

Glacier National Park Fisheries Inventory and Monitoring

Program Report 2010-2012



Glacier National Park Fisheries Inventory and Monitoring Program Report 2010-2012





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April, 2013



Suggested citation:

Downs, C.C., M. Woody, and B. McKeon. 2013. Glacier National Park Fisheries Inventory and Monitoring Report, 2010--2012. National Park Service, Glacier National Park, West Glacier, Montana.

Front cover photo caption: Gill net sampling of Cosley Lake.

Inside photo captions (top to bottom): native mountain whitefish, fishery technician Melissa Brickl and GIS technician Kim Lindstrom retrieving a deep water benthic minnow trap in St. Mary Lake, fisheries technician Billy Brann collecting plankton samples from Lake McDonald for AIS monitoring, fisheries technicians Kieth Joint and Melvin Woody conducting a depletion population estimate for juvenile bull trout in Boulder Creek.

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Native Fish Population Monitoring

ABSTRACT

In 2009, Glacier National Park began development of a monitoring program for native salmonids inhabiting park streams. The intent of the program is to establish baseline abundance levels in key streams that will serve as useful benchmarks for monitoring changes in populations over time. We continued this work in 2010-2012, conducting sampling in 13 streams. Wild Creek (St. Mary drainage) had the highest density of age-1 and older westslope cutthroat trout (wct) *Oncorhynchus clarkii lewisi*, estimated at 24.4 fish/100m² in 2012. This is an unusually high estimate and was driven by a very strong age-1 year class. The next highest density estimate for 2010-2012 for wct in any streams was in Muir Creek (Middle Fork Flathead drainage) in 2010 (13.6 fish/100m²). Boulder Creek (St. Mary River drainage) had the highest density of age-1 and older bull trout, estimated at 6.6/100m². Many sites also supported tailed frogs *Ascaphus truei* and spotted frogs *Rana pretiosa*. Some culverts along the Camas Road may be important isolating mechanisms protecting genetically pure wct from downstream rainbow trout introgression. Similarly, Autumn Creek (tributary to Bear Creek) is protected by a large culvert under the railroad line. Management and conservation of native fish in the North and Middle forks Flathead River remains complicated due to the presence of migratory and resident populations of native salmonids, and expanding distribution of non-native fish species. Until recently, Midvale Creek supported the only known remaining genetically pure wct population in the Upper Missouri River drainage in the park. Recent genetic data show the population was recently compromised by rainbow trout and suppression actions were conducted in 2010 and 2011. Wild Creek (St. Mary tributary) supports one of the only remaining genetically pure wct populations on the east side of the park, yet maintains seasonal connectivity to the mainstem river. Evaluation of seasonal connectivity of Wild Creek to the St. Mary would provide valuable information when considering management and conservation needs of this population. In general, the largest threat currently facing migratory native fish species on the east side of the park is the unscreened St. Mary water diversion, part of the Milk River Irrigation Project.

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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 (ebt) <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.

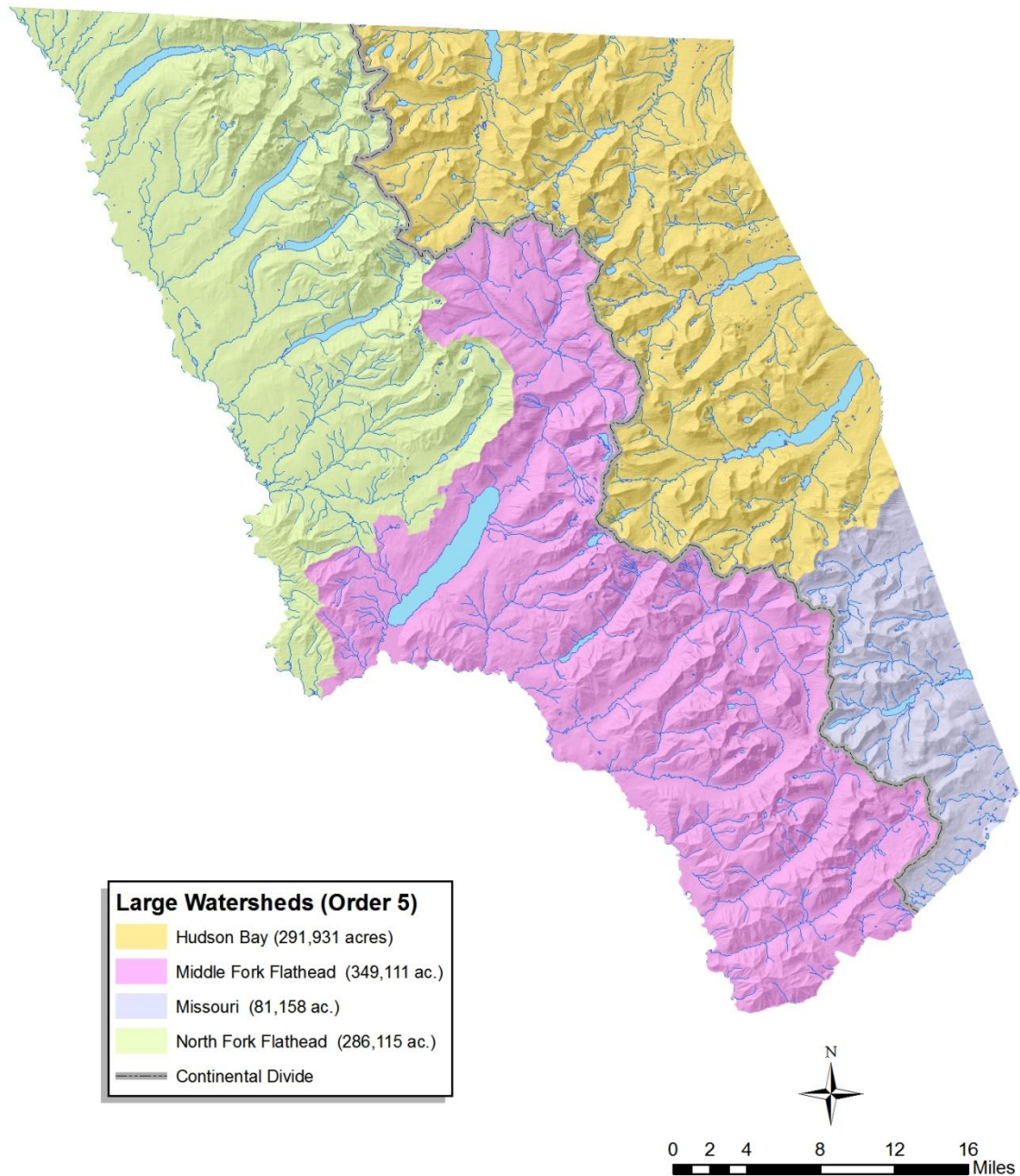


Figure 1. Major watersheds of Glacier National Park, Montana.

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 2010-2012 (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 sampling equipment 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.

We used the removal (depletion) method (Zippin 1958) to estimate the abundance and size structure of fish populations in the streams surveyed. 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.

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. In addition, in smaller, streams similar to those of headwater areas of GNP, wct fry may not even emerge from the gravel until mid-August (Scarnecchia and Bergersen 1986, Downs 1995). Scarnecchia and Bergersen (1986) indicated few cutthroat trout from headwater systems in Colorado exceeded lengths of 30-35 mm before they 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. Similarly, Polacek and James (2003) working in the Yakima River Basin, considered age-0 bull trout to be between 22-81mm total length (TL). Fishes of these respective 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.

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.

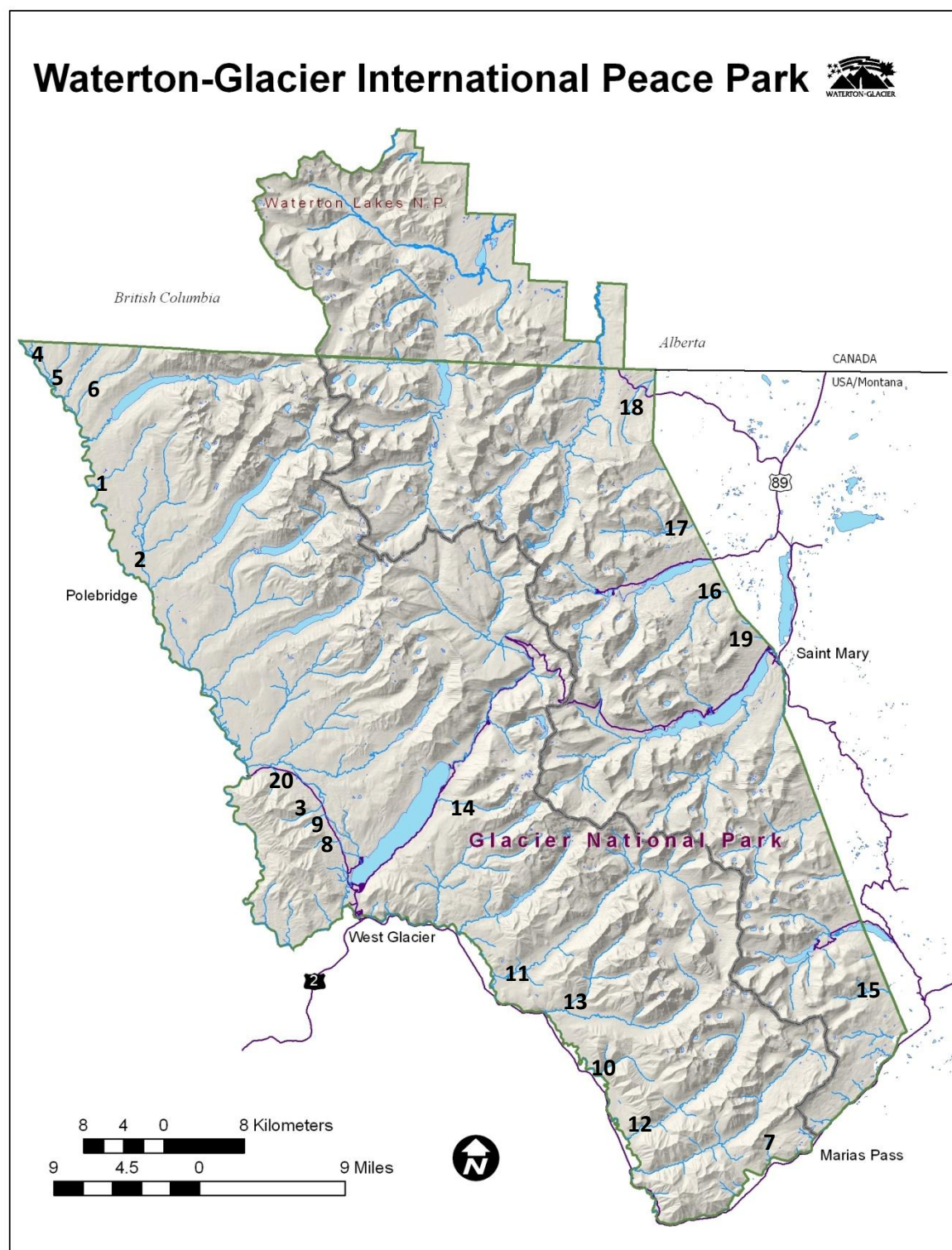


Figure 2. 2009-2012 stream sampling sites for depletion population estimates, Glacier National Park, Montana. Site numbers from Table 3.

Table 3. Stream name, site location, and date sampled for 2010-2012 sampling sites.

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	8/3/09; 7/30/10	138
	Ford (2)	11U 0694157N, 5417724W	7/20/09	93
		11U 0693673N, 5416862W	7/20/10	96
	(Top of reach)	11U 0693774N, 5416977W	8/2/11; 7/23/12	120
	McGhee (3)	11U 0718256N, 5386976W	7/10/09	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	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	90
	Fish (9)	11U 0718665N, 5386000W	7/10/09; 7/18/10; 7/19/11; 7/19/12	112
	Muir (10)	12U 0303790N, 5360495W	7/23/09; 8/4/10; 8/16/11; 8/9/12	80
Missouri	Midvale (15)	12U 0332437N, 5368436W	7/22/09	
		12U 0330666N, 5368622W*	7/28/10	
St. Mary (Hudson Bay)	Boulder (16)	12U 0314860N, 5408095W	8/12/09; 8/18/11	131
	Lee (18)	12U 0307481N, 5428586W	7/21/09; 8/10/11; 8/15/12	125
	Wild (19)	12U 0319740N, 5403793W	7/15/09;	128
		12U 0319740N, 5403793W	8/9/11; 8/13/12	95

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

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.

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.

This report represents a summary of data collected during 2010-2012 field seasons. We compared these data with data collected in 2009 (Downs et al. 2011) to provide a longer-term perspective on the sampling.

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). 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.

Previous voltage (700v) was increased to 800v for sampling in 2010 while maintaining a 30 hertz frequency, with a 9% duty cycle and a 3 ms pulse width. Species observed in Akokala Creek included wct, mwf, and sculpin. The density estimate for age-1 and older wct was 0.38 wct/100m² (Table 4). First pass CPUE₁ for the same fish was estimated at 4.7 fish/hr and 0.28 fish/100m². Average condition factor (K) was estimated at 0.97 (SD = 0.057) (n = 3) while mean total length of age-1 and older wct was 105.3 mm. No age-0 wct were captured during sampling. We also collected genetic samples from wct for future analysis.

Table 4. Mean length (TL; mm), mean weight (g), standard deviation (SD), and sample size (n) of age-1 and older wct, as well as mean length and mean weight of all individuals of other species captured in Akokala Creek in 2010. Length range represents all fish captured.

Species	Mean length (SD) (n)	Length range (all individuals captured)	Mean weight (SD) (n)	Fulton Condition Factor (K)
wct	105.3 (15.95) (3)	92-123	11.7 (4.7) (3)	1.1
mwf	56 (0) (1)	56	3 (0) (1)	1.7
sculpin	95 (0) (1)	95	9 (0) (1)	1.0

The length-frequency histogram of wct from the reach of Akokala Creek sampled is truncated at 120mm (Figure 3). This sampling reach is located approximately one kilometer upstream of the confluence with the North Fork Flathead River, in what is presumed to be migratory wct spawning and rearing habitat. Our data is consistent with this presumption. We would expect to find multiple year-classes of wct, including adult fish in the size range 150-200 mm (Downs 1995) if we were sampling a resident wct population, which we did not.

Although a total of four wct were captured in 2010 one escaped prior to length and weight measurement. Consequently we are unable to classify it as either young-of-the-year (age-0) or age-1 and older wct, therefore the estimated abundance of age-1 or older wct is 3. Because fish were captured on the first pass only during 2010 sampling, we report the total number captured as the population estimate without confidence intervals. In 2009 the same 148 m-long section was estimated to have 9 (95% CI: 7-11) wct (Figure 4).

Tailed frogs (*Ascaphus truei*) were present, but rare. Gravel was estimated to be the dominant substrate type followed by sand in the reach. Water temperature at the time of sampling was 13°C. Dissolved oxygen was measured at 8.41 mg/L, pH at 6.95, and conductivity at 56.5µS. Mean wetted width remained 7.7m. Mean channel slope was assumed to be unchanged from sampling in 2009 at 1.7%.

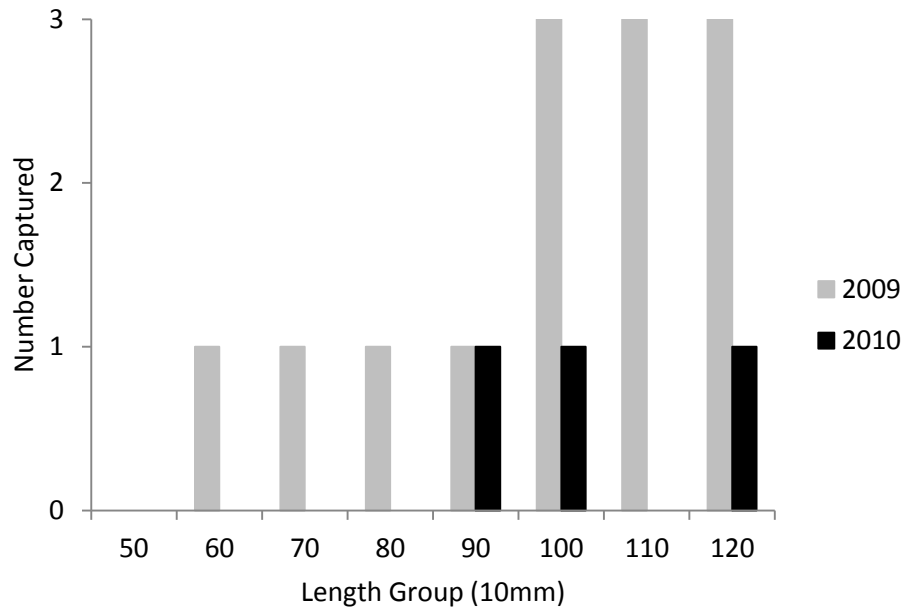


Figure 3. Length-frequency histogram for wct captured in Akokala Creek, GNP, 2009-2010.

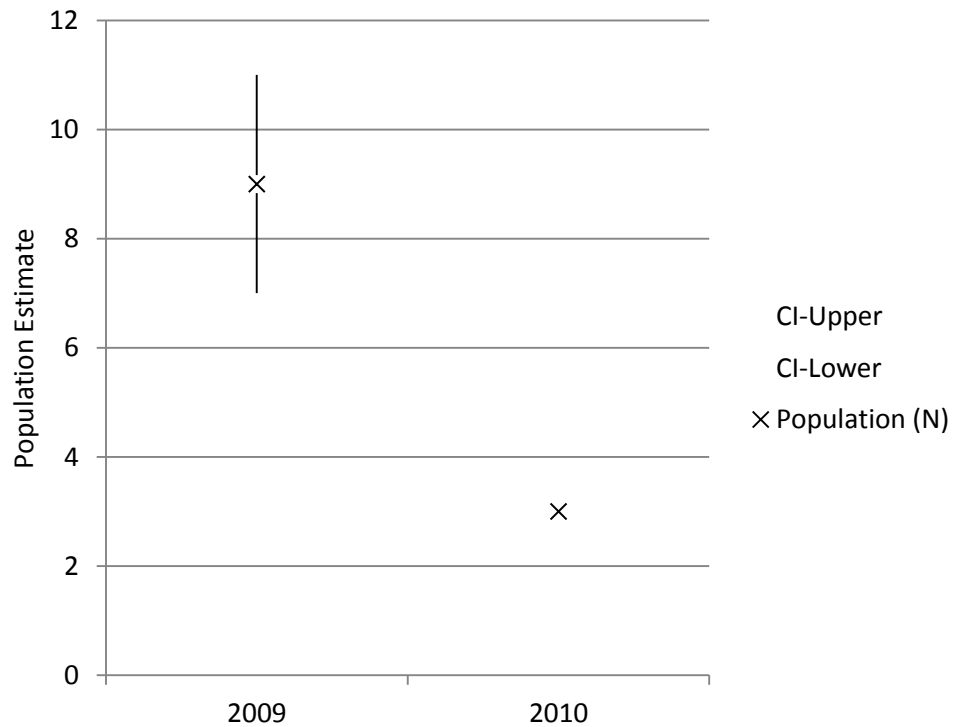


Figure 4. Estimated population abundance of age-1 and older wct in Akokala Creek, GNP, 2009-2010.

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 65 μ S, an average channel slope of 1.4%, and bed material composition of 30% rubble, 30% gravel, 30% fines, and 10% boulder. Channel debris was noted as “low” in both studies. We found the same species in this reach of Akokala Creek as was found in the 1980 study, however our density estimate was lower for wct. Using snorkeling, the authors estimated 1.8 age-1 and older wct/100m², considerably higher than our estimates of 0.79/100 m² and 0.38/100 m² derived using depletion population estimates in 2009 and 2010, respectively. Muhlfeld et al. (2009a) estimated a density of 1.0 age-1 and older wct/100m² in Akokala Creek, using multiple pass depletion methods, similar to ours. It should be noted that these estimates were not derived using the same sampling methodology (single pass snorkeling compared to a multiple pass depletion estimates conducted downstream into a block net) or in precisely the same locations, but in combination the more recent data suggest lower densities of age-1 and older wct than were previously estimated. Continuation of the depletion population estimate is recommended for long-term monitoring because of the three methods previously used to estimate fish density in Akokala Creek, it is the least sensitive to changes in technique and personnel.

Ford Creek

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

In an effort to continue monitoring the status of wct and rainbow trout hybridization, genetic samples were collected from fish captured during sampling in 2010 and 2011. Location of stream sections varied by year, but were located in the general vicinity of the Kintla Lake Road crossing (Table 3). We employed 650 volts at 30 hertz, with a 12% duty cycle and a 3 ms pulse width to capture fish in Ford Creek in 2010. Maintaining all other previous settings, voltage was increased to 665 volts in 2011 and 2012.

In 2010 bull trout, as well as wct and sculpin, were observed in Ford Creek while only sculpin and wct were observed in 2011 and 2012. The single bull trout observed in 2010 was 171 mm long and weighed 45 g. In 2010 a total of five age-0 (TL<45mm) wct were captured with an average length of 31.6mm whereas only four age-0 wct were observed in 2012 (mean 29 mm). No wct less than 55mm total length were captured in 2011 (Table 5, Figure 5). Average condition factor (K) ranged from 0.92-1.1.

In 2010 no fish were captured on the second pass therefore we used the total number captured on the first pass as the estimate of population size (Table 6). Density estimates ranged from a low of 2.9 to a high of 8.0 wct/100m². Although estimated wct abundance and density was higher in more recent sampling years, it should be noted that a different and slightly longer stream reach was sampled in 2011 and 2012. Similarly, CPUE of age-1 and older wct was different in all three sections over the four years sampled, ranging from 26.6 fish/hr in 2010 to 45.0 fish/hr in 2012.

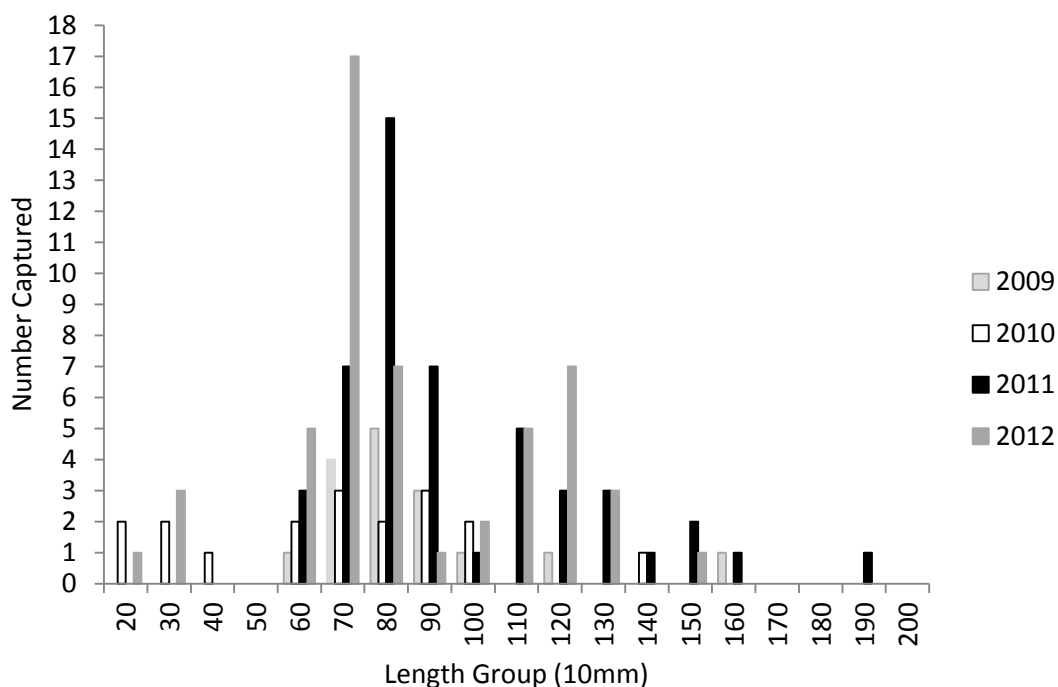


Figure 5. Length-frequency histogram for wct captured in Ford Creek, GNP, 2009-2012.

Table 5. Mean total length (TL; mm), mean weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older bull trout and wct; mean total length and mean weight of all individuals of other species captured in Ford Creek. Length range represents all fish captured.

Species		Mean Length (SD) (n)	Length Range (all individuals captured)	Mean Weight (SD) (n)	Fulton Condition Factor (K)
BLT					
	2010	171(0)(1)	171	45(0)(1)	0.90
	2011	---	---	---	---
	2012	---	---	---	---
WCT					
	2009	82.3(24.8)(16)	58-160	7.3(9.5)(16)	1.0
	2010	88.6(20.4)(13)	27-142	8.6(7.3)(13)	1.1
	2011	100.7(28.2)(49)	65-193	12.9(12.9)(49)	1.0
	2012	93.5(25.0)(48)	22-158	9.3(8.1)(47)	0.92
Sculpin					
	2010	64.8(21.4)(5)	40-87	6(1.7)(3)	1.15
	2011	74.6(25.6)(18)	40-115	6.9(6.4)(18)	1.27
	2012	55.7(17.4)(28)	36-112	2.7(4.0)(28)	1.11

Table 6. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct and bull trout captured in Ford Creek, GNP, 2009 - 2012. Section lengths/locations varied between some years (Table 1).

Species	Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE ₁ (fish/100m ²)	CPUE ₁ (fish/hr)
BLT					
	2010	1	0.2	0.2	2.1
WCT					
	2009	17(12-22)	4.8	3.1	28.7
	2010	13	2.9	2.9	26.6
	2011	49(47-51)	8.0	6.1	39.2
	2012	48(46-50)	7.7	4.8	45.0

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. They also estimated a conductivity of 105µS and a stream gradient of 3.4%. 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. These estimates are not directly comparable because the precise locations and methods varied.

Both adult and juvenile tailed frogs were observed in Ford Creek in 2010 but none were noted in 2011 or 2012. Cobble was estimated to be the dominant substrate type followed by gravel in all sampling reaches. Water temperature at the time of samplings ranged from 13-17 °C (Table 7). Muhlfeld et al. (2009a) reported a maximum summer water temperature of 15.9C.

Table 7. Stream attributes and water chemistry data for Ford Creek, GNP, 2009 - 2012.

	2009	2010	2011	2012
Date Sampled	7/20/09	7/20/10	8/2/11	7/23/12
Temperature (°C)	13	13	15	16.6
Dissolved Oxygen (mg/L)	8.82	9.63	5.55	N/A
pH	7.58	8.15	N/A	N/A
Conductivity (µS)	81.8	81.5	82.7	91.4
Average Slope (%)	4.0	1.2	1.7	1.7
Average wetted width (m)	3.8	4.7	5.3	5.2
Sample Reach Length (m)	93	96	115	120

McGhee Creek

McGhee Creek is a third order tributary to Camas Creek in the N. Fk. Flathead River drainage. It is historical wct habitat, but is threatened by hybridization with rbt. Muhlfeld et al. (2009a) documented hybridization between rbt and wct in Dutch Creek, a large tributary to Camas Creek. Genetic samples collected from lower McGhee Creek in 2008 indicated hybridization was occurring (MFWP, unpublished data). There are no known fish passage barriers in the drainage that would preclude advancing rbt. The

culvert under the Camas Road may preclude upstream passage to the headwater reaches of McGhee Creek due to its length and drop out of the pipe, but this remains uncertain.

Although the lower limit of all sampling conducted on McGhee Creek (2009-2012) remained the same (BR: 11U 0718256N 5386976W) the upper limits fluctuated resulting in varying section lengths (Table 10). Densities of these fish ranged from 1.1- to 5.3 wct/100m². and a first pass CPUE of 3.7 fish/100m² (25.4 fish/hr) (Table 8). Mean fish length ranged from 122 to 162 mm (Table 9). Condition factors at or over one suggest good feeding/growth conditions.

Table 8. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured on McGhee Creek, GNP, 2009 -2012. Section lengths/locations varied between some years (Table 1).

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
2009	13	5.3	5.3	39.1
2010	7(6-8)	4.4	3.7	25.4
2012	8(7-9)	1.1	0.98	12.0

Table 9. 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 McGhee Creek, GNP, 2009 – 2012. Length range represents all fish captured.

Year	Year	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT					
	2009	121.9(33.7)(13)	80-180	25.9(23.0)(13)	1.1
	2010	134.4(23.7)(7)	107-174	26.9(15.4)(7)	1.0
	2012	161.6(22.8)(8)	53-185	51(19.7)(8)	1.2

Initial sampling in 2011 in McGhee Creek failed to capture any fish. Due to time constraints and park wide sampling objectives no attempt was made to resample McGhee in 2011. Subsequent sampling in 2012 for the 50 m reach also failed to capture any fish. We reassessed the sample reach length in 2012 and increased the total sampling length from the 50m section to 154 m beginning at the same downstream location (Table 3). Fish were captured in the longer reach, resulting in a population estimate of only 8 wct with a density of 1.12 fish/100m². First pass CPUE for age-1 and older wct was calculated to be 12.02 fish/hr or 0.98 fish/100m². Although the 2009 stream reach represented a substantially smaller sampling area it had the largest number of wct present (Table 8). The longer section sampled in 2012 (154m) had relatively few fish (8) but contained more larger fish, versus observations for the two smaller sampling reaches, in 2009 and 2010 (Figure 6). Poor recruitment from 2010 and 2011 spawning years may be responsible for the lack of smaller fish in the section in 2012. Future sampling will likely focus on the longer stream reach so that comparisons can be made for the same sampling area.

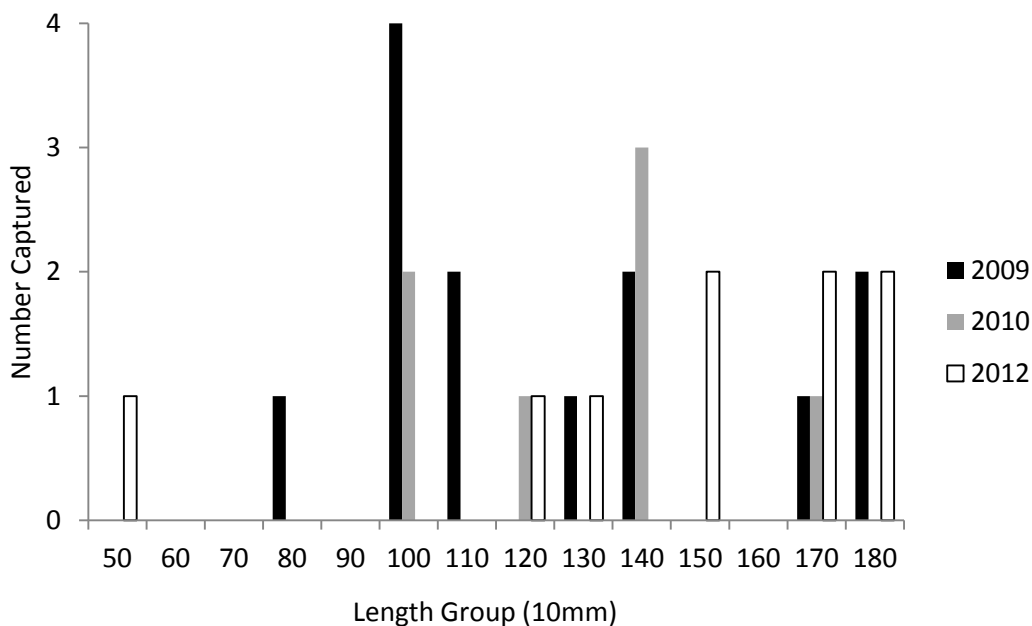


Figure 6. Length-frequency histogram for wct captured in McGhee Creek, GNP, 2009, 2010, 2012.

We collected genetic samples for wct in 2010 for future analysis. We employed 500 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in McGhee Creek in 2010. Except for voltage, which was 600v, all other settings remained the same in 2012. The presence of tailed frogs was not noted in 2010 nor 2012. Cobble was estimated to be the dominant substrate type within the reach followed by gravel. Water chemistry and physical habitat measurements varied by year (Table 10).

Table 10. Stream attributes and water chemistry data for McGhee Creek, GNP, 2009, 2010 and 2012.

	2009	2010	2012
Temperature (°C)	8	7	8.4
Dissolved Oxygen (mg/L)	10.19	9.55	N/A
pH	5.3	7.92	N/A
Conductivity (µS)	111.6	123.1	114.5
Sample Reach Length (m)	75.7	49.5	143.7
Average Slope (%)	2.5	2.6	1.4
Average wetted width (m)	3.25	N/A	4.97

Spruce Creek

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

Conducted in 2010, the most recent sampling of Spruce Creek yielded capture of both wct and sculpins. No age-0 wct were captured in the 122.5 m-long section. Total abundance of age-1 and older wct was estimated at 9 (95% CI: 8-10) and density was estimated at 1.89 wct/100m². Age-1 and older wct first pass CPUE was estimated at 11.37 fish/hr and 1.68 fish/100m². Average Fulton Condition Factor (K) was estimated at 1.0 for age-1 and older wct (SD = 0.04) (n = 9). Average length for the same group was 134.2 mm (Table 11). In 2009, Downs et al. (2011) estimated the total abundance of age-1 and older wct for the same stream reach at 14, averaging 183.3 mm in length with a density of 3.1 wct/100m². Size structure suggests a resident wct population (Figure 7). Westslope cutthroat trout genetics samples were collected for future analysis.

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 and all sculpins captured on Spruce Creek, GNP, 2009 – 2012. Length range represents all fish captured.

Species	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT				
2009	183.3(20.1)(14)	107-193	27.6(13.2)(14)	1.0
2010	134.2(34.1)(9)	80-180	28.8(18.87.3)(9)	1.0
Sculpin				
2010	77.5(24.7)(2)	60-95	2(4.2)(2)	1.2

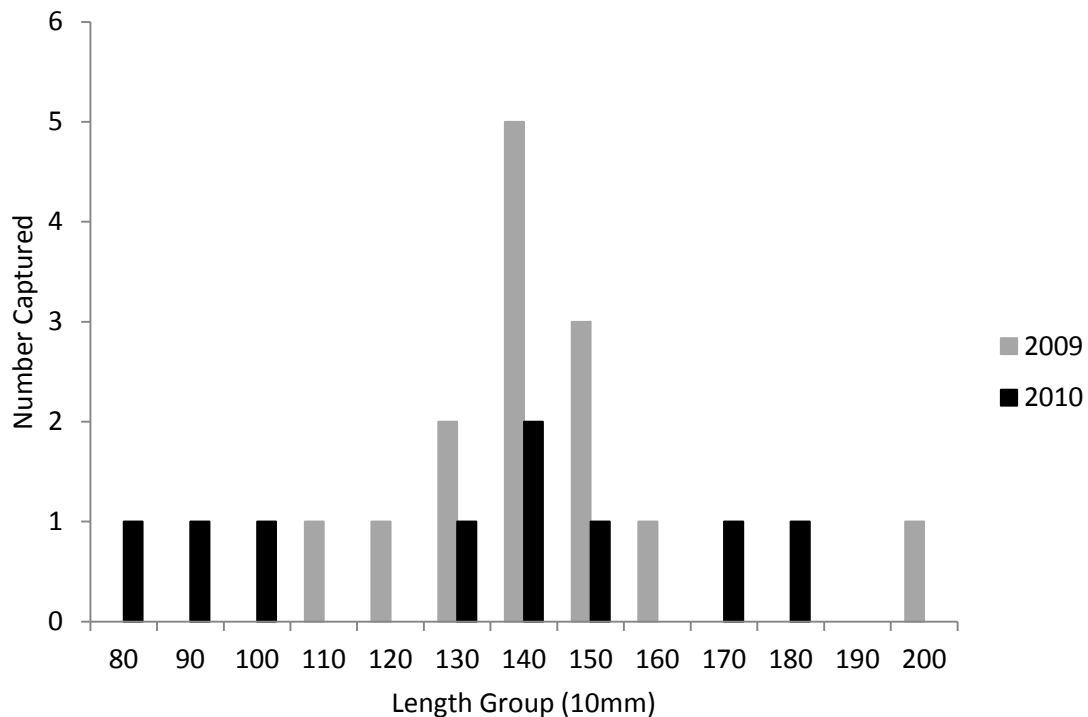


Figure 7. Length-frequency histogram for wct captured in Spruce Creek, GNP, 2009-2010.

For comparison Read et al. (1983) estimated 3.6 age-1 and older wct/100m² using snorkeling in the lower reaches of Spruce Creek. This estimate is not directly comparable to our sampling nor that conducted in 2009, as the sites were not duplicated and methods of enumeration differed, but they represent a range of wct densities observed in Spruce Creek.

We employed 600 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Spruce Creek. Cobble was estimated to be the dominant substrate type within the reach followed by sand. Water chemistry and physical habitat measurements varied by year (Table 12). No amphibians were observed in Spruce Creek.

Table 12. Stream attributes and water chemistry data for Spruce Creek, GNP, 2009-2010.

	2009	2010
Temperature (°C)	12	10
Dissolved Oxygen (mg/L)	7.8	8.9
pH	8.2	7.3
Conductivity (µS)	95	69.8
Sample Reach Length (m)	122.5	100
Average Slope (%)	2.7	1.3
Average wetted width (m)	3.7	4.8

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 crosses under the road flowing in northerly direction. (hereafter referred to as No-Name Creek). 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. To our knowledge no previous sampling has been conducted on No-Name Creek.

We conducted presence-absence sampling in July 2011 just downstream from the outlet of the culvert running under the Camas Road and determined that wct were present. A 90m section of stream with a lower reach beginning approximately 100m upstream from the culvert was established for monitoring. In 2011 we captured fish representing a variety of age classes, ranging from age-0 to age-3+. Based on the length frequency histogram and the downstream presence of the road culvert, we concluded this is a resident population (Figure 8). This is an important discovery, as genetically pure westslope cutthroat trout populations are threatened by hybridization in the Camas Creek drainage (as well as the larger N. Fk. Flathead River drainage).

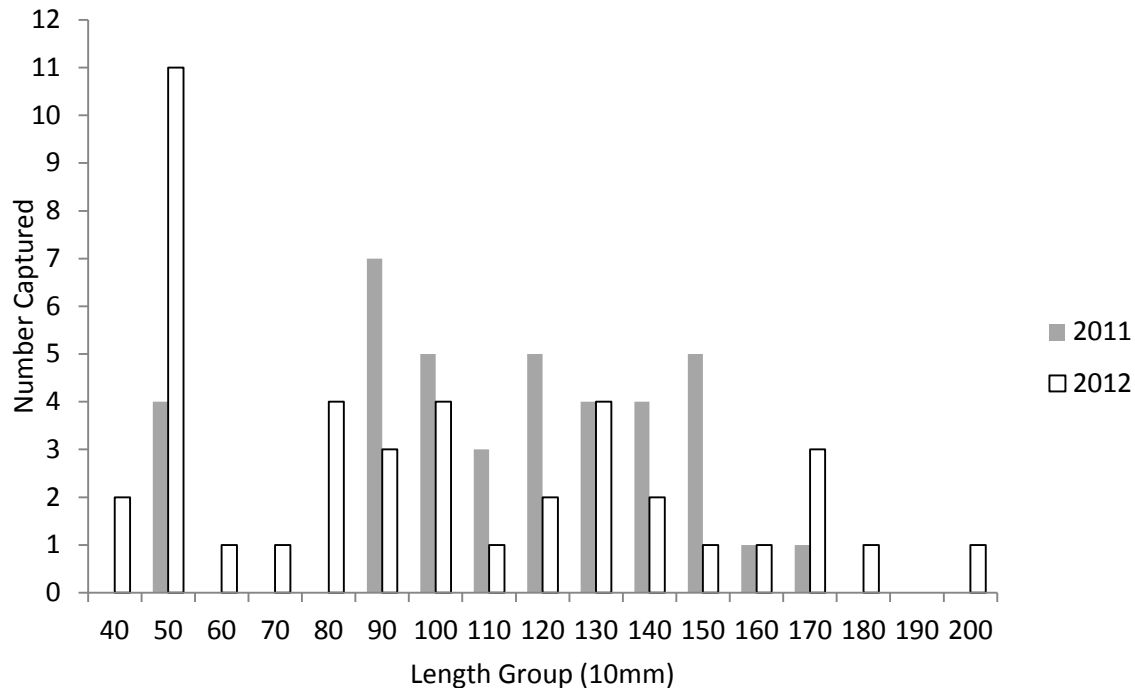


Figure 8. Length-frequency histogram for wct captured in No Name Creek, GNP, 2011-2012.

The mean length and weight for age-1 and older wct in No Name Creek sampled in 2012 was similar to that found in 2011 (Table 13). Similarly, Fulton Condition Factor (K) ranged from 1.1 in 2011 to 1.0 in 2012. Estimated abundance of wct in 2011 was 39 fish while in 2012 it was 35 (Table 14). The population density (wct/100m²) compared to other park streams remained consistently high in both 2011 (12.5) and 2012 (11.5). Age-1 and older catch per unit effort for the first pass was also higher than most other park streams, calculated at 8.0 wct/100m² in 2011 and 8.8 wct/100m² in 2012.

Table 13. 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, 2011-2012. Length range represents all fish captured.

Species		Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT					
	2011	118.6(29.11)(38)	50-174	21.2(14.16)(38)	1.1
	2012	113.7(41.71)(34)	46-201	22.5(23.06)(34)	1.0

Table 14. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured on No Name Creek, GNP, 2011 – 2012.

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE ₁ (fish/100m ²)	CPUE ₁ (fish/hr)
2011	39(36-42)	12.5	8.0	68.3
2012	35(31-39)	11.5	8.8	69.1

No Name Creek was determined to have a dominant substrate of cobble and a subdominant gravel substrate. The sampling reach length was approximately 90m while average wetted width was 3.5m. Water temperature and conductivity for 2011 was 9°C and 130 µS, respectively, and 10.4°C and 122.7 µS for 2012. Due to instrument malfunction, dissolved oxygen was not available for either sample. In 2011 electrofishing settings were 600 volts at 30 Hertz on a 9% duty cycle and a 3 millisecond pulse width. All settings remained the same in 2012 with the exception of voltage (550v). Tailed frogs were note as abundant in both 2011 and 2012.

Starvation Creek

Starvation Creek is a third order tributary to the N. Fk. Flathead River (Read et al. 1982). It flows from Canada into the park prior to entering the N. Fk. Flathead River approximately 6 km south of the Canadian border. 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, wct, mountain whitefish, and sculpin in Starvation Creek (Read et al. 1982, Downs et al. 2011).

A limited field season prevented electrofishing of Starvation Creek from 2010-2012. On July, 27 2012 hook and line sampling was conducted on a lower reach of the stream. Sampling began at the ford of the Kishenehn Trail and Starvation Creek about 50 meters upstream from its confluence with the N. Fk. Flathead River, extending upstream approximately one kilometer. Using a number 12 dry-fly with 4 weight line, two individuals moved upstream alternating casting with equal effort. Collectively both fishermen captured approximately 60-65 westslope cutthroat trout within 126 minutes. All fish were released.

A capture tally was maintained by one individual reflecting wct size to the nearest inch (Table 15). Of the number recorded a majority of fish were within the six inch class (140-165mm). The maximum size observed was 10.5 in (267mm). Considering a cumulative effort of approximately 65 wct, CPUE can be estimated around 32wct/hr. Downs et al. (2011) estimated a density of age-1 and older wct to be 11.2 fish /100m² in Starvation Creek. Taken together, these data indicate a robust population of wct in Starvation Creek.

Table 15. Angler tally by size class (rounded to the nearest inch) for hook-and-line sampling on Starvation Creek, GNP, 2012.

	Size class (rounded to the nearest inch, mm)					
	5in (4.5-5.4)	6 in (5.5-6.4)	7 in (6.5-7.4)	8 in (7.5-8.4)	9 in (8.5-9.4)	10in (9.5-10.4)
# Captured	3	13	11	2	1	1

Middle Fork Flathead River Drainage

Autumn Creek

Autumn Creek is a third order tributary to Bear Creek in the Middle Fork of the Flathead River Drainage. Autumn Creek is located near Marias Pass, and is isolated from Bear Creek by a perched culvert under the Burlington Northern Railroad that parallels Bear Creek to its confluence with the Middle Fork Flathead River (Downs et al. 2011). The culvert is approximately 70 m long, 4.5 m wide, and 5 m tall. During 2009 sampling the culvert was perched approximately 2.2 m above the water surface of Autumn Creek at the outlet, and the drop pool maximum depth was 50 cm. Based on these measurements, the culvert was determined to be a year-around fish passage barrier.

Westslope cutthroat trout were the only fish species observed in Autumn Creek from 2009-2012. In both 2009 (Downs et al 2011) and 2010, fifty percent of the age-1 and older individuals were presumed sexually mature ($TL \geq 150\text{mm}$) (Downs et al. 1997). For 2011, the proportion of mature fish sampled decreased slightly to 45% however by 2012 sexually mature fish only comprised 11% of the age-1 and older sampled individuals (Figure 9). The average length of age-1 and older wct also declined over the three years from 148.6mm in 2010 to 138.1mm in 2011 and 95.3mm in 2012 (Table 16). The average Fulton Condition Factor (K) for wct fluctuated for all three samples (Table 16).

We estimated the total abundance of age-1 and older wct in the 110 m-long section for each year sampled. Previous sampling in 2009 resulted in an estimated population abundance of 6 (Downs et al. 2011). The estimate increased to 10 in 2010 but had more than quadrupled by 2011 (45). Estimated abundance from sampling remained relatively the same from 2011 into 2012 (Table 17, Figure 10). Voltage settings on the electrofisher was increased from 550v in 2009 to 600v in 2010, and to 665v for 2011 and 2012. It is possible that increased sampling efficiency resulting from the changes in voltage could account for the higher population estimates in later years, but within years capture efficiency was high.

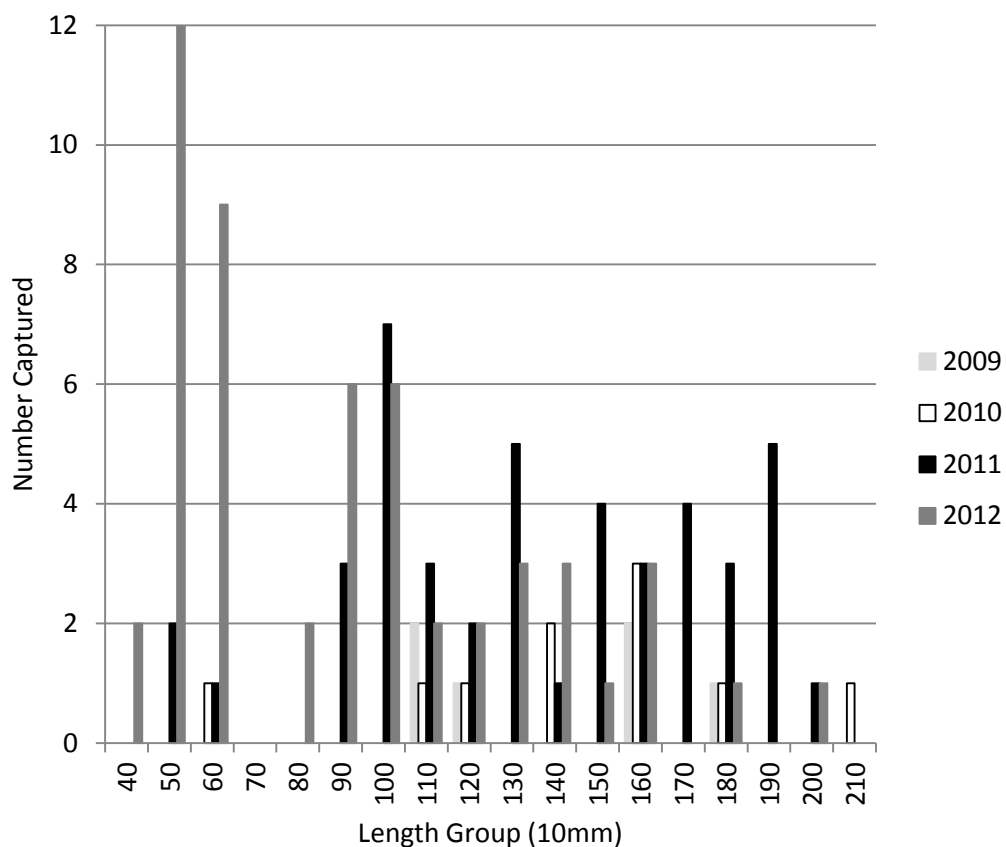


Figure 9. Length-frequency histogram for wct captured in Autumn Creek, GNP, 2009-2012.

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 Autumn Creek, GNP, 2009 – 2012. Length range represents all fish captured.

Species	Year	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT					
	2009	145.2(30.4)(6)	110-186	41.5(22.2)(6)	1.04
	2010	148.6(41.4)(10)	67-218	41.9(24.1)(9)	1.06
	2011	138.1(41.1)(44)	50-200	29.4(21.5)(44)	0.91
	2012	95.3(41.2)(53)	47-207	13.7(18.1)(53)	0.98

Table 17. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured in Autumn Creek, GNP, 2009 – 2012.

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
2009	6(5-7)	1.4	1.2	15.2
2010	10(9-11)	2.4	2.1	21.3
2011	47(40-54)	8.4	6.1	66.9
2012	55(50-60)	12.2	7.8	55.1

Based on the population size structure and the presence of a physical barrier to upstream movement, we conclude this population is resident. We collected genetic samples from wct for future analysis. We employed 600 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Autumn Creek during the 2010 sampling. Voltage was increased to 665 volts for the 2011 and 2012 sampling events.

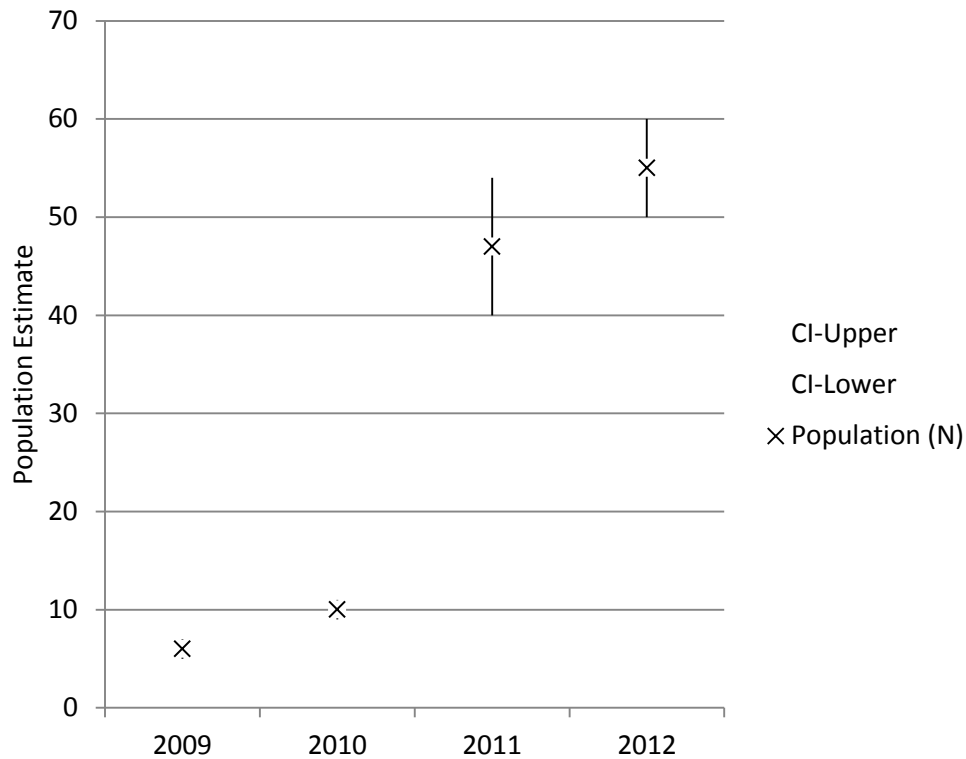


Figure 10. Estimated population abundance of age-1 and older wct in Autumn Creek, GNP, 2009-2012.

Both adult and juvenile tailed frogs were present and noted as common. Cobble was estimated to be the dominant substrate type followed by boulder. Water chemistry and physical habitat measurements varied by year (Table 18).

Table 18. Stream attributes and water chemistry data for Autumn Creek, GNP, 2009 – 2012.

	2009	2010	2011	2012
Temperature (C)	12.5	7.5	8.5	12.4
Dissolved Oxygen (mg/L)	7.96	7.61	13.3	NA
pH	6.05	7.25	NA	NA
Conductivity (μS)	115.7	101.6	62.9	119
Average Slope (%)	5.0	2.7	2.7	2.7
Average wetted width (m)	3.9	4.2	5.2	4.0

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 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.

We captured only wct in Fern Creek. The average length of age-1 and older wct as well as length range was relatively constant over the four year study period suggesting stable growth, recruitment and mortality rates (Table 19). Fish condition was relatively high across all years.

Table 19. 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, 2009 – 2012. Length range (mm) represents all fish captured.

Species	Year	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT					
	2009	122.5(30.7)(15)	74-179	24.2(18.1)(14)	1.11
	2010	110.8(38.7)(15)	66-177	20.8(19.8)(14)	1.05
	2011	119.9(16.1)(17)	96-154	20.5(9.2)(17)	1.13
	2012	103.5(50.8)(24)	52-182	25.3(23.1)(14)	0.97

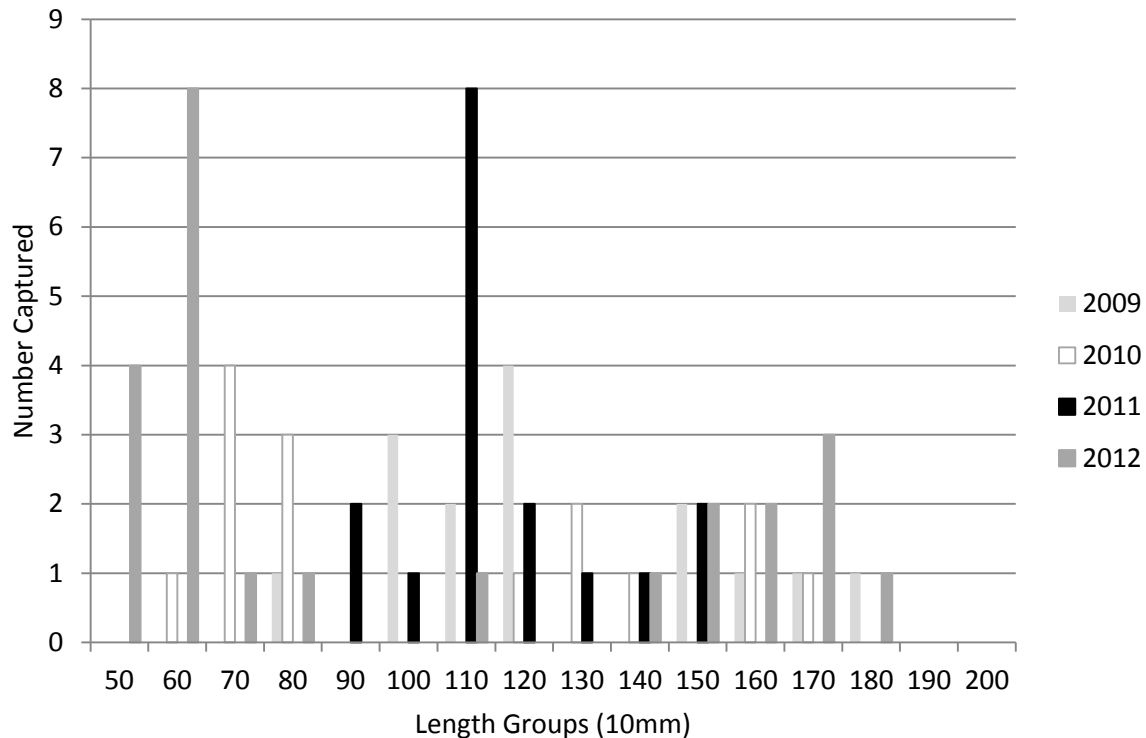


Figure 11. Length-frequency histogram for wct captured in Fern Creek, GNP, 2009-2012.

Abundance estimates remained fairly constant from 2009-2011, and then increased in 2012 (Figure 12). Density ranged from 3.8 to 7.1 fish/ 100m² over the same time period (Table 20). During our observations age-1 and older first pass catch per unit effort was highest (3.2 wct/100m², 25.5 wct/hr) in 2012.

Based on the size structure of the population, the location of the sampling site in the drainage, and the likelihood that the culvert under the Camas Road is at least a partial upstream barrier, it is likely this population is resident. We collected genetic samples from Fern Creek wct for future analysis. Electrofishing settings were 30 hertz, 9% duty cycle, and a 3 ms pulse width. Voltage settings ranged from 400 to 600 volts across all years.

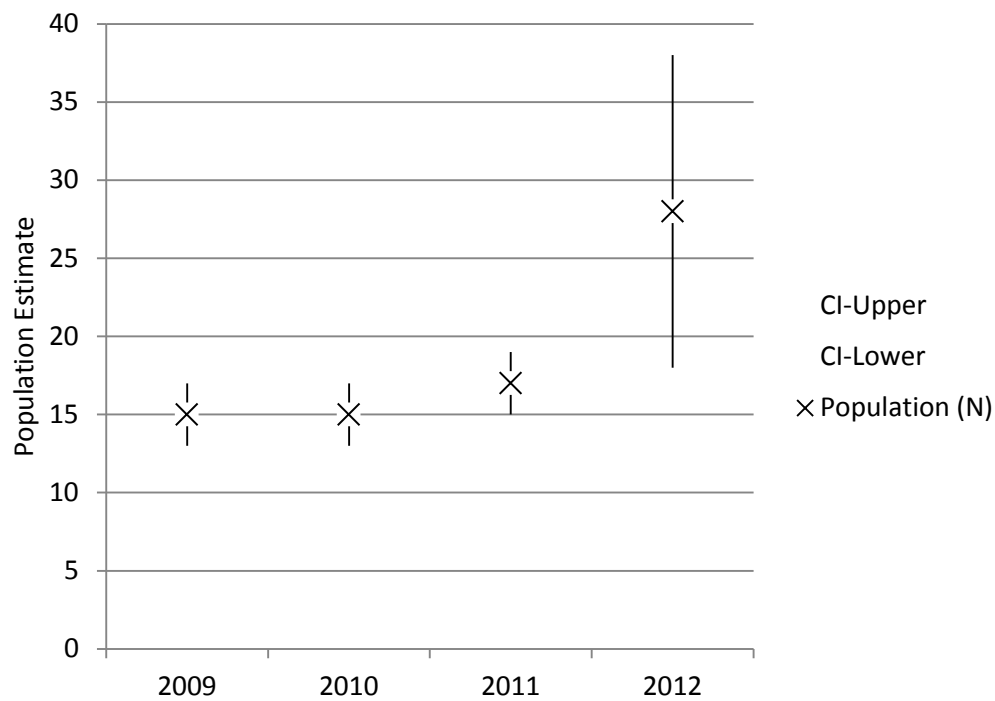


Figure 12. Estimated population abundance of age-1 and older wct in Fern Creek, GNP, 2009-2012

Table 20. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured in Fern Creek, GNP, 2009 – 2012.

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
2009	15(13-17)	4.3	2.8	29.7
2010	15(13-17)	5.3	2.8	16.3
2011	17(15-19)	3.8	2.9	17.3
2012	28(18-38)	8.2	3.2	25.5

Both adult and juvenile tailed frogs were present in all years sampled. Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Physical and chemical measurements varied across sampling years (Table 21).

Table 21. Stream attributes and water chemistry data for Fern Creek, GNP, 2009 – 2012.

	2009	2010	2011	2012
Temperature (C)	7.5	9	9	13.8
Dissolved Oxygen (mg/L)	8.3	10.07	14.5	NA
pH	5.98	7.96	-	NA
Conductivity (μ S)	101.5	85	85	90.6
Average Slope (%)	5.4	2.6	2.6	2.6
Average wetted width (m)	3.93	4.1	4.5	3.8
Reach length (m)	89.6	70	100	89.5

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. Westslope cutthroat trout genetic status has not been assessed in Fish Creek.

Age-1 and older wct (TL $\geq$ 45mm) were the only fishes captured in the 112 m long sample section of Fish Creek. Estimated abundance of these fish in 2009 was 32 (Downs et al. 2011) while our estimates from 2010 through 2012 ranged between 24-27 wct (Table 22; Figure 13). Fish density decreased marginally from 2009 to that observed in 2010.

Table 22. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured on Fish Creek, GNP, 2009 – 2012.

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
2009	32(28-36)	12.2	9.2	89.7
2010	24(20-28)	5.3	2.8	16.3
2011	27(25-29)	7.6	5.0	27.7
2012	25(23-27)	7.2	5.8	51.1

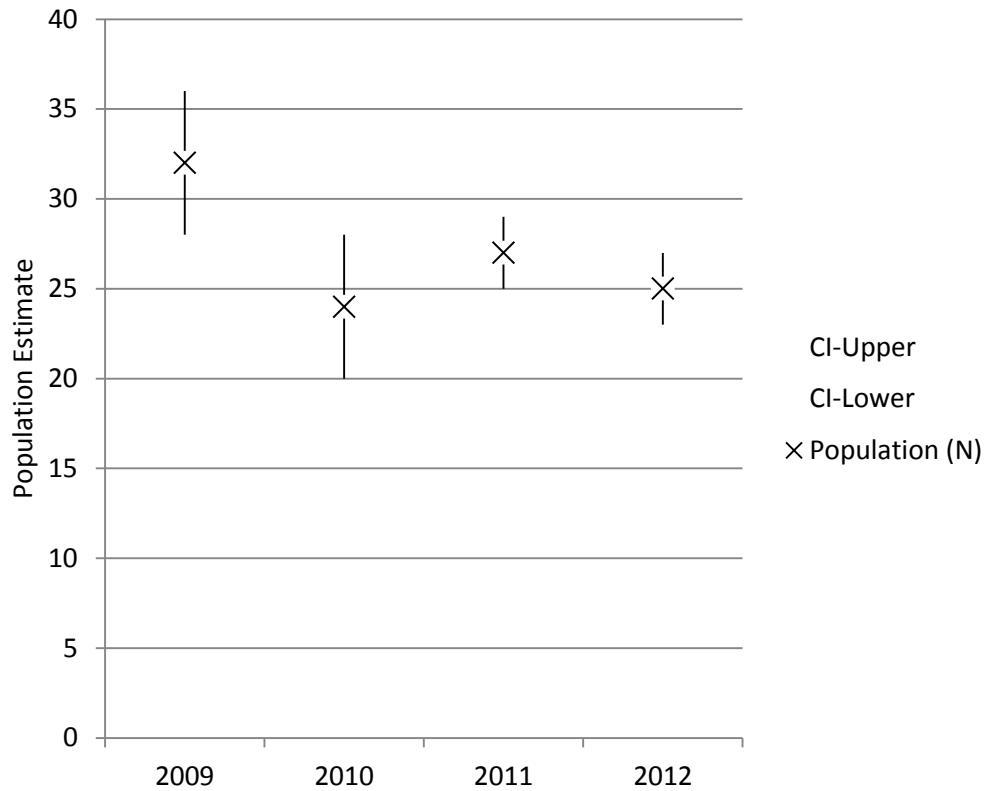


Figure 13. Estimated population abundance of age-1 and older wct in Fish Creek, GNP, 2009-2012.

In 2010 and 2011, we did not capture any fish greater than 150 mm, whereas in 2009 and 2012 the size distribution included larger fish (Table 23, Figure 14). Similarly, we did not capture any age-1 wct in 2012. This could be the result of recruitment/reproduction failures in some years. Based on the size structure of the population and the location of the sampling site in the drainage it is likely this population is resident. We collected genetic samples from Fish Creek wct in 2010 for future analysis.

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 captured in Fish Creek, GNP, 2009 – 2012. Length range represents all fish captured.

Species		Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT					
	2009	116.9(35.5)(31)	63-206	21.6(21.1)(31)	1.07
	2010	96.4(21.8)(23)	70-141	10.7(7.7)(23)	1.02
	2011	107.4(15.5)(27)	87-143	14.3(6.5)(27)	1.10
	2012	146.2(19.7)(25)	120-198	34.5(15.9)(25)	1.04

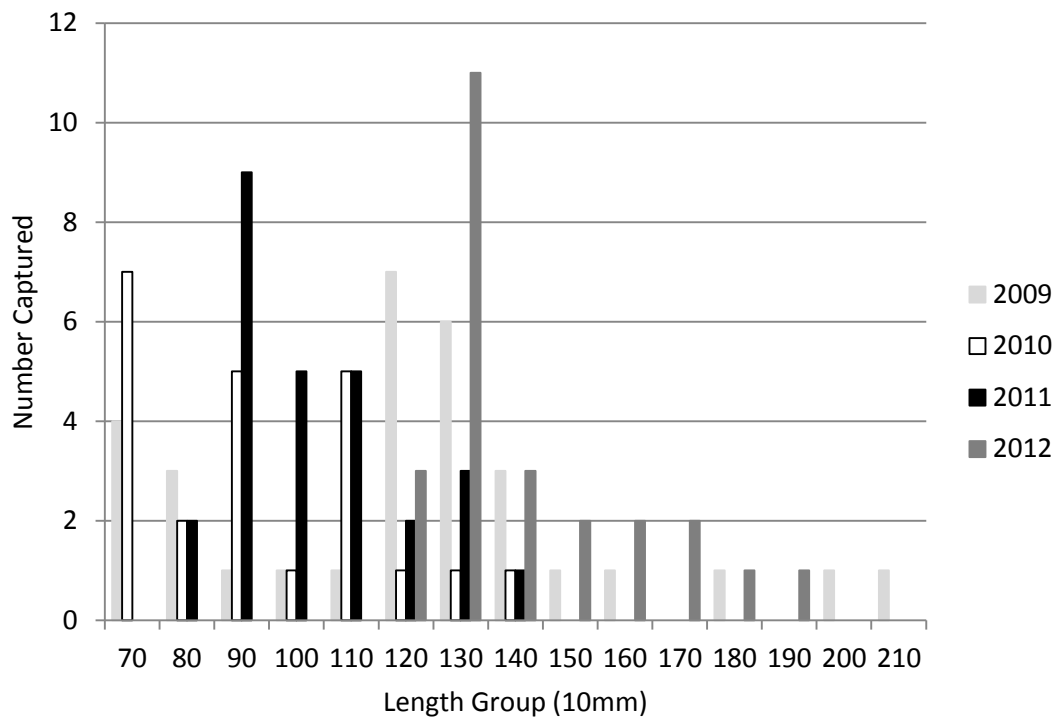


Figure 14. Length-frequency histogram for wct captured in Fish Creek, GNP, in 2009-2012.

Due to a lack of fish response, electrofishing voltage was increased from 2010 through 2012. Each year 30 hertz with a 9% duty cycle and a 3 ms pulse width was employed however initial voltage in 2010 was 550 v, then 665v in 2011 and finally 700 v in 2012. Both adult and juvenile tailed frogs were present and common within the section of Fish Creek that was sampled. Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Physical and chemical measurements varied across sampling years (Table 24).

Table 24. Stream attributes and water chemistry data for Fish Creek, GNP, 2009 – 2012.

	2009	2010	2011	2012
Temperature (°C)	8.0	12	9	12.5
Dissolved Oxygen (mg/L)	8.5	8.08	12.8	NA
pH	6.0	7.9	-	NA
Conductivity (µS)	63.6	77.4	29.5	66.5
Average Slope (%)	1.8	1.7	1.7	1.7
Average wetted width (m)	2.5	2.5	3.2	3.0

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). Westslope cutthroat trout genetic status has not been assessed in Muir Creek. The only species of fish documented in Muir Creek by previous researchers have been westslope cutthroat trout and a single adult bull trout. Juvenile bull trout have not been captured in Muir Creek (Weaver et al. 1983).

Bull trout were captured in Muir Creek in 2010 and 2011 but none were observed in 2012. While wct were sampled in all three years, a single mountain whitefish was also captured in 2011 (287mm TL ;209g). During the three years of monitoring no age-0 bull trout (TL<60mm) or wct (TL<45mm) were observed. Six age-1 and older bull trout averaging 133.7mm and 21.2g were observed in 2010 (Table 25, Figure 15). Only two bull trout were captured in 2011 (Table 25). It is possible that Muir Creek may provide some limited spawning and rearing habitat for migratory bull trout from Flathead Lake. Analysis shows that the average body condition (K) was similar for bull trout each year.

Table 25. 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 in Muir Creek, GNP, 2009 – 2012. Length range represents all fish captured.

Species	Year	Mean Length (mm) (SD) (n)	Length Range (mm) (all individuals captured)	Mean Weight (g) (SD) (n)	Fulton Condition Factor (K)
BLT					
	2010	133.7(6.2)(6)	123-139	21.2(3.4)(6)	0.88
	2011	163(45.2)(2)	131-195	42(32.53)(2)	0.86
WCT					
	2009	124.6(51.9)(28)	56-242	29.6(33.4)(28)	0.99
	2010	122.7(40.6)(49)	62-254	26.7(26.7)(47)	1.08
	2011	154.3(37.0)(30)	74-233	46.3(33.93)(30)	1.07
	2012	169.5(5.0)(2)	166-173	49.5(6.4)(2)	1.01

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. Population estimates across sampling years ranged from 30-51 for wct and from 2-6 for blt (Table 26, Figure 16).

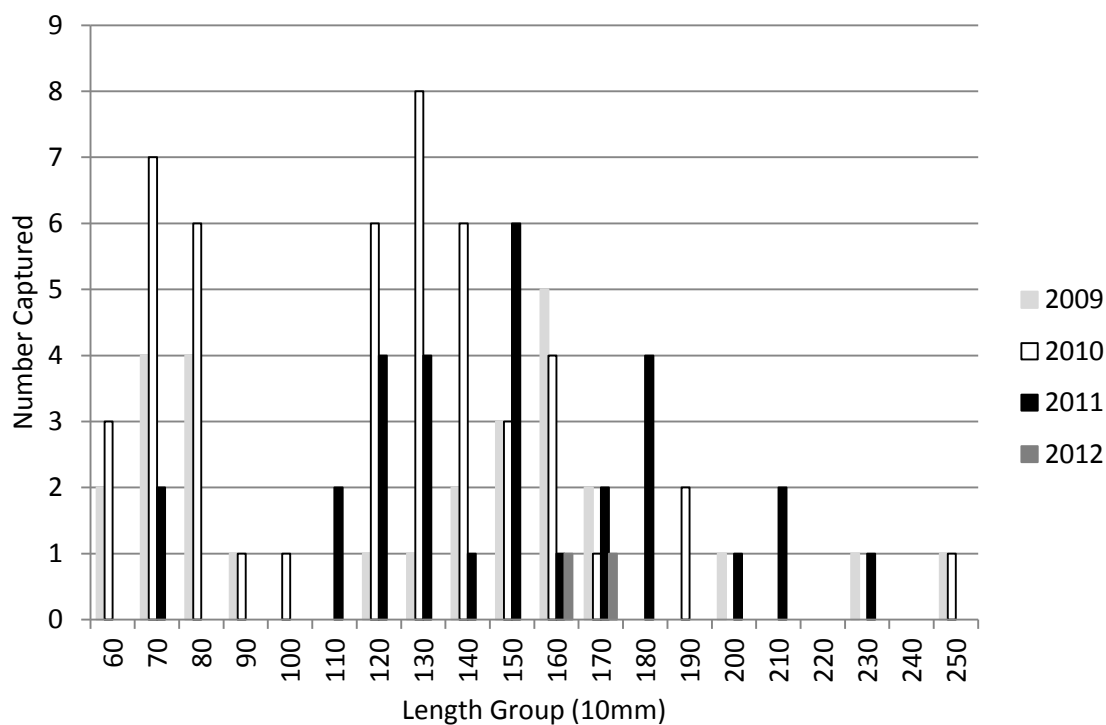


Figure 15. Length-frequency histogram for wct captured in Muir Creek, GNP, 2009-2012.

Table 26. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older bull trout and wct captured on Muir Creek, GNP, 2009-2012.

Species/Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
BLT				
2010	6(5-7)	1.6	1.3	10.0
2011	2	0.5	0.24	2.41
WCT				
2009	30(24-36)	7.6	5.3	60.9
2010	51(46-56)	13.6	10.4	78.0
2011	31(27-35)	7.4	4.08	41.0

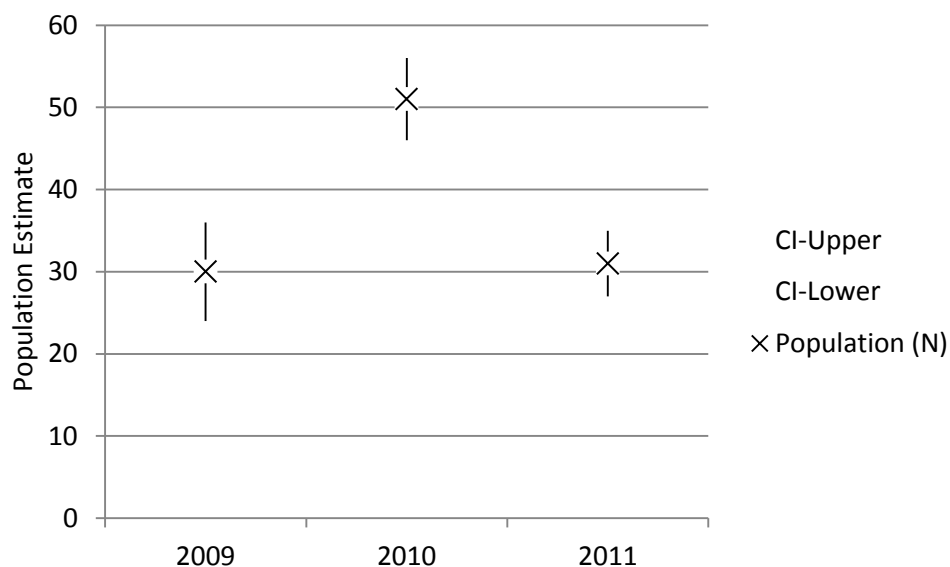


Figure 16. Estimated population abundance of age-1 and older wct in Muir Creek, GNP, 2009-2011.

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 sampling results are similar (13.6 wct/100m²), estimated abundance declined thereafter. The estimates 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.

We employed 600 volts at 30 hertz, 9% duty cycle, and a 3 ms pulse width to capture fish in Muir Creek during 2011 sampling. We did not see or capture any fish until the very end of the first pass (against the block net) in 2012, so we increased the electrofishing settings to 700v, 40 hertz, 9% duty cycle and a 3 ms pulse width. These higher settings did not improve the catch and we concluded abundance of fish was dramatically lower than in previous years. Cobble was estimated to be the dominant substrate type followed by gravel in the reach. No amphibians were observed in Muir Creek. Physical and chemical measurements varied across sampling years (Table 27).

Table 27. Stream attributes and water chemistry data for Muir Creek, GNP, 2009 – 2012.

	2009	2010	2011	2012
Temperature (°C)	11	9.5	11	11
Dissolved Oxygen (mg/L)	8.45	9.09	12.28	NA
pH	8.3	8.47	NA	NA
Conductivity (µS)	86.4	83.3	68.5	111
Average Slope (%)	4.3	2.5	2.5	2.5
Average wetted width (m)	6.2	4.7	5.2	4.4

Missouri River Drainage

Midvale_Creek

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

Single pass sampling was conducted in 2010 and both westslope cutthroat trout and suspected wctxrbt hybrids were captured. We sampled from the headwaters of the drainage (upper-most distribution of fish) downstream to the diversion dam (approximately 1.75 miles). We removed any suspected wctxrbt hybrids from the stream. We collected fin clips from 73 wct, and 6 were determined to be hybrids. Previous genetics sampling in 2002 (MFWP, personal communication) did not detect any hybridization. Based on discussions with the operators of the diversion dam and the recent appearance of hybridization, we concluded that rainbow trout or hybrids were allowed to access the stream upstream of the diversion during a recent dam sluicing operation when the sluice gates were opened and became stuck open for an extended period of time. All individuals captured were greater than 55 mm TL (age-1 and older). This is not unexpected as we were sampling (July 28) prior/close to wct fry emergence. Of the fish for which we have data, the mean length was 186.1mm (SD=39.8, n=40) while the mean weight was 82.5g (SD=45.8, n=34). The average condition factor, K, was 1.07. A majority of the fishes captured were greater than 150mm TL (Figure 17).

No physical parameters beyond basic water chemistry were recorded, therefore no specific density or CPUE can be estimated. Water temperature was 9.2°C, with 7.0 mg/L of dissolved oxygen. Conductivity was 60.2 μ S and pH was 8.26. Electrofishing settings were 800 volts at frequency of 30Hz on 9% duty cycle and a pulse width of 3ms. Boulder was estimated to be the dominant substrate type followed by cobble in the reach. No amphibians were noted.

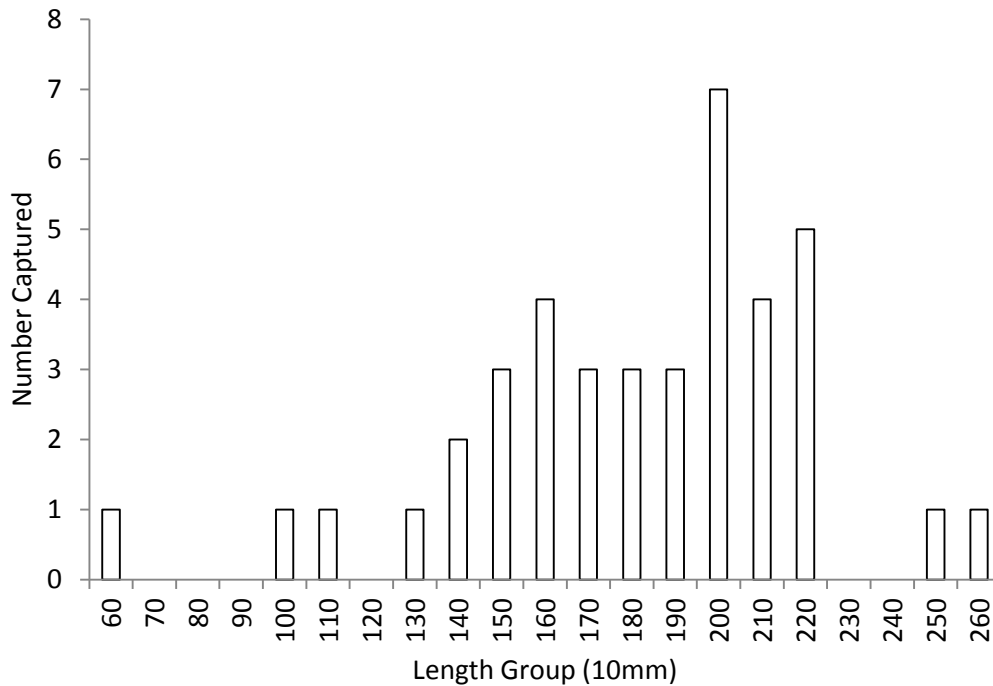


Figure 17. Length-frequency histogram for wct/hybrids captured in Midvale Creek, GNP, in 2010.

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 could be characterized as pristine.

Boulder Creek supports a run of migratory bull trout from the St. Mary River (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 rainbow trout, westslope cutthroat trout and their hybrids are present. Brook trout are also present at low densities .

Bull trout, wct (and apparent hybrids), and one mountain white fish (254mm, 145g) were captured in Boulder Creek in 2011. Two age-0 bull trout (TL<60mm) averaging 52.5mm and weighing less than 1g were captured during sampling. Although the range of individual blt captured was very

wide (470.7) one adult approximately 520 mm in total length accounted for most of the variability (Table 28). The remaining length distribution fell between 50 and 213 mm total length (Figure 18). Average length of age-1 and older blt was 111.7 mm (Table 28). Average condition factor (K) was estimated at 1.01 for age-1 and older blt (SD = 0.14) (n = 74).

Table 28. 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, 2009 and 2011. 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				
2009	112.2 (18.0) (92)	47-182	13.3 (9.0) (55)	0.89
2011	111.7 (53.7) (75)	50-520	14.2 (13.3) (74)	1.0
WCT/hybrids				
2009	117.3 (49.8) (18)	57-235	15.0 (12.2) (10)	1.0
2011	115.5 (47.8) (21)	62-206	27.7 (34.7) (21)	1.0

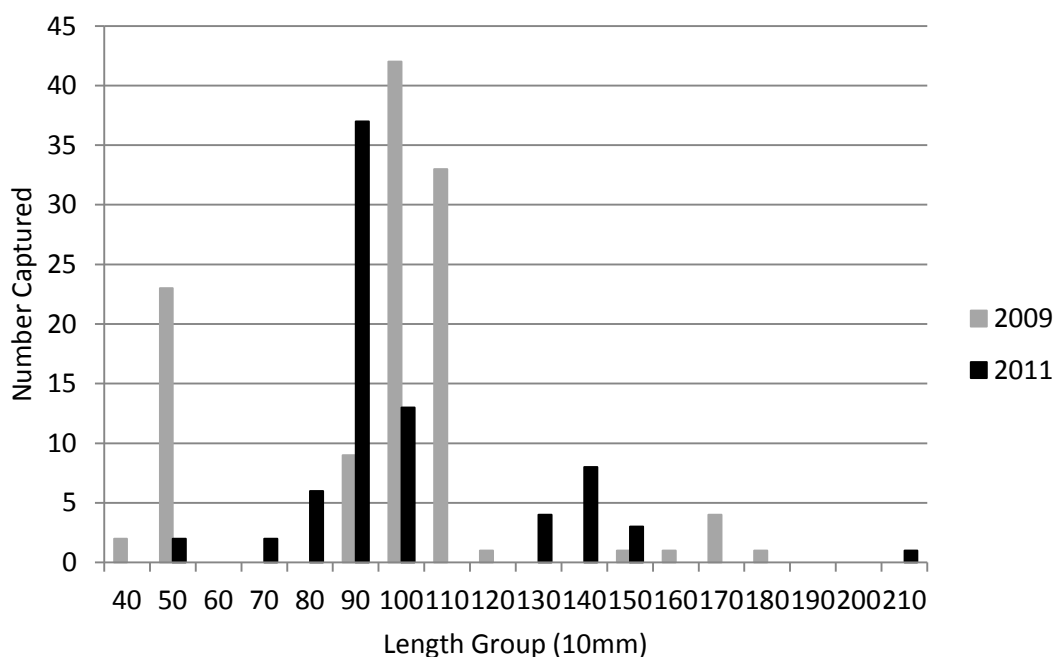


Figure 18. Length-frequency histogram for blt captured in Boulder Creek, GNP, 2009 and 2011.

Within the 131m long stream reach sampled in 2011 we estimated the total abundance of age-1 and older bull trout to be 82 (Table 29). The density estimate for these same fish was 6.6 blt/100m². First pass CPUE for age-1 and older blt was estimated at 59.0 fish/hr and 4.6 fish/100m².

Table 29. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older blt and wct/hybrids captured in Boulder Creek, GNP, 2009 and 2011.

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
BLT				
2009	102(90-114)	9.0	4.9	86.5
2011	82(72-92)	6.6	4.6	59.0
WCT/hybrids				
2009	18(16-20)	1.6	0.89	15.7
2011	23(16-30)	1.9	1.2	15.5

Prior sampling conducted by Downs et al. (2011) in 2009 resulted in an estimated blt abundance of 102 fish compared to the 82 blt noted above (Figure 19). Both the previous density estimate and first pass CPUE rout for bull (Downs et al. 2011) were higher than our estimates. However, the fish sampled in 2009 had a lower average body condition (0.89) compared to that estimated in 2011 (1.01).

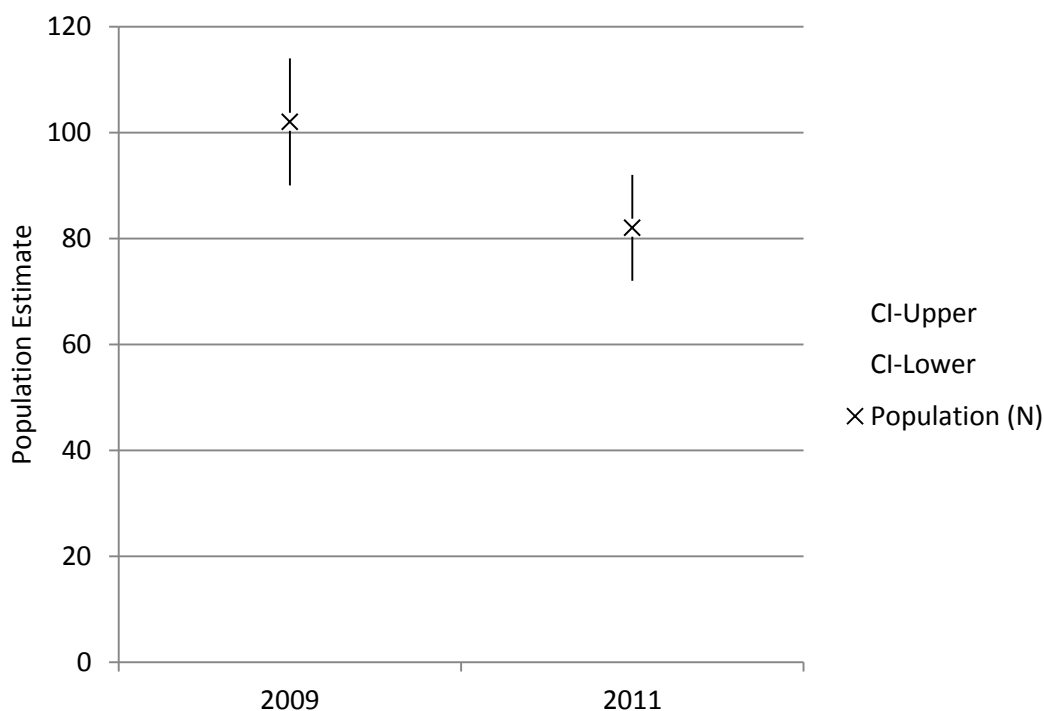


Figure 19. Estimated population abundance for age-1 and older bull trout in Boulder Creek, GNP, 2009 and 2011.

It appeared as though many of the wct captured were hybridized with rainbow trout. Mogen and Kaeding (2004) reported capturing wct hybrids, but also noted that wct captured in one of Boulder Creek's upper tributaries were 95% genetically pure. It is difficult to distinguish hybrids from genetically pure wct in the field, particularly for small fish. Therefore, we treated them collectively as one group (wct/hybrids) in our analyses.

We did not capture any age-0 wct/hybrids (Figure 20). In 2011 the estimated total abundance of age-1 and older wct/hybrids was 23 (Table 29, Figure 21). The density estimate for these same fish was 1.9 wct/100m². First pass CPUE for age-1 and older wct/hybrids was estimated at 15.5 fish/hr and 1.2 fish/100m². Average length of age-1 and older wct/hybrids was 115.5 mm (Table 28). Average Fulton Condition Factor (K) was also estimated at 1.01 for age-1 and older wct/hybrids (SD = 0.23) (n = 20).

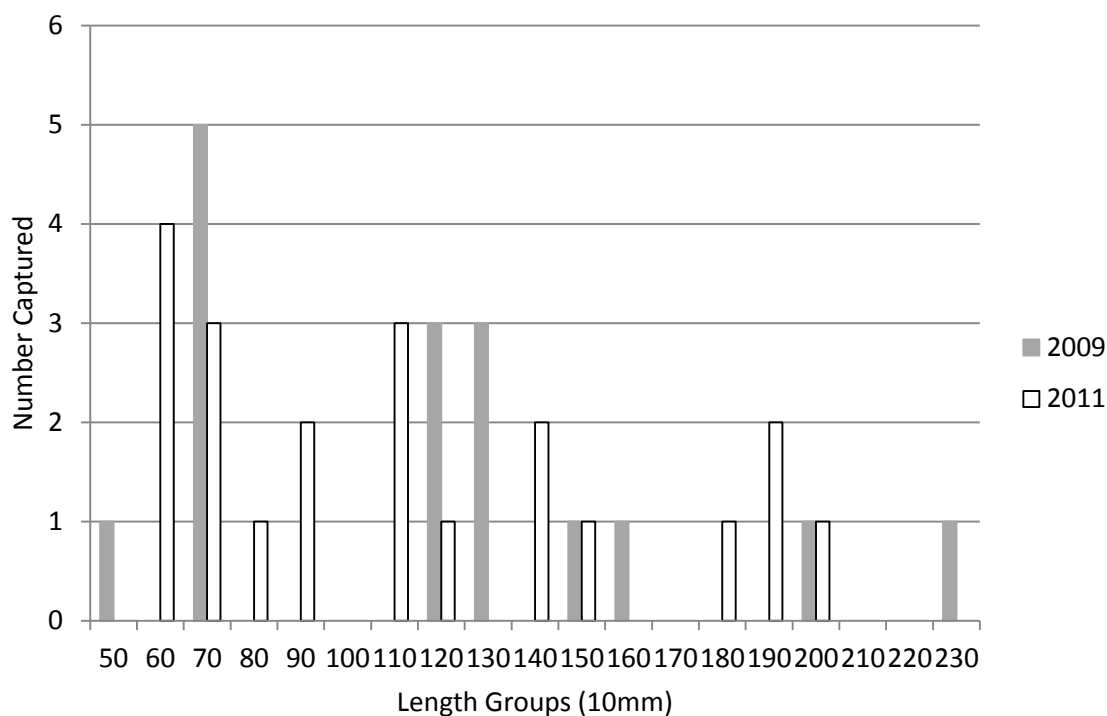


Figure 20. Length-frequency histogram for wct/hybrids captured in Boulder Creek, GNP, 2009 and 2011.

We employed 650 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Boulder Creek. Cobble was estimated to be the dominant substrate type followed by boulder in the study reach. Water temperature at the time of sampling was 12°C. Dissolved oxygen was 10.65 mg/L. and conductivity was measured at 120µS. Mean channel slope was 2.2% and the average wetted width was 9.43m for the study reach. No amphibians were observed.

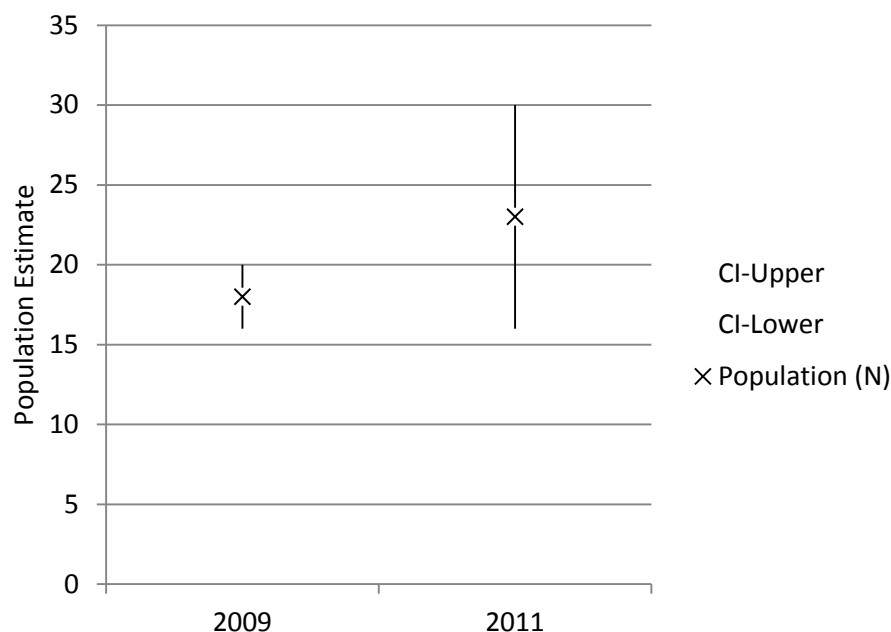


Figure 21. Estimated population abundance for age-1 and older wct/hybrids in Boulder Creek, GNP, 2009 and 2011.

Lee Creek

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

Lee Creek supports a run of migratory bull trout from the St. Mary River (Mogen and Kaeding 2004). Redd counts were initiated in 2011 and conducted again in 2012 (counts were 15 and 31, respectively). In addition to blt, wct hybrids and mwf are present in Lee Creek (Mogen and Kaeding 2004).

Both bull trout and wct/hybrids were captured in Lee Creek in 2011 and 2012. This includes one adult bull trout (425mm, 500g) that was captured in 2011. No age-0 fishes (blt<60mm, wct<45mm) were captured in 2011, however young-of-the-year were observed for both species in 2012. These age-0 blt comprised 21% (n=8, average length 46mm, average weight<1g) of the total number of blt captured in 2012 (38) (Figure 22). Only one wct/hybrid less than 45mm long was observed in 2012 (41mm,<1g; Figure 23).

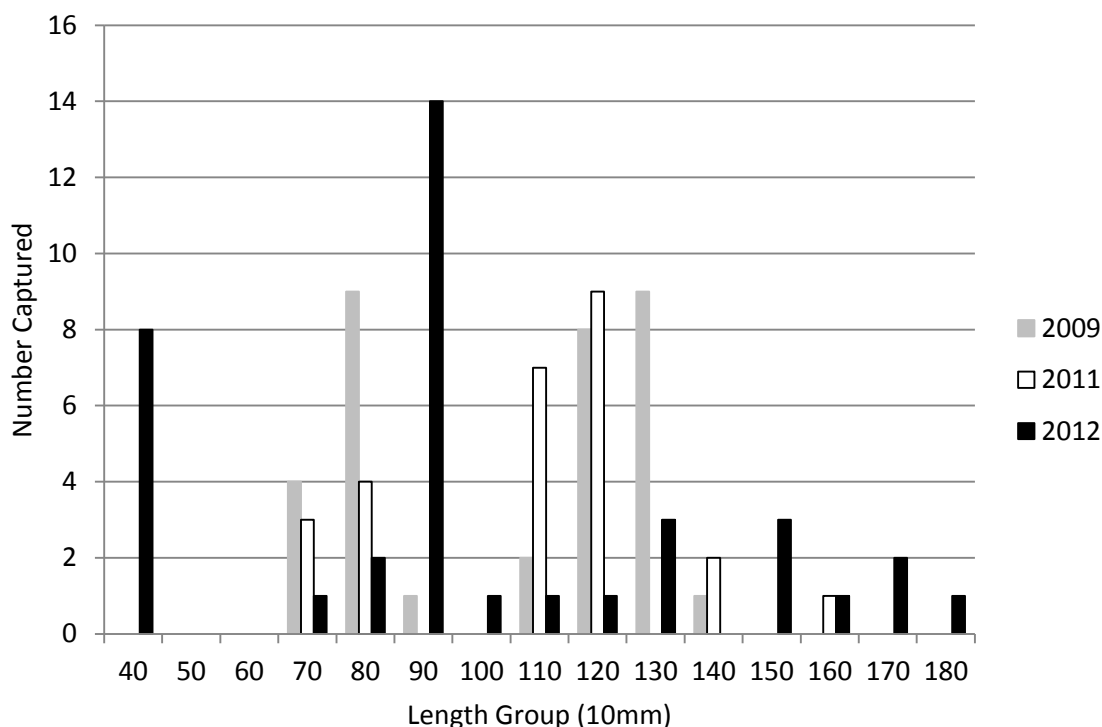


Figure 22. Length frequency histogram for bull trout captured in Lee Creek, GNP, 2009, 2011-2012.

The average length of age-1 and older blt was 123.5 mm in 2011 and 115.5mm in 2012 (Table 30), whereas it had been 101.5mm in 2009 (Downs et al 2011). For the same age-1 and older fish the average condition factor (K) was estimated at 0.88 in 2011 and 0.92 in 2012.

Table 30. 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, 2009, 2011-2012. Length range represents all fish captured.

Species	Year	Mean Length (mm) (SD) (n)	Length Range (mm) (all individuals captured)	Mean Weight (g) (SD) (n)	Fulton Condition Factor (K)
BLT					
	2009	101.5(24.8)(34)	63-133	10.5(6.5)(34)	0.85
	2011	123.5(64.26)(27)	73-425	32.0(93.9)(27)	0.88
	2012	115.5(32.3)(30)	42-186	16.8(13.8)(30)	0.92
WCT					
	2009	121.5(36.7)(26)	73-240	22.5(25.0)(26)	0.98
	2011	147.9(41.3)(17)	80-234	42.7(35.5)(17)	1.05
	2012	126.3(54.0)(26)	41-223	30.3(32)(26)	0.92

The length frequency distribution of wct/hybrids varied between 2009, 2011-2012 (Figure 23). The mean length of wct/hybrids was greatest in 2011 which corresponded with the largest mean weight

(42.7) and highest Fulton Condition Factor (1.05). Length range was similar for westslope cutthroat trout and their hybrids across all sampling years.

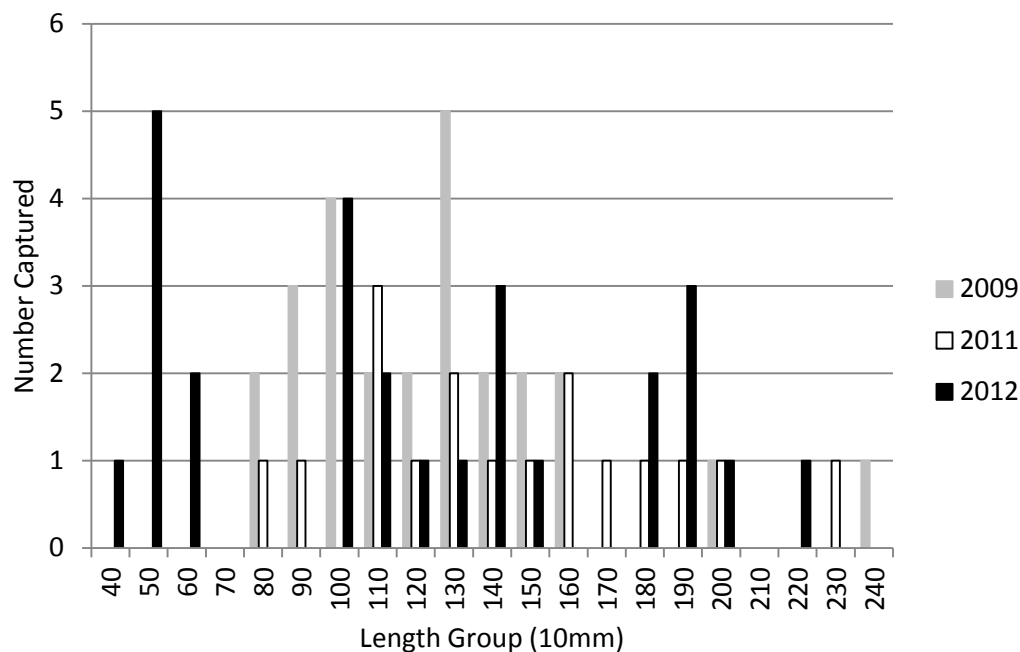


Figure 23. Length frequency histogram for wct/hybrids captured in Lee Creek, GNP, 2009, 2011-2012.

Population estimates were very consistent for bull trout (Table 31). Sampling in 2011 yielded an estimated total abundance of age-1 and older bull trout of 29 while 2012 sampling produced a population estimate of 31 (Table 31, Figure 24). Capture efficiency of bull trout at least 60mm long was lowest in 2012 (2.7 blt/100m², 24.9 blt/hr).

Table 31. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older blt and wct/hybrids captured in Lee Creek, GNP, 2009, 2011 and 2012.

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
BLT				
2009	35(32-38)	7.0	5.0	35.0
2011	29(23-35)	4.5	3.1	28.2
2012	31(27-35)	4.9	2.7	24.9
WCT/hybrids				
2009	29(21-37)	5.8	2.6	18.0
2011	17(15-19)	2.6	2.2	19.7
2012	29(21-37)	4.6	1.9	17.5

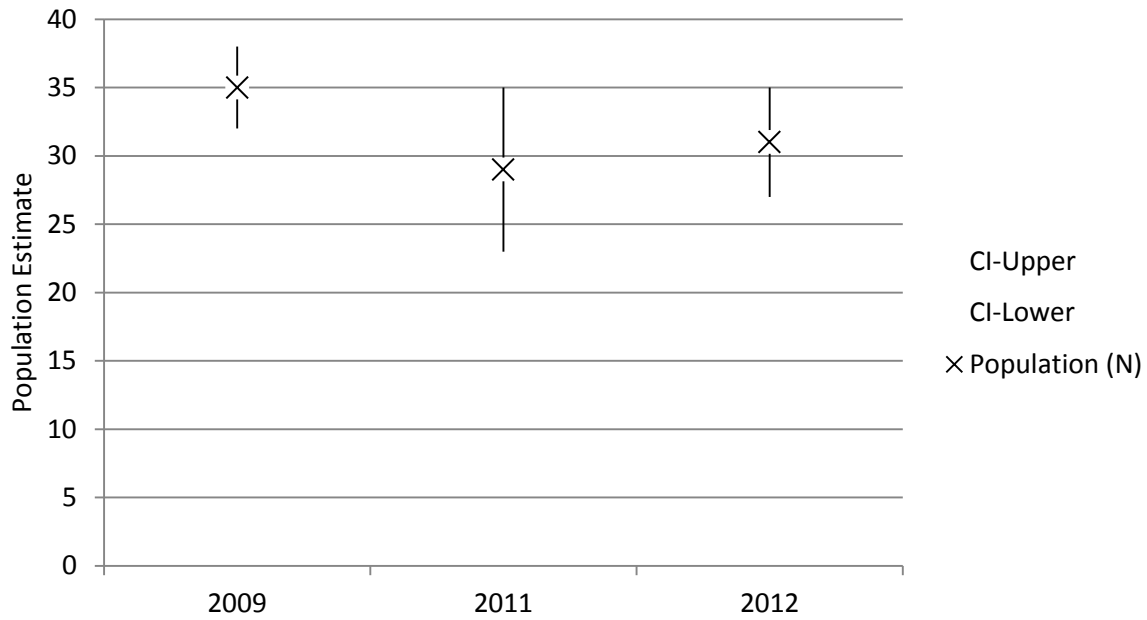


Figure 24. Estimated population abundance for age-1 and older bull trout in Lee Creek, GNP, 2009, 2011-2012.

The population abundance estimate for wct/hybrids in 2012 was identical to that observed in 2009 ($n=29$, 95%CI 21-37). The estimate for 2011 was considerably lower (Figure 25).

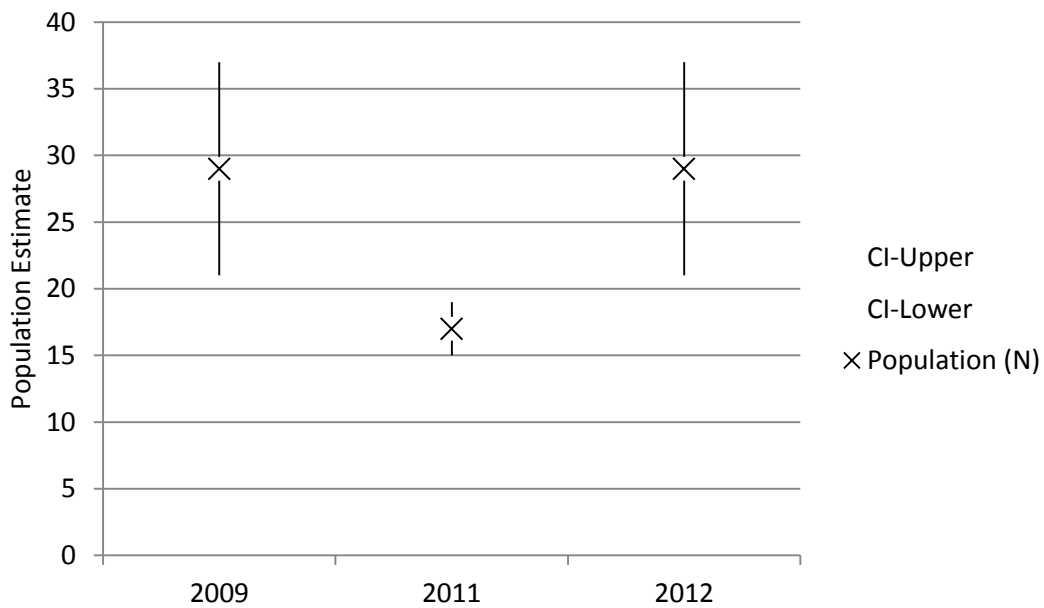


Figure 25. Estimated population abundance for age-1 and older wct/hybrids in Lee Creek, GNP, 2009, 2011-2012.

We employed 450 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width to capture fish in Lee Creek. Cobble was estimated to be the dominant substrate type followed by gravel in the study reach. No amphibians were observed. Physical and chemical measurements varied across sampling years (Table 32). A thermograph was installed approximately 50m downstream of the Chief Mountain Highway crossing, and water temperatures reached a high of 14.7°C on August 7, 2012 (Figure 26).

Table 32. Stream attributes and water chemistry data for Lee Creek, GNP, 2009, 2011-2012.

	2009	2011	2012
Temperature (C)	9.5	6.5	9.3
Dissolved Oxygen (mg/L)	8.08	9.58	NA
pH	6.15	-	NA
Conductivity (μ S)	331	334	296
Average Slope (%)	3.7	2.5	3.7
Average wetted width (m)	4.0	5.2	4.7

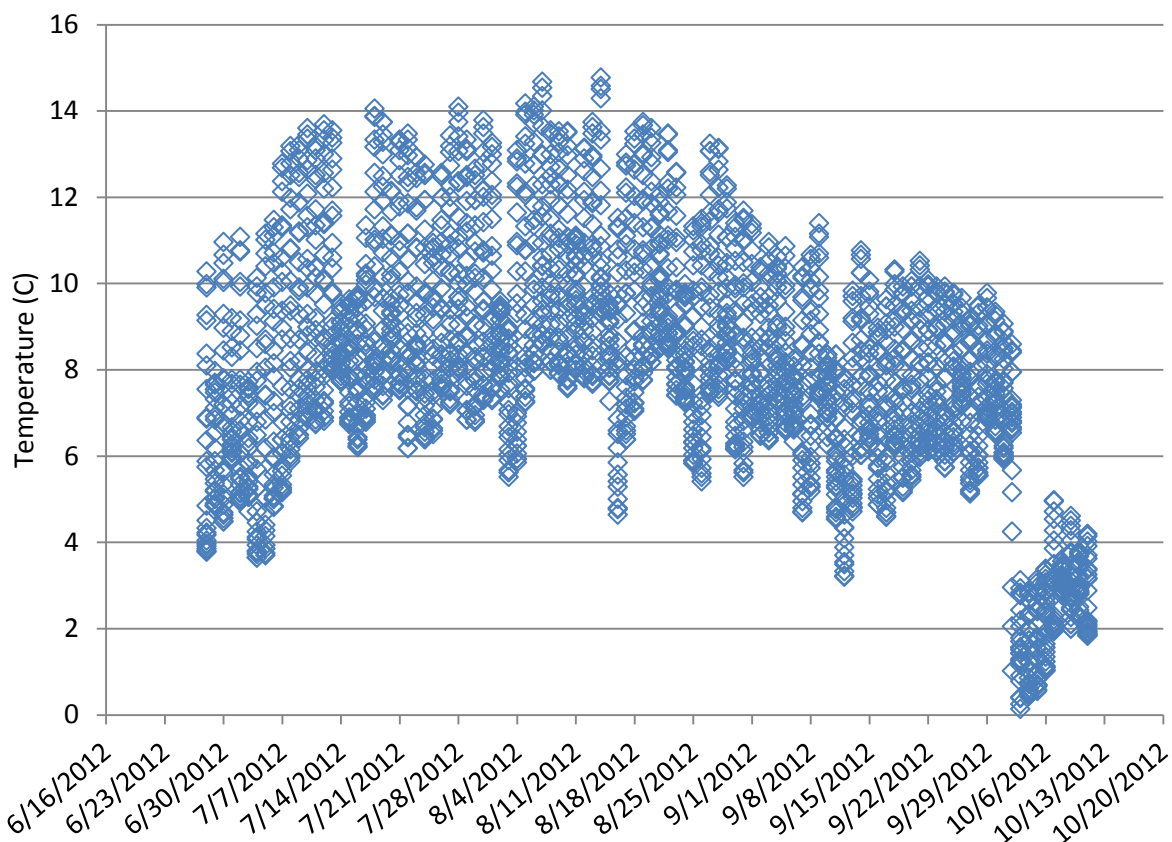


Figure 26. Water temperatures recorded in 2012 in Lee Creek, Glacier National Park, Montana.

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.

Downs et al. (2011) sampled Wild Creek in mid-July (7/15/09) whereas subsequent samplings (2011 and 2012) were conducted in August. The earlier sampling date in 2009 precluded possible observation of age-0 fish, but some age-0 were observed in later years (Table 33).. Except for a single bull trout captured in 2011 (203mm, 73g) only wct were observed in Wild Creek. Two wct less than 30 mm were observed during sampling in 2011 but escaped prior to measurement. During 2012, age-0 wct (TL<45mm) were observed on the edges of the stream channel. In an effort to minimize stress to fry we did not attempt to sample the stream edges in water less than 5cm deep.

Table 33. Stream attributes and water chemistry data for Wild Creek, GNP, 2009, 2011-2012.

	2009	2011	2012
Date	7/15/09	8/9/11	8/13/12
Temperature (°C)	6.9	9	14.5
Dissolved Oxygen (mg/L)	9.97	9.18	NA
pH	5.06	NA	NA
Conductivity (µS)	180.4	151	170.3
Average Slope (%)	5.1	5.1	5.1
Average wetted width (m)	4.9	4.9	4.9
Sample Reach Length (m)	127.7	95	96

A wide range of sizes were captured across all years. Length range and condition factor remained relatively consistent across sampling years (Table 34). Age-1 and age-2 fish dominated the catch in all years (Figure 27). Based on the length-frequency histogram (Figure 27), Wild Creek supports a resident population of westslope cutthroat trout.

Table 34. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older wct captured on Wild Creek, GNP, 2009, 2011-2012. Length range represents all fish captured.

Year	Mean Length (mm) (SD) (n)	Length Range (mm) (all individuals captured)	Mean Weight (g) (SD) (n)	Fulton Condition Factor (K)
2009	98.4(40.7)(64)	45-212	24.5(22.1)(37)	1.1
2011	116.2(31.2)(30)	45-182	22.9(16.8)(30)	1.2
2012	94.2(40.1)(95)	58-224	15.7(25.5)(95)	1.0

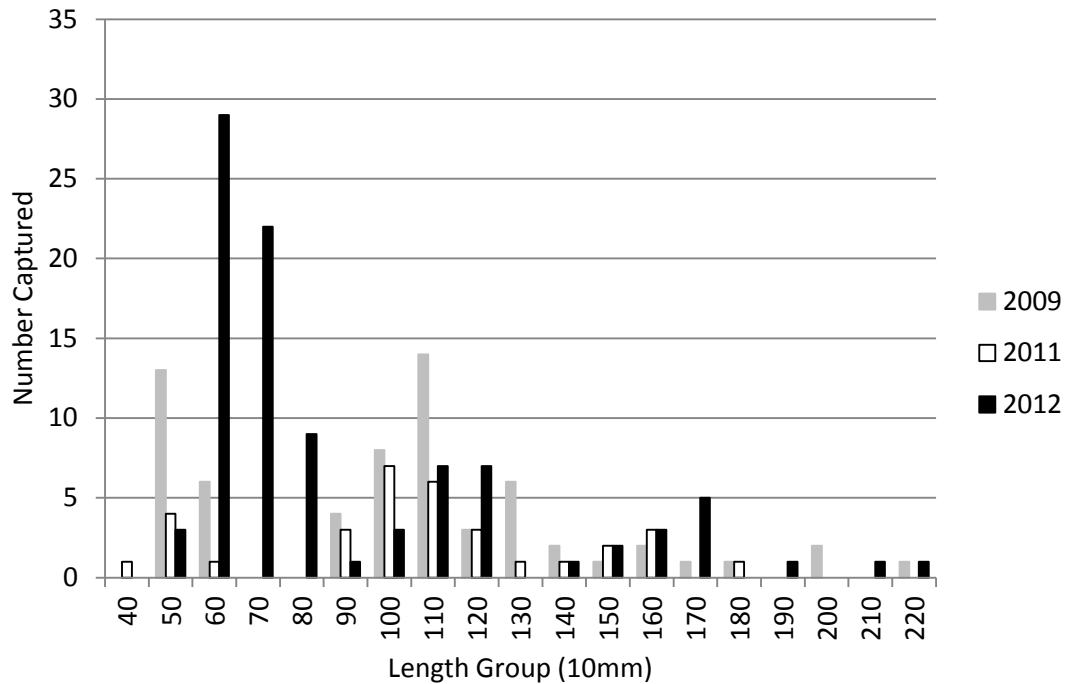


Figure 27. Length frequency histogram for wct captured in Wild Creek, GNP, 2009, 2011-2012.

Estimated total abundance of Wild Creek age-1 and older wct was highly variable across sampling years (Table 35, Figure 28). The 2012 estimate was very high, resulting in part from a strong year class of age-1 fish. It should be noted that although we attempted to sample the same reach using photos and GPS coordinates, the length of the stream reach sampled in 2009 was reported to be 128m whereas the total reach length sampled for 2011 and 2012 was 96m.

Table 35. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured on Wild Creek in 2009, 2011-2012.

Year	Population Estimate (95%CI)	Density (fish/100m ²)	CPUE (fish/100m ²)	CPUE (fish/hr)
2009	70(60-80)	11.1	7.6	61.1
2011	34(30-38)	7.4	5.6	48.4
2012	101(92-110)	21.4	15.9	108.1

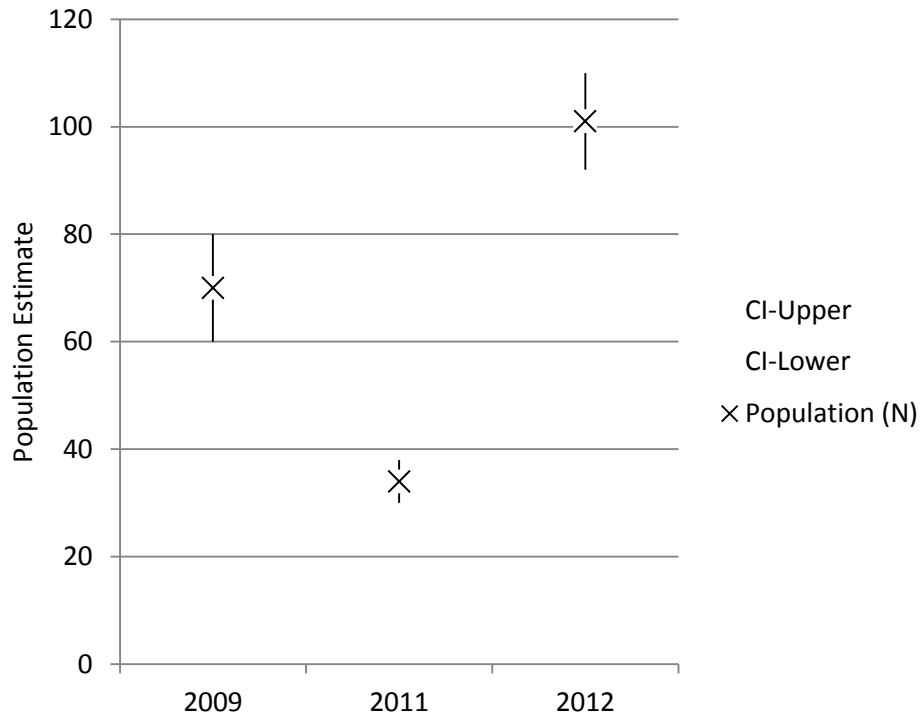


Figure 28. Estimated population abundance for age-1 and older wct in Wild Creek, GNP, 2009, 2011-2012.

Although voltage for electroshocking was increased marginally by 50v in 2012, versus the 500v used in 2009 and 2011, conductivity was similar (Table 33). All other setting remained the same: 30 hertz, with 9% duty cycle and 3 ms pulse width. Dissolved oxygen measurement was not recorded in 2012, but was measured at 9.18mg/L in 2011. Mean channel slope remained 5.1% with a dominant cobble and subdominant boulder substrate. No amphibians were observed.

Other Stream Temperature Monitoring

We also installed thermographs on Ole Creek (Middle Fork Flathead River drainage) and Starvation Creek (North 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 and the thermograph on Starvation Creek was located near the confluence with the North Fork Flathead River, about 20m upstream of the trail crossing. Stream temperature reached a high of 14.8°C in Ole Creek and 17.5°C in Starvation Creek (Figures 29 and 30, respectively).

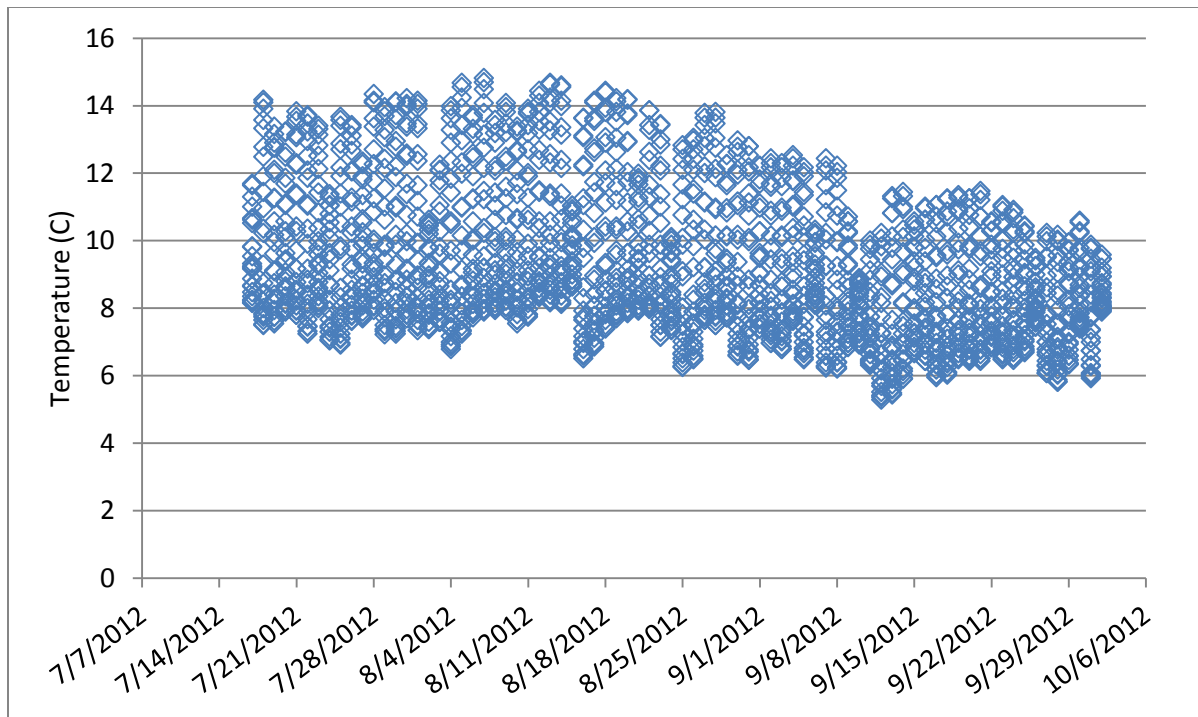


Figure 29. Ole Creek stream temperatures, Glacier National Park, Montana.

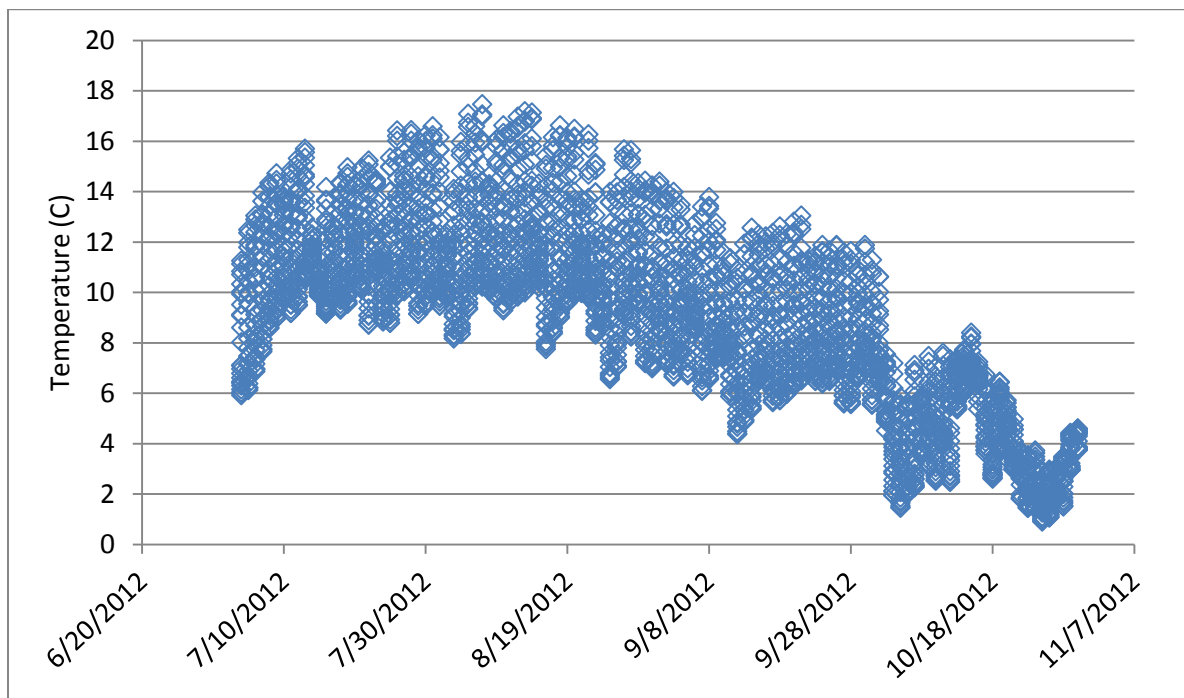


Figure 30. Stream temperature in Starvation Creek, Glacier National Park, Montana.

ACKNOWLEDGEMENTS

The authors wish to thank NPS employees Keith Joint, Billy Brann, Melissa Brickl, and Kim Lindstrom for assistance with field sampling and Mark Biel for reviewing this report. The NPS IMRICO Program provided partial funding for this work.

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Cosley Lake Gill Net Sampling

ABSTRACT

We sampled Cosley Lake with four sinking experimental mesh gill nets from July 30-31, 2012 to characterize the fish community, collect baseline information, and establish a standardized fishery sampling dataset. We captured lake trout (45), mountain whitefish (44), and a single rainbow trout. Length-at-capture ranged from 327-869 mm (TL) for lake trout and 180-363 mm for mountain whitefish. Mean fish condition (K) was estimated 0.89 and 0.87 for lake trout and mountain whitefish, respectively. Size distributions and length-weight relationships for both lake trout and mountain whitefish remained similar between 1986 and 2012 suggesting relatively stable growth and mortality conditions over time.

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INTRODUCTION

Cosley Lake is a 221 acre lake located in the Belly River drainage in Glacier National Park (Figure 1). The lake lies at an elevation of 1,476m and has approximately 5.6 km of shoreline (Morton 1961). Maximum depth is unknown, but our fishery surveys documented depths of 83' during netting. It is likely the lake is considerably deeper than our measurements. The lake is approximately 1.9 km long, with a maximum width of 0.7 km. The drainage is roadless, and the lake is accessed by trail from the Belly River campground. The physical habitat is pristine. Cosley Lake has one backcountry campground, and receives relatively little fishing pressure. However, it is known for providing fishing opportunities for large lake trout (McClung 1998).

Little baseline data exists with regard to native fish population size or fish community structure in Cosley Lake. There has only been one cursory gill net survey conducted previously on Cosley Lake, conducted in 1986 by the U.S. Fish and Wildlife Service using a single experimental gill net (R. Wagner, USFWS, personal communication). Beyond these data, knowledge of the Cosley Lake fishery has been limited to infrequent creel survey and anecdotal fishing reports. We desired to set a series of repeatable gill nets to serve as a baseline for future fishery monitoring efforts.

The status of native fish in Cosley Lake remains unclear. The Mokowanis Valley that contains Cosley Lake may have been filled with ice as the glaciers retreated, with a classic hanging valley formed downstream by Gros Ventre Falls, thereby preventing colonization of Cosley Lake by native fish. This appears to have been the case for the neighboring Belly River drainage (Elizabeth Lake), which is known only to support introduced species Arctic grayling *Thymallus arcticus* and rainbow trout *O. mykiss*. Historical stocking records dating back to 1921 indicate a number of species as having been planted in Cosley Lake, including brook, rainbow and cutthroat trout. However, there are no records of any entity ever stocking lake trout or mountain whitefish in the lake (or any record of stocking mountain whitefish anywhere in the park). Both species are native to the adjacent Waterton and St. Mary river drainages. NPS literature dating back to 1921 (Morton 1961) describes cutthroat and lake trout as well as mountain whitefish as being "present" in Cosley Lake. A well-documented fishery for trophy-sized lake trout had been documented in Cosley Lake as early as the 1940's (McClung 1998). A more recent genetic analysis (Wilson and Hebert 1997) of almost 100 lake trout populations from across the U.S. and Canada partitioned out lake trout in Cosley Lake, Waterton Lake, and Twin Lakes (disjunct native lake trout in Southwest Montana) as a distinct genetic haplotype that likely persisted in a localized glacial refugia. If Cosley Lake lake trout were not native, genetic evidence would suggest they originated from nearby Waterton Lake (or perhaps St. Mary Lake). However, we are unaware of any lake trout culture/stocking programs using lake trout from either of these lakes. Prior to the area becoming a National Park in 1910, it was managed by the U.S. Forest Service as a Forest Reserve (1896-1910). Prior to that, it was part of the Blackfeet Indian Reservation (ceded strip). Any stocking would most likely have been done by/at the request of the Great Northern Railway, which had an early tent camp for visitors on the shore of Cosley Lake, and would have occurred between 1896-1910. A search of NPS historical archives failed to find any evidence that the Great Northern Railway stocked fish at Cosley Lake. In addition, it is questionable as to whether private individuals would have gone through the effort to also transplant mountain whitefish into the system. The presence of mountain whitefish suggests potential access for native fish at some point. This issue deserves further consideration and research.

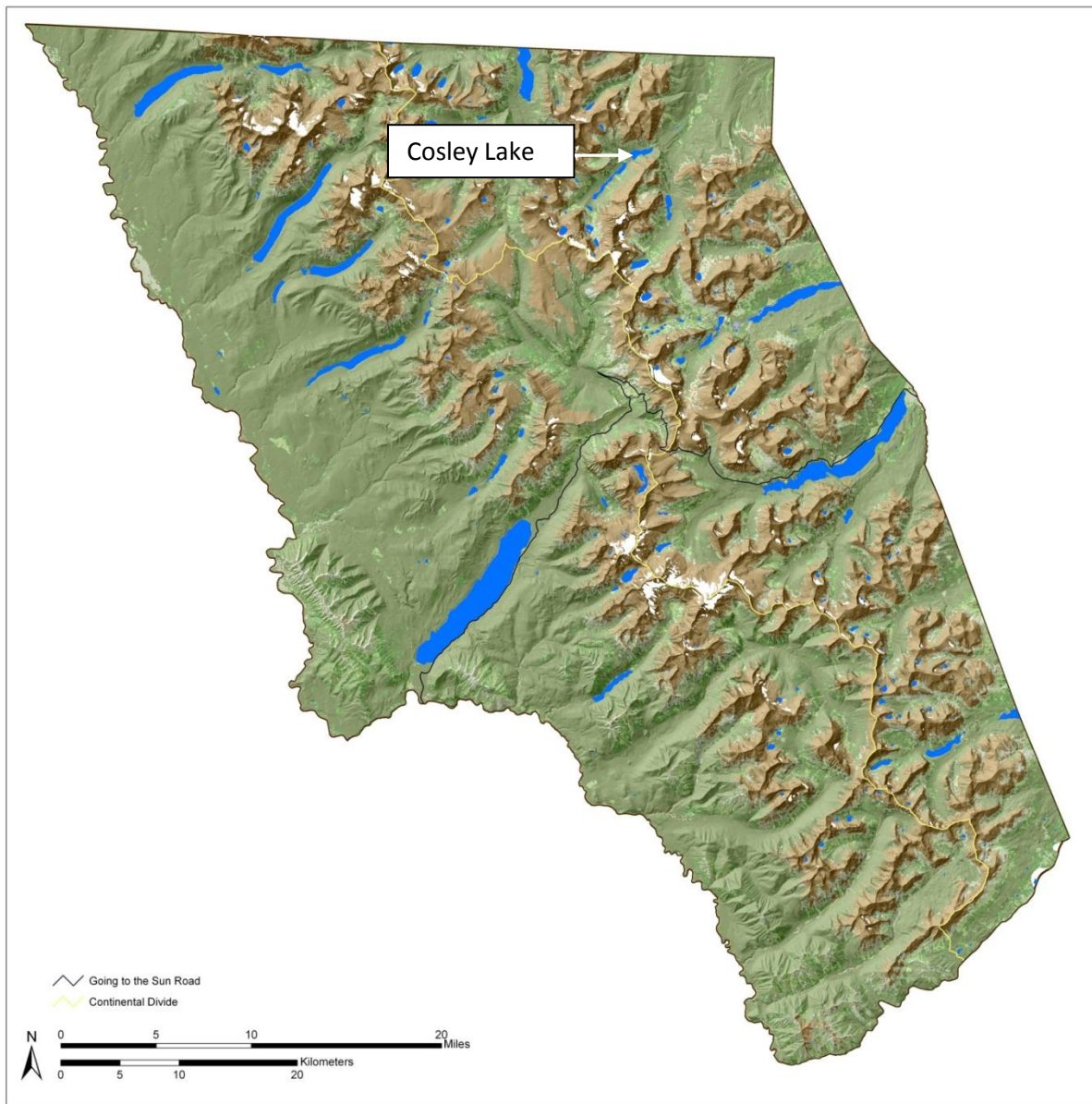


Figure 1. Location of Cosley Lake, Belly River drainage, Glacier National Park.

METHODS

We four used sinking multi-filament experimental mesh gill nets to sample Cosley Lake from July 30-31, 2012 (Figure 2). 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.

COSLEY LAKE - GILL NET LOCATIONS 2012

July 30-31, 2012



Date: 1/3/2013

Lake	UTM_X	UTM_Y	TimeSet	DatePulled	TimePulled	DepthFt	Depth2Ft	Net number	Date set
Cosley	298240	5422787	1915	7/31/2012	845	10	44	COS1	7/30/2012
Cosley	298126	5422726	2000	7/31/2012	900	60	72	COS2	7/30/2012
Cosley	297399	5422706	1910	7/31/2012	845	11	70	COS3	7/30/2012
Cosley	297471	5422683	1947	7/31/2012	1900	73	85	COS4	7/30/2012

Figure 2. Locations of gill net sets in Cosley Lake, July, 2012.

Nets were set from small two-person inflatable rafts in late afternoon/early evening and allowed to soak overnight. They were retrieved at sunrise the following day. Nets were set with the smallest mesh towards shore.

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 estimated relative weight (W_r) (Anderson and Neuman 1996) for selected fish species to evaluate growth conditions across the sampled waters, using standard weight equations for lake trout (Picolo et al. 1993) and mountain whitefish (Rogers et al. 1996).

RESULTS AND DISCUSSION

We captured a total of 90 fish comprising three species during gill netting efforts (Table 1). In 1986 only two species were reported captured, lake trout and what was reported as lake whitefish. However, based on our sampling observations, we believe the whitefish captured in 1986 were actually mountain whitefish. CPUE estimated as fish captured/net night was similar between 1986 and 2012 sampling for lake trout but considerably lower in 2012 than in 1986 (R. Wagner, USFWS, unpublished data) for mountain whitefish (Table 1). This is likely the result of setting only a single gill net in 1986 (insufficient sample size), rather than any changes in mountain whitefish abundance.

Table 1. Gill net catch in Cosley Lake 1986 (R. Wagner, USFWS, unpublished data) and 2012.

Species	1986		2012	
	Catch	CPUE (fish/net night)	Catch	CPUE (fish/net night)
Lake trout	13	13	45	11.3
Mountain whitefish	49	49	44	11.0
Rainbow trout	0	0	1	0.3

Mean length for lake trout captured in 2012 was 515.2mm (Table 2). Mean weight and mean body condition (W_r) were 1,571.3g and 93.4, respectively. Mean length for mountain whitefish was 293.6mm. Mean weight and body condition (W_r) were 242g and 86.1, respectively. Lake trout sample size from the 1986 sampling is too small to draw comparisons. However, mountain whitefish mean length was similar between years. Length distributions were also similar, although the 1986 sampling captured more, larger fish (Figure 3). The length-weight relationships for both lake trout and mountain whitefish have remained constant over time, suggesting forage and growth conditions have remained fairly stable (Figures 4 and 5). Lake trout in Cosley Lake have the greatest average length, the best body condition (highest relative weight), and the highest CPUE of any of the large lakes previously surveyed using standardized methods in the park.

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 2012 using gill nets in Cosley Lake, Glacier National Park.

Species	Mean length (SD) (n)	Length range	Mean weight (SD) (n)	Mean Wr (SD) (n)
Lkt	515.2 (172.3) (44)	327-869	1,571.3 (1,718.4) (40)	93.4 (9.3) (40)
Mwf	293.6 (54.4) (44)	180-363	242 (112) (44)	86.1 (7.8) (44)

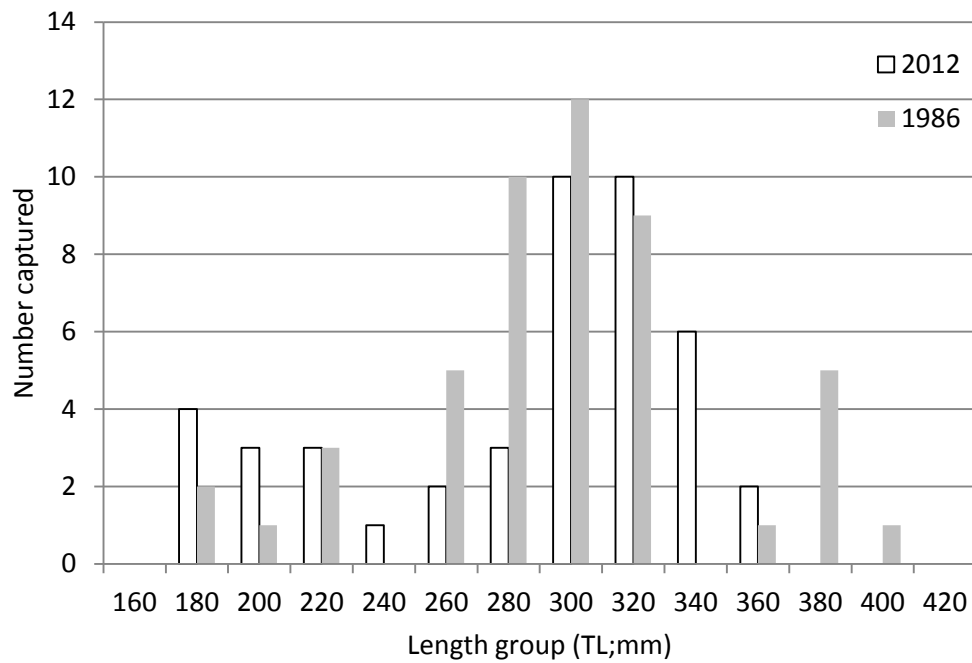


Figure 3. Length distribution of mountain whitefish captured in Cosley Lake, Glacier National Park, 1986 and 2012.

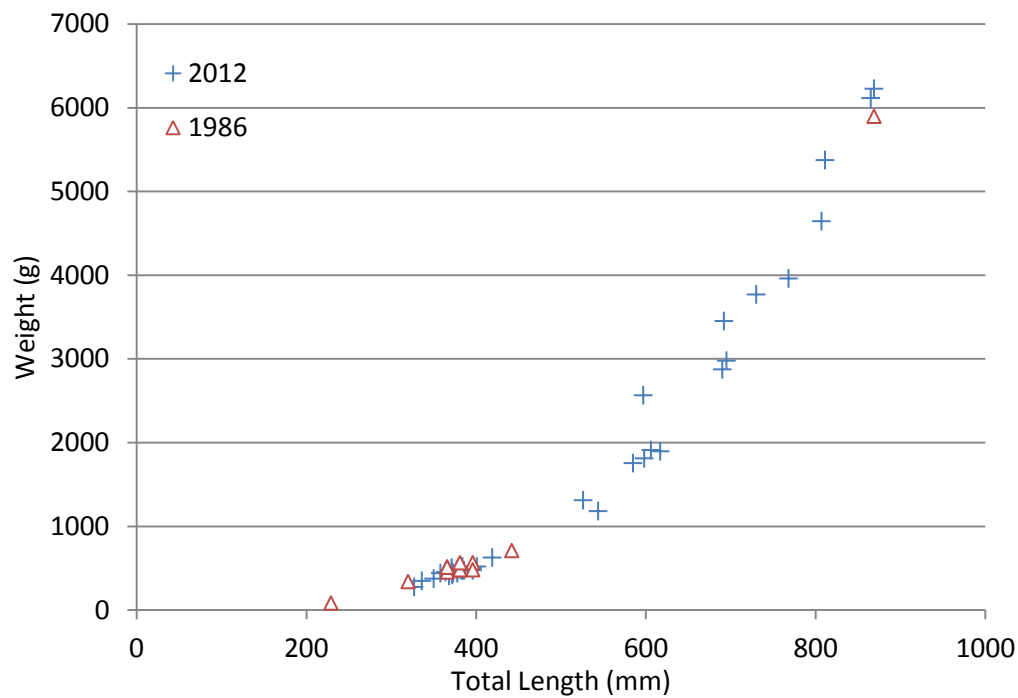


Figure 4. Comparison of length-weight relationships for lake trout captured in Cosley Lake, Glacier National Park, Montana.

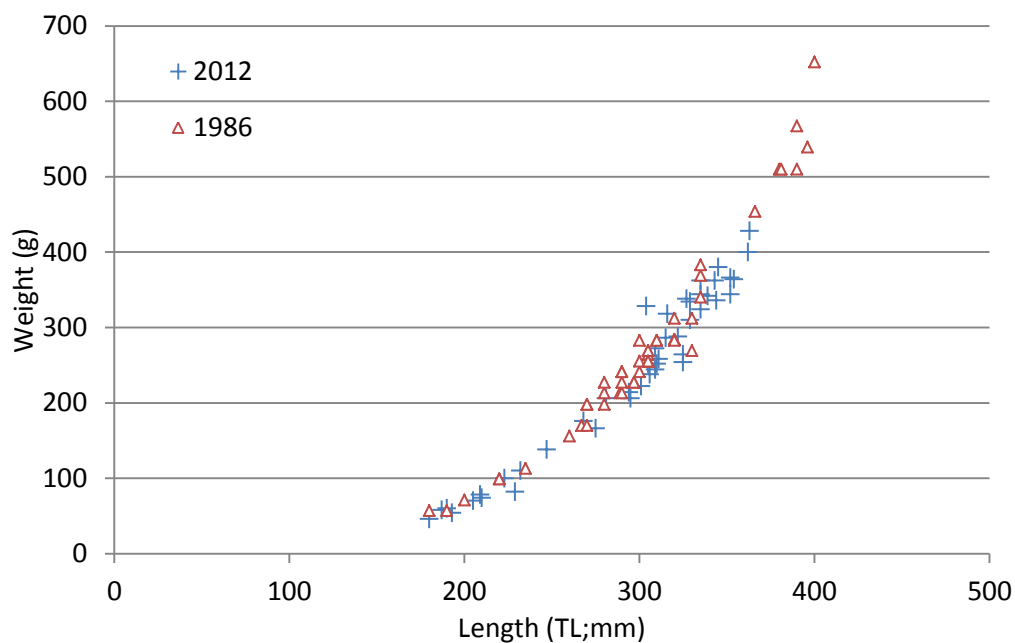


Figure 5. Comparison of length-weight relationships for mountain whitefish captured in Cosley Lake, Glacier National Park, Montana.

Existing park fishing regulations permit the harvest of two lake trout in waters east of the Continental Divide. Given the number of large fish in our catch, remote access, and the need for a boat/float tube to reliably catch lake trout in Cosley Lake, overharvest of lake trout does not appear to be a concern at this time. Future surveys should be conducted every 10-15 years in response to any changing habitat/visitor use conditions.

ACKNOWLEDGEMENTS

The author wishes to thank Melvin Woody, Billy Brann, and Craig Stafford for assistance with gill netting. Mark Biel USFWS provided a constructive review of this report. The NPS IMRICO Program provided funding for this work.

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Harrison and Lower Quartz Lakes Trend Gill Net Survey

ABSTRACT

We sampled Harrison and Lower Quartz lakes in Glacier National Park in 2011 to characterize the fish community and to assess any fish community changes over time. with earlier gill net efforts. We used sinking experimental mesh gill nets, set overnight, to sample both lakes. We captured eight species (five native) in Harrison Lake and six species (five native) in Lower Quartz Lake. Mountain whitefish were the most abundant species in our catch for both lakes. Lake trout appear to be increasing in abundance in both lakes, while native bull trout continue to decline. Harrison Lake supports both non-native lake trout and brook trout, both of which have adverse impacts on native bull trout. Unlike most other bull/lake trout lakes on the west side of the park, the combination of non-native species in Harrison Lake poses predation and competition risks in both the lake environment (lake trout) and hybridization and competition risks in the tributaries (brook trout) for bull trout. This creates a situation that makes it more difficult for bull trout to maintain their abundance/genetic integrity and also complicates any potential restoration efforts.

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INTRODUCTION

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

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

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Arctic grayling <i>Thymallus arcticus</i>	--	--	I
Brook trout <i>Salvelinus fontinalis</i>	I	I	I
Bull trout (BLT) <i>S. confluentus</i>	N	--	N
Kokanee (KOK) <i>Oncorhynchus nerka</i>	I	--	I
Lake trout (LKT) <i>S. namaycush</i>	I	--	N
Lake whitefish (LWF) <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 <i>O. mykiss</i>	I	I	I
Westslope cutthroat trout (WCT) <i>O. clarkii lewisi</i>	N	N	N
Yellowstone cutthroat trout (YCT) <i>O. c. bouvieri</i>	I	I	I

Although GNP represents some of the last best wild areas in North America, recent studies have demonstrated that GNP is not immune to anthropogenic impacts such as exotic species and airborne contaminants. Recent fisheries studies (Marnell et al. 1987, Fredenberg 2002, Hitt et al. 2003, Muhlfeld et al. 2009) have demonstrated the ecological damage and genetic consequences associated with non-native fish species in park waters. For example, Fredenberg (2002) documented the replacement of bull trout as the top predator by non-native lake trout in sympatric lakes on the west side of the park during the last several decades, and lake trout are continuing to expand their numbers and range on the west

side of the park. In response to the lake trout threat the park has recently initiated collaborative efforts to secure remaining at-risk bull trout populations. Monitoring data from park lakes will be key in evaluating any future shifts in fish community structure that may result from additional impacts of non-native species, climate change, or other habitat perturbations.

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

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

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Northern pikeminnow (NPM) <i>Ptychocheilus oregonensis</i>	N	--	--
Peamouth (PEA) <i>Mylocheilus caurinus</i>	N	--	--
Redside shiner (RSS) <i>Richardsonius balteatus</i>	N	--	--
Longnose sucker (LNS) <i>Catostomus catostomus</i>	N	N	N
Largescale sucker (CSU) <i>C. macrocheilus</i>	N	--	--
White sucker (WSU) <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 (BUR) <i>Lota lota</i>	--	--	N
Northern pike (NPI) <i>Esox lucius</i>	--	--	N
Trout-perch <i>Percopsis omiscomaycus</i>	--	--	N

METHODS

We used sinking multi-filament experimental mesh gill nets to sample Harrison and Lower Quartz lakes (Figure 1). All nets were 38.1m (125') X 1.83m (6') experimental multifilament nylon with five 7.62m (25') panels consisting of 19mm (3/4") (#104 twine), 25mm (1") (#139), 32mm (1-1/4") (#104), 1-1/2" (38mm)(#104), 51mm (2") (#139) bar measured mesh. 30# leadcore and polyfoam line was used to hold the net upright, while maintaining contact with the lake bottom. Harrison Lake was sampled August 22-24, 2011 while Lower Quartz was sampled from July 25-27, 2011.

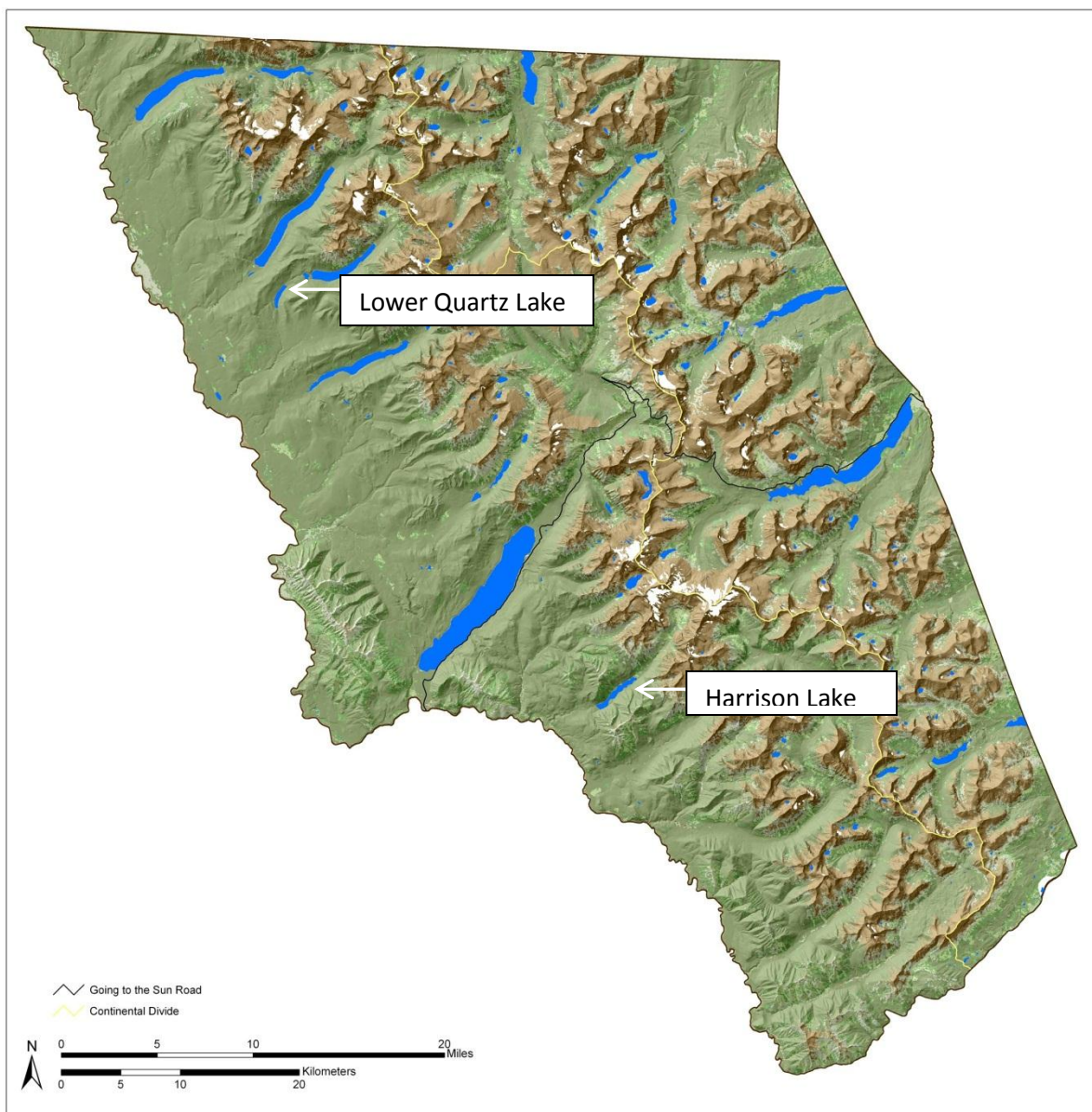


Figure 1. Locations of Harrison and Lower Quartz lakes, Glacier National Park, Montana.

On Harrison Lake, we attempted to repeat set locations and depths used in earlier sampling (USFWS unpublished data, Meeuwig et al. 2007) (Table 3). Set locations are similar but some nets were dragged deeper than desired during the setting process resulting in slightly deeper sets (Figure 2). In general, the inshore end of nets should be set at approximately 10'-15' of depth. On Lower Quartz Lake, we covered a similar range of sample depths, but in a desire to improve habitat type coverage, net set locations varied across years (Figure 3). We deployed nets in similar locations as to those used in 2001 due to the improved spatial coverage over 2005 sampling. The small size and shallow nature of Lower Quartz Lake likely minimized any chance for significant bias caused by changes in individual set locations between years.

Table 3. Previous gill net set dates, locations, and depths for Harrison Lake, Glacier National Park.

Net	UTM Zone	UTM Easting	UTM Northing	Set Date 2000	Inshore d (m)	Set Date 2005	Inshore d (m)
1	12	293812	5376455	8/28/00	3.1	8/26/05	8
2	12	294010	5376520	8/28/00	4.6	8/26/05	3.8
3	12	294719	5377019	8/29/00	4.2	8/26/05	6.1
4	12	294907	5377206	8/29/00	3.1	8/26/05	4.5
5	12	295579	5378014	8/28/00	3.1	8/27/05	4.1
6	12	295828	5378098	8/28/00	3.1	8/27/05	5.2
7	12	296557	5378537	8/29/00	3.1	8/27/05	2.4
8	12	296878	5378735	8/29/00	3.7	8/27/05	7.9
9	12	296053	5377901	N/A	N/A	8/28/05	4.6
10	12	296010	5377847	N/A	N/A	8/28/05	5.1

Nets were retrieved in the morning, labeled, and taken to shore to be cleared of fish. Fish were identified to species, weighed (g) and measured (total length (TL); mm). Age structures and (otoliths) were collected from incidental bull trout mortalities.

We estimated catch-per-unit-effort (CPUE) as the number of fish captured per net-night of sampling. We calculated average length and weight for each species to facilitate comparisons of size structure between populations, and also provide information useful in comparing changes in all of the population and sampling metrics through time. We estimated relative weight (*Wr*) (Anderson and Neuman 1996) for selected fish species to evaluate growth conditions across the sampled waters. We used standard weight equations for lake trout (Picolo et al. 1993) and mountain whitefish (Rogers et al. 1996) to estimate *Wr* for each water. Fulton-type (*K*) condition factors were used for other species (Anderson and Neuman 1996). We compared our current catch composition and CPUE with that of earlier sampling to evaluate trends over time.

HARRISON LAKE - GILL NET LOCATIONS 2011

August 22-24, 2011

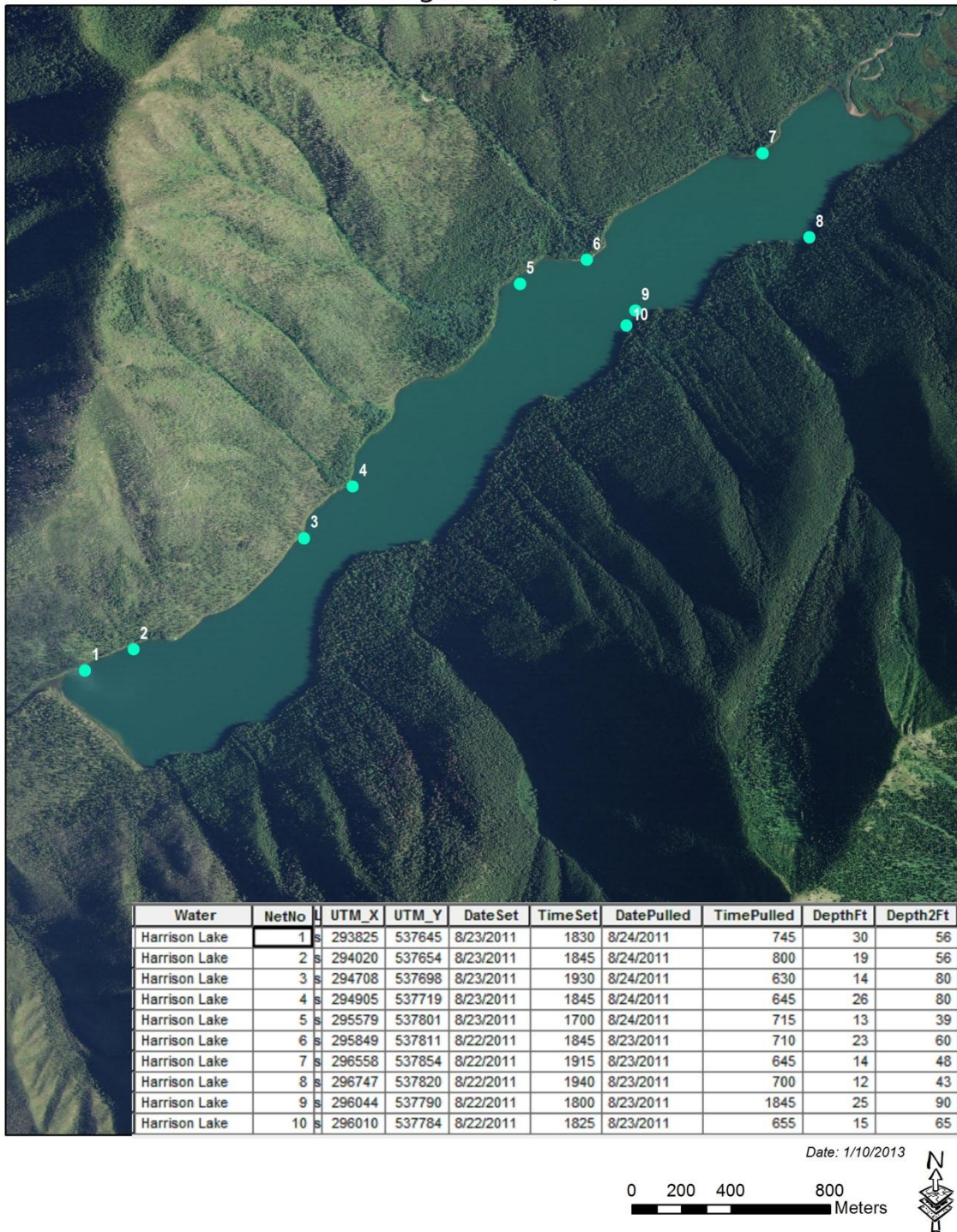


Figure 2. 2011 gill net set locations on Harrison Lake, Glacier National Park.

LOWER QUARTZ LAKE - GILL NET LOCATIONS 2011 July 25-27, 2011

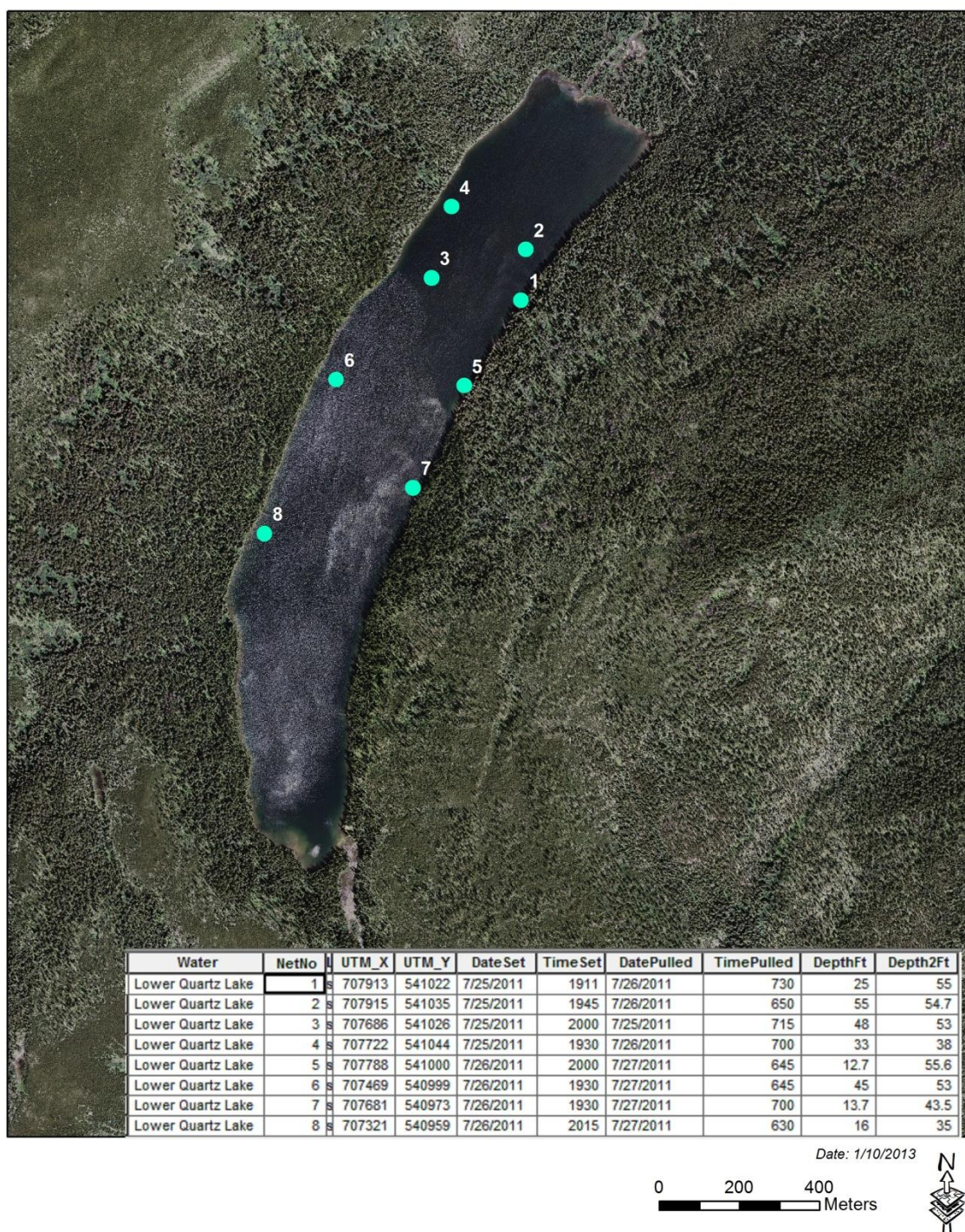


Figure 3. 2011 gill net set locations on Lower Quartz Lake, Glacier National Park.

RESULTS AND DISCUSSION

We set 10 nets on Harrison Lake (five nets each on 8/22 and 8/23) and 8 nets on Lower Quartz Lake (four nets each on 7/25 and 7/26). Species diversity was highest in Harrison Lake (Table 4). CPUE varied by species in each lake, with mountain whitefish being the most abundant in our catch for both lakes. Netting data from both Harrison and Lower Quartz lakes indicates an increasing relative abundance of lake trout and decreasing relative abundance of bull trout in both lakes (Figures 4 and 5). CPUE of net sets over time (USFWS, unpublished data; Meuwig 2008) suggests that lake trout are increasing in abundance fairly rapidly in Harrison Lake while bull trout are in decline. Patterns are less clear in Lower Quartz Lake, where lake trout appear to be increasing and bull trout do not show a consistent trend. However, it should be noted that Lower Quartz Lake is a relatively small and shallow lake (maximum depth of 62') with limited stream spawning and rearing habitat. Its ability to support a robust self-sustaining population of lake trout without immigration from upstream/downstream sources is questionable. CPUE varied across species and years for both Harrison (Table 5) and Lower Quartz Lake (Table 6).

Table 4. Catch composition for Harrison and Lower Quartz lakes, Glacier National Park, 2011.

Water	Species	Number captured	CPUE (fish/net night)
Harrison Lake	Brook trout (brk)	1	0.1
	Bull trout (blt)	5	0.5
	Coarsescale sucker (csu)	15	1.5
	Kokanee (kok)	3	0.3
	Lake trout (lkt)	14	1.4
	Longnose sucker (lns)	50	5.0
	Mountain whitefish (mwf)	137	13.7
	Westslope cutthroat trout (wct)	4	0.4
Lower Quartz Lk.	Bull trout	2	0.3
	Coarsescale sucker	2	0.3
	Lake trout	3	0.4
	Longnose sucker	17	2.1
	Mountain whitefish	28	3.5
	Westslope cutthroat trout	18	2.3

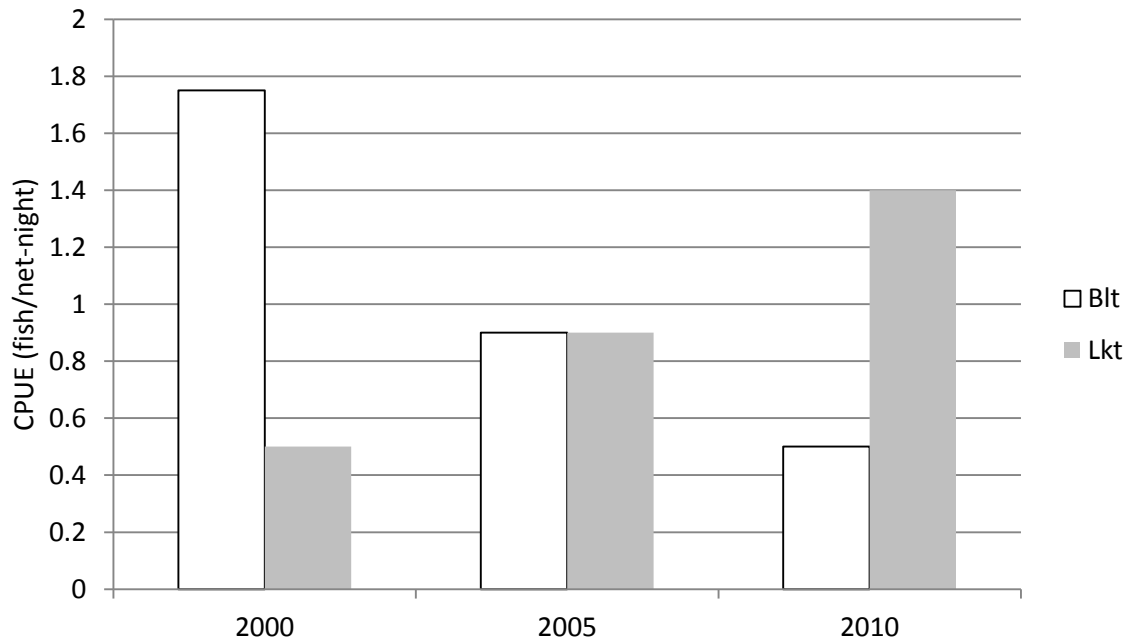


Figure 4. Gill net catch-per-unit-effort (CPUE; fish/net-night) for bull and lake trout over time in Harrison Lake, Glacier National Park.

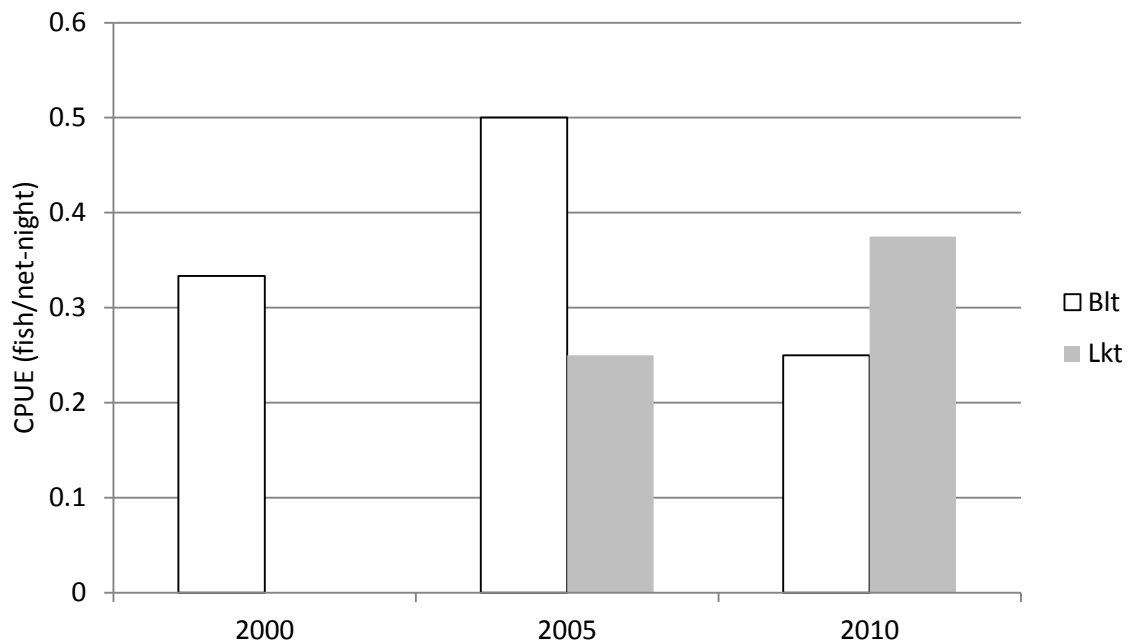


Figure 5. Gill net catch-per-unit-effort (CPUE; fish/net-night) for bull and lake trout over time in Lower Quartz Lake, Glacier National Park.

Table 5. Catch-per-unit-effort (CPUE; fish/net night) over time for species captured during gill net sampling, Harrison Lake, Glacier National Park.

Species	2000	2005	2011
Bull trout	1.9	0.9	0.5
Brook trout	0.3	0.1	0.1
Coarsescale sucker	1.3	0.0	1.5
Kokanee	0.0	0.4	0.3
Lake trout	0.5	0.9	1.4
Longnose sucker	6.0	5.1	5.0
Mountain whitefish	25.1	33.8	13.7
Northern pikeminnow	0.1	0.0	0.0
Westslope cutthroat trout	0.0	1.2	0.4

Table 6. Catch-per-unit-effort (CPUE; fish/net night) over time for species captured during gill net sampling, lower Quartz Lake, Glacier National Park.

Species	2000	2005	2011
Bull trout	0.3	0.5	0.3
Coarsescale sucker	0.1	0.0	0.3
Lake trout	0.0	0.3	0.4
Longnose sucker	0.8	13.1	2.1
Mountain whitefish	4.8	23.8	3.5
Redside shiner	0.0	0.3	0.0
Westslope cutthroat trout	0.7	5.6	2.3

Table 7. Mean length (TL;mm), length range, and weight (g) for species captured in gill net sets in Harrison Lake, Glacier National Park.

Species	Mean length (SD) (n)	Length range (TL;mm)	Mean weight (SD) (n)	Mean Wr/K* (SD) (n)
Brook trout	197.0	N/A	76.0	N/A
Bull trout	284.0 (127.4)(5)	204-510	84 (17.1)(3)	0.78* (0.04)(3)
Coarsescale sucker	180.9 (82.1)(15)	111-450	115.1 (251.6)(15)	1.04* (0.09)(15)
Kokanee	279.7 (72.2)(3)	235-363	228.0 (181.8)(3)	0.91* (0.003)(3)
Lake trout	481.8 (131.8)(14)	139-592	963.1(602.6)(13)	77.6 (8.9)(13)
Longnose sucker	249.0 (106.6)(50)	110-510	108.1(140.4)(35)	1.04* (0.06)(35)
Mountain whitefish	226.1 (46.1)(137)	128-396	104.5 (77.4)(135)	78.6 (6.4)(135)
Westslope cutthroat trout	223.3 (119.5)(4)	100-377	261.0(306.9)(2)	0.93* (0.05)(2)

Table 8. Mean length (TL;mm), length range, and weight (g) for species captured in gill net sets in Lower Quartz Lake, Glacier National Park.

Species	Mean length (SD) (n)	Length range (TL;mm)	Mean weight (SD) (n)	Mean Wr/K* (SD) (n)
Bull trout	601.5 (118.1)(2)	518-685	1,871.0 (920.7)(2)	0.83* (0.07)(2)
Lake trout	748.0 (137.2)(2)	651-845	2,855 (881.1)(2)	65.8 (18.5)(2)
Longnose sucker	351.9 (50.2)(17)	247-444	516.3 (198.8)(13)	1.04* (0.08)(13)
Mountain whitefish	205.2 (49.4)(27)	163-435	89.1 (169.8)(26)	73.4 (9.9)(17)
Westslope cutthroat trout	261.1 (79.5)(18)	182-436	247.1 (273.4)(13)	0.87* (0.17)(13)

Comparison of length and weight between waters is limited by sample size. However, mature bull and westslope cutthroat trout were captured in both lakes (Tables 7 and 8). Based on Wr and K, most species evaluated appeared to have less than optimal body condition (Tables 7 and 8). Lake trout condition in Harrison Lake was similar to other lowland park lakes (Downs et. al. 2011), but considerably lower than Cosley Lake. A Wr of 100 represents the 75th percentile of average weight for a given length across a large number of populations for a particular species. In concept, a Wr of 100 is generally representative of good physiological and feeding conditions, and has been shown to be positively correlated with fat content in fish (Anderson and Neuman 1996, Renne and Verdon 2008) and prey

availability (Renne and Verdon 2008). Ellis et al. (1992) evaluated trophic status for a number of GNP lakes, including McDonald and St. Mary lakes and concluded all of the waters they sampled were either oligotrophic or ultra-oligotrophic. It would not be unexpected to find lower fish condition in such unproductive waters. This conclusion is supported by the findings of Stafford et al. (2002) who found lake trout from Lake McDonald grew considerably slower than those from the more productive waters of Flathead Lake.

ACKNOWLEDGEMENTS

The author wishes to thank Melvin Woody, Billy Brann, and Craig Stafford for assistance with gill netting. Mark Biel provided a constructive review of this report.

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Aquatic Invasive Species Prevention and Monitoring

ABSTRACT

In 2011 and 2012, Glacier National Park (GNP) implemented a program of boat inspection and launch permitting for all non-hand powered boats entering GNP. In 2012, GNP began an 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. Again, six boats were denied permits to launch for reasons ranging from standing water in the boat to dried vegetation adhered to the hull. However, no AIS were found on any of the boats inspected in either year. Boats entered GNP from 24 States and 3 Canadian Provinces over the two years. Fifteen of those twenty-four States have populations of zebra and/or quagga mussels, demonstrating the risk of introduction. In addition, we expanded 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 in 2011 and 2012, boaters were informed by signage and/or entrance gate staff that a boat inspection and launch permit was required for all boats launching within the park. The launch permit remained valid as long as the boat did not leave the park. Re-inspection was required upon re-entry into the park. Inspections/permits were 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 2010. The Crown of the Continent Learning Center at GNP developed a Resource Bulletin intended for the public addressing AIS threats to GNP (<http://www.nps.gov/glac/naturescience/ccrlc.htm>). The bulletin is brief (two pages) and is intended to provide key information regarding the status of AIS in GNP, as well as how the public can help protect the park from additional AIS. GNP also added AIS prevention content to its website (<http://www.nps.gov/glac/planyourvisit/outdooractivities.htm>) so boaters and other visitors would be exposed to the “clean, drain, dry” message before visiting the park. The AIS message was also incorporated into the Waterton-Glacier Guide brochure, available to the public. In addition, GNP 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.

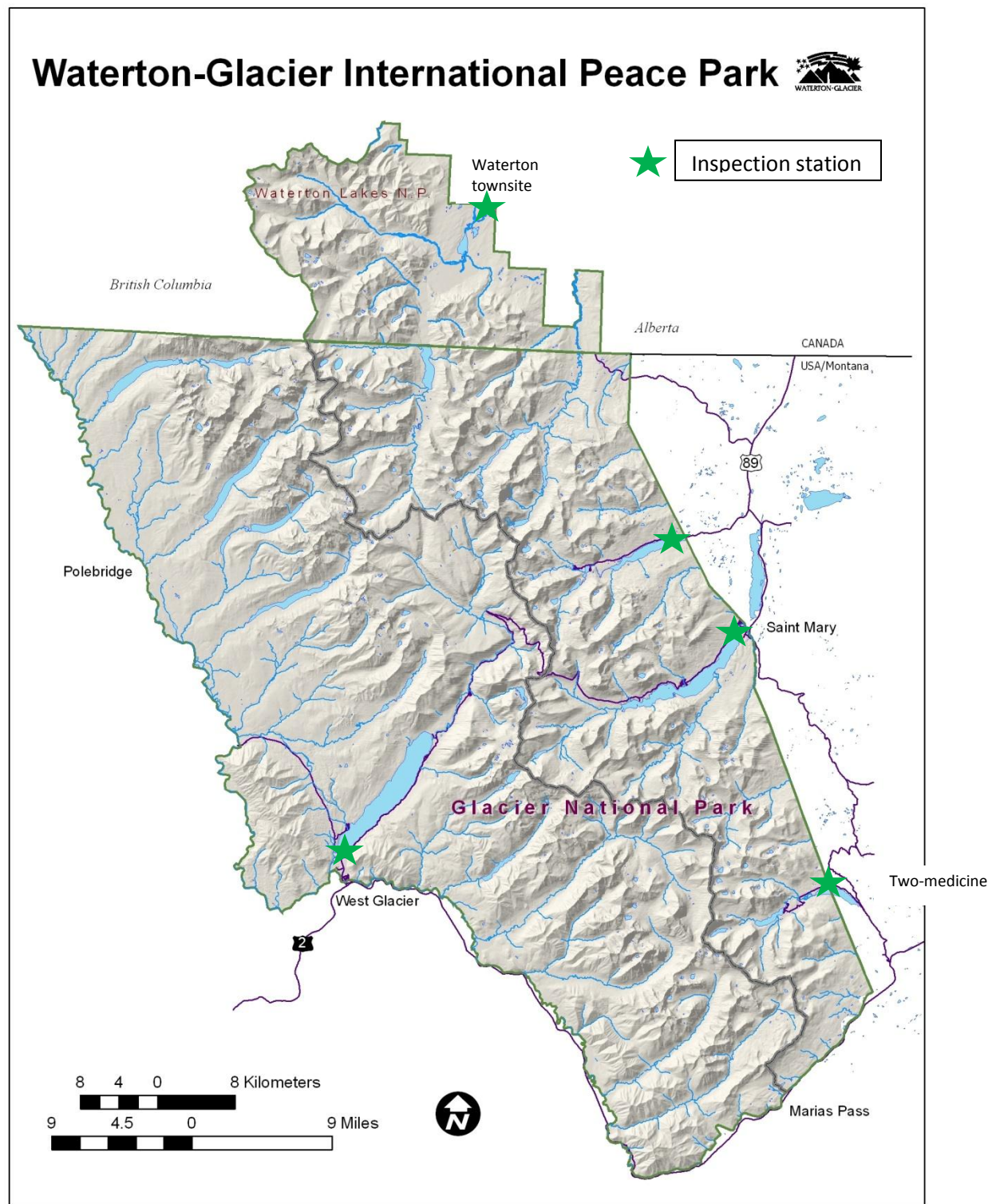


Figure 1. AIS boat inspection and launch permitting locations in Glacier National Park, 2011-2012.

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.

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).

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 over the two years. Seventy-eight percent of the boats were registered in Montana. Thirteen of the nineteen States have populations of zebra and/or quagga mussels, demonstrating the risk of introduction (Table 2).

Aquatic Invasive Species (AIS) Sampling 2011-2012 Waterton-Glacier International Peace Park

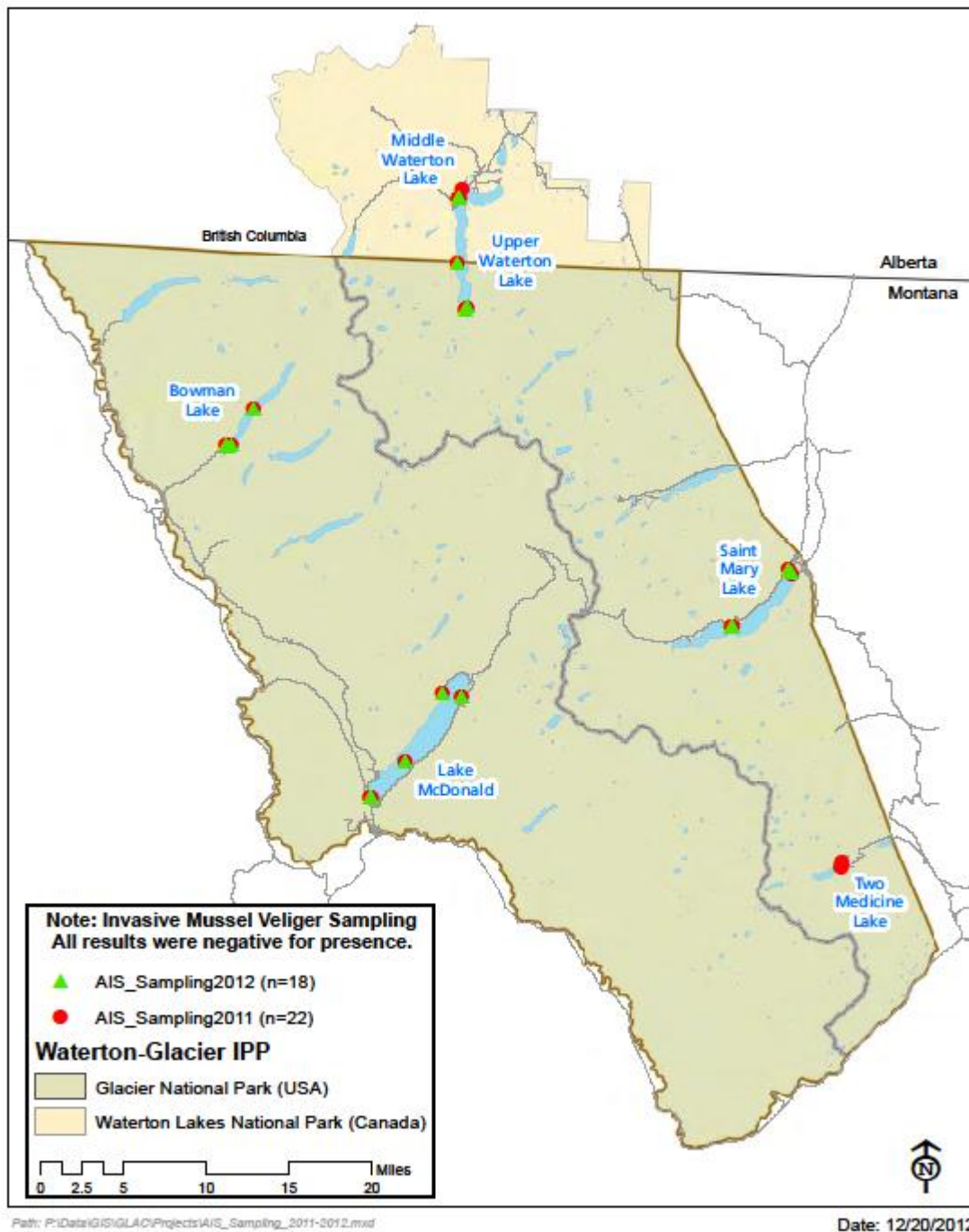


Figure 2. Invasive mussel larvae (veliger) sampling in Waterton-Glacier International Peace Park, 2011-2012.

Table 1. Invasive mussel veliger sampling locations in GNP, 2011-2012.

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	48.61980	-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	48.95850	-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
8/7/2012	48.74224	-113.45276	St. Mary Lake
8/6/2012	48.52895	-113.99020	Lake McDonald
8/6/2012	48.62322	-113.90263	Lake McDonald
8/6/2012	48.52990	-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

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

State	2011	2012	Total
Alabama	0	1	1
Arizona	8	8	16
California	19	14	33
Colorado	3	2	5
Illinois	1	2	3
Indiana	1	1	2
Michigan	0	1	1
Minnesota	5	3	8
Missouri	3	0	3
Nevada	1	0	1
New York	0	2	2
Oklahoma	0	1	1
Texas	1	2	3
Utah	1	3	4
Wisconsin	3	1	4

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 and 2012 (Figures 3-9). 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 occurred from August 6-11. 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. 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.

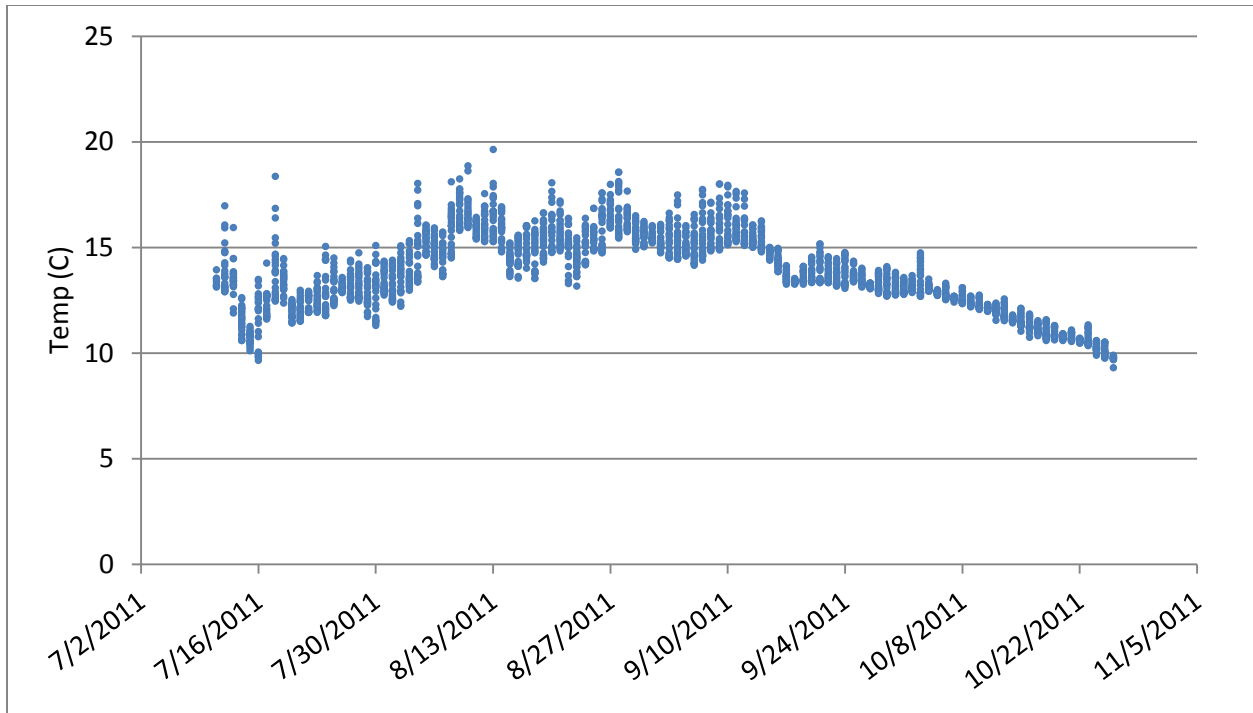


Figure 3. 2011 Bowman Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in approximately four feet of water.

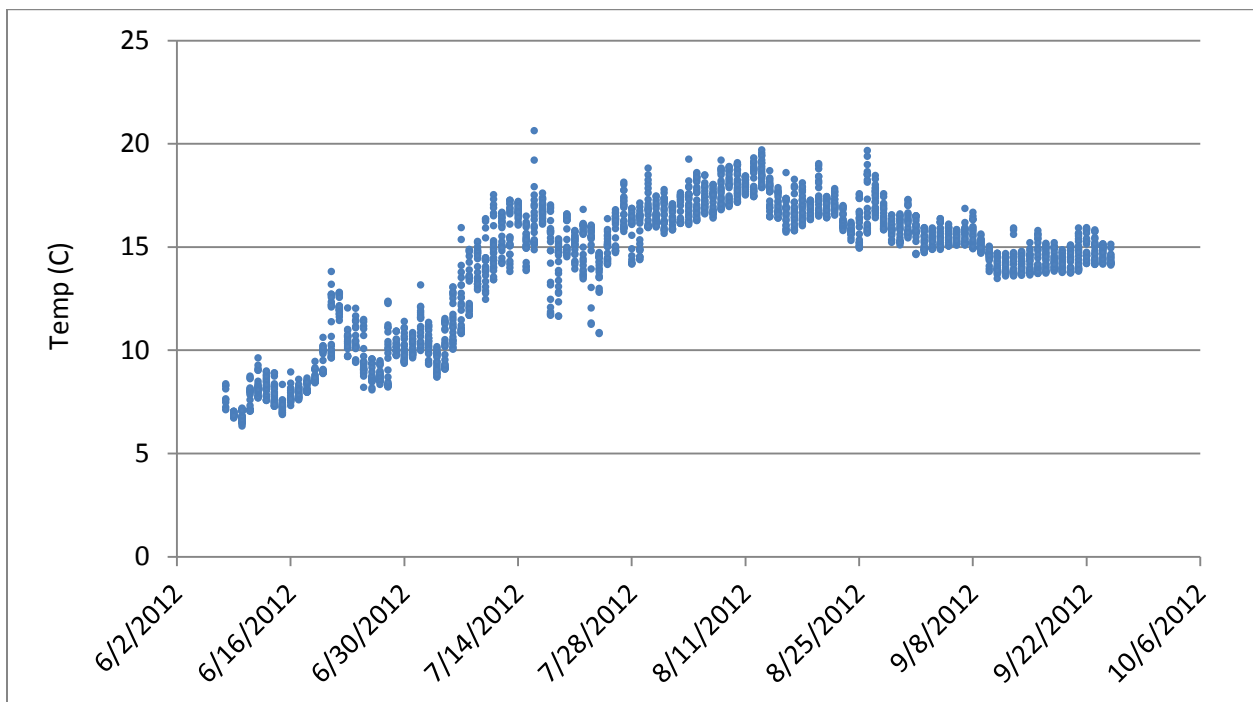


Figure 4. 2012 Bowman Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in approximately four feet of water.

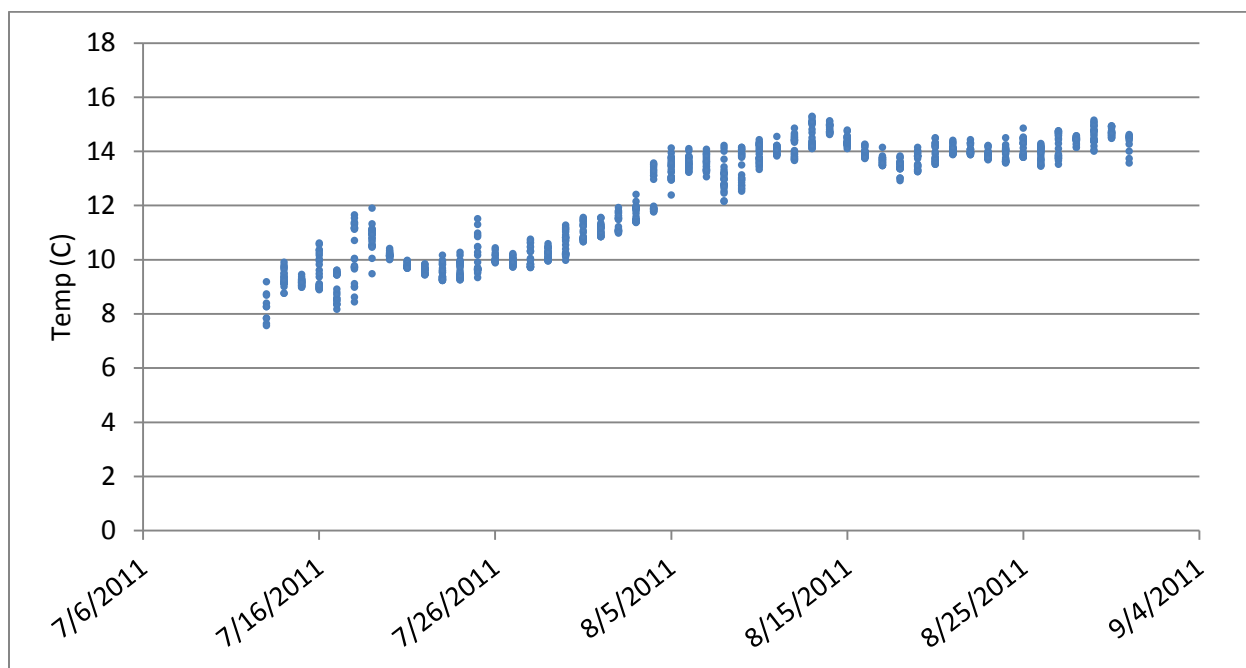


Figure 5. 2011 Two-Medicine Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock.

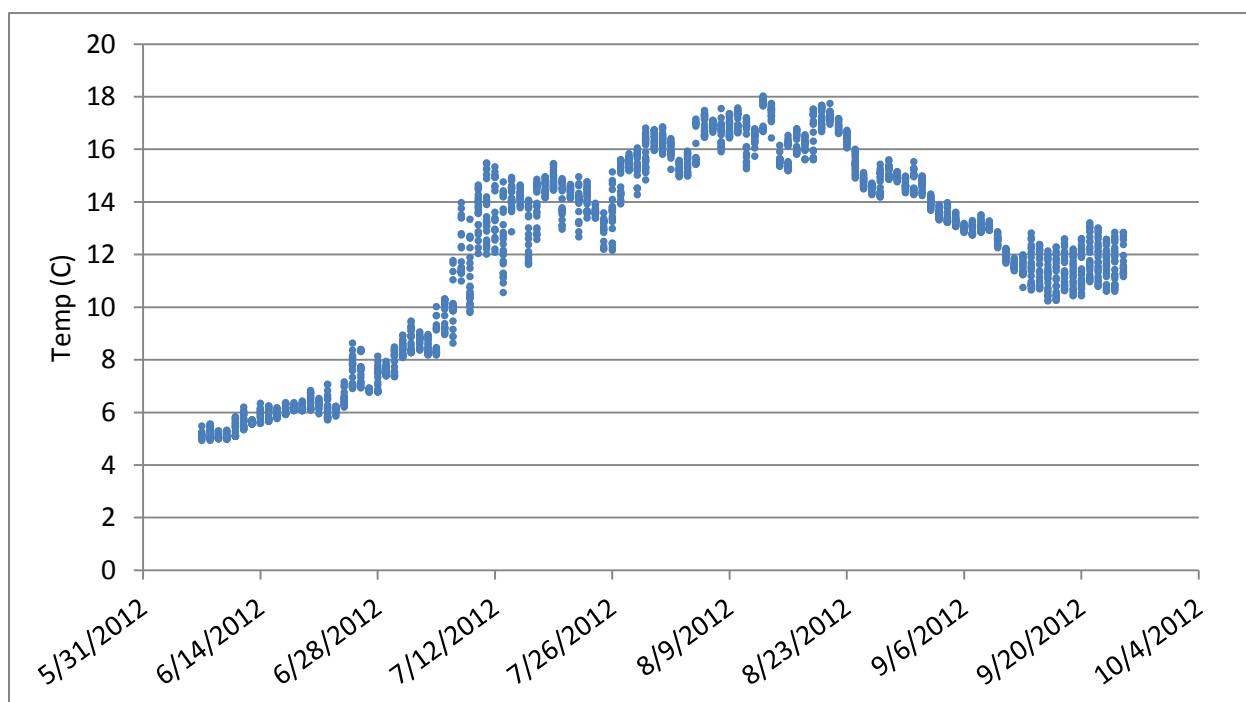


Figure 6. 2012 Two-Medicine Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock.

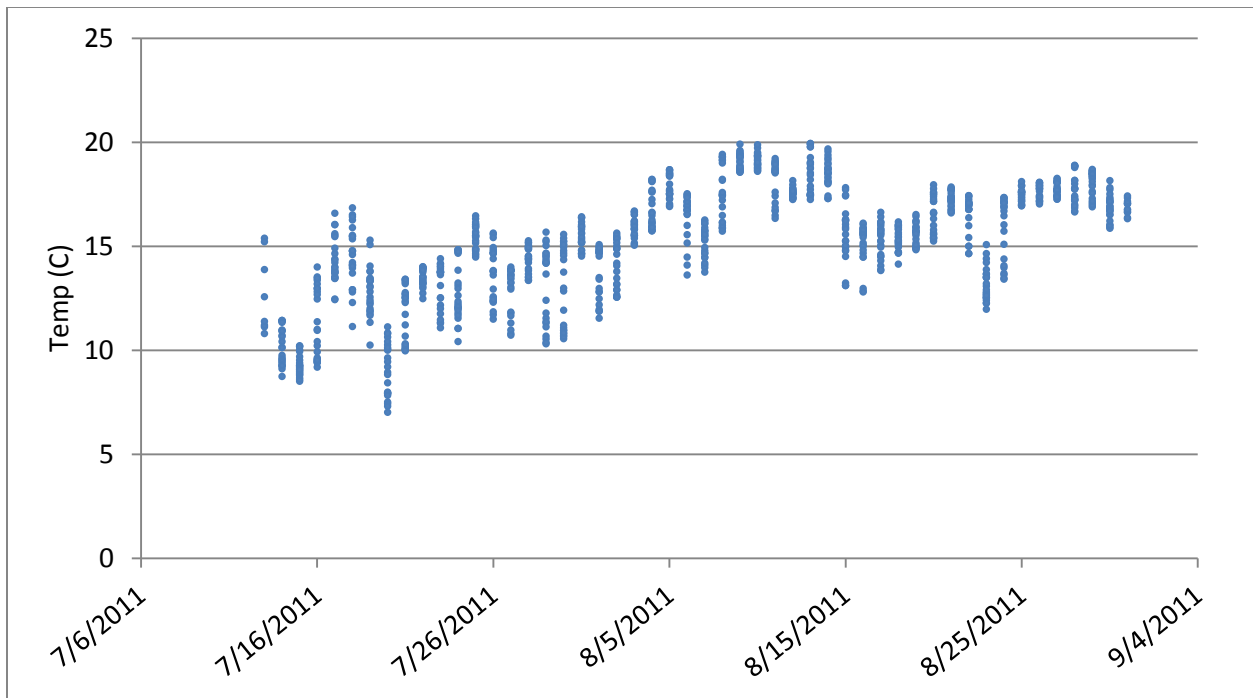


Figure 7. 2011 Lake McDonald Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock.

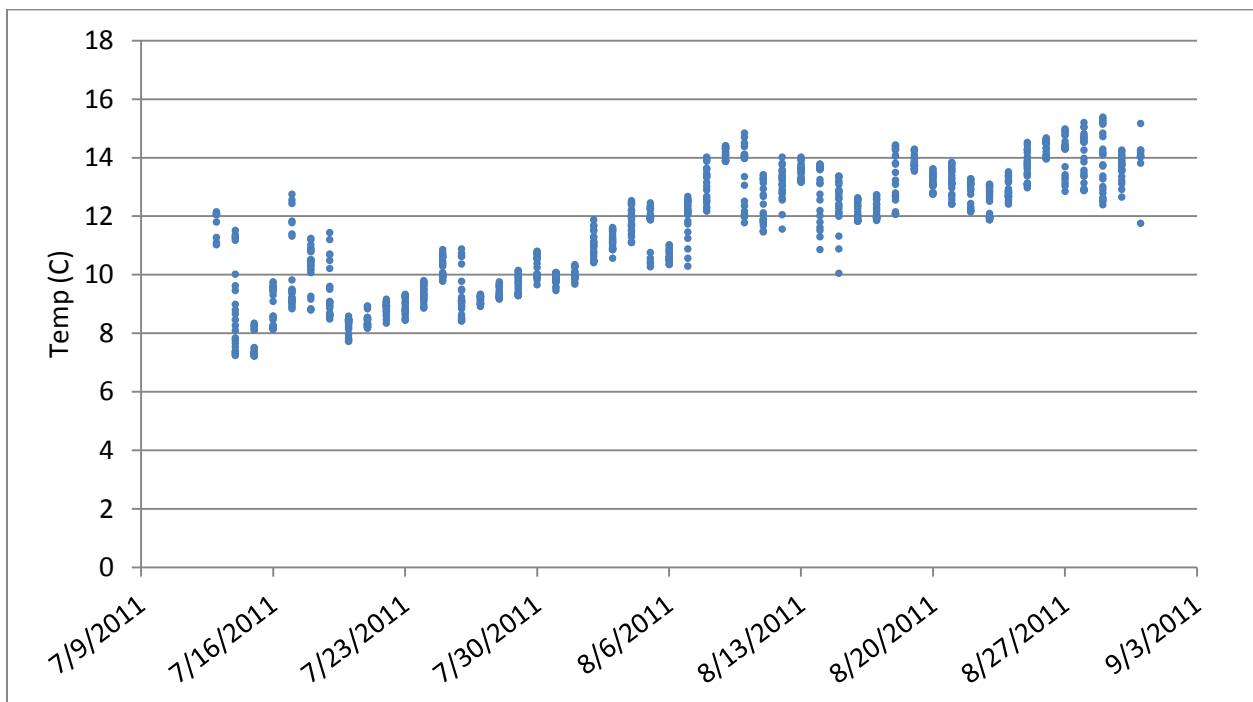


Figure 8. 2011 St. Mary Lake water temperature recorded near the lake bottom at the end of the boat dock.

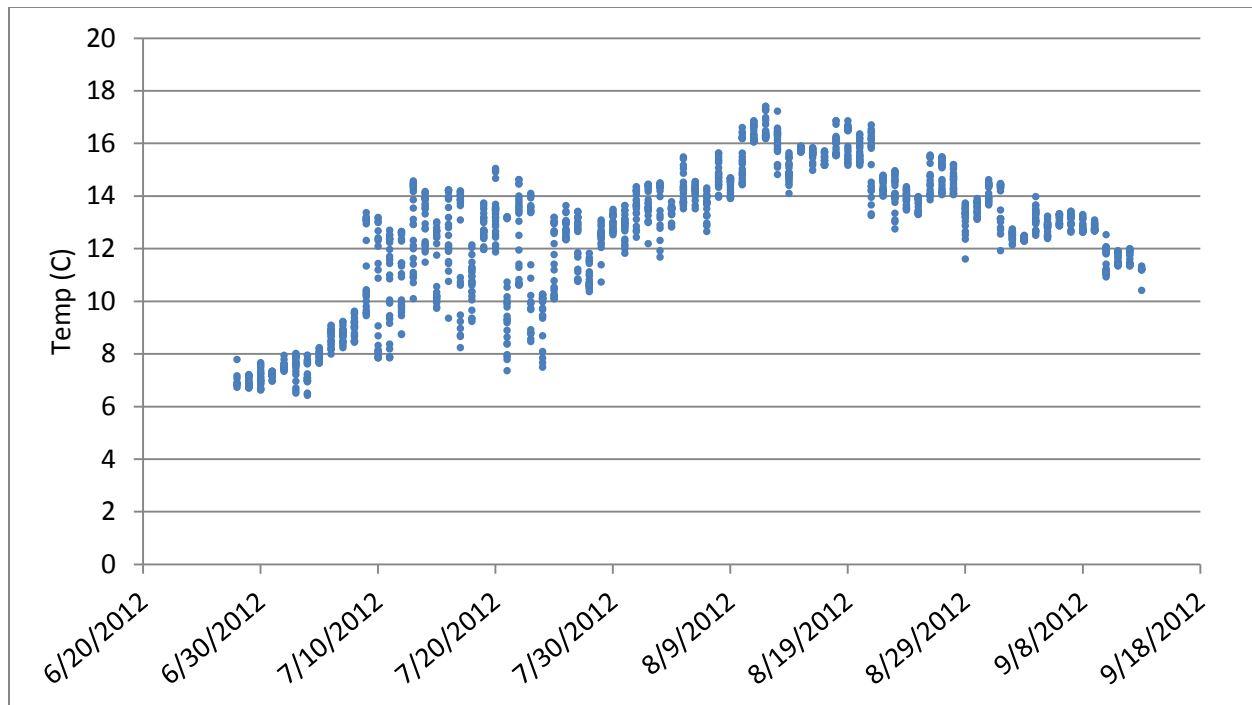


Figure 9. 2012 St. Mary Lake water temperature recorded near the lake bottom at the end of the boat dock.

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 is 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 watercraft (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

The authors wishes to thank fisheries technicians Melvin Woody and Billy Brann for assistance with field sampling. Mark Biel of the NPS provided constructive review of this report.

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Glacier National Park Bull Trout Redd Counts

ABSTRACT

We conducted bull trout *Salvelinus confluentus* redd counts in 11 streams/stream reaches in Glacier National Park in 2012. Five streams/stream reaches were surveyed in the N. Fk. Flathead River drainage, four were surveyed in the M. Fk. Flathead River drainage, and three were surveyed in the St. Mary River drainage. We counted a total of 218 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 33 redds. 2012 redd counts for Flathead Lake migratory bull trout populations spawning in the Middle Fork Flathead River tributaries in the park were at or above long-term averages. Redd counts for bull trout populations spawning in the St. Mary drainage were mixed, with Boulder and Lee creeks having strong counts while the count in Kennedy Creek continued to be 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 2012. 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 (Daniel 1990), to test for trends in “count year” versus “redd count” in the long-term redd count data set (Rieman and Myers 1997). We used tau-b to compensate for any bias caused by ties in the data, and noted statistical significance at the $\alpha = 0.05$ level (Rieman and Myers 1997).

RESULTS AND DISCUSSION

GNP, USGS, USFWS, and MFWP staff surveyed five stream reaches in the N. Fk. Flathead River drainage and four in the M. Fk. Flathead River drainage. In addition, three 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 (Figure 2; Appendix A). Redd counts in 2012 were average for Boulder Creek, and below average for Kennedy Creek. We recently initiated redd counts on Lee Creek and counted 31 redds, more than double the 2011 count of 15. 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 (-0.39; $p < 0.05$), 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.

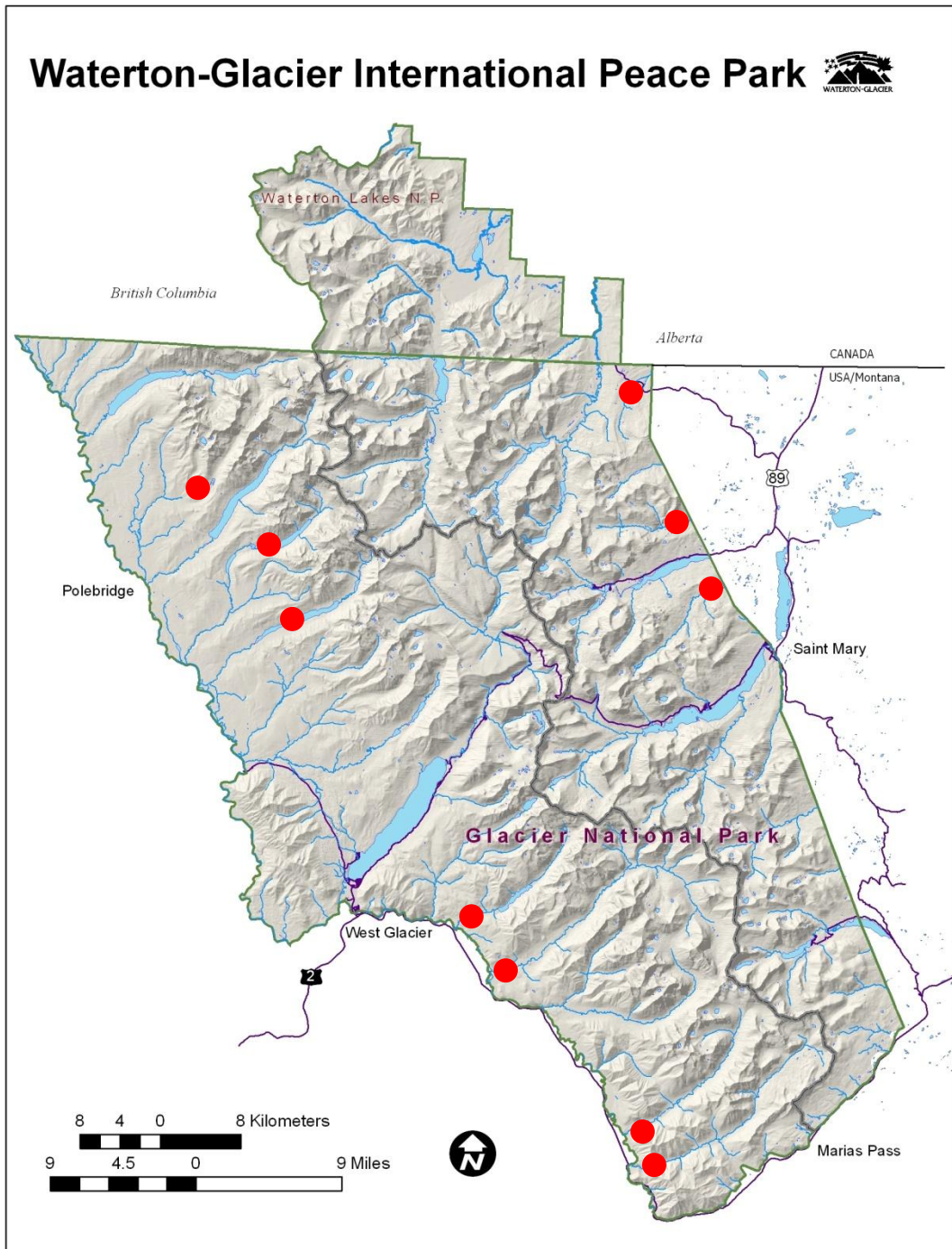


Figure 1. Drainages monitored for bull trout spawning activity (red circles) in Glacier National Park, Montana in 2012.

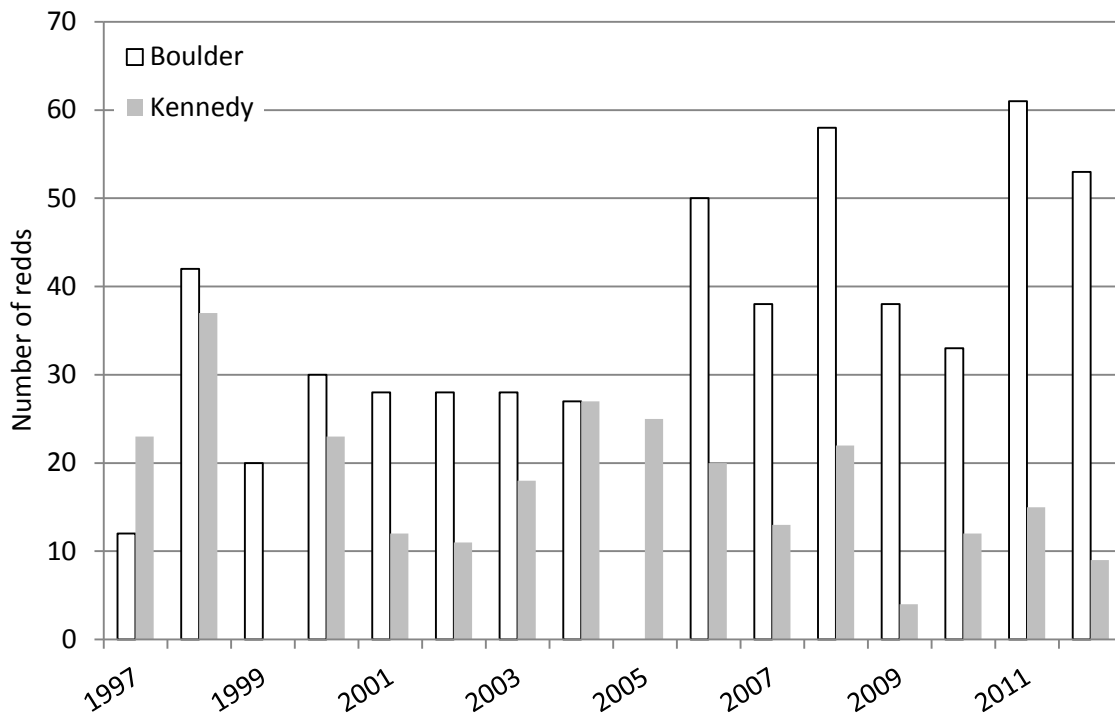


Figure 2. Bull trout redd counts for Boulder and Kennedy creeks, Hudson Bay Drainage, Glacier National Park.

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 dewateres Sherburne 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 loss of bull trout and other native fish from the system (Mogen and Kaeding 2001). The St. Mary Diversion Dam, used to provide water into the irrigation canal, creates an approximately 6’ high impediment to upstream migration of bull trout during the migration season (Mogen and Kaeding 2005). The BOR has recently initiated formal consultation with the USFWS and has formed stakeholder working groups to identify issues and develop alternatives for consideration in the National Environmental Policy Act (NEPA) process to address fishery issues associated with the Milk River Irrigation Project. Addressing the fishery impacts of this project will significantly improve migration conditions as well as survival of migratory bull trout.

On the west side of GNP, both migratory stocks of bull trout from Flathead Lake as well as populations that reside entirely within the park (known locally as “disjunct” migratory populations) are monitored (Appendix A). Flathead Lake migratory bull trout stocks underwent dramatic declines starting in about 1990, and declines are believed to have been the result of the introduction of Mysis shrimp *Mysis relicta* into the system and resulting major alterations in trophic dynamics (i.e. rapidly expanding lake trout population) in the lake, as well as drought conditions (Weaver et al. 2006). As the spawning and stream rearing habitats for the Flathead Lake populations that use GNP are largely located within the park, the traditional land-use threats to habitat quality (i.e. road building, timber harvest, residential development) are not the primary issue of concern for these individual populations. One of the most significant contemporary threats to these populations is predation with and competition by non-native fish species in both the migratory 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 2012 redd count for Ole Creek of 53 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 = 0.09, $p > 0.05$), but over the last 10 years there has been a statistically significant positive trend in spawner abundance in Ole Creek (tau-b = 0.69, $p > 0.05$) (Figure 3, Appendix A). The 2012 redd count on both Nyack and Park creeks were very similar to their long-term average (Figure 3, Appendix A). Although the redd count trend correlations remain negative, no statistically significant trend was detected in the long-term redd count data set for Nyack (tau-b = -0.25, $p > 0.05$) or Park creeks (-0.10, $p > 0.05$). 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.

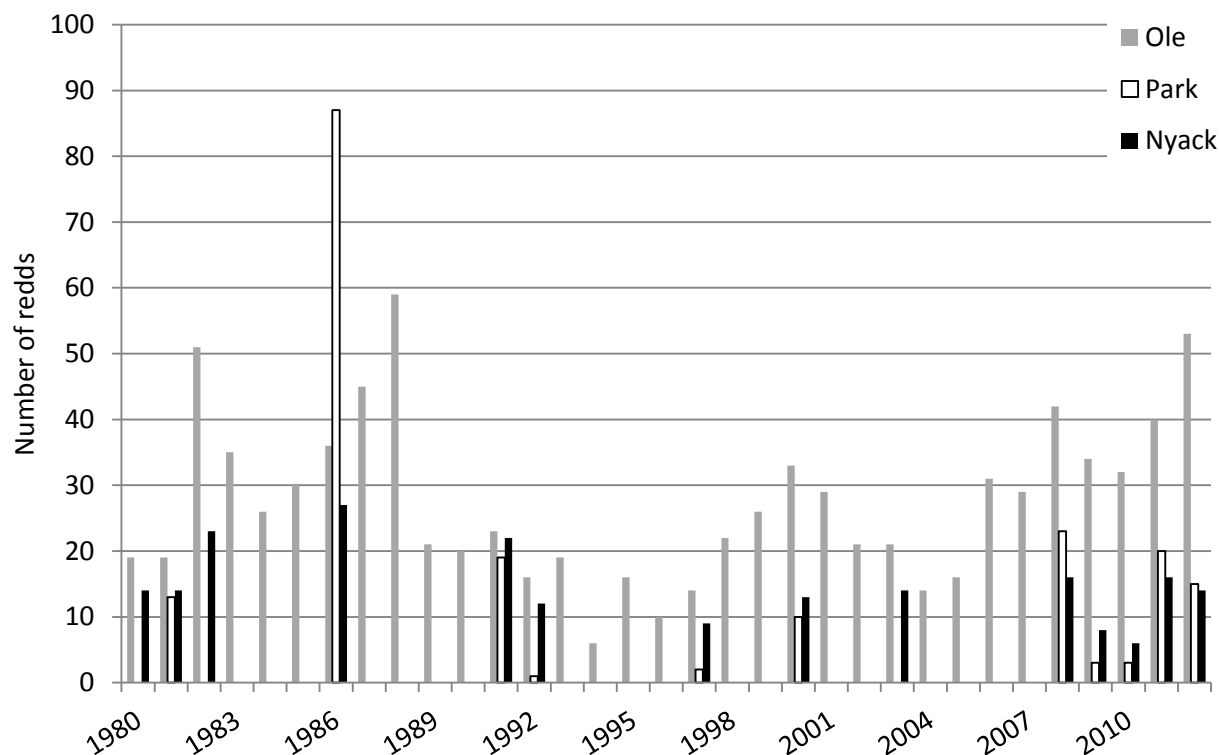


Figure 3. Bull trout redd counts conducted in Ole, Park, and Nyack creeks, Middle Fork Flathead River Drainage, Glacier National Park.

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.

Redd counts on Quartz Creek, the primary spawning area for bull trout from Quartz Lake have not shown a statistically significant trend ($\tau\text{-}b = 0.02$, $p > 0.05$) since monitoring began, suggesting lake trout have yet to reach levels where they are impacting bull trout population viability (Figure 4, Appendix A). Similarly, bull trout redd counts have remained fairly constant since the experimental suppression program began, suggesting gill net bycatch mortality of bull trout is not having a measurable impact on spawning adult abundance.

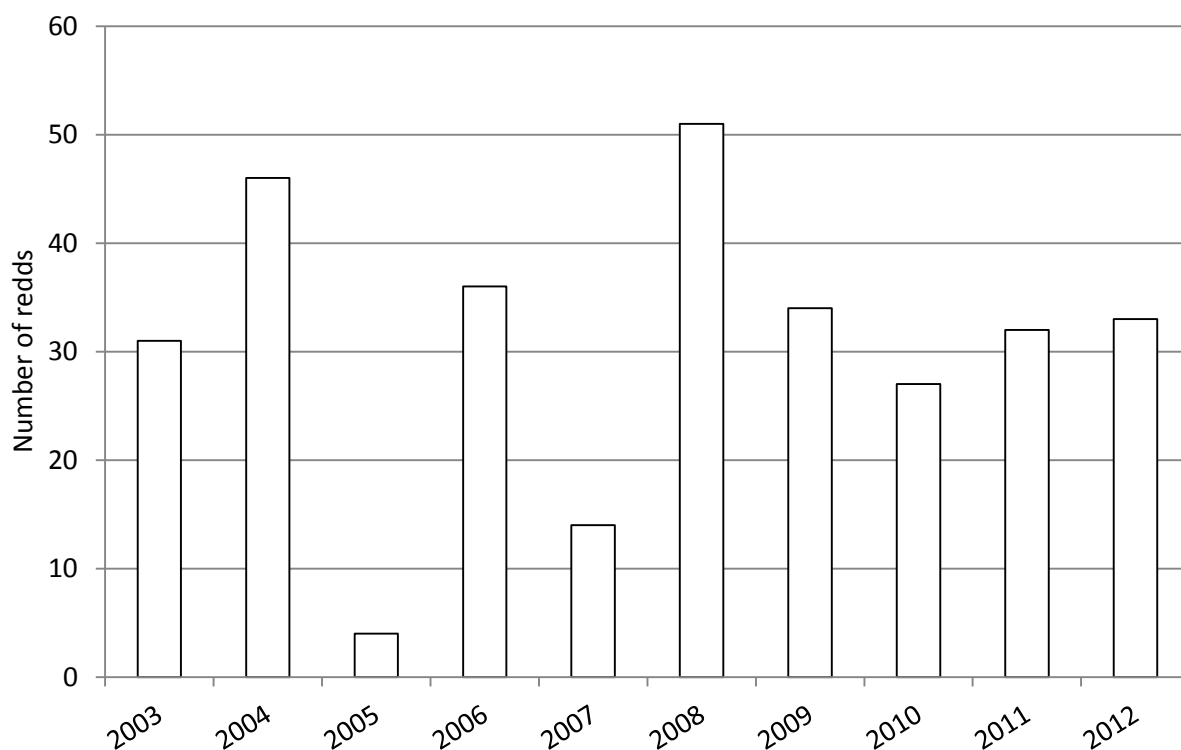


Figure 4. Bull trout redd counts in Quartz Creek, Glacier National Park, Montana.

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.

ACKNOWLEDGEMENTS

Thanks to Jim Mogen of the USFWS, Toby Tabor of the Blackfeet Nation Fish and Wildlife Department, John Fraley and John Cavigli of Montana Fish, Wildlife, and Parks, Clint Muhlfeld and Joe Giersch of the USGS, and Ben Galloway of Montana State University, as well as Phil Wilson and John Waller of GNP for their assistance with redd counts. The NPS IMRICO Program provided partial funding for this work. Mark Biel of the NPS provided constructive review of this report.

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

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

Stream	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996
<i>Hudson Bay Drainage</i>																	
Boulder Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Kennedy Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
<i>N. Fk. Flathead Drainage</i>																	
Akokala Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Agassi Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Bowman Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Harrison Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Jefferson Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Logging Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (lower)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (middle)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (upper)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Rainbow Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Upper Kintla outlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	52	--	--
Upper Kintla inlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
<i>M. Fk. Flathead Drainage</i>																	
Ole Cr.	19	19	51	35	26	30	36	45	59	21	20	23	16	19	6	16	10
Nyack Cr.	14	14	23	--	--	--	27	--	--	--	--	22	12	--	--	--	--
Park Cr.	--	13	0	--	--	--	87	--	--	--	--	19	1	--	--	--	--
Starvation Cr.	1	1	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--

Table A.1. Continued.

Stream	1997	1998	1999	2000	2001	2002	2003	2004 f	2005 b,f	2006	2007 a,c	2008 c,d,e	2009 g	2010 h	2011	2012
Hudson Bay Drainage																
Boulder Cr.	12	42	20	30	28	28	28	27	--	50	38	58	38	33	61	53
Kennedy Cr.	23	37	--	23	12	11	18	27	25	20	13	22	4	12	15	9
Lee Cr.															15	31
N. Fk. Flathead Drainage																
Akokala Cr.	--	--	--	--	--	--	--	--	--	--	--	11	6	1	4	5
Agassi Cr.	--	--	--	--	--	--	--	--	--	--	--	0	--	--	--	--
Bowman Cr.	--	--	--	--	--	0	0	0	0	2	1	0	0	1	3	--
Harrison Cr.	--	--	--	--	--	--	--	4	0	8	15	14	1	6	--	1
Jefferson Cr.	--	--	--	--	--	--	--	--	--	--	--	0	0	--	--	--
Logging Cr.	--	--	--	--	--	--	--	3	20	0	--	5	0	3	3	1
Quartz Cr. (lower)	--	--	--	--	--	--	--	1	3	2	2	3	2	2	5	3
Quartz Cr. (middle)	--	--	--	--	--	--	0	0	0	0	0	0	3	0	--	--
Quartz Cr. (upper)	--	--	--	--	--	--	31	46	4	36	14	51	34	27	32	33
Rainbow Cr.	--	--	--	--	--	--	--	--	--	--	--	28	12	4	9	--
Upper Kintla outlet	--	--	--	--	--	--	--	--	--	--	--	0	--	25	--	--
Upper Kintla inlet	--	--	--	--	--	--	--	--	--	--	--	0	--	--	--	--
M. Fk. Flathead Drainage																
Ole Cr.	14	22	26	33	29	21	21	14	16	31	29	42	34	32	40	53
Nyack Cr.	9	--	--	13	--	--	14	--	--	--	--	16	8	6	16	14
Park Cr.	2	--	--	10	--	--	0	--	--	--	--	23	3	3	20	15
Starvation Cr.	0	--	--	0	--	--	--	--	--	--	--	--	--	--	--	--

a = spawning activity on Upper Quartz likely inhibited by weir at mouth.

b = minimum count due to high flows in Upper Quartz.

c = count accuracy may have been compromised due to kokanee spawning activity in Harrison.

d = cumulative count based on multiple survey events in Upper Quartz.

e = count conducted by helicopter on Park.

f = minimum count on Ole as high flows may have obliterated some redds.

g = Kennedy count does not include 3 additional redds counted upstream of the index section, or 2 redds counted in unnamed tributary flowing from Yellow Mountain.

h=very poor visibility in Bowman Creek due to glacial runoff from Jefferson Creek

