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# Quantitative Uncertainty Analysis for the Human Fish Ingestion Pathway: Select Radionuclide and Chemical Contaminants in the Clinch River/Watts Bar Reservoir System

A Thesis

Presented for the

**Master of Science**

Degree

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**Jana S. Hammonds**

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## Abstract

The goal of this study was to perform a quantitative uncertainty analysis on the excess health risk associated with the ingestion of select radionuclide and chemical contaminants as the result of human consumption of contaminated fish. A quantitative uncertainty analysis is an efficient method of ranking contaminants and pathways and of determining the most sensitive parameters to the overall estimate of uncertainty in the model result; thus, providing a powerful tool in decision making processes. Subjective probability distributions were determined for each parameter in the risk assessment equations. These distributions were defined by using subjective judgment after extensively reviewing the literature and in some cases, after eliciting expert opinion. The Monte Carlo approach used to propagate the subjective probability distributions for the model parameters was Latin Hypercube Sampling. Subjective confidence intervals were obtained for the potential detriment from the human ingestion of fish contaminated with Co-60, Sr-90, Cs-137, Pu-239, methyl-mercury (MeHg), chlordane, Aroclor-1254, and Aroclor-1260. The results indicate that the potential risk from the chemical contamination in the fish of the Clinch River/Watts Bar Reservoir system is greater than the potential risk from radionuclides. The primary contaminants for further study are Aroclor-1260, Aroclor-1254, MeHg, and chlordane. The results of the sensitivity analysis indicate that the overall estimate of uncertainty in the total risk estimates would be most efficiently reduced by taking local surveys to obtain better subjective probability distributions for parameters such as the number of fish-meals per day and the exposure duration instead of taking more fish samples.

# Table of Contents

# List of Tables

# List of Figures

# List of Acronyms

| | |
|---|---|
| DOE | Department of Energy |
| GSD | Geometric Standard Deviation |
| IAEA | International Atomic Energy Agency |
| ICRP | International Commission on Radiation Protection |
| LCL | Lower Confidence Limit |
| LHS | Latin Hypercube Sampling |
| LOAEL | Lowest Observed Adverse Effect Level |
| MeHg | Methyl-mercury |
| NCRP | National Council on Radiation Protection and Measurements |
| NOAEL | No Observed Adverse Effect Level |
| NRC/NAS | National Research Council of the National Academy of Sciences |
| PCB | Poly-chlorinated Biphenyl |
| PDF | Probability Distribution Function |
| ORNL | Oak Ridge National Laboratory |
| SRS | Simple Random Sampling |
| UCL | Upper Confidence Limit |
| UF | Uncertainty Factor |
| UNSCEAR | United Nations Scientific Committee on the Effects of Atomic Radiation |
| USEPA | United States Environmental Protection Agency |

## List of Parameters

| | |
|---|---|
| $AT$ | Averaging Time (days) |
| $BM$ | Body Mass (kg) |
| $C$ | Concentration of contaminant in fish (Bq/kg or mg/kg) |
| $DCF$ | Dose Conversion Factor (Sv/Bq) |
| $ED$ | Exposure Duration (years) |
| $EF$ | Exposure Frequency (days/year) |
| $F_m$ | Frequency of fish-meals eaten from fish caught in the Clinch River/Watts Bar Reservoir system (fish-meals/day) |
| $HI$ | Hazard Index (unitless) |
| $HQ$ | Hazard Quotient (unitless) |
| $I$ | Daily intake rate of fish from the Clinch River/Watts Bar Reservoir system (kg/day) |
| $LCL_{5\%}$ | 5th percentile value of the risk (lower confidence limit) (unitless) |
| $LR_{chem}$ | Excess Lifetime Cancer Risk from carcinogenic chemicals (unitless) |
| $LR_{rad}$ | Excess Lifetime Cancer Risk from radionuclides (unitless) |
| $M_w$ | Amount of fish eaten per fish-meal (kg/fish-meal) |
| $RCF$ | Risk Conversion Factor (risk/Sv) |
| $RfD$ | Reference dose for the chemical of interest (mg/kg-day) |
| $SF$ | Slope factor for the chemical of interest [per (mg/kg-day)] |
| $S_{g,R}$ | Geometric Standard Deviation of the result |
| $SI$ | Sensitivity Index (unitless) |
| $UCL_{95\%}$ | 95th percentile value of the risk (upper confidence limit) (unitless) |

| | |
|---|---|
| $UCL_{mean}$ | 95% upper confidence limit on the estimated arithmetic mean concentration (Bq/kg or mg/kg) |
| $UF_{mean}$ | Uncertainty factor for the estimated arithmetic mean concentration (unitless) |
| $X_{est.}$ | Estimated arithmetic mean |
| $X_{g,R}$ | Geometric mean of the result |
| $\sigma_R^2$ | Variance of the logarithms |
| $\mu_R$ | Sum of the means of logarithms |

# 1.0 Introduction

When hazardous substances are inadvertently or purposely introduced into the environment, a primary concern is the possible effect on the environment and on human health. For human health concerns, baseline risk assessments are therefore performed to quantify the potential detriment to exposed individuals given the absence of any institutional controls prior to determining the need for and type of remediation measures. The United States Environmental Protection Agency (USEPA), in its current Risk Assessment Guidance document for Superfund sites (USEPA, 1989), recommends that either a qualitative or quantitative uncertainty analysis accompany risk assessments so that the confidence in the results can be expressed.

Most uncertainty analyses to date have been restricted to a qualitative statement of confidence in the result. Uncertainty less than one order of magnitude is "low", uncertainty between one and two orders of magnitude is "moderate", and uncertainty more than two orders of magnitude is "high" (USEPA, 1989). However, unless the person doing the analysis is an expert on the performance of the model and the comparison of model results with data from conditions similar to those in which the model is to be used, these "qualitative" statements are difficult to substantiate.

A quantitative uncertainty analysis, on the other hand, provides subjective confidence intervals for the risk estimates so that the contaminants and pathways can be ranked. Thus, a quantitative approach provides a major tool for use in decision-making processes. The present report focuses on a quantitative uncertainty analysis for the determination of the potential detriment associated

with the ingestion of selected radionuclides and chemicals as a result of human consumption of contaminated fish.

### 1.1 Background on USEPA Methods for Risk Assessment

For a baseline risk assessment, the USEPA (1989) uses the following equation for determining the lifetime risk of excess cancer from ingesting radionuclides:

$$LR_{rad} = C \cdot I \cdot ED \cdot EF \cdot DCF \cdot RCF \qquad (1.1)$$

where,

$LR_{rad}$ = excess lifetime cancer risk from radionuclides (unitless),

$C$ = concentration in the contaminated medium (Bq/kg),

$I$ = estimated intake of the contaminated medium (kg/day),

$ED$ = exposure duration (30 yrs),

$EF$ = exposure frequency (350 days/yr),

$DCF$ = dose conversion factor for the radionuclide of interest (Sv/Bq),

and $RCF$ = risk conversion factor (0.073/Sv) (ICRP, 1991; NCRP, 1993).

The generic equations used in USEPA methods for determining the risk from chemicals are different for non-carcinogens and carcinogens. For non-carcinogens a quantity called the Hazard Quotient ($HQ$) is calculated for each chemical using the following formula.

$$HQ = \frac{C \cdot I \cdot ED \cdot EF}{BM \cdot AT \cdot RfD} \tag{1.2}$$

where,

$HQ$ = hazard quotient (unitless),

$C$ = concentration in the contaminated medium (mg/kg),

$I$ = estimated intake rate of the contaminant (kg/day),

$ED$ = exposure duration (years),

$EF$ = exposure frequency (350 days/year),

$BM$ = body mass (70 kg),

$AT$ = averaging time (days),

and $RfD$ = reference dose for the chemical of interest (mg/kg-day).

If the $HQ$ is below 1, then it is highly unlikely that chronic exposure to the chemical would lead to a health effect. If the $HQ$ is greater than unity, remediation of the area may be warranted. The $HQ$s for various chemicals are summed for each exposure pathway to obtain a Hazard Index ($HI$) for a given area (USEPA, 1989).

For carcinogenic chemicals, a lifetime cancer risk is calculated from the following formula.

$$LR_{chem} = \frac{C \cdot I \cdot ED \cdot EF \cdot SF}{BM \cdot AT} \tag{1.3}$$

where,

$LR_{chem}$ = excess lifetime cancer risk from chemicals (unitless),

$C$ = concentration in the contaminated medium (mg/kg),

$I$ = estimated intake of the contaminated medium (kg/day),

$ED$ = exposure duration (30 yrs),

$EF$ = exposure frequency (350 days/year),

$SF$ = slope factor (or cancer potency factor) for the chemical of interest [per (mg/kg-day)],

$BM$ = body mass (70 kg),

and $AT$ = averaging time (25,550 days).

A primary difference in evaluating risk between non-carcinogens and carcinogens is the threshold and no threshold assumption, respectively. For non-carcinogens, toxic effects occur after an exposure above a certain threshold value has occurred. It is assumed that any exposure below this threshold has no detrimental effect to the organism. For carcinogens, however, it is assumed that there is no threshold below which a risk is assumed to be zero. A carcinogen is defined as a substance for which any exposure results in a potential excess cancer risk.

These equations yield point estimates of the risk which obscure the inherent uncertainty in the calculations (see Example 1 in Appendix A). If decisions concerning remediation of the contaminated area are based on point estimates, and if the true risk is either grossly overestimated or is associated with high uncertainty, the result may be inappropriate remedial efforts and/or

misallocation of limited resources (see Example 2 in Appendix A). Therefore, quantitative uncertainty analysis should be performed to quantify the degree of confidence in the estimate of risk. This requires analysis of the uncertainty associated with each parameter in the USEPA risk equations; the bounds of uncertainty, expressed as subjective probability distributions, must be estimated for each parameter. The term subjective probability distribution is used because subjective judgment is used in the absence of any site specific data.

## 1.2 Problem Statement

Over the years, various contaminants have been released into the Clinch River/Watts Bar Reservoir system by Department of Energy (DOE) operations at the Oak Ridge National Laboratory (ORNL), the Y-12 Plant, and the K-25 Site. A thorough risk assessment accompanied by an uncertainty analysis will be performed for various contaminants in order to determine the potential risk to the surrounding population. The first step of a risk assessment is to limit the scope of the problem by using one or more screening techniques, which permits the identification of a subset of contaminants and exposure pathways of greatest potential concern. In most cases, a screening analysis results in a smaller number of contaminants and exposure pathways for which a formal quantitative uncertainty analysis is necessary.

The screening calculations for the contaminants in the Clinch River/Watts Bar Reservoir system were completed in Phase 1 of the remedial investigation (Cook et al., 1992). The next step in the risk assessment will be partially fulfilled in the present research. A quantitative uncertainty analysis for the fish-human pathway will be performed for select radionuclides and chemicals as determined

in the Phase 1 screening to be of potential concern. The radionuclide of greatest potential concern for the fish ingestion pathway was cesium-137. Cobalt-60, strontium-90, and plutonium-239 are also of interest due to their substantial historical use at the DOE facilities. In addition to these radionuclides, various chemicals were identified as contaminants of concern. These chemicals are polychlorinated biphenyls (PCBs, i.e., Aroclor-1254 and Aroclor-1260), chlordane, and methyl-mercury (MeHg).

The goal for this project, therefore, is a quantitative uncertainty analysis of the risk to human health (specifically, to the maximally exposed individual) resulting from the ingestion of fish from the Clinch River/Watts Bar Reservoir system. Contaminants evaluated are cobalt-60, strontium-90, cesium-137, plutonium-239, Aroclor-1254, Aroclor-1260, chlordane, and MeHg (the total mercury measured in fish). The results of this uncertainty analysis will provide valuable information in determining where further study is required and where financial resources should be allocated for the fish ingestion pathway.

### 1.3 Report Structure

The methods of uncertainty analysis are presented in the following section along with a general set of guidelines that may be used to perform an uncertainty analysis. The subjective probability distributions for each of the model parameters are defined in Section 3.0. The results of the quantitative uncertainty analysis and the sensitivity analysis are provided in Section 4.0. These results are briefly summarized and compared to background levels of contamination in Section 5.0. A brief conclusion and future recommendations for further study are also provided.

# 2.0 Methods for Uncertainty Analysis

If all of the various exposure pathways for every hazardous substance found at a contaminated site were considered, the risk assessment would be an extremely complex task. An uncertainty analysis of every parameter involved in an exposure scenario is often impractical if not impossible. Therefore, the first step in any risk assessment should be to narrow the scope of the problem.

The objective(s) of the assessment should be clearly defined, and a screening procedure should be used to identify the contaminants and exposure pathways which require more detailed analysis (Hoffman and Gardner, 1983; NCRP, 1989; Hoffman et al., 1991). Screening can be considered a first-step approach to uncertainty analysis. Once the list of contaminants and pathways has been narrowed, uncertainty analysis can be used to assess the extent of confidence in the estimate of health risk.

## 2.1 Model Uncertainty

Model validation is perhaps the best method for assessing model uncertainties (Hoffman et al., 1984). This is accomplished by comparing model predictions to numerous independent data sets. For human health risk assessment equations, model validation is not an easy task. Various data sets of human health effects from exposure at low doses over long time periods would be necessary. This type of study is usually precluded because the incipient incidence of disease will obscure effects occurring at low doses. Therefore, other methods of uncertainty analysis are considered.

In this study, it is assumed that the uncertainty in the estimate of risk can be calculated from an estimate of uncertainty in each of the parameters used in the risk assessment equations. This approach is referred to as a "parameter uncertainty analysis" (IAEA, 1989). The technique of parameter uncertainty analysis provides a quantitative way to estimate the uncertainty in the model result assuming the structure of the model is correct. *Any additional uncertainty due to model structure should be represented either by alternative equations or by the addition of parameters to the risk assessment model.* In this study, the bounds of uncertainty for each model parameter were defined so that the true but unknown value will be within two limits. No additional parameters are used to reflect uncertainty in the model structure. Therefore, for this analysis, it is assumed that the correct values for the model parameters will produce the correct value for the model result.

## 2.2 General Approach to Parameter Uncertainty Analysis

The following general guidelines are given for a parameter uncertainty analysis (IAEA, 1989):

1. List all uncertain parameters (include additional parameters to represent uncertainty in model structure).
2. Specify the maximum conceivable range of each parameter. (Section 3.0)
3. Specify a subjective probability distribution for values occurring within this range. ( Section 3.0)
4. Account for dependencies and/or correlations between parameters.

5. Using either analytical or numerical procedures, propagate the subjective probability distribution functions (pdf) of the uncertain model parameters to generate a subjective probability distribution for the model result. (Section 4.0)

6. Derive quantitative statements of uncertainty in terms of a subjective confidence interval for the true but unknown value representing the prediction endpoint, e.g., excess cancer risk or Hazard Index. (Section 4.0)

7. Rank the parameters by the significance of their contribution to uncertainty in the model prediction. (Section 4.0)

8. Present and interpret the results of the analysis. (Section 4.0)

These guidelines are generally used in the present research. The uncertain parameters for this assessment include the concentrations of the contaminants in fish ($C$), the daily intake of fish ($I$), the body mass ($BM$, chemicals only), the exposure duration ($ED$), the exposure frequency ($EF$), the averaging time ($AT$, chemicals only), the dose conversion factors ($DCF$s, radionuclides only), the risk conversion factor ($RCF$, radionuclides only), the slope factors ($SF$s, chemicals only), and the reference doses ($RfD$s, chemicals only).

Correlation effects were not considered in this study. Throughout the literature review, no substantial evidence of correlations between model parameters was determined. One might suggest that a correlation exists between body mass and the amount of fish ingested, but as Example 3 in Appendix A shows, accounting for this correlation (if it exists) has little to no effect on the model result due to the relative insensitivity of the uncertainty in the body mass parameter to the uncertainty in the risk estimate.

Maximum conceivable ranges and subjective probability distributions (steps 2 and 3) were determined through examination of available data from the open literature and from discussions with various experts and are presented in

Section 3.0 of this report. Monte Carlo simulation [specifically, Latin Hypercube Sampling (LHS)] was used to propagate the uncertainty and determine subjective confidence limits on the end point, and the results are presented in Section 4.0. For more information on the various methods of performing a quantitative uncertainty analysis, see Hoffman and Hammonds (1992), IAEA (1989), and Morgan and Henrion (1990).

### 2.3 Analytical Methods for Uncertainty Analysis

For relatively simple equations, uncertainty analysis can be performed using analytical methods (algebraic equations) for statistical error propagation. A type of analytical approach that is frequently used for uncertainty analysis is variance propagation (IAEA, 1989).

To best demonstrate an example of variance propagation, a simple additive model is desired. In an additive model, the mean of the result is equal to the sum of the means of the parameters, and the variance of the result, assuming statistical independence among the parameters, is equal to the sum of the variances of the parameters (Hoffman and Gardner, 1983; IAEA, 1989).

$$\mu_R = \sum_{i=1}^{p} \mu_i \tag{2.1}$$

$$\sigma_R^2 = \sum_{i=1}^{p} \sigma_i^2 \tag{2.2}$$

where p = the number of parameters in the model.

However, the basic form of USEPA risk assessment models (Equations 1.1, 1.2, and 1.3) is a multiplicative chain of parameters for each contaminant and exposure pathway. Multiplicative models can be reduced to additive form by logarithmically transforming the variables. For example:

$$Y = a\ b\ c \tag{2.3}$$

$$\ln(Y) = \ln(a) + \ln(b) + \ln(c). \tag{2.4}$$

Therefore, the distribution of Y will tend to be approximately lognormal even when the parameters are assigned distribution shapes other than lognormal (Gardner, 1988; O'Neill et al., 1981). For multiplicative calculations, one can estimate the median value (or geometric mean) for the risk by simply summing the means of the logarithms for the various parameters and exponentiating the sum. This relationship is presented below in equation form.

$$X_{g,R} = e^{\mu_R} \tag{2.5}$$

where $X_{g,R}$ = the geometric mean of the result

and $\mu_R$ = the resulting sum of the means of logarithms.

In addition, the geometric standard deviation of the result is found by taking the square root of the sum of the variances obtained for the logarithms of the parameters and exponentiating (Hoffman and Gardner, 1983; IAEA, 1989). This formula is represented in the equation given below.

$$S_{g,R} = e^{\sqrt{\sigma_R^2}} \tag{2.6}$$

where $S_{g,R}$ = the geometric standard deviation of the result

and $\sigma_R^2$ = the variance of the logarithms.

Confidence limits for the true but unknown value are determined by multiplying the median value by the square (or some other power) of the exponentiated standard deviation of logarithms to obtain an upper bound and then dividing the median by the square (or some other power) of the exponentiated standard deviation of the logarithms to obtain the lower bound. The exponential standard deviation of logarithms is often referred to as the geometric standard deviation (GSD). The use of the square of the GSD will lead to a 95% confidence interval, assuming that the distribution of the model prediction will be lognormal. Taking the GSD to a power of 1.65 will lead to a 90% confidence interval for a lognormal distribution. For example, the following equations would lead to a 90% confidence interval:

$$X_{95}^{HQ} = X_{g,HQ} \cdot S_{g,HQ}^{1.65} \tag{2.7}$$

$$X_{5}^{HQ} = \frac{X_{g,HQ}}{S_{g,HQ}^{1.65}} \tag{2.8}$$

The formula one uses to estimate the mean and variance of logarithms for each uncertain parameter depends on the type of subjective probability distribution chosen to represent the uncertain parameter. Formulas for the mean and variance of logarithms of log-normal, log-uniform, and log-triangular distributions are provided in Appendix B.

As one can see, variance propagation is a straight-forward process for simple additive and multiplicative models in which the parameters are statistically independent. For more complex calculations, variance propagation techniques are more difficult to derive analytically, and, in some cases, their derivation may not be practical. A demonstration of this technique is given in Example 4 of Appendix A.

## 2.4 Numerical Methods for Uncertainty Analysis

To overcome problems encountered with variance propagation equations, various numerical methods are useful for performing an uncertainty analysis with the aid of a computer. Two such numerical techniques are Monte Carlo simulation (Hoffman and Gardner, 1983; IAEA, 1989; Rubinstein, 1981) and deterministic uncertainty analysis (Worley, 1987). Monte Carlo simulation usually employs two random sampling processes: Simple Random Sampling (SRS) and Latin Hypercube Sampling (LHS) (McKay et al., 1979; Iman and Conover, 1980; 1982; Iman and Shortencarier, 1984; IAEA, 1989).

In SRS, a random number is obtained from each distribution specified for an uncertain parameter, and an estimate of the desired endpoint is calculated. This process is repeated for a specified number of samples or iterations. The result is a subjective probability distribution of the model endpoint. SRS, however, is less efficient than its counterpart, LHS, when the sample size is less than a few thousand.

In LHS, the distribution for each parameter is divided into sections of equal probability. The number of sections depends on the number of samples or iterations to be made in the Monte Carlo simulation. During the run, the random

numbers are selected by chance within a section, but only one random number is chosen from each section. Once a random number has been selected from a section, that section is excluded from the rest of the analysis. The distributions are thus represented more efficiently than with SRS, and it takes less sampling effort to reach a stable mean and variance of the prediction endpoint (IAEA, 1989).

Monte Carlo analysis may be performed in many ways. One may write a numerical code or utilize one of several currently available software packages. Several Monte Carlo simulation programs that are available are listed below with their references.

| | |
|---|---|
| MOUSE | Klee (1986) |
| TAM3 | Kanyar and Nielsen (1989) and Gardner (1988) |
| PRISM | Gardner and Trabalka (1985) and Gardner et al. (1983) |
| Crystal Ball | Decisioneering (1992) |
| @RISK | Palisade Corporation (1991) |
| ORMONTE | Williams and Hudson II (1989) |

Once familiarization with the Monte Carlo simulation software package is accomplished, this technique becomes extremely quick. Even though a risk analysis becomes more complicated, the Monte Carlo technique does not. The fact that the variance propagation techniques can become complicated and time-consuming for more involved risk analyses is one reason that Monte Carlo calculations are more useful. Setting up simulations to run on the computer is much more efficient and accurate than performing hand calculations. Demonstrations of this technique for more complicated risk analysis situations are presented in Examples 5 and 6 in Appendix A.

## 3.0 Quantification of Uncertain Model Parameters

The subjective probability distributions used for each of the uncertain parameters in Equation 1.1, 1.2, and 1.3 reflect the degree of belief that the true, but unknown, value for each parameter does not exceed a specified value for that parameter. For equations of the general form of Equation 1.1, 1.2, and 1.3, the exact shape of the probability distribution assigned to a given parameter will have minimal effect on the mean, variance, and general shape of the distribution of the model prediction (Gardner, 1988; O'Neill et al., 1981), if the mean and variance of the probability distribution are held constant. This will be true if no single parameter dominates the overall uncertainty in the model prediction and if the extent of interdependence among the model parameters is small.

Where data are limited but uncertainty is relatively low (less than a factor of 10), a range is used to specify a uniform distribution. If there is knowledge about a most likely value or midpoint, in addition to a range, a triangular distribution is assigned. When the range of uncertainty exceeds a factor of 10, a log-uniform or log-triangular distribution is often used. The use of normal, log-normal, or empirical distributions is usually based on the available data. Piece-wise uniform or log-uniform distributions can also be used and are especially common when eliciting subjective judgment from a group of experts (IAEA, 1989). Other distribution types suitable for Monte Carlo analysis include the gamma, beta, Poisson, Weibull, and a variety of discrete distributions (Decisioneering, 1992; Palisade Corp., 1991).

In this section, the first parameter discussed is the contaminant concentration in fish, followed by discussion of exposure factors that are independent within a class (radionuclides, non-carcinogenic chemicals, and

carcinogenic chemicals) of contaminants which include the daily intake, the exposure frequency, the exposure duration, the body mass, and the averaging time. The last two topics discussed in this section are the uncertainties involved with radionuclide specific parameters, the dose conversion factors and the risk conversion factor, and with chemical specific parameters, the slope factors and the reference doses.

### 3.1 Concentration in fish (*C*), Bq/kg

The Phase 1 fish data for the remedial investigation of the Clinch River/Watts Bar Reservoir system are summarized in Cook et al. (1992). For this uncertainty analysis, only those concentrations where at least one measurement per reach (a section of the river or reservoir) was above detection limits are considered. Both normal and log-normal significance tests were performed (Cook et al., 1992). The results of these tests indicated that a log-normal distribution gave the best representation of the observed data.

The uncertainty bounds on the estimated arithmetic mean concentration were quantified for each selected contaminant in each reach. A distribution was obtained that represents the subjective uncertainty of the estimated arithmetic mean concentration. The following equation was used to estimate the uncertainty factor for the arithmetic mean concentrations.

$$UF_{mean} = \frac{UCL_{mean}}{\overline{X}_{est.}} \tag{3.1}$$

where $UF_{mean}$ = the uncertainty factor for the arithmetic mean,

$UCL_{mean}$ = the 95% upper confidence limit on the mean,

and $\bar{X}_{est.}$ = the arithmetic mean.

### 3.1.1 Radionuclide Concentrations

The values for the arithmetic mean concentration and the UCL on the mean were obtained from the Clinch River/Watts Bar Reservoir system database (Brandt, 1993) and are summarized for radionuclides in Table 3.1.

The methodology is the same for estimating the uncertainty factor on each arithmetic mean concentration; therefore, the calculation will be demonstrated in detail only for one radionuclide. The calculation for Cs-137 in reach 1 is shown below, and a summary of all calculations is presented in Table 3.1. All values for the UFs were rounded to the nearest tenth.

$$UF_{mean} = \frac{1.54}{1.11} = 1.4$$

*A uniform distribution was used to represent the uncertainty of the estimated arithmetic mean concentration except for three Co-60 concentrations as indicated in Table 3.1.* Due to their large degree of uncertainty, a log-uniform distribution of the true but unknown mean was used for these three concentrations. The maximum value for the distribution was obtained by multiplying the estimated arithmetic mean by the uncertainty factor, and the minimum value for the distribution was obtained by dividing the estimated arithmetic mean by the uncertainty factor. These values are also presented in Table 3.1.

Table 3.1. Summary statistics and estimates of uncertainty for radionuclide contaminant concentrations in fish where at least one measurement was above the detection limits.

| Reach | Radionuclide | Estimated Arithmetic Mean (Bq/kg) | $UCL_{mean}$ (Bq/kg) | $UF_{mean}$ | Maximum (Bq/kg) | Minimum (Bq/kg) |
|---|---|---|---|---|---|---|
| 1 | Cs-137 | 1.1E+00 | 1.5E+00 | 1.4 | 1.6E+00 | 7.9E-01 |
| | Co-60[a] | 5.0E-01 | 2.4E+00 | 4.8 | 2.4E+00 | 1.0E-01 |
| | Sr-90[b,c] | 3.7E-02 | 3.7E-02 | -- | -- | -- |
| 2 | Cs-137 | 1.4E+01 | 2.2E+01 | 1.6 | 2.3E+01 | 8.8E+00 |
| | Sr-90[b] | 1.7E-01 | 2.9E-01 | 1.7 | 2.9E-01 | 1.0E-01 |
| 3 | Cs-137 | 2.3E+00 | 3.0E+00 | 1.3 | 3.0E+00 | 1.8E+00 |
| | Sr-90[b] | 1.1E-01 | 1.5E-01 | 1.3 | 1.5E-01 | 8.7E-02 |
| | Co-60[a] | 4.7E-01 | 2.9E+00 | 6.1 | 2.9E+00 | 7.7E-02 |
| 4 | Cs-137 | 3.9E+00 | 4.7E+00 | 1.2 | 4.6E+00 | 3.2E+00 |
| | Sr-90[b] | 9.5E-02 | 1.3E-01 | 1.4 | 1.3E-01 | 6.8E-02 |
| 5 | Cs-137 | 1.9E+00 | 2.6E+00 | 1.4 | 2.7E+00 | 1.4E+00 |
| | Sr-90[b] | 8.9E-02 | 1.2E-01 | 1.4 | 1.3E-01 | 6.4E-02 |
| 10 | Cs-137 | 1.1E+00 | 1.6E+00 | 1.5 | 1.6E+00 | 7.2E-01 |
| | Sr-90[b] | 6.3E-02 | 9.4E-02 | 1.5 | 9.4E-02 | 4.2E-02 |
| | Co-60[a] | 1.8E-01 | 8.0E-01 | 4.5 | 8.0E-01 | 3.9E-02 |
| 13 | Cs-137 | 2.1E+00 | 3.5E+00 | 1.7 | 3.5E+00 | 1.2E+00 |
| | Co-60 | 1.9E+00 | 2.7E+00 | 1.4 | 2.6E+00 | 1.3E+00 |
| | Sr-90[b] | 7.7E-02 | 1.9E-01 | 2.4 | 1.9E-01 | 3.2E-02 |

a A log-uniform distribution was used for these concentrations instead of a uniform distribution due to the large uncertainty range for these concentrations.

b These values are the concentrations in the bones of the fish.

c The arithmetic mean concentration and the UCL for this radionuclide were given as the same value; therefore, this radionuclide was not considered in the uncertainty analysis.

The values for the $UF_{mean}$ in Table 3.1 are around 1.4 for most concentrations. The three exceptions are for Co-60 concentrations where only one out of more than ten measurements was above the limit of detection. This implies that the uncertainties among the fish concentrations are relatively consistent if measurable concentrations are obtained.

The excess lifetime risk of cancer from ingestion of fish contaminated with plutonium was not considered in Phase I of the Clinch River/Watts Bar Reservoir remedial investigation and measured concentrations were not available (Cook et al., 1992). An uncertainty analysis for plutonium-239 in fish was included in the present research. The value for the concentration of plutonium in fish of 1 Bq/kg was held constant throughout the uncertainty analysis to give a preliminary estimate of uncertainty in the risk estimate in terms of a unit concentration.

### 3.1.2 Chemical Concentrations

The selected chemical contaminants for this uncertainty analysis are MeHg, chlordane, Aroclor-1254, and Aroclor-1260. Of these chemicals, MeHg is a non-carcinogen, chlordane is both a non-carcinogen and a carcinogen, and Aroclor-1254 and -1260 are carcinogens. A non-carcinogen is defined as a substance for which a toxic reaction results from exceeding a threshold level of exposure. A carcinogen is a substance that is thought to cause cancer at any level of exposure. Carcinogens are classified according to the evidence of their carcinogenicity. Radionuclides are classified as "A" which denotes there is ample evidence derived from human epidemiological data. PCBs and chlordane are

classified as "B2" carcinogens which implies no information from humans and all data comes from animals.

#### 3.1.2.1 Non-Carcinogens

The values for the estimated arithmetic mean concentration and the UCL on the mean were obtained from the Clinch River/Watts Bar Reservoir system database (Brandt, 1993) and are summarized for MeHg and chlordane in Table 3.2. The $UF_{mean}$s, maximum concentrations, and minimum concentrations were calculated as described in the previous section and are also presented in Table 3.2.

As with the radionuclide contaminants, *a uniform distribution was used to represent the uncertainty of the arithmetic mean concentrations except for the concentrations with UF values greater than 2.5 as indicated in Table* 3.2. Due to their large degree of uncertainty in the estimate of the mean, a log-uniform distribution was used for these two concentrations of chlordane. The values for the $UF_{mean}$ in Table 3.2 are fairly consistent. This indicates that the MeHg contamination is reasonably easy to measure with relatively consistent results.

#### 3.1.2.2 Carcinogens

The estimated arithmetic mean concentrations and the UCLs on the mean for Aroclor-1254, Aroclor-1260, and chlordane were also obtained from the Clinch River/Watts Bar Reservoir system database (Brandt, 1993) and are summarized in Table 3.3. The $UF_{mean}$s, maximum concentrations, and minimum

Table 3.2. Summary statistics and estimates of uncertainty for select non-carcinogenic contaminant concentrations in fish where at least one measurement was above the detection limits.

| Reach | Contaminant | Estimated Arithmetic Mean (mg/kg) | $UCL_{mean}$ (mg/kg) | $UF_{mean}$ | Maximum (mg/kg) | Minimum (mg/kg) |
|---|---|---|---|---|---|---|
| 1 | MeHg | 6.4E-02 | 8.2E-02 | 1.3 | 8.3E-02 | 4.9E-02 |
| | chlordane[a] | 4.0E-02 | 1.1E-01 | 2.7 | 1.1E-01 | 1.5E-02 |
| 2 | MeHg | 8.3E-02 | 1.4E-01 | 1.6 | 1.3E-01 | 5.2E-02 |
| | chlordane[a] | 2.8E-02 | 8.2E-02 | 2.9 | 8.0E-02 | 9.5E-03 |
| 3 | MeHg | 5.0E-01 | 6.2E-01 | 1.2 | 6.0E-01 | 4.2E-01 |
| | chlordane | 5.4E-02 | 7.5E-02 | 1.4 | 7.6E-02 | 3.9E-02 |
| 4 | MeHg | 2.7E-01 | 3.9E-01 | 1.5 | 4.0E-01 | 1.8E-01 |
| | chlordane | 5.0E-02 | 9.7E-02 | 1.9 | 9.5E-02 | 2.6E-02 |
| 5 | MeHg | 8.1E-02 | 1.1E-01 | 1.3 | 1.1E-01 | 6.2E-02 |
| | chlordane | 2.1E-02 | 4.6E-02 | 2.1 | 4.5E-02 | 1.0E-02 |
| 10 | MeHg | 2.1E-01 | 2.8E-01 | 1.3 | 2.8E-01 | 1.6E-01 |
| | chlordane | 9.7E-03 | 1.6E-02 | 1.6 | 1.6E-02 | 6.0E-03 |
| 13 | MeHg | 1.7E-01 | 3.7E-01 | 2.2 | 3.6E-01 | 7.5E-02 |
| | chlordane | 6.5E-02 | 1.2E-01 | 1.8 | 1.2E-01 | 3.6E-02 |

[a] A log-uniform distribution was used for these concentrations instead of a uniform distribution due to the large uncertainty range for these concentrations.

Table 3.3. Summary statistics and estimates of uncertainty for select carcinogenic chemical contaminant concentrations in fish where at least one measurement was above the detection limits.

| Reach | Contaminant | Estimated Arithmetic Mean (mg/kg) | $UCL_{mean}$ (mg/kg) | $UF_{mean}$ | Maximum (mg/kg) | Minimum (mg/kg) |
|---|---|---|---|---|---|---|
| 1 | Aroclor-1254[a] | 7.7E-02 | 2.0E-01 | 2.6 | 2.0E-01 | 2.9E-02 |
| | Aroclor-1260 | 4.4E-01 | 9.0E-01 | 2.0 | 8.8E-01 | 2.2E-01 |
| | chlordane[a] | 4.0E-02 | 1.1E-01 | 2.7 | 1.1E-01 | 1.5E-02 |
| 2 | Aroclor-1254[a] | 2.5E-01 | 7.5E-01 | 3.1 | 7.6E-01 | 7.9E-02 |
| | Aroclor-1260 | 8.8E-01 | 1.4E+00 | 1.6 | 1.4E+00 | 5.5E-01 |
| | chlordane[a] | 2.8E-02 | 8.2E-02 | 2.9 | 8.0E-02 | 9.5E-03 |
| 3 | Aroclor-1254 | 1.3E-01 | 1.9E-01 | 1.4 | 1.9E-01 | 9.6E-02 |
| | Aroclor-1260 | 5.6E-01 | 7.7E-01 | 1.4 | 7.8E-01 | 4.0E-01 |
| | chlordane | 5.4E-02 | 7.5E-02 | 1.4 | 7.6E-02 | 3.9E-02 |
| 4 | Aroclor-1254[a] | 1.4E-01 | 3.8E-01 | 2.7 | 3.8E-01 | 5.2E-02 |
| | Aroclor-1260[a] | 1.1E+00 | 2.8E+00 | 2.6 | 2.8E+00 | 4.2E-01 |
| | chlordane | 5.0E-02 | 9.7E-02 | 1.9 | 9.5E-02 | 2.6E-02 |
| 5 | Aroclor-1254[a] | 1.5E-01 | 4.4E-01 | 2.9 | 4.5E-01 | 5.3E-02 |
| | Aroclor-1260 | 5.3E-01 | 8.9E-01 | 1.7 | 9.0E-01 | 3.1E-01 |
| | chlordane | 2.1E-02 | 4.6E-02 | 2.1 | 4.5E-02 | 1.0E-02 |
| 10 | Aroclor-1260 | 3.2E-02 | 7.3E-02 | 2.3 | 7.3E-02 | 1.4E-02 |
| | chlordane | 9.7E-03 | 1.6E-02 | 1.6 | 1.6E-02 | 6.0E-03 |
| 13 | Aroclor-1254 | 1.6E-01 | 2.0E+04 | | | |
| | Aroclor-1260 | 3.0E-01 | 4.9E-01 | 1.6 | 4.8E-01 | 1.9E-01 |
| | chlordane | 6.5E-02 | 1.2E-01 | 1.8 | 1.2E-01 | 3.6E-02 |

a A log-uniform distribution was used for these concentrations instead of a uniform distribution due to their large uncertainty ranges.

concentrations were calculated as described in section 3.1.1 and are also presented in Table 3.3.

As with the previously discussed contaminants, *a uniform distribution was used to represent the uncertainty in the estimate of the arithmetic mean concentrations (except for the concentrations with UF values above 2.5 as indicated in Table 3.3).* Due to their relatively large degree of uncertainty in the estimate of the mean, a log-uniform distribution was used for the seven concentrations with UF values exceeding 2.5.

## 3.2 Exposure Factors Independent of the Contaminant Type

Five exposure factors are used in the risk assessment models which are independent of specific radionuclides and chemicals: fish ingestion (daily intake) ($I$), exposure frequency ($EF$), exposure duration ($ED$), body mass ($BM$), and averaging time ($AT$). The subjective uncertainty bounds and probability distributions for these parameters were chosen such that the true but unknown value pertaining to the maximally exposed individual lies within the interval. A literature review and moderate knowledge of the local behavior were used to determine the best estimate of uncertainty for each of these parameters.

### 3.2.1 Fish Ingestion ($I$), kg/day

The fish ingestion rate was defined in the following manner:

$$I = M_w \cdot F_m \tag{2.6}$$

where $M_w$ = the mass of fish consumed per meal (kg/fish-meal)

and $F_m$ = the frequency of meals consisting of fish caught in the Clinch River/Watts Bar Reservoir system (fish-meal/day).

A subjective probability distribution was formulated to represent the uncertainty in each individual parameter, i.e., distributions for $M_w$ (same for both carcinogens and non-carcinogens) and $F_m$ (different for carcinogens and non-carcinogens) were determined.

The amount of fish eaten per meal was estimated to be 0.230 kg/fish-meal for the Clinch River/Watts Bar Reservoir system. The 95th percentile value given for fin fish by the USEPA (1989) is 0.284 kg/fish-meal, obtained from a study by Pao et al. (1982). In addition, Rupp et al. (1980) analyzed fish consumption from several surveys performed in the late 1960s to early 1970s. Perhaps the most relevant of these was a survey done in 1966-67 on the Columbia River. This survey was restricted to households with at least one fisherman, and memory recall was used to estimate the fish consumption rate over a period of one year. The average fish meal was determined to be 0.200 kg, and the maximum fish meal was estimated at 0.313 kg.

After substantial consideration of these surveys, *a triangular distribution was used to represent the subjective uncertainty for* $M_w$. For the avid fisherman, the most likely value (the mode) was estimated at 0.230 kg/fish-meal (this corresponds to about one-half of a pound). The range of the uncertainty was assessed from the Columbia River study because of its direct applicability to the present assessment's target individual (the avid fisherman). A maximum value of 0.320 kg/fish-meal and a minimum value of 0.170 kg/fish-meal was used for

the triangular distribution. The minimum value was derived by dividing 0.230 kg/fish-meal by 0.320 kg/fish-meal, multiplying this ratio by 0.230 kg/fish-meal, and rounding to the nearest hundredth.

#### 3.2.1.1 Carcinogens

For the parameter $F_m$, the uncertainty was based primarily on subjective judgment. Because each value was equally likely to be the true but unknown number of meals per day, *a log-uniform distribution was used to represent the uncertainty*. An upper bound value was determined by assuming that a fisherman consumes two meals per week of fish caught from the Clinch River/Watts Bar Reservoir system. For the avid fisherman, this seems to be an valid assumption. This results in a maximum frequency value of 0.286 fish-meals/day. A lower bound was formulated by assuming that a fisherman will consume one meal per month of fish caught from the Clinch River/Watts Bar Reservoir system. This value translates to 12 meals/yr, which closely matches the average frequency of 14 meals/yr obtained in the Columbia River survey (Rupp et al., 1980). This value results in a minimum frequency value of 0.033 fish-meals/day. For carcinogens, the estimate of the uncertainty in $F_m$ should be likely values to occur over a 30 year period. Therefore, $F_m$ is sufficiently represented by the log-uniform distribution.

These estimates exclude values for subsistence fishermen who may be obtaining a major portion of their dietary requirements from fish caught from the Clinch River/Watts Bar Reservoir system. At the present time, the actual number of individuals in this category is not likely to be large and will not be considered in this uncertainty analysis.

### 3.2.1.2 Non-Carcinogens

For non-carcinogens, where toxic effects are the result above a threshold exposure, the uncertainty bounds for $F_m$ must take into account the possibility of an avid fisherman consuming a substantial amount of fish during a one year period. It is possible that a fisherman would eat an average of 2 fish-meals/week from the Clinch River/Watts Bar Reservoir system for one year. Because of the lack of site specific data, *a log-uniform distribution was chosen to represent the subjective uncertainty in* $F_m$. The uncertainty bounds were obtained by doubling the estimated values for the carcinogen consideration so that an account of someone who chooses to participate in unusual eating habits for one year is obtained.

The uncertainty values for the $M_w$ and the $F_m$ parameters for carcinogens and non-carcinogens are summarized in Table 3.4.

Table 3.4. Summary information for the uncertainty in the parameters used to calculate daily intake.

| Class | Parameter | Distribution | Minimum | Maximum | Mode |
|---|---|---|---|---|---|
| Both | $M_w$ | Triangular | 0.170 | 0.320 | 0.230 |
| Carcinogens | $F_m$ | Log-Uniform | 0.033 | 0.286 | ------ |
| Non-Carcinogens | $F_m$ | Log-Uniform | 0.067 | 0.571 | ------ |

For the Sr-90 concentrations, the bones of the fish must be included in "fish patties" in order to obtain a significant exposure. Although this scenario seems unlikely, the fish-meals in this analysis for Sr-90 are assumed to be "fish patties".

### 3.2.2 Exposure Frequency (*EF*), 350 days/year

The *EF* represents the number of days out of a year that a person is exposed to the daily intake rate. The USEPA (1991) currently recommends the use of 350 days/year. This value takes into account 2 weeks of vacation per year during which the individual will not be exposed to the contaminant/pathway being studied.

To determine a lower bound for the exposure frequency, it was assumed that in addition to the 2 weeks of vacation, half of the weekends are spent vacationing, resulting in an *EF* of about 300 days/year. The maximum value can not be larger than 365 days/year. Because it seems more likely that someone would spend two weeks away on vacation than a total of 65 days away on vacation, *a triangular distribution was used to represent the subjective uncertainty about the EF,* with a mode of 350 days/year, a minimum of 300 days/year, and a maximum of 365 days/year.

### 3.2.3 Exposure Duration (*ED*)

The *ED* represents the number of years that an individual will be exposed to the given exposure conditions (daily intake rate and exposure frequency), i.e., an estimate of the projected length of time that a person will live in the area under consideration.

For all carcinogens, USEPA (1989) recommends a value of 30 years for the *ED*. This value represents the 90th percentile based on national statistics. Israeli and Nelson (1992) carried out an in-depth study of the distribution and expected time of residence for U.S. households, using housing surveys published by the

Bureau of Census and the U.S. Department of Housing and Urban Development. The total residence time is defined as the time between moving into and out of a residence, and it depends both on the household category or group (e.g., urban, rural, etc.) and on whether the resident is an owner or a renter. The total residence times for rural areas are larger than those for urban areas. Because most of the land surrounding the Clinch River/Watts Bar Reservoir system is rural, the information from Israeli and Nelson (1992) pertaining to the total residence time for rural residents was used to estimate the uncertainties in the exposure duration.

Values are available for the 95th, 90th, 75th, 50th, and 25th percentiles (Israeli and Nelson, 1992). Because our target individual is an avid fisherman likely to reside in the area for some length of time, the 75th percentile value of 9.1 years for rural households was used as an estimate for the lower bound on the exposure duration. Although the 95th percentile for rural households is 32.3 years, a value of 50 years was used as the upper bound on the reasonable exposure for a "maximally exposed" individual. This value was chosen as a maximum to include the possibility of one living his/her entire adult life in the same area. For the local community under consideration, this does not seem to be an unlikely possibility. Since each value between the upper and lower bounds was thought to be equally likely to occur and due to the absence of site-specific data, a *uniform distribution was used to represent the subjective uncertainty in the exposure duration*.

For non-carcinogens, immediate toxic effects from over-exposure are of primary concern. These effects typically occur when a person has ingested too much of a contaminated medium. Since it is unlikely that someone would

maintain excessive eating habits for more than one year, a fixed value of one year was chosen for the non-carcinogen *ED*.

### 3.2.4 Body Mass (*BM*), 70 kg

The "average" value for the body mass of Reference Man is 70 kg (ICRP, 1975). However, as one can observe, the estimate of this "average" value has considerable uncertainty. The subjective probability distribution for this parameter was intended to target the "average male." The first consideration was the minimum value that a man might weigh. For this assumption, a value of 55 kg was chosen (about 121 pounds). Next, a maximum value of body mass for the target individual was set at 110 kg (about 242 pounds). Because the true value of the body mass for the average male seems less likely at the minimum and maximum values than around the value of 70 kg, *a triangular distribution was chosen to represent the uncertainty in this parameter*. A mode of 70 kg, a minimum of 55 kg, and a maximum of 110 kg were used to describe the distribution.

### 3.2.5 Averaging Time (*AT*), days

The averaging time is defined as the period of time over which an exposure is averaged. Because it serves as a prorating parameter (USEPA, 1989), the values for the averaging time were fixed for this uncertainty analysis. For non-carcinogens the averaging time is equal to the exposure duration. Therefore, *AT* for non-carcinogens is set at 365 days. For carcinogenic substances, where the effect from a single exposure could occur several years after the received dose,

the exposure is averaged over a lifetime (70 years). Therefore, *AT* for carcinogens equals 25,550 days.

## 3.3 Radionuclide Specific Factors

### 3.3.1 Dose Conversion Factor (*DCF*), Sv/Bq

The committed effective dose per unit intake is defined as the *DCF* for this report. The committed effective dose (hereafter referred to as "dose") is defined by the International Commission on Radiological Protection (ICRP, 1989) as the time integral of the dose-equivalent rate in a particular tissue that is received by an individual following an intake of radioactive material into the body. A 70-year lifetime was assumed in this calculation. For an adult, a body weight corresponding to Reference Man [70 kg (ICRP, 1975)] and an averaging time of 50 years were assumed (ICRP, 1989; ICRP, 1991; NCRP, 1993).

Dosimetric models and the parameters that are used to determine the "dose" vary with the intake pathway, the radionuclide, age at the time of exposure, gender, etc. Recent studies have used various biokinetic data and in some cases (such as Pu) radionuclide-specific models to obtain age-dependent doses to members of the public (ICRP, 1989; ICRP, in press).

Recently, members of the National Council on Radiation Protection and Measurements (NCRP) scientific committee 57-16 were asked to estimate the uncertainty factors associated with the ingestion and inhalation "dose" factors for a variety of radionuclides (NCRP, in press). These uncertainty factors were estimated for the application of ICRP-30 "dose" factors and might not apply to the newly projected factors listed in ICRP-56 (ICRP, 1989) or to recent revisions of

the ICRP-56 values (ICRP, in press). Uncertainty factors pertaining to Co-60, Sr-90, Cs-137, and Pu-239 were obtained by expert judgment (Eckerman, 1993). The "dose" factors and their associated uncertainty factors from both sources are summarized in Table 3.5.

Table 3.5. Dose conversion factors for the ingestion pathway and corresponding uncertainty factors for Co-60, Sr-90, Cs-137, and Pu-239.

| Contaminant | *DCF* for Ingestion[a] (Sv/Bq) | Uncertainty Factor[b] A[c] | B[d] |
|---|---|---|---|
| Co-60 | 3.8E-09 | 10 | 10 |
| Sr-90 | 4.0E-08 | 3 | 2 |
| Cs-137 | 1.4E-08 | 2 | 2 |
| Pu-239 | 2.5E-07 | 10 | 10 |

a ICRP (in press).
b These uncertainty factors apply to the adult male.
c NCRP (in press).
d Eckerman (1993).

The uncertainty factor values from both sources are the same except for Sr-90; for Sr-90, the more recently determined value of 2 was used (Eckerman, 1993). The primary source of uncertainty for Cs-137 and Sr-90 is inaccuracy in the biokinetic data used to estimate the *DCF*. The primary source of uncertainty for Co-60 is the absence of a physiological model, and for Pu-239, the uncertainty in the amount absorbed through the gastro-intestinal tract.

*A log-triangular distribution was used to represent the subjective uncertainty in the DCF values.* The *DCF* value itself was used as the mode of the triangle. The maximum value was obtained by multiplying the *DCF* by the UF, and the

minimum value by dividing the *DCF* by the UF. The modes, the maximums, and the minimums are summarized in Table 3.6.

Table 3.6. Parameters required to establish the log-triangular distribution used to describe the uncertainty in the *DCF*s.

| Contaminant | Mode | Maximum | Minimum |
|---|---|---|---|
| Co-60 | 3.8E-09 | 3.8E-08 | 3.8E-10 |
| Sr-90 | 4.0E-08 | 8.0E-08 | 2.0E-08 |
| Cs-137 | 1.4E-08 | 2.8E-08 | 7.0E-09 |
| Pu-239 | 2.5E-07 | 2.5E-06 | 2.5E-08 |

### 3.3.2 Risk Conversion Factor (*RCF*)

The *RCF* as determined by the ICRP (1991) and the NCRP (1993) is 0.073/Sv; this is based primarily on extensive analyses of the atomic bomb survivors in Nagasaki and Hiroshima, Japan, by such scientific bodies as the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) and the National Research Council of the National Academy of Sciences (NRC/NAS). The *RCF* serves as a method of predicting the excess lifetime risk for genetic defects (1.3%/Sv), dying of cancer (5%/Sv), and other occurrences of cancer (1%/Sv) (NCRP, 1993; ICRP, 1991). The *RCF* is defined by the ICRP's definition of "detriment" where the risk of genetic defects and occurrences of cancer incidence are weighted against the number of fatalities (ICRP, 1991).

There are several sources of uncertainty in the *RCF*, but two sources tend to dominate the overall estimate of uncertainty (NCRP, 1993). These are (1) the uncertainty in the *RCF* evaluated at high dose and high dose rate exposures, and (2) the uncertainty in extrapolating the *RCF* to low dose and low dose rate

exposures. Of these, the uncertainty in extrapolating to low dose and low dose rate exposures appears to be the predominant uncertainty in the *RCF* at low doses (NCRP, 1993).

An uncertainty factor of about 2 is recommended for the uncertainty in the *RCF* at high dose and high dose rate exposures, and a UF of greater than or equal to 2 is recommended for the uncertainty involved in the extrapolation to low doses (NCRP, 1993). The possibility of the minimum value for the *RCF* being **zero** cannot be excluded, due to the absence of sufficient data to allow a significant analysis of the risk for exposure at low doses (NRC/NAS, 1990; NCRP, 1993). However, for this uncertainty analysis, a minimum value of zero will not be used, on the assumption that the underlying model for the risk associated with radiation exposure is correct and there is no existence of a threshold dose.

Based on the above information, an uncertainty factor of **4** was used for this analysis. *A log-triangular distribution was used to represent this subjective uncertainty.* The recommended *RCF* value of 0.073/Sv was used as the mode, the maximum was determined by multiplying the *RCF* by 4, and the minimum was obtained by dividing the *RCF* by 4. These values are summarized below.

| | | |
|---|---|---|
| Mode | = | 0.073 |
| Maximum | = | 0.292 |
| Minimum | = | 0.018 |

## 3.4 Chemical Specific Factors

### 3.4.1 Non-Carcinogens - Reference Doses

The risk from exposure to non-carcinogens is evaluated by calculating a quantity referred to as the Hazard Quotient (equation 1.2). The potential exposure to an individual is divided by a value called the reference dose (*RfD*) to obtain this quotient. "A *RfD* is defined as an estimate of a daily exposure level (chronic) for the human population that is likely to be without an appreciable risk of deleterious effects" (USEPA, 1989). Therefore, when the *HQ* is greater than 1, further study and perhaps remediation is recommended.

The *RfD* is obtained by dividing a "No Observed Adverse Effect Level" (NOAEL), which is obtained from the analysis of a human or more often an animal data set, by an uncertainty factor and a modifying factor (USEPA, 1989). An uncertainty factor of 10 is applied to protect the most sensitive population, a value of 10 is used when extrapolating from animals to humans, a value of 10 is used when a NOAEL is obtained from a subchronic study, and a factor of 10 is applied when a "Lowest Observed Adverse Effect Level" (LOAEL) is obtained instead of a NOAEL (USEPA, 1989). Finally, a modifying factor is applied based on best professional judgment.

The following sections will discuss the uncertainties for the MeHg and chlordane *RfD*s.

### 3.4.1.1 Methyl-mercury

The *RfD* for MeHg is 3.0E-04 mg/kg-day, and a safety factor of 10 is reported (USEPA, 1992). This value was based on an animal study that was performed in the 1950s (Canady, 1993).

There have been three major outbreaks of MeHg poisoning. The first two occurred in Minamata and Niigata, Japan during the late 1950s to early 1960s and were caused by industrial release of MeHg and other mercury compounds into Minamata Bay and the Agano River (WHO, 1976). Although there were numerous people affected, this study has proven ineffective in obtaining a better *RfD* because of the inability to relate an exposure level to the observed effect (Canady, 1993). The third and largest recorded outbreak occurred in Iraq during the winter of 1971-1972 (WHO, 1976). The people of Iraq used wheat and other grain seed which was treated with MeHg to make bread. Excessive ingestion of this bread due to their economic conditions resulted in a major poisoning epidemic. Studies have been performed to relate the concentration in hair and blood to the actual exposure (USEPA, 1984; WHO, 1976).

The USEPA is currently reevaluating the *RfD* for MeHg based on this Iraqi data (Canady, 1993). A LOAEL value from the Iraq data set which approximates the NOAEL obtained in the rat study is under review. However, this LOAEL (which would be divided by 10 to obtain a value close to the current *RfD*) occurs with severe neurological effects. Less obvious effects to the central nervous system could be occurring that are being overlooked (Canady, 1993). Therefore, a lower *RfD* value may be derived by the USEPA.

The current *RfD* is used as the mode of the subjective probability distribution, and *a log-triangular distribution is used to represent the uncertainty in the RfD*. To protect against more subtle neurological effects, a minimum value was obtained by dividing the *RfD* by the safety factor of 10 (3.0E-05 mg/kg-day). A maximum value was determined by multiplying the *RfD* by the safety factor of 10 (3.0E-03 mg/kg-day).

### 3.4.1.2 Chlordane

The *RfD* for chlordane is 5.0E-05 mg/kg-day, and a safety factor of 1000 is given (USEPA, 1988b). There is only one study that is reported with significant information to derive an *RfD* for chlordane (USEPA, 1988b). The *RfD* was based on an unpublished report by the Velsicol Chemical Corporation (1983) describing the results obtained from a rat study. A LOAEL of 0.045 mg/kg-day which caused a significant increase in hepatocellular necrosis (liver tumors) in males was used to derive the *RfD* (USEPA, 1988b; Velsicol Chemical Corp., 1983).

Because there is limited information pertaining to the *RfD*, an uncertainty factor was obtained by taking the square root of the reported safety factor. The *RfD* was multiplied by this UF to obtain a maximum value and was divided by the UF to obtain a minimum value. The reported *RfD* was used as the most likely value to occur (mode). *Therefore, a log-triangular distribution was used to represent the uncertainty in the RfD for chlordane with a mode of 5.0E-05 mg/kg-day, a minimum of 1.6E-06 mg/kg-day, and a maximum of 1.6E-03 mg/kg-day.*

### 3.4.2 Carcinogens - Slope Factors

A risk from exposure to a carcinogenic substance is believed to exist no matter how small the level of exposure. Since this "no threshold" concept is recognized, the potential excess lifetime risk of cancer from exposure to carcinogens is evaluated by relating a potential dose to risk (equation 1.3). This "conversion" is determined by applying a quantity called the Slope Factor (*SF*).

There are several sources of uncertainty associated with the derivation of these *SF*s. Some of these include accuracy of the data in "sensitive" species at high doses, the extrapolation of a response at high doses to low doses, and extrapolation from animals to humans.

The following sections describe the subjective probability distributions chosen to represent the uncertainty in the *SF*s for PCBs and chlordane.

#### 3.4.2.1 Poly-chlorinated Biphenyls

The *SF* for PCBs is reported as 7.7 per (mg/kg-day) (USEPA, 1992). This value is the highest determined *SF* from 3 rat studies using Aroclor-1260 (USEPA, 1988a). One large area of uncertainty in the *SF* for PCBs is that a rat population is exposed to a known concentration of a specific Aroclor with a certain outcome. In reality, no one knows what form the human population is exposed to because of the transformation of PCBs in the environment (Cogliano, 1993). In addition, Aroclor-1260 was chosen to be representative of all of the Aroclors (USEPA, 1988a). However, recent studies suggest that this assumption

may not be correct (Silberhorn, et al., 1990). For this uncertainty analysis the *SF*s for Aroclor-1254 and -1260 are considered separately.

Aroclor-1260 has been evaluated in three separate rat studies. From these studies, *SF*s of 3.9, 5.7, and 7.7 per (mg/kg-day) have been determined (USEPA, 1988a). These values differ by about a factor of 1.3-1.5. A UF of 1.3 was multiplied by the reported *SF* to obtain a maximum value. Because the only significant evidence of carcinogenicity exists in rats, the possibility of Aroclor-1260 not being a carcinogen in humans can not be ignored. Therefore, *a triangular distribution with a mode of 7.7 per (mg/kg-day), a minimum of 0 per (mg/kg-day), and a maximum of 10 per (mg/kg-day) was used to represent the uncertainty in the SF for Aroclor-1260.*

The evaluation of a *SF* for Aroclor-1254 has been performed in only one rat study. The *SF* obtained from this study was 2.6 per (mg/kg-day) (USEPA, 1988a). Because this value is sufficiently lower than the reported *SF* for PCBs, a different mode was used for the Aroclor-1254 subjective probability distribution. In order to provide a conservative estimate for the mode, an uncertainty factor of 1.5 (obtained by dividing the lowest Aroclor-1260 *SF* by the Aroclor-1254 *SF*) was multiplied by the value of 2.6. This results in a value of about 4.0 per (mg/kg-day) for the most likely *SF*. Therefore, *a triangular distribution with a mode of 4.0 per (mg/kg-day), a minimum of 0 per (mg/kg-day), and a maximum of 10 per (mg/kg-day) was chosen to describe the uncertainty in the SF for Aroclor-1254.*

### 3.4.2.2 Chlordane

The reported *SF* for chlordane is 1.3 per (mg/kg-day) (USEPA, 1992). Like PCBs, there is considerable uncertainty due to the transformation of chlordane in the environment.

Slope factors have been evaluated for 4 mice studies and 1 rat study (USEPA, 1986). The largest *SF* obtained from the 4 mice studies was 4.7 per (mg/kg-day), and the smallest *SF* was 0.3 per (mg/kg-day) (USEPA, 1986). The USEPA *SF* of 1.3 per (mg/kg-day) was derived by taking the geometric mean of these 4 mice studies. This value compares to the *SF* of 1.1 per (mg/kg-day) which was derived from the rat study (USEPA, 1986).

Because the studies were done for mice and rats exposed to technical chlordane, as with PCBs, the possibility of no carcinogenic effect in humans can not be ignored. A maximum uncertainty estimate of 5 was obtained by rounding the largest *SF* derived from the mice studies. Therefore, *a triangular distribution with a mode of 1.3 per (mg/kg-day), a minimum of 0 per (mg/kg-day), and a maximum of 5 per (mg/kg-day) was used to describe the uncertainty in the SF for chlordane.*

# 4.0 Quantitative Uncertainty Analysis

This section presents the results of a quantitative uncertainty analysis in which Monte Carlo simulation was used to propagate the subjective probability distributions for the parameters involved in the risk assessment into a subjective probability distribution for both the excess lifetime cancer risk and the *HI*. The approach used for the Monte Carlo simulation was Latin Hypercube Sampling.

The inputs required for the Monte Carlo simulations are the subjective probability distributions and uncertainty bounds for each parameter presented in Section 3.0. Using the various input distributions, a Monte Carlo simulation provides an estimate of the model result in terms of a subjective probability distribution. From this distribution, a subjective confidence interval for the true but unknown result is obtained. *The term "subjective confidence interval" means that the probability distributions specified for the uncertain model parameters were derived using subjective judgment in the absence of perfect data.*

In addition to the subjective confidence intervals obtained for the model endpoint, the results of a sensitivity analysis are presented in this section. A sensitivity analysis permits the identification of the parameter having the greatest effect on the total result. The sensitivity analysis was accomplished by holding a potentially sensitive parameter constant while allowing all others to vary. The results obtained for each parameter were compared and ranked according to the amount of influence the parameter had on the uncertainty in the result. The parameters having the greatest effect are considered to be the most sensitive. Although not employed in this analysis, scatter plots of the input parameters and statistical regression techniques are other methods of performing sensitivity analyses. Descriptions of statistical approaches to sensitivity analysis

using regressions of the randomly selected values of the uncertain parameters on the values produced for the model predictions can be found in Iman et al. (1981a; 1981b), IAEA (1989), and Iman and Helton (1991).

## 4.1 Analysis for Radionuclides

### 4.1.1 Subjective Confidence Intervals Obtained from the Uncertainty Analysis

The assumptions used for this uncertainty analysis are discussed in Section 3.0. These assumptions were used to find the median, the lower 5% subjective confidence limit (LCL), and the upper 95% subjective confidence limit (UCL) for the excess lifetime cancer risk associated with human ingestion of fish from the Clinch River/Watts Bar Reservoir system which are contaminated with Co-60, Sr-90, and Cs-137. These values were obtained by using 500 iterations of LHS. Subjective confidence intervals were obtained for each estimated arithmetic mean concentration. Subjective confidence intervals for the total lifetime cancer risk were obtained for each reach or section of the river system.

The subjective confidence intervals are summarized in Table 4.1. The median risk for Co-60 in reach 1, for instance, is 2.9E-08; there is 90% confidence that the true but unknown risk lies between 1.8E-09 and 4.3E-07 (or there is 95% confidence that the true but unknown value does not exceed 4.3E-07). The median risk for Cs-137 in reach 1 is 2.5E-07, and there is 90% confidence that the true but unknown risk lies between 4.9E-08 and 1.4E-06. The median value for the total risk from reach 1 is 3.0E-07, and there is 90% confidence that the true but unknown total risk for reach 1 lies between 5.7E-08 and 1.8E-06.

Table 4.1. Results obtained for the excess lifetime cancer risk from radionuclide contaminants in the Clinch River/Watts Bar Reservoir system from 500 iterations of LHS using the assumptions presented in Section 3.0.

| Reach | Radionuclide | 5% Subjective Confidence Limit | Median | 95% Subjective Confidence Limit |
|---|---|---|---|---|
| 1 | Co-60 | 1.8E-09 | 2.9E-08 | 4.3E-07 |
| | Cs-137 | 4.9E-08 | 2.5E-07 | 1.4E-06 |
| | Total Risk[a] | 5.7E-08 | 3.0E-07 | 1.8E-06 |
| 2 | Sr-90 | 1.8E-08 | 1.1E-07 | 6.5E-07 |
| | Cs-137 | 6.4E-07 | 3.4E-06 | 1.6E-05 |
| | Total Risk[a] | 6.6E-07 | 3.6E-06 | 1.7E-05 |
| 3 | Co-60 | 2.2E-09 | 2.9E-08 | 5.1E-07 |
| | Sr-90 | 1.4E-08 | 6.8E-08 | 3.8E-07 |
| | Cs-137 | 1.0E-07 | 5.2E-07 | 2.7E-06 |
| | Total Risk[a] | 1.2E-07 | 6.5E-07 | 3.5E-06 |
| 4 | Sr-90 | 1.1E-08 | 5.9E-08 | 3.5E-07 |
| | Cs-137 | 1.6E-07 | 8.8E-07 | 4.2E-06 |
| | Total Risk[a] | 1.7E-07 | 9.4E-07 | 4.5E-06 |
| 5 | Sr-90 | 9.6E-09 | 5.8E-08 | 3.2E-07 |
| | Cs-137 | 8.1E-08 | 4.4E-07 | 2.2E-06 |
| | Total Risk[a] | 9.3E-08 | 5.0E-07 | 2.5E-06 |
| 10 | Co-60 | 7.9E-10 | 9.6E-09 | 1.5E-07 |
| | Sr-90 | 7.4E-09 | 4.1E-08 | 2.2E-07 |
| | Cs-137 | 4.6E-08 | 2.5E-07 | 1.2E-06 |
| | Total Risk[a] | 6.0E-08 | 3.1E-07 | 1.8E-06 |
| 13 | Co-60 | 1.1E-08 | 1.1E-07 | 1.1E-06 |
| | Sr-90 | 1.0E-08 | 6.0E-08 | 3.9E-07 |
| | Cs-137 | 8.8E-08 | 4.9E-07 | 3.0E-06 |
| | Total Risk[a] | 1.3E-07 | 7.4E-07 | 4.4E-06 |

[a] The values for the total risk might not be directly additive because of the random selection process in the calculation.

Examples of the subjective probability distributions obtained in the uncertainty analysis are given in Figures 4.1, 4.2, and 4.3, respectively, for the excess lifetime cancer risk for the concentrations of Co-60 and Cs-137 in reach 1 and the total lifetime risk for reach 1.

The 5% and 95% subjective confidence limits are marked with triangles on the horizontal axis in the figures. The uncertainty in the estimates of the excess lifetime cancer risk is log-normally distributed; therefore, the most likely excess lifetime cancer risk to the maximally exposed individual occurs near the LCL.

The highest potential excess lifetime cancer risk occurs in Reach 2, with a median of 3.6E-06. There is 95% confidence, given the assumptions used in this analysis, that the true but unknown total risk from radionuclides in reach 2 does not exceed 1.7E-05. The primary source of this risk is from Cs-137.

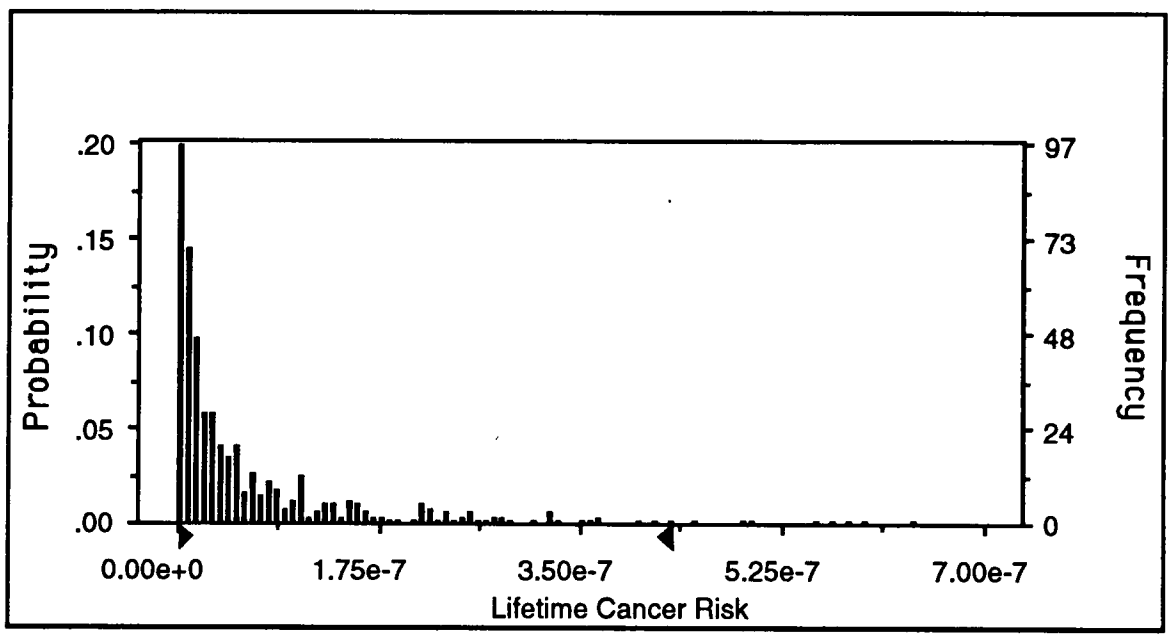


Figure 4.1. Subjective probability distribution of the excess lifetime cancer risk obtained for the Co-60 concentration in reach 1 after 500 iterations of LHS.

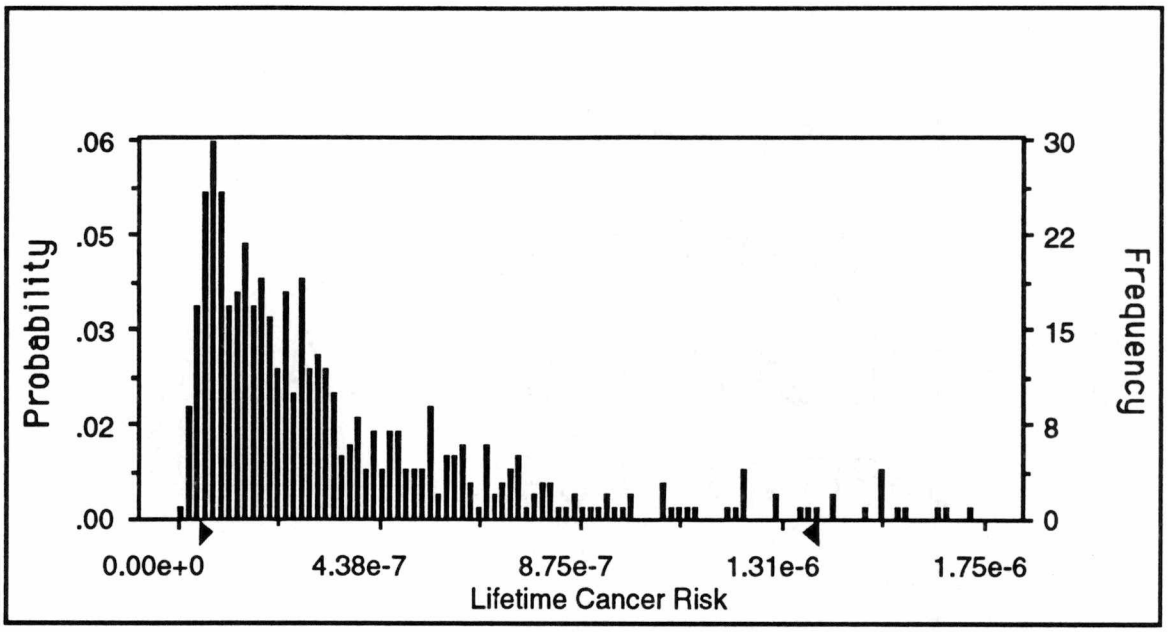


Figure 4.2. Subjective probability distribution of the excess lifetime cancer risk obtained for the Cs-137 concentration in reach 1 after 500 iterations of LHS.

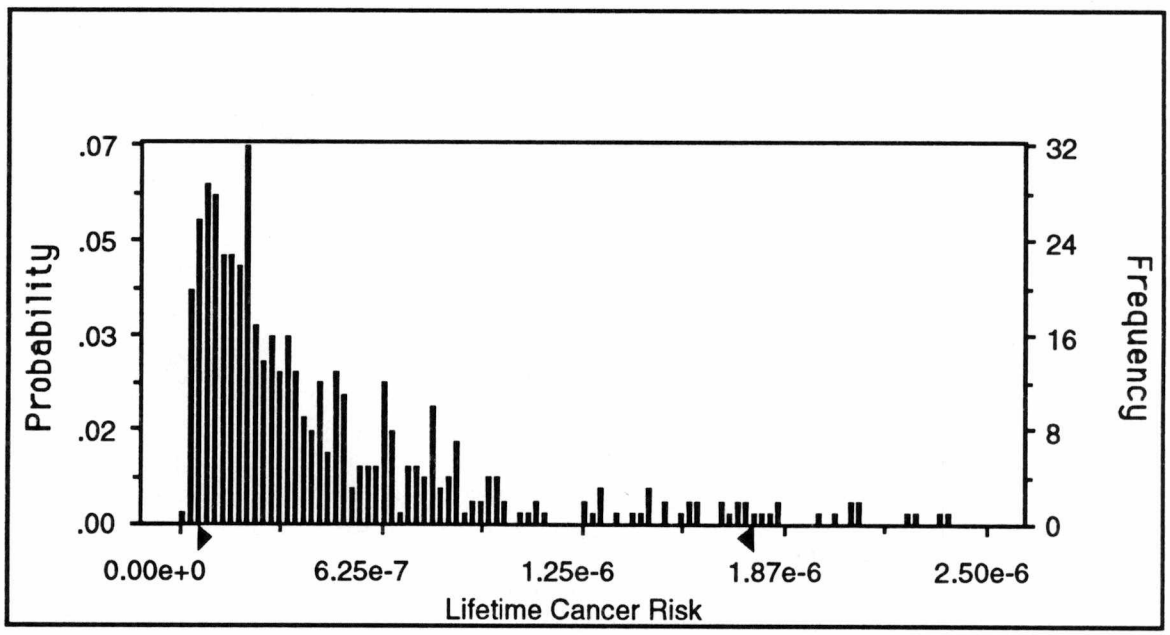


Figure 4.3. Subjective probability distribution of the total excess lifetime cancer risk obtained from radionuclides in reach 1 after 500 iterations of LHS.

The wide ranges of the excess lifetime cancer risk for Co-60 in reaches 1, 3, and 10 are partially due to the large uncertainty associated with the arithmetic mean concentrations. This large uncertainty occurs because of the low number of measurements that were actually above detection limits in comparison with the number of samples taken. Since all Co-60 estimates are at or below a 1.0E-06 lifetime cancer risk, it is recommended that this radionuclide be given low priority for further investigation.

The subjective confidence intervals given in Table 4.1 provide valuable information that can be used to guide decision-making. For example, if a 5% lower confidence limit is above a regulatory standard of concern, then remediation is most likely needed. If the 95% upper confidence limit is below the standard, remediation is most likely not required (all Reaches). If the 95% upper confidence limit is above the standard, but the 50$^{th}$ percentile is below the standard, further study should be recommended for those parameters which dominate the overall uncertainty. However, if the 50$^{th}$ percentile is above the standard, further study may still be recommended, although under some circumstances it may be useful to proceed with remediation. The decision as to whether or not to proceed with remediation will be driven by a cost versus benefits analysis.

### 4.1.2 Comparison of Results Obtained with the Uncertainty Analysis with those Presented as Screening Calculations

The results of conservative and nonconservative screening for the excess lifetime cancer risk to adults from radionuclides in fish are summarized in Table 4.2 (Cook et al., 1992).

Table 4.2. Summary of screening results for the excess lifetime cancer risk to adults from radionuclides in fish as reported in Cook et al. (1992).

| Reach | Element | 5% Subj. Confidence Limit | Nonconservative Screening Estimate | 95% Subj. Confidence Limit | Conservative Screening Estimate |
|---|---|---|---|---|---|
| 1 | Co-60 | 1.8E-09 | 7.1E-09 | 4.3E-07 | 3.4E-07 |
| | Cs-137 | 4.9E-08 | 5.9E-08 | 1.4E-06 | 4.0E-07 |
| | Total | 5.7E-08 | 6.6E-08 | 1.8E-06 | 7.4E-07 |
| 2 | Sr-90 | 1.8E-08 | 1.2E-08 | 6.5E-07 | 8.7E-08 |
| | Cs-137 | 6.4E-07 | 5.7E-07 | 1.6E-05 | 5.8E-06 |
| | Total | 6.6E-07 | 5.9E-07 | 1.7E-05 | 5.8E-06 |
| 3 | Co-60 | 2.2E-09 | 4.8E-09 | 5.1E-07 | 4.1E-07 |
| | Sr-90 | 1.4E-08 | 7.2E-09 | 3.8E-07 | 5.1E-08 |
| | Cs-137 | 1.0E-07 | 1.1E-07 | 2.7E-06 | 7.8E-07 |
| | Total | 1.2E-07 | 1.2E-07 | 3.5E-06 | 1.2E-06 |
| 4 | Sr-90 | 1.1E-08 | 6.8E-09 | 3.5E-07 | 4.5E-08 |
| | Cs-137 | 1.6E-07 | 2.1E-07 | 4.2E-06 | 1.2E-06 |
| | Total | 1.7E-07 | 2.2E-07 | 4.5E-06 | 1.3E-06 |
| 5 | Sr-90 | 9.6E-09 | 5.9E-09 | 3.2E-07 | 4.1E-08 |
| | Cs-137 | 8.1E-08 | 9.8E-08 | 2.2E-06 | 6.9E-07 |
| | Total | 9.3E-08 | 1.0E-07 | 2.5E-06 | 7.3E-07 |
| 10 | Co-60 | 7.9E-10 | 2.6E-09 | 1.5E-07 | 1.1E-07 |
| | Sr-90 | 7.4E-09 | 4.5E-09 | 2.2E-07 | 3.1E-08 |
| | Cs-137 | 4.6E-08 | 5.9E-08 | 1.2E-06 | 4.1E-07 |
| | Total | 6.0E-08 | 6.6E-08 | 1.8E-06 | 5.6E-07 |
| 13 | Co-60 | 1.1E-08 | 6.2E-08 | 1.1E-06 | 3.8E-07 |
| | Sr-90 | 1.0E-08 | 5.4E-09 | 3.9E-07 | 3.7E-08 |
| | Cs-137 | 8.8E-08 | 1.2E-07 | 3.0E-06 | 9.0E-07 |
| | Total | 1.3E-07 | 1.9E-07 | 4.4E-06 | 1.3E-06 |

The screening calculations (Cook et al., 1992) are within the subjective confidence intervals obtained in the uncertainty analysis. The nonconservative estimates of the excess lifetime cancer risk are very similar to the LCLs. The conservative estimates of the excess lifetime cancer risk are somewhat similar to but less than the UCLs.

The most significant differences between the conservative estimates and the UCL values are for the Sr-90 calculations. This difference primarily results from the different methods used for converting the exposure to lifetime cancer risk. The screening estimates were calculated using slope factors published in the Health Effects Assessment Summary Tables (USEPA, 1992). The LCL and the UCL values were estimated using the most recent *DCFs* (ICRP, in press) and the *RCF* (ICRP, 1991; NCRP, 1993). The two "slope factors" are summarized in Table 4.3 for each radionuclide. The difference between the two factors for Sr-90 is about a factor of 3.

Table 4.3. Slope factors used to convert exposure to lifetime cancer risk.

| Radionuclide | Derived Slope Factor[a] | HEAST Slope Factor[b] |
|---|---|---|
| Co-60 | 2.8E-10 | 4.1E-10 |
| Sr-90 | 2.9E-09 | 9.7E-10 |
| Cs-137 | 1.0E-09 | 7.6E-10 |

a Derived by multiplying the *DCF* value (ICRP, in press) and the *RCF* value (ICRP, 1991; NCRP, 1993).

b USEPA (1992).

Contrary to popular belief, the slope factors for radionuclides recommended by the USEPA are not always conservative. The use of recommended EPA slope factors could lead to an underestimate of the excess lifetime cancer risk for radionuclides.

### 4.1.3 Results of the Sensitivity Analysis

A sensitivity analysis was performed for each "Total Lifetime Cancer Risk". For each risk under evaluation, each parameter in turn was held constant while all others were allowed to vary. In order to obtain an accurate comparison of the "sensitive study" risk and the total risk, 500 iterations of LHS with an initial seed value of 1.0 were used. Setting an initial value of the random number seed sets the simulation for the same starting point so that the results are reproducible given the same assumptions.

A sensitivity index (SI) was calculated for each parameter. The equation for the sensitivity index is given below:

$$SI = 1 - \frac{\left(\frac{UCL_{95\%}}{LCL_{5\%}}\right)_{Sens.}}{\left(\frac{UCL_{95\%}}{LCL_{5\%}}\right)_{Total}} \qquad (4.1)$$

where SI = the sensitivity index; the higher the sensitivity index, the more sensitive the parameter,

$UCL_{95\%}$ = the 95th percentile value of the risk (upper confidence limit),

and $LCL_{5\%}$ = the 5th percentile value of the risk (lower confidence limit).

The ratio of the UCL and the LCL for the lifetime cancer risk when all parameters are allowed to vary is the denominator in equation 4.1. The ratio of the UCL and the LCL for the lifetime cancer risk when the parameter being studied is held constant and the others are allowed to vary is the numerator in equation 4.1.

The sensitivity indices for the total risk in reach 3 (the location with the most uncertain fish concentrations) and the total risk in reach 13 (the location containing all three contaminants with the least uncertain fish concentrations) are presented in Table 4.4. The sensitive parameters are ranked according to priority for further study.

The four most sensitive parameters for the total lifetime cancer risk from reaches 3 and 13 are consistent. These parameters are the number of fish-meals per day, the risk conversion factor, the exposure duration, and the amount of fish eaten per fish-meal. For reach 13, the overall estimate of uncertainty was also moderately sensitive to the Cs-137 concentration.

These results indicate that the uncertainty involved with the excess lifetime cancer risk from the human ingestion of fish in the Clinch River/Watts Bar Reservoir system would be most effectively lowered by taking local surveys of consumption patterns, etc., rather than further sampling of fish. Such surveys would aid in the development of more definitive estimates of subjective confidence intervals for the number of fish-meals per day, the exposure duration, and the amount of fish eaten per fish-meal. For example, if the number of fish-meals per day were known exactly, then the uncertainty in the overall excess lifetime cancer risk would be decreased by almost a factor of two. If the number

Table 4.4. Results obtained for the sensitivity analysis performed for excess lifetime risk from radionuclide contamination associated with the human ingestion of fish from the Clinch River/Watts Bar Reservoir system.

| Reach | Sensitive Parameter | Sensitivity Index | Rank |
|---|---|---|---|
| #3 Total Risk | **$F_m$: meal/day** | **0.53** | **1** |
| | ***RCF*: risk/Sv** | **0.50** | **1** |
| | ***ED*: years** | **0.40** | **3** |
| | **$M_w$: kg/meal** | **0.18** | **4** |
| | Sr-90 Conc. | 0.08 | |
| | Co-60 Conc. | 0.07 | |
| | Cs-137 *DCF* | 0.07 | |
| | *EF*: days/yr | 0.05 | |
| | Cs-137 Conc. | 0.04 | |
| | Co-60 *DCF* | -0.02 | |
| | Sr-90 *DCF* | -0.13 | |
| #13 Total Risk | **$F_m$: meal/day** | **0.54** | **1** |
| | ***RCF*: risk/Sv** | **0.51** | **1** |
| | ***ED*: years** | **0.41** | **3** |
| | **$M_w$: kg/meal** | **0.23** | **4** |
| | **Cs-137 Conc.** | **0.22** | **4** |
| | *EF*: days/yr | 0.11 | |
| | Sr-90 Conc. | 0.11 | |
| | Co-60 Conc. | 0.10 | |
| | Co-60 *DCF* | 0.03 | |
| | Cs-137 *DCF* | 0.03 | |
| | Sr-90 *DCF* | -0.10 | |

of fish-meals per day, the exposure duration, and the amount of fish eaten per meal were known exactly, then the uncertainty in the overall excess lifetime cancer risk would be decreased by a factor of almost 3.

The uncertainty in the overall excess lifetime cancer risk is also very sensitive to the risk conversion factor. However, until better estimates of the risk at high dose and high dose rate exposures and/or significant data at low dose and low dose rate exposures are obtained, it is unlikely that the uncertainty factor for the risk conversion factor will be lowered.

### 4.1.4 Results of Uncertainty and Sensitivity Analysis for Plutonium-239

The median excess lifetime cancer risk per Bq/kg obtained for Pu-239 is 4.0E-06. For 1 Bq/kg of Pu-239, there is 90% confidence that the true but unknown excess lifetime cancer risk is between 4.2E-07 and 3.7E-05.

The results of a sensitivity analysis for Pu-239 are summarized below. The most sensitive parameter for the overall estimate of uncertainty is the dose conversion factor for Pu-239. If the DCF were known exactly, the overall uncertainty in the excess lifetime cancer risk could be reduced by a factor of almost three. The other sensitive parameters for the excess lifetime risk for Pu-239 are the number of fish-meals/day and the risk conversion factor, similar to the results from the sensitivity analyses performed for the Clinch River/Watts Bar Reservoir fish data.

| | Sensitive Parameter | SI | Rank |
|---|---|---|---|
| Pu-239 | **Pu-239 *DCF*** | **0.71** | **1** |
| | **$F_m$: meal/day** | **0.43** | **2** |
| | ***RCF*: risk/Sv** | **0.25** | **3** |
| | *ED*: years | 0.03 | |
| | *EF*: days/yr | -0.09 | |
| | $M_W$: kg/meal | -0.18 | |

## 4.2 Analysis for Non-Carcinogenic Chemicals

### 4.2.1 Subjective Confidence Intervals Obtained from the Uncertainty Analysis

The assumptions used for the uncertainty analysis for non-carcinogens are discussed in Section 3.0. These assumptions were used to find the median, the lower 5% subjective confidence limit (LCL), and the upper 95% subjective confidence limit (UCL) for the health hazard associated with human ingestion of MeHg and chlordane in fish from the Clinch River/Watts Bar Reservoir system. These values were obtained by using 500 iterations of LHS. Subjective confidence intervals were obtained for each estimated arithmetic mean concentration. Subjective confidence intervals for the total *HI* were obtained for each reach or section of the river system.

The subjective confidence intervals are summarized in Table 4.5. The median *HQ* for MeHg in reach 1, for instance, is 1.2E-01; there is 90% confidence that the true but unknown *HQ* lies between 1.7E-02 and 9.5E-01 (or there is 95% confidence that the true but unknown value does not exceed 9.5E-01). The median *HQ* for chlordane in reach 1 is 4.7E-01, and there is 90% confidence that

Table 4.5. Results obtained for the hazard quotients from select carcinogenic chemical contaminants in the Clinch River/Watts Bar Reservoir system from 500 iterations of LHS using the assumptions presented in Section 3.0.

| Reach | Contaminant | 5% Subjective Confidence Limit | Median | 95% Subjective Confidence Limit |
|---|---|---|---|---|
| 1 | MeHg | 1.7E-02 | 1.2E-01 | 9.5E-01 |
| | chlordane | 3.1E-02 | 4.7E-01 | 5.4E+00 |
| | Total *HI*[a] | 7.6E-02 | 6.8E-01 | 6.1E+00 |
| 2 | MeHg | 2.2E-02 | 1.7E-01 | 1.2E+00 |
| | chlordane | 2.3E-02 | 3.2E-01 | 4.5E+00 |
| | Total *HI*[a] | 8.4E-02 | 6.4E-01 | 5.2E+00 |
| 3 | MeHg | 1.3E-01 | 8.9E-01 | 6.5E+00 |
| | chlordane | 4.5E-02 | 6.6E-01 | 7.4E+00 |
| | Total *HI*[a] | 3.4E-01 | 2.2E+00 | 1.2E+01 |
| 4 | MeHg | 7.4E-02 | 5.0E-01 | 4.1E+00 |
| | chlordane | 4.8E-02 | 6.1E-01 | 7.9E+00 |
| | Total *HI*[a] | 2.3E-01 | 1.5E+00 | 9.9E+00 |
| 5 | MeHg | 2.0E-02 | 1.5E-01 | 1.2E+00 |
| | chlordane | 1.9E-02 | 2.9E-01 | 3.5E+00 |
| | Total *HI*[a] | 8.1E-02 | 5.7E-01 | 3.9E+00 |
| 10 | MeHg | 5.6E-02 | 4.1E-01 | 3.1E+00 |
| | chlordane | 8.9E-03 | 1.2E-01 | 1.4E+00 |
| | Total *HI*[a] | 1.1E-01 | 7.1E-01 | 3.9E+00 |
| 13 | MeHg | 4.9E-02 | 3.9E-01 | 2.8E+00 |
| | chlordane | 5.9E-02 | 8.0E-01 | 1.0E+01 |
| | Total *HI*[a] | 2.0E-01 | 1.6E+00 | 1.1E+01 |

a The values for the total risk might not be directly additive because of the random selection process in the calculation.

the true but unknown *HQ* lies between 3.1E-02 and 5.4E+00. The median value for the total *HI* from reach 1 is 6.8E-01, and there is 90% confidence that the true but unknown total *HI* for reach 1 lies between 7.6E-02 and 6.1E+00. The uncertainty in the estimates of the *HI* is log-normally distributed; therefore, the most likely *HI* to the maximally exposed individual occurs near the LCL.

The highest potential *HI* occurs in reaches 3, 4, and 13, with medians of 2.2, 1.5, and 1.6, respectively. There is 95% confidence, given the assumptions used in this analysis, that the true but unknown total *HI* from non-carcinogens in reaches 3, 4, and 13 does not exceed 12.1, 9.9, and 11.0, respectively. The primary source of these *HIs* appears to be chlordane; however, the large uncertainty associated with chlordane's *RfD* may be contributing to the conservative UCL. The *HIs* from MeHg are also substantial in these reaches.

The wide ranges of the *HQs* for chlordane in all of the reaches are primarily due to the large uncertainty associated with its *RfD*. The subjective confidence intervals given in Table 4.5 provide valuable information that can be used to guide decision-making. For example, if a 5% lower confidence limit is above a regulatory standard of concern, then remediation is most likely needed. If the 95% upper confidence limit is below the standard, remediation is most likely not required. If the 95% upper confidence limit is above the standard, but the 50$^{th}$ percentile is below the standard (Reaches 1, 2, 5, and 10), further study should be recommended for those parameters which dominate the overall uncertainty. However, if the 50$^{th}$ percentile is above the standard (Reaches 3, 4, and 13), further study may still be recommended, although under some circumstances it may be useful to proceed with remediation.

### 4.2.2 Comparison of Results Obtained with the Uncertainty Analysis with those Presented as Screening Calculations

The results of conservative and nonconservative screening for the *HQs* for adults from MeHg and chlordane in fish are summarized in Table 4.6 (Cook et al., 1992).

The screening calculations (Cook et al., 1992) are within the subjective confidence intervals obtained in the uncertainty analysis. The nonconservative estimates of the *HQs* are very similar to the LCLs. The conservative estimates of the *HQs* are consistently less than the UCLs. The most significant differences between the conservative estimates and the UCL values are for the chlordane calculations. This difference primarily results from the large uncertainty associated with the reference dose. The conservative estimates for the *HQs* resulting from MeHg are also significantly different from the UCLs. This indicates that the assumptions used for the conservative screening of non-carcinogens may need to be re-evaluated.

### 4.2.3 Results of the Sensitivity Analysis

A sensitivity analysis was performed for each "Total Hazard Index". As previously discussed, for each *HI* under evaluation, each parameter in turn was held constant while all others were allowed to vary. In order to obtain an accurate comparison of the "sensitive study" risk and the total *HI*, 500 iterations of LHS with an initial seed value of 1.0 were used. A sensitivity index (SI) was calculated for each parameter by using equation 4.1.

The sensitivity indices for the total *HI* in reach 2 (the location with the most uncertain fish concentrations) and the total *HI* in reach 3 (the location with

Table 4.6. Summary of screening results for the *HQ*s for adults from MeHg and chlordane in fish as reported in Cook et al. (1992).

| Reach | Element | 5% Subj. Confidence Limit | Nonconservative Screening Estimate | 95% Subj. Confidence Limit | Conservative Screening Estimate |
|---|---|---|---|---|---|
| 1 | MeHg | 1.7E-02 | 1.6E-02 | 9.5E-01 | 2.5E-01 |
| | chlordane | 3.1E-02 | 2.5E-02 | 5.4E+00 | 1.6E+00 |
| | Total | 7.6E-02 | 4.1E-02 | 6.1E+00 | 1.9E+00 |
| 2 | MeHg | 2.2E-02 | 1.9E-02 | 1.2E+00 | 4.1E-01 |
| | chlordane | 2.3E-02 | 1.7E-02 | 4.5E+00 | 1.2E+00 |
| | Total | 8.4E-02 | 3.6E-02 | 5.2E+00 | 1.6E+00 |
| 3 | MeHg | 1.3E-01 | 1.3E-01 | 6.5E+00 | 1.8E+00 |
| | chlordane | 4.5E-02 | 6.0E-02 | 7.4E+00 | 1.1E+00 |
| | Total | 3.4E-01 | 1.9E-01 | 1.2E+01 | 2.9E+00 |
| 4 | MeHg | 7.4E-02 | 5.6E-02 | 4.1E+00 | 1.2E+00 |
| | chlordane | 4.8E-02 | 4.3E-02 | 7.9E+00 | 1.5E+00 |
| | Total | 2.3E-01 | 9.9E-02 | 9.9E+00 | 2.7E+00 |
| 5 | MeHg | 2.0E-02 | 2.0E-02 | 1.2E+00 | 3.2E-01 |
| | chlordane | 1.9E-02 | 1.6E-02 | 3.5E+00 | 6.8E-01 |
| | Total | 8.1E-02 | 3.6E-02 | 3.9E+00 | 1.0E+00 |
| 10 | MeHg | 5.6E-02 | 5.7E-02 | 3.1E+00 | 8.4E-01 |
| | chlordane | 8.9E-03 | 1.2E-02 | 1.4E+00 | 2.4E-01 |
| | Total | 1.1E-01 | 6.9E-02 | 3.9E+00 | 1.1E+00 |
| 13 | MeHg | 4.9E-02 | 3.2E-02 | 2.8E+00 | 1.1E+00 |
| | chlordane | 5.9E-02 | 2.1E-02 | 1.0E+01 | 1.1E+00 |
| | Total | 2.0E-01 | 5.3E-02 | 1.1E+01 | 2.2E+00 |

the least uncertain fish concentrations) are presented in Table 4.7. The sensitive parameters are ranked according to priority for further study.

The two most sensitive parameters for the total *HI* from reaches 2 and 3 are consistent. These parameters are the *RfD* for chlordane and the number of fish-meals per day. For reach 2, the overall estimate of uncertainty was moderately sensitive to the chlordane concentration, and for reach 3, the overall estimate of uncertainty was moderately sensitive to the MeHg *RfD*.

These results indicate that the uncertainty involved with the excess lifetime cancer risk from the human ingestion of fish in the Clinch River/Watts Bar Reservoir system would be most effectively lowered by taking local surveys of consumption patterns, etc., rather than further sampling of fish and by further analysis of the *RfDs* for chlordane and MeHg.

## 4.3 Analysis for Carcinogenic Chemicals

### 4.3.1 Subjective Confidence Intervals Obtained from the Uncertainty Analysis

The assumptions used for this uncertainty analysis are discussed in Section 3.0. As in the previous two sections, these assumptions were used to find the median, the lower 5% subjective confidence limit (LCL), and the upper 95% subjective confidence limit (UCL) for the excess lifetime cancer risk associated with human ingestion of fish which are contaminated with PCBs (Aroclor-1254 and -1260) and chlordane from the Clinch River/Watts Bar Reservoir system. These values were obtained by using 500 iterations of LHS. Subjective confidence intervals were obtained for each estimated arithmetic mean

Table 4.7. Results obtained for the sensitivity analysis performed for the *HI* from select non-carcinogens associated with the human ingestion of fish from the Clinch River/Watts Bar Reservoir system.

| Reach | Sensitive Parameter | Sensitivity Index | Rank |
|---|---|---|---|
| #2 Total *HI* | **chlordane *RfD*** | **0.71** | **1** |
| | **$F_m$: meal/day** | **0.30** | **2** |
| | **chlordane Conc.** | **0.14** | **3** |
| | MeHg *RfD* | 0.07 | |
| | *BM*: kgs | -0.09 | |
| | MeHg Conc. | -0.13 | |
| | *EF*: days/yr | -0.13 | |
| | $M_W$: kg/meal | -0.23 | |
| #3 Total *HI* | **chlordane *RfD*** | **0.52** | **1** |
| | **$F_m$: meal/day** | **0.46** | **2** |
| | **MeHg *RfD*** | **0.36** | **3** |
| | MeHg Conc. | -0.01 | |
| | *BM*: kgs | -0.04 | |
| | chlordane Conc. | -0.12 | |
| | *EF*: days/yr | -0.12 | |
| | $M_W$: kg/meal | -0.18 | |

concentration. Subjective confidence intervals for the total lifetime cancer risk for these contaminants were obtained for each reach or section of the river system.

The subjective confidence intervals are summarized in Table 4.8. The median risk for Aroclor-1254 in reach 1, for instance, is 3.6E-05; there is 90% confidence that the true but unknown risk lies between 5.1E-06 and 2.0E-04 (or there is 95% confidence that the true but unknown value does not exceed 2.0E-04). The median risk for Aroclor-1260 in reach 1 is 3.3E-04, and there is 90% confidence that the true but unknown risk lies between 5.3E-05 and 1.3E-03. The median risk for chlordane in reach 1 is 7.5E-06, and there is 90% confidence that the true but unknown risk lies between 1.2E-06 and 5.4E-05. The median value for the total risk from reach 1 is 3.8E-04, and there is 90% confidence that the true but unknown total risk for reach 1 lies between 7.4E-05 and 1.4E-03. The uncertainty in the estimates of the excess lifetime cancer risk is log-normally distributed; therefore, the most likely excess lifetime cancer risk to the maximally exposed individual occurs near the LCL.

The highest potential excess lifetime cancer risk occurs in reaches 2 and 4, with medians of 7.2E-04 and 7.3E-04, respectively. There is 95% confidence, given the assumptions used in this analysis, that the true but unknown total risk from carcinogenic chemicals in reaches 2 and 4 does not exceed 2.9E-03 and 3.7E-03, respectively. The primary source of this risk is from Aroclor-1260.

Table 4.8. Results obtained for the excess lifetime cancer risk from select carcinogenic chemical contaminants in the Clinch River/Watts Bar Reservoir system from 500 iterations of LHS using the assumptions presented in Section 3.0.

| Reach | Contaminant | 5% Subjective Confidence Limit | Median | 95% Subjective Confidence Limit |
|---|---|---|---|---|
| 1 | Aroclor-1254 | 5.1E-06 | 3.6E-05 | 2.0E-04 |
| | Aroclor-1260 | 5.3E-05 | 3.3E-04 | 1.3E-03 |
| | chlordane | 1.2E-06 | 7.5E-06 | 5.4E-05 |
| | Total Risk[a] | 7.4E-05 | 3.8E-04 | 1.4E-03 |
| 2 | Aroclor-1254 | 1.8E-05 | 1.1E-04 | 6.0E-04 |
| | Aroclor-1260 | 9.7E-05 | 5.6E-04 | 2.2E-03 |
| | chlordane | 6.6E-07 | 5.3E-06 | 3.5E-05 |
| | Total Risk[a] | 1.5E-04 | 7.2E-04 | 2.9E-03 |
| 3 | Aroclor-1254 | 1.1E-05 | 6.4E-05 | 2.9E-04 |
| | Aroclor-1260 | 6.4E-05 | 3.5E-04 | 1.4E-03 |
| | chlordane | 1.6E-06 | 1.1E-05 | 5.2E-05 |
| | Total Risk[a] | 9.8E-05 | 4.3E-04 | 1.6E-03 |
| 4 | Aroclor-1254 | 1.1E-05 | 6.2E-05 | 3.4E-04 |
| | Aroclor-1260 | 9.9E-05 | 6.2E-04 | 3.5E-03 |
| | chlordane | 1.5E-06 | 1.2E-05 | 6.0E-05 |
| | Total Risk[a] | 1.4E-04 | 7.3E-04 | 3.7E-03 |
| 5 | Aroclor-1254 | 8.7E-06 | 7.1E-05 | 3.5E-04 |
| | Aroclor-1260 | 5.7E-05 | 3.4E-04 | 1.6E-03 |
| | chlordane | 7.4E-07 | 5.2E-06 | 2.4E-05 |
| | Total Risk[a] | 1.0E-04 | 4.3E-04 | 1.9E-03 |
| 10 | Aroclor-1260 | 3.5E-06 | 2.4E-05 | 1.2E-04 |
| | chlordane | 2.8E-07 | 2.3E-06 | 8.8E-06 |
| | Total Risk[a] | 4.6E-06 | 2.7E-05 | 1.3E-04 |
| 13 | Aroclor-1260 | 3.3E-05 | 2.0E-04 | 8.0E-04 |
| | chlordane | 2.1E-06 | 1.5E-05 | 7.0E-05 |
| | Total Risk[a] | 4.0E-05 | 2.2E-04 | 8.6E-04 |

a The values for the total risk might not be directly additive because of the random selection process in the calculation.

The subjective confidence intervals given in Table 4.8 provide valuable information that can be used to guide decision-making. For example, if a 5% lower confidence limit is above a regulatory standard of concern (Reaches 2, 4, and 5), then remediation is most likely needed. If the 95% upper confidence limit is below the standard, remediation is most likely not required. If the 95% upper confidence limit is above the standard, but the 50$^{th}$ percentile is below the standard (Reach 10), further study should be recommended for those parameters which dominate the overall uncertainty. However, if the 50$^{th}$ percentile is above the standard (Reaches 1, 3, and 13), further study may still be recommended, although under some circumstances it may be useful to proceed with remediation.

### 4.3.2 Comparison of Results Obtained with the Uncertainty Analysis with those Presented as Screening Calculations

The results of conservative and nonconservative screening for the excess lifetime cancer risk to adults from select carcinogenic chemicals in fish are summarized in Table 4.9 (Cook et al., 1992).

The screening calculations (Cook et al., 1992) are within the subjective confidence intervals obtained in the uncertainty analysis. The nonconservative and conservative estimates of the excess lifetime cancer risk are very similar to the LCLs and UCLs, respectively. The non-conservative and conservative estimates for Aroclor-1254 are somewhat higher than the LCLs and UCLs obtained in the uncertainty analysis. This difference primarily results because of the different mode used to represent the subjective probability distribution for Aroclor-1254's *RfD*.

Table 4.9. Summary of screening results for the excess lifetime cancer risk to adults from select carcinogenic chemicals in fish as reported in Cook et al. (1992).

| Reach | Element | 5% Subj. Confidence Limit | Nonconservative Screening Estimate | 95% Subj. Confidence Limit | Conservative Screening Estimate |
|---|---|---|---|---|---|
| 1 | Aroclor-1254 | 5.1E-06 | 1.2E-05 | 2.0E-04 | 3.0E-04 |
| | Aroclor-1260 | 5.3E-05 | 8.8E-05 | 1.3E-03 | 1.3E-03 |
| | chlordane | 1.2E-06 | 9.9E-07 | 5.4E-05 | 2.7E-05 |
| | Total | 7.4E-05 | 1.0E-04 | 1.4E-03 | 1.6E-03 |
| 2 | Aroclor-1254 | 1.8E-05 | 4.3E-05 | 6.0E-04 | 1.1E-03 |
| | Aroclor-1260 | 9.7E-05 | 2.4E-04 | 2.2E-03 | 2.1E-03 |
| | chlordane | 6.6E-07 | 6.9E-07 | 3.5E-05 | 2.0E-05 |
| | Total | 1.5E-04 | 2.8E-04 | 2.9E-03 | 3.2E-03 |
| 3 | Aroclor-1254 | 1.1E-05 | 3.5E-05 | 2.9E-04 | 2.8E-04 |
| | Aroclor-1260 | 6.4E-05 | 1.4E-04 | 1.4E-03 | 1.1E-03 |
| | chlordane | 1.6E-06 | 2.4E-06 | 5.2E-05 | 1.9E-05 |
| | Total | 9.8E-05 | 1.8E-04 | 1.6E-03 | 1.4E-03 |
| 4 | Aroclor-1254 | 1.1E-05 | 1.9E-05 | 3.4E-04 | 4.0E-04 |
| | Aroclor-1260 | 9.9E-05 | 1.5E-04 | 3.5E-03 | 4.1E-03 |
| | chlordane | 1.5E-06 | 1.7E-06 | 6.0E-05 | 2.4E-05 |
| | Total | 1.4E-04 | 1.7E-04 | 3.7E-03 | 4.5E-03 |
| 5 | Aroclor-1254 | 8.7E-06 | 2.1E-05 | 3.5E-04 | 4.8E-04 |
| | Aroclor-1260 | 5.7E-05 | 1.2E-04 | 1.6E-03 | 1.3E-03 |
| | chlordane | 7.4E-07 | 6.4E-07 | 2.4E-05 | 1.1E-05 |
| | Total | 1.0E-04 | 1.4E-04 | 1.9E-03 | 1.8E-03 |
| 10 | Aroclor-1260 | 3.5E-06 | 7.8E-06 | 1.2E-04 | 1.1E-04 |
| | chlordane | 2.8E-07 | 5.0E-07 | 8.8E-06 | 3.9E-06 |
| | Total | 4.6E-06 | 8.3E-06 | 1.3E-04 | 1.1E-04 |
| 13 | Aroclor-1260 | 3.3E-05 | 1.0E-04 | 8.0E-04 | 5.8E-04 |
| | chlordane | 2.1E-06 | 8.2E-07 | 7.0E-05 | 1.8E-05 |
| | Total | 4.0E-05 | 1.0E-04 | 8.6E-04 | 6.0E-04 |

### 4.3.3 Results of the Sensitivity Analysis

A sensitivity analysis was performed for each "Total Lifetime Cancer Risk". As previously described, for each risk under evaluation, each parameter in turn was held constant while all others were allowed to vary. In order to obtain an accurate comparison of the "sensitive study" risk and the total risk, 500 iterations of LHS with an initial seed value of 1.0 were used. A sensitivity index (SI) was calculated for each parameter by using equation 4.1.

The sensitivity indices for the total risk in reach 1 (the location with the most uncertain fish concentrations) and the total risk in reach 3 (the location with the least uncertain fish concentrations) are presented in Table 4.10. The sensitive parameters are ranked according to priority for further study.

The two most sensitive parameters for the total lifetime cancer risk from select carcinogenic chemicals for reaches 1 and 3 are consistent. These parameters are the number of fish-meals per day and the exposure duration. For reach 1, the overall estimate of uncertainty was moderately sensitive to the Aroclor-1260 concentration and the PCB slope factors.

These results agree with the previous two sections in that the uncertainty involved with the excess lifetime cancer risk from the human ingestion of fish from the Clinch River/Watts Bar Reservoir system would be most effectively lowered by taking local surveys of consumption patterns, etc., rather than further sampling of fish.

Table 4.10. Results obtained for the sensitivity analysis performed for excess lifetime risk from select carcinogenic chemicals associated with the human ingestion of fish from the Clinch River/Watts Bar Reservoir system.

| Reach | Sensitive Parameter | Sensitivity Index | Rank |
|---|---|---|---|
| #1 Total Risk | **$F_m$: meal/day** | **0.41** | **1** |
| | ***ED*: years** | **0.26** | **2** |
| | **Aroclor-1260 *SF*** | **0.15** | **3** |
| | **Aroclor-1260 Conc.** | **0.11** | **3** |
| | **Aroclor-1254 *SF*** | **0.11** | **3** |
| | Aroclor-1254 Conc. | 0.06 | |
| | chlordane *SF* | 0.03 | |
| | *BM*: kgs | 0.02 | |
| | chlordane Conc. | -0.03 | |
| | *EF*: days/yr | -0.14 | |
| | $M_W$: kg/meal | -0.14 | |
| #3 Total Risk | **$F_m$: meal/day** | **0.49** | **1** |
| | ***ED*: years** | **0.24** | **2** |
| | Aroclor-1260 *SF* | 0.07 | |
| | Aroclor-1260 Conc. | -0.01 | |
| | chlordane Conc. | -0.02 | |
| | Aroclor-1254 Conc. | -0.08 | |
| | Aroclor-1254 *SF* | -0.08 | |
| | *EF*: days/yr | -0.11 | |
| | *BM*: kgs | -0.13 | |
| | $M_W$: kg/meal | -0.14 | |
| | chlordane *SF* | -0.16 | |

# 5.0 Summary

## 5.1 Radionuclides

A quantitative uncertainty analysis on the excess lifetime cancer risk resulting from human ingestion of radionuclide-contaminated fish from the Clinch River/Watts Bar Reservoir system was performed. A potential exposure to the Co-60, Sr-90, and Cs-137 concentrations present in fish results in a 95% UCL of 1.0E-06 lifetime cancer risk for most reaches. However, the most likely values for the lifetime cancer risk are 1.0E-07 or below. The only exception is for reach 2, where the Cs-137 contamination drives the 95% UCL to 1.7E-05.

The results of the uncertainty analysis were compared to the screening estimates of the lifetime cancer risk (Cook et al., 1992). The nonconservative and conservative screening estimates for the radionuclides were within the 90% subjective confidence intervals obtained from the uncertainty analysis. However, the "conservative" estimates, in some cases, were not as conservative as the 95% subjective confidence interval. These underestimates of the 95% subjective confidence value for the potential lifetime cancer risk are because of the different methods used to convert a potential exposure to the excess risk associated with that exposure. By using a risk conversion factor representing the excess "detriment" to the individual (ICRP, 1991) and a dose conversion factor (ICRP, in press), "slope factors" were derived that in some cases (Sr-90) were up to a factor of three greater than the USEPA's recommended values (USEPA, 1992). Therefore, caution should be used when using the USEPA's slope factors for radionuclides so that the upper bound on the potential excess lifetime cancer risk will not be underestimated.

The sensitivity analysis showed that the overall uncertainty in the estimate of the total lifetime cancer risk from radionuclides is most affected by the number of fish-meals per day, the risk conversion factor, the exposure duration, and the amount of fish consumed per meal. When the total lifetime cancer risk is dominated by the Cs-137 concentration, it becomes a moderately sensitive parameter (ranked fifth). The best way to lower the overall uncertainty in the risk estimate is to perform local surveys to obtain further information about the frequency of fish-meals, the amount of fish consumed per meal, and the extent of time over which the intake persisted.

Because of the extremely low estimates of lifetime cancer risk to an avid fisherman from radionuclides in fish from the Clinch River/Watts Bar Reservoir system, further consideration of the potential lifetime cancer risk from human ingestion of Co-60- and Sr-90-contaminated fish in the Clinch River/Watts Bar Reservoir system should not be warranted. This conclusion is made even for the assumption that fish is consumed in the form of a "fish patty" to maximize the intake of Sr-90, which will be associated with bone as opposed to fish flesh. In the Clinch River/Watts Bar Reservoir system, the primary radionuclide of concern is Cs-137. However, because of the high confidence in the low risk values obtained for Cs-137 and the relatively low sensitivity value for the fish concentrations, further investigation may not be warranted for this contaminant either.

## 5.2 Carcinogenic Chemicals

The primary result from the quantitative uncertainty analysis for the select carcinogenic chemicals was that the 1E-04 recommended level for remediation

was exceeded in every reach at the 95% subjective confidence level, and in every reach except reach 10 at the median level. The highest potential excess lifetime cancer risk from the select chemicals occurs in reaches 2 and 4 with medians of 7.2E-04 and 7.3E-04 and 95% confidence limits of 2.9E-03 and 3.7E-03, respectively. The primary source of this risk was Aroclor-1260.

The values obtained in the uncertainty analysis are very similar to those obtained in the non-conservative and conservative screening estimates (Cook et al., 1992). The values for Aroclor-1254 are slightly higher for the conservative estimates. This observation is probably due to the lower mode used for the Aroclor-1254 *SF* in the uncertainty analysis.

The sensitivity analysis showed that the overall uncertainty in the estimate of the total lifetime cancer risk is most affected by the number of fish-meals per day and the exposure duration. In addition, the *SF*s for Aroclor-1254 and Aroclor-1260 are moderately sensitive parameters. As with the radionuclides, the overall uncertainty in the estimate of the total lifetime cancer risk would be reduced most efficiently with local surveys.

It is evident from this analysis that further study is needed for Aroclor-1254 and Aroclor-1260. Health advisories should be considered for levels of fish consumption in some reaches. Further study of the risk from chlordane should be given a lower priority than Aroclor-1254 and Aroclor-1260 because of the high confidence in its lower risk values as a carcinogen. It appears that the primary contaminant for further consideration is Aroclor-1260.

## 5.3 Non-Carcinogens

Although further study on the excess lifetime cancer risk from chlordane contamination may not be warranted, further evaluation of the risk from toxic effects is warranted for chlordane. The results of the uncertainty analysis performed for the *HI* for MeHg and chlordane indicate that the threshold value of 1.0 is exceeded in all reaches with 95% subjective confidence and in reaches 3, 4, and 13 with median values. The primary source of these excessive levels is chlordane. However, in reaches 3, 4, and 13 the *HQ*s for MeHg are also substantial. The large $HQ$ values for chlordane may be due to the large uncertainty in the estimate of uncertainty in its *RfD*.

When these results were compared to those obtained in the screening procedures (Cook et al., 1992), the non-conservative estimates were similar, but the conservative estimates were consistently less than the 95% subjective confidence levels. This indicates that the screening parameters used for the non-carcinogen conservative estimates may need reevaluation. In agreement with the uncertainty analysis, the screening estimates produced the largest *HI*s in reaches 3, 4, and 13.

The results of the sensitivity analysis were as expected. The most sensitive parameter was the *RfD* for chlordane, followed by the number of fish-meals per day. For the reach with the most uncertain fish concentrations, the chlordane concentration was a moderately sensitive parameter. In addition, when the MeHg contamination is a significant part of the total *HI*, the *RfD* for MeHg becomes a sensitive parameter.

Further study of the chlordane and MeHg contamination in all reaches is warranted. However, the primary parameters of study should be the corresponding *RfD*s. In addition, health advisories may be considered for the ingestion of fish from reaches 3, 4, and 13 until further study is completed.

## 5.4 Comparison to Risks from Background Contamination

The primary purpose of this section is to provide a manner of understanding the magnitude of risks resulting from contamination in the Clinch River/Watts Bar Reservoir system. The risks from background contamination are not intended to trivialize the excess lifetime cancer risk determined in this study but are presented to provide a more complete set of information from which the public can draw their own conclusions. Various risk information for contaminants in east Tennessee that are not a result of specific DOE facilities is presented in Table 5.1. Some of these unavoidable risks that are encountered daily include cosmic radiation, indoor radon, terrestrial gamma rays that are naturally occurring in the materials present on Earth, air pollution, and various chemicals with which man has contaminated the environment.

As one can see from Table 5.1, the excess lifetime cancer risks from background radiation appear to exceed the risk levels obtained from the radionuclide contamination in the fish from the Clinch River/Watts Bar Reservoir system. Although the background PCB risk level is lower than the risks obtained in this study, this is because the background risk was calculated for an average individual who would only be exposed to these concentrations for a period of seven years. The risks from chlordane seem to be similar.

Table 5.1. Life-time Risks from Background Contamination.

| Action | Annual Risk or Dose | Notes | Lifetime Risk | Source |
|---|---|---|---|---|
| Air Pollution | $2.5 \times 10^{-4}$ (100% variability) | Lung cancer deaths from sulfates in the atmosphere. | $2 \times 10^{-2}$ | Crouch and Wilson. 1982. *Risk/Benefit Analysis.* Ballinger Publishing, Cambridge, MA. |
| Natural Background Radiation: | | Cancer Incidence. | | |
| - Radon from earth materials | 2.00 mSv/year | | $1 \times 10^{-2}$ [a] | NCRP No. 93 (1987). |
| - Terrestrial gamma rays | 0.28 mSv/year | | $1 \times 10^{-3}$ [a] | NCRP No. 93 (1987). |
| - Ingestion of natural radionuclides | 0.39 mSv/year | | $2 \times 10^{-3}$ [a] | Clarke and Southwood (1989). |
| - Cosmic Rays[b] | 0.26 mSv/year | | $1 \times 10^{-3}$ [a] | NCRP No. 93 (1987). |
| Cosmic Radiation in Denver, CO (Elevation 5, 280 feet) | 0.5 mSv/year | Cancer Incidence. | $3 \times 10^{-3}$ [a] | NCRP No. 93. and Brill et al. 1982. *Low-level radiation effects.* Society of nuclear medicine, Inc. New York. |
| Background Concentration of PCBs in East Tennessee[c] | $7 \times 10^{-5}$ | Cancer Incidence. | $5 \times 10^{-4}$ | STORET data base. |
| Background Concentration of Chlordane in East Tennessee[c] | $3 \times 10^{-7}$ | Cancer Incidence. | $2 \times 10^{-6}$ | STORET data base. |
| Background Concentration of Arsenic in East Tennessee[c] | $6 \times 10^{-7}$ | Cancer Incidence. | $4 \times 10^{-6}$ | STORET data base. |

[a] A risk conversion factor of 0.07 per Sv (ICRP, 1991) and a 70 year lifetime of exposure was used in converting annual exposure to lifetime risk.

[b] The value used to calculate lifetime risk for cosmic rays was the number given for the exposure at sea level.

[c] For these materials, background concentration refers to the concentration present in fish that is not a result of DOE facilities' operations. These do not occur naturally, but they are evident in background due to various other uses. An intake value of 33 g/day for 7 years averaged over a 70 year lifetime was used to calculate these lifetime risks. An exposure period of 7 years is more representative of an average individual.

# 6.0 Conclusions

A quantitative uncertainty analysis on the risk resulting from human ingestion of select radionuclide- and chemical-contaminated fish from the Clinch River/Watts Bar Reservoir system was performed. Several conclusions resulted from this study.

- Further study is not warranted for the radionuclide contamination in fish located in the Clinch River/Watts Bar Reservoir system due to the low magnitude of the upper bound of the uncertainty in the risk estimate for the maximally exposed individual.

- PCB contamination in the fish does warrant further study. Because the excess lifetime cancer risk estimates exceed a 1E-04 level of concern at the 95% subjective confidence limits and, in most cases, at the median values, health advisories are warranted for limiting the ingestion of the contaminated fish. However, because the only evidence of cancer incidence from PCB exposure is in rats, caution should be used in assessing the importance of this contaminant to the overall excess lifetime cancer risk for humans.

- Further study of the toxic effects from MeHg and chlordane is also recommended (especially in reaches 3, 4, and 13). Although the total risk from these contaminants appears to be driven by chlordane, the reference dose for chlordane is associated with large uncertainty.

- Use of USEPA recommended slope factors for radionuclides may lead to estimates of the excess lifetime risk from certain radionuclides that differ substantially from the estimates using methods recommended by ICRP and NCRP. The dose conversion factors used in this research were multiplied by the risk conversion factor to obtain comparable "slope factors". The difference between the "slope factors" obtained in this study and the USEPA radionuclide slope factors was evaluated. Differences between these two values were noted for each radionuclide considered in the uncertainty analysis with the greatest difference occurring for Sr-90 (in which the EPA estimate was lower by a factor of 3).

- Although the screening estimates of risk obtained from Cook et al. (1992) and the subjective confidence limits obtained from the uncertainty analysis were similar for the radionuclide and carcinogenic chemicals, the conservative estimate and the 95% subjective confidence limit were not similar for the non-carcinogenic chemicals. In most cases, the conservative screening estimates for the select non-carcinogens were substantially smaller than the 95% subjective UCL. This result indicates that a reevaluation of the parameter values used in the screening estimates for non-carcinogens is warranted.

- Results of the sensitivity analysis indicate that the overall estimate of uncertainty in the total risk estimates would be most efficiently reduced by taking local surveys to obtain better subjective probability distributions for parameters such as the numbers of fish-meals eaten per day and the exposure duration for individuals qualifying as being "maximally exposed".

- The reference dose for chlordane was the most sensitive parameter in the non-carcinogenic assessment.

This study used a quantitative uncertainty analysis to efficiently rank the select contaminants of concern and to determine the most sensitive parameters in the overall uncertainty of the risk. The research demonstrates that uncertainty analysis can be a powerful tool for decision making and provides information that could not be obtained from the standard point estimate approach. If decisions concerning remediation are based on point estimates, and if the true risk is either grossly overestimated or is associated with high uncertainty, the result may be inappropriate remedial efforts or misallocation of limited resources. Therefore, to borrow a phrase from Dr. Adam Finkel (1993), "*a choice made with 'uncertainty-colored glasses' on rather than blinders is a superior choice.*"

# 7.0 Recommendations

Although this research was for the maximally exposed individual, regulators may be interested in a different assessment endpoint (e.g., the 95th percentile of a population). The definition of the assessment endpoint is critical to the uncertainty analysis approach. Although not considered in this study due to the definition of the assessment endpoint, a distinction between stochastic variability and deterministic uncertainty (uncertainty about a true but unknown value) would be necessary if the assessment endpoint was the 95th percentile of the population. Many professionals are still confusing these two issues; therefore, this would be a major area of study for future consideration. In addition, the procedure for eliciting judgment from a group of experts and accounting for this information is another area for further study. Most uncertainty analysis specialists believe that the only way to obtain a defensible estimate of subjective probability distributions is to elicit information from a **group** of experts. Although expert opinion was used to obtain estimates of uncertainty for some of the parameters considered in this research, formal elicitation from a group of experts was not possible due to time and money constraints.

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# Appendices

# Appendix A

## Examples Used to Demonstrate Methods of Uncertainty Analysis

## Appendix A: Examples Used to Demonstrate Methods of Uncertainty Analysis

Please note that the examples presented in this section are entirely hypothetical.

Example 1

An inadvertent release of methyl mercury has contaminated a nearby lake. After further testing, a 95% UCL of the mean concentration of 1.57E-01 mg/kg was obtained for the concentration in the fish. What would be the risk to a maximally exposed individual from eating fish? This is determined by asking: What is the Hazard Quotient for this concentration and what is its interpretation?

*Solution:*

A value of 70 kg (Reference Man) will be used for body mass. Assuming 2 fish meals per week, 50 weeks a year, and 230 grams per fish meal, averaged over one year, one obtains a value of 0.065 kg/day for the daily ingestion rate. Finally, the reference dose value of 3.00E-04 mg/kg-day is obtained from the Health Effects Assessment Summary Tables (USEPA, 1992). Using Equation 1.2 one obtains a Hazard Quotient of 0.49. From this value, one would conclude that it is highly unlikely that an exposure to methyl mercury in fish in this case would warrant remedial action.

Example 2

Upon investigation of a potentially contaminated site, it was discovered that a nearby lake and surrounding sediment were contaminated with methyl-mercury and inorganic mercury, respectively. The 95% UCL values for the concentrations of inorganic mercury in soil is found to be 700 mg/kg, and the 95% UCL for the concentration of methyl-mercury in fish is 3.05E-01 mg/kg.

Which is the most hazardous pathway to a hypothetical maximally exposed individual?

*Solution:*

A summary of the values obtained for this example is provided in Table A.1. The values for the HQs for the two pathways will be compared with each other for two situations: (1) by using EPA's generic equations and (2) by incorporating uncertainty analysis. By manipulating Equation 1.2, the HQ for the fish and soil pathways are calculated below.

$$HQ_{SP} = \frac{C_{soil} \cdot I_{soil} \cdot EF}{BM \cdot RfD_{IM}} = \frac{(700)(1.0E-04)(0.7)}{(70)(3.0E-04)} = 2.33$$

$$HQ_{FP} = \frac{C_{fish} \cdot I_{fish}}{BM \cdot RfD_{MM}} = \frac{(3.05E-01)(6.5E-02)}{(70)(3.0E-04)} = 0.94$$

Table A.1. Information for Example 2.

| Parameter | Distribution | Minimum | Maximum | Average* | Standard Deviation | Units |
|---|---|---|---|---|---|---|
| Fish Concentration | Log-Normal | | | 2.06E-1 | 4.22E-2 | mg/kg |
| Intake of Fish | Log-Uniform | 2.00E-2 | 1.30E-1 | 6.50E-2 | | kg/day |
| Soil Concentration | Log-Normal | | | 3.11E+2 | 1.50E+2 | mg/kg |
| Intake of Soil | Log-Uniform | 5.00E-5 | 2.00E-4 | 1.00E-4 | | kg/day |
| Exposure Frequency | Log-Uniform | 2.70E-1 | 7.00E-1 | 7.00E-1 | | |
| Body Mass | Log-Triangle | 4.50E+1 | 1.20E+2 | 7.00E+1 | | kg |
| Inorganic Mercury RfD | Log-Uniform | 3.00E-4 | 3.00E-2 | 3.00E-4 | | mg/kg-day |
| Methyl Mercury RfD | Log-Triangle | 1.50E-4 | 3.00E-3 | 3.00E-4 | | mg/kg-day |

* The value given for the intake, body mass, and RfD is the most likely value (mode).

From these calculations, one would conclude that the risk to the maximally exposed individual results from the soil ingestion pathway. However, by incorporating the uncertainties for the parameters and by using Monte Carlo simulation, one obtains different results. A summary of the results obtained from the EPA's methods and the uncertainty analysis is provided in Table A.2. The results from the uncertainty analysis imply that the fish ingestion pathway is the source of most of the risk given to the maximally exposed individual. The reversal of the ranking from the EPA calculations is primarily due to the large uncertainty associated with the RfD for inorganic mercury. If the uncertainty of this parameter had not been taken into account, an inaccurate conclusion and possibly an inappropriate course of action would have resulted.

Table A.2. Summary of results obtained for both the generic method and the uncertainty analysis values for the fish and soil pathways.

| | | Uncertainty Analysis Result | | |
|---|---|---|---|---|
| Pathway | Generic EPA Method | 5% | 50% | 95% |
| Fish | 9.4E-1 | 6.5E-2 | 2.9E-1 | 1.2E+0 |
| Soil | 2.3E+0 | 4.0E-3 | 5.4E-2 | 7.2E-1 |

Example 3

The purpose of this example is to study the effect of the correlation between body mass and intake on the Total Cancer Risk and the Total Hazard Index for the situation given in Example 6. First, assume that a minimum correlation of 0.3 has been determined between body mass and intake, and second, compare the results with those obtained with correlations of 0.5 and 0.7.

*Solution:*

In this case, rank correlations are used (Decisioneering, 1992) to account for interdependencies between body mass and intake. As can be seen from Table A.3, where the results are produced from 500 iterations using Latin Hypercube Sampling, the correlation coefficients do not have a dramatic effect on the total risk. The values for the total cancer risk are virtually the same. A slight difference is detected in the 95% upper confidence limits for the Total Hazard Index. The values obtained with a correlation coefficient applied to the analysis are slightly lower than the value calculated without the correlation applied. One reason that the correlation does not have an obvious effect on the results is that the body mass is not a sensitive parameter.

Table A.3. Results obtained for correlations between body mass and intake for the situation described in Example 6.

| | Rank Correlation Coefficient | | |
|---|---|---|---|
| Total Cancer Risk | 0.3 | 0.5 | 0.7 |
| 5% | 8.9E-4 | 9.1E-4 | 9.6E-4 |
| 50% | 3.4E-3 | 3.4E-3 | 3.4E-3 |
| 95% | 1.3E-2 | 1.3E-2 | 1.3E-2 |
| Total Hazard Index | | | |
| 5% | 1.2E-1 | 1.3E-1 | 1.2E-1 |
| 50% | 6.8E-1 | 7.0E-1 | 7.0E-1 |
| 95% | 3.4E+0 | 3.3E+0 | 3.3E+0 |

Example 4

Let us assume, that there has been an accidental spill of methyl-mercury in a nearby lake. Using the technique of variance propagation obtain a 90% confidence interval for the true but unknown Hazard Index to a maximally

exposed individual. After reviewing the literature, available data, and consulting with other experts, the subjective probability distributions obtained for this problem are provided in Table A.4.

Table A.4. Information for Example 4.

| Parameter | Distribution | Minimum | Maximum | Average* | Standard Deviation | Units |
|---|---|---|---|---|---|---|
| Fish Concentration | Log-Normal | | | 7.10E-2 | 3.43E-2 | mg/kg |
| Intake | Log-Uniform | 2.00E-2 | 1.30E-1 | 6.50E-2 | | kg/day |
| Body Mass | Log-Triangle | 4.50E+1 | 1.20E+2 | 7.00E+1 | | kg |
| RfD | Log-Triangle | 1.50E-4 | 3.00E-3 | 3.00E-4 | | mg/kg-day |

* The values given for the intake, body mass, and RfD are the most likely values (modes). The distributions represent subjective confidence about the uncertainty associated with estimating a true but unknown value for each parameter.

*Solution:*

The form of the equation used for this problem is a determined by manipulating Equation 1.2 and is as follows:

$$HQ = C \times I \times (BM)^{-1} \times (RfD)^{-1}.$$

By log-transforming, this equation becomes:

$$\ln(HQ) = \ln(C) + \ln(I) - \ln(BM) - \ln(RfD).$$

Therefore, the logarithmic mean and variance, assuming independence between the parameters, of the HQ is found by using Equations 2.1 and 2.2. The equations given in Appendix B must be utilized to find the mean and variance of logarithms for each of the model parameters. Using Equations B.1 and B.2, one obtains a mean of the logarithms for the fish concentration of -2.75, and the variance of the logarithms for the fish concentration 0.21.

Equations B.3 and B.4, given for the log-uniform distribution, will be used to find μ (the mean of logarithms) and σ2 (the variance of logarithms) for the intake. These values are calculated below:

$$\mu_I = \frac{\ln(0.02) + \ln(0.13)}{2} = -2.98$$

$$\sigma_I^2 = \frac{\left[\left(\frac{0.13}{0.02}\right)\right]^2}{12} = 0.29$$

Equations B.5 and B.6, given for the log-triangular distribution, are the ones that are used for the body mass and RfD. To demonstrate, the μ and σ2 for the body mass are calculated below.

$$\mu_{BM} = \frac{1}{3}[\ln(70) + \ln(120) + \ln(45)] = 4.28$$

$$\sigma_{BM}^2 = \frac{1}{18}\left[\begin{array}{l}(\ln(45))^2 + (\ln(120))^2 - (\ln(45))(\ln(120)) + (\ln(70))^2 - \\ (\ln(70))(\ln(45) + \ln(120))\end{array}\right]$$

$$\sigma_{BM}^2 = 0.04$$

This same process is performed for the RfD from which one obtains a mean value of -7.58 and a variance of 0.41. We are now able to find the mean of logarithms, and thus, the geometric mean of the HQ.

$$\mu_{HQ} = (-2.75) + (-2.98) + (-4.28) + (7.58) = -2.43$$

$$X_{g,HQ} = e^{\mu_{HQ}} = e^{-2.43} = 0.09$$

The variance of the HQ and consequentially the geometric standard deviation of the HQ is found below.

$$\sigma_{HQ}^2 = 0.21 + 0.29 + 0.04 + 0.41 = 0.95$$

$$S_{g,HQ} = e^{\sqrt{\sigma_{HQ}^2}} = e^{\sqrt{0.95}} = 2.65$$

The 95 and 5% subjective confidence limits for a 90% subjective confidence interval are calculated below:

$$X_{95}^{HQ} = X_{g,HQ} \cdot S_{g,HQ}^{1.65} = (0.09)(2.65)^{1.65} = 0.45$$

$$X_5^{HQ} = \frac{X_{g,HQ}}{S_{g,HQ}^{1.65}} = \frac{(0.09)}{(2.65)^{1.65}} = 0.02$$

Therefore, there is high confidence (at a subjective level of 90%) that the HI should lie between 0.02 and 0.45.

Example 5

Use the scenario presented in Example 4, to demonstrate the use of Monte Carlo simulation. What is the risk to the maximally exposed individual with 90% subjective confidence?

*Solution:*

To begin an uncertainty analysis, one must describe the uncertainty about each variable with a subjective probability distribution. This is done through judgment after extensive review of all relevant data. The information presented

in Table A.4 is also utilized as inputs for a Monte Carlo simulation for this problem.

When running a Monte Carlo technique, values are selected at random from each uncertain variable to produce a prediction. This procedure is repeated for a specified number of iterations and forms a distribution of predicted values. A sample of randomly selected values obtained by running LHS for 500 iterations for this problem is provided in Table A.5.

Table A.5. A sample of random values obtained from 500 iterations of LHS for Example 5.

| Sample Number | Fish Concentration mg/kg | Intake kg/day | Body Mass kg | Reference Dose mg/kg/day | Hazard Quotient |
|---|---|---|---|---|---|
| 1 | 1.01E-01 | 3.40E-02 | 4.71E+01 | 5.25E-04 | 1.38E-01 |
| 2 | 1.14E-01 | 1.19E-01 | 7.68E+01 | 5.64E-04 | 3.15E-01 |
| 3 | 8.11E-02 | 1.05E-01 | 6.78E+01 | 1.76E-04 | 7.10E-01 |
| 4 | 6.51E-02 | 3.63E-02 | 7.50E+01 | 3.00E-04 | 1.05E-01 |
| . | . | . | . | . | . |
| . | . | . | . | . | . |
| . | . | . | . | . | . |
| 499 | 9.40E-02 | 9.21E-02 | 7.04E+01 | 2.71E-03 | 4.53E-02 |
| 500 | 8.60E-02 | 2.66E-02 | 8.15E+01 | 8.96E-04 | 3.13E-02 |

This process yields a subjective probability distribution for the Hazard Quotient from which a quantitative representation of the Hazard Quotient can be formed. Figure A.1 contains the result as obtained from Crystal Ball (Decisioneering, 1992) for the risk after 500 iterations using LHS.

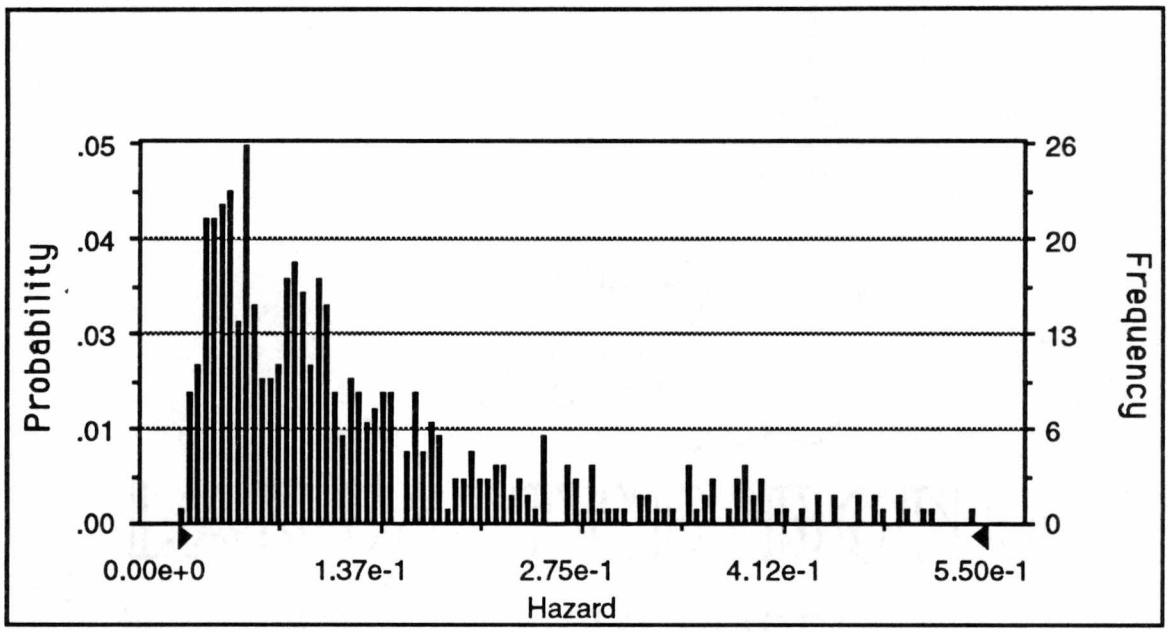


Figure A.1. Subjective probability distribution of the Hazard Quotient for Example 5.

From this Monte Carlo simulation a 90% confidence interval of [ 1.70E-2, 4.17E-1] is obtained. This implies that after taking into account the uncertainties on the parameters, one is highly confident (at a subjective level of 90%) that the true HI should lie between 1.70E-2 and 4.17E-1. Since the 95% upper confidence limit is still below the value of unity, one would conclude that there is little potential for risk to the maximally exposed individual for this scenario and remediation is not warranted.

Example 6

Let us assume that as the result of waste management practices, a mixture of contaminants is released inadvertently to the environment. Eventually through various pathways, this contamination is transported to aquatic systems such as rivers and lakes where various fish and biota are exposed. After further investigation, it is discovered that the contaminants released were Aroclor-1254,

Aroclor-1260, chlordane, and methyl-mercury. Suppose that a fisherman catches some contaminated fish and eventually, eats them. The assessment problem is as follows: What is the health risk to the maximally exposed individual?

In order to perform this risk assessment, a form of Equations 1.2 and 1.3 will be used. To quantify the uncertainty associated with each of the parameters introduced in these equations, one must derive with the use of a considerable amount of judgment subjective probability distributions from very limited sets of data and other relevant facts in the published literature. Once these distributions have been specified, one can utilize Monte Carlo techniques to obtain a probability distribution of the Hazard Index and cancer risk. From this propagated distribution, a subjective confidence interval (90%) can be obtained to set limits for a true but unknown result that are useful for decision making.

Table A.6 contains values for the estimates of uncertainty on each of the parameters that would be used in an environmental risk assessment of Aroclor-1254, Aroclor-1260, chlordane, and methyl-mercury in the fish potentially harvested from a contaminated fresh water system.

*Solution:*

The values given in Table A.6 were used to find the median, the lower 5% subjective confidence limit, and the upper 95% subjective confidence limit for the non-carcinogen Hazard Quotient for chlordane and methyl-mercury, and for the cancer risk involved with the given concentrations in fish of Aroclor-1254, Aroclor-1260, and chlordane. These values were obtained by using a Monte Carlo technique with 500 iterations of Latin Hypercube Sampling and are presented in Table A.7. The subjective probability distributions obtained for the total cancer risk and the total Hazard Index are provided in Figure A.2 and A.3, respectively.

Table A.6. Subjective probability distributions specified for the Monte Carlo Analysis of Example 6.

| Chemical | Parameter | Subjective Probability Distribution | Minimum | Maximum | Mean (Mode) | Standard Deviation | Units |
|---|---|---|---|---|---|---|---|
| Aroclor-1254 | Fish Conc. | Log-Normal | 4.00E-03 | 3.79E+00 | 5.34E-01 | 2.26E+00 | mg/kg |
| | Intake | Log-Uniform | 1.65E-02 | 8.25E-02 | | | kg/day |
| | Body Mass | Log-Triangle | 4.50E+01 | 1.20E+02 | 7.00E+01 | | kg |
| | Slope Factor | Triangle | 0.00E+00 | 1.00E+01 | 7.70E+00 | | $(mg/kg\text{-}day)^{-1}$ |
| Aroclor-1260 | Fish Conc. | Log-Normal | 3.19E-01 | 2.29E+00 | 9.75E-01 | 5.16E-1 | mg/kg |
| | Intake | Log-Uniform | 1.65E-02 | 8.25E-02 | | | kg/day |
| | Body Mass | Log-Triangle | 4.50E+01 | 1.20E+02 | 7.00E+01 | | kg |
| | Slope Factor | Triangle | 0.00E+00 | 1.00E+01 | 7.70E+00 | | $(mg/kg\text{-}day)^{-1}$ |
| Chlordane (carcinogen) | Fish Concentration | Log-Normal | 3.96E-02 | 3.06E-01 | 1.27E-01 | 6.98E-02 | mg/kg |
| | Intake | Log-Uniform | 1.65E-02 | 8.25E-02 | | | kg/day |
| | Body Mass | Log-Triangle | 4.50E+01 | 1.20E+02 | 7.00E+01 | | kg |
| | Slope Factor | Triangle | 0.00E+00 | 5.00E+00 | 1.30E+00 | | $(mg/kg\text{-}day)^{-1}$ |
| Chlordane (non-carc) | Fish Concentration | Log-Normal | 3.96E-02 | 3.06E-01 | 1.27E-01 | 6.98E-02 | mg/kg |
| | Intake | Log-Uniform | 2.00E-02 | 1.30E-01 | | | kg/day |
| | Body Mass | Log-Triangle | 4.50E+01 | 1.20E+02 | 7.00E+01 | | kg |
| | RfD | Log-Triangle | 3.00E-05 | 1.90E-03 | 6.00E-05 | | mg/kg-day |
| Methyl Mercury | Fish Concentration | Log-Normal | 2.55E-02 | 1.57E-01 | 7.10E-02 | 3.43E-02 | mg/kg |
| | Intake | Log-Uniform | 2.00E-02 | 1.30E-01 | | | kg/day |
| | Body Mass | Log-Triangle | 4.50E+01 | 1.20E+02 | 7.00E+01 | | kg |
| | RfD | Log-Triangle | 1.50E-04 | 3.00E-03 | 3.00E-04 | | mg/kg-day |

Table A.7. Results obtained from Monte Carlo Simulation Using Values in Table A.6.

| Chemical | Type of Result | Median | 5% Subjective Confidence | 95% Subjective Confidence |
|---|---|---|---|---|
| Total Cancer Risk* | | 3.5E-03 | 9.6E-04 | 1.3E-02 |
| Aroclor-1254 | cancer risk | 3.4E-04 | 6.0E-05 | 7.5E-03 |
| Aroclor-1260 | cancer risk | 2.5E-03 | 5.7E-04 | 8.5E-03 |
| Chlordane | cancer risk | 1.0E-04 | 2.0E-05 | 4.4E-04 |
| Total Hazard Index* | | 6.3E-01 | 1.3E-01 | 3.8E+00 |
| Chlordane | non-carcinogen Hazard Index | 5.3E-01 | 6.7E-02 | 3.2E+00 |
| Methyl Mercury | non-carcinogen Hazard Index | 8.6E-02 | 1.7E-02 | 3.9E-01 |

* Summed over all contaminants.

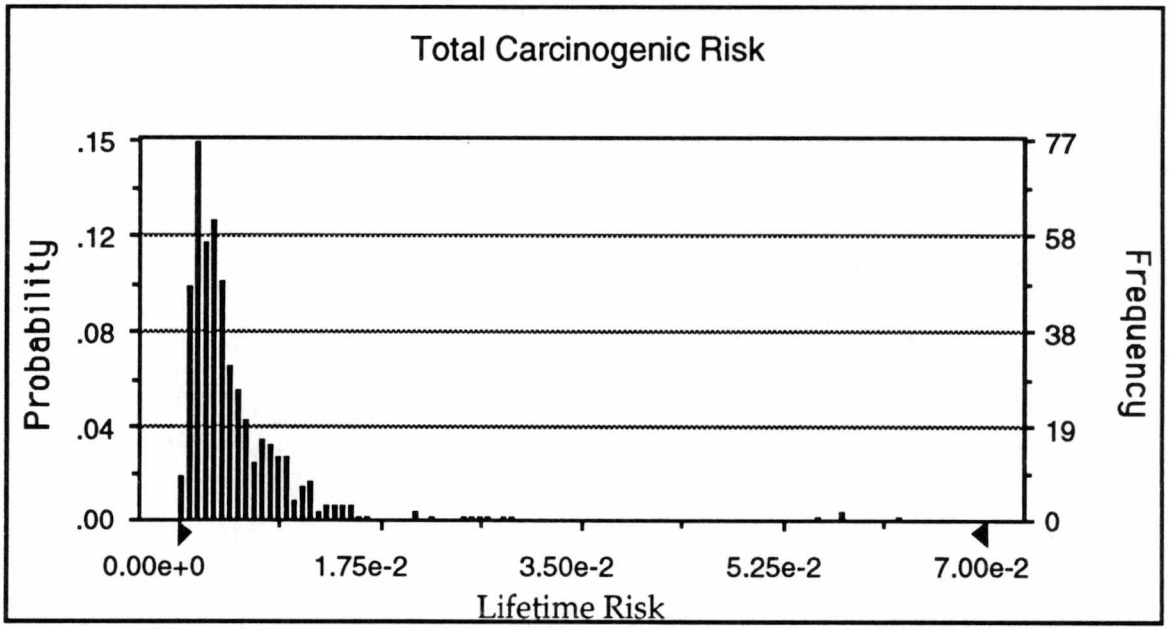


Figure A.2. Results obtained from 500 simulations of LHS for the Total Carcinogenic Risk presented in Table A.7.

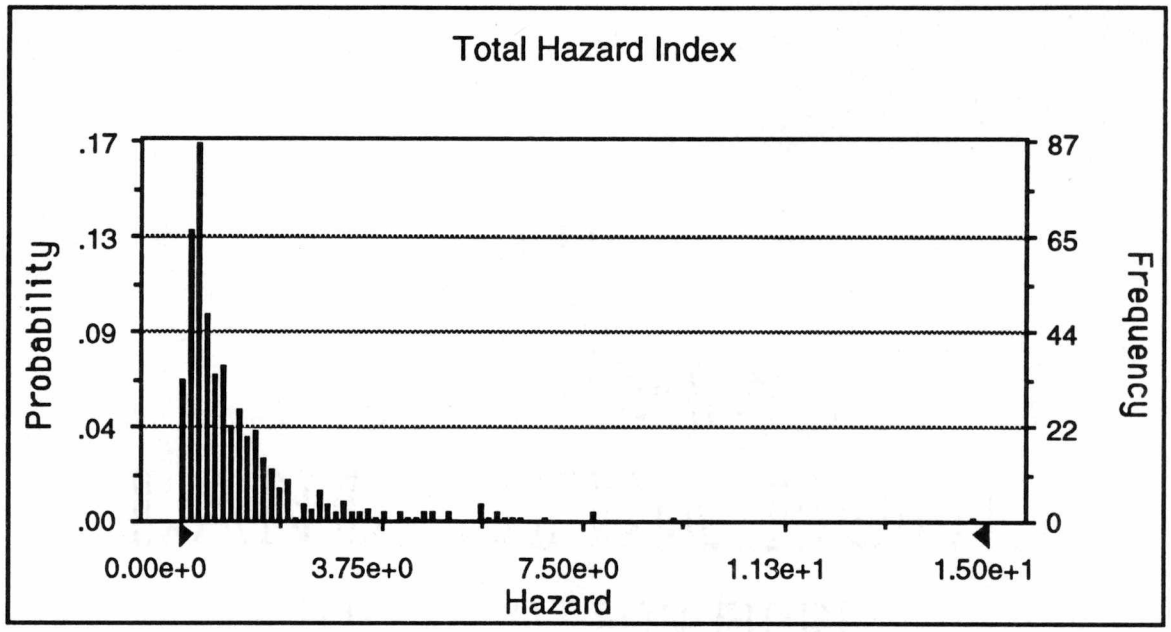


Figure A.3. Results obtained from 500 simulations of LHS for the Total Hazard Index presented in Table A.7.

As one can see from Table A.7, the primary chemical contributing to the total cancer risk is Aroclor-1260, and the chemical contributing the majority of the total Hazard Index is chlordane. By performing a sensitivity analysis, one can determine which parameter has the most affect on the total result. This is done by simply holding a potentially sensitive parameter constant while varying the others. After this process is repeated for all of the parameters of interest, the different results obtained for each parameter are compared and ranked according to the parameters having the most influence on the uncertainty in the result. These parameters are said to be the most sensitive parameters. Descriptions of statistical approaches to sensitivity analysis using regressions of the randomly selected values of the uncertain parameters on the values produced for the model predictions can be found in Iman et al. (1981a and b), IAEA Safety Series No. 100 (1989), and Iman and Helton (1991).

For the total cancer risk, a sensitivity analysis shows that the amount of fish ingestion has the most effect followed by the concentration of Aroclor-1254 in the fish. One might not expect the latter result, but the uncertainty involved with the Aroclor-1254 fish concentration is much greater than that involved with the fish concentration for Aroclor-1260.

For the total Hazard Index, a sensitivity analysis shows that the two parameters that are the most significant contributors to the total uncertainty are the amount of fish ingested and the reference dose for chlordane.

# Appendix B

## Equations for Log-Normal, Log-Uniform, Log-Triangular, Normal, Uniform, and Triangular Distributions

## Appendix B: Equations for Log-Normal, Log-Uniform, Log-Triangular, Normal, Uniform, and Triangular Distributions

The following distributions are suggested for subjective probability distributions in analysis of multiplicative models.

Log-Normal Distribution:

$\mu$ = the mean of the logarithms

$\sigma^2$ = the variance of the logarithms.

However, if you have a situation where you are given only the arithmetic mean and arithmetic variance, then $\mu$ and $\sigma^2$ can be estimated with the following equations (Hoffman and Gardner, 1983):

$$\mu = \ln\left\{\frac{\bar{x}}{\left[1+\left(\frac{s}{\bar{x}}\right)^2\right]^{0.5}}\right\} \qquad \text{(B.1)}$$

$$\sigma^2 = \ln\left[1+\left(\frac{s}{\bar{x}}\right)^2\right] \qquad \text{(B.2)}$$

where $\bar{x}$ = the arithmetic mean of the distribution

and s = the standard deviation of the distribution.

Log-Uniform Distribution (Hoffman and Gardner, 1983):

$$\mu = \frac{[\ln(\min) + \ln(\max)]}{2} \tag{B.3}$$

$$\sigma^2 = \frac{\left[\ln\left(\frac{\max}{\min}\right)^2\right]}{12} \tag{B.4}$$

Asymmetrical Log-Triangular Distribution (Beauchamp, 1991; Johnson and Kotz, 1970):

$$\mu = \frac{1}{3}\left[\ln\left(H^*\right) + \ln(b) + \ln(a)\right] \tag{B.5}$$

$$\sigma^2 = \frac{1}{18}\left\{\begin{array}{l}[\ln(a)]^2 + [\ln(b)]^2 - [\ln(b)][\ln(b)] + \left[\ln(H^*)\right]^2 - \\ \left[\ln(H^*)\right][\ln(a) + \ln(b)]\end{array}\right\} \tag{B.6}$$

where $H^*$ = the mode of the triangular distribution,

b = the maximum of the triangular distribution,

and a = the minimum of the triangular distribution.

The following distributions are suggested for use as subjective probability distributions in analysis of additive models.

Normal Distribution:

The mean value of the normal distribution is simply the value at the 50 percentile. With a normal distribution the median, mode, and mean are the same. The variance of the normal distribution is the second central moment of the variable or the standard deviation squared.

Uniform Distribution:

$$\bar{x} = \frac{(\min + \max)}{2} \tag{B.7}$$

$$s^2 = \frac{\left[\left(\frac{\max}{\min}\right)^2\right]}{12} \tag{B.8}$$

Asymmetrical Triangular Distribution (Beauchamp, 1991; Johnson and Kotz, 1970):

$$\bar{x} = \frac{1}{3}\left(H^* + b + a\right) \tag{B.9}$$

$$s^2 = \frac{1}{18}\left[(a)^2 + (b)^2 - (a)(b) + (H^*)^2 - (H^*)(a+b)\right] \tag{B.10}$$

In addition to these suggested distributions, a few more distributions that one may use are custom designed, Poisson, Weibull, gamma, beta distributions, and any number of discrete distributions (Decisioneering, 1992; Palisade, 1991).

# Vita

Jana Sue Hammonds was born in Harriman, Tennessee on August 20, 1969. When she was eight years old, her family moved to Knoxville, Tennessee. She went to West High School where she graduated as co-salutatorian in May, 1987. The following September she entered The University of Tennessee (UT) at Knoxville. She worked part time at the Y-12 Plant (which is operated by Martin Marietta Energy Systems, Inc.) in Oak Ridge, Tennessee during her junior and senior years at UT. In May, 1991 she received the degree of Bachelor of Science in Nuclear Engineering (GPA 3.64). After taking the summer off, she reentered UT in August, 1991. After working under the direction of Dr. F. Owen Hoffman at the Oak Ridge National Laboratory for one year, she was hired by Dr. Hoffman (who began his own company) as a full time employee of SENES Oak Ridge, Inc. In August, 1991 she received a Master of Science degree in Nuclear Engineering (GPA 4.0).

Her research interests include: (1) analysis of quantitative methods used to perform human health risk assessments for chemicals and radionuclides, (2) application and evaluation of quantitative uncertainty analyses for human health risk assessments, (3) methods of improving and applying internal dosimetry techniques for radionuclides and chemicals beyond the specifications for "reference man," and (4) research regarding new implementations of nuclear pharmaceuticals and positron emission tomography for therapeutic and diagnostic nuclear medicine. Her hobbies include gardening, reading, volleyball, softball, and fishing.