DEPARTMENT OF
ECOLOGY
State of Washington

Supplemental Information to Support the Fish Consumption Rates Technical Support Document

July 20, 2012

Toxics Cleanup Program
Washington State Department of Ecology
Olympia, Washington

Contents

Technical Issue Paper

Estimating Annual Fish Consumption Rates Using Data from Short-Term Surveys

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Table of Contents

List of Tables

Introduction

Ecology received comments on the draft *Fish Consumption Rate Technical Support Document* (TSD) expressing concern with the way short-term fish consumption surveys were used to extrapolate consumption habits over the long term.

In some populations, fish are consumed frequently and in large quantities (USEPA 2011). Estimating the rate of fish consumption for high-fish consuming groups is important; these groups may be at greater health risk if the fish they consume are from contaminated water bodies.

Studies to estimate fish consumption rates for specific target populations are generally one of two types: creel surveys and interviews/mail surveys.[1] In a creel survey, fishermen are asked, among other things, how many fish they have caught and the number of family members with which they will share their catch. Creel survey data do not represent usual behavior because a fisherman may not have the same fishing success over time. As a result, results from creel studies have often been misinterpreted (USEPA 2011).

Mail surveys, personal interviews, or telephone interviews ask participants to recall how much each family member ate over a certain period of time. The recall period determines whether the survey characterizes long-term (i.e., usual) intake or short-term consumption. For risk assessment, estimates of long-term consumption are needed, but long recall periods are associated with generally higher reporting errors (i.e., it is harder for people to remember what and how much they ate over a longer period of time). Short-term studies, on the other hand, may underestimate the number of people who consume fish and may overestimate long-term consumption. This is particularly true for high end consumers, because short-term studies tend to underestimate the number of days when respondents do not consume fish.

Ebert et al. (1994) describe the problem as follows:

> *Although an individual may consume at a rate in the upper 5th percentile of the distribution during a specific two-week period, it is not necessarily true that the same angler will be an upper 5th percentile consumer throughout the season. Rather, that individual may fish only occasionally, may only be interested in consuming certain species when they are available, and is not likely to be equally successful on every trip. The same uncertainty exists for anglers who have had no activity or success during a single two-week period but may, in fact, have different behavior at other times. It is likely that activity and consumption by individual anglers are highly variable through the season due to weather, fishing regulations, differences in species availability, and fluctuations in success rates for the individual angler. Although, much of this variability tends to be averaged out in longer-term estimates, extrapolation from single-day or short-term measurements can result in an over- or*

[1] Additional information on fish consumption survey methodology is provided in Consumption Surveys for Fish and Shellfish: A Review and Analysis of Survey Methods (USEPA 1992) and Guidance for Conducting Fish and Wildlife Consumption Surveys (USEPA 1998).

> *underestimation in the inter-individual variation of annual intake. Thus, short-term surveys may be useful for characterizing the central tendency in consumption rates but not the variance within the population.*

Attempts to account for the variance and uncertainty associated with the use of short-term consumption studies have generally included qualitative evaluation of data from a range of sources, coupled with consideration of the intended use of the data.

In the draft Fish Consumption Rate TSD, Ecology identified four fish consumption surveys as appropriate for use in establishing a technically defensible default fish consumption rate for Washington:

- Fish Consumption Survey of the Umatilla, Nez Perce, Yakama, and Warm Springs Tribe of the Columbia River Basin (Columbia River Inter-Tribal Fish Commission [CRITFC] 1994).
- Fish Consumption Survey of the Tulalip and Squaxin Island Tribes of the Puget Sound Region (Toy et al. 1996).
- Fish Consumption Survey of the Suquamish Indian Tribe of the Port Madison Indian Reservation, Puget Sound Region (The Suquamish Tribe 2000).
- Asian and Pacific Islander Seafood Consumption Study, King County (Sechena et al. 1999).

These four studies relied on personal interviews and food frequency surveys to obtain dietary recall information. The Suquamish survey quantified consumption rates using both a 24-hour recall and a food frequency survey.

The draft TSD also included information on estimated United States per capita fish consumption. These national fish consumption rates were based on participant responses to 24-hour dietary recall surveys conducted on two nonconsecutive days.

Public review comments on the draft TSD identified several issues of concern. Many reviewers noted correctly that the length of the survey period can have a significant effect on the resulting fish consumption rates. Some reviewers suggested that estimates of annual fish consumption rates be adjusted to account for lifetime consumption. One reviewer was concerned about the difference in mean consumption rates measured in the 24-hour recall portion of the Suquamish survey compared to the food frequency survey.

This Technical Issue Paper discusses the limitations of using short-term studies to estimate long-term consumption, and presents options for adjusting short-term results to better characterize long-term consumption habits. It is a targeted examination of the issues raised by review comments received on the draft Fish Consumption Rates TSD, and was prepared within a limited time frame. Therefore, it may not include all available information on this subject.

Limitations of short-term dietary studies

USEPA (2011) has acknowledged that short-term dietary records present problems when estimating long-term consumption rates. AMEC (2003), in a paper prepared for the Northwest Pulp and Paper Association, summarized the limitations of short-term dietary studies. Key elements of this summary are presented below.

I. AMEC summary of short-term study limitations

The length of the recall period in dietary studies can significantly affect the estimate of long-term fish consumption. For example, dietary data compiled by the U.S. Department of Agriculture (USDA) were collected from survey participants during two non-consecutive 24-hour periods (USEPA 2000). Because of the way in which sampling was conducted, the actual fish consumption behaviors reported are strongly biased toward those respondents who consume fish with a high frequency.

All of the individuals included as fish consumers in the USDA estimate consumed fish at least once during the 2-day sampling period. To use these data to estimate long-term consumption rates, it is necessary to assume that the consumption behavior that occurred during the 2-day period is the same as the consumption behavior that occurs throughout every other 2-day period during the year. Thus, if an individual reported eating one fish meal during the sampling period, the extrapolation necessary to estimate long-term consumption requires the assumption that the individual continues to eat fish with a frequency of once every two days, or as many as 183 meals per year. If an individual eats one-half pound (227 grams) of fish per meal, this results in a consumption rate of 114 grams/day.

However, the individual who consumed fish during that sampling period may not actually be a regular fish consumer. In fact, that fish meal may have been the only fish meal that the individual consumed in an entire year. Thus, that person's fish consumption rate would be substantially overestimated. Unfortunately, because of the way that the USDA data are collected, there is no way to determine if the behaviors reported by survey respondents during the sampling period were representative of their long-term behaviors. Thus, for the "consumers" in the population who were reported in these data, the reported consumption rates must have a minimum of one meal every two days.

Conversely, individuals who did not consume fish during the 2-day sampling period were assumed to be non-consumers of fish when instead, those individuals may have been fish consumers who coincidentally did not consume fish during the 2-day sampling period. Because there are no data on which to base consumption estimates for these individuals, they must be assumed to consume 0 gram/day. However, they may in fact consume fish with a frequency ranging from as little as 0 meals per year to as much as one meal per day (or even more than one meal per day) on all days except the two that USDA conducted the survey. As with the high consumers identified in the USDA database, there is no way to determine whether 0 gram/day consumers are actually non-consumers or just individuals who consume with less frequency than once every two days.

To demonstrate the effect that the length of the sampling period can have on resulting fish consumption rates, the findings of other short-term dietary studies can be compared to long-term studies. For example, a USDA survey reported by Mertz and Kelsay (1984) asked 29 people to track the types and amounts of food they ate for a one-year period. Because the daily dietary records kept by the study subjects could be condensed into 52 discrete one-week periods, it was possible to investigate the relationship between annual and weekly average fish consumption rates. The mean annual fish consumption rate from the Mertz and Kelsay (1984) survey data was estimated by summing the entire quantity of fish consumed by each survey respondent during the year and dividing by 365 days. The mean per capita "365-day" fish consumption rate developed using this approach was 26 grams/day. In addition, the mean daily fish consumption rate averaged over a one-week period, the "7-day" fish consumption rate, was estimated to be 26 grams/day. Thus, the mean per capita consumption rate did not appear to be affected substantially by the recall period. Although this study included only 29 participants, results suggest that the mean is a fairly robust and meaningful measure of the average consumption rate, regardless of the survey period.

The same cannot be said, however, of the upper percentiles of the fish consumption rate distribution. When comparing the 7-day intake rates collected by Mertz and Kelsay (1984) with the 365-day intake rates, the upper percentiles were very different. For example, when looking at the 7-day intake rates, the maximum value reported is 228 grams/day. However, when the 365-day averages are developed, by combining all of the 7-day periods throughout the year, the maximum consumption rate is 78 grams/day. Thus, the short-term estimate overstates the actual long-term maximum by a factor of three. Similarly, when comparing the 95th percentiles reported for these two periods, the 7-day daily average (87.7 grams/day) substantially overestimated the 365-day average (51.1 grams/day) by 72 percent, again demonstrating that the 7-day recall period did not provide a reliable surrogate for long-term consumption behavior at the upper end of the distribution. It is very likely that extrapolating from a 2-day sampling period would further overestimate long-term behavior.

This problem has also been demonstrated and discussed by Ebert et al. (1994), who compared reported rates of self-caught fish consumption based on the duration of the recall period. Ebert et al. reported that when a one-day recall period was used by Pierce et al. (1981) and Puffer et al. (1981), "high-end" (95th percentile) intakes ranged up to 339 grams/day for consumers. When Pao et al. (1982) used a 3-day recall period, the 95th percentile intake for consumers was reported to be 128 grams/day. Using a 30-day recall period, Javitz (1980) reported a 95th percentile intake of 42 grams/day, and when a recall period of one year (365 days) was used (Fiore et al., 1989; Ebert et al. 1993), the 95th percentile estimates for sport-caught fish consumers ranged from 26 to 37 grams/day.

II. Comparison of short-term and longer-term survey data

USEPA has acknowledged that short-term dietary records are problematic when attempting to estimate long-term rates of consumption. In its *Exposure Factors Handbook* (USEPA 2011), USEPA stated that "percentiles of the distribution of average daily intake reflective of long-term consumption patterns cannot in general be estimated using short-term (e.g., one week) data. Such data can be used to adequately estimate mean average daily intake rates (reflective of short- or long-term consumption); in addition, short-term data can serve to validate estimates of usual intake based on longer recall."

As part of its compilation of fish consumption survey data for the 2011 *Exposure Factors Handbook*, USEPA obtained the raw data from a study of Michigan sport anglers that included both a long-term and short-term component (the West et al. 1989 study). The long-term component asked respondents about the frequency of fish meals during each of the four seasons, among other questions. The short-term component was a recall survey of fish meals consumed by all household members during the past 7 days. USEPA used the short-term data to validate the results of the longer recall part of the survey. The results of the analysis showed that there was general agreement between mean consumption estimates made using the 1-year recall and estimates made using the 7-day recall (14.4 grams/day and 14.0 grams/day for consumption of sport-caught fish, respectively), although there was some tendency for infrequent fish consumers to underestimate their usual frequency of fish consumption.

A follow-up survey of Michigan sport anglers was conducted in 1993 (West et al. 1993); this survey used a one-week recall period. The mean consumption rate for sport-caught fish was 16.7 grams/day, similar to the 1989 study. In addition, USEPA (1995) calculated an overall 95th percentile fish consumption rate based on the 1993 7-day recall data; this value, 77.9 grams/day, was about double the 95th percentile estimated using the yearlong consumption data from the 1989 Michigan survey (38.7 grams/day). USEPA states that "because this survey only measured fish consumption over a short (1 week) interval, the resulting distribution will not be indicative of the long-term fish consumption distribution, and the upper percentiles reported form the [1995] U.S. EPA analysis will likely considerably overestimate the corresponding long-term percentiles" (USEPA 2011).

In the Suquamish Tribal fish consumption survey, one of the four studies selected as appropriate for use in developing a Washington State default fish consumption rate in the draft TSD, respondents provided information about fish consumption frequency during the period when specific fish and shellfish were "in season" and "during the rest of the year." In addition, respondents were asked about fish or shellfish they had consumed in the previous 24 hours. All respondents indicated that they are consumers of seafood; however, 55 percent of respondents indicated that they had not consumed fish or shellfish in the previous 24 hours. Correspondingly, the mean consumption rate measured in the 24-hour recall portion of the study (1.5 g/kg-day) was nearly half the consumption rate estimated in the food frequency and portion size survey (2.7 g/kg-day). The authors concluded that "lower mean consumption rate for dietary recall suggests that a brief set of questions does not uncover all forms of consumption."

This conclusion is not consistent with the West et al. (1989) study described above, or with other literature on dietary surveys, which show that while a 24-hour recall does not capture day-to-day variability, on a population level it may provide a more accurate account of the consumption rate than the food frequency survey (Exponent 2012). Some studies indicate that longer-term diet history surveys, such as the Suquamish food frequency survey which looked back over a year, may be more likely to overestimate usual consumption than a 24-hour recall (Rasanen 1979). Others indicate that the upper percentile estimates of fish and shellfish intakes based on a 30-day daily average are lower than those based on two- or three-day daily averages (Tran et al. 2004).

III. Consumption frequency vs. consumption rate estimates

Dietary studies may be conducted for two purposes: (1) to determine the proportion of individuals in a population who are at risk, or (2) to evaluate long-term intake rates of a specific food or food group.

To determine the proportion of individuals in a population who are at risk, habitual intake must be estimated (Rutishauser 2005). In this case, a single 24-hour recall survey is not adequate. Instead, at least two, preferably non-consecutive, days of dietary recall data are needed. Thus, a survey such as the USDA 1994–1996 and 1998 Continuing Survey of Food Intakes by Individuals (CSFII) may be appropriate for evaluating the proportion of individuals in the U.S. population who consume fish; however, this will not provide reliable estimates of individual consumption rates. This limitation is recognized by the authors of the USEPA (2002) *Estimated Per Capita Fish Consumption in the United States* (which used the CSFII data), who note that the limited time period for collecting information on dietary intake does not produce habitual intake estimates. Because short-term studies do not reflect habitual intake, the results of the 24-hour recall portion of the four principle studies used by Ecology were used to *qualitatively* evaluate the habitual intakes reported in the results of the longer-term food consumption surveys, rather than to develop a proposed fish consumption rate.

To evaluate long-term (habitual) intake of a specific food or food group, a food frequency survey is preferable to short-term dietary recall data (Rutishauser 2005). One disadvantage of self-administered surveys (such as mail surveys) is often the lack of detail that is provided. Using trained interviewers to collect diet histories rather than relying on self-administered questionnaires can overcome this problem. The surveys used in the four principle studies used by Ecology were all conducted by personal interview.

Another disadvantage of food frequency surveys for general nutrition studies is the inability to accurately track ingestion of a large range of food types. Rutishauser (2005) recommends restricting the use of food frequency surveys to estimating long-term intake for a limited number of foods. The surveys used in the four principle studies used by Ecology included only questions regarding patterns of habitual seafood consumption.

Basiotis et al. (1987) conducted a statistical analysis on data collected in the year-long USDA Beltsville Human Nutrition Research Center study. The analysis determined the average number of days of dietary intake data needed in order to estimate a "true" average of intakes for a group or for an individual. (In this case, the parameters of interest were energy and various nutrients.) An estimate was determined to be "precise" if it fell within $\pm$ 10% of the average from the 365-day dietary study for that individual or group, 95% of the time. The number of days required varied significantly by the numbers of individuals in the group, as well as by the nutrient of interest. For estimating "true" average daily energy intakes, the average number of days of intake data required ranged from 27 to 35 for individuals, with only three days of data required for a group.

Quantitative methods for adjusting short-term dietary recall data

I. Adjustment of consumption frequency estimates

Carrington and Bolger (2001, 2002) developed a mathematical model to estimate the number of annual servings from short-term dietary surveys. Their model is based on the hypothesis that using results from short-term surveys to project food consumption over longer periods of time may overestimate the amount of food individuals consume and underestimate the number of individuals who consume the food.

First, the algorithm decreases the number of seafood consumption events and increases the number of consumers using a Long-Term to Short-Term Consumer Ratio (LTSTCR). Second, the algorithm adjusts the LTSTCR for frequent seafood consumers using an exponential function that reduces the LTSTCR as the number of servings increases. An example of this function for adjusting daily serving data to projected annual servings is provided as:

$$\text{LTS} = \frac{\text{STS x } 365}{\text{LTSTCR}^{(\alpha/\text{DS})\beta}}$$

Where

LTS = projected annual servings (long-term estimate)
STS = daily servings from short-term survey
LTSTCR = long-term to short-term consumer ratio
DS = daily serving
α = adjustable parameter inversely related to consumption frequency to keep average consumption frequency constant
β = adjustable parameter to determine the shape of the function and keep the low end of the curve consistent with short-term estimates

Estimates based on this model were validated using the resulting seafood consumption rates in an exposure model to relate seafood consumption to levels of mercury in blood and hair. Using data from the 1989–1991 CSFII (USDA 1998), which recorded consumption over a 3-day period, the projected annual seafood servings were estimated as:

$$\text{LTS} = \frac{\text{D3S x } 122}{\text{LTSTCR}^{(x/\text{D3S})}}$$

Where

D3S = 3-day servings from short-term survey
LTSTCR = long-term to short-term consumer ratio = projected % consumers in the total population ÷ % consumers recorded in short-term survey = 70 to 90% ÷ 33.5% = 2.1 to 2.7
x = exponential slope set to maintain the mean per capita consumption near the value measured in the short-term survey

In this assessment, a value of 1.5 was used for *x* to result in a mean per capita seafood consumption that was within 3% of the value reported in the 3-day survey. As an example, for individuals who report consuming 3 servings of seafood over the course of the 3-day survey, the projected annual consumption was calculated as:

$$LTS = \frac{3 \text{ servings/day x } 122}{2.1 \text{ to } 2.7^{(1.5/3)}} = 227 \text{ to } 257 \text{ servings/year}$$

Use of the consumption rates projected in this manner along with Monte Carlo simulation of species distribution and mercury concentration resulted in a close approximation (within a factor of 2 up to the 90th percentile) to mercury biomarker (blood and hair) survey data.

II. Adjustment of consumption rate estimates

Lambe et al. (2000a) evaluated the influence of survey duration on estimates of food intakes using consumption rate data for 32 different foods (including fish) collected for 14 consecutive days from 948 teenagers. This evaluation supported the conclusion that shorter-term surveys used to predict long-term food consumption may overestimate the amount of food individuals consume; however, estimates of group mean intakes are reasonably independent of the survey duration once the sample size is sufficient. The results for fish ingestion are representative of most of the other 31 foods evaluated. Table 1 shows the survey results for fish consumption rates reported after 1, 3, 5, 7, 10, and 14 days.

Table 1. Mean population intake, consumer-only intake, and percent consumers of fish after 1, 3, 5, 7, 10, and 14 days of data collection

Study Length	Mean Population Intake (g/day)	Consumer-only Intake (g/day)	Percent Consumers
1-Day Diary[a]	13	112	12%
3-Day Diary[a]	14	46	30%
5-Day Diary[a]	13	31	41%
7-Day Diary[a]	12	24	49%
10-Day Diary[a]	12	21	56%
14-Day Diary	12	18	63%

Source: Lambe et al. 2000a

[a] Data reported for the first 1, 3, 5, 7, and 10 days of a 14-day diary survey.

While the the mean population intakes showed little difference between the various survey durations, food chemical exposure assessments generally require consumption rates for *consumers* rather than the total population (Renwick 1996), and the consumer-only rates were strongly influenced by survey duration.

Using these data, Lambe et al. (2000b) investigated whether combining the data from a short-term (e.g., 3-day) dietary survey with a long-term food frequency questionnaire could produce consumer-only consumption rate estimates comparable to those calculated from a 14-day food diary based on the premise that consumer-only intakes are really the mean total population intakes adjusted for % consumers. As shown in Table 2 for fish ingestion, Lambe et al. demonstrated that the mean total population intakes, based on a 3-day food diary, can be divided by total % consumers, based on a food frequency questionnaire, to provide values comparable to the mean consumer-only intakes from a 14-day food diary.

The adjusted mean consumer-only fish intake calculated as the total (consumers and non-consumers) fish intake from the 3-day diary (14 g/day) divided by the total % consumers from the food frequency questionnaire (57%) is 25 g/day and is similar to the consumer-only intake reported from the 14-day diary survey of 18 g/day.

This approach has been applied to the results of the short-term study data reported by The Suquamish Tribe (2000) in Table 3 and to the national (CSFII) data reported by USEPA (2002) in Table 4.

Table 2. Adjustment of short-term (3-day) to long-term estimates and comparison to longer-term (14-day) fish consumption rates (Lambe et al. 2000b)

Parameter	3-Day Diary[a]		14-Day Diary	Long-Term % Consumers[b]
Mean Population Intake	14 g/day		12 g/day	--
Consumer-only Intake	46 g/day		18 g/day	--
Percent Consumers	30%		63%	57%
Consumer-only 3-day recall adjusted for long-term exposure	14 g/day / 57% consumers	=	25 g/day	

Source: Lambe et al. 2000b

[a] Data reported for the first 3 days of a 14-day diary survey.

[b] From food frequency survey.

-- = intake rates not calculated from food frequency questionnaire

Table 3. Adjustment of short-term (24-hour recall) to long-term estimates and comparison to long-term fish consumption rates - Suquamish Tribe study

Parameter	24-hour Recall		Long-Term Recall	Long-Term % Consumers[b]
Mean Population Intake	119 g/day		214 g/day	--
Consumer-only Intake	263[a] g/day		214 g/day	--
Percent Consumers	45%		100%	97%
Consumer-only 1-day intake adjusted for long-term exposure	119 g/day / 97% consumers	=	123 g/day	

Source: The Suquamish Tribe 2000

[a] Calculated as 119 g/day/45%

[b] Upper-bound of non-consumers reported as 3% (The Suquamish Tribe 2000)

-- = intake rates not calculated from food frequency survey

Table 4. Adjustment of short-term (2-day recall) to long-term estimates and comparison to long-term fish consumption rates – General Population (CSFII Data)

Parameter	2-day Recall	Long-Term Recall	Long-Term % Consumers[b]
Mean Population Intake	19.9 g/day	--	--
Consumer-only Intake	74.3[a] g/day	--	--
Percent Consumers	27%	--	85.4%
Consumer-only 2-day recall adjusted for long-term exposure	$\frac{19.9 \text{ g/day}}{85.4\% \text{ consumers}}$	=	23.3 g/day

Source: USEPA 2002

[a] Calculated as 74.3 g/day/27%.

[b] Percent of people who never eat meat reported as 14.6 (+ 3%) in (Stahler 2006)

-- = not reported

Applying this approach results in minor upward adjustments of the Suquamish 24-hour recall (from 119 g/day to 123 g/day) and United States 2-day recall (from 19.9 g/day to 23.3 g/day) survey data. The relatively small impact of the adjustment is due to the assumption that a large percentage of the population eats fish (97% for the Suquamish population and 85.4% for the U.S. population).

III. Statistical reanalysis of national fish consumption data

Ecology conducted a statistical reanalysis of national and regional-specific fish consumption data to derive consumption estimates for different fish species groups (e.g., shellfish, anadromous, non-anadromous), estimates of fish consumed based on harvest patterns, and long-term national fish consumption estimates statistically derived from short-term episodic[2] dietary data. Detailed descriptions of the statistical methods used to reevaluate the national and regional-specific data are presented in *Draft Statistical Analysis of National and Washington State Fish Consumption Data* (Polissar et al. 2012), a companion document to the Fish Consumption Rate TSD.

The methodology developed by Tooze et al. (2006), which estimates the usual intake of episodically consumed foods such as fish, was used to reassess data from individual responses to the 2003–2006 National Health and Nutrition Examination Survey (NHANES).

These methods were not applied to the regional-specific fish consumption data due to differences in data collection methods: the national data were based on two 24-hour recall surveys, while the regional data were based on food frequency questionnaires.

Detailed descriptions of the methodology are provided in the following technical literature:

[2] "Episodic" refers to consumption that is intermittent or occasional.

- Tooze et al. 2006. A New Statistical Method for Estimating the Usual Intake of Episodically Consumed Foods with Application to Their Distribution. *Journal of the American Dietetic Association,* October 2006, 106: 1575-1587
- Kipnis et al. 2009. Modeling Data with Excess Zeros and Measurement Error: Application to Evaluating Relationships between Episodically Consumed Foods and Health Outcomes. *Biometrics*, December 2009, 65: 1003-1010.
- Subar et al. 2006. The Food Propensity Questionnaire: Concept, Development, and Validation for Use as a Covariate in a Model to Estimate Usual Food Intake. *Journal of the American Dietetic Association*, October 2006, 106: 1556-1563.
- Keogh and White, 2011. Allowing for never and episodic consumers when correcting for error in food record measurements of dietary intake. *Biostatistics* (2011), 12, 4, PP. 624-636.
- Dodd et al. 2006. Statistical Methods for Estimating Usual Intake of Nutrients and Foods: A Review of the Theory. *Journal of the American Dietetic Association*, October 2006, 106: 1640-1650.

The fish consumption estimates resulting from this reanalysis are presented below. For comparison, national estimates were also derived based on standard statistical methods that did not account for estimating the usual, or long-term, fish dietary consumption from episodically consumed foods. These estimates are also presented below.

Table 5. Adult U.S. National Fish Consumption Rates, Consumers Only (From NHANES 2003–2006)

Usual Intake Based on Tooze et al. Methodology (Data from NHANES 2003-2006)				
Population National	Species Group	Descriptive Statistics (grams/day)		
		50th Percentile	90th Percentile	95th Percentile
National U.S. Population	all	12.7	42.5	56.6
	finfish	9.0	31.8	43.3
	shellfish	2.4	13.2	20.5
Adapted from Table 4, Polissar et al., 2012				

Observed Consumption on Two Days (Data from NHANES 2003-2006)				
Population National	Species Group	Descriptive Statistics (g/day)		
		50th Percentile	90th Percentile	95th Percentile
National U.S. Population	all	37.9	127.9	168.3
	finfish	34.6	115.3	149.8
	shellfish	25.7	100.5	146.6
Adapted from Table 3, Polissar et al., 2012				

Summary

Because food intake rates from short-term surveys are highly variable and result in a wide range of fish consumption patterns, these surveys tend to overestimate the prevalence of low and high intake rates. This variation is particularly relevant for assessments of food chemical exposure where the parameters of interest are at the extremes of the exposure distribution rather than at the center (Lambe 2002).

Attempts to account for the variance and uncertainty associated with the use of short-term consumption studies have generally included qualitative evaluation of data from a range of sources, coupled with consideration of the intended use of the data. To evaluate long-term (habitual) seafood intake, longer-term survey data are preferable to short-term dietary survey data.

While there are mathematical approaches that attempt to adjust short-term recall data for longer-term consumption estimates, such adjustments are complex, do not provide a clear improvement in the fish consumption rate estimate, and are themselves subject to uncertainty. If the percentage of fish consumers is high in the population being considered, these adjustments result in a relatively minor change in the estimated fish consumption rate.

Ecology conducted a statistical reanalysis of short-term national fish consumption data to estimate long-term (usual) national fish consumption rates, using the methodology of Tooze et al. (2006). National fish consumption rate estimates based on this reanalysis are significantly lower than estimates based on simple extrapolation of the short-term fish consumption data. Additional information is provided in Polissar et al. (2012).

References

AMEC. 2003. Evaluation of the Fish Consumption Rate Selected by Oregon DEQ for the Development of Ambient Water Quality Criteria. Submitted to Northwest Pulp and Paper Association, Bellevue, WA. Prepared by Ellen S. Ebert and Paul Anderson, AMEC Earth & Environmental. May 2003.

Basiotis, P., Welsh, S., Cronin, F., Kelsay, J., and Mertz, W. 1987. Number of Days of Food Intake Records Required to Estimate Individual and Group Nutrient Intakes with Defined Confidence. *The Journal of Nutrition*, May 1987 http://jn.nutrition.org/content/117/2/local/ed-board.pdf.

Carrington, C.D., Bolger, M.P. 2001. Methods for Projecting Long-Term Dietary Exposure from Short-Term Survey Data for Environmental Contaminants. Center for Food Safety and Applied Nutrition, U.S. Food and Drug Administration, College Park Maryland, USA. Toxicology and Industrial Health 2001 17:176-179.

Carrington, C.D., Bolger, M.P. 2002. An exposure assessment for methylmercury from seafood for consumers in the United States. Center for Food Safety and Applied Nutrition, U.S. Food and Drug Administration, Washington, DC 20204, USA. *Risk Anal.* August 22(4):689-99.

CRITFC (Columbia River Inter-Tribal Fish Commission). 1994. A fish consumption survey of the Umatilla, Nez Perce, Yakima, and Warm Springs Tribes of the Columbia Basin. Technical Report 94-3. October 1994.

Ebert, E.S., Harrington, N.W., Boyle, K.J., Knight, J.W., and Keenan, R.E. 1993. Estimating consumption of freshwater fish among Maine anglers. *N. Am. J. Fish. Mgt.* 13:737-745.

Ebert, E.S., Price, P.S., and Keenan, R.E. 1994. Selection of Fish Consumption Estimates for Use in the Regulatory Process. *J. Expos. Anal. Environ. Epidemiol.* 4(3):373-393.

Exponent. 2012. Letter from Mark W. Johns and Michael R. Garry, Exponent, to Washington State Department of Ecology, Re: Comments on Publication No. 11-09-050, Fish Consumption Rates Technical Support Document, A Review of Data and Information about Fish Consumption in Washington. January 17, 2012.

Fiore, B.J., Anderson, H.A., Hanrahan, L.P., Olson, L.J., and Sonzogni, W.C. 1989. Sport fish consumption and body burden levels of chlorinated hydrocarbons: A study of Wisconsin anglers. *Archives of Environmental Health* 44(2):82-88. As cited in AMEC 2003.

Javitz, H. 1980. Seafood Consumption Data Analysis: Final Report. Prepared by SRI International for USEPA Office of Water Regulations and Standards. September 24, 1980. As cited in AMEC 2003.

Keogh and White. 2011. Allowing for never and episodic consumers when correcting for error in food record measurements of dietary intake. *Biostatistics* (2011), 12, 4, PP. 624-636.

Kipnis, V., Midthune, D., Buckman, D.W., Dodd, K.W., Guenther, P.M., Krebs-Smith, S.M., Subar, A.F., Tooze, J.A., Carroll, R.J., and Freedman, L.S. 2009. Modeling Data with Excess Zeros and Measurement Error: Application to Evaluating Relationships between Episodically Consumed Foods and Health Outcomes. Biometrics, December 2009, 65: 1003-1010.

Lambe, J., Kearney, J., Leclercq, C., Zunft, H., De Henauw, S., Lamberg-Allardt, C., Dunne, A., and Gibney, M. 2000a. The Influence of Survey Duration on Estimates of Food Intakes and its Relevance for Public Health Nutrition and Food Safety Issues. *European Journal of Clinical Nutrition* (2000) 54, 166-173.

Lambe, J., Kearney, J., Leclercq, C., Berardi, D., Zunft, H., Sulzer, S., De Henauw, S., De Volder, M., Lamberg-Allardt, Karkkainen, M., C., Dunne, A., and Gibney, M. 2000b. Enhancing the Capacity of Food Consumption Surveys of Short Duration to Estimate Long Term Consumer-Only Intakes by Combination with a Qualitative Food Frequency Questionnaire. *Food Additives and Contaminants*, 2000, Vol. 17, No. 3, 177-187.

Lambe, J. 2002. The Use of Food Consumption Data in Assessments of Exposure to Food Chemicals Including the Application of Probabilistic Modeling.

Mertz, W. and Kelsay, J. 1984. Rationale and design of the Beltsville one-year dietary intake study. *Am. J. Clin. Nutr.* 40(Suppl. 6):1323-1326.

Pao, E.M., Fleming, K.H., Guenther, P.M., and Mickle, S.J. 1982. Foods commonly eaten by individuals: amount per day and per eating occasion. Home Economic Report No. 44. U.S. Department of Agriculture, Washington, DC. As cited in USEPA 2011.

Pierce, D., Noviello, D., and Rogers, S.H. 1981. Commencement Bay Seafood Consumption Study: Preliminary Report. Tacoma-Pierce County Health Department, Seattle, WA. December 1981. As cited in AMEC 2003.

Polissar, N.L., Neradilek, M., Aravkin, A.Y., Danaher, P., and Kalat, J. *Draft Statistical Analysis of National and Washington State Fish Consumption Data.* Prepared for Washington State Department of Ecology by The Mountain-Whisper-Light Statistics, Seattle, Washington. June 2012.

Puffer, J.W., Azen, S.P., Duda, M.J., and Young, D.R. 1982. Consumption rates of potentially hazardous marine fish cuaght in the metropolitan Los Angeles area. Grant R 807 120010. University of Southern California School of Medicine for Environmental Research Laboratory. As cited in AMEC 2003.

Rasanen, L. Nutrition survey of Finnish rural children. VI. Methodological study comparing the 24-hour recall and the dietary history interview. *Am. J. Clin. Nutr.* 32:2560-2567. December 1979.

Renwick, A. G., 1996. Needs and Methods for Priority Setting for Estimating the Intake of Food Additives. *Food Additives and Contaminants*, 13, 467-475.

Rutishauser, I. 2005. Dietary Intake Measurements. *Public Health Nutrition*: 8(7A), 1100-1107.

Sechena, R., Nakano, C., Liao, S., Polissar, N., Lorenzana, R., Truong, S., and Fenske, R. Asian and Pacific Islander Seafood Consumption Study (EPA 910/R-99-003). May 27, 1999.

Stahler, C. 2006. How Many Adults Are Vegetarian? Results of a survey conducted by Harris Interactive® on behalf of The Vegetarian Resource Group. *Vegetarian Journal* 2006 Issue 4. http://www.vrg.org/journal/vj2006issue4/vj2006issue4poll.htm

Subar, A.F., Dodd, K.W., Guenther, P.M., Kipnis, V., Midthune, D., McDowell, M. Tooze, J.A., Freedman, L.S., Krebs-Smith, S.M. 2006. The Food Propensity Questionnaire: Concept, Development, and Validation for Use as a Covariate in a Model to Estimate Usual Food Intake. *Journal of the American Dietetic Association,* October 2006, 106: 1556-1563.

The Suquamish Tribe 2000. Fish Consumption Survey of the Suquamish Indian Tribe of the Port Madison Indian Reservation.

Dodd, K.W., Guenther, P.M., Freedman, L.S., Subar, A.F., Kipnis, V., Midthune, D., Tooze, J.A., Krebs-Smith, S.M. 2006. Statistical Methods for Estimating Usual Intake of Nutrients and Foods: A Review of the Theory. *Journal of the American Dietetic Association*, October 2006, 106: 1640-1650.

Tooze, J.A., Midthune, D., Dodd, K.W., Freedman, L.S., Krebs-Smith, S.M., Subar, A.F., Guenther, P.M., Carroll, R.J., Kipnis, V. 2006. A New Statistical Method for Estimating the Usual Intake of Episodically Consumed Foods with Application to Their Distribution. Journal of the American Dietetic Association, October 2006, 106: 1575-1587

Toy, K.A., Polissar, N.L., Liao, S., Mittelstaedt, G.D. 1996. A fish consumption survey of the Tulalip and Squaxin Island Tribes of the Puget Sound Region. Tulalip Tribes, Department of the Environment, Marysville, Washington.

Tran, N.L., Barraj, L., Smith, K., Javier, A., and Burke, T.A. 2004. Combining Food Frequency and Survey Data to Quantify Long-Term Dietary Exposure: A Methyl Mercury Case Study. *Risk Analysis*, Vol. 24, Issue 1, pp 19-30. February 2004.

USEPA (U.S. Environmental Protection Agency). 1992. Consumption Surveys for Fish and Shellfish: A Review and Analysis of Survey Methods. U.S. Environmental Protection Agency, Office of Water. EPA 822/R-92-001. February 1992.

USEPA. 1995. Fish consumption estimates based on the 1991-1992 Michigan sport anglers fish consumption study. Final Report. Prepared by SAIC for the Office of Science and Technology, Washington, DC. As cited in USEPA 2011.

USEPA. 1998. Guidance for Conducting Fish and Wildlife Consumption Surveys. U.S. Environmental Protection Agency, Office of Water. EPA-823-B-98-007. November 1998.

USEPA. 2000. Methodology for deriving ambient water quality criteria for the protection of human health. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology. EPA-822-B-00-004. October 2000.

USEPA 2002. Estimated Per Capita Fish Consumption in the United States. EPA-821-C-02-003.

USEPA. 2011. Exposure Factors Handbook: 2011 Edition. EPA/600/R-090/052F. U.S. Environmental Protection Agency Office of Research and Development, Washington, DC. September 2011.

West, P.C., M.J. Fly, R. Marans, and F. Larkin. 1989. Michigan sport anglers fish consumption survey. A report to the Michigan Toxic Substance Control Commission. Michigan Department of Management and Budget, Contract No. 87-20141. As cited in AMEC 2003.

West, P.C., Fly, J.M., Marans, R., Larkin, F., and Rosenblatt, D. 1993. 1991-92 Michigan Sport Anglers Fish Consumption Study. Prepared by the University of Michigan, School of Natural Resources, for the Michigan Department of Natural Resources. Technical Report No. 6. May 1993. As cited in AMEC 2003.

Technical Issue Paper

Recreational Fish Consumption Rates

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Recreational Fish Consumption Rates

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DEPARTMENT OF
ECOLOGY
State of Washington

Toxics Cleanup Program
Washington State Department of Ecology
Olympia, Washington

Table of Contents

Appendices

List of Tables

Tables

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Introduction

The draft *Fish Consumption Rates Technical Support Document* (TSD) identified four fish/shellfish dietary surveys as appropriate for use in establishing a technically defensible default fish consumption rate for Washington. These surveys included two Native American fish/shellfish dietary surveys for three tribal populations in Puget Sound, one Native American finfish dietary survey for four tribal populations in and around the Columbia River basin, and one Asian and Pacific Islander fish/shellfish dietary survey from King County. In addition, the draft TSD provided information on United States per capita fish consumption, based on the U.S. Department of Agriculture's 1994–1996 and 1998 food intake surveys.

The draft TSD did not provide specific information about consumption of fish and shellfish by recreational fishers.[1] Ecology received review comments requesting the inclusion of additional fish consumption information for this receptor group. This issue paper summarizes regional-specific and selected national fish dietary information for recreational fishers. It is a targeted examination of the issues raised by review comments received on the draft Fish Consumption Rates TSD, and was prepared within a limited time frame. Therefore, it may not include all available information on this subject.

Although data for the general population is useful for evaluating fish consumption rates, data on recreational fishing are needed to assess exposure to individuals with potentially higher fish consumption levels. Recreational fishers may consume fish more frequently, and may consume larger portions at each meal, than the general population. In addition, they may frequently fish from a single contaminated source. These factors may put recreational fishers at higher risk of exposure to contaminants in fish and shellfish.

[1] In this issue paper, the term "fisher" denotes a person who fishes for any type of seafood by any method, including finfish and shellfish. The term "angler" refers to a person who fishes with hook and line. Some fish consumption surveys reviewed during preparation of this issue paper use the terms "fisher", "fisherman", and "angler" interchangeably.

Marine Recreational Studies

This section describes fish consumption rate studies for marine and estuarine finfish and shellfish. Four recreational fish consumption studies have been conducted in marine/estuarine areas of Washington State. Three of these were conducted in the early to mid-1980s, over 25 years ago (Pierce et al. 1981; McCallum 1985; Landolt et al. 1985, 1987), and the fourth was conducted in 2007. All four studies were creel surveys, in which anglers were interviewed while fishing. They were asked questions about how much fish they had caught on the day of the survey, and the number of family members with which they intended to share the catch. These data do not represent usual behavior because an angler does not catch the same number of fish on each fishing occasion.

In a creel study, the target population is typically anyone who fishes at the location being studied. Creel studies may not provide data representative of the target population, because the probability of being sampled is not the same for all members of the target population (USEPA 2011). The Environmental Protection Agency's (EPA) Exposure Factors Handbook (2011) describes the problem as follows:

> "For instance, if the survey is conducted for 1 day at a site, then it will include all persons who fish there daily, but only about 1/7 of the people who fish there weekly, 1/30 of the people who fish there monthly, etc. In this example, the probability of being sampled (or inverse weight) is seen to be proportional to the frequency of fishing. However, if the survey involves interviewers revisiting the same site on multiple days, and persons are only interviewed once for the survey, then the probability of being in the survey is not proportional to frequency; in fact, it increases less than proportionally with frequency."

In most creel studies, there is no mention of sampling weights; by default, all weights are set to one, implying equal probability of sampling. Because the sampling probabilities in a creel survey, even with repeated interviewing at a site, are highly dependent on fishing frequency, the fish intake distributions reported for these surveys may overestimate intake rates of the total fishing population that uses the sampled water body (USEPA 2011).

Price et al. (1994) proposed using the inverse frequency of fishing to adjust data derived from creel surveys to statistically "correct" for the increased probability of encountering frequent fishers (relative to infrequent fishers). Price's reanalysis of the Commencement Bay Seafood Consumption Study data is described below (Section I).

Keill and Kissinger (1997, pp. 21-22) noted that multiplying the number of infrequent anglers by the inverse of the number of days that they fish per year results in greater weight being given to the infrequent angler. For example, an individual who fishes once a year would be assigned a weight 365 times an individual who fishes daily. Keill and Kissinger (1997) maintained that this adjustment obscures and negates the importance of the subpopulation most likely to incur exposure to contaminated fish. They noted that some researchers have specifically excluded infrequent anglers from surveys, and/or nonconsumers from analyses, in order to avoid characterizing consumption for a population that is not at risk.

CalEPA (2001) stated that many factors can affect the relationship between fishing frequency and the probability of sampling, including the actual timing and frequency of sampling, the methods for selecting respondents, and whether repeat interviews occur. In addition, a fisher's frequency of fishing may be correlated with the time of day, day of week, climate and seasonal factors, and other parameters that may not be equivalent across sampling days. These factors are not accounted for in the basic weighting adjustment used by Price et al. (1994).

Other limitations associated with creel studies were described in the draft Fish Consumption Rate TSD, as well as by CalEPA (2001) and Kissinger (2010). These include:

- Difficulties in eliciting year-round consumption information;
- Difficulties in using visual aids or other materials to assist in quantifying consumption;
- Inability to obtain a random, unbiased sample of the study population;
- Willingness (or unwillingness) of the interviewee to provide information in a field situation when the interviewee would rather be fishing;
- Concerns about accurate transmission of fishing information due to trust issues;
- Language and communication issues if anglers do not speak English;
- Inadequate measurement of fish consumption over the geographic area of concern;
- Inadequate assessment of fishing activity at all times when fishing might occur;
- Underestimation of catch associated with measuring catch prior to the end of the fishing activity; and
- Problems in relating an angler's catch to consumption rates of the angler's family and community.

The Washington State studies are discussed below. Additional studies of marine/estuarine fish consumption rates are summarized in Table A-1 (Appendix A of this Technical Issue Paper). These include other studies on the West Coast, particularly along the California coast, as well as studies from New York/New Jersey, Texas, and Delaware.

I. Commencement Bay Seafood Consumption Study (Pierce et al. 1981)

Pierce et al. (1981) conducted a creel survey in 1980 for the Tacoma-Pierce County Health Department to study seafood consumption patterns and demographics of sport fishermen in Commencement Bay, near Tacoma. The objectives of the survey were to: determine the extent to which local species of fish and shellfish were used as food; determine which species were most commonly consumed; and assess methods of preparation of the catch. Only successful anglers were interviewed, and salmon were excluded from the study because they were considered to have minimal contact with Commencement Bay pollutants.

Surveys of shore and boat anglers were conducted in the mornings and evenings, and each subarea was sampled five times during the summer and four times during the fall, except for one area that was sampled only twice during the fall. The boat fishing area was sampled four times during the fall only. Interviews were conducted only with anglers who had successfully caught fish, and each angler was interviewed only once during the survey period. The following data were recorded: species, wet weight, size of the family, place of residence, fishing frequency, planned uses of the fish, age, sex, and race. Sampling periods lasted as late as 1:00 a.m.; however, the study authors noted that a considerable portion of the total catch was suspected to have been obtained during all-night fishing (CalEPA 2001).

A total of 304 interviews of shore anglers were conducted during the summer of 1980 (July to September), and 204 interviews of shore and boat anglers were conducted in the fall (September to November). About 60 percent of the anglers interviewed were white, 20 percent were black, 19 percent were Asian, and the remainder was Hispanic or Native American (USEPA 2011). The dominant species caught were Pacific hake and walleye pollock. Pierce et al. (1981) did not present a distribution of fish intake or a mean fish intake rate.

In 1994, Price et al. obtained the raw data from this survey and re-analyzed it using sampling weights proportional to inverse fishing frequency. Price et al. (1994) calculated a median intake rate of 1.0 g/day and a 90th percentile rate of 13 g/day. When equal weights were applied, the median intake rate was 19 g/day and the 90th percentile was 155 g/day. According to USEPA, all of these values are probably underestimates, because the sampling probabilities are less than proportional to fishing frequency; thus, the true median for the target population is probably somewhat above 1.0 g/day, and the true 50th percentile of the resource utilization distribution is probably higher than 19 g/day (USEPA 2011).

Limitations to this study included the following (CalEPA 2001):

- Local seafood contamination in this area may have suppressed the reported fish consumption rate;
- Salmon were excluded from the fish consumption rate;
- Anglers who had not caught any fish at the time of the interview were not included; and
- Sampling was conducted in summer and fall only; no samples were collected in winter or spring.

II. Seafood Catch and Consumption in Urban Bays of Puget Sound (McCallum 1985)

McCallum (1985) studied shore-based recreational fishers in Elliott Bay, Everett Harbor (Port Gardner), and Sinclair Inlet. The objective of the study was to collect information on the species consumed, the amount consumed, the frequency of fish or shellfish consumption, and the location of collection. Of particular concern was whether Southeast Asian immigrants or other low income groups were frequent users of urban fishing sites and whether they consumed the more contaminated species (e.g., sole) or the more contaminated parts (e.g., fish liver, crab hepatopancreas).

Shore-based fishing, crabbing, or clamming sites within contaminated bays were sampled. On-site interviews were conducted between July 1983 and June 1984. Initial survey attempts were conducted on different days of the week and at different hours of the day. Subsequently, survey effort was adjusted to high use days/times. The survey schedule was also modified to reflect seasonal changes in fishing, crabbing, or clamming. Only those fishers seeking bottomfish, crabs, or clams were fully interviewed. Anglers fishing for salmon or squid were excluded, as these species are highly migratory and were not expected to be impacted by contaminants in the urban bays.

Tetra Tech (1988) calculated a 95th percentile consumption rate of 24.3 g/day based on the data collected during this study.

Limitations to this study included (Kissinger 2010):

- Language barrier problems in conducting interviews;
- Many individuals refused to be interviewed; and
- Interviews were not conducted at night (midnight to 6 a.m.), a time period when fishing may have occurred.

III. 1983-1984 Puget Sound Survey (Landolt et al. 1985, 1987)

Landolt et al. (1985, 1987) conducted a two-year study of recreational fish and shellfish catch and consumption from four urban embayments of Puget Sound: Commencement Bay, Elliott Bay, Sinclair Inlet, and Edmonds. The objectives of the study were to: identify the most commonly consumed species; assess the demographic characteristics of the fishing population; describe patterns of consumption; and estimate the quantity of selected chemicals consumed by anglers and their families.

Personal interviews were used to obtain species-specific catch and consumption information. Over 4,000 shore fishers were interviewed during the first year (November 1983 through November 1984). Sampling times were initially selected at random, and interviews were conducted during all times of the day. After the preferred fishing times were identified, sampling focused on those times when the largest number of fishers were expected (CalEPA 2001). The second year of the study focused primarily on chemical analysis of tissue samples, but catch and consumption patterns for 437 boat anglers at two of the four embayments (Elliott Bay, Commencement Bay) were also evaluated during February to October 1985.

Calculations of consumption rates were based on estimates of the weight of the catch (fish in hand) divided by the number of consumers in the household, and by the number of days since fish caught at the same site were last eaten. This value was multiplied by a cleaning factor of 0.3 for fish and 0.49 for squid and crab, to derive the mean daily grams of available edible portion per person. Geometric means were then calculated for each embayment and ethnic group. An overall geometric mean daily fish consumption rate of 11 g/day for all ethnic groups and species was calculated.

Geometric means ranged from 8 to 14 g/day per person. Consumption rates for the most common species were significantly higher than the average. The most common species, squid, was consumed at a geometric mean rate of 39 g/day, but only during the fall. Landolt et al. (1987) noted that boaters fished primarily for salmon, and consumed 51.7 g/day of King salmon from the two bays where boat anglers were surveyed (CalEPA 2001). Correlations were made between ethnicity and fishing mode, time of fishing, household size, parts of fish consumed, and seasonality of fishing.

Limitations to this study included:

- Shore anglers were not interviewed at the end of their fishing trips, potentially underestimating consumption; and
- Measurement of shellfish intake was limited to crab.

IV. Survey of Fish Consumption Patterns of King County Recreational Anglers (Mayfield et al. 2007)

Mayfield et al. (2007) summarized results of two fish consumption surveys conducted among recreational anglers at marine and estuarine sites in King County.[2]

The first survey was conducted during a 10-week period from June 1997 to August 1997 at marine and estuarine public parks and boat launches throughout Elliott Bay and the Duwamish River. Simmonds et al. (1998) originally summarized the results of the survey but did not perform a detailed analysis of fish and shellfish consumption rates. Surveys were conducted on weekends and 10 randomly selected weekdays, over a period of 10 weeks. Locations were visited at least twice a day (morning and afternoon) between the hours of 5:00 am and 10:00 pm. A total of 807 and 152 anglers were surveyed in Elliott Bay and the Duwamish River, respectively.

The second survey was conducted between March 2001 and March 2002 at marine locations in North King County and Snohomish County. This study followed a random stratified design and was conducted on both weekdays and weekends, typically between 7:00 am and 8:00 pm. Locations were visited randomly, and surveyors attempted to interview as many anglers as possible within a 1-hour site visit. The interviews typically were 5 to 10 minutes each. A total of 228 anglers were surveyed in North King and Snohomish Counties.

Data were collected on fishing location preferences, fishing frequency, consumption amounts, species preferences, cooking methods, and whether family members would also consume the catch. Respondent demographic data were also collected. Consumption rates were estimated using information on fishing frequency, weight of the catch, a cleaning factor, and the number of individuals consuming the catch. The majority of anglers surveyed were over male, 15 years and older, and were either Caucasian or Asian/Pacific Islander.

[2] Mayfield eta l. (2007) also summarized results of a fish consumption survey among recreational anglers at freshwater locations; these results are discussed in the Freshwater Recreational Studies section of this issue paper.

Mayfield et al. (2007) used the harvest method to calculate fish consumption rates; this method estimates consumption by combining information on fishing frequency and the weight of fish caught during the time of the interview. Due to the limited number of fish actually measured during the time of the interviews, a mean value for total fish weight was used to calculate consumption. The mean weights for uncleaned fish and shellfish caught by anglers were 1,574 and 1,053 g/catch at Elliott Bay, 544 and 821 g/catch at Duwamish River sites, and 1,035 and 683 g/catch in North King and Snohomish Counties. The North King County survey did not query for the number of fish consumers in the family; an average family size of 2.5 was used.

Mean recreational marine fish and shellfish consumption rates were 53 g/day and 25 g/day, respectively (Table 1). Mayfield et al. (2007) also reported differences in intake according to ethnicity. Mean marine fish intake rates were 73, 60, 50, 43, and 35 g/day for Native American, Caucasian, Asian and Pacific Islander, African American, and Hispanic/Latino respondents, respectively.

Table 1. Fish and Shellfish Consumption Rates (g/day) for Marine Recreational Fishers in King County, WA

Location	Sample Size	Mean	SD	SE	50th Percentile	90th Percentile	95th Percentile
Marine Fish Consumption							
Duwamish River	50	8	13	2	2	23	42
Elliott Bay	377	63	91	5	31	145	221
North King County	67	32	40	5	17	85	102
All Locations	494	53	83	4	21	121	181
Shellfish Consumption							
Duwamish River	16	20	33	8	3	77	123
Elliott Bay	49	28	33	5	14	74	119
North King County	31	22	33	6	12	62	132
All Locations	96	25	33	3	11	60	119

Source: Mayfield et al. (2007)

SD = Standard deviation; SE = Standard error

Limitations to this study included:

- The Elliott Bay survey was conducted over 10 weeks during peak summer fishing activity;
- A substantial number of individuals refused to have their catch weighed, and/or refused to be interviewed;
- Surveys were very brief (5 to 10 minutes); and
- The study does not address how language issues were dealt with in administering the survey to non-English speakers.

Freshwater Recreational Studies

This section describes fish consumption rate studies for freshwater fish and shellfish. Three recreational fish consumption studies have been conducted in the freshwater areas of Washington State: Lake Roosevelt (WDOH 1997), Lake Whatcom (WDOH 2001), and King County lakes (Mayfield et al. 2007, Parametrix 2003).

Additional studies of freshwater fish and shellfish consumption rates are summarized in Table A-2. These include studies in Oregon, California, South Carolina, Tennessee, Minnesota, North Dakota, Indiana, Georgia, Connecticut, Alabama, Michigan, New York, Maine, and Wisconsin.

I. Consumption Patterns of Anglers Who Frequently Fish Lake Roosevelt (WDOH 1997)

A fish consumption survey was conducted at Lake Roosevelt during August through November 1994 and May through September 1995. The purpose of the survey was to evaluate the consumption patterns of anglers who repeatedly fished from the lake, in order to assess the public health impacts associated with ingestion of chemically-contaminated fish (WDOH 1997). The study included a creel survey (trip length information, fish species, length, and weight) and shoreside interviews of boat anglers upon return from their fishing trips. The consumption survey form (used during the shoreside interviews) was separate from the creel survey. Lake Roosevelt was divided into three sections (upper, middle, and lower); morning and afternoon survey locations were randomly selected from a total of 48 possible locations. Anglers who did not consume Lake Roosevelt fish and anglers who were previously surveyed were excluded during data analysis. A total of 348 survey responses were used to estimate fish consumption rates.

Surveyed individuals were primarily older adult Caucasian males that are part of two-adult households in which both individuals consume fish. Results indicate that surveyed anglers consume an average of 42 meals per year, with over 90 percent consuming 103.2 meals or less per year (2 meals per week). Fish are consumed primarily as fillets, and rainbow trout and walleye are preferred above kokanee and bass. No sturgeon, sucker, or whitefish were reported caught and consumed. Consumption rates were estimated based on an assumed 8-ounce meal size (Table A-2).

Limitations of this study include:

- Reported consumption rates may have been depressed due to the presence of chemical contamination in Lake Roosevelt;
- The survey questions were not adequate for determining fish meal size;
- Combining the consumption interviews with the creel survey resulted in a survey that was too lengthy;
- Some anglers were anxious to depart the boat launch facilities at the completion of their fishing trip, and may have provided responses in haste.

II. Lake Whatcom Residential and Angler Fish Consumption Survey (WDOH 2001)

WDOH conducted a survey of Lake Whatcom residents and anglers in July 2000, to gather information on consumption of fish caught in Lake Whatcom and perceptions related to fish advisories (WDOH 2001). A retrospective on-site study design with a four-week recall period was selected for this survey. Three populations were surveyed: residents who live on or near the lake or in developments with direct access to the lake; boat anglers accessing the lake at public boat launch facilities; and shore anglers at public access points. Surveys were conducted through door-to-door interviews, at frequently used boat launch facilities, and at popular shore-fishing locations. A total of 220 surveys were completed, however only 10 shore anglers and 16 boat anglers were surveyed; 40 percent and 6 percent of these, respectively, reported eating fish in the previous four weeks. Of the 194 residents that were surveyed, 22 percent identified themselves as fishers, and only 2 percent reported eating fish in the previous four weeks.

The primary species caught and consumed were cutthroat trout and smallmouth bass. All anglers reported consuming Lake Whatcom caught fish three or fewer months per year. Average meal size for shore and boat anglers was 256 grams (9 ounces). The mean fish consumption rate for consumers was estimated at 6.0 g/day (ODEQ 2008).

Limitations of this survey include the following (ODEQ 2008):

- Small sample size and low number of people who reported consuming fish during the previous four weeks;
- Reports of mercury in Lake Whatcom fish may have affected consumption rates;
- Accurate frequency of meals per week or month was not clearly presented;
- The calculated fish consumption rate was based on a number of assumptions, including average fillet weight for each species.

III. Survey of Fish Consumption Patterns of King County Recreational Anglers (Parametrix 2003)

Parametrix (2003) conducted a series of freshwater surveys between 2002 and 2003 at locations around Lake Sammamish, Lake Washington, and Lake Union. Mayfield et al. (2007) included this study in their survey of fish consumption patterns of King County recreational fishers.

This study followed a random stratified design and was conducted on both weekdays and weekends, typically between 7:00 am and 8:00 pm. Locations were visited randomly, and surveyors attempted to interview as many anglers as possible within a 1-hour site visit. The interviews typically were 5 to 10 minutes each (Mayfield et al. 2007).

A total of 212 individuals were interviewed at these locations. The majority of participants were male, 18 years and older, and were either Caucasian or Asian and Pacific Islander. Data were collected on fishing location preferences, fishing frequency, consumption amounts, species preferences, cooking methods, and whether family members would also consume the catch. A visual representation of typical serving sizes was used to estimate portion sizes, and respondents estimated how many times they had eaten self-caught fish during the previous 30 days. Respondent demographic data were also collected.

Approximately 98 percent of anglers interviewed sought finfish rather than other types of aquatic organisms, and over 50 percent reported that they fished only from the location where they were interviewed. This indicates that a majority of anglers consistently fish from a single unique location (Parametrix 2003). Reported fishing frequencies were 6, 16, and 20 days/year at Lake Union, Lake Sammamish, and Lake Washington, respectively.

Consumption rates were estimated using responses on fish meal frequency and meal size. Mean fishing frequency in the King County lakes was 19 days/year, with a 95th percentile value of 74 days/year. Most anglers (94 percent) reported eating fillets without skin. Meal sizes were generally in the 6- to 8-ounce range (64 percent of anglers), with over 30 percent of anglers reporting an average meal size greater than 8 ounces (Parametrix 2003). The mean reported family size (including respondents) sharing the anglers' catch was 4.1.

The mean recreational freshwater fish consumption rates were 10 g/day for all respondents and 7 g/day for the children of survey respondents (Table 2). Mayfield et al. (2007) also reported differences in intake according to ethnicity. Mean freshwater fish intake rates were 26, 13, 8, and 6 g/day for African American, Asian and Pacific Islander, Caucasian, and Hispanic/Latino respondents, respectively.

Table 2. Fish Consumption Rates (g/day) for Freshwater Recreational Anglers in King County, WA

Location	Sample Size	Mean	SD	SE	50th Percentile	90th Percentile	95th Percentile
Lake Washington	93	11	26	2.7	5.7	NA	30
Lake Sammamish	35	9.1	18	3.1	0	NA	57
King County Lakes (all respondents)	128	10	24	2.2	0	23	42
King County Lakes (children of respondents)	81	7.2	20	2.2	0	17	29

Source: Parametrix (2003); Mayfield et al. (2007)

SD = Standard deviation; SE = Standard error

Limitations to this study included:

- A substantial number of individuals refused to have their catch weighed, and/or refused to be interviewed;
- Surveys were very brief (5 to 10 minutes); and
- The study does not address how language issues were dealt with in administering the survey to non-English speakers.

Summary

Results of recreational fish consumption surveys are presented in Tables A-1 and A-2. It should be noted that the fish consumption surveys identified in these appendices were conducted for a variety of purposes, used different methods of estimating fish and shellfish consumption, and had different target populations. The fish consumption rates listed in Tables A-1 and A-2 typically include both finfish and shellfish, although several studies reported separate finfish and shellfish consumption rates. Some studies estimated consumption rates for fish consumers only, while others included nonconsumers. These differences are noted in the tables, as appropriate.

A summary of the recreational fish consumption rates listed in Appendix A is provided in Table 3 below.

Table 3. Summary of Recreational Fish Consumption Rates

	Mean (g/day)	Median (g/day)	Upper (90th to 95th) Percentile (g/day)
Marine Fish/Shellfish Consumption Rates			
Washington State	11 – 53	1.0 – 21	13 – 246
Other West Coast Studies	1.2 – 50	2.9 – 37	6.8 – 225
Studies in Other Regions	12 – 48	9.1	5.1 – 68
Freshwater Fish Consumption Rates			
Washington State	6.0 – 22	NA	42 – 67
Other West Coast Studies	27	3.8 – 24	5.7 – 127
Studies in Other Regions	4.9 – 55	1.1 – 24	18 – 235

Mean estimated marine/estuarine consumption rates for recreational fishers ranged from 1.2 to 53 g/day, with similar results reported in Washington State and other regions. Upper percentile (90 to 95th percentile) marine fish/shellfish consumption rates ranged from 5.1 to 246 g/day for all studies reviewed; consumption rates were somewhat higher in studies conducted along the West Coast (Washington, Oregon, and California) than in studies conducted in other regions.

For freshwater fish, mean consumption rates ranged from 4.9 to 55 g/day. Results from Washington and other West Coast states were somewhat lower (6.0 to 27 g/day), however studies along the West Coast were very limited (three studies in Washington, and one each in Oregon and California). The fish consumption dataset from studies of freshwater fish consumption in other regions is more robust (16 studies), and the upper end of the range of mean consumption rates is very similar to that estimated for marine fish/shellfish (50 to 60 g/day).

Upper percentile freshwater fish consumption rates ranged from 5.7 to 235 g/day in all studies reviewed, and from 42 to 67 g/day in studies conducted in Washington State. The highest reported freshwater fish consumption rates were from a study conducted by Burger (2002b); this study, which included both freshwater and marine sport-caught fish and was based on a one-year recall survey at a sportsman's event in South Carolina, likely overestimates freshwater fish consumption rates by recreational anglers. The next highest upper percentile result is 136 g/day, which is based on a 3-month recall survey of minority recreational anglers in Indiana. With the exception of the Burger (2002b) study, results of freshwater fish consumption rates in other regions are similar to those observed along the West Coast.

In conclusion, fish consumption surveys reviewed during preparation of this issue paper can be used to provide a conservative estimate of mean and upper (90th to 95th) percentile marine/estuarine and freshwater fish consumption rates for recreational fishers in Washington State, as follows:

- Mean consumption rates for both freshwater and marine/estuarine fish and shellfish are in the range of 20 to 60 g/day;

- Upper percentile consumption rates are in the range of 200 to 250 g/day for marine/estuarine fish and shellfish, and in the range of 100 to 150 g/day for freshwater fish.

Based on these studies, recreational fish consumption rates developed in Washington State and elsewhere are consistent with the preliminary default fish consumption rate of 157 to 267 g/day presented in the draft TSD. In other words, a default fish consumption rate in this range would be protective of marine and freshwater recreational fishers.

References

ADEM (Alabama Department of Environmental Management). 1994. Estimation of daily per capita freshwater fish consumption of Alabama anglers. As cited in USEPA 2011.

Adolfson Associates. 1996. Technical Memorandum on the Results of the 1995 Fish Consumption and Recreational Use Surveys – Amendment No. 1. From Chee Choy, City of Portland, to Jean Ochsner, Adolfson Associates. April 19, 1996.

Alcoa (Aluminum Company of America). 1998. Draft report for the finfish/shellfish consumption study Alcoa (Point Comfort)/Lavaca Bay Superfund Site, Volume B7b: Bay System Investigation Phase 2. Point Comfort, TX. As cited in USEPA 2011.

Balcom, N., Capacchione, C., and Hirsch, D.W. 1999. Quantification of seafood consumption rates for Connecticut. Prepared for the Connecticut Department of Environmental Protection, Office of Long Island Sound Programs. As cited in USEPA 2011.

Benson, S., Crocker, C., Erjavec, J., Jensen, R.R., Nyberg, C.M., Wixo, C.Y., and Zola, J.M. 2001. Fish consumption survey: Minnesota and North Dakota. Prepared for the U.S. Department of Energy by the Energy and Environmental Research Center, University of North Dakota. As cited in USEPA 2011.

Burger, J., 2002a. Consumption patterns and why people fish. *Environ Res* 90:125-135. As cited in USEPA 2011.

Burger, J. 2002b. Daily consumption of wild fish and game: exposures of high end recreationists. *Environ Health Res* 12:343-354. As cited in USEPA 2011.

Burger, J. 2000. Sex differences in meal patterns: Role of self-caught fish and wild game in meat and fish diets. *Environ Res* 83:140-149. As cited in USEPA 2011.

Burger, J., Stephens, W.L., Boring, C.S., Kuklinski, M., Gibbons, J.W., and Gochfeld, M. 1999. Factors in exposure assessment: Ethnic and socioeconomic differences in fishing and consumption of fish caught along the Savannah River. *Risk Anal* 19(3):427-438. As cited in USEPA 2011.

Burger, J., Sanchez, J., and Gochfeld, M. 1998. Fishing, consumption, and risk perception in fisherfolk along an East Coast estuary. *Environ Res* 77:25-35. As cited in USEPA 2011.

CalEPA. 2001. Chemicals in Fish: Consumption of Fish and Shellfish in California and the United States. Final Report. Pesticide and Environmental Toxicology Section, Office of Environmental Health Hazard Assessment, California Environmental Protection Agency. October 2001.

Campbell, K., Dickey, R., and Sexton, R. 2002. Fishing along the Clinch River arm of Watts Reservoir adjacent to the Oak Ridge Reservation, Tennessee: Behavior, knowledge, and risk perception. *Sci Total Environ* 299:145-161. As cited in USEPA 2011.

ChemRisk. 1992. Consumption of freshwater fish by Maine anglers, a Technical Report. Revised July 24, 1994. As cited in USEPA 2011.

Chiang, A. 1998. A seafood consumption survey of the Laotian community of West Contra Costa County, CA. Asian Pacific Environmental Network, Oakland, CA. As cited in USEPA 2011.

Connelly, N.A., Knuth, B.A., and Brown, T.L. 1996. Sportfish consumption patterns of Lake Ontario anglers and the relationship to health advisories. *N Am J Fish Manage* 16:90-101. As cited in USEPA 2011.

Connelly, N.A., Knuth, B.A., and Bisogni, C.A. 1992. Effects of the health advisory and advisory changes on fishing habits and fish consumption in New York sport fisheries. Human Dimension Research Unit, Department of Natural Resources, New York State College of Agriculture and Life Sciences, Cornell University. As cited in USEPA 2011.

Fiore, B.J., Anderson, H.A., Hanrahan, L.P., Olsen, L.J., and Sonzogni, W.C. 1989. Sport fish consumption and body burden levels of chlorinated hydrocarbons: A study of Wisconsin anglers. *Arch Environ Health* 44:82-88. As cited in USEPA 2011.

KCA Research Division. 1994. Fish consumption of Delaware recreational fishermen and their households. Prepared for the State of Delaware, Department of Natural Resources and Environmental Control, in support of the Delaware Estuary Program. As cited in USEPA 2011.

Keill, L. and Kissinger, L. 1997. Analysis and Selection of Fish Consumption Rates for Washington State Risk Assessments and Risk-Based Standards. Washington State Department of Ecology. Review draft. October 1997. As cited in CalEPA 2001.

Kissinger, L. 2010. Email from Lon Kissinger, U.S. Environmental Protection Agency, to Sharon Quiring and Laura Scheffler, URS Corporation, Re: Sport Fisher Seafood Consumption Rates. March 10, 2010.

Landolt, M.L., Hafer, F.R., Nevissi, A., Van Belle, G., Van Ness, K., and Rockwell, C. 1985. Potential Toxicant Exposure Among Consumers of Recreationally Caught Fish from Urban Embayments of Puget Sound. NOAA Technical Memorandum NOS OMA 23. November 1985.

Landolt, M.L., Kalman, D.L., Nevissi, A., Van Belle, G., Van Ness, K., and Hafer, F.R. 1987. Potential toxicant exposure among consumers of recreationally caught fish from urban embayments of Puget Sound. NOAA Technical Memorandum NOS OMA 33. As cited in Tetra Tech 1988.

Mayfield, D.B., Robinson, S., and Simmonds, J. 2007. Survey of fish consumption patterns of King County (Washington) recreational anglers. *Journal of Exposure Analysis and Environmental Epidemiology*, 17:604-612.

McCallum, M. 1985. Recreational and Subsistence Catch and Consumption of Seafood from Three Urban Industrial Bays of Puget Sound: Port Gardner, Elliott Bay and Sinclair Inlet. Washington State Division of Health, Epidemiology Section. January 1985.

Moya, J. 2004. Overview of Fish Consumption Rates in the United States. *Human and Ecological Risk Assessment*, 10:1195-1211.

Moya, J. and Phillips, L. 2001. Analysis of consumption of home-produced foods. *J Expo Anal Environ Epidemiol* 11(5):398-406. As cited in USEPA 2011.

NMFS (National Marine Fisheries Service). 1993. Data tapes for the 1993 NMFS provided to U.S. EPA, National Center for Environmental Assessments, Washington, DC. As cited in USEPA 2011.

ODEQ (Oregon Department of Environmental Quality). 2008. Human Health Focus Group Report, Oregon Fish and Shellfish Consumption Rate Project. Oregon Department of Environmental Quality, Water Quality Division. June 2008.

Parametrix. 2003. Results of a Human Use Survey for Shoreline Areas of Lake Union, Lake Washington, and Lake Sammamish. Sammamish-Washington Analysis and Modeling Program (SWAMP). Prepared for King County Department of Natural Resources. September 2003.

Pierce, D., Noviello, D.T., and Rogers, S.H. 1981. Commencement Bay Seafood Consumption Study. Preliminary Report. Tacoma-Pierce County Health Department, Tacoma, Washington. December 1981.

Price, P., Su, S., and Gray, M. 1994. The effects of sampling bias on estimates of angler consumption rates in creel surveys. *J Expos Anal Environ Epidemiol* 4:355-371. As cited in USEPA 2011.

Puffer, H.W., Azen, S.P., Duda, M.J., and Young, D.R. 1981. Consumption rates of potentially hazardous marine fish caught in the metropolitan Los Angeles area. As cited in USEPA 2011.

SFEI (San Francisco Estuary Institute). 2000. Technical report: San Francisco Bay Seafood Consumption Report. California Department of Health Services. San Francisco, CA. As cited in USEPA 2011.

Shilling, F., White, A., Lippert, L., and Lubell, M. 2010. Contaminated fish consumption in California's Central Valley Delta. *Environ Res* 110(4):334-344. As cited in USEPA 2011.

Simmonds, J., Munger, S., Strand, J., Homan, C., Robinson, S., Toll, J., Wisdom, C., Seidel, P., Greer, H., and Shroy, J. Results of a survey on seafood collection and consumption from the shores of the Duwamish River and Elliott Bay. Paper presented at the Puget Sound Research Conference, Seattle, WA 1998: pp. 194-200.

SMBRP (Santa Monica Bay Restoration Project). 1994. Santa Monica Bay Seafood Consumption Study. Final Report. Prepared by Southern California Coastal Water Research Project and MBC Applied Environmental Sciences. June 1994.

Tetra Tech. 1988. Health Risk Assessment of Chemical Contaminants in Puget Sound Seafood. Prepared by Tetra Tech, Inc. for U.S. Environmental Protection Agency, Region 10. September 1988.

USEPA (U.S. Environmental Protection Agency). 2011. Exposure Factors Handbook: 2011 Edition. Chapter 10, Intake of Fish & Shellfish. U.S. Environmental Protection Agency, National Center for Environmental Assessment. EPA/600/R-09/052F. September 2011.

WDOH (Washington State Department of Health). 1997. Consumption Patterns of Anglers Who Frequently Fish Lake Roosevelt. Washington State Department of Health, Office of Environmental Health Assessments. September 1997.

WDOH. 2001. Lake Whatcom Residential and Angler Fish Consumption Survey. Data Report. Washington State Department of Health. April 2001.

West, P.C., Fly, J.M., Marans, R., Larkin, F., and Rosenblatt, D. 1993. 1991-1992 Michigan sport anglers fish consumption study. Prepared by the University of Michigan, School of Natural Resources, for the Michigan Department of Natural Resources. Technical Report No. 6. May 1993. As cited in USEPA 2011.

West, P.C., Fly, M.J., Marans, R., and Larkin, F. 1989. Michigan sport anglers fish consumption survey. A report to the Michigan Toxic Substance Control Commission. As cited in USEPA 2011.

Williams, R., O'Leary, J., Sheaffer, A., and Mason, D. 2000. An examination of fish consumption by Indiana recreational anglers: An onsite survey. Department of Forestry and Natural Resources, Purdue University. As cited in USEPA 2011.

Williams, R., O'Leary, J., Sheaffer, A., and Mason, D. 1999. Consumption of Indiana sport caught fish: Mail survey of resident license holders. Department of Forestry and Natural Resources, Purdue University. As cited in USEPA 2011.

Appendices

Appendix A. Fish Consumption Rates for Recreational Fishers

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Table A-1. Recreational Marine Fish/Shellfish Consumption Rates

Source	Purpose of Study	Location	Population/Subgroup	Type of Fish/Shellfish	Sample Size	Median (g/day)	Mean (g/day)	Upper Percentile (g/day)	Notes
Washington State Studies									
Survey of Fish Consumption Patterns of King County Recreational Anglers (Mayfield et al. 2007)	To support environmental analyses of proposed capital improvement projects.	Elliott Bay, Duwamish River, and north King County	Households with members who fish; consumers and nonconsumers	Marine fish; species include sea perch, salmon, rockfish, sole, herring, flounder.	494	21	53	181 (95th)	Creel survey/personal interviews at public parks and boat launches; consumption rate calculated using information on fishing frequency, mean weight of catch at each location (Elliott Bay, Duwamish River, North King County), a cleaning factor, and number of individuals consuming the catch.
				Shellfish, primarily crabs and shrimp.	96	11	25	119 (95th)	
1983-1984 Puget Sound Survey (Landolt et al. 1985, 1987); reanalysis by Tetra Tech 1988	To identify most commonly consumed species, demographic characteristics of the fishing population, and patterns of consumption.	Commencement Bay, Elliott Bay, Sinclair Inlet, Edmonds	Recreational fishers and their households	Fish and shellfish: market squid, Chinook salmon, Coho salmon, unidentified salmon, Pacific hake, Pacific cod, pile perch	4,437		11		Creel survey; reported value is a geometric mean. Based on weight of catch, number of consumers in household, number of days since fish caught at the same site were last eaten, and a cleaning factor.
			Recreational fishers (shore anglers) and their households			26.1		246 (95th)	Data reanalyzed by Tetra Tech (1988).
			Recreational fishers (boat anglers) and their households			12.3		95.1 (95th)	Data reanalyzed by Tetra Tech (1988).
Seafood Catch and Consumption in Urban Bays of Puget Sound (McCallum 1985); reanalysis by Tetra Tech 1988	To assess the number of people with repeated, long-term exposure to fish and shellfish from contaminated urban bays.	Port Gardner, Elliott Bay, Sinclair Inlet	Shore-based recreational fishers; consumers and nonconsumers	Sablefish, striped perch, pile perch, Pacific cod, Dungeness crab, red rock crab, butter clams, littleneck clams; salmon and squid excluded	702	1.9	NA	24.3 (95th)	Creel survey. Intake rates calculated by Tetra Tech (1988); methodology and assumptions not specified.
Commencement Bay Seafood Consumption Study (Pierce et al. 1981); reanalysis by Price et al. (1994)	To examine seafood consumption patterns and demographics of sport fishermen in Commencement Bay.	Commencement Bay	Shore and boat anglers; surveyed fishers only	Common species included Pacific hake, walleye pollock; salmon excluded	508	19	NA	155 (90th)	Creel survey. Intake rates calculated by Price et al. (1994).
			Shore and boat anglers; total recreational fisher population			1.0	NA	13 (90th)	Creel survey; results weighted inversely proportional to sampling probability (Price reanalysis).
Other West Coast Studies									
San Francisco Bay Seafood Consumption Report (SFEI 2000)	To conduct a comprehensive study of San Francisco Bay anglers.	San Francisco Bay, California	Recreational fishers; consumers only	Marine finfish, including striped bass, halibut, jacksmelt, sturgeon, and white croaker.	465	16	23	80 (95th)	Questionnaire; consumers are respondents who reported eating fish in the previous 4 weeks.
Seafood Consumption Survey of the Laotian Community of West Contra Costa County, CA (Chiang 1998)	To obtain data on fishing and fish consumption activities of the Laotian community.	San Francisco Bay, California	Members of the Laotian community; consumers only	Marine finfish, including catfish and striped bass.	199	9.1	21.4	85.1 (95th)	Questionnaire; the proportion of reported fish consumption that was self-caught is not clear.
Seafood Consumption Habits of Recreational Anglers in Santa Monica Bay (SMBRP 1994)	To study the seafood consumption habits of recreational anglers in Santa Monica Bay.	Santa Monica Bay, California	Recreational fishers; consumers only	Marine finfish, including barred sand bass, Pacific barracuda, kelp bass, rockfish species, Pacific bonito, and California halibut.	555	21.4	49.6	161 (95th)	Field survey: 28-day recall.

Table A-1. Recreational Marine Fish/Shellfish Consumption Rates

Source	Purpose of Study	Location	Population/Subgroup	Type of Fish/Shellfish	Sample Size	Median (g/day)	Mean (g/day)	Upper Percentile (g/day)	Notes
National Marine Fisheries Service (NMFS 1993); USEPA analysis	To estimate the size of the recreational marine finfish catch by location, species, and fishing mode, and to estimate the total number of participants in marine recreational finfishing and the total number of fishing trips.	Oregon and California (Pacific Region)	Adult recreational fishers; consumers and nonconsumers	Marine finfish	1,827	NA	2.0	6.8 (95th)	Intake rates calculated by USEPA. Consumption rate calculated from field intercept (creel) survey; estimated yearly amount of fish caught (based on weight of catch) was adjusted by 0.5 assumed edible fraction and 2.5 intended consumers per catch. No data for Washington.
		Oregon			284	NA	2.2	8.9 (95th)	
Intake Rates of Potentially Hazardous Marine Fish Caught in the Metropolitan Los Angeles Area (Puffer et al. 1981); reanalysis by Price et al. (1994)	To evaluate intake rates of potentially hazardous marine fish/shellfish by local, non-professional fishermen.	Los Angeles, California	Recreational fishers; surveyed fishers only	Finfish and shellfish	1,059	37	NA	225 (90th)	Creel survey.
			Total recreational fisher population			2.9	NA	35 (90th)	Creel survey; results weighted inversely proportional to sampling probability (Price reanalysis).
Studies in Other Regions									
Consumption Patterns and Why People Eat Fish (Burger 2002a)	To evaluate fishing behavior and consumption patterns	Newark Bay and New York-New Jersey Harbor estuary	Recreational fishers; consumers only	Marine/estuarine fish	111	NA	22.2	NA	Interviews with recreational fishers at various fishing locations. Fish consumption rate is based on number of reported fish meals per month, an average portion size (8 ounces), and the number of months per year that self-caught fish are consumed. Crab consumption rate based on reported number of crabs eaten per month, an estimated average edible portion weight (70 grams), and the number of months per year that self-caught crabs are eaten.
				Marine/estuarine crabs	110	NA	15.8	NA	
Analysis of Consumption of Home-Produced Foods (Moya and Phillips 2001)	To estimate consumption of self-caught fish.	United States	Households with members who fish	Freshwater and saltwater fish	220	NA	11.8	NA	Based on USDA Nationwide Food Consumption Survey. Intake calculated for fish taken into the household; no correction applied for edible fraction.
Fishing, Consumption, and Risk Perception in Fisherfolk along an East Coast Estuary (Burger et al. 1998)	To examine fishing behavior, consumption patterns, and risk perceptions of people fishing/ crabbing in Barnegat Bay.	Barnegat Bay, New Jersey	Adult recreational fishers; consumers and nonconsumers	Marine finfish and shellfish, including bluefish, fluke/summer flounder, weakfish, and crabs.	515	NA	48.3	NA	Field survey: one-month recall.
Finfish/Shellfish Consumption Study - Alcoa (Point Comfort)/ Lavaca Bay Superfund Site (Draft) (Alcoa 1998)	To evaluate the quantity and species of finfish and shellfish consumed by individuals who fish at Lavaca Bay.	Lavaca Bay, Texas	Adult male recreational fishers; consumers and nonconsumers	Saltwater fish , including red drum, speckled sea trout, and flounder.	1,979	NA	24.8	68.1 (90th)	Telephone and mailed surveys: 28-day recall. Consumption rates for women, small children, and youths also reported.
				Shellfish, including oysters, blue crab, and shrimp.		NA	1.2	5.1 (95th)	
Fish Consumption of Delaware Recreational Fishermen and Their Households (KCA Research Division 1994)	To obtain information on the number of fishing trips, and the amount of fish caught, kept, and eaten by fishing households.	Delaware	Marine recreational fishers and their households	Marine finfish; some freshwater fish and crabs	867	NA	17.5	NA	Telephone survey: 30-day recall.

Table A-2. Recreational Freshwater Fish Consumption Rates

Source	Purpose of Study	Location	Population/Subgroup	Type of Fish/Shellfish	Sample Size	Median (g/day)	Mean (g/day)	Upper Percentile (g/day)	Notes
Washington State Studies									
Survey of Fish Consumption Patterns of King County Recreational Anglers (Parametrix 2003; Mayfield et al. 2007)	To support environmental analyses of proposed capital improvement projects.	King County: Lake Washington, Lake Sammamish	Households with members who fish; consumers and nonconsumers	Freshwater fish including yellow perch, trout, salmon, bass, bullhead	128	0	10.3	41.7 (95th)	Personal interviews at public parks and boat launches; consumption rate calculated based on reported fish meals in the previous month and meal size estimates based on visual aid.
Consumption Patterns of Anglers Who Frequently Fish Lake Roosevelt (WDOH 1997)	To determine the consumption patterns of anglers who repeatedly fish this lake.	Lake Roosevelt	Recreational fishers; consumers only	Rainbow trout, walleye, kokanee, bass	348	NA	22	67 (95th)	Interviews at fishing locations; consumption rate based on number of fish meals reported per week and assumed 8-ounce meal size (L. Kissinger, personal comm., 6/6/2012)
Lake Whatcom Residential and Angler Fish Consumption Survey (WDOH 2001)	To gather exposure information for use in assessing the human health implications associated with the consumption of mercury-contaminated fish from Lake Whatcom.	Lake Whatcom	Boat anglers, shore anglers, and residents living on or near the lake	Smallmouth bass, cutthroat trout, yellow perch, kokanee	220	NA	6.0	NA	Personal interview, 4-week recall. Elevated mercury levels and media reports may have affected fish consumption. Calculated based on number of fish per meal, how many months fish were consumed, and average fillet weight for each species (as presented in ODEQ 2008).
Other West Coast Studies									
Fish Consumption and Recreational Use Surveys - Columbia Slough (Adolfson Associates 1996)	To provide information on the fishing habits of people angling in the Columbia Slough.	Columbia Slough, Portland, Oregon	Columbia Slough fishermen, consumers only	Resident finfish and shellfish, fresh and estuarine	84	24	NA	36 (75th)	Creel survey; fish consumption rates based on fish weights, assumed edible fraction, and household size.
		Sauvie Island, Portland, Oregon	Sauvie Island fishermen, consumers only		35	3.8	NA	5.7 (75th)	
Contaminated Fish Consumption in California's Central Valley (Shilling et al. 2010, as cited in USEPA 2011)	To assess consumption of contaminated fish in a region with high subsistence fishing rates.	Sacramento-San Joaquin River Delta, California	Recreational and subsistence anglers	Striped bass, salmon, shad, catfish	373	19.7	27.4	127 (95th)	Interviews at fishing locations; 30-day recall.
Studies in Other Regions									
Daily Consumption of Wild Fish and Game (Burger 2002b, as cited in USEPA 2012)	To determine consumption patterns for wild-caught fish and game by high-end recreationists.	South Carolina	High-end recreational anglers	Sport-caught fish, freshwater and marine	458	17.6	50.2	216 (95th)	Personal interviews at sportsman event. One-year dietary recall.
			High-end recreational anglers, male		308	21.3	55.2	235 (95th)	
			High-end recreational anglers, female		149	11.6	39.1	172 (95th)	
Fishing Along the Clinch River Arm of Watts Bar Reservoir (Campbell et al. 2002, as cited in USEPA 2011)	To examine consumption habits of anglers fishing along the Clinch River near the USDOE Oak Ridge Reservation.	Clinch River, Tennessee	Recreational anglers, consumers only	Locally-caught freshwater fish, including crappie, striped bass, white bass, sauger, and catfish	202	NA	20.3	NA	Personal interviews at fishing locations. Annual reported fish consumption divided by 365.
			Recreational anglers, consumers who eat fish from the study area		77	NA	37.5	NA	
Analysis of Consumption of Home-Produced Foods (Moya and Phillips 2001, as cited in USEPA 2011)	Analysis of data from the household component of the USDA's 1987-1988 National Food Consumption Survey.	National	Fishers, consumers only	Self-caught fish, freshwater and saltwater	220 households	NA	6.0	NA	

Table A-2. Recreational Freshwater Fish Consumption Rates

Source	Purpose of Study	Location	Population/Subgroup	Type of Fish/Shellfish	Sample Size	Median (g/day)	Mean (g/day)	Upper Percentile (g/day)	Notes
Minnesota and North Dakota Fish Consumption Survey (Benson et al. 2001, as cited in USEPA 2011)	To evaluate fish consumption by the general population, licensed anglers, and members of Native American tribes.	Minnesota	Licensed anglers	Sport-caught fish, freshwater and marine	2,020	3.9	NA	30.4 (95th)	Mail survey.
				Fish from all sources	2,020	13.2	NA	64.5 (95th)	
		North Dakota	Licensed anglers	Sport-caught fish, freshwater and marine	1,101	4.5	NA	30.8 (95th)	
				Fish from all sources	1,101	14.0	NA	76.2 (95th)	
Examination of Fish Consumption by Indiana Recreational Anglers (Williams et al. 2000, as cited in USEPA 2011)	To evaluate fish consumption by Indiana recreational anglers as a function of ethnicity and income.	Indiana	Caucasian recreational anglers	Sport-caught fish	177	7.6	20	113 (95th)	On-site surveys at fishing locations; 3-month recall period.
			Minority recreational anglers	Sport-caught fish	143	7.6	27.2	136 (95th)	
Role of Self-Caught Fish and Wild Game in Meat and Fish Diets (Burger 2000, as cited in USEPA 2011)	To evaluate sex differences in consumption patterns of self-caught fish and wild game in a meat and fish diet.	South Carolina	High-end recreational anglers, male	Sport-caught fish, freshwater and marine	457	NA	27.5	NA	Personal interviews at sportsman event. Consumption rates calculated based on reported monthly meal frequency and meal size.
			High-end recreational anglers, female	Sport-caught fish, freshwater and marine		NA	11.6	NA	
Consumption of Indiana Sport-Caught Fish (Williams et al. 1999, as cited in USEPA 2011)	To estimate consumption of sport-caught fish by licensed Indiana anglers.	Indiana	Recreational anglers, consumers only	Sport-caught fish	1,045	9.5	19.8	60.5 (95th)	Mail survey, 3-month recall period.
Ethnic and Socioeconomic Differences in Fishing and Consumption of Fish Caught Along the Savannah River (Burger et al. 1999, as cited in USEPA 2011)	To examine the differences in fishing rates and fish consumption of people fishing along the Savannah River as a function of demographics.	Savannah River, Georgia	Recreational anglers, consumers and nonconsumers	Sport-caught fish	252	NA	48.7	NA	Personal interviews at fishing locations. Consumption rate calculated based on reported servings per month and meal size.
Quantification of Seafood Consumption Rates for Connecticut (Balcom et al. 1999, as cited in USEPA 2011)	To evaluate seafood consumption rates for Connecticut residents	Connecticut	Sport-fishing households, consumers and nonconsumers	Sport-caught fish, freshwater and saltwater	502	NA	51.1	NA	Mail survey with portion size models, and 10-day diary. Consumption rate reflects cooked weights.
			Sport-fishing households, consumers only		487	NA	52.7	NA	
Sportfish Consumption Patterns of Lake Ontario Anglers and the Relationship to Health Advisories (Connelly et al. 1996, as cited in USEPA 2011)	To provide accurate estimates of fish consumption among Lake Ontario anglers, and to evaluate the effect of health advisory recommendations.	Lake Ontario	Licensed anglers in six counties bordering Lake Ontario	Sport-caught finfish	366	2.2	4.9	17.9 (95th)	Consumption rates based on self-recorded diary information plus follow-up telephone calls. Health advisories in effect, which may have impacted consumption rates.
				Fish from all sources	366	14.1	17.9	42.3 (95th)	
Estimation of Daily Per Capita Freshwater Fish Consumption of Alabama Anglers (ADEM 1994, as cited in USEPA 2011)	To estimate fish consumption for sport-fishing Alabama anglers.	Alabama	Recreational anglers, consumers only	Sport-caught fish	563	NA	43.1	NA	Personal interviews at fishing locations; consumption based on "harvest method" which used actual harvest of fish and dressing method.
					1,313	NA	44.8	NA	Personal interviews at fishing locations; consumption rate estimates based on "4-ounce serving method."

Table A-2. Recreational Freshwater Fish Consumption Rates

Source	Purpose of Study	Location	Population/Subgroup	Type of Fish/Shellfish	Sample Size	Median (g/day)	Mean (g/day)	Upper Percentile (g/day)	Notes
Follow-up to Michigan Sport Anglers Fish Consumption Survey (West et al. 1993, as cited in USEPA 2011)	To provide short-term recall data of fish consumption by recreational anglers over a full year.	Michigan	Recreational anglers, consumers only	Sport-caught fish	2,475	NA	16.7	77.9 (95th)	Mail survey based on 7-day recall; 95th percentile (calculated by EPA) is believed to considerably overestimate long-term percentiles.
Effects of Health Advisory and Advisory Changes on Fishing Habits and Fish Consumption in New York Sports Fishers (Connelly et al. 1992, as cited in USEPA 2011)	To assess the awareness and knowledge of New York anglers about fishing advisories and contaminants, and their fishing and fish consuming behaviors.	New York	Recreational anglers, consumers only	Sport-caught fish	1,030	NA	27.4	NA	Mail survey based on one-year recall. Assumes 8-ounce meal size. Fish advisories in effect.
Consumption of Freshwater Fish by Maine Anglers (Chemrisk 1992, as cited in USEPA 2011)	To characterize the rates of freshwater fish consumption among Maine residents.	Maine - flowing and standing water bodies	Licensed anglers and their households, consumers only	Salmon, white perch, brook trout	1,053	2.0	6.4	26.0 (95th)	Mail survey based on one-year recall; calculated from estimated weight of fish caught, edible fraction, and number of intended consumers.
			Licensed anglers and their households, consumer and nonconsumers		1,369	1.1	5.0	21.0 (95th)	
Michigan Sport Anglers Fish Consumption Survey (West et al. 1989, as cited in USEPA 2011)	To evaluate sport angler fish consumption by Michigan residents with fishing licenses.	Michigan	Licensed anglers and their households	Sport-caught fish	738	10.9	14.4	38.7 (95th)	Survey based on one-year recall (usual intake). Serving size assumed to be 8 ounces. Consumption rates calculated by EPA.
						NA	14.0	NA	Survey based on short-term (7-day) recall; serving size based on comparison to pictures of an 8-ounce portion. Consumption rate calculated by EPA.
				Fish from all sources	738	24.2	27.7	58.1 (95th)	Survey based on one-year recall (usual intake). Serving size assumed to be 8 ounces. Consumption rates calculated by EPA.
Sport Fish Consumption and Body Burden Levels of Chlorinated Hydrocarbons (Fiore et al. 1989, as cited in USEPA 2011)	To assess sociodemographic factors and sport-fishing habits of anglers, to evaluate anglers' comprehension of fish advisories, and to examine the relationship between body burden levels of PCBs/DDE with consumption of sport-caught fish.	Wisconsin	Licensed anglers, consumers and nonconsumers	Sport-caught fish	801	6.2	11.2	37.2 (95th)	Based on one-year recall survey. Intakes calculated by EPA, assuming an 8-ounce meal size.
			Licensed anglers, consumers only	Sport-caught fish	729	NA	12.3	37.3 (95th)	

Technical Issue Paper

Health Benefits and Risks of Consuming Fish and Shellfish

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Health Benefits and Risks of Consuming Fish and Shellfish

July 20, 2012

Toxics Cleanup Program
Washington State Department of Ecology
Olympia, Washington

Table of Contents

Page

List of Figures and Tables

Figures

Tables

Acronyms & Abbreviations

AD	Alzheimer's disease
AHA	American Heart Association
ALA	alpha-linoleic acid
AMD	age-related macular degeneration
BMD	bone mineral density
CHD	coronary heart disease
CKD	chronic kidney disease
CVD	cardiovascular disease
DDT	dichloro diphenyl trichloroethane
DEHP	di-ethylhexyl phthalate
DHA	docosahexaenoic acid
DPA	docosapentaenoic acid
Ecology	Washington State Department of Ecology
EPA	eicosapentaenoic acid
FAO	Food and Agriculture Organization of the United Nations
IOM	Institutes of Medicine
LCP	long-chain polyunsaturated fatty acid
n-3 PFA or PUFA	Omega-3 polyunsaturated fatty acid
NSAID	nonsteroidal anti-inflammatory drug
PAH	polycyclic aromatic hydrocarbon
PBT	Persistent Bioaccumulative Toxics
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PTMI	provisional tolerable monthly intake
TCDD	2,3,7,8-tetrachlorodiobenzo-*p*-dioxin
TERA	Toxicology Excellence for Risk Assessment
TEQ	toxic equivalent
TSD	Fish Consumption Rates Technical Support Document
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
WDOH	Washington Department of Health
WHO	World Health Organization

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Introduction

Multiple comments received by the Washington State Department of Ecology (Ecology) on the draft *Fish Consumption Rates Technical Support Document* (TSD) (Ecology 2011) asked why the TSD did not include a discussion of the relative benefits of consuming fish and shellfish or address the potential public health risks if people consume less fish. Others also suggested that the TSD should include some information regarding the contaminant concentrations in Washington State fish and shellfish.

Fish is a good source of protein and, unlike fatty meat products, it is not high in saturated fat. Fish is also a good source of omega-3 fatty acids. Omega-3 fatty acids benefit the heart of healthy people, and those at high risk of, or who have, cardiovascular disease. While some types of fish may contain high levels of mercury, PCBs (polychlorinated biphenyls), dioxins and other environmental contaminants, the American Heart Association recommends eating fish (particularly fatty fish) at least two times (two servings) per week.[1]

According to the Food and Agriculture Organization of the United Nations, worldwide fish consumption reached an all-time high in 2011, with an average consumption of 17 kg per person.[2]

This Technical Issue Paper summarizes the known health benefits of consuming seafood and the risks associated with a set of common contaminants in Washington State seafood, and compares these health benefits and risks. It is a targeted examination of the issues raised by review comments received on the draft TSD, and was prepared within a limited time frame. Therefore, it may not include all available information on this subject.

Throughout this paper, use of the word "seafood" denotes fish and shellfish. A review of Washington State seafood contaminant concentrations and potential human exposures is presented in another Technical Issue Paper (*Chemical Contaminants in Dietary Protein Sources)*.

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[1] http://www.heart.org/HEARTORG/GettingHealthy/NutritionCenter/HealthyDietGoals/Fish-and-Omega-3-Fatty-Acids_UCM_303248_Article.jsp

[2] http://www.fao.org/news/story/en/item/50260/icode/

Analysis

I. Benefits associated with eating fish and shellfish

The most recent guidance from the American Heart Association (AHA) recommends eating fish, particularly fatty fish (e.g., salmon, anchovy, herring), at least two times per week (AHA 2012). The following sections describe why fish and shellfish are recommended as an important part of the diet.

A. Seafood as a high quality protein source

Fish is a good source of high quality protein that is not high in saturated fat (USDA 2010; AHA 2012). Saturated fats have a chemical makeup in which the carbon atoms are saturated with hydrogen atoms and are typically solid at room temperature. Eating foods that contain saturated fats raises the level of cholesterol in the blood. High levels of blood cholesterol increase the risk of heart disease and stroke. Many foods high in saturated fats are also high in cholesterol. Shellfish contain more cholesterol than most types of fish, but are very low in saturated fat (AHA 2012). A significant proportion of dietary fish intake is comprised of protein (Undeland et al. 2009). Fish proteins are considered easily digestible and are rich in essential amino acids (the building blocks of proteins) (review in Costa 2007). Essential amino acids are those that cannot be made by the body and must be supplied in the diet.

B. Seafood as a source of essential fatty acids

Omega-3 (n-3 polyunsaturated) fatty acids (n-3 PFAs) are essential fats that the body needs to function properly but cannot make on its own. Humans must get n-3 PFAs from foods. The n-3 PFAs that are particularly important in human nutrition include: alpha-linoleic acid (ALA), eicosapentaenoic acid (EPA), docosapentaenoic acid (DPA), and docosahexaenoic acid (DHA) (review in Mahaffey et al. 2011). EPA and DHA are important for human neurological development. These two n-3 PFAs play a role in: (1) cell membrane formation, integrity, and functions; (2) functioning of brain, retina, liver, kidney, adrenal glands, and gonads; and (3) local hormone production for the regulation of blood pressure and immune and inflammatory responses.

Humans can synthesize some of their requirement for long-chain PFAs from ALA (a short-chain PFA), but generally not in the amounts sufficient to meet dietary needs (review in Mahaffey et al. 2011). Fish is rich in the very long-chain n-3 PFAs EPA and DHA. Fish and shellfish acquire these PFAs by consuming algae (or other fish). Certain fatty fish (e.g., salmon, herring, sardines, mackerel) are also high in n-3 PFAs (Table 1). Some that are low in fat (e.g., trout and shrimp) are also good sources of n-3 PFAs.

Table 1. Frequency of consumption rank and concentrations of n-3 PFAs

Rank	Species	EPA + DHA (mg/100g fish)*
1	Shrimp	390
2	Tuna (all, average) Tuna canned – light (skipjack) Tuna canned – white (albacore) Tuna fresh – Bluefin (7 kg) Tuna fresh – Skipjack (3 kg) Tuna fresh – Yellowfin (5-20 kg)	630 128 – 270** 862 1,173 – 1,504** 256 – 328** 100 – 120**
3	Breaded fish products	0.26
4	Salmon	1,590
5	Crabs	36
6	Catfish	280
7	Other fish	54
8	Scallops	270
9	Lobster	360
10	Clams	240
11	Cod	240
12	Oysters	350
13	Other shellfish	310
14	Flatfish	15
15	Unknown fish	53
16	Pollock	260
17	Mussels	350
18	Trout	580
19	Haddock	180
20	Crayfish	380
21	Perch	300
22	Sardines	980
23	Swordfish	580
24	Bass (freshwater)	640
25	Sea bass	490
26	Pike	140
27	Mackerel – except King King mackerel	1,790 401
28	Shark	220
29	Walleye	530
30	Porgy	210

Source: Adapted from Mahaffey et al. 2011
*100 grams is equivalent to a 3.5-ounce serving
** range shown is raw vs. cooked

C. Seafood as a source of nutrients (vitamins and minerals)

Many Americans do not eat the variety and amounts of foods necessary to supply needed nutrients (USDA 2010). Seafood supplies a number of these essential vitamins and minerals, including: vitamins A, B3, B6, and B12, and D, and the minerals calcium, iron, selenium, and zinc (summarized from USEPA and TERA 1999; USDA 2010, others).

a. Vitamins

Vitamin A (retinol, retinal, retinoic acid) is a fat-soluble vitamin that is critical for vision, growth, bone development and maintenance, and immune function, among others. In general, fish is a better source of Vitamin A than beef, pork, or chicken, and higher-fat fish species contain more of this vitamin than lower-fat species. Some foods, such as milk, are fortified with this vitamin.

Vitamins B3 and B6 are water soluble – they must be supplied daily in the diet because the body stores these vitamins only briefly. Vitamin B3 (niacin, nicotinic acid, and nicotinamide) is involved in hydrogen transfer reactions and deficiency in the diet causes a condition called Pellagra, which is characterized by diarrhea, dermatitis, and dementia. Vitamin B6 exists as several different chemical structures that serve as important co-enzymes in various chemical reactions in the body that are involved predominantly in metabolism. Deficiency in Vitamin B6 is very rare in the United States. Vitamin B12 is important in normal functioning of the brain and nervous system and for the formation of blood. It is produced by microorganisms in animals and does not occur naturally in plant foods. Deficiencies of Vitamin B12 generally occur if absorption from food is impaired. Sport-caught fish (salmon, trout, channel catfish) are excellent sources of this vitamin.

Vitamin D is a fat-soluble vitamin that occurs naturally in several forms and can be synthesized in the skin when sun exposure is adequate. Vitamin D acts like a steroid hormone, controlling blood levels of calcium to affect bone, kidney, and intestinal tissues. Deficiency of Vitamin D leads to bone demineralization and a condition called osteomalcia (in children it is called rickets). Fish is among the best food sources of this vitamin. Some foods, such as milk and some packaged orange juices, are fortified with this vitamin.

b. Minerals

Calcium is necessary for bone health as well as nerve transmission, constriction and dilation of blood vessels, and muscle contraction. Low calcium intake is associated with low bone mass and risk of bone fractures. Dairy products provide about one-half of dietary calcium in the United States. Fish with soft bones (walleye, bass, and yellow perch), small fish eaten whole (sardines, smelts), and canned fish with bones (salmon) can contribute substantial amounts of calcium to the diet.

Iron is an essential part of many proteins and enzymes, including the proteins involved in oxygen transport (hemoglobin, myoglobin). It is also important in many enzyme oxidation/reduction reactions. Iron deficiency is the most common nutritional deficiency in the United States The heme form of iron, which is the most readily absorbed form, can be found in animal foods (meat fish). Seafood is a good source of heme iron.

Selenium, an essential nutrient, is present in the cells of all mammals. When bound to certain proteins, selenium acts as an antioxidant by detoxifying free radicals. Selenium must be supplied through dietary sources. Seafood and organ meats such as liver are the best sources of dietary selenium.

Zinc is part of many enzymes and deficiencies can cause stunted growth and delayed sexual maturation. In general, foods rich in protein are also good sources of zinc. Insufficient zinc levels are common in North America, particularly in vegetarians and adult women.

D. Seafood consumption and improved health outcomes

Seafood consumption is linked with improvements in health conditions including cardiovascular disease, arthritis, and cancer. The following sections summarize reviews of the published evidence regarding the various health benefits of seafood consumption. Where the evidence to support a link between seafood consumption and a specific health benefit is considered preliminary and further research is needed (e.g., with treatment of mood disorders), this is noted in the review for that condition.

There has been a tremendous amount of research into the health benefits of seafood consumption in recent years. A Google Scholar® search[3] of scientific literature using the keywords seafood +benefits +risk +review and limiting the results of the search to the last two years turned up over 2,700 articles and reports. McManus et al. (2010) provide a recent overview of health benefits from seafood consumption, although this is aimed toward benefits in the elderly. For this issue paper, the McManus et al. (2010) review was used as the basis for most of the discussions of health benefits and was augmented using primary literature and other reviews.

a. Cardiovascular disease/stroke

Over 81 million Americans (37 percent of the population) have cardiovascular disease (CVD) (AHA 2010). Major risk factors for CVD include high levels of blood cholesterol and other lipids, type 2 diabetes, hypertension (high blood pressure), metabolic syndrome, obesity, physical inactivity, and tobacco use. Hypertension affects 34 percent of Americans and is a major risk factor for heart disease, stroke, congestive heart failure, and kidney disease.

From McManus et al. 2010: "Strong evidence exists supporting the assertion that fish intake significantly contributes to the maintenance of heart health, protecting against cardio-vascular diseases, particularly ischemic stroke. Even a small amount of fish can provide a protective health effect for seniors. For example, 1 to 2 serves a week of oily fish (sardines, salmon, trout) is associated with a reduced rate of hospitalization and mortality, with the highest evidence for older women."

Consumption of 1 to 2 fish meals per week lowers the relative risk of mortality from coronary heart disease (CHD) by 20 to 30 percent (Costa 2007). In fact, moderate evidence indicates that consumption of about 8 ounces of seafood per week is associated with reduced cardiac deaths among individuals with and without pre-existing CVD (AHA 2010). DHA and EPA in the diet likely reduce the risk of fatal CHD by reducing cardiac arrhythmias (Harris et al. 2009). An

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[3] http://scholar.google.com/

arrhythmia is a problem with the rate or rhythm of the heartbeat – too slow, too fast, or with an irregular rhythm. A review by Dórea (2008) found two studies indicating that the cardiovascular benefits of fish are due not only to the n-3 PFAs but also to the fish proteins. Even short-term DHA + EPA consumption (> 1 to 2 g/day) favorably affects many physiological measures of cardiovascular risk including blood pressure, resting heart rate, triglyceride levels, and possibly heart rate variability (Harris et al. 2009). Other reviews have concluded that n-3 PFAs alone do not have a clear effect on total mortality or combined CVD events and that positive effects were only seen when taking into account fish-based studies (Undeland et al. 2009). This suggests that other, non-n-3 PFA compounds in fish contribute to cardio-protective and neuro-protective value of fish consumption. For example, Vitamin D insufficiency has been related to CVD, and reduced Vitamin D status has been associated with several CVD risk factors including blood pressure and body-mass index.

Djoussé et al. (in press) conducted a meta-analysis[4] of cohort studies that evaluated the association between fish consumption or n-3 PFA and heart failure. The authors retained seven prospective studies conducted in the United States (four studies) or Europe (three studies) with 176,441 participants in whom 5,480 incident heart failures occurred. The average duration of follow-up in the studies was 13.33 years. The analysis found that higher fish consumption and a higher dietary or plasma concentration of EPA/DHA were each associated with about a 15 percent lower risk of heart failure compared with the lower exposure category (i.e., those who ate less fish or had less EPA/DHA in their plasma). The authors noted that their conclusions were based on a rather small number of observational studies (albeit with a large number of individuals when combined), and stressed that the findings need to be confirmed with a large randomized trial.

Dekelbaum and Calder (2012) discuss reviews and meta-analyses favorable to n-3 PFAs in lowering the risk for CHD and myocardial infarction (heart attack), in contrast to some recent clinical trials that found no or very little benefit from EPA and DHA. These experts concluded that, although the results of recent trials have been mixed, "it seems safe and sensible to follow the American Heart Association's recommendations that all adults eat fish (especially fatty fish) at least twice a week…" Eussen et al. (2011) theorized that recent failures of secondary prevention trials to demonstrate a beneficial effect of n-3 PFAs on cardiovascular outcomes may be due to the growing use of statin drugs since the mid-1990s. When the patients in one of those trials (the Alpha Omega Trial) were divided into statin-users and statin non-users, it was found that statins modify the effects of n-3 PFAs on major cardiovascular events (fatal cardiovascular diseases, non-fatal myocardial infarction, non-fatal cardiac arrest, non-fatal stroke and cardiac interventions) after myocardial infarction. In statin users, supplementation with n-3 PFAs had no effect. In the statin non-users, n-3 PFA supplementation (EPA-DHA plus ALA) had 54 percent fewer major cardiovascular events than those on placebo.

Evidence for effects on stroke is mixed. Meta-analyses indicate a 30 percent lower risk of ischemic stroke with fish intake of ≥ 1 meal/week compared with < 1 meal/month (Harris et al. 2009). Ischemic stroke involves blockage of the arteries leading to the brain caused by

[4] A meta-analysis is a study in which the results of other studies, all published on the same topic, are combined so as to gain more information regarding the shared area of inquiry; the larger size of the pooled data lends more evidence to any conclusions that are drawn.

artherosclerosis and thrombosis (Undeland et al. 2009). The mechanisms protecting against ischemic stroke are believed to be the same as those for CHD.

b. Arthritis

Rheumatoid arthritis is a chronic inflammatory autoimmune disease of the joints and bones. The n-6 PFA arachidonic acid is the precursor of inflammatory compounds (eicosanoids), which are involved in rheumatoid arthritis. The n-3 PFAs EPA and DHA decrease the arachidonic acid content of cells involved in autoimmune responses, decreasing the production of the inflammatory eicosanoids (summary in Miles and Calder 2012). In a case control study, Rossell et al. (2009) found that the risk for developing rheumatoid arthritis was 20 percent lower for those who consumed oily fish one to seven times per week compared to those who never or seldom consumed oily fish. These researchers found no effect of n-3 PFA supplements on incidence of rheumatoid arthritis.

Studies in Japan and of Eskimos have suggested that consumption of oily fish reduces the risk of rheumatoid arthritis (review in Lahiri et al. 2012). Miles and Calder (2012) reviewed the results of 23 randomized controlled studies of patients with rheumatoid arthritis and found evidence "for a fairly consistent, but modest, benefit of marine n-3 PFAs on joint swelling and pain, duration of morning stiffness, global assessments of pain and disease activity, and use of non-steroidal anti-inflammatory drugs."

Greater intake of Vitamin D is also associated with lower risk for rheumatoid arthritis and improvement in rheumatoid arthritis patients (Undeland et al 2009). Vitamin D could have an immunosuppressive role in these effects, but more research is needed (Undeland et al. 2009). A more recent review by Lahiri et al. (2012) found that the effect of Vitamin D deficiency, with respect to rheumatoid arthritis, remains unclear.

c. Cancer

McManus et al. (2010) state that "high fish intake has been associated with significantly reduced risk of ovarian and colorectal cancer. Furthermore, findings from a recent United Kingdom Women's Cohort Study of 35,372 women supports the assertion that postmenopausal women who consumed fish experienced a significantly reduced risk of breast cancer when compared with red meat consumers, indicating reduced risk in older women who prefer fish as a primary protein source to the exclusion of red meat. High level evidence supports fish consumption as protective in reducing the risk of prostate and lung cancers in males. Increased consumption of seafood also confers protection against the development of esophageal cancer in males aged 45 years and older in large population-based studies." A systematic review by Gerber (2012) found that there is increasing evidence suggesting a reduced risk of breast cancer associated with long-chain PFAs from fish consumption. In a cohort study of women who had been diagnosed with early stage breast cancer, Patterson et al. (2011) found that women with higher intakes of EPA and DHA from food had an approximately 25 percent lower risk of additional breast cancer events compared to women in the group with the lowest intake. These women also had a dose-dependent reduced risk of all-cause mortality. EPA and DHA intake from supplements, however, was not associated with breast cancer outcomes in this study.

Undeland et al. (2009), summarizing the benefits of seafood consumption, stated that observational studies[5] on fish consumption have shown protection for risk of digestion tract cancers (oral cavity and pharynx, esophagus, stomach, colon, and rectum) and also for ovary, pancreas, larynx, and endometrial cancers. Prospective[6] and case-control[7] studies, however, either do not show an association between fish intake and cancer risk, or only show reduced risk at high fish intake levels. Harris et al. (2009) states that the preponderance of evidence for anticarcinogenic effects of EPA + DHA is weak despite persuasive data from animal models and cultured tumor cell lines. However, a review and meta-analysis of fish consumption and prostate cancer risk that found no strong evidence of a protective association between fish consumption and prostate cancer risk did show a significant 63 percent reduction in prostate cancer-specific mortality (Scymanski et al. 2010). Cockbain et al. (2012) reviewed the research to date on n-3 PFAs and colorectal cancer and found that there is growing epidemiological, experimental, and clinical evidence that n-3 PFAs have anti-colorectal cancer activity and that they may play a role in primary prevention as well as treatment of colorectal cancer and advanced metastatic disease. A meta-analysis of 22 prospective case studies and 19 case-control studies by Wu et al. (2012) found that fish consumption decreased colorectal cancer incidence by 12 percent. However, a review by Gerber (2012), indicated that evidence is too limited to draw firm conclusions on the effect of long-chain n-3 PFAs and colorectal cancer.

d. Macular degeneration

n-3 PFAs are found in photoreceptor outer segment membranes and are a potentially important nutrient for eye health (Lim et al. 2012). Depletion of DHA may impair retinal function and influence the development of a condition known as age-related macular degeneration (AMD) (Souied 2012). AMD is a deterioration or breakdown of the eye's macula and is a common eye condition among people age 50 and older. The macula is the part of the eye that provides sharp, central vision needed for seeing objects clearly. AMD is a leading cause of vision loss in older adults. Cardiovascular disease is a system risk factor for AMD and cardiovascular risk factors such as hypertension have been linked with AMD (Lim et al. 2012). As noted above, intake of seafood and n-3 PFAs can improve cardiovascular health.

In a European study, consumption of oily fish at least once per week reduced the risk of neovascular AMD by one-half compared with consumption less than once per week (review in Souied 2012). Recently, a cohort study of female medical professionals (over 38,000 with average age of 54) found regular consumption of DHA and EPA and fish was associated with significantly decreased risk of incident AMD and may be of benefit in primary prevention of AMD (Christen et al. 2011). Women who consumed one or more servings of fish per week, compared with those who consumed less than one serving per month, had a relative risk of AMD of 0.58. According to the review by Souied (2012), the growing body of evidence shows that the regular consumption of oily fish or a higher intake of n-3 PFAs is associated with a 30 to 50

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[5] An observational study draws inferences about the possible effect of a treatment on subjects, where the assignment of subjects into a treated group versus a control group is outside the control of the investigator.

[6] An analytical study designed to determine the relationship between a condition and a characteristic shared by some members of a group.

[7] An analytical study which compares individuals who have a specific disease ("cases") with a group of individuals without the disease ("controls").

percent risk reduction in the occurrence or progression of AMD. However, Souied (2012) also noted that the majority of evidence is observational and further research involving cohort studies and randomized clinical trials is needed.

e. Cognitive function

According to McManus et al. (2010), “omega-3 PUFAs in seafood play an important role in neurological structure and function. Docosahexaenoic acid (DHA), a long-chain marine PUFA found in seafood, is a catalyst for the slowing of early stage progression of dementia. Further study is expected to shed light on how DHA potentially prevents the neurological damage that results from dementia.” The review further states that “research from marine and human epidemiological studies suggest that ‘higher fish consumption is associated with better cognitive function in later life’, enabling resistance to cognitive decline. Recent evidence strongly associates a dietary profile in which fish features prominently, with lower risk of developing Alzheimer’s disease (AD) and maintaining cognitive function. Evidence increasingly supports the assertion that marine source omega-3 PUFAs in fish play a role in delaying onset and arresting the progression of AD, though further studies are needed to investigate the mechanism involved.”

The EPA and DHA review by Harris et al. (2009) found that the available evidence for protective effects of long-chain PFAs from fish on the risk of dementia is promising but limited. Epidemiological evidence is positive for benefits of just one fish meal per week on the risk of Alzheimer’s disease, dementia, and cognitive decline. Animal evidence supports a protective relationship of n-3 PFAs on neurodegeneration of the brain with aging, and several studies in progress in the United States and Europe should shed more light on this relationship. Cunnane et al. (2012) note that the negative results of clinical trials with DHA supplements do not agree with the largely protective link of fish and DHA that is seen in prospective epidemiological studies. In a review that included seven studies of the effects of supplements containing long-chain n-3 PFAs (LCP), Dangour et al. (2012) found that only one identified any potential benefits on cognitive function. These authors conclude that “Whether this lack of evidence results from insufficient thought into designing studies of appropriate size and duration, or whether it relates to the selection of study populations who may benefit most from n-3 supplementation, such as those with low n-3 LCP status at study entry, or finally whether it suggests that despite the epidemiological and mechanistic evidence n-3 LCP supplementation does not affect cognitive function, remains open to question.”

f. Hearing loss

Gopinath et al. (2010) found an inverse association between higher intakes of long-chain n-3 PFAs and regular weekly consumption of fish and hearing loss. Higher dietary intake of long-chain n-3 PFAs was associated with a 24 percent decreased risk of developing incident hearing loss. Regular consumption of fish in the diet was negatively associated with the 5-year incidence and progression of hearing loss in older adults. These data suggest that n−3 fatty acids and fish have a role in maintaining healthy auditory function.

g. Depression/mood

According to McManus et al. (2010), “intake of omega-3 PUFA rich seafood is linked to increased dispositional optimism in the elderly, and has, in some long term studies, been linked to reduced depression, with a recent meta-analytic review of polyunsaturated fatty acid levels in patients with depression concluding that ‘n-3 polyunsaturated fatty acids play a role in the pathogenesis of depression’. Therefore, omega-3 PUFA rich seafood could benefit individuals suffering from depression. Further research on the possible role of seafood consumption in moderating depression is required for these findings to be substantiated.”

Hegarty and Parker (2011) reviewed the evidence implicating marine n-3 PFAs in the etiology [causes or origin] of depressive and bipolar disorders and the effect of n-3 PFA supplementation in the treatment of those disorders. The authors focused primarily on studies conducted within the previous 5 years, and found the following: (1) the evidence suggests a contributory etiological role of n-3 PFA deficiency to depressive and bipolar disorders; (2) a growing body of evidence implicates inflammatory processes in the etiology and/or progression of depression; (3) n-3 PFAs play a role in modulating the n-6 PFA derived pro-inflammatory molecules and are able to suppress the expression of a wide variety of inflammatory genes; (4) a recent meta-analysis concluded that levels of EPA, DHA, and total n-3 PFAs are significantly lower in depressed patients compared to controls; (5) trials of EPA-rich n-3 PFA supplements were more effective in treating depressive symptoms than DHA-rich preparations; and (6) an increasing number of trials do not support n-3 PFA supplementation as an effective treatment for major depressive disorder or postnatal depression. Hagarty and Parker (2011) concluded that more research focusing on EPA-rich supplements is warranted and that, even with many questions remaining, two fish servings per week would be recommended with supplementation for those with depressive or bipolar disorders. They believe it is possible that n-3 PFA supplementation may be of benefit for mood disorders but that research has not yet established the right dose and/or the right constituent ratio. A review of the effects of n-3 PFA supplementation on behavior and neuropsychiatric disorders by Ortega et al. (2012) found that most of the existing studies do not meet the requirements to elucidate the effect of n-3 PFAs, for example, by failing to control for dietary intake.

h. Osteoporosis

McManus et al. (2010) state that “seafood is a rich source of both calcium and Vitamin D, important bone-building micronutrients. Vitamin D rich seafood can play an important role in the maintenance of bone mineral density as people age. Potential reduced sun exposure and an increased requirement of Vitamin D in older people underpins the need for high quality, bioavailable Vitamin D. Seniors also have a reduced capacity to ‘synthesize provitamin D3 in skin and to hydroxylate vitamin D3 in kidneys. It is widely recognized that a diet high in oily fish prevents vitamin D deficiency; and commonly consumed, affordable sources of seafood such as Australian salmon and silver perch contain more than double the recommended daily intake of Vitamin D in a 150 g serve. A 150 g serve of Australian Salmon will also deliver more than half the recommended daily intake of calcium. Calcium requirements increase with age and seafood presents rich serves of calcium combined with optimal amounts of Vitamin D to aid its absorption, protecting bone mineral density (BMD).

Loss of calcium through urinary excretion is of concern to bone health. Evidence is emerging showing lower fractures and higher bone mineral density with the consumption of adequate levels of calcium rich, high protein seafood. This may be due to increased intestinal absorption, which negates the impact of urinary excretion. When calcium and vitamin D intake is adequate, dietary protein at moderate levels is beneficial to total body BMD particularly for seniors.

Seafood is a good source of calcium, vitamin D, and protein [and] therefore can favorably contribute to BMD."

However, according to a review by Orchard et al. (2012), a small number of epidemiological studies investigating the relationship of fish consumption or dietary n-3 PFA consumption to fracture risk have yielded mixed results. For example, in a study of post-menopausal Chinese women, high intake of sea fish was associated with greater bone mass and lower osteoporosis risk, especially those consuming more than 250 grams of seafood per week (Chen et al. 2010). But in a study of elderly Japanese women, consuming fish 3 to 4 times per week was associated with decreased risk of hip fracture but eating more than 4 fish meals per week did not improve relative risk (summary in Orchard et al. 2012). Orchard et al. (2012) concluded that "there is a need for further large scale investigation of the differential effects of various n-3 PFAs in relation to skeletal outcomes, particularly fracture."

i. Recent Research

The following summaries do not represent the results of an exhaustive literature search, which is outside the scope of this review, but are articles of interest that were found when searching for recent reviews on the benefits of seafood consumption. These articles are too new to have made it into the review papers that were published recently.

Gopinath et al. (2011) investigated the association between dietary intakes of PFA (n-3, n-6 and α-linoleic acid), fish and the prevalence of chronic kidney disease (CKD). Due to the anti-inflammatory properties of PFA, it has been suggested that they may protect against kidney damage in adults. These researchers found that an increased dietary intake of long-chain n-3 PFA and fish reduced the prevalence of CKD. Therefore, a diet rich in n-3 PFA and fish could have a role in maintaining healthy kidney function.

In a case-control study of maternal dietary patterns and congenital heart defects, a dietary pattern characterized by the high intake of fish and seafood was associated with a reduced risk of congenital heart defects in offspring (Obermann-Borst et al. 2011). Based on these findings, the researchers suggested further investigation in a randomized intervention trial was warranted.

Changing human dietary patterns in the last 100–150 years have given rise to an imbalance in the dietary ratio of n-6 and n-3 PFAs (review in Candela et al. 2011). This ratio used to be around 1–2:1 but in Western diets is now as high as 20–30:1. n-3 PFA intake is much lower today due to factors such as decreased fish consumption while n-6 PFA intake is higher due to factors such as increases in n-6 rich grains and vegetable oils. Although n-6 and n-3 PFAs exert opposing effects on inflammatory activity (n-3 PFAs reduce inflammation), high n-6:n-3 ratios have not been correlated with high inflammatory marker levels. Some researchers point to the need to decrease n-3 PFA intakes while others call for increasing n-3 PFA intakes to mediate the imbalance in the ratio. Beneficial effects on asthma (an inflammatory disease) have been seen at ratios of 5:1 with adverse effects seen at ratios of 10:1 (review in Candela et al. 2011). With regard to cancers, a high n-6:n-3 ratio was associated with increased prostate cancer risk in a recent study (Williams et al. 2011).

II. Risks associated with eating fish and shellfish

Risks associated with eating fish and shellfish include the risk from chemical contaminants (fish and shellfish) and pathogens or biotoxins (generally in shellfish).

According to the U.S. Environmental Protection Agency (USEPA) 2010 listing of fish consumption advisories (USEPA 2011), all 50 states have fish consumption advisories in place to protect their residents from the potential health risks of eating contaminated fish caught in local waters. As of 2010, 42 percent of the nation's total lake acreage and 36 percent of the nation's total river miles were under advisories. There were consumption advisories for 33 different environmental pollutants in the United States in 2010—a total of 4,598 fish consumption advisories were in effect. Three pollutants—methylmercury, PCBs, and dioxins—were responsible for nearly 90 percent of the advisories. These contaminants accumulate in fish tissue at concentrations many times higher than concentrations in the water and can persist for years in sediments. Fish consumption advisories are designed to "avoid" risks from consuming different contaminated fish. For example, the metrics of advice prescribes meal frequency and size to reduce exposures to mercury or other fish contaminants. All states, except Nebraska, offer meal frequency advice in terms of fish meals to avoid per week, month, year or some combination thereof (Scherer et al. 2008).

Some shellfish species feed by taking in large volumes of water and filtering out the food particles (WDOH 2012). These filter-feeding species, such as oysters, clams, and mussels, can accumulate natural biotoxins as well as pathogens (e.g., norovirus, fecal coliforms, and vibrio) in their edible tissues. The Washington Department of Health (WDOH) Shellfish Program monitors for biotoxins such as paralytic shellfish poison and domoic acid in shellfish and closes harvest areas when levels pose a threat to public health. The Shellfish Program also classifies shellfish growing areas, monitors marine water for pollution, licenses and inspects commercial growers, and provides recreational harvesters with information about where, when, and what type of shellfish are safe to harvest. Washington State shellfish consumers can minimize their risks of becoming ill from shellfish biotoxins and pathogens by avoiding raw shellfish (proper cooking kills pathogens although it does not neutralize biotoxins) and complying with shellfish harvesting restrictions when the Shellfish Program closes harvest areas. This issue paper focuses on chemical contaminants in seafood and so pathogens and biotoxins are not discussed further.

A. Brief summary of potential health effects

Toxic contaminants enter Washington's ecosystems via water (e.g., river or stream inputs, industrial discharges, stormwater runoff), the atmosphere, and in the bodies of migrating organisms. Some chemicals in the environment pose an immediate health threat. Others gradually build up in the environment and, in humans, can cause disease long after exposure. Contaminants in sediment and water may accumulate in fish and shellfish through processes called bioaccumulation and bioconcentration. Bioaccumulation is the accumulation of chemicals in the tissue of fish or shellfish through respiration, ingestion, or direct contact with contaminated water, sediment, and pore water in the sediment. Bioconcentration is the process of accumulation of water-borne chemicals by fish or shellfish through non-dietary routes. Some contaminants are biomagnified up the food chain. This occurs when the tissue concentration increases at each trophic level in the food chain when there is efficient uptake of the contaminant but slow elimination. Hydrophobic chemicals—those that adhere to organic particles in sediments—tend to be bioaccumulative. Two types of contaminants that may occur in seafood are metals and organics. These are described below.

a. Metals

Metals include essential trace elements required for human health, such as zinc and copper, and those without any known human dietary need, such as lead and mercury. Many metals, even those considered essential in the diet, can be toxic at high enough doses. Some metals are widespread in the environment due to natural processes (e.g., arsenic) but have increased locally or globally due to human activities (Eisler 2000). Metals contamination occurs throughout the state from a variety of sources such as industrial releases, deterioration or wear of roofing and other materials, batteries, paints, and dyes. Methylmercury, an organic form of mercury, is included in a class of contaminants termed Persistent Bioaccumulative Toxics (PBTs).[8] PBTs are of concern because they resist degradation, bioconcentrate from the environment (e.g., water, sediments) into organisms, and may biomagnify up food chains (PSAT 2007).

b. Organics

Organic contaminants are natural or manmade chemicals that include several PBTs: dioxin and dioxin-like compounds (polychlorinated-dibenzo-p-dioxins [PCDDs], polychlorinated-dibenzo-furans [PCDFs], and polychlorinated biphenyls [PCBs]), polycyclic aromatic hydrocarbons (PAHs), and organochlorine pesticides (e.g., chlordane, DDT).

PAHs are a group of organic chemicals that have a fused ring structure of two or more benzene rings, and are formed during the incomplete combustion of organic materials. PAHs are ubiquitous in nature, largely as the result of natural processes such as forest fires and microbial synthesis. Anthropogenic sources of PAHs include vehicle exhaust; manufacturing of coal tar pitch and asphalt; petroleum refining; and open burning (Eisler 2000 and others). The largest sources of PAHs in Washington are wood burning stoves and fireplaces. Other major sources include creosote-treated wood, vehicles, leaks and improper disposal of motor oil, and small engines (e.g., lawn mowers and garden equipment). PAHs in fish and shellfish are a result of contamination of water and sediment while PAHs in livestock (e.g., beef) are from consumption of contaminated pasture and vegetation. PAHs may accumulate in shellfish but generally do not biomagnify because most are readily metabolized by other species such as fish and humans (Eisler 2000).

Dioxin-like compounds include PCDDs, PCDFs, and certain PCBs. PCDDs and PCDFs have natural sources but are also created and are considered unwanted impurities in some industrial processes and products such as pesticides (Eisler 2000). Common sources of dioxins in Washington State include waste incinerators, pulp mills, industrial processes, and backyard burn barrels. PCBs are a group of 209 synthetic organic chemicals that were used as insulating and cooling agents in the electricity generating industry, and that had a number of other industrial applications (e.g., as additives to caulks and paints to improve durability). The production and

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[8] Two metals (cadmium and lead) are considered PBT metals of concern in Washington State (Washington Administrative Code 173-333) but these metals are not always analyzed in seafood tissue samples and do not often appear in articles and reports regarding PBTs in seafood in Washington.

sale of PCBs was banned in the United States in the late 1970s, but these compounds have persisted in the environment in soils, sediments, and organisms. Dioxins and PCBs bioaccumulate and biomagnify in the food chain.

c. Primary chemicals of concern in Washington fish and shellfish

Fish and shellfish in Washington State have been exposed to pollutants discharged into waters including sewage, pulp and paper industry wastes, petroleum products, heavy metals, and synthetic organic chemicals (e.g., pesticides, PCBs). Accidental spills of dangerous materials and past business practices have contaminated land and waters. In 2006, WDOH reviewed Puget Sound fish tissue data from the Washington Department of Fish and Wildlife to determine which contaminants have the potential to cause public health concern (WDOH 2006). The following chemicals were detected in 10 percent or more of the samples: alpha chlordane, arsenic, benzyl alcohol, copper, DDT and degradation products, DEHP, mercury, and PCBs. Of these chemicals, only PCBs and mercury were detected with sufficient frequency and at high enough levels that WDOH believed an assessment of health risk was warranted at that time. WDOH did not include shellfish in the risk assessment. Some shellfish are known to accumulate PAHs due to their inability to metabolize these compounds. Therefore, because shellfish may be a significant source of PAHs for shellfish consumers, PAHs were included in this Technical Issue Paper. Note that the Puget Sound Assessment and Monitoring Program did not analyze tissues for PCDDs or PCDFs. However, data from the fish and shellfish studies in the Ecology Environmental Information Management Database indicate that these chemicals are found in Washington State fish and shellfish. This Technical Issue Paper focuses on the health risks associated with mercury, PAHs, dioxins/furans, and PCBs.

B. Adverse health effects associated with primary contaminants of concern

a. Mercury

Mercury and mercury compounds have no known biological function and their presence in biological organisms is undesirable and potentially hazardous (review in Eisler 2000). Most atmospheric mercury is in elemental or inorganic forms while the mercury in water, soil, plants, and animals is generally in organic forms. Bacteria in the environment convert inorganic mercury to organic forms such as methylmercury. Methylmercury is considered the most hazardous mercury compound due to its high stability, its lipid solubility, and its ability to penetrate membranes in living organisms. Methylmercury can bioconcentrate in organisms and biomagnify up food chains. Organic mercury can cross the blood-brain barrier[9] and the placenta. For all organisms tested, early developmental stages were the most sensitive to mercury poisoning, and organic forms were more toxic than inorganic forms. Mercury is not concentrated in fat but is associated with protein and therefore in animal-based foods it is found in the meat/fillet.

[9] The blood-brain barrier is a semi-permeable membrane that allows some materials to cross but prevents others from crossing; it protects the brain from foreign substances, protects the brain from hormones and neurotransmitters from other parts of the body, and maintains a constant environment for the brain.

Mercury is a known teratogen (causing birth defects), mutagen (causing DNA mutations), and carcinogen (review in Eisler 2000). Mercury can adversely affect reproduction, growth and development, behavior, blood serum chemistry, motor coordination, vision, hearing, histology, and metabolism at low concentrations. In several studies, high maternal exposure to mercury during pregnancy have been associated with impaired child cognitive development and achievement of developmental milestones as measured by various tests including language, motor skills, and intelligence (Oken et al. 2008; Cabelli et al. 2009; review in FAO/WHO 2010).

b. Arsenic

Arsenic is bioconcentrated but does not biomagnify in food chains (Eisler 2000). Therefore, arsenic concentrations are usually low in most organisms. Exposure to inorganic arsenic is generally from contaminated drinking water (WDOH 2009). Arsenic in food sources including seafood is generally in organic forms, primarily arsenobetaine. Arsenobetaine is very stable; after ingestion, it is rapidly excreted unchanged in the urine (summary in Vahter 1994).

Acute or subacute exposure to arsenic compounds can cause appetite loss, reduced growth, hearing loss, dermatitis, blindness, degenerative changes in the kidney and liver, cancer, chromosomal damage, birth defects, and death (review in Eisler 2000). Chronic exposure to arsenic compounds is associated with liver, kidney, and heart damage, hearing loss, brain-wave abnormalities, and impaired resistance to viral infection. However, the probability of chronic poisoning from continuous ingestion of small doses is rare due to the body's ability to detoxify and excrete arsenic rapidly.

In mammals, inorganic arsenic compounds are more toxic than organic arsenic compounds, and trivalent species (As^{+3}) are more toxic than pentavalent (As^{+5}) species (review in Eisler 2000). Ingestion of inorganic arsenic increases the risk of cancer of the skin, lungs, bladder, and kidneys. Early developmental stages are the most sensitive to arsenic. Inorganic arsenic can cross the placenta and produce mutagenic, teratogenic, and carcinogenic effects in offspring. However, at environmentally relevant levels and routes of exposure, humans are not at risk of birth defects from arsenic.

c. PAHs

Exposure to PAHs may occur through the lungs, stomach, or skin. Of these pathways, dietary intake is a major source of exposure for PAHs (Diggs et al. 2011). Chronic exposure can cause dermatitis and hyperkeratosis. Benzo(a)pyrene is considered to be one of the most toxic PAH compounds and is the PAH with the most available health effects data. The USEPA classifies benzo[a]pyrene as a probable human carcinogen based on multiple studies with rats and mice. According to a Joint Food and Agriculture Organization of the United Nations (FAO) / World Health Organization (WHO) Expert Committee report (WHO 2006), 13 PAHs are clearly carcinogenic and 15 PAHs are genotoxic in experimental animals. Cancer associated with exposures to PAH containing mixtures occurs mainly in the lung (inhalational exposure) and skin (dermal exposure). Ingestion of PAHs has been associated with esophageal cancer (Diggs et al. 2011).

d. PCDD/Fs

Dioxin-like compounds (referred to hereafter as dioxins) include PCDDs, PCDFs, and PCBs. Dioxins and furans are discussed here, while PCBs are discussed below. Dioxins affect the immune system and cause dermal and hepatic (liver) toxicity, a variety of endocrine (hormone) effects, and cancer (summary in Birnbaum and Fenton 2003; others). The embryo and fetal stage of development may be especially susceptible to dioxin effects (Birnbaum 2005). Dioxins are tumor promoters that cause tumors to grow and enhance the incidence and multiplicity of tumors at multiple sites in the body. Dioxins are persistent, accumulating and lasting in the body for years. For maternal to offspring exposure to dioxins (e.g., through placenta and breast milk), the majority, if not all, of the effects are associated with *in utero* exposure (Birnbaum 2005). Nursing leads to greater infantile exposure through the breast milk, but this does not have long-term effects on the adult body burden.

The mechanism of action for dioxin effects on vertebrate species is through activation of the aryl hydrocarbon receptor (Ah-receptor) (White and Birnbaum 2009). The Ah-receptor is believed to play key roles in development, aging, hypoxia, and circadian rhythms. Laboratory test animals that are missing the Ah-receptor[10] are not healthy, which highlights that the receptor and its controlled activation are necessary for well-being (summary in Tuomisto and Tuomisto 2012).

Experiments with animals have shown dioxin to be a multi-site carcinogen, but there is limited evidence for the carcinogenicity of dioxin in humans at environmentally relevant concentrations, and this issue is extremely controversial. Epidemiological studies of dioxin exposures and effects are complicated by the fact that everyone has some exposure to dioxins and to other chemicals, and carry body burdens of a suite of chemicals. Recently, Boffetta et al. (2011) conducted a critical review of the epidemiologic studies on human exposures to 2,3,7,8-tetrachlorodiobenzo-*p*-dioxin (TCDD) and cancer and concluded that "recent epidemiological evidence falls far short of conclusively demonstrating a causal link between TCDD exposure and cancer risk in humans." This finding is mirrored in the review by Tuomisto and Tuomisto (2012) who concluded that "Occupational cohorts with the highest exposures imply that there is a small risk of all cancers combined, but it is difficult to pinpoint the confounding effect of the main chemicals. Studies after major accidents do not unequivocally confirm this risk. The risks to populations *at the current dioxin levels*[11] seem trivial if present at all."

e. PCBs

Similar to dioxins, PCBs are lipophilic and move through the placenta and into milk (review in Santerre 2008). The USEPA has classified PCBs as a probable human carcinogen (USEPA 2012). Only a few PCBs are structurally similar to dioxin and therefore are considered to be dioxin-like in their toxic effects (i.e., toxicity mediated by induction of the Ah-receptor). The non-cancer effects of PCBs may include: immune system suppression, reproductive effects such as reduced birth weight, neurological effects such as learning deficits and changes in activity, and endocrine effects such as changes in thyroid hormone levels (USEPA 2012). The different health effects of PCBs may be interrelated, as alterations in one system may have significant implications for the other regulatory systems of the body. PCBs and dioxins are rarely found in environmental and biological samples in the absence of one another (White and Birnbaum 2009).

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[10] Animals that are bred specifically without the receptor to study drug toxicity and the biological function of the receptor.

[11] Emphasis added.

III. Comparison of health benefits and risks

This section describes the metrics used in evaluating the risks associated with contaminants and presents comparisons of health benefits to risks of seafood consumption. The risk assessments are presented by date order, with the most recent described first.

A. Metrics for evaluating risks

Contaminants can cause a variety of health effects including cancer and non-cancer health effects. The metrics used to evaluate risks from contaminants differ depending on if the toxic endpoint is cancer or not cancer (e.g., neurotoxicity, immunotoxicity). The following paragraphs, modified from USEPA and TERA (1999), describe the most commonly used metrics for these two types of endpoints.

Cancer slope factors are estimates of risk that are derived from dose-response data from laboratory animal or human epidemiology studies. Traditionally, a linearized multi-stage model has been used to extrapolate from what is observed at high experimental concentrations to lower environmental exposure levels. This cancer potency is estimated as the 95 percent upper confidence limit of the slope of the dose-response curve in the low dose region. This is an upper estimate of risk and the actual risk may be much lower or even approach zero. USEPA proposed revised cancer guidelines in 1996 and additional proposed guidance in 1998, which recommend that the mode of action be considered. The guidance recommends that a linear extrapolation should be used if the chemical is believed to act via a genotoxic mode of action, if the mode of action is expected to be linear at low doses, or (as a default) if no mode of action data are available. The guidance also recommends that a non-linear approach to extrapolation to low doses should be used when sufficient information on mode of action warrants.

For non-cancer endpoints, a reference dose is identified at which one would not expect to see adverse effects in a population (including sensitive subgroups). A single estimate of a “safe” dose is identified from animal or human data, using the No Observed Adverse Effect Level (NOAEL). This is divided by uncertainty factors to account for extrapolation from animals to humans, variability in the human population, and deficiencies in the database of studies on the substance. The resulting reference dose (RfD) is defined as “an estimate (with uncertainty perhaps spanning an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime.”

B. Risk/Benefit Comparisons

a. FAO/WHO (2011)

In January 2010, the FAO and WHO convened a Joint Expert Consultation on the Risks and Benefits of Fish Consumption (FAO/WHO 2011). The tasks of the Expert Consultation were to review data on levels of nutrients (long-chain n-3 PFAs) and two chemical contaminants (methylmercury and dioxins) in a range of fish species and to compare the health benefits of fish consumption and nutrient intake with the health risks associated with contaminants present in fish[12].

After reviewing the literature, the Expert Consultation decided to compare the effects of (1) prenatal exposure to long-chain n-3 PFAs (EPA and DHA) and methylmercury on child IQ and (2) exposure to long-chain n-3 PFAs and dioxins on mortality. The rationale for this choice was based on the common health end-points and relatively robust evidence to establish dose–response relationships from multiple cohort studies.

Using data on over 75 species of fish and shellfish, the Expert Consultation classified the content of n-3 PFAs (as EPA + DHA) by total mercury content and by dioxin content (as total TEQs) (Tables 2 and 3). The resulting matrices were produced using those classifications:

Table 2. Classification of the content of EPA plus DHA by total mercury content in 96 finfish and shellfish species

		EPA + DHA			
		x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
Mercury	x ≤ 0.1 μg/g	Fish: butterfish; catfish; cod, Atlantic; cod, Pacific; croaker, Atlantic; haddock; pike; plaice, European; pollock; saithe; sole; tilapia Shellfish: clams; cockle; crawfish; cuttlefish; oysters; periwinkle; scallops; scampi; sea urchin; whelk	Fish: flatfish; John Dory; perch, ocean and mullet; sweetfish; wolf fish Shellfish: mussels; squid	Fish: redfish; salmon, Atlantic (wild); salmon, Pacific (wild); smelt Shellfish: crab, spider; swimcrab	Fish: anchovy; herring; mackerel; rainbow trout; salmon, Atlantic (farmed); sardines; sprat Fish liver: cod, Atlantic (liver); saithe (liver) Shellfish: crab (brown meat)
	0.1 < x ≤ 0.5 μg/g	Fish: anglerfish; catshark; dab; grenadier; grouper; gurnard; hake; ling; lingcod and scorpionfish; Nile perch; pout; skate/ray; snapper, porgy and sheepshead; tuna, yellowfin; tusk; whiting Shellfish: lobster; lobster, American	Fish: bass, freshwater; carp; perch, freshwater; scorpion fish; tuna; tuna, albacore Shellfish: crab; lobster, Norway; lobsters, spiny	Fish: bass, saltwater; bluefish; goatfish; halibut, Atlantic (farmed); halibut, Greenland; mackerel, horse; mackerel, Spanish; seabass; seabream; tilefish, Atlantic; tuna, skipjack	Fish: eel; mackerel, Pacific; sablefish
	0.5 < x ≤ 1 μg/g	Fish: marlin; orange roughy; tuna, bigeye	Fish: mackerel, king; shark	Fish: alfonsino	Fish: tuna, Pacific bluefin
	x > 1 μg/g		Fish: swordfish		

[12] The U.S. Food and Drug Administration published a draft report on the benefits and risks of food consumption in .January 2009. The FAO/WHO Expert Consultation reviewed that draft report and incorporated some of the analyses in their assessment.

Table 3. Classification of the content of EPA + DHA by dioxin content in 76 finfish and shellfish species

		EPA + DHA			
		x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
Dioxins	x ≤ 0.5 pg TEQ/g	Fish: anglerfish; catshark; cod, Atlantic; grenadier; haddock; hake; ling; marlin; orange roughy; pollock; pout; saithe; skate/ray; sole; tilapia; tuna, bigeye; tuna, yellowfin; tusk; whiting Shellfish: cockle; clams; crawfish; cuttlefish; periwinkle; scallops; scampi; sea urchin	Fish: flatfish; John Dory; perch, ocean and mullet; shark; sweetfish; tuna, albacore	Fish: redfish; salmon, Pacific (wild); tuna, skipjack	
	0.5 < x ≤ 4 pg TEQ/g	Fish: catfish; dab; gurnard; plaice, European Shellfish: lobster; oysters; scallops; whelk	Fish: scorpion fish; swordfish; tuna Shellfish: mussels; squid	Fish: alfonsino; goatfish; halibut, Atlantic (farmed); halibut, Greenland; mackerel, horse; salmon, Atlantic (wild); seabass; seabream	Fish: anchovy; herring; mackerel; mackerel, Pacific; rainbow trout (farmed); salmon, Atlantic (farmed); tuna, Pacific bluefin Shellfish: crab (brown meat)
	4 < x ≤ 8 pg TEQ/g			Shellfish: crab, spider	Fish: sardines; sprat
	x > 8 pg TEQ/g			Fish: bluefish	Fish: eel Fish liver: cod, Atlantic (liver); saithe (liver)

For the risk-benefit comparisons, the Expert Consultation created matrices for the calculations of (1) the effects on child IQ as a result of the mother consuming one, two, four, or seven servings of fish per week with different n-3 PFA and methylmercury concentrations (Table 4), and (2) the effects on mortality as a result of consuming one, two, four, or seven servings of fish per week with different n-3 PFA and dioxin concentrations (Table 5). The matrices for the effects of methylmercury show estimates of changes in IQ. In the calculation, IQ points are lost as a result of methylmercury exposure and gained as a result of n-3 PFA. Positive numbers in green show where the net effect on IQ is positive—the gain in IQ points (using the upper bound estimate) is greater than the loss; negative numbers in red show where the net effect on IQ points is negative. Cells where the net effect is negative are shaded. The matrices for the effects of dioxins show the net of estimates of lives lost as a result of dioxin exposure and lives saved (due to reduction in coronary heart disease) as a result of n-3 PFA exposure. Where the net effect is positive (lives gained is greater than lives lost) this is represented by positive, green numbers. Shaded cells with negative numbers represent estimates of where the net effect is negative—lives lost are greater than lives saved.

Table 4. Estimated changes in child IQ resulting from maternal consumption of fish with different methylmercury and EPA plus DHA contents[a]

(a) One serving per week (3.5 ounces per week, or 14.3 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Methylmercury	x ≤ 0.1 µg/g	0.05	+0.68	+2.0	+4.3	+5.7
	0.1 < x ≤ 0.5 µg/g	0.3	+0.3	+1.6	+3.9	+5.3
	0.5 < x ≤ 1 µg/g	0.75	–0.43	+0.9	+3.2	+4.6
	x > 1 µg/g	1.5	–1.5	–0.2	+2.1	+3.5

(b) Two servings per week (7 ounces per week, or 28.6 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Methylmercury	x ≤ 0.1 µg/g	0.05	+1.3	+4.0	+5.6	+5.6
	0.1 < x ≤ 0.5 µg/g	0.3	+0.6	+3.3	+4.9	+4.9
	0.5 < x ≤ 1 µg/g	0.75	–0.8	+1.9	+3.5	+3.5
	x > 1 µg/g	1.5	–3.2	–0.5	+1.1	+1.1

(c) Four servings per week (14 ounces per week, or 57.1 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Methylmercury	x ≤ 0.1 µg/g	0.05	+2.8	+4.0	+5.5	+5.5
	0.1 < x ≤ 0.5 µg/g	0.3	+1.2	+3.9	+3.9	+3.9
	0.5 < x ≤ 1 µg/g	0.75	–1.6	+1.1	+1.1	+1.1
	x > 1 µg/g	1.5	–6.2	–3.5	–3.5	–3.5

(d) Seven servings per week (25 ounces per week, or 100 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Methylmercury	x ≤ 0.1 µg/g	0.05	+4.9	+5.3	+5.3	+5.3
	0.1 < x ≤ 0.5 µg/g	0.3	+2.1	+2.5	+2.5	+2.5
	0.5 < x ≤ 1 µg/g	0.75	–2.8	–2.4	–2.4	–2.4
	x > 1 µg/g	1.5	–10.9	–10.5	–10.5	–10.5

a Fish serving size was estimated to be 100 g. Ratio of DHA to EPA + DHA was assumed to be 0.67. Maternal body weight was assumed to be 60 kg. IQ points gained from DHA exposure was estimated using the coefficient of 4 IQ points for 100 mg of DHA intake. The maximum positive effect from DHA was estimated at 5.8 points. Yellow shaded cells represent the estimates where the net effect on child IQ, using the upper-bound estimate for methylmercury, is negative.

Table 5. Estimated changes in mortality per million people from consuming fish with different dioxin and EPA plus DHA contents [a]

(a) One serving per week (3.5 ounces per week, or 14.3 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Dioxins	x ≤ 1.0 pg/g	0.2	+4,500	+12,450	+26,150	+39,750
	1.0 < x ≤ 4.0 pg/g	2.5	+3,950	+11,900	+25,600	+39,200
	4.0 < x ≤ 8.0 pg/g	6.0	+3,150	+12,100	+24,800	+38,400
	x > 8.0 pg/g	20.0	–250	+7,700	+21,400	+35,000

(b) Two servings per week (7 ounces per week, or 28.6 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Dioxins	x ≤ 1.0 pg/g	0.2	+9,000	+24,900	+39,700	+39,700
	1.0 < x ≤ 4.0 pg/g	2.5	+7,900	+23,800	+38,600	+38,600
	4.0 < x ≤ 8.0 pg/g	6.0	+6,200	+22,100	+36,900	+36,900
	x > 8.0 pg/g	20.0	–400	+15,500	+30,300	+30,300

(c) Four servings per week (14 ounces per week, or 57.1 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Dioxins	x ≤ 1.0 pg/g	0.2	+18,010	+39,610	+39,610	+39,610
	1.0 < x ≤ 4.0 pg/g	2.5	+15,800	+37,400	+37,400	+37,400
	4.0 < x ≤ 8.0 pg/g	6.0	+12,500	+34,100	+34,100	+34,100
	x > 8.0 pg/g	20.0	–800	+20,800	+20,800	+20,800

(d) Seven servings per week (25 ounces per week, or 100 g/day)

			EPA + DHA			
			x ≤ 3 mg/g	3 < x ≤ 8 mg/g	8 < x ≤ 15 mg/g	x > 15 mg/g
		Median	2	5.5	11.5	20
Dioxins	x ≤ 1.0 pg/g	0.2	+31,570	+39,470	+39,470	+39,470
	1.0 < x ≤ 4.0 pg/g	2.5	+27,700	+35,600	+35,600	+35,600
	4.0 < x ≤ 8.0 pg/g	6.0	+21,900	+29,800	+29,800	+29,800
	x > 8.0 pg/g	20.0	–1,400	+6,500	+6,500	+6,500

a Fish serving size was estimated to be 100 g (3.5 ounces). Mean population body weight was assumed to be 60 kg. The numbers of lives lost from dioxin exposure was estimated using upper-bound estimates of risk. Estimates of the numbers of lives saved were due to reduction in coronary heart disease risk from EPA + DHA intake. The maximum positive effect from EPA + DHA was estimated to occur at 250 mg/day. Positive green numbers occur where lives saved due to EPA + DHA is greater than lives lost to dioxin. Yellow shaded cells with a red number represent the estimates where the net effect is negative; lives lost are greater than lives saved.

Note that none of the fish species evaluated by FAO/WHO fell into the maximum dioxin/minimum n-3 PFA category where risk was greater than benefit (see Table 3), and relatively few fish species fell into the mercury range where risk was greater than benefit (see Table 2).

It should be noted that Tables 4 and 5 assume a serving size of 100 grams (3.5 ounces). A study by Smiciklas-Wright et al. (2002) estimated mean and 90th percentile fish meal size for general population adults using data gathered in the 1994–1996 USDA continuing survey of food intakes by individuals. The average amount of finfish (other than tuna) per eating occasion was 114 grams; males, age 40 to 59, had the highest meal size, with a mean of 130 grams (4.6 ounces) and a 90th percentile value of 243 grams (8.6 ounces). Studies of recreational fishers have reported average meal sizes up to 376 grams (13.3 ounces; Burger et al. 1999), while studies of Native American tribal fish consumers also report meal sizes significantly larger than 100 grams.

Concentrations of mercury and dioxins in finfish and shellfish collected in Washington State during the last 10 years are summarized in another Technical Issue Paper (*Chemical Contaminants in Dietary Protein Sources*). Average mercury concentrations range from 0.011 to 0.24 µg/g (11 to 240 µg/kg) in Washington state finfish, and from 0.0074 to 0.067 µg/g (7.4 to 67 µg/kg) in Washington state shellfish. Mean dioxin TEQs[13] range from 0.054 to 1.7 pg/g in Washington State finfish; mean concentrations in shellfish range from 0.080 to 6.1 pg/g. These concentrations indicate that consumption of seafood results in a net health benefit.

For example, maternal consumption of 25 ounces per week (or about three 8-ounce servings per week) of seafood contaminated with methylmercury at 0.24 µg/g (the high end of the average concentration in Washington state finfish) would be expected to result in a net increase in child IQ. An estimated 5.4 IQ points would be gained due to DHA exposure, compared to a loss of up to 3.3 IQ points due to methylmercury exposure (Table 4).

Similarly, consumption of 25 ounces per week (or about three 8-ounce servings per week) of seafood contaminated with dioxins/furans at a concentration of 1.7 pg/g TEQ (the high end of the average dioxin concentration in Washington state finfish) would be expected to result in a net increase in health benefits. An estimated 31,900 lives would be saved due to a reduction in coronary heart disease, compared to a loss of up to 4,200 lives due to dioxin toxicity, for each million people exposed (Table 5).

The Expert Consultation[14] summarized a large body of information available on the health benefits and risks associated with fish consumption including:

- Consumption of fish provides energy, protein, and a range of other important nutrients, including the long-chain n-3 polyunsaturated fatty acids (long-chain n-3 PFAs). Optimal health benefits can be maximized by consuming fish with higher long-chain n-3 PFA content and lower methylmercury content. The risk of coronary heart disease mortality is significantly increased by not eating fish.

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13 The TEQ calculation is for dioxin and furan compounds only and does not include dioxin-like PCB congener data, which were not available for marine fish in the Ecology EIM database.

14 Some of the statements taken from the Expert Consultation have been edited here for clarity or brevity.

- The benefits of fish consumption, demonstrated in numerous studies across a wide range of populations, reflect the sum of benefits and risks from all of the constituents in the fish.
- Based on observed dose-response relationships and other factors, it is very unlikely that the benefits of fish consumption are explained to any large extent by the replacement of less "healthy" foods with fish. If this were the case, however, it would still represent a causal effect of fish consumption.
- Among the general adult population, consumption of fish, particularly fatty fish, lowers the risk of mortality from coronary heart disease. There is an absence of probable or convincing evidence of risk of coronary heart disease associated with methylmercury. While convincing evidence that high dioxin exposure increases the risk of cancer, there is currently insufficient evidence that typical levels of dietary dioxins (such as seen in fish and other dietary sources) increase the risk of cancer. Potential cancer risks associated with dioxins are well below established coronary heart disease benefits from fish consumption.
- When comparing the benefits of long-chain n-3 PFAs with the risks of methylmercury among women of childbearing age, maternal fish consumption lowers the risk of suboptimal neurodevelopment in their offspring compared with the offspring of women not eating fish in most circumstances evaluated.
- At levels of maternal exposure to dioxins (from fish and other dietary sources) that do not exceed the provisional tolerable monthly intake (PTMI) of 70 pg/kg body weight[15] established by Joint Expert Council (for PCDDs, PCDFs, and coplanar PCBs), neurodevelopmental risk for the fetus is negligible. At levels of maternal exposure to dioxins (from fish and other dietary sources) that exceed the PTMI, neurodevelopmental risk for the fetus may no longer be negligible.
- Among infants, young children, and adolescents, the available data are currently insufficient to derive a quantitative framework of the health risks and health benefits of eating fish. However, healthy dietary patterns that include fish consumption and are established early in life influence dietary habits and health during adult life.

FAO/WHO note that other biological endpoints have been used to measure dioxin risks that are more conservative than the gross endpoint of mortality. For example, various studies in human populations have examined the association of gestational and/or lactational exposure to dioxins and non-dioxin-like PCBs with neurobehavioral development. A problem inherent in these studies is that dioxins and PCBs (including dioxin-like and non-dioxin-like PCBs) coexist with each other (and with a host of other contaminants) in biological systems. Therefore, the relative contribution of the various contaminant components to developmental toxicity cannot be delineated.

For methylmercury, other measures of developmental effects besides IQ have been used in various studies, but the Expert Consultation chose to use IQ based on the common health endpoint for methylmercury and DHA+EPA exposure and because the dose-response relationships from multiple cohort studies had been established by multiple meta-analyses.

[15] By comparison, the USEPA RfD for 2,3,7,8-TCDD is 0.7 pg/kg/day, or roughly 210 pg/kg/month assuming a 30-day month.

b. Tsuchiya et al. (2008–2009)

Japanese and Korean populations in Washington State consume fish at higher rates than the national average. The University of Washington's Institute for Risk Analysis and Risk Communication and the Washington State Department of Health conducted a longitudinal study[16] examining the mercury exposure in women of childbearing age in these populations (Tsuchiya et al. 2008a,b, 2009). The study populations included 106 Japanese and 108 Korean women of childbearing age (18 to 45) who resided in the Puget Sound area of Washington for at least 6 months.

The women provided hair samples along with urine, blood, and toenail samples, and completed a food-frequency questionnaire that was open ended and spanned fish consumption over the previous year. The fish consumption survey was based on surveys previously conducted for several other Pacific Northwest fish-consuming populations (tribal surveys and Sechena et al., 1999). The food frequency questionnaire was a validated dietary tool used and developed by the Fred Hutchinson Cancer Research Center and was self administered by the participants of this study. As part of the fish dietary survey, participants were provided a pictorial fish booklet containing pictures with names of various fish species commonly consumed by Japanese and Koreans and seafood commonly found in the Pacific Northwest. Interview questions included frequency of consumption and servings sizes (based on fish models of fish steaks, fillets, sushi pieces, and shellfish samples). Also, participants were asked if they consumed any other fish not listed in the fish booklet.

The Japanese cohort was interviewed three times and the Korean cohort twice over a 14-month period. Participants that returned for second and third visits were administered dietary recall surveys spanning the 2-week period prior to the date of the visit. Data were obtained for 106, 90, and 85 Japanese individuals on the first, second, and third visits, respectively. The Korean group had 108 individuals participate on the first visit with 63 returning a second time. Mercury fish tissue concentrations were determined from fish commonly consumed by Japanese and Korean communities in the Puget Sound area from local Asian grocery stores. Analysis was conducted on skinless edible portions consisting of steaks or fillets.

Both Japanese and Korean respondents from this survey consume almost the same amounts of finfish (mean fish consumption of 60 grams/day for Japanese and 59 grams/day for Korean). Also, this similarity in fish consumption for Japanese and Koreans is reflected in the finfish consumption distribution with 95th percentiles being 159 grams/day for Japanese and 147 grams/day for Koreans. Differences in amounts of total fish consumption for these two fish-consuming populations are due to the Koreans consuming nearly 70 percent more shellfish on a daily basis (22.7 grams/day/person) compared to the Japanese (13.5 grams/day/person). The mean total fish consumption for Japanese (73 grams/day) and Koreans (82 grams/day) is almost identical to the 95th percentile estimates from CSFII and NHANES national fish dietary data.

Table 6 provides the fish species consumed (percent of total fish consumption) for the Japanese population. The list of fish species most consumed by the Japanese group did not change much over the course of the study.

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[16] Longitudinal studies make repeated observations or take repeated measurements of the same variable(s) over time.

Table 6. Fish Species Most Consumed (>4 percent of total) for Japanese Population

Species	Fish Tissue Mercury Concentrations (µg/kg)	Percent of total consumption		
		Visit 1 (n=106)	Visit 2 (n=90)	Visit 3 (n=85)
Salmon	72	29.0	28.0	36.3
Mackerel	40	9.1	18.1	10.6
Black Cod	97	6.5	4.5	2.6
Squid	39	5.6	5.5	3.1
Light Tuna (Canned)	127	5.0	4.8	8.3
Halibut	216	4.4	0.6	3.3
Ahi	185	4.2	4.3	6.7
Cod	115	1.2	8.7	5.5
Total		65	75	76
Mean individual fish intake (grams/day		59.5	33.7	31.3

Adapted from Tsuchiya et al. 2009, Table 1.

Table 7 provides the mercury intakes (mercury fish tissue concentrations for species consumed) for the Japanese population.

Table 7. Fish Species with Greatest Mercury Intake (>5 percent of total) for Japanese Population

Species	Fish Tissue Mercury Concentrations (µg/kg)	Percent of total mercury for species consumed		
		Visit 1 (n=106)	Visit 2 (n=90)	Visit 3 (n=85)
Salmon	72	17.0	25.0	32.0
White (Albacore) Tuna (Canned)	361	8.7	5.7	2.8
Halibut	216	7.9	1.6	8.7
Ahi	185	6.5	9.7	15.1
Light tuna	127	5.3	7.3	12.8
Black cod	97	5.3	5.3	3.2
Red snapper	221	3.8	5.3	3.4
Mackerel	40	3.0	8.8	5.1
Cod	115	1.1	12.2	7.8
Total		59	81	91
Mean individual estimate mercury intake (µg/day)		7.2	2.7	2.5

Adapted from Tsuchiya et al. 2009, Table 2.

Table 8 provides the fish species consumed (percent of total fish consumption) for the Korean population.

Table 8. Fish Species Most Consumed (>4 percent of total) for Korean Population

Species	Fish Tissue Mercury Concentrations (µg/kg)	Percent of total consumption		
		Visit 1 (n=108) (all participants)	Visit 1 (n=63) (subsample)	Visit 2 (n=63) (subsample)
Squid	39	23.0	21.2	10.2
Mackerel	40	12.0	13.1	16.4
Yellow croaker	53	11.0	13.1	15.4
Salmon	72	9.1	8.2	7.8
Flounder/sole	147	6.3	6.0	9.1
Light tuna (Canned)	127	5.6	4.7	12.2
Black cod	97	4.8	5.3	0.5
Pike mackerel	30	4.3	4.6	5.1
Pollack	22	3.5	4.1	4.4
Ahi	185	2.9	2.3	4.4
Total		83	83	86
Mean individual fish intake (grams/day		59.1	71.7	29.1

Adapted from Tsuchiya et al. 2009, Table 3.

For the Koreans, the average estimated mercury exposures for the first visit were 5.3 µg/day (for all 108 participants) and 5.1 µg/day (for the subsample of 63 participants) (Table 9). The average estimated mercury exposure for the second visit was 2.6 µg/day. There was a significant difference in total fish consumption estimates between those participants who returned for a second visit and those who did not. However, the percentage that each species contributed to the total fish consumption estimates was similar between visits.

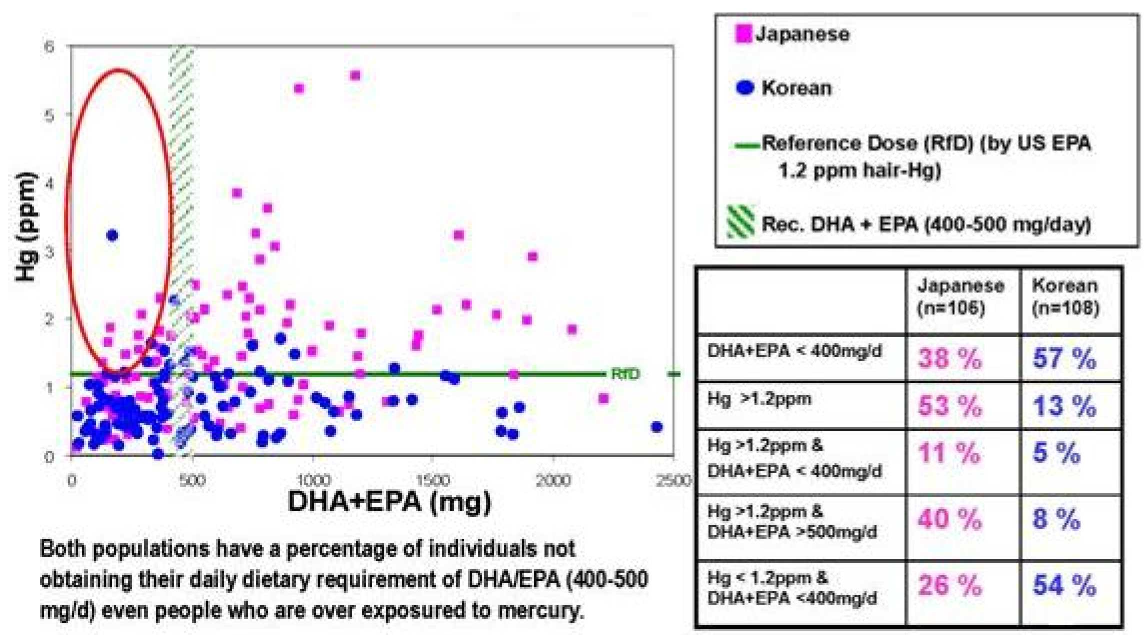
Table 9. Fish Species with Greatest Mercury Intake (>5 percent of total) for Korean Population

Species	Fish Tissue Mercury Concentrations (µg/kg)	Percent of total mercury for species consumed		
		Visit 1 (n=108) (all participants)	Visit 1 (n=63) (subsample)	Visit 2 (n=63) (subsample)
White (Albacore) tuna (Canned)	361	14.0	12.7	15.2
Flounder/sole	147	10.0	12.6	14.5
Squid	39	10.0	11.8	4.2
Light tuna (Canned)	127	7.9	8.6	17.0
Salmon	72	7.4	8.5	6.1
Yellow croaker	53	6.4	9.9	9.1
Ahi	185	4.9	6.0	9.1
Mackerel	40	4.5	7.5	6.6
Total		65	78	82
Mean individual estimate mercury intake (µg/day)		5.3	5.1	2.6

Adapted from Tsuchiya et al. 2009, Table 4.

At the time of the first visit, over half of the Japanese participants were overexposed to mercury, based on hair-mercury levels exceeding the established RfD of 1.2 mg/kg. For the Koreans, approximately 13 percent of the participants were overexposed to mercury. This is because the average mercury content of the species consumed was lower in this group. Japanese participants who were overexposed had three times higher hair mercury levels and had fish consumption rates1.5 to 2 times higher than those not overexposed. Overexposed Korean participants had hair mercury levels 2.3 to 3 times higher than those not overexposed. For both cohorts, those not overexposed consumed fish at a rate of 40–60 grams/day. After the first-year interviews were completed, participants were given education materials and advised to switch to alternative fish species with lower mercury concentrations. However, the fish species responsible for most of the mercury body burden in both populations did not change over time.

The researchers also estimated n-3 PFA intake for the study participants. This was done using DHA+EPA concentrations available in the literature for each species consumed, and then deriving DHA+EPA intake values for each participant using their species-specific intake rates. Figure 1 shows n-3 PFA intake plotted against hair mercury concentrations.



	Japanese (n=106)	Korean (n=108)
DHA+EPA < 400mg/d	38 %	57 %
Hg >1.2ppm	53 %	13 %
Hg >1.2ppm & DHA+EPA < 400mg/d	11 %	5 %
Hg >1.2ppm & DHA+EPA >500mg/d	40 %	8 %
Hg < 1.2ppm & DHA+EPA <400mg/d	26 %	54 %

Figure 1. n-3 PFA intake and mercury exposure (hair-mercury) compared to the mercury RfD and recommended daily allowance of DHA=EPA

Source: Modified from Faustman 2011, based on Tschuyia et al. 2008a.

Note that some participants exceeded the mercury RfD and still did not meet the recommended dose for n-3 PFAs. The authors conclude that both mercury and n-3 PFA content of fish are important in developing consumption guidance. In addition, the study found a 100 percent difference in the fish intake between the open-ended and 2-week recall surveys and that the open-ended survey data better represented mercury intake as determined from hair mercury concentrations.

c. Institutes of Medicine (2007)

The National Oceanic and Atmospheric Administration, which regulates U.S. marine fisheries, asked the Institutes of Medicine (IOM) of the National Academies to convene a committee to examine the scientific evidence on the nutritional benefits and risks from seafood. The IOM committee was charged to:

- identify and prioritize adverse health effects from both naturally occurring and introduced toxicants in seafood;
- assess evidence on availability of specific nutrients in seafood compared to other food sources;
- determine the impact of modifying food choices to reduce intake of naturally occurring and introduced toxicants on nutrient intake and nutritional status within the U.S. population;
- develop a decision path for U.S. consumers to balance their seafood choices to obtain nutritional benefits while minimizing exposure risks; and
- identify data gaps and recommend future research.

The IOM committee concentrated on issues affecting marine species and published its final report in 2007 (Nesheim and Yaktine 2007).

For the benefit/risk analysis, the IOM committee evaluated changes in benefits and risks associated with changing consumption patterns that may occur as a result of guidance provided to consumers regarding fish consumption. The IOM committee examined the impact of food-choice trade-offs involving calories, saturated fat, EPA/DHA, selenium, and iron. Contaminants evaluated, due to availability of data, were methylmercury, dioxins, and dioxin-like compounds. This information was used to develop guidance on seafood consumption tailored to four distinct population groups (Figure 2): **(**1) females who are or may become pregnant or who are breastfeeding; (2) children up to age 12; (3) adolescent males, adult males, and females who will not become pregnant; (4) adult males and females who are at risk of cardiovascular disease.

Seafoodhealthfacts.org, a joint project of Oregon State University, Seafood Consumer Center, Cornell University, and the Universities of California, Delaware, Florida, and Rhode Island, modified one of the sample public risk communication graphics from the IOM report when it created a seafood health reference guide for healthcare providers (Seafoodhealthfacts.org, undated). The Seafoodhealthfacts.org figure includes the FDA action level for mercury, which puts the mercury content of the various seafood species into context (Figure 3). This figure shows estimated n-3 PFA intake (as EPA + DHA) in a 3-ounce serving of fish against the average mercury levels for each species. Those species considered to be low risk for mercury exposure are colored green. Yellow species have high average mercury levels that exceed or approach the FDA action level of 1 part per million (ppm, or mg/kg) and should be avoided by sensitive groups (women who may become pregnant, pregnant and breastfeeding women, and young children) according to recommendations by several groups (USFDA, USEPA, IOM, etc.). The blue colored species (canned albacore tuna) is considered a good source of n-3 PFAs but consumption should be limited due to mercury levels in this species.

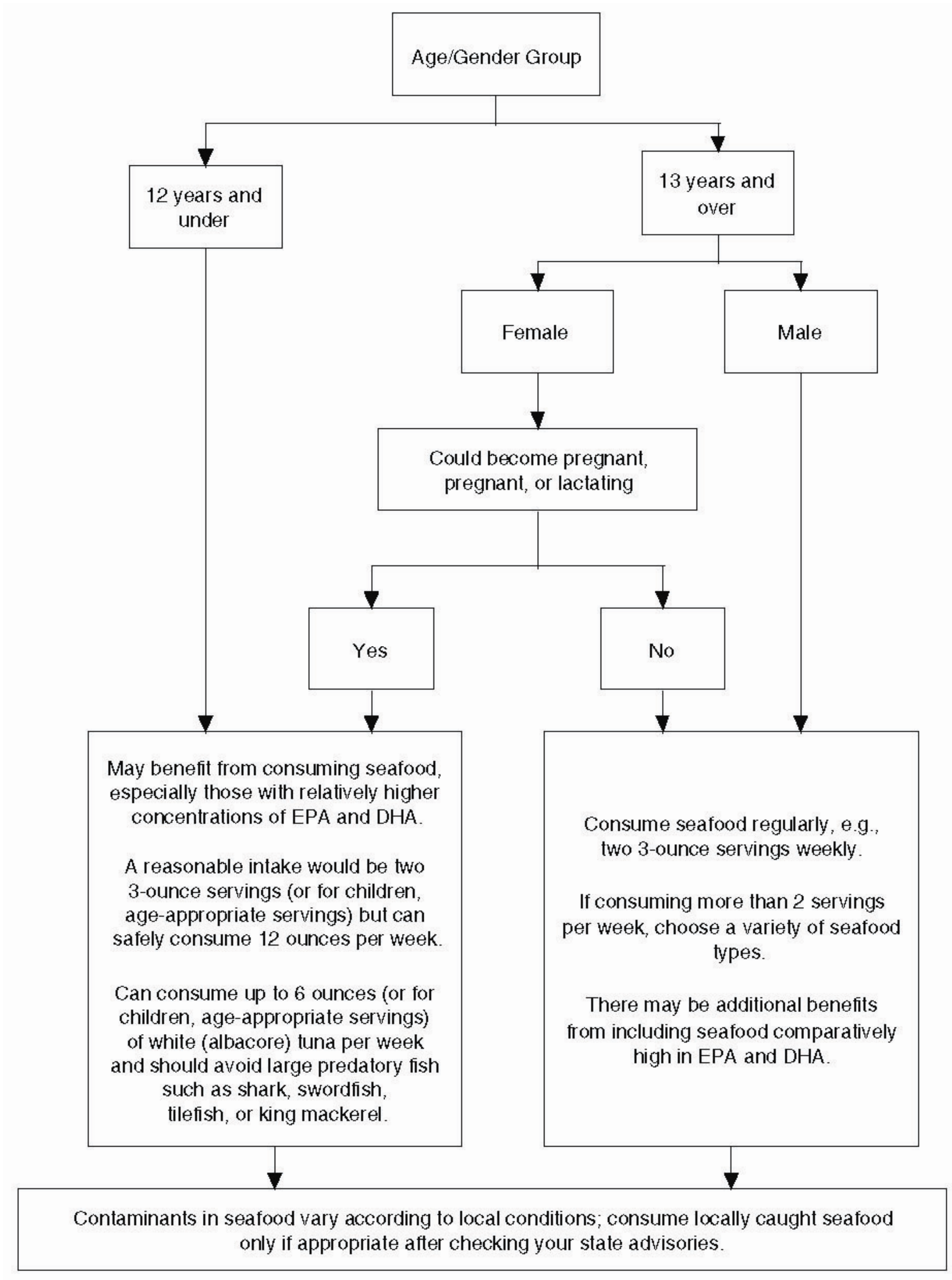


Figure 2. Decision pathway representing the balance between benefits and risks associated with seafood consumption

Source: Nesheim and Yaktine 2007

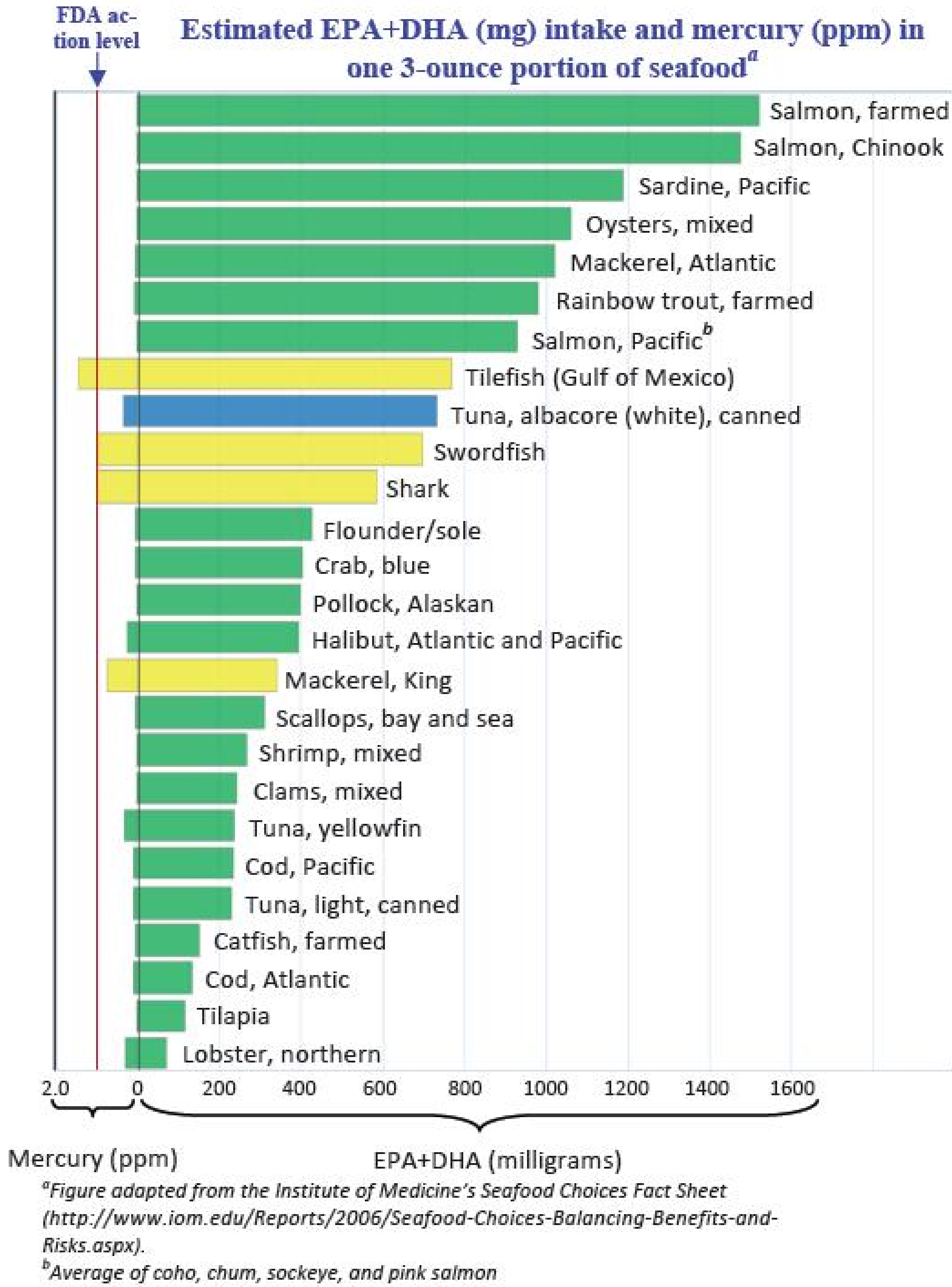


Figure 3. EPA+DHA and mercury content in representative seafood portions

Source: Seafoodhealthfacts.org.

d. Mariën and Patrick (2001)

In 2001, the Washington State Department of Health conducted a methylmercury exposure analysis for five populations of fish consumers in Washington: freshwater recreational anglers, saltwater recreational anglers, and members of the Tulalip, Squaxin Island, and Suquamish Indian Tribes. Fish-consumption data for freshwater recreational anglers were taken from anglers fishing at Lake Roosevelt. This lake is bordered by two tribal reservations (Coleville and Spokane) and is visited by over 1 million people annually. Data for saltwater anglers were obtained from surveys of Puget Sound anglers, including both shoreside anglers and boating anglers. Fish consumption rates for the three tribes were obtained from surveys. Mercury data were obtained from existing datasets for the fishery that each specific population was using.

The authors calculated a Tolerable Daily Limit (TDI) to compare to estimated methylmercury intakes determine if any the five consumption groups were at risk for overexposure to methylmercury. The TDI represents the daily intake of methylmercury that is unlikely to cause any adverse health effects. The WDOH TDI was based upon the results of three long-term cohort studies of the neurological effects in children whose mothers were exposed to methylmercury in their diet – the Faroe Islands, New Zealand, and Seychelles Islands studies. The TDI developed by WDOH fell within the range of values from 0.035 to 0.08 microgram per kilogram of body weight per day (µg/kg/day). Note that the upper end of this range is close to the USEPA RfD of 0.1 µg/kg/day.

Freshwater Anglers

The daily intake of mercury for walleye, kokanee, rainbow trout, and smallmouth bass was calculated as the product of the number of meals per month when that species was consumed, the usual number of fillets consumed at a meal, the average weight of a fillet of that species, and the average fish tissue mercury concentration for that species. The total daily intake was then estimated for each person by summing the estimated intakes due to each of the four species. This value was then converted to units of µg/kg/day assuming an average adult body weight of 70 kg. Of the 343 individual anglers surveyed, most (298 individuals, or 87 percent) had estimated mercury intake levels below the upper bound of the TDI. Nearly all of the individuals exceeding the TDI were adult males over the age of 50, who are not considered a population of concern (women of childbearing age). Assuming the spouses of these more highly exposed older males had similar intakes and were of similar age, they also would not represent a population of concern.

Saltwater Anglers

For saltwater anglers, populations with the highest consumption rates for a particular fish were categorized and mercury intake levels for each population were calculated. Estimated daily mercury intake for Puget Sound shore anglers ranged from about 0.002 µg/kg/day (tomcod and Pacific Hake) to as high as 0.023 µg/kg/day (Pacific cod). Boat angler consumption data were calculated for individuals consuming fish from three locations in Puget Sound. Estimated mercury intake for the boat anglers ranged from 0.003 µg/kg/day (rock sole) to 0.0095 µg/kg/day (Pacific cod). Total mercury intake levels for all fish species among both shore and boat anglers was less than the TDI even though the intake calculations assumed fish were consumed throughout the year. Species availability for most species is not year round due to fishing regulations that restrict when certain species can be caught. The authors separately evaluated mercury exposure for anglers consuming salmon using tissue data that had been unavailable when the original surveys were conducted. Using median values for coho and Chinook salmon (0.04 and 0.1 mg/kg, respectively), intake values were calculated as 0.02 and 0.09 µg/kg/day, respectively. Anglers consuming Chinook at rates greater than 0.8 g/kg/day would have exposures slightly exceeding the TDI.

Tulalip, Squaxin, and Suquamish Tribes

For the tribal exposures, the 90th percentile fish consumption rate for each category (anadromous, pelagic, bottom) and species of fish were combined with Washington Department of Wildlife median mercury fish concentrations to determine intake levels (Table 10). Salmonid consumption was the primary cause for intake values exceeding the TDI. The TDI was exceeded by 8 to 14 percent of the Tulalip Tribe population and 10 to 25 percent of the Squaxin Island Tribe population when consuming salmon only. The percent of the Suquamish Tribe population exceeding the TDI could not be determined from the available data although 25 percent of the surveyed population was at or above the TDI.

Table 10. Estimated mercury intake from fish consumption for Tulalip Tribes, Squaxin Island Tribe, and Suquamish Tribes

Finfish group consumed	*N*	Consumption in g/kg/day (90th% population values)	Fish mercury level median of composite means (mg Hg/kg fish)	Intake (μg Hg/kg/day)[a]
Tulalip Tribes				
Anadromous	73	1.429	0.1 (chinook salmon)	0.14
			0.05 (coho salmon)	0.07
Anadromous	73	0.63 (86th percentile)[c]	0.1 (chinook salmon)	0.06
			0.05 (coho salmon)	0.03
Pelagic	73	0.156	0.29 (quillback rockfish)	0.05
			0.17 (copper rockfish)	0.03
Bottom	73	0.111	0.06 (English sole)	0.01
Squaxin Island Tribe				
Anadromous	117	1.639	0.1 (chinook salmon)	0.16
			0.05 (coho salmon)	0.08
Pelagic	117	0.106	0.29 (quillback rockfish)	0.03
			0.17 (copper rockfish)	0.02
Bottom	117	0.176	0.06 (English sole)	0.01
Suquamish Tribe				
Salmon	92	1.680	0.1 (chinook salmon)	0.17
			0.05 (coho salmon)	0.08
Halibut/sole/rockfish/flounder/red snapper	76	0.392	0.29 (quillback rockfish)	0.11
			0.17 (copper rockfish)	0.07
			0.06 (English sole)	0.02
Tuna	83	0.346	0.17 (canned tuna)	0.06

a Intake values are based on the assumption that the fish type consumed from a particular group (anadromous, pelagic, bottom) is of one type only (provided in fourth column).

b Data with footnote ‘b’ not included in this revised version of the table.

c Consumption value based on 86th percentile rate for anadromous fish category.

The authors concluded that most of the individuals within the Tulalip Tribe could reduce their mercury intake below the TDI by consuming other anadromous fish as alternatives to Chinook (e.g., coho, chum, sockeye, steelhead, or pink salmon). Squaxin Island and Suquamish tribe members could also reduce their mercury intake by consuming salmonids other than Chinook. Another recommendation was that the Suquamish Tribe individuals that were consuming elevated quantities should reduce their intake of these species. The authors also noted that background mercury levels present in salmon were resulting in populations being exposed to mercury above the TDI.

e. USEPA/TERA (1999)

The USEPA and Toxicology Excellence for Risk Assessment (TERA) summarized what was known about health benefits and risks from fish consumption in a report published in 1999. The report compared the possible health risks of consuming contaminated fish while considering the potential health benefits lost by not eating fish, and proposed a framework for comparing the benefits and risks quantitatively.

Benefits described by the USEPA/TERA report included discussions of the beneficial protein, fatty acid profiles, and nutrients provided by fish and the various studies to date on CHD and heart attack, and other health endpoints which have been described in detail in Section II. At the time, however, the authors believed further study was needed to resolve whether fish consumption provided significant protection against CHD or heart attack. Contaminants considered included DDT (an organochlorine pesticide now banned in the United States) and its metabolites, methylmercury, dioxin, PCBs, chlordane (an organochlorine pesticide now banned in the United States), and chlorpyrifos (an organophosphate pesticide).

The USEPA/TERA report concluded that consuming uncontaminated fish (or at least fish that are smaller, younger, or in general less contaminated) may provide health benefits, but without the potential health risks associated with contamination. Before eating any contaminated fish, consumers should consider fish supplies from cleaner water bodies, eating smaller, less contaminated fish, and cooking and cleaning methods that reduce contaminants. The eating of such "cleaner" fish rather than more contaminated fish would maximize the net benefit of fish consumption. However, better estimations of benefits were needed for the general population and its sensitive subgroups and that better risk information is needed on the chemicals that commonly contaminate fish. The data gaps identified were sufficiently large so as to prevent any definitive conclusions from the study or any overall recommendations regarding existing fish consumption advisory programs of the United States or other countries. The authors believed further study was needed to confirm and extend these preliminary findings.

Conclusions

For those who consume meats, replacing meats with vegetable alternatives (e.g., beans) or fish is one strategy to replace saturated fats with monounsaturated fats and reduce cholesterol (Lichtenstein et al. 2006; USDA and USDHHS 2010). Increasing seafood consumption of most species will lead to increased n-3 PFA intake and reduced risk for major health conditions such cardiovascular and coronary heart disease. Health benefits associated with consumption of fish, particularly fatty fish such as salmon, are well documented.

While exposures to methylmercury and persistent organic pollutants may have negative human health impacts, there are considerable uncertainties about estimates of these health risks to the general population at levels present in commercially obtained seafood (Nesheim and Yaktine 2007).

High rate fish consumers such as certain ethnic groups (Japanese, American Indian) can accumulate mercury levels that approach or exceed reference doses (Tsuchiya et al. 2008a,b).

However, Mahaffey et al. (2008) suggest that data on methylmercury and n-3 PFA concentrations seafood can be used to guide the selection of individual fish and shellfish species that are higher in n-3 PFA and low in methylmercury content (see Figure 2), thereby reducing mercury exposure. Further, the risks from lipophilic compounds can be reduced by trimming fat, using cooking methods that reduce fat such as broiling, and by eating a variety of species.

A recurrent theme in recent reviews and analyses is that reducing fish consumption can negatively impact the health status of vulnerable populations (Dórea 2008). The evidence suggests that the fetus and infants may be among the principal beneficiaries from certain nutrients in seafood. Few data are available about the extent to which beneficial components of seafood, such as selenium and omega-3 fatty acids, might mitigate risks associated with contaminants in seafood (Nesheim and Yaktine 2007).

Where there are local instances of excessively high levels of contaminants such as PCBs in recreationally caught fish, it is important that consumers, especially those in sensitive populations, consult with their local health department before consuming locally caught fish.

References

AHA (American Heart Association). 2012. *Fish and Omega-3 Fatty Acids*. http://www.heart.org/HEARTORG/GettingHealthy/NutritionCenter/HealthyDietGoals/Fish-and-Omega-3-Fatty-Acids_UCM_303248_Article.jsp#.T39O3tn4J4o (Accessed March 25, 2012).

Birnbaum, L.S. 2005. Dioxins: are we at risk? PowerPoint presentation. National Health and Environmental Effects Research Laboratory, U.S. Environmental Protection Agency, Research Triangle Park, NC. Presented at the Grace A. Dow Memorial Library in Midland, MI, July 12, 2005.

Burger, J., W.L. Stephens, C.S. Boring, M. Kuklinski, J.W. Gibbons, and M. Gochfeld. 1999. Factors in exposure assessment: Ethnic and socioeconomic differences in fishing and consumption of fish caught along the Savannah River. *Risk Analysis* 19(3): 427-438. As cited in USEPA 2011.

Cabelli, S., S. Thurston, D. Bellinger, C. Amarasiriwardena, L. Altshul, and S. Korrick. 2009. The effects of prenatal exposure to PCBs and MeHg on memory and learning in school-aged children. *Epidemiology*. 20(6): S163-S164.

Candela, C.G., L.M.B. López, and V.L. Kohen. 2011. Importance of a balanced omega 6/omega 3 ratio for the maintenance of health: nutritional recommendations. *Nutrición Hospitalaria.* 26(2): 323-329.

Christen, W.G., D.A. Schaumberg, R.J. Glynn, and J.E. Buring. 2011. Dietary Ω-3 Fatty Acid and Fish Intake and Incident Age-Related Macular Degeneration in Women. *Archives of Ophthalmology*. 129(7): 921-929.

Cunnane, S.C., R. Chouinard-Watkins, C.A. Castellano, and P. Barberger-Gateau. In press. Docosahexaenoic acid homeostasis, brain aging and Alzheimer's disease: Can we reconcile the evidence? *Prostaglandins, Leukotrienes and Essential Fatty Acids.* Corrected proof available online May 8, 2012.

Dangour, A.D., E. Sydenham, R. Uauy, and V.A. Andreeva. 2012. Omega 3 fatty acids and cognitive health in older people. *British Journal of Nutrition.* 107(Suppl. 2): S152-S158.

Deckelbaum, R.J., and P.C. Calder. 2012. Different outcomes for omega-3 heart trials: Why? *Current Opinion in Clinical Nutrition and Metabolic Care.* 15(2): 97-98.

Diggs, D.L., A.C. Huderson, K.L. Harris, J.N. Myers, L.D. Banks, P.V. Rekhadevi, M.S. Niaz, and A. Ramesh. 2011. Polycyclic aromatic hydrocarbons and digestive tract cancers: A perspective. *Journal of Environmental Science and Health - Part C Environmental Carcinogenesis and Ecotoxicology Reviews.* 29(4): 324-357.

Djoussé, L., A.O. Akinkuolie, J.H. Wu, E.L. Ding, and J.M. Gaziano. In press. Fish consumption, omega-3 fatty acids and risk of heart failure: A meta-analysis. *Clinical Nutrition.* doi:10.1016/j.clnu.2012.05.010.

Dórea, J.G. 2008. Persistent, bioaccumulative and toxic substances in fish: Human health considerations. *Science of The Total Environment.* 400(1-3): 93-114.

Ecology. 2011. Fish Consumption Rates Technical Support Document: A Review of Data and Information About Fish Consumption in Washington. Washington State Department of Ecology. Publication No. 11-09-050, Version 1.0 (Draft). September 2011.

Eisler, R. 2000. *Handbook of Chemical Risk Assessment: Health Hazards to Humans, Plants, and Animals.* Boca Raton, FL: Lewis Publishers.

Eussen, S.R.B.M., J.M. Geleijnse, E.J. Giltay, C.J.M. Rompelberg, O.H. Klungel, and D. Kromhout. 2011. Statins modify the effects of *n*-3 fatty acids on major cardiovascular events in patients after myocardial infarction. In *The Food-Pharma interface: Consequences of combined use of functional foods and statins (Thesis).* Eussen, S.R.B.M. Utrecht: Utrecht University. 86-96.

FAO/WHO. 2011. Report of the Joint FAO/WHO Expert Consultation on the Risks and Benefits of Fish Consumption: Rome, 25-29 January 2010. FAO Fisheries and Aquaculture Report No. 978. A joint publication of the Food and Agriculture Organization of the United Nations and the World Health Organization, Rome. 2011.

Faustman, E.M. What's the public health issue, and why is it important? Presentation. December, 12. Dr. Elaine Faustman, University of Washington School of Public Health.

Gerber, M. 2012. Omega-3 fatty acids and cancers: A systematic update review of epidemiological studies. *British Journal of Nutrition.* 107(Suppl. 2): S228-S239.

Gopinath, B., G. Burlutsky, P. Mitchell, D.C. Harris, and V.M. Flood. 2011. Consumption of long-chain n-3 PUFA, -linoleic acid and fish is associated with the prevalence of chronic kidney disease. *British Journal of Nutrition.* 105(9): 1361-1368.

Gopinath, B., V.M. Flood, E. Rochtchina, C.M. McMahon, and P. Mitchell. 2010. Consumption of omega-3 fatty acids and fish and risk of age-related hearing loss. *The American Journal of Clinical Nutrition.* 92(2): 416-421.

Lahiri, M., C. Morgan, D.P.M. Symmons, and I.N. Bruce. 2012. Modifiable risk factors for RA: Prevention, better than cure? *Rheumatology.* 51(3): 499-512.

Lichtenstein, A.H., L.J. Appel, M. Brands, M. Carnethon, S. Daniels, H.A. Franch, B. Franklin, P. Kris-Etherton, W.S. Harris, B. Howard, N. Karanja, M. Lefevre, L. Rudel, F. Sacks, L. Van Horn, M. Winston, J. Wylie-Rosett, and C. American Heart Association Nutrition. 2006. Diet and lifestyle recommendations revision 2006: a scientific statement from the American Heart Association Nutrition Committee. *Circulation.* 114(1): 82-96.

Lim, L.S., P. Mitchell, J.M. Seddon, F.G. Holz, and T.Y. Wong. 2012. Age-related macular degeneration. *The Lancet.* 379(9827): 1728-1738.

Mahaffey, K.R., R.P. Clickner, and R.A. Jeffries. 2008. Methylmercury and omega-3 fatty acids: Co-occurrence of dietary sources with emphasis on fish and shellfish. *Environmental Research.* 107(1): 20-29.

Mahaffey, K.R., E.M. Sunderland, H.M. Chan, A.L. Choi, P. Grandjean, K. Mariën, E. Oken, M. Sakamoto, R. Schoeny, P. Weihe, C.H. Yan, and A. Yasutake. 2011. Balancing the benefits of n-3 polyunsaturated fatty acids and the risks of methylmercury exposure from fish consumption. *Nutrition reviews.* 69(9): 493-508.

Mariën, K., and G.M. Patrick. 2001. Exposure analysis of five fish-consuming populations for overexposure to methylmercury. *Journal of Exposure Analysis and Environmental Epidemiology.* 11(3): 193-206.

McManus, A., M. Merga, W. Newton, and A. Trzesinski. 2010. Seafood is Nutritional Gold for Seniors. *Australasian Medical Journal.* 3(13): 855-859.

Nesheim, M.C., and A.L. Yaktine. 2007. *Seafood choices : balancing benefits and risks*. Washington, D.C.: Institute of Medicine, National Academies Press.

Obermann-Borst, S.A., M. Vujkovic, J.H. de Vries, M.F. Wildhagen, C.W. Looman, R. de Jonge, E.A.P. Steegers, and R.P.M. Steegers-Theunissen. 2011. A maternal dietary pattern characterised by fish and seafood in association with the risk of congenital heart defects in the offspring. *BJOG: An International Journal of Obstetrics & Gynaecology.* 118(10): 1205-1215.

Oken, E., J.S. Radesky, R.O. Wright, D.C. Bellinger, C.J. Amarasiriwardena, K.P. Kleinman, H. Hu, and M.W. Gillman. 2008. Maternal fish intake during pregnancy, blood mercury levels, and child cognition at age 3 years in a US cohort. *American Journal of Epidemiology.* 167(10): 1171-1181.

Orchard, T.S., X. Pan, F. Cheek, S.W. Ing, and R.D. Jackson. 2012. A systematic review of omega-3 fatty acids and osteoporosis. *British Journal of Nutrition.* 107(Suppl. 2): S253-S260.

Ortega, R.M., E. Rodriguez-Rodriguez, and A.M. Lopez-Sobaler. 2012. Effects of omega 3 fatty acids supplementation in behavior and non-neurodegenerative neuropsychiatric disorders. *British Journal of Nutrition.* 107(Suppl. 2): S261-S270.

Patterson, R.E., S.W. Flatt, V.A. Newman, L. Natarajan, C.L. Rock, C.A. Thomson, B.J. Caan, B.A. Parker, and J.P. Pierce. 2011. Marine fatty acid intake is associated with breast cancer prognosis. *The Journal of Nutrition.* 141(2): 201-206.

Rosell, M., A.M. Wesley, K. Rydin, L. Alfredsson, and L. Klareskog. 2009. Dietary fish and fish oil and the risk of rheumatoid arthritis. *Epidemiology.* 20(6): 896-901.

Scherer, A.C., A. Tsuchiya, L.R. Younglove, T.M. Burbacher, and E.M. Faustman. 2008. Comparative analysis of state fish consumption advisories targeting sensitive populations. *Environmental Health Perspectives.* 116(12): 1598-1606.

Seafoodhealthfacts.org. Undated. Seafood for health: information for healthcare providers. A joint project of Oregon State University, Seafood Consumer Center, Cornell University, and the Universities of California, Delaware, Florida, and Rhode Island. Undated, accessed on April 23, 2012. http://www.seafoodhealthfacts.org/pdf/final-seafood-health-reference-guide-for-professionals.pdf.

Sechena, R., C. Nakano, S. Liao, N. Polissar, R. Lorenzana, S. Truong, and R. Fenske. 1999. Asian and Pacific Islander Seafood Consumption Study in King County, Washington. EPA 910/R-99-003. May 1999. http://www.epa.gov/region10/pdf/asian_pacific_islander_seafood_consumption_1999.pdf.

Smiciklas-Wright, H., D.C. Mitchell, S.J. Mickle, A.J. Cook, and J.D. Goldman. 2002. Foods commonly eaten in the United States 1994-1996: Quantities consumed per eating occasion and in a day. U.S. Department of Agriculture. Agricultural Research Center NFS Report No. 96-5, 264 pp. As cited in USEPA 2011.

Souied, E.H. 2012. Nutritional supplements and age-related macular degeneration – Focus on Omega-3 fatty acids. *European Journal of Ophthalmology.* 6(1): 43-47.

Tsuchiya, A., T.M. Burbacher, E.M. Faustman, T.A. Hinners, and K. Mariën. 2008a. Mercury exposure from fish consumption within the Japanese and Korean communities. *Journal of Toxicology and Environmental Health - Part A: Current Issues.* 71(15): 1019-1031.

Tsuchiya, A., T.M. Burbacher, E.M. Faustman, J. Hardy, and K. Mariën. 2008b. Fish intake guidelines: Incorporating n-3 fatty acid intake and contaminant exposure in the Korean and Japanese communities. *American Journal of Clinical Nutrition.* 87(6): 1867-1875.

Tsuchiya, A., F. Krogstad, T.M. Burbacher, E.M. Faustman, T.A. Hinners, J.W. White, and K. Mariën. 2009. Longitudinal mercury monitoring within the Japanese and Korean communities (United States): Implications for exposure determination and public health protection. *Environmental Health Perspectives.* 117(11): 1760-1766.

Undeland, I., H. Lindqvist, Y. Chen-Yun, E. Falch, A. Ramel, M. Cooper, A. Gildberg, J.B. Luten, E. Stenberg, H.H. Nielsen, and E. Elvevoll. 2009. Seafood and health: what is the full story? In *The Nordic Network for Marine Functional Food (MARIFUNC)*. Luten, J.B. Wageningen, The Netherlands: Wageningen Academic Publishers. 17-87.

USEPA. 2011. Exposure Factors Handbook: 2011 Edition. U.S. Environmental Protection Agency, Office of Research and Development. EPA/600/R-090/052F. September 2011.

USEPA. 2012. Health effects of PCBs. U.S. Environmental Protection Agency. http://www.epa.gov/epawaste/hazard/tsd/pcbs/pubs/effects.htm (Accessed April 24, 2012; last updated April 3, 2012)

USEPA and TERA. 1999. Comparative dietary risks: balancing the risks and benefits of fish consumption. U.S. Environmental Protection Agency Office of Water and Toxicology Excellence for Risk Assessment (TERA). August 1999.

Vahter, M. 1994. What are the chemical forms of arsenic in urine, and what can they tell us about exposure? *Clinical chemistry*. 40(5): 679-680.

WDOH (Washington State Department of Health). 2006. Human health evaluation of contaminants in Puget Sound fish. Publication No. 334-104. Washington Department of Health, Division of Environmental Health, Office of Environmental Health Assessments, Olympia, WA. October 2006.

WDOH. 2012. WDOH Office of Shellfish and Water Protection Shellfish Program webpage. (Accessed April 11, 2012). http://www.doh.wa.gov/ehp/sf/default-sf.htm

White, S.S., and L.S. Birnbaum. 2009. An overview of the effects of dioxins and dioxin-like compounds on vertebrates, as documented in human and ecological epidemiology. *Journal of Environmental Science and Health - Part C Environmental Carcinogenesis and Ecotoxicology Reviews.* 27(4): 197-211.

Williams, C.D., B.M. Whitley, C. Hoyo, D.J. Grant, J.D. Iraggi, K.A. Newman, L. Gerber, L.A. Taylor, M.G. McKeever, and S.J. Freedland. A high ratio of dietary n-6/n-3 polyunsaturated fatty acids is associated with increased risk of prostate cancer. *Nutrition research.* 31(1): 1-8.

Wu, S., B. Feng, K. Li, X. Zhu, S. Liang, X. Liu, S. Han, B. Wang, K. Wu, D. Miao, J. Liang, and D. Fan. 2012. Fish consumption and colorectal cancer risk in humans: A systematic review and meta-analysis. *The American Journal of Medicine.* 125(6): 551-559.

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Technical Issue Paper

Chemical Contaminants in Dietary Protein Sources

Technical Issue Paper

Chemical Contaminants in Dietary Protein Sources

July 20, 2012

Toxics Cleanup Program
Washington State Department of Ecology
Olympia, Washington

Table of Contents

Page

Appendices

List of Figures and Tables

Figures

Page

Tables

Page

Acronyms & Abbreviations

ATSDR	Agency for Toxic Substances and Disease Registry
CDD	chlorinated dibenzo-*p*-dioxin
CDF	chlorinated dibenzofuran
DHHS	U.S. Department of Health and Human Services
EIM	Ecology Environmental Information Management (database)
Ecology	Washington State Department of Ecology
FAO	Food and Agriculture Organization
MTCA	Model Toxics Control Act
n-3 PFA	Omega-3 polyunsaturated fatty acid
NMFS	National Marine Fisheries Service
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
pg/g	picogram per gram
TEQ	toxic equivalent
TSD	Fish Consumption Rates Technical Support Document
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USFDA	U.S. Food and Drug Administration
WDOH	Washington Department of Health
WHO	World Health Organization

Introduction

Ecology received comments on the draft *Fish Consumption Rates Technical Support Document* (TSD) (Ecology 2011), which noted that other protein sources, such as beef, chicken, pork, and dairy products, may contain contaminants that would be considered to pose unacceptable health risks under the Model Toxics Control Act (MTCA). Some reviewers requested that the TSD address the potential health risk if people eat less fish. Others suggested that the TSD should include some information regarding the contaminant concentrations in Washington State fish and shellfish.

This Technical Issue Paper summarizes contaminant concentrations in animal protein sources (meat, eggs, dairy products, seafood), including contaminant data for Washington State fish and shellfish collected within the past 10 years, and discusses the relative contribution of dietary protein sources to human exposures associated with these contaminants. It is a targeted examination of the issues raised by review comments received on the draft TSD, and was prepared within a limited time frame. Therefore, it may not include all available information on this subject.

The health benefits and risks associated with consuming seafood are reviewed in *Technical Issue Paper: Health Benefits and Risks of Consuming Fish and Shellfish*. That Technical Issue Paper also describes the toxicity of the major contaminants commonly found in fish and shellfish.

Analysis

I. Sources of dietary protein

Protein in the diet provides a source of calories and amino acids that assist in building and preserving body muscle and tissues (USDA and DHHS 2010). Protein is found in both animal and plant foods. Animal sources include seafood, meat, poultry, eggs, and milk and other dairy products. Plant sources of protein include beans, peas, nuts, seeds, and soy products. Table 1 summarizes protein and fat content for common animal food sources, which are the focus of this Technical Issue Paper.

Over the past three decades in the United States, per capita consumption of chicken has increased, beef consumption has decreased, and pork consumption has remained fairly stable (Figure 1). Consumption of all three meat categories decreased somewhat over the period 2005 to 2011. Per capita fish consumption in the United States has increased slightly in the last decade, but as of 2010, fish and shellfish consumption is at about the level of the late 1980s (Figure 2). Per capita consumption of fish and shellfish is much lower than the consumption of chicken, beef, or pork.

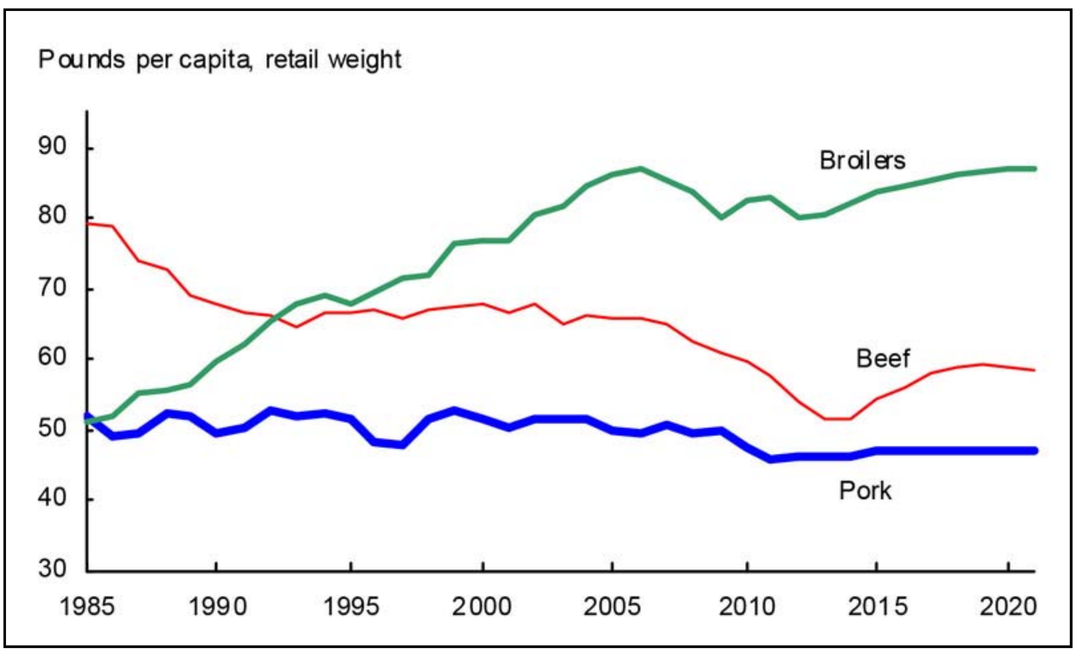

Figure 1. U.S. per capita meat consumption (historical and forecast to 2021)

Source: USDA 2012

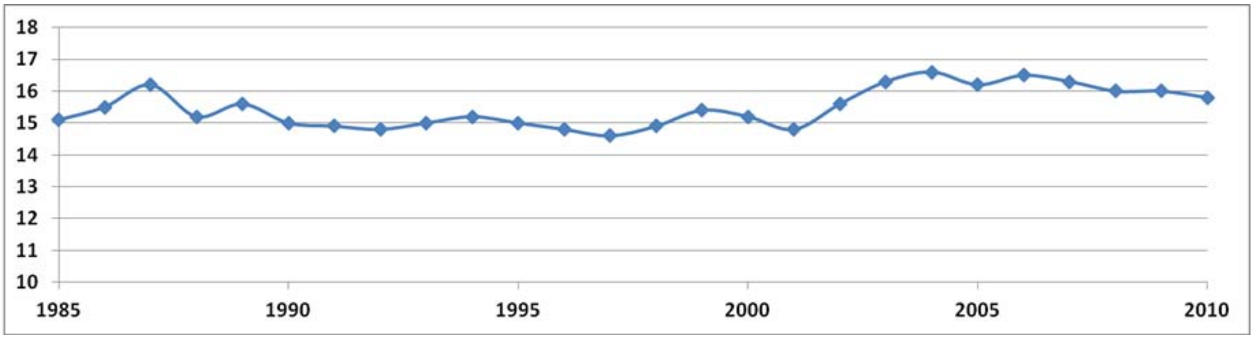

Figure 2. Annual per capita fish and shellfish consumption in the United States (pounds)

Source: NMFS 2011

Table 1. Protein and fat content of selected protein food sources

Food	Serving Measure	Weight (g)	Protein (g)	Total Fat	Fatty Acids			Cholesterol (mg)
					Saturated (g)	Mono-unsaturated (g)	Poly-unsaturated (g)	
Meats								
Beef, ground 83% lean (broiled)	3 oz	85	22	14	5.5	6.1	0.5	71
Beef, roast lean only (oven cooked)	3 oz	85	23	11	4.2	4.5	0.3	68
Lamb, leg, lean only (roasted)	3 oz	85	24	7	2.3	2.9	0.4	76
Pork, loin chops, lean only (broiled)	3 oz	85	26	7	2.5	3.1	0.5	70
Pork, cured bacon, cooked	3 slices	19	6	9	3.3	4.5	1.1	16
Pork, cured ham, lean (roasted)	3 oz	85	21	5	1.6	2.2	0.5	47
Poultry								
Chicken, breast meat, skinless (roasted)	half-breast	86	27	3	0.9	1.1	0.7	73
Chicken, drumstick, skinless (roasted)	1 drumstick	44	12	2	0.7	0.8	0.6	41
Chicken, liver (simmered)	1 liver	20	5	1	0.4	0.3	0.2	126
Turkey, dark meat (roasted)	3 oz	85	24	6	2.1	1.4	1.8	72
Turkey, ground (cooked)	1 patty	82	22	11	2.8	4	2.6	84
Turkey, light meat (roasted)	3 oz	85	25	3	0.9	0.5	0.7	59
Seafoods								
Crab, king (moist heat cooked)	3 oz	85	16	1	0.1	0.2	0.5	45
Fish, cod (cooked dry heat)	3 oz	85	20	1	0.1	0.1	0.3	40
Fish, flatfish, flounder/sole (cooked dry heat)	3 oz	85	21	1	0.3	0.2	0.5	58
Fish, rockfish (cooked dry heat)	3 oz	85	20	2	0.4	0.4	0.5	37
Fish, salmon, sockeye (cooked dry heat)	3 oz	85	23	9	1.6	4.5	2	74
Fish, salmon, Chinook (smoked)	3 oz	85	16	4	0.8	1.7	0.8	20
Fish, tuna, light, canned in water, drained	3 oz	85	22	1	0.2	0.1	0.3	26
Mollusks, clams (raw)	3 oz	85	11	1	0.1	0.1	0.2	29

Food	Serving Measure	Weight (g)	Protein (g)	Total Fat	Fatty Acids			Cholesterol (mg)
					Saturated (g)	Mono-unsaturated (g)	Poly-unsaturated (g)	
Mollusks, oyster (raw)	6 medium	84	6	2	0.6	0.3	0.8	45
Shrimp, canned, drained solids	3 oz	85	20	2	0.3	0.2	0.6	147
Dairy Products and Eggs								
Butter, salted	1 tbsp	14.2	trace	12	7.2	3.3	0.4	31
Cheese, cheddar	1 oz	28	7	9	6	2.7	0.3	30
Cheese, cottage (2% low fat)	1 cup	226	31	4	2.8	1.2	0.1	19
Cheese, mozzarella part-skim, low moisture	1 oz	28	8	5	3.1	1.4	0.1	15
Cheese, ricotta, part skim milk	1 cup	246	28	19	12.1	5.7	0.6	76
Milk, 2% reduced fat	1 cup	244	8	5	2.9	1.4	0.2	18
Yogurt, plain low fat	8 oz	227	12	4	2.3	1	0.1	14
Egg, whole	1 large	50	6	5	1.6	1.9	0.7	213

Modified from Gebhardt and Thomas 2002

http://www.ars.usda.gov/SP2UserFiles/Place/12354500/Data/hg72/hg72_2002.pdf

The serving sizes used in this table are generally standardized units of measure and may not represent the portion of food a person actually eats on one occasion.

tbsp = tablespoon; oz = ounce;

II. Contaminants of interest

A. Metals

For the purposes of this Technical Issue Paper, the two metals of interest include mercury and arsenic.

a. Mercury

The presence of mercury and mercury compounds in biological organisms is undesirable and potentially hazardous (Eisler 2000). Mercury is widespread in the environment due to natural and anthropogenic releases. Inorganic mercury is converted to the organic form methylmercury by bacteria. Methylmercury is the primary form of mercury in fish and shellfish tissues. Methylmercury is of concern due to its negative health effects (described in *Technical Issue Paper: Health Benefits and Risks of Consuming Fish and Shellfish*), and its ability to biomagnify in food chains in aquatic environments. It is most concentrated in larger and long-lived predatory species such as sharks and tuna.

b. Arsenic

Arsenic is a relatively common element present in air, water, soil, plants, and all living tissues (Eisler et al. 2000). Natural sources of arsenic are geologic, and some rocks and soils have naturally high arsenic content. Large quantities of arsenic-containing compounds (arsenicals) are released into the environment as a result of industrial and agricultural activities. Low levels of arsenic are commonly found in foods. Organic forms of arsenic may be transformed into organic forms by bacteria, fungi, algae, and plants. The primary organic form is arsenobetaine, which is rapidly excreted in the urine and generally considered non-toxic (ATSDR 2007). Arsenic (primarily the inorganic forms of arsenic) is of concern due to its negative health effects (described in *Technical Issue Paper: Health Benefits and Risks of Consuming Fish and Shellfish*).

B. Organics

Organic contaminants of interest that occur in foods include polycyclic aromatic hydrocarbons (PAHs) and dioxin-like compounds—polychlorinated dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), and biphenyls (PCBs).

a. PAHs

There are thousands of PAH compounds, and these compounds are ubiquitous in nature, largely as a result of natural processes such as forest fires and microbial synthesis. Anthropogenic sources include vehicle exhaust, manufacturing of coal tar pitch and asphalt, petroleum refining, and open burning. The largest sources of PAHs in Washington are wood burning stoves and fireplaces. Other major sources include creosote-treated wood, vehicles, leaks and improper disposal of motor oil, and small engines (e.g., lawn mowers and garden equipment). PAHs have been detected in animal and plant tissues, sediments, soils, air, surface water, drinking water, industrial effluents, river water, well water, and groundwater. PAH concentrations in marine clams and mussels tend to be highest in industrialized areas. PAH levels in fish and most crustaceans, however, are usually low because these organisms can rapidly metabolize PAHs.

Therefore, while most aquatic organisms rapidly accumulate PAHs from the ambient medium (water, sediments), PAHs show little tendency to biomagnify because most are rapidly metabolized (Eisler 2000). PAHs are of concern because several PAHs are carcinogenic, mutagenic, or teratogenic to fish and other aquatic life, amphibians, birds, and mammals, including humans.

b. Dioxins

Dioxin-like compounds include PCDDs, PCDFs, and certain PCBs (discussed below). The term *dioxins* refers to 75 congeners of PCDD and 135 congeners of PCDF (Fiedler 1998). PCDD/Fs enter the environment naturally through forest fires or volcanic activity (Eisler 2000). Human sources include municipal incinerators, pulp and paper mills that use chlorine for bleaching, and aerial application of some types of pesticides. (PCDDs are trace impurities and their presence in manufactured chemicals is not intentional or desired.) PCDD/Fs are found as mixtures of individual congeners in environmental matrices such as soil, sediment, air, and plants and lower animals (Fiedler 1998). Dioxins are of concern due to their toxicity (described in *Technical Issue Paper: Health Benefits and Risks of Consuming Fish and Shellfish*) and ability to bioaccumulate and biomagnify in food chains.

c. PCBs

PCBs are a group of 209 synthetic organic chemicals that were used as insulating and cooling agents in the electricity generating industry, and that had a number of other industrial applications (e.g., as additives to caulks and paints to improve durability). The production and sale of PCBs was banned in the United States in the late 1970s, but these compounds have persisted in the environment in soils, sediments, and organisms. PCBs are of concern due to their toxicity (described in *Technical Issue Paper: Health Benefits and Risks of Consuming Fish and Shellfish*) and ability to bioaccumulate and biomagnify in food chains.

C. Relative contribution of diet to overall exposure

a. Mercury

Dietary intake is the most important source of non-occupational exposure to mercury (ATSDR 1999). The other major source of mercury body burden is inhalation of elemental mercury volatilized from dental amalgams. Other sources are ingestion of drinking water, inhalation of vapors, medical treatments (e.g., mercury compounds used as preservatives in vaccines), and even some cosmetics and tattoo inks. Individuals living in the vicinity of former primary production or mining sites or current secondary production sites, chloralkali plants, pulp and paper mills, coal-fired power plants, or other facilities where mercury is released (e.g., municipal waste incinerators), or hazardous waste sites may be exposed to mercury through several exposure pathways, including inhalation, dermal, and oral exposures. Individuals living near municipal and medical waste incinerators, power plants fired by fossil fuels (particularly coal fired plants), or hazardous waste sites may inhale vapors or particulates contaminated with mercury from ambient outdoor air.

b. Arsenic

Estimates of inorganic arsenic intake in the U.S. adult population identified drinking water and food as the two largest sources (Figure 3) (Meacher et al. 2002; CDC 2009). Dietary exposure to inorganic arsenic occurs naturally and is unavoidable. Due to the relatively high use of water in the preparation of food, individuals in regions with higher arsenic concentrations in water likely also have higher dietary intake of inorganic arsenic (Yost et al. 2004). Background arsenic in drinking water in the United States varies by region, depending on local geology.

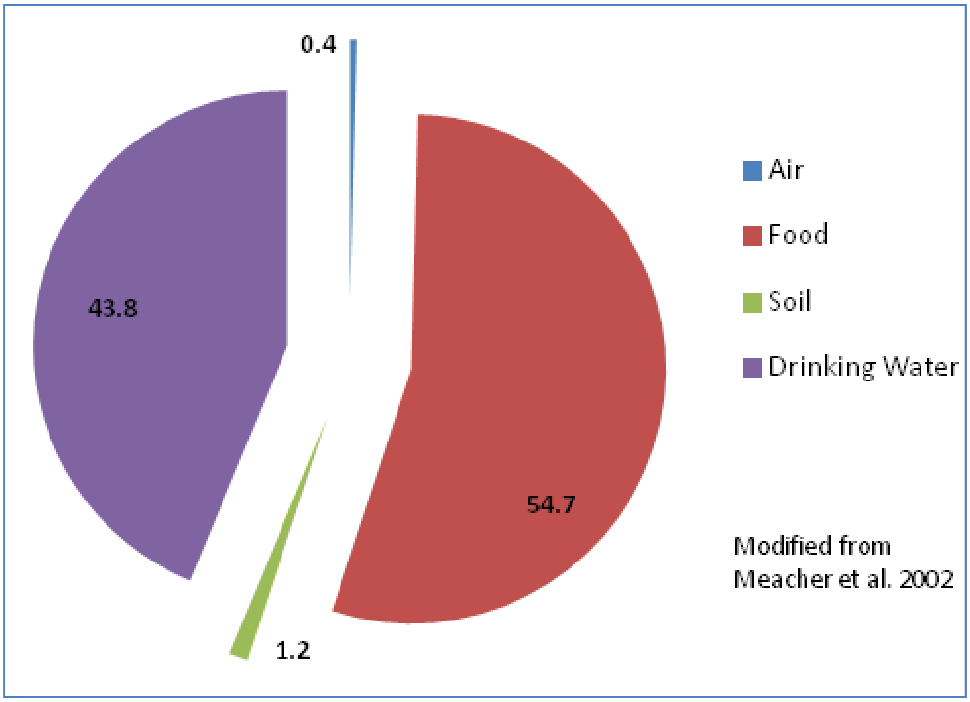

Figure 3. Relative contribution of inorganic arsenic exposure by source

c. PAHs

Food is the major contributor to total intake of PAHs in the general population, with smaller contributions from water and air (FAO/WHO 2006). Smokers and people exposed occupationally will have additional exposures to PAHs. For smokers, the PAH intake from smoking may be of similar magnitude to intake from the diet (European Food Safety Authority 2008). Most foods contain 1 to 10 µg total PAHs/kg fresh weight. Cereals and cereal products are major contributors of PAHs to the diet in many countries due to the high consumption rates of these foods. Vegetable fats and oils are also major contributors due to the high PAH concentrations in this food group. PAHs form directly during food processing (drying and smoking) and cooking (grilling, roasting, frying). Grilled and barbequed foods can have high PAH levels; smoking and barbequing fish and meats increases total PAH content by up to 100-fold. PAHs form on or near meat surfaces rather than the interior; therefore, foods cooked without being exposed to smoke do not show significant levels of PAHs (Kazerouni et al. 2001). Some population groups may have higher intakes of PAHs, for example, people who regularly cook food over open fires or barbeques (ATSDR 1995).

d. Dioxins

Inhalation, water consumption, soil ingestion, dermal contact, and vegetable fat ingestion contribute only a small percentage of overall exposure to dioxins (Birnbaum 2005). Figure 4 shows the relative contributions of dietary and other sources to dioxin toxic equivalents (TEQs). In the U.S. Environmental Protection Agency (USEPA) Dioxin Reassessment, dose estimates (that included the dioxin-like compounds PCBs as well as PCDD/Fs) were provided for inhalation, soil ingestion, soil dermal contact, water ingestion, and for 10 food ingestion categories including beef, pork, poultry, other meats (game, lamb, unidentified meat in casseroles, etc.), eggs, milk, dairy, marine fish, freshwater fish, and vegetable oils (Lorber et al. 2009). The dose estimates were dominated by ingestion of animal food products (beef, pork, poultry, other meats, dairy, eggs, milk, and fish), which comprised 93 percent of total exposures.

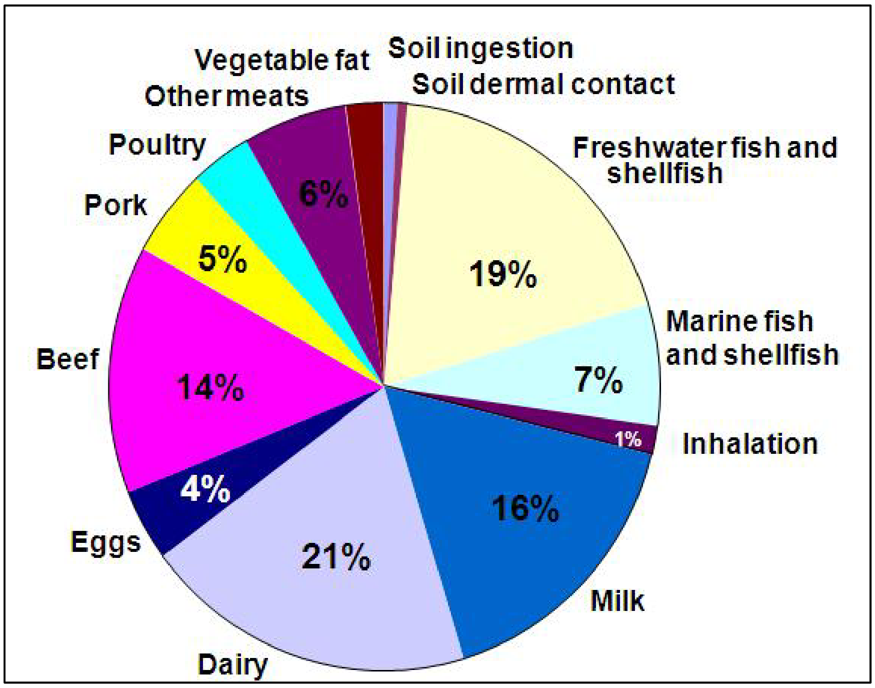

Figure 4. Sources of U.S. average daily intake of PCDD/PCDF/PCB TEQs

Source: Birnbaum 2005

D. Relative contribution of each type of protein source

a. Mercury

The U.S. Food and Drug Administration (USFDA) Total Diet Study found that nearly all of the mercury in the U.S. diet comes from meat, fish, and poultry, with fish and shellfish being the principal source (Mahaffey 2009). Methylmercury in fish is the primary route of organic mercury acquisition in humans (Dórea 2008). A United Kingdom dietary study found that fish supplied most of the mercury in the average diet (25 percent); in that study, mercury concentrations ranged from less than 5 µg/kg in meat, eggs, and milk, up to 54 µg/kg in fish (Ysart et al. 1999). However, the use of fishmeal in farm animal feeds is likely to impact human foods by increasing exposure to methylmercury (Dórea 2008). Researchers in Sweden found that mercury in animal products raised on fish meal can correlate with hair mercury levels (a widely used biomarker of fish consumption) in non-fish eaters (review in Dórea 2008). [Note that SAIC attempted to acquire published paper(s) with the metals data from the Total Diet Study but only found papers estimating intake by source; the papers did not include the actual ranges found in the food products that were used to estimate intakes (e.g., Gunderson 1995).]

b. Arsenic

Fish and shellfish contain the highest concentrations and are the largest dietary source of arsenic, generally in organic forms (Yost et al. 2004; ATSDR 2007). As noted in *Technical Issue Paper: Health Benefits and Risks of Consuming Fish and Shellfish*, organic forms such as those found in seafood are generally considered non-toxic. Yost et al. (2004) presented total and inorganic arsenic concentrations for animal-based protein sources (Figure 5; data supporting this figure are presented in Table 11). While saltwater fish and shrimp had total arsenic concentrations greatly exceeding those of other sources, the inorganic arsenic concentrations of all of the sources were small and within one order of magnitude of each other. Inorganic arsenic was not detected in tuna or milk.

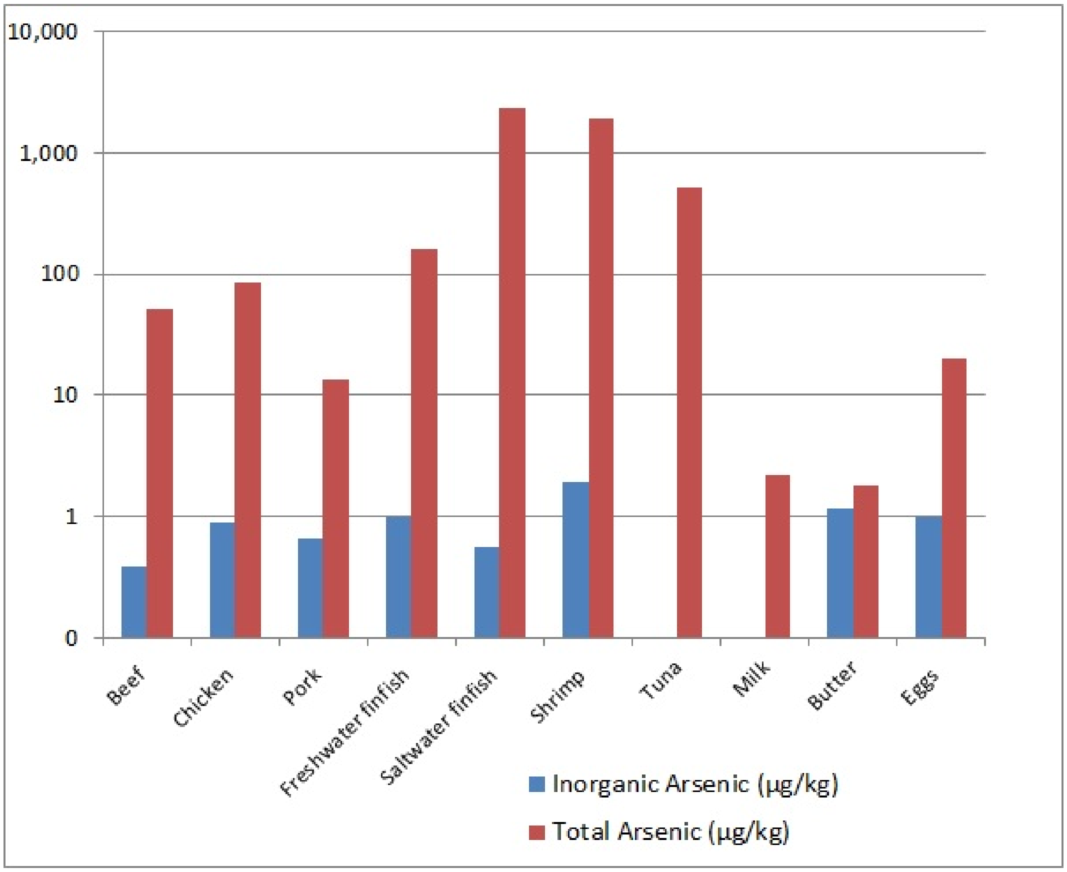

Figure 5. Total and inorganic arsenic concentrations of selected protein sources (from data in Yost et al. 2004)

Yost et al. (2004) examined intake of inorganic arsenic in children ages 1 to 6 years and found that fish and shellfish contributed less than 1 percent of the daily dietary intake (Figure 6, data table presented in Appendix A). Meat and poultry contributed slightly more (just over 1 percent) and dairy products another 14 percent. Young children were investigated because they are often considered to be a sensitive sub-population that is vulnerable to arsenic exposures from soil and subject to higher exposures on a body weight basis. Inorganic arsenic was specifically evaluated because it is considered the most toxicologically significant. Grain and grain products contributed nearly 50 percent of the inorganic arsenic in the diet of young children. Of the animal protein sources, dairy products contributed the most inorganic arsenic (14 percent), followed by meat and poultry (1.2 percent), eggs (0.5 percent), and then fish/shellfish (0.1 percent).

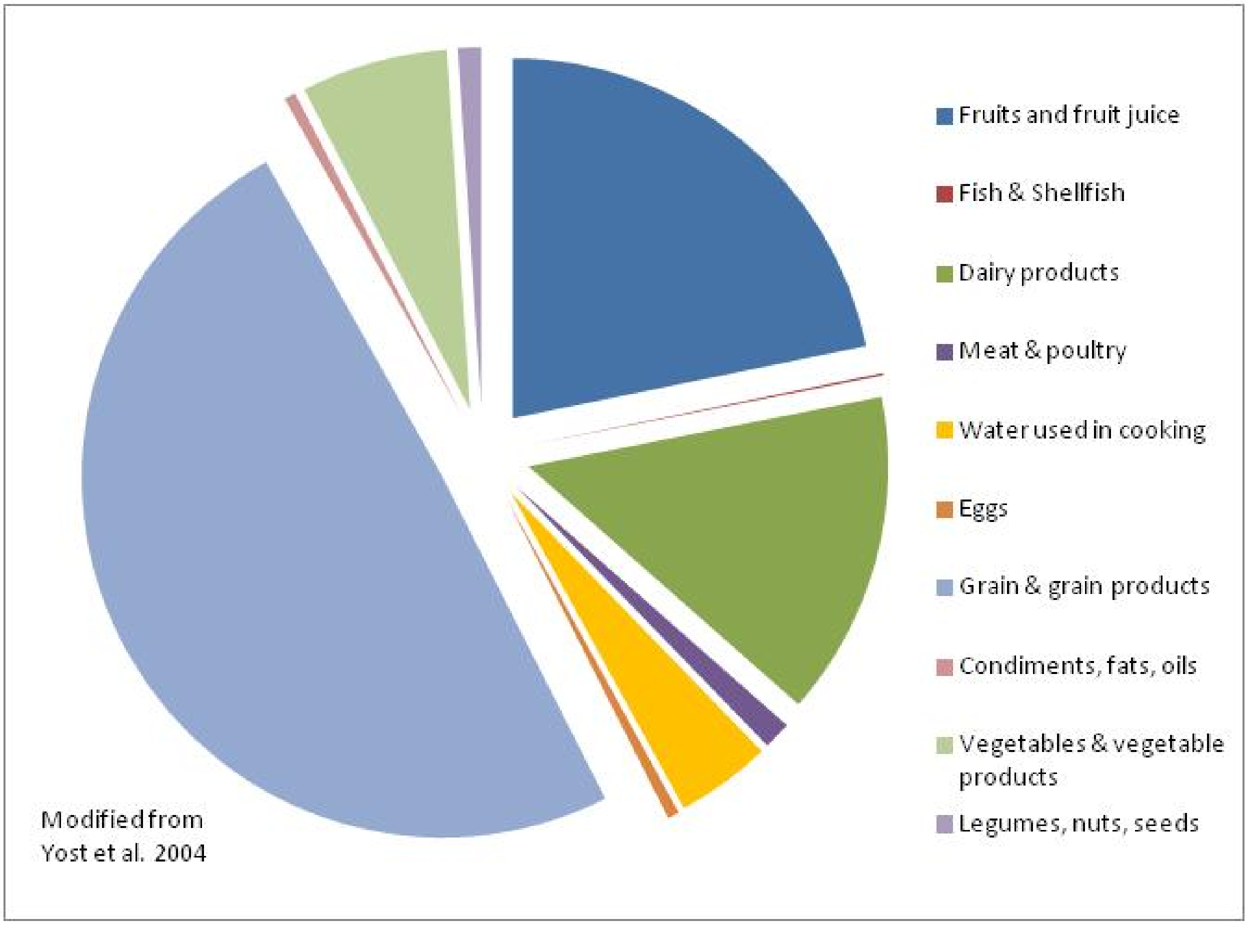

Figure 6. Sources of dietary inorganic arsenic for children 1 to 6 years (% contribution)

c. PAHs

Kazerouni et al. (2001) measured concentrations of benzo(a)pyrene in 200 food items. Table 2 summarizes some of the results for animal-based protein sources. Cooking method greatly affected the concentration of benzo(a)pyrene in foods, with grilled foods generally containing more than broiled or fried foods and very well-done grilled meats containing more than medium or well-done meats.

Table 2. Benzo(a)pyrene in animal-based protein sources

Food	Cooking Method	Portion Size (g)	Concentration (µg/kg)*	ng/portion
Hamburger	Oven-broiled - very well	85	0.01	0.8
Hamburger	Grilled - medium	85	0.09	8.0
Hamburger	Grilled – well	85	0.56	48
Hamburger	Grilled - very well	85	1.52	129
Steak	Oven-broiled - very well	112	0.01	1.0
Steak	Grilled - very well	112	4.86	544
Chicken, boneless	Oven-broiled - very well	96	0.48	46
Chicken, boneless	Grilled - well	96	0.39	37
Chicken, boneless	Grilled - very well	96	0.40	38
Crab	Not given	38	0.1	3.8
Perch fillet	Grilled - well	85	0.19	16
Pork chops	Oven-broiled - well	112	0.01	1.0
Pork chops	Pan-fried - well	112	0.01	1.0
Butter	NA	10	ND	ND
Cheese, cottage	NA	113	0.07	8.0
Eggs	NA	100	0.03	3.0
Milk, whole	NA	244	0.02	5.0
Yogurt, flavored/frozen	NA	227	0.18	41

Modified from Kazerouni et al. 2001

*ND = not detected; limit of detection was 0.005 µg/kg

The European Food Safety Authority (2008) recently presented data for 15 priority PAHs (PAH15) in food samples taken throughout Europe. The PAHs included: benz(a)anthracene, benzo(b)fluoranthene, benzo(j)fluoranthene, benzo(k)fluoranthene, benzo(ghi)perylene, benzo(a)pyrene, chrysene, cyclopenta(cd)pyrene, dibenz(a,h)anthracene, dibenzo(a,e)pyrene, dibenzo(a,h)pyrene, dibenzo(a,i)pyrene, dibenzo(a,l)pyrene, indeno(1,2,3-)pyrene, and 5-methylchrysene. The highest concentrations were reported for processed fish, molluscs, and smoked molluscs (Figure 7; data presented in Appendix A).

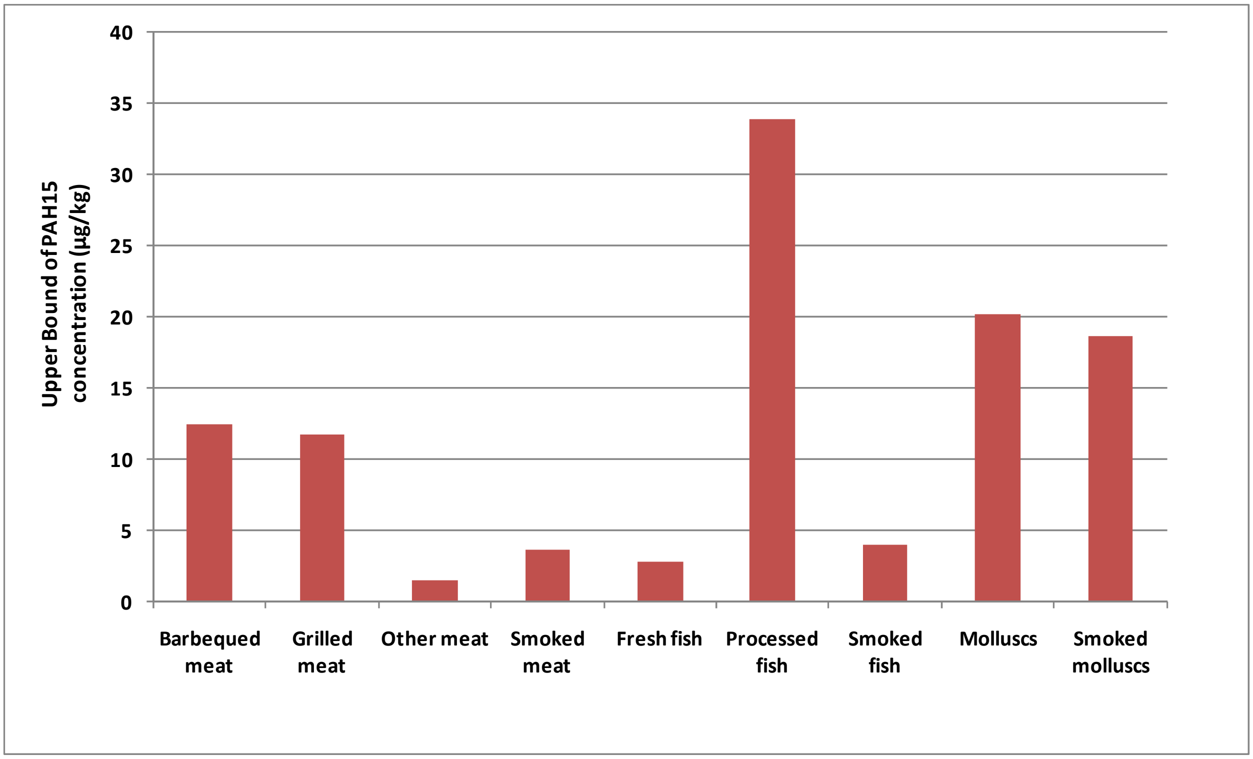

Figure 7. Upper bound PAH15 concentrations (µg/kg) by food group

Modified from European Food Safety Authority 2008

WDOH evaluated the PAH8 data in the European Food Safety Authority study (White 2012) and estimated that PAHs in shellfish could account for up to 20 percent of the PAH intake for a shellfish consumer's diet. PAH8 priority PAHs included the following carcinogenic compounds: benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(ghi)perylene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-)pyrene.

In a total daily diet intake study, Kazerouni et al. (2001) found that barbecued meats contributed the most benzo(a)pyrene, followed by dairy products, and non-barbecued meats (Figure 8).

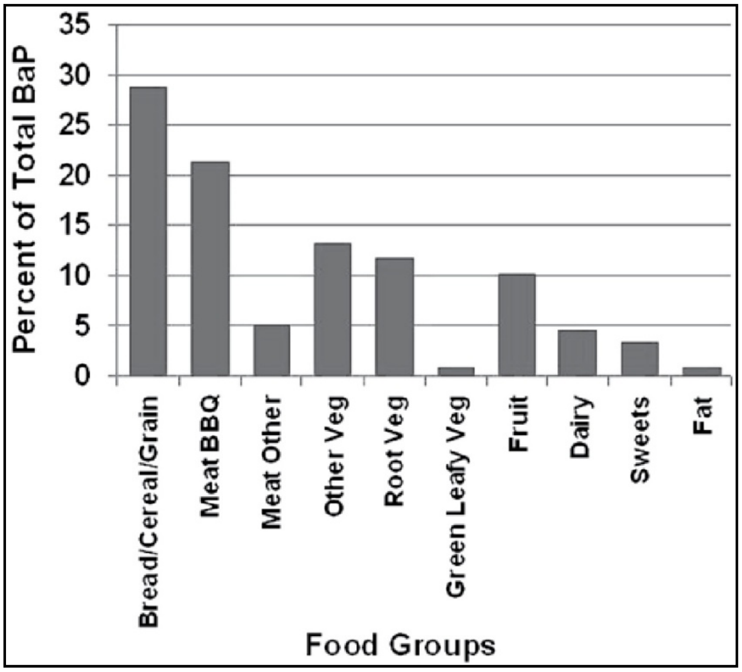

Figure 8. Percent of total daily benzo(a)pyrene intake in a control group of 228 subjects

Source: Kazerouni et al. 2001

d. Dioxin

Dougherty (2000) estimated that fish and shellfish contributed nearly 90 percent of the dioxins in the average U.S. consumer's diet while beef contributed less than 10 percent. More recently, Lorber et al. (2009) estimated the total TEQ contributions of dioxin-like compounds (including dioxin-like PCBs) from meat, seafood, egg, and dairy products in the diet (Figure 9; data presented in Appendix A). This was achieved using average contaminant concentrations in foods and food consumption rates. The contribution to TEQs from fish in the diet, as calculated by Lorber et al. (2009), was less than 32 percent.

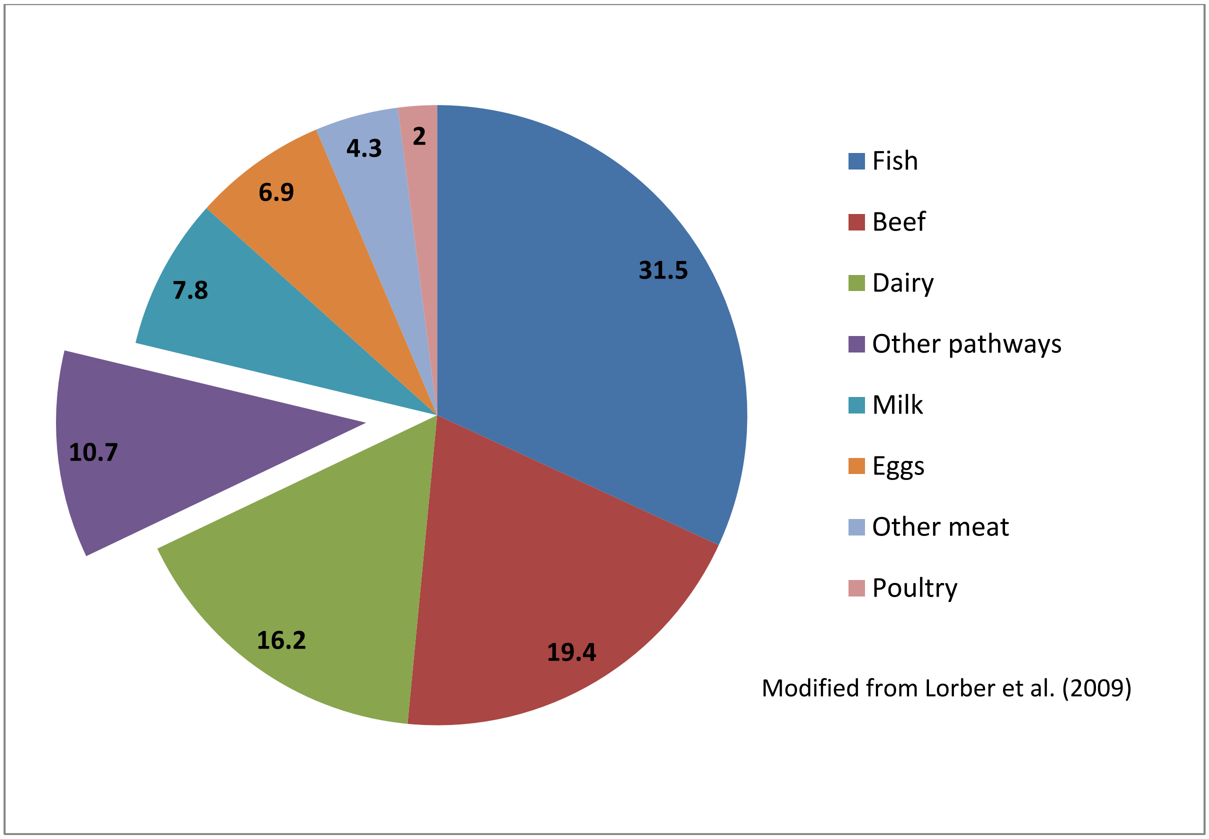

Figure 9. Exposure pathway for PCDD/PCDF/PCB TEQ intakes from protein sources (percent contribution by source)

"Other pathways" includes: consumption of water, inhalation of air, ingestion of soil, dermal contact, and vegetable fat intake.

e. PCBs

Dougherty (2000) estimated that fish and shellfish contributed nearly 100 percent of the PCBs in the average U.S. consumer's diet. However, Schecter et al. (2010), using samples of 31 foods collected in 2009 from a market-basket survey, calculated that meat sources contributed more than seafood, dairy, eggs, and vegetable products combined to daily dietary intake of PCBs (data for animal sources provided in Tables 3, 4, and 5). This may have been due to the relative quantities consumed, because PCBs were found in fish at similar concentrations (and with additional PCB congeners) than in meats. Each of the 209 PCB congeners has been assigned a congener number (e.g., PCB-52, which is 2,2',5,5'-tetrachlorobiphenyl). The specific congeners analyzed by Schecter et al. (2010) are denoted by their congener number in the tables below. Note that the representative PCBs were not detected in any meats except hamburger, in any fish except salmon and canned sardines, or in any dairy products or eggs. Researchers in Japan (Hirai et al. 2004), however, found that exposure of farm animals to coplanar PCBs through fish meal is an important source of PCBs in meat and milk. Variations in feed-grass concentrations of coplanar PCBs were insufficient for explaining levels of these congeners in meat and milk. When fish meal was included in modeling calculations, the predicted congener profile came close to actual measurements.

Table 3. Levels of marker PCBs in composite meat samples (ng/g ww or LOD)

Marker	Hamburger	Bacon	Turkey	Sausages	Ham	Chicken breast	Roast beef
Lipid Content (%)	21.7	36.1	2	23.9	4.3	4.7	4.6
PCB-52	ND (0.1)	ND (0.09)	ND (0.04)	ND (0.1)	ND (0.05)	ND (0.05)	ND (0.04)
PCB-101	ND (0.4)	ND (0.3)	ND (0.1)	ND (0.4)	ND (0.2)	ND (0.2)	ND (0.1)
PCB-118	ND (0.2)	ND (0.1)	ND (0.06)	ND (0.2)	ND (0.07)	ND (0.08)	ND (0.09)
PCB-153	1.2	ND (0.4)	ND (0.2)	ND (0.5)	ND (0.2)	ND (0.3)	ND (0.2)
PCB-138	ND (0.7)	ND (0.4)	ND (0.2)	ND (0.6)	ND (0.2)	ND (0.3)	ND (0.2)
PCB-180	0.21	ND (0.10)	ND (0.04)	ND (0.1)	ND (0.04)	ND (0.06)	ND (0.05)

ww = wet weight; ND = not detected; LOD = level of detection

Table 4. Levels of marker PCBs in composite fish samples (ng/g ww or LOD)

Marker	Salmon	Canned tuna	Fresh catfish fillet	Tilapia	Cod	Canned sardines	Frozen fish sticks
Lipid Content (%)	11.9	14.8	11.6	1.6	0.3	10.3	10.3
PCB-52	0.28	ND (0.06)	ND (0.1)	ND (0.09)	ND (0.07)	0.28	ND (0.06)
PCB-101	0.51	ND (0.2)	ND (0.3)	ND (0.3)	ND (0.2)	0.67	ND (0.2)
PCB-118	0.43	ND (0.07)	ND (0.2)	ND (0.1)	ND (0.1)	0.8	ND (0.09)
PCB-153	1.21	ND (0.3)	ND (0.5)	ND (0.4)	ND (0.3)	1.83	ND (0.3)
PCB-138	0.93	ND (0.3)	ND (0.5)	ND (0.3)	ND (0.2)	1.8	ND (0.3)
PCB-180	0.44	ND (0.06)	ND (0.1)	ND (0.07)	ND (0.06)	0.49	ND (0.07)

ww = wet weight; ND = not detected; LOD = level of detection

Table 5. Levels of marker PCBs in composite dairy and egg samples (ng/g ww or LOD)

Marker	Butter	American cheese	Other cheese	Whole milk	Yogurt	Cream Cheese	Eggs
Lipid Content (%)	91.4	25.3	30.1	3.8	2.9	34	2.9
PCB-52	ND (0.2)	ND (0.1)	ND (0.09)	ND (0.05)	ND (0.03)	ND (0.08)	ND (0.05)
PCB-101	ND (0.2)	ND (0.3)	ND (0.3)	ND (0.2)	ND (0.08)	ND (0.02)	ND (0.1)
PCB-118	ND (0.1)	ND (0.2)	ND (0.2)	ND (0.07)	ND (0.04)	ND (0.02)	ND (0.08)
PCB-153	ND (0.5)	ND (0.4)	ND (0.4)	ND (0.2)	ND (0.1)	ND (0.04)	ND (0.2)
PCB-138	ND (0.5)	ND (0.4)	ND (0.4)	ND (0.2)	ND (0.1)	ND (0.04)	ND (0.2)
PCB-180	ND (0.1)	ND (0.1)	ND (0.10)	ND (0.05)	ND (0.03)	ND (0.09)	ND (0.05)

ww = wet weight; ND = not detected; LOD = level of detection

E. Sensitive subpopulations

There are two types of populations that may be considered as *sensitive* populations when considering contaminant exposures: those that are exceptionally sensitive to a particular contaminant and those who may receive greater exposure to a contaminant due to age, gender, health status, genetic differences, and lifestyle choices such as diet.

Native tribes and other ethnic groups may eat fish/shellfish at rates significantly higher than most consumers. For example, Korean and Japanese Americans in the Puget Sound region have seafood consumption rates that are among the highest in the nation (Tsuchiya et al. 2008). As discussed in the TSD, many Native Americans in Washington State also consume more seafood than the national average. A mercury intake biometric study by Tsuchiya et al. (2008) found that one in two persons in the Japanese community was overexposed to methylmercury based on their diet and evidence from biological samples analyzed for mercury content. The Korean community, however, had about one in ten persons who exceeded the reference dose, even though total seafood consumption was about the same as the Japanese cohort. While both populations consumed a large amount of seafood, their species preferences (e.g., Japanese consumed more salmon and the Koreans consumed more squid) influenced their mercury exposure.

Children are unique in their contaminant exposures for many reasons. They drink more fluids, eat more food, and breathe more air per kilogram of body weight, and have a larger skin surface in proportion to their body volume than do adults. Their diets often differ from that of adults. Nutritional requirements change with age: from placental nourishment to breast milk or formula to the diet of older children who eat more of certain types of foods than adults. Behavior and lifestyle differences also influence a child's contaminant exposure. Children crawl on the ground, put things in their mouths that an adult would not, and sometimes eat inappropriate things such as dirt or paint chips. Children may also spend more time outdoors than adults. As one example, children drink more milk on a per body weight basis than do adults. This significantly increases their exposures to milk-borne contaminants compared to adults.

The developing fetus and young children are also sensitive subpopulations because they are more susceptible to the toxic effects of some contaminants such as methylmercury. Therefore, women of childbearing age, pregnant or nursing women, and young children are considered sensitive populations for neurodevelopmental risks from exposure to contaminants in foods. As FAO/WHO (2010) note, these same groups are also sensitive populations for neurodevelopmental risks from not consuming fish.

III. Contaminant concentrations in dietary protein sources

A. Fish and shellfish — Washington State contaminant concentrations

The following tables and figures present contaminant concentrations in Washington State seafood. Where available, data were selected that reflect different regions of the state. Due to the industrial development prevalent around Puget Sound, there are myriad studies of seafood contamination from this region. Coastal areas (e.g., Figures 10 and 11) and eastern Washington data (Spokane River in Table 8) are also represented in this Technical Issue Paper. Other data were selected because they showed regional comparisons of contaminant concentrations (e.g., Figure 14).

Appendix A provides tables of mercury, total PCBs, and dioxin TEQ concentrations that were summarized from data taken from the Ecology Environmental Information Management (EIM) database. The EIM data included marine, anadromous, and freshwater species, and statistical summaries pooled the data by the ecological group and tissue type sampled. Additional information on background levels of PCBs and dioxins in Washington State freshwater fish is available in Ecology Publication No. 10-03-007 (Ecology 2010). Data analyzed in that report that were included in the EIM database are contained within the statistical summaries in Appendix A. The tables and figures in this section represent a subset of the contaminant data available for different regions of Washington State.

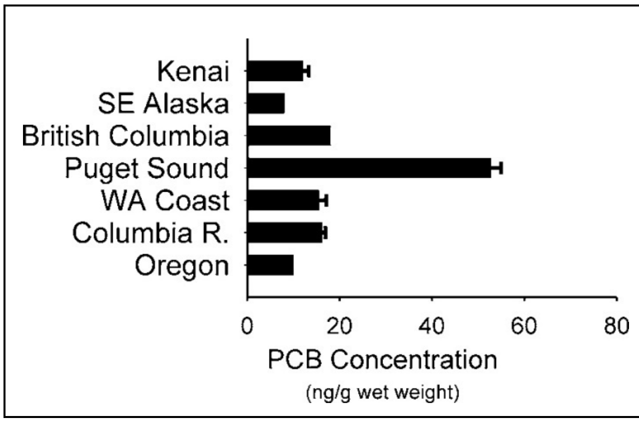

Figure 10. Mean PCB concentration in Chinook salmon fillets

Source: O'Neill and West 2009

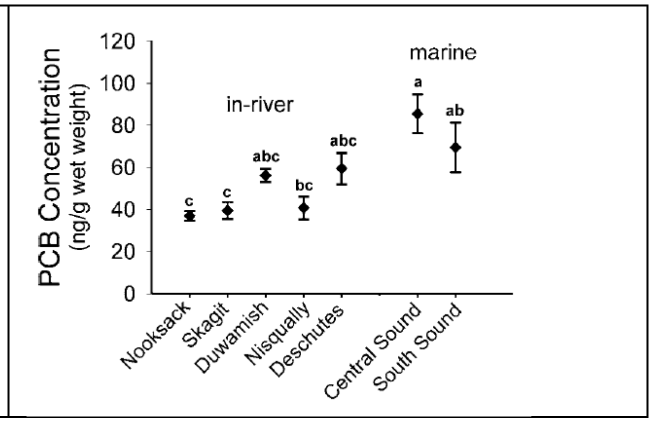

Figure 11. Mean PCB concentration in adult Chinook salmon returning to Puget Sound (in-rivers versus marine waters*)

Source: O'Neill and West 2009
Error bars represent standard error around the means.
*"In-river" samples were taken from fish collected in the rivers noted, whereas "marine" samples were taken from fish collected in marine waters of the central or south Puget Sound, as noted.

Table 6. Average PCBs in Chinook salmon from in-river versus marine areas of the Puget Sound basin

Location Type	Puget Sound basin	Location	*N*f	*N*fa	% SWA1	*N*s	PCBs* (µg/kg)	% Lipids	Fish Age (years)	SW Age (years)	Length (mm)
In-river	North	Nooksack River	133	120	3.3	28	37	3.45	3.6	2.5	741
	North	Skagit River	125	114	3.5	29	40	4.83	4.1	2.6	816
	Central	Duwamish River	171	159	12.6	65	56	7.34	3.8	2.4	763
	South	Nisqually River	92	90	5.6	20	41	3.76	3.4	2.3	732
	South	Deschutes River	113	77	0.0	34	59	1.74	3.9	2.4	789
		All in-river sites	634	560	5.9	176	49	4.82	3.8	2.4	770
Marine	Central	Central sound	60	60	76.7	12	86	5.74	2.8	1.3	599
	South	South sound	69	68	2.9	16	69	4.15	3.5	2.3	747
		All marine sites	129	128	37.5	28	76	4.83	3.2	1.9	683
Total		All sites	763	688	11.8	204	53	4.82	3.7	2.3	758

Modified from O'Neill and West 2009

* wet weight basis

Nf= number of salmon collected; Nfa = number of fish in the sample whose age was estimated (i.e., “aged fish”); %SWA1 = percentage of aged fish that spent one winter in saltwater; Fish Age = average age of the fish in the sample;
SW Age = average salt water age of sample; length is the average length of the fish in the sample

Table 7. Maximum concentration of contaminants detected in fish and shellfish sampled at Fidalgo Bay in Anacortes, Washington

Chemicals	Contaminant maximum concentration (ppm)					
	Horse Clams	Manila Clams	Bent nose Clams	Macoma Clams	Starry Flounder	English Sole
Arsenic total	2.82	3.14	3.84	n/a	1.35	3.1
Mercury	0.009	0.03	0.02	n/a	0.071	0.0077
Total Dioxin TEQ	7.0E-8	1.2E-7	7.6E-7	6.9E-7	1.75E-7	1.68E-7
	Red Rock Crab			Dungeness Crab		
	Tissue	Other*	Hepato-pancreas	Tissue	Other*	Hepato-pancreas
Arsenic total	5.95	4.69	7.31	10.47	6.81	6.54
Mercury	0.058	0.026	0.060	0.088	0.051	0.049
Total Dioxin TEQ	1.8E-7	1.53E-6	6.57E-6	1.44E-7	5.82E-6	1.05E-5

Source: WDOH 2010 Health Consultation for Fidalgo Bay

Total Dioxin TEQ – sum of dioxin/furans toxic equivalent (TEQ)

PPM – parts per million, na – not available

* Other soft tissues (viscera)

Table 8. Mean total PCB concentrations for fish from the Spokane River, Spokane, WA

Location	Species	Total mean PCBs (μg/kg, ww)
Fillet samples		
Plante Ferry	Rainbow Trout	55
Mission Park	Rainbow Trout	153
Mission Park	Mountain Whitefish	234
Ninemile	Rainbow Trout	73
Ninemile	Mountain Whitefish	139
Upper Long Lake	Mountain Whitefish	43
Upper Long Lake	Brown Trout	130
Upper Long Lake	Smallmouth Bass	37
Lower Long Lake	Mountain Whitefish	76
Lower Long Lake	Smallmouth Bass	67
Whole body samples		
Stateline	Largescale Sucker	56
Plante Ferry	Largescale Sucker	122
Mission Park	Largescale Sucker	1,823
Ninemile	Bridgelip Sucker	69
Upper Long Lake	Largescale Sucker	327
Lower Long Lake	Largescale Sucker	254

Source: WDOH 2007; ww - wet weight

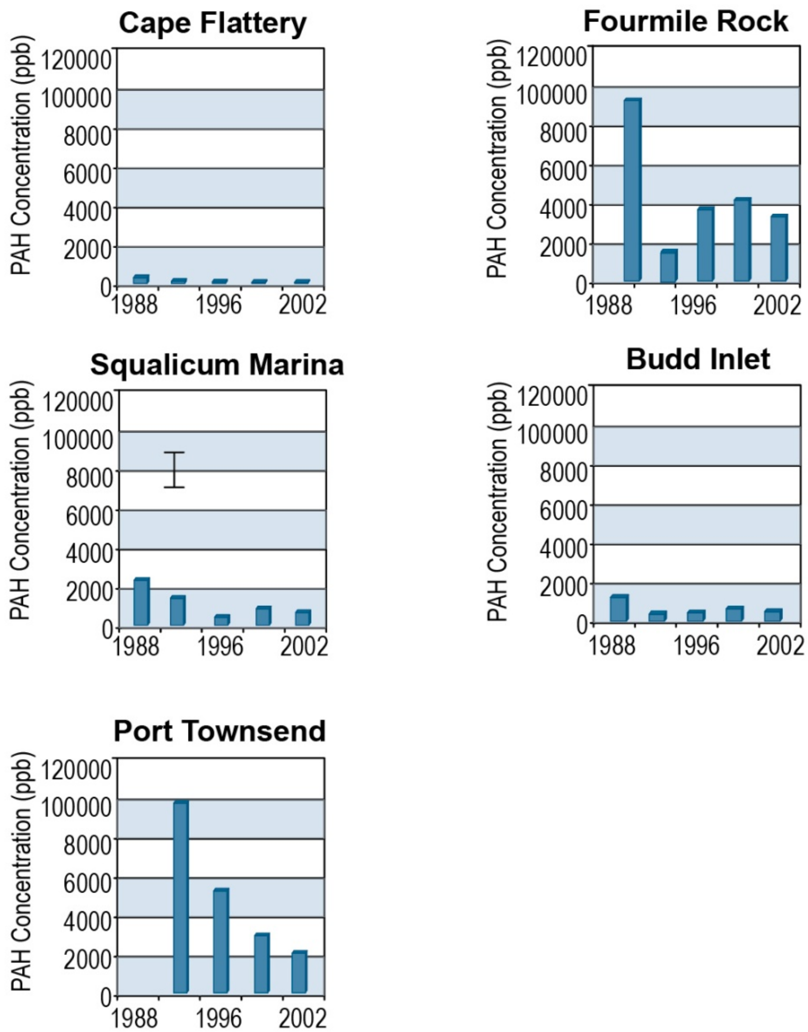

Figure 12. Total PAHs in Washington State mussels (μg/kg dry weight)

Source: Puget Sound Action Team 2007.

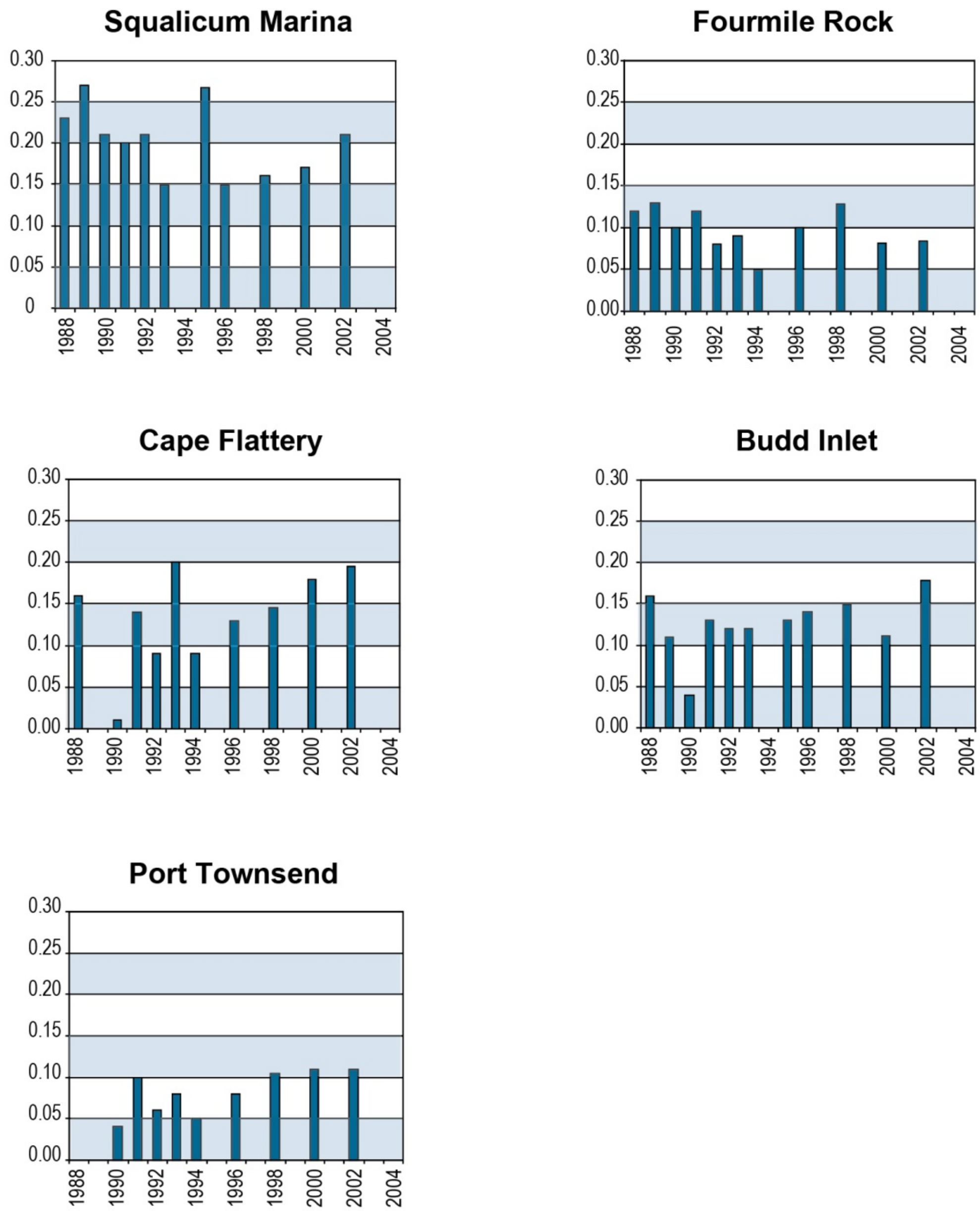

Figure 13. Total mercury concentrations in Washington State mussels (µg/kg dry weight)

Source: Puget Sound Action Team 2007.

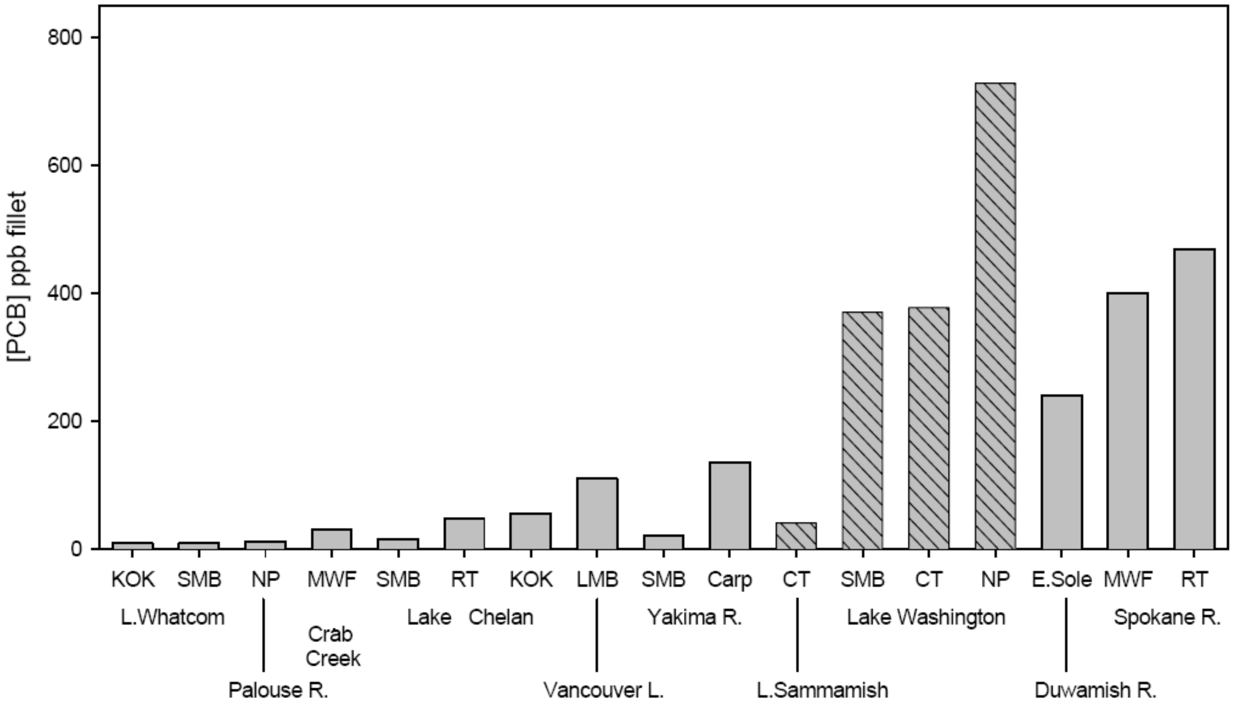

Figure 14. PCBs in large fish from Washington State lakes and rivers

Source: Hardy and McBride 2004

KOK=kokanee, SMB=smallmouth bass, NP=northern, pikeminnow, MWF=mountain whitefish, RT=rainbow trout, CT=cutthroat trout.

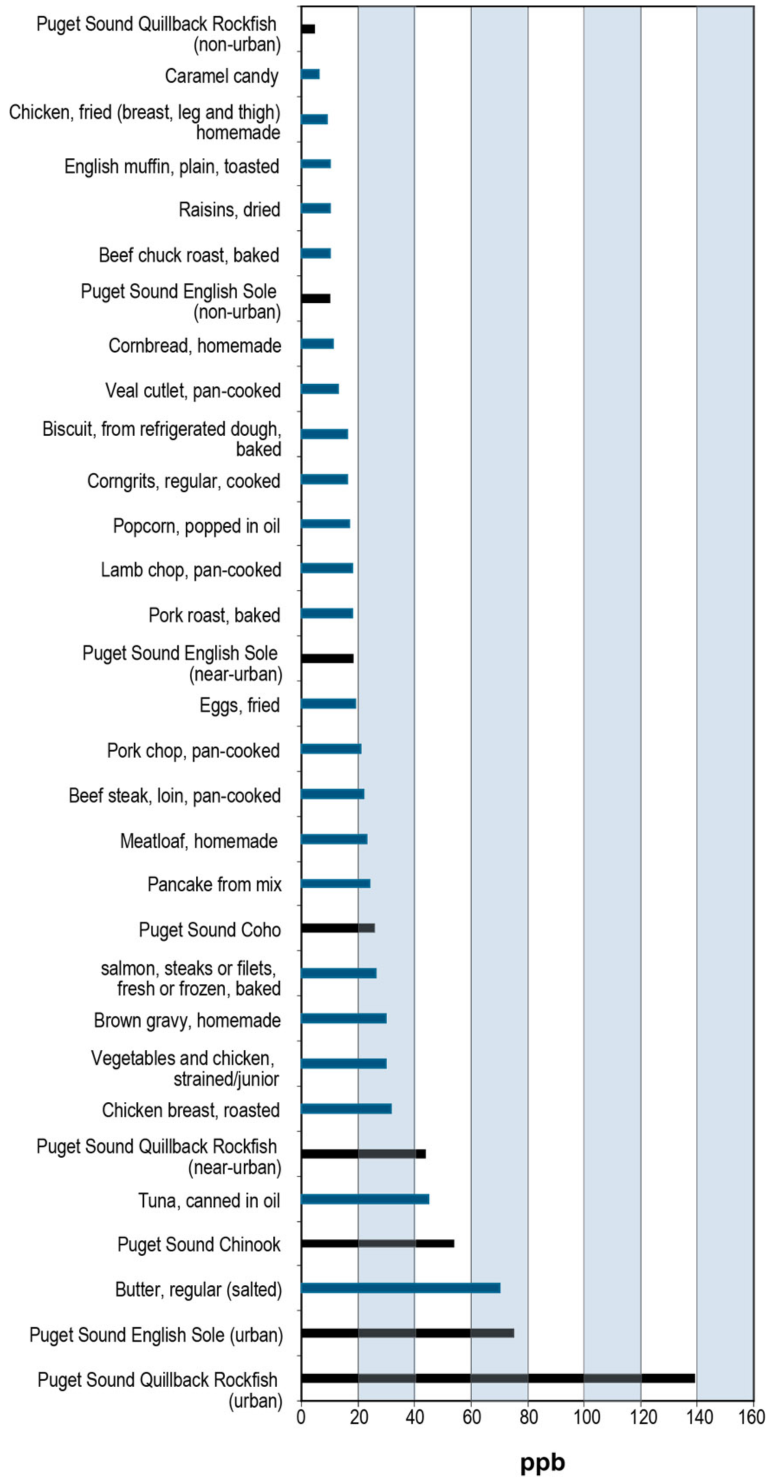

Figure 15. PCBs (µg/kg) in fish from Puget Sound compared to other common food sources

Samples include fish from Puget Sound and results are reported in micrograms per kilogram sampled. Commercial foods were sampled as part of the U.S. Food and Drug Administration's total diet study and market-basket survey. In most cases, data are limited by small sample sizes.

Source: Puget Sound Action Team 2007

Table 9. Summary of PCB data (µg/kg wet weight) in hake and pollock from seven Puget Sound Basins

ΣPCBs	Pacific Hake								Walleye Pollock		
	Str. Juan de Fuca	Strait of Georgia	Hood Canal	Whidbey Basin	Elliott Bay	Main Basin	South Sound		Hood Canal	Elliott Bay	Main Basin
n[a]	3/3	3/3	10/10	10/10	11/11	12/12	3/3		3/3	1/1	4/4
min	2.7	3.4	9	29	29	23	32		6.9	120	11
max	6.6	6.6	38	67	72	68	34		8.4	120	26
mean	4.2	5.3	15	44	45	33	33		7.6	nc	15
10th %.	2.7	3.4	9	30	30	24	32		6.9	nc	11
Median	3.3	5.7	10	41	44	28	33		7.4	nc	12
90th %	6.6	6.6	33	65	67	52	34		8.4	nc	26

[a] Number of samples with detected values/number of samples analyzed

"nc" indicates statistics that were not calculated because there were fewer than three detected values

In cases where no analyte was detected, the minimum (min) and maximum (max) values are presented as "<" the average limit of quantitation for that analyte.

B. Other protein sources (meat, eggs, dairy)

Very little information was available regarding contaminant concentration in other protein sources in Washington State. Therefore, this section summarizes available data from various U.S. and international sources.

a. Mercury

As described above, the U.S. Total Diet Study found that nearly all methylmercury in the U.S. diet is from fish and shellfish (Mahaffey 2009), but concerns have been raised about the potential for fish meal to increase contaminant levels in farm animals (Dórea 2008). This may be less of a concern for beef and cow's milk, however, because cows can demethylate methylmercury in the rumen and therefore absorb less mercury (ATSDR 1999). A study conducted in Pakistan (Shah et al. 2010) showed mercury uptake in chicken tissues from mercury in the chicken feed (Table 10).

Table 10. Concentration of total mercury in chicken feed and the different tissues of boiler chickens (µg/kg)

Poultry farms	Feed	Age groups (weeks)	Chicken tissues			
			Leg	Breast	Liver	Heart
Farm-1	15.1 ± 0.77	1–3	3.24 ± 0.23	3.49 ± 0.26	4.94 ± 0.49	3.27 ± 0.32
		4–6	2.66 ± 0.36	2.93 ± 0.22	3.55 ± 0.51	2.19 ± 0.26
Farm-2	12.8 ± 0.36	1–3	2.87 ± 0.21	2.59 ± 0.34	3.98 ± 0.38	2.84 ± 0.41
		4–6	2.15 ± 0.17	2.47 ± 0.21	3.43 ± 0.37	2.50 ± 0.41
Farm-3	16.5 ± 0.25	1–3	2.47 ± 0.14	3.86 ± 0.09	5.54 ± 0.33	2.93 ± 0.24
		4–6	2.14 ± 0.23	2.53 ± 0.26	4.34 ± 0.49	2.27 ± 0.32
Farm-4	12.2 ± 0.61	1–3	2.14 ± 0.23	2.79 ± 0.26	4.04 ± 0.49	3.17 ± 0.32
		4–6	1.57 ± 0.21	2.11 ± 0.34	3.28 ± 0.38	2.44 ± 0.41
Farm-5	8.57 ± 0.47	1–3	1.97 ± 0.21	2.11 ± 0.34	3.28 ± 0.38	2.44 ± 0.41
		4–6	1.27 ± 0.14	1.86 ± 0.09	2.54 ± 0.33	2.13 ± 0.24

Modified from Shah et al. 2010

*dry weight basis

b. Arsenic

Inorganic arsenic concentrations appear to occur in a relatively narrow concentration range across protein types, as shown by the data from Yost et al. (2004) (Table 11).

Table 11. Mean total and total inorganic arsenic concentrations in foods (µg/kg wet weight)

Food	Mean Total Arsenic	Mean Total Inorganic Arsenic[3]
Meat and Poultry		
Beef	51.5	0.39 J
Chicken	86.4	0.89 J
Pork	13.5	0.67 J
Seafoods		
Freshwater finfish	160	1.0 J
Saltwater finfish	2,356	0.55 J
Shrimp	1,890	1.9 J
Tuna	512	1.0 U
Dairy products and Eggs		
Milk[1,2]	2.2	1.0 U
Butter	1.8	1.17 J
Eggs	20	0.98 J

Modified from Yost et al. 2004

[1] both whole and skim milk were analyzed; results combined and applied to all milk products

[2] As3+ concentration was 0.18 µg/kg J

[3]Where no arsenic was detected (after blank-correcting), one-half the value of the method detection limit was given with a U designation. When the concentration of arsenic in food (after blank-correcting) was detected above the blank concentration but below the method detection limit, the value was given a J designation. Undetected samples have been included at one-half of the detection limits. All averaged values were computed as follows: (1) If one or more, but not all, values to be averaged were non-detected, 50 percent of the detection limit(s) was used in calculating the average concentration and (2) Mean values have a U or a J qualifier if all values used to calculate the mean were U or J qualified, respectively.

c. PAHs

As shown in Table 2 and described in Section 2.3.2, PAHs in protein sources are highly influenced by cooking methods. Schaum et al. (2003) evaluated cow's milk samples from locations across the United States for PBTs including PAHs. The PAHs analyzed were acenaphthylene, anthracene, benzo(a)pyrene, fluorene, naphthalene, phenanthrene, and pyrene (Table 12). Acenaphthene and benzo(a)pyrene were not detected in any samples analyzed by Schaum et al. (2003).

Table 12. Concentrations of PAHs in milk from western U.S. locations (ng/kg)*

PAH	July 2000	January 2001
Acenaphthylene	44.6	39.8
Anthracene	ND	40.8 J
Fluorene	57.6 J	127.2
Naphthalene	947.1	882.3
Phenanthrene	431.0	640.8
Pyrene	98.9	151.2

Modified from Schaum et al. 2003

West locations included: Portland, Tacoma, Spokane, Sacramento, San Francisco, Las Vegas, and Honolulu

Data from the original paper was in ng/L, which is roughly equivalent to ng/kg (density of milk is about 1.003)

Detected PAHs only; acenaphthene and benzo(a)pyrene were not detected in any samples from any locations in the study

d. Dioxins/PCBs

Table 13 presents the dioxin concentrations in milk samples taken across the United States. The Far West location included samples from Washington, Oregon, California, and Hawaii.

Table 13. TEQ concentrations of CDD/CDFs and PCBs in milk (pg/kg)

	CDD/CDF			PCB			Total[a]
Composite location	July 2000	January 2001	Mean [b]	July 2000	January 2001	Mean [b]	Mean [b]
New England	12.85	8.89	10.87	9.53	8.29	8.91	19.78
Mid-Atlantic	11.67	14.21	12.94	14.37	8.3	11.34	24.28
South Central	17.53	19.14	18.34	6.93	9.57	8.24	26.59
North Central	20.48	10.35	15.42	11.36	7.13	9.25	24.66
West Central	17.94	18.59	18.27	5.31	5.16	5.24	23.5
Southwest	13.27	6.21	9.74	6.71	6.29	6.5	16.24
Far South	18.13	35.82	26.98	8.07	7.59	7.83	34.81
Far West	22.5	13.2	17.85	8.8	8.33	8.57	26.42

Modified from Schaum et al. 2003

CDD/CDF = chlorinated dibenzo-*p*-dioxin/ chlorinated dibenzofuran

Data from the original paper was in pg/L, which is roughly equivalent to pg/kg (density of milk is about 1.003)

[a]This is the sum of the CDD/CDF seasonal mean and the PCB seasonal mean.

[b]Average of summer and winter concentrations.

Table 14 presents dioxin (PCDDs, PCDFs, plus a subset of four dioxin-like PCB congeners) concentrations (as mean TEQs) in uncooked meat samples from the United States.

Table 14. Mean TEQs of 17 PCDD/Fs and 4 dioxin-like PCBs in U.S. meats (ng/kg lipid)

Contaminant	beef N = 139	pork N = 136	chicken N = 151	turkey N = 84
mean PCDD/F TEQ	0.55 (0.51)	0.14 (0.04)	0.12 (0.04)	0.36 (0.34)
mean dioxin-like-PCB* TEQ	0.11 (0.11)	0.02 (0.01)	0.05 (0.05)	0.25 (0.25)

Adapted from Huwe et al. 2009
*PCBs analyzed were the coplanar congeners 77, 81, 126, and 169

Summary

Contaminants occur in many food sources and for some, food is the major contributor to intake in the general population (i.e., those not exposed occupationally). Mercury, arsenic, PAHs, dioxins, and PCBs all have food as the major source of intake.

- Over the past three decades, per capita chicken has increased, beef consumption has decreased, and pork and seafood consumption has remained fairly stable.
- In the United States, fish and shellfish are the primary dietary source of mercury and arsenic (organic forms). Shellfish may contribute significantly (~20 percent) to the PAH intake for shellfish consumers.
- Inorganic arsenic (generally considered to be the more toxic form) is present in low concentrations in most animal-based protein sources, but dietary intake is largely from cereals, water used in cooking, vegetables, and fruits and fruit juices.
- Nearly all of the dietary intake of mercury in the United States comes from animal-based protein sources, with fish and shellfish being the principal sources. PAH concentrations in foods are largely influenced by cooking methods.
- Dietary intake studies show dioxin TEQs come largely from animal-based protein sources, with fish contributing roughly one-third of total TEQs.
- If fish and shellfish consumption increases in the future, exposure to methylmercury could increase. Actual increases would depend on the species consumed, as some contain higher average concentrations of mercury (for more on this topic, see *Technical Issue Paper: Health Benefits and Risks of Consuming Fish and Shellfish*).

Direct comparison of contaminant concentrations in dietary protein sources is difficult due to variations in the way these data are presented in the literature. Concentrations are presented as wet weight or dry weight, cooked or uncooked, lipid-normalized or total, and using a variety of units. Data were collected during different time periods, in different locations, and using different analysis methods.

References

ATSDR. 1995. Toxicological profile for polycyclic aromatic hydrocarbons. U.S. Dept. of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA. August 1995.

ATSDR. 1999. Toxicological profile for mercury. U.S. Dept. of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA. March 1999.

ATSDR. 2007. Toxicological profile for arsenic. U.S. Dept. of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA. August 2007.

Birnbaum, L.S. 2005. Dioxins: are we at risk? PowerPoint presentation. National Health and Environmental Effects Research Laboratory, U.S. Environmental Protection Agency, Research Triangle Park, NC. Presented at the Grace A. Dow Memorial Library in Midland, MI, July 12, 2005.

CDC (Centers for Disease Control). 2009. Arsenic - fact sheet. November 2009. http://www.cdc.gov/biomonitoring/pdf/Arsenic_FactSheet.pdf.

Dórea, J.G. 2008. Persistent, bioaccumulative and toxic substances in fish: Human health considerations. *Science of The Total Environment.* 400(1-3): 93-114.

Dougherty, C., S. Holtz, J. Reinert, L. Panyacosit, D. Axelrad, and T. Woodruff. 2000. Dietary exposures to food contaminants across the United States. *Environmental Research.* 84(2): 170-185.

Ecology. 2010. An Assessment of the PCB and Dioxin Background in Washington Freshwater Fish, with Recommendations for Prioritizing 303(d) Listings. Washington State Department of Ecology, Environmental Assessment Program, Statewide Coordination Section. Publication No. 10-03-007. January 2010. http://www.ecy.wa.gov/pubs/1003007.pdf

Ecology. 2011. Fish Consumption Rates Technical Support Document: A Review of Data and Information About Fish Consumption in Washington. Washington State Department of Ecology. Publication No. 11-09-050, Version 1.0 (Draft). September 2011.

Eisler, R. 2000. *Handbook of Chemical Risk Assessment: Health Hazards to Humans, Plants, and Animals. Vol. 2, Organics*. Boca Raton, FL: Lewis Publishers.

European Food Safety Authority. 2008. Polycyclic aromatic hydrocarbons in food: Scientific Opinion of the Panel on Contaminants in the Food Chain. *The EFSA Journal.* 724: 1-114.

FAO/WHO. 2006. Evaluation of certain food contaminants: Sixty-fourth report of the Joint FAO/WHO Expert Committee on Food Additives. WHO Technical Report Series No. 930. Food and Agriculture Organization of the United Nations and the World Health Organization, Geneva. 2006.

Fiedler, H. 1998. Sources and environmental impact of PCDD/PCDF. Proceedings of the Subregional Awareness Raising Workshop on Persistent Organic Pollutants (POPs) Kranjska Gora, Slovenia, 11-14 May 1998. http://www.chem.unep.ch/pops/POPs_Inc/proceedings/slovenia/FIEDLER1.html

Hardy, J., and D. McBride. 2004. Evaluation of contaminants in fish from Lake Washington, King County, Washington. Washington Department of Health, Division of Environmental Health, Office of Environmental Health Assessments, Olympia, WA. September, 2004.

Hirai, Y., S.i. Sakai, N. Watanabe, and H. Takatsuki. 2004. Congener-specific intake fractions for PCDDs/DFs and Co-PCBs: modeling and validation. *Chemosphere.* 54(10): 1383-1400.

Huwe, J., M. Lorentzsen, D. Pagan-Rodriguez, N. Abdelmajid, K. Dearfield, N. Clinch, D. Gordon, J. Holterman, and E. Zaki. 2009. Survey of polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans, and non-ortho-polychlorinated biphenyls in U.S. meat and poultry, 2007-2008: Effect of New Toxic equivalency factors on toxic equivalency levels, patterns, and temporal trends. *Journal of Agricultural and Food Chemistry.* 57(23): 11194-11200.

Kazerouni, N., R. Sinha, C.-H. Hsu, A. Greenberg, and N. Rothman. 2001. Analysis of 200 food items for benzo[a]pyrene and estimation of its intake in an epidemiologic study. *Food and Chemical Toxicology.* 39(5): 423-436.

Lorber, M., D. Patterson, J. Huwe, and H. Kahn. 2009. Evaluation of background exposures of Americans to dioxin-like compounds in the 1990s and the 2000s. *Chemosphere.* 77(5): 640-651.

Mahaffey, K.R. 2009. Response to Commentary on "Studies of fish consumption as source of methylmercury should consider fishmeal-fed farmed fish and other animal foods" by Professor Dórea. *Environmental Research.* 109(1): 133-134.

Meacher, D.M., D.B. Menzel, M.D. Dillencourt, L.F. Bic, R.A. Schoof, L.J. Yost, J.C. Eickhoff, and C.H. Farr. 2002. Estimation of multimedia inorganic arsenic intake in the U.S. population. *Human and Ecological Risk Assessment.* 8(7): 1697-1721.

NMFS (National Marine Fisheries Service). 2011. Fisheries of the United States 2010. Current Fisheries Statistics No. 2010. National Marine Fisheries Service, Office of Science and Technology, Fisheries Statistics Division, Silver Spring, MD. August 2011.

O'Neill, S.M., and J.E. West. 2009. Marine distribution, life history traits, and the accumulation of polychlorinated biphenyls in Chinook salmon from Puget sound, Washington. *Trans. Am. Fish. Soc. Transactions of the American Fisheries Society.* 138(3): 616-632.

Puget Sound Action Team. 2007. 2007 Puget Sound Update: Ninth report of the Puget Sound assessment and monitoring. PSAT 07-02. Office of the Governor, Olympia, WA. 260 pp.

Schaum, J., L. Schuda, C. Wu, R. Sears, J. Ferrario, and K. Andrews. 2003. A national survey of persistent, bioaccumulative, and toxic (PBT) pollutants in the United States milk supply. *Journal of Exposure Analysis and Environmental Epidemiology*. 13(3): 177-186.

Schecter, A., K. Patel, J. Colacino, D. Haffner, M. Opel, O. Papke, and L. Birnbaum. 2010. Perfluorinated compounds, polychlorinated biphenyls, and organochlorine pesticide contamination in composite food samples from Dallas, Texas, USA. *Environmental Health Perspectives.* 118(6): 796-802.

Shah, A.Q., T.G. Kazi, J.A. Baig, H.I. Afridi, G.A. Kandhro, S. Khan, N.F. Kolachi, and S.K. Wadhwa. 2010. Determination of total mercury in chicken feed, its translocation to different tissues of chicken and their manure using cold vapour atomic absorption spectrometer. *Food and Chemical Toxicology*. 48(6): 1550-1554.

Tsuchiya, A., T.M. Burbacher, E.M. Faustman, J. Hardy, and K. Marien. 2008. Fish intake guidelines: Incorporating n-3 fatty acid intake and contaminant exposure in the Korean and Japanese communities. *American Journal of Clinical Nutrition.* 87(6): 1867-1875.

USDA (U.S. Department of Agriculture). 2012. USDA agricultural projections to 2021. Long-term Projections Report OCE-2012-1. Interagency Agricultural Projections Committee, Washington, DC. February 2012.

USDA, and USDHHS. 2010. Dietary guidelines for Americans 2010. U.S. Department of Agriculture, U.S. Department of Health and Human Services, Washington, DC. December 2010.

WDOH (Washington Department of Health). 2007. Health consultation: Evaluation of PCBs, PBDEs and Selected Metals in the Spokane River, Including Long Lake Spokane, Washington. Washington Department of Health Office of Environmental Health Assessments, prepared under a cooperative agreement with the Agency for Toxic Substances and Disease Registry, Olympia, WA. August 28, 2007.

WDOH. 2010. Health consultation: Fidalgo Bay, Anacortes, Skagit County, Washington. Washington Department of Health Office of Environmental Health Assessments, prepared under a cooperative agreement with the Agency for Toxic Substances and Disease Registry, Olympia, WA. February 25, 2010.

White, J. 2012. Estimate of PAH8 dietary intake from shellfish. Provided by email from R. Duff (Washington Department of Health) to C. McCormack (Washington Department of Ecology), May 16, 2012.

Yost, L., S.H. Tao, S. Egan, L. Barraj, K. Smith, J. Tsuji, Y. Lowney, R. Schoof, and N. Rachman. 2004. Estimation of dietary intake of inorganic arsenic in U.S. children. *Human and Ecological Risk Assessment.* 10(3): 473-483.

Ysart, G., P. Miller, H. Crews, P. Robb, M. Baxter, C.D. L'Argy, S. Lofthouse, C. Sargent, and N. Harrison. 1999. Dietary exposure estimates of 30 elements from the UK Total Diet Study. *Food Additives and Contaminants.* 16(9): 391-403.

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Appendices

Appendix A. Contaminant Data Tables Supporting Figures 6, 7, and 9

Table A-1. Sources of dietary inorganic arsenic for children 1 to 6 years (% contribution)

Exposure Pathway	Percent Contribution
Fruits and fruit juice	20.9
Fish & Shellfish	0.1
Dairy products	14
Meat & poultry	1.2
Water used in cooking	4.1
Eggs	0.5
Grain & grain products	47.3
Condiments, fats, oils	0.5
Vegetables & vegetable products	6.3
Legumes, nuts, seeds	1

Modified from Yost et al. 2004

Table A-2. Lower bound and upper bound PAH15 concentrations (µg/kg) by food group

Food Group	N	Lower Bound	Upper Bound
Barbequed meat	31	12.18	12.48
Grilled meat	21	11.27	11.77
Other meat	49	0.96	1.43
Smoked meat	167	2.97	3.61
Fresh fish	6	0.19	2.82
Processed fish	29	31.9	33.86
Smoked fish	153	3.51	4.03
Molluscs	167	19.57	20.14
Smoked molluscs	11	18.23	18.65

N: number of samples

PAHs analysed included: benz(a)anthracene, benzo(b)fluoranthene, benzo(j)fluoranthene, benzo(k)fluoranthene, benzo(ghi)perylene, benzo(a)pyrene, chrysene, cyclopenta(cd)pyrene, dibenz(a,h)anthracene, dibenzo(a,e)pyrene, dibenzo(a,h)pyrene, dibenzo(a,i)pyrene, dibenzo(a,l)pyrene, indeno(1,2,3-)pyrene and 5-methylchrysene.

Processed fish included fish canned in oil.

Table A-3. Estimated TEQ intake from dietary protein sources

Exposure Pathway	PCDD/PCDF/PCB TEQ Intakes (pg/day)[1]	Percent Contribution
Fish	10.9	31.5
Beef	6.7	19.4
Dairy	5.6	16.2
Other sources[2]	3.7	10.7
Milk	2.7	7.8
Eggs	2.4	6.9
Other meat[3]	1.5	4.3
Poultry	0.7	2.0
Total:	34.5	--

Modified from Lorber et al. (2009)

Notes:

1. Non-detects set at zero

2. Other sources included consumption of water, air inhalation, ingestion of soil, soil dermal contact, and vegetable fat intake

3. Other meats included game, lamb, and other unidentified meats.

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Appendix B. Washington State Contaminant Data Summarized from the EIM Database

The following tables of mercury, total PCBs, and dioxin TEQ concentrations were summarized from data taken from the Ecology Environmental Information Management (EIM) database. The EIM data included marine and freshwater species. Statistical summaries pooled the data by marine vs. freshwater, species type, and ecological niche (deposit feeding vs. filter feeding shellfish, pelagic vs. demersal fish vs. salmonids, etc.). Further breakdown of the data was by tissue type analyzed (e.g., whole body vs. fillet for fish, muscle vs. hepatopancreas for crabs). Data included in the summary statistics were restricted to samples collected within the last 10 years. PCB congeners were summed to create *Total PCB* values; however, not all possible congeners were analyzed in every sample. Species were grouped as follows:

Anadromous

Bull trout (can also occur as a freshwater species)
Chinook salmon
Coho salmon
Cutthroat trout (can also occur as a freshwater species)
Steelhead (can also occur as a freshwater species [rainbow trout])
Sockeye salmon (can also occur as a freshwater species [kokanee])
White sturgeon (can also occur as a freshwater species)
American shad (this may be a freshwater occurring species too)
Pacific lamprey

Freshwater

Salmonids (if EIM data assigned FW, then kept FW)

Brook trout
Brown trout (aka German brown trout)
Cutthroat trout (can also occur as an anadromous species)
Rainbow trout (can also occur as an anadromous species [steelhead])
Kokanee (can also occur as an anadromous species [sockeye salmon])
American shad (usually an anadromous species)
Lake trout
Mountain whitefish
Lake whitefish
Pygmy whitefish
Smelt

Bass

Largemouth bass
Smallmouth bass

Other freshwater fish

White sturgeon (can also occur as an anadromous species)

Northern pike
Walleye
Northern pikeminnow
Black crappie
Bluegill
Pumpkinseed
Peamouth
Rock bass (sunfish family)
Yellow perch
Burbot
Bridgelip sucker
Common carp
Grass carp
Brown bullhead
Channel catfish
Largescale sucker
Unidentified sucker
White sucker

Marine Demersal

Arrowtooth flounder
Butter sole
English sole
Flathead sole
Pacific dover sole
Pacific sand sole
Pacific sanddab
Pacific slender sole
Rock sole
Speckled sanddab
Starry flounder
unidentified flounder

Other Marine Fish

Lingcod
Pile perch
Shiner perch
Striped seaperch

Invertebrates

Bivalves – deposit feeding: macoma clam, "unidentified" clams
Bivalves – filter feeding: softshell, Manilla, horse, littleneck, Asiatic, geoduck, and butter clams; blue mussels, Pacific oysters, Kumomoto oysters
Bivalves – freshwater: Asiatic (Corbicula) clam
Crabs: Dungeness, rock, graceful
Crayfish: Pacific signal crayfish

Table B-1. Contaminant concentrations in selected Washington State freshwater fish

	Count	Minimum	Maximum	Mean
BASS				
Fillet, skin on				
dioxin TEQ (ng/kg)	35	0.028	0.57	0.18
mercury (µg/kg)	403	20	1,800	234
PCB, Sum of Congeners (µg/kg)	20	0.25	25	6.5
Fillet, skin off				
mercury (µg/kg)	115	26	790	240
FRESHWATER SALMONID				
Fillet, skin on				
dioxin TEQ (ng/kg)	12	0.081	1.0	0.40
mercury (µg/kg)	60	17	250	93
PCB, Sum of Congeners (µg/kg)	49	0.29	384	25
Fillet, skin off				
mercury (µg/kg)	1	64	64	64
Whole Organism				
PCB, Sum of Congeners (µg/kg)	66	0.94	450	35
OTHER FRESHWATER FISH				
Fillet, skin on				
dioxin TEQ (ng/kg)	84	0.032	11	0.44
mercury (µg/kg)	265	11	1,900	187
PCB, Sum of Congeners (µg/kg)	31	0.044	611	41
Fillet, skin off				
dioxin TEQ (ng/kg)	9	0.21	1.1	0.49
mercury (µg/kg)	12	17	410	152
PCB, Sum of Congeners (µg/kg)	1	2.4	2.4	2.4
Whole Organism				
dioxin TEQ (ng/kg)	22	0.14	0.65	0.38
mercury (µg/kg)	24	24	290	160
PCB, Sum of Congeners (µg/kg)	12	28	253	97

Data provided by Laura Inouye, Washington Department of Ecology,

- queried from the Environmental Information Management Database
- limited to the most recent 10 years (2002 - 2012)
- non-detects not included in calculations

Table B-2. Contaminant concentrations in selected Washington State invertebrates

	Count	Minimum	Maximum	Mean
BIVALVE, DEPOSIT FEEDER				
Whole organism, not shell				
dioxin TEQ (ng/kg)	24	0.02	20	1.7
BIVALVE, FILTER FEEDER				
Muscle				
dioxin TEQ (ng/kg)	12	0.016	0.16	0.1
mercury (µg/kg)	34	5	33	11
Whole organism, not shell				
dioxin TEQ (ng/kg)	73	0.015	1.6	0.2
mercury (µg/kg)	197	3.2	82	9.2
Whole organism, not shell, not gut				
dioxin TEQ (ng/kg)	13	0.017	0.35	0.1
mercury (µg/kg)	72	3.6	61	18
Skin				
mercury (µg/kg)	20	1	36	7.4
Viscera, abdominal and thoracic				
dioxin TEQ (ng/kg)	5	0.055	0.099	0.1
mercury (µg/kg)	77	4	74	20
Gut ball				
dioxin TEQ (ng/kg)	4	0.17	0.44	0.3
mercury (µg/kg)	2	45	48	47
FRESHWATER BIVALVE				
Whole organism, not shell				
dioxin TEQ (ng/kg)	29	0.41	5.6	1.2
mercury (µg/kg)	25	5	16	9.4
CRAB				
Muscle				
dioxin TEQ (ng/kg)	46	0.027	0.74	0.2
mercury (µg/kg)	26	20	70	43
Muscle, visceral				
dioxin TEQ (ng/kg)	9	0.028	0.38	0.2
mercury (µg/kg)	9	31	110	67
Whole organism, not exoskeleton				
dioxin TEQ (ng/kg)	7	0.14	1.5	0.6
Whole organism, not exoskeleton, not gut				
dioxin TEQ (ng/kg)	10	0.011	0.3	0.1
Hepatopancreas				
dioxin TEQ (ng/kg)	54	0.18	41	6.1
mercury (µg/kg)	22	20	220	55

Table B-2. Contaminant concentrations in selected Washington State invertebrates

	Count	Minimum	Maximum	Mean
CRAYFISH				
Whole organism, not exoskeleton, not gut				
PCB, Sum of Congeners (µg/kg)	1	0.87	0.87	0.9
SHRIMP				
Whole organism				
dioxin TEQ (ng/kg)	6	0.16	0.34	0.2
mercury (µg/kg)	6	23	50	37
SEA CUCUMBER				
Muscle, somatic				
mercury (µg/kg)	18	16	88	32

Data provided by Laura Inouye, Washington Department of Ecology,

- queried from the Environmental Information Management Database
- limited to the most recent 10 years (2002 - 2012)
- non-detects not included in calculations

Table B-3. Contaminant concentrations in selected Washington State marine/anadromous fish

	Count	Minimum	Maximum	Mean
ANADROMOUS FISH				
Fillet, skin on				
dioxin TEQ (ng/kg)	107	0.044	4.6	0.37
mercury (µg/kg)	136	5.8	610	103
Fillet, skin off				
mercury (µg/kg)	2	37	54	46
Whole organism				
dioxin TEQ (ng/kg)	17	0.51	1.5	0.78
mercury (µg/kg)	35	29	100	54
Whole organism, not fillets				
mercury (µg/kg)	10	22	41	33
MARINE DEMERSAL FISH				
Fillet, skin on				
mercury (µg/kg)	43	5	130	37
Fillet, skin off				
dioxin TEQ (ng/kg)	25	0.086	1.0	0.34
mercury (µg/kg)	28	20	190	78
Muscle				
dioxin TEQ (ng/kg)	35	0.011	0.36	0.054
mercury (µg/kg)	3	10	40	23
Whole organism				
dioxin TEQ (ng/kg)	27	0.071	1.7	0.50
mercury (µg/kg)	50	5	56	17
Whole organism, not shell				
dioxin TEQ (ng/kg)	26	0.023	0.30	0.077
mercury (µg/kg)	23	4	27	11
Whole organism, not fillets				
dioxin TEQ (ng/kg)	16	0.16	1.2	0.66
Viscera, abdominal and thoracic				
dioxin TEQ (ng/kg)	20	0.029	0.18	0.088
Skeleton				
mercury (µg/kg)	8	18	62	38
OTHER MARINE FISH				
Fillet, skin on				
dioxin TEQ (ng/kg)	2	0.084	0.092	0.088
mercury (µg/kg)	4	40	97	69
Whole organism				
mercury (µg/kg)	25	10	30	22

Table B-3. Contaminant concentrations in selected Washington State marine/anadromous fish

	Count	Minimum	Maximum	Mean
Whole organism, not fillets				
dioxin TEQ (ng/kg)	2	0.14	0.17	0.16
mercury (µg/kg)	2	56	220	138

Data provided by Laura Inouye, Washington Department of Ecology,

- queried from the Environmental Information Management Database
- limited to the most recent 10 years (2002 - 2012)
- non-detects not included in calculations

Table B-4. Mercury concentrations (μg/kg) in selected Washington State fish

	Count	Minimum	Maximum	Mean
FRESHWATER FISH				
BASS				
Fillet, skin on	403	20	1,800	234
Fillet, skin off	115	26	790	240
FRESHWATER SALMONID				
Fillet, skin on	60	17	250	93
Fillet, skin off	1	64	64	64
OTHER FRESHWATER FISH				
Fillet, skin on	265	11	1,900	187
Fillet, skin off	12	17	410	152
Whole organism	24	24	290	160
MARINE/ANADROMOUS FISH				
ANADROMOUS FISH				
Fillet, skin on	136	5.8	610	103
Fillet, skin off	2	37	54	46
Whole organism	35	29	100	54
Whole organism, not fillets	10	22	41	33
MARINE DEMERSAL FISH				
Fillet, skin on	43	5	130	37
Fillet, skin off	28	20	190	78
Muscle	3	10	40	23
Whole organism	50	5	56	17
Whole organism, not shell	23	4	27	11
Skeleton	8	18	62	38
OTHER MARINE FISH				
Fillet, skin on	4	40	97	69
Whole organism	25	10	30	22
Whole organism, not fillets	2	56	220	138

Data provided by Laura Inouye, Washington Department of Ecology,

- queried from the Environmental Information Management Database
- limited to the most recent 10 years (2002 - 2012)
- non-detects not included in calculations

Table B-5. Mercury concentrations (µg/kg) in selected edible Washington State invertebrates

	Count	Minimum	Maximum	Mean
BIVALVE, FILTER FEEDER				
Muscle	34	5	33	11
Whole organism, not shell	197	3.2	82	9.2
Whole organism, not shell, not gut	72	3.6	61	18
Skin	20	1	36	7.4
Viscera, abdominal and thoracic	77	4	74	20
Gut ball	2	45	48	47
FRESHWATER BIVALVE				
Whole organism, not shell	25	5	16	9.4
CRAB				
Muscle	26	20	70	43
Muscle, visceral	9	31	110	67
Hepatopancreas	22	20	220	55
SHRIMP				
Whole organism	6	23	50	37
SEA CUCUMBER				
Muscle, somatic	18	16	88	32

Data provided by Laura Inouye, Washington Department of Ecology,
- queried from the Environmental Information Management Database
- limited to the most recent 10 years (2002 - 2012)
- non-detects not included in calculations

Table B-6. Dioxin TEQ concentrations (ng/kg) in selected Washington State fish and invertebrates

	Count	Minimum	Maximum	Mean
FRESHWATER FISH				
BASS				
Fillet, skin on	35	0.028	0.57	0.18
FRESHWATER SALMONID				
Fillet, skin on	12	0.081	1.0	0.40
OTHER FRESHWATER FISH				
Fillet, skin on	84	0.032	11	0.44
Fillet, skin off	9	0.21	1.1	0.49
Whole organism	22	0.14	0.65	0.38
MARINE/ANADROMOUS FISH				
ANADROMOUS FISH				
Fillet, skin on	107	0.044	4.6	0.37
Whole organism	17	0.51	1.5	0.78
MARINE DEMERSAL FISH				
Fillet, skin off	25	0.086	1.0	0.34
Muscle	35	0.011	0.36	0.054
Whole organism	27	0.071	1.7	0.50
Whole organism, not shell	26	0.023	0.30	0.077
Whole organism, not fillets	16	0.16	1.2	0.66
Viscera, abdominal and thoracic	20	0.029	0.18	0.09
OTHER MARINE FISH				
Fillet, skin on	2	0.084	0.092	0.09
Whole organism, not fillets	2	0.14	0.17	0.16
INVERTEBRATES	292	0.011	41	1.5
BIVALVE, DEPOSIT FEEDER				
Whole organism, not shell	24	0.020	20	1.7
BIVALVE, FILTER FEEDER				
Muscle	12	0.016	0.16	0.056
Whole organism, not shell	73	0.015	1.6	0.22
Whole organism, not shell, not gut	13	0.017	0.35	0.14
Viscera, abdominal and thoracic	5	0.055	0.099	0.068
Gut ball	4	0.17	0.44	0.27
FRESHWATER BIVALVE				
Whole organism, not shell	29	0.41	5.6	1.2

Table B-6. Dioxin TEQ concentrations (ng/kg) in selected Washington State fish and invertebrates

	Count	Minimum	Maximum	Mean
CRAB				
Muscle	46	0.027	0.74	0.16
Muscle, visceral	9	0.028	0.38	0.19
Whole organism, not exoskeleton	7	0.14	1.5	0.62
Whole organism, not exoskeleton, not gut	10	0.011	0.30	0.080
Hepatopancreas	54	0.18	41	6.1
SHRIMP				
Whole organism	6	0.16	0.34	0.21

Data provided by Laura Inouye, Washington Department of Ecology,

- queried from the Environmental Information Management Database
- limited to the most recent 10 years (2002 - 2012)
- non-detects not included in calculations

Table B-7. PCB concentrations (total congeners, μg/kg) in selected Washington State freshwater fish and shellfish

	Count	Minimum	Maximum	Mean
FRESHWATER FISH				
BASS				
Fillet, skin on	20	0.25	25	6.5
FRESHWATER SALMONID				
Fillet, skin on	49	0.29	384	25
Whole organism	66	0.94	450	35
OTHER FRESHWATER FISH				
Fillet, skin on	31	0.044	611	41
Fillet, skin off	1	2.4	2.4	2.4
Whole organism	12	28	253	97
INVERTEBRATES				
CRAYFISH				
Whole organism, not exoskeleton, not gut	1	0.87	0.87	0.9

Data provided by Laura Inouye, Washington Department of Ecology,

- queried from the Environmental Information Management Database
- limited to the most recent 10 years (2002 - 2012)
- non-detects not included in calculations

Technical Issue Paper

Salmon Life History and Contaminant Body Burdens

Technical Issue Paper

Salmon Life History and Contaminant Body Burdens

July 20, 2012

DEPARTMENT OF ECOLOGY
State of Washington

Toxics Cleanup Program
Washington State Department of Ecology
Olympia, Washington

Table of Contents

Page

List of Figures and Tables

Figures

Page

Tables

Page

Acronyms & Abbreviations

DDT	dichlorodiphenyltrichloroethane
DEHP	di-2-ethylhexyl phthalate
DNA	deoxyribonucleic acid
DPE	dialkyl phthalate ester
Ecology	Washington State Department of Ecology
EDC	endocrine disrupting chemical
FAC	fluorescent aromatic compound
FCR	fish consumption rate
HCH	hexachlorocyclohexane
LDW	Lower Duwamish Waterway
PAH	polycyclic aromatic hydrocarbon
PBDE	polybrominated diphenyl ether
PBDE	polybrominated diphenyl ether
PBT	persistent bioaccumulative toxics
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PCN	polychlorinated napthalene
PFC	perfluoroalkyl compound
POP	persistent organic pollutant
PSAMP	Puget Sound Assessment and Monitoring Program
SaSI	Salmonid Stock Inventory
TEQ	toxic equivalent
tPAH	total polycyclic aromatic hydrocarbon
tPCB	total polychlorinated biphenyl
TSD	Technical Support Document
WDFW	Washington Department of Fish and Wildlife

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Introduction

The Washington State Department of Ecology (Ecology) received many and divergent comments about including or excluding salmon from the default fish consumption rate for Washington State fish-consuming populations. Consideration of salmon consumption depends on several different technical- and policy-related factors, including salmon life cycle, salmon chemical contaminant body burden, and Native American treaty reserved rights. New technical information and policy-related questions were provided to Ecology to further evaluate issues related to the consumption of salmon.

This Technical Issue Paper integrates information provided in the public review comments on the salmon life cycle and survival strategies with the information currently in the draft *Fish Consumption Rates Technical Support Document* (TSD). This integration and analysis considers, in part, where salmon contaminant body burdens are obtained in relation to the salmon life cycles and survival strategies. This Technical Issue Paper also examines factors that may affect decisions to include or exclude consumption of salmon in site-specific sediment risk management and cleanup decisions. It is a targeted examination of the issues raised by review comments received on the draft TSD, and was prepared within a limited time frame. Therefore, it may not include all available information on this subject.

Analysis

This section presents a review of salmon life history, a summary of literature on where and when salmonids obtain contaminant body burdens, and an assessment of the adequacy of currently available information to support site-specific salmon consumption criteria or advisories. Each topic includes a brief summary of the information presented in the draft TSD, followed by a summary of the public comments received on that topic. The analysis then provides further reviews or information summaries for additional clarity on each technical topic.

I. Salmon life history review

The draft TSD, specifically Appendix E, provides background information about the life cycle and survival strategies of those salmonids that both occur in Washington State waters and are consumed in quantity by its residents. Reviewers of the draft TSD provided very few comments with respect to new or missing life history information. Rather, comments largely focused on where and when body burdens are accumulated in a given salmon's life. Body burden accumulation is discussed in the next section of this Technical Issue Paper. However, in an effort to provide additional species-specific information, a summary of available information comparing life histories is included here. While this section is not an exhaustive literature review for Pacific Northwest salmonids, it is a suitable summary for salmonid life history comparison that provides a foundation for the following section on salmonid body burdens.

A. Summary of life history information in the draft TSD

Appendix E of the draft TSD provides background information about the life cycle and survival strategies for Chinook, coho, pink, sockeye, and chum salmon, as well as steelhead and cutthroat trout (Ecology 2011). The draft TSD Appendix E tables provide a general overview of the salmonids reviewed. Table E-1 of the draft TSD summarizes general rearing habitats and migrational behaviors, while Tables E-5 and E-6 provide summary information on seasonal occurrence, habitat use, and the range of life expectancy for each of the seven salmonids. Tables E-7 through E-10 provide creel data for freshwater and marine systems. Lastly, Tables E-11 through E-13 provide Salmonid Stock Inventory (SaSI) summaries, including stock-specific status for salmonids occurring near the Port Angeles Harbor and adjacent areas. SaSI is the Washington Department of Fish and Wildlife's (WDFW) standardized, uniform approach to identifying and monitoring the status of Washington's salmonid stocks.

B. Summary of additional life history information from TSD comments

Very little new information on the life history of salmonids was provided in comments received on the draft TSD. However, one reviewer (NCASI 2012) referred to Fresh et al. (2005) for a summary of juvenile salmonid life history strategies. Even though this table provides a different display of available life history information, it was technically consistent with the tables provided in Appendix E of the draft TSD. However, one of the important points noted in their comments was that Chinook salmon not only display more residency, but, as they mature, Chinook salmon tend to eat more fish than other salmon. This point is relevant for the following sections of this document that discuss the regional and species-specific differences between salmon in their uptake of toxins.

Additionally, a few other comments received (Weyerhaeuser 2012; Nippon Paper 2012) referred to their support of the 2003 AMEC paper titled *Evaluation of the Fish Consumption Rate Selected by Oregon DEQ for the Development of Ambient Water Quality Data*. This paper, when referring to salmonid bioaccumulation in the Columbia River basin, stated that "most anadromous species spend only a small fraction of their lifetime in the Columbia River," with the majority accumulated in marine habitats. While this is true for some species, or some "types" of species, this is an oversimplification due to the complex life history strategies of salmonids. For a more-detailed description of salmonid life histories and general periods of occurrence, please refer to the following section titled *Enhanced summary of Puget Sound salmonid life history strategies*.

Although the draft TSD was not intended to be a comprehensive review of the variable life history strategies of all Pacific Northwest salmonids, by watershed, and the degrees by which these differences could play a role in body burden accumulation, these comments do indicate that some additional life history summary would be helpful in this process. In an effort to more fully describe these differences, a slightly more comprehensive review of these life history strategies is provided in the following sections.

C. Enhanced summary of Puget Sound salmonid life history strategies

This summary is not intended to be an exhaustive literature review for Pacific Northwest salmonids, but rather a summary of relevant information to aid in the interpretation of contaminant body burdens and other issues discussed in subsequent sections of this Technical Issue Paper. It is not intended to provide a comprehensive review of habitat requirements, such as dissolved oxygen, stream depth/velocity, temperature, or turbidity, and the potential effects these may have on salmonid life stages. Seven species of salmonids were briefly reviewed in the draft TSD (Chinook, coho, sockeye, chum and pink salmon, and steelhead and cutthroat trout). Due to their relative absence as a human food resource compared to the other salmonids, this Technical Issue Paper does not include a life history review summary for cutthroat trout (similar to the previous exclusion of bull trout in the draft TSD).

Although each of the six species reviewed here are salmonids, their life history strategies can vary substantially. Table E-1 of the draft TSD indicates, for these six species, whether juveniles rear in rivers, estuaries, or lakes. It also indicates the general areas of migration: nearshore, continental shelf, or mid-oceanic. Table E-5 of the draft TSD provided additional life history detail for a variety of salmonid species, including general life expectancy. Additionally, Table E-6 provides a summary of the complex differences in habitat use for rearing and spawning life stages of the various Pacific Northwest salmonids. However, even with species-specific differences in their life histories, an aspect that each of these salmonids have in common is that they are anadromous. That is, they hatch from eggs in fresh water, mature in marine waters, then return to fresh water as adults to spawn. What follows is a more detailed species-specific summary than was provided in the draft TSD. Knowledge of where fish hatch, rear, migrate, and forage is important to understand how each of these species may differ in their accumulation of toxic body burdens.

When information is too voluminous for inclusion, the reader will be referred to the source for further review. For example, in 2002 WDFW updated the SaSI report by classifying the status of 489 Washington State salmonids and steelhead stocks (WDFW 2002). These SaSI status reports, along with other references, were briefly reviewed for general stock-specific background information. For a detailed description of the data and summaries of these salmonid stocks, please see the WDFW SaSI web page (http://wdfw.wa.gov/fish/sasi/).

Some of the life history questions that arose during the draft TSD review process include seasonal occurrence, habitat use, distribution, foraging, and life expectancy. Relative to more resident species such as perch and starry flounder, salmonids have large and highly variable distributions over their lifespan; occurrence and habitat use are a function of the needs of a given lifestage. However, even range and age at out-migration can be highly variable within a given species (e.g., steelhead and coho juveniles, and resident Chinook). Relative to fish species that are much longer lived such as English sole, spiny dogfish, white sturgeon, sablefish, yelloweye rockfish, and rougheye rockfish, estimated to live as long as 22, 75, 104, 114, 121, and 205 years, respectively (Love 1996; Berkeley et al. 2004), salmonids have short life spans, generally living from 2 to 6 years. In addition, compared to other bony fish species, salmonids have very large eggs. This provides the developing embryos and alevins with sufficient food resources to survive until they begin foraging as fry. However, egg size is highly variable between salmonids. This difference is important as body burden of lipophilic toxins can be transferred from adult salmon to their young via the fatty content of the eggs. The species-specific life history information that follows is intended to supplement the information previously provided in the draft TSD. For an in-depth summary of Chinook salmon life histories, please refer to Myers et al. (1998), Healey (1991), and Quinn (2005), among many others.

a. Chinook Salmon

Chinook salmon (*Oncorhynchus tshawytscha*) is the largest of the *Oncorhynchus* species, typically reaching 8 to 10 kg, although Chinook salmon have been documented in excess of 45 kg (Healey 1991; Quinn 2005). Resident Puget Sound Chinook salmon, however, are typically on the smaller end of this scale. Chinook salmon occur throughout the northeastern coast of the Pacific Ocean, and are considered one of the most sought after of the salmon species by commercial and tribal fishermen, sport anglers, and human consumers.

Due to their relatively large size, Chinook salmon generally spawn in larger rivers or streams than other salmonids (Healey 1991; Quinn 2005). Chinook salmon can be highly variable between and within given watersheds. They have various in-migration (e.g., spring versus fall) and out-migration (e.g., ocean-type versus stream-type) times that can vary within a given system, stock, or run of fish (WDFW 2002; Healey 1991; Myers et al. 1998; Duffy 2003, 2009; Duffy et al. 2005; Redman et al. 2005; Quinn 2005). Spring Chinook typically produce stream-type juveniles, whereas fall Chinook typically produce ocean-type juveniles (Quinn 2005). In general, Chinook salmon adults return to their natal streams in spring and fall where they can hold for several months prior to spawning (WDFW 2002; Quinn 2005). With some exceptions, for Chinook populations in the state of Washington, ocean-type Chinook occur within Puget Sound and the lower Columbia River tributaries, while spring-type Chinook occur in tributaries further up the Columbia River (Myers et al. 1998).

The life history habits of juvenile Chinook salmon vary widely, and a full understanding and classification of these differences is still in progress (Redman et al. 2005). For this summary, a four strategy classification of behavior in emerging Chinook salmon is described here, with one strategy for stream-type, and three strategies for ocean-type.

Egg

Chinook salmon have the largest eggs compared to the other Pacific Northwest salmonids, ranging from 210 to 420 mg (Allen and Hassler 1986; Beacham and Murray 1993). The eggs of Puget Sound Chinook stocks are larger eggs of Chinook that occur in most other watersheds (Myers et al. 1998). Adult ocean-type Chinook salmon, which are generally larger in size than stream-type Chinook, also appear to produce larger eggs (Myers et al. 1998). As a result of the relatively larger eggs, Chinook fry are also relatively large upon emergence, averaging 33 to 37 mm in length and ranging from 0.12 to 0.47 gram (Healy 1991; Myers et al. 1998; Quinn 2005).

Fry

After emergence, stream-type Chinook spend a year or more in the river before migrating downstream (Healey 1991; Myers et al. 1998; Duffy 2003, 2009; Duffy et al. 2005; Quinn 2005). Once entering the marine environment, stream-type Chinook spend very little time in the estuaries before migrating toward coastal waters (Duffy 2003, 2009; Duffy et al. 2005). Ocean-type fry display a very different type of behavior. Myers et al. (1998) summarized literature documenting three distinct phases of ocean-type juveniles migrating to marine habitats. One phase, or strategy, is the immediate type that migrates to the ocean soon after yolk resorption at 30 to 45 mm in length. During years of poor environmental conditions, another behavioral phase may occur, with ocean-type juveniles remaining in fresh water for a year, although this is relatively uncommon. The most common behavior for ocean-type fry migrants is to migrate to marine habitats at 60 to 150 days post-hatching, or as fingerling migrants, migrating downstream in the late summer or autumn of their first year. Emergent Chinook fry in freshwater habitats, like fry of other Pacific salmonids, depend on shaded nearshore habitat with slow-moving currents, where they forage on drift organisms, including insects and zooplankton (Healey 1991). In general, ocean-type parr (the freshwater stage of juvenile salmon, which usually occurs in the first one to two years of life) usually migrate to estuarine areas from April through July with some variability (peak out-migration occurring from May to early July), becoming smolts (juveniles that have transitioned from fresh water to salt water) soon after entering marine waters. Duffy et al. (2005) found that wild ocean-type Chinook out-migrate to Puget Sound waters from March to July, while hatchery Chinook occupy nearshore Puget Sound waters soon after release and in pulses from May to June. Once reaching the marine environment, they then spend a few weeks or longer rearing in the estuary (Duffy 2003, 2009; Duffy et al. 2005).

Juvenile

Due to their longer residence time in freshwater systems prior to out-migration, stream-type Chinook are much larger, averaging 73 to 134 mm, than ocean-type Chinook upon reaching the estuary (review in Myers et al. 1998). Entering the estuary at a larger size allows spring-type juveniles to move offshore more quickly than ocean-type juveniles (Healey 1991).

Growth rates of juvenile Chinook salmon are dependent on the most abundant and available food resources. Ocean-type juveniles can grow at a faster rate than stream-type individuals when food is not limited (review in Myers et al. 1998). However, Duffy (2009) found that insects, a staple for stream-type juveniles, can provide a higher caloric content, and therefore promote faster growth than other food resources such as decapod larvae when food resources are equally abundant. Upon entering marine habitats, hatchery-produced smolts occur at a larger size than do their wild counterpart (Duffy 2009). Once reaching the marine waters of Puget Sound, juvenile ocean-type Chinook occupy nearshore habitats from April to June, before shifting to more offshore habitats from July to September (Duffy 2009). Estuaries are extremely important for foraging and nearshore-migrating juvenile salmon (Simenstad et al. 1982; Simenstad and Cordell 2000). Further, juvenile salmonids do not limit their use of estuarine habitats to their natal estuaries, as juvenile salmonids have also been found to enter and utilize non-natal estuaries during their marine nearshore migration (Shaffer et al. 2008). Duffy (2009) found that the size of ocean-type juveniles immediately prior to this offshore movement was strongly related to survival. By early fall, it was presumed that juvenile Chinook salmon in Puget Sound either migrate towards the Pacific Ocean or to deeper waters of Puget Sound (Duffy 2009). Coastal rearing populations of juvenile Chinook salmon are largely piscivorous (fish eating), but also consume euphausiids, amphipods, copepods, pteropods, and cephalopods (Brodeur 1990).

Subadult

Once reaching the ocean, Chinook salmon are distributed in both coastal and offshore waters (Quinn 2005). In the ocean, ocean-type Chinook are distributed along coastal waters, whereas stream-type can be found offshore and throughout the central Pacific (Healey 1991; Myers et al. 1998). The diet of oceanic maturing Chinook consists mainly of squid, with fish, euphausiids, and pteropods also present (Brodeur 1990). Those salmonids that mature and occur within coastal waters have a diet primarily composed of fish, with euphausiids, decapod larvae, and cephalopods also present (Brodeur 1990). Salmonids mature in oceanic and coastal waters from 1 to 6 years, although 2 to 4 years is more typical, before returning to their natal streams to spawn (Myers et al. 1998).

Adult

Mature adult Chinook salmon can return to their natal streams months in advance of spawning; however, both spring-run and fall-run fish begin spawning in the fall (Quinn 2005). Within much of the Puget Sound region, adult fall-run Chinook salmon start entering estuarine and riverine waters as early as August and begin spawning in their natal rivers and streams from mid-September to November, but this can continue through January (Myers et al. 1998; Quinn 2005). Spring-run Chinook salmon enter Puget Sound and Columbia River waters in April and May, although spawning does not begin until August and September (Myers et al. 1998). Upon entering these freshwater systems, ocean-type adults are larger than spring-type adults. It has been postulated that this is in part due to the additional 3 to 5 months longer that ocean-type adults remain in the marine environment prior to entering their natal streams (Myers et al. 1998; Duffy 2003, 2009; Duffy et al. 2005). It is likely that both of these factors play a role in the larger size of ocean-type adults.

Fat Content of Chinook

Fat content between salmonid stocks within a given system can vary substantially between years, seasons, sexes, and runs. Seasonal abundance of food in the ocean can alter size at entry as well as fat content upon entry. Within the Klamath River, Hearsey (2011) found that spring Chinook salmon, entering rivers months in advance of spawning, had much higher overall fat content compared to fall-run Chinook. However, due to their long period of freshwater residence prior to spawning, upon spawning, fall-run fish generally had a higher overall fat content. Hearsey (2011) found that male and female Chinook generally had the same fat content upon entry, but the females had a much lower fat content by the time spawning occurred. This is attributed to females using their fat reserves (muscle lipids and triacyglycerol deposits) during upstream migration and gonadal development of the eggs. Lipid content is an important factor as lipophilic contaminants can occur at higher numbers in these fish. Then as the female converts this energy during gonadal development, the fatty eggs from females with higher body burdens receive maternal transferred contaminants at elevated levels.

b. Coho Salmon

Like other salmonids in Washington State, coho salmon (*Oncorhynchus kisutch*) are hatchery-reared and released throughout the state. Although some of the cited literature reviews hatchery-released fish, this life history summary is intended to focus primarily on *natural spawning* or *wild* fish behaviors.

Along the northeastern coast of the Pacific, coho salmon range from Monterey Bay, California, to Point Hope, Alaska, and the Aleutian Islands (Laufle et al. 1986). Coho salmon typically reach 3.5 to 4.5 kg (Sandercock 1991), although they have been documented to reach as many as 14 kg (Laufle et al. 1986). Sandercock (1991) found that adult coho returning to British Columbia waters ranged from 53 to 67 cm (21 to 26 inches). For populations in and around Washington State, returning adult coho salmon are generally 3-year-olds, and spend approximately 18 months in fresh water and 18 months in marine habitats (Sandercock 1991). A small percentage of sexually mature males (*jacks*) return to fresh water to spawn after only 5 to 7 months in the ocean. Compared to Chinook salmon, coho tend to spawn in smaller streams of modest gradient (Quinn 2005).

In general, coho salmon do not have the broad variation in life history habits that occurs in other salmonids such as Chinook and sockeye salmon and steelhead. With some variation, adult coho salmon generally return to their natal streams to spawn in fall (Laufle et al. 1986; Sandercock 1991; Quinn 2005). After emerging, the fry generally remain within freshwater streams for a year or two before migrating downstream. A more detailed summary of the life history characteristics of coho salmon is provided below. For a more detailed, in depth summary of coho salmon life histories, please refer to Laufle et al. (1986), Sandercock (1991), and Quinn (2005), among others.

Egg

For coho salmon populations, there is considerable variation in egg size within and among populations (Fleming and Gross 1990). In general, longer female body length within a population resulted in a larger egg produced. Along the Pacific coast, coho salmon have been shown to have a significant negative relationship between latitude and egg size (Fleming and Gross 1990). In other words, all other factors equal, coho salmon in California have larger, albeit fewer, eggs than those in Puget Sound, which are larger than those in systems further north. In general, coho salmon eggs within Washington State range from 160 to 295 mg, depending on location, and from 4.5 to 6.0 mm in diameter (Fleming and Gross 1990; Laufle et al. 1986; Quinn 2005).

Fry

Following hatching, coho alevin do not emerge from the gravel until 2 to 3 weeks after hatching (Laufle et al. 1986). Emergence has been detected from March to July (Laufle et al. 1986). Coho salmon fry are approximately 30 mm upon emergence (Quinn 2005). Although some fry migrate to marine waters soon after emergence, the majority disperse both up- and downstream, remaining in streams to rear as juveniles for one to two years before migrating downstream (Laufle et al. 1986; Quinn 2005). Following emergence, coho fry begin feeding on a variety of insects, including dipterans, ephemopterans, and plecopterans (Laufle et al. 1986). Fry that are not displaced (washed out) by freshets grow larger, reside longer, and establish territories within their natal freshwater systems (Sandercock 1991). Over-wintering habitats, particularly off-channel flood plain habitats, are critical for growth and survival of young coho salmon.

Juvenile

While residing in freshwater streams, the maturing juvenile coho diet progressively includes larger prey items to sustain their growth. At this stage their diet begins to include crustaceans and fish, including co-occurring salmonids (i.e., pink and chum salmon) (Laufle et al. 1986). In fact, when co-occurring with other similar-sized juvenile salmonids, juvenile coho are more aggressive and outcompete other salmonids, including juvenile Chinook, for food resources (Laufle et al. 1986; Sandercock 1991). As a result, juvenile coho grow at a faster rate than co-occurring salmonids of other species. Both age and size of juvenile coho salmon are believed to be triggers for downstream migration (Laufle et al. 1986). Within this region, coho smolts typically leave fresh water and migrate to marine habitats to enter the smolting process in the spring (April to June) (Sandercock 1991). Once entering marine waters, coho smolts spend little time rearing in estuaries, instead migrating toward coastal waters (Quinn 2005). Juvenile, adult, and subadult coho diets in marine waters tend to vary with fish size and coastal region occupied. Juvenile coho diets, while occurring in nearshore sublittoral habitats, primarily consist of decapod larvae, euphausiids, amphipods, polychaetes, and crustaceans (Fresh et al. 1981, as summarized by Laufle et al. 1986; Quinn 2005; Sandercock 1991). Juvenile and subadult coho occurring further offshore generally consume decapod larvae, euphausiids, amphipods, and fish (principally herring) (Fresh et al. 1981, as summarized by Laufle et al. 1986; Brodeur 1990).

Subadult

Although some coho salmon move to offshore waters, typically subadults continue to feed and mature in these coastal waters of the northeast Pacific (Quinn 2005). An additional group appears to rear in the Strait of Georgia, and possibly Puget Sound (Laufle et al. 1986). For those coho that move offshore, this migration appears to begin in July and August (Laufle et al. 1986). Coho captured in oceanic waters tended to have diets dominated by fish and squid, although euphausiids, amphipods, and pteropods were also present (Brodeur 1990). The majority of coho originating from Washington streams migrate to coastal waters off Oregon and Washington, with low numbers occurring in Oregon and British Columbia waters (Laufle et al. 1986). Very few Washington coho appear to migrate to Alaskan coastal waters (Laufle et al. 1986). Both subadult and adult coho salmon usually occur in the top 10 meters of marine water, except when these surface waters become too warm (Laufle et al. 1986).

Adult

Young adult coho occurring offshore of the Columbia River are generally larger than their Puget Sound conspecifics, which may be due in part to their preying almost exclusively on fish (i.e., anchovy, surf smelt, whitebait smelt, herring, and juvenile Chinook and rockfish), although diets of mature coastal coho can also include euphausiids, crab larvae, amphipods, and squid (Laufle et al. 1986; Brodeur 1990). While some adult male coho salmon return after spending only one summer at sea, the majority of coho return after spending two, and sometimes three, summers at sea (Laufle et al. 1986; Quinn 2005). There are some run timing differences between coastal and inland Washington stocks of coho salmon, but adults begin returning to estuaries and outlets of their natal streams from July to September (Laufle et al. 1986). Except for earlier arriving jacks, sexually mature adults migrate up natal streams and rivers from August to November and spawn from November to December, and occasionally as late as February or March (Laufle et al. 1986; Sandercock 1991). In general, coho spawn in streams and small rivers that have relatively fast flow (Laufle et al. 1986; Sandercock 1991; Quinn 2005).

c. Sockeye Salmon

This section describes the general life history of anadromous forms of sockeye salmon (*Oncorhynchus nerka*); it does not describe life history patterns of land-locked sockeye salmon, known as kokanee. Mature adult sockeye salmon are small relative to other species in this region, reaching weights between 2 and 4 kg (4.4 and 8.8 lbs) (Quinn 2005). Along the coast of the northeast Pacific Ocean, sockeye salmon populations range from the Sacramento River in California to the Arctic (Burgner 1991). Within the state of Washington, a number of stocks exist including Lake Pleasant, Ozette Lake, Lake Washington, Baker Lake, and a few runs in the Columbia River drainage including Lake Wenatchee. Sockeye salmon have one of the most diverse patterns of life history among Pacific Northwest salmon species. For example, age at out-migration to marine systems from their natal streams not only varies between systems, and within systems, but can vary among related individuals. As a result, the summary presented here is considered a general overview of sockeye salmon life histories, not an all-inclusive review.

Sockeye salmon migrate from marine waters to natal freshwater systems in the summer months. Many times these systems include a *nursery lake*, in which juveniles rear and adults may hold prior to spawning. Adult sockeye salmon return to Lake Washington from June through August, with peaks from mid-June to mid-July (Newell 2005). When conditions are right, the adults migrate to suitable spawning habitat to dig a nest (redd) and spawn. In some systems sockeye salmon spawning can occur on lake beaches, inlets or outlets of lakes, and in outwash fans of lake tributaries (Gustafson et al. 1997). For an in-depth summary of sockeye salmon life histories please refer to Pauley et al. (1989), Burgner (1991), and Gustafson et al. (1997).

Egg

The number of eggs produced by a female sockeye salmon is relatively high (averaging 3,500), yet sockeye salmon produce some of the smallest eggs of the Pacific salmon (5.3 to 6.6 mm in diameter) (Pauley et al. 1989). As a result, when sockeye fry emerge, they are also small, averaging 26 to 29 mm (Quinn 2005). Lake Washington sockeye salmon eggs hatch approximately 51 to 124 days after fertilization. Hendry et al. (1998) found that incubation duration is strongly linked to water temperatures, with eggs incubating in warmer (12.5°C) waters hatching in less than half the time as those incubating in colder water (5°C).

Fry

The hatched alevin then take an additional 24 to 60 days to emerge from the gravel as fry, with warmer temperatures reducing the time for emergence. Sockeye salmon emerge as fry generally in April or May (Pauley et al. 1989), with some variability associated with temperature. As might be expected, fry length and weight have been found to be positively correlated with egg weight (Hendry et al. 1998). While there is some variability with post-emergent behavior (Gustafson et al. 1997), after emergence, fry generally migrate toward a lake that they will use as a nursery (and will migrate either upstream or downstream to reach a lake). However, some forms will remain in the natal stream for a portion of their rearing. Regarding their entry into marine waters, two *types* of sockeye salmon occur: the ocean-type (or *sea-type*) that migrates to marine waters in the first year of their life, and the stream-type that may rear in rivers and lakes for a year or more before migrating to marine habitats (Gustafson et al. 1997).

Juvenile

Once entering a lake system, juvenile sockeye begin preying upon available zooplankton and insect larvae. Juvenile sockeye in Washington generally migrate from their nursery lakes to marine habitats in March and continuing through June, with peak out-migration occurring in April and May (Appendix C-5 in Gustafson et al. 1997). Upon entering marine waters, estuarine use by juvenile sockeye salmon (smolts at this point) is limited, although some ocean-type sockeye may use these habitats before migrating toward coastal waters. Once entering salt water, juvenile sockeye salmon migrate northward to the Gulf of Alaska in coastal waters, but then by autumn move away from coastal habitats and use offshore waters for growth to maturation (Gustafson et al. 1997). In offshore marine waters, sockeye juveniles tend to disperse and not form defined schools (Burgner 1980, as cited in Pauley et al. 1989).

Subadult

Sockeye spend 2 to 4 years at sea before returning to their natal systems to spawn (Burgner 1991; Quinn 2005). Sockeye have long gill rakers to allow foraging on a variety of plankton species while in the ocean. As there is some dietary overlap between sockeye and pink salmon while at sea, during even-numbered years, when pink salmon abundance is relatively low, maturing sockeye salmon that migrated to the Bering Sea saw their diet of euphausiids, copepods, fish, and squid increase (Davis et al. 2005). For all years, and many regions, larger sockeye tended to have greater proportions of fish and squid in their diet (Brodeur 1990; Davis et al. 2005). For sockeye populations off the Oregon and Washington coasts, larval fishes (Osmeridae) and euphausiids can be dominant prey resources (Brodeur 1990). In other nearby regions, amphipods, copepods, and euphausiids were dominant (Brodeur 1990). As these fish reach maturity, they begin their return migrations from offshore and coastal waters towards the estuaries and river mouths of their natal river, streams, and lakes.

Adult

Upstream migration of sockeye stocks is highly variable depending on region and other factors, such as day length and water temperature (Burgner 1991). Adult sockeye salmon return to Lake Washington from June through August, with peaks from mid-June to mid-July (Newell 2005); however, these fish delay spawning until fall. Upon returning to these systems, reproductively mature adults average between 2 and 4 kg (4.4 and 8.8 lbs) (Quinn 2005). However, as reproductive sockeye that enter these systems exhibit little or no feeding behavior, some reduction in size may occur prior to their upstream migration to spawn in the fall. Within Lake Washington there are variations in sockeye salmon breeding behavior. Some of the returning adults hold deep in the lake for up to six months before spawning in the Cedar River that flows into the south end of the lake (Newell 2005). Other smaller populations, those that spawn in tributaries that flow into the north end of the lake or on the lake's beaches, spawn at different times (Newell 2005). This is believed to be a strategy for maximizing the survival of eggs, alevins, and fry as spawning timing is correlated with emergence timing, and therefore the best environmental conditions to optimize survival for the next generation (Brannon 1987, as cited in Newell 2005).

d. Chum Salmon

Chum salmon (*Oncorhynchus keta*) have the broadest distribution of all salmonid species (Pauley et al. 1988) and range along the Northeast Pacific coast from Monterey Bay, California, to the Arctic Ocean (Pauley et al. 1988; Salo 1991; Johnson et al. 1997). Similar to pink salmon, adult chum salmon prefer to spawn in the lower reaches of their natal streams (Pauley et al. 1988; Quinn 2005). Chum salmon produce larger eggs than do the smaller sockeye and pink salmon (Beacham and Murray 1993; Quinn 2005). Similar to pink salmon or ocean-type Chinook, juvenile chum migrate from their freshwater redds to marine waters almost immediately after emergence (Pauley et al. 1988; Salo 1991; Johnson et al. 1997; Quinn 2005). Chum salmon generally live 3 to 5 years and are relatively large compared to other salmonids, second only to Chinook; chum typically average 3.6 to 6.8 kg (8 to 15 lbs) (Salo 1991; Quinn 2005) and records indicate they can attain weights of up to 20.8 kg (46 lbs) (Johnson et al. 1997). Adult chum lengths typically average 53 to 79 cm (21 to 31inches), although they have been reported to reach sizes in excess of 108 cm (42.5 in) (Johnson et al. 1997). What follows is a summary by lifestage for chum salmon, with a focus on those populations that occur in Washington State. For an in-depth summary of chum salmon life histories please refer to Pauley et al. (1988), Salo (1991), and Johnson et al. (1997).

Egg

Female chum salmon lay between 900 and 8,000 eggs (Pauley et al. 1988) that range in weight from 248 to 349 mg (Beacham and Murray 1993). These eggs are extremely sensitive to changes in the environment, with a high degree of mortality (up to 90 percent) in the developing eggs (Pauley et al. 1988). Depending on environmental conditions, the eggs hatch to become alevins 50 to 130 days later. The alevins remain in the gravel another 30 to 50 days, until their yolk sac is absorbed (Pauley et al. 1988). Having depleted the energy stores in their yolk sacs, alevins emerge from the gravel as fry in the spring (Pauley et al. 1988).

Fry

Emerging fry are relatively large, ranging from 32 to 38 mm (Quinn 2005). While their time in freshwater is limited, these fry rely on aquatic insects as their primary food resource after their yolk-sac is absorbed. Most chum salmon fry spend only a few days to a few weeks rearing in fresh water before migrating toward marine habitats from March to May (Pauley et al. 1988; Salo 1991; Johnson et al. 1997; Quinn 2005). A much smaller number of fry may rear in freshwater streams but migrate to marine waters by the end of their first summer. Chum salmon utilize estuarine habitats for a few more weeks before migrating to coastal, then offshore waters.

Once reaching marine habitats, chum salmon form dense schools as they rear in nearshore habitats (Pauley et al. 1988; Salo 1991). The Hood Canal shoreline is said to serve as a nursery and rearing habitat for a significant portion of all chum salmon originating from Washington State rivers (Pauley et al. 1988). While in this environment, chum fry stay in very shallow, nearshore habitats and consume a number of epibenthic invertebrates, including gammaridean amphipods, harpacticoid copepods, cumaceans, and mysids (Pauley et al. 1988). This environment not only provides them with a readily available food resource, but also protects them from larger predators such as cutthroat trout. Most chum fry enter estuaries by June and leave them by mid to late summer.

Juvenile

By the time chum fry leave estuaries and begin moving to coastal waters, size becomes a key factor for their survival. Chum salmon that enter these waters when they are less than 55 mm are less likely to survive than larger juveniles (Pauley et al. 1988). For those fish that remain in estuaries through the summer, more typical sizes are 150 to 225 mm (Pauley et al. 1988). At this size juvenile chum generally begin to move to offshore neritic habitats, feeding on euphausiids, calanoid copepods, hyperiid amphipods, larvaceans, and fish larvae (Simenstad and Kinney 1978). A number of age 2 chum salmon do occur within Puget Sound waters (Pauley et al. 1988), although the absence of age 3 chum suggests that all chum salmon spend time rearing in the Pacific Ocean (Pauley et al. 1988).

Subadult

Once reaching coastal waters, the juvenile chum disperse and are widely distributed. Dietary studies of chum salmon along the British Columbia coast vary, with dominant food items of younger, maturing chum salmon including amphipods, euphausiids, pteropods, and fishes (Brodeur 1990). As chum grow larger and mature, their diet shifts to larger prey items, including amphipods, euphausiids, and crustaceans, but also an increase in squid and fishes (Brodeur 1990). In general, chum salmon originating from Washington streams and rivers, and rearing in the open ocean, do not return as mature adults until age 3 or 4 (Pauley et al. 1988).

Adult

Chum salmon can be reproductively mature as 3, 4, or 5-year-old fish (Pauley et al. 1988). In some systems, chum salmon are the last of the salmon to migrate upstream to spawn (Pauley et al. 1988). While most systems support only a summer and fall run of returning adult chum, Puget Sound rivers produce *early* (mid-August – October), *normal* (November – December), and *late* (January – March) runs (Pauley et al. 1988). In systems such as Hood Canal, chum salmon migration and spawning times are separated by up to a month, resulting in summer-run fish (spawning in early September to mid-October) and fall-run fish (spawning in early November to late December) (Johnson et al. 1997). Even though data are limited, coastal runs of chum salmon occur in October and November, with spawning taking place as late as December (Johnson et al. 1997).

As with other salmonids, adult chum may mill at a river's mouth, or estuary, for a period before initiating their upstream migration (Johnson et al. 1997). Once beginning their upstream migration, adult chum no longer continue to feed (Pauley et al. 1988). Chum salmon survival is limited once they enter fresh water. Studies have indicated that, once entering fresh water, adult chum may die within 11 days but may survive as long as a month (Pauley et al. 1988). Although chum salmon in Washington State generally spawn in the lower portions of rivers and streams, in some systems, such as the Skagit River, chum salmon migrate 170 km or more (Hendrick 1996, as cited in Johnson et al. 1997). After the female digs out a redd, and lays somewhere between 900 and 8,000 eggs that are immediately fertilized by a male, the pair defends the redd(s) until eventually dying in the following days.

e. Pink Salmon

Pink salmon (*Oncorhynchus gorbuscha*) are the most abundant salmon along the coast of the northeast Pacific Ocean and are also the smallest at maturity (Bonar et al. 1989; Heard 1991; Quinn 2005). Mature adult pink salmon are approximately 1.0 to 2.8 kg (2.2 to 6.2 lbs) when they return to their natal streams to spawn (Bonar et al. 1989; Heard 1991; Quinn 2005). Pink salmon only live for 2 years, with very little variability. In systems south of the Fraser River, most pink salmon adults return to their natal streams in odd years (Bonar et al. 1989; WDFW 2002; Quinn 2005). Twelve of the thirteen pink salmon populations in Washington spawn only in odd years, with an additional even-year population occurring in the Snohomish River (Hard et al. 1996). Pink salmon are more abundant in the northern river systems (e.g., Nooksack, Skagit, Stillaguamish, and Snohomish Rivers) than in southern river systems (e.g., Puyallup and Nisqually Rivers) (Hard et al. 1996). Though less abundant, populations also occur in Hood Canal (e.g., Hamma Hamma, Duckabush, and Dosewallips Rivers) and the Strait of Juan de Fuca systems (e.g., upper Dungeness, lower Dungeness, and Elwha Rivers) (Hard et al. 1996).

As pink salmon adults spawn near river mouths, and fry migrate downstream immediately after emergence, this salmon species spends the least amount of time in fresh water (Bonar et al. 1989; Heard 1991). Although some smaller coastal and Columbia River runs occur, within Washington State two of the rivers supporting the largest pink salmon runs are the Snohomish and Puyallup. These two rivers support commercial netting and sport fishing, activities that occur in marine, estuarine, and freshwater portions of the respective systems.

Within four to six weeks of entering estuaries, the adults go through a transformation, with the males developing a large hump on their backs, enlarged heads, and the development of large teeth (Bonar et al. 1989; Heard 1991). The hump in male pink salmon is so pronounced compared to other salmonids it has resulted in the species being nicknamed *humpies*. Following spawning, the vast numbers of spawned out pink salmon begin to die off, with dead and dying adult pink salmon littering the banks and shallows. A variety of predators and scavengers (e.g., eagles, bears, and gulls) utilize this readily available food resource (Heard 1991). The remaining carcasses decay, with their bodies providing nutrients in the nearby downstream vicinity of their spawning grounds. A summary description of the life history characteristics of pink salmon is provided below. For an in-depth summary of pink salmon life histories please refer to Bonar et al. (1989), Heard (1991), and Quinn 2005).

Egg

The eggs produced by pink salmon are smaller (about 6 mm in diameter) than other larger salmonid species (Bonar et al. 1989; Quinn 2005). Depending on the region and in-stream temperatures, fertilized pink eggs can take from five to eight months before fry emerge and migrate downstream (Heard 1991). Accounting for some variability between years and systems, pink salmon eggs laid in August to October generally hatch between late January and early March (Hard et al. 1996). Newly hatched alevin remain in the gravel for a few weeks utilizing their yolk sac. These alevin can average 21.3 mm in length.

Fry

With some variability, pink salmon fry emerge from the gravel primarily in March and April (Hard et al. 1996). Once the yolk sac is depleted, the alevins emerge as fry some 41 to 64 days (average 52 days) post-hatching. These newly emerged fish range from 29 to 33 mm long with an average length 31.5 mm (Heard 1991; Quinn 2005). There is very little or no freshwater rearing as pink salmon fry migrate seaward upon emergence from the gravel, and so their downstream migration also occurs in March and April (Hard et al. 1996). Pink salmon originating from Puget Sound and Hood Canal streams and rivers appear to use nearshore areas extensively for early rearing during their first few weeks of entry into marine habitats (Jewell 1966, as cited in Hard et al. 1996). While little is known about their behavior as the fry are exiting Puget Sound proper, Hiss (1994, as cited in Hard et al. 1996) found that fry occurrence in Dungeness Bay (near Sequim) peaked in April and they were gone by late May. These fish ranged in length from 35 to 75 mm. Findings suggest that most out-migrating pink salmon enter the open ocean by late summer or early fall (Hard et al. 1996). However, like some Chinook and coho, a small portion of the pink salmon population appears to adopt residency in Puget Sound for the marine phase of the life cycle (Hard et al. 1996).

Juvenile/Subadult

Pink salmon fry migrate so rapidly to the sea, and so little is known about the juvenile and subadult behavior at sea, that for pink salmon the juvenile and subadult life history categories are combined. Once reaching estuarine and marine habitats, pink salmon migrate toward the open ocean within the first couple of months. By September the majority of pink salmon migrate hundreds of miles out in the open sea to grow and mature (Hartt and Dell 1986, as cited in Hard et al. 1996; Bonar et al. 1989; Heard 1991). Stomach contents of ocean-caught pink salmon were found to include fish, amphipods, euphausiids, pteropods, salps, squid, crustacean larvae, chaetognaths, and polychaetes (Bonar et al. 1989; Brodeur 1990; Pearcy et al. 1984). They spend approximately eighteen months rearing in the open ocean before their eastward migration to their natal streams and rivers (Bonar et al. 1989; Heard 1991).

Adult

Adult pink salmon begin migrating toward the river mouth of their natal stream from June to September, with peaks in some systems, such as the Snohomish, in the month of August (Heard 1991; Hard et al. 1996). Most pink salmon returning to Washington rivers and streams are believed to travel east through the Strait of Juan de Fuca, with some additional use of the Johnstone Strait/Strait of Georgia corridor (Hard et al. 1996). Once reaching their natal streams, adult pink salmon, in general, do not migrate far upstream, preferring to spawn within a few miles of the estuary. However, in the Snohomish River pink salmon spawn as far upstream as the Sultan River (approximately 23 miles), in the Puyallup River primarily above 12.5 river miles, and in the Nisqually River main stem between river mile 22 and 40 (Hard et al. 1996). In Washington and southern British Columbia, spawning generally occurs from August to October (Hard et al. 1996). Adult pink salmon spawn in dense numbers near riffles with clean gravel, an activity that takes from one to eight days (Heard 1991). After spawning the adults aggressively defend their respective redds until they eventually die, 11 to 21 days later (Heard 1991).

f. Steelhead

Steelhead (*Oncorhynchus mykiss*) range from central California to the Bering Sea and Bristol Bay, Alaska (Pauley et al. 1986). This species can occur as both anadromous (steelhead) and resident (rainbow trout) fish (Pauley et al. 1986). This brief life history summary focuses on naturally spawning, or *wild*, steelhead populations, and not the resident rainbow trout or hatchery produced stock of either form. A significant difference between steelhead and salmon is that steelhead, in fresh water, can maintain a life history as rainbow trout, and steelhead can be repeat spawners. Adult spawners of either steelhead or rainbow trout can produce either, or both, freshwater or anadromous juveniles (Quinn 2005). The discussion below focuses on the anadromous, or steelhead, form of the species.

Comparing the size of an average steelhead to a given salmon species needs to take into account the age of a given steelhead. Unlike some of the larger salmonids that die after spawning, steelhead can return to spawn in multiple years. As a result, steelhead can spawn up to four times and have been documented to live as long as 8 or 9 years (Pauley et al. 1986). Because steelhead grow larger in the productive marine environment, fish that stay in these habitats longer are typically larger. Studies investigating this have found that steelhead range in size from 47 cm (18.5 inches) for a 1-year saltwater resident to 88 cm (34.6 inches) for a 4-year saltwater resident (Maher and Larkin 1954, as cited in Pauley et al. 1986). Steelhead are prevalent throughout streams and tributaries of the Puget Sound and Columbia River (Pauley et al. 1986). Both winter and summer steelhead types, or races, occur within Washington State streams and rivers.

Typically adult steelhead return to streams and rivers in the winter or summer and spawn in the spring and summer, with fry emerging in just a few weeks. Upon emergence, steelhead typically rear in the freshwater streams and rivers between 1 and 3 years. Following their downstream migration to marine waters, these fish rear and mature in the ocean for 1 to 3 years before returning to freshwater systems as adults to spawn (Pauley et al. 1989; Quinn 2005). Because steelhead can be repeat spawners, the age and size of returning adults varies considerably. A summary description of the life history characteristics of steelhead is provided below. For an in-depth summary of steelhead life histories please refer to Pauley et al. (1986) and Quinn (2005).

Egg

Steelhead eggs are typically smaller than salmon eggs, with size varying based on maternal size and body condition. Though timing for egg laying and hatching varies between river systems and temperature conditions, eggs of both summer- and winter-run fish begin to hatch from April to June, 4 to 7 weeks following adult spawning. Alevins emerge soon after, rapidly absorb their yolk, and are free swimming within 3 to 7 days (Pauley et al. 1986). These newly emerged fish are active foragers on microscopic organisms, moving toward insects and larger prey items as they grow.

Fry

By the summer of their first year, steelhead fry are consuming bottom-dwelling aquatic and terrestrial insects (Pauley et al. 1986). As they continue to grow, young steelhead fry become parr. Steelhead parr can remain in fresh water from 1 to 4 years, although 2 to 3 years is more common. Steelhead parr actively forage from spring to fall but become relatively inactive during winter months (Bustard and Narver 1975, as cited in Pauley et al. 1986). It is at the parr stage that steelhead begin their downstream migration toward marine habitats.

Juvenile

Juvenile steelhead migrate downstream typically in spring (although this can be from January to July) of their second, third, or fourth years in freshwater habitats, finally entering marine waters and becoming smolts. Size at smoltification in steelhead has been correlated with survival. Small steelhead entering marine habitats do not survive long. Ideally a steelhead smolt should be at least 14 to 16 cm (5.5 to 6.3 inches) to survive this critical period (Pauley et al. 1986). Once entering estuaries, juveniles migrate toward coastal waters for a short period before moving offshore and into the open ocean. While in the ocean, juvenile steelhead diet shifts again and includes amphipods, euphausiids, a variety of fish, and squid (Pauley et al. 1986; Brodeur 1990).

Subadult

While in the ocean, subadult steelhead grow rapidly. However, upon entering the ocean, instead of forming schools they disperse widely. As a result, little is known about their distribution in marine habitats, other than distribution conforming between the 5°C isotherm to the north and the 15°C isotherm to the south (Pauley et al. 1986). Subadult steelhead diet while in the ocean appears to vary between regions, which may be a factor of available food resources. For those fish that occur off Oregon and Washington, cephalopods represent nearly 80 percent of the diet with fish representing only 10 percent (Brodeur 1990). Similar aged steelhead in the Gulf of Alaska had diets that consisted of 63 percent fish and 31 percent squid (Brodeur 1990). Although Columbia River summer-run steelhead generally only spend one summer in the ocean, other stocks, including winter-run, spend up to 4 years in the ocean, although 2 to 3 years is more common (Pauley et al. 1986).

Adult

Stream-maturing (summer) adult steelhead typically enter rivers from August to September, remain in the river through fall and winter, then spawn the following spring (Pauley et al. 1986; Quinn 2005). Ocean-maturing (winter) steelhead in Washington return during March to April and may spawn within a month of entering fresh water (Pauley et al. 1986; Busby et al. 1996, as cited in Quinn 2005). Returning adult steelhead rarely eat and grow little if at all once entering fresh water to spawn (Maher and Larking 1954, as cited in Pauley et al. 1986). Unlike salmon, steelhead do not die after spawning and can spawn up to four times, although two to three is more common (Pauley et al. 1986). As a result, post-spawn adult steelhead migrate back toward marine habitats within a relatively short period of time after spawning.

II. Salmonid body burdens

Over the course of a given salmonid lifespan, these fish occupy a variety of habitats that meet specific needs for each lifestage. As they grow from one lifestage to the next, and move from one habitat type to the next, their growth is based on the available food resources (Healey 1991, Quinn 2005; Duffy 2009). During each period of growth, each lifestage has the potential to accumulate environmental toxins. This section reviews and summarizes the information provided in the draft TSD, and the comments received on the draft TSD. Finally, this section includes a summary review of salmonid body burden accumulation. This review refers back to the life history summaries described above in an effort to determine whether it may be appropriate to attribute body burden accumulation to specific sites.

A. Summary of salmonid body burden information in the draft TSD

The draft TSD was written to address human consumption of fish and not the biology of Washington State salmonids. However, some explanation of salmonid life history and general region of body burden accumulation is warranted. Appendix E of the draft TSD, titled *The Question of Salmon*, summarized the life cycle and survival strategies of selected Washington State salmonids.

Appendix E acknowledges that, by the year 2000, an estimated 58 percent of all salmon consumed worldwide was farmed salmon. However, the draft TSD indicated people of Washington State not only consume more salmon than other populations, but much of this salmon comes directly from wild-caught resources in the Pacific Northwest region. As a result, accumulation of body burden by non-farmed salmonids in Washington State is relevant when evaluating human health risks associated with consumption of the resource.

The Appendix E section titled *Salmonid Contaminant Body Burden* summarizes the role that salmonids play in the distribution of bioaccumulative pollutants. Appendix E acknowledges that "Pacific salmon exposure to PBTs, and PCBs in particular are, in part, contingent on migratory patterns, residency time in Puget Sound, proximity of the salmon to contaminated sediments, waste sites, and different behavior and dietary patterns as the fish mature." Included in this summary is the finding of O'Neill et al. (1998) "…that chinook and coho salmon accumulate most of their PCB body-burden in the marine waters of Puget Sound and the ocean, and because chinook salmon live longer and stay at sea longer than coho salmon they accumulate higher PCB concentrations in their muscle tissues." Further, more than 98 percent of the final body weight of most salmon is attained at sea. Appendix E cites the finding of O'Neill et al. (2006) that coastal migrants (coho and ocean-type Chinook) have greater body burdens than do more oceanic migrants (chum, pink, and sockeye). Further, this study found that resident Chinook, which tend to be ocean-type, fall-run fish, were found to have 2 to 6 times the amount of polychlorinated biphenyls (PCBs) as non-residents, and 5 to 17 times the polybrominated diphenyl ether (PBDE) burden. A final point made in Appendix E with respect to regional variations in accumulation is that tributaries in the far southern reaches of Puget Sound lengthen the time that these fish occur in the Sound. This is due to greater distances from natal streams to the ocean for both out-migrants and in-migrants.

B. Summary of salmonid body burden information from the draft TSD comments

Public review comments on the draft TSD represented a wide range of positions with respect to where and when in a salmon's life cycle it accumulates significant portions of its toxic body burden, and how this information should be incorporated into the draft TSD. These comments are summarized below.

One reviewer (Yakama 2012) noted that salmon have been shown to acquire contaminants in waters that are under Washington State jurisdiction. They pointed out that accumulations occur within fresh waters of the Columbia River for juveniles during their out-migration (LCREP 2007) and in estuarine and inland marine waters of Washington (O'Neill 2011, as cited in Yakama 2012).

Another reviewer (Exponent 2012) pointed out the draft TSD Appendix E appeared to support the conclusion that, for most salmon, body burden of bioaccumulative chemicals (e.g., PCBs, toxins, mercury) derives mostly from marine waters. They noted that "Washington waters/sediments may contribute to body burden depending on the species, run, chemical, life cycle characteristics, and range of environmental physical characteristics," and concluded that "variability, dominated by a lack of significant contribution, argues for evaluating the situation on a site-specific basis with the exclusion of salmon being the default."

A few comments received on the draft TSD (Weyerhaeuser 2012; Nippon Paper 2012) referred to their support of the 2003 AMEC paper titled *Evaluation of the Fish Consumption Rate Selected by Oregon DEQ for the Development of Ambient Water Quality Data* and this paper's relevance to the reviewed draft TSD. One of the points made by the AMEC (2003) document with respect to freshwater and estuarine occurring fish was that "If there is a permitted discharge to a freshwater body, the consumption of estuarine fish and shellfish is likely to be irrelevant. Similarly, if there is a discharge to an estuarine area, the freshwater fish upstream will likely not be affected. Thus the inclusion of rates of consumption of freshwater and estuarine finfish and shellfish is a very conservative assumption for these specific applications" (page 5).

The AMEC (2003) paper called into question the inclusion of anadromous fish, which are consumed by Columbia River Tribal members at rates of three times higher than resident fish species, according to the CRITFC (1994) fish consumption survey. They noted that many juvenile Chinook salmon in the Columbia River spend several months in the system before migrating to a marine feeding area, and that when they return as adults after 2 to 6 years, they generally do not feed during their upstream spawning migration, and therefore do not reach equilibrium with the surrounding environment. In other words, though potentially occurring within a discharge area, migratory fish species such as Chinook salmon do not occur in any site-specific region for a sufficient length of time to achieve equilibrium with regulated compound concentrations in the associated water column.

One reviewer (NCASI 2012) provided extensive comments, citing many of the same references that were cited in the draft TSD. They point out that different species and different runs of the same salmon species have very different life history strategies and will accumulate body burdens to differing degrees. They state that literature supports the position that the major fraction of persistent bioaccumulative toxic (PBT) burden carried by adult salmon originates from the open ocean. Their comments noted that resident Puget Sound Chinook, with higher body burdens, are an exception as they do not utilize open ocean habitats. Due to this difference, they noted that "…it might be appropriate to assess risk to select Puget Sound residents as a separate activity, and inclusion of salmon in an FCR [fish consumption rate] used in such a risk assessment may well be warranted." However, due to the large differences in life history, the reviewer felt that it is inappropriate to lump all species together as *salmon*, and should instead apportion numbers between salmon species. Their conclusion was that, based on the information provided above, the consumption of salmon should be excluded from any state-wide default fish consumption rate.

C. Salmonid body burden accumulations by life stages, rates, and location

This section includes a summary of the detailed technical literature review comments provided by one specific reviewer, and an additional literature review concerning salmonid body burden accumulation conducted during preparation of this Technical Issue Paper.

a. NCASI review comments (Should salmonid body burden be included as part of the fish consumption rate?)

A key question asked by a number of reviewers of the draft TSD was whether salmonid body burden accumulation is sufficiently understood to be included as part of the default fish consumption rate. One reviewer in particular, NCASI (2012), provided a summary of their literature review on where salmonids accumulate PBT chemicals. A summary of their review is included here, as well as an expanded review of additional literature, to further understand what the current knowledge is for the rate of uptake, locations where uptake occurs, and those salmonid species and life stages for which this information is available.

The NCASI (2012) comments include a table taken from Fresh et al. (2005), that compared the life history variations for six Washington State salmonid species (NCASI [2012] cited the Fresh paper as NOAA 2005). NCASI (2012) comments stated that… "These differences are potentially significant in that they may lead to differences in the mass (burden) of the chemical contaminants (e.g., PBT chemicals) ultimately accumulated by the salmon, and in the fraction of this ultimate burden accumulated in freshwater vs. saltwater." Section I.C of this Technical Issue Paper was prepared to provide an even greater understanding of what those life history differences are, where these six salmonid species occur at given stages of their life, and the principal food items consumed during each life stage. This Technical Issue Paper is not an exhaustive summary, but one that allows for comparison on an appropriate scale.

NCASI (2012) stated that, taken to the extreme, the life history differences of these salmonids imply that each run of each salmonid should be evaluated independently to determine where contaminants are accumulated. They further stated that this comparison may not be necessary, as the work of O'Neill et al. (1998), West and O'Neill (2007), and O'Neill and West (2009) indicated that the open ocean is the dominant pathway for PBT chemical uptake by salmonids. The NCASI review provided additional references that are included to show that the majority of contaminant burden in anadromous fish occurs in marine systems and not freshwater systems. A final, albeit very notable, distinction made in their review was the finding by O'Neill et al. (1998) that indicated a greater uptake of PBTs in Chinook, in both Puget Sound proper and its tributaries, relative to other salmonids. Findings such as this are likely attributable to two factors: (1) resident Puget Sound Chinook are generally ocean-type (rearing in marine waters rather than extensive periods in fresh water), and (2) they eat more fish, including contaminated herring stocks, as a part of their food resource than other salmonids.

b. Summary of investigations concerning salmonid body burden accumulation

There is growing evidence in the literature that Puget Sound fish, including salmonids, are experiencing biological effects attributable to contaminant exposure (McCain et al. 1990; Stein et al. 1995; Landahl et al. 1997; O'Neill et al. 1998; West and O'Neill 2007; O'Neill and West 2009; Hope 2012). Although there is evidence that adult salmon returning to spawn in Washington waters have elevated body burdens of these contaminants, the origin of these contaminants is of central importance for resource management and risk reduction. What follows is a summary of selected papers that investigated body burden accumulation in salmonids.

Juvenile accumulation

In situ biomonitoring of caged, juvenile Chinook salmon (*Oncorhynchus tshawytscha*) in the Lower Duwamish Waterway (Kelley et al. 2011):

> Kelley et al. (2011) conducted a series of 8 to 10 days in situ caged fish exposure studies in the Lower Duwamish Waterway (LDW) each July, over a 4-year period (2004–2007). Juvenile ocean-type Chinook salmon (*Oncorhynchus tshawytscha*) were obtained as eggs or fry from the University of Washington hatchery and reared at the NOAA Fisheries marine lab in Mukilteo, WA. They were raised in freshwater and acclimated to seawater in June of each year. The exposure period was chosen to be well below previous studies, indicating that residence time in these waters for naturally occurring juvenile Chinook tends to be much greater (ranging from 1 week to 2 months). Among other investigative methods, they used fluorescent aromatic compounds (FACs) in Chinook salmon bile to evaluate the relationship between contaminated sediment and ecological receptors for fish caged in the LDW. At each caged fish location they collected sediment and water sample data.
>
> Their findings indicated that biomarkers among field-exposed Chinook were greater than those in control sites. In fact, deoxyribonucleic acid (DNA) adduct analysis indicated there were significant differences between the test subjects and the controls, with the exception of 2007 data, a period when in-water contaminant levels were at their lowest. Further, tissue body burden of yearly mean and the standard error of mean (SEM) polycyclic aromatic hydrocarbon (tPAHs) were significantly different during the 2005 study period when some of the highest tPAHs were detected in water samples (Figure 1), which happened to coincide with the only time total PCBs (tPCBs) were detected in water samples. The findings of Kelley et al. (2011) indicated that juvenile Chinook, occurring for a relatively short period of time in a contaminated estuarine environment, can experience elevated contaminant body burden. Similar conclusions were reported by other authors, indicating that contaminant accumulation by salmonids can occur within freshwater and estuarine systems (Giesy 1999; Meador et al. 2002; Hardy and McBride 2004; Sethajintanin et al. 2004; Fresh et al. 2005; Johnson et al. 2007; Yanagida 2012).

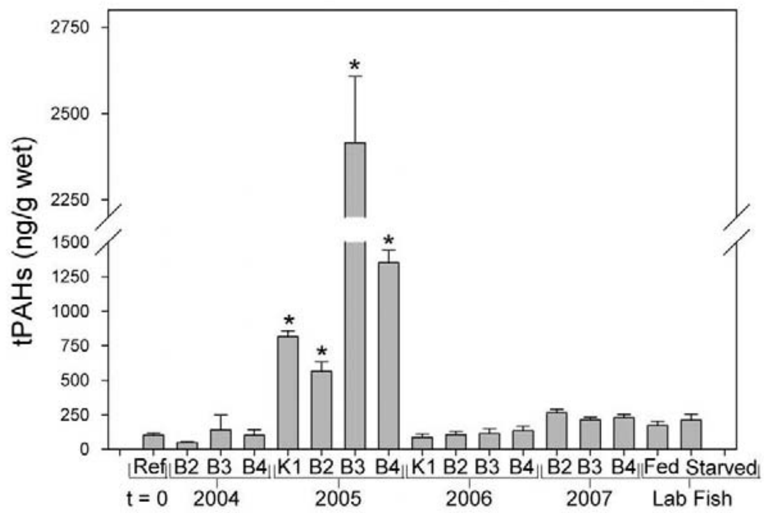

Fig. 5. Mean and SEM total PAHs detected in juvenile Chinook salmon tissue composites, illustrating the significant spike across stations in 2005, following an 8–10 days caged exposure (depending on study year) in the Lower Duwamish Waterway, compared to fish that were not exposed ($t = 0$). Fish that were caged at the Mukilteo lab ± feeding were also included. Typically 3–8 fish per were used per composite and $n = 4$ for each treatment group. *Statistically significant difference from control.

Figure 1. Mean and SEM total PAHs detected in juvenile Chinook salmon tissue composites

Source: Kelly et al. 2011

Studies of uptake, elimination, and late effects in Atlantic Salmon (*Salmo salar*) dietary exposed to di-2-ethylhexyl phthalate (DEHP) during early life (Norman et al. 2007):

> Norman et al. (2007) investigated the uptake, elimination, and late effects in Atlantic salmon of hatchery origin where juveniles were exposed to orally administered, and food contaminated with, di-2-ethylhexyl phthalate (DEHP). They found diet was the major route for uptake of lipophilic compounds. In addition, they found that for this compound and its metabolite, salmon were able to excrete the compounds, with body burden concentrations approaching background within one week of exposure. The exposure levels in this study demonstrated that dietary exposure to DEHP could temporarily increase body burden, but if no longer exposed can excrete DEHP to levels more consistent with background, and have no latent effects on growth or survival to adulthood. However, they did find that repeated exposure to very high doses that did not allow the fish to return to background levels could alter gonadal development in males.

In situ biomonitoring of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) using biomarkers of chemical exposures and effects in a partially remediated urbanized waterway of the Puget Sound, WA (Browne et al. 2010):

> Browne et al. (2010) investigated the contaminant effects of Duwamish Waterway PAHs and PCBs on caged juvenile Chinook salmon using in situ biomonitoring and molecular biomarker analysis. In July 2007, approximately 20 pre-smolt Chinook were placed in four cages for eight days at adjacent sites in the Duwamish. Both subject and control fish were of hatchery origin. Hatchery-maintained control reference Chinook were not fed because juvenile fish caged in the Duwamish system have limited access to prey in the water column and do not feed. Biomarkers were compared to control cages of fish at an upstream hatchery. Prior to

cage placement, sediment samples were collected and analyzed at the selected cage locations. High concentrations of total PAHs in sediments were detected at one site, whereas two other sites had relatively low concentrations. Waterborne PAHs at all of the sampling sites were relatively low (<1 ng/L). Sediment PCBs at the sites ranged from a low of 421 ng/g to 1,160 ng/g, and there were no detectable waterborne PCBs at any of the sites (detection limit = 10 ng/L).

Browne et al. (2010) found no significant differences (<0.05) in biomarker gene expression in the Duwamish-caged fish relative to controls, although there was a pattern of gene expression suppression at the most heavily PAH-enriched site. The lack of a marked perturbation of mRNA biomarkers was consistent with relatively low levels of gill PAH-DNA adduct levels that did not differ among caged reference and field fish. Browne et al. (2010) found that there was low bioavailability of sediment pollutants for experimentally caged juvenile Chinook. Their conclusion was that these findings were potentially reflecting low waterborne exposures occurring at contaminated sites within the Duwamish waterway that have undergone partial remediation.

However, a limitation of the Browne et al. (2010) study may be duration of exposure. As the majority of Chinook populations in Puget Sound are ocean-type, fall-run fish, they generally enter estuaries at a smaller size and stay for a longer period than would spring type Chinook. A critical reviewer of this paper may propose an exposure period more consistent with naturally occurring pre-smolt Chinook to determine if the findings of no significant difference of PAHs and PCBs would stand.

Contaminant exposure in outmigrant juvenile salmon from Pacific Northwest estuaries of the United States (Johnson et al. 2007a):

Johnson et al. (2007a) investigated contaminant uptake in out-migrant juvenile salmon in the Pacific Northwest, including concentrations of PCBs, DDTs, PAHs and organochlorine pesticides in tissues and prey of juvenile Chinook and coho salmon from selected estuaries and hatcheries in the U.S. Pacific Northwest. Subyearling Chinook were collected using beach seines from five Washington and seven Oregon bays and estuaries from 1996 to 2001. To ensure field sampling captured wild fish instead of hatchery-reared fish, the authors attempted to collect fish from field sites prior to releases from hatcheries or other programs. The few hatchery fish that were captured in beach seines were not included in their analyses. Additional specimens were collected from hatcheries. Juvenile coho were collected from two Washington and three Oregon estuaries in 1998. In addition to tissue testing, bile metabolites and stomach contents were evaluated.

Johnson et al. (2007a) found that Chinook salmon had the highest whole body contaminant concentrations, typically 2 to 5 times higher than coho salmon from the same sites. Believed to be a function of high lipid content, hatchery Chinook body burdens of PCBs and DDTs were higher than estuarine Chinook salmon. Of the twelve estuaries where sampling occurred, concentrations of PCBs were highest in Chinook salmon from the Duwamish Estuary, the Columbia River, and Yaquina Bay. Each of these systems had fish exceeding the 2,400 ng/g lipid NOAA threshold for adverse health effects on fish. No significant differences were observed in PCB concentrations in coho salmon between sampling sites, though sample size was small.

Johnson et al. (2007a) found DDT concentrations in juvenile Chinook were highest in the Columbia River and Nisqually Estuary. PAH metabolites concentrations in bile were highest in Chinook captured in the Duwamish Estuary and Grays Harbor. When coho and Chinook salmon collected from the same sites were compared, ∑DDT concentrations were much lower in coho salmon. As PCBs, PAHs, and DDTs were consistently present in stomach contents at concentrations correlated with contaminant body burdens from the same sites, it is believed that contaminants in estuarine food resources contribute to out-migrant body burden.

Persistent organic pollutants in outmigrant juvenile Chinook salmon from the Lower Columbia Estuary, USA (Johnson et al. 2007b):

Johnson et al. (2007b) investigated exposure to several persistent organic pollutants (PAHs, PCBs, DDTs, and other organochlorine pesticides) in out-migrant juvenile fall Chinook salmon in the Lower Columbia River. They also evaluated the potential for adverse effects on salmon and the estuarine food web. Three regions were chosen for the collection of whole body and gut content samples: Columbia River estuary, Longview, and the confluence of the Willamette and Columbia Rivers. Approximately 30 juvenile Chinook were collected via beach seine in the summers of 2001 and 2002. Additional fish were captured using a purse seine in the Columbia River estuary. Chinook captured included a mix of both hatchery and naturally spawned fish. Once captured, fish were sacrificed for whole body and gut content analysis. The entire stomach content was collected post-mortem.

Johnson et al. (2007b) found average whole body concentrations of PCBs in juvenile Chinook ranged from 1,300 to 14,000 ng/g lipid (Figure 2). Average whole body concentrations of DDT were extremely high, ranging from 1,800 to 27,000 ng/g lipid (Figure 3). Whole body PCB and DDT concentrations tended to be associated with fish size, not distribution by river mile. The authors concluded that this finding was a correlation with estuarine residence time, as larger fish required longer residence time. With respect to prey items collected from gut contents, PCBs, DDTs, and PAHs were all found in salmon stomach samples, indicating that prey is a source of exposure. The authors concluded that since whole body contaminant concentrations were poorly correlated with nearby sediment concentrations, pelagic as well as benthic sources are important in determining salmon exposure. The findings of Johnson et al. (2007b) indicate that concentrations of DDTs and PCBs are elevated in out-migrant juvenile Chinook salmon in the Lower Columbia River, relative to salmon from undeveloped Pacific Northwest estuaries.

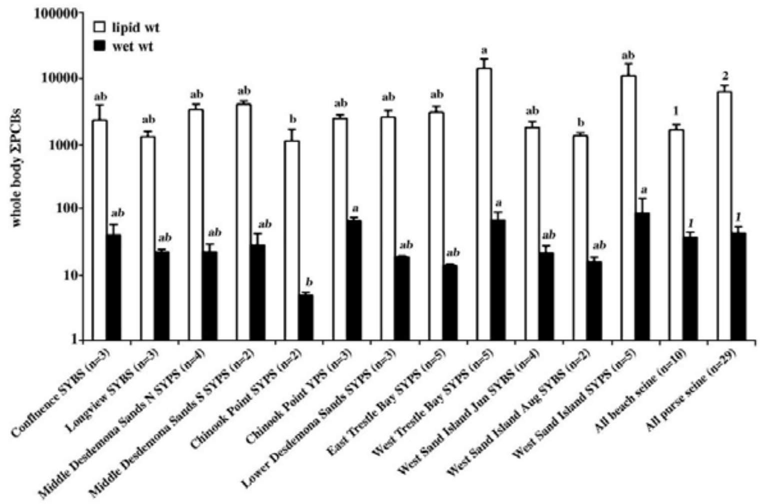

Fig. 4. Mean concentrations (±SE) of ΣPCBs (ng/g wet weight and ng/g lipid) in whole bodies of juvenile chinook salmon collected from the Lower Columbia River and Estuary. Beach seine samples are composites of 5–6 fish each, while purse seine samples are individual fish. Purse seine samples were collected in 2001 and beach seine fish were collected in 2002. Means with different letter designations are significantly different (ANOVA, Tukey–Kramer multiple range test, $p<0.05$); lower-case letters indicate comparisons among sites; upper-case letters indicate comparison between all beach seine and all purse seine fish.

Figure 2. Mean concentrations of total PCBs (ng/g wet weight and ng/g lipid) in whole bodies of juvenile Chinook salmon from the Lower Columbia River and Estuary

Source: Johnson et al. 2007b

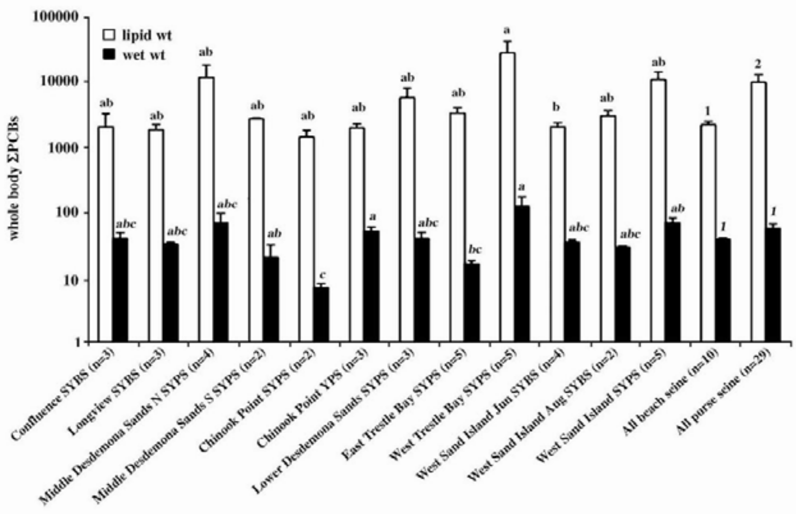

Fig. 6. Mean concentrations (±SE) of ΣDDTs (ng/g wet weight and ng/g lipid) in whole bodies of juvenile chinook salmon collected from the Lower Columbia River and Estuary. Beach seine samples are composites of 5–6 fish each, while purse seine samples are individual fish. Purse seine fish were collected in 2001 and beach seine fish were collected in 2002. Means with different letter designations are significantly different (ANOVA, Tukey–Kramer multiple range test, $p<0.05$); lower-case letters indicate comparisons among sites; upper-case indicate comparison between all beach seine and all purse seine fish.

Figure 3. Mean concentrations of total DDTs (ng/g wet weight and ng/g lipid) in whole bodies of juvenile Chinook salmon from the Lower Columbia River and Estuary

Source: Johnson et al. 2007b

Adult uptake and transfer of contaminants

Lipid reserve dynamics and magnification of persistent organic pollutants in spawning sockeye salmon (*Oncorhynchus nerka*) from the Fraser River, British Columbia (Kelly et al. 2007):

> Though accumulations of salmonid body burdens can and do occur during freshwater and estuarine occurrence of salmonids, the relative proportion of total adult body burden accumulated during juvenile stages has been questioned by some authors (O'Neill and West 2009). Kelly et al. (2007) investigated spawning sockeye salmon in the Fraser River. From June to October of 2001, they collected returning adult fish from as far downstream as 10 km, and as far upstream as approximately 1,200 km. Unfortunately, their complete methodology was not included in the paper, though their target analytes included PCBs and polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Their study found that the Fraser River is not a major source of sockeye salmon body burden accumulation of PCBs and PCDD/PCDFs, and that marine food sources and pathways are the important factors of persistent organic pollutants (POPs) in returning adults. However, other important findings were that magnification of toxics occurs during the development of eggs. Findings showed a three-fold increase of lipid toxic equivalents (TEQs) in eggs from open-ocean samples to those taken during spawning. This suggests that lipid reserve depletion during upstream migration can increase PCB and PCDD/PCDF concentrations in eggs as the adult fish prepare to spawn.
>
> Kelly et al. (2007) conclude by noting that other globally distributed dioxin-like compounds and/or endocrine disrupting chemicals (EDCs) such as polychlorinated napthalenes (PCNs), polybrominated diphenyl ethers (PBDEs), hexachlorocyclohexanes (HCHs), endosulfans, dialkyl phthalate esters (DPEs), and perfluoroalkyl compounds (PFCs) are undoubtedly present in tissues of these salmon. Should magnification and maternal transfer of these compounds also occur during spawning, toxicological impacts would affect returning adults and early life stage development. Unlike legacy pollutants such as PCBs and PCDD/PCDFs, many of these other current use chemicals of concern can be extensively discharged into urban/agricultural receiving waters such as the Fraser River. Future research should, therefore, focus on accumulation patterns and the cumulative and/or synergistic effects of PCBs, PCDD/PCDFs, and other EDCs of emerging concern on the reproductive health and population dynamics of Pacific salmon.

Acquisition of polychlorinated biphenyls (PCBs) by Pacific chinook salmon: An exploration of various exposure scenarios (Hope 2012):

> Hope (2012) used models to examine 16 different scenarios to identify where Oregon State water quality standards in place for the protection of human health might be effective at reducing PCBs in adult fall Chinook salmon. Model scenarios assumed a 130-day juvenile residency in fresh water, 50-day juvenile residency in estuaries, and adult fall Chinook returning after three to four winters in the ocean (1,860 days). A simplified summary of what Hope (2012) applied includes a bioenergetics model and a bioaccumulation model with estimates of prey consumption rates to estimate contaminant levels in a given fish on a given day at a given location. The model scenarios assumed that prey of adult salmonids included 20 to 60 percent herring, a substantial PCB source for adult Chinook. However, it was not clear if the various model scenarios took into account the region-specific differences in

herring PCB burdens described by West et al. (2008). While acknowledging that interpreting models must be done with caution, Hope (2012) concludes that his model scenarios suggest a limited ability of the Oregon water quality standards to meet the expectation of reducing contaminant loadings in anadromous species (in this case fall Chinook).

Marine distribution, life history traits, and the accumulation of polychlorinated biphenyls in Chinook salmon from Puget Sound, Washington (O'Neill and West 2009):

One of the most important papers for understanding body burden accumulations of Chinook salmon in Puget Sound is O'Neill and West (2009). They focused on three primary objectives; (1) compare PCB levels from Puget Sound Chinook salmon with other West Coast populations, (2) evaluate whether PCB accumulation mainly occurred in the freshwater or marine habitats, and (3) quantify the relative importance of fish age, fish size (fork length), lipid content, and saltwater age (the number of winters spent in salt water) on PCB concentration. Maturing and subadult Chinook salmon were sampled in August and September from 1992 to 1996. Muscle samples (total of 204) were collected from 763 individual Chinook. Only a portion of the fish sampled were of naturally spawning runs, with hatchery fish representing 98 percent of fish that migrated to salt water as yearlings (30 percent of the total fish sampled). As a result, a comparison of PCB levels between hatchery and wild fish was not possible. An aroclor analysis of the tissues was the selected method.

Though varying widely among samples, average concentration of PCBs measured in samples of skinless muscle tissue from subadult and maturing Puget Sound Chinook salmon was 53 ng/g fish tissue, which was three to five times higher than average concentrations reported for adult Chinook salmon from six other populations on the West Coast of North America (Figure 4) (O'Neill and West 2009). Chinook samples collected in the south and central portions of Puget Sound had the highest average concentrations (Figure 5). Rivers in the northern portion of Puget Sound, while still averaging approximately 40 ng/g, had the lowest concentrations of the seven separate groups within the Puget Sound sampling area (Figure 5). Although the proportion of Chinook that are caught in Puget Sound relative to the other six regions was not represented, estimates have indicated that approximately 14.5 percent of the Oregon and Washington commercial Chinook landings were from Puget Sound, with Treaty-Indian landings ranging from 72 to 95 percent of Puget Sound-caught Chinook salmon from 1991 to 2005 (O'Neill 2006). It is important to note that these estimates do not include sport-caught Chinook.

When comparing regions of body burden accumulation, the analysis of O'Neill and West (2009) indicated that, even in the most highly PCB-contaminated river draining into Puget Sound, the Duwamish River, the vast majority (>96 percent) of PCB accumulation occurred in the marine environment, with little freshwater contribution. They note that these findings are not surprising, given that Chinook salmon typically gain 99 percent of their total mass in marine habitats. They further estimated that yearling out-migrants (possibly stream-type fish) would acquire a similar percentage (>98.7 percent) of their total mass in marine habitats.

O'Neill and West (2009) found wide variations in contaminant burden for Puget Sound Chinook, suggesting that behavioral differences varied considerably among individual Puget Sound Chinook salmon. Variation may be related to fish size, and diet shifts that occur with size. They suggest that two mechanisms appear to be at work here: (1) growth dilution of

PCB associated with the addition of weight accumulated by older fish, and (2) a reduction in dietary PCB inputs associated with feeding on cleaner prey resources offshore for these older Chinook.

That the variation of contaminant burden within Puget Sound is region-specific is an indication that there may be minimal straying, or some geographic isolation that occurs, between north and south sound resident Chinook. Because the pelagic food web in Puget Sound is more heavily contaminated than that in the coastal waters (West et al. 2008), resident Puget Sound Chinook salmon, particularly those that occur within inner Puget Sound (an urbanized basin), experience a much more contaminated environment than non-resident populations. The urbanization of the inner Sound and the extended residence in these waters for Chinook originating from South Sound tributaries contributes to the findings that fish residing and feeding in central and southern Puget Sound probably would be exposed to higher PCB levels than fish feeding in northern areas of the sound.

In an investigation of inland marine herring stocks, a substantial portion of subadult and adult Chinook salmon diets, West et al. (2008) found that Pacific herring from central Puget Sound are more highly contaminated with PCBs than those from northern Puget Sound and the southern Strait of Georgia. As a result, O'Neill and West (2009) conclude that the wide range of PCB levels observed for Puget Sound Chinook salmon reflects their degree of residency and distribution while feeding within the inland marine waters. Although the authors concluded that Puget Sound Chinook PCB burdens appear to be a function of duration of residency and distribution while feeding within the inland marine waters, they also stated that behavioral differences of individual fish with respect to diet, overwintering, and movement within the inland marine waters was not fully understood.

As indicated by O'Neill et al. (2006) and Cullon et al. (2009), the body burden of Puget Sound salmonid populations has implications for populations utilizing these fish as a substantial portion of their diet. Although these studies focused on the effects salmonid body burdens would have on southern resident killer whale populations, the findings can be used as an indicator of health risks associated with the consumption of contaminated food resources from Puget Sound waters.

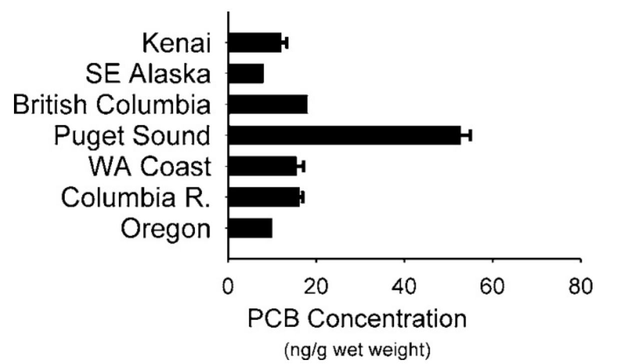

Figure 4. Mean PCB concentration in Chinook salmon fillets

Source: O'Neill and West 2009[1]

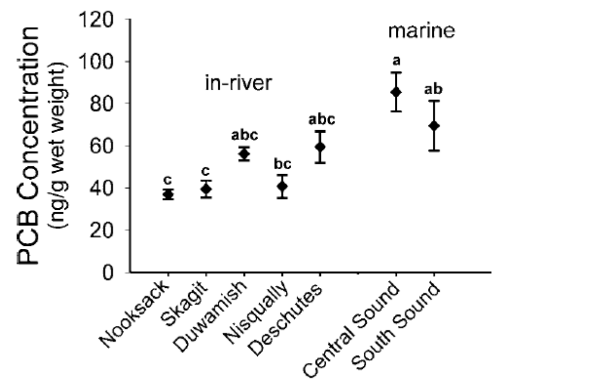

Figure 5. Mean PCB concentration in adult Chinook salmon returning to Puget Sound (in-rivers versus marine waters)

Source: O'Neill and West 2009
Error bars represent standard error around the means.

Persistent organic pollutants in Chinook salmon (*Oncorhynchus tshawytscha*); implications for resident killer whales of British Columbia and adjacent waters (Cullon et al. 2009):

> In a study designed to investigate the effects of contaminated food resources for Southern Resident Killer Whales, Cullon et al. (2009) measured POP concentrations in Chinook salmon (*Oncorhynchus tshawytscha*). A total of nine smolts and 24 adult Chinook were collected in 2000 and 2001 over an area ranging from Johnstone Strait to southern Puget Sound, at the Deschutes hatchery. Adult Chinook were collected from the central Strait of Georgia, whereas Washington fish were collected at the mouth of the Duwamish River and at a Deschutes River hatchery. Once collected, samples were prepared and analyzed for stable isotope analysis, contaminant analysis, and lipid determination. Scale analysis showed that all but three of the adult Chinook appeared to migrate to marine waters in their first year.

[1] Science Panel requested finding out what percentage of the total number of Chinook salmon consumed is represented by each of these seven regions. This information was requested from both authors of O'Neill and West (2009). Mr West suggests Ms. O'Neill is the most appropriate respondent. Ms. O'Neill is out of the country until approximately July 2012, but was able to respond in brief. She stated that this may be available with sufficient time to coordinate with the CTC, PFMC, and WDFW for the relevant publications, then create a summary.

Table 1. Estimated body burdens of persistent organic pollutants in returning adult Chinook salmon and Chinook smolts

Table 4. Estimated body burdens of persistent organic pollutants in returning adult chinook salmon (*Oncorhynchus tshawytscha*) from Johnstone Strait (n = 6) and Lower Fraser River (n = 6) (British Columbia, Canada); Duwamish River (n = 6) and Deschutes River (n = 6) (Washington, USA); and chinook smolts (n = 6) from the Strait of Georgia (British Columbia, Canada) and Puget Sound (n = 1 pool of 12) (Washington, USA). Body burden estimates calculated using POP lipid weight concentrations and whole fish lipid mass. Values represent mean ± standard error of the mean. One-way analysis of variance (ANOVA) tests were used to assess significant differences (α = 0.05) between the four adult salmon groups (ν = 23) and three adult groups for total dichlorodiphenyltrichloroethane (ΣDDT) and total hexachlorocyclohexane (ΣHCH) (ν = 17). Tukey post hoc tests were used to assess which groups differed (results in italics)

	Body burden (μg) (except for ΣPCDD and ΣPCDF)						
Sum congeners/ isomers[a] Group:	Strait of Georgia smolts	Johnstone Strait adults 1	Lower Fraser River adults 2	Puget Sound smolts	Duwamish River adults 3	Deschutes River adults 4	ANOVA test *(Tukey test)*
ΣPCBs as % returning burden	1.12 ± 0.40	141.54 ± 30.48 99.21%	537.58 ± 99.01 99.79%	0.03	216.32 ± 56.88 99.99%	339.62 ± 108.82 99.99%	0.017 *(1-2)*
ΣPCDDs (ng) as % returning burden	0.13 ± 0.04	8.99 ± 1.44 98.56%	10.14 ± 3.18 98.72%	0.005	5.50 ± 1.34 99.91%	9.76 ± 3.07 99.95%	0.194
ΣPCDFs (ng) as % returning burden	0.21 ± 0.10	8.07 ± 2.30 97.40%	23.94 ± 8.05 99.12%	0.001	8.21 ± 1.87 99.99%	11.48 ± 2.01 99.99%	0.018 *(1-2; 2-3)*
ΣPBDEs	NA[b]	NA	165.75	NA	54.34	NA	ND[c]
ΣDDT[d] as % returning burden	0.15 ± 0.08	24.11 ± 6.30 99.38%	44.46 ± 8.00 99.66%	0.01	107.96 ± 28.08 99.99%	NA	0.003 *(1-3)*
ΣHCH[e] as % returning burden	0.04 ± 0.02	35.01 ± 3.74 99.88%	6.68 ± 1.21 99.40%	0.001	9.32 ± 1.55 99.99%	NA	0.000 *(1-2; 1-3)*

[a] PCB = polychlorinated biphenyl; PCDD = polychlorinated dibenzo-*p*-dioxin; PCDF = polychlorinated dibenzofuran; PBDE = polybrominated diphenyl ether; DDT = dichlorodiphenyltrichloroethane; DDD = dichlorodiphenyldichloroethane; DDE = dichlorodiphenyldichloroethylene; HCH = hexachlorocyclohexane.
[b] NA = not analyzed.
[c] ND = statistical comparison not possible.
[d] ΣDDT includes DDT (*o,p'*-DDT, *p,p'*-DDT), DDD (*o,p'*-DDD, *p,p'*-DDD), and DDE (*o,p'*-DDE, *p,p'*-DDE).
[e] ΣHCH includes (α-, β-, γ-) HCH.

Source: Cullon et al. 2009

Based on their analysis, Cullon et al. (2009) concluded that 97 to 99 percent of PCBs, PCDDs, PCDFs, dichlorodiphenyltrichloroethane (DDT), and hexachlorocyclohexane (HCH) in returning adult Chinook were acquired during their time at sea, not in fresh water or estuaries (Table 1). However, it is important to note that the sample size for this investigation, considering the large area investigated, was very limited. Additionally, although the samples originated from both hatchery and wild-caught fish, the study's results did not differentiate between the two types (possibly due to limited sample size). An interesting finding was that even though they were found to have lower lipid concentrations, Chinook collected further south had higher POP concentrations than Chinook sampled in more northern areas.

Thirty years of persistent bioaccumulative toxics in Puget Sound; Time trends of PCBs and PBDE flame retardants in three fish species (West and O'Neill 2007):

West and O'Neill (2007) were interested in evaluating Puget Sound PCB tissue concentration trends in English sole, Pacific herring, and coho salmon. They combined, screened for comparable methodologies, and analyzed Puget Sound Assessment and Monitoring Program (PSAMP) data that had been collected over a 30-year period. As the needs of this issue paper are for salmonid accumulations, a review of the English sole is not presented. However, as herring are an important food resource for salmonids in Puget Sound, their summary is included. Data were regionally organized into North, Central, and South Puget Sound. The analysis of PCB concentrations in herring showed that northern Puget Sound fish had much lower concentrations than those in central and southern portions of the sound. Concentrations of PBDEs in Pacific herring were roughly one-half to one-third of their PCB concentrations, with concentrations from 2001 to 2006 of central Puget Sound herring ranging from 40 to 85 ng/g wet wt., total PBDEs. The authors note that there was no apparent time trend in any stock, as early data were very limited.

The summary review by West and O'Neill (2007) of historic data indicated that PCBs in muscle tissue of coho salmon in central Puget Sound decreased over time from over 200 ng/g wet weight in one composite sample from 1975 to 86 ng/g in 1987, and generally less than 50 ng/g in the 1990s. Data were not presented for coho occurring in southern or northern portions of the sound. In comparison to coho, Pacific herring from central and southern Puget Sound stocks have remained nearly three times that of coho salmon from central Puget Sound. No species analyzed were found to have a reduction in PCBs from 1990 through 2005. West and O'Neill (2007) hypothesized that the lack of a declining trend in PCBs over the past 15 years is the result of biotic recycling of PCBs through the food web.

Spatial extent, magnitude, and patterns of persistent organochlorine pollutants in Pacific herring (*Clupea pallasi*) populations in Puget Sound (USA) and Strait of Georgia (Canada) (West et al. 2008):

West et al. (2008) investigated the geographic distribution and magnitude of three POPs in three Puget Sound populations and three Strait of Georgia populations of Pacific herring. Their summary on herring is included due to the herring's importance as a food resource for selected species of adult salmonids. Methods included mid-water trawls from 1999 to 2004, collecting a total of 1,055 pre-spawn herring for all locations combined. They found Puget Sound herring were 3 to 9 times more contaminated with PCBs, and 1.5 to 2.5 times more contaminated with DDTs, than Strait of Georgia herring (Figure 6). The authors concluded, considering the distinct differences between the populations, that the stocks appeared to be isolated from each other geographically, allowing for the environmental exposure to contaminants to be regionally expressed. Therefore, although herring represent a mobile source of POPs, they appear to stay regionally isolated. This is relevant to Puget Sound salmonid populations as heavily burdened Puget Sound herring are a notable source of bioavailable POPs in the Puget Sound food web, and their regional distribution may have a corresponding regional effect on their predators, including resident Puget Sound Chinook.

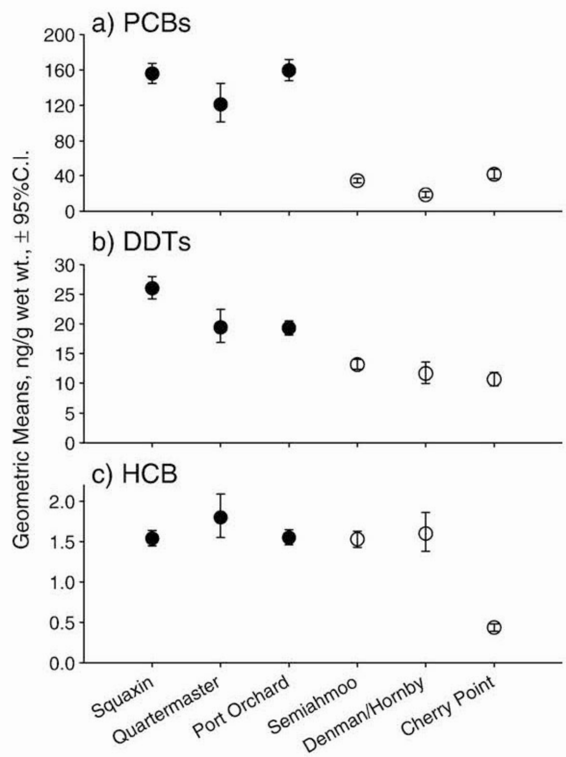

Fig. 2 – (a–c). Concentration of a) PCBs, b) DDTs, and c) hexachlorobenzene (HCB) in six herring populations from the Puget Sound (filled circles) and Strait of Georgia (open circles). Symbols indicate geometric means, and vertical bars define 95% confidence intervals.

Figure 6. Concentration of PCBs, DDTs, and hexachlorobenzene in six herring populations from Puget Sound and the Strait of Georgia

Source: West et al. 2008

III. Can salmonid contaminant body burden be attributed to a specific location?

A central question related to site-specific consumption is: can a salmonid's body burden be attributed to a location in which it occurs for a specific time in its life?

The life history strategies of different salmon species, and runs within given salmon species, are highly variable. Life history variation between salmonids begins at a very early age. Slight differences occur between species due to the time it takes for a given egg to hatch, with pre-hatching duration differences affected by environmental conditions, as well as species-specific differences. In general, eggs take some number of weeks to hatch, and the alevins remain in the gravel for a few more weeks until they consume their yolk sac reserves. It is at emergence from the gravel where life history patterns begin to vary substantially between and within species. Some salmon, such as pink, chum, and ocean-type Chinook salmon, migrate downstream within the first few weeks, with some exceptions. Other species, such as stream-type Chinook, coho, and sockeye salmon and steelhead, remain in freshwater streams or rivers to rear for 1 to 3 years before migrating toward marine habitats.

The differing behavior within species (e.g., stream-type vs. ocean-type Chinook) illustrates how difficult it is to make generalizations, even within a given salmonid species. Estuarine rearing is also highly variable between and within salmonids, with some species and types rearing for prolonged periods in these habitats, while others spend a few days to a few weeks during their shoreline out-migration. Salmonid food resources are also very different, which is likely a life history strategy that allows co-occurring species to partition the available resources and minimize direct competition. Some salmonids, both as juveniles and adults, use available planktonic food resources, some use epibenthic resources, while others as they grow larger are mainly piscivorous. Age at maturity varies between species; some species (pink salmon) reach maturity at age 2 with little variation. Other species are much more variable, reaching maturity at ages 3, 4, 5, or even 6 years of age (e.g., Chinook and chum).

A few salmonids (Chinook, coho, and pink) that originate from Puget Sound show some form of residency behavior (Laufle et al. 1986; Duffy 2003, 2009; Duffy et al. 2005). All or portions of these runs and/or stocks remain in Puget Sound waters for rearing and maturing, never migrating to coastal or open-ocean waters. However, likely because their residency forms are less abundant than their ocean-going forms, much less is known about possible resident coho and pink salmon than resident Puget Sound Chinook. While non-resident Chinook may gain as much as 98 percent of their total weight while in the ocean (Quinn 2005), resident Chinook complete their entire life from egg to returning adult in their natal stream and Puget Sound. Therefore, body burden accumulation for resident Chinook salmon may be an indicator of environmental conditions within Puget Sound.

Due to widely varying life histories, most notably the regions where salmonids occur and their dietary preferences, it is not appropriate to group all salmonids into a single category to serve as site-specific indicators for contaminants. In fact, as a number of authors indicate, almost all salmonids accumulate the vast majority of their body burden at sea; accumulation at juvenile life stages in freshwater and estuarine habitats contributes a very limited proportion of the total accumulation. There is some regional potential, however, for resident Puget Sound Chinook salmon.

It is not surprising that O'Neill and West (2009) found greater than 96 percent of PCB accumulation in Puget Sound Chinook occurred in marine habitats. Even more noteworthy, when taking into account the life history of these fish, is the finding of O'Neill and West (2009) that Puget Sound Chinook have three to five times higher PCB concentrations than those of other west coast populations. Considering that the majority of salmonids originating in Puget Sound are ocean-type, meaning they rear only for a short time in fresh water before migrating to estuarine and marine habitats, the total mass of a returning resident adult Chinook is primarily obtained in Puget Sound marine habitats. In addition, Puget Sound Chinook return to their natal streams to spawn having spent fewer years in marine habitats than some Chinook populations elsewhere. Therefore, Puget Sound resident Chinook salmon spend less time accumulating contaminants than many other populations, yet have a higher total body burden.

Another factor potentially relevant to this topic, briefly mentioned by a few authors but not investigated in detail, is the lack of understanding of whether there was a threshold response effect on juvenile salmonids exposed to contaminants while in freshwater and estuarine habitats. It is possible that the subadult and adult salmonids sampled for body burden analysis were those fish that did not experience behavioral and physiological abnormalities, post-exposure, that would have reduced their survival to adulthood. In other words, subadult and adult fish sampled may not be entirely representative of the naturally occurring juvenile population.

O'Neill and West (2009) showed that there is regional variation within Puget Sound waters and resident Chinook in more urbanized waters have greater body burdens. These findings suggest that a relationship does exist between contaminants and body burden in these fish. Nonetheless, due to their transient nature, these fish will probably not be able to serve as site-specific indicators. However, a more complete understanding of their duration of residency and distribution while feeding within specific regions of the inland marine waters may enable resident Puget Sound Chinook to serve as region-specific indicators in the future.

Summary

A review of the literature on northeast Pacific Ocean salmon body burden accumulation found that there were very few investigations of the origin of steelhead body burden and contaminants compared to other salmonids. However, given that steelhead can live many more seasons in marine environments than other co-occurring salmonids, with multiple freshwater entries for spawning, it would stand to reason that steelhead may have an even higher body burden accumulation than these other species. Although other salmonids along the west coast and in Puget Sound have lower body burden concentrations of contaminants, Puget Sound Chinook salmon have elevated PCB levels that appear to correlate with specific regions within Puget Sound. Nearly 22 percent of the maturing and subadult Chinook salmon samples collected from Puget Sound by Meador et al. (2002) had PCB concentrations above an effects threshold identified for salmonid fishes (i.e., 2,400 ng/g lipids), which included endpoints such as reduced growth, altered enzyme and hormone levels, and increased mortality.

O'Neill and West (2009) summarized that the most contaminated Puget Sound Chinook salmon in their investigation, believed to be Puget Sound residents, had concentrations comparable to Baltic Sea Atlantic salmon. Only Great Lakes salmon had higher concentrations of PCBs in North America, where Chinook salmon and coho salmon populations in the early 1990s were 20 to 30 times more contaminated with PCBs than the Puget Sound Chinook salmon investigated by O'Neill and West (2009). It is alarming that Puget Sound Chinook spend fewer years in marine habitats than many other salmonids, have a lower fat content in which lipophilic contaminants can be stored than spring-run Chinook, yet have much higher body burdens of PCB contaminants. These observations suggest that the food web within Puget Sound has a much higher concentration of PCBs than other rivers and estuaries in this region.

To understand where, when, and how salmonids accumulate their body burden, it is important to first note their life history, summarized in Section I, *Salmon Life History*. Regional occurrence is important: where do salmon occur over what portion of their life, and what are they doing in a given region that contributes to contaminant uptake (principally, what are the salmon eating and where are these prey residing (i.e., getting their exposure)?). While there are many differences between salmonids originating from Washington State streams and rivers, salmonid diet and where subadults/adults occur while they are maturing appear to be the most important factors. The dietary resource of a maturing salmon when it is maximizing growth (by mass) appears to be the principal factor for its eventual total body burden. That does not mean that salmonids do not accumulate toxins as fry or smolts, as various authors have shown this occurs. However, relative to the total body burden by the time a salmon reaches a harvestable size, juvenile contaminant uptake does not appear to be a significant factor.

Salmonid diet is important to consider when studying the uptake of contaminants because some salmonids prey on food resources higher up the food chain than others, increasing the total biomagnification. For example, a large portion of the pink and sockeye salmon diet is composed of planktonic invertebrates. By the time these short-lived prey become food resources, they have not lived long enough, or consumed a sufficient number of other prey items, to increase their potential for toxic loads. However, the larger species, notably coho and Chinook, will consume prey items higher on the food chain (e.g., Osmeridae and Clupeidae), increasing their exposure risk. Maturing Chinook salmon, more than any other salmonid, is the most piscivorous. The diet of a subadult or adult Chinook salmon includes a substantial amount of fish species, herring (planktivores) in particular. Feeding higher on the food chain may take advantage of a higher caloric resource, but it also means prey items have lived longer (2 to 3 years) and consumed more organisms. As a result, prey resources such as herring have been exposed to more contaminants and have an elevated body burden (West et al. 2008). For those salmonids preying on herring stocks in Puget Sound and the Strait of Georgia, the volume and origin of their food resources exposes them to the highest risk.

Studies have shown that the region in which a salmonid occurs during its growth from juvenile to subadult to adult is the most important factor for origin of contaminants in a given fish. For those salmonids that mature at sea, at least 96 percent of their POP body burden has been described as originating from oceanic waters. Studies have also shown that salmonids with the greatest burden are those that mature in polluted waters. Although some very small numbers of coho and pink salmon may not migrate to oceanic waters, the stock of greatest note is the resident Puget Sound Chinook. A number of authors showed that tissue analysis of this stock has a much higher body burden than ocean-migrating stocks. In addition, resident Puget Sound Chinook originating from southern Puget Sound tributaries have greater burdens than those to the north.

As noted above in Section III, the life histories of salmonids are too variable between and within species and the habitats they utilize to be used for site-specific evaluation. Ocean-maturing fish accumulate the vast majority of their body burden while at sea. However, for resident Puget Sound Chinook there does appear to be a correlation between salmon body burden and region of occurrence within the inland marine waters. Though the life history of these fish does not make them ideal candidates to serve as site-specific indicators, further understanding of their region-specific differences and origin of burden may enable them to serve as region-specific indicators in the near future.

References

Allen, M. A. and Hassler T.J. 1986. Species Profiles: Life Histories and Environmental Requirements of Coast Fishes and Invertebrates (Pacific Southwest) - Chinook Salmon. Report # U.S. Fish Wildl. Serv. Bio. Rep. 82(11.49). U.S. Army Corps of Engineers, TR EL-82-4.

AMEC Earth & Environmental 2003. Evaluation of the Fish Consumption Rate Selected by Oregon DEQ for the Development of Ambient Water Quality Criteria. Submitted to Northwest Pulp and Paper Association, Bellevue, WA.

Beacham, T.D., and C.B. Murray. Fecundity and egg size in North American Pacific salmon (Oncorhynchus). Journal of Fish Biology 42: 485-508.

Berkeley, S.A., M.A. Hixon, R.J. Larson, and M.S. Love. 2004. Fisheries Sustainability via Protection of Age Structure and Spatial Distribution of Fish Populations. Fisheries 29(8): 23-32.

Bonar, S.A., G.B. Pauley, and G.L. Thomas. 1989. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest)--pink salmon. U.S. Fish Wildl. Serv. Biol. Rep. 82(11.88). U.S. Army Corps of Engineers, TR EL-82-4. 18 pp.

Brodeur, R.D. 1990. A synthesis of the food habits and feeding ecology of salmonids in marine waters of the North Pacific. (INPFC Doc.) FRI-UW-9016. Fish. Res. Inst., University of Washington, Seattle. 38 pp.

Browne, E., M. Kelley, G-D. Zhou, L.Y. He, T. McDonald, S. Wang, B. Duncan, J. Meador, K. Donnelly, and E. Gallagher. 2010. In situ biomonitoring of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) using biomarkers of chemical exposures and effects in a partially remediated urbanized waterway of the Puget Sound, WA. Environmental Research 110(2010): 675-683.

Burgner, R.L. 1991. Life History of sockeye salmon (*Oncorhynchus nerka*). Pp 3-115 *In* Groot, C., and L. Margolis (Eds.) Pacific Salmon Life Histories. UBC Press, University of British Columbia, Vancouver, Canada.

Cullon, D.L., M.B. Yunker, C. Alleyne, N.J. Dangerfield, S. O'Neill, M. Whitcar, and P.R. Ross. 2009. Persistent organic pollutants in Chinook salmon (*Oncorhynchus tshawytscha*); implications for resident killer whales of British Columbia and adjacent waters. Envorn. Toxicol. and Chem. 28(1): 148-161.

CRITFC (Columbia River Inter-Tribal Fish Commission). 1994. A Fish Consumption Survey of the Umatilla, Nez Perce, Yakama, and Warm Springs Tribes of the Columbia River Basin. Technical Report 94-3. October 1994. 183 pp.

Davis, N.D., M. Fukuwaka, J.L. Armstrong, and K.W. Myers. 2005. Salmon food habits studies in the Bering Sea, 1960 to present. North Pacific Anadromous Fish Commission Technical Report No. 6. 5 pp.

Duffy, E.J. 2003. Early marine distribution and trophic interactions of juvenile salmon in Puget Sound. University of Washington, Seattle. Master's Thesis. 186 pp.

Duffy, E.J. 2009. Factors during early marine life that affect smolt-to-adult survival of ocean-type Puget Sound Chinook salmon (*Oncorhynchus tshawytscha*). University of Washington, Seattle. Doctoral Thesis. 164 pp.

Duffy, E.J., D.A. Beauchamp, N.C. Overman, and R.L Buckley. 2005. Hatchery Scientific Review Group Research Grant Project Annual Report I. Marine Distribution and Trophic Interactions of Juvenile Salmon in Puget Sound: A Synthesis of Trends among Basins. August 29, 2005. 28 pp.

Ecology. 2011. Fish Consumption Rates Technical Support Document. A Review of Data and Information about Fish Consumption in Washington. Publication No. 11-09-050, Version 1.0. Washington State Department of Ecology, Toxics Cleanup Program, Olympia, WA. September 2011.

Fleming, I.A., and M.R. Gross. 1990. Latitudinal clines: a trade-off between egg number and size in Pacific salmon. Ecology 7(1): 1-11.

Fresh, K.L., E. Casillas, L.L. Johnson, and D.L. Bottom. 2005. Role of the estuary in the recovery of Columbia River basin salmon and steelhead: an evaluation of the effects of selected factors on salmonid population viability. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-69, 105 p.

Giesy, J.P., K. Kannan, J.A. Kubitz, L.L. Williams, and M.J. Zabik. 1999. Polychlorinated Dibenzo-p-Dioxins (PCDDs) and Dibenzofurans (PCDFs) in muscle and eggs of salmonid fishes from the Great Lakes. Arch. Environ. Contam. Toxicol. 36:432-446.

Gustafson, R.G., Wainwright, T.C., Winans, G.A., Waknitz, F.W., Parker, L.T., and Waples, R.S. 1997. Status review of sockeye salmon from Washington and Oregon. NOAA Tech. Mem. NMFSNWFSC-33.

Hard, J. J., R.G. Kope, W.S. Grant, F.W. Waknitz, L.T. Parker, and R.S. Waples. 1996. Status review of pink salmon from Washington, Oregon and California. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-25. Seattle, Washington. 131 pp.

Hardy and McBride. 2004. Evaluation of Contaminants in Fish from Lake Washington, Final Report. Prepared by the Division of Environmental Health for the Washington State Department of Health. September, 2004. 80 pp.

Healey, M.C. 1991.Life History of Chinook salmon (Oncorhynchus tshawytscha). Pp 313-393 In Groot, C., and L. Margolis (Eds.) Pacific Salmon Life Histories. UBC Press, University of British Columbia, Vancouver, Canada.

Heard, W.R. 1991. Life History of pink salmon (*Oncorhynchus gorbuscha*). Pp 121-230 *In* Groot, C., and L. Margolis (Eds.) Pacific Salmon Life Histories. UBC Press, University of British Columbia, Vancouver, Canada.

Hearsey, J.W. 2011. Run timing, fat content and maturity of spring- and fall-run Klamath River Chinook Salmon (*Oncorhynchus tshawytscha*). Master's Thesis, Humboldt State University, Humboldt, California. 81 pp.

Hendry, A.P., J.E. Hensleigh, and R.R. Reisenbichler. 1998. Incubation temperature, developmental biology, and the divergence of sockeye salmon (*Oncorhynchus nerka*) within Lake Washington. Can. J. Fish. Aquat. Sci. 55: 1387-1394.

Hope, B.K. 2012. Acquisition of polychlorinated biphenyls (PCBs) by Pacific chinook salmon: An exploration of various exposure scenarios. Integr Environ Assess Manag

Johnson, O.W., W.S. Grant, R.G. Kope, K. Neely, F.W. Waknitz, and R.S. Waples. 1997. Status review of chum salmon from Washington, Oregon, and California. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-32, 280 pp.

Johnson, L.L. G.M. Ylitalo, M.R. Akoosh, A.N. Kagley, C. Stafford, J.L. Bolton, J. Buzitis, B.F. Anulacion, and T.K. Collier. 2007a. Contaminant exposure in outmigrant juvenile salmon from Pacific Northwest estuaries of the United States. Environ Monit. Assess. 124(1-3): 167-194.

Johnson, L.L., G.M. Ylitalo, C.A. Sloan, B.F. Anulacion, A.N. Kagley, M.R. Arkoosh, T.A. Lundrigan, K. Larson, M.D. Siipola, T.K. Collier. 2007b. Persistent organic pollutants in outmigrant juvenile Chinook salmon from the Lower Columbia Estuary, USA. Science of the Total Environment, 374(2007):342-366.

Kelley, M.A., A. Gillespie, G-D. Zhou, S. Zhang, J.P. Meador, B. Duncan, K.C. Donnelly, and T.J. McDonald. 2011. In situ biomonitoring of caged, juvenile Chinook salmon (*Oncorhynchus tshawytscha*) in the Lower Duwamish Waterway. Mar. Pollut. Bull. 62(2011): 2520-2532.

Kelly, B.C., S.L. Gray, M.G. Ikonomou, J.S. Macdonald, S.M. Bandiera, and E.G. Hrycay. 2007. Lipid reserve dynamics and magnification of persistent organic pollutants in spawning sockeye salmon (*Oncorhynchus nerka*) from the Fraser River, British Columbia. Environ. Sci. Technol. 41: 3083-3089.

Landahl, J.T., L.L. Johnson, J.E. Stein, T.K. Collier, and U. Varanasi. 1997. Approaches for determining effects of pollution on fish populations of Puget Sound. Trans. Of the A. Fish. Soc. 126: 519-535.

Laufle, J.C., Pauley, G.B. and Shepard, M.F. 1986. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (Pacific Northwest) – Coho Salmon. U.S. Fish and Wildlife Service. Biological Report 82(11.48). U.S. Army Corps of Engineers, TR EL-82-4.

LCREP (Lower Columbia River Estuary Partnership). 2007. Lower Columbia River and Estuary Ecosystem Monitoring: Water Quality and Salmon Sampling Report.

Love, M. 1996. Probably more than you want to know about the fishes of the Pacific coast. Really big press, Santa Barbara, California, 381 pp.

Meador, J. P., T. Collier, and J. Stein. 2002. Use of tissue and sediment-based threshold concentrations of polychlorinated biphenyls (PCBs) to protect juvenile salmonids listed under the U.S. Endangered Species Act. Aquatic Conservation 12:493–516.

McCain, B.B., D.C. Malins, M.M. Krahn, D.W. Brown, W.D. Gronlund, L.K. Moore, and S-L. Chan. 1990. Uptake of aromatic and chlorinated hydrocarbons by juvenile Chinooks salmon (*Oncorhynchus tshawytscha*) in an urban estuary. Arch. Environ. Toxicol. 19:10-16.

Myers, J.M., R.G. Kope, G.J. Bryant, D. Teel, L.J. Lierheimer, T.C. Wainwright, WS. Grant, F.W. Waknitz, K. Neely, S.T. Lindley, and R.S. Waples. 1998. Status review of chinook salmon from Washington, Idaho, -Oregon, and California. US**.** Dept. Commer., NOAA Tech, Memo. NMFS-NWFSC-35,443 p.

Newell, J. 2005. Migration and Movement Patterns of Adult Sockeye Salmon (*Oncorhynchus nerka*) in Lake Washington. Masters Thesis, University of Washington, Seattle. 91 pp.

Norman, A., H. Börjeson, F. David, B. Tienpont, and L. Norrgren. 2007. Studies of uptake, elimination, and late effects in Atlantic Salmon (*Salmo salar*) dietary exposed to di-2-ethylhexyl phthalate (DEHP) during early life. Arch. Environ. Contam. Toxicol. 52: 235-242.

O'Neill, S.M. 2006. Draft Overview of commercial and recreational fisheries. Prepared for Department of Health staff by Sandie O'Neill. June 2006. 5 pp.

O'Neill, S.M., and J.E. West. 2009. Marine distribution, life history traits, and the accumulation of polychlorinated biphenyls in Chinook salmon from Puget Sound, Washington. Transactions of the American Fisheries Society 138: 616-632.

O'Neill, S.M., G.M. Ylitalo, J.E. West, J. Bolton, C.A. Sloan, and M.M. Krahn. 2006. Regional patterns of persistent organic pollutants in five Pacific salmon species (*Oncorhynchus spp*) and their contributions to contaminant levels in northern and southern resident killer whales (*Orcinus orca*). 2006 Southern Resident Killer Whale Symposium, NOAA Fisheries Service Northwest Regional Office April 3-5, 2006. Seattle, WA. Extended Abstract. 5pp.

O'Neill, S.M., J.E West, and J.C. Hoeman. 1998. Spatial trends in the concentration of polychlorinated biphenyls (PCBs) in Chinook (*Oncorhynchus tshawytscha*) and coho salmon (*O. kisutch*) in Puget Sound and factors affecting PCB accumulation: results from the Puget Sound Ambient Monitoring Program. Puget Sound Research 1998 Proceedings.

Pauley, G.B., B.M. Bortz, and M.F. Shepard. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest) -- steelhead trout. U.S. Fish Wildl. Serv. Biol. Rep. 82(11.62). USACE, TR EL-82-4. 24 pp.

Pauley, G.B., K.L. Bowers, and G.L. Thomas. 1988. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest) -- chum salmon. U.S. Fish Wildl. Serv. Biol. Rep. 82(11.81) U.S. Army Corps o f Engineers, TR EL-82-4. 17 pp.

Pauley, G.B., R. Risher, and G.L. Thomas. 1989. Species Profile: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest) – Sockeye salmon. USFWS Biol. Rep 82(11.116) USACE, TR EL-82-4. 22 pp.

Quinn, T.P. 2005. The Behavior and Ecology of Pacific Salmon and Trout. University of Washington Press, Seattle, WA.

Pearcy, W., T. Nishiyama, T. Fuji, and K. Masuda. 1984. Diel variations in the feeding habits of Pacific salmon caught in gill nets during a 24-hour period in the Gulf of Alaska. Fishery Bulletin 82(2): 391-399.

Redman, S., D. Myers, and D. Averill. 2005. Regional Nearshore and Marine Aspects of Salmon Recovery in Puget Sound. Prepared for Shared Strategy for Puget Sound for inclusion in their regional salmon recovery plan. June 28, 2005. 321 pp.

Salmonid Stock Inventory (SaSI). 2002. Salmonid Stock Inventory (SaSI). http://wdfw.wa.gov/conservation/fisheries/sasi/

Salo, C.E. 1991. Life History of chum salmon (*Oncorhynchus keta*). Pp 233-309 *In* Groot, C., and L. Margolis (Eds.) Pacific Salmon Life Histories. UBC Press, University of British Columbia, Vancouver, Canada.

Sandercock, F.K. 1991. Life History of coho salmon (*Oncorhynchus kisutch*). Pp 397-445 *In* Groot, C., and L. Margolis (Eds.) Pacific Salmon Life Histories. UBC Press, University of British Columbia, Vancouver, Canada.

Sethajintanin, D., E.R. Johnson, B.R. Loper, and K.A. Anderson. 2004. Bioaccumulation profiles of chemical contaminants in fish from the lower Willamette River, Portland Harbor, Oregon. Arch. Environ. Contam. Toxicol. 46: 114-123.

Shaffer, A., P. Crain, T. Kassler, and J. Schilke. 2008. Draft report: Juvenile Chinook use of the nearshore central and western Strait of Juan de Fuca. 16 pp. Report dated June 26, 2008.

Simenstad, C.A. and J.R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmon habitat in Pacific Northwest estuaries. Ecological Engineering 15:283-302.

Simenstad, C.A., and W.J. Kinney. 1978. Trophic relationships of outmigrating chum salmon in Hood Canal, Washington, 1977. Fish. Res. Inst. Univ. Wash., Seattle. Rep. Fri-UW-7810. July 1978. 75 pp.

Simenstad, C.A., K.L. Fresh, and E.O. Salo. 1982. The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function. Pp 343-364 *In* V.S. Kennedy (Ed.). Estuarine Comparisons. Academic Press, New York..

Stein, J.E., T. Hom, T.K. Collier, D.W. Brown, and U. Varanasi. 1995. Contaminant exposure and biochemical effects in outmigrant juvenile Chinook salmon from urban and nonurban estuaries of Puget Sound, Washington. Environ. Toxicol. And Chem. 14:1019-1029.

WDFW (Washington Department of Fish and Wildlife). 2002. Salmonid stock inventory (SaSI) 2002. Maps and stock reports. Washington Department of Fish and Wildlife, Olympia, WA. http://wdfw.wa.gov/fish/sasi/.

West, J.E., and S.M. O'Neill. 2007. Thirty years of persistent bioaccumulative toxics in Puget Sound; Time trends of PCBs and PBDE flame retardants in three fish species. 2007 Research in the Georgia Basin and Puget Sound Conference. Vancouver, B.C.: Puget Sound Action Team.

West, J. E., S. M. O'Neill, and G. M. Ylitalo. 2008. Spatial extent, magnitude, and patterns of persistent organochlorine pollutants in Pacific herring (*Clupea pallasi*) populations in Puget Sound (USA) and Strait of Georgia (Canada). Science of the Total Environment 394:369–378.

Yanagida, G.K., B.F. Anulacion, J.L. Bolton, D. Boyd, D.P. Lomax, O.P. Olson, S.Y. Sol, M. Willis, G.M. Tlitalo, and L.L. Johnson. 2012. Polycyclic aromatic hydrocarbons and risk to threatened and endangered Chinook salmon in the lower Columbia River estuary. Arch. Environ. Contam. Toxicol. 62(2): 282-295.

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