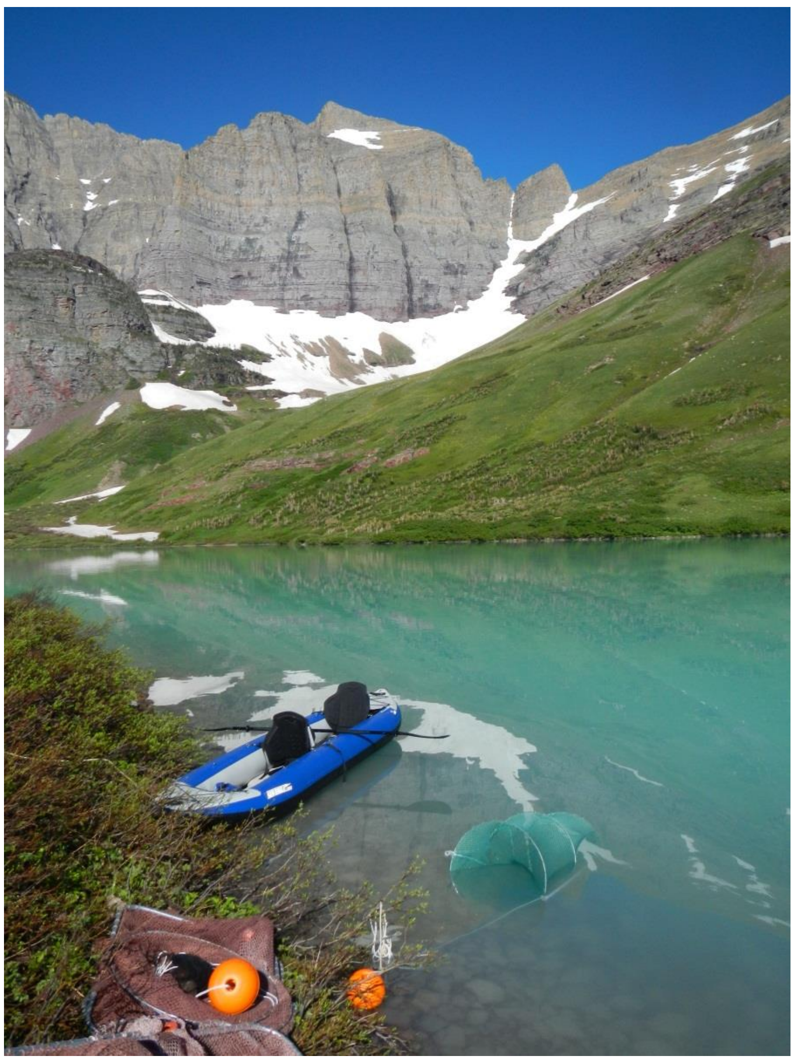
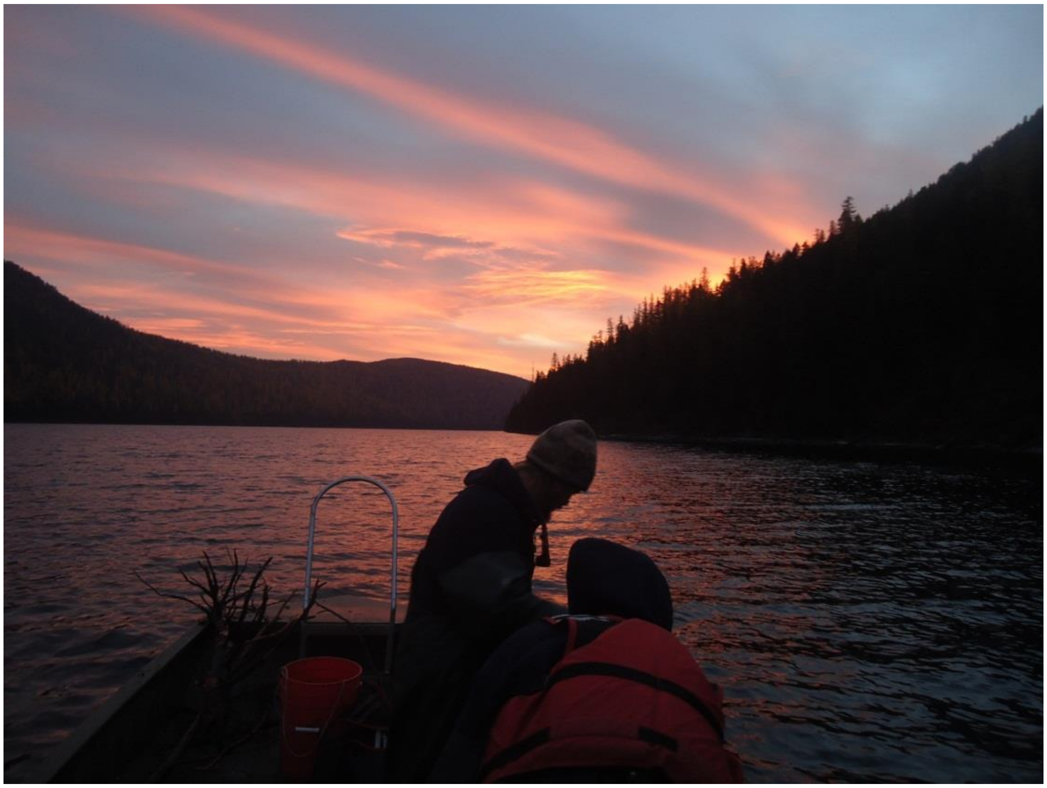
Glacier National Park Fisheries Monitoring and Management

Program Report 2014





Prepared by:

Christopher C. Downs[1], Carter Fredenberg[1], V. D'Angelo[2], C. Muhfeld[2] and Brian McKeon[1]
National Park Service, West Glacier, Montana
January 2015

Suggested citation:

Downs, C.C.[1], C.R. Fredenberg[1], V. D'Angelo[2], C. Muhlfeld[2], and B. McKeon[1]. 2015. Glacier National Park Fisheries Monitoring and Management Report, 2014. National Park Service, West Glacier, Montana.

[1] National Park Service, Glacier National Park, West Glacier, MT 59901
[2] U.S. Geological Survey, Northern Rockies Science Center, Glacier Field Station, West Glacier, MT 59901

Front cover photo caption: Fyke net survey on Cracker Lake.

Inside photo captions (top to bottom): Collecting data after netting, fish passage barrier on Quartz Creek, snorkel survey in classic west side creek, and gillnetting lake trout on Quartz Lake.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF TABLES

LIST OF FIGURES

Native Fish Population Monitoring

LIST OF TABLES

LIST OF FIGURES

LIST OF TABLES

LIST OF FIGURES

Glacier National Park Bull Trout Redd Counts

LIST OF FIGURES

APPENDIX A

List of Tables

Quartz Lake Lake Trout Suppression Project

ABSTRACT

Glacier National Park (GNP) supports approximately one-third of the remaining natural lake core habitat areas supporting threatened bull trout *Salvelinus confluentus*. However, bull trout populations have recently declined and are at high risk of extirpation in several lakes in western GNP due to the establishment of invasive lake trout. In 2009, the U.S. Geological Survey and the National Park Service began suppressing lake trout in Quartz Lake (352 ha) to reduce impacts to native bull trout. In 2014, we removed a total of 1,640 lake trout. Both adult and juvenile lake trout gill net catches declined in 2014, suggesting lake trout abundance in Quartz Lake is declining. Similarly, angler catch rates for lake trout continue to decline while bull trout catch rates remain relatively stable. Lake trout population size structure continues to decline. A record bull trout redd count was observed in 2014. Each of these metrics suggests lake trout suppression is achieving the intended objective of reducing lake trout abundance to protect bull trout and other native fish.

INTRODUCTION

Invasive species threaten the biodiversity of aquatic ecosystems worldwide, and are considered the second greatest threat to biodiversity in North America (Vitousek et al. 1997). Invasions of introduced taxa often disrupt the structure and function of ecosystems, reduce biological diversity among native species, and impose huge economic costs. Therefore, understanding how to reduce or eliminate exotic species in native ecosystems is critical for conserving native aquatic species and ecosystems.

Lake trout are large, long-lived, top-level predators native to deep, cold, oligotrophic lakes of Canada and northern parts of the United States, including the Great Lakes (Behnke 2002). During the late 19th and early 20th century, lake trout were widely introduced into lakes and reservoirs outside their native range (Crossman 1995). More recently, the species has expanded its range in the western United States through dispersal and unauthorized translocations (Behnke 2002). Bull trout and lake trout have similar morphologies, diets, and growth rates, and occupy similar trophic positions as top-level predators, resulting in strong potential for competitive or predatory interactions. In the Pacific Northwest, native bull trout populations have declined in all systems where lake trout have been introduced (Donald and Alger 1993; Fredenberg 2002; Martinez et al. 2009).

Introduced to Flathead Lake in 1905, non-native lake trout have dispersed to many additional lakes in the wider Flathead watershed, including nine lakes west of the Continental Divide in GNP that historically supported bull trout (Fredenberg 2002; Downs et al. 2011). Fredenberg (2002) reported a broad bull trout decline and a corresponding increase in non-native lake trout from 1969 to 2000 in four lakes (Logging, Bowman, Harrison, Kintla) in GNP. In the same study, bull trout catch data in Quartz Lake remained constant because it was the only lake where lake trout were absent (Fredenberg 2002). Similarly, recent bull trout redd surveys (2003-2010) show significant declines in adult abundance in many GNP streams, with some counts representing less than 10 reproducing individuals (Downs et al. 2011). Combined, these data show that several native bull trout populations have drastically declined in western GNP with several adfluvial populations at imminent risk of extirpation due to adverse interactions with invasive lake trout.

STUDY AREA

Quartz Lake is a glacially formed lake located in the headwaters of the Columbia River Basin, Montana. Quartz Lake is the fifth largest lake west of the Continental Divide in GNP (Fredenberg et al. 2007), with a surface area of 352 ha and a maximum depth of 83 m (Figure 1). The lake is at an elevation of 1,346 m and is positioned in a narrow glaciated valley that is supplied by perennial flow from snow and glacial runoff from the Lewis Range. The limnetic zone substrate is dominated by a mixture of cobble and boulder. Quartz is an oligotrophic, dimictic lake with stratification occurring in late June and destratification in early October. Three avalanche chutes along the northern shore line have deposited large angular cobble substrates during the spring and winter months.

The native fish assemblage in the Quartz drainage consists of bull trout, westslope cutthroat trout *Oncorhynchus clarkii lewisi*, mountain whitefish *Prosopium williamsoni*, longnose sucker *Catostomus catostomus*, largescale sucker *Catostomus macrocheilus*, slimy sculpin *Cottus cognatus*, and redside shiner *Richardsonius balteatus*. The lake trout is the only non-native fish species in the drainage and was first detected in Lower Quartz Lake in 2003 and in Quartz Lake in 2005 (Fredenberg et al. 2007).

There are no natural putative fish barriers in the Quartz drainage, although several high gradient cascades serve as potential intermittent barriers between Lower Quartz and Middle Quartz lakes. The discovery of lake trout in Lower Quartz Lake prompted construction of an artificial gabion barrier approximately 100 m downstream of Middle Quartz Lake in 2004 to conserve the upstream native fish assemblages in Middle Quartz, Quartz, and Cerulean lakes. Unfortunately, lake trout were detected in Quartz Lake in 2005 before the barrier was completed. In 2006, the structure was damaged by high water, and served as a partial barrier to fish dispersal through 2012. The long term functionality of the barrier was critical to the success of lake trout suppression in Quartz Lake, and consequently, the NPS repaired the structure in 2012 to deter further invasions.

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

METHODS

Telemetry

During spring gillnetting, experienced lake trout anglers attempted to capture adult sized lake trout to implant with sonic telemetry tags to aid in determining the location and timing of the spawn. Anglers were asked to record the number of hours fished and the species caught during all angling activity. Tag implantation and telemetry methods were identical to those described in Fredenberg (2014).

Gillnetting

All lake trout captured during 2014 were measured for total length (TL, mm). We recorded weight (g) and removed otoliths and genetic samples from all mature lake trout and approximately 10 fish per cm interval (when present) during spring gillnetting. Additionally, otoliths, genetic samples, and weight were taken from all bull trout and westslope cutthroat trout bycatch mortalities. Otoliths were cleaned and prepped for dry storage similar to the methods reported in Stafford et al. (2002). All lake trout were killed and their airbladders were punctured before they were returned to the lake as biomass.

Spring

The 2014 lake trout suppression project followed the same general schedule conducted in 2010-2013. Gillnetting was comprised of two specific netting periods; the spring (27 May-25 June) and fall (2 October-28 October). During spring gillnetting, crews netted from Monday-Friday for a five week period. Juvenile and subadult lake trout were captured using randomly placed small mesh (19 mm, 26 mm, and 32 mm bar measure) 91 m monofilament gill-nets. Nets were set at all hours of the day and night and were generally deployed for 4-6 hours in depths greater than 30 m. The short deployment interval and deeper net sets were purposefully used to reduce incidental bycatch mortality of bull trout and westslope cutthroat trout. In previous years 28 mm bar mesh nets were used, however, selectivity analysis in 2013 revealed the peak length selected for by the mesh sizes used overlapped, therefore, the use of 28 mm bar measure mesh was stopped in 2014. In 2013, catch rates of short (33 m) experimental 19 mm bar measure gill-nets with the standard twine diameter of 0.20 mm were compared to catch rates of 19 mm bar measure gill-nets containing finer 0.10 mm twine diameter (Fredenberg 2014). It was determined the finer 0.10 mm twine diameter had a significantly higher catch per unit effort than the 0.20 mm twine diameter (Fredenberg 2014). Therefore, in 2014, full length (91 m) nets consisting of 19 mm bar mesh and 0.10 mm twine diameters were incorporated into the fleet of nets used in the spring. Spring netting objectives were: (1) to remove as many juvenile lake trout as possible and (2) to implant sonic telemetry tags into mature size lake trout.

Fall

The second netting event (fall netting) targeted mature adult lake trout on the spawning grounds, which were identified by radio tagged adult lake trout 2009-2014. Fall netting in 2014 occurred from 2 October-28 October and was nearly continuous. Adult lake trout congregated on the two primary spawning locations, identical to those observed 2009-2013 (Fredenberg 2014), and were captured with large mesh (57 mm, 64 mm, and 70 mm bar measure), 91 m monofilament gill-nets. Fall netting objectives were: (1) to identify the timing and location of lake trout spawning and (2) to remove as many adults as possible, thereby reducing the potential for further recruitment.

RESULTS AND DISCUSSION

Telemetry

Sonic tags were surgically implanted in four presumably mature lake trout caught by experienced lake trout anglers during spring 2014. An additional fish tagged during 2012 was tracked in the fall of 2014 bringing the total number of tagged fish to five. Lake trout were relocated throughout October, and four of five (80%) sonic tagged adult lake trout were recaptured with gill nets. The one fish that was not recaptured expired after tagging in the spring of 2014. As the suppression years have progressed from 2010-2014, anglers have found it increasingly difficult to capture mature size lake trout for implantation of sonic radio tags. While lake trout catch per angling hour has drastically declined since 2010, the bull trout catch per angler hour has remained generally stable, or increased in 2014 (Figure 2). Although this is largely observational data, it further corroborates the adult gillnetting data, suggesting we have successfully removed large portions of the mature lake trout population via targeted fall netting.

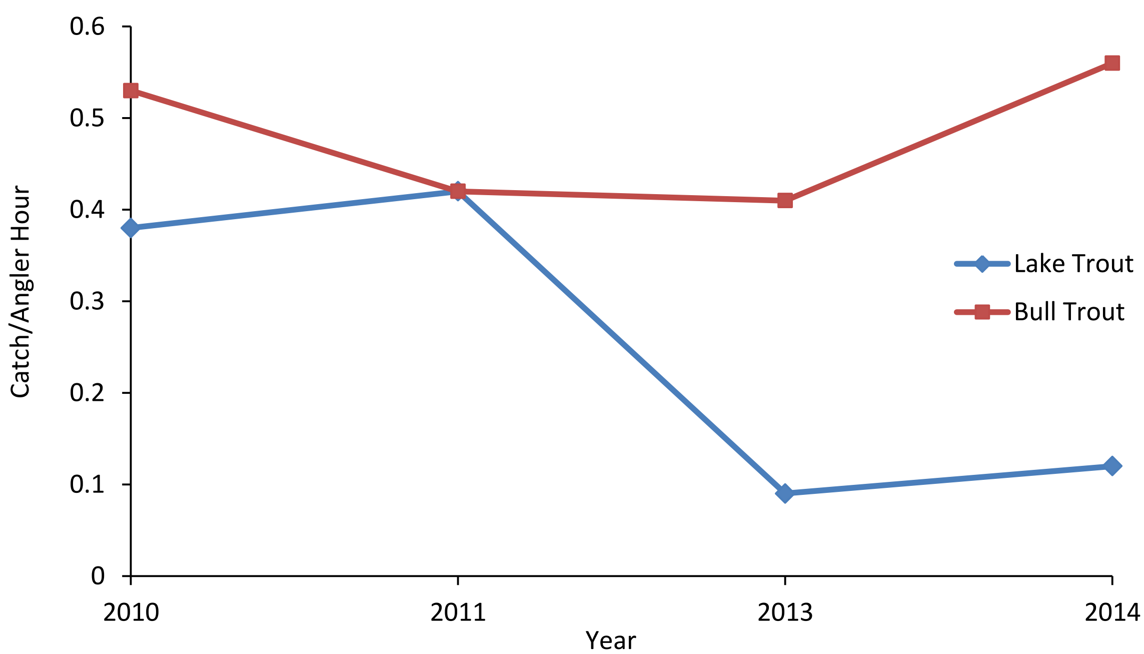


Figure 2. Catch per angler hour for lake trout and bull trout in Quartz Lake, Montana 2010-2014.

In addition to the two known spawning locations described in Fredenberg (2014), a third area where tagged fish were often relocated along the southern shore was netted. Although sonic tagged adults were shown to frequent the area, observation of the substrate in this area indicated spawning was not likely to occur. The frequent relocations of telemetered fish in this area are not understood. However, upon removal and examination of lake trout from this area, stomach contents revealed a high percentage of the lake trout had prey fish in their stomach or gullet. Therefore, it is theorized that this area is occupied by large schools of prey fish and may be a feeding area.

Gillnetting

Approximately 35.1 km of nets were deployed in 2014, including 17.8 km during the spring and 17.2 km during the fall, resulting in a total of 1,640 lake trout being removed (1,614 juveniles, 26 adults).

Although the number of days fished has remained similar, the kilometers of net deployed has varied from year to year. During spring netting in 2014 crews deployed the fourth lowest amount of net; in contrast, fall netting was the third highest amount of net deployed (Table 1). The low quantity of net deployed in the spring is likely a function of the sheer volume of fish and the time needed to pick the fish out of the fragile 0.10 mm twine diameter 19 mm bar measure nets.

Table 1. Annual kilometers of net set during spring and fall gillnetting from 2009-2104 in Quartz Lake, Montana.

Year	Spring	Fall	Total
2009	--	38	38
2010	23.3	27.3	50.6
2011	28.1	12.8	40.9
2012	18.7	10.2	28.9
2013	17.6	14.0	31.6
2014	17.8	17.3	35.1

Spring

Spring netting removed a total of 1,608 juvenile lake trout. Spring gillnetting was highly successful, the addition of 19 mm bar mesh nets containing twine diameters of 0.10 mm vastly increased juvenile catch from previous years (Figure 3). The cumulative length distribution of juveniles caught from 2010-2014 suggests large proportions of the juvenile fish targeted with the 26 mm and 32 mm bar measure nets have been removed, and the size distribution of juvenile lake trout has continued to shift, with the majority of the catch comprised of smaller younger individuals (Figure 4). In addition to this shift, the incorporation of the smaller 19 mm bar measure nets with 0.10 mm twine diameter vastly increased the catch of juvenile lake trout and extended the length range harvestable to smaller individuals in 2013 and 2014. Of the juvenile lake trout caught in 2014, the 19 mm bar mesh nets captured approximately 89% (*N* = 1,436) while the 26 mm and 32 mm bar measure nets caught 9% (*N* = 137), and 2% (*N* = 35), respectively.

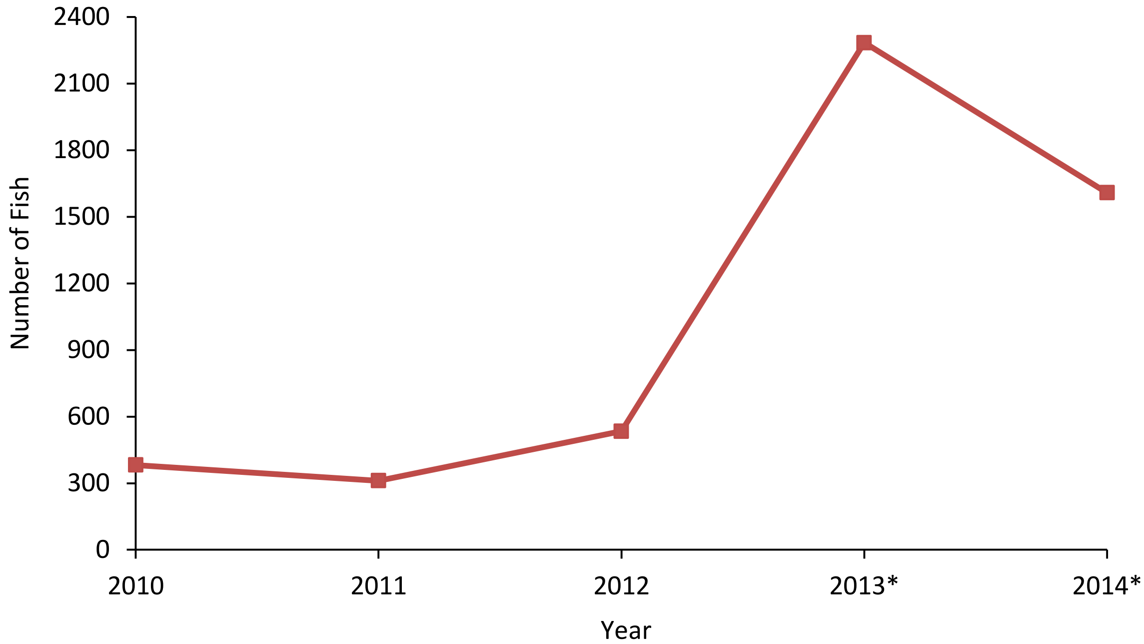


Figure 3. Number of lake trout caught annually (2010-2014) during spring netting in Quartz Lake, Montana. Net effectiveness improved starting in 2013 with the use of gill-nets containing 0.10 mm twine diameter and 19 mm bar measure.

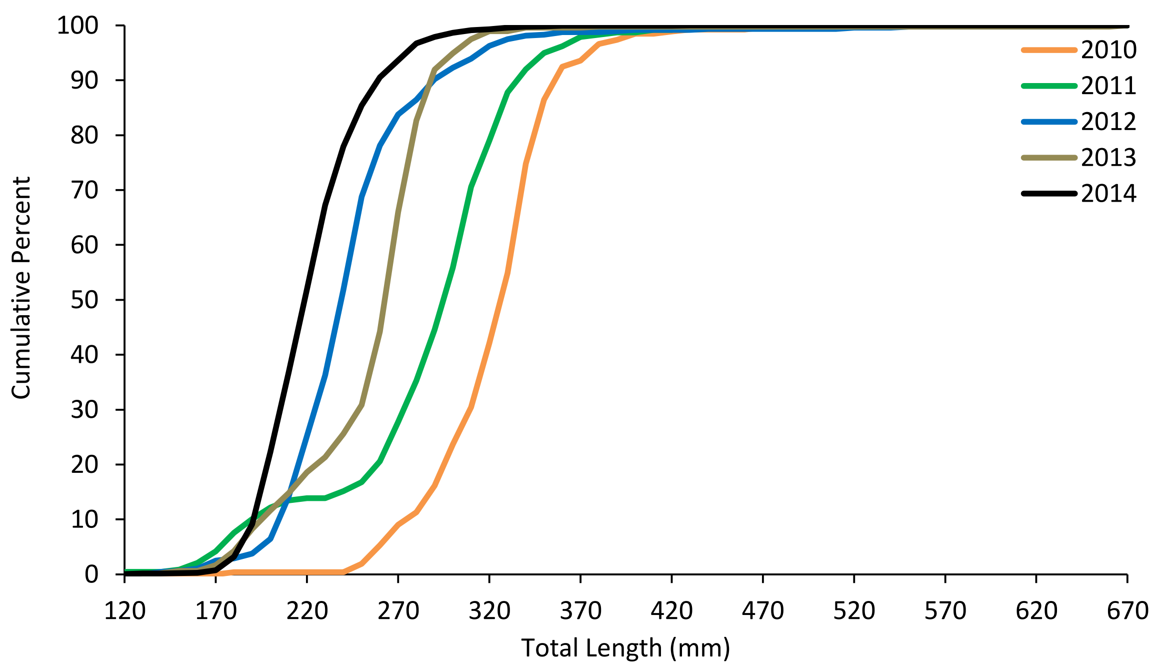


Figure 4. Annual cumulative length frequency distribution for lake trout caught during spring gillnetting in Quartz Lake, Montana.

Fall

Fall netting removed a total of 32 lake trout, including 6 immature and 26 mature lake trout. Results from fall 2014 were promising. Since the projects inception in 2009, the number of mature lake trout captured during fall netting has decreased by approximately 81% hitting an all-time low in 2014 (Figure 5). Additionally, the adult length distribution has decreased from 2009. Cumulative length frequencies from 2009-2014 for lake trout captured during fall netting has shifted to smaller, younger individuals, suggesting netting efforts have effectively removed the larger older individuals from the population (Figure 6). Truncation of length and age distributions in fish populations is a common response to intense size selective harvest (Coleman et al. 2000).

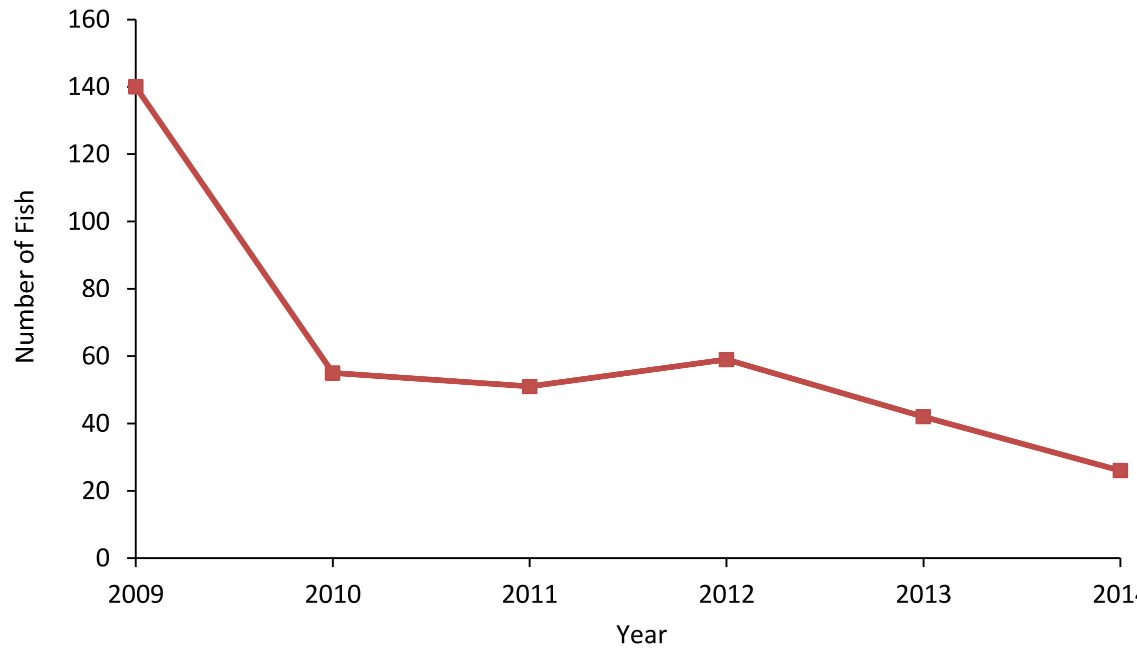


Figure 5. Number of mature lake trout caught during fall netting (2009-2014) in Quartz Lake, Montana.

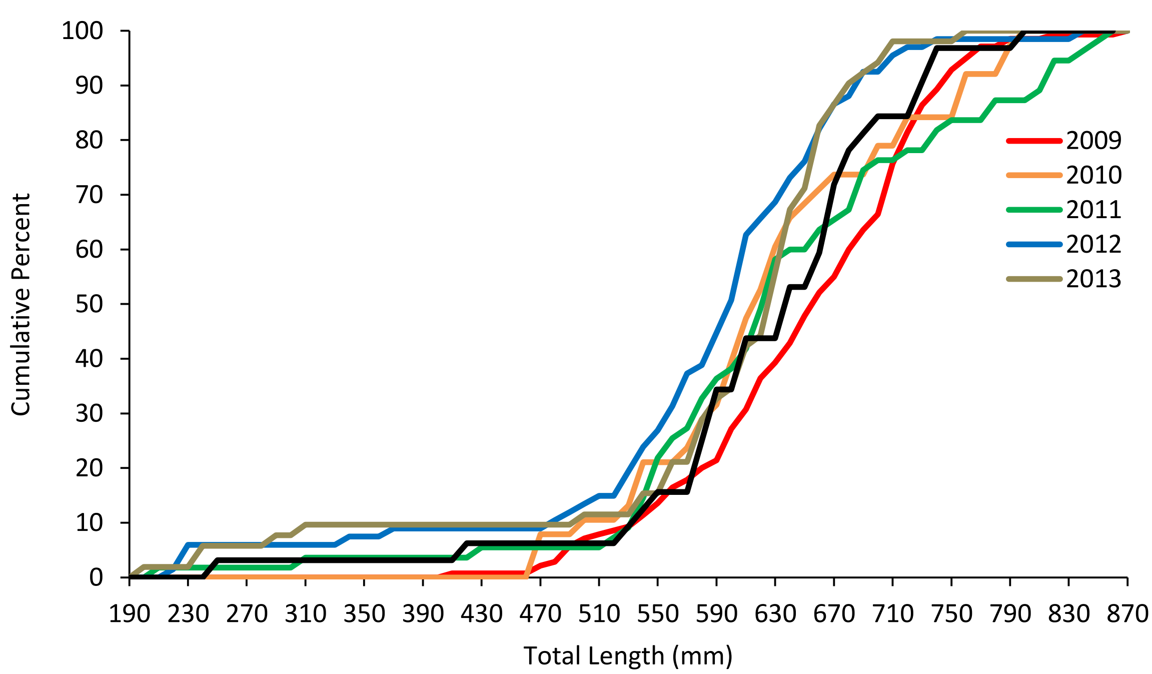


Figure 6. Annual cumulative length frequency distribution for lake trout caught during fall gillnetting in Quartz Lake, Montana.

Bycatch

Bull trout bycatch remains a concern in this project, and is therefore closely monitored. Redd counts have continued and are monitored closely to discern whether gillnetting activities have negatively impacted the adult bull trout population. Redd count analysis from 2003-2013 showed no significant trend, indicating the adult bull trout population in Quartz Lake remains stable (Fredenberg 2014). Additionally, the redd count from 2014 resulted in a historic high count (*N* = 66), further corroborating the results of the redd analysis (Figure 7).

To minimize bull trout bycatch, soak times for spring and fall netting have been kept relatively short to give inadvertently captured bull trout the best chances of survival. An adaptive management framework has been used in an attempt to decrease the number of bull trout incidentally caught, and to decrease bull trout mortality. Although gill-net mesh sizes and efforts have been generally standardized to allow year to year comparisons, we have made slight modifications over the years to try to reduce bull trout bycatch. An example of this includes the removal of the 45 mm and 51 mm bar measure gill-nets in 2011-2014. Analysis from 2009 and 2010 revealed that these mesh sizes accounted for the greatest number of adult bull trout mortalities, while 57 mm, 64 mm, and 70 mm bar mesh gill-nets accounted for the vast majority of the adult lake trout catch. Furthermore, in 2015 all of our juvenile gill-net meshes will be changed from the standard 0.20 mm twine diameter to 0.15 mm twine diameter. This modification should increase lake trout catch rates and may allow any incidentally caught mature bull trout to tear the mesh and free themselves prior to incidental mortality.

A total of 58 bull trout were captured via gillnetting in 2014, 27 (47%) of which were captured during spring and 31 (53%) during fall. Of the 58 bull trout captured, 34 (59%) were classified as adults (≥ 400mm) and 24 (41%) were juveniles or subadults. Additionally 36 adult sized bull trout were captured during spring angling. Eleven bull trout were incidentally killed in gillnetting efforts throughout 2014 (19% of total bull trout captured), including 5 adults (14% of captured adults) and 6 juveniles (26% of captured juveniles). The lower mortality rate incurred by adult bull trout when compared to juvenile bull trout is likely explained by the short gill-net set periods, the shallower set depths, and the colder mean water temperatures during fall netting when adults are typically captured. Bull trout bycatch and bycatch mortality in 2014 was near the lowest observed since the projects inception (Table 2).

Bycatch of non-target species was similar to the previous five years with the exception of an increase in mountain whitefish bycatch (Table 3). Non-target bycatch in 2014 included: longnose suckers (*N* = 176, length range, 145-615 mm), largescale suckers (*N* = 152, length range 140-650 mm), mountain whitefish (*N* = 1,258, length range 110-410 mm), and westslope cutthroat trout (*N* = 40, length range 240-370 mm). Bycatch mortality for these species were as follows: three longnose suckers (2% of total), two largescale suckers (1% of total), >143 mountain whitefish (> 12% of total), and eight westslope cutthroat trout (20% of total). The high bycatch and bycatch mortality of mountain whitefish in 2014 when compared to previous years can most likely be explained by the incorporation of 19 mm bar measure gill-nets used during spring gillnetting.

Table 2. Annual bycatch and bycatch mortality for juvenile (mature) bull trout in Quartz Lake, Montana for years 2009-2014.

Year	Bycatch	Mortality
2009	38(108)	14(13)
2010	28(95)	13(30)
2011	10(46)	3(8)
2012	20(45)	3(9)
2013	50(38)	22(7)
2014	24(34)	6(5)

Table 3. Annual bycatch of bull trout (BLT), westslope cutthroat trout (WCT), mountain whitefish (MWF), largescale sucker (LSS), and longnose sucker (LNS) during spring (fall) netting events from 2009-2014 in Quartz Lake, Montana.

Year	BLT	WCT	MWF	LSS	LNS
2009	--(146)	--(28)	--(498)	--(2)	--(351)
2010	53(70)	11(33)	185(55)	24(52)	216(249)
2011	29(27)	0(7)	100(17)	26(24)	285(27)
2012	24(41)	0(35)	124(51)	18(30)	143(74)
2013	52(38)	8(29)	229(25)	88(48)	237(18)
2014	27(31)	3(37)	1,237(21)	112(40)	135(41)

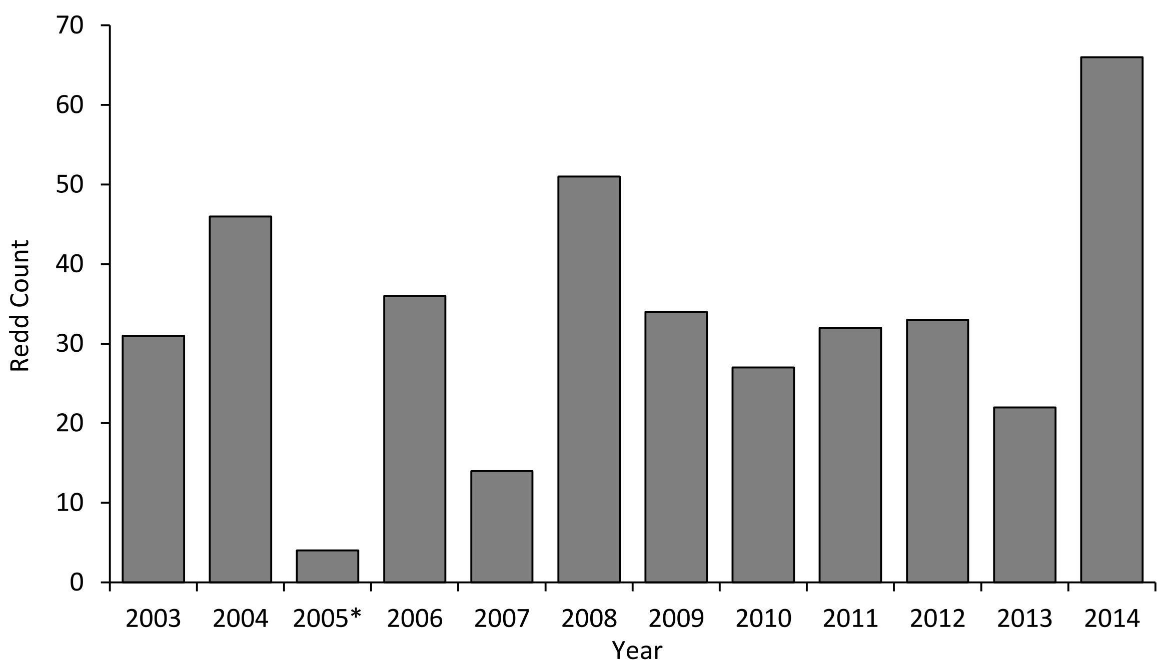


Figure 7. Bull trout redd counts from 2003 through 2014 in Quartz Creek upstream of Quartz Lake, Glacier National Park, Montana. *Redds were obscured by high flows in 2005 (Meeuwig and Guy 2007).

ACKNOWLEDGEMENTS

We thank Andrew Lamonte of the USGS, Larry Lockard, Wade Fredenberg, and Dan Brewer of the USFWS and Nate Muhn and Jon McCubbins of the National Park Service, for assistance in the field. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U.S. Government.

LITERATURE CITED

Behnke, R. J. 2002. Trout and salmon of North America. The Free Press, New York.

Crossman, E. 1995. Introduction of the lake trout in areas outside its native distribution: a review. Journal of Great Lakes Research 21(Supplement 1):17-29.

Donald, D.B. and D.J. Alger. 1993. Geographic distribution, species displacement, and niche overlap for lake trout and bull trout in mountain lakes. Canadian Journal of Zoology 71:238-247.

Downs, C.C., C. Stafford, H. Langner, and C.C. Muhlfeld. 2011. Glacier National Park fisheries inventory and monitoring bi-annual report, 2009-2010. National Park Service, Glacier National Park, West Glacier, Montana.

Fredenberg, C. R. 2014. Efficacy of suppressing non-native lake trout in an isolated backcountry lake in Glacier National Park. Master's Thesis. Montana State University, Bozeman.

Fredenberg, W. A. 2002. Further evidence that lake trout displace bull trout in mountain lakes. Intermountain Journal of Sciences 8:1-11.

Fredenberg, W., M. H. Meuwing, and C. S. Guy. 2007. Action plan to conserve bull trout in Glacier National Park. Final Action Plan Report. U.S. Fish and Wildlife Service, Creston, Montana and the U.S. Geological Survey, Montana Cooperative Fishery Research Unit, Montana State University, Bozeman.

Martinez, P. J., P. E. Bigelow, M. A. Deleray, W. A. Fredenberg, B. S. Hansen, N. J. Horner, S. K. Lehr, R. W. Schneidervin, S. A. Tolentino, and A. E. Viola. 2009. Western lake trout woes. Fisheries 34:424-442.

Meeuwig, M. H., C. S. Guy. 2007. Evaluation and action plan for protection of 15 threatened adfluvial populations of bull trout in Glacier National Park, Montana. Final scientific report to US Fish and Wildlife Service, Kalispell, Montana.

Muhlfeld, C. C. and C. R. Fredenberg. 2010. Preservation of native bull trout in Glacier National Park: experimental suppression of non-native lake trout in Quartz Lake. U.S. Geological Survey, Northern Rocky Mountain Science Center, Glacier Field Station, Glacier National Park.

Stafford, C. P., J. A. Stanford, F. R. Hauer, and E. B. Brothers. 2002. Changes in lake trout growth associated with *Mysis relicta* establishment: a retrospective analysis using otoliths. Transactions of the American Fisheries Society 131: 994-1003.

Vitousek, P. M., H. A. Mooney, J. Lubchenco , and J. M. Melillo. 1997. Human domination of Earth's ecosystems. Science 277:494–499.

Grace Lake Translocation

ABSTRACT

Glacier National Park (GNP) supports approximately one-third of the remaining natural lake habitat core areas supporting ESA listed bull trout *Salvelinus confluentus*. However, many bull trout populations have declined over recent decades and some are at high risk of extirpation in western GNP due to the establishment of invasive lake trout. Nine of seventeen lake-dwelling populations of bull trout located on the west side of GNP have been compromised by lake trout, and lake trout have been documented replacing bull trout as the dominant predator in these waters. Some populations appear to be persisting at dangerously low numbers (e.g. Logging, and Harrison lakes), and are at high risk of extirpation. Prior to the lake trout invasion, Logging Lake was one of the strongest bull trout populations on the west side of the Continental Divide in GNP. In 2014, NPS and USGS staff began translocating juvenile bull trout from Logging Creek to the upstream waters of the Grace Lake system where lake trout are unable to colonize due to a barrier falls between the lakes. Between 19 August and 8 October, Logging Creek upstream of Logging Lake was electrofished for 16 days. A total of 114 juvenile bull trout were captured during electrofishing, and 111 of these were translocated from Logging Creek upstream into Grace Lake. In addition, a two-way fish trap was used from September-October in an effort to capture adult bull trout for egg take and rearing at the Creston fish hatchery. One juvenile and one adult male were captured, neither were translocated. Zero adult females were observed or captured. The lack of spawning adult bull trout further exemplifies the urgency and importance of this translocation project to conserve the Logging Lake imperiled bull trout population.

INTRODUCTION

Non-native species are one of the most serious threats to global biodiversity (Vitousek et al. 1997), and are a leading cause of freshwater fish extinctions in western North America (Wilcove et al. 1998; Clavero and Garcia-Berthou 2005). Forty species of freshwater fish were known to be extirpated in North America between 1889 and 1989, and introductions of non-native species contributed to 68% of those extinctions (Miller and Williams 1989). Over the past century, non-native fish species have been widely introduced throughout the United States for aquaculture and recreational fisheries, and in many cases have eliminated native fish species, homogenized freshwater fish faunas, and reduced regional biodiversity (Miller and Williams 1989; Rahel 2000). Non-native fishes have impacted many species and populations of native fishes through competition, predation, and hybridization (Miller and Williams 1989; Rahel 2000).

Threatened bull trout *Salvelinus confluentus* populations within the Columbia River Basin have been shown to be particularly vulnerable to the negative effects associated with the introduction of non-native species. Of the approximate 100 lakes in the contiguous United States with native adfluvial (lake-dwelling) bull trout populations, about half are in undammed ecosystems. Glacier National Park supports approximately one-third of the remaining natural lake habitat core areas for bull trout in the United States (Fredenberg et al. 2007), and represents one of the last remaining strongholds for bull trout across their native range. Adverse interactions with non-native lake trout on the west side of GNP currently represent the single greatest threat to bull trout populations in these areas of the park. Recent studies have documented the replacement of bull trout by lake trout as the dominant predator in the majority of the large lakes on the west side of GNP in just the past 30 years (Downs et al. 2011).

Logging Lake is the fourth largest lake west of the continental divide in GNP and historically supported one of the most productive bull trout populations in the park (Fredenberg et al. 2007). However, following the documentation of lake trout in 1984, the bull trout population declined precipitously, and within less than 30 years, was replaced by lake trout as the dominant piscivore in the Logging Lake system. Currently, the bull trout population in Logging Lake persists at dangerously low numbers. Without immediate action, this population is at risk of imminent extirpation.

To mitigate the negative effects caused by non-native lake trout within the Logging Lake drainage, National Park Service(NPS) and U.S. Geological Survey (USGS) staff began a multiple step bull trout conservation project within the Logging Lake drainage. Step one was to translocate juvenile bull trout from Logging Creek to the isolated upstream waters of Grace Lake to conserve the Logging Lake bull trout population. Grace Lake is isolated from future lake trout invasion by a 7 m vertical falls located approximately 1.1 km upstream of Logging Lake. Grace Lake was identified in 2013 as possessing all of the physical properties needed to support a translocated population of bull trout (Galloway 2013). Following the successful translocation and conservation of the Logging Lake bull trout population, the project will proceed to step two in 2015; the suppression of lake trout within Logging Lake using gill nets.

Establishing a translocated bull trout population in Grace Lake will be key to preserving the genetic diversity associated with the Logging Lake bull trout population. The establishment of a bull trout population in Grace Lake should be valuable to serve as a donor population following lake trout suppression in Logging Lake in 2015 and beyond. This report represents a summary of data collected during the 2014 translocation field season.

STUDY AREA

Logging Lake is a glacially formed lake located in the headwaters of the Columbia River Basin, Montana. Logging Lake is the fourth largest lake west of the Continental Divide in GNP (Fredenberg et al. 2007), with a surface area of 451 ha and a maximum depth of 60 m (Figure 1). The lake is positioned in a narrow glaciated valley that is supplied by perennial flow from snow and glacial runoff from the Lewis Range. A large 7 m vertical falls; approximately 1.1 km upstream of Logging Lake isolates upstream Grace Lake from non-native fish immigration. Grace Lake has a surface area of approximately 33 ha and a maximum depth of 15 m. Historically, Grace was a fishless lake; however, it was stocked with 101,000 Yellowstone cutthroat trout (YCT) *Oncorhynchus clarkia bouvieri* in 1925 and continues to support a strong population.

Figure 1. Location of Grace and Logging lakes in the North Fork of the Flathead River Drainage, Glacier National Park.

The native salmonid assemblage in Logging Lake consists of bull trout, westslope cutthroat trout *Oncorhynchus clarkii lewisi* and mountain whitefish *Prosopium williamsoni*. The lake trout is the only non-native salmonid in the drainage and was first detected in Logging Lake in 1984 (Fredenberg et al. 2007). No native salmonids were present in Grace Lake prior to the introduction of bull trout.

METHODS

<u>Electrofishing</u>

In 2014, we began translocating bull trout from Logging Creek into the upstream waters of Grace Lake. Translocation involved electroshocking Logging Creek inlet to approximately 1.1 km upstream to the vertical falls. 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 four person crew sampled moving upstream, carefully working back and forth across the channel to effectively sample the entire creek.

Upon capture, juvenile bull trout were anesthetized, measured (total length (TL); mm) , weighed (g), fin clipped for genetic identification, and if the fish was ≥ 100 mm passive integrated transponder (PIT) tags were inserted into the abdominal cavity (Columbia Basin Fish and Wildlife Authority 1999). Fish were allowed to recover their equilibrium in a holding pen. Following a recovery period, black garbage bags were filled with water, instant ice packs were placed into the bags to ensure water temperatures remained within a suitable range. The remaining space in the bag was supersaturated with pure oxygen and the bags were hiked upstream, where fish were released within the creek draining into Grace Lake.

A stationary two looped PIT-tag antenna was fixed to the bottom and spanned the Grace Lake outlet. When the PIT-tagged fish swam over the antenna, the date, time and direction of passage were recorded. The antenna was powered by two deep cycle batteries with solar chargers.

<u>Trapping</u>

A two-way fish trap was used from September-October in an effort to capture adult bull trout for gamete take. The trap was checked daily, five days a week. On the weekends when crews were not present, the trap was left open to allow bull trout passage to the spawning area. Gamete take was part of a coordinated effort by NPS, USGS, and US Fish Wildlife Service (USFWS) staffs to take eggs and sperm from mature bull trout for propagation of young within Creston Fish Hatchery to preserve the genetic integrity of the bull trout population within the Logging system. The progeny were to be planted back into Logging Lake following planned lake trout removal in coming years, as well as to augment the effort to establish a bull trout population in Grace Lake.

RESULTS AND DISCUSSION

The translocation effort was deemed highly successful following the translocation of 111 juvenile bull trout. Although one concern was that the removal of juveniles from the Logging Lake system would likely threaten the downstream donor population within Logging Lake, it was deemed necessary to rescue the imperiled population of bull trout prior to their imminent demise. Past gill netting (Fredenberg 2002; Meeuwig et al. 2008; Downs et al. 2011) and redd count data (Downs et al. 2011) indicated that bull trout in Logging Lake exist at low densities and were at high risk of extirpation if action was not taken. Although it is uncertain at this time, the translocation of bull trout from Logging Lake to the waters of Grace Lake may have conserved the Logging Lake bull trout population from imminent extirpation.

Electrofishing

We began electrofishing Logging Creek to capture and translocate juvenile bull trout on 19 August and ended on 8 October. We sampled 16 days during this period. We captured a total of 114 juvenile bull trout using electrofishing and 111 of these were translocated from Logging Creek upstream into Grace Lake. We had a total of three juvenile bull trout mortalities during the electrofishing capture efforts. Lengths of translocated bull trout varied from 52-200 mm (mean 67 mm; 95% CI 63-71 mm) (Figure 2). Catch per unit effort varied from 4-25 bull trout per hour (Table 1).

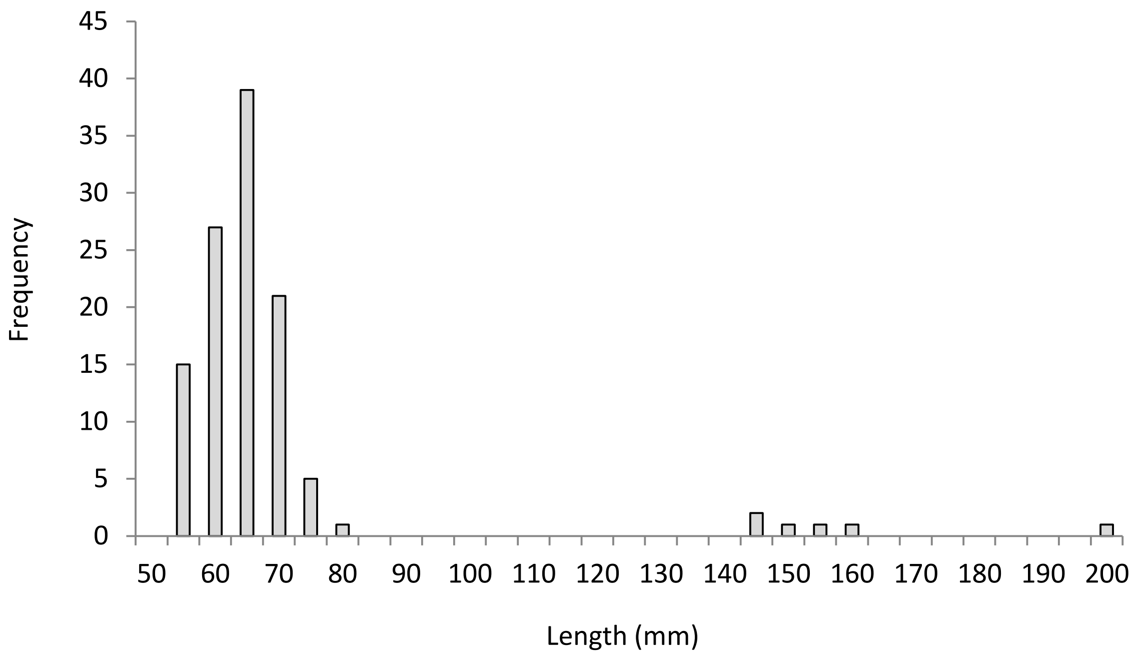


Figure 2. Length frequency of bull trout translocated from Logging Creek to Grace Lake in 2014, GNP, Montana.

Six bull trout were PIT-tagged and released into the Grace Lake inlet stream. The PIT-tagged fish varied in length from 143-200 mm in length (Table 2). The PIT-tag antenna recorded one PIT-tagged bull trout moving to the outlet of Grace Lake. On 8 September at approximately 23:07, the 200 mm bull trout (ID 22597) was detected at the outlet of Grace Lake. Detection duration lasted 21 hours until 9 September at 21:00 hours, and the bull trout did not leave the system.

Table 1. Shocking date, number of bull trout (BLT) captured, shock time in seconds, and catch per unit effort (CPUE) of bull trout per hour, in Logging Creek, GNP 2014.

Date	Total BLT Caught	Shock Time (Seconds)	CPUE BLT (Fish/Hr)
8/19/2014	6	--	--
8/20/2014	3	1130	9.6
8/20/2014	7	1330	19.0
8/21/2014	15	5907	9.1
8/26/2014	4	2274	6.3
8/27/2014	4	1894	7.6
8/28/2014	11	--	--
9/2/2014	4	--	--
9/3/2014	22	3918	20.2
9/9/2014	10	1428	25.2
9/10/2014	2	1834	3.9
9/11/2014	4	6460	2.2
9/16/2014	3	891	12.1
9/17/2014	10	1715	21.0
9/23/2014	2	340	21.2
9/24/2014	2	1805	4.0
10/8/2014	4	1409	10.2

Table 2. Date, PIT-tag identification number, and length of bull trout captured via electrofishing in Logging Creek that were translocated to the inlet of Grace Lake.

Date	Fish ID	Length (mm)
8/20/2014	22597	200
8/20/2014	519893	143
8/21/2014	519804	144
9/03/2014	519885	151
9/09/2014	519932	146
9/16/2014	520056	156

Trapping

The two-way fish trap was set beginning 24 September and removed 24 October. The trap was checked daily during weekdays, and left open during the weekend to allow bull trout to migrate to the spawning areas. Although the trap was left open on the weekends, no mature bull trout were visually identified within the spawning reach throughout the trap-netting period. Additionally, no bull trout spawning redds were identified within the spawning reach. One mature male bull trout (570 mm) was captured on 7 October and zero mature female bull trout were caught throughout the trap-netting period. Additionally, one juvenile bull trout was caught moving downstream in the trap, however it escaped prior to translocation to Grace Lake. The lack of spawning adults further exemplifies a bull trout population on the brink of functional extinction.

In 2015, NPS and USGS staffs will continue translocation attempts of juvenile bull trout. Additionally, a lake trout suppression program is anticipated to begin in 2015 in Logging Lake. The suppression program will be focused on removing large proportions of the lake trout population from Logging Lake to inhibit deleterious effects of lake trout on the native bull trout population. Future work in Grace Lake will be focused on determining the survival of translocated bull trout via trap netting. Finally, gamete take of spawning bull trout in nearby Quartz Lake is planned for fall of 2015. Eggs and sperm will be taken to the Creston Fish Hatchery, where FWS staff will oversee the rearing of the bull trout prior to their reintroduction into the Logging Lake system. The population of bull trout in Quartz Lake is considered stable (Fredenberg 2014), is genetically similar to that of Logging Lake (Meeuwig et al. 2010), and represents the "nearest neighbor" population to Logging Lake. It is therefore likely to have undergone similar environmental adaptation pressures and should be similarly suited to Grace Lake.

ACKNOWLEDGEMENTS

We thank Andrew Lamonte and Maddy Cochrane of the USGS and Nate Muhn and Jon McCubbins of the National Park Service, for assistance in the field.

LITERATURE CITED

Columbia Basin Fish and Wildlife Authority. 1999. PIT tag marking procedures manual. PIT tag steering committee, Portland, Oregon.

Clavero, M., and E. Garcia-Berthou. 2005. Invasive species are a leading cause of animal extinctions. Trends in Ecology & Evolution 20:110.

Downs, C. C., C. Stafford, H. Langner, and C. C. Muhlfeld. 2011. Glacier National Park fisheries inventory and monitoring bi-annual report, 2009-2010. National Park Service, Glacier National Park, West Glacier, Montana.

Fredenberg, C. R. 2014. Efficacy of suppressing non-native lake trout in an isolated backcountry lake in Glacier National Park. Master's Thesis. Montana State University, Bozeman.

Fredenberg, W. A., M. Meeuwig, and C. S. Guy. 2007. Action plan to conserve bull trout in Glacier National Park. USFWS Creston Fish and Wildlife Center, Kalispell, MT.

Galloway, B.T. 2013. Feasibility assessment for translocation of imperiled bull trout populations in Glacier National Park, Montana. Master's Thesis. Montana State University, Bozeman.

Meeuwig, M. H., C. S. Guy, S. T. Kalinowski, and W. A. Fredenberg. 2010. Landscape influences on genetic differentiation among bull trout populations in a stream-lake network. Molecular Ecology 19:3620-3633.

Miller, R. R., and J. D. Williams. 1989. Extinctions of North American fishes during the past century. American Fisheries Society 14:22-38.

Rahel, F. J. 2000. Homogenization of fish faunas across the United States. Science 288:854-856.

Vitousek, P. M., C. M. D'Antonio, L. L. Loope, and R. Westerbrooks. 1997. Biological invasions as global environmental change. American Scientist 84:468–478.

Wilcove, D. S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. American Institute of Biological Sciences 48:607-615.

Native Fish Population Monitoring

ABSTRACT

In 2009, Glacier National Park (GNP) began development of a monitoring program for native salmonids inhabiting park streams. The intent of the program is to establish baseline abundance levels in a variety of park waters to serve as useful benchmarks for monitoring changes in populations over time. We continued this work through 2014, conducting sampling in seven waterbodies. We conducted electrofishing depletion removals on Fern (Middle Fork Flathead drainage), McGee (North Fork Flathead drainage), Un-named tributary to Camas Creek (North Fork Flathead drainage), and Lee (St. Mary drainage) creeks; snorkel surveys on Muir (Middle Fork Flathead drainage) and Akokala (North Fork Flathead drainage) creeks, and live trap netting in Cracker Lake (St. Mary drainage). We noted presence or absence along with relative abundance of amphibians in our surveys. In general, the largest threat currently facing migratory native fish species on the east side of the park is the unscreened St. Mary River irrigation diversion near Babb, part of the Milk River Irrigation Project. Other threats include non-native species (walleye) found in the St. Mary River downstream of the U.S./Canada border. The most significant threat to native fish on the west side of the park comes from invasive non-native species, such as rainbow, brook, and lake trout. Some culverts along the Camas Road on the west side of the park may be important isolating mechanisms, protecting genetically pure westslope cutthroat from downstream populations of hybrid westslope cutthroat-rainbow trout. However, management and conservation of native fish in the North and Middle forks Flathead River remains complicated due to the presence of migratory and resident populations of native salmonids, and expanding distribution of non-native fish species.

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 (Table 1; Table 2).

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.

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 quantitative, repeatable historic fisheries data exists, and most of it was collected by Montana Fish, Wildlife, and Parks in the Flathead River drainages of the park (Read et al. 1982, Weaver et al. 1983). There were also a couple of efforts in the late 1960’s and 1970’s (NPS files, unpublished data) to conduct standardized gill-net surveys in some of the large lowland lakes in the North Fork Flathead Drainage of the park by the USFWS, and these data have served as useful baseline to evaluate fish species composition changes over time (e.g. lake trout versus bull trout). Later work by Mogen and Kaeding (2004) on bull trout populations in the St. Mary drainage represented the start of limited, but more intensive monitoring efforts for bull and westslope cutthroat trout in that drainage.

Establishing baseline data sets will be key to understanding how populations are responding to a changing climate and the associated changes in water temperatures, storm frequency, runoff timing, increase fire frequency, etc. Developing population estimate sections spread across the park should be valuable to monitor changes over time in fish community composition and abundance in response to expanding non-native fish species populations and climate change. This report represents a summary of data collected during 2014 field season.

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

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Arctic grayling *Thymallus arcticus*	--	--	I
Brook trout (bkt) *Salvelinus fontinalis*	I	I	I
Bull trout (blt) *S. confluentus*	N	--	N
Kokanee *Oncorhynchus nerka*	I	--	I
Lake trout *S. namaycush*	I	--	N
Lake whitefish *Coregonus clupeaformis*	I	--	N
Mountain whitefish (mwf) *Prosopium williamsoni*	N	N	N
Pygmy whitefish *P. coulteri*	N	--	N
Rainbow trout (rbt) *O. mykiss*	I	I	I
Westslope cutthroat trout (wct) *O. clarkii lewisi*	N	N	N
Yellowstone cutthroat trout *O. c. bouvieri*	I	I	I
Fathead minnow *Pimephales promelas*	--	--	--
Northern pikeminnow *Ptychocheilus oregonensis*	N	--	--
Peamouth *Mylocheilus caurinus*	N	--	--
Redside shiner *Richardsonius balteatus*	N	--	--

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

Species	Columbia Drainage	Missouri Drainage	Hudson Bay Drainage
Longnose sucker *Catostomus catostomus*	N	N	N
Largescale sucker *C. macrocheilus*	N	--	--
White sucker *C. commersoni*	--	N	N
Deepwater sculpin *Myoxocephalus thomsoni*	--	--	N
Mottled sculpin *Cottus bairdi*	--	N	N
Slimy sculpin *C. cognatus*	N	--	--
Shorthead sculpin *C. confuses*	N	--	--
Spoonhead sculpin *C. ricei*	--	--	N
Burbot *Lota lota*	--	--	N
Northern pike *Esox Lucius*	--	--	N
Trout-perch *Percopsis omiscomaycus*	--	--	N

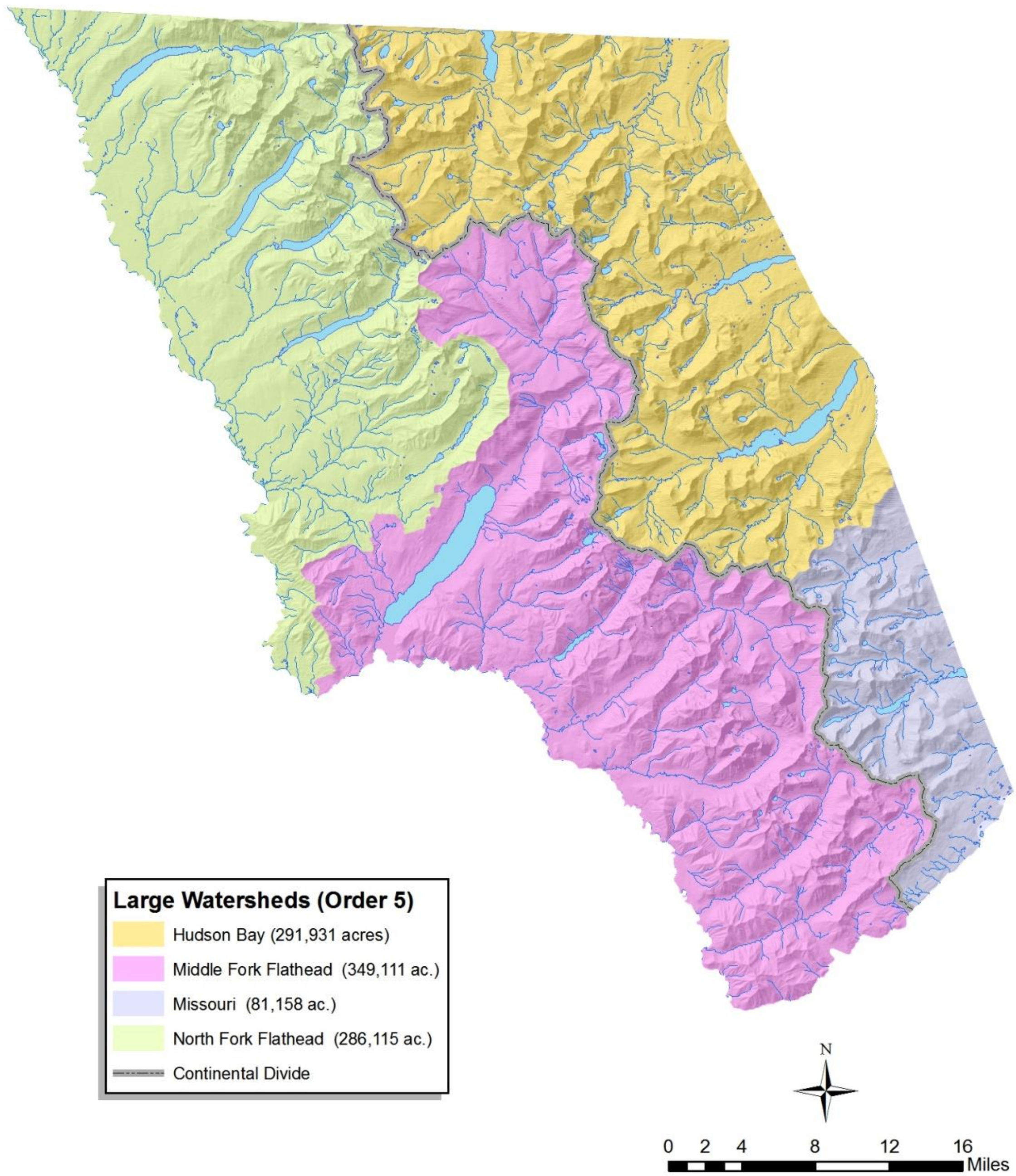


Figure 1. Major watersheds of Glacier National Park, Montana.

METHODS

Electrofishing

In 2014, we conducted removal (depletion) estimates (Zippin 1958) on four park streams (Figure 2, Table 3). Removal estimates involved identifying a representative reach of stream approximately 100 m long, and placing a 1.22 m X 9.15 m X 6.35 mm block net (minnow seine) at the downstream end of the section to minimize the likelihood of fish moving into or out of the section. The upstream end of the section was located at a high gradient riffle break or other natural drop in the stream channel bed. We used a Smith-Root model 15-B battery powered electrofisher, using pulsed DC current to capture the fish. Settings were adjusted to use the minimum amount of power required to capture fish while minimizing fish injury. Settings were generally set at 30 htz., 3 ms pulse width, and between 400 and 700 volts, depending on stream temperature and conductivity. A two to three person crew sampled moving downstream, carefully working back and forth across the channel to effectively sample the entire reach. Repeated downstream passes were made through the section until the catch on the most recent pass was reduced to 30% or less of the catch on the first pass for age-1 and older juvenile salmonids.

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

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

Netting

We used custom-made miniature fyke and hoop nets to sample Cracker Lake, located in the Many Glacier Valley (Figure 2). Fyke nets were constructed of 12mm (bar measure) brown nylon mesh. Each net measured 3 m in length and had leads 7 m long X 0.66 m tall, with a leadcore and polyfoam line used to hold the lead upright, while maintaining contact with the lake bottom. Their rectangular opening at the mouth of the fyke net measured 66 cm tall by 1 meter wide. The remainder of the trap consisted of a single throat (approximately 15 cm opening at the small end) and four hoops. Fyke nets were not baited. We used these same mini fyke nets to sample Logging Lake, located in the North Fork Flathead River drainage.

We also used hoop nets to sample Cracker Lake. These nets consisting of 12mm (bar measure) green nylon mesh with three supporting metal hoops, each measuring 65 cm in diameter. The overall length of the traps were 1.5 meters with a single throat with a cod-end diameter of approximately 10cm. Most hoop nets were baited with dry cat food (salmon flavor).

An inflatable kayak was used transport the nets to the trap locations and set the nets as necessary. Fyke nets were also set by staff wading into the deepest water they could without overtopping their chest waders and dropping the cod end of the trap to the bottom anchored by a weight. The lead end was then pulled to shore to set the trap. The shallow end of the lead was set where it would obstruct the entire depth of the water column (about 0.66m), which was typically within 1-2 m of shore. Fyke nets were set angled along the shoreline such that the entrance to the trap was generally set in <1-m deep water. These shallow sets were used because test netting failed to catch any fish in deeper water. Overnight sets were used and nets were typically set in late afternoon and retrieved early the following morning.

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 blt (Anderson and Neumann 1996). Fish were allowed to recover their equilibrium and were released back into the stream.

Snorkeling

In addition to electrofishing and trapping, we also used snorkeling to assess species composition and size structure in some streams (Figure 2, Table 3). Surveying for fish using standardized snorkeling techniques has long been understood to be a valid method to survey abundance, species composition, size structure, and habitat use (Thurow 1994). We defined three size classes of fish; Class I: <45mm TL, Class II: 46-149mm TL, Class III: >150mm. Surveyors snorkeled in an upstream direction. At least one observer walked alongside the snorkeler measuring stream width, recording time snorkeler was surveying, species found, and to watch for safety hazards. When creek size allowed, two snorkelers moved upstream side by side. Fish were not counted in the survey until the snorkeler had moved past them in order to avoid duplicate counting of fish. Snorkel estimates involved identifying a representative reach of stream approximately 100-200 m long. We used a high-gradient riffle break or other natural drop in the stream channel as section starting and ending points. When possible, snorkel surveys were performed within previously sampled electrofishing or historical (e.g. Read et al. 1982) snorkeling sections.

Due to the cold water temperatures generally found in the park, the timing of our sampling, and previous length-frequency data, we assumed bull trout (blt) ≥60 mm and westslope cutthroat trout (wct) ≥45 mm were age-1 and older for estimation of abundance and catch-per-unit effort (CPUE). This was confirmed by examining the length-at-capture data from our 2009 field sampling. These assumptions are further supported by other studies. In some cold systems similar to GNP, wct fry may not even emerge from the gravel until mid-August (Scarnecchia and Bergersen 1986, Downs 1995). Scarnecchia and Bergersen (1986) indicated few cutthroat trout from headwater systems in Colorado exceeded lengths of 30-35 mm before the entered their first winter. Therefore using a lower limit (i.e. 45 mm) as a cutoff for inclusion in estimates of age-1 and older wct is more appropriate for most park waters containing rearing wct. Fishes of these sizes (blt≥60mm and wct≥45mm) 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 Starvation, Ole, and Lee creeks.

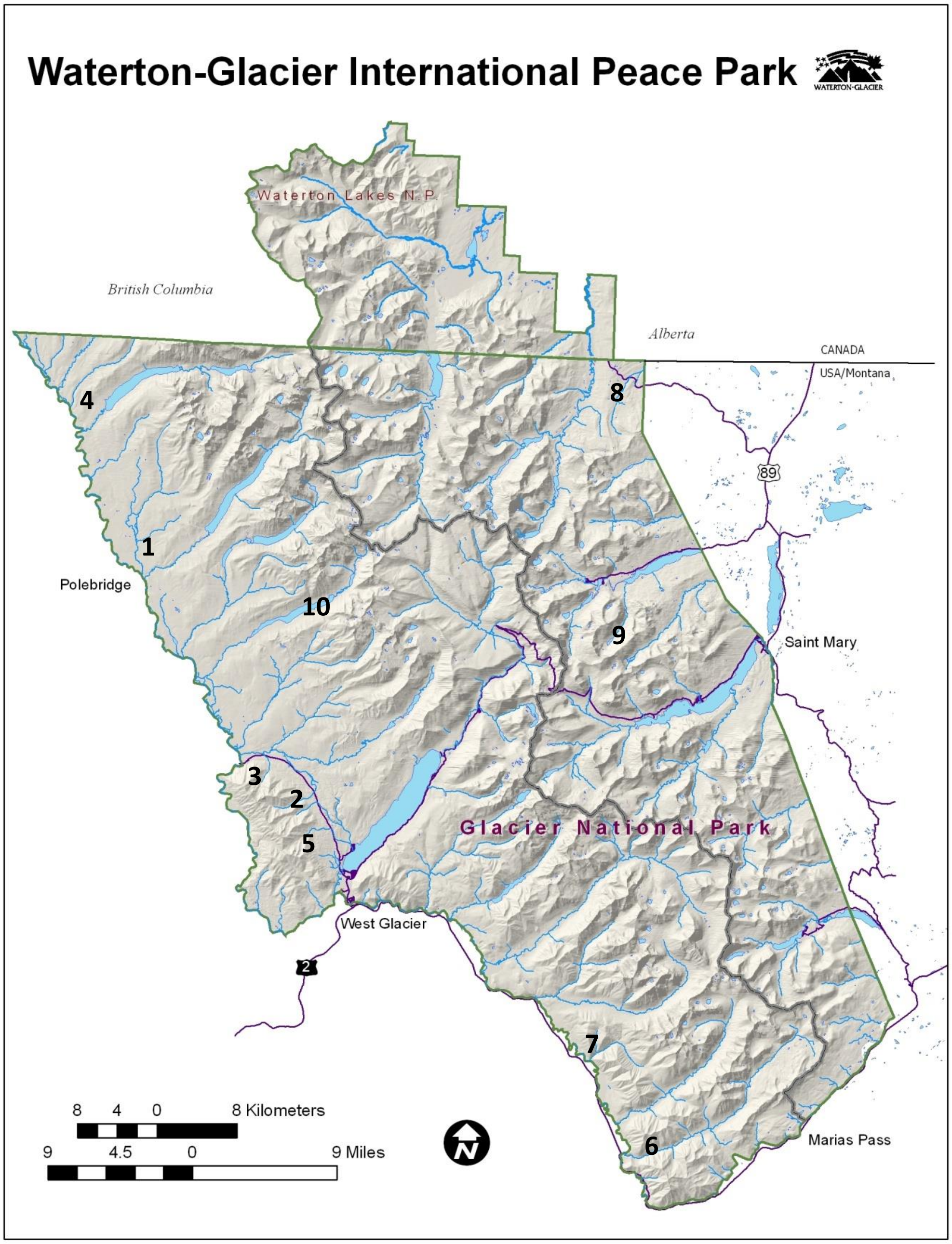


Figure 2. 2014 stream sampling sites for depletion population estimates, temperature monitoring, snorkeling, and trapping efforts in Glacier National Park, Montana. Site numbers from Table 3.

For both electrofishing and snorkeling surveys, GPS coordinates of the upstream and downstream end of the sections were recorded in UTM using the WGS 84 datum. Digital photographs were taken of the upstream and downstream ends of each section, and wetted widths were measured at approximately 20 m intervals to calculate the wetted stream area sampled. Channel gradient was estimated in percent with a clinometer by taking multiple measurements (four or more) of slope in each study reach and averaging them. Stream temperature was recorded at the time of sampling 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.

Table 3. Location information for native fish population monitoring efforts in 2014.

Major Drainage	Water name (Site No.)	Method	Location of downstream end of reach (WGS 84, UTM)	Dates sampled	Sample Reach Length (m)
North Fork Flathead	Akokala Cr (1)	Snorkel	11U 698703, 5409823	8/1/2013; 9/2/14	159
	McGee Cr (2)	Electrofish	11U 0718256N, 5386857W	7/10/09; 7/29/13; 8/11/14	134
	No Name Cr (3)	Electrofish	11U 0713098N, 5390261W	8/4/11; 7/17/12; 7/31/13; 7/28/14	90
	Starvation Cr (4)	Thermograph only	11U 0690925N, 5423640W	2014	
	Logging Lake (10)	Mini fyke nets	Multiple sites	2014	
Middle Fork Flathead	Fern Cr (5)	Electrofish	11U 0720216N, 5383473W	7/9/09; 7/15/10; 7/18/11; 7/18/12; 7/2/13; 7/17/14	90
	Ole Cr (6)	Thermograph only	12U 0307191N, 5350741W	2014	
	Muir Cr (7)	Snorkel	12U 0303790N, 5360495W	8/21/2013; 8/13/14	160
Saint Mary (Hudson Bay)	Lee Cr (8)	Electrofish	12U 0307481N, 5428586W	7/21/09; 8/10/11; 8/15/12; 8/16/13; 8/27/14	125
	Cracker Lake (9)	Mini fyke nets	Multiple sites	7/7-7/10/14	

RESULTS AND DISCUSSION

North Fork Flathead River Drainage

Akokala Creek

Akokala Creek is a third order tributary to the North Fork of the Flathead River. The lower reaches of Akokala Creek are known to support migratory westslope cutthroat trout spawning (Muhlfeld 2009b), while the upper reaches of the drainage support “disjunct” migratory populations of wct and bull trout using Akokala Lake as adult habitat (Meeuwig et al. 2007, Meeuwig 2008). Resident wct are also found in the upper stream portions of the watershed. Migratory wct from Flathead Lake or the mainstem Flathead River reproduce in the lower gradient stream reaches from the time of peak streamflow through the descending limb of the hydrograph (Muhlfeld 2009b). Hybridization between rainbow and wct was recently detected in the lower reaches of Akokala Creek, but we do not believe it has progressed into headwater areas. A recent radio-telemetry study documented hybridized westslope cutthroat trout entering Akokala Creek from the North Fork Flathead River in the spring (Muhlfeld et al. 2009). Hybridization has also been recently detected in the lower reaches of Akokala Creek (MFWP, unpublished data). Longbow Creek, a tributary to Akokala was sampled for westslope cutthroat genetic status in 2008, and did not show any evidence of hybridization (MFWP, unpublished data). Similarly, genetic testing in 2008 failed to detect hybridization at the NPS trail crossing in the upper portions of the drainage. We desired to sample Akokala Creek in the migratory spawning reach to develop an index of abundance that would reflect the health of the migratory native wct population. Migratory wct populations are the most vulnerable wct life-history because they depend on large connected lake-river systems, which have become increasingly rare. They are also more likely to be adversely impacted by non-native fish species due to the highly modified fish assemblages of major lake systems across their native range.

A 159m long reach of Akokala Creek was snorkeled on 9/2/2014 in approximately the same location as in 1980 (Read et al. 1982) and 2013 (Downs et al. 2013). Read et al. (1982) reported surveying lower Akokala Creek on 8/19/80. We observed 35 wct <45mm, 45 wct between 45-149 mm, and 19 wct $\geq$150 mm (Table 4). By excluding wct<45mm, we estimated a density of 5.24 age-1 and older wct/100m^2 present in the section in 2014. This is considerably higher than that reported by Read et al. (1982), but very similar to our 2013 results (Table 4). Read et al. (1982) used a similar size range as we used (<40mm) to identify age-0 wct and (<50mm) age-0 bull trout, so the results are reasonably comparable. It appears that wct abundance of age-1+ has increased from 1981 to present in Akokala Creek. We also observed one juvenile bull trout in the section in 2014.

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

Table 4. Size class, species, and fish count for snorkel surveys conducted in Akokala Creek.

Species	Length	1981	2013	2014
WCT	<45 mm		15	35
	45-149 mm		65	45
	≥150 mm		19	19
	Age 1+ /100m2	2.4	5.7	5.2
MWF	<45 mm		6	69
	45-149 mm		6	15
	≥150 mm		2	11
	Age 1+/100m2	present	0.9	2.1
BLT	<50 mm	0	0	0
	50-149 mm	0	0	0
	≥150 mm	0	0	1
	Age 1+/100m2	0.0	0.0	0.1

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. No amphibians were observed. Water temperature at the time of sampling was 13°C. Physical habitat characteristics were comparable for this reach to those reported by Read et al. (1982), who sampled the same general area on 8/19/80. The authors measured conductivity at 65µS (80µS in 2014) and bed material composition of 30% rubble, 30% gravel, 30% fines, and 10% boulder. Channel debris was noted as "low" in both studies**.** This stretch of Akokala Creek is ideal for snorkeling with its wide channel, numerous pools, and low density of woody debris.

McGee Creek

McGee Creek is a third order tributary to Camas Creek in the N. Fk. Flathead River drainage. It is historical wct habitat, but is threatened by hybridization with rbt. Muhlfeld et al. (2009a) documented hybridization between rbt and wct in Dutch Creek, a large tributary to Camas Creek. Genetic samples collected from lower McGee Creek in 2008 indicated hybridization was occurring (MFWP, unpublished data). The culvert under the Camas Road may preclude upstream passage to the headwater reaches of McGee Creek due to its length and drop out of the pipe, but this remains uncertain.

Within the 134m long stream reach, the estimated total abundance of age-1 and older wct was 19 (95% CI = 17-21). The density estimate for these same fish was 4.3 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 27.1 fish/hr and 3.4 fish/100m^2. Average length of age-1 and older wct was 110.5 mm (Table 5; Figure 3). Average Fulton Condition Factor (K) was also estimated at 1.1 for age-1 and older wct (SD = 0.13) (n = 18). Population densities have remained relatively constant in the study section over time (Table 6).

Table 5. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton condition (K) of age-1 and older for wct captured in McGee Creek, GNP, 2014. Length range represents all fish captured.

Species	Mean Length (SD) (n)	Length Range (TL;mm)	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT	110.5 (28.9)(19)	61-168	18.7(15.6)(18)	1.1

Table 6. Population estimates conducted over time in McGee Creek, Glacier National Park.

Year	Estimate (95% CI)	Density (fish/100m2)
2009	13 (N/A-no fish captured on second pass)	5.3
2013	14 (12-16)	3.4
2014	19 (17-21)	4.3

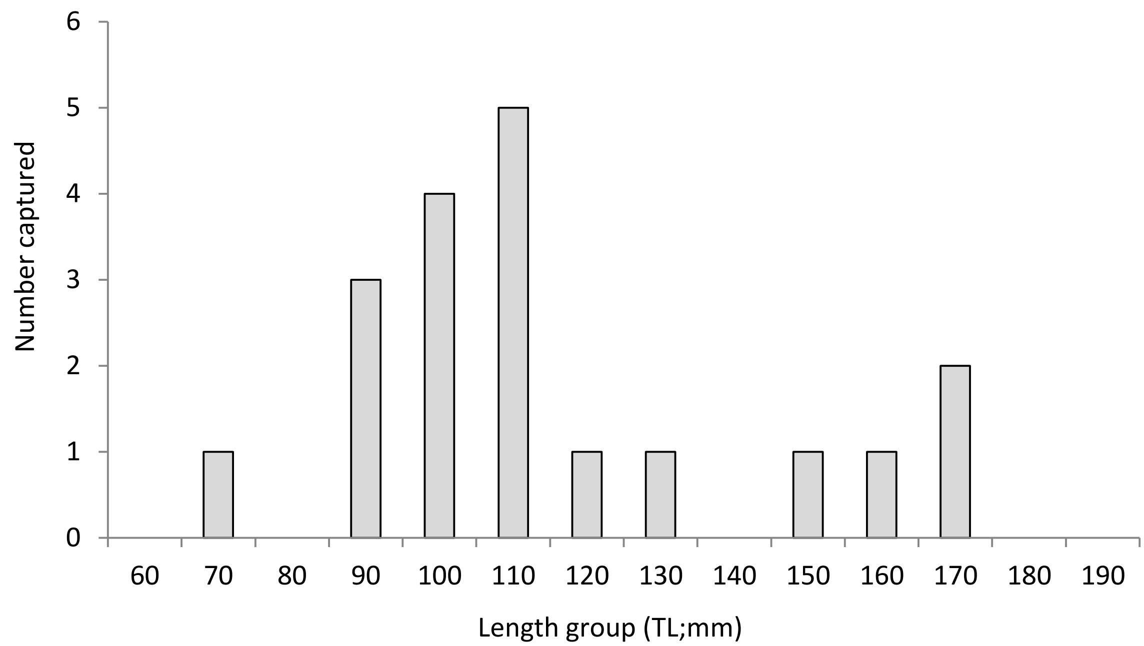


Figure 3. Length-frequency histogram for wct captured in McGee Creek, Glacier National Park, in 2014.

Cobble was estimated to be the dominant substrate type within the reach followed by gravel. Water temperature was measure to be 9.8 °C and conductivity was measured at 120.5 µS. We employed 670 volts at 30 hertz, with a 9% duty cycle and a 3 ms pulse width. The presence of tailed frogs was noted in 2014. Average wetted with measured 3.3 m over the 134m reach.

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

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

Average length of age-1 and older wct was 107.3 mm (Table 7, Figure 4). Average Fulton Condition Factor (K) was also estimated at 1.0 for age-1 and older wct (SD = 0.09) (n = 13) (Table 7). Within the 99 m long stream reach the estimated total abundance of age-1 and older wct was 13 (Table 8). The density estimate for these same fish was 4.7 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 18.5 fish/hr and 3.6 fish/100m^2. The density of wct in the study area remained fairly stable for the first three years of the study and appears to have declined substantially in 2014 (Table 9).

Table 7. 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, 2014. Length range represents all fish captured.

Species	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Fulton Condition Factor (K)
WCT	107.3(33.5)(13)	66-157	16.7(13.7)(13)	1.0

Table 8. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older wct captured in No-name Creek, GNP, 2014.

Species	Population Estimate (95%CI)	Density (fish/100m^2)	$CPUE_1$ (fish/100m2)	$CPUE_1$ (fish/hr)
WCT	13 (11-15)	4.7	3.6	18.5

Table 9. Population estimates conducted over time in No Name Creek, Glacier National Park.

Year	Estimate (95% CI)	Density (fish/100m2)
2011	39 (36-42)	12.5
2012	35 (31-39)	11.5
2013	21 (19-23)*	13.3*
2014	13 (11-15)	4.7

*shorter, 50m subsection used for this estimate

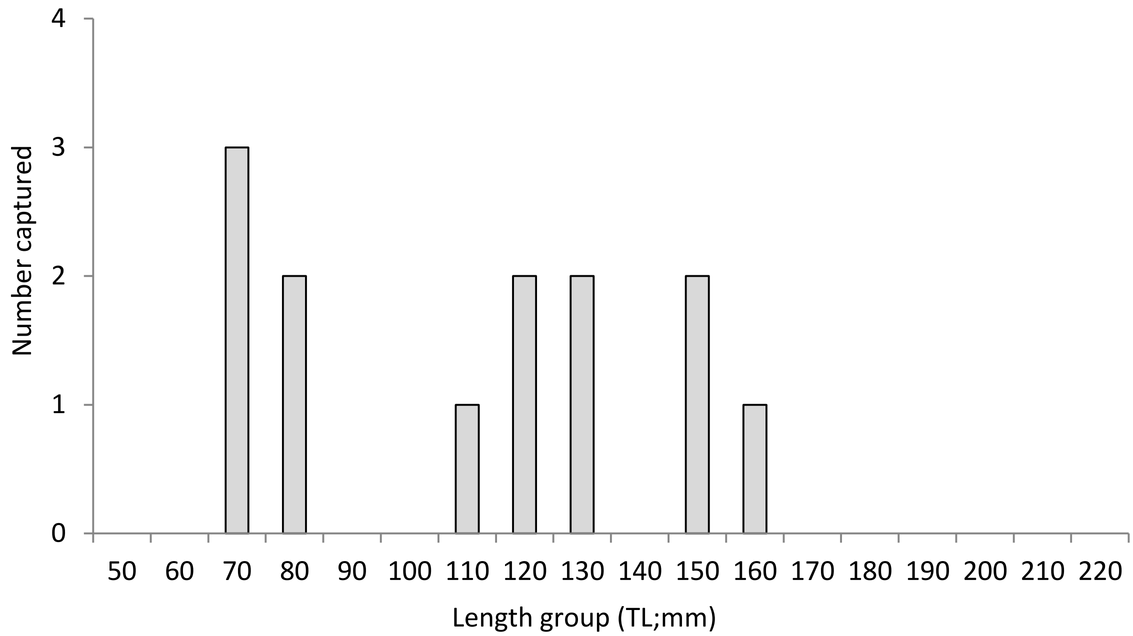


Figure 4. Length-frequency histogram for wct captured in No-name Creek, Glacier National Park, in 2014.

The dominant substrate was determined to be cobble and a subdominant gravel substrate. No Name Creek had water temperature and conductivity of 9.9°C and 131 µS, respectively. Our electrofishing settings were 700 volts at 30 Hertz on a 9% duty cycle and a 3 ms pulse width. The sampling reach length was 99m while average wetted width was 2.8m. Tailed frogs were noted as abundant. We did not snorkel this reach due to small stream size and habitat complexity.

Middle Fork Flathead River Drainage

Fern Creek

Fern Creek is a second order tributary to Fish Creek in the McDonald Creek drainage. The downstream end of the sampling site is located approximately 50 meters upstream of the Camas Road crossing. A perched culvert under the Camas Road may isolate wct upstream of the culvert from downstream fish, protecting the genetic integrity of the upstream fish (Downs et al. 2011). Westslope cutthroat trout hybridization has not been assessed downstream of the culvert in Fern Creek. The culvert is approximately 60' long, with a relatively flat slope. When measured by Downs et al. (2011), the drop out of the culvert to the water surface was approximately two feet. The area burned in 2003 as part of the Robert Fire, and much of the riparian area is dominated by shrub-type cover as well as standing and downed burned timber.

Based on the size structure of the population, the location of the sampling site in the drainage, and the likelihood that the culvert under the Camas Road is at least a partial upstream barrier, it is likely this population is resident. Based on samples collected from upper Fern Creek in 2008, Fern Creek upstream of the Camas Road contain genetically pure wct (C. Muhlfeld, USGS, personal communication). We collected additional genetic samples from Fern Creek wct for future analysis.

We captured a total of seven wct over two passes. The complexity of the habitat made shocking difficult. We only captured larger individuals. Length ranged from 133 to 169 mm (Figure 5). Average length of age-1 and older wct was 148.3 mm (Table 10). Average Fulton Condition Factor (K) was also estimated at 1.2 for age-1 and older fish.

The population estimate for age-1 and older wct in 88m long section was 7 (95%CI=5-9). The associated density estimate is 2.0 fish/100m2 with first pass CPUE of 1.4 age-1 and older wct/100m2 and 15.7 age-1 and older wct/hr. Population estimates have been conducted on this section of Fern Creek almost annually since 2009, with the exception of 2013 (Table 11). Population density dropped off substantially between 2013 and 2014.

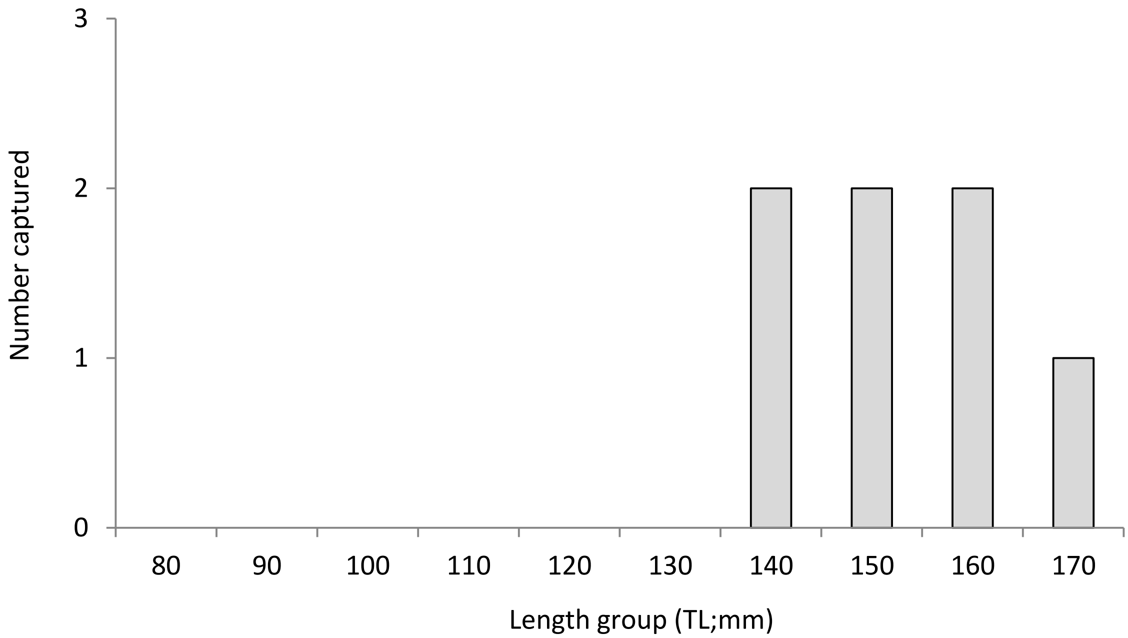


Figure 5. Length-frequency histogram for wct captured in Fern Creek, Glacier National Park, in 2014.

Table 10. 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, 2014. Length range (mm) represents all fish captured.

Species	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Weight Range	Fulton Condition Factor (K) (SD)(n)
WCT	148.3 (13.1)(7)	133-169	41.1(11.1)(7)	31-57	1.24 (0.11)(7)

Table 11. Population estimates conducted over time in Fern Creek, Glacier National Park.

Year	Estimate (95% CI)	Density (fish/100m2)
2009	15 (13-17)	4.3
2010	15 (13-17)	5.3
2011	17 (15-19)	3.8
2012	28 (18-38)	8.2
2014	7 (5-9)	2.0

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. The sampling reach length was approximately 88m while average wetted width was 4.1m. Water temperature and conductivity for 2014 was 11°C and 80µS, respectively. Our electrofishing settings were 650 volts at 30 Hertz on a 9% duty cycle and a 3 millisecond pulse width. Both adult and juvenile tailed frogs were abundant.

Muir Creek

Muir Creek is a third order tributary to the M. Fk. Flathead River (Weaver et al. 1983). It flows largely southwest and drains approximately 34.8 km^2 of land area (Weaver et al. 1983). Genetic samples collected in 2010 indicated the lower reaches of Muir Creek contained 99% genetically pure westslope cutthroat trout while upper reaches contained 100% genetically pure wct (C. Muhlfeld, USGS, personal communication). The only species of fish documented in Muir Creek by previous researchers (Weaver et al. 1983, Downs et al. 2011).

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

We did not electrofish Muir Creek in 2013 but rather snorkeled a much longer, 196.9m stretch beginning at the bottom end of our electrofishing reach and extending upstream to a 6m tall cascade/falls. Sixteen wct were observed by snorkelers in 2013 (8/21/2013) in this larger reach, substantiating the decline in abundance of wct observed using electrofishing in 2012 (Table 12). We repeated the snorkeling effort in 2014 (8/13/2014) and observed a total of 31 age-1 and older wct, a considerable improvement over the 2013 observations (Table 12).

We do not know why densities of wct in the lowermost reaches of Muir Creek have dropped off in recent years, but it is possible environmental factor(s), predation, or illegal harvest may have contributed.

Table 12. Size class, species, and count for fish observed during snorkel surveys in Muir Creek in 2013 and 2014.

Species	Lengths	2013	2014
WCT	<45 mm	0	0
	45-149 mm	5	11
	>150 mm	11	20
	Age 1+ /100m2	0.02	5.2
MWF	<45 mm	0	0
	45-149 mm	0	1
	>150 mm	0	0
	Age 1+ /100m2	0	0.01
BLT	<50 mm	0	0
	50-149 mm	0	0
	>150 mm	1	0
	Age 1+ /100m2	0.01	0

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/100m^2 in their lower-most sampling reach. Although higher than our recently observed 5.2 age-1 and older wct/100m^2, both estimates suggest a relatively strong density of wct. These data suggest Muir Creek remains an important conservation area for wct within GNP but one where the population dynamics are not fully understood.

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Average wetted-width measured 4.1m. Water temperature was measure a 10^{o}C at the time of sampling. No amphibians were observed in Muir Creek in 2013 or 2014, however tailed frogs likely remain present.

St. Mary River Drainage

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 roads in the Lee Creek drainage within the park, although the Chief Mountain Highway crosses Lee Creek near the Canadian border.

Lee Creek supports a run of migratory bull trout from the St. Mary River/St. Mary Reservoir (Mogen and Kaeding 2004). Redd counts have been conducted by the NPS annually since 2011. In addition to blt, wct and mwf are present in Lee Creek (Mogen and Kaeding 2004).

Both bull trout and wct were captured in Lee Creek 2014. The average length of age-1 and older blt was 118.8 mm in 2014 (Table 13; Figure 6). Average length of age-1 and older wct was 108.3 mm (Table 13; Figure 7). Average Fulton Condition Factor (K) for both age-1 and older blt and wct was estimated at 0.9 (Table 13).

Within the 120.5m long sample reach we estimated the total abundance of age-1 and older bull trout to be 24 (Table 14). The density estimate for these same fish was 4.4 blt/100m^2. First pass CPUE for age-1 and older blt was estimated at 38.1 fish/hr and 2.8 fish/100m^2.

In 2014 the estimated total abundance of age-1 and older wct was 11 (Table 14). The density estimate for these same fish was 2.0 wct/100m^2. First pass CPUE for age-1 and older wct was estimated at 17.8 fish/hr and 1.3 fish/100m^2.

Table 13. 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, 2011-2014. 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)	0.9
	2011	123.5(64.26)(27)	73-425	32.0(93.9)(27)	0.9
	2012	115.5(32.3)(30)	42-186	16.8(13.8)(30)	0.9
	2013	112.6 (30.1)(20)	68-211	17.7 (16.5)(19)	1.0
	2014	118.8 (21.3)(23)	44-156	15.5 (7.5)(23)	0.9
WCT					
	2009	121.5(36.7)(26)	73-240	22.5(25.0)(26)	1.0
	2011	147.9(41.3)(17)	80-234	42.7(35.5)(17)	1.1
	2012	126.3(54.0)(26)	41-223	30.3(32.0)(26)	0.9
	2013	117.3 (36.8)(29)	61-219	24.3 (27.5)(29)	1.1
	2014	108.3 (32.8)(11)	64-154	14.0 (10.7)(11)	0.9

Table 14. First pass catch per unit effort (CPUE), population estimates and densities for age-1 and older blt and wct captured in Lee Creek, GNP, 2009, 2011-2014.

Species	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
	2013	21 (17-25)	4.1	3.0	33.2
	2014	24 (20-28)	4.4	2.8	38.1
WCT					
	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
	2013	30 (26-34)	5.9	4.7	50.2
	2014	11 (9-13)	2.0	1.3	17.8

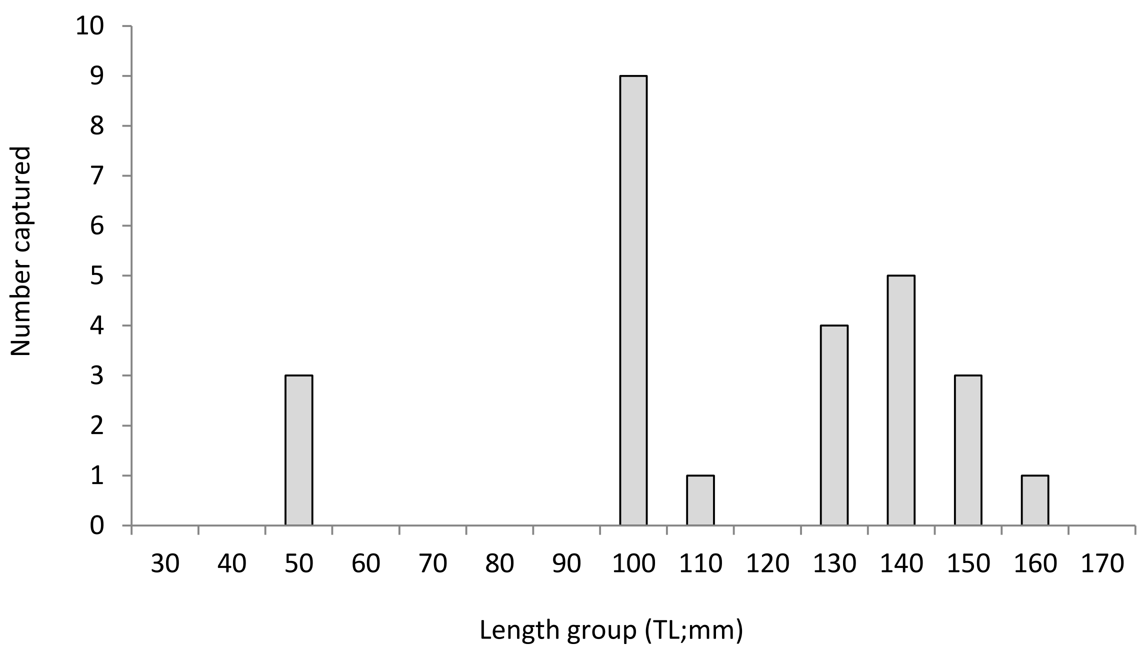


Figure 6. Length-frequency histogram for blt captured in Lee Creek, Glacier National Park, in 2014.

Cobble was estimated to be the dominant substrate type followed by gravel in the study reach. We employed 600 volts at 30 hertz, with a 25% duty cycle and a 3 ms pulse width to capture fish in Lee Creek. No amphibians were observed. A thermograph was installed approximately 100m downstream of the Chief Mountain Highway crossing, and water temperatures reached a high of 12.3^{0}C on August 27, 2014 (Figure 8).

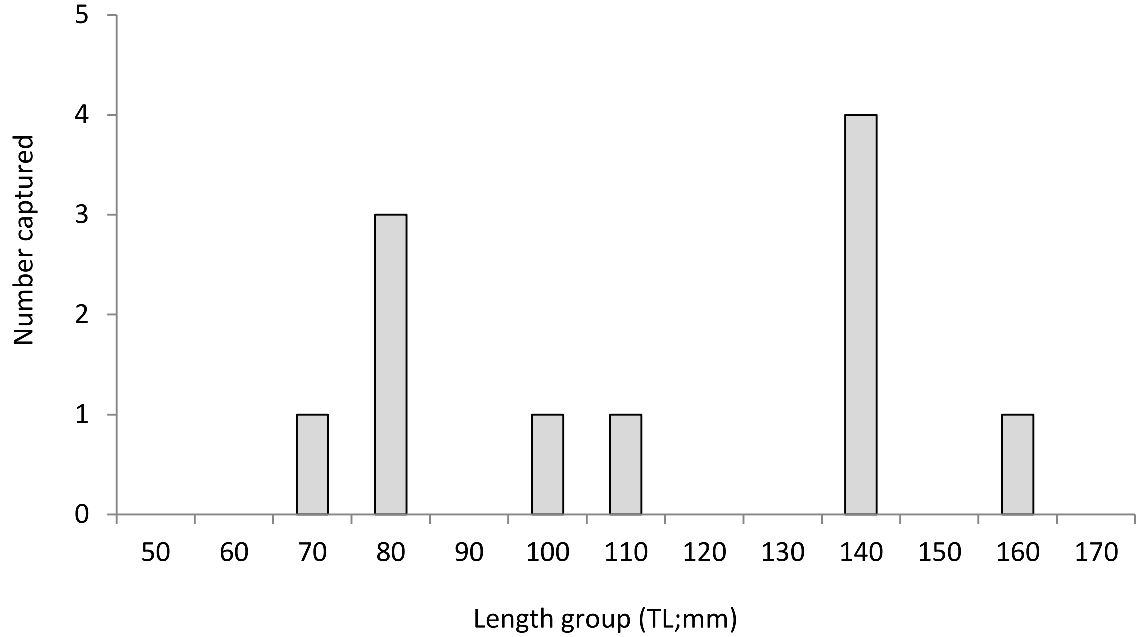


Figure 7. Length-frequency histogram for wct captured in Lee Creek, Glacier National Park, in 2014.

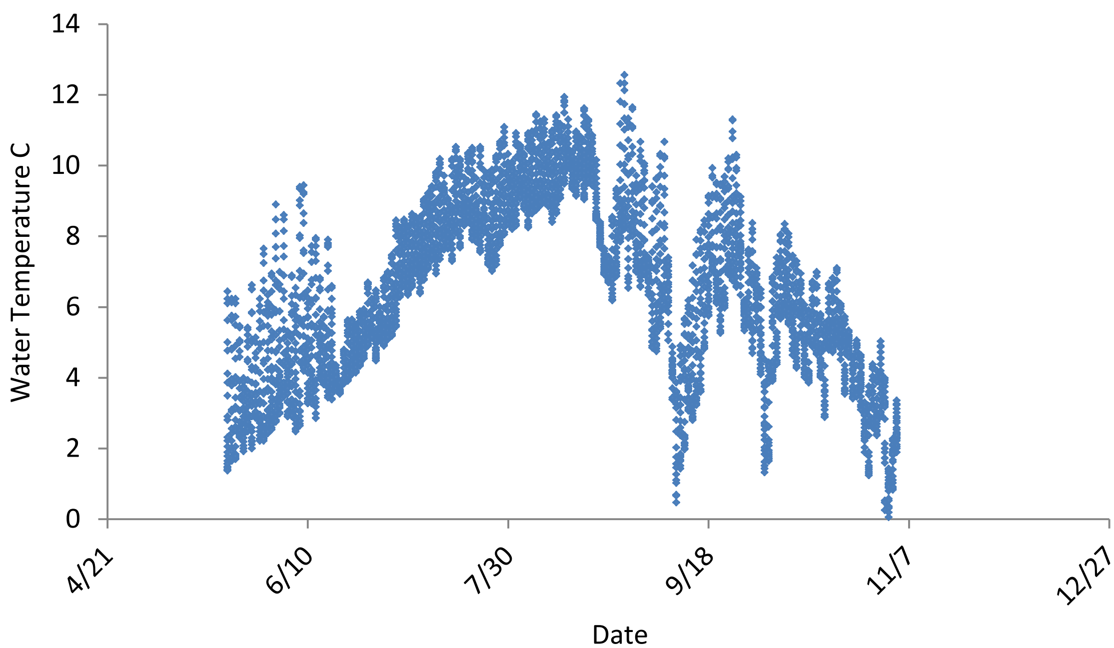


Figure 8. Water temperature recorded in 2014 in Lee Creek, Glacier National Park, Montana.

Other Stream Temperature Monitoring

Ole Creek

We also installed a thermograph in Ole Creek (Middle Fork Flathead River drainage). The thermograph on Ole Creek was located approximately 20m upstream of the suspension bridge at the trail crossing on lower Ole Creek . Stream temperature reached a high of 13.9 ^{o}C in Ole Creek on August 13, 2014 (Figure 9).

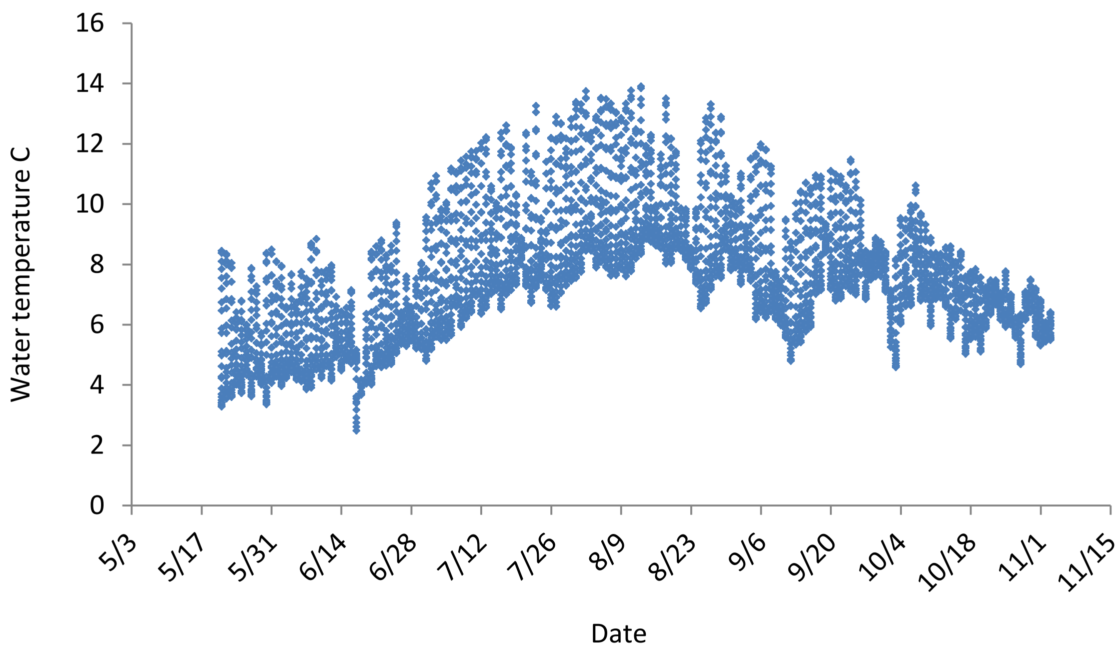


Figure 9. Ole Creek stream temperatures during 2014, Glacier National Park, Montana.

Starvation Creek

Starvation Creek is a third order tributary to the N. Fk. Flathead River (Read et al. 1982). It flows from Canada into the park prior to entering the N. Fk. Flathead River 6 km south of the Canadian border. Westslope cutthroat trout genetic samples were collected in 2008 and did not show any evidence of hybridization (MFWP, unpublished data). Previous researchers have documented bull trout, westslope cutthroat, mountain whitefish, and sculpin in Starvation Creek (Read et al. 1982, Downs et al. 2011).

We also installed a thermograph in Starvation Creek located near the confluence with the North Fork Flathead River, about 100m upstream of the trail crossing. Stream temperature peaked at 17.9 ^{o}C on July 26, 2013 (Figure 10).

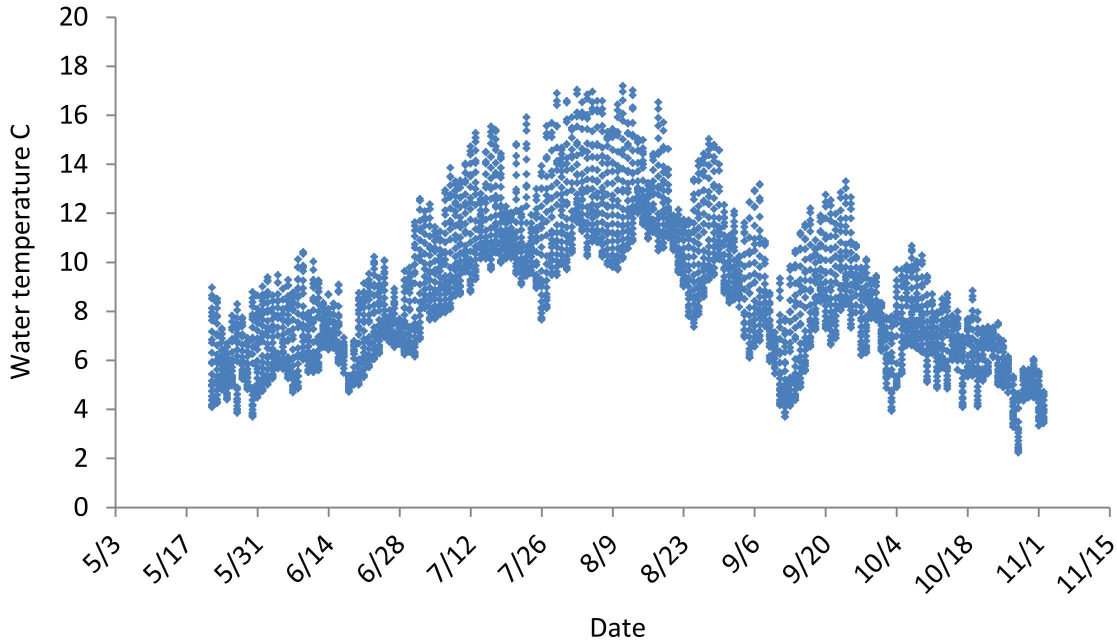


Figure 10.Stream temperature in Starvation Creek during 2014, Glacier National Park, Montana.

Cracker Lake Netting

Cracker Lake is a 40.1 acre lake located in the Swiftcurrent Creek drainage in Glacier National Park (Figure 2). The lake lies at an elevation of 1801 m and has a maximum depth 18.3 m. The lake is approximately 970 m long, with a maximum width of 258 m. The drainage is roadless, and the lake is accessed by trail from the Many Glacier Hotel (9km). The physical habitat is pristine, yet fairly unique for bull trout. The lake lies at the headwall of a large cirque basin and receives meltwater from Siyeh Glacier. As a result of suspended glacial flour, the lake is an opaque, turquoise-blue color year-around. We measured a secchi depth reading of only 0.66m at the time of sampling (7/7/14). Cracker Lake has one backcountry campground and the lake is closed to fishing.

Little is known about the life-history of bull trout in Cracker Lake. Bull trout in Cracker Lake rarely reach lengths in excess of 275mm and are generally a steel gray in color. Inlet spawning habitat is limited to approximately 100m of small stream habitat. It is not known if these fish spawn in the inlet stream, the outlet stream (Canyon Creek) or along the lakeshore.

As we had never sampled Cracker Lake previously, we considered the first night of sets to be experimental in nature and were not used in the analysis of CPUE for future monitoring purposes. The first nights sets were set fairly deep and failed to capture any fish. We had also cut a hole in the top of each trap to allow any small mammals accidentally captured in the traps to escape. We failed to capture any fish during the first night and subsequently set the traps in shallower water and closed up the escape holes in each trap. Sets were distributed around the lake where shallow shelf habitat could be accessed (Figure 11; Table 15). We were very effective at capturing bull trout the second night of sets in shallow water.

We captured a total of 64 bull trout during netting efforts. This resulted in a CPUE of 7.1 bull trout/net night. Mean length for bull trout captured was 220.7mm (Table 16). Length at capture ranged from 126 to 384 mm (Figure 12).   Mean weight and mean Wr were 93.2g and 0.80, respectively.   The Wr of the Cracker Lake bull trout is considerably lower than the 95.3 Wr observed in Lake Isabel in 2013 (Downs et al. 2013). Lake Isabel, located in the Middle Fork Flathead River Drainage of the park , supports similar sized “dwarf” form bull trout with similar fyke net catch rates (5.6 blt/net night) (Downs et al. 2013). Water clarity is very good in Lake Isabel compared to Cracker Lake, likely making it a more productive system resulting in better fish condition.

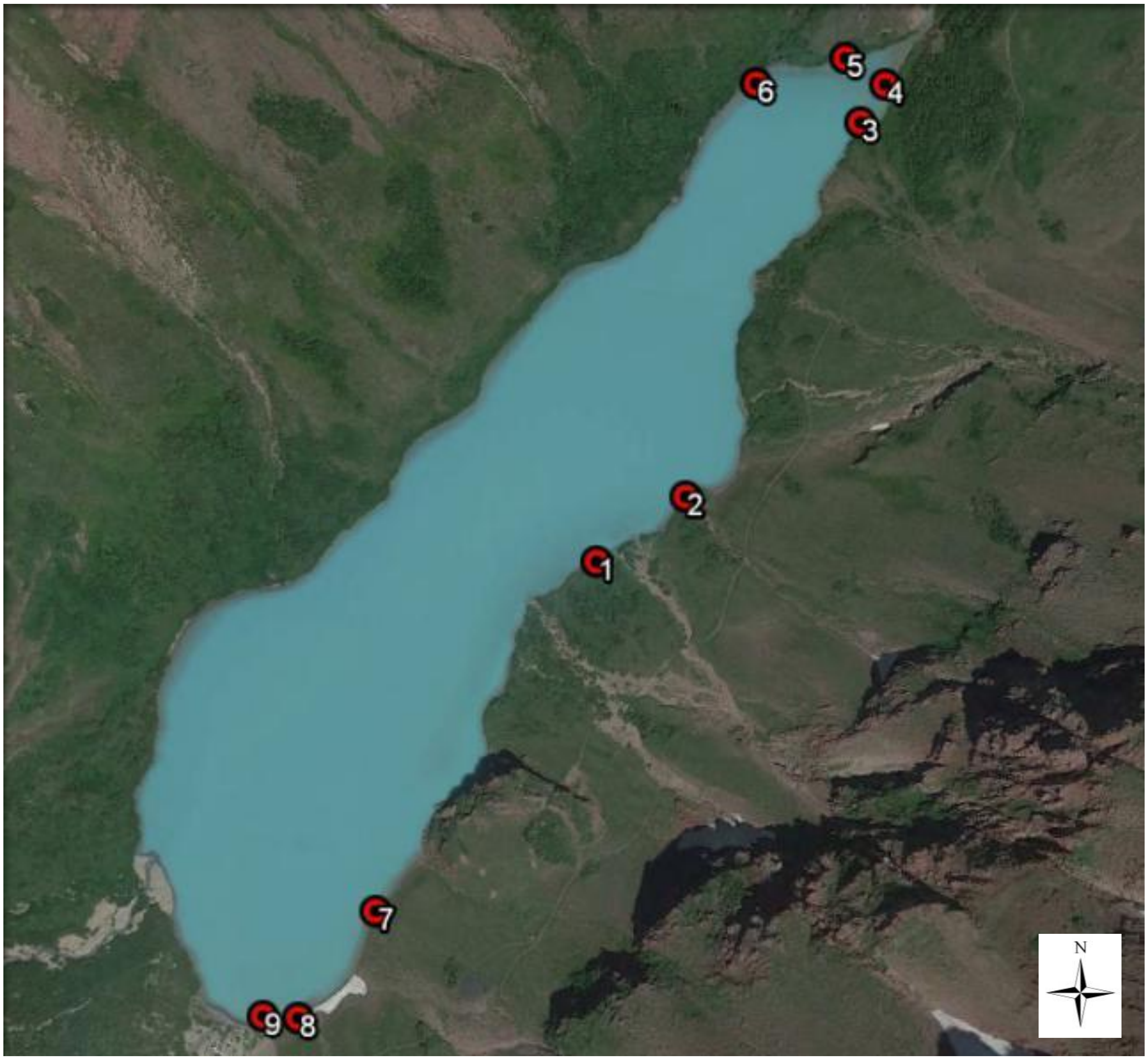


Figure 11. Locations of trap net sets in Cracker Lake, July 8-9, 2014.

We observed erosion on the lower edge of the caudal fin on several bull trout. We assumed this was related to redd construction and we used this as an indication of size at maturity. The smallest fish we observed with an eroded lower caudal fin was 227mm.

Baseline information on fish abundance and growth in Cracker Lake is particularly timely, given the shrinking size of Siyeh Glacier(predicted to be gone within 30 years). When the glacier and the glacially-derived rock flour are gone, the lake will clear. This will require adaptation by bull trout to a rapidly changing environment.

Table 15. UTM's and set parameters of Cracker Lake live trapping. For Fyke (trap) nets, "Depth" refers to the depth of the trap entrance.

Net	UTM_X	UTM_Y	Date Set	Time Set	Date Pulled	Time Pulled	Depth (feet)	No. blt captured
Fyke Net 4	305713	5402357	7/8/14	1330	7/9/14	1130	2	7
Fyke Net 5	305790	5402409	7/8/14	1400	7/9/14	1115	2	7
Fyke Net 6	305948	5402718	7/8/14	1435	7/9/14	0805	2	1
Fyke Net 7	305970	5402749	7/8/14	1453	7/9/14	0825	3	7
Fyke Net 8	305937	5402772	7/8/14	1459	7/9/14	0920	2	9
Fyke Net 9	305861	5402752	7/8/14	1518	7/9/14	1018	2	7
Fyke Net 10	305517	5402070	7/9/14	1509	7/10/14	0610	2	9
Fyke Net 11	305449	5401983	7/9/14	1521	7/10/14	0625	2	10
Fyke Net 12	305419	5401985	7/9/14	1530	7/10/14	0640	2	7
Hoop Net 1	305519	5402096	7/7/14	1700	7/8/14	0730	21	0
Hoop Net 2	305410	5401982	7/7/14	1740	7/8/14	0745	10	0
Hoop Net 3	305340	5402112	7/7/14	1750	7/8/14	0750	8	0
Hoop Net 4	305481	5402034	7/7/14	1800	7/8/14	0800	9	0
Hoop Net 5	305696	5402354	7/8/14	1342	7/9/14	1145	2	0
Hoop Net 6	305803	5402411	7/8/14	1405	7/9/14	1115	2	0
Hoop Net 7	305937	5402704	7/8/14	1442	7/9/14	1130	2	0
Hoop Net 8	305875	5402758	7/8/14	1505	7/9/14	1145	2	0

Table 16. Mean length (TL; mm), weight (g), standard deviation (SD), sample size (n) and Fulton Condition Factor (K) of age-1 and older blt captured in Cracker Lake, GNP, 2014. Length range (mm) represents all fish captured.

Species	Mean Length (SD) (n)	Length Range	Mean Weight (SD) (n)	Weight Range	Fulton Condition Factor (K) (SD)(n)
WCT	220.7 (40.9)(64)	126-384	93.2(55.8)(63)	25-446	0.80 (0.08)(63)

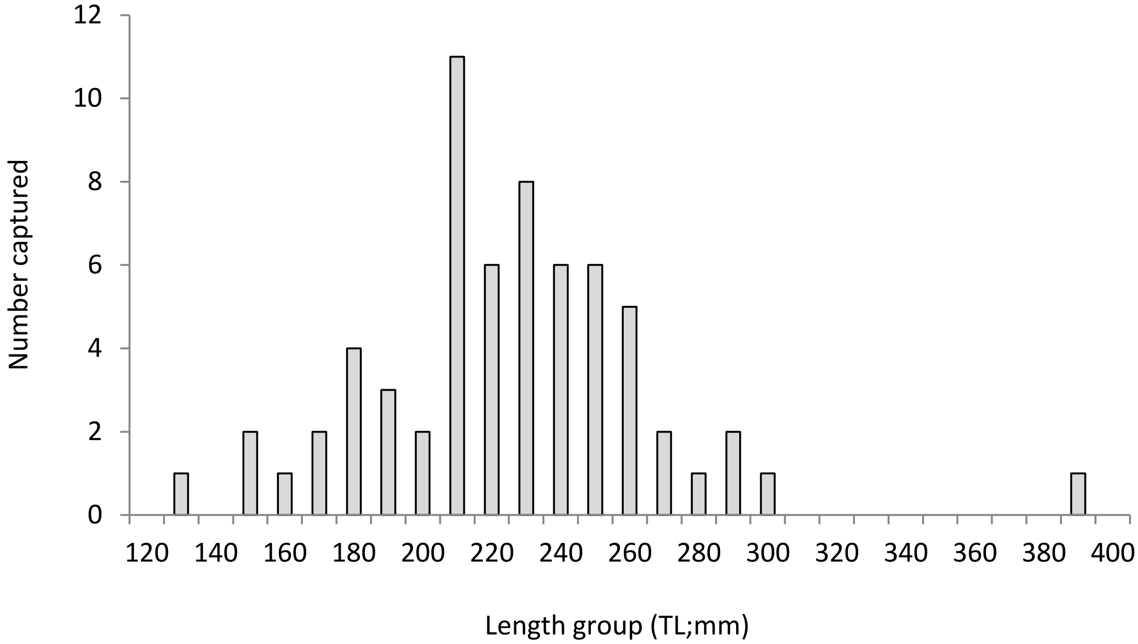


Figure 12. Length-frequency histogram for blt captured in Cracker Lake, Glacier National Park, in 2014.

Logging Lake Netting

Logging Lake is a 1,114 acre lake located in the North Fork Flathead River drainage in Glacier National Park (Figure 2). The lake lies at an elevation of 1,161 m and has a maximum depth 60.4 m. The lake is approximately 8.5 km long, with a maximum width of 700 m. The drainage is roadless, and the lake is accessed by trail from the Inside North Fork Road (8.4km). The physical habitat is pristine, however, lake trout have compromised the lake over the past 30 years and have increased in abundance to the point where they are likely approaching carrying capacity.

The NPS has recently initiated a project to reduce lake trout abundance in Logging Lake to benefit native bull trout, which are approaching functional extinction in the lake. We began the project with an initial translocation effort to move as many of the remaining bull trout in Logging Creek upstream into Grace Lake where lake trout are not present. We also conducted some limited experimental trapping using miniature fyke nets in an attempt to capture subadult bull trout in Logging Lake itself. We report the results of the experimental fyke netting here. Although no bull trout were captured in these efforts, the data collected will add to the baseline of information available to evaluate the impacts of the lake trout suppression project.

We fished a total of five fyke net nights between 8/19/14-8/21/14. Traps were set such that the trap entrance was set between 2 and 5' of depth (Table 17). We captured a total of 127 fish comprising five species (Table 18). Northern pikeminnow were the most abundant species captured. Water temperature was fairly warm (18^{o}C) and as a result, we only captured a total of two trout (both westslope cutthroat).

Table 17. UTM's and set parameters of Logging Lake live trapping. For Fyke (trap) nets, "Depth" refers to the depth of the trap entrance.

Net	UTM_X	UTM_Y	Date Set	Time Set	Date Pulled	Time Pulled	Depth (feet)
Fyke Net 1	717459	5405643	8/19/14	1930	8/20/14	0820	4
Fyke Net 2	717459	5405643	8/19/14	1930	8/20/14	0910	5
Fyke Net 3	718657	15406674	8/19/14	2018	8/20/14	0830	2
Fyke Net 4	718247	5406541	8/20/14		8/21/14	0750	3
Fyke Net 5	718180	5406501	8/20/14		8/21/14	0818	4

Table 18. Fyke trap catch statistics for Logging Lake sampling.

Species	Number Captured	CPUE (fish/net-night)
BLT	0	0.0
LNS	10	2.0
LSS	52	10.4
MWF	3	0.6
NPM	60	12.0
WCT	2	0.4

ACKNOWLEDGEMENTS

The authors wish to thank NPS employees Nate Muhn and Jon McCubbins for assistance with field sampling.

LITERATURE CITED

Anderson, R.O. and R.M. Neumann. 1996. Length, weight, and associated structural indices. Pages 447-482 *in* B.R. Murphy and D.W. Willis, editors. Fisheries Techniques, second edition. American Fisheries Society, Bethesda, Maryland.

Downs, C.C. 1995. Age determination, growth, fecundity, sexual maturity, and longevity for isolated, headwater populations of westslope cutthroat trout. MS Thesis. Montana State University, Bozeman.

Downs, C.C., C. Stafford, H. Langner, and C.C. Muhlfeld. 2011. Glacier National Park Fisheries Inventory and Monitoring Bi-Annual Report, 2009-2010. National Park Service, Glacier National Park, West Glacier, Montana.

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.

Meeuwig, M., C. Guy, W.A. Fredenberg. 2007. Research Summary for Action Plan to Conserve Bull Trout in Glacier National Park, Montana. Montana State University, Bozeman.

Meeuwig, M.H. 2008. Ecology of lacustrine-adfluvial bull trout populations in an inter-connected system of natural lakes. Ph.D. Dissertation. Montana State University, Bozeman.

Mogen, J.T. and L.R. Kaeding. 2004. Bull trout (Salvelinus confluentus) use of tributaries of the St. Mary River, Montana. Report to the Bureau of Reclamation by the U.S. Fish and Wildlife Service. Bozeman, Montana.

Muhlfeld, C.C., T.E. McMahon, M.C. Boyer, and R.E. Gresswell. 2009. Local habitat, watershed, and biotic factors influencing the spread of hybridization between native westslope cutthroat trout and introduced rainbow trout. Transactions of the American Fisheries Society 138: 1036-1051.

Muhlfeld, C.C., T.E. McMahon, D. Belcer, and J.L. Kershner. 2009b. Spatial and temporal spawning dynamics of native westslope cutthroat trout *Oncorhynchus clarkii lewisi*, introduced rainbow trout *O. mykiss,* and their hybrids. Canadian Journal of Fisheries and Aquatic Sciences 66:1153-1168.

Read, D., B.B. Shepard, and P.J. Graham. 1982. Fish and habitat inventory of streams in the North Fork Drainage of the Flathead River. Montana Department of Fish, Wildlife and Parks, Helena.

Scarnecchia, D.L. and E.P. Bergersen. 1986. Production and habitat of threatened and endangered greenback cutthroat and Colorado River cutthroat trout in Rocky Mountain headwater streams. Transactions of the American Fisheries Society 115:382-391.

Thurow, R.F. 1994. Underwater Methods for Study of Salmonids in the Intermountain West. US Department of Agriculture. Forest Service, Intermountain Research Station. General Technical Report INT-GTR-307.

Weaver, T.M., J.J. Fraley, and P. Graham. 1983. Fish and habitat inventory of streams in the Middle Fork Drainage of the Flathead River. Montana Department of Fish, Wildlife, and Parks, Helena.

Zippin, C. 1958. The removal method of population estimation. Journal of Wildlife Management. 22(1): 82-90.

Aquatic Invasive Species Prevention and Monitoring

ABSTRACT

In 2014, Glacier National Park (GNP) continued to implement a program of mandatory boat inspection and launch permitting for all non-hand powered boats entering GNP as well as 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. Six boats were denied permits to launch for reasons ranging from standing water in the boat to dried vegetation adhered to the hull. In 2013, GNP conducted inspections on 1,174 issued 1,164 permits to launch boats. Ten boats were denied permits to launch for reasons ranging from standing water in the boat to dried vegetation adhered to the hull. In 2014, 1,085 boats were inspected for AIS, 1,079 launch permits were issued and six boats were denied launch permits . Reasoning for launch denial ranged from standing water in the boat to boat was launched within the last 30 days in mussel infected waters. Boats entered GNP from 17 States in 2014, 13 of which have populations of invasive mussels demonstrating the continued risk to park waters. No AIS were found on any of the boats inspected over the past four years. In addition, we continued the parks monitoring program for AIS, including both artificial substrates to monitor for adult mussels and water sampling for the presence of mussel veligers. No AIS have been detected in any park waters.

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 loostrife *Lythrum salicaria*, and others are present within a three hour drive of the park, and New Zealand mudsnails are present in southwest Montana. Taken together, the potential transport and establishment of additional AIS into park waters is a serious threat. In response, park managers are taking proactive steps to reduce the risk.

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

METHODS

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

A public education effort regarding the risks of AIS to GNP waters continued in 2014. 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.

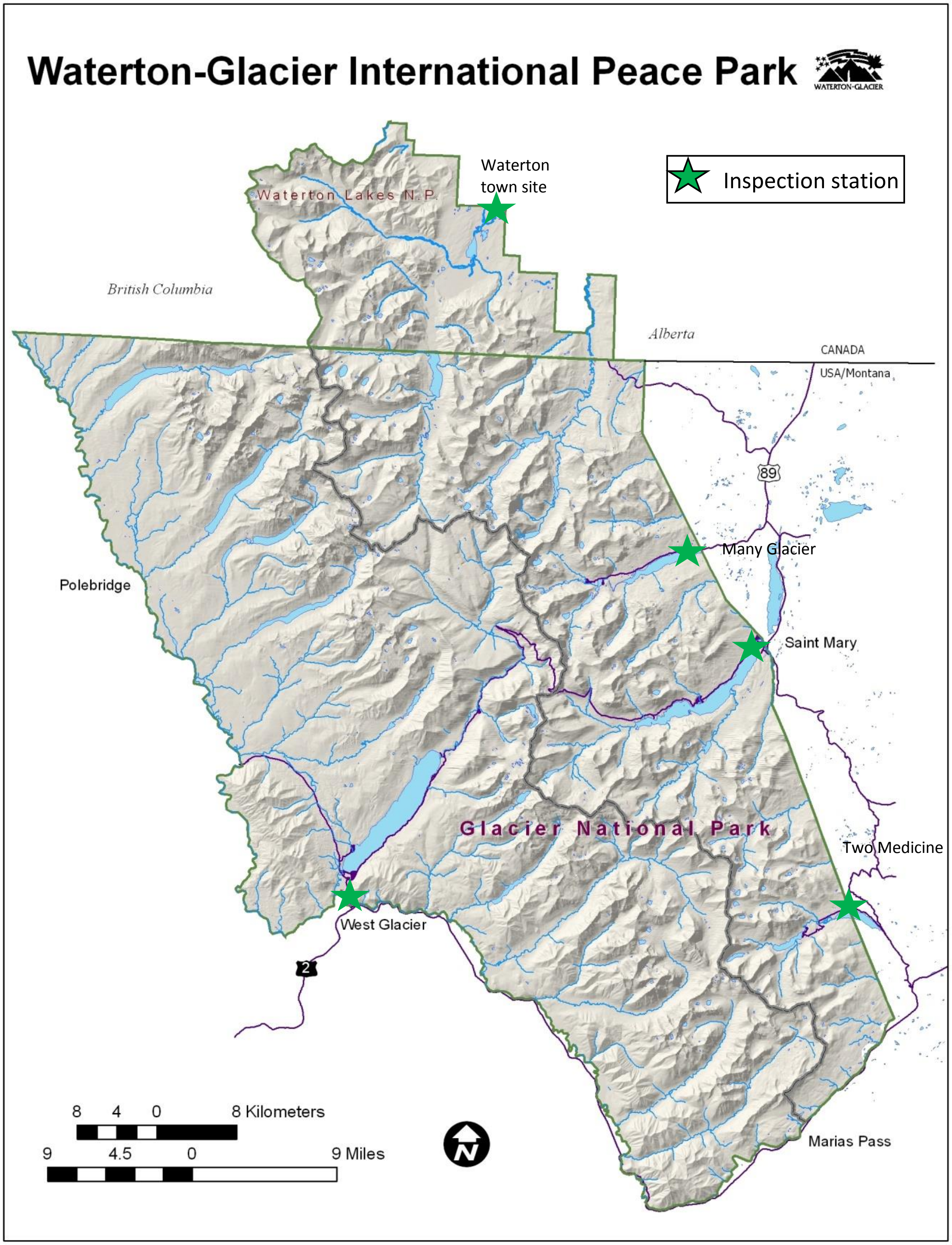


Figure 1. AIS boat inspection and launch permitting locations in Glacier National Park, 2014.

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, and Two Medicine 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 during the fall. Upon retrieval, artificial substrates were inspected for the presence of adult and sub-adult mussels.

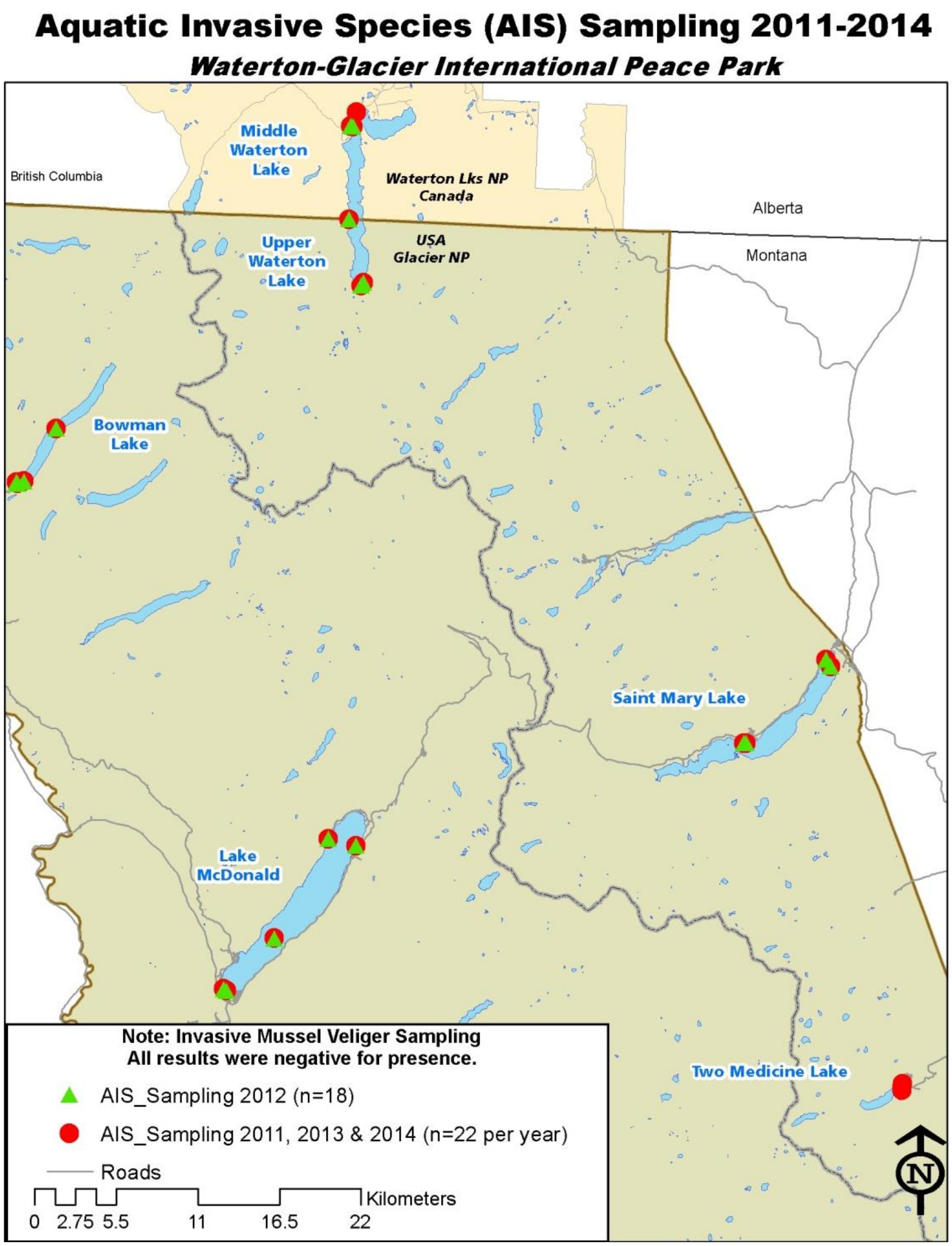


Figure 2. Invasive mussel larvae (veliger) sampling in Waterton-Glacier International Peace Park, 2011-2014.

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^{0}F (11^{0}C). We used a standard 64 µm mesh vertical plankton sampling net and we made triplicate vertical hauls at each site. The net was lowered into the water to a maximum depth of 20m or to within 1m of the lake bottom (if sample site depth < 20m) and then slowly retrieved to the water surface. Samples were preserved in 100% non-denatured ethanol, using a 1:2 sample:ethanol ratio. Veliger monitoring sites typically included a sample at the foot of the lake near the boat launch, one at mid-lake, and one at the upper end of the lake. GIS coordinates were taken for all sampling locations (Table 1)**.**

Table 1. Invasive mussel veliger sampling locations in GNP, 2014.

Date Collected	UTM X	UTM Y	Water Body
7/29/2014	279246	5379419	Lake McDonald
7/29/2014	286109	5389646	Lake McDonald
7/29/2014	279054	5379533	Lake McDonald
7/29/2014	287973	5389188	Lake McDonald
7/29/2014	282478	5382948	Lake McDonald
7/30/2014	288334	5426879	Upper Waterton Lake
7/30/2014	288479	5427078	Upper Waterton Lake
7/30/2014	287529	5431368	Upper Waterton Lake
7/30/2014	287794	5437616	Upper Waterton Lake
7/30/2014	287627	5437649	Upper Waterton Lake
7/30/2014	288034	5438579	Upper Waterton Lake
7/30/2014	324816	5372720	Two Medicine Lake
7/30/2014	324819	5372910	Two Medicine Lake
7/30/2014	324849	5373164	Two Medicine Lake
7/30/2014	314133	5396082	St. Marry Lake
7/30/2014	314278	5396090	St. Marry Lake
7/30/2014	320008	5401233	St. Marry Lake
7/30/2014	319682	5401708	St. Marry Lake
7/31/2014	705896	5412597	Bowman Lake
7/31/2014	705500	5412409	Bowman Lake
7/31/2014	705397	5412481	Bowman Lake
7/31/2014	707768	5416289	Bowman Lake

RESULTS AND DISCUSSION

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

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

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

In 2014, GNP conducted inspections on 1,085 boats and issued 1,079 permits to launch. Six boats were denied permits to launch, reasoning for launch denial ranged from standing water in the boat to the boat having been launched within the last 30 days in mussel infected waters. No AIS were detected. Approximately eighty 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 (9%), Two-Medicine Ranger Station (8%), and the Polebridge Ranger Station (1%), and Many Glacier (<2%). Boats entered GNP from 17 States and 3 Canadian Provinces this year (Table 2). Twelve of the 17 States have populations of zebra and/or quagga mussels, demonstrating the continued risk of introduction. Eighty-five percent of all boats inspected were registered in Montana.

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

State	2011	2012	2013	2014	Total
Alabama	0	1	0	0	1
Arizona	8	8	4	12	32
Arkansas	0	0	1	0	1
California	19	14	8	8	49
Colorado	3	2	4	4	13
Illinois	1	2	1	0	4
Indiana	1	1	2	1	5
Louisiana	0	0	1	0	1
Maryland	0	0	1	0	1
Michigan	0	1	1	4	6
Minnesota	5	3	1	3	12
Missouri	3	0	1	0	4
Nevada	1	0	2	1	4
New Mexico	0	0	2	0	2
New York	0	2	0	2	4
North Dakota	0	0	2	1	3
Oklahoma	0	1	0	1	2
Texas	1	2	1	1	4
Utah	1	3	0	0	4
Wisconsin	3	1	1	0	5
Wyoming	0	0	6	0	6

Water temperatures were recorded throughout the high boating-use summer months on waters with developed boat ramps. Zebra and Quagga mussels will spawn when water temperatures reach and remain above 11°C (52°F). All waters sampled had extended time-periods when water temperatures remained consistently above this level in 2014 (Figures 3-5). Water temperature measurements suggest a potential summer spawning season for invasive mussels in park lakes lasting from two months (Two Medicine Lake) to three months (Lake McDonald).

We timed our invasive mussel veliger sampling (plankton hauls) to correspond with peak water temperatures. Sampling in 2014 occurred from 29 July-31 July. We sampled all of the primary boating waters in the park (Figure 2). No invasive mussel larvae were detected. Furthermore, invasive mussel substrates were removed at the end of the boating season and inspected for the presence of invasive mussels each year. None were found.

Bowman Lake reached a maximum temperature of 21.3 °C on 8/11/2014 (Figure 3). Two Medicine Lake reached a maximum temperature of 17.3 °C on 8/07/2014 (Figure 4). Lake McDonald reached a maximum temperature of 22.5 °C on 8/1/2014 (Figure 5).

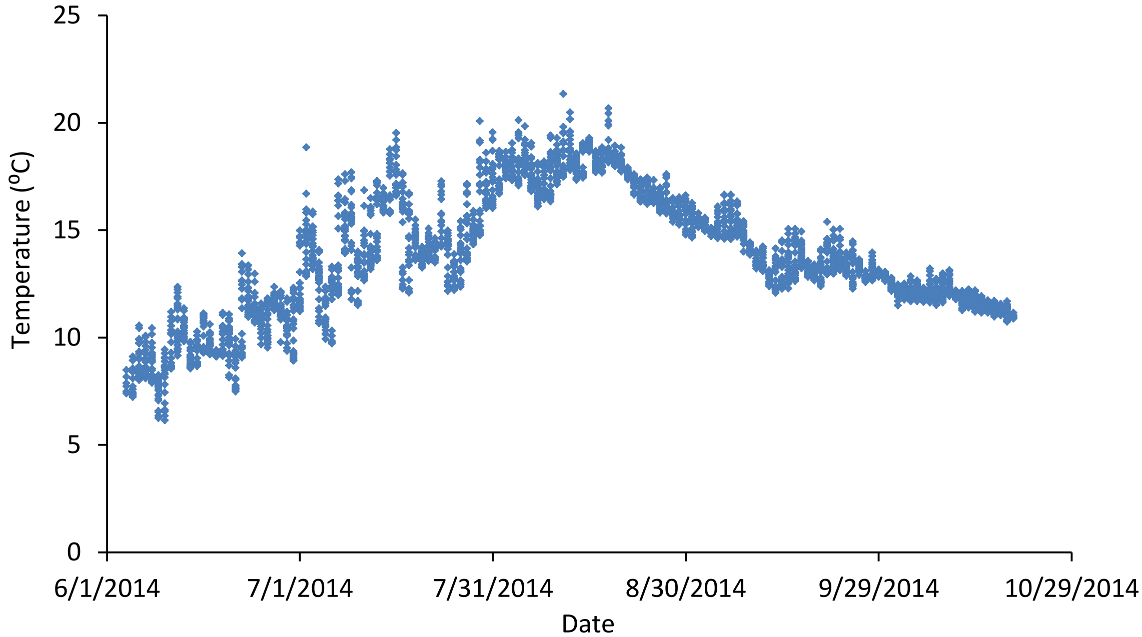


Figure 3. 2014 Bowman Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in approximately four feet of water.

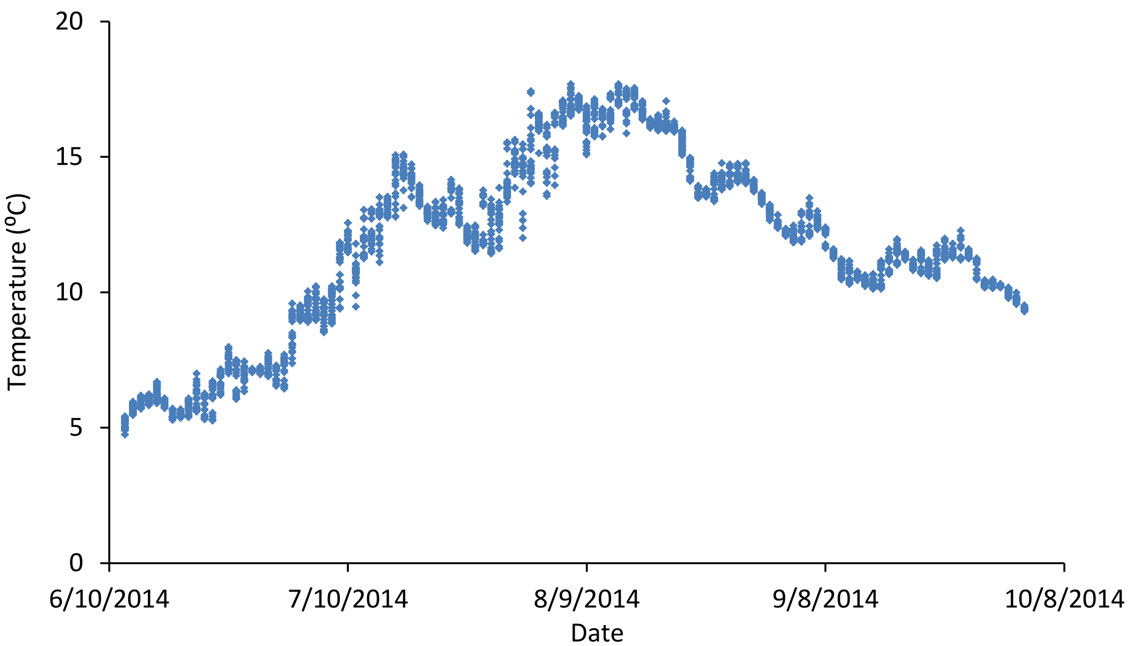


Figure 4. 2014 Two-Medicine Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock.

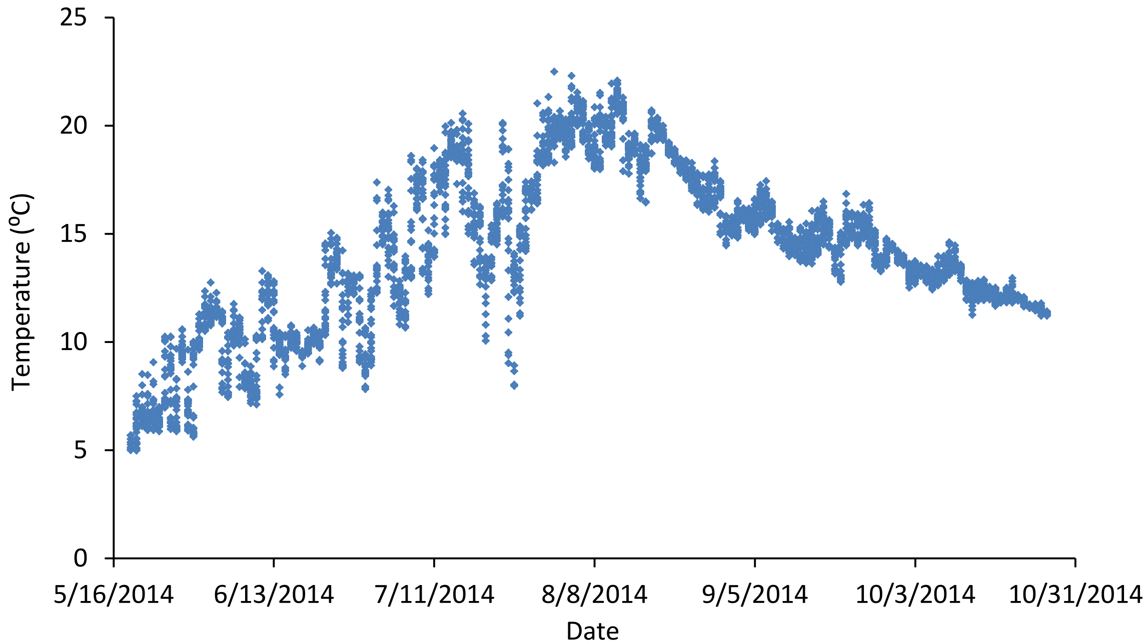


Figure 5. 2014 Lake McDonald Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock.

2010 saw the beginning of more aggressive prevention and monitoring efforts by the NPS aimed at preventing additional AIS from colonizing park waters. Initiation of this effort was very timely given the westward movement of AIS such as zebra and quagga mussels and the program should continue into the future. The current GNP AIS prevention program of inspecting all non-hand-propelled water craft (largely motorized watercraft), along with self-inspection and certification of hand-propelled watercraft such as canoes and kayaks can be expected to significantly reduce the risk of unintended transport of AIS by boats into park waters. As new AIS issues and threats arise, park managers will continue to be challenged in finding a balance between accommodating visitor use of GNP and providing vigilant ecosystem protection and conservation.

ACKNOWLEDGEMENTS

The authors wish to thank NPS employees Nathan Muhn and Jon McCubbins for assistance with field sampling.

LITERATURE CITED

Connelly, N.A., C.R. O'Neill Jr., B.A. Knuth, T.L. Brown. 2007. Economic impact of zebra mussels on drinking water treatment and electric power generation facilities. Environmental Management 40: 105-112.

Fredenberg, W. 2002. Further evidence that lake trout displace bull trout in mountain lakes. Intermountain Journal of Sciences 8:143-151.

Griffiths, R.W., D.W. Schloesser, J.H. Leach, and D.P. Kovalak. Distribution and dispersal of the zebra mussel Dreisenna polymorpha in the Great Lakes Region. Canadian Journal of Fisheries and Aquatic Sciences 48: 1381-1388.

Ludyanksy, M.L., D. McDonald, and D. MacNeill. 1993. Impact of the zebra mussel, a bivalve invader. Bioscience 43: 533-544.

Montana Fish , Wildlife, and Parks. 2010. Officials test suspicious larvae found in Flathead Lake. MFWP Press Release. Helena, MT

Montana Fish , Wildlife, and Parks. 2010. Variety of tests finds no evidence of exotic mussels in Flathead Lake. MFWP Press Release. Helena, MT

Mills, E.L., G. Rosenburg, A.P. Spidle, M. Ludyanskiy, Y. Pligin, and B. May. 1996. A review of the biology and ecology of the Quagga mussel (*Dreissena bugensis*), a second species of freshwater Dreissenid introduced to North America. American Zoologist 36:271-286.

Ricciardi, A., R.J. Neves, and J.B. Rasmussen. 1998. Impending extinctions of North American mussels (*Unionoida*) following the zebra mussel (*Dreissena polymorpha*) invasion. Journal of Animal Ecology 67:613-619.

Vitousek, P.M., C.M. D'Antonio, L.L. Loope, and R. Westbrooks. 1996. Biological invasions as global environmental change. American Scientist 84: 468-478.

Glacier National Park Bull Trout Redd Counts

ABSTRACT

We conducted bull trout *Salvelinus confluentus* redd counts in 17 streams/stream reaches in Glacier National Park in 2014. Seven 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 259 redds across the park. Quartz Lake remains the strongest monitored bull trout population residing wholly within the park, with a total of 66 redds. 2014 redd counts for Flathead Lake migratory bull trout populations spawning in the Middle Fork Flathead River tributaries in the park were mixed compared to long-term averages. Redd counts for bull trout populations spawning in the St. Mary drainage were positive, with Boulder and Kennedy creeks having strong, above average counts. 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.

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^{0}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 (Thurow 1997; Bonneau and Scarnecchia 1998). Excess fine sediment holds the potential not only to reduce egg and embryo survival, but might also limit juvenile bull trout abundance in streams by reducing the amount of interstitial spaces available for overwinter habitat. Channel stability, habitat complexity, and connectivity are all important components in bull trout population persistence (Rieman and McIntyre 1993).

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

Redd counts, or spawning nest counts, are used across the range of bull trout to monitor population trends. They are typically used as an index of abundance to gauge the relative strength of adult escapement from year to year. They can also be used to estimate actual adult escapement by expanding the redd counts to fish numbers using various spawner to redd ratios. Redd counts require far less effort to conduct than other traditional monitoring methods such as trapping, and yet provide valuable information on bull trout at the watershed and/or population scale. However, redd counts are not without 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 2014. Redd surveys generally occur during the first full three weeks of

October. Surveys occurred between September 27-October 28. 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) suggest using at least 10 years of redd count data for trend analysis. Both Kennedy and Boulder creeks on the east side of the park, as well as Ole, Park, Nyack, and Quartz creeks on the west side of the park meet the criteria. We used a nonparametric rank-correlation procedure, Kendall's tau-b (Daniel 1990), to test for trends in "count year" versus "redd count" in the long-term redd count data set and noted statistical significance at the $\alpha = 0.05$ level (Rieman and Myers 1997).

RESULTS AND DISCUSSION

GNP, USGS, USFWS, and MFWP staff surveyed ten stream reaches in the N. Fk. Flathead River, including two new reaches, one above Trout Lake (beginning UTM 286418,5397044; end UTM 287564, 5398453) and the other upstream of Rogers Lake (beginning UTM 283801,5394284; end UTM 285132, 5395031), both are in the Camas drainage. Additionally four redd counts were conducted in the M. Fk. Flathead River drainage and, three streams were surveyed in the St. Mary River drainage by the GNP, USFWS, and Blackfeet 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 2014 were above average for Boulder and Kennedy creeks (Figure 2). We recently initiated redd counts on Lee Creek and counted 16 redds, up from 2013's count of 12. Correlations in "count year" versus "redd count" identified a statistically significant positive trend in the redd counts on Boulder Creek (tau-b = 0.61; $p<0.05$), 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 resulted in a non-statistically significant negative trend (-0.34; $p>0.05$), suggesting declining adult abundance. Bull trout habitat quality is higher in Boulder Creek. Streambank stability and bed stability are also higher. The primary spawning and rearing reach of Kennedy Creek can be characterized as relatively unstable and dominated by cobble-sized substrate. Spawning gravel is very limited. On the west side of the park, the only statistically significant trend was in the short-term redd count data (most recent 10 years) on Ole Creek, which was significantly positive (0.49; $p<0.05$).

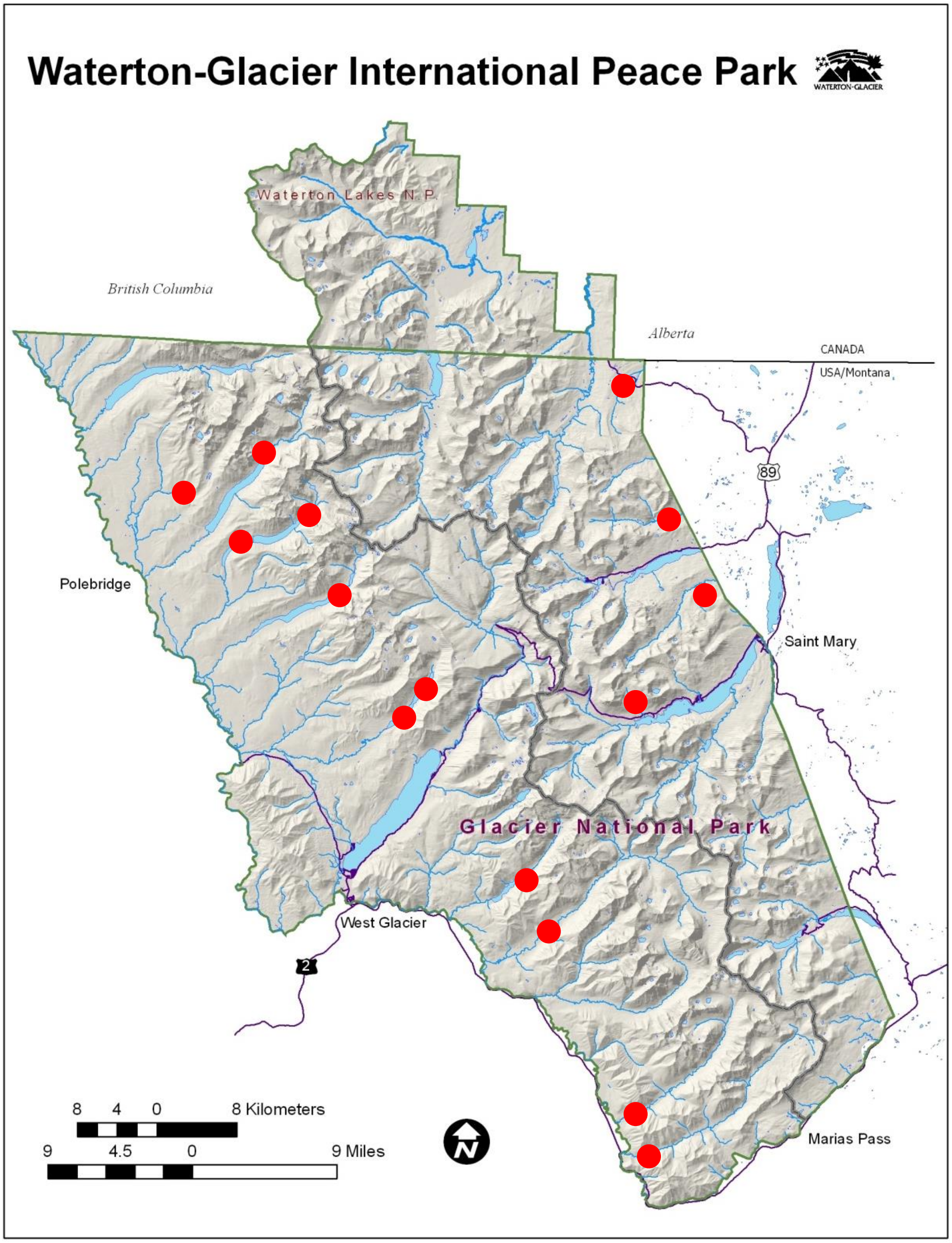


Figure 1. Drainages monitored for bull trout spawning activity (red circles) in Glacier National Park, Montana in 2014.

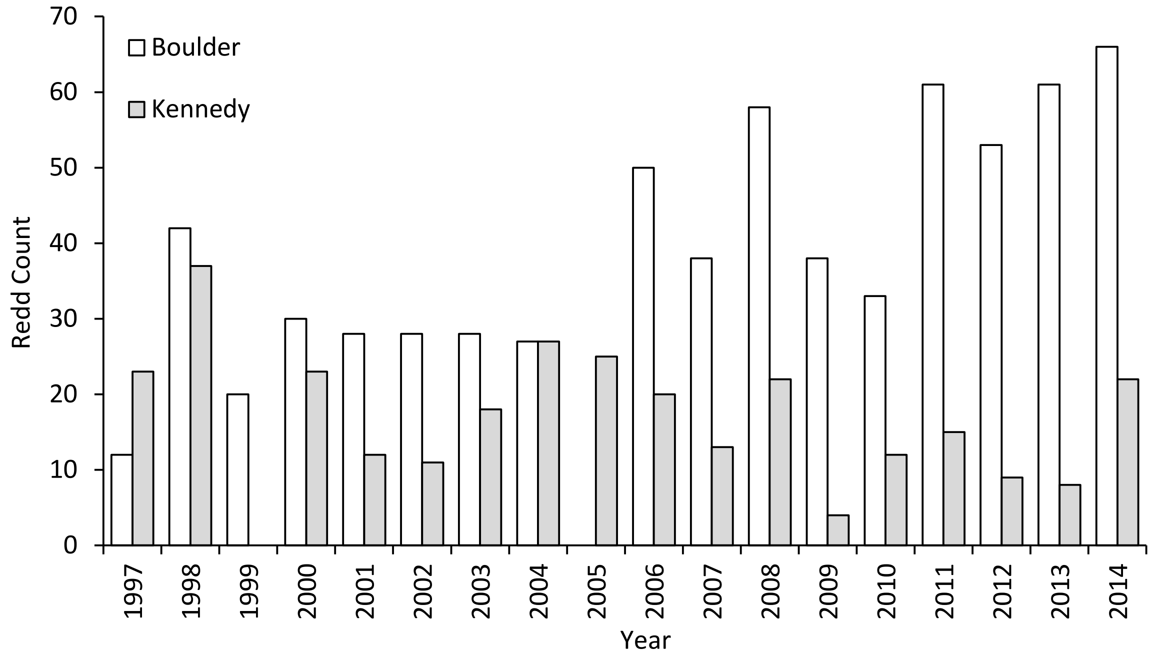


Figure 2. Bull trout redd counts for Boulder and Kennedy creeks, Hudson Bay Drainage, Glacier National Park.

Documentation of progress towards meeting recovery objectives in the St. Mary River drainage established by the USFWS (USFWS 2002) will be difficult without expanded monitoring of bull trout population abundance and trends in GNP. The four recovery criteria focus on quantitative measures of adult bull trout abundance and population trends. Recovery Criteria 1 calls for the presence of nine stable local bull trout populations in the St. Mary-Belly River Recovery Unit, well distributed across the landscape. Recovery Criteria 2 calls for documentation of at least one population in each of the six Core Areas supporting at least 100 adults annually. Recovery Criteria 3 calls for documenting a stable or increasing population of bull trout in the Recovery Unit over time, using at least 10 years of trend data. Recovery Criteria 4 addresses the need for resolution to operational issues associated with Sherburne Dam and the St. Mary Irrigation Canal operated by the U.S. Bureau of Reclamation (BOR). The most cost-effective way to evaluate progress against the first three criteria may be through bull trout redd counts, but existing efforts focusing on monitoring only 3 of the 6 core area populations may limit our ability to adequately evaluate local populations against established recovery criteria.

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

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

On the west side of GNP, both migratory stocks of bull trout from Flathead Lake as well as populations that reside entirely within the park (known locally as “disjunct” migratory populations) are monitored (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 diluviana* into the system and resulting major alterations in trophic dynamics (i.e. rapidly expanding lake trout population) in the lake, as well as drought conditions (Weaver et al. 2006). One of the most significant contemporary threats to these populations is predation with and competition by non-native fish species in both the migratory and rearing habitats of the Flathead River and Flathead Lake (Deleray et al. 1999, Muhlfeld et al. 2008).

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

The 2014 redd count for Ole Creek of 32 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 = 0.14, $p > 0.05$), but over the last 10 years there has been a statistically significant positive trend in spawner abundance in Ole Creek (tau-b = 0.73, $p > 0.05$) (Figure 3, Appendix A). The 2014 redd count for Park Creek was four, well below the long-term average of 14 (Figure 3, Appendix A). Sufficient data does not exist to analyze short-term (10 year) trends for either Nyack or Park creeks due to the intermittency of the counts.

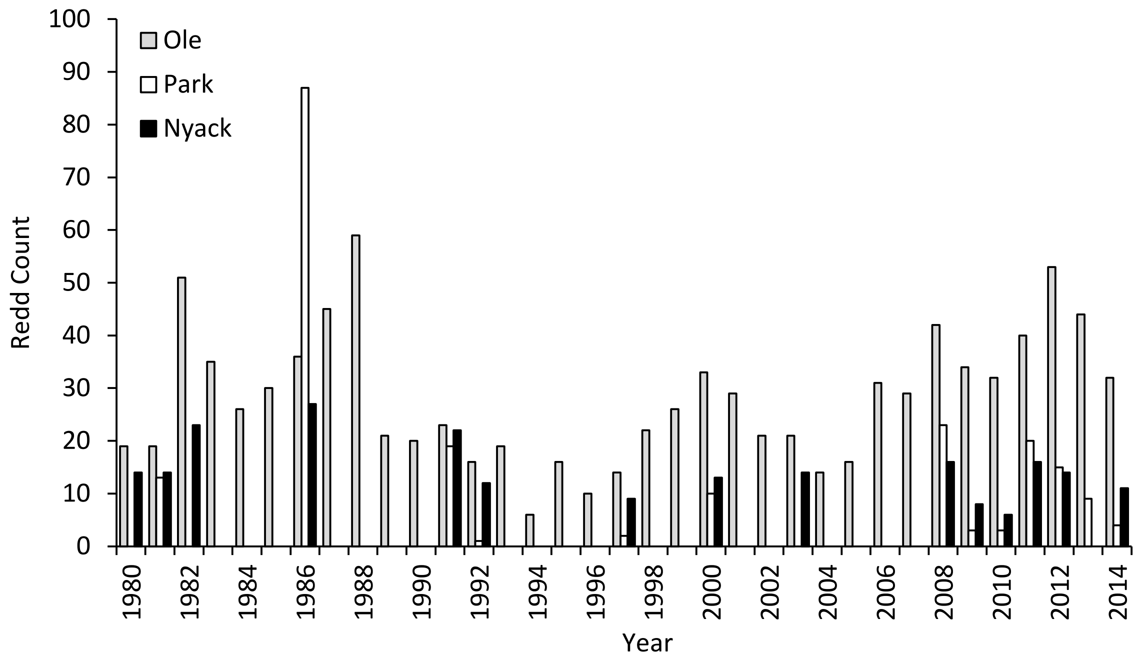


Figure 3. Bull trout redd counts conducted in Ole, Park, and Nyack creeks, Middle Fork Flathead River Drainage, Glacier National Park.

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

Expanding populations of lake trout have colonized almost all of the accessible lake habitats on the west side of GNP, and now threaten the persistence of the majority of the “disjunct” migratory bull trout populations remaining on the west side of GNP. Nine of seventeen lake-dwelling populations of bull trout located on the west side of GNP have been compromised by lake trout (Fredenberg et al. 2007), and lake trout have been documented replacing bull trout as the dominant predator in these waters, where long-term data on fish populations exists (Fredenberg 2002; Downs et al. 2011). Some populations appear to be persisting at dangerously low numbers (e.g. Bowman, Logging, and Harrison lakes), and interactions with non-native lake trout are likely the driving force behind the declines and the precarious status of bull trout in these systems (Donald and Alger 1993). In 2009, an experimental lake trout suppression program was initiated on Quartz Lake to preserve fish native populations, including bull trout and is showing promise in reducing adult lake trout abundance.

Redd counts on Quartz Creek, the primary spawning area for bull trout from Quartz Lake have not shown a statistically significant trend (tau-b = 0.09, $p > 0.05$) since monitoring began, suggesting lake trout have yet to reach levels where they are impacting bull trout population viability (Figure 4, Appendix A). The 2014 redd count was encouraging , as it was the highest historical redd count on file. However, the 2013 redd count was the lowest count since the experimental lake trout suppression began. Continued monitoring will be necessary to determine if the low count in 2013 was the result of natural variability in adult abundance, lake trout impacts, or potential gill netting/handling impacts.

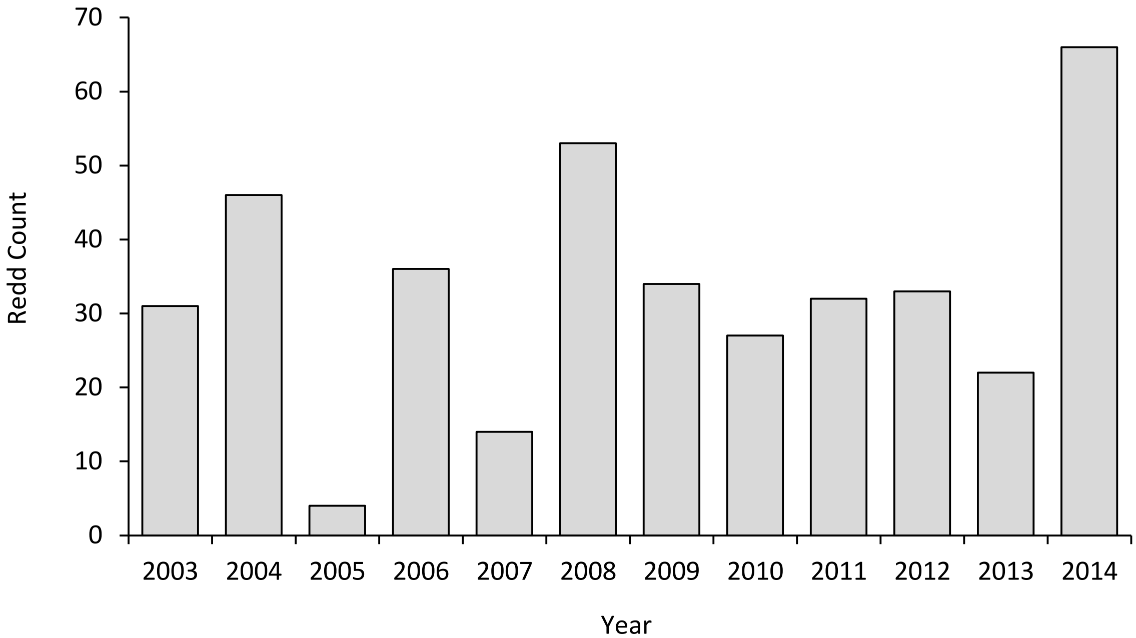


Figure 4. Bull trout redd counts in Quartz Creek, Glacier National Park, Montana.

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

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

ACKNOWLEDGEMENTS

Thanks to Jim Mogen and Robbin Wagner of the USFWS; Blackfeet Nation Fish and Wildlife Department; John Fraley of Montana Fish, Wildlife, and Parks; Andrew Lamont of the USGS; Pat VanEimeran of the USFS; and John Waller and Terry Peterson of GNP for their assistance with redd counts. Mark Biel of the NPS provided constructive review of this report.

LITERATURE CITED

Baxter, C.V. and F.R. Hauer. 2000. Geomorphology, hyporheic exchange and selection of spawning habitat by bull trout (Salvelinus confluentus). Canadian Journal of Fisheries and Aquatic Sciences 57:1470-1481.

Block, D.G. 1953. Trout migration and spawning studies on the North Fork Drainage of the Flathead River. Master's Thesis, Montana State University, Bozeman.

Bonar, S.A., M. Divens, and B. Bolding. 1997. Methods for sampling the distribution and abundance of bull trout and Dolly Varden. Washington Department of Fish and Wildlife. Olympia.

Bonneau, J.L. and D.L. Scarnecchia. 1998. Seasonal and diel changes in habitat use by juvenile bull trout (Salvelinus confluentus) and cutthroat trout (Oncorhynchus clarki) in a mountain stream. Canadian Journal of Zoology 76:783-790.

Daniel, W.W. 1990. Applied nonparametric statistics. PWS-KENT Publishing Company. Boston, Massachusetts.

Deleray, M., L. Knotek, S. Rumsey, and T. Weaver. 1999. Flathead Lake and river system fisheries status report. DJ Report No. F-78-R-1-R-5, SBAS Project No. 3131. Montana Fish, Wildlife, and Parks, Kalispell, Montana.

Donald, D.B. and D.J. Alger. 1993. Geographic distribution, species displacement, and niche overlap for lake trout and bull trout in mountain lakes. Canadian Journal of Zoology 71:238-247.

Downs, C.C., D. Horan, E. Morgan-Harris, and R. Jakubowski. 2006. Spawning demographics and juvenile dispersal of an adfluvial bull trout population in Trestle Creek, Idaho. North American Journal of Fishery Management 26: 190-200.

Downs, C.C. and R. Jakubowski. 2006. Lake Pend Oreille/Clark Fork River fishery research and monitoring 2005 progress report. Report Number 06-41. Prepared for Avista Corporation by the Idaho Department of Fish and Game. Boise.

Downs, C.C., C. Stafford, H. Langner, and C.C. Muhlfeld. 2011. Glacier National Park Fisheries Inventory and Monitoring Bi-Annual Report, 2009-2010. National Park Service, Glacier National Park, West Glacier, Montana.

Dunham, J., B. Rieman, and K. Davis. 2001. Sources and magnitude of sampling error in redd counts for bull trout. North American Journal of Fishery Management 21: 343-352.

Fraley, J.J. and B.B. Shepard. 1989. Life history, ecology, and population status of migratory bull trout (Salvelinus confluentus) in the Flathead Lake and river system, Montana. Northwest Science 63:133-143.

Fredenberg, W. 2002. Further evidence that lake trout displace bull trout in mountain lakes. Intermountain Journal of Sciences 8:143-151.

Fredenberg, W., M. Meeuwig, and C. Guy. 2007. Action Plan to Conserve Bull Trout in Glacier National Park. USFWS Creston Fish and Wildlife Center, Kalispell, MT.

Gregory, J.S., and B. Gamett. 2009. Cattle trampling of simulated bull trout redds. North American Journal of Fisheries Management 29:361-366.

Meeuwig, M., C. Guy, W.A. Fredenberg. 2007. Research Summary for Action Plan to Conserve Bull Trout in Glacier National Park, Montana. Montana State University, Bozeman.

Meeuwig, M.H. 2008. Ecology of lacustrine-adfluvial bull trout populations in an inter-connected system of natural lakes. Ph.D. Dissertation. Montana State University, Bozeman.

Mogen, J.T. and L.R. Kaeding. 2001. Population biology of bull trout (Salvelinus confluentus) in the St. Mary River Drainage. U.S. Fish and Wildlife Service, Bozeman, MT.

Mogen, J.T. and L.R. Kaeding. 2004. Bull trout (Salvelinus confluentus) use of tributaries of the St. Mary River, Montana. U.S. Fish and Wildlife Service, Bozeman, MT.

Mogen, J.T. and L.R. Kaeding. 2005. Large-scale, seasonal movements of radiotagged, adult bull trout in the St. Mary River drainage, Montana and Alberta. Northwest Science 79(4): 246-253.

Montana Bull Trout Scientific Group. 1998. The relationship between land management activities and habitat requirements of bull trout. Montana Fish, Wildlife, and Parks, Helena.

Muhlfeld, C.M., M.L. Taper, D.L. Staples, and B.B. Shepard. 2006. Observer error structure in bull trout redd counts in Montana streams: Implications for inference of true redd count numbers. Transactions of the American Fisheries Society 135: 643-654.

Muhlfeld, C.M., D.H. Bennet, R.K. Steinhorst, B. Marotz, and M. Boyer. 2008. Using bioenergetics modeling to estimate consumption of native juvenile salmonids by non-native northern pike in the upper Flathead River system, Montana. North American Journal of Fisheries Management 28:636-648.

National Park Service. 2006. Management Policies 2006. US Government Printing Office, Washington, DC.

Pratt, K. 1984. Pend Oreille trout and char life history study. Report to the Idaho Department of Fish and Game and the Lake Pend Oreille Idaho Club. Boise, Idaho.

Rieman, B.E. and J.D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. General Technical Report INT-302. USDA Forest Service, Intermountain Research Station. Boise, Idaho.

Rieman, B.E. and D.L. Myers. 1997. Use of redd counts to detect trends in bull trout (Salvelinus confluentus) populations. Conservation Biology 11:1015-1018.

Rieman, B.E. and J.D. McIntyre. 1996. Spatial and temporal variability in bull trout redd counts. North American Journal of Fisheries Management 16:132-141.

Saffel, P.D. and D.L. Scarnecchia. 1995. Habitat use by juvenile bull trout in Belt-series geology watersheds of Northern Idaho. Northwest Science 69: 304-317.

Schill, D.J., R. Thurow, and P. Kline. 1994. Wild trout evaluations, Job 2. Seasonal movement and spawning mortality of fluvial bull trout in Rapid River, Idaho. Job Performance Report IDFG 94-13 for Sport Fish Restoration. Idaho Department of Fish and Game, Boise.

Tennant, L.B. 2010. Spawning and early life-history characteristics of bull trout in a headwater lake ecosystem. MS Thesis. Montana State University, Bozeman.

Thurow, R.F. 1997. Habitat utilization and diel behavior of juvenile bull trout (Salvelinus confluentus) at the onset of winter. Ecology of Freshwater Fish 6:1-7.

U.S. Fish and Wildlife Service. 2002. Chapter 25, St. Mary-Belly River Recovery Unit, Montana. 134p. *In*: U.S. Fish and Wildlife Service Bull Trout (Salvelinus confluentus) Draft Recovery Plan. Portland, Oregon.

Weaver, T., M. Deleray, and S. Rumsey. 2006. Flathead Lake and River System Fisheries Status Report. DJ Report No. F-113-R-1-R-4, SBAS Project No. 3130, Montana Fish, Wildlife, and Parks, Kalispell, Montana.

APPENDIX A

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
Hudson Bay Drainage																	
Boulder Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Kennedy Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Lee Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Rose Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
N. Fk. Flathead Drainage																	
Akokala Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Agassi Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Bowman Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Harrison Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Jefferson Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Logging Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (lower)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (middle)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Quartz Cr. (upper)	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Rainbow Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Upper Kintla outlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	52	--	--
Upper Kintla inlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Trout Lake inlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Arrow Lake inlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
M. Fk. Flathead Drainage																	
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	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
								f	b,f		a,c	c,d,e	g	h				
Hudson Bay Drainage																		
Boulder Cr.	12	42	20	30	28	28	28	27	--	50	38	58	38	33	61	53	61	66
Kennedy Cr.	23	37	--	23	12	11	18	27	25	20	13	22	4	12	15	9	8	20
Lee Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	15	31	12	16
Rose Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	0	--
N. Fk. Flathead Drainage																		
Akokala Cr.	--	--	--	--	--	--	--	--	--	--	--	11	6	1	4	5	--	12
Agassi Cr.	--	--	--	--	--	--	--	--	--	--	--	0	--	--	--	--	--	--
Bowman Cr.	--	--	--	--	--	0	0	0	0	2	1	0	0	1	3	--	1	4
Jefferson Cr.	--	--	--	--	--	--	--	--	--	--	--	0	0	--	--	--	--	0
Logging Cr.	--	--	--	--	--	--	--	3	20	0	--	5	0	3	3	1	27	0
Quartz Cr. (lower)	--	--	--	--	--	--	--	1	3	2	2	3	2	2	5	3	5	4
Quartz Cr. (middle)	--	--	--	--	--	--	0	0	0	0	0	0	3	0	--	--	--	1
Quartz Cr. (upper)	--	--	--	--	--	--	31	46	4	36	14	51	34	27	32	33	22	66
Rainbow Cr.	--	--	--	--	--	--	--	--	--	--	--	28	12	4	9	--	4	5
Starvation Cr.	0	--	--	0	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Upper Kintla outlet	--	--	--	--	--	--	--	--	--	--	--	0	--	25	--	--	--	--
Upper Kintla inlet	--	--	--	--	--	--	--	--	--	--	--	0	--	--	--	--	--	--
Trout Lake inlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	8
Arrow Lake inlet	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	0
M. Fk. Flathead Drainage																		
Ole Cr.	14	22	26	33	29	21	21	14	16	31	29	42	34	32	40	53	44	32
Nyack Cr.	9	--	--	13	--	--	14	--	--	--	--	16	8	6	16	14	--	11
Park Cr.	2	--	--	10	--	--	0	--	--	--	--	23	3	3	20	15	9	4
Harrison Cr.	--	--	--	--	--	--	--	4	0	8	15	14	1	6	--	1	--	10

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

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

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

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

e = count conducted by helicopter on Park.

f = minimum count on Ole as high flows may have obliterated some redds.
g = Kennedy count does not include three additional redds counted upstream of the index section, or two redds counted in unnamed tributary flowing from Yellow Mountain.
h=very poor visibility in Bowman Creek due to glacial runoff from Jefferson Creek