

## 7.0 Lead Risk Assessment

Lead health risks are presented separately because lead health risk methods are unique owing to the ubiquitous nature of lead exposures and the reliance on blood lead concentrations to describe lead exposure and toxicity. Lead risks are characterized by predicting blood lead levels with models and guidance developed by EPA available from the following web site:

<http://www.epa.gov/superfund/programs/lead/prods.htm> - software. In this assessment, lead exposure from fish consumption is added to all other likely sources of lead exposure to predict a blood lead level. Both the Integrated Exposure Uptake Biokinetic Model (IEUBK) for children and the EPA Adult Lead Model for the fetus predict blood lead levels from a given set of input parameters. There is no other model for lead exposures except the Adult Lead Model, so it is used for children and fetuses.

In contrast to risk assessments for cancer or non-cancer risks, lead risk assessments typically use central tendency exposure values to predict a central tendency (geometric mean) blood lead level. The predicted geometric mean blood lead level is then used in conjunction with a modeled log-normal distribution to estimate the probability of exceeding a target blood lead level of 10 µg/dl. Blood lead levels are a measure of internal dose that has been related to many adverse health effects (NRC, 1993). The emphasis on blood lead integrates exposure, toxicity and risk, which are more distinct in other types of risk assessment. For other chemicals, risk is described in terms of an external dose (e.g. mg/kg-day).

The IEUBK Model was used to predict blood lead levels in children up to 72 months of age (USEPA, 1994a,b). The EPA Adult Lead Model was used to predict blood lead levels in fetuses (USEPA, 1996b). This section on lead risk assessment is organized into separate discussions of the two lead models. Each of the two lead models was run using both central tendency and high end rates of fish ingestion. Central tendency rates of fish ingestion were used to predict both geometric mean blood lead levels and the probability of exceeding a blood lead level of 10 µg/dl in both children and fetuses. For the high end fish ingestion rates, only the most likely blood level could be predicted; it is not appropriate to predict the probability of exceeding 10 µg/dl associated with high end fish consumption.

### 7.1 Lead Concentrations in Fish

Study sites, collection methods, analytical methods, and quality assurance plans are discussed in Section 1; concentrations of lead in fish are discussed in Section 2. Whole fish had substantially higher lead levels because lead tends to concentrate in the bones and gills (Ay et al., 1999). Note that the maximum in the concentration scale for whole fish is 500 µg/kg and 100 µg/kg for fillets (Table 2-14). The highest individual sample was 1200 µg/kg in a fall chinook salmon taken from Station 14 on the Columbia River. For fish tissue samples with undetected lead concentrations, a value of half the detection limit was used (5 µg/kg) in all risk estimates.

## 7.2 Overview of Lead Risk Assessment Approach

Risk assessment methods for lead differ from other types of risk assessment because they integrate all potential sources of exposure to predict a blood lead level. Lead in the blood reflects all sources of lead exposure, regardless of its origin. Lead risk assessments reflect the widespread distribution of lead in the environment. Common sources of lead in the environment include residual contamination from past uses of lead in gasoline, paint, agricultural chemicals, and industrial sources including lead mining and smelting (NRC, 1993). People are exposed to lead through ingestion of soil and dust, inhalation of lead from the air, and consuming food with background concentrations of lead. Lead can enter drinking water through contamination of surface and groundwater as well as leaching from lead pipes and solder in plumbing systems. All of these sources and exposure pathways are included in the models used to assess lead risks. The IEUBK model is used to simulate lead exposures from air, water, diet, soil, and house dust. The Adult Lead Model accounts for the same sources of lead exposure by using a baseline blood lead level derived from the National Health and Nutrition Examination Survey (USEPA, 1996b).

Risk assessment methodologies for substances other than lead utilize a combination of central tendency and high end exposure values to estimate an aggregate reasonable maximum exposure scenario. A point value for risk derived using a reasonable maximum exposure scenario is accepted as being protective of public health. Public health protection using lead risk assessment methodology derives from a limit on the acceptable predicted blood lead values. An acceptable risk for lead exposure typically equates to a predicted probability of no more than 5% greater than the 10 µg/dl level (USEPA, 1998b)

Risk, expressed as predicted blood lead levels, was calculated in two ways for children and fetuses. The first, and more typical, method used median fish ingestion rates to predict: 1) a geometric mean blood lead level and 2) the corresponding risk of exceeding a blood lead level of 10 µg/dl. The probability of exceeding 10 µg/dl was calculated with a log-normal risk model based on the model's output (the geometric mean blood lead level) and an assumed geometric standard deviation. In the second method, high-end fish ingestion rates were used to predict blood lead levels for children or mothers who consume large amounts of fish. Because the resultant high-end fish ingestion prediction does not represent a geometric mean blood lead level, the geometric standard deviation could not be applied to predict the probability of exceeding 10 µg/dl. Predicted blood lead levels resulting from high-end fish consumption scenarios represent the most likely blood lead levels associated with high-end consumption rates.

The adverse health effects of lead have been related to blood lead concentrations in units of micrograms of lead per deciliter of whole blood (µg/dl). As a result, blood lead levels have evolved as measures of exposure, risk, and toxicity. Since 1991, the national level of concern for young children and fetuses has been 10 µg/dl (CDC, 1991). An analogous level has not been defined for other groups, but children and the developing fetus are accepted as being especially vulnerable to lead because lead interferes with the development of the central nervous system (NRC, 1993). Lead risks were evaluated by comparing predicted blood lead levels to the 10 µg/dl standard and by determining the expected percentage to exceed the 10 µg/dl criterion.

Adverse health effects observed at a blood lead level of 10 µg/dl are sub-clinical, meaning that, these effects cannot be diagnosed in an individual. The adverse health effects include cognitive deficits in IQ and learning, based on numerous scientific studies involving comparisons of large groups of children to control for confounding factors and account for the natural variability in cognitive function (NRC, 1993; USDHHS, 1999; CDC, 1991). The studies have incorporated both cross-sectional and longitudinal designs. The importance of primary prevention of lead exposure has been highlighted by recent studies suggesting adverse health effects at blood lead levels less than 10 µg/dl and the failure of chelation treatment to prevent cognitive impairments in treated children (Lanphear et al., 2000; Rogan et al., 2001; Rosen and Mushak, 2001).

Children are the population of greatest concern for lead exposure. Blood lead levels tend to peak in children as they become more mobile and begin to explore their surroundings. Blood lead levels normally peak at approximately 30 months of age when children are especially vulnerable to neuro-behavioral deficits (Rodier, 1995;Goldstein, 1990). The adverse effects of low-level lead poisoning can result from relatively short-term exposures on the order of months, as opposed to periods of years or longer for other chemicals. The fetus is vulnerable to the same developmental and neuro-behavioral effects as children. Although lead is harmful to fetuses, children are a greater concern because they generally have higher exposures than fetuses. Fetal exposures are lower because exposures to mothers are typically lower than exposures to children. These and other health effects are described in further detail in Appendix C (Toxicity Profiles).

### **7.3 Method for Predicting Risks to Children**

In contrast to risk assessment methodologies for predicting cancer or non-cancer risks, the lead models rely on central tendency exposure values to predict a central tendency (geometric mean) blood lead level. The predicted geometric mean blood lead level is then used in conjunction with an assumed geometric standard deviation to estimate the probability of exceeding a target blood lead level of 10 µg/dl established by the Centers for Disease Control (CDC, 1991). In this way, central tendency exposure estimates are used to estimate upper percentile blood lead levels. An example graph of an IEUBK Model run depicting the geometric mean and percent greater than 10 µg/dl is shown in Figure 7-1. In the IEUBK model, a geometric mean blood lead level of 4.6 µg/dl corresponds to a 5% chance of exceeding 10 µg/dl using the default geometric standard deviation of 1.6 (USEPA, 1994b). Although lead risk assessment methods differ from that employed for other chemicals, the goal of protecting highly exposed individuals remains the same.

The geometric standard deviation accounts for the variation in blood lead observed in children exposed to similar environmental concentrations of lead. The variation in observed blood lead levels is attributed to differences in the children (behavior and metabolism); not the environment. Because the geometric standard deviation accounts for behaviors that determine exposure levels to lead, applying the geometric standard deviation to high contact rate behaviors, including fish ingestion, would over-estimate the variability and over-predict the probability of exceeding 10 µg/dl.

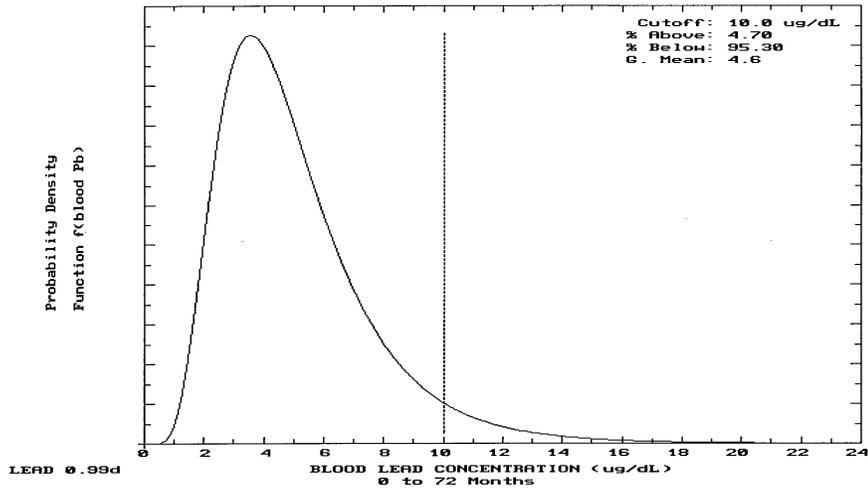


Figure 7-1. Sample IEUBK Model for Lead Output Graph.

Running the IEUBK Model with high-end fish consumption rates predicts the most likely blood lead levels for people eating large amounts of fish, although, the result does not correspond to the geometric mean of a population consuming different amounts of fish. Blood lead predictions for highly exposed individuals facilitate comparison of lead risks to risks from other chemicals, but results from high-end exposure inputs preclude application of the geometric standard deviation to calculate risks of exceeding a 10  $\mu\text{g}/\text{dL}$  blood lead level. Risks to highly exposed individuals are typically characterized by the 95<sup>th</sup> percentile of the blood lead distribution centered around the predicted geometric mean blood lead rather than using the high-end fish ingestion values.

The IEUBK Model was run with all exposure parameters set to default levels with the addition of dietary lead intake attributable to lead in fish tissue for the full range of lead concentrations observed. Default exposure parameters are based on national average levels of lead in air, water food, soil, and dirt (Table 7-1) and described in detail in EPA guidance (USEPA, 1994b).

<b>Input Parameter</b>	<b>Value</b>
Soil lead concentration	200,000 µg/kg
House dust lead concentration (proportion of soil in dust = 0.7)	140,000 µg/kg
Combined soil and dust ingestion rate by age:	
0-11 months	85 mg/day
12-23 months	135 mg/day
24-35 months	135 mg/day
36-47 months	135 mg/day
48-59 months	100 mg/day
60-71 months	90 mg/day
Lead concentration in Air	0.10 : g/cubic meter
Lead concentration in drinking water	4 : g/liter

The default concentrations of lead in soil and house dust are representative of average, national conditions. The default concentrations for lead in soil and house dust are 200,000 µg/kg and 140,000 µg/kg respectively (USEPA, 1994b). These values are appropriate for urban areas and are likely to exceed the expected concentrations in rural areas surrounding the Columbia River because lead levels increase with urbanization. A recent survey of 50 homes from small, rural towns in Northern Idaho found soil lead concentrations less than 100,000 µg/kg (Spalinger et al., 2000). These concentrations would not account for severe lead paint contamination. Lack of data on specific soil and house dust concentrations remains a large source of uncertainty in this evaluation because soil and dust in the home account for a large proportion of lead exposure in young children (Manton et al., 2000) (Lanphear et al., 1998).

The IEUBK model has the capability to simulate exposures to locally grown vegetables, game, and fish. The IEUBK default values for soil, house dust, air, diet, and water were used in conjunction with an age-specific median fish ingestion rate of 16.2 g/day based on the fish consumption survey of CRITFC's member tribes (CRITFC, 1994). Fish ingestion was specified as the percentage of meat (Table 7-2) consisting of locally caught fish and the lead concentrations in the fish. There are other ways to simulate fish ingestion in the IEUBK Model (e.g. by specifying dietary lead intakes as µg/day), but it was preferred to specify fish ingestion as a percentage of meat to preserve the caloric and protein intake assumptions of the model. This approach substitutes fish for other protein sources rather than adding fish to the default diet. This approach conforms with IEUBK body weight and biokinetic assumptions and is described in EPA guidance (USEPA, 1994b).

<b>Age Range (months)</b>	<b>Meat Consumption grams/day</b>
12-24	87
25-36	96
37-48	102
49-60	107
61-72	112
Average	101

The CRITFC study examined Columbia River fish consumption in young children as surveyed by their parents. This study was selected as the most relevant study to assess the Columbia River lead hazard for all children because it is specific to the place, CRITFC's member tribes, and the age range specified by the IEUBK (CRITFC, 1994). The tribal ingestion rates are likely to overestimate fish consumption for non-tribal members. Because the CRITFC study presents consumption rates for children up to 72 months of age, the IEUBK Model was run for the same age range.

To facilitate comparisons between risks from lead and other chemicals presented in Section 6, the ingestion rates used for other chemicals are summarized in Table 7-3. Fish ingestion rates used to estimate risks from chemicals other than lead are based on mean and 99<sup>th</sup> percentiles of both the CRITFC survey and national data for the general public described in Section 4 of this report.

The distribution of child fish consumption rates from the CRITFC study is statistically skewed because it included individuals with very high fish consumption rates relative to others. For skewed data, the arithmetic mean is not an appropriate measure of central tendency because it is highly influenced by the individuals with large fish consumption rates. The median (50<sup>th</sup> percentile) is a preferred central tendency measure of skewed data because it is less sensitive to extreme values. The fish consumption data for CRITFC's member tribes (CRITFC, 1994) were re-analyzed to omit children who did not consume fish from the data set (Kissinger and Beck, 2000). The re-analysis calculated a median consumption rate occurred between 13 and 16.2 g/day, the 39<sup>th</sup> and 65<sup>th</sup> percentiles, respectively (see Table 7-4). Rather than interpolate a median value of 14.4 g/day between the 39<sup>th</sup> and 65<sup>th</sup> percentiles, the higher value was selected as a protective central tendency consumption rate.

**Table 7-3. Fish Ingestion Rates (grams/day) Used to Assess Risk for Lead and other Chemicals**

Target Population				Non-lead		Non-lead	
Assessment	Lead		Native American		General Public		
Population	Native American						
Exposure Level	Central	High End	Central	High End	Central	High	
	Mother and Fetus		Adult		Adult		
Ingestion Rate	39.2	389	63.2	389	7.5	142.4	
Basis	50 <sup>th</sup> CRITFC	99 <sup>th</sup> CRITFC	Mean CRITFC	99 <sup>th</sup> CRITFC	Mean EPA	99 <sup>th</sup>	
Age Range	Children < 72 Months		Children < 72 Months		Children < 15 years		
Ingestion Rate	16	101	24.8	162	2.83	77.95	
Basis	50 <sup>th</sup> CRITFC	IEUBK MAX*	Mean CRITFC	99 <sup>th</sup> CRITFC	Mean	99 <sup>th</sup>	

\* A fish ingestion rate of 101 g/day assumes that locally caught fish comprise 100% of all dietary protein sources and represents an upper constraint of the IEUBK Lead Model for Children

**Table 7-4. Percentages of Child Fish Consumption Rates for Consumers of Fish From (Kissinger and Beck, 2000) analysis of (CRITFC, 1994)**

Grams/day	Cumulative Percent	Grams/day	Cumulative Percent	Grams/day	Cumulative Percent
0.4	1%	8.1	33%	32.4	84%
0.8	1%	9.7	35%	48.6	89%
1.6	5%	12.2	38%	64.8	93%
2.4	5%	13.0	39%	72.9	95%
3.2	9%	16.2	65%	81.0	97%
4.1	14%	19.4	66%	97.2	98%
4.9	16%	20.3	67%	162.0	100%
6.5	18%	24.3	70%		

### 7.4 Risk Characterization for Children

Predicted blood lead levels spanning the full range of observed fish tissue concentrations are shown in Figure 7-2. Predicted geometric mean blood lead levels are plotted on the left axis with a solid line. The corresponding probabilities of exceeding 10 µg/dl are shown as percentages on the right axis with a dashed line. Each of the 11 pairs of points represents a separate IEUBK Model run at successively increasing concentrations of lead in fish. These results indicate that for fish containing lead up to 500 µg/kg, the probability of achieving a blood lead level greater than 10 µg/dl is no more than 5% and the predicted geometric mean blood lead level is 4.6 µg/dl. For comparison, only the average concentration of whole body eulachon had a lead concentration of 500 µg/kg. The next highest whole fish species is fall chinook, with an average lead concentration of 220 µg/kg. Average lead concentrations in all other whole fish and fillet samples occur well below 500 µg/kg and concentrations in fillets averaged 200 µg/kg (Table 2-14).

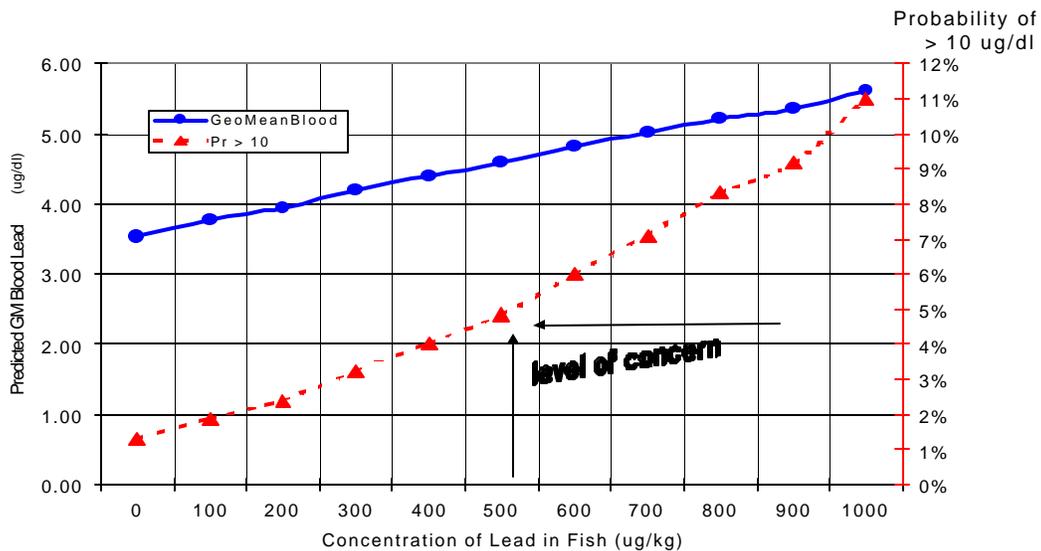


Figure 7-2. Predicted blood lead levels for children who consume of fish collected from the Columbia River Basin assuming fish is 16% of dietary meat.

To explore the effect of an extremely high fish consumption rate in children, the IEUBK Model was run assuming that fish replaced 100% meat in the diet (101 g/day) (Figure 7-3). The IEUBK Model was run repeatedly to determine the fish tissue concentration associated with a predicted blood lead level of 10 µg/dl. A lead concentration of 500 µg/kg in fish tissue corresponded to a predicted blood lead concentration of 10 µg/dl. This is the same concentration associated with a 5% risk of exceeding 10 µg/dl under the 16.2 g/day fish consumption scenario described in the previous paragraph.

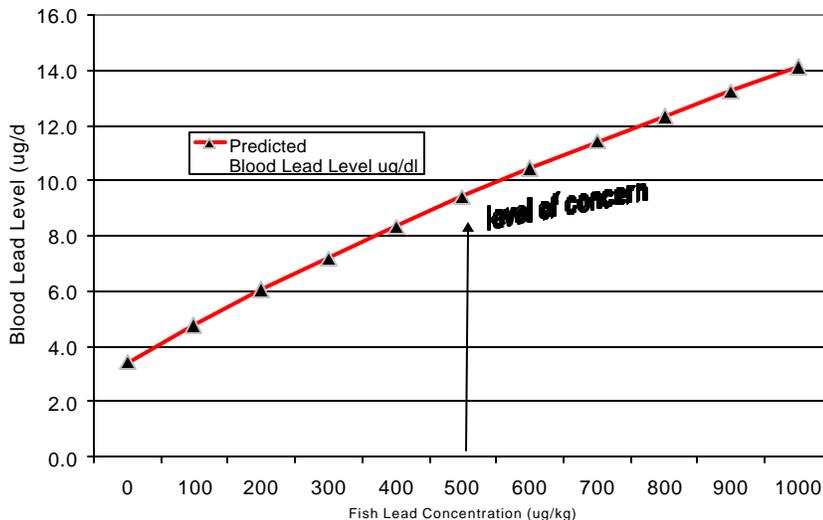


Figure 7-3. Predicted blood lead levels for children (0-72 months) who consume 101 g/day of fish collected from the Columbia River Basin, 1996-1998.

## 7.5 Uncertainties in risk estimates for Children

Lead risk assessment methods are unique because they use cumulative exposures to predict blood lead levels in contrast to methods used for other chemicals which generally limit evaluation of exposures to discreet sources. Because lead risks are cumulative, uncertainties are compounded by the many sources of exposure in addition to uncertainties arising from fish consumption. In children, lead exposure occurs primarily from lead in soil and house dust rather than from typical dietary sources (Manton et al., 2000). Sources of lead exposure common to children and fetuses include industrial or agricultural sources, occupational exposures, and environmental lead originating from gasoline or leaded paint. Occupational exposures can track contaminants from the workplace into the home, potentially spreading exposure among children and adults in a household (Fenske et al., 2000). A major source of uncertainty in this risk assessment may be attributable to sources of lead other than Columbia River fish. The magnitude of lead exposure from fish consumption varies with selection of fish parts eaten (e.g. whole versus fillet), species of fish, and the study site of the fish relative to sources of lead contamination.

The IEUBK model is normally used to simulate blood lead levels for children up to 84 months of age. However, because the fish consumption data from the CRITFC study were reported for children up to 72 months of age, IEUBK evaluation was limited to 72 months. A 72-month

model run predicts higher blood lead concentrations than an 84-month model run because blood lead levels peak during the first 36 months. In the absence of data to estimate specific, concurrent residential exposures, the default concentrations of lead in soil and house dust represent a large source of uncertainty in the IEUBK evaluation because these sources are expected to account for most of the lead exposure to young children. However, the default soil and dust concentrations are unlikely to underestimate average levels of lead in the homes.

## 7.6 Method for Predicting Risks to Fetuses

The Adult Lead Model begins with a baseline blood lead level for adult women and then predicts an incremental increase in blood lead levels associated with an increase in exposure that is not included in the baseline blood lead levels (USEPA, 1996b and USEPA, 1999a). In the Adult Lead Model, fetal blood lead levels are set equal to 90% of the mother's blood lead level. If the baseline blood lead reflects the modeled incremental exposure, then the exposure is counted twice and the modeled blood lead level would be too high. In this study, the Adult Lead Model was used to evaluate fish ingestion as the source of incremental exposure greater than the baseline blood lead level.

The assumptions used in this approach include:

- 1) Lead exposures from all sources except consuming fish from the Columbia River are captured in the baseline blood lead level, based on high end estimates from national blood lead surveys, and
- 2) incremental ingestion of fish is not included in the baseline blood lead level.

Selection of a high baseline blood lead level minimized the possibility of underestimating risk. The lead ingested from fish is converted to a blood lead level by using a constant ratio of an increase in blood lead concentration associated with a mass of absorbed lead. This ratio is the Biokinetic Slope Factor (BKSF). The baseline blood lead level, the blood level in the absence of lead exposure via Columbia River fish ingestion, is critical to this calculation. A complete listing of all the Adult Lead Model input values is included in Table 7.5.

The equations used in the Adult Lead Model are (USEPA 1999b):

*Equation 7-1*

*Adult Blood Lead Level = Baseline Blood Lead Level + Increase in Blood Lead*

*Equation 7-2*

*Increase in Blood Lead =*

*[(BKSF) \* Fish Ingestion Rate \* Fish Concentration \* Absorbed Fraction for Fish]*

*Equation 7-3*

*Fetal Blood Lead = Adult Blood \* 0.9*

*Equation 7-4*

Probability that Fetal Blood Lead is greater or equal to 10 µg/dl using the z-value where:  

$$z = \ln(10) - \ln(\text{Fetal Blood Lead}) / \ln(\text{Geometric Standard Deviation})$$

Analysis of the lead hazard associated with adult consumption of Columbia River fish was conducted using the formula:

$$\text{Equation 7-5 } PbB_{adult, central} = PbB_{adult,0} + BKSF * (PBF * IR_F * AF_F * EF_F) / AT$$

**Table 7-5. Input Parameters Used for the EPA Adult Lead Model**

Variable	Description	Value Used
PbB <sub>adult,0</sub>	Adult blood lead concentration in the absence of other lead exposure.	Central 1.7 µg/dl High End 2.2 µg/dl
BKSF	Biokinetic slope factor relating the (quasi-steady state) increase in blood lead concent	
PbF	Fish lead concentration	full range of values: 0-1000 µg/kg
IR <sub>F</sub>	Intake rate of fish in g/day median of CRITFC Adult Consumption	39.2 g/day
AF <sub>F</sub>	Absolute gastrointestinal absorption factor for ingested lead in fish (dimensionless)	0.10
EF <sub>F</sub>	Exposure frequency for ingestion of fish (days of exposure during the averaging period); may be taken as days per year in continuing long term exposures.	365 days per year
AT	Averaging time, the total period during which exposure may occur	365 days per year

Because study site-specific baseline blood lead levels and geometric standard deviations are not available for consumers of Columbia River fish, the Adult Lead Model was run using both central tendency and high-end estimates of the baseline blood lead level and the geometric standard deviation described in (USEPA, 1996b). The larger baseline blood lead level increased the predicted blood lead levels. An increase in the Geometric Standard Deviation increased the probability of exceeding 10 µg/dl. All input parameters are listed in Table 7.6.

**Table 7-6. Adult Lead Model Baseline Blood Lead and Geometric Standard Deviations**

Input Parameter	Baseline Blood Lead Level	Geometric Standard Deviation
Central Values	1.7 µg/dl	1.8
High End Values	2.2 µg/dl	2.1

Fish ingestion rates for adult consumers of Columbia River fish are based on the median ingestion rate of 39.2 g/day interpolated from Table 10 of the 1994 CRITFC consumption survey (CRITFC, 1994). Consumption rates were reported as 38.9 g/day and 40.5 g/day for the 49<sup>th</sup> and 53<sup>rd</sup> percentiles respectively (CRITFC, 1994). For comparison, EPA provides a mean estimate of national per capita fish consumption of 7.5 g/day (USEPA, 2000b). The Model was also run using the 99<sup>th</sup> percentile ingestion rate from the CRITFC survey (389 g/day) to facilitate comparison with the risks from chemicals other than lead (Table 7.1).

## 7.7 Risk Characterization for Fetuses

The Adult Lead Model was used to evaluate potential lead risks to the fetus following maternal consumption of Columbia River fish. Predicted fetal geometric mean blood lead levels and associated probabilities of exceeding the 10  $\mu\text{g}/\text{dl}$  for a range of lead levels in fish are summarized in Figures 7-4 and 7-5. Figure 7-4 shows results using the maximum recommended exposure parameters for the baseline blood lead level of 2.2  $\mu\text{g}/\text{dl}$  and geometric standard deviation of 2.1 (USEPA, 1996b). Figure 7-5 is identical to Figure 7-4, but uses central tendency estimates of baseline blood lead level of 1.7  $\mu\text{g}/\text{dl}$  and geometric standard deviation of 1.8. Although, the predicted risks of exceeding 10  $\mu\text{g}/\text{dl}$  are substantially higher in Figure 7-4, the fish concentration associated with a 5% risk of exceeding 10  $\mu\text{g}/\text{dl}$  is 700  $\mu\text{g}/\text{kg}$ . Average fish concentrations in whole fish and fillets were 0.12 and 0.02 respectively. The highest lead concentrations were found in whole-body samples of eulachon with an average fish tissue concentration of 500  $\mu\text{g}/\text{kg}$  lead. For the fetus of an adult consuming 39.2 grams of whole fish per day (129  $\mu\text{g}/\text{kg}$ ), the Adult Lead Model predicts that fetal blood lead levels will exceed 10  $\mu\text{g}/\text{dl}$  less than 2% of the time using the high end values for baseline blood lead level and geometric standard deviation. Using high end values for baseline blood lead level and geometric standard deviation with the 389 g/day ingestion rate results in a predicted fetal blood lead level at a fish concentration of 600  $\mu\text{g}/\text{kg}$ .

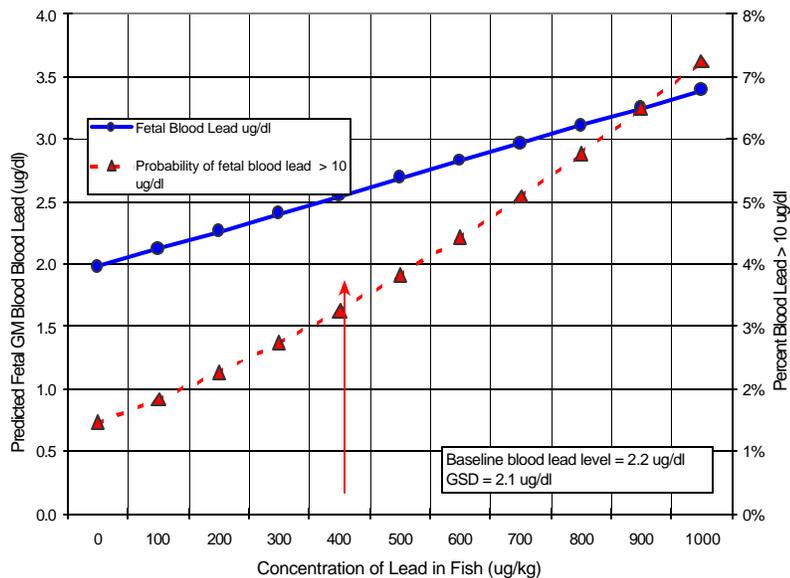


Figure 7-4. Predicted fetal blood lead levels with maternal fish ingestion rate of 39.2 g/day with baseline blood lead level at 2.2  $\mu\text{g}/\text{dl}$  and GSD = 2.1  $\mu\text{g}/\text{dl}$ .

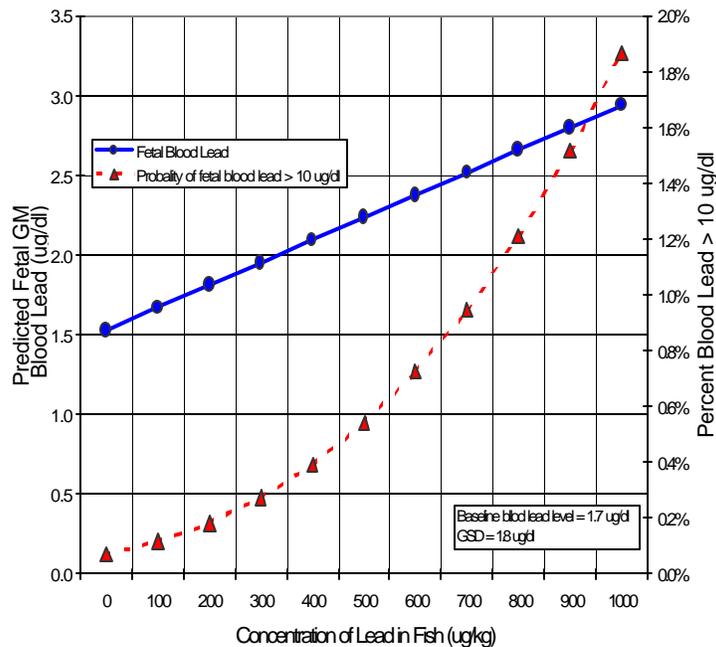


Figure 7-5. Predicted fetal blood lead level with maternal fish ingestion rate of 39.2 g/day with baseline blood lead level at 1.7  $\mu\text{g}/\text{dl}$  and GSD = 1.8  $\mu\text{g}/\text{dl}$ .

## 7.8 Uncertainty Analysis for Risk to Fetuses

Fetal risk estimates share common sources of uncertainties with the estimates for child risks including the assumed fish lead concentrations and fish consumption rates. Uncertainties unique to the Adult Lead Model include the assumed baseline blood lead level and geometric standard deviation parameters from the National Health and Nutrition Examination Survey (USEPA, 1996b). The results are based on the highest recommend values for the baseline blood lead levels and the geometric standard deviation. They are unlikely to underestimate risk.

## 7.9 Conclusions

Despite uncertainties in this assessment, lead levels in fish analyzed from the Columbia River occur at levels unlikely to cause a blood level greater than 10  $\mu\text{g}/\text{dl}$ . Risks to children from fish consumption are unlikely to exceed 5% at lead concentrations less than 500  $\mu\text{g}/\text{kg}$  (Figure 7-2, 7-3). Similarly, fetal risks are unlikely to exceed 5% at concentrations less than 700  $\mu\text{g}/\text{kg}$  (Figure 7-4, 7-5). These levels of concern occur at lead concentrations near the maximum values of the samples. This conclusion is supported by several analyses using health protective exposure assumptions that are unlikely to underestimate risks from fish consumption. The exposure assumptions are based on default and high end exposure parameters recommended by EPA lead risk assessment guidance used in conjunction with fish ingestion rates from the CRITFC fish consumption survey (CRITFC, 1994).

## 8.0 Radionuclide Assessment

### 8.1 Radionuclide Data Reporting and Use

A unique characteristic of some radionuclide analytical data is the occurrence of numerically negative results. Radionuclide analyses usually require the subtraction of an instrument background measurement from a gross sample measurement. Both results are positive, and when sample activity is low (close to background), random variations in measurements can cause the resulting net activity to be less than zero. Although negative activities have no physical significance, they do have statistical significance, as for example in the evaluation of trends or the comparison of groups of samples. Good practice for laboratory reporting of radionuclide analysis results therefore dictates reporting results as generated: whether positive, negative, or zero, together with associated uncertainties.

This is consistent with EPA guidance (USEPA, 1980a), which states: “When making measurements near background levels, one can expect to frequently obtain values that are less than the estimated lower limit of detection or minimum detectable concentration. If these values are not recorded and used in making average estimates, then these estimates are always going to be greater than the “true” representation in the environment. Therefore it is recommended that every measurement result should be recorded and reported directly as found.”

The general principles for evaluation of radionuclide data for this project were:

- a. It is generally best to use reported values plus the associated uncertainties.
- b. Reported values are better estimates of actual concentrations than are minimum detectable concentrations.
- c. J-qualified (estimated) data should not be used for quantitative purposes where unqualified data is available to substitute.
- d. All reported data (including U-qualified (nondetect) data, should be used in averages.
- e. Quantitative analyses should only be performed for those radionuclides which have at least one positive unqualified result reported.
- f. For gamma data, the EPA’s National Air and Radiation Exposure Laboratory (NAREL) reported minimum detectable concentration values for certain radionuclides of interest even in cases where the radionuclide was not detected and no value was reported. If these minimum detectable concentrations are used for quantitative analyses, the results should clearly note the use of minimum detectable concentration-based input. If minimum detectable concentrations are to be used for quantitative purposes, the minimum detectable concentrations may need additional decay corrections where holding times exceeded 10 half lives. This should not be an issue since no radionuclide with a half-life

less than 10% of holding time was detected in any of the gamma analyses and therefore these short-lived radionuclides would not be used for analytical purposes.

## 8.2 General Information on Radiation Risk

Radiation is a known human carcinogen. As such, the models used to estimate risk from radiation exposure assume that at low levels of exposure, the probability of incurring cancer increases linearly with dose, and without a threshold.

All of the epidemiological studies used in the development of radiation risk models involve high radiation doses delivered over relatively short periods of time. Evidence indicates that the response per unit dose at low doses and dose rates from low-linear energy transfer radiation (primarily gamma rays) may be overestimated if extrapolations are made from high doses acutely delivered. The degree of overestimation is often expressed in terms of a dose and dose rate effectiveness factor that is used to adjust risks observed from high doses and dose rates for the purpose of estimating risks from exposures at environmental levels. EPA models for radiation risk include a dose and dose rate effectiveness factor of 2 applicable to most low-linear energy transfer radiation exposure. For high-linear energy transfer radiation (e.g. alpha particles), the differences in relative biological effect are accounted for in weighting factors applied in the calculation of dose and risk.

In addition to cancer risk, radiation can also represent a risk for hereditary effects. Radiation-induced genetic effects have not been observed in human populations, however, and cancers generally occur more frequently than genetic effects. The radiation-related risk of severe hereditary effects in offspring is estimated to be smaller than that for cancer. The risk of severe mental retardation from radiation exposure to the fetus is estimated to be greater per unit dose than the risk of cancer in the general population, but the period of susceptibility is very much shorter. Based on these considerations, EPA generally considers the risk of cancer to be limiting and uses it as the sole basis for assessing radiation-related human health risks.

The risk coefficients used in this risk assessment are derived using age-specific models and are age-averaged. This means that the risk coefficients are appropriate for use in estimating exposure over a lifetime, since they are derived by taking into account the different sensitivities to radiation as a function of age. The risk coefficients in this assessment may be used to assess the risk due to chronic lifetime exposure of an average individual to a constant environmental concentration. The risk estimates in this report are intended to be prospective assessments of estimated cancer risks from long-term exposure to radionuclides in the environment. The use of the risk coefficients listed for retrospective analyses of radiation exposures to populations should be limited to estimation of total or average risks in large populations. The risk coefficients are not intended for application to specific individuals or to specific subgroups.

Estimates of lifetime risk of cancer to exposed individuals resulting from radiological and chemical risk assessments may be summed to determine the overall potential human health hazard. It is standard practice, however, to tabulate the two sets of risk estimates separately. This

is due to important differences in the two kinds of risk estimates. For many chemical carcinogens, laboratory experiments and animal data are the basis for estimates of risk. In the case of radionuclides, however, the data come primarily from epidemiological studies of exposure to humans. Another important difference is that the risk coefficients used for chemical carcinogens generally represent an upper bound or 95<sup>th</sup> percent upper confidence level of risk, while radionuclide risk coefficients are based on best estimate values.

### 8.3 Risk Calculations

Data qualifiers assigned during the data verification and validation process were used in making decisions about numerical values for input into risk calculations. Reported values were used with the following exceptions: zero was used where negative values were reported and one half of the reported minimum detectable concentration was used where the result was reported as minimum detectable concentration.

The naturally-occurring radionuclide potassium-40 (K-40) is a special case in the risk calculations. Potassium is an essential nutrient which contains the naturally radioactive isotope potassium-40, which has a half-life of more than one billion years. K-40 constitutes 0.01% of natural potassium which as a result has a specific activity of approximately 800 pCi/g of potassium. Variations in diet have little effect on the radiation dose received, since the amount of potassium in the body is under close hemostatic control. Although K-40 is the predominant source of radiation exposure from food, calculation of dose or risk for specific food pathways is not meaningful since the biological control of potassium content in the body (and hence the radiation dose due to potassium) means that the dose is independent of intake. Therefore, K-40 concentrations were not included in the calculations of cumulative risk from radionuclides in samples. K-40 concentrations and risks are discussed separately for comparison.

Quantitative analyses were performed only for those radionuclides which had at least one positive unqualified result reported. Those radionuclides and their associated risk coefficients are:

<u>Radionuclide</u>	<u>Risk Coefficient (risk/Bq)</u>
Uranium -234 (U-234)	2.58 x 10 <sup>-9</sup>
Uranium-235+D (U-235+D)	2.63 x 10 <sup>-9</sup>
Uranium-238+D (U-238+D)	3.36 x 10 <sup>-9</sup>
Strontium-90+D (Sr-90+D)	2.58 x 10 <sup>-9</sup>
Plutonium-239 (Pu-239)	4.70 x 10 <sup>-9</sup>
Bismuth-212 (Bi-212)	included in Th-228+D coefficient
Bismuth-214 (Bi-212)	included in Ra-226+D coefficient
Cesium-137+D (CS-127+D)	1.01 x 10 <sup>-9</sup>
Potassium-40 (K-40)	9.26 x 10 <sup>-10</sup>
Lead-212(Pb-212)	included in Th-228+D coefficient
Lead-214(Pb-214)	included in Ra-226+D coefficient
Raon-224(Ra-224)	included in Th-228+D coefficient
Thorium-228+D (Th-228+D)	1.14 x 10 <sup>-8</sup>
Radon-226+D (Ra-226+D)	1.39 x 10 <sup>-8</sup>
Tellurim-208 (Tl-208)	included in Th-228+D coefficient

Risks

for individual radionuclides were calculated using morbidity coefficients for dietary intake from EPA guidance (USEPA 1999c). Many of the radionuclides detected are members of important naturally-occurring decay chains (e.g. Ra-226 series, Th-228 series). For these radionuclides, risks were calculated based on risk from the entire decay series in secular equilibrium. Risk coefficients representing the entire decay series (identified with “+D” designation) were derived by summing the risk coefficients for all decay chain members. For some decay series members (e.g. Po-218) no data is available in EPA guidance and these radionuclides were not included in the calculation of risk coefficients (USEPA, 1999d). Based on data for these radionuclides reported in HEAST the risks from radionuclides which are not included in EPA guidance are insignificant in comparison to the risks from the other members of the decay series for which EPA guidance provides data (USEPA, 1994c; USEPA, 1999d).

The general approach used in selecting data for input into decay series calculations was to:

- 1) use measured data wherever possible,
- 2) prioritize measured data in accordance with assigned data qualifiers, and
- 3) to use minimum detectable concentration values ( minimum detectable concentrations) for input only when other sources of data were not available.

In selecting the value to use for the concentration of the radionuclide at the head of the chain, decay products were used as surrogates. This is consistent with the physical principles of radioactive decay and secular equilibrium. Where more than one decay product was available to act as surrogate, positive values were selected over nondetect. The largest positive value was used where two or more otherwise equally suitable results were available.

In cases where Tl-208 was used as a surrogate for the Th-228 decay series, the branching ratio of the Bi-212 decay (36% decaying to Tl-208) was taken into account. If no decay chain member data is available, one-half of the minimum detectable concentration value for Ra-226 was used for input into the calculation for the Ra-226+D subchain. Similarly, one-half the minimum detectable concentration for Ra-228 was used as input into the Th-228+D subchain calculation where necessary. In the case of Cs-137, if no gamma peak was reported, one-half of the Cs-137 minimum detectable concentration was used as input for this radionuclide.

If there was a choice between uranium data from uranium alpha analyses and from gamma analyses (e.g. U-235), the uranium alpha analysis data was used. Alpha analysis for uranium is a more sensitive technique than gamma analysis. In particular, U-235 analysis by gamma spectroscopy involves additional analytical uncertainty resulting from Ra-226 interference with the spectral line used to quantify U-235. If only the gamma data was available, it was used with appropriate consideration of data qualifiers.

Analytical results used for risk calculations included three samples which had a total of six “J” qualified (estimated) results among them. Five of these estimated values represented uranium isotopes which are expected to be present, and for which the estimated values represent the best available data for input into the risk calculation. In one case the estimated value used represented a result for Pu-239. These estimated values were included in the calculations for completeness,

and their inclusion did not significantly alter the magnitude of the risks calculated.

## **8.4 Composite Study site Results**

Plutonium, strontium and uranium analyses were not performed on all samples sent for radionuclide analysis. For some of the composite groups of samples (composites 53 (study site Columbia River 9U), 24 (study site Columbia River 7), and 25 (study site Columbia River 8), only gamma analyses were performed. Risks were calculated based on the gamma component of these samples only. Risks were calculated based on a nominal consumption rate of 1 gram per day and also for consumption rates of 7.5 g/day (average public consumption), 142.4 g/day (99<sup>th</sup> percentile public consumption), 63.2 g/day (average CRITFC's member tribe consumption) and 389 g/day (99<sup>th</sup> percentile CRITFC's member tribe consumption). These consumption rates are the same as used for the nonradionuclide risk analysis. Risks were calculated for a 70 year lifetime. Composites of particular interest include Composite 54 (study site -K-Basin ponds) and 30 (study site Snake River 13). Table 8-1 presents a summary of the calculated risks for each consumption rate.

### **8.4.1 Potassium-40 Results**

As expected, the results for K-40 analyses are very consistent throughout the samples and represent one of the most prominent sources of radioactivity in all samples analyzed. The concentrations in samples ranged between 1.7 pCi/g and 3.7 pCi/g with an average value of 2.8 pCi/g. If this value were used to calculate risk in the same manner as the other radionuclides detected, the resulting calculated average risk would be  $1 \times 10^{-3}$ . As noted previously, however, although K-40 is the predominant source of radiation exposure from food, calculation of dose or risk for specific food pathways is not meaningful since the biological control of potassium content in the body (and hence the radiation dose due to potassium) means that the dose is independent of intake. Therefore, K-40 concentrations were not included in the calculations of cumulative risk from radionuclides in samples. K-40 concentrations and risks are presented separately for the purposes of comparison.

## **8.5 Background**

As anticipated, many of the radionuclides present in naturally-occurring background were also present in the samples analyzed. The sampling and analysis for radionuclides was not designed to provide the statistical power necessary to quantitatively define background. The mobile nature of the species sampled together with normal regional and local variations in concentrations of naturally-occurring radionuclides in the environment make such an effort impractical in the context of this project. However, an effort was made to obtain data that would provide a qualitative perspective on background concentrations in fish. To this end, samples were taken from the Snake River (composite group number 30; study site Snake River 13) to represent fish that would not be affected by the operations of nuclear facilities in the Tri-Cities area. Examination of the analytical results for the Snake River samples shows that in none of the samples was there any Pu-239 or Sr-90 detected. Cs-137 was detected, as could be expected from

the worldwide distribution of this radionuclide as a result of the atmospheric testing of nuclear weapons during the 1950's and early 1960's. In addition, naturally occurring radionuclides in the uranium and thorium decay series were also detected.

**Table 8-1. Composite risks for consumption of fish contaminated with radionuclides from the Columbia River Basin for the general public and CRITFC's member Tribes .**

Composite number (study sites)	Species	Unit (1 g/d)	Fish Consumption Rates			
			Average Public (7.5 g/d)	High Public (142.4 g/d)	Average CRITFC's member tribe (63.2 g/d)	High CRITFC's member tribe (389 g/d)
52 (9E,9F)	Largescale sucker	$6 \times 10^{-7}$	$5 \times 10^{-6}$	$9 \times 10^{-5}$	$4 \times 10^{-5}$	$2 \times 10^{-4}$
53 (9F,9H)	Largescale sucker	$9 \times 10^{-7}$ *	$7 \times 10^{-6}$ *	$1 \times 10^{-4}$ *	$6 \times 10^{-5}$ *	$4 \times 10^{-4}$ *
54 (9K)	White sturgeon	$6 \times 10^{-7}$	$5 \times 10^{-6}$	$9 \times 10^{-5}$	$4 \times 10^{-5}$	$2 \times 10^{-4}$
24 (7A)	White sturgeon	$1 \times 10^{-6}$ *	$8 \times 10^{-6}$ *	$1 \times 10^{-4}$ *	$6 \times 10^{-5}$ *	$4 \times 10^{-4}$ *
25 (8F)	White sturgeon	$8 \times 10^{-7}$ *	$6 \times 10^{-6}$ *	$1 \times 10^{-4}$ *	$5 \times 10^{-5}$ *	$3 \times 10^{-4}$ *
29 (8E,8B)	White sturgeon	$6 \times 10^{-7}$	$5 \times 10^{-6}$	$9 \times 10^{-5}$	$4 \times 10^{-5}$	$2 \times 10^{-4}$
84 (8F)	Channel catfish	$8 \times 10^{-7}$	$6 \times 10^{-6}$	$1 \times 10^{-4}$	$5 \times 10^{-5}$	$3 \times 10^{-4}$
85 (8F,8I)	Largescale sucker	$9 \times 10^{-7}$	$7 \times 10^{-6}$	$1 \times 10^{-4}$	$6 \times 10^{-5}$	$3 \times 10^{-4}$
86 (8C)	Channel catfish	$6 \times 10^{-7}$	$5 \times 10^{-6}$	$9 \times 10^{-5}$	$4 \times 10^{-5}$	$3 \times 10^{-4}$
30 (13E,13F)	White sturgeon	$8 \times 10^{-7}$	$6 \times 10^{-6}$	$1 \times 10^{-4}$	$5 \times 10^{-5}$	$3 \times 10^{-4}$
87 (9I)	White sturgeon	$7 \times 10^{-7}$	$5 \times 10^{-6}$	$1 \times 10^{-4}$	$4 \times 10^{-5}$	$3 \times 10^{-4}$
88 (9I)	White sturgeon	$7 \times 10^{-7}$	$5 \times 10^{-6}$	$1 \times 10^{-4}$	$4 \times 10^{-5}$	$3 \times 10^{-4}$
78 (9Q,9P)	Mountain whitefish	$8 \times 10^{-7}$	$6 \times 10^{-6}$	$1 \times 10^{-4}$	$5 \times 10^{-5}$	$3 \times 10^{-4}$
79 (9O,9N)	Mountain whitefish	$6 \times 10^{-7}$	$5 \times 10^{-6}$	$9 \times 10^{-5}$	$4 \times 10^{-5}$	$2 \times 10^{-4}$
82 (9D,9B,9A)	White sturgeon	$8 \times 10^{-7}$	$6 \times 10^{-6}$	$1 \times 10^{-4}$	$5 \times 10^{-5}$	$3 \times 10^{-4}$
83 (9A)	White sturgeon	$5 \times 10^{-7}$	$4 \times 10^{-6}$	$7 \times 10^{-5}$	$3 \times 10^{-5}$	$2 \times 10^{-4}$

\*Composites 53, 24, and 25 did not have uranium, strontium or plutonium analyses performed, and the composite risks do not include contributions from those radionuclides .

## 8.6 Uncertainties

The uncertainty associated with cancer risk estimates for ingestion of fish contaminated with radionuclides includes contributions from the analytical uncertainties of the reported results, and risk coefficients. The analytical uncertainties associated with the laboratory results are reported at the two standard deviation level. For radionuclide analyses, uncertainties related to counting statistics depend on the number of counts obtained, which varies with the analytical technique used as well as the concentrations of radionuclide in the sample. As a percentage of the reported result, their magnitude typically varies from a few percent in the case of gamma results which are significantly greater than detection limits (e.g. K-40 results), to 20-40% for uranium results, to more than 100% in cases of reported results which are classified as non-detect.

Some analytical results are qualified as estimated values due to interferences from other radionuclides in the analysis. Additional uncertainty results from the use of some radionuclides as surrogates for other radionuclides in decay series, the assumption of secular equilibrium, and the use of minimum detectable concentration data in calculating risk. These uncertainties likely result in overestimates of risk.

The uncertainties associated with the risk coefficients are likely to be larger than those due to analytical uncertainties. EPA guidance does not provide specific quantitative uncertainty estimates of the cancer risk coefficients (USEPA 1999d). National Council on Radiation Protection and Measurements. (NCRP) Report 126 (NCRP, 1997), examined the question of uncertainties in risk coefficients for the relatively simple case of external radiation exposure to low linear energy transfer (primarily gamma) radiation. The conclusion was that the 90% confidence interval encompassed a range approximately a factor of 2.5 to 3 higher and lower than the value of the risk estimate. Since estimates of risk from ingestion of food necessarily involve the added complexity of modeling of physiological processes to determine dose and risk, the uncertainties in this context are likely to be even greater.

The National Academy of Sciences Committee on the Biological Effects of Ionizing Radiation (BEIR), in their report, addressed the issue of uncertainty in risk estimates for low doses from low linear energy transfer radiation (NAS, 1990). BEIR V considered the assumptions inherent in modeling such risks and concluded that at low doses and dose rates it must be acknowledged that the lower limit of the range of uncertainty in the risk estimates extends to zero.

## 8.7 Discussion

Considering the number of samples, the mobility of the fish, and the range of results obtained, it does not appear to be possible to attribute results to specific sources. Most of the radionuclides detected are known to be present naturally in the environment. Cs-137 is also widespread in the environment and was detected in many samples without apparent pattern. There were three samples in the vicinity of the Hanford Reach (Columbia River study site 9U) which showed positive detection results for Sr-90.

Sr-90, like Cs-137, is a widespread radionuclide resulting from atomic testing in the atmosphere. It is also associated with Hanford operations and is known from other environmental studies to be present in Columbia River sediments near Hanford.

The estimated risks are similar across all composite groups (Table 8-1). This is consistent with the observation that the majority of the estimated risk is generally due to radionuclides which are members of naturally occurring decay chains.

## **8.8 Conclusions**

The risks calculated for fish consumption (Table 8-1) are small relative to the estimated risks associated with radiation from naturally-occurring background sources, to which everyone is exposed. In the US, the average annual effective dose equivalent is approximately 300 millirem including exposure to radon. The lifetime risk associated with this background dose can be estimated to be approximately  $1 \times 10^{-2}$ , or 1%.