulting in higher mercury contamination in the limited sample of small fish. Collectively, it appears that no major differences exist in mercury contamination exist across the low lying, oligotrophic lakes of the park. This pattern suggests that aerial deposition (rather than local sources) is the major source of mercury contamination to park waters as has generally been found in other remote locations.

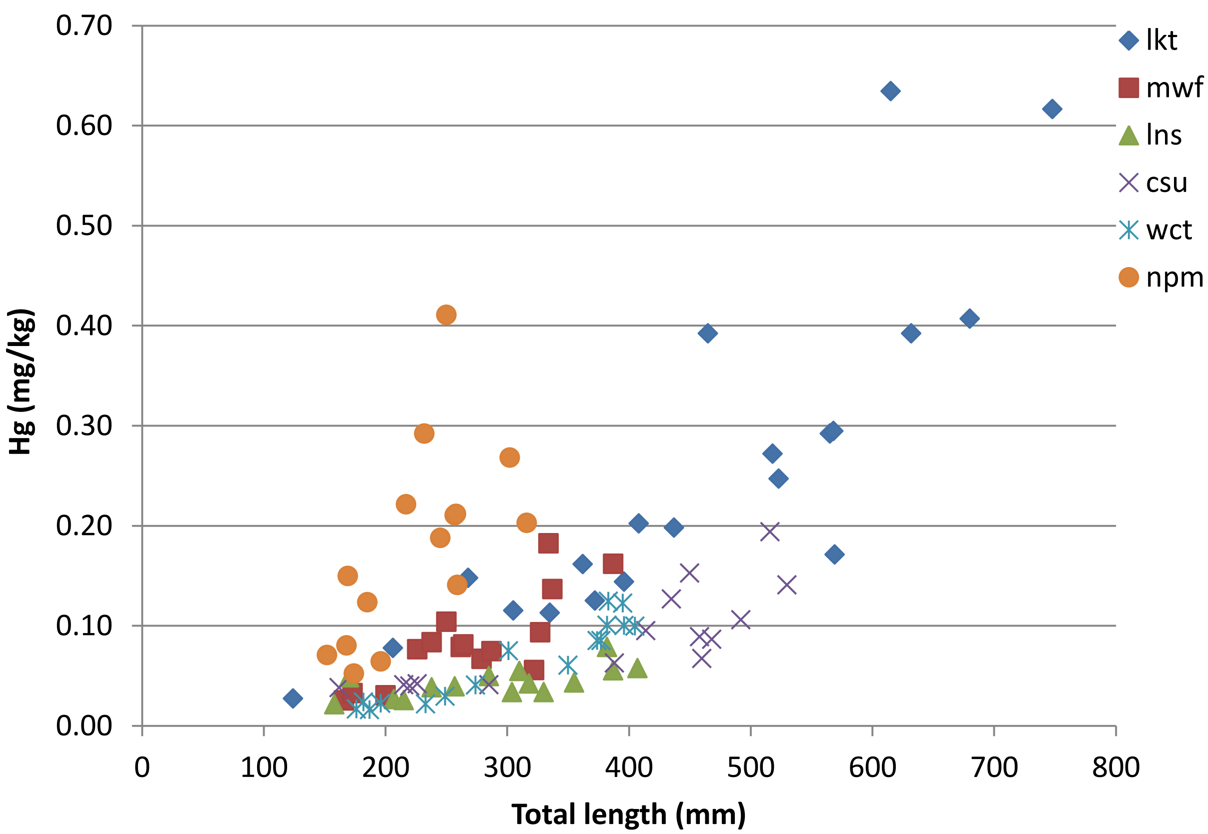

Figure 8. Mercury concentrations derived using fillet tissue (mg/kg, wet-weight) for various species of fish from Logging Lake, Glacier National Park, Montana collected in 2010 (csu = coarsescale sucker, lkt = lake trout, lns = longnose sucker, mwf = mountain whitefish, npm = northern pikeminnow, wct = westslope cutthroat trout).

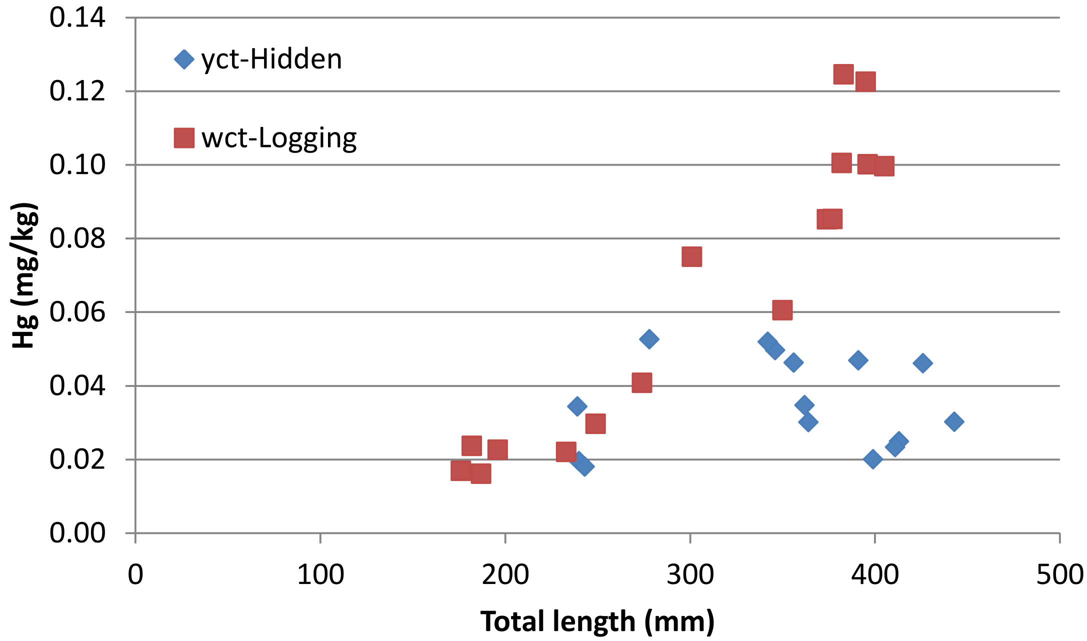

Figure 9. Hg concentrations derived using fillet tissue (mg/kg, wet-weight) for westslope cutthroat trout from Logging Lake and Yellowstone cutthroat trout from Hidden Lake, Glacier National Park.

Watras et al. (1995) found similar total Hg concentrations in lakes in GNP compared to lakes in Wisconsin and New York , but a lower fraction of MeHg in GNP waters. The authors speculated that lower rates of biological methylation activity or weaker organic binding (lower residence time) may be responsible. Water temperature influences bacterial activity (Watras et al. 1995), and it is possible that colder water temperatures explain some of the difference in Hg levels. More recent studies point to the importance of sunlight in breaking down MeHg (Sellers et al. 1996) which may take on particular importance in the clear waters of GNP especially at high elevations. Despite the substantially lower MeHg levels in GNP lake waters relative to New York and Wisconsin, fish mercury levels are only moderately lower in GNP (Watras et al. 1995) suggesting that MeHg moves very efficiently through the oligotrophic food webs in park lakes. The findings of Watras et al. also point to the possibility that lower MeHg levels may exist in the high elevation lakes in the park, and may help explain the lower levels of mercury in larger cutthroat from Hidden Lake relative to those from Logging Lake. In combination, the process of slower MeHg formation and higher demethylation rates in higher elevation waters likely explain some of the differences in fish tissue concentrations between park waters. However, the reason for the lack of an obvious length-MeHg concentration relationship in these waters remains largely speculative.

To put the current GNP Hg results into context, a comparison of Hg values from other nearby lake trout populations was made. Lake trout collected by the U.S. Environmental Protection Agency (EPA) from Upper Two-Medicine Lake in GNP had moderate levels of Hg in fish tissue (0.136 mg/kg; length range 371-406 mm; mean length 389 mm) (T. Selch, MFWP, personal communication). Hg levels in Upper Two-Medicine Lake were similar to those for similar sized lake trout from the west side of the Continental Divide in GNP, but considerably lower than similar sized lake trout from St. Mary Lake. We also compared our data to data from Flathead Lake (Stafford et al. 2004), Yellowstone Lake (Koel et al.

2008), and Swan Lake (L. Rosenthal, MFWP, unpublished data) for lake trout (Figure 11). The comparison reveals that the GNP lakes are similarly contaminated as other nearby, less pristine systems (Swan and Flathead).

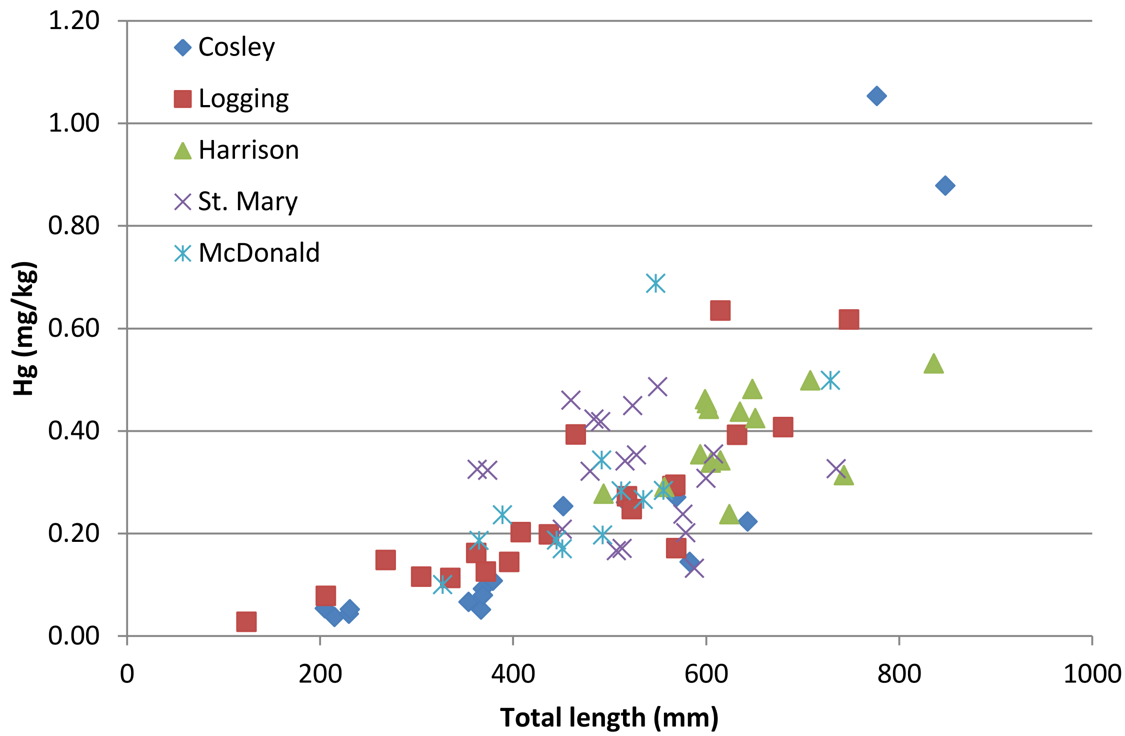

Figure 10. Hg concentrations derived using fillet tissue (mg/kg, wet-weight) for lake trout from various Glacier National Park lakes collected in 2008 and 2010.

EPA provides reference dose data for fish consumption by humans based on fish tissue Hg concentrations (USEPA 2001). Montana Fish, Wildlife, and Parks (MFWP) has taken these data and developed fish consumption guidelines based on fish length and fish tissue Hg concentrations (MFWP 2011). When we apply the MFWP consumption guidance criteria (T. Selch, MFWP, personal communication) to our samples, we can offer some general guidance to anglers who wish to consume fish caught in GNP waters, but also want to minimize their intake of Hg. Scatter plots of the data for lake trout collected from the three west side GNP waters suggested the relationship between fish length and Hg concentration was fairly consistent across these waters (Figure 12). Therefore, we combined the samples across these west-side waters into one single data set and developed guidance based on the combined data. In general, smaller lake trout contained less Hg than larger ones. We used the Hg data collected to adjust existing or develop revised fish consumption guidance for park waters (Table 7). In general, consumption of smaller individuals of a species, or consuming species that feed at lower trophic levels (e.g. cutthroat trout) is a reasonable approach to reducing Hg intake from GNP fish.

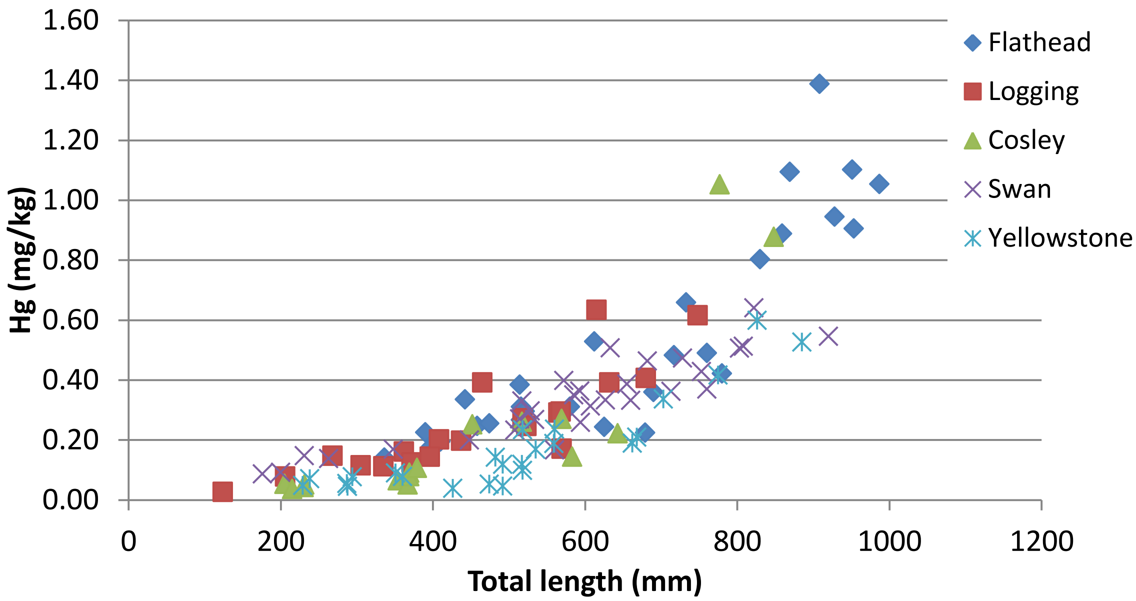

Figure 11. Hg concentrations for lake trout from regional waters compared to Glacier National Park lakes.

We utilized literature-based thresholds to assess risk to piscivorous wildlife from consumption of Hg contaminated fish. Logging and St. Mary lakes were chosen for evaluation to represent fish species assemblages on each side of the park, however adverse weather conditions prohibited adequate sampling on St. Mary Lake. Because Hg concentration varies with fish size and species, we also utilized literature values to estimate an average fish size that could be reasonably be expected to be consumed by piscivorous wildlife (Erlinge 1969, Swenson 1978, Watson et al. 1991, Barr 1996). We estimated whole fish Hg concentrations, and chose standard lengths of 150mm, 200mm, and 300mm to evaluate fish Hg concentrations against established Hg diet thresholds for piscivorous wildlife (Table 8). Because the sample of both sucker species contained few small individuals and appeared to show similar mercury vs. size relationships in the small fish we pooled the suckers for this assessment. Muscle fish tissue concentrations converted to whole fish concentrations were generally low (Table 9). We developed a weighted mean Hg prey value (fish tissue), based on their relative abundance in our netting and used this to represent potential diet of piscivorous wildlife (Table 9; Figure 13). Westslope cutthroat trout would be underrepresented in the analysis because the catch in the sinking nets is biased against westslope cutthroat trout due to behavioral differences between species.

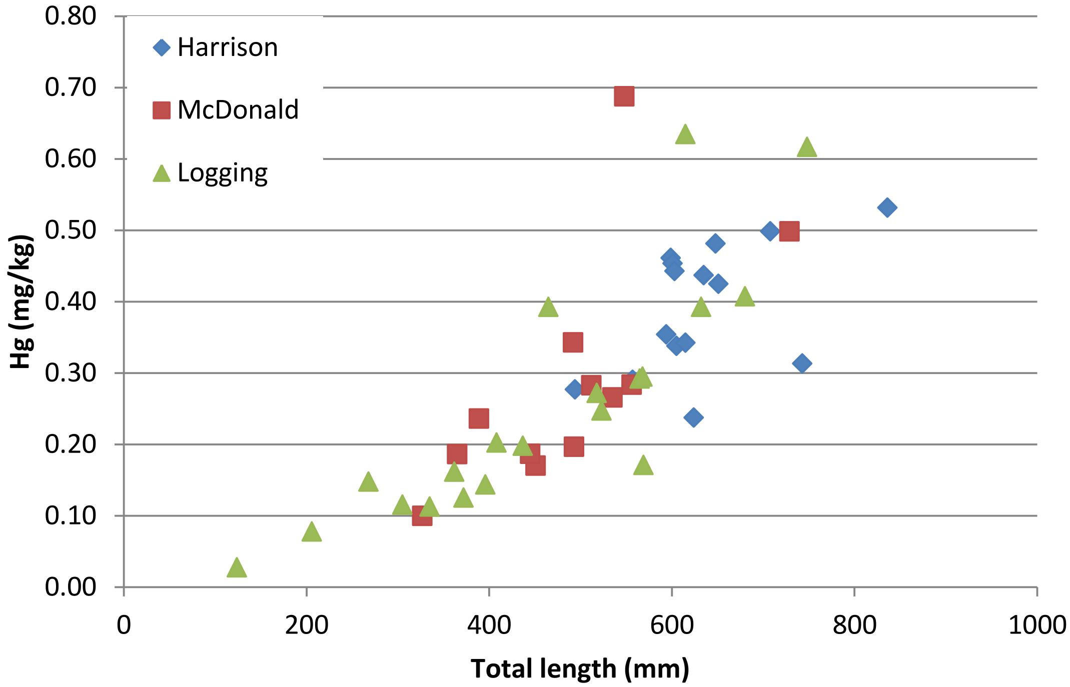

Figure 12. Hg concentrations derived using fillet tissue (mg/kg, wet-weight) for lake trout from lakes west of the Continental Divide in 2008 and 2010, Glacier National Park.

Table 7. Fish consumption guidance (meals per month) for Glacier National Park waters (data collected in 2008 and 2010) based on Montana Fish, Wildlife and Parks Hg consumption guidelines (T. Selch, MFWP, personal communication). U = unlimited.

Water	Species	Sample length range (TL;mm) (sample size)	Mean Hg concentration (mg/kg)	Consumption guidance (M)[a]	Consumption guidance (WC)[b]
Cosley Lk.	lkt	205-379 (9) (8”-15”)	0.065	U	U
		452-643 (5) (17”-26”)	0.230	12	4
		777-848 (2) (30”-34”)	0.965	2	1

Table 7.Continued.

Water	Species	Sample length range (TL;mm) (sample size)	Mean Hg concentration (mg/kg)	Consumption guidance (M)[a]	Consumption guidance (WC)[b]
Lakes west of the Continental Divide	lkt	268-335 (4) (10”-14”)	0.119	U	10
		362-451 (9) (14”-18”)	0.179	U	6
		465-557 (11) (18”-22”)	0.322	8	3
		565-651 (15) (22”-26”)	0.384	7	3
		680-836 (6) (26”-33”)	0.478	5	2
Hidden Lk.	yct	239-443 (17) (9”-16”)	0.035	U	U
Logging Lk.	wct	176-377 (11) (7”-15”)	0.043	U	U
		382-405 (5) 15”-16”	0.109	U	11
	mwf	226-327 (9) (8”-13”)	0.079	U	U
		334-387 (3) (13”-16”)	0.160	U	7
McDonald Lk.	lwf	428-535 (10) (16”-22”)	0.132	U	8
Sherburne Res.	npi	815-996 (7) (32”-40”)	0.768	3	1
	lwf	322-377 (10) (12”-15”)	0.152	U	7
St. Mary Lk.	bur	225-378 (3) (8”-15”)	0.072	U	U
		405-583 (7) (15”-23”)	0.192	U	6
		596-890 (3) (23”-36”)	0.433	6	2
	lkt	363-735 (19) (14”-29”)	0.316	9	3
	lwf	397-524 (16) (15”-21”)	0.147	U	8

[a] Maximum number of meals per month recommended for men and women not of reproductive age. Based on a single 8-ounce fillet and a 150 lb. person.
[b] Maximum number of meals per month recommended for women of child-bearing age and children under 6. Based on a single 6-ounce fillet.

Table 8. Effect levels for consumption of mercury contaminated fish by piscivorous wildlife.

Species	Measure	Prey concentration (wet weight; whole fish, mg/kg)	Citation
Common loon (chick) *Gavia immer*	No observed effect level	0.08	Review in Ambio 36 (1):12-18
Common loon (adult)	Reduced reproductive success	0.21	Burgess and Meyer 2008
Common loon (adult)	Reduced reproductive success	0.16	Evers et al. 2007
Belted kingfisher *Ceryle alcyon*	Protective Wildlife Value	0.03	Lazorcheck et al. 2003
Mink *Mustela vison*	Protective Wildlife Value	0.07	Lazorcheck et al. 2003
Otter *Lutra canadensis*	Protective Wildlife Value	0.10	Lazorcheck et al. 2003

Table 9. Predicted Hg concentrations (wet weight, whole body) at 150mm, 200mm and 300mm (TL) for various fish species from Glacier National Park. Size groups are defined as fish sizes reasonably assumed to be consumed by the specified predator groups.

		kingfisher/loon/mink/otter		osprey/eagle
Water	Species	Predicted 150 mm Hg (mg/kg)	Predicted 200 mm Hg (mg/kg)	Predicted 300 mm Hg (mg/kg)
Logging	lkt	0.040	0.048	0.072
	mwf	0.021	0.030	0.062
	npm	0.054	0.081	0.180
	sucker spp.	0.022	0.026	0.038
	wct	0.011	0.016	0.034
	Weighted Mean Concentration	**0.030**	**0.042**	**0.082**
St. Mary	bur	N/A	0.040	0.056
	lns	0.019	0.022	0.030
	mwf	N/A	0.021	0.050

Our wildlife risk assessment suggests that most fish species collected in the waters in this study (Logging and St. Mary lakes) had Hg concentrations at the size ranges typically consumed by piscivorous wildlife that do not appear to pose a significant mercury risk to piscivorous wildlife (Table 9). One notable exception was northern pikeminnow. Even small northern pikeminnow had Hg levels that could be cause for concern if significant prey selection for this species occurred (Figure 8, Table 9). However, it is more likely that they are part of a diet containing other less-contaminated species such as suckers and westslope cutthroat trout. A broad diet, reasonably approximated by our estimated species composition from netting is a key assumption of our analysis. Barr (1996) reported suckers to be readily captured and consumed by loons. It is also likely that westslope cutthroat trout make up a more significant proportion of piscivorous wildlife than our sampling suggests, due to their upper water column habitat preference. Small westslope cutthroat trout had some of the lowest Hg concentrations of any fish species we evaluated. We believe this proportional diet approach to estimating Hg risk would provide a more appropriate context than a single fish species best or worst case scenario approach. Actual piscivorous wildlife diet data would be useful in validating our assumptions.

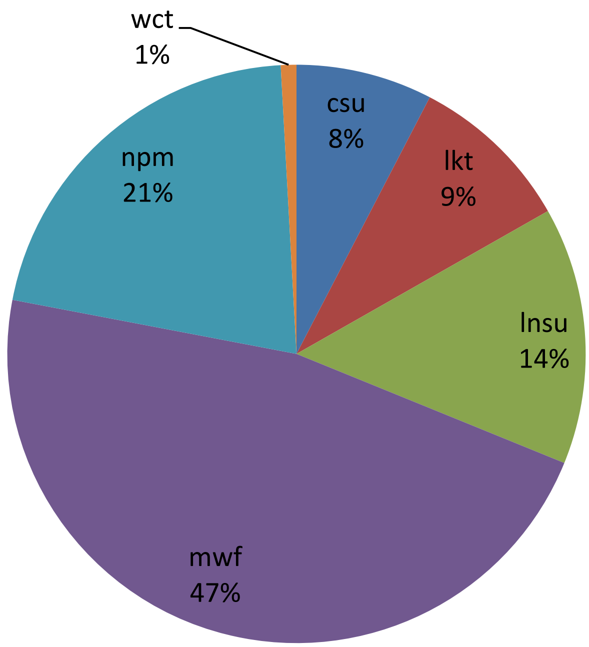

Figure 13. Catch composition in sinking gill nets from Logging Lake, Glacier National Park.

Landers et al. (2008) concluded that whole fish from Oldman and Snyder lakes exceeded the protective Hg consumption threshold of 0.03 mg/kg for kingfishers (Lazorcheck et al. 2003), but it is unclear if the size of fish sampled would be likely to be consumed by kingfishers. The relationship between fish size and Hg concentration in Hidden Lake was weak (Figure 10), and our whole fish mean Hg level across all sizes in Hidden Lake was 0.026 mg/kg. These fish ranged in size from 239-443mm, considerably larger than would be expected to be regularly consumed by kingfishers. Despite the weak length-Hg relationship in Hidden Lake, it is likely that smaller fish do have lower Hg levels.

Protective wildlife value (WV) thresholds (Lazorcheck et al. 2003) are conservative estimates of Hg concentrations below which the organism being studied is assumed to be protected from adverse impacts. These are not necessarily levels immediately above which adverse impacts occur. Development of prey-based WV's is limited to a few species, and we relied on additional literature to assess risk to other species**.** Evers et al. (2004) concluded that an average of 0.16 mg/kg wet-weight in the diet of common loon was a threshold for adverse reproductive impacts. We did not find similar prey-concentration Hg literature values available for other species such as bald eagle and osprey.

Our analysis assumes that piscivorous wildlife consume prey based on its availability, and that our netting was representative of the fish community available to wildlife. Although not without limitation, making these assumptions is a reasonable starting place for assessment of potential impacts. We were unable to find prey-based protective wildlife criteria for bald eagle *Haliaetus leucocephalus* and osprey *Pandion halaietus*. Birds vary in their vulnerability to adverse impacts from Hg for multiple reasons, and developing protective consumption limits based on prey concentrations, such as those developed by Lazorcheck et al. (2003,) would facilitate a better assessment of risk.

We also measured wet-weight Se tissue concentrations in a sub-sample of fish. Se, an essential element, can also be toxic to wildlife at elevated levels (Presser et al. 1994). Se can be leached from soil/parent material at an accelerated rate in disturbed environments, such as agricultural and mining areas which can result in biota being exposed to harmful concentrations (Presser et al. 1994, Palace et al. 2004). However, concentrations were well below aquatic toxicity thresholds of about 8 mg/kg (dry weight) in fish muscle tissue (Lemly 1993, Essig and Kosterman 2008) (Table 6). Baseline information regarding this element in fish has relevancy in light of previously proposed mining activity in the North Fork Flathead basin. In addition, our data provides the first baseline information on Se levels in fish tissue from park lakes.

Our main impetus for determining Se levels in GNP fish is the growing scientific evidence that Se greatly affects the fate of mercury in aquatic food chains, and ultimately moderates its toxicity (Cuvin-aralar and Furness 1991, Raymond and Ralston 2004, Yang et al. 2008, Peterson et al. 2009). It has been hypothesized that Se combines with MeHg in tissues to form highly stable compounds that are biologically inactive, although multiple mechanisms may be at work (Yang et al. 2008). Thus, Se has the potential to moderate MeHg toxicity and at the same time cause potential Se deficiencies in organisms (Peterson et al. 2009). Protective effects of Se against Hg toxicity are premised on sufficient Se in the diet to provide not only for potential binding with ingested Hg, but also to provide for healthy function of various systems within the organism.

In an assessment of Hg toxicity risk in stream fishes across the west, Peterson et al. (2009) used a Se:Hg molar ratio >1 in fish tissues (whole fish) as a threshold for potential protection of both the fish and potential fish consumers. We therefore calculated molar ratios for all fish samples analyzed for both Se and Hg. Almost all of our Se:Hg ratios were >1.0, which we presume is favorable to reduced Hg toxicity (Figure 14), however our analysis was conducted on fish fillets (muscle tissue) only. As wildlife generally consume the entire fish, our analysis could be improved by estimating the Se:Hg molar ratio from whole fish tissue. Of the species and sizes analyzed, only the largest lake trout had Se:Hg ratios < 1.0. This makes sense as Hg concentrations generally increase with size, while Se concentrations do not. This would also be consistent with typical fish consumption guidance that suggests smaller fish pose lower risk of adverse impacts from MeHg consumption.

At some point, science may advance to a point where Se and Hg together will be used to determine "risk" associated with consuming fish, and these data will be useful in future assessments. Interestingly, Se addition has been evaluated as a method to remediate elevated Hg concentrations in lake systems in order to reduce fish tissue concentrations (Rudd et al. 1980, Rudd et al. 1983, Paulsson and Lundbergh 1989). Studies to date suggest a protective relationship from Hg toxicity when Se is present in sufficient amounts. However, it is not entirely clear at what concentrations Se may provide this protection, and if consuming prey species with a >1 molar ratio of Se:Hg protects predatory species from Hg toxicity. This would have significant bearing on Hg risk assessments in aquatic and terrestrial foodchains, and would be of value for resource managers currently evaluating Hg risk based on Hg tissue concentrations alone.

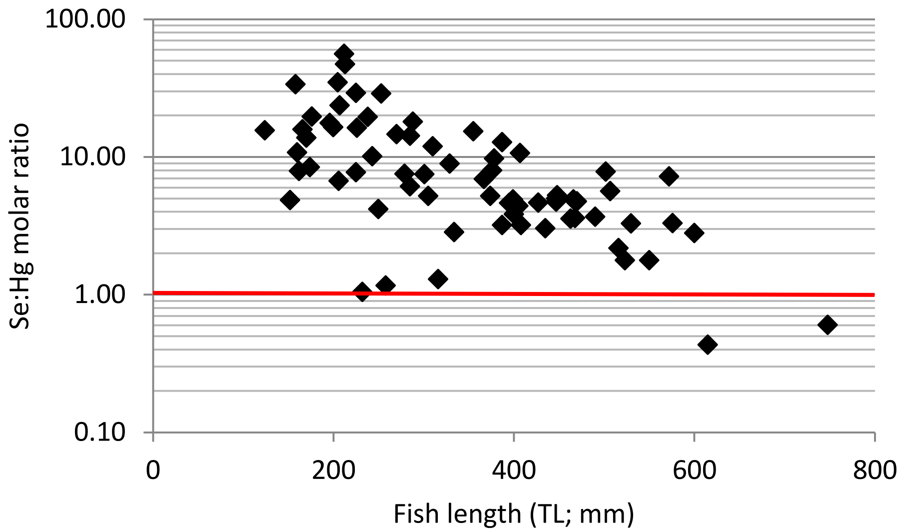

Figure 14. Se:Hg molar ratios (fillet tissue) for fish collected from Logging and St. Mary lakes, Glacier National Park.

While not exhaustive, our study suggests that for piscivorous wildlife, fish tissue concentrations in the waters evaluated in this study were near or below levels that would be expected to cause impairment or adverse population impacts. However, these conclusions are limited in that we were unable to sample smaller fish from all species and waters (i.e. St. Mary Lake) and we only tested Se levels in two lakes. Given the variation we observed between Logging Lake and St Mary Lake it is likely that Se values in fish are variable across other waters. Hg levels Impacts to juvenile birds are not well addressed in this study (or other studies), largely due to a lack of information on risk thresholds and diet composition. Further refinement of piscivorous wildlife diet, establishment of protective wildlife values for other piscivorous wildlife species, and development of regional protective wildlife values would improve the utility of our data, facilitate additional risk assessment, and also assist in validating our conclusions.

These results suggest that aerial deposition of mercury is of concern in GNP, yet risk to piscivorous wildlife based on our analysis appears relatively low. However, unproductive water bodies such as those in GNP may be at greater risk for biomagnification of mercury because the slow growth rates of fish reduce biodilution (Thomann 1989, Stafford and Haines 2001; Stafford et al. 2004), and because there is less organic matter in these systems to immobilize the contaminants (Pickhardt et al. 2002) and facilitate its burial in the sediments. Organic biomagnified contaminants should also be considered for testing especially given the moderately high levels of DDE, chlordane's, and dieldrin found recently in GNP cutthroat trout (Landers et al. 2008). These results are particularly concerning as cutthroat presumably have a very low propensity to biomagnify organic contaminants due to their small size, low fat content, and invertebrate feeding pattern. It is possible that levels of organic contaminants are much higher in large lake trout given their piscivorous feeding behavior high fat content, and longevity and future testing should be a priority.

The processes of Hg release, transport, deposition, transformation, biomagnification, and biological impact are complex and not completely understood. Studies such as this provide baseline information on Hg and Se concentrations in a protected area. These data will be useful in monitoring impacts to GNP resources as Hg sources and emissions change over time. They also may serve useful in larger-scale assessments of Hg patterns and impacts. Should relationships between Se and Hg become better understood with regard to biological impact, GNP will have data to further assess risks to humans and wildlife.

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