The Importance of Weight-Normalized Exposure Data when Issuing Fish Advisories for Protection of Public Health

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Public health protection from environmental contaminants requires an understanding of the extent of contamination and of the extent of exposure to the contamination. My argument here is that weight-normalized, species-specific, individual-consumption pattern data are vital for determining exposure levels used to ascertain health protection measures and impacts from consuming contaminated fish. This study demonstrates the importance of adequate consumption pattern data for determining exposure distributions used for public health protection by examining three populations exposed to methylmercury through fish consumption: one recreational angler population and two Native-American populations. I compared exposure distributions derived from empirically derived species-specific, individual-consumption data from the three populations and exposure distributions derived, in part, from summary statistics for populations. In so doing, I conducted sensitivity analyses and population-specific probabilistic assessments of exposure. Although the goals of present-day accepted practices—using exposure distributions derived partly from point-estimate-based consumption and body-weight values—are laudable, results presented here indicate that weight-adjusted intake values for a population of concern are warranted when determining exposure distributions and should not be neglected in a health assessment instigated by available data on contaminant concentrations. If individual intake data are unobtainable, raw data from similar populations or tabulated values providing contaminant intake normalized for body weight may be viable alternatives to default values, and can be used to adequately protect public health. Without weight-normalized consumption pattern data to determine exposure, health assessment conclusions can mislead the public and have diminishing protective value. ***Key words:*** **consumption data, contaminant data, exposure data, fish, fish advisories, mercury, tolerable daily intake.** ***Environ Health Perspect*** **110:671–677 (2002). [Online 28 May 2002]**
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When contaminants are found in fish, concern regarding the concentrations of these compounds in the fish can raise the question "Are these fish safe to eat?" In response, local health jurisdictions or other health and environmental agencies frequently use contaminant concentrations along with point estimate-based values for consumption and body weight to determine exposure levels to the population. However, actual rates and values for the population of concern may differ considerably from the default values used, especially when subpopulations, such as those who are more sensitive or high fish consumers, are considered.

Under certain circumstances, this approach may be acceptable because minimizing exposure to contaminants is the primary goal of an assessment. However, with respect to fish contaminants such as mercury, the reduction in exposure must be weighed against the impact on fish consumption patterns of populations that rely on fish as a protein source. Also, cultural, spiritual, and historical practices of the populations must be considered. Weighing these various aspects is not a trivial undertaking, because negative effects from mercury exposure have been well documented, as have positive effects from eating fish. Exposure through diet has resulted in increased body burdens of methylmercury in human populations (*1,2*). Catastrophic exposures in communities in Japan and Iraq produced severe toxic and teratogenic effects (*3*). Prenatal exposure of the fetus can lead to central nervous system damage, which can produce neurotoxic effects in children (*1,2*). These consequences from overexposure must be weighed against the benefits of fish consumption because it is an excellent source of protein that is low in saturated fats and high in essential nutrients, including vitamin D and ω_3 fatty acids. Also, fish consumption has been linked to reduction of cardiovascular disease and osteoporosis. As a result, it becomes imperative that health assessors determine *a*) acceptable levels of exposure that do not cause adverse health effects and *b*) estimates of exposure for populations of concern that best represent actual exposure values. Only with these variables properly quantified can assessors prevent overexposure to mercury in fish while minimizing the impact on beneficial consumption of fish.

I conducted this study to aid in proper quantification of these two variables by examining and comparing various approaches to determine exposure to mercury using species-specific, individual-consumption data, default values, and probabilistic approaches. In suggesting an alternative approach to those using default values solely, or to probabilistic assessments that rely in part on point estimates, a clear public health benefit must be provided; the alternative presented here requires more time, effort, and financial resources than if default values are used. I also discuss the use of surrogate data from similar populations and tabulated data because they may provide alternatives when species-specific, individual-consumption data are not obtainable. I argue that assessments conducted using contaminant data without adequate consumption data can lead to intervention strategies that have negative public health impacts.

Methods

I determined exposure distributions and rates for three populations using contaminant data from consumed fish species and from calculated exposure values. I derived these exposure values from species-specific, individual-consumption data and from exposure values based on approaches using or relying on default values. I describe the various aspects and data sets below.

Contaminant data. I obtained mercury concentrations in fish and shellfish for several water bodies in Washington State from existing data sets (*4–14*). This allowed me to use mercury concentrations from the fishery that each specific population was consuming. These fish tissue mercury concentration data were compiled previously and have been discussed elsewhere (*15*).

Consumption rates. I obtained consumption rates for three populations, which have been previously described (*15,16*). In summary, I obtained consumption data on freshwater fish from 343 surveyed anglers from a study at Franklin D. Roosevelt Lake (Lake Roosevelt) in Washington State, which is visited by over 1 million people annually (*15*).

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The study was conducted during 1994–1995 to determine consumption patterns of anglers who repeatedly fish the lake with the presumption that these individuals catch and consume the greatest amount of fish. I obtained information on saltwater fish consumption practices of Native Americans from a survey of the Tulalip and Squaxin Island Tribes of the Puget Sound region in Washington State (*15,16*). In 1994, 190 adults were interviewed (72 and 117 from Tulalip and Squaxin Island Tribes, respectively) to determine fish and shellfish consumption rates (*16*). Fish species consumed were categorized as anadromous (e.g., chinook and coho salmon), bottom (e.g., sole), pelagic (e.g., quillback and copper rockfish), and shellfish. Fish species consumption rates were provided in grams of fish consumed per kilogram of body weight, and the distributions of fish intake for each population were provided as tabulated grouped data and presented as histograms (*16*). The raw consumption pattern data were made available to the Washington State Department of Health (DOH).

Extent of exposure. I derived exposure distributions and various rates for exposure to mercury, as described below. In general, I determined exposure values by dividing daily mercury intake (milligrams Hg per day) by body weight (kilograms). Body-weight values were actual or estimated with a default value (e.g., 70 kg and 80 kg values). I determined mercury intake values by multiplying the amount of a fish species consumed (kilograms fish) with the observed mercury concentration in that fish (milligrams Hg per kilogram fish).

I used data obtained from the Tulalip and Squaxin Island Tribes to determine weight-adjusted species-specific, individual-consumption exposure values. I determined these values on an individual basis for a particular fish species or a combination of species consumed. I derived exposure values for the combination of fish consumed from the amounts and types of fish species consumed by each individual. Thus, I determined combination fish contaminant levels as a weighted average based on an empirical combination of species representing an individual's consumption pattern. These exposure values determined from eating a combination of fish species are still estimates because contaminant data were not available for all fish species (*15*). When contaminant data were unavailable for a particular species (e.g., cod, pollock, etc.), I used existing contaminant data for a species (e.g., rockfish) from the same fish category (e.g., pelagic) to develop the distributions. To determine the sensitivity of body weight and contaminant concentration, I established weight-adjusted, individual-consumption exposure distributions by varying body-weight values or fish concentrations for all fish consumed by ± 10% and ± 20% from their actual values.

For purposes of comparison, I determined non-weight–normalized, species-specific, individual-consumption exposure values for methylmercury for all three populations. Thus, I did not divide the methylmercury intake levels (micrograms Hg per day) for each individual by actual individual weights as above, but divided by default point estimates (70 kg and 80 kg) to derive mercury exposure values (micrograms Hg per kilogram per day). Regarding the recreational anglers consuming freshwater fish species from Lake Roosevelt, contaminant data were available for all types of fish consumed. However, 70 kg was used for body weight because data for this variable were not obtained in the fish consumption survey. I used the approximate average weight (80 kg) of the Native-American population surveyed along with 70 kg to determine distributions of exposure using species-specific, individual-consumption values for each of the two tribes.

For the two populations for which I knew individual weights (the Tulalip and Squaxin Island Tribes), I again determined distributions for individual exposure to methylmercury using consumption point estimates but normalized for weight. Various point-estimate values for consumption are available for use. For example, the average consumption of fish and shellfish from estuarine and fresh waters by the general U.S. population in 1995 was considered to be 6.5 g/day (*17*). This value has been previously used, along with a body-weight value of 60 kg, by the U.S. Environmental Protection Agency (EPA) to establish water quality criteria. The U.S. EPA has also used a 14 g/day estimate, which represents the average consumption of fish and shellfish from marine, estuarine, and fresh waters by the general U.S. public (*17*). The U.S. EPA 30 g/day ingestion value is an estimate of the 50th percentile of recreational fishermen. The 140 g/day value represents the 90th percentile of recreational fishermen (*17*) and can be used as a population-protective value. Although the U.S. EPA recommends that states always evaluate any type of consumption pattern that reasonably could be occurring at a particular location, these and similar point estimates are often used in lieu of actual consumption pattern data (*18*). Further, when attempts are made to protect the vast

Table 1. Exposure distributions derived from weight-adjusted, species-specific, individual-consumption-pattern data.

Calculated Hg intake (µg Hg/kg/day)	No. (%) individuals consuming each species					No. (%) consuming combination of species (total)[a]
	King	Coho	Quillback	Copper	Sole	
Tulalip Tribes						
< 0.035	69 (95)	71 (98)	73 (100)	73 (100)	73 (100)	47 (64)
0.036 ≤ 0.08	3 (4)	1 (1)				16 (22)
0.081 ≤ 0.12	1 (1)	1 (1)				5 (7)
0.13 ≤ 0.15						2 (3)
0.16 ≤ 0.30						3 (4)
> TDI (0.08)	1 (1)	1 (1)	0 (0)	0 (0)	0 (0)	10 (14)
Squaxin Island Tribe						
< 0.035	100 (85)	112 (96)	114 (97)	116 (99)	116 (99)	62 (53)
0.036 ≤ 0.08	11 (9)	5 (4)	2 (2)	1 (1)	1 (1)	33 (28)
0.081 ≤ 0.12	3 (3)		1 (1)			11 (9)
0.13 ≤ 0.15	1 (1)					1 (1)
0.16 ≤ 0.30	1 (1)					8 (7)
0.31 ≤ 1.00	1 (1)					2 (2)
> TDI (0.08)	6 (6)	0 (0)	1 (1)	0 (0)	0 (0)	22 (19)

Mercury exposure values were determined from consumption rates of members of the Tulalip (n = 73) and Squaxin Island (n = 117) Tribes for specific fish species consumed and for the combination of fish consumed by each individual using actual individual body weights. Fish consumed were king and coho salmon, quillback and copper rockfish, and English sole. The table provides the total number of individuals along with the percentage of the total number surveyed (in parentheses) for each distribution.
[a]Mercury exposure values determined from individual consumption rates for the combination of fish consumed by each individual. Combination total = [g king/kg body weight x 0.1 mg Hg/kg + g coho/kg body weight × 0.05 mg Hg/kg + (chum + pink + steelhead + other + smelt) × 0.05 mg Hg/kg] + [(cod + pollock + sablefish + rockfish + greenling + herring + spiny + dogfish + perch) × (0.29 mg Hg/kg)] + [(halibut – sole/flounder + sturgeon) × 0.06 mg Hg/kg].

majority of a population through the use of point estimates, 90th percentile values are frequently applied. As a result, I used each of the three highest ingestion rates (14, 30, and 140 g/day) with the individual body weights to attain separate distributions of mercury intakes for the various fish species.

Tolerable daily intake. A tolerable daily intake (TDI) for methylmercury (0.035–0.08 µg/kg/day) was derived based on studies investigating sensitive end points in children of mothers who consume fish over prolonged periods of time (*15,19*). This TDI is unlikely to result in adverse health effects. I determined study results by pairing distribution data with the TDI.

Data analysis. The software Stata (Stata Corp., College Station, TX), Excel (Microsoft Corporation, Redmond, WA), and Crystal Ball (Decisioneering, Inc., Denver, CO) generated exposure distributions and mercury intake values for each population of anglers. Also, I used these programs to develop summary statistics of the methylmercury intakes and the percentage of individual intakes above the TDI.

Results and Discussion

I conducted this study to ascertain if weight-normalized, species-specific, individual-consumption pattern data are vital for determining exposure levels used to establish health protection measures and impacts from consuming contaminated fish. In arguing that these types of data are necessary for determining exposure by which all other metrics of exposure must be compared, I examined and compared various approaches to determine exposure. In so doing, I conducted sensitivity analyses and population-specific probabilistic assessments of exposure.

Approaches and outcomes. Table 1 shows mercury exposure distribution data obtained by combining mercury fish concentrations with weight-normalized, species-specific, individual-consumption pattern data. Table 1 provides intake distribution values for species consumed on an individual basis as well as a total of the combination of species consumed. Distributional data indicate that 14% of the Tulalip Tribes and 19% of the Squaxin Island Tribe consume a combination of fish species in sufficient quantities to exceed the TDI. In both populations, consumption pattern rates suggest that if exposure from the consumption of an individual fish species were considered in place of a combination of fish, only a small percentage of individuals would have mercury intake levels exceeding the TDI (0–6%). Accordingly, these population data indicate the importance of considering all species consumed because recommendations made to populations based on single or a few fish species can differ from those that consider the exposure to contaminants from all consumed fish species.

Table 2 provides distribution data for mercury exposure determined from individual consumption rates for the combination of fish consumed by each individual using default body weights and includes exposure values using actual individual body weights and consumption rates from Table 1 (combination total). The use of point-estimate body weights (70 kg or 80 kg) resulted in 23–29% of the individuals surveyed within the Tulalip Tribes exceeding the TDI, whereas 37–42% of the Squaxin Island Tribe exceeded the TDI. These value ranges are approximately 2-fold greater than those obtained using the actual body-weight values, which were 14% and 19% for the Tulalip and Squaxin Island Tribes, respectively. With the use of body-weight summary statistics for these populations providing for a 2-fold increase in the number of individuals exceeding the TDI, the conclusions, recommendations, and restrictions made to either of these populations based on distributions using point-estimate body-weight values would be different from those provided using weight-normalized results.

For recreational anglers, exposure determinations using the default body-weight value of 70 kg (Table 2) indicate that approximately one in eight individuals (13%) had mercury intakes exceeding the TDI. Because individual body weights were not obtained for these anglers, the same comparison between non-weight-adjusted exposure rates and weight-adjusted exposure rates could not be made. The lack of body-weight values is cause for concern because these data have been used, along with similar data in this and other states, for public health protection. The contrasting difference between the tribal weight-adjusted, species-specific, individual-consumption exposure values and those exposure values derived using summary statistics for these populations brings into question the accuracy of using point-estimate body-weight values for other populations. Although the recreational angler population cannot be compared with the Native-American populations surveyed, the results suggest that the percentage of recreational anglers above or below the TDI could be greatly different if individual weights were available and that present public health guidance provided to the recreational anglers may be based on imprecise conclusions.

I performed sensitivity analyses on two variables: mercury fish concentrations and body weight. Table 3 provides results from distributions obtained for mercury exposure when the body-weight values or fish concentrations for all fish consumed are changed by ± 10% or ± 20% from their actual values (combination total). For example, I obtained distributions for mercury exposure by increasing all fish concentrations by 10%. The same was done with respect to body weights in that actual body weights of all

Table 2. Comparison of exposure distributions [combination total (%)] using default point estimates of body weight and actual body weight data.

	Tulalip Tribes[a] (n = 73)			Squaxin Island Tribe[a] (n = 117)			Recreational[b] anglers (n = 377)
Exposure µg Hg/kg/day	Individual weights	Assuming 80 kg bw	Assuming 70 kg bw	Individual weights	Assuming 80 kg bw	Assuming 70 kg bw	Assuming 70 kg bw
< 0.035	47 (64)	27 (37)	14 (19)	62 (53)	36 (31)	24 (21)	228 (67)
0.036 ≤ 0.08	16 (22)	29 (40)	38 (52)	33 (28)	37 (32)	43 (37)	70 (20)
0.081 ≤ 0.12	5 (7)	9 (12)	12 (17)	11 (9)	27 (23)	27 (23)	24 (7)
0.13 ≤ 0.15	2 (3)	2 (3)	2 (3)	1 (1)	7 (6)	8 (7)	9 (3)
0.16 ≤ 0.30	3 (4)	6 (8)	7(9)	8 (7)	5 (4)	10 (8)	12 (3)
0.31 ≤ 1.00				2 (2)	5 (4)	5 (4)	
> TDI (0.08)	10 (14)	17 (3)	21 (29)	22 (19)	44 (37)	50 (42)	45 (13)

bw, body weight. Data are for individuals within the Tulalip and Squaxin Island Tribes and for recreational anglers. Mercury exposure values were determined from individual consumption rates for the combination of fish consumed by each individual using default and actual individual body weights. I used default body weight values of 70 kg and 80 kg because 70 kg represents a recognized default value whereas 80 kg was the average body weight of the Native Americans surveyed. Mercury exposure values determined from consumption rates for the combination of fish consumed by each individual using actual individual body weights are those from Table 1 (combination total, individual weights). The table presents the total number of individuals along with the percentage of the total number surveyed, and the total number of individuals along with the percentage of the total number surveyed above the TDI (0.08 µg Hg/kg/d) for each distribution.

[a]Mercury exposure values determined are described in Table 1. [b]Mercury exposure values determined from individual consumption rates for all fish consumed by each individual recreational angler (combination total) = (kokanee × 0.04 mg Hg/kg) + (rainbow × 0.04 mg Hg/kg) + (walleye × 0.17 mg Hg/kg) + (bass × 0.28 mg Hg/kg).

individuals in a population were, for example, decreased by 20% and then used to recalculate mercury exposure.

For the Tulalip population, decreasing weight by 10% resulted in the percentage of the number of individuals above the TDI increasing from 14% to 19%; decreasing weight by 20% resulted in the percentage of individuals above the TDI increasing from 14% to 20%. For the Squaxin Island population surveyed, this 10% and 20% decrease in body weight resulted in 26% and 31% of the individuals, respectively, exceeding the TDI (compared with 19% obtained when using actual individual weight values). Thus, in these two populations, decreasing the body weight led to an approximate 50% increase in the number of individuals exceeding the TDI. Although the percentage of individuals exceeding the TDI would not be specifically known when decreasing body-weight values, an increase in the percentage of individuals exceeding the TDI would be expected with a decrease in body weight. That is, representing mercury intake versus number of individuals on an *x–y* coordinate system with intake on the abscissa, the distributions would be shifted right, resulting in a greater number of individuals exceeding a particular intake level (the TDI). As with the results presented in Table 2, this percentage increase in the number of individuals exceeding the TDI supports the conclusion that a lack of weight-normalized data could result in public health agencies initiating overprotective intervention strategies.

Results indicate that varying fish concentrations increased the number of individuals exceeding the TDI. However, an increase of 20% in mercury concentration in every fish species consumed by each individual is required before achieving a 50% increase in the number of individuals exceeding the TDI. This would require significant increases in mercury concentrations in all fish species consumed, which is not biologically feasible on a short-term basis without a very large and unexpected mercury release. Accordingly, variations in this variable would be of significant impact to exposure rates only under certain circumstances, for example, if analytical mistakes are made, if obtained consumption patterns have large errors, or if consumption pattern rates significantly increase. Thus, incorrectly estimating the appropriate default value for mercury fish tissue concentration in a particular fish species may not as greatly affect public health recommendations because exposure values may not be significantly altered.

Body-weight values are important, as demonstrated in the examination of the effect of changes in fish concentrations and individual body weights on probabilistic distributions determining mercury exposure. Incorrect assumptions of body-weight values can lead to changes in estimated exposures exceeding those obtained from changing fish concentration levels. This is not to suggest that obtaining fish concentration levels is not valuable, merely that adequate consumption pattern data that are weight-adjusted are also very important.

Another means by which to test the significance of weight-normalized, species-specific, individual-consumption rates is by comparing distributions of exposure values derived from such rates with those obtained by combining default consumption rates with the individual weight values (Table 4). Although the combination of point-estimate consumption values with actual individual weights for a given population would be an uncommon scenario, resulting distributions provide insight into the impact produced from the replacement of species-specific, individual-consumption rates with point-estimate consumption values. I used default ingestion values of 14, 30, and 140 g/day, which have been previously used by the U.S. EPA to calculate the individual mercury intake levels (micrograms Hg per kilogram per day) for each species consumed. I also determined exposure values based on the consumption of a combination of fish species. Results indicate that a consumption rate of 140 g/day yields mercury intake distributions having 65–100% of the population with intake levels above the TDI. Nearly all fish consumed at rates of 14 g/day and 30 g/day yielded intake values with 3% or less of the individuals exceeding the TDI (e.g., coho salmon ingested at 30 g/day resulted in no individuals exceeding the TDI; results not shown). The application of the default ingestion rates to the combination of fish species consumed results in at least 97% of the individuals having intake levels below the TDI when consuming the combination of species at 14 g/day or at 30 g/day; the ingestion rate of 140 g/day results in 100% of the individuals exceeding the TDI (Table 4). In contrast to these mercury intake levels obtained using point-estimate consumption rates, the weight-normalized, species-specific, individual-consumption rates yielded mercury intake distributions having 14% and 19% of the Tulalip and Squaxin Island populations above the TDI, respectively (combination total).

Results from using default ingestion rates indicate that assuming an intake level of 140 g/day instead of using actual consumption pattern data can lead to a 5–7-fold increase in the estimated number of individuals exceeding the TDI. To achieve the 14% and 19% values above the TDI, distributions using actual weight values would require that the Tulalip and Squaxin Island Tribes consume 43 g of fish of various species per day (distribution data not shown). This disparity in consumption rates suggests that the 30-g/day estimate, which represents the 50th percentile consumption rate for fish and shellfish from marine, estuarine, and fresh waters for recreational fishermen, would have been a better proxy with which to protect the health of these individuals, and

Table 3. Comparisons of exposure distributions determined using ± 10% and ± 20% of individual body weights and fish concentrations.

Exposure	Fish concentration					Individual weight				
(µg Hg/kg/day)	–20%	–10%	+10%	+20%	100%[a]	–20%	–10%	+10%	+20%	100%[a]
Tulalip Tribes										
> TDI (%)	7 (9)	9 (12)	14 (19)	15 (20)	10 (14)	15 (20)	14 (19)	9 (12)	7 (10)	10 (14)
Mean	0.03	0.04	0.04	0.05	0.04	0.05	0.04	0.04	0.03	0.04
SD	0.04	0.04	0.05	0.05	0.05	0.06	0.05	0.04	0.04	0.05
95th percentile	0.15	0.17	0.20	0.22	0.18	0.23	0.20	0.17	0.15	0.18
Squaxin Island Tribe										
> TDI (%)	16 (14)	17 (14)	28 (24)	33 (28)	22 (19)	36 (31)	30 (26)	18 (15)	16 (14)	22 (19)
Mean	0.05	0.05	0.06	0.07	0.06	0.07	0.06	0.05	0.05	0.06
SD	0.07	0.08	0.10	0.11	0.09	0.11	0.10	0.08	0.07	0.09
95th percentile	0.33	0.37	0.45	0.49	0.41	0.51	0.45	0.37	0.34	0.41

I derived weight-adjusted, individual-consumption mercury exposure values using ± 10% and ± 20% of individuals body weight values. We also obtained distributions by using ± 10% and ± 20% of actual mercury concentrations in the combination of fish species consumed by each individual. The 100% values presented are those total mercury exposure values using actual individual body-weights from Table 1 (combination total). Data are for individuals within the Tulalip Tribes (n = 73) and Squaxin Island Tribe (n = 117). The table shows the total number of individuals along with the percentage of the total number surveyed above the TDI (0.08 µg Hg/kg/day), mean and standard deviations of the distributions (µg/Hg/kg/day), and 95th percentile values for each distribution. Fish consumed were king and coho salmon, quillback and copper rockfish, and English sole.
[a]Combination totals; mercury exposure values as described in Table 1.

that the 140-g/day value, which represents the 90th percentile consumption rate for recreational fisherman and represents a "conservative" elevated consumption rate used to protect most of the population, provided for results that greatly misrepresented observed findings. These results suggest that the assumption of surrogate point estimates in deriving exposure distributions and levels can lead to initiating intervention strategies that could mislead the public regarding actual human health impact from consuming fish.

Of concern is the finding that individual weights provide for probabilistic exposure distributions that more accurately depict the observed number of individuals that exceeded the TDI when used with a 43 g/day consumption rate (in place of the more frequently used population-protective value of 140 g/day). The U.S. EPA, as well as other agencies, has recently adopted an approach that provides recommended fish consumption limits (number of 8-oz portions) of fish containing various levels of mercury for individuals weighing 72 kg so that these individuals do not exceed a recommended "safe level" (reference dose) (*18*). That is, assuming meal size (8 oz, 200 g) and body weight (72 kg), the U.S. EPA provides recommendations on a sliding scale of how many meals can be eaten given various mercury fish contaminant levels. The U.S. EPA's adjusting scale indicates that if mercury fish tissue concentrations are, for example, > 0.08–0.12 mg/kg, then eight 8-oz portions can be consumed monthly. This is approximately equal to 50 g/day. For all Native Americans surveyed, I examined individual consumption pattern rates to determine the mercury fish concentration of the combination of fish consumed by each individual. I summed and averaged the mercury fish concentrations for each individual. Mean fish tissue concentrations for the fish consumed by each individual for the Tulalip and Squaxin Island Tribes were 0.10 mg/kg and 0.08 mg/kg, respectively. Thus, under the U.S. EPA paradigm, these individuals can consume 50 g/day. Within the Tulalip Tribes, 23% (17 individuals) of the individuals consumed more than 50 g/day, whereas 38% (45 individuals) of the Squaxin Island Tribe exceeded the 50 g/day level. For the Tulalip Tribes, this percentage value of individuals exceeding the TDI (23%) is 60% greater than the 14% value (10 individuals) derived using weight-normalized, species-specific consumption pattern data (Table 1). For the Squaxin Island Tribe, the 38% value is twice the 19% (22 individuals) value derived using weight-normalized, species-specific consumption pattern data (Table 1).

The U.S. EPA's approach is an example of using contaminant values as a foundation from which to carry out assessments and provide consumption guidelines and recommendations without possessing complete consumption data. Although this approach may be mathematically plausible, the results provided herein bring the public health implications of such an approach into question, for two reasons. First, as discussed above, choosing point-estimate weight values affects outcome and can lead to inappropriate health intervention strategies. Any conclusion based on the weight estimate of 72 kg (158 pounds) is protective only of these individuals and of all those weighing more. Yet for those individuals weighing more, consumption recommendations based on a 72-kg weight value would result in an unnecessary reduction in a beneficial food source. Also, these meal consumption guidelines are not protective of individuals weighing less than 72 kg, because more restrictive consumption guidelines would be required if, for example, a 120-pound female was in need of protective recommendations. Accordingly, the use of this body-weight value would lead to intervention strategies insufficiently protective for one half of the community and overly protective for the remaining half. Second is the issue of fish intake. Under the U.S. EPA's adjusting scale paradigm, the Tulalip and Squaxin Island Tribes can consume 50 g/day. Within the Tulalip Tribes, the data indicate that 23% (17 individuals) of the individuals consumed more than 50 g/day, whereas 38% (45 individuals) of the Squaxin Island Tribe exceeded the 50 g/day level; this suggests that these percentage values represent the portion of the population that may be consuming fish in quantities so as to exceed the "safe level" or reference dose. However, as stated, only 14% and 19% of these two populations surveyed exceeded the TDI when using weight-normalized, species-specific consumption pattern data (Table 1). Neither the 50 g/day limit, nor its equivalent, 8-oz portions per month, both of which come from a sliding scale for consumption that are based on point-estimate portion sizes and body weights, is a metric that allows for an accurate representation of distributions determined from weight-normalized, species-specific, individual consumption pattern data. As a result, their use may not confer the appropriate and intended level of protection.

Previously, tabulated weight-adjusted exposure levels provided as grouped data and presented using histograms (*16*) were used to determine the extent to which the two Native-American populations have exposures exceeding the TDI (*15*). Results of that work suggest that 14% and 25% of the Tulalip and Squaxin Island tribal members, respectively, have mercury intakes above the TDI. These results compare favorably with exposure distribution data obtained using the raw weight-adjusted, species-specific, individual-consumption data (Table 1), which indicate that 14% of the Tulalip Tribes and 19% of the Squaxin Island Tribe exceed the TDI.

Table 4. Comparison of exposure distributions derived using point-estimate consumption rates versus actual consumption rates.

Exposure (µg Hg/kg/day)	Quillback (30)	Quillback (140)	Copper (140)	King (140)	Coho (140)	Sole (140)	Combination total[a]
Tulalip Tribes							
> TDI (%)	67 (92)	73 (100)	73 (100)	73 (100)	52 (71)	65 (89)	10 (14)
Mean	0.11	0.53	0.31	0.18	0.09	0.11	0.04
SD	0.02	0.11	0.06	0.04	0.02	0.23	0.05
95th percentile	0.17	0.81	0.48	0.28	0.14	0.17	0.18
Squaxin Island Tribe							
> TDI (%)	108 (92)	117 (100)	117 (100)	117 (100)	76 (65)	103 (88)	22 (19)
Mean	0.11	0.53	0.31	0.18	0.09	0.11	0.06
SD	0.03	0.13	0.08	0.04	0.02	0.23	0.09
95th percentile	0.21	0.96	0.56	0.33	0.17	0.20	0.41

For each population, point-estimate ingestion values of 14, 30, and 140 g/day were used to calculate the individual mercury intake levels (µg Hg/kg/day) for each species consumed. Results are not provided for fish consumed by individuals in such quantity that nearly all individuals (≥ 97%) were below the TDI (e.g., coho salmon ingested at 30 g/day resulted in no individuals exceeding the TDI). Actual weights were used in place of point estimates to derive distributions. Total mercury exposure values used actual individual body-weights and actual consumption rates as presented in Table 1 (combination total). Data for individuals within the Tulalip Tribes (n = 73) and Squaxin Island Tribe (n = 117) are number and percentage of individuals exceeding the TDI (0.08 µg Hg/kg/day) with mean, SD, and 95th percentile values for each distribution obtained. Point-estimate ingestion values (g/day) are provided in parentheses below each fish species. Fish consumed were king and coho salmon, quillback and copper rockfish, and English sole.
[a]Mercury exposure values determined as described in Table 1.

Conclusions from using the tabulated grouped data from Toy et al. (*16*) indicate that for the Tulalip Tribes, most if not all of the individuals within this population could achieve mercury intake levels at or below the TDI while maintaining their cultural heritage by consuming other anadromous fish as alternatives to chinook (*15*). For the Squaxin Island Tribe, Washington State DOH recommended that women of child-bearing age within this population be encouraged through educational efforts to consume salmonids other than chinook. Although this recommendation leads to reduced mercury intake levels, changing consumption of salmonid species would still result in intake levels above the TDI. Thus, the potential for an adverse outcome remained from exposure to mercury through fish consumption. Suggesting that fish be consumed in lesser quantities would be simple. This, however, is not necessarily a prudent public health recommendation. Recommending a change in diet away from nutritional foods such as fish does not imply that the replacement will be equally beneficial. Also, cultural, spiritual, and historical practices must be considered. As a result, the DOH recommended that educational efforts be provided to encourage the broadening of fish species consumed to include freshwater fish species or pelagic fish that possibly have lower mercury concentrations (e.g., cod, pollock, perch). Hair levels of methylmercury in the women of child-bearing age who consume fish in excess of 1.0 g/kg/day should be monitored because hair analysis is presently the exposure metric most frequently used to determine mercury exposure. These data could be used along with educational efforts as excellent tools for properly protecting the health of this population.

The use of the raw species-specific, individual-consumption data to determine weight-adjusted exposure levels in place of exposure levels obtained from grouped data displayed as histograms provides comparable results. Although the raw data are preferred and provide for a more accurate representation of population percentages exceeding the TDI, using grouped data as provided by Toy et al. (*16*) would not change the final recommendations. This provides further support for the argument that weight-adjusted consumption data are vital when providing consumption pattern results from surveys. Also, if non-weight-adjusted values had been relied upon (e.g., those from Table 2, which suggest that the percentage of individuals exceeding the TDI is 2-fold higher), recommendations to these fish-consuming populations would have included severe restrictions on fish consumption and would have lead to dietary changes.

Weight-adjusted compiled consumption data such as these could be used for similar populations for which raw data are unobtainable. This suggestion is made with the understanding that fish consumption rates, habits, and patterns can vary among tribes and even among subpopulations. Despite these variations, weight-adjusted, compiled consumption data from similar populations should be the preferred alternative to using probabilistic distributions based on default values, because results based on such distributions can lead to inaccurate fish advisories.

Shortcomings. Arguments presented in this article would be strengthened with further supporting exposure data that could be normalized for weight. Data sets from other studies that can be weight-adjusted should be examined to appraise these findings, especially because the two populations for which body-weight values were available are similar. Lake Roosevelt anglers, along with other recreational anglers, should be surveyed again to obtain weight-normalized consumption data. These data could determine the adequacy of the non-weight-normalized consumption results that are used to guide public health strategies.

The lack of complete contaminant data is also an issue. The use of rockfish, which are highly contaminated, to determine concentrations in pollock or cod, which are also consumed in quantity and are considered to be less contaminated, can lead to exposure levels that are unrepresentative of the populations' intake levels and could lead to inappropriate recommendations. Contaminant data for actual species consumed is enormously beneficial.

Summary and Conclusions

Observations presented in this study suggest that actual consumption pattern data by species are important for two reasons. First, population-protective consumption values such as 140 g/day do not provide for probabilistic exposure distributions that accurately depict the observed number of individuals that exceed the TDI; rather, a 43 g/day consumption rate provides distributions that more accurately depict the observed number of individuals that exceed the TDI (Table 4). Second, actual consumption pattern data from a combination of fish species is necessary, because recommendations based on the consumption of one or a few species may not be sufficiently protective (Table 1). Consumption pattern data allows for the determination of the average fish tissue concentration of the combination of fish consumed by each individual, as well as the type and quantity of each species that is consumed by each individual. Such determinations provide powerful tools that cannot be gathered from contaminant data pertaining to one or a few species, especially when consumption data are lacking or unavailable.

This study also demonstrates the importance of having consumption pattern data that is weight-normalized. For example, using fixed body-weight levels to determine exposure levels within the Native-American populations resulted in an increase in the estimated number of individuals exceeding the TDI when compared with results obtained from using actual body-weight values (Table 2). In addition, altering body-weight values by 10% and 20% and obtaining exposure distributions produced alterations in the number of individuals exceeding the TDI and resulted in a greater impact on the number of individuals exceeding the TDI than if similar alterations were made to fish concentration levels (Table 3).

Results suggest that the U.S. EPA, when not using weight-normalized consumption pattern data, provides estimates of exposure that can lead to incorrect conclusions. Further, intervention strategies would be insufficiently protective or overly protective when using a default weight value (72 kg) in the U.S. EPA's sliding scale approach to limiting consumption.

Many nations, including the United States, are beginning to aggressively evaluate the public health impact of contaminants found in fish that are regularly consumed by their various constituents. As ecosystems are being investigated for contaminant levels in sediments, biota, and fish, these data are increasingly needed to best meet the public health needs of exposed populations. In the United States, fish advisories have in the past been provided by state agencies that address consumption limitations on one fish type, such as canned tuna, or address limitations on consumption for particular species from a given water body, such as bass or trout from particular rivers or lakes. Although the outcomes of this study are preliminary, results indicate that weight-normalized consumption pattern data are crucial. Relying on distributions derived, in part, on point-estimate based consumption and body-weight values to determine exposure when setting fish advisories should be reconsidered, because this can lead to inaccurate health assessment conclusions. This is especially true in cases where overprotection could have deleterious consequences such as the removal of a food source of considerable benefit. With the impact of using adequate consumption pattern data revealed to be substantial, state agencies responsible for protecting as well as promoting health should understand that both contaminant and consumption data are vital when protecting public health.

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