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PCBs in the Environment near the Oak Ridge Reservation– a Reconstruction of Historical Doses and Health Risks

Submitted to the Tennessee Department of Health by

ChemRisk
A Service of McLaren/Hart

OAK RIDGE HEALTH STUDIES
OAK RIDGE DOSE RECONSTRUCTION

– TASK 3 REPORT –

PCBs IN THE ENVIRONMENT NEAR THE OAK RIDGE RESERVATION– A RECONSTRUCTION OF HISTORICAL DOSES AND HEALTH RISKS

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McLaren/Hart-ChemRisk

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EXECUTIVE SUMMARY

This report presents the findings of a multi-year assessment of the releases of polychlorinated biphenyls (PCBs) from the Oak Ridge Reservation (ORR) and the potential for adverse effects in the local populations. PCBs were used and released from the ORR, and large numbers of individuals received doses by multiple routes of exposure. These routes included exposure to PCBs in air, water, sediments, biota (fish and turtles), and possibly from the sale and use of contaminated surplus oil. For many individuals, the exposures resulted in dose rates that are of little toxicological concern. However, for the farm families that lived along East Fork Poplar Creek (EFPC), and for persons who ate fish from Watts Bar and the Clinch River, the exposures could have resulted in both carcinogenic and noncarcinogenic effects. An accurate evaluation of exposures to the farm families is not possible due to the absence of relevant measurements of PCBs in the soils of the farms. However, sufficient data were available to assess the risks to anglers and their families.

A conservative estimate of the carcinogenic risks posed by the ORR releases to consumers of fish from Watts Bar Reservoir and the Clinch River range from less than 1 in a 1,000,000 to 2 in 10,000. Based on these risk estimates and the number of anglers using the fisheries, three or less excess cases of cancer would be expected to occur in the populations of consumers of fish from Watts Bar Reservoir and Clinch River since the ORR began operations in the 1940s. Because the estimates are conservative, the actual risks and expected number of cases are likely to be smaller and could be zero.

Persons who consumed large amounts of fish from the Clinch River and Watts Bar were at risk from the noncarcinogenic effects of PCBs. However, for Watts Bar these risks were primarily due to sources of PCBs other than that of the ORR; the releases from the ORR appear to have placed an additional one to two percent of the total number of fish consumers potentially at risk. This percentage corresponds to approximately 1,000-2,000 fish consumers over the last 50 years. The number of individuals actually experiencing adverse effects as a result of the ORR releases would be much smaller. The current data are insufficient to characterize the exact number of individuals experiencing adverse effects or the nature and severity of the effects. This uncertainty is largely due to the fundamental limitations in the understanding of the toxicology of PCBs in humans.

The project team recommends a number of additional areas of study that could reduce the uncertainty in the estimates of carcinogenic and noncarcinogenic risks. These include additional soil and sediment sampling, additional modeling of PCB exposures to Watts Bar fish consumers, and additional modeling of PCB dose responses in humans.

Beginning with early operations in the 1940s, PCBs were used extensively on the ORR. PCBs were present at Oak Ridge for several reasons:

- The ORR was one of the largest users of electrical energy in the United States. Because of their superior insulating properties and thermal stability, PCBs were present in many electrical components (such as transformers and capacitors) that were used at the government complexes.

- The ORR was also the site of a number of machining operations for the fabrication of metal weapons parts and related process equipment. PCBs were used as cutting fluids for lubrication and cooling during certain metal working operations.

Preliminary screening analyses conducted during the first phase of the Oak Ridge Health Studies, the Dose Reconstruction Feasibility Study, indicated that PCBs potentially represented the most important non-radioactive cancer-causing chemical historically released from the ORR. In the Feasibility Study, the study team also found that sources of PCB releases to the environment and the resulting exposures to local populations were poorly characterized. As a result, Task 3 of the Oak Ridge Dose Reconstruction was initiated to reconstruct PCB doses to human populations living in and around Oak Ridge.

The objectives of Task 3 were to:

- investigate historical releases of PCBs from the government complexes at Oak Ridge,
- evaluate PCB levels in environmental media in the ORR area,
- describe releases of PCBs from other sources in the Oak Ridge area, and
- evaluate potential human exposures and health impacts associated with the historical presence of these contaminants in the environment.

During the first 30 years of operations at the ORR, little or no attention was paid to the use, disposal, or contamination of the environment with PCBs. Few attempts were made to control the release of PCBs to the environment during this period, and minimal effort was made to track or document the amounts of PCBs used, disposed of on site, or released off site. This was because the carcinogenicity of PCBs in laboratory animals was not discovered until the 1970s. In 1977, the manufacture of PCBs was banned in the U.S. because of evidence that PCBs accumulated in the environment and caused harmful effects.

In the absence of detailed historical records regarding PCB use and disposal at the ORR, it was necessary for the project team to identify and evaluate all available information regarding processes and disposal practices at the ORR that might have resulted in the release of PCBs to the environment. Data were obtained from a variety of sources. Publicly available documents prepared by ORR contractors, the Tennessee Valley Authority (TVA), and the USEPA were obtained and reviewed. Historical records maintained at the ORR were also reviewed to identify relevant processes, accidental spills, and general disposal practices that might have resulted in releases of PCBs. In addition, information regarding undocumented historical events was obtained through interviews with active and retired employees of the ORR and residents of Oak Ridge living adjacent to the facilities. A detailed system of data management was maintained by the project team to ensure that the information collected was thoroughly evaluated and that the sources were clearly identified.

In general, it appears that the primary uses of PCBs at the ORR were electrical equipment (i.e., transformers and capacitors), hydraulic fluids, heat-transfer fluids, and cutting oils. In addition, PCBs were present in relatively low levels (less than a few percent) in many products including paints, coatings, adhesives, inks, and gaskets. PCB uses, as well as the potential impact of off-site releases, differed among the three main complexes on the ORR. Therefore, separate analyses are presented for each of the facilities.

PCB contamination at Y-12 can be traced to several general sources, including electrical systems (i.e., transformers and capacitors), the use of PCB-containing cutting oils for the machining of enriched uranium, and the "Z-oil" cooling system that was associated with the electromagnetic process for enrichment of uranium. In addition, PCBs were introduced to some of the hydraulic systems throughout the plant. Although it is possible that PCB-contaminated oils from transformers were disposed of at on-site burial grounds, it appears that the majority of the oils were recycled whenever possible. This indicates that transformers were probably not a significant source of PCB releases.

Limited information is available regarding waste disposal activities during the early years of operation at Y-12. It has been suggested that liquid wastes generated at Y-12 prior to 1950 were disposed at burial facilities at X-10. In the early 1950s, Y-12 began operation of a number of waste management units at the Bear Creek Disposal Area. Although records indicate that some portion of the waste oils generated at Y-12 may have been sold to the public at K-25, it appears that the majority was disposed at the Bear Creek Disposal Area. Throughout the course of its operation, the Bear Creek Disposal Area contained three principle disposal sites, including the S-3 Ponds, the Burial Grounds, and the Oil Landfarm.

The most likely routes of exposure to off-site populations from PCBs used at Y-12 were due to transport of PCBs to the sediments, surface water, and biota of Bear Creek and East Fork Poplar Creek (EFPC). Historical concentrations in Bear Creek were likely related to the disposal of PCB-containing oily wastes at the Bear Creek Disposal Area, while those measured in EFPC were likely due to unregulated outfalls discharging to the creek and incidental leaks and drips from the Z-oil system. Of the streams, EFPC was the more likely source of exposure, as it flowed through non-restricted and residential portions of the city of Oak Ridge, while most of Bear Creek was contained within the restricted access area.

Potential exposures to PCBs in EFPC could have occurred via a number of pathways, including incidental ingestion of and dermal contact with contaminated water and sediments during recreational activities, consumption of contaminated fish and other aquatic organisms, consumption of meats and dairy products from livestock grazed on floodplain soils, consumption of vegetables produced on floodplain soils, and dermal contact while gardening on flood plain soils. Although the creek has never been large enough to support a substantial recreational fishery, interviews conducted with Oak Ridge residents have indicated that fishing for minnows and other small fish from EFPC was a relatively common activity, and that fish caught in EFPC may have been consumed. Additionally, turtles from the creek were occasionally consumed.

Inhalation of airborne PCBs could have occurred as a result of the open pit burning activities at Burial Ground A. Low temperature combustion, such as occurred in the Burn Pit and the Burn Tank, may have resulted in aerial transportation of limited quantities of both vapor phase and particulate-bound PCBs. In addition, incineration of PCBs under poorly controlled conditions could have resulted in the formation of chlorinated dioxins and furans. Based upon the data collected, it is unlikely that oils containing high concentrations of PCBs were incinerated. Waste oils containing high concentrations of PCBs are nonflammable and would have been disposed in burial pits. In addition, the only documented wastes with high concentrations of PCBs (the cutting fluids) were disposed in the 1970s after the practice of burning of waste oils had been discontinued. It is possible, however, that wastes containing lower concentrations of PCBs (up to several hundred parts per million) could have been burned at the facility, potentially resulting in PCB levels in ambient air and also causing the formation of low levels of chlorinated dioxins and furans.

<u>The K-25 Plant for enrichment of uranium by gaseous diffusion</u> began operating in 1945 and continued to operate until the 1980s. The primary use of PCBs at K-25 was in the electrical power system for the gaseous diffusion cascades. Very large amounts of electrical power were brought into K-25 from a number of sources and distributed to the pumps that supported the gaseous diffusion process. Transformers, capacitors, and other types of electrical equipment were filled with dielectric fluids comprised of either mineral oils or mixtures of PCBs and other nonflammable compounds called Askarels. Though relatively costly, PCB oils were used in a number of types of electrical equipment, such as the cascade transformers, due to their fire retardant properties.

Based on plant records and interviews, approximately 125,000 gallons of PCB oils were contained in about 200 electric transformers and 10,000 capacitors used at K-25 during its period of operation. The majority of these PCB fluids were removed and incinerated off-site in 1989 and 1991. During this process, it was estimated that between 5,000 and 10,000 gallons of Askarel fluid either remained behind in the carcasses of the transformers or was otherwise unaccounted for.

A large number of electrical transformers were also used at the electrical switchyards at K-25. Although the fluids used in the switchyard transformers were mineral oil-based, they contained PCBs due to cross-contamination with Askarel fluids. In addition, some of the ancillary equipment at the switchyards contained PCBs.

Felt gaskets impregnated with PCBs connected sections of ventilation systems in the gaseous diffusion plant buildings. Lubrication oil, which did not contain PCBs, condensed on the inside of the ventilation systems, leached PCBs from these gaskets and, in certain instances, dripped onto the floors. While the total volume of lubrication oil in the ventilation duct gaskets in the three process buildings was estimated at 232,000 gallons, the potential for off-site migration of these PCBs was low. The oil dripped from gaskets located indoors and was cleaned up with dry sorbants that were subsequently landfilled. In addition, a trough system was installed in 1989 to capture any leaking oil. Gaskets are still in place due to a moratorium on removal of PCB equipment.

Historical waste areas that likely contributed to off-site releases of PCBs from K-25 included the burial grounds, holding ponds, leaking electrical equipment, and outdoor storage areas where surface runoff could have transported PCBs to Poplar Creek and the Clinch River.

PCBs were likely released from a number of sources at K-25 and likely entered Poplar Creek and the Clinch River through numerous discharge points. These releases resulted from a wide range of activities. Although the relative sizes of the releases are unclear, it appears that PCBs were primarily released from K-25 in small quantities associated with their widespread use in electrical equipment. In addition, PCBs likely migrated off-site as a result of storm water runoff, drainage from process areas, discharges from waste water in on-site holding ponds, and flooding events at the waste storage areas. The proximity of K-25 to Poplar Creek and the Clinch River suggests that the PCBs released from K-25 may represent a significant portion of the total aquatic releases of PCBs from the ORR. Investigations of burial grounds, burn areas, holding ponds, electrical switchyards, and outside storage areas indicated that, although the majority of PCBs released from these areas would have been contained on-site, off-site migration via surface runoff, waste water discharges, and volatilization to air likely occurred. Based on review of the historical uses and disposal practices at K-25, potential off-site exposures are likely to have been associated with the presence of PCBs in sediments and biota of Poplar Creek and the Clinch River. Potential exposures to PCBs near these water bodies could have occurred via a number of pathways, including dermal and oral exposure to contaminated water and sediments during recreational activities and consumption of contaminated fish and other biota. Interviews conducted with Oak Ridge residents have indicated that fishing was a relatively common activity and that many of the fish caught were consumed.

<u>The X-10 site</u> was built in 1943 and served as a pilot plant for demonstrating chemical techniques of plutonium separation. A network of underground storage tanks and pipelines was constructed in 1943 to handle and store the radioactive and chemical waste liquids generated by these separation operations. Once a laboratory wholly dedicated to nuclear technology research and development, X-10 presently includes multidisciplinary efforts in non-nuclear technologies and sciences. Historical activities at X-10 required electrical equipment such as capacitors, transformers, pumps, and electric motors. Lubricating and cooling oils associated with this equipment likely contained PCBs. The primary use of PCBs at X-10 was in the form of dielectric oils in electrical transformers.

Although records of the last 15 years indicate that releases from the facility have been negligible, measurable levels of PCBs exist in White Oak Creek Embayment and White Oak Lake. This indicates that PCBs have been released from X-10 operations. It is not clear whether these observed levels have resulted from releases that occurred prior to the late 1970s or from ongoing low level releases.

Warning signs and physical barriers restrict access to White Oak Creek, White Oak Embayment and White Oak Lake. These restrictions have been in place since operations began at the ORR. Posting the area likely prevented frequent use of these water bodies by the public, and exposures to sediment and surface water would have been minimal.

The migration of PCBs off-site from all of the ORR complexes was reduced due to the chemical properties of PCBs. PCBs are not mobile in groundwater and, when released to surface water, quickly bind to sediment. Because of these properties, the majority of PCBs placed in burial grounds or pits have remained in or near these units. Small lakes, ponds, and lagoons have been an integral part of the storm water and waste water management systems at the ORR. These surface water bodies have served as traps for PCBs, PCB-contaminated oils, and PCB-contaminated sediments, and have limited the movement of PCBs off the reservation. Finally, the sediments of Bear Creek, White Oak Creek, and other streams located on the ORR have entrapped a portion of the PCBs released from Y-12 and X-10 and have reduced the amount of PCBs migrating off the reservation.

Although releases to surface water and sediment transport represent the primary transport routes of PCBs to off-site locations, it also necessary to consider other, less significant pathways. For example, there is evidence that burning of PCB-contaminated material associated with both the Y-12 Burn Yards and the Toxic Substances Control Act (TSCA) Incinerator at K-25 may have resulted in the air releases of PCBs, as well as dioxins and furans produced from the partial incineration of PCBs. In addition, there is evidence to suggest that materials containing PCBs, such as used oils or electrical equipment, may have been sold and transported off-site.

It has long been recognized that PCBs were used in a large number of facilities throughout the watershed of the Tennessee River and its tributaries. Because PCBs have been detected in sediment and fish from the Tennessee River above its confluence with the Clinch River and in the Clinch River upstream of the ORR, the project team collected data on other sources of PCBs entering Watts Bar Reservoir. Available records on PCB use identified more than 22 facilities that managed PCB-containing wastes on portions of the Tennessee River above the Clinch River and on the Clinch River above the ORR.

Two independent approaches were utilized by HydroQual, Inc. in the early 1990s to evaluate the relative fraction of PCBs in Watts Bar fish attributable to Oak Ridge. One approach involved a straightforward analysis of spatial trends in fish monitoring data. The other approach entailed the development of a sediment transport model, which was used in conjunction with PCB sediment core data to predict sources and transport of PCBs at various times and locations within the watershed. Based on the results of their analyses, HydroQual concluded that historical releases of PCBs from Oak Ridge were responsible for less than 9 to 13 percent of the currently observed levels in Watts Bar fish. HydroQual also concluded that this estimate could be further reduced if sources of PCBs above Melton Hill Dam were considered. In addition, because of the approximate agreement between these two independent measurements, HydroQual concluded that there was strong evidence that the vast majority of PCBs currently detected in fish in the lower Watts Bar occurred as a result of releases to the Tennessee River upstream of the Clinch River. The analyses also indicated that, with the exception of three periods of elevated discharges, PCB releases to the Clinch River from all ORR sources were relatively constant over time, and the total magnitude of annual PCB releases from the 1940s through the 1990s were on the order of nine kilograms per year.

Based on the information obtained regarding releases of PCBs to the environment, the project team identified potential off-site exposure pathways that were primarily associated with releases to surface water and to air. Releases to surface water are primarily associated with White Oak Creek, Bear Creek, EFPC, Poplar Creek, the Clinch River, and Watts Bar Reservoir. In general, exposure pathways associated with releases to surface water have included fish consumption, dermal contact with surface water and sediments, and incidental ingestion of surface water and sediments. In addition, based on the available information regarding historical activities in the area, direct contact with flood plain soil as well as pathways associated with bioaccumulation of PCBs in vegetation and animals have been identified as complete exposure pathways for EFPC. Exposure pathways associated with bioaccumulation of PCBs in animals have been identified for Jones Island and the Clinch River. Exposures related to PCB releases in air have included both direct pathways, such as inhalation, and indirect pathways such as bioaccumulation of PCBs in vegetation and animals. These pathways are also likely complete for dioxins and furans that may have been formed during the incineration of PCBs.

The project team identified potential exposures associated with the historical sale of PCB-containing materials. Waste oils containing less than 500 ppm of PCBs may have been sold by the ORR facilities in the late 1940s. Such oil could have been used by local individuals for fuel, dust suppression, or vegetation control. Exposure pathways considered included direct contact with contaminated soil.

Based on the available information, the project team determined that developing quantitative estimates of PCB releases from specific release points as a function of time (often called “source terms”) would be difficult, if not impossible, for the following reasons. The first problem is the widespread use of PCBs on the ORR and the absence of documentation of releases. From the initial construction of Oak Ridge through the early 1970s, PCBs were viewed as nontoxic, inert substances that offered no particular hazard to workers, the general public, or the environment. As a result, there was no attempt to manage or track the use, release, and disposal of PCBs in any systematic manner. A second problem is that a PCB release that may have happened as a result of a specific event may result in an extended, low-level source of contamination to a body of water. Once PCBs enter a body of water, they may remain localized until a storm event or a change in the fundamental hydrology results in remobilization and additional transport. Rather than basing the Task 3 risk assessments on quantitative estimates of quantities of PCBs historically released, the project team estimated past exposures largely based on available environmental measurements of PCBs. Air-related pathways were an exception. These pathways were evaluated using estimates of releases and air dispersion models.

The assessment of exposures from the releases of PCBs requires information on the toxicity of the compounds and in particular the doses that are associated with adverse effects in animals. Section 4.0 presents a summary of the available information on the dose response of PCBs in humans and test animals. Dose-response assessment is the process of characterizing the relationship between the dose of an agent administered or received and the incidence of an adverse health effect in an exposed population. Dose-response relationships are developed based on animal studies and models of what might occur in humans or on human epidemiological evidence when such data are available. The toxicity value for the assessment of carcinogenic effects is the cancer slope factor (CSF) and the reference dose (RfD) is an estimate of daily exposure that is without appreciable risk of adverse noncarcinogenic effects. The RfD is significantly below the No-Observed-Adverse-Effect-Level (NOAEL) and well below the Lowest-Observed-Adverse-Effect-Level (LOAEL).

The regulatory processes used in setting the toxicity values are intended to be conservative in the face of uncertainty. As a result, the estimates of the RfD and CSF for PCBs are biased. That is, they are intended to be values that have a high probability of overestimating actual risks. Section 4.0 discuses the sources and magnitude of the biases.

An exposure pathway identified in Section 3.0 is the accumulation of PCBs in fish and the resulting exposures to the anglers and their families who consume the fish. An important issue in evaluating fish consumption is the frequency and amount of fish that a person consumes. Fish consumption varies greatly across the local population, with some individuals consuming no fish and others obtaining a large amount of their protein needs from the consumption of fish. The types of anglers vary as well and include commercial, recreational and subsistence anglers.

Section 5.0 evaluates the information available on historical fishing activities on the water bodies of interest, identifies potentially exposed populations of fish consumers, and derives estimates of fish consumption rates for the populations that were likely exposed as a result of their fishing activities. Based upon this assessment, eight distinct populations may have received exposure to PCBs through consumption of fish from water bodies in proximity to the ORR. These populations include: commercial anglers who fished Watts Bar or Clinch River/Poplar Creek; recreational anglers who fished Watts Bar, Clinch River/Poplar Creek, or EFPC; and subsistence individuals who may have fished any of these water bodies. Section 5.0 also discusses the data and methodologies used to develop both the point estimates of consumption used in the level I evaluation (Section 6.0) and the distributions used in the level II and III evaluations (Sections 7.0 and 8.0).

As described above, potential exposure pathways that were identified included direct exposure to PCBs in water, sediment, flood plain soils, and air, as well as indirect exposure through the ingestion of contaminated food (such as fish, vegetables, beef, and milk). The project team collected site-specific demographic information regarding farming, fishing, and recreational activities through interviews with current and past residents of Oak Ridge. The project team used this site-specific information, as well as measured levels of PCBs in the various media of concern, to confirm which of the possible exposure pathways actually resulted in exposures to off-site populations. Those that were determined to be “complete” exposure pathways were considered in the level I evaluation to identify the pathways most likely associated with risks to off-site populations.

In the level I evaluation, described in Section 6.0 of this report, the project team characterized the exposure pathways by medium, selected conservative upper bound exposure parameter values, and developed exposure point concentrations in order to estimate potential exposure intakes. These intake estimates were then combined with toxicity values to estimate the risks associated with each pathway. These estimates of risks were compared to screening criteria to determine which pathways represented the pathways most likely to result in risks to off-site populations. The screening criteria or decision guides used were an excess cancer risk of 1 in 10,000 and a nominal hazard quotient (the estimated dose divided by the USEPA Reference Dose) equal to 1 for noncancer health effects. If risk estimates for pathways were below the decision guides, these pathways were set aside from further evaluation. Likely off-site populations were identified for those pathways for which the estimated risks exceeded the decision guides. Because of the conservative exposure and toxicity assumptions used in deriving these estimates of risk, the findings of the level I evaluation can not be taken as evidence of actual risk. The estimates are best viewed as indications of pathways that warrant additional study.

Exposure point concentrations (EPCs) for the level I evaluation were based on historical data obtained from a variety of sources including studies performed by TVA and DOE. These historical data were limited, however, particularly for periods prior to the 1970s. Because the evaluation was a retrospective analysis, concentrations reported for any year were treated equally. Soil and sediment samples taken at depth were also considered because these samples may have represented historical EPCs. Because of the conservative nature of the level I evaluation, the EPCs for soil, sediment, surface water, drinking water, and aquatic biota were defined as the maximum total PCB concentration for each medium for each water body. The EPCs for the direct air pathways were modeled using a conservative Gaussian air dispersion model, SCREEN3. For the indirect air pathways, the EPCs were derived by predicting the concentrations in vegetables, beef, or milk based on measured or estimated concentrations in various media.

Using site-specific information pertaining to historical activities and the pathways identified, the project team identified five off-site populations potentially exposed via the pathways identified during the screening evaluation:

- farm families that raised beef, dairy cattle, and vegetables on the flood plain of EFPC;
- individuals who may have purchased beef and milk from cattle raised in the EFPC flood plain;
- commercial and recreational fish consumers;
- individuals that may have consumed turtles; and
- users of surface water for recreational activities.

The sizes of these populations vary greatly. The number of people eating fish from EFPC and the number of farm families are expected to have been small, perhaps less than 20 individuals. In contrast, it is estimated that the number of individuals (anglers and their families) who consumed fish caught in Watts Bar and the Clinch River in the years since the ORR activities began is greater than 100,000 individuals.

The exposure pathways that exceeded the decision guides (a nominal hazard quotient of 1 or cancer risk of 1×10^{-4} or one in ten thousand) in the level I evaluation are listed in Table ES-1. These pathways were further evaluated in a second assessment (level II evaluation), which is described in Section 7.0. In focussing on the pathways that yielded doses above the decision guides, no judgement was made on the acceptability of the risks associated with the pathways that yielded doses below those guide values.

Instead, the project team focussed its efforts to further refine exposures and risks on those pathways that exceeded the decision guides in order to focus project resources on those sources of exposure that had the highest potential for harm.

The level I evaluation used a conservative estimate of intake and risk to identify those pathways potentially associated with risks to off-site populations. This analysis was based on a determination of whether there was evidence that an individual in an exposed population could have a risk greater than the decision guides. No determination was made on how likely such a risk would actually occur or the fraction of the population exposed by a pathway would be affected. A proper evaluation of the risks to populations exposed by one or more pathways should focus on the heterogeneity of the population and the uncertainty in the estimates of exposure and risk.

Table ES-1: Exposure Pathways Retained for the Level II Evaluation

Location	Direct Exposure Pathways	Indirect Exposure Pathways
East Fork Poplar Creek	Ingestion of sediment Ingestion of soil Skin contact with soil	Ingestion of fish Ingestion of beef contaminated from soil Ingestion of beef contaminated from pasture grown on contaminated soil Ingestion of milk contaminated from soil Ingestion of milk contaminated from pasture grown on contaminated soil Ingestion of vegetables from contaminated soil
Poplar Creek		Ingestion of fish
Clinch River		Ingestion of fish
Watts Bar Reservoir		Ingestion of fish

All of the exposed populations have large amounts of variation (heterogeneity) in the dose that specific individuals receive. Certain individuals may receive trivial exposures (e.g., a child who visited EFPC once but did not come in contact with contaminated sediments) or be significantly exposed (e.g., a child who regularly played in the creek). Therefore, it is critical to determine the fraction of the exposed population that received dose rates associated with levels of concern.

A second issue is the level of confidence that can be attributed to an estimate of exposure and risk. For certain pathways, there is a high confidence that exposure actually occurred; however, for others the evidence is suggestive but incomplete. PCB exposure from consumption of contaminated fish is well documented. In contrast, exposures to the families on farms that bordered EFPC are quite uncertain, because the extent of historical levels of PCB contamination in soils at the affected farms is largely unknown. As discussed above, the dose-response estimates for PCBs are also highly uncertain. Because of these uncertainties, it is important to determine the uncertainty in the estimates of risk made for various portions of the exposed populations.

The project team characterized the uncertainty and variation in risk to the individuals in the exposed populations in Sections 7.0 and 8.0. Section 7.0, the level II evaluation, characterizes the range of doses that plausibly occurred in the exposed populations given the available data. Similar to the level I evaluation, simple dose rate models were constructed for each pathway. Dose estimates were determined across populations using a numerical simulation technique called Monte Carlo analysis.

The level II evaluation handled uncertainty and variation in different ways. Variability was directly modeled and estimates of the distribution of doses in the population were directly determined. Uncertainty that occurred due to the lack of knowledge was addressed by using estimates of intake parameters that were biased with respect to uncertainty. Specifically, the parameter values were selected from the upper end of the range of plausible alternative values. The evaluation used the same toxicological criteria (reference dose and cancer slope factor) as used in the level I evaluation. As discussed in Section 4.0, these values overestimate cancer and noncancer risks. Thus, the distribution of cancer and noncancer risks across the population are believed to be overestimates of the distribution of actual risks in the populations.

The goal of the level II evaluation was to identify those exposed populations where there was some chance that a small fraction (five percent) of the population received risks in excess of the decision guides. Populations where this was not true were set aside and the remaining populations were subject to additional analyses. The additional analyses included a quantification of the uncertainty in the hazard assessment. In certain populations with risks in excess of the guides, there was insufficient information to allow additional assessments. When this occurred, the lack of information was identified as a critical data gap and included in the recommendations for additional research (Section 10.0).

The level II assessment evaluated risks to recreational fish consumers, commercial anglers, farm families, and recreational users of surface water bodies near the ORR. Where a population was exposed via multiple pathways, this analysis estimated the total intake received from all pathways. The project team developed distributions for those exposure parameters believed to make a significant contribution to the variation in the dose rate, based on the range of available data. The distributions were developed by fitting the available data to various distribution types (e.g., normal, lognormal) according to accepted methods. Examples and discussion of the different distribution types are presented in Appendix D. A summary of the exposure parameter values and distributions used for each population, as well as the assumptions and rationale on which they were based, are presented in Tables 7-2 through 7-6.

The level II evaluation demonstrated that there was considerable variation in both noncarcinogenic and carcinogenic risk estimates for all of the populations evaluated. The risks to adults and children did not differ greatly and, in most cases, the ranges in the risk estimates overlapped. Adults tended to have slightly higher cancer risks because their longer exposure durations resulted in higher lifetime average daily doses. Noncancer risks also tended to be slightly higher. The estimates of risk for the median (50th percentile) and 95th percentile of the cumulative distribution of exposures in each population are presented in Tables ES-2 and ES-3. These results are later revised based on a new approach developed in this work. The majority of the exposures for certain populations occurred from PCB sources other than the ORR. Estimates of cancer risk from a source of PCBs are directly proportional to the relative contribution of that source. Table ES-2 includes estimates of cancer risks associated with the ORR releases as well as all sources for certain populations. It is more difficult to separate the relative contribution of the ORR releases for estimates of noncancer risks, because the impact of the additional exposure is a function of the level of exposure from other sources.

Because the size of the populations of fish consumers were determined and the distribution of risks was calculated for each population, the assessment also determined the number of excess cases of cancer that would be expected to occur in the populations. Three or fewer excess cases of cancer are expected to occur in the populations of recreational consumers of fish from the Clinch River and Watts Bar since the late 1940s. Because the carcinogenic potency of PCBs used in this assessment is expected to overestimate risk, the actual number of cases are expected to be smaller and may be zero. No cases are expected to occur in populations other than the recreational fish consumers because of the small size of these populations.

The dose reconstruction results for each population of interest, stated in terms of the estimates of health risk given in Tables ES-2 and ES-3, can be summarized as follows:

- <u>Recreational Fish Consumers:</u> Cancer and noncancer risks for recreational fish consumers at EFPC were lower than for the other bodies of water. Risks for both adults and children were below the decision guides (with the exception of children at the 95th percentile where the nominal hazard quotient was 2 in level II evaluation). The lower fish consumption rates for EFPC, based on its poor quality as a fishery, accounted for the lower risk estimates.

  The risks for recreational fish consumers using Clinch River/Poplar Creek were higher than those at EFPC. The cancer risk for adults at the 95th percentile was 3×10^{-4}. The cancer risk estimate for children at the 95th percentile was less than the decision guide of 1×10^{-4}. The nominal hazard quotient exceeded the noncancer guide of one for both adults and children at the median and 95th percentile in level II evaluation.

  Recreational fish consumers using Watts Bar had the highest cancer and noncancer risks of the three water bodies due to higher levels of PCBs in the fish and greater fish consumption rates. The cancer risk estimate for adults was 6×10^{-4} at the 95th percentile and for children was 1×10^{-4} at the 95^{th} percentile.

  Similar to the Clinch River/Poplar Creek analysis, the nominal hazard quotients in the level II evaluation exceeded the noncancer guide of one for both adults and children at the median and 95th percentile. As discussed earlier, the contribution of ORR PCB releases likely represented 8 to 13 percent of the total amount of PCBs. Therefore, at most, 13 percent of the total cancer risk for recreational fish consumers using Watts Bar was attributable to the ORR. Because 13 percent of 6×10^{-4} is less than 1×10^{-4}, the ORR releases do not appear to have resulted in cancer risks that exceed the guides.

Table ES-2. Summary of Cancer Risks[1] for the Populations Evaluated in the Level II Analysis

	Waterway					
	Adult			Child		
Population	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek
Recreational Fish Consumer (All Sources)						
95th Percentile[2]	**--**[3]	**6 x 10^{-4}**	**3 x 10^{-4}**	**--**[3]	**1 x 10^{-4}**	**5 x 10^{-5}**
Median	**--**[3]	**4 x 10^{-5}**	**2 x 10^{-5}**	**--**[3]	**1 x 10^{-5}**	**5 x 10^{-6}**
Recreational Fish Consumer (ORR Releases)						
95th Percentile	**1 x 10^{-5}**	**8 x 10^{-5}**	**2 x 10^{-4}**	**4 x 10^{-6}**	**1 x 10^{-5}**	**3 x 10^{-5}**
Median	**8 x 10^{-7}**	**5 x 10^{-6}**	**8 x 10^{-6}**	**3 x 10^{-7}**	**1 x 10^{-6}**	**2 x 10^{-6}**
Commercial Angler (All Sources)						
95th Percentile	**--**[4]	**4 x 10^{-5}**	**2 x 10^{-5}**	**--**[4]	**9 x 10^{-5}**	**2 x 10^{-5}**
Median	**--**[4]	**4 x 10^{-6}**	**2 x 10^{-6}**	**--**[4]	**1 x 10^{-5}**	**1 x 10^{-6}**
Commercial Angler (ORR Releases)						
95th Percentile	**--**[4]	**5 x 10^{-6}**	**1 x 10^{-5}**	**--**[4]	**1 x 10^{-5}**	**8 x 10^{-6}**
Median	**--**[4]	**5 x 10^{-7}**	**8 x 10^{-7}**	**--**[4]	**1 x 10^{-6}**	**7 x 10^{-7}**
Farm Family						
95th Percentile	**2 x 10^{-3}**			**9 x 10^{-4}**		
Median	**1 x 10^{-4}**			**1 x 10^{-4}**		
Recreational User						
95th Percentile	**4 x 10^{-7}**			**2 x 10^{-7}**		
Median	**3 x 10^{-8}**			**2 x 10^{-8}**		

[1] Estimated risks for all sources includes sources of PCBs other than the ORR.

[2] Numbers are expressed in scientific notation; for example 5x10^{-8} equals 0.00000005.

[3] There are no identified sources of PCB releases to EFPC other than the Y-12 Plant.

[4] There has been no commercial fishing of EFPC.

Table ES-3. Summary of Noncancer Risks[1] (Hazard Quotients) for the Populations Evaluated in the Level II Analysis

	Waterway					
	Adult			Child		
Population	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek
Recreational Fish Consumer						
95th Percentile	**1**	**40**	**20**	**2**	**60**	**30**
Median	**0.1**	**4**	**2**	**0.2**	**5**	**2**
Commercial Angler						
95th Percentile		**40**	**6**		**50**	**7**
Median		**4**	**0.6**		**5**	**0.7**
Farm Family						
95th Percentile	**100**			**200**		
Median	**20**			**40**		
Recreational User						
95th Percentile	**0.05**			**0.09**		
Median	**0.005**			**0.01**		

[1] Hazard quotients reflect the contribution of PCB sources other than the ORR and are based on the Reference Dose for Aroclor 1254 established by EPA.

The percentage of PCBs in the Clinch River that is attributable to the ORR releases is not well defined. Up to one half of the PCBs present in Clinch River fish may have been contributed by other sources on the Clinch River. However, even if one-half of the PCBs were contributed by non-ORR releases, the ORR releases appear to have resulted in risks above the cancer decision guide.

- Commercial Anglers In general, the risks to commercial anglers were similar to, but slightly lower than, risks estimated for recreational fish consumers. At the 95th percentile, all of the cancer risks for these populations were equal to or less than 1×10^{-4}. The nominal hazard quotients for the 95th percentile adult and child were greater than one for the Clinch River/Poplar Creek anglers and were higher for the Watts Bar Reservoir anglers in the level II evaluation.

  As with recreational fish consumers, the risk estimates for the commercial anglers are affected by other sources of PCBs along the rivers. Because concerns for carcinogenic risk are already below the decision guides, further reduction by accounting for different sources would not change the findings of the analysis.

- EFPC Farm Family: The estimates of cancer and noncancer risks were higher for the farm family population than the populations of fish consumers. Carcinogenic risk estimates at the 95th percentile for both adults and children exceeded the cancer decision guide. Nominal hazard quotients exceeded the noncancer decision guide for the entire range of the uncertainty analysis in level II. It should be noted, however, that the actual concentrations of PCBs in soil at the farms were highly uncertain, and this uncertainty could not be characterized because of a lack of data. Thus, these estimates may not be representative of true hazards associated with those populations.

- EFPC Recreational User: Both the noncancer and cancer risks for the recreational user population on EFPC were below the decision guides; at the 95th percentile, cancer risks for adults and children were 4×10^{-7} and 2×10^{-7}, respectively. Similarly, the nominal hazard quotients for adults and children were both less than one.

Based on the results of the level II evaluation, the following conclusions were reached:

Populations exposed from recreational use of EFPC do not appear to warrant additional assessment. The conservative estimates of carcinogenic and noncarcinogenic risks developed in the level II evaluation did not exceed the decision guides at the 95th percentile.

Exposures to PCBs from the consumption of fish in EFPC were also low. Cancer risks were well below the cancer guide. The adult fish consumer was at the noncancer guide and the child was slightly above the guide. Based on these findings, the team considered performing additional analyses on the population. However, given the extremely limited productivity of the creek and the uncertainty in the estimates of fish consumption, no additional work on this population was performed.

Cancer and noncancer risks for the farm families greatly exceeded the decision guides. Thus, this population could be at risk from their PCB exposures and should be considered in additional analyses. However, the estimates of PCB exposures used in the assessment are highly uncertain due to the limited data on PCB levels in the farm soils. PCB levels in the sediments and flood plain of EFPC have been characterized to at least some degree in recent remedial investigations; however, the ranges of PCB concentrations historically present in the soils of the fields, pastures, gardens, and areas around the farm houses themselves were not investigated. Because of this data gap, Task 3 investigators were unable to further assess risks to the families.

Risks to the commercial and recreational fish consumers using Watts Bar and the Clinch River were similar, suggesting that future assessments need not separate the groups. The carcinogenic risks to each population were above the decision guide. However, when the contribution of the ORR was considered, risks at the 95th percentile were smaller and in the case of Watts Bar were below the decision guide. The Clinch River fish consumers at the 95th percentile exceeded the guide by a factor of less than two. In addition, the estimate of the total number of cases expected for all populations is small (less than three). Based on these findings, the team did not believe that additional effort to characterize the uncertainty in the carcinogenic risk estimates was warranted. Therefore, no additional work was performed on assessing the carcinogenic effects for these fish consumer populations.

Section 8.0 presents the level III evaluation, which was performed on recreational consumers of fish from Watts Bar Reservoir and the Clinch River, both adults and children. No additional modeling was performed for the commercial angler because their risks in the level II evaluation appeared to be similar or slightly lower than those estimated for recreational fish consumers. Thus, the level III evaluation for recreational fish consumers is assumed to be applicable to commercial anglers as well.

The assessment of noncancer effects in the level II evaluation suggests that the majority of commercial and recreational fish consumers may have been at risk. The vast majority of fish consumers had nominal hazard quotients that were greater than one. However, the current noncancer decision guide is based on the assumption that any dose that exceeds the RfD is of some toxicological concern. As discussed in Section 4.0, most RfDs represent doses that are substantially smaller than the actual doses that are protective of even sensitive individuals. As a result, there is a high degree of certainty that there is no risk with a dose below the RfD and that doses well above the RfD may also be without risk. For example, none of the populations (at the 95th percentile) received doses that were greater than the doses shown to actually cause adverse effects in test animals. Therefore, it is not clear that findings of nominal hazard quotients greater than one (i.e., a dose greater than the RfD) imply the occurrence of adverse effects.

Section 8.0 presents a characterization of the population threshold for the noncarcinogenic effects of PCBs and uses the characterization to better estimate noncarcinogenic risk. A population threshold is the highest dose that does not cause an adverse effect in an individual who is uniquely sensitive to PCBs. The project team characterized this threshold using EPA's methodology for setting RfDs and replacing the safety factors with distributions. The distributions were based on a review of the available literature on generic approaches and by the development of a PCB-specific distribution. The PCB-specific distribution is believed to present the best use of the available toxicological data on PCBs.

As discussed above, the level II evaluation was biased with respect to the uncertainty in the estimate of dose rates and dose response. In the level III evaluation, this uncertainty is quantitatively evaluated along with the information on the variation in dose rates developed in the level II evaluation. Over the past five years, techniques have been developed for modeling both the uncertainty and variability of dose rates in exposed populations. One technique, called two-dimensional Monte Carlo, uses a “nested loop” programming technique that requires additional model development and computer resources. The result is the distribution of risk estimates across the exposed populations and the uncertainty bounds on those estimates. These estimates take into account the uncertainty in the estimate of both dose and the population threshold of PCBs. The results of the two-dimensional Monte Carlo analysis differ substantially from the earlier assessments of noncarcinogenic risk. In the earlier assessments, the risks were defined in terms of the nominal hazard quotient (the ratio of the fish consumer’s dose to the reference dose set by the USEPA). In the two-dimensional assessment, the risks are characterized as the ratio of the dose received by a person to the actual population threshold. This ratio is referred to as the true hazard quotient. Because the estimate of the population threshold and the dose received by a fish consumer are uncertain, the true hazard quotient is described in terms of probability. For example, a fish consumer may be described as having a 50 percent chance of having a true hazard quotient of less than 0.1 and a 95 percent chance of having a true hazard quotient less than 0.5.

The two-dimensional analysis of uncertainty and variation in the noncancer risk estimates, sought to further characterize risks by: 1) determining the distribution of true hazard quotients across the population; 2) calculating the fraction of the population receiving doses above the threshold of PCB effects; and 3) evaluating the incremental contribution made by the ORR releases to the risks associated with exposure to PCBs from other sources.

Table ES-4 presents the results of the level III evaluation. As discussed above, the model calculated the true hazard quotients for the percentiles of the populations. These estimates of the true hazard quotients provide a more unbiased estimate of noncancer risk. As a result, the values tend to be much lower than the estimates of the nominal hazard quotients. The true hazard quotients indicate that the typical fish consumer (represented by the median) for either water body is not at risk (true hazard quotient is less than one). The highly exposed fish consumer (represented by 95th percentile) may be at risk, because the estimates of the true hazard quotients could exceed one.

The confidence intervals of the hazard indices calculated for each of the reference populations addressed in Level II or Level III assessments of exposures to PCBs are shown in Figure ES-1. These 90% confidence intervals are plotted along with NOAEL and LOAEL values that have been used by the USEPA as bases for reference doses for Aroclor 1254 and Aroclor 1016.

Table ES-4. Summary of Noncancer Risks (Hazard Quotients)[1] for the Populations Evaluated in the Level III Analysis

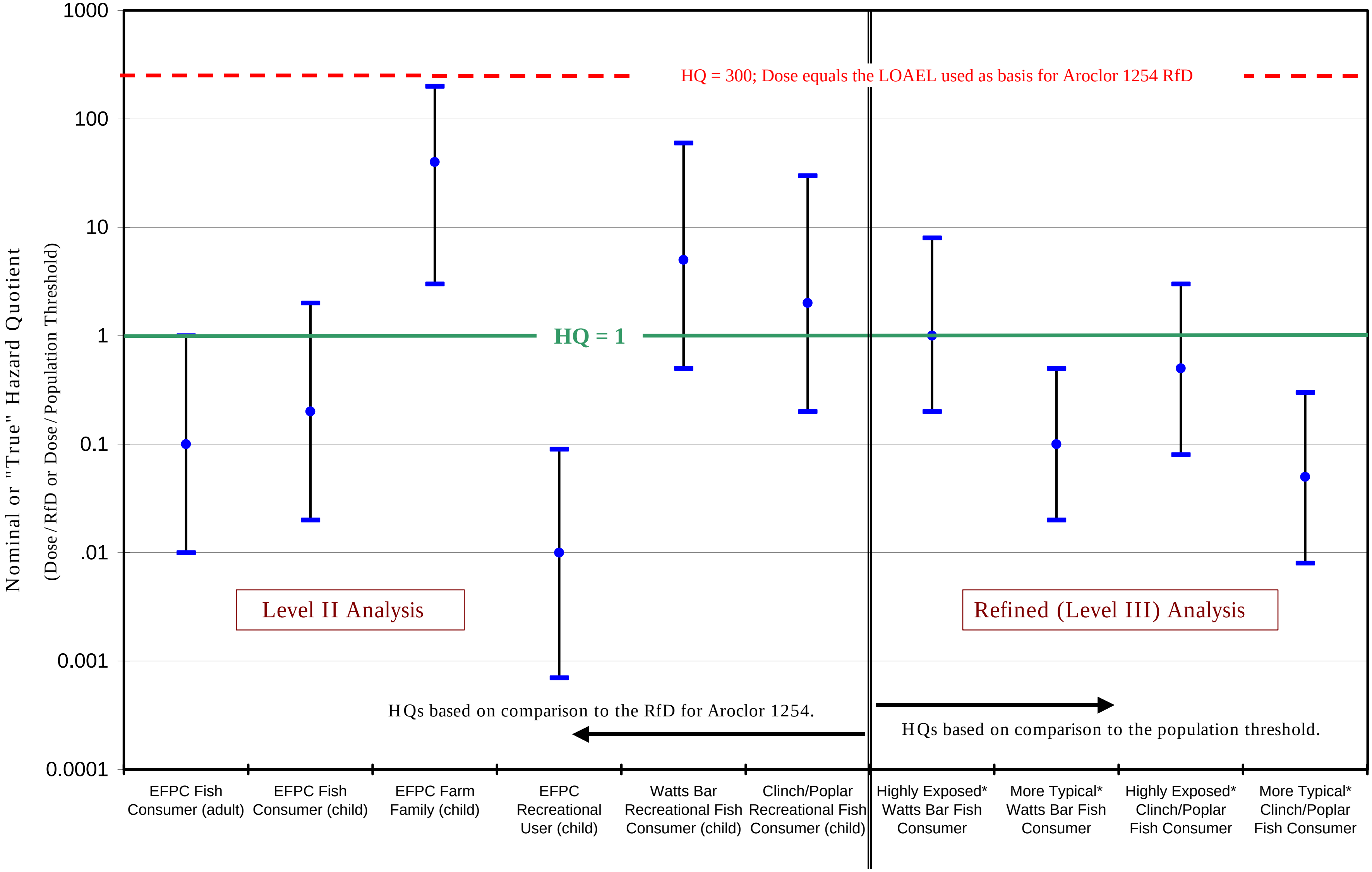
Population	Waterway Watts Bar Reservoir	 Clinch River/ Poplar Creek
Recreational and Commercial Fish Consumers		
95th Percentile	**1 (0.2-8)[2]**	**0.5 (0.08-3)**
Median	**0.1 (0.02-0.5)**	**0.05 (0.008-0.3)**

[1] Hazard quotients reflect the contribution of PCB sources other than the ORR and are based on the population threshold for PCBs.

[2] Estimate of the hazard quotient with 90% confidence limits.

Figure ES-1: Confidence Intervals of Hazard Quotients (HQs) Calculated for PCB Exposures

(Confidence intervals are represented by 5th, 50th, and 95th percentile values)



*The highly exposed fish consumer is represented by the 95th percentile of the population; the more typically exposed consumer is represented by the 50th percentile.

While the risk of noncancer effects can be evaluated in terms of true hazard quotients, the risks can also be evaluated more directly in terms of the fraction of the population that have received doses that exceed the population threshold (Table ES-5). The results of this assessment indicate that small portions of the fish consumer populations with very high fish intakes from Watts Bar Reservoir and Clinch River/Poplar Creek may have received doses in excess of the population threshold. However, it is not possible to determine the fraction of the population that actually experienced adverse effects.

Because of inter-individual variation in the tolerance to PCBs, the dose necessary to cause an adverse effect in the typical adult or child is higher than the dose that affects sensitive individuals. As a result, only a small fraction of those who receive a dose above the population threshold are expected to be affected.

The noncarcinogenic risks associated with the ORR releases are a function of both the size of the releases and the presence of other sources. In order to investigate the impact of the ORR releases, the exposure model calculations were done with and without the ORR contribution to PCB levels in Clinch River and Watts Bar fish and incremental changes in the ratios were determined. The analysis indicates that PCB contamination from background (non-ORR sources) resulted in a portion of fish consumers receiving doses greater than the population threshold. The releases from the ORR resulted in a small increase (one to two percent of the fish consuming population) in the fraction of individuals receiving doses greater than the population threshold (Table ES-5). The analysis also indicated that had the ORR releases happened in the absence of the other background sources, the releases would have been unlikely to result in adverse effects. Section 10.0 recommends additional studies of fish consumption rates, which would reduce the uncertainty and provide a clearer understanding of the fraction of the population receiving doses that exceeded the threshold.

Based upon the overall findings of Task 3, the project team recommended a number of additional areas of study that could reduce the uncertainty in the estimates of carcinogenic and noncarcinogenic risks (Section 10.0). These recommendations include additional soil and sediment sampling, additional modeling of PCB exposures to Watts Bar fish consumers, and additional modeling of PCB dose responses in humans.

Table ES-5. Percent of Population Receiving a Dose Above the PCB Population Threshold

Watts Bar Fish Consumer (Adult)			
	Background[1, 2]	Background + ORR	Change Due to ORR
Refined Empirical Distribution[3]	5.0 (0.61-39)	6.6 (0.82-43)	1.6
Watts Bar Fish Consumer (Child)			
	Background	Background + ORR	Change Due to ORR
Refined Empirical Distribution	7.5 (0.92-44)	8.9 (1.4-48)	1.4
Clinch River/Poplar Creek Fish Consumer (Adult)			
	Background	Background + ORR	Change Due to ORR
Refined Empirical Distribution	0.55 (0-9.7)	2.2 (0-21)	1.7
Clinch River/Poplar Creek Fish Consumer (Child)			
	Background	Background + ORR	Change Due to ORR
Refined Empirical Distribution	0.97 (0-15)	3.8 (0-29)	2.8

[1] Background reflects risks associated with PCB releases attributable to sources other than ORR. Based on the HydroQual analysis, 87% and 50% of the total PCB concentration in Watts Bar and the Clinch River/Poplar Creek, respectively, is assumed to be associated with other sources.

[2] Median value (90% confidence limits); values reported to 2 significant figures

[3] A set of distributions that reflect information on PCB toxicity (see Section 8.0)

Acute exposures- Short-term exposures. Acute exposures are typically defined as less than 14 days.

ADD - Average Daily Dose. Exposure expressed as the mass of a substance received by an individual per unit body weight per unit time (i.e., mg/kg-day), averaged over the exposure duration.

Adherence factor - The mass of soil retained on a given area of skin after contact with the ground.

Adipose tissue - Bodily connective tissue that contains stored cellular fat.

Antineoplastic – Having cancer-fighting properties.

Angler- Usually a person who fishes with line and hook. Used in this report to mean any fish consumer, including members of a family who ate fish caught by another person.

Aroclors- Commercial mixtures of polychlorinated biphenyls.

Askarels - Commercial mixtures of polychlorinated biphenyls and chlorinated benzenes.

Averaging time - The period of time over which a dose is averaged. Usually determined as a function of the toxicological effect being evaluated.

Benchmark dose modeling - A technique that uses the lower confidence limits of an estimate of the ED_{10} instead of the NOAEL.

Bioaccumulation - The net accumulation of a substance by an organism as a result of uptake from all environmental sources.

Biotransfer factors - Measure of an organic chemical's potential to accumulate in the tissues of an organism relative to the total uptake of the chemical.

Body burdens - Concentrations of a chemical accumulated in the tissues of an exposed organism.

Carcinogen - A substance capable of increasing the occurrence rate of cancer in either animals or humans.

CERCLA - Comprehensive Environmental Response, Compensation, and Liability Act

Chronic - Persisting over a long period of time. Chronic exposures are generally greater than 1/10 of expected lifetime, e.g., chronic exposures for humans are defined as greater than seven years (lifetime = 70 years).

Commercial angler - An individual who fishes as an occupation.

Congeners – Specific PCB molecules.

CSF - Cancer Slope Factor. The slope factor is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a carcinogen.

Demographic - Pertaining to the study of vital statistics of human populations, e.g., size, growth, density, and distribution.

Dermal - Of or relating to the skin.

Detection limit - The lowest chemical concentration in an environmental media that can be accurately quantified based on a specified analytical method.

Directed searches - Document searches aimed at collecting specific data or other relevant information identified by the project team.

DOE – The United States Department of Energy

ED_{10} - Effective Dose. The dose level at which 10 percent of the test subjects exhibit an expected response.

Edible tissue - That portion of the fish that is commonly consumed.

EFPC - East Fork Poplar Creek

Endpoint - The effect resulting from exposure to a chemical or physical agent.

Extrapolation - To infer or estimate by projecting or extending known information.

EPC - Exposure Point Concentration. The concentration of a chemical that will be contacted over the exposure period. The EPC is determined using available monitoring data for each specified media, or through chemical fate and transport models.

Epidemiology - The study of diseases and their occurrence in human populations.

Exposure - Contact of organism with a chemical or physical agent. Exposure is quantified as the amount of the agent available at the exchange of boundaries of the organism (e.g., skin, lungs, gut) and available for absorption. Exposure to the chemical may occur through direct means (e.g., ingestion of contaminated sediment, inhalation of contaminated air) or indirectly (e.g., consumption of fish exposed to contaminated sediments).

Exposure duration - The length of time (i.e., years) over which an exposure occurs.

Exposure frequency - The rate at which a particular exposure occurs (e.g., days/year).

Exposure parameter - Term or variable in an equation used to calculate the dose rate that occurs as a result of exposure to a contaminant.

Exposure parameter distribution - Range of numbers representing the possible values and associated probability of occurrence representing a particular exposure parameter. Exposure parameter distributions can occur in a number of forms, including triangular, lognormal, normal, cumulative, etc.. Each of these forms refers to the exact shape of the curve defined by the distribution. Mathematical and graphic descriptions for each form are provided in Appendix D.

Exposure pathway - The mechanism by which an agent reaches an organism. Each exposure pathway includes a source of releases to the environment, a process by which the contaminant reaches an individual, and a set of behaviors that define an individual’s interaction with the contaminants and the resulting dose received by the individual.

Incidental ingestion - Consumption of non-food material (i.e., soil, sediment, and water) that occurs during defined activities.

Intake rate - A measure of exposure expressed as the mass of a substance entering an individual per unit body weight per unit time (e.g., mg chemical/kg-day).

LADD - Lifetime Average Daily Dose. Exposure expressed as the mass of a substance received per unit body weight per unit time (i.e. mg/kg-day), averaged over an individual’s lifetime (i.e., 70 years).

LOAEL – Lowest Observed Adverse Effect Level. The lowest dose in a toxicological study at which there are statistically significant differences between the frequencies of adverse effects observed in exposed and control populations of test animals.

Lung deposition fraction - The fraction of inhaled particulate matter that is retained in the lung.

MF- Modifying factor

Monte Carlo - Numerical simulation technique that characterizes the range of values associated with the output of a model based on information regarding the uncertainty associated with each of the input parameters.

NOAEL - No Observed Adverse Effect Level. The highest dose in a toxicological study at which there are no statistically significant differences between the frequencies of adverse effects observed in exposed and control populations of test animals.

Nominal hazard quotient - The ratio of the calculated intake rate to a known or 'safe' dose such as the chronic RfD. In this report, the hazard quotient was determined based on EPA's chronic RfD of $2x10^{-5}$ mg/kg-day established for Aroclor 1254.

Noncarcinogenic - Health effects other than cancer.
Ocular - Of or relating to the eye.

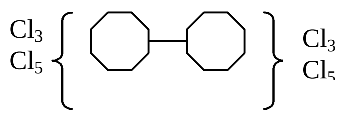
ORGDP - Oak Ridge Gaseous Diffusion Plant

ORNL - Oak Ridge National Laboratory

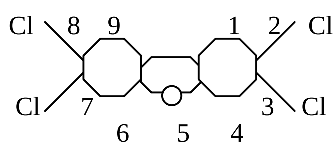
ORR - Oak Ridge Reservation

Particle emission factor - Relates the contaminant concentration in soil with the concentration of respirable particles in the air due to fugitive dust emissions.

PCB - Polychlorinated Biphenyl



PCDF - Polychlorinated Dibenzofuran



3,7,8 – Tetrchlorodibenzofuran (TCDF)

Photodegradation constant - Describes the rate of decay of a contaminant associated with exposure to sunlight.

Population threshold – The highest dose that does not cause a deleterious effect in the most sensitive individual in a population.

Population risks - An expression of risks in terms of the impacts on a defined population. Usually expressed as the number of cases of adverse effects occurring in a defined population.

Productivity - The amount of living tissue produced per unit time in an ecological system.

ppm - Parts per million

Pyranol - A commercial trade name for dielectric fluids that contain PCBs and transformers that contain such fluids.

QA/QC - Quality Assurance / Quality Control

Quantal data - Data associated with a toxicological effect that is characterized by an 'all or none' response, e.g., death of an individual.

Recreational angler - An individual primarily catching fish as a recreational activity.

Recreational user - An individual participating in outdoor activities that might result in exposure to contaminated soil, water, or sediment (e.g., swimming, picnicking, hiking, etc.).

RfD - Reference Dose. An estimate of the daily exposure to a chemical that is likely to be without appreciable risk of deleterious effects during a lifetime. Established by the U.S. Environmental Protection Agency.

Retrospective analysis - An historical rather than predictive analysis.

RM - River Mile; zero mileage is at the mouth of river.

SIOU - Surface Impoundment Operable Unit
Source term - An estimate of the mass of a substance released from a source per unit time.

SRA - Shonka Research Associates, Inc.

Steady-state - Not changing with time.

Subchronic exposure - Exposure to a chemical for a duration greater than acute, but less than chronic.

SWMU - Solid Waste Management Units

SWSA - Solid Waste Storage Areas

Systematic search – A document search aimed at ensuring that no potentially relevant information was overlooked.

Threshold – The highest dose that does not cause deleterious effects in an individual.

Total uncertainty - Characterization of uncertainty due to lack of information and variability in a population.

True hazard quotient – The ratio of the calculated intake rate to an estimate of the population threshold. In this report, the true hazard quotient was determined based on probabilistic model of EPA's chronic RfD of $2x10^{-5}$ mg/kg-day established for Aroclor 1254.

True uncertainty – Characterization of uncertainty due to lack of knowledge about the true value of some parameter.

Variability - Variations in a measured parameter that occur as the result of the natural heterogeneity associated with the parameter.

TSCA - Toxic Substances Control Act

TVA - Tennessee Valley Authority

Two-dimensional analysis - A model of uncertainty in risk where uncertainty and variability are separated.

UF - Uncertainty Factor (Safety Factor)

WAGs - Waste Area Groupings

1.0 INTRODUCTION

In July 1991, the Department of Energy (DOE) signed an agreement with the State of Tennessee to fund an independent health study of the population living around the Oak Ridge Reservation (ORR; Figure 1-1). The purpose of the study was to estimate exposures to chemicals and radioactive materials released at the ORR since 1942. The Dose Reconstruction Feasibility Study, the first phase of the project, began in May 1992 and was completed in September 1993. A goal of the Feasibility Study was to identify chemicals and radionuclides released from the ORR in the past 50 years, with the greatest potential for causing adverse health effects in individuals living off-site. The study was designed to determine the feasibility of estimating the doses of these contaminants, given the quality of the available information. The results indicated that a significant amount of information was available to reconstruct the historical releases and potential off-site doses; thus, the second phase or the Dose Reconstruction Study was initiated. This phase was designed to estimate the actual amounts or the "doses" of the contaminants received by people as a result of off-site releases. Contaminants that were identified during the Dose Reconstruction Feasibility Study were addressed as separate tasks during the Dose Reconstruction Study. This report presents the results of Task 3, which investigated the historical uses and releases of polychlorinated biphenyls (PCBs) from the ORR and off-site exposures to those compounds.

PCBs were used extensively at the ORR for several purposes. First, the ORR was one of the largest consumers of electrical energy in the United States during the 1940s to the 1980s. Because of their superior insulating properties and thermal stability, PCBs were used in transformers, capacitors, and other electrical equipment at the ORR. PCBs were also used as cutting fluids in the metal component fabrication operations at the ORR. Consequently, PCBs and mixtures containing PCBs were stored and used in large quantities at the ORR.

During most of the 50 years of PCB production and use in the U.S., little was known about the toxicological properties of PCBs. The carcinogenicity of PCBs in laboratory animals was not discovered until the 1970s; in 1977, the manufacture of PCBs was banned in the U.S. because of evidence that PCBs accumulated in the environment and caused harmful effects (EPA, 1986). Prior to the 1970s, no special precautions were taken with handling or disposal of PCBs or PCB-contaminated equipment at the ORR.

Because of the relatively high potential for possible releases, as well as the lack of information regarding actual uses or disposal activities, the screening analyses conducted during the Dose Reconstruction Feasibility Study indicated that PCBs potentially represented the most important non-radioactive carcinogenic chemical historically released from the ORR (ChemRisk, 1993). Although the Dose Reconstruction Feasibility Study identified PCBs as a chemical of concern to off-site populations, it did not determine the extent to which Oak Ridge area residents may have been exposed to PCBs released from the site.

Task 3 of the Dose Reconstruction Study was designed to reconstruct the PCB doses to human populations living in and around Oak Ridge. The objectives of Task 3 were to 1) investigate historical releases of PCBs from the ORR, 2) evaluate PCB levels in the ORR area, 3) describe releases of PCBs from other sources in the Oak Ridge area, and 4) evaluate the potential human exposures and health impacts associated with the presence of these contaminants in the environment.

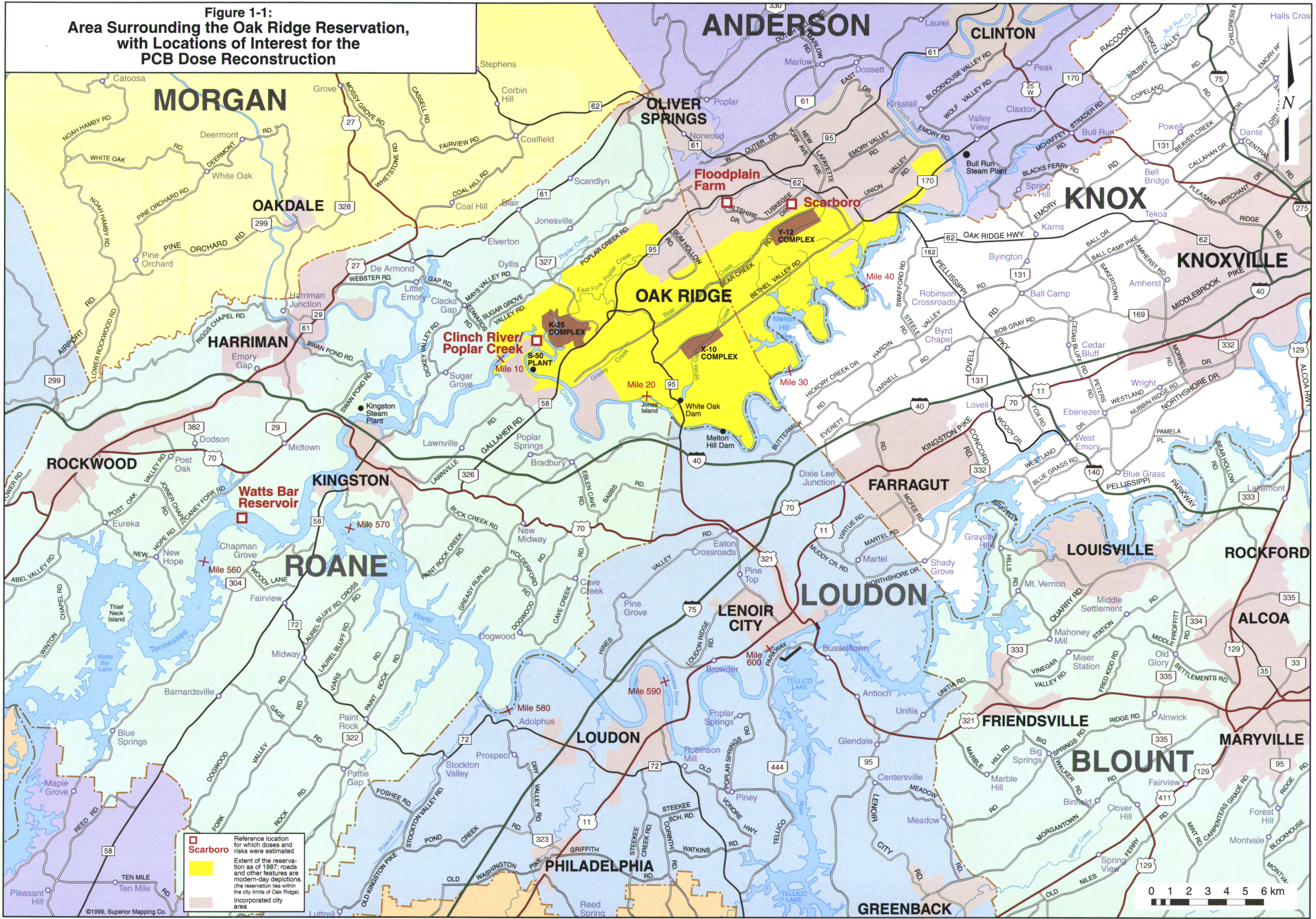
To achieve these objectives, the project team first characterized the history of PCB usage at the ORR. Investigators characterized the primary sources of PCBs, the mechanisms by which the PCBs were released, the media that received the PCBs, and the mechanisms by which the PCBs moved to off-site media to which humans could be exposed. The potential exposure pathways identified were then evaluated through an iterative risk assessment process, designed to produce refined estimates of exposure and associated risks from PCBs. Based on this iterative process, the project team provided a characterization of the potential magnitude and uncertainty of doses received by individuals in the off-site areas.

This document is organized into eleven sections, including this Introduction (Section 1.0) and References (Section 11.0). Section 2.0 describes the data collection and management methods used by the project team to organize and document the historical information on the uses and releases of PCBs at the ORR, as well as demographic information and data on environmental levels of PCBs. The history of PCB usage at the ORR is presented in Section 3.0. This section also identifies the mechanisms by which PCBs left the ORR and discusses the difficulties in characterizing specific locations and dates for PCB releases. Section 4.0 describes the toxicity of PCBs as characterized by studies in humans and test animals. This information is used to derive measures of hazard associated with low level exposures to PCBs such as the reference dose (RfD) (IRIS, 1998a,b) and the minimal risk level (MRL) (ATSDR, 1996). The toxicity data are also used to characterize the potential for carcinogenic effects (EPA, 1996a). In addition, Section 4.0 discusses the sources and magnitude of the uncertainty and bias in the toxicity estimates. Section 5.0 presents a detailed analysis of fish consumption rates for anglers fishing from waterbodies affected by the ORR's releases of PCBs.

Sections 6.0, 7.0 and 8.0 present an iterative quantitative risk assessment process for the evaluation of PCB exposure. The objective of this iterative process was to identify populations in the vicinity to the ORR that may have been at risk from exposure to PCBs and to determine the degree of risk to those populations. The project team began the process by aggressively identifying the potential routes of exposure and the populations affected by each route. The goal of this evaluation was to include all pathways that could be reasonably characterized.

Once identified, the pathways were subjected to a series of screening evaluations. The initial evaluation (level I) focused on those exposure pathways that resulted in the highest doses in off-site populations. To avoid the exclusion of pathways that deserved additional study, the process intentionally overestimated risks by relying on conservative assumptions for various exposure parameters. The process led to two separate categories of exposure pathways: those that resulted in low levels of risk, and pathways that resulted in higher doses that warranted additional analyses. The project team did not take a position on the acceptability of any specific level of risk, but used decision guides (a hazard quotient of 1 and 1×10^{-4} excess lifetime cancer risk) to

Figure 1-1:
Area Surrounding the Oak Ridge Reservation, with Locations of Interest for the PCB Dose Reconstruction



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determine which pathways would be subject to additional assessments and those which would be deferred from additional assessment at this time. In certain instances, pathways and associated populations were deferred from additional analysis if there were insufficient data to meaningfully reduce the uncertainty in the estimates of exposure or risk. In these cases the absence of data was identified as a data gap and included in the recommendations for additional studies. For example, recommendations were made for additional sampling of areas used for pasture around EFPC and for additional sediment sampling in the Clinch River.

Following level I evaluation, a second evaluation (level II) was conducted to estimate the distribution of doses and associated risks that would occur in the populations exposed via the relevant pathways. Populations were screened based on the estimate of risk to the 95^{th} percentile of a cumulative dose distribution. If an individual at the 95^{th} percentile of the population had a dose associated with a risk below the decision guide, that population was deferred from further consideration.

The risk estimates developed during the level I and II evaluations were intended to overestimate the risks and should be viewed with caution. The true risks to individuals in the populations are believed to be less than the values predicted by these assessments. In fact, risk estimates for exposure pathways and populations retained for additional analyses were generally lower in the subsequent assessments.

Once the initial evaluations were concluded, the remaining pathways and exposed populations were assessed in a two-dimensional risk assessment (Section 8.0). Unlike the initial evaluations, this analysis attempted to provide an unbiased estimate of the distribution of risks across a population and to fully disclose the uncertainty in those risk estimates. This evaluation considered the uncertainty in the estimate of the distribution of dose (exposure) and the uncertainty in toxicity estimates.

Findings and conclusions are discussed in Section 9.0. Based on our findings, the project team prepared a series of recommendations for additional research (Section 10.0). These recommendations focus on filling the data gaps identified in the analysis and on reducing the uncertainty in the final estimates of risk.

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2.0 DATA COLLECTION AND MANAGEMENT

The objective of Task 3 was to investigate all potential current and historical exposures to PCBs associated with activities at the ORR. In the absence of detailed historical records regarding PCB use and disposal at ORR contractors, it was necessary for the project team to identify and evaluate all available, relevant information regarding processes and disposal practices at the ORR that might have resulted in the release of PCBs to the environment. Data were obtained from a wide variety of sources to ensure that all relevant information was identified. Publicly available documents prepared by ORR contractors, Tennessee Valley Authority (TVA), and the USEPA were obtained and reviewed. General records maintained at the ORR were also considered to identify relevant processes, accidental spills, and general disposal practices that might have resulted in releases of PCBs. In addition, information regarding undocumented historical events was obtained through interviews with current and former employees of the ORR and residents of Oak Ridge living adjacent to the facilities.

Due to the amount of data involved in this analysis, it was necessary to maintain a detailed system of data management to ensure that the information collected was thoroughly evaluated and that the sources were clearly identified. In addition, because this process involved the reconstruction of historical events not previously recorded, it was critical that all components of the process (e.g., data evaluated, assumptions made, results) were clearly organized and documented so that the approach and resulting conclusions could be easily reviewed and validated.

2.1 DOCUMENT REPOSITORY SEARCHES

Numerous document repositories are maintained at the ORR. A list and detailed summary of each repository is provided in the Task 5 Plan (ChemRisk, 1994). With the guidance and support of Task 5, the project team conducted both directed and systematic searches of these repositories for information regarding past activities associated with PCBs. The purpose of the directed searches was to locate specific data or documents pertaining to PCBs or related activities for use by the project team. In contrast, systematic searches involved a methodical review of records contained within a specified repository so that no information relevant to dose reconstruction was overlooked.

2.1.1 Directed Searches

The purpose of directed searches was to collect specific data or other relevant information pertaining to a process or disposal practice identified by the project team. Each directed search focuses on a repository associated with the plant at which the activity of interest occurred. The project team reviewed all available listings regarding the contents of the repository to identify relevant documents. When a document or group of documents was identified, the project team submitted a document request form (Figure 2-1) to the respective site coordinator.

Figure 2-1. Example of Document Request Form

ChemRisk/Shonka Research Associates, Inc., Document Request Form

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(This section to be completed by subcontractor requesting document)

_______________ / _______________________________

Requestor *Document Center (is requested to provide the following document)*

Date of request __________ *Expected receipt of document* __________

Document number __________ *Date of document* __________

Title and author (if document is unnumbered)

____

____

(This section to be completed by Document Center)

Date request received _______________________

Date submitted to ADC _______________________

Date submitted to HSA Coordinator _______________________

(This section to be completed by HSA Coordinator)

Date submitted to CICO _______________________

Date received from CICO _______________________

Date submitted to ChemRisk/Shonka and DOE _______________________

(This section to be completed by ChemRisk/Shonka Research Associates, Inc.)

Date document received _______________________

Signature _______________________

2.1.2 Systematic Searches

While directed searches were conducted in response to an identified data need, systematic searches followed a random approach to ensure that no potentially relevant information was overlooked. Systematic searches focused on those repositories at Oak Ridge identified as having the most potential to yield useful information for the dose reconstruction (ChemRisk, 1994). Emphasis was placed on reviewing historical documentation not reviewed during the Dose Reconstruction Feasibility Study for information relevant to PCBs or PCB-associated activities. Systematic searches were conducted according to the methodologies specified in the Task 5 Plan (ChemRisk, 1994). Specifically, upon being directed to a specific repository by the Task 5 Coordinator, the project team evaluated any available listings regarding the contents of the repository to make an initial selection of documents and/or boxes to review for relevant information. A record of all boxes examined was maintained by the Task 5 Coordinator. In addition, a random sample was conducted of all non-selected documents and/or boxes, to increase the likelihood that all relevant information was identified. To aid in the systematic search, Task 3 project team members developed a list of data needs and relevant key words to assist with the identification of information relevant to PCBs (Table 2-1). This list was periodically updated to reflect new information and to fill data gaps as they became apparent.

2.2 INTERVIEWS

The collection and review of available data and written reports provided useful information regarding environmental levels of PCBs, as well as regulated uses of PCBs throughout the facilities. In order to obtain other relevant but anecdotal information, such as uses of PCBs prior to the implementation of regulations regarding their use, or land use in the vicinity of the ORR, the project team conducted interviews with active and retired personnel from ORR operations, management staff who occupied key positions at the ORR, and residents living near the ORR. These interviews were conducted either in person or by telephone. A written summary of each interview was generated by the interviewer and submitted for entry into the project database.

2.2.1 On-site Interviews

The Task 3 project team conducted on-site interviews with current and former employees of the ORR. These interviews were arranged by the site coordinator for the specific facility where the employee currently or historically worked. All original notes associated with these interviews were submitted to the appropriate ORR personnel for review for public release. In some cases, the interviewee also reviewed these reports. Following their release, the notes were returned to the interviewer, who prepared a written description for entry into the project database. Each summary form was carefully reviewed for accuracy prior to entry into the database.

2.2.2 Telephone Interviews

In addition to the on-site interviews conducted by the project team, some interviews with Oak Ridge personnel were conducted via telephone. These interviews were reserved primarily for

Table 2-1. Data Needs Summary and Guide for Task 3

Data Needs

- Information on the volume of PCBs purchased or used at ORR; trade names for PCBs include: Aroclor, Phenoclor, Fenclor, Askarel, Pyranol, Clophen, and Kanechlor
- Information about PCB disposal, spills, leaks, other releases at K-25, Y-12 or X-10, including transformer failures
- Environmental PCB measurements from Clinch River, Poplar Creek, Mitchell Branch, White Oak Creek, White Oak Embayment, and White Oak Lake. Also measurements from building outfalls, sumps or drain systems
- Information on disposal practices for oils or oily wastes
- Information on disposal of electrical equipment

Keywords

- Polychlorinated biphenyls OR PCBs OR Aroclor OR Askarel OR Pyranol OR Phenoclor OR Fenclor OR Clophen OR Kanechlor
- dielectric fluids
- non-flammable AND oils/paint/coatings
- transformers

those individuals who could readily discuss unclassified environmental data. However, due to the difficulty in arranging all necessary interviews during on-site visits, information on historical use and releases of PCBs was also collected during some telephone interviews. As with the on-site interviews, all original notes were submitted to the appropriate ORR personnel for classification review, and in some cases, the interviewee also reviewed these reports. Following release of the notes, the interviewer prepared a written description of the interview and submitted it for entry into the project information database. Each summary form was carefully reviewed for accuracy prior to entry into the database.

2.3 DATA MANAGEMENT SYSTEM

The Task 3 project team created a data management system to organize data, documents, document summary forms and interview summary forms. The system provided: 1) a means of organizing documents for easy storage and retrieval; 2) documentation of the specific information obtained; and, 3) a way of tracking information by source.

2.3.1 Documents and Document Summary Forms

The project team established document review procedures (Figure 2-2) to ensure that all documents were handled in a consistent and thorough manner. All documents were reviewed by a member of the project team, and a document summary form was submitted to the project information database. Copies of all documents, as well as the document summary forms, were maintained in a library dedicated to ORR references. A bibliographic database listing the primary author, year of publication, and location referenced (e.g., X-10 or Y-12) was maintained to track all documents obtained (Table 2-2).

2.3.2 Data Management

Due to the scarcity of historical documentation regarding specific sources of PCBs at the ORR, Task 3 investigators had to rely heavily on the use of environmental measurements. For the purpose of tracking documents containing relevant data, the project team created a spreadsheet summarizing available analytical data (Table 2-3). This spreadsheet summarized the data available in each document according to type (i.e., individual data points or summary data), the media (i.e., fish, sediment, soil, water), and the locations sampled.

All relevant data identified were entered into separate databases according to media. For example, fish tissue data were summarized as to the waterbody sampled, river mile, year of sampling, species, and PCB concentration reported. The document and page numbers on which the data were reported were also recorded. Table 2-4 presents an example of the databases maintained for each medium. All data spreadsheets were subjected to review for accuracy.

Figure 2-2. Document Review Procedures Established for Task 3

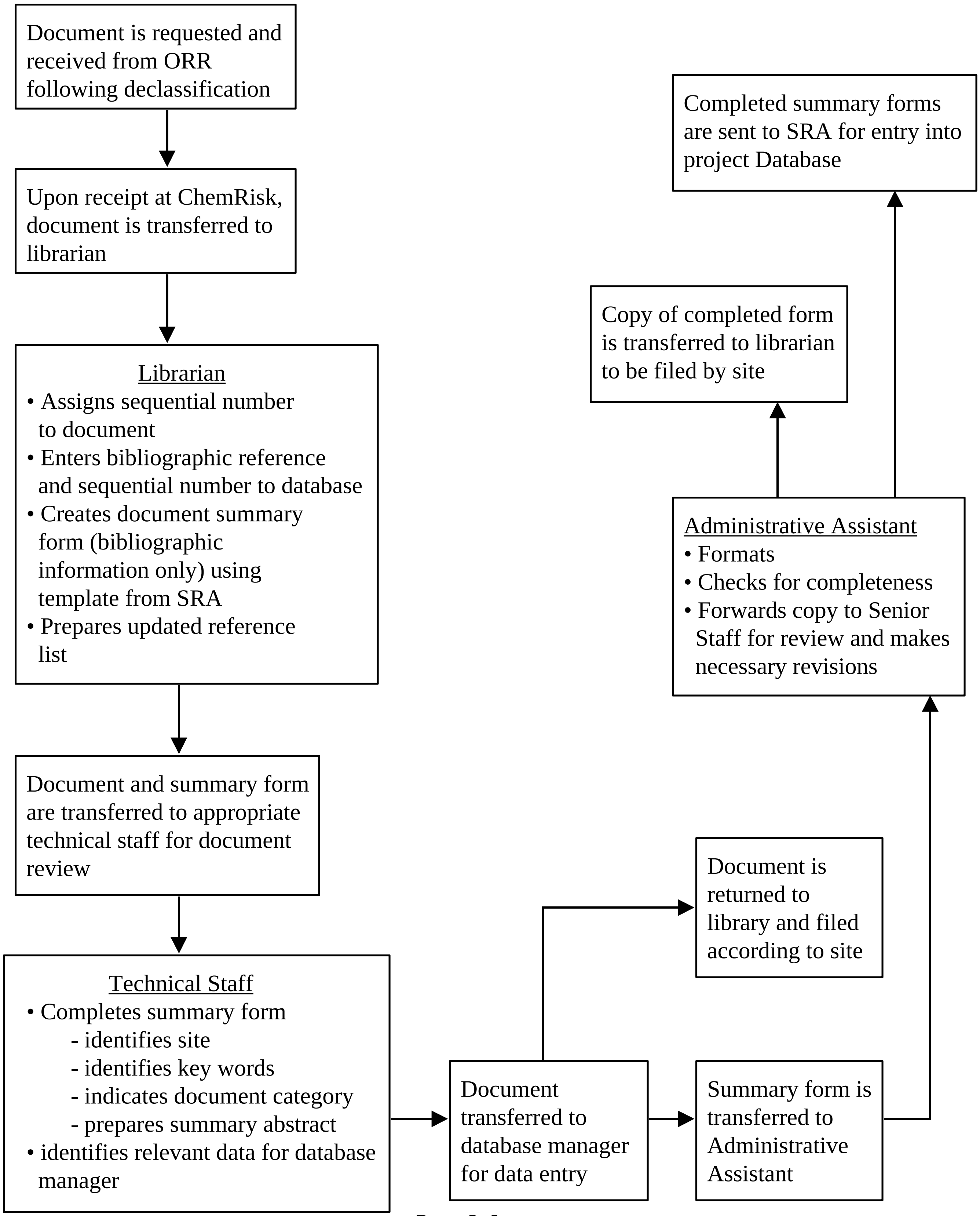


Table 2-2. Document Summary Reference List Example

Author	Year	Document	Location	Missing
Ashwood, T.L., et al.	1986	1	K-25	
Bailey, Z.C. and Lee	1991	2	Y-12	
Blaylock, B.G., et al.	1993	3	WOC	
Boyle, J.W. et al.	1982	4	X-10	
Dycus, D.L.	1989	5	OTHER	
Dycus, D.L.	1990	6	MELT	SUMMARY
Dycus, D.L.	1990	7	OTHER	
Hoffman, F.O., et al.	1991	8	ORR	
Kimbrough, C.W.	1986	9	Y-12	
Kornegay, F.C., et al.	1991	10	ORR	
Kornegay, F.C., et al.	1992	11	ORR	
Loar, J.M., et al.	1981	12	K-25	
MMES.	1985	13	ORR	
MMES.	1986	14	ORR	
MMES.	1990	15	ORR	
MMES.	1991	16	K-25	
Napier, J.M.	1989	17	Y-12	
Oakes, T.W., et al.	1987	18	ORR	
Olsen, C.R., et al.	1990	19	ORR	

Table 2-3. Example of Spreadsheet Summarizing Data Contained in Reviewed Documents

Document Number	Year Sampled	Type (a)	Media					Location																		
			Fish	Sediment	Soil	GW	SW	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S
1	1986	R		x											x											
2		N																								
3	1984,1990	S	x	x			x	x																		
3	1993	S		x			x	x																		
4	1982	N																								
5	1989	R	x							x								x								
6	1990	R	x											x				x		x	x					
6	1990	S	x							x				x				x		x	x					
7	1988	R	x								x															
8	1991	S	x	x															x							
8	1982	R	x									x														
9	1986	R		x		x	x									x	x									
10	1991	S	x	x	x	x	x												x							
11	1991	S	x	x	x	x	x	x	x	x									x							
12	1981	R	x							x		x														
12	1981	S	x							x		x														
13	1984	S	x				x	x		x			x													
14	1985	S	x			x	x			x							x		x							
15	1990	S	x					x		x		x		x		x										
16	1991	N																								
17		N																								
18	1987	S	x														x									
19		N																								

a. R=Raw, S=Summary, N=None
b. Location Code:

A. White Oak Creek and White Oak Lal
B. ORNL
C. Clinch River
D. Tennessee River
E. Poplar Creek
F. EFPC
G. Watts Bar Res.
H. K-25
I. Y-12
J. Bear Creek
K. Melton Hill Reservoir
L. General ORR
M. Fort Loudoun Reservoir
N. Tellico Reservoir
O. Little River
P. Powell River
Q. Little Tennessee River
R. Emory River
S. New Hope Pond

Table 2-4. Example of Data Summary Tables

Document Number	Year	Location		Fish Information				Aroclor Mixtures of PCBs (mg/kg)						
		Waterbody	River mile	Species	wt (lbs)	L (in)	sex	1242	1254	1221	1232	1248	1260	1016
31	1984	Clinch River	2	Channel Catfish	1.11	14.7	f	0.1u	0.2	0.1u	0.1u	0.1u	0.5	0.1u
31	1984	Clinch River	11	Channel Catfish	1.65	17	m	0.1u	0.7	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Clinch River	2	Channel Catfish	1.06	15.2	f	0.1u	0.7	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Clinch River	6	Channel Catfish	1.1	15.3	f	0.1u	0.8	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Clinch River	2	Channel Catfish	3.26	20.6	m	0.1u	0.4	0.1u	0.1u	0.1u	0.5	0.1u
31	1984	Clinch River	11	Channel Catfish	0.96	51.1	m	0.1u	0.9	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Clinch River	6	Channel Catfish	1.12	15.2	f	0.1u	1	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Clinch River	6	Channel Catfish	1.27	16.7	m	0.1u	0.5	0.1u	0.1u	0.1u	0.6	0.1u
31	1984	Clinch River	11	Channel Catfish	1.58	16.9	f	0.1u	1.1	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Clinch River	11	Channel Catfish	2.03	16.3	f	0.1u	0.8	0.1u	0.1u	0.1u	0.7	0.1u
31	1984	Clinch River	2	Channel Catfish	1.49	17.2	f	0.1u	0.6	0.1u	0.1u	0.1u	0.9	0.1u
31	1984	Clinch River	11	Channel Catfish	1.57	17.3	f	0.1u	1.1	0.1u	0.1u	0.1u	1	0.1u
31	1984	Clinch River	6	Channel Catfish	1.09	15.2	f	0.1u	0.6	0.1u	0.1u	0.1u	1	0.1u
31	1984	Clinch River	11	Channel Catfish	1.78	18.5	m	0.1u	0.5	0.1u	0.1u	0.1u	1.2	0.1u
31	1984	EFPC	13.8	Carp	3.38	19.7	m	0.1u	0.66	0.1u	0.1u	0.1u	1.3	0.1u
31	1984	EFPC	13.8	Carp	2.6	21.1	m	0.1u	1.7	0.1u	0.1u	0.1u	2	0.1u
31	1984	EFPC	4	Channel Catfish	1.68	17.3	f	0.1u	0.3	0.1u	0.1u	0.1u	0.3	0.1u
31	1984	EFPC	13.8	Largemouth Bass	0.59	10.7	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	13.8	Largemouth Bass	0.61	11.4	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.54	0.1u
31	1984	EFPC	13.8	Largemouth Bass	0.87	11.9	f	0.1u	0.26	0.1u	0.1u	0.1u	0.66	0.1u
31	1984	EFPC	4	North Hogsucker	0.24	11.9	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	8.8	Rock Bass	0.41	8.4	f	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	8.8	Rock Bass	0.26	7.1	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	8.8	Rock Bass	0.34	7.5	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	8.8	Rock Bass	0.34	7.5	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	8.8	Rock Bass	0.43	8.1	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	8.8	Rock Bass	0.26	6.9	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	8.8	Rock Bass	0.23	6.5	f	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	EFPC	4	Spotted Sucker	0.98	13.5	f	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Poplar Creek	0.2	Channel Catfish	1.49	16.5	m	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u	0.1u
31	1984	Poplar Creek	0.2	Channel Catfish	4.02	21.9	m	0.1u	0.4	0.1u	0.1u	0.1u	1.4	0.1u
31	1984	Poplar Creek	0.2	Channel Catfish	0.95	14.3	f	0.1u	0.4	0.1u	0.1u	0.1u	1.7	0.1u
31	1984	Poplar Creek	0.2	Channel Catfish	1.48	16.5	m	0.1u	0.6	0.1u	0.1u	0.1u	1.9	0.1u
31	1984	Poplar Creek	0.2	Channel Catfish	2.35	18.5	f	0.1u	0.6	0.1u	0.1u	0.1u	2.8	0.1u

u = not detected at a detection limit of 0.1 mg/kg

2.3.3 Interview Summaries

Similar to document review, the project team established procedures to ensure consistency in the methods used in conducting and documenting interviews (Figure 2-3). Section 2.2 discusses the procedures followed when the team conducted interviews either in person or by telephone. As described, these interviews were documented and the summaries submitted for entry into the project database. In addition, a copy of each interview summary form was filed according to subject. The project team created a spreadsheet to track information including the date and type of interview (telephone or on-site), the name and position of the individual interviewed, the interviewer(s), the issues discussed, and all contacts to whom the interviewer was referred during the interview. An example of this spreadsheet is presented in Table 2-5.

Figure 2-3. Interview Documentation Procedures Established for Task 3

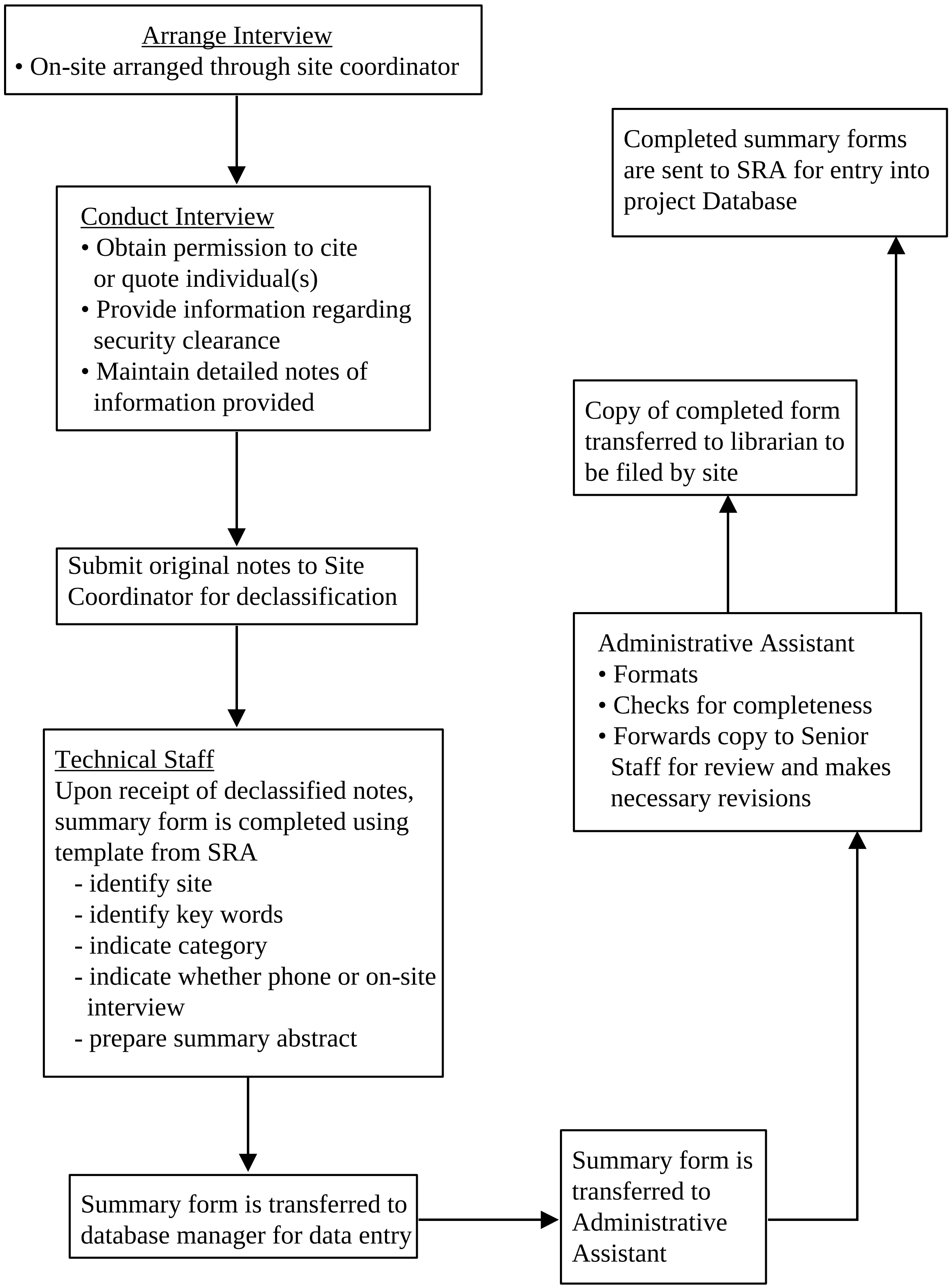


Table 2-5. Interview Summary Database - Example

Date	Site	Interviewee	Position	Interviewers	Phone or On-site	Time	Issue	Contacts Identified: Name	Affiliation	Area of Expertise
1/18/95	ORR	Gordon Blaylock	SENES	Bonnevie	P	3:30 PM	contacts at ORR	Les Hook		data
								Larry Voorhees		data
								Craig Brandt	X-10	database
								Don Dycus	TVA	historical data
								Mark Bevelhimer	X-10	PCB data
1/19/95	ORR	George Southworth		Curry	P	3:00 PM	data and sources	Sig Christiansen	X-10	data
								Tom Collins	Y-12	remed. at Bear Creek
								Bob Cook	X-10	Clinch River RI
								Don Dycus	TVA	data
1/19/95	ORR	Sara Welch	Central Comp.	Curry	P	3:00 PM	data and sources	Craig Rightmire		water data
1/19/95	ORR	Mark Bevelhimer	ORNL/data	Bonnevie	P	4:00 PM	data and sources	Jim Loar	ORNL	BMAP data
								George Southworth	ORNL	BMAP data
								Les Hook	data manager/ER	env. restoration
								Craig Brendt		database
								Tom Ashwood	ORNL	data
								Larry Wilson	TN Tech U	creel surveys
1/19/95	ORR	Mark Peterson	ORNL	Curry	P	2:00 PM	data and sources	Sig Christiasen	ORNL	data
1/20/95	Off site	Don Dycus	TVA	Bonnevie	P	10:30 AM	data	Greg Dentin	Dep. Wat. Poll. Cont.	data
								Dave McKinney	TN Wildlife Res.	data
								Dale Parkhurst	Shoals Alabama	data
								Dave Bruggink	TVA	STORET
								Carol Ann Davis	TVA	records clerk
								Joe Fehring	TVA?	sources
1/20/95	ORR	Sig Christiansen	ORNL/BMAP	Bonnevie	P	9:30 AM	data	Gordy Thompson		data/groups
								David Herr	OREIS	data
									Cent. Doc. Manag.	data
									Env. Restor.	data
								Roxanna Hinzeman	Doc. Center	monitoring studies
1/25/95	ORR	David Herr	OREIS	Bonnevie	P	11:15 AM	OREIS	Teresa James	OREIS	PCB data
1/25/95	Off-site	Carol Ann Davis	TVA	Bonnevie	P	10:15 AM	TVA records			
1/25/95	ORR	Bob Cook	ORNL	Bonnevie	P	4:00 PM	Clinch River RI		BMAP	reports
1/25/95	Off-site	Dave Bruggink	TVA	Bonnevie	P	10:45 AM	STORET data			
1/25/95	ORR	Craig Brandt	ORNL	Bonnevie	P	3:30 PM	database Clinch RI			
1/26/95	Y-12	Mick Wiest	ORNL	Curry	P	1:25 PM	sources/releases	Stacey Rathke	Y-12	PCBs at ORR
								Jim Loar	ORNL	BMAP
								George Southworth	ORNL	BMAP
1/26/95	ORR	Teresa James	OREIS	Bonnevie	P	10:30 AM	OREIS database			
1/26/95	ORR	Danielle Long	ER	Bonnevie	P	11:30 AM	ER Doc. Center			
1/27/95	ORR	Jeff Murphy	K-25	Curry	P	10:10 AM	PCB data	Cheryl Baker	PCB Coord. K-25	
								Wade Hollinger	K-25	PCBs K-25
								Stacey Rathke	Y-12	PCBs-Y-12
								Robert Johnson	PCB Coord. Y-12	PCBs Y-12

3.0 THE HISTORY OF PCBS ON THE OAK RIDGE RESERVATION

The results of the Dose Reconstruction Feasibility Study (ChemRisk, 1993) stated that the origin of PCB releases to the environmental media at the ORR and the resulting exposures to local populations were poorly characterized. As a result, the first step in the Task 3 evaluation was to characterize the historical uses and releases of PCBs and ways in which exposures to the populations living near the ORR could have occurred. This investigation was conducted to specify the primary sources of contaminants at a site, the mechanisms by which the contaminants were released, the media that received the contaminants, and the mechanisms by which the contaminants moved to off-site media to which humans may have been exposed.

The project team first searched historical records and interviewed current and past personnel to identify the different historical activities and processes involving PCBs at the ORR. Based on the sources identified, the team characterized the likely sources and releases of PCBs to environmental media and identified complete exposure pathways for potential human receptors. The project team collected and organized the available monitoring data for PCBs in relevant environmental media (i.e., soil, sediment, surface water and biota). In addition, the team evaluated the available information on other sources of PCBs in surface water bodies near the ORR. Finally, the team studied the potential for dioxin and furan formation during the burning of oily wastes containing PCBs.

The results of the historical investigation provided the working hypotheses upon which the remainder of Task 3 was based. The complete exposure pathways identified in this section are evaluated further in Section 6.0.

3.1 HISTORICAL USES AND RELEASES OF PCBS AT THE ORR FACILITIES

In comparison to the other chemicals investigated in the Dose Reconstruction Study, information regarding the use and disposal of PCBs at the ORR is very limited. This is attributed to the fact that prior to the early 1970s, PCBs were not considered a hazard to human health or the environment. During the first thirty years of operations at the ORR, little or no attention was paid to the use, disposal, or contamination of the environment with PCBs. Few attempts were made to control the release of PCBs to the environment during this period, and minimal effort was made to track or document the amounts of PCBs used or released to the environment. However, it is important to note that, unlike other materials such as solvents, PCBs were not routinely replaced. Due to their relatively high cost, as well as the fact that they were typically used for specialized purposes, PCBs were considered less disposable than most oils. As a result, generation of PCB waste was usually associated with equipment disposal or maintenance. In the 1970s and 1980s, due to the increased knowledge regarding the toxicity of PCBs and the resulting environmental regulations, the operators of the ORR facilities initiated efforts to identify and control PCB releases to the environment. As a result, information has been collected since the mid-1970s on the presence of PCBs in electrical and other equipment and in environmental media and biota. Based on this information, it is possible to obtain a reasonable picture of the long-term uses of PCBs at the ORR in situations where PCB-containing fluids are still in use. In addition, because PCBs are persistent in the environment and bind readily to soils and sediments, recent measurements of PCBs in soils, sediments, and biota also provide insight into historical releases.

In general, it appears that the primary uses of PCBs at the ORR were electrical equipment (i.e., transformers and capacitors), hydraulic fluids, heat-transfer fluids, and cutting oils. In addition, PCBs were present in relatively low levels (less than a few percent) in many products including paints, coatings, adhesives, inks, and gaskets. PCB uses, as well as the potential impact of releases on off-site receptors, differed among the three facilities at the ORR. Therefore, separate analyses are presented here for each of the facilities.

3.1.1 Y-12

Y-12 is located at the eastern end of the Bear Creek Valley, about 1/2 mile from the center of the City of Oak Ridge (Figures 3-1 and 3-2). The plant is bordered on the southern side by Chestnut Ridge and on the north side by Bear Creek Road and Pine Ridge (Turner et al., 1988). Discharges from Y-12 potentially impacted the watersheds of two streams, Bear Creek and East Fork Poplar Creek (EFPC). Bear Creek originates at the west end of the plant and flows for 8 miles in a southwesterly direction to its confluence with EFPC (Turner et al., 1988). The headwaters of EFPC are contained in a series of underground collection pipes that extend along the western and southern ends of the facility. The above ground portion of the stream begins along the central portion of the southern boundary of the plant, flows in a northwest direction through a gap in Pine Ridge, and continues through commercially zoned areas in the town of Oak Ridge before meandering west towards its confluence with Poplar Creek (SAIC, 1994).

During World War II, Y-12's primary mission was the separation and enrichment of uranium-235 from natural uranium by an electromagnetic process using separation units called "calutrons" (OREP, 1992). Large magnets, called "D-magnets", provided magnetic fields for this process and were positioned between back-to-back calutrons to form units known as "separators" (ChemRisk, 1993; Banic, 1995a). The separators were grouped in banks called "tracks" and arranged in pairs, either in an oval or a rectangular shape. The separation and enrichment of uranium-235 required two distinct types of dual-ion source separators, an "alpha" separator for the first stage enrichment and a "beta" separator for final stage enrichment (ChemRisk, 1993; Banic, 1995a). There were a total of 290 alpha separators (288 operating units plus 2 experimental units) housed in five buildings (designated as 9201-1 to 9201-5) and a total of 866 beta separators (864 operating units plus 2 experimental units) divided among four buildings (designated 9204-1 to 9204-4).

Once K-25's gaseous diffusion plant became operational in 1945, the need for electromagnetic separation diminished, and most of the tracks at Y-12 were placed in a standby mode for future needs (OREP, 1992). Today, only one of the original tracks is in operation, enriching isotopes for use in the medical industry (Banic, 1995a). After the war efforts, the Y-12 Plant was transformed in the late 1940s to a high-tech plant for processing of nuclear materials and production of weapons components. In the early 1950s, the United States launched a crash program to produce enriched lithium deuteride (enriched in its lithium-6 component) for thermonuclear weapons. One form of hydrogen fuel used in the bombs was lithium deuteride (UCCND, 1983). Y-12's assignment was to separate high-purity ^{6}Li from natural lithium to

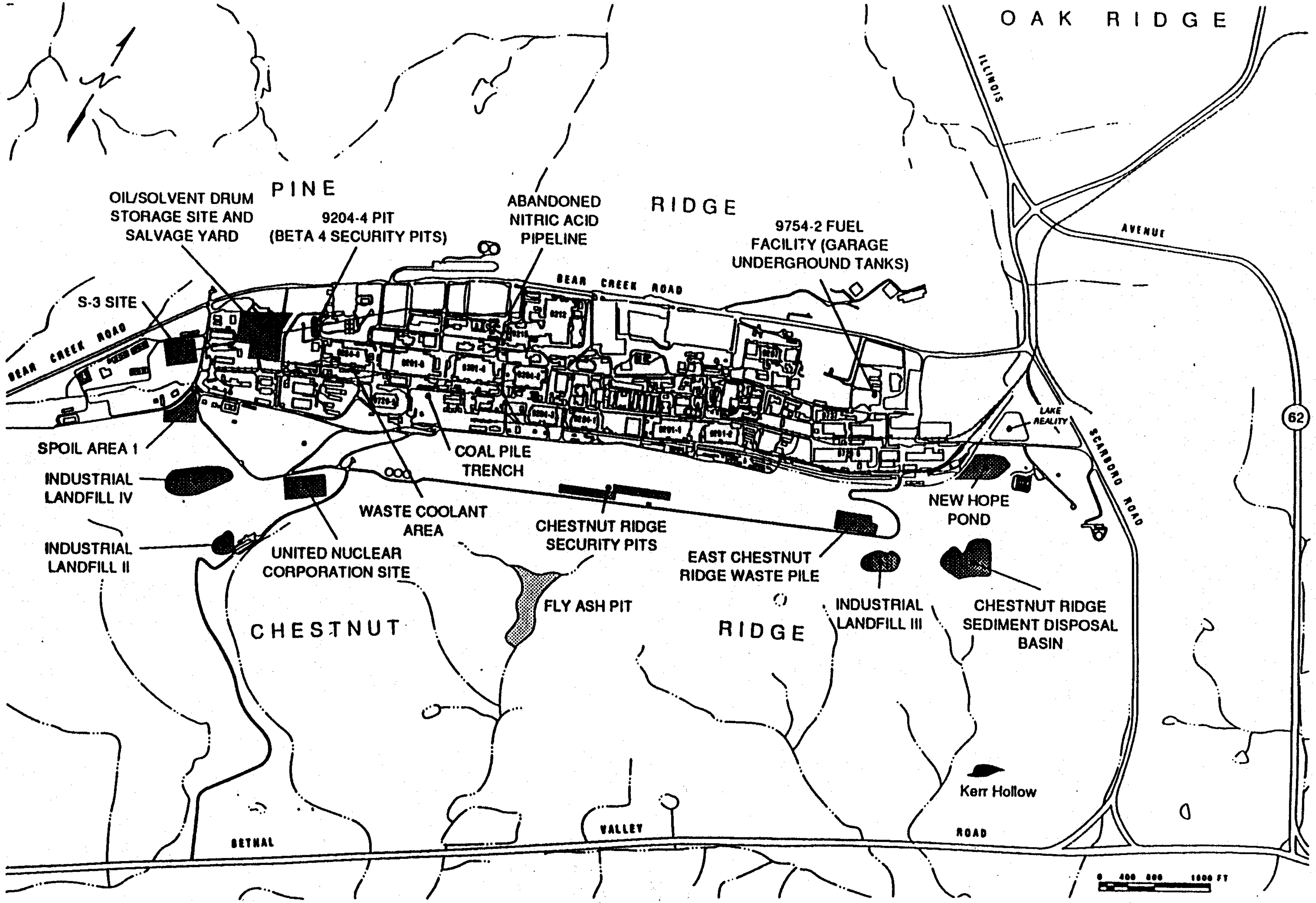


Figure 3-1. Map of the Y-12 Plant Site-Main Production/Support Area

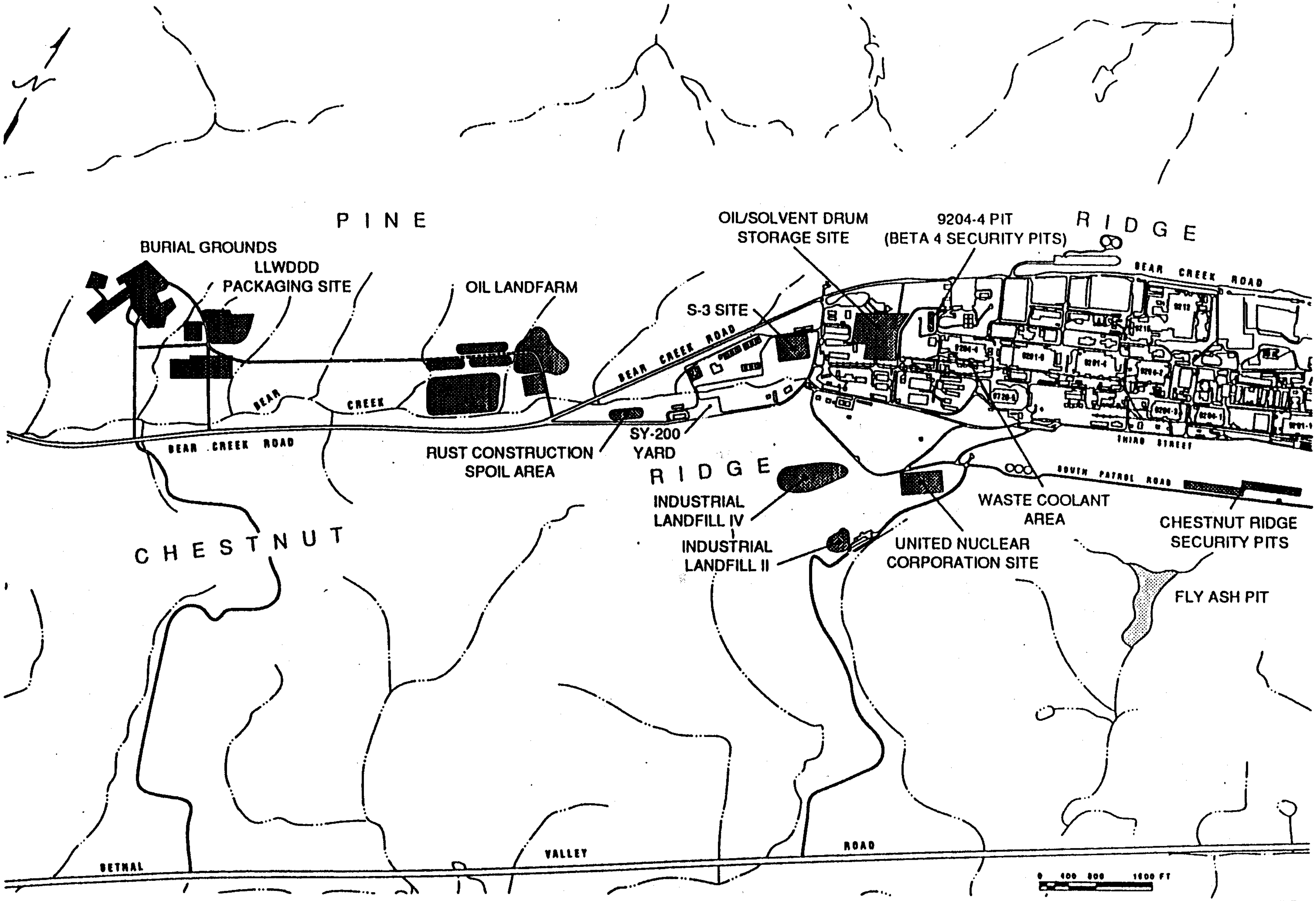


Figure 3-2. Map of the Y-12 Plant Site-Western End of the Plant

produce enriched ^{6}Li deuteride for use in more powerful and efficient weapons (UCCND, 1983). Lithium separation at Y-12 involved pilot plant facilities from 1950 to 1955, and production facilities from 1953 to 1963. Two large-scale production facilities were completed between December 1953 and September 1955.

3.1.1.1 Historical Uses and Releases Involving PCBs at Y-12

PCB contamination at Y-12 can be traced to several general sources, including electrical systems (i.e., transformers and capacitors), the use of PCB-containing cutting oils for the machining of enriched uranium, and a cooling system known as the Z-oil system, which was associated with the electromagnetic separation process. In addition, PCBs were introduced to some of the hydraulic systems throughout the plant. Each of these sources is discussed in more detail below.

<u>Electrical Equipment</u>

Although transformers and capacitors containing PCBs were used at the Y-12 facility, only limited information regarding the numbers and exact time period involved is available from existing records; details regarding concentrations of PCBs in individual pieces of equipment were not recorded. In addition, it is important to note that although Askarel PCB transformers and capacitors were purchased for specific purposes, other equipment, such as mineral oil transformers may have contained low levels of PCBs due to cross-contamination from manufacturing as well as maintenance activities. As a result of environmental regulations on the use, storage and disposal of PCB-contaminated equipment, efforts were initiated in the 1980s to identify and remove all electrical equipment containing PCBs at Y-12. Available information indicates that much of the equipment currently in place is original, therefore concentrations measured recently in the equipment are likely indicative of historical PCB levels.

Available information indicates that approximately 90 large Askarel transformers, each containing, on average, 600 gallons (2,300 liters), were utilized by the electrical maintenance system at locations throughout Y-12, servicing the primary power needs of the plant (Health, Safety, Environment, and Accountability Division, 1992; Blalock, 1995). As of 1973, it was estimated that there were approximately 53,511 gallons of PCBs used in these transformers which were located throughout the plant (Jordan et al., 1977). Approximately half of these transformers were located outdoors, with the remainder situated inside buildings throughout the plant (Health, Safety, Environment, and Accountability Division, 1992). As a result of increasing PCB regulations in the 1980s, the outdoor transformers were replaced with non-PCB transformers and the indoor transformers were retrofilled (i.e., drained, flushed, and refilled) with a non-PCB insulating fluid (Health, Safety, Environment, and Accountability Division, 1992). The discarded PCB transformers and PCB-contaminated oil were disposed of in accordance with PCB regulations.

There were also numerous mineral oil transformers throughout the Y-12 Plant, ranging from large (i.e., 100 to 800 gallon or 378 to 3,000 liter) pad-mounted transformers to smaller (i.e., 15 to 100 gallon or 57 to 378 liter) pole- or pad-mounted transformers (Health, Safety, Environment, and Accountability Division, 1992). A survey in 1991 indicated that the majority of the transformers on-site at that time were small pole-mounted transformers (Author Unknown, 1991). Concentrations of PCBs in the mineral oil transformers have typically been reported to be below 500 ppm (Author Unknown 1991; Health, Safety, Environment, and Accountability Division, 1992), although transformers stored in the basement of Building 9204-3 were found to contain concentrations ranging from 2 to 880 ppm (Ashwood, 1986). However, anecdotal information indicates that none of the mineral oil transformers had PCB concentrations greater than 1,000 ppm (Blalock, 1995).

In addition to the transformers used for general power needs, there were also transformers associated with specific systems. The electromagnetic process for the enrichment of uranium-235 required large amounts of power; each calutron was supplied by a minimum of eight transformers with fluid capacity varying from about 10 to 200 gallons (38 to 760 liters). The capacity of the combined transformers for the beta units equaled approximately 400 gallons, while those of the alpha units totaled approximately 600 gallons (2,271 liters; Banic, 1995a, b, c).

Due to the size and number of the transformers associated with the electromagnetic process, this system presented a significant potential for the releases of PCBs due to leakage and during disposal of PCB oils. In 1987, transformers associated with the system were sampled and found to have concentrations ranging from 24 to 487 ppm (MMES, Unk; Banic 1995b,c). In addition, transformers stored in the basement of Beta 3 are believed to have been associated with the electromagnetic separation process (White, 1995). In 1992, PCB concentrations in these transformers were determined to be under 1000 ppm (MMES, Unk).

In addition to transformers, numerous capacitors were used at many different locations at Y-12 for accumulating and holding electrical charges (Health, Safety, Environment, and Accountability Division, 1992). Capacitors contained varying amounts of oils, but typically less than 1.3 gallons (5 liters) of oil. Many of the outdoor capacitors at Y-12 were Pyranol capacitors; these used fluids containing 50 to 60 percent Aroclor 1242. Currently, all outdoor capacitors at Y-12 have been removed (Health, Safety, Environment, and Accountability Division, 1992). A large number of capacitors, many of them Pyranol, have been inventoried in the Beta 3 building (9204-3) (MMES, Unk). Specifically, 60 to 80 medium and large capacitors, most Pyranol, were stored for re-use in the basement in an area with a 6-8 inch dike (MMES, Unk). Unlike transformers, capacitors were factory sealed with no drainage ports. Therefore, capacitors are not believed to have been a source of PCB releases during their use. PCBs could, however, have been released following the disposal of capacitors.

To date no information regarding major accidents or releases of PCB-contaminated oils from electrical equipment at Y-12 has been discovered. It appears that the primary release of oils from transformers resulted from incidental leaks and drips that occurred while the transformers were on line. In the event of minor leaks or spills, sorbant materials were wrapped around the base of the transformer to absorb the oils, and were ultimately disposed of as oil wastes at the Burial Grounds (Health, Safety, Environment, and Accountability Division 1992; Blalock, 1995). In addition, many old or failing transformers were kept for spare parts at various lineyards and scrap yards throughout the plant (Ashwood, 1986; ORNL, 1987; MMES, Unk). While minor releases of PCB-contaminated oils from transformers in these areas have occurred, the concentrations of PCBs in these transformers are reportedly under 20 ppm, and soil concentrations in the vicinity have been found to be less than 1 ppm (ORNL, 1987; MMES, Unk).

In addition to the transformers stored in the scrap yards and lineyards, it has been reported that some transformer carcasses were sold as scrap at the Elza Gate scrap yard (Banic, 1995a; Blalock, 1995). It is possible that some of the oils from these transformers also may have been sold to the general public as used oils; however, there are no records to indicate amounts that may have been disposed in this manner (Banic, 1995a; Blalock, 1995).

Due to the lack of regulation regarding the disposal of PCB wastes prior to 1970, there are no detailed records regarding the exact disposal methods for used transformer oils; the first record of PCB-contaminated transformer oil disposed of at the Burial Grounds was from the early 1970s (Bailey, 1986). In general, used transformer oils were sent to a reclamation center at Y-12 to be cleaned and reused (Blalock, 1995). There is also anecdotal information to indicate that waste oils from transformers may have been introduced into the "dirty line" of the Z-oil system or disposed of at the Burial Grounds (Blalock, 1995; Hummel, 1995).

In summary, although it is possible that PCB-contaminated oils from transformers were disposed of at the Burial Grounds, it appears that the majority of the oils were recycled whenever possible. This indicates that transformers were probably not a significant source of PCBs.

<u>Cutting Oils</u>

As previously discussed, Y-12's primary mission following World War II became the fabrication and assembly of various types of nuclear weapon parts. This involved processes ranging from basic chemical conversions of raw materials to machining of weapons parts. Cutting or cooling fluids were a necessary component of the machining processes. These fluids were sprayed directly on the cutting tool to provide lubrication and remove heat. In order to minimize risk for a catastrophic event when working with enriched uranium, cutting oils containing high levels of chlorine to absorb neutrons are particularly valuable. Available records indicate that a perchloroethylene-based cutting oil, Shell Vitrea or Shell Vitriol, was used initially in the machining of uranium at Y-12 (Author Unknown, 1980; Spence, 1991). In the early 1960s, however, researchers at Y-12 began to investigate the possibility of using PCB-based cutting fluids for the machining of enriched uranium. Specific characteristics of PCBs, including low chemical reactivity, fire resistance, and high chlorine content made them very attractive for use as a cutting oil (Spence, 1991; Napier, 1995). In 1964, the machining process for enriched uranium in Building 9215's "M-Wing" began using a PCB-based mixture (60% Aroclor 1248 / 40% perchloroethylene) as a cutting fluid (Author Unknown, 1980; Everett, 1980; Bailey, 1986; Spence, 1991). In 1968, the Aroclor 1248 was replaced with a similar PCB mixture called Therminol (Author Unknown, 1980).

In the early 1970s, manufacturers of PCB-containing equipment and material issued letters to their clients alerting them to the suspected carcinogenic status of PCBs (Hibbs, 1971; Bailey, 1986). Based on this information, the cutting fluid system was drained and replaced with a non-PCB perchloroethylene mixture cutting fluid (Hibbs, 1971; Author Unknown, 1980; Bailey, 1986; Spence, 1991). Available information provided conflicting information regarding the exact dates of the changeout of the system. A letter dated November 19, 1971 indicated the intention of Union Carbide to discontinue use of Monsanto's Therminol (PCB-based) product. The first reported disposal of drums containing M-Wing coolant at the Burial Grounds occurred in July of 1970 (Bailey, 1986). Approximately 40 drums, comprising a total of 1,200 to 2,200 gallons (4,500 to 8,300 liters) of oil, were disposed of at the Y-12 Burial Grounds from July 1970 to July 1971 (Bailey, 1986). The concentration of PCBs in the system following the changeout is unclear; while it has been reported that measured concentrations were 5,000 ppm (Spence, 1991), no documentation of the analyses exists (Everett, 1980).

In late 1979, new PCB regulations were enacted specifying that open systems could not contain more than 500 ppm PCB. Sampling of the M-Wing cutting oil initiated as the result of this new regulation revealed that in 1979 the system still contained approximately 3,000 ppm PCBs (Author Unknown, 1980; Everett, 1980; Spence, 1991). In addition, samples collected in an exhaust filter house showed PCB concentrations of up to 8,906 ppm (Everett, 1980; Spence, 1991). As a result, the system was shut down on April 3, 1980. Throughout the following week, the system was repeatedly drained and flushed, until PCB concentrations were determined to be less than 50 ppm (Author Unknown, 1980; Everett, 1980; Spence, 1991); however, due to residual concentrations, as well as PCB impregnation of gaskets and filters in the system, concentrations continued to gradually increase (Author Unknown, 1980; Napier, 1995). Between 1980 and 1983, the M-Wing system was drained 17 times in attempts to maintain PCB concentrations below 50 ppm (Spence, 1991). In 1985 it was decided to switch to a cutting fluid comprised of an equal mixture of water and propylene glycol due to the volatility of perchloroethylene (Spence, 1991). Because PCBs were not soluble in the water / propylene glycol mixture, the system required no additional draining to maintain PCB concentrations below 50 ppm (Napier, 1995).

Available information indicates that the use of these PCB-based coolants at Y-12 was limited to the M-Wing of Building 9215 (Author Unknown, 1980; Napier, 1995). The system required approximately 12,000 gallons (45,420 liters) of coolant to operate (Author Unknown, 1980), therefore providing a relatively large volume of oil with high concentrations of PCBs. Incidental releases of PCBs probably did occur when employees, who had frequent contact with oil-coated parts, washed their hands in workplace sinks that discharged to EFPC. However, due to the value of enriched uranium, extreme efforts were made to minimize the potential for loss during the machining operations; for example, rooms in which machining processes occurred did not have floor drains (Napier, 1995). Because of these precautions, releases of oils containing PCBs directly to EFPC from enriched uranium operations were likely minimal.

Based on the available information, it does not appear that cutting oils from the M-Wing were disposed of at the Burial Grounds during regular operations. As previously discussed, cutting oils were filtered and recycled, reducing the amount of waste oil generated (Napier, 1995). The first recorded disposal of coolant from the M-Wing occurred in 1970 (Bailey, 1986). It is possible that small quantities of waste oils, such as those collected in catch pans associated with the system, may have been disposed of at the Burial Grounds. It does not appear, however, that waste oils generated from the 1980 changeout of the M-Wing machining system were disposed there. Bailey (1986) and Spence (1991) reported that between April 1980 and December 1985, approximately 23,743 gallons (89,870 liters) of M-Wing coolant were removed and placed into storage tanks on Third Street; these drums were later disposed off-site as prescribed by PCB regulations.

In summary, it appears that despite the large volume of PCB oils used in the system, the contribution of PCBs from the M-Wing cooling system was primarily limited to the single disposal event that occurred during the changeout in the early 1970s. This event resulted in the transport of 1,200 to 2,100 gallons (4,500 to 8,000 liters) of waste oils containing large amounts of PCBs to the Burial Grounds.

The Z-oil System

Because of the large amount of heat generated in the electromagnets used in the calutrons, a central, oil-based coolant system, referred to as the "Z-oil system", was designed for all of the buildings that housed the electromagnetic separation process (Wing, 1980a; Health, Safety, Environment, and Accountability Division, 1992; Banic, 1995a; Hummel, 1995). This system was comprised of an intricate, closed network of connected pipelines, originating at a 110,000 gallon (416,000 liter) storage tank located in building number 9620-2 at the east end of the facility (Campbell and Cloyd, 1995; Napier, 1995). From there, the oil was pumped across EFPC through a three inch "clean line" that ran west along the southern edge of the facility to holding tanks at each of the nine buildings associated with the process (Hummel, 1995). Each building had its own pump system and associated cooling towers that circulated oil from the tanks through the D-magnets (Banic, 1995a; Hummel, 1995; MMES, Unk). In addition, a six inch "dirty line" connected to each of the buildings, carried used oil back to building 9620-2 where it was filtered and returned to the central storage tank (Hummel, 1995). Anecdotal information indicates that in addition to used Z-oil, the "dirty line" also may have been used to dispose a variety of waste oils, including used transformer oils (Hummel, 1995).

As previously discussed, the electromagnetic separation process was significantly reduced at Y-12 following the war, with tracks in all but one of the buildings (Building 9204-3) dismantled (Banic, 1995a). The extensions of the pipeline that had serviced the inactive tracks were closed, although anecdotal information indicates that these buildings may have continued to draw Z-oil from the pipeline on occasion for other purposes, such as for use in mineral oil transformers (Bailey, 1995a; Hummel, 1995). In addition, it is likely that these buildings continued to use the "dirty line" to dispose of waste oils. Throughout the 1980s, efforts were made to eliminate abandoned portions of the Z-oil system (Ashwood, 1986; ORNL, 1987). For example, Ashwood (1986) reported that in 1986, 12,000 gallons (45,000 liters) of oil remaining in an abandoned Z-oil piping system of Building 9201-2 was in the process of being drained by the Maintenance & Utilities Division of Y-12. In 1991, an action plan was developed and implemented to gradually drain and remove the entire system, with the exception of a modified section associated with Building 9204-3 (Health, Safety, Environment, and Accountability Division, 1992).

The Z-oil system used approximately 142,000 gallons (537,500 liters) of a light oil described as being similar to the mineral oil used in transformers (Wing, 1980a; Bailey, 1986; Hummel, 1995). The oil used likely did not contain PCBs when it was purchased (Bailey and Thornton, 1995; Banic, 1995a; Hummel, 1995; Napier, 1995); however, PCB concentrations of about 100 ppm were discovered in 1979 during a routine sampling of oil systems at the plant (Wing, 1980a;

Health, Safety, Environment, and Accountability Division, 1992). The most probable source of PCBs in the Z-oil system was related to the use of the "dirty line" for disposal of oils, as previously discussed (Bailey and Thornton, 1995; Hummel, 1995). In addition, it is possible that a shipment of contaminated oil may have been added to the system; it was reported that in the 1970s, Y-12 received notification from Shell Oil that a 5,000 gallon (19,000 liter) shipment of mineral oil, purchased for use in the Z-oil system, may have contained PCBs (Napier, 1995).

Efforts to reduce PCB contamination in the Z-oil system were initiated in the early 1980s as a result of the increasing regulatory controls on PCBs which required that open systems containing PCBs be drained and flushed prior to May 1, 1980 (Wing, 1980a; Health, Safety, Environment, and Accountability Division, 1992). Due to the costs associated with draining and flushing the system, as well as the potential difficulties associated with disposal of such a large quantity of PCB-contaminated oils, Y-12 requested a waiver from the regulation, suggesting that a new technique for reducing PCB concentrations be implemented (Wing, 1980a,b; Keller, 1980a,b). As a result, a contract was granted to Sun Ohio to clean the Z-oil using a patented mobile filtration system capable of reducing PCB concentrations to less than 5 ppm (Wing, 1980a; Health, Safety, Environment, and Accountability, 1992; Hummel, 1995; Napier, 1995). A portion of the treated oil was sold, while the remainder was placed back into the system along with new, clean oil (Health, Safety, Environment, and Accountability Division, 1992).

Due to the size and extent of the pipeline, its proximity to EFPC, and a history of leakage in parts of the system, it is likely that oils were occasionally spilled or leaked into the waterway from the Z-oil system. For example, during a 1990 inspection of Building 9204-3, it was reported that hundreds of feet of piping containing Z-oil were located in the building (MMES, Unk). Although no leaks were observed, there were catch pans with oil in them under much of the piping in the basement, indicating that oils were expected to leak at specific locations. Consequently, minor releases of Z-oil to the creek may have occurred via floor drains within nearby buildings serviced by the system (MMES, Unk).

In addition, although releases of oils from the D-magnets were likely to have been negligible while the magnets were operational, many of the large magnets were stored on-site after the tracks were dismantled. For example, thirty D-magnets containing oils with PCB concentrations ranging from 3 to 110 ppm were stored in the lineyard east of Building 9204-1 (MMES, Unk). While three of these magnets were observed to leak PCB oil onto the gravel (MMES, Unk), releases of oil were not reported to be significant.

Despite the possibility of minor releases of oil over time, there are no records of significant spills of oil from the Z-oil system, probably as the result of the fundamental design of the system (Health, Safety, Environment, and Accountability, 1992). It has been reported that occasional ruptures or leaks in the pipeline did occur; in the event of a rupture, however, the affected section of pipe could be isolated and drained until the pipeline was repaired, thus limiting the amount of oil released (Health, Safety, Environment, and Accountability, 1992; Hummel, 1995). Although records have not been identified that report the volumes of oil released during these occurrences, anecdotal information indicates that in the event of a spill, efforts were made to reclaim as much of the oil as possible for reuse in the system (Hummel, 1995). In addition, while in use the system was carefully monitored and maintained in an effort to prevent any substantial ruptures or leaks.

Information on historical disposal of oil from the Z-oil system is limited. Available information indicates that, due to the cost of the oil and the design of the system, it was recycled as much as possible; therefore, disposal was probably relatively minimal (Bailey and Thornton, 1995; Hummel, 1995; Napier, 1995). Following the enactment of PCB regulations, the Z-oil was disposed of as PCB waste (ORNL, 1987; Bailey, 1995a,b; Napier, 1995). While it is possible that some quantity of the oil associated with the system could have been disposed of at the Burial Grounds, the relatively low concentrations of PCBs measured in the system indicate that any contribution of PCBs from this source would have been minimal.

Hydraulic Systems

The Y-12 plant has thousands of hydraulic systems utilized for various purposes. Although there are no records to indicate that PCBs were intentionally introduced to the oils contained within these systems, PCBs have been discovered at measurable levels. In 1981, as part of the initial PCB inventory, hydraulic and lubrication systems with a volume greater than 100 gallons (378 liters) were sampled and analyzed for PCBs (Case, 1981). Of the 101 systems tested, only 10 had concentrations greater than 50 ppm; more than three-fourths were non-detect. Concentrations greater than 50 ppm have, however, been identified, primarily in hydraulic systems located in Buildings 9212, 9215, and 9998 (Bailey, 1986). For example, concentrations of up to 16,600 ppm were found in portions of a milling operation in Building 9215 (Case, 1981). Elevated PCB concentrations (e.g., 1000 to 5000 ppm) have also been associated with three foundry systems in Building 9998 (Case, 1981). In addition, an abandoned hydraulic system containing PCB concentrations ranging from 5,000 to 1,000,000 ppm was found in Building 9206 (Rathke, 1987).

As a result of TSCA regulations regarding PCB concentrations in open systems, hydraulic systems found to contain concentrations of PCBs greater than 50 ppm were flushed and refilled (Rathke, 1995). As of 1992, the concentrations in the milling and foundry systems described above were reported to be below 50 ppm (Health, Safety, Environment, and Accountability, 1992). In 1991 a plan was developed for the sampling of several hydraulic systems at the Y-12 plant to verify that concentrations had remained within acceptable limits (Rathke, 1991). Information was not available regarding the outcome of that sampling.

There is little information regarding the potential for spills or releases of PCB-contaminated oils from hydraulic systems. Although it is noted that the milling and foundry operations that have been identified were not diked (Health, Safety, Environment, and Accountability 1992), no record of spills or leaks has been found. Because there were floor drains located near the foundry operations, PCB-contaminated oils may have entered EFPC (Health, Safety, Environment, and Accountability, 1992). Similarly, a 1989 release of PCB-contaminated water from a drainage trough around the Building 9202 press was reported (Spence, 1991). Levels of PCBs in the extrusion fluid associated with the press were low (6 ppm); however, smear samples indicated concentrations of 1400 g/cm^2 in the trough. Based on this information, while it was assumed that historical concentrations of PCBs in the fluid were higher, records do not exist to verify this assumption (Spence, 1991).

In addition to direct releases to EFPC, it is likely that oils used in the various hydraulic systems may have comprised a portion of the waste oils disposed at the Burial Grounds. Unlike the Z-oil or cutting fluids previously discussed, hydraulic oils were frequently changed and would likely have been treated in the same manner as used motor oils. As a result, these oils may have represented an ongoing source of PCBs in the wastes disposed of in the Burial Grounds.

3.1.1.2 Waste Disposal of PCBs at Y-12

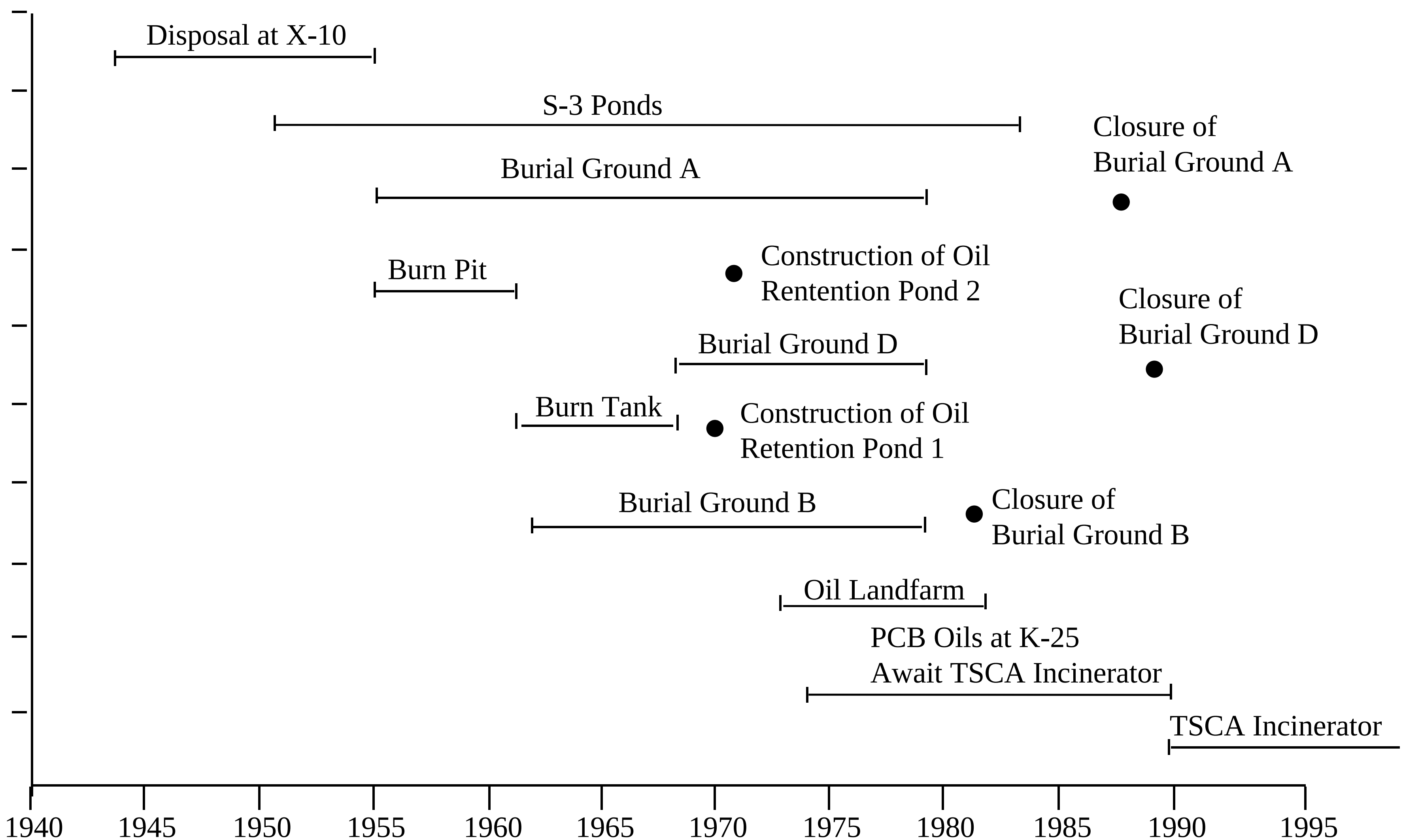
Industrial activities at Y-12 generated large quantities of liquid and solid waste, including thousands of gallons of waste oils. A large proportion of the waste oils from Y-12 did not contain PCBs, but rather were comprised of mineral oils, water soluble coolants, anti-freeze, motor oils, and products such as "Gulf petroleum," which was a mixture of 95% water and 5% emulsifiable oils (McCauley, 1984b). Based on the available information, it appears that the majority of the PCB-contaminated waste oils generated at Y-12 were those oils used in the machining of enriched uranium (M-Wing coolant), hydraulic systems, and electrical transformers.

Limited information is available regarding waste disposal activities during the early years of operation at Y-12. It has been suggested that liquid wastes generated at Y-12 prior to 1950 were disposed at burial facilities at X-10 (Geraghty & Miller, 1985; Bailey and Thornton, 1995); however, no specific documentation of these activities has been found. In the early 1950s, Y-12 began operation of a number of waste management units at the Bear Creek Disposal Area. Although records indicate that some portion of the waste oils generated at Y-12 may have been sold to the public at K-25, it appears that the majority were disposed at the Bear Creek Disposal Area. Throughout the course of its operation, the Bear Creek Disposal Area contained three principle disposal sites, including the S-3 Ponds, the Burial Grounds, and the Oil Landfarm. A timeline of operation for each of these sites is presented in Figure 3-3.

<u>S-3 Ponds: 1951-1984</u>

Constructed in 1951, the S-3 Ponds, located at the headwaters of Bear Creek, were the first operational site at the Bear Creek Disposal Area (Turner et al., 1988; OREP, 1992). Made up of four adjoining, unlined seepage pits, each with an approximate capacity of 2.5 million gallons (9.5 million liters) (Bailey, 1986), the S-3 Ponds were used primarily for disposal of mop waters, machine coolants, byproducts from uranium mining operations, caustic wastes, and acidic wastes with a pH of less than 2.0. The Ponds remained operational until 1984, at which time the annual quantity of liquid waste disposed there had reached approximately 2.7 million gallons (10 million liters) (Turner et al., 1988). After 1984, the ponds were allowed to settle, and the resulting supernatant was pumped to the S-3 Ponds Treatment Facility for removal of trace metals and organics. Closure and filling of the S-3 Ponds began in 1988.

Figure 3-3. Timeline of Oil Waste Disposal Activities at Bear Creek



The Burial Grounds: 1955-1979

Located adjacent to Bear Creek and approximately two miles west of the Y-12 plant, the Burial Grounds consisted of several waste disposal units designated Burial Grounds A, B, C, D, E, and J. Each unit consisted of a series of trenches used for the disposal of liquid and solid wastes. The first trench in Burial Ground A was excavated in 1955 for the disposal of solid waste, including uranium-contaminated solids. Burial Grounds B and C were excavated in 1962, and were used primarily for disposal of beryllium, beryllium oxide, thorium, and solid waste contaminated with these materials. Burial Ground D was excavated in 1968 in order to accommodate depleted uranium metals and oxides after Burial Ground B had reached capacity (Geraghty & Miller, 1985). Burial Grounds B and D were also likely recipients of low quantities of waste oils that covered depleted uranium metal and oxides. These oils were placed in unlined trenches and covered with soil. Burial Ground B was closed in 1982, and Burial Ground D was closed in June of 1991 (OREP, 1992). Burial Grounds E and J were used primarily for the disposal of uranium metal and oxides.

Of the individual units, Burial Ground A was the recipient of the majority of waste oils and liquids. For example, beginning in 1959, mop waters generated from floor cleaning operations at Y-12 were drained into standpipes, rock pits, or trenches installed vertically in Burial Ground A. There is documentation that 600,000 gallons (2.3 million liters) of mop waters were disposed annually in this fashion from 1971 to 1978 (Geraghty & Miller, 1985); the amounts disposed prior to 1971 were not recorded. In addition to the mop waters, approximately 100,000 gallons (380,000 liters) of waste solvents were poured onto gravel piles and waste trenches in the southern part of Burial Ground A between 1970 and 1981 (Turner et al., 1988). Between 1972 and 1979, approximately 460,000 gallons (1.7 million liters) of waste oils and coolants were poured onto a rock pit in the northwest portion of Burial Ground A (Geraghty & Miller, 1988). Disposal of waste oils in Burial Ground A ceased in 1979 (McCauley, 1984a), and the area was remediated and capped in 1989 (OREP, 1992).

Beginning in late 1971, approximately 132,000 gallons (499,600 liters) of oil and mop waters were pumped into a standpipe located at the northeast corner of Burial Ground A (McCauley, 1984). This practice was abandoned when oil was observed leaking from the east end of the northern-most trench. To retain the oil leaking from the Burial Ground, two small ponds were built, as described in the section on the Oil Retention Ponds. Anecdotal information indicates that oils collected in the retention ponds were periodically removed and returned to the trenches in Burial Ground A.

Burning of Waste Oils

In addition to the disposal of oils in the trenches, waste oils and solvents were burned at Burial Ground A during two separate periods. From 1955 to 1961, a portion of the waste oils disposed at Burial Ground A were poured over solid waste and burned in an area known as the Burn Pit (Turner et al., 1988). These oils are believed to have been comprised of non-PCB containing fluids such as mineral oils, motor oils, anti-freeze, and cutting fluid from the depleted uranium machining operations (Bailey, 1995a,b). There is no documentation available that describes the exact quantities or origins of waste oils that were burned in the Burn Pit. In 1961, a surface tank, known as the Burn Tank, was installed in Burial Ground A to collect waste oils and coolants to be burned; liquids that did not burn were drained into adjacent trenches. An estimated 180,000 gallons (681,300 liters) of waste oils were burned in the Burn Tank from 1961 to 1968 (Turner et al., 1988), when the practice was terminated due to regulations imposed under the Clean Air Act.

The available documentation suggests that significant quantities of PCBs were not burned at Burial Ground A. The primary source of waste oils containing high concentrations of PCBs was related to the disposal of M-Wing coolants which occurred in 1970, 2 years after the practice of burning had been discontinued (Bailey, 1995a,b). However, because there is no documentation that describes either the specific origin of the waste oils disposed of at the Burial Grounds prior to 1970 or their range of PCB concentrations, it is not possible to state with certainty that PCB-containing waste oils were not burned there. It is possible that small amounts of transformer oils and hydraulic fluids (both of which contained PCBs) may have been burned. If oils with high levels of PCBs were disposed at the Burial Grounds during the time that burning was taking place, it is likely that the elevated PCB content in those oils would have rendered them nonflammable.

<u>Oil Retention Ponds</u>

Beginning in 1969, oils and coolants were poured through standpipes vertically installed in the trenches in the Burial Grounds (Turner et al., 1988). In 1971, oils were observed seeping from the western end of Burial Ground A into small tributaries that emptied into Bear Creek. In order to trap these oils, two oil retention ponds were constructed on either side of the Burial Grounds. Oil Retention Pond 1 was excavated in the southwest corner of Burial Ground A in 1971. A ditch was also excavated in order to divert surface water away from the pond and reduce the potential for overflow. The pond was equipped with overflow pipes to permit water discharge while retaining floating oils in the impoundment. Oil Retention Pond 2 was constructed on the northeast corner of Burial Ground A in 1972, in order to collect and retain oils observed seeping into an intermittent stream along its east side.

The majority of oils flowing to the Retention Ponds were trapped by Oil Retention Pond 1, which had a maximum depth of six feet (1.8 meters) and received runoff from an area of approximately eight acres (Bailey, 1986). From 1974 to 1984, approximately 38,000 gallons (143,830 liters) of oil were skimmed from Oil Retention Pond 1 (Bailey, 1986). This oil was found to be contaminated with PCBs. Of this quantity, 15,000 gallons (57,000 liters) were disposed of in the Oil Landfarm in 1974 (see below); 5,000 gallons (19,000 liters) were sprayed on nearby trees to control an infestation of pine beetles in 1975, and in 1979, 18,000 gallons (68,000 liters) were placed in an underground tank near Building 9754 at the Y-12 plant (Bailey, 1986). In the mid 1980s, close out plans were prepared to drain the water from the ponds, remove the PCB-contaminated sediment, and install a collection system for seeps coming from the old trenches (Napier and Hancher, 1988). By 1992, remediation and capping of the Oil Retention Ponds had been completed (OREP, 1992).

The Oil Landfarm: 1973-1982

In 1973, an innovative disposal technology, in which oily wastes were biodegraded following application to nutrient adjusted soils, was initiated in an area known as the Oil Landfarm, located immediately to the west of the Burial Grounds. Over 1 million gallons (3 million liters) of wastes were disposed of in this fashion from 1973 to 1982, after which operations were suspended pending negotiations with the Tennessee Department of Health (Bailey, 1986). While landfarming was demonstrated to be very effective in degrading the organic components of oil (Francke et al., 1974), there was little information regarding the effectiveness of this approach for chlorinated compounds (Francke et al., 1974).

Waste oils originating primarily from Y-12 and K-25 were disposed at the Oil Landfarm from April to October, and were frequently tilled into the soil in order to maintain aerobic conditions for enhanced biodegradation. It is estimated that the oily wastes disposed at the Oil Landfarm were composed of approximately 85 percent oil and 15 percent water. These oils are known to have been contaminated with a variety of compounds including beryllium and beryllium oxides, depleted uranium, tetrachloroethene, 1,1,1-trichloroethane, and PCBs (Turner et al., 1988); however, the precise PCB content of these oils could not be determined. Although regulations prohibited the disposal of oils contaminated with concentrations of PCBs in excess of 5 ppm (Bailey, 1986), high concentrations of PCBs reported in Turner et al. (1988) from the Oil Landfarm (up to 110 ppm) indicate the possibility that oil disposed at the Oil Landfarm may have contained PCBs in excess of the allowable level.

Salvage Yard/Solvent Drum Storage Area

In addition to the Bear Creek Disposal Area, waste was stored at the Salvage Yard/Solvent Drum Storage Area located at the northwest section of the Y-12 facility. Chemical storage drums were stored and crushed for disposal in this area. In addition, approximately 175,000 gallons (660,000 liters) of waste oils, including chlorinated organic materials and solvents, were stored in this yard. Two tanks were installed in this area in the late 1970s to store approximately 11,000 gallons (41,600 liters) of PCB-contaminated oils. Any spills occurring in this area were released to the storm drain system (Kornegay et al., 1991).

Storage and Eventual Incineration of PCB Waste Oils: 1982 - Present

After 1982, waste oils were stored at several tank farms at Y-12 awaiting incineration at the K-25 TSCA incinerator. The oils were generally separated according to their PCB content; waste oils containing greater than 5 ppm were kept separate from those containing lower concentrations. In 1987, this concentration limit was decreased to 2 ppm. Some of the waste oils containing less than the concentration limits were sent off-site for commercial disposal. It is estimated that from 1982 to 1991, approximately 150,000 gallons (568,000 liters) of PCB-containing waste oils had accumulated at Y-12. In 1991, when the K-25 incinerator began operations, these oils were sent to the K-25 incinerator and, by 1995, most of these oils had been burned (Bailey, 1995).

3.1.1.3 Potential Off-site Exposures to PCBs from Y-12

Based on the review of the historical uses and releases of PCBs from Y-12, the most likely routes of exposure to off-site populations were due to transport of PCBs to the sediments, surface water, and biota of Bear Creek and EFPC. Historical concentrations in Bear Creek were likely related to the disposal of PCB-containing oily wastes at the Bear Creek Disposal Area (Turner et al., 1988), while those measured in EFPC were likely due to unregulated outfalls discharging to the creek and incidental leaks and drips from the Z-oil system (SAIC, 1994). Of the streams, EFPC was the more likely source of exposure, as it flowed through non-restricted and residential portions of the city of Oak Ridge, while most of Bear Creek was contained within the restricted access area (Turner et al., 1988).

Potential exposures to PCBs in EFPC could have occurred via a number of pathways, including incidental ingestion of and dermal contact with contaminated water and sediments during recreational activities, consumption of contaminated fish and other aquatic organisms, consumption of meats and dairy products from livestock grazed on floodplain soils, consumption of vegetables produced on floodplain soils, and dermal contact while gardening on floodplain soils. Although the creek has never been large enough to support a substantial recreational fishery, interviews conducted with Oak Ridge residents have indicated that fishing for minnows and other small fish from EFPC was a relatively common activity, and that fish caught in EFPC may have been consumed. Additionally, turtles and frogs from the creek were occasionally consumed (Mills et al., 1995).

Prior to 1963, when an oil retention pond was constructed at the northern end of the plant, releases to EFPC could have resulted in the transport of oils and associated PCBs off-site. However, New Hope Pond, later replaced by Lake Reality, likely prevented the migration of a significant fraction of the total residual oil in EFPC to off-site portions of the creek (Napier, 1995). Thus, it is not surprising that the limited data describing PCB concentrations in sediments and fish from EFPC indicate that levels are relatively low.

Inhalation of airborne PCBs could also have occurred as a result of the open pit burning activities at Burial Ground A. Low temperature combustion, such as occurred in the Burn Pit and the Burn Tank, may have resulted in aerial transportation of limited quantities of both vapor phase and particulate bound PCBs. In addition, incineration of PCBs under poorly controlled conditions could have resulted in the formation of chlorinated dioxins and furans (Jansson and Sundström, 1982; Erickson, 1985). Based upon the data collected, it is unlikely that oils containing high concentrations of PCBs were incinerated. Waste oils containing high concentrations of PCBs are nonflammable and would have been disposed in burial pits. In addition, the only documented wastes with high concentrations of PCBs (the cutting fluids) were disposed in the 1970s after the practice of burning of waste oils had been discontinued. It is possible, however, that wastes containing lower concentrations of PCBs (up to several hundred parts per million) could have been burned at the facility, potentially resulting in PCB levels in ambient air and also causing the formation of low levels of chlorinated dioxins and furans.

3.1.1.4 Conclusions

PCBs were used in electrical equipment and as coolants in the machining of enriched uranium. They also occurred as contaminants in oils used in a heat exchange system for calutrons in the electromagnetic separation of uranium (the Z-oil system), and in hydraulic systems. PCB migration to areas beyond the perimeter of the plant occurred via discharges to Bear Creek and EFPC, and from the burning of waste oils at the burial grounds. In addition, waste oils containing PCBs may have been sold to the public and used for dust suppression, as a pesticide or herbicide, or for a variety of other uses. The results of this investigation indicate that exposure of residential populations to PCBs originating from Y-12 was minimal and was confined to potential inhalation exposures during burning periods and exposures to residents living on or near the floodplain of EFPC. Although high sediment and floodplain soil concentrations have been detected at the headwaters of Bear Creek, these areas generally were not accessible to individuals other than ORR workers, and have not resulted in historical and current residential exposures. For the residents around EFPC, consumption of contaminated meats, dairy products, and vegetables grown on floodplain soils, as well as dermal contact and incidental ingestion of contaminated soils/sediments and consumption of recreationally-caught fish or biota are pathways that may have resulted in potential exposures.

3.1.2 X-10

The X-10 site was built in 1943 and served as a pilot plant for demonstrating chemical techniques of plutonium separation. A network of underground storage tanks and pipelines was constructed in 1943 to handle and store the radioactive and chemical waste liquids generated by these separation operations. Once a laboratory wholly dedicated to nuclear technology research and development, X-10 presently includes multidisciplinary efforts in non-nuclear technologies and sciences.

X-10 is located on the southern border of the ORR. The valley floor is highly developed within the central site area and the surrounding terrain is wooded. The facility discharges to two small streams on-site, First and Fifth Creeks, which discharge to White Oak Creek. White Oak Creek passes near south of the developed area, leaves the valley through a gap in Haw Ridge and then enters Melton Valley. In Melton Valley, White Oak Creek flows into White Oak Lake, which was formed by White Oak Lake Dam, located 1.7 miles upstream from the confluence of White Oak Creek and the Clinch River, and built by the Tennessee Valley Authority (TVA) in 1943. White Oak Creek Embayment lies between White Oak Lake and the Clinch River.

3.1.2.1 Historical Uses and Releases Involving PCBs at X-10

Historical activities at X-10 required electrical equipment such as capacitors, transformers, pumps, and electric motors. Lubricating and cooling oils associated with this equipment likely contained PCBs. The primary use of PCBs at X-10 was in the form of dielectric oils in electrical transformers at concentrations ranging from <5 to 1 million ppm (ORNL, 1991). Prior to the 1970s, there was little information regarding these transformers. In the early 1970s, a number of these transformers were drained and filled with non-PCB oil, and by 1982, the electrical distribution system at X-10 included only 25 transformers containing PCB dielectric material or contaminated oil at 50 ppm or greater (Boyle et al., 1982; Bruce, 1995). In 1986, approximately 12 transformers, each containing 500 to 600 gallons (1,890 to 2,271 liters) of PCBs, were removed by Westinghouse and incinerated off-site (Bruce, 1995). These transformers had been housed inside Buildings 4500S, 6000, 4509, 7900, and 2026. All remaining transformers at the site (approximately 300) were sampled by 1986 and, by the summer of 1995, each transformer containing PCBs at concentrations greater than 50 ppm had been drained and refilled with non-PCB oil (Bruce, 1995). The PCB-contaminated oil was disposed in accordance with PCB regulations.

Documentation of historical spills of PCBs and PCB Annual Reports for X-10 reporting accidental spills inside X-10 buildings indicate that floor drains and outfalls from these buildings may have been sources of PCB releases to White Oak Creek. A history of oil spills at X-10 reported approximately 99 spills, varying in size from one ounce (28 grams) to five gallons (19 liters), at Building 4500S between 1981 and 1994 (ORNL, 1995). The project team assumed that these oils contained some level of PCBs and that the frequency and quantity of the spills were indicative of those that might have occurred historically at this building. Building 4500S was a potential source of PCBs to White Oak Creek. Building 2026, which housed PCB oil-filled transformers, was defined in the 1984 PCB Annual Report for X-10 as a "building or area that either contains > 500 ppm PCB or that does/will serve as storage areas for PCBs". The outfall area for any releases from Building 2026 would have been First Creek. In addition, in 1986 a transformer containing 100 percent PCBs was removed from Building 4509, a building which discharged waste directly to White Oak Creek (PCB Annual Report for ORNL, 1985). Similarly, Building 6000 for which the outfall area was White Oak Creek, contained six PCB oil-filled transformers until 1986 (PCB Annual Report for ORNL, 1985). In addition to PCB use in electrical equipment, the Annual PCB Inventory Report for X-10 (ORNL, 1993) reported that PCBs were used in the heat transfer and hydraulic systems at X-10. The extent of these uses was not reported.

3.1.2.2 Waste Disposal of PCBs at X-10

Historically, waste disposal at X-10 occurred at on-site burial grounds or at on-site surface impoundments. Unfortunately, records of disposal of PCB-containing materials at these sites, prior to environmental regulation, were not found. Likewise, interviews with retired personnel, who might have had knowledge of disposal practices between the beginning of operations and the late 1970s, failed to uncover additional information. The lack of information on PCB waste disposal at X-10 was likely a result of the lack of awareness of the potential hazards associated with PCBs prior to the 1970s. As a result, this analysis has focused on recent studies conducted on disposal and treatment sites at X-10 in an effort to determine the types of waste present and their possible sources. The following sections summarize the information provided by these reports.

<u>Burial Grounds</u>

Limited information is available pertaining to the original Burial Grounds located along White Oak Creek (Figure 3-4). The precise locations of these grounds were not uncovered in the literature; only general information on the types of wastes received by the Burial Grounds was

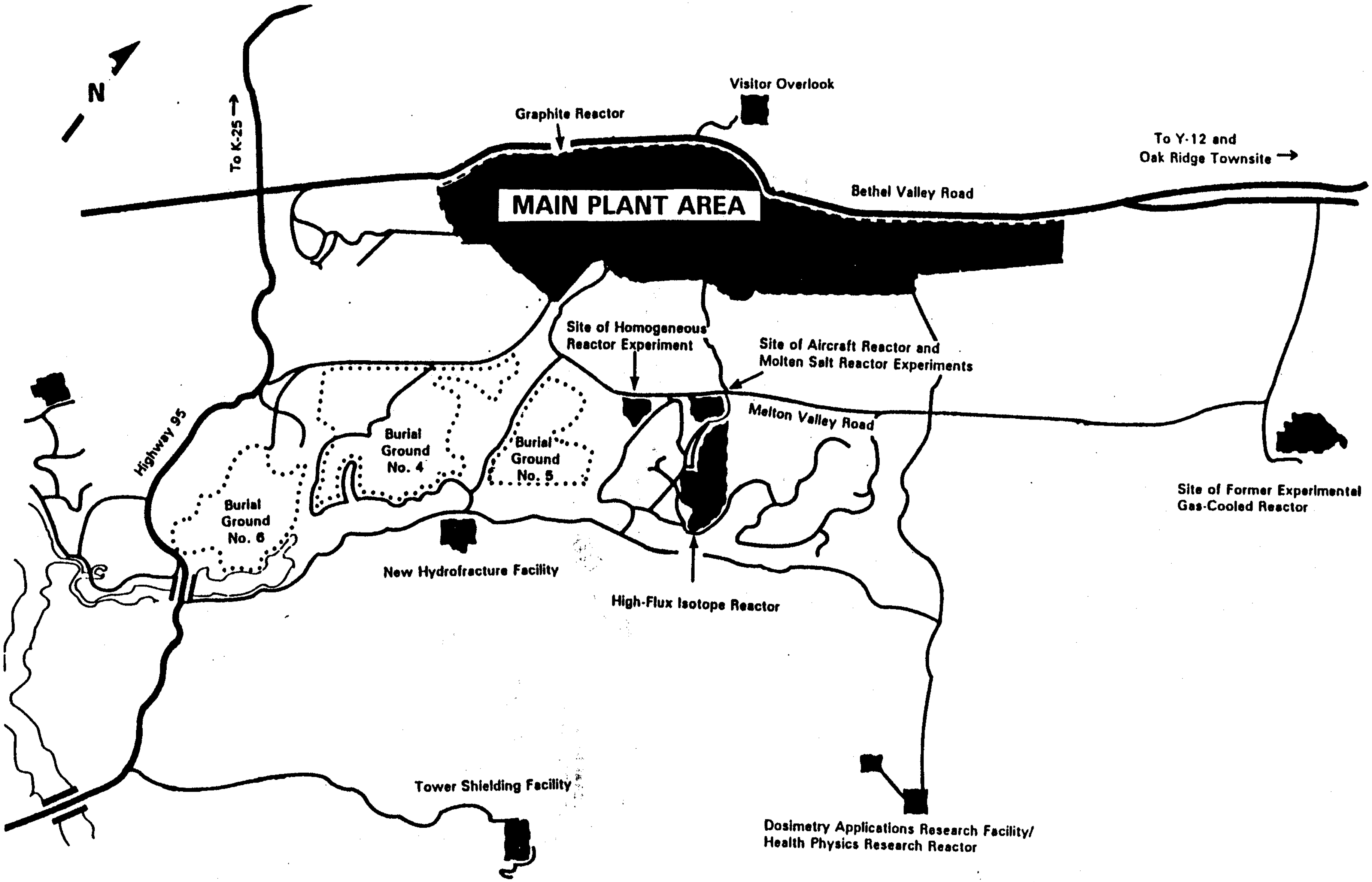


Figure 3-4. Map of the Oak Ridge National Laboratory Outlying Areas

obtained. Burial Grounds 1, 2, 3, and 5 were primarily limited to radioactive waste. Burial Ground 4 received waste from government agencies and private companies, while chemical and biological wastes were disposed in unlined trenches located at Burial Ground 6. In addition to Burial Grounds 1 through 6, Waste Pit 1 was built in 1951. Although discharges were terminated only months later due to leaks, drains in the decontamination building continued to discharge to Waste Pit 1 until 1964.

<u>Waste Area Groupings</u>

Waste Area Groupings (WAGs), which delineated areas of contamination, were defined along White Oak Creek, as well as other areas of X-10, in 1987. Due to the large number of sites and the complex hydrology and geology at X-10, WAGs were developed in response to regulatory requirements pertaining to remediation of the site (Clapp et al., 1994). Because the WAGs delineated areas of similar contamination as historical Burial Grounds, their arrangement likely reflected that of the original Burial Grounds. The WAGs were generally defined by local geographic watersheds or drainage basins that contain contiguous and similar remedial action sites. Hydrologic interaction among some sites within a WAG makes individual sites hydrologically inseparable. A total of 21 WAGs have been identified at ORNL, 14 of which have been identified as candidates for further action. With the exception of WAGs 2 and 21 (groundwater operable unit), WAGs are sources of contaminants to other areas of the site and are termed “contaminant source WAGs”. WAG 2, which consists of White Oak Creek below the 7500 Bridge monitoring system at Melton Valley Road, Melton Branch and associated floodplains, White Oak Lake, and the White Oak Creek Embayment at the confluence of the Clinch River, is downgradient from all WAGs in the watershed. Because it receives and integrates the contaminants released from the other WAGs in the watershed that move through the surface water system, WAG 2 is termed an “integrator” WAG and serves as a conduit for contaminants from the X-10 WAGs to off-site areas (Clapp et al., 1994). Each WAG is made up of one or several solid waste management units (SWMUs), and the wastes received by these units are very specific (e.g., radioactive versus organic). Certain WAGs are known to contain PCBs or are believed to contain PCBs based on the type of wastes received.

Several SWMUs at the Main Plant Area (WAG 1) are associated with PCB usage and storage. Building 2018N is a PCB storage area and Building 2525 houses two holding tanks for waste oil, the contents of which have not been sampled. The outfall area of both of these sites is First Creek. Other potential sources of PCBs at WAG 1 include Building 3087, which houses the Oak Ridge Research Reactor Heat Exchanger, and Hazardous Waste Treatment and Storage Facilities, including Buildings 7652, 7653, 7654, and 7651. Fifth Creek likely receives discharge from Building 3087, while the Hazardous Waste Treatment and Storage facilities discharge to Melton Hill Lake.

Surface impoundment operable unit (SIOU) is part of WAG 1 and consists of Impoundments 3513, 3524, 3539, and 3540. Impoundment 3524 was built in 1943 as an equalization basin and served as backup storage for Gunite tanks in South Tank Farm, where radiochemical and metal wastes were stored as the first step in the waste management process. From 1944 to 1949, Impoundment 3513 was added to provide settling space and for diluting partially decayed and dissolved radioactive wastes. Between the years of 1954 and 1957, Impoundments 3513 and 3524 received process wastewater from various facilities.

In 1964**,** Impoundments 3539 and 3540 were constructed to hold process wastewater from the Building 4500 complex. Although the Building 4500 complex has not been identified as a source of PCBs (ORNL, 1985), the history of oil spills at X-10 indicated that at least 129 spills have occurred at Buildings 4500S and 4500N since 1972 (ORNL, 1995). While the specific type of oil is not provided for each spill reported, the presence of PCBs is a possibility in each case. If PCB-containing used oil was disposed in the wastewater stream of the 4500 complex, this may explain the origin of PCBs in Impoundments 3539 and 3540.

In 1976, Impoundment 3513 was removed from service. Impoundment 3524 continued as an equalization basin until 1989, when tanks were constructed to store incoming waste. In 1990, effluent from the Building 4500 complex was rerouted to the waste storage tanks and Impoundments 3539 and 3540 were removed from service. Currently, Impoundments 3524, 3539, and 3540 are used for emergency waste storage and will be removed from that service when new tanks are constructed to provide emergency storage.

Investigations into the presence of PCBs at the SIOU did not occur until 1984. Although detailed information on the material received by the impoundments was not identified in the literature, the presence of PCBs has been confirmed in part of the area (Jacobs ER Team, 1995).

WAG 2 receives groundwater seepage and surface drainage from nonpoint sources such as the solid waste storage areas (SWSAs) (WAGs 3, 4, 5, and 6), the liquid waste seepage pits and trenches area (WAG 7), and the experimental reactor facilities in Melton Valley (WAGs 8 and 9). Results of weir pool sediment sampling at WAG 2 indicate widespread contamination of stream sediments in the Melton Valley area with Aroclors 1254 and 1260. PCBs at T2A weir at WAG 2 could come directly from stream discharge of the WAG 4 tributary, the Intermediate Holding Pond, or the main plant area because T2A is inundated by White Oak Lake during flood events (Clapp et al., 1994).

WAG 3 is located just east of the northwest tributary of White Oak Creek. ORNL (1987) suggests that hazardous chemical wastes, such as PCBs, were placed in WAG 3. Although burial of solid waste ceased at this site in 1951, the area continued to be used for above-ground scrap metal storage until 1979. Burial records for this site were destroyed in a fire in 1961 (Coobs and Gissel, 1986).

WAG 4 is located south of the main plant area and is adjacent to the west side of White Oak Creek. SWSAs at WAG 4 were in use from 1943 to 1963 (ORNL, 1987). Wastes including paper, clothing, equipment, filters, animal carcasses, and related laboratory wastes, were placed in trenches, shallow auger holes, and in piles on the ground surface to be covered at a later date. Early records on the amount of waste disposed in WAG 4 were destroyed by a fire and later records did not include information on the proportion of each type of waste buried at the site. However, ORNL (1987) reported that approximately half of the total volume buried in WAG 4 was from outside sources. This information indicates the possibility of buried organic waste, including PCBs, at the site.

WAG 11, also known as the White Wing Scrap Yard, is a largely wooded area located on the western edge of East Fork Ridge, northwest of X-10. This site is distant from X-10 and is contained within K-25 grid coordinates. The White Wing Scrap Yard was used for the above-ground storage of contaminated material from X-10, K-25 and Y-12 plants. Starting in the 1950s, steel tanks, drums, trucks, earth-moving equipment, assorted pieces of glass, steel, stainless steel, aluminum and reaction vessels used in Building 3019, and electrical capacitors and transformers were stored at this site (ORNL, 1987; CDM Federal Programs Corporation, 1994). In 1966, work was begun to clean up the site and cleanup continued until October 1970 (CDM Federal Programs Corporation, 1994). Most of the large surface scrap was buried in X-10's Solid Waste Storage Area 5. Approximately 6000 cubic yards of contaminated soil was removed from the site and disposed in accordance with PCB regulations (CDM Federal Programs Corporation, 1994).

An Interim Remedial Measures Study was issued in 1992 and an Interim Record of Decision (IROD) required that all surface debris be removed and disposed at WAG 6 (CDM Federal Programs Corporation, 1994). A surface radiological investigation of WAG 11 conducted between late 1989 and early 1990 uncovered an old transformer/capacitor device that was partially coated with an oily substance. During the IROD, other capacitors were found; studies have confirmed that the oily substance contained PCBs (concentrations in surrounding soil were reported to be as high as 250,000 ppm) (CDM Federal Programs Corporation, 1994). Because WAG 11 is distant from the X-10 facility, it is unlikely that WAG 11 is associated with PCB contamination of X-10 or its water bodies.

WAG 15 is located at the Y-12 plant. Building 9201-2 of WAG 15 contains Z-oil, known to be contaminated with PCBs. Another site at WAG 15 includes areas that house PCB transformers (Buildings 9204-1, 9204-3, and SY200 Yard).

WAG 17, the major craft and machine shop area for X-10, is located approximately one mile east of the Main Plant Area. Two underground and three above-ground oil storage tanks are located at this WAG. All of the above-ground and one of the underground tanks collect waste oils from vehicle maintenance and cutting oils from machining operations. Most of the above-ground tanks are diked and no leaks, spills, or other means of release of oil or hazardous materials from any of the tanks have been reported (ORNL, 1987). However, this area has been in use since X-10 operations began in 1943 and it is probable that some spills or leaks of waste oils and solvents have occurred but have not been documented (ORNL, 1987). Although Tank 7030E (installed in 1975) contains waste fuel reported to contain PCBs (ORNL, 1987), no data are available relative to the presence of PCBs in the waste oil of other storage tanks.

WAG 19 contains six SWMUs that represent X-10 hazardous waste treatment and storage facilities. One cluster of SWMUs includes the permitted hazardous and mixed waste storage units. Spills or leaks have not been recorded at any of the sites in WAG 19. While all of the facilities have been designed to contain any spills or leaks that occur in the waste containers handled or stored, historical releases of contaminants may have occurred. Drainage from WAG 19 enters Beaver Creek and Melton Hill Reservoir (upstream of White Oak Lake's confluence with the Clinch River) and does not impact the White Oak Creek watershed.

In summary, there is little documentation of the amounts of PCBs placed in the various WAGs at X-10. Management of PCB-containing wastes between the time X-10 operations began and when environmental regulations took effect in the late 1970s was minimal and no records were kept. In contrast, since the late 1970s, PCB releases have been handled according to stringent federal regulations and the policies set by the ORR. During the 1970s, 1980s, and 1990s, surveys of PCBs in environmental media found widespread low level contamination in the areas around and downstream of X-10. Although records of the last 15 years indicate that releases from the facility have been negligible, measurable levels of PCBs exist in White Oak Creek Embayment and White Oak Lake. This suggests that PCBs have been released from X-10 operations. It is not clear whether these observed levels have resulted from releases that occurred prior to the late 1970s or from ongoing low level releases.

3.1.2.3 Potential Off-site Exposures to PCBs from X-10

PCBs have been detected in water bodies downstream of X-10. However, warning signs and physical barriers restrict access to White Oak Creek, White Oak Embayment and White Oak Lake. These restrictions have been in place since operations began at the ORR. Posting the area likely prevented frequent use of these water bodies by the public, and exposures to sediment and surface water would have been minimal. Thus, although it appears that releases of PCBs from X-10 occurred, it is unlikely that individuals were exposed to off-site releases of PCBs. Because the exposure pathways are incomplete, exposures to PCBs associated with releases from X-10 are not considered further. It should be noted that PCBs likely entered the Clinch River from White Oak Creek. This contribution was included in the evaluation of exposures from the consumption of Clinch River fish.

3.1.2.4 Conclusions

At X-10, PCBs were primarily used in electrical equipment and in heat transfer and hydraulic systems. Although PCB migration to areas beyond the perimeter of the plant occurred via discharges to White Oak Creek, public access to the area has been prohibited since the beginning of operations at the ORR. Thus, it is unlikely that individuals were exposed to off-site releases of PCBs originating from X-10. However, the contribution of PCBs from White Oak Creek to the Clinch River was included in the evaluation of exposures from the consumption of Clinch River fish.

3.1.3 K-25

K-25 was the name for the Oak Ridge Gaseous Diffusion Plant (ORGDP) at the ORR. The ORGDP is located on 640 acres near the junction of Poplar Creek and the Clinch River. Poplar Creek flows from northeast to southwest, approximately transecting the center of the ORGDP. The complex is accessible by Blair Road from the north and Tennessee Highway 58 from the northeast and southwest (Figure 3-5).

K-25 began operating in 1945 and continued to operate until the 1980s. Its initial purpose was to provide slightly enriched uranium as feedstock for the calutrons at Y-12. After the end of World

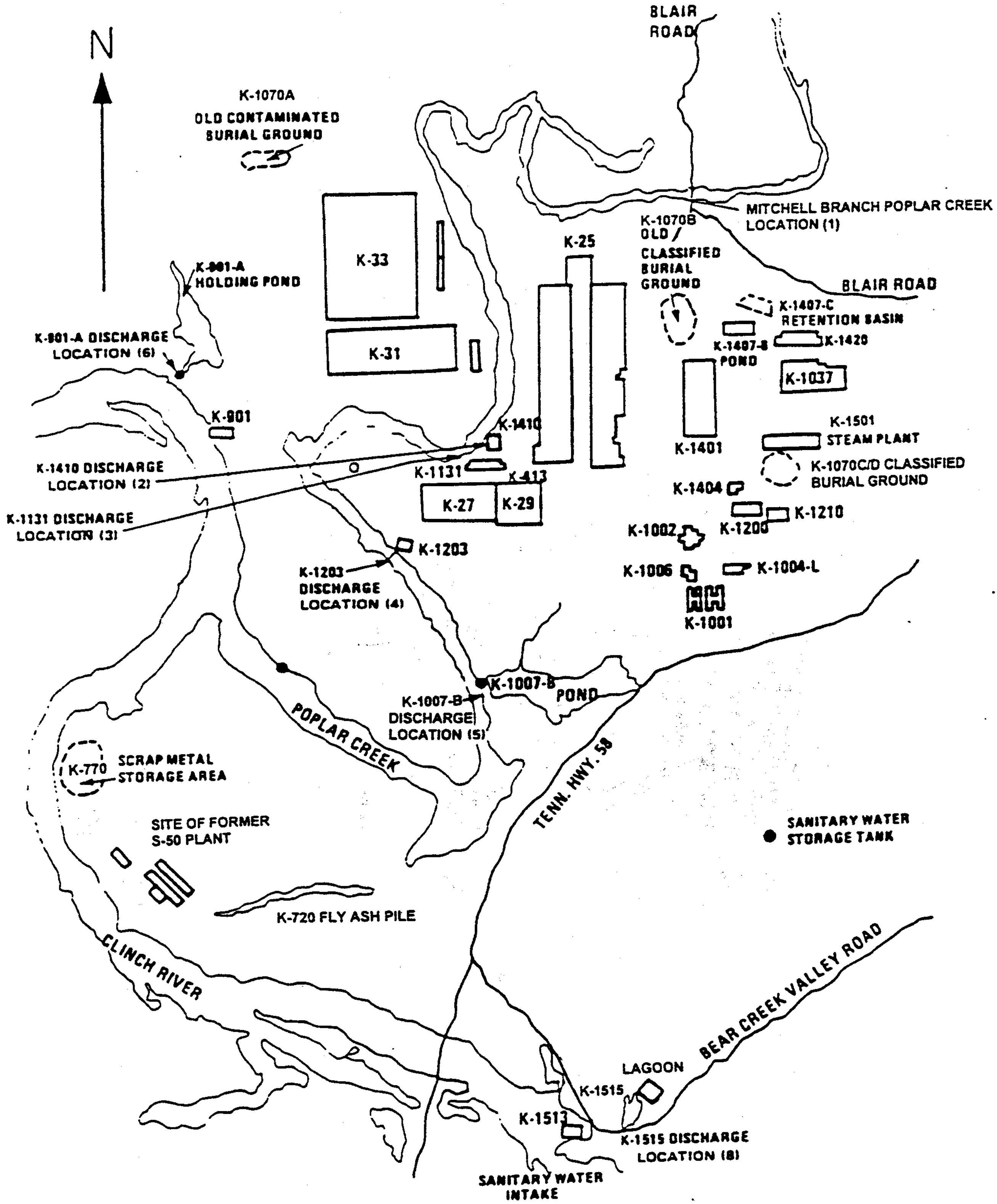


Figure 3-5. Oak Ridge Gaseous Diffusion Plant (ORGDP) Site Map

War II, the demand for enriched uranium decreased at Y-12; as a result, K-25 began providing it to other facilities. By the 1950s, K-25 supplied all the enriched uranium used in the United States for commercial and military purposes until the demand for weapons grade uranium decreased in 1964 (OREP, 1992). From 1964 until 1985, K-25 enrichment operations were limited to production of lower grade uranium for commercial reactors, and in 1984 the plant was shut down (OREP, 1992). Currently, K-25 primarily functions as headquarters for waste storage, treatment, and disposal activities for the ORR (OREP, 1992).

3.1.3.1 Historical Uses and Releases Involving PCBs at K-25

The primary use of PCBs at K-25 was in the electrical power system for the gaseous diffusion cascades (Jordan, 1976). Very large amounts of electrical power were brought into K-25 from a number of sources and distributed to the pumps that were a part of the gaseous diffusion process. Transformers, capacitors, and other types of electrical equipment were filled with dielectric fluids comprised of either mineral oils or PCB-based oils (Askarels). Though relatively costly, PCB oils were used in a number of types of electrical equipment, such as the cascade transformers, due to their fire retardant properties.

Based on plant records and interviews with the ORR personnel, approximately 200 transformers, representing about 50 percent of the electrical transformers that provided energy for the gaseous diffusion process, were filled with PCB fluids (Haymore, 1988; Keeling, 1992; Mitchell, 1993). In addition, approximately 10,000 capacitors, each containing approximately 1.3 gallons (5 liters) of fluid, were filled with Askarel. Altogether the surveys indicated that approximately 125,000 gallons (473,000 liters) of electrical PCB-type oils were contained in electric transformers and capacitors used at K-25 during its period of operation. The majority of these PCB fluids were removed and incinerated off-site in 1989 and 1991 (Keeling, 1992). During this process, it was estimated that between 5,000 and 10,000 gallons (19,000 to 38,000 liters) of Askarel fluid either remained behind in the carcasses of the transformers or were otherwise unaccounted for (Nolan, 1995).

A large number of electrical transformers were also used at the electrical switchyards at K-25. Although the fluids used in the switchyard transformers were mineral oil-based, they contained PCBs due to cross-contamination with Askarel fluids (Nolan, 1991). In addition, some of the ancillary equipment at the switchyards contained PCBs (Blanchard, 1995). This source of PCBs was a major concern for PCB management at K-25 in the 1980s and 1990s (Perry, 1988; Keeling, 1993).

In 1951, two transformer explosions and a fire occurred at the K-762 switchyard (also known as the K-31 substation) near the K-31 process area (MMES 1991). This undoubtedly resulted in the release of PCBs. In 1982, there was an equipment failure at the K-732 switchyard (also called the K-27 substation), located in the process area west of the K-27 and K-29 buildings (MMES, 1991). Records indicated that approximately 2,900 gallons (11,000 liters) of PCB-contaminated mineral oil were released at that time. An oily film, observed on Poplar Creek a few days after this release, was traced to a sump pump valve and storm drain near the switch house that drained to Poplar Creek (MMES, 1991). Another transformer failure was reported at the K-709 switchyard (also known as the K-25 substation). This incident occurred in 1950; however, information was not available as to the extent of any PCB releases (MMES, 1991).

Because PCBs persist in sediments and soils, trace levels remain in the pipes and sumps of waste water systems for long periods of time. As a result, recent monitoring data can provide information on historical releases. Measurable levels of PCBs have been found in the sediments and waste water discharges of the storm sewers draining the switchyards, providing confirmatory evidence of historical releases of PCBs from this area (Goddard, 1995). These data do not, however, provide information on the amount of PCBs released or the dates of the release. In addition, it remains unclear whether the switchyards were a major or a minor source of PCBs released from K-25. Regardless, any releases of PCBs from the switchyards would have directly entered Poplar Creek and been transported downstream with sediments and surface water to the Clinch and Tennessee Rivers (Goddard, 1995).

Felt gaskets impregnated with PCBs connected sections of ventilation systems in the K-29, K-31, and K-33 gaseous diffusion plant buildings (Perry, 1988; Latham, 1995). Lubrication oil that condensed on the inside of the ventilation systems leached PCBs from these gaskets and, in certain instances, leaked through the gaskets and dripped onto the floors (Latham, 1995). The total volume of lubrication oil in the ventilation duct gaskets in the three process buildings was estimated at 232,000 gallons (15,000 gallons [57,000 liters] at K-29; 85,000 gallons [321,000 liters] at K-31; and 132,000 gallons [500,000 liters] at K-33) (Perry, 1988). While sampling results of the oil indicated relatively high levels of PCBs (Perry, 1988), the potential for off-site migration of these PCBs was low. The oil dripped from gaskets located indoors and was cleaned up with dry sorbants that were subsequently landfilled. In addition, a trough system was installed in 1989 to capture any leaking oil. Gaskets are still in place due to a moratorium on removal of PCB equipment (Nolan, 1995).

Other nonelectrical sources of PCBs at K-25 included various commercial products such as oil-based paints, coatings, lubricants, adhesives, sealants, inks, and copy paper. From the 1950s to the 1970s, many of these products contained varying levels of PCBs, which were often added as a nonflammable component in the product formulations (Versar, 1976; Hollinger, 1995). The presence of PCBs in these miscellaneous products used at K-25 probably resulted in numerous low-level releases to the environment.

3.1.3.2 Waste Disposal of PCBs at K-25

Even before TSCA regulations took effect in the late 1970s, management of PCBs at K-25 was an important issue. In 1971, the Department of the Army established a PCB standards committee, and in 1972, PCB handling and disposal guidelines were issued for the ORR (Winkel, 1973). A PCB task force, established in 1973 to oversee PCB handling at the site, was charged with four responsibilities: 1) prepare an inventory of PCB-containing material; 2) determine corrective measures and associated costs needed in order to comply with existing and anticipated regulations; 3) establish a schedule for corrective action; and 4) follow through with the outlined tasks (Winkel, 1973). This indicated that there was a heightened consciousness at the ORR regarding the potential hazards of PCBs and that measures had been put in place in the early 1970s to facilitate proper handling and disposal of PCB-containing materials.

There was limited information available regarding PCB management and disposal practices at K-25 prior to the 1970s. It is known that there were few reports of catastrophic events, such as electrical explosions, fires, or massive equipment failures, that would have resulted in significant releases of PCBs. It is also known that PCB-based oils used in electrical equipment required minimal maintenance, thus minimizing their handling (Versar, 1976) and, because PCB-based fluids were relatively costly, they would not have been handled or disposed of carelessly. In fact, indications suggested that PCB oils were conserved as a routine practice and reused whenever possible (Nolan, 1995).

Nevertheless, PCBs were released into the environment at low levels from such sources as small leaks and spills from active and stored electrical equipment, routine transfer of PCB-based oils, and landfilling of PCB-contaminated equipment and drummed waste containing PCBs (MMES, 1995). Such releases likely occurred throughout the plant's operating history, although the waste management practices instituted in the 1970s as a result of PCB legislation resulted in stricter handling and storing practices (Perry, 1988; Keeling, 1993). Historical waste areas that likely contributed to off-site releases of PCBs from K-25 included the burial grounds, holding ponds, leaking electrical equipment, and outdoor storage areas where surface runoff could have transported PCBs to Poplar Creek and the Clinch River (OREP, 1992).

<u>Burial Grounds</u>

Burial Ground K-1070-D was the largest of a number of burial grounds that existed at K-25 and was known to contain PCBs. K-1070-D, located east of K-25, was used for the disposal and storage of classified materials generated by the gas centrifuge program and other plant operations. At least four disposal trenches existed at K-1070-D. Wastes included low-level radioactive materials and non-radioactive waste materials and equipment, including PCBs. The PCBs entered the Burial Ground from the disposal of electrical equipment, PCB fluids, and from commercial products that contained PCBs (Goddard, 1995; SAIC and J.E.R. Team, 1995).

A Remedial Investigation (RI) for the K-1070-D Burial Grounds reported the presence of PCBs in soil, sediment and groundwater (SAIC and J.E.R. Team, 1995). Currently, leachate from the K-1070 Burial Grounds is intercepted by storm sewers at K-25, and measurable levels of PCBs have been detected in the discharges of the sewers (Goddard, 1995). The sewer water discharges into a pond and into Mitchell Branch, both of which discharge into Poplar Creek (Goddard, 1995).

The K-770 scrap metal yard, located on 25-35 acres of land in the bend of the Clinch River, represents another burial ground at K-25 (Cofer, 1993). This scrap yard operated from the 1960s until the initiation of a waste management tracking system in 1977. Any metal that originated in a process building was brought to the scrap yard. Radioactive materials, PCBs, and mercury were stored in this area. OREP (1992) reported that the scrap yard contained an estimated 40 thousand tons of low-level radioactive scrap metal.

Burn Areas

In addition to the burial grounds, other areas existed where waste materials were stored and burned. The K-1064 Burn Area, approximately three acres in size and located on a peninsula that extends northward into Poplar Creek about three miles above Poplar Creek's confluence with the Clinch River (ORGDP, 1987), was used to store and burn waste solvents, paint wastes, other organic wastes, and organic- and radioactive-contaminated waste oil. Open burning of solvents occurred during the 1950s and 1960s (MMES, 1991). In the late 1960s and early 1970s, drums of solvents, organics (including PCBs), and radioactive-contaminated waste oil were stored. Records indicated that 1,838 drums of waste, amounting to 90,000 gallons (341,000 liters), were stored at the unit. These drums were removed and the unit was closed during 1979. Runoff from the K-1064 area flows either directly into Poplar Creek or collects in storm drains which discharge to Poplar Creek. PCBs have been detected in the soil around the area and in the storm drains at levels of 1 to 15 ppm (ORGDP, 1987).

The K-1085 Firehouse Burn Area, located south of K-25 near the intersection of Highway 58 and Bear Creek Road, originally consisted of a work camp and support facilities during the construction of the K-25 facility (ORGDP, 1988). An existing farmhouse on the site was converted into a firehouse that was operated by the Roane - Anderson Company from 1944 until 1947 when Union Carbide Corporation took over operations. J.A. Jones Construction Company built and operated a garage and a field station in the K-1085 Burn Area until they were dismantled in 1947. The underground storage tanks associated with these operations were covered with concrete pads and left in place. Waste oil was burned in metal pans placed on these pads for fire training exercises. In addition to the waste oil, solvents, heavy metals, and uranium-contaminated oils were burned. An unlined pit in the K-1085 Burn Area also was used for fire training (ORGDP, 1989). Contaminated oil, originating from the process buildings, was placed in the pit and burned. At the end of each day, unburned oil remaining in the pit was extinguished with water and the resulting mixture was released to Poplar Creek.

Accidental Spills

Based on document reviews and interviews with the ORR personnel, there were approximately 10,000 capacitors containing Askarel at the K-25 facility (Hollinger, 1995; Nolan, 1995). Information obtained during equipment inventories and inspections indicated that the vast majority of capacitors remained intact. If there were accidental spills or releases from capacitors, small volumes of PCBs would have been released into the K-25 waste water system and discharged to Poplar Creek and the Clinch River (Nolan, 1995).

In the late 1970s, two 4000-gallon storage tanks were installed in the south end of Unit 8 at K-25. This installation coincided with an upgrade in the power capacity of the cascade system. These tanks stored Askarel fluid that was drained from the transformers prior to sending them for upgrade. The transformers were then refilled with Askarel from the storage tanks upon their return. Two 500-gallon storage tanks "on wheels" in Units 6 and 8 transported Askarel fluid between transformers and the 4000-gallon storage tanks. During the upgrading process, it is possible that small quantities of PCBs were released from the transfer of Askarel fluids (Nolan, 1995).

Drummed waste was another likely source of PCB release to the environment from K-25. A recorded incident occurred in 1991 at Building K-711 which is located near the Clinch River. A drum containing PCB-contaminated waste developed a pin hole leak as a result of internal corrosion. Between 40 and 50 gallons (150 to 190 liters) of material leaked into a diked area located on the concrete floor of the building (Site Program Management Organization, 1992). During an inspection of the storm drain system servicing the building, an oil sheen was observed on the Clinch River. The oil sheen was later determined to contain PCBs. K-25 emergency response personnel performed containment and cleanup procedures and notified the proper regulatory agencies. While obtaining background data to assess closure options, levels of PCBs greater than 600 ppm were also discovered in a separate area away from the incident (Site Program Management Organization, 1992). These data indicated the likelihood of past spills of PCB-contaminated materials and the potential for releases to the Clinch River.

Holding Ponds

At K-25, three holding ponds, K-901A, K-1007B, and K-1407B, were used for the settling and dilution of chemical wastes. Investigations of these ponds confirmed the presence of PCBs in their sediments and in fish living in the ponds, indicating that wastes discharging from the ponds were probably sources of PCBs to the Clinch River.

The K-1407B Pond, located on the north side of K-25 approximately 200 feet south of Poplar Creek and immediately south of the Mitchell Branch, was an unlined pit that contained hazardous materials for more than 40 years (MMES, 1991). The Pond, which operated from 1943 to 1988, received process water from the K-1407A neutralization pit; wastewater discharges from the K-1302 recirculating cooling water supply, the K-1503 neutralization pit, and the K-1421 incinerator drain; and uranium compounds, transuranics, organic degreasers, and oils, some containing PCBs from K-1420 (MMES, 1991). The wastewaters that entered the Pond from these various sources contained organic compounds, metal hydroxides, uranium compounds, degreasers, and oils containing PCBs (MMES, 1991). These wastewaters were discharged to Mitchell Branch and in turn to Poplar Creek (Goddard, 1995). In 1988 the sludge was removed from the Pond and the unit was closed in 1992 (MMES, 1991).

The K-1007B Pond, constructed in the 1940s, covered approximately 25 acres and was located outside of the security fence of the K-25 facility (ORGDP, 1988). From the beginning of plant operations until 1985, chemical by-products of routine analytical laboratory operations were discharged to the K-1007B Pond at a rate of approximately 2,200 gallons (8,300 liters) per year (ORGDP, 1988). The Pond also received drainage from the switchyards, process area, and storm drains (Goddard, 1995). The K-1007B Pond was permitted to discharge to Poplar Creek under National Pollutants Discharge Elimination System (NPDES) Regulations from 1974 until 1985 when releases to the Pond ceased (Goddard, 1995).

The K-901A Pond, located west of the K-31 and K-33 buildings, received wastewater discharge from the uranium enrichment operations at K-31 and K-33, beginning in the late 1950s (MMES, 1991). The Pond began as a marsh-like area; in 1965 a dam was constructed to create the Pond. The discharge consisted largely of sludge and blowdown water from cooling operations. It contained many heavy metals including chromium (MMES, 1991). Sampling in this area indicated PCBs below the detection limit of 1 ppm in sediments. Discharge into the Pond was discontinued in 1985 (MMES, 1991).

3.1.3.3 Potential Off-site Exposures to PCBs from K-25

As previously discussed, the primary use of PCBs at K-25 was in electrical transformer and capacitor fluids. PCB fluids were relatively costly and thus were routinely conserved (Nolan, 1995). In addition, because PCB oils used in electrical equipment did not generally require maintenance or replacement, they were handled minimally, reducing the opportunity for spills and losses. The total volume of PCB-based fluids historically used in electrical equipment at K-25 has been estimated at 125,000 gallons (473,000 liters) (Nolan, 1995). The majority of these fluids were removed and incinerated off-site in 1989 and 1991 (Keeling, 1992). During the history of plant operations, however, it is likely that incidental releases occurred and migrated off-site via surface runoff and storm sewer discharge.

PCBs also were likely available for release and migration off-site from sources other than electrical equipment. Investigations of burial grounds, burn areas, holding ponds, electrical switchyards, and outside storage areas indicated that, although the majority of PCBs released from these areas would have been contained on-site, off-site migration via surface runoff, waste water discharges, and volatilization to air likely occurred. Specific incidents that documented PCB releases at K-25 included an explosion and fire that occurred in 1951 near the K-31 process area (MMES, 1991). In addition, two accidental spills were documented at K-25. One spill involved a leaking storage drum at K-711 in 1991 that resulted in a release of 40 to 50 gallons (150 to 190 liters) of PCB fluids to a diked area on-site, with a portion of that release migrating to the Clinch River via storm water drains (Site Program Management Organization, 1992). The second incident involved an equipment failure at the K-732 switchyard which resulted in approximately 2,000 gallons (7,600 liters) of PCB-contaminated mineral oil being released via a storm drain to Poplar Creek (MMES, 1991).

Based on this review of the historical uses and disposal practices at K-25, potential exposures to off-site receptors are likely to have been associated with the presence of PCBs in sediments and biota of Poplar Creek and the Clinch River. Potential exposures to PCBs near these water bodies could have occurred via a number of pathways, including dermal and oral exposure to contaminated water and sediments during recreational activities and consumption of contaminated fish and other biota. Interviews conducted with Oak Ridge residents have indicated that fishing was a relatively common activity and that many of the fish caught were consumed (Mills et al., 1995).

3.1.4 Summary of Potential Releases of PCBs from the ORR

In general, although there was no conscious attempt to control the discharge of PCBs to the environment before the 1970s, several factors limited the amount of PCBs released from the ORR. First, although incidental releases may have occurred to EFPC or Poplar Creek as the result of spills, the project team identified no evidence that oily wastes were routinely discharged directly to the Clinch River or its tributaries. As discussed for each of the facilities, PCBs were typically treated similarly to other waste oils and were disposed of by a variety of methods including burning them in pits and tanks at Y-12 and K-25, landfarming at Y-12, and land disposal in pits, trenches, and landfills at all three facilities (Turner et al., 1988).

Second, the migration of PCBs off-site was reduced due to the chemical properties of PCBs. PCBs are not mobile in groundwater and, when released to surface water, quickly bind to sediment (ATSDR, 1996). Because of these properties, the majority of PCBs placed in burial grounds or pits have remained in or near these units. Third, small lakes, ponds, and lagoons have been an integral part of the stormwater and waste water management systems at the ORR. These surface water bodies have served as leaky traps for PCBs, PCB-contaminated oils, and PCB-contaminated sediments, and have limited the movement of PCBs off the reservation. Finally, the sediments of Bear Creek, White Oak Creek, and other streams located on the ORR have entrapped a portion of the PCBs released from Y-12 and X-10 and have reduced the amount of PCBs migrating off the reservation (Turner et al., 1988; SAIC, 1994).

While several factors limited the amount of PCBs moving off-site, there is evidence that discharges of PCBs from the ORR resulted in contamination of water, sediment, and fish at off-site locations (TVA, 1988; 1989; 1990; Loar, 1994a). Due to the tendency of PCBs to bind to sediment and soil and to resist degradation, PCBs have persisted in off-site sediments decades after the original releases. Furthermore, contaminated sediments in ponds, stream beds, waste water systems and soils adjacent to surface water bodies have acted as reservoirs for on-going low level releases of PCBs to the Clinch River (Ashwood et al., 1986; Blaylock et al., 1993; Loar, 1994a; 1994b). Because of these contaminated sediments, fish in the Clinch River or waters entering the Clinch River, such as White Oak Creek or Poplar Creek, are also contaminated.

Although releases to surface water and sediment transport represented the primary transport routes of PCBs to off-site locations, it was also necessary to consider other, less significant pathways. For example, there is evidence that burning of PCB-contaminated material associated with the Y-12 burn yards, K-25 burn yards and the TSCA Incinerator at K-25 may have resulted in air releases of PCBs, as well as dioxins and furans. In addition, there is evidence to suggest that materials containing PCBs, such as used oils or electrical equipment, may have been sold and transported off-site.

3.2 RELATIVE SOURCES OF PCBS ENTERING THE CLINCH AND TENNESSEE RIVERS

It has long been recognized that PCBs were used in a large number of facilities throughout the watershed of the Tennessee River and its tributaries (TVA, 1981; 1982; 1983a; 1984; 1987; 1988). The extensive use of PCBs occurred because of the availability of electrical power from the TVA in the 1940s-60s, the growth of industries that used the power, and the concurrent commercial development of PCB products. Since PCBs have been detected in sediment and fish from the Tennessee River above its confluence with the Clinch River and in the Clinch River upstream of the ORR, the project team, collected data on other sources of PCBs entering Watts Bar Reservoir. Available records on PCB use identified more than 22 facilities that managed PCB-containing wastes on portions of the Tennessee River above the Clinch River and on the

Clinch River above the ORR (Table 3-1). It is difficult; however, to discern what fraction of the PCBs in fish in the vicinity of the ORR may have been contributed by these other facilities.

To address this issue, Martin Marietta and Union Carbide retained HydroQual of Mahway, NJ in 1993 to conduct a quantitative evaluation of the relative sources of PCBs in the Watts Bar Reservoir (HydroQual 1995; Appendix A). Two independent approaches were utilized by HydroQual to evaluate the relative fraction of PCBs in Watts Bar fish attributable to Oak Ridge. One approach involved a straightforward analysis of spatial trends in fish monitoring data. The other approach entailed the development of a sediment transport model, which was used in conjunction with PCB sediment core data to predict sources and transport of PCBs at various times and locations within the watershed. A manuscript describing this model was subsequently published (Ziegler and Nisbet, 1995). Based on the results of their analyses, HydroQual concluded that historical releases of PCBs from Oak Ridge were responsible for less than 9 to 13 percent of the currently observed levels in Watts Bar fish. HydroQual also concluded that this estimate could be further reduced if sources of PCBs above Melton Hill Dam were considered. In addition, because of the approximate agreement between these two independent measurements, HydroQual concluded that there was strong evidence that the vast majority of PCBs currently detected in fish in the lower Watts Bar occurred as a result of releases to the Tennessee River upstream of the Clinch River. The analyses also indicated that, with the exception of three periods of elevated discharges, PCB releases to the Clinch River from all ORR sources were relatively constant over time, and the total magnitude of annual PCB releases from the 1940s through the 1990s were on the order of nine kilograms per year (HydroQual, 1995; Appendix A). The agreement of two independent measures, as well as the acceptance of the sediment model methodology in the peer-reviewed literature, provides credibility to HydroQual's conclusions.

3.3 EXPOSURE PATHWAYS FOR OFF-SITE POPULATIONS

Based on the information obtained regarding releases of PCBs to the environment, potential exposure pathways for off-site receptors were likely associated with releases to surface water and to air. Releases to surface water were primarily associated with White Oak Creek, Bear Creek, EFPC, Poplar Creek, the Clinch River, and the Tennessee River (i.e., Watts Bar Reservoir). Due to its proximity to X-10, access to White Oak Creek has typically been restricted. Available anecdotal information indicates that fishing and other recreational activities have been limited in this area (Blalock et al., 1993). Therefore, potential exposures to PCBs in this water body are not considered further. The contribution of PCB levels in the Clinch River from White Oak Creek was included in the assessment of exposure from the consumption of Clinch River fish.

In general, exposure pathways associated with releases to surface water have included fish consumption, dermal contact with surface water and sediments, and incidental ingestion of surface water and sediments. In addition, based on the available information regarding historical activities in the area, direct contact with floodplain soil as well as pathways associated with bioaccumulation of PCBs in vegetation and animals have also been identified as complete exposure pathways for EFPC. Exposure pathways associated with bioaccumulation of PCBs in animals have also been identified for Jones Island (Figure 1-1) and the Clinch River. Exposures related to PCB releases to air have included both direct pathways, such as inhalation, and indirect

Table 3-1. Other Facilities Identified as Potential Sources of PCBs

- TVA Power Plants
  Melton Hill, Fort Loudoun, Bull Run, and Kingston
- Aluminum Company of America
  Alcoa
- Cementation Company of America
  Mascot
- IT Technology Development Lab
  Knoxville
- Harriman Utility Board
  Harriman
- Lafollette Utilities
  Lafollette
- Pellissippi Center
  Knoxville
- Transfer Station
  Knoxville
- City of Maryville Operations
  IT Analytical SVC
- Kindrick Trucking Company
  Harriman
- Lenoir City Utilities
  Lenoir City
- Sevier County Electric System
  Sevierville
- Transport America, Inc.
  Knoxville
- General Electric
  Knoxville
- IT Environ. Tech. Development Ctr.
  Oak Ridge
- Knoxville Utility Board
  Knoxville
- Loudon Utility Pollution Control Plant
  Loudon
- Southern Alloys and Metal
  Harriman
- TVA Singleton Laboratory
  Maryville

pathways such as bioaccumulation of PCBs in vegetation and animals. These pathways also are likely for dioxins and furans that may have been formed during the incineration of PCBs.

The project team also identified potential exposures associated with the historical sale of PCB-containing materials. Waste oils containing less than 500 ppm of PCBs may have been sold by the ORR facilities in the late 1940s. Such oil could have been used by local individuals for fuel, dust suppression, or vegetation control. Exposure pathways considered include direct contact with contaminated soil and pathways associated with bioaccumulation of PCBs in vegetation.

3.4 SOURCE TERMS AND EXPOSURE ESTIMATES

Based on the available information, the project team determined that developing quantitative estimates of the source terms for PCBs would be difficult, if not impossible, for the following reasons. First, a release of PCBs to the environment can be a complicated event. For example, a spill of PCB fluids from a transformer may result in an immediate release to the environment. However, the PCBs themselves may become mixed with soils near the spill, and movement away from the spill site may happen over an extended time as contaminated soil is eroded. When PCBs enter a storm water or waste water management system, PCBs may be retained in sumps or entrained in sediments for considerable periods of time. Thus, a release that may have happened as a result of a specific event may be translated to an extended, low-level source of contamination to a body of water. Finally, once PCBs enter a body of water, they may remain localized until a storm event or a change in the fundamental hydrology results in remobilization and additional transport. Therefore, a source term for PCBs cannot be constructed in the same way as a source term for an air release, such as radioactive iodine.

A second problem is the widespread use of PCBs and the absence of documentation of releases. From the initial construction of Oak Ridge through the early 1970s, PCBs were viewed as nontoxic, inert substances that offered no particular hazard to workers, the general public, or the environment. As a result, there was no attempt to manage or track the use, release, and disposal of PCBs in any systematic manner. In addition, PCBs were used in literally tens of thousands of separate pieces of electrical equipment and in a variety of non-electrical processes. Finally, PCBs were used and PCB-containing wastes were managed at numerous locations on the ORR.

4.0 DOSE-RESPONSE ASSESSMENT FOR PCBS

Dose-response assessment is the process of characterizing the relationship between the dose of an agent administered or received and the incidence of an adverse health effect in an exposed population (Preuss and Ehrlich, 1987). Dose-response relationships are developed on the basis of animal studies and theoretical precepts about their implications for humans, or based on human epidemiological evidence when adequate data are available. The result of the dose-response assessment is the determination of human uptake levels that provide a certain measure of protection to exposed persons for carcinogenic and noncarcinogenic endpoints. The toxicity value for the assessment of carcinogenic effects is the cancer slope factor (CSF) and the reference dose (RfD) is an estimate of daily exposure that is without appreciable risk of adverse noncarcinogenic effects (EPA, 1991).

4.1 CARCINOGENIC DOSE-RESPONSE ASSESSMENT FOR PCBS

The assessment of carcinogenic effects is a two-step process which consists of (a) assigning a qualitative weight-of-evidence classification based on evidence of carcinogenicity in humans or in test animals and (b) deriving a toxicity value based on human epidemiology studies and/or animal bioassay data. Human epidemiological studies have not established a causal dose-response relationship between exposure to PCBs and adverse carcinogenic effects in humans. Consequently, carcinogenic dose-response values for PCBs have been developed based on animal studies.

In September 1996, the EPA published a reassessment of the carcinogenic potential of PCBs (EPA, 1996a). That reassessment considered all available cancer studies of commercial PCB mixtures including a recent oncogenicity feeding study conducted by Brunner et al. (1996). This feeding study is the most comprehensive study of the carcinogenicity of PCBs to date. Conducted in male and female Sprague-Dawley rats using four PCB mixtures with varying degrees of chlorination (Aroclors 1016, 1242, 1254, 1260), the animals were dosed for seven days a week for 24 months. The cancer slope factors generated from the results of this study are substantially lower than the previous CSF of 7.7 $(mg/kg\text{-}day)^{-1}$.

In consideration of the Brunner et al. (1996) study and all prior carcinogenicity data, EPA's reassessment specified a new set of CSFs for PCBs, including both upper-bound cancer potency estimates and central-tendency estimates. The CSFs depend on the exposure pathway and the degree of chlorination of the PCBs. For PCB mixtures other than those containing minimal amounts of the more highly chlorinated congeners, the central-tendency CSFs range from 0.3 to 1 $(mg/kg\text{-}day)^{-1}$ and the upper-bound CSFs range from 0.4 to 2 $(mg/kg\text{-}day)^{-1}$ (EPA, 1996a). The lower end of these ranges is to be used for vapor inhalation, dermal exposures (if no dermal absorption factor is applied), and water ingestion, and the upper end is to be used for soil or sediment ingestion, dermal exposures (if an absorption factor is applied), dust inhalation, and food chain exposures. Lower CSF -- 0.04 and 0.07 $(mg/kg\text{-}day)^{-1}$ for central-tendency and upper-bound estimates, respectively -- are prescribed for PCB mixtures in which congeners with more than four chlorines comprise less than 0.5 percent of the total PCBs.

The upper-bound CSF of 2 $(mg/kg\text{-}day)^{-1}$ specified for more highly chlorinated PCB mixtures was used to assess the carcinogenic potential of PCBs for all pathways evaluated in this analysis. The low end of the upper-bound CSF range of 0.4 $(mg/kg\text{-}day)^{-1}$ would have been appropriate to use in the Level 1 evaluation (Section 6.0) for ingestion of surface water and inhalation of vapor. However, because the Level 1 evaluation was considered a screening analysis, the project team used the more conservative CSF for all the pathways. In the subsequent evaluations throughout the analysis, the CSF of 2 $(mg/kg\text{-}day)^{-1}$ was appropriate for the pathways that were retained for further study.

4.2 UNCERTAINTY IN THE CANCER SLOPE FACTOR

In the late 1970s, toxicologists in the US became increasingly concerned about exposure to chemical carcinogens. Data from radiation and certain models of carcinogenicity led researchers and regulatory scientists to conclude that no threshold existed for carcinogens (ILSI, 1996) and that the traditional approaches for setting "safe levels" of intake would result in unacceptable levels of risk. Therefore, dose response models were used to estimate virtually safe doses. The nature of these models has varied over time from simple hit models (Anderson et al., 1983), to multistage models (Crump, 1981), to loosely defined models of additivity, to background levels (EPA, 1996b). The evaluation of cancer has relied either on animal studies in highly inbred test species or on epidemiology studies that focused on average dose and frequency of response. Neither approach provides much information on interindividual variation in pharmacokinetics or dynamics.

The process in performing cancer dose response assessments was to estimate the response rate as a function of long-term dose rates in humans. This was referred to as the carcinogenic potency or the dose response slope. Once the response slope was determined from the experimental data, an estimate of the response slope in the general human population was made. In developing these estimates, the agencies took into consideration experimental uncertainty and interspecies differences. The approach used was to estimate the highest dose response slope that was consistent with the available data. Where data were available from multiple species preference was given to studies of a higher quality.

Characterizing the carcinogenic slope for chemicals involves the management of a number of sources of uncertainty. In order to manage these uncertainties, regulatory agencies have established policies that traditionally seek to characterize the upperbound of the estimate of the true carcinogenic potency for compounds that occur at a low dose (Evans et al., 1994; EPA, 1996b). In the case of PCBs, the major sources of uncertainty are low dose extrapolation and interspecies extrapolation.

<u>Low Dose Extrapolation</u>

As discussed above, human epidemiology provides negative or inconclusive evidence for the carcinogenic potency of PCBs. Therefore, EPA has relied upon animal testing using doses that are much higher than the levels of exposure that can be reasonably expected to occur in the population around the ORR. The animal studies established that PCBs cause liver cancer at doses where other toxicological effects including liver effects occur. The relevance of these effects to lower doses, which do not cause other toxicological effects, is not clear. When PCBs induce carcinogenicity through a process that involves one or more thresholds, then the dose response seen in the studies may not be a reasonable prediction of effects that occur at lower doses in the exposed population. Current EPA guidance states that the use of linear models between the ED_{10} (the dose that causes 10% of animals tested to develop cancer) and a zero dose can be regarded as a conservative upperbound on the plausible range of low-dose uncertainties. This slope is assumed applicable in humans when it is adjusted for the differences between body weights. The true dose-response curve for PCBs may be much lower than the slope calculated using this method (EPA, 1996b). As a result the actual risks are likely to be lower then the risks predicted using this method and may be zero.

Interspecies Extrapolation

Because PCB carcinogenic potency is established in rats, it is not clear that humans will respond in the same fashion. Studies of interspecies differences in carcinogenic effects of chemicals suggest that there is a wide range of responses across species to a given chemical. The agency has concluded that it is prudent to assume that test animals are likely to be less sensitive than humans and that an upper bound to this decreased sensitivity can be estimated by using the ratio of the individual's body weight to the 3/4 power (EPA, 1992).

For both sources of uncertainty, the current carcinogenic potency likely overestimates the true potential risks to populations. Therefore, carcinogenic potency estimates in this analysis should be regarded as a screening value. Unlike other investigations of the Oak Ridge Dose Reconstruction, we have not attempted to quantify this uncertainty in the carcinogenic potency of PCBs. Had this been done, we would expect the upper confidence limits on the uncertainty in a predicted risk to be similar to the analysis conducted using the carcinogenic slope factors.

4.3 NONCARCINOGENIC DOSE-RESPONSE ASSESSMENT FOR PCBS

Human epidemiological studies for PCBs have suggested that a number of adverse effects may be associated with PCBs. However, because of uncertainty in dose rates, potential confounding factors, and inconsistencies in studies, a clear relationship between specific dose and adverse effects has not been established in humans. Therefore, animal studies have been used to evaluate the potential noncarcinogenic health effects of PCBs. Overall, the findings from these studies indicate that the noncarcinogenic toxicity of PCBs is dependent on the degree of chlorination of the PCB mixture.

Table 4-1 presents a summary of noncarcinogenic dose response data for the most sensitive endpoints for various PCB (Aroclor) mixtures. The table indicates NOAELS and LOAELS for PCB mixtures as low as 5 µg/kg-day. The most sensitive endpoints for PCB noncancer effects are as follows: immunological, developmental, reproductive, dermal/ocular, hepatic, and neurological. In addition, the studies indicate that primates tend to be more sensitive than rodents. The effects associated with the reported LOAELs range in terms of severity of effect, from changes in immunological parameters or subtle behavioral effects to more clinically significant effects such as alopecia and reproductive failure. It is worth noting that the most sensitive effects, that is, those effects associated with the lowest LOAELs and NOAELs, are

Table 4-1. Summary of Noncarcinogenic Dose Response Data for Polychlorinated Biphenyls (PCBs)

Target Organ	Route/ Aroclor	Effect	Species	NOAEL (mg/kg/day)	LOAEL	Exposure Duration (months)	Source	Comments
Dermal Hemotological Hepatic	(1254)	nail loss; facial edemia; anemia; hepatocyte necrosis	monkey	-	0.2	12	Tryphonas et al., 1989a; Arnold et al., 1993a,b	Basis for Aroclor 1254 RfD
Dermal Gastrointestinal Hematological Hepatic	oral (1254)	severe normocytic anemia; hypertrophic gastropathy; necrosis nail loss; gingival necrosis	monkey	-	0.2	28	Tryphonas et al., 1989b; Arnold et al., 1993a,b	Basis for Aroclor 1254 RfD
Dermal/Ocular Reproductive	oral (1248)	alopecia; acne; periorbital edema; decreased spematogenesis and libido	monkey	-	0.1	12	Allen and Norback 1976	
Dermal	oral (1254)	alopecia; facial edema	rat	1.25	2.5	24	NCI, 1978	34% decreased survival at the LOAEL reported
Immunological	oral (1254)	decreased IgG and IgM levels in response to sheep red blood cells (SRBC)	monkey	-	0.005	12	Tryphonas, et al., 1989, 1991	Basis for ATSDR MRL of 20 ng/kg/day
Immunological	oral (1254)	decreased natural killer cells; decreased by thymus weight	rat	1	10	1	Smialowicz et al., 1989	
Developmental	oral (1016)	18% reduction in birth weight	monkey	0.007	-	12	Barsotti and Van Miller, 1984	Basis for Aroclor 1016 RfD
Developmental	oral (1016)	decreased performance in spatial discrimination problems	monkey	0.008	0.03	16	Schantz et al., 1989	
Developmental	oral (1254)	increased relative liver weight	rat	0.13	1.3	6	Overman et al., 1987	50% neonatal death reported for 13.5 mg/kg/day dose group

Table 4-1. Summary of Noncarcinogenic Dose Response Data for Polychlorinated Biphenyls (PCBs)

Target Organ	Route/ Aroclor	Effect	Species	NOAEL (mg/kg/day)	LOAEL	Exposure Duration (months)	Source	Comments
Reproductive	oral (1254)	increased estrus, decreased receptivity; decreased litter size	rat	-	30	1	Brezner et al., 1984	
Reproductive	oral (1254)	no conception, post-implant bleeding and abortion	monkey	-	0.2	9	Arnold et al., 1990	
Neurological	oral (1016, 1260)	decreased levels of dopamine in specific areas of the brain	monkey	-	0.8	5	Seegal et al., 1990, 1991	
Neurological	oral (1254)	decreased dopamine levels in the brain	rat	-	500	single dose	Seegal et al., 1986	

those that can be considered less serious, such as the behavioral effects (decreased performance in spatial discrimination problems in offspring of maternally exposed animals (Schantz et al., 1989)) and immunological changes (decreased IgG and IgM levels in response to SRBC (Tryphonas et al., 1989, 1991b)). In fact, there is ongoing scientific debate as to whether some of these effects, including those which provide the basis for the noncarcinogenic toxicity criteria, can be truly considered adverse or whether they would be more appropriately viewed as biomarkers of exposure.

Currently, there are three noncarcinogenic toxicity criteria available for PCB mixtures. The EPA has published oral RfDs for Aroclor 1016 and Aroclor 1254 of 7×10^{-5} mg/kg-day (70 ng/kg-day) and 2×10^{-5} mg/kg-day (20 ng/kg-day), respectively (IRIS, 1998a,b). ATSDR has also developed a chronic oral Minimal Risk Level (MRL) for PCBs of 2×10^{-5} mg/kg-day (ATSDR, 1996). The basis of the RfD for Aroclor 1016 is the study by Barsotti and Van Miller (1984). This study investigated reproductive and developmental effects in rhesus monkeys and reported decreased birth weights in the offspring of animals receiving approximately 0.04 mg Aroclor 1016/kg body weight-day. EPA (1988b) used the no-observable-adverse-effect level (NOAEL) of 0.007 mg/kg-day from this study to derive the RfD. The EPA calculated the RfD of 7×10^{-5} mg/kg-day by dividing the NOAEL of 0.007 mg/kg-day from the Barsotti and Van Miller (1984) study by a safety factor of 100. The basis of the 100-fold safety factor is as follows: a 3-fold factor to account for sensitive individuals; a factor of 3 for extrapolation from animals to humans; a 3-fold factor for limitations in the data base; and a 3-fold factor to account for extrapolation from subchronic exposure to a chronic RfD (IRIS, 1998a). The product of the four safety factors were calculated and rounded up to a factor of 100.

An external technical review workshop was conducted by the EPA to determine the adequacy of the evidence which provides the basis for the Aroclor 1016 RfD (EPA, 1994c). Based on an assessment of four main issues related to the RfD: 1) selection of the critical study; 2) selection of critical effects; 3) selection of safety factors; and 4) weight of evidence conclusions, the technical review panel made specific recommendations to the EPA. The more substantive of these included a recommendation to reevaluate the available data regarding characteristics of the parental monkeys in the Barsotti and Van Miller (1984) study, so that the critical effect (reduced birth weight) could be adequately assessed (EPA, 1994c). The panel also recommended that the NOAEL for the study be revised using the analytical data for the monkey chow, rather than calculating the NOAEL based on the amount of Aroclor 1016 added to the diet. This would lower the NOAEL from 0.25 ppm to 0.17 ppm (EPA, 1994c). To date, the EPA has not revised the RfD for Aroclor 1016 in response to the recommendations of the technical review panel.

Other issues can be raised regarding this RfD. The application of a safety factor for subchronic to chronic exposure duration in the derivation of the Aroclor 1016 RfD does not appear to be necessary. In general, use of a safety factor for exposure duration is only necessary when there is evidence that chronic exposure would increase the severity of the effect or decrease the dose necessary to elicit the effect. This is not the case for reproductive effects where an increase in the maternal exposure duration does not necessarily result in an increased risk of reproductive effects. The critical exposure period for causing reproductive and developmental effects is the period immediately prior to conception to birth (Johnson, 1989; Paustenbach, 1989). Maternal exposures that occur outside this critical period are not thought to increase the risk of deleterious effects to the offspring. Because the experimental animals were dosed for the entire critical period, it is not necessary that they be dosed for their entire or even the majority of their lifetime. In addition, the adult monkeys had reached steady state conditions and an increased exposure duration would not have increased the body burden of PCBs in the offspring. For the purpose of characterizing and comparing risks to the fetus from maternal exposure in the Barsotti and Van Miller (1984) study to a biologically plausible human exposure scenario, the exposure duration of about 1 year in monkeys prior to birth appears to be more than adequate for extrapolating to humans. Therefore, the use of the 3-fold safety factor for extrapolating from a subchronic exposure to a chronic RfD appears to be inappropriate.

The RfD for Aroclor 1254 is based on a study of rhesus monkeys conducted by Tryphonas et al. (1989, 1991a,b) and Arnold et al. (1993a,b). In this study, general systemic, reproductive, and immunological effects were investigated during the pre-breeding phase of a study of rhesus monkeys exposed to 0, 0.005, 0.02, 0.04, or 0.08 mg/kg-day Aroclor 1254 in the diet for five years (Tryphonas et al., 1989; 1991a,b; Arnold et al., 1993a,b). Following 37 months of exposure, general health and clinical pathology results were reported by Arnold et al. (1993a,b); results of immunologic evaluations after 23 and 55 months of exposure were reported by Tryphonas et al. (1989; 1991a,b).

Dose-related dermal and ocular effects, including ocular exudate, inflamed Meibomian glands, and changes in finger and toe nails were reported at 0.005, 0.02, 0.04, and 0.08 mg/kg-day (Arnold et al., 1993a,b). In the immunologic portion of the study, Tryphonas et al. (1989) reported a statistically significant ($P<0.05$) dose-related decrease in antibody titers of the IgM and IgG isotypes to sheep red blood cells (SRBCs) for all dose groups after 23 months of exposure, and a significant dose-related decrease in IgM after 55 months of exposure (Tryphonas et al., 1991a).

For the purposes of developing an RfD, EPA (IRIS, 1998b) identified a LOAEL of 0.005 mg/kg-day based on clinical effects and immunologic alterations, and applied a safety factor of 300 to derive the RfD of 2×10^{-5} mg/kg-day (20 ng/kg-day). The safety factor included factors of 3 each for interspecies extrapolation, use of a LOAEL, and subchronic exposure. In addition, a factor of 10 was applied for interindividual variability, and the total safety factor of 270 was rounded to 300.

A number of questions can be raised in EPA's selection and evaluation of the studies from which the Aroclor 1254 RfD was derived. Despite the changes in immunologic parameters reported by the researchers, clinical relevance of these changes has not been demonstrated. In addition, the rhesus monkey is not an appropriate model for dermal, ocular and nail effects of PCBs in humans. A comparison of effects and the body burdens (blood serum levels) seen in workers exposed to PCBs and the effects and associated body burdens reported in rhesus monkey studies indicates that PCBs produce nail changes, ocular effects, and dermal effects at much lower doses in rhesus monkeys than in humans (Table 4-2). These findings suggest that rhesus monkeys are significantly more sensitive to the effects than humans exposed to PCBs.

There is also limited evidence to indicate that PCBs are metabolized differently in humans than in rhesus monkeys and that the metabolism of PCBs may be critical to the overall expression of

Table 4-2. PCB Concentration in Blood

			Plasma (ppb)[a]			
Study	**Description**	**PCB Exposure**	**Low[b] Homolog**	**High[b] Homolog**	**Total PCBs**	**Reported Oculodermal Effects**
Tryphonas et al., 1989 a, b;	Rhesus monkey	5-80 µ/kg-day	NR	NR		Ocular exudate, inflamed Meibomian glands,
Arnold et al., 1993 a,b	study - Aroclor 1254	controls	NR	NR	1.3[c]	changes in finger and toe nails in all PCB dose groups
Wolff et al., 1982;	Epidemiology Study	High				
Fischbien et al., 1985	Capacitator Workers	male	161	25		No significant correlation between oculodermatological
		female	89	18		findings and plasma/serum PCB concentrations
		Medium				
		male	45	19		
		female	39	11		
		Low				
		male	57	28		
		female	78	25		
		Retired				
		male	54	95		
		female	49	27		
Taylor et al., 1988	Epidemiology Study	Direct exposure	269	33	302	NR
	Capacitor Workers	Indirect exposure	50	11	61	
		Reference group	7	9	16	
Smith et al., 1982	Epidemiology Study	F-30 are exposed	502	44		
	Capacitor Workers	G-44 area exposed	237	51		
Emmett et al., 1988a,b	Epidemiology Study	Exposed			9.7	No effects associated with PCB exposure
	Capacitor Workers	Nonexposed			4.6	
Ouw et al., 1979	Epidemiology Study	Exposed Aroclor 1242	394			1 case chloracne (unconfirmed)
	Capacitor Workers					
Lawton et al., 1985	Epidemiology Study	Exposed to Aroclors 1016,	363	30		No chloracne reported
	Capacitor Workers	1242, 1254				

a. Concentrations are geometric mean values, unless otherwise noted.
b. Low homologs include tri-, tetra-, and half of the penta-congeners; high homologs include half of the penta-, hexa-, hepta-, and octa-congeners.
c. Calculated from reported whole blood concentrations, assuming plasma PCB concentration is equivalent to 1.3 the concentration in whole blood.

PCB toxicity (Gillis and Price, 1996). Evidence of this difference can be seen in the patterns of PCB congeners that accumulate in adipose and hepatic tissues of rhesus monkeys chronically exposed to Aroclor 1254 and how they differ from patterns of congener retention in humans exposed to PCBs. Humans produce a retention pattern similar to that observed in *in vitro* studies of P450B2 enzyme activity, referred to as P450B2-like metabolism (Brown et al., 1989; Brown 1994). Humans produce a second pattern when exposed to mixtures of PCBs and furans (Masuda et al., 1978; Kunita et al., 1984). This pattern results from metabolism of PCBs by a combination of P450B2-like and P450A1-mediated metabolic pathways. It appears that, in the absence of concurrent exposures to dioxins and furans, PCBs do not induce the P450A1 enzymes in humans. Furthermore, studies of PCB induction of P450A1 in rodents suggest that such induction, if it occurs in humans, would require exposures of PCBs far higher than have occurred from environmental or historical occupational exposures (Brown et al., 1991).

In contrast to the metabolism of PCBs in humans (in the absence of concurrent exposures to dioxins and furans), a different pattern is observed in rhesus monkeys. Metabolism patterns of PCBs in rhesus monkeys indicate that PCBs are metabolized by means of the P450A1 pathway and a second pathway known as the P450RH pathway, which appears to be unique to the rhesus monkey (Brown, 1994). The specific enzymes responsible for metabolizing PCBs in the unusual P450RH pattern observed in monkeys are unclear at this time. However, the differences in enzyme systems supports the finding that PCBs are metabolized differently in rhesus monkeys and humans, and suggest that the rhesus monkey is a poor model for endpoints associated with the activation of P450A1.

Several studies have demonstrated that PCB metabolism is critical to the expression of PCB toxicity in humans (Brown, 1994; Brown et al., 1989; 1991; 1994). For example, induction of P450A1 at low PCB doses is associated with dermal, ocular, and nail effects in animals (Brown et al., 1994). In humans, Yusho victims, who were exposed to both PCBs and furans and experienced many of these effects, also displayed P450A1 metabolism. Conversely, metabolism of PCBs under the P450B2-like pathway in occupationally exposed human populations is not associated with these effects.

Because the evidence indicates that the rhesus monkey is significantly more sensitive than humans, EPA's safety factor of 3 to account for interspecies sensitivity appears to be inappropriate. A safety factor of 1 or less is more appropriate to address interspecies uncertainty. In deriving the RfD, EPA applied a safety factor of 3 to adjust for study duration. In the case of Aroclor 1254, the monkeys were dosed for greater than 25 percent of their lifetime and steady state PCB body burdens had been achieved (Arnold et al., 1993a,b). This suggests that a longer exposure duration would not result in an increased toxic response. This issue is investigated further in Section 8.0.

In its toxicological profile for PCBs, the ATSDR (1996) also used the Tryphonas et al., (1989, 1991a,b) immunotoxicity studies to derive an MRL of 20 ng/kg-day. In calculating the MRL, the LOAEL of 0.005 mg/kg-day was divided by a safety factor of 300 (10 for extrapolation from a NOAEL to a LOAEL, 3 for extrapolation from animals to humans, and 10 for human variability).

4.4 INTERPRETATION OF NONCARCINOGENIC RISK ASSESSMENT RESULTS

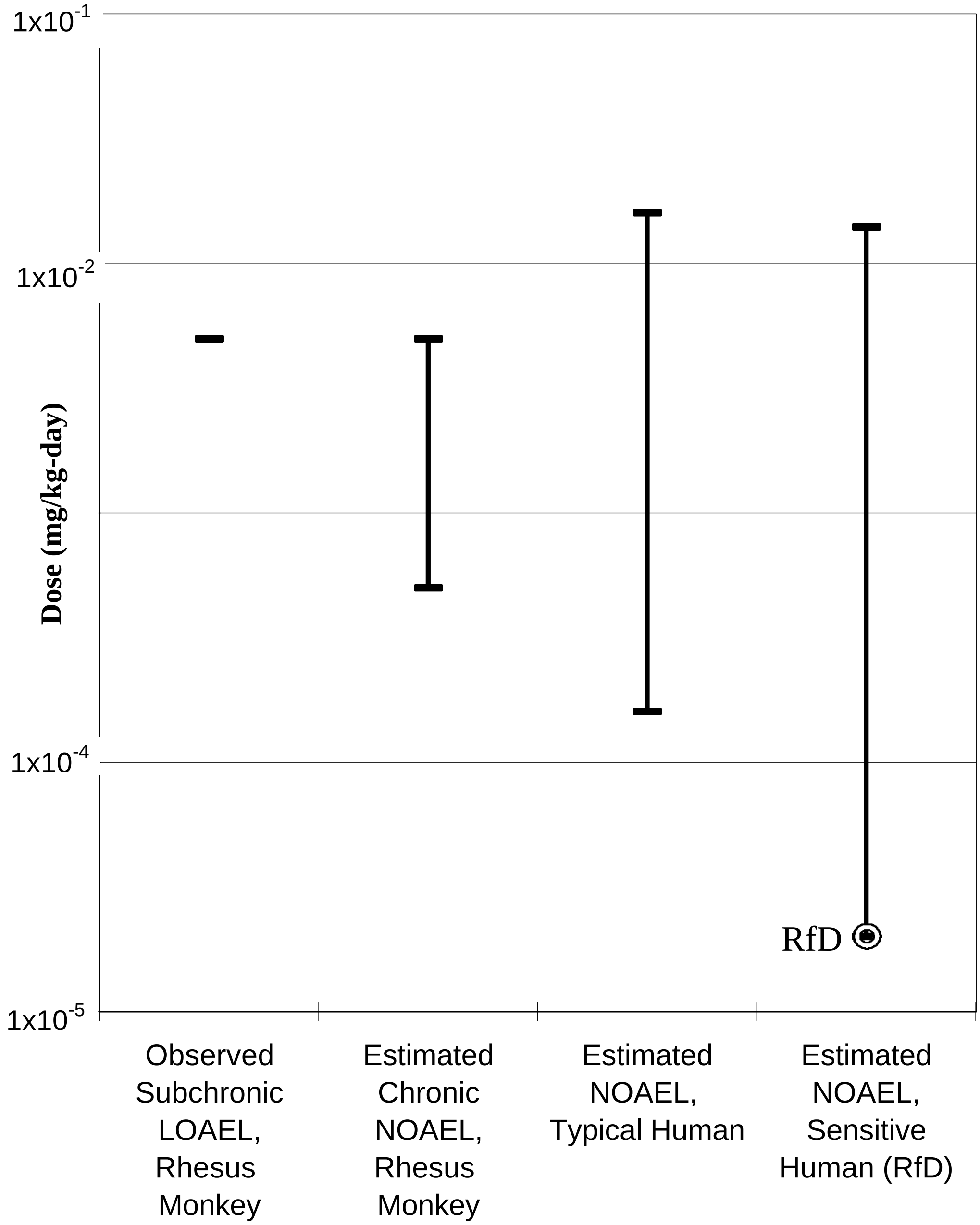
The RfDs and MRL established for PCBs are benchmark levels used as guidance for "safe" or "acceptable" levels of exposure. Doses below these benchmark levels are not expected to result in appreciable risk even in sensitive individuals. Unfortunately, the methodology that established the benchmarks was not designed to evaluate the magnitude of risks that would occur at levels of exposure above the RfD or MRL. However, some insight on the potential for adverse effects can be derived from the definition of the RfD and MRL, and the system of safety factors used in setting RfDs and MRLs.

The RfD and MRL are intended to be doses that will not cause an adverse effect in any individual in both the general population and in sensitive individuals (EPA, 1986; Barnes and Dourson, 1988). It has long been recognized that individuals respond to chemicals at different doses, with certain individuals responding at low doses (sensitive individuals) and others at much higher doses (resistant individuals). The RfD and MRL are intended to be a dose lower than that which causes an adverse effect in a sensitive individual. Therefore, doses slightly in excess of the RfD or MRL can be viewed as presenting some risk to sensitive individuals, but not to the general population. As doses increase significantly beyond the RfD, it is reasonable to expect that some members of the general population would begin to be affected. However, doses in excess of the RfD and MRL do not imply that all or even the majority of individuals will be affected. Rather, doses in excess of the standard should be viewed as resulting in some level of risk of an adverse effect in a small percentage of individuals. As the doses increase higher and higher above the MRL and RfD, the percentage of the population affected and the severity of the effects are expected to increase.

RfDs and MRLs are derived from the results of toxicological studies and application of one or more safety factors (Barnes and Dourson, 1988). These safety factors are intended to account for the extrapolation from the estimates of "safe" or "low risk" doses in toxicology studies to levels protective of adverse effects in human populations. In the case of PCBs, large safety factors (300 fold) are used in setting the RfDs and MRLs. The methodology used in setting the safety factors is designed to account for worst case chemicals. Worst case chemicals are those which: 1) are uniquely toxic to humans e.g., more toxic to humans than the test species, and 2) significantly more toxic to certain human subpopulations. Based upon available data (Weil and McCollister, 1963, Dourson and Starra, 1983, Lewis et al., 1990, Baird et al., 1996), the majority of chemicals do not behave in the worst-case manner assumed by regulatory agency guidelines. As a result most RfDs and MRLs represent doses that are substantially smaller then the actual dose that is protective to sensitive individuals (Baird et al., 1996; Swartout et al., 1998).

The potential for the RfD and MRL to predict low values increases with the magnitude of the safety factors used in setting the standard (Baird et al., 1996). Because the safety factors for the MRL and RfDs for PCBs are relatively large (300 fold), it is likely that the actual dose that would result in adverse effects in the exposed populations would be much higher than the MRL or RfDs. Some perspective on the magnitude of the potential for underestimating the dose that is protective of sensitive individuals is given in Figure 4-1.

Figure 4-1: Results of Tryphonas et al. (1989) and the Cumulative Uncertainty in the RfD for Aroclor 1254



This figure presents the conservative nature of each step in the development of the RfD for Aroclor 1254. The first column in the figure presents the LOAEL of 5 x 10^{-3} mg/kg-day from the animal studies used to set the RfD (Tryphonas et al., 1989, 1991a,b; Arnold et al., 1993a,b). EPA applied two safety factors to convert the subchronic LOAEL to a chronic NOAEL of 5 x 10^{-4} mg/kg-day. However, it is possible that had a lower dose group been used and had the study continued for a longer period of time, a chronic NOAEL greater than 5 x 10^{-4} mg/kg-day may have resulted. In fact, because the endpoints evaluated are not expected to increase in severity with duration and because several of the effects are of little toxicological significance, it is possible that the chronic NOAEL would be similar to the subchronic LOAEL. The second column in Figure 4-1 represents the uncertainty by indicating that the plausible estimates of the chronic NOAEL could range from 5 x 10^{-3} mg/kg-day to 5 x 10^{-4} mg/kg-day.

As discussed previously, the EPA applied a third safety factor to convert the animal chronic NOAEL to a chronic NOAEL in typical humans of 2 x 10^{-4} mg/kg-day. This conversion is based on the assumption that humans are more sensitive than monkeys. However, because the monkeys may be more sensitive than humans, the chronic NOAEL for typical humans may be higher than for the test animal. The third column in Figure 4-1 represents this possibility by indicating that the true value of the chronic NOAEL for typical humans may range from a value three fold higher to a value three fold lower than the estimate of the animal NOAEL i.e., 2 x 10^{-2} mg/kg-day to 2 x 10^{-4} mg/kg-day.

The EPA applied a final safety factor to convert the chronic NOAEL for typical humans to a chronic NOAEL for humans uniquely sensitive to the effects of PCBs (the sensitive human chronic NOAEL or RfD). This sensitivity, for example, may be due to age-related differences, genetic subpopulations, concurrent disease states, or pregnancy. However, it is possible that sensitive individuals exposed to PCBs do not respond differently than typical individuals. Consequently, the chronic NOAEL for sensitive individuals may be quite similar to the chronic NOAEL for typical individuals. The fourth column in Figure 4-1 represents this possibility by indicating that the chronic NOAEL for sensitive humans (the RfD) could range from 2 x 10^{-2} mg/kg-day to 2 x 10^{-5} mg/kg-day. In other words, the chronic NOAEL for sensitive humans could be three orders of magnitude higher than the RfD established by the EPA.

In summary, the characterization of noncarcinogenic risks using the RfD methodology is associated with considerable uncertainty. The project team estimated this uncertainty in Section 8.0.

5.0 CHARACTERIZATION OF FISH CONSUMPTION RATES

Fish consumption is an important exposure pathway when considering potential PCB exposure, due to the tendency of the more highly chlorinated PCBs to bioaccumulate in the tissues of aquatic biota. As demonstrated in numerous studies many of the fish sampled from Watts Bar, Clinch River/Poplar Creek, and EFPC have detectable levels of PCBs in their tissues. As a result, since the 1950s individuals who caught and consumed fish from any of these water bodies received some exposure to PCBs. The level of exposure received is directly related to three factors: the species of fish consumed, the concentration of PCBs in those fish, and the frequency with which those fish were eaten. Concentrations of PCBs in fish tissues by species have been identified through historical sampling and are discussed in Sections 7.0 and 8.0. This section evaluates the information available on historical fishing activities on the water bodies of interest, identifies potentially exposed populations of anglers, and derives estimates of fish consumption rates for the populations that were likely exposed as a result of their fishing activities.

There are eight distinct populations that may have received exposure to PCBs through consumption of fish from water bodies in proximity to the ORR. They include: commercial anglers who fished Watts Bar or Clinch River/Poplar Creek; recreational anglers who fished Watts Bar, Clinch River/Poplar Creek, or EFPC; and subsistence individuals who may have fished any of these water bodies. The following sections present a description of the three types of anglers, their usage of the water bodies of interest, and the species targeted by them.

In addition, this section discusses the data and methodologies used to develop both the point estimates of consumption used in the level I evaluation conducted in Section 6.0 and the distributions used in the level II and III evaluations discussed in Sections 7.0 and 8.0, respectively. Generally, point estimates were based on conservative estimates of mean consumption rates reported for surveyed populations of fish consumers that were considered most relevant to the populations identified.

Studies of fish consumption indicate that, in general, fish consumption distributions are positively skewed (Puffer et al., 1981; Landolt et al., 1985; Ebert et al., 1993; 1994; SCCWRP and MBC, 1994) in that most consumption rates lie closer to the minimum than to the maximum value. Thus, these distributions are best represented by lognormal distribution models. To develop distributions of values to be used in the level II and III evaluations, lognormal distribution models were selected to represent the shape of each distribution of fish consumption rates identified for each relevant angler population and water body.

A lognormal distribution is defined by the arithmetic mean and standard deviation of the distribution.[1] Thus, for each distribution, the arithmetic mean consumption rate for each population of interest was used as the point of central tendency of the distribution. Because, in most cases, a standard deviation could not be derived using the available data, an estimated standard deviation needed to be derived. Although standard deviations are reported for other data sets, to use those standard deviations with arithmetic means that are substantially different

[1] The lognormal distribution is often defined by the geometric mean and geometric standard deviation. However, a lognormal distribution can also be defined by an arithmetic mean and standard deviation.

would alter the shape of the distribution. Instead, it is important to preserve the relationship between the mean and standard deviation for different data sets. Thus, for this analysis, standard deviations were estimated based on the observed relationships between the means and standard deviations reported for other fish consumption studies. The fish consumption study conducted by Ebert et al. (1993) reported a mean consumption rate of 6.4 g/person-day with a standard deviation of 16 g/person-day, resulting in a coefficient of variance of 2.5. A similar coefficient of variance was found in the results of a study of freshwater anglers in New York (Connelly et al., 1996). The mean (6.36 g/person-day) and standard deviation (14.32 g/person-day) reported from that study resulted in a coefficient of variance of 2.25.[2] While most other fish consumption studies do not provide standard deviations, and thus can not be used to calculate coefficients of variance, the differences between the reported median and arithmetic mean values in those studies, and the magnitudes of their 95th percentiles compared to the arithmetic means indicate that these distributions are also positively skewed (Javitz 1980; Rupp et al., 1980; Pierce et al., 1981; Puffer et al., 1981; Cox et al., 1987; Fiore et al., 1989; CRITFC, 1994; Price et al., 1994; Stern et al., 1996). Thus, this approach appears to be reasonable and appropriate.

Averaging the coefficients of variance from the Ebert et al. (1993) and Connelly et al. (1996) studies yields a value of 2.38. This average value was multiplied by the mean consumption rate estimated for each distribution to derive an estimated arithmetic standard deviation for that distribution. (While this is not a true standard deviation because the data set is not normally distributed, it does provide a reasonable upper bound on the mean.) This value was used along with the arithmetic mean to define the shape of each fish consumption distribution used.

5.1 COMMERCIAL ANGLERS

Historical information indicates that commercial fishing harvest in the Tennessee River Valley has increased steadily since the 1940s (Eschmeyer and Tarzwell, 1941; TVA, 1944, 1945, 1947, 1959, 1960, 1961, 1962, 1963, 1967; Morgan and Hubert, 1974; and Todd, 1990). Reports on commercial fishing activities in the 1970s and 1980s indicate that there historically have been two types of individuals who held commercial fishing licenses (Hargis, 1968; Morgan and Hubert, 1974; Hubert et al., 1975; Todd, 1990): full-time anglers who fished as a primary source of income, and part-time anglers who fished for supplemental income or who wanted the opportunity to use gill nets or other commercial fishing gear as part of their recreational fishing. For this analysis of commercial anglers, only data concerning full-time commercial anglers were considered.

5.1.1 Watts Bar Reservoir

Hubert et al. (1975) reported on commercial fishing activity in Upper East Tennessee during 1973. Although the report did not provide specific data for commercial activity at Watts Bar, it did indicate that some of the anglers interviewed for the survey fished Watts Bar. Overall, Hubert et al. reported that, of a total of 206,975 lbs. (94,079 kg) of fish commercially harvested

[2] To the casual reader, a finding of a standard deviation greater than the mean may appear to be unusual and may, at first glance, suggest that the distribution includes negative fish consumption rates. However, in the case of lognormal distributions, the dispersion in the distribution that determines the size of the standard deviation is asymmetrical; that is, the upper end of the distribution is much further from the mean than the lower end. All of the fish intakes in the distribution are greater than zero but the upper ends of the distribution are sufficiently large to produce the large standard deviation.

by 29 commercial anglers in Upper East Tennessee that year, 201,111 lbs. were sold to dealers or individuals, leaving 5,864 lbs. (2,665 kg) potentially available for personal use. If these fish were evenly distributed over 29 anglers, had edible portions of 30 percent (EPA, 1989), and were consumed by 3.2 individuals (the average household size in Roane County in 1970), the resulting average consumption rate can be estimated to be 24 g/person-day. This equates to roughly three half-pound fish meals per month or less than one fish meal per week.

A lower consumption rate for this population of commercial anglers was estimated using the data provided by Todd (1990) concerning the fishing activity of Watts Bar commercial anglers in 1989. In that analysis, it was reported that a total of 18,418 pounds of fish were harvested from Watts Bar by four full time commercial anglers. Todd reported that full-time commercial anglers in Tennessee kept only one percent of their catch for personal use. If this retention rate was also applicable to the Watts Bar anglers, the result was 184 pounds of fish that may have been kept for consumption purposes by commercial anglers and their families. Assuming that these fish were evenly distributed among the commercial anglers, had edible portions of 30 percent (EPA, 1989), and were consumed by 2.5 individuals (the average of household sizes in Roane and Anderson Counties in 1990), the resulting average consumption rate can be estimated to be 7 g/person-day. This equates to approximately one half-pound fish meal per month.

Each of these data sets has its limitations. Data provided by Hubert et al. (1975) provide a conservative but supportable basis for deriving estimated historical consumption rates for full-time commercial anglers using Watts Bar; however, these data likely overestimate actual consumption of fish by commercial anglers and their families as it is assumed, in this analysis, that all fish not commercially sold were retained for personal consumption. It is more likely that only a portion of those fish were retained for personal consumption and the remainder were given away, discarded, used for bait, or fed to pets. Unfortunately, there are no data available to account for these other uses when evaluating this data set. At the same time, the assumption that fish were shared equally among all household members may bias these estimates of consumption rates low because not all household members may have been consumers of these fish and it is known that children are less likely to consume freshwater fish than are adults (Rupp et al., 1980). Thus, adult fish consumers in a given household may in fact have consumed larger amounts of fish than has been estimated here.

While the data reported by Todd (1990) provide an alternative estimate of consumption and are specific to commercial anglers who fished Watts Bar, it is important to note that these data were collected after the fish advisories were posted for Watts Bar in 1989 (Denton, 1992). Thus, at the time these data were collected, consumption of fish by commercial angler families may have been suppressed, due to the presence of the advisory, and may not be representative of the fish consumption behaviors of commercial anglers prior to the advisories.

In light of these uncertainties, it is reasonable to assume that the mean consumption rate derived from the Hubert et al. data represents a plausible but conservative upper bound estimate of average consumption by these individuals in the years before the advisory was posted. Thus, the mean consumption rate estimated from this study, 24 g/person-day, was the consumption estimate used for the level I evaluation discussed in Section 6.0.

In addition, a distribution of consumption rates based on the Hubert et al. data was derived for use in the level II evaluation discussed in Section 7.0. As discussed previously, it was assumed that the consumption rates of this population were lognormally distributed. To define the distribution, the arithmetic mean of 24 g/person-day from the Hubert et al. data was used. An estimated standard deviation of 57 g/day was derived using the coefficient of variance of 2.38 discussed previously.

It is important to put annualized daily fish consumption estimates into perspective. As shown in Table 5-1, a mean consumption rate of 24 g/day equates to approximately one meal per week while an upper bound fish consumption rate of 80 g/person-day equates to approximately two meals per week. To supply fish to support a rate of 24 g/day for one person, an angler would need to harvest 64 pounds of whole fish per year. For two persons to consume at this rate, a total of 128 lbs. of fish per year would need to be harvested while a harvest rate of 193 lbs/year would be necessary to support this level of consumption by three household members.

Species

The species targeted by commercial anglers have been primarily driven by fluctuations in the market values of various fish (TVA, 1959, 1960, 1961, 1962; and Alexander and Peterson, 1982). The principal species targeted by commercial anglers since the 1940s have been catfish, paddlefish (flesh and roe), buffalo fish, carpsucker, carp, and drum (TVA, 1959; Hargis, 1968; Hubert et al., 1975; Alexander and Peterson, 1982; and Todd, 1990). For this analysis, it will be assumed that full time commercial anglers ate a combination of these species, as well as other species in Watts Bar that may not have been commercially marketable.

5.1.2 Clinch River/Poplar Creek

It is unlikely that the Clinch River/Poplar Creek area has been commercially fished to any great degree due to the limited access for larger boats and the proximity of the Watts Bar commercial fishery. If these water bodies have been fished by full-time commercial anglers, it is likely that the mass of fish harvested from them has been substantially smaller than the mass harvested from the larger, more productive, and more highly accessible Watts Bar Reservoir.

Todd (1990) reported that of the 166 full-time commercial anglers statewide, only 33 (20 percent) fished rivers and for those individuals, only about 31 percent of their time were spent fishing rivers. Todd (1990) also reported that for all commercial anglers, 91 percent of catch was from reservoirs and nine percent was from rivers. If this percentage is applied to the estimated mean consumption rate for full-time commercial anglers, 24 g/person-day, the result is an estimated rate of consumption from Clinch River/Poplar Creek of 2.2 g/person-day. This equates to approximately 3.5 half-pound fish meals per year for commercial anglers and their families. This mean is used in the level I evaluation discussed in Section 6.0.

Table 5-1. Comparison of Fish Consumption Rates and Their Meal and Harvest Equivalents

Fish Consumption Rate (g/person-day)	Meal Equivalents (meals/year)*	Meal Equivalents (meals/week)*	Harvest Equivalents (pounds whole fish per year)		
			For 1 Consumer	For 2 Consumers	For 3 Consumers
1	1.6	0.03	2.7	5.4	8.0
10	16	0	27	54	80
20	32	1	54	107	161
24	39	1	64	128	193
30	48	1	80	161	241
40	64	1	107	214	321
50	80	2	134	268	402
60	96	2	161	321	482
70	113	2	187	375	562
80	129	2	214	428	642
90	145	3	241	482	723
100	161	3	268	535	803
110	177	3	294	589	883
120	193	4	321	642	964
130	209	4	348	696	1044
140	225	4	375	749	1124

*Assumes average meal size of 1/2 lb (227 g)

This mean is also used as the basis for the distribution of fish consumption rates for commercial Clinch River/Poplar Creek anglers in the level II evaluation (Section 7.0). As with the consumption rate distribution developed for Watts Bar commercial anglers, an estimated standard deviation of 5.2 g/person-day was derived for use in developing the lognormal distribution of consumption rates for Clinch River/Poplar Creek commercial anglers, using the estimated mean (2.2 g/person-day) and the coefficient of variance (2.38) discussed previously.

Species

Full-time commercial anglers who fished as their primary source of income would have targeted species that were commercially marketable and would have used techniques suitable for catching those species. For this reason, it is very likely that the fish harvested would have been the same species as those harvested from Watts Bar including catfish, paddlefish, buffalo fish, carpsucker, carp, and drum. In addition, these anglers may have also consumed other non-marketable fish species that were harvested incidentally.

5.2 RECREATIONAL ANGLERS

Recreational anglers are those individuals who fish during their recreational time and do not depend on fishing as their main source of income. These anglers exhibit a variety of individual behaviors ranging from those who fish once or twice per year to those who fish with high frequency and routinely eat the fish that they catch. The vast majority of recreational anglers fish occasionally and consume small amounts of fish that are successfully harvested during those trips; however this group will include the few anglers who eat large amounts of fish. Such anglers, by choice, may be supplying a large amount of their dietary protein by fishing. The goal of this analysis is to capture the wide range of consumption behaviors that are exhibited within a given recreational angler population.

A high percentage of those individuals who hold commercial licenses are, in fact, recreational anglers who are willing to pay higher license fees in order to gain the use of commercial fishing gear (Hubert et al., 1975). For this reason, this analysis of recreational anglers includes part-time commercial anglers as well as individuals who hold recreational licenses.

5.2.1 Watts Bar Reservoir

Watts Bar Reservoir has been used by recreational anglers since it was impounded. Eschmeyer and Tarzwell (1941) reported a total of 8,045 angler days for Watts Bar Reservoir. While little information is available on the early years of recreational fishing at Watts Bar Reservoir, available data indicate that the Tennessee Valley reservoirs and their tailwaters have always been productive recreational fisheries. A 1944 report on Guntersville Dam tailwater (TVA, 1944) indicated that within five days of opening the area to fishing, 1,000 anglers had fished there and that one area had received 300 anglers daily.

More recently, Todd (1990) reported that a total of 26,681 lbs. (12,128 kg) of fish were harvested from Watts Bar by the 33 part-time commercial anglers who fished there. Although specific information on percent of harvest retained for consumption was not provided for those anglers, it can be estimated, based on data that were provided by Todd (1990), that they may have retained 11 percent of their harvest for personal use. Assuming that 30 percent of the total fish mass was edible, that all of that fish was kept for consumption purposes, and that each angler shared his or her fish with other family members (2.5 persons per household), an annualized consumption rate of 13 g/person-day can be derived. As with commercial anglers, however, it is important to remember that the Todd data were collected after the implementation of advisories so that consumption activities may have been suppressed in response to those advisories.

To evaluate longer term trends in fishing activities, the Tennessee Wildlife Resources Agency (TWRA, Unk.) reported statistics for the Watts Bar recreational fishery between the years of 1977 and 1991. That report included 15 years of data on a species-specific basis concerning catch rates, mean weights of catch, and the number of fish harvested. Because catch rates (fish/hour) were not reported for the years prior to 1988, it was not possible to calculate and compare consumption rates on a year-by-year basis. However, while the estimated hours per trip, trips per acre, and hours per acre were variable over this time period, there was no discernable trend in the intensity of fishing activity; consequently, there was no indication that the data from a particular year would be preferable to the data for other years. For this reason, it was considered appropriate to average the data over the 15-year period in order to develop a mean consumption rate for recreational anglers using Watts Bar.

Averaging the data over 15 years resulted in the average weights per fish and average number of fish harvested per hour for each species reported (Table 5-2). Using these data and the average trip length of 4.5 hours, estimates of weight of fish per trip were derived. Using the average number of trips to lakes and reservoirs (14.6 trips per year) reported for Tennessee anglers by the U.S. Fish and Wildlife Service (USFWS) (USDOI, 1993) and a 30 percent edibility factor (EPA, 1989), resulted in the estimated edible mass of fish per year for each species. Dividing that by an average household size of 2.7 individuals (average of the mean household sizes in Loudon, Meigs, Rhea, and Roane Counties for 1980 and 1990), resulted in species-specific consumption rates ranging from 0.022 to 7.2 g/day.

If one assumed that an undefined population of anglers harvested all of the species of fish listed in Table 5-2 at the average harvest rates listed, the result is a total edible fish mass harvested of 27 kg/year. Assuming again that the average household size was 2.7 individuals, the annualized daily rate of consumption can be estimated at 28 g/person-day. This is a highly conservative estimate and is not likely to represent fishing success and consumption behavior for any individual angler, due to the fact that anglers typically target certain species of fish during their recreational activities and use fishing gear that is appropriate to the targeted species. As a result, they would not harvest all of the species of fish listed by TWRA or harvest at the average rate for all species, during each fishing trip.

To define a more reasonable estimate, one could assume that a typical angler might regularly harvest the most frequently harvested species (largemouth bass, channel catfish, white crappie, and white bass were consistently harvested in the greatest numbers each year) at the rates reported during the year and derive a consumption rate based on those species alone. Summing the annualized daily consumption rates for largemouth bass (1.4 g / person-day), channel catfish

Table 5-2. Estimation of Rates of Consumption of Recreationally-Caught Fish from Watts Bar Reservoir Between 1977 and 1991[a]

Species	Mean Weight[b] (kg/fish)	Harvest Rate[b] (fish/hr)	Harvest Weight (kg/hr)	Trip Duration (hr/trip)	Trip Weight[c] (kg/trip)	Edible trip wt.[d] (kg edible/trip)	Trip Rate[e] (trips/yr)	Edible Mass per year[f] (edible kg/year)	Household size[g] (persons)	Annual Ingestion Rate[h] (g/person-yr)	Daily Ingestion Rate[i] (g/person-day)
Largemouth bass	0.59	0.12	0.07	4.5	0.32	0.096	14.6	1.4	2.73	510	1.4
Smallmouth bass	0.72	0.16	0.12	4.5	0.52	0.16	14.6	2.3	2.73	840	2.3
Spotted bass	0.55	0.002	0.0011	4.5	0.0049	0.0015	14.6	0.022	2.73	7.9	0.022
Blue catfish	0.86	0.11	0.095	4.5	0.43	0.13	14.6	1.9	2.73	690	1.9
Channel catfish	0.73	0.32	0.23	4.5	1	0.31	14.6	4.6	2.73	1700	4.6
Black crappie	0.27	0.023	0.0063	4.5	0.028	0.0085	14.6	0.12	2.73	45	0.12
White crappie	0.27	0.75	0.2	4.5	0.92	0.28	14.6	4	2.73	1500	4
Bluegill	0.09	1.9	0.17	4.5	0.78	0.23	14.6	3.4	2.73	1200	3.4
Sauger	0.55	0.24	0.13	4.5	0.59	0.18	14.6	2.6	2.73	950	2.6
White bass	0.36	1	0.36	4.5	1.6	0.49	14.6	7.2	2.73	2600	7.2
Total[j]			1.4	4.5	6.3	1.9	14.6	27	2.73	10,000	28

a. Based on species-specific average harvest rates and weights of fish reported over 15 year period

b. Mean of average values reproted between 1977 and 1991

c. Trip weight = harvest weight (kg/hr) x trip duration (hr/trip)

d. Assuming 30% of harvest weight is edible (EPA, 1989)

e. Mean number of trips to lakes or reserviours by Tennessee anglers in 1991 (USDOI, 1993)

f. Edible mass per year = Edible trip weight (kg/trip) x Trip rate (trips/year)

g. Mean of average household sizes in Loudon, Meigs, Rhea, and Roane Counties in 1980 and 1990 (U.S. Census Data)

h. Annual ingestion rate = Edible mass per year/household size

i. Species-specific daily ingestion rate = annual ingestion rate/365 days

j. Total derived by summing across all species

(4.6 g/person-day), white crappie (4.0 g/person-day) and white bass (7.2 g/person-day), results in a total annualized consumption rate of 17 g/person-day.

In order to provide an upper bound estimate to ensure that consumption by the recreational angler population was not being underestimated, the data for 1991, the year for which the highest level of harvest was reported, were evaluated. Using those data, along with the assumptions outlined above, it was assumed that a single angler would not consume all species listed but instead only consume the most harvested species (largemouth bass at 1.5 g/day, channel catfish at 7.2 g/day, blue catfish at 7.2 g/day, and white bass at 13 g/day), the resulting consumption rate was estimated to be 29 g/person-day. This equates to slightly less than one sport-caught fish meal per week.

Based on available data, it appears that 30 g/day is a plausible but conservative estimate of the mean consumption rate to be used in evaluating recreational anglers at Watts Bar. Mean consumption rates of this magnitude have also been reported in other surveys of recreational fish consumption in productive areas of the United States (FIMS and FAA, 1994; SCCWRP and MBC, 1994). Thus, a consumption rate of 30 g/person-day is used in the level I evaluation discussed in Section 6.0.

In addition, this mean is used as the basis for the lognormal distribution of consumption rates for this population in the level II evaluation discussed in Section 7.0. Multiplying the derived coefficient of variance (2.38), based on Ebert et al. (1993) and Connelly et al. (1996) data, by the mean of 30 g/person-day results in an estimated standard deviation of 71 g/person-day. This mean and arithmetic standard deviation were used in defining the distribution of consumption rates for this population of recreational anglers.

As discussed previously in this section, because this distribution is highly skewed, the arithmetic mean of 30 g/person-day overestimates consumption for the vast majority of recreational anglers who use Watts Bar. Typically, arithmetic means of fish consumption distributions are actually representative of approximately the 75th or 80th percentile of the angler population. For example, Ebert et al. (1993) reported that the arithmetic mean consumption rate of 6.4 g/person-day represented the 77th percentile of the distribution (76 percent of the consuming anglers in that study ate fish at lower rates of consumption) and the median (50th percentile) of consumption rates was, in fact, 2 g/person-day. Figure 5-1 and Table 5-3 provide information on the range of consumption rates for Watts Bar recreational anglers.

Species

Eschmeyer and Tarzwell (1941) reported that just after the impoundment of Watts Bar Reservoir, the catch consisted primarily of bass, white bass, bluegill, crappie, and food fish (species not specified). Data from 1977 to 1991 at Watts Bar (TWRA, unk.) indicated that the primary species

Figure 5-1. Percentiles for Fish Consumption Distributions

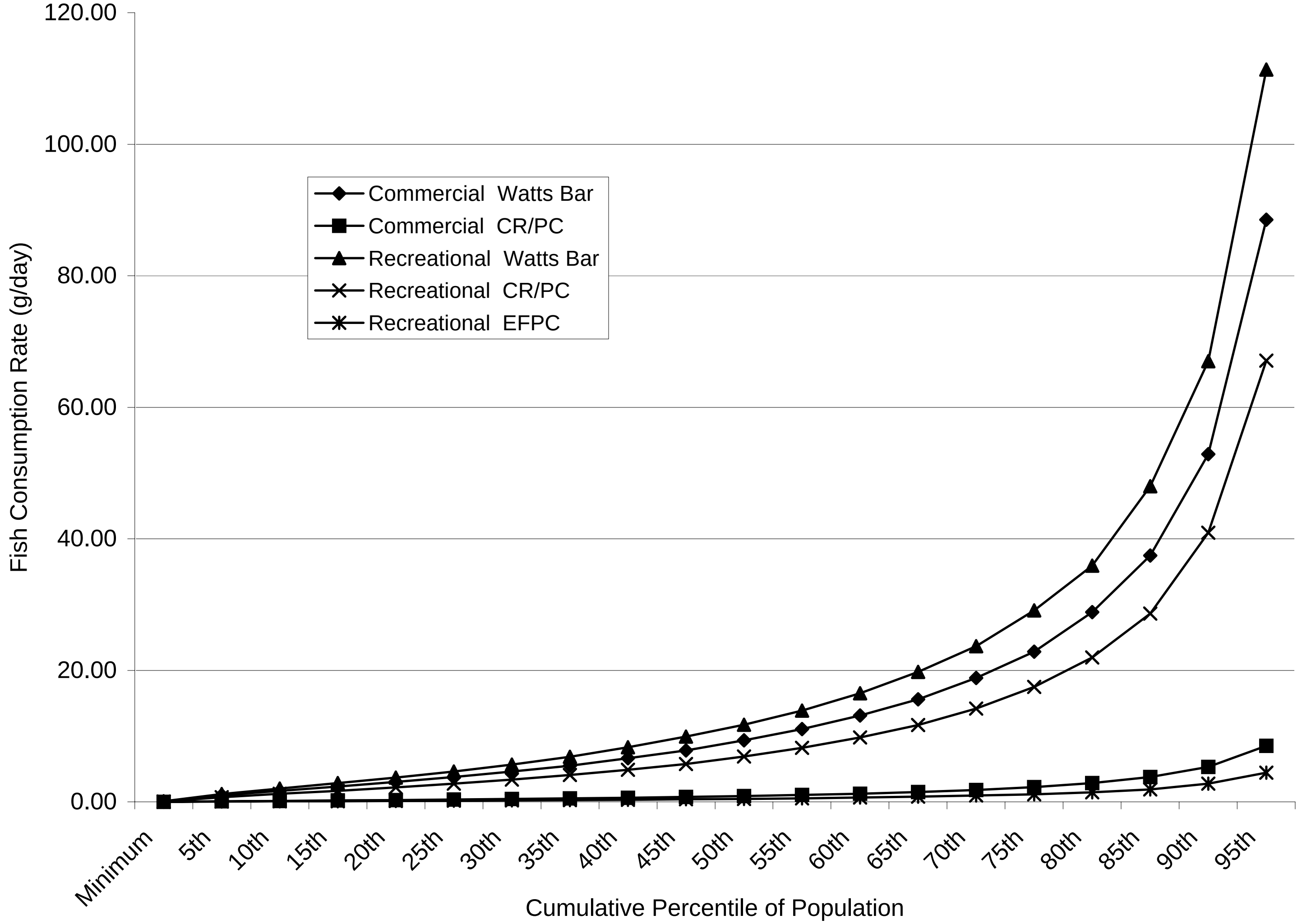


Table 5-3. Percentiles for Fish Consumption Rates (g/person-day) by Population and Water Body

Consumption Percentile	Commercial Watts Bar	Recreational Watts Bar	Commercial CR/PC	Recreational CR/PC	Recreational EFPC
Minimum	0.03	0.08	0.003	0.02	0.003
Arith. Mean	24	30	2.2	18	1.2
Std. Deviation	57	71	5.2	43	2.9
10^{th}	1.7	2.0	0.15	1.2	0.08
20^{th}	3.0	3.7	0.27	2.2	0.14
30^{th}	4.6	5.7	0.42	3.4	0.22
40^{th}	6.6	8.3	0.61	4.9	0.32
50^{th}	9.4	12	0.88	6.9	0.45
60^{th}	13	17	1.2	9.8	0.65
70^{th}	19	24	1.8	14	0.95
80^{th}	29	36	2.8	22	1.4
90^{th}	53	67	5.3	41	2.8
95^{th}	89	110	8.5	67	4.4

harvested during that period were largemouth bass, smallmouth bass, spotted bass, catfish, crappie, bluegill, sauger, and whitefish. As it appears that there may have been a wide variety of fish species available in Watts Bar over the past 50 years, all available fish tissue data from game species, panfish, and food fish have been used to evaluate potential exposures to this population.

5.2.2 Clinch River/Poplar Creek

In the information provided to date, there has only been anecdotal information concerning recreational fishing activities and practices on the Clinch River or Poplar Creek. Because access to the Clinch River/Poplar Creek is considerably less than that afforded by the many public areas of Watts Bar, it can be expected that angler activity on these two water bodies would be less than that on Watts Bar. Statistics from the 1991 USFWS survey (USDOI, 1993) indicate that Tennessee anglers, in general, made an average of 8.9 trips per year to rivers and streams in the state. If the consumption rate recommended for Watts Bar anglers is multiplied by 0.6, the ratio of yearly river trips over yearly lake trips (8.9/14.6), the resulting consumption rate is 18 g/person-day. This consumption rate has been used in the level I evaluation presented in Section 6.0. In addition, it has been used as the mean for the lognormal distribution of consumption rates for Clinch River/Poplar Creek recreational anglers. A standard derivation of 43 g/person-day, based on the product of this mean and the coefficient of variance (2.38) discussed previously, has been used to define the shape of the distribution for the level II evaluation (Section 7.0).

Species

Both of these rivers are of substantial size and could be expected to contain many of the same species contained in Watts Bar. For this reason, the location-specific fish concentration data for the same species indicated for Watts Bar recreational anglers have been used throughout these analyses.

5.2.3 East Fork Poplar Creek

While it is possible that recreational anglers may have spent a portion of their fishing activity at EFPC, the levels of activity and success likely have been low due to the limited access, the nature of the creek itself, and the ready availability of higher quality fisheries nearby. It is possible, however, that some anglers may have used EFPC on an infrequent basis, particularly if those anglers lived near the creek. In its draft *Estimating Exposures to Dioxin-Like Substances*, EPA (1994) recommended using fish ingestion rates ranging from 1.2 to 4.1 for estimating consumption by recreational anglers fishing small ponds or streams. Due to the small size and limited habitat of EFPC, the lower end of the range, 1.2 g/person-day, was used in the level I evaluation (Section 6.0). It also was used as the mean for the lognormal distribution of consumption rates for recreational anglers using EFPC. Multiplying this mean by the coefficient of variance discussed previously (2.38), resulted in an estimated standard deviation of 2.9 g/person-day. These were the values used to generate the lognormal distribution used in the level II evaluation in Section 7.0.

While this mean consumption rate is low and would equate to roughly two fish meals per year, such a consumption rate seems reasonable given the productivity of EFPC. As discussed in

Appendix B, productivity of EFPC is very low with a sustainable yield of only 112 kg of edible sized whole fish per year. If it is assumed that 30 percent of that fish mass is edible, the edible fish mass produced per year is 34 kg. This means that a maximum of 76 individuals could consume fish from EFPC at the average consumption rate of 1.2 g/person-day.

Species

The species that were reported by individuals who historically fished EFPC were primarily crappie, sunfish varieties, and carp. Location-specific data for these species were used to evaluate consumption by this population of fish consumers.

5.3 SUBSISTENCE ANGLERS

The definition and identification of a subsistence population has traditionally been extremely ambiguous with a few notable exceptions: Native American populations that have subsistence and treaty rights to certain fisheries, like the Columbia River (CRITFC, 1994), and Arctic Inuits who, because of tradition and their remote location, rely heavily on native foods obtained from the sea (Kinloch et al., 1992). Beyond these fairly well-defined and well-characterized populations, the definition of a subsistence angler is less clear.

There are two ways in which a subsistence angler population might be defined for the dose reconstruction project. The use of the word “subsistence” implies that an individual or family has a low income and therefore has little choice but to rely on self-caught fish as a major or sole source of dietary protein. Such individuals are estimated to have mean fish intakes as high as 180 g/d (EPA, 1989; McCormack and Cleverly, 1990). While it may be convenient to define the existence of such a population in this manner, there is no easy way of determining either the probability that such a population historically existed for a specific body of water or the size of such a population. An alternative approach is to define the subsistence anglers as the populations of individuals who, for cultural, ethnic, socioeconomic, or preferential reasons, may have consumption behaviors that mimic that which would be assumed for truly subsistence individuals. The potential subpopulations that were considered in this manner for the dose reconstruction included the following:

- individuals with low incomes who did not have the resources to purchase adequate amounts of food for themselves or their families and thus may have relied on self-caught fish or game to provide adequate nutrition;

- groups with distinct ethnic or cultural backgrounds that had access to large amounts of fish

- and, because of cultural traditions, historically relied upon fish as a major component of their diets;

- commercial anglers who, because of high levels of activity and the use of commercial fishing gear, had access to large amounts of fish and may have consumed the fish that they were not able to market successfully; and

- recreational anglers who chose, through personal preference, to consume large amounts of fish instead of other sources of dietary protein.

Commercial anglers and recreational anglers have already been targeted as populations of interest for the dose reconstruction, and distributions of their consumption behavior have been provided and discussed previously in this chapter. As can be seen in Figure 5-1, while the vast majority of these individuals consumed fish at rates at or below 30 g/person-day, a small number of individuals likely consumed fish at higher rates, and some consumed fish at rates exceeding 100 g/person-day. This is consistent with the findings of other surveys that have reported small fractions of angler populations who consumed fish at rates above 100 g/person-day (Ebert et al., 1993; SCCWRP and MBC, 1994). Thus, potential high consumers within these two populations have already been taken into consideration in this analysis and are not considered further under subsistence populations.

Low income anglers or anglers whose ethnic or cultural traditions have resulted in high levels of fish consumption may or may not, however, already be included in the recreational or commercial angler populations discussed above. Thus, it is necessary to consider whether there have historically been any of these populations living in proximity to the ORR and, if so, whether their consumption behaviors have differed from the behaviors of the general recreational or commercial angler populations in the area that are already being considered.

Income level

While there could be selected individuals who rely on fish as their primary source of protein, the survey literature has demonstrated no correlation between low income levels and increased levels of fish consumption either among anglers (West et al., 1989, 1991; NYSDEC, 1990; Anderson and Rice, 1993; Ebert et al., 1993; SCCWRP and MBC, 1994) or within the general U.S. population (USDA, 1941; Javitz, 1980). USDA (1955) data on the general U.S. population indicate that the quantities of meat, poultry and fish (combined) consumed increased as income increased. A study conducted in New York State further indicated that there were no substantial differences in consumption rates by income level. In that study, Wendt (1986) analyzed the fish consumption habits of low income families living in New York State to determine their levels of locally-caught freshwater fish consumption. For those who reported eating fish, the annual fish consumption rates (assuming a 227 g meal size) ranged from 0.6 to 60 g/person-day with a median rate of 5.3 g/person-day and a mean rate of 11 g/person-day. These rates were similar to the rates reported in various surveys of recreational anglers in the Northeastern U.S. (NYSDEC, 1990; Connelly et al., 1992, 1996; Ebert et al., 1993, 1996), and thus indicated that there were no substantial differences between the consumption rates of low income families and the general angler population in that region.

A similar conclusion can be drawn by comparing consumption of fish in the southeastern U.S. Kreiss et al. (1981) reported that low income individuals living at the confluence of Indian Creek and the Tennessee River in Triana, Alabama ate 4.3 fish meals per month. If a 227 g portion size is assumed, the resulting estimate of consumption is 32 g/day. Thus, the average intake rates for these anglers and the Watts Bar recreational anglers discussed above are very similar.

Based on available information, it appears that low income, in and of itself, may not have been a predisposing factor resulting in high levels of sport-caught fish consumption. While it is likely that there were low income populations located near the ORR, it does not necessarily follow that they had higher rates of fish consumption than other anglers living in the area. Even if such individuals were to rely on self-caught fish as a source of dietary protein, it is very likely that fish would not have been the sole source; rather, it would be expected that a portion of their dietary protein would have been provided by home-produced livestock or hunted game. Thus, while it is certain that recreational anglers in the area had a range of income levels, there is no need to consider any income-defined group as a subpopulation requiring separate evaluation in this analysis.

Ethnic background

There are data indicating that certain ethnic and cultural groups located in certain geographic regions may have higher rates of consumption than the general population. Several studies have reported that native peoples, particularly in the Pacific Northwest, Canada, and the Arctic Circle, rely more heavily on fish as a staple of their diets than does the general population (Wolfe and Walker, 1987; Dewailly et al., 1989; NYSDOH, 1993; Richardson and Currie, 1993; Coad, 1994; CRITFC, 1994). However, the available census and demographic data in the counties near the ORR during the past 50 years indicate that there has been no concentration of Native Americans living near the ORR (Bureau of the Census, 1952;1963a,b; 1973; 1983a,b; 1992a,b).

In evaluating fish consumption behaviors, it is important to determine if there were any ethnic or culturally-defined subpopulations of the recreational angler population that exhibited different consumption behaviors from those demonstrated by the angler population as a whole. The available census data indicate that the two ethnic populations that have historically been present in the area of the ORR are Caucasians and African Americans, with Caucasians comprising between 95 and 97 percent of the populations of Roane and Anderson Counties between 1960 and 1990. It is likely that the recreational angler population in this area generally reflected local demographics and was, therefore, predominantly Caucasian with a small percentage (three to five percent) of African Americans. Because of the high percentage of Caucasian anglers, it is expected that the estimated fish consumption rate distribution for the population as a whole can be considered representative of that segment of the population. It is important, therefore, to determine whether there may have been differences in the fish consumption behaviors of the smaller number of African Americans who were included in that population.

Historical data on the specific fish consumption patterns of Caucasians and African Americans are extremely limited but suggest that consumption rates for the groups are similar. USDA (1941) reported on the results of a 7-day study of dietary patterns and indicated that mean rates of fish consumption by Caucasians in the Southeastern U.S. ranged from 14 to 31 g/day while mean consumption by African Americans in that region ranged from 28 to 44 g/day. More similar rates of consumption for the two groups were reported by USDA (1950) for farm families in Georgia. In that analysis, it was reported that mean rates of consumption for Caucasians ranged from 23 to 29 g/person-day and mean rates for African Americans ranged from 21 to 29 g/person-day. Although these data are not specific to recreational anglers, they do indicate that there may have been small or no differences in the consumption patterns for these two groups in the Southeastern U.S.

More specific to the consumption behavior of anglers, Kreiss et al. (1981) studied exposure via fish consumption in Triana, Alabama. Triana is a rural town of 500 persons located at the confluence of Indian Creek and the Tennessee River. The individuals living in that town were predominantly African Americans (96.9 percent) of low socioeconomic status who had been employed in agriculture and had ready access to a substantial fishery. As stated previously, the median fish consumption rate reported for these individuals was estimated to be 4.3 fish meals per month or approximately one meal per week. If it is conservatively assumed that fish consumption was constant throughout the year for these individuals and that the average meal size was 1/2 lb. (227 g), this meal frequency results in an annualized consumption rate of 32 g/person-day. This is very similar to the mean consumption rate estimated for the predominately Caucasian recreational anglers at Watts Bar. For this reason, it is reasonable to assume that the distribution of consumption rates provided for the general recreational angler population is representative of both Caucasian and African American anglers and, as a result, it is not necessary to consider the African American subpopulation in a separate analysis.

5.4 CONSUMPTION BY CHILDREN

For the level I evaluation, a mean fish consumption rate of 30 g/person-day, the same as the mean consumption rate for Watts Bar recreational anglers, was used and was assumed to be representative of consumption by children as well as adults. This assumption is likely to have resulted in over estimates of intake for children. Most small children who eat fish do not consume adult size portions (1/2 pound). Thus, this consumption rate would likely overstate their actual level of consumption. Conversely, however, children have lower body weights than adults. Because delivered dose is defined as the measure of intake per kilogram of body weight, children may have received a higher dose than adults despite the fact that their portion sizes were smaller.

Data provided by Rupp et al. (1980) have been evaluated to account for the potential difference in intake between adults and children. Rupp et al. (1980) reported on the age-related consumption habits of individuals who participated in the one month diary study conducted by NPD Research, Inc. Based on the NPD data, Rupp reported that the mean per capita rates of freshwater fish consumption were 0.37 kg/yr (1.0 g/day) for 1-11 year old children, 0.37 kg/yr (1.0 g/day) for 12-18 year old teenagers, and 0.85 kg/yr (2.3 g/day) for adults in the East South Central region of the U.S., which included Tennessee. Dividing these consumption rates by appropriate age-related body weights of 23, 54, and 69 kg, respectively (EPA, 1989), results in body weight adjusted fish intake rates of 0.044 g fish/kg bw-day, 0.019 g fish/kg bw-day, and 0.033 g fish/kg bw-day, respectively. These estimated rates indicate that small children may consume 33 percent more freshwater fish on a body weight basis than adults, and that teenagers may consume 42 percent less fish than adults on a body weight basis. These differences were taken into consideration for the subsequent analyses of potential risks to children.

To account for differences in fish consumption behaviors by age group, an adjustment factor was introduced to estimate consumption by young children in the analysis of potential risks to children presented in Section 7.0. For 1-11 year old children, fish consumption rates selected randomly from the adult fish consumption rate distributions were multiplied by a factor of 1.3 to derive an age-appropriate estimate of delivered dose. While it appears that teenagers consume less fish on a per body weight basis than adults, no adjustment was made for this age group and their consumption rates and exposure potentials were assumed to be adequately represented by the adult consumption distributions and exposure analyses.

5.5 DISCUSSION AND SUMMARY

The consumption rate estimates discussed in this chapter are generally based on data collected since 1970. While it would have been preferable to also use data obtained between 1945 and 1970, adequately detailed data from that period have not been identified to support such an estimate. It is likely, however, that the use of more recent data has overestimated consumption in some situations or is comparable to what might have been in earlier years.

The estimates provided for commercial anglers are generally based on data collected during the past 25 years. While it cannot be stated with certainty, it is likely that the harvesting success of commercial anglers has remained fairly constant over the years and that commercial anglers have always sold as much of their harvest as possible. Thus, there is no reason to suspect that they may have eaten substantially greater amounts of fish in earlier years.

Similarly, it is likely that consumption by recreational anglers may have increased over earlier years, due to the fact that the fishing season is longer now than it was just after impoundment, that fishing gear is always improving, and that the number of fishing trips taken by typical U.S. anglers increased 24 percent between 1955 and 1985 (USDOI, 1988). In addition, household sizes have decreased steadily. (Similar amounts of fish harvested by anglers will yield larger portion sizes and consumption rates for the smaller number of household members who consume them.) For these reasons, it is likely that consumption rate estimates may be overestimated for recreational anglers who have used the resources over the last 50 years, and thus provide very conservative estimates of consumption during the entire dose reconstruction period.

The values used in this analysis are given in Table 5-3. Estimated means have been used in the level I evaluation discussed in Section 6.0 while the means and standard deviations were used to develop lognormal distributions of fish consumption rates for use in the level II evaluation presented in Section 7.0.

5.6 SUMMARY

Based on the information evaluated to date, it appears that anglers can be grouped into five different intake distributions including those developed for commercial anglers who fished Watts Bar or Clinch River/Poplar Creek, and recreational anglers who fished Watts Bar, Clinch River/Poplar Creek, or EFPC. The recreational angler distribution is applicable to individuals of all income and ethnic backgrounds. Evaluation of the ranges of intakes by each of these groups is the best means of characterizing both typical consumers and consumers with high intakes of fish within each population.

6.0 LEVEL I EVALUATION OF EXPOSURE AND ASSOCIATED RISKS– IDENTIFICATION OF EXPOSURE PATHWAYS AND POPULATIONS

The next three sections (Sections 6.0, 7.0 and 8.0) present an iterative quantitative risk assessment for the evaluation of PCB exposure. In this section, potential routes of exposure to PCBs are identified and estimates are made of the risks associated with each pathway. The project team then compared these risk estimates to decision criteria and determined which pathways represented the pathways most likely to result in risks to off-site populations. If risk estimates for the pathways were below the screening criteria, these pathways were set aside from further evaluation. If the risk estimates exceeded the criteria, (a hazard quotient of 1 or a risk of 1 x 10^{-4}) the pathways were subject to additional analyses (Sections 7.0 and 8.0). In order to avoid the elimination of pathways that deserved additional study, the screening process used input values that overestimated the actual risks that occurred. Examples of inputs where conservative values were applied include estimates of exposure duration, food consumption rates, and the historical levels of PCBs in environmental media and biota. The toxicity measures used in the analyses are biased (ATSDR, 1996; EPA, 1996a; EPA, 1998a,b), and also substantially overestimate the actual potential for adverse effects. As a result, the estimates of risk developed in this section should be viewed with caution.

For those pathways where risk estimates exceeded the criteria, the project team identified populations that could have been exposed via one of the pathways. We determined the size of each population and the time periods of exposure.

6.1 EXPOSURE PATHWAYS

Based on the results of Section 3.0, the project team identified a number of exposure pathways that may have resulted in off-site exposures to PCBs or PCB-combustion products. These pathways included direct exposure to PCBs in water, sediment, flood plain soils, and air, as well as indirect exposure through the ingestion of contaminated food (e.g., fish, vegetables, beef, and milk). Based on this information, the project team collected site-specific demographic information regarding farming, fishing, and recreational activities through interviews with current and past residents of Oak Ridge. The project team used this site-specific information, as well as measured levels of PCBs in the various media of concern, to confirm which of the proposed exposure pathways actually resulted in exposures to off-site populations. Those that were determined to be complete were considered in the level I evaluation.

For the purpose of the level I evaluation, the complete exposure pathways were characterized according to environmental media. The identified exposure pathways were further divided into two categories: direct and indirect. The direct pathways involve exposure by inhalation, ingestion, or dermal contact with a contaminated media, such as air, soil, sediment or water. Indirect pathways are those that involve the adsorption of the contaminant by plants or animals and the subsequent consumption of those plants or animals in the human diet. A summary of the pathways relevant to each medium is provided below.

6.1.1 Exposure Pathways Associated with Releases to Surface Water

Historical releases of PCBs into EFPC, Poplar Creek, Clinch River, and Watts Bar provided potential sources of exposure to residents living near the ORR. Although releases of PCBs to White Oak Creek occurred, access to this waterbody was restricted from the time that operations began at the ORR; therefore, significant exposures to off-site populations due to PCBs in sediments, biota and surface water of White Oak Creek were not considered in this analysis. PCBs were released from White Oak Creek and entered the Clinch River. This source of PCBs contributed to PCB levels in the Clinch River fish. This contribution was included in the assessment of exposures from the consumption of Clinch River fish.

Surface water is not the primary source of exposure to PCBs in aquatic systems. While some PCBs are retained in the water column, most are associated with suspended particulate matter or adsorbed to the bottom sediment. This is particularly true for the more highly chlorinated PCB mixtures which have a high affinity for organic carbon. During flooding events, contaminated sediments may be deposited on flood plain soils. Thus, exposure to PCBs associated with releases to surface water primarily involves pathways associated with sediment and flood plain soils and, to a much smaller extent, contact with water.

Using information on recreational and residential activities obtained through interviews with current and former residents, the project team identified both direct and indirect exposure pathways for each surface water body (Table 6-1). Direct exposures to sediment and surface water via incidental ingestion and dermal contact likely occurred during recreational activities such as fishing, swimming, and wading at EFPC, Poplar Creek, Clinch River, and Watts Bar. In contrast, the distance of residences from Poplar Creek, Clinch River, and Watts Bar, and the lower potential for flooding along these water bodies, suggested that exposure to flood plain soil was only important for EFPC. Therefore, incidental ingestion and dermal contact with flood plain soil were only considered for EFPC. Similarly, ingestion of drinking water was considered a complete pathway only for the Clinch River and Watts Bar because available data indicated that neither EFPC nor Poplar Creek were sources of drinking water.

The level I evaluation also considered indirect exposures to contaminated surface water, sediment, and flood plain soil. Based on historical information collected during interviews, the project team confirmed that several farm families lived along EFPC. These families raised their own vegetables in backyard gardens, raised beef cattle for home consumption, and owned dairy cows for milk (Anonymous, 1995a,b; Brooks, 1995; Clark, 1995; Sturm, 1995; DaMassa, 1996a; Waller, 1996). The project team assumed that the pastures where the cattle grazed were located within the contaminated flood plain, and that the vegetables produced by these families were also grown in contaminated flood plain soil. Once ingested by cattle, PCBs are stored in the muscle and fatty tissues of the animal. Thus, the project team assumed that PCBs were present in beef and in the fat contained in the milk produced from the animals. Similarly, because PCBs have some potential to accumulate in plants, the project team assumed that PCBs were present in the homegrown vegetables. Specifically, complete indirect pathways associated with flood plain soils at EFPC

Table 6-1. Exposure Pathways Evaluated in Screening Risk Assessment

Pathways	East Fork Poplar Creek	Poplar Creek	Clinch River	Watts Bar	Use of Waste Oil for Dust Control	Exposures to Air Emissions
Surface water						
Incidental ingestion	X	X	X	X		
Dermal contact	X	X	X	X		
Ingestion of drinking water			X	X		
Beef ingestion	X					
Milk ingestion	X					
Soil						
Incidental ingestion	X				X	
Dermal contact	X				X	
Inhalation of particulates	X				X	
Vegetable ingestion	X					
Beef ingestion	X					
Pasture to beef ingestion	X					
Milk ingestion	X					
Pasture to milk ingestion	X					
Sediment						
Incidental ingestion	X	X	X	X		
Dermal contact	X	X	X	X		
Biota						
Fish ingestion	X	X	X	X		
Turtle ingestion	X	X	X	X		
Air [a]						
Inhalation of vapors						X
Vegetable ingestion						X
Beef ingestion						X
Pasture to beef ingestion						X
Milk ingestion						X
Pasture to milk ingestion						X

a. Evaluated for both PCBs and PCDFs potentially formed during the burning of PCBs at the K-25 burn area, Y-12 burn tank and burn pit, and TSCA incinerator.

included consumption of beef and milk from cattle raised on farms in the flood plain and homegrown vegetables raised in flood plain soils.

For the Clinch River, ingestion of PCB-contaminated beef from cattle grazed on Jones Island was initially considered as a potential exposure pathway. Although flood plain soil data were not available for Jones Island, preliminary information suggested that beef cattle sold for local consumption were grazed on pastures treated with PCB-contaminated dredge spoils from the Clinch River. Through additional investigation, the project team determined that while beef cattle were raised on Jones Island, they grazed there prior to the time when dredged material from the Clinch River was deposited on the island (Sutton, 1996; Wade, 1996; Waller, 1996). As a result, any cattle that grazed on Jones Island were not exposed to PCBs; thus, potential human exposure via this pathway was not evaluated further.

PCBs in sediment and surface water enter the aquatic food chain through sediment dwelling organisms (benthic macroinvertebrates) and accumulate in aquatic species, such as fish and turtles, that consume them. Based on information regarding fishing activities, consumption of fish exposed to PCBs in sediments and surface water was evaluated for EFPC, Poplar Creek, Clinch River, and Watts Bar. In addition, preliminary anecdotal information on recreational activities for EFPC indicated that turtles from the stream may have been consumed occasionally by local residents (Mills et al., 1995). Assuming that turtles were caught and consumed from the other water bodies, consumption of turtles was also considered as a potential exposure pathway.

6.1.2 Exposure Pathways Associated with Air Releases

Historical data indicated that waste oil was burned at the K-25 Burn Area, the Y-12 Burial Ground A Burn Pit and at the Y-12 Burn Tank. This waste oil may have contained PCBs and, as a result of the burning activities, PCBs may have been released into the ambient air. It is also likely that polychlorinated dibenzofurans (PCDFs) were released because these compounds can be formed when PCBs are burned in an uncontrolled manner such as burning in open pits (Rappe et al., 1982; Erickson et al., 1985). In addition, in the late 1980s, burning of wastes at the TSCA incinerator at K-25 was initiated. Although not a source of historical releases, there has been public concern regarding releases from this facility. Therefore, as part of the level I evaluation, the project team evaluated the potential health impacts of these releases as well.

To evaluate these potential air exposures, the project team modeled the releases of PCBs and PCDF using the air dispersion model, SCREEN3 (95250). SCREEN3 is recommended by the USEPA as a screening level tool for estimating potential worst-case impacts resulting from single source emissions (EPA, 1995). The analysis was conducted using the following steps:

- identified locations where PCB combustion may have occurred,
- determined the composition and quantity of the material burned,
- determined the manner in which the material was burned,
- identified the PCB destruction efficiency, potential products of the combustion process, associated release rate, and other physical parameters affecting the releases,
- described the local surroundings affecting releases dispersion,
- identified the closest receptor located outside the ORR, and

- performed air dispersion modeling to estimate the potential impacts at the receptor.

The project team evaluated direct inhalation of air releases as well as several indirect routes of exposure to both PCB and PCDF releases. The following indirect pathways were evaluated:

- ingestion of beef from cattle that inhaled the airborne contaminants;
- ingestion of beef exposed to contaminants from deposition onto pasture;
- ingestion of milk from cattle that inhaled the airborne contaminants;
- ingestion of milk from dairy cattle exposed to contaminants from deposition onto pasture; and
- ingestion of vegetables exposed to contaminants by direct deposition of contaminants.

For each of these pathways, the air concentration at the point of exposure estimated by the SCREEN3 (EPA, 1995) model was used to predict the initial chemical concentration from each source (see Appendix C). These concentrations were then combined to derive an average air concentration for the purpose of calculating potential risks to the off-site populations evaluated in the level I evaluation. To determine the concentration of PCBs and PCDF in vegetables and pasture vegetation, air to plant transfer factors were applied to these average concentrations as described in Section 6.4.8**.** Similarly, for the pathways evaluating risks due to ingestion of beef or milk from cattle exposed through either inhalation or through consumption of exposed vegetation, transfer factors were used to estimate the fraction of PCBs or PCDFs inhaled or ingested by the cattle that would partition into the beef and milk (Fries et al., 1973; Travis and Arms, 1988).

6.1.3 Sale of Waste Oil

Interviews with retired personnel indicated that waste oils, comprised primarily of used motor oils and solvents, were sold to the general public by the ORR during the 1950s (Banic, 1995a). The manner in which these oils were used by the buyer is unclear; however, uses may have included engine lubrication, rust prevention, supplemental fuel, vegetation control, and application to dirt roads for the control of dust. While the relatively small volume of use, high value, and recycling potential of PCBs made it unlikely that commercial PCB mixtures, such as transformer fluids or specialty cutting fluids, were sold as waste oils, the possibility remains that waste oils contaminated with PCBs may have been sold, resulting in exposures to off-site populations.

The project team selected the use of oils for dust suppression as a representative scenario to evaluate the potential exposure to PCBs from the possible uses of waste oil. Dust suppression was selected for several reasons. First, this activity could have contaminated large areas and thereby resulted in the exposure of more individuals. Second, children could potentially have been exposed. Third, the spreading of waste oil could have resulted in daily exposures over considerable periods of time. Finally, several pathways, including ingestion, dermal contact, and inhalation may have been involved. In contrast, exposures from other possible uses of waste oils would have been limited to adults and would only have occurred over relatively short periods of time.

It should be noted that, while the use of waste oil to suppress dust likely represents the scenario with the greatest potential of exposure compared to other uses, interviews with town officials indicated that Oak Ridge did not typically oil roads for dust control (Personal Communication, Robert Collier, April 1997). Although the Roane County highway department occasionally sprayed oil on roads in response to specific complaints from residents, the oils typically used were waste oils generated by the department, rather than oils purchased from other sources (Personal Communication, Robert Collier, April 1997). Despite the limited potential for the county to have used waste oils from the ORR for this purpose, it is possible that local individuals could have purchased waste oil from the ORR and applied it to private roads or driveways.

6.2 INTAKE RATE EQUATIONS

This section describes the equations used to estimate the potential daily intake of PCBs by adults and children resulting from the exposure pathways identified in Section 6.1. The intake rate is described as the lifetime average daily dose rate (LADD) when calculating carcinogenic risks for chemicals. For noncarcinogenic risks, it is described as the average daily dose rate (ADD). The same equation is used to calculate the LADD and ADD; the difference between the two measures is the value used to represent the averaging time. The ADD is averaged over the period of exposure duration, while the LADD is averaged over an individual's lifetime (i.e., 70 years). Both the LADD and the ADD are expressed in units of milligrams of chemical per kilogram body weight per day (mg/kg-d). For each intake rate equation, definitions for variables that remain constant across pathways are provided for the first pathway only and are not repeated. Parameters that are unique to a pathway are defined for that specific pathway.

6.2.1 Air Releases

As previously discussed, direct and indirect exposures associated with releases of PCBs and PCDFs were evaluated. For the indirect pathways, such as the consumption of vegetables, beef, or milk, biotransfer factors were applied to the estimated long-term average concentration of PCBs (or PCDFs) in air to estimate the resulting concentration associated with the bioaccumulation of these compounds.

Inhalation of Vapors

$$Intake = \frac{C_a \ x \ U_a \ x \ f_t x \ EF \ x \ ED}{AT \ x \ BW}$$

Where:

Intake	=	LADD or ADD (mg/kg-d)
C_a	=	Concentration in air (mg/m^3)
U_a	=	Quantity of air inhaled per day (m^3/d)
f_t	=	Fraction time exposed to contaminated air (unitless)
EF	=	Exposure frequency (d/y)
ED	=	Exposure duration (y)

AT	=	Averaging time (d)
BW	=	Body weight (kg)

Ingestion of Vegetables Contaminated by Direct Deposition

$$Intake = \frac{C_v \ x \ WD \ x \ U_v \ x \ f \ x \ EF \ x \ ED}{AT \ x \ BW}$$

Where:

C_v	=	Concentration of PCBs or PCDFs in vegetables (mg/kg dry wt)
WD	=	Dry to fresh weight conversion factor (unitless)
U_v	=	Average daily consumption of vegetables (kg wet wt/d)
f_{cv}	=	Fraction of vegetables consumed that was contaminated (unitless)

Ingestion of Beef from Cattle Exposed via Inhalation

$$Intake = \frac{C_a \ x \ Q_{air(b)} \ x \ B_b \ x \ U_b \ x \ f_{cb} \ x \ EF \ x \ ED}{AT \ x \ BW}$$

Where:

$Q_{air(b)}$	=	Daily inhalation rate of beef cattle (m^3/d)
B_b	=	Beef biotransfer factor (d/kg)
U_b	=	Average daily consumption of beef (kg/d)
f_{cb}	=	Fraction of beef consumed that was contaminated (unitless)

Ingestion of Beef from Cattle Exposed via Consumption of Contaminated Vegetation

$$Intake = \frac{C_p \ x \ Q_{feed(b)} \ x \ f_{pb} \ x \ B_b \ x \ U_b \ x \ f_{cb} \ x \ EF \ x \ ED}{AT \ x \ BW}$$

Where:

C_p	=	Concentration of PCBs or PCDFs in pasture (mg/kg)
$Q_{feed(b)}$	=	Daily consumption of feed by beef cattle (kg/d)
f_{pb}	=	Fraction of feed consumed by beef cattle that was contaminated pasture (unitless)

<u>Ingestion of Milk from Dairy Cattle Exposed via Inhalation</u>

$$Intake = \frac{C_a \ x\ Q_{air(d)} \ x\ B_m \ x\ U_m \ x\ f_{cm} \ x\ EF \ x\ ED}{AT \ x\ BW}$$

Where:

$Q_{air(d)}$	=	Daily inhalation rate of dairy cattle (m^3/d)
B_m	=	Milk biotransfer factor (d/kg)
U_m	=	Average daily consumption of milk (kg/d)
f_{cm}	=	Fraction of milk consumed that was contaminated (unitless)

<u>Ingestion of Milk from Dairy Cattle Exposed via Consumption of Contaminated Vegetation</u>

$$Intake = \frac{C_p \ x\ Q_{feed(d)} \ x\ f_{pd} \ x\ B_m \ x\ U_m \ x\ f_{cm} \ x\ EF \ x\ ED}{AT \ x\ BW}$$

Where:

C_p	=	Concentration of PCBs or PCDFs in pasture (mg/kg)
$Q_{feed(d)}$	=	Daily consumption of feed by dairy cattle (kg/d)
f_{pd}	=	Fraction of feed consumed by dairy cattle that was contaminated pasture (unitless)

6.2.2 Surface Water

The project team evaluated direct exposures to surface water via incidental ingestion and dermal contact during recreational activities at each of the water bodies. In addition, the project team assumed that Watts Bar and the Clinch River were sources of drinking water. Thus, ingestion of drinking water was evaluated for these two bodies of water. Indirect exposures associated with the consumption of beef or milk from cattle potentially drinking from impacted water bodies were also evaluated. Biotransfer factors were used to estimate the fraction of PCBs ingested by beef or dairy cattle that would have partitioned into the beef or milk.

<u>Incidental Ingestion of Surface Water</u>

$$Intake = \frac{C_{sw} \ x\ U_{w(r)} \ x\ f_{cw} \ x\ EF \ x\ ED}{AT \ x\ BW}$$

Where:

C_{sw}	=	Concentration in surface water (mg/L)
$U_{w(r)}$	=	Consumption of surface water during recreational activities (L/d)
f_{cw}	=	Fraction of surface water consumed that was contaminated (unitless)

Dermal Contact with Surface Water

$$Intake = \frac{C_{sw}\ x\ PC\ x\ SA\ x\ f_{cw}\ x\ ET\ x\ EF\ x\ ED}{AT\ x\ BW}$$

Where:

PC	=	Permeability coefficient (cm/h)
SA	=	Skin surface area (cm^2)
f_{cw}	=	Fraction of surface water contacted that was contaminated (unitless)
CF	=	Conversion factor (L/cm^3)

Ingestion of Beef from Cattle Exposed to Surface Water

$$Intake = \frac{C_{sw}\ x\ Q_{water(b)}\ x\ f_{cw}\ x\ B_b\ x\ U_b\ x\ f_{cb}\ x\ EF\ x\ ED}{AT\ x\ BW}$$

Where:

$Q_{water(b)}$	=	Beef cattle water consumption rate (L/d)

Ingestion of Milk from Dairy Cattle Exposed to Surface Water

$$Intake = \frac{C_{sw}\ x\ Q_{water(d)}\ x\ f_{cw}\ x\ B_m\ x\ U_m\ x\ EF\ x\ ED}{AT\ x\ BW}$$

Where:

$Q_{water(d)}$	=	Dairy cattle water consumption rate (L/d)

Ingestion of Drinking Water

$$Intake = \frac{C_{sw}\ x\ U_{water(d)}\ x\ f_{cd}\ x\ EF\ x\ ED}{AT\ x\ BW}$$

Where:

C_{dw}	=	Concentration in surface water (mg/L)
$U_{water(d)}$	=	Average daily consumption of drinking water (L/d)
f_{cd}	=	Fraction of drinking water consumed that was contaminated

6.2.3 Soil and Sediment

Pathways associated with exposures to soil and sediment were similar. The project team evaluated direct exposures via incidental ingestion, dermal contact, and inhalation of dust as well as indirect pathways associated with consumption of potentially contaminated beef and milk. The equations used to describe direct contact with soil were also used in the scenario to evaluate exposures to soils amended with waste oils. As discussed for the air and surface water pathways, biotransfer factors were used to estimate the fraction of chemical to which the cattle were exposed that would partition into beef or milk.

Inhalation of Dust

$$Intake = \frac{C_s\ x\ U_p\ x\ LD\ x\ 1/PEF\ x\ f_t\ x\ ET\ x\ EF\ x\ ED}{AT\ x\ BW}$$

Where:

C_s	=	Concentration in soil or sediment (mg/kg dry weight)
U_p	=	Daily inhalation rate of particulates (m^3/h)
LD	=	Lung deposition fraction (unitless)
PEF	=	Particle emission factor (m^3/kg)

Ingestion of Soil/Sediment

$$Intake = \frac{C_s\ x\ U_s\ x\ f_{sc}\ x\ EF\ x\ ED\ x\ CF}{AT\ x\ BW}$$

Where:

U_s	=	Average daily ingestion of soil or sediment (mg/d)
f_{sc}	=	Fraction of soil/sediment ingested that was contaminated (unitless)
CF	=	Conversion factor (kg/mg)

Dermal Contact with Soil/Sediment

$$Intake = \frac{C_s \ x \ SA \ x \ AF \ x \ Ab \ x \ f_{sd} \ x \ EF \ x \ ED \ x \ CF}{AT \ x \ BW}$$

Where:

SA	=	Skin surface area (cm^2)
AF	=	Adherence factor (mg/cm^2-d)
Ab	=	Dermal bioavailability (unitless)
f_{sd}	=	Fraction of soil/sediment that was contaminated (unitless)
CF	=	Conversion factor (kg/mg)

Ingestion of Vegetables Grown in Contaminated Soil

$$Intake = \frac{C_s \ x \ B_v \ x \ U_v \ x \ f_{cv} \ x \ WD \ x \ EF \ x \ ED}{AT \ x \ BW}$$

Where:

B_v	=	Bioaccumulation of contaminants by vegetables (unitless)
U_v	=	Average daily consumption of vegetables (kg wet weight/d)
f_{cv}	=	Fraction of vegetables consumed that was contaminated (unitless)

Ingestion of Beef from Cattle Exposed via Ingestion of Contaminated Soil/Sediment

$$Intake = \frac{C_s \ x \ Q_{soil(b)} \ x \ f_{cs(c)} \ x \ B_b \ x \ U_b \ x \ f_{cb} \ x \ EF \ x \ ED}{AT \ x \ BW}$$

Where:

$Q_{soil(b)}$	=	Daily ingestion of soil by beef cattle (kg/d)
$f_{cs(c)}$	=	Fraction soil/sediment contaminated (unitless)

Ingestion of Beef Grazed on Vegetation Grown in Contaminated Soil/Sediment

$$Intake = \frac{C_s \ x \ B_p \ x \ Q_{feed(b)} \ x \ f_{pb} \ x \ B_b \ x \ U_b \ x \ f_{cb} \ x \ EF \ x \ ED}{AT \ x \ BW}$$

Where:

B_p	=	Bioaccumulation of contaminants by pasture (unitless)

Ingestion of Milk from Dairy Cattle Exposed via Ingestion of Contaminated Soil/Sediment

$$Intake = \frac{C_s \ x\ Q_{soil(d)} \ x\ f_{cs(c)} \ x\ B_m \ x\ U_m \ x\ f_{cm} \ x\ EF \ x\ ED}{AT \ x\ BW}$$

Where:

$Q_{soil(d)}$	=	Daily ingestion of soil by dairy cattle (kg/d)

Ingestion of Milk from Dairy Cattle Grazed on Vegetation Grown in Contaminated Soil/Sediment

$$Intake = \frac{C_s \ x\ B_p \ x\ Q_{feed(d)} \ x\ f_{pd} \ x\ B_m \ x\ U_m \ x\ f_{cm} \ x\ EF \ x\ ED}{AT \ x\ BW}$$

6.2.4 Biota

The project team estimated exposures resulting from the consumption of aquatic biota (i.e., fish and turtles) exposed to PCBs in sediments and surface water using the following equations.

Ingestion of fish

$$Intake = \frac{C_f \ x\ U_f \ x\ EF \ x\ ED \ x\ f_{cf} \ x\ CF}{AT \ x\ BW}$$

Where:

C_f	=	Concentration in fish (mg/kg)
U_f	=	Average daily consumption of fish (g/d)
f_{cf}	=	Fraction of fish consumed that was contaminated (unitless)
CF	=	Conversion factor (kg/g)

Ingestion of turtles

$$Intake = \frac{C_t \ x\ U_t \ x\ EF \ x\ ED \ x\ f_{ct} \ x\ CF}{AT \ x\ BW}$$

Where:

C_t	=	Concentration in turtles(mg/kg)
U_t	=	Average daily consumption of turtles (g/d)
f_{ct}	=	Fraction of turtle consumed that was contaminated (unitless)
CF	=	Conversion factor (kg/g)

6.3 EXPOSURE PARAMETERS

For the screening assessment, parameter values were selected to represent reasonable upper bound estimates of exposure to ensure that the intakes were not underestimated. Although many of the parameter values were based on standard default assumptions, where possible, values were based on historical site-related information obtained through interviews with current and former residents. Age-specific parameter values are presented in Table 6-2. Parameter values common to both adults and children are presented in Table 6-3. Unless otherwise noted, the values apply to all exposure pathways and water bodies.

6.4 EXPOSURE POINT CONCENTRATIONS

Exposure point concentrations (EPCs) for the screening analysis were based on historical data obtained from a variety of sources including studies performed by TVA and DOE. These historical data were limited, however, particularly for time periods prior to the 1970s. However, the available data indicate that fish levels have been relatively constant throughout the years analyzed. In addition, the HydroQual evaluation provides evidence that sediment levels have not varied significantly over time. Because of these factors and the fact that our evaluation is a retrospective analysis, all available sampling data were considered regardless of the year in which the data were collected. Temporal variations in measured concentrations were not considered. Soil and sediment samples taken at depth were also considered. Although PCBs present in buried sediments are no longer available to the current food chain, these samples may represent historical levels of PCBs.

Only those data pertaining to areas where off-site exposures likely occurred were used to develop the EPCs. For example, samples taken within the Bear Creek Disposal Area or on-site disposal areas or settling ponds were not considered because off-site populations had no access and hence no exposure to PCBs in these areas. Data evaluated for the Clinch River were limited to samples collected downstream of Melton Hill Dam. In Poplar Creek, only those samples collected between the stream's confluence with EFPC and its confluence with the Clinch River were considered. For EFPC, data collected within or upstream of the Y-12 facility boundary (such as measurements from New Hope Pond or Lake Reality) were not included due to lack of public access to these areas. Data for Watts Bar Reservoir included samples collected from the Tennessee River between river mile (RM) 530 (Watts Bar dam) and RM 570 (vicinity of confluence with the Clinch River).

Selecting data for the EPCs was hampered by variations in the data quality and level of detail reported for each data set. For example, some investigations evaluated total PCBs, while others evaluated specific Aroclors. Frequently, only summary statistics (i.e., minimum, maximum,

Table 6-2. Age-Specific Exposure Parameters Used to Calculate Daily Intakes

Parameter	Rationale	Adult Value	Child Value	Units	Reference
Body weight (BW)	Default value based on data from the Exposure Factors Handbook.	70	16	kg	EPA, 1997
Exposure duration (ED)					
All pathways except oiled road and inhalation scenario	Based on a maximum duration of 50 years of operations at the ORR (1945-1995). Assumed 5 years spent as a child.	45	5	years	Site specific
Oiled road scenario	Based on period during which contaminated waste oils may have been sold (mid 1940s to late 1960s). Due to regulatory pressure in the late 1960s, it is unlikely that PCB contaminated waste oils were sold after 1970. Assumed 5 years spent as a child.	20	5	years	Site specific
Inhalation scenario	Based on period of time that burn pit was operational. Assumed 5 years of that time spent as a child.	10	5	years	Site specific
Surface area (SA)					
Exposed to soil	25 percent of the total body (hands, forearms, lower legs, and feet) was assumed to be exposed to soil.	5,800	2,500	cm^2	Site specific
Exposed to surface water for all waterbodies except Poplar Creek	During swimming activities at EFPC, Clinch River and Watts Bar, the upper end estimate of the total body surface area was selected as the surface area.	19,400	7,280	cm^2	Site specific
Exposed to surface water in Poplar Creek	Swimming at Poplar Creek was not considered viable. Surface area was calculated for a wading angler assuming hands, forearms, lower legs, and feet.	5,800	2,500	cm^2	Site specific
Intake Rates					
Inhalation rate (vapor) (U_a)	Default EPA values.	20	12	m^3/day	EPA, 1989a
Inhalation rate (dust) (U_p)	Default EPA values.	0.83	0.5	m^3/hr	EPA, 1989a
Soil ingestion (U_s)	Upper bound values.	100	250	mg/day	Lepow et al., 1975; EPA, 1989a
Sediment ingestion (U_s)	Soil ingestion rates used in absence of published sediment ingestion rates.	100	250	mg/day	Lepow et al., 1975; EPA, 1989a
Drinking water ingestion ($U_{water(d)}$)	Upper bound tapwater intake rates.	2.2	1.3	L/day	Ershow and Cantor, 1989
Incidental surface water ingestion ($U_{w(r)}$)	Default EPA value.	0.05	0.05	L/hr	EPA, 1989a
Beef consumption rate (U_b)	Conservative estimate based on average total intake of beef during 1955 and 1965.	0.1	0.05	kg/day	Rupp, 1980
Milk consumption rate (U_m)	Conservative estimate.	1	1	L/day	Rupp, 1980
Vegetable consumption rate (U_v)	Conservative estimate based on average total intake of fresh produce during 1955 and 1965.	0.5	0.4	kg/day	Rupp, 1980
Fish consumption rate (U_f)	Based on harvest statistics for Watts Bar collected between 1977 and 1991.	30	30	g/day	TWRA, 1993
Turtle consumption rate (U_t)	Based on one 0.5 lb turtle meal per week.	32	32	g/day	Personal communication, F. Miller, 2/20/97

Table 6-3. Common Exposure Parameters Used to Calculate Daily Intakes

Parameter	Rationale	Value	Units	Reference
Exposure frequency (EF)				
All pathways except oiled road scenario, dermal contact and incidental ingestion of surface water, and dermal contact and incidental ingestion of sediment for Poplar Creek/Clinch River	Conservative assumption.	365	days/yr	Professional Judgement
Oiled road scenario	Exposure assumed to occur for 3 days per week during ten months of the year. No exposure was expected to occur in wet weather and winter.	130	days/yr	Professional Judgement
Dermal contact and incidental ingestion of surface water at EFPC	In comparison to other more appealing waterbodies, swimming at EFPC was expected to occur infrequently. National average for swimming was reduced by 50 percent.	4	days/yr	Professional Judgement; EPA, 1989a
Dermal contact and incidental ingestion of surface water at Poplar Creek; dermal contact and incidental ingestion of sediment at Poplar Creek and Clinch River	Individuals expected to fish Poplar Creek and Clinch River 1 day/month throughout the year.	12	days/yr	Professional Judgement
Dermal contact and incidental ingestion of surface water at Clinch River and Watts Bar	Consistent with national average for swimming.	7	days/yr	EPA, 1989a
Exposure time (ET)				
Oiled road scenario	Based on time an adult may walk on a dirt road or a child may play by the side of a dirt road.	2	hrs/day	Professional Judgement
Dermal contact and incidental ingestion of surface water at EFPC	Assumed that swimming at EFPC occurred infrequently.	0.5	hrs/day	Professional Judgement
Dermal contact and incidental ingestion of surface water at Poplar Creek	Based on time spent fishing in a given day.	4	hrs/day	Professional Judgement
Dermal contact and incidental ingestion of surface water at Clinch River and Watts Bar	Consistent with national average for swimming.	2.6	hrs/day	EPA, 1989a
Adherence factor (AF)	Conservative estimate based on data from hand measurements.	0.5	mg/cm^2-day	Lepow et al., 1975
Dermal bioavailability (Ab)	EPA default value	6	percent	EPA, 1992
Permeability coefficient (PC)	Estimated value from dermal assessment document.	7.1×10^{-1}	cm/hr	EPA, 1992
Averaging time (noncarcinogenic) (AT)	Exposure duration x 365 days.	scenario specific	days/year	EPA, 1989a
Averaging time (carcinogenic) (AT)	Average dose rate over 70 year lifetime.	25,550	days	EPA, 1989a
Beef cattle intake rates				
Feed ($Q_{feed(b)}$)	Upper bound of data reported for dry matter intake.	10	kg/day	IAEA, 1994
Soil ($Q_{soil(b)}$)	Based on a range of soil ingestion rates for both beef and dairy cattle of 0.1 to 0.71 kg/day.	0.5	kg/day	McKone, 1988
Water ($Q_{water(b)}$)	Upper bound estimate based on a range of 37.6 to 50 L/day.	50	L/day	McKone, 1988
Vapor ($Q_{air(b)}$)	Based on average cattle inhalation rate of 122 m^3/day and a range of 85 to 150 m^3/day.	130	m^3/day	McKone, 1988
Dairy cattle consumption rates				
Feed ($Q_{feed(d)}$)	Upper bound estimate for cattle raised in an unmanaged feeding regime.	16	kg/day	Koranda, 1965
Soil ($Q_{soil(d)}$)	Same as beef cattle.			
Water ($Q_{water(d)}$)	Upper bound estimate based on range of 37.5 to 60 L/day.	60	L/day	McKone, 1988

Table 6-3 cont. Common Exposure Parameters Used to Calculate Daily Intakes

Parameter	Rationale	Value	Units	Reference
Dairy cattle consumption rates (cont.)				
Vapor ($Q_{air(d)}$)	Same as beef cattle.	130	m^3/day	McKone, 1988
Fraction feed as contaminated pasture-beef cattle (f_{pb})	Conservative assumption.	80	percent	Professional Judgement
Fraction feed as contaminated pasture-dairy cattle (f_{pd})	Conservative assumption.	50	percent	Professional Judgement
Fraction contaminated				
Air (f_t)	Conservative assumption.	80	percent	Professional Judgement
Soil (f_{sc})	Conservative assumption.	100	percent	Professional Judgement
Surface water (f_{caw})	Conservative assumption.	100	percent	Professional Judgement
Drinking water (f_{cd})	Conservative assumption.	50	percent	Professional Judgement
Beef (f_{cb})	Conservative assumption.	80	percent	Professional Judgement
Milk (f_{cm})	Conservative assumption.	100	percent	Professional Judgement
Vegetables (f_{cv})	Conservative assumption.	60	percent	Professional Judgement
Fish (f_{cf})	Conservative assumption.	100	percent	Professional Judgement
Turtles (f_{ct})	Conservative assumption.	100	percent	Professional Judgement
Biotransfer factors				
Beef (B_b)	Calculated as the concentration in beef (mg/kg) divided by the daily intake of organic compounds (mg/day).	0.0525	day/kg	Travis and Arms, 1988
Milk (B_m)	Calculated as the concentration in milk (mg/kg) divided by the daily intake of organic compounds (mg/day).	0.01122	day/kg	Travis and Arms, 1988
Uptake from soil by pasture and vegetables (B_p, B_v)	Conservative estimate based on upper bound BCF for carrots.	4	percent	Iwata and Gunther, 1976; Moza et al., 1979; O'Connor et al., 1990
Dry weight to fresh weight conversion factor (WD)	EPA default value.	0.85	unitless	EPA, 1994b
Lung deposition fraction (LD)	Based on data that indicates that only 25% of inhaled particles are deposited in lower passages of lung. Of this 25%, half is retained in lungs with a half-life of 120 days. The other half is swallowed within first 24 hours.	12.5	percent	Paustenbach et al., 1992
Particle emission factor (PEF)	EPA default value.	4.63×10^9	m^3/kg	EPA, 1991

arithmetic mean) were reported. In addition, detection limits were high, by current standards, and varied substantially among investigations. Finally, many studies focused on sample collection near known sources (e.g., discharge pipes, chemical spills) and in areas with limited public access. These concentrations are not included in the calculation of EPCs because they were not likely to be relevant to actual off-site exposures.

In this conservative, screening level analysis, the EPCs for soil, sediment, surface water, drinking water, and aquatic biota were defined as the maximum total PCB concentration for each medium for each water body. In the absence of data on total PCBs in a sample, the maximum values reported for each Aroclor were summed to provide an estimated maximum total PCB concentration, regardless of whether the reported concentrations were measured in the same sample. This approach overestimated the actual total PCB concentration because it assumed that the maximum concentrations of each Aroclor were reported in the same sample. One-half the reported detection limit was used for values reported as not detected. If all Aroclors were reported as not detected, the estimated total PCB concentration was conservatively assumed to be equivalent to the maximum reported detection limit.

The EPCs for the direct air pathways were modeled using a conservative gaussian air dispersion model, SCREEN3 (EPA, 1995). For the indirect air pathways, the EPCs were derived by predicting the concentrations in vegetables, beef, or milk based on measured or estimated concentrations in various media. For example, the concentration of PCBs in beef from cattle that were pastured downwind of the K-25 burn area was derived based on the amount of PCBs estimated to be released from the burn area (using an air dispersion model), a model of air-to-pasture transfer of PCBs, and a model of bioaccumulation in beef fat.

A summary of the data reviewed for each environmental medium is presented below. Table 6-4 presents the EPC estimated for each medium.

6.4.1 Soil

The project team evaluated soil data only for the flood plain of EFPC. The most extensive survey of flood plain soil was conducted by SAIC (1994) during the East Fork Poplar Creek - Sewer Line Beltway Remedial Investigation. This survey reported the results of samples taken from nine transects along EFPC. The exact location of these samples in relation to gardens and pastures along EFPC was not investigated. However, these data were assumed to be representative of exposure conditions. Concentrations of Aroclor 1260 ranged from 0.012 mg/kg to 3.8 mg/kg, while concentrations of Aroclor 1254 ranged from 0.03 mg/kg to 3 mg/kg. An estimated maximum total PCB concentration of 6.8 mg/kg was determined based on the sum of these data. However, an earlier study conducted by researchers at Y-12 (Welch, Unk.) reported a maximum soil concentration in the vicinity of EFPC of 18 mg/kg. Although the specific location of this sample was not reported, the project team selected 18 mg/kg as a conservative EPC for this screening analysis.

Table 6-4. Exposure Point Concentrations in Media Evaluated in Screening Risk Assessment

Media	Value PCBs	PCDF[a]	Units	Rationale	Reference
Air				Results of SCREEN3 air dispersion modeling[b].	Appendix C
Direct	4.1E-03	3.6E-06	µg/m³		
Indirect	3.7E-03	2.9E-06	µg/m³		
Soil	18	NA	mg/kg	Maximum soil concentration reported for EFPC	Welch, Unk.
Sediment					
EFPC	6	NA	mg/kg	Sum of maximum Aroclor concentrations reported for EFPC.	TVA, 1985
Poplar Creek	4.3	NA	mg/kg	Total PCB concentration reported at area near Blair Road, a potential access point for recreational fishing.	TVA, 1983b
Clinch River	14.3	NA	mg/kg	Maximum of total PCB concentrations reported from the mouth of the river to Melton Hill Dam.	Hoffman et al., 1991; Suter, 1991; Cook et al., 1992
Watts Bar	0.1	NA	mg/kg	Limited data indicate that all sediment concentrations were below detection limits of 0.1 mg/kg.	Suter, 1991; Cook et al., 1992
Surface Water [b]					
EFPC & Watts Bar	0.1	NA	µg/L	All reported concentrations below detection limits of 0.1 to 0.5 µg/L.	TVA, 1983b; Kornegay et al., 1991; Welch, Unk.; Cook et al., 1992; MMES, 1985
Poplar Creek & Clinch River	0.1	NA	µg/L	All detected concentrations associated with spill events or collected near discharge areas and are not expected to be associated with actual off-site exposure concentrations.	Suter, 1991; Kornegay, 1992
Waste Oil	6.9	NA	mg/kg	Calculated based on soil concentrations.	Table 4-5
Fish					
EFPC	3.7	NA	mg/kg	Maximum total PCB concentration in carp.	TVA, 1986
Poplar Creek	8.5	NA	mg/kg	Measured total PCB concentration in longnose gar.	Loar et al., 1981
Clinch River	12	NA	mg/kg	Estimated maximum total PCB concentration measured in carp.	MMES, 1985
Watts Bar	7.5	NA	mg/kg	Maximum reported PCB concentration in channel catfish.	Dycus, 1986
Turtles					
EFPC	0.032	NA	mg/kg	Maximum reported PCB concentration in turtle muscle.	VanAudenhove, 1997
Poplar Creek	3.38	NA	mg/kg	Maximum reported PCB concentration in turtle muscle.	VanAudenhove, 1997
Clinch River	0.814	NA	mg/kg	Maximum reported PCB concentration in turtle muscle.	VanAudenhove, 1997
Watts Bar	0.922	NA	mg/kg	Maximum reported PCB concentration in turtle muscle.	VanAudenhove, 1997
Vegetables	0.00012	2.6E-07	mg/kg	Calculated based on modeled air concentrations[d].	EPA, 1994a,b; Smith et al., 1995
Pasture	0.012	2.6E-05	mg/kg	Calculated based on modeled air concentrations[d].	EPA, 1994a,b; Smith et al., 1995

NA - Not applicable; TCDD exposure not complete for these pathways.

Note: Concentrations may include releases from sources other than the ORR.

a. Total concentration of PCDF expressed in units of TCDD equivalents. TCDD equivalents are estimates of concentrations of 2,3,7,8-TCDD that would offer risks similar to the PCDF.

b. See appendix C.

c. Includes the EPC used to evaluate drinking water in the Clinch River and Watts Bar Reservoir.

d. Calculated based on the equations presented in Section 6.4.8.

6.4.2 Sediment

Sediment exposures were evaluated for EFPC, Watts Bar Reservoir, Poplar Creek, and the Clinch River. Three studies were evaluated regarding sediment concentrations in EFPC (TVA, 1983b,1985; SAIC, 1994). In general, reported concentrations were below 1 mg/kg for individual Aroclors, with a total PCB concentration reported at 1.74 mg/kg, based on a sample collected between the confluence with Poplar Creek and RM 5 (TVA, 1983b). However, because TVA (1985) reported maximum concentrations of 4 mg/kg Aroclor 1260 and 2 mg/kg for Aroclor 1254, an estimated EPC of 6 mg/kg was selected as a maximum instead for EFPC, based on the sum of these concentrations.

In Poplar Creek, the majority of sediment data had no detectable level of PCBs (Hoffman et al., 1991; Suter, 1991; Cook et al., 1992). One study (TVA, 1983b) reported a total PCB concentration of 4.29 mg/kg collected between RM 3 and RM 6. This area approximately corresponded with the area near Blair Road which had been identified as an access point for recreational fishing (Mills et al., 1995). Consequently, a value of 4.3 mg/kg was selected as the EPC for Poplar Creek.

To determine a sediment EPC for the Clinch River, data pertaining to the section of the River from its mouth to Melton Hill Dam were considered (Hoffman et al., 1991; Suter, 1991; Cook et al., 1992; ESD, 1995). Aroclors 1260 and 1254 predominated, although additional Aroclors were detected in some samples. A conservative EPC of 14.3 mg/kg was selected, based on the sum of the maximum concentrations reported for Aroclors 1260 (5.7 mg/kg), 1254 (5.7 mg/kg) and 1248 (2.9 mg/kg).

Limited sediment data were available for Watts Bar Reservoir, and all reported concentrations were below detection limits (Suter, 1991; Cook et al., 1992). Because a detection limit of 0.1 mg/kg was reported for most samples, this value was selected as the EPC for sediment in this water body.

6.4.3 Surface Water

Exposure point concentrations for surface water were calculated for EFPC, Clinch River, Poplar Creek, and Watts Bar Reservoir based on limited data. In Watts Bar Reservoir and EFPC, all reported concentrations were below detection limits that ranged from 0.1 µg/L to 0.5 µg/L. Typically, the higher detection limits were associated with historical studies and were, therefore, likely reflective of less accurate analytical techniques rather than actual concentrations. For this reason, a value of 0.1 µg/L was applied to all non-detect samples; this value represented the most reasonable choice. As a result, EPCs for EFPC and Watts Bar Reservoir were assigned a value of 0.1 µg/L. This EPC was used for both the incidental ingestion and drinking water exposures evaluated for Watts Bar.

While detected concentrations were reported for the Clinch River (Kornegay, 1992) and Poplar Creek (Suter, 1991), those samples in which PCBs were detected typically were either associated with spill events or were collected in the vicinity of discharge areas. These samples were, therefore, not considered representative of actual off-site concentrations. As a result, the EPCs for Poplar Creek and the Clinch River were also assigned a value of 0.1 µg/L, representing the most reasonable detection limit, as discussed above. This value was used for both the incidental ingestion pathway and the drinking water pathway evaluated for the Clinch River.

6.4.4 Waste Oil

The project team evaluated potential exposures to waste oils sold to the general public by investigating a scenario where PCB-contaminated oils were assumed to be regularly applied to a road directly in front of a residence, resulting in PCB contamination of the soil. Adults and children were assumed to contact soil when walking or playing barefoot along the road. The results of such a scenario were believed to provide a reasonable upper bound on the likely potential for exposures to PCBs associated with the use of waste oil.

In general, the scenario was based on an evaluation conducted for the Wide Beach Superfund Site (EPA, 1985b) in which PCB concentrations were measured following application of PCB-contaminated oils for the purpose of dust control. Because of the relatively small volume of use, high value, and recycling potential of PCBs, it is unlikely that commercial PCB mixtures, such as transformer fluids or specialty cutting fluids, were sold as waste oils at the ORR. Rather, it is likely that the ORR sold hydraulic fluids or other oil types that would have contained only minimal amounts of PCBs. However, because transformer oils containing up to 500 ppm PCBs occasionally could have been included in the waste oils, 300 ppm was selected as an upper bound estimate of the possible PCB concentration in waste oil. Based on this PCB concentration, the project team modeled a soil EPC of 6.9 mg/kg taking into consideration the likely application rate and the mixing with soil. The assumptions used in deriving this EPC are recorded in Table 6-5.

6.4.5 Fish

Fish tissue concentrations in the vicinity of the ORR have been investigated by a number of researchers. As a result, a substantial amount of fish tissue data have been collected and analyzed for various chemical contaminants, including PCBs. Some studies reported data for individual fish, while others reported only summary statistics (i.e., minimum, maximum, average) by species. Due to the volume of data available, as well as inconsistencies among data sets pertaining to data quality, EPCs for fish tissue were based primarily on summary statistics rather than data for individual fish. Most studies reported total PCB concentrations; when total concentrations were not available, estimated total concentrations were derived by summing the reported values for the individual Aroclors, as described above. As previously discussed, the project team identified the maximum total PCB concentration, regardless of species, as the EPC for each water body. It should be noted that based on the available data, PCB levels in fish were relatively constant across species for the water bodies and time period evaluated.

The project team considered only those studies that evaluated fish collected from areas where exposures to off-site receptors were likely to have occurred. These areas included EFPC, from the mouth of the creek to the vicinity of New Hope Pond and Lake Reality; Poplar Creek from the mouth to just above the confluence with EFPC; the Clinch River downstream of Melton Hill Dam; and the Tennessee River arm of the Watts Bar Reservoir, extending from Watts Bar Dam

Table 6-5. Derivation of the Exposure Point Concentration (EPC) for PCBs in Soils Under the Oiled Road Scenario (µg/g)

$$\text{EPC} = \frac{\text{Mass of PCB } (\mu g)}{\text{Mass of Oil (g)} + \text{Mass of Soil (g)}}$$

Where:

Mass of Soil (g) = Soil volume (cm^3) x Soil density (g/cm^3)

Mass of Oil (g) = Oil volume (cm^3) x Oil density (g/cm^3)

Mass of PCB (µg) = PCB concentration (µg/g) x Oil mass (g)

Soil volume (cm^3) = Length of road (cm) x Width of road (cm) x Depth of PCB layer (cm)

Oil mass (g) = Oil volume (cm^3) x Oil density (g/cm^3)

Parameter	Value	Units	Reference
Length of road	170,000	cm	EPA, 1985b
Width of road	457.2	cm	Site specific
Depth of PCB contaminated layer	5.08	cm	Site specific
Soil bulk density	1.5	g/cm^3	EPA, 1993
Volume of oil applied per year	1.55×10^7	cm^3	EPA, 1985b
Oil density	0.9	g/cm^3	Genium Publishing Co., 1991
PCB concentration	300	µg/g	Site specific
Oil mass	1.395×10^7	g	EPA, 1985b; Genium Publishing Co., 1991
EPC	6.9	µg/g or mg/kg	

(RM 530) to the vicinity of the confluence with the Clinch River (RM 570). In addition, while PCBs have been measured in a number of fish species found in these water bodies, the project team only considered those species described by the Tennessee Angler Guide (TWRA, 1992) as typically targeted by commercial or recreational anglers and reported to be consumed at least occasionally.

For EFPC, the most data were available for redbreast sunfish, bluegill, and carp while limited data were available for channel catfish, carp, largemouth bass, northern hogsucker, rock bass, and spotted sucker. Maximum total PCB concentrations in all species ranged from 0.07 mg/kg to 3.7 mg/kg (TVA, 1986). Based on the available data, an EPC of 3.7 mg/kg was selected for EFPC.

In Poplar Creek, bluegill, channel catfish and gizzard shad were the most commonly sampled species, although limited data were also available for white bass, largemouth bass, white crappie, longnose gar, freshwater drum, spotted gar, sunfishes, and spotted sucker. The maximum PCB concentration reported was 8.5 mg/kg in longnose gar (Loar et al., 1981). According to the Tennessee Department of Fisheries (pers. communication, 8/96), longnose gar is not frequently consumed, however there may be some individuals that occasionally catch it. Therefore, this value was conservatively selected as the fish tissue EPC for Poplar Creek.

More than 20 studies of fish were conducted on the Clinch River. Channel catfish and bluegill were the most commonly sampled species, although limited data were also available for carp, gizzard shad, striped bass, white bass, largemouth bass, sauger, sunfishes, other bass, and crappie. The estimated maximum total PCB concentrations ranged from 0.02 mg/kg to 12 mg/kg (MMES, 1985). Based on these data, a maximum EPC of 12 mg/kg was selected for the Clinch River.

In general, channel catfish has been the most commonly sampled species from Watts Bar Reservoir, although data were also available for largemouth bass, sauger, striped bass, white bass, white crappie, black crappie, blue catfish, carp, shad, and smallmouth buffalo. Total PCB concentrations in these species ranged from not detected to 7.5 mg/kg. The maximum detected concentration was conservatively used as the EPC in fish tissue for the analysis of Watts Bar.

6.4.6 Turtles

Turtle tissue concentrations in the vicinity of the ORR were recently reported by the Tennessee Department of Environment and Conservation DOE Oversight Division (Van Audenhove, 1997). PCB concentrations in turtles from each of the water bodies were measured. For the purpose of this analysis, the project team selected the maximum value for each water body as the EPC. These EPCs are presented in Table 6-4.

6.4.7 Beef and Milk

Exposure point concentrations for beef and milk were modeled indirectly from the exposure point concentrations derived for soil, surface water, and air. As described in Section 6.2, the biotransfer factors accounted for the fraction of PCBs that may have been bioaccumulated by cattle following exposure via ingestion of soil, water, vegetation, and inhalation of air. Actual EPCs for beef and milk were not directly calculated.

6.4.8 Vegetables/Pasture

Exposure point concentrations in vegetables produced for human consumption, as well as vegetation consumed by beef and dairy cattle (i.e., pasture) were based on the EPCs derived for soil and air. Biotransfer factors (Iwata and Gunther, 1976; Moza et al., 1979; O'Connor et al., 1990) were applied to soil concentrations to estimate concentrations in vegetables grown in contaminated soil.

The concentration of PCDF in plants resulting from vapor air releases was derived based on a recent study by Smith et al. (1995). This approach, based on the premise that a gas behaves as very small particles, estimates the resulting concentration in plants according to the following equation:

$$C_p = \frac{C_{pa} \ x \, V_d \ x \, (1 - e^{-kw(t)})}{CF \ x \, kw \, x \, Y_j}$$

Where:

C_p	=	Concentration of PCDF in plant (mg/kg dry weight)
C_{pa}	=	Concentration of PCDF in vapor phase (mg/m)
V_d	=	Deposition velocity (m/d)
kw	=	Photodegradation constant (1/d)
t	=	Time to harvest (d)
Y_j	=	Crop density (kg/m^2)
CF	=	Conversion factor (1000 µg/mg)

This model was derived for predicting TCDD levels in pasture. As discussed by EPA, the lower surface to volume ratios of vegetables require a correction factor (EPA 1994a,b). Following agency guidance, the estimates of TCDD in vegetables is assumed to be 1/100 of the level predicted by this equation for pasture.

The concentration of PCDFs (calculated as TCDD equivalents) in vegetables and pasture was determined to be 2.6 x 10^{-7} mg/kg and 2.6 x 10^{-5} mg/kg, respectively, using the maximum vapor concentration of PCDFs that is anticipated to occur at a local farm.

Smith et al. (1995) only evaluated dioxin and furan compounds. In the absence of verification that their approach is applicable to PCBs, the default EPA (1994a,b) approach for estimating the air to plant transfer from vapor releases was used to derive the concentration of PCBs in plants (mg/kg)

$$C_p = \frac{C_v \ x \, B_v \ x \, Vg_{ag}}{p_a}$$

Where:

C_p	=	Concentration of PCB in plant (mg/kg dry weight)
C_v	=	Concentration of PCB in vapor phase (mg/m^3)
B_v	=	Air to plant transfer factor ([mg_{PCB}/ kg_{plant}] / [μg_{PCB} / g_{air}])
Vg_{ag}	=	Correction factor for transfer into bulky vegetation rather than leaves (unitless)
p_a	=	Density of air (g/m^3)

Based on this equation, the concentrations of PCBs in vegetables and pasture were determined to be 1.2 x 10^{-4} mg/kg and 1.2 x 10^{-2} mg/kg, respectively, using the concentration of PCBs in vapor calculated using the SCREEN3 air dispersion model of the farm receiving the highest air levels of PCBs.

6.4.9 Air

The project team estimated ambient air concentrations of PCBs and PCDFs at receptor locations outside of the ORR using SCREEN3**,** a screening-level gaussian air dispersion model (EPA, 1995) as described in Appendix C. The SCREEN3 model simulates 52 different meteorological conditions covering six atmospheric stability classes and twelve wind speeds. The model determines which meteorological condition would result in maximum downwind impacts, and calculates one-hour average concentrations for user-defined distances (EPA, 1995). One-hour average ambient air concentrations are converted to annual averages by multiplying the one-hour average value by 0.08 (EPA, 1992). The results of the SCREEN3 modeling are likely to overestimate the airborne concentrations at the exposure points, providing a conservative exposure point concentration.

For the purpose of this analysis, the project team evaluated releases from four sources at the ORR: the Y-12 Burn Pit, Y-12 Burn Tank, the K-25 Burn Pit, and the TSCA Incinerator (Appendix C). Table C-1 in Appendix C presents the predicted long-term concentrations for a residential receptor (direct exposures) and a farm receptor (indirect exposure). As the table indicates, the highest exposure predicted to occur for a residential receptor (Scarboro) is 4.1 x 10-^{6} mg/m^3 for PCBs and 3.6 x 10^{-9} mg/m^3 of PCDFs (calculated as TCDD equivalents). The highest concentrations that occur at a farm (EFPC) are 3.3 x 10^{-6} mg/m^3 for PCBs and 2.9 x 10^{-9} mg/m^3 of PCDF (calculated as TCDD equivalents).

6.5 Screening Estimates of Risks

Carcinogenic and noncarcinogenic human health risks associated with exposures to PCBs were evaluated for both adults and children living near the ORR. Exposures (LADDs and ADDs) were based on the exposure parameter values and EPCs described above. Carcinogenic risks for each pathway were calculated by multiplying the LADD by the recently revised cancer slope factor (CSF) for PCBs of 2.0 (mg/kg-d)$^{-1}$ (EPA, 1996a). Noncarcinogenic risks were calculated by dividing the ADD by EPA's reference dose (RfD) of 2 x 10^{-5} mg/kg-d for Aroclor 1254 (IRIS, 1998b). Historically, most dioxin risk assessments have used a CSF of 156,000 (mg/kg-day)$^{-1}$ based on a two-year oncogenicity study of tetrachlorodibenzodioxin (TCDD) in female rats conducted by Kociba et al. (1978). In 1990, the rat liver slides from the Kociba et al. (1978)

study were re-evaluated by the Pathology Working Group (PWG) using new criteria for classifying proliferative rat liver lesions (PWG, 1990a,b). Using a cross-species scaling approach based on body weight combined with the results of the PWG reanalysis, U.S. Food and Drug Administration (FDA) derived a CSF for TCDD of 9,000 $(mg/kg\text{-}day)^{-1}$. This CSF was used is this analysis to estimate carcinogenic risks for TCDD.

6.5.1 Risk Results

The results of the risk assessment were evaluated based on the following screening criteria. Pathways were set aside from further consideration if the noncarcinogenic nominal hazard quotient was less or equal to 1.0 and the carcinogenic risk was less or equal to 1×10^{-4}.

As presented in Table 6-6, values for all air-related pathways (except milk consumption), pathways associated with exposures to waste oil, dermal contact with sediment, incidental ingestion of sediment (Poplar Creek, Clinch River, Watts Bar), ingestion of drinking water, and dermal contact and ingestion of surface water were below the screening criteria and those pathways were set aside from further evaluation. Screening values for pathways that exceeded the criteria are presented in Table 6-7. These pathways included dermal contact and incidental ingestion of flood plain soil, incidental ingestion of sediment from EFPC, consumption of vegetables, beef, milk from farms adjacent to EFPC, consumption of fish and turtles from EFPC, and ingestion of fish and turtles from Poplar Creek, Clinch River, and Watts Bar. It was concluded that these pathways represented the pathways most likely to result in risks to off-site populations and these pathways were evaluated in the level II evaluation presented in Section 7.0. We evaluated ingestion of turtle meat only rather than turtle meat and fat. A consumption rate for turtle fat was unavailable. Also, because the risk estimate for consumption of turtle meat exceeded the screening criteria, the pathway was carried forward. Thus, inclusion of turtle fat would not have impacted the outcome.

6.6 IDENTIFICATION OF EXPOSED POPULATIONS

Using site-specific information pertaining to historical activities and the pathways identified, the project team identified five off-site populations potentially exposed via the pathways identified during the level I evaluation: farm families that raised beef, dairy cattle, and vegetables on the flood plain of EFPC; local individuals who may have purchased beef and milk from cattle raised on farms in the EFPC flood plain; commercial and recreational anglers; individuals that may have consumed turtles; and recreational users who used surface water for recreational activities. The existence of each of these populations along the relevant water bodies was evaluated.

Land use and demographic information support the existence of farms along EFPC throughout the time period of the ORR’s operation (Anonymous, 1995a,b; Brooks, 1995; Clark, 1995; Sturm, 1995; DaMassa, 1996a,b; Huber, 1996; Sutton, 1996; Waller, 1996). Therefore, the farm family population was evaluated for this water body. Due to farming and residential activities, individuals within the farm families likely were exposed to PCBs via several of the relevant pathways. In addition to direct contact with the flood plain soil and sediment, the farmer and family members may have consumed contaminated homegrown vegetables, milk, and beef.

Table 6-6. Pathways Set Aside from Further Evaluation Based on Screening Estimates of Risk[a]

Location and Pathway	Hazard Quotient Adult	Hazard Quotient Child	Carcinogenic Risk Adult	Carcinogenic Risk Child
EFPC				
Direct				
Dermal contact with sediment	0.7	1	2×10^{-5}	4×10^{-6}
Dermal contact with surface water	0.005	0.009	1×10^{-7}	2×10^{-8}
Incidental ingestion of surface water	0.02	0.09	5×10^{-7}	2×10^{-7}
Inhalation of dust	0.000001	0.000004	4×10^{-11}	1×10^{-11}
Indirect				
Ingestion of beef from cattle exposed to contaminated air[b]	0.002	0.003	3×10^{-7}	3×10^{-7}
Ingestion of beef from cattle exposed to pasture contaminated via direct deposition[b]	0.3	0.6	7×10^{-6}	7×10^{-6}
Ingestion of beef from cattle exposed to contaminated water	0.02	0.03	4×10^{-7}	9×10^{-8}
Ingestion of milk from cows exposed to contaminated air[b]	0.003	0.02	7×10^{-7}	2×10^{-6}
Ingestion of milk from cows exposed to contaminated water	0.05	0.21	1×10^{-6}	6×10^{-7}
Ingestion of vegetables contaminated via direct deposition[b]	0.02	0.02	1×10^{-7}	5×10^{-8}
Scarboro				
Inhalation of vapor[b]	0.04	0.1	1×10^{-6}	1×10^{-6}
Poplar Creek				
Dermal contact with sediment	0.08	0.1	2×10^{-6}	4×10^{-7}
Dermal contact with surface water	0.2	0.3	4×10^{-6}	9×10^{-7}
Incidental ingestion of sediment	0.04	0.5	1×10^{-6}	1×10^{-6}
Incidental ingestion of surface water	0.002	0.009	5×10^{-8}	2×10^{-8}
Clinch River				
Dermal contact with sediment	0.3	0.5	6×10^{-6}	1×10^{-6}
Dermal contact with surface water	0.05	0.08	1×10^{-6}	2×10^{-7}
Incidental ingestion of sediment	0.2	1	4×10^{-6}	5×10^{-6}
Incidental ingestion of surface water	0.0002	0.0008	5×10^{-9}	2×10^{-9}
Ingestion of drinking water	0.08	0.2	2×10^{-6}	6×10^{-7}
Inhalation of vapor	0.005	0.01	3×10^{-8}	3×10^{-8}
Watts Bar				
Dermal contact with sediment	0.01	0.02	3×10^{-7}	7×10^{-8}
Dermal contact with surface water	0.05	0.08	1×10^{-6}	2×10^{-7}
Incidental ingestion of sediment	0.007	0.08	2×10^{-7}	2×10^{-7}
Incidental ingestion of surface water	0.0002	0.0008	5×10^{-9}	2×10^{-9}
Ingestion of drinking water	0.08	0.2	2×10^{-6}	6×10^{-7}
Sale of Waste Oil				
Dermal contact with soil	0.3	0.6	4×10^{-6}	2×10^{-6}
Ingestion of soil	0.2	1	2×10^{-6}	6×10^{-6}
Inhalation of dust	8×10^{-8}	2×10^{-7}	9×10^{-13}	6×10^{-13}

Note: Fraction of estimated risks may be attributable to sources of PCBs other than the ORR.

a. Pathways with a cancer risk less than or equal to 1×10^{-4} or a hazard quotient less than or equal to 1.

b. For both PCBs and PCDFs formed during the burning of PCB contaminated waste oil.

Table 6-7. Pathways Retained for Level II Evaluation[a]

Location and Pathway	Hazard Quotient		Carcinogenic Risk	
	Adult	Child	Adult	Child
EFPC				
Direct				
Incidental ingestion of sediment	0.4	5	1×10^{-5}	1×10^{-5}
Incidental ingestion of soil	1	14	3×10^{-5}	4×10^{-5}
Dermal contact with soil	2	4	6×10^{-5}	1×10^{-5}
Indirect				
Ingestion of fish	79	347	2×10^{-3}	10×10^{-4}
Ingestion of beef from cattle exposed to contaminated soil	27	59	7×10^{-4}	2×10^{-4}
Ingestion of beef from cattle exposed to pasture grown in contaminated soil	17	38	4×10^{-4}	1×10^{-4}
Ingestion of milk from cows exposed to contaminated soil	72	316	2×10^{-3}	9×10^{-4}
Ingestion of milk from cows consuming pasture grown in contaminated soil	46	202	1×10^{-3}	6×10^{-4}
Ingestion of milk from cows consuming pasture contaminated by air emissions of PCBs and PCDFs	0.7	3.0	4×10^{-5}	8×10^{-5}
Ingestion of vegetables grown in contaminated soil	154	540	4×10^{-3}	1×10^{-3}
Poplar Creek				
Ingestion of fish	182	797	5×10^{-3}	2×10^{-3}
Clinch River				
Ingestion of fish	257	1130	7×10^{-3}	3×10^{-3}
Watts Bar				
Ingestion of fish	161	703	4×10^{-3}	2×10^{-3}

Notes:

1. Fraction of estimated risks may be attributable to sources of PCBs other than the ORR.
2. Screening values for ingestion of turtles exceeded criteria, but the pathway was not retained for further analysis (see Section 6.6 for rationale).

a. Pathways with a cancer risk greater than 1×10^{-4} or a hazard quotient greater than 1.

The project team also considered the possibility that milk and beef from cattle raised along EFPC may have been sold to local residents, providing a route of exposure to those individuals. However, additional investigations have indicated that farms along EFPC did not produce beef and milk for local consumption. Rather interviews with current and past residents have indicated that all beef or milk produced was either consumed by the farm families or sold to distributors outside of the Oak Ridge area (DaMassa, 1996b). It was assumed that beef and milk sold to regional distributors would be dispersed across a wide population and would not pose a significant risk to any single individual. Therefore, the population of beef and milk consumers was deferred from additional analyses.

The assessment of exposure to PCB releases resulted in noncancer risks to children for the indirect pathway of air to pasture to dairy products (nominal hazard quotient of three). However, this assessment was based on a number of conservative assumptions that likely overestimated the risks. These assumptions included the assumption that two ridges between the Y-12 Burn Tank and the farms did not exist. The elevated risks were limited to the farms located along EFPC. The analysis did not predict risks above the risk criterion for the farms near Union /Lawnville (across the Clinch River). Because the risks exceeded the criterion only for the farms along EFPC, and because the contamination of milk from PCBs via soils to pasture to dairy products is higher than the levels from air deposition, noncancer risks to children from PCBs in milk via air deposition were not considered in later analyses.

Populations of commercial and recreational anglers were identified for Watts Bar. Watts Bar was used as a commercial fishery and supported a small number of full-time commercial anglers. Watts Bar has also been used by recreational anglers since it was impounded. Available data indicate that this reservoir as well as other reservoirs on the Tennessee and Clinch River have always been popular recreational fisheries.

Populations of commercial and recreational anglers were also identified for the Clinch River. While the Clinch River likely was not commercially fished to any great degree, due to the limited access for large boats and the proximity of a better quality fishery on Watts Bar, commercial fishing may have occurred and commercial anglers were identified as an exposed population. Recreational fishing was also known to occur on the Clinch River and the lower portion of Poplar Creek. Because it was difficult to distinguish between the angler populations for the Clinch River and Poplar Creek, the angler populations for both water bodies were considered together.

A recreational angler population was associated with EFPC given information regarding fishing activity in the area. Although the level of fishing activity was likely low due to limited access, low productivity of the creek, and the availability of higher quality fisheries nearby, it is likely that some individuals fished the creek on an infrequent basis.

The screening assessment of the consumption of turtles indicated that this exposure route could have resulted in risks above the decision criteria. However, the consumption rate for turtle meat used in the level I evaluation was a very conservative estimate, based on data from a commercial turtle meat distributor. Site-specific information indicated turtles were eaten infrequently, and that the actual consumption rate would likely have been much lower than that used. In addition, there are little data on PCB concentrations in turtles from EFPC. Because of the lack of information on the consumption rate or frequency of this practice, it is not feasible to perform additional quantitative analyses of the uncertainty in the doses and associated risks or to determine the size of the population. Recommendations for collecting additional data pertaining to this pathway are discussed in Section 10.0.

Anecdotal reports suggest that individuals, particularly children living in nearby neighborhoods, occasionally played in EFPC and would have been in direct contact with the sediment and flood plain soil. Therefore, the recreational user population was also evaluated.

6.7 ESTIMATION OF POPULATION SIZES

While population sizes are often used to calculate population risks (excess cases of cancer in a given population), for this analysis population sizes were developed for the purpose of determining whether an epidemiologic study could successfully detect an elevation in disease occurrence rates within each population. Unfortunately, population sizes have been difficult to calculate for many of the off-site populations exposed historically to PCBs at the ORR. In many cases, there were very limited data on the size and turnover rates for the populations in the early years of the ORR operations. When evaluating population sizes, the project team differentiated between the total number of individuals in each population over time, and the number in the population at any given time. Typically, the total population was estimated by estimating the size of the population at any given time, determining an annual population turnover rate, and then multiplying by the number of years that the population was exposed. For example, while an elementary school may have only 200 students at any one time, it might reasonably be expected that each year one-sixth of the population changed due to children moving on to another school. Thus, over a 30-year period, a total of five separate sets of children would have used the school, resulting in a total population of 1,000. In this assessment, the project team used a similar methodology to estimate the total size of each potentially exposed population over 50 years.

6.7.1 Population Size for Farm Families

An accurate estimate of population size for farm families was difficult to make. At best, the data suggested the presence of approximately ten farms in the EFPC flood plain. Based on this limited number of farms, the number of individuals exposed over the history of the ORR activities was estimated to range from 30 to 50.

6.7.2 Anglers

Based on historical and current information on fishing behavior and activities, the project team identified two populations of anglers: commercial and recreational.

6.7.2.1 Commercial Anglers

<u>Watts Bar</u>

The number of full-time commercial anglers fishing Watts Bar Reservoir is very small. Although there are no records of the numbers of full-time commercial anglers who might have fished Watts Bar before the 1960s, Hargis (1968) reported that in 1967, there were a total of seven full-time commercial anglers in Rhea, Meigs, Roane, Anderson, and Loudon Counties, combined. Todd (1990) reported that there were four full-time commercial anglers using Watts Bar in 1989. Other sources indicate that the numbers of commercial anglers fishing the TVA reservoirs in the eastern portion of Tennessee were very small (Hargis, 1968; Morgan and Hubert, 1974; Hubert et al., 1975). Because commercial fishing activity may have been affected by the advisories that were issued in the 1980s, it is reasonable to assume that the numbers reported by Hargis (1968) may have been representative of commercial fishing activity prior to the advisories and that the numbers reported by Todd (1980) may have been representative after the issuance of advisories. If it is conservatively assumed that there were a total of seven full-time commercial anglers fishing Watts Bar in a given year, and that each year one angler stopped activity and another commenced activity, the resulting estimate of the total commercial angler population potentially exposed between 1945 and 1995 may have been as large as 57 anglers and their families. Assuming an average household size of 3.1 individuals (average of household sizes between 1960 and 1990) results in an estimate of 180 as the total number of individuals in this population over the duration of historical ORR operations.

<u>Clinch River/Poplar Creek</u>

It is very likely that the size of the full-time commercial angler population using Clinch River/Poplar Creek is extremely small. As discussed previously, Todd (1990) reported that only 20 percent of commercial anglers fished rivers. If this percentage is applied to the seven anglers estimated for Watts Bar Reservoir, the resulting estimate is that there may be one commercial angler using Clinch River/Poplar Creek in a given year. If it is conservatively assumed that every seven years another angler began to fish the area, the resulting angler population size estimate would be eight individuals between 1945 and 1995. Assuming 3.1 individuals in the typical angler household results in an estimated population size of 25 individuals for the total number of commercial anglers and household members who consumed fish from Clinch River/Poplar Creek during the operation at the ORR.

6.7.2.2 Recreational Anglers

<u>Watts Bar</u>

There are no data available to estimate the actual population size for recreational anglers using Watts Bar. While data are available on the small number of part-time commercial anglers who use the reservoir (Hubert et al., 1975; Todd, 1990), there are no reported estimates of the numbers of sport-licensed anglers. This is due to two factors. First, fisheries managers are generally not concerned with the number of anglers using a resource but rather are interested in the total amount of effort expended, regardless of the number of individuals exerting that effort.

Thus, they do not generally collect data on the number of individual anglers who use a given fishery. Second, historical licensing records are not available. While recent license records can provide information on the total number of individuals who purchased sport licenses in Tennessee in a given year, these data are not available on either a county-specific or waterbody-specific basis. Thus, they can not provide information on the specific water bodies fished by licensed anglers and are, therefore, not useful in predicting the number of anglers using a specific water body during a given year.

The only way in which population size estimates can reasonably be made is to apportion the level of effort (total trips) over an estimate of the number of trips that the average angler might take in a year in order to estimate the population size. TWRA (Unk.) reported 133,887 trips in 1990 for Watts Bar. According to U.S. Fish and Wildlife statistics for 1990 (USDOI, 1993), Tennessee anglers took an average of 14.6 trips per year to fish lakes and reservoirs. If the total number of trips taken to Watts Bar in 1990 (133,887 trips/year) is divided by 14.6 trips/year-angler, the result is an estimated 9,170 anglers using the reservoir that year.

The total population of Anderson, Loudon, Meigs, Rhea, and Roane Counties (the counties adjacent to Watts Bar) during 1990 was 179,109 individuals. Thus, the estimated number of individuals who fished Watts Bar, 9,170 anglers, represented approximately five percent of the population of those counties. A slightly higher percentage of the local population is estimated if one evaluates the data available for 1980, the previous census year. In that year, TWRA reported 150,698 fishing trips to Watts Bar. Assuming again that anglers who fished Watts Bar averaged 14.6 trips per year, it can be estimated that a total of 10,321 anglers fished Watts Bar that year. When comparing this estimate to the total estimated population for the five counties of interest, 168,780 persons, it appears that Watts Bar anglers represented approximately six percent of that population. This higher percentage of the population was used to estimate population sizes at various points of time, based on census data.

Assuming that six percent of the relevant county-wide populations fished Watts Bar in a given year, the number of anglers who may have fished Watts Bar during each census year can be estimated. As shown in Table 6-8, population sizes for each of the relevant counties have increased steadily since 1950 and the total population of the five counties combined has increased from 136,375 in 1950 to 179,109 in 1990. Applying a factor of 0.06 to the population sizes in 1950 and 1960 results in estimated angler population sizes of 8,183 and 8,637 for those years, respectively. Thus, it appears that the total angler population size during that decade increased by 454 anglers. Similar increases of 115, 1,375, and 620 new anglers, based on general increases in regional population, can be estimated for 1970, 1980, and 1990, respectively. If estimates of the number of new anglers in each of the 10-year census periods are then multiplied by the mean household sizes of the counties of interest of the appropriate decade an estimate can be made of the total population of fish consumers, that is each angler and members of his or her household. The result is an estimated 40,482 individuals who may have consumed recreationally-caught Watts Bar fish over the period of interest.

It is not reasonable, however, to assume that every angler who begins to fish Watts Bar in a given year will continue to fish it every year thereafter. Anglers may die, move away, or cease fishing for a number of reasons. Thus, the above estimate does not likely provide an accurate

Table 6-8. Estimate of Population Size for Watts Bar Anglers and Their Families

Year	Total Population[a]	Fraction Who Fished Watts Bar[b]	Estimated Number of Anglers	Number of New Anglers	Mean Household Size[c]	Total Exposed
1950	136,375	0.06	8,183	8,183	4.0	32,882
1960	143,945	0.06	8,637	454	3.8	1,703
1970	145,868	0.06	8,752	115	3.2	368
1980	168,780	0.06	10,127	1,375	2.8	3,905
1990	179,109	0.06	10,747	620	2.6	1,624
					Total	40,482

a. The total population of Anderson, Loudin, Meigs, Rhea, and Roane Counties.

b. Based on ratio of the estimated number of Watts Bar anglers in 1980 and the total population of Anderson, Loudin, Meigs, Rhea, and Roane Counties in that year.

c. Based on census data for appropriate counties in that year.

picture of the total number of individuals who may have consumed recreationally-obtained Watts Bar fish since 1945. Rather, it is appropriate that there is a certain level of turnover in the angler population and that anglers who have ceased their angling activities are replaced by other anglers, so that the actual number of anglers who used Watts Bar over time is substantially larger than the above estimate.

Watts Bar is a large fishery that is accessible from several counties. Thus, even if anglers moved from one county to another, they may have continued to fish Watts Bar. As a result, their duration of fishing effort may have been substantially longer than occurs on smaller, localized fisheries. As a conservative measure, the residence times reported by Israeli and Nelson (1992) for farm families have been doubled to reflect the lower rate of inter-regional mobility to generate a distribution of mobility rates for this population. After truncating the distribution at a reasonable maximum of 75 years, the distribution results in a mean exposure duration of 31 years. Thus, it can be assumed that in any given year, 1/31 of the population may turn over. If it is assumed that 1/31 of the populations estimated (including new additions in each ten-year period) turn over, it can be estimated that approximately 84,300 individuals may have consumed recreationally-caught fish from Watts Bar between 1945 and 1995 (Table 6-9).

<u>Clinch River/Poplar Creek</u>

There are no data available to provide estimates of the number of anglers who may have used Clinch River/Poplar Creek as a fishery. U.S. Fish and Wildlife (USDOI, 1993) data for Tennessee indicate that a total of 479,600 state residents fished large lakes or reservoirs during 1991. In that same year, 338,300 anglers fished the state's rivers or streams. Based on those data, it appears that the number of anglers who fished rivers and streams was approximately 70 percent of the number of anglers who fished lakes and reservoirs. Applying this percentage to the estimated 84,300 persons consuming recreationally-caught fish from Watts Bar, results in an estimated size of 59,000 persons living in the area who may consume fish caught by river anglers. However, the area of interest on Clinch River/Poplar Creek is very small and has extremely limited access, making it unlikely that such large numbers of individuals would have obtained fish for consumption from this area.

To adjust for this limited size and accessibility, an adjustment factor was developed based on the relative surface areas of the Watts Bar and Clinch River/Poplar Creek fisheries. According to Hoffman et al. (1991), Watts Bar Reservoir encompasses an area that is 38,600 acres in size. There are approximately 22.3 river miles on the Clinch River between Melton Hill Dam and the beginning of Watts Bar Reservoir. If it is assumed that the Clinch River averages 200 feet in width over its length, the result is 540 acres of surface water. Thus, the surface area of the affected reaches of Clinch River/Poplar Creek is approximately 1.4 percent of the size of Watts Bar. If the number of individuals who consume fish from a given area is the same in the Clinch River and in Watts Bar, the number of anglers would be proportionately smaller than the number of individuals consuming fish from Watts Bar. Applying 0.88 percent to the estimated population size for Watts Bar fish consumers, 84,300 persons, results in an estimated population size for Clinch River/Poplar Creek of 1,200 persons. Applying this same percentage to the estimated number of individuals who may have consumed fish from rivers in the area, 62,000 individuals, results in an estimated population size of 900 persons. These estimates may be

Table 6-9. Size of Population Potential Exposed as a Result of Recreational Angling Activity, Based on Population Mobility Rates, Increases in Regional Populations, and Sizes of Angler Families

Year	Population Size[a]	Number of New Anglers Due to Turnover of 1/31 of Population Each Year	New Anglers Due to Population Increase	Total New During 10-year Period[b]	Household Size Multiplier[c]	Total[d] Exposed
1950-1960	8,183	2,640	8,183	10,823	4.0	43,399
1960-1970	8,637	2,786	454	3,240	3.8	12,150
1970-1980	8,752	2,823	115	2,938	3.2	9,402
1980-1990	10,127	3,267	1,375	4,642	2.8	13,183
1990-1995	10,747	1,733	620	2,353	2.6	6,166
						84,300

a. Assumes that 6 percent of the populations of Anderson, Loudon, Meigs, Rhea and Rhone Counties fish Watts Bar in a given year fish Watts Bar.

b. New anglers due to increase in local population and yearly turnover.

c. Based on the average of the census data for mean household sizes for the five-county area in first year of decade.

d. Total individuals exposed at end of decade based on the product of the total angler population for that 10-year period and the average household multiplier for the first year of the 10-year period.

underestimated, because the number of anglers using a body of water is a function of many factors such as access and productivity.

EFPC

Given the size and characteristics of EFPC and its low productivity, it is unlikely that a substantial number of anglers used it as a fishery. Anecdotal information indicates that EFPC has not always supported viable fish populations and, as such, may not have been a useable fishery for a number of the years being evaluated. There are no firm estimates of the number of individuals who may have fished there over time.

To estimate the population size for EFPC recreational anglers, an evaluation of sustainable recreational fish harvest in the publicly accessible portions of EFPC was completed (Appendix B). In that evaluation, data collected between 1987 and 1996 by the Y-12 Plant Biological Monitoring and Abatement program on the species and size-class-specific fish biomass in publicly accessible reaches of the creek were evaluated. Only species and sizes of fish likely to be retained and consumed (bluegill, largemouth bass, rock bass, white sucker, redbreast sunfish, green sunfish, largemouth, red sunfish, and hybrid sunfish greater than five inches in length) were considered. Total biomass of fish available for consumption was calculated for the publicly accessible areas of the creek. It was assumed that 50 percent of the calculated biomass was a reasonable sustainable harvest. This resulted in a maximum sustainable yield for EFPC of 112 kg/year or 33.6 kg/year assuming 30 percent of total mass is edible. Assuming that each angler who fished EFPC consumed fish at the mean rate of 1.2 g/person-day, only 77 individuals could have consumed fish from the creek without exceeding the sustainable yield. Alternatively, if it is assumed that EFPC anglers fished the creek an average of nine times per year and harvested one 227 g meal per trip, the creek could have supported only 16 fish consumers. Thus, it appears likely that the population size for EFPC anglers was less than 100 persons.

6.7.3 Recreational User Population Size

Similar to the farm family, the size of the recreational user population was difficult to estimate. The project team assumed that the adults participating in recreational activities along EFPC resided in the vicinity of the stream, because EFPC was not a recreational destination that would have attracted individuals living any distance away from the creek. Based on this information, the population for adult recreational users was assumed to range from 30 to 100 individuals.

Information was available on the number of school age children who lived in the communities that bordered EFPC. Based on four elementary schools within one mile of EFPC that served 700 pupils each and a junior high that bordered the creek that served 2,000 students, an estimate of 4,100 school-age children lived in the area of the creek (personal communication, C. DaMassa, 4/17/96). It was more difficult to determine the fraction of these children who regularly used the stream for recreational purposes. The attractiveness of EFPC for recreational purposes is limited in many areas due to the varying depth, width, and steepness of the banks. In addition, access would have been limited in some areas due to proximity to private properties and roads. A footpath used by schoolchildren to cross the stream at the Junior High was identified (personal communication, D. Page, 5/20/97); however, the existence of a bridge along the path limits the likelihood of contact with the sediments and surface water of the stream. Based on these considerations, the project team assumed that the children regularly using the stream were those living in close proximity to the stream including those in private homes as well as multiple occupancy and public housing units. The project team further assumed that between 5 and 20 elementary school age children lived near the stream at any one time and would have used it. It was assumed that a new set of children began using the stream every 6 years, resulting in an estimate of approximately 100 to 300 children over the period of the ORR activities.

7.0 LEVEL II EVALUATION OF EXPOSED POPULATIONS BASED ON CONSERVATIVE ESTIMATES OF VARIABILITY IN EXPOSURE

This section presents the level II evaluation of individuals exposed to PCBs at the ORR. The primary goal of the section is to determine whether individuals in "exposed populations" received doses of PCBs that produced risks in excess of the decision criteria. Although our assessment has shifted from focusing on the pathways of exposure (Section 6.0) to the exposed populations, the risk estimates for these populations are based on the total exposure from multiple pathways. In situations where a single pathway resulted in different risks at different locations, we treated the individuals exposed at different locations as separate populations.

The approach used in level II evaluation characterizes the ranges of doses that plausibly occurred in the exposed populations given the available data. Similar to level I evaluation, simple dose rate models were constructed for each pathway. The dose estimates that occurred across populations were determined using a numerical simulation technique called Monte Carlo analysis.

As discussed by many researchers, estimates of dose rates are subject to both variability and uncertainty (Morgan and Henrion, 1990). Uncertainty is the measure of the imprecision in an estimate that occurs as a result of a lack of complete knowledge. Variability is a property of what is being measured; in this case, the interindividual variation in doses rates across individuals in the different populations. Level II evaluation handled uncertainty and variation in two ways. Variability was directly modeled and estimates of the distribution of doses in the population were directly determined. Uncertainty that occurred due to the lack of knowledge was modeled by using estimates of intake parameters that were biased with respect to uncertainty. Specifically, the parameter values were selected from the upper end of the range of plausible alternative values. The evaluation used the same toxicological criteria (reference dose (RfD) and cancer slope factor (CSF)) as used in the level I evaluation. As discussed in Section 4.0, these values overestimate cancer and noncancer risks. Therefore, the estimates of interindividual variation in risk overestimate the true range of risks received by each of the populations.

7.1 METHODOLOGY AND APPROACH

As discussed above, the purpose of this evaluation was to estimate risks to the exposed populations identified in Section 6.0. These populations included recreational anglers, commercial anglers, farm families, and recreational users of surface water bodies near the ORR. The populations and associated exposure pathways evaluated for each water body are summarized in Table 7-1.

The project team used Monte Carlo analysis, a numerical simulation technique that allows any parameter in an equation or model to be represented by a range (distribution) of values, to investigate the uncertainty in risk estimates. During the simulation, the exposure equation was calculated repeatedly, selecting the value for each parameter randomly from the distribution of values for that parameter, in accordance with an assigned probability of occurrence for that particular value. The result was a set of exposure estimates whose range and distribution

Table 7-1. Summary of Populations and Associated Exposure Pathways Evaluated in Total Uncertainty Analysis

Population	Waterway: East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek
Recreational fish consumer[a]			
Ingestion of Fish	**X**	**X**	**X**
Ingestion of Sediment	**X**		
Commercial angler[a]			
Ingestion of Fish		**X**	**X**
Farm Family[b]			
Ingestion of Beef	**X**		
Ingestion of Milk	**X**		
Ingestion of Vegetables	**X**		
Ingestion of Soil	**X**		
Dermal Contact with Soil	**X**		
Recreational User[a]			
Ingestion of Sediment	**X**		
Dermal Contact with Sediment	**X**		

a. Age groups evaluated include adults and children aged 6 to 10 years.

b. Age groups evaluated include adults and children aged 1 to 10 years.

reflected the combined variability in the input parameters (Morgan and Henrion, 1990; Kangas, 1996). The analysis was performed on an Excel (Version 5.0, Microsoft, 1994) spreadsheet using a commercially available software package, @Risk (Version 3.0; Palisade Corp. 1996).

An exposed population warranted additional study if the estimate of the dose rate to the 95th percentile of the cumulative dose rate distribution resulted in cancer and noncancer risks that were above the decision criteria[3]. If we concluded that the risks associated with these doses did not exceed the health criteria then the populations were set aside and no additional assessment was performed. The results of such estimates are presented graphically and in tabular form and are included in the final results of the task as conservative estimates of the range of risks in the exposed populations.

7.1.1 Exposure Parameter Distributions

In performing the Monte Carlo analysis, the project team developed distributions for those exposure parameters believed to make a considerable contribution to the total variation in dose rates of a population. The distributions were developed by fitting the available data to various distribution types (e.g., normal, lognormal) according to accepted methods (Finley et al., 1994; Kangas, 1996). Examples and discussion of the different distribution types are presented in Appendix D. A summary of the exposure parameter values and distributions used for each population, as well as the assumptions and rationale on which they were based, are presented in Tables 7-2 through 7-6.

Distributions presented in the Exposure Factors Handbook (EPA, 1997b) or other standard guidance references were applied to those parameters that are similar across most populations and not likely to be affected by site-specific factors (e.g., body weight). The remaining distributions were developed based on-site- or region-specific data. In general, an effort was made to focus on data representative of rural populations in the southeastern United States during the specific period of interest (i.e., 1945 to 1995). Other factors influencing the selection of the distributions included the specific populations and water bodies being evaluated. For example, distributions for fish consumption rates for the commercial and recreational angler populations along each of the impacted water bodies were determined based on a variety of sources, including regional catch statistics, historical local and regional creel surveys, and general information pertaining to consumption of freshwater fish among rural southeastern populations. As discussed above, when the available data were not adequate to define the distribution of a parameter, we intentionally biased the distribution towards those distributions that would result in higher exposures. A detailed discussion regarding the derivation and basis for each site-specific distribution is provided below.

7.1.2 Exposure Duration

Due to the fact that the ORR has been in existence for approximately 50 years, the maximum possible exposure duration to off-site populations from releases associated with the ORR is also

[3] Because the estimates are intended to overestimate the actual doses, this dose rate probably corresponds to a percentile higher than the 95th percentile and may in fact be higher than the dose received by any individual.

Table 7-2. Exposure Parameter Distributions Associated with All Pathways

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Body weight (kg)									
adult	Cumulative					35	130	Average of values reported for adult men and women.	EPA, 1997
child (1-10)	Cumulative					10	65	Age-specific body weights associated with each percentile were averaged over the age group and gender to develop one distribution representative of the age group range.	EPA, 1997
child (6-10)	Cumulative					18	65	Age-specific body weights associated with each percentile were averaged over the age group and gender to develop one distribution representative of the age group range.	EPA, 1997
Averaging time (days)									
carcinogenic	Point Estimate	25,550						The carcinogenic averaging time was based on an assumed average 70 year lifespan.	EPA, 1989a
noncarcinogenic	Custom	Variable[a]						For non-carcinogenic risks, the pathway-specific period of exposure for each identified age group was averaged over the exposure duration for that age group.	EPA, 1989a

a. Derived according to the following equation: Exposure duration (years) x 365 (days/year)

Table 7-3. Exposure Parameter Distributions for the Commercial Angler

Exposure Parameters	Distribution Type	Point Estimate or Custom	Arithmetic Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Fraction from contaminated source (unitless)	Point Estimate	1						Conservative assumption.	Professional Judgement
Exposure frequency (days/year)	Point Estimate	365						Based on use of an annualized fish consumption rate.	
Exposure duration (years)									
adult	Discrete Uniform[a]	[93, 7] [(1, 8), (35,50)]						Based on historical commercial fishing activity. Assumed that 93% of commercial anglers fished 1 to 8 years and 7% fished 35 to 50 years.	Hargis, 1968; Morgan and Hubert, 1974; Hubert et al., 1975; Todd, 1990
child (6-10)	Triangular			5		1	5	Duration of exposure assumed to range from 1 to 5 years, with a most likely duration of 5 years.	Professional Judgement
Fraction of PCBs remaining after cooking (unitless)	Triangular			0.5		0.4	0.8	Studies indicate range of cooking losses of 20 to 60%.	ChemRisk, 1992; Connelly et al., 1992; Sherer and Price, 1993
Fish consumption rate (g/person-day)									
adult									
Watts Bar Reservoir	Lognormal		24		57			Mean based on regional commercial harvest data. Standard deviation derived by assuming that the coefficient of variation would be similar to that reported for other studies.	Hubert et al., 1975; Ebert et al., 1993; Connelly et al., 1996
Clinch River/Poplar Creek	Lognormal		2.2		5.2			Mean based on Watts Bar data indicating that 9% of regional commercial harvest was from rivers. Standard deviation derived by assuming that the coefficient of variation would be similar to other studies.	Todd, 1990; Ebert et al., 1993; Connelly et al., 1996
child (6-10)									
Watts Bar Reservoir	Lognormal/Custom	Variable[b]						Assumed that children consume 33 percent more fish than adults on a per body weight basis.	Rupp et al., 1980
Clinch River/Poplar Creek	Lognormal/Custom	Variable[b]						Assumed that children consume 33 percent more fish than adults on a per body weight basis.	Rupp et al., 1980

a. Values presented represent the discrete uniform distribution as follows [p_1, p_2], [(min_1, max_1), (min_2, max_2)] where p = probability

b. Derived according to the following equation: [Adult Fish Consumption Rate (g/day) x Child Body Weight (kg) x 1.3] / [70 (kg)]

Table 7-4. Exposure Parameter Distributions for the Recreational Angler

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	References
Common Parameters									
Fraction from contaminated source (unitless)	Point Estimate	1						Conservative Assumption.	Professional Judgement
Exposure duration (years)[a]									
adult									
East Fork Poplar Creek	Cumulative					0.5	55	Assumed to be employees at one of the ORR plants. Duration based on occupational tenure information. Mean and standard deviation were doubled to account for the typically longer tenure observed at the ORR.	Israeli and Nelson, 1992; Finley et al., 1994
Watts Bar Reservoir and								Used twice the mean and standard deviation reported for residential occupancy.	Israeli and Nelson, 1992
Clinch River/Poplar Creek	Cumulative					0	50		
child (6-10)	Triangular			5		1	5	Conservative assumption.	Professional Judgement
Pathway Specific Parameters									
Ingestion of Fish									
Fish consumption rate (g/person-day)									
adult									
East Fork Poplar Creek	Lognormal		1.2		2.9			Recommended ingestion rates for consumption by recreational anglers fishing small ponds or streams. Standard derivation derived as explained for commercial anglers.	Ebert et al., 1993; EPA, 1994b; Connelly et al., 1996
Watts Bar Reservoir	Lognormal		30		71			Based on catch statistics for the years from 1977 to 1991. Assumed that anglers targeted the most harvested species. Standard deviation derived as explained for commercial anglers.	TWRA, UNK; Ebert et al., 1993; Connelly et al., 1996
Clinch River/Poplar Creek	Lognormal		18		43			Consumption rate for Watts Bar Reservoir multiplied by 0.6 (ratio of trips to rivers / trips to lakes for Tennessee anglers). Standard deviation was derived as explained for commercial anglers.	Ebert et al., 1993; USDOI, 1993; Connelly et al., 1996
child (6-10)									
East Fork Poplar Creek	Lognormal/ Custom	Variable[b]						Assumed that children consume 33 percent more fish than adults on a per body weight basis.	Rupp et al., 1980
Watts Bar Reservoir	Lognormal/ Custom	Variable[b]						Assumed that children consume 33 percent more fish than adults on a per body weight basis.	Rupp et al., 1980
Clinch River/Poplar Creek	Lognormal/ Custom	Variable[b]						Assumed that children consume 33 percent more fish than adults on a per body weight basis.	Rupp, 1980

Table 7-4. Exposure Parameter Distributions for the Recreational Angler

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	References
Fraction of PCBs remaining after cooking (unitless)	Triangular			0.5		0.4	0.8	Same basis as described for commercial angler.	ChemRisk, 1992; Connelly et al., 1992; Sherer and Price, 1993
Exposure frequency (days/year)	Point Estimate	365						Based on the use of annualized fish consumption rate.	Professional Judgement
Ingestion of Sediment									
Sediment ingestion rate (mg/day)									
adult	Cumulative					2.5	100	Assumed that adults consume approximately 50% of the sediment ingested by 1-5 year old children.	Stanek and Calabrese, 1992
child (6-10)	Cumulative					2.5	100	Assumed that older children consume approximately 50% of the sediment ingested by 1-5 year old children.	Stanek and Calabrese, 1992
Exposure frequency (days/year)	Custom	Variable[c]						Based on total annual consumption (g/year) divided by an average harvest rate per trip of 400 g/trip (day).	Turcotte, 1983

a. A value of 50 was assigned each time the model selected an exposure duration greater than 50 years.

b. Derived according to the following equation: [Adult Fish Consumption Rate (g/day) x Child Body Weight (kg) x 1.3] / [70 (kg)].

c. Derived according to the following equation: [Fish Consumption Rate (g/day) x 365 (days/year) / 400 (g/trip) where each trip is assumed equal to one day.

Table 7-5. Exposure Parameter Distributions for the Recreational User

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Geometric Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Common Parameters										
Fraction from contaminated source (unitless)	Triangular				0.1		0	1	Assumed that only a limited portion of the time spent participating in recreational activities would occur in the vicinity of EFPC due to its physical characteristics.	Professional Judgement
Exposure frequency (days/year)										
adult	Uniform						20	78	Assumed a minimum of 1 day every 2 weeks during spring, summer, and fall and maximum of 2 days/week during spring, summer, and fall.	Professional Judgement
child	Uniform						26	91	Assumed a minimum of 1 day every 2 weeks throughout the year, and maximum of 2 days/week during spring, summer, and fall and 1 day/week during the winter.	Professional Judgement
Exposure duration (years)[a]										
adult	Cumulative						0.5	55	Assumed to be employees at one of the ORR plants. Duration based on occupational tenure information. Mean and standard deviation doubled to account for the typically longer tenure observed at the ORR.	Finley et al., 1994
child (6-10)	Triangular				5		1	5	Conservative assumption.	Professional Judgement
Pathway Specific Parameters										
Ingestion of Soil										
Soil ingestion rate (mg/day)										
adult	Cumulative						2.5	100	Assumed that adults consume approximately 50% of the soil ingested by 1-5 year old children.	Stanek and Calabrese, 1992
child (6-10)	Cumulative						2.5	100	Assumed that older children consume approximately 50% of the soil ingested by 1-5 year old children.	Stanek and Calabrese, 1992

Table 7-5. Exposure Parameter Distributions for the Recreational User

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Geometric Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Dermal Contact with Soil										
Skin surface area exposed (cm^2)										
adult	Cumulative						1050	17808	Assumed that forearms, hands, lower legs, and feet exposed during spring, summer, and fall; forearms and hands exposed during winter.	EPA, 1997
child (6-10)	Cumulative						304	11524	Assumed that forearms, hands, lower legs, and feet exposed during spring, summer, and fall; forearms and hands exposed during winter.	EPA, 1997
Adherence factor (mg/cm^2-day)	Lognormal			0.24		3.45[b]			Data collected from studies where fingertips were placed in pre-weighed amounts of soil and the difference in mass pre/post was measured.	Duggan and Williams, 1977; Que Hee et al., 1985; Driver et al., 1989; Finley et al., 1994
Dermal bioavailability (unitless)	Triangular				0.032		0.006	0.06	Based on range reported by EPA, adjusted for site-specific total organic carbon content.	EPA, 1992

a. A value of 50 was assigned each time the model selected an exposure duration value greater than 50 years.

b. Geometric standard deviation

Table 7-6. Exposure Parameter Distributions for the Farm Family

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Geometric Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Common Parameters										
Exposure duration (years)[a]										
adult	Cumulative						0.25	70	Based on a residential occupancy period distribution for farms.	Israeli and Nelson, 1992
child (1-10)	Triangular				10		1	10	Conservative assumption.	Professional Judgement
Pathway Specific Parameters										
Milk Consumption										
Fraction from contaminated source (unitless)	Triangular				1		0.04	1	Based on data regarding the average fraction of home produced milk consumed as reported in the National Food Consumption Study (NFCS).	USDA, 1955a,b; 1966; 1983
Milk consumption rate (kg/day)										
adult	Lognormal		0.6			0.908			Based on data reported by the NFCS. Assumed that 85% of the total milk consumed was fresh milk. Standard deviation estimated from data reported by NFCS for 1978.	USDA, 1955a,b;1966; 1983
child (1-10)	Lognormal		0.6			0.908			Assumed to equal ingestion by adults as reported in NFCS.	USDA, 1983
Milk biotransfer factor (days/kg)	Discrete[b]	0.0104; 0.0106; 0.0107; 0.0107; 0.0109; 0.011; 0.0117; 0.0118; 0.0129							Derived based on data presented in Fries et al., 1973.	Fries et al., 1973; Travis and Arms, 1988
Exposure frequency (days/year)	Point Estimate	365							Based on use of annualized milk consumption rates.	Professional Judgement
<u>Via Soil</u>										
Dairy cattle soil ingestion rate (kg/day)	Custom/Uniform						Variable[c]	Variable[d]	Based on pasture providing 2% total feed intake during season of high pasture and 14% total feed intake during times of low pasture.	Fries et al., 1982; Thorton and Abraham, 1982; Zach and Mayoh, 1983; Sumerling et al., 1984
Fraction from contaminated soil (unitless)	Point Estimate	1							Conservative assumption.	Professional Judgement

Table 7-6. Exposure Parameter Distributions for the Farm Family

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Geometric Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Via Soil to Pasture										
Uptake by pasture (unitless)	Uniform						0.0004	0.0005	PCB bioconcentration factors for plants range from 0.04 percent for roots to 0.05 percent for surface vegetation.	Sawhney and Hankin, 1984
Feed consumption rate (kg/day)	Normal		14			2.49			Based on compilation of studies of feed rates of non-commercial dairy cattle.	Shor and Fields, 1980; Boone et al., 1981; Dreicer et al., 1990
Fraction consumption pasture (unitless)	Uniform						0.3	1	Minimum value assumed that 20% of diet was fresh pasture for 40% of the year (low pasture season) and 80% was fresh pasture for 60% of the year (high pasture season). Maximum value assumed 100% fresh pasture throughout the year.	Koranda, 1965
Beef Consumption										
Fraction from contaminated source (unitless)	Triangular				1		0	1	Based on average fraction of home produced beef consumed as reported in NFCS.	USDA, 1955a,b; 1966; 1983
Beef consumption rate (kg/day)										
adult	Lognormal		0.158			0.186			Assumed that beef comprised 40% of the total meat ingestion rate reported by the NFCS. Standard deviation based on 1978 NFCS data.	USDA, 1955a,b; 1966; 1983
child (1-10)	Custom/ Lognormal	Variable[e]							Assumed that children consume the same amount of beef as an adult on a per body weight basis.	Professional Judgement
Beef biotransfer factor (days/kg)	Discrete[b]	0.032; 0.040; 0.043; 0.046; 0.049; 0.049; 0.067; 0.068; 0.075							Based on data presented by Fries et al. (1973).	Fries et al., 1973; Travis and Arms, 1988
Exposure frequency (days/year)	Point Estimate	365							Based on the use of annualized beef consumption rates.	Professional Judgement
Via Soil										
Beef cattle soil ingestion rate (kg/day)	Custom/ Uniform						Variable[c]	Variable[d]	Reported soil ingestion rates for cattle range from a low of 2% total feed intake during season of high pasture, to a high of 14% total feed intake during times of low pasture.	Fries et al., 1982; Thorton and Abraham, 1982; Zach and Mayoh, 1983; Sumerling et al., 1984
Fraction from contaminated soil (unitless)	Point Estimate	1							Conservative assumption.	Professional Judgement

Table 7-6. Exposure Parameter Distributions for the Farm Family

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Geometric Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Via Soil to Pasture										
Uptake by pasture (unitless)	Uniform						0.0004	0.0005	PCB bioconcentration factors for plants range from 0.04 percent for roots to 0.05 percent for surface vegetation.	Sawhney and Hankin, 1984
Feed consumption rate (kg/day)	Truncated Normal		12			4.4	6.1	17.5	Based on a compilation of studies.	McKone, 1988
Fraction consumption pasture (unitless)	Uniform						0.3	1	For the minimum value, it was estimated that fresh pasture comprised 20 % of diet for 40% of the year (low pasture season) and 80 % of diet for 60% of the year (high pasture season). Maximum value assumed 100 % pasture throughout the year.	Koranda, 1965
Vegetable Consumption										
Vegetable consumption rate (kg/day)										
adult	Lognormal		0.495			0.425			Average total vegetable intake rate for a rural southern farmer reported in NFCS. Standard deviation based on 1978 NFCS data.	USDA, 1955a,b; 1966; 1983
child (1-10)	Custom/ Lognormal	Variable[f]							Assumed that children consumed same amount of vegetables as adults on a per body weight basis.	Professional Judgement
Fraction from contaminated source (unitless)	Triangular				0.93		0	1	Based on NFCS data regarding the average fraction of vegetables consumed that were home-produced.	USDA, 1955a,b; 1966
Uptake by vegetables (unitless)	Triangular				0.002		0.0004	0.04	The minimum and maximum represent the bioconcentration factor for roots and carrots respectively. The most likely value is the weighted average based on relative contribution of each plant species to total vegetable intake.	Iwata and Gunther, 1976; Moza et al., 1979; Sawhney and Hankin, 1984; O'Connor et al., 1990
Exposure frequency (days/year)	Point Estimate	365							Based on the use of annualized vegetable consumption rates.	Professional Judgement

Table 7-6. Exposure Parameter Distributions for the Farm Family

Exposure Parameters	Distribution	Point Estimate or Custom	Arithmetic Mean	Geometric Mean	Most Likely	Standard Deviation	Minimum	Maximum	Rationale	Reference
Ingestion of Soil										
Fraction from contaminated source (unitless)	Point Estimate	1							Conservative assumption.	Professional Judgement
Soil ingestion rate (mg/day)										
adult	Cumulative						2.5	100	Assumed that adults would consume approximately 50% of the soil ingested by 1-5 year old children.	Stanek and Calabrese, 1992
child (1-10)	Cumulative						3.8	150	Assumed to be the average of the rate reported for young children (i.e., 1-5 years) for five years and that assumed for adults and older children for five years.	Stanek and Calabrese, 1992; Professional Judgement
Exposure frequency (days/year)	Triangular				260		104	365	Assumed that exposure occurred a minimum of 2 days per week. Maximum assumed 7 days/week. Most likely value assumed to be 5 days/week.	Professional Judgement
Dermal Contact with Soil										
Fraction from contaminated source (unitless)	Point Estimate	1							Conservative assumption.	Professional Judgement
Skin surface area exposed (cm^2)										
adult	Cumulative						1,050	17,808	Assumed exposure to forearms, hands, lower legs, and feet during spring, summer, and fall, and only forearms and hands in winter.	EPA, 1997
child (1-10)	Cumulative						304	11,524	Assumed exposure to forearms, hands, lower legs, and feet during spring, summer, and fall, and only forearms and hands in winter.	EPA, 1997
Exposure frequency (days/year)	Triangular				260		104	365	Assumed exposure ranged from a minumum of 2 days/week to 7 days/week with a most likely value of 5 days/week.	Professional Judgement
Adherence factor (mg/cm^2-day)	Lognormal			0.24		3.45[g]			Data collected from studies where fingertips were placed in pre-weighed amounts of soil and the difference in mass pre/post was measured.	Duggan and Williams, 1977; Que Hee et al., 1985; Driver et al., 1989; Finley et al., 1994.
Dermal bioavailability (unitless)	Triangular				0.032		0.006	0.06	Based on data provided by EPA, adjusted for site-specific total organic carbon content.	EPA, 1992

a. A value of 50 was assigned each time the model selected an exposure duration value greater than 50 years.
b. Values presented have equal probability (p=0.011) of being selected in the model simulation.
c. Derived according to the following equation: Feed Consumption Rate (kg/day) x 0.02.
d. Derived according to the following equation: Feed Consumption Rate (kg/day) x 0.14.
e. Derived according to the following equation: [Adult Beef Consumption Rate (kg/day) x Child Body Weight (kg)] / [70 (kg)].
f. Derived according to the following equation: [Adult Vegetable Consumption Rate (kg/day) x Child Body Weight (kg)] / [70 (kg)].
g. Geometric standard deviation.

50 years. However, the exposure duration distributions developed for several of the populations had maximum durations in excess of 50 years. Therefore, the model was designed such that when an exposure duration of greater than 50 years was selected, a value of 50 was assigned for that iteration.

Commercial Anglers

Historical information on commercial fishing activities in the Tennessee River Valley during the 1970s and 1980s indicated that there were two types of individuals who held commercial fishing licenses (Hargis, 1968; Morgan and Hubert, 1974; Hubert et al., 1975; Todd, 1990): full-time anglers who fished as a primary source of income throughout their lifetimes (i.e., 35 to 50 years), and part-time anglers who fished for supplemental income or to use commercial gear during their recreational activities. For the purpose of this assessment, it was assumed that part-time anglers would typically fish commercially for relatively short periods of time (i.e., one to eight years).

Available information indicated that the number of full-time commercial anglers fishing Watts Bar or Clinch River/Poplar Creek was typically very small for any given time period. For example, in 1967, a total of seven full-time commercial anglers were reported in Rhea, Meigs, Roane, Anderson, and Loudon Counties, combined (Hargis, 1968), while Todd (1990) reported four full-time anglers using Watts Bar in 1989. Based on this information, the project team assumed that the majority of commercial anglers would fish for short periods of time. As discussed in Section 5.0, the project team defined the commercial angler population by assuming that in any given year there were a total of seven full-time commercial anglers, and that each year one angler stopped activity and another commenced activity. Based on this assumption, the project team defined exposure duration for commercial anglers associated with Watts Bar and Clinch River/Poplar Creek as a discrete distribution, with a 93 percent chance of fishing for one to eight years (i.e., part-time), and a seven percent chance of fishing for 35 to 50 years (i.e., full-time).

To evaluate children associated with the commercial angler scenario, it was assumed that children living in commercial angler households between the ages of six to ten years might participate in fishing activities with their parent. It was assumed that children less than six years old were too young to participate in fishing activities. A triangular distribution ranging from one to five years with a most likely value of five years was used as a conservative estimate of exposure duration for the six to ten year old child.

Recreational Angler

The distribution of exposure durations for recreational anglers was based on the assumption that the number of years an angler fished in a particular water body was a function of the number of years the angler resided in the area. This assumption is supported by studies of angler behavior. For example, more than 80 percent of anglers surveyed in Tennessee reported traveling less than 50 miles from their residence for in-state fishing trips (USDOI, 1993). Based on this assumption, a cumulative distribution of exposure duration was modeled for Watts Bar and Clinch River/Poplar Creek recreational anglers using the mean and standard deviation for residence time for farm families, as reported by Israeli and Nelson (1992). Farm families were considered representative due to the rural nature of the counties near Watts Bar and Clinch River/Poplar Creek. It was noted, however, that because Watts Bar and Clinch River/Poplar Creek areas were large fisheries that were accessible from many counties, even if anglers moved from one county to another, they may have continued to fish in these areas. Thus, as a conservative measure, the residence times reported by Israeli and Nelson (1992) for farm families were doubled to reflect the lower rate of inter-regional mobility. Based on these assumptions, a cumulative distribution ranging from zero to 50 years was used.

In contrast to Watts Bar and Clinch River/Poplar Creek, EFPC is a small, less productive fishery that would be unlikely to attract a large number of anglers from the surrounding areas. It was assumed, therefore, that the recreational anglers along EFPC were Oak Ridge residents living in the immediate vicinity of the creek. In general, residency in Oak Ridge could be linked to employment at one of the Oak Ridge plants. Therefore, a cumulative distribution, based on occupational tenure information presented by Finley et al. (1994), was derived to represent exposure duration for this scenario. This distribution assumes that most individuals within the general population change jobs throughout the course of their employment history. However, anecdotal information indicates that individuals employed by the ORR often transferred from one facility to another within the ORR. Therefore, as an added conservatism, the mean and standard deviation reported by Finley et al. were doubled to account for the typically longer occupational tenure observed at the ORR. Based on these assumptions, a cumulative distribution ranging from 0.5 to 55 years was used.

To evaluate children associated with the recreational angler scenario, it was assumed that children living in recreational angler households between the ages of six to ten years might participate in fishing activities with their parent. Therefore, a triangular distribution ranging from one to five years with a most likely value of five years was used as a conservative estimate of exposure duration for all water bodies evaluated.

Farm Families

The exposure duration for the farm family adult was a cumulative distribution based on the residential occupancy period distribution for farms developed by Israeli and Nelson (1992). This distribution was defined by a minimum of 0.25 years and a maximum of 70 years. For children, the duration was conservatively assumed to be a triangular distribution ranging from one to ten years, with a most likely value of ten years.

Recreational User

The recreational user scenario was only evaluated for EFPC. Similar to the recreational anglers along EFPC, it was assumed that most of the adult recreational users lived in Oak Ridge and were employed by one of the plants. Therefore, the exposure duration was a cumulative distribution based on occupational tenure information presented by Finley et al. (1994). As discussed for the recreational angler, the mean and standard deviation reported by Finley et al. (1994) were doubled to account for the typically longer tenure observed at the ORR. Based on these assumptions, a cumulative distribution ranging from 0.5 to 55 years was used.

Children associated with the recreational user scenario were assumed to range in age from six to ten years, based on the assumption that participation of children younger than six would be more limited due to the increased need for parental supervision. Therefore, a triangular distribution ranging from one to five years with a most likely value of five years, was used as a conservative estimate of exposure duration.

7.1.2.1 Fish Consumption Rate

A detailed discussion of the data upon which the fish consumption rate distributions were based is provided in Section 5.0. While the available data pertained to consumption of fish by adults only, Rupp et al. (1980) provided regional information on age-related fish consumption habits. Based on these data, it was estimated that children (aged 1-11 years) consumed approximately 33 percent more fish on a per body weight basis than adults. Therefore, to derive distributions for children, an adjustment for body weight was applied to the adult consumption rates and a factor of 1.3 was applied to account for age-related differences in fish intake, as described by the following equation:

$$Cr_{child} = \frac{Cr_{adult} \times BW_{child} \times 1.3}{BW_{adult}}$$

Where:

Cr_{child} = Consumption rate for child (g/person-day)
Cr_{adult} = Consumption rate for adult (g/person-day)
BW_{child} = Body weight of child (kg)
BW_{adult} = Body weight of adult (kg)

<u>Commercial Angler</u>

The distribution of fish consumption rates for Watts Bar commercial anglers was assumed to be a lognormal distribution, based on a mean rate of 24 g/person-day derived from regional commercial harvest data reported by Hubert et al. (1975) (Section 5.0). The standard deviation (57 g/person-day) was derived by assuming that the coefficient of variation would be similar to that reported for other fish consumption studies (Ebert et al., 1993; Connelly et al., 1996) as discussed in Section 5.0.

Available data indicated that only nine percent of the commercial harvest in the region was obtained from rivers (Todd, 1990); based on this information, the mean consumption rate for commercial anglers on the Clinch River/Poplar Creek was assumed to be nine percent of the estimated rate of consumption for Watts Bar commercial anglers. Therefore, for Clinch River/Poplar Creek commercial anglers, a lognormal distribution with a mean consumption rate of 2.2 g/person-day was used. The standard deviation (5.2 g/person-day) was derived by assuming that the coefficient of variation would be similar to that reported for other fish consumption studies (Ebert et al., 1993; Connelly et al., 1996) as discussed in Section 5.0.

Recreational Angler

For Watts Bar recreational anglers, a mean consumption rate of 30 g/person-day was derived based on an evaluation of recreational catch statistics reported by the Tennessee Wildlife Resource Agency (TWRA) for the years from 1977 to 1991 (TWRA UNK) as described in Section 5.0. The standard deviation (71 g/person-day) was derived by assuming that the coefficient of variation was similar to that reported for other fish consumption studies (Ebert et al., 1993; Connelly et al., 1996) as discussed in Section 5.0.

Statistics from the 1991 USFWS survey (USDOI, 1993) indicated that the ratio of trips made by Tennessee anglers to rivers versus lakes was 0.6. Thus, to estimate rates of consumption by recreational anglers fishing Clinch River/Poplar Creek this ratio was applied to the consumption rate derived for Watts Bar anglers. This resulted in a mean consumption rate of 18 g/person-day for Clinch River/Poplar Creek recreational anglers. The standard deviation (43 g/person-day) was derived by assuming that the coefficient of variation was similar to that reported for other fish consumption studies (Ebert et al., 1993; Connelly et al., 1996) as discussed in Section 5.0. A lognormal distribution based on these values was used for this analysis.

While it is possible that recreational anglers could have spent a portion of their time fishing at EFPC, the level of activity was likely to be very low due to the limited available access, the nature of the creek itself, and the ready availability of higher quality fisheries nearby. EPA (1994b) has recommended using fish ingestion rates ranging from 1.2 to 4.1 g/person-day when estimating consumption by recreational anglers fishing small ponds or streams. Due to the small size and limited habitat of EFPC, a lognormal distribution based on a mean consumption rate of 1.2 g/person-day was used to evaluate recreational anglers who may have fished the creek. The standard deviation (2.9 g/person-day) was derived by assuming that the coefficient of variation was similar to that reported for other fish consumption studies (Ebert et al., 1993; Connelly et al., 1996) as discussed in Section 5.0.

7.1.2.2 Fraction of PCBs Remaining After Cooking

It has been demonstrated that cooking results in a reduction in concentrations of PCBs in fish fillets. Data in the published literature indicate that the percent of PCBs lost varies between 20 and 60 percent depending on the cooking method used (ChemRisk, 1992; Connelly et al., 1992; Sherer and Price, 1993). Thus, the fraction of the original PCB concentration remaining in the fish after cooking varies between 40 and 80 percent. Based on this information, the project team assigned a triangular distribution for this factor with a minimum of 0.4, a maximum of 0.8 and a most likely value of 0.5 to describe the fraction of PCBs remaining after cooking.

7.1.2.3 Fraction from Contaminated Source

Farm Families

Milk, Beef, Produce

To evaluate risks to the farm families, it was necessary to define the portion of milk, beef, and vegetables consumed by those families that were home produced and therefore potentially contained PCBs. As part of the 1955, 1965 and 1978 National Food Consumption Study (NFCS) (USDA, 1955a,b; 1966; 1978; 1983) participants provided information regarding the fractions of milk, beef, and produce that they consumed that were home grown. Based on these data, triangular distributions were derived to define the fractions of the milk, beef, and vegetables consumed by the farm families that were from a contaminated source (i.e., home produced). For beef and milk consumption, a most likely value of one (100 percent) was used; for vegetables, the most likely value was 0.93 (93 percent). The minimum and maximum values for all beef and vegetables were assumed to be 0 and 1, respectively. For milk, the minimum and maximum values were 0.04 and 1, respectively.

Recreational User

For the recreational user, the fraction from contaminated source represented the fraction of time that the individual was assumed to engage in recreational activities within the flood plain of EFPC. Due to the physical characteristics of the creek and the availability of larger, higher quality recreational water bodies nearby, it was assumed that only a limited portion of the time spent participating in recreational activities would have occurred in the vicinity of EFPC. Based on this assumption, a triangular distribution ranging from zero to one was used, with a most likely value of 0.1, or ten percent of total recreational activity.

7.1.2.4 Soil and Sediment Ingestion Rate

Farm Families, Recreational Angler, and Recreational User

Incidental ingestion of soil or sediment was evaluated for the farm families, recreational angler, and recreational user. For the recreational angler, all incidental ingestion exposures were conservatively assumed to be associated with sediments. For the farm family and recreational user, ingestion of soil only was evaluated. For the purpose of this analysis, it was assumed that the rate of sediment ingestion was the same as that for soil for all age groups.

While incidental soil ingestion occurs at all ages as a result of hand-to-mouth activities and deposition of airborne soil particles on foods, it is widely believed that younger children (i.e., two to five years old) are the only age group that consumes significant amounts of soil (Finley et al., 1994). As a result, there have been numerous studies of soil ingestion by younger children. Calabrese et al. (1989) quantitatively evaluated eight tracer elements in the feces and urine of children aged one to five years for the purpose of predicting soil ingestion. These data were subsequently validated using a model to predict the precision of the results (Calabrese and

Stanek, 1992). Based on this validation, soil ingestion rates for young children were estimated to range from a minimum value of 5 mg/d to approximately 241 mg/d.

In contrast, there are no reliable quantitative data available on the soil ingestion rates of adults and older children. In the absence of such data, it was conservatively assumed that adults and older children consumed soil at approximately half the rate of soil reportedly ingested by younger children. Based on this assumption, a cumulative distribution with a minimum ingestion rate of 2.5 mg/d and a maximum of 100 mg/d was derived for adults and children aged six to ten years old, based on the data reported by Stanek and Calabrese (1992). For children aged one to ten years (i.e., farm family) an average of the values used for older children/adults and the rates reported by Stanek and Calabrese (1992) were used to derive a minimum value of 3.8 mg/d and a maximum value of 150 mg/d.

<u>Dairy and Beef Cattle</u>

The rate of soil ingestion by cattle has been found to vary depending upon the rate of consumption of fresh pasture and the time of year, as well as site-specific factors such as farm management practices, pasture type, soil type, and number of cattle grazing a given area (Healy, 1968; Fries et al., 1982; Thornton and Abrahams 1983; Zach and Mayoh, 1983; Sumerling et al., 1984). Soil ingestion rates per unit of fresh pasture consumed have been reported to be significantly higher during low pasture season (i.e., winter months when pasture growth is sparse) than during high pasture season. Reported soil ingestion rates for cattle range from a low of two percent total feed intake during high pasture season, to a high of 14 percent total feed intake during low pasture season, based on Zach and Mayoh (1983) and Fries et al. (1982). A custom uniform distribution was derived with a minimum value equivalent to two percent of the feed consumption rate and a maximum value of 14 percent of the feed consumption rate.

7.1.2.5 Exposure Frequency

<u>Recreational User</u>

The adult recreational user was assumed to participate in outdoor recreational activities at a minimum of one day every two weeks during nine months of the year when the weather was mild (i.e., spring, summer, and fall). The project team assumed a maximum frequency of two days per week for the same nine months (i.e., spring, summer, and fall). Based on these assumptions, a uniform distribution, with a minimum of 20 days/year and a maximum of 78 days/year, was developed.

The child recreational user was assumed to participate in outdoor recreational activities at a minimum of one day every two weeks throughout the year. The project team assumed a maximum frequency of two days per week for nine months (i.e., spring, summer, and fall) and one day per week for the remaining three months (i.e., winter). This is a very conservative estimate of the possible exposure frequency. Based on these assumptions, a uniform distribution, with a minimum of 26 days/year and a maximum of 91 days/year, was developed.

Recreational Angler

For the recreational angler, the exposure frequency associated with sediment exposure was dependent on the number of fishing trips taken by the angler. In the absence of site-specific data regarding the number of annual trips taken by anglers in the vicinity of Oak Ridge, a customized distribution was derived based on the assumed fish consumption rate and a reasonable assumption about the amount of fish likely to have been obtained during each trip. It was assumed that an angler caught an average of 400 g of edible fish per fishing trip. This value was derived based on reported harvest data rates (fish mass/trip) for the Savannah River in Georgia (Turcotte, 1983) and the assumption that 30 percent of harvested fish was edible tissue. The number of trips per year were then back calculated as follows to derive an exposure frequency for the dermal contact and sediment ingestion pathways for the recreational angler:

$$EF = \frac{U_f \text{ x } 365\ (days/year)}{400\ (g/trip)}$$

Where:

EF	=	Exposure frequency (days/year when one trip is assumed to be equivalent to one day)
U_f	=	Consumption rate (g/person-day)

For the fish consumption pathway, an exposure frequency of 365 days/year was used. Because the fish consumption rate estimates are based on annual fish harvest statistics averaged over 365 days to derive a daily rate, it is appropriate to use a frequency of 365 days/year.

Farm Families

For the purpose of evaluating exposure frequency for the soil/sediment ingestion and dermal contact pathways for farm families, a triangular distribution was used. It was assumed that exposure to soil occurred at a minimum of two days per week, or 104 days/year, while the individual participated in typical, outdoor residential activities. A conservative maximum value of 365 days/year was assumed as it is conceivable that an individual living on a farm could be engaged in outdoor activities seven days per week. The most likely value was estimated to be equal to five days per week, or 260 days/year.

For the beef, milk, and vegetable consumption pathways, an exposure frequency of 365 days/year was used. As with the fish consumption pathway, it is appropriate to use this frequency when assuming annualized rates of consumption.

7.1.2.6 Skin Surface Area Exposed

A cumulative distribution of surface areas for evaluating dermal exposures to soil or sediment was determined based on data presented by EPA (1996b). For the farm family and the recreational user, it was assumed that for nine months of the year (i.e., spring, summer, and fall) exposed skin surfaces included forearms, hands, lower legs, and feet. It was assumed that only forearms and hands were exposed for the remaining three months of the year. The 95th percentile values, reported by EPA (1996b), were the basis for the maximum exposure areas of 17,808 cm^2/d for adults and 11,524 cm^2/d for children. The minimum values of 1,050 cm^2/d and 304 cm^2/d for adults and children, respectively, were derived based on the 5th percentile values reported by EPA (1996b).

7.1.2.7 Milk Consumption Rate

The 1955 and 1965 NFCS reported total milk consumption rates for one-person households of 0.82 L/d and 0.67 L/d, respectively (USDA 1955a,b; 1966). To estimate a fresh milk ingestion rate for the 1950s and 1960s based on the NFCS data, it was assumed that approximately 85 percent of the total milk consumed was fresh milk. Based on this assumption, the estimated average daily ingestion rate of milk for a rural farmer was assumed to be 0.6 L/d. The standard deviation for the distribution, estimated from the total dairy ingestion rate for rural farmers in the South from the 1978 NFCS data, was 0.908 (USDA, 1983). A lognormal distribution based on these values was derived. The ingestion rate for children (i.e., one to ten years) was assumed to be equal to that for adults. This is comparable to data reported for children by the USDA for 1978 (USDA, 1983). For the purpose of this assessment, one liter of milk was assumed to weigh 1 kg. Therefore, the consumption rate is presented in units of kg/d.

7.1.2.8 Beef Consumption Rate

The 1955 and 1965 NFCS reported meat consumption as consumption of total meats; this estimate included beef, pork, poultry, fish, eggs, legumes, and nuts (USDA, 1955a,b; 1966). According to USDA data, 73 percent of total meats is comprised of beef, poultry, and fish. The project team assumed that 50 percent of the beef, poultry, and fish fraction was represented by beef. Therefore, it was estimated that approximately 40 percent of the total meat (as classified by NFCS) was beef.

Based on this estimated fraction of the total meat ingestion rate, the beef ingestion rate for a rural farmer in the South during the 1955 and 1965 NFCS was estimated to be 0.128 and 0.188 kg fresh beef per day, respectively. The average daily ingestion rate of beef for the 1950s and 1960s was defined as a lognormal distribution with an arithmetic mean of 0.158 kg/d, the average of those two rates. The standard deviation used for the distribution, 0.186 kg/d, was estimated from the total meat ingestion rate reported in the 1978 NFCS for rural farmers in the South (USDA, 1983).

The 1955 and 1965 NFCS data did not provide food ingestion rates for children; rather, child food ingestion rates were combined with adult food ingestion rates and presented as a total household food ingestion rate. Due to the lack of data, it was assumed that children consumed the same amount of beef as adults, on a per body weight basis, according to the following relationship:

$$Cr_{child} = \frac{Cr_{adult} \ x \ BW_{child}}{BW_{adult}}$$

Where:

Cr_{child}	=	Consumption rate for child (g/d)
Cr_{adult}	=	Consumption rate for adult (g/d)
BW_{child}	=	Body weight of child (kg)
BW_{adult}	=	Body weight of adult (kg)

7.1.2.9 Vegetable Consumption Rate

Vegetable consumption rates were derived from the 1955 and 1965 NFCS data, which reported vegetable consumption as total vegetables (USDA, 1955a,b; 1966). Total vegetables included potatoes, sweet potatoes, dark green and deep yellow vegetables, other greens, and tomatoes, as well as mixtures and soups. The average total vegetable ingestion rates for a southern rural farmer were reported at 0.51 and 0.48 kg fresh vegetables per day, respectively, resulting in an average consumption rate of 0.495 kg/d. Based on this information, a lognormal distribution was derived with a mean of 0.495 kg/d. A standard deviation of 0.425 kg/d was estimated based on total vegetables ingested by rural southern farmers in 1978 (USDA, 1983).

The NFCS did not provide information on vegetable ingestion rates for children; rather, their ingestion rates were combined with adult food ingestion rates and presented as a total household vegetable ingestion rate. To define a vegetable ingestion rate distribution for children, it was assumed that children consumed the same amount of vegetables as adults on a per body weight basis. Therefore, the vegetable ingestion rate distribution for children was derived according to the same relationship described in Section 7.1.1.9 for consumption of beef.

7.1.2.10 Fraction of Pasture Consumed

In the Oak Ridge area, it was assumed that high pasture season occurred for eight months (i.e., mid-February to mid-October), comprising approximately 60 percent of the year, and that low pasture season occurred for the remaining four months. Pasture consumption fractions were assumed to be approximately 80 percent and 20 percent of total feed during high and low pasture seasons, respectively (Koranda, 1965). Based on this information, a uniform distribution was used where the minimum value, 0.3, represented the weighted average of these fractions, and the maximum value of 100 percent assumed that the cattle’s entire diet was from grazing.

7.1.3 Exposure Point Distributions

Exposure point distributions were based on the data used to select the point estimate EPCs for the level I evaluation. As discussed for that analysis, only those data pertaining to areas where off-site exposures were likely to have occurred were evaluated. Samples from all years and depths were considered.

7.1.3.1 Soil and Sediment

Due to the limited nature of the available soil and sediment data, uniform distributions were selected to best represent PCB concentrations in these media. The minimum values represented the lowest reported values for the specified water body and medium, based on the available detection limits. With the exception of soils at EFPC, the maximum values were equivalent to the maximum values used for the level I evaluation. For EFPC, additional investigations indicated that the value of 18 mg/kg originally used for the level I evaluation to represent soil concentrations was associated with a localized spill event and was not representative of concentrations in other sections of the creek. Although considered appropriate for the purposes of a conservative screening assessment, the project team decided to use a more realistic value for the level I evaluation. Therefore, the maximum concentration of 6.8 mg/kg, based on data reported during the Sewer Line Beltway Remedial Investigation (SAIC, 1994), was used. The exposure point distributions for each of the water bodies are presented in Table 7-7.

The distribution of the average soil concentrations that individuals would have come in contact with over their lifetimes would have better characterized the interindividual variation in the dose rates. Because average PCB concentrations come from a number of different soils that an individual may contact, the distribution of PCB levels should have less variability than the distribution of values taken at specific locations. In addition, because the distribution of most environmental contaminants tend to follow a lognormal distribution, the use of a uniform distribution likely overestimates the number of individuals that have doses close to the maximum levels. For these reasons, the use of a uniform distribution likely overestimated dose rates for individuals in the upper portion of the distribution.

7.1.3.2 Fish

The project team derived distributions of PCB concentrations in fish for each water body from those data compiled to determine the point estimate EPCs discussed in Section 6.4.5. As previously reported, some studies reported data for individual fish, while others reported only summary statistics (i.e., average or composite concentration) by species. The number of fish analyzed to derive the average or composite concentration varied among the studies evaluated; therefore, the project team calculated weighted averages based on the number of fish analyzed in each study for each commonly consumed species for each water body to insure that the variation among individual fish was properly evaluated. The distribution of exposure point concentrations for fish were derived based on the following assumptions. Because an individual consumes many fish over his or her lifetime, the inter-fish variation in levels of contamination tends to average out. For example, if two anglers fish the same body of water and have similar preferences in the type of fish they consume, the long-term average concentration in the fish they consume would be similar even if the concentration in specific fish differs. Consequently, the inter-fish variation in PCB concentration is not directly relevant to the distribution in interindividual exposure point concentrations.

Because of this tendency for concentrations to average out, the greatest source of interindividual variation in long-term average fish concentrations is species preference. In other words, if one angler prefers a species with higher or lower than average PCB levels, then their long-term average fish concentration will differ. For this reason, we have based the distribution of exposure point concentrations on different species. Specifically, we have assumed that the highest and lowest values of fish concentrations are given by the highest and lowest species-specific average concentrations. A triangular distribution was developed for each water body based on the weighted averages calculated for each species, using the minimum, maximum and

Table 7-7. Exposure Point Distributions for Soil and Sediment Exposures

Exposure Parameters	Distribution	Minimum	Maximum
PCB Concentration (mg/kg)			
Clinch River/Poplar Creek[a]			
Sediment	Uniform	0.01	14.3
Watts Bar Reservoir			
Sediment	Uniform	0	0.1
East Fork Poplar Creek			
Soil	Uniform	0.005	6.8
Sediment	Uniform	0.2	6

Note: Concentrations used may represent releases from additional sources other than the ORR.

a. Clinch River and Poplar Creek data were combined for the purpose of developing this EPC.

mode values as the minimum, maximum, and most likely values, respectively for the distribution for all species. For recreational anglers, all weighted averages for all game species were used in determining the triangular distribution. For commercial anglers, only those species known to be targeted for commercial sale were evaluated. The exposure point distributions for each of the water bodies are presented in Table 7-8.

In this assessment, as in the level I evaluation, the project team assumed that the monitored fish concentrations in each species are indicative of tissue concentrations over the last 40 years. This finding is based on the TVA data for the Clinch River and Watts Bar which have been relatively constant over the last 15 years. In addition, cores from the Clinch River and Watts Bar also suggest that PCBs levels in sediments and by extension in fish have been relatively stable over the last 35 years. There is limited evidence in the sediment cores that PCB levels spiked in the Clinch River on three occasions in the 1960s and 1970s (See Appendix A). However, these increases spanned relatively short periods (one to three years) and would not have had a significant effect on the estimates of lifetime average daily doses used in the carcinogenic risk assessment. The increases could have had an effect on the estimates of chronic exposure used in the noncarcinogenic daily doses. The need for additional data on historical sediment levels from sediment cores is discussed in section 7.4.

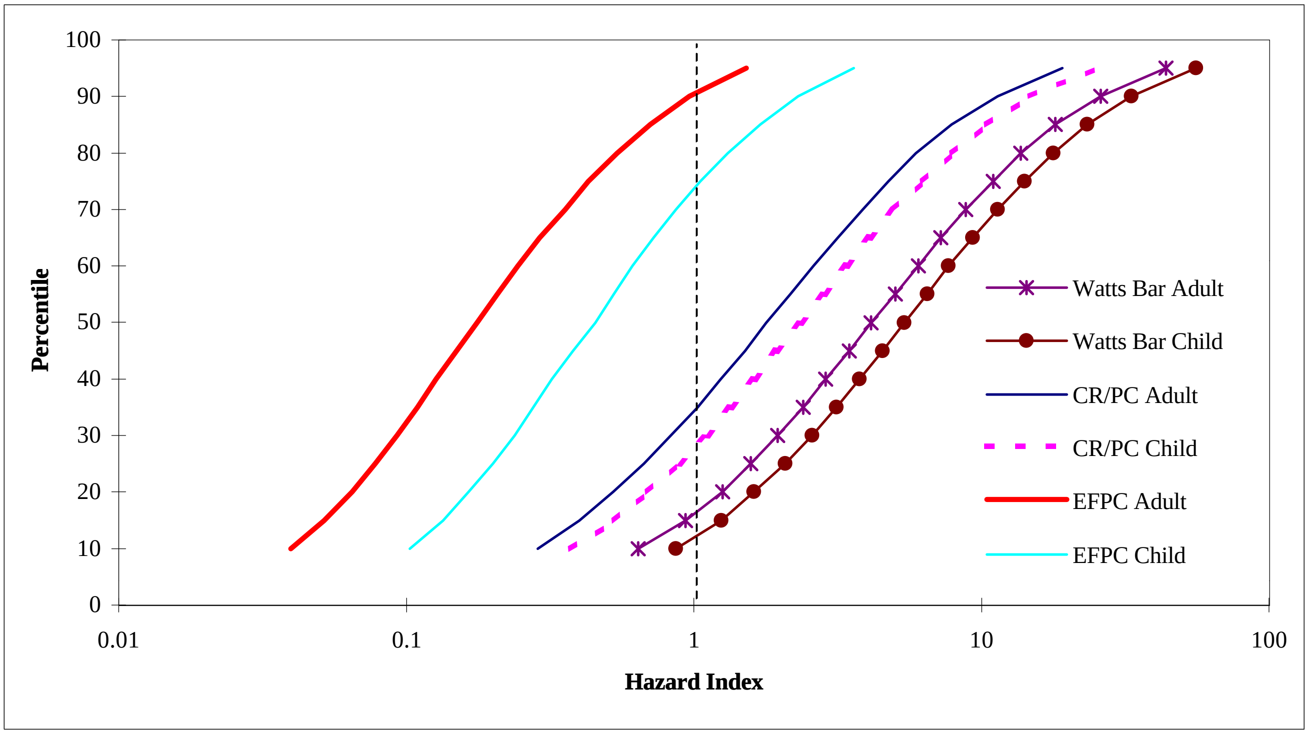
7.2 PRELIMINARY CHARACTERIZATION OF POPULATION VARIABILITY

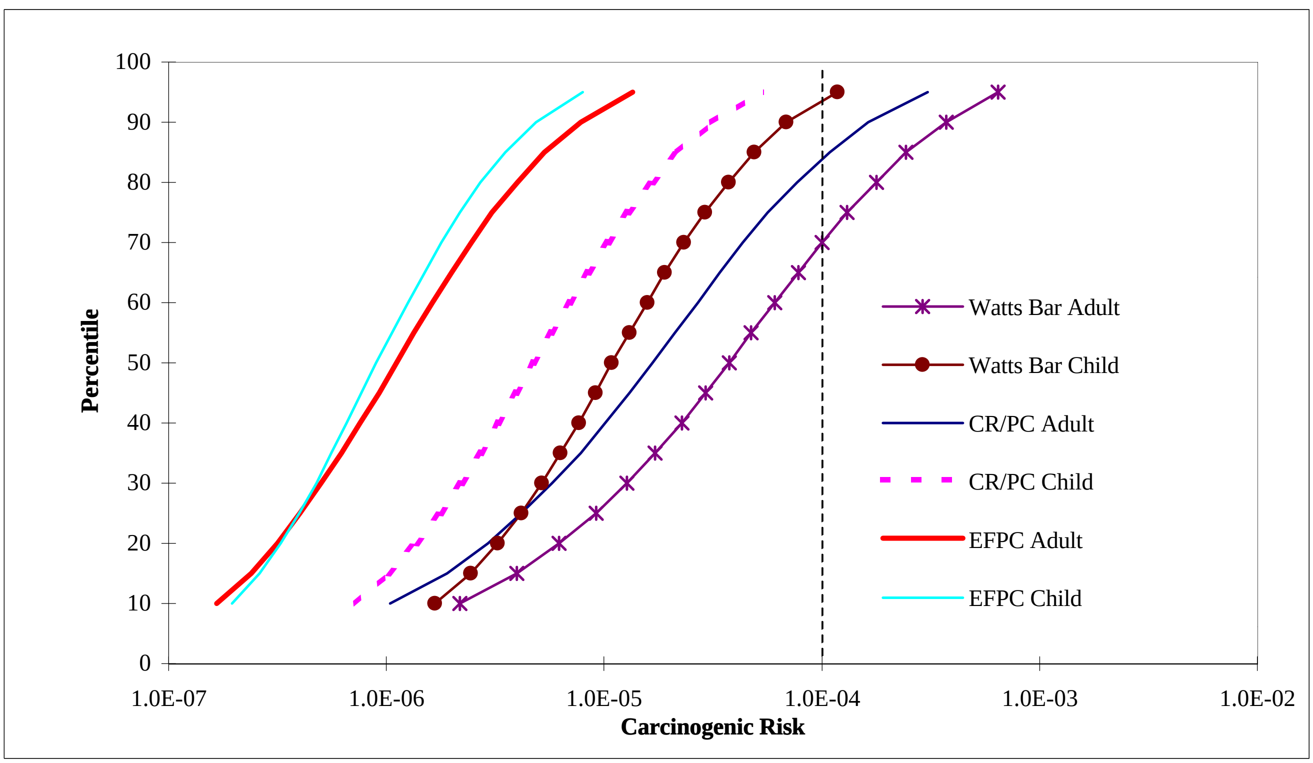
As described above, the purpose of characterizing interindividual variation in dose was to develop distributions of risk that accounted for uncertainty and variability in exposure estimates. The results of the analysis demonstrated that there was considerable variability in both noncarcinogenic and carcinogenic risk estimates for all of the populations evaluated. The risks to adults and children did not differ greatly and, in some cases, the ranges in the risk estimates greatly overlapped. Adults tended to have slightly higher cancer risks because of their longer exposure durations, resulting in higher lifetime average daily doses. Noncancer risks also tended to be slightly higher. The results for the 5th percentile, mean, median (50th percentile), and 95th percentile of each estimated risk distribution are presented in Tables 7-9 and 7-10 and in Figures 7-1 through 7-4. A percentile represents the fraction of the population that falls at or below a particular risk estimate. For example, a risk estimate of 1×10^{-6} at the 5th percentile indicates that five percent of the population has a total incremental lifetime risk of 1×10^{-6} or less. It should be noted that the estimates of exposure and toxicity are biased towards overestimating risks.

7.2.1 Recreational Anglers

Figure 7-1 presents the cancer and noncancer risks for the recreational anglers at EFPC, Watts Bar Reservoir, and Clinch River/Poplar Creek. The cancer and noncancer risks for the recreational anglers at EFPC were lower than for the other bodies of water. Risks for both adults and children were below the decision criteria (with the exception of children at the 95th percentile where noncancer risk exceeded 1). The lower fish consumption rates for EFPC, based on its poor quality as a fishery, accounted for the lower risk estimates.

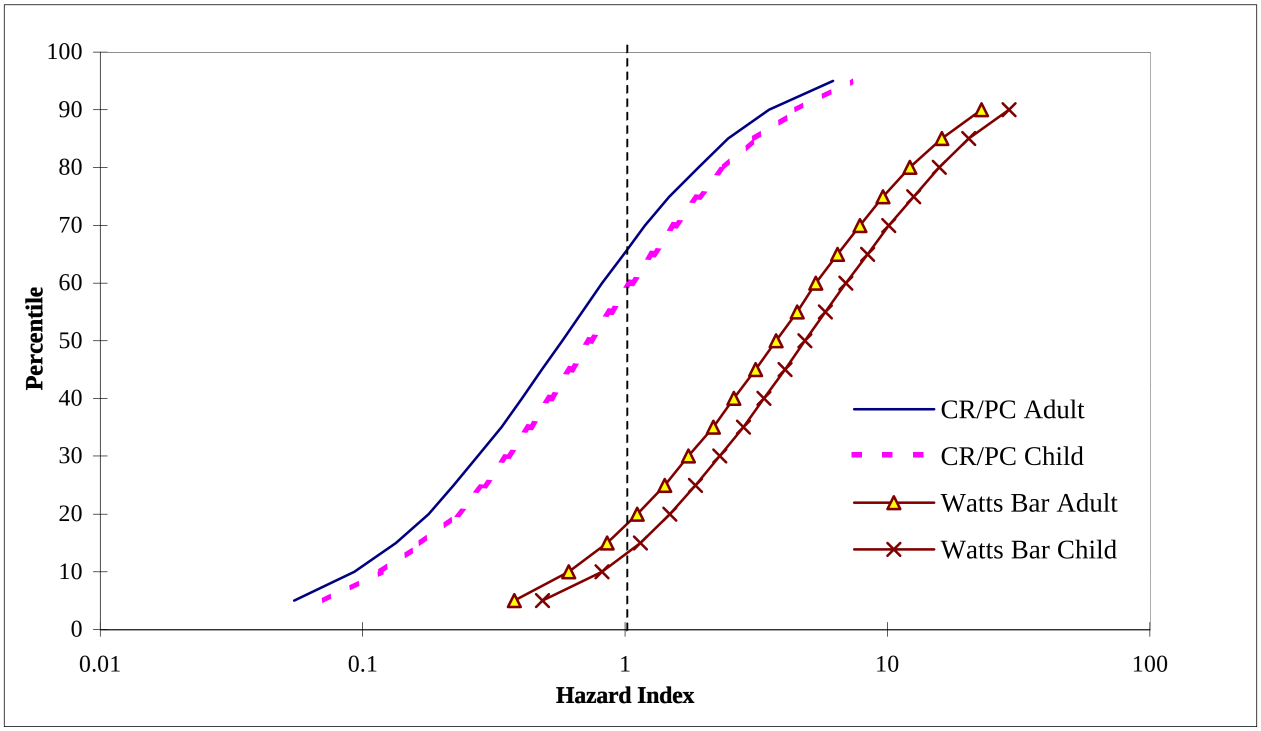
Figure 7-1. Lifetime Risks to Recreational Fish Consumers

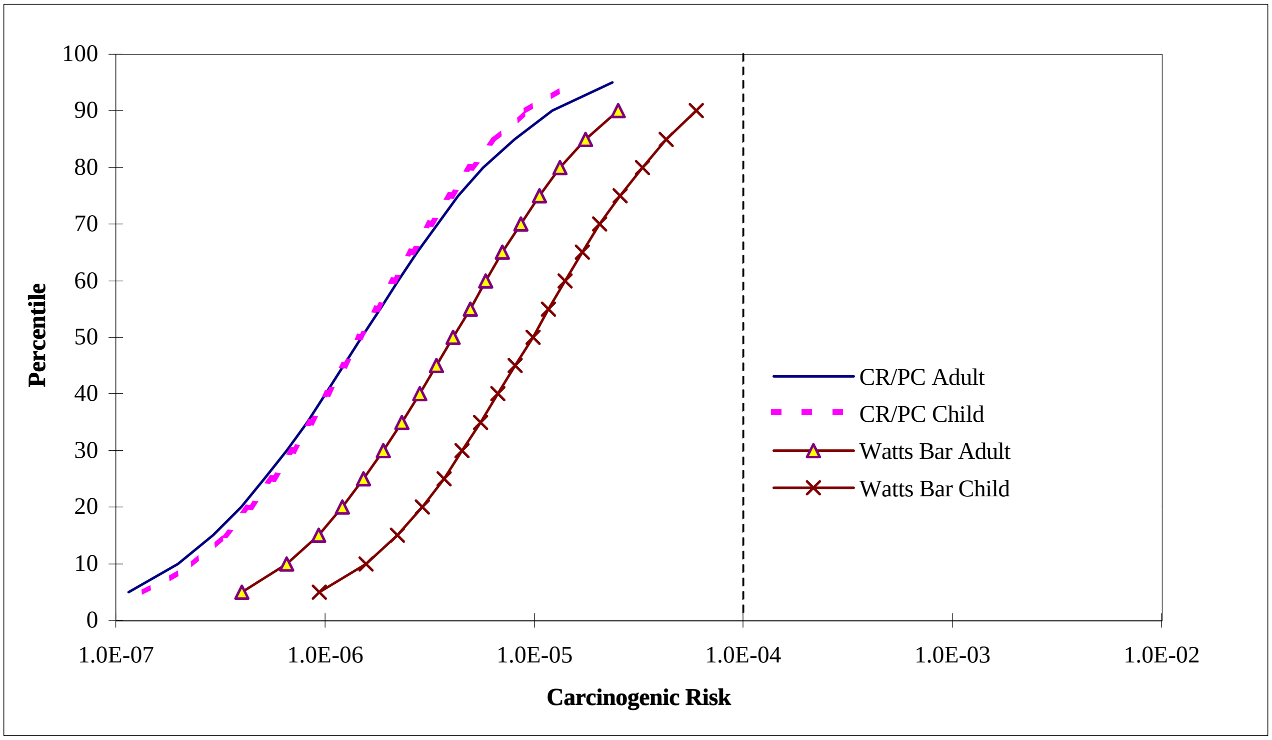




EFPC = East Fork Poplar Creek
CR/PC = Clinch River/Poplar Creek

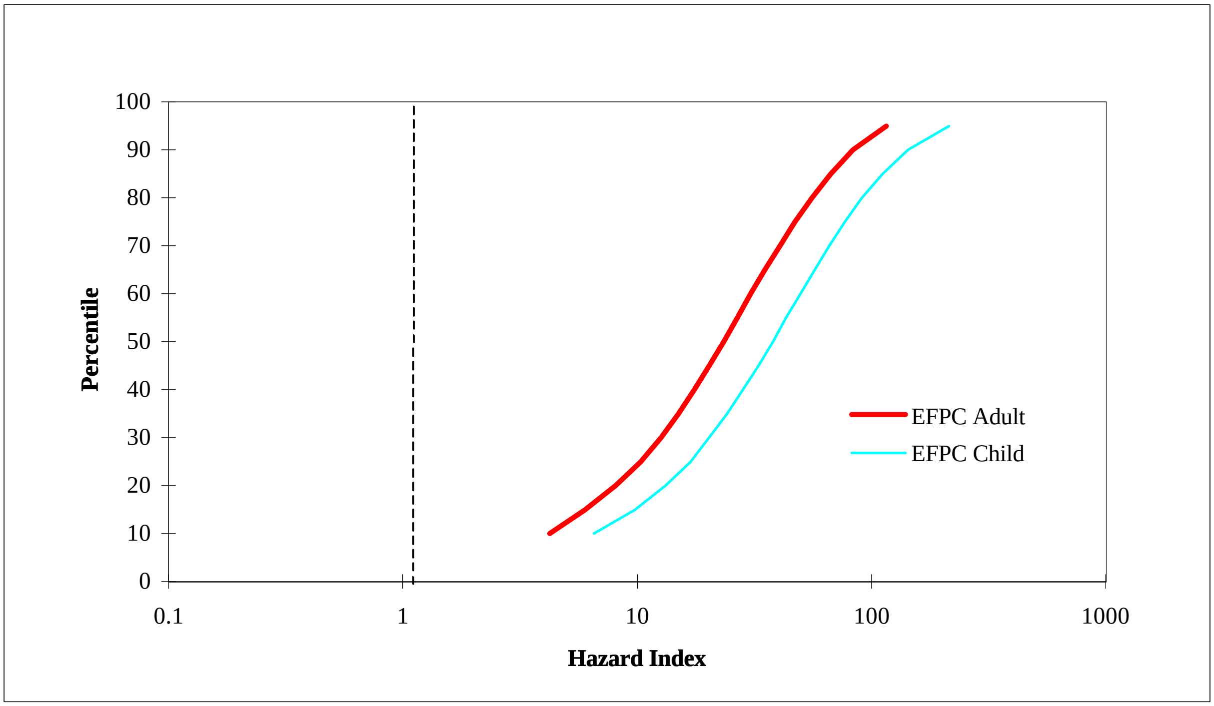
Figure 7-2. Lifetime Risks to Commercial Anglers

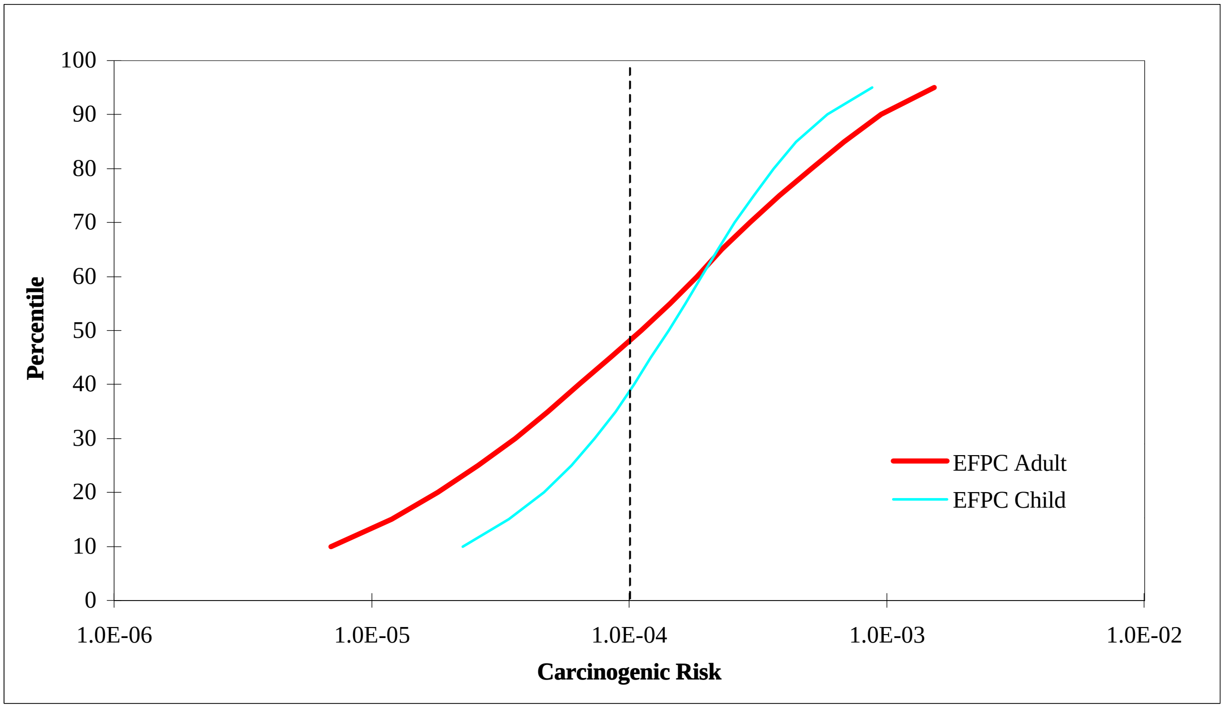




Note: CR/PC = Clinch River/Poplar Creek

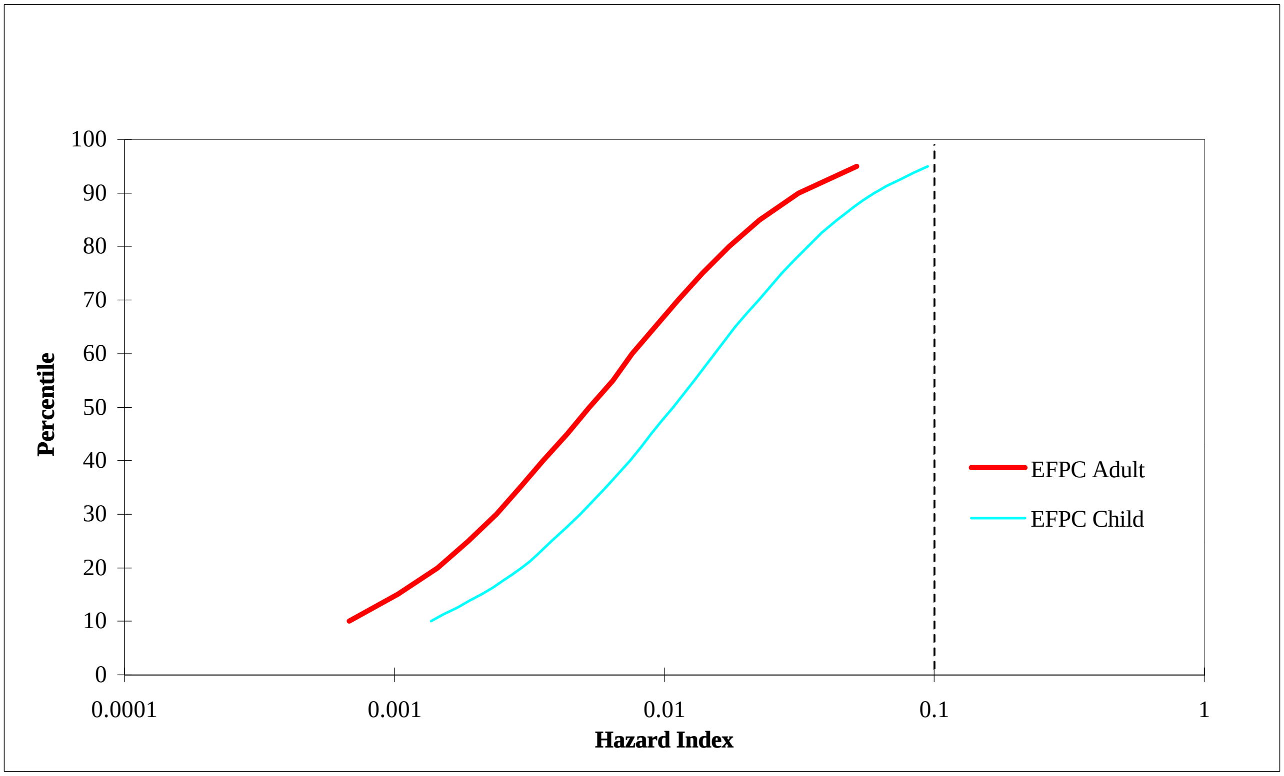
Figure 7-3. Lifetime Risks to Farm Families

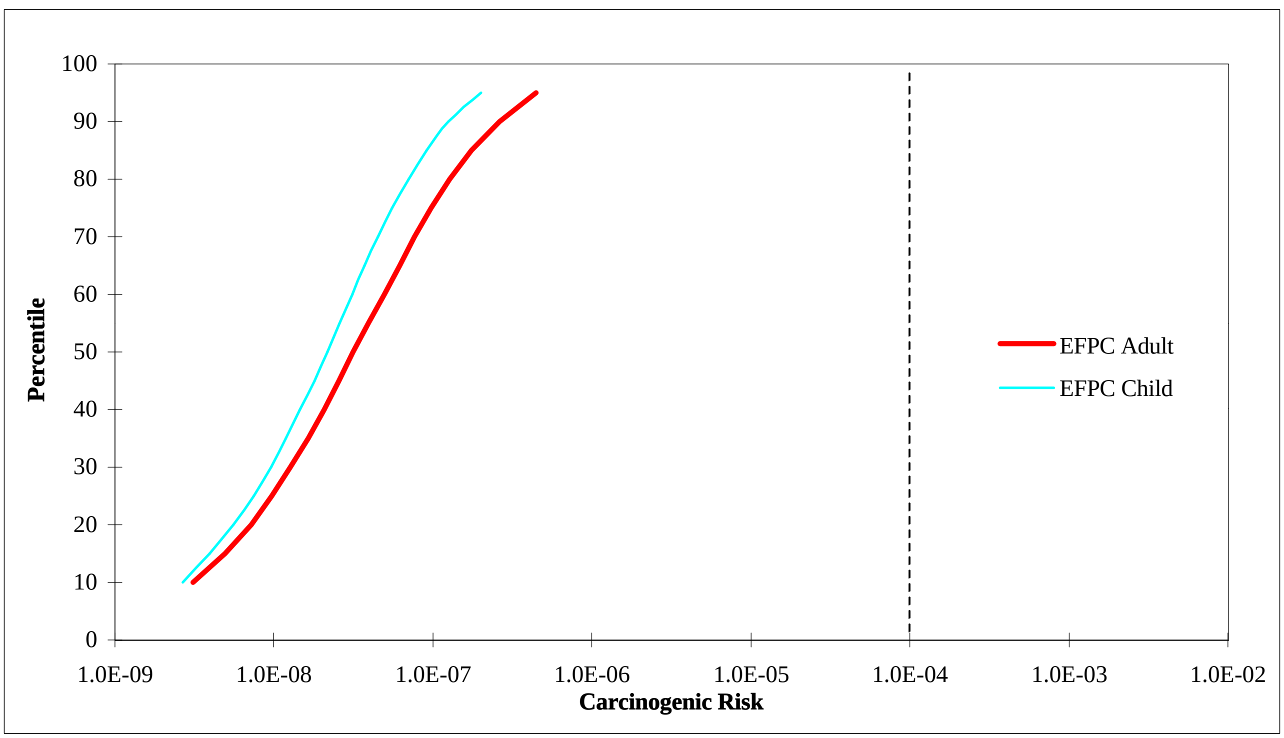




Note: EFPC = East Fork Poplar Creek
CR/PC = Clinch River/Poplar Creek

Figure 7-4. Lifetime Risks to Recreational Users





Note: EFPC = East Fork Poplar Creek

Table 7-8. Exposure Point Distributions for Fish Consumption

Exposure Parameters	Distribution	Minimum	Most Likely	Maximum
PCB Concentration in Fish (mg/kg)				
Clinch River/Poplar Creek				
Commercial [a]	Triangular	1.03	1.07	3.2
Recreational [b]	Triangular	0.20	0.66	1.2
Watts Bar Reservoir				
Commercial [a]	Triangular	0.45	1.30	1.35
Recreational [b]	Triangular	0.29	1.17	1.35
East Fork Poplar Creek				
Recreational [b]	Triangular	0.49	0.71	1.04

Note: Concentrations used may represent releases from additional sources other than the ORR.

a. Only species known to be targeted by commercial anglers were evaluated. Included were: carp, catfish, smallmouth buffalo, and longnose gar.

b. All game species, except longnose gar, were evaluated. Included were: bass, blue gill, carp, catfish, crappie, gizzard shad, largemouth bass, sunfish sp., sauger, striped bass, white bass, redbreast sunfish, and black crappie. Longnose gar was not included because this fish is consumed very infrequently (personal communication with TN Dept. of Fisheries, 8/96).

Table 7-9. Summary of Cancer Risks for the Populations Evaluated in the Total Uncertainty Analysis

	Waterway					
	Adult			Child		
Population	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek
Recreational Fish Consumer						
5th Percentile	**5×10^{-8}**	**9×10^{-7}**	**4×10^{-7}**	**3×10^{-8}**	**1×10^{-6}**	**4×10^{-7}**
Mean	**3×10^{-6}**	**2×10^{-4}**	**7×10^{-5}**	**1×10^{-7}**	**3×10^{-5}**	**1×10^{-5}**
Median	**8×10^{-7}**	**4×10^{-5}**	**2×10^{-5}**	**3×10^{-7}**	**1×10^{-5}**	**5×10^{-6}**
95th Percentile	**1×10^{-5}**	**6×10^{-4}**	**3×10^{-4}**	**4×10^{-6}**	**1×10^{-4}**	**5×10^{-5}**
Commercial Angler						
5th Percentile		**4×10^{-7}**	**1×10^{-7}**		**9×10^{-7}**	**1×10^{-7}**
Mean		**1×10^{-5}**	**6×10^{-6}**		**3×10^{-5}**	**4×10^{-6}**
Median		**4×10^{-6}**	**2×10^{-6}**		**1×10^{-5}**	**1×10^{-6}**
95th Percentile		**4×10^{-5}**	**2×10^{-5}**		**9×10^{-5}**	**2×10^{-5}**
Farm Family						
5th Percentile	**3×10^{-6}**			**1×10^{-5}**		
Mean	**4×10^{-4}**			**3×10^{-4}**		
Median	**1×10^{-4}**			**1×10^{-4}**		
95th Percentile	**2×10^{-3}**			**9×10^{-4}**		
Recreational User						
5th Percentile	**1×10^{-9}**			**1×10^{-9}**		
Mean	**1×10^{-7}**			**5×10^{-8}**		
Median	**3×10^{-8}**			**2×10^{-8}**		
95th Percentile	**4×10^{-7}**			**2×10^{-7}**		

Note: Estimated risk values include those attributable to sources of PCBs other than the ORR.

Table 7-10. Summary of Noncancer Risks for the Populations Evaluated in the Total Uncertainty Analysis

	Waterway					
	Adult			Child		
Population	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek	East Fork Poplar Creek	Watts Bar Reservoir	Clinch River/ Poplar Creek
Recreational Fish Consumer						
5th Percentile	**0.01**	**0.4**	**0.2**	**0.02**	**0.5**	**0.2**
Mean	**0.4**	**10**	**5**	**0.5**	**15**	**6**
Median	**0.1**	**4**	**2**	**0.2**	**5**	**2**
95th Percentile	**1**	**40**	**20**	**2**	**60**	**30**
Commercial Angler						
5th Percentile		**0.4**	**0.05**		**0.5**	**0.07**
Mean		**10**	**2**		**13**	**2**
Median		**4**	**0.6**		**5**	**0.7**
95th Percentile		**40**	**6**		**50**	**7**
Farm Family						
5th Percentile	**2**			**3**		
Mean	**40**			**70**		
Median	**20**			**40**		
95th Percentile	**100**			**200**		
Recreational User						
5th Percentile	**0.0003**			**0.0007**		
Mean	**0.01**			**0.03**		
Median	**0.005**			**0.01**		
95th Percentile	**0.05**			**0.09**		

Note: Estimated risk values include those attributable to sources of PCBs other than the ORR.

The risks for recreational anglers using Clinch River/Poplar Creek were higher than those at EFPC. The cancer risk for adults at the 95th percentile was 3×10^{-4}. The cancer risk estimate for children at the 95th percentile was less than the benchmark of 1×10^{-4}. Nominal hazard quotients exceeded the noncancer criterion of one for both adults and children at the mean, median and 95th percentile.

Recreational anglers using Watts Bar had the highest cancer and noncancer risks of the three water bodies due to higher levels of PCBs in the fish and greater fish consumption rates. The cancer risk estimate for adults was 6×10^{-4} at the 95th percentile and 1×10^{-4} for children. Similar to the Clinch River/Poplar Creek analysis, the nominal hazard quotients exceeded the noncancer criterion of one for both adults and children at the mean, median and 95th percentile. The median estimates of the nominal hazard quotients were greater than one and the 95th percentiles exceeded 20.

PCBs have entered Watts Bar from releases to both the Clinch River and the Tennessee River above the reservoir. Based on work presented in Appendix A on the contribution of PCBs from sources other than the ORR, it appears that the majority of PCBs in Watts Bar likely came from sources on the Tennessee River. The contribution of the ORR releases represented 8 to 13 percent of the total amount of PCBs. Therefore, at most, 13 percent of the total cancer risk for recreational anglers using Watts Bar was attributable to the ORR. Because 13 percent of 6×10^{-4} is less than the decision criterion of 1×10^{-4}, additional study of the cancer risks to Watts Bar recreational anglers is deferred at this time. In contrast to the estimate of carcinogenic risks, noncarcinogenic risks associated with the ORR releases would still be a concern. If the contribution from the ORR to fish was as low as eight percent and if the ORR releases were the only source of PCBs in the fish, then the estimate of the nominal hazard quotient would still exceed the noncancer risk criterion of 1.0.

The percentage of PCBs in the Clinch River that is attributable to the ORR releases is not well defined. As discussed in Appendix A**,** up to one half of the PCBs present in Clinch River fish may have been contributed by other sources on the Clinch River. However, even if one-half of the PCBs were due to non-ORR sources, the ORR releases would have resulted in risks above the cancer decision criterion. In addition, even if one-half of the PCBs in Clinch River fish were due to the ORR releases, the nominal hazard quotients would still exceed the noncancer decision criterion.

7.2.2 Commercial Angler

Figure 7-2 presents the results of the cancer and noncancer risks for commercial anglers from Watts Bar Reservoir and Clinch River/Poplar Creek. In general, the risks were similar to, but slightly lower than risks estimated for recreational anglers. At the 95th percentile, all of the cancer risks for these populations were equal to or less than 1×10^{-4}. The noncancer risks for the 95th percentile adult and child were greater than five for the Clinch River/Poplar Creek anglers and were approximately ten times higher for the Watts Bar Reservoir anglers.

As with recreational anglers, the risk estimates for the commercial anglers are affected by other sources of PCBs along the rivers. Because the cancer risks for this population are already below the cancer criterion, further reduction by accounting for different sources would not change the findings of the analysis. The noncancer risk estimates, while reduced if other sources are taken into consideration, still exceed the noncancer decision criterion.

7.2.3 Farm Family

The estimates of cancer and noncancer risks were higher for the farm family population than the angler populations (Figure 7-3; Tables 7-9 and 7-10). Carcinogenic risk estimates at the 95th percentile for both adults and children exceeded the cancer decision criterion. Nominal hazard quotients exceeded the noncancer decision criterion for the entire range of the uncertainty analysis. It should be noted, however, that the actual concentrations of PCBs in soil at the farms were highly uncertain, and this uncertainty was not considered in the assessment. The exact location of the EFPC soil samples in relation to current and historical gardens and pastures within the flood plain is unknown. It is unclear whether the samples included soils from areas that may have actually supported gardens or pastures. Thus, risk estimates based on the available data may not be representative of actual hazards associated with those areas.

7.2.4 Recreational User

The risk estimates for the recreational user population at EFPC are presented in Figure 7-4. Both the noncancer and cancer risks were below the decision criteria; at the 95th percentile, cancer risks for adults and children were 4×10^{-7} and 2×10^{-7}, respectively. Similarly, the nominal hazard quotients for adults and children were 0.05 and 0.09, respectively. These findings indicate that PCBs in the sediment at EFPC were not likely to cause adverse health effects in individuals using that area for recreational purposes.

7.3 SUMMARY AND CONCLUSIONS

This characterization of variation in risks to the exposed population represents a further refinement of the level I evaluation presented in Section 6.0. In general, results of the level II evaluation confirmed the findings of the level I evaluation in that the majority of the populations that exceeded decision criteria in the level I assessment also had risk estimates at the 95th percentile that exceeded the decision criteria. However, the results of the level II evaluation indicated that the risks to recreational users of EFPC were below levels of concern.

Based on the results of the level II evaluation, the following conclusions have been made:

- The portion of the PCBs released from the ORR at the 95^{th} percentile of risk did not reach the 1×10^{-4} risk level for the population of nearly a hundred thousand who ate fish caught by recreational anglers in Watts Bar Reservoir. For those exposed from the Clinch River, 95^{th} percentile risk reached that threshold. The PCB releases resulted in nominal hazard quotients larger than 1 for a large share of the population. (Note that exposure at the level yielding a hazard quotient of 1.0 is considered by EPA to be assuredly safe even for members of sensitive subpopulations.)

Commercial anglers on Watts Bar and Clinch River did not have predicted cancer risks in excess of 1 x 10^{-4} but most anglers had nominal hazard quotients greater than 1 as a result of intake of PCBs from all sources (ORR and non-ORR sources). The fraction of PCBs released from the ORR would also have resulted in nominal hazard quotients greater than 1 for some commercial anglers. The total number of commercial anglers and their families are believed to be less than 180 individuals.

- This analysis found that a large percentage of recreational anglers on Watts Bar and Clinch River had predicted cancer risks in excess of 1 x 10^{-4} and nominal hazard quotients greater than 1 as a result of intake of PCBs from all sources (ORR and non-ORR sources).

- The fraction of PCBs released from the ORR was unlikely to have resulted in cancer risks in excess of 1 x 10^{-4} among recreational anglers. However, such releases would have resulted in nominal hazard quotients greater than 1 for many recreational anglers. The number of recreational anglers and their families who consumed fish since the 1940s is likely to be greater than 100,000.

- If the assumptions concerning PCBs in flood plain soils are correct, families who lived on affected farms had the highest potential for exposure to PCBs. Populations exposed at the doses estimated in this analysis were potentially at risk from both carcinogenic and noncarcinogenic effects of PCBs. Given the considerable uncertainty in the available information on historical PCB concentrations in the soils of the farm properties, there is little or no certainty in the risk estimates. The number of individuals in the population is believed to be small (less than 50).

7.4 ADDITIONAL SOURCES OF UNCERTAINTY

The level II evaluation suggested that angler and farm populations were at risk from PCBs and should be investigated further. The key issues that require additional study are as follows.

<u>The significance of exceeding the Reference Dose (RfD).</u> The current noncancer decision criterion is based on the assumption that any dose that exceeds the RfD is of some toxicological concern. As discussed in Section 4.0, there is a high degree of certainty that there is no risk with a dose below the RfD, that doses in excess of the RfD (2-10 times higher) are likely to be without risk, and even doses that exceed the RfD by orders of magnitude may be without any appreciable effect. Therefore, it is clear that findings of nominal hazard quotients greater than one (i.e., a dose greater than the RfD) do not imply that adverse effects will occur.

<u>The exposures to farm families.</u> The risk estimates for the farm family population are the highest for any of the groups potentially exposed. However, the risk estimates are based on soil concentrations that were taken from areas adjacent to the EFPC. It is not clear whether these concentrations are representative of most or even part of the soils in the areas historically used for pasture at the farms. In addition, there are a number of conservative assumptions concerning uptake of PCBs by cattle and human consumption rates of dairy and beef that may have resulted in overestimated estimates of dose and risk.

<u>The role of Oak Ridge releases on past and current levels of PCBs in fish.</u> A key issue in determining the impact of Oak Ridge releases is the fraction of PCBs in fish that occurred as a result of the ORR releases. The existing studies on the impacts from other PCB sources provided insights into the relative role of sources on the Clinch and Tennessee Rivers but did not establish the fraction of the PCBs in the Clinch that are due to Oak Ridge releases. In addition, there were only limited data on whether the existing monitoring data taken in the 1980s and 1990s reflected levels that occurred in the 1950s, 60s, and 70s.

<u>Characterization of uncertainty.</u> The current model makes some conservative (biased) estimates with respect to uncertainty. As a result, the assessment can not answer questions such as what is the uncertainty in the dose and risk to individuals in the top five percent of the exposed population.

Options to address these key areas are as follows:

1. Develop refined risk measurements for noncarcinogenic effects.

   The RfD setting methodology can be extended to determine the uncertainty in an estimate of the population threshold for the noncarcinogenic effects of PCBs (Baird et al., 1996; Swartout et al., 1998). This approach can be used to calculate the probability of an individual receiving a dose that could affect sensitive individuals. It should be noted that the term threshold is used in the sense of a threshold for concern. Currently, toxicologists are divided on whether there is a true threshold for the noncarcinogenic effects of PCBs in humans.

   Section 8.0 presents the characterization of the uncertainty in the threshold for the noncarcinogenic effects of PCBs, based on current techniques developed by the project team and the EPA.

2. Collect additional data on PCBs in selected media and biota.

   Because PCBs are highly persistent compounds, it is possible to measure current levels of PCBs and use the data to estimate historical exposures. Examples of additional measurements could include: soil samples from current and historical farm properties near EFPC; sediment core samples from the Clinch River and Poplar Creek to identify upstream sources and better define historical trends in PCBs over the last 40 years; and biological samples (blood and adipose tissue) from cattle grazing in the flood plain areas of EFPC.

   Section 9.0 presents recommendations for additional sampling of soils, sediment, and biota that will assist in reducing the uncertainty in the assessment.

3. Perform a refined uncertainty analysis that separates uncertainty and variation.

   Over the past five years, techniques have been developed for modeling both the uncertainty and variability of dose rates in exposed populations. One technique, called two-dimensional Monte Carlo, uses a “nested loop” programming technique that requires additional model development and computer resources. The result is an estimate of the distribution of risk across the exposed populations and the uncertainty bounds on those estimates.

   Section 8.0 presents a two-dimensional Monte Carlo model (level III evaluation) for noncarcinogenic risks for the recreational anglers. No additional modeling was performed for the commercial anglers, because their risks appear to be similar or slightly lower than those estimated for recreational anglers and their population sizes are considerably smaller. In addition, because the Todd (1990) data provide a lower bound estimate of average consumption which the mean of 7 g/person-day is very similar to the mean consumption rates reported by Ebert et al. (1993) and Connelly et al. (1996), it is believed that the range of consumption rates provided by the recreational angler distributions are also representative of the rates of consumption by commercial anglers. Thus, the refined assessment for recreational anglers is assumed to be applicable to commercial anglers as well.

No additional modeling was conducted for the farm families. The greatest source of uncertainty in the risk estimates for the farm families was the assumed EPC in the soil used to grow pasture and the resulting PCB levels in beef and dairy products from the cattle that grazed on that pasture. This uncertainty should be addressed by additional sampling at the sites of current and former farms. Until such sampling is complete, additional modeling of uncertainty and variation in the noncancer and cancer risks associated with this scenario is not warranted. As discussed in Appendix B, the number of anglers that EFPC could support is very small. Because of the limited potential for fishing, no additional modeling was conducted on anglers using EFPC.

The findings of the noncancer risk assessment raise a number of risk management questions concerning the importance of the ORR releases of PCBs. In the cancer risk assessment, it is possible to separate the contribution of risk by source. However, the noncarcinogenic effects are influenced by the dose rates of PCBs received from other sources. It is useful to ask what the impact was of PCBs released from the ORR on anglers already exposed to other sources of PCBs. The analysis in Section 8.0 will examine this incremental risk by evaluating the risks associated with background sources of PCBs alone and the risks associated with both background and the PCBs released from the ORR.

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8.0 LEVEL III EVALUATION OF UNCERTAINTY AND VARIATION IN RISK ESTIMATES AMONG RECREATIONAL ANGLERS

Much attention has been given in recent years to the importance of separating variability and uncertainty in chemical risk assessment (Hoffman and Hammonds, 1994; Price et al., 1995; Frey and Rhodes, 1996). When these factors are treated separately, a risk manager has insight into the level of risk to various members of the exposed population, as well as an understanding of the level of confidence that can be attributed to those estimates. This section presents a quantitative assessment of the variation and uncertainty in noncancer risks posed by PCBs to recreational anglers (adults and children) fishing the Clinch River/Poplar Creek and Watts Bar Reservoir. A two-dimensional Monte Carlo model was used to separately characterize the uncertainty and variability associated with these risk estimates. This model incorporated the uncertainty associated with the toxicological threshold of PCB effects in the exposed population and available information on the uncertainty and variation in the estimate of exposure. In order to investigate the incremental impact from the ORR, we carried out two assessments for the recreational anglers. First, we performed an analysis assuming no release from the ORR. The analysis was then repeated with both background sources of PCBs and the ORR releases. We compared the two analyses and determined the incremental change in risk estimates associated with the ORR releases.

8.1 METHODOLOGY

The approach used in this analysis was an extension of the current method of characterizing noncarcinogenic risks used in Sections 6.0 and 7.0. The risks were characterized in terms of the fraction of the populations who received a dose rate of PCBs in excess of the population threshold of PCBs. The population threshold is defined as the highest dose rate that will not cause an adverse effect in any individual even those that are sensitive to the effects of PCBs. The population threshold is determined using a variation of EPA’s process for setting RfDs (Swartout et al., 1998). If the fraction of the population which is exposed to a dose rate greater than the population threshold, is zero, then there is no potential for adverse effects. If, however, that fraction of the population is greater than zero, then there is a potential for adverse effects.

If a person receives a dose in excess of the population threshold, it is not certain (or even likely) that they will experience adverse effects. Because an individual’s ability to tolerate exposure to PCBs varies, the personal threshold may be considerably higher than the population threshold. If this is the case, an individual may be unaffected by the dose that he/she receives. Therefore, the fraction of the population experiencing adverse effects will be smaller than the fraction receiving doses above the population threshold.

The probability of exceeding the population threshold is expressed as the ratio of the individual’s dose and the true but unknown population threshold. In this report, this ratio is referred to as the true hazard quotient. The project team investigated the uncertainty and variation in the dose rates in the populations of anglers and the uncertainty in those distributions as well as the uncertainty in the population threshold. The result was an estimate of the uncertainty and variation in the true hazard quotient.

The evaluation was conducted using a two-dimensional Monte Carlo analysis. The model was run with and without the ORR's contribution to PCB levels in Clinch River and Watts Bar fish and the incremental changes in the true hazard quotients were determined. As discussed in Section 7.0, Monte Carlo analysis is a numerical, simulation technique that uses information on the uncertainty or variation in the parameters of a model to estimate the uncertainty and variation in risk estimates. It is performed by randomly selecting parameter values from distributions that reflect either the uncertainty or variation in the specified parameter for the exposed population (Morgan and Henrion, 1990; Kangas, 1996). Once a value has been selected for each parameter, the resulting solution is calculated and saved. This process is repeated thousands of times and produces a distribution of results.

In the case of two-dimensional Monte Carlo analysis, the uncertainty and variation in each of the exposure parameters is separately characterized (Hoffman and Hammonds, 1994; Frey and Rhodes, 1996). A parameter in two-dimensional Monte Carlo can have either an uncertain component, a variable component, or both. Variability results from the fact that no single parameter value represents all individuals in a population. For example, no single value can represent the fish consumption rate for an entire population of anglers; rather individual anglers are likely to have a wide range of consumption rates. Therefore, this exposure parameter can only be described by a distribution or range of values including high rates for people who consume large amounts of fish, and low rates for people who consume small amounts of fish. In contrast, uncertainty results from insufficient knowledge of either the value of the parameter or the shape of a distribution of parameter values. For example, while there is one true distribution of fish consumption rates for Watts Bar, one cannot be certain about the exact distribution because surveys of the variation of intakes among these anglers have not been performed.

Developing uncertainty and variability distributions calls for a certain amount of professional judgement. The parameters fall into three categories; parameters that only have uncertainty, parameters with variability but little or no uncertainty, and parameters that are both variable and uncertain. There is a single true (but unknown) value for the estimate of the PCB threshold; thus, the threshold was only considered subject to uncertainty. For body weight, while there was substantial variability within populations, there was little uncertainty in those estimates because the variation in body weights has been well characterized; thus, only the variability component was considered for this parameter. Exposure parameters that had both uncertainty and variability components included fish consumption rates and cooking loss. The concentration of PCBs in fish had relatively little variation or uncertainty. Details of the distributions selected for both uncertain and variable parameters are provided in Section 8.1.1.

The two-dimensional Monte Carlo approach to the characterization of uncertainty and variability in noncancer risk estimates was based on the following equation:

$$tHQ = \frac{U_f \ x \ F_{cook} \ x \ FC_{PCB}}{BW \ x \ threshold}$$

Where:

tHQ	=	True hazard quotient;
U_f	=	Fish consumption rate (kg/d);
F_{cook}	=	Fraction of chemical retained after cooking (unitless);
FC_{PCB}	=	Concentration of PCBs in fish (mg/kg);
BW	=	Body weight (kg); and
Threshold	=	The highest dose that does not cause adverse effects in sensitive individuals (mg/kg-d).

Parameter distributions are described in Sections 8.1.1 and 8.1.2.

In this analysis, a two-dimensional Monte Carlo model that separately characterized uncertainty and variation was constructed using the nested loop approach described by Hoffman and Hammonds (1994). The approach used an iterative procedure (the uncertainty loop) to select values for the uncertain parameters. During each iteration of the uncertainty loop, the model used a second nested iterative procedure (the variation loop) that characterized the human variation in fish consumption exposure in the modeled populations. Figure 8-1 presents a flow chart for the model.

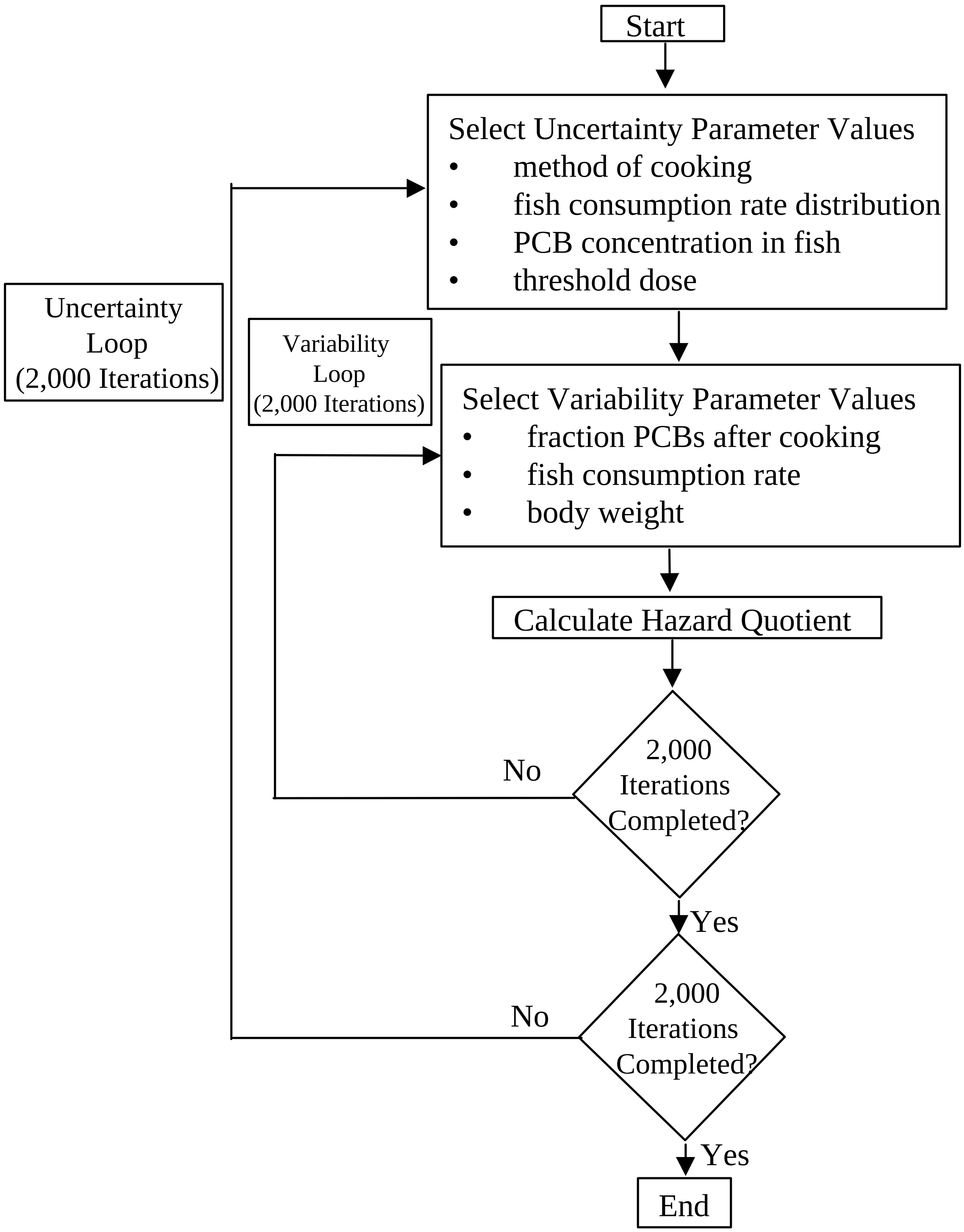
The model begins by selecting a set of values for the uncertainty parameters for the first iteration of the uncertainty loop, including the distributions for fish consumption rates, PCB levels in fish, body weight, fraction of PCBs remaining after cooking (based on the cooking method selected), and the value of the threshold. Once these values and parameter distributions are selected, the model enters the variation loop.

The variation loop then models a distribution of dose rates in the exposed population based on random selection of the available values for each parameter as determined by the distributions and values defined in the uncertainty loop. In each iteration of the variation loop, the model randomly extracts values from the distributions that describe the variability in individuals' fish consumption rates, fraction of PCBs remaining after cooking, and body weight.

The selected values for the exposure parameters are then used to calculate a dose rate for the individual consuming fish from the specified water body. This dose is then divided by the value of the threshold dose selected in the uncertainty loop to derive a true hazard quotient for that iteration of the variability loop. This process is repeated for 2,000 iterations of the variability loop before returning to the uncertainty loop to begin the next iteration. The results of the model are 2,000 sets of modeled true hazard quotients for 2,000 anglers or 4,000,000 separate estimates of true hazard quotients.

The output of the model does not include the results of all 4,000,000 true hazard quotients, due to the associated data management difficulties. Rather, each time the end of the uncertainty loop is reached, the model determines the minimum value, maximum value, mean, and selected percentiles (5^{th} to the 95^{th}) for the results of the 2,000 iterations of the variability loop. These values are stored for the final output, and the next uncertainty loop, including 2,000 iterations of the variability loop, is initiated. Thus, the model's output is a set of 2,000 values for the

Figure 8-1. Two-Dimensional Monte Carlo Analysis



minimum, maximum, mean, and selected percentiles which represents the uncertainty in each of these estimates.

The output of the model is presented in graphic and tabular forms. The graphic description is in the form of a cumulative distribution of interindividual variation in true hazard quotients. As described by Hoffman and Hammonds (1994), if all the runs of the inner loop were graphed as cumulative distributions, the result would appear as a fuzzy band rather than a single curve. The width of the band would be a measure of the uncertainty in the estimates of the true hazard quotients for each percentile of the population. As stated above, the model saves data on selected percentiles of the variation distribution. The uncertainty in these percentiles is a measure of the width of the uncertainty band at the percentiles. This uncertainty can be characterized by selecting the median and 90 percent confidence limits of the values of the percentiles. The graphic form of the model output consists of a plot of three cumulative distributions. The first distribution is the set of estimates of the median values of the 5th through the 95th percentiles of the distribution of chronic exposures of the population. The remaining two distributions are the upper and lower 90th percent confidence limits (the 5th and 95th percentiles of the uncertainty distribution) of the estimates of the true hazard quotient for each percentile. The tabular form presents the true hazard quotients that correspond to selected percentiles of the uncertainty and variation distributions.

As discussed above, the PCBs in the fish in the Clinch River and Watts Bar occurred as the result of releases from a number of sources other than activities at the ORR. In order to determine the relative contribution from the ORR, we have investigated the noncancer risks that would have occurred had the ORR not released any PCBs, and how those risks would have increased as a result of the ORR releases.

Two other sources of uncertainty, scenario and modeling uncertainty, were not addressed in the level III evaluation (EPA, 1992b). Scenario uncertainty is the uncertainty in defining the pathways with which individuals are exposed. An example of this may be a population that consumes fish in a poorly understood fashion that is not reflected in the distribution of intake. Model uncertainty is the uncertainty in the models used to characterize dose and response. An example of this may be the presence of some factor in fish that ameliorates or exacerbates the effects of PCBs. These issues can not be readily quantified in the uncertainty analysis. Future assessments of PCB exposure should consider these sources.

8.1.1 Uncertainty and Variability in Exposure Parameters

The characterization of the uncertainty and/or variability in the parameters used in the above equation are presented in Table 8-1 and are discussed in this section.

Fish Consumption Rate (U_f)

In this model, the fish consumption rate is viewed as having both uncertainty and variability. As discussed above, fish consumption varies from angler-to-angler. The uncertainty stems from the fact that studies of fish consumption practices have not been conducted on either Watts Bar and Clinch River/Poplar Creek angler populations.

Table 8-1. Exposure Parameter Distributions Used in the Two Dimensional Analysis of the Recreational Angler

Model Inputs	Distribution Type	Minimum	Mode	Maximum	Source
VARIABILITY					
Fish consumption rate (kg/day)[a]	Cumulative				
Clinch River/Poplar Creek					
minimum distribution		0.000001		0.12	Ebert et al., 1993
maximum distribution		0.0006		0.58	TWRA, unk; Ebert et al., 1993; USDOI, 1993; Connelly et al., 1996
Watts Bar					
minimum distribution		0.000023		0.217	Ebert et al., 1993
maximum distribution		0.001		0.97	TWRA, unk; Ebert et al., 1993; USDOI, 1993; Connelly et al., 1996
Fraction of PCBs remaining after cooking (unitless)	Triangular	0.4	Selected in uncertainty section	0.8	ChemRisk, 1992; Connelly et al., 1992; Sherer and Price, 1993
Body weight (kg)	Cumulative				
adult		35		130	EPA, 1989a
child (6-10)		18		65	EPA, 1989a
PCB concentration in fish (mg/kg)	Triangular				Based on a compilation of available data
Clinch River/Poplar Creek [b,c]		0.20	0.66	1.2	
Watts Bar [b]		0.29	1.17	1.35	
UNCERTAINTY					
Fish consumption rate (kg/day)	Triangular	Nth percentile from the lower bound fish consumption rate distribution	Nth %tile from the upper bound fish cons. rate dist.	Nth percentile from the upper bound fish consumption rate distribution	Professional judgement
Mode of the fraction of PCBs remaining after cooking (unitless)	Uniform	0.4		0.8	Professional judgement
Reference Dose (μg/kg-day)$^{-1}$					
Reference	Cumulative	0.0046		0.41	Swartout et al., 1998
Empirical	Cumulative	0.000044		13	Gillis et al., 1997; Schmidt et al., 1997; Swartout et al., 1997
Refined Empirical	Cumulative	0.027		14	Gillis and Price, 1996

a. Adult fish consumption rate distribution is presented. The fish consumption rate for a child (6-10 years old) is based on the adult consumption rate as follows: Child fish consumption rate = Adult fish consumption rate x child body weight x 1.3/70

b. For recreational anglers, all game species as indicated on Table 5-8 were included.

c. Longnose gar were not included; personal communication with TN Department of Fisheries (8/96) indicated that this fish is very infrequently consumed.

The upper-bound distribution of estimated consumption rates for anglers using Watts Bar was based on site-specific creel data and information on the coefficient of variation reported in surveys of recreational anglers conducted in other areas of the U.S. (as discussed in Section 7.1.1.2). To define the lower uncertainty bounds for these estimates, the project team selected a study of fish consumption rates of freshwater recreational anglers conducted elsewhere. For Watts Bar anglers, the lower bound distribution was derived from consumption rates for consuming anglers in Maine (Ebert et al., 1993). Table 8-1 and Figure 8-2 present the bounding distributions for Watts Bar. The angler survey from Maine is believed to underestimate the consumption rate for Watts Bar anglers, because the colder weather in Maine results in a shorter fishing season than that of Watts Bar. In addition, Rupp et al. (1980) have demonstrated that consumption of freshwater fish in the northeastern region of the U.S. is generally lower than consumption in the southeastern region. The upper bound distribution likely overestimates intakes due to the conservative assumptions used in deriving that distribution.

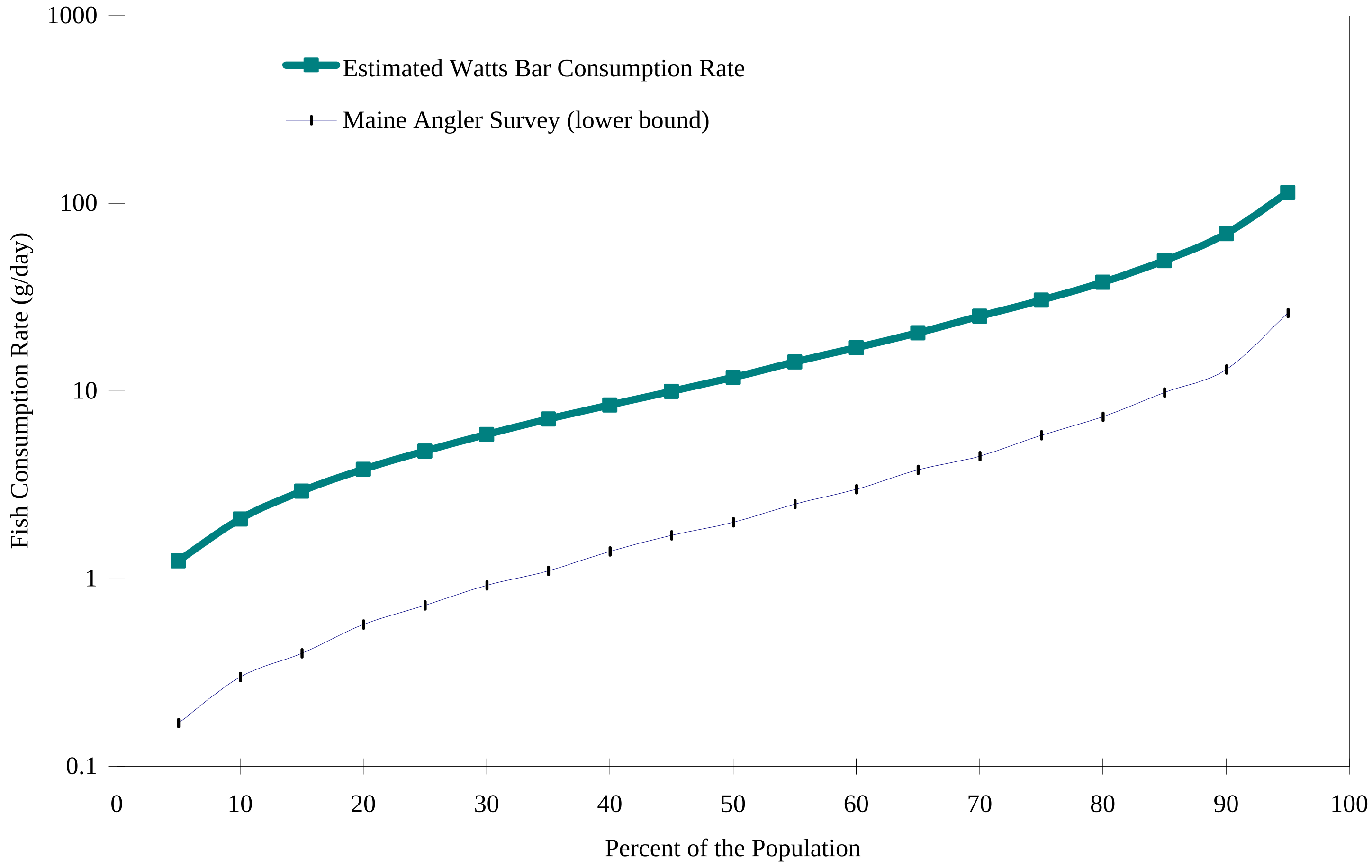
For Clinch River/Poplar Creek anglers, the distributions were developed in a similar manner. Estimated upper bound consumption rates were based on Watts Bar estimates, as discussed in Section 7.1.1.2. Consumption rates for consuming anglers from Maine who fished rivers and streams were used to derive the lower bound distribution. The consumption rates provided by these studies were reduced by 40 percent to reflect the reduced capacity of the fishing resource at Clinch River/Poplar Creek compared to Watts Bar.

The project team assumed that the actual distribution of consumption rates for these two populations fell between the two bounds, but was more likely to fall near the upper bound. Therefore, the probability of the percentile falling at an intake between the two bounds was modeled using a triangular distribution with a maximum and mode equal to the upper bound and the minimum equal to the lower bound. At the beginning of each uncertainty loop the model was required to select a distribution of fish intakes. This distribution was generated by using a series of points that fell at the same proportional distance between the two bounding estimates, based on a value from 0 to 1.0. For example, if the value selected was 0, the lower bound distribution was selected for that iteration of the uncertainty loop. A selected value of 1.0 resulted in use of the upper bound distribution. Selection of a value of 0.5 resulted in the use of the distribution of values that fell at the midpoint between the upper and lower bound distributions. Once a value between 0.0 to 1.0 was selected in the uncertainty loop, the corresponding distribution of interindividual variation in fish consumption rate was used in the variability loop. The distribution of values was triangular with a minimum of 0 and a maximum and mode of 1.

PCB Concentration in Fish (FC_{PCB})

While the levels of PCBs vary from fish to fish, over long periods of time anglers, and especially those with high fish consumption rates, consume hundreds of fish. Consumption of large numbers of fish tends to average out the inter-fish variation so that the PCB concentration consumed over time closely resembles the mean concentration. However, anglers that favor one species over another tend to have different long-term average concentrations of PCB in the fish they consume. This range of concentrations can be conservatively estimated based on the range of species-specific average concentrations (see Section 7.1.2.2). There is some uncertainty in

Figure 8-2. Uncertainty in Fish Consumption Rate for Watts Bar



this distribution. However, given the relatively small differences between the different species, the uncertainty is unlikely to have a significant impact on the assessment. As a result, the distribution of fish concentrations was modeled as being variable but not uncertain. Therefore, as described in Section 7.0, weighted averages were derived for each commonly consumed game species for each specified water body by combining relevant data from all available investigations. The minimum, mode and maximum reported values for each water body were used in a triangular distribution to describe the variability in the long-term average concentration of PCBs for the recreational anglers on those bodies of water.

Fraction of PCBs Remaining After Cooking (F_{cook})

Published data indicate that the percent of PCBs remaining in the fish after cooking varies from 80 to 40 percent depending on the cooking method used (ChemRisk, 1992; Connelly et al., 1992; Sherer and Price, 1993; Harrington et al., 1998). As a result, the range of PCB concentrations ingested by anglers evaluated in this analysis was defined by the cooking method selected for each iteration. Because most anglers will vary their choice of cooking method from meal-to-meal based on the species consumed, it is unlikely that anglers would consistently choose the same cooking method. Thus, the project team modeled the variation in this parameter as a triangular distribution with fixed minimum and maximum values. Because the mode of the distribution was viewed as being uncertain, the model allowed the mode to vary with equal probability among all possible values between the minimum and the maximum values of the distribution. The value of the mode was selected in the uncertainty loop and the resulting distribution of cooking loss across anglers was used in the subsequent variation loop.

Body Weight (BW)

The variability associated with the body weight distribution was also characterized in this model. The adult and child body weight distributions were represented by cumulative distributions provided by the *Exposure Factors Handbook* (EPA, 1997). Because this distribution was based on data collected from large numbers of individuals, it was assumed that the uncertainty in the distribution was minimal and did not warrant inclusion in the model.

8.1.2 Uncertainty in the PCB Threshold Dose

Background

Currently, regulatory agencies evaluate the risks of noncarcinogenic effects using a system that is based on the RfD. The RfD is defined by the USEPA (Barnes and Dourson 1988; EPA, 1989a) as "an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without appreciable risk of deleterious effects during a lifetime". That is, the RfD is considered to be at or below an exposure level associated with negligible risks in the human population ("negligible risk level"). For compounds with thresholds (minimum dose levels required to produce a given effect), the RfD has been described as a dose that has a high probability of being below the population threshold.

The RfD is derived using the following ratio:

$$\frac{NOAEL}{UF \ x \ MF}$$

where the NOAEL is the no-observed-adverse-effect-level in a population of test animals, UF is a composite safety[4] factor representing multiple safety factors, and MF is a situation-specific modifying factor.

The safety factors in the RfD address both the uncertainty associated with extrapolating from the results of toxicology studies conducted in laboratory animals to draw conclusions about effects in humans, and uncertainty arising from limitations in the available toxicological data. Primary safety factors are interspecies variation and interindividual variation (Table 8-2). The secondary safety factors are used to account for LOAEL to NOAEL extrapolation, subchronic to chronic extrapolation, and database uncertainty. Each safety factor can be defined as a loose approximation of the upper bound of the distribution of dose ratios associated with different toxicological endpoints and chemicals (Swartout et al., 1998). Safety factors have historically been assigned values of either ten or three.

The establishment of an RfD requires the consideration of both variability and uncertainty in the estimate of human toxicological responses to chemicals. Variability in human response must be addressed in order to derive an estimate of a dose that is sufficiently low so as to be protective of individuals who are particularly sensitive to a compound. Uncertainty must be addressed because estimates of threshold in humans, which are based on NOAELs in test-animal studies, require a number of imprecise and uncertain extrapolations. Therefore, the RfD must be set low enough such that there is little chance that it will be above the true threshold. As a result, the RfD can be thought of as the lower confidence limit on a compound’s threshold in humans.

If the existing values of the safety factors are replaced with distributions that reflect the inter-chemical variation in the appropriate ratios, the result will be an uncertainty distribution for the threshold of a compound (Swartout et al., 1998). This distribution should not be viewed as a representation of the uncertainty in the RfD, but rather as an estimate of the true but unknown threshold where the RfD is some point on the lower end of the distribution.

Toxicological Data on the RfD

The RfD for Aroclor 1254 was used in this assessment to characterize the uncertainty in the noncancer effects of PCB. As discussed in Section 4.0, the Aroclor 1254 RfD is based on a chronic study in which rhesus monkeys were fed capsules containing 0, 5, 20, 40, or 80 µg/kg-d of Aroclor 1254 (Tryphonas, 1989; 1991a,b; Arnold, 1993a,b). Effects observed in the monkeys, and which provide the basis for the RfD include: ocular exudate, inflamed and prominent meibomian glands, distortion in the growth of nails, and decreased immune response to sheep erythrocytes. The dose of 5 µg/kg-d was identified as a LOAEL; however, no NOAEL was

[4] The term “safety” factor will be used in this report in place of the more common term “uncertainty” factor. This change in nomenclature is solely to avoid confusion with the other uses of the term “uncertainty” in this report.

Table 8-2. Toxicological Extrapolation Associated with Each of the Safety Factors

Safety Factor	Measured Endpoint	Estimated Endpoint
Data Base (UFD)	NOAEL in any chronic study	The lowest NOAEL observed in a set of chronic and reproductive (or developmental) toxicity studies
LOAEL (UF_L)	LOAEL in a study	NOAEL in a study
Subchronic (UF_S)	NOAEL in a subchronic study	NOAEL in a chronic study
Interspecies (UF_A)	NOAEL in a test species	NOAEL in a typically healthy human population
Interindividual (UF_H)	NOAEL in the general population	NOAEL in a population of sensitive individuals

(Barnes and Dourson, 1988; Jarebeck, 1994; Dourson, 1994) identified. In deriving the RfD, one safety factor of 10 (for sensitive humans, UF_H) and three factors of three each (for interspecies extrapolation, UF_A; LOAEL to NOAEL extrapolation, UF_L; and subchronic to chronic extrapolation, UF_S) were applied to the LOAEL, resulting in an RfD of 0.02 µg/kg-d or 2×10^{-5} mg/kg-d (IRIS, 1998b).

Characterizing the Uncertainty in the PCB Threshold

In this analysis, three safety factor distributions were derived for the population threshold of PCBs. Each of the three distributions was calculated based on the equation previously described and distributions for each of the four safety factors used in setting the Aroclor 1254 RfD (UF_H, UF_A, UF_L, and UF_S). Distributions of interchemical variation have been proposed for the interspecies sensitivity, interindividual sensitivity, and subchronic to chronic extrapolation factors (Gillis et al., 1997; Schmidt et al., 1997; Swartout, 1998); no distribution has been proposed for the LOAEL-NOAEL extrapolation.

The project team investigated whether the LOAEL-NOAEL factor could be replaced by using a benchmark dose approach to estimate the NOAEL (Crump, 1984). Benchmark dose modeling allows for the prediction of an acceptable animal dose (i.e., a substitute for the NOAEL) using available dose-response data. To investigate this, benchmark dose model for quantal data (THRESH, ICF Kaiser Engineers, Inc., K.S. Crump Group) was used to analyze the Arnold et al. (1995) data on reproductive effects in rhesus monkeys. These data do not monotonically increase. In fact, for several endpoints the 80 ng/kg-day dose group (the next to the highest of the four dose groups) appears to be a NOAEL. As a result, the software could not find a statistically valid method of fitting the data of Arnold et al. (1995); therefore, benchmark dose modeling could not be used to evaluate the study's results.

In the absence of a distribution or a benchmark dose to replace the LOAEL, this analysis used EPA's safety factor value of three as a point estimate to extrapolate from the LOAEL to the NOAEL; thus, the uncertainty in this extrapolation was not represented by a distribution. The equation used for calculating the population threshold for PCBs was as follows:

$$PCB_{threshold} = \frac{NOAEL_e}{UF_A \; x \; UF_H \; \; x \; UF_S}$$

Where:

$PCB_{threshold}$	=	The population threshold of PCBs
$NOAEL_e$	=	Estimated no-observed-adverse-effect level equal to the LOAEL in the test animal divided by 3
UF_A	=	Interspecies extrapolation distribution
UF_H	=	Interindividual variation distribution
UF_S	=	Subchronic to chronic variation distribution

The result of this calculation was a distribution of the uncertainty in the estimate of the threshold of adverse effects of PCBs in humans. Sensitive individuals receiving doses above this threshold were considered to be at risk for some level of adverse effects, although the nature, severity or frequency of the adverse effects were not known. It could be assumed, however, that if an individual was very sensitive to PCBs, then a dose slightly above the threshold could cause adverse effects, less sensitive individuals would require higher doses to be affected.

8.1.3 Safety Factor Distributions

This assessment examined three sets of distributions (Figure 8-3). The first set was the "reference" distribution, which was based on conceptual information about the historical design and use of safety factors (Swartout et al., 1998). The second set was comprised of "empirical" distributions, which were data-derived distributions designed to apply to any chemical. The third set was based on the empirical distributions but was tailored to PCBs by the application of PCB-specific toxicological data, resulting in a set of "refined empirical" distributions. Both the first and second sets of distributions are generic, i.e., they are applicable to any chemical. Such generic approaches do not reflect a consideration of chemical-specific toxicological information. Each set of distributions is described below.

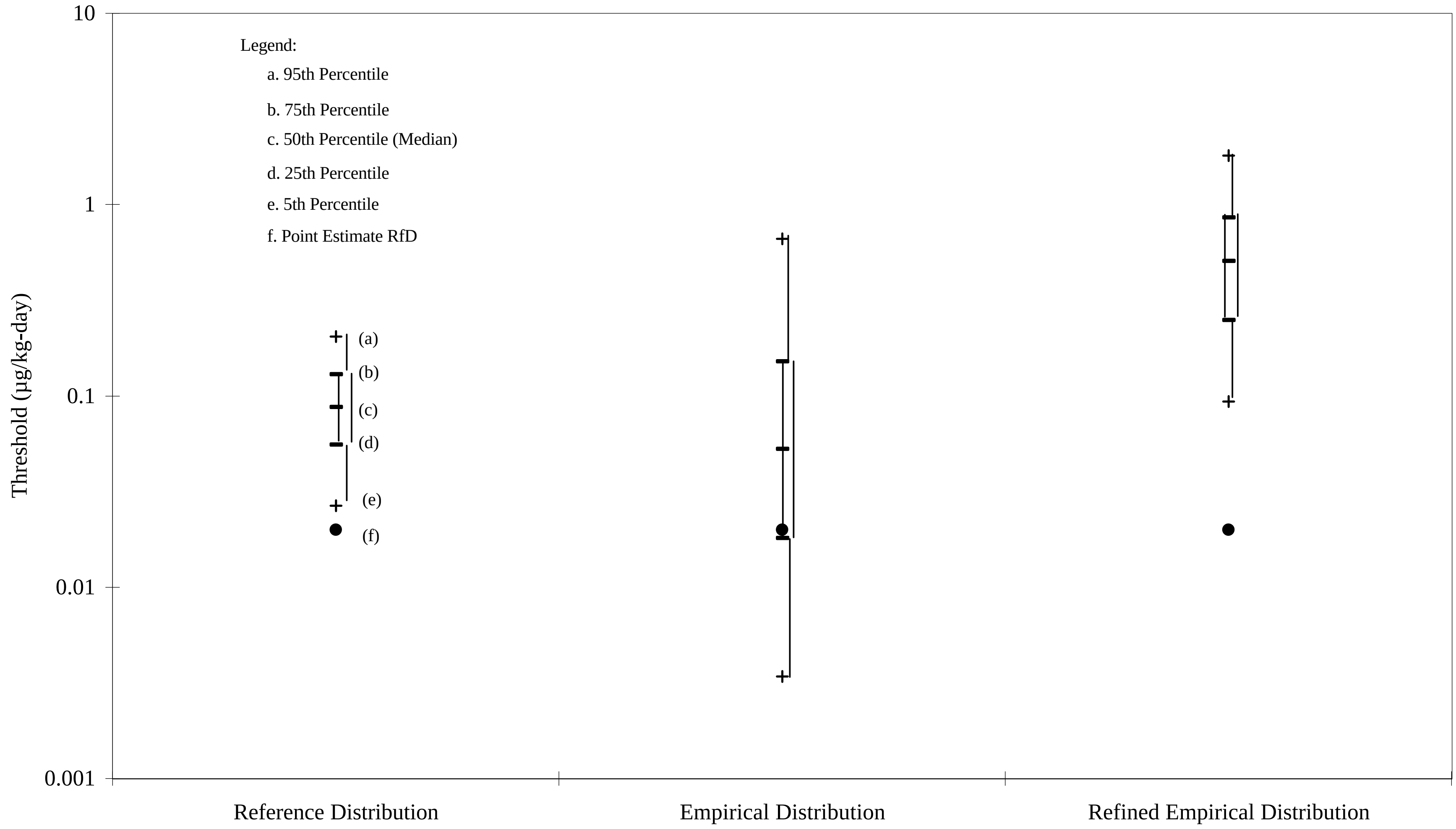
Reference Distribution. First, the safety factors were characterized using a generic "reference" distribution proposed by Swartout et al. (1998). This reference distribution was developed based on the existing concepts underlying the current system of values for safety factors (i.e., that the value of 10 is a loose upper bound, and that the UF cannot be smaller than 1.0). Using these concepts, and the understanding that toxicological data are generally believed to be lognormally distributed, Swartout et al. (1998) proposed a three-parameter lognormal distribution with a geometric mean of 2.16, a geometric standard deviation of 2.38, a 95th percentile value of 10, and an offset (t) of 1.0. This distribution was used for UF_H.

Because EPA used safety factors of three for both subchronic and interspecies effects, the project team modified the reference distribution derived by Swartout et al. (1998) such that the 95th percentile was equivalent to three, with a lower bound of one. The resulting distribution had a geometric mean of 2.43 and a geometric standard deviation of 1.18. This distribution was used for UF_A and UF_S.

Empirical Distributions. Preliminary empirical distributions for UF_A, UF_H, and UF_S developed by Gillis et al. (1997), Schmidt et al. (1997), and Swartout (1998) were used in a separate evaluation of uncertainty in the RfD. These preliminary distributions were designed to apply to a multitude of chemicals. Schmidt et al. (1997) developed preliminary distributions for UF_A based on animal-to-human ratios of toxicological endpoints for antineoplastic agents. Distributions were developed for mouse-, rat-, dog-, and monkey-to-man ratios. The monkey-to-man distribution was used in this assessment, because the RfD for Aroclor 1254 is based on a monkey study. The ratios in this distribution ranged from 0.1 to 17, with a reported mean value of 3.5. The distribution was simulated as a cumulative distribution using all of the individual ratios.

Gillis et al. (1997) developed an empirical distribution for UF_H based on ratios of highest to lowest therapeutic doses for pharmaceutical agents randomly selected from the Physicians Desk Reference. The therapeutic range was assumed to represent one geometric standard deviation (GSD) in the distribution of human responses to chemicals. As discussed by Gillis et al. (1997), estimates of the GSD were used to derive a distribution for UF_H. The resulting distribution had

Figure 8-3. Distributions of PCB Thresholds



values that ranged from 1.2 to 64 with a median value of nine. The distribution was simulated as a cumulative distribution using the individual ratios.

Swartout (1998) developed a distribution for the exposure duration uncertainty factor (UF_S) based on ratios of subchronic to chronic toxicological endpoints. Geometric mean NOAELs and LOAELs from subchronic studies were compared with those of chronic studies for a variety of chemicals (mostly pesticides). The resulting ratios formed the basis for a lognormal distribution for UF_S with a geometric mean of 2.17 and a geometric standard deviation of 3.16. The ratios ranged from 0.1 to 57, with a 90th percentile of 10 and 95th percentile of 15. The distribution was simulated as an unbounded lognormal distribution with the aforementioned parameters.

Refined Empirical Distributions. In order to develop the most appropriate distributions for PCBs in this analysis, it was critical to take into account the toxicological data that were available on PCBs. This was done by revising the generic empirical distributions to reflect specific knowledge about PCBs. The distribution for animal-to-man extrapolation (UF_A) was altered based on information indicating that monkeys are generally more sensitive to the reproductive effects of PCBs than humans (Gillis and Price, 1996). Given this information, the distribution for UF_A was assumed to reach a maximum at one (1) (indicating that humans and monkeys are equally sensitive). Due to the paucity of data points below 1 in the antineoplastic toxicity database, it was not possible to use the original distribution truncated at the value of 1. Rather, the preliminary distribution was parameterized as lognormal, and the resulting lognormal distribution was truncated to values below 1.

The subchronic to chronic UF_S safety factor was set equal to 1. The subchronic to chronic safety factor examines the impact of increased duration of dosing on test animals. It is intended to account for the uncertainty introduced by using the results from short-term studies to predict the risks from long-term exposures. The reason for this is that toxicologists believe that the risk of reversible effects of PCBs, such as the dermal, ocular and acute reproductive effects, are a function of the average body burden of the compound over the period of time necessary to cause the effect. In the case of the rhesus monkey study, the animals had reached quasi-steady state (Arnold, 1993a,b); thus, increasing the duration of exposure would not have changed their body burdens or observed effects.

The distribution for UF_H, which addresses interindividual differences in responsiveness to chemicals, was not modified. This was because there is little available information regarding interindividual variation in response to PCBs.

Figure 8-3 presents the PCB thresholds developed using the three types of distributions and shows that the estimates of the thresholds derived from the three sets of distributions overlapped. In all three cases, the distributions included dose rates that were higher than the established RfD. The reference distribution gave the most narrow confidence limits and the refined empirical distributions gave higher estimates of the threshold than the other distributions.

The characterization of uncertainty in the PCB threshold is an evolving field of study, and the data available with which to develop empirically-based safety factor distributions are not perfect. Thus, the reference distribution was developed as an initial means of quantifying the concepts underlying the RfD derivation. While the empirical distribution had some empirical support and represented the information currently available on the uncertainty in "generic" chemicals, the refined empirical distribution took into account toxicological data specific to PCBs. As a result, the project team viewed that distribution to be more reasonable and scientifically-defensible representation of the uncertainty in the threshold for PCBs than either the reference or empirical distributions.

8.2 CHARACTERIZING INCREMENTAL RISK

We modeled the incremental risks associated with the ORR releases to anglers using the Clinch River and Watts Bar in the following way. First, the models were performed using the fish concentrations discussed in the previous section. The resulting estimates of the true hazard quotients reflect the total amount of PCBs in the fish that occurred as a result of all sources of release. To determine the risks associated with non-ORR sources (background), the levels in the fish in Watts Bar were reduced by 13 percent and the levels in fish in the Clinch River by 50 percent. These levels were based on the conservative estimates of the relative contribution of the ORR releases to these bodies of water. We recalculated the risks for these lower concentrations and determined the differences between the two estimates.

Figures 8-4 to 8-9 present the uncertainty and variability of the estimated true hazard quotients for the adults in each population evaluated. For each population, the cumulative distribution associated with the 5th, 50th (i.e., median), and 95th percentiles of the uncertainty in the true hazard quotients for selected percentiles of population are presented. The distance between these curves along the x-axis represents the uncertainty in the estimate of the true hazard quotients. The 50th percentile curve represents the median estimate, indicating that there is an equal probability that the true cumulative distribution of true hazard quotients could be above or below the estimated curve. In this sense, it can be viewed as an unbiased estimate. The 5th and 95th percentile curves can be viewed as the upper and lower 90 percent confidence limits of the estimate of the true hazard quotient of each percentile. The smaller the distance between the 5th and 95th percentile curves, the less uncertainty is associated with the median estimate of the true hazard quotient distribution. For example, in Figure 8-4, the distance between the distribution curves for the 5th and 95th percentile spans approximately one order of magnitude.

The variation associated with each of the distribution curves is depicted along the y-axis. For example, in Figure 8-4, the variation in the true hazard quotients across the population (50th percentile) ranges from less than 0.1 to almost 10. The fraction of the population who will have true hazard quotients of a certain value also can be determined from these graphs. However, the estimates of the fraction of the population and the degree that individuals exceed the population threshold are very uncertain. This uncertainty is due both to the uncertainty in the estimate of dose and the estimate of the population threshold. The size of the uncertainty in the population threshold is also uncertain as reflected in the differences in the estimates of true hazard quotients that were obtained using the three sets of safety factor distributions. While all three sets of safety factor distributions have some justification, the project team believes that the refined empirical distribution is the most appropriate for PCBs because the distribution uses compound-specific data. Consequently, the following sections focus on the results of the analyses developed using the refined empirical distribution.

Figure 8-4. Watts Bar Adult Recreational Angler - Reference RfD Distribution

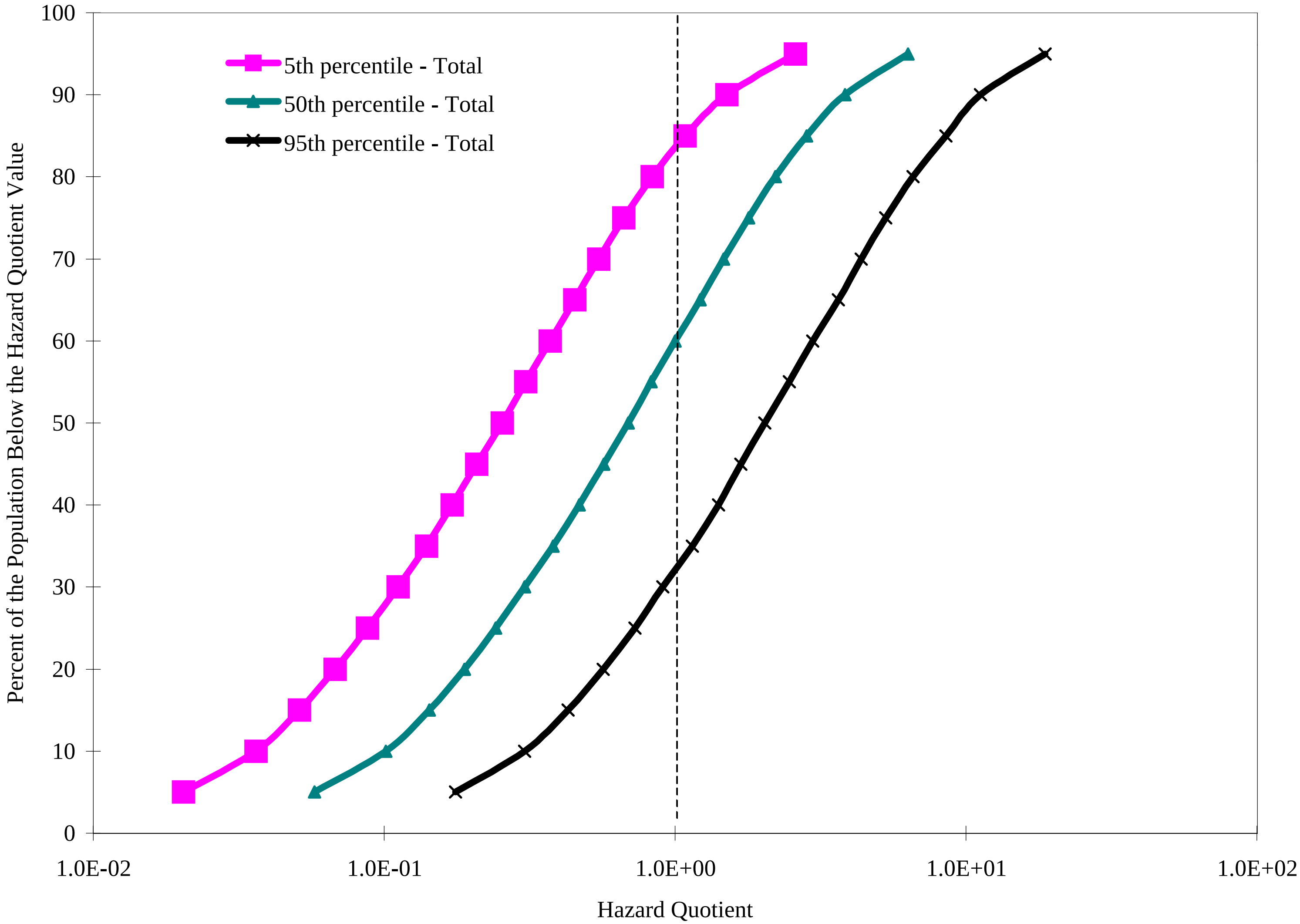


Figure 8-5. Watts Bar Adult Recreational Angler - Empirical RfD Distribution

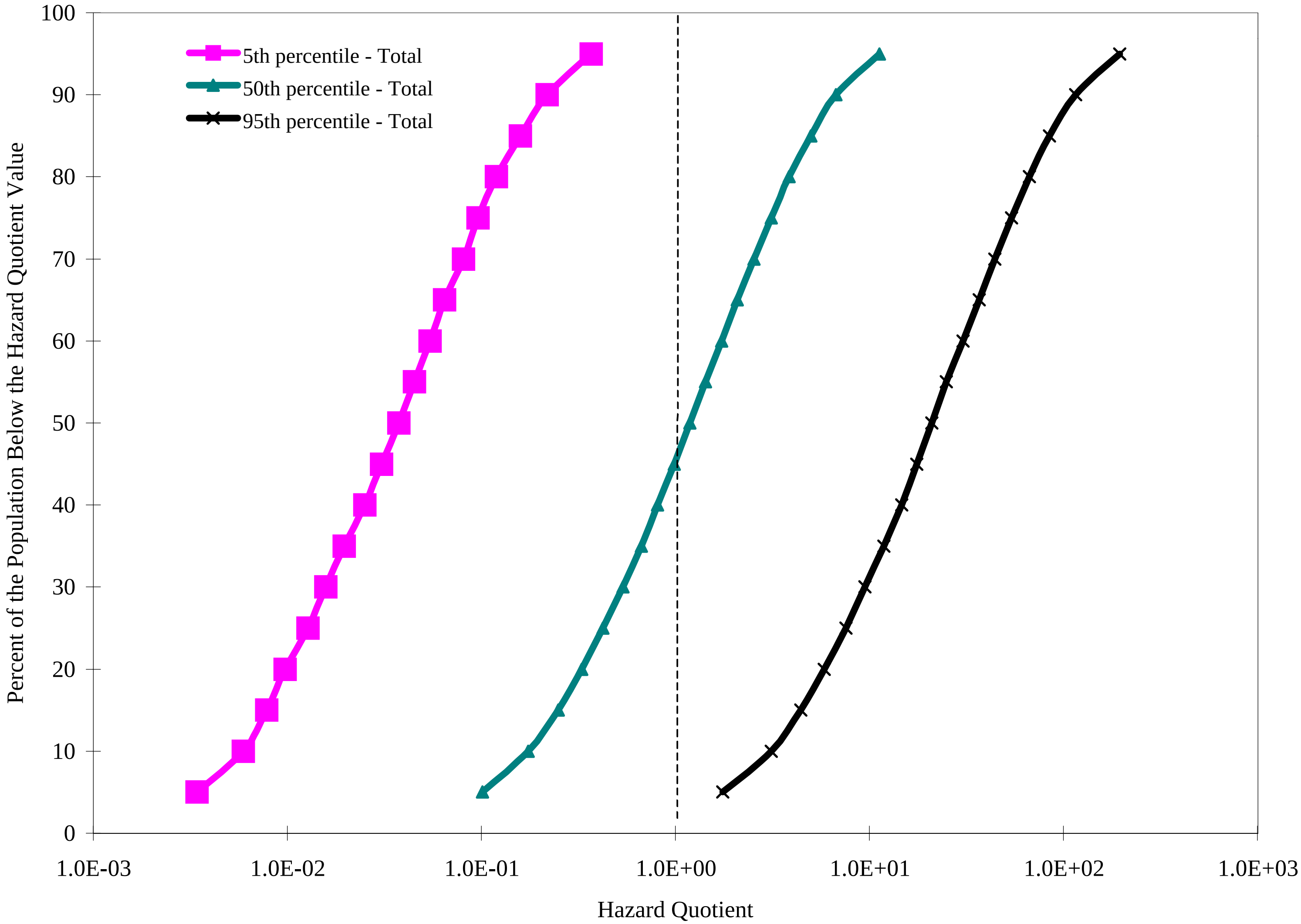


Figure 8-6. Watts Bar Adult Recreational Angler - Refined Empirical RfD Distribution

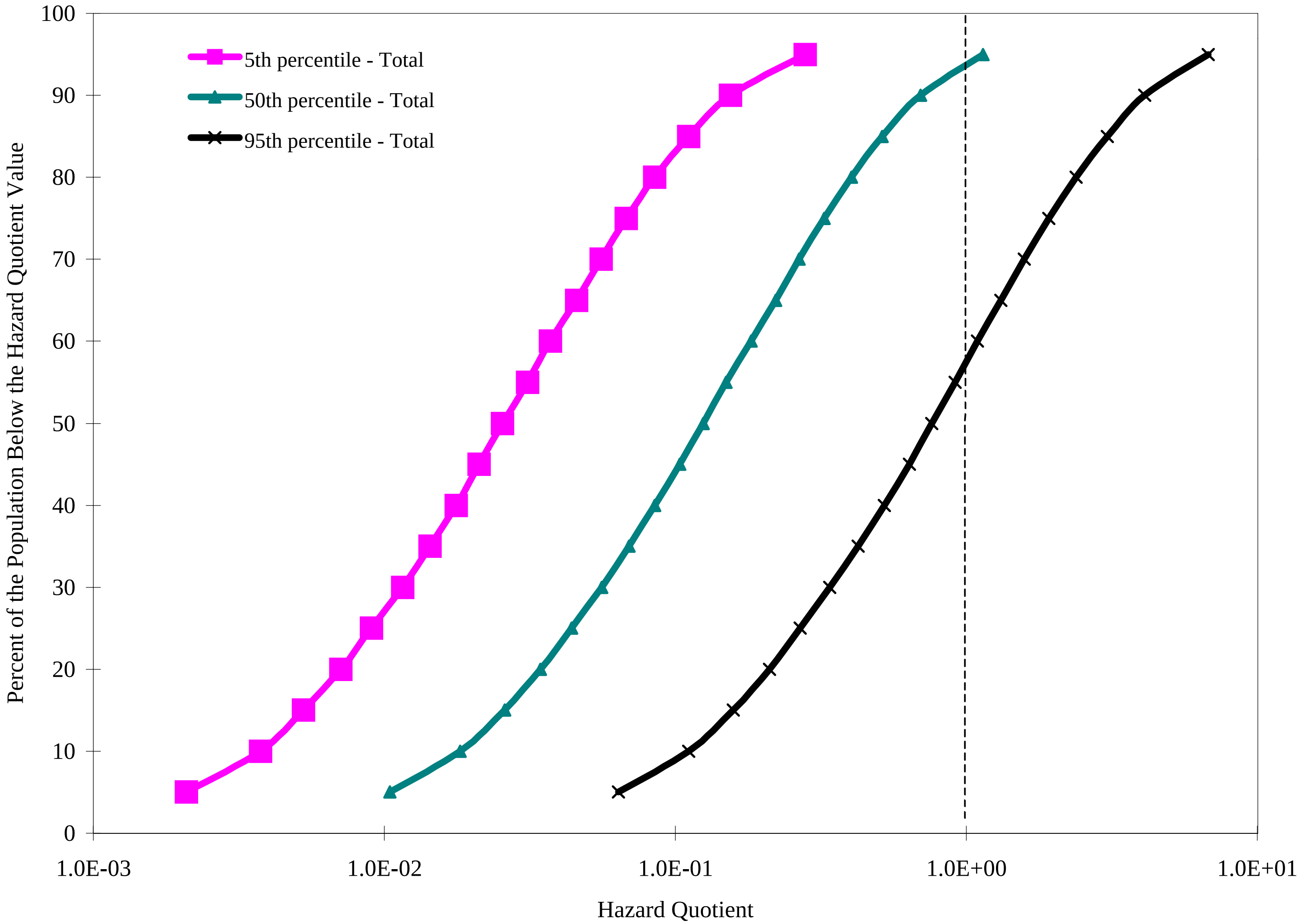


Figure 8-7. Clinch River Adult Recreational Angler - Reference RfD Distribution

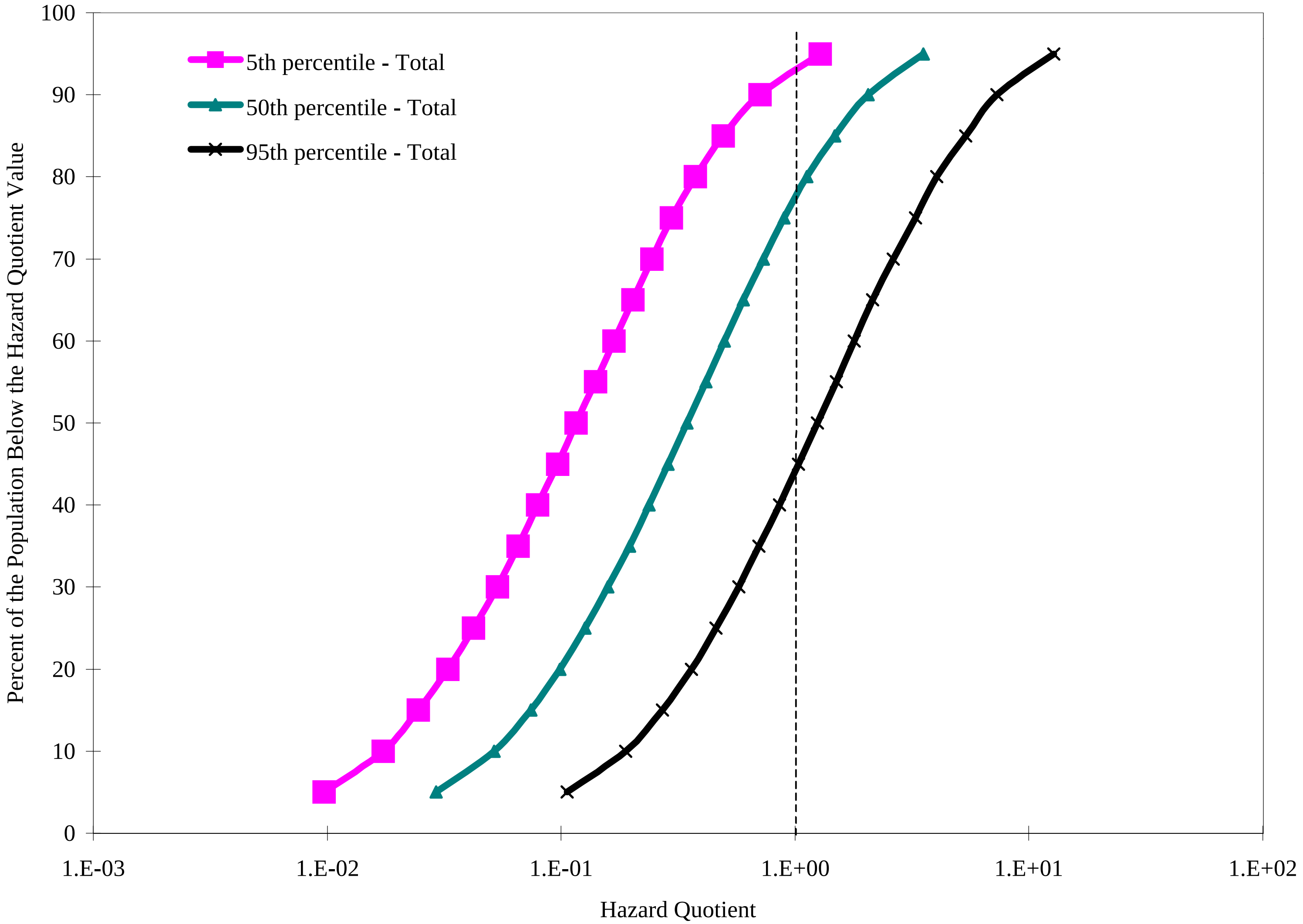


Figure 8-8. Clinch River Adult Recreational Angler - Empirical RfD Distribution

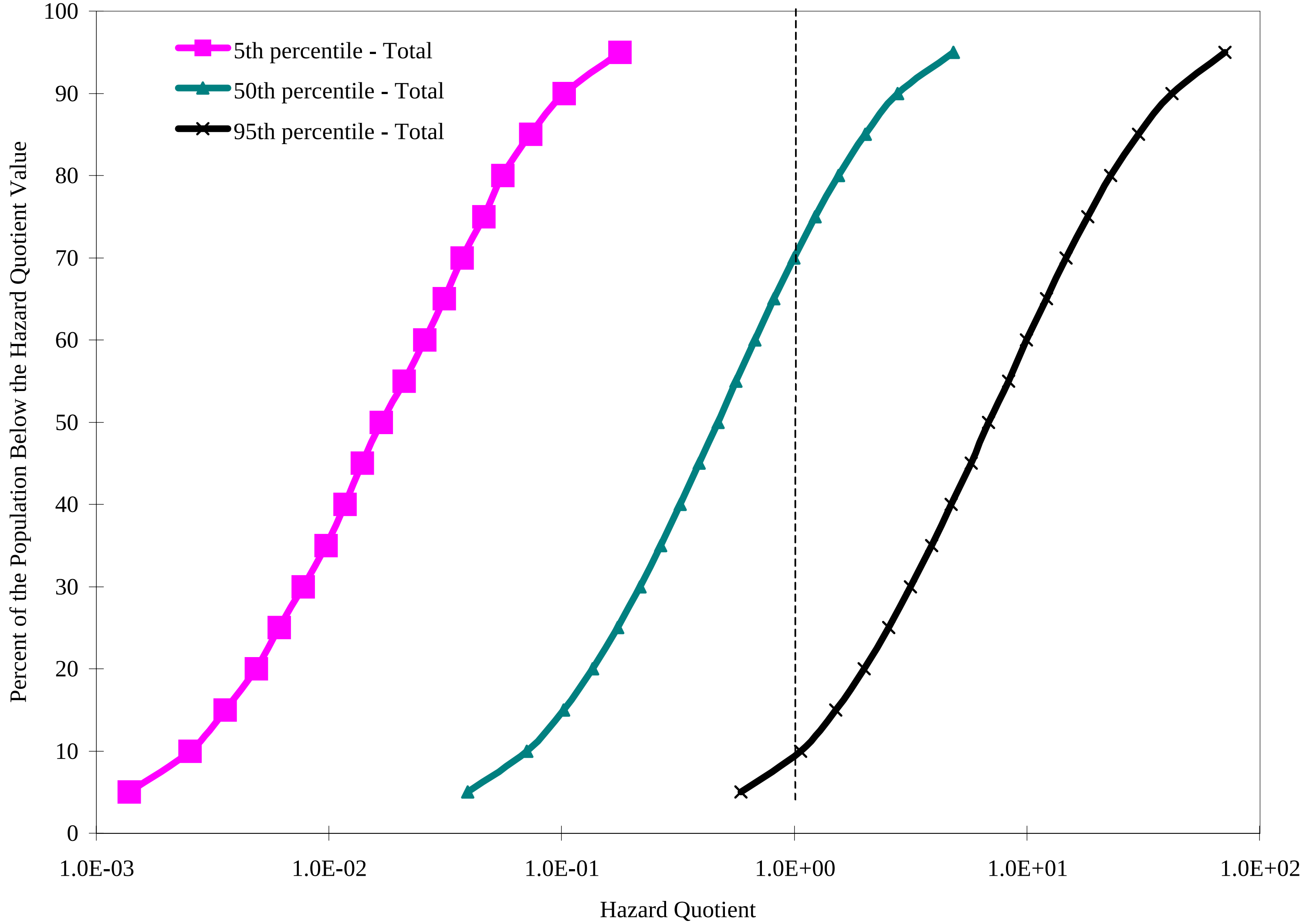
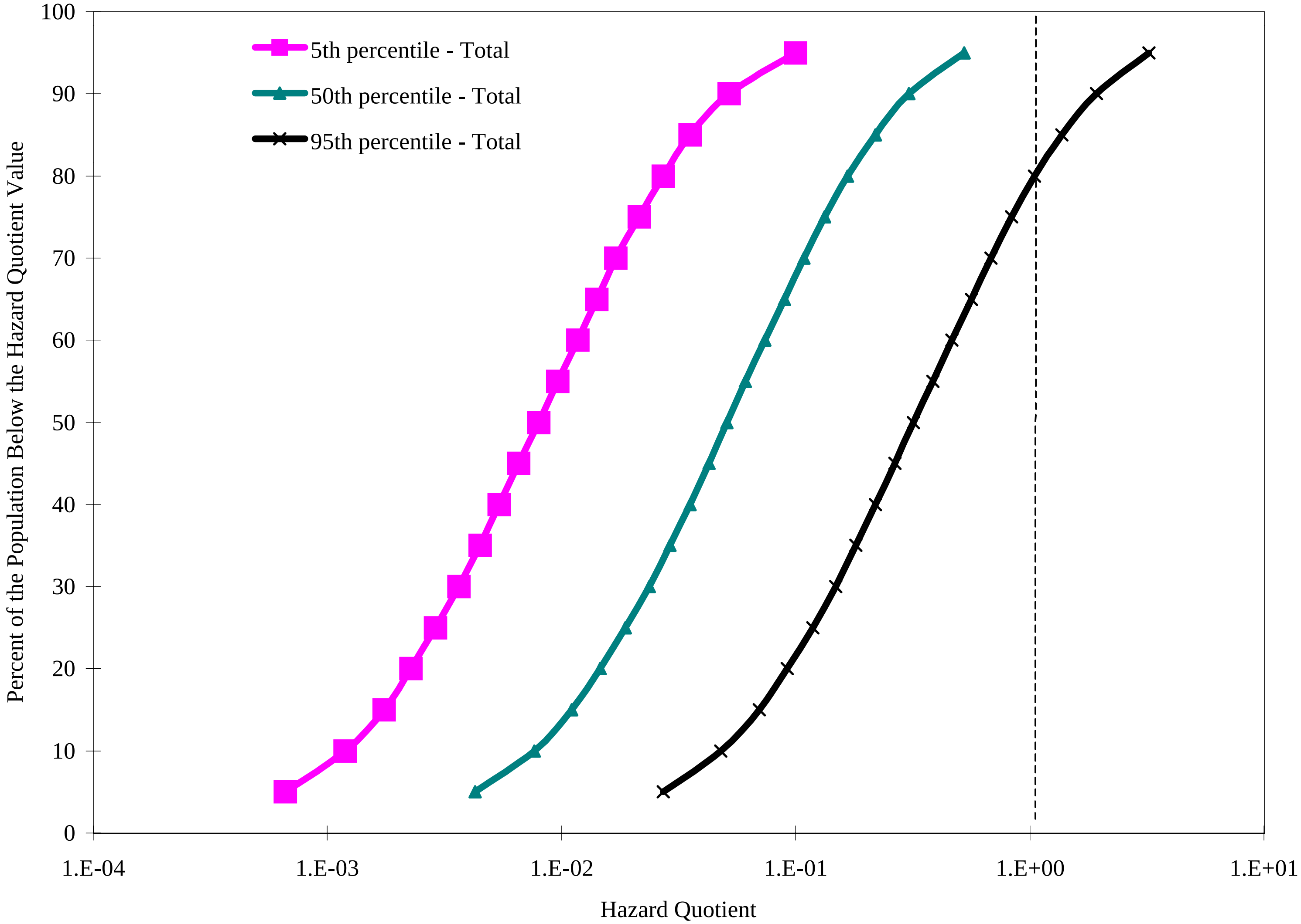


Figure 8-9. Clinch River Adult Recreational Angler - Refined Empirical RfD Distribution



8.2.1 Watts Bar

Results of the two-dimensional Monte Carlo analysis of uncertainty and variability in noncancer hazards to adult recreational anglers at Watts Bar from background and for the sum of background and the ORR releases are shown in Figures 8-4 through 8-6; the numerical estimates for selected percentiles are given in Table 8-3. These results demonstrate that the true hazard quotient is both highly variable and very uncertain. As shown in the Table 8-3, the median estimate of true hazard quotient for the 95th percentile adult consumer was estimated at 2, using the reference distribution, 20, using the empirical distribution, and 0.8, using the refined empirical distribution. The upper confidence limit (95%) on the true hazard quotient for the 95th percentile adult consumer ranged from 7 for the refined empirical distribution to 200 for the empirical distribution. However, the estimates of true hazard quotients were consistently lower than the estimates in the level II evaluation. As Table 8-3 also indicates, median estimates of the true hazard quotient of the Watts Bar child angler showed patterns similar to those of adults.

All of the distributions of the threshold indicated that some fraction of the population were receiving doses above the population threshold (true hazard quotient > 1.0). In addition, it appears that the individuals with the highest rates of fish intake may have had doses that exceeded the threshold by several fold.

As discussed in Section 7.2 and Appendix A, the fraction of PCBs in Watts Bar fish that occurred as a result of the releases from the ORR ranges between 6 and 13 percent. Assuming a value of 13 percent, it is possible to determine the incremental change in true hazard quotients that resulted from the ORR releases. Figure 8-10 presents the distributions of true hazard quotients for the background (non-ORR sources) as well as background plus the ORR sources. The figures demonstrate that the contribution of the ORR was sufficient to increase the percent of individuals with true hazard quotients in excess of 1.0 by one to two percent. In other words, the ORR's releases caused an additional one to two percent of the exposed population to receive doses in excess of the population threshold. Similar incremental findings were found for children. The incremental risks were also found to be independent of the choice of distributions for safety factors.

Finally, the analysis determined the values of the true hazard quotients that would have occurred in the absence of other sources of PCBs. These values suggest that in the absence of other sources, the ORR releases would have had little potential for adverse effects.

8.2.2 Clinch River/Poplar Creek

Results of the two-dimensional Monte Carlo analysis of uncertainty and variability in noncancer hazards to recreational anglers using Clinch River/Poplar Creek are shown in Figures 8-7 through 8-9; the numerical estimates for selected percentiles are summarized in Table 8-4. As shown in the table, the median estimate of hazard quotient for the 95th percentile adult consumer was 1 using the reference distribution, 7 using the empirical distribution, and 0.3 using the refined empirical distribution. The upper confidence limit (95%) on the true hazard quotient for this percentile ranged from 3 for the refined empirical distribution to 70 for the empirical

Figure 8-10. Watts Bar Adult Recreational Angler - Refined Empirical RfD Distribution

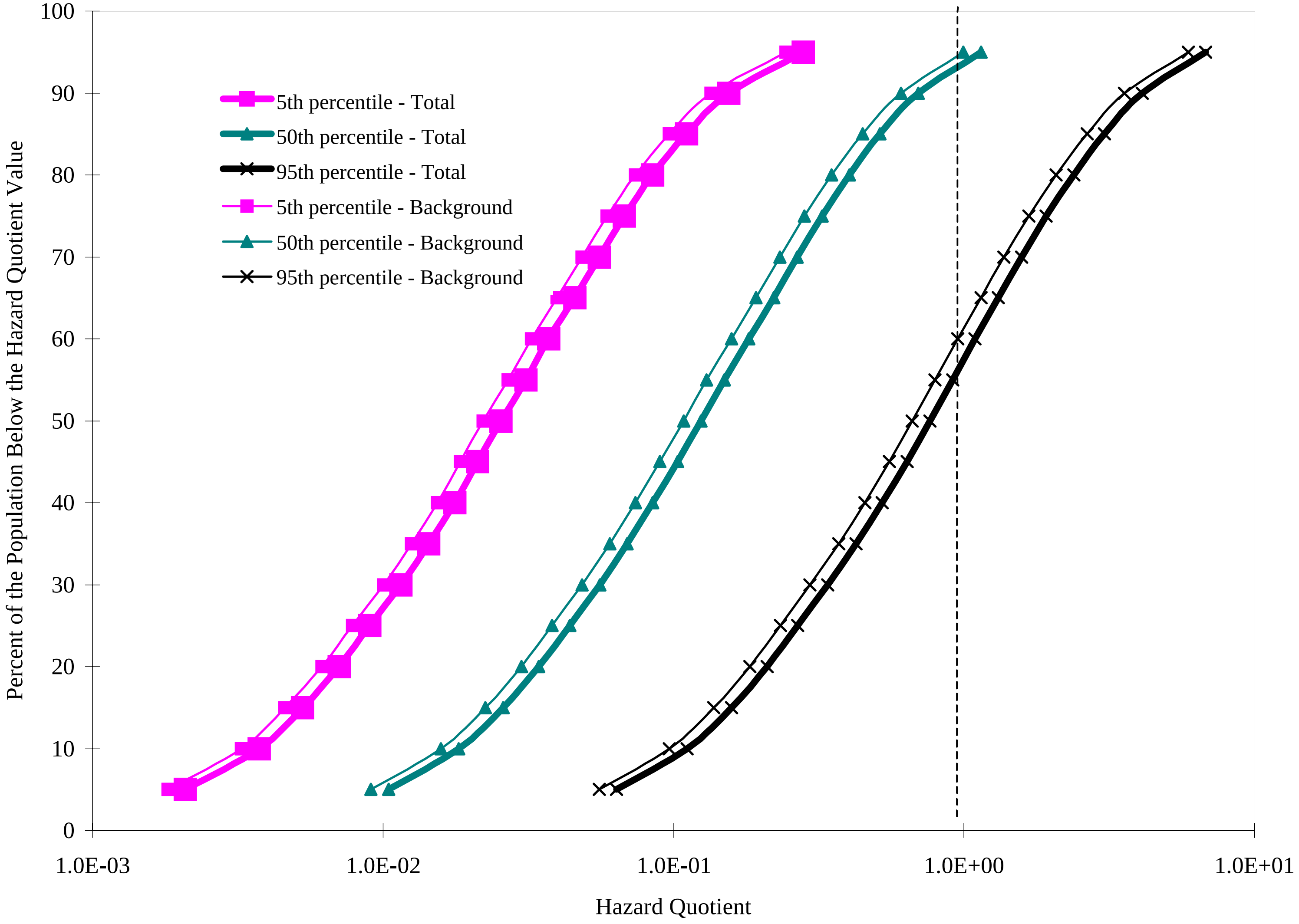


Table 8-3. Hazard Quotients Associated with Recreational Anglers from Watts Bar

Adult

Total Exposures									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.02	0.06	0.2	0.003	0.1	2	0.002	0.01	0.06
50th	0.3	0.7	2	0.04	1	20	0.03	0.1	0.8
95th	3	6	20	0.4	10	200	0.3	1	7

Releases from ORR[1]									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.0026	0.0078	0.026	0.00039	0.013	0.26	0.00026	0.0013	0.0078
50th	0.039	0.091	0.26	0.0052	0.13	2.6	0.0039	0.013	0.104
95th	0.39	0.78	2.6	0.052	1.3	26	0.039	0.13	0.91

Child

Total Exposures									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.03	0.08	0.3	0.005	0.2	3	0.003	0.01	0.08
50th	0.3	0.9	3	0.05	2	30	0.03	0.2	0.9
95th	3	8	30	0.5	20	200	0.4	1	8

Releases from ORR[1]									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.0039	0.0104	0.039	0.00065	0.026	0.39	0.00039	0.0013	0.0104
50th	0.039	0.117	0.39	0.0065	0.26	3.9	0.0039	0.026	0.117
95th	0.39	1.04	3.9	0.065	2.6	26	0.052	0.13	1.04

1. ORR releases are assumed to represent 13 percent of the hazard quotient estimated for Watts Bar.

Table 8-4. Hazard Quotients Associated with Recreational Anglers from the Clinch River/Poplar Creek

Adult

Total Exposures									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.01	0.03	0.1	0.001	0.04	0.6	0.0007	0.004	0.03
50th	0.1	0.3	1	0.02	0.5	7	0.008	0.05	0.3
95th	1	4	10	0.2	5	70	0.1	0.5	3

Releases from ORR[1]									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.005	0.015	0.05	0.0005	0.02	0.3	0.00035	0.002	0.015
50th	0.05	0.15	0.5	0.01	0.25	3.5	0.004	0.025	0.15
95th	0.5	2	5	0.1	2.5	35	0.05	0.25	1.5

Child

Total Exposures									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.01	0.1	1	0.004	0.06	1	0.001	0.0066	0.039
50th	0.03	0.4	4	0.04	0.7	10	0.013	0.08	0.45
95th	0.1	2	10	0.4	7	100	0.15	0.8	4.4

Releases from ORR[1]									
	Uncertainty Percentiles								
Variability	Reference Distribution			Empirical Distribution			Refined Empirical Distribution		
Percentiles	5th	50th	95th	5th	50th	95th	5th	50th	95th
5th	0.005	0.05	0.5	0.002	0.03	0.5	0.0005	0.0033	0.0195
50th	0.015	0.2	2	0.02	0.35	5	0.0065	0.04	0.225
95th	0.05	1	5	0.2	3.5	50	0.075	0.4	2.2

1. ORR releases are assumed to represent 50 percent of the hazard quotient estimated for the Clinch River.

distribution. Similarly, for the Clinch River/Poplar Creek child angler, median estimates of the true hazard quotient to the 95th percentile consumer ranged from 1 to 10, and upper confidence limit estimates of the true hazard quotient for this consumer ranged from 7 to 100. All of the threshold distributions used indicated that some fraction of the population was receiving doses above the threshold and that the individuals with the highest levels of exposure may have had doses that exceeded the threshold by several fold.

As discussed in Section 7.2 and Appendix A, the fraction of PCBs in Clinch River fish that occurs as a result of the releases from the ORR is approximately 50 percent. Table 8-5 and Figure 8-11 present the distributions of true hazard quotients for the background (non-ORR sources) as well as the total of background and the ORR sources. Figure 8-11 demonstrates that the contribution of the ORR was sufficient to increase the percent of individuals with true hazard quotients in excess of 1.0 by about one to two percent. The analysis also determined the values of the true hazard quotients that would have occurred in the absence of other sources of PCBs. These values suggest that the ORR releases would have resulted in some fraction of the anglers having true hazard quotients greater than 1.0.

8.3 CONCLUSIONS

The results of the level II evaluation presented in Section 7.0 indicated a high probability that anglers consuming fish from Watts Bar Reservoir and Clinch River/Poplar Creek were exposed to doses of PCBs in excess of the RfD. However, interpretation of these results is difficult, because it is not clear whether the risk estimates demonstrate that the anglers were at risk or merely reflect the bias in the doses and in the estimates of toxicity.

In the level III evaluation (two-dimensional analysis of uncertainty and variation in the noncancer risk estimates), the project team sought to further characterize those risks by: 1) determining the distribution of true hazard quotients across the population and the uncertainty in those estimates; and, 2) calculating the fraction of the population receiving doses above the population threshold of PCB effects. The team also sought to characterize the uncertainty associated with the estimate of the population threshold that resulted from alternative approaches to estimating the distributions of the safety factors used to derive the RfD.

The results of this assessment suggested that some portion of the angling populations on Watts Bar Reservoir and Clinch River/Poplar Creek were at risk for noncarcinogenic effects of PCBs. It was shown that some fraction of the anglers fishing both of these water bodies had a high probability of receiving doses of PCBs that exceeded the threshold for adverse effects. However, the estimate of the size of the fraction was very uncertain. For example, the fraction of the Clinch River/Poplar Creek angler population that exceeded the population threshold could have been less than 1 percent or greater than 50 percent, depending on the RfD distribution used. The analysis also suggested that children were, at most, at only slightly higher risk than adults. These findings have implications for other populations. The level III evaluation found that many individuals with nominal hazard quotients that were slightly above 1.0 had true hazard quotients below 1.0. This finding suggests that had a level III evaluation been performed on EFPC anglers, their risks would have been small. In contrast, the large nominal hazard quotient found in the level II evaluation for the farm families almost certainly would have remained a concern.

Figure 8-11. Clinch River Adult Recreational Angler - Refined Empirical RfD Distribution

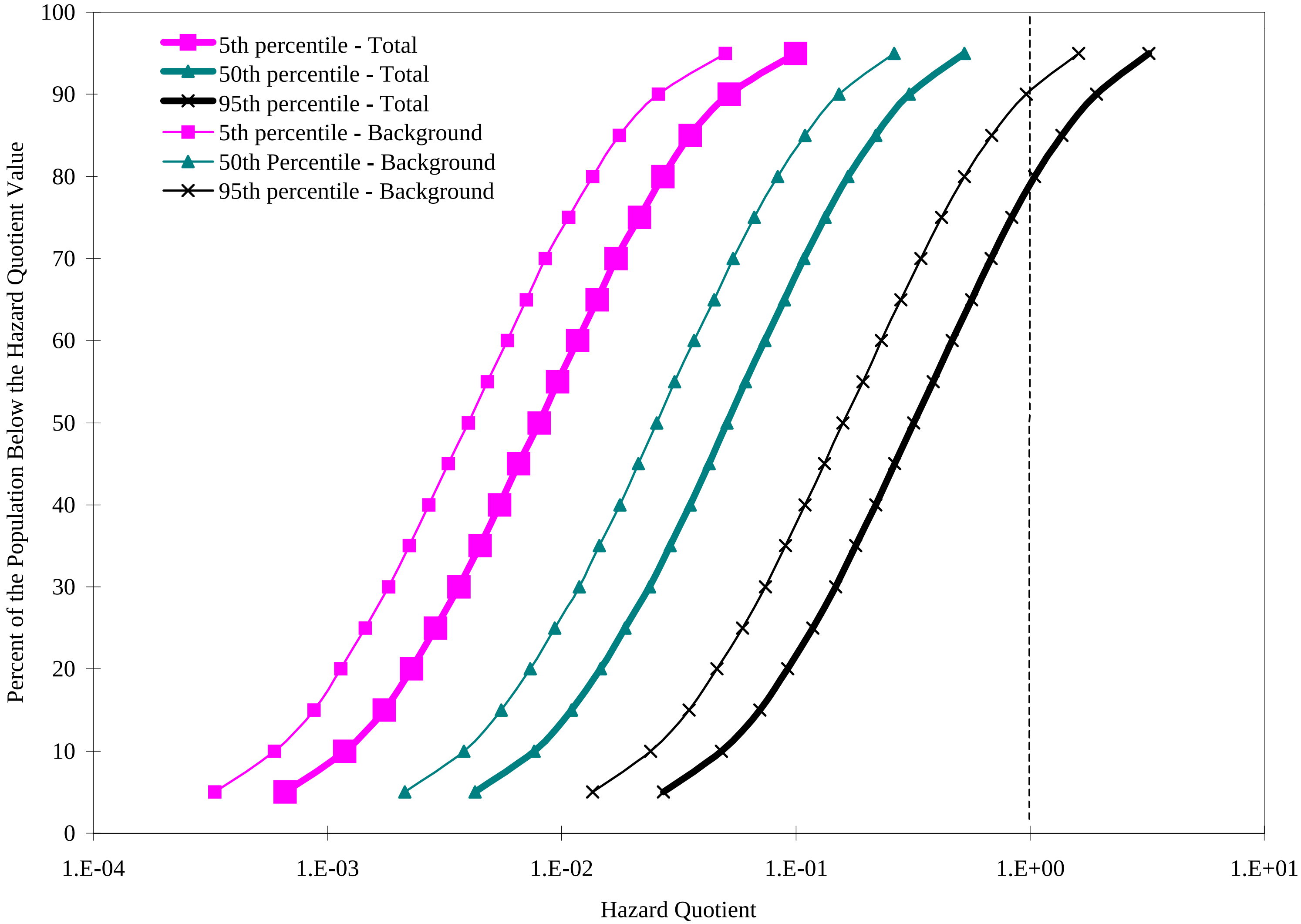


Table 8-5. Percent of Population for Which Hazard Quotient Exceeds 1

Watts Bar Angler (Adult)			
	Background[1]	Background + ORR	Change Due to ORR
Reference Distribution	37 (14-68)[2]	40 (17-71)	3.5
Empirical Distribution	51 (0.68-97)	55 (0.86-97)	3.7
Refined Empirical Distribution	5.0 (0.61-39)	6.6 (0.82-43)	1.6
Watts Bar Angler (Child)			
	Background	Background + ORR	Change Due to ORR
Reference Distribution	44 (19-76)	48 (22-78)	3.7
Empirical Distribution	61 (1.4-98)	64 (2.0-98)	3.2
Refined Empirical Distribution	7.5 (0.92-44)	8.9 (1.4-48)	1.4
Clinch River/Poplar Creek Angler (Adult)			
	Background	Background + ORR	Change Due to ORR
Reference Distribution	11 (3.3-37)	23 (7.5-56)	12
Empirical Distribution	15 (0-80)	30 (0-91)	15
Refined Empirical Distribution	0.55 (0-9.7)	2.2 (0-21)	1.7
Clinch River/Poplar Creek Angler (Child)			
	Background	Background + ORR	Change Due to ORR
Reference Distribution	12 (4.0-42)	25 (8.6-61)	13
Empirical Distribution	24 (0.35-90)	41 (2.2-95)	18
Refined Empirical Distribution	0.97 (0-15)	3.8 (0-29)	2.8

[1] Background reflects risks associated with PCB releases attributable to sources other than ORR. Based on the HydroQual analysis, 87% and 50% of the total PCB concentration in Watts Bar and the Clinch River/Poplar Creek, respectively, is assumed to be associated with other sources.

[2] Median value (90% confidence limits); values reported to 2 significant figures

Finally, the analysis suggests that the releases from the ORR would have resulted in an increase of one to two percent in the number of individuals receiving doses in excess of the population threshold in anglers on Clinch River/Poplar Creek.

9.0 FINDINGS AND CONCLUSIONS

In Task 3 of the Dose Reconstruction Study, the project team performed an assessment of the risks posed by PCB releases from the ORR to individuals living near the reservation and affected water bodies. The goal of this task was to define, to the extent possible, the extent of releases of PCBs from the ORR, the potential for exposures to the general population living near the reservation as a result of those releases, and the potential for adverse effects in those exposed populations.

To complete this evaluation, the project began with an aggressive search for information on the uses and releases of PCBs (as discussed in Section 3.0 of this report). Once those releases were characterized, the next step was to determine all potential pathways of PCB exposure to off-site populations (Section 6.0). After identification, these pathways were grouped into three categories: pathways associated with contamination of surface water bodies, pathways associated with air releases, and pathways associated with exposures to PCBs in waste oils. In addition, the risks associated with the exposure to dioxin and furans were evaluated due to the identified potential for the formation of dioxins and furans during the combustion of PCBs in various areas of the ORR.

In a preliminary evaluation of potential risks, the project team performed a screening assessment of the identified pathways to determine those pathways that presented potential risks to off-site populations. Those pathways that did not exceed the decision guide of 1×10^{-4} or the nominal hazard quotient of 1.0 were set aside from the analysis and investigated no further. Based on this conservative screening level analysis, many of the pathways associated with releases to surface water, all but one of the pathways associated with releases to air, and all pathways associated with waste oils were set aside.

In setting aside these pathways, no decision was made on the acceptability of the risks associated with those pathways. Instead, the project team limited its efforts to further refine exposures and risks to those pathways that exceeded the decision criteria in order to focus project resources on the sources of exposure that produced the highest potential for risk of harm.

Once the level I evaluation was completed, the project team identified the types, locations, and sizes of populations that might have been associated with each of the remaining pathways. Four populations were identified: recreational anglers fishing from EFPC, Poplar Creek, Clinch River, and/or Watts Bar; commercial anglers fishing Watts Bar and/or the Clinch River; recreational users of EFPC; and farm families residing along EFPC. The largest of these populations was estimated to be recreational anglers who fished Watts Bar.

The uncertainty in the estimates of cancer risk and nominal hazard quotient was determined through the level II evaluation (Monte Carlo analysis) for each of the populations (Section 7.0). The goal of this analysis was to determine whether those populations had a high likelihood of receiving risks in excess of the decision guide. Where a population was exposed by multiple pathways, the level II evaluation considered the uncertainty in the sum of the doses for all of the relevant pathways. Any scenario for which five percent or more of the population was found to exceed a 1×10^{-4} cancer risk or nominal hazard quotient of 1.0 was regarded as needing additional assessment. Those scenarios for which less than a five percent of the estimates exceeded the decision guide were set aside.

While recreational anglers at EFPC, commercial anglers at all water bodies, and farm families along EFPC did not meet either one or both of the aforementioned criteria, these populations were not evaluated further for other reasons. EFPC recreational anglers were set aside due to the limited productivity of that water body and the associated assumption that it could only support a small number of recreational anglers. Commercial anglers were not evaluated further because the population size was small and it was believed that exposures to recreational anglers were comparable to those experienced by commercial anglers. Finally, farm families along EFPC were not evaluated further due to the small number of potentially affected individuals and the high level of uncertainty associated with historical PCB concentrations in farm soils.

As a final step in the Dose Reconstruction, the uncertainty and variability of the doses received by recreational anglers were modeled using a two-dimensional Monte Carlo model, taking into account the uncertainty in the estimate of the threshold of PCB toxicity (level III evaluation). The result was a model of the probability of exceeding the actual threshold of adverse effects in the exposed populations (Section 8.0).

The results of the level III evaluation suggested that there was a reasonable chance, but not a certainty, that anglers with high rates of fish consumption from Watts Bar or Poplar Creek/Clinch River exceeded the population threshold for PCBs' noncarcinogenic adverse effects. However, neither the fraction of the population at risk nor the actual number of individuals who may have experienced adverse effects could be precisely determined. The uncertainty in the risk estimates would be improved if better information on fish consumption rates and body burdens of PCBs in these anglers were available.

The majority of the risks to Watts Bar anglers appear to be due to PCBs coming from upstream sources other than the ORR. We determined the incremental risks posed by the ORR releases to anglers already exposed to other sources. The ORR releases resulted in an additional one to two percent of anglers receiving doses in excess of the population threshold. If there had been no releases from other Tennessee River sources, the ORR releases would not have resulted in doses that exceeded the population threshold level for Watts Bar anglers. The same was not true, however, for Poplar Creek/Clinch River, where it appears that the ORR discharges were likely to have resulted in some anglers receiving doses in excess of the population threshold for adverse effects.

A conservative estimate of the carcinogenic risks posed by the ORR releases to anglers on Watts Bar and the Clinch River range from less than 1 in a 1,000,000 to 2 in 10,000. Using these risk estimates and the number of anglers in this population of anglers using the fisheries, we estimate that approximately three excess cases of cancer would occur. Because the estimates are conservative, the actual risks and the number of cases are likely to be smaller.

In performing the assessment for anglers using Poplar Creek/Clinch River or Watts Bar, the project team used fish tissue PCB data collected in the 1980s. As a result, the level I evaluation may not have been representative of exposures to anglers prior to the early 1980s. As described in Appendix A, there is limited evidence to suggest that sediment levels of PCBs in the Clinch River were relatively constant from the late 1950s to the present, with the exception of three periods of time when levels were elevated by a factor of two to three. The periods were two to three years in duration and occurred around 1964, 1972, and 1976. These short-term increases in PCB levels in sediments may have resulted in elevated PCB levels in fish. Because cancer risks are believed to be a function of lifetime average exposures, short-term increases would not have a major impact on the estimated lifetime average exposures. Thus, it is reasonable to assume that the estimates of cancer risks produced in this analysis were also applicable to anglers who fished those water bodies in the 1950s, 60s, and 70s. However, short-term elevations in fish concentrations would have impacted noncancer doses. The use of monitoring data taken in the 1980s may have resulted in an underestimation of the nominal hazard quotients for Clinch River anglers who consumed fish during periods when there were elevated fish concentrations. As a result, the absence of data on earlier periods of exposure should be viewed as a significant data gap.

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10.0 RECOMMENDATIONS

This assessment was successful as in identifying key populations at risk from PCBs released from the ORR. During this assessment, however, the project team identified a number of data gaps and informational needs that, if filled, would improve the certainty in risk estimates. This section presents a series of recommendations for additional studies and analyses that could be used to fill those data gaps and provide improved estimates of risk to the exposed populations. The specific recommendations are as follows.

10.1 RECOMMENDATIONS FOR THE COLLECTION OF ADDITIONAL DATA

Characterize Fish Consumption Rates for Poplar Creek/Clinch River and Watts Bar Reservoir

Studies of anglers have shown that recreational angling has continued to increase in popularity as a sport between the 1940s and the present. As a result, the project team believes that site-specific surveys of current angler behavior at Poplar Creek/Clinch River and Watts Bar could provide a reasonable and perhaps conservative model of the historical fish consumption rates of anglers who used these bodies of water. Such a survey could provide additional information on the number and types of anglers using each body of water, the species consumed, rates of consumption on a "per individual" basis, and demographic characteristics of the targeted population. Therefore, it is recommended that surveys of fish consumption rates and other relevant demographic data be performed for anglers on these water bodies.

Collect Core Samples from Poplar Creek/Clinch River and Watts Bar Reservoir

Because of the persistence of PCBs in sediment, cores could provide information on the history of PCB concentrations in sediments of Poplar Creek/Clinch River and Watts Bar. The collection of sediment cores would allow the determination of whether sediment and fish concentrations were higher in the 1950s, 1960s and 1970s than they were in the 1980s, when the currently available monitoring data were collected. In addition, collection of cores above Melton Hill Dam may also provide a better understanding of the fraction of PCBs in the Clinch River that were attributable to activities at the ORR.

Perform Additional Sampling of Soils near EFPC

The level I and II evaluations conducted for the farm families in this analysis identified a large potential for exposures to PCBs. However, because these estimates were based upon limited sampling data taken from the areas directly adjacent to EFPC, it is not clear how these measurements related to actual soil concentrations in areas where cattle were grazed and vegetables grown on these farms. Because of the persistence of PCBs, measurements of current levels would allow the delineation of the limits of historical PCB contamination on these farms, and provide insight into the average level of PCBs in the areas historically used as pasture. This site-specific data could then be used to refine estimates of risks to this population.

Measure PCB levels in cattle currently grazing near EFPC

The PCB exposures to farm families via the consumption of beef and dairy products were estimated based on grazing and biotransfer models. The uncertainties associated with this approach could be eliminated by taking direct measurements of PCBs in the fat of cattle grazing on affected pastures. Such data, coupled with the site-specific soil concentrations, would allow derivation of a site-specific bioconcentration factor to be used to refine risk estimates for these exposure pathways.

10.2 RECOMMENDATIONS FOR ADDITIONAL ANALYSES

Revise Risk Estimates to Reflect Additional Survey Data

The evaluations performed in this assessment will need to be revised and extended as additional measurements of PCB levels in fish, soil and sediment are collected. These additional analyses could range from simply updating the existing risk estimates to performing more refined assessments such as MicroExposure Event Modeling (Price et al., 1996). Microexposure event modeling is an approach that models exposure to PCBs for each fish meal that an angler consumes. The doses for each meal are summed to determine the daily, monthly, and annual doses that occur over an angler's life. The approach, while more complex, is very useful for determining peak dose rates (or body burdens) that could occur over an individual's life or the peak exposures that could occur to subpopulations such as children or pregnant women.

Model Body Burdens of PCBs

The assessment of exposures performed in this dose reconstruction have focused on characterizing administered doses. Toxicologists have often found that measurements of body burdens are more accurate measures of the potential for adverse effects than measurements of administered doses. Pharmacokinetic models have been developed to evaluate the impact of exposure and uptake of PCBs on individual's blood levels. These models allow the incorporation of information on background sources of PCBs as well as local exposures and could be used to determine of the impact of PCBs from the consumption of fish or beef and dairy products on the total body burdens of exposed individuals. In addition, such models could be validated using data from the recommended study of blood levels in recreational anglers or from the recent ATSDR sampling.

Estimate Response Rates for Noncarcinogenic Effects

Over the last year, several techniques for estimating the rates of response to noncarcinogenic compounds have been proposed (Baird et al., 1997; Price et al., 1997). These techniques may be useful for providing upper-bound estimates of the number of cases of adverse effects that could be expected to occur within the exposed populations. These estimates would be useful in determining whether or not future epidemiologic studies would be able to detect observable increases in adverse effects.

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APPENDIX A

SUMMARY OF HYDROQUAL INVESTIGATIONS

APPENDIX A-1

CHEMRISK MEMORANDUM REGARDING FEBRUARY 29, 1996 MEETING

MEMORANDUM

To: **Tom Widner**

From: **Paul Price**
Nancy Bonnevie

CC: **Tom Mongan**
Owen Hoffman
Jack Buddenbaum

Date: **April 12, 1996**

Subject: **Trip Report: Thursday, February 29 meeting at HydroQual Headquarters in Mahwah, NJ**

Nancy Bonnevie and Paul Price of the Portland office met with Mr. Kurt Ziegler of HydroQual on February 29, 1996 to discuss the modeling work performed by HydroQual on behalf of Union Carbide and Martin Marietta regarding PCBs in sediments. During this meeting, Mr. Ziegler explained the purpose, approach, and results of HydroQual's modeling work on the Clinch and Tennessee Rivers. He also indicated that HydroQual was available for additional modeling work if ChemRisk was interested. The following is a summary of the key points of the meeting.

Background

HydroQual was retained in November, 1993 to provide technical support for a lawsuit that had been brought by marina owners located on lower Watt's Bar against Martin Marietta and Union Carbide in the early 1990s. The marina owners asserted that releases of PCBs from Oak Ridge Reservation (ORR) had resulted in elevated levels of PCBs in Watts Bar fish. According to the suit, the contamination of fish and resulting fish advisories had suppressed fishing activities, adversely affecting the livelihoods of the marina owners. HydroQual's research was based upon existing data from the Clinch River studies and published information on PCB concentration in fish. As a result, only limited amounts of information on PCB levels in sediments were available for the project.

HydroQual's evaluation was completed in May, 1994, and the results were presented to a group of scientists at Oak Ridge for peer review. Overheads for that presentation are available. In addition, a series of depositions reporting the results of their work were prepared. Shortly before the scheduled trial date in August, 1994, the case was settled out of court and the task was closed, therefore, no final report of the analysis was ever prepared. However, a manuscript describing the sediment transport model developed for the Watts Bar reservoir was subsequently published as the attached article entitled, "Long-term Simulation of Fine-grained Sediment Transport in a Large Reservoir" in the November, 1995 edition of Journal of Hydraulic Engineering. This article is attached with the permission of ASCE.

Project Goals

The strategy pursued by Union Carbide and Martin Marietta in this case was to demonstrate that: 1) the contribution of ORR to PCBs measured in Watts Bar fish was small in relation to the contributions from other sources upstream of Watts Bar on the Tennessee and Clinch Rivers; and, 2) that releases from ORR alone would not have resulted in PCB concentrations in fish that would warrant a fish advisory. Therefore, the focus of the HydroQual effort was to develop technical evidence on the relative contribution of releases from Oak Ridge versus other sources of PCBs entering Watts Bar.

Approach Used by HydroQual

Two independent approaches were utilized by HydroQual to evaluate the relative fraction of PCBs in Watts Bar fish attributable to Oak Ridge. One approach involved a straightforward analysis of spatial trends in fish monitoring data. The other approach entailed the development of a sediment transport model, which was used in conjunction with PCB sediment core data to predict sources and transport of PCBs at various times and locations within the watershed.

<u>Data Analysis</u>
One of the limiting factors for this evaluation was the paucity of information regarding sediment concentrations of PCBs in the Clinch and Tennessee Rivers. Therefore, for the purpose of the data analysis approach, HydroQual obtained fish tissue data collected by TVA throughout the previous 20 years. Fortunately, surveys performed by TVA provided excellent information on the mean concentration of PCBs in fish at different locations along the Tennessee and Clinch Rivers. Lipid-normalized PCB concentrations were summarized and averaged for various discrete areas within the watershed including Watts Bar Reservoir, Melton Hill Reservoir, and Fort Loudoun Reservoir. It was assumed that these measured tissue levels were proportional to average concentrations of PCBs present in surficial and suspended sediments in the various bodies of water that made up the Tennessee River and its tributaries. The mass loading of PCBs from each of these areas was assumed to be proportional to the average fish tissue concentration times the average sediment loading rates of the respective waterbodies.

Using this approach HydroQual was able to construct a strong argument demonstrating that loadings of PCBs to Watts Bar from the Clinch River were lower than those from the Tennessee River.

Evaluation of fish tissue concentrations alone indicated that measured concentrations of PCBs in fish were lower in the Clinch than in Watts Bar or the Tennessee River. In addition, the sediment loading rate of the Tennessee is significantly higher than that of the Clinch. Based on this analysis, HydroQual estimated that approximately 13 percent of the PCBs measured in Watts Bar originated from the Clinch River. The analysis further demonstrated that approximately half of this 13 percent originated from sources upstream of the Melton Hill Dam. This analysis provided powerful evidence that, for the years when the fish monitoring data were available, ORR had played a very minor role as a source of PCBs in fish in the Watts Bar.

Sediment Transport Model

The data analysis described above provided a strong argument that contributions of PCBs to Watts Bar from ORR can account for approximately 13 percent of the PCBs present throughout the last 20 years. However, the issue of historical contributions of PCBs to Watts Bar remained unanswered. Such historical releases might not be reflected in fish concentrations recently collected. In order to deal with this issue, HydroQual constructed a sediment and contaminant transport model for the Clinch and Tennessee Rivers from the Melton Hill Dam to the lower Watts Bar.

Excellent information on changes in sediment depths was available for various locations on the Clinch and Tennessee Rivers from TVA, providing a history of sediment movement in these bodies of water over the last 40 years. In addition, HydroQual was able to obtain information regarding suspended sediment levels and flow rates for several locations throughout the watershed. Based upon this information, a model of sediment movement from the Clinch River to the Watts Bar was constructed. This work was the basis for the publication in the Journal of Hydraulic Engineering. Because of the large amounts of data available on sediments, flow rates, and levels of suspended sediments, the results of this sediment transport model are expected to be quite robust.

PCBs are highly hydrophobic chemicals, therefore, their transport in rivers is largely a function of sediment transport. As constructed, HydroQual's sediment transport model only required information on the concentration of PCBs in the suspended and surficial sediments at various times in the past in order to make predictions of the historical loadings from the Clinch River to the Watts Bar and the concentration of PCBs in Watts Bar sediment that resulted from ORR releases. These predicted sediment concentrations could be used with a BSAF factor to predict the associated concentration in fish. Information regarding releases of PCBs from ORR were not available, therefore, it was necessary to derive loading estimates from a limited number of sediment cores collected by researchers at ORNL as part of the Clinch River survey. Of the five cores in which PCB data were available, three did not contain detectable levels of PCBs. The remaining two showed levels of PCBs that fluctuated by a factor of three to five at different depths. The core data also included measurements of cesium and mercury. The historical levels of cesium and mercury released from Oak Ridge are relatively well known and assisted in the dating of the different depths of the core. In addition, information on the dating of the core sediments was derived from sedimentation rates reported for the locations where the cores were collected.

HydroQual calibrated its model of PCB transport using the core furthest upstream (Clinch River mile 9.5). The model was then verified using the core taken at Clinch River mile 0.5. Good agreement was found between predicted concentrations at Clinch River mile five, and the concentrations measured in this sediment core. The model also predicted that the concentrations in the three additional sediment cores taken on the Tennessee River would occur at levels that were below the analytical detection limits of the methods used to analyze for PCBs. As a result, the absence of detectable levels of PCBs in the remaining sediment cores provides an additional confirmation of the model.

The sediment transport model developed by HydroQual predicted that if the Clinch River were the sole source of PCBs entering Watts Bar then the fish concentrations in Watts Bar would be approximately 9 percent of the observed levels. Based on these results, HydroQual concluded that historical releases of PCBs from Oak Ridge were responsible for less than 9 percent of the currently observed levels in Watts Bar fish, an estimate that could be further reduced if sources of PCBs above Melton Hill dam are considered. In addition, because of the approximate agreement between these two independent measurements, HydroQual concluded that there was strong evidence that the vast majority of PCBs currently detected in fish in the lower Watts Bar occurred as a result of releases to the Tennessee River upstream of the Clinch River.

The approaches used to model sediment transport and PCB transport appear to be reasonable. However, the reliance upon measurable data in only two cores suggest that specific predictions about concentrations of any time in the past are highly uncertain.

Implications for Oak Ridge Dose Reconstruction

HydroQual's investigation provides a number of findings that are relevant to the PCB dose reconstruction evaluation. First, the results of the HydroQual investigation imply that the measured PCB concentrations in fish from Watts Bar reflect many sources in addition to ORR. Therefore, in performing a dose reconstruction it is appropriate to reduce the total exposure to PCBs from the consumption of fish taken from Watts Bar by some apportioned factor.

Second, the HydroQual dating of the two cores provides the best available evidence on historical levels of PCBs in sediments and, by extrapolation, in fish. Based on the results of the model, it appears that levels of PCBs in sediments deposited from the early 1950s to the present have been relatively constant with the exception of three episodes, each of one to two years in duration, where concentrations were elevated by a factor of two to five. These findings imply that the levels of PCBs observed in fish collected in monitoring studies over the last 20 years are likely to be similar to concentrations in fish occurring in the early 1950s.
The HydroQual modeling effort also collected data on historical releases of cesium and mercury which may be of use to other tasks in the dose reconstruction effort.

Recommendations

One of the most limiting factors associated with the PCB dose reconstruction task is the lack of historical emissions release rates for PCBs. As a result, the investigation has been forced to rely upon measurements of PCBs in environmental media, limiting our ability to predict doses to those years where fish concentrations were actually measured (i.e., the late 1970s to the present). The HydroQual data provides the only identified source of information on levels of PCBs prior to that time period.

It is important to note that additional sediment core data for PCBs on the Clinch River would provide valuable information. Currently, only two cores with detectable levels of PCBs have been identified. It will be critical to determine if other cores were taken and if the data for the cores confirms the patterns of historical levels in surficial sediments identified in the currently available cores. The team should make every effort to identify the existence of other core data and should flag for the ORHASP committee collection and analysis of core data as a high priority research need.

In the absence of this additional information regarding historical level of PCBs we propose to assume that historical levels were similar to current levels with the exception of three periods where levels might have been higher. This suggests that for the evaluation of acute effects or subchronic effects of PCBs, the current levels in fish should be multiplied by a factor of two or three to account for potentially elevated historical levels. Levels used in carcinogenic risk estimates however should probably be set at the levels

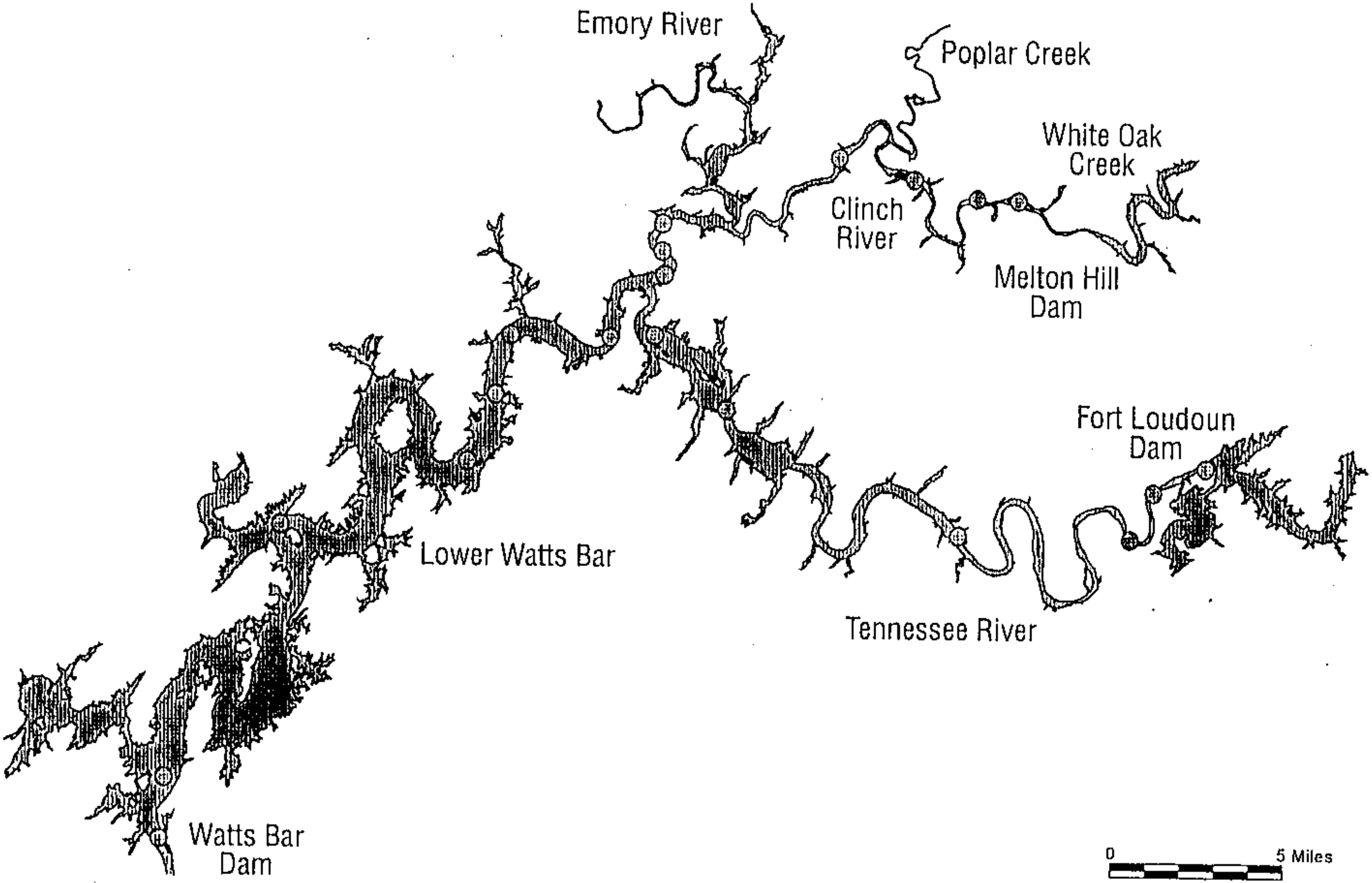
APPENDIX A-2

BACKGROUND MATERIALS
RECEIVED FROM HYDROQUAL

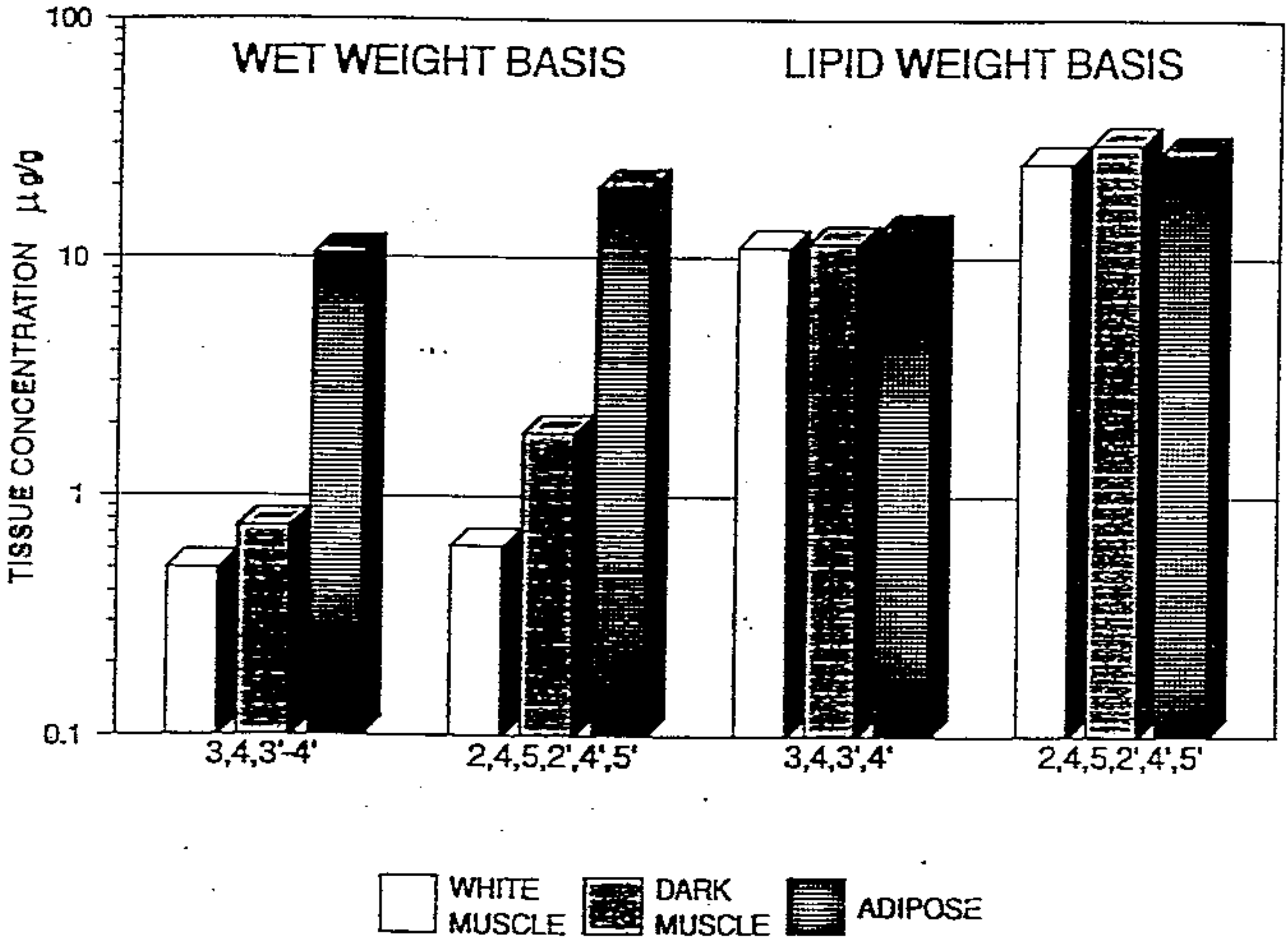
-K. ZIEGLER AND D. DITORO
PRESENTATION TO ORNL, 5/5/94

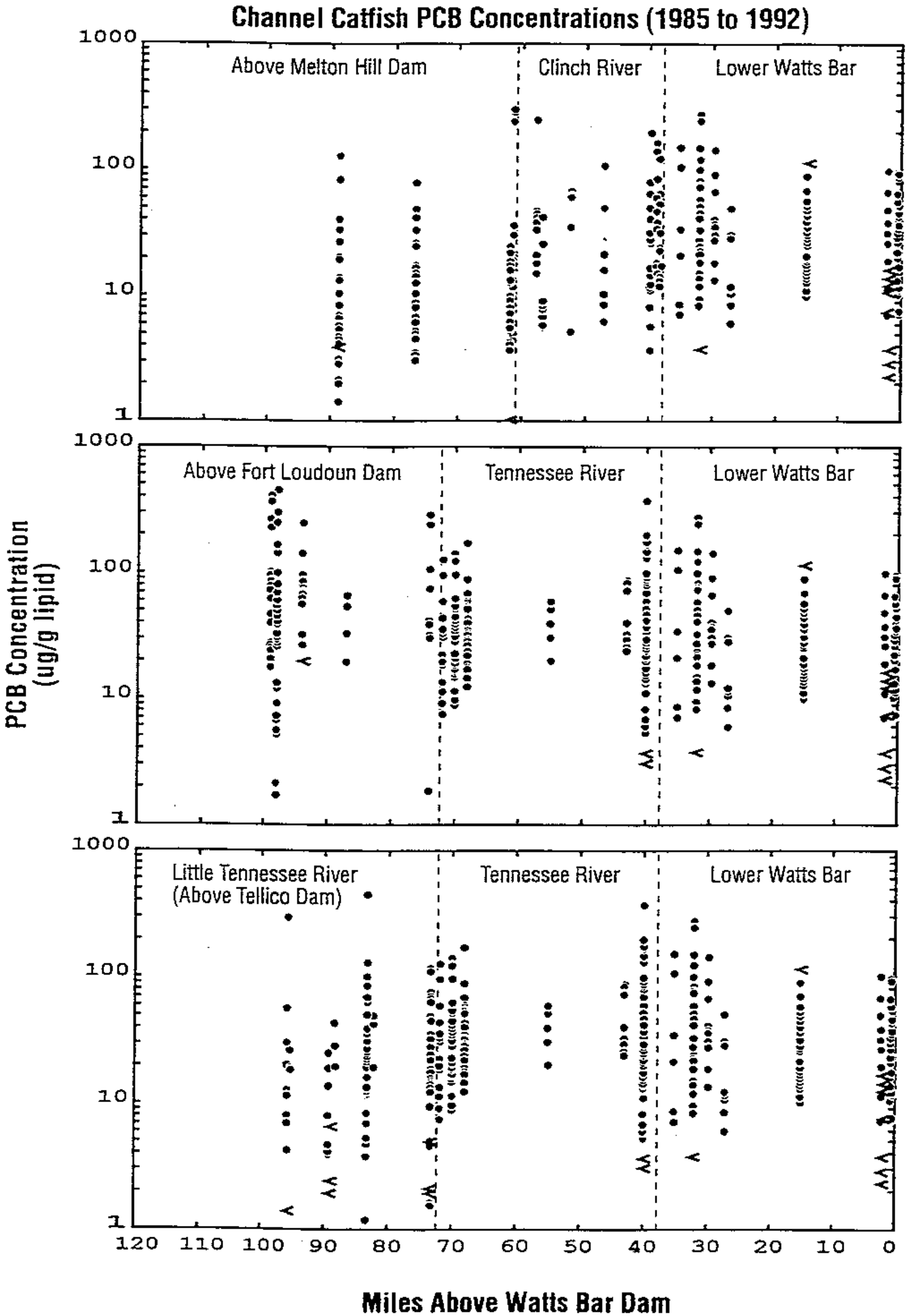
-TESTIMONY EXHIBITS FOR D. DITORO

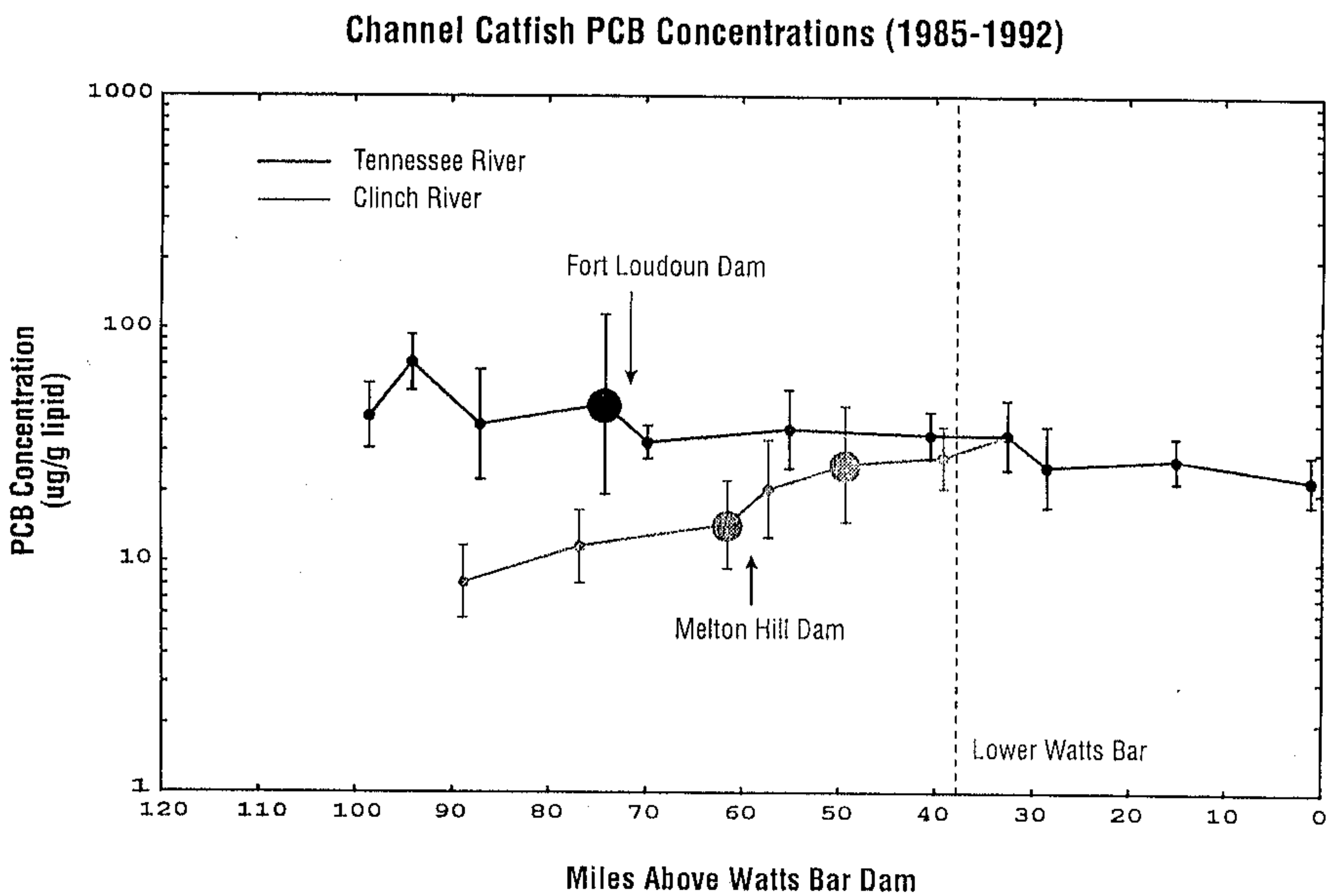
Fish Sampling Areas (1985 To 1992)



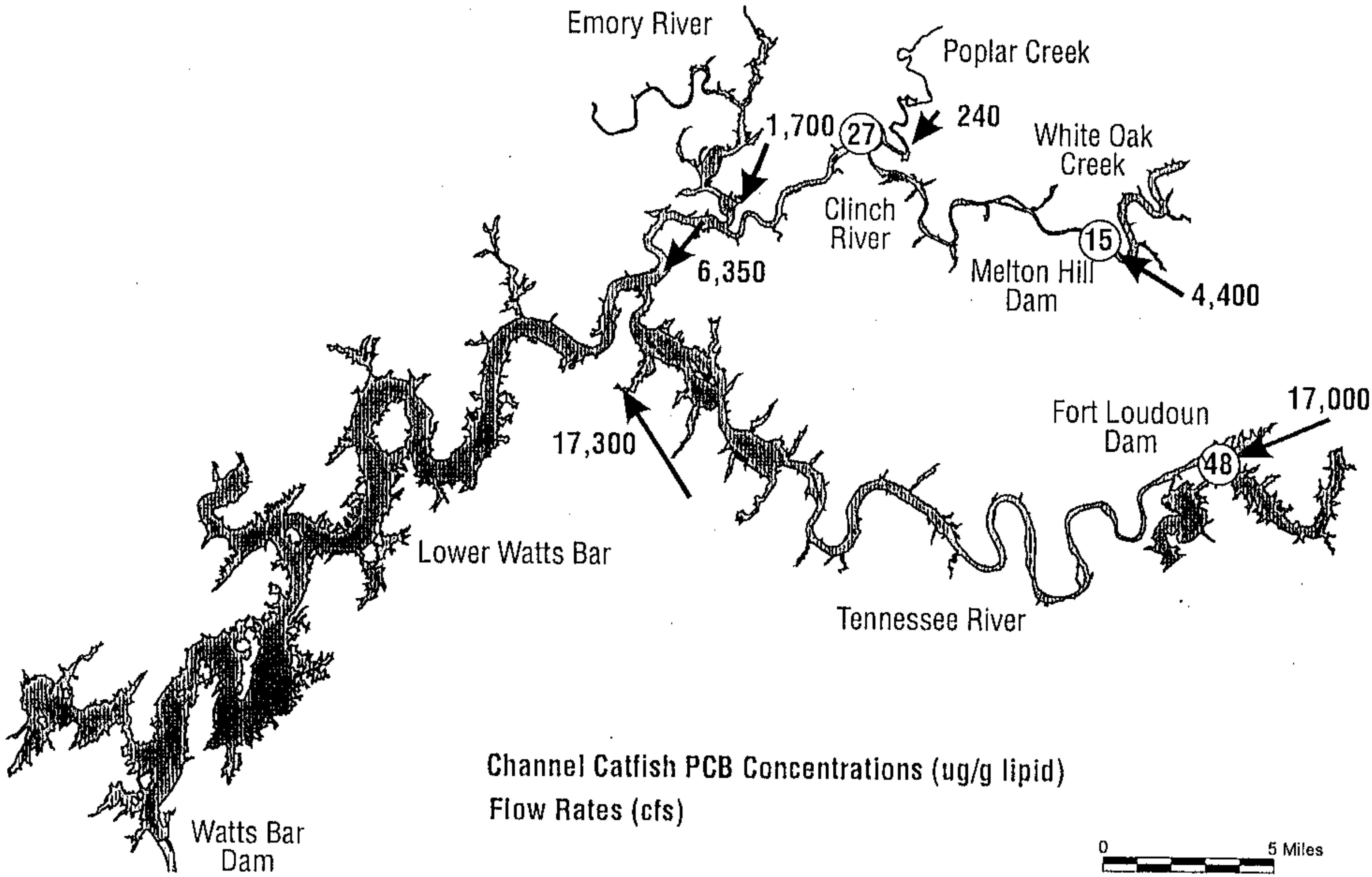
CONCENTRATIONS OF CHLOROBIPHENYLS IN TISSUE OF JUVENILE COHO SALMON FED A CONTAMINATED DIET FOR 117 DAYS (GRUGER, et al,, 1975)



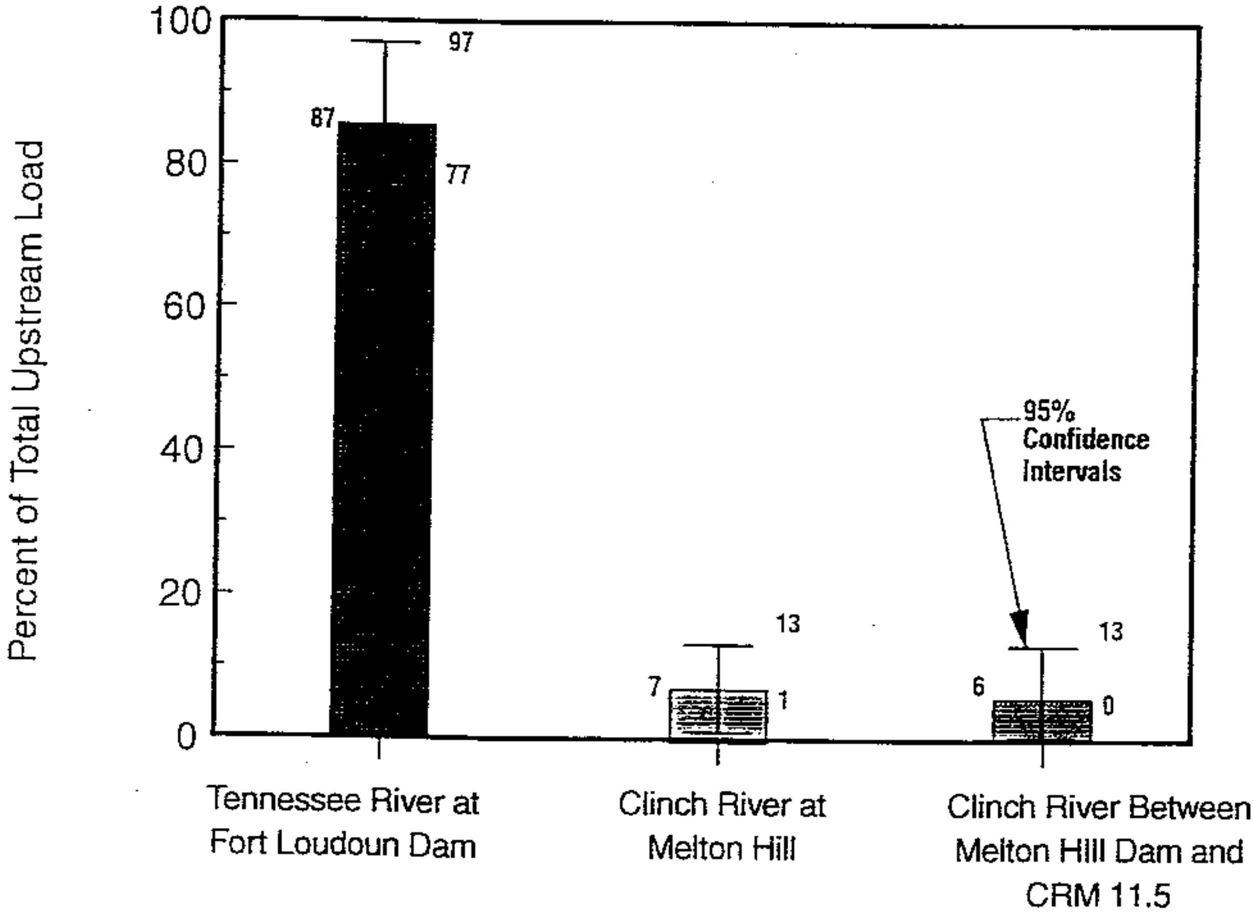

Channel Catfish PCB Concentrations (1985 to 1992)
Above Melton Hill Dam
Clinch River
Lower Watts Bar
Above Fort Loudoun Dam
Tennessee River
Lower Watts Bar
Little Tennessee River
(Above Tellico Dam)
Tennessee River
Lower Watts Bar
PCB Concentration
(ug/g lipid)
1000
100
10
1
120
110
100
90
80
70
60
50
40
30
20
10
0
Miles Above Watts Bar Dam


Channel Catfish PCB Concentrations (1985-1992)
1000
100
10
1
PCB Concentration (ug/g lipid)
Tennessee River
Clinch River
Fort Loudoun Dam
Melton Hill Dam
Lower Watts Bar
120
110
100
90
80
70
60
50
40
30
20
10
0
Miles Above Watts Bar Dam

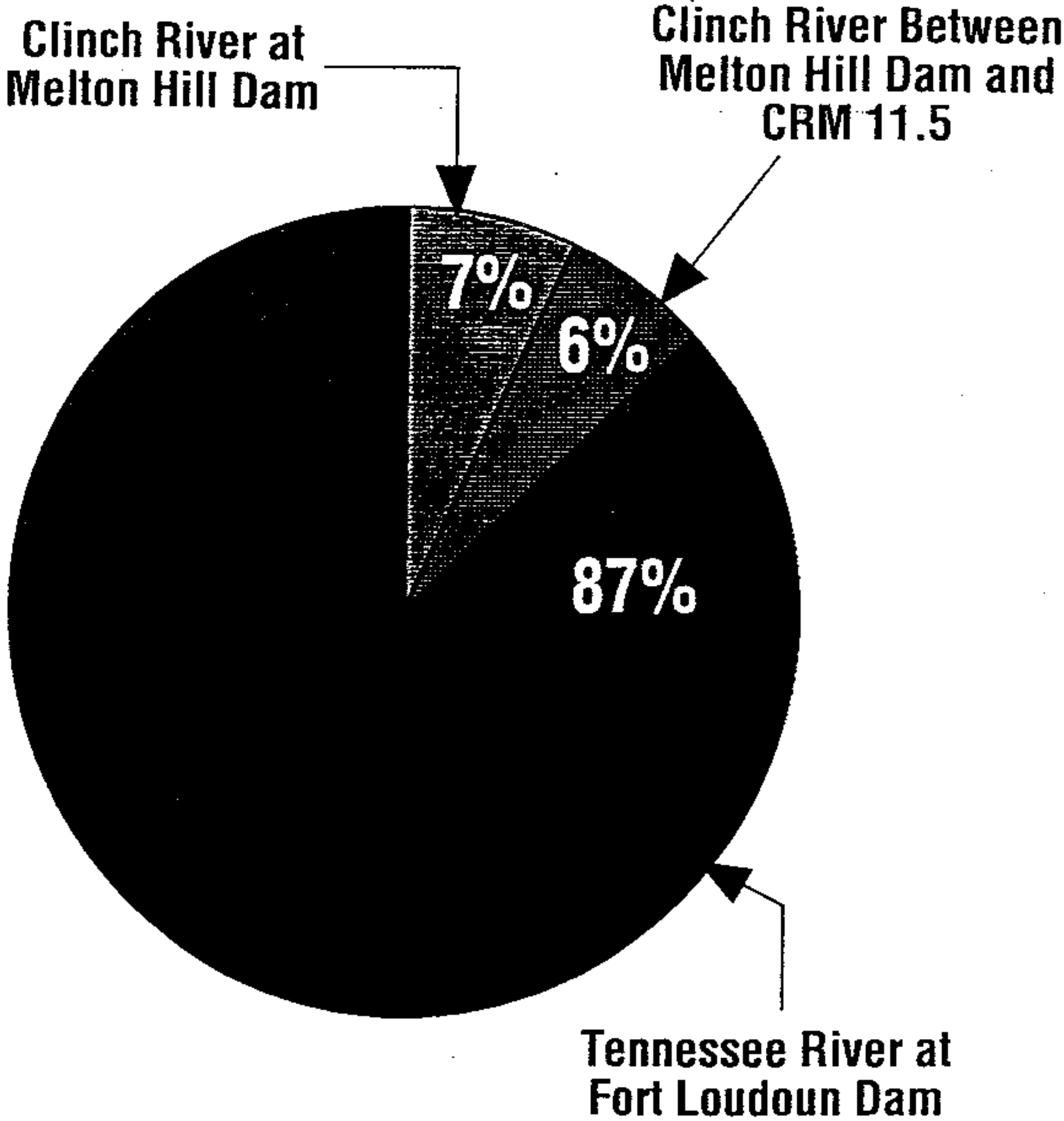
Mean Flow Rates and Catfish PCB Concentrations for 1985 - 1992



Relative Contributions of Upstream Sources To Watts Bar Reservoir PCB Load (1985 to 1992)

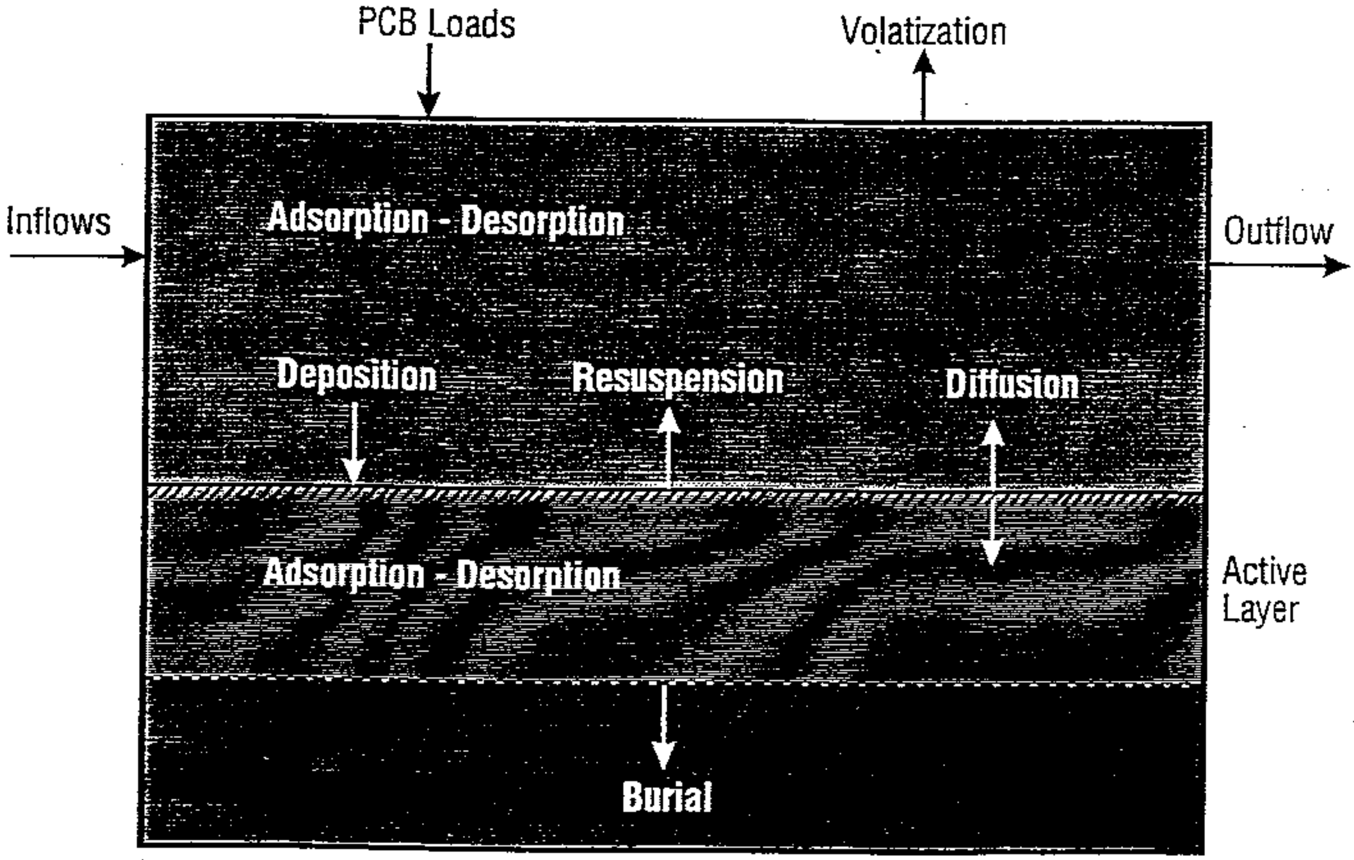


Relative Contributions of Upstream Sources to Watts Bar Reservoir PCB Load (1985 to 1992)



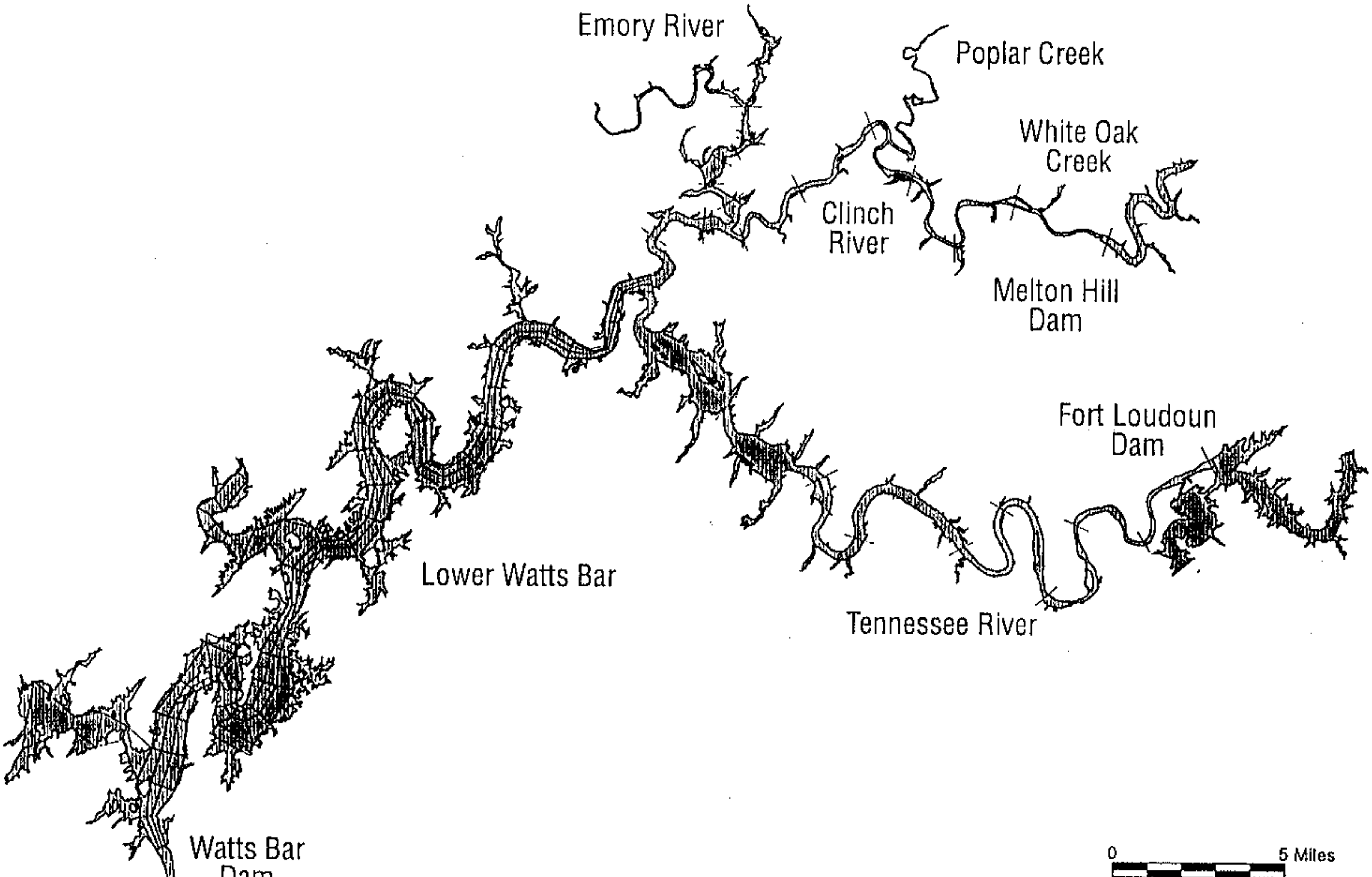
Relative PCB Loads are Based on Measured Flow Rates and Channel Catfish PCB Concentrations

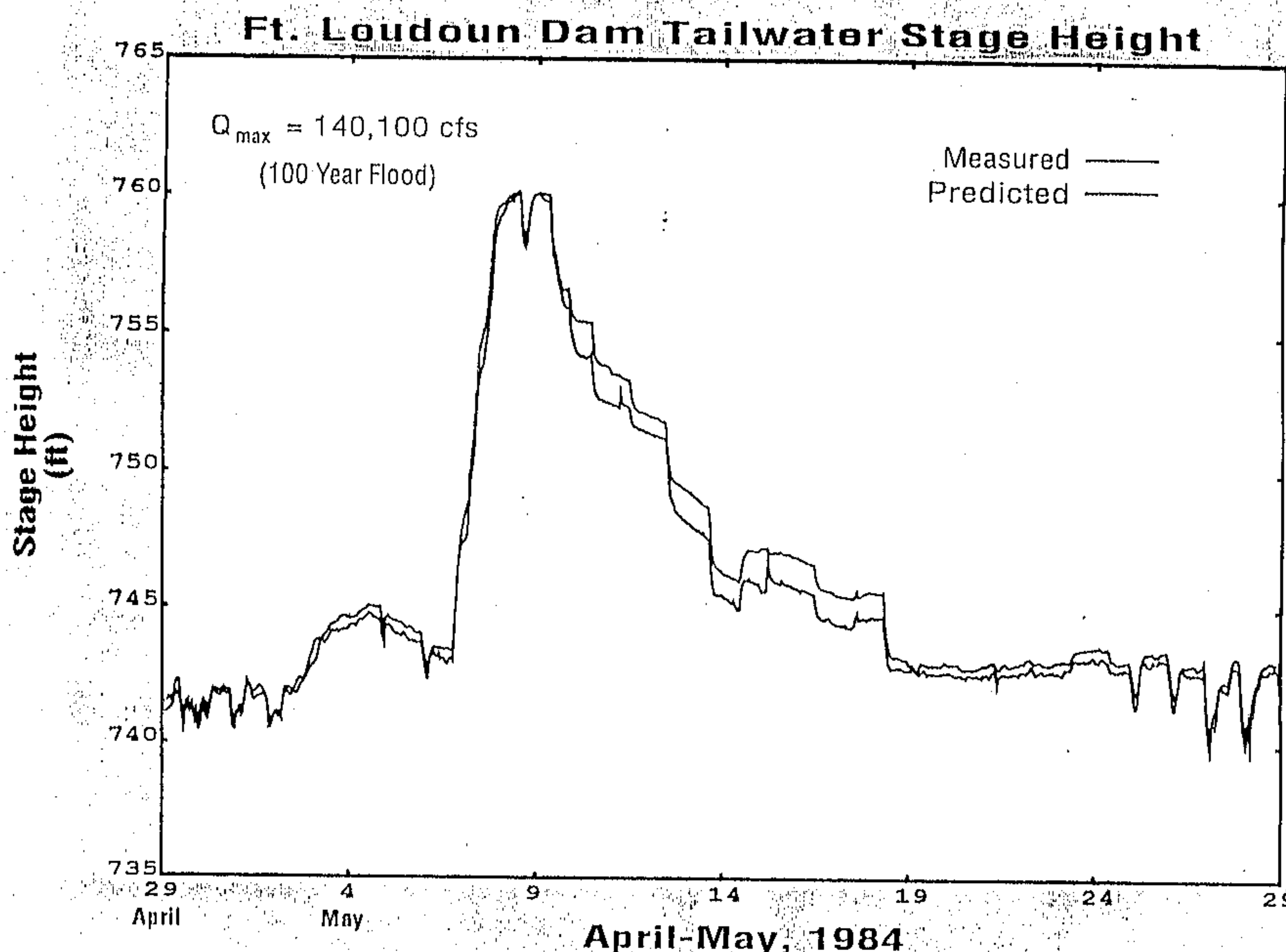
Predicting PCB Transport and Fate in Watts Bar Reservoir



- Sediment transport model calculates water depths, flow rates, deposition/burial rates and resuspension rates
- Parameter values controlling adsorption-desorption, volatization, and diffusion processes were estimated using results of various experimental studies
- Active layer thickness and PCB loads were determined using relevant Watts Bar Reservoir data

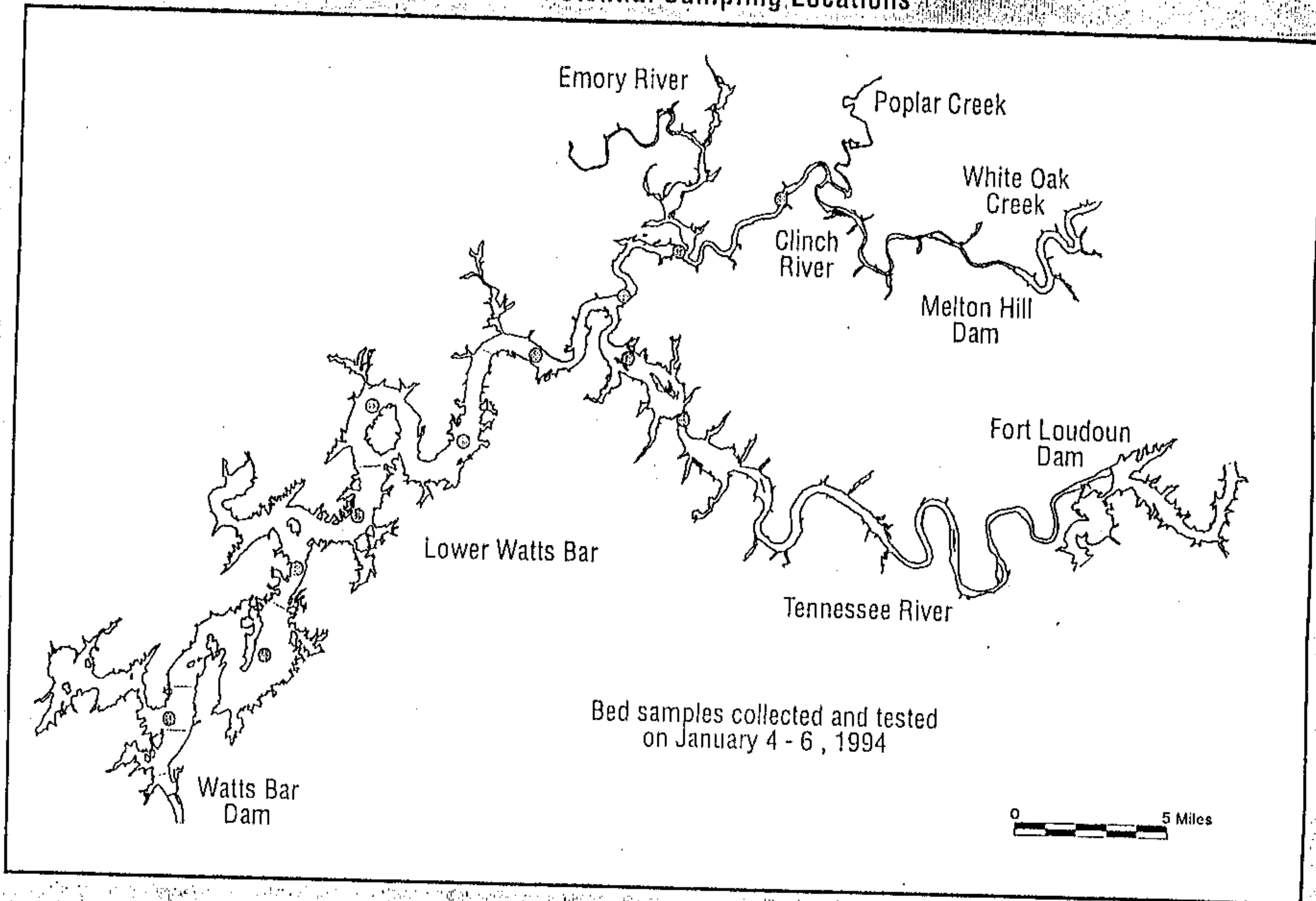
Numerical Grid for Lower Watts Bar Reservoir



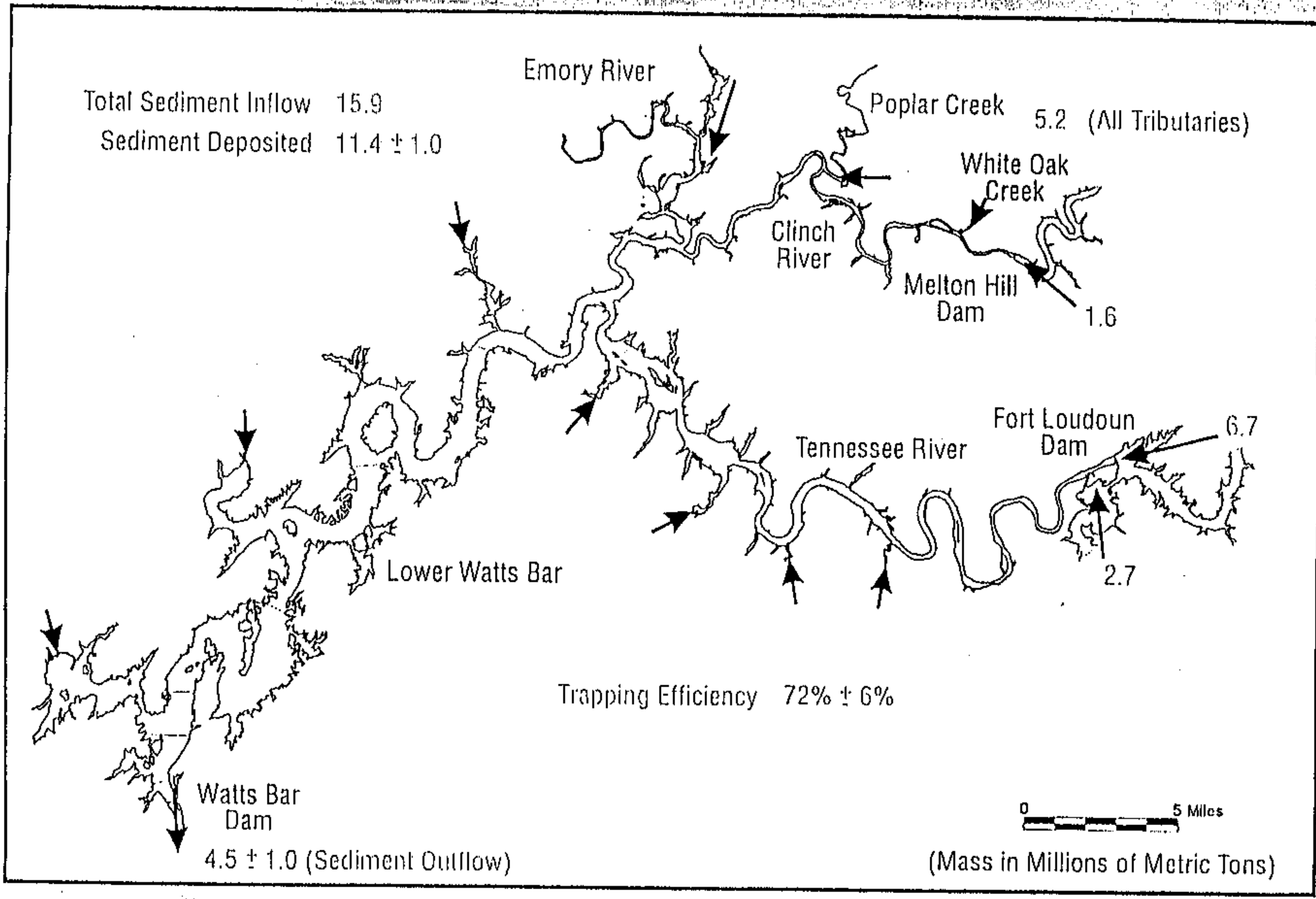


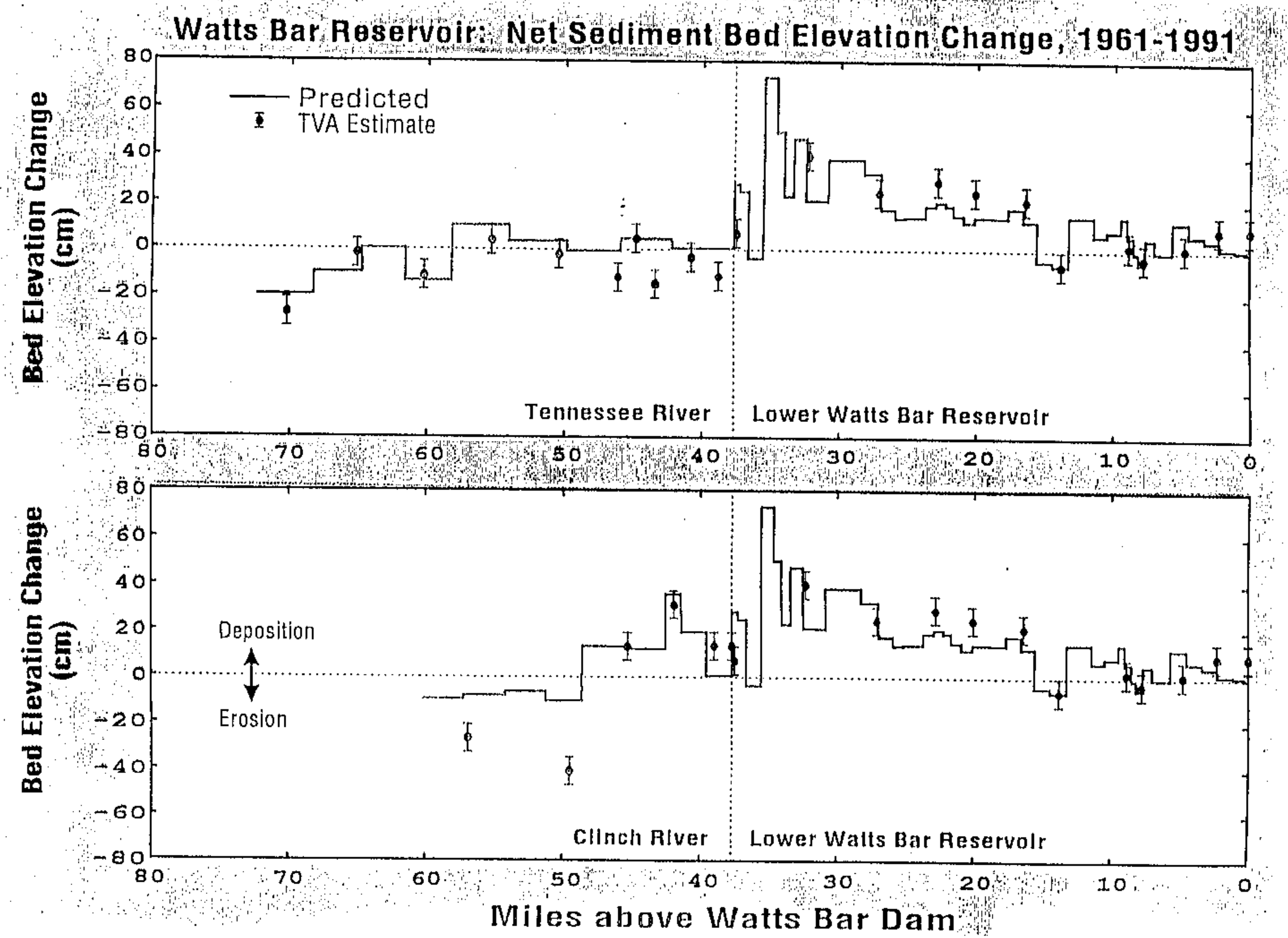
Ft. Loudoun Dam Tailwater Stage Height
Q max = 140,100 cfs
(100 Year Flood)
Measured
Predicted
Stage Height (ft)
765
760
755
750
745
740
735
29 April
4 May
9
14
19
24
29
April-May, 1984

Erosion Potential Sampling Locations

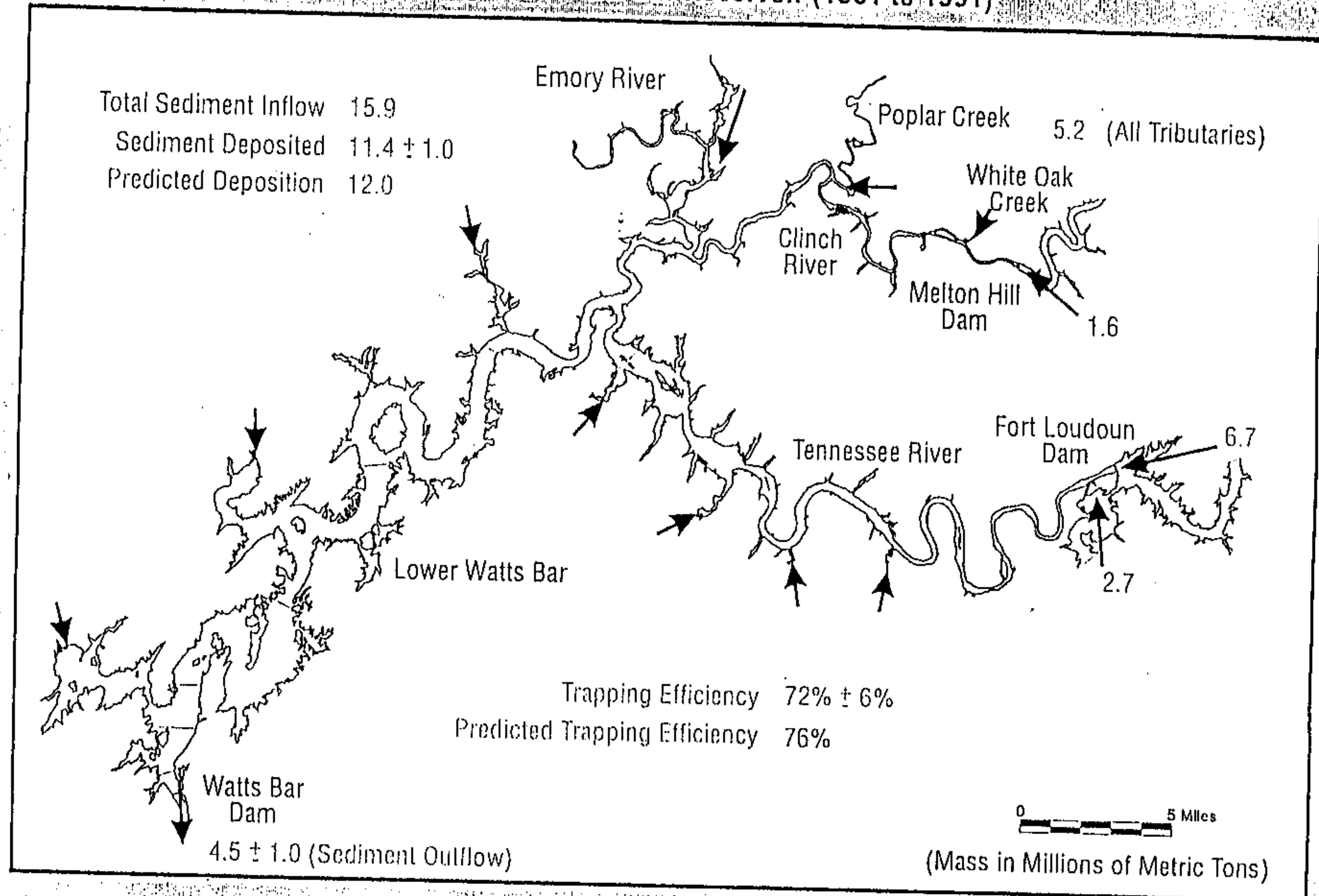


Sediment Budget for Watts Bar Reservoir (1961 to 1991)

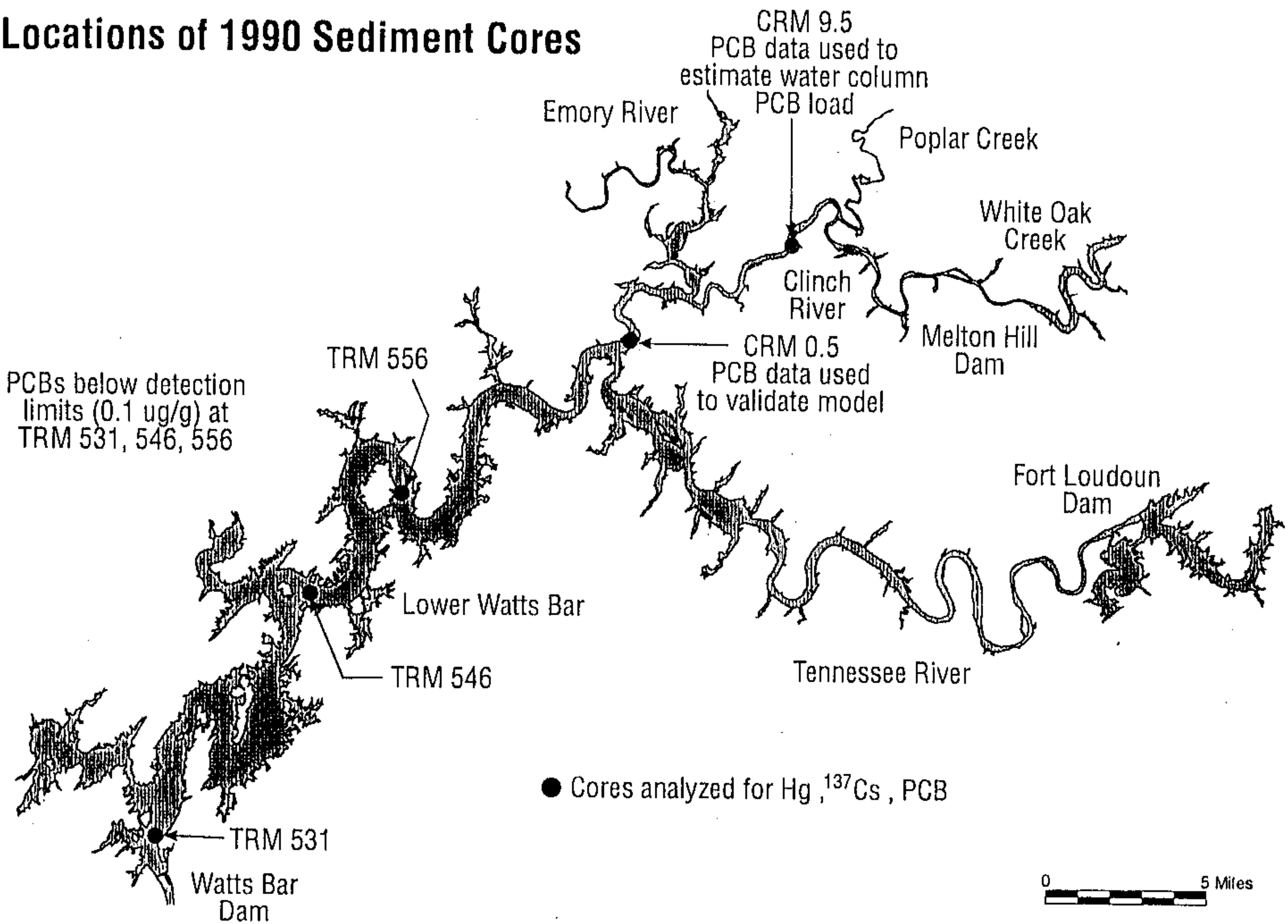



Watts Bar Reservoir: Net Sediment Bed Elevation Change, 1961-1991
Predicted
TVA Estimate
Bed Elevation Change (cm)
Tennessee River
Lower Watts Bar Reservoir
Deposition
Erosion
Clinch River
Lower Watts Bar Reservoir
Miles above Watts Bar Dam

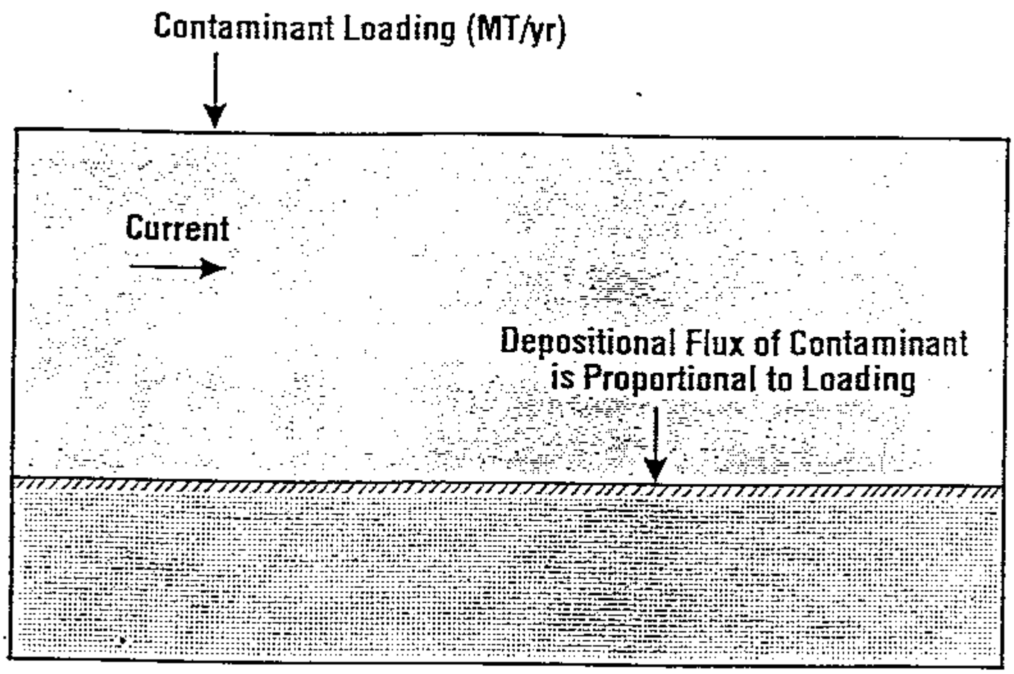
Sediment Budget for Watts Bar Reservoir (1961 to 1991)



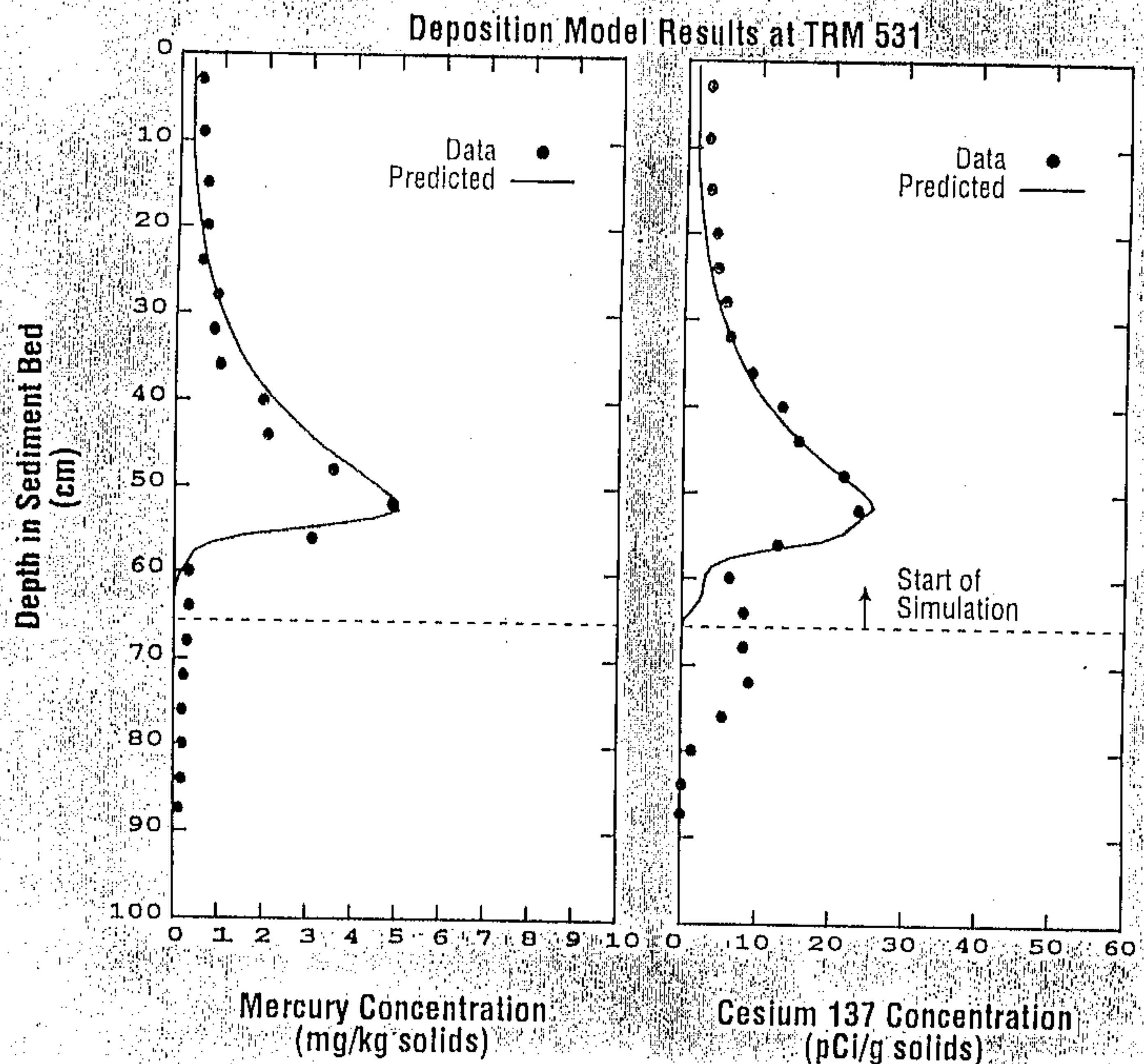
Locations of 1990 Sediment Cores



Simulating Deposition of Bed Contaminants

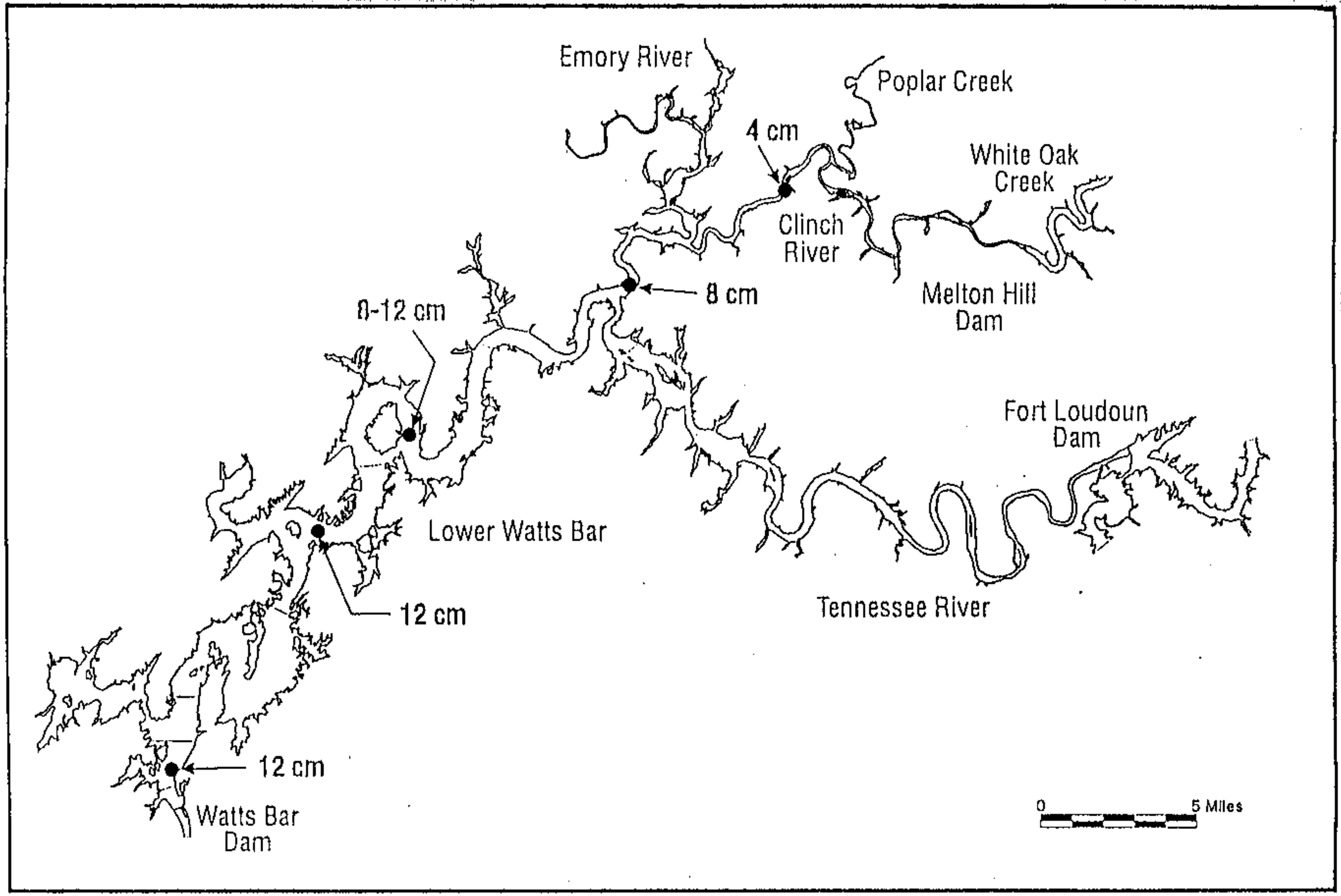


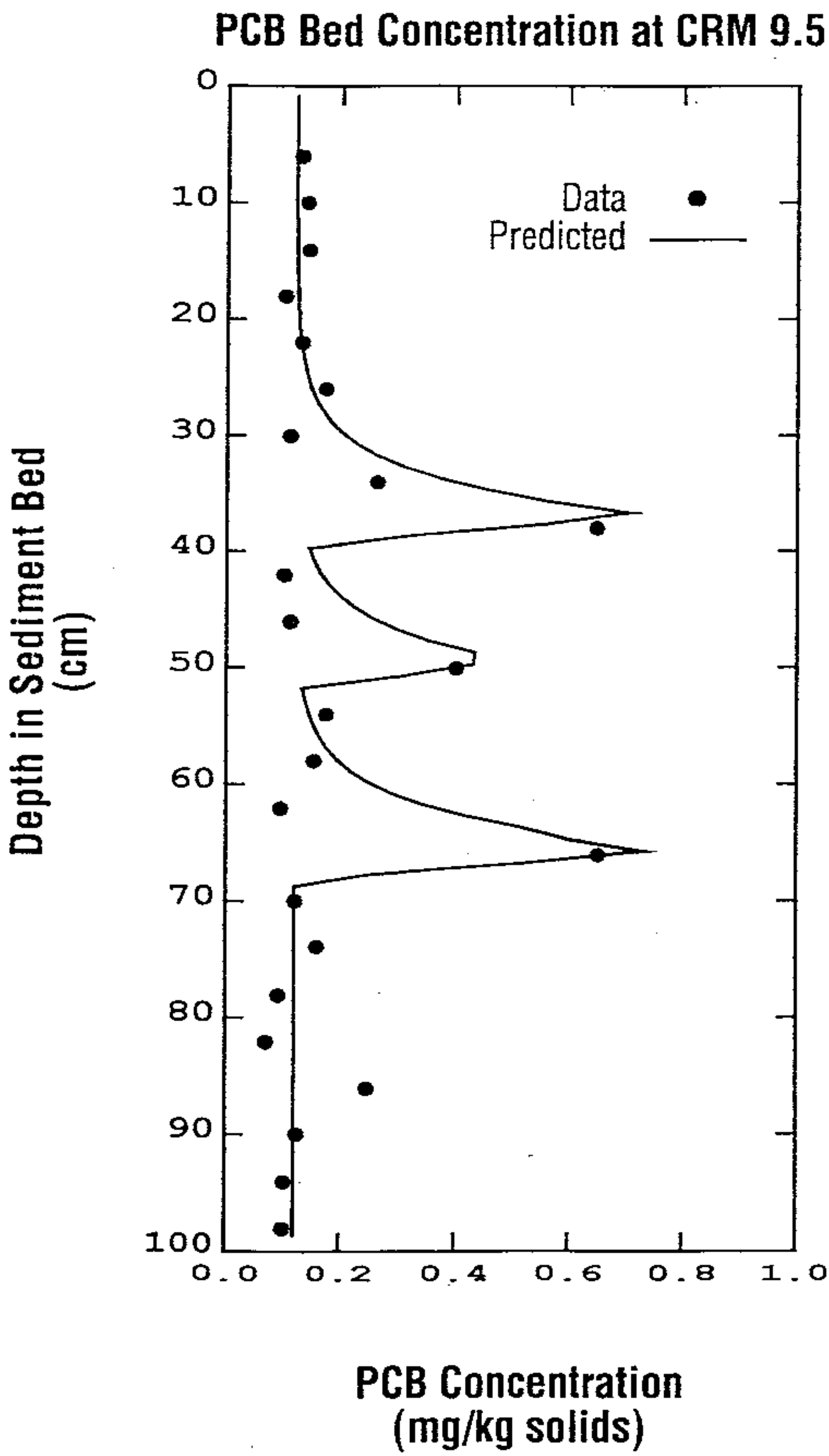
- Discharge histories of contaminants (PCBs, Hg, ^{137}Cs) are recorded in the sediment bed of the reservoir
- Deposition model is used to predict vertical distribution of contaminant concentrations in the sediment bed (model reproduces contaminant record in sediment bed)
- Mercury (Hg) and Cesium (^{137}Cs) data were used to calibrate the depostion model (produced active layer thickness)
- Deposition model was used in inverse mode to predict historical PCB loading in the Clinch River

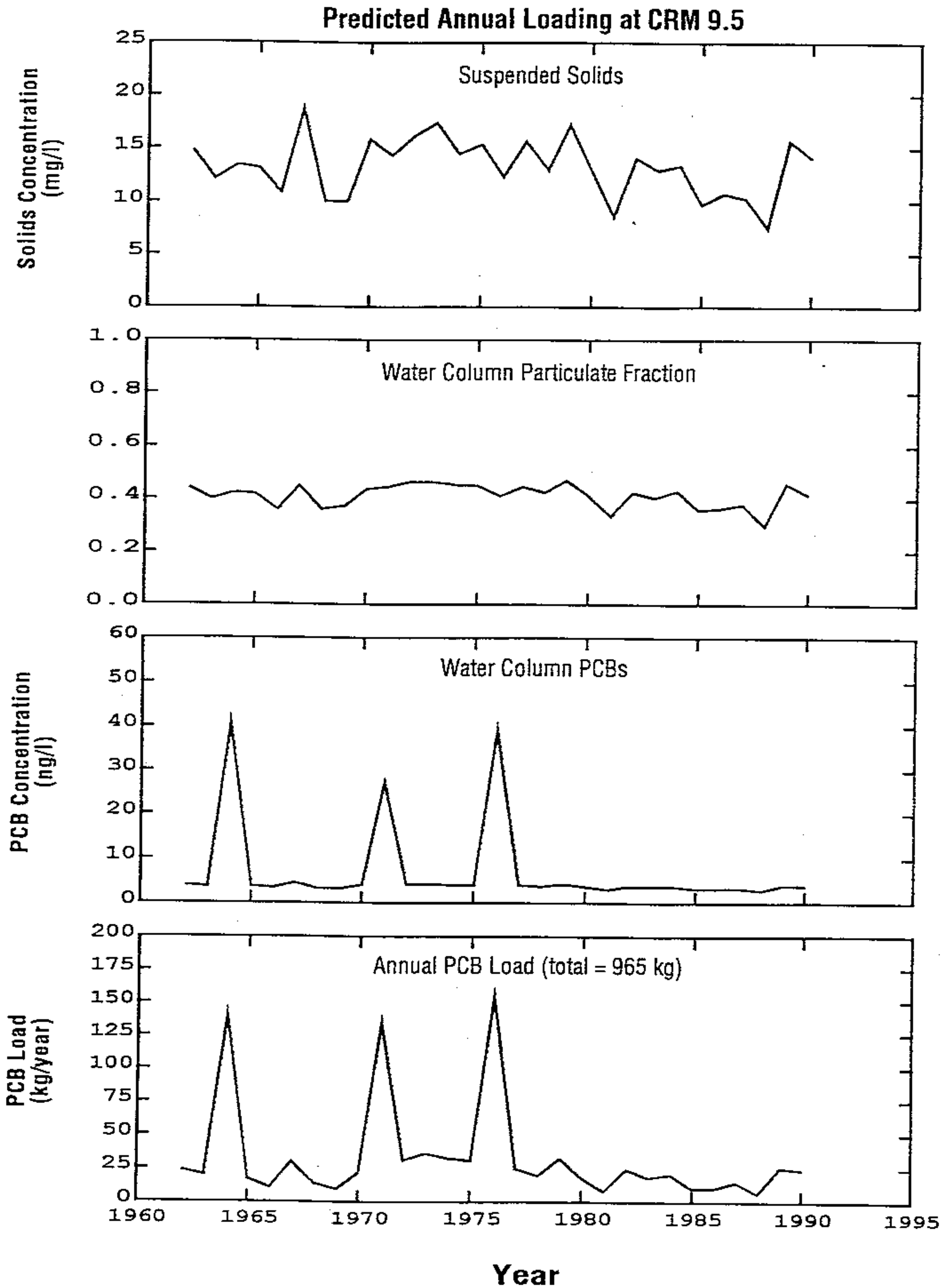


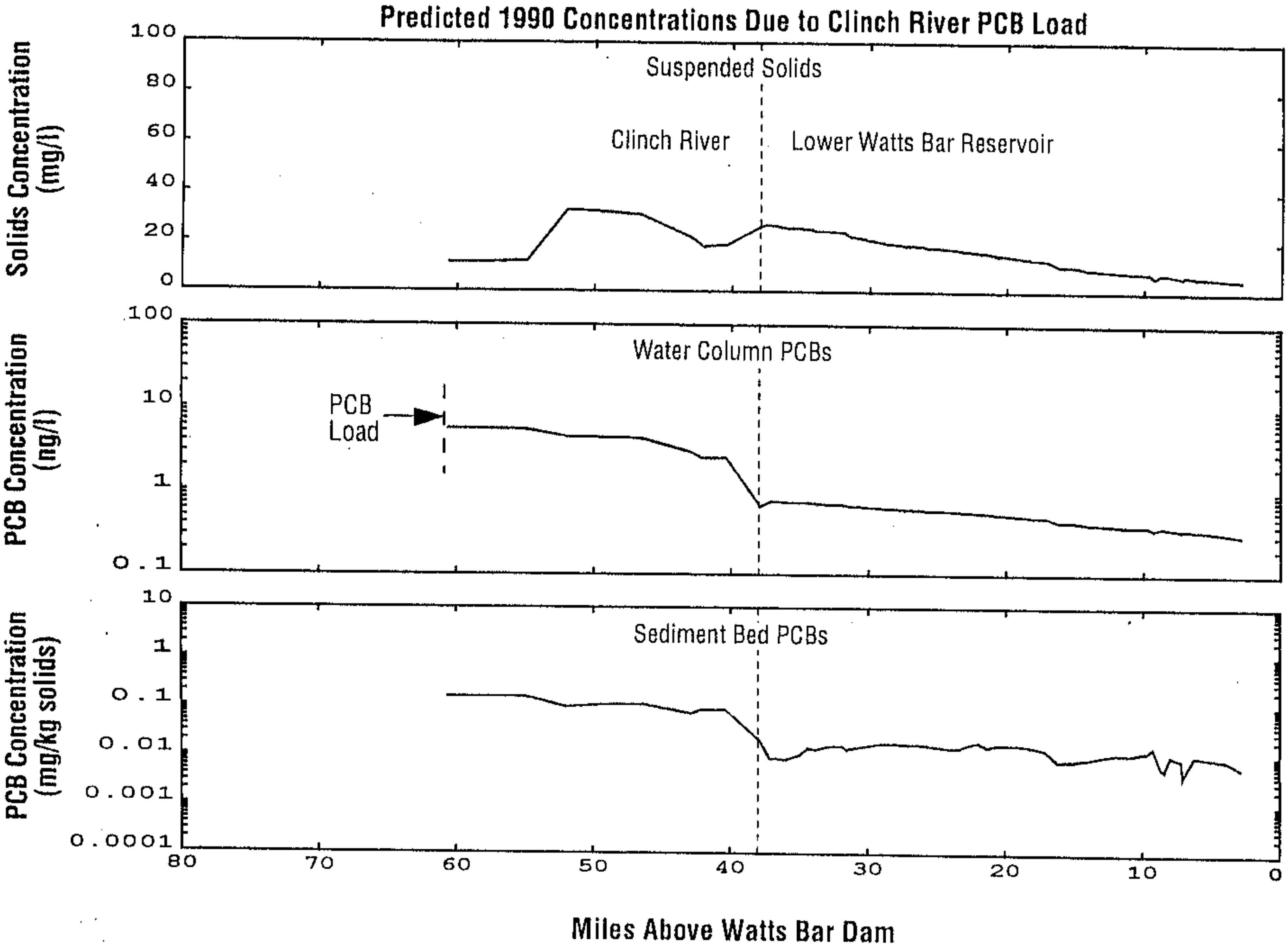
Deposition Model Results at TRM 531
Depth in Sediment Bed (cm)
Data
Predicted
Start of Simulation
0
10
20
30
40
50
60
70
80
90
100
0 1 2 3 4 5 6 7 8 9 10
0 10 20 30 40 50 60
Mercury Concentration (mg/kg solids)
Cesium 137 Concentration (pCi/g solids)

Estimated Active Layer Thickness

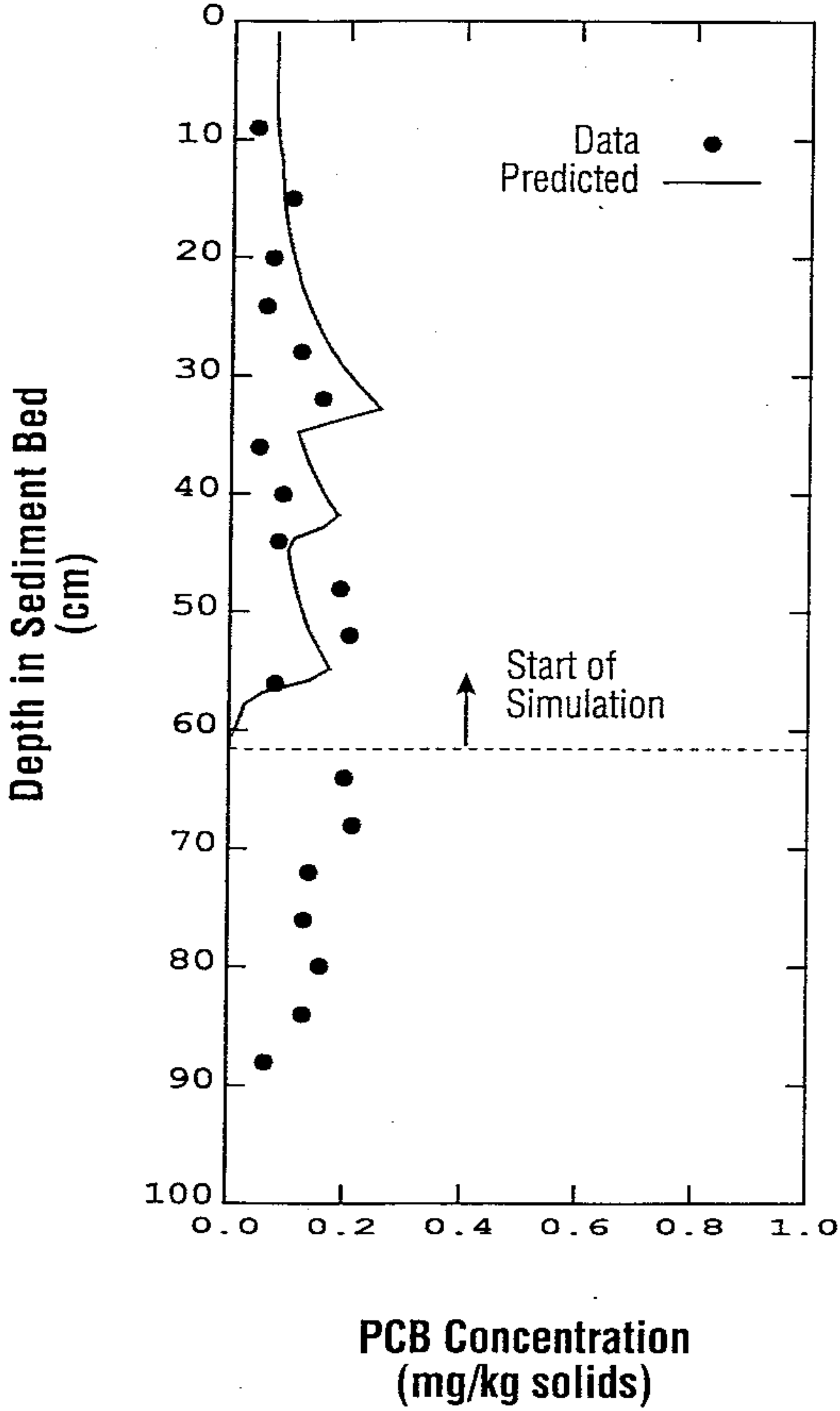


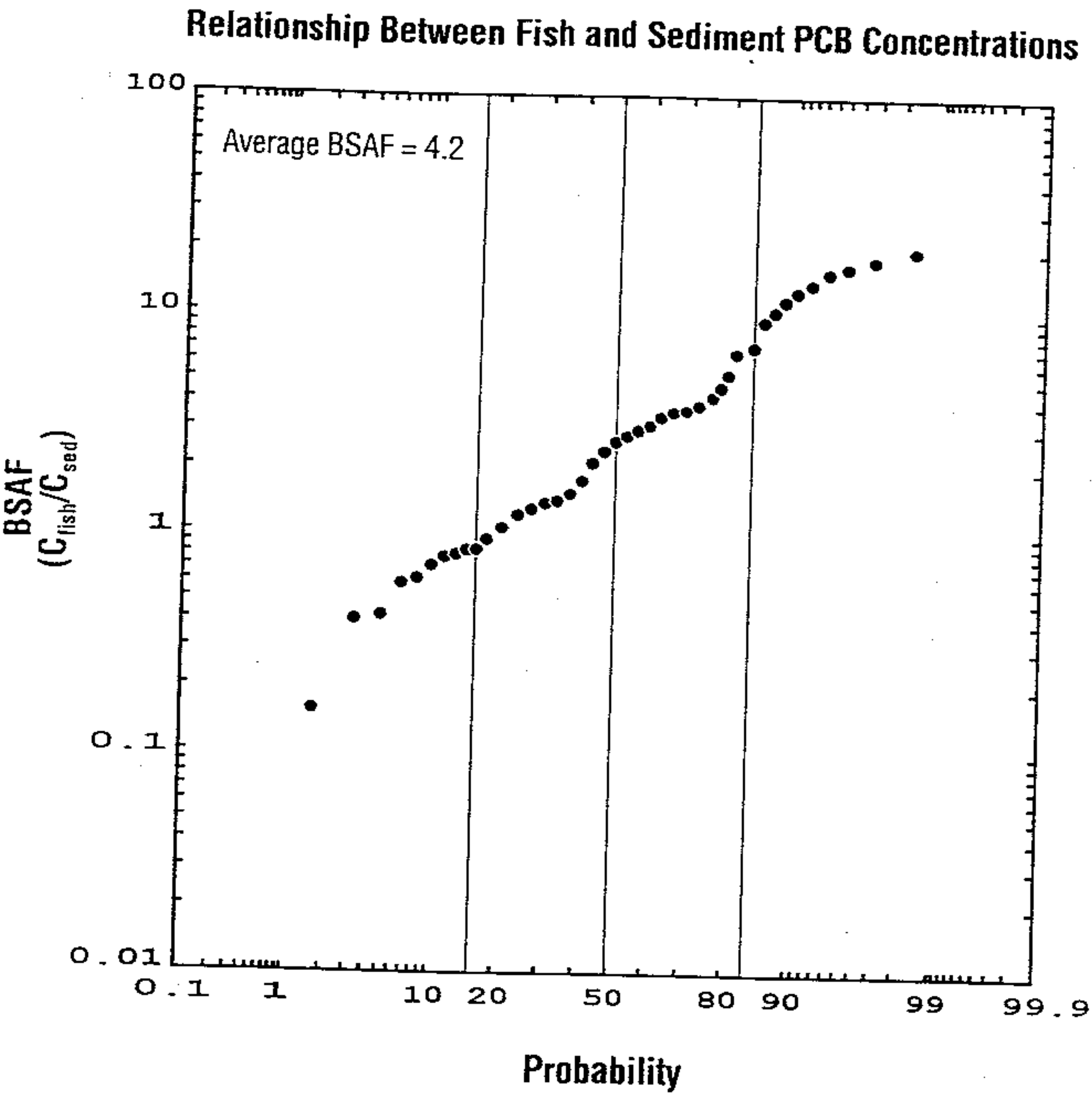

PCB Bed Concentration at CRM 9.5
Data
Predicted
Depth in Sediment Bed
(cm)
0
10
20
30
40
50
60
70
80
90
100
0.0
0.2
0.4
0.6
0.8
1.0
PCB Concentration
(mg/kg solids)


Predicted Annual Loading at CRM 9.5
Suspended Solids
Solids Concentration (mg/l)
Water Column Particulate Fraction
Water Column PCBs
PCB Concentration (ng/l)
Annual PCB Load (total = 965 kg)
PCB Load (kg/year)
Year


Predicted 1990 Concentrations Due to Clinch River PCB Load
Suspended Solids
Clinch River
Lower Watts Bar Reservoir
Solids Concentration (mg/l)
100
80
60
40
20
0
Water Column PCBs
PCB Load
PCB Concentration (ng/l)
100
10
1
0.1
Sediment Bed PCBs
PCB Concentration (mg/kg solids)
10
1
0.1
0.01
0.001
0.0001
80
70
60
50
40
30
20
10
0
Miles Above Watts Bar Dam

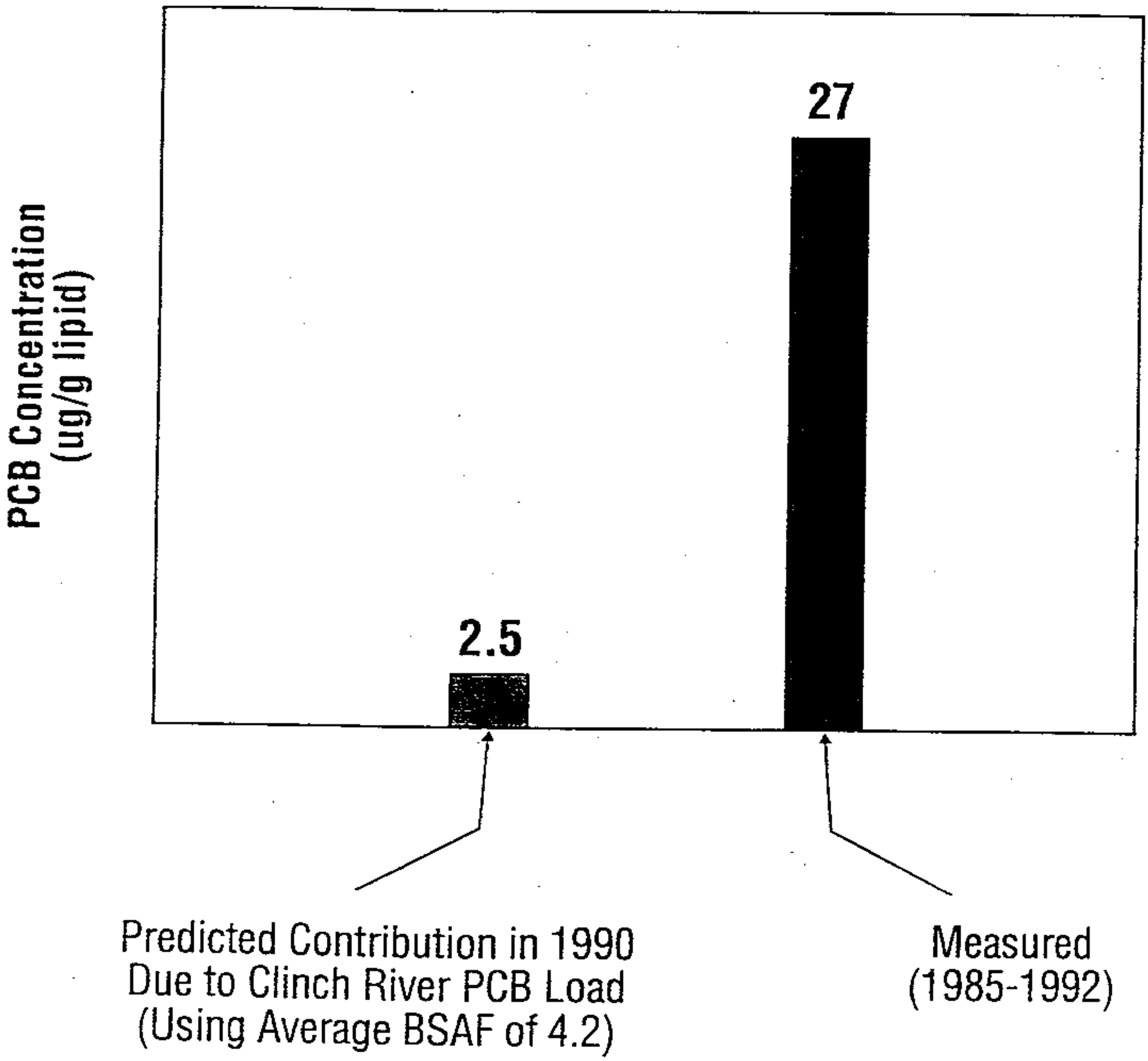
PCB Bed Concentration at CRM 0.5



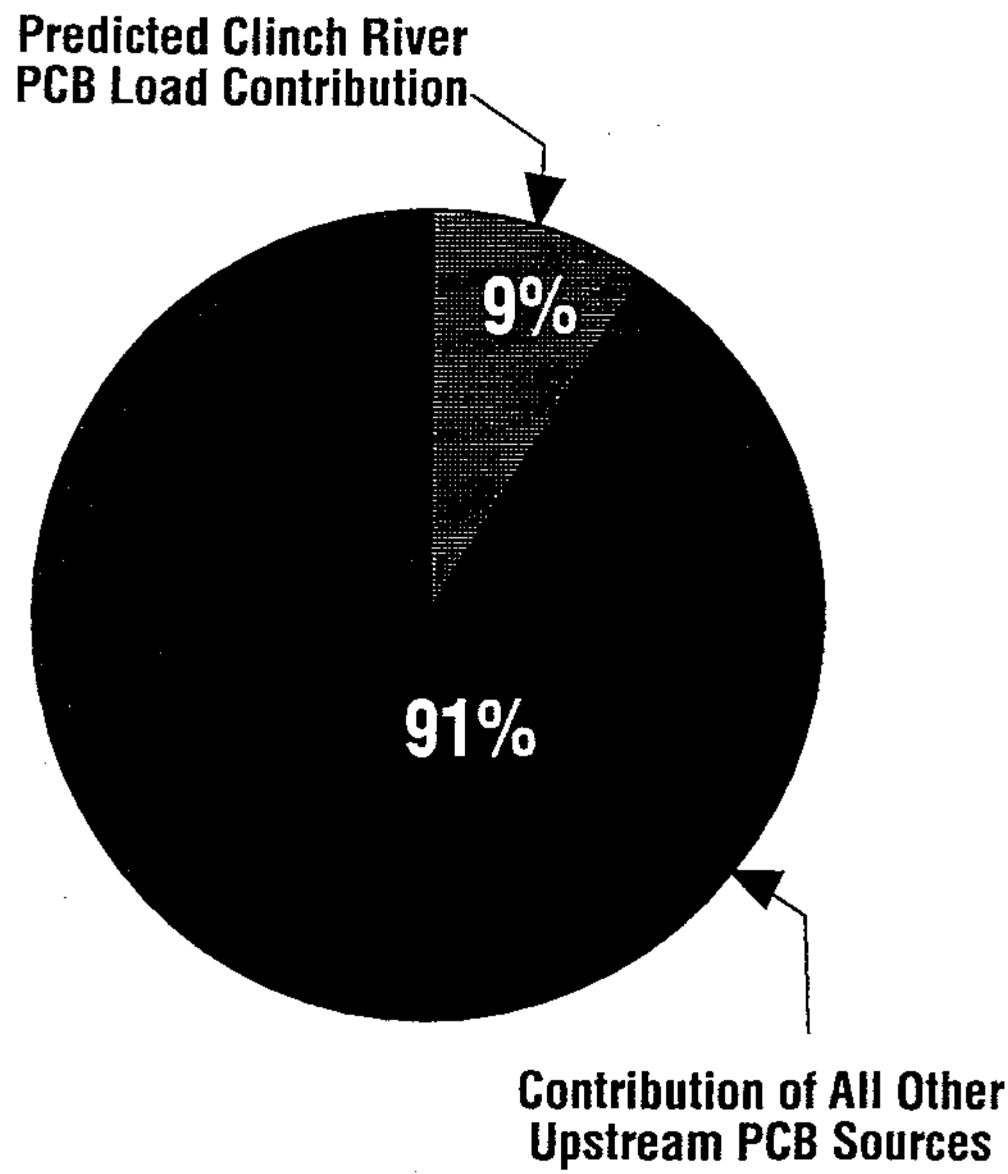


Relationship Between Fish and Sediment PCB Concentrations
Average BSAF = 4.2
BSAF
(C_{fish}/C_{sed})
100
10
1
0.1
0.01
0.1
1
10
20
50
80
90
99
99.9
Probability

Catfish PCB Concentrations in Lower Watts Bar Reservoir



Relative Contribution of Clinch River PCB Load to Catfish PCB Concentrations in Lower Watts Bar Reservoir



APPENDIX B

EVALUATION OF SUSTAINABLE RECREATIONAL FISH HARVEST IN EAST FORK POPLAR CREEK

MEMORANDUM

To: Ellen Ebert

From: Larry Barnthouse and Stephen Deppen

CC: Files

Date: September 16, 1996

Subject: Consumption of Fish from East Fork Poplar Creek

We've completed an initial evaluation of sustainable recreational fish harvest in the publicly accessible portions of East Fork Poplar Creek (EFPC). This memo documents our methods and results.

Site Description and Data Sources

The EFPC drainage basin is located near the northern boundary of the U.S. Department of Energy Oak Ridge Reservation. The creek has a drainage area of 77.2 km^2 from its headwaters above Lake Reality (formerly New Hope Pond) to the mouth into Poplar Creek. The study area for fish consumption includes only those reaches of EFPC that allow public access. Public access is possible from East Fork Poplar Creek Kilometer (EFK) 23.4 just below Lake Reality to approximately EFK 7.8 where EFPC reenters the Oak Ridge Reservation (Hinzman, 1996).

The data used in this analysis were collected during the spring of 1987, 1988, 1994, 1995 and 1996 by the Y-12 Plant Biological Monitoring and Abatement Program. Three study sites in public accessible reaches of the creek were used: EFK 18.2/18.7, EFK 13.8, and EFK 10.0. EFK 18.2/18.7 is located below an area of intensive commercial and limited light industrial development and just above an area of high mercury contamination. This sampling site was located at EFK 18.2 in 1987 and 1988; sampling was moved to EFK 18.7 after 1988 due to construction activities near the original site. EFK 13.8 is located approximately 400 meters above the outfall of the Oak Ridge Wastewater Treatment Facility. EFK 10.0 is located approximately 900 meters below Gum Hollow Road bridge and 3.4 km below the Oak Ridge Wastewater Treatment Facility (Hinzman, 1993 and Loar, 1992). The fourth

BMAP study site, at EFK 6.3 was not used in this analysis because that site is within the reservation boundary.

Estimation of Recreational Fish Abundance

The BMAP sampling procedure involves closing the sampled reach with upstream and downstream seines and then performing multiple passes with electroshocking equipment. This procedure permits estimation of the absolute number of fish present, by species and size class. Species and size-class-specific estimates of fish biomass at each study site were estimated using a computer program developed for BMAP by Railsback et al. (1989). The program uses a maximum weighted likelihood method to estimate population sizes by length class according to the method of Carle and Strub (1978). Annual production and biomass by age class are estimated using to the method of Garman and Waters (1983). Although the program can estimate annual production rates by species and size class, the numbers of fish caught per sampling event were too small to permit such an evaluation. All calculations were performed by Michael Ryon at Oak Ridge National Laboratory.

Only species and size classes likely to be retained and consumed were considered. The species included in the analysis were bluegill *(Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), rock bass (*Ambloplites rupestris*), white sucker (*Catostomus commersoni*), redbreast sunfish (*Lepomis auritis*), green sunfish (*Lepomis cyanellus*), warmouth (*Lepomis gulosis*), redear sunfish *(Lepomis microlophus*), and hybrid sunfish (*Lepomis* sp.). Only fish with a length greater than 5 inches (12.7 cm) were considered.

The total biomass of fish available for consumption, expressed as grams of fish per year per study site, was calculated for each year and sample location. These estimates were then converted to g/m^2 using the total surface area of each sample reach. To estimate the total biomass present in the publicly accessible portion of EFPC, the creek was divided into three reaches, each centered as closely as possible on one of the three study sites. The upper reach extends from EFK 23.4 to EFK 16.3 (midway between sites 18.7 and 13.8). The middle reach extends from EFK 16.3 to EFK 11.8 (midway between sites EFK 13.8 and EFK 10.0). The lower reach extends from EFK 11.8 to the Oak Ridge Reservation boundary at EFK 7.8. The stream lengths for each of the three reaches are, respectively, 7.1 km, 4.4 km and 4.1 km. Information presented in the East Fork Poplar Creek RI Report (SAIC 1993) was used to calculate the total surface area of each reach: 40,000 m^2 for the upper reach, 47,000 m^2 for the middle reach, and 36,000 m^2 for the lower. The harvestable biomass estimate for each study site was scaled up to a total reach biomass using the ratio of the study site surface area to the total reach surface area.

Results, by species, year, site, and reach, are presented in Tables B-1 through B-3.

Estimation of Sustainable Harvest

Ricker (1975) summarized the basic methods for estimating the maximum sustainable harvest or yield (MSY) of fish from a population that can be obtained without causing significant reductions in productive capacity. Two basic approaches are available: the “stock-recruitment” approach, in which quantitative relationships are developed between the numbers and ages of fish reproducing in a given year and the number of offspring produced by those fish, and the “direct estimation” approach, in which relationships are developed between the number of fish caught and the intensity of fishing. The stock-recruitment approach requires, for each population being evaluated, multiple years of observations of the total number of spawning fish (the stock) and the total number of offspring produced by those spawners (the recruits). These data are not available for any of the fish populations present in EFPC. The direct estimation approach requires multiple years of observations of the number of fish caught and the total fishing effort (e.g., fisherman-hours) expended in catching those fish. Since there currently is no fishing in EFPC, this approach is inapplicable. Hence, it is not possible to develop a site-specific estimate of the maximum sustainable harvest of recreational fish from EFPC. However, Ricker (1975) calculated values of MSY, for a range of plausible fish life history characteristics, for the two most common stock-recruitment models (Ricker 1975, Tables 11.6 and 11.7). Depending on life history, the maximum yield according to both models would be obtained when between 20% and 80% of the population were being harvested annually. The lower value would be characteristic of species with very low rates of growth and reproduction (e.g., sturgeon) that are highly vulnerable to overfishing; the higher value would be characteristic of short-lived, highly fecund species (e.g., anchovy or herring) that can sustain very high rates of fishing. Typical recreational fish species examined in this study - primarily bluegills and other related members of the family Centrarchidae - do not fit either of these extreme life history types. An annual harvest of 50% of the average standing biomass should provide a good first approximation to a sustainable rate of harvest for these fish.

Using this value, the maximum sustainable harvest of fish in the upper, middle and lower reaches of EFPC would, averaged over all of the years for which data are available, be approximately 18 kg/yr, 70 kg/yr and 24 kg/yr respectively (Tables B-1 through B-3). The maximum sustainable yield for the length of EFPC that is accessible by the public would be 112 kg/yr.

These estimates reflect the conditions present during the spring sampling events in 1987, 1988, 1994, 1995, and 1996. Inclusion of data from the fall sampling events would provide a more representative estimate of average annual biomass; because the abundance of most species is typically lower in the fall than in the spring inclusion of the fall data should result in lower estimates of sustainable harvest.

These estimates reflect the beneficial effects of improvements in water treatment by both the City of Oak Ridge and the Y-12 plant during the 1970s and early 1980s. No data on fish community abundance or composition are available for years prior to the initiation of the BMAP program, however, it would be expected that the population sizes of most species of recreational interest would have been substantially lower than at present. Most fish species that are attractive to anglers, including bluegill, rock bass, largemouth bass, and most sunfishes, feed on other fish or on large insects such stoneflies and caddisflies. As noted by Karr (1991), in streams affected by organic pollutants these species are usually replaced by omnivorous fish (e.g., carp and green sunfish) that can tolerate low oxygen levels and feed on plants and small, sediment-dwelling invertebrates. To the extent that water quality in EFPC was poorer during the 1950s and 1960s than it is today, the abundance of typical recreational species should have been lower than during the years covered by the BMAP surveys.

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TABLE 1-1
Fish Production for East Fork Poplar Creek
for Creek Mile 18.2/18.7 Spring Samples

Species	Estimated Population by Class >12 cm	Estimated Weight by Class >12 cm	Estimated Total Biomass by Species (g)	Total Biomass of Consumable Fish (g/year)	Annual Total Biomass of Consumable Fish (g/m^2/year)	Upper Reach Biomass of Consumable Fish (g/m^2/year)	Sustainable Biomass of Consumable Fish (g/m^2/year)
1987							
Redbreast Sunfish	2	113.5	227				
Redbreast Sunfish	2	193.5	387				
Warmouth Sunfish	2	122.5	245				
Sucker	4	12.5	50				
Sucker	2	29	58				
Bluegill	3	31.8	95.4				
Bluegill	3	75.3	225.9				
Hybrid	1	84.5	84.5				
Total Biomass				**1372.8**	**1.06**		
1988							
Redbreast Sunfish	1	33.5	33.5				
Sucker	1	63.2	63.2				
Total Biomass				**96.7**	**0.09**		
1994							
Redbreast Sunfish	5	36.9	184.5				
Redbreast Sunfish	3	63.5	190.5				
Redbreast Sunfish	5	99.8	499				
Redbreast Sunfish	3	162	486				
Bluegill	2	38	76				
Total Biomass				**1436**	**1.89**		
1995							
Redbreast Sunfish	4	39.2	156.8				
Redbreast Sunfish	2	65.1	130.2				
Redbreast Sunfish	2	101	202				
Total Biomass				**489**	**0.69**		
1996							
Redbreast Sunfish	4	40.4	161.6				
Redbreast Sunfish	2	64.4	128.8				
Redbreast Sunfish	2	71.2	142.4				
Sucker	1	162	162				
Total Biomass of Consumable Fish				**594.8**	**0.68**	**0.88**	**0.44**

TABLE 1-2
Fish Production for East Fork Poplar Creek
for Creek Mile 13.8 Spring Samples

Species	Estimated Population by Class >12 cm	Estimated Weight by Class >12 cm	Estimated Total Biomass by Species (g)	Total Biomass of Consumable Fish (g/year)	Annual Total Biomass of Consumable Fish (g/m^2/year)	Upper Reach Biomass of Consumable Fish (g/m^2/year)	Sustainable Biomass of Consumable Fish (g/m^2/year)
1987							
Sucker	1	255	255				
Rockbass	1	72	72				
Redbreast Sunfish	5	45.7	228.5				
Redbreast Sunfish	2	96	192				
Total Biomass				**747.5**	**1.05**		
1988							
Rockbass	1	92.4	92.4				
Total Biomass				**92.4**	**0.13**		
1994							
Sucker	1	76	76				
Sucker	1	265	265				
Sucker	1	530	530				
Rockbass	1	78.5	78.5				
Redbreast Sunfish	5	62.8	314				
Redbreast Sunfish	2	104	208				
Redbreast Sunfish	2	134	268				
Warmouth Sunfish	1	123	123				
Bluegill	1	119	119				
Total Biomass				**1981.5**	**2.61**		
1995							
Sucker	5	124.8	624				
Sucker	3	335.3	1005.9				
Sucker	1	675	675				
Rockbass	1	44.5	44.5				
Rockbass	1	74.2	74.2				
Redbreast Sunfish	4	37.5	150				
Redbreast Sunfish	7	68.4	478.8				
Redbreast Sunfish	7	110.5	773.5				
Redbreast Sunfish	6	139.8	838.8				
Green Sunfish	1	43.7	43.7				
LM Bass	1	291	291				
Readear Sunfish	1	208	208				
Total Biomass				**5207.4**	**7.23**		

TABLE 1-2
Fish Production for East Fork Poplar Creek
for Creek Mile 13.8 Spring Samples

Species	Estimated Population by Class >12 cm	Estimated Weight by Class >12 cm	Estimated Total Biomass by Species (g)	Total Biomass of Consumable Fish (g/year)	Annual Total Biomass of Consumable Fish (g/m^2/year)	Upper Reach Biomass of Consumable Fish (g/m^2/year)	Sustainable Biomass of Consumable Fish (g/m^2/year)
1996							
Sucker	1	42.3	42.3				
Sucker	1	310	310				
Sucker	1	642	642				
Rockbass	1	151	151				
Redbreast Sunfish	3	36.3	108.9				
Redbreast Sunfish	3	65.8	197.4				
Redbreast Sunfish	6	100.1	600.6				
Redbreast Sunfish	2	138.5	277				
Green Sunfish	1	26.7	26.7				
Bluegill	2	72.7	145.4				
Readear Sunfish	1	51.7	51.7				
LM Bass	2	214.5	429				
Total Biomass of Consumable Fish				**2982**	**3.73**	**2.95**	**1.48**

TABLE 1-3
Fish Production for East Fork Poplar Creek
for Creek Mile 10 Spring Samples

Species	Estimated Population by Class >12 cm	Estimated Weight by Class >12 cm	Estimated Total Biomass by Species (g)	Total Biomass of Consumable Fish (g/year)	Annual Total Biomass of Consumable Fish (g/m^2/year)	Upper Reach Biomass of Consumable Fish (g/m^2/year)	Sustainable Biomass of Consumable Fish (g/m^2/year)
1987							
Redbreast Sunfish	5	64.2	321				
Warmouth Sunfish	1	48	48				
Total Biomass				**369**	**0.44**		
1988							
Redbreast Sunfish	3	95.9	287.7				
Warmouth Sunfish	4	69.8	279.2				
Rockbass	4	51	204				
Rockbass	1	132.3	132.3				
Bluegill	1	278	278				
Sucker	1	39	39				
Total Biomass				**1220.2**	**1.47**		
1994							
Rockbass	3	46.9	140.7				
Warmouth Sunfish	1	135	135				
Bluegill	1	92.3	92.3				
LM Bass	1	872	872				
Total Biomass				**1240**	**1.63**		
1995							
Rockbass	2	74.6	149.2				
Rockbass	1	223	223				
Redbreast Sunfish	2	63.5	127				
Redbreast Sunfish	5	103.8	519				
Bluegill	3	34.3	102.9				
Total Biomass of Consumable Fish				**1121.1**	**1.72**	**1.32**	**0.66**

APPENDIX C

DERIVATION OF EXPOSURE POINT CONCENTRATIONS FOR AIR PATHWAYS

APPENDIX C: DERIVATION OF EXPOSURE POINT CONCENTRATIONS FOR AIR PATHWAYS

The project team estimated ambient air concentrations of PCBs and PCDF at receptor locations outside of the ORR using SCREEN3, a screening-level gaussian air dispersion model (EPA, 1995). The SCREEN3 model simulates 52 different meteorological conditions covering six atmospheric stability classes and twelve wind speed. The model determines which meteorological condition would result in maximum downwind impacts, and calculates one-hour average concentrations for user-defined distances (EPA, 1995). One-hour average ambient air concentrations are converted to annual averages by multiplying the one-hour average value by 0.08 (EPA, 1992). The results of the SCREEN3 modeling are likely to overestimate the airborne concentrations at the exposure points, providing a conservative exposure point concentration. For the purpose of this analysis, we evaluated direct exposures (inhalation of air) for the nearest residential communities and we evaluated indirect exposures for the nearest farm families.

Residential exposures to PCBs and their by-products are impacted by the complex terrain surrounding the ORR and the prevailing meteorological conditions that exist there. The Y-12 Burial Grounds are located within Bear Creek Valley, with Pine Ridge and Chestnut Ridge rising sharply to the north and south, respectively. General air movement within the valley limits dispersion and migration of airborne chemicals to either up valley (northeasterly) or down valley (southwesterly) movement; intervalley air exchange is limited, but some cross valley transport does occur.

The K-25 Burn Tank was located on a peninsula that extends northward into Poplar Creek, approximately three miles above its confluence with the Clinch River. The TSCA incinerator is located in close proximity to the site of the former Burn Tank. Black Oak Ridge rises to the north of the site and Pine Ridge rises to the south. These ridges effectively enclose the Burn Tank area and provide physical barriers to the north and south. The assessment of risks from air pathways requires the establishment of receptor locations. Air releases in the vicinity of the ORR occurred from the northern portion of K-25 (the K-25 Burn Area and the TSCA incinerator) and from the Bear Creek disposal areas (Y-12 Burn Pit and Y-12 Burn Tank). We evaluated inhalation exposure for receptors that lived in the community of Hartland Estates located north of the K-25 facility. This location was selected because it was the closest community to K-25. The receptors for direct exposure from the sources at Y-12 was the community of Scarboro. The receptors for indirect exposures (consumption of vegetables, milk, beef) were the nearest farms. In the case of K-25, we estimated that the nearest farms were across the Clinch River, near the town of Union. In the case of Y-12, the nearest farms were those located along EFPC.

Y-12 Burial Ground A Burn Pit Air Releases

To estimate the EPC of unburned PCBs and PCDFs from the Y-12 Burial Ground A Burn Pit, the project team assumed that the waste oils burned were similar to the Z-oils used at the Y-12 facility. The Z-oil is described as a mineral oil similar in nature to the oil used in transformers and other electrical equipment (Wing, 1980; Bailey, 1986). Initially, the Z-oils did not contain PCBs (Banic, 1995a; Napier, 1995; Bailey and Thorton, 1995; Hummel, 1995), but in 1979 PCBs were detected at concentrations of about 100 ppm in the Z-oil during routine sampling of the oil system at the plant (Wing, 1980; Health, Safety, Environment, and Accountability Division, 1992). The project team assumed that the material combusted at the Burial Ground A Burn Pit was consistent with the Z-oil at the Y-12 facility, and that it contained approximately 100 ppm PCBs.

There are no records regarding the precise volume of waste liquids burned at the Y-12 Burn Pits between 1955 and 1961. However, data from 1961 to 1968 show that approximately 180,000 gallons of waste oils were burned in the burn tank during this time period (Turner et al., 1988). For the Burn Pit scenario, the project team assumed that during 1955 to 1961, waste oils were accumulated and burned at the same rate as that reported between 1961 and 1968. Given this assumption, approximately 22,500 gallons of waste oils were burned per year at the Burn Pit. Assuming 260 days of operation per year, about 86.5 gallons of oil were combusted during each day of operation.

While no records exist describing the content of the releases from the Burn Tank, laboratory experiments have been performed to measure PCB destruction efficiency and the formation of PCDFs from the combustion of PCB-containing mineral oils under excess oxygen conditions (Erickson, 1985; Addis, 1986). Addis (1986) reported the PCB destruction efficiency and creation of PCDF formed during the combustion of PCB-contaminated mineral oils, and quantified the formation of PCDFs as a percent of the PCB content in the unburned mineral oil. The experiments were performed iteratively to find the conditions for optimum PCDF formation. Addis (1986) used PCB concentrations in mineral oil of 50, 500, and 5000 ppm, resulting in total PCDF formation of 0.656 percent, 0.814 percent, and 1.47 percent, respectively. The PCB destruction efficiency at these concentrations ranged from 82 to 90 percent. Because the estimated PCB content of the waste oil, 100 ppm, was closest to the 50 ppm mineral oil mixture used by Addis (1986), the project team assumed that PCDF formation during combustion of waste oils equaled 0.656 percent, and that the PCB destruction efficiency was 88 percent (the low end of the range of PCB destruction efficiencies reported for the 50 ppm PCB in mineral oil by Addis (1986)). Further, Addis (1986) found that PCDF formation during the burning of the 50 ppm PCB in mineral oil was characterized by approximately 30 percent trichlorodibenzofuran, 30 percent tetrachlorodibenzofuran, and 5.6 percent pentachlorodibenzofuran. The project team assumed that the formation of the PCDF isomers while burning 100 ppm oil was equivalent to that measured by Addis (1986).

The project team calculated the International Toxicity Equivalence Factor (ITEF) of the tri-, tetra- and pentachlorinated dibenzofuran isomers using the guidance provided in EPA (1989). The ITEF method is a procedure that simplifies risk assessments associated with complex mixtures of chlorinated furans and dioxins by normalizing the toxicity of the 2,3,7,8-substituted dioxin and furan congeners to that of 2,3,7,8-tetrachlordibenzodioxin (2,3,7,8-TCDD) (EPA, 1989). This allows the risk assessor to use one value in assessing risk; the 2,3,7,8- TCDD toxicity equivalent (TEQ).

Trichlorodibenzofurans are not 2,3,7,8-substituted and are much less biologically active than the 2,3,7,8-substituted congeners (EPA, 1989). EPA (1989) has eliminated the non-2,3,7,8-substituted congeners from the ITEF calculations because of their limited biological activity. Therefore, the fraction of the PCDF that is made up of the trichlorodibenzofuran was not considered further. The tetra- and pentachlorodibenzofurans, however, can be 2,3,7,8-substituted congeners; as a result, they were included in this analysis.

Based on the work of Rappe et al. (1982), the project team calculated the fraction of the tetra- and pentachlorinated dibenzofurans that may have been 2,3,7,8- substituted. Based on the analysis of soot from the combustion of PCB laden mineral oil, Rappe et al. (1982) found that 9.38 percent of the tetrachlorinated dibenzofurans, and 26.47 percent of the pentachlorinated dibenzofurans produced were 2,3,7,8-substituted. Therefore, to calculate the release rate of 2,3,7,8-TCDD TEQ, the project team calculated the amount of 2,3,7,8-substituted tetra- and pentachlorinated dibenzofurans produced, based on the proportions reported by Rappe et al., and then applied the ITEF factors to derive a final estimated 2,3,7,8-TCDD TEQ.

Combustion of waste oils was assumed to occur for 260 days per year for 8 hours per day. This resulted in a combustion rate of 10.8 gallons of waste oil per hour. With a PCB content of 100 ppm, the resulting PCB combustion rate was 1137 g PCB/sec. Assuming a PCB destruction efficiency of 88 percent, the resulting PCB release rate was estimated at 1.1 x 10-4 g PCB/sec. Further, based on the formation rates summarized above, the release rate of 2,3,7,8-TCDD TEQ was estimated to be 1.2 x 10-7 g/sec.

The releases from the Burn Pit were modeled as a volume release source because the fire was likely to have risen out of the pit and into the surrounding air. It was assumed that the height of flame zone above ground level was three meters. The source was modeled at a unit release rate of 1 g/sec, providing a χ/Q value for each receptor location. The χ/Q value represents the air concentration at that receptor for each gram of chemical emitted per second (g sec/m3 g). The χ/Q value was multiplied by the estimated PCB and TCDD TEQ release rates (1.4 x 10^{-4} and 1.2 x 10^{-7}g/sec, respectively) to yield their airborne concentrations at the receptor.

Y-12 Burn Tank Air Releases

Waste oils and solvents were burned in an open steel tank at the Burial Ground A area from 1961 to 1968. Although the content and origin of the material burned in the steel tank are not known, the contents of the liquid wastes are believed to have consisted of non-PCB containing mineral oils, motor oils, anti-freeze, and cutting oils (Bailey, 1995a). Like the waste liquid burned in the Burn Pit, it is possible that the liquid may have been contaminated with PCBs during its use, storage, or disposal. Therefore, the project team evaluated the potential for the releases of unburned PCBs and the creation of PCDFs during combustion of the waste.

The project team assumed that the liquid burned in the Steel Tank was similar to the Z-oil used in the Y-12 facility. The oil was assumed to be contaminated with PCBs at a concentration of 100 ppm (Wing, 1980; Health, Safety, Environment, and Accountability Division, 1992). The bulk of the Z-oil was assumed to be consistent with mineral oil (Wing, 1980; Bailey, 1986).

Records indicated that approximately 180,000 gallons of waste oil were burned in the Steel Tank from 1961 to 1968. The project team assumed that the waste was generated at a constant rate over that eight-year period, resulting in an annual combustion rate of 22,500 gallons per year. It was assumed that combustion in the steel tank took place 8 hours per day, 5 days per week, for 52 weeks each year. Based on these assumptions, the resulting combustion rate of waste oil was approximately 10.8 gallons per hour.

Using the same logic as described for the Burial Ground A Burn Pit, the project team estimated an release rate of PCBs at 1.1 x 10-4 g PCB/sec, and a 2,3,7,8-TCDD TEQ release rate of approximately 1.2 x 10-7 g/sec from the liquid waste fire.

Using EPA's SCREEN3 Gaussian air dispersion model, the project team estimated the contributions of PCBs and 2,3,7,8-TCDD TEQ to ambient air at an off-site receptor resulting from the Steel Tank combustion releases. The project team modeled the Steel Tank releases as a volume release source, assuming that the dimensions of the tank were approximately 12 meters long by 4 meters wide with a flame height of 3 meters.

K-25 Drum Storage and Burn Area (Unit K-1064)

To estimate EPCs of PCBs and PCDFs from the K-25 Burn Area, the project team assumed that the composition of the waste oil burned during the 1950s had the same PCB content as the wastes stored in drums at K-25 during the 1960s and 1970s. Data regarding the PCB content of these wastes were evaluated to estimate an average chemical makeup of the drummed liquids.

In 1979, over 700 drums of waste oil were tested for PCB content. Of the 700 drums, 34 (4.9%) were found to contain PCBs at concentrations less than 50 ppm, 31 (4.4%) contained between 50 and 500 ppm PCB, and 13 (1.9%) contained PCBs at concentrations greater than 500 ppm (Long, 1980s). To estimate the average PCB concentration in the material burned at the Burn Tank, it was conservatively assumed that those drums with less than 50 ppm PCB contained 50 ppm of PCBs, the drums with PCBs between 50 and 500 ppm contained 500 ppm PCB, and those with concentrations greater than 500 ppm contained 1,000 ppm PCBs. The percent volume for each PCB level (4.9%, 4.4 %, or 1.9%) was multiplied by the assigned PCB concentration (50 ppm, 500 ppm, or 1,000 ppm) to derive a weighted average PCB concentration of 43 ppm. This weighted average was used to estimate the concentration of PCBs found in the waste that may have been burned at K-25.

There are no records regarding the precise volume of waste liquids burned at the K-25 Burn Tank during the 1950s. Data from 1960 to 1979 indicated that 1,838 drums, with a total capacity of 90,000 gallons, were stored at the facility during it's use as a drum storage facility (MMES, 1991). For the Burn Tank scenario, the project team assumed that between 1950 and 1960, waste oils were accumulated and burned at the same rate as was recorded between 1960 and 1979. Given this assumption, approximately 4,750 gallons of waste were burned per year at the Burn Tank.

No records regarding the burning schedule was found, so the project team assumed that combustion of waste oils occurred 5 days per week or 260 days per year. This resulted in a combustion rate of 18.2 gallons of waste oil per day. The project team assumed that this volume of waste was burned every day for eight hours. With a PCB content of 43 ppm, the resulting PCB combustion rate was estimated to be 824 g PCB/sec. Assuming a PCB destruction efficiency of 88 percent, the resulting PCB release rate was estimated at 9.9 x 10-5 g PCB/sec. Further, using the same assumptions as those used for the Y-12 burn areas, regarding thermochemical formation of furans with subsequent conversion to 2,3,7,8-TCDD TEQ, the project team calculated an release rate of 2,3,7,8-TCDD-TEQ of 8.6 x 10-8 g/sec.

The Burn Tank was modeled as a volume release source because the fire was likely to have risen out of the tank and into the surrounding air. It was assumed that the height of flame zone above ground level was three meters. The project team set the model to calculate a one-hour average impact. The one-hour impact was chosen with respect to the assumed daily combustion rate as described previously. The source was modeled at a unit release rate of 1 g/sec, providing a /Q value that was multiplied by the actual PCB and 2,3,7,8-TCDD TEQ release rates (9.9 x 10-5 and 8.6 x 10-6 g/sec, respectively). The resulting products were assumed to be the ambient air PCB and 2,3,7,8-TCDD TEQ concentrations at the receptor location.

K-25 TSCA Incinerator

The TSCA incinerator was designed to destroy solid and liquid uranium-contaminated PCB waste and hazardous organic waste materials (Joyner, 1991). The incinerator utilizes a rotary kiln (primary combustion chamber) and secondary combustion chamber to destroy the waste materials fed into the system. The rotary kiln operates at a temperature of approximately 1,800F and the secondary combustion chamber operates at 2,200F. Both the rotary kiln and the secondary combustion chamber use either natural gas or oil as an auxiliary fuel source. Solids are fed into the rotary kiln using a hydraulic ram feeder, while liquids and sludges are injected directly to the rotary kiln. Off gas from the rotary kiln is directed to the secondary combustion chamber where it undergoes final combustion. The residence time in the secondary combustion chamber is four seconds (Engineering Science, 1988).

According to Engineering Science (1988), the TSCA incinerator is used to dispose of PCB waste and hazardous materials originating from the following DOE facilities:

- Oak Ridge Gaseous Diffusion Plant,
- Paducah Gaseous Diffusion Plant,
- Y-12 Plant,
- Portsmouth Gaseous Diffusion Plant,
- Feed Materials Production Center (Fernald),
- Oak Ridge National Laboratory, and
- RMI Extrusion Plant (Ashtabula, Ohio).

The waste materials burned include:

- PCB-contaminated wastes (liquid and solid),
- Hazardous organic wastes (liquid and solid),
- Organic and inorganic wastes contaminated with low-assay uranium, and
- Direct burn materials (highly reactive or non-compatible liquids).

According to the analytical results for the 1988 trial burn, the TSCA incinerator operates with a PCB destruction and removal efficiency (DRE) of 99.99997 percent (Engineering Science, 1988). The trial burn conducted in 1988 used feed materials consisting of PCB-contaminated soil, capacitor oils, and shredded capacitors. For the six tests performed in the trial burns, the PCB feed rates ranged from 45,227,123 mg PCB/hr to 203,513,438 mg PCB/hr. The resulting PCB output from the incinerator measured in the stack releases ranged from 2 mg PCB/hr to 132 mg PCB/hr for the six test runs (Engineering Science, 1988). It should be noted that although samples of the ash sump solids were collected and found to contain PCBs, these results were not used to calculated the DRE for PCBs.

Based on the results of the trial burn, Engineering Sciences (1988) estimated the release rate of PCBs to be approximately 31 mg/hr (8.6 x 10-6 g/s). The basis of this release rate was a PCB feed rate of approximately 1.4 x 108 mg/hr (7200 lbs/day). However, the reported daily PCB incineration rates for the years 1992, 1993, and 1994 were much lower; 36, 84, and 183 lbs/day, respectively. As part of the 1988 trial burn, Engineering Science (1988) monitored for the presence of chlorinated dioxins and furans in the stack effluent. The testing performed during the trial burn indicated that no reportable quantities of chlorinated dioxins or furans attributable to the incinerator were measured. Moreover, data collected from the incinerator during 1987 showed that dioxins and furans were not detected in the stack effluent (1987 Dioxin Results). Detection limits were typically lower than the nanogram/dscm range.

To simplify the dispersion modeling task, the project team assumed that all non-detected dioxin and furan isomers were present at one-half their respective detection limits. Based on this assumption and the ITEF scheme, the project team estimated a TCDD-TEQ release rate of approximately 1.1 x 10-8g/s. Again, it should be noted that this release rate was conservatively based on a 7200 lb/day trial burn feed rate of PCBs.

Ambient air concentrations of PCBs and PCDF were estimated at receptors located outside of the ORR using the SCREEN3 air dispersion model. Because the releases from the incinerator were from a stack, the project team modeled the incinerator releases as a point source. Input parameters used in the model are described below.

The gaseous effluent from the incinerator is directed to the air pollution control train and subsequently to the stack. Physical features of the stack and the dynamics of the effluent gasses affect the rise of the plume from the stack. The physical stack and effluent gas parameters reported in the Engineering Science (1988) report are as follows:

- approximate stack base elevation - 230 meters mean sea level (MSL),
- height - 26.5 m,
- interior stack diameter - 1.4 m,
- stack gas exit velocity - 7.2 m, and
- exit gas temperature - 348.6K.

The project team modeled the releases using the USEPA's screening level air dispersion model, SCREEN3 (95250) and inputting the physical stack parameters summarized above.

Calculation of Final Air EPC

For the purpose of this analysis, the project team evaluated releases from four sources at the ORR: the Y-12 Burn Pit, Y-12 Burn Tank, the K-25 Burn Pit, and the TSCA Incinerator. As discussed above, both the Y-12 and K-25 sites are bounded by ridges that tend to direct air releases up and down the valleys. Because of the complexities of modeling such movements, the project team decided to model the nearest receptors without consideration of terrain. This overestimates the actual air concentrations for the receptors that do not reside in the same valley as the source. Table C-1 presents the predicted long-term concentrations for both direct exposures (residential receptors) and for indirect exposures (farm families). The highest exposure predicted to occur for a residential receptor was 4.1 x 10-6 mg/m3 for PCBs and 3.6 x 10-9 mg/m3 for PCDF (TCDD equivalents). The highest concentrations estimated for the farm family were 3.3 x 10-6 mg/m3 for PCBs and 2.9 x 10-9mg/m3 for PCDF (TCDD equivalents).

Table C-1. Receptor Locations and Air Concentrations[1] for Direct and Indirect Exposure Pathways (mg/m^3)

	Direct Exposure (Nearest Residences)			Indirect Exposure (Nearest Farms)		
		Air Concentrations			Air Concentrations	
Sources	**Location**	**PCB**	**PCDF[2]**	**Location**	**PCBs**	**PCDF**
K-25						
K-25 Burn Tank	Hartland Estates	3.0×10^{-7}	2.6×10^{-10}	Union, TN	2.5×10^{-7}	2.2×10^{-10}
TSCA Incinerator	Hartland Estates	6.5×10^{-9}	8.3×10^{-12}	Union, TN	6.1×10^{-9}	1×10^{-5}
Y-12 (Bear Creek Disposal Area)						
Y-12 Burn Pit	Scarboro	3.7×10^{-6}	3.3×10^{-9}	EFPC Farms	3.1×10^{-6}	2.7×10^{-9}
Y-12 Burn Tank	Scarboro	4.1×10^{-6}	3.6×10^{-9}	EFPC Farms	3.3×10^{-6}	2.9×10^{-9}

[1] Determined assuming a flat terrain (no ridges)
[2] TCDD equivalents

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APPENDIX D

EXAMPLE EXPOSURE PARAMETER DISTRIBUTIONS

APPENDIX D: EXAMPLE EXPOSURE PARAMETER DISTRIBUTIONS

This Appendix briefly defines distributions and statistical terms used to describe distributions, and provides descriptions and visual representation of the distribution types used in the total uncertainty and two-dimensional analyses. A distribution can be defined as the collection of all values that a single variable or parameter, such as body weight, can take on in a population of individuals. A probability density function organizes the data in a distribution by the frequency with which any given value occurs in the collection of data. In a probability density function, the data are sorted from smallest to largest and the frequency with which each value occurs in the distribution is plotted. The resulting frequency plot or *histogram* reveals the *shape* of the distribution. Distributions can be of many shapes, including the familiar "bell curve" of a normal distribution. A variety of statistical terms are used to describe the shape of a distribution, as well as its *location* (i.e., where its numerical values lie on a numerical scale). Some of the more common terms are described below using an example of body weights.

A population of 100 people are given a body weight survey, and body weights ranging from 105 pounds up to 210 pounds are recorded. The *minimum* of this distribution would be 105 pounds and the *maximum* would be 210 pounds. The *mode* or *most likely* value of this distribution would be the weight reported by the highest number of people. Thus, if 10 people reported their weight to be 170 pounds, and no other single weight was reported by more than 10 people, then this value would represent the mode. The *median* or the *50th percentile* is the value that splits the population in two; for example, exactly half of the people would weigh less than the median of 160 pounds, and exactly half would weigh more. Other percentiles can be interpreted likewise; 95% of the population weighs less than the *95th percentile*, and 5% weighs more, while 25% weighs less than the *25th percentile*, and 75% weigh more.

Many distributions can be described by two important parameters which measure *central tendency* and *variability* or *spread*. Central tendency refers to that part of the distribution containing values representing most of the population. In the famous "bell curve" or normal distribution, the central tendency corresponds to the middle of the bell. Variability refers to how large the range of values is, or, intuitively, how widely spread the shape is. For example, a group of body weights ranging from 85 pounds to 350 pounds would be more variable than the previous example. The most common measures of central tendency and variability are the *mean* or *average value* and the *standard deviation*, respectively.

One final distinction that bears discussion is between *continuous* and *discrete* distributions. In a continuous distribution, there is an infinite number of possible values, because between any two values there are fractional values that can also be selected. In a discrete distribution, there is a finite number of values that the parameter can take. For example, distributions of car types would be discrete, because there cannot be five and one-half Jeep Cherokees.

Frequency plots (histograms) of the distribution types used in the total uncertainty and two-dimensional analyses are shown in Figures D-1 through D-5. Brief descriptions of the features of individual distributions are provided below.

A normal distribution is pictured in Figure D-1. The normal distribution, familiar to many as the “bell curve”, is entirely symmetrical and can take any value, positive or negative. In a perfect normal distribution, the mean and median values are identical.

A sample uniform distribution is pictured in Figure D-2. The uniform distribution is the most simple distribution type, and is described entirely by its minimum and maximum values. Values between the minimum and maximum all occur with equal frequency.

A sample triangular distribution is pictured in Figure D-3. The triangular distribution can be described by three parameters: the minimum, the mode or most likely value, and the maximum. This distribution is often used to approximate a distribution when sufficient actual data are not available.

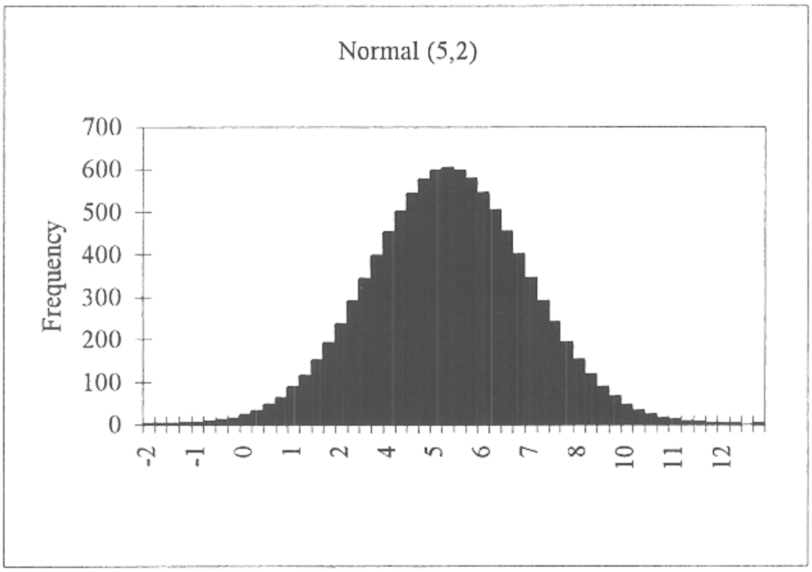
A sample lognormal distribution is pictured in Figure D-4. The lognormal distribution is generally described by the mean and standard deviation. Unique features of the lognormal distribution are the long “tail” representing values that are part of the dataset but that occur infrequently, and the lower bound at zero, indicating that all values in a lognormal distribution are positive. The lognormal gets its name from the fact that a distribution composed of the natural logarithms of individual points in a lognormal distribution will in fact be normally distributed (“bell” shaped).

A sample cumulative distribution is pictured in Figure D-5. A cumulative distribution is defined empirically when the available data suggest that “x-percent of the values are less than y”. For example, if we have a small sample of persons reporting body weight, there may not be enough information to define a distribution type, but there may be sufficient information to say that 10%

of the weights are less than 130 pounds, 50% of the weights are less than 170 pounds, 75% of the weights are less than 190 pounds, and 100% of the weights are less than or equal to 210 pounds. A cumulative distribution can be defined from this information, and typically appears as a series of bars, with values uniformly distributed between each pair of data points.

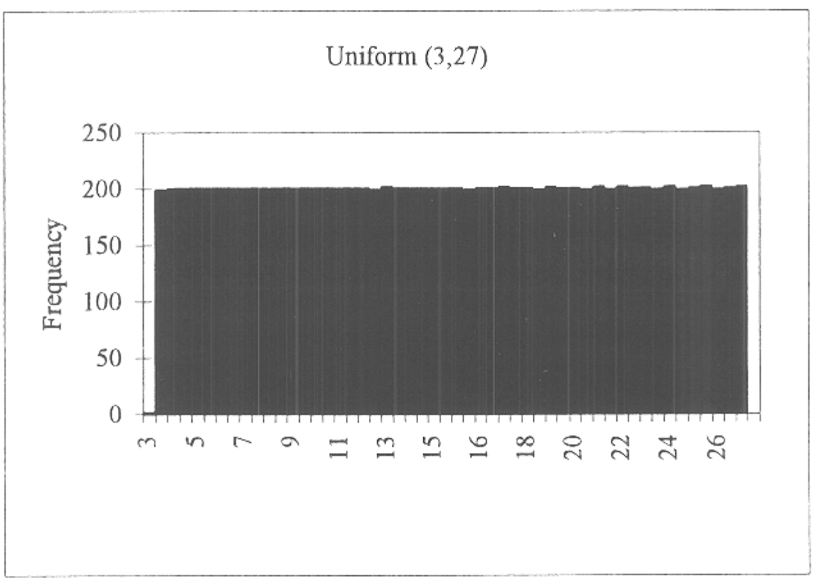
Distribution Type: Normal
Parameters: Mean = 5
Standard Deviation = 2

Figure D-1



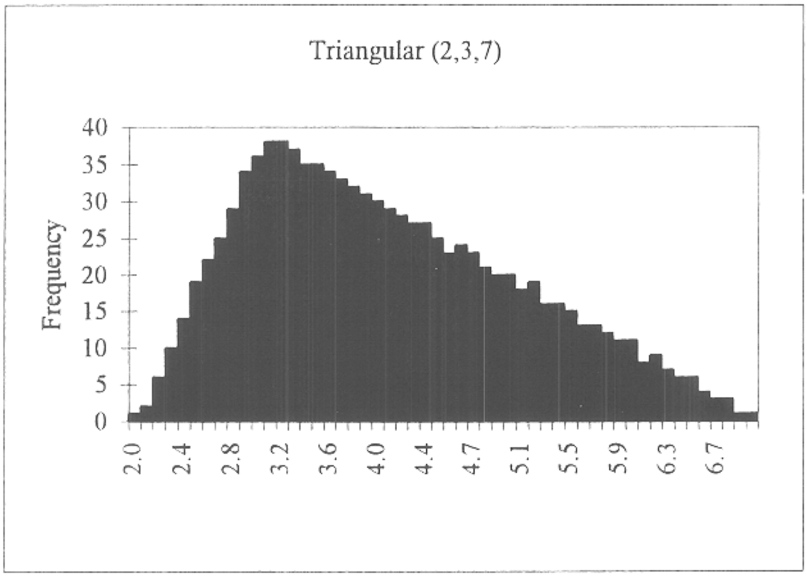
Distribution Type: Uniform
Parameters: Minimum = 3
Maximum = 27

Figure D-2



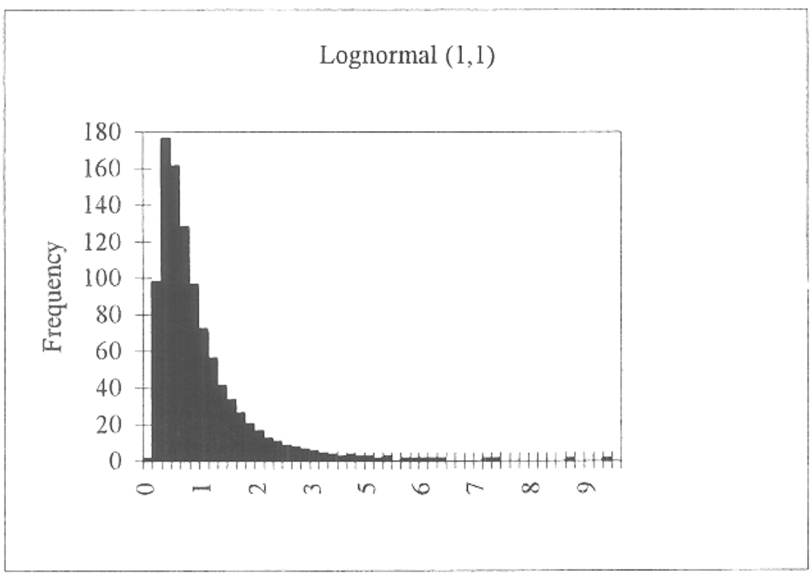
Distribution Type: Triangular
Parameters: Minimum = 2
Mode = 3
Maximum = 7

Figure D-3



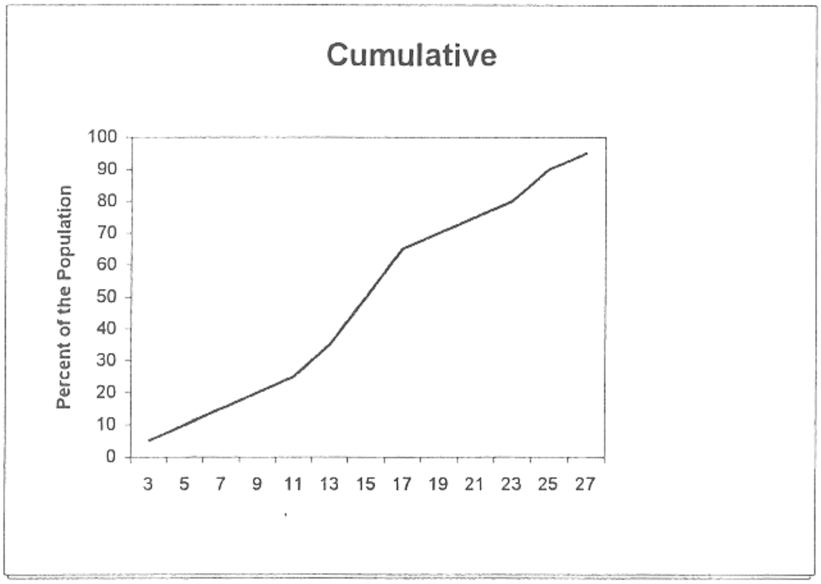
Distribution Type Lognormal
Parameters Mean = 1
Standard Deviation = 1

Figure D-4



Distribution Type: Cumulative
Parameters: Empirically Defined

Figure D-5



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