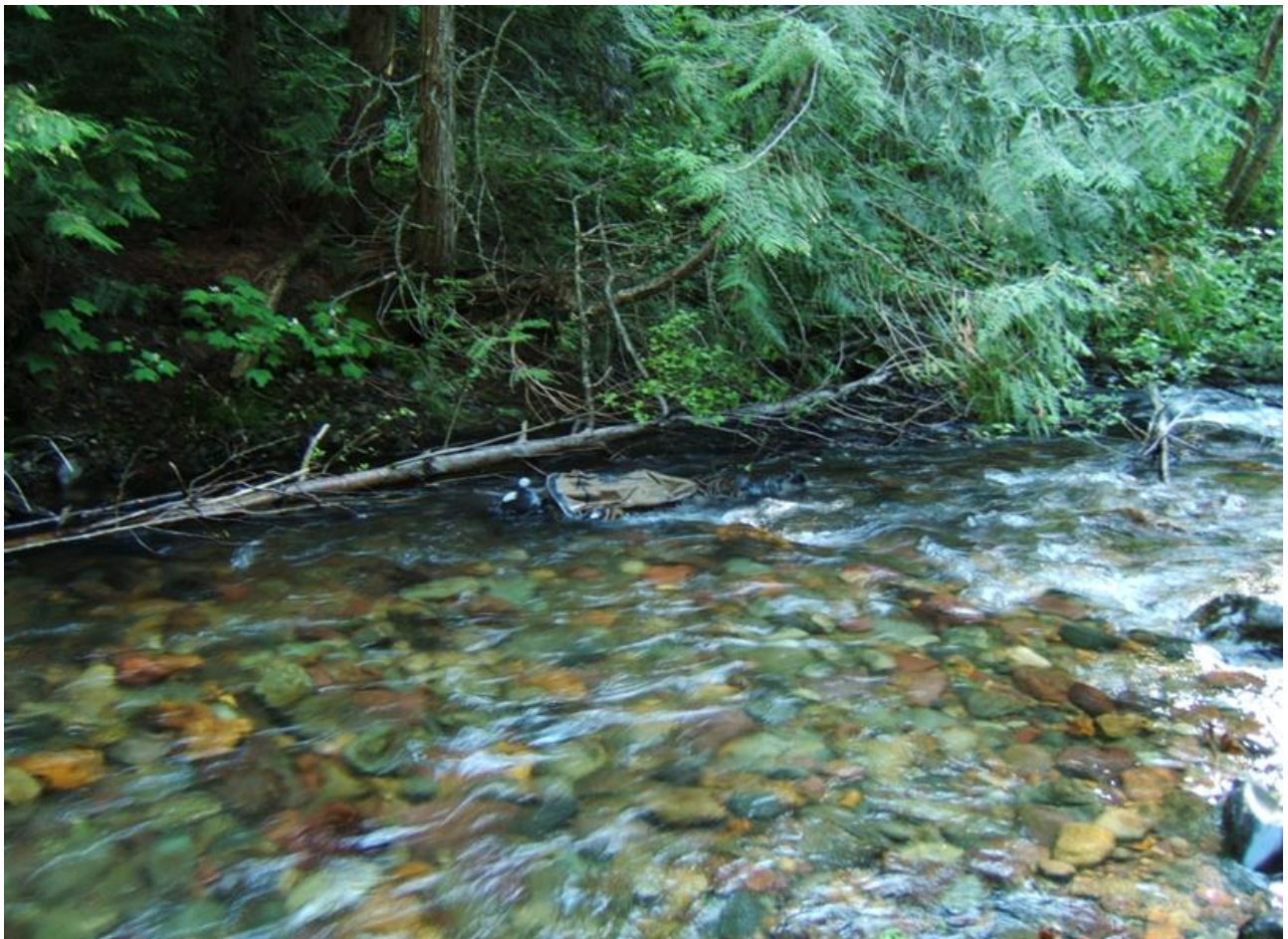
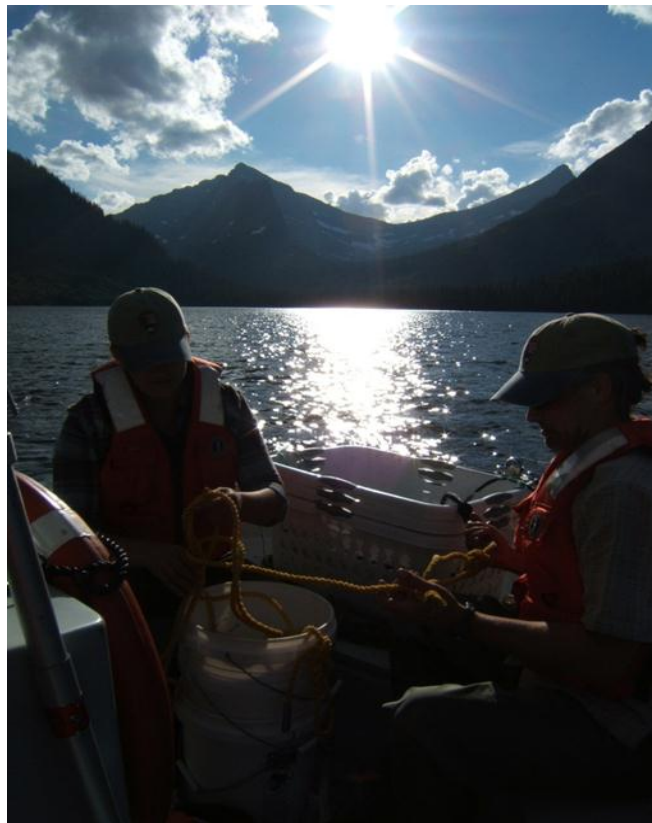


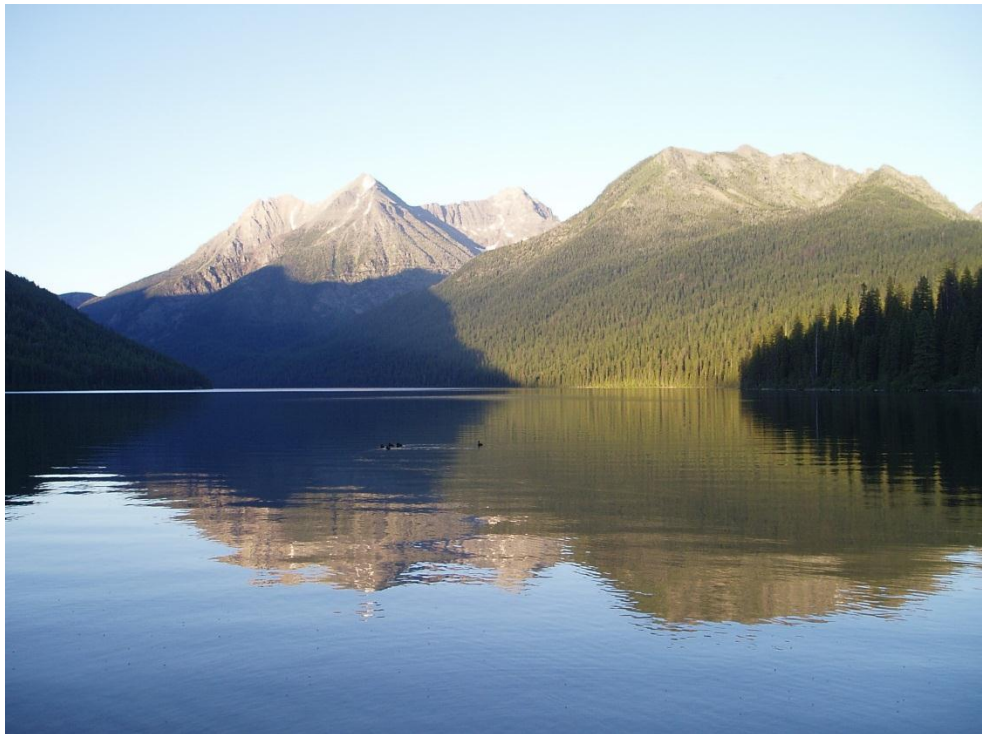
Glacier National Park Fisheries Inventory and Monitoring

Program Report 2013



**Glacier National Park Fisheries Inventory and Monitoring
Program Report 2013**





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Front cover photo caption: Snorkel survey on lower Fish Cr.

Inside photo captions (top to bottom): Collecting data after electrofishing, trend gill net sampling on Two Medicine Lake, snorkel survey in classic west side creek, and view from the patrol cabin on Quartz Lake.

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Native Fish Population Monitoring

ABSTRACT

In 2009, Glacier National Park (GNP) 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 continued this work through 2013, conducting sampling in 13 streams. Many of the streams also support amphibians, such as tailed frogs *Ascaphus truei* and spotted frogs *Rana pretiosa*. We noted presence or absence along with relative abundance of amphibians in our surveys. These data will prove useful in evaluating changes in distribution of these amphibians over time. 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 St. Mary River. In general, the largest threat currently facing migratory native fish species on the east side of the park is the unscreened St. Mary River irrigation diversion near Babb, part of the Milk River Irrigation Project. The most significant threat to native fish on the west side of the park comes from invasive non-native species, such as rainbow and lake trout. Some culverts along the Camas Road on the west side of the park may be important isolating mechanisms, protecting genetically pure westslope cutthroat from downstream populations of rainbow trout. However, 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. Additionally, research this year included testing and developing snorkeling techniques to monitor stream-dwelling fish. We snorkeled eight streams to evaluate the feasibility, effectiveness, and efficiency of snorkeling compared to electrofishing. Smaller streams with greater habitat complexity (i.e. lots of large woody debris) were more difficult to effectively survey using snorkeling techniques than larger streams with lower habitat complexity. Streams such as upper McDonald Creek and the lower reaches of Akokala Creek appear well suited to snorkel surveys.

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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 inhabits 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 <i>Thymallus arcticus</i>	--	--	I
Brook trout (bkt) <i>Salvelinus fontinalis</i>	I	I	I
Bull trout (blt) <i>S. confluentus</i>	N	--	N
Kokanee <i>Oncorhynchus nerka</i>	I	--	I
Lake trout <i>S. namaycush</i>	I	--	N
Lake whitefish <i>Coregonus clupeaformis</i>	I	--	N
Mountain whitefish (mwf) <i>Prosopium williamsoni</i>	N	N	N
Pygmy whitefish <i>P. coulteri</i>	N	--	N
Rainbow trout (rbt) <i>O. mykiss</i>	I	I	I
Westslope cutthroat trout (wct) <i>O. clarkii lewisi</i>	N	N	N
Yellowstone cutthroat trout <i>O. c. bouvieri</i>	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 <i>Pimephales promelas</i>	--	--	--
Northern pikeminnow <i>Ptychocheilus oregonensis</i>	N	--	--
Peamouth <i>Mylocheilus caurinus</i>	N	--	--
Redside shiner <i>Richardsonius balteatus</i>	N	--	--
Longnose sucker <i>Catostomus catostomus</i>	N	N	N
Largescale sucker <i>C. macrocheilus</i>	N	--	--
White sucker <i>C. commersoni</i>	--	N	N
Deepwater sculpin <i>Myoxocephalus thomsoni</i>	--	--	N
Mottled sculpin <i>Cottus bairdi</i>	--	N	N
Slimy sculpin <i>C. cognatus</i>	N	--	--
Shorthead sculpin <i>C. confusus</i>	N	--	--
Spoonhead sculpin <i>C. ricei</i>	--	--	N
Burbot <i>Lota lota</i>	--	--	N
Northern pike <i>Esox lucius</i>	--	--	N
Trout-perch <i>Percopsis omiscomaycus</i>	--	--	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. 2009a, 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 (Mogen and Kaeding 2004). 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. This report represents a summary of data collected during 2013 field season.

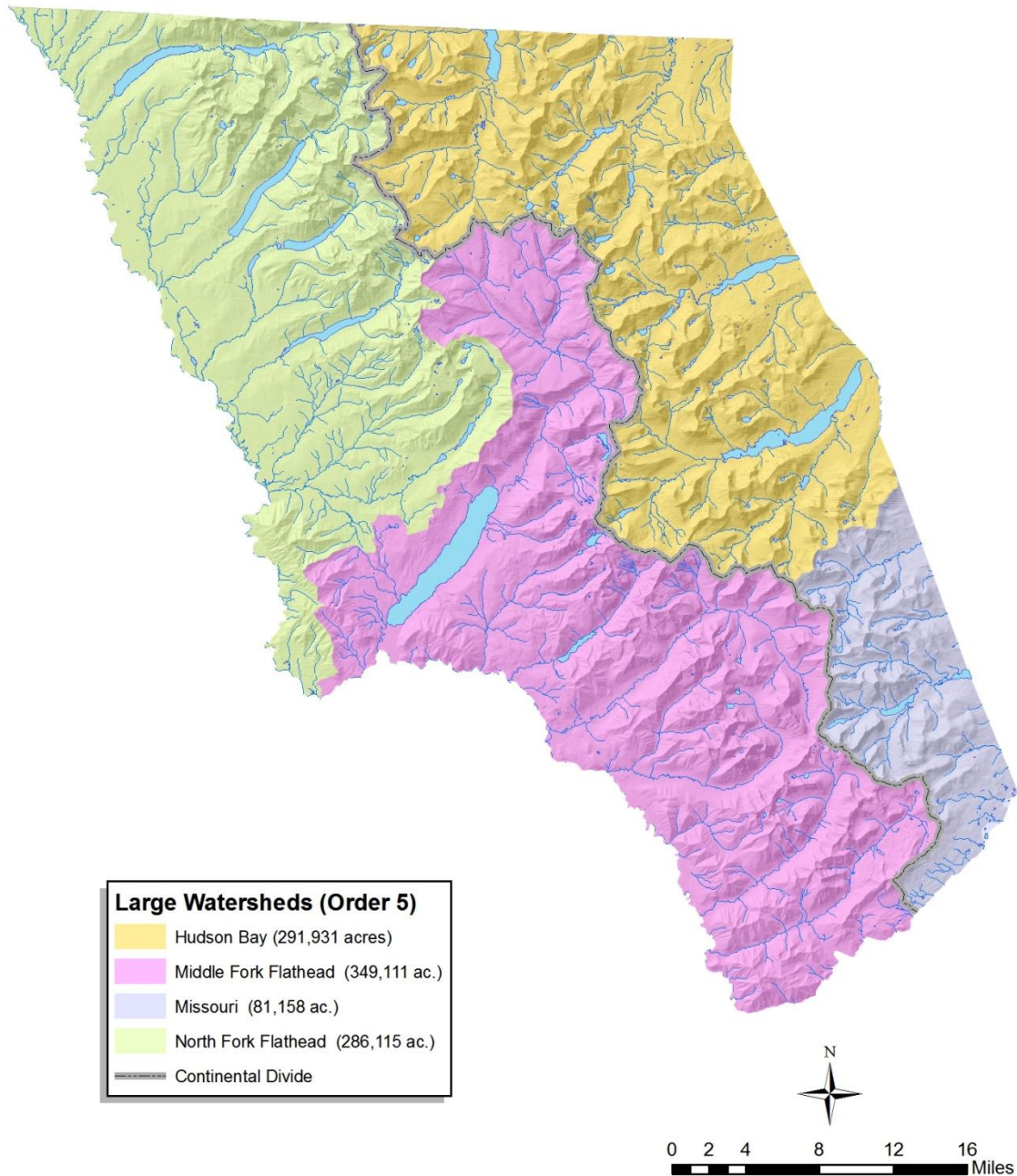


Figure 1. Major watersheds of Glacier National Park, Montana.

METHODS

In 2009, we selected 19 streams for initial survey based on their previous sampling history, species composition, and accessibility (Figure 2). Some of these streams were not sampled in subsequent years for a variety of reasons including logistics and depletion feasibility, resulting in 13 streams being sampled from 2010-2013 (Table 3). We identified potential sample streams using a variety of means including previous sampling history, native fish population strength, and those that could be sampled using backpack electrofishing equipment or snorkeled by a two-person crew in one day. The intent is to identify a group of core streams to monitor across the park to characterize native fish population trends into the future.

Electrofishing to conduct removal (depletion) estimates (Zippin 1958) has been our primary stream sampling method, but in 2013 we began conducting snorkel surveys (Thurrow 1994) to establish abundance trend data sets. 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 30% or less of the catch on the first pass for age-1 and older juvenile salmonids.

Population estimates from the depletion data were calculated using 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/100m² and fish captured/hr of electrofishing to facilitate comparison with other sampling efforts occurring in the park. 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.

Fish were anesthetized, identified to species, measured (total length (TL); mm) and weighed (g). Fulton Condition Factor (K) was calculated for age-1 and older wct and blt (Anderson and Neumann 1996). 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.

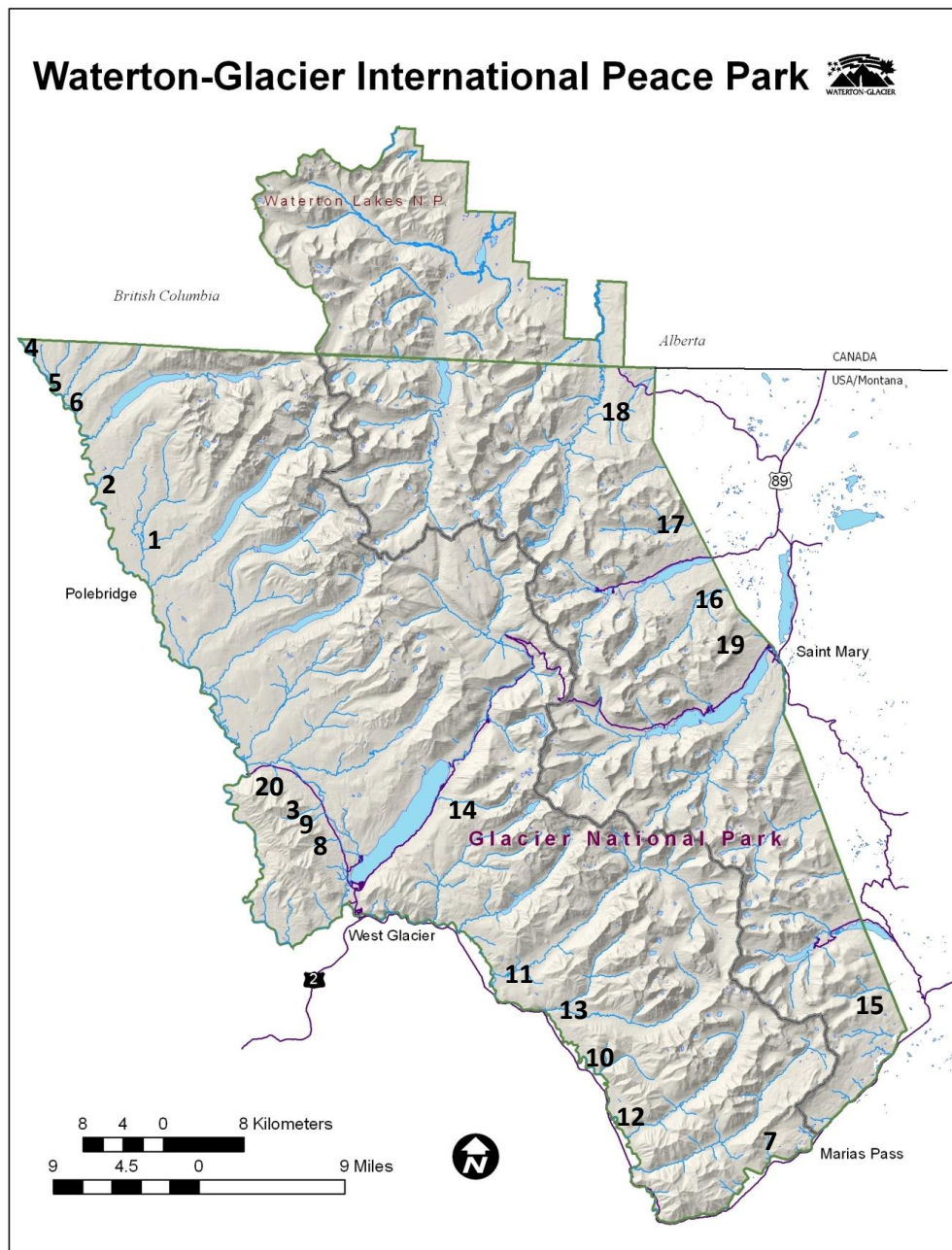


Figure 2. 2009-2013 stream sampling sites for depletion population estimates, Glacier National Park, Montana. Site numbers from Table 3.

Table 3. Stream name, site location, and date sampled with electrofishing for 2009-2013 sampling sites.

Major drainage	Stream name (Site No.)	Location of downstream end of reach (WGS 84, UTM)	Dates sampled	Sample Reach Length (m)
N. Fk. Flathead	Akokala (1)	11U 0699251N, 5408441W	8/3/09; 7/30/10	138
	Ford (2)	11U 0694157N, 5417724W	7/20/09	93
		11U 0693673N, 5416862W	7/20/10; 7/30/13	96
	(Top of reach)	11U 0693774N, 5416977W	8/2/11; 7/23/12	120
	McGee (3)	11U 0718256N, 5386976W	7/10/09; 7/29/13	76
		11U 0718256N, 5386976W	7/18/10	50
		11U 0718256N, 5386976W	7/26/12	144
	Spruce (5)	11U 0688239N, 5427453W	7/27/09; 7/21/10	122
	No Name (20)	11U 0713098N, 5390261W	8/4/11; 7/17/12; 7/31/13	90
	Starvation (6)	11U 0695225N, 5428464W	7/29/09	99
M. Fk. Flathead	Autumn (7)	12U 0320621N, 5352007W	7/15/09; 7/14/10; 7/20/11; 7/24/12	110
	Fern (8)	11U 0720216N, 5383473W	7/9/09; 7/15/10; 7/18/11; 7/18/12; 7/2/13	90
	Fish (9)	11U 0718665N, 5386000W	7/10/09; 7/18/10; 7/19/11; 7/19/12; 7/31/13	112
	Muir (10)	12U 0303790N, 5360495W	7/23/09; 8/4/10; 8/16/11; 8/9/12	80
	Midvale (15)	12U 0332437N, 5368436W	7/22/09	
Missouri		12U 0330666N, 5368622W*	7/28/10	
St. Mary (Hudson Bay)	Boulder (16)	12U 0314860N, 5408095W	8/12/09; 8/18/11; 8/15/13	131
	Lee (18)	12U 0307481N, 5428586W	7/21/09; 8/10/11; 8/15/12; 8/16/13	125
	Wild (19)	12U 0319740N, 5403793W	7/15/09	128
		12U 0319740N, 5403793W	8/9/11; 8/13/12; 8/14/13	95

*Represents the bottom of the suppression reaches. Entire stream length was sampled.

2013 was the first year that snorkeling has been seriously pursued in Glacier to monitor fish populations. We surveyed eight creeks within the park (Figure 3, Table 4). Surveying for fish using standardized snorkeling techniques has long been understood to be a valid method to survey abundance, diversity, and size of fish, along with observing habitat usage and distribution in streams (Thurow 1994). Snorkel surveys have some limitations, including the inability to collect detailed length and weight information on fish. Snorkeling can be more suitable for backcountry work, requiring less cumbersome equipment (D'Angelo 2010). Moreover, though electrofishing is fairly innocuous, it is possible to cause injury or death to fish.

Standard snorkeling protocols for GNP were formulated in 2012 based on research published by Thurow (1994). We defined three size classes of fish; Class I: <45mm TL, Class II: 46-149mm TL, Class III: >150mm. Surveyors snorkeled in an upstream direction. At least one observer walked alongside the snorkeler measuring stream width, recording time snorkeler is surveying, species found, and to watch for safety hazards. When creek size allowed, two snorkelers moved upstream side by side. Fish were not counted in the survey until the snorkeler had moved past them in order to avoid duplicate counting of fish. Snorkel estimates involved identifying a representative reach of stream approximately 100 m long. We used a high gradient riffle break or other natural drop in the stream channel bed as section starting and ending points. When possible, snorkel surveys were performed within previously sampled electrofishing or historical snorkeling (e.g. Read et al. 1982) sections.

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) ≥ 45 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. These assumptions are further supported by other studies. In some cold systems similar to 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. 45 mm) as a cutoff for inclusion in estimates of age-1 and older wct is more appropriate for most park waters containing rearing wct. Fishes of these sizes (blt ≥ 60 mm and wct ≥ 45 mm) can be efficiently sampled with electrofishing gear and thereby provide for estimation of abundance and catch per unit effort (CPUE). Water temperatures were continuously recorded over the course of the summer using temperature loggers in Ole and Lee creeks.

For both electrofishing and snorkeling surveys, GPS coordinates of the upstream and downstream end of the sections were recorded in UTM using the WGS 84 datum. 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. 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.

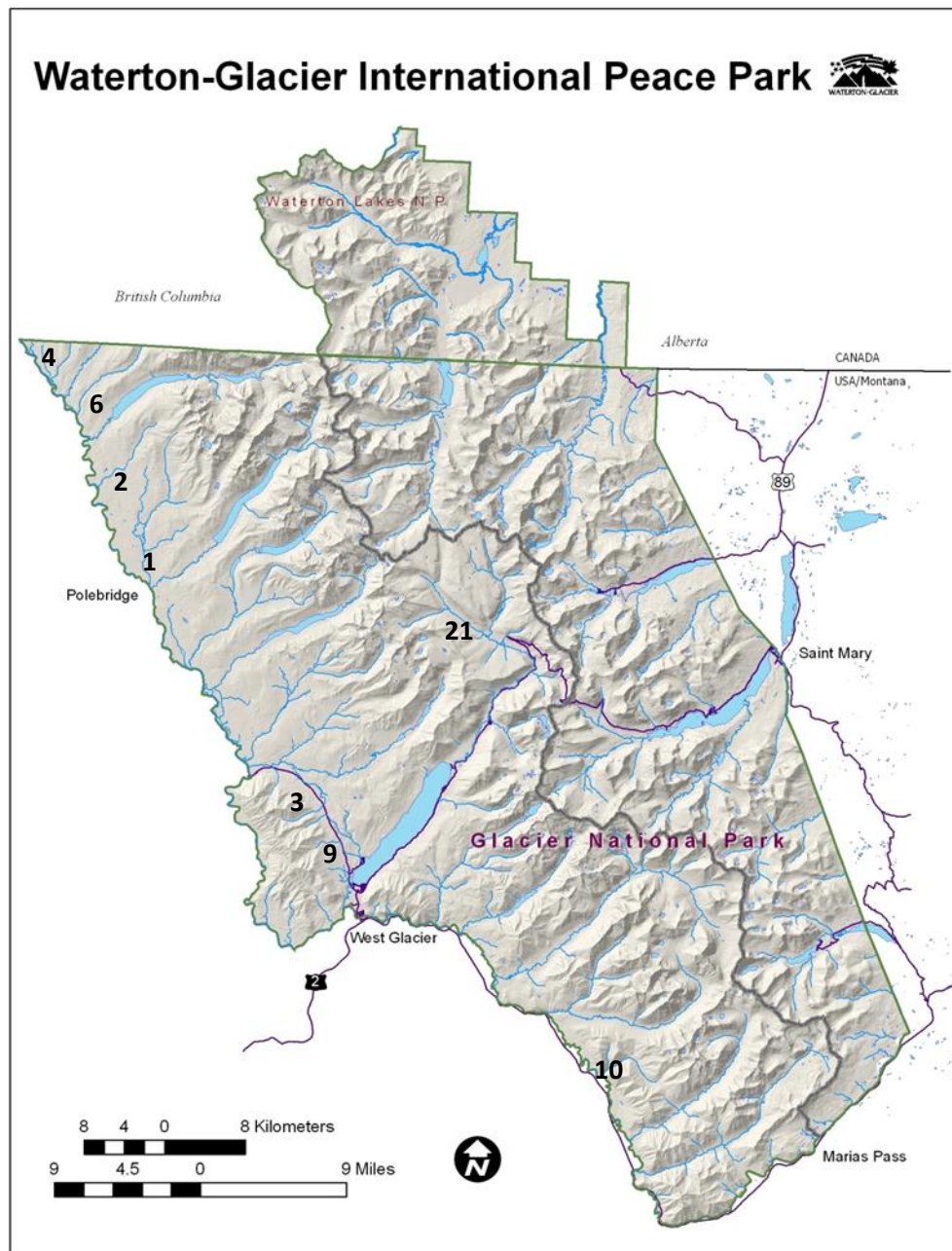


Figure 3. 2013 stream sampling sites for snorkel surveys, Glacier National Park, Montana. Site numbers are from Table 4.

Table 4. Stream name, site location, and date sampled with snorkeling in 2013.

Major drainage	Stream name (Site No.)	Location of downstream end of reach (WGS 84, UTM)	Date sampled	Sample Reach Length (m)
N. Fk. Flathead	Akokala (1)	11U 0699251N, 5408441W	8/1/13	143
	Ford (2)	11U 0693673N, 5416862W	7/30/13	121
	McGee (3)	11U 0718256N, 5386976W	7/29/13	60
	Sage (4)	11U 0685857N, 5430016W	8/19/13	250
	Starvation (6)	11U 0695225N, 5428464W	8/13/13	200.3
	Fish (9)	11U 0718665N, 5386000W	7/31/13	112
	Muir (10)	12U 0303790N, 5360495W	8/21/13	196.6
	Upper McDonald Cr (21)	12 U 0292335N, 5403907W	8/20/13	329

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 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). Hybridization between rainbow and wct was recently detected in the lower reaches of Akokala Creek, but we do not believe it has progressed into headwater areas. 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. 2009). Hybridization has also been recently detected in the lower reaches of Akokala Creek (MFWP, unpublished data). 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). Similarly, genetic testing in 2008 failed to detect hybridization at the NPS trail crossing in the upper portions of the drainage. 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 wct population. Migratory wct populations are the most vulnerable wct life-history because they depend on large connected lake-river systems which have become increasingly rare. They are also more likely to be adversely impacted by

non-native fish species due to the highly modified fish assemblages of major lake systems across their native range.

A 143m reach of Akokala Creek was snorkeled on 8/1/2013 in approximately the same location as historic snorkel sampling (Read et al. 1982). Two snorkelers were used to survey given the width of the creek. Read et al. (1982) reported a density of age-1 and older of 1.8 wct/100m². We observed 15 wct <45mm, 65 wct between 46-149 mm, and 19 wct >150 mm (Table 5). Using the same average wetted width as measured in 1981, we estimated a density of 5.65 age-1 and older wct/100m². Furthermore, 6 mwf <49 mm were observed, along with 6 mwf between 46-149mm, and 2 mwf >150 mm (Table 5).

Table 5. Size class, species, and count in a side by side snorkel survey in Akokala Creek in 2013.

<u>Species</u>	<u>Diver 1</u>	<u>Diver 2</u>
WCT <45 mm	10	5
WCT 46-149 mm	46	19
WCT >150 mm	11	8
MWF <45 mm	0	6
MWF 46-149 mm	2	4
MWF >150 mm	1	1

This sampling reach is located at a historic snorkeling reach (Read et al. 1982) approximately one kilometer upstream from the inside North Fork Road, in what is presumed to be migratory wct spawning and rearing habitat. Previous studies (Muhlfeld et al. 2009b) have documented use of Akokala Creek by migratory westslope cutthroat trout from Flathead Lake/River .

Gravel was estimated to be the dominant substrate type followed by sand in the reach. Tailed frogs (*Ascaphus truei*) were present, but rare. Water temperature at the time of sampling was 11°C.

Physical habitat characteristics were comparable for this reach to those reported by Read et al. (1982), who sampled 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. This stretch of Akokala Creek is ideal for snorkeling with its wide channel, numerous pools, and low density of woody debris.

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, Downs et al. 2011, Downs et al. 2012), and both westslope cutthroat trout and sculpin were detected. Genetic analysis conducted by Muhlfeld et al. (2009a) did not detect hybridization between wct and rainbow trout in Ford Creek.

2013 was the first time Ford Cr has been snorkeled since Read et al. (1982). Snorkeling was done in our electrofishing reach which is lower in the drainage than the area sampled by Read et al. (1982). Since we were going to electrofish the reach after snorkeling we set up a bottom reach block net. Snorkeling was performed over a reach of 121.1 m. A total of 5 wct were observed, although only 4 were age-1 and older wct (Table 6). There was a substantial discrepancy between the snorkel count and the electrofishing sample. One factor that might have contributed to such an inconsistent count was the presence of several expansive log jams with multiple pools that were difficult to effectively snorkel.

Table 6. Size class, species, and count of all individuals of other species observed by snorkeling in Ford Creek in 2013.

<u>Species</u>	<u>Diver 1</u>
WCT <45 mm	1
WCT 46-149 mm	4
WCT >150 mm	0

Once snorkeling was completed we sampled the creek with the electrofisher. We captured one bull trout (185 mm and 59 g) in the two passes administered. Of the 50 wct captured, 41 were age-1 and older. Furthermore, 38 sculpin were also captured.

Wct ranged in length from 41 to 163 mm (Figure 4). Average length of age-1 and older wct was 96.2 mm (Table 7). Average Fulton Condition Factor (K) was also estimated at 1.1 for age-1 and older wct (SD = 24.3) (n = 41) (Table 7).

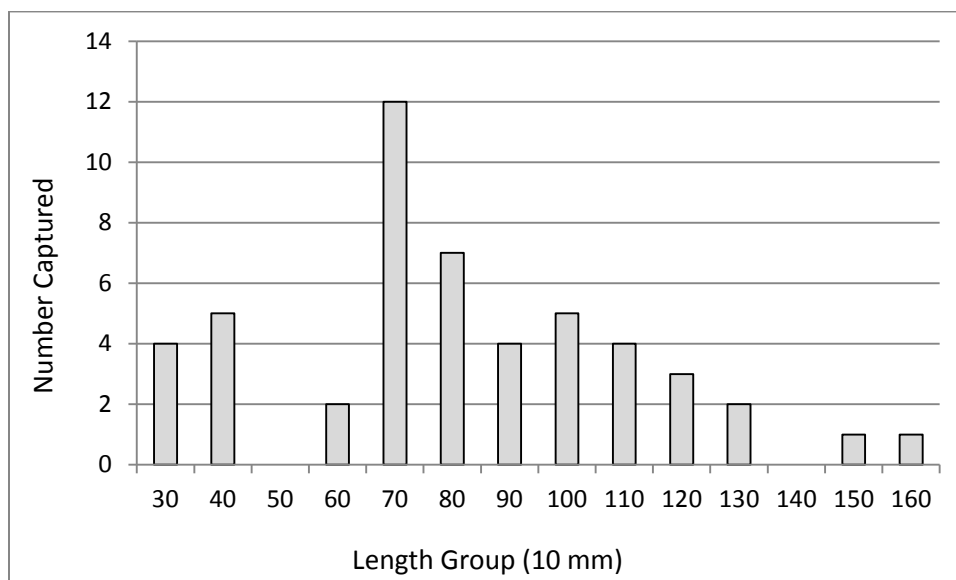


Figure 4. Length-frequency histogram for wct captured in Ford Creek, Glacier National Park, in 2013.

Table 7. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton condition (K) of age-1 and older for wct, blt, and sculpin (scp) captured electrofishing in Ford Cr., GNP, 2013. Length range represents all fish captured.

Species	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT	96.2(24.3)(41)	66-163	9.8(8.3)(41)	1.1
BLT	185(-)(1)	185	59(-)(1)	0.93
SCP	62.5(17.8)(38)	18-89	3.8(2.9)(32)	1.58

Within the 121.1m long stream reach sampled in 2013 the estimated total abundance of age-1 and older wct was 49 (Table 8). The density estimate for these same fish was 7.7 wct/100m² (Table 7). First pass CPUE for age-1 and older wct was estimated at 48.7 fish/hr and 5.3 fish/100m² (Table 7).

In 2013 we estimated the total abundance of age-1 and older bull trout to be 1 (Table 8). First pass CPUE for age-1 and older blt was estimated at 1.7 fish/hr and 0.19 fish/100m² (Table 8). Average Fulton Condition Factor (K) was also estimated at 0.93 for age-1 and older blt (n = 1) (Table 7).

Table 8. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct and bkt captured in Ford Creek, GNP, 2013.

Species	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE ₁ (fish/100m ²)	CPUE ₁ (fish/hr)
WCT	49 (34-64)	7.7	5.3	48.7
BLT	1 (-)	0.19	0.19	1.7

During snorkeling surveys Read et al. (1982) estimated a density of 9.8 age-1 and older wct/100 m² in the lower reaches of Ford Creek. Using similar methods, we estimated a density of wct > 45 mm to be 0.8 wct/100 m².

Using electrofishing depletion population estimation, Muhlfeld et al. (2009a) estimated a density of 8 wct >75 mm/100m² 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 reaches we sampled. The electrofishing density estimates are comparable between years, but it should be noted that we did not precisely duplicate the sampling site location or sample length of Muhlfeld et al. (2009a).

Cobble was estimated to be the dominant substrate type followed by gravel in all sampling reaches. Water temperature at the time of samplings was 11 °C, conductivity was measured at 85.5µS, along with pH of 8.2. Average width of the creek was estimated to be 4.4. Muhlfeld et al. (2009a) reported a maximum summer water temperature of 15.9 °C. Read et al. (1982) measured a conductivity of 105µS and a stream gradient of 3.4%. We employed 700 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width.

McGee Creek

McGee 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. (2009a) documented hybridization between rbt and wct in Dutch Creek, a large tributary to Camas Creek. Genetic samples collected from lower McGee Creek in 2008 indicated hybridization was occurring (MFWP, unpublished data). The culvert under the Camas Road may preclude upstream passage to the headwater reaches of McGee Creek due to its length and drop out of the pipe, but this remains uncertain.

Within the 134m long stream reach sampled in 2013 the estimated total abundance of age-1 and older wct was 14. The density estimate for these same fish was 3.4 wct/100m² (Table 9). First pass CPUE for age-1 and older wct was estimated at 27.6 fish/hr and 2.7 fish/100m² (Table 9). Average length of age-1 and older wct was 119.9 mm (Table 10; Figure 5). Average Fulton Condition Factor (K) was also estimated at 1.3 for age-1 and older wct (SD = 38.6) (n = 14) (Table 10).

Table 9. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured in McGee Creek, GNP, 2013.

Species	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE ₁ (fish/100m ²)	CPUE ₁ (fish/hr)
WCT	14 (12-16)	3.4	2.7	27.6

Table 10. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton condition (K) of age-1 and older for wct captured in McGee Creek, GNP, 2013. Length range represents all fish captured.

Species	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT	119.9(38.6)(14)	69-199	22.14(6.09)(14)	1.3

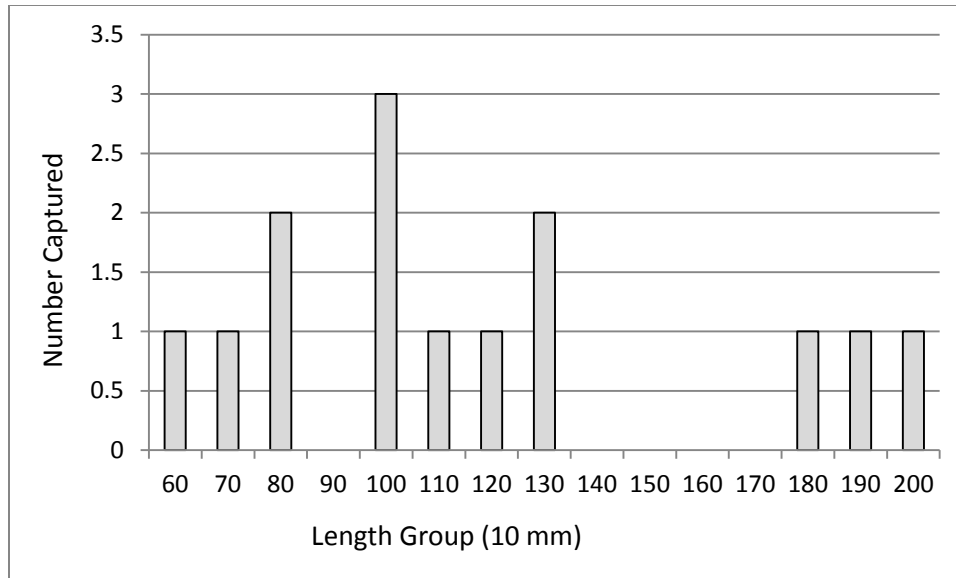


Figure 5. Length-frequency histogram for wct captured in McGee Creek, Glacier National Park, in 2013.

We snorkeled the bottom end of the electrofishing reach (60m) and did not observe any fish. Habitat in McGee Creek is complex with numerous and continual log jams, which made snorkeling observation difficult. Additionally, we only snorkeled a portion of the electrofishing section which may have influenced the opportunity to observe fish in the stream.

Cobble was estimated to be the dominant substrate type within the reach followed by gravel. Water temperature was measure to be 7.5 °C and conductivity was measured at 106.8 μ S. We employed 670 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width. The presence of tailed frogs was not noted in 2013. Average wetted with measured 3.0 m over the 134m reach.

Sage Creek

Sage Creek is a third order tributary to the N. Fk. Flathead River. It is the northern most creek on the west side of the divide and flows from Canada into the park prior to entering the N. Fk. Flathead River near the Canadian border. We are unaware of any wct genetics data for Sage Creek.

We snorkeled Sage Creek on 8/19/2013 and observed only one >150 mm wct over the entire 250 m reach. We started snorkeling at the trail crossing which was also the bottom of the historical snorkeling reach for Read et al. (1982). Read et al. estimated wct densities to be 11.1 wct/100m², while we estimated it to be 0.03 wct/100 m². We did not electrofish the creek after snorkeling. Read et al. (1982) documented wct, mwf, and sculpin throughout the reach. Sage Creeks physical habitat is ideal for snorkeling with lots of pools and a consistent depth allowing continuous submersion.

The dominate substrate was cobble followed by gravel as the subdominant. Average wetted with measured 16.3 m. Water temperature was measured to be 13 °C, conductivity of 165.3 μ S, and the pH was 8.3.

Un-named tributary of Camas Creek (No-Name Creek)

Approximately two kilometers east of the Camas Creek entrance on the Camas Road, an unnamed second order tributary to Camas Creek (hereafter referred to as No-Name Creek) crosses under the road flowing in a northerly direction. Although we did not obtain detailed measurements to evaluate the culvert it appears that culvert length and drop height to the stream below would prevent upstream fish passage. Genetic samples collected upstream of the culvert in 2011 indicated a wct population with 99.9% genetic purity.

Average length of age-1 and older wct was 109.2 mm (Table 11, Figure 6). Within the 50m long stream reach the estimated total abundance of age-1 and older wct was 21 (Table 12). The density estimate for these same fish was 13.3 wct/100m² (Table 12). First pass CPUE for age-1 and older wct was estimated at 72.5 fish/hr and 8.2 fish/100m² (Table 12). Average Fulton Condition Factor (K) was also estimated at 1.5 for age-1 and older wct (SD = 10.3) (n = 21) (Table 11).

Table 11. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older wct captured in No Name Creek, GNP, 2013. Length range represents all fish captured.

Species	Year	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT	2013	109.2(10.3)(21)	51-221	19.9(5.9)(21)	1.5

Table 12. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured in No-name Creek, GNP, 2013.

Species	Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE ₁ (fish/100m2)	CPUE ₁ (fish/hr)
WCT	2013	21 (19-23)	13.3	8.2	72.5

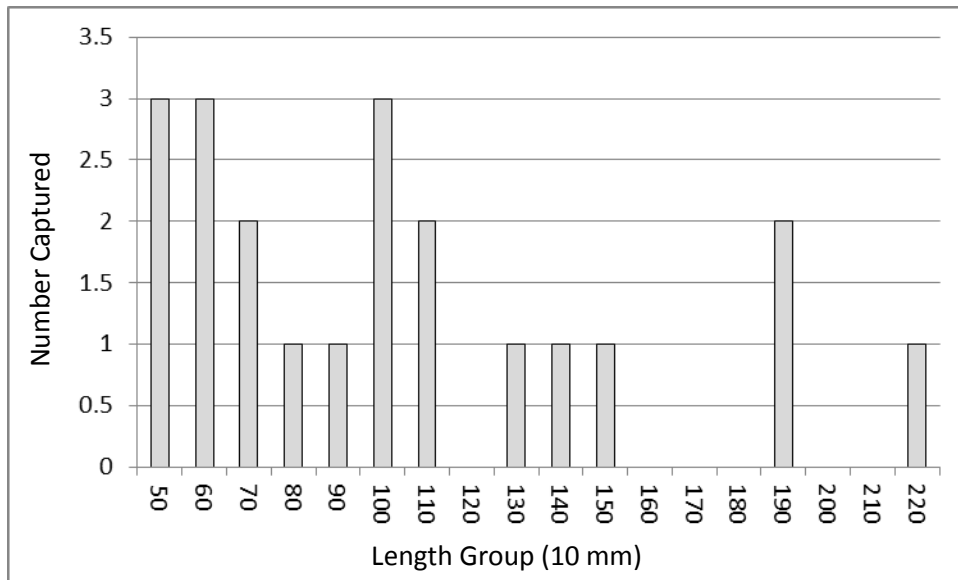


Figure 6. Length-frequency histogram for wct captured in No-name Creek, Glacier National Park, in 2013.

The dominant substrate was determined to be cobble and a subdominant gravel substrate. No Name Creek had water temperature and conductivity was 8°C and 136 μ S, respectively. Our electrofishing settings were 700 volts at 30 Hertz on a 9% duty cycle and a 3 millisecond pulse width. The sampling reach length was 50m while average wetted width was 3.2m. Tailed frogs were noted as abundant. We did not snorkel this reach due to small stream size and habitat complexity.

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 6 km south of the Canadian border. Westslope cutthroat trout 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, mountain whitefish, and sculpin in Starvation Creek (Read et al. 1982, Downs et al. 2011).

We snorkeled Starvation Creek on 8/13/2013. With an average width of 9.8 m one technician snorkeled the creek over the entire 200.3 m reach. The reach starting point was approximately the same as the historic start of Read et al. (1982). One class I, 4 class II, and 1 class III wct were observed (Table 13). Furthermore, three sculpin were observed. Read et al. (1982) observed blt, wct, mwf, and scp. They also noted that spawning bull trout have been documented. Our observed snorkel density of wct was 0.3 wct/100 m², while Read et al. (1982) observed a higher snorkel density of 2.0 wct/100 m².

Table 13. Size class, species, and count of all individuals seen in Starvation Creek in 2013.

<u>Species</u>	<u>Diver 1</u>
WCT <45 mm	1
WCT 46-149 mm	4
WCT >150 mm	1

Starvation Creek was determined to have a dominant substrate of cobble and a subdominant gravel substrate. The sampling reach length was approximately 200.3 m while average wetted width was 9.8 m. Water temperature and conductivity for 2013 was 13°C and 137 μ S, respectively. No amphibians were observed. We did not electrofish Starvation Cr.

We also installed a thermograph in Starvation Creek located near the confluence with the North Fork Flathead River, about 100m upstream of the trail crossing. Stream temperature peaked at 17.9 °C on July 26, 2013 (Figure 7).

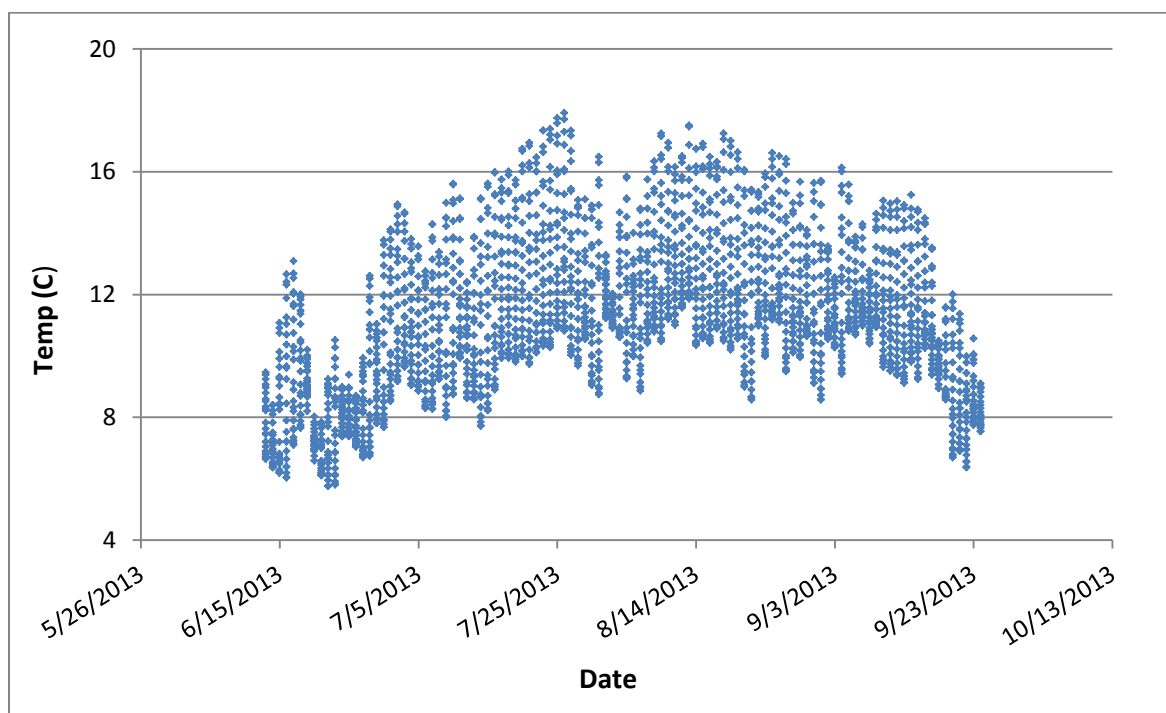


Figure 7. Stream temperature in Starvation Creek, Glacier National Park, Montana.

Middle Fork Flathead River Drainage

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 wct upstream of the culvert from downstream fish, protecting the genetic integrity of the upstream fish (Downs et al. 2011). Westslope cutthroat trout hybridization has not been assessed downstream of the culvert in Fern Creek. The culvert is approximately 60' long, with a relatively flat slope. When measured by Downs et al. (2011), the drop out of the culvert to the water surface was approximately two feet. 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.

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. Based on samples collected from upper Fern Creek in 2008, Fern Creek upstream of the Camas Road is reported to contain genetically pure wct (C. Muhlfeld, USGS, personal communication). We collected additional genetic samples from Fern Creek wct for future analysis.

Only the bottom 37 m of the historic reach was surveyed with one pass. Length ranged from 84 to 176 mm (Figure 8). Average length of age-1 and older wct was 119.2 mm (Table 14). Average Fulton Condition Factor (K) was also estimated at 2.2 for age-1 and older (Table 14). We captured 13 wct resulting in a density of 89.5 wct/100 m², a CPUE (fish/100m²) of 8.6, and a CPUE (fish/hr) of 8.6 (Table 15). Length ranged from 84 to 176 mm (Figure 8). Average length of age-1 and older wct was 119.2 mm (Table 14). Average Fulton Condition Factor (K) was also estimated at 2.2 for age-1 and older (Table 14).

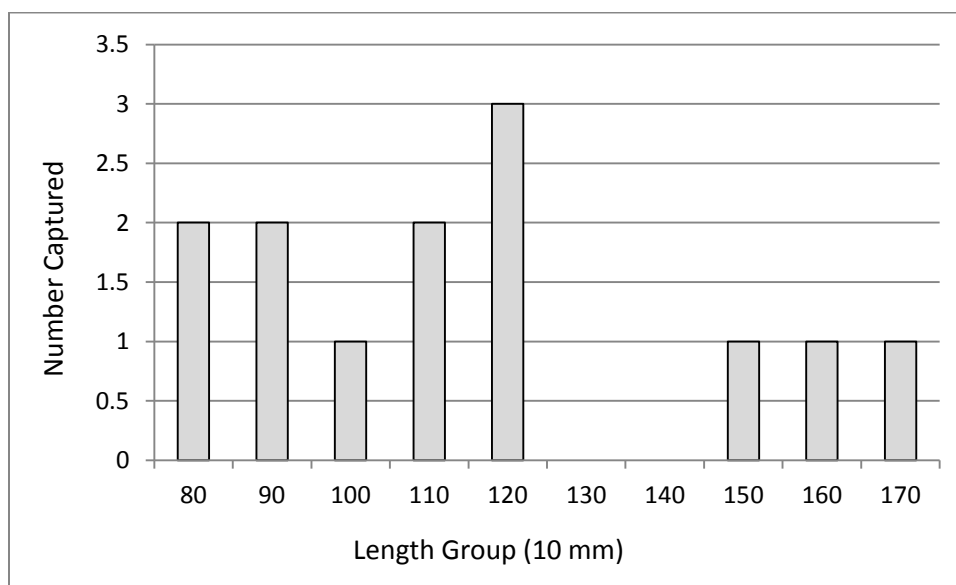


Figure 8. Length-frequency histogram for wct captured in Fern Creek, Glacier National Park, in 2013

Table 14. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older wct captured in Fern Creek, GNP, 2013. Length range (mm) represents all fish captured.

Species	Year	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Weight Range	Fulton Condition Factor (K)
WCT	2013	119.2 (28.9)(13)	84-176	25.8 (30.5)(13)	4-97	2.16

Table 15. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older blt and wct captured in Fern Creek, GNP, 2013

Species	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
WCT	N/A	89.5	8.6	8.6

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. The sampling reach length was approximately 37 m while average wetted width was 4.1 m. Water temperature and conductivity for 2013 was 12°C and 59 µS, respectively. Our electrofishing settings were 700 volts at 30 Hertz on a 9% duty cycle and a 3 millisecond pulse width. Both adult and juvenile tailed frogs were abundant.

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 it. 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. Downfall within the sampling reach is significant, approximately 107 logs over 100 meters of stream length. Based on samples collected from upper Fish Creek in 2008, Fish Creek both upstream and downstream of the Camas Road is reported to contain 99% genetically pure wct (C. Muhlfeld, USGS, personal communication).

We snorkeled Fish Creek using a single snorkeler and made a pass lasting 30 minutes. No fish were observed. To our knowledge Fish Creek above the Camas road has not been snorkeled in the past. After snorkeling, one pass was made with the electrofisher. Only three fish were captured (Table 16). We did not attempt a second pass due to the amount of wood in and across the channel.

Table 16. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older wct captured in Fish Creek, GNP, 2013. Length range represents all fish captured.

Species	Year	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT	2013	122.3 (52.9)(3)	81-182	25 (31.3)(3)	1.36

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Water temperature, conductivity, and pH for 2013 were 11°C, 68.7 µS, and 8.2, respectively. Due to instrument malfunction, dissolved oxygen was not available. The sampling reach length was approximately 112 m, while average wetted width was 2.5 m. Our electrofishing settings were 670

volts at 65 Hertz on a 9% duty cycle and a 3 millisecond pulse width. Both adult and juvenile tailed frogs were present and common within the section of Fish Creek that was sampled.

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 km² of land area (Weaver et al. 1983). Genetic samples collected in 2010 indicated the lower reaches of Muir Creek contained 99% genetically pure westslope cutthroat trout while upper reaches contained 100% genetically pure wct (C. Muhlfeld, USGS, personal communication). The only species of fish documented in Muir Creek by previous researchers have been westslope cutthroat trout and bull trout. Juvenile bull trout have also been captured in Muir Creek (Downs et al, 2011).

Sampling in 2012 resulted in the capture of only two wct for the entire 80m stream reach. This was surprising as wct were abundant in the sampling reach in previous years. As recently as 2011, 30 wct and 2 blt were captured in the sample reach. Population estimates across earlier sampling years ranged from 30-51 for wct and from 2-6 for blt.

We did not electrofish Muir Creek in 2013 but rather snorkeled 196.9 m stretch beginning at the bottom end of our electrofishing reach. Snorkeler #1 sampled the entire electrofishing section (97.9m), only seeing 1 wct >150 mm (Table 17). Snorkeler #2 began at the top of the electrofishing section and continued for 98.7 m to the base of a small water fall measuring approximately 2 m tall. Snorkeler #2 counted 5 wct between 50-149mm, 10 wct >150mm and 1 blt >150mm (Table 17).

Table 17. Size class, species, and count of all individuals of other species seen in Muir Creek in 2013.

<u>Species</u>	<u>Diver 1</u>	<u>Diver 2</u>
WCT <45 mm	0	0
WCT 46-149 mm	0	5
WCT >150 mm	1	10
BLT <45 mm	0	0
BLT 46-149 mm	0	0
BLT >150 mm	0	1

We do not know why densities of wct in the lowermost reaches of Muir Creek have dropped off so drastically. It is possible environmental factor(s), predation, or illegal fishing may have contributed.

Within the lower reaches of Muir Creek our sampling took place approximately 1 km downstream of the lower-most snorkel section conducted by Weaver et al. (1983). Using snorkeling, Weaver et al. (1983) estimated densities of 11.6 age-1 and older wct/100m² in their lower-most sampling reach. Although 2010 electrofishing sampling results are similar (13.6 wct/100m²), abundance declined thereafter. In 2013, we used only snorkeling to sample the stream and estimated a density of 1.99 wct/100 m² over the 196.6 m long snorkel section. The surveys conducted by Weaver et al. (1983) 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. This data set suggests that Muir Creek remains an important conservation area for wct within GNP but one where the population dynamics are not fully understood.

Cobble was estimated to be the dominant substrate type followed by gravel in the reach (196.6m) where the average width measured 4.1m. Water temperature was measure to be 8.5 °C, conductivity 101.4 µS, and pH 8.43. No amphibians were observed in Muir Creek in 2013.

Upper McDonald Creek (*upstream of the confluence with Mineral Cr*)

Upper McDonald Creek above the confluence with Mineral Creek is a third order tributary to the M. Fk. Flathead River. It flows southeast primarily, until it joins Mineral Creek at which time it flows southwest to Lake McDonald. In 2013 we snorkeled from the confluence with Mineral Cr upstream for 329m. Sections below Lake McDonald have been surveyed previously by Weaver et al (1983), however, we do not believe that others have snorkeled above the Mineral Creek confluence. McDonald Creek upstream of McDonald Falls is known to contain genetically pure wct (C. Muhlfeld, USGS, personal communication).

Only wct were observed in this reach of McDonald Creek. Fish in all size groups were observed (Table 18). A density of 1.44 wct/100 m² was observed in this 329 m stretch. This is an ideal snorkel section with very little woody debris, reasonable stream gradient, and desirable pool/run ratio. This section should be considered for long term snorkel monitoring.

Table 18. Size class, species, and count of all individuals seen in Upper McDonald Creek in 2013.

<u>Species</u>	<u>Diver 1</u>
WCT <45 mm	11
WCT 46-149 mm	31
WCT >150 mm	23

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. The average width was 11.43m. Water temperature was measure at 13 °C, conductivity at 90.3 µS, and pH at 8.16. No amphibians were observed in Upper McDonald Creek.

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 from Boulder Lake, near Siyeh Pass along the Continental Divide. It flows northeast out of GNP 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 can be characterized as pristine. Mogen and Kaeding (2004) reported

capturing wct hybrids, but also noted that wct captured in one of Boulder Creek's upper tributaries were 95% genetically pure. Genetic samples collected in 2009 from the study section indicated a 93% pure wct population.

Boulder Creek supports a run of migratory bull trout from the St. Mary River/Lower St. Mary Lake (Mogen and Kaeding 2004). Since redd counts began in 1997 there has been an average of 36 redds observed annually. Using documented spawner: redd ratios of approximately 3:1 (Fraleigh 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 and westslope cutthroat trout, brook trout are also present at low densities.

Bull trout, westslope cutthroat, and brook trout were captured in Boulder Creek in 2013. Fifty-one bull trout, 13 wct, and one brook trout were captured. Five additional blt approximately 470, 550, 500, 325, 400mm were netted from within the section and immediately released below the downstream block net, we did not collect weights for these individuals (Figure 9).

We were unable to estimate the abundance of bull trout in Boulder Creek in 2013 due to a non-descending catch pattern. First pass CPUE for age-1 and older blt was estimated at 30.4 fish/hr and 1.8 fish/100m² (Table 20). Average length of age-1 and older blt was 161.2 mm (Table 19, Figure 9). Average Fulton Condition Factor (K) was also estimated at 1.11 for age-1 and older blt (SD = 111.6) (n = 43) (Table 19).

Average length of age-1 and older wct was 167.5 mm (Figure 10). Average Fulton Condition Factor (K) was also estimated at 1.33 for age-1 and older wct/hybrids (SD = 38.8) (n = 13) (Table 19). In 2013 the estimated total abundance of age-1 and older wct was 13. The density estimate for these same fish was 1.2 wct/100m² (Table 20). First pass CPUE for age-1 and older wct was estimated at 15.2 fish/hr and 0.9 fish/100m² (Table 20).

Table 19. Mean length (TL; mm), weight (g), standard deviation (SD), and sample size (n) of age-1 and older blt and wct captured in Boulder Creek, GNP, 2013. Length range represents all fish captured for all species.

Species/year	Mean length (SD) (n)	Length range (all individuals of species)	Mean weight (SD) (n)	Fulton Condition Factor (K)
BLT	161.2 (111.6)(43)	95-550	20.7 (11.69)(38)	1.11
WCT	167.5 (38.8)(13)	123-245	62.5 (46.6)(13)	1.33

Table 20. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older blt and wct captured in Boulder Creek, GNP, 2013 (*non-descending removal pattern)

Species	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
BLT	N/A*	N/A*	1.8	30.4
WCT	13 (11-15)	1.18	.9	15.2
BKT	1	.09	.091	1.5

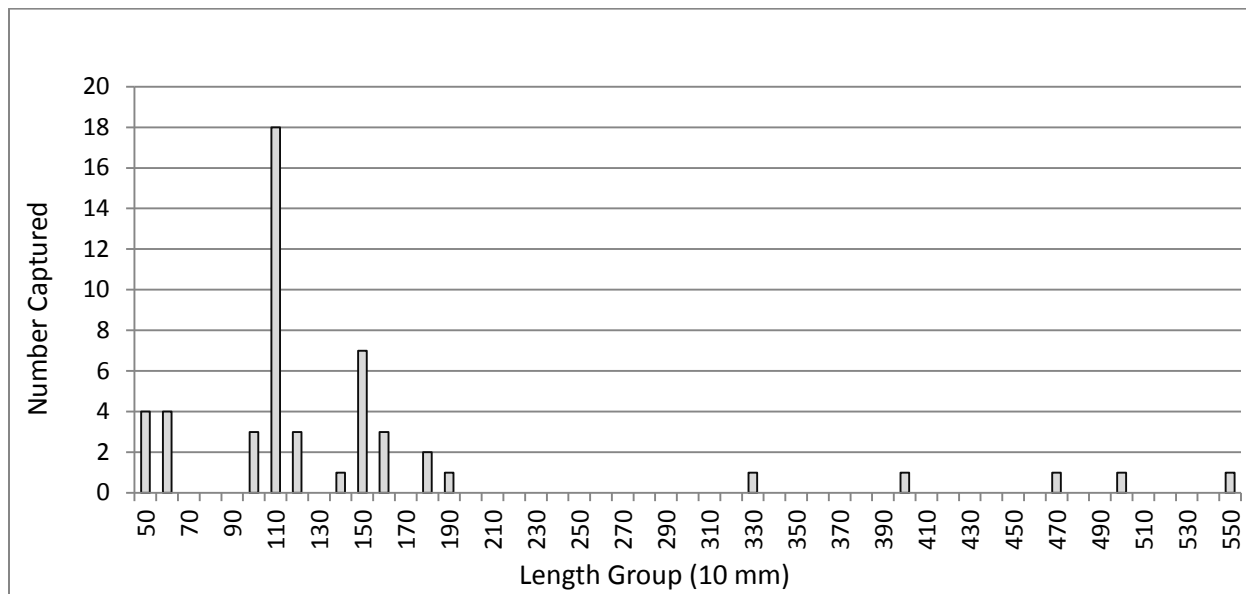


Figure 9. Length-frequency histogram for blt captured in Boulder Creek, Glacier National Park, in 2013.

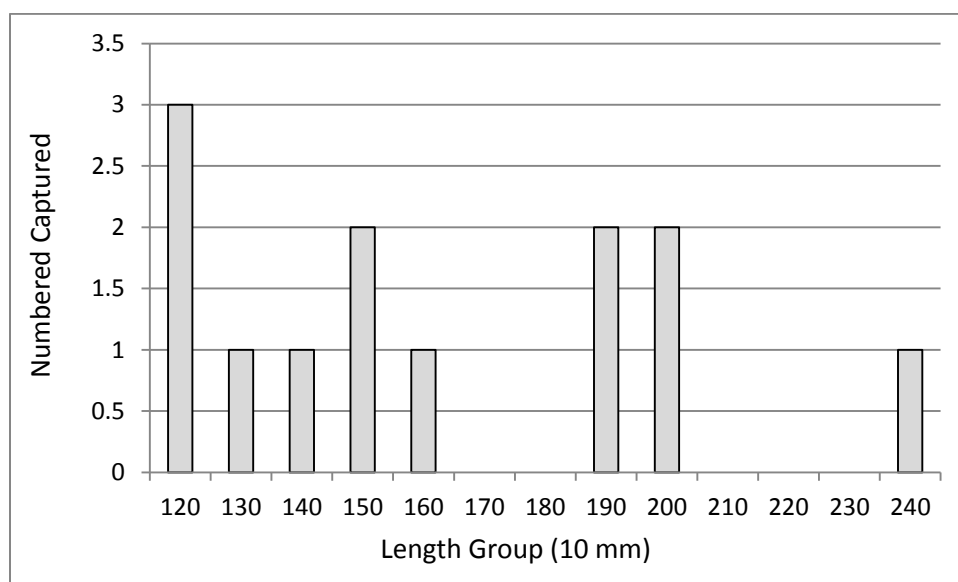


Figure 10. Length-frequency histogram for wct captured in Boulder Creek, Glacier National Park, in 2013.

Cobble was estimated to be the dominant substrate type followed by boulder in the study reach. Water temperature at the time of sampling was 10°C. Conductivity was measured at 141 μ S. Average wetted width was 8.4 m for the study reach. One small spotted frog was observed along the

shore though no other amphibians were observed. We employed 800 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Boulder Creek.

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/St. Mary Reservoir (Mogen and Kaeding 2004). Redd counts have been conducted annually since 2011). In addition to blt, wct and mwf are present in Lee Creek (Mogen and Kaeding 2004).

Both bull trout and wct were captured in Lee Creek 2013. Two age-0 blt but no age-0 wct (blt<60mm, wct<45mm) were captured in 2013 (Figures 11 and 12). The average length of age-1 and older blt was 112.6 mm in 2013 (Table 21).

Within the 120.5m long sample reach we estimated the total abundance of age-1 and older bull trout to be 21 (Table 22). The density estimate for these same fish was 4.05 blt/100m² (Table 22). First pass CPUE for age-1 and older blt was estimated at 33.2 fish/hr and 3.04 fish/100m² (Table 22). Average Fulton Condition Factor (K) was also estimated at 1.2 for age-1 and older blt (Table 21).

In 2013 the estimated total abundance of age-1 and older wct was 30. The density estimate for these same fish was 4.05 wct/100m² (Table 22). First pass CPUE for age-1 and older wct/hybrids was estimated at 50.9 fish/hr and 4.7 fish/100m² (Table 22). Average length of age-1 and older wct was 117.3 mm (SD = 36.8) (n = 29) (Table 21). Average Fulton Condition Factor (K) was also estimated at 1.5 for age-1 and older wct (Table 21).

Table 21. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older bull trout and wct captured on Lee Creek, GNP, 2013. Length range represents all fish captured.

Species	Mean Length (mm) (SD) (n)	Length Range (mm) (all individuals captured)	Mean Weight (g) (SD) (n)	Fulton Condition Factor (K)
BLT	112.6 (30.1)(20)	68-211	17.4 (16.15)(20)	1.2
WCT	117.3 (36.8)(29)	61-219	24.3 (27.54)(29)	1.5

Table 22. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct and bkt captured in Lee Creek, GNP, 2013.

Species	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE ₁ (fish/100m ²)	CPUE ₁ (fish/hr)
BLT	21 (17-25)	4.05	3.04	33.2
WCT	30 (26-34)	5.9	4.7	50.9

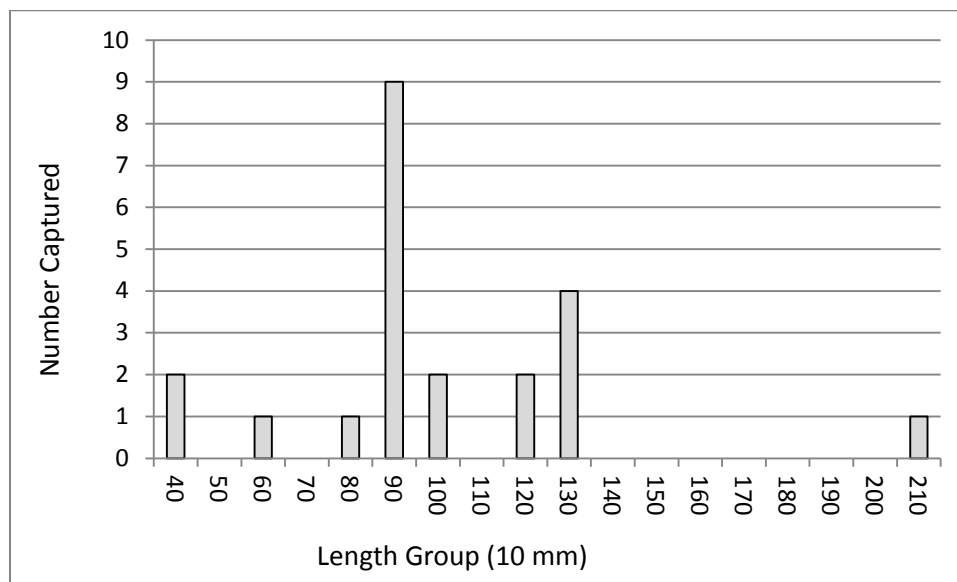


Figure 11. Length-frequency histogram for blt captured in Lee Creek, Glacier National Park, in 2013.

Cobble was estimated to be the dominant substrate type followed by gravel in the study reach. We employed 700 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Lee Creek. No amphibians were observed. A thermograph was installed approximately 100m downstream of the Chief Mountain Highway crossing, and water temperatures reached a high of 14.6⁰C on July 26th, 2013 (Figure 13).

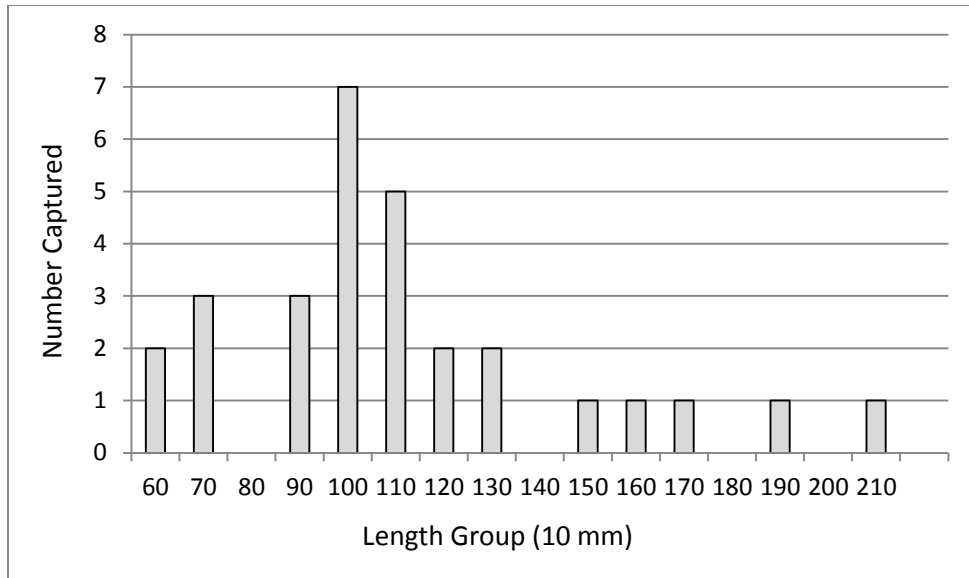


Figure 12. Length-frequency histogram for wct captured in Lee Creek, Glacier National Park, in 2013.

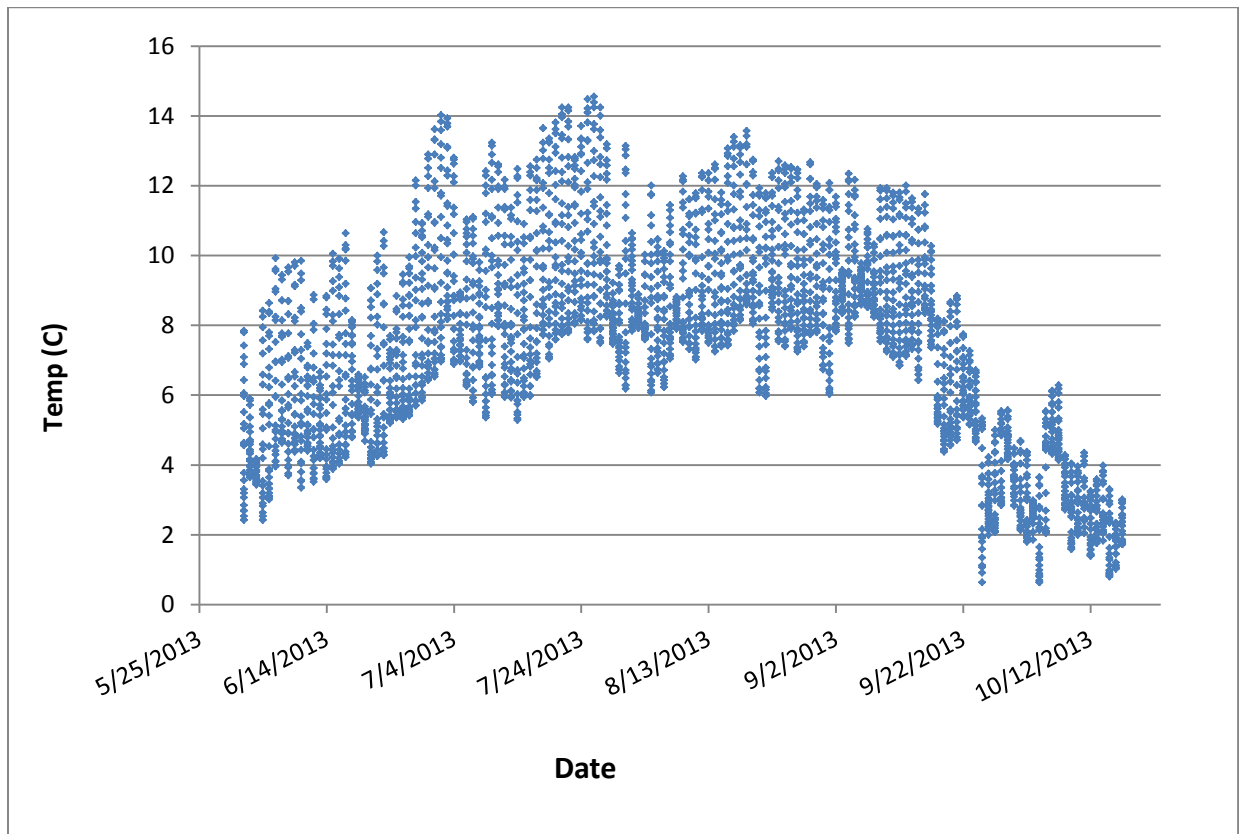


Figure 13. Water temperatures recorded in 2013 in Lee Creek, Glacier National Park, Montana.

Wild Creek

Wild Creek is a third 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. Genetic samples collected in 2009 indicated a genetic purity of over 99%.

Within the 85 m long study reach, the estimated total abundance of age-1 and older wct was 57 (Table 24). The density estimate for these same fish was 12.5 wct/100m² (Table 24). First pass CPUE for age-1 and older wct/hybrids was estimated at 76.6 fish/hr and 9.6 fish/100m² (Table 24). Average length of age-1 and older wct/hybrids was 102.7 mm (SD = 45.2) (n = 51) (Table 23, Figure 14). Average Fulton Condition Factor (K) was also estimated at 1.2 for age-1 and older wct/hybrids (Table 23).

In 2013 we estimated the total abundance of age-1 and older brook trout to be 3 (Table 24). The density estimate for these same fish was 0.7 bkt/100m² (Table 24). First pass CPUE for age-1 and older bkt was estimated at 3.9 fish/hr and 0.5 fish/100m² (Table 24). Average length of age-1 and older bkt was 128 mm (SD = 10.8) (n = 3) (Table 23). Average Fulton Condition Factor (K) was also estimated at 1.08 for age-1 and older bkt (Table 23).

Table 23. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older wct and bkt captured on Wild Creek, GNP 2013. Length range represents all fish captured.

Species	Mean Length (mm) (SD) (n)	Length Range (mm) (all individuals captured)	Mean Weight (g) (SD) (n)	Fulton Condition Factor (K)
WCT	102.7 (45.2)(51)	50-360	13.2 (13.1)(51)	1.2
BKT	128(10.8)(3)	119-140	22.7(10.8)(3)	1.08

Table 24. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct and bkt captured in Wild Creek, GNP, 2013.

Species	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE ₁ (fish/100m ²)	CPUE ₁ (fish/hr)
WCT	57 (48-66)	12.5	9.6	76.6
BKT	3 (0-6)	0.7	0.5	3.9

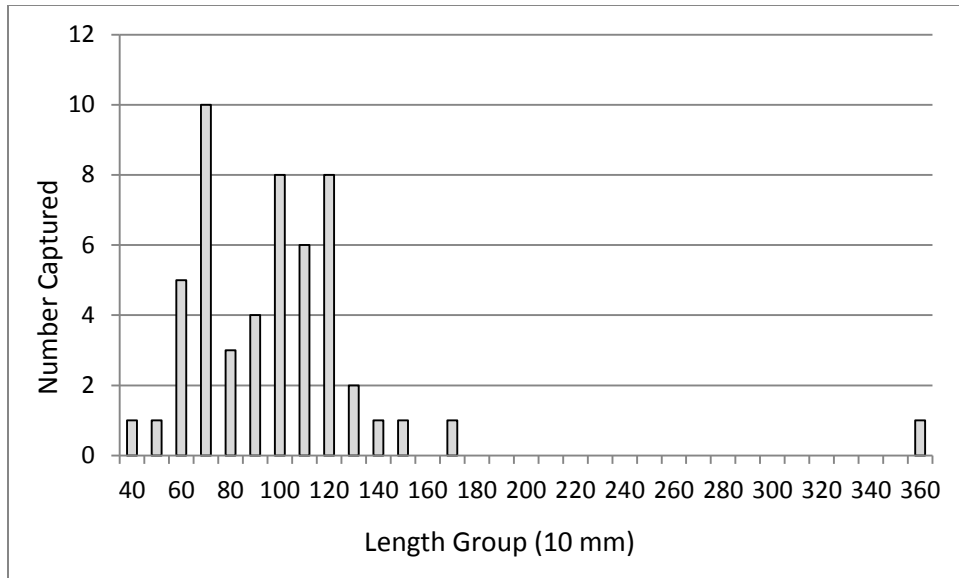


Figure 14. Length-frequency histogram for wct captured in Wild Creek, Glacier National Park, in 2013.

Cobble and boulder were the dominant and subdominant substrates, respectively. Water temperature was 12.5 °C, conductivity was 147.2 and pH of 6.74. We employed 700 volts the first pass and 650 the second pass at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Lee Creek. No amphibians were observed.

Other Stream Temperature Monitoring

We also installed a thermograph in Ole Creek (Middle Fork Flathead River drainage). The thermograph on Ole Creek was located approximately 20m upstream of the suspension bridge at the trail crossing on lower Ole Creek. Stream temperature reached a high of 15.2 °C in Ole Creek on July 24, 2013. We found the temperature logger partially out of the water on July 27th, but based on the data, it appears as though the sensor continued to record reasonable stream temperatures. The logger was repositioned further into the middle of the creek.

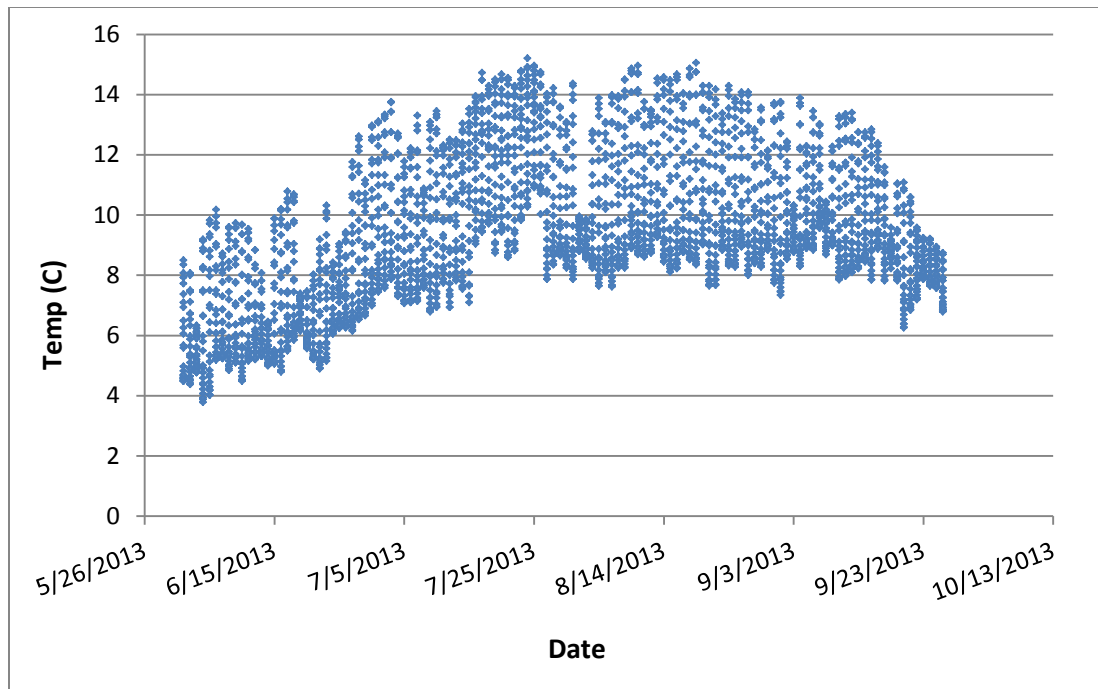


Figure 15. Ole Creek stream temperatures, Glacier National Park, Montana.

ACKNOWLEDGEMENTS

The authors wish to thank NPS employee Katie Rayfield for assistance with field sampling and Mark Biel for reviewing this report.

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Two Medicine Lake Gill Net Sampling

ABSTRACT

We sampled Two Medicine Lake with eight sinking experimental mesh gill nets from July 8-9, 2013 to characterize the fish community, collect baseline information, and establish a standardized fishery sampling dataset. We captured lake trout (60), brook trout (22), and rainbow trout (4). Length-at-capture ranged from 190-767 mm (TL) for lake trout, from 166-344 mm for brook trout, and from 172-471 mm (TL) for rainbow trout. Previous gill net surveys conducted in 1970 in Two Medicine Lake did not capture any lake trout and the NPS has no record of stocking lake trout into Two Medicine Lake or upstream waters.

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INTRODUCTION

Two Medicine Lake is a 424.2 acre lake located in Glacier National Park which feeds into the Missouri River drainage (Figure 1). The lake lies at an elevation of 1573.987 m and has approximately 8.26 km of shoreline. Maximum depth is 51 m. The lake is approximately 3.6 km long, with a maximum width of 652 m. Since the Two Medicine valley is easily accessed by motor vehicle and small motorized boats may be used, Two Medicine Lake is among the more popular fishing destinations in the park. Two Medicine Lake is believed to have been historically fishless due to the presence of Running Eagle Falls (Trick Falls), located approximately 1.5 km downstream of Two Medicine Lake. NPS stocking records show that rainbow trout, cutthroat trout, brook trout, and Arctic grayling were stocked into Two Medicine Lake between 1919 and 1969. Rainbow and brook trout were stocked almost annually during that timeframe. Grayling were stocked into the lake in 1924 and 1926 while cutthroat were stocked in 1939 and 1940. The lake became well known for its brook trout fishery.

Gill net surveys conducted in Two Medicine Lake in 1970 did not capture any Lake trout, yet lake trout now dominate the fishery. Lake trout were documented in Upper Two Medicine Lake in gill net surveys conducted in 2003 (D. Skaar, MFWP, personal communication). We have no stocking records for lake trout in the drainage.

METHODS

We used eight sinking multi-filament experimental mesh gill nets to sample Two Medicine Lake from July 8-9, 2013 (Figure 2 and Table 1). Nets were set between 1830 and 1930 hours and were pulled the following day between 0625 and 0715 hours. 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. This is the same net type/mesh sizes used in the 1970 sampling. Nets were set with the smallest mesh towards shore.

Some fish captured alive were removed from the net, counted in the net catch total, and released. All other fish were measured (TL; mm) and weighed (g). Lake trout were sexed, maturity status was determined, otoliths were removed, and a genetic sample was collected. We also collected subsample of heads from lake trout, brook trout, and a couple rainbow trout for whirling disease testing.

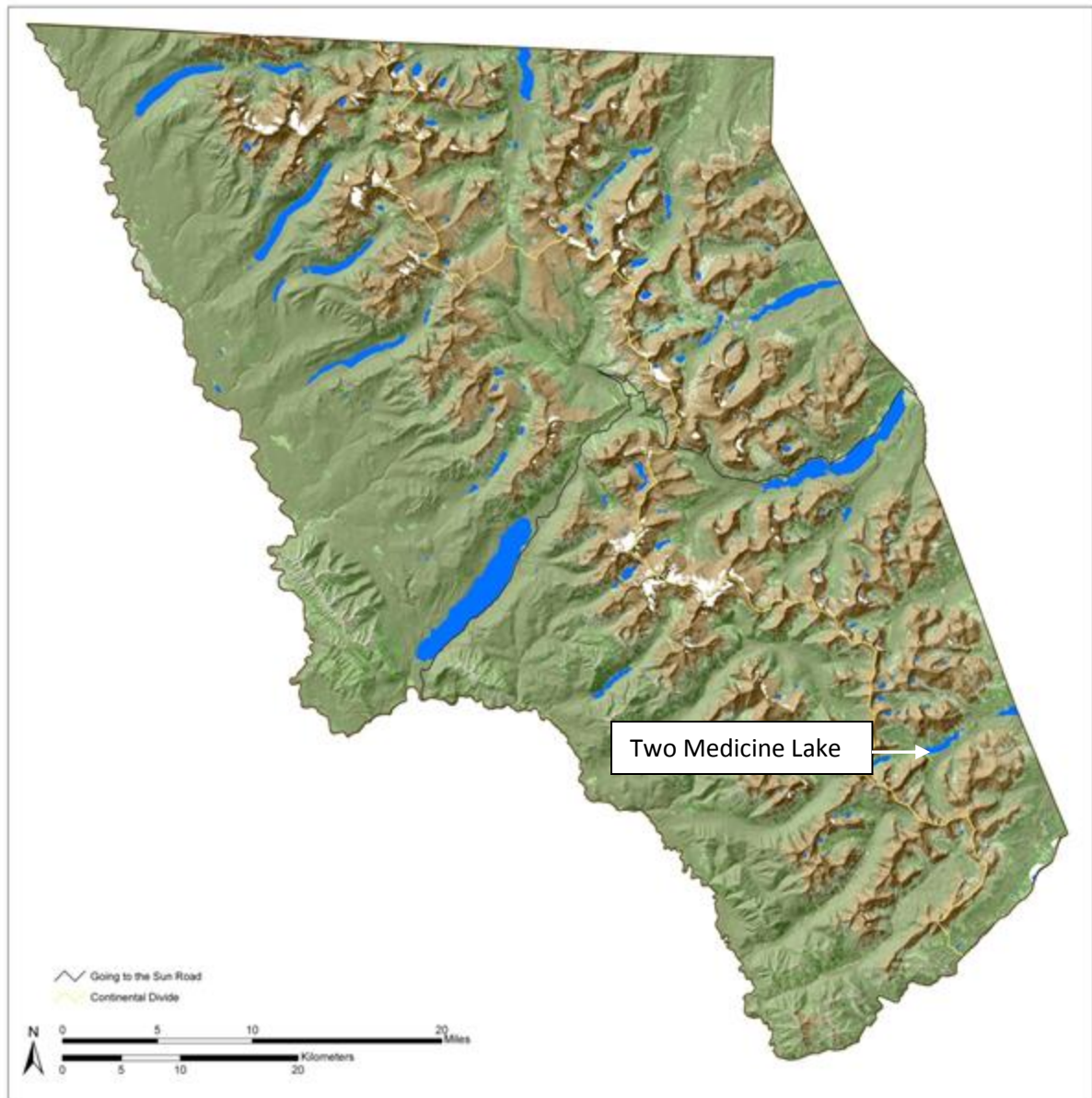


Figure 1. Location of Two Medicine Lake, Missouri River drainage, Glacier National Park.

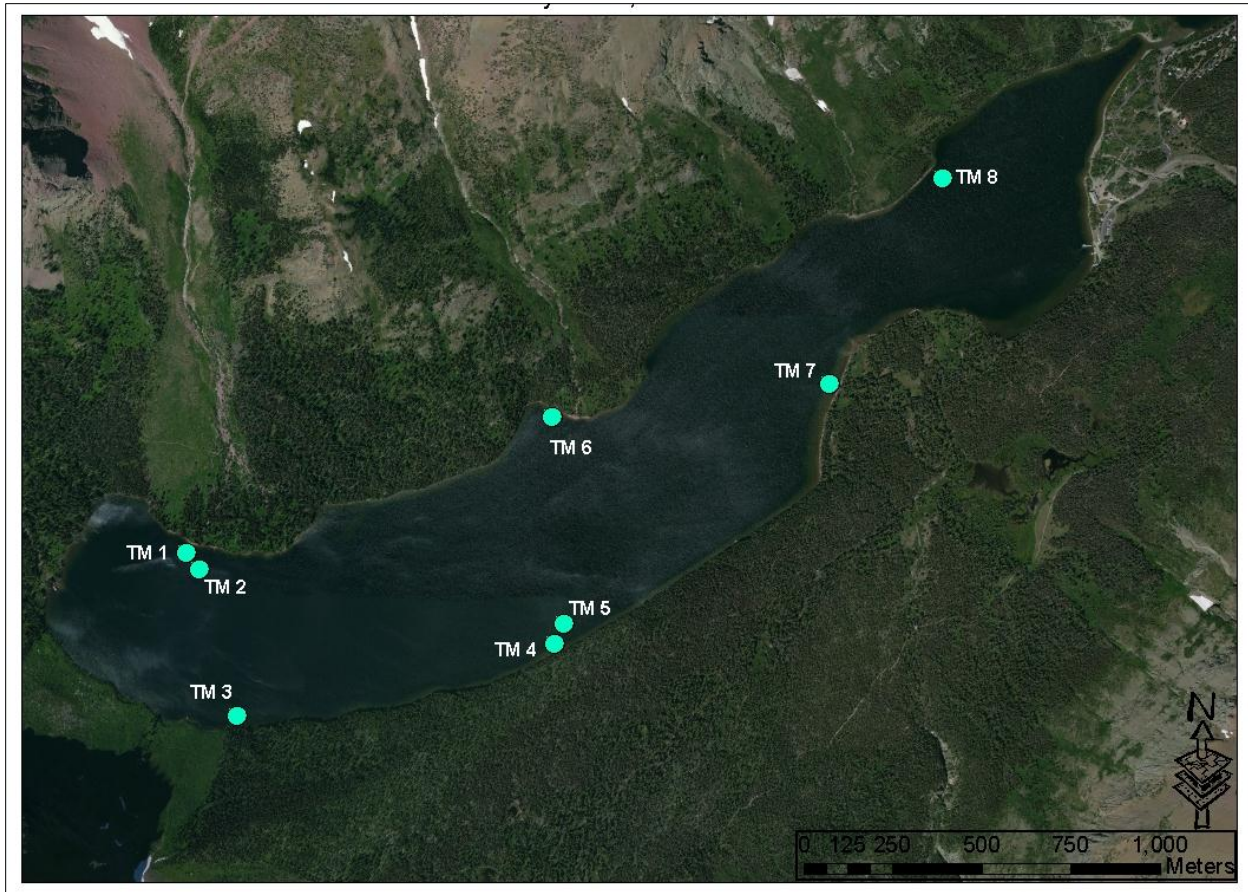


Figure 2. Locations of gill net sets in Two Medicine Lake, July, 2013.

Table 1. UTM's and set parameters of gill net sets in Two Medicine Lake, July, 2013.

Lake	Net #	UTM_X	UTM_Y	Date Set	Time Set	Date Pulled	Time Pulled	Starting Depth (ft)	Ending Depth (ft)
Two Medicine lake	1	322348	5371914	7/8/2013	1830	7/9/2013	700	21	69
Two Medicine lake	2	322383	5371869	7/8/2013	1845	7/9/2013	715	73	83
Two Medicine lake	3	322490	5371459	7/8/2013	1900	7/9/2013	630	7	22
Two Medicine lake	4	323378	5371659	7/8/2013	1915	7/9/2013	630	8	55
Two Medicine lake	5	323404	5371715	7/8/2013	1930	7/9/2013	645	56	79
Two Medicine lake	6	323372	5372294	7/9/2013	1845	7/10/2013	625	5	69
Two Medicine lake	7	324150	5372386	7/9/2013	1915	7/10/2013	630	5	52
Two Medicine lake	8	324466	5372962	7/9/2013	1930	7/10/2013	635	29	109

RESULTS AND DISCUSSION

We captured a total of 86 fish comprising 60 lkt, 22 bkt, and 4 rbt during gill netting efforts. CPUE as fish/net night ranged from 7.5, 2.75, and 0.5 for lkt, bkt, and rbt, respectively (Table 2). While CPUE/Fish ranged from 0.87, 0.32, and 0.06 for lkt, bkt, and rbt, respectively (Table 2). In 1970 the NPS only fished four nets for one night. A total of 141 fish were captured, 123 bkt and 17 rbt. CPUE as fish/net night in 1970 ranged from 31 to 1.51 for bkt and rbt, respectively (Figure 3) while CPUE as fish/hr ranged from 1.512 and 0.21 for bkt and rbt, respectively (Figure 4).

Table 2. Catch composition for Two Medicine Lake, Glacier National Park, 2013.

Species	Count	CPUE Fish/Net Night	CPUE Fish/Hour
LKT	60	7.5	0.87
BKT	22	2.8	0.32
RBT	4	0.5	0.06

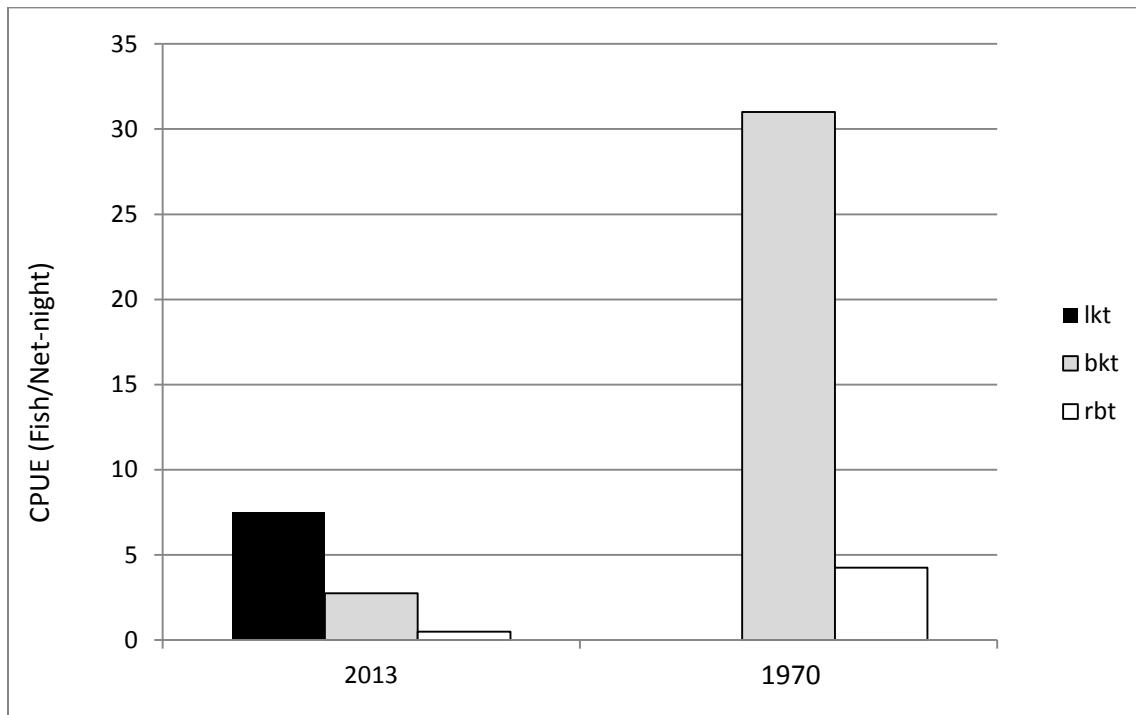


Figure 3. CPUE (Fish/Net-night) comparison from 1970 and 2013 gill netting in Two Medicine Lake, Glacier National Park.

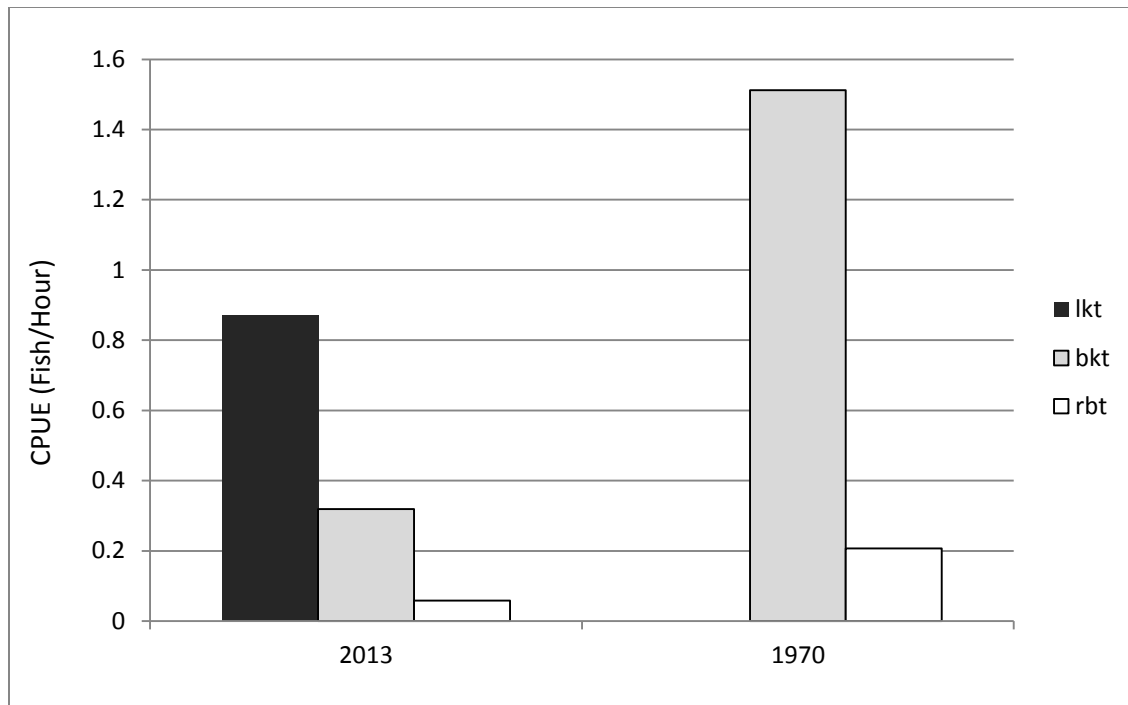


Figure 4. CPUE (Fish/Hour) comparison from 1970 and 2013 gill netting in Two Medicine Lake, Glacier National Park.

Mean length for lake trout captured was 421.8 mm (Table 1) (Figure 5). Length ranged from 172-471 mm for rainbow trout and from 166-344 mm for brook trout (Table 3). The scale malfunctioned so no weights were recorded. Mean length for brook trout was 224 mm (Table 3). Mean length for rainbow trout was 288 mm (Table 3).

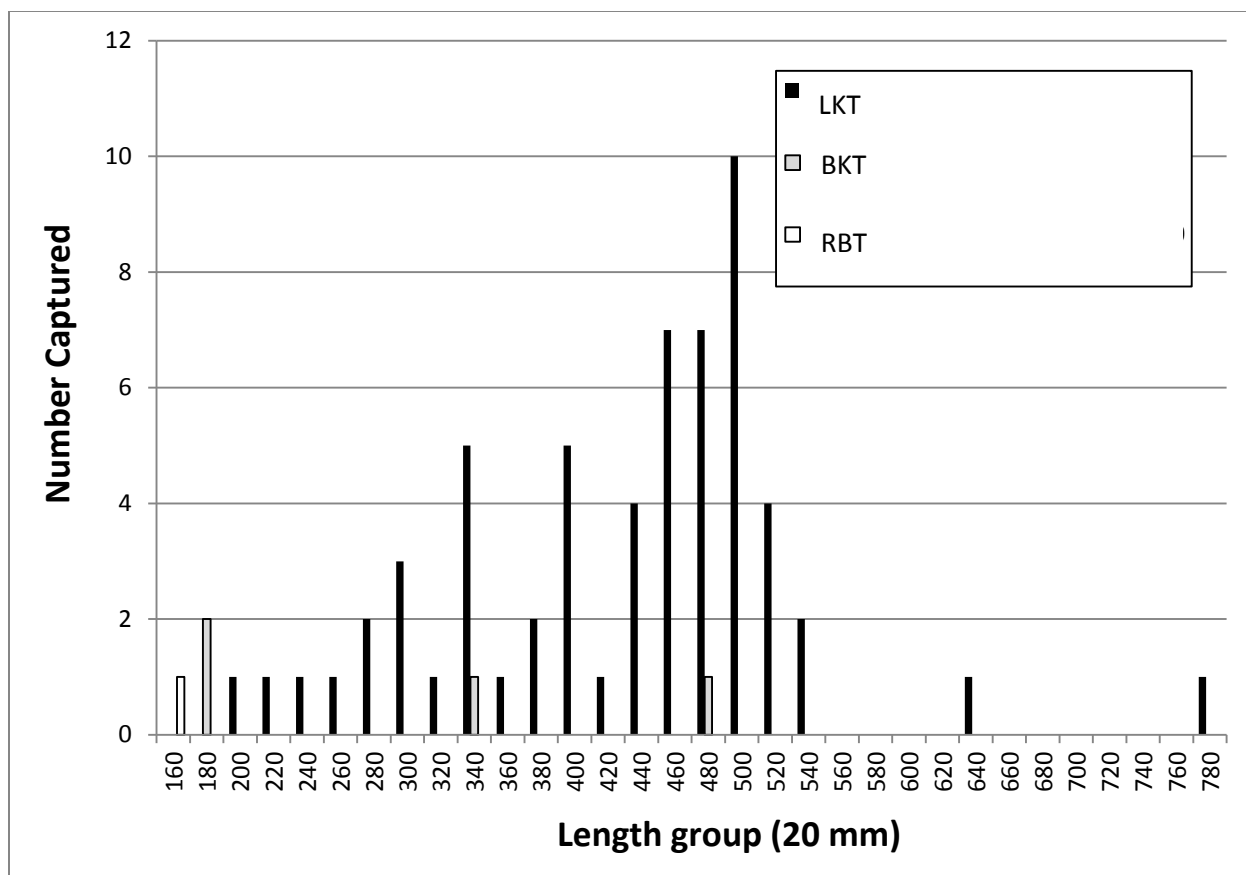


Figure 5. Length frequency for LKT, BKT, and RBT captured in gill nets in Two Medicine Lake in 2013.

Table 3. Mean length (TL; mm), length range, mean weight (g), mean Relative Weight (Wr) standard deviation (SD), and sample size(n) for fish captured in 2013 using gill nets in Two Medicine Lake, Glacier National Park.

Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)	Mean Wr (SD) (n)
LKT	421.8 (102.4)(60)	190-767	N/A	N/A
BKT	224(64.8)(22)	166-344	N/A	N/A
RBT	288(141.8)(4)	172-471	N/A	N/A

Sampling with similar gear in 1970 did not capture any lake trout. Lake trout were captured in Upper Two Medicine lake in gill nets in 2003. These fish ranged in size up to 20". Based on age and growth data from Swan Lake (Cox 2010), a 20" lake trout would likely be 5-7 years old. Lake trout were likely illegally introduced to Two Medicine Lake between 1970 and 1990.

Mean length of both brook and lake trout captured in 1970 (243.4mm and 280.5, respectively) was similar to the mean lengths for these species we observed in 2013. However, catch rates described as both fish captured per net-night and fish captured per-hour were lower for our sampling than that observed for both lake and rainbow trout in 1970. This is not particularly surprising due to the

introduction and establishment of lake trout as a new predator combined with a cessation of stocking of rainbow and brook trout in the system by the late 1960's.

Future surveys should be conducted every 5-10 years to monitor any changes in the fish community resulting from the establishment of lake trout, visitor use, and climate change.

ACKNOWLEDGEMENTS

The authors wish to thank to Katie Rayfield for assistance with gill netting and Mark Biel for reviewing this report.

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Lake Isabel Hoop and Trap Net Sampling

ABSTRACT

Lake Isabel is located in the Middle Fork Drainage of Glacier National Park. It supports isolated populations of westslope cutthroat and bull trout. Isabel Lake is one of the few remaining park lakes with a native fish assemblage that is secure from invasion by non-native fish species. It is protected by several waterfalls along the course of Park Creek. Lake Isabel had been sampled with gill nets in the past, but our goal was to develop and implement an effective non-destructive sampling approach for these unique populations. We used custom-made miniature trap and hoop nets, along with angling to sample the fishery. We captured 65 trout in five trap net sets and eight hoop net sets over the course of two sampling nights. We captured only bull trout (45) and westslope cutthroat (20) in the traps. Bull trout lengths ranged from 158 – 286 mm, while westslope cutthroat trout ranged from 131-302 mm. The trap nets were more effective despite fewer net sets, catching a total of 43 (66%) of the fish. We captured only westslope cutthroat trout using angling (spinning gear).

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INTRODUCTION

Lake Isabel is a 45.2 acre lake located in the Park Creek drainage in Glacier National Park (Figure 1). The lake lies at an elevation of 1742 m and has approximately 1.65 km of shoreline. Maximum depth is 16 m (Meeuwig et al 2008). The lake is approximately 580 m long, with a maximum width of 468 m. The drainage is roadless, and the lake is accessed by trail from the Walton Ranger Station (16 miles) or the Two Medicine Valley via Two Medicine Pass (12 miles). The physical habitat is pristine though fairly recently burned from the Rampage Fire in 2003. Lake Isabel has one backcountry campground, and receives relatively little fishing pressure. However, it is known for providing quality fishing opportunities for westslope cutthroat.

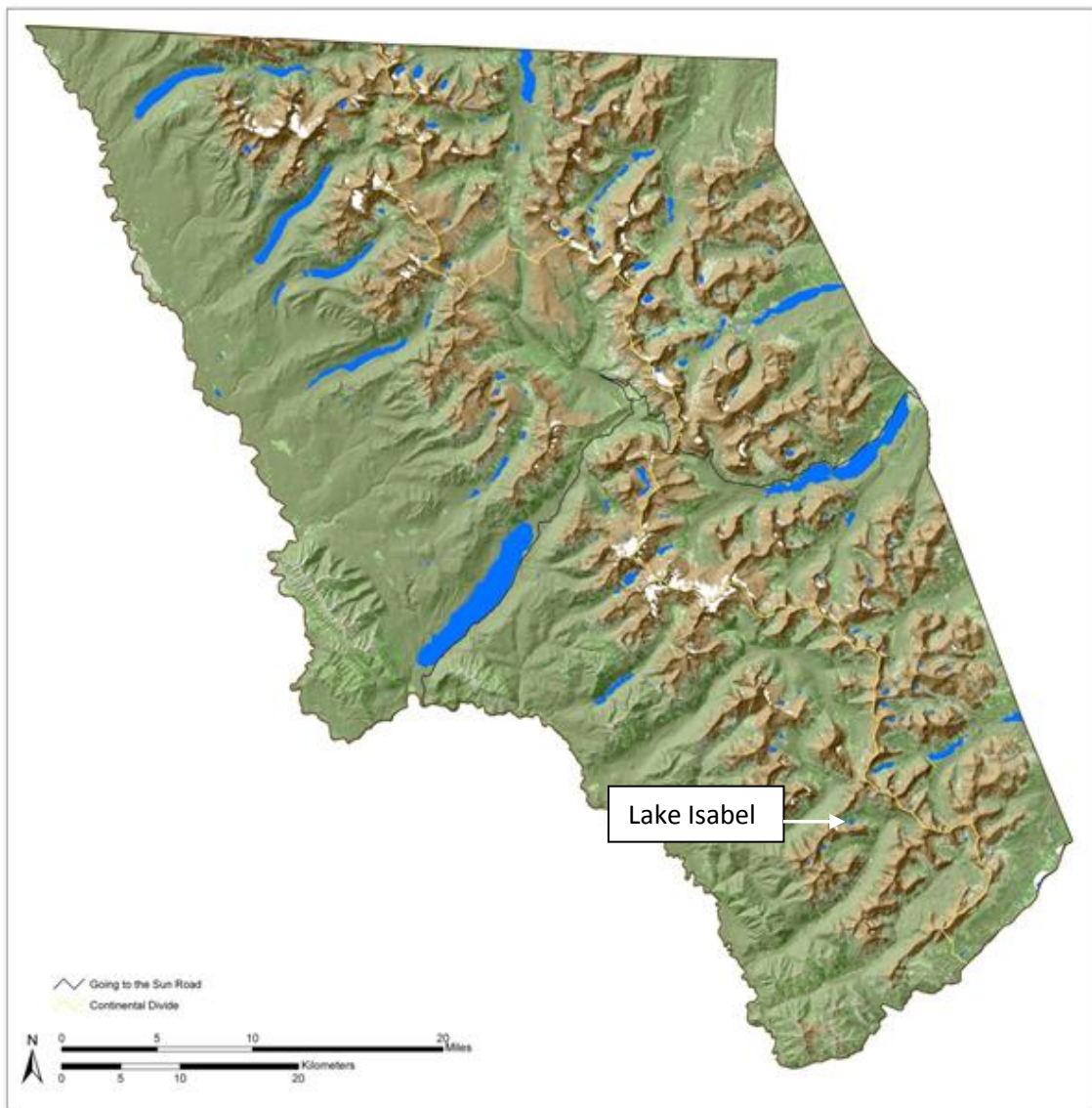


Figure 1. Location of Lake Isabel, Middle Fork Flathead River drainage, Glacier National Park

METHODS

We used custom-made mini hoop nets consisting of 1/2" (bar measure) green nylon mesh with three supporting metal hoops, each measuring 650 cm in diameter. The overall length of the traps were 1.5 meters with a single throat with a cod-end diameter of 200cm. Most hoop nets were baited with either dry cat food (salmon and chicken flavor) or various types of Power Bait. We also deployed mini trap nets made of 1/2" (bar measure) brown nylon mesh. Each net measured 3 m in length and had leads 7 m long with a leadcore and polyfoam line used to hold the lead line upright, while maintaining contact with the lake bottom. Their rectangular opening at the mouth of the trap net measures 700 cm tall by 1 meter wide with two rectangular throats 1 m tall by 300 cm wide to guide fish into the trapping area. Trap nets were generally not baited.

Nets were set from small two-person inflatable rafts in the afternoon and allowed to soak overnight. Nets were set at a variety of depths along the North shore of the lake (Figure 2 and Table 1). Due to the bulky nature of the nets and the difficulty in transporting live fish across the lake, we concentrated our efforts near the campground where access was good. They were retrieved at sunrise the following day. All fish were measured (TL; mm), weighed (g), and released.



Figure 2. Locations of Hoop and Trap net sets in Lake Isabel, July 23-26, 2013.

Table 1. UTM's and set parameters of Isabel live trapping. For trap nets, "Depth" refers to the depth of the trap itself. Leads were extended towards shore.

Lake	Net	UTM_X	UTM_Y	Date Set	Time Set	Date Pulled	Time Pulled	Depth (feet)
Isabel	Trap Net 1	315613	5366411	7/23/2013	1000	7/24/2013	820	4
Isabel	Trap Net 2	315641	5366423	7/23/2013	1200	7/24/2013	815	5.3
Isabel	Trap Net 3	315571	5366429	7/24/2013	1150	7/25/2013	815	5
Isabel	Trap Net 4	315609	5366413	7/24/2013	1200	7/25/2013	845	4.5
Isabel	Trap Net 5	315641	5366423	7/24/2013	1200	7/25/2013	830	5
Isabel	Hoop Net 1	315535	5366429	7/23/2013	1120	7/24/2013	900	15
Isabel	Hoop Net 2	315579	5366395	7/23/2013	1120	7/24/2013	950	19
Isabel	Hoop Net 3	315636	5366391	7/23/2013	1100	7/24/2013	1015	12
Isabel	Hoop Net 4	315510	5366428	7/23/2013	1200	7/24/2013	900	5.5
Isabel	Hoop Net 5	315482	5366414	7/24/2013	1200	7/25/2013	925	21
Isabel	Hoop Net 6	315522	5366417	7/24/2013	1200	7/25/2013	930	20
Isabel	Hoop Net 7	315571	5366406	7/24/2013	1200	7/25/2013	945	16
Isabel	Hoop Net 8	315668	5366379	7/24/2013	1200	7/25/2013	1000	16.3

Angling was also used to sample fish in Lake Isabel. Light spinning tackle and a small silver spoon were employed both from shore and from a small inflatable boat. Fishing time as well as fish length was recorded.

We used Relative Weight (Wr) (Anderson and Neuman 1996) to evaluate fish condition. We used the Standard Length equation for bull trout as: $\log_{10}(Ws) = -5.327 + 3.115 * \log_{10}(\text{total length})$ (Hyatt and Hubert 2000) and for westslope cutthroat trout (lentic cutthroat trout) as: $\log_{10}(Ws) = -5.681 + 3.246 * \log_{10}(\text{total length})$ (Kruse and Hubert 1997).

RESULTS AND DISCUSSION

We captured a total of 65 fish comprising two species during netting efforts (Table 2). Mean length for bull trout captured was 249.5mm (Table 2, Figure 3). Mean weight and mean Wr were 135.9 g and 95.3, respectively. Mean length for cutthroat was 261.8 mm. Mean weight and mean Wr for WCT were 158.7 g and 0.89, respectively (Table 2, Figure 3).

Table 2. Mean length (TL; mm), length range, mean weight (g), mean Relative Weight (Wr), standard deviation (SD), and sample size(n) for fish captured in Lake Isabel, Glacier National Park in 2004 and 2013. 2004 sampling was conducted using gill nets while 2013 sampling employed trap and hoop nets.

Sample year	Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)	Mean Relative weight (Wr) (SD)(n)
2013	BLT	249.5 (3.7)(45)	158-286	135.9 (33.3)(42)	95.3(8.4)(42)
2004	BLT	253.7(29.5)(57)	115-297	143.5 (36.3)(57)	96.6 (8.6)(57)
2013	WCT	261.8 (46.0)(20)	131-302	158.7 (53.4)(20)	103.7 (17.2)(20)
2004	WCT	270.9 (35.1)(93)	165-310	185.6 (50.5)(93)	112.5 (34.9)(93)

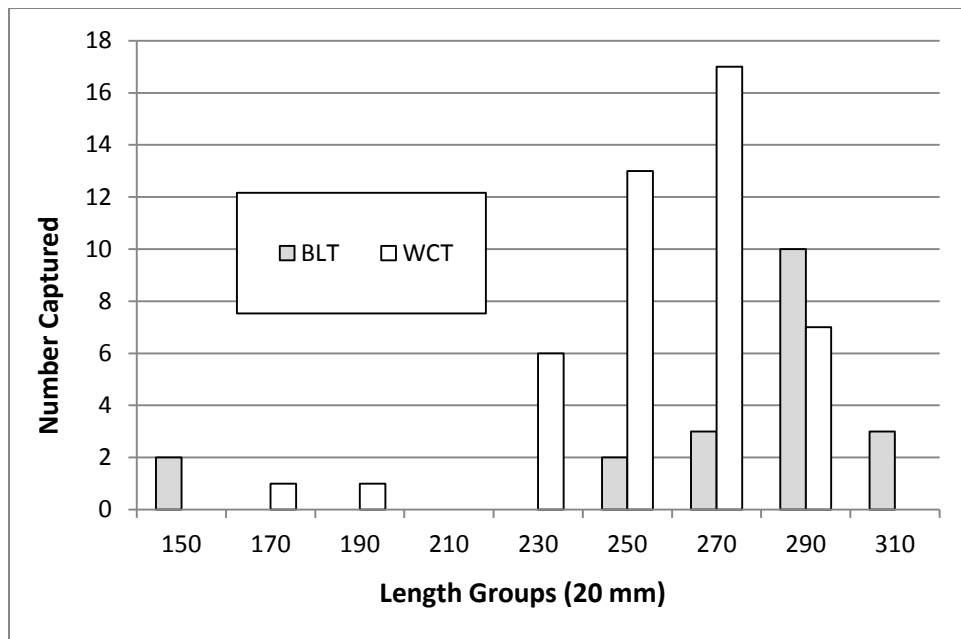


Figure 3. Length-frequency histogram for blt and wct captured in Lake Isabel, Glacier National Park, in 2013.

The unbaited trap nets appeared more effective than baited hoop nets, catching 11 more blt and 10 more wct, despite fishing fewer net nights (Figure 4). CPUE as fish/net night for wct and blt in hoop nets (n=8) was 0.63 and 2.13, respectively (Table 3). CPUE as fish/net night for wct and blt with trap nets (n=5) was 3 and 5.6, respectively (Table 3). Furthermore, CPUE as fish/hour for wct and blt in hoop nets was 0.03 and 0.10, respectively as compared to a CPUE as fish/hour for wct and blt in trap nets of 0.14 and 0.27, respectively (Table 3).

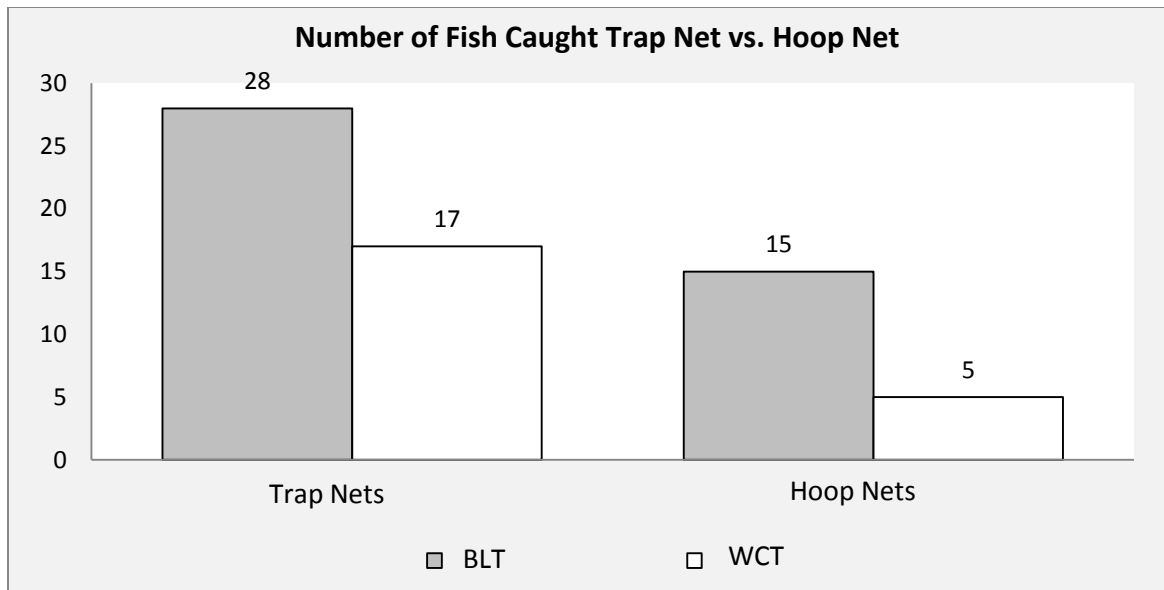


Figure 4. Catch comparison for trap (n=5) and hoop nets (n=8) at Lake Isabel, Glacier National Park, 2013.

Table 3. Catch composition for Lake Isabel, Glacier National Park, 2013.

Net Type	Species	Number Captured	CPUE/Net night	CPUE/Hour
Hoop Nets				
	BLT	17	2.13	.10
	WCT	5	0.63	.03
Trap Nets				
	BLT	28	5.6	.27
	WCT	15	3	.14

While letting our nets soak two technicians sampled the lake using hook and line and silver spoons. They fished for a total of 3.75 hours and captured 25 wct for a catch rate of 6.75 fish/hr. The average fish length was 229 mm (SD=55.6, n=24), ranging from 124 to 302 mm (Figure 5). Upper Isabel Lake was also fished and several westslope cutthroat were caught. We did not fish sufficient time to develop an angling CPUE estimate for Upper Isabel Lake.

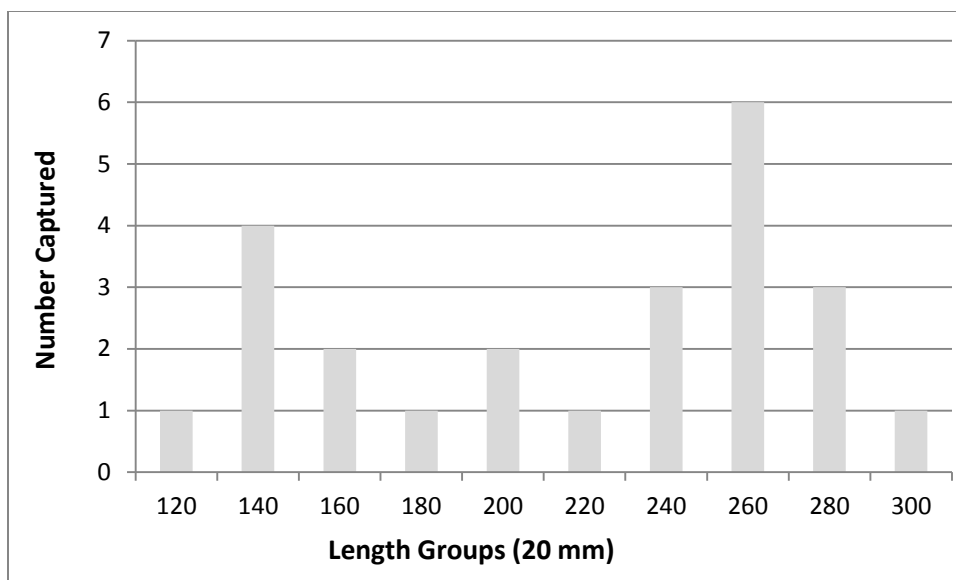


Figure 5. Length frequency of wct captured by angling at Lake Isabel in 2013.

When we compare our bull trout catch data to previous sampling conducted in 2004 (M. Meeuwig, Oregon State University, personal communication of unpublished data), we see a similar range of fish sizes captured and relative abundance of various size classes (Figure 6). Similarly, when we combine our wct capture data (angling and netting) and compare to that collected in 2004, we see similar length range coverage across sampling gears (Figure 7). However, on average, wct sampled in 2013 were slightly smaller than those sampled in 2004 (Table 2). This is likely a result of differences in sampling techniques and sample sizes, with gill nets catching more wct and a higher proportion of larger wct. Fish condition within species was also similar across sampling years (Table 2).

Our traps proved to be an effective alternative to gill netting. Although gill netting is an effective sampling tool, it results in high mortality (often >75%) to captured fish. We only had one mortality in all of the traps we set. Determining the size structure of populations in live entrapment gear may be biased due to predation of smaller individuals in the nets, differences in habitat use, or size selectivity of the trap mesh. However, based on the relatively small size of the adult bull and westslope cutthroat trout we captured, it is unlikely significant predation was occurring in our traps. Based on comparisons of the data, we conclude our live capture methods provide an appropriate alternative to gill netting for long-term trend monitoring in Lake Isabel.

Existing park fishing regulations permit the fishing of Lake Isabel but no harvest. We saw no evidence in the sampling data that recreational fishing is having an adverse impact on fish populations in Lake Isabel. Future surveys should be conducted every 5-10 years in response to any changing habitat/visitor use conditions.

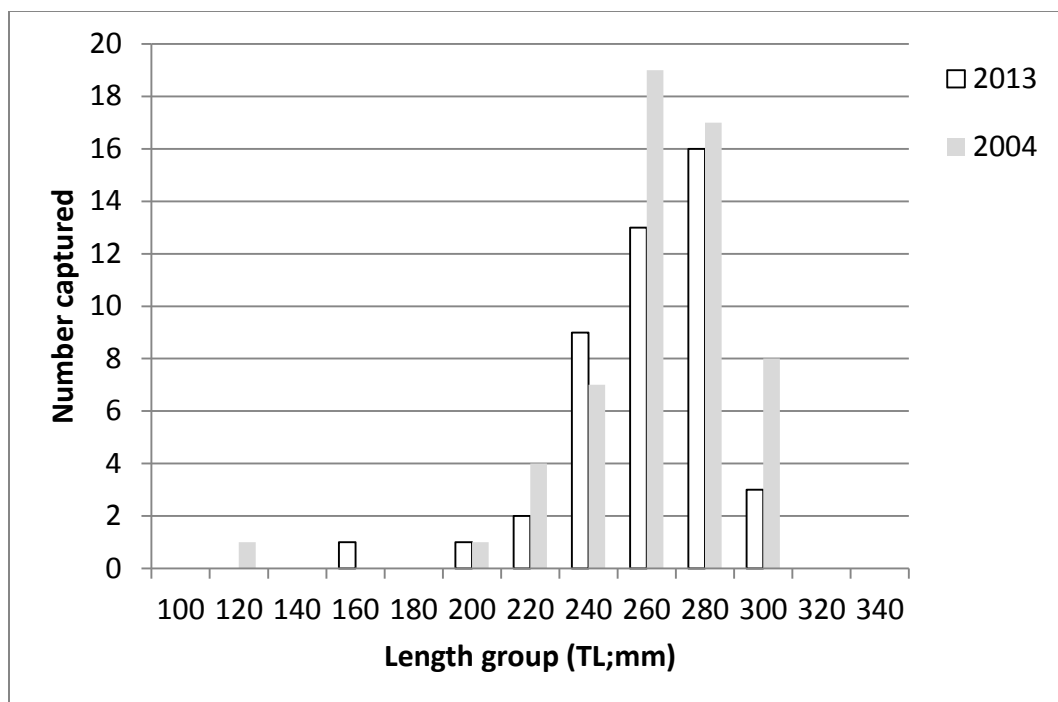


Figure 6. Length-frequency histogram for bull trout captured in Lake Isabel, Glacier National Park using gill netting (2004) and trap/hoop netting (2013).

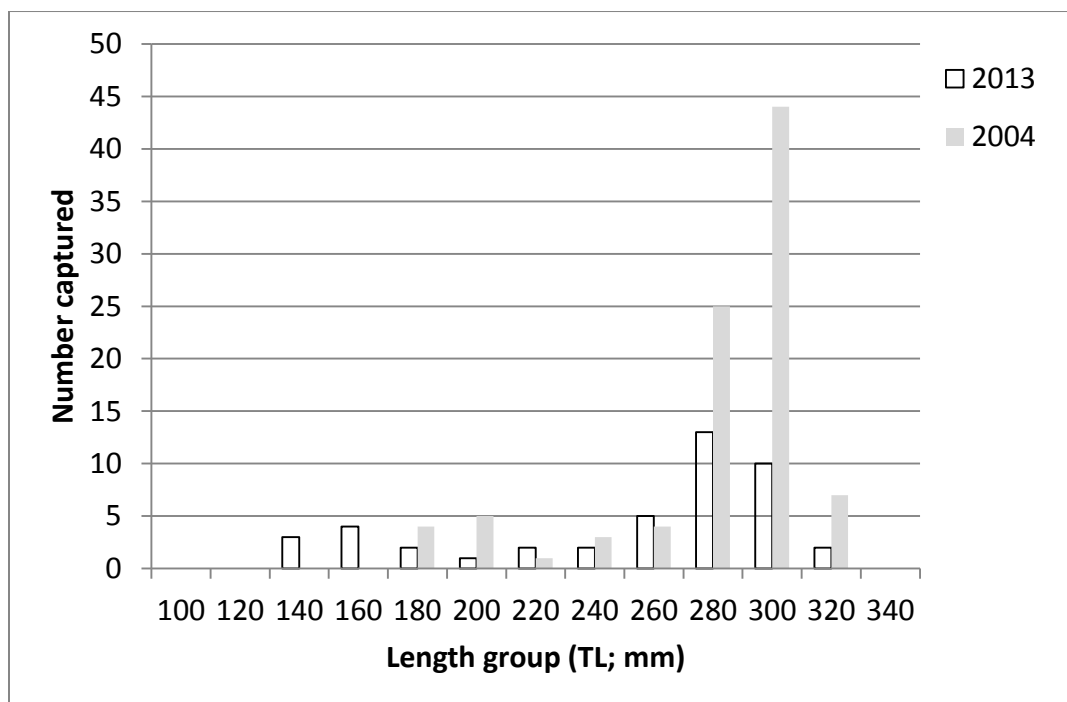


Figure 7. Length-frequency histogram for westslope cutthroat trout captured in Lake Isabel, Glacier National Park using gill netting (2004) and trap netting, hoop netting, and angling (2013).

ACKNOWLEDGEMENTS

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Aquatic Invasive Species Prevention and Monitoring

ABSTRACT

In 2013, Glacier National Park (GNP) continued to implement a program of mandatory boat inspection and launch permitting for all non-hand powered boats entering GNP as well as a AIS self-certification program for all hand-powered watercraft. In 2011, 1,257 boat inspections were performed and six boats were denied launch permits for reasons ranging from standing water present in internal areas of the boat to dried vegetation adhered to the boat hull. In 2012, GNP conducted inspections on 1,107 boats and issued 1,101 permits to launch boats. Six boats were denied permits to launch for reasons ranging from standing water in the boat to dried vegetation adhered to the hull. In 2013, GNP conducted inspections on 1,174 issued 1,164 permits to launch boats. Ten boats were denied permits to launch for reasons ranging from standing water in the boat to dried vegetation adhered to the hull. No AIS were found on any of the boats inspected over the past three years. Boats entered GNP from 24 States in 2013, 17 of which have populations of invasive mussels demonstrating the continued risk to park waters. In addition, we continued the parks monitoring program for AIS, including both artificial substrates to monitor for adult mussels and water sampling for the presence of mussel veligers. No AIS have been detected.

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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 of those ecosystems. 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, Idaho, 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). A live mussel was also found on a trailered sailboat at a marina on Flathead Lake. The boat had recently arrived from the southwest U.S., had reportedly been decontaminated, and was not launched in Montana.

Other AIS threaten GNP as well. Plant species such as Eurasian watermilfoil *Myriophyllum spicatum*, purple loosestrife *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.

In 2009, GNP initiated a project to evaluate the risk of introduction and establishment of AIS in park waters (Downs et al. 2011) which documented the risk of potential AIS introduction and establishment. This evaluation, along with heightened awareness of the ecological, financial, and social impacts that AIS such as zebra and quagga mussels cause, prompted GNP to begin a limited boat inspection and launch permitting program in 2010. The initial program required a permit to launch motorized watercraft in park waters. In order to qualify for a launch permit, non-resident boaters were required to submit to an inspection of their boat for the presence of AIS. Because zebra and quagga mussels are not present in Montana, an AIS-free self-certification program for resident motorized watercraft was implemented (rather than requiring NPS inspection). Resident boaters qualified for the launch permit by completing the AIS free self-certification form. However, due to the expanding nature of the AIS threat, including live mussels found on boats in Montana and invasive aquatic plants found in Flathead county, in 2012 the park expanded the program to require inspection and permitting of all non-hand propelled watercraft before launch in any park water. The updated program included also an AIS-free self-certification requirement for all hand-powered non-motorized watercraft (e.g. canoes, kayaks) as they presumably present a lower risk of infestation and transport of AIS.

METHODS

Upon entry to the park boaters are informed by signage and/or entrance gate staff that a boat inspection and launch permit was required for all non-handpowered boats launching within the park. The launch permit remains valid as long as the boat does not leave the park. Re-inspection is required upon re-entry into the park. Inspections/permits are generally offered in the immediate area of water bodies that permit boating use (Figure 1). Boaters intending to boat on Bowman Lake were encouraged to have their boats inspected in West Glacier because it is difficult to keep a boat clean for inspection at Bowman Lake under wet road conditions on the North Fork Road, as well as staffing limitations. In the West Glacier vicinity, the busiest boat launch area (Lake McDonald), inspectors were station at the Backcountry Permit office in the “shoulder” boating seasons, and at the Park Headquarters for the peak summer-use months. Boat inspections were conducted by trained NPS staff. Waterton Lakes NP in Canada controls access to Waterton Lake, which spans the US-Canadian border. Waterton Lakes NP does not operate an identical inspection and permitting program but does inspect boats from the U.S., as well as from mussel-infested Canadian Provinces.

A public education effort regarding the risks of AIS to GNP waters continued in 2013. 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 recently collaborating with the NPS Rocky Mountain Cooperative Ecosystem Studies Unit, the University of Montana and the Crown Managers Partnership to produce a color AIS pocket guide.

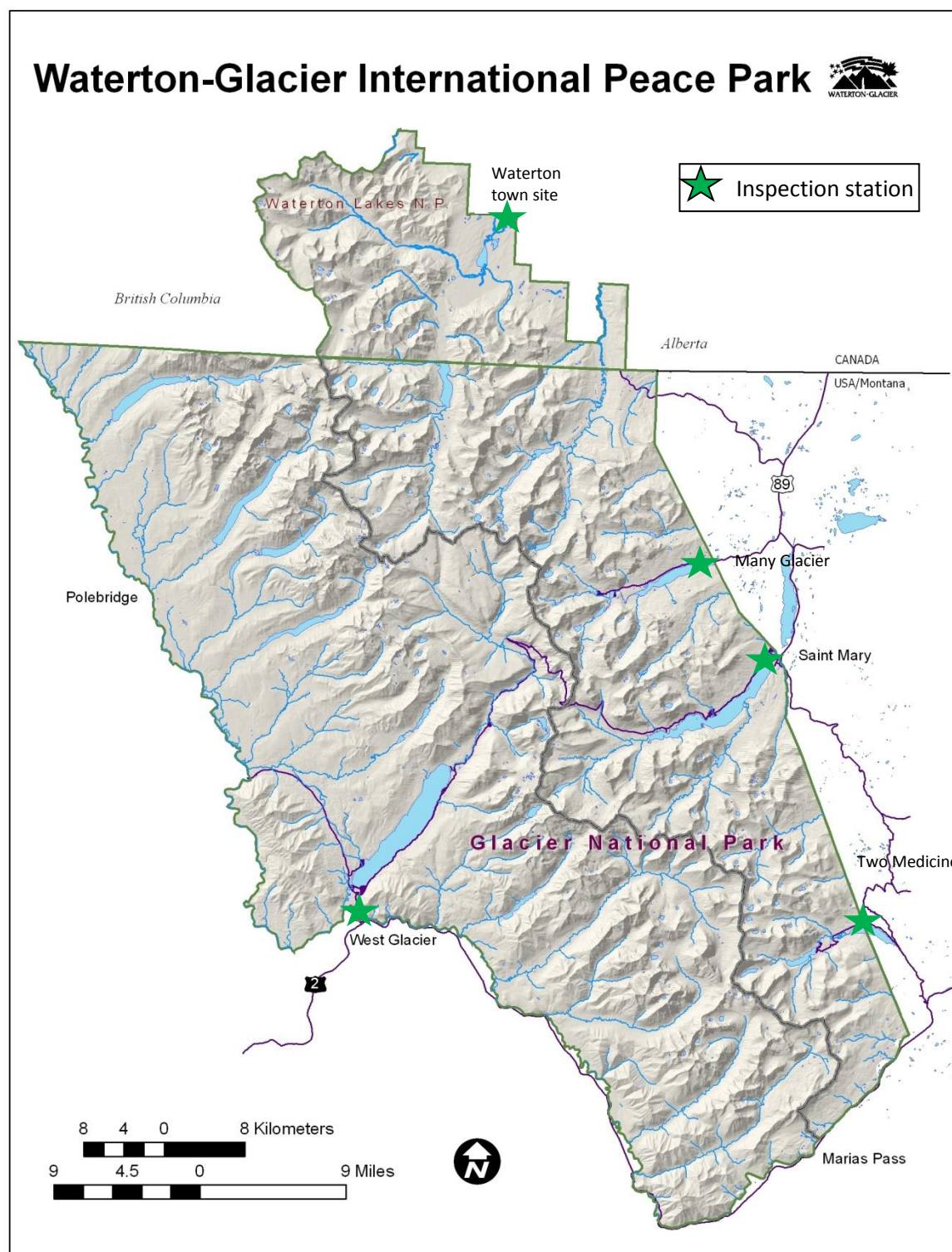


Figure 1. AIS boat inspection and launch permitting locations in Glacier National Park, 2013.

We used artificial substrates and plankton sampling to monitor for the presence of invasive mussels (Figure 2). We deployed artificial substrates in Bowman Lake, Lake McDonald, Two Medicine Lake, and St. Mary Lake. Artificial substrates were generally deployed near the lake bottom in boat launch areas at depths between 3' and 10'. 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 artificial substrates were generally deployed in June and retrieved at the end of summer. Upon retrieval, artificial substrates were inspected for the presence of adult and sub-adult mussels.

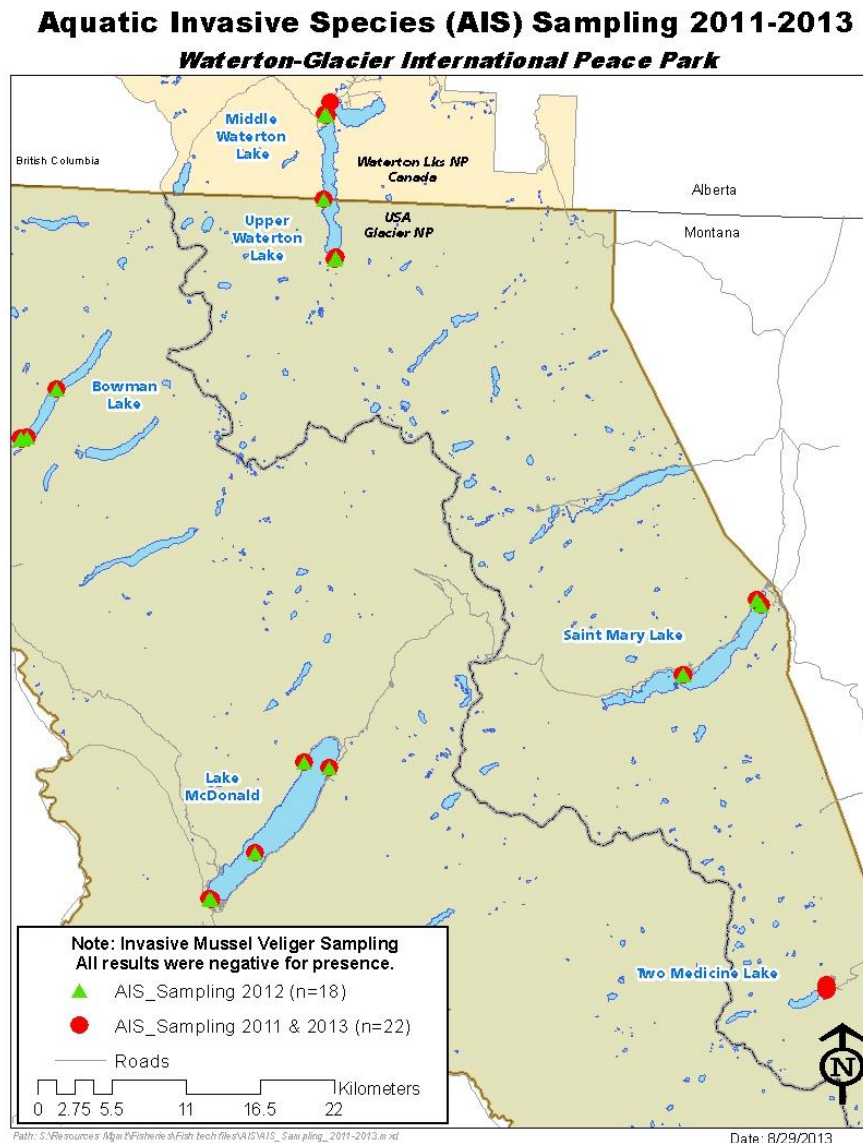


Figure 2. Invasive mussel larvae (veliger) sampling in Waterton-Glacier International Peace Park, 2011-2013.

Sampling for the presence of the larval form of the invasive mussels (i.e. veligers) was conducted according to established MFWP protocols (E. Ryce, MFWP, personal communication). Sampling

occurred during peak surface water temperatures, after surface water temperatures had reached at least 52°F (11°C). We used a standard 64 µm mesh vertical plankton sampling net and we made triplicate vertical hauls at each site. The net was lowered into the water to a maximum depth of 20m or to within 1m of the lake bottom (if sample site depth < 20m) and then slowly retrieved to the water surface. Samples were preserved in 100% non-denatured ethanol, using a 1:2 sample:ethanol ratio. Veliger monitoring sites typically included a sample at the foot of the lake near the boat launch, one at mid-lake, and one at the upper end of the lake. GIS coordinates were taken for all sampling locations (Table 1).

Table 1. Invasive mussel veliger sampling locations in GNP, 2011-2013.

Date Collected	Latitude	Longitude	Water Body
8/8/2011	48.52902	-113.99021	Lake McDonald
8/8/2011	48.62328	-113.90263	Lake McDonald
8/8/2011	48.52997	-113.99287	Lake McDonald
8/8/2011	48.6198	-113.87714	Lake McDonald
8/8/2011	48.56185	-113.94833	Lake McDonald
8/9/2011	48.48318	-113.37076	Two Medicine Lake
8/9/2011	48.48489	-113.37079	Two Medicine Lake
8/9/2011	48.48718	-113.37049	Two Medicine Lake
8/9/2011	48.69008	-113.52563	St. Mary Lake
8/9/2011	48.69019	-113.52367	St. Mary Lake
8/9/2011	48.73809	-113.44813	St. Mary Lake
8/9/2011	48.74227	-113.45276	St. Mary Lake
8/10/2011	48.95856	-113.89165	Upper Waterton Lake
8/10/2011	48.96039	-113.88978	Upper Waterton Lake
8/10/2011	48.99861	-113.90498	Upper Waterton Lake
8/10/2011	49.05483	-113.90463	Upper Waterton Lake
8/10/2011	49.05507	-113.90693	Upper Waterton Lake
8/10/2011	49.06357	-113.90185	Middle Waterton Lake
8/11/2011	48.83218	-114.19427	Bowman Lake
8/11/2011	48.83062	-114.19975	Bowman Lake
8/11/2011	48.83131	-114.20112	Bowman Lake
8/11/2011	48.86473	-114.16692	Bowman Lake
8/8/2012	48.9585	-113.89165	U. Waterton Lake
8/8/2012	48.96034	-113.88978	U. Waterton Lake
8/8/2012	48.99855	-113.90498	U. Waterton Lake
8/8/2012	49.05477	-113.90463	U. Waterton Lake
8/8/2012	49.05501	-113.90693	U. Waterton Lake
8/7/2012	48.69004	-113.52563	St. Mary Lake
8/7/2012	48.69016	-113.52367	St. Mary Lake
8/7/2012	48.73806	-113.44813	St. Mary Lake

Table 1. Continued.

Date Collected	Latitude	Longitude	Water Body
8/7/2012	48.74224	-113.45276	St. Mary Lake
8/6/2012	48.52895	-113.9902	Lake McDonald
8/6/2012	48.62322	-113.90263	Lake McDonald
8/6/2012	48.5299	-113.99287	Lake McDonald
8/6/2012	48.61974	-113.87714	Lake McDonald
8/6/2012	48.56179	-113.94833	Lake McDonald
8/6/2012	48.83213	-114.19427	Bowman Lake
8/6/2012	48.83057	-114.19975	Bowman Lake
8/6/2012	48.83125	-114.20111	Bowman Lake
8/6/2012	48.86468	-114.16692	Bowman Lake
8/8/2013	48.52902	-113.99021	Lake McDonald
8/8/2013	48.62328	-113.90263	Lake McDonald
8/8/2013	48.52997	-113.99287	Lake McDonald
8/8/2013	48.6198	-113.87714	Lake McDonald
8/8/2013	48.56185	-113.94833	Lake McDonald
8/6/2013	48.48318	-113.37076	Two Medicine Lake
8/6/2013	48.48489	-113.37079	Two Medicine Lake
8/6/2013	48.48718	-113.37049	Two Medicine Lake
8/6/2013	48.69008	-113.52563	St. Mary Lake
8/6/2013	48.69019	-113.52367	St. Mary Lake
8/6/2013	48.73809	-113.44813	St. Mary Lake
8/6/2013	48.74227	-113.45276	St. Mary Lake
8/7/2013	48.95856	-113.89165	Upper Waterton Lake
8/7/2013	48.96039	-113.88978	Upper Waterton Lake
8/7/2013	48.99861	-113.90498	Upper Waterton Lake
8/7/2013	49.05483	-113.90463	Upper Waterton Lake
8/7/2013	49.05507	-113.90693	Upper Waterton Lake
8/7/2013	49.06357	-113.90185	Middle Waterton Lake
8/5/2013	48.83218	-114.19427	Bowman Lake
8/5/2013	48.83062	-114.19975	Bowman Lake
8/5/2013	48.83131	-114.20112	Bowman Lake
8/5/2013	48.86473	-114.16692	Bowman Lake

RESULTS AND DISCUSSION

In 2011, 1,257 boats were inspected and issued launch permits. Six other boats were denied launch permits for reasons ranging from standing water present in internal areas of the boat to dried vegetation adhered to the boat hull. Seventy-five percent of the inspections occurred at either Headquarters or the Apgar Backcountry permit office (Lake McDonald area). The remaining inspections took place at the St. Mary Visitor Center (11%), Two-Medicine Ranger Station (9%), and the Polebridge Ranger Station (4%). Eighty-eight percent of the boats were registered in Montana. Boats entered the park from three Canadian Provinces and nineteen states, eleven of which have populations of zebra and/or quagga mussels (Table 2). No AIS were found during any of the inspections.

In 2012, GNP conducted inspections on 1,107 boats and issued 1,101 permits to launch. As in 2011, six boats were denied permits to launch for reasons ranging from standing water in the boat to dried vegetation adhered to the hull to the presence of internal ballast tanks that could not be inspected. Again, 75% of the inspections occurred at either Headquarters or the Apgar Backcountry permit office (Lake McDonald area). The remaining inspections took place at the St. Mary Visitor Center (12%), Two-Medicine Ranger Station (10%), and the Polebridge Ranger Station (2%), and Many Glacier (<1%). Boats entered GNP from 19 States and 3 Canadian Provinces in 2012. Thirteen of the nineteen States have populations of zebra and/or quagga mussels. Seventy-eight percent of the inspected boats were registered in Montana.

In 2013, GNP conducted inspections on 1,174 boats and issued 1,164 permits to launch. Ten boats were denied permits to launch for reasons ranging from standing water in the boat to dried vegetation adhered to the hull to the presence of internal ballast tanks that could not be inspected. No AIS were detected. Seventy-nine percent of the inspections occurred at either Headquarters or the Apgar Backcountry permit office (Lake McDonald area). The remaining inspections took place at the St. Mary Visitor Center (11%), Two-Medicine Ranger Station (8%), and the Polebridge Ranger Station (2%), and Many Glacier (<1%). Boats entered GNP from 24 States and 3 Canadian Provinces this year (Table 2). Seventeen of the 24 States have populations of zebra and/or quagga mussels, demonstrating the continued risk of introduction. Eighty-six percent of all boats inspected were registered in Montana.

Table 2. Summary of boats inspected in GNP from States with zebra and/or quagga mussels.

State	2011	2012	2013	Total
Alabama	0	1	0	1
Arizona	8	8	4	20
Arkansas	0	0	1	1
California	19	14	8	41
Colorado	3	2	4	9
Illinois	1	2	1	4
Indiana	1	1	2	4
Louisiana	0	0	1	1
Maryland	0	0	1	1
Michigan	0	1	1	2
Minnesota	5	3	1	9
Missouri	3	0	1	4
Nevada	1	0	2	3
New Mexico	0	0	2	2
New York	0	2	0	2
North Dakota	0	0	2	2
Oklahoma	0	1	0	1
Texas	1	2	1	4
Utah	1	3	0	4
Wisconsin	3	1	1	5
Wyoming	0	0	6	6

Water temperatures were recorded throughout the high boating-use summer months on waters with developed boat ramps. Zebra and Quagga mussels will spawn when water temperatures reach and remain above 11°C (52°F). All waters sampled had extended time-periods when water temperatures remained consistently above this level in 2011-2013 (Figures 3-6). Water temperature measurements suggest a potential summer spawning season for invasive mussels in park lakes lasting from one month (St. Mary Lake) to three months (Bowman Lake).

We timed our invasive mussel veliger sampling (plankton hauls) to correspond with peak water temperatures. Sampling in 2013 occurred from August 5-8. We sampled all of the primary boating waters in the park, as well as Lower Waterton Lake located solely in Waterton Lakes NP (Figure 2). No invasive mussel larvae were detected. Furthermore, invasive mussel substrates were removed at the end of the boating season and inspected for the presence of invasive mussels each year. None were found.

Two Medicine Lake reached a maximum temperature of 19.0 °C on 8/14/2013 (Figure 4), while St. Mary Lake reached a maximum temperature of 19.3 °C on 8/11/2013 (Figure 6). Lake McDonald reached a maximum temperature of 21.9 °C on 8/8/2013 (Figure 5). Bowman Lake reached a maximum temperature of 22.2 °C on 7/1/2013 (Figure 3).

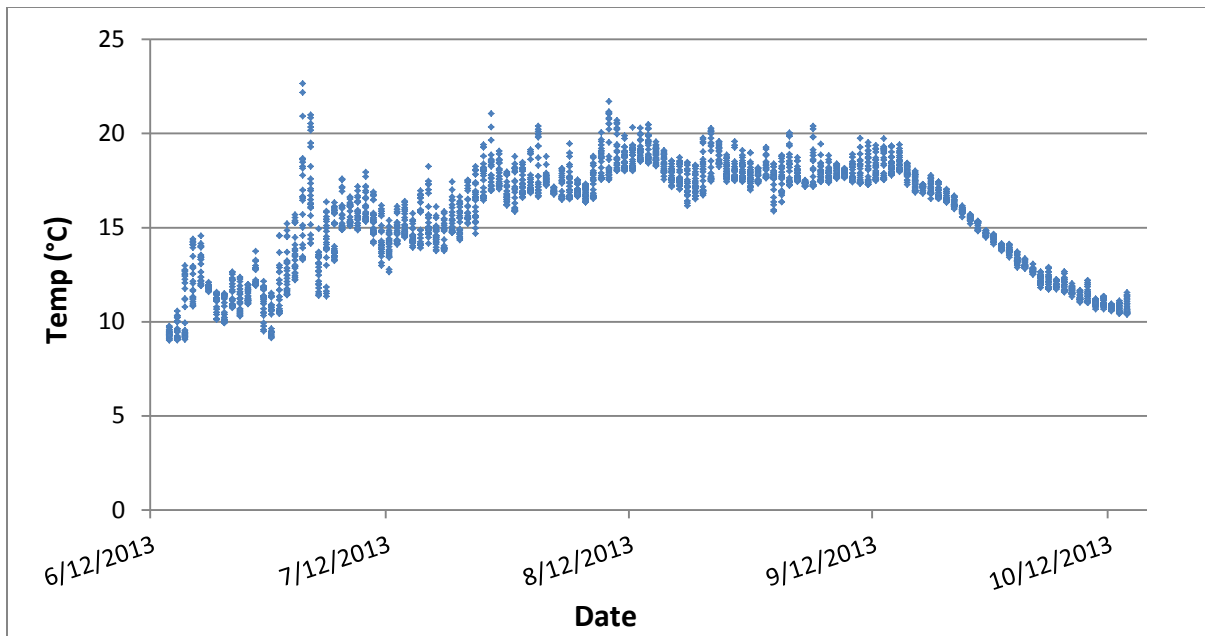


Figure 3. 2013 Bowman Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in approximately four feet of water.

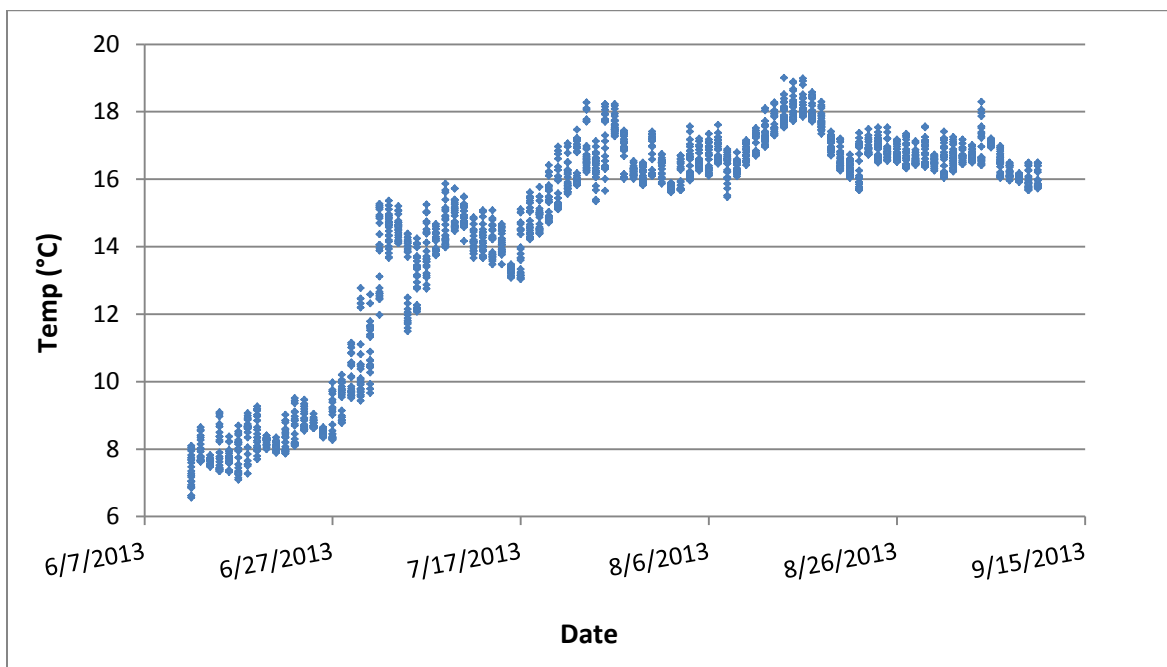


Figure 4. 2013 Two-Medicine Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock.

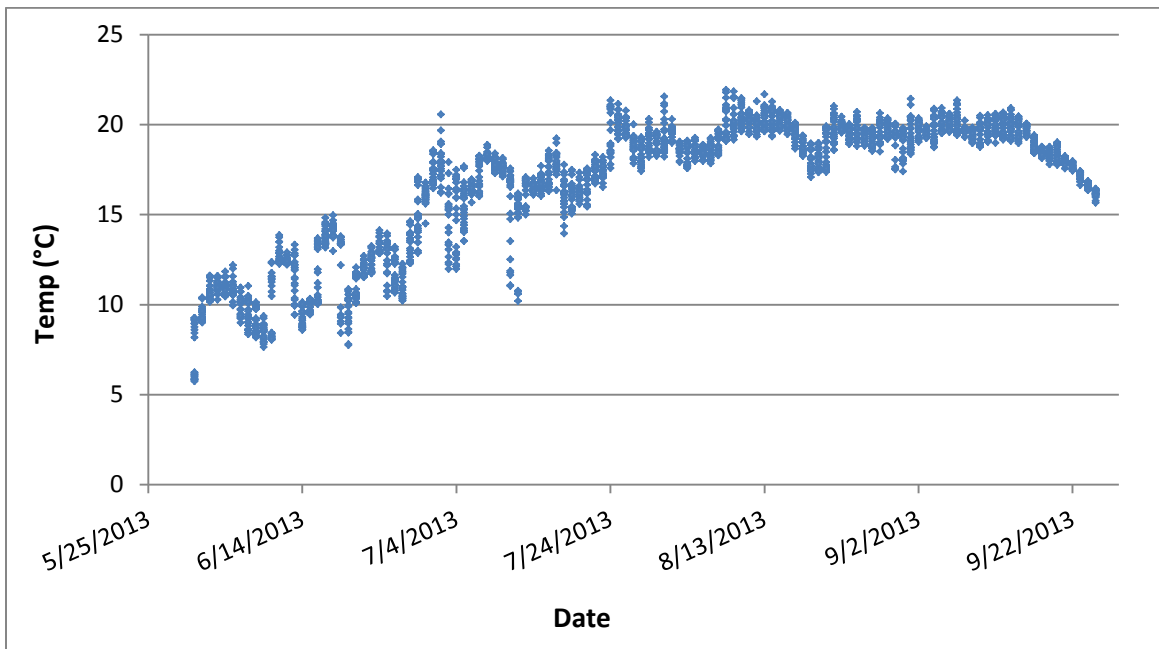


Figure 5. 2013 Lake McDonald Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock.

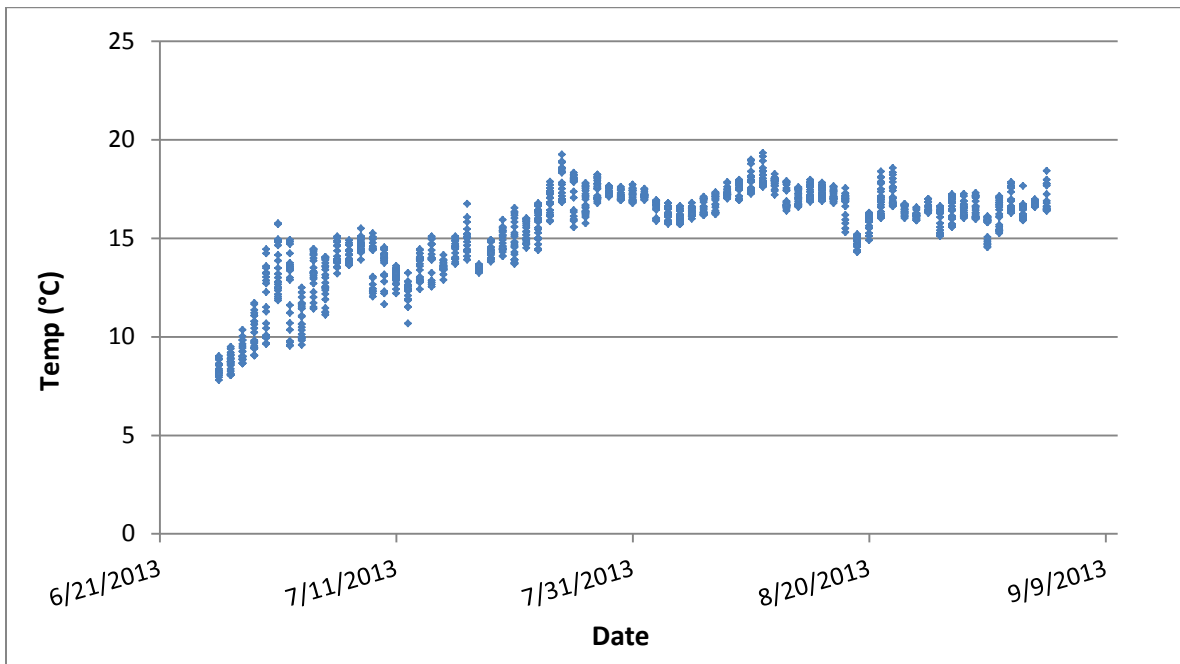


Figure 6. 2013 St. Mary Lake water temperature recorded near the lake bottom at the end of the boat dock.

2010 saw the beginning of more aggressive prevention and monitoring efforts by the NPS aimed at preventing additional AIS from colonizing park waters. Initiation of this effort was very timely given the westward movement of AIS such as zebra and quagga mussels and the program should continue into the future. The current GNP AIS prevention program of inspecting all non-hand-propelled water craft (largely motorized watercraft), along with self-inspection and certification of hand-propelled watercraft such as canoes and kayaks can be expected to significantly reduce the risk of unintended transport of AIS by boats into park waters. 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.

ACKNOWLEDGEMENTS

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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 2013. Seven streams/stream reaches were surveyed in the N. Fk. Flathead River drainage, two were surveyed in the M. Fk. Flathead River drainage, and four were surveyed in the St. Mary River drainage. We counted a total of 166 redds across the park. The Quartz/Cerulean lakes complex remains the strongest monitored bull trout population residing wholly within the park, with a total of 26 redds. 2013 redd counts for Flathead Lake migratory bull trout populations spawning in the Middle Fork Flathead River tributaries in the park were mixed compared to long-term averages. Redd counts for bull trout populations spawning in the St. Mary drainage were also mixed, with Boulder creek having a strong, above average count, while the count in Kennedy Creek continued to be well below average. In general, bull trout populations in west-side park lakes continue to show very low escapement levels, reflecting the adverse impacts of non-native lake trout on native fish populations. Some 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, 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 the adfluvial life history strategy. These bull trout grow to maturity in the lakes, and then spawn in tributaries or lake outlets. 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 September and October in the Flathead River/Lake system (Block 1953; Fraley and Shepard 1989; Meeuwig 2008), including Glacier National Park lakes (Tennant 2010). 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°C) 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 (Thurrow 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 limitation, 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), 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) 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 prudent to establish index redd count monitoring for additional 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 2013. Redd surveys generally occur during the first full three weeks of

October. Surveys occurred between September 27-October 18. 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, Nyack, and Quartz creeks on the west side of the park meet the criteria. We used a nonparametric rank-correlation procedure, Kendall’s tau-b (Daniel 1990), to test for trends in “count year” versus “redd count” in the long-term redd count data set and noted statistical significance at the $\alpha = 0.05$ level (Rieman and Myers 1997).

RESULTS AND DISCUSSION

GNP, USGS, USFWS, and MFWP staff surveyed five stream reaches in the N. Fk. Flathead River drainage and four in the M. Fk. Flathead River drainage. In addition, four other streams were surveyed in the St. Mary River drainage by the GNP, USFWS, and Blackfoot Tribe personnel (Figure 1).

East of the Continental Divide, bull trout redd counts continue to remain relatively strong, although few populations are monitored (Table 1). Redd counts in 2013 were average for Boulder Creek, and below average for Kennedy Creek (Figure 2). We recently initiated redd counts on Lee Creek and counted 12 redds, down from 2012’s count of 31. Correlations in “count year” versus “redd count” identified a statistically significant positive trend in the redd counts on Boulder Creek (tau-b = 0.50; $p < 0.05$), suggesting spawner numbers are increasing in the population. However in Kennedy Creek, a tributary located downstream of the St. Mary Irrigation Diversion, the same correlation measure was both statistically significant and negative (-0.45; $p < 0.05$), suggesting declining adult abundance. Bull trout habitat quality is higher in Boulder Creek. Streambank stability and bed stability are also higher. The primary spawning and rearing reach of Kennedy Creek can be characterized as relatively unstable and dominated by cobble-sized substrate. Spawning gravel is very limited. On the west side of the park, the only statistically significant trend was in the short-term redd count data (most recent 10 years) on Ole Creek, which was significantly positive (0.73; $p < 0.05$).

Table 1. Bull trout redd counts conducted in Glacier National Park, 1994 to present.

	Akokala Cr (o)	Bowman Cr (c,l)	Harrison Cr (d)	Logging Cr(n)	Lower Quartz	Middle Quartz (h)	Quartz Cr (a,b,e,m)	Rainbow Cr	Upper Kintla (m)
1994	--	--	--	--	--	--	--	--	52
1995	--	--	--	--	--	--	--	--	--
1996	--	--	--	--	--	--	--	--	--
1997	--	--	--	--	--	--	--	--	--
1998	--	--	--	--	--	--	--	--	--
1999	--	--	--	--	--	--	--	--	--
2000	--	--	--	--	--	--	--	--	--
2001	--	--	--	--	--	--	--	--	--
2002	--	0	--	--	--	--	--	--	--
2003	--	0	--	--	--	0	31	--	--
2004	--	0	4	3	1	0	46	--	--
2005	--	0	0	20	3	0	4	--	--
2006	--	2	8	0	2	0	36	--	--
2007	--	1	15	--	2	0	14	--	--
2008	11	0	14	5	3	0	51	28	0
2009	6	0	1	0	2	3	34	12	--
2010	1	1	6	3	2	0	27	4	25
2011	4	3	N/R	3	5	--	32	9	--
2012	5	--	1	1	3	--	33	--	--
2013	0	1	--	N/R	5	--	22	4	--

a=high flows may have obliterated some redds in 2005

b=spawning activity may have been impacted by weir placement near the mouth in 2007

c=minimum count during 2005 survey due to poor conditions

d=redd count accuracy may have been compromised by kokanee spawning activity in 2007 and 2008.

e=2008 data is a cumulative count based on multiple survey events

g=high flows may have obliterated some redds in 2004, 2005

h = old cutthroat redds still visible and may have affected count reliability

i = 3 additional redds were identified upstream of index reach in 2009

j=count was conducted by helicopter in 2008

l=poor counting conditions due to higher water levels and high turbidity (glacial) from Jefferson Creek in 2010

N/R=unreliable count due to kokanee spawning

m=2 redds observed in Quartz above confluence of rainbow creek in both 2008 and 2013 and included in total. This area (~400m) not counted in other years

n=2013 count likely inflated due to large (14-18") kokanee spawning at the time of the count. 14 redds not included that were presumed to be kokanee redds due to the observation of kokanee in the immediate vicinity of these redds

o=beaver dam located at creek mouth may have blocked bull trout access for spawning in 2013

Table 1. Continued.

Middle Fork Flathead Tributaries				St. Mary Tributaries			
	Ole Cr (g)	Park Cr (j)	Nyack Cr	Boulder Cr	Kennedy Cr (i)	Unnamed Kennedy Cr Trib.	Lee Cr
1980	19	--	14	--	--	--	--
1981	19	13	14	--	--	--	--
1982	51	0	23	--	--	--	--
1983	35	--	--	--	--	--	--
1984	26	--	--	--	--	--	--
1985	30	--	--	--	--	--	--
1986	36	87	27	--	--	--	--
1987	45	--	--	--	--	--	--
1988	59	--	--	--	--	--	--
1989	21	--	--	--	--	--	--
1990	20	--	--	--	--	--	--
1991	23	19	22	--	--	--	--
1992	16	1	12	--	--	--	--
1993	19	--	--	--	--	--	--
1994	6	--	--	--	--	--	--
1995	16	--	--	--	--	--	--
1996	10	--	--	--	--	--	--
1997	14	2	9	12	23	--	--
1998	22	--	--	42	37	--	--
1999	26	--	--	20	--	--	--
2000	33	10	13	30	23	--	--
2001	29	--	--	28	12	--	--
2002	21	--	--	28	11	--	--
2003	21	0	14	28	18	--	--
2004	14	--	--	27	13	--	--
2005	16	--	--	--	25	--	--
2006	31	--	--	50	20	--	--
2007	29	--	--	38	13	--	--
2008	42	23	16	58	22	--	--
2009	34	3	8	38	5	2	--
2010	32	3	6	33	12	0	--
2011	40	20	16	61	15	1	15
2012	53	15	14	53	9	--	31
2013	44	9	--	61	8	--	12

f = helicopter count in 2008.

g = high water may have obliterated some redds and poor count conditions in 2004 and 2005.

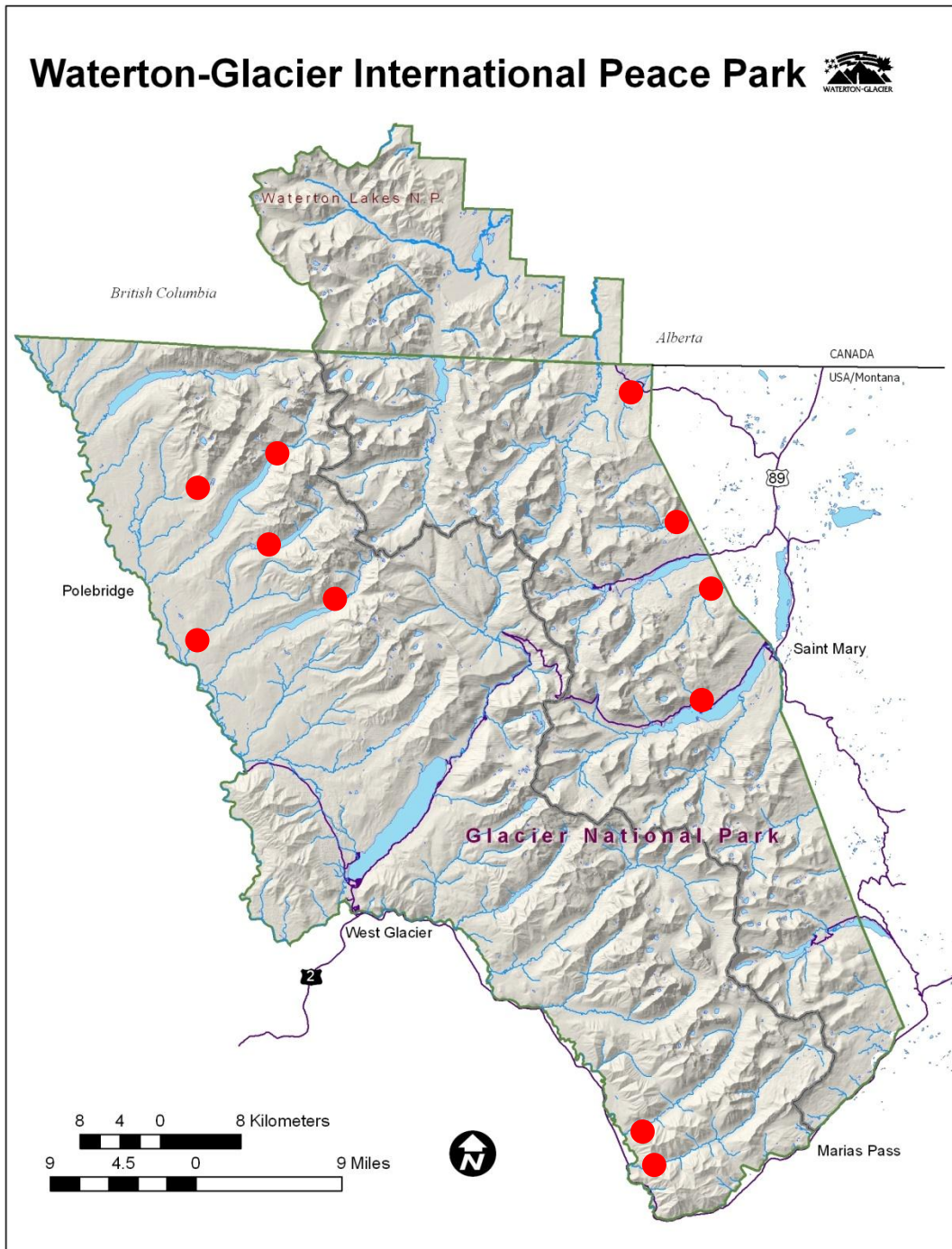


Figure 1. Drainages monitored for bull trout spawning activity (red circles) in Glacier National Park, Montana in 2013.

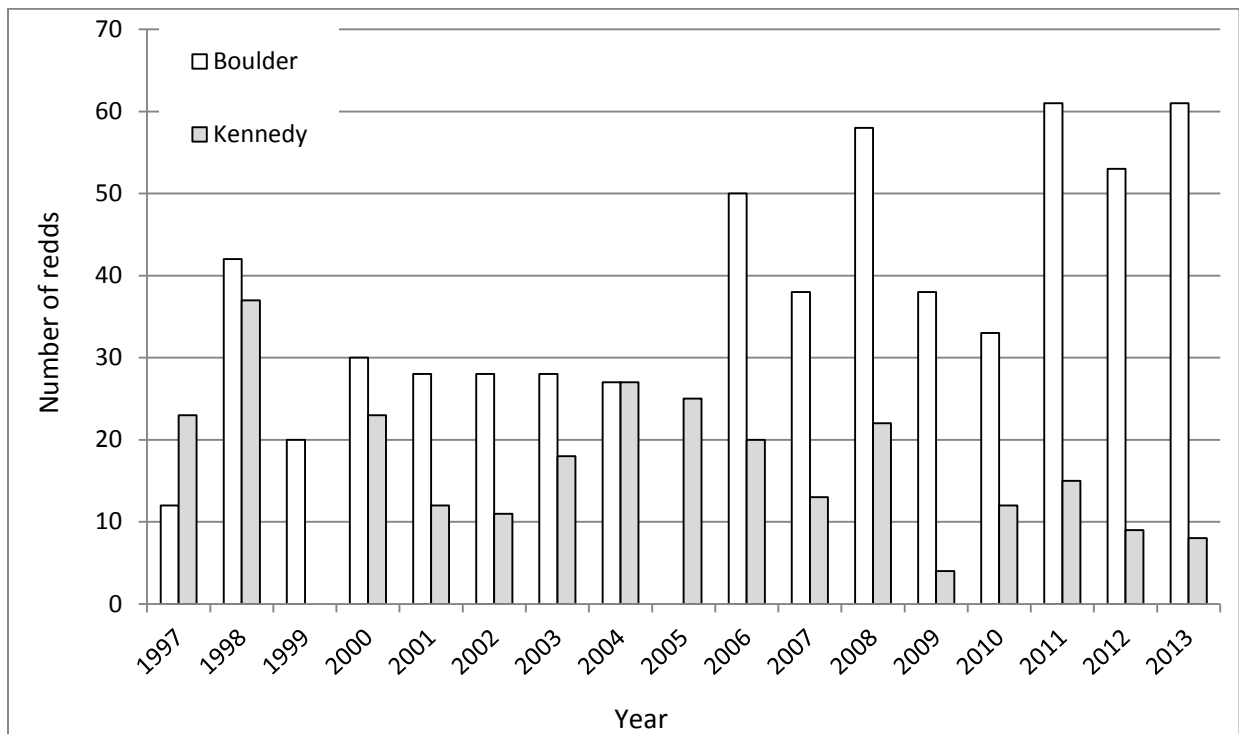


Figure 2. Bull trout redd counts for Boulder and Kennedy creeks, Hudson Bay Drainage, Glacier National Park.

Documentation of progress towards meeting recovery objectives in the St. Mary River drainage established by the USFWS (USFWS 2002) will be difficult without expanded monitoring of bull trout population abundance and trends in GNP. The four recovery criteria focus on quantitative measures of adult bull trout abundance and population trends. Recovery Criteria 1 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 3 of the 6 core area populations may limit our 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 and Lee creek drainages and the construction and operation of Sherburne Dam and the Milk River Irrigation Project (USFWS 2002). 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), as well as in Lee Creek (C. Downs, NPS, personal communication). 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 completely dewatered Swiftcurrent Creek downstream of the dam in the winter 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 permanent loss of hundreds of bull trout and thousands of other native fish from the system each year (J. Mogen, USFWS, personal communication). 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 (Table 1). 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). One of the most significant contemporary threats to these populations is predation with and competition by non-native fish species in both the migratory and rearing habitats of the Flathead River and Flathead Lake (Deleray et al. 1999, Muhlfeld et al. 2008).

The only populations that have been monitored for 10 years or more with redd counts on the west side of GNP are Ole, Nyack, Park, and Quartz creeks. Bull trout redd counts in Ole Creek have been monitored annually by MFWP since 1980 (Weaver et al. 2006). While Ole and Quartz creeks are 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 2013 redd count for Ole Creek of 44 was higher than the long-term average of 28 redds. No statistically significant trend is evident over the long-term (full data set; tau-b = 0.13, $p > 0.05$), but over the last 10 years there has been a statistically significant positive trend in spawner abundance in Ole Creek (tau-b = 0.73, $p > 0.05$) (Figure 3, Table 1). The 2013 redd count for Park Creek was 9, well below long-term average of 15 (Figure 3, Table 1). Sufficient data does not exist to analyze short-term (10 year) trends for either Nyack or Park creeks due to the intermittency of the counts.

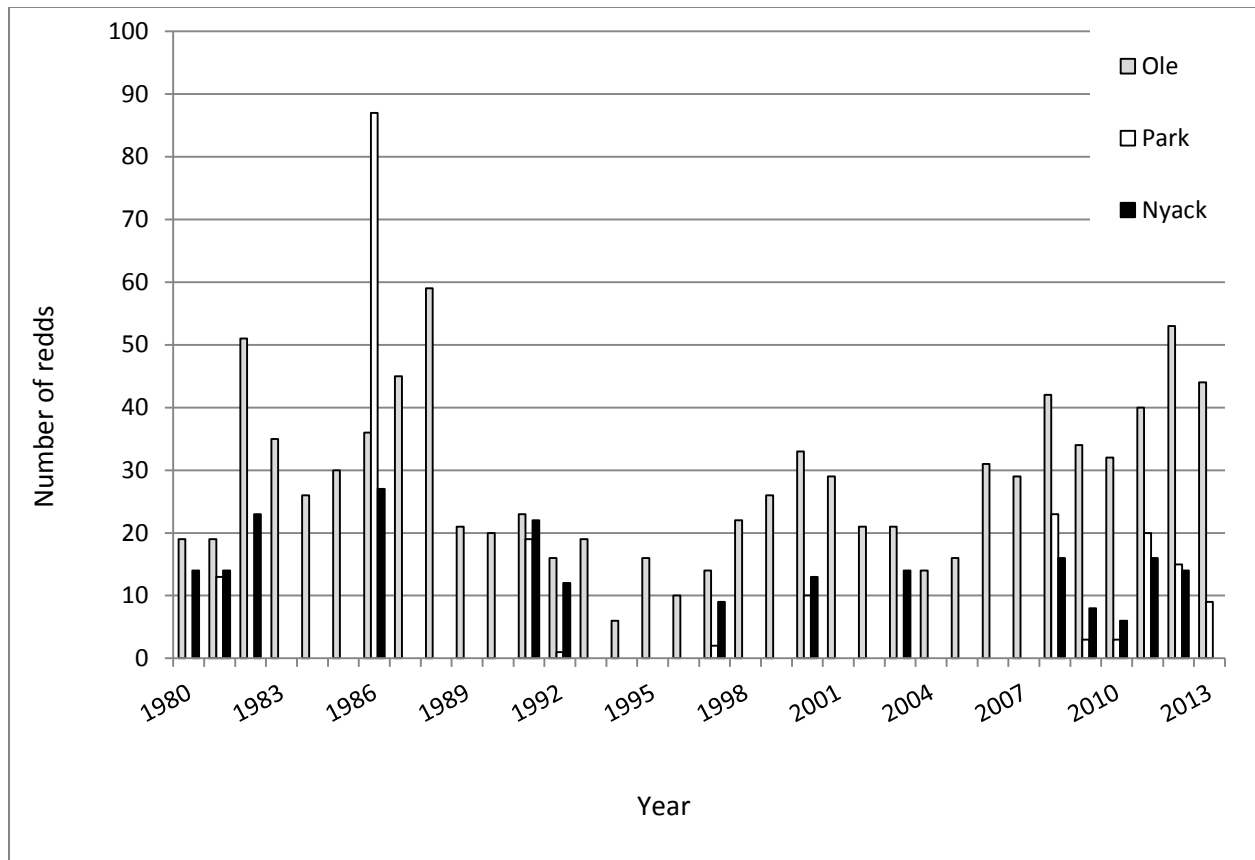


Figure 3. Bull trout redd counts conducted in Ole, Park, and Nyack creeks, Middle Fork Flathead River Drainage, Glacier National Park.

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 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; Downs et al. 2011). 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). In 2009, an experimental lake trout suppression program was initiated on Quartz Lake to preserve fish native populations, including bull trout and is showing promise in reducing adult lake trout abundance.

Redd counts on Quartz Creek, the primary spawning area for bull trout from Quartz Lake have not shown a statistically significant trend ($\tau\text{-}b = -0.09$, $p > 0.05$) since monitoring began, suggesting lake trout have yet to reach levels where they are impacting bull trout population viability (Figure 4, Table 1). However the 2013 redd count was the lowest count since the experimental lake trout suppression began. Continued monitoring will be necessary to determine if the low count is the result of natural variability in adult abundance, lake trout impacts, or potential gill netting/handling impacts.

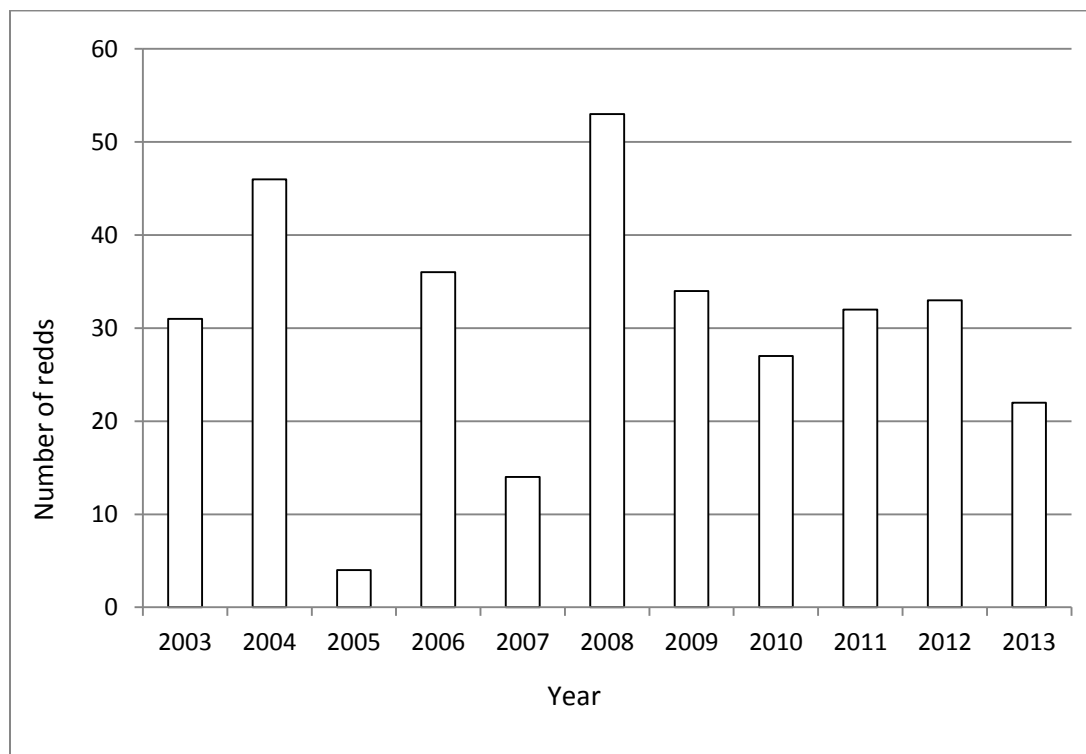


Figure 4. Bull trout redd counts in Quartz Creek, Glacier National Park, Montana.

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, the feasibility of population assessment using snorkeling should be evaluated in a variety of park streams. In addition, 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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