Glacier National Park Fisheries Monitoring and Management

Program Report 2015

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Front cover photo caption: Construction of Akokala Creek fish barrier.

Inside photo captions (top to bottom): construction of Akokala barrier, trend netting Bowman Lake, seining Moran's Bathtub and snorkel survey in classic west side creek.

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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 2015, we removed 570 lake trout. Both adult and juvenile lake trout gill net catches declined in 2015, 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. Bull trout redd counts remained stable at 39, which remains above the ten year average of 30 redds. 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 83m (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.

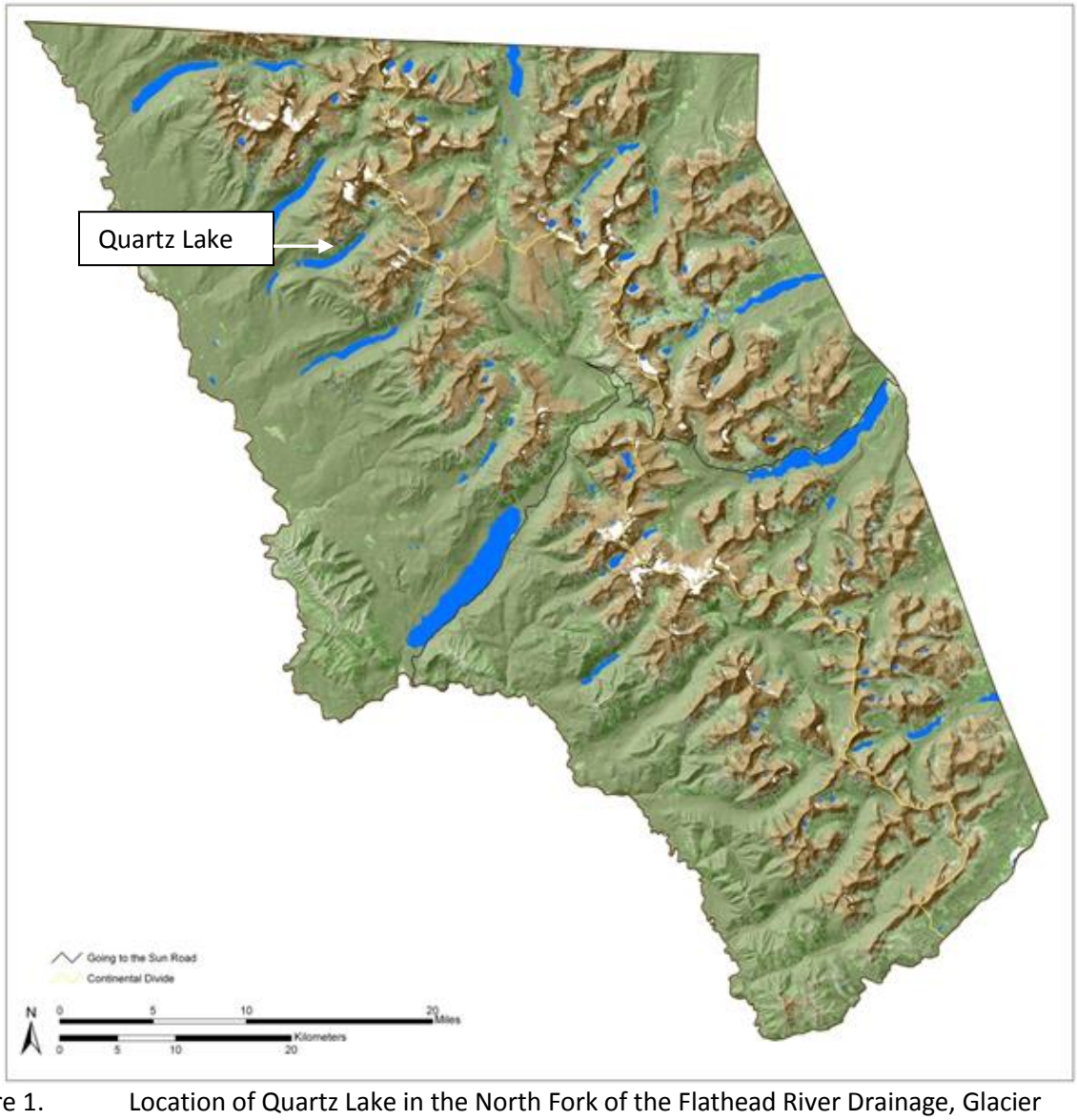

Figure 1. Location of Quartz Lake in the North Fork of the Flathead River Drainage, Glacier National Park.

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 2015 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 2015 lake trout suppression project followed the same general schedule conducted in 2010-2014. Gillnetting was comprised of two specific netting periods; the spring (25 May-26 June) and fall (5 October-26 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-long 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 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). However, the durability of the 0.10 mm nets was problematic and the cost of replacing nets would be unreasonable, therefore, beginning in 2014, full length (91 m) nets of 19 mm, 26 mm and 32 mm bar measure mesh and 0.15 mm twine diameters were 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, primarily on the two known spawning areas as determined by prior radio-telemetry efforts (2009-2014). 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. Fall netting in 2015 occurred from 5 October-26 October and was nearly continuous. Adult lake trout were captured with large mesh (57 mm, 64 mm, and 70 mm bar measure), 91 m-long monofilament gill-nets.

RESULTS AND DISCUSSION

Telemetry

One sonic tag was surgically implanted in a presumably mature lake trout, caught by experienced lake trout anglers during spring 2015. The tagged lake trout was not recaptured because it died shortly following tagging. As the suppression years have progressed from 2010-2015, 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, until 2015 (Figure 2). The decreased number of bull trout caught in 2015 during angling efforts may be in part due to the early warming of the lake. The spring of 2015 was unusually warm and could potentially have biased both lake trout and bull trout catch per unit effort, as fish often will migrate to cooler waters as the surface temperature warms.

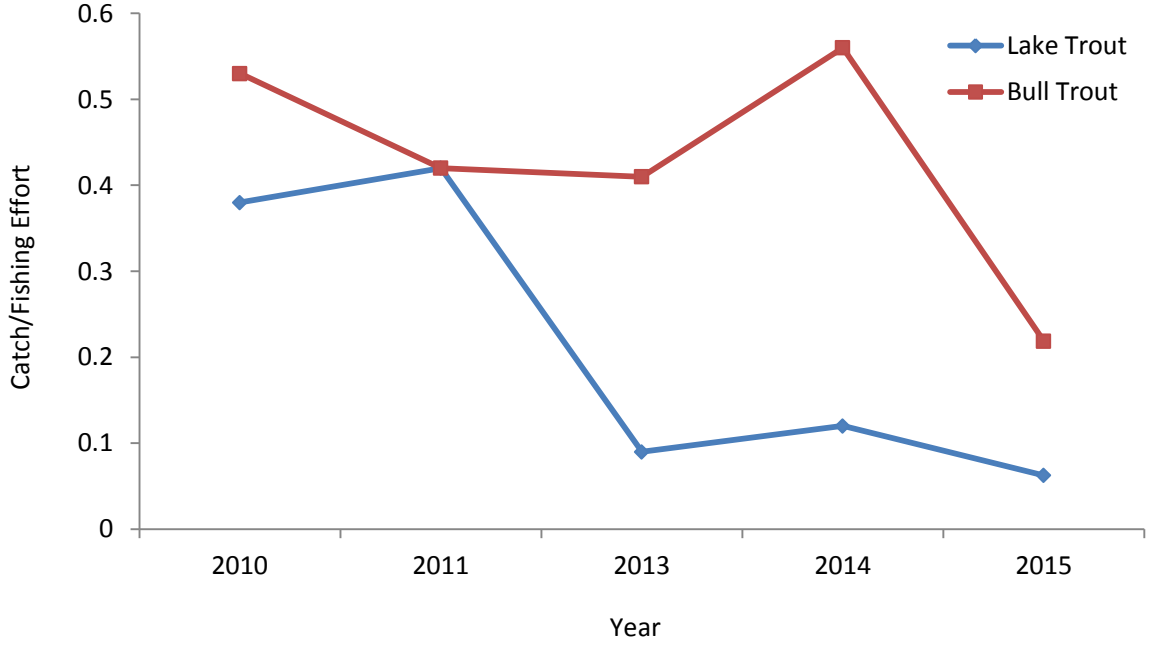

Figure 2. Catch per angler hour for lake trout and bull trout in Quartz Lake, Montana 2010-2015.

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 may be a feeding area.

Gillnetting

Approximately 28.8 km of nets were deployed in 2015, including 16.9 km during the spring and 11.9 km during the fall, resulting in a total of 570 lake trout being removed (554 juveniles and subadults, 16 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 2015 crews deployed the lowest amount of net;

fall netting produced the second lowest 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 0.15 mm twine diameter 19 mm bar measure nets. In 2015, crews found it increasingly difficult to capture adult lake trout while avoiding native salmonids. Due to this difficulty, deployment of spawner nets fell sharply in an attempt to avoid bycatch mortality of westslope cutthroat trout and bull trout (Table 1). Although this is a promising indication that lake trout numbers continue to decline, it increases the difficulty of targeting primarily lake trout.

Table 1. Annual kilometers of net set during spring and fall gillnetting from 2009-2105 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
2015	16.9	11.9	28.8

Spring

Spring netting removed a total of 551 juvenile lake trout. Spring gillnetting saw a decrease in the number of juveniles removed. The decrease in juvenile catch is promising, it may indicate that large proportions of these year classes were removed in 2013 and 2014 (Figure 3). Juvenile length at capture has largely decreased from 2010-2015 suggesting large proportions of the juvenile fish targeted with the 26 mm and 32 mm bar measure nets have been removed Of the juvenile lake trout caught in 2015, the 19 mm bar mesh nets captured approximately 71% (N = 394) while the 26 mm and 32 mm bar measure nets caught 24% (N = 132), and 5% (N = 25), respectively.

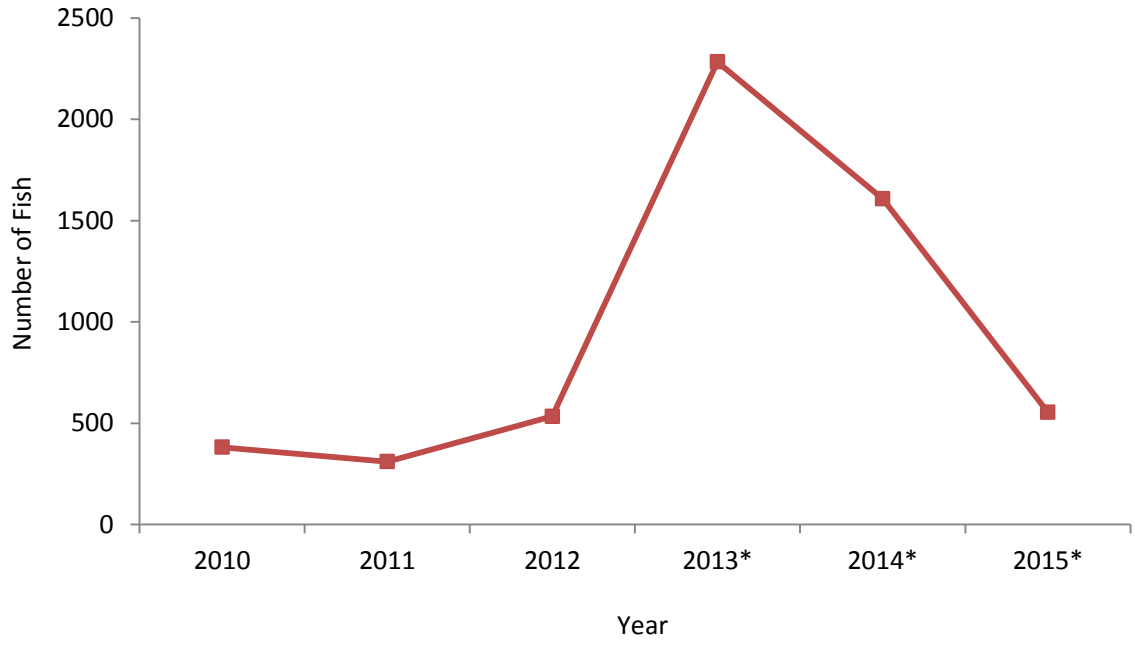

Figure 3. Number of lake trout caught annually (2010-2015) during spring netting in Quartz Lake, Montana. Net effectiveness improved starting in 2013 with the use of gill-nets containing 0.10 mm twine diameter and 19 mm bar measure. In 2015 all spring gill nets were changed to twine diameters of 0.15 mm.

Fall

Fall netting removed a total of 19 lake trout, including 3 immature and 16 mature lake trout. Results from fall 2015 were promising. Since the projects inception in 2009, the number of mature lake trout captured during fall netting has decreased by approximately 89% hitting an all-time low in 2015 (Figure 4). Additionally, the adult length distribution has decreased from 2009. 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. Truncation of length and age distributions in fish populations is a common response to intense size selective harvest (Coleman et al. 2000).

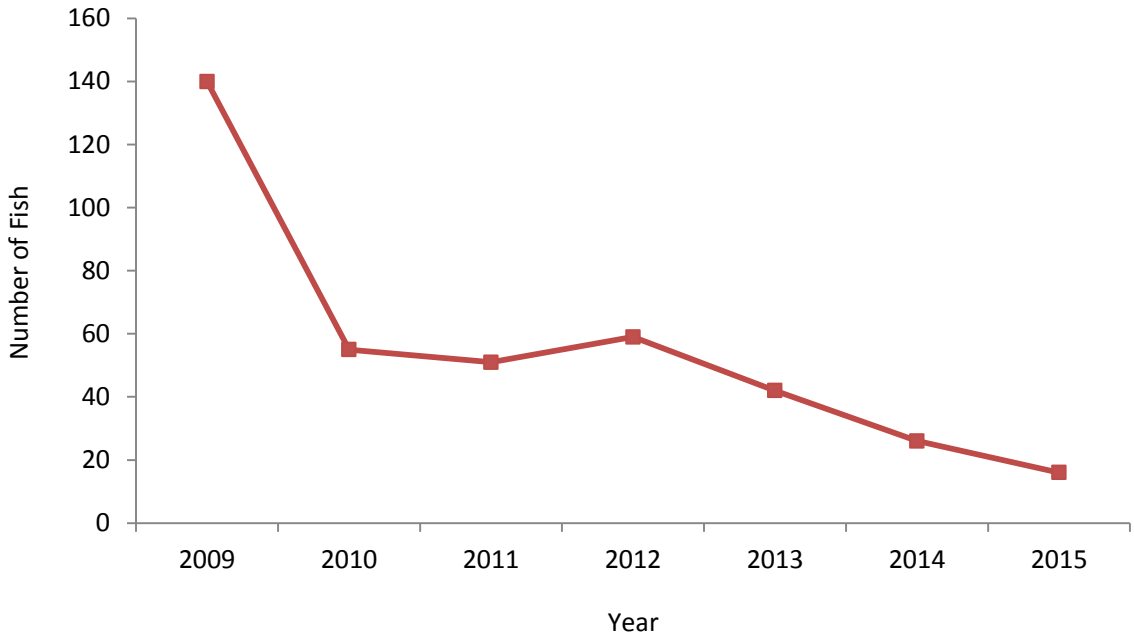

Figure 4. Number of mature lake trout caught during fall gillnetting (2009-2015) in Quartz Lake, Montana.

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 affected the adult bull trout population. Redd count analysis from 2003-2013 showed no significant trend, suggesting the adult bull trout population in Quartz Lake remains stable (Fredenberg 2014). Additionally, the redd count from 2014 resulted in a historic high (*N* = 66) and in 2015 the count remained above the ten year average of 30 redds (*N* = 39), further corroborating the results of the redd analysis (Figure 5).

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 2011-2015. 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 were 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 20 bull trout were captured via gillnetting in 2015, 16 (80%) of which were captured during spring and 4 (20%) during fall. Of the 20 bull trout captured, 11 (55%) were classified as adults (≥ 400mm) and 9 (45%) were juveniles or subadults. Additionally 21 adult sized bull trout were captured during spring angling. Two bull trout were incidentally killed in gillnetting efforts throughout 2015 (10% of total bull trout captured), including 1 adult (9% of captured adults) and 1 juvenile (11% 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 2015 was the lowest observed since the projects inception (Table 2).

Bycatch of non-target species was similar to the previous five years (Table 3). Non-target bycatch in 2015 included: longnose suckers (*N* = 158, length range, 145-615 mm), largescale suckers (*N* = 53, length range 140-650 mm), mountain whitefish (*N* = 690, length range 110-410 mm), and westslope cutthroat trout (*N* = 25, length range 240-370 mm). Bycatch mortality for these species were as follows: two longnose suckers (1% of total), one largescale suckers (2% of total), >200 mountain whitefish (> 33% of total), and seven westslope cutthroat trout (28% of total). The high bycatch and bycatch mortality of mountain whitefish in 2015 when compared to previous years can most likely be explained by the incorporation of fine-diameter 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 2009-2015.

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)
2015	9(11)	1(1)

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 2009-2015 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)
2015	16(4)	4(22)	690(1)	49(4)	153(5)

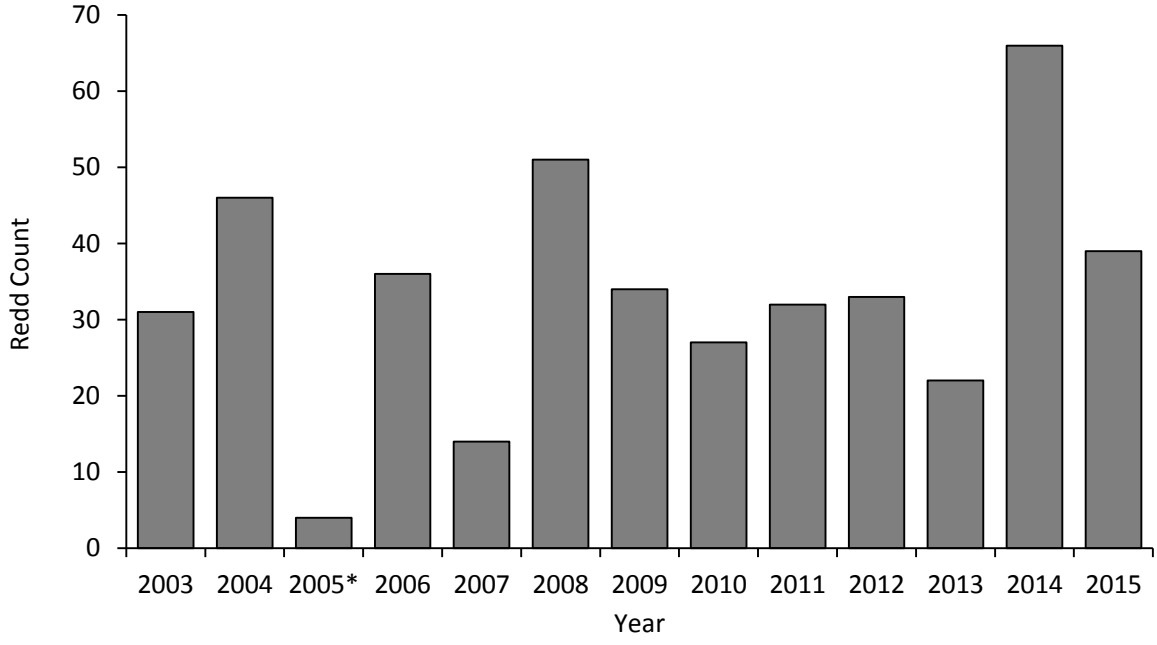

Figure 5. Bull trout redd counts from 2003 through 2015 in Quartz Creek upstream of Quartz Lake, Glacier National Park, Montana. *Redds were obscured by high flows in 2005 (Meeuwig and Guy 2007).

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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 translocated 111 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. In 2015, NPS and USGS staff attempted to translocate additional juvenile bull trout, however, after sampling Logging Creek from 2 September-3 September, 2015 they only captured one additional bull trout which was PIT tagged and translocated successfully. The lack of spawning adult bull trout from 2014 and the decline in juvenile bull trout sampled in 2015 further exemplifies the urgency and importance of this translocation project to conserve the imperiled Logging Lake 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 affected 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 in 2014 (Downs et al. 2015), the project proceeded 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 2015 translocation field season. For additional information pertaining to the gillnet suppression activities, please refer to the USGS' 2015 annual report (D'Angelo et al. 2015)

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.

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 piscivorous salmonid in Logging Lake and was first detected in 1984 (Fredenberg et al. 2007). No native salmonids were present in Grace Lake prior to the introduction of bull trout.

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

METHODS

Electrofishing

In 2015, we attempted to translocate 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, "spot shocking" was conducted to sample the best habitat within the reach.

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-loop PIT-tag antenna was fixed to the stream 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.

RESULTS AND DISCUSSION

The translocation effort was deemed highly successful following the translocation of 111 juvenile bull trout in 2014. The success of the translocation in 2014 was followed by the successful translocation of one individual in 2015. The lack of redds and adult bull trout observed in 2014 and the lack of juveniles within the sampling reach in 2015 further substantiates GNP's stance on the immediate need of the project to occur to preserve bull trout within the Logging Lake chain. 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 gillnetting and redd count data (Fredenberg 2002; Meeuwig et al. 2010; Downs et al. 2014) 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 2 September and ended on 3 September. We captured one juvenile bull trout using electrofishing and translocated it from Logging Creek upstream into Grace Lake. There were no juvenile bull trout mortalities during the electrofishing capture efforts. Lengths of translocated bull trout from 2014-2015 have varied from 52-200 mm (mean 67 mm; 95% CI 63-71 mm) (Figure 2).

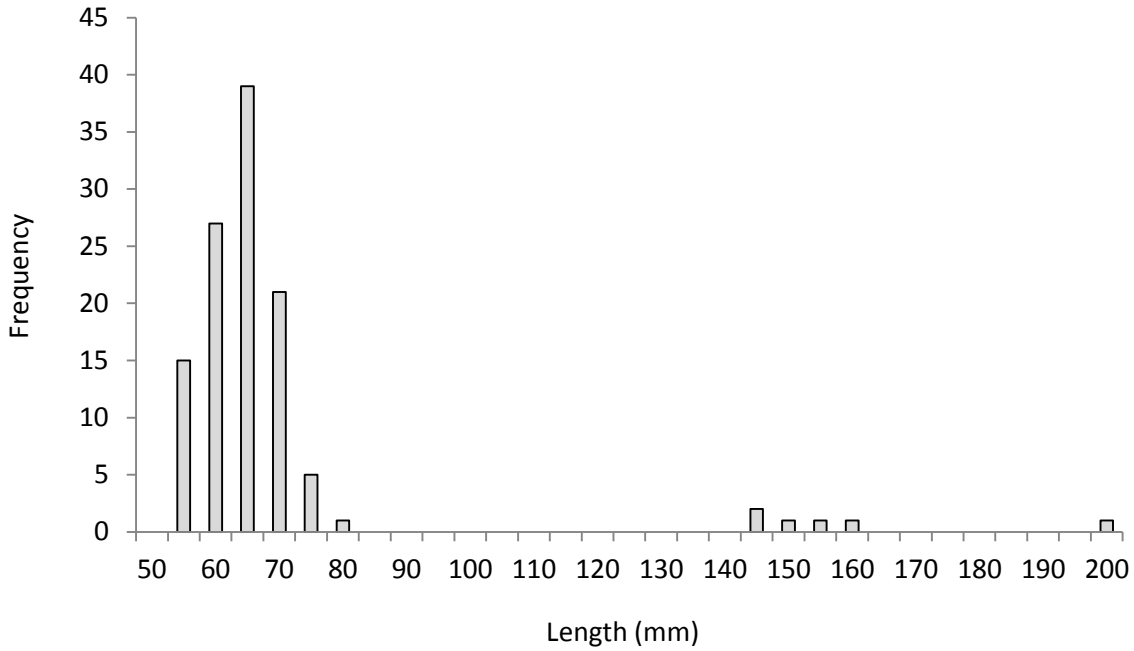

Figure 2. Length frequency of bull trout translocated from Logging Creek to Grace Lake in 2014, GNP, Montana.

One additional bull trout was PIT-tagged and released into the Grace Lake inlet stream in 2015 bringing the total number of PIT-tagged juveniles to seven. The PIT-tagged fish vary in length from 132-200 mm in length (Table 1). The PIT-tag antenna recorded a total of three of the seven PIT-tagged bull trout moving to the outlet of Grace Lake. In September 2014, bull trout number 597 was detected in the vicinity of the Grace Lake outlet but appeared to remain in the lake. In 2015, from 5 June through 11 June bull trout number 056 was detected at the outlet of Grace Lake and potentially left the system. In addition, on 28 September, 2015 bull trout number 79061 was also detected in the Grace Lake outlet, but the fish appeared to remain in the lake.

Table 1. 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 from 2014-2015.

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
9/2/2015	20239	132

ACKNOWLEDGEMENTS

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Akokala Fish Passage Barrier Construction

ABSTRACT

Over the past 30 years, non-native lake trout have replaced native bull trout as the dominant predator in most of the large lakes on the west side of Glacier National Park (GNP). There are 17 lake dwelling populations of bull trout identified on the west side of GNP. Nine of these lakes have been compromised by lake trout, one has non-native brook trout, and two (Akokala and Cerulean) are at-risk of invasion by lake trout. The Akokala Lake bull trout population is significantly differentiated from other bull trout populations on the west side of GNP, and the history of lake trout invasion in GNP suggested that the installation of a fish passage barrier in Akokala Creek downstream of the outlet of Akokala Lake would be a prudent step to protect this lake from invasion by exotic species. In addition, Akokala Lake also supports genetically pure westslope cutthroat trout which are threatened by hybridization from downstream areas. Although relatively small, the uncompromised status of the bull trout population in Akokala Lake represented a unique opportunity for managers to isolate the population prior to lake trout colonization and subsequent negative impacts. However, GNP is managed as a wilderness and Akokala Lake is situated six and a half miles in the backcountry, creating unique obstacles in the building of a barrier. In 2015, the NPS began constructing a fish passage barrier approximately 200 m downstream of the Akokala outlet. The construction of a fish barrier of this magnitude had never been attempted within the backcountry of GNP. The barrier was designed and built to withstand a twenty five to fifty year flood event. The materials used in construction were primarily taken from the surrounding area and all labor (with exception of chainsaws and a wood drill) were non-mechanized, manual labor. The purpose of the barrier was to remove the imminent threat that lake and rainbow trout hybrids posed to this small isolated population of native bull and westslope cutthroat trout within Akokala Lake.

INTRODUCTION

Over the past 30 years, non-native lake trout have replaced native bull trout as the dominant predator in most of the large lakes on the west side of GNP. The decline of bull trout in GNP is directly attributed to the invasion and establishment of nonnative lake trout, which consistently displace bull trout in systems where introduced (Donald and Alger 1993, Fredenberg 2002). There are 17 lake dwelling populations of bull trout identified on the west side of GNP. Nine of these lakes have been compromised by lake trout, one by non-native brook trout, and two (Akokala and Cerulean) are at-risk of invasion by lake trout.

Genetic evidence suggests that the Akokala Lake bull trout population is significantly differentiated from other bull trout populations on the west side of GNP (Meeuwig 2008), and the history of lake trout invasion in GNP (Fredenberg 2002) suggests the installation of a fish passage barrier in Akokala Creek downstream of the outlet of Akokala Lake would be a prudent step to protect this lake from invasion by exotic species. Additionally, rainbow trout/hybrids are emerging as a significant threat to native westslope cutthroat trout in these same systems. Rainbow trout appear to be invading the N. Fk. Flathead River drainage in a stepping stone invasion pattern (Hitt et al. 2003, Boyer et al. 2008), and an adult rainbow-westslope cutthroat trout hybrid was recently identified (and likely spawned) in the lower reaches of Akokala Creek (Muhlfeld et al. 2012). Juvenile westslope cutthroat-rainbow trout hybrids were recently detected in lower Akokala Creek. Akokala Lake, located in the headwaters of the Akokala Creek drainage, appears to support a healthy population of genetically pure westslope cutthroat trout, with adequate spawning habitat provided by the inlet stream. Bull trout redd counts conducted by GNP staff identified a spawning reach for bull trout located in the inlet stream, and it is likely this same reach would also be used by westslope cutthroat trout for spawning. It is clear that the threat of colonization of Akokala Lake by both lake and rainbow trout exists, and specific to rainbow trout colonization, the threat appears imminent.

Currently, there are multiple large-scale lake trout suppression projects underway across the western U.S., including efforts in Glacier by GNP and the USGS on Quartz and Logging lakes, Yellowstone National Park on Yellowstone Lake, the Idaho Department of Fish and Game on Lake Pend Oreille and Upper Priest Lake, and the Montana Department of Fish, Wildlife, and Parks (MFWP) on Swan Lake. Each of these efforts is aimed at conserving bull trout or other native fish through lake trout removal, and highlights the seriousness of the lake trout invasion problem across the west.

Unless additional conservation measures were implemented, lake trout would likely continue to expand their abundance and distribution in GNP, further reducing the distribution and abundance of bull trout and other native fish in GNP. Based on cursory habitat surveys, habitat suitability for lake trout in Akokala lake appears low. However, given the expense of removing lake trout and uncertain outcome of such efforts, the substantially reduced number of secure lakes supporting bull and westslope cutthroat trout, and the threat posed by rainbow trout hybrids, the most conservative management approach for native species conservation in the system is to construct a fish passage barrier to prevent non-native fish from entering the system. Due to its small size (8.9 ha) and shallow maximum depth (7 m), even colonization and maintenance of a lake trout population solely by emigration could potentially cause harm to the small migratory bull (likely less than 50 reproducing adults) and westslope cutthroat populations in the system. Lake trout can live considerably longer than either bull or westslope cutthroat trout (Schram and Fabrizio 1998, Downs et al. 1998, Downs et al. 2006) and long-term predation by lake trout even with only periodic recruitment could have adverse impacts to these small populations.

In addition to threats to bull trout posed by lake trout invasion, the Akokala Lake system may lend itself well to future invasion by brook trout *(S. fontinalis)* (Fredenberg 2002). Brook trout can hybridize with bull trout (Kanda et al. 2002), and are hypothesized to out-compete cutthroat trout at young ages in stream systems (Peterson et al. 2004). Brook trout are not known to currently inhabit the N. Fk. Flathead drainage, but are present in the M. Fk. Flathead River drainage further downstream. Low gradient, broad valleys, similar to the lower reaches of Akokala Creek, appear to offer habitat suitable for brook trout population establishment. In the long-term, a barrier would also serve to keep the system secure from brook trout invasion as well.

Due to the imminent risk of either hybridization or non-native invasion, the National Park Service began building a fish passage barrier at the Akokala Lake outlet in 2015. Prior to constructing the barrier GNP staff conducted lake live trapping to assess the natural fish assemblage within the lake. Beginning in early July, fisheries crews set nine trap nets to live capture fish species within the lake. Immediately following the sampling, the construction of the fish passage barrier began.

Study Area

Akokala Creek is a third order tributary to the North Fork of the Flathead River (Figure 1). The lower reaches of Akokala Creek are known to support migratory westslope cutthroat trout spawning (Muhlfeld et al. 2012), while the upper reaches of the drainage support a "disjunct" migratory population 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 et al. 2012). Hybridization between rainbow and westslope cutthroat trout was recently detected in a sample of juvenile fish collected from 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.

Akokala Lake is relatively small (8.9 ha) and shallow (7.0 m) when compared to other lakes supporting populations of bull trout within GNP. Although one of the smaller and shallower lakes, bull trout densities in Akokala Lake are relatively high (Fredenberg et al. 2007). The native fish assemblage in Akokala Lake is comprised of wct, blt, and mwf.

Figure 1. Location of Akokala Lake in GNP.

METHODS AND RESULTS

Netting

We used custom-made miniature fyke nets to sample Akokala Lake, located in the North Fork of the Flathead River drainage (Figure 1). Fyke nets were constructed of 12 mm (bar measure) brown nylon mesh. Each net measured 3m in length and had leads 7 m long X 0.66 m tall, with a lead core 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 m 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. Fyke nets were 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 0.66 m), 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. Overnight sets were used, nets were typically set in late afternoon and retrieved early the following morning. Three nets were set per night for three consecutive nights for a total of nine sampling events (Table 1; Figure2).

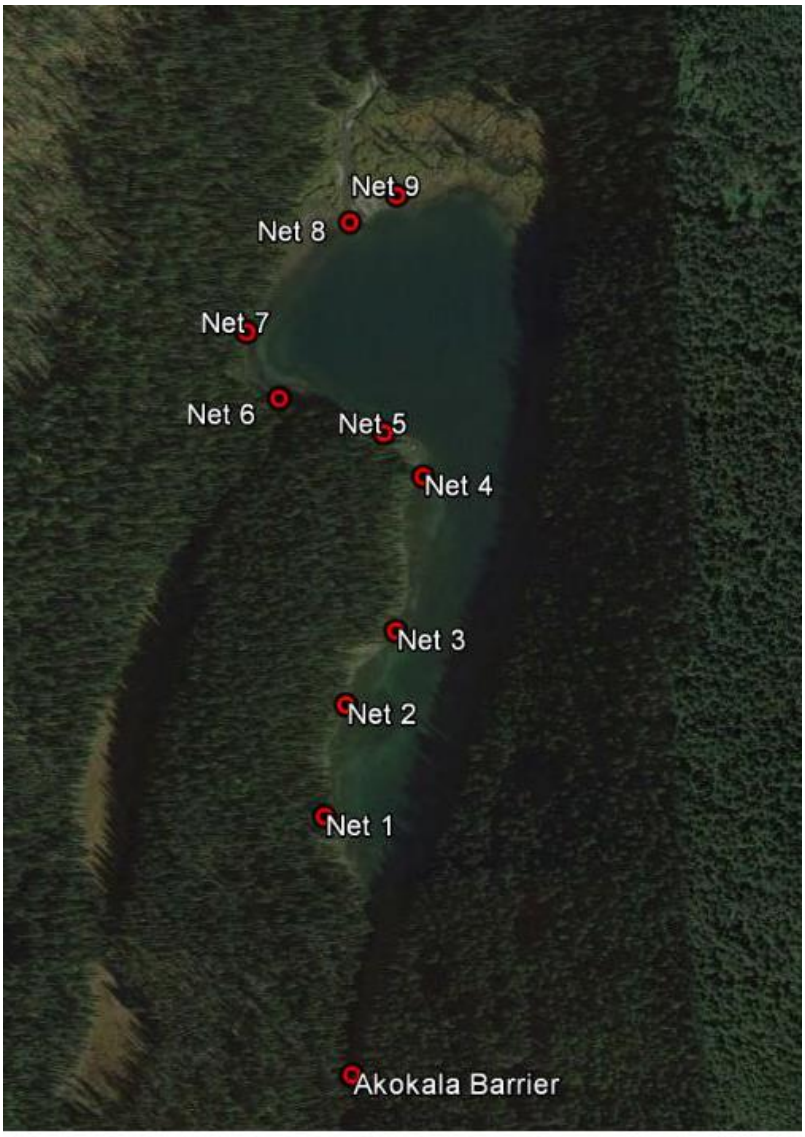

Figure 2. Location of trap net sets and barrier placement on Akokala Lake 2015.

We captured eleven bull trout, thirty four westslope cutthroat trout and fifty nine mountain whitefish during trap netting efforts. This resulted in a CPUE of 1.22 bull and 6.56 westslope cutthroat trout/net night and 3.78 mountain whitefish/net night. Length at capture for bull and westslope cutthroat trout ranged from 137 mm to 764 mm (mean 256.4 mm) and 127 mm to 241 mm (mean 165.76 mm) respectively (Figures 3 and 4).

Table 1. UTM's and set parameters of Akokala 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	11U 0705297	5417487	6/29/15	18:47	6/30/15	9:06	4
Fyke Net 2	11U 0705317	5417581	6/29/15	19:16	6/30/15	9:30	4
Fyke Net 3	11U 0705361	5417643	6/29/15	19:32	6/30/15	10:00	4
Fyke Net 4	11U 0705388	5417774	6/30/15	19:02	7/1/15	8:40	4
Fyke Net 5	11U 0705355	5417813	6/30/15	19:14	7/1/15	9:00	4
Fyke Net 6	11U 0705266	5417844	6/30/15	19:24	7/1/15	9:15	4
Fyke Net 7	11U 0705240	5417901	7/1/15	18:41	7/2/15	11:00	4
Fyke Net 8	11U 0705330	5417996	7/1/15	19:08	7/2/15	11:30	4
Fyke Net 9	11U 0705371	5418019	7/1/15	19:27	7/2/15	10:24	4

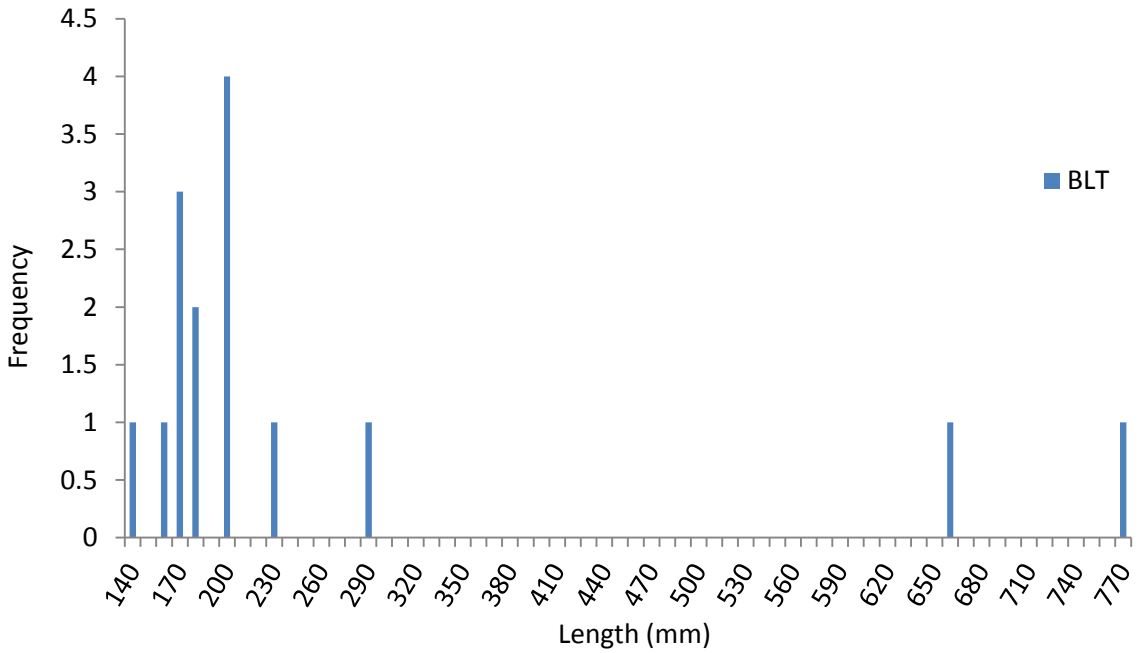

Figure 3. Length frequency of BLT caught in miniature Fyke nets in Akokala Lake from 6/30-7/2/15 in GNP, Montana.

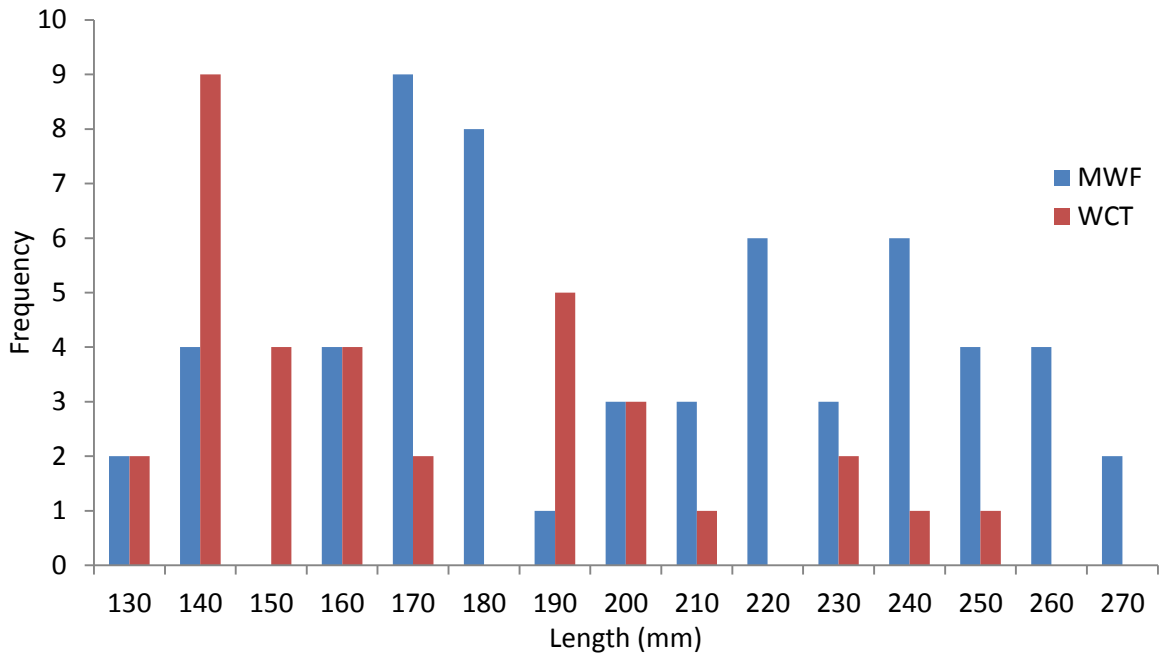

Figure 4. Length frequency of WCT and MWF caught in miniature Fyke nets in Akokala Lake from 6/30-7/2/15 in GNP, Montana.

Barrier Construction

National Park Service and USGS staff collaboratively identified several potential locations for the construction of a fish passage barrier. The best site to construct the fish passage barrier was at the outlet of Akokala Lake. The site was selected because it offered a constricted channel, steep gradient, large boulders at the channel margins for structural support, good access, and a sufficient supply of large rock and trees for construction. Other sites evaluated located further downstream in the drainage afforded more habitat protection, but posed high-risk from a barrier constructability, durability, and long-term effectiveness perspective. Engineering plans were constructed based on the 100 year flood event. Although the engineered plans were based on the 100 year flood event, the backcountry location, short field season and management of the surrounding area as wilderness limited the construction of the barrier to withstand the twenty five to fifty year flood events.

Following the site identification and plan drafting by a private contractor (DJ & A, Missoula, Montana) the barrier was constructed of harvested timber and native cobble-sized rock located in the surrounding area of the barrier. The construction phase of the project was undertaken by both NPS trail and fish crews. A supervisor familiar with the plans and knowledge needed to direct construction was on site during most of the construction phase. Mechanized tools were used sparingly in an attempt to maintain the wilderness experience for visitors in the area. Most tools were battery operated with the exception of chainsaws and a wood drill. Trees were felled by certified sawyers, stripped of bark and moved into place via highline ropes and pully's. Four “cribs” were built behind the barrier face, and filled with cobble and boulder substrates to increase stability and rigidity. The barrier was constructed as a spillover, allowing the water to pool on the upstream side of the barrier (Figures 5 and 6). The middle portion of the face is lower than the rest of the structure, allowing the pooled water to flow over the top of the barrier onto gabions filled with cobble, creating a waterfall with a splash pad to deter erosion and prevent the formation of a jump pool for fish. The drop over the structure at base-flow is

approximately 6'. Four trail crew personnel were used for five hitches (each hitch was six days of work and two days of travel) and the fisheries crew worked an additional twenty days including ten travel days where only partial work could be conducted in the remaining hours.

Future management will consist of a multipronged approach. First, NPS staff will assess the barrier during August of each year to determine whether any repairs need to be conducted prior to the next year's high water. Second, the NPS will conduct genetic analysis on bull and westslope cutthroat trout caught within the lake using trap nets and electroshocking to determine whether the genetic status of the populations remain stable and to test whether translocation of new individuals is needed to maintain genetic diversity. Last, in 2015 and every other year thereafter, NPS staff will shock approximately one mile of stream habitat below the barrier site, any bull trout caught will be translocated above the barrier to ensure the loss of bull trout from emigration will not be significant.

Figure 5. Photo of completed fish passage barrier on Akokala Creek, Glacier National Park, Montana.

Figure 6. Photo of completed fish passage barrier on Akokala Creek, Glacier National Park, Montana.

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Native Fish Population Monitoring

ABSTRACT

In 2009, Glacier National Park (GNP) began development of a monitoring program for native salmonids inhabiting park streams. The intent of the program is to establish baseline abundance levels in a variety of park waters to serve as useful benchmarks for monitoring changes in populations over time. We continued this work through 2015, sampling in seven waterbodies. We conducted snorkel surveys on Mineral (Middle Fork Flathead drainage) and Upper McDonald (North Fork Flathead drainage) creeks, trap netting in Akokala and Upper Kintla lakes (North Fork Flathead drainage) and trap netting and seining in Swiftcurrent Ridge 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 species, such as rainbow, brook, and lake trout. However, management and conservation of native fish in the North and Middle forks of the Flathead River remains complicated due to the presence of migratory and resident populations of native salmonids, and expanding distribution of invasive 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 approximately 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 perennial 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 continental scale watersheds (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, while the northeastern portions of the park drain into both the Arctic and the North Atlantic oceans 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 decades ago 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 2015 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 (ebt) *Salvelinus fontinalis*	I	I	I
Bull trout (blt) *S. confluentus*	N	--	N
Kokanee *Oncorhynchus nerka*	I	--	--
Lake trout *S. namaycush*	I	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 (lns) *Catostomus catostomus*	N	N	N
Largescale sucker (lss) *C. macrocheilus*	N	--	--
White sucker (wsu) *C. commersoni*	--	N	N
Deepwater sculpin *Myoxocephalus thomsoni*	--	--	N
Mottled sculpin *Cottus bairdi*	--	N	N
Slimy sculpin *C. cognatus*	N	--	--
Shorthead sculpin *C. confuses*	N	--	--
Spoonhead sculpin *C. ricei*	--	--	N
Burbot *Lota lota*	--	--	N
Northern pike *Esox Lucius*	--	--	N
Trout-perch *Percopsis omiscomaycus*	--	--	N
Lake Chub *Couesius plumbeus*	--	N	N

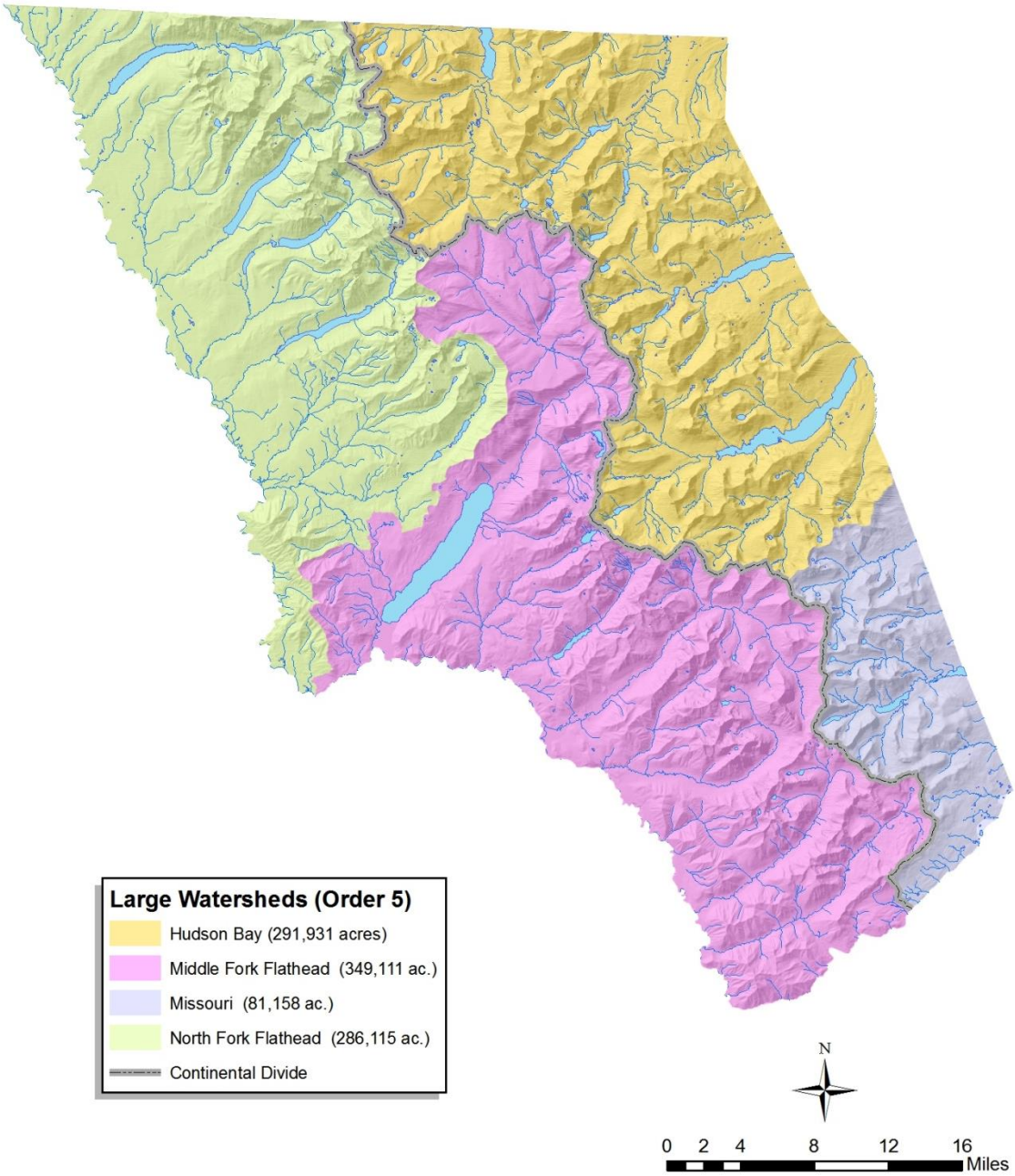

Figure 1. Major watersheds of Glacier National Park, Montana.

METHODS

Netting

We used custom-made miniature fyke nets to sample Akokala and Upper Kintla lakes, located in the North Fork drainage and Swiftcurrent Ridge Lake in the Saint Mary drainage (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 m 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 also used a seine to sample Swiftcurrent Ridge Lake located in the Hudson River drainage.

Fyke nets were 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 0.66 m), 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.

Our seine measured 10’ x 100’ and was constructed of white ¼” nylon mesh. ten Seines were generally pulled through a maximum depth of in 5’. The seine was set parallel to shore, spanning a bay from one shore to the opposite shore. While ensuring the bottom of the seine remained on the lake floor, personnel simultaneously pulled the ends of the seine in toward the shore until the seine was shallow enough to retrieve the captured fish.

Snorkeling

In addition to seining 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) $\geq$60 mm and westslope cutthroat trout (wct) $\geq$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.

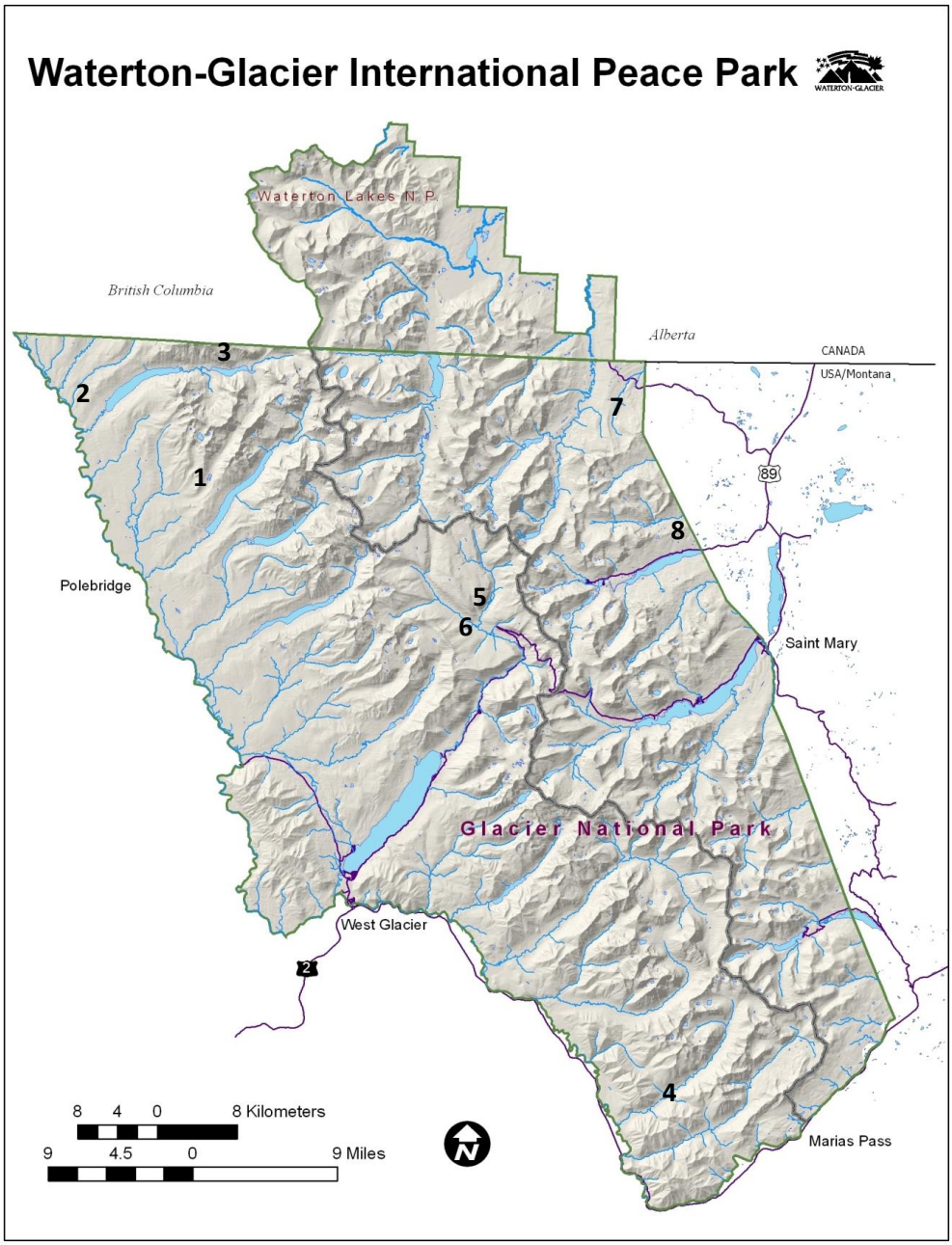

Figure 2. Stream and lake sampling sites for temperature monitoring, snorkeling, and trapping efforts in GNP, Montana 2015. Site numbers from Table 3.

For 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 20m 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. 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 2015.

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 Lake (1)	Mini Fyke Nets	Multiple sites	7/1/- 7/2/15	
	Starvation Cr (2)	Thermograph only	11U 0690925N, 5423640W	2015	
	Upper Kintla Lake (3)	Mini fyke nets	Multiple sites	7/28-7/29/15	
Middle Fork Flathead	Ole Cr (4)	Thermograph only	12U 0307191N, 5350741W	2015	
	Mineral Cr (5)	Snorkel	12U 0303790N292587, 5360495W5404462W	8/21/2013; 8/13/14; 8/11/15	377
	Upper McDonald Creek (6)	Snorkel	12U 292335N, 5403907W	8/20/13; 8/12/15	329
Saint Mary (Hudson Bay)	Lee Cr (7)	Thermograph only	12U 0307481N, 5428586W	2009; 2011; 2012; 2013; 2014; 2015	
	Swiftcurrent Ridge Lake (8)	Mini fyke nets	Multiple sites	7/21-7/22/15	

RESULTS AND DISCUSSION

North Fork Flathead River Drainage

Akokala Lake

Akokala Lake is located in the North Fork of the Flathead River Drainage. 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). 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 juvenile cutthroat trout 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. 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.

Miniature fyke nets were used to live trap mountain whitefish, bull and westslope cutthroat trout in Akokala Lake in 2015.

Middle Fork Flathead River Drainage

Mineral Creek

Mineral Creek is a second order tributary to McDonald Creek in the McDonald Lake drainage (Figure 2). The downstream end of the sampling site is located at the confluence of Mineral Creek and upper McDonald Creek.

Based on the size structure of the population and the location of the sampling site in the drainage it is likely this population is resident. Mineral Creek contains genetically pure wct based on samples collected from Mineral Creek in 2008.

In 2015 wct were the only species of fish observed in this reach of Mineral Creek. Fish in all size groups were observed (Table 4). One hundred and fifty three wct were observed by snorkelers in 2015 (8/11/2015) resulting in a density estimate of 2.7 wct/100 m^2. This section was difficult to snorkel due to shallow water and a lack of pools. We will revisit the location of this sampling site in 2016.

Large cobble was estimated to be the dominant substrate type followed by gravel in the reach. Average wetted-width measured 11.3 m in 2015. Water temperature was measured at 19.5 ^{o}C at the time of sampling in 2015. No amphibians were observed in Mineral Creek in 2015.

Table 4. Size class, species, and count for fish observed during snorkel surveys in Mineral Creek in 2015.

Species	Lengths	2015
WCT	<45 mm	38
	46-149 mm	94
	≥150 mm	21
	Age 1+ /100m2	2.7
MWF	<49 mm	0
	50-149 mm	0
	≥150 mm	0
	Age 1+ /100m2	0

Upper McDonald Creek

Upper McDonald Creek above the confluence with Mineral Creek is a third order tributary to the M. Fk. Flathead River (Figure 2). It flows primarily southeast, until it joins Mineral Creek at which time it flows southwest to Lake McDonald. Upper McDonald Creek drains approximately 28,700 ha. In 2013 and 2015 we snorkeled from the confluence with Mineral Creek upstream for 329m, the elevation at the sampling site was approximately 1,147m. Sections below Lake McDonald have been surveyed previously by Weaver et al (1983), however, we do not believe that others have snorkeled above the Mineral Creek confluence. McDonald Creek upstream of McDonald Falls is known to contain genetically pure wct (NPS, unpublished data).

In both 2013 and 2015 wct were the only species of fish observed in this reach of McDonald Creek. Fish in all size groups were observed (Table 5). Sixty five wct were observed by snorkelers in 2013 (8/20/2013) resulting in a density estimate of 1.4 wct/100 m^2. We repeated the snorkeling effort in 2015 (8/12/2015) and observed a total of forty wct resulting in a density estimate of 0.62 wct/100 m^2. Observed cutthroat trout density was lower in 2013 than in 2015 (Table 5). Although the 2013 estimate was higher, both the 2013 and 2015 wct density estimates suggest upper McDonald Creek contains relatively low density of cutthroat trout. This is an ideal snorkel reach with very little woody debris, reasonable stream gradient, and desirable pool/run ratio. These low densities of wct within upper McDonald Creek are expected due to the low productivity and nutrient-poor nature of the system.

Cobble was estimated to be the dominant substrate type followed by gravel in the reach. Average wetted-width measured 11.43 m in 2013 and 10.78 m in 2015. Water temperature was measured at 13 ^{o}C in 2013 and 12°C at the time of sampling in 2015. No amphibians were observed in McDonald Creek in 2013 or 2015.

Table 5. Size class, species, and count for fish observed during snorkel surveys in McDonald Creek in 2013 and 2015.

Species	Lengths	2013	2015
WCT	<45 mm	11	18
	46-149 mm	31	19
	≥150 mm	23	3
	Age 1+ /100m2	1.44	0.62

St. Mary River Drainage

Swiftcurrent Ridge Lake (Moran's Bathtub) Netting and Seining

Swiftcurrent Ridge Lake is a 15.2 acre lake located in the Swiftcurrent Creek drainage in Glacier National Park (Figure 2). The lake lies at an elevation of 1858 m. The lake is approximately 395 m long, with a maximum width of 217 m. The drainage is roadless, and the lake is accessed by trail from the Many Glacier Entrance Station. The lake has no visible inlet or outlet present. Grayling were planted within the lake in the 1920s and 30s, however, the illegal introduction of competing suckers and minnows, and the lack of suitable spawning habitat for grayling culminated in the loss of the grayling population. Swiftcurrent Ridge Lake has no backcountry campground.

Little was known about the fishery resource in Swiftcurrent Ridge Lake. Numerous reports and records documented the planting of Arctic grayling throughout the 1920s and brook trout in the late 1930s and early 1940s occurred. Further, it was presumed that one sucker species and two minnow species were introduced into the lake via fisherman using them in an attempt to angle for arctic grayling. The sucker and minnow species introduced have persisted, although the species of sucker and minnows within the lake were not definitively identified until this investigation. It was suspected that the redbelly-finescale dace hybrid may have been one of the introduced species of minnow within the lake. This hybrid dace is a Species of Concern within Montana, and we intended to collect samples for positive identification by Dr. Robert Bramblet at Montana State University.

As we had never sampled Swiftcurrent Ridge Lake previously, we were not sure what to expect. Therefore, both fyke nets and a seine were utilized to sample the lake. We set two trap nets overnight and completed one seine haul within the lake. Trap nets were placed on the northern side of the lake where shallow shelf habitat could be accessed (Figure 3; Table 6). Approximately 15 individuals of each of the two species collected were sent to Montana State University for species identification. Samples were identified as white sucker and lake chub. No redbelly-finescale dace were captured. However, we were very effective at capturing suckers with the fyke nets and both suckers and minnows while seining.

We captured a total of 121 white suckers and three lake chub during fyke trap netting efforts. This resulted in a CPUE of 60.5 white sucker/net night and 1.5 lake chub/net night. Mean length for white suckers captured via netting was 216.5 mm. Length at capture ranged from 110 to 470 mm (Figure 4). We captured eighty three white suckers and thirty nine lake chub in the single seine haul. The mean length of white suckers caught in the seine was 72.6 mm and lengths ranged from 53 mm to

132 mm (Figure 5). The mean length of lake chub caught in the seine was 82.2 mm, and the length of captured chub ranged from 51 mm to 120 mm (Figure 6).

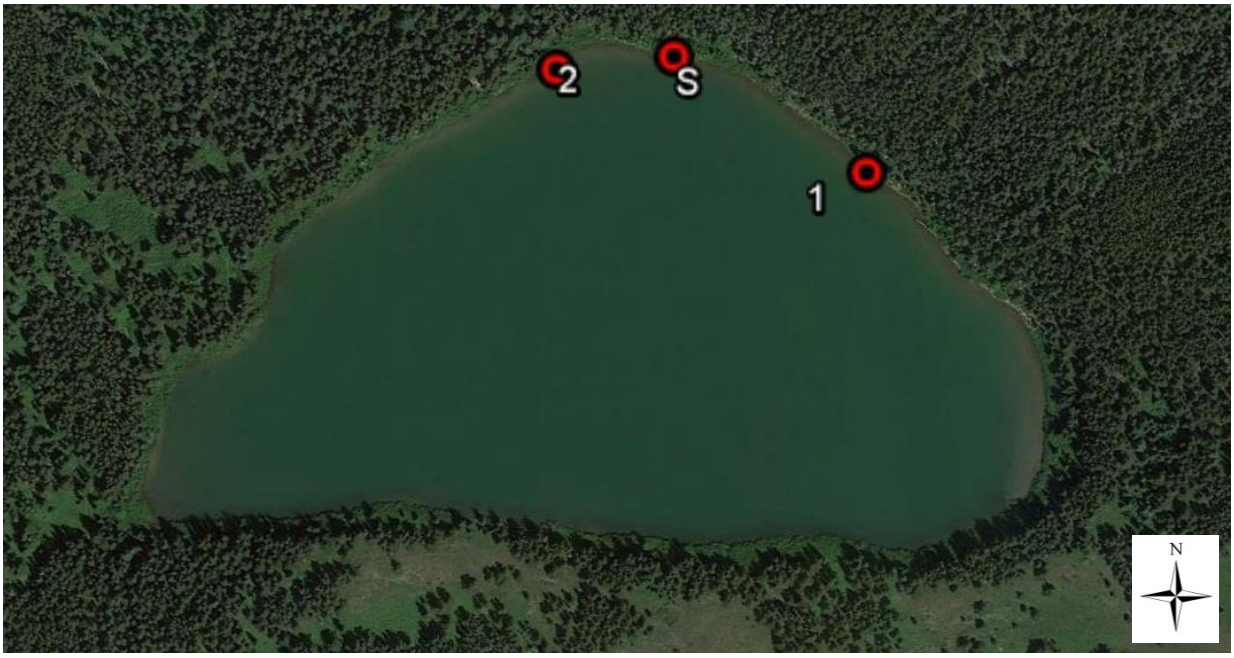

Figure 3. Locations of fyke net sets (1 and 2) and the seine haul (S) in Swiftcurrent Ridge Lake, July 21-22, 2015.

Table 6. UTM's and set parameters of Swiftcurrent Ridge Lake live trapping and siening. For fyke nets, "Depth" refers to the depth of the trap entrance, for seine depth refers to the maximum depth in the haul.

Net	UTM_X	UTM_Y	Date Set	Time Set	Date Pulled	Time Pulled	Depth (feet)
Fyke Net 1	12U 0310463	5412960	7/21/15	14:09	7/22/15	10:14	4
Fyke Net 2	12U 0310330	5413008	7/21/15	14:18	7/22/15	10:14	4
Seine	12U 0310381	5413012	7/21/15	12:50	NA	NA	5

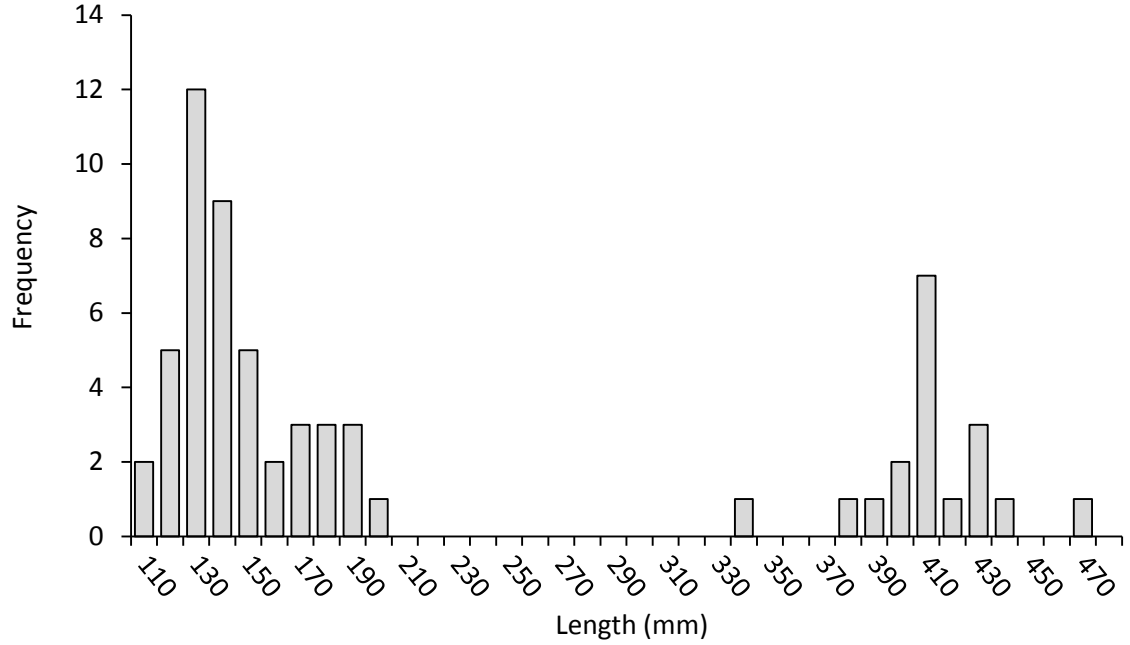

Figure 4. Length-frequency histogram for WSU captured in trap nets in Swiftcurrent Ridge Lake, Glacier National Park, in 2015.

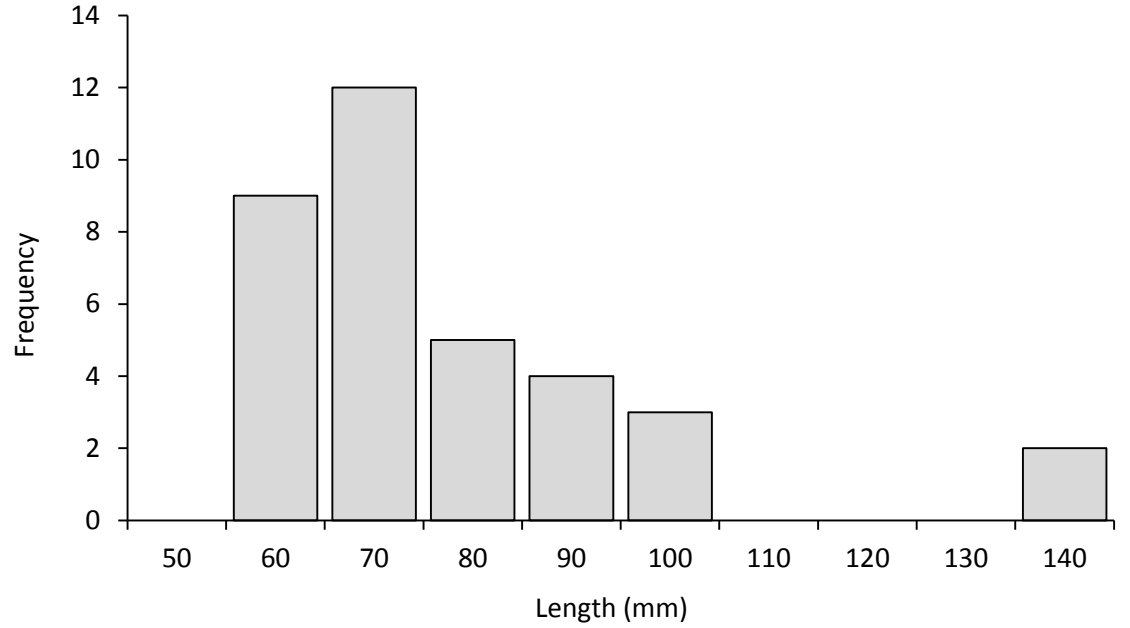

Figure 5. Length-frequency histogram for WSU captured in seine haul in Swiftcurrent Ridge Lake, GNP, 2015.

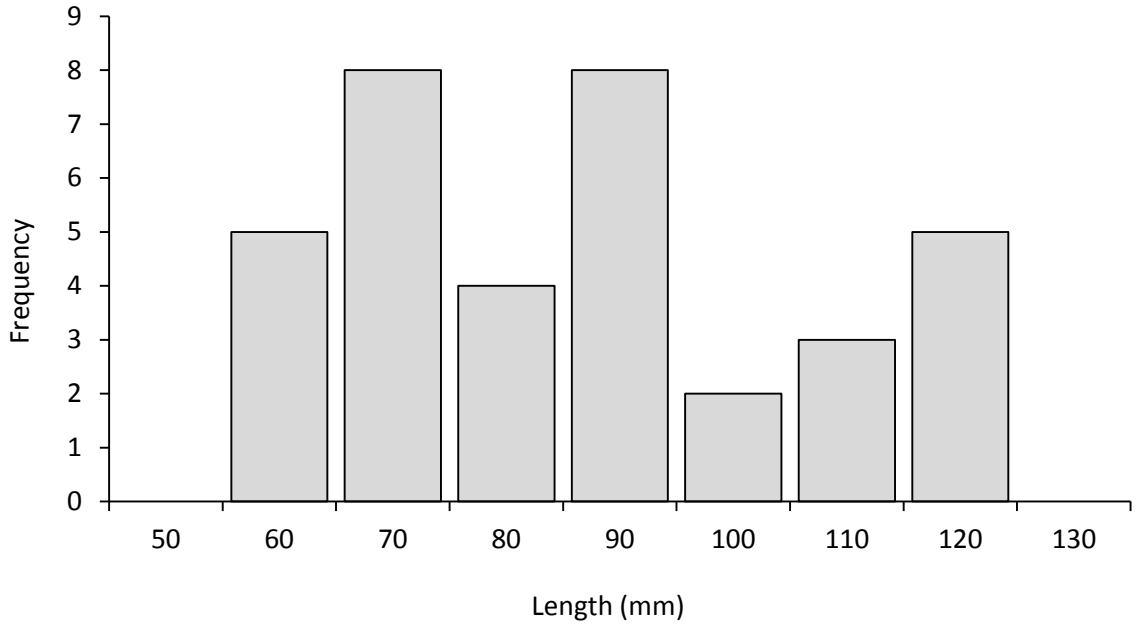

Figure 6. Length-frequency histogram for LCU captured in seine haul in Swiftcurrent Ridge Lake, GNP, 2015.

Stream Temperature Monitoring

Ole Creek

We installed a thermograph in Ole Creek (Middle Fork Flathead River drainage). The thermograph on Ole Creek was located approximately 20m upstream of the suspension bridge at the trail crossing on lower Ole Creek. Stream temperature reached a high of 15.7 ^{o}C in Ole Creek on August 1, 2015 (Figure 7).

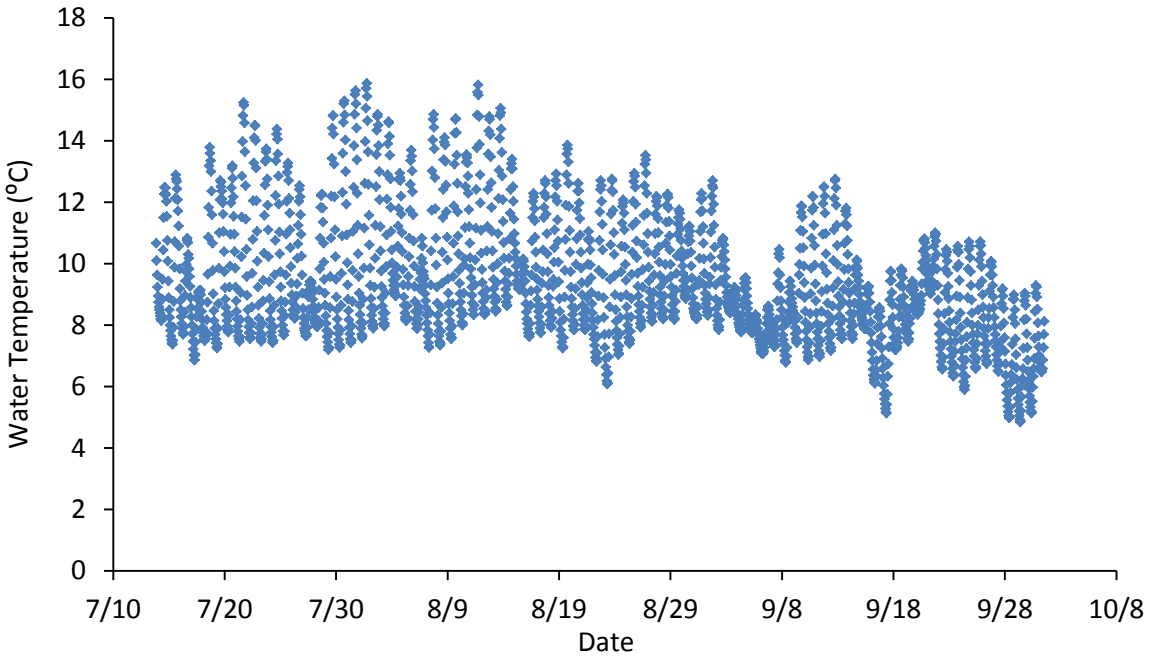

Figure 7. Ole Creek stream temperatures during 2015, Glacier National Park, Montana.

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 (Figure 2). 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 installed a thermograph in Starvation Creek located near the confluence with the North Fork Flathead River, about 100 m upstream of the trail crossing. Stream temperature peaked at 18.7 °C on August 11, 2015 (Figure 8).

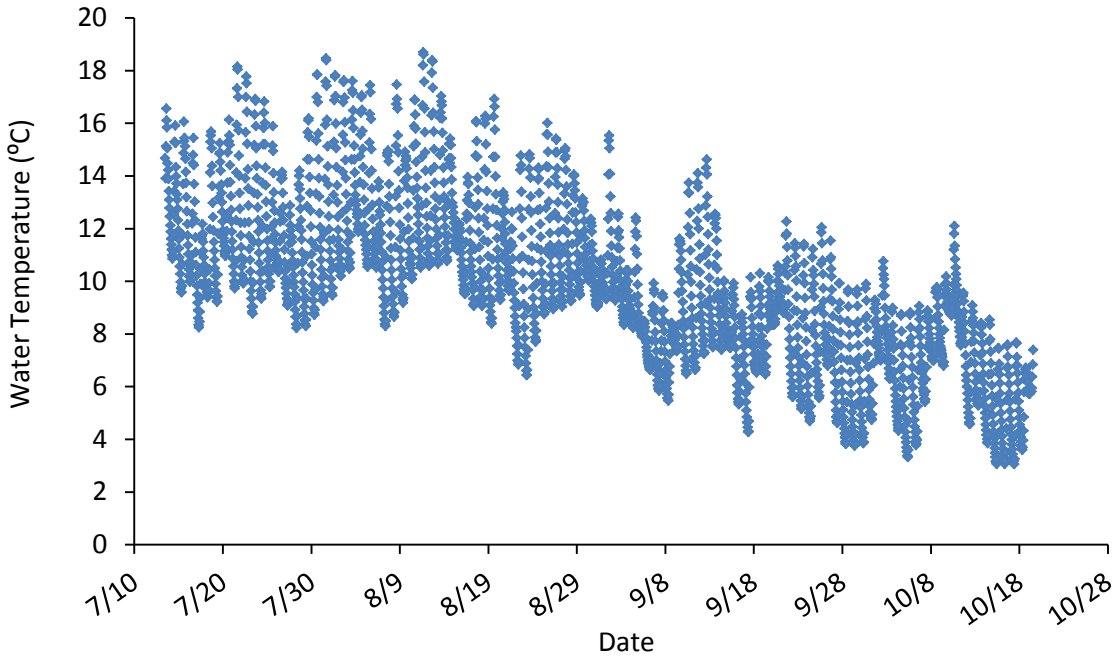

Figure 8. Stream temperature in Starvation Creek during 2015, Glacier National Park, Montana.

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Lake Fisheries Monitoring

ABSTRACT

We sampled Kintla, Bowman and McDonald lakes in GNP in 2015 to characterize and assess fish communities and trends occurring within the fish communities over time. We used sinking multi-filament experimental mesh gill nets, set overnight, to sample all lakes from late July through late August, when the lakes were thermally stratified. Relative abundance of most species have remained stable when compared to historical sampling. Decline in bull trout relative abundance has been occurring since introduced lake trout migrated from downstream areas into each of these three lakes. Following the dramatic decline in bull trout abundance throughout the 1970's and 1980's, bull trout have remained at extremely low numbers when compared to historic abundance estimates. Although vastly reduced in numbers, bull trout persist within all three of the lakes sampled in 2015.

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 approximately 1,000,000 acres, providing high-quality lentic and lotic fish habitat. Glacier National Park supports over 700 perennial lakes/ponds, ranging in size from less than an acre, up to Lake McDonald, covering almost 7,000 surface acres. Glacier National Park also provides over 2,200 km of high-quality perennial stream habitat for aquatic species. A diversity of native and introduced fish species inhabit park waters.

Although GNP represents some of the last best wild areas in North America, recent studies have demonstrated that GNP is not immune to anthropogenic impacts such as exotic species and airborne contaminants. Recent fisheries studies (Marnell et al. 1987, Fredenberg 2002, Hitt et al. 2003, Muhlfeld et al. 2009) have demonstrated the ecological damage and genetic consequences associated with non-native fish species in park waters. For example, Fredenberg (2002) documented the replacement of bull trout as the top predator by non-native lake trout in sympatric lakes on the west side of the park during the last several decades, and lake trout are continuing to expand their numbers and range on the west side of the park (Downs et al. 2015). In response to the lake trout threat the park has recently initiated collaborative efforts to secure remaining at-risk bull trout populations. Monitoring data from park lakes will be key in evaluating any future shifts in fish community structure that may result from additional impacts of non-native species, climate change, or other habitat perturbations.

The purpose of the study was to repeat long-term gill net trend monitoring in Kintla, Bowman and McDonald lakes of GNP (Figure 1). This information will be used to assess the status of native fish populations in park lakes over time and to aid managers in preserving the native species existing within the pristine waters of GNP.

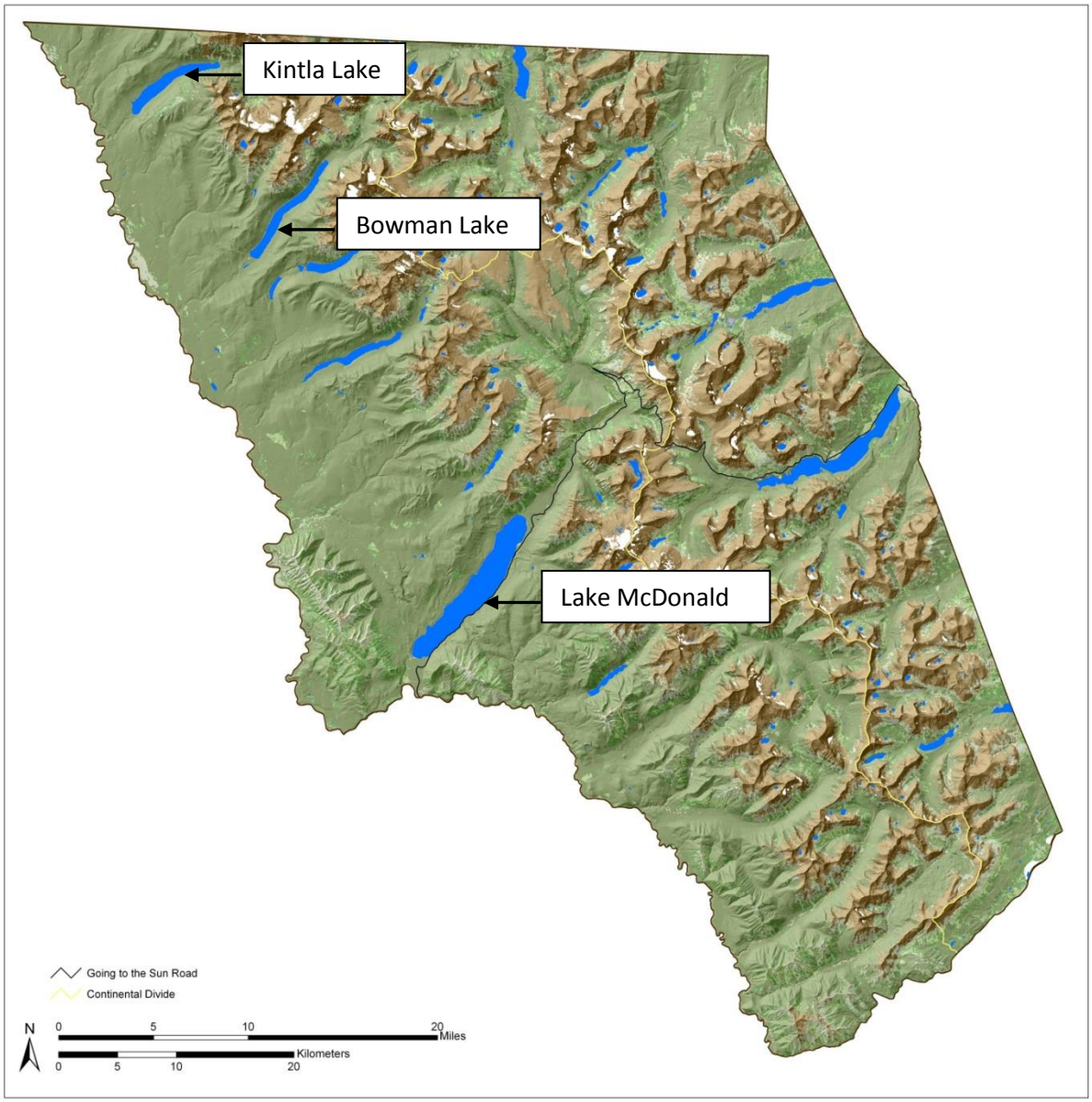

Figure 1. Location of Kintla, Bowman and McDonald lakes, Glacier National Park.

METHODS

We used sinking multi-filament experimental mesh gill nets to sample the lakes. Nets were 38.1 m (125') X 1.83 m (6') experimental multifilament nylon with five 7.62 m (25') panels consisting of 19 mm (3/4"), 25 mm (1"), 32 mm (1-1/4"), 38 mm (1-1/2") , 51 mm (2") bar measured mesh. Thirty pound leadcore and polyfoam line was used to hold the net upright, while maintaining contact with the lake bottom. Two nets were ganged together to form a single 250' net by joining the large and small ends of the net. Kintla Lake was sampled 29-31 July 2015, while Bowman and McDonald lakes were netted 17-19 August 2015 and 24-26 August 2015 respectively. We attempted to repeat set locations

and depths used historically to monitor the lake fish communities (Fredenberg et al. 07 ;Table 1). Nets were set with the smallest mesh towards shore.

Table 1. Gillnet locations and set and retrieval dates for Kintla, Bowman, and McDonald lakes in GNP, 2015.

	Net	UTM Zone	UTM Easting	UTM Northing	Set Date	Retrieve Date
Kintla Lake	1*	11	0694976	5423835	7/30/15	7/31/15
	2	11	0694852	5424931	7/30/15	7/31/15
	3	11	0695476	5425867	7/30/15	7/31/15
	4	11	0698166	5427871	7/29/15	7/30/15
	5	11	0701249	5428564	7/29/15	7/30/15
	6	11	0700128	5427773	7/29/15	7/30/15
	7	11	0697283	5426317	7/29/15	7/30/15
	8	11	0696719	5425698	7/30/15	7/31/15
	9	11	0695790	5424856	7/30/15	7/31/15
	10	11	0697135	5427269	7/29/15	7/30/15
Bowman Lake	1	11	0707196	5415910	8/18/15	8/19/15
	2	11	0707371	5414681	8/18/15	8/19/15
	3	11	0706600	5414339	8/18/15	8/19/15
	4	11	0708277	5416474	8/17/15	8/18/15
	5	11	0709371	5418108	8/17/15	8/18/15
	6	11	0710974	5420710	8/17/15	8/18/15
	7	11	0709944	5417904	8/17/15	8/18/15
	8	11	0705538	5412826	8/18/15	8/19/15
	9	11	0707149	5414111	8/18/15	8/19/15
	10	11	0707950	5416140	8/17/15	8/18/15
Lake McDonald	1	12	0280377	5379629	8/25/15	8/26/15
	2	12	0279965	5381307	8/25/15	8/26/15
	3	12	0283160	5384871	8/25/15	8/26/15
	4	12	0285743	5388532	8/24/15	8/25/15
	5	12	0286332	5389986	8/24/15	8/25/15
	6	12	0288558	5390531	8/24/15	8/25/15
	7	12	0287582	5388444	8/24/15	8/25/15
	8	12	0286501	5385956	8/24/15	8/25/15
	9	12	0282950	5382275	8/25/15	8/26/15
	10	12	0280617	5381949	8/25/15	8/26/15

* Denotes netting locations that differed from historical gillnetting locations.

Gillnets were set in late afternoon/early evening and allowed to soak overnight. They were retrieved at sunrise the following day. Nets were set with the smallest mesh towards shore. Net locations were replicates of those described by Meeuwig et al. (2007) to allow for historical comparisons. All fish captured alive were removed from the net, counted in the net catch total, and were released. All other fish were measured (TL; mm) and weighed (g). Bull trout were sexed, maturity

status was determined, otoliths were removed, and a genetic sample was collected if fish had expired prior to net removal.

Catch-per-unit effort (CPUE), defined as the number of fish captured per net-night, was used as a the primary measure of catch rate over time. In addition, as our comparison across years involves identical numbers of nets set each year, we compared the total number of fish captured for each species as a reflection of relative abundance. We calculated average length and weight for each species to facilitate comparisons of size structure between populations as well as over time within waterbodies. We estimated relative weight (*Wr*) (Anderson and Neuman 1996) for selected fish species to evaluate growth conditions across the sampled waters. We used standard weight equations for lake trout (Picolo et al. 1993), and lake whitefish (Rennie and Verdon 2008) to estimate *Wr* for each water. Fulton-type (K) condition factors were used for other species (Anderson and Neuman 1996).

RESULTS AND DISCUSSION

Ten nets (five per evening) were set in each of the three lakes between 07/29 and 8/25 2015. Species diversity was highest in Lake McDonald (Table 2). Cath-per-unit-effort varied by species in each lake, with mountain whtefish being the most abundant in all lakes (Table 2). Netting data from Kintla, Bowman, and McDonald lakes indicates an increasing relative abundance of lake trout and a decreasing relative abundance of bull trout when compared to historical sampling data (Figures 2, 4 and 6). Catch-per-unit-effort of net sets over time (USFWS, Meeuwig 2008) suggests that lake trout are increasing in abundance fairly rapidly in all three lakes while bull trout abundances are in decline.

Kintla Lake

Kintla Lake was netted from 29 July-1 August, 2015. Five nets were set each night resulting in ten net sets. Four hundred and sixty fish were captured, consisting of eight different species. Mountain whitefish were the predominant catch (N = 262), followed by lake trout (N = 53). Catch results from 2015 were similar to years 1969-2010 (Figure 3). However, bull trout and longnose sucker numbers have decreased over the years, while lake trout, peamouth, and large scale sucker numbers have risen and remained at relatively high numbers when compared to historic catch rates (Figure 3). The increase in largescale sucker numbers is likely due to past identification error. Within the three lakes sampled, catch per-unit effort of lake trout was highest in Kintla Lake (5.3 fish/net night) while bull trout catch per-unit effort was lowest (0.4 fish/net night) (Table 2).

Table 2. Catch composition and catch per unit effort for Kintla, Bowman and McDonald lakes, GNP, 2015.

Water	Species	Number Captured	CPUE (fish/net night)
Kintla Lake	Bull trout (BLT)	4	0.4
	Lake trout (LKT)	53	5.3
	Longnose sucker (LNS)	33	3.3
	Largescale sucker (LSS)	44	4.4
	Mountain Whitefish (MWF)	262	26.2
	Northern pikeminnow (NPM)	1	0.1
	Peamouth (PMTH)	44	4.4
	Westslope cutthroat trout (WCT)	19	1.9
Bowman Lake	Bull trout	6	0.6
	Lake trout	41	4.1
	Longnose sucker	41	4.1
	Largescale sucker	3	0.3
	Mountain whitefish	314	31.4
	Redside shiner	8	0.8
	Westslope cutthroat trout	22	2.2
Lake McDonald	Bull trout	8	0.7
	Eastern brook trout (EBT	1	0.1
	Lake trout	15	1.4
	Longnose sucker	30	2.7
	Largescale sucker	21	1.9
	Lake whitefish (LWF)	29	2.6
	Mountain whitefish	80	7.3
	Northern pikeminnow	176	16.0
	Peamouth	45	4.1
	Redside shiner	18	1.6

Bowman Lake

Bowman Lake was netted from 17 August-19 August, 2015. Five nets were set each night for two nights, resulting in a total of ten net sets. We captured 293 fish comprising seven species during gill-netting efforts (Table 2). Bull trout catch (*N* = 6) and catch rate (0.6 fish per net-night) remained low when compared to historical gill-netting data (Figure 4). Comparisons with the historical catch data continue to show a bull trout population at or near its lowest population level since sampling began. Besides declining bull trout and increasing lake trout abundances, the catch of all other species has remained fairly consistent in Bowman Lake, with the largest proportion of the catch coming from the mountain whitefish (*N*= 314) followed by lake trout and longnose suckers (*N = 41*) (Figure 5). Bowman Lake CPUE for lake trout (4.1 fish/net night) and bull trout (0.6 fish/net night) was second highest for both when compared to the other two lakes (Table 2).

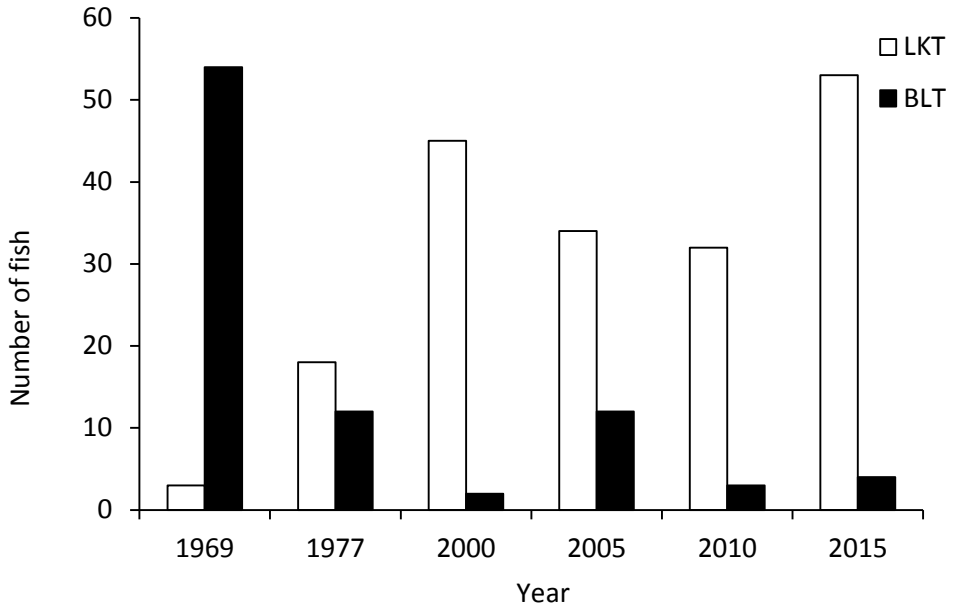

Figure 2. Number of bull trout (BLT) and lake trout (LKT) caught historically, in relation to 2015 standardized gill nets in Kintla Lake, GNP.

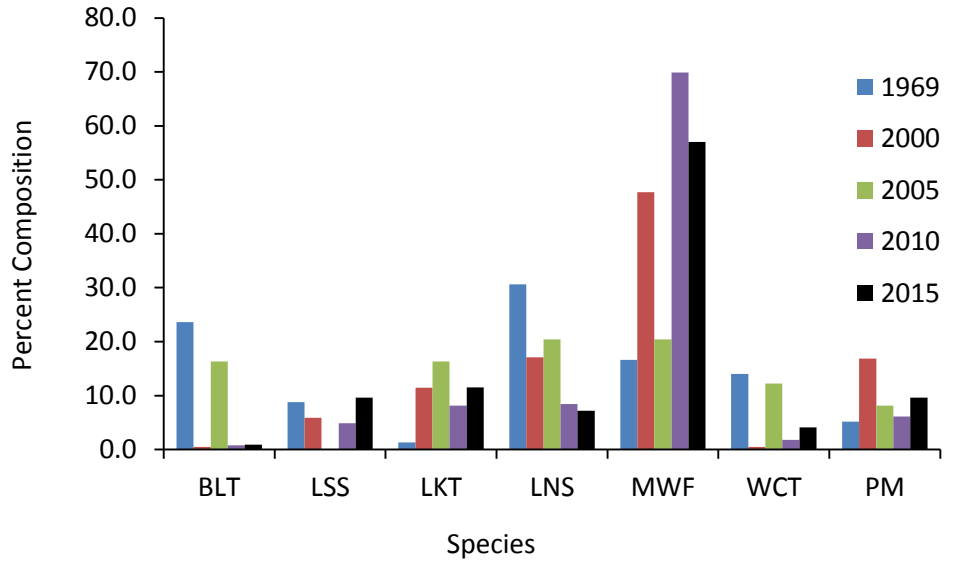

Figure 3. Historical catch composition in relation to 2015 standardized gill net sets in Kintla Lake, GNP.

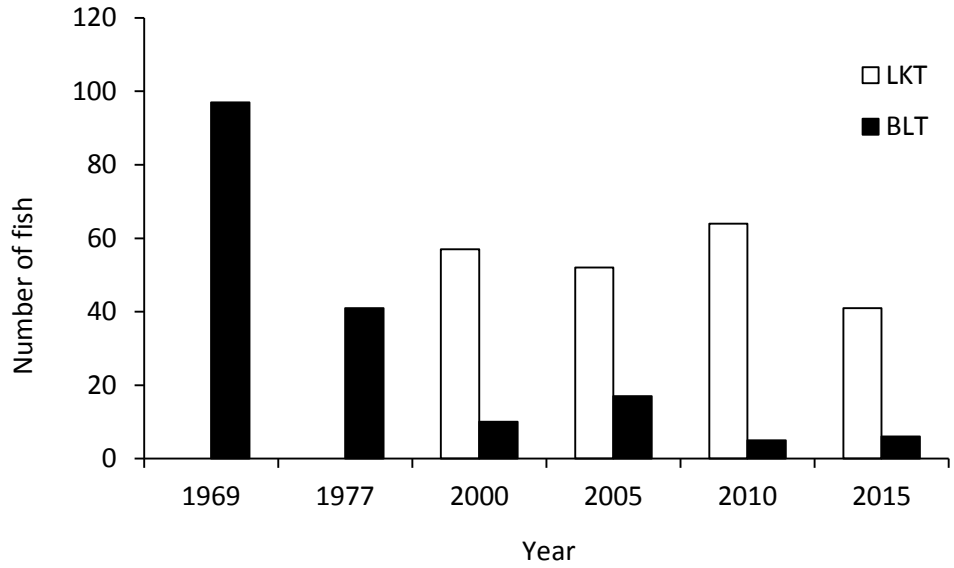

Figure 4. Number of bull trout (BLT) and lake trout (LKT) caught historically, in relation to 2015 standardized gill nets in Bowman Lake, GNP

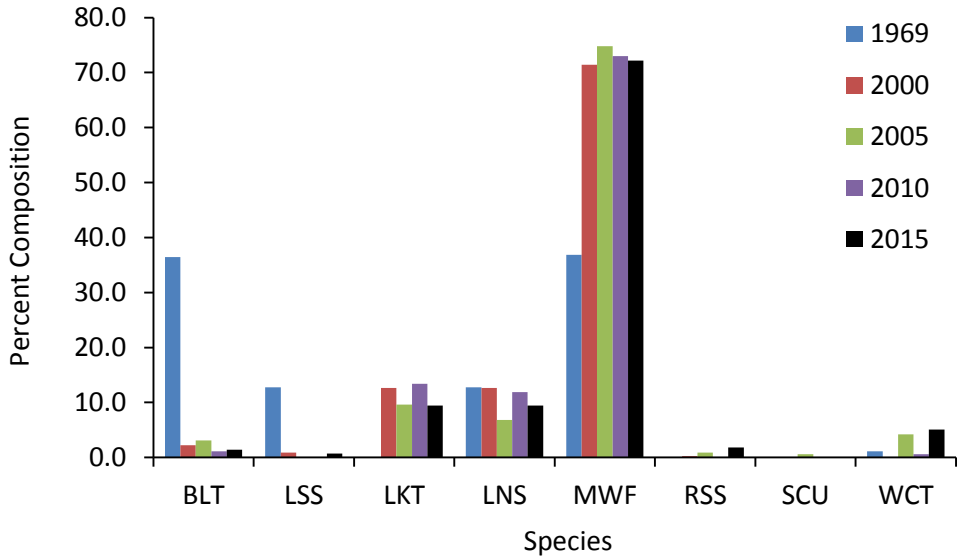

Figure 5. Historical catch composition in relation to 2015 standardized gill net sets in Bowman Lake, GNP

Lake McDonald

Lake McDonald was netted 24 August- 26 August, 2015. We captured 377 fish comprising nine different species during gill-netting efforts in Lake McDonald in 2015. Northern pike minnow were the predominant catch in Lake McDonald (*N = 176*), followed by mountain whitefish (*N = 80*) (Figure 7). Similar to bull trout and lake trout populations in both Kintla and Bowman lakes, bull trout abundance has greatly decreased since the establishment of lake trout within Lake McDonald (Figure 6). Catch per-

unit effort for both lake trout (1.4 fish/net night) and bull trout (0.7 fish/net night) declined since McDonald was last sampled in 2010, however, of the three lakes sampled bull trout CPUE was highest in Lake McDonald (Table 2). Conversely, lake trout CPUE declined since the last sampling event and represented the lowest CPUE for lake trout among the three lakes sampled.

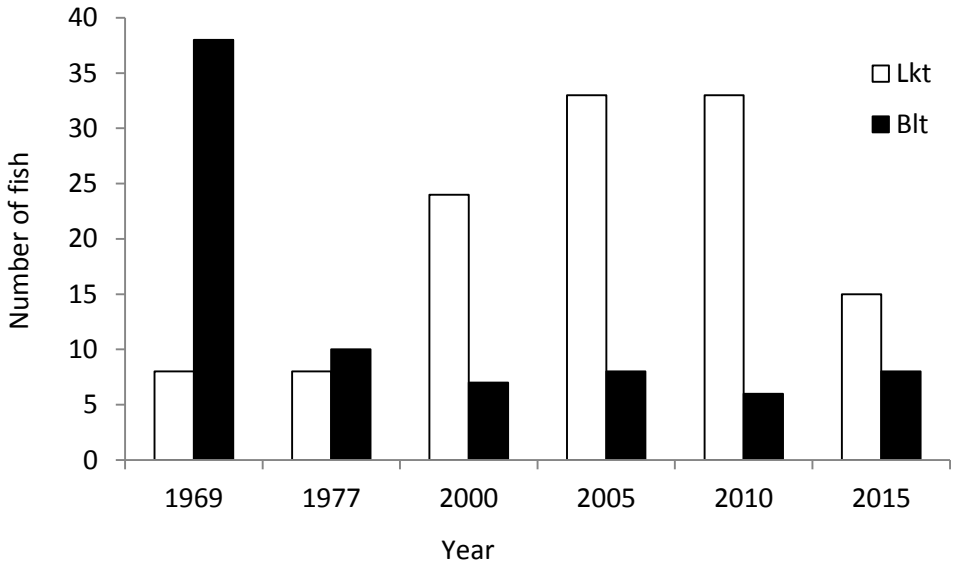

Figure 6. Number of bull trout (BLT) and lake trout (LKT) caught historically, in relation to 2015 standardized gill nets in Lake McDonald, GNP.

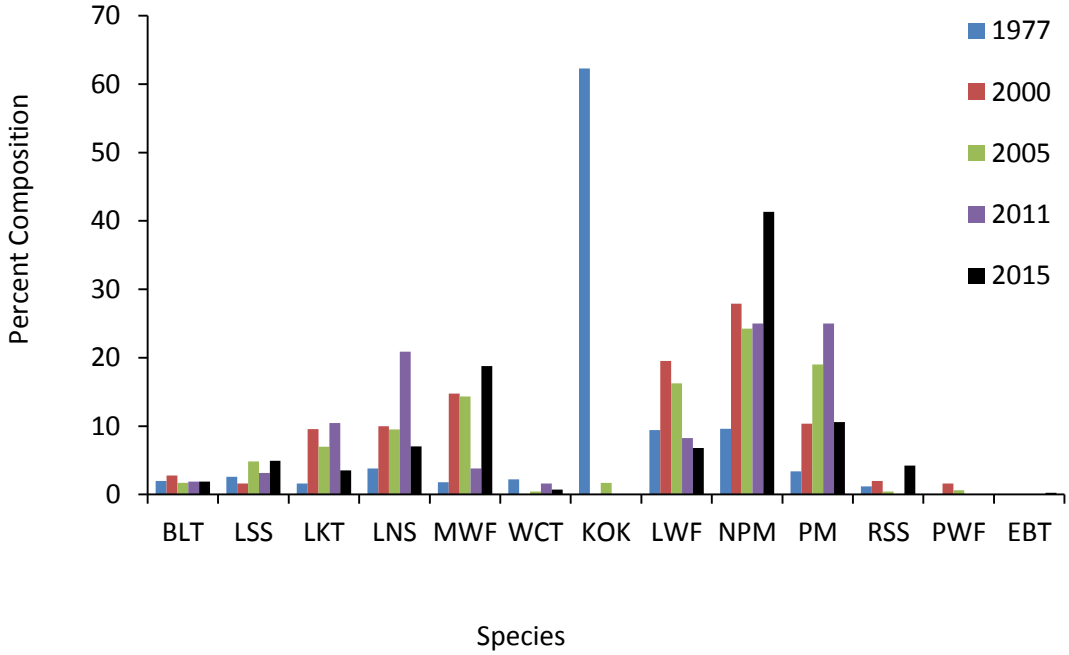

Figure 7. Historical catch composition in relation to 2015 standardized gill net sets in Lake McDonald, GNP.

Both catch rate and total catch of fish varied by water (Table 2). Bowman Lake had the second highest CPUE for lake trout. We also captured six bull trout in Bowman Lake in 2015. This is troubling, as Bowman Lake once supported a strong bull trout population (Fredenberg 2002). Historic assessment of standardized gill net catch data document the replacement of bull trout by lake trout as the dominant aquatic predator in the large-interconnected lakes on the west side of the park (Figures 2, 4 and 6). Trends in catch composition of other species in Bowman Lake do not show a consistent trend (Figure 5). However, sampling conducted in 1969 captured more suckers and fewer mountain whitefish than later years. With limited historical data, it is difficult to speculate as to whether this is an artifact of differences in sampling timing (1969 sampling was consistently 1-2 months earlier than 2000-2015 sampling), or if it represents a shift in species composition. Although we captured westslope cutthroat trout in gill net sets, sinking nets don't effectively sample the upper 6'-10' of the water column where westslope cutthroat trout are most active. A floating gill net series would be a better choice for sampling westslope cutthroat.

Low numbers of bull trout were also captured in both Kintla (*N* =4) and McDonald (*N* = 8) lakes as well, but these lakes do not appear to have the same potential to support a reproducing bull trout population such as Bowman, due to limited tributary spawning and rearing habitat. Meeuwig (2008) evaluated genetic population structuring in GNP, and determined that Kintla Lake and Lake McDonald were not significantly different from one another, had high genetic diversity, and that they were most similar to a composite genetic sample of bull trout spawners from Flathead Lake (Meeuwig et al. 2007). This suggests that these two "populations" could be comprised of individuals from a number of migratory populations supporting Flathead Lake, rather than their own distinct bull trout populations. Meeuwig (2008) further demonstrated genetic differentiation in GNP bull trout was positively correlated with tributary distance. Both Kintla and Lake McDonald are located in close proximity to the mainstem of the North and Middle Forks of the Flathead River, making movement between these lakes and the larger Flathead system fairly easy. Expansion patterns of lake trout within the system further support the idea that these lakes are more readily accessed by migratory fish from Flathead Lake. Gill-netting data suggest Lake McDonald and Kintla Lake were the first lakes to be colonized by lake trout from Flathead Lake (Fredenberg 2002). While Kintla Lake had the highest CPUE of lake trout (5.3 fish/net night), Lake McDonald had the lowest (1.4 fish/net night).

Based on Wr and K, all species evaluated appeared to have less than optimal body condition (Table 3). Overall, lake trout condition was similar across all sampled waters, suggesting less than optimal feeding and/or temperature conditions may exist. A Wr of 100 represents the 75th percentile of average weight for a given length across a large number of populations for a particular species. In concept, a Wr of 100 is generally representative of good physiological and feeding conditions, and has been shown to be positively correlated with fat content in fish (Anderson and Neuman 1996, Renne and Verdon 2008) and prey availability (Renne and Verdon 2008). Ellis et al. (1992) evaluated trophic status for a number of GNP lakes, including McDonald and St. Mary lakes and concluded all of the waters they sampled were either oligotrophic or ultra-oligotrophic. It would not be unexpected to find lower fish condition in such unproductive waters. This conclusion is supported by the findings of Stafford et al. (2002) who found lake trout from Lake McDonald grew considerably slower than those from the more productive waters of Flathead Lake.

Table3. Mean length (TL;mm), mean weight (g), Fulton-type condition factor (K), and relative weight (Wr) for species captured in overnight sinking gill net sets conducted during late summer in Glacier National Park, 2015 (BLT = bull trout, LSS = largescale sucker, LKT = lake trout, LNS = longnose sucker, LWF = lake whitefish, MWF = mountain whitefish, NPM = northern pikeminnow, PMTH = peamouth and WCT = westslope cutthroat trout.

Water Body	Species	Mean Length (SE) (N)	Mean Weight (SE) (N)	Fulton Condition (K)	Relative Weight (Wr)
Bowman Lake	BLT	339.3 (59.23) (6)	561.6 (225.12)	0.84 (0.05)	NA
	LKT	444.8 (22.8) (40)	928.7 (1.23)	0.73 (0.01)	79.22 (1.23)
	LNS	210.8 (8.39) (41)	120.34 (15.6)	1.13 (0.09)	NA
	MWF	253 (4.98) (169)	149.79 (9.03)	0.78 (0.02)	78.4 (1.70)
	WCT	245.82 (19.76) (22)	154.67 (42.18) (18)	0.93 (0.04)	84.0 (3.11)
Kintla Lake	BLT	430 (82.39) (4)	701 (447.93) (3)	0.82 (0.01)	NA
	LKT	402.89 (21.61) (53)	693.02 (124.38) (52)	0.72 (0.01)	80.38 (1.39)
	LNS	216.91 (11.41) (33)	144.48 (23.33) (33)	1.08 (0.02)	NA
	LSS	250.23 (11.55) (44)	222 (27.68) (44)	1.11 (0.01)	NA
	MWF	248.70 (3.44) (262)	152.92 (6.76) (262)	0.79 (0.01)	78.73 (0.92)
	PMTH	178.41 (5.83) (24)	53.01 (5.90) (24)	0.88 (0.02)	NA
	WCT	252.79 (19.17) (19)	190 (46.08) (17)	1.0 (0.02)	90.24 (2.19)
Lake McDonald	BLT	449.38 (52.39) (8)	1025.14 (280.42) (7)	0.85 (0.02)	NA
	LKT	498.33 (36.63) (15)	1186.93 (248.39) (15)	0.76 (0.02)	79.63 (2.26)
	LNS	311.01 (19.74) (30)	455.59 (67.72) (29)	1.16 (0.01)	NA
	LSS	369.57 (20.96) (21)	651.05 (77.73) (21)	1.18 (0.12)	NA
	LWF	372.59 (21.59) (29)	601.24 (88.49) (29)	0.89 (0.02)	89.97 (1.95)
	MWF	245.28 (8.24) (80)	154.59 (14.06) (80)	0.83 (0.01)	82.79 (0.84)
	NPM	228.21 (5.92) (124)	142.32 (13.01) (124)	0.93 (0.01)	NA
	PMTH	166.69 (5.64) (45)	50.27 (4.93) (45)	0.95 (0.02)	NA
	WCT	289.67 (43.97) (3)	308 (159.65) (3)	1.05 (0.06)	92.61 (4.33)

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Aquatic Invasive Species Prevention and Monitoring

ABSTRACT

In 2015, 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 2015, 959 boat inspections were performed and eighteen boats were denied launch permits for reasons ranging from standing water present in internal areas of the boat to dried organic matter adhered to the boat hull. Boat inspections have declined from 2014 by 126 watercraft. Furthermore, boat launch inspections have declined since the implementation of the AIS watercraft inspection stations. In 2011, 1,257 boat inspections were performed, since 2011 the number of inspections have declined annually culminating in the lowest number of vessels inspected in 2015. Boats entered GNP from seventeen states and four Canadian Provinces in 2015, nine of the seventeen states 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, aside from invasive fishes, 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-hand powered 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 2015. 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.

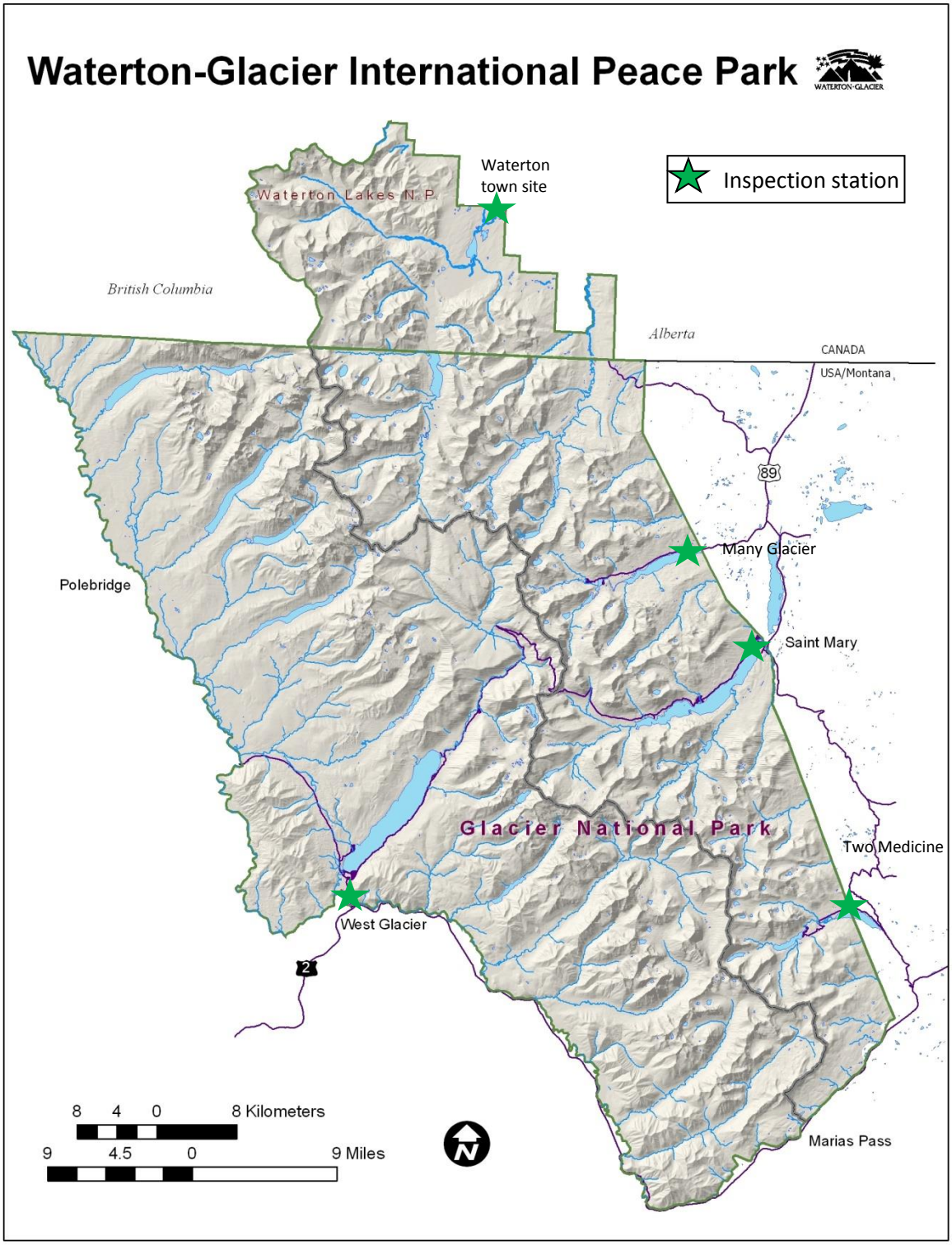

Figure 1. AIS boat inspection and launch permitting locations in Glacier National Park, 2015.

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.

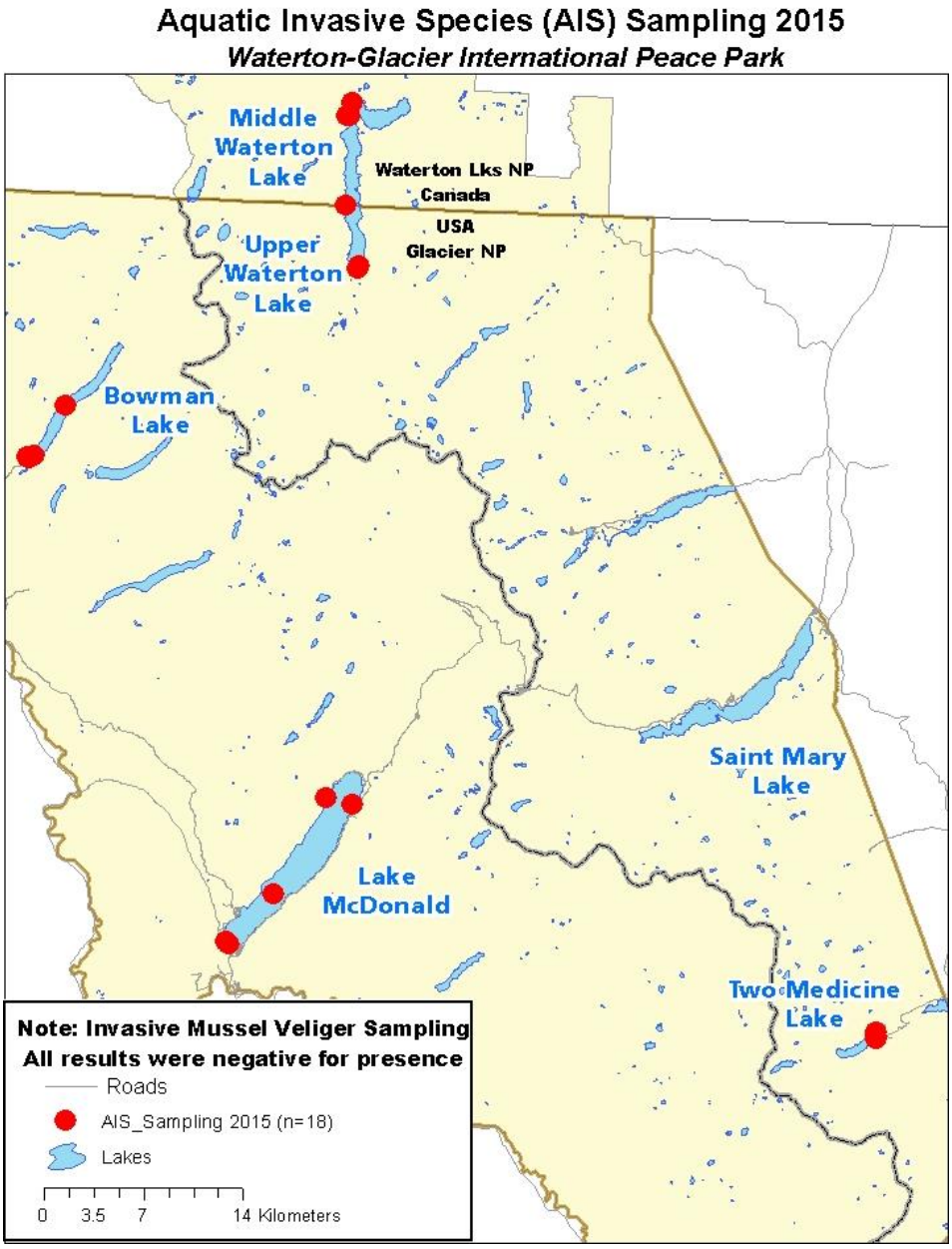

Figure 2. Invasive mussel larvae (veliger) sampling in Waterton-Glacier International Peace Park, 2011-2015.

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 52^0F (11^0C). 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, 2015.

Date Collected	UTM X	UTM Y	Water Body
8/24/2015	279246	5379419	Lake McDonald
8/24/2015	286109	5389646	Lake McDonald
8/24/2015	279054	5379533	Lake McDonald
8/24/2015	287973	5389188	Lake McDonald
8/24/2015	282478	5382948	Lake McDonald
8/6/2015	288334	5426879	Upper Waterton Lake
8/6/2015	288479	5427078	Upper Waterton Lake
8/6/2015	287529	5431368	Upper Waterton Lake
8/6/2015	287794	5437616	Upper Waterton Lake
8/6/2015	287627	5437649	Upper Waterton Lake
8/6/2015	288034	5438579	Upper Waterton Lake
8/5/2015	324816	5372720	Two Medicine Lake
8/5/2015	324819	5372910	Two Medicine Lake
8/5/2015	324849	5373164	Two Medicine Lake
8/31/2015	705896	5412597	Bowman Lake
8/31/2015	705500	5412409	Bowman Lake
9/1/2015	705397	5412481	Bowman Lake
9/1/2015	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.

In 2015, GNP conducted inspections on 959 boats and issued 941 permits to launch. Eighteen 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 infested waters. No AIS were detected. Approximately eighty 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 (8%), Two-Medicine Ranger Station (6%), and the Polebridge Ranger Station (<1%), and Many Glacier (1%). Boats entered GNP from 17 States and four Canadian Provinces this year (Table 2). Nine 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	2015	Total
Alabama	0	1	0	0	0	1
Arizona	8	8	4	12	0	32
Arkansas	0	0	1	0	0	1
California	19	14	8	8	13	62
Colorado	3	2	4	4	5	18
Illinois	1	2	1	0	3	7
Indiana	1	1	2	1	1	6
Louisiana	0	0	1	0	0	1
Maryland	0	0	1	0	0	1
Michigan	0	1	1	4	0	6
Minnesota	5	3	1	3	2	14
Missouri	3	0	1	0	3	7
Nevada	1	0	2	1	0	4
New Mexico	0	0	2	0	0	2
New York	0	2	0	2	0	4
North Dakota	0	0	2	1	0	3
Ohio	0	0	0	0	2	2
Oklahoma	0	1	0	1	0	2
Texas	1	2	1	1	1	5
Utah	1	3	0	0	0	4
Wisconsin	3	1	1	0	1	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 2015 (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 2015 occurred from 5 August-31 August. 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 20.72 °C on 8/03/2015 (Figure 3). Two Medicine Lake reached a maximum temperature of 17.6 °C on 8/13/2015 (Figure 4). Lake McDonald reached a maximum temperature of 21.4 °C on 8/03/2015 (Figure 5).

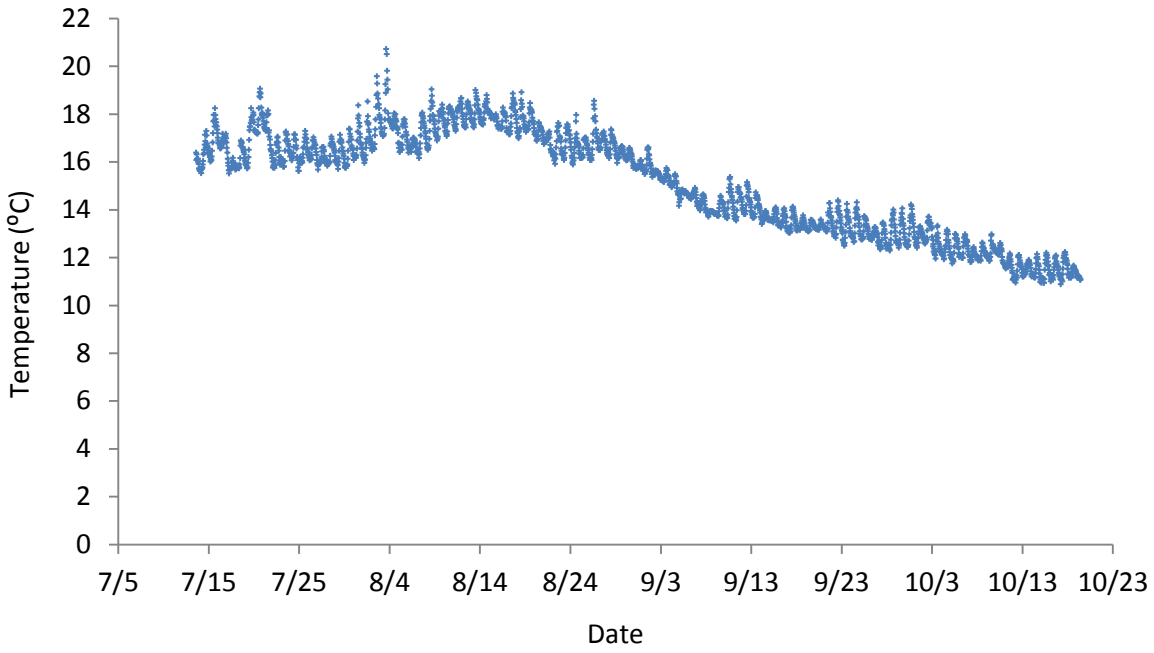

Figure 3. Bowman Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in approximately four feet of water in 2015.

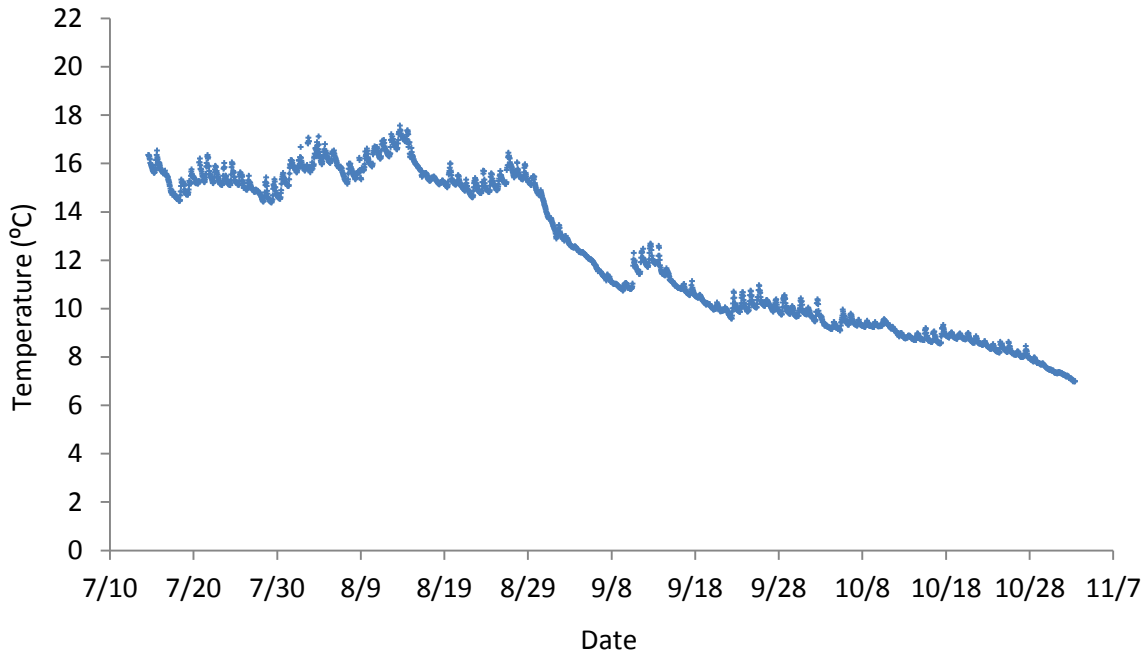

Figure 4. Two-Medicine Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in 2015.

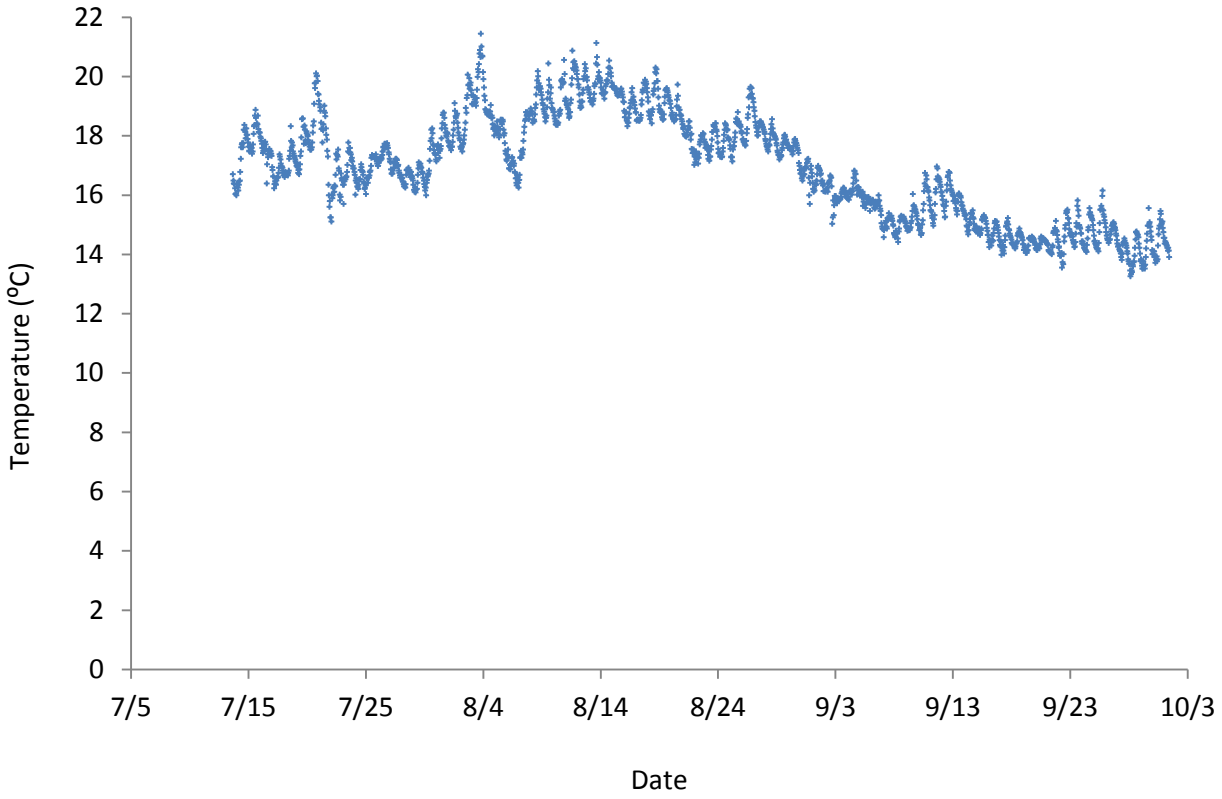

Figure 5. Lake McDonald Lake water temperatures recorded near the lake bottom at the end of the NPS boat dock in 2015.

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

ACKNOWLEDGEMENTS

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Glacier National Park Bull Trout Redd Counts

ABSTRACT

We conducted bull trout *Salvelinus confluentus* redd counts in eleven streams/stream reaches in Glacier National Park in 2015. Six streams/stream reaches were surveyed in the N. Fk. Flathead River drainage, two were surveyed in the M. Fk. Flathead River drainage, and three were surveyed in the St. Mary River drainage. We counted a total of 194 redds across the park. Quartz Lake remains one of the strongest monitored bull trout population residing wholly within the park, with a total of 39 redds. 2015 redd counts for Flathead Lake migratory bull trout populations spawning in the Middle Fork Flathead River tributaries in the park were mixed compared to long-term averages. Redd counts for bull trout populations spawning in the St. Mary drainage were also mixed, with Boulder Creek remaining strong while Kennedy and Lee creeks having below 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 14^0C) and with the number of pocket pools in stream reaches (Saffel and Scarnecchia 1995). Unembedded cobble substrate is an important overwinter habitat type for juvenile bull trout (Thurow 1997; Bonneau and Scarnecchia 1998). Excess fine sediment holds the potential not only to reduce egg and embryo survival, but might also limit juvenile bull trout abundance in streams by reducing the amount of interstitial spaces available for overwinter habitat. Channel stability, habitat complexity, and connectivity are all important components in bull trout population persistence (Rieman and McIntyre 1993).

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

understanding bull trout population dynamics in the park, and in establishing management programs to benefit native fish.

Redd counts, or spawning nest counts, are used across the range of bull trout to monitor population trends. They are typically used as an index of abundance to gauge the relative strength of adult escapement from year to year. They can also be used to estimate actual adult escapement by expanding the redd counts to fish numbers using various spawner to redd ratios. Redd counts require far less effort to conduct than other traditional monitoring methods such as trapping, and yet provide valuable information on bull trout at the watershed and/or population scale. However, redd counts are not without 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). 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. Bull trout redd counts have been conducted annually 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, 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 and the USFWS identified and enumerated bull trout redds in 2015. Redd surveys generally occur during the first full three weeks of October. Surveys occurred between October 5-October 20 in 2015. 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. We analyzed trends for the most recent 10-year count period for Kennedy and Boulder creeks on the east side of the park, as well as Ole and Quartz creeks on the west side of the park. 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 six stream reaches in the N. Fk. Flathead River. Additionally two 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, USGS, and USFWS (Figure 1).

East of the Continental Divide, relatively few bull trout populations are monitored using redd counts(Figure 2; Appendix A). Boulder is clearly the strongest migratory population, averaging 41 redds/yr. Correlation analysis indicated a statistically significant positive 10 year trend (Tau-b=0.55; $p<0.05$) in the number of bull trout spawning nests (redds) in Boulder Creek (Figure 2; Appendix A). Both Lee and Kennedy creeks had below average red counts, although no statistically significant trends were identified. The redd count of five on Lee Creek is well below the five-year average of sixteen redds. The reasons for the continued strength of the Boulder Creek bull trout population are unclear, but there are tributary spawning and rearing habitat quality differences between the systems. Overall, bull trout habitat quality is higher in Boulder Creek as indicated by stable streambanks, a stable bedform, and abundant spawning gravel. Kennedy Creek can be characterized as relatively unstable and dominated by cobble-sized substrate. Spawning gravel is very limited. Lee Creek is considerably smaller than both Boulder and Kennedy creeks, but habitat quality is high. Cold water temperatures along with sufficient large woody debris and spawning gravel should afford adequate spawning and rearing habitat to support this population. However, we have no information on the fate of juveniles or adults once they move downstream and into Canada.

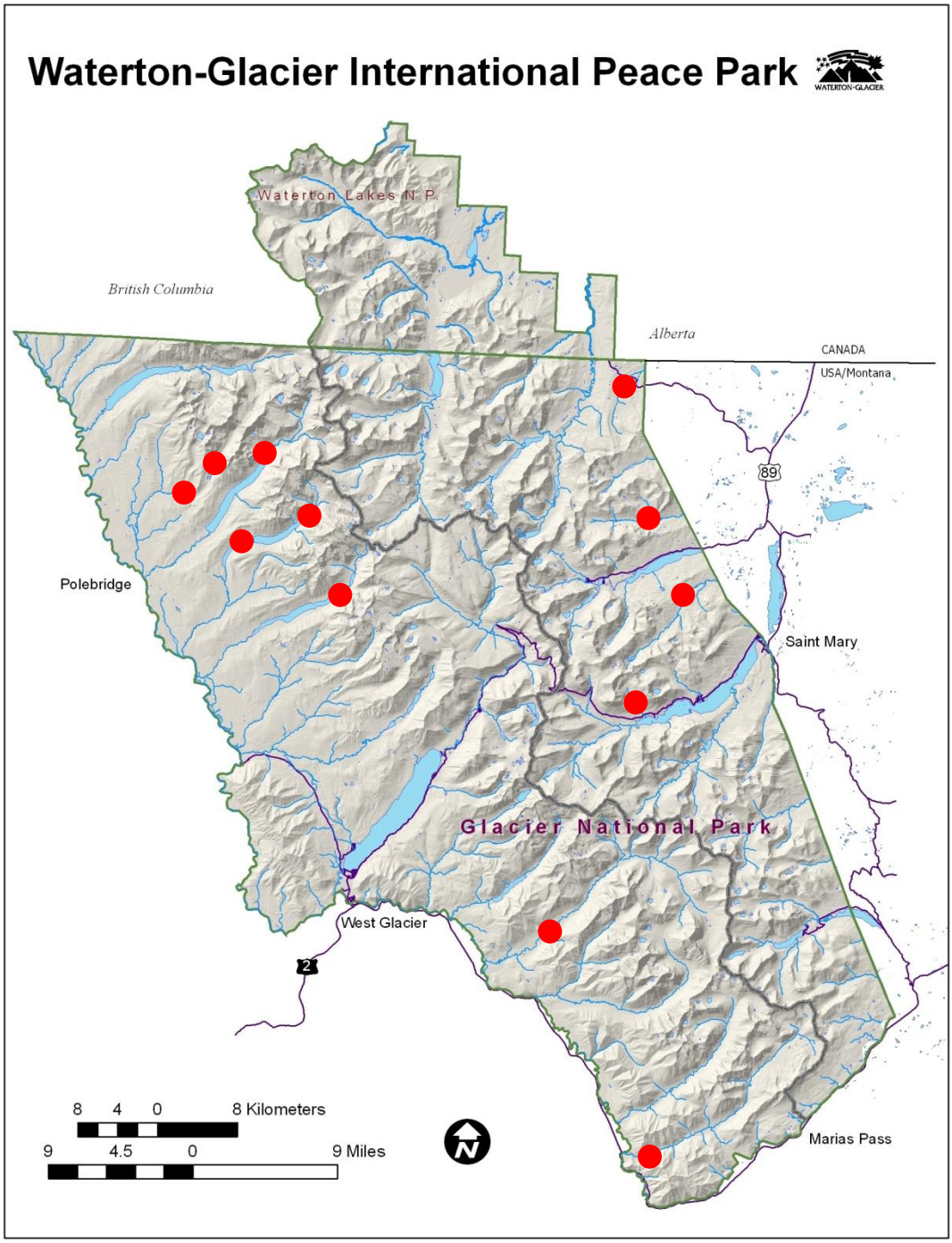

Figure 1. Drainages monitored for bull trout spawning activity (red circles) in Glacier National Park, Montana in 2015.

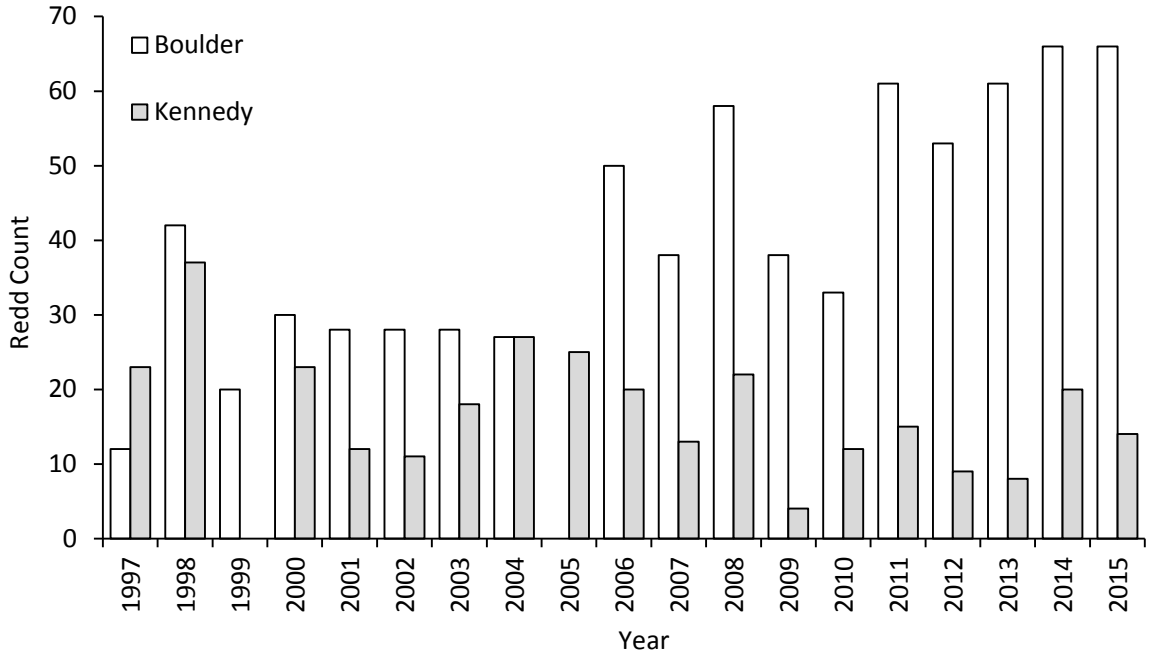

Figure 2. Bull trout redd counts for Boulder and Kennedy creeks, Hudson Bay Drainage, Glacier National Park.

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 threat to 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 death 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). Addressing the fishery impacts of this project would 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).

On the west side of the park, redd counts for the migratory populations from Flathead Lake using Middle Fork Flathead tributaries were near their historical averages. Ole Creek appears to be at or near all-time spawning highs, while Nyack Creek is about 50% lower than its historic high counts that occurred from 1982-1991 (Figure3; Appendix A). Park Creek, the other regularly monitored Middle Fork tributary in the park, was not counted in 2015. No statistically significant trends were identified in these data. The reasons for the differences in trends between Ole and Nyack creeks are not entirely clear, but it is noteworthy that the amount and quality of bull trout spawning and rearing habitat is higher in Ole Creek (and brook trout are absent). As such, it would not be unexpected that Ole Creek would be a more resilient and consistent spawning stream.

Disjunct lake bull trout populations in the park's North Fork Flathead River drainage showed mixed results. No redds were counted in either Bowman or Logging creeks. Bull trout are at or near functional extinction in these systems. Efforts to move juvenile bull trout from Logging Creek/Lake upstream Into Grace Lake is the only option at this point to conserve this population and its genetic legacy.

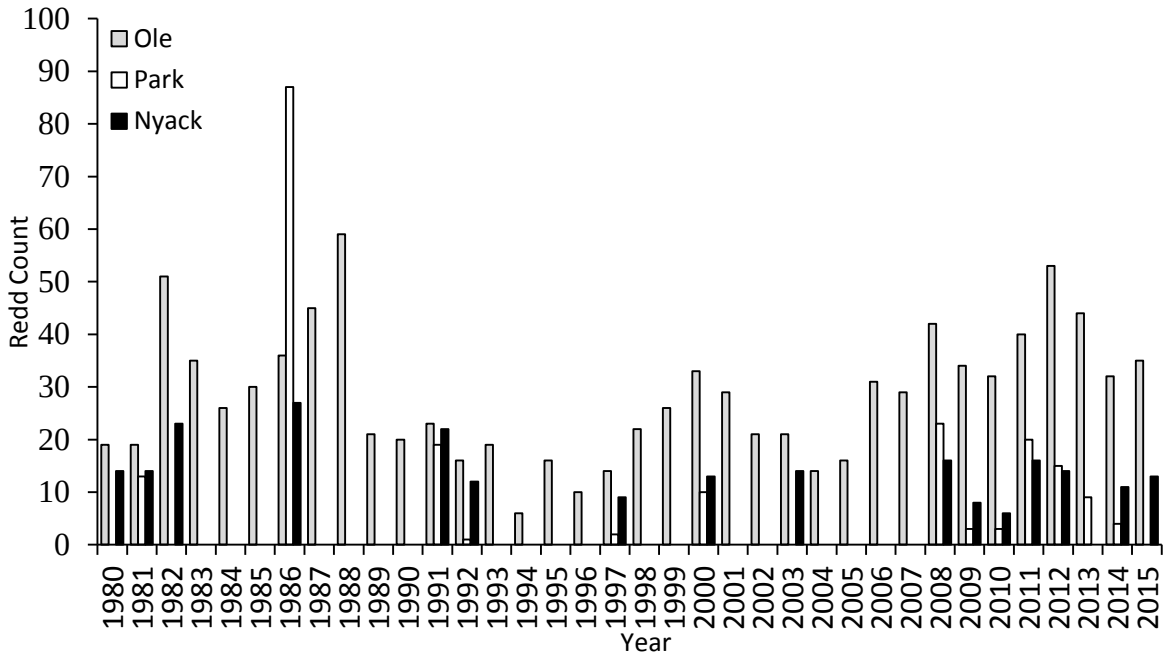

Figure 3. Bull trout redd counts conducted in Ole, Park, and Nyack creeks, Middle Fork Flathead River Drainage, Glacier National Park.

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. Similarly, a lake trout removal project was initiated on Logging Lake and was successful at removing large numbers of adult lake trout in 2015. Both of these projects are anticipated to continue through at least 2022.

Redd counts on Quartz Creek, the primary spawning area for bull trout from Quartz Lake have not shown a statistically significant trend of increasing or decreasing abundance (tau-b = 0.11, $p > 0.05$; Figure 4; Appendix A). We did not anticipate a positive trend since the lake trout invasion was relatively

new and bull trout remained abundant. However, a declining trend would suggest either bycatch mortality in gill nets was having a substantial adverse impact on bull trout, or lake trout suppression efforts were not being successful at preventing lake trout population expansion. The 2015 redd count was encouraging, as it was higher than the historical redd count average of 34 redds. The redd count from 2015 was the second year in a row that the count was higher than the historical average. The high redd counts in Quartz Creek reflect positively on the ongoing lake trout population suppression, indicating that the suppression activity is not significantly reducing the adult bull trout population. Continued monitoring will be necessary as a key measure of bull trout population strength in the system.

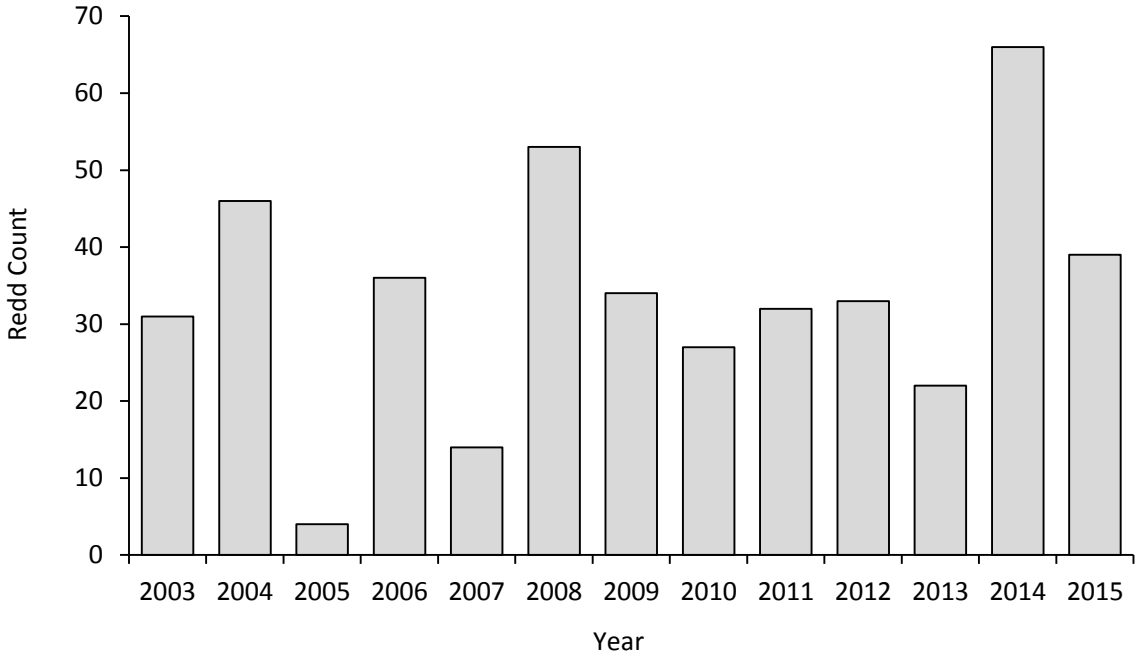

Figure 4. Bull trout redd counts in Quartz Creek, Glacier National Park, Montana.

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

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

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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 f	2005 b,f	2006	2007 a,c	2008 c,d,e	2009 g	2010 h	2011	2012	2013	2014	2015
Hudson Bay Drainage																			
Boulder Cr.	12	42	20	30	28	28	28	27	--	50	38	58	38	33	61	53	61	66	66
Kennedy Cr.	23	37	--	23	12	11	18	27	25	20	13	22	4	12	15	9	8	20	14
Lee Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	15	31	12	16	5
Rose Cr.	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	0	--	--
N. Fk. Flathead Drainage																			
Akokala Cr.	--	--	--	--	--	--	--	--	--	--	--	11	6	1	4	5	--	12	14
Agassi Cr.	--	--	--	--	--	--	--	--	--	--	--	0	--	--	--	--	--	--	--
Bowman Cr.	--	--	--	--	--	0	0	0	0	2	1	0	0	1	3	--	1	4	0
Jefferson Cr.	--	--	--	--	--	--	--	--	--	--	--	0	0	--	--	--	--	0	--
Logging Cr.	--	--	--	--	--	--	--	3	20	0	--	5	0	3	3	1	27	0	0
Quartz Cr. (lower)	--	--	--	--	--	--	--	1	3	2	2	3	2	2	5	3	5	4	1
Quartz Cr. (middle)	--	--	--	--	--	--	0	0	0	0	0	0	3	0	--	--	--	1	0
Quartz Cr. (upper)	--	--	--	--	--	--	31	46	4	36	14	51	34	27	32	33	22	66	39
Rainbow Cr.	--	--	--	--	--	--	--	--	--	--	--	28	12	4	9	--	4	5	7
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	35
Nyack Cr.	9	--	--	13	--	--	14	--	--	--	--	16	8	6	16	14	--	11	13
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