HUMAN HEALTH RISK CHARACTERIZATION FOR DIETARY EXPOSURE TO POLYCHLORINATED BIPHENYLS (PCBs) IN FISH FROM THE COLUMBIA BASIN IRRIGATION PROJECT:

A PROBABILISTIC APPROACH

BY

CHRISTOPHER L. MAGAN

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To the Faculty of Washington State University:

The members of the Committee appointed to examine the thesis of CHRISTOPHER L. MAGAN find it satisfactory and recommend that it be accepted.

Allan Felsot, Ph.D., Chair

John Strand, Ph.D.

Vincent Hebert, Ph.D.

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Abstract

By Christopher L. Magan, M.S.
Washington State University
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Chair: Allan Felsot

The Columbia Basin Irrigation Project (CBIP) provides water to over 671,000 acres of agricultural land in the Columbia River Basin in central Washington. Located within a region characterized by a climate of long summers, mild winters and frequent sunshine, the CBIP offers an excellent resource for local recreational fishing. With this increase in fishing opportunity, potential hazards now exist from the ingestion of fish tissue contaminated with polychlorinated biphenyls (PCBs). The purpose of this study was to assess PCB concentrations in fish tissue and to probabilistically characterize their risks to human health. Twelve sites for sampling fish were chosen within the jurisdictions of the CBIP and the Columbia Basin National Wildlife Refuge. Exposure of two age-related demographic groups to PCBs was assessed for risks of both non-cancer and cancer effects. Exposure was probabilistically modeled using the Monte Carlo sampling technique. Risks were calculated using EPA-developed functions recommended for assessing chemical contaminant data to be used in fish advisories. Results from the sensitivity analysis showed chemical concentration in fish tissue, exposure frequency, and ingestion rates to be the most important factors contributing to risks from PCBs exposure from contaminated fish. The perspective of risk derived from a probabilistic analysis of exposure contrasted with the perspective derived from the deterministic analyses associated with fish consumption advisories that are issued by State regulatory agencies. The probabilistically characterized risks of carcinogenic or non-carcinogenic hazards associated with consumption of CBIP fish by the general population of recreational fishermen were small and below tolerable regulatory levels of concern.

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Dedication

To my daughter Sierra Skye you can do anything you put your mind to; the world is yours.

Introduction and Literature Review

History and Surrounding Area

The Columbia Basin Irrigation Project (CBIP) began in 1933 as a federal relief project with the allocation of funds for the construction of Grand Coulee Dam on the Columbia River in Central Washington State (Figure 1). The dam and hydroelectric power structures were nearly complete by 1941, but the start of World War II postponed completion of the irrigation project. In 1943, work on the project resumed with the excavation of the water supply canal (Main Canal) to help establish irrigated lands in the future. The first development of irrigated lands began in 1948 in the southern part of the project with water being pumped from the Snake and Columbia Rivers. In the northern part of the project irrigation water was delivered through the Main Canal system beginning in 1952. Most of the irrigated acreage was developed between 1952 and 1959, with some additional acreage added during the mid-1960s (U.S. Bureau of Reclamation 1989).

Following the construction of Grand Coulee Dam, control of CBIP was the responsibility of the United States Bureau of Reclamation. In 1969 all of the responsibility for operations and maintenance of the basic irrigation system was transferred to three irrigation districts: the South Columbia Basin Irrigation District, the Quincy Columbia Basin Irrigation District, and the East Columbia Basin Irrigation District (U.S Bureau of Reclamation, 1976). The CBIP has now grown to be one of the largest agricultural irrigation projects in the western United States, irrigating about 671,000 acres in central Washington State.

Dominating the geographical setting of the "Big Bend Region," the CBIP extends north to south from the Ephrata-Soap Lake area to the Tri-Cities, and east to west from Connell and Warden to the Columbia River. Water for the Project is pumped from Franklin D. Roosevelt Lake, which forms behind Grand Coulee Dam on the Columbia River, into Banks Lake. From Banks Lake, the Main Canal carries water to the northern end of the Project, where secondary canals deliver water throughout the irrigated area. The Potholes Reservoir, in the central portion of the Project, captures return flows from the northern Project area and, in turn, provides water to the southern Project area. Additional return flows enter the Columbia River within the Wanapum, Priest Rapids, and McNary Pools (Figure 1).

Environmental Setting

The CBIP is a multi-purpose project providing irrigation water to agricultural land in the Columbia River basin in central Washington. The CBIP drainage area is more than 4,000 square miles and is bounded on the west by the Columbia River and on the south by the Snake River, and it extends east and north to include all lands considered economically irrigable from the project canal system (Weaver 1999). The main geologic formation in the CBIP is the Columbia Plateau River Basalt Group that exhibits various extrusions of basalt lava that helped form the Columbia Plateau between 6 and 16.5 million years ago. This area also has overlaying deposits of unconsolidated sediment and silt across the Basin. The Columbia Plateau River Basalt Group flows are estimated to be more than 14,000 ft thick near Pasco, Washington. Parts of the CBIP also contain loess, a wind-deposited silt covering various area sand dunes located throughout the project (Drost et. al. 1990).

Figure 1. The extent of the Columbia Basin Irrigation Project in 2008. (Source: http://www.usbr.gov/pn/project/columbia_index.html)

Major land use and economy in the CBIP is agriculture, followed by livestock production and food processing. The land is primarily rural, but five towns in the CBIP area have populations greater than 5,000: Pasco, Othello, Moses Lake, Ephrata, and Quincy. Pasco is the largest town, with a population of about 32,000 (U.S. Census Bureau 2004).

The CBIP has few natural perennial streams, and stream flow in the project area is enhanced and partially regulated by seasonal delivery of irrigation water (mainly April to October) to the area. Irrigation water is pumped from Lake Roosevelt, stored in Banks Lake, and distributed throughout the CBIP by a network of regulated reservoirs and conveyance canals to irrigated lands. Surface-water applied to irrigated lands eventually returns to a wasteway, which creates an irrigation-return flow directly to all major streams and rivers in the area. Because of the large quantities of irrigation water delivered to farm units in the CBIP, large fluctuations in irrigation-return flows to various drainage basins throughout the area can increase sediment loading and transport during the irrigation season.

Hydrology

The hydrology of the region is dominated by the irrigation system, as nearly all surface water is contained in canals and agricultural drains. The diversion of water to the Columbia Basin from the Columbia River has caused the groundwater table to rise tens to hundreds of feet, increasing base flow of shallow groundwater to many of the intermittent waterways in the area. Water flow during the irrigation season is dominated by surface water diversions. Base flow conditions occur during winter months when groundwater is draining out of the soil. Land use is predominantly irrigated and dry land agriculture and rangeland. Major crops grown in the CBIP include alfalfa, wheat, corn, potatoes, and tree fruit (Williamson et. al. 1998).

Climate

Climate within the basin is mostly arid, with annual rainfall in the range of 6-10 inches and most of the precipitation occurring during the winter months. Seasonal temperatures within the CBIP range from below freezing during winter to over 34^{o}C during the mid-summer months (Williamson et. al. 1998).

Polychlorinated Biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) are a group of man-made, chlorinated organic chemicals that were first introduced into commercial use in 1929 as insulating fluids for electric transformers and capacitors. In the United States, PCBs were produced by the Monsanto Company in Sauget, IL and given the trade name of Aroclor. After initial commercialization of PCBs, many other applications were developed including use as hydraulic fluids, paint additives, plasticizers, adhesives, and fire retardants. Production of PCBs in the United States was halted in 1977 because of uncertainty regarding toxicity of the ubiquitous and persistent environmental residues (ATSDR 2000).

Chemistry

Aroclors are commercial formulations of polychlorinated biphenyl congeners (Figure 2) defined by a four-digit number to describe the various constituent chlorine content. For example, the first two digits of Aroclor 1242 represent the 12 carbon atoms in biphenyl, and the second two digits represent the percentage by weight of chlorine (42%) within the mixtures. Aroclor 1016 is an exception to the general naming scheme, containing 12 carbons with 41% by weight chlorine content.

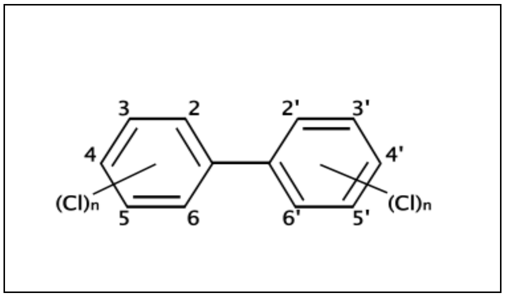

Figure 2. Basic PCB structure.

PCBs exist as mixtures containing 209 possible structural configurations (i.e., congeners) that vary by number and location of chlorine atoms on the base biphenyl structure. Number and ring substitution patterns of the chlorine atoms define each homologous congener. Only 36 congeners are considered relevant for human and ecological health based on their toxicity and occurrence in the environment (McFarland and Clark, 1989). In general, PCB persistence and toxicity increase with the degree of chlorination within the commercial mixture (e.g., Aroclor 1254 is more persistent than Aroclor 1232) (ATSDR 2000). Octanol-water partition coefficients (K_{ow}) tend to increase and water solubilities tend to decrease as PCB chlorination increases. Because of slow metabolism in tissue and high K_{ow}, PCBs accumulate in fatty tissues and can be biomagnified in the food chain (Abramowicz 1995).

PCB congeners are named by their IUPAC designated nomenclature and by a system of sequential numbering that has become known as the BZ number. In the IUPAC system the numbers in the beginning of the name specify the positions where chlorines are located on the two phenyl rings. The second and most common system developed by Ballshmiter and Zell (1980) assigns a separate number ranging from 1-209 for each of the 209 specific PCB congeners. This correlates to the structural arrangement of the PCB congeners in an ascending order of the number of chlorine substitutions within each sequential homolog. For example, if each of the rings has a chlorine in the 3,3' and the 4,4' position then the IUPAC name would be 3,3',4,4'-tetrachlorobiphenyl and the assigned BZ number is 77. If the chlorine substitution for each of the phenyl rings occurs on the 3,4 and the 4',5 position the IUPAC name would be 3,4, 4',5-tetrachlorobiphenyl and the assigned BZ number is 81 (Table 1). This example also points out that differences among BZ numbers may indicate a position change of the chlorines on the phenyl rings and not necessarily the total chlorine number.

Table 1. Toxic PCB Congeners and associated IUPAC nomenclature and assigned BZ numbers.

Toxic Polychlorinated Biphenyl (PCB) Congeners		
IUPAC Name	*Empirical Formula*	*Ballschmiter(BZ) Congener Number*
3,3',4,4'-Tetrachlorobiphenyl	$C_{12}H_6Cl_4$	77
3,4,4',5-Tetrachlorobiphenyl	$C_{12}H_6Cl_4$	81
2,3,3',4,4'-Pentachlorobiphenyl	$C_{12}H_5Cl_5$	105
2,3,4,4',5-Pentachlorobiphenyl	$C_{12}H_5Cl_5$	114
2,3',4,4',5-Pentachlorobiphenyl	$C_{12}H_5Cl_5$	118
2,3',4,4',5'-Pentachlorobiphenyl	$C_{12}H_5Cl_5$	123
3,3',4,4',5-Pentachlorobiphenyl	$C_{12}H_5Cl_5$	126
2,3,3',4,4',5-Hexachlorobiphenyl	$C_{12}H_4Cl_6$	156
2,3,3',4,4',5'-Hexachlorobiphenyl	$C_{12}H_4Cl_6$	157
2,3',4,4',5,5'-Hexachlorobiphenyl	$C_{12}H_4Cl_6$	167
3,3',4,4',5,5'-Hexachlorobiphenyl	$C_{12}H_4Cl_6$	169
2,3,3',4,4',5,5'-Heptachlorobiphenyl	$C_{12}H_3Cl_7$	189

Biochemical Toxicology

A subset of PCB congeners with chlorine constituents in the ortho positions of each phenyl group are restricted in ring rotation and thus cannot assume a co-planar structure. PCBs without ortho substitution or just mono ortho substitution can assume a co-planar structure. Co-planarity causes a PCB congener to resemble the polychlorinated dibenzo-p-dioxin known as TCDD (2,3,7,8-tetrachlorodibenzo-*p*-dioxin) and thus imparts increased toxicity. Specifically, the TCDD binds to the aryl hydrocarbon receptor (Ah) and induces transcription of the cytochrome P-450 A1A gene. Increased synthesis of cytochrome P-450 A1A is associated with a host of biochemical reactions leading to various toxicological effects. Dioxin-like PCBs also have the potential to bind to (Ah) and by extension, tend to be more toxic than non-coplanar PCBs (Safe 1994;Van den Berg et. al. 1998).

To assess potential cancer risks from exposure to mixtures of PCB congeners, toxic equivalency factors (TEFs) are used to quantify the potential for each congener to exhibit dioxin-like toxicity. TEF values are used in combination with chemical residue data to calculate TCDD toxic equivalent concentrations (TEQs) in a variety of environmental samples including animal tissue, soil, sediment, and water. Individual dioxin-like congeners are converted to TEQs by multiplying their initial concentrations by established Toxic Equivalency Factors (TEFs). TEQs are used in human health risk assessments in order to assess the impacts of a mixture of dixon-like compounds, including chlorinated dibenzodioxins (CDDs), chlorinated dibenzofurans (CDFs), and co-planer polychlorinated biphenyls (Van den Berg et. al. 1998).

The World Health Organization (WHO) has assigned TEFs to each of the dioxin-like PCB congeners to indicate their toxicity relative to 2,3,7,8-TCDD. TCDD itself has been assigned a TEF of 1.0. A PCB congener with a TEF of 0.01 is considered to be one hundred times less toxic than 2,3,7,8-TCDD. WHO first established the TEF values in 1994 for application to humans and mammals. In 1998 WHO expanded and revised the TEFs to include separate values for fish and birds as well as for humans and mammals(Van den Berg et. al. 1998). In 2006 WHO reevaluated TEFs for dioxins and dioxin-like compounds (Van den Berg et al. 2006) (Table 2).

Table 2. Current animal toxic equivalency factors for dioxin-like PCBs that have been evaluated by the World Health Organization (Van den Berg et al. 2006).

BZ/IUPAC Number	IUPAC Prefix	2005 WHO TEFs		
		Humans& Mammals	*Fish*	*Birds*
PCB-77	3,3',4,4'-Tetrachlorobiphenyl	0.0001	0.0001	0.05
PCB-81	3,4,4',5-Tetrachlorobiphenyl	0.0003	0.0005	0.1
PCB-105	2,3,3',4,4'-Pentachlorobiphenyl	0.00003	<0.000005	0.0001
PCB-114	2,3,4,4',5-Pentachlorobiphenyl	0.00003	<0.000005	0.0001
PCB-118	2,3',4,4',5-Pentachlorobiphenyl	0.00003	<0.000005	0.00001
PCB-123	2,3',4,4',5'-Pentachlorobiphenyl	0.00003	<0.000005	0.00001
PCB-126	3,3',4,4',5-Pentachlorobiphenyl	0.1	0.005	0.1
PCB-156	2,3,3',4,4',5-Hexachlorobiphenyl	0.00003	<0.000005	0.0001
PCB-157	2,3,3',4,4',5'-Hexachlorobiphenyl	0.00003	<0.000005	0.0001
PCB-167	2,3',4,4',5,5'-Hexachlorobiphenyl	0.00003	<0.000005	0.00001
PCB-169	3,3',4,4',5,5'-Hexachlorobiphenyl	0.03	0.00005	0.001
PCB-189	2,3,3',4,4',5,5'-Heptachlorobiphenyl	0.00003	<0.000005	0.00001

Sources

An estimated 2,000,000 tons of PCBs have been produced worldwide. A small fraction of these are still in use today in closed, highly regulated systems. Historical unregulated use of PCBs caused their widespread release and dissemination on land and water. Once released into the environment, PCBs bind intensively to soils and sediments but nevertheless partition from aqueous phases into air and thus continue to disperse worldwide. PCBs are deposited on vegetation and water surfaces by both precipitation and airborne particles. In addition to environmental releases of commercial PCB formulations, various congeners can be formed during incomplete combustion of organic materials burned naturally or during agricultural burning (Eckhardt et al. 2007).

The PCBs mainly bind to organic matter but have established an equilibrium with water that allows them to accumulate throughout the food chain, concentrating in the fatty tissues of birds, marine mammals, and fish (Alcock et al. 1994). For the general public worldwide, contaminated fish consumption is the major source of PCB exposure.

Environmental Fate

PCBs enter the environment as commercially produced Aroclors, but the compositional mixture of congeners changes over time. Transformation in congener composition occurs through weathering, environmental partitioning, chemical transformation, bioaccumulation, and various biodegradation processes (Callahan 1979). The persistence of PCBs lengthens as the degree of chlorination increases. For example, mono-, di- and trichlorinated biphenyls will biodegrade faster than tetra- and heptachlorinated biphenyls. However, persistence varies among the environmental compartments in which the PCBs are located (Table 3). Under laboratory conditions, bacteria that can metabolize PCBs to lower chlorinated congeners have been isolated, but under environmental conditions these organisms may not be functional, or alternatively, ambient conditions may not favor their transformation. Within Hudson River sediments a combination of aerobic and anaerobic bacteria could transform higher chlorinated PCB congeners to lower, less toxic forms (Abramowicz 1995; U.S. EPA 2000a). These bacteria exhibit oxidation-reduction reactions that reduce the chlorine content, which in turn would reduce the potential health risk associated with PCB contamination. Nevertheless PCB concentrations vary so greatly from one location to another that risk characterizations can be highly variable and uncertain.

Table 3. Half-lifes of PCBs in different environmental compartments. (Mackay et. al.1992.)

	Mean Half-life/Hours Air	Mean Half-life/Hours Water	Mean Half-life/Hours Soil	MeanHalf-life/Hours Sediment
Dichloro-PCB	170 (~1 week)	5,500 (~8 months)	17,000 (~2 years)	17,000 (~ 2 years)
Heptachloro-PCB	5,500 (~8 months)	55,000 (~6 years)	55,000 (~6 years)	55,000 (~6 years)

Health Concerns

Health concerns from PCBs arise from their prolonged persistence in the environment and within organisms, and thus the life-long duration of exposure in human populations. Because PCBs consist of many different compounds that have varied toxicological endpoints in addition to physiochemical properties, assessment of health effects are not as straight-forward as for well defined contaminants like methyl mercury. However, PCBs are perceived to pose an equal or greater risk than methyl mercury to exposed populations. Epidemiological studies of PCBs in electrical capacitor manufacturing workers have found increased rates of melanomas, liver cancer, gall bladder cancer, biliary tract cancer, gastrointestinal tract cancer, and brain cancer (Johnson 1999). PCBs could also cause developmental effects in children whose mothers were exposed to PCBs before or during pregnancy. These effects have included significant neurological and motor control problems, comparatively lower IQ scores, and poor short-term memory at the highest levels of maternal exposure (Jacobson and Jacobson 1996).

The Environmental Protection Agency (EPA) predicted a significant increase in cancer among people eating certain species of fish from specific locations within the U.S. For example, during the Hudson River PCB Reassessment study, EPA scientists concluded an increased cancer risk as high as 1 in 2500 (4×10^{-4}) from the consumption of contaminated fish. The projected increased risk is nearly a hundred times higher than the level set by the EPA for the protection of human health (i.e., a risk of 1×10^{-6}) (U.S. EPA 2000b).

PCBs in Fish

The results of many studies throughout the country have demonstrated levels of environmental contaminants in a variety of fish species that suggest excessive health risks. In a global approach, Hites et al. (2004) assessed the possible human impacts from various organic contaminants, including PCBs, in farmed and wild salmon from around the world. They concluded that the contaminants were higher in farm-raised salmon from the Atlantic Ocean than in wild stocks of the Pacific Northwest. Using the methodology recommended by the EPA for assessing carcinogenic risk from fish consumption, Hites et. al. (2004) concluded that farmed salmon could pose a significant health risk, which could offset any beneficial nutritional aspects of eating fish.

Extending the work of Hites et. al. (2004), Foran et. al. (2005) used a risk based approach to assess cancer and non-cancer health risks from exposure to dioxins and dioxin-like compounds in farmed and wild salmon stocks. Their analysis was based on acceptable intake levels of dioxin-like compounds established by the World Health Organization (WHO) and with risk estimates for exposure to dioxins developed by the EPA. Foran et. al. (2005) concluded that consumption of farmed salmon, even at low rates, could increase dioxin exposure sufficiently to increase the risk of adverse human health effects.

Other fish tissues in the Great Lakes area have declined in total PCBs levels over time (Hebert et. al. 1999, Jackson et. al. 1998, Madenjian et. al 1999), but most studies have not examined individual PCB congeners. In the mid 1990's, research by Jackson et. al. (2001) looked at PCBs congeners and their trophic level concentrations. He concluded that PCBs in fish and macroinvertbrates are closely related and show concentration differences of 20 to 30 times between lower and higher trophic levels. This research helps further our understanding of how complex contaminant mixtures act and flow through the environment and potentially impact human health.

Within Washington state between 2001 and 2005, 150 fish tissue samples where collected from over 70 sites and were analyzed for various contaminants, including PCB's (Seiders and Kinney 2004, Seiders et. al. 2006, 2007). Even after recent field sampling of fish and testing for toxic contaminants by federal and state agencies, many areas in Washington have not been assessed. Thus, more information is needed to characterize the levels of toxic contaminants within freshwater fish and their potential impact on health of humans and wildlife.

Efforts have increased to sample more sites within eastern Washington and to expand analysis to include PCB contaminants. In 2006 and 2007 the Washington State Toxics Monitoring Program (WSTMP) issued reports listing monitoring sites located in the eastern part of the state. From those studies, only a small number fell within the Columbia Basin Irrigation Project. These sites included Banks Lake, Crab Creek, Moses Lake, Potholes Reservoir, and Scooteney Reservoir. Although the locations represent only a small portion of the CBIP, the results should help in the understanding of the full extent of PCB levels within the irrigation project. Of the sampling sites listed above, PCBs in fish tissue from Moses Lake, Scooteney Reservoir, and Potholes Reservoir were high enough in 2004 to trigger an "impaired" listing (http://apps.ecy.wa.gov/wats08/).

Fish Consumption Advisories

The occurrence and persistence of PCBs in fish tissue is a world-wide problem. Within the United States contaminants in tissues of sport-caught fish have resulted in many consumption advisories. The main focus of these advisories is to alert local residents of potential health risks from eating contaminated fish caught in their local water. In 2002, the number of advisories reached 3,089 mostly because of mercury and PCB contamination (U.S. EPA 2003). In 2004, 3,221 advisories were issued (http://www.epa.gov/fishadvisories/advisories/2004/fs2004.pdf). Within Washington State, most advisories prior to the 1990's focused mainly on mercury contamination with comparatively few attributed to PCBs (McBride 2008).

All 50 states have fish-advisory programs to some degree with each state setting its own criteria for when a consumption advisory should be issued and for what content the advisory may include (Scherer et. al. 2008). Some may have specific listings that include coastal waters, rivers or lakes, while others will include a combination of all three. The content could also be different between advisories with some including recommendations to limit or avoid eating specific kinds of fish from specific water bodies, while others target sensitive population groups like pregnant women, nursing mothers, and children (McBride 2008). These inconsistencies likely result from fish advisories being completely voluntary state recommendations that are not regulated by the Federal government. Although most advisories are issued by individual state agencies, the EPA does provide guidance.

Little information about PCBs in fish tissue was collected in Washington during the mid 1980's through the early 1990's. It was not until 2000 that monitoring of toxic chemicals in freshwater environments increased and included sampling fish tissue, sediments, water, and wildlife in Washington State. From this increased effort two fish advisories were issued within the past seven years in the Puget Sound region. Currently, the Washington State Department of Health (DOH) lists 12 site-specific consumption advisories for finfish and shellfish due to PCB's and mercury (Figure 3).

Figure 3. Washington State Fish Consumption Advisory Locations 2008.
(http://www.doh.wa.gov/ehp/oehas/fish/advisoriesmap.htm)

In Washington State the Department of Health (DOH) works in conjunction with the Washington Department of Ecology (WDOE) in collecting and conducting the assessments needed to help identify possible risks to human health. To help identify problem areas, WDOE uses the National Toxics Rule and those screening values as a base for possible human health problems from contaminants detected in fish tissue.

Washington's water quality standards criteria for toxic contaminants were issued to the state in EPA's 1992 National Toxics Rule (NTR) (40CFR131.36). The human health-based NTR criteria are designed to minimize the risk of effects occurring to humans from chronic (lifetime) exposure to substances through the ingestion of drinking water and consumption of fish obtained from surface waters. The NTR criteria are thresholds that, when exceeded, may lead to regulatory action. When water quality criteria are exceeded, the federal Clean Water Act requires that the waterbody be put on a 303D list and that a water cleanup plan (also know as a TMDL) be developed for the pollutant causing the problem. Ecology uses this TMDL program to control sources of a particular pollutant to bring the waterbody back into compliance with the water quality standards.

While DOH supports Ecology's use of the NTR criteria for identifying problems and controlling pollutant sources so that water quality standards are met, DOH does not use the NTR criteria to establish fish consumption advisories (McBride 2008). For assessing mercury, PCBs, and other contaminants, DOH uses an approach similar to that in EPA's *Guidance for Assessing Chemical Contaminant Data for use in Fish Advisories Vol. 1-4* (U.S. EPA 2000g). These guidance documents provide established procedures that states can use to evaluate fish tissue data and help in the development of fish consumption advisories. The methodology recommended by EPA follows the risk analysis paradigm that encompasses risk assessment, risk management, and risk communication.

The EPA risk analysis paradigm has the potential for overestimating risks from contaminants in fish that could potentially scare the general public from consuming fish and ultimately reduce the associated health benefits (Mozaffarin and Rimm, 2006). Fish are a good source of omega-3 fatty acids that can help lower blood presser and heart rate, and improve cardiovascular condition. Risks associated with heart disease, the leading cause of death in both men and woman, can also be reduced from the consumption of fish. Pregnant women, mothers who are breastfeeding, and women of child bearing age, could benefit from fish consumption that supplies DHA, which is a specific omega-3 fatty acid that is beneficial for infant brain development (Mozaffarin and Rimm, 2006).

To assess potential risks of PCBs to human health, advisories issued by most DOH regions are based on deterministic or point estimates of exposure that could vary greatly from one estimate to the next. For example, when many assessments are conducted the procedure is to take a “worse case” approach focusing on the high end of the exposed population and also use high-end values for physical parameter variables like body weight, exposure duration, and ingestion rate of fish tissue consumed. This approach results in estimated exposures that conservatively attempt to address both variability and uncertainty with the end result being very conservative (WHO 1999). Thus, most assessments conducted within Washington State are likely to be over protective then under protective when issuing State fish consumption advisories. However, any characterization of risk could be misleading because the probability of a broad range of data possibilities does exist. To help assess inherent uncertainties created by variability in the input parameters, which is inherent to most risk assessment data, a probabilistic approach to exposure is needed.

Probabilistic modeling is increasingly being used to assess the variability and uncertainty associated with human exposure to contaminants via different exposure routes. (Schleier et al. 2008, Dinkins 2008, Maddalena 2004). Probabilistic techniques based on Monte Carlo analysis are time consuming and expensive, so they are not utilized very often. The main difference between a deterministic and probabilistic exposure assessment is the estimations by the latter of range and likelihood of specific events. These events, also called input parameters, are then addressed as exposure factors which are defined as distributions instead of specific point estimates like the deterministic approach. A probabilistic approach will enable the characterization of a broad range of exposure scenarios, including a prediction of the uncertainty associated with the range of PCB concentrations generated from fish samples collected within the CBIP.

Purpose of the Study

As a result of worst-case deterministic exposure assessments, it is hypothesized that PCB levels in fish from the CBIP could pose a significant risk to human health. The purpose of this paper is to characterize PCB concentrations in fish tissue and human exposure by subjecting existing available data to probabilistic modeling.

Methods and Materials

Problem formulation

Probabilistic one and two-dimensional modeling employing Crystal Ball® software is used to assess the potential carcinogenic and non-carcinogenic effects of PCBs to two population groups consuming contaminated fish collected in aquatic systems under the jurisdiction of the CBIP. First, a sensitivity analysis is used to determine the possible impacts of each input variable on the estimated output values and focus attention on the input values of most importance. Second, one-dimensional modeling is used to simulate probabilistic exposure and characterize risk of adverse effects. Third, two-dimensional analysis is applied to the input datasets so that uncertainty from variations in chemical concentrations can be separated from variability (i.e. range in exposure durations, ingestion rates, body weights, etc.) and thus improves the estimation of risk.

For many of these types of assessments it is important to clearly distinguish between uncertainty and variability. Separating these concepts in a simulation lets the researcher more accurately detect variations in a forecast due to lack of knowledge and the variation caused by natural variability in a measurement or population. The primary output of the two dimensional process is a chart depicting a series of cumulative frequency distributions that describe the possible range of risks associated within the population in question.

This assessment examines exposure to two demographic groups based on age-related differences--combined adult male and females (18-65 years) and children (<15 years). Within these groups three scenarios, Children, Adults from the Lake Roosevelt area, and Adults throughout the Columbia Basin, were analyzed using different ingestion rates adopted from two previously published human health risk assessments relevant to the Columbia Basin (EPA 2002b; Patrick 1997). Use of these ingestion rates reduces the uncertainty from the lack of data on angler activity and eating habits within the defined study area.

Site Selection

Sampling sites included those within the jurisdiction of the CBIP and the Columbia Basin National Wildlife Refuge. The criteria used in site selection were accessibility, potential for recreational use, and fish species present. Fish were collected at 12 sites: Banks Lake, Billy Clapp Lake, Crab Creek, Frenchmen Hills wasteway, Lind Coulee wasteway, Moses Lake, Potholes Reservoir, Royal Lake, Saddle Mountain Lake, Scooteney Reservoir, WB10 wasteway lakes, and Winchester wasteway (Figure 4). In addition, PCB data from fish samples collected by State and Federal agencies, including the Washington Department of Ecology (WDOE), Washington Department of Fish & Wildlife (WDFW), and the U.S. Environmental Protection Agency (EPA), were included in exposure characterization. The other governmental monitoring studies had sampled fish from some of the same 12 CBIP sites. Combining PCB data from different studies increased the sample size and helped define the variability in residue data.

Field activities

The three irrigation districts that make up the CBIP and the United States Bureau of Reclamation (USBR) contracted with the United States Fish and Wildlife Service (USFWS) in Spokane, WA, to coordinate the fish sampling effort. Field activities took

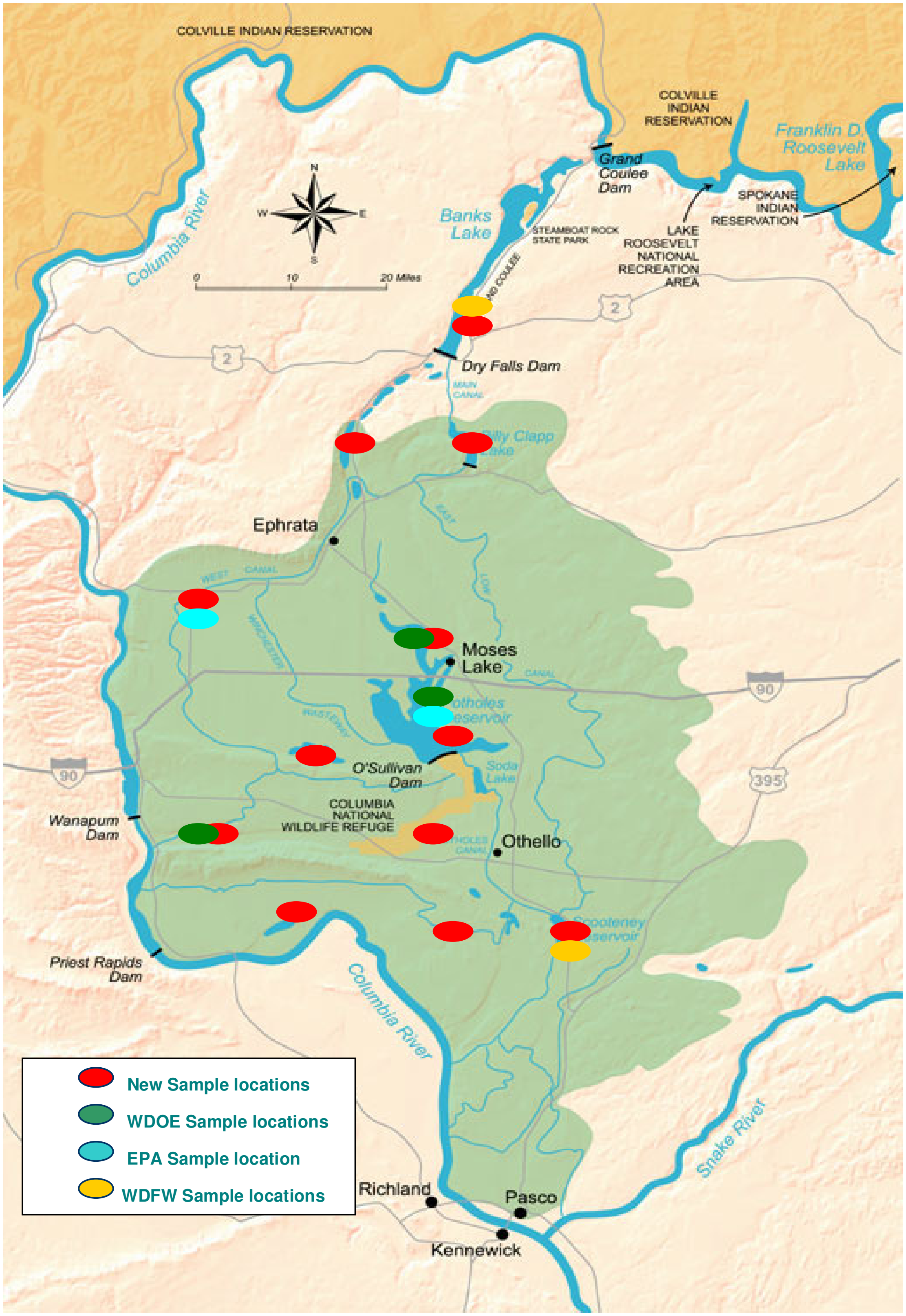

Figure 4. Fish sampling locations for the CBIP monitoring study and studies conducted by the WDOE and EPA.

place in the Fall of 2005 and 2006 with the target species being common carp (*Cyprinus carpio*), largemouth bass (*Micropterus salmoides*) and smallmouth bass (*Micropterus dolomieu*). These species where selected because they are common in all of the site locations and are the most popular target fish for local anglers.

Fish Capture

Site-specific collection methods were pre-determined according to shoreline habitat, access, bottom substrate, and depth contours along the shore at each site. At Saddle Mountain Lake, the largemouth bass and common carp were collected by hook and line and electro-fishing. In the WB10 wasteway ponds, largemouth bass were collected by hook and line, and the common carp were collected using electro-fishing and bow fishing. Largemouth bass were collected in Royal Lake by hook and line, with common carp collected by bow fishing. Bow fishing for common carp was the selected sampling method for all other sites, except Bank's Lake, where electro-fishing was the preferred method used for capture.

Sample Collection and Preparation

Samples of fish tissue were collected using EPA methods for sampling and analysis (EPA 2000b). At each site, composite samples of equal numbers of fish of the same species with similar size were collected. The fish in each sample were weighed, and total length (TL) was measured. Each fish was also examined externally for any abnormalities using procedures outlined by the United States Geological Survey (USGS) (Schmitt et al. 1999). Scales were also collected from each fish to determine age at a later date if needed. All fish samples were labeled and kept frozen until shipped to the laboratory for analysis. Project specific data forms were completed for all fish after each sample was tagged with information identifying sample site, sample number, sample composite number, date, and species. Separate data forms were filled out for each individual fish noting any external anomalies along with the specific information for that individual.

Laboratory methods

All tissue samples collected were processed through the USFWS Analytical Control Facility (ACF), Sheperdstown, West Virginia and then sent to various approved commercial laboratories. The whole bodies of common carp and bass fillets were analyzed for organochlorine (OC) pesticides and PCBs with metals being analyzed only in the bass fillets (Table 4). PCB analysis included identification and quantification of Aroclors 1242, 1248, 1254, 1260, 1268 and 159 individual congeners.

Whole common carp were sent to the Geochemical and Environmental Research Group (GERG) at the Texas A&M Research Foundation, College Station, Texas, for analysis of OC compounds, which included PCBs. The bass fillets were sent to the Trace Element Research Laboratory (TERL) at Texas A&M Research Foundation for metals analysis. PCB Aroclor analyses in samples collected by the WDOE were completed by the agency's Manchester Environmental Laboratory (MEL). Samples for PCB congener analysis were sent to Pacific Rim laboratories, Inc. in Surry, B.C., Canada. Table 5 describes the analytical methods and detection limits applicable to the fish tissues.

Table 4. List of Fish Tissue Analytes.

Organochlorines	PCBs	Metals
Aldrin	PCB-1242	Silver (Ag)
Alpha-hexachlorocyclohexane (Alpha BHC)	PCB-1248	Boron (B)
Alpha chlordane	PCB-1254	Barium (Ba)
Beta-hexachlorocyclohexane (Beta BHC)	PCB-1260	Beryllium (Be)
Chlorpyrifos	PCB-1268	Cadmium (Cd)
Cis-nonachlor	Congeners**	Cobalt (Co)
Delta-hexachlorocyclohexane (Delta BHC)		Chromium (Cr)
Dieldrin		Copper (Cu)
Endosulfan II		Iron (Fe)
Endrin		Mercury (Mg)
Gamma-hexachlorocyclohexane (Gamma BHC)		Magnesium (Mg)
Gamma chlordane		Aluminum (Al)
HCB		Arsenic (Ar)
Heptachlor		Magenese (Mn)
Heptachlor epoxide		Molybdenum (Mo)
Mirex		Nuckel (Ni)
o,p'-DDD		Lead (Pb)
o,p'-DDE		Selenium (Se)
o,p'-DDT		Strontium (Sr)
Oxychlordane		Titanium (Ti)
p,p'-DDD		Vanadium (Va)
p,p'-DDE		Zinx (Z)
p,p'-DDT		Phosphorus (P)
Pentachloro-anisole		Sulfur (S)
Trans-nonachlor		
1,2,3,4 Tetrachlorobenzene		
1,2,3,5 Tetrachlorobenzene		

** Only 159 PCB congeners analyzed

Data Collection

Analytical results for the PCBs in fish tissues from all the monitoring programs were combined in a Microsoft Excel database. In addition to results from the CPIB study data were also extracted from the *National Study of Chemical Residues in lake Fish Tissue* (EPA 2005) and WDOE (Hopkins 1991, Serdar et. al. 1994, Davies et. al. 1998, Seiders et. al. 2006, and Seiders et. al. 2007). Five sites from these additional sampling programs matched those from the CBIP study--Banks lake, Moses Lake, Crab Creek, Frenchman Hills Wasteway, Potholes Reservoir, and Scooteney Reservoir. The species of fish varied among the studies, but all fish collected are potential targets of anglers for food consumption and recreation.

Table 5. Analytical methods for all fish tissue samples collected by CBIP personnel.

Parameter	Description	Method	Reporting Limit
PCB Aroclors	GC/ECD[1]	EPA 8082	0.5 µg/kg, wet wt.
PCB Congeners	HI Res GC/MS[2]	EPA 1668A	0.02-0.08 µg/kg, wet wt.
Chlorinated pesticides	GC/ECD[1]	EPA 8081	0.25-15 µg/kg, wet wt.
Mercury (total mercury)	CVAA[3]	EPA 245.6	0.017 mg/kg, wet wt.
Lipids-percent	gravimetric	Mel SOP 700009	0.1 percent

[1] (GC/ECD) Gas Chromatography With Electron Capture Detection.
[2] (HI Res GC/MS) High Resolution Gas Chromatography/ Mass Spectrometry.
[3](CVAA) Cold Vapor Atomic Absorption.

Quality Control of Data

PCB data were screened to establish consistency and comparability between results from the different agencies that produced the information. For PCB congener data used to calculate total PCBs, results below detection limits were treated as a no detection (ND) and assigned a value of zero. PCB congeners detected at the detection limit were divided by two, and the resulting concentration was used to calculate total PCBs present in that sample. If the final total PCB concentration equaled zero, the detection limit was used for the final result for that sample site (EPA 1989, 1990, 2002d). These same procedures were also applied to PCB homolog and Aroclor data that were used to calculate total PCBs. Statistical means, standard deviations, and ranges were calculated for total PCBs (homolog, congener, and Aroclors) for all sample sites. These transformed data were used to establish the distribution factor applied to the total PCBs concentrations estimated in the Monte Carlo analysis of exposure.

Exposure Assessment

Ideally, exposure assessment should consider all possible sources of PCB residues, including air, water, soil, and biota. However, EPA's health risk assessment for PCBs in the Hudson River concluded that the major pathway of concern for PCB exposure was through fish consumption, and all other possible pathways had little impact (EPA 2000a). This conclusion is generally supported in other reviews of PCB risk assessments (Shields 2006). Assessment of exposure required integration of data about fish consumption habits by different demographic groups with data on PCB concentrations in fish. Thus, fish tissue residues and consumption data were incorporated to estimate exposure.

Little information had been collected on fish consumption behavior for populations within the area of the CBIP. However, the WDOH measured consumption pattern of anglers in the Lake Roosevelt area. Because of the proximity of this population to the CBIP, these data were used as the basis for characterizing distribution of fish consumption patterns.

<u>Exposure Calculations</u>

To assess for possible cancer and non-cancer risks associated with fish consumption within the CBIP a probable total daily exposure from fish consumption was calculated. An exposure equation was developed for estimating the Average Daily Dose (ADD) for non-cancer estimates and a Lifetime Average Daily Dose (LADD) for cancer estimates (EPA 2000b, EPA 2002b).

The exposure function included all plausible independent variables that could affect body dose of PCBs and were calculated using the equation below.

$$(L)ADD = \frac{C \times CF \times IR \times EF \times FR \times ED}{BW \times AT} \qquad \text{(Eq. 1)}$$

where:

(L)ADD = (Lifetime) Average Daily Dose of total PCBs from CBIP Fish tissue (mg/kg-day)
C = Chemical concentrations in fish tissue (mg/kg)
CF = Conversion factor (kg/g)
IR = Ingestion (consumption) rate (g/day)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
FR= Fraction of PCBs remaining after cooking (i.e. cooking loss factor
BW=Bodyweight (kg)
AT = Averaging time for exposure duration (days)

Assumptions

The magnitude and assumed distribution of variables in Eq. 1 that were subjected to Monte Carlo analysis are defined in Table 6. Total PCB concentrations were used to predict cancer risk instead of the TEQ method. The TEQ method could not be applied because of lack of available data in the PCB database (EPA 2000f). Concentrations in the WDOE database only expressed results as total Aroclors with no individual congener information.

Total PCB concentrations in sampled fish ranged from 0.00031 mg/kg (0.31 ppb) to 0.051 mg/kg (51ppb). The variability of PCB levels in fish from the CBIP was assumed to represent a log-normal distribution with the mean of the measured residues equivalent to the mean of the distribution. Assuming that estimates of the true mean were normally distributed, the standard error of the analytical data was used as the standard deviation for the distribution.

Table 6. Assumptions used in Monte Carlo analysis for exposure to PCBs through consumption of fish tissue.

Parameter	Acronym	Distribution	Distribution Parameters	Reference
Concentration	C	Log-Normal[1,2]	Minimum=0.000319	[1]USEPA 1997a
(mg/kg)			Mean= 0.0129	[2]VanLandingham et al 2004
			Std=0.0143	
			Maximum=0.051	
Ingestion rate	IR	Log-Normal[1,3,4]	Adult 18-65	[1]EPA 2002b
(g/day)		SD (+-) 0.75	Mean=7.5 (0-142.4)[1]	[2]WDOH 1997
		SD (+-) 4.2	Mean=42 (0-90)[2]	[3]EPA 1997d
		SD (+-) 1.56	Children<15[1,3]	[4]ODEQ 1998
			Minimum=0.0	
			Mean=2.83	
			Maximum=77.95(99th)	
Exposure frequency	EF	Triangular[1,2]	Adult/Child[1,2]	[1]EPA 2002b
(day/yr)			Minimum=0.0	[2]EPA 1997a-c
			Mean=182.50	
			Maximum=365	
Exposure duration	ED_{carc}	Triangular[1,2]	Adult (Carcinogenic)[1,2]	[1]EPA 1997a-c
(yrs)			Minimum=0.0	[2]ODEQ 1998
			Mean=35	
			Maximum=70	
	$ED_{non\text{-}carc}$	Triangular[1,2]	Adult(non-Carcinogenic)[1,2]	
			Minum=0.0	
			Mean=15	
			Maximum=30	
	ED_{Child}	Triangular[1]	Children[2]	[1]EPA 2000e
			Minimum=.5	[2]EPA 1997c
			Mean=7.5	
			Maximum=15	
Body weight	BW	Log-Normal[1,2,3]	Adult[2,3]	[1]EPA 1997a
(kg)			Minimum= 96	[2]Finely,et al. 1994
		SD (+-)2.5	Mean= 98.5 (95th %)[4]	[3]ODEQ 1998
			Maximum=101	[4]EPA 2002b
		Log-Normal [1,2,3]	Children[2]	[1]EPA 1997c
		SD (+-) 2.30	Minimum=11.65	[2]Schleier III, et al. 2008
			Mean=23.03	[3]ODEQ 1998
			Maximum=43.28	
Cooking loss	FR	Normal	Minimum=0.0	[1]Zabik, et.al. 1995
%			Mean=0.68 [1,2]	[2]Wilson et.al 1998.
			Maximum=1.00	
Averaging time	AT_{carc}	Triangular[2]	Mean=12775 (0-25550)[1]	[1]EPA 2002b
(days)	$AT_{non\text{-}carc}$	Triangular[2]	Mean=5475 (0-10950)[1]	[2]ODEQ 1998
	AT_{Child}	Triangular[2]	Mean=2737.50 (0-5475)[1]	

For this assessment scenario, PCB levels were conservatively assumed to remain stable throughout a person's life. PCB residues were adjusted by cooking loss (FR), which can range from 0-100% depending on fish species, cooking method, and if skin was left on or off (Zabik et. al. 1995 and Wilson et. al. 1998). Cooking loss (FR) was assigned a normal distribution with a mean of 68%.

The major population of concern were anglers that use the local lakes and waterways for recreational and subsistence fishing. Within the CBIP, however, specific use patterns by anglers have not been characterized. Therefore, fish consumption rates for recreational fishers were adopted from the Lake Roosevelt anglers survey (WDOH) and the Columbia Basin Fish Contaminant Survey (EPA 2002b). Average and high-end consumption rates were 42 and 90 g per day, respectively, for Lake Roosevelt anglers compared to the mean and maximum consumption rates of 7.5 g and 142 g per day, respectively, for the general public (EPA 2000b). Ingestion rate was assigned a log normal distribution (ODEQ 1998) for both adult and children under 15 years of age (Table 6). Exposure frequency (EF) and duration (ED) were based on EPA recommendations (1997b, 1997c) for exposure assessment of children and adults (for both carcinogenic and non-carcinogenic endpoints). EF and ED were modeled with a triangular distribution using means calculated from the EPA data.

Body weight (BW) for adults (male and female) was calculated using data from EPA (1989) and from Finley et. al. (1994), and it ranged from 62 kg to 101 kg with a median of 70 kg and a 95^{th} percentile of 99 kg. Weights for children ranged from 12 kg to 23kg (EPA 1997c). For both children and adult BW was assigned a log-normal distribution (Shleier 2008). For all scenarios, averaging time was assigned a triangular distribution using data from both EPA (2002b) and ODEQ (1998).

Risk Characterization

Carcinogen

Cancer risk is assessed using the cancer slope factor, which is an upperbound estimate of excess lifetime cancer risks per unit dose or exposure to a carcinogen. The units are $(mg/kg\text{-}day)^{-1}$. The highest slope factor for ingestion exposure (i.e. non comercial fish consumption and fish advisories) is 2.0 per mg/kg-day, which was used in this study. Cancer risk was calculated using the equation below.

$$\text{Cancer Risk} = \text{ADD} * \text{PCB Slope Factor} \quad \text{(Eq. 2)}$$

Where:

ADD = Average daily dose (mg/kg-day) of PCBs in fish tissue.
PCB Slope factor = 2/mg/kg-day

Cancer risk was classified as probable if the modeled result was greater than 10E-06 (EPA 1996).

Non-Carcinogen

Risk of non-cancer health effects assumes an exposure threshold below which adverse effects are unlikely to occur. In this assessment, the evaluation of non-cancer health effects compares average daily exposure (ADD) to PCBs in fish tissue with the EPA reference doses (RfD) of 0.00002 mg/kg/day. The reference dose is an estimate of the daily exposure to a chemical having a reasonable certainty of causing no observable adverse effects. The RfD is estimated as the no observable adverse effect level (NOAEL) for a defined toxicological endpoint divided by an uncertainty factor of 100. Potential health risks from non-cancer effects for a specific chemical are expressed as a hazard quotient (HQ), which is the ratio of the calculated exposure relative to the RfD for PCBs. The equation for HQ is:

$$HQ = \frac{ADD}{RfD} \qquad \text{(Eq. 3)}$$

Where:

HQ = Chemical-specific hazard quotient (unitless)
ADD = Average daily dose (mg/kg-day)
RfD = Chemical-specific oral reference dose (mg/kg-day)

Both the estimated average daily doses from consuming fish and the RfDs are expressed in units of amount (in milligrams) of a chemical ingested per kilogram of body weight per day (mg/kg-day). An HQ of >1.0 would mean that the estimated exposure is greater than the RfD and could pose a significant health risk (EPA 2002b).

Probabilistic Analysis

A Monte-Carlo simulation (Crystal Ball ® 7.3.1; Decisioneering, Denver, CO.) was used to evaluate HQ and cancer risk. The (L)ADD was calculated from the various input variables listed in Table 6. Each input variable was randomly sampled by the software enough times to reproduce the shape of the distribution of the statistical outputs. Each input was assigned to categories of either variability or uncertainty for later two-dimensional analysis. The modeled output distribution of each input variable was substituted into the (L)ADD equation to calculate a probabilistic distribution of exposures. For each model iteration, the output (L)ADD was used to calculate risk, resulting in a distribution of cancer risks and HQs.

Sensitivity Analysis

Sensitivity analysis was performed using a one-dimensional probabilistic analysis with 20,000 iterations for the (L)ADD calculation parameters. Sensitivity analysis showed which parameters contributed the most to the overall exposure. For one dimensional exposure analysis, the input variables were sampled 20,000 times each to create one final distribution, and to achieve a more stable final exposure distribution.

Two-Dimensional Analysis

Two-dimensional analysis was performed using 10,000 iterations of variability and 100 iterations of uncertainty to calculate the median and standard deviation (SD) at 95% confidence for the estimate of risk for both cancer and non-cancer endpoints. For the two-dimensional analysis body weight, ingestion rate, exposure frequency, exposure duration, averaging time, and cooking loss factor were placed in the variable category. Uncertainty was only assigned to the fish PCB residues.

Results

CBIP Field Sampling

A total of 59 fish were collected between the 2005 and 2006 season. Among the 12 sites sampled the mean lengths and weights for common carp (*Cyprinus carpio)* ranged from 397-711 mm and 860-5988 g, respectively (Table 7). For largemouth bass (*Micropterus salmoides*) and smallmouth bass (*Micropterus dolomieu*), the mean lengths ranged from 209 - 341 mm, and the mean weights ranged from 213-584 g (Table 8).

Table 7. Mean and ranges of lengths and weights of whole body carp composites (n=5) from the Columbia Basin Irrigation Project field collection 2005-2006.

Sample location	Range of Lengths (mm)	Mean length (mm)	Range of weights (g)	Mean Weight (g)
Banks Lake (BLK)	379-717	521	804-5221	2393
Billy Clapp Lake (BCL)	488-623	564	1670-3700	2714
Moses Lake (MOL)	573-833	672	3100-9100	5110
Lind Coulee Wasteway (LND)**	585-680	640	1589-3895	2696
Winchester Wasteway (WWW)	510-602	575	1814-3400	2744
Frenchman Wasteway	505-550	524	1180-2268	1792
Potholes Reservoir (PRS)	554-636	592	2497-3759	3063
Scootenay Reservoir (SNR)	553-615	584	2800-3000	2880
Crab Creek (CCL)	361-519	445	681-1589	1135
Royal Lake (ROY)	577-816	711	2723-11340	5988
WB10 Wasteway (WBW)	370-418	397	601-999	860
Saddle Mountain Lake (SML)	614-726	654	3500-5080	4235

** Only four carp were in this composite sample

Table 8. Mean and ranges of lengths and weights of bass fillet composites (n=5) from the Columbia Basin Irrigation Project field collection 2005-2006.

Sample location	Range of Lengths (mm)	Mean length (mm)	Range of weights (g)	Mean Weight (g)
Royal Lake (ROY)	136-242	209	172-252	213
WB10 Wasteway (WBW)	24-366	341	500-690	584
Saddle Mountain Lake (SML)	233-328	299	220-690	522

PCB congeners were detected in fish tissue at every sample location. Of the six possible PCB Aroclors, only 1254 and 1260 were detected in the common carp samples but none was identified in the bass tissue. Aroclor 1254 was detected in eight of the common carp samples, and Aroclor 1260 was detected in seven of the samples.

Forty-two of the 159 PCB congeners included in the analytical method were detected in common carp, and no PCB congeners were detected in bass tissue. One “dioxin-like” PCB congener (PCB-105) was detected in seven carp samples with the highest concentration from Moses Lake. No “dioxin-like” congeners were detected in bass samples. The Billy Clapp Lake common carp sample had the highest number (32) of PCB congeners detected. The carp sample from Crab Creek had only two congeners (PCB #132/153/168 and 170/190) (Table 9). The Moses Lake and Banks Lake samples each contained 30 and 23 PCB congeners, respectively, and the Winchester wasteway sample had four congeners. The remaining six samples had between 9 and 21 PCB congeners above analytical detection limits.

PCB Concentration vs. Body Weight of Fish

In order to determine if there was a linear relationship between PCB concentrations in the samples collected and the mean body weights of fish, regression analysis was employed. Figure 5 shows the scatter plot between PCB concentrations and

Table 9. Number of PCB congeners detected in common carp tissue at each sample location during the 2005-2006 Columbia Basin fish tissue sampling.

Sample Location	# of PCB Congeners Detected [1]
Billy Clapp Lake (BCL)	32
Banks Lake (BLK)	23
Crab Creek (CCL)	2
Frenchman Wasteway (FRW)	14
Lind Coulee Wasteway (LDW)	9
Moses Lake (MOL)	30
Potholes Reservoir (PHR)	17
Royal Lake (ROY)	19
Saddle Mountain Lake (SML)	15
Scooteney Reservoir (SNR)	21
WB10 Wasteway (WBW)	0
Winchester Wasteway (WWW)	4

[1]PCBs residues that could not be completely resolved by the analytical method used were grouped together and counted as one congener.

body weights of fish. The PCB concentration variable was log transformed because of the skewness of the data. The regression model showed a slight correlation (r^2=0.27, p=<0.05) even though the sample size was so small (n=9) and the outliers where taken out after performing the kurtosis test for outliers.

The most frequently detected congeners in common carp were PCB 101/113, 132/153/168 and 138/158 (n=12 samples). Of the total PCB congeners detected seven congeners were detected in only one sample each. Of those seven congeners detected in only one sample, three of the congeners (128/162, 141,178) were detected only in fish from Banks Lake (Table 10).

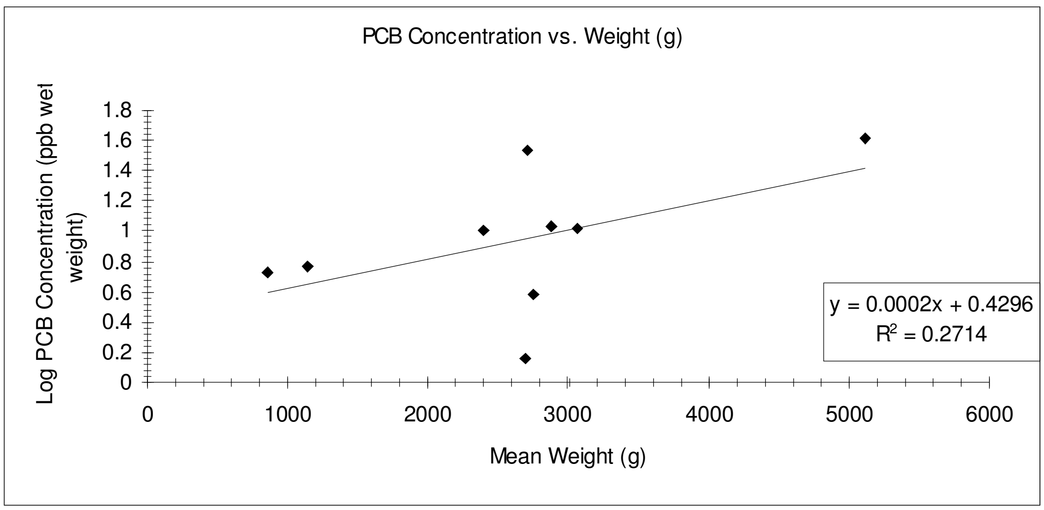

Figure 5. Regression line for log PCB concentration and mean weight.

Table 10. Means (ppb [µg/kg] wet weight), ranges, and number of detections of PCBs in common carp from the Columbia Basin Project during 2005 and 2006 (n=12 sampling locations).

PCB Congener	Mean	Range	Number of detections
Aroclor-1254	8.40	<1.58-37.10	8
Aroclor-1260	3.31	<1.58-11.31	7
PCB# 28/31	0.35	<0.062-1.24	6
PCB# 42	0.06	<0.079-0.20	1
PCB# 43/52	0.55	<0.079-2.42	7
PCB# 44	0.13	<0.079-0.63	3
PCB# 47/48/62/65/75	0.13	<0.079-0.58	3
PCB# 49	0.26	<0.079-0.94	7
PCB# 63	0.13	<0.079-0.58	2
PCB# 66	0.38	<0.079-2.07	3
PCB# 70	0.37	<0.079-2.01	4
PCB# 74	0.20	<0.079-1.07	2
PCB# 83/109	0.07	<0.079-0.17	2
PCB# 84/92	0.26	<0.079-1.16	6
PCB# 87/111	0.36	<0.079-1.70	8
PCB# 88/95	0.09	<0.079-0.36	3
PCB# 91	0.08	<0.079-0.48	1
PCB# 99	0.71	<0.079-2.73	9
PCB# 101/113	1.13	<0.079-4.93	10
PCB# 105	0.21	<0.079-0.86	7
PCB# 110	0.53	<0.079-3.20	8
PCB#128/162	0.06	<.079-0.2	1
PCB# 112/119	0.06	<0.079-0.17	2
PCB# 132/153/168	0.88	<0.079-4.26	11
PCB# 135	0.06	<0.079-0.27	2
PCB# 138/158	0.67	<0.079-3.69	10
PCB# 139/149	0.35	<0.079-1.79	8
PCB#141	0.05	<0.079-0.11	1
PCB# 144	0.05	<0.079-0.14	2
PCB# 160/163/164	0.20	<0.079-1.01	8
PCB# 170/190	0.14	<0.079-0.42	10
PCB# 174/181	0.07	<0.079-0.29	4
PCB# 175	0.23	<0.079-1.11	5
PCB# 177	0.05	<0.079-0.14	2
PCB#178	0.05	<0.079-0.12	1
PCB# 180/193	0.26	<0.079-0.97	9
PCB# 182/187	0.13	<0.079-0.37	8
PCB# 183	0.05	<0.079-0.10	2
PCB# 194	0.06	<0.079-0.20	2
PCB# 196/203	0.11	<0.079-0.46	4
PCB# 206	0.05	<0.079-0.14	1
PCB# 209	0.06	<0.079-0.18	1

Among all CBIP fish, total PCB congeners concentration ranged from 0.31 ppb wet weight in Crab Creek samples to 40.82 ppb wet weight in Moses Lake samples. Saddle Mountain Lake (common carp) and WB-10 Wasteway (common carp and bass) samples were the only sites with all sample PCB residues below detection limits and thus not included in the total PCB calculations used for the probabilistic assessment (Table 11).

Table 11. Total PCB congeners concentration (ppb wet weight) in fish from the Columbia Basin Irrigation Project field sampling during 2005 and 2006.

Sample Location	Total PCBs (ppb)
Billy Clapp Lake (BCL)	34.79
Banks Lake (BLK)	10.35
Crab Creek (CCL)	0.31
Frenchman Hills Wasteway (FRW)	3.84
Lind Coulee Wasteway (LDW)	1.45
Moses Lake (MOL)	40.82
Potholes Reservoir (PHR)	10.59
Royal Lake (ROY)	5.94
Saddle Mountain Lake (SML)bass	<0.097
Saddle Mountain Lake (SML) carp	5.29
Scootenay Reservoir (SNR)	10.89
WB10 Wasteway (WBW)bass	<0.093
WB10 Wasteway (WBW)carp	<0.093
Winchester Wasteway (WWW)	0.63

<u>PCB Accumulation Patterns CBIP Fish</u>

Homolog profiles of polychlorinated congeners varied in their contribution to the overall total PCB concentration depending on sample location. For the CBIP data, between species differences could not be determined because of lack of positive results from bass sample composites. From most sites penta- and hexachlorobiphenyls were the predominant homolog groups. Crab creek data differed from this pattern with heptachlorophenyls representing 64% of the overall total PCB homolog profile (Figure 6).

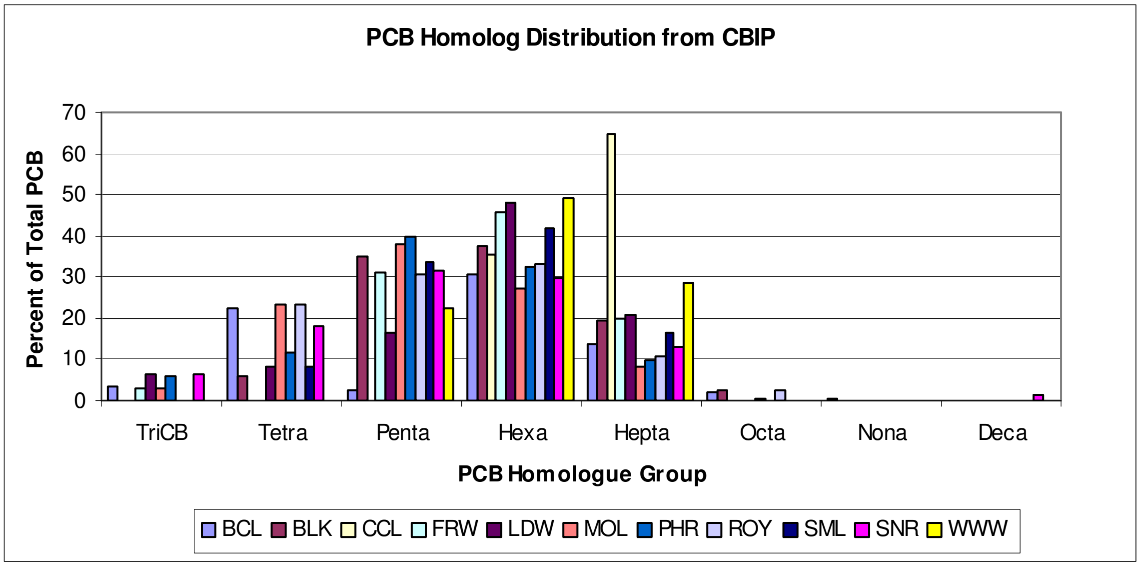

Figure 6. PCB homolog distribution for common carp within the CBIP 2005-2006.

Washington Department of Ecology (WDOE) Field Sampling Studies

Of the 12 possible CBIP sample sites only five (Banks Lake, Moses Lake, Crab Creek, Frenchman Hills Wasteway, and Potholes Reservoir) were sampled by WDOE between 1992 and 2005. Species of fish collected included channel catfish, walleye, rainbow trout, lake whitefish, and large-scale suckers (Table 12). Aroclors 1254 and 1260 were detected in ten WDOE fish samples. Total Aroclors concentrations ranged from 5.2-51 ppb wet weight (Table 12). With the exception of the Potholes Reservoir lake white fish sample, no individual PCB congeners were separately reported by the WDOE. The lake white fish from Potholes Reservoir sample reported total PCB congeners 6 ppb. Only the total Aroclor data was used when conducting the Monte Carlo analysis portion of this study.

Table 12. Washington Department of Ecology sample locations and total Aroclor Concentrations (ppb wet weight) in fish from the Columbia Basin Irrigation Project 1992-2005.

Sample Location	Collection Date	Species	Matrix	Aroclor-1254	Aroclor-1260	Total Aroclors
Banks Lake[1]	10/16/2003	Lake white fish	Tissue	17	16	33.00
Banks Lake[1]	10/16/2003	Rainbow trout	Tissue	8.3	8.5	16.80
Banks Lake[1]	10/15/2003	Walleye	Tissue	3.1	3.6	6.70
Moses Lake[1]	9/29/2003	Channel Catfish	Tissue	3.8	3.5	7.30
Moses Lake[2]	10/23/2002	Largemouth Bass	Fillet, skin off	14	3.9	17.9
Moses Lake[2]	10/23/2002	Rainbow Trout	Fillet, skin on	9.25	3.0	12.25
Crab creek[2]	10/23/2002	Walleye	Fillet, skin on	3.7	0.0	3.7
Crab creek[3]	9/15/1992	Largescale sucker	Tissue/whole	26	25	51.00
Crab creek[3]	9/15/1992	Largescale sucker	Tissue/whole	19	24	43.00
Frenchman Wasteway[3]	9/15/1992	Mountain Whitefish	Tissue/Fillet	14	16	30.00
Potholes Reservoir[4]	10/25/2005	Lake white fish	Tissue/whole	N/A	N/A	17.00
Potholes Reservoir[4]	10/25/2005	Walleye	Tissue/Fillet	N/A	N/A	5.20

[1]Seiders et. al (2006).
[2] Seiders et. al. (2004).
[3]Davis et. al. (1998).
[4]Seiders et. al. (2007).

EPA 2000 and 2001 Field Sampling Study

Frenchman Hills Lake (a.k.a. wasteway), and Potholes Reservoir were the only sites sampled by the U.S. EPA during their National Study of Chemical Residues in Lake Fish between 2000 and 2004. Collected fish were not listed by species but were categorized as either bottom-dwellers or predators. Bottom dwellers where processed as whole fish and predators as fillets prior to homogenization for analysis.

The results of the U.S. EPA study closely matched the CBIP data with individual congeners and homologs reported and used in the final total PCB calculations. Total PCB homologs in bottom-dwellers ranged from 12.03-14.48-ppb wet weight. The total PCBs in predatory fish were much lower, from 1.16-1.64 ppb wet weight (Table 13). Lengths and weights were not present in the data files so correlation between lengths/weights and PCB concentrations could not be determined.

Table 13. U.S EPA National Study of Chemical Residues in Lake Fish sampling locations and total PCB homolog concentrations (ppb wet weight (EPA 2005).

Sample Location	Year	Species	Total PCBs Homolog
Frenchman Hills Lake	2000	Bottom-dweller	14.48
Frenchman Hills Lake	2000	Predator	1.64
Potholes Reservoir	2001	Bottom-dweller	12.03
Potholes Reservoir	2001	Predator	1.16

PCB Accumulation Patterns EPA Collected Fish

PCB homolog profiles varied in their contribution to the overall total PCB concentration depending on sample location and species sampled. In the Frenchman Hills (FRW) bottom dweller samples, hexachlorobiphenyls and pentachlorobiphenyls represented 38% and 31.5%, respectively, of the total PCB homologs (Figure 7). The Potholes Reservoir (PHR) sample location had similar bottom dweller patterns of hexachlorobiphenyls (36.4%) and pentachlorobiphenyls (30.8%) (Figure 7). PHR predator samples showed a slightly different pattern than bottom dwellers, having predominantly higher pentachlorobiphenyls (35.3%) compared to hexachlorobiphenyl (31.9%). Predator samples from the (FRW) followed a similar pattern as their bottom dweller counter part with hexachlorobiphenyls being predominant at 35.4% and pentachlorobiphenyls at 34.1% (Figure 7).

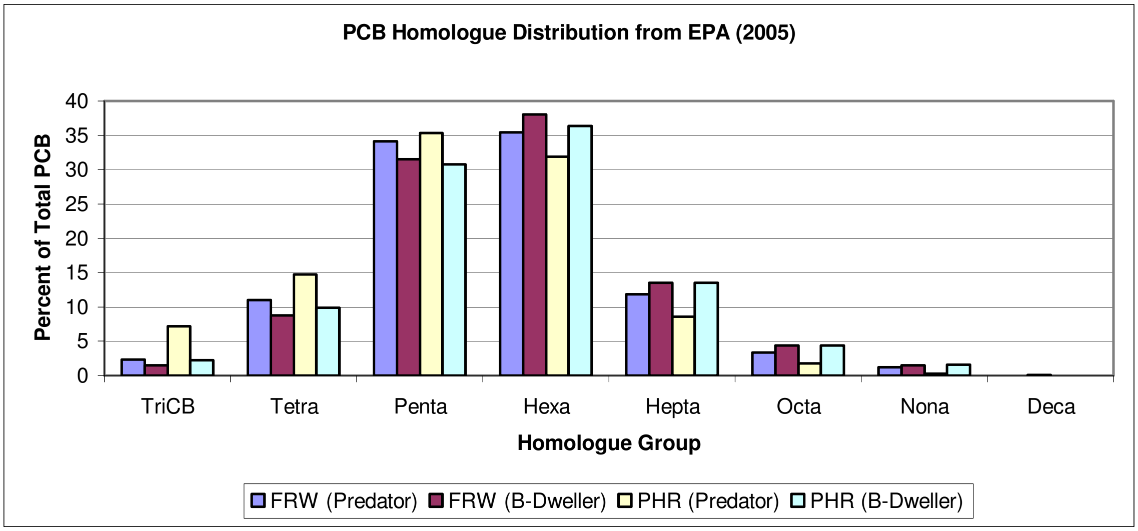

Figure 7. PCB Homolog distribution between bottom dwellers (B-dwellers) and predators from EPA (2005).

Probabilistic Analysis

The results from the fish tissue monitoring projects were used to examine exposure to two demographic groups based on age-related differences--combined adult male and females (18-65 years) and children (<15 years). Within these groups three probabilistic exposure scenarios were completed for children less than 15 years of age, adults from the Lake Roosevelt area, and adults within the Columbia Basin. A deterministic calculation was also completed to help assess the differences, if any, between the current regulatory methodology and the probabilistic approach used in this study. Carcinogenic and non-carcinogenic hazards were considered in the risk characterization.

<u>Carcinogenic Risk Analysis</u>

Risk levels varied greatly between adult and child exposure scenarios in a one-dimensional probabilistic analysis. Small differences in risk level between adult populations were observed in the one-dimensional analysis, but the two dimensional analysis revealed larger differences. In Columbia Basin adults the median risk level (at p=0.05) ranged from 4.24E-07 to 6.13E-07 for the one- and two-dimensional analysis, respectively (Appendix A). Lake Roosevelt adult risk ranged from 2.30E-06 to 3.89E-06 for the one- and two-dimensional analysis, respectively (Appendix B). Children less than 15 years old followed similar trends with risk estimates for the one-dimensional analysis of 5.91E-07 and the two-dimensional analysis of 9.23E-07 (Table 14;Appendix C). Thus, in all exposure scenarios, the one-dimensional Monte Carlo analysis yielded a higher carcinogenic risk estimate than the two-dimensional analysis. Lake Roosevelt adults were estimated to have higher risks than the Columbia Basin adults, and children less than 15 years old had the lowest risks for carcinogenic effects.

The deterministic results drastically differed from the probabilistic results. Risk of carcinogenic effects based on a deterministic analysis was 12 to 300 times greater then risk estimated by the probabilistic analysis (Table 14).

Table 14. Median carcinogenic risk level and standard deviations estimated for p=0.05 each population group studied

Analysis Scenario	Columbia Basin Adult	Lake Roosevelt Adult	Children<15
One-Dimensional	4.24E-07	2.30E-06	5.91E-07
SD(+/-)	1.92E-06	1.20E-06	3.77E-06
Two-Dimensional	6.13E-07	3.89E-06	9.23E-07
SD(+/-)	2.73E-06	3.86E-06	1.12E-06
Point Estimate (Deterministic)	7.27E-05	4.60E-05	1.70E-04

Non-carcinogenic Analysis

Median hazard quotients (HQ) ranged from 0.01 to 0.08 depending on the demograhic group being assessed (Table 15). In the probabilistic analysis none of the groups exceeded the level of concern of 1.0 or above. The Lake Roosevelt adults had the highest HQ, ranging between 0.06 and 0.08 (Appendix B). The Columbia Basin adults were slightly lower with a range of 0.01 to 0.04 (Appendix A). The children had an HQ that ranged from 0.01 to 0.02 (Appendix C). In contrast to the probabilistic analysis, all deterministic HQ for adult and children exposures were greater than one with children facing the greatest risk (Table 15).

Table 15. Non-carcinogenic effects hazard quotient (HQ) medians with standard deviations (SD) at p=0.05 for each population group studied.

Analysis Scenario	Columbia Basin Adult	Lake Roosevelt Adult	Children<15
One-Dimensional	0.01	0.06	0.01
SD(+-)	0.05	0.86	0.13
Two-Dimensional	0.04	0.08	0.02
SD(+-)	0.05	0.11	0.02
Point Estimate (Deterministic)	1.82	1.15	4.26

Sensitivity Analysis

The sensitivity analysis demonstrated that chemical concentrations had the largest influence on risk to adults, accounting for 52-53% of the variance associated with exposure. Exposure factor, exposure duration, and averaging time contributed 14.6-16.1% of the variance in risk for the adult groups. Ingestion rate, fraction cooking loss and body weight contributed less than 1.0% of the variance (Table 16).

For children under 15 ingestion rates contributed the largest percentage (34.2%) to the estimated risk with PCB concentration as the second largest factor (31.5%). Exposure factor and averaging time had the same contribution of 11.0%, and exposure duration only influenced 8.8% of the variance in risk. Body weight and fraction cooking loss had the least influence on the risk estimation with contributions of only 2.9% and 0.6%, respectively (Table 16).

Table 16. Sensitivity analysis for the percentage contributed by all input variables to the output variance for all population groups studied.

	Columbia Basin Adult	Lake Roosevelt Adult	Children<15
Chemical Concentration	51.3	53.0	31.5
Ingestion Rate	0.9	0.6	34.2
Exposure Factor	15.8	15.0	11.0
Exposure Duration	15.6	16.1	8.8
Fraction Cooking Loss	0.9	0.6	0.6
Body Weight	0.0	0.0	2.9
Averaging Time	15.6	14.6	11.0

Discussion

PCB Residue Characterization

The accumulation of co-planar (non- and mono-ortho) PCB congeners in aquatic food webs is of special interest due to their dioxin-like characteristics. Fish collected within the CBIP showed a significantly higher proportion of PCBs occurring in the pentachlorobiphenyls and hexachlorobiphenyls homolog groups. Other researchers have observed this same shift to higher chlorinated homologs in biota at higher trophic levels of the food web (Willman et. al. 1997; Feldman and Titus, 2001). Large predatory fish tend to accumulate higher PCB levels than smaller fish (Stow and Qian, 1998). Even with a small sample size (n=9) the CBIP fish data followed this same pattern with a slight correlation (r^2=0.27) between PCB concentration and body size.

PCB isomer compositions in fish from the CPIB were very similar to isomer compositions in fish collected from other fresh water (Carlson et. al. 2006;Gewurts et. al. 2007) and marine environments (Fikslin et. al. 2003). For the CBIP data only one dioxin-like congener, PCB 105, was observed in common carp collected from seven of the twelve study sites. Other researchers have found in fish tissue from fresh water environments of the Great Lakes region that PCB 105 and PCB 118 represented at the 75^{th} quartile 2.4% and 6.2%, respectively, of the total PCB abundance (Bhavsar et. al 2007). The contrast in dioxin-like congener proportions between CBIP and Great Lakes fish could have been due to differences in method detection limits between the different studies.

Another possible explanation for congener profile variation is the differences in the type of ecosystems from which the fish are collected. For example, the CBIP is a constructed facility and was created for the purposes of water storage and irrigation and do not follow the same processes as natural systems. Within these systems, normal limnological processes do not occur. Instead, water flow is seasonally managed by the irrigation district. Water is added in early March from Grande Coulee Dam and is sent throughout the entire system. Sediment and impounded water is flushed out of the system to prepare for the delivery of new irrigation water to farmers. Seasonal flushing of the system likely affects the chemical dynamics of PCBs, including their accumulation and transfer throughout the food web. Trophic level makeup and interaction among species, especially predator-prey relationships, could be altered and behave differently than in natural systems. This change could potentially cause variations in feeding preferences that ultimately would alter how PCBs accumulate up the food chain and therefore potentially impact human health.

Probability Assessment

In the past, risk assessments have traditionally summarized results in the form of a single estimate of risk. These assessments more often then not used conservative assumptions to help combat uncertainty inherent with these types of assessments. To help alleviate the impact from uncertainty and variability a more complete characterization for risk can be conducted using probability distributions to describe the likelihood of different possible risk values.

For this assessment a quantitative approach was used to help estimate risk from ingestion of fish tissue from the Columbia Basin Irrigation Project. First, a sensitivity analysis was conducted on the models used to estimate cancer and non-cancer risks. Results from this analysis helped to determine how much influence specific assumptions within the models had on the final output value. Knowing how each input parameter affected the model enabled refinement of risk estimates and greatly increased the accuracy of the results. The sensitivity analysis enabled me to ignore or discard some assumptions, while changing distributions types in others until the refining process was complete. As a result, I was able to construct a more realistic spreadsheet model in less time with more accuracy.

Sensitivity Analysis

Many times assumptions might have a high degree of uncertainty yet have little effect on the final forecast. This is due to certain parameters not contributing enough to influence the outcome of the overall model. The opposite could also hold true; an assumption could have a low degree of uncertainty but in the end could have a fairly high influence on the overall results of the model. The results from the models did show some variability between the two adult population (Columbia Basin and Lake Roosevelt) and the children (< 15 years) exposure scenarios for most of the input variables. Such variability is due to different exposure factors used for adults and children that was necessitated by differences in physical body parameters and consumption behavior between these demographic groups.

The analysis revealed that chemical concentrations (C) had the largest influence on risk for adults, accounting for 52-53% of the variance associated with exposure. Other researchers running Monte Carlo simulations found that consumption frequencies (EF) were the greatest contributors to the variance (Harris and Jones, 2008), probably due to an overestimate of yearly fish consumption rates. Another possible problem is a difference in interpretation between consumption/ingestion rates and exposure frequency, which could cause outliers within the sensitivity analysis parameters.

Other parameters such as exposure factor, exposure duration, and averaging time contributed 14.6-16.1% of the variance to the models for all adult groups studied. Ingestion rate, fraction cooking loss and body weight contributed less than 1.0% of the variance. For children under 15, ingestion rates contributed the largest percentage (34.2%) to the estimated risk, and PCB concentrations were the second largest factor (31.5%). The high sensitivity of the model to ingestion rate could have been caused by the use of empirical data obtained during creel surveys from fisherman that could have overestimated consumption of fish for the year. When exposure distributions follow a lognormal distribution, the exposure could be overestimated for percentile distributions above the median if empirical consumption survey data are used (Stanek et. al. 1998). The overestimation results from the survey data covered a much shorter exposure period than the actual biological exposure period. In support of this hypothesis, researchers have found that sport fisherman interviewed during seasonal fishing periods may overestimate their consumption for the rest of the year (Shatenstein et. al 1999).

For the CBIP little information had been collected on fish consumption behavior for most local populations. However, the DOH did conduct creel surveys which measured consumption pattern of anglers in the Lake Roosevelt area. Because of the proximity of this population to the CBIP, these data would be the most current for the local population of sport fisherman, but little attention was made to assess women or children. Researchers have found that women do consume far less fish than males (Burger et. al., 2000). This could be a source of uncertainty when assessing the risk from consumption of fish for women in the CBIP region. Other surveys conducted by CRITFC (Columbia River Inter-Tribal Fish Commission) in 1994 did include children, but because it was over such a large area the data obtained might not be representative of the fish consumption around the CBIP.

One-dimensional and Two-dimensional analysis

Overall, risk to the general population of recreational fisherman within the CBIP from either carcinogenic or non-carcinogenic hazards was small and below tolerable levels of concern. Tolerable levels of concern for carcinogenic hazards are classified as probable if the model risk estimate is greater than 10E-06 (EPA 1996). Between the different scenarios, the one-dimensional Monte Carlo analysis yielded a slightly higher carcinogenic risk estimate than the two-dimensional analysis. This difference in risk estimates was caused by the isolation of the chemical concentration factor in the two-dimensional models to better reflect variability of PCB concentrations among fish samples. Once the chemical concentrations in the fish tissue were isolated and defined as uncertain for the two-dimensional analysis a better characterization of risk was accomplished with some variations between scenarios.

Carcinogenic risk levels varied greatly between adult and child exposure scenarios in a one-dimensional probabilistic analysis. Small differences in risk level between adult populations were observed in the one-dimensional analysis, but the two dimensional analysis showed that Columbia Basin adults faced higher risks than Lake Roosevelt adults. Children less than 15 years old were estimated to have lower risks than either adult population. In all exposure scenarios, however, the one-dimensional Monte Carlo analysis yielded a higher carcinogenic risk estimate than the two-dimensional analysis.

Most of the variability in the estimated risk levels was due to differences in ingestion rates and exposure duration among the groups studied. Children had the lowest ingestion rate with a mean of 2.8 g/day and Columbia Basin adults had a mean of 7.5 g/day. The highest mean ingestion rate came from the Lake Roosevelt group at 42 g/day. Exposure duration also varied with means ranging from 35 years for adults to 7.5 years for children. It is this difference that impacted the sensitivity analysis the most between the two scenario models and showed how important physical differences are when conducting exposure assessments. Being directly correlated with age (ODEQ 1998), body weight and ingestion rate also had some impact on the differences between the adult and children scenario models. Because of children's fast growth rate it is difficult to generally quantify specific intake rates. But with the fast growth rate and the inherent variability associated with ingestion rates in children a proportionally bigger difference in exposure would be possible.

Monte Carlo versus Deterministic Risk Characterization

The deterministic risk results drastically differed from the probabilistic results. Risk of carcinogenic effects based on a deterministic analysis was 12 to 300 times greater then risks estimated by the probabilistic analysis. For adults living within the CBIP the deterministic risk levels varied from 7.27E-05 for the Columbia Basin population to 4.60E-05 for the Lake Roosevelt population. This translates into an average of 6 excess cancer tumors per100,000 persons exposed versus 0.22 per 100,000 persons estimated in the 2-dimensional Monte Carlo analysis. Most of this variation can be attributed to the differences in PCB concentrations and annual consumption rates used by the two methodologies.

Children less than 15 years old also showed an elevated risk estimate of 1.70E-04 for the deterministic approach compared to the 2-dimentional estimate of 9.23E-07. This would translate into an average of 0.2 excess cancer tumors per100,000 persons exposed versus 0.09 per 100,000 persons estimated by the 2-dimensional Monte Carlo analysis. The deterministic risk estimates associated with children exposure was likely inflated compared to the probabilistic risk estimate as a result of an assumption of shorter exposure periods.

The results for non- carcinogenic effects followed a pattern similar to the carcinogenic results. Median hazard quotients (HQ) ranged from 0.01 to 0.08 depending on the demographic group being assessed. In the probabilistic analysis none of the groups exceeded the level of concern of 1.0 or above. All HQs were at least ten fold lower than the regulatory benchmark of 1.0, and children's risk was lower than adults. In contrast to the probabilistic analysis, all deterministic estimates of HQ were greater than one, but children had the highest risk estimate.

The higher carcinogenic risk estimates by the deterministic method compared to the Monte Carlo method resulted from the use of conservative worst-case values for exposure inputs. These input values usually fall above the 95th, 98th, or even the 99.9th percentile, which can significantly overestimate true exposure and risks to people (Burmaster and Harris, 1993; Cullen, 1994). These worst-case scenario values were established to alleviate some of the variability and uncertainty in risk assessment by using simple equations and single point values to estimate risk.

In contrast to the deterministic method of using point estimates for input data, the Monte Carlo approach uses exposure factor values obtained from an entire distribution of many possible randomly selected variables for each input parameter. Variability and uncertainty are defined for each parameter and exposure values are assessed within a particular statistical distribution (i.e. lognormal, normal, constant, etc). When appropriate Monte Carlo methods are used, a full range of exposure values are available over all percentiles and their associated risks can be evaluated with a broader range of probable outcomes. Unlike the deterministic techniques that give a single point estimate, Monte Carlo techniques will give a more comprehensive risk characterization based on a probability distribution of risk over a given range. Probabilistic analysis gives risk managers a better tool with more information that helps make a realistically informed decision about populations at risk. But the value of Monte Carlo simulations are only as good as the data being defined and if one or more of the parameters are poorly defined, the end result could have a low confidence value.

Uncertainties

Other sources of uncertainty in risk characterizations for human health come from the choice of dose-response criteria applied to estimate risk. The dose-response criteria (i.e. cancer slope factor and reference dose values) used in this assessment were developed by the U.S. EPA (1996) based on very conservative results from animal studies (Brunner et. al., 1996; Norback and Weltman, 1985). In those studies, the administered dose far exceeded those encountered by adults and children consuming contaminated fish. Uncertainties for the cancer slope factor include species-to-species comparisons, extrapolation of high dose values reduced to fit much lower doses for human exposure, and data analysis techniques designed to provide upper bound values to fit a more deterministic methodology. Similarly, the RfD values to characterize non-cancer hazards were also developed from animal study data that required application of conservative safety factors to deal with uncertainty about extrapolating animal test results to humans and the range of exposures likely encountered (IRIS, 2002). This high degree of protectiveness has ultimately impacted how fish advisories are implemented across our region.

At the present time, most regional and local PCB fish advisories that are in effect are based on risk estimates usually using the RfD values for Aroclor 1254. Within Washington State, most environmental assessments conducted for PCBs measure Arolcor levels and not specific congeners. An advisory is issued when total PCB levels based on Aroclor analysis are 0.2 ppm or greater (WADOH, 2003). Relying on Aroclor residues likely results in a gross overestimate of the true levels of PCBs in fish tissue, making local fish advisories overly protective. Because composition of PCB congeners in the environment do not resemble the original Aroclor mixtures after weathering has occurred, the use of RfD values based mainly on the commercial mixtures would be inappropriate.

Scientific opinion is divided about the validity of extrapolating PCB-associated cancer risk based on rodent bioassays to human environmental exposures (Silberhorn et. al. 1990; ATSDR, 2000; IRIS, 2002). These mixed opinions within the toxicology community could come from the uncertainty of choosing which cancer slope factor (CSF) to use for a given situation among the several available to choose from. Within the past few years most studies in the scientific literature have varied greatly in their use of the CSF as a toxicological benchmark. In some cases, the chosen benchmarks were inaccurately applied according to applicable guidelines, giving a false assessment of risk. For Example, Hites et. al. (2004) used an upper bound CSF (USEPA 2000g) in his assessment of farmed raised salmon from around the world. According to the USEPA Office of Water (2000g), the slope factor used by these researchers should be for non-commercial fish and the development of fish advisories and not used to characterize the risk from consumption of commercial fish by the general population. Instead, the central estimate of 1.0 for the CSF should have been used because it is applicable to assessing the risk to the general population (IRIS 2002). This inconsistency in choice of the CSF variable makes it difficult to chose a proper assessment benchmark for determining cancer risk in humans. Further research is needed to define better CSF values that can be used for specific food types and address individuality among the exposed populations of interest.

Conclusion

Consumption of fish caught within the CBIP is a major source of PCB exposure and therefore contributes to the potential for adverse effects on human health. In this study, carcinogenic and non-carcinogenic risks from fish consumption by adults and children within the CBIP were estimated using a probabilistic model based on Monte Carlo sampling techniques. This research represents the first attempt to quantify the potential heath hazards within the CBIP using the Monte Carlo methodology. Risk characterization based on output from probabilistic modeling was compared to the more traditional deterministic risk characterization methods favored by regulatory agencies that set health advisories for consumption of contaminated fish. The results from the modeling provided a more complete and clearer picture of the differences between the deterministic and probabilistic approach to estimating risk. The noted differences in risk characterization between the two methods and the associated divergent perceptions in population associated risks suggest a need to revise how fish advisories are established in Washington State.

Currently, most regional fish advisories are based on deterministic methodologies wherein exposures to the highest PCB residues are assumed to pose the same hazard as exposure to Aroclor residues. Assuming that all PCB congeners pose the same hazard as determined for the Aroclors could overestimate risk by discounting the different composition of the highly weathered environmental residues. Furthermore, summing PCB residues and assuming they behave toxicologically as Aroclors over estimate risks when few dioxin-like congeners are actually detected.

Overestimates of risks that are associated with fish consumption could potentially scare the general public from consuming fish and ultimately reduce the health benefits from fish. Fish are a good source of omega-3 fatty acids that can help lower blood pressure and heart rate, and generally improve cardiovascular condition. Risks associated with heart disease, the leading cause of death in both men and women, can also be reduced from the consumption of fish. Pregnant women, mothers who are breastfeeding, and women of child bearing age, could benefit from fish consumption that supplies DHA, which is a specific omega-3 fatty acid that is beneficial for brain development in infants.

My analysis has shown that chemical concentration, exposure frequency, and ingestion rates are the most important factors in predicting exposure hazards to PCBs from fish consumption in the CBIP. The probabilistically characterized risks of carcinogenic or non-carcinogenic hazards associated with consumption of CBIP fish by the general population of recreational fishermen were small, below tolerable levels of concern, and substantially less than the risks calculated deterministically. Over time, the use of probabilistic techniques could cause a reduction in the number of fish advisories for PCBs in Washington State and thereby increase the consumption of fish and associated health benefits.

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APPENDIXES

APPENDIX A.
Frequency distribution output charts for Columbia Basin adults.

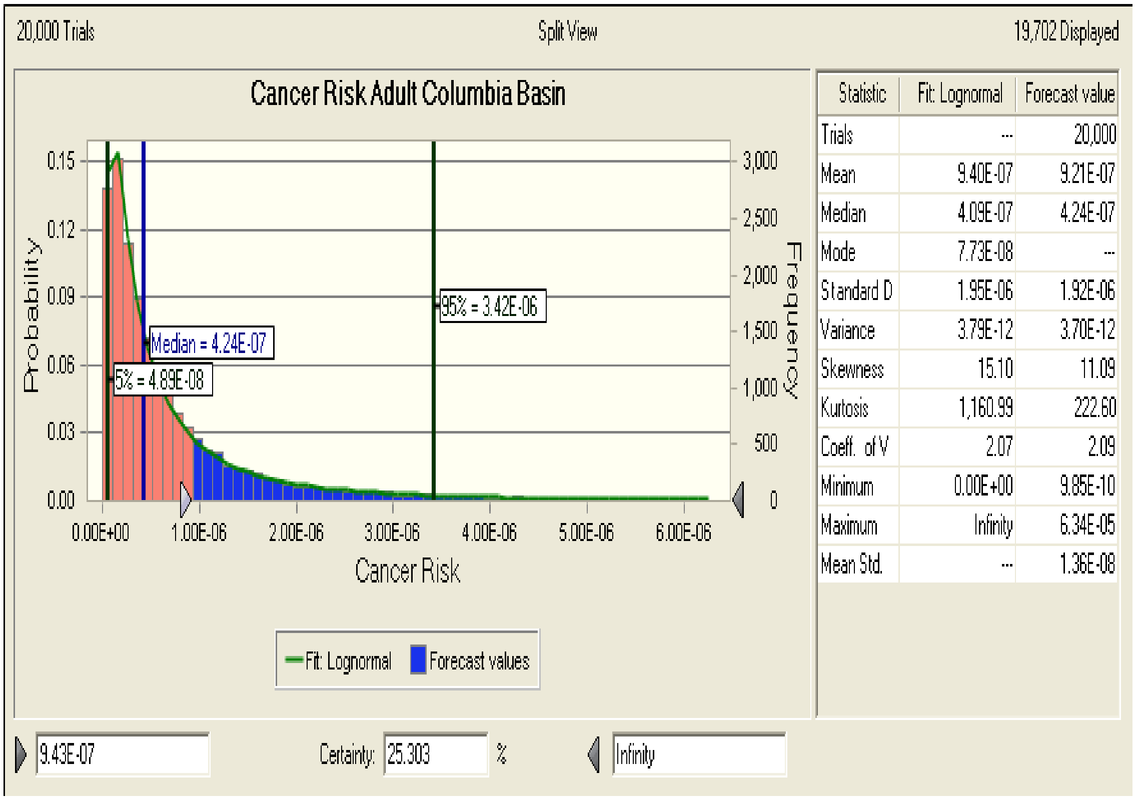


Figure A-1. Frequency distribution cancer risk Columbia Basin adult 1D.

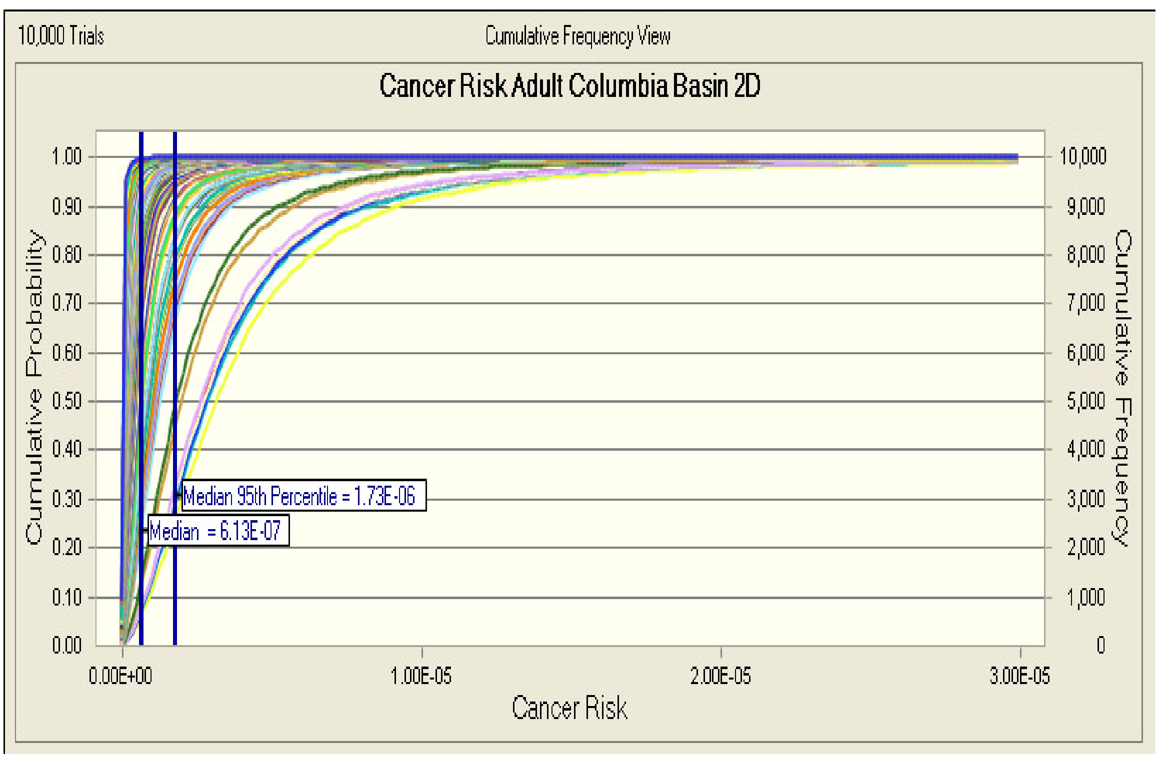


Figure A-2. Cumulative probability distribution cancer risk Columbia basin adult 2D.

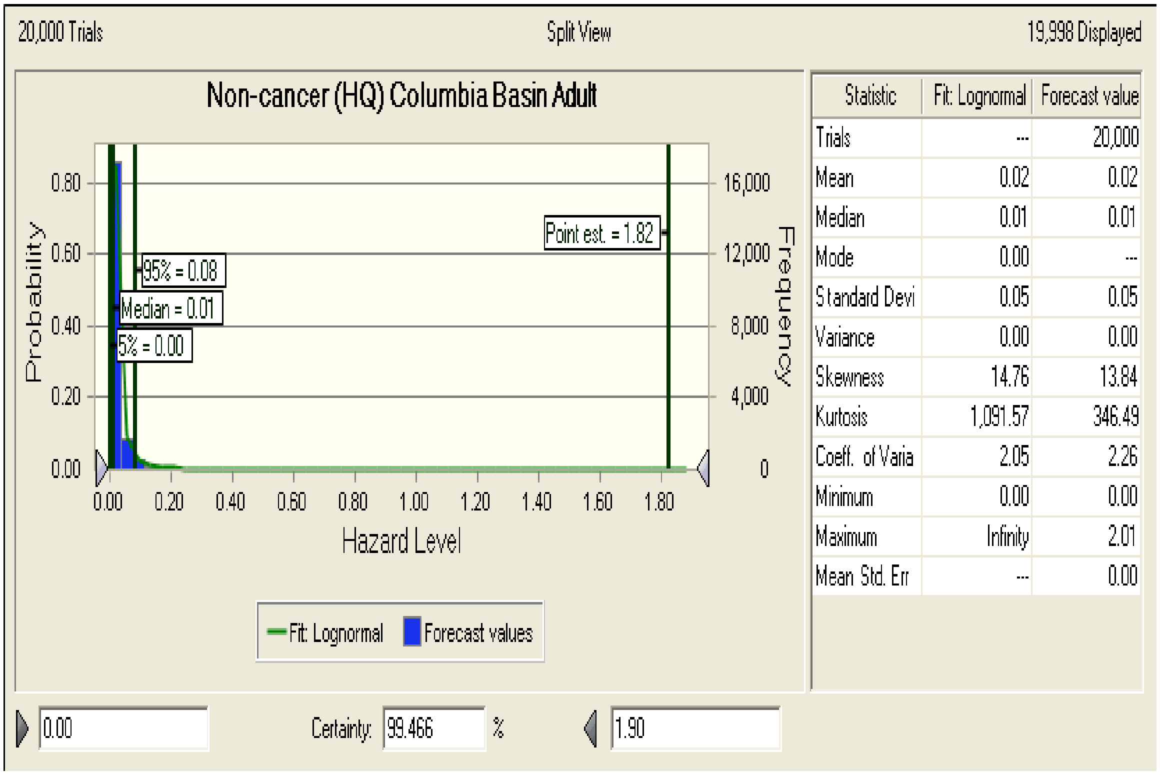


Figure A-3. Frequency distribution non-cancer risk Columbia Basin adult 1D.

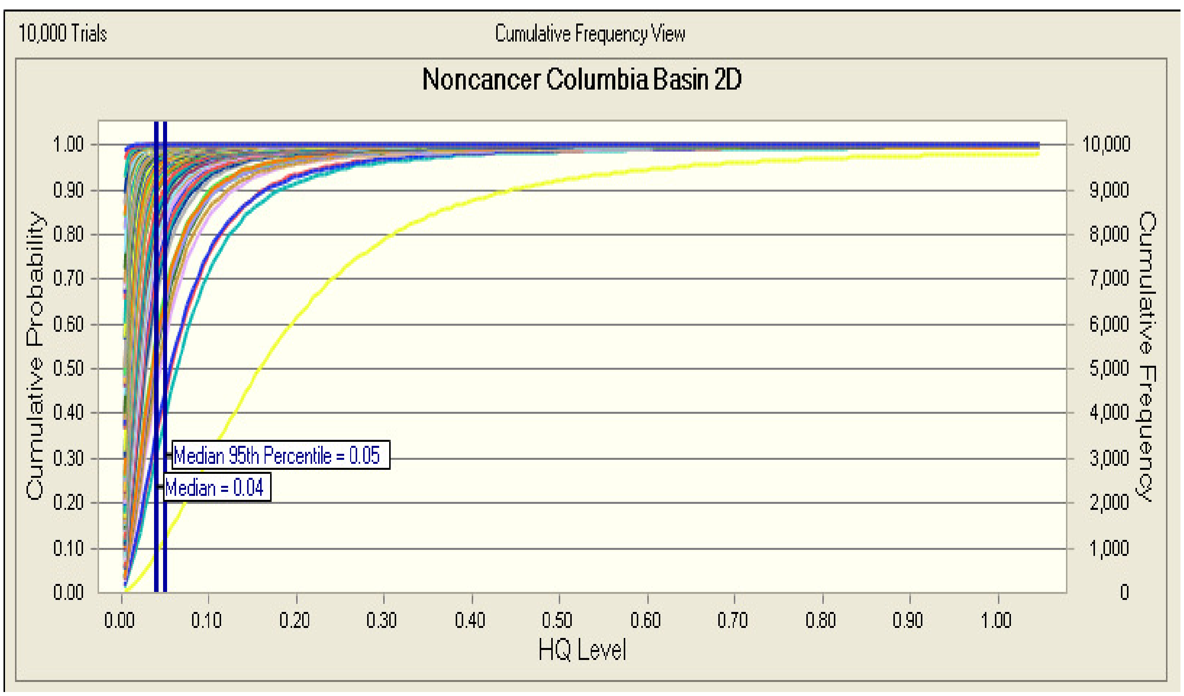


Figure A-4. Cumulative probability distribution non-cancer Columbia Basin adult 2D.

APPENDIX B.
Frequency distribution output charts for Lake Roosevelt Adults.

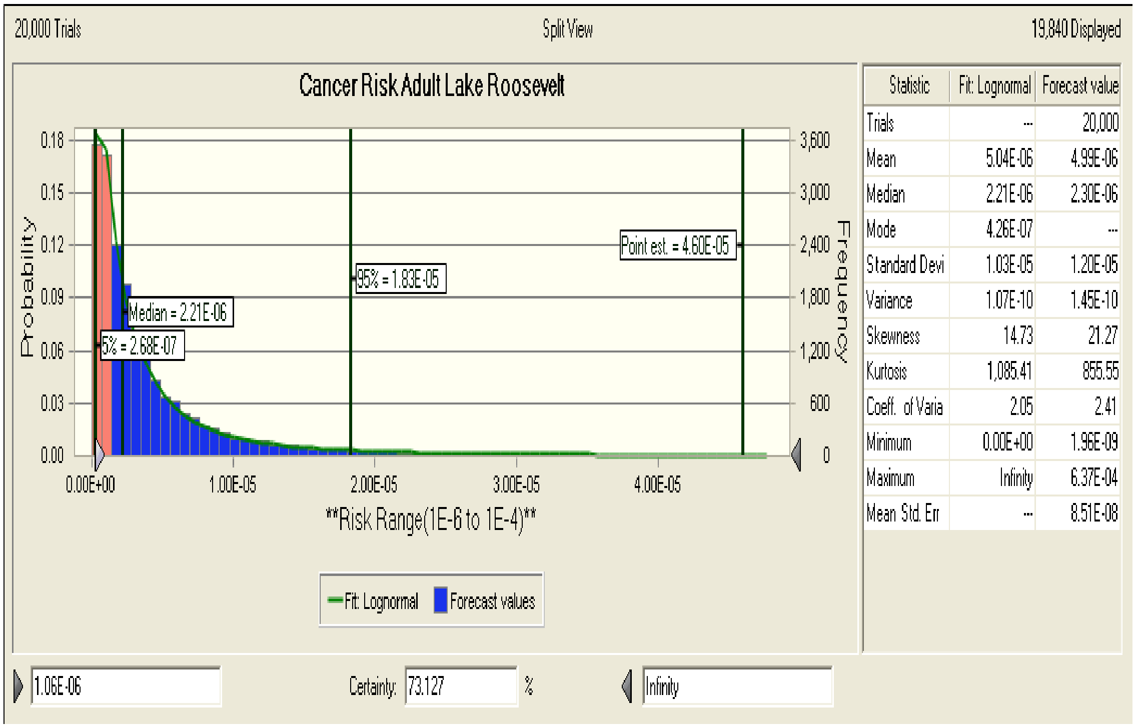


Figure B-1. Frequency distribution cancer risk Lake Roosevelt adult 1D.

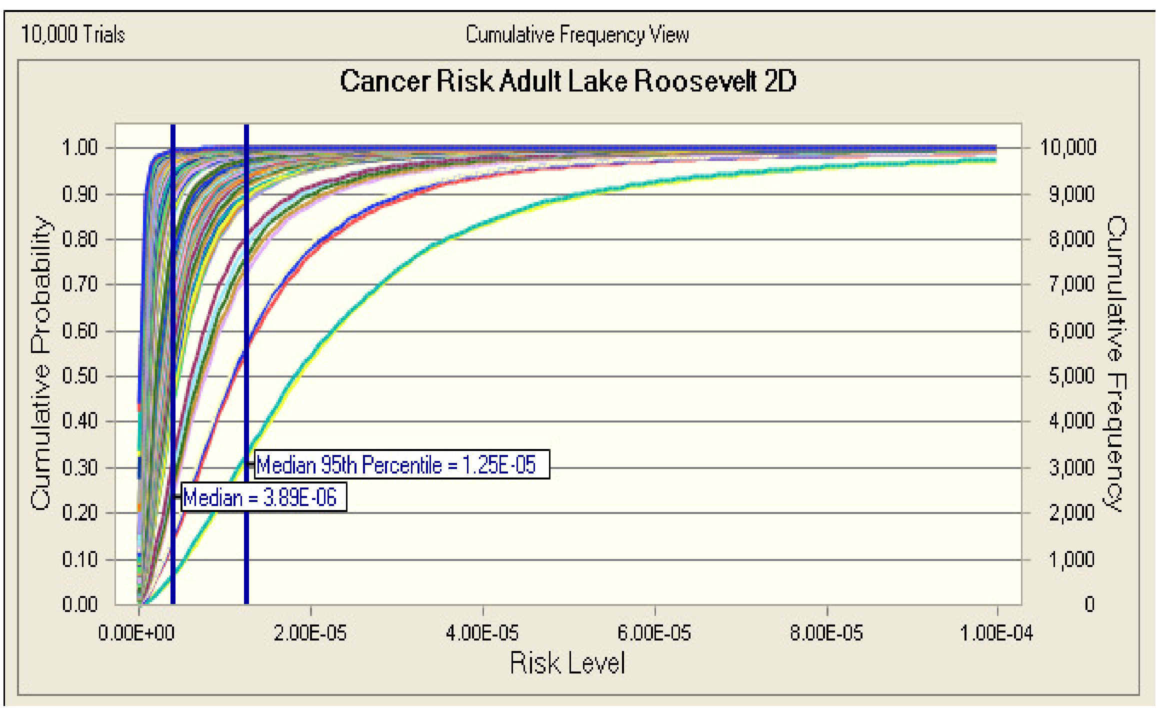


Figure B-2. Cumulative probability distribution cancer risk Lake Roosevelt adult 2D.

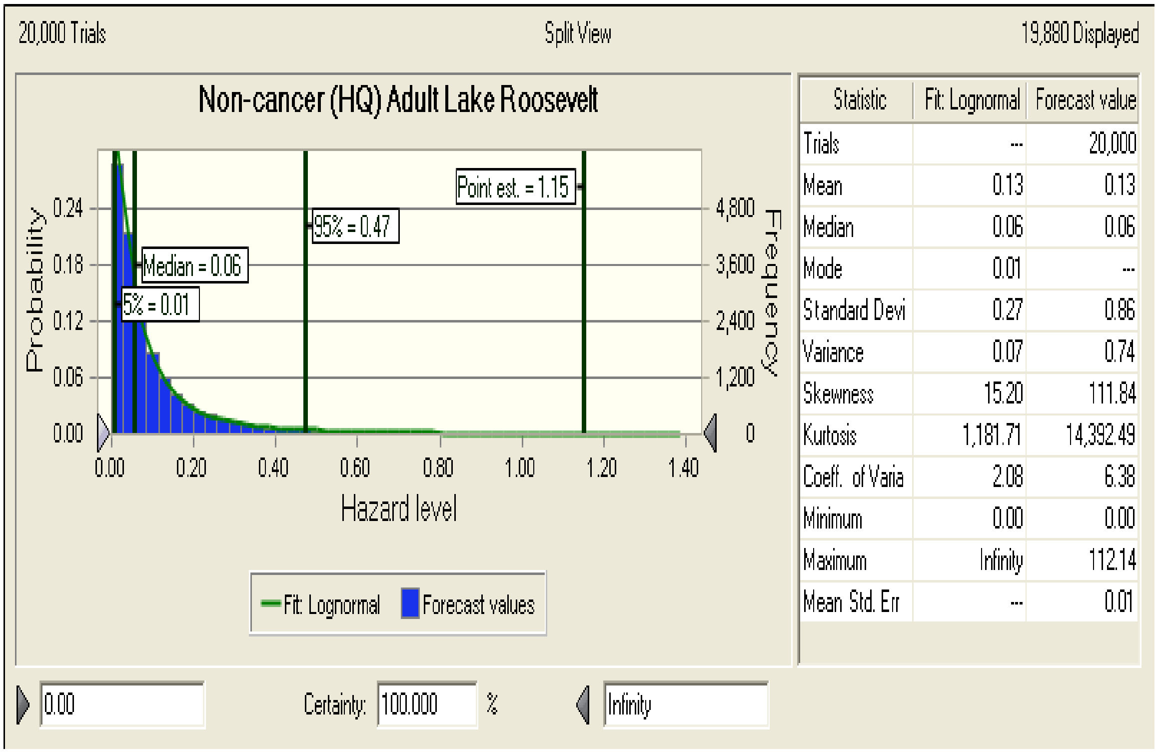


Figure B-3. Frequency distribution non-cancer lake Roosevelt adult 1D.

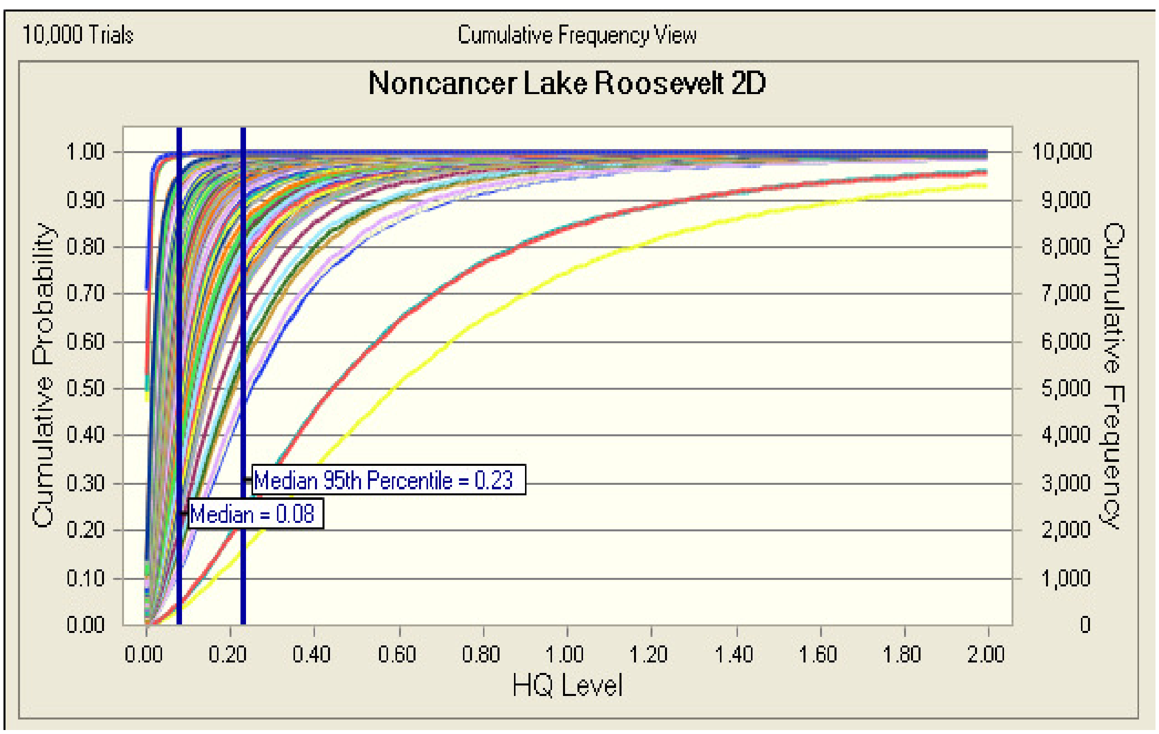


Figure B-4. Cumulative probability distribution non-cancer Lake Roosevelt adult 2D.

APPENDIX C.

Frequency distribution output charts for Children less than 15 years of age.

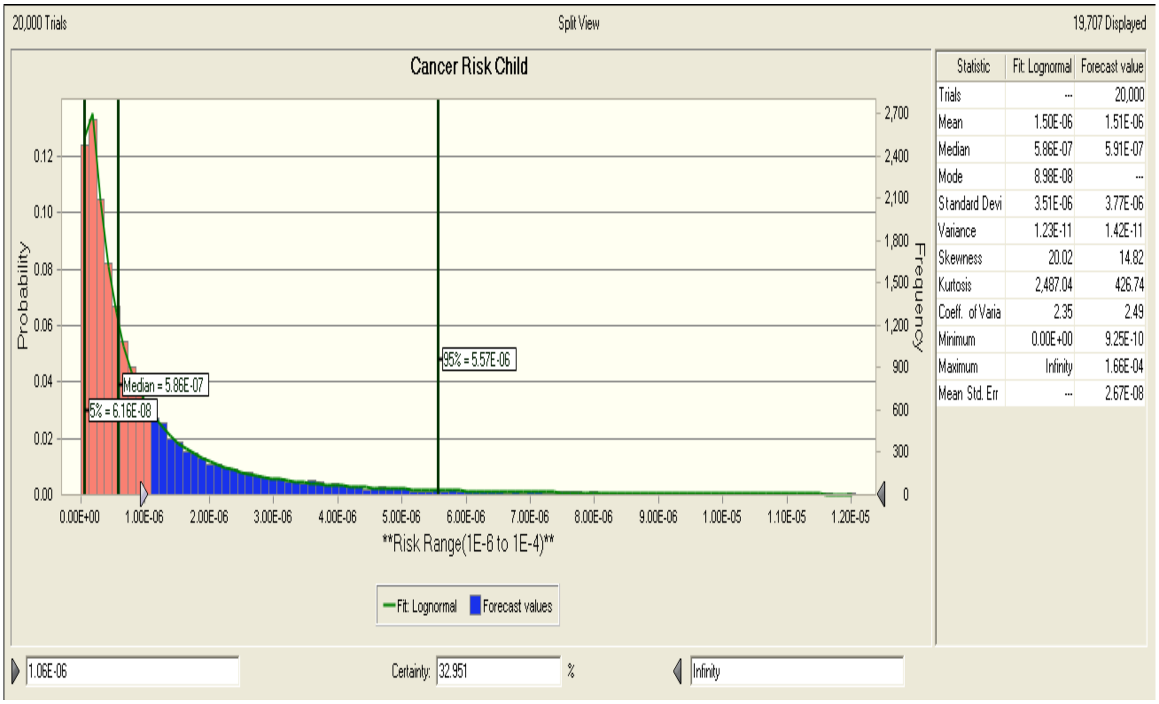
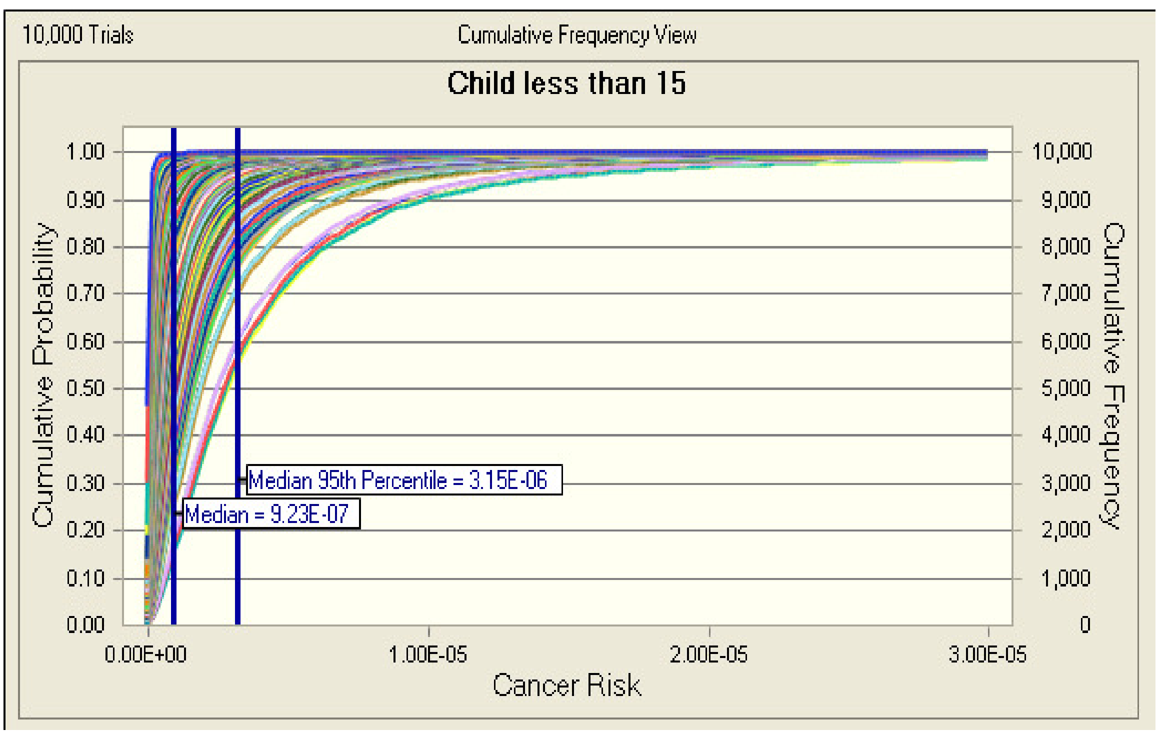


Figure C-1. Probability frequency cancer risk children (< 15 years) 1D.



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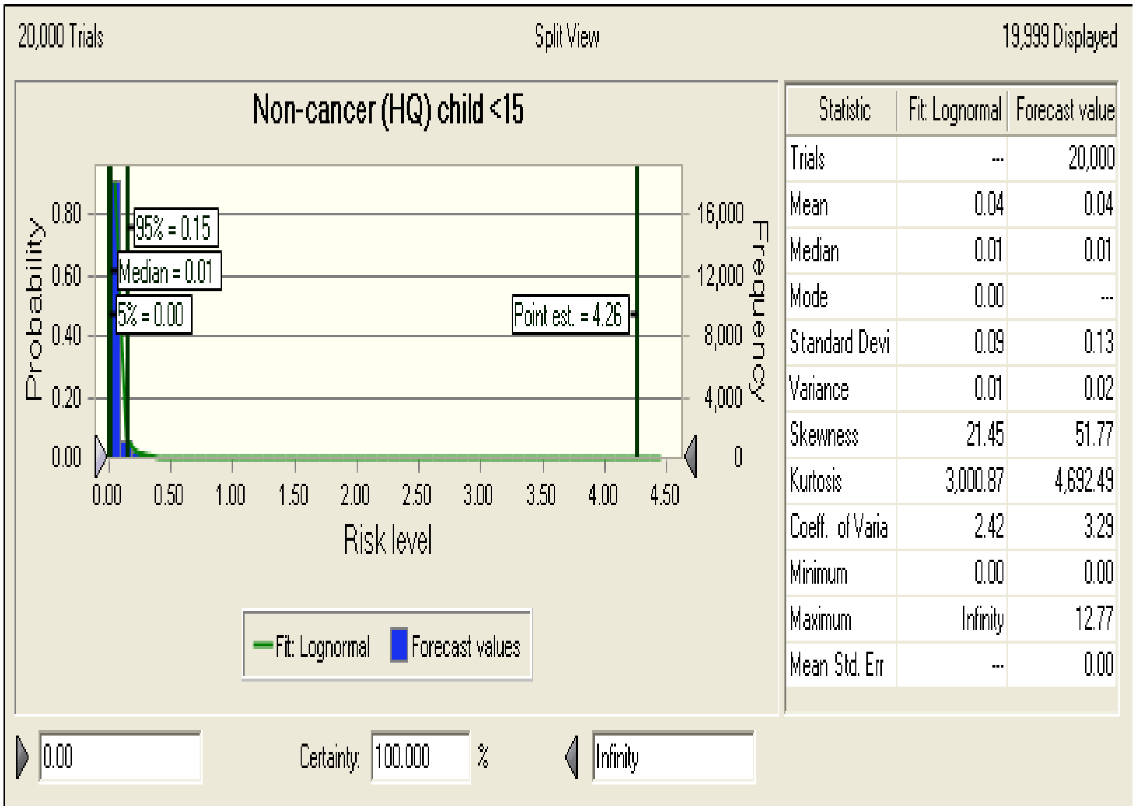

Figure C-3. Probability frequency distribution cancer risk children (< 15 years) 1D.

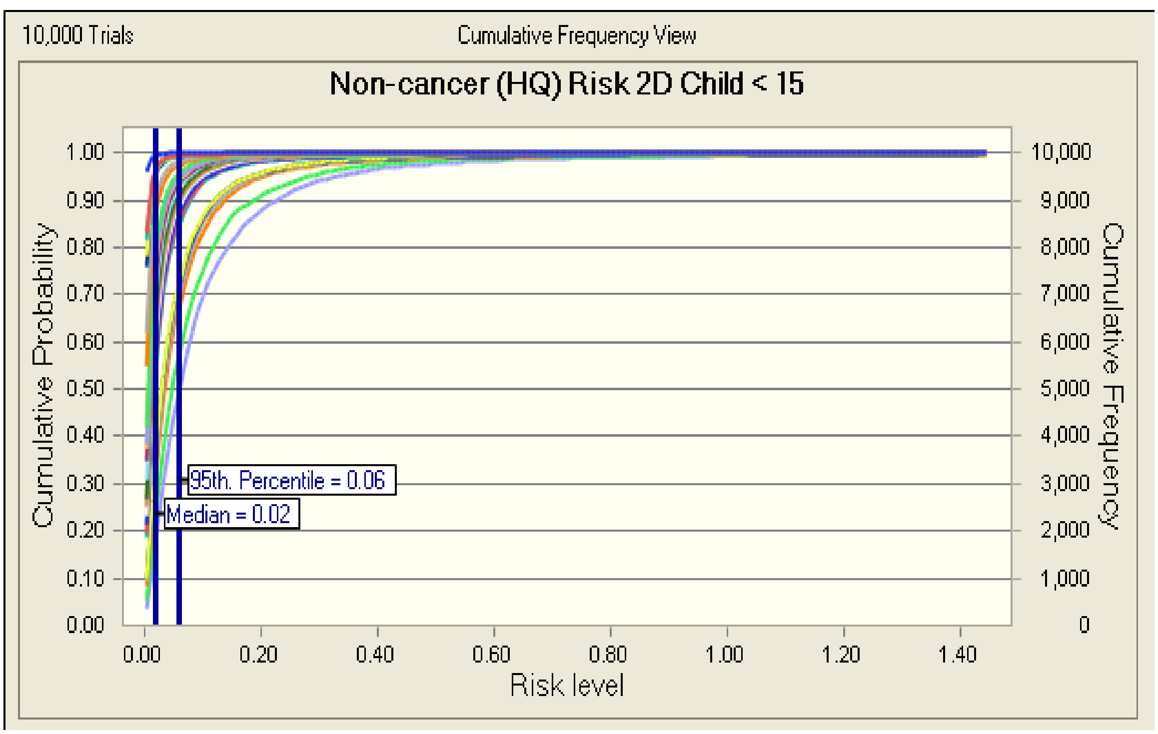

Figure C-4. Cumulative probability frequency non-cancer risk children (< 15 years) 2D