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Mercury in Fishes from 21 National Parks in the Western United States—Inter- and Intra-Park Variation in Concentrations and Ecological Risk

Open-File Report 2014–1051

U.S. Department of the Interior
U.S. Geological Survey

Cover: Reflection Lake at Mount Rainier National Park. Photograph taken by Collin A. Eagles-Smith, U.S. Geological Survey, July 31, 2012.

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Conversion Factors

SI to Inch/Pound

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
kilometer (km)	0.6214	mile (mi)
	Area	
acre (ac)	0.0015625	square mile (mi^2)
	Mass	
gram (g)	0.03527	ounce, avoirdupois (oz)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F=(1.8\times°C)+32$$

Abbreviations and Acronyms

dw	dry weight
ww	wet weight
Hg	mercury
THg	total mercury
MeHg	methylmercury
LSM	least-square mean
NP	national park
NPS	National Park Service
CARE	Capitol Reef National Park
CRLA	Crater Lake National Park
DENA	Denali National Park
GLBA	Glacier Bay National Park
GLAC	Glacier National Park
GRCA	Grand Canyon National Park
GRTE	Grand Teton National Park
GRBA	Great Basin National Park
GRSA	Great Sand Dunes National Park
LACL	Lake Clark National Park
LAVO	Lassen Volcanic National Park
MEVE	Mesa Verde National Park
MORA	Mount Rainer National Park
NOCA	North Cascades National Park
OLYM	Olympic National Park
ROMO	Rocky Mountain National Park
SEKI	Sequoia – Kings Canyon National Park
WRST	Wrangell – St. Elias National Park
YELL	Yellowstone National Park
YOSE	Yosemite National Park
ZION	Zion National Park

Mercury in Fishes from 21 National Parks in the Western United States—Inter- and Intra-Park Variation in Concentrations and Ecological Risk

By Collin A. Eagles-Smith[1], James J. Willacker[1], Jr., and Colleen M. Flanagan Pritz[2]

Abstract

Mercury (Hg) is a global contaminant and human activities have increased atmospheric Hg concentrations 3- to 5-fold during the past 150 years. This increased release into the atmosphere has resulted in elevated loadings to aquatic habitats where biogeochemical processes promote the microbial conversion of inorganic Hg to methylmercury, the bioavailable form of Hg. The physicochemical properties of Hg and its complex environmental cycle have resulted in some of the most remote and protected areas of the world becoming contaminated with Hg concentrations that threaten ecosystem and human health. The national park network in the United States is comprised of some of the most pristine and sensitive wilderness in North America. There is concern that via global distribution, Hg contamination could threaten the ecological integrity of aquatic communities in the parks and the wildlife that depends on them. In this study, we examined Hg concentrations in non-migratory freshwater fish in 86 sites across 21 national parks in the Western United States. We report Hg concentrations of more than 1,400 fish collected in waters extending over a 4,000 kilometer distance, from Alaska to the arid Southwest. Across all parks, sites, and species, fish total Hg (THg) concentrations ranged from 9.9 to 1,109 nanograms per gram wet weight (ng/g ww) with a mean of 77.7 ng/g ww. We found substantial variation in fish THg concentrations among and within parks, suggesting that patterns of Hg risk are driven by processes occurring at a combination of scales. Additionally, variation (up to 20-fold) in site-specific fish THg concentrations within individual parks suggests that more intensive sampling in some parks will be required to effectively characterize Hg contamination in western national parks.

Across all fish sampled, only 5 percent had THg concentrations exceeding a benchmark (200 ng/g ww) associated with toxic responses within the fish themselves. However, Hg concentrations in 35 percent of fish sampled were above a benchmark for risk to highly sensitive avian consumers (90 ng/g ww), and THg concentrations in 68 percent of fish sampled were above exposure levels recommended by the Great Lakes Advisory Group (50 ng/g ww) for unlimited consumption by humans. Of the fish assessed for risk to human consumers (that is, species that are large enough to be consumed by recreational or subsistence anglers), only one individual fish from Yosemite National Park had a muscle Hg concentration exceeding the benchmark (950 ng/g ww) at which no human consumption is advised. Zion, Capital Reef, Wrangell-St. Elias, and Lake Clark National Parks all contained sites in which most fish exceeded benchmarks for the protection of human and wildlife health. This finding is particularly

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concerning in Zion and Capitol Reef National Parks because the fish from these parks were speckled dace, a small, invertebrate-feeding species, yet their Hg concentrations were as high or higher than those in the largest, long-lived predatory species, such as lake trout. Future targeted research and monitoring across park habitats would help identify patterns of Hg distribution across the landscape and facilitate management decisions aimed at reducing the ecological risk posed by Hg contamination in sensitive ecosystems protected by the National Park Service.

Introduction

Mercury (Hg) is among the most widespread environmental contaminants, distributed at a global scale from natural sources, such as volcanic eruptions and from anthropogenic sources such as emissions followed by atmospheric deposition (Driscoll and others, 2013) and at local or regional scales from current and historic mining activities (Singer and others, 2013). When reactive forms of inorganic Hg are deposited or transported to aquatic habitats, microbial processes can convert the inorganic Hg to methylmercury (MeHg), an organic Hg form that efficiently biomagnifies through food webs (Lavoie and others, 2013) and is a potent neurotoxin (Clarkson and Magos, 2006; Bennett and others, 2009; Wiener and others, 2012). The prevalence and toxicity of Hg make it a serious threat to both wildlife and humans. More than 16 million lake acres and 1 million river miles are under fish consumption advisories due to Hg contamination in the United States, and 81 percent of all fish consumption advisories were issued because of Hg contamination (U.S. Environmental Protection Agency, 2013). Mercury affects numerous physiological processes in vertebrate wildlife and humans, particularly neurological function, but it also can have a deleterious influence on cardiovascular, renal, and endocrine systems. Although the toxicological responses are particularly damaging to developing individuals, elevated Hg exposure also can have fitness consequences in adults through impairment of reproduction (Clarkson and Magos, 2006). Symptoms can occur at exposure levels commonly observed in the environment, and can result in reduced foraging efficiency, survival, or reproductive success (Burgess and Meyer, 2008; Crump and Trudeau, 2009; Wiener and others, 2012).

The widespread transport of Hg through atmospheric and hydrologic pathways, coupled with its potential for environmentally induced “activation” to a highly bioaccumulative and toxic species, is important because it suggests that even remote and protected environments may be at risk to ecological harm from Hg (Lamborg and others, 2002; Schuster and others, 2002). In fact, increasing evidence indicates that Hg contamination of remote and protected environments is not uncommon. For example, total Hg (THg) concentrations in arctic char (*Salvelinus alpinus*) and lake trout (*S. namaycush*) from lakes in Nunavut Canada have up to 1,400 nanograms per gram wet weight (ng/g ww) (Swanson and Kidd, 2010), and THg concentrations in mean northern pike (*Esox lucius*) from remote sites in Alaska have been reported above 1,800 ng/g ww (Duffy and others, 1999). At some of these remote sites, THg concentrations are routinely above the highest benchmarks for risk to human consumers (Jewett and Duffy, 2007). Similarly, Hg concentrations in fish from several western national parks have previously been reported to exceed human and wildlife health thresholds (Landers and others, 2008). Because many remote and protected areas provide refuge from development and human disturbance, they often support high densities of rare and sensitive wildlife species. As a result, the ecological consequences of elevated Hg in these locations could be particularly harmful from a conservation perspective.

National Park Service (NPS) lands are among the most pristine natural areas in the United States, protecting more than 84 million acres of diverse aquatic and terrestrial habitats. This is particularly true for the Western United States and Alaska, which only contain about 32 percent of national park units, yet account for approximately 89 percent of the total land area managed by the NPS. Because such a large proportion of NPS land is in the West, ecological stressors that are pervasive across the region could disproportionately affect NPS resources. Mercury contamination is among those stressors, because many areas of the Western United States are subject to Hg contamination at levels of concern for both human and ecosystem health (Peterson and others, 2007). Moreover, recent work has shown that some remote, high-altitude water bodies in western national parks contain fish with Hg concentrations that exceed toxicological thresholds for both humans and wildlife (Schwindt and others, 2008). However, the extent of contamination across western parks still lacks a systematic characterization and assessment of risk. Furthermore, it remains unclear whether variation in Hg concentrations in fish from remote national parks is more related to local, site- or park-specific characteristics, or broader regional characteristics. Additionally, spatially expansive baseline information is needed in order to evaluate the response of isolated environments to the Mercury and Air Toxics Standards (MATS), implemented by the U.S. Environmental Protection Agency (EPA), which aims to reduce Hg emissions from powerplants by 90 percent. Understanding how variation in fish Hg concentrations is distributed across local and regional scales will facilitate a better understanding of potential mitigation and management options.

As the first step toward better understanding the distribution of Hg across aquatic communities in western national parks, we measured Hg concentrations in fish from 21 different national parks in the Western United States and Alaska (fig. 1). We primarily targeted fish from remote, high altitude lake habitats with limited hydrological connectivity to upstream watersheds in order to minimize watershed scale processes that may potentially confound our assessment of local fish Hg exposure and the relative influence of local versus regional and global processes. Our primary objectives were as follows:

- *Objective 1:* Compare fish Hg concentrations among western national parks, and among sites within western national parks
- *Objective 2:* Determine whether variation in fish Hg concentrations in western national parks is more evident at local site and park scales, compared to broader regional scales;
- *Objective 3:* Evaluate fish Hg concentrations in western national parks with respect to a range of wildlife and human health exposure thresholds.

Because lakes with limited upstream connectivity were preferentially sampled, estimates of variability among lakes may be reduced relative to a random sampling of water available bodies, particularly at local and park scales. Furthermore, the hierarchical spatial design of the current study, combined with the gradient in landscape types, altitudes, and fish species that occur over such an extensive study area, preclude examination of the specific drivers of Hg variation at various scales, and investigating these factors was beyond the scope of the current study.

Methods

Study Sites and Sample Collection

This study focused on remote, fish-bearing aquatic habitats in the Western United States. We sampled 86 individual sites from 21 national parks in 10 Western States (fig. 1, table 1). Our primary target habitats were remote, high-altitude lakes containing non-migratory fish populations in order to reduce the likelihood that fish Hg concentrations are reflective of exposure that occurred elsewhere. However, some parks, especially Great Basin National Park (NP) and those in the arid Southwest, lacked similar aquatic habitats; therefore, we also sampled flowing rivers and streams in a few parks where lake habitats were unable to be sampled.

Fish were collected in 2008, 2009, 2011, and 2012 by NPS personnel following standardized protocols as described in this paragraph below. The intensive sampling at Mount Rainier in 2012 was conducted by U.S. Geological Survey (USGS) staff as part of a focused study in that park, examining a range of food web components. Whereas most of the sites were only sampled during 1 of the 4 years, 10 lakes within four parks (Lake Clark, Olympic, Rocky Mountain, Sequoia-Kings Canyon) were sampled in 2 separate years in order to examine temporal variation within a subset of lakes (table 1). Depending on the characteristics of a site, and its associated access logistics, fish were sampled using variable-mesh or fixed-mesh gillnets, hook and line, or beach seines. Because of ecological limits to the distribution of fish species across the entire western region of North America, identical species did not exist at all sites. Instead, we focused on targeting four primary species (brook trout [*S. fontinalis*], cutthroat trout [*Oncorhynchus clarkii*], rainbow trout [*O. mykiss*], and lake trout) that occurred across as many sites as possible. In sites where those species did not occur, we sampled the most abundant resident species that were found, trying where possible to match the ecological niche of at least one of our primary species. Once fish were collected, they were immediately wrapped in chemically cleaned aluminum foil and sealed in individually labeled polyethylene zip bags. Fish in the field were immediately placed on wet or dry ice until transport to the laboratory, where they were stored at -20°C until processing and analysis.

Sample Processing and Mercury Determination

In the laboratory, we thawed each fish to room temperature, measured standard length (SL) to the nearest millimeter, and mass to the nearest 0.1 g. From each fish with greater than 100 mm SL, we dissected 5–10 g of skinless axial muscle using nitric acid and deionized-water-rinsed scalpels, scissors, and forceps. We then rinsed each muscle sample in deionized water to remove any surface contaminants, blotted them dry with clean, lint-free wipes, and weighed them on an analytical balance to the nearest 0.0001 g. Fish with less than 100 mm SL were processed as whole-body samples. For these, we removed stomach contents and thoroughly cleaned the surface of the fish with deionized water, then blotted dry and weighed as described above for muscle tissue. We chose to process and analyze fish less than 100 mm SL as whole fish because they often lacked adequate muscle tissue for analysis. Moreover, robust conversion equations exist for translating between whole-body fish and muscle THg concentrations (Peterson and others, 2005, 2007); thus, the whole-body samples were still comparable to the muscle samples. Once muscle or whole-body samples were cleaned and weighed, we placed them in a drying oven at 50 °C until a constant mass was achieved, typically 48 h. We subsequently removed the sample from the drying oven and allowed them to cool to room temperature in a desiccator. Once cool, we measured the dry mass of each sample to the nearest 0.0001 g, and homogenized each to a fine powder with scissors and a ceramic mortar and pestle. The homogenized samples were stored in a desiccator until chemical analysis.

We determined THg concentrations of each sample following EPA method 7473 (U.S. Environmental Protection Agency, 2000) on a Milestone tri-cell DMA-80 Direct Mercury Analyzer (Milestone, Inc., Monroe, Connecticut, U.S.A.). Briefly, we used an integrated sequence of drying (250 °C for 30 s), thermal decomposition (650 °C for 90s), catalytic conversion, and then amalgamation in a gold trap, followed by cold vapor atomic absorption spectroscopy. Quality-assurance measures included analysis of two certified reference materials (either dogfish muscle tissue [DORM-4; National Research Council of Canada, Ottawa, Canada], or dogfish liver [DOLT-3; National Research Council of Canada, Ottawa, Canada]), two system and method blanks, and two duplicates-per-batch of 30 samples. Recoveries (± standard error) averaged 102.8 ± 0.6 percent (n = 88) and 97.83 ± 1.06 percent (n = 128) for certified reference materials and calibration checks, respectively. Absolute relative percent difference for all duplicates averaged 3.31 ± 0.40 percent.

Statistical Analyses

Unless otherwise noted, all statistical analyses were conducted on dry-weight THg concentrations to minimize inconsistencies resulting from variable moisture contents of individual fish. However, to facilitate comparisons with other work, we present wet weight THg concentrations in the text and figures. Fish length is often positively correlated with Hg concentrations, which can confound direct comparisons of fish Hg concentrations among locations with different sized fish. In order to make spatial comparisons among and within parks, we first evaluated whether size adjusting THg concentrations was necessary for each species of fish at each sampling site. To make this determination, we developed decision rules based on a combination of the strength of regression coefficients (R^2) and the small sample-size corrected Akaike's Information Criterion (AIC_c) for length versus THg concentration (dry weight, $\log_{10}$-transformed) regression models from each species-site combination. No size adjustment was made for models with R^2 less than 0.20. For regression models with R^2 greater than 0.20, we compared the regression model with a null model using AIC to assess whether including fish length as a variable improved the likelihood of predicting fish THg concentrations. If the difference between the regression and null AIC_c values (ΔAIC_c) was less than 2, we proceeded with size adjusting individual THg concentrations using linear regression. We also used AIC-based model assessment to determine whether sampling year (where available) influenced the relationship between size and Hg concentrations by including a year-by-size interaction. The year-by-size interaction failed to improve the plausibility of the size-Hg relationship in all cases; thus for size adjustment, we lumped all fish from a population regardless of year sampled.

The variation in sizes and species of fish collected across sites complicated direct comparisons of size-adjusted THg concentrations for all fish simultaneously. Therefore, using length-frequency histograms, we classified fish species into three different size categories (50, 200, and 400 mm SL), thus minimizing extrapolation of THg concentrations outside the range of sizes observed for each species. Mercury concentrations in smallest size category of fishes (speckled dace [*Rhinichthys osculus*], threespine stickleback [*Gasteristeus aculeatus*], and torrent sculpin [*Cottus rhotheus*]) were adjusted to the concentration predicted at 50 mm SL for each individual. Mercury concentrations in the intermediate size category of fishes (arctic grayling [*Thymallus arcticus*], brook trout, brown trout [*Salmo trutta*], cutthroat trout, Dolly Varden [*S. malma malma*], golden trout [*O. mykiss aquabonita*], kokanee [*O. nerka*], rainbow trout), were adjusted to the concentration predicted at 200 mm SL for each individual. Mercury concentrations in the largest size category of fishes (bull trout [*S. confluentus*], lake trout, lake whitefish [*Coregonus clupeaformis*], and northern pike) were adjusted to the concentration predicted at 400 mm SL for each individual. To standardize THg concentrations to the appropriate sizes, we used the linear regression equations for each site-species combination, then added the residuals from each individual fish to its predicted concentration in order to incorporate the unexplained variation back into the estimate. For those fish sampled from sites where there was no meaningful Hg-size relationship, we used the un-adjusted Hg concentrations for each individual.

To compare Hg concentrations among parks and sites within parks, we constructed separate mixed effects, nested general linear models for each of the three size classes (50, 200, and 400 mm SL) with park and site (nested within park) included as fixed effects, and species included as a random effect in each model to statistically account for variation due to different species being collected across sites. In addition to comparing model-adjusted fish THg concentrations among parks and sites within parks, we used the model-adjusted THg concentrations at sites and parks to examine how variation in THg concentrations was partitioned across a gradient of spatial scales. To do this, we evaluated the coefficient of variation (CV) for (1) all fish within a site using individual fish THg concentrations, (2) all sites within a park using model-generated least-square mean (LSM) THg concentrations for each site, and (3) all parks using model-generated LSM Hg concentrations for each park. By comparing the CVs from each of these scales with t-tests (two-sample in comparison 1 and 2, one-sample in comparison 3), we evaluated whether within-lake processes, within-catchment processes, or within-region processes accounted for the most variation in THg concentrations.

In order to evaluate temporal variation in the subset of 10 sites that were sampled in more than 1 year, we used analysis of covariance (ANCOVA), which tested whether fish Hg concentrations were different between years for each lake, while statistically controlling for fish length. Therefore, we included standard length as a covariate in the models. The initial sampling event for each site varied between 2008, 2009, and 2011 depending on the park. The final sampling for all sites was 2012. Because of this, our goal was not to test whether there were consistent year effects, but simply to evaluate how Hg concentrations within a site might differ between two distinct years regardless of which years those were.

Risk Benchmark Values

In addition to examining spatial and temporal variation in fish THg concentrations across parks, we also evaluated the threat Hg might pose at each site to the health of fish, piscivorous wildlife, and humans by comparing measured and modeled THg concentrations to a range of established benchmarks of toxicity to different taxonomic groups.

Fish

We compared whole-body Hg concentrations in fish to a no-observed-effects-residue (NOER) of 200 ng/g ww (Beckvar and others, 2005) and a lowest-observed-effects-residue (LOER) of 300 ng/g ww (Sandheinrich and others, 2011) to assess potential risks of Hg exposure to fish themselves. The NOER threshold identifies the Hg concentration in fish tissues below which fish should not experience deleterious effects of Hg exposure on reproduction, growth, or survival. In contrast, the LOER threshold indicates the concentration above which sublethal endpoints of Hg exposure, including alterations to reproductive health, have been documented in laboratory and field studies of fish.

Birds

Birds are sensitive to the toxicological effects of MeHg exposure, and piscivorous birds are among the taxa most commonly at risk because of their high trophic position. However, robust data on the dietary Hg exposure thresholds that result in deleterious effects exist for very few bird species. Moreover, there is considerable evidence of substantial variability in the sensitivity of different bird species to Hg exposure (Heinz and others, 2009). Therefore, accurately extrapolating risk across a range of unique species can be difficult and may result in low confidence estimates. This is particularly problematic in studies such as this one, where no single piscivorous bird species with well-documented toxicological responses to Hg exposure extends across such an expansive spatial extent, nor over such a range of habitats. Therefore, our approach was to establish a baseline of risk in a well-studied avian piscivore, then evaluate the likely range of risk for arbitrary taxa that are more and less sensitive to Hg than the baseline species. Arguably the most robust information on avian piscivore toxicological responses to Hg exposure across a range of scales of ecological organization exists for the common loon (*Gavia immer*), which has been intensively studied in field and laboratory settings for several decades (Kenow and others, 2008; Mitro and others, 2008; Scheuhammer and others, 2008; Kenow and others, 2011). This extensive research on the behavioral and reproductive impacts of Hg contamination has resulted in the development of robust dietary benchmarks for identifying ecological risk of Hg to common loons at a range of deleterious endpoints. These studies, coupled with other work on interspecific sensitivity to Hg exposure (Heinz and others, 2009), suggest that loons are moderately sensitive to Hg exposure in comparison to some other taxa. Therefore, we utilized the common loon dietary screening benchmark of 180 ng Hg/g ww in whole-body prey-fish (Depew and others, 2012b) as our baseline indicator of Hg risk to moderately sensitive avian consumers. Without solid quantitative comparisons of relative sensitivity among species, we chose a conservative approach of scaling the loon screening value down and up by 50 percent in order to estimate potential risk to bird species that are more or less sensitive to Hg exposure than the common loon. Thus, we assessed risk to avian consumers by comparing Hg concentrations in whole-body fish to 90 (high sensitivity), 180 (moderate sensitivity), and 270 ng Hg/g ww (low sensitivity) benchmarks. We recognize that this scaling is somewhat arbitrary in that it is not calibrated to specific avian species responses, but we chose this approach to better encompass the range of potential risk depending on the species that might occur at any given area. Much work is still needed to refine the scientific understanding of interspecies differences in sensitivity, but we use Heinz and others (2009) as the baseline for identifying some taxonomic groups that may fall into each classification. Based on that study of 27 bird species, piscivorous taxa that can be categorized as high sensitivity and that are found in some of the parks studied include osprey (P*andion haliaetus*), white ibis (*Eudocimus albus*), and snowy egret (*Egretta thula*). Species that can be categorized as low sensitivity include hooded merganser (*Lophodytes cucullatus*) and double-crested cormorant (*Phalacrocorax auritus*). Besides common loon, other species with probable moderate sensitivity include common tern (*Sterna hirundo*), royal tern (*Sterna maxima*), and great egret (*Ardea alba*).

Humans

We assessed risk of Hg exposure to human consumers of fish by comparing Hg concentrations in fish muscle to the EPA’s tissue residue criterion for Hg (300 ng/g ww) and two benchmarks recommended by the Great Lakes Advisory Group (GLAG) for limiting the risk of Hg exposure to anglers (Great Lakes Advisory Group, 2007). The lower benchmark (50 ng/g ww) represents the concentration of Hg in fish muscle below which unrestricted consumption of fish poses no risk to consumers. The higher benchmark (950 ng/g ww) represents the concentration of Hg in fish muscle above which GLAG recommends that no consumption occur.

Risk Estimation

We assessed risk to fish, avian consumers, and humans using two approaches. First, we compared whole-body (fish and avian risk assessments) or muscle (human risk assessment) Hg concentrations in fish to each of the reference values discussed above (two fish, three avian, and three human benchmarks) and enumerated the proportion of individuals exceeding each benchmark at each site, park, and across parks. This analysis provides an overview of Hg risk in all parks and sites sampled, but does not account for other factors that influence Hg risk. In particular, because Hg concentrations can be strongly influenced by fish size, it is important to understand how risk varies across the size spectra of the fish communities. Therefore, in the populations (that is, species-site combinations) where size was correlated with Hg concentrations, we used a second approach utilizing the size-Hg relationships to model the Hg concentration of each fish sampled from a population at 1 mm increments over the range of sizes observed in that population. We then compared these modeled fish Hg concentrations to each of the eight benchmarks at 1 mm increments to determine the proportion of individuals exceeding each benchmark at a given size. Uncertainty in these size-specific risk estimates was preserved by comparing the bounds of a 95 percent prediction interval for each fitted value to the benchmarks. The resulting size-specific risk profiles present changes in the risk of Hg across the size of individuals sampled from a population and can be used to identify the size at which a given proportion of individuals in a population likely exceed a specified threshold, or conversely, the proportion of individuals in a population that exceed a threshold at a given size. Because Hg concentrations in fish, and thus the risk of Hg to fish, wildlife, and humans, varies with fish size, these size-specific risk profiles better approximate the range of risks present in a population and provide important insights into how risks can be managed.

Results

Over a 4 year time span (2008, 2009, 2011, and 2012), we sampled 1,486 fish representing 16 different species from 86 individual sites across 21 national parks (table 1). Throughout all parks and species, fish muscle THg concentrations ranged over two orders of magnitude from 9.9 to 1,109 ng/g ww. The overall geometric mean (95-percent confidence interval in parentheses) of all fish sampled was 77.7 (73.6–80.8) ng/g ww. However, fish THg concentrations varied substantially among individual parks, with park-specific geometric mean values ranging from 32.6 (28.3–37.6) ng/g ww in Grand Teton NP to 325.6 (281.4–376.9) ng/g ww in Capitol Reef NP (table 2; fig. 2). Furthermore, within parks there was considerable variation in geometric mean fish THg concentrations among sites (table 3, figs. 2 and 3). The greatest range in site-specific geometric mean fish THg concentrations within parks occurred in Mount Rainier and Rocky Mountain NPs, where we measured approximately 11- and 8-fold differences, respectively, between the sites within each park with the lowest and highest THg concentrations (table 2, fig. 3). Importantly, the species and size distribution of fishes differed among parks and lakes, confounding direct comparisons across locations because of the influence that these factors have on THg concentrations (fig. 3). Therefore, we subsequently normalized the THg concentrations to account for the effects of size and species on fish THg concentrations, thus allowing for more meaningful spatial comparisons.

Model-Adjusted Mercury Concentrations

Inter-Park Comparisons

In order to control for the influence of size and species on fish Hg concentrations, we first assessed the relationships between fish length and THg concentrations on a site and species-specific basis as described in section, “Methods.” Size was correlated with THg concentrations in 61 of the 104 site-species combinations (table 1; Significant Size Regression). For those 61 combinations, we normalized each individual THg concentration to a standardized length (50, 200, or 400 mm SL), which was dependent on the taxonomic grouping to which it was assigned (see section, “Methods”). For the site-species combinations where there were no relationships between length and Hg concentrations, we used unadjusted Hg values for each fish as it was assigned to its respective size category (table 1).

After accounting for the effects of size and species, LSM THg concentrations of each fish size class differed among parks (50 mm SL: $F_{3,107} = 100.15$, $p < 0.001$; 200 mm SL: $F_{14,56.7} = 79.60$, $p < 0.001$; 400 mm SL: $F_{4,124} = 79.08$, $p < 0.001$; fig. 4). Fish species assigned to the 400 mm SL size class only occurred in 5 of the 21 parks, primarily the northernmost parks—Denali, Wrangell-St. Elias, Lake Clark, Glacier, and Yellowstone NPs. Among those five parks, LSM THg concentrations in fish ranged from 47.4 ± 1.6 to 278.9 ± 7.4 ng/g ww. Fish THg concentrations were highest at Wrangell-St. Elias NP followed by Glacier, Lake Clark, Yellowstone, and Denali NPs (table 2, figs. 4 and 5).

Fish assigned to the 200 mm SL size class were collected from the greatest number of parks, and representatives of this size class are absent only from Denali, Lake Clark, Glacier, Capitol Reef, Zion, and Mesa Verde NPs. Mercury concentrations in fish from the 200 mm SL size class had a much narrower range of LSMs across parks than the 400 mm SL fish with concentrations ranging from 26.6 ± 2.6 to 119.9 ± 10.2 ng/g ww. Within this size class, Yellowstone NP had the highest fish THg concentrations, followed by Olympic, Glacier Bay, Grand Canyon, Wrangell-St. Elias, North Cascades, Yosemite, Sequoia-Kings Canyon, Lassen Volcanic, Great Sand Dunes, Mount Rainier, Rocky Mountain, Great Basin, Grand Teton, and Crater Lake NPs (fig 4, table 2).

Similar to the largest size class of fishes, we only collected samples falling into our smallest size class of fishes (50 mm SL) from a limited subset of four parks, mostly in the Southwestern region. Least-square mean THg concentrations in this size class ranged from 74.9 ± 2.9 to 242.5 ± 5.4 among parks. Among the 50 mm SL size class, fish Hg concentrations were highest at Zion NP, followed by Capitol Reef, Mount Rainier, and Mesa Verde NPs. The LSM values of 50 mm SL fish from Zion and Capitol Reef NPs are comparable to those in parks with the highest 400 mm SL fish (figs. 4 and 5).

Intra-Park Comparisons

In addition to regional differences in THg concentrations across parks, we also found differences in fish THg concentrations among sites nested within parks for each of the three size classes (50 mm SL: $F_{5,\,107} = 12.22$, $p < 0.001$; 200 mm SL: $F_{57,\,679.9} = 41.72$, $p < 0.001$; 400 mm SL: $F_{3,\,124} = 31.56$, $p < 0.001$; fig 6, table 3). The magnitude of site differences within a park varied among parks and size classes. For the 400 mm SL size class, there were only two parks with multiple sites containing fish assigned to this size class. The sites with the highest THg concentrations were 1.33- and 3.35-fold higher than the sites with the lowest THg concentrations for Wrangell-St. Elias and Lake Clark NPs, respectively. For the 200 mm SL size class, the highest site within a park averaged 4.4 times higher than the lowest site within a park. The differences between the highest and lowest sites within parks ranged from 1.27- to 22.61-fold in Great Basin and Mount Rainier NPs, respectively. For the 50 mm SL size class, the highest site within a park was on average 1.7 times higher than the lowest site within a park, and this difference between highest and lowest sites within a park ranged from 1.05- to 2.25-fold in Zion and Mount Rainier NPs, respectively. The values above provide insight into the overall range in concentrations within and among NPs, but do not adequately address the overall variance that was measured at each scale.

Variation Across Spatial Scales

To evaluate whether variation in fish THg concentration was most prevalent at the site, park, or regional level, we examined the CV at three scales: (1) CV among fish within individual sites, (2) CV among lakes within individual parks, and (3) CV among parks across the .Western United States, including Alaska. The CV among fish within individual sites ranged from 0.097 at Death Canyon Creek in Grand Teton NP to 0.819 at The Loch in Rocky Mountain NP, and averaged 0.303 across all sites (table 3). The CV among lakes within individual parks ranged from 0.121 in Great Basin NP to 0.865 on Yosemite NP (table 3). The mean CV among sites within parks (0.493) was greater than the average CV among fish within sites ($t = 3.87$, $df = 18$, $p = 0.001$). The CV among parks was 0.673 (table 3); significantly higher than the mean CV among sites within parks ($t = -3.798$, $df = 16$, $p = 0.002$). These data suggest that variation in THg concentrations increases at progressively larger landscape scales, with the greatest variation resulting from regional factors, followed by local and within-lake biological factors.

Inter-Annual Variation

There were no consistent patterns in inter-annual variability across the 10 sites sampled over separate years. Significant differences in mean Hg concentrations were observed at 5 of 10 sites. Fish THg concentrations increased from the first sampling year to the second at two sites: Sun Up Lake (Olympic NP; $F_{1,27} = 6.35$, $p = 0.018$; fig. 7) and Center Basin Lake (Sequoia-Kings Canyon NP; $F_{1,42} = 22.87$, $p < 0.0001$; fig. 7). Conversely, fish THg concentrations decreased from the first sampling event to the second in three sites—Upper Lena Lake (Olympic NP; $F_{1,24} = 7.12$, $p = 0.014$, fig. 7), Poudre Lake (Rocky Mountain NP; $F_{1,27} = 4.77$, $p = 0.038$, fig. 7), and Ypsilon Lake (Rocky Mountain NP; $F_{1,27} = 4.36$, $p = 0.046$, fig. 7). Finally, we found no differences among years at the remaining five sites—Gladys Lake (Olympic NP; $F_{1,26} = 1.13$, $p = 0.30$), Lake Nanita (Rocky Mountain NP; $F_{1,20} = 0.004$, $p = 0.89$), Mirror Lake (Rocky Mountain NP; $F_{1,27} = 0.21$, $p = 0.65$), Kern Point (Sequoia-Kings Canyon NP; $F_{1,27} = 1.47$, $p = 0.23$), and Lake Clark (Lake Clark NP; $F_{1,24} = 0.037$, $p = 0.85$). The magnitude of differences between years was 19.6 percent in Sun Up Lake (Olympic NP), 24.4 percent in Ypsilon Lake (Rocky Mountain NP), 24.8 percent in Upper Lena Lake (Olympic NP), 26.5 percent in Poudre Lake (Rocky Mountain NP), and 43.6 percent in Center Basin Lake (Sequoia-Kings Canyon NP).

Toxicological Risk

Estimating toxicological risk of Hg within ecosystems can be subjective and dependent on the endpoints selected, the uncertainty surrounding individual- and species-specific sensitivity estimates, and whether risk is based on dietary exposure or tissue-based Hg concentrations. Because of the geographic scale of this study, the wide range of species sampled, and the somewhat limited sample sizes within sites, we evaluated ecological and human health risk at a relatively coarse scale, by comparing measured, unadjusted fish THg concentrations to a gradient of pre-established benchmarks individually for fish health, avian (piscivorous bird) health, and human health. However, in addition to simply comparing the proportion of sampled fish to these thresholds, we modeled Hg risk to each group across the range of fish sizes for each site in order to evaluate how risk to these groups varies with fish size (fig. 8; see section, “Methods”).

Fish Risk

Across all parks, species, and sizes of fish, 5 percent of individuals had whole-body THg concentrations above which Hg exposure may affect fish health (NOER; 200 ng/g ww; table 4). Fewer than 2 percent of individuals had whole-body THg concentrations above which deleterious effects are likely (LOER; 300 ng/g ww; table 4).

The distribution of risk was not evenly distributed across all parks. Thirteen of the 21 parks sampled contained no fish exceeding the 200 ng/g NOER benchmark. In the eight parks (38 percent) with fish exceeding 200 ng/g THg, the proportion of individual fish exceeding this benchmark was highest in Capitol Reef NP, where 49 percent of fish sampled had concentrations above 200 ng/g ww, followed by Lake Clark, Zion, Wrangell-St. Elias, Yosemite, Lassen Volcanic, Rocky Mountain, and Mount Rainier NPs (table 4). Of the 21 parks sampled, only 6 (Capitol Reef, Lake Clark, Wrangell-St. Elias, Zion, Yosemite, and Rocky Mountain NPs; 29 percent) contained fish with THg concentrations exceeding the 300 ng/g ww LOER benchmark (table 4), the value above which deleterious effects to fish are likely. Twenty-two percent of fish from Capitol Reef NP exceeded this concentration, whereas Lake Clark, Wrangell-St. Elias, Zion, Yosemite, and Rocky Mountain NPs had progressively lower proportions fish exceeding this benchmark (table 4).

When examined on a site-specific basis, fish THg concentrations exceeded 200 ng/g ww in 13 sites (15 percent) across the 8 parks containing fish with values above this benchmark. More than one-half of the individual fish exceeded the NOER threshold at Tanada Lake (85 percent) in Wrangell-St. Elias NP, Fremont River site 1 (67 percent) and site 7 (60 percent) in Capitol Reef NP, and Lake Clark (54 percent) in Lake Clark NP (table 4). Other sites with fish exceeding the NOER benchmark include East Fork Virgin River (Zion NP), MORA_1614 (Mount Rainier NP), Fremont River site 4 (Capitol Reef NP), Lake Kontrashibuna (Lake Clark), Mildred Lake (Yosemite NP), Fall River (Rocky Mountain NP), Mirror Lake (Rocky Mountain NP), North Fork Virgin River (Zion NP), Horseshoe Lake (Lassen Volcanic NP). The 300 ng/g LOER was exceeded in only 8 individual sites (9 percent). Two sites in Capitol Reef NP (Fremont River sites 1 and 7) had the highest proportions of individuals exceeding this benchmark, 33 and 27 percent, respectively (table 4). Other sites with fish exceeding the LOER benchmark include Tanada Lake (Wrangell-St. Elias NP), Lake Clark (Lake Clark NP), Fremont River site 4 (Capitol Reef NP), Mildred Lake (Yosemite NP), North Fork Virgin River (Zion NP), and Fall River (Rocky Mountain NP).

Size-Specific Risk

We modeled risk across the observed size range for all 61 populations with size-Hg relationships. This approach was implemented so that instead of simply calculating the proportion of fish exceeding benchmarks from the sampled size range (which is not necessarily representative of the size distribution of fish within the lake), we could estimate specific lengths at which most fish exceed each benchmark. Taking this approach, our models estimated that fish would exceed the 200 ng/g ww NOER benchmark in 21 (34 percent) of the populations modeled (fig. 9). In eight of those populations, at least one-half of all individuals are predicted to exceed the NOER at some length along their size spectra (fig. 9). Four of these populations were speckled dace from three sites in Capitol Reef NP and one site in Zion NP. In these four populations, the mean size at which 50 percent of the individuals were modeled to exceed the benchmark was 61 mm SL, ranging from 52 mm SL in Capitol Reef's Fremont River site 1, to 67 mm SL in Fremont River sites 4 and 7 (fig. 9). For the other four populations, our models predict that 50 percent of the individuals will exceed the NOER at 264 and 302 mm in brook trout from Mildred Lake in Yosemite NP and Horseshoe Lake in Lassen Volcanic NP, respectively, 323 mm SL in suckers from Fall River in Rocky Mountain NP, and 412 mm SL in lake trout from Tanada Lake in Wrangell-St. Elias NP (fig. 9).

Model predictions estimate that fewer fish populations (12 of 61 populations modeled; 20 percent) are expected to contain individuals exceeding the 300 ng/g LOER over the range of fish sizes sampled in each population than estimated for the NOER (fig. 9). At least 50 percent of the individuals in five of the populations were predicted to exceed this benchmark at some point within their measured size spectra. Four of those populations were specked dace from Capitol Reef and Zion NPs, where the estimated size at which 50 percent exceeded the benchmark was 71 mm SL (fig. 9). Additionally, at least one-half of the lake trout population greater than 526 mm SL from Tanada Lake in Wrangell-St. Elias NP are projected to exceed the LOER (fig. 9).

Avian Risk

When examining the unadjusted whole-body THg concentrations, 35 percent of all fishes sampled had THg concentrations above our benchmark for risk to birds with high sensitivity to Hg exposure (90 ng/g ww; table 4). Tissue concentrations in 12 percent of individual fish exceeded the 180 ng/g ww benchmark for reproductive impairment in species with moderate sensitivity, and 5 percent of individual fish exceeded 270 ng/g ww, our benchmark for risk to birds with low sensitivity to Hg exposure (table 4).

Among the 21 parks, the potential risk to piscivorous birds varied substantially. In 7 parks (33 percent), none of the fish analyzed exceeded the 90 ng/g ww benchmark for highly sensitive bird species. In the remaining 14 parks (67 percent), more than one-half of sampled fish were above this concentration, and more than three-quarters of sampled fish exceeded this value in Lake Clark, Zion, Capitol Reef, and Glacier NPs (76, 90, 98, and 100 percent, respectively). The benchmark for reproductive impairment in moderately sensitive avian species was exceeded in nine (43 percent) parks. The proportion of fish within these nine parks that exceeded this benchmark, ranged from 1 percent of individual fish at Yellowstone NP to 56 percent of individual fish at Capitol Reef NP, which was the only park from which more than one-half of the fish sampled exceeded the moderate sensitivity benchmark (table 4). Fourteen of the 21 parks had no fish with THg concentrations above the benchmark for birds with low Hg sensitivity (table 4), and the proportion of fish exceeding this threshold was less than 10 percent in all remaining seven parks (33 percent) except Capitol Reef, where 33 percent of fish sampled exceeded the threshold (table 4).

On a site-specific level, 41 of the 86 sites sampled had no fish with THg concentrations above the benchmark for the most sensitive avian species (table 4). However, at the remaining 45 sites (52 percent), 63 percent of fish exceeded this benchmark, and at 7 sites, 100 percent of fish sampled were above the high sensitivity avian benchmark. At an additional seven sites, between 50 and 93 percent of fish had THg concentrations greater than the high sensitivity avian threshold (table 4). The loon reproductive impairment threshold was exceeded at 18 sites (21 percent), with more than one-half of the fish surpassing this benchmark at four sites (table 4). Nine individual sites (representing seven parks) contained fish with THg concentrations above the benchmark for birds with low sensitivity to Hg (table 4).

Size-Specific Risk

Size-specific Hg risk to piscivorous birds was modeled for 61 populations. Of these, whole-body Hg concentrations in fish from 14 populations (23 percent) did not exceed the 90 ng/g ww benchmark for risk to the most sensitive avian species, while the remaining 47 populations (77 percent) were modeled to exceed this threshold at some size (fig. 10). However, 100 percent of lake trout from Tanada and Copper Lakes in Wrangell-St. Elias NP and lake whitefish from Lake McDonald in Glacier NP were predicted to exceed this benchmark over the entire range of fish sizes (fig. 10). The size at which 50 percent of the fish in a population were predicted to exceed the benchmark for risk to highly sensitive piscivorous birds ranged from less than 40 mm SL in speckled dace from Fremont River site 1 in Capitol Reef NP to 414 mm SL in brown trout from the Colorado River in Rocky Mountain NP. Among the smaller "forage fish" (that is, speckled dace, torrent sculpin, and threespine stickleback), the mean size at which 50 percent of the individuals within a population exceeded the threshold was 59 mm SL (n = 6 populations). For brook trout, cutthroat trout, and rainbow trout, the mean size at which 50 percent of the individuals within a population exceeded the threshold was 223 mm SL (n = 15 populations).

The majority of individual populations modeled (36 of 61 populations, 59 percent) were predicted to have no individual fish with THg concentrations exceeding the benchmark for reproductive impairment in moderately sensitive avian consumers across the range of fish lengths, whereas in 25 parks (41 percent), some fish were modeled to exceed this benchmark (fig. 10). However, in 10 populations, more than one-half of the individual fish in each population were predicted to have THg concentrations above the common loon reproductive impairment benchmark, and lake trout from Tanada Lake in Wrangell-St. Elias NP were modeled to exceed this threshold over the entire range of sizes sampled (fig. 10). Four of the 10 populations with greater than 50 percent of fish predicted to exceed the loon risk concentration at some length within the population were speckled dace. The mean size at which 50 percent of these speckled dace populations was predicted to exceed the loon risk benchmark is 58 mm SL. The mean size at which 50 percent of the individuals in a population was modeled to exceed the loon risk threshold was 261 mm SL for three brook trout populations (Horseshoe Lake in Lassen Volcanic NP, Mirror Lake in Rocky Mountain NP, and Mildred Lake in Yosemite NP).

Our models predicted that fish from 12 populations sampled (20 percent) would exceed the 270 ng/g ww threshold for birds with the lowest Hg sensitivity at some point along their size spectrum (fig. 10). In six of the populations, 50 percent or more of the individuals were modeled to exceed this threshold at some size. Four of these six populations were speckled dace from Capitol Reef and Zion NPs, and the mean length at which 50 percent of the population was predicted to exceed the threshold was 68 mm SL, only 10 mm greater than the same exceedance level for loon risk. The other two populations were suckers from the Fall River in Rocky Mountain NP and lake trout from Tanada Lake in Wrangell-St. Elias NP. In those populations, the model-predicted size at which 50 percent of the population exceeded the threshold was 353 and 496 mm SL, respectively.

Human Consumption Risk

As mentioned in section, "Methods," human health risk was assessed using fish muscle THg concentrations, not whole-body concentrations as was done for wildlife risk. In the unadjusted data across all parks, species and sizes of fish, 32 percent of fish sampled had THg concentrations in their muscle below the 50 ng/g ww threshold established by the GLAG for unlimited consumption of fish, indicating that some human consumption guidance would apply to 68 percent of the individuals measured in this study (table 4). However, only 4 percent of fish had muscle THg concentrations exceeding the EPA human health criterion of 300 ng/g ww, and less than 1 percent had concentrations above the level at which GLAG recommends that no fish be consumed (950 ng/g ww).

As with the other benchmarks, the proportion of fish exceeding human risk thresholds varied greatly among parks. Grand Teton and Great Basin NPs had the lowest proportion of fish exceeding the GLAG unlimited consumption threshold (18 percent), whereas 100 percent of fish analyzed from Glacier NP exceeded this threshold (table 4). In 17 parks (81 percent), more than one-half of sampled fish had concentrations above the GLAG threshold. In 11 (61 percent) of the 18 parks from which species consumed by humans (that is, excluding speckled dace, threespine stickleback, and torrent sculpin) were analyzed, no fish had muscle THg concentrations exceeding the EPA human health criterion. In the remaining 7 of 18 parks (39 percent) with fish exceeding the EPA human health criterion, the proportion of fish with concentrations above 300 ng/g ww was typically only 1–3 percent of individuals, with the exception of Lake Clark, Wrangell-St. Elias, and Yosemite NPs, where 40.7, 18.3, and 6.7 percent of fish exceeded the guideline, respectively (table 4). Yosemite was the only park in which any fish exceeded the GLAG no consumption guideline, with 3 percent of the analyzed fish having muscle concentrations over the guideline (table 4).

When examined at the finer site scale, 6 (8 percent) of the 79 sites from which fish species consumed by humans were sampled had no fish exceeding the GLAG unlimited consumption guideline, whereas at 23 of these sites (30 percent) all fish analyzed exceeded this guideline. At 56 sites (71 percent) at least 50 percent of the fish analyzed exceeded the unlimited consumption guideline. Thirteen sites (17 percent) had fish exceeding the EPA human health criterion, with more than 50 percent of individual fish from Lake Clark (80 percent) in Lake Clark NP and Tanada Lake (85 percent) in Wrangell–St. Elias NP exceeding the guideline. Mildred Lake in Yosemite NP was the only lake from which fish (7 percent) exceeded the GLAG no consumption guideline.

Size-Specific Risk

We used the same approach presented for fish and birds to model risk to humans across the size spectra of 54 populations with size-THg relationships. These models predicted that muscle THg concentrations in fish from every population would exceed the GLAG unlimited consumption guideline at some length over the observed range of sizes (fig. 11). The sole exception was cutthroat trout from Death Canyon Creek in Grand Teton NP, where even the largest fish sampled (260 mm SL) were predicted to have muscle concentrations below this level. In 40 of the 54 populations (74 percent) modeled, the unlimited consumption threshold was predicted to be exceeded by some fish across the entire range of observed sizes (fig. 11). In 8 populations (15 percent), all fish were expected to exceed this threshold across the entire range of sizes sampled. In the remaining populations, the average size at which 50 percent of the fish in a population were expected to exceed the threshold varied substantially, ranging from 40 mm SL in brook trout from Golden Lake in Mount Rainier NP to nearly 400 mm SL in northern pike from Lake Chilchukabena in Denali NP. When limited to brook, rainbow, and cutthroat trout, which comprise nearly 65 percent of the fish analyzed, the average size at which 50 percent of the fish in a population were modeled to exceed the unlimited consumption guideline was 179 mm SL (n = 36 populations).

Eighteen populations were modeled to exceed the EPA criterion for human consumption at some point across the measured size spectra (fig. 11). In six of these populations, models estimated that 50 percent or more of the individual fish at a certain size threshold within a population were predicted to exceed the EPA human health criterion, and all lake trout down to the smallest size (410 mm SL) from

Tanada Lake in Wrangell-St. Elias NP were predicted to exceed the EPA human health criterion (fig. 11). For the three brook trout populations where at least one-half of the fish at a given size were predicted to exceed the EPA criterion (Horseshoe Lake in Lassen Volcanic, Golden Lake in Mount Rainier, and Mildred Lake in Yosemite), the average size at which 50 percent of the individuals exceeded it was 258 mm SL.

After modeling risk across the entire sampled fish size range, nine populations (17 percent of all populations) are predicted to include some fish that exceed the GLAG no consumption guideline (fig. 11). Of these nine populations, the models predicted that only two populations (suckers from the Fall River in Rocky Mountain NP and brook trout from Mildred Lake in Yosemite NP) were expected to have more than 50 percent of individuals in the population exceeding the threshold at some length along the size spectra for each population. The sizes at which 50 percent of individuals exceeded this threshold were 353 and 250 mm SL in suckers from the Fall River and Mildred Lake, respectively (fig. 11).

Discussion

Our results indicate that Hg bioaccumulation and risk to aquatic ecosystems of national parks in the Western United States is widespread, yet highly variable. Total Hg concentrations in individual fish ranged over two orders of magnitude across more than 1,400 individuals analyzed, with the lowest concentrations near 10 ng/g ww and the highest concentrations exceeding 1,100 ng/g ww. Importantly, this variation in Hg concentrations not only occurred across parks, but also among different sites within parks, even in light of site selection criteria that included restricting sites to mostly oligotrophic, headwater systems. These results highlight the significance of both local and regional-scale processes in determining Hg risk. As a result of this variability, many of the parks contained fish that were all below even the most conservative benchmarks of risk to fish (13 of 21 parks) and piscivorous birds (7 of 21 parks). Moreover, although all parks contained one or more fish that exceeded the GLAG unlimited consumption benchmark, most parks (14 of 21) did not have any fish in exceedance of the 300 ng/g ww EPA fish tissue criterion. Conversely, for those parks with fish that did exceed ecological and human health benchmarks, a substantial proportion of the fish had concentrations that were above those thresholds of concern. This was particularly true in the case of avian risk, where more than one-half of the fish sampled in several locations exceeded thresholds for risk to reproductive impairment in highly sensitive and moderately sensitive bird species. Although we did not directly assess risk to mammalian wildlife, the benchmarks chosen for avian consumers correspond closely with the no-observed-affect-level (NOAEL) benchmark for dietary Hg in mammalian wildlife (110 ng/g ww; Basu and Head 2010). Thus, our data suggest that mammalian wildlife also are likely to be at risk across these parks. Finally, although the geographic distributions in species occurrence impeded direct comparisons across all parks simultaneously, our data identify two regions of particular concern—(1) some parks in Alaska, as well as Glacier NP, had THg concentrations in large top predator fish species that approached or exceeded human consumption guidelines, and (2) some of the smallest "forage fish" sampled in the arid Southwest region had among the highest THg concentrations of the entire study, greatly exceeding wildlife risk thresholds for reproductive impairment. Although many plausible explanations exist for these patterns, investigating the potential mechanisms was beyond the scope of this study. Thus, more work is needed to better understand the landscape drivers of THg concentrations in these fish, as well as any deleterious effects these Hg concentrations might be having on resident biota in these parks. Additionally, information regarding the species composition, size ranges, and consumption rates of fish consumed by subsistence fishers in Alaskan parks is important for better characterizing human health risk in those parks.

THg concentrations in fish from this study generally are consistent with those of other studies in freshwater habitats of protected lands in the Western United States. For example, the Western Airborne Contaminants Assessment Project (WACAP) measured fish THg concentrations in 14 lakes from eight western national parks, six of which also were included in this study (Landers and others, 2008). Mean whole-body concentrations ranged from less than 40 ng/g ww in Glacier NP to slightly more than 200 ng/g ww in Noatak NP, Alaska (Landers and others, 2008). However, because the WACAP study was designed specifically as a screening study, there was limited replication within parks for evaluating site-specific variability in relation to regional variability. The need for such replication is indicated by a comparison of Hg concentrations reported from Glacier NP by Landers and others (2008) and the current study. Landers and others (2008) found Hg concentrations in cutthroat trout from two lakes in Glacier NP to be among the lowest measured in their study, whereas in the current study, concentrations from a third lake in Glacier were among the highest observed.

We specifically addressed this issue by sampling two parks (Mount Rainier and Rocky Mountain) intensively, with 17 and 19 different sites in each park, respectively. This higher resolution sampling revealed up to 20-fold variation in size-normalized fish THg concentrations among lakes within an individual park. This level of variation within a park is important to demonstrate and incorporate into park-level assessments, but is not unexpected given the well-established linkages between site-level characteristics and both methylation and bioaccumulation processes (Hall and others, 2008; Mitchell and Gilmour, 2008; Gabriel and others, 2009; Rolfhus and others, 2011; Hsu-Kim and others, 2013). Regardless, relatively little information exists about this variability in remote and high-altitude habitats that were the focus of this study. Recent studies do suggest that Hg bioaccumulation in osprey nesting in alpine habitats is regulated by a combination of local processes and atmospheric deposition (Guigueno and others, 2012). Similarly, Hg loadings to western Canadian alpine lakes was most strongly related to site-specific processes such as organic matter supply and catchment weathering (Phillips and others, 2011). Moreover, a recent study of THg concentrations in fish from sub-alpine lakes in the Wallowa-Whitman National Forest in northeastern Oregon and western Idaho focused on among-lake variation in fish THg concentrations within a relatively localized (Eagles-Smith and others, 2013). Consistent with the current study, mean fish THg concentrations among lakes varied by more than an order of magnitude (25–443 ng/g ww), even though all lakes were within 100 km of one another. Conifer density (basal area) within the catchment of each lake was the strongest predictor of fish THg concentrations among lakes (Eagles-Smith and others, 2013), highlighting the importance of watershed characteristics on downstream Hg cycling processes. At an even broader scale, the percentage of land covered by coniferous forests (coupled with wet Hg deposition) across six States in the south-central U.S. was an important factor influencing fish THg concentrations (Drenner and others, 2013). The hierarchical spatial design of the current study, combined with the gradient in landscape types, altitudes, and fish species that occur over the approximately 4,000 km distance, complicate a similar landscape scale analysis without even greater data density and replication among sites. However, our results support the notion that Hg risk is driven by a dynamic balance of local, regional, and global processes. What is unclear, is how the relative importance of these processes changes across locations, which should be addressed in focused future research in order to facilitate more informed management of Hg risk.

Although we do not formally address risk to the unique ecological communities across the parks studied, the model-adjusted fish THg concentrations provide a valuable index of relative risk to fish, wildlife, and human health. Our results suggest that although risk is low in many locations, there are substantial concerns about the impact of Hg on ecological health in other locations. In particular, THg concentrations in speckled dace, a small invertebrate-foraging minnow from Zion and Capitol Reef NPs, commonly exceeded some of the highest values in top predators, such as lake trout, from other parks. Size-normalized (50 mm) mean THg concentrations in these fish exceeded 200 ng/g ww in both parks, and mean concentrations in one site exceeded 300 ng/g ww. Moreover, across the range of fish sampled in these parks (approximately 70–100 mm SL), most fish exceeded these values. Because speckled dace occupy intermediate trophic positions, they serve as potential prey items for predatory fish and piscivorous birds, and dietary Hg concentrations at these levels are associated with biochemical and reproductive effects in fish (Depew and others, 2012a), as well as reproductive impairment in birds (Depew and others, 2012b). Moreover, it suggests that other invertebrate-feeding organisms in the park could be at risk. Amphibians and riparian- or aquatic invertebrate-feeding birds are two groups in particular that are both sensitive to Hg, and likely to be exposed due to their foraging ecology (Bergeron and others, 2011a, 2011b; Jackson and others, 2011; Metts and others, 2012; Willson and Hopkins, 2013).

THg concentrations in fish from Wrangell-St. Elias, Lake Clark, and Glacier NPs approached or exceeded all three benchmarks associated with reproductive impairment in piscivorous birds (90, 180, and 270 ng/g ww), and the EPA criterion for protection of human health. It is unclear whether these elevated concentrations are the result of enhanced MeHg production in these systems, or simply the fact that these were large fish (400–650 mm SL) that were likely many years old and had reached a growth asymptote in systems with only moderate Hg availability. This information gap is an important component to address with future work because it will help better gauge risk to park visitors and wildlife. Both Wrangell-St. Elias and Lake Clark support subsistence hunting and fishing activities for native and non-native rural Alaskans. If toxicological risk is mostly restricted to large, older fish, then guidelines can be better established to inform subsistence users of how to minimize exposure risk. Moreover, relatively few wildlife species are capable of consuming these large individuals, suggesting fewer species would be at risk from fish consumption and the species that are would likely be more easily monitored. Thus, understanding the relative risk posed across the size spectra of a population provides important insights into how that risk can be monitored and managed. Conversely, if certain habitats or locations within these parks are particularly sensitive to Hg methylation, the food web contamination may be more widespread, potentially necessitating a broader communication and monitoring strategy that might incorporate a more comprehensive array of subsistence foods and other wildlife taxa. Progress is already being made to address this potential concern, and USGS and Wrangell-St. Elias NP are currently completing a study to assess THg concentrations in several additional locations and species, as well as to evaluate the age structure of the fish being sampled. Although subsistence harvest is not a concern in Glacier NP, one of the species we measured, the bull trout, is listed as federally threatened under the endangered species act. Hg concentrations that we measured in bull trout from Lake McDonald in Glacier NP approached levels of concern for tissue-based fish toxicity. Because we only sampled fish from a single lake in Glacier NP, it is unclear if this is a widespread concern for toxicity to bull trout or their predators, but our data suggest that a follow-up study is warranted.

The unprecedented spatial breadth of this study facilitates the most thorough understanding of Hg bioaccumulation patterns in fish of western national parks to date. Our analysis and data compilation provide novel insight into the broad-scale distribution of Hg in fish communities of remote, protected habitats. Of note, many locations show limited Hg risk, and our data confirm that even with continued increases in global Hg releases (Sunderland and Selin, 2013), many protected NPS lands in the United States still exhibit relatively low Hg risk. However, our results also raise new questions and can be used to guide the development of future research to help manage the unexpected number of locations in this study where Hg contamination could be an ongoing or emerging threat to ecosystem health. The high degree of variation within parks and that the parks with the most intensive sampling had the greatest range in THg concentrations. These findings strongly suggest that more intensive research and monitoring efforts within parks will undoubtedly reveal both low and high risk environments. Thus, monitoring programs should not focus on a small number of "indicator" sites and we recommend that where feasible, newly established monitoring or investigation efforts focus on sampling at the largest spatial scale possible within a park to better understand this variability.

Additionally, our focus on high-altitude habitats provided little insight regarding the magnitude of risk in lower altitude lakes and wetlands. Recent work in Mount Rainier NP suggests that Hg bioaccumulation in wetlands and lakes is substantially higher in the lowest-altitude habitats (C.A. Eagles-Smith, U.S. Geological Survey, unpub. data, 2013). Additional monitoring in low-altitude habitats that integrate a broader range of watershed processes is warranted.

Beyond monitoring, a more quantitative understanding of the biogeochemical and food web factors that regulate within-park variability will facilitate greater certainty in predicting areas across these parks with the greatest risk likelihood. Future interdisciplinary research across a range of replicated aquatic habitat types that couples key biogeochemical process-level measurements with detailed food web processes and quantification of bioaccumulation pathways would provide valuable insight for developing these predictive models. A new collaborative, citizen-science effort between USGS, NPS, and University of Maine will make initial progress in this area by evaluating Hg bioaccumulation in biosentinel dragonfly larvae in relation to a range of biogeochemical and landscape drivers. One benefit of this upcoming effort is that the use of dragonflies as biosentinels will inform the parks about risk to aquatic habitats that are not fish-bearing. Another impending contribution in this realm will be provided by an empirical model being developed by the USGS Mercury Research Laboratory that will predict aqueous MeHg concentrations in surface water based on water chemistry and watershed structure. Coupling this model with biological assessments may provide more predictive power to assess ecological risk.

Although these suggestions will greatly increase the understanding of the drivers of Hg cycling in national parks, estimates of risk will continue to be speculative and of limited confidence without empirical research on toxicological responses in resident park biota. This lack of robust toxicological benchmarks for many taxa was a particular concern in the current study because few taxa extended across the large spatial extent and ecologically diverse habitats sampled. Although our choice of toxicological benchmarks were appropriate given the constraints of the current study, generalized benchmarks, such as our bird risk thresholds, do not provide definitive estimates of risk to specific park resources. Specifically, the data presented here suggest that toxicological assessments of key ecological endpoints in Capitol Reef, Zion, Lassen Volcanic, Yosemite, Wrangell-St. Elias, Lake Clark, and Glacier NPs are warranted. Direct assessments of the physiological, behavioral, and reproductive responses of wildlife and fish in these parks will be tremendously valuable for calibrating risk estimates that have been derived on different taxa in unrelated habitats, and provide a baseline from which meaningful benchmarks for evaluating the integrative risk of Hg toxicity to whole parks or ecosystems could be developed.

National Park Summaries

Capitol Reef National Park

All fish sampled from Capitol Reef NP were speckled dace, a small insectivorous "forage fish." Despite their intermediate position in the food web, these fish contained some of the highest Hg concentrations measured in this study, exceeding those measured in many of the largest predatory fish. As smaller fish, speckled dace serve as potential prey items for predatory fishes and piscivorous birds, and dietary Hg concentrations at these levels are associated with biochemical and reproductive effects in fish (Depew and others, 2012a), as well as reproductive impairment in birds (Depew and others, 2012b). Moreover, such high Hg concentrations suggest that other invertebrate-feeding organisms in the park could be at risk. Amphibians and riparian- or aquatic invertebrate-feeding birds are two groups in particular that are both sensitive to Hg, and likely to be exposed due to their foraging ecology (Bergeron and others, 2011a, 2011b; Jackson and others, 2011; Metts and others, 2012; Willson and Hopkins, 2013). However, we only sampled a single species at three sites in Capitol Reef NP, all of which were along the Freemont River. Therefore, we are unable to speculate whether the fish communities throughout the park contain similarly high concentrations, or whether these values were the result of a localized process. Future sampling of additional sites and species within the park is necessary to fill this important knowledge gap.

Crater Lake National Park

Kokanee and rainbow trout were sampled from two sites in Crater Lake NP. The two species had similar Hg concentrations and were among the lowest measured in this study, ranging from 16.8 to 76.0 ng/g ww. Overall our results indicate that Hg concentrations in fish from Crater Lake are unlikely to put fish or avian consumers at risk.

Denali National Park

We sampled northern pike from a single lake in Denali NP. Despite their large size and position as top predators, northern pike from Denali NP had the lowest size-normalized Hg concentrations in the 400 mm SL size class (47.4 ng/g ww at 400 mm SL) and some of the lowest observed in the entire study. In fact, the THg concentrations in northern pike from Denali were similar or less than THg concentrations in smaller fish species from the other parks in Alaska (Wrangell-St. Elias, Lake Clark, Glacier Bay). Moreover, past work in Denali found similarly low THg concentrations in burbot (*Lota lota*) and whitefish (Landers and others, 2008). Such low Hg concentrations may be the result of this site's isolated nature. However, at other similarly isolated sites in Alaska, northern pike have been found to have consistently higher Hg concentrations (110–1,500 ng/g ww on average; Jewett and Duffy, 2007). Any additional sampling from Denali NP should aim to assess whether these data are representative of other water bodies and species in the park.

Glacier Bay National Park

We sampled Dolly Varden from two lakes and one stream in Glacier Bay NP. Mean THg concentrations across these three sites (90.2 ng/g ww) were slightly above the global mean concentration in fish across all sites and parks in this study (77.7 ng/g ww). When modeled at 200 mm, the mean Hg concentration in fish from Glacier Bay NP (102.3 ng/g ww) was among the highest in the size class. However, we also observed a 3-fold variation in the mean concentrations among the three lakes indicating that local processes are likely important drivers of Hg bioaccumulation in fish within this park. The THg concentrations measured in Dolly Varden from Glacier Bay NP were similar to concentrations (40–230 ng/g ww) found in Dolly Varden from other sites in Alaska (Jewett and Duffy, 2007).

Glacier National Park

Lake whitefish and bull trout were sampled from Lake McDonald in Glacier NP. Both species had high Hg concentrations relative to the mean across all fish in the study, and after accounting for the effects of size and species, fish from Glacier NP were some of the highest in the large (400 mm SL) size class. Mercury concentrations in Glacier NP fish approached or exceeded the EPA criterion for protection of human health and the level at which reproductive impairment to piscivorous birds could occur. Additionally, Hg concentrations in many individuals exceeded the level at which tissue-based toxicity to fish is a concern. This is particularly important considering bull trout is federally listed as threatened under the endangered species act. Because we only sampled fish from a single lake, we cannot assess whether these data are representative of other lakes in the park. However, Landers and others (2008) found that fish from two other lakes within Glacier NP had the lowest Hg concentrations among the 14 lakes measured across national parks in the West. These contrasting results suggest that substantial variation in Hg concentrations exists among water bodies within Glacier NP and more detailed analysis of local processes may be necessary to effectively manage risk.

Grand Canyon National Park

At the three sites we sampled in Grand Canyon NP, unadjusted mean THg concentrations (76 ng/g ww) were similar to the average across all fish in the study. However, size adjusted (200 mm SL) THg concentrations were among the highest in the size class (101.2 ng/g ww) across all parks. Moreover, size-adjusted THg concentrations in Shinumo Creek (178.8 ng/g ww) were more than 2-fold higher than those in the other two sites (71.5 and 81.0 ng/g ww), suggesting that Hg exposure within the park is variable.

Grand Teton National Park

Mercury concentrations in fish from the two lakes and one stream sampled in Grand Teton NP were consistently among the lowest measured in this study. It is unclear whether these low concentrations are the result of low local inputs of Hg, the Hg methylation potential of the water bodies measured, or some biological characteristic of the fish.

Great Basin National Park

We sampled brook trout from two creeks and a lake in Great Basin NP. Among these three sites Hg concentrations were uniformly among the lowest measured in the current study suggesting that at the time of sampling the ecological risk posed by Hg in these systems is likely low. Atmospheric dry deposition in Great Basin NP was among the highest measured in six western national parks (Wright and others, 2014).

Great Sand Dunes National Park

Mercury concentrations in fish from one of the two sites (Medano Lake; 70.1 ng/g ww) sampled in Great Sand Dunes NP were similar to the average across parks, whereas fish from the second site (Sand Creek; 52.7 ng/g ww) were considerably lower. Based on these data, the risk of Hg exposure to wildlife is likely low at these sites; however, data from these two sites may not adequately characterize risk at other locations within the park.

Lake Clark National Park

Lake trout from three lakes in Lake Clark NP were included in this study. Two of these sites (Lake Clark and Lake Kontrashibuna) were sampled in both 2011 and 2012, although Hg concentrations were not significantly different between the 2 years. Although THg concentrations across all sites within the park were among the highest observed in the 400 mm SL size class, there was considerable variation among lakes within the park. The mean THg concentration in Lake Clark (365.2 ng/g ww) was more than 3-fold higher than that in Telaquana Lake (109.0 ng/g ww) and 1.8 than that in Lake Kontrashibuna (204.0 ng/g ww). At the lower end of this range, THg concentrations pose limited risk to park wildlife and human users. However, the concentrations observed in Lake Kontrashibuna exceeded the benchmark for reproductive impairment to piscivorous birds, and the tissue-based criterion for fish toxicity. Moreover, fish from Lake Clark exceeded these benchmarks and the EPA criterion for protection of human health.

Lassen Volcanic National Park

There was greater than 4-fold variation in mean fish THg concentrations among the three lakes that brook trout were sampled from in Lassen Volcanic NP. Mercury concentrations were lowest in Summit Lake (43.7 ng/g ww), whereas Ridge Lake (111.3 ng/g w) and Horseshoe Lake (180.7 ng/g ww) had concentrations above the mean concentration for all fish in this study. Some of this variation in Hg concentrations among lakes in the park was due to the size of fish. After adjusting fish THg concentrations to a 200 mm SL fish size, the variation among lakes was substantially reduced. Size-based risk assessments were conducted for Horseshoe and Summit Lakes, and the resulting risk profiles contrast sharply. Our modeled risk estimates suggest that there is limited toxicological risk to fish, wildlife or humans across the fish size range in Summit Lake. Conversely, brook trout from Horseshoe Lake were modeled to reach concentrations of concern for tissue-based fish toxicity and risk to even the least sensitive species of avian consumers within the range of sizes sampled. Furthermore, a high proportion of fish from Horseshoe Lake were modeled to exceed the EPA criterion for protection of human health at sizes exceeding 280 mm SL.

Mesa Verde National Park

Speckled dace were sampled from a single location, the Mancos River, in Mesa Verde NP. Despite the small size of these fish (43–69 mm SL), their THg mean concentration (74.9 ng/g ww) was to the study-wide mean value. However, the THg concentrations from speckled dace from the lone site in Mesa Verde NP were much lower than those measured in dace from nearby Capitol Reef and Zion NPs. This difference suggests that the high concentrations observed in dace from these other parks is not solely due to some aspect of this species biology, but may reflect environmental processes occurring at the site or watershed scale.

Mount Rainer National Park

Mount Rainer NP is one of two parks in which we sampled a large number of sites in order to characterize intra-park variation. Although the average fish THg concentration across the 17 sites from which fish were sampled (71.5 ng/g ww) was similar to the study-wide mean across all fish, we found a 12-fold range in mean concentrations among sites before accounting for the effects of size. Size-normalized Hg concentrations in the 200 mm SL size class were even more variable among sites. The highest mean concentration at a site (193.2 ng/g ww) was nearly 23-fold higher than the lowest site mean concentration (8.5 ng/g ww). Such extreme variation among sites in Mount Rainer NP suggests that local-scale factors play an important role in determining the ecological risk of Hg and emphasizes the need to sample from multiple locations in order to accurately characterize Hg risk to park resources as a whole.

North Cascades National Park

All three of the sites sampled in North Cascades NP had mean Hg concentrations below the average of all fish sampled in this study. After standardizing the fish to 200 mm SL, the mean Hg concentration in fish from North Cascades NP (73.3 ng/g ww) was similar to the mean of 200 mm fish across all parks. These data suggest that the ecological risk of Hg at the three sites sampled in North Cascades NP is low relative to many of the other parks. However, given the variation demonstrated among sites in this study, the limited sampling it is difficult to generalize across other sites in the park without more extensive sampling.

Olympic National Park

Mercury concentrations in fish across the five lakes sampled in Olympic NP (85.0 ng/g ww) were slightly higher than the mean concentration across the entire study. However, we found substantial variability in fish THg concentrations among lakes. In particular, size-adjusted THg concentrations in fish from Hoh Lake (253.0 ng/g ww) were more than 3.5-fold higher than those in Gladys Lake (71.5 ng/g ww), which had the lowest site-specific THg concentrations in the park. Our models of Hg risk to consumers reflects these higher concentrations and suggest that fish in Hoh Lake are likely to approach or exceed the EPA criterion for protection of human health and the avian benchmark for reproductive impairment above 180 mm. These results are similar to those reported by Landers and others (2008), who found Hg concentrations in fish from Hoh Lake were among the highest measured during their screening of national parks.

Rocky Mountain National Park

We sampled 19 sites in Rocky Mountain NP, allowing us to more rigorously examine within park variability in Hg concentrations among sites. Overall, the mean Hg concentration in fish from Rocky Mountain NP (ng/g ww) was slightly lower than the study-wide mean across all parks. Across sites within Rocky Mountain NP, concentrations varied by more than 6.5-fold between the sites with the lowest (Lake Haiyaha; 19.8 ng/g ww) and highest (Mirror Lake; 121.2 ng/g ww) concentrations. Our models of Hg risk suggest that most of the sites we sampled from Rocky Mountain NP had fish THg concentrations below the benchmarks of risk that we selected. However, brook trout from Mirror Lake and suckers from the Colorado and Fall Rivers are likely to exceed both the EPA human health criterion level and the avian reproductive impairment benchmark when the fish exceed 250 mm SL. In addition, we examined inter-annual variation at four sites in Rocky Mountain NP. At two of these sites (Nanita Lake and Mirror Lake), we found that there was no significant difference between the years sampled, whereas fish THg concentrations at Poudre Lake and Ypsilon Lake were significantly lower in 2012 compared to 2009. Additionally, two of the lakes sampled in this study (Mills Lake and Lone Pine Lake) also were sampled for another study in 2003 (Landers and others, 2008). Rainbow trout THg concentrations in Mills Lake were higher (87.5 ng/g) than in 2003 (less than 60 ng/g ww). Whereas THg concentrations in brook trout from Lone Pine Lake were lower (57.8 ng/g ww) than in 2003 (approximately 70 ng/g ww). Consistent with the results from Mount Rainer NP, our more intensive sampling at Rocky Mountain NP indicates substantial variation among sites, highlighting the importance of sampling multiple sites when characterizing the risk of Hg exposure within a national park.

Sequoia–Kings Canyon National Park

Four high-altitude lakes were sampled from Sequoia–Kings Canyon NPs. The mean fish THg concentration across all these sites (43.8 ng/g ww) was below the study-wide average across all fish analyzed in this study. Size-adjusted (200 mm SL) THg concentrations remained similarly low in all but Center Basin Lake, where the mean THg concentrations in golden trout increased to 207.3 ng/g ww. However, based on the size distribution of our sample of 30 individuals, 200 mm fish are likely rare in this population. Furthermore, two of the lakes from Sequoia–Kings Canyon NPs were sampled in 2008 (Center Basin Lake) or 2009 (Kern Point Lake) and again in 2012. Comparisons between sampling years indicate that Hg concentrations in fish from Kern Point Lake were remained similar between years, whereas Hg concentrations in Center Basin Lake fish were higher in 2012 than in 2008. Landers and others (2008) reported Hg concentrations higher than those observed in the current study from two additional high-altitude lakes in Sequoia–Kings Canyon NPs (Pear and Emerald Lakes), demonstrating that even among high-altitude lakes with potentially limited Hg inputs, fish THg concentrations can be quite variable.

Wrangell-St. Elias National Park

Lake-specific mean fish THg concentrations from the three lakes sampled at Wrangell-St. Elias NP spanned nearly 8-fold between rainbow trout from Summit Lake (53.3 ng/g ww) and Lake Trout from Tanada Lake (416.6 ng/g ww), which had the highest mean Hg concentration of any site from all of the parks examined in this study. After size-adjusting fish THg concentrations, we found that concentrations in 200 mm SL fish from Copper Lake (147.4 ng/g ww) were substantially higher than those from Summit Lake (65.4 ng/g ww). The mean size-adjusted THg concentration for 400 mm SL fish from Copper Lake (242.0 ng/g ww) and Tanada Lake (321.2 ng/g ww) approached or exceeded the

EPA fish tissue criterion, the avian reproductive impairment benchmark, and the tissue-based criterion for fish toxicity. Tanada Lake was one of the few lakes in the present study in which a relatively high proportion of fish were expected to exceed upper benchmarks for fish toxicity, reproductive impairment in low sensitivity birds, and at the largest sizes sampled, the no-consumption guideline proposed by the GLAG. The cause of such high concentrations in some fish from Wrangell-St. Elias NP is unclear and may reflect some combination of local biogeochemical processes that stimulate methylmercury production, the ecological structure of the lake's food web, or physiological characteristics of the fish themselves. For example, fish with high Hg concentrations may be exceptionally old and thus may have accumulated large quantities of Hg over their extended lives. Alternatively, fish in such northern lakes may grow slowly compared to fish in warmer, more productive systems resulting in little biodilution of ingested Hg through somatic growth.

Yellowstone National Park

Across all lakes, mean fish THg concentrations in Yellowstone NP (101.2 ng/g ww) were above the study-wide mean value from all parks. Size adjusted THg concentrations for 200 mm SL fish (119.8 ng/g ww) were among the highest measured in this study. Importantly, natural geothermal Hg sources occur throughout Yellowstone, though it is unclear if those sources play a role in fish bioaccumulation.

Yosemite National Park

The two sites sampled at Yosemite NP display starkly contrasting patterns of Hg concentrations. At one site, Spillway Lake, fish THg concentrations were uniformly low; the site mean (38.7 ng/g ww) was one-half of the study-wide mean across all parks. In contrast, THg concentrations in fish from Mildred Lake (174.4 ng/g ww) were more than twice the study-wide mean. Moreover, individual fish concentrations had a 15-fold range within the lake. Importantly, the highest individual fish THg concentration in the entire study (1,109 ng/g ww) was measured in a brook trout from Mildred Lake. This was the only fish that exceeded the no-consumption guideline proposed by the GLAG, and was 1.6-times greater than the next highest concentration measured in this study. The source of such variation within a lake is unclear, but future efforts should aim to identify the factors regulating accumulation in individuals and determining whether this variability is representative of other lakes in the park.

Zion National Park

As in Capitol Reef NP, all fish sampled from Zion NP were speckled dace and displayed high Hg concentrations (241.5 ng/g ww) for a small, low trophic level fish. At both sites along the Virgin River, mean Hg concentrations were more than 3-fold higher than the study-wide mean concentration. Furthermore, variation within and between these isolated sites was very low. As in Capitol Reef, the potential for dace to serve as prey for higher trophic position consumers, and indicators of Hg exposure in other invertebrate-feeding taxa, suggest that more research should examine factors regulating Hg accumulation in the lower trophic levels of these systems.

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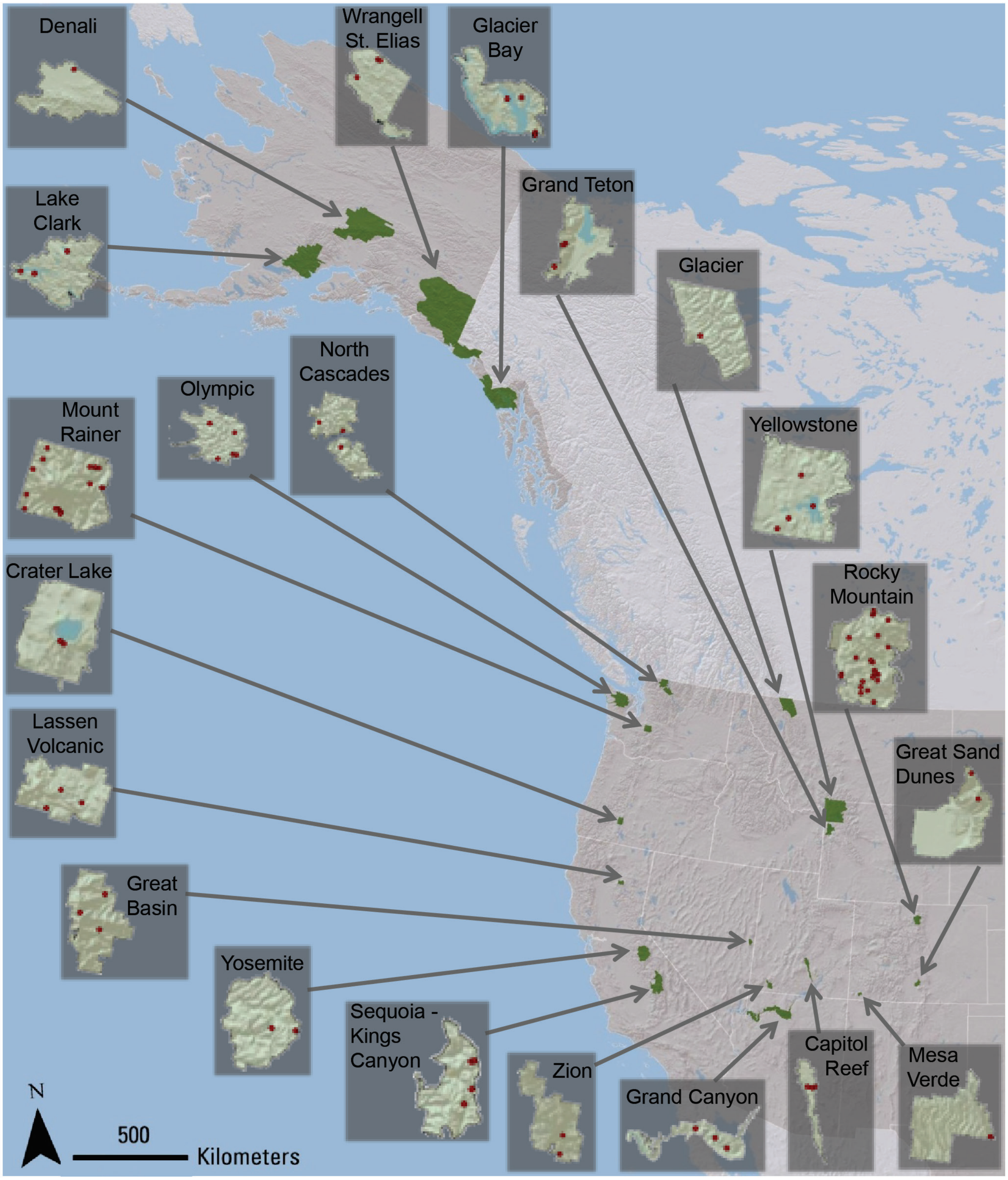

Figure 1. Spatial distribution of the 21 national parks in the Western United States sampled in this study between 2008 and 2012. Shaded polygons on base map indicate park boundaries. Red circles within each inset park boundary indicate specific sampling locations for each park.

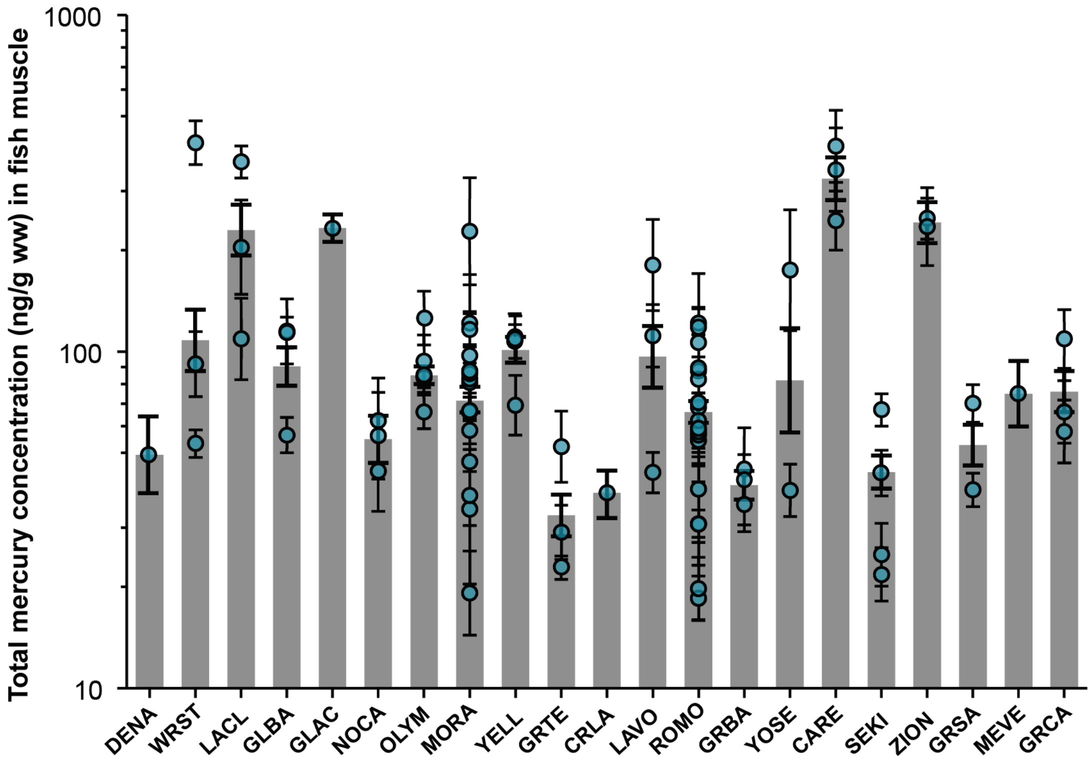

Figure 2. Total mercury (THg) concentrations (ng/g ww) in fish muscle from 21 national parks in the Western United States. Bar height and bolded error bars represent the park-wide geometric mean THg concentration and 95-percent confidence interval, respectively. Colored circles and thin error bars represent site-specific geometric mean THg concentration and 95-percent confidence interval, respectively. Parks are ordered by decreasing latitude, and park abbreviations can be referenced to park names in table 1.

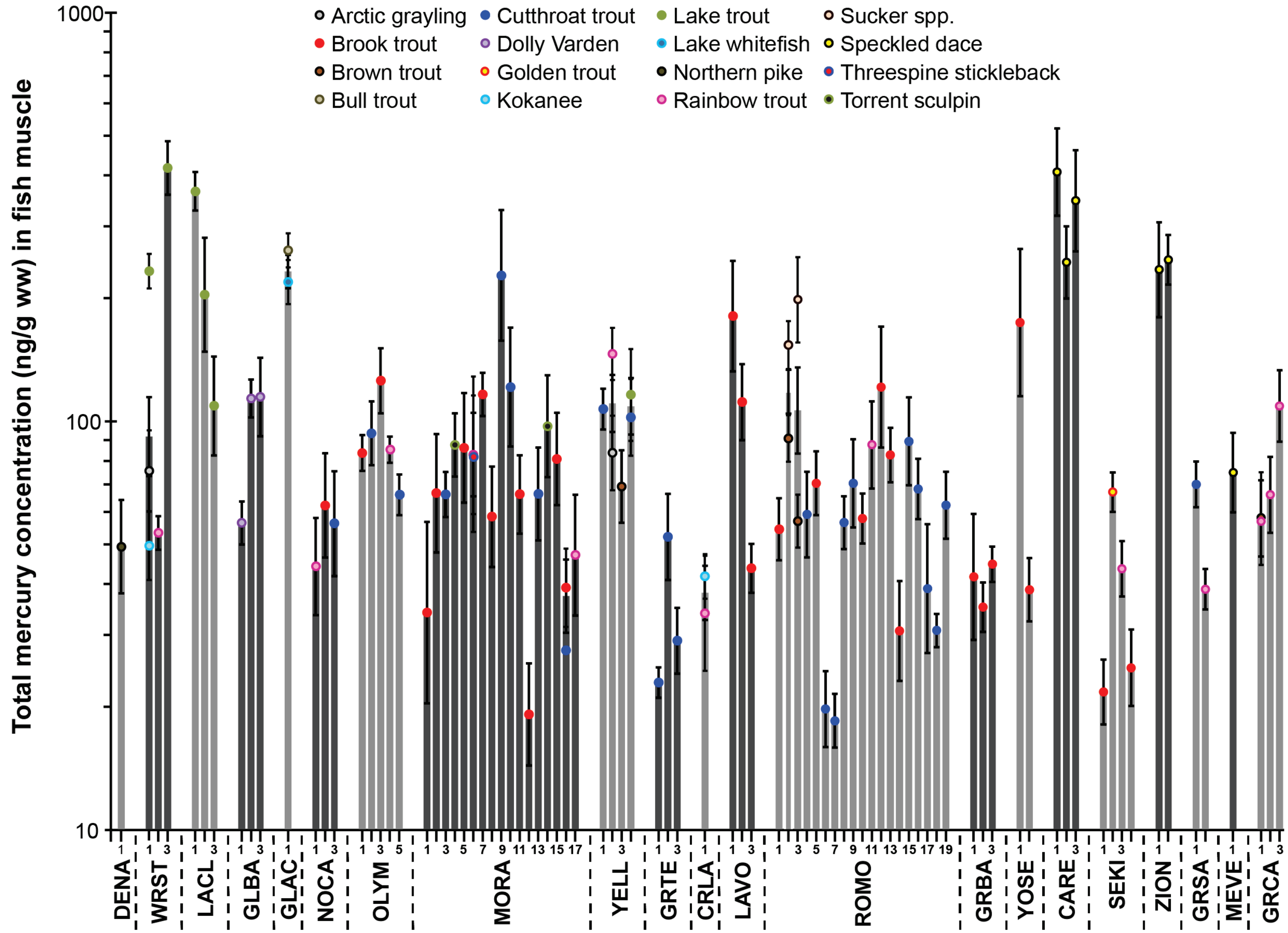

Figure 3. Total mercury (THg) concentrations (ng/g ww) in fish muscle from individual sites within 21 national parks in the Western United States. Bar height and bolded error bars represent site-specific geometric means and 95-percent confidence intervals. Circles represent species-specific geometric means in 95-percent confidence intervals within each site. Parks are ordered by decreasing latitude, and park abbreviations and site numbers can be referenced to park name in table 1.

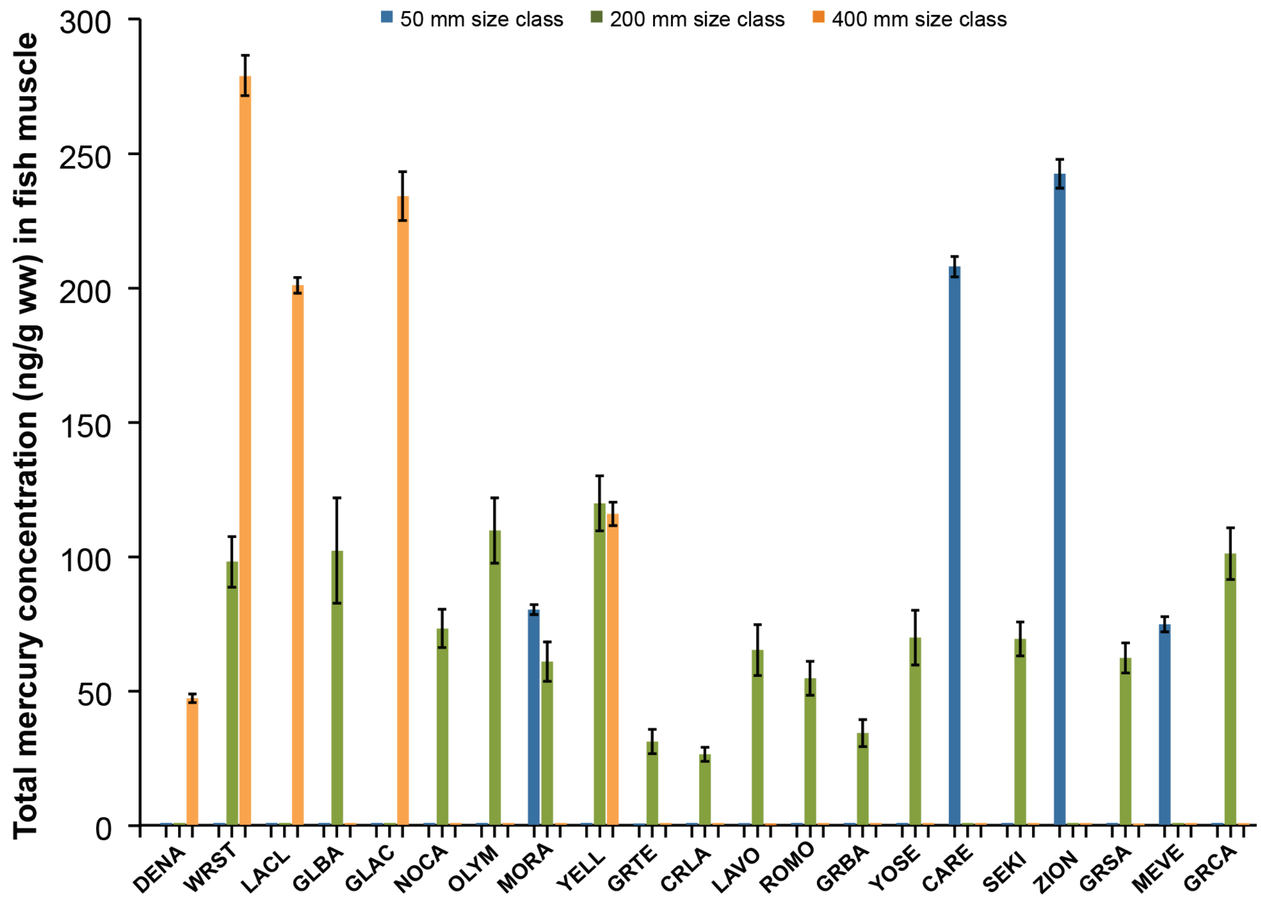

Figure 4. Size-normalized least-square (LS) total mercury (THg) concentrations in fish from 21 national parks in the Western United States. LS mean concentrations control for the influence of fish species and site within each park. Blue bars are LS means for fish species normalized to 50 mm size, green bars are LS means for fish species normalized to 200 mm size, orange bars are LS means for fish species normalized to 400 mm size. Parks are ordered by decreasing latitude, and park abbreviations can be referenced to park name in table 1. Species comprising each size category is discussed in section, “Methods.”

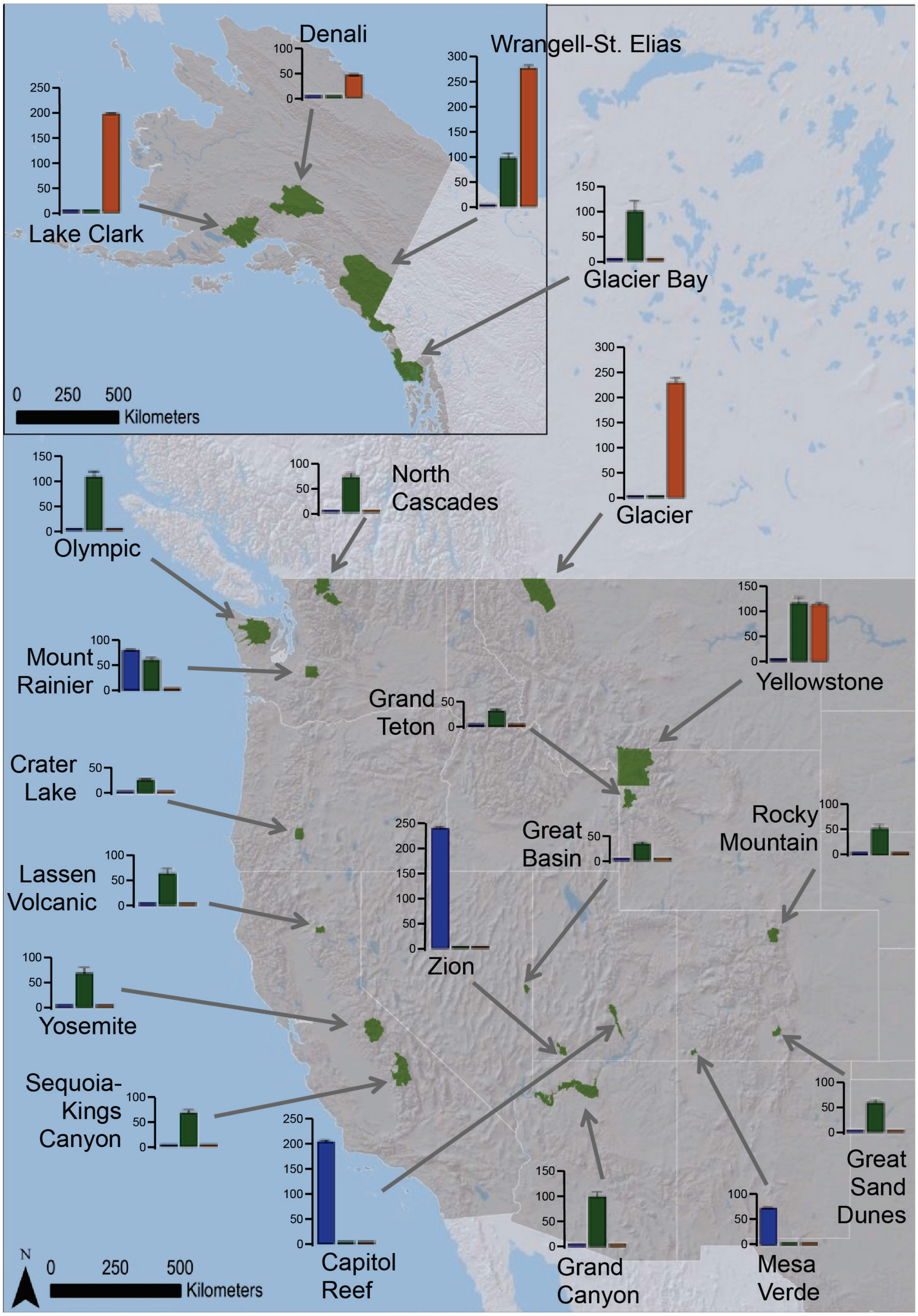

Figure 5. Least-square mean fish muscle total Hg concentrations (ng/g ww) in three size-normalized classes (blue/left = 50 mm, green/middle = 200 mm, orange/right = 400 mm) from 21 national parks in the Western United States.

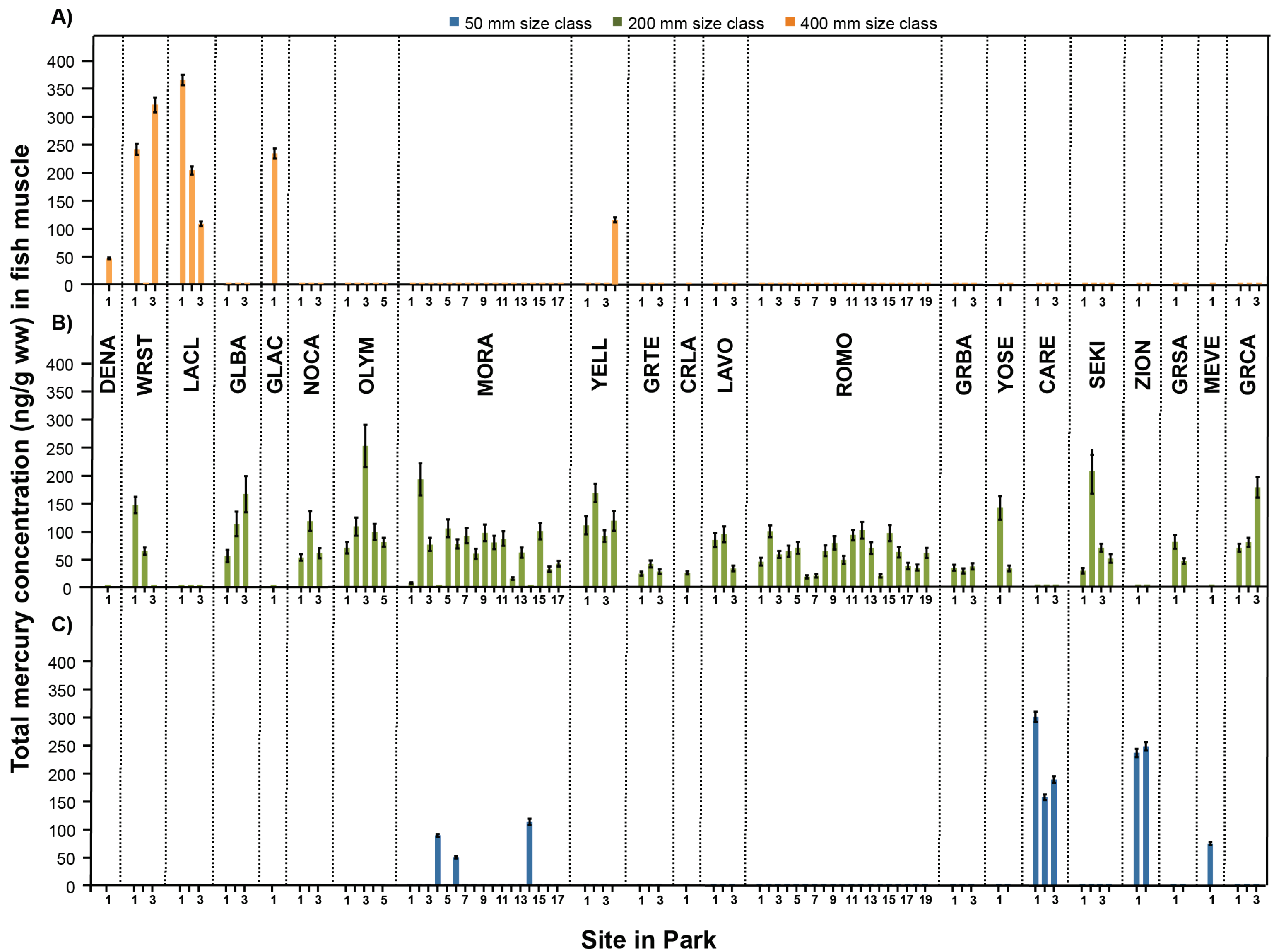

Figure 6. Least-square mean (whiskers = standard error) total mercury (THg) concentrations in fish muscle from individual sites in 21 national parks in the Western United States. Concentrations are size normalized for (A) 400 mm SL, (B) 200 mm SL, (C) 50 mm SL fish. Park abbreviations and site numbers can be referenced to park name in table 1.

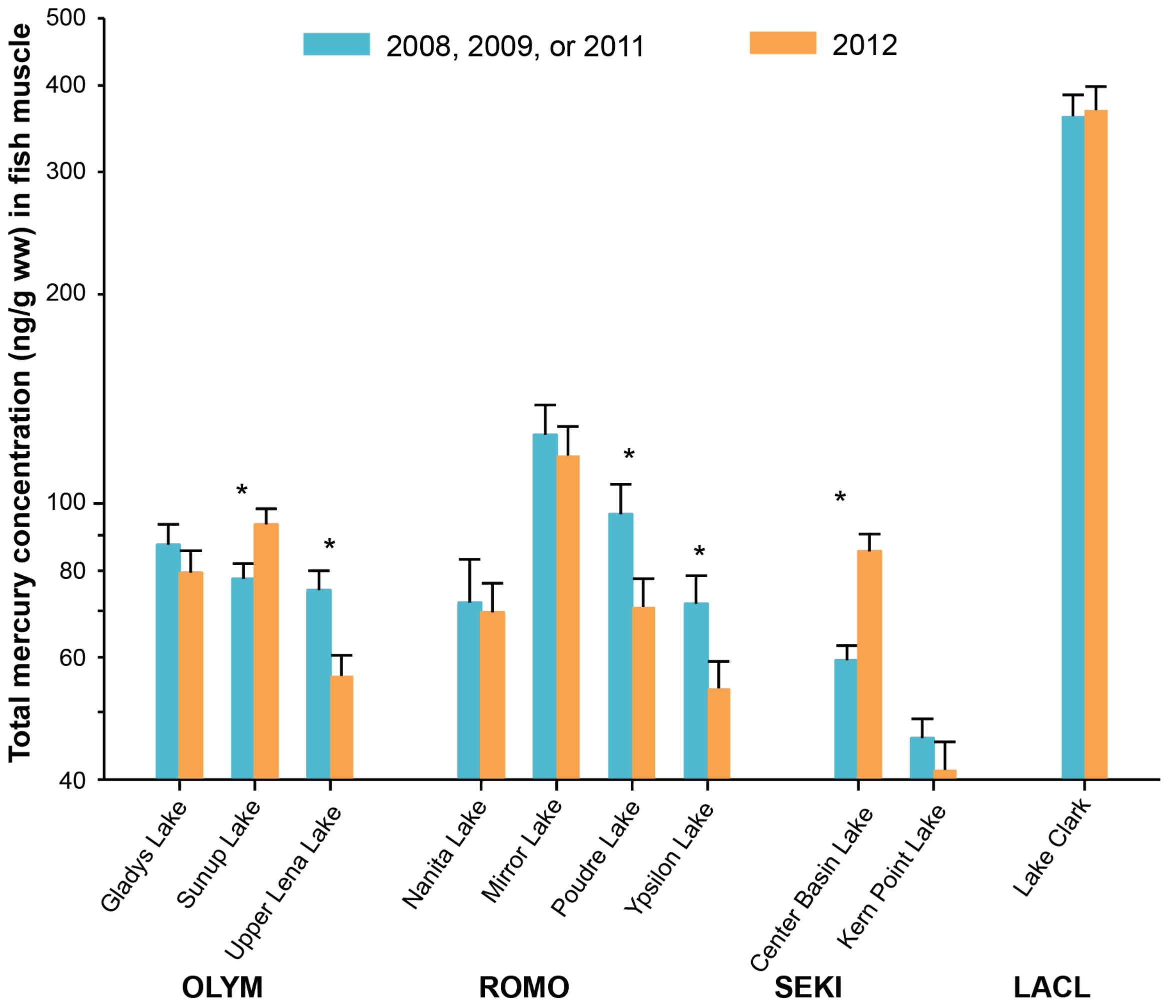

Figure 7. Inter-annual comparisons of fish total mercury concentrations (ng/g ww) in muscle tissue from 10 sites in four national parks in the Western United States. Error bars represent standard error. Asterisks indicate significant differences ($p<0.05$) between years within a site. OLYM, Olympic NP, ROMO, Rocky Mountain NP; SEKI, Sequoia-Kings Canyon NP; and LACL, Lake ClarkNP.

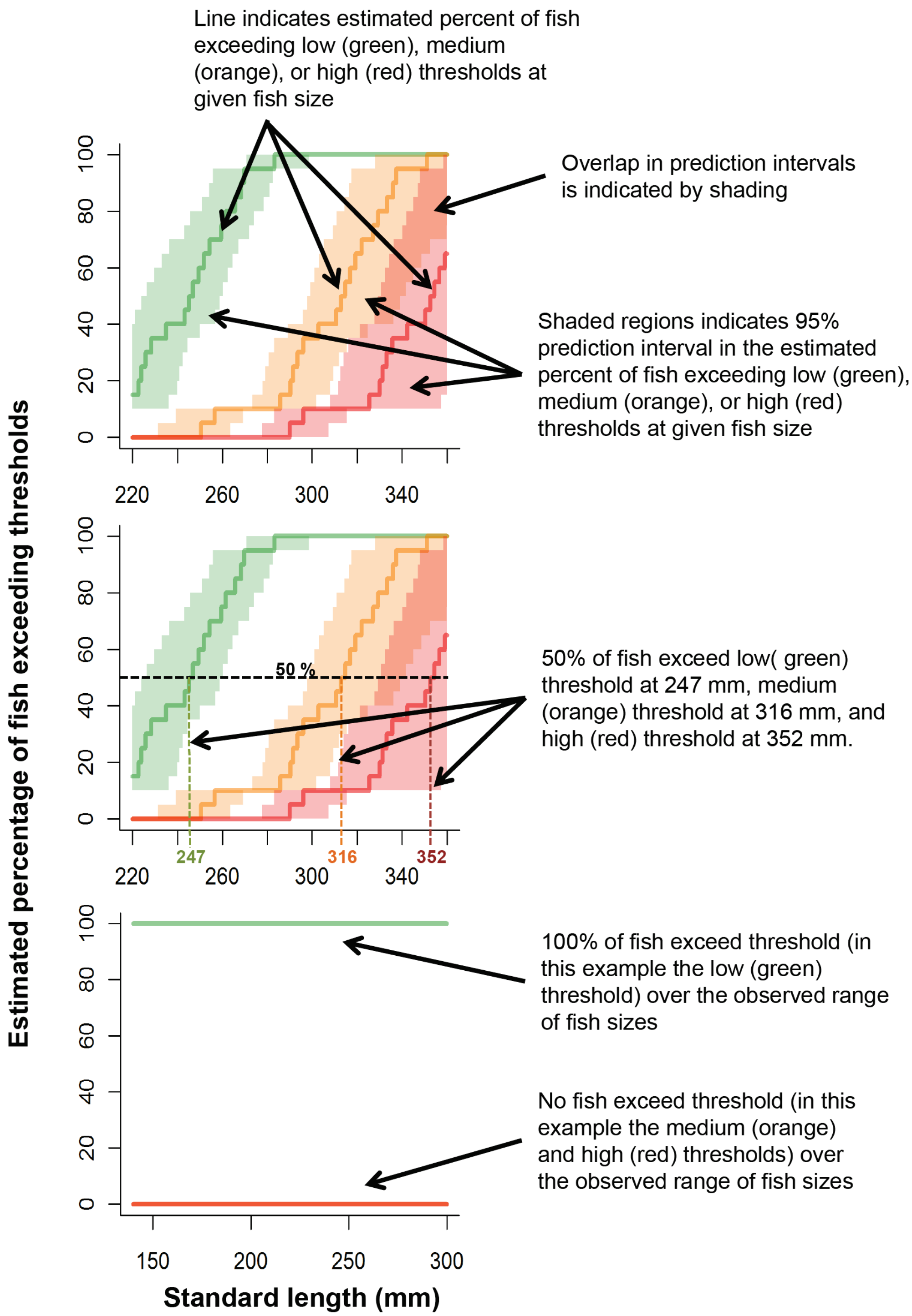

Figure 8. Example size-risk profile to evaluate proportion of fish exceeding defined toxicological benchmarks. Each color represents a defined risk benchmark. Solid lines indicated the estimated proportion of fish exceeding benchmarks across the size spectrum. Shaded regions represent the 95-percent prediction interval around the benchmark. A solid line at 0 or 100 percent indicates that all no fish or all fish, respectively, exceed the specified benchmark across the size spectra.

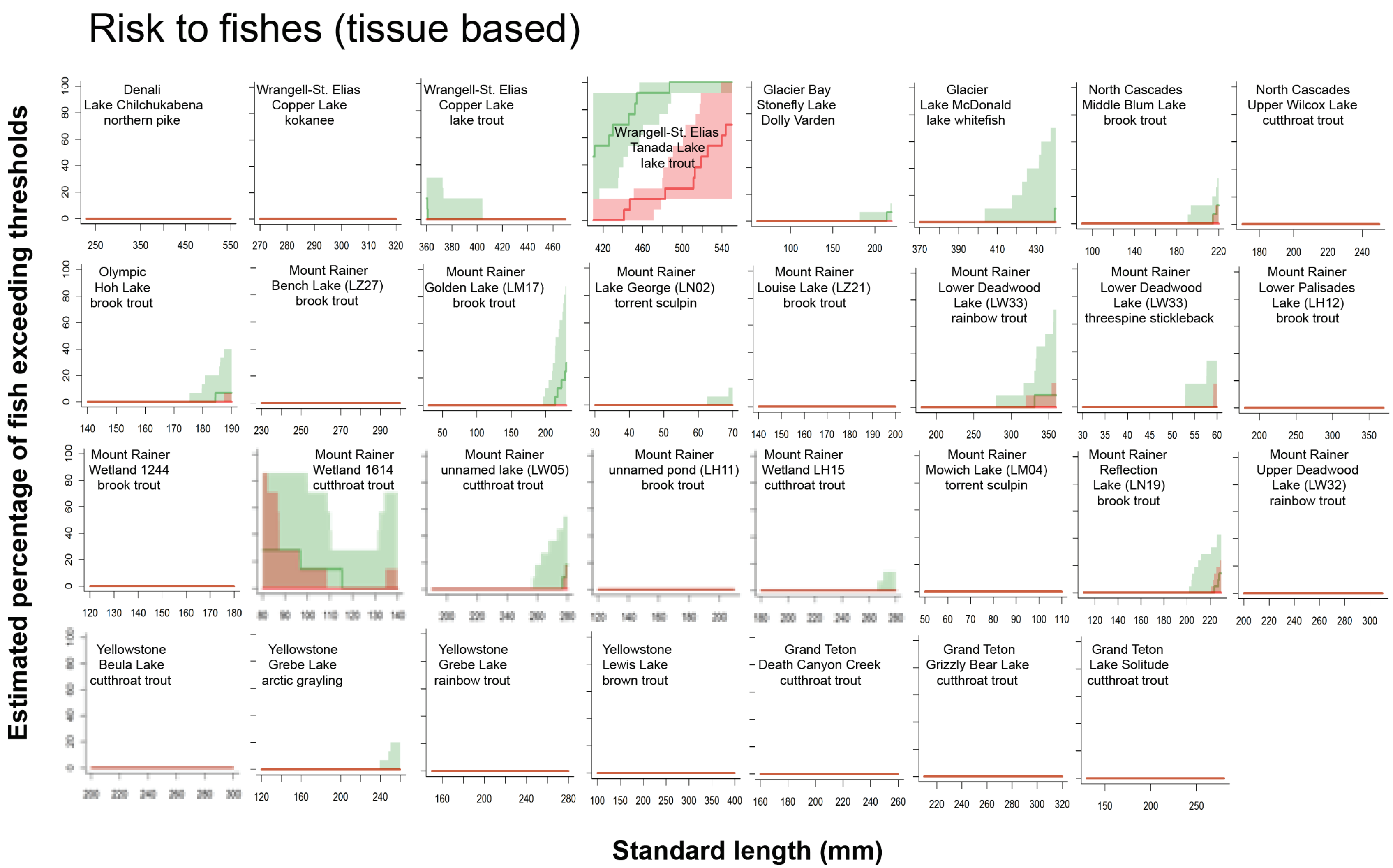

Figure 9. Modeled percentage of fish across the size range sampled in each population with whole-body total mercury concentrations that exceed benchmarks at a given size for a generic no-observed-effects residue (green; 200 ng/g ww), and lowest-observed-effects-residue (red; 300 ww) for fish health at 21 national parks in the Western United States. Parks are ordered by decreasing latitude. Figure continues on next page.

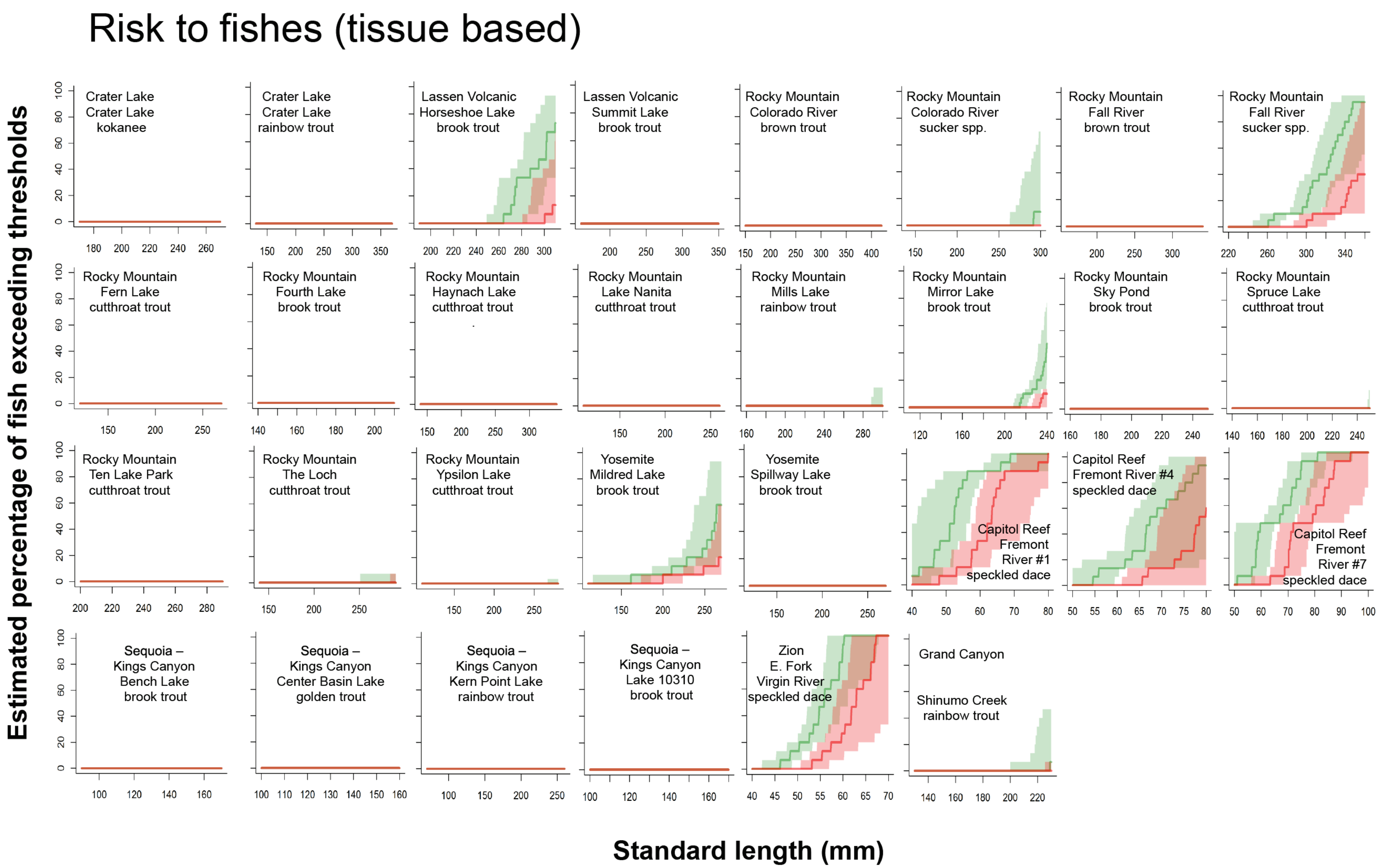

Figure 9.—Continued. Modeled percentage of fish across the size range sampled in each population with whole-body total mercury concentrations whole-body total mercury concentrations that exceed benchmarks for a generic no-observed-effects residue (green; 200 ng/g ww), and lowest-observed-effects-residue (red; 300 ng/g ww) for fish health in 21 national parks in the Western United States. Parks are ordered by decreasing latitude.

Risk to birds (diet based)

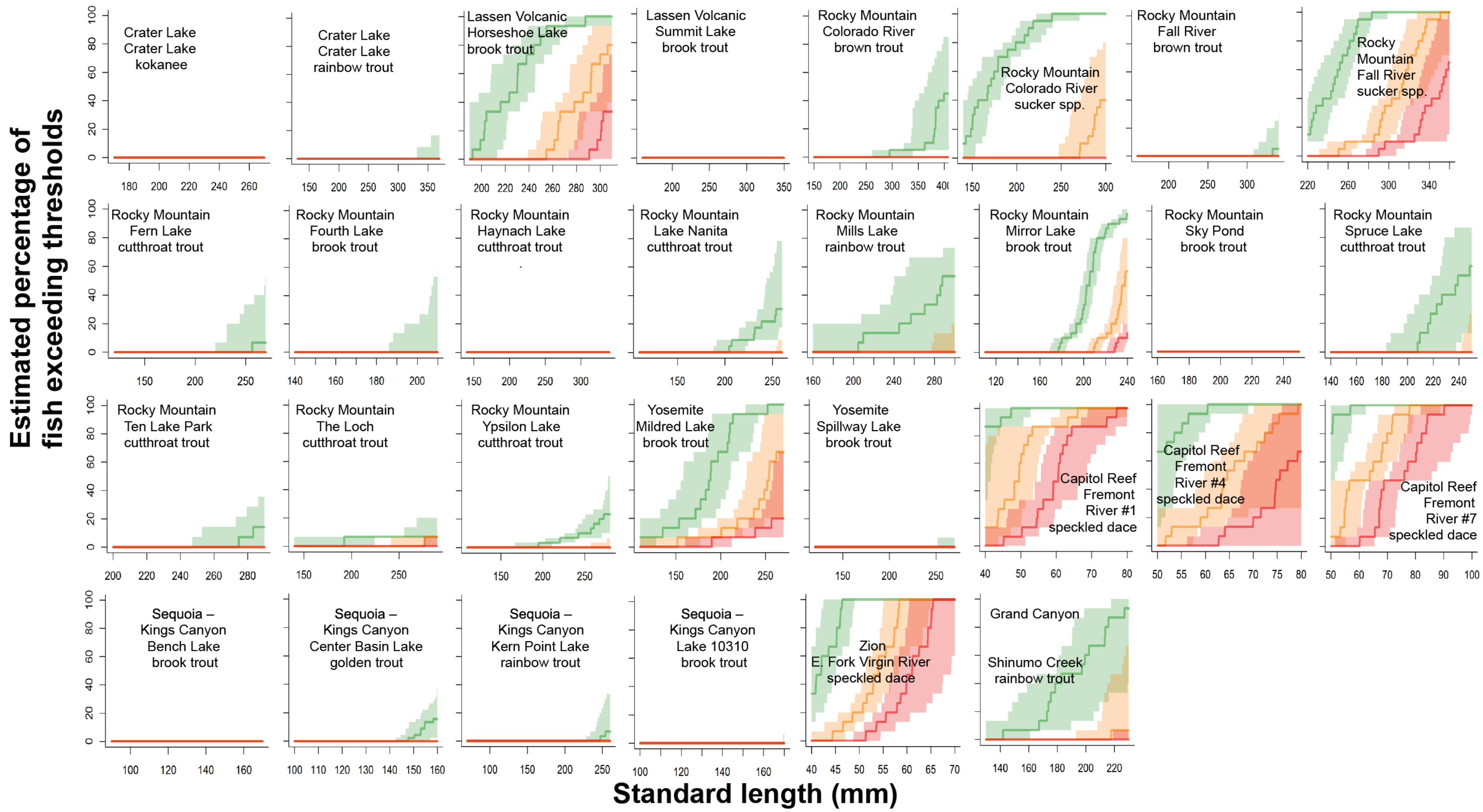

Figure 10. Modeled percentage of fish across the size range sampled in each population with whole-body total mercury concentrations that exceed dietary toxicity benchmarks for piscivorous birds that can be classified as high (green; 90 ng/g ww), medium (orange; 180 ng/g ww; common loon reproductive impairment), or low (red; 270 ng/g ww) sensitivity in 21 national parks in the Western United States. Parks are ordered by decreasing latitude. Figure continues on next page.

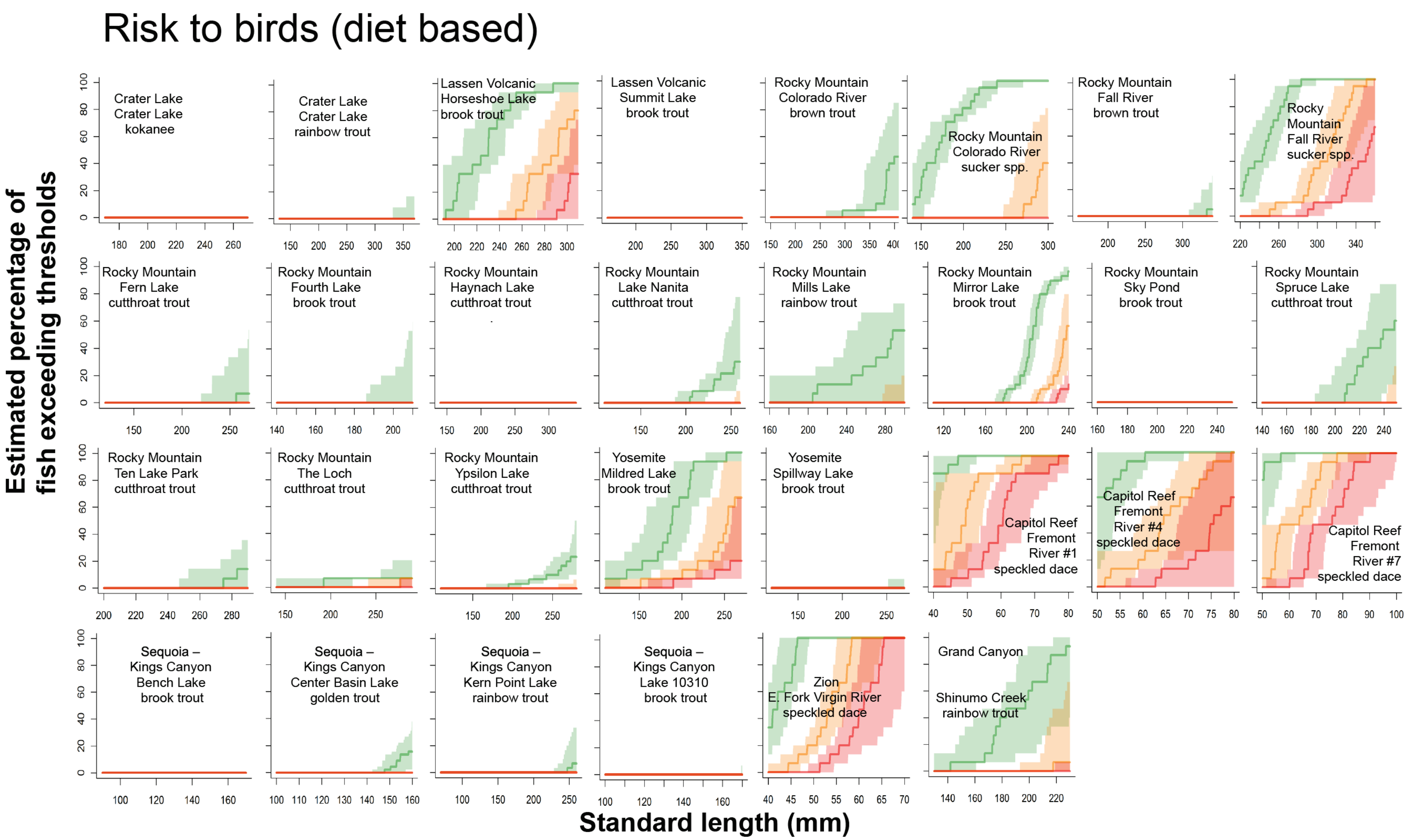

Figure 10.—Continued. Modeled percentage of fish across the sampled size range in each population with whole-body total mercury concentrations that exceed dietary toxicity benchmarks for piscivorous birds that can be classified as high (green; 90 ng/g ww), medium (orange; 180 ng/g ww; common loon reproductive impairment), or low (red; 270 ng/g ww) sensitivity in 21 national parks in the Western United States. Parks are ordered by decreasing latitude.

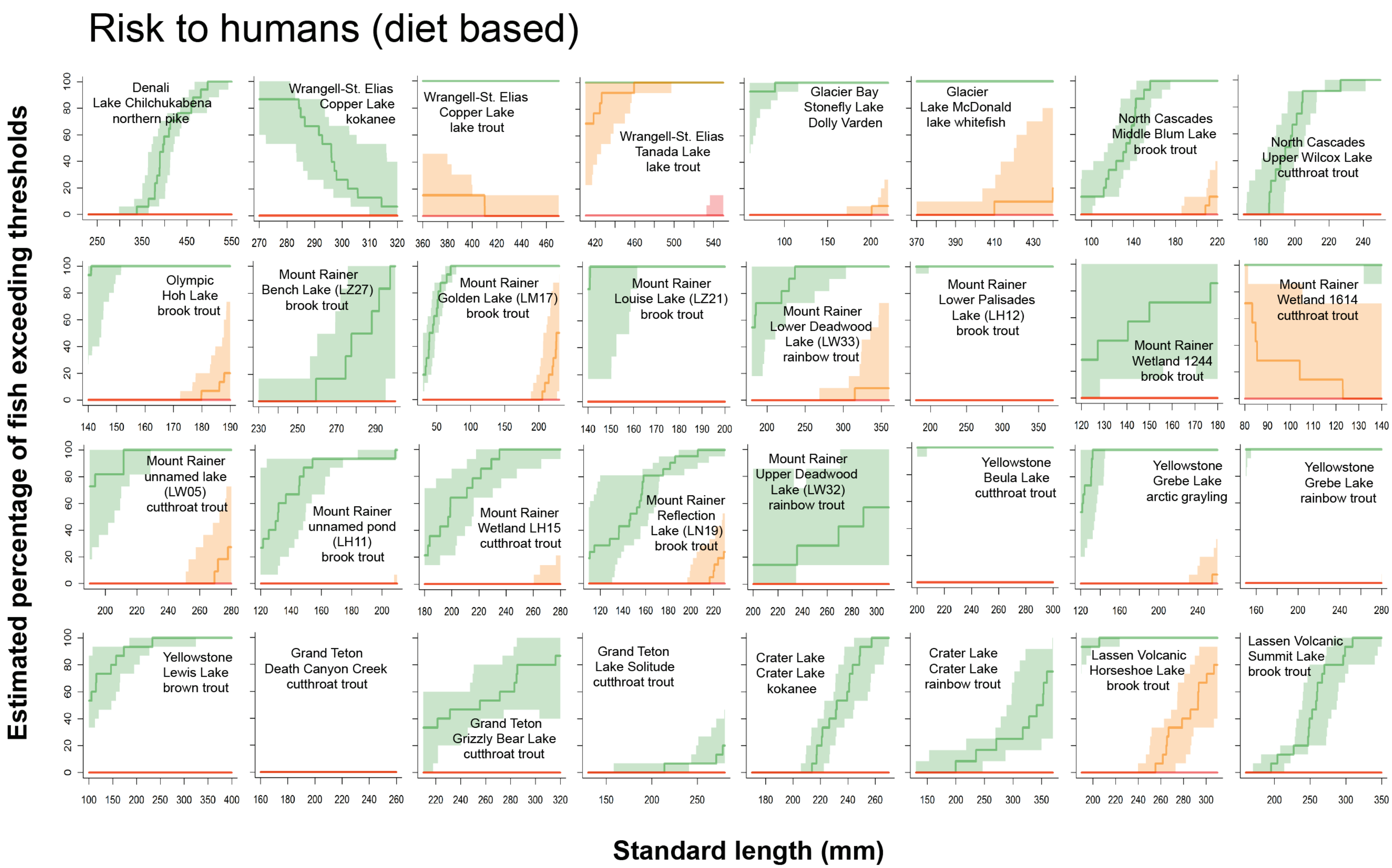

Figure 11. Modeled percentages of fish across the sampled size range in each population with muscle total mercury concentrations exceeding the Great Lakes Advisory Group's (GLAG) unlimited consumption threshold (green; 50 ng/g ww), the Environmental Protection Agency's fish consumption criterion (orange; 300 ng/g ww), and the GLAG's no-consumption threshold (red; 950 ng/g ww) in 21 national parks in the Western United States. Parks are ordered by decreasing latitude.

Risk to humans (diet based)

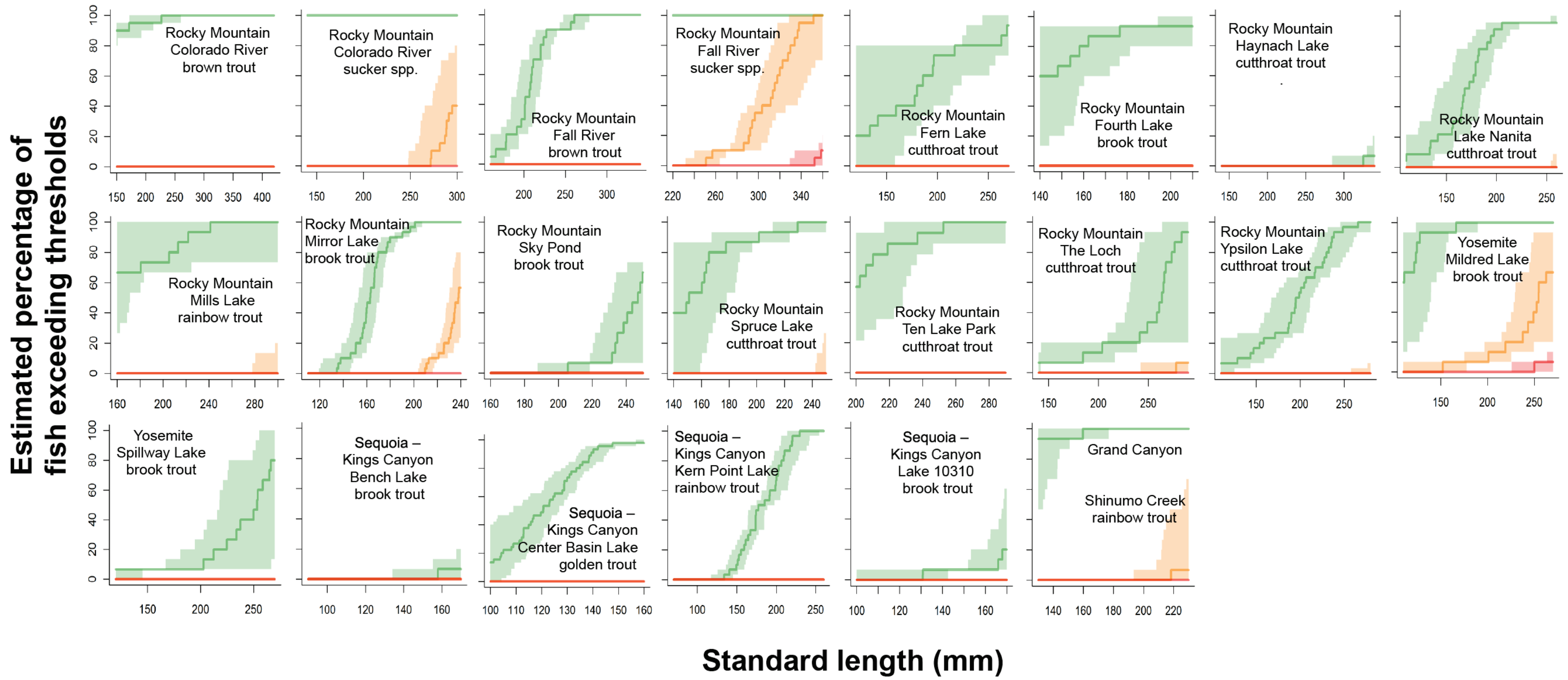

Figure 11.—Continued. Modeled percentages of fish across the sampled size range in each population with muscle total mercury concentrations exceeding the Great Lakes Advisory Group's (GLAG) unlimited consumption threshold (green; 50 ng/g ww), the Environmental Protection Agency's fish consumption criterion (orange; 300 ng/g ww), and the GLAG's no consumption threshold (red; 950 ng/g ww) in 21 national parks in the Western United States. Parks are ordered by decreasing latitude.

Table 1. Site locations and sampling information for 21 national parks in the Western United States sampled between 2008 and 2012.

Park / Site	Site ID	Latitude	Longitude	Altitude (m)	Habitat	Species	Year Sampled	n	Standard Length (mm) Mean	Range	SR[a]
Capitol Reef											
Fremont River #1	1	38.2827	-111.1157	1502	river	speckled dace	2012	15	58	46-74	Y
Fremont River #4	2	38.2815	-111.1794	1577	river	speckled dace	2012	15	61	55-71	Y
Fremont River #7	3	38.2856	-111.2416	1649	river	speckled dace	2012	15	67	53-99	Y
Crater Lake											
Crater Lake	1	42.9057	-122.1017	1884	lake	kokanee	2011	15	207	172-265	Y
						rainbow trout	2011	12	254	137-370	Y
Denali											
Lake Chilchukabena	1	63.9153	-151.5083	191	lake	northern pike	2012	17	407	232-546	Y
Glacier											
Lake McDonald	1	48.5691	-113.9362	963	lake	bull trout	2008	5	434	386-494	N
Glacier Bay											
Falls Creek	1	58.4443	-135.5988	226	creek	Dolly Varden	2012	15	96	73-127	N
North Skidmore Lake	2	58.8451	-136.6839	98	lake	Dolly Varden	2012	15	161	128-203	N
Stonefly Lake	3	58.9674	-136.3379	38	lake	Dolly Varden	2012	15	141	61-219	Y
						lake whitefish	2008	10	396	371-431	Y
Grand Canyon											
Bright Angel Creek	1	36.1228	-112.0800	887	lake	brown trout	2012	2	139	104-174	N
						rainbow trout	2012	10	131	96-263	N
Havasu Creek	2	36.2980	-112.7402	621	creek	rainbow trout	2012	15	200	140-268	N
Shinumo Creek	3	36.2515	-112.33186	794	creek	rainbow trout	2012	15	171	131-222	Y
Grand Teton											
Death Canyon	1	43.6702	-110.87452	2564	creek	cutthroat trout	2012	15	214	162-253	Y
Grizzly Bear Lake	2	43.8027	-110.8107	2814	lake	cutthroat trout	2012	15	254	219-315	Y
Lake Solitude	3	43.7926	-110.8448	2758	lake	cutthroat trout	2012	15	227	137-278	Y
Great Basin											
Baker Lake	1	38.9577	-114.3100	3243	lake	brook trout	2011	9	230	200-252	N
Lehman Creek	2	38.9246	-114.2381	2404	creek	brook trout	2011	15	150	127-167	N
Snake Creek	3	39.01468	-114.2428	2281	creek	brook trout	2011	15	154	122-204	N
Great Sand Dunes											
Medano Lake	1	37.8564	-105.4847	3513	lake	cutthroat trout	2009	15	165	110-212	N
Sand Creek	2	37.9350	-105.5230	3365	creek	rainbow trout	2009	14	190	153-258	N

Park / Site	Site ID	Latitude	Longitude	Altitude (m)	Habitat	Species	Year Sampled	n	Standard Length (mm) Mean	Range	SR[a]
Lake Clark											
Lake Clark[b]	1	60.0187	-154.7574	77	lake	lake trout	2011	16	476	364-648	N
						lake trout	2012	12	478	428-580	N
Lake Kontrashibuna	2	60.1795	-154.2223	140	lake	lake trout	2011	14	358	315-398	N
						lake trout	2012	2	383	378-387	N
Telaquana Lake	3	60.9520	-153.7625	372	lake	lake trout	2011	15	483	412-586	N
Lassen Volcanic											
Horseshoe Lake	1	40.4724	-121.3358	1996	lake	brook trout	2009	15	249	197-305	Y
Ridge Lake	2	40.4138	-121.4630	1984	lake	brook trout	2009	17	187	143-210	N
Summit Lake	3	40.4929	-121.4233	2038	lake	brook trout	2009	15	221	165-346	Y
Mesa Verde											
Mancos River	1	37.2204	-108.3410	1886	river	speckled dace	2012	10	56	43-69	N
Mount Rainier											
Bench Lake (LZ27)	1	46.7622	-121.6986	1386	lake	brook trout	2012	6	262	237-298	Y
Golden Lake (LM17)	2	46.8887	-121.9002	1372	lake	brook trout	2012	16	73	36-229	Y
Green Lake (LC07)	3	46.9769	-121.8590	974	lake	cutthroat trout	2012	6	257	214-285	N
Lake George (LN02)	4	46.7917	-121.9040	1308	lake	torrent sculpin	2012	16	49	37-62	Y
Louise Lake (LZ21)	5	46.7704	-121.7157	1405	lake	brook trout	2012	6	171	145-192	Y
Lower Deadwood Lake (LW33)	6	46.8871	-121.5227	1602	lake	rainbow trout	2012	11	239	185-353	Y
						stickleback[c]	2012	6	42	34-59	Y
Lower Palisades Lake (LH12)	7	46.9533	-121.5903	1682	lake	brook trout	2012	19	277	180-365	Y
Wetland 1244	8	46.7704	-121.7259	1487	lake	brook trout	2012	7	158	122-179	Y
Wetland 1614	9	46.7401	-121.8836	669	lake	cutthroat trout	2012	7	108	89-135	Y
Unnamed Lake (LW05)	10	46.9561	-121.5719	1649	lake	cutthroat trout	2012	11	238	195-278	Y
Unnamed Pond (LH11)	11	46.9554	-121.5932	1666	lake	brook trout	2012	15	159	122-201	Y
Wetland 583	12	46.8918	-121.5948	1125	lake	brook trout	2012	9	164	131-188	N
Unnamed wetland (LH15)	13	46.9517	-121.6178	1679	lake	cutthroat trout	2008	14	218	187-276	Y
Mowich Lake (LM04)	14	46.9339	-121.864	1506	lake	torrent sculpin	2008	7	99	92-105	Y
						torrent sculpin	2012	2	56	53-59	Y
Reflection Lake (LN19)	15	46.7693	-121.7294	1483	lake	brook trout	2012	21	175	114-227	Y
Snow Lake	16	46.7570	-121.6992	1432	lake	brook trout	2012	13	210	179-239	N
						cutthroat trout	2012	2	217	193-241	N
Upper Deadwood Lake (LW32)	17	46.8890	-121.5248	1599	lake	rainbow trout	2008	7	261	203-305	Y
North Cascades											
George Lake	1	48.7061	-121.1722	268	lake	rainbow trout	2011	10	189	171-226	N
Middle Blum Lake	2	48.7500	-121.4956	1529	lake	brook trout	2008	15	142	91-220	Y
Upper Wilcox Lake	3	48.6011	-121.1717	1569	lake	cutthroat trout	2008	12	204	170-248	Y

Park Site	Site ID	Latitude	Longitude	Elevation (m)	Habitat	Species	Year Sampled	n	SL (mm)[a] Mean	Range	SR[b]
Olympic											
Gladys Lake[b]	1	47.8767	-123.3578	1646	lake	brook trout	2011	15	166	159-180	N
						brook trout	2012	15	167	150-182	N
Hagen Lake	2	47.6192	-123.2680	1446	lake	cutthroat trout	2012	15	212	175-238	N
Hoh Lake	3	47.8988	-123.7863	1387	lake	brook trout	2012	15	165	142-185	Y
Sun Up Lake[b]	4	47.5314	-123.4862	1227	lake	cutthroat trout	2011	15	242	232-257	N
						cutthroat trout	2012	15	250	219-269	N
Upper Lena Lake[b]	5	47.6331	-123.2080	1389	lake	rainbow trout	2011	15	171	149-187	N
Rocky Mountain											
Black Lake	1	40.2653	-105.6410	3242	lake	brook trout	2012	15	188	126-226	N
Colorado River	2	40.2773	-105.8504	2647	river	brown trout	2012	20	272	152-413	Y
						sucker spp	2012	20	193	145-292	Y
Fall River	3	40.4014	-105.6063	2590	river	brown trout	2012	20	229	166-338	Y
						sucker spp	2012	20	273	220-359	Y
Fern Lake	4	40.3368	-105.6763	2913	lake	cutthroat trout	2012	15	209	126-270	Y
Fourth Lake	5	40.2217	-105.6861	3167	lake	brook trout	2012	15	178	146-206	Y
Haynach Lake	6	40.3467	-105.7646	3378	lake	cutthroat trout	2012	15	237	141-340	Y
Lake Haiyaha	7	40.3047	-105.6622	3120	lake	cutthroat trout	2012	15	255	182-309	N
Lake Louise	8	40.5083	-105.6206	3369	lake	cutthroat trout	2012	14	271	222-314	N
Lake Nanita[b]	9	40.2560	-105.7165	3291	lake	cutthroat trout	2008	8	231	187-258	Y
						cutthroat trout	2012	15	188	111-260	Y
Lone Pine Lake	10	40.2327	-105.7232	3134	lake	brook trout	2012	14	141	91-178	N
Mills Lake	11	40.2895	-105.6417	3033	lake	rainbow trout	2012	15	226	167-295	Y
Mirror Lake[b]	12	40.5380	-105.6984	3364	lake	brook trout	2009	15	211	153-237	Y
						brook trout	2012	15	190	115-238	Y
Poudre Lake[b]	13	40.4222	-105.8089	3282	lake	brook trout	2009	15	202	129-242	N
						brook trout	2012	15	203	108-264	N
Sky Pond	14	40.2782	-105.6688	3315	lake	brook trout	2012	15	215	162-246	Y
Spruce Lake	15	40.3423	-105.6871	2950	lake	cutthroat trout	2012	15	206	148-242	Y
Ten Lake Park	16	40.2104	-105.7194	3415	lake	cutthroat trout	2012	14	239	206-288	Y
The Loch	17	40.2925	-105.6570	3109	lake	cutthroat trout	2012	15	218	148-285	Y
Upper Hutcheson Lake	18	40.1736	-105.6470	3417	lake	cutthroat trout	2012	15	208	183-250	N
Ypsilon Lake[b]	19	40.4436	-105.6633	3218	lake	cutthroat trout	2009	15	215	156-275	Y
Sequoia-Kings Canyon											
Bench Lake	1	36.9488	-118.4651	3224	lake	brook trout	2009	15	131	92-165	Y
Center Basin Lake[b]	2	36.7305	-118.3960	3517	lake	golden trout	2008	30	136	103-156	Y
						golden trout	2012	15	138	100-157	Y
Kern Point Lake[b]	3	36.6013	-118.4319	3281	lake	rainbow trout	2009	15	166	108-224	Y
Lake 10310	4	36.9591	-118.4354	3297	lake	brook trout	2012	15	143	101-168	Y

Park Site	Site ID	Latitude	Longitude	Elevation (m)	Habitat	Species	Year Sampled	n	SL (mm)[a] Mean	Range	SR[b]
Wrangell-St. Elias											
Copper Lake	1	62.4256	-143.5550	886	lake	arctic grayling	2008	15	326	274-374	N
						kokanee	2008	15	295	278-311	Y
						lake trout	2008	13	417	365-470	Y
Summit Lake	2	61.3156	-144.1919		lake	rainbow trout	2009	15	250	123-405	N
Tanada Lake	3	62.4258	-143.3717	880	lake	lake trout	2009	13	466	418-542	Y
						rainbow trout	2012	12	180	151-198	N
Yellowstone											
Beula Lake	1	44.1598	-110.7670	2261	lake	cutthroat trout	2012	15	239	204-300	Y
Grebe Lake	2	44.7516	-110.5578	2450	lake	arctic grayling	2012	15	166	128-255	Y
						rainbow trout	2012	15	234	152-273	Y
Lewis Lake	3	44.3068	-110.6305	2375	lake	brown trout	2012	15	187	109-394	Y
Yellowstone Lake	4	44.4659	-110.3330	2361	lake	cutthroat trout	2012	15	266	243-294	N
						lake trout	2012	15	273	240-325	N
						cutthroat trout	2012	15	221	115-262	Y
Yosemite											
Mildred Lake	1	37.8202	-119.4407	2916	lake	brook trout	2009	15	204	117-269	Y
Spillway Lake	2	37.8408	-119.2321	3189	lake	brook trout	2009	15	192	124-263	Y
						rainbow trout	2012	15	170	73-252	Y
Zion											
E. Fork Virgin River	1	37.1594	-112.9686	1192	river	speckled dace	2012	15	50	40-61	Y
N. Fork Virgin River	2	37.2413	-112.9588	1294	river	speckled dace	2012	15	39	34-44	N

[a] Size regression used: Y-yes, N-no.
[b] Lakes used in inter-annual comparison. Lake Kontrashibuna was excluded from intra-annual comparisons because only two fish were collected in 2012.
[c] Threespine stickleback.

Table 2. Total mercury concentrations (THg; ng/g ww) in muscle of fish from 21 national parks in the Western United States.

[Values are reported as geometric means, 95percent confidence intervals (CI), and range, and least-square mean (LSM) concentrations controlling for the effects of size, species, and site at 50, 200, and 400 mm standardized lengths]

Park	Park Abbreviation	Total n	Geometric Mean	95percent CI	Range	50 mm			200 mm			400 mm		
						n	LSM	SE	n	LSM	SE	n	LSM	SE
Capitol Reef	CARE	45	325.6	281.4 - 376.9	136.7 - 805.4	45	207.9	3.7		-			-	
Crater Lake	CRLA	27	38.0	32 - 44.3	16.8 – 76.0		-		27	26.6	2.6		-	
Denali	DENA	17	49.3	37.9 - 64.1	23.7 - 88.8		-			-		17	47.4	1.6
Glacier	GLAC	15	232.4	211.6 - 255.3	167.3 - 297.9		-			-		15	234.1	9.1
Glacier Bay	GLBA	45	90.2	79.1 - 102.9	41.0 - 203.1		-		45	102.3	19.6		-	
Grand Canyon	GRCA	42	76.0	66.1 - 87.5	31.8 - 241.2		-		42	101.2	9.6		-	
Grand Teton	GRTE	45	32.6	28.3 - 37.6	16.2 – 99.0		-		45	31.3	4.5		-	
Great Basin	GRBA	39	40.0	36.3 - 44.2	19.0 - 85.5		-		39	34.5	5.0		-	
Great Sand Dunes	GRSA	29	52.7	45.8 - 60.6	29.1 - 136.6		-		29	62.4	5.6		-	
Lake Clark	LACL	59	229.3	192.8 - 272.8	35.8 – 691.0		-			-		59	201.0	2.9
Lassen Volcanic	LAVO	47	96.4	78.1 - 119	28.6 - 493.4		-		47	65.3	9.5		-	
Mesa Verde	MEVE	10	74.9	59.9 - 93.7	45.7 - 115.4	10	74.9	2.9		-			-	
Mount Rainier	MORA	207	71.5	65.8 - 78.5	12.5 - 384.5	35	80.4	1.9	172	61.0	7.3		-	
North Cascades	NOCA	37	54.9	46.7 - 64.4	22.4 - 209.3		-		37	73.3	7.2		-	
Olympic	OLYM	117	85.0	80 - 90.3	34.9 - 208.8		-		337	109.7	12.1		-	
Rocky Mountain	ROMO	385	66.1	61.4 - 71.1	9.9 - 528.3		-		385	54.8	6.3		-	
Sequoia - Kings Canyon	SEKI	105	43.8	39.1 - 49	13.3 - 124.5		-		105	69.5	6.3		-	
Wrangell-St. Elias	WRST	71	107.9	87.4 - 133.1	27.8 - 632.3		-		45	98.15	9.41	26	278.8	7.4
Yellowstone	YELL	90	101.2	92.7 - 110.5	44.6 - 312.2		-		75	119.8	10.2	15	116.0	4.3
Yosemite	YOSE	30	82.1	57.5 - 117.2	24.9 - 1108.6		-		30	70.0	10.2		-	
Zion	ZION	30	241.5	209.7 - 278.2	95.9 - 532.6	30	242.5	5.4		-			-	

Table 3. Total mercury concentrations (THg; ng/g ww) and coefficients of variation (CV) in fish muscle from 86 sites in 21 national parks in the Western United States.

[Values are reported as geometric means and 95 percent confidence intervals (95% CI), and least-square mean (LSM) concentrations controlling for the effects of size, species, and site at 50 mm, 200 mm, and 400 mm standardized lengths]

Park / Site	Geometric Mean	95% CI	50 mm[b] LSM	50 mm[b] SE	200 mm[b] LSM	200 mm[b] SE	400 mm[b] LSM	400 mm[b] SE	CV
All Parks	**77.7**	**73.6 - 80.8**							**0.67**
Capitol Reef	**325.6**	**281.4 - 376.9**	**207.9**	**3.7**					**0.88**
Fremont River #1	407.3	318.3 - 521.1	301.0	9.4					0.31
Fremont River #4	244.7	199.8 - 299.7	157.8	4.9					0.31
Fremont River #7	346.5	260.4 - 461.1	189.4	5.9					0.30
Crater Lake	**38.0**	**32 - 44.3**			**26.6**	**2.6**			**NA**
Crater Lake	38.0	32.7 - 44.3			26.6	2.6			0.28
Denali	**49.3**	**37.9 - 64.1**					**47.4**	**1.6**	**NA**
Lake Chilchukabena	49.3	37.9 - 64.1					47.4	1.6	0.23
Glacier	**232.4**	**211.6 - 255.3**					**234.1**	**9.1**	**NA**
Lake McDonald	232.4	211.6 - 255.3					234.1	9.1	0.15
Glacier Bay	**90.2**	**79.1 - 102.9**			**102.3**	**19.6**			**0.49**
Falls Creek	56.4	50 - 63.6			56.4	10.9			0.22
North Skidmore Lake	113.7	102.3 - 126.4			113.7	22.1			0.18
Stonefly Lake	114.7	91.9 - 143.1			167.0	32.4			0.27
Grand Canyon	**76.0**	**66.1 - 87.5**			**101.2**	**9.6**			**0.54**
Bright Angel Creek	57.8	46.7 - 71.7			71.5	7.0			0.35
Havasu Creek	66.0	53.3 - 81.8			81.0	8.2			0.50
Shinumo Creek	108.9	89.1 - 133.2			178.8	18.0			0.28
Grand Teton	**32.6**	**28.3 - 37.6**			**31.3**	**4.5**			**0.28**
Death Canyon	22.9	21 - 25			25.1	3.7			0.10
Grizzly Bear Lake	52.1	40.9 - 66.4			42.4	6.3			0.40
Lake Solitude	29.0	24.1 - 34.9			28.8	4.3			0.29
Great Basin	**40.0**	**36.3 - 44.2**			**34.5**	**5.0**			**0.12**
Baker Lake	41.6	29.2 - 59.3			35.6	5.4			0.52
Lehman Creek	35.1	30.5 - 40.3			30.0	4.4			0.24
Snake Creek	44.7	40.4 - 49.4			38.3	5.6			0.16
Great Sand Dunes	**52.7**	**45.8 - 60.6**			**62.4**	**5.6**			**0.37**
Medano Lake	70.1	61.6 - 79.7			81.8	12.1			0.28
Sand Creek	38.8	34.6 - 43.5			47.6	4.8			0.21
Lake Clark	**229.3**	**192.8 - 272.8**					**201.0**	**2.9**	**0.57**
Lake Clark	365.2	327.4 - 407.4					365.2	9.2	0.31
Lake Kontrashibuna	204.0	147.9 - 281.4					204.0	7.3	0.41
Telaquana Lake	109.0	82.5 - 144					109.0	4.1	0.56

Park Site	Geometric Mean	95% CI[a]	50 mm[b] LSM	50 mm[b] SE	200 mm[b] LSM	200 mm[b] SE	400 mm[b] LSM	400 mm[b] SE	CV
Lassen Volcanic	**96.4**	**78.1 - 119**			**65.3**	**9.5**			**0.45**
Horseshoe Lake	180.7	132.3 - 246.9			84.8	12.5			0.29
Ridge Lake	111.3	89.9 - 137.9			95.4	14.0			0.39
Summit Lake	43.7	38.1 - 50.1			34.5	5.1			0.12
Mesa Verde	**74.9**	**59.9 - 93.7**	**74.9**	**2.9**					**NA**
Mancos River	74.9	59.9 - 93.7	74.9	2.9					0.30
Mount Rainier	**71.5**	**65.8 - 78.5**	**80.4**	**1.9**	**61.0**	**7.3**			**0.55**
Bench Lake	34.0	20.4 - 56.7			8.5	1.3			0.29
Golden Lake	66.6	47.7 - 93			193.2	28.5			0.14
Green Lake	66.2	58.3 - 75.2			77.3	11.9			0.12
Lake George	87.5	73.2 - 104.5	89.8	2.7					0.27
Louise Lake	86.1	63.2 - 117.3			105.9	16.2			0.19
Lower Deadwood Lake	82.9	65.5 - 104.8	50.6	2.0	77.9	8.0			
Lower Palisades Lake	116.3	103 - 131.4			92.9	13.6			0.21
MORA_1244	58.4	44 - 77.4			60.6	9.2			0.34
MORA_1614	227.4	157.5 - 328.2			97.8	14.9			0.30
MORA_230	121.2	86.7 - 169.4			80.7	12.0			0.32
MORA_234	66.2	53.1 - 82.5			87.4	12.9			0.29
MORA_583	19.2	14.4 - 25.5			16.4	2.5			0.46
MORA_LH15	66.4	51.1 - 86.2			62.5	9.2			0.36
Mowich Lake	97.2	72.9 - 129.6	114.1	5.6					0.17
Reflection Lake	80.8	62.3 - 104.9			101.0	14.8			0.44
Snow Lake	37.3	30.4 - 45.9			33.3	4.8			0.37
Upper Deadwood Lake	47.0	33.5 - 66.1			42.9	4.6			0.36
North Cascades	**54.9**	**46.7 - 64.4**			**73.3**	**7.2**			**0.45**
George Lake	44.1	33.5 - 58			54.1	5.6			0.34
Middle Blum Lake	62.2	46.3 - 83.5			118.5	17.5			0.36
Upper Wilcox Lake	56.2	41.8 - 75.5			61.5	9.1			0.18
Olympic	**85.0**	**80 - 90.3**			**109.7**	**12.1**			**0.60**
Gladys Lake	83.5	75.5 - 92.4			71.5	10.4			0.28
Hagen Lake	93.5	78.1 - 111.9			109.1	16.1			0.30
Hoh Lake	125.7	104.7 - 150.9			253.0	37.3			0.26
Sun Up Lake	85.2	79.1 - 91.7			99.4	14.5			0.19
Upper Lena Lake	66.1	58.9 - 74.1			81.1	8.0			0.27
Rocky Mountain	**66.1**	**61.4 - 71.1**			**54.8**	**6.3**			**0.43**
Black Lake	54.4	45.7 - 64.8			46.6	6.9			0.31
Colorado River	118.0	104.3 - 133.6			100.4	10.6			0.40
Fall River	106.2	83.3 - 135.5			59.3	6.3			0.43
Fern Lake	59.1	46.4 - 75.3			65.3	9.6			0.35
Fourth Lake.	70.4	58.8 - 84.3			71.6	10.6			0.27
Haynach Lake	19.8	15.9 - 24.5			19.4	2.9			0.26
Lake Haiyaha	18.5	15.9 - 21.5			21.6	3.2			0.26
Lake Louise	56.5	48.7 - 65.5			65.9	9.7			0.25
Lake Nanita	70.5	55 - 90.4			80.0	11.7			0.34
Lone Pine Lake	57.8	50.2 - 66.5			49.5	7.3			0.26
Mills Lake	87.5	68.5 - 111.9			94.0	9.5			0.37
Mirror Lake	121.2	86.2 - 170.5			102.4	14.9			0.36

Park Site	Geometric Mean	95% CI	50 mm[b]		200 mm[b]		400 mm[b]		CV
			LSM	SE	LSM	SE	LSM	SE	
Rocky Mountain cont.									
Poudre Lake	82.6	70.9 - 96.2			70.8	10.3			0.50
Sky Pond	30.7	23.2 - 40.6			21.4	3.2			0.30
Spruce Lake	89.3	69.7 - 114.4			97.5	14.4			0.30
Ten Lake Park	68.3	57.5 - 81			63.2	9.4			0.28
The Loch	38.9	27.1 - 56			38.6	5.7			0.82
Upper Hutcheson Lake	30.8	28 - 33.8			35.9	5.3			0.16
Ypsilon Lake	62.3	51.6 - 75.3			61.7	9.0			0.45
Sequoia - Kings Canyon	**43.8**	**39.1 - 49**			**69.5**	**6.3**			**0.38**
Bench Lake.	21.7	18.1 - 26.1			30.3	4.5			0.32
Center Basin Lake	67.0	60 - 74.9			207.3	39.8			0.28
Kern Point Lake	43.5	37.2 - 50.9			71.4	7.0			0.24
Lake 10310	24.9	20.1 - 30.9			52.1	7.7			0.41
Wrangell-St. Elias	**107.9**	**87.4 - 133.1**			**98.2**	**9.4**	**278.8**	**7.4**	**0.57**
Copper Lake	91.7	73.4 - 114.5			147.4	14.5	242.0	9.8	0.24
Summit Lake	53.3	48.5 - 58.5			65.4	6.6			0.18
Tanada Lake	416.6	358.4 - 484.2					321.2	13.0	0.19
Yellowstone	**101.2**	**92.7 - 110.5**			**119.8**	**10.2**	**116.0**	**4.3**	**0.23**
Beula Lake	107.1	95.5 - 120.1			111.1	16.4			0.18
Grebe Lake	110.5	94.2 - 129.7			168.8	16.4			0.19
Lewis Lake	69.2	56.5 - 84.9			92.3	10.1			0.25
Yellowstone Lake	108.9	92.8 - 127.7			119.3	17.6	116.0	4.3	
Yosemite	**82.1**	**57.5 - 117.2**			**70.0**	**10.2**			**0.86**
Mildred Lake	174.4	115.1 - 264.1			142.4	21.0			0.58
Spillway Lake	38.7	32.3 - 46.2			34.4	5.1			0.31
Zion	**241.5**	**209.7 - 278.2**	**242.5**	**5.4**					**0.57**
E. Fork Virgin River	234.9	180 - 306.5	236.8	7.4					0.29
N. Fork Virgin River	248.3	215.8 - 285.7	248.3	7.7					0.32

Table 4. Proportion of fish with wet weight mercury concentrations exceeding toxicity benchmarks from 86 sites in 21 western national parks in the Western United States.

[Comparisons to fish (no observed effects residue [NOER] = 200 ng/g, lowest observed effects residue [LOER] = 300 ng/g) and bird (high sensitivity = 90 ng/g, moderate sensitivity = 180 ng/g, low sensitivity = 270 ng/g) benchmarks were made using whole-body concentrations whereas comparisons to human consumption (unlimited consumption = 50 ng/g, EPA criterion = 300 ng/g, no consumption = 950 ng/g) benchmarks used concentrations in axial muscle tissue]

Park	Fish Toxicity		Bird Toxicity			Human Consumption		
Site	NOER	LOER	High Sens.	Moderate Sens.	Low Sens.	Unlimited Consumption	EPA Criterion	No Consumption
Among all parks	5	2	35	12	5	68	4	1
Capitol Reef	49	22	98	56	33	NA	NA	NA
Fremont River #1	67	33	100	80	60	NA	NA	NA
Fremont River #4	20	7	93	27	7	NA	NA	NA
Fremont River #7	60	27	100	60	33	NA	NA	NA
Crater Lake	--	--	--	--	--	19	--	--
Crater Lake	--	--	--	--	--	19	--	--
Denali	--	--	--	--	--	59	--	--
Lake Chilchukabena	--	--	--	--	--	59	--	--
Glacier	--	--	100	--	--	100	--	--
Lake McDonald	--	--	100	--	--	100	--	--
Glacier Bay	--	--	18	--	--	87	--	--
Falls Creek	--	--	--	--	--	60	--	--
North Skidmore Lake	--	--	20	--	--	100	--	--
Stonefly Lake	--	--	33	--	--	100	--	--
Grand Canyon	--	--	10	--	--	79	--	--
Bright Angel Creek	--	--	--	--	--	58	--	--
Havasu Creek	--	--	7	--	--	73	--	--
Shinumo Creek	--	--	20	--	--	100	--	--
Grand Teton	--	--	--	---	--	18	--	--
Death Canyon	--	--	--	--	--	--	--	--
Grizzly Bear Lake	--	--	--	--	--	47	--	--
Lake Solitude	--	--	--	--	--	7	--	--
Great Basin	--	--	--	--	--	18	--	--
Baker Lake	--	--	--	--	--	33	--	--
Lehman Creek	--	--	--	--	--	7	--	--
Snake Creek	--	--	--	--	--	20	--	--

Park	Fish Toxicity		Bird Toxicity			Human Consumption		
Site	NOER	LOER	High Sens.	Moderate Sens.	Low Sens.	Unlimited Consumption	EPA Criterion	No Consumption
Great Sand Dunes	--	--	--	--	--	59	--	--
Medano Lake	--	--	--	--	--	100	--	--
Sand Creek	--	--	--	--	--	14	--	--
Lake Clark	31	7	76	42	10	98	41	--
Lake Clark	54	14	100	79	21	100	75	--
Lake Kontrashibuna	19	--	81	19	--	94	19	--
Telaquana Lake	--	--	27	--	--	100	--	--
Lassen Volcanic	2	--	40	2	2	72	2	--
Horseshoe Lake	7	--	87	7	7	93	7	--
Ridge Lake	--	--	35	--	--	100	--	--
Summit Lake	--	--	--	--	--	20	--	--
Mesa Verde	--	--	--	--	--	NA	NA	NA
Mancos River	--	--	--	--	--	NA	NA	NA
Mount Rainier	1	--	15	2	--	72	2	--
Bench Lake	--	--	--	--	--	17	--	--
Golden Lake	--	--	19	--	--	56	6	--
Green Lake	--	--	--	--	--	100	--	--
Lake George	--	--	6	--	--	NA	NA	NA
Louise Lake	--	--	--	--	--	100	--	--
Lower Deadwood Lake	--	--	18	--	--	94	--	--
Lower Palisades Lake	--	--	21	--	--	100	--	--
MORA_1244	--	--	--	--	--	57	--	--
MORA_1614	29	--	100	29	--	100	29	--
MORA_230	--	--	55	--	--	100	--	--
MORA_234	--	--	7	--	--	73	--	--
MORA_583	--	--	--	--	--	--	--	--
MORA_LH15	--	--	7	--	--	64	--	--
Mowich Lake	--	--	11	--	--	NA	NA	NA
Reflection Lake	--	--	19	5	--	71	5	--
Snow Lake	--	--	--	--	--	27	--	--
Upper Deadwood Lake	--	--	--	--	--	29	--	--
North Cascades	--	--	5			51	--	--
George Lake	--	--	--	--	--	40	--	--
Middle Blum Lake	--	--	7	--	--	53	--	--
Upper Wilcox Lake	--	--	8	--	--	58	--	--

Park	Fish Toxicity		Bird Toxicity			Human Consumption		
Site	NOER	LOER	High Sens.	Moderate Sens.	Low Sens.	Unlimited Consumption	EPA Criterion	No Consumption
Olympic	--	--	6			94	--	--
Gladys Lake	--	--	3	--	--	100	--	--
Hagen Lake	--	--	7	--	--	100	--	--
Hoh Lake	--	--	33	--	--	100	--	--
Sun Up Lake	--	--	--	--	--	100	--	--
Upper Lena Lake	--	--	--	--	--	74	--	--
		--				--		
Rocky Mountain	2	--	15	3	1	66	3	--
Black Lake	--	--	--	--	--	60	--	--
Colorado River	--	--	28	3	--	100	3	--
Fall River	10	3	33	10	5	80	10	--
Fern Lake	--	--	--	--	--	80	--	--
Fourth Lake	--	--	--	--	--	87	--	--
Haynach Lake	--	--	--	--	--	--	--	--
Lake Haiyaha	--	--	--	--	--	--	--	--
Lake Louise	--	--	--	--	--	57	--	--
Lake Nanita	--	--	17	--	--	70	--	--
Lone Pine Lake	--	--	--	--	--	71	--	--
Mills Lake	--	--	13	--	--	80	--	--
Mirror Lake	10	--	57	17	--	77	17	--
Poudre Lake	--	--	17	--	--	100	--	--
Sky Pond	--	--	--	--	--	13	--	--
Spruce Lake	--	--	20	--	--	87	--	--
Ten Lake Park	--	--	--	--	--	93	--	--
The Loch	--	--	--	--	--	27	--	--
Upper Hutcheson Lake	--	--	--	--	--	--	--	--
Ypsilon Lake	--	--	10	--	--	60	--	--
Sequoia - Kings Canyon	--	--	--	--	--	46	--	--
Bench Lake	--	--	--	--	--	--	--	--
Center Basin Lake	--	--	--	--	--	73	--	--
Kern Point Lake	--	--	--	--	--	43	--	--
Lake 10310	--	--	--	--	--	13	--	--
Wrangell-St. Elias	16	4	39	18	7	76	18	--
Copper Lake	--	--	35	5	--	74	5	--
Summit Lake	--	--	--	--	--	60	--	--
Tanada Lake	85	23	100	85	38	100	85	--
Yellowstone	--	--	**20**	**1**	**--**	**94**	**1**	**--**
Beula Lake	--	--	7	--	--	100	--	--
Grebe Lake	--	--	30	--	--	97	--	--
Lewis Lake	--	--	7	--	--	80	--	--
Yellowstone Lake	--	--	23	3	--	97	3	--

Park	Fish Toxicity		Bird Toxicity			Human Consumption		
Site	NOER	LOER	High Sens.	Moderate Sens.	Low Sens.	Unlimited Consumption	EPA Criterion	No Consumption
Yosemite	**7**	**3**	**27**	**7**	**3**	**60**	**7**	**3**
Mildred Lake	13	7	53	13	7	100	13	7
Spillway Lake	--	--	--	--	--	20	--	--
Zion	**20**	**3**	**90**	**27**	**3**	**NA**	**NA**	**NA**
E. Fork Virgin River	30	--	80	40	--	NA	NA	NA
N. Fork Virgin River	7	7	100	13	7	NA	NA	NA

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