

9.0 Comparisons of Fish Tissue Chemical Concentrations

9.1 Comparison by Chemical Concentration

In this section the fish tissue residues from our study are compared to other food types and studies of contaminants in fish reported in literature. This section also includes a comparison of fish tissue concentration data for smallmouth bass and channel catfish in addition to the 13 fish species which were the main focus of this report.

9.1.1 Chlordane

Chlordane was used as a pesticide from the 1940's until the late 1980's. Until 1983 it was used on corn and citrus fruits, lawns and gardens. It was banned in 1988.

Like most of the other cyclodiene pesticides (heptachlor, heptachlor epoxide, aldrin, dieldrin, endrin, and endosulfans I and II) chlordane degrades very slowly. Various of its metabolites can stay in the soil for over 20 years and can bioaccumulate in tissues of higher organisms.

Exposure to chlordane occurs largely from eating contaminated foods, such as root crops, meats, fish, and shellfish, or from touching contaminated soil. In the early 1980's chlordane was detected in 4 of 324 food composites: 3 potato composites ranging from trace to 2 µg/kg, and 1 garden fruit composite at a trace level (Gartrell et al., 1986). In the 1980 U.S. Food and Drug Administration (USFDA) market basket survey of infant and toddler diet samples, chlordane was detected at 5 µg/kg in one of 143 toddler food composites (Gartrell et al., 1985).

Chlordane concentrations of 118 to 290 µg/kg were measured in various estuarine fish in coastal states surveyed (Butler and Schutzmann, 1978). In a more recent survey, Munn and Gruber (1997) reported fish concentrations of 140 - 610 µg/kg of the sum of chlordane in composite samples of whole body fish from the Central Columbia Plateau.

The average concentrations of total chlordane found in anadromous fish tissue from our study ranged from <4 µg/kg in eulachon and coho salmon to 43 µg/kg in Pacific lamprey (Table 2-3). Egg samples from spring chinook sample had the highest average concentration (66 µg/kg) in our study (Table 2-3). The average concentrations of total chlordane in the resident fish species in our study ranged from < 2.4 µg/kg in rainbow trout and bridgelip sucker to 29 µg/kg in white sturgeon (Table 2-3).

9.1.2 Total DDT

The legal use of DDT in agriculture has been banned in the United States since 1972. DDT and its derivatives are persistent, bioaccumulative compounds which are ubiquitous in the organisms, sediments, and soils.

Exposure to DDT and its structural analogs (DDE, DDD) occurs primarily from eating contaminated foods, such as root and leafy vegetables, meat, fish, and poultry. From 1967 to 1972 the concentrations of total DDT in meat, fish and poultry decreased from 3,200 µg/kg to 900 µg/kg (IARC, 1978). From 1970 to 1973, DDE residues decreased only 27%, compared to a decrease of 86% and 89% for DDT and DDD, respectively (USEPA, 1980).

Based on data from the US Fish and Wildlife Service National Pesticides Monitoring Program (Schmitt et al., 1981), the DDT concentrations in fish ranged from 100 to 11,000 µg/kg.

DDT was detected in meats (0.3 µg/kg) and raw berries (2.0 µg/kg) consumed by indigenous residents of the Canadian Arctic (Berti et al., 1998).

The maximum concentration of DDE in the fish from several USGS surveys was in a whole body composite sample of carp (3,300 µg/kg) from the Brownlee Reservoir on the Snake River, Idaho (Table 9-1). The maximum concentration of DDE in our study was in the whole body composite sample of white sturgeon (1400 µg/kg) from the Hanford Reach of the Columbia River (study site 9U). The maximum concentrations of DDE in bridgelip sucker, rainbow trout, and largescale sucker levels in our study were higher than levels found by Munn and Gruber (1997) in the Central Columbia Plateau (Table 9-1). The largescale sucker levels in our study were similar to the largescale sucker levels reported by Clark and Maret (1998) for the Snake River Basin.

Table 9-1. Comparison of range concentrations of sum of DDE (o,p' & p,p') in whole body composite fish samples Columbia River Basin.

Fish	µg/kg	Location	Reference
carp	3300	Brownlee Reservoir, Snake River, Idaho	Clark and Maret ,1998
bridgelip sucker	87	Palouse River, Central Columbia Plateau	Munn and Gruber, 1997
bridgelip sucker	120-340	Northern Desert, Central Columbia	Munn and Gruber ,1997
bridgelip sucker	347 - 612	Columbia River Basin	Our study, 1996-1998
rainbow trout	9.5-32	Northern Desert, Central Columbia	Munn and Gruber, 1997
rainbow trout	5-89	Columbia River Basin	Our study, 1996-1998
largescale sucker	33-1300	Snake River Basin	Clark and Maret ,1998
largescale sucker	120-400	Palouse River, Central Columbia Plateau	Munn and Gruber, 1997
largescale sucker	29-1312	Columbia River Basin	Our study, 1996-1998

9.1.3 PCBs

PCBs, are stable, man-made chemicals that only degrade at very high temperatures. They do not conduct electricity and most of the various types of PCBs and PCB mixtures take the form of liquids. For these reasons, PCBs have been used extensively in much of the world as electrical insulating fluids, especially in capacitors and transformers which deliver high voltage in critical devices and situations where fire prevention is of great concern. PCBs have also been used extensively as hydraulic fluids, as well as in the manufacture of carbonless copy paper, etc. Environmental contamination with PCBs has resulted from industrial and domestic discharges, landfills, and atmospheric transport of incompletely incinerated PCBs.

Under environmental conditions, PCBs are extremely stable and slow to chemically degrade

(Eisler, 1986b). PCBs enter the environment as mixtures containing a variety of individual components (congeners) and impurities that vary in toxicity. The chlorinated nature of the various PCB molecules also makes them more fat soluble, and thus capable of bioaccumulating in aquatic food webs. The lipid solubility of the PCBs increases with increased chlorine substitution. This lipophilicity also tends to increase resistance to biodegradation.

Because of the relatively great environmental persistence and lipophilicity of this group of pollutants, low-level PCB contamination is now a global phenomenon, with PCB residues occurring almost universally in human milk, other human tissues, food, etc. For the general population, likely routes of ongoing chronic exposure to PCBs are primarily from food (Table 9-2).

Table 9-2. PCB residues in raw agricultural commodities, 1970-76.
(Source: Duggan et al, 1971)

Food Type	Number of samples	Percent Detected	Average ($\mu\text{g}/\text{kg}$)
fish	2,901	46	892
eggs	2,302	9.6	72
milk	4,638	4.1	67
cheese	784	0.9	11
red meat	15,200	0.4	8
poultry	11,340	0.6	6

The estimated PCB content of a typical teenage boy's diet was about 15 $\mu\text{g}/\text{day}$ in 1971, decreasing by 1975, to about 8.1 $\mu\text{g}/\text{day}$ (IARC, 1978). The levels of PCBs have declined in ready-to-eat foods from 1978 to 1982 (Table 9-3). However, the human body burden remains high. The body burden of PCBs in human fat ranged between 500 and 1,500 $\mu\text{g}/\text{kg}$ in 1987 (USEPA, 1987).

Table 9-3. The declining trends in PCBs in ready-to-eat foods collected in markets of a number of US cities (Source: Duggan et al., 1971).

Year	Number of samples	Percent Detected	Average ($\mu\text{g}/\text{kg}$)
1978	360	9	trace - 50
1979	360	4	<1 - 2
1980	360	2	2
1981- 82	324	2	1

In the 1980 -1981 USFWS survey of PCBs in fish from 107 locations the geometric was 530 $\mu\text{g}/\text{kg}$ (Schmitt et al., 1985). This was lower than mean PCB levels from previous monitoring efforts, in which geometric means for PCBs were 880 $\mu\text{g}/\text{kg}$ (1976-1977) and 850 $\mu\text{g}/\text{kg}$ from (1978- 1979) (Schmitt et al., 1985).

In a 1976-1980 EPA survey of PCB residues in finfish from the Chesapeake Bay watershed, the concentrations ranged from non detects to 4,640 $\mu\text{g}/\text{kg}$ (Tale 9-5). There was no trend over time as was observed in the USFWS Pesticide Monitoring Program.

Table 9-4. The 1976-80 ranges for PCB residues from 547 finfish from the Chesapeake Bay and its tributaries (Source: USEPA, 1987a).

<u>Year</u>	<u>µg/kg</u>
1976	ND - 980
1977	30 - 510
1978	60 - 4,640
1979	10 - 1,600
1980	3 - 1,450

In later studies concentrations of total PCBs in a variety of fish tissue types ranged from 10 µg/kg in white sucker fillets in Saginaw Bay, Lake Huron, Michigan to 14,500 µg/kg in fish from the Spokane River, Washington (Table 9-5). Measurements of Aroclor 1254 and 1260 in white croaker muscle in California ranged from 1 µg/kg to 713 µg/kg (Table 9-6).

Table 9-5. Total PCB concentrations in fish tissue from studies reported in the literature from 1978-1994.

<u>Species & Tissue type</u>	<u>µg/kg</u>	<u>Location/date of study</u>	<u>Reference</u>
fish livers	132 - 772	near the outfall for the Los Angeles County wastewater treatment plant 1980-81,	Gossett et al., 1983.
750 fish samples	70 - 14,500	11 major lakes and rivers in Alberta, Canada	Chovelon et al., 1984
25 white suckers fillets	10-180	Saginaw Bay, Lake Huron, 1979-1980	Kononen, 1989
freshwater fish (whole body) mean = 36 maximum =930		Spokane River, WA, 1999	Johnson, 2001

Table 9-6. Concentrations Aroclor 1254 & 1260 in white croaker muscle tissue from California water bodies in the spring of 1994. (Source: Fairey et al., 1997)

<u>µg/kg</u>	<u>Location</u>
137 - 613	13 locations throughout San Francisco Bay
1	Southern California Dana Point,
757	Malibu

The concentration of Aroclor 1254 ranged from 480 µg/kg to 9,930 µg/kg in lake trout from lakes in Michigan (Table 9-7). The concentration of Aroclor 1254 in resident fresh water species from our study ranged from 10 µg/kg in rainbow trout to 930 µg/kg in mountain whitefish.

Table 9.7. Concentrations of Aroclor 1254 in lake trout from lakes in Michigan during 1978-82 (Devault et al., 1986).

<u>µg/kg</u>	<u>Location</u>
5630 - 9930	Lakes Michigan
2100 - 3660	Lake Huron
480-1890	Lake Superior

The concentration of Aroclors in chinook salmon eggs from Lake Michigan were much higher

than the levels found in our study (Table 9-8).

Table 9-8. Aroclor concentrations in chinook salmon eggs reported for Lake Michigan, Michigan, compared to our study of Aroclors in the chinook salmon eggs.

$\mu\text{g}/\text{kg}$	N	salmon	Location/date of study
Aroclor 1254			
5,400		chinook	Lake Michigan, 1982 (Jaffet et al., 1985)
<i>12</i>	<i>1</i>	<i>fall chinook</i>	<i>Columbia River Basin, 1996-1998</i>
<i>15 - 20</i>	<i>6</i>	<i>spring chinook</i>	<i>Columbia River Basin, 1996-1998</i>
Aroclor 1260			
1,100		chinook	Lake Michigan, 1982 (Jaffet et al., 1985)
<i><19</i>	<i>1</i>	<i>fall chinook</i>	<i>Columbia River Basin, 1996-1998</i>
<i><18</i>		<i>spring chinook</i>	<i>Columbia River Basin, 1996-1998</i>

< = detection limit

Concentrations of PCBs measured in fish from our study were compared to other fish surveys in Lake Roosevelt on the upper Columbia River in Washington (Table 9-9). The maximum concentration of Aroclors 1254 and 1260 in walleye and rainbow trout were lower in our study of the Columbia River Basin than the EPA (USEPA, 1998c) and USGS (Munn, 2000) surveys of Lake Roosevelt, Washington. Concentrations of the Aroclors in white sturgeon were higher in our study than the EPA study of Lake Roosevelt, Washington (Table 9-9).

Table 9-9. Concentrations of Aroclors 1254 and 1260 in composite samples of fish fillets from Lake Roosevelt, Washington compared concentrations measured in our study of the Columbia River Basin.

Fish Species	$\mu\text{g}/\text{kg}$	N	Location	Reference
Aroclor 1254				
small walleye	30 - 10	9	Lake Roosevelt, 1994	USEPA, 1998c
large walleye	35 - 89	2	Lake Roosevelt, 1994	USEPA, 1998c
<i>walleye</i>	<i>12 - 14</i>	<i>7</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
white sturgeon*	15 - 77	2	Lake Roosevelt, 1994	USEPA, 1998c
<i>white sturgeon*</i>	<i>10 - 190</i>	<i>16</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
rainbow trout	13 - 45	10	Lake Roosevelt, 1994	USEPA, 1998c
rainbow trout	3 - 49	16	Lake Roosevelt, 1998	Munn, 2000
<i>rainbow trout</i>	<i>10 - 20</i>	<i>7</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
smallmouth bass	ND - 8	9	Lake Roosevelt, 1994	USEPA, 1998c
<i>smallmouth bass</i>	<i>38 - 83</i>	<i>3</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
kokanee	28 - 40	4	Lake Roosevelt, 1994	USEPA, 1998c
lake whitefish	31 - 51	3	Lake Roosevelt, 1994	USEPA, 1998c
Aroclor 1260				
small walleye	4 - 13	9	Lake Roosevelt, 1994	USEPA, 1998c
large walleye	23 - 32	2	Lake Roosevelt, 1994	USEPA, 1998c
<i>walleye</i>	<i><19</i>	<i>7</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
white sturgeon*	13 - 102	2	Lake Roosevelt, 1994	USEPA, 1998c
<i>white sturgeon*</i>	<i>13 - 200</i>	<i>16</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
rainbow trout	5 - 72	10	Lake Roosevelt, 1994	USEPA, 1998c
<i>rainbow trout</i>	<i><18</i>	<i>7</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
smallmouth bass	3 - 6	9	Lake Roosevelt, 1994	USEPA, 1998c
<i>smallmouth bass</i>	<i>68 - 220</i>	<i>3</i>	<i>Columbia River Basin, 1996-1998</i>	<i>our study</i>
kokanee	10 - 14	4	Lake Roosevelt, 1994	USEPA, 1998c
lake whitefish	16 - 29	3	Lake Roosevelt, 1994	USEPA, 1998c

N - number of samples < = detection limit *White sturgeon were individual fillets without skin

9.1.4 Chlorinated Dioxins and Furans

Because of their chlorination and specific chemical structures, most chlorinated dioxins and furans are highly fat soluble, and difficult for the body to quickly degrade and excrete. They are similar to some of the other persistent chlorinated residues like DDT and PCBs. Also like PCBs and DDTs, chlorinated dioxins and furans can bioaccumulate in fish. The amount of furans in fish can sometimes be tens of thousands times higher than the levels in the surrounding water.

The chlorinated dibenzodioxins and chlorinated dibenzofurans are not produced intentionally by industrial processes. Rather, most chlorinated dioxins and furans are generated in very small amounts as unwanted impurities during the manufacture of several chlorinated chemicals and consumer products, including certain wood treatment chemicals, some metals, and paper products. When the waste water, sludge, or solids from these processes are released into waterways or soil in dump sites, the sites may become contaminated with chlorinated dioxins and furans. These unwanted contaminants also enter the environment from burning municipal and industrial waste in incinerators, as well as from gasoline exhaust, and the burning of coal, wood, or oil for home heating and production of electricity. Other production chemicals which can generate unwanted trace amounts of 2,3,7,8-TCDD have included the forestry herbicide 2,4,5-trichlorophenoxy propionic acid (Silvex), and the industrial chemical 2,4,5-trichlorophenol. Unwanted trace amounts of some of the higher-chlorinated dioxins, especially the hexa and octa isomers, have also been associated with the production of the widely used wood preservative, pentachlorophenol.

Many of the various chemicals and processes which significantly produce chlorinated dioxins and furans in the environment are either being slowly phased out or are strictly controlled. It is currently believed that chlorinated dioxin and furan emissions associated with incineration and combustion activities are the predominant environmental source of these contaminants (USEPA, 2000e). Chlorinated dioxins and furans also arise from natural processes in the environment such as forest fires and volcanos.

TCDF is often found in fish tissue because of its affinity for lipids and because of its formation as a by-product in the industrial processes, especially pulp and paper mills (USEPA, 2000e). The concentration of 2,3,7,8-TCDF was measured in a variety of fish species from Lake Roosevelt, Washington by the USEPA in 1994 (Table 9-10). The concentrations of 2,3,7,8-TCDF in walleye ranged from 0.0001 to 0.0063 $\mu\text{g}/\text{kg}$ (Table 9-10). The maximum concentration from our study was lower than the maximum reported for Lake Roosevelt, Washington. The white sturgeon 2,3,7,8-TCDF maximum concentration in our study was higher than the maximum from the 1994 Lake Roosevelt study (Table 9-10). The rainbow trout 2,3,7,8-TCDF concentrations were similar in both studies.

Table 9-10. Concentrations of 2,3,7,8-TCDF in composite samples of fish filets collected from Lake Roosevelt, Washington in 1994 compared with our 1996-1998 survey of the Columbia River Basin.

Fish	µg/kg	N	Collection date	Reference
small walleye	0.0001 - 0.0016	9	Lake Roosevelt, 1994	USEPA, 1998c
large walleye	0.0007 - 0.0063	2	Lake Roosevelt, 1994	USEPAc 1998c
<i>walleye</i>	0.0006 - 0.00085	3	Columbia River Basin, 1996-98	our study
white sturgeon	0.016 - 0.025	2	Lake Roosevelt, 1994	USEPA, 1998c
<i>white sturgeon</i>	0.0025 - 0.054	16	Columbia River Basin, 1996-98	our study
small rainbow trout	0.000098 - 0.0015	6	Lake Roosevelt, 1994	USEPA, 1998c
large rainbow trout	0.0015 - 0.00188	10	Lake Roosevelt, 1994	USEPA, 1998c
<i>rainbow trout</i>	0.0001 - 0.0003	7	Columbia River Basin, 1996-98	our study
kokanee	0.0028 - 0.0031	4	Lake Roosevelt, 1994	USEPA, 1998c
smallmouth bass	0.00001 - 0.0041	9	Lake Roosevelt, 1994	USEPA, 1998c
lake whitefish	0.0038 - 0.01610	3	Lake Roosevelt, 1994	USEPA, 1998c

N= number of samples

In the USEPA National Dioxin Survey (USEPA, 2000d) background levels of toxicity equivalence concentrations for chlorinated dioxins, furans, and dioxin-like PCB congeners were 0.00116 ± 0.00121 µg/kg in fish and 0.00046 ± 0.00099 µg/kg in beef. In our study the average toxicity equivalence concentrations ranged from a low of 0.0004 µg/kg in fall chinook salmon to the highest average concentration of 0.0063 µg/kg in mountain whitefish.

9.1.5 Metals

The metals measured in our study are naturally occurring substances. Some of these metals are essential at trace levels for survival of vertebrates. These chemicals may combine with other chemicals to form compounds, (e.g. methylmercury, dimethylarsenic, arsenocholine, arsenosugars) which alters their bioavailability and toxicity. Most can become toxic if sufficiently high levels are encountered in the environment. Many of the metals which are taken up by fish tend to increase in concentration as the organisms age and increase in body size (Wiener and Spry, 1996, reported in Clark and Maret, 1998).

Information about barium, beryllium, cobalt, and manganese and are not included in this section. Background information on these chemicals is included in the Toxicity Profiles (Appendix C)

9.1.6 Aluminum

Aluminum is the most common and widely distributed metal in the earth's crust. Concentrations as high as 150,000 - 600,000 mg/kg have been reported in soil. The average ingestion of aluminum by humans has been estimated at 30 - 50 mg/day (Bjorksten, 1982). This estimate may be low, in light of a 1997 United Kingdom (UK) total diet study involving 20 different food groups from 20 representative towns, for the general UK population, where the highest mean concentrations of aluminum were found in the bread (6,600 µg/kg) and fish (6,100 µg/kg) (Ysart et al., 2000). Aluminum is present in the natural diet, in amounts varying from very low in animal products to relatively high in plants.

In our study the basin-wide average aluminum concentrations ranged from non-detect in coho salmon (whole body and fillet) to 69,000 µg/kg in whole body largescale sucker. The maximum concentration was 190,000 µg/kg in the largescale sucker composite sample from the main-stem Columbia River (study site 8).

9.1.7 Arsenic

Arsenic is found widely in nature, and occurs most abundantly in sulfide ores. Arsenic levels in the earth's crust average about 5,000 µg/kg. Arsenic is found in trace amounts in aquatic environments. As was described in Section 5, arsenic exists in both organic and inorganic forms. The most common combined form of arsenic is the inorganic compound, arsenopyrite (FeAsS). The organic arsenic compounds are less toxic than the inorganic arsenic compounds.

Arsenic does not readily bioconcentrate in aquatic organisms. It is typically water soluble and does not combine with proteins. Since, aquatic invertebrates accumulate arsenic more readily than fish biomagnification is unlikely (Spehar et al., 1980). Planktivorous fish are more likely to concentrate arsenic than omnivorous or piscivorous fishes (Hunter et al., 1981). Eisler (1988a) found no evidence that biomagnification occurs in aquatic food chains. In 1995, Robinson et al., found no evidence of arsenic uptake or accumulation from water in both rainbow and brown trout. The rainbow trout in our study had the lowest arsenic concentrations (<25 µg/kg fillet; 120 µg/kg whole body) of the fish species sampled.

In a 1997 UK study, dietary exposures to arsenic were estimated to be about 65 µg /day (Ysart et al., 2000). The “fish” food group had the highest mean arsenic concentration (400 µg/kg; Ysart et al., 2000).

Arsenic levels recorded for fish tissues seem to be quite variable. Fish taken from the Great lakes contained 5.6 - 80 µg/kg arsenic; primarily in the lipid fraction of the fish tissue (Lunde, 1970). In a study of African tilapia fish, muscle tissue contained arsenic levels ranging from 110 µg/kg (Ikdu and Marget Lakes) to one specimen with 10,500 µg/kg (Abu Quir Bay) (El Nabawi et al., 1987). Ashraf and Jaffar (1988) measured arsenic levels of 2,880 µg/kg and 2510 µg/kg in two tuna species from the Arabian Sea. The authors noted that increased arsenic content was proportional to increased weight in the tuna species.

The average arsenic levels in resident, fresh water fish species in our study ranged from not detect in rainbow trout fillet to 490 µg/kg in whole body walleye (Table 2-14). The average concentrations in anadromous species from our study ranged from 310 µg/kg in Pacific lamprey fillet to 890 µg/kg in whole body eulachon. There was no correlation between lipid and arsenic in fish in our study, as was observed in the Great Lakes study (Lunde, 1970) or body weight and arsenic as observed by Asraf and Jaffar (1988).

9.1.8 Cadmium

Cadmium naturally occurs in the aquatic environment, but is of no known biological use and is considered one of the most toxic metals. While cadmium is released through natural processes, anthropogenic cadmium emissions have greatly increased its presence in the environment. In aquatic systems, cadmium quickly partitions to sediment, but is readily remobilized through a variety of chemical and biological processes (Currie et al., 1997). Cadmium does not bioconcentrate significantly in fish species, but does tend to accumulate more readily in invertebrates. Omnivorous and insectivorous predators tend to accumulate cadmium in their tissues more than piscivorous predators (Scheuhammer, 1991). Saiki et al., (1995) found no evidence of biomagnification of cadmium in steelhead on the Upper Sacramento River. Eisler (1985a) also maintains that evidence for cadmium biomagnification suggests that only the lower trophic levels exhibit biomagnification. Cadmium tends to form stable complexes with metallothionein (a sulfhydryl-rich protein). The resulting cadmium complexes have long half-lives and a tendency to accumulate with age in exposed organisms. As such, long lived species tend to be at a higher risk from chronic low-level dietary cadmium exposure.

People who are smokers are exposed to significant levels of inhaled cadmium. The major exposure route for the non-smoking human population is via food. In a 1997 UK study, the mean population dietary exposures to cadmium was estimated to be about 12 $\mu\text{g}/\text{kg}/\text{day}$ for the general UK population (Ysart et al., 2000). Cadmium concentrations were highest in the viscera and trimmings of animals (77 $\mu\text{g}/\text{kg}$), and nuts (59 $\mu\text{g}/\text{kg}$), while the bread and potato food groups made up the greatest contributions (both 25%) to dietary exposure of the general population.

Certain cruciferous vegetable crops are known to be able to sequester elevated cadmium levels if grown in sufficiently contaminated soils. Queiroloa et al. (2000) reported ranges of 0.2 to 40 $\mu\text{g}/\text{kg}$ for cadmium, with highest levels being found in potato skin in a study of vegetables (broad beans, corn, potato, alfalfa and onion) from farming villages in Northern Chile.

The WHO (1992) indicates that marine organisms generally contain higher cadmium residues than their freshwater and land-dwelling counterparts. In our study the highest cadmium levels were in whole body samples of largescale sucker (250 $\mu\text{g}/\text{kg}$) followed by spring chinook salmon (170 $\mu\text{g}/\text{kg}$) and Pacific lamprey (150 $\mu\text{g}/\text{kg}$).

Average cadmium concentrations ranged from non detect in fillet samples of walleye, coho salmon, and fall chinook salmon to 120 $\mu\text{g}/\text{kg}$ in whole body spring chinook salmon. The maximum concentration (250 $\mu\text{g}/\text{kg}$) was in the largescale sucker composite sample from the Hanford Reach of the Columbia River (study site 9U).

9.1.9 Chromium

Chromium is widely distributed in the earth's crust, with an average concentration of about 125,000 $\mu\text{g}/\text{kg}$. It is found in small amounts in all soils and plants. Most of the chromium present in food is in the trivalent form [Cr(III)], which is an essential nutrient. The hexavalent

form is more toxic, but is not normally found in food. In freshwater environments, hydrolysis and precipitation are the most important processes in determining the environmental fate of chromium, while absorption and bioaccumulation are considered minor. Chromium (VI) is highly soluble in water and thus very mobile in aquatic systems (Ecological Analysts, 1981).

The mean daily dietary intake of chromium from air, water, and food, is estimated to be about 0.2 - 0.4 µg, 2.0 µg, and 60 µg, respectively (ATSDR, 2000). The predicted intakes from air chromium are probably exceeded considerably in the case of smokers, and those who are occupationally exposed.

In a 1997 UK study, meat products contained the highest mean chromium concentration (230 µg/kg), but beverages made the greatest dietary contribution (19%) to the population exposure to chromium (Ysart et al., 2000). The US Food and Nutrition Board has recommended a safe and adequate dietary intake of chromium of 0.05 - 0.20 µg/day (Seller and Sigel, 1988).

Chromium was found in fish sampled from 167 lakes in the northeast United States at levels ranging from 30-1,460 µg/kg with a mean of 190 µg/kg (Yeardley et al., 1998). Seaweeds have been shown to sequester total chromium by a bioaccumulation factor of about 100 times greater than ambient levels in seawater (Boothe and Knauer, 1972). Snails showed an accumulation factor of 1×10^6 for total chromium (Levine, 1961).

In our study, basin-wide average chromium concentrations ranged from <100 µg/kg in eulachon to 360 µg/kg in the whole body white sturgeon (Table 2-14). The maximum concentration (1000 µg/kg) was measured in the whole body white sturgeon sample from the main-stem Columbia River (study site 8)

9.1.10 Copper

Because of its ubiquitous occurrence in the environment, and its essentiality for life, copper is found naturally at trace levels in aquatic and terrestrial organisms. Copper is not strongly bioconcentrated in vertebrates, but is more strongly bioconcentrated in invertebrates. In salmonids the accumulation of copper in muscle, kidney, and spleen tissues occurred at copper concentrations ranging from 0.52-3 µg/L in both seawater and freshwater (freshwater hardness=46-47 mg/L)(Camusso and Balestrini, 1995; Peterson et al., 1991; Saiki et al., 1995). The concentrations of copper in fish tissues reflect the amount of bioavailable copper in the environment. Baudo (1983, Wren et al. (1983), and Mance (1987) have all concluded that copper, along with zinc and cadmium do not biomagnify in the aquatic environment.

Intake of copper from food tends to be about one order of magnitude greater than intake from drinking water (USEPA, 1987). Exceptions to this are in relatively rare situations involving consumption of “soft” drinking water sources supplied by copper pipes; which can result in daily individual drinking water intakes of copper in excess of 2 mg/day. In a 1997 UK diet study, copper was highest in viscera and trimmings (50,000 µg/kg) and nuts (8,500 µg/kg), with mean concentrations in the other food groups ranging from 50 to 2,100 µg/kg (Ysart et al., 2000).

In our study, the copper concentrations ranged from 250 µg/kg in white sturgeon fillet sample to 4500 µg/kg in whole body Pacific lamprey. The maximum concentration (14,000 µg/kg) was in the whole body fall chinook salmon composite sample from the main-stem Columbia River (study site 14).

9.1.11 Lead

Lead is a naturally occurring, ubiquitous compound that can be found in rocks, soils, water, plants, animals, and air. Lead is the fifth most prevalent commercial metal in the US. Lead is found naturally in all plants, with normal concentrations in leaves and twigs of woody plants of about 2,500 µg/kg, pasture grass 1,000 µg/kg, and cereals from 100 -1,000 µg/kg (IARC, 1980).

Absorption of lead by aquatic animals is affected by the age, gender and diet of the organism, as well as the particle size, chemical species of lead, and presence of other compounds in the water (Eisler, 1988b; Hamir et al., 1982). Although inorganic lead is poorly accumulated in fish, it has been shown to bioconcentrate in aquatic species. Invertebrates tend to have higher lead bioconcentration factors than vertebrates. A bioconcentration factor of 42 was observed in brook trout embryos (Eisler, 1988b). Bioconcentration factors decrease as waterborne lead concentrations increase, thus suggesting accelerated depuration or saturation of uptake mechanisms (Hodson et al., 1984). Exposures of rainbow trout to 3.5-51 µg/L tetramethyl lead from 7 - 14 days resulted in rapid accumulation of lead. However, once the fish were removed to clean water, lead decreased rapidly from organs, followed by a slower release from other body components, until baseline levels were reached. An increase in dietary calcium of 0-8400 µg/kg reduced the uptake of waterborne lead in coho salmon, possibly due to interactions with gill membrane permeability (Hodson et al., 1984). In vertebrates, lead concentrations tend to increase with age and localize in hard tissues such as bone or teeth.

The primary exposure route for lead is food (Table 9-11). Foods which are likely to have elevated lead levels are dried foods, liver, canned food, and vegetables which have a high area-to-mass ratio. Historic use of soldered food cans greatly increased the lead content of prepared and processed foods. Sherlock (1987) reported that while ravioli from welded (no lead) cans contained 30 µg/kg lead, ravioli from a 98% lead soldered can was found to contain a mean content of 150 µg/kg lead.

Table 9-11. Lead concentrations in food purchased in five Canadian cities between 1986 - 1988 (Source: Dabeka and McKenzie, 1995).

category	% contribution to dietary intake	mean $\mu\text{g}/\text{kg}$	maximum $\mu\text{g}/\text{kg}$
fruits and fruit juice	13.9	44.4	372.7
miscellaneous	6.1	41.7	178.9
vegetables	16.8	24.4	331.7
meat and poultry	7.6	20.2	523.4
<i>fish</i>	0.7	19.3	72.8
sugar and candies	1.5	18.3	111.6
soups	4.5	15.5	48.7
bakery goods and cereals	20.6	13.7	66.4
beverages	20.9	9.9	88.8
fats and oils	0.3	9.6	19.7
milk and milk products	7.1	7.7	44.7
canned and raw cherries			203
canned citrus fruit			126
canned beans			158
canned luncheon meats			163

The basin-wide average lead concentrations in fish from our study of the Columbia River Basin ranged from non detect in fillets of Pacific lamprey, walleye, and rainbow trout to 500 $\mu\text{g}/\text{kg}$ in whole body eulachon (Table 2-14). The maximum concentration (1200 $\mu\text{g}/\text{kg}$) in our study was in the whole body fall chinook salmon from the main-stem Columbia River (study site 14).

9.1.12 Mercury

While mercury does occur naturally in small amounts in aquatic environments, the cycling of mercury prolongs the influence of man-made mercury compounds (Hudson et al., 1995). Mercury is cycled through the environment through an atmospheric-oceanic exchange. This cycling is facilitated by the volatility of the metallic form of mercury. Natural bacterial transformation of mercury results in stable, lipid soluble, alkylated compounds such as methyl mercury (Beijer and Jernelov, 1979). In sediments, mercury is usually found in its inorganic forms, but aquatic environments are a major source of methyl mercury (USEPA, 1985). In background freshwater systems, mercury occurs naturally at concentrations of 0.02-0.1 $\mu\text{g}/\text{L}$ (Moore and Ramamoorthy, 1984).

Mercury has been shown to bioconcentrate in a variety of aquatic organisms. Aquatic predators face the greatest danger of bioconcentrating mercury, and thus their tissue concentrations best reflect the amount of mercury available to aquatic organisms in the environment. Fish have been shown to concentrate mercury as methyl mercury even when they are exposed to inorganic mercury. Fish, such as rainbow trout, have been found to accumulate mercury in the form of methyl mercury at aquatic concentrations as low as 1.38 ng/L (Ponce and Bloom, 1991).

Some evidence supports the biomagnification of mercury in aquatic food chains. When comparing benthic feeding fish, fish that feed on plankton, invertebrates, and vertebrates, the

greatest mercury concentrations were found in piscivorous fishes. Thus, the authors of this study concluded that mercury content in fish increased with higher trophic levels (Wren and MacCrimmon, 1986).

Freshwater ecosystems historically associated with heavy gold mining activity have often been impacted by elevated mercury levels in fish. This is in large part due to the use of liquid elemental mercury, or quicksilver, as a means of separating out gold during the mining process, especially during historic times.

Dietary sources greatly exceed other media like air and water as a source of human mercury exposure and uptake. In a 1997 UK diet study, fish contained the highest mean concentration (43 µg/kg), and made the greatest contribution (33%) to the population dietary exposure estimate (Ysart et al., 2000). The World Health Organization, EPA, and others indicate that risk to humans from mercury contamination via ocean fish is mainly through the consumption of predator species like swordfish, king mackerel, and shark (WHO, 1976).

In a monitoring study of fish in British Columbia, Canada, mercury concentrations in muscle tissue of various fish ranged from 40 µg/kg in rainbow trout to 2,860 µg/kg in lake trout (Table 9-12). In our study, rainbow trout the average mercury concentrations ranged from 73 µg/kg in whole body samples to 77 µg/kg in the fillet samples (Table 2-14).

Fish Species (study location)	µg/kg
Rainbow trout (Tezzeron Lake)	40
herring	70
dolly varden or char (Carpenter Lake)	410-1,940
dogfish or shark (English Bay)	1,080
lake trout (Pinchi Lake)	2,860

A 1984 EPA national survey of fish tissue found mercury ranging from 50 µg/kg in salmon to 610 µg/kg in pike (Table 9-13). In our study average mercury concentrations in fillet samples of salmon was 84 µg/kg in fall chinook, 100 µg/kg in spring chinook, and 120 µg/kg in coho. (Table 2-14).

Table 9-13. EPA 1984 survey of total mercury concentrations in edible fish tissue, shrimp, and prepared foods. (Source USEPA, 1984b)

Fish Species	µg/kg	Invertebrates	µg/kg	Prepared food	µg/kg
salmon	50	shrimp	460	fish sticks	210
whiting	50			canned tuna	240
sardines	60				
flounder	100				
snapper	450				
bass	210				
catfish	150				
trout	420				
pike	610				

In a more recent EPA national survey of mercury in fish tissue, median mercury levels ranged from 1 µg/kg in largemouth bass, channel catfish, bluegill sunfish, and common carp to 8,940 µg/kg in largemouth bass (Table 9-14). The concentrations of mercury fillets of fish tissue in our study were 380 - 470 µg/kg in smallmouth bass, 160 - 200 µg/kg in walleye, and 240 - 280 µg/kg in channel catfish (Table 9-27). All of these fish species had lower concentrations in our study than in the EPA 1990-1995 survey (USEPA, 1999e).

Table 9-14. Mercury concentrations from an EPA 1990 - 1995 national survey of fish fillets (Source : USEPA, 1999e).

Species	µg/kg
largemouth bass	1 - 8,940
Smallmouth bass	8 - 3,340
walleye	8 - 3,000
northern pike	100 - 4,400
channel catfish	1 - 2,570
bluegill sunfish	1 - 1,680
common carp	1 - 1,800
white sucker	2 - 1,710
yellow perch	10 - 2,140

In 1999, May et al. (2000) collected 141 samples of fish from reservoir and stream areas in the Bear and South Yuba River watersheds in the Sierra Nevada of Northern California (Table 9-15). Fish concentrations in the California survey ranged from 20 µg/kg to 1,500 µg/kg (Table 9-15). Rainbow trout mercury concentrations in fillets ranged from 45 - 150 µg/kg (Table 9-27). Channel catfish mercury concentrations ranged from 240 - 280 µg/kg (Table 9-27).

Table 9-15. USGS survey of mercury concentrations in fish tissue from reservoirs and streams in Northern California. (Source: May et al, 2000). Fish were fillets without skin

Reservoir	µg/kg
largemouth bass	20 - 1,500
Reservoir sunfish	< 100 - 410
channel catfish	160 - 750
Streams	µg/kg
Brown trout	20 - 430
rainbow trout	60 - 380

Several recent surveys in Washington measured concentrations of mercury in resident fish species (Table 9-16). The walleye samples from our study were within the range of the samples from Munn and Short (1997) and Munn (2000). Smallmouth bass from our study were within the range of the studies by Munn et al. (1995) and Sedar et al. (2001) although the maximum concentrations in our smallmouth bass were lower than the levels found in Lake Roosevelt, Washington (Munn et al., 1995) and Lake Whatcom (Serdar et al., 2001). Serdar et al., (2001) reported a mean concentration of (70 µg/kg) in most fish species in Washington State. The authors found higher concentrations of mercury in 6 of 8 fillets with the skin off. In our study all the fillets, except white sturgeon, were analyzed with skin. There was also no consistent pattern between fillets with skin or whole body. Rainbow trout concentrations from our study were also within the range observed in rainbow trout from Lake Roosevelt, Washington, although the maximum was lower than the maximum observed in Lake Roosevelt (Munn et al, 1995).

Table 9-16. Mercury concentrations in fish fillets collected in Lake Whatcom and Lake Roosevelt, Washington compared to our study of the Columbia River Basin .

Fish species	Tissue Type	µg/kg	N	Location	
walleye	composite	110 - 440	34	Lake Roosevelt, 1994	Munn and Short 1997
walleye	individual	110 - 150	8	Lake Roosevelt, 1998	Munn 2000
walleye	composite	160 - 200	3	Columbia River Basin, 1996-1998	our study
smallmouth bass	composite	160 - 620	5	Lake Roosevelt, 1994	Munn et al., 1995
smallmouth bass	individual	100 - 1840	96	Lake Whatcom, 2000	Serdar et al., 2001
smallmouth bass	composite	380 - 470	3	Columbia River Basin, 1996-1998	our study
rainbow trout	individual	110 - 240	6	Lake Roosevelt, 1994	Munn et al., 1995
rainbow trout	composite	45 - 150	7	Columbia River Basin, 1996-1998	our study
perch	individual	120 - 290	30	Lake Whatcom, 2000	Serdar et al., 2001
kokanee	individual	100 - 130	30	Lake Whatcom, 2000	Serdar et al., 2001
pumpkinseed	individual	70 -120	30	Lake Whatcom, 2000	Serdar et al., 2001
cutthroat trout	individual	60 - 80	30	Lake Whatcom, 2000	Serdar et al., 2001
brown bullhead	individual	70 - 440	30	Lake Whatcom, 2000	Serdar et al., 2001

N= Number of samples

9.1.13 Nickel

Nickel occurs naturally in rocks and soils and can leach into aquatic environments. However, weathering of nickel-containing substrates results in only small amounts of nickel entering into aquatic systems. Manmade sources of nickel include mining, combustion of coal, petroleum and tobacco, manufacture of cement and asbestos, food processing, textile and fur fabrication,

laundries, and car washes (USEPA, 1983). The National Academy of Sciences reports that fish contain nickel at a maximum of 1,700 µg/kg (NAS, 1975).

Nickel concentrations the maximum nickel concentration was 17,000 µg/kg in a whole body steelhead sample from the Klickitat River (study site 56). This sample was an anomaly since the other samples from this site were 170 and 520 µg/kg. The average concentrations in fillet samples ranged from 15 µg/kg in Pacific lamprey to 260 µg/kg in walleye; whole body ranged from 50 µg/kg in eulachon to 1200 µg/kg in Coho salmon.

9.1.14 Selenium

While selenium is ubiquitous in the earth's crust, only trace levels normally occur in aquatic environments. Selenium enters aquatic habitats from a number of anthropogenic and natural sources. Elevated levels in aquatic systems are found in regions where soil is selenium-rich or where soils are extensively irrigated (Dobbs et al., 1996). As an essential micronutrient, selenium is used by animals for normal cell functions. However, the difference between useful amounts of selenium and toxic amounts is small. Selenium at low levels in the diet is an essential element for humans. At elevated dose levels, it exhibits toxicity (selenosis). Organic and reduced forms of selenium (e.g. seleno-methionine and selenite) are generally more toxic and will bioaccumulate (Besser et al., 1993; Kiffney and Knight, 1990). Bioconcentration of selenium may be modified by water temperature, age of receptor organism, organ and tissue specificity, and mode of administration (Eisler, 1985a). Fish bioconcentrate selenium in their tissues with particularly high concentrations observed in ovaries when compared to muscle tissues (Lemly, 1985; Hamilton et al., 1990) and milt (Hamilton and Waddall, 1994). Selenium that is bioconcentrated appears to occur in its most harmful concentrations in predator species such as chinook salmon (Hamilton et al., 1990). Bioconcentration factors (BCFs) in rainbow trout range from 2-20 after exposure to 220-410 µg/L selenium. The magnitude of the BCFs appeared to be inversely related to exposure concentrations (Adams and Johnson, 1977). Biomagnification of selenium has also been well documented. The magnitude of the biomagnification ranges from 2-6 times between producers and lower consumers (Lemly and Smith, 1987). Piscivorous fish accumulate the highest levels of selenium and are generally one of the first organisms affected by selenium exposure, followed by planktivores and omnivores (Lemly, 1985).

Selenium has been frequently detected in a great variety of commonly consumed foods. In a 1997 UK diet study the mean selenium concentrations in the viscera and trimmings was estimated to be 490 µg/kg and 250 µg/kg in nuts (Ysart et al., 2000). Meat products (15%), fish (13%), and bread (13%) groups make the greatest contributions to diet (Ysart et al., 2000).

In the US infant diet the average concentration of selenium was highest in grains and cereals followed by fish (Table 9-17).

Table 9-17. Selenium concentrations in US infant diet. (Source: Gartrell et al., 1985 and 1986).

Food Group	1979 $\mu\text{g}/\text{kg}$	1981-1982 $\mu\text{g}/\text{kg}$
other dairy products	2	15
potatoes	2	2
beverages	2	
whole milk	4	9
vegetables	4	7
sugars and adjuncts	11	
oils and fats	12	5
meat, fish and poultry	107	112
grain and cereals	156	192

Selenium is well known to accumulate in living tissues. Selenium has been found in marine fish meal at levels of about 2,000 $\mu\text{g}/\text{kg}$, which is about 50,000 times greater than the selenium levels in seawater (Wilbur, 1980). Table 9-18 is a list of selenium concentrations in a variety of fish tissue types.

Table 9-18. Concentrations of selenium in fish reported in the literature.

Fish type	$\mu\text{g}/\text{kg}$	Location and date	Reference
Mean			
Razorback sucker eggs	3,700 - 10,600	Utah (1992)	Hamilton and Waddell, 1994
largemouth bass and bluegills gonads	2,630 - 4,640	power plant cooling reservoirs (1994)	Baumann and Gillespie, 1986
rainbow trout, edible portion	270	Toronto Harbor, Canada 1980	Davies, 1990
northern pike, edible portion	250	Toronto Harbor, Canada 1980	Davies, 1990
Geometric mean			
freshwater fish	560	112 selected US monitoring stations during from 1976-1979	Lowe et al., 1985
	460		
	470		
brown trout liver	6,290	South Platte River Basin in 1992 -93	Heiny and Tate, 1997
carp liver	8,130	South Platte River Basin in 1992 -93	Heiny and Tate, 1997
white sucker liver	17,900	South Platte River Basin in 1992 -93	Heiny and Tate, 1997
lake trout	500 to 860	Lake Huron from 1980 - 85	Great Lakes Water Quality Board, 1989
walleye and splake /backcross lake trout	650 to 790	Lake Huron 1980 - 85	Great Lakes Water Quality Board, 1989
walleye and splake /backcross lake trout	700 to 790	Lake Huron 1979 and 1985,	Great Lakes Water Quality Board, 1989
Maximum			
carp	3,650	Colorado River 1978 -79,	Lowe et al., 1985

The average concentrations of selenium in our study ranged from 220 $\mu\text{g}/\text{kg}$ in a rainbow trout fillet to 1,100 $\mu\text{g}/\text{kg}$ in the white sturgeon fillet (Table 2-14). The maximum concentration (2700 $\mu\text{g}/\text{kg}$) was in a white sturgeon fillet sample from the Hanford Reach of the Columbia River (study site 9U).

9.1.15 Vanadium

Vanadium is found in vegetables from about 0.5 to 2 µg/kg, with an average of about 1 µg/kg (Beyerrum, 1991). Veal and pork have been found to contain about 0.1 µg/kg. According to ATSDR (1992), foods containing the highest levels of vanadium include ground parsley, 1,800 µg/kg; freeze-dried spinach, 533 - 840 µg/kg; wild mushrooms, 50 - 2,000 µg/kg; and oysters, 455 µg/kg. Intermediate levels are found in certain cereals, like maize (0.7 µg/kg), and Macedonian rice 30 µg/kg). Also vanadium has been found in beef at 7.3 µg/kg, and in chicken at about 38 µg/kg. Seller and Sigel (1988) indicate that beverages, fats, oils, and fresh fruits and vegetables contained the least vanadium, ranging from less than 1 to about 5 µg/kg. Grains, seafoods, meats, and dairy products were generally from about 5 to 30 µg/kg. Prepared food ranged from 11 to 93 µg/kg, and dill seed and black pepper contained 431 and 987 µg/kg vanadium, respectively. ATSDR (ATSDR, 1992) indicates that in general, seafoods have been found to contain somewhat higher levels of vanadium than do tissues from terrestrial animals.

Mackeral has been found to contain about 3.5 µg/kg of vanadium, with 28 µg/kg in freeze-dried tuna (ATSDR, 1992). Konasewich et al. (1978) found vanadium in whole-fish samples of burbot and bloater chub taken from Lake Huron at concentrations of 75 µg/kg and 260 µg/kg, respectively. The same authors also found vanadium in whole samples of lake trout from Lake Superior, at 85 µg/kg. Nakamoto and Hassler (1992) found vanadium in the carcasses of male and female bluegill taken from the Merced River and the Salt Slough, California, at mean concentrations of 2,200 and 1,700 µg/kg, respectively.

In our study the average vanadium concentrations ranged from 5 µg/kg in fillet samples of spring chinook salmon and walleye to 310 µg/kg in whole body largescale sucker. The maximum concentration (770 µg/kg) was in a whole body rainbow trout composite sample from the Umatilla River (study site 101).

9.1.16 Zinc

Zinc occurs naturally in the earth's crust at an average concentrations of about 70,000 µg/kg. It is introduced into aquatic systems via leaching from igneous rocks. Zinc is found in all living organisms and is an essential element for growth, development and reproduction. However aquatic animals tend to accumulate excess zinc which can result in growth retardation, hyperchromic anemia, and defective bone mineralization. Because zinc combines with biomolecules in target species and most of these species accumulate more than they need for normal metabolism, data showing bioconcentration factors for target receptors may be misleading. Bioconcentration factors (BCF's) reported by EPA ranged from 51 in Atlantic salmon (*Salmo salar*) to 1,130 for the mayfly (*Ephemera grandis*) (USEPA, 1987c). Little to no evidence exists indicating the successive biomagnification of zinc in tissues of fish and avian receptors (USEPA, 1987c).

In the ATSDR survey of food groups the levels for zinc ranged from 29,200 µg/kg in fish/meat/poultry to 2,300 µg/kg in leafy vegetables (Table 9-19).

Table 9-19. Concentrations of zinc in food groups. (Source: ATSDR, 1993)

Food Group	µg/kg	Food Group	µg/kg
meat/fish/poultry	29,200	dairy products	4600
grain/cereals	8,700	legumes	8300
legumes	8,300	leafy vegetables	2300
legumes	8,300		

The average concentrations of zinc in whole body fish tissue from our study ranged from 3800 µg/kg in the white sturgeon fillet to 30,000 µg/kg in the whole body coho salmon (Table 2-14). The maximum concentration (40,000 µg/kg) was in the whole body mountain whitefish from the Deschutes River (study site 98).

9.2 Comparisons By Fish Species

This section includes general descriptions of each of the chemicals measured in this study followed by brief comparisons of these chemicals with data reported in databases or other studies. More information about each chemical is provided in Appendix C (Toxicity Profiles). In addition to chemical descriptions, this section includes a summary of the life history of the fish species. This brief discussion of the habitat preferences and feeding habits is intended to provide some understanding of how the fish may be exposed to pollutants. Appendix B (Fish Life Histories) contains detailed information on each fish species.

The chemical levels measured in fish tissue from our study in largescale and bridgelip sucker, mountain whitefish, rainbow trout, channel catfish, smallmouth bass, fall and spring chinook, and coho were compared with levels reported in 4 databases and two other similar studies in the Columbia River Basin. Only those concentrations which had more than a 10 fold difference are discussed.

Information on white sturgeon, walleye, steelhead, eulachon, and Pacific lamprey was not found in these databases or reports. However their life histories and a synopsis of the literature information described in Section 9.1 are added to this section to complete the summary for all species from this study.

The 4 databases were developed by:

- 1) the USGS, National Contaminant Biomonitoring Program (NCBP) database (Schmitt et al., 1999a),
- 2) the USGS, Biomonitoring of Environmental Status and Trends (BEST) database (Schmitt et al., 1999b)
- 3) the State of Washington, Puget Sound Ambient Monitoring Program (PSAMP) (West et al., 2001 and
- 4) EPA's 1994 survey of literature reports on chemical data from the Columbia River

Basin (USEPA 1994d)

The NCBP database includes data on persistent organochlorine insecticides, industrial chemicals, herbicides, and potentially toxic contaminants that may threaten fish and wildlife resources (Schmitt et al., 1999a). The NCBP database, from the early 1960's through 1986, contains measured values of average whole-body composite fish samples where each composite sample was comprised of five individual fish samples.

The BEST database includes data from the smallmouth bass sampled from the Mississippi River drainage during August-December 1995 (Schmitt et al., 1999b). Fish tissue data consisted of whole body composite samples, where, ideally, each composite sample consisted of 10 individual fish samples.

The PSAMP database consists of measured chemical concentrations in fillet (without skin) composites of adult chinook and coho salmon (West et al., 2001). Composite samples include 2-5 individual fish, with five individual fish per composite being the most common.

EPA's 1994 database includes a compilation of data from 1984 to 1994 on chemical concentrations in fish tissue and sediments from the Columbia River Basin. The information in the database includes individuals and agencies contacted, data sources, abstracts for contaminant studies, and an overview of future or ongoing studies (USEPA, 1994d).

The data from two surveys of chemicals in fish from the Columbia River Basin were also compared to fish tissue residues from our study:

- 1) The Lower Columbia River Bi-State Water Quality Program (Tetra Tech, 1996) and
- 2) Willamette River Human Health Technical Study (EVS, 2000)

The Lower Columbia River Bi-State Water Quality Program (Tetra Tech, 1996) characterized potential human health risks associated with consuming fish from the lower Columbia River, below the Bonneville Dam. The Bi-State study was conducted during two periods: 1991-1993 and 1995. Data from 1991-1993 consisted of data that measured chemical contaminant concentrations in fillet tissues of five different resident target fish species (largescale sucker, carp, peamouth, white sturgeon, and crayfish). Five individual fish were composited to form single composite samples. Data from 1995 included measured chemical concentrations in fillet fish tissue from largescale sucker, smallmouth bass, chinook salmon, and coho salmon. Fish tissue data for these species consists of range and mean data from three composite samples where each sample was made up of eight fish.

The Willamette River Human Health Technical Study (EVS, 2000) included data from four fish species of which smallmouth bass and largescale sucker were used for comparisons with our study. Data were compared for both fillet with skin and whole body tissue. All samples from the

Willamette study were composite samples formed by homogenizing tissue from five to eight individual fish.

9.2.1 Largescale Sucker (*Catostomus macrocheilus*) and Bridgelip Sucker (*C. columbianus*)

The largescale sucker is native to the Pacific Northwest in tributaries to the Pacific Ocean from the Skeena River in British Columbia to the Sixes River in Oregon (Scott and Crossman 1973). Largescale suckers are abundant throughout the Columbia River and are the most common resident fish species collected in the Hanford Reach (Gray and Dauble 1977).

Dauble (1986) found that algal periphyton was the major food item for fry, juvenile, and adult largescale suckers in the Columbia River. The stomachs of adults may also contain crustaceans, aquatic insect larvae, snails, fish eggs, sand, and bottom debris (Dauble 1986, Scott and Crossman 1973). Stream fish appear to feed upon more algae, diatoms, and aquatic insect larvae other than Chironomidae, whereas lake fish include Amphipoda and Mollusca (Carl 1936).

The bridgelip sucker is found in the Fraser and Columbia river basins from British Columbia to southeastern Oregon, including the Harney basin, below Shoshone Falls in the Snake River, and in northern Nevada (Scott and Crossman 1973, Lee et al. 1980). Throughout its range it coexists and hybridizes with the largescale sucker (*C. macrocheilus*) (Dauble and Buschbom 1981).

The life history and behavior of the bridgelip sucker are poorly understood. According to Scott and Crossman (1973), this fish usually inhabits small, swift, cold-water rivers with gravel to rocky substrates, whereas Wydoski and Whitney (1979) report it inhabits quiet backwater areas or the edges of the main current of rivers with sand or mud bottoms. In the Yakima River, Patten et al. (1970) found this fish in warm flowing waters. In the mid Columbia River during the day, Dauble (1980) found that subadult and adult bridgelip suckers were common in the tailouts of pools, at the end of riffles, and above boulders in the main current. At night, these fish were more abundant near shore in flowing water 0.6 to 1.5 m deep.

The diet of *C. columbianus* is almost entirely periphyton during all seasons. This fish has an expanded cartilaginous lower lip on its mouth that enables it to efficiently crop algae attached to the bottom. However, like almost all other suckers, this species also feeds to some extent on aquatic insect larvae and crustaceans (Dauble 1978, Wydoski and Whitney 1979). Mammals and some birds prey on this species (Scott and Crossman 1973).

Chemical concentrations in largescale sucker fish tissue were compared for arsenic, cadmium, copper, mercury, lead, selenium, zinc, p,p'-DDE, p,p'-DDT, Aroclor 1254, and Aroclor 1260. Data were compared in the NCBP databases and the Bi-State and Willamette River studies (Table 9-20a).

While the metal concentrations in largescale sucker from our study were within the range of the other studies and databases examined, the maximum concentrations of metals were higher or

lower depending on the chemical (Table 9-20a). Cadmium concentrations were 25 times higher in our study than in the Willamette River study and National NCBP database. Lead in largescale sucker from our study was 9 times higher than in largescale sucker from the NCBP National database.

The organic chemical comparisons in largescale sucker were also quite variable (Table 9-20a). With exception of the Aroclors the organic chemical concentrations in our study were all within the range of the other databases and studies. However, the maximum concentrations were different. The maximum concentration of p,pDDE in largescale sucker was 9 times higher in our study than in the Bi-State study, and 14 times higher than in the NCBP Columbia River station 98.

The maximum Aroclor 1254 concentrations in largescale sucker were higher in the Columbia River NCBP stations (from 8x to 46x) than in our study. The detection limits were too high in the National NCBP database to discern a difference in Aroclor 1254 and our study.

With the exception of cadmium, the Willamette River study results for metals and organic chemicals were similar to our study.

The concentrations of chemicals in bridgelip sucker were within the range found in largescale sucker, except the largescale sucker had higher maximum concentrations (Table 9-20a,b).

Table 9-20a. Comparison of chemical concentrations in composites samples of whole body largescale sucker.

Station	USGS- NCBP- Columbia River Basin				USGS- NCBP	Willamette	Bi-State		EPA	
	Columbia (46)	Columbia (47)	Columbia (98)	Snake (41,42,96)	National		Our study			
Chemical	range	range	range	range	range	single composite	mean	max	ave	range
	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$	$\mu\text{g/kg}$
Arsenic	<50 - 870	130 - 290	111 - 333	<50 - 260	40 - 270	120	8	385	160	74- 320
Cadmium	<50 - 160	<50 - 600	50 - 410	<50 - 260	<5 - 9	10	37	66	55	13-250
Copper	850 - 1340	1070 - 1283	720 - 1150	490- 4318	600 - 1010	1780	912	1230	1400	800-5600
Lead	90 - 390	100 - 520	160 - 2570	10 - 290	20 - 120	37	171	860	170	27-1100
Mercury	50 - 320	<10 - 160	20 - 130	10 - 230	10 - 370	121	122	264	130	<58-250
Selenium	60 - 430	60 - 386	190 - 250	170 - 450	80 - 340	ND	132	260	310	<180-500
p,p'-DDE	20 - 2000	20 - 1100	10-90	50 - 560	10 - 970	835	59	150	370	28-1300
p,p'-DDT	10 - 270	10 - 430	10-70	10 - 440	10 - 190	190	10	56	33	<1-180
Aroclor 1254	100 - 2100	5 - 3000	100 - 600	<5 - 500	<100	53	176	270	30	<14-65
Aroclor 1260	100 - 700	<5 - 100	100 - 300	<5 - 300	<100 - 300	36	35	1300	38	<12-100

Min= minimum; Max = maximum, Ave = average <= detection limit

NCBP = USGS National Contaminant Biomonitoring Program 1969-1986. Range of average whole body composites. Station numbers are in parentheses.

Willamette = composites without replication, EVS, 2000.

Bi-State = whole body concentrations of fish collected during 1991-1993 from the lower Columbia River, below Bonneville Dam. Mean and maximum (max) TetraTech, 1996

EPA- Our study = range of composite fish samples from sites in the Columbia River Basin. See table 1-1 and 1-2 for description of sites.

Table 9-20b . Comparison of ranges of chemical concentration in composite samples of whole body bridgelip sucker.

Station Chemical	USGS - NCBP- Columbia River Basin			NCBP	EPA
	Salmon (43) <i>µg/kg</i>	Snake (96) <i>µg/kg</i>	Columbia (98) <i>µg/kg</i>	National <i>µg/kg</i>	Our Study <i>µg/kg</i>
Arsenic	160 - 330	No Data	180 - 270	60	260 - 300
Cadmium	20 - 50	No Data	70 - 280	<50 - 60	22 - 32
Copper	680 - 1900	No Data	No Data	No Data	880 - 1800
Lead	100 - 220	No Data	530 - 1000	<100 - 110	37 - 78
Mercury	40 - 80	120	20 - 70	80 - 160	<40 - 53
Selenium	200 - 470	No Data	200 - 260	No Data	280
p,p''-DDE	10 - 30	340 - 440	<10 - 40	200 - 350	310 - 560
p,p''-DDT	<10 - 20	190 - 200	<10 - 40	180 - 380	37 - 52
PCB1254	<100	<100 - 500	<100	1000 - 2800	18 - 32
PCB1260	<100	<100	<100 - 4800	No Data	27 - 49

< = detection limit

NCBP = USGS National Contaminant Biomonitoring Program 1969-1986 Range of average whole body composites. Station numbers are in parentheses.

EPA- Our Study = range of composites from the Yakima River (study site 48).

9.2.2 Mountain Whitefish (*Prosopium williamsoni*)

The mountain whitefish is native to cold water rivers and lakes in western North America, both east and west of the Continental Divide (Scott and Crossman 1973). Seven-year old fish range in length and weight from 307 to 387 mm and from 475 to 890 g, respectively, while the ranges for 8-year old fish are 330 to 410 mm and 501 to 944 g (Scott 1960, Pettit and Wallace 1975, Thompson and Davies 1976). Mountain whitefish feed primarily on immature forms of bottom-dwelling aquatic insects such as Diptera (true flies and midges), Trichoptera (caddisflies), Ephemeroptera (mayflies), and Plecoptera (stoneflies) (Wydoski and Whitney 1979, Cirone et al. 2002).

The ranges of chemical concentrations in the whole body mountain whitefish, from the present study were compared with mountain whitefish data from the NCBP database (Table 9-21). There was no consistent pattern between the metal concentrations in our study of mountain whitefish and NCBP database (Table 9-21). The maximum arsenic and cadmium levels were similar in our study and the NCBP database. The maximum copper concentrations in mountain whitefish in our study were 6 to 9 times higher than the concentrations in the NCBP database. Lead concentrations were higher in the NCBP database. The maximum mercury levels measured in the Salmon River in NCBP database were higher than the levels measured in our study; the levels in the NCBP Snake River mountain whitefish were lower. The maximum selenium concentrations were lower in the NCBP database than in our study.

The maximum p,p' DDE concentrations in mountain whitefish in our study were 700 times higher than the concentrations in mountain whitefish from the NCBP Salmon River station. The Aroclor concentrations were not comparable because of the higher detection limits in the NCBP database.

Table 9-21. Comparison of ranges chemical concentrations in composite samples of whole body mountain whitefish.

Station	USGS -NCBP - Columbia River Basin			EPA
	Salmon (43)	Snake (96)	Columbia (97)	Our Study
Chemical	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$
Arsenic	120	No data	No data	120 - 180
Cadmium	40	No data	No data	<4 - 54
Copper	840	590	No data	620 - 5000
Lead	100	103	No data	10 - 72
Mercury	290	65	190	<47 - 130
Selenium	680	472	No data	590 - 1800
p,p'-DDE	<10	590	1410	13 - 770
p,p'-DDT	20	30	350	<2 - 49
Aroclor 1254	<100	100	<100	<21 - 140
Aroclor 1260	<100	100	100	<18 - 130

<= detection limit

NCBP = USGS National Contaminant Biomonitoring Program 1969-1986. Range of average whole body composites. Station numbers are in parentheses.

EPA- Our Study = range of composite fish samples from sites in the Columbia River Basin. See table 1-1 and 1-2 for description of sites

9.2.3 White Sturgeon (*Acipenser transmontanus*)

White sturgeon is native to the Pacific Northwest where it has evolved life history characteristics that have allowed them to thrive for centuries in large, dynamic river systems containing diverse habitats. These characteristics include opportunistic food habits, delayed maturation, longevity, high fecundity, and mobility (Beamesderfer and Farr 1997). White sturgeon may attain lengths and weights of more than 6 m and 580 kg, respectively, during a life span of over 100 years (Scott and Crossman 1973). White sturgeon body weight ranged from 9 to 34 kg.

White sturgeon take advantage of scattered and seasonal food sources by moving between different riverine habitats. They feed on a wide range of food items including zooplankton, molluscs, amphipods, aquatic larvae, benthic invertebrates, and fish (McCabe et al. 1993). White sturgeon are more predaceous than any other North American sturgeon (Semakula and Larkin 1968) and can capture and consume large prey (Beamesderfer and Farr 1997). Seasonal migrations occur in the Lower Columbia River where sturgeon move to feed on eulachon (*Thaleichthys pacificus*), northern anchovy (*Engraulis mordax*), American shad (*Alosa sapidissima*), moribund salmonids, amphipods, and other invertebrates (DeVore et al. 1995).

Concentrations of the Aroclors and 2,3,7,8-TCDF and in white sturgeon from our study of the Columbia River Basin were higher than the EPA 1994 (USEPA, 1998c) studies of Lake Roosevelt, Washington (Tables 9-9 and 9-10).

9.2.4 Walleye (*Stizostedion vitreum*)

The original range of the walleye generally east of the Rocky Mountains was expanded when it was introduced to the Columbia River below Roosevelt Dam in the 1940's or 50's (Wydoski and Whitney 1979). This species shows a preference for large, semi-turbid waters, but is capable of inhabiting a large range of physical and chemical conditions (Colby et al. 1979).

Feeding usually occurs near or at the bottom, and walleye may move into shallow water to feed. Walleye fry feed on rotifers, copepods, and cladocerans. Juvenile and adult walleye are largely piscivorous, but invertebrates (e.g., mayfly nymphs and amphipods) may be a large part of their diet in the late spring and early summer. Cannibalism is common with this species (Colby et al. 1979, Eschmeyer 1950). Prey for this species in the Columbia River includes mainly cottids, cyprinids, catostomids, and percopsids; out migrating juvenile salmonids were a smaller part of their diet (Zimmerman 1999).

Adult walleye are not usually preyed upon by other fish. However, in its native range northern pike and muskellunge do prey on this fish (Colby et al. 1979). They are also probably preyed upon by fish eating birds and mammals (Sigler and Sigler 1987).

The maximum concentration of Aroclors 1254 and 1260 and 2,3,7,8-TCDF in walleye were lower in our study of the Columbia River Basin than levels found in surveys of Lake Roosevelt, Washington, (USEPA, 1998c; Munn, 2000) (Tables 9-9 and 9-10).

9.2.5 Channel catfish (*Ictalurus punctatus*)

The original range of the channel catfish, east of the Rock Mountains was expanded when it was introduced to Idaho waters in 1893, but the date of its introduction to Washington waters is unknown (Wydoski and Whitney 1979, Simpson and Wallace 1982).

Young channel catfish tend to feed primarily on aquatic insects and bottom arthropods, but after attaining about 100 mm in length they are usually omnivorous or piscivorous (Carlander 1969). Adult channel catfish consume a wide variety of plant and animal material including clams, snails, crayfish, pondweed, and small terrestrial vertebrates (Eddy and Underhill 1976, Moyle 1976).

Young channel catfish are prey to a variety of fishes and piscivorous birds but the adults, due to their size and bottom occurrence, are probably free of predation (Scott and Crossman 1973, Schramm et al. 1984).

The concentrations of chemicals measured in channel catfish our study were compared to levels reported in the NCBP database (Table 9-22). The concentrations of metals were higher in the National and Columbia Basin NCBP databases with two exceptions. The maximum concentrations of arsenic and selenium concentrations in channel catfish were 10 times higher in our study than the NCBP Willamette station. The concentrations of the following metals were higher in the NCBP national database: cadmium 29x, lead 60x, mercury 14x, and selenium 4 times higher.

The concentrations of organic chemicals were higher in the NCBP National database than in our study. The maximum concentrations of the following chemicals in channel catfish from the National NCBP database were higher than the levels in channel catfish in our study: p,p'DDE 47x, p,p'DDT 166x, Aroclor 1260 672x, and Aroclor 1260 42 times higher. The concentrations

of p,p' DDT in the NCBP Columbia Basin stations were 5 - 23 times higher than in our study. The maximum concentrations of Aroclor 1254 in channel catfish was from the NCBP Columbia Basin Stations were 24 to 76 times higher than in our study.

Table 9-22. Comparison of ranges of chemical concentrations in whole body channel catfish tissue from our study with the USGS-NCBP database.

Station	USGS - NCBP			EPA	
	Willamette (45)	Snake (96)	National	Our Study	
Chemical	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$	ave	$\mu\text{g}/\text{kg}$
Arsenic	<50	<50 - 610	10 - 630	230	110 - 430
Cadmium	<50	<50	3 - 760	17	13 - 26
copper	no data	no data	no data	510	410 - 590
Lead	100	<100 - 210	30 - 2000	21	12 - 33
Mercury	290	80 - 900	<10 - 4500	210	140 - 320
Selenium	60	70 - 180	<50 - 2500	500	410 - 630
p,p'-DDE	570	<10 - 1050	10 - 42300	570	280 - 900
p,p'-DDT	<10 - 1050	<10 - 220	<5 - 7500	21	0.8 - 45
Aroclor 1254	4400	<10 - 1400	<50 - 39000	38	25 - 58
Aroclor 1260	No Data	<100 - 500	<50 - 5900	77	32 - 140

*Samples are fillet with skin;

Ave= average

NCBP = USGS National Contaminant Biomonitoring Program 1969-1986. Range of average whole body composites. Station numbers are in parentheses.

EPA-Our Study = whole body composite samples from the Columbia River (study site 8) and the Yakima River (study site 8)

9.2.6 Smallmouth Bass (*Micropterus dolomieu*)

The range of the smallmouth bass, originally restricted to freshwaters of eastern-central North American, was expanded by plantings in the Pacific Northwest in the late 1800s and early 1900s. In Washington, smallmouth bass are most numerous in the Columbia and Snake rivers (Wydoski and Whitney 1979, Simpson and Wallace 1982).

Smallmouth bass fry initially eat copepods and cladocerans and at lengths of 2 to 5 cm change to a diet of insects and small fish (Hubbs and Bailey, 1938). Tabor et al. (1993) found that salmonids made up from 4 to 59% (by weight) and from 19 to 30% (by volume) of the diet of smallmouth bass in the Columbia River Basin. The authors concluded that predation rates on salmonids were high during the spring and early summer when subyearling salmon were abundant and of suitable forage size and shared habitat with the smallmouth bass.

Smallmouth bass in the Columbia River grow at a rate equal to or better than that of bass from other locations in the United States. In a 1952 study, the weights and total lengths of the Columbia River fish at age four were 510 g and 32 cm; age six, 794 g and 38 cm; age eight, 1,304 g and 43 cm; and at age ten, 1,814 g and 47 cm, respectively (Henderson and Foster 1957, Wydoski and Whitney 1979). The body weight of smallmouth bass in our study ranged from 1300 to 1400 g.

Smallmouth bass from our study were compared to data reported in the BEST and NCBP databases (Table 9-23). The concentrations of all chemicals in smallmouth bass from the NCBP National database were higher than in our study. In particular, Aroclor 1254 was higher (68x) in

the NCBP National database. The Aroclor concentrations in Columbia River Basin NCBP stations had higher detection limits than in our study.

Table 9-23. Comparison of ranges of chemical concentrations in whole body smallmouth bass.

Chemical	USGS- NCBP					USGS	EPA
	Yakima (44)	Snake (42)	Salmon (43)	Willamette(45)	National	BEST	Our Study
Chemical	$\mu\text{g}/\text{kg}$						
Arsenic	No data	50 - 60	<30 - 50	250	40 - 670	<178 - 263	160 - 170
Cadmium	No data	10 - 50	6 - 60	50	2 - 50	<36 - 43	5 - 19
Copper	No data	380	1182	No data	257 - 1950	445 - 591	500 - 560
Lead	No data	<100	100 - 170	120	10 - 320	8 - 100	10 - 140
Mercury	140 - 270	150 - 280	210 - 360	130	60 - 1200	80 - 280	220 - 360
Selenium	No data	440	606 - 830	No data	80 - 1260	203 - 491	480 - 710
p,p'-DDE	940 - 1660	80 - 2540	280 - 690	60	10 - 950	10 - 65	970 - 1700
p,p'-DDT	200 - 420	80 - 170	80 - 170	20	<5 - 590	10 - 84	44 - 80
Aroclor 1254	100 - 600	<100	<50 - 400	<400	<50 - 6400	No data	46 - 94
Aroclor 1260	200	<100 - 800	<50 - 100	<200	<50 - 1300	No data	80 - 190

NCBP = USGS National Contaminant Biomonitoring Program 1969-1986. Range of average whole body composites. Station numbers are in parentheses.

BEST = USGS Biomonitoring of Environmental Status and Trends Program - 1995 Fish Samples from the Mississippi Delta.

EPA- Our Study = whole body composite samples from the Yakima River (study site 48)

9.2.7 Rainbow and Steelhead (*Oncorhynchus mykiss*)

Oncorhynchus mykiss are native to the Pacific Northwest and appear in two forms: the resident rainbow trout and the anadromous steelhead, both of which occur in the Columbia River Basin. It also has the greatest diversity of life history patterns of any Pacific salmonid species (Wydoski and Whitney 1979, Pauley et al. 1986). This diversity includes degrees of anadromy, differences in reproductive biology, and plasticity of life history between generations (Peven 1990, Busby et al. 1996).

The diet of rainbow trout and juvenile steelhead changes seasonally, depending on food availability. They may feed on aquatic insects, amphipods, leaches, snails, and fish eggs. The steelhead's diet in the ocean includes crustaceans, squid, herring, and other fish (Withler, 1966; Wydoski and Whitney, 1979). Adult non-migratory rainbow trout average 0.9 to 1.8 kg in weight and usually have a life span of 5 to 6 years (Simpson and Wallace, 1982; Sigler and Sigler, 1987). Steelhead can achieve 9 years of age, weights of 16 kg, and lengths to 122 cm (Scott and Crossman, 1973; Wydoski, and Whitney, 1979). The average body weight of rainbow trout in our study ranged from 47 - 571g. The steelhead average body weight ranged from 1633 to 6440g.

The chemical residues in rainbow trout measured in our study were compared to the NCBP databases (Table 9-24). The maximum concentration of p,p' DDE in rainbow trout was 300 times higher in the NCBP Columbia River Basin station (Snake River) than in our study.

Steelhead concentrations of metals in fish tissue were within the range of rainbow trout (Table 9-24). The maximum concentrations of arsenic and lead were higher (4x and 2x respectively) in the steelhead, while p,p'DDE was lower in the steelhead than the rainbow trout.

Table 9-24. Comparison of ranges of chemical concentrations in composite samples of whole body rainbow trout.

Station Chemical	USGS - NCBP		EPA (Our Study)	
	Snake (41) µg/kg	National µg/kg	rainbow trout µg/kg	steelhead
Arsenic	<50 - 145	<50 - 260	<50 - 560	290 - 1200
Cadmium	5 - 50	10 - 70	<4 - 58	29 - 88
Copper	680 - 3130	1130 - 4620	900 - 5000	1900 - 6800
Lead	9 - 100	10 - 650	<10 - 88	<10 - 360
Mercury	30 - 130	10 - 270	<33 - 380	<50 - 420
Selenium	220 - 540	170 - 3000	230 - 790	460 - 940
p,p'-DDE	80 - 25400	10 - 140	3 - 84	5 - 33
p,p'-DDT	5 - 70	5 - 40	<2 - 12	<1 - 6
Aroclor 1254	100 - 600	<50 - 300	<10 - 20	9 - 29
Aroclor 1260	<50	<50 - 100	<6 - 22	<6 - 21

NCBP = USGS National Contaminant Biomonitoring Program 1969-1986. Range of average whole body composites. Station numbers are in parentheses.

EPA- Our study = range of composite fish samples from sites in the Columbia River Basin. See table 1-1 and 1-2 for description of sites.

9.2.8 Chinook Salmon (*Oncorhynchus tshawytscha*)

Chinook salmon are the largest of the Pacific salmon and have a variable life history. Timing of migration and spawning, and the duration of freshwater, estuarine, and ocean residencies varies for this species (Meehan and Bjornn 1991). 'Stream-type' and 'ocean-type' chinook are the two main races. Stream-type chinook are also referred to as spring or summer chinook salmon, and ocean-type as fall chinook salmon. Most (78%) of the chinook salmon in the Columbia River are ocean-type and they spawn from mid-September to late December. Ocean-type juveniles migrate to the estuary at 3 to 6 months of age when they are 70 to 90 mm in length (Meehan and Bjornn 1991). In the estuary, these juveniles prefer low banks and subtidal refuge areas and their diet consists of insect and crab larvae and small fish (Healey 1991). Stream-type juveniles overwinter in freshwater before out migrating as yearlings from April to June. Some will spend two winters in freshwater. Deep pools with rock crevices provide over wintering habitat. In freshwater, juvenile diet is primarily insects, both aquatic larvae and terrestrial adults. During outmigration, yearling smolts spend a brief period in the estuary where they occupy the outer part of the estuary, thus, their habitat does not overlap with the smaller ocean type chinook (Healey 1991).

Chemical concentrations of metals and organic chemicals measured in fall chinook salmon from our study of the Columbia River Basin were compared to fall chinook salmon measurements in PSAMP database and the Bi-State study (Table 9-25).

The concentration of arsenic in chinook salmon was similar in our study, PSAMP, and the EPA 1994 database, while the Bi-State arsenic concentrations were lower (48x for fall chinook salmon; 52x for spring chinook salmon). The cadmium levels in chinook salmon were higher (13x fall chinook salmon; 3x spring chinook salmon) in the EPA 1994 database than our study. The maximum lead concentrations were higher in the spring chinook salmon in our study than in the Bi-State study (14x). Fall chinook and spring chinook salmon from our study had higher concentrations of Aroclor 1254 than the Bi-State study (35x and 24x, respectively).

The chemical concentrations in fall and spring chinook salmon from our study were similar to each other with the exception of cadmium, lead, and mercury which were higher in spring chinook (15x, 8x, and 5x, respectively; Table 9-25).

Table 9-25. Comparison of chemical concentrations in chinook salmon fillet with skin.

Station	EPA 1994		EPA					
	Database	PSAMP	Bi-State		Our Study			
	range µg/kg	range µg/kg	ave µg/kg	max µg/kg	fall chinook salmon		spring chinook salmon	
Chemical					ave µg/kg	range µg/kg	ave µg/kg	range µg/kg
Arsenic	20 - 1110	570 - 1600	13	23	810	530 - 1100	850	560 - 1200
Cadmium	20 - 50	No data	2	2.5	<2	<4	2	<4 - 15
Copper	240 - 1900	370 - 1200	860	1010	640	540 - 760	790	240 - 1000
Lead	20 - 40	no data	7	10	7	<10 - 16	14	<10 - 140
Mercury	62 - 164	58 - 160	100	130	84	<50 - 150	100	<83 - 510
Selenium	360 - 370	no data	280	340	330	280 - 380	350	290 - 430
p,p'-DDE	no data	4 - 48	8.5	11	12	4 - 26	12	6 - 18
p,p'-DDT	3	0.5 - 4	1.5	3	2.5	<2 - 8	4	3 - 8
Aroclor 1254	18 - 20	5 - 88	0.9	0.9	17	9 - 35	16	9 - 24
Aroclor 1260	16 - 30	1 - 72	10	15	9.9	<19	11	<18
2,3,7,8-TCDD	0.00014	no data	0.0002	0.0006	0.00002	<0.00001-0.00005	0.00002	<0.00001-0.00005
2,3,7,8-TCDF	0.0009	no data	0.0016	0.00027	0.00068	<0.00003-0.0014	0.0006	0.0004-0.00074

Ave = average; max = maximum <= detection limit

EPA 1994 database = EPA survey of data from the Columbia River Basin from 1983-1994. Does not differentiate between spring and fall chinook salmon

Bi-State = 1995 concentrations in fillets of fish from the lower Columbia River, below Bonneville Dam. Does not differentiate between fall and spring chinook salmon (Tetra Tech, 1996).

PSAMP = 1992-1995, data is for fillet without skin. Does not differentiate between fall and spring chinook salmon

EPA- Our study = range of composite fish samples from sites in the Columbia River Basin. See table 1-1 and 1-2 for description of sites

9.2.9 Coho Salmon (*Oncorhynchus kisutch*)

Coho salmon are one of the five Pacific salmon species in North America. The life span of most coho is three years, during which they attain average weights ranging from about 3,000 to 6,000g (Wydoski and Whitney 1979). The average body weight of the coho salmon in our study was 2,855g to 3,960g.

The coho salmon fish typically spend up to 21 months in freshwater followed by approximately 16 months in the ocean before returning to freshwater where they will spawn and die. These fish rarely feed on non-moving food or off the bottom in streams (Sandercock 1991). Juveniles consume insects (larvae, pupae, and adults), worms, small fish, and fish eggs. In reservoirs, coho juveniles feed primarily on zooplankton and emerging insects (Wydoski and Whitney 1979).

Samples of coho salmon from our study were compared to data from PSAMP and the Bi-State study (Table 9-26). The maximum concentrations of several chemicals were higher in coho salmon from our study than the coho salmon from the Bi-State study: arsenic (85x), lead (25x), and Aroclor 1254 (19x).

Table 9-26. Comparison of chemical concentrations in coho salmon fillet with skin.

Station	PSAMP	Bi-State		EPA - Our study	
	range	mean	max	ave	range
Chemical	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$	$\mu\text{g}/\text{kg}$
Arsenic	570 - 1600	2.7	7	540	450 - 600
Cadmium	No data	3	5		<4
Copper	410 - 1010	810	850	1700	680 - 3600
Lead	No data	4	9	81	<10 - 230
Mercury	58 - 160	44	48	120	110 - 120
Selenium	No data	168	188	290	270 - 310
p,p'-DDE	1.3 - 26	3	5	33	29 - 35
p,p'-DDT	0.52 - 1.4	0.8	1	2	<2 - 4
Aroclor 1254	2 - 66	0.6	0.9	16	12 - 19
Aroclor 1260	1 - 32	3	4		<18
2,3,7,8-TCDD	No data	0.0003	0.0009	0.000017	<0.00001 - 0.00004
2,3,7,8-TCDF	No data	0.0007	0.0009	0.0005	0.0004 - 0.0005

Ave = average; max = maximum; < = detection limit

PSAMP = 1992-1995, data is for fillet without skin

Bi-State = 1995 whole body concentrations of fish from the lower Columbia River, below Bonneville Dam. (TetraTech, 1996)

EPA - Our study = range of composite fish samples from sites in the Columbia River Basin. See table 1-1 for site descriptions.

9.2.10 Pacific Lamprey (*Lampetra tridentata*)

The Pacific lamprey is a native anadromous fish with a widespread distribution in the Columbia River Basin (Wydoski and Whitney 1979).

The adults overwinter in freshwater, do not feed during this time, and spawn the following spring (Beamish 1980). Larvae (ammocoetes) leave the gravel approximately 2 to 3 weeks after hatching, drift down current, settle in slow back water areas, burrow in soft substrates with organic debris, and take up a filter feeding existence (Pletcher 1963, Kan 1975). The ammocoete life stage may range from 4 to 7 years, during which time they remain buried in the sediment (Beamish and Levings 1991, Close et al. 1995). Ammocoetes are reported to feed on vegetative material (Clemens and Wilby 1967), diatoms and desmids (Pletcher 1963), and detritus and algae suspended above and within the substrate (Moore and Mallatt 1980). Juvenile lampreys play an important role in the diets of many freshwater fishes, including channel catfish, northern pike minnow, and several species of cyprinids and cottids. Salmonid fry prey upon lamprey eggs, but do not feed on the ammocoetes. The larvae are also taken by several species of gulls and terns (Pletcher 1963, Close et al. 1995).

Metamorphosis occurs from July to October. Shortly thereafter, the downstream migration of young adult lampreys begins usually at night and with an abrupt increase in river flow. Pacific lampreys migrate to salt water where they take up a parasitic life, but feeding may start in freshwater (Pletcher 1963, Beamish 1980, Beamish and Levings 1991).

The ocean phase of the adult life cycle may last 3.5 years (Beamish 1980). In ocean and estuarine areas, adults are important prey for several pinniped species. After entering the Columbia River they become a prey item for white sturgeon (Wydoski and Whitney 1979, Roffe and Mate 1984, Close et al. 1995).

There were no comparable studies of Pacific lamprey in the literature.

9.2.11 Eulachon (*Thaleichthys pacificus*)

The eulachon occurs only on the west coast of North America, including the Columbia River Basin (Scott and Crossman 1973). This anadromous species spawns in the main channel of the Columbia River and periodically in the Grays, Cowlitz, Kalama, Lewis, and Sandy Rivers (Smith and Saafeld 1955).

It is believed that developing larvae do not feed in freshwater, but rely on their yolk sac for nourishment until they reach the ocean (Smith and Sallfeld 1955, Scott and Crossman 1973). At sea, post-larval eulachon move into deeper water as they grow. They feed on plankton, mysids, ostracods, copepods and their eggs, and barnacle, cladoceran, and polychaete larvae (Hart 1973). Juvenile and adult fish feed primarily on euphausiid shrimp, crustaceans, and cumaceans. Adults do not feed after they return to freshwater (Barraclough 1964).

As are other smelts, *T. pacificus* is a very important food item for a wide variety of predators. Adults are fed on by many piscivorous fishes including Pacific salmon and white sturgeon, marine mammals ranging from the harbor seal to the finback whale, seabirds, waterfowls, and gulls (Scott and Crossman 1973). The larval and post larval stages contribute modestly to the diet of small salmon off the Fraser River (Hart 1973).

There were no comparable studies of eulachon in the literature.

9.3 Comparisons across all species

9.3.1 Resident Fish

White sturgeon, mountain whitefish, whole body walleye, largescale sucker, smallmouth bass, and channel catfish had the highest concentrations of organic chemicals of all the species tested in this study (Table 9-27a,b). Bridgelip sucker and walleye fillet samples had much lower chemical residues, similar to the salmonids and eulachon.

The largescale sucker was the fish species with the most frequent detection of PAHs (Table 2-1a). The phenols were detected in only one white sturgeon sample from the main-stem Columbia River (study site 8) (Table 2-1a).

The basin-wide average concentrations of total DDT (Table 2-4) in the salmonids (chinook, coho, rainbow trout, and steelhead) and eulachon were much lower than, white sturgeon, mountain whitefish, largescale sucker, and smallmouth bass. The maximum concentrations p,p'DDE was found in whole body smallmouth bass followed by white sturgeon fillet, channel catfish fillet, and whole body largescale sucker (Table 9-27a).

The white sturgeon, mountain whitefish, whole body walleye, and smallmouth bass had the

highest concentrations of Aroclors. The maximum concentration of TCDF was in the white sturgeon (Table 9-27a,b). The next highest average concentration was in the mountain whitefish.

The maximum concentrations of metals (arsenic, cadmium, copper, lead, mercury, selenium) were lower in the resident species than in the anadromous species, except for largescale sucker which had the highest concentration of cadmium (Table 9-27a,b). When doing a comparison of fish tissue across all species it is important to not only consider the maximum concentrations but also some measure of the variability. In this study, the average concentration is a measure of variability. While the maximum mercury and selenium concentrations were in the spring chinook salmon, the basin-wide average concentrations of mercury were highest in the largescale sucker, walleye, and white sturgeon.

The higher concentration of organic chemicals may be attributed to size in some species or lipid content. The white sturgeon were some of the largest fish measured in the study. The samples included only single fish. It is also known to have a very long life span. Thus, it is not clear whether the high levels of organic chemicals in this fish may be due to an anomaly in the few fish that were sampled, their size, or their age.

The association of organic chemical concentrations in the tissues of resident species and percent lipid was not particularly evident in this study. There was an association with lipid in the white sturgeon samples from one study site (study site 6). The difference in chemical content between the whole body walleye and the fillet was also associated with lipid. However, there were no other clear associations of whole body and fillet with lipid and organic chemicals in fish tissue.

There was an indication of high concentrations of organic chemicals in the resident fish collected from the Hanford Reach of the Columbia River (study site 9U). However, there is no information in this study to explain the levels in fish from this study site.

9.3.2 Pacific lamprey and eulachon

Of the anadromous fish species, Pacific lamprey had maximum concentration of organic chemicals (DDE and Aroclor 1254; Table 9-27b). The high concentration of organic chemicals in the Pacific lamprey may have been due to its high lipid content.

The metals content of the Pacific lamprey was not consistent across different metals. For example when compared to the other anadromous species, the arsenic concentrations were low for Pacific lamprey while concentrations of copper, lead, mercury, and selenium were within the range of the range of these other fish species.

While eulachon also had a high lipid content, they had some of the lowest levels of organic chemicals of all the species test. Aroclors and chlordane were not detected in the eulachon. Eulachon had the highest average concentration of arsenic and lead.

9.3.3 Salmonids

The salmonids had the lowest concentrations of organic chemicals with a few exceptions. There were no semi-volatile chemicals detected in the fall chinook salmon or coho salmon tissue samples. Pyrene was found at the highest concentrations of all the PAHs in a rainbow trout collected from the upper Yakima River (study site 49). The fillet or whole body samples of rainbow trout, eulachon, and coho salmon had no detectable concentrations of any of the chlordanes compounds.

The concentrations of metals in the chinook salmon and steelhead were higher than the other resident or anadromous fish species. Steelhead had the maximum concentration of arsenic. When doing a comparison of fish tissue across all species it is important to not only consider the maximum concentrations but also some measure of the variability. In this study, the average concentration is a measure of variability. Thus, while steelhead had the maximum concentration of arsenic, the average concentrations were higher in eulachon, and chinook salmon (Table 2-14). From this study, the salmon, steelhead, and eulachon had higher concentrations of arsenic than the resident species and Pacific lamprey. Fall chinook salmon had the maximum concentration of lead (Table 9-27b). The average concentrations of lead were highest in eulachon, fall chinook salmon, and whole body walleye (Table 2-14).

Although the egg samples from the salmon and steelhead had high percent lipid, the concentration of organic compounds was generally lower than the fish tissue of the anadromous or resident fish with a few exceptions. The highest concentrations of total chlordanes were in egg samples from the spring chinook salmon. The maximum concentrations of copper and selenium were in egg samples from the salmon and steelhead (Table 9-27b). The basin-wide average concentrations of copper were highest in the egg samples from the salmon and steelhead followed by the whole body Pacific lamprey. The basin-wide average concentrations for selenium were highest in spring chinook salmon egg samples followed by white sturgeon and mountain whitefish. The high concentration of selenium may also be associated with the high percent lipid in the egg samples.

Table 9-27a. Range of chemical concentrations in resident fish tissue samples from our study of the Columbia River Basin, 1996-1998.

Chemical	T	largescale	Bridgelip	rainbow	mountain	white	walleye	channel	smallmouth
		sucker	sucker	trout	whitefish	sturgeon**		catfish	bass
		µg/kg	µg/kg	µg/kg	µg/kg	µg/kg	µg/kg	µg/kg	µg/kg
<i>N-FS</i>		19		7	12	16	3	5	
<i>N-WB</i>		23	3	12	12	8	3	6	
Arsenic	FS	50 - 100	NS	<50	51 - 140	150 - 640	290 - 400	50 - 330	110 - 170
	WB	74 - 320	260 - 300	<50 - 560	120 - 180	<200 - 640	480 - 510	110 - 430	160 - 170
Cadmium	FS	<4 - 24*	NS	<4 - 5*	<4 - 14*	<4 - 6*	<4	ND	ND
	WB	13 - 250	22 - 32	<4 - 58	4 - 54	15 - 95	100 - 110	13 - 26	5 - 19
Copper	FS	430 - 870	NS	440 - 610	510 - 840	<210 - 410	500 - 600	310 - 360	510 - 560
	WB	800 - 5600	880 - 1800	900 - 5000	620 - 5000	260 - 1800	730 - 5700	410 - 590	500 - 560
Lead	FS	10 - 140	NS	<10	<10 - 26	<10 - 29*	<10	10 - 11*	10 - 55
	WB	27 - 1100	37 - 78	<10 - 88	10 - 72	27 - 330	<10 - 490	12 - 33	10 - 140
Mercury	FS	71 - 370	NS	45 - 150	<49 - 140	38 - 430	160 - 200	240 - 280	380 - 470
	WB	<58 - 250	40 - 53	<33 - 380	<47 - 130	73 - 250	120 - 220	140 - 320	220 - 360
Selenium	FS	130 - 400	NS	180 - 250	300 - 720	310 - 2700	380 - 400	240 - 500	450 - 530
	WB	<180 - 500	<280	230 - 790	590 - 1800	<420 - 1100	410 - 540	410 - 630	480 - 710
p,p'-DDE	FS	14 - 740	NS	4 - 54	8 - 910	100 - 1400	44 - 52	330 - 1300	480 - 1200
	WB	28 - 1300	310 - 560	3 - 84	13 - 770	400 - 1100	350 - 440	280 - 900	970 - 1700
p,p'-DDT	FS	<2 - 92*	NS	<2 - 5*	<2 - 58	2 - 31	<2 - 3	2 - 87	23 - 48
	WB	<1 - 180	37 - 52	<2 - 12*	<2 - 49	<4 - 38	7 - 12	0.8 - 45	44 - 80
Aroclor 1254	FS	10-46	NS	10 - 20	<16 - 930	10 - 190	12 - 14	29 - 69	38 - 83
	WB	<14 - 65	18 - 32	<7 - 30	<21 - 140	38 - 120	54 - 98	25 - 58	46 - 94
Aroclor 1260	FS	<11 - 75	NS	<18	<9 - 190	<13 - 200	<19	37 - 130	68 - 220
	WB	<12 - 100	27 - 49	<6 - 22*	<18 - 130	41 - 160	47 - 61	32 - 140	80 - 190
2,3,7,8-TCDD	FS	<0.00001 - 0.00007	NS	<0.0000 - 0.00015	<0.00001 - 0.00021	0.0001 - 0.0014	0.00007 - 0.00008	0.001 - 0.0014	NA
	WB	<0.00001-0.00021	0.00006-0.00008	<0.00001 - 0.0002	<0.00001 - 0.00023	0.00006 - 0.0013	0.00036 - 0.00042	0.0010 - 0.0014	NA
2,3,7,8-TCDF	FS	0.0001 - 0.0015	NS	0.00014 - 0.00028	0.00014 - 0.014	0.0025 - 0.054	0.0006 - 0.00075	0.0022 - 0.0034	NA
	WB	0.0008 - 0.0036	0.0008 - 0.001	<0.0004 - 0.00048	0.0002 - 0.012	0.008 - 0.047	0.0038 - 0.0055	0.0022 - 0.0034	NA

N=number of samples; FS- Fillet with Skin; WB = whole body;E=egg; NA = not analyzed; < detection limit; * detection frequency was less than 50% of the samples

**whitesturgeon were single fish and fillets without skin.

Table 9-27b. Range of chemical concentrations (µg/kg) in anadromous fish tissue samples from our study of the Columbia River Basin.

	T	steelhead	fall chinook salmon	spring chinook	coho salmon	eulachon	Pacific lamprey
<i>N-Egg</i>		1	1	6	3		
<i>N-FS</i>		21	15	24	3		3
<i>N-WB</i>		21	15	24	3	3	9
Arsenic	E	ND	240	<410 - 510	310 - 360		
	FS	280 - 1500	530 - 1100	560 - 1200	450 - 600	NS	280 - 360
	WB	290 - 1200	610 - 1000	570 - 1100	450 - 560	860 - 930	150 - 370
Cadmium	E	34	<4	22 - 72	<4		
	FS	<4 - 9	<4	<4 - 15	<4	NS	16 - 30
	WB	29 - 88	5 - 10	6 - 170	19 - 27	9 - 10	56 - 150
Copper	E	18,000	5800	5300 - 6600	4100 - 5000		
	FS	540 - 940	540 - 760	240 - 1000	680 - 3600	NS	1100 - 1400
	WB	1900 - 6800	1000 - 14000	1100 - 2300	720 - 2400	920 - 970	3700 - 5500
Lead	E	41	<10	<10 - 50*	<10		
	FS	<10 - 23*	<11 - 16	<10 - 140	<10 - 230	NS	<10
	WB	<10 - 360	11 - 1200	<10 - 92	11 - 20	370 - 680	<10 - 69*
Mercury	E	<43	<50	<79	<100		
	FS	70 - 210	<50 - 150	<83 - 510*	110 - 120	NS	<110
	WB	<50 - 420	<50 - 200	<71 - 130*	11 - 20	<35	<91 - 210
Selenium	E	4500	2400	3700 - 5500	1100 - 1300		
	FS	<250 - 500	280 - 380	290 - 430	270 - 310	NS	410 - 450
	WB	460 - 940	<380 - 570	360 - 680	330 - 420	270 - 300	520 - 760
p,p'-DDE	E	7	7	10 - 16	31 - 33		
	FS	5 - 28	4 - 26	6 - 18	29 - 35	NS	46 - 55
	WB	5 - 33	5 - 53	11 - 22	31 - 37	10 - 11	35 - 77
p,p'-DDT	E	<2	<2	4 - 7	<2		
	FS	<1 - 5	<2 - 8	<2 - 7	<2 - 4	NS	28 - 38
	WB	<1 - 6	<2 - 7	3 - 8	<2 - 4	<4	6 - 29
Aroclor 1254	E	15	12	15 - 20	11 - 17		
	FS	8 - 21	9 - 35	9 - 24	12 - 19	NS	80 - 100
	WB	9 - 29	10 - 47	13 - 26	18 - 19	<37	60 - 150
Aroclor 1260	E	<20	<19	<18	<18		
	FS	<6 - 21*	<19	<18	<18	NS	<19
	WB	<6 - 21*	<19	<18	<18	<37	<13 - 20*
2,3,7,8-TCDD	E	<0.00003	<0.00004	<0.00001 - 0.00004	<0.00001-0.00005		
	FS	<0.00001 - 0.00008	<0.00001 - 0.00005	<0.00001-0.00005	<0.00001-0.00004		0.00001-0.00006
	WB	<0.00001-0.00006	<0.0000 - 0.00006	<0.00001 - 0.0001	<0.00001	<0.00005-0.0001	0.00002 - 0.0007
2,3,7,8-TCDF	E	<0.00022	0.00043	0.00036 - 0.00065	0.00029-0.00066		
	FS	<0.00018-0.00065	<0.00003-0.0014	0.0004-0.00074	0.00035-0.00054		0.0012-0.0017
	WB	<0.00025-0.0006	0.00043-0.0014	0.00057 - 0.0011	0.00036-0.00049	0.00058-0.00078	0.0011-0.0032

10.0 Uncertainty Evaluation

There are many uncertainties in completing a survey of contaminants in fish tissue and in estimating risks from consumption of these fish. This section provides a summary of the assumptions and uncertainties in evaluating the fish contaminant data and preparing the risk assessment. Some of the types of uncertainty which were encountered in this study include:

- 1) errors in sampling, fish preparation, and chemical analysis,
- 2) variability in fish tissue concentrations within fish, across species and tissue types, and among stations,
- 7) lack of comparable data-sets for comparisons, and
- 3) lack of knowledge regarding human exposure and toxicity.

10.1 Fish Tissue Collection

Uncertainty in toxic chemical levels is primarily associated with variability in fish tissue concentrations over space and time as well as errors in chemical analytical methods. The temporal (seasonal, annual) range of chemical concentrations in fish species was not known.

There was some measure of spatial variability in certain fish species which were collected at a number of sites (largescale sucker, white sturgeon, mountain whitefish, rainbow trout, chinook salmon, steelhead, Pacific lamprey). Coho salmon, bridgelip sucker, and eulachon were each only collected at one location, therefore there was no measure of spatial variability in these species. Pacific lamprey and walleye were only collected at two locations. Therefore, there were gaps in our information on contaminant levels in these species from other sections of the Columbia River Basin. In addition to a limited number of sampling locations, some of the sites included large stream reaches (Table 1-1). Therefore, the average concentrations from these sites represent sampling areas of several miles.

Individual fish tissue were composited to obtain a representative sample of the mean concentrations of fish tissue. However, by compositing the fish there is a loss of certainty in the variance among individual fish samples. To reduce some of the uncertainty associated with composites, an attempt was made to collect fish: 1) at the same time and 2) of the same size.

To maintain uniformity in sample size within composites the smallest individual within a composite was supposed to be no less than 75% of the total length of the largest individual. Seventy-nine percent of the composites were within this guideline. Of the composite samples not meeting the guideline, roughly one-half were within 70% of the total length of the largest individual. The compositing goals were not fully met in all samples because:

- 1) larger fish (rainbow trout and mountain whitefish) were added to some composites to gain enough fish tissue for analyses,
- 2) tribal members requested that small fall chinook salmon (jacks) be added to samples of larger adults, or
- 3) spatial and temporal variability in fish species limited the number of fish available for sampling.

To maintain uniformity across composites the relative difference between the average length of the individuals in the smallest-sized composite (i.e., the one with the smallest average body lengths) was to be within 10% of the average length of the largest-sized composite. Eighty-nine percent of the composites were within the 10% guideline. Of the 11% not meeting the guideline, 5 composites were steelhead, and one each were walleye, largescale sucker, rainbow trout, and spring chinook salmon.

In addition to collecting composites of the same size an attempt was made to collect replicate samples at each study site to provide a more accurate estimate of the variance in tissue analyses. The goal of collecting at least three replicate composite samples for each sample type from each study site was met at 92% of the study sites. Only two replicates or less were collected at 8% of the study sites. Replication was limited at study site 30 on the Umatilla River because the electro-fishing boat broke down, which prohibited additional collections of walleye and largescale sucker. There were a low number of rainbow trout available from study site 98 in the Deschutes River.

The uncertainty in the tissue concentrations is also associated with the sampling design. The fish type, tissue type, and sample location were all predetermined during the planning conference. This type of sampling is biased with unequal sample sizes and predetermined sample locations rather than a random design. This bias is to be expected when attempting to provide information for individuals or groups based on their preferences. The results of this survey should not be extrapolated to any other fish or fish from other locations.

EPA's guidance for preparing fish tissue for chemical analysis recommends scaling fish (USEPA, 2000f). However, CRITFC's member tribes do not typically scale their fish (CRITFC tribes, personal communication). The results of some of the chemical analyses in this report may be affected by the amount of certain chemicals (e.g. metals) which may be concentrated in the fish scales.

The homogeneity of ground fish tissue can vary considerably, depending upon the nature of the tissue sample and the grinding procedures. In this project we attempted to minimize variability of chemical measurements by specifying the fish grinding procedure (See Volume 5) and by monitoring the homogeneity of composite samples.

With the exception of white sturgeon, fish tissue chemical residues were measured in fillet with skin and whole body. White sturgeon were the only species which were analyzed as fillet without skin. As discussed in Section 2, whole body fish tissue samples tend to be somewhat higher in

lipids than fillet with skin samples for some fish species. This difference in lipids between whole body and fillet fish samples was not consistent across species. This was not surprising since the preparation of fillets with skin usually left a thin layer of subcutaneous fat remaining under the skin.

The fillet and whole body samples were not from the same fish. Therefore, any comparisons between them will be affected by the natural variability in fish samples as well as the tissue type.

10.2 Chemical Analyses

All data quality objectives established for this project were met. However, there were uncertainties in the chemical analysis due to interferences, detection limits, and method development.

A number of problems were encountered in the measurement of target compounds. For dioxins/furans, dioxin-like PCBs, non-acid labile chlorinated pesticides, and Aroclors, the primary analytical problem encountered by the laboratories was the interference of chlorinated and brominated non-target compounds in extracts of project fish samples. For dioxin-like PCBs, many sample extracts had to be diluted and re-measured because of high levels of dioxin-like PCB target compounds in some samples.

The metallic equipment used to grind fish samples was tested prior to sample analysis for possible interferences. The results indicated that lead, manganese, nickel, copper, aluminum, zinc, and PCB 105 were found in the rinsate blanks from the fish grinder. The levels of manganese, nickel, copper, aluminum, zinc, and PCB 105 were in negligible quantities and should not affect the study results. However, the lead levels (77 µg/l) in the rinsate were higher; therefore, the results reported in this study for lead may be increased over levels that would be found in tissue samples.

Modifications to digestion procedures for high levels of lipids in some project samples improved measurements of metals and mercury using EPA methods 200.8 and 251.6. The chemical analysis of chlorinated phenolics (EPA Method 1653) and neutral semi-volatiles (EPA Method 8270) had the largest number of data which were not acceptable due to high quantitation limits.

For this project, analytical methods were chosen to provide detection or quantitation limits which were as low as possible given available analytical methods and resources. The true value of chemicals which were “not detected” is actually somewhere between the reported detection limit and zero. For this study ½ the detection limit was used to estimate chemical concentrations. Appendix E lists each chemical concentration as equal to: 1) the detection limit, 2) zero, and 3) one-half the detection limit. The use of ½ the detection limit may have over or underestimated the true fish tissue concentration.

In the quality assurance review of the chemical data, certain chemical concentrations were qualified with a “J”. The “J” qualifier designates a concentration which is estimated. EPA

recommends that the J-qualified concentrations be treated in the same way as data without this qualifier with acknowledgment that there is more uncertainty associated with “estimated” data (USEPA, 1989). We chose to use these data in this assessment without conditions. Use of this data to calculate fish tissue concentrations may overestimate the true concentration since these levels may be incorrect. The data qualifiers are listed with each data point in Appendix D of Volume 1 and in Volume 4.

The percent difference in field duplicates was estimated for all chemicals analyzed. There was less than 10% difference between most of the duplicate samples. The samples with greater than 10% difference are shown in Table 10-1. The maximum difference was 157% in cobalt concentrations in fall chinook from study site 48 (Table 10-1). There was no consistent pattern of error in field duplicate by study site, chemical, or fish species.

The difference in duplicate fillets from the same fish is an indication of the variability of chemicals within fish tissue, since the fillets were from the opposite sides of the same fish. In this study, the duplicate values were averaged. By averaging the concentration of the duplicate samples fish tissue concentrations and risk estimates may be lower than the actual exposure that would occur if the higher fish tissue concentration was used.

Table 10-1 . Percent difference in field duplicate samples from the Columbia River Basin. Fish are listed with study site ID in parentheses. The maximum percent difference is given for the chemical within a chemical group.

Species (study sites)	Percent difference for analytes (greater than 10%)			
	Dioxins & Furans	Metals	PCBs	Pesticides
steelhead (96)	46 (OCDD)	68 (Ba)	56 (PCB 123)	67 (DDT)
spring chinook (94)	13 (HxCDF)	62 (Cd)	17 (PCB 189)	15 (DDT)
fall chinook (8)		29 (Hg)	14 (PCB 157)	11 (DDD)
fall chinook (48)	18 (TCDF)	107 (Cr); 157 (Co)	28 (PCB 126); 18 (Aroclor 1254)	
mountain whitefish (98)	29 (TCDD)	70 (Pb)	32 (PCB 167); 32 (Aroclor 1254)	35 (DDE)
white sturgeon (13)	29 (HxCDF)	54 (Hg)	15 (PCB 118); 11 (Aroclor 1260)	124 (nonaolcor)
white sturgeon (6)	57 (TCDF & HxCDF)	42 (Co)	39 (PCB 105); 109 (Aroclor 1254)	119 (DDT)
white sturgeon (9)	50 (OCDD)	144 (Co)	27 (PCB 169)	59 (oxychlordan)

10.2.1 Lipid analyses

All samples were measured for percent lipids according to the procedure described in EPA Method 1613B. Other percent lipid procedures such as the three extraction methods described in EPA Method 8290 would have produced different percent lipid results because of the different extraction solvents used and different extraction conditions. While the lipid values reported in our study were consistent because the analyses were all done within one laboratory using one

method, there would be considerable uncertainty in comparing the lipid levels measured in this study with other data generated by different methods or different laboratories.

10.3 Comparing Chemical Data Across Fish Species and with Other Studies

The comparison of this study with other studies is confounded by the methods that were used to collect the samples, the tissue type, number of samples, and species as well as the inconsistency in chemical methods. In particular, methods for analyzing fish tissue for dioxins, furans, and PCB congeners have changed recently. Thus, chemical analysis of fish tissue data for these particular chemicals from the 1970's through the early 1990's will not necessarily give the same results as were seen in this study.

10.4 Risk Assessment

Uncertainties can occur in all parts of the risk assessment--exposure assessment, toxicity assessment, and risk characterization. An uncertainty evaluation has been done as a part of this risk assessment to show how the risk characterization could be affected if alternative assumptions had been made and/or different parameters had been used to calculate the cancer risks and non-cancer hazard indices.

10.4.1 Exposure Assessment

10.4.1.1 Contaminant Concentrations in Fish Tissue

As discussed earlier in this report, the fish species collected and the sampling study sites selected were based primarily on data from CRITFC's Fish Consumption Report (CRITFC, 1994) and discussions with tribal staff. Although samples were taken from the study sites used most frequently by the tribes, many other study sites used for fishing were not sampled. In addition, as discussed in Section 4.5, there were limited data on the species collected and fishing locations used by non-tribal populations in the Columbia River Basin. Therefore, while the concentrations of chemicals in fish tissue have been used to characterize risk for the general public in this study, this characterization was uncertain due to the lack of data on fishing practices for the general public.

Another source of uncertainty for this risk assessment involves the use of the average chemical concentrations for fish collected over a short period of time to estimate human exposure over 30 and 70-year durations. If average chemical concentrations in fish tissue have changed over time, or were likely to change in the future, the risk estimates presented in this report may either underestimate or overestimate the risk to individuals. The relatively small amount of existing historical data on chemical contaminants in fish within the Columbia River Basin was insufficient to reliably evaluate trends in chemical concentrations. The seasonal range of chemical concentrations in the target species evaluated in this risk assessment is also not known.

Thus, the risk estimates presented in this report could increase or decrease depending upon how

concentrations vary over location and time.

As discussed in Section 1.7.5, to calculate average contaminant levels in fish, a value of one-half the detection limit was used in some cases for non-detected chemicals. Risk characterization based upon one-half the detection limit could be either an overestimate or an underestimate of the actual risks.

10.4.1.2 Tissue Type

For this study, both whole fish and fillets were analyzed when possible. The fillet and whole body sample types were chosen based on the fish consumption survey for CRITFC's member tribes (CRITFC, 1994). In this study, respondents were asked to identify the fish parts they consume for each species. For most of the fish species sampled as a part of this study, 50% or more of the respondents said that they consume fish skin. A smaller proportion of the tribal members consumed other fish parts (head, eggs, bones and organs). In addition to the question of people consuming fish parts, some chemicals preferentially accumulate in fat or internal organs, thus having both whole body and fillet fish tissue samples provides a more comprehensive picture of the amount of chemical accumulated throughout the fish tissue. Fillets were analyzed with skin because most tribal members consumed the skin with the muscle tissue.

Information on the portions of fish that are consumed most frequently by the general public were not available. However, respondents to the qualitative fish consumption survey of people from Wheatland Ferry to Willamette Falls Reach of the Willamette River, Oregon indicated that they consume primarily fish fillets as well as other fish parts and the whole body (EVS, 1998).

In Section 6.2.4, the ratios of the estimated hazard indices and cancer risks for whole body to filleted fish samples were calculated to determine the possible impact of tissue type on the risk characterization. These results were calculated for those species that had both fillet and whole body samples analyzed at a given site. For non-cancer effects, whole body to fillet ratios were calculated for the total hazard index as well as for the endpoints of immunotoxicity and reproduction. The number of whole body to fillet ratios that were greater than 1 compared to the total number of samples was also shown. These calculations (Table 6-23) did not show a consistent pattern in whole body to fillet ratios for the total hazard indices, the immunotoxicity hazard indices, or cancer risks at a given site for a species. The whole body to fillet ratios ranged from 0.2 to greater than 1 for a few species/sites (e.g. high of a ratio 6.6 for fall chinook, immunotoxicity hazard index). For reproductive effects, the ratios of the hazard indices for reproductive effects in whole body to fillet samples appear to be less than 1 more frequently than those for the other hazard indices or cancer risks. This may be because the hazard index for reproductive effects is based largely upon the contaminant mercury which is not lipophilic and binds strongly to protein (e.g., muscle tissue).

Any conclusions, however, on the results of whole body to fillet samples are limited by the small sample sizes (usually 3 or less) at each site and by the fact that whole body samples were always from a composite of fish different than those used for the whole body samples (i.e., fillet and

whole body samples are not from the same fish).

10.4.1.3 Exposure Duration

Exposure duration is defined as the time period over which an individual is exposed to one or more contaminants. For adults, two different exposure durations were used for the risk assessment: 70 years, which represents the approximate average life expectancy of all individuals born in the United States in the late 1960s; and 30 years, which represents the 90th percentile length of time that an individual stays at one residence (USEPA, 1997b).

The value of 70 years was assumed for lifetime exposure in this risk assessment because it is the value commonly assumed for the general population in most EPA risk assessments. Also, 70 years is the primary assumption used in the derivation of many of the cancer slope factors found in IRIS (USEPA, 2000c).

As was discussed in Section 4, changes in exposure duration do not impact the exposures estimated for calculating non-cancer health impacts. This is because the product of the exposure frequency (EF) times exposure duration (ED) is always equivalent to the averaging time (AT) (see Equation 4-1 in Section 4.3).

However, since the averaging time for estimating exposure for cancer risks is always a person's lifetime, changing exposure duration does impact the estimated risk. The cancer risk estimates for an individual who consumes fish over an exposure duration that differs from the exposure durations used in this report (ED_{new}) can be determined using the following equation:

$$(Equation 10-1) \quad ECR_{new} = ECR_{70} \times ED_{new}/ED_{70}$$

where:

- ECR_{new} = Excess cancer risk for the new exposure duration
- ECR_{70} = Excess cancer risk estimate for a lifetime exposure duration of 70 years
- ED_{new} = Individual exposure duration in years
- ED_{70} = Default lifetime exposure duration of 70 years

Equation 10-1 shows that the excess cancer risk will change in direct proportion to the ratio of the new and default exposure durations. For example, if an exposure duration of 9 years was selected, which is the median length of time an individual stays at one residence, the lifetime exposure cancer risk estimates would be multiplied by a factor of 0.13 (9 years ÷ 70 years = 0.13) to obtain revised cancer risk estimates for a 9-year exposure duration. Thus, all total excess cancer risk estimates for 70 years exposure duration for the fish species and tissue types evaluated in this report would decrease by approximately an order of magnitude (i.e. ten-fold) for an exposure duration of 9 years.

10.4.1.4 Consumption Rate

In this risk assessment, exposures were estimated for both the general public and for members of CRITFC's member tribes. For the general public, adequate quantitative information on fish consumption rates for those areas of the Columbia River Basin sampled in this study was not available. Therefore, the ingestion rates assumed for those individuals in this risk assessment

were based on a national report of fish consumption (USEPA, 2000b). For CRITFC's member tribes, ingestion rates were taken from CRITFC's fish consumption study (CRITFC, 1994). For both the general population and the tribes, mean and a 99th percentile ingestion rates for children and adults were selected to evaluate potential risks over a range of possible ingestion rates.

It is not known if the ingestion rates selected for this risk assessment are representative of the actual consumption practices of individuals consuming fish from the study area. The exposures estimated in this report are likely to be higher than those expected for a recreational fisherman who infrequently fishes at any of the study sites. On the other hand, as discussed in Section 4, Harris and Harper (1997) suggest that an ingestion rate of 540 g/day is more appropriate for a tribal member who pursues a traditional lifestyle. This is higher than the 99th percentile CRITFC member tribal fish consumption rate of 389 g/day used in this report.

10.4.1.5 Multiple-Species Consumption Patterns

The hazard indices and cancer risk estimates in this report were primarily based upon the consumption of individual fish species and tissue types. However, these estimates which are based upon individual fish species may not be an adequate representation of risk for most individuals since most people likely eat a diet composed of multiple fish species. Therefore, as a part of the risk characterization, a hypothetical multiple-species diet was also evaluated using tribal fish consumption data from CRITFC's fish consumption study. For this hypothetical multiple-species diet, information from Table 17 of the CRITFC fish consumption study (CRITFC, 1994) was used. This table from the CRITFC consumption survey provides information on the percentage of adults that consumed 10 fish species evaluated in the study (CRITFC, 1994). As was shown in Table 6-24 and Figures 6-35 and 6-36 the resultant cancer risk and non-cancer hazards of the multiple species diet reflect the proportion of the different types of fish in the diet and the contaminant levels in those fish. Therefore, the estimated cancer risks and non-cancer hazards from consuming fish from the Columbia River Basin for any one individual depend upon the types and amounts of fish they eat and may be very different from those estimated in this report for individual species.

As part of this uncertainty analyses, an estimate of the total cancer risks and non-cancer hazards from a multiple species diet using data from Table 18 in the CRITFC fish consumption study in addition to that in Table 17 was calculated (CRITFC, 1994). Table 18 provides average consumption rates (grams per day) for each species for those adult respondents in the survey who consume fish. These rates were determined by combining the average consumption rate for each individual who consumed a particular species with the average serving size in ounces for that individual and then calculating the mean of all of the individual consumption rates. The differences in the consumption rates for the hypothetical multiple diet using the two CRITFC tables (Table 17 versus Table 18) are shown in Table 10-2. As can be seen from Table 10-2, the

consumption rates, cancer risks and total hazards for each individual fish species differ using the results from the two different tables in the CRITFC consumption study (CRITFC, 1994). However, the total estimated cancer risks and total non-cancer hazard indices from consuming all species are approximately the same using either table.

Table 10.2. Comparison of estimated total cancer risks and hazard indices for a hypothetical multiple species diet using data from Table 17 and Table 18 in the CRITFC fish consumption report (Source: CRITFC, 1994).

Fish Species	T	Results using Table 17 in the CRITFC fish consumption study ⁽¹⁾				Results using Table 18 in the CRITFC fish consumption study		
		Percentage of Hypothetical Diet	Consumption Rate (grams/day)	Total Cancer Risk	Non-Cancer Effects (total HI)	Consumption Rate (grams/day)	Total Cancer Risk	Non Cancer Effects (total HI)
salmon	FS	27.7%	17.5	6E-05	0.6	25.7	8E-05	0.9
trout	FS	21.0%	13.3	3E-05	0.3	9.6	2E-05	0.2
whitefish	FS	6.8%	4.3	9E-05	0.7	8.9	2E-04	1.5
smelt	WB	15.6%	9.9	3E-05	0.1	4.8	2E-05	0.0
lamprey	FS	16.3%	10.3	1E-04	0.7	4.7	5E-05	0.3
walleye	FS	2.8%	1.8	4E-06	0.1	3.8	9E-06	0.2
sturgeon	FW	7.4%	4.7	7E-05	0.6	3.3	5E-05	0.4
sucker	FS	2.3%	1.5	9E-06	0.1	2.8	2E-05	0.2
Totals		100.0%	63.2	4E-04	3.2	63.6	4E-04	3.8

(1) These results are those presented in Section 6.2.5 and Table 6-24
 FS = fillet with skin FW = fillet without skin WB = whole body

T= tissue type
 HI = hazard index

10.4.1.6 Effects of Cooking

It was assumed for this risk assessment, that (with the exception of skinless white sturgeon fillets) the skin and fatty areas of the fish are not removed during preparation, and that there is no net reduction in contaminant concentrations during cooking. Anglers who prepare fillets by skinning and trimming away the fatty area may reduce their exposure to chemicals (such as organochlorines) that accumulate in fatty areas. It has also been shown that cooking the fish may affect exposure concentrations of such chemicals, depending on the cooking method.

EPA's guidance (USEPA, 2000a) provides a summary of the effects on organochlorine (e.g., PCBs, DDT, chlordane, dioxins/furans) contaminant levels in fish as a result of fish preparation and cooking. This summary shows that the reductions in chemical concentrations vary considerably among the different studies because of different fish species, contaminants, cooking methods, etc. In these studies most of the percent reductions in chemical concentrations ranged from about 10 to 60%. However, much higher losses were also seen as were net gains of one contaminant (PCBs). Overall, these studies support the conclusion that organochlorines can be lost during cooking. But, based on the available information, it is difficult to quantify these losses for use in a risk assessment since the actual losses from cooking depend upon the cooking method (i.e., baking, frying, broiling, etc.), the cooking duration, the temperature during cooking, preparation techniques (i.e., trimmed or untrimmed, with or without skin), the lipid content of the fish, the fish species, and the contaminant levels in the raw fish.

Also as discussed in EPA guidance (USEPA, 2000a), several studies indicate that some organo-metal compounds bind to different fish tissues than the tissue which bind organochlorines. Mercury, for example, binds strongly to protein, thereby concentrating in the muscle tissue of fish. Mercury also concentrates in liver and kidney, though at generally lower rates. Thus, preparations such as trimming and gutting, can actually result in a greater average concentration of mercury in the remaining tissues compared with the concentration in the whole fish (Gutenmann and Lisk, 1991). As discussed previously in the discussion on effects of sample type on the risk characterization (Section 6.2.4 and Table 6-23), the ratios of the hazard indices for reproductive effects in whole body to fillet samples appear to be less than 1 more frequently than the ratios for the total hazard index, hazard index for immunotoxicity, and cancer risks. This may be because the hazard index for reproductive effects is based largely upon the contaminant mercury which is not lipophilic and binds strongly to protein (e.g., muscle tissue). However, any conclusions based on the ratios of whole body to fillet samples are limited by the small sample sizes (usually 3 or less) at each site and by the fact that whole body samples were always from a composite of fish different than those used for the whole body analysis (i.e., fillet and whole body samples are not from the same fish).

The impact of cooking on mercury levels was studied by Morgan et al., 1997. They found that mercury concentrations (wet weight basis) in pan-fried, baked and boiled walleye fillet ranged from 1.1 to 1.5 times higher than in the corresponding raw portions; in lake trout the range was 1.5 to 2.0 times higher.

10.4.2 Toxicity Assessment

There are also uncertainties in the toxicity assessment. These include uncertainties (1) in the toxicity values (i.e., reference doses and cancer slope factors) used; (2) in the toxicity equivalence factors developed for dioxins/furans and dioxin-like PCBs and in the relative potency factors used for PAHs; (3) in the lack of toxicity data for some of the chemicals that were detected in fish, and; (4) in the manner in which certain chemicals (Aroclors, dioxin-like PCBs, DDT/DDE/DDD, and arsenic) were evaluated.

10.4.2.1 Toxicity Values

As discussed in Section 5.0, the majority of the toxicity factors used in estimating hazard indices and cancer risks were taken from EPA's IRIS database which is a database of human health effects that may result from exposure to various substances found in the environment. For a small number of chemicals whose toxicity factors were not available in IRIS, toxicity factors developed by NCEA were used. Although the development of the IRIS toxicity factors has been reviewed by a group of EPA health scientists using consistent chemical hazard identification and dose-response assessment methods, there are still several sources of uncertainty in these factors and their relevance to the populations for which the risk assessment is being conducted. As discussed in EPA's guidance (USEPA, 1989), some of these uncertainties may include:

- using dose-response information from effects observed at high doses to predict the

adverse effects that may occur in humans following exposure to the lower levels expected from human exposure in the environment;

- using dose-response information from short-term studies to predict the effects of long-term exposures;
- using dose-response information from animal studies to predict effects in humans; and
- using dose-response information from homogenous populations or healthy human populations to predict the effects likely to be observed in the general population consisting of individuals with a wide range of sensitivities.

In addition to the uncertainties in developing reference doses and cancer slope factors based upon the data that are available, there are also uncertainties in the fact that specific types of effects data are often not available for a given chemical. Some examples include the lack of data on a chemical's cancer and non-cancer impact on vulnerable populations (e.g., children) and a lack of information for some chemicals on non-cancer endpoints such as reproductive, developmental, and endocrine disruption. However, the lack of data on non-cancer effects is usually considered when determining what uncertainty factors and modifying factors should be used to develop a reference dose for a given chemical. The lack of data on cancer is partially addressed by using conservative assumptions (e.g., upper confidence levels, the most sensitive species) in estimating cancer slope factors. All of these assumptions are intended to provide a margin of safety to ensure that the health impacts for an individual chemical are not likely to be underestimated.

To better understand the uncertainties associated with the toxicity factors for each of the chemicals evaluated in this risk assessment, refer to the Toxicity Profiles in Appendix C. These profiles review the data upon which the reference doses and cancer slope factors were developed.

10.4.2.2 Toxicity Equivalence Factors for Dioxins, Furans, and Dioxin-like PCB Congeners and Relative Potency Factors for PAHs

Toxicity equivalence factors were used for the chlorinated dioxins and furans and the dioxin-like PCBs measured in this study to calculate toxicity equivalence concentration. These toxicity equivalence factors were calculated using all of the available data and were selected to account for uncertainties in the available data and to avoid underestimating risk (Van den Berg et al., 1998). Alternative approaches, including the assumption that all dioxin-like PCBs carry the toxicity equivalence of 2,3,7,8-TCDD, or that all chlorinated dioxins, furans, and dioxin-like PCB congeners other than 2,3,7,8-TCDD can be ignored, have been generally rejected as inadequate for risk assessment purposes by EPA and many other countries and international organizations. These toxicity equivalence factors are order-of-magnitude estimates relative to the toxicity of 2,3,7,8-TCDD. Therefore, their use creates uncertainty in the risk assessment, especially since chlorinated dioxins/furans and dioxin-like PCBs contribute significantly to the cancer risks estimated in this risk assessment.

Also, it should be noted that the cancer slope factor for 2,3,7,8-TCDD is being re-evaluated as part of a current review by EPA (USEPA, 2000e). A review of the most current draft document suggests that this cancer slope factor may increase. This change would affect both the cancer risk estimates associated with 2,3,7,8-TCDD as well as those risk estimates calculated for the other chlorinated dioxins, furans, and dioxin-like PCB congeners having toxicity equivalence factors. If the slope factor increases, cancer risks estimated for these classes of compounds would also increase.

As discussed in Section 5, EPA has developed provisional guidance on estimating risk from exposure to PAHs (USEPA, 1993). A cancer slope factor is available for only one PAH, benzo(a)pyrene. In this provisional guidance, relative potency factors have been developed for six PAHs relative to benzo(a)pyrene. These relative potency factors were used to estimate cancer risk from PAHs in this risk assessment. As with the toxicity equivalence factors these relative potency factors are order-of-magnitude estimates and, therefore, have inherent uncertainties. However, unlike the toxicity equivalence factors, these relative potency factors for the PAHs are considered to be more uncertain because they do not meet all of the criteria for the application of toxicity equivalence factors to mixtures.

In our study, with the exception of one composite sample of largescale sucker taken at study site 13 (see discussion in Section 6.2), PAHs do not contribute significantly to the levels of contaminants in fish or to cancer risk estimates from consuming fish. Therefore, the uncertainties in the use of relative potency factors for PAHs should not greatly impact the overall risks characterized in this report.

10.4.2.3 Chemicals Without Quantitative Toxicity Factors

As shown in Table 5-1, there were 23 chemicals that were analyzed for in fish tissue that do not have a cancer slope factor or reference dose. Of the 23 chemicals without toxicity values, the following 14 chemicals were not detected in any fish species: delta-BHC, dibenzofuran, gamma-chlordene, tetrachloroguaiacol, 4-bromophenyl-phenylether, 4-chloroguaiacol, 4-chlorophenyl-phenylether, 3,4-dichloroguaiacol, 4-chloro-3-methylphenol, 4,5-dichloroguaiacol, 4,6-dichloroguaiacol, 3,4,5-trichloroguaiacol, 3,4,6-trichloroguaiacol, and 3,5,6-trichloroguaiacol. Six additional chemicals were detected in less than 3% of the samples: acenaphthylene, alpha-chlordene, benzo(ghi)perylene, phenanthrene, retene, and 1-methyl-naphthalene. Of the remaining 3 chemicals, DDMU was detected less than 10%; 2-methyl-naphthalene and pentachloroanisole were detected greater than 10% of the time.

As discussed in the Toxicity Profiles (Appendix C), the toxicity and mechanism(s) of action(s) of pentachloroanisole are similar to those of its parent chemical, pentachlorophenol. However, methylation of the chlorophenols makes them more polar, and thus likely to be somewhat less reactive in biological systems. Thus the extent of both acute and chronic toxicity of pentachloroanisole can be reasonably anticipated to be somewhat less than its chlorinated parent, PCP. DDMU is a breakdown product of the DDT. Little information is available on DDMU or 2-methyl-naphthalene.

It is impossible to predict how the lack of toxicity information on these 23 chemicals might impact the characterization of risk in this report. However, given the fact that only 2 of these chemicals (2-methyl-naphthalene and pentachloroanisole) were detected in greater than 10% of the samples, any underestimation of cancer risk and non-cancer hazards is unlikely to be great.

There are no EPA consensus reference doses available for the chlorinated dioxins and furans and the dioxin-like PCB congeners, therefore, the possible non-cancer health effects from exposure to these chemicals from fish consumption could not be estimated in this report. From the most recent draft of EPA's reassessment of the toxicity of these compounds (USEPA, 2000e), it is clear that these compounds can cause non-cancer effects at very low levels of exposure. The inability to characterize the non-cancer hazards from these compounds may result in an underestimate of the non-cancer hazards calculated in this report.

10.4.2.4 Risk Characterization for PCBs

As discussed in Section 1, two different measurements were used in this study to determine PCB concentrations in fish tissue: 1) analysis of Aroclors which are commercial mixtures of both dioxin-like and non-dioxin-like PCB congeners, and 2) analysis of individual dioxin-like PCB congeners. The Aroclor methodology included the analysis of 7 Aroclors: Aroclor 1016, Aroclor 1221, Aroclor 1232, Aroclor 1242, Aroclor 1248, Aroclor 1254, and Aroclor 1260. Only Aroclors 1242, 1254, and 1260 were detected. Eleven dioxin-like PCB congeners that exert toxicity similar to 2,3,7,8-TCDD were also measured. PCB 170 and PCB 180, though measured, were not considered in the risk assessment as dioxin-like PCB congeners because they do not currently have associated toxicity equivalence factors.

Cancer Risks for PCBs

Because Aroclors are a mixture of both dioxin-like and non-dioxin-like PCB congeners, calculating and summing the risk associated with both Aroclors and with individual dioxin-like PCB congeners would likely overestimate cancer risk by accounting for the dioxin-like PCB congener risk both individually and within the risk estimates for Aroclors. Therefore, before using the Aroclor fish concentrations to calculate cancer risk, an adjustment was made to the Aroclor concentrations by subtracting the concentration of dioxin-like PCB congeners from the total Aroclor concentrations for each sample. This resulted in what is called the "adjusted Aroclor" value.

To estimate the impact of using this method on the cancer risk, a comparison was made for estimates of cancer risk from PCBs using different methods. The excess cancer risks calculated with these methods (using basin averages) for each fish species are shown in Table 10-3. The risk from dioxin-like PCB congeners alone ranged from 0.5 (coho salmon) to 3.5 (rainbow trout) times (column B/A) the risk calculated for total unadjusted Aroclors alone. Because the mass of dioxin-like PCB congeners is so small compared to that of the Aroclors, the risk estimated for adjusted Aroclors (subtracting the concentration of dioxin-like PCB congeners from the total Aroclor concentrations) (column C) is only slightly lower than that for total unadjusted Aroclors

(Column A). Characterizing PCB risks by combining either total Aroclors plus dioxin-like PCB congeners (A + B) or adjusted Aroclors plus dioxin-like PCB congeners (B + C) is approximately the same. The PCB risks estimated from using “adjusted Aroclors plus dioxin-like PCB congeners” is from 1.5 to 4.3 times that estimated from using total unadjusted Aroclors alone (Column B+C /A).

Table 10-3. Estimated Cancer Risks for PCBs Using Different Methods of Calculation. CRITFC’s member tribal adult, average fish consumption, 70 years exposure using average Columbia River Basin-wide chemical concentrations.

	A	B	B/A	C	A+B	B+C	(B+C)/ (A+B)	(B+C)/A
	Total unadjusted Aroclors	Dioxin- like PCB congeners	Risk Ratio	Adjusted Aroclors only	Total Aroclors plus dioxin- like PCB congeners	Adjusted Aroclors plus dioxin-like PCB congeners	Risk Ratio	Adjusted Aroclors plus dioxin- like PCB congeners / total unadjusted Aroclors
bridgelip sucker	1.1E-04	1.2E-04	1.1	1.0E-04	2.3E-04	2.3E-04	0.98	2.1
largescale sucker	7.6E-05	1.1E-04	1.4	7.1E-05	1.8E-04	1.8E-04	0.97	2.4
mountain whitefish	3.5E-04	7.7E-04	2.2	3.0E-04	1.1E-03	1.1E-03	0.96	3.1
white sturgeon	2.0E-04	1.7E-04	0.8	1.9E-04	3.7E-04	3.6E-04	0.97	1.8
walleye	2.3E-05	2.6E-05	1.1	2.1E-05	4.9E-05	4.6E-05	0.95	2.0
rainbow trout	2.5E-05	8.7E-05	3.5	2.2E-05	1.1E-04	1.1E-04	0.97	4.3
coho	4.6E-05	2.5E-05	0.5	4.5E-05	7.0E-05	7.0E-05	0.99	1.5
fall chinook	3.1E-05	3.6E-05	1.2	3.0E-05	6.8E-05	6.6E-05	0.98	2.1
spring chinook	2.9E-05	4.8E-05	1.7	2.8E-05	7.7E-05	7.6E-05	0.98	2.6
steelhead	4.4E-05	7.5E-05	1.7	4.2E-05	1.2E-04	1.2E-04	0.99	2.7
eulachon	ND	9.5E-06	NA	ND	9.5E-06	9.5E-06	1.00	NA
Pacific lamprey	1.6E-04	3.3E-04	2.1	1.5E-04	4.8E-04	4.7E-04	0.98	3.0

ND = not detected NA = not applicable

Non-Cancer Effects from Aroclors

The immunological endpoint was based upon the toxicity of Aroclors. However, only one of the three Aroclors detected in the fish samples has a reference dose - Aroclor 1254. Therefore, two possible methods were available to estimate the non-cancer hazard for the immunotoxicity endpoint.

- (A) - estimate the hazard index using the concentration of Aroclor 1254 only and the reference dose for Aroclor 1254, or
- (B) - assume that the reference dose for Aroclor 1242 and 1260 are equivalent to that for Aroclor 1254; estimate the hazard index by summing all three Aroclor concentrations and use this sum with the reference dose for Aroclor 1254.

Method B was used in this risk assessment. To show the potential uncertainties with using Method B, the hazard indices calculated with both methods (using basin averages) for each fish species are shown in Table 10-4.

Table 10-4. Comparison of Hazard Indices for the Immunological Endpoint Based on Alternative Treatments of Aroclor Data. CRITFC's member tribal adult, average fish consumption, using average Columbia River Basin-wide chemical concentrations.

	Endpoint specific hazard index for immunotoxicity		(B/A) Ratio of the hazard index for the sum of Aroclors to the hazard index for Aroclor 1254 only
	(A) Aroclor 1254	(B) sum of Aroclors 1242, 1254, and 1260	
bridgelip sucker	1.1	2.7	2.5
largescale sucker	0.8	1.9	2.4
mountain whitefish	5.1	8.7	1.7
white sturgeon	2.6	5	1.9
walleye	0.6	0.6	1.0
rainbow trout	0.6	0.6	1.0
coho salmon	0.7	1.1	1.6
fall chinook salmon	0.8	0.8	1.0
spring chinook salmon	0.7	0.7	1.0
steelhead	0.7	1.1	1.6
eulachon	ND	ND	ND
Pacific lamprey	3.9	3.9	1.0

ND = Not Detected

Table 10-4 also shows the ratio of the hazard index calculated using (A) Aroclor 1254 concentrations only or (B) the sum of all three Aroclors. For walleye, rainbow trout, spring chinook, fall chinook, and Pacific lamprey, the method used has no impact on the hazard index calculated for the immunotoxicity endpoint. This is because for these five species, only Aroclor 1254 was detected in the fish sampled. For the other species, the hazard index based on Method B (using the sum of all Aroclor concentrations) is from 1.6 to 2.5 times higher than the hazard index based upon Aroclor 1254 alone (column B/A).

10.4.2.5 Non-Cancer Effects from DDT, DDD, and DDE

DDT and its derivatives, DDD and DDE, were measured in fish tissue samples; however, only DDT has a reference dose. The reference dose for DDT is based upon its toxic effects on the liver (hepatotoxicity). For the non-cancer hazard assessment done in this report, two possible methods for the estimation of the hazard quotient and hazard index from these chemicals were possible:

- (A) - estimate the hazard quotient using the concentrations of DDT only and the reference dose for DDT, or
- (B) - assume that the reference doses for DDD and DDE are equivalent to that for DDT. Therefore, first sum the concentrations of all of the DDD, DDE and DDT species in each sample and utilize the reference dose for DDT to estimate the hazard quotient from the summed concentrations of DDD, DDE, and DDD

Table 10-5. Comparison of Hazard Quotients and Hazard Indices for the Hepatic Health Endpoint Based on Alternative Treatments of DDT, DDD, and DDE Data. CRITFC's member tribal adult, average fish consumption, using average Columbia River Basin-wide chemical concentrations.

Species	Hazard quotient		Hazard Index for hepatic endpoint			
	A	B	(B/A)		(D/C)	
	DDT only	Total DDT	HQ (Total DDT)/ HQ (DDT)	DDT only	sum of DDT, DDE, and DDD	HI (Total DDT)/ HI (DDT)
bridgelip sucker	0.08	0.95	11	0.13	1.00	7.5
largescale sucker	0.04	0.44	11	0.10	0.50	5.0
mountain whitefish	0.03	0.76	27	0.19	0.93	4.8
white sturgeon	0.02	1.04	52	0.36	1.38	3.9
walleye	0.00	0.10	28	0.47	0.57	1.2
rainbow trout	0.01	0.05	8	0.04	0.09	2.1
coho salmon	0.00	0.01	4	0.06	0.07	1.2
fall chinook	0.00	0.03	7	0.08	0.10	1.4
spring chinook	0.01	0.04	4	0.08	0.11	1.3
steelhead	0.00	0.03	8	0.07	0.10	1.4
eulachon	ND	0.02	NA	0.05	0.07	1.4
Pacific lamprey	0.06	0.17	3	0.22	0.33	1.5

ND = not detected; NA = not applicable
 HS = hazard quotient
 HI = Hazard index
 Total DDT = sum of DDT, DDD, DDE

Method B was used to characterize non-cancer health effects in this study. Because DDT has been identified as having a hepatic (liver) toxicity endpoint, the treatment of DDT and its derivatives will affect not only the hazard quotient for these species, but also the hazard index for the hepatic (liver) toxicity endpoint.

Table 10-5 compares the hazard quotients for DDT and its derivatives (in columns A and B) as well as the hazard indices for the hepatic endpoint (in columns C and D) using the two methods. As can be seen from Table 10-5, the hazard quotient increased from about 3 times for Pacific lamprey to 52 times for white sturgeon when all three species (DDT, DDE, DDD) are summed to calculate the hazard quotient compared to calculating the hazard quotient using DDT data alone. The impact on the hepatic endpoint is less because for some fish species other chemicals in addition to DDT and its derivatives are included in the calculation of the hazard index for hepatotoxicity. The ratio between the hepatic hazard index using DDT, DDE, and DDD to the hepatic hazard index using DDT alone ranges from between 1.2 for coho salmon to 7.5 for bridgelip sucker, with the highest ratios seen in some of the resident fish species. Thus, the endpoint specific hazard indices for hepatotoxicity that are discussed in Section 6 may be an overestimate if DDE and DDD are less toxic to the liver than DDT. This is primarily true for several of the resident species.

10.4.2.6 Risk Characterization for Arsenic

As discussed in Section 5.3.3, total arsenic was measured in fish tissue samples in this study. Because a reference dose and cancer slope factor are available for only inorganic arsenic, an

assumption about the percent of inorganic arsenic in fish had to be made to estimate the non-cancer hazards and cancer risks. The non-cancer hazards and cancer risks discussed in Section 6.2.1 and 6.2.2, respectively, assumed that for all fish species (resident fish and anadromous fish) caught in this study, 10% of the total arsenic was inorganic arsenic. The data in Section 5.3.3 also suggests that an alternative assumption for anadromous fish species should be considered - the assumption that 1% of the total arsenic is inorganic. Therefore in Section 6.2.6, the non-cancer hazards and cancer risks were recalculated for anadromous fish species using basin data assuming that 1% of the total arsenic was inorganic.

This comparison of the results from using the two different assumptions (1% versus 10%) for arsenic in fish shows that the reduction of the non-cancer hazards is less than 12% for all anadromous fish species, except eulachon which had about a 50% reduction. However, the impact is greater on the estimates of cancer risk. With the exception of lamprey for which cancer risks were reduced by only 6%, the reductions in cancer risks for steelhead were about 29%. The cancer risks for the other anadromous fish species were reduced from about 40% to 50%. Thus, the assumptions used for percent inorganic arsenic have the most impact on the cancer risks estimated for salmon, steelhead and eulachon and on the non-cancer hazards for eulachon.

10.4.3 Risk Characterization

10.4.3.1 Cancer Risk Estimates

As recommended by EPA's guidance on mixtures (USEPA, 2000g), the total cancer risk from a sample is calculated by summing the risk of individual carcinogenic compounds in that sample. This approach for carcinogens (response addition) assumes independence of action by the components in a mixture (i.e., that there are no synergistic or antagonistic interactions among the carcinogens in fish and that all chemicals produce the same effect, cancer). If these assumptions are incorrect, over- or under-estimation of the actual risks could result. The underlying biological basis for assuming synergism is that cancer is a multistage process where a series of events transforms a normal cell into a malignant tumor. If two carcinogens act at different stages, their combined effect can be greater than either acting alone. For example, initiation-promotion studies have demonstrated synergistic effects for some pairs of carcinogens. On the other hand, similar-acting carcinogens can compete with each other to result in antagonism. For example, the presence of one metal can decrease the absorption or effectiveness of a similar metal. Interactions can be quite complex and can depend on dose or other factors, including background exposures to other carcinogens. In general, available information seldom allows quantitative inferences to be made about potential interactions among carcinogens. In the absence of such information, the practice is to assume additivity, particularly at low doses for mixtures.

Summation of carcinogenic risks for substances with different weights-of-evidence for human carcinogenicity is also an uncertainty. The cancer risk equation for multiple substances sums all carcinogens equally, giving as much weight to class B or C as to class A carcinogens. Using the assumption of additivity gives equal weight to all slope factors without regard to their basis from human data. In this assessment, only arsenic is in the class A carcinogen group (human carcinogen based on human data) and all of the other major contributors to cancer risk (e.g., DDT

and DDE, DDD, Aroclors, dioxin-like PCB congeners and chlorinated dioxins and furans) are in the class B2 group (probable human carcinogen based on sufficient evidence in animals and inadequate or no evidence in humans). It should be noted, however, that EPA's most recent draft document on the toxicity of 2,3,7,8-TCDD and related compounds (USEPA, 2000e) characterizes the complex mixtures of dioxins to which humans are exposed as "likely human carcinogens".

The cancer slope factors used in this risk characterization are primarily from EPA's database, IRIS. Most of the IRIS cancer slope factors are considered to be plausible upper bounds to the actual lifetime excess cancer risk for a given chemical. Concern has often been raised that adding multiple carcinogens, whose slope factor are upper bound estimates, will lead to unreasonably high estimates of the actual risk. Statistical examination of this issue suggests that the error in the simple addition of component upper bounds is small compared to other uncertainties, and that as the number of mixture components increases, summing their upper bounds yields an inflated but not misleading estimate of the overall risk (Cogliano, 1997). In fact, division by a factor of two can be sufficient to convert a sum of upper bounds into a plausible upper bound for the overall risk. If one or two carcinogens predominate the risk, however, this is not of concern.

10.4.3.2 Non-Cancer Health Effects

In Section 6, non-cancer health impacts were evaluated in several ways. First, the hazard quotient was calculated. The hazard quotient, which is the ratio between an individual's estimated exposure to a chemical compared to the reference dose for that chemical, assumes that there is a level of exposure (i.e., the reference dose) below which it is unlikely for even sensitive populations to experience adverse health effects. As a rule, the greater the value of the hazard quotient, the greater the level of concern. However, it is important to emphasize that the level of concern does not increase linearly as the reference dose is approached or exceeded for each chemical because reference doses for different chemicals do not have equal accuracy or precision and are not based on the same severity of toxic effects. Therefore, the possible health impacts resulting from exposures greater than the reference dose can vary widely depending upon the chemical.

Based on EPA guidance (USEPA, 1986a; USEPA, 1989; USEPA, 2000g), the hazard quotients calculated for each chemical in a sample were then summed to give a hazard index. This approach of adding all of the hazard quotients regardless of endpoint (dose addition) has several uncertainties because it assumes that all compounds in a mixture have similar uptake and pharmacokinetics (absorption, distribution, and elimination in the body) and it results in combining chemicals with reference doses that are based upon very different critical effects, levels of confidence, uncertainty/modifying factors, and dose-response curves. Since the assumption of dose additivity is most properly applied to compounds that induce the same effect by the same mechanism of action, EPA guidance recommends that when the total hazard index for a mixture exceeds 1, the chemicals in that mixture should be segregated by effect and mechanism to derive endpoint-specific hazard indices (USEPA, 1986a).

Although deriving endpoint specific hazard indices, as was done for this risk assessment, likely reduces the uncertainty in the non-cancer hazard evaluation in this risk assessment, these

uncertainties are not eliminated. For example, calculation of endpoint specific hazard indices may still be incorrect estimates of non-cancer health impacts. Although two chemicals may affect the same organ (e.g. the liver), they may not necessarily do so by the same specific toxicological process.

However, it should be noted that in this assessment the majority of the estimated non-cancer hazards resulted from a limited number of chemicals: Aroclors, mercury, total DDTs, and arsenic. The highest endpoint specific hazard indices were for immunotoxicity (due to Aroclors), central nervous system and reproduction/developmental (due to mercury), liver (due primarily to DDT, DDE and DDD), and hyperpigmentation/cardiovascular (due to arsenic). These endpoint specific hazard indices are based in large part on a single chemical or class of chemical (e.g. total DDTs). Therefore, the many uncertainties regarding calculation of endpoint specific hazard indices using a mixture of chemicals should not play a major role in the characterization of non-cancer hazards.

10.4.3.3 Cumulative Risk from Chemical and Radionuclide Exposure

Risks were combined for all carcinogens to equal a total cancer risk. However, radionuclides were not included in this estimate because radionuclide analyses were not completed for all species in this assessment.

10.5 Risk Characterization for Consumption of Fish Eggs

As discussed in Section 4.5, a small number of egg samples were collected for some of the anadromous fish species. Although the fish consumption studies discussed in this report suggest that both CRITFC's member tribes and some of the general public consume eggs, none of these studies provided information on the amount of eggs consumed. Therefore, a risk characterization of eggs was not included in Section 6. However, to provide information on the potential risks from consuming eggs, the average fish ingestion rates for adults and children (general public and CRITFC's member tribes) were used for estimating cancer risk (adults only) and non-cancer hazards (adults and children) for eggs. These estimates for eggs, which are shown in Appendix P, are very uncertain but they serve as a useful comparison to the results for fish consumption.

Three samples of eggs were collected from coho salmon (Umatilla), fall chinook (Columbia, site 8), and steelhead (Columbia, site 8) and six egg samples were collected from spring chinook (3 at the Umatilla and 3 at Looking Glass Creek).

Endpoint specific and total hazard indices for eggs were calculated using the average fish ingestion rates for each population (adult and child, general public and; adult and child, CRITFC's member tribes)(Tables 1.1 and 1.2 (coho salmon), 2.1 and 2.2 (fall chinook salmon), 3.1 and 3.2 (spring chinook salmon), 4.1 and 4.2 (steelhead)). This provides estimates of the non-cancer hazards for two ingestion rates for adults (7.5 and 63.2 g/day) and children (2.83 g/day, up to age 6; and 24.8 g/day, up to age 15). No endpoint specific hazard indices and no total hazard indices greater than 1 were found using the average fish consumption rate for the general public, adult or child. At the average consumption rate for CRITFC's member tribal adults and children,

some of the total hazard indices were greater than 1 for eggs, the highest being approximately 4 for steelhead eggs at the average fish consumption rate for CRITFC's member tribal children. Endpoint specific hazard indices greater than 1 (high of 2) for liver, immunotoxicity, and selenosis were seen for CRITFC's member tribal child, average ingestion rate for spring chinook and steelhead; an immunotoxicity endpoint specific hazard index of approximately 1 was seen for coho. Endpoint specific hazard indices greater than 1 were due to exposures greater than the reference dose for total Aroclors (immunotoxicity) and selenium (selenosis and liver).

Cancer risks for eggs were calculated using the average fish ingestion rates for both adult populations (general public adult and CRITFC's member tribal adult) for both 30 and 70 years of exposure. These results are found in the tables in Appendix P (Tables 1.3 (coho salmon), 2.3 (fall chinook salmon), 3.3 (spring chinook salmon), and 4.3 (steelhead)). As can be seen from these tables, cancer risks from consumption of eggs ranged from 4×10^{-6} for both fall chinook and steelhead at the lowest exposures (general public adult, average fish ingestion rate, 30 years exposure) to a high of 8×10^{-5} for the highest exposure calculated (average fish consumption rate, CRITFC's member tribal adult, 70 years of exposure). For these same exposures, coho salmon eggs ranged from 7×10^{-6} to 1×10^{-4} and spring chinook eggs from 9×10^{-6} to 2×10^{-4} .