Mercury Hazard Assessment for Piscivorous Wildlife in Glacier National Park

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Abstract

We examined the mercury hazard posed to selected piscivorous wildlife in Glacier National Park (GNP), Montana. Logging Lake was our focal site where we estimated the dietary mercury concentrations of wildlife (common loon [*Gavia immer*], American mink [*Neovison vison*], river otter [*Lontra canadensis*], and belted kingfisher [*Megaceryle alcyon*]) by assuming that fishes were consumed in proportion to their relative abundances. To evaluate if Logging Lake provided a suitable baseline for our study, we made geographic comparisons of fish mercury levels and investigated the distribution and abundance of high mercury fishes within GNP. We complimented our assessment by examining selenium:mercury molar ratios in fishes from Logging Lake and Saint Mary Lake. Our results suggest fish consumption does not imperil wildlife from Logging Lake based on published thresholds for adverse mercury effects, but some hazard may exist particularly if there is strong feeding selectivity for the most contaminated species, northern pikeminnow (*Ptychocheilus oregonensis*). The geographic comparisons of fish mercury levels, together with the distribution and abundance of high mercury fishes within GNP, suggest that Logging Lake provided a relatively protective baseline among our study lakes. Risk may be further reduced by the molar excess of selenium relative to mercury, particularly in the smaller fishes typically consumed by GNP wildlife. Our findings contrast with studies from northeastern US and southeastern Canada where greater mercury hazard to wildlife exists. An emergent finding from our research is that waterborne concentrations of methylmercury may provide limited insight into regional differences in fish mercury levels.

Keywords: Fish mercury, fish selenium, air pollution, piscivorous wildlife, Glacier National Park

Introduction

The western U.S. contains a diverse array of protected natural areas, yet these areas are not immune from the impacts of aerially delivered toxicants. For example, recent studies in Glacier National Park (GNP), Montana, have documented the presence of PCBs, pesticides, herbicides, fire retardants, and mercury in precipitation, lake water, lake sediments, and fish (Watras et al. 1995, Krabbenhoft et al. 2002, Ackerman et al. 2008, USEPA 2009, Landers et al. 2008, Schwindt et al. 2008, Mast et al. 2010, Eagles-Smith et al. 2014). The presence of these pollutants poses potential hazards to aquatic biota and their consumers. Based on a large scale survey of airborne toxicants in western national parks (including GNP), Landers et al. (2008) concluded that mercury likely posed one of the greatest ecological threats.

Although mercury naturally occurs in the environment, investigations of lake sediments (Swain et al. 1992), glacial cores (Schuster et al. 2002), and preserved animal tissues (Dietz et al. 2009) have revealed large increases since the onset of the Industrial Revolution. The primary source of elevated mercury levels in most areas is increased atmospheric deposition, particularly in remote headwater settings such as GNP (Fitzgerald et al. 1998). Currently, coal burning is the largest source of mercury to the atmosphere followed by artisanal gold mining (Streets et al. 2011). Glob-

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ally, it is estimated that the mercury flux from the atmosphere has tripled since the Industrial Revolution (Lindberg et al. 2007). Locally, sediment cores from four GNP lakes show that modern fluxes of mercury averaged 3.2 times higher than pre-industrial (Mast et al. 2010).

Given the importance of the atmosphere in delivering mercury, regional concentrations in precipitation and the associated mercury deposition rates are relevant when evaluating mercury hazards. The National Atmospheric Deposition Program (NADP) regularly collects precipitation samples from sites throughout most of the US, including GNP, which are analyzed for total mercury. The NADP maps generated from these data show that mercury deposition (which integrates mercury concentration and precipitation amount) is greatest in the Southeast and high elevation mountains of the West (NADP 2016). In GNP, the increasing amount of precipitation with elevation gain results in a large mercury flux to the park's mountainous landscape (GNP elevation range = 949 to 3190 m) (Figure 1). The extent to which mercury deposition patterns present an ecological hazard is influenced strongly by the speciation of mercury in the environment.

Although atmospheric deposition is dominated by inorganic mercury, methylmercury is typically the form of greatest ecotoxicological concern. Methylmercury primarily is produced in lakes and their watersheds by sulfate reducing bacteria (Gilmour and Henry 1991), although some is deposited aerially (Fitzgerald et al. 1991). Methylmercury biomagnifies in aquatic food chains and generally comprises a larger fraction of the total mercury burden with increasing trophic level (Mason and Sullivan 1997). These food chain dynamics together with the high toxicity

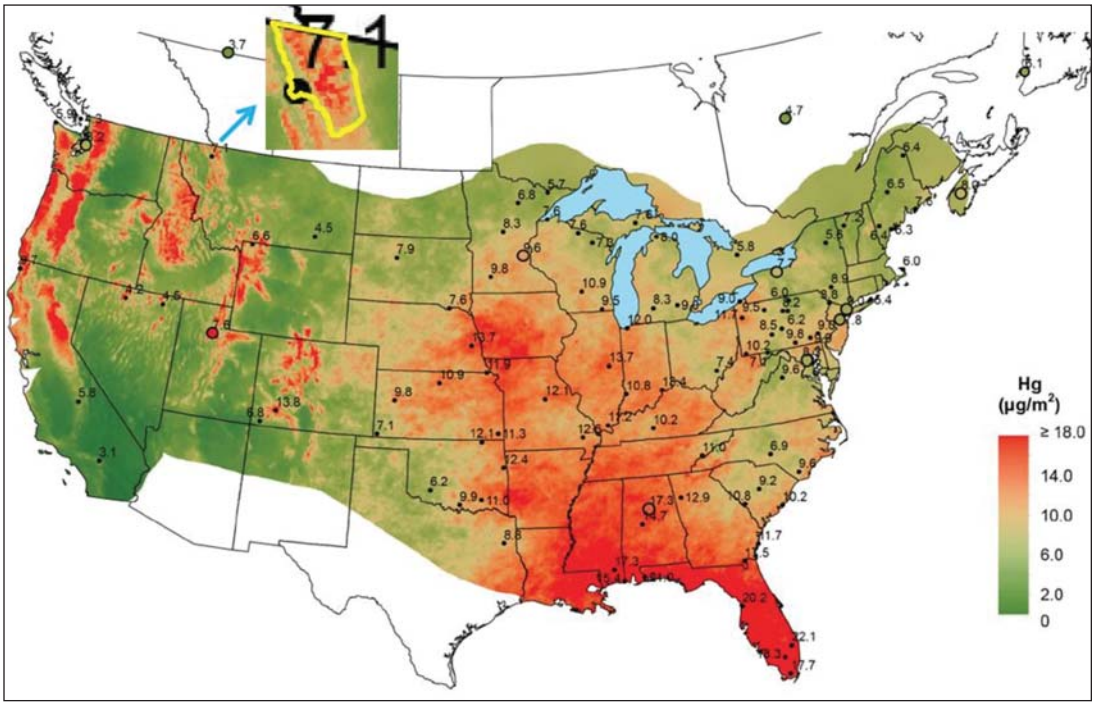

Figure 1. Annual amount of total mercury deposited by precipitation per unit area (µg/m^2) in the US and parts of southern Canada for 2014. Black dots are denoted with the observed data at each monitoring station, while colors represent values based on a modeled, spatial interpolation. The inset denoted with the blue arrow is an enlargement of the Glacier National Park area, including the West Glacier precipitation monitoring station and the park boundary (yellow). All data and the associated map are from the National Atmospheric Deposition Program (NADP 2016), and we have modified their map with the inset and cropping.

of methylmercury create toxicological concerns, particularly for high level consumers such as piscivorous wildlife.

Fish consumption is the mercury exposure pathway of greatest concern to wildlife (Scheuhammer et al. 2007). Methylmercury is a potent neurotoxin that has a high mobility in the body because it readily crosses biological membranes, and this mobility also facilitates prenatal exposure (Hiroshi et al. 1985). Further, in mammals, postnatal exposure can occur through consumption of breast milk (Knott et al. 2012). In piscivorous birds, the demographic factor most sensitive to methylmercury exposure is reproduction, and this also may apply to piscivorous mammals and fishes (Scheuhammer et al. 2007). In North America, mercury impacts to piscivorous wildlife especially have been reported in the northeastern US and southeastern Canada, and are concomitant with elevated fish mercury concentrations (Stafford and Haines 1997, Evers et al. 2003, Depew et al. 2013). These impacts have been well documented common loon (*Gavia immer*), American mink (*Neovison vison*), and river otter (*Lontra canadensis*) (Scheuhammer et al. 2007), all of which occur in GNP.

Given that atmospheric mercury deposition is higher in GNP than the northeastern US, a potential issue is that fish mercury levels are negatively impacting the park's piscivorous wildlife. This is particularly concerning if the high deposition rates are accompanied by concomitantly high concentrations of methylmercury and/or the fish species present have a high propensity to accumulate methylmercury in the environments of GNP. Waterborne methylmercury concentrations in lakes from GNP are low relative to other US locations (Watras et al. 1995, Krabbenhoft et al. 2002), relieving the former concern, however, the later possibility remained largely unexamined.

The propensity of fishes to accumulate methylmercury often varies substantially among and within species due to numerous factors. Risks for elevated fish mercury levels at a given body size include: feeding high on the food chain, feeding on long food chains, habitat preferences for warm water, oligotrophic habitats, low dietary selenium, and slow, inefficient growth (Cabana and Rasmussen 1994, Stafford and Haines 1997, Simoneau et al. 2005, Driscoll et al. 2007, Yang et al. 2008, Peterson et al. 2009a, Ward et a. 2010). Accordingly, several aspects of GNP could allow fish mercury concentrations to reach levels of concern. Piscivorous fishes occur throughout most of GNP's low elevation lakes, particularly lake trout (*Salvelinus namaycush*) and bull trout (*Salvelinus confluentus*). A warm water piscivore, northern pikeminnow (*Ptychocheilus oregonensis*), is of particular concern for mercury accumulation but is only present in a few GNP lakes (Schultz 1941, Meeuwig et al. 2008, Peterson et al. 2009a). Other potential risks for elevated fish mercury levels are the low rates of fish growth and the associated low levels of primary productivity in the cold, oligotrophic waters that characterize most of GNP (Dux 2005, Simoneau et al. 2005, Driscoll et al. 2007).

Existing mercury data for GNP fishes is limited. Mercury testing has been conducted for cutthroat trout (*Oncorhynchus clarkii*) from Snyder Lake and Oldman Lake (Watras et al. 1995, Landers et al. 2008, Schwindt et al. 2008), lake trout from Upper Two Medicine Lake (USEPA 2009), and bull trout as well as lake whitefish (*Coregonus clupeaformis*) from Lake McDonald (Eagles-Smith et al. 2014). Additionally, Waterton Lake (which straddles GNP and the adjacent Waterton Lakes National Park in Alberta) has been tested using lake trout and lake whitefish (Brinkmann 2007). Collectively, these previous investigations reveal muscle mercury levels (wet weight) ranging from ~ 0.03 mg kg^{-1} in "small" cutthroat trout (Watras et al. 1995) from Snyder Lake to 1.41 mg kg^{-1} in a 510 mm fork length (~ 556 mm total length [TL]) lake trout from Waterton Lake (Brinkmann 2007). Although these previous investigations show that mercury concentrations in GNP salmonids can reach levels of toxicological concern, more information was needed to better assess the hazard to piscivorous wildlife. In particular, many fishes in GNP (especially non-salmonids) had not been tested for mercury, the tested species sometimes lacked data for the smaller sizes primarily consumed by wildlife, and the geographic coverage was insufficient to assess spatial mercury variation

within fish species. Because of these limitations, the mercury risk to GNP's piscivorous wildlife largely was unknown.

Mercury risks to fish consumers may be ameliorated by adequate dietary selenium. Selenium naturally occurs in aquatic environments and the associated biota, but can be increased by various human activities (Lemly 2004). Selenium, like mercury, can reach levels of toxicological concern in aquatic ecosystems (Lemly 2004). However, unlike mercury, selenium is an essential nutrient and thus beneficial at appropriately low dosages. Numerous reviews have identified the protective effects of adequate selenium against methylmercury toxicity using a wide range of taxa (Cuvin-Aralar and Furness 1991, Chapman and Chan 2000, Yang et al. 2008, Peterson et al. 2009b). Further, failure to consider selenium levels in a few studies of human fish consumers has been attributed to a paradox wherein higher methylmercury exposure yielded less toxicological effects (Zhang et al. 2014a). In contrast, some instances are known where the toxic effects of methylmercury were not reduced or increased due to selenium (Chapman and Chan 2000). An additional concern is that some uncertainty exists regarding the protective effect of selenium during long term exposure to methylmercury (Bjørklund 2015).

Selenium has an enormous propensity to bind to mercury (Dyrssen and Wedborg 1991) potentially reducing mercury toxicity, but the exact dietary Se:Hg molar ratio that provides optimal protection has not been well quantified (Zhang et al. 2014a). Consequently, selenium levels have not been explicitly incorporated into mercury toxicity thresholds for wildlife, at least to our knowledge. Nevertheless, given that adequate selenium generally appears to reduce mercury toxicity, measuring both elements in the same food items provides additional guidance regarding mercury risk and may aid future investigations that explicitly incorporate selenium.

Our current investigation addressed the research question: *do lentic fishes from GNP have sufficiently high mercury levels to pose a potential hazard to piscivorous wildlife?* Our first objective was to evaluate this hazard using a low elevation lake in which we most thoroughly quantified fish mercury concentrations. Specifically, in Logging Lake we incorporated mercury levels in fishes and their relative abundances to assess the potential dietary hazard posed to selected piscivorous wildlife based on published thresholds for adverse effects. The second objective was to evaluate if geographic variation in mercury levels within fish species could alter our interpretation of risk based on Logging Lake. Accordingly, we compared fish mercury levels between Logging Lake and other lentic waters in GNP. Our third objective was to evaluate if inter-species variation in fish mercury levels could alter our interpretation of risk based on Logging Lake. Specifically, we identified species with elevated mercury levels at a given size and evaluated if their distribution and abundance in GNP were likely to create greater concerns for piscivorous wildlife than inferred from Logging Lake. Our final objective was to preliminarily evaluate if Se:Hg molar ratios in GNP were sufficiently high to potentially reduce the mercury toxicity to wildlife. Accordingly, we examined Se:Hg molar ratios in fishes from Logging Lake and, on the other (east) side of the Continental Divide, from Saint Mary Lake to broaden our inference.

Methods

Study Area

We conducted our investigation on seven lentic water bodies in GNP (Figure 2). We chose Logging Lake as our focal site as it was scheduled for routine gill net monitoring, providing an opportunity for intensive sampling without additional impacts to the fishery. Five sites (including Logging Lake) are at low elevations at or near the transition between the mountain valleys and the downstream lowlands, one is at high elevation in a headwater location (Hidden Lake), and one is transitional between these two landscape settings (Cosley Lake). We selected these water bodies to obtain a wide geographic coverage within GNP, with a focus on low elevations where piscivorous wildlife tends to be concentrated. Another relevant aspect of low elevation lakes is that piscivorous fishes (which tend to have elevated mercury levels) generally

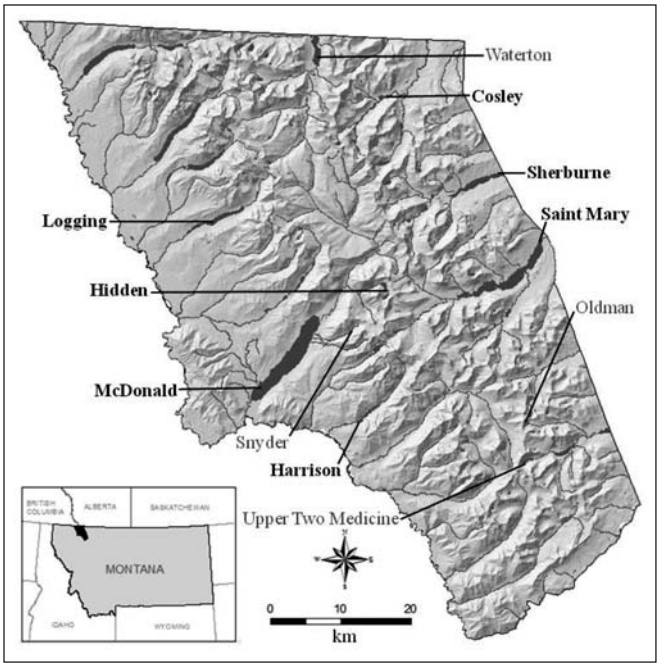

Figure 2. Glacier National Park water bodies sampled for fish in the current assessment. Water bodies in bold were sampled for this assessment, while those in grey represent sites used in previous investigations by others.

are present, contrasting with high elevation GNP lakes. Six of the study sites are typical of GNP lakes in that they are oligotrophic, relatively cold, and vary considerably in their sizes and maximum depths. Our remaining site, Sherburne Reservoir (completed in 1921), is also oligotrophic and relatively cold, but represents GNP's only true reservoir (Table 1). These sites contain moderately varied fish communities, with some overlapping species, reflecting the limited natural dispersal through the mountainous terrain and historical fish stocking in GNP and the nearby connected waters.

TABLE 1. Site, geographic location, surface area, maximum depth, and surface elevation for the lentic water bodies used in this assessment. Area, depth, and elevation for Sherburne Reservoir (the only reservoir site) are based on the spillway crest elevation although water levels usually are lower.

Site	Location (decimal degree)	Area (ha)	Depth (m)	Elev. (m)
Cosley	48.922, – 113.758	90	36	1479
Harrison	48.516, – 113.772	163	41	1126
Hidden	48.674, – 113.744	110	20	1944
Logging	48.756, – 114.076	451	60	1161
McDonald	48.569, – 113.939	2781	141	961
Saint Mary	48.687, – 113.513	1571	75	1367
Sherburne	48.817, – 113.578	696	21	1464

Fish Capture and Muscle Collection

We captured fish with variable mesh gill nets (primarily 38.1 x 1.8 m experimental multifilament nylon with five 7.6 m panels consisting of bar measured mesh of 19, 25, 32, 38, and 51 mm graded sequentially from small to large mesh), and additionally 15 cutthroat trout from Logging Lake were collected by angling in September 2010. Most nets were set on the bottom perpendicular to shore with the small mesh end inshore and deployed overnight, including in our focal water body, Logging Lake. Here we joined two nets at the large and small mesh ends of each net, and a total of 10 net pairs were set throughout the lake. Elsewhere we deployed fewer nets or net pairs, but still distributed them widely in each water body. The exception was Hidden Lake, our second smallest water body, where we only set nets in the vicinity of the outlet. In Logging Lake, we retained subsamples of all fishes captured ($n = 6$) for chemical analyses. Other fishes are known to occur in Logging Lake (Meeuwig et al. 2008) but apparently were too rare (bull trout) or small (redside shiner [*Richardsonius balteatus*] and sculpin [*Cottus spp.*]) to be captured in our nets. Elsewhere, we collected fewer fishes per water body (range = 1 to 5). The total sample we retained from all water bodies contained 10 fishes, and lake trout were the most represented species ($n = 5$ lakes). Fish collections were made in August and September of 2008 and 2010. Each water body was sampled at least once and, sometimes, twice to increase sample sizes. Details regarding the dates of fish capture, species collected, and their length ranges for each water body are provided in Table 2.

TABLE 2. Site, sampling date, fish species collected for mercury analysis, fish length (TL) range, number of individuals tested for mercury, muscle mercury concentration (Hg) versus total length (TL) relationship, associated r^2 and P values, and predicted muscle mercury concentration at 250 mm total length (PHg 250). If no significant muscle mercury versus total length relationship existed, the mean mercury concentration and total length values are presented, respectively, and the associated r^2 and P values are based on the exponential equation. Scientific names not provided in the text are as follows: coarsescale sucker (*Catostomus macrocheilus*), longnose sucker (*Catostomus catostomus*), and burbot (*Lota lota*).

Site	Date (mm/yr)	Species	TL range (mm)	n	Hg versus TL (mg kg^{-1}, mm)	r^2	P	PHg 250 (mg kg^{-1})
Cosley	8/2010	lake trout	205–848	16	$0.0145*e^{0.00492*TL}$	0.89	<0.001	0.050
Harrison	8/2008	lake trout	494–836	15	0.000585*TL+0.0211	0.27	0.046	-------
Hidden	8/2010	cutthroat trout	239–443	15	0.035, 350	0.02	0.631	0.035
Logging	8/2010,	coarsescale sucker	162–530	15	$0.0168*e^{0.00398*TL}$	0.80	<0.001	0.045
	9/2010	cutthroat trout	176–405	16	$0.00414*e^{0.00825*TL}$	0.94	<0.001	0.033
		lake trout	124–748	20	$0.0309*e^{0.00412*TL}$	0.82	<0.001	0.087
		longnose sucker	160–466	15	$0.0189*e^{0.00366*TL}$	0.84	<0.001	0.047
		mountain whitefish	166–387	15	$0.00984*e^{0.00748*TL}$	0.68	<0.001	0.064
		n. pikeminnow	152–316	15	$0.0227*e^{0.00848*TL}$	0.50	0.003	0.189
McDonald	8/2008	bull trout	431–564	5	0.158, 489	0.29	0.346	-------
		lake trout	327–729	12	$0.0414*e^{0.00372*TL}$	0.60	0.003	-------
		lake whitefish	428–535	10	0.132, 485	0.39	0.052	-------
Saint Mary	9/2008,	burbot	225–890	13	$0.00000157*TL^{1.88}$	0.69	<0.001	0.051
	9/2010	lake trout	363–735	19	0.316, 522	0.02	0.588	-------
		lake whitefish	397–524	16	– 0.000495*TL+0.368	0.34	0.018	-------
		longnose sucker	158–407	15	$0.0173*e^{0.00302*TL}$	0.48	0.004	0.037
		mountain whitefish	205–329	8	$0.00440*e^{0.00935*TL}$	0.80	0.003	0.046
Sherburne	9/2010	lake whitefish	322–377	10	0.152, 351	0.00	0.973	-------
		northern pike	190–996	8	$0.0760*e^{0.00258*TL}$	0.88	0.001	0.145
		longnose sucker	180–553	5	$0.0158*e^{0.00511*TL}$	0.93	0.009	0.057

We processed fish immediately after removal from the nets. Fish were weighed, measured for TL, and a piece of skinless dorsal muscle was removed and placed into a re-sealable plastic bag. Muscle samples were frozen in the field using dry ice and stored in a freezer at -18 °C until analysis.

Mercury and Selenium Analyses

Muscle was analyzed for total mercury and total selenium in the Environmental Biogeochemistry Laboratory at the University of Montana, Missoula. Fish from 2008 and 2010 were analyzed for mercury, and a subset of the 2010 fish from Logging Lake and Saint Mary Lake was used for selenium determination. We used total mercury as a surrogate for methylmercury because the methyl form predominates (typically 80–99%) in fish (Downs et al. 1998), consistent with previous observations from GNP (Watras et al. 1995). For all mercury and selenium determinations, the frozen samples were warmed at room temperature to near 0 °C. A piece then was cut from the center of each sample, weighed, and used for the analyses as wet tissue. Mercury in samples from 2010 was analyzed without digestion using a Milestone Inc. model DMA-80 Direct Mercury Analyzer following USEPA Method 7473. Approximately 0.25 g of tissue was used for each mercury determination, resulting in a practical quantitation limit of 10 µg kg^{-1}. For all other analyses we placed ~ 0.5 g of tissue into 50 mL digestion vials (Savillex 0202, made of perfluoroalkoxy and pre-cleaned as described by DeWild et al. [2002]) then added 3.3 mL of concentrated trace-metal grade HNO_3 and 2 mL of deionized water. These mixtures were left at room temperature for 30 min before refluxing at 80–85 °C for 4 h. After cooling, we added 1 mL of 30% concentrated ACS-grade H_2O_2 then heated the mixtures to 60–70 °C for 30 min. The mixtures were cooled again and then deionized

water was added to produce a total volume of 50 mL. The resulting digests were analyzed for total mercury using a Leeman Hydra AF cold vapor atomic fluorescence spectrometer using USEPA method 245.7, and the associated practical quantitation limit was 0.2 μg kg^{-1}. For selenium determination we used a Perkin-Elmer model Elan DRCe ICP-mass spectrometer following USEPA Method 6020, producing a practical quantitation limit of 50 μg kg^{-1}. Mass 82 (^{82}Se isotope) was used for quantitation without any collision/reaction cell gas as we did not detect any significant interference issues at the selenium concentrations present in our samples (> 0.5 μg L^{-1} in digest).

Quality control samples were within the limits required for the respective USEPA methods. We included at least one DORM-3 Standard Reference Material (Canadian Research Council) for each set of 20 samples. Mean mercury concentrations of DORM-3 reference material was 105% (SD = 13.7%, n = 5) of the certified value for the digestion/atomic fluorescence method and 107% (SD = 4.0%, n = 7) for the direct method. Mean measured selenium concentrations of DORM-3 reference samples were 98% (SD = 3.5%, n = 4) of the listed value. Duplicate analyses of separate muscle subsamples from the same fish yielded a mean difference of 11% (SD = 5.4%, n = 6) for mercury and 11% (SD = 4.3%, n = 4) for selenium.

Se:Hg ratios are reported as mole/mole and all mercury and selenium concentrations are reported on a wet weight basis. The evaporable content in our raw muscle samples averaged 79.6% (n = 26, range = 73.3 to 85.1%) based on a subsample of our 2008 collection. Data from species/lake combinations that had less than five fish were not included in this analysis (n = 2).

Hazard Assessment for Piscivorous Wildlife from Logging Lake

Our primary hazard assessment was conducted on Logging Lake where we estimated the mercury concentration in the diets of piscivorous wildlife. Empirical diet data from GNP was lacking for our selected wildlife species, but should bear some correspondence to the fish community composition. Thus we characterized diet by assuming fishes were consumed in proportion to their net catch biomasses from Logging Lake. In essence, these biomass proportions are equivalent to relative abundances that have been adjusted for the modest differences among fishes in their mass versus length relationships. Specifically, within each fish length category (150, 200, 250, and 300 mm), we multiplied the relative abundance of each fish species by their predicted body mass at the corresponding length. The resulting biomass calculations were summed for all six fishes, and the contribution of each fish species was expressed as a (biomass) proportion. Because the catch of smaller fish was low for three species in Logging Lake, we used the number caught from in the 150–300 mm range to calculate the biomass proportions for each of our four length categories. Next, we used muscle mercury concentration versus length relationships for each fish species to predict their mercury concentrations in each length category (Table 2). Mercury calculations for 150 mm required minor extrapolation (mean = 13 mm) for five fish species. For the Logging Lake hazard assessment only, muscle mercury concentrations were converted to (lower) whole fish concentrations using Peterson et al.'s (2005) formula ($\log_{10}$ [whole-body Hg] = 0.9005 $\log_{10}$ [muscle Hg] − 0.2712) and the results were expressed arithmetically. The final step was to weight the whole body mercury concentrations by the biomass proportions. Specifically, for each species/length category combination, the biomass proportion was multiplied by the corresponding whole body mercury concentration. Within each length category, these biomass weighted mercury concentrations were then summed for all fish species to estimate the average dietary mercury concentration associated with fish consumption. As a counterpoint to this community based assessment, we examined a scenario wherein wildlife consumed only the most contaminated fish species present in Logging Lake. The goal here was to place an upper boundary on dietary mercury exposure caused by wildlife selectivity for fish species.

We evaluated the risk to piscivorous wildlife in Logging Lake by comparing our estimated mercury concentration associated with fish con-

TABLE 3. Piscivorous wildlife species used in this assessment, their toxicological thresholds based on dietary mercury concentration (wet weight), and the fish total length (TL) used to characterize their diet. The threshold for loon is based on the lowest observed adverse effect level for impaired behavior (Depew et al. 2012) while all others are protective wildlife values (Lazorchak et al. 2003). The fish lengths consumed by wildlife were based on literature sources (see text).

Species	Hg dietary threshold (mg kg^{-1})	Diet fish TL (mm)
kingfisher	0.03	80
loon	0.10	150
mink	0.07	150
otter	0.10	300

sumption at a given length to dietary thresholds for adverse effects (Table 3). We selected four wildlife piscivores that occur widely in GNP: belted kingfisher (*Megaceryle alcyon*), common loon, American mink, and river otter. These species can have mercury concentrations of toxicological concern in North America (Lazorchak et al. 2003, Scheuhammer et al. 2007) and are primarily piscivorous (Sample and Suter 1999, USEPA 1993, Gingras and Paszkowski 2006). We characterized the average fish sizes consumed for each species based on literature sources: 80 mm TL for kingfisher (USEPA 1993), 150 mm TL for loon (Barr 1996), 150 mm TL for mink (Sample and Suter 1999), and 300 mm TL for otter (Sample and Suter 1999). For dietary thresholds, we primarily used the mercury wildlife values of Lazorchak et al. (2003) to help provide a consistent basis for evaluating risk among species. These values are protective thresholds "expected to protect the viability of piscivorous wildlife", and were based on species-specific test doses that were converted into mercury mass ingested per body mass with the assumption that diet composition is 100% fish. Lazorchak et al.'s approach follows that of the USEPA (1995) except uptake of mercury from the water was assumed to be nonexistent and the exposure component was expressed as mercury concentration in fish. As Lazorchak et al. (2003) did not assess loon, we also included Depew et al. (2012) who derived three benchmarks of fish mercury concentrations based on an extensive literature survey, and these benchmarks are equivalent to the lowest observed adverse effect level (Depew et al. 2013). We use the lowest of these benchmarks, impaired behavior.

Geographic Assessment of Fish Mercury Levels in GNP

To evaluate how our primary hazard assessment from Logging Lake could be influenced by intra-species variation in fish mercury levels, we compared muscle mercury levels in fishes from Logging Lake to our other study lakes. These geographic comparisons were matched by species and length, and based on predicted mercury at 250 mm using the equations in Table 2. However, because the most widely tested species (lake trout) often lacked smaller fish in our collections, we also included 400 and 550 mm categories (which are less relevant for most piscivorous wildlife in GNP). Each comparison was made by subtracting the predicted Logging Lake mercury concentration from that of the other lake, and within each of the three length categories these differences were expressed as a mean.

To expand the scope of our geographic assessment, we separately examined fish mercury data (all were total mercury based on wet weight) from other studies in GNP and its adjacent lakes (locations provided in Figure 2). Brinkmann (2007) reported a muscle mercury versus fork length relationship in lake trout ($n = 29$, collected in 2005) for Waterton Lake (elevation = 1280 m). Brinkmann used a factor of 1.09 to convert lake trout fork length to TL, thus a fork length of 229 mm is equivalent to 250 mm TL. At this size the predicted mercury concentration is 0.176 mg kg^{-1}. Muscle mercury data for lake trout data from Upper Two Medicine Lake (elevation = 1647 m) were collected by the USEPA (2009). Fish ranging from 371 to 406 mm TL (average = 387 mm, $n = 5$) were captured in September of 2003 and analyzed as a composite which yielded 0.136 mg kg^{-1} (raw data provided by USEPA). Mercury levels in whole cutthroat trout were reported by Landers et al. (2008) and Schwindt et al. (2008) for Oldman Lake (elevation = 2026 m) and Snyder Lake

(elevation = 1600 m) based on collections from 2005. Fish in both lakes averaged 0.037 mg kg^{-1}, while fork length averaged 382 mm in Oldman Lake (n = 10) and 181 mm in Snyder Lake (n = 15). Whole fish mercury data were converted to muscle values using Peterson et al. (2007), resulting in 0.054 mg kg^{-1}. We used a value of 1.023 to convert cutthroat trout fork length to TL, rendering an average TL of 185 mm for Snyder Lake and 391 mm for Oldman Lake. To compare the fish mercury concentrations from other studies to our Logging Lake data, we subtracted the predicted Logging Lake concentration (Table 2) from that of the other lakes using species/length matched comparisons. Because the use of data collected and summarized by other studies created some uncertainties, we opted to keep this inquiry separate from the geographic assessment component that used only our fish mercury data.

Inter Species Assessment of Fish Mercury Levels in GNP

Even if mercury values within a fish species vary little geographically, wildlife risk may still differ from Logging Lake due to differences in species composition across GNP. In this context, high mercury species are of particular concern. Accordingly, we identified fishes with elevated mercury risk by making comparisons with lake trout at 250 mm, but also used larger lengths if data from smaller fish were unavailable. We then qualitatively evaluated if the abundance and geographic distribution of these high mercury species could alter our interpretation of hazard to piscivorous wildlife based on Logging Lake.

Assessment of Fish Se:Hg Molar Ratios in Two GNP Lakes

To help guide our hazard assessment, we evaluated if selenium levels in GNP fishes from two lakes were adequate to potentially reduce the toxic effects of methylmercury. In general, a molar excess of selenium relative to mercury (i.e. Se:Hg molar ratio > 1) has been interpreted as sufficiently high to reduce methylmercury risk (Peterson et al. 2009b). Accordingly, we used our data from Logging Lake and Saint Marys Lake to calculate Se:Hg molar ratios and the associated regressions with fish length within each lake. The ratios are based on our muscle data rather than calculated whole body ratios. This approach was necessary because we were unable to locate well developed equations to estimate whole body selenium levels from muscle data that were based on either a broad range of fishes or our specific study species, particularly at or near the muscle selenium concentrations we observed (mean = 0.297 mg kg^{-1} wet weight).

Statistical Analysis

All statistical analyses were conducted using SPSS version 22. For fish mercury versus length relationships, our primary interest was to estimate fish mercury concentrations at a given body size. Accordingly, within each species/water body combination we based model selection on the ability to produce minimized, unbiased residuals over the range of observed lengths. This goal was generally achieved using an exponential model, but the use of linear (n =3) and power (n = 1) functions was required in four instances. If no significant (P = 0.05) mercury versus length relationship existed, we present the mean mercury value and the reported r^2 and p values were based on the exponential model. For the Se:Hg molar ratio versus length relationships, we $\log_{10}$ transformed Se:Hg molar ratios because of strong heteroscedasticity.

Results

Hazard Assessment for Piscivorous Wildlife from Logging Lake

In Logging Lake, the ~ 150–300 mm fish most relevant for our wildlife assessment generally exhibited moderate variation among species in their muscle mercury concentrations. The most notable exception was the markedly higher levels in northern pikeminnow, especially as length increased. Lake trout was the next most contaminated species, while cutthroat trout had the lowest mercury levels—particularly at smaller lengths (Figure 3).

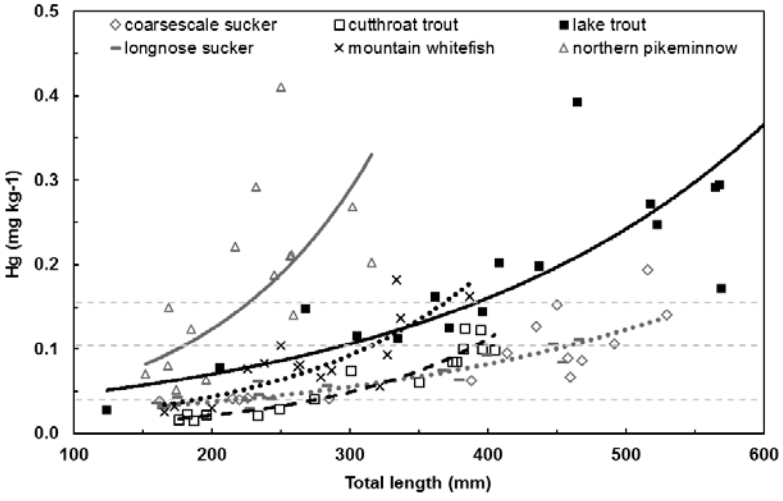

Figure 3. Muscle mercury concentration versus length for fishes collected in Logging Lake. Dashed horizontal lines represent the wildlife dietary thresholds for mercury (Table 3) after conversion from whole fish concentration to muscle concentration using Peterson et al. (2007) and are supplied for visual reference only. Horizontal line key (parenthesis denote typical fish length consumed): bottom = kingfisher (80 mm), middle = mink (150 mm), and top = loon (150 mm) and otter (300 mm). Trend line key: cutthroat trout = black dashes, lake trout = black solid, mountain whitefish = black dots, northern pikeminnow = gray solid, and gray dots represent the nearly identical lines for both sucker species. See Table 2 for the fish mercury versus length equations. Four larger lake trout are not shown to aid visualization of the remaining data, see Figure 4 for these data.

The risks to wildlife from the comparatively high mercury levels in northern pikeminnow were offset substantially in the wildlife hazard assessment by the other less contaminated fishes, and the conversion of muscle to whole body mercury reduced concentrations for all species (Table 4). Northern pikeminnow was the second most common fish species (32.9% of catch); however, the remaining species with their lower contamination substantially reduced the estimated mercury concentration in wildlife diets. Mountain whitefish (*Prosopium williamsoni*) played a particularly important role in this reduction both due to their high abundance (57.6% of catch) and moderate mercury levels within our Logging Lake collection.

The GNP wildlife piscivores did not exceed their dietary mercury thresholds based on our fish community based assessment from Logging Lake (Tables 3 and 4). Loon and mink were substantially below their thresholds in the 150 mm fish category, while at 300 mm the threshold for otter was nearly reached. Kingfisher consume fishes smaller than those in our Logging Lake collection. However, based on the calculated fish mercury concentration of 0.035 mg kg^{-1} in the 150 mm category and the diminishing mercury levels with fish size, it seems unlikely that their 0.03 mg kg^{-1} threshold was exceeded at 80 mm.

If wildlife consumed exclusively northern pikeminnow, the most contaminated fish at a given length in Logging Lake, dietary mercury exposure would increase by 60% at 150 mm and 79% at 300 mm. Under this scenario loon and mink still did not exceed their dietary mercury thresholds. However, the otter threshold of 0.10 mg kg^{-1} was surpassed by the northern pikeminnow value of 0.175 mg kg^{-1} at 300 mm. For kingfisher, the prognosis is uncertain. Given the diminishing northern pikeminnow mercury concentrations with size, a plausible result is that the kingfisher threshold was slightly exceeded, i.e. by < 0.01 mg kg^{-1} (Tables 3 and 4).

Geographic Assessment of Fish Mercury Levels in GNP

Geographic comparisons of muscle mercury levels matched by species and length show that, based on our collections, Logging Lake fishes had moderately higher contamination in the smaller length category most relevant for wildlife (Table 5). At 250 mm, fishes from Logging Lake averaged 0.016 mg kg^{-1} (37.5%) higher than in our other lakes. Comparisons based on 400 mm and 550 mm indicate that mercury levels were more typical in Logging Lake, and individual comparisons tended to show larger deviations than at 250 mm. The largest deviations at 400 mm were lake trout from Saint Mary Lake and cutthroat trout from Hidden Lake, both of which lacked significant mercury versus length relationships (Table 2). At

TABLE 4. Fish species captured in Logging Lake, total number caught in the 150–300 mm total length (TL) range, and predicted whole body mercury concentrations (mg kg^{-1}) as well as biomass proportions (prop.; predicted body mass of a species at a given length * number caught then expressed as proportion relative to all fishes in the same length category; in essence, relative abundance that has been adjusted for modest differences among species in their mass at a given length) in four fish length categories. At the bottom is the estimated dietary mercury concentration (mg kg^{-1}) of piscivorous wildlife associated with each fish length category. These concentrations were calculated by multiplication of the whole body mercury concentration and the associated proportional mass contribution for each species which was then summed within each length category.

Species	n	150 mm TL mg kg^{-1}	prop.	200 mm TL mg kg^{-1}	prop.	250 mm TL mg kg^{-1}	prop.	300 mm TL mg kg^{-1}	prop.
coarsescale sucker	5	0.023	0.016	0.028	0.016	0.033	0.016	0.040	0.015
cutthroat trout	2	0.012	0.006	0.017	0.006	0.025	0.006	0.036	0.006
lake trout	6	0.041	0.012	0.049	0.012	0.059	0.012	0.071	0.012
longnose sucker	20	0.025	0.066	0.029	0.064	0.034	0.062	0.040	0.061
n. pikeminnow	114	0.056	0.348	0.082	0.341	0.120	0.336	0.175	0.332
mountain whitefish	200	0.023	0.553	0.032	0.561	0.045	0.568	0.063	0.574
dietary Hg (mg kg^{-1})		0.035		0.049		0.069		0.098	

TABLE 5. Fish species, site, and species matched difference (dif.) in predicted muscle mercury concentration relative to Logging Lake (site concentration minus Logging Lake concentration; mg kg^{-1}) in three total length categories for species/lake combinations used in the geographic assessment of mercury contamination within Glacier National Park using only our data.

Species	Site	dif. 250 mm (mg kg^{-1})	dif. 400 mm (mg kg^{-1})	dif. 550 mm (mg kg^{-1})
cutthroat trout	Hidden	0.002	– 0.077	
lake trout	Cosley	– 0.037	– 0.057	– 0.081
lake trout	Harrison			0.045
lake trout	McDonald		0.023	0.022
lake trout	Saint Mary		0.155	0.018
longnose sucker	Saint Mary	– 0.010	– 0.024	
mountain whitefish	Saint Mary	– 0.018		
mean of differences		– 0.016	0.004	0.001

550 mm, the largest deviation was from Cosley Lake lake trout. The moderate geographic variation in fish mercury levels relative to Logging Lake was coherent with the inter lake patterns in our most widely tested species, lake trout (n =5 sites, Figure 4).

Expanding the comparison of mercury levels in Logging Lake to lakes tested by others shows that Logging Lake fishes were more contaminated in two of four instances. After conversion of whole fish mercury to muscle, cutthroat trout at 391 mm were 0.050 mg kg^{-1} lower in Oldman Lake, while at 185 mm values were 0.035 mg kg^{-1} higher in Snyder Lake. Relative to muscle mercury values from Logging Lake, lake trout at 387 mm were 0.016 mg kg^{-1} lower in Upper Two Medicine Lake, and at 250 mm were 0.090 mg kg^{-1} higher in Waterton Lake. We note that the predicted mercury concentration in Waterton Lake lake trout (0.176 mg kg^{-1} at 250 mm) is elevated relative to the lake trout values we found among our study sites (Figure 4).

Inter Species Assessment of Fish Mercury Levels in GNP

Comparisons of all sites showed that at a fish length of 250 mm predicted mercury concentrations were highest in northern pikeminnow

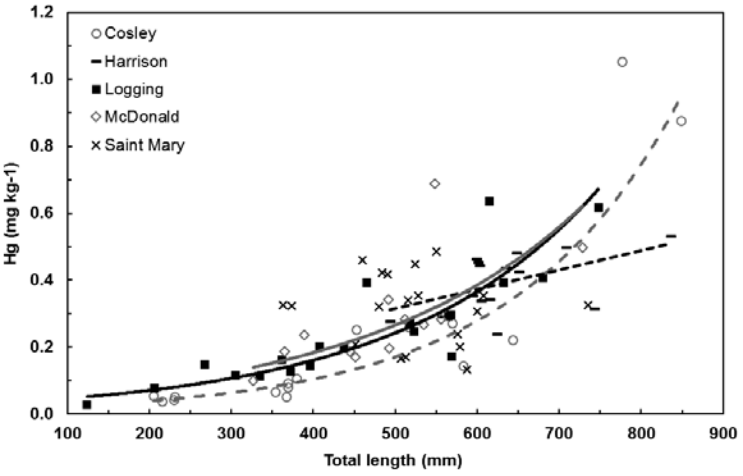

Figure 4. Muscle mercury concentration versus length for lake trout from five lakes in Glacier National Park. Trend line key: Cosley Lake = gray dashes, Harrison Lake = black dashes, Logging Lake = black solid, Lake McDonald = gray solid, and Saint Mary Lake showed no significant relationship. See Table 2 for the associated equation information.

An allopatric comparison was used to evaluate mercury levels in lake whitefish from Sherburne Reservoir, as no lake trout were present. Lake whitefish in Sherburne Reservoir had 0.152 mg kg^{-1} mercury at 351 mm, higher than the corresponding 0.117 mg kg^{-1} average for lake trout based Cosley Lake, Logging Lake, and Lake McDonald (Table 2). An additional basis for elevated mercury levels is that lake whitefish in Sherburne Reservoir were more contaminated at a given length than those from Waterton Lake (Brinkmann 2007).

(Logging Lake) at 0.189 mg kg^{-1} followed by northern pike (*Esox lucius*, Sherburne Reservoir) at 0.145 mg kg^{-1}. Appreciably lower was the third most contaminated species, lake trout at 0.069 mg kg^{-1}, based on the average of Logging Lake and Cosley Lake (Table 2).

Based on length matched comparisons with sympatric lake trout in and near GNP, we found no evidence that our collections lacking fish 250 mm or less had higher mercury levels than those previously identified. In Lake McDonald, the mercury concentration at 489 mm for bull trout was 0.158 mg kg^{-1} versus 0.255 mg kg^{-1} for lake trout (Table 2). Data from Swan Lake, Montana (70 km south of Lake McDonald) show that these two species had similar mercury levels in the 560–808 mm TL range (Leo Rosenthal, Montana Fish, Wildlife, & Parks, personal communication). Sympatric comparisons of lake whitefish reveal lower mercury levels relative to lake trout in Lake McDonald and Saint Mary Lake, but at sizes substantially greater than 250 mm (Table 2). Using a wider range of sizes, studies from Flathead Lake, Montana (75 km south of Lake McDonald) and Waterton Lake found that lake whitefish were less contaminated than sympatric lake trout, and the former study included fishes < 250 mm TL (Stafford et al. 2004, Brinkmann 2007).

Assessment of Fish Se:Hg Molar Ratios in Two GNP Lakes

Fish Se:Hg molar ratios in muscle diminished with increasing length in both lakes but almost always remained substantially greater than 1:1 (Figure 5). In general, lower ratios were present in Logging Lake than Saint Mary Lake. Only in two large lake trout from Logging Lake did the molar ratios drop below 1:1. Among the smaller fish in our collections, only northern pikeminnow had Se:Hg molar ratios near 1:1 (mean = 1.16, $n = 3$).

Discussion

Hazard Assessment for Piscivorous Wildlife from Logging Lake

The hazard assessment from Logging Lake suggests that the piscivorous wildlife we evaluated was not at high risk for mercury toxicity. Loon and mink were substantially below their dietary mercury thresholds based on fishes at 150 mm. For mink in particular, their actual mercury exposure likely is lower than the other piscivores due to greater consumption of less contaminated terrestrial food items (Sample and Suter 1999, USEPA 1993, Gingras and Paszkowski 2006). Otter were just below their mercury threshold at a fish size of 300 mm, largely owing to their

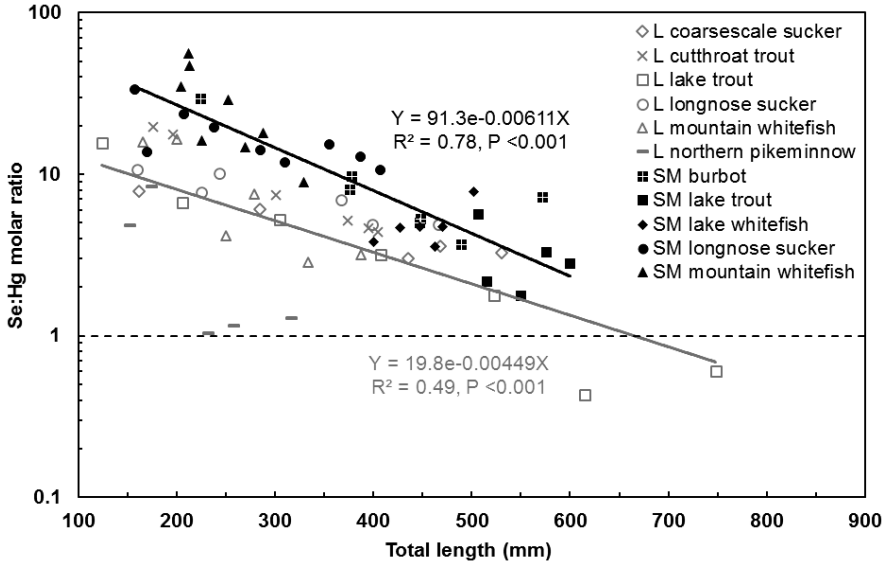

Figure 5. Muscle Se:Hg molar ratio versus length for fishes from Logging Lake (L, gray line) and Saint Mary Lake (SM, black line). The dashed horizontal line denotes a molar ratio of 1:1.

preferences for these bigger, more contaminated fish. For kingfisher, our findings are somewhat uncertain as we assumed fish mercury concentrations continued to diminish from 150 to 80 mm used to characterize their diet. Further, we did not collect redside shiner and sculpin and these small fishes could be consumed by kingfisher. Other studies have shown redside shiner has typical mercury levels at a given size relative to other sympatric fishes (Lowe et al. 1985), while sculpins tend to have moderately elevated levels (Power et al. 2002, McIntyre and Beauchamp 2007). The omission of redside shiner and sculpin, however, did not overtly affect our findings for the other wildlife species as the maximum sizes of these fishes are less than 150 mm TL in our study region (Brown 1971).

A relevant aspect of our hazard assessment from Logging Lake is that we did not have empirical diet data for our wildlife piscivores, but rather assumed that fishes were consumed in proportion to their relative abundance in the gill nets. A particular concern is if wildlife select for high mercury fishes (e.g. northern pikeminnow, and to a lesser extent lake trout). This may be true for northern pikeminnow, particularly given their preference for shallow water, but unlikely for lake trout due to preferences for deep water (Elrod and Schneider 1987). Conversely, our approach probably underestimated the actual intake of cutthroat trout as the sinking nets we used do not effectively capture these surface oriented fish, potentially overestimating risk as cutthroat trout had the lowest mercury values in Logging Lake. The various biases associated with each fish species create uncertainties in our Logging Lake hazard assessment. However, when no local wildlife diet are available, we believe our approach represents an improvement compared to studies that make no consideration regarding the relative abundance of prey fishes or their consumed sizes.

Our scenario where wildlife consume only northern pikeminnow in Logging Lake puts an upper bracket on dietary mercury exposure caused by feeding selectively on fish species. Even with the associated increase in dietary mercury concentrations, loon and mink would still not exceed their thresholds. In contrast, thresholds for otter would be appreciably surpassed (0.075 mg kg^{-1}) and that of kingfisher probably would be exceeded slightly. The emergent pattern is that strong selectivity for northern pikeminnow could alter our interpretation of dietary mercury hazard

to wildlife, and the concern under this scenario particularly is for otter.

Geographic Assessment of Fish Mercury Levels in GNP

Within our study sites, the geographic comparisons show that Logging Lake had moderately elevated fish mercury levels in the size category most relevant for wildlife. At 250 mm, the within species comparisons show that mercury levels averaged 37.5% higher in Logging Lake relative to our other sites. Accordingly, this component of the geographic assessment suggests that Logging Lake tended to provide a protective basis for evaluating risk to piscivorous wildlife among our study lakes.

By geographically comparing our fish mercury data from Logging Lake to other investigations, we expanded our inference and identified situations where contamination appears elevated within GNP. Lake trout at 387 mm from Upper Two Medicine Lake and cutthroat trout at 391 mm from Oldman Lake were both lower than Logging Lake, coherent with the general patterns using only our data. In contrast, lake trout from Waterton Lake were substantially more contaminated at 250 mm than Logging Lake. We attribute this finding to the presence of *Mysis diluviana* which increases food chain length and thus elevates lake trout mercury levels (Cabana and Rasmussen 1994). However, the results from Waterton Lake appear to be idiosyncratic for our assessment as no other GNP waters are known to contain Mysis. The higher cutthroat trout mercury levels in Snyder Lake may be part of a broader pattern not evident in our data. Eagles-Smith et al. (2016) found that salmonid mercury concentrations were higher when subalpine lakes (primarily in northeastern Oregon) were smaller and contained larger conifer basal areas in their watersheds. Snyder Lake is 2.6 ha, considerably smaller than any of our study lakes, but inspection of satellite imagery shows that conifer basal area is not high. Thus small, subalpine GNP lakes with a high conifer basal area may contain fish with mercury levels that are even greater at a given length than Snyder Lake. However, subalpine and alpine lakes containing fish in GNP almost exclusively hold non-piscivorous salmonids with a limited propensity to accumulate mercury, presumably reducing the exposure to piscivorous wildlife.

Inter Species Assessment of Fish Mercury Levels in GNP

The inter-species comparisons revealed that northern pikeminnow was the most contaminated species we tested at a given length, and unlike most other waters in GNP, this species comprised a large portion of Logging Lake's fish community. Northern pikeminnow are only known to occur in a few other lakes in the Pacific Drainage of GNP (i.e. Rogers Lake, Lake McDonald, and Harrison Lake) where they comprise a similar or smaller portion of the fish community relative to Logging Lake (Schultz 1941, Meeuwig et al. 2008). Thus it is possible that other GNP waters have fish mercury levels of concern for piscivorous wildlife due to the presence of northern pikeminnow, but this issue is of limited geographic scope in GNP.

A scenario that could alter our assessment of low wildlife risk is if other waters contain high mercury fishes not present in Logging Lake, particularly if they are abundant. However, comparisons at 250 mm show that northern pike was the only high mercury species we tested that was absent from Logging Lake. Further, within GNP northern pike are only known to occur in Sherburne Reservoir in any substantial abundance (Shultz 1941, C. Downs, unpublished gill net data). For fishes that lacked smaller individuals in our collections, it does not appear that bull trout or lake whitefish pose a major mercury risk to wildlife in GNP based on length matched comparisons with lake trout. An exception was that some elevated risk may exist for lake whitefish in Sherburne Reservoir. These findings, and those for the other fishes collected at this site, may have been influenced by features of the reservoir environment. For example, other studies have shown that widely fluctuating water levels can increase fish mercury levels (Evers et al. 2007), a hydrological pattern present in Sherburne Reservoir.

We were unable to test all GNP fishes for mercury, so a concern is that untested fishes that are high in mercury could alter our interpretation of wildlife risk based on Logging Lake. In general, fish species that feed at a high trophic

position or exhibit preferences for warmer water have elevated mercury concentrations (Stafford and Haines 1997, Peterson et al. 2009a). Based on these criteria, obvious candidates for untested, high mercury fishes are lacking in GNP. Another scenario associated with elevated mercury levels is when fishes mature at a small size and subsequently reduce body growth (Hammar et al. 1993, Christensen et al. 2006). For example, longnose dace (*Rhinichthys cataractae*) is a small invertebrate feeder but can have relatively high mercury levels for its size (Christensen et al. 2006). However, Brinkmann and Rasmussen (2012) reported muscle mercury concentrations for longnose dace (mean fork length = 52 mm, = ~ 56 mm TL) in the nearby Oldman River of southwestern Alberta. Mercury concentrations in the colder, upstream waters were ~ 0.02 mg kg^{-1} (wet weight), providing an initial indication that longnose dace is not a high mercury species in the cold waters of GNP. Further, longnose dace is known to occur only in a few lower elevation waters on the east side of GNP and rarely exceeds 102 mm TL in Montana (Schultz 1941, Brown 1971). A few other GNP fishes have a similar life history pattern, but also have a limited distribution or do not reach large enough sizes (Schultz 1941, Brown 1971) to be likely candidates for high mercury levels. Nevertheless, small fishes that mature early may pose some hazard to piscivorous wildlife in GNP, particularly mercury sensitive wildlife that consumes smaller prey items like kingfisher.

Assessment of Fish Se:Hg Molar Ratios in Two GNP Lakes

Our interpretation that the piscivorous wildlife we examined is not imperiled by mercury in our study sites is further complimented by the fish Se:Hg molar ratios in Logging Lake and Saint Mary Lake. Ratios > 1:1 indicate a molar excess of selenium relative to mercury, and have been interpreted to reduce the toxic effects of methylmercury (Peterson et al. 2009b). In the size ranges typically consumed by wildlife, the Se:Hg molar ratio of 1:1 was far exceeded in all fishes from both lakes with the exception of northern pikeminnow—similar to stream dwelling fishes collected across the western US (Peterson et al. 2009a). Given that northern pikeminnow both have elevated mercury levels at a given size and molar ratios that were not appreciably greater than 1:1, this species may be of particular concern at the few GNP sites where it is present. Overall, however, our findings from Logging Lake and Saint Marys lakes provide initial evidence that adequate selenium generally exists in smaller GNP fish to potentially reduce mercury risk to piscivorous wildlife.

Potential Differences in Fish Mercury Levels with Flow and Elevation

Given the diversity of waters in GNP, relevant considerations include the role of flow (lentic versus lotic) and elevation in influencing fish mercury levels. Differences in fish mercury concentrations associated with lentic and lotic habitats do not appear to be a major limitation for our study as Kamman et al. (2005) reported similar contamination between these habitats. Further, Evers et al. (2005) found no significant differences in bald eagle (*Haliaeetus leucocephalus*) or kingfisher mercury levels between rivers and lakes. Elevation is another geographic consideration which could be particularly relevant the mountainous terrain of GNP. Zhang et al. (2014b) found no relationship in the Tibetan Plateau and also noted that other studies in mountainous terrain have found varied patterns. Regionally, Eagles-Smith et al. (2016) reported that elevation is not a major predictor of salmonid mercury levels in subalpine lakes (mostly located in northeast Oregon), which seems coherent with our findings for fish sizes typically consumed by GNP wildlife. Specifically, comparisons of our highest and lowest elevation lakes for cutthroat trout (Hidden Lake and Logging Lake, respectively) and lake trout (Cosley Lake and Logging Lake, respectively) at 250 mm reveal similar contamination for cutthroat trout and moderately lower levels for lake trout at the higher elevation site (Table 5).

Wildlife Mercury Hazard in GNP Relative to Other Areas

Our assessment of mercury hazard to piscivorous wildlife in GNP contrasts to some other areas of North America, such as the northeastern US and

southeastern Canada, and is coherent with geographic differences in fish mercury levels. Among fishes, a relevant consideration is the scarcity of warm water species in GNP, particularly piscivores which tend to have elevated contamination (Stafford and Haines 1997). Within species, mercury values for lake trout in GNP overall were similar to those reported for Flathead Lake and Swan Lake (although GNP values were lower at small lengths and increased more rapidly with body size) (Stafford et al. 2004, L. Rosenthal, personal communication), but were less than half the values reported for northeastern North America (Stafford and Haines 1997, Kamman et al. 2005). For yellow perch (*Perca flavescens*), Depew et al. (2013) found that modeled mercury concentrations increased from west to east in Canada. Consistent with this geographic pattern, Evers et al. (2003) reported that mercury concentrations of in loon eggs increased from west to east in northern North America, and specifically that contamination in Montana ($n = 5$) was relatively low.

An emergent aspect of our research is that methylmercury concentrations in water may provide limited insight regarding regional differences in fish contamination. Although GNP receives a relatively large amount of total mercury from precipitation, local environmental conditions lead to low concentrations of waterborne methylmercury in GNP lakes compared to Wisconsin, New York, and other western national parks (Watras et al. 1995, Krabbenhoft et al. 2002). However, the methylmercury present in the water of GNP lakes water appears to strongly biomagnify. For example, Watras et al. (1995) showed that waterborne concentrations of methylmercury were approximately four times higher in Wisconsin Lakes compared to those in GNP. In contrast, comparisons of lake trout and lake whitefish indicate moderately elevated concentrations of mercury exist in GNP relative to a state-wide summary from Wisconsin that used primarily skin-on fillets (Schrank 2014). Comparisons using only skin-on fillets collected from 2001 to 2015 also reveal higher mercury levels in the GNP fishes at a given length (raw data provided by Candy Schrank, Wisconsin Department of Natural Resources), regardless of the Wisconsin water body type and the slightly lower mercury values in skin-on relative to skin-off fillets (Zhang et al. 2013). Potential reasons for the high biomagnification of methylmercury in GNP lakes include the slow rates of fish growth (Dux 2005, Simoneau et al. 2005) and increased biomagnification under oligotrophic conditions (Driscoll et al. 2007, Clayden et al. 2013).

Effects of Multiple Contaminants in GNP

An unaddressed concern is the possible additive or synergistic effects of mercury with other environmental toxicants. Mercury is only one of several airborne toxicants of concern that have been documented in GNP, yet testing of these other contaminants has been limited in GNP fishes. Studies of two small headwater lakes in GNP have shown the presence of a wide variety of semi-volatile organic compounds (SOCs) of concern in cutthroat trout. Overall, these concentrations were high in Snyder Lake relative to lakes in other national parks of the western US, while those from Oldman Lake were more typical (Ackerman et al. 2008, Landers et al. 2008). It is likely that SOCs of concern are higher for fatty, piscivorous species in GNP such as larger lake trout, but to our knowledge only one composite sample of lake trout (from Upper Two Medicine Lake, average fish TL = 387 mm) has been tested recently (USEPA 2009). Further, lakes with substantial upstream glaciers (unlike Snyder Lake and Oldman Lake) are probably receiving elevated concentrations of historically used SOCs such as PCBs and DDT from glacial melt water, as glacial retreat has been shown to accelerate this process in other locations (Bogdal et al. 2009). This issue seems particularly relevant in GNP where the remaining glaciers are melting at a rapid rate (Hall and Fagre 2003). Waterton Lake may be additionally of concern due to the increased biomagnification of biologically persistent contaminants associated with Mysis (Rasmussen et al 1990, Cabana and Rasmussen 1994)

Suggestions for Future Mercury and SOC Testing of Lentic Fishes in GNP

Given that this investigation was intended as an initial screening of mercury hazard to piscivorous

wildlife in the lentic waters of GNP, we do not consider our findings exhaustive. However, our work does help identify areas of uncertainty or concern, providing guidance for any future mercury investigations in GNP. Specifically, we suggest mercury testing of small, early maturing fishes and (the limited) populations of northern pikeminnow. Elevated mercury levels are evident in Waterton Lake lake trout and thus may be present in other sympatric fishes, particularly those that are part of the Mysis food chain. Given our observations and those from other studies, more data are needed to quantify the mercury patterns in fishes from Sherburne Reservoir and very small GNP lakes. We also identify the need for additional SOCs testing in GNP, particularly fatty and piscivorous species such as lake trout in watersheds containing glaciers, so that piscivorous wildlife risk can be more holistically evaluated.

Conclusions

Based on this initial screening, GNP lentic fishes generally do not appear to pose a major mercury risk to piscivorous wildlife. Our hazard assessment from Logging Lake suggests that fish mercury levels were insufficient to imperil the piscivorous wildlife we evaluated. However, greater risk may exist if wildlife (particularly otter) exhibits a high selectivity for northern pikeminnow, the most contaminated GNP fish we found at a given length. Logging Lake seems to provide a relatively protective benchmark among our study lakes based on geographic comparisons of mercury levels within fish species, the high relative abundance of northern pikeminnow, and the limited distribution and abundance of high mercury fishes we identified in other GNP lentic waters. Further, our findings from two GNP lakes suggest that mercury risk to piscivorous wildlife may be reduced by the presence of adequate selenium in fishes, particularly smaller individuals. Although untested fishes that are high in mercury and/or site specific conditions that elevate fish mercury levels likely create situations in GNP where wildlife risk is substantially greater than in Logging Lake, our investigations suggest this will be uncommon. Our assessment from GNP generally contrasts to those from the northeastern US and southeastern Canada where mercury risk to piscivorous wildlife is of greater concern. Low methylmercury concentrations in the water of GNP lakes presumably help reduce fish mercury levels, but appear to be offset substantially by high biomagnification rates. Accordingly, the more important consideration for explaining the limited mercury risk to piscivorous wildlife may be the scarcity of warm water fishes in GNP, particularly piscivorous varieties that tend to have elevated mercury levels.

This research should not be misconstrued to indicate that consumption of GNP fishes poses no mercury risk to humans as many of our specimens (particularly larger piscivores) substantially exceed the USEPA's human health criterion of 0.30 mg kg^{-1}. Further, any protective effect of selenium to human consumers may be less evident in larger GNP fishes as Se:Hg molar ratios diminished with length.

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