



Risk Assessment Guidance for Superfund:

Volume III - Part A,
Process for Conducting
Probabilistic Risk Assessment





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Risk Assessment Guidance for Superfund: Volume III - Part A, Process for Conducting Probabilistic Risk Assessment

**Office of Emergency and Remedial Response
U.S. Environmental Protection Agency
Washington, DC 20460**



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DISCLAIMER

This document provides guidance to EPA Regions concerning how the Agency intends to exercise its discretion in implementing one aspect of the CERCLA remedy selection process. The guidance is designed to implement national policy on these issues.

Some of the statutory provisions described in this document contain legally binding requirements. However, this document does not substitute for those provisions or regulations, nor is it a regulation itself. Thus, it cannot impose legally-binding requirements on EPA, States, or the regulated community, and may not apply to a particular situation based upon the circumstances. Any decisions regarding a particular remedy selection decision will be made based on the statute and regulations, and EPA decision makers retain the discretion to adopt approaches on a case-by-case basis that differ from this guidance where appropriate.

Interested parties are free to raise questions and objection about the substance of this guidance and the appropriateness of the application of this guidance to a particular situation, and the Agency welcomes public input on this document at any time. EPA may change this guidance in the future.

ABOUT THE REVISION

WHAT IT IS EPA's *Process for Conducting Probabilistic Risk Assessment* is an update of the 1989 *Risk Assessment Guidance for Superfund (RAGS)*. It is Volume III, an update to the existing two-volume set of RAGS. Volume III: Part A provides policy and guidance on conducting probabilistic risk assessment for both human and ecological receptors.

WHO IT'S FOR RAGS Volume III: Part A is written primarily for risk assessors. Risk assessment reviewers, remedial project managers, and risk managers involved in Superfund site cleanup activities will also benefit from this addition to RAGS.

WHAT'S NEW RAGS Volume III: Part A provides guidance on applying probabilistic analysis to both human health and ecological risk assessment. New information and techniques are presented that reflect the views of EPA Superfund program. A tiered approach is described for determining the extent and scope of the modeling effort that is consistent with the risk assessment objectives, the data available, and the information that may be used to support remedial action decisions at Superfund hazardous waste sites.

RAGS Volume III: Part A contains the following information:

- For the risk assessor—updated policies and guidance; discussion and examples of Monte Carlo modeling techniques for estimating exposure and risk.
- For the risk manager and the remedial project manager—an introduction to PRA, a chapter on communicating methods and results of PRA with the public, and a chapter on the role of PRA in decision making.

TABLE OF CONTENTS

Table of Contents	iv
Acronyms and Abbreviations	xvi
Preface	i
1.0 What is the Purpose of RAGS Volume 3 Part A?	ii
2.0 What is Probabilistic Risk Assessment and how is it used in Risk Characterization? ...	ii
3.0 What are the Advantages and Disadvantages of PRA for Remedial Decisions?	iii
4.0 How is <i>RAGS Volume 3, Part A</i> Organized?	iii
5.0 What are the Key Guiding Concepts in <i>RAGS Volume 3: Part A</i> ?	iii
References for Preface	v
Chapter 1 Overview of Probabilistic Approach to Risk Assessment	1-1
1.0 Introduction	1-1
1.1 The Role of Risk Assessment in Superfund	1-4
1.1.1 Risk Assessment in the United States	1-4
1.1.2 Risk Assessment at EPA	1-5
1.1.3 Risk Assessment in Superfund	1-5
1.1.4 Probabilistic Risk Assessment and Its Role in Superfund	1-7
1.2 Basic Concepts of Probabilistic Risk Assessment	1-9
1.2.1 What is PRA?	1-10
1.2.2 What is a Monte Carlo Simulation?	1-13
1.2.3 Why is Variability Important in Risk Assessment? How is it Addressed by the Point Estimate and Probabilistic Approaches?	1-15
1.2.4 Why is Uncertainty Important in Risk Assessment? How is Uncertainty Addressed by the Point Estimate and Probabilistic Approaches?	1-17
1.2.5 Reasonable Maximum Exposure at the High-end	1-21
1.3 Advantages and Disadvantages of Point Estimate and Probabilistic Approaches ...	1-21
1.4 Conducting an Acceptable PRA	1-24
1.4.1 Key Policies for Applying PRA at Superfund Sites	1-26
1.5 Organization of the Guidance	1-27
1.6 Next Steps for PRA Implementation	1-30
References for Chapter 1	1-31
Exhibit 1-1 Definitions for Chapter 1	1-2
Exhibit 1-2 Nine Criteria for Evaluation of Cleanup Alternatives	1-6
Exhibit 1-3 Cancer and Noncancer Risk Models	1-11
Exhibit 1-4 Use a PDF and CDF To Display:	1-12
Exhibit 1-5 Quantifying Variability and Uncertainty	1-20
Exhibit 1-6 Advantages and Disadvantages of Point Estimate Approach	1-22
Exhibit 1-7 Advantages and Disadvantages of Probabilistic Risk Assessment	1-23

Figure 1-1	Example of a normal distribution that characterizes variability in adult body weight	1-12
Figure 1-2	Conceptual model of Monte Carlo analysis	1-14
Figure 1-3	Example of a probability distribution for risk illustrating the 95 th percentile and two different risk levels of concern (A and B)	1-16
Figure 1-4	Illustration of “Vertical” and “Horizontal” Confidence Intervals (or limits) on a risk estimate	1-19
Chapter 2	Workplan and The Tiered Approach	2-1
2.0	Introduction	2-1
2.1	Workplan	2-1
2.2	Special Administrative Considerations in PRA	2-4
2.2.1	Scoping of PRA	2-4
2.2.1.1	PRA Scope of Work for Fund-lead Sites	2-4
2.2.1.2	PRP Scope of Work for PRP-Lead Sites	2-5
2.2.2	Development of Probability Distributions	2-5
2.2.3	EPA Review of PRA Documents	2-6
2.2.4	Peer-Review	2-6
2.2.5	Response to Comments on PRA	2-6
2.2.6	Administrative Record	2-6
2.2.7	Communication with Stakeholders	2-6
2.2.8	Communication with EPA Management	2-7
2.3	Overview of the Tiered Approach	2-7
2.3.1	Getting Started	2-11
2.3.2	Tier 1	2-11
2.3.3	Tier 2	2-14
2.3.4	Tier 3	2-17
2.3.5	Flexibility in Defining Tiers	2-18
References for Chapter 2		2-19
Exhibit 2-1	Definitions for Chapter 2	2-2
Exhibit 2-2	Examples of Important Contents of A PRA Workplan	2-4
Exhibit 2-3	Stakeholders Potentially Involved in EPA’s Decision-Making Process for PRA	2-8
Exhibit 2-4	Typical Elements of Tier 1 Risk Assessment	2-11
Exhibit 2-5	Typical Elements of Tier 2 Risk Assessment	2-15
Exhibit 2-6	Typical Elements of Tier 3 Risk Assessment	2-17
Figure 2-1	Schematic Diagram of Tiered Approach	2-9
Figure 2-2	Schematic diagram of deliberation/decision cycle in the tiered process for PRA	2-10
Chapter 3	Using Probabilistic Analysis in Human Health Assessment	3-1
3.0	Introduction	3-1
3.1	Characterizing Variability In Exposure Variables	3-1
3.1.1	Developing Distributions For Exposure Variables	3-5
3.1.2	Characterizing Risk Using PRA	3-6
3.2	Role of the Sensitivity Analysis	3-9
3.3	Exposure Point Concentration Term	3-10
3.4	Characterizing Uncertainty in Exposure Variables	3-11

3.4.1	Parameter Uncertainty	3-11
3.4.2	Scenario and Model Uncertainty	3-17
3.5	Example of PRA for Human Health	3-17
References for Chapter 3		3-27
Exhibit 3-1	General Equation for Exposure	3-1
Exhibit 3-2	Definitions for Chapter 3	3-2
Exhibit 3-3	Equation for Cancer Risk	3-7
Exhibit 3-4	Equation for Noncancer Hazard Quotient	3-7
Exhibit 3-5	Using the Tiered Process for PRA Hypothetical Case Study for Human Health Risk Assessment	3-18
Exhibit 3-6	Risk Equations	3-23
Figure 3-1	Example of a frequency distribution for adult drinking water ingestion rates	3-4
Figure 3-2	Hypothetical PRA results showing a PDF and CDF	3-8
Figure 3-3	CDFs of risk based on Monte Carlo simulations described in Table 3-2.	3-16
Figure 3-4	CDFs of risk based on Monte Carlo simulations described in Table 3-2.	3-16
Figure 3-5	Site map for future wildlife refuge	3-22
Figure 3-6	Results of sensitivity analysis for preliminary 1-D MCA (Tier 2)	3-26
Table 3-1	Methods for characterizing parameter uncertainty with Monte Carlo simulations	3-12
Table 3-2	Example of 1-D MCA and 2-D MCA	3-14
Table 3-3	Concentrations in Surface Soil (mg/kg)	3-22
Table 3-4	Exposure Parameters used in Point Estimate Analysis.	3-24
Table 3-5	Point Estimate Risks and Exposure Pathway Contributions	3-24
Table 3-6	Input Distributions for Exposure Variables used in 1-D MCA for Variability	3-25
Table 3-7	1-D MCA Risk Estimates using Preliminary Inputs	3-25
Table 3-8	Exposure Duration Survey Results.	3-26
Table 3-9	Refined Point Estimate and 1-D MCA Risk Estimates	3-26
Chapter 4	Probabilistic Analysis in Ecological Risk Assessment	4-1
4.1	Introduction	4-1
4.1.1	Basic Approach for Performing Ecological Risk Assessments	4-1
4.1.2	Predictive vs Observational Approaches	4-6
4.1.3	Potential Advantages and Limitations of Probabilistic Methods in ERA	4-7
4.1.4	Focus of This Chapter	4-8
4.2	Deciding If and When to Use PRA in Ecological Risk Assessment	4-8
4.2.1	Technical Considerations	4-9
4.2.2	Cost and Schedule Considerations	4-11
4.3	Problem Formulation	4-11
4.4	Modeling Variability in Exposure	4-12
4.4.1	Characterizing Variability in Dose	4-12
4.4.2	Characterizing Variability in Exposure Concentration	4-15
4.5	Modeling Variability in Toxicity	4-15
4.5.1	Variability in Response Among Members of a Population	4-15
4.5.2	Variability in Response Among Species	4-20
4.6	Modeling Variability in Risk	4-22
4.6.1	Variability in Hazard Quotient	4-22

4.6.2	Variability in Response	4-26
4.6.3	Joint Probability Curves	4-30
4.7	Modeling Uncertainty in Ecological Risk Assessments	4-31
4.7.1	Uncertainty in Exposure	4-31
4.7.2	Uncertainty in Toxicity	4-32
4.7.4	Uncertainty in Response	4-34
4.7.3	Uncertainty in Hazard Quotient	4-35
4.8	Interpreting Results of an Ecological PRA	4-37
4.9	Guidelines For Planning And Performing a Probabilistic ERA	4-39
4.9.1	Planning an Ecological PRA	4-39
4.9.2	Evaluating an Ecological PRA	4-41
4.10	Example of the Tiered Process in ERA	4-41
References for Chapter 4		4-49
Exhibit 4-1	Definitions for Chapter 4	4-3
Exhibit 4-2	Ecological Risk Assessment Guidance and Policy Directives	4-4
Exhibit 4-3	Modeling Variability in Response for a Dichotomous Endpoint	4-17
Exhibit 4-4	Modeling Variability in Response for a Continuous Endpoint	4-19
Exhibit 4-5	Hypothetical Species Sensitivity Distribution	4-21
Exhibit 4-6	Modeling Variability in a Dichotomous Response	4-27
Exhibit 4-7	Modeling Variability in a Continuous Response	4-29
Exhibit 4-8	Example Elements of a Workplan for Ecological PRA	4-40
Exhibit 4-9	Checklist for Including a PRA as Part of the ERA	4-41
Exhibit 4-10	Refined Screening Point Estimate Inputs and Results	4-43
Exhibit 4-11	Screening Level PRA Calculations of HQ Distribution	4-45
Exhibit 4-12	Simulated Distribution of Responses	4-47
Figure 4-1	Ecological Risk Assessment Framework (U.S. EPA, 1992a)	4-1
Figure 4-2	Eight-step Ecological Risk Assessment Process for Superfund	4-5
Figure 4-3	Example of cases where use of PRA may be helpful	4-10
Figure 4-4	Example Graphical Presentations of Dose Distributions.	4-14
Figure 4-5	Example Comparison of Exposure Distribution to TRV.	4-22
Figure 4-6	Example Distribution of HQ Values.	4-23
Figure 4-7	Example Presentation of Species Sensitivity Distribution.	4-25
Figure 4-8	Example Joint Probability Curve.	4-30
Figure 4-9	Example Presentation of Uncertainty in Exposure.	4-31
Figure 4-10	Example Presentation of Uncertainty in Response.	4-35
Figure 4-11	Example Presentation of Uncertainty in Exposure and TRV.	4-36
Figure 4-12	Example Presentation of Uncertainty in HQ Estimates	4-37
Chapter 5	Probabilistic Risk Assessment and Preliminary Remediation Goals	5-1
5.0	Introduction	5-1
5.1	General Concepts Regarding EPCs and PRGs	5-4
5.1.1	Sources of Uncertainty in the EPC	5-5
5.1.2	Pre- and Post-Remediation Exposure Point Concentrations	5-6
5.1.3	Remediation Action Levels and 95% UCL Calculation Methods	5-6
5.1.4	Consideration of Risk from Acute Toxicity	5-7

5.1.5	Characterization of Uncertainty in the EPC: Point Estimates and Distributions	5-8
5.1.6	Multiple Chemicals	5-8
5.2	When to Use PRA for Developing PRGs	5-9
5.3	Methods for Developing PRGs	5-19
5.4	Backcalculation	5-10
5.4.1	Difficulties with Backcalculation	5-11
5.5	Iterative Methods	5-11
5.5.1	Iterative Reduction	5-12
5.5.2	Iterative Truncation	5-13
5.5.3	Example of Iterative Methods	5-14
5.5.4	Multiple Exposure units and Iterative Methods	5-17
5.6	PRGs for Groundwater	5-18
5.7	PRGs for Other Contaminated Media	5-19
5.8	Measurement of Attainment	5-21
5.9	Summary of Recommended Methods	5-23
References for Chapter 5		5-24
Exhibit 5-1	Summaries of Some Key Terms	5-1
Exhibit 5-2	Definitions for Chapter 5	5-2
Exhibit 5-3	Criteria for Iterative Truncation	5-14
Exhibit 5-4	Example of Iterative Methods	5-16
Exhibit 5-5	Evaluation of Alternative RALs Using Iterative Truncation	5-20
Figure 5-1	A hypothetical example of the use of iterative methods	5-12
Figure 5-2	Lognormal probability plot of soil concentrations, including 4 nondetects	5-16
Figure 5-3	Hypothetical example of a mixed, bimodal distribution.	5-18
Table 5-1	Soil sample	5-16
Table 5-2	Pre- and Post-Remediation EPCs (95% UCLs) for Chemical X in Surface Soil Samples	5-17
Table 5-3	Summary of Potential Methods for PRG Development by Environmental Medium	5-23
Chapter 6	Communicating Risks and Uncertainties in Probabilistic Risk Assessments	6-1
6.0	Introduction	6-1
6.1	Stakeholder Involvement	6-4
6.2	Communication and Presentation	6-5
6.2.1	Communication of PRA With Concerned Citizens, Other Stakeholders, and Managers: An Overview	6-6
6.2.2	Steps for Communication of the Results of the PRA	6-7
6.3	Communicating Differences Between Point Estimate and PRA	6-10
6.4	Graphical Presentation of PRA Results to Various Audiences	6-11
6.4.1	Public Meeting	6-11
6.4.2	EPA Senior Staff	6-17
6.4.3	Press Releases	6-19
6.5	Perception of Risk And Uncertainty	6-19
6.6	Trust and Credibility	6-21

6.7	Communication Issues for RPMs	6-21
References for Chapter 6		6-23
Supplemental References		6-24
Exhibit 6-1	Definitions for Chapter 6	6-2
Exhibit 6-2	Stakeholders Potentially Involved in the Decision-Making Process for PRA	6-4
Exhibit 6-3	Important Steps for Communicating PRA Results	6-7
Exhibit 6-4	Key Considerations in Developing Understandable Material	6-8
Figure 6-1	Hypothetical PRA results showing a PDF and CDF	6-12
Figure 6-2	Results of a sensitivity analysis shown as a pie chart and tornado plot.	6-16
Figure 6-3	The results of a 2-D MCA	6-17
Table 6-1	Examples of Graphics for Communicating PRA Concepts in this Guidance Document	6-14
Chapter 7	Role of the PRA in Decision Making	7-1
7.0	Introduction	7-1
7.1	General Principles of Risk-Based Decision Making In Superfund	7-1
7.2	Interpreting A Risk Distribution	7-3
7.2.1	What Is A Distribution Of Risk And What Does It Look Like?	7-3
7.2.2	What Is the RME Range?	7-4
7.2.3	Relating the Risk Distribution to the Risk Management Goal for Human Health	7-4
7.2.4	Relating the Risk Distribution to the Risk Management Goal for Ecological Risk Assessment	7-6
7.3	Factors to Consider in Choosing the Percentile for the RME	7-6
7.4	Uncertainty Associated with the Use of the 99.9 th Percentile	7-11
7.5	Moving From A PRG To A Remedial Goal	7-11
References for Chapter 7		7-15
Exhibit 7-1	Definitions for Chapter 7	7-2
Exhibit 7-2	Examples of Demographic, Cultural, and Behavioral Factors that Can Affect Exposure	7-7
Exhibit 7-3	Examples of Physical or Geographical Factors that Can Affect Exposure	7-7
Exhibit 7-4	Examples of Toxicity Considerations	7-9
Figure 7-1	Hypothetical PRA results showing a CDF for lifetime excess cancer risk.	7-3
Figure 7-2	Example of a probability distribution for risk illustrating the 95 th percentile	7-5
Figure 7-3	Box and whisker plots characterizing uncertainty in the RME	7-10
Figure 7-4	Example of graphic showing variability in risk (i.e., RME range, or 90 th to 99.9 th percentiles) associated with different choices of PRG for plutonium in soil (pCi/g).	7-14
Figure 7-5	Example of graphic showing uncertainty in a 95 th percentile of the risk distribution associated with the same choices of PRG as Figure 7-4.	7-14
Appendix A	Sensitivity Analysis: How Do We Know What's Important?	A-1

A.0	Introduction	A-1
A.1.0	Utility of Sensitivity Analysis	A-3
A.2.0	Common Methods of Sensitivity Analysis	A-10
A.2.1	Tier 1 Approaches	A-11
A.2.1.1	Percentage Contribution of Exposure Pathways to Total Risk	A-12
A.2.1.2	Inspection of Risk Equation	A-13
A.2.1.3	Sensitivity Ratio (SR)	A-13
A.2.1.4	Sensitivity Score	A-19
A.2.2	Tier 2 Approaches	A-21
A.2.2.1	Graphical Techniques	A-21
A.2.2.2	Correlation Coefficients	A-21
A.2.2.3	Focusing on the RME Range of the Risk Distribution	A-27
A.2.2.4	Inspection	A-27
A.3.0	Advanced Concepts in Sensitivity Analysis	A-28
A.3.1	Relating the Change in Risk to the Change in Input Variable X	A-28
A.3.2	Normalized Partial Derivative	A-31
A.3.3	Regression Analysis: R^2 , Pearson r , and Partial Correlation Coefficients	A-32
A.3.3.1	Calculations of R^2 and Adjusted R^2	A-33
A.3.3.2	Relative Partial Sum of Squares (RPSS)	A-35
A.3.3.3	Spearman's Rank Correlation Coefficient (Rho)	A-36
	References for Appendix A	A-37
Exhibit A-1	Definitions for Appendix A	A-2
Exhibit A-2	Utility of Sensitivity Analysis	A-3
Exhibit A-3	Some Key Indices of Sensitivity Analysis	A-10
Exhibit A-4	Categories of Solutions for Sensitivity Ratios of Multiplicative or Additive Equations	A-17
Exhibit A-5	Simplifying Assumptions in Regression Analysis	A-32
Figure A-1	Results of 2-D MCA in which parameters of input distributions describing variability are assumed to be random values.	A-9
Figure A-2	Scatterplots of simulated random values from a 1-D MCA of variability. The output from the model is a contaminant concentration in soil (C) that corresponds with a prescribed (fixed) level of risk for a hypothetical population	A-23
Figure A-3	Scatterplots of simulated random values from a 1-D MCA of variability for example in Section A.2.0	A-24
Figure A-4	Top panel - bar graph showing the r^2 values (square of Spearman rank correlation coefficient), a metric for the dependence of HI on exposure factors based on 1-D MCA for variability. Bottom panel - bar graph, sometimes referred to as "tornado plot", showing rank correlation coefficient.	A-25
Figure A-5a	Hypothetical 2-D response surface for Y given one input variable: $Y=F(X)$.	A-29
Figure A-5b	Hypothetical 3-D response surface for Y given two input variables: $Y=f(X_1, X_2)$	A-30
Figure A-5c	Hypothetical 3-D response surface when Y is a linear function of two input variables: $Y=f(X_1, X_2)$	A-30
Table A-1	Overview of Sensitivity Analysis Methods Applicable in Tiers 1, 2, and 3 of a PRA	A-4
Table A-2	Point estimates and probability distributions for input variables used in the hypothetical example of HI associated with occupational exposure via water and soil ingestion.	A-11
Table A-3	Percent contribution of exposure pathways to HI for the example in Section A.2	A-12

Table A-4	Results of the Sensitivity Ratio (SR) approach applied to the hypothetical example of RME HI given in Section A.2.0. Includes <i>both</i> soil ingestion and tap water ingestion pathways	A-14
Table A-5	Results of the Sensitivity Ratio (SR) approach applied to the hypothetical example of RME HI given in Section A.2.0. Includes <i>only</i> tap water ingestion pathway	A-15
Table A-6	Examples of algebraic solutions to Sensitivity Ratio calculations for additive and multiplicative forms of risk equations	A-17
Table A-7	Calculation of coefficient of variation ($CV = SD / \text{Mean}$) for the hypothetical example of RME HI given in Section A.2.0	A-19
Table A-8	Results of the Sensitivity Score (Score) approach applied to the hypothetical example of RME HI given in Section A.2.0	A-20
Table A-9	Results of Tier 2 sensitivity analyses applied to hypothetical example in Section A.2.0: Pearson product moment correlations and Spearman rank correlations	A-22
Appendix B	Selection and Fitting of Distributions	B-1
B.0	Introduction	B-1
B.1.0	Conceptual Approach for Incorporating a Probability Distribution in a PRA	B-3
B.2.0	Preliminary Sensitivity Analysis	B-4
B.3.0	What Does The Distribution Represent?	B-5
B.3.1	Concepts of Population and Sampling	B-6
B.3.2	Considering Variability and Uncertainty in Selecting and Fitting Distributions	B-12
B.4.0	Do Data Exist To Select Distributions?	B-13
B.4.1	What are Representative Data?	B-14
B.4.2	The Role of Expert Judgment	B-15
B.5.0	Fitting Distributions to Data	B-16
B.5.1	Considering the Underlying Mechanism	B-17
B.5.2	Empirical Distribution Functions (EDFs)	B-22
B.5.3	Graphical Methods for Selecting Probability Distributions	B-22
B.5.4	Parameter Estimation Methods	B-24
B.5.5	Dealing with Correlations among Variables or Parameters	B-26
B.5.6	Censored Data	B-28
B.5.7	Truncation	B-30
B.6.0	Assessing Quality of the Fit	B-31
B.6.1	What is a Goodness-of-Fit Test?	B-31
B.6.2	What are some common Goodness-of-Fit Techniques?	B-33
B.6.3	Cautions Regarding Goodness-of-Fit Tests	B-34
B.6.4	Accuracy of the Tails of the Distribution	B-34
B.7.0	Selecting Probability Distributions Based on State of Knowledge	B-35
References for Appendix B		B-49
Exhibit B-1	Definitions for Appendix B	B-2
Exhibit B-2	General Strategy for Selecting and Fitting Distributions	B-3
Exhibit B-3	Factors to Consider in Selecting a Probability Distribution	B-16
Exhibit B-4	Variations of the EDF	B-22
Exhibit B-5	Estimating the area of a hypothetical exposure unit	B-24
Exhibit B-6	Criteria for Evaluating Parameter Estimation Methods	B-25
Exhibit B-7	Parameter Estimation Methods	B-25

Exhibit B-8	Correlation of Input Variables for 1-D MCA of Variability	B-27
Exhibit B-9	Steps for Simulating Uncertainty in Linear Regression Equation Using a Bivariate Normal Distribution to Correlate Parameters (β_0 , β_1)	B-47
Figure B-1	(page 1 of 2). Conceptual approach for incorporating probability distributions for variability in PRA	B-7
Figure B-1	(page 2 of 2). Conceptual approach for incorporating probability distributions for variability in PRA	B-8
Figure B-2a	(page 1 of 3). Conceptual approach for quantifying model and parameter uncertainty in PRA	B-9
Figure B-2a	(page 2 of 3). Conceptual approach for quantifying model and parameter uncertainty in PRA	B-10
Figure B-2a	(page 3 of 3). Conceptual approach for quantifying model and parameter uncertainty in PRA	B-11
Figure B-3	Comparison of step-wise EDF and linearized EDF for ingestion rate	B-38
Figure B-4	Graphical assessment of beta and lognormal distributions fit to the cumulative distribution reported in the literature (circles)	B-39
Figure B-5	Histograms of lead concentrations in quail breast muscle	B-41
Figure B-6	Lognormal probability plots of lead in mourning dove breast tissue	B-43
Figure B-7	Histograms of meal size	B-44
Figure B-8	Probability plot of meal size data	B-45
Figure B-9	Simple linear regression of zinc concentrations in soil and dust	B-48
Figure B-10	Results of Monte Carlo simulation	B-49
Table B-1	Examples of Preliminary Distributions Based on Information Available	B-5
Table B-2	Examples of Selected Probability Distributions for PRA	B-18
Table B-3	Theoretical bounds and parameter values for selected distributions	B-30
Table B-4	Strategies for conducting PRA based on available information	B-36
Table B-5	Selected statistics for reported and fitted distributions for ingestion rate (mg/day). .	B-38
Table B-6	Sample values of lead concentration (ppm) in quail breast muscle	B-41
Table B-7	Parameter estimates for lognormal distribution of lead concentrations (ppm).	B-42
Table B-8	Meal size (g/meal)	B-44
Table B-9	Zinc concentrations in paired (i.e., co-located) soil and dust samples (ppm) for n=21 locations	B-48

Appendix C Characterizing Variability and Uncertainty in the Concentration Term C-1

C.0	The Concentration Term and the Exposure Unit	C-1
C.1.0	Variability in PRA	C-1
C.1.1	Temporal Variability	C-2
C.1.2	Spatial Variability	C-3
C.1.3	Example of Temporal and Spatial Variability	C-4
C.1.4	Spatial and Temporal Variability for Different Exposure Media	C-5
	C.1.4.1 Variability of Concentrations in Soil	C-5
	C.1.4.2 Variability of Concentrations in Groundwater	C-5
	C.1.4.3 Variability of Concentrations in Surface Water	C-5
	C.1.4.4 Variability of Concentrations in Sediment	C-5
	C.1.4.5 Variability of Concentrations in Fish	C-5
	C.1.4.6 Examples of Temporal and Spatial Variability in the Concentration Term for Selected Exposure Media	C-6
C.2.0	Nonrandom Exposures	C-7

C.3.0	Sources of Uncertainty in the Concentration Term	C-8
C.3.1	Quantification of Uncertainty Based on the Size of the Exposure Unit	C-8
C.3.1.1	When the Exposure Unit Is Smaller than the Site	C-8
C.3.1.2	When the Exposure Unit is the Same Size as the Site	C-9
C.3.1.3	When the Exposure Unit is Larger than the Site	C-9
C.4.0	Summary of Recommendations for the Concentration Term	C-10
C.5.0	Methods for Estimating Uncertainty in the Mean Concentration	C-10
C.5.1	Quantifying Uncertainty without Information About Locations of Samples and Receptors	C-12
C.5.2	Quantifying Uncertainty with Information About Locations of Samples and Receptors	C-12
References for Appendix C		C-14
Figure C-1	Spatial and temporal variability in contaminant concentrations in groundwater	C-7
Table C-1	Examples of temporal and spatial variability in selected media for the concentration term in common exposure scenarios	C-6
Table C-2	Summary of factors that may be considered in developing an EPC	C-10
Appendix D Advanced Modeling Approaches for Characterizing Variability and Uncertainty D-1		
D.0	Introduction	D-1
D.1.0	Expressing Variability and Uncertainty Simultaneously	D-1
D.2.0	Two-Dimensional Monte Carlo Analysis (2-D MCA)	D-3
D.3.0	Microexposure Event Analysis	D-6
D.4.0	Geospatial Statistics	D-10
D.4.1	Correlation and Spatial Autocorrelation	D-11
D.4.2	Effective Sample Size (n^*) and Degrees of Freedom	D-12
D.4.3	Assessment of Additional Site Sampling	D-13
D.4.4	Map Generalization	D-15
D.4.5	Implementation Issues Related to Georeferenced Data	D-16
D.5.0	Expert Judgment and Bayesian Analysis	D-16
References for Appendix D		D-25
Exhibit D-1	Definitions for Appendix D	D-3
Exhibit D-2	Positive Spatial Autocorrelation	D-10
Exhibit D-3	Examples of Risk Assessment Issues Linked to Geospatial Statistics	D-10
Exhibit D-4	Effect of Spatial Autocorrelation (r) on Effective Sample Size (n^*)	D-13
Exhibit D-5	Components of Bayes Theorem in PRA	D-17
Figure D-1	Panel A shows a family of 20 CDFs for a hypothetical random variable. Panel B shows the “90% credible interval” for the CDF based on 2500 <i>simulations</i>	D-2
Figure D-2	Diagram showing of a 2-D Monte Carlo model	D-4
Figure D-3	Output from a 2-D MCA showing the estimated mean Hazard Quotient (HQ) and the 90% confidence interval	D-5
Figure D-4	Time Step for MEE	D-7
Figure D-5	Flowchart showing general approach for Microexposure Event (MEE) analysis.	D-8
Figure D-6	Hypothetical example showing the effect of model time step on the probability distribution for soil and dust ingestion rate in children over a 1-year period	D-9
Figure D-7	Effect of an outlier on measured correlation	D-12
Figure D-8	Conceptual model of Bayesian Monte Carlo analysis	D-18

Figure D-9	Expected Loss associated with various types of information incorporated into a generic uncertainty analysis	D-21
Figure D-10	Conceptual model for evaluating the expected value of including uncertainty in a Bayesian Monte Carlo analysis	D-23
Appendix E	Definitions of Terms Relevant to PRA and References for Further Reading ...	E-1
E.0	Definitions of Terms	E-1
E.1	Additional Information	E-14
	References for Appendix E	E-15
	References for Further Reading	E-16
Appendix F	Workplan and Checklist for PRA	F-1
F.0	Introduction	F-1
F.1.0	Workplan	F-1
F.2.0	Focal Points for PRA Review	F-2
F.3.0	Checklist for Reviewers	F-2
F.4.0	Internal and External Review	F-3
	References for Appendix F	F-6
Exhibit F-1	Examples of Elements of the Workplan for PRA	F-1
Exhibit F-2	Key Focal Points for PRA Review	F-2
Table F-1	Example of a Generic Checklist for Reviewers	F-4
Appendix G	Frequently Asked Questions for PRA	G-1
	References for Appendix G	G-6
Appendix H	Index	H-1

ACRONYMS AND ABBREVIATIONS

1-D MCA	One-dimensional Monte Carlo analysis
2-D MCA	Two-dimensional Monte Carlo analysis
95% UCL	95% upper confidence limit
AM	Arithmetic mean
ARARs	Applicable or relevant and appropriate requirements
AT	Averaging time
AWQC	Ambient water quality criterion
BCa	Bias correction acceleration method
BMD	Benchmark dose
BMDS	Benchmark dose software
BMR	Benchmark Response
BTAG	Biological Technical Assistance Group
BW	Body weight
C	Concentration
CAG	Community advisory group
CDF	Cumulative distribution function
CI	Confidence interval
CIC	Community involvement coordinator
CIP	Community involvement plan
CLT	Central limit theorem
COC	Chemical of concern
CQR	Continuous quadratic regression
CSF	Cancer slope factor
CTE	Central tendency exposure
CV	Coefficient of variation
DI	Daily intake
DQO	Data quality objectives
EC ₀	Exposure concentration that produces zero effect
EC ₂₀	Concentration that causes a 20% effect
ECDF	Empirical cumulative distribution function
ED	Exposure duration
ED ₁₀	Dose that causes a 10% effect
EDF	Empirical distribution function
EF	Exposure frequency
EPA	U.S. Environmental Protection Agency
EPC	Exposure point concentration
ERA	Ecological risk assessment
ERAF	Risk Assessment Forum
ERAGS	Ecological Risk Assessment Guidance for Superfund
EU	Exposure unit
EVIU	Expected value of including uncertainty
EVOI	Expected value of information
EVPI	Expected value of perfect information
EVSI	Expected value of sample information
GIS	Geographical Information Systems
GM	Geometric mean
GoF	Goodness-of-Fit
GSD	Geometric standard deviation
HEAST	Health effects assessment summary table
HHEM	Human Health Evaluation Manual
HI	Hazard Index

HQ	Hazard Quotient
IR	Iterative reduction
Irsd	Soil and dust ingestion rate
IRIS	Integrated Risk Information System
LADD	Life-time average daily intake
LCL	Lower confidence limit
LED ₁₀	Lowest effect dose - lower confidence bound for dose that causes a 10% effect
LHS	Latin hypercube sampling
LOAEL	Lowest-observed-adverse-effect level
LOD	Limit of detection
LOEC	Lowest-observed-effect-concentration
MCA	Monte Carlo analysis
MCL	Maximum contaminant levels
MDC	Maximum detected concentration
MEE	Microexposure Event Analysis
MLE	Maximum Likelihood Estimation
MoMM	Method of Matching Moments
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NOAEL	No-observed-adverse-effect level
NOEC	No-observed-effect-concentration
OLS	Ordinary least squares
PBPK	Physiologically-based pharmacokinetic
PCBs	Polychlorinated biphenyls
pCi/g	Picocuries/gram
PDF	Probability density function
PDFu	Probability distribution for variability
PDFv	Probability distribution for uncertainty
PMF	Probability mass function
PPT	Parts per trillion
PRA	Probabilistic risk assessment
PRG	Preliminary remediation goal
PRP	Potentially responsible party
QAPP	Quality Assurance Project Plan
RAGS	Risk Assessment guidance for Superfund
RAL	Remedial action level
RBC	Risk based concentration
RCRA	Resource Conservation and Recovery Act
RfC	Reference concentration
RfD	Reference dose
RG	Remediation goal
RI/FS	Remedial Investigation/Feasibility Study
RME	Reasonable maximum exposure
RMSE	Root mean squared error
ROD	Record of decision
ROS	Rank order statistic
RPSS	Relative partial sum of squares
RPM	Remedial project manager
RSS	Regression sum of squares
SCM	Site conceptual model
SD	Standard deviation

SE	Standard error
SMDP	Scientific/Management Decision Point
SOW	Statement of Work
SR	Sensitivity ratio
SSD	Species sensitivity distribution
SSE	Sum of squares due to error
SSR	Sum of squares due to regression
SST	Sum of squares for total (regression plus error)
TAB	Technical Assistance to Brownfields Community
TAG	Technical assistance grant
TOSC	Technical outreach services for communities
TRV	Toxicity reference value
TSS	Total sum of squares
UCL	Upper confidence limit
VOI	Value of information

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PREFACE

Risk Assessment Guidance for Superfund (RAGS) Volume III: Part A (hereafter referred to as RAGS Volume 3: Part A) provides technical guidance on the application of probabilistic risk assessment (PRA) methods to human health and ecological risk assessment in the U.S. Environmental Protection Agency (EPA) Superfund program. *RAGS Volume 3: Part A* supplements existing human health and ecological assessment guidance provided in the RAGS series. This guidance focuses on Monte Carlo analysis (MCA) as a method of quantifying variability and uncertainty in risk. Primarily geared toward the risk assessor, it is intended, both in content and format, to be most accessible to those readers who are familiar with risk assessment and basic statistical concepts. Chapters 1, 2, 6, and 7 are also directed towards risk managers. The term risk manager is used in this guidance to refer to individuals or entities that serve as the decision makers at hazardous waste sites. The term is used to emphasize the separation between risk assessment and risk management activities. Risk managers may include individual remedial project managers (RPMs), site partnering teams, senior EPA managers (Section Chiefs, Branch Chiefs, or Division Directors), or other decision makers.

An attempt has been made in this document to define all relevant technical terms using plain language and to illustrate concepts with examples. An exhibit at the beginning of each chapter provides definitions of terms used in that chapter. In addition, a comprehensive definition of terms is provided in Appendix E. Other useful information has been presented in exhibits placed throughout each chapter. Bullets are used throughout the text to emphasize important concepts and policy statements related to the use of PRA. References are listed at the end of each chapter.

RAGS Volume 3: Part A was developed by the Superfund Probabilistic Risk Assessment Workgroup and the Ecological Risk Assessment Forum (ERAF); both are intra-Agency workgroups that have focused on improving the Risk Assessment Guidance for Superfund and implementing Superfund Reform activities. The guidance has undergone extensive review by Superfund and other programs within the Agency. In February 2000, a draft of the guidance was announced in the *Federal Register* to provide an opportunity for public comment (U.S. EPA, 2000a). In August 2000, a notice of peer review was announced in the *Federal Register* (U.S. EPA, 2000b), and in November 2000, *RAGS Volume 3: Part A* received a formal peer review from panelists outside the Agency.

The Agency may incorporate PRA under fund-lead and Potentially Responsible Party (PRP)-lead risk assessments. Implementation of successful PRAs requires careful planning. EPA strongly recommends that PRPs involve the Agency in all decisions regarding the planning, submittal, and technical details of any PRA. Coordinating with EPA early in the process will help ensure that PRAs conform to the recommended guidelines as part of the Superfund risk assessment process for protecting human and ecological health. PRPs should submit workplans for Agency review before initiating any PRA. Similarly, when EPA chooses to use PRA for an EPA-lead risk assessment, a PRA workplan will assist in directing site investigation and risk assessment activities, whether conducted by EPA or an EPA contractor. A workplan specifies contractor activities in the risk assessment and provides risk assessors and risk managers with an opportunity to obtain internal feedback from knowledgeable EPA staff, prior to initiating work on the assessment.

A tiered approach to PRA is advocated, which begins with a point estimate risk assessment. Important considerations include the time required to perform the PRA, the additional resources involved in developing the PRA, the quality and extent of data on exposure that will be used in the assessment, and

the value added by conducting the PRA. Project scoping is an essential component of all risk assessments and is especially important in PRA.

Implementation of a PRA usually requires special computer software that may be commercially available or that may need to be custom-designed for a specific application. Although commercial software packages are noted in this guidance, any mention or use of a particular product in *RAGS Volume 3: Part A* does not constitute an endorsement of that product by the Agency.

1.0 WHAT IS THE PURPOSE OF RAGS VOLUME 3 PART A?

RAGS Volume 3: Part A addresses the technical and policy issues associated with the use of PRA in EPA Superfund program. This guidance builds upon basic concepts of risk assessment outlined in *RAGS Volume I* (U.S. EPA, 1989a; 2001), recent guidance for ecological risk assessment (U.S. EPA, 1992, 1994, 1997a, 1998a; 1999), and the Agency Probabilistic Analysis Policy document (U.S. EPA, 1997b). *RAGS Volume 3: Part A* addresses the use of PRA for both human health and ecological risk assessments. *RAGS Volume 3: Part A* was developed to provide risk assessors and risk managers with basic guidelines for incorporating PRA into Superfund site-specific risk assessments. It is not intended to be a detailed technical reference on PRA methods, however, it does direct the reader to appropriate literature on important technical subjects. A primary purpose of *RAGS Volume 3: Part A* is to help prevent misuse and misinterpretation of PRA.

2.0 WHAT IS PROBABILISTIC RISK ASSESSMENT AND HOW IS IT USED IN RISK CHARACTERIZATION?

PRA is a risk assessment that uses probability distributions to characterize variability or uncertainty in risk estimates. In a PRA, one or more variables in the risk equation is defined as a probability distribution rather than a single number. Similarly, the output of a PRA is a range or probability distribution of risks experienced by the receptors. The evaluation of variability and uncertainty is an important component of the risk characterization of all risk assessments. As stated in the 1995 Risk Characterization memorandum from Administrator Carol Browner (U.S. EPA, 1995),

... we must fully, openly, and clearly characterize risks. In doing so, we will disclose the scientific analyses, uncertainties, assumptions, and science policies which underlie our decisions... There is value in sharing with others the complexities and challenges we face in making decisions in the face of uncertainty.

In addition, the 1997 EPA Policy for Use of Probabilistic Analysis in Risk Assessment (U.S. EPA, 1997b) states:

☞ It is the policy of the U.S. Environmental Protection Agency that such probabilistic analysis techniques as Monte Carlo analysis, given adequate supporting data and credible assumptions, can be viable statistical tools for analyzing variability and uncertainty in risk assessments.

A more extensive general discussion of PRA can be found in Chapter 1 of the guidance. The use of PRA in Superfund remedial decision making is presented in Chapter 7 of the guidance.

3.0 WHAT ARE THE ADVANTAGES AND DISADVANTAGES OF PRA FOR REMEDIAL DECISIONS?

The primary advantage of PRA within the Superfund program is that it can provide a quantitative description of the degree of variability or uncertainty (or both) in risk estimates for both cancer and non-cancer health effects and ecological hazards. The quantitative analysis of uncertainty and variability can provide a more comprehensive characterization of risk than is possible in the point estimate approach.

Another significant advantage of PRA is the additional information and potential flexibility it affords the risk manager. Superfund remedy decisions are often based on an evaluation of the risk to the individual at the reasonable maximum exposure (RME) level (U.S. EPA, 1990). The RME represents the highest exposure reasonably likely to occur (U.S. EPA, 1989a). When using PRA, the risk manager can select the RME from the high-end range of percentiles of risk, generally between the 90th and 99.9th percentiles, referred to in this guidance as the *RME range*.

However, PRA may not be appropriate for every site. Disadvantages of PRA are that it generally requires more time, resources, and expertise on the part of the assessor, reviewer, and risk manager than a point estimate approach.

4.0 HOW IS *RAGS VOLUME 3, PART A* ORGANIZED?

Although the primary audience of this guidance is the risk assessor, Chapter 1 provides a basic overview of PRA for risk assessors and risk managers. The centerpiece of *RAGS Volume 3: Part A* is the tiered approach described in Chapter 2. The tiered approach is a framework that enables the risk manager to decide if and when to undertake a PRA and to determine the appropriate level of complexity for the PRA. Chapter 3 provides a description of using PRA for human health risk assessment. Chapter 4 discusses the issues of using PRA for ecological risk assessment. Chapter 5 presents a discussion of using PRA to determine preliminary remediation goals. Chapter 6 details issues associated with communicating risk estimates developed with PRA. Chapter 7 provides information for risk managers choosing to base remedial decisions on the results of a PRA.

Eight appendices to this guidance expand on technical aspects of topics important to PRA, such as sensitivity analysis and selecting and fitting probability distributions.

5.0 WHAT ARE THE KEY GUIDING CONCEPTS IN *RAGS VOLUME 3: PART A*?

- (1) *Use a tiered approach to incorporating PRA into site risk assessments.*
- (2) *Submit a workplan for Agency review prior to initiating work on a PRA.*
- (3) *Perform a point estimate assessment prior to considering a PRA.*
- (4) *While PRA can provide a useful tool to characterize and quantify variability and uncertainty in risk assessments, it is not appropriate for every site.*
- (5) *PRA generally requires more time, resources, and expertise on the part of the assessor, reviewer, and risk manager than a point estimate risk assessment.*

- (6) The decision to use PRA is site-specific and is based on the complexity of the problems at the site, the quality and extent of site-specific data, and the likely utility of the result.*
- (7) If the additional information provided from a PRA is unlikely to affect the risk management decision, then it may not be prudent to proceed with a PRA. However, if there is a clear value added from performing a PRA, then the use of PRA as a risk assessment tool generally should be considered despite the additional resources that may be needed.*
- (8) Communicating the results of a PRA will be more challenging than communicating the results of a point estimate risk assessment because PRA and its perspective will be new to most participants.*
- (9) If the decision is made to conduct a PRA, it is important to include community in the planning process. Communication on PRA may involve: providing the community with a basic understanding of the principles of PRA, discussing the proposed workplan and inviting comments on the proposed approach, discussing site-specific data, and communicating the final results and how they impact decisions for the site.*

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CHAPTER 1

OVERVIEW OF PROBABILISTIC APPROACH TO RISK ASSESSMENT

1.0 INTRODUCTION

This chapter is intended for risk managers and risk assessors as an overview of the probabilistic approach to risk assessment in the context of the Superfund program at the U.S. Environmental Protection Agency (EPA). The goals of this chapter are to provide the reader with information about (1) the role of risk assessment in the Superfund program; (2) the basic concepts of probabilistic risk assessment (PRA); (3) important policies and guiding principles for PRA, as outlined throughout this guidance; and (4) the next steps that will be undertaken in the Superfund program to provide guidance on PRA.

Section 1.1 (1.1.1–1.1.3) describes the role of risk assessment from three perspectives, including the role of risk assessment in areas external to EPA, Agency-wide, and within Superfund. Section 1.1 (1.1.4) also introduces PRA and identifies its place in the Superfund program. Section 1.2 introduces the basic concepts of PRA, including the key terms of variability, uncertainty, Monte Carlo analysis (MCA), and reasonable maximum exposure (RME). PRA concepts are presented using a comparison between PRA and the traditional point estimate approach. Sections 1.2.4 and 1.3 summarize the advantages and disadvantages of PRA and point estimate risk assessment. Section 1.4 provides a summary of policies and guiding principles for using PRA in the Superfund program. EPA's policies on conducting PRA are highlighted throughout the guidance using pointers and are linked to more detailed policy discussions in other chapters in the guidance. Section 1.5 outlines the organization of this document and provides a brief summary of the content of each subsequent chapter and appendix. Section 1.6 presents EPA's next steps for PRA implementation in the Superfund program.

Key terms used throughout this guidance include: Probabilistic Risk Assessment (PRA), Monte Carlo Analysis (MCA), Probability Density Function (PDF), Cumulative Distribution Function (CDF), Reasonable Maximum Exposure (RME), Sensitivity Analysis, Tiered Approach, Variability, Uncertainty, and Preliminary Remediation Goal (PRG). Terms and their definitions are identified in an exhibit at the beginning of each chapter. Terms and definitions relevant to Chapter 1 are presented in Exhibit 1-1. In addition, a glossary of terms used throughout the guidance is given in Appendix E.

EXHIBIT 1-1

DEFINITIONS FOR CHAPTER 1

Central Tendency Exposure (CTE) - A risk descriptor representing the average or typical individual in a population, usually considered to be the mean or median of the distribution.

Confidence Interval - A range of values that are likely to include a population parameter. Confidence intervals may describe a parameter of an input variable (e.g., mean ingestion rate) or output variable (e.g., 95th percentile risk). When used to characterize uncertainty in a risk estimate, it is assumed that methods used to quantify uncertainty in the model inputs are based on statistical principles such as sampling distributions or Bayesian approaches. For example, given a randomly sampled data set, a 95% confidence interval for the mean can be estimated by deriving a sampling distribution from a Student's t distribution.

Confidence Limit - The upper or lower value of a confidence interval.

Countably Infinite - Used to describe some discrete random variables, this term refers to a set of numbers that can be counted with integers (e.g., one, two, three) and that has no upper limit. Examples include the number of tosses required for a coin to show a head—we can count each toss, but it is possible that at least one more toss is needed. The number of dust particles in a volume of air is another example. Countably finite implies there is an upper limit (e.g., days of work per year).

Credible Interval - A range of values that represent plausible bounds on a population parameter. Credible intervals may describe a parameter of an input variable (e.g., mean ingestion rate) or output variable (e.g., 95th percentile risk). The term is introduced as an alternative to the term confidence interval when the methods used to quantify uncertainty are not based entirely on statistical principles such as sampling distributions or Bayesian approaches. For example, multiple estimates of an arithmetic mean may be available from different studies reported in the literature - using professional judgment, these estimates may support a decision to describe a range of possible values for the arithmetic mean.

CTE Risk - The estimated risk corresponding to the central tendency exposure.

Cumulative Distribution Function (CDF) - Obtained by integrating the PDF, gives the cumulative probability of occurrence for a random independent variable. Each value *c* of the function is the probability that a random observation *x* will be less than or equal to *c*.

Expected Value of Information (EVOI) - The expected increase in the value (or decrease in the loss) associated with obtaining more information about quantities relevant to the decision process. EVOI is a measure of the importance of uncertainty in risk and the potential for changing a risk management decision if uncertainty is reduced (see Appendix D).

Frequency Distribution or Histogram - A graphic (plot) summarizing the frequency of the values observed or measured from a population. It conveys the range of values and the count (or proportion of the sample) that was observed across that range.

Monte Carlo Analysis (MCA) or Monte Carlo Simulation - A technique for characterizing the uncertainty and variability in risk estimates by repeatedly sampling the probability distributions of the risk equation inputs and using these inputs to calculate a range of risk values.

Numeric Stability - Stochastic variability, or "wobble" associated with random sampling, calculated as the average percent change in the model output after rerunning Monte Carlo simulations with the same set of input assumptions. Used as a metric for evaluating the adequacy of the number of iterations in a MCA.

Parameter - A value that characterizes the distribution of a random variable. Parameters commonly characterize the location, scale, shape, or bounds of the distribution. For example, a truncated normal probability distribution may be defined by four parameters: arithmetic mean [location], standard deviation [scale], and min and max [bounds]. It is important to distinguish between a variable (e.g., ingestion rate) and a parameter (e.g., arithmetic mean ingestion rate).

Point Estimate - In statistical theory, a quantity calculated from values in a sample to estimate a fixed but unknown population parameter. Point estimates typically represent a central tendency or upper bound estimate of variability.

EXHIBIT 1-1

DEFINITIONS FOR CHAPTER 1—*Continued*

Point Estimate Risk Assessment - A risk assessment in which a point estimate of risk is calculated from a set of point estimates for exposure and toxicity. Such point estimates of risk can reflect the CTE, RME, or bounding risk estimate depending on the choice of inputs.

Probabilistic Risk Assessment (PRA) - A risk assessment that yields a probability distribution for risk, generally by assigning a probability distribution to represent variability or uncertainty in one or more inputs to the risk equation.

Probability Density Function (PDF) - A function representing the probability distribution of a continuous random variable. The density at a point refers to the probability that the variable will have a value in a narrow range about that point.

Probability Distribution - A mathematical representation of the function that relates probabilities with specified intervals of values for a random variable. Also called a *probability model*.

Probability Mass Function (PMF) - A function representing the probability distribution for a discrete random variable. The mass at a point refers to the probability that the variable will have a value at that point.

Random Variable - A variable that may assume any value from a set of values according to chance. Discrete random variables can assume only a finite or countably infinite number of values (e.g., number of rainfall events per year). A random value is continuous if its set of possible values is an entire interval of numbers (e.g., quantity of rain in a year).

Reasonable Maximum Exposure (RME) - The highest exposure that is reasonably expected to occur at a site (U.S. EPA, 1989a). The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures.

Remedial Investigation/Feasibility Study (RI/FS) - Studies undertaken by EPA to delineate the nature and extent of contamination, to evaluate potential risk, and to develop alternatives for cleanup.

RME Risk - The estimated risk corresponding to the reasonable maximum exposure.

Sensitivity Analysis - Sensitivity generally refers to the variation in output of a model with respect to changes in the values of the model's input(s). Sensitivity analysis can provide a quantitative ranking of the model inputs based on their relative contributions to model output variability and uncertainty. Common metrics of sensitivity include:

- ▶ Pearson Correlation Coefficient - A statistic r that measures the strength and direction of linear association between the values of two quantitative variables. The square of the coefficient (r^2) is the fraction of the variance of one variable that is explained by the variance of the second variable.
- ▶ Sensitivity Ratio - Ratio of the change in model output per unit change in an input variable; also called *elasticity*.
- ▶ Spearman Rank Order Correlation Coefficient - A "distribution free" or nonparametric statistic r that measures the strength and direction of association between the ranks of the values (not the values themselves) of two quantitative variables. See Pearson (above) for r^2 .

Stochastic Dominance - Implies no intersection between two or more CDFs. For example, if the CDF for A and B do not overlap and the CDF for A is greater than the CDF for B, then at every cumulative percentile, the value of A is greater than that of B. Therefore, it can be stated that distribution A stochastically dominates distribution B. It should be noted that even when the CDFs for A and B do not overlap, the PDFs for A and B can overlap.

Uncertainty - Lack of knowledge about specific variables, parameters, models, or other factors. Examples include limited data regarding the concentration of a contaminant in an environmental medium and lack of information on local fish consumption practices. Uncertainty may be reduced through further study.

Variability - True heterogeneity or diversity that characterizes an exposure variable or response in a population. Further study (e.g., increasing sample size, n) will not reduce variability, but it can provide greater confidence (e.g., lower uncertainty) in quantitative characterizations of variability).

1.1 THE ROLE OF RISK ASSESSMENT IN SUPERFUND

The role of risk assessment in the Superfund program today is built upon a foundation of scientific and management principles, policies, and laws that have been established over the past two decades. Since the enactment of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) in 1980 the risk assessment policies and guidance documents have evolved to reflect advances in science and changes in federal regulations.

1.1.1 RISK ASSESSMENT IN THE UNITED STATES

Risk assessment has a long history beginning in 1940. In 1983, the National Research Council published *Risk Assessment in the Federal Government: Managing the Process* (NRC, 1983) which outlines the four steps of risk assessment (hazard identification, dose-response, exposure assessment, and risk characterization) that are used today.

The NRC addressed three main objectives in risk assessment: (1) assessment of the benefits of separating the analytical process of risk assessment from the regulatory process of risk management; (2) consideration of the feasibility of creating a single regulatory agency for the purpose of conducting all government risk assessments; and (3) consideration of the feasibility of creating uniform guidelines for risk assessment (NRC, 1983).

The Committee concluded that regulatory agencies should maintain a conceptual distinction between risk assessment and risk management, and develop uniform inference guidelines in risk assessment for use by all federal regulatory agencies. The Committee also recommended that Congress establish a Board on Risk Assessment Methods in order to ensure that risk assessment procedures be continuously reviewed and modified as the science advances. The Committee rejected the proposal for a single federal risk assessment agency based on inadequate evidence to show that one administrative structure would be more advantageous (NRC, 1983).

Since 1983, there have been ongoing advancements in the field of risk assessment. These include: (1) a continued increasing role for risk assessment in the decision-making process of many regulatory agencies, as exemplified by several bills introduced by the 103rd and 104th Congresses in 1994-1995; (2) an increased awareness of the need for uncertainty analysis and for quantifying and communicating uncertainties in risk estimates (*Science and Judgement in Risk Assessment*, NRC, 1994); (3) guidance about more inclusive approaches to risk assessment, as exemplified by environmental health legislation such as the Food Quality Protection Act (FQPA) of 1996 and the Presidential/Congressional Commission on Risk Assessment and Risk Management (1997); and (4) setting the stage for a more open decision-making process through stakeholder involvement in the risk management process, as outlined in *Improving Risk Communication* (NRC, 1989).

1.1.2 RISK ASSESSMENT AT EPA

EPA has refined the risk paradigm through deliberations of the Risk Assessment Forum, Science Policy Council, and other Agency-wide bodies. Such deliberations have led to consensus in guidance, policies, and memoranda that respond to the requirements set out by various environmental statutes. Individual offices have also developed regulations, guidance, and other supporting documents to aid in the implementation of particular environmental statutes.

In 1986, EPA issued final guidelines relating to risk assessment for cancer, mutagenic effects, developmental effects, exposure assessment, and chemical mixtures. Since 1986, EPA has updated or issued revised final guidelines for developmental toxicity, exposure assessment, reproductive toxicity, neurotoxicity, and ecological risk assessment; and is now revising carcinogen risk assessment guidelines. (See <http://www.epa.gov/ncea/raf/rafguid.htm> for details on *guidelines*.)

Other notable documents that guide risk assessment at EPA include:

- *Framework for Ecological Risk Assessment* (U.S. EPA, 1992b)
- *Guidelines for Ecological Risk Assessment* (U.S. EPA, 1998)
- *Guidance for Risk Characterization* (U.S. EPA, 1995a)
- *Policy for Risk Characterization* (U.S. EPA, 1995c)
- *Policy on Evaluating Health Risks to Children* (U.S. EPA, 1995d)
- *Policy for Use of Probabilistic Analysis in Risk Assessment* (U.S. EPA, 1997g)
- *Use of Probabilistic Techniques (including Monte Carlo Analysis) in Risk Assessment* (U.S. EPA, 1997g)
- *Guidance on Cumulative Risk Assessment. Part 1. Planning and Scoping* (U.S. EPA, 1997e)
- *Risk Characterization Handbook* (U.S. EPA, 2000)

1.1.3 RISK ASSESSMENT IN SUPERFUND

The activities and publications described above have provided a strong foundation for the development of risk assessment guidance on conducting human health—and ecological risk assessments in the Superfund program. EPA uses risk assessment (NRC, 1983, 1994) to carry out CERCLA, as amended by the Superfund Amendments and Reauthorization Act of 1986 (SARA). Under CERCLA/SARA, EPA's Superfund program is authorized to protect human health and the environment from current and potential threats posed by releases of hazardous substances, pollutants, or contaminants. The blueprint for the Superfund program is the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) (U.S. EPA, 1990). Among other things, the NCP calls for the identification and mitigation of environmental impacts at hazardous waste sites, and for the selection of remedial actions to protect human health and the environment. An important part of the NCP is the implementation of a Remedial Investigation and Feasibility Study (RI/FS), which is designed to support risk management decisions within the Superfund program. A risk assessment is an integral part of the RI/FS, and is

generally conducted at a site to determine the need for action and to ensure that a selected remedy will be protective. The NCP also establishes some benchmarks for protectiveness and lays out nine criteria (some risk-based) against which each cleanup option should be evaluated (see Exhibit 1-2).

Guidance for risk assessment in the Superfund program has been developed to facilitate consistent site-specific responses. Early major guidance documents developed by EPA included: *Risk Assessment Guidance for Superfund (RAGS): Volume I. Human Health Evaluation Manual (HHEM) (Part A, Baseline Risk Assessment)* (U.S. EPA, 1989a) and *Risk Assessment Guidance for Superfund. (RAGS): Volume II. Environmental Evaluation Manual* (U.S. EPA, 1989b). *RAGS Volume I: Part A* provides an approach for conducting site-specific baseline (i.e., without remediation or institutional controls) human health risk assessments. *RAGS Volume II*, aimed at site managers, provides a framework for considering environmental effects at sites. More recently, EPA developed guidance for conducting ecological risk assessments within the Superfund program. This guidance, *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (U.S. EPA, 1997a), discusses scientific methods and stakeholder input.

Over the years, the Superfund program has expanded RAGS to include the following documents relating to human health:

- *RAGS Volume I, Part B: Development of Risk-based Preliminary Remediation Goals (Risk Equations and Parameters)* (U.S. EPA, 1991b)
- *RAGS Volume I, Part C: Risk Evaluation of Remedial Alternatives* (U.S. EPA, 1991c)
- *RAGS Volume I, Part D: Standardized Planning, Reporting, and Review of Superfund Risk Assessments* (U.S. EPA, 2001a)
- *RAGS Volume I, Part E: Supplemental Guidance for Dermal Risk Assessment* (U.S. EPA, 2001b)

Additional ecological guidance documents include:

- *Role of the Ecological Risk Assessment in the Baseline Risk Assessment*. OSWER Directive No. 9285.7-17 (U.S. EPA, 1994a)
- *Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites*. OSWER Directive 9285.7-28 P (U.S. EPA, 1999)
- *The Role of Screening-Level Risk Assessments and Refining Contaminants of Concern in Baseline Risk Assessments*. 12th Intermittent Bulletin, ECO Update Series. (U.S. EPA, 2001d)

EXHIBIT 1-2

NINE CRITERIA FOR EVALUATION OF CLEANUP ALTERNATIVES (U.S. EPA, 1990)

Threshold Criteria

1. Overall protection of human health and the environment
2. Compliance with ARARs

Balancing Criteria

3. Long-term effectiveness and permanence
4. Reduction in toxicity, mobility, or volume through treatment
5. Short-term effectiveness
6. Implementability
7. Cost

Modifying Criteria

8. State acceptance
9. Community acceptance

This document (*RAGS Volume 3: Part A*) provides guidance for probabilistic approaches for both human health and ecological risk assessment.

The Superfund program has also issued supplementary documents, including:

- *Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors"* (U.S. EPA, 1991a)
- *Supplemental Guidance to RAGS: Calculating the Concentration Term* (U.S. EPA, 1992d)
- *Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions* (U.S. EPA, 1991d)
- *Use of IRIS (Integrated Risk Information System) Values in Superfund Risk Assessment* (U.S. EPA, 1993)
- *Final Soil Screening Guidance, May 17, 1996. Soil Screening User's Guide* (U.S. EPA, 1996)
- *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* (U.S. EPA, 2001c).

EPA will continue to develop Superfund guidance and tools to improve the practice of risk assessment. Superfund guidance documents are available from EPA's Superfund publications web site (<http://www.epa.gov/superfund/pubs.htm>).

The role of risk assessment in Superfund, described above, can be summarized by a number of principles that are followed and developed in *RAGS Volume 3: Part A*, including:

- The Superfund risk assessment process should rely on early problem formulation, planning, and scoping for improved remedial investigations and feasibility studies, risk assessments, and risk management decisions.
- The use of a tiered process in Superfund risk assessment and management is beneficial in that it promotes an efficient allocation of resources and improved decision-making.
- Early and continuing involvement of stakeholders throughout the Superfund risk assessment process provides an opportunity to build stakeholder trust and meet stakeholder needs, which can result in improved risk assessments and faster, more-informed risk management decisions.

1.1.4 PROBABILISTIC RISK ASSESSMENT AND ITS ROLE IN SUPERFUND

RAGS Volume I (U.S. EPA, 1989a) and supporting guidance describe a point estimate approach to risk assessments in the Superfund program. Point estimate risk assessments use single values (point estimates) to represent variables in a risk equation. The output of the risk equation in a point estimate risk assessment is, therefore, a point estimate of risk, which can be a central tendency exposure (CTE) estimate of risk (e.g., the average expected risk) or reasonable maximum exposure (RME) estimate of risk (e.g., the risk expected if the RME was to occur), depending on the input values used in the risk equation. *RAGS Volume 3: Part A* describes a probabilistic approach to risk assessment. Probabilistic risk assessment uses probability distributions for one or more variables in a risk equation in order to quantitatively characterize variability and/or uncertainty. The output of a PRA is a probability distribution of risks that reflects the combination of the input probability distributions. If the input

distributions represent variability, then the output risk distribution can provide information on variability in risk in the population of concern. If the input distributions reflect uncertainty, then the output risk distribution can provide information about uncertainty in the risk estimate. Information from a PRA can be used to make statements about the likelihood of exceeding a risk level of concern, given the estimated variability in elements of the risk equation. Since the results of point estimate methods generally do not lend themselves to this level of risk characterization (e.g., quantitative uncertainty assessment), PRA can provide unique and important supplemental information that can be used in making Superfund risk management decisions at Superfund sites.

Monte Carlo Analysis (MCA) is perhaps the most widely used probabilistic method in PRA. MCA is a specific probabilistic method that uses computer simulation to combine multiple probability distributions in a risk equation (see Section 1.2.2 for further discussion of Monte Carlo simulation). Monte Carlo methods have been in use in modeling since 1946 when Stanislaw Ulam used MCA to conduct uncertainty analysis at Los Alamos during the conceptual stage of the hydrogen bomb project. The history of the use of MCA (from the 1940s to the present) can be found in Rugen and Callahan, 1996.

The application of probabilistic analysis to human health and ecological risk assessment is a relatively recent development that was facilitated by development of statistical sampling techniques to obtain a probabilistic approximation to the solution of a mathematical equation and/or model, and increased speed and capacity of modern computers which can support the intensive computational requirements of MCA. Desktop computers and commercial software are currently available which enable risk assessors to make, in minutes, PRA calculations that only a few years ago would have required days.

The potential value of PRA to support risk-based decisions has become increasingly apparent over the last several years. This has prompted the need for appropriate policy and guidance documents that define the role of PRA in the Superfund program and that promote and facilitate the highest quality and consistent application of PRA in the Program where appropriate. EPA previously issued guidance that addresses the use of quantitative uncertainty analysis in risk assessment. *RAGS Volume I* (U.S. EPA, 1989a) and the *Final Guidelines for Exposure Assessment Guidelines* (U.S. EPA, 1992a) emphasize the importance of assessing variability and uncertainty in risk estimates conducted in the Superfund program. Guidance is also available for characterizing the 95% upper confidence limit (UCL) for the mean exposure concentration (U.S. EPA, 1992d, 1997f). At the regional level, EPA Regions 3 and 8 issued guidance on the appropriate use of probabilistic methods in risk assessment (U.S. EPA, 1994b, 1995e). The importance of adequately characterizing variability and uncertainty is addressed in the 1995 memorandum on *Risk Characterization Policy and Guidance* (U.S. EPA, 1995b). In the spring of 1997, EPA released the memorandum, *Use of Probabilistic Techniques (including Monte Carlo Analysis) in Risk Assessment* (U.S. EPA, 1997g). According to the Policy Statement of the memorandum, probabilistic analysis techniques, "given adequate supporting data and credible assumptions, can be viable statistical tools for analyzing variability and uncertainty in risk assessments." As such, a PRA, "will be evaluated and utilized in a manner that is consistent with other risk assessments submitted to the Agency." Along with this Policy Statement, the Agency released a set of guiding principles for use and review of probabilistic analyses (U.S. EPA, 1997g). Hence, both RAGS and Agency-wide guidance emphasize the importance of review of the scientific and technical merit of a probabilistic analysis to determine whether or not the assessment is of sufficient quality to support a remedial decision.

Currently, EPA's Office of Emergency and Remedial Response (OERR) is implementing PRA as part of its Superfund reform activities. This guidance, *RAGS Volume 3: Part A*, provides risk assessors with comprehensive guidance on when and how it may be appropriate to conduct PRAs using Monte

Carlo analysis within the Superfund program. It describes basic concepts in PRA, an approach for conducting MCA, and EPA's policy for implementing PRA in the Superfund program. The Agency also intends to supplement this guidance with additional examples and case studies in PRA (see Section 1.6).

1.2 BASIC CONCEPTS OF PROBABILISTIC RISK ASSESSMENT

This section describes what a PRA is and compares and contrasts it to the more familiar point estimate methods for human health risk assessment (U.S. EPA, 1989a) and ecological risk assessment (U.S. EPA, 1997a). A risk assessment performed using probabilistic methods is very similar in concept and approach to the point estimate method, with the main difference being the methods used to incorporate variability and uncertainty into the risk estimate. A variety of modeling techniques can be used to characterize variability and uncertainty in risk. This guidance focuses on MCA, perhaps the most common probabilistic method that risk assessors will encounter. Basic concepts on how to use MCA to propagate variability and uncertainty in exposure through a risk model are presented. Many of the concepts presented in this guidance are applicable to other probabilistic approaches to risk assessment.

At some sites, probabilistic analysis can provide a more complete and transparent characterization of the risks and uncertainties in risk estimates than would otherwise be possible with a point estimate approach. However, a PRA is not necessary or desirable for every site. The tiered approach presented in Chapter 2 highlights important scientific and management decisions for determining if PRA is appropriate at a specific site. The decision to perform PRA is appropriate only after the risk assessor and the remedial project manager (RPM) at the site determine whether a PRA will enhance decision making at the site. If a PRA is conducted, the assumptions and inputs to the probabilistic model should be sufficiently documented so that the results can be independently reproduced.

An essential concept in PRA that will be important throughout this section and the rest of the guidance is the distinction between "variability" and "uncertainty". **Variability** refers to true heterogeneity or diversity. For example, among a population that drinks water from the same source and with the same contaminant concentration, the risks from consuming the water may vary. This may be due to differences in exposure (i.e., different people drinking different amounts of water, having different body weights, exposure frequencies, and exposure durations) as well as differences in response (e.g., genetic differences in resistance to a chemical dose). Differences among individuals in a population are referred to as inter-individual variability, while differences for one individual over time are referred to as intra-individual variability.

Uncertainty occurs because of a lack of knowledge. For example, we can be very certain that different people drink different amounts of water, but we may be uncertain about how much variability there is in water intakes among the population. Uncertainty can often be reduced by collecting more and better data, while variability is an inherent property of the population being evaluated. Variability can be better characterized with more data, but it cannot be reduced or eliminated.

Sometimes there can be confusion about whether data are representative of variability or uncertainty, especially when the distinction depends on how the problem is framed. For example, one of the exposure variables that may be considered in a risk assessment of workers exposed via inhalation to an indoor air contaminant is the fraction of time spent indoors on site. Assume that time-activity information is available from surveys of a representative population of workers. This data set may be used to define a probability distribution (e.g., empirical, normal) that characterizes inter-individual

variability in exposure times among workers. Sources of uncertainty would include the choice of the probability distribution used to characterize variability, as well as the parameter estimates that are based on a finite data set. Using the same data set, uncertainty in a parameter, such as the arithmetic mean exposure time, may also be defined by a probability distribution. Efforts to clearly distinguish between variability and uncertainty are important for both risk assessment and risk communication. Section 1.2.4 and Chapter 3, Section 3.4 present an overview of the different sources of uncertainty. Guidance on selecting and fitting probability distributions is given in Appendices B and C, and advanced methods for characterizing both variability and uncertainty are discussed in Appendix D.

1.2.1 WHAT IS PRA?

Probabilistic risk assessment is a general term for risk assessments that use probability models to represent the likelihood of different risk levels in a population (i.e., variability) or to characterize uncertainty in risk estimates.

A risk assessment performed using probabilistic methods would rely on the same fundamental exposure and risk equations as do point estimate approaches. U.S. EPA guidance, including *RAGS Volume I: Part A* (U.S. EPA, 1989a), the *Standard Default Exposure Factors Guidance* (U.S. EPA, 1991a), *Supplemental Guidance for Developing Soil Screening Levels* (U.S. EPA, 2001c), and *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (U.S. EPA, 1997a) present methods for estimating risk using standardized exposure and risk models. Examples of typical exposure and risk equations that would be used in risk calculations, in this case, for a drinking water exposure scenario, are provided in Exhibit 1-3:

EXHIBIT 1-3

CANCER AND NONCANCER RISK MODELS

Exposure Model:

$$CDI = \frac{C \times IR \times EF \times ED}{BW \times AT}$$

Cancer Risk Model:

$$Risk = CDI \times CSF$$

Noncancer Risk Model:

$$HQ = \frac{CDI}{RfD}$$

CDI	=	chronic daily intake of the chemical (mg/kg-day)
C	=	concentration of the chemical in an exposure medium (e.g., mg/L)
IR	=	ingestion rate (e.g., L/day for water, mg/day for soil, etc.)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
BW	=	body weight (kg)
HQ	=	hazard quotient
AT	=	averaging time (equal to ED x 365 days/year for noncarcinogens and 70 years x 365 days/year for carcinogens)
CSF	=	cancer slope factor (linear low-dose cancer potency factor) for the chemical (mg/kg-day) ⁻¹
RfD	=	reference dose for the chemical for assessing noncancer health effects (mg/kg-day)

In the point estimate approach, a single numerical value (i.e., point estimate) is chosen for each variable shown in Exhibit 1-3. For example, point estimates may include a drinking water ingestion rate of 2 L/day and a body weight of 70 kg for an adult. Based on the choices that are made for each individual variable, a single estimate of risk is calculated. In the probabilistic approach, inputs to the risk equation are described as *random variables* (i.e., variables that can assume different values for different receptors in the population) that can be defined mathematically by a probability distribution. For continuous random variables, such as those in Figure 1-1 (body weight), the distribution may be described by a PDF, whereas for discrete random variables (e.g., number of fish meals per month), the distribution may be described by a probability mass function (PMF). The key feature of PDFs and PMFs is that they describe the range of values that a variable may assume, and indicate the relative likelihood (i.e., probability) of each value occurring within that range for the exposed population. For example, the distribution of tap water ingestion (mL/day) among the general U.S. population might be characterized by a lognormal distribution with a log-mean of 6.86 and a log-standard deviation of 0.575 (Table 3-11 of U.S. EPA 1997b). One might use a PDF to show how approximately half the population drinks more than 1 L/day of tap water, but only 10% of the population drinks more than 2 L/day. After determining appropriate PDF types and parameter values for selected variables, the set of PDFs is combined with the toxicity value in the exposure and risk equations given in Exhibit 1-3 to estimate a distribution of risks. Guidance on selecting and fitting distributions for variables in risk equations is provided in Appendix B.

In human health risk assessments, probability distributions for risk should reflect variability or uncertainty in exposure. In ecological risk assessments, risk distributions may reflect variability or uncertainty in exposure and/or toxicity (see Sections 1.4 and 1.4.1, Item 3).

A continuous probability distribution can be displayed in a graph in the form of either a PDF or corresponding CDF; however, for clarity, it is recommended that both representations be presented in adjacent (rather than overlaid) plots. Figure 1-1 illustrates a PDF and CDF for a normal probability distribution for adult body weight. Both displays represent the same distribution, but are useful for conveying different information. Note that it is helpful to include a text box with summary statistics relevant to the distribution (e.g., mean, standard deviation). The types of information that PDFs and CDFs are most useful for displaying are presented in Exhibit 1-4.

EXHIBIT 1-4

USE A PDF AND CDF TO DISPLAY:

PDF

- The relative probability of values
- The most likely values (e.g., modes)
- The shape of the distribution (e.g., skewness, kurtosis, multimodality)
- Small changes in probability density

CDF

- Percentiles, including the median
- High-end risk range (e.g., 90th to 99th percentiles)
- Confidence intervals for selected percentiles
- Stochastic dominance (i.e., for any percentile, the value for one variable exceeds that of any other variable)

Source: U.S. EPA, 1997g

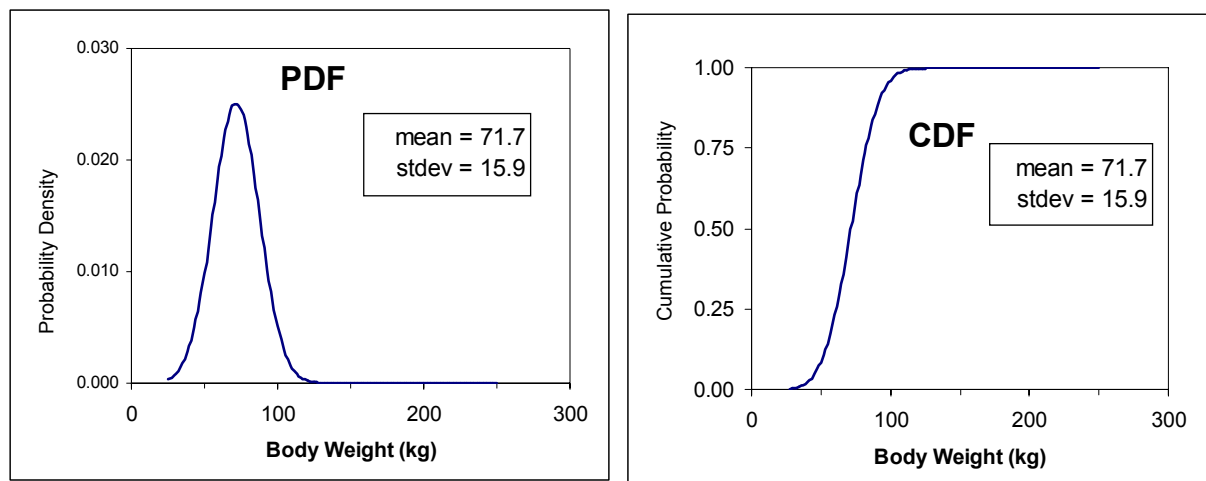



Figure 1-1. Example of a normal distribution that characterizes variability in adult body weight (males and females combined). Arithmetic mean=71.7 kg, standard deviation=15.9 kg (Finley and Paustenbach, 1994). Body weight may be considered a continuous random variable. The left panel shows a bell-shaped curve and represents the PDF, while the right panel shows an S-shaped curve and represents the CDF. Both displays represent the same distribution (including summary statistics), but are useful for conveying different information.

The CDF for risk can be especially informative for illustrating the percentile corresponding to a particular risk level of concern (e.g., 95th percentile=1E-06). A text box may also be included on the graph to highlight important summary statistics, such as the parameters of the input distribution, or selected percentiles of the output distribution for risk. For example, a clear description of the parameters for the probability distribution should be given, as well as an indication of whether the distribution represents variability or uncertainty.

1.2.2 WHAT IS A MONTE CARLO SIMULATION?

Perhaps the most common numerical technique for PRA is Monte Carlo simulation. Monte Carlo simulation has been widely used to explore problems in many disciplines of science as well as engineering, finance, and insurance (Rugen and Callahan, 1996). The process for a Monte Carlo simulation is illustrated in Figure 1-2. In its general form, the risk equation can be expressed as a function of multiple exposure variables (V_i) and a toxicity term: $\text{Risk} = f(V_1, V_2, \dots, V_n) \times \text{Toxicity}$. Solutions for equations with PDFs are typically too complex for even an expert mathematician to calculate the risk distribution analytically. However, numerical techniques applied with the aid of computers can provide very close approximations of the solution. This is illustrated here for the simplified case in which the assessment variables are statistically independent, that is, the value of one variable has no relationship to the value of any other variable. In this case, the computer selects a value for each variable (V_i) at random from a specified PDF and calculates the corresponding risk. This process is repeated many times (e.g., 10,000), each time saving the set of input values and corresponding estimate of risk. For example, the first risk estimate might represent a hypothetical individual who drinks 2 L/day of water and weighs 65 kg, the second estimate might represent someone who drinks 1 L/day and weighs 72 kg, and so forth. Each calculation is referred to as an iteration, and a set of iterations is called a simulation.

 *A convenient aid to understanding the Monte Carlo approach for quantifying variability is to visualize each iteration as representing a single individual and the collection of all iterations as representing a population.*

Each iteration of a Monte Carlo simulation should represent a plausible combination of input values (i.e., exposure and toxicity variables), which may require using bounded or truncated probability distributions (see Appendix B). However, risk estimates are not intended to correspond to any one person. The “individuals” represented by Monte Carlo iterations are virtual and the risk distributions derived from a PRA allow for inferences to be made about the likelihood or probability of risks occurring within a specified range for an exposed human or ecological population. A simulation yields a set of risk estimates that can be summarized with selected statistics (e.g., arithmetic mean, percentiles) and displayed graphically using the PDF and CDF for the estimated risk distribution. Often the input distributions are assumed to be independent, as shown in Figure 1-2. More complex Monte Carlo simulations can be developed that quantify a dependence between one or more input distributions by using conditional distributions or correlation coefficients (see Appendix B, Section B.5.5 for a discussion of correlated input distributions).

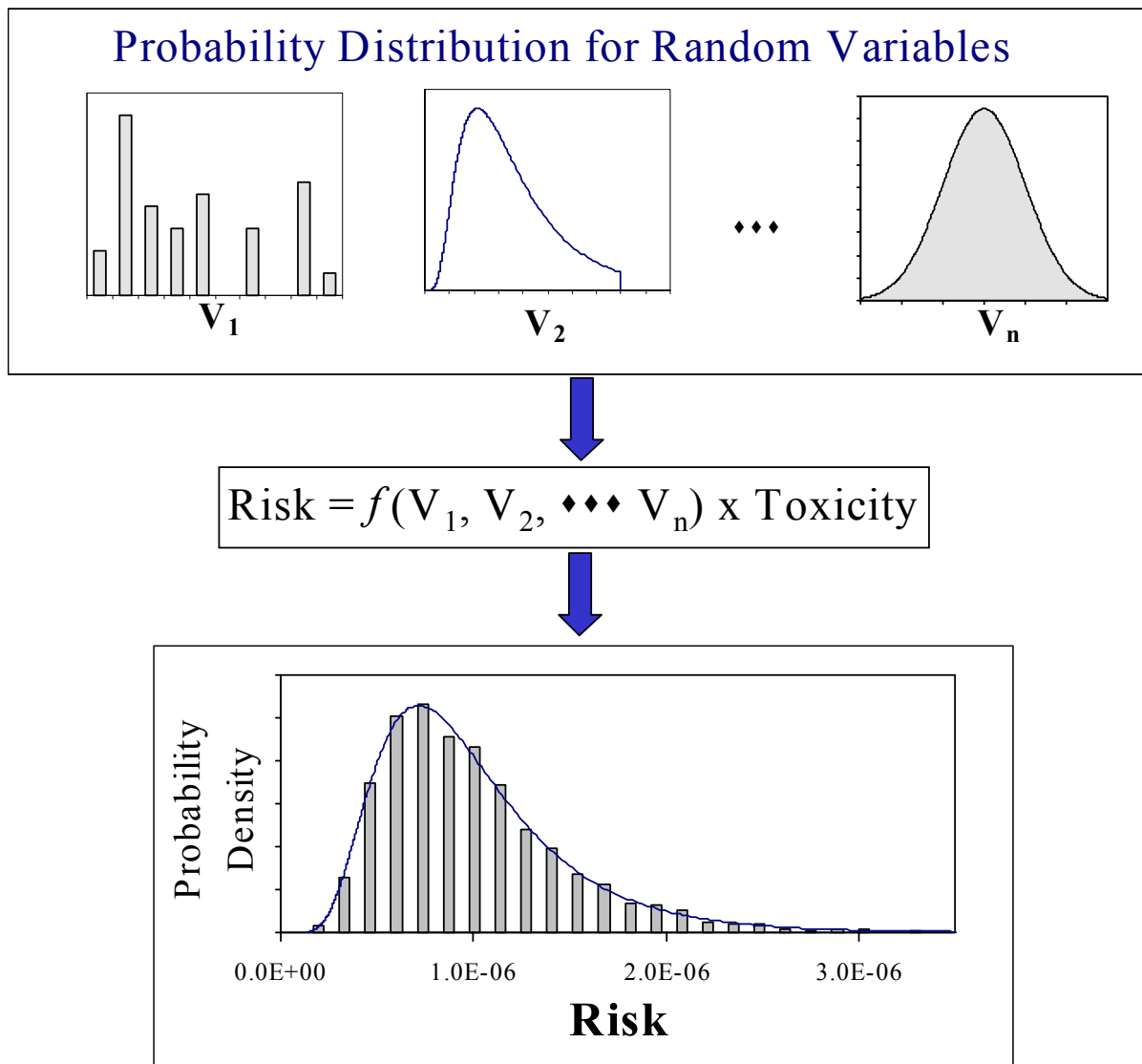


Figure 1-2. Conceptual model of Monte Carlo analysis. Random variables ($V_1, V_2, \dots V_n$) refer to exposure variables (e.g., body weight, exposure frequency, ingestion rate) that are characterized by probability distributions. A unique risk estimate is calculated for each set of random values. Repeatedly sampling (V_i) results in a frequency distribution of risk, which can be described by a PDF. In human health risk assessments, the toxicity term should be expressed as a point estimate. In ecological risk assessment (see Sections 1.4 and 1.4.1) the toxicity term may be expressed as a point estimate or as a probability distribution.

The rapid evolution in computing power has greatly reduced concerns among regulators regarding the number iterations needed in MCA.

☞ While this guidance does not prescribe specific criteria or set an arbitrary “minimum” number of iterations needed for PRA, a general rule of thumb is that a sufficient number of iterations should be run to obtain numerical stability in percentiles of the output (e.g., risk distribution) that are important for decision making.

Numerical stability refers to the stochastic variability, or “wobble” associated with random sampling, and can be evaluated by running multiple simulations with the same set of input assumptions and calculating the average percent change in a specified percentile of the output (e.g., Maddalena et al., 2001). For example, it may be determined that 5,000 iterations are sufficient to achieve numerical stability in the 50th percentile, but insufficient for the 95th percentile risk estimate when a criteria of $\pm 1\%$ is applied for multiple simulations. As discussed in Section 1.4, one of the eight conditions specified by EPA for the acceptance of PRA is that the numerical stability of the output be presented and discussed, since it will vary depending on what percentile of the risk distribution is evaluated. While some commercial software now have a feature to automatically stop simulations after a specified criterion for numerical stability is achieved (Burmaster and Udell, 1990), care should be taken to understand how this criterion is implemented across the entire range of the output distribution.

1.2.3 WHY IS VARIABILITY IMPORTANT IN RISK ASSESSMENT? HOW IS IT ADDRESSED BY THE POINT ESTIMATE AND PROBABILISTIC APPROACHES?

As noted previously, variability refers to true heterogeneity or diversity that occurs within a population or sample. Factors that lead to variability in exposure and risk include variability in contaminant concentrations in a medium (air, water, soil, etc.), differences in ingestion rates or exposure frequencies, or in the case of ecological assessments, inter- and intra-species variability in dose-response relationships. *Risk Assessment Guidance for Superfund Volume I* (Section 6.1.2 of U.S. EPA, 1989a) and the *NCP Preamble* (U.S. EPA, 1990) state that human health risk management decisions at Superfund sites will generally be based on an individual that has RME. Likewise, RME estimates of risk are the most appropriate basis for decision making using an ecological risk assessment. Use of the RME and CTE risk descriptors in ecological risk assessment are discussed in Chapter 4. The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures based on both quantitative information and professional judgment (Sections 6.1.2 and 6.4.1 of U.S. EPA, 1989a). In addition, the Agency released guidance in 1992 (U.S. EPA, 1992c) recommending the inclusion of a “central tendency” exposure estimate to an individual, as well as a high-end exposure estimate, in the risk assessment. Generally, the CTE is considered to be a measure of the mean or median exposure. The difference between the CTE and the RME gives an initial impression of the degree of variability in exposure or risk between individuals in an exposed population.

Depending on assessment needs at a site, a range of point estimates of risk can be developed to represent variability in exposures. To support the evaluation of RME risk estimates using the point estimate approach described in Section 1.3, the Superfund program developed guidance with recommended default values for exposure variables as inputs to the risk equations (U.S. EPA, 1992a, 1996, 1997a, 2001d). These standardized values are a combination of average (e.g., body weight, skin surface area) and high-end exposure assumptions (e.g., drinking water intake, exposure duration). A CTE risk estimate is based on central estimates (e.g., mean, 50th percentile) for each of the exposure variables.

Available site-specific data on plausible mean and upper range values for exposure variables should be used to support CTE and RME risk estimates. The point estimate approach to risk assessment does not determine where the CTE or RME risk estimates lie within the risk distribution. For example, the RME risk estimated with the point estimate approach could be the 90th percentile, the 99.9th percentile, or some other percentile of the risk distribution. Without knowing what percentile is represented by the RME risk estimate, the risk manager might be unsure about the likelihood of the RME risk occurring or being exceeded in the receptor population and about what level of remedial action is justified or necessary to achieve the protective objectives of CERCLA.

In a PRA, distributions used as inputs to the risk equations can characterize the inter-individual variability inherent in each of the exposure assumptions. By characterizing variability with one or more input distributions, the output from the Monte Carlo simulation is a distribution of risks that could occur in that population (Figure 1-3). The central tendency of the risk distribution (e.g., arithmetic mean, geometric mean, 50th percentile) may be characterized as the CTE risk estimate. Similarly, the high-end of the risk distribution (e.g., 90th to 99.9th percentiles) is representative of exposures to the RME individual. In addition to providing a better understanding of where the CTE and RME risks occur in the distribution, a PRA can also provide an estimate of the probability of occurrence associated with a particular risk level of concern (e.g., cancer risk of 1E-05). A PRA that quantifies variability can be used to address the question, “What is the likelihood (i.e., probability) that risks to an exposed individual will exceed 1E-05?” Based on the best available information regarding exposure and toxicity, a risk assessor might conclude, “The estimated distribution for variability in risk across the target population indicates that 10% of the individuals exposed under these circumstances have a risk exceeding 1E-05.” This type of evaluation can be achieved using a technique known as one-dimensional Monte Carlo Analysis (1-D MCA). Guidelines for interpreting the high-end of the risk distribution in terms of the RME risk estimate are discussed further in Section 1.4.1 and Chapter 7.

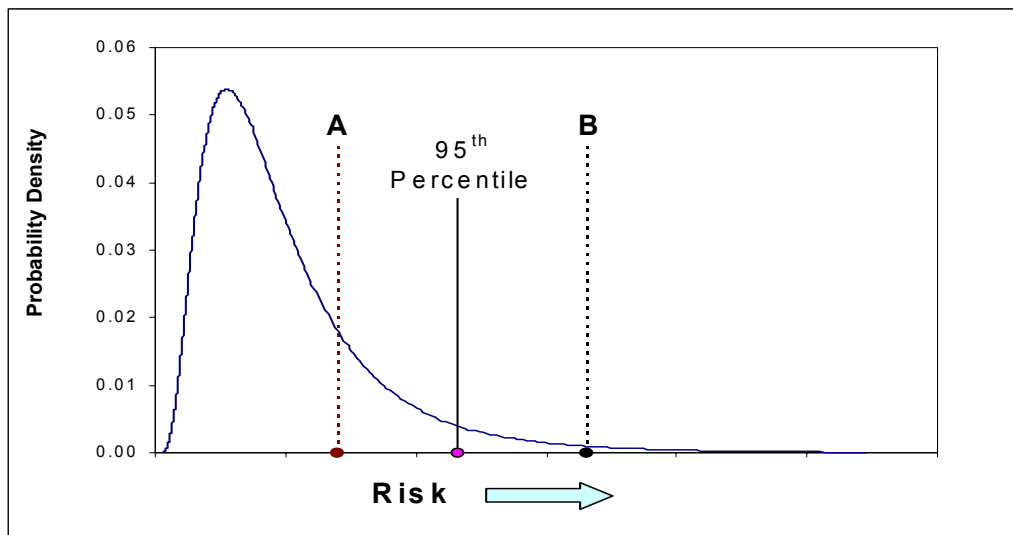


Figure 1-3. Example of a probability distribution for risk illustrating the 95th percentile and two different risk levels of concern (A and B). Assuming the 95th percentile corresponds to the RME, the need for remedial action depends on how the RME risk compares with the risk level of concern. For Case A (RME > level of concern), remedial action may be warranted. For Case B (RME < level of concern), remedial action may be unnecessary.

The agreement (or lack of agreement) between the results of the point estimate calculations and the PRA calculations is expected to vary as a function of the form of the exposure or risk model and the attributes of the input variables. In general, if the terms in the denominator of the exposure or risk equation have low variability and do not approach zero, then the CTE point estimate is likely to agree quite well with the arithmetic mean from the PRA simulation, and the RME point estimate is likely to correspond to the high-end of the risk distribution (see discussion of RME range in Section 1.2.5). However, if the exposure or risk model has terms in the denominator that are a significant source of variability, or if the terms approach zero, then the agreement between the point estimate values and the PRA values may be more substantial. In addition, since the RME point estimate of risk reflects a combination of central tendency and high-end input values, it is difficult to anticipate what percentile of a distribution of variability it represents.

☞ If results of PRA calculations differ substantially from point estimate calculations, a risk manager may benefit from understanding the reasons for the differences and the relative strengths of the different approaches.

Since point estimate and PRA approaches may yield different estimates of CTE and RME risks, the two approaches also may support different risk management decisions. This does not imply that either approach is invalid. Likewise, a correspondence between the point estimate and PRA results does not imply a greater accuracy or certainty in the modeling assumptions and inputs. Simply stated, PRA, based on the same risk equations and data as the point estimate approach, provides a different means of characterizing variability and uncertainty. Potential sources of variability and uncertainty in risk estimates should be identified, discussed, and to the extent practicable, quantified. Advantages and disadvantages of PRA and point estimate risk assessment are discussed in Section 1.2.4 and 1.3.

1.2.4 WHY IS UNCERTAINTY IMPORTANT IN RISK ASSESSMENT? HOW IS UNCERTAINTY ADDRESSED BY THE POINT ESTIMATE AND PROBABILISTIC APPROACHES?

Uncertainty derives from a lack of knowledge. Various taxonomies of uncertainty relevant to risk assessment have been presented (Finkel, 1990; Morgan and Henrion, 1990; Cullen and Frey, 1999). U.S. EPA guidance, including the *Final Guidelines Exposure Assessment Guidelines* (U.S. EPA, 1992a), *Exposure Factors Handbook* (U.S. EPA, 1997b,c,d), and *Guiding Principles for Monte Carlo Analysis* (U.S. EPA, 1997g) describe a variety of different types of uncertainty in risk assessment as well as modeling strategies for quantifying uncertainties. Potential sources of uncertainty in risk assessment can be divided into one of three broad categories:

- (1) *Parameter uncertainty* - uncertainty in an estimate of an input variable in a model. In PRA, this may refer specifically to a statistical concept of uncertainty in estimates of population parameters (e.g., arithmetic mean, standard deviation) from random samples, due to the quality, quantity, and representativeness of available data as well as the statistical estimation method.
- (2) *Model uncertainty* - uncertainty about a model structure (e.g., exposure equation) or intended use, including the relevance of simplifying assumptions to the endpoint of the risk assessment, the choice of probability distribution to characterize variability, and interpolation or extrapolation beyond the scale used to calibrate a model from empirical data.

- (3) *Scenario uncertainty* - uncertainty regarding missing or incomplete information to fully define exposure. This may include descriptive errors regarding the magnitude and extent of chemical exposure or toxicity, temporal and spatial aggregation errors, incomplete analysis (i.e., missing exposure pathways), and potential mis-specification of the exposed population or exposure unit.

Sources of uncertainty described by these categories are important because they can influence risk management decisions in both point estimate and probabilistic risk assessment. As additional sources of uncertainty are quantified and included in the risk assessment, uncertainty in risk estimates may appear to increase, suggesting there may be little confidence in a risk management decision. This situation may appear to be counterintuitive for those managers who expect confidence to increase as uncertainty is quantified. However, as discussed below and in Chapter 6 (see Section 6.4.2), uncovering and quantifying these sources of uncertainty may help to provide perspective, and make the decisions using the tiered process more transparent. In PRA, there are a variety of methods that can be used to effectively quantify uncertainty as well as communicate confidence in risk estimates (see Chapter 3, Section 3.4; Chapter 6, Section 6.4, and Section 6.5).

Parameter uncertainty may be the most readily recognized source of uncertainty that is quantified in site-specific risk assessments at hazardous waste sites. Parameter uncertainty can occur in each step of the risk assessment process from data collection and evaluation, to the assessment of exposure and toxicity. Sources of parameter uncertainty may include systematic errors or bias in the data collection process, imprecision in the analytical measurements, inferences made from a limited database when that database may or may not be representative of the variable under study, and extrapolation or the use of surrogate measures to represent the parameter of interest.

In the point estimate approach, parameter uncertainty is addressed in a qualitative manner for most variables. For example, the uncertainty section of a point estimate risk assessment document might note that a soil sampling plan yielded a small sample size that may not be representative of overall contaminant concentrations and, as a result, the risk estimate may over- or under-estimate actual risk. Uncertainty in the concentration term is addressed quantitatively to a limited extent in a point estimate approach by using the 95% UCL for the arithmetic mean concentration in both CTE and RME risk estimates; this accounts for uncertainty associated with environmental sampling and site characterization (U.S. EPA, 1992d, 1997f). The 95% UCL is combined in the same risk calculation with various central tendency and high-end point estimates for other exposure factors.

Some examples of the models that EPA uses in the risk assessment process are the equations used to calculate exposure and risk, the linearized multistage model used to estimate cancer dose-response relationships, and media-specific models to estimate contaminant concentrations. All models are simplified, idealized representations of complicated physical or biological processes. Models can be very useful from a regulatory standpoint, as it is generally not possible to adequately monitor long term exposure for populations at contaminated sites. However, models that are too simplified may not adequately represent all aspects of the phenomena they were intended to approximate or may not capture important relationships among input variables. Other sources of model uncertainty can occur when important variables are excluded, interactions between inputs are ignored, or surrogate variables that are different from the variable under study are used.

In most probabilistic assessments, the first step of analysis is usually an analysis of variability in exposure or risk. However, PRA methods may also be used to characterize uncertainty around the best estimate of the exposure or risk distribution. This is done using "2-dimensional" MCA (2-D MCA) (see Appendix D). One convention that has been used to distinguish between probability distribution

functions for variability and uncertainty is to use subscripts "v" and "u" to indicate PDFs that characterize variability (PDFv) or uncertainty (PDFu). Figure 1-4 shows an example of the results of this type of 2-D MCA. This analysis can provide a quantitative measure of the *confidence in the fraction of the population with a risk exceeding a particular level*; which is sometimes referred to as a *vertical confidence interval* (Figure 1-4). For example, a conclusion based on this type of output might be, "While the best estimate for the variability distribution for risk across the target

population indicates that 10% of the individuals exposed under these circumstances have a risk exceeding 1E-06, the uncertainty is such that we can only be reasonably certain (e.g., 95% sure) that no more than 20% of the exposed population has a risk that exceeds 1E-06."

Additionally, the output from a 2-D MCA can provide a quantitative measure of the *confidence in the risk estimate* for a particular fraction of the population; which is sometimes referred to as a *horizontal confidence interval*. This type of output might support the following type of conclusion, "While the best estimate for the variability distribution for risk across the target population indicates that 10% of the individuals exposed under these circumstances have a risk exceeding 1E-06, the uncertainty is such that we can only be reasonably certain (e.g., 95% sure) that the risk for this group of individuals does not exceed 2E-06." The term "confidence interval" is used loosely in this context to convey information about uncertainty; however, it is not the same as a statistical confidence interval that one might obtain by estimating a population parameter from a sample. The vertical and horizontal bars shown in Figure 1-4 represent a range of possible estimates for the percentile given one or more sources of uncertainty that were included in the simulation. If the target audience for this graphic has a greater understanding of statistics, it may be less confusing if alternative phrases are used to describe the results, such as "credible interval" or "probability band".

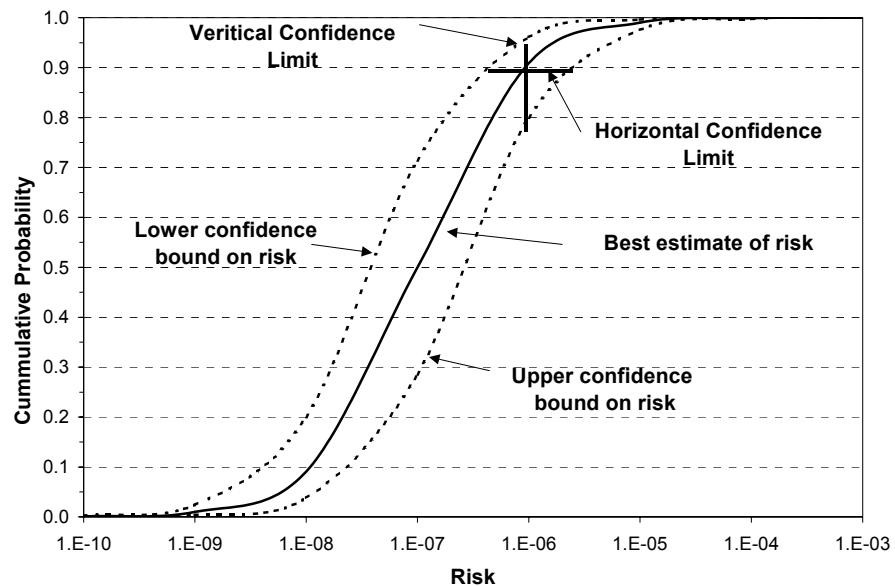


Figure 1-4. Illustration of "Vertical" and "Horizontal" Confidence Intervals (or limits) on a risk estimate. This type of output can be produced from a 2-D MCA in which probability distributions of uncertainty are introduced into the risk equation. See Chapter 3 and Appendix D for further discussion of 2-D MCA in quantitative uncertainty analysis.

In general, one should avoid developing input distributions to a PRA model that yield a single risk distribution that intermingles, or represents both variability and uncertainty. By separately characterizing variability and uncertainty, the output from a PRA will be easier to understand and communicate. A number of tools can aid in evaluating the uncertainty in estimated distributions for variability. Both simple and very complex approaches have been applied to this problem. Two basic

methods for quantifying variability and parameter uncertainty simultaneously are described in Exhibit 1-5. PRAs that use these approaches can provide quantitative estimates of uncertainty in percentiles of the risk distribution based on confidence intervals or credible intervals for one or more parameter estimates. Techniques for characterizing both variability and uncertainty in PRA are discussed in more detail in Chapters 3, 4, 5, and 7, and Appendices A, C, and D.

A common apprehension concerning the utility of PRA is that it may require more information and data than are available to generate credible PDFs. Risk assessors may feel that they can't specify a PDF because they don't have enough information to choose a distribution type, estimate parameters, or evaluate the representativeness to the site population of concern. However, if sufficient information exists to support a meaningful point estimate evaluation (i.e., if some sort of central tendency and upper bound values are available for each input variable), then it is usually possible to perform a screening level, or preliminary 1-D MCA that may provide additional useful information regarding variability. Likewise, an initial two-dimensional analysis may be performed that does not require collection of any new data, but simply characterizes uncertainty in the existing data. The results of such a 2-D MCA can help to identify the main sources of uncertainty in the risk results, and can support decisions to collect more data and/or proceed with additional tiers of analysis in order to improve the assessment. As with a preliminary 1-D MCA, the decision to conduct a more advanced probabilistic analysis does not always result in added data requirements.

EXHIBIT 1-5

QUANTIFYING VARIABILITY AND UNCERTAINTY

1. Single source of uncertainty

Run multiple one-dimensional Monte Carlo simulations (1-D MCA) in which each simulation uses a different point estimate for a parameter selected from an uncertainty distribution, combined with PDFv's for one or more variables. For example, separate simulations can be run in which the mean of the exposure concentration variability distribution is represented by either the 95% lower or upper confidence limit on the mean. A comparison of the output of these simulations would provide a partial characterization of the quantitative impact of uncertainty in the mean exposure concentration on the risk estimate (provided that certain conditions hold; i.e., risk increases with increasing exposure concentration) (see Chapter 3, Section 3.3.1).

2. Multiple sources of uncertainty

Run a single two-dimensional Monte Carlo simulation (2-D MCA), in which separate probability distributions are specified for variability and parameter uncertainty and values from these distributions are randomly selected and used in each iteration of the Monte Carlo simulation (see Appendix D).

Use of probabilistic methods (e.g., MCA) to propagate variability and uncertainty through risk models offers five key advantages over point estimate approaches in addressing uncertainty in risk estimates:

- (1) Probabilistic methods may often provide a more complete and informative characterization of variability in exposure or risk than is usually achievable using point estimate techniques.
- (2) Probabilistic methods can provide a more quantitative expression of the confidence in risk estimates than the point estimate approach.
- (3) Sensitivity analysis methods using PRA may help risk assessors to better identify influential exposure factors.

- (4) Probabilistic methods can account for dependencies between input variables (e.g., body weight and skin surface area).
- (5) Probabilistic methods provide quantitative estimates of the expected value of additional information that might be obtained from data collection efforts (Morgan and Henrion, 1990). The importance of quantifying uncertainty in an *expected value of information* (EVOI) framework is discussed in Appendix D.

Since both point estimate and probabilistic approaches in risk assessment are applied to the same conceptual models (i.e., the same exposure and risk models), uncertainties in the conceptual model are generally addressed in the same manner. If other models are available to explain or characterize a given phenomenon, the risk estimates associated with each of those conceptual models could be compared to determine the sensitivity of the risk to the uncertainty in the choice of a model (see Chapter 2 and Appendix A). For example, when deciding on a contaminant concentration term for tetrachloroethylene in groundwater for a residential exposure assessment 10 years in the future, it would be appropriate to compare and contrast several fate and transport models and their results before deciding on a concentration term.

1.2.5 REASONABLE MAXIMUM EXPOSURE AT THE HIGH-END

Risk management decisions at Superfund sites should be based on an estimate of the risk to a reasonably maximum exposed receptor, considering both current and future land-use conditions. The RME is defined as the highest exposure that is reasonably expected to occur at a site. In general, risks corresponding to the 90th to 99.9th percentiles of the risk distribution estimated from a PRA are considered plausible high-end risks, and the RME risk should be selected within this range (see Section 1.2.4, Section 1.4.1, and Chapter 7 for further discussion). In comparison with point estimate risk assessments, PRA can provide the entire range of estimated risks as well as the likelihood of values within the range (i.e., the frequency distribution)

As noted in Chapter 7, estimates of risk become more uncertain at very high percentiles (e.g., the 99.9th), so results of PRA calculations at these extreme values should be used with caution. Risk frequency distributions toward the 99.9th percentile may be numerically unstable due to the uncertainties embedded in the input exposure assumptions. This guidance recommends that a risk manager select the RME in consultation with a risk assessor. One item for discussion should be the numerical stability of the high-end RME risk value (i.e., a stable value on the frequency distribution within the high-end range that could be reproduced in successive Monte Carlo simulations.)

1.3 ADVANTAGES AND DISADVANTAGES OF POINT ESTIMATE AND PROBABILISTIC APPROACHES

As discussed in Chapter 2, a PRA should not be conducted until adequate point estimate calculations have been completed. Once this has been done, the potential benefits of proceeding to a PRA evaluation should be based on an understanding of the potential advantages and limitations in each approach. Potential advantages and disadvantages of point estimate calculations are summarized in Exhibit 1-6 and potential advantages and disadvantages of PRA are listed in Exhibit 1-7.

In general, compared to a point estimate risk assessment, a PRA based on the same state of knowledge may offer a more complete characterization of variability in risk, can provide a quantitative evaluation of uncertainty, and may provide a number of advantages in assessing if and how to proceed to higher levels of analysis. However, there are also some real and perceived disadvantages regarding additional effort on the part of both the risk assessor and the risk manager, and the potential to cause confusion if the effort is not clearly presented.

In general, the key question to consider in deciding whether a PRA should be performed is whether or not the PRA analysis is likely to provide information that will help in the risk management decision making. For some sites, the additional information provided by a PRA will not affect the decision that would have been made with a point estimate approach alone, and a PRA will not be useful. However, when the decision whether or not to take action is not completely clear, PRA may be a valuable tool. The tiered process for PRA (Chapter 2) introduces the concept of scientific management decision points (SMDPs) to guide the complexity of analysis that may be needed for decision making. An SMDP marks a point in the process in which the potential that another analysis may influence the risk management decision is evaluated based on the problem formulation, the information available to define input variables, the results of previous analyses, and the feasibility of a subsequent analysis.

☞ *A point estimate approach is conducted for every risk assessment; a probabilistic analysis may not always be needed.*

EXHIBIT 1-6

ADVANTAGES AND DISADVANTAGES OF POINT ESTIMATE APPROACH

Advantages

- Calculations are simple and do not require any advanced software.
- EPA has established default inputs and methods to help standardize point estimate calculations between sites.
- Useful as a screening method—may allow risk management decisions with no additional work.
- Central tendency and RME estimates of risk provide a semi-quantitative measure of variability.
- Method is easily described and communicated.
- Requires less time to complete; not as resource intensive.

Disadvantages

- Computational simplifications may result in deviations from target values.
- Results are often viewed as “the answer”; importance of uncertainty is sometimes lost.
- Information from sensitivity analysis is generally limited to dominant exposure pathways and chemicals of concern; may not highlight the key exposure variables and uncertain parameters.
- Does not provide a measure of the probability that risk exceeds a regulatory level of concern, or the level of confidence in a risk estimate.
- Provides fewer incentives for collecting better or more complete information.
- May not utilize all available data for characterizing variability and uncertainty in risk estimates.

EXHIBIT 1-7

ADVANTAGES AND DISADVANTAGES OF PROBABILISTIC RISK ASSESSMENT

Advantages

- Can make more complete use of available data when defining inputs to the risk equation.
- Can provide a more comprehensive characterization of variability in risk estimates.
- Can provide a more comprehensive characterization of uncertainty in inputs, which may support statements regarding confidence in risk estimates. Communication of uncertainty in the risk assessment can help to build trust among stakeholders.
- Sensitivity analysis can identify the exposure variables, probability models, and model parameters that influence the estimates of risk.
- Puts the risk assessment in a *Value-of-Information* framework (see Appendix D). Can identify data gaps for further evaluation/data collection and can use wider variety of site-specific information.
- Allows available site-specific information to inform the choice of high-end percentile from the risk distribution that corresponds with RME risk.

Disadvantages

- Concepts and approaches may be unfamiliar; there is often apprehension regarding added costs and potential for inadvertent error and/or intentional misrepresentation.
- Places more burden on risk assessors to ensure the PRA is done correctly and on managers to understand and make decisions within a range of alternatives.
- May require more time and resources to select and fit probability distributions, and may require greater effort to communicate methodology and results.
- May convey false sense of accuracy when data are sparse.
- Complexities of the PRA approaches may obscure important assumptions or errors in basic exposure or risk models.
- If communication of the more complex PRA is unsuccessful, then it may generate mistrust of the assessment and risk management decisions.

1.4 CONDUCTING AN ACCEPTABLE PRA

In 1997, EPA issued a memorandum which contained its policy statement on PRA (U.S. EPA, 1997g). The 1997 EPA Policy Statement is as follows:

It is the policy of the U.S. Environmental Protection Agency that such probabilistic analysis techniques as Monte Carlo analysis, given adequate supporting data and credible assumptions, can be viable statistical tools for analyzing variability and uncertainty in risk assessments. As such, and provided that the conditions described below are met, risk assessments using Monte Carlo analysis or other probabilistic techniques will be evaluated and utilized in a manner that is consistent with other risk assessments submitted to the Agency for review or consideration. It is not the intent of this policy to recommend that probabilistic analysis be conducted for all risk assessments supporting risk management decisions. Such analysis should be a part of a tiered approach to risk assessment that progresses from simpler (e.g., deterministic) to more complex (e.g., probabilistic) analyses as the risk management situation requires. Use of Monte Carlo or other such techniques in risk assessments shall not be cause, *per se*, for rejection of the risk assessment by the Agency. For human health risk assessments, the application of Monte Carlo and other probabilistic techniques has been limited to exposure assessments in the majority of cases. The current policy, Conditions for Acceptance and associated guiding principles are not intended to apply to dose response evaluations for human health risk assessment until this application of probabilistic analysis has been studied further. In the case of ecological risk assessment, however, this policy applies to all aspects including stressor and dose-response assessment.

In support of this policy statement, EPA has outlined eight *conditions for acceptance* (in italics below), and good scientific practice of PRA. A PRA that is submitted to the Agency for review and evaluation should generally comply with each condition in order to ensure that adequate supporting data and credible assumptions are used in the assessment. These conditions are as follows:

- (1) *The purpose and scope of the assessment should be clearly articulated in a "problem formulation" section that includes a full discussion of any highly exposed or highly susceptible subpopulations evaluated (e.g., children, the elderly). The questions the assessment attempts to answer are to be discussed and the assessment endpoints are to be well defined.*
- (2) *The methods used for the analysis (including all models used, all data upon which the assessment is based, and all assumptions that have a significant impact upon the results) are to be documented and easily located in the report. This documentation is to include a discussion of the degree to which the data used are representative of the population under study. Also, this documentation is to include the names of the models and software used to generate the analysis. Sufficient information is to be provided to allow the results of the analysis to be independently reproduced.*

Possible sources of bias inherent in the input distributions should be discussed along with the expected impacts on the resulting risk estimates. For example, if a site-specific study of fish consumption indicated consumption rates are five to ten times higher than other studies from similar populations, this possible bias or inaccuracy should be discussed in the document. Computer programs should generally

be described in sufficient detail to allow the reviewer to understand all aspects of the analysis. Computer code/spreadsheets should provide adequate documentation and annotation.

- (3) *The results of sensitivity analyses are to be presented and discussed in the report. Probabilistic techniques should be applied to the compounds, pathways, and factors of importance to the assessment, as determined by sensitivity analyses or other basic requirements of the assessment.*

Sensitivity analysis is a valuable tool in any tier of a PRA.

- (4) *The presence or absence of moderate to strong correlations or dependencies between the input variables is to be discussed and accounted for in the analysis, along with the effects these have on the output distribution.*
- (5) *Information for each input and output distribution is to be provided in the report. This includes tabular and graphical representations of the distributions (e.g., probability density function and cumulative distribution function plots) that indicate the location of any point estimates of interest (e.g., mean, median, 95th percentile). The selection of distributions is to be explained and justified. For both the input and output distributions, variability and uncertainty are to be differentiated where possible.*
- (6) *The numerical stability of the central tendency and the higher end (i.e., tail) of the output distributions are to be presented and discussed.*

As discussed in Section 1.2.5, numerical stability refers to the observed numerical changes in parameters of the output distribution (e.g., median, 95th percentile) from a Monte Carlo simulation as the number of iterations increases. Because most risk equations are linear and multiplicative, distributions of risk will generally be right-skewed, and approximate a lognormal distribution. Values in the tails of the distribution typically are less stable than the central tendency, and the rate of convergence for the tails will depend on the form of the risk model, the skewness of the probability distributions selected for input variables and the numerical methods used to simulate probability distributions. Provided that appropriate numerical methods are employed, numerical stability is generally not a concern for most 1-D MCA models, which can be run with a sufficient number iterations in minutes with modern high speed computers; however, it can be an important consideration for more complex simulations, such as with 2-D MCA models.

- (7) *Calculations of exposures and risks using deterministic (e.g., point estimate) methods are to be reported if possible. Providing these values will allow comparisons between the probabilistic analysis and past or screening level risk assessments. Further, deterministic estimates may be used to answer scenario specific questions and to facilitate risk communication. When comparisons are made, it is important to explain the similarities and differences in the underlying data, assumptions, and models.*

If results of PRA calculations differ substantially from point estimate calculations, a risk manager may benefit from understanding the reasons for the differences and the relative strengths of the different approaches. Sometimes, a closer look at uncertainties in the underlying data, assumptions, and models will lead a risk assessor to revisit parts of the assessment in order to provide a more consistent basis for comparison.

- (8) *Since fixed exposure assumptions (e.g., exposure duration, body weight) are sometimes embedded in the toxicity metrics (e.g., Reference Doses, Reference Concentrations, Cancer risk factors), the exposure estimates from the probabilistic output distribution are to be aligned with the toxicity metric.*

1.4.1 KEY POLICIES FOR APPLYING PRA AT SUPERFUND SITES

EPA's recommended process for conducting an acceptable PRA generally follows the policy and guiding principles presented above. In addition, this section highlights four key policies for conducting acceptable PRAs at hazardous waste sites.

(1) *Follow the Tiered Approach to PRA*

In accordance with the *1997 EPA Policy Statement* (U.S. EPA, 1997g), this guidance recommends using a tiered approach when considering PRA to help with risk management decisions. A tiered approach begins with a relatively simple analysis and progresses stepwise to more complex analyses. The level of complexity should match the site-specific risk assessment objectives and the risk management goals. The tiered approach, with helpful suggestions on risk communication, is presented in Chapter 2. A brief introduction is given below.

The premise for recommending a tiered approach is that there is a balance between the benefits of conducting a more complex analysis, and the cost in terms of additional time, resources, and challenges for risk communication. PRA may require additional resources compared with the point estimate approach, and may not be used routinely for screening level assessment. At more complex hazardous waste sites, PRA may not be warranted if the investment of time and resources is unlikely to provide information on variability and uncertainty in risk that will affect the risk management decision.

This guidance recommends that a point estimate risk assessment be conducted in the first tier after completing the remedial investigation (RI) planning, site scoping, problem formulation, data collection, and the development of a site conceptual model. In general, when site decision making would benefit from additional analysis beyond the point estimate risk assessment, and when the risk manager needs more information to complete the RI/FS process, the risk manager would proceed to higher tiers. Sensitivity analysis should be conducted in each tier to guide decisions regarding data collection and the complexity of the analysis needed to characterize variability and/or uncertainty in risk. Sensitivity analysis can also play an important role in risk communication by supporting decisions to continue characterizing less influential variables with point estimates in higher tiers.

(2) *Select the RME Risk from the RME Risk Range (90th to 99.9th percentile)*

The RME is defined as the highest exposure that is reasonably expected to occur at a site. *Final Guidelines for Exposure Assessment* (EPA, 1992a) states that the "high-end" of exposure for a population occurs between the 90th and 99.9th percentiles, with the 99.9th percentile considered a bounding estimate. Using a point estimate approach, the calculation of the RME risk would be based on high-end input values in combination with average input values. For example, for estimation of risks from the ingestion of groundwater, default exposure is based on a high-end water intake rate (2 L/day), a high-end exposure frequency and duration (350 days/year for 30 years), and an average body weight (70 kg).

With the probabilistic approach, the calculation of the RME risk would be based on a range of input values, or frequency distributions, including low, average, and high-end values for each of the input exposure factors. For example, for estimation of risks from ingestion of groundwater, exposure would be based on the combination of lognormal distributions for water intake rate, body weight, and exposure duration (each using a specified mean and standard deviation) and a triangular distribution for exposure frequency (using a specified minimum, most likely value, and maximum). As a result, the RME risk would become a probability distribution ranging from low- to high-end values based on varying a combination of input values. In PRA, a recommended starting point for risk management decisions regarding the RME is the 95th percentile of the risk distribution (see Chapter 7).

(3) *Use PRA for Dose-Response in Ecological Assessment, not in Human Health Assessment*

Approaches to characterizing variability and uncertainty in toxicological information should reflect both the latest developments in the science of hazard and dose-response evaluation and consistent application of EPA science policy. This statement is consistent with the *1997 EPA Policy Statement* presented in Section 1.4 above (U.S. EPA, 1997g). Probabilistic approaches to ecological dose-response assessment may be explored, as discussed and demonstrated in Chapter 4. This guidance does not develop or evaluate probabilistic approaches for dose-response in human health assessment and, further, *discourages undertaking such activities on a site-by-site basis*. Such activities require contaminant-specific national consensus development and national policy development. Parties wishing to undertake such activities should contact the OERR to explore ways in which they might contribute to a national process for the contaminant of interest to them.

(4) *Prepare a Workplan for EPA Review and Approval*

A workplan should be developed and submitted for review before commencement of a PRA. The workplan should document the combined decisions of the RPM and risk assessor involved in the risk assessment, and positions of the stakeholders. The workplan should address conditions and policies presented in this section of *RAGS Volume 3: Part A*, the software to be used, the exposure routes and models, and the input probability distributions and their basis, including appropriate literature references. The workplan is discussed in more detail in Chapter 2.

A checklist of some of the key considerations to assist in the review of a PRA is provided in Appendix F.

1.5 ORGANIZATION OF THE GUIDANCE

Subsequent chapters of *RAGS Volume 3: Part A* focus on the following topics:

Chapter 2 - The Tiered Approach to PRA

Chapter 2 includes information regarding organizational issues that may need to be considered by the RPM in developing a PRA. Examples, include: workplans, involvement of the Community Involvement Coordinator (CIC), additional meetings with communities, and review of PRA documents.

Chapter 2 also presents the tiered approach in full detail. The approach begins with RI planning, scoping, problem formulation, and data collection. Tier 1 entails a point estimate risk assessment and sensitivity analysis. Tier 2 proceeds with additional data collection, a MCA to characterize variability

and/or uncertainty, and a more in-depth sensitivity analysis. More advanced techniques are used in Tier 3 to simultaneously characterize variability and uncertainty. The endpoint of the tiered approach is to provide information that helps risk managers complete the RI/FS process.

Chapter 3 - Probabilistic Human Health Risk Assessment

Chapter 3 provides a discussion of how PRA approaches may be utilized in human health risk assessment. Probabilistic approaches focus on the exposure assessment, and an example is included to illustrate the application of the tiered approach to a human health risk assessment.

Chapter 4 - Probabilistic Ecological Risk Assessment

Chapter 4 provides a discussion of how PRA approaches may be utilized in ecological risk assessment. This includes a discussion of basic tactics, such as how to decide if, and when, a PRA is needed, along with technical discussions and examples of how to model variability and/or uncertainty in exposure, toxicity, and risk (characterized both as hazard quotients and responses) for different types of ecological receptors, both within and between species. The chapter also provides a discussion of how the results of an ecological PRA can be used in risk management decision making, and provides guidelines for planning and performing an ecological PRA.

Chapter 5 - PRA and Preliminary Remediation Goals (PRGs)

This chapter provides a discussion about issues associated with deriving PRGs from both point estimate risk assessment and PRA. Issues and limitations associated with back calculation are highlighted, along with an explanation and recommendation regarding the iterative forward calculations.

Chapter 6 - Communicating Risks and Uncertainties in PRA

Chapter 6 provides a basic overview of the current Superfund guidance on communicating with the public. With this as a basis, the chapter provides specific information regarding continuous involvement of stakeholders in the PRA process, various tools that may be useful in communicating the principles of PRA, organizational issues regarding planning of communication strategies, and examples of procedures that may be helpful at individual sites. This chapter also provides references to various documents on current approaches for communicating risk to the public.

Chapter 7 - Role of PRA in Decision Making

This chapter provides guidance on how to interpret the results of a PRA to determine if an unacceptable risk is present, and criteria to consider when moving from a risk-based PRG to a remedial goal.

Appendix A - Sensitivity Analysis

Important information from PRA includes the results of sensitivity analysis. This appendix outlines the methodology and interpretation of statistical methods used to conduct sensitivity analysis with point estimate and probabilistic models.

Appendix B - Selecting and Fitting Distributions

One of the more challenging aspects of PRA is choosing appropriate probability distributions to represent variability and uncertainty in the input variables. This appendix presents a process for selecting and fitting distributions to data, including hypothesizing families of distributions, parameter estimation techniques, and goodness-of-fit tests.

Appendix C - Exposure Point Concentration (EPC)

An important variable in most risk assessments is the concentration term. This appendix presents the basic principles of the EPC, and different methods for quantifying both variability and parameter uncertainty in the EPC.

Appendix D - Advanced PRA Models

Sometimes a more complex modeling approach can be used to improve the representativeness of the probabilistic risk estimates. These approaches are generally anticipated to be applied in Tier 3 of the tiered approach. Examples include the use of Microexposure Event modeling, geostatistics, and Bayesian Monte Carlo analysis.

Appendix E - Definitions

A list of definitions is provided at the beginning of each chapter. This appendix provides a compilation of all definitions presented in the guidance.

Appendix F - Generic Checklist

After a PRA has been submitted to the Agency, an efficient process is needed to evaluate the accuracy and clarity of the results. This appendix suggests a series of elements of the review process that can be adopted to structure the review of PRAs for both human health and ecological risk assessment.

Appendix G - Frequently Asked Questions (FAQ) about PRA

Risk assessors and risk managers who read *RAGS Volume 3: Part A* will find that probabilistic risk assessment covers a wide variety of topics ranging from statistical theory to practical applications and policy decisions. U.S. EPA OERR plans to maintain and periodically update a list of frequently asked questions and responses on an EPA Superfund web page at <http://www.epa.gov/superfund/index.htm>. This appendix provides a preliminary list of anticipated questions.

Appendix H - Index

This index includes keywords and concepts used throughout this guidance document. They are listed alphabetically with numbers indicating the appropriate chapter and page number(s) within each chapter. Commas separate page numbers within a chapter or appendix, while semi-colons separate chapters and appendices. For example: probability density function, 1-5, 6-8; 4-3, 10-12; C-1, 8-10. This would indicate Chapter 1, page 5, and pages 6-8; Chapter 4, page 3, and pages 10-12; Appendix C, page 1 and pages 8-10.

1.6 NEXT STEPS FOR PRA IMPLEMENTATION

This guidance has presented the current principles, including the tiered approach, and examples to aid in conducting acceptable PRAs at Superfund sites. Policies and practices will change over time as scientific advances continue in the future. The PRA Workgroup intends to keep current and provide new information on EPA Superfund web page at <http://www.epa.gov/superfund/index.htm>. EPA expects to make the following PRA support items available on-line in the near future:

- *RAGS Volume 3: Part B*: A workbook that serves as a companion to *RAGS Volume 3: Part A*; it will include case studies and examples in PRA.
- *Guidance on Probability Distributions*: Documents and/or spreadsheets to aid in selecting and fitting probability distributions for input variables.
- *Guidance on Data Representativeness*: A ranking methodology to evaluate data representativeness for various exposure scenarios.
- *Hands-On Training*: Basic MCA training materials, and limited computer hands-on training sessions available to Regional EPA and State staff.
- *Access to PRA Workgroup*: A workgroup to provide support on PRA to EPA regional risk assessors.
- *FAQs*: A list of Frequently Asked Questions (FAQs) about PRA and responses from the PRA Workgroup, maintained and periodically updated on-line.

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CHAPTER 2

WORKPLAN AND THE TIERED APPROACH

2.0 INTRODUCTION

While probabilistic risk assessment (PRA) can provide useful information for risk management, not all site decisions will benefit from probabilistic approaches. Similarly, not all PRAs need involve complex models and quantitative uncertainty analysis methods; often, very useful information can be obtained by taking the point estimate approach one step further to explore variability in selected input variables. The level of effort and complexity of the risk assessment should match site-specific needs. The use of a tiered approach for moving from a point estimate risk assessment to PRAs of varying levels of complexity is recommended (Figure 2-1 and 2-2). This chapter outlines the basic steps of a tiered approach for including PRA in a site risk assessment. The major feature of the tiered approach is an iterative evaluation of the risk estimates developed at each tier to determine if they are sufficient for risk management decisions. Built into the tiered approach are opportunities for communication with stakeholders with a view to saving time and costs, and facilitating a successful remedial process.

2.1 WORKPLAN

In practice, the potential value of PRA may be considered at various planning stages of a risk assessment. For some sites, PRA and point estimate risk assessment approaches may be discussed in the initial scoping of the risk assessment. For other sites, PRA may become a viable option only after the point estimate risk assessment results are available. Ideally, PRA should be considered as early as possible in the planning of risk assessment activities at a site so that sampling plans and data collection efforts may be appropriately directed. Initial PRA discussions should be included as part of the risk assessment workplan. If a PRA is being considered following completion of a point estimate risk assessment, the original workplan for the point estimate assessment should be expanded to include needs that are unique to PRA.

The methods and procedures used to prepare a workplan to gather additional information for a baseline point estimate risk assessment are documented in RAGS Volume I: Part A (U.S. EPA, 1989). This chapter of RAGS Volume 3: Part A describes the procedures that would be used to prepare a workplan to gather additional information to conduct a PRA. Separate workplans may be warranted for human health and ecological risk assessments.

Like the quality assurance project plan (QAPP), the workplan for a PRA should document the combined decisions of the remedial project manager (RPM) and the risk assessor. Meaningful involvement of stakeholders early in the decision-making process also will save time and effort.

EXHIBIT 2-1

DEFINITIONS FOR CHAPTER 2

Central Tendency Exposure (CTE) - A risk descriptor representing the average or typical individual in a population, usually considered to be the mean or median of the distribution.

Countably Infinite - Used to describe some discrete random variables, this term refers to a set of numbers that can be counted with integers (e.g., one, two, three) and that has no upper limit. Examples include the number of tosses required for a coin to show a head—we can count each toss, but it is possible that at least one more toss is needed. The number of dust particles in a volume of air is another example. Countably finite implies there is an upper limit (e.g., days of work per year).

CTE Risk - The estimated risk corresponding to the central tendency exposure.

Monte Carlo Analysis (MCA) or Monte Carlo Simulation - A technique for characterizing the uncertainty and variability in risk estimates by repeatedly sampling the probability distributions of the risk equation inputs and using these inputs to calculate a range of risk values.

Parameter - A value that characterizes the distribution of a random variable. Parameters commonly characterize the location, scale, shape, or bounds of the distribution. For example, a truncated normal probability distribution may be defined by four parameters: arithmetic mean [location], standard deviation [scale], and min and max [bounds]. It is important to distinguish between a variable (e.g., ingestion rate) and a parameter (e.g., arithmetic mean ingestion rate).

Point Estimate - In statistical theory, a quantity calculated from values in a sample to estimate a fixed but unknown population parameter. Point estimates typically represent a central tendency or upper bound estimate of variability.

Point Estimate Risk Assessment - A risk assessment in which a point estimate of risk is calculated from a set of point estimates for exposure and toxicity. Such point estimates of risk can reflect the CTE, RME, or bounding risk estimate depending on the choice of inputs.

Potentially Responsible Party (PRP) - PRPs are individuals, companies, or any other party that are potentially liable for payment of Superfund cleanup costs.

Preliminary Remediation Goal (PRG) - Initially developed chemical concentration for an environmental medium that is expected to be protective of human health and ecosystems. PRGs may be developed based on applicable or relevant and appropriate requirements (ARARs), or exposure scenarios evaluated prior to or as a result of the baseline risk assessment. (U.S. EPA, 1991a, 1991b).

Probabilistic Risk Assessment (PRA) - A risk assessment that yields a probability distribution for risk, generally by assigning a probability distribution to represent variability or uncertainty in one or more inputs to the risk equation

Probability Density Function (PDF) - A graph that shows the probability of occurrence of an unknown or variable quantity. A PDF is used to characterize a continuous random variable, X. PDFs can be used to display the shape of the distribution for an input variable or output variable of a Monte Carlo simulation. The term *density* comes from the concept that a probability at a point, x, for a continuous distribution is equal to the area under the curve of the PDF associated with a narrow range of values around x.

Probability Distribution - A mathematical representation of the function that relates probabilities with specified intervals of values for a random variable. Also called a *probability model*.

Probability Mass Function (PMF) - A function representing the probability distribution for a discrete random variable. The mass at a point refers to the probability that the variable will have a value at that point.

Random Variable - A variable that may assume any value from a set of values according to chance. Discrete random variables can assume only a finite or countably infinite number of values (e.g., number of rainfall events per year). A random value is continuous if its set of possible values is an entire interval of numbers (e.g., quantity of rain in a year).

Reasonable Maximum Exposure (RME) - The highest exposure that is reasonably expected to occur at a site (U.S. EPA, 1989). The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures.

Remedial Investigation/Feasibility Study (RI/FS) - Studies undertaken by EPA to delineate the nature and extent of contamination, to evaluate potential risk, and to develop alternatives for cleanup.

EXHIBIT 2-1

DEFINITIONS FOR CHAPTER 2—Continued

RME Risk - The estimated risk corresponding to the reasonable maximum exposure.

Scientific/Management Decision Point (SMDP) - A point during the tiered process in PRA when the risk assessor communicates results of the assessment to the risk manager. At this point, the risk manager determines whether the information is sufficient to arrive at a decision or if additional data collection or analysis is needed. SMDPs provide a tool for transitioning to a subsequent tier or for exiting the tiered process.

Sensitivity Analysis - Sensitivity generally refers to the variation in output of a model with respect to changes in the values of the model's input(s). Sensitivity analysis can provide a quantitative ranking of the model inputs based on their relative contributions to model output variability and uncertainty. Common metrics of sensitivity include:

- ▶ Pearson Correlation Coefficient - A statistic r that measures the strength and direction of linear association between the values of two quantitative variables. The square of the coefficient (r^2) is the fraction of the variance of one variable that is explained by the variance of the second variable.
- ▶ Sensitivity Ratio - Ratio of the change in model output per unit change in an input variable; also called *elasticity*.
- ▶ Spearman Rank Order Correlation Coefficient - A "distribution free" or nonparametric statistic r that measures the strength and direction of association between the ranks of the values (not the values themselves) of two quantitative variables. See Pearson (above) for r^2 .

Uncertainty - Lack of knowledge about specific variables, parameters, models, or other factors. Examples include limited data regarding the concentration of a contaminant in an environmental medium and lack of information on local fish consumption practices. Uncertainty may be reduced through further study.

A PRA workplan should be developed early in the risk assessment planning process for the site, regardless of who will actually develop the PRA (e.g., Environmental Protection Agency (EPA), EPA contractor, or potentially responsible party (PRP)). If a PRP performs the PRA, the workplan should be submitted to EPA for review and approval prior to commencing the PRA. It should describe the intended PRA in sufficient detail so that EPA can determine if the work products will adequately address risk assessment and management needs (see Exhibit 2-2 for contents of a typical workplan). It is important that the risk assessor and RPM discuss the scope of the probabilistic analysis and the potential impact it may have on the remedial investigation/feasibility study (RI/FS).

Given the time and effort that can be expected to be invested in conducting a PRA, it is important that a workplan undergo review and approval by EPA, prior to proceeding with the assessment.

In general, regions should not accept probabilistic analysis when a workplan for the analysis has not been submitted to the Agency, and approved by the regional risk assessor and RPM.

The tiered process for PRA, described in Section 2.3, is an iterative process. As new information becomes available, it should be used to evaluate the need to move to a higher tier. The decision to move an assessment to a higher tier of complexity should result in a revised workplan reflecting the greater complexity and demands of the higher tier. The proposed probabilistic sensitivity analysis developed at the lower tier should be included in the revised workplan, along with a point estimate risk assessment based on any data collected as part of a lower tier. The probabilistic methods used in a PRA can often be restricted to the chemicals and pathways of concern that contribute the greatest risk. The less sensitive chemicals and exposure pathways should still remain in the PRA using point estimates, unless there is a compelling reason to exclude them from the assessment altogether. As stated in Appendix A (Section A.1, *Risk Communication*), the decision to represent an input variable with a point estimate, rather than a probability distribution, will generally be made on a case-by-case basis. The decision will reflect an

attempt to balance the benefits of simplifying the analysis (e.g., easier to communicate; focuses discussion on more critical areas) with the potential for arbitrarily reducing the variance in the output distribution (e.g., discounting variability in multiple variables with negligible contributions to risk may end up having a non-negligible effect on the RME percentile).

Throughout the process of developing the PRA, EPA risk assessor and other contributors to the assessment should have a continuing dialogue to discuss the elements of the workplan and their potential impacts on the assessment. This dialogue, along with interim deliverables, will help to ensure that the risk assessment report will meet the needs of the Agency and that any problems are identified and corrected early in the process.

2.2 SPECIAL ADMINISTRATIVE CONSIDERATIONS IN PRA

Inclusion of a PRA in the RI/FS will generate certain administrative activities for the RPM. The scope of these activities will depend on whether the PRA is conducted by EPA and its contractors or by the PRP. The following sections provide practical advice for the RPM who is considering applications of PRA at a site.

2.2.1 SCOPING OF PRA

The RPM will generally be involved in the discussions among EPA project team, as well as PRPs and other stakeholders, regarding the level of PRA that is appropriate for the site. As outlined in the tiered approach (see Section 2.3), the scope and complexity of the PRA should satisfy the risk assessment and management decision making needs of the site. Team members should meet to discuss the scope of the PRA, the anticipated community outreach, and the required level of review. These discussions can be useful for ascertaining the level of contractor involvement, specific requirements for deliverables from PRPs, and the anticipated number of responses to comments. These meetings should include consideration of funding, resources, and availability of personnel to work on the PRA.

2.2.1.1 PRA SCOPE OF WORK FOR FUND-LEAD SITES

A Statement of Work (SOW) should be developed before any work is started on a PRA, regardless of whether the PRA is to be submitted to the Agency or developed by the Agency. The SOW should outline the general approach that EPA and its contractor will use in developing the PRA. The SOW should include the general approaches for the following PRA items: selection of input probability distributions, documentation of methods and results, selection of computer programs, submission of

EXHIBIT 2-2

EXAMPLES OF IMPORTANT CONTENTS OF A PRA WORKPLAN

1. Statement of the ecological assessment endpoints and/or human risk
2. Summary of the point estimate risk assessment
3. Potential value added by conducting a PRA and proceeding to the subsequent tiers
4. Discussion of adequacy of environmental sampling for PRA or moving to a successive tier (e.g., data quality issues)
5. Description of the methods and models to be used (e.g., model and parameter selection criteria)
6. Proposal for obtaining and basis for using exposure factor distributions or ecological toxicity distributions
7. Methods for deriving the concentration term
8. Proposal for probabilistic sensitivity analysis
9. Software (e.g., date and version of product, random number generator)
10. Bibliography of relevant literature
11. Proposed schedule, discussion points, and expertise needed

computer codes and outputs, comparison of the results from the point estimate and probabilistic assessments, and the format for presenting the final PRA in the RI/FS document. The SOW should be sufficiently detailed to support a milestone schedule, which should be submitted as part of the SOW. Based on the complexity of the PRA, and consistent with the RAGS Volume I: Part D principles of involving the risk assessor early and often in the risk assessment process (U.S. EPA, 2001), it may be appropriate to obtain submission of interim deliverables to allow the risk assessor the opportunity to identify potential problems early in the process.

Within the RI/FS workplan, additional resources may be required to hold additional meetings, to respond to comments specific to the PRA, and to develop handouts describing PRA in terms accessible to a wider audience than risk assessors. Where appropriate, these additional resource requirements should be included in the SOW along with interim and final deliverable dates. Chapter 6 provides guidance on communicating concepts and results of PRA to various audiences.

2.2.1.2 PRP SCOPE OF WORK FOR PRP-LEAD SITES

The SOW for PRP-lead sites should follow the same general outline as the SOW for fund-lead sites (Section 2.2.1.1). Legal documents such as Unilateral Orders, Administrative Orders of Consent, and Consent Decrees should contain language requiring the PRP to submit a workplan before any work on the PRA is started. It is also important that interim deliverables, including computer code or spreadsheet models, be submitted so that EPA can review and verify the results of the PRA. A comparison of the results of the PRA and the point estimate assessment should be included in the final RI/FS.

Depending on the complexity of the site and the anticipated PRA, the RPM may be involved in more extensive negotiations with the PRPs. These negotiations may involve both EPA staff and contractor support. These activities may need to be included in the appropriate SOWs.

If warranted by the complexity of the PRA, the RPM may consider the need to expand oversight contracts to include additional resources for the contractor to review and comment on the interim deliverables and finalize the PRA. This may require a specialized level of expertise that will need to be discussed with the contractor. Further, the contract section regarding community involvement may also need to be expanded to include additional resources for developing handouts describing PRA in terms accessible to a wider audience than risk assessors and for holding additional community meetings.

2.2.2 DEVELOPMENT OF PROBABILITY DISTRIBUTIONS

A key component of any PRA is the selection of representative probability distributions. The information available to support the characterization of variability or uncertainty with probability distributions may be an important factor in the decision to conduct a PRA. In some cases, this may require resources to conduct exploratory data analysis or to collect site-specific information. As part of this process, a PRA using preliminary distributions based on the available information may be considered to identify the variables and exposure pathways that may have the strongest effect on the risk estimates. Appendix B (Section B.2.0) provides a more detailed description of preliminary distributions and their potential role in the tiered process. All of these activities may require extensive discussions with the PRPs and the community. In addition, for PRP-lead sites, they may require additional resources to critically review the proposed distributions. The RPM should consider these potential activities in developing the SOW and legal documents to assure adequate resources are available to address them.

2.2.3 EPA REVIEW OF PRA DOCUMENTS

The review of PRA documents may require more time than is usually allocated for point estimate risk assessments. In part, the additional time is needed for reviewing and discussing input distributions, for developing and running computer simulations, and for discussing outcomes of the assessment with the PRP or EPA contractor. The early involvement of an EPA risk assessor may reduce the time needed for review of the final risk assessment documents, although additional review time may still be required, depending on the complexity of the PRA conducted.

In addition to EPA's review, it may also be important to include external reviewers with specialized expertise in PRA to aid in the review. This additional support may involve resources and time to review documents and verify simulation results, as well as additional contractual arrangements. As stated in Chapter 1, Section 1.4 (Conducting an Acceptable PRA), it is important that negotiations with the PRP address the assurance that adequate details will be included in the submission so that the methods can be evaluated, and the results independently reproduced.

2.2.4 PEER-REVIEW

Depending on the level of complexity of the PRA, and whether new science is being used, it may be necessary to conduct a peer review of the document. The Agency's guidance on peer review (U.S. EPA, 2000b) should be consulted for information regarding the criteria for determining whether or not a peer review is appropriate and, if it is, the process that should be followed.

2.2.5 RESPONSE TO COMMENTS ON PRA

The time and resources needed to respond to comments on a PRA may vary depending on the complexity of the PRA. In developing the SOW, workplan, and schedule for the RI/FS, it is important that the RPM include adequate resources and time for the thorough evaluation of the PRA. In developing the response to comments, it may be necessary to consider alternative PRAs submitted by reviewers. The RPM should plan for sufficient time and resources needed for such activities.

2.2.6 ADMINISTRATIVE RECORD

Criteria should be established for documentation to be included in the administrative record. Examples may include documentation regarding the basis for selection of input distributions, a description of the design of the PRA conducted, the computer codes used in simulations, how tiering decisions are made, and the results of the PRA. The RPM should consider using technologies such as a CD-ROM to document the appropriate information for the record.

2.2.7 COMMUNICATION WITH STAKEHOLDERS

Chapter 6 provides details regarding the goal of early involvement of the public in the PRA process. For example, Section 6.1 of Chapter 6 provides additional topics for consideration in development of community involvement plans (CIPs) where PRA is considered. In general, early involvement of the community in the RI/FS process is important, but such involvement should meet the site-specific needs. Important considerations include resources, funding, and the level of effort appropriate for the site.

2.2.8 COMMUNICATION WITH EPA MANAGEMENT

Communication with EPA managers regarding PRA is discussed in Chapter 6. The RPM may need to consider allocating additional resources for prebriefings of appropriate management levels, development of handouts, and follow-up to the management meetings. Coordination with appropriate EPA staff and contractors may be necessary to assure the communication is effective.

2.3 OVERVIEW OF THE TIERED APPROACH

The tiered approach presented in this guidance is a process for a systematic, informed progression to increasingly more complex risk assessment methods including PRA. A schematic presentation of the tiered approach is shown in Figure 2-1 and Figure 2-2. Higher tiers reflect increasing complexity and, in many cases, will require more time and resources. Higher tiers also reflect increasing characterization of variability and/or uncertainty in the risk estimate, which may be important for making risk management decisions. Central to the concept of a systematic, informed progression is an iterative process of evaluation, deliberation, data collection, work planning, and communication (see Figure 2-2). All of these steps should focus on deciding (1) whether or not the risk assessment, in its current state, is sufficient to support risk management decisions (a clear path to exiting the tiered process is available at each tier); and (2) if the assessment is determined to be insufficient, whether or not progression to a higher tier of complexity (or refinement of the current tier) would provide a sufficient benefit to warrant the additional effort.

The deliberation cycle provides an opportunity to evaluate the direction and goals of the assessment as new information becomes available. It may include evaluations of both scientific and policy information. The risk manager, in the decision-making process, is encouraged to seek input on a regular basis from EPA staff and other stakeholders. Exhibit 2-3 lists some of the potential stakeholders that may contribute to the deliberation process.

Although PRA may involve technical dialogue between EPA and outside “experts”, input from members of the general public who may have an interest in the outcome of the remedial process should also be sought at appropriate stages of the process. Frequent and productive communication between EPA and stakeholders throughout the risk assessment process is important for enhancing the success of a PRA.

EXHIBIT 2-3

**STAKEHOLDERS POTENTIALLY INVOLVED IN
EPA’S DECISION-MAKING PROCESS FOR PRA**

- EPA risk assessors and managers
- Members of the public
- Representatives from state or county environmental or health agencies
- Other federal agencies (e.g., health agencies, Natural Resource Damage Assessment trustees, etc.)
- Tribal government representatives
- Potentially responsible parties and their representatives
- Representatives from federal facilities (e.g., Department of Defense, Department of Energy, etc.)

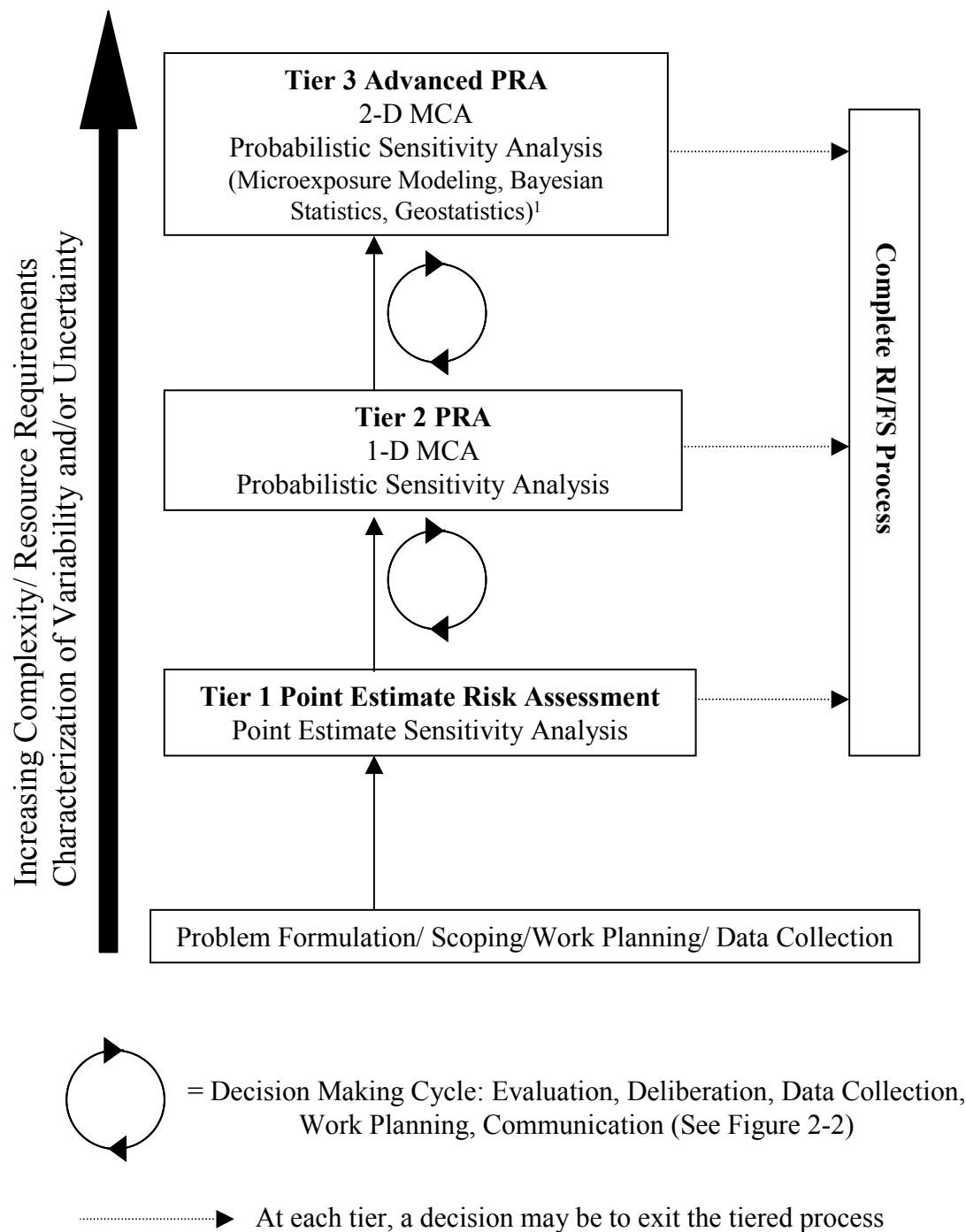


Figure 2-1. Schematic Diagram of Tiered Approach.

¹ Examples of advanced methods for quantifying temporal variability, spatial variability, and uncertainty (see Appendix D)

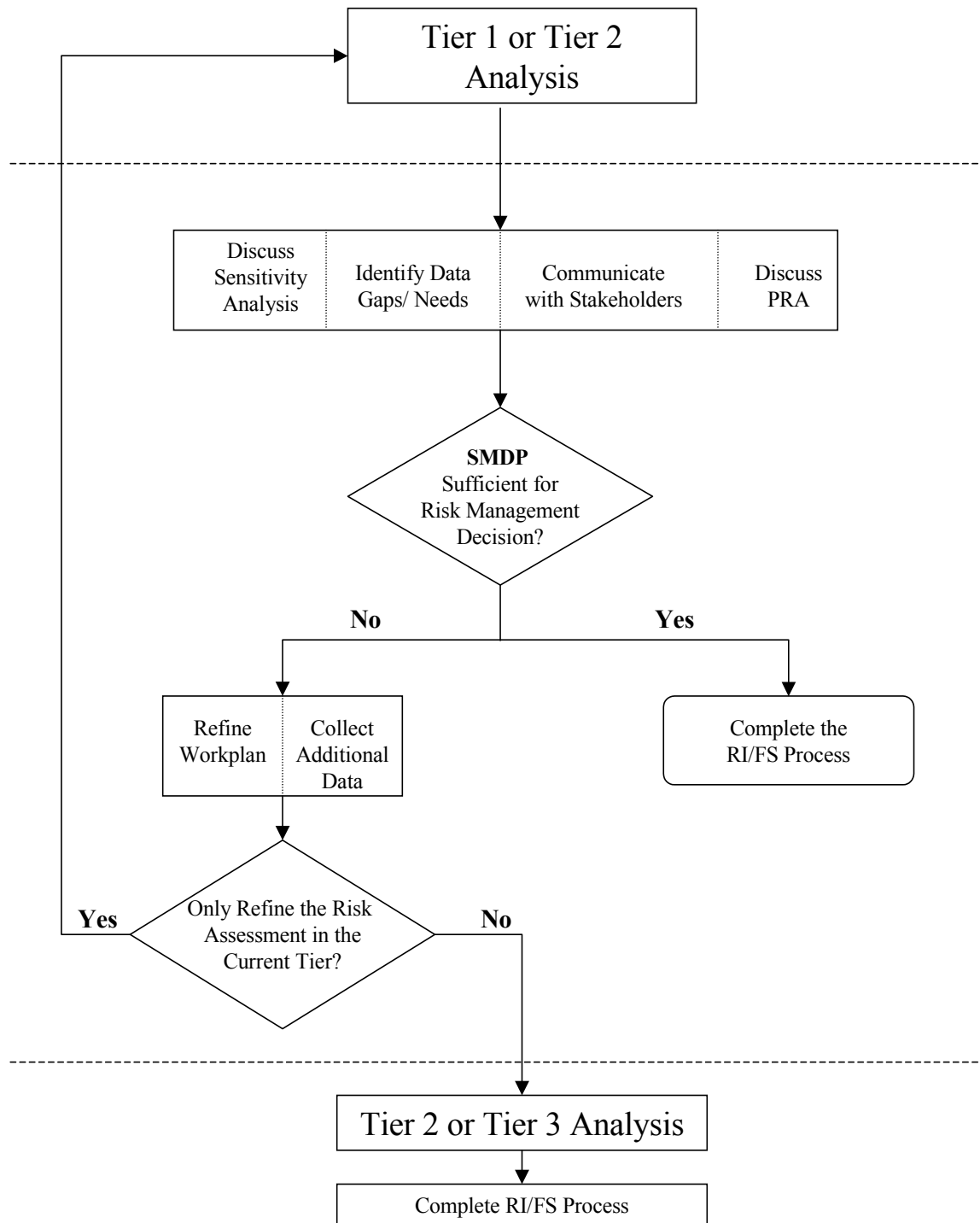


Figure 2-2. Schematic diagram of deliberation/decision cycle in the tiered process for PRA. SMDP refers to a scientific/management decision point, which implies that the decision involves consideration of not only the risk assessment, but also Agency policy, stakeholder concerns, cost, schedule, feasibility and other factors.

2.3.1 GETTING STARTED

All risk assessments should begin with problem formulation, scoping, preparation of a workplan (Section 2.1), and data collection. Problem formulation generally is an iterative process where reevaluation may occur as new information and data become available. The RPM should convene a scoping meeting prior to any risk assessment activities. Depending on the site-specific factors, discussion of performing a PRA may be appropriate at this initial scoping meeting. Alternatively, this discussion may be more productive at a later stage of the tiered process.

The risk manager should initiate discussions with EPA staff and other stakeholders early in the process, well before planning a risk assessment. Early communication with risk assessors or other EPA staff can help the risk manager evaluate the adequacy of the current information and plan additional data-gathering activities. Early communication with communities and other stakeholders should establish trust and facilitate a successful remedial process (see Chapter 6 on risk communication).

Generally, once the appropriate steps have been taken to adequately formulate and identify the problem and complete a workplan (Section 2.1), data collection efforts towards the point estimate risk assessment may begin. The process for conducting a point estimate risk assessment (Tier 1) is documented elsewhere in various RAGS volumes and related Superfund risk assessment guidance documents (e.g., U.S. EPA, 1989, 2001).

2.3.2 TIER 1

Tier 1 consists of the well-established process for planning and conducting human health and ecological point estimate risk assessments. Typical elements of a Tier 1 risk assessment, as they relate to higher tiers, are presented in Exhibit 2-4. A more detailed discussion of these elements can be found in Chapters 3 and 4 and Appendix A (Sensitivity Analysis).

A more detailed schematic presentation of the tiered process, showing the various elements of the deliberation/decision cycle and their linkage to higher tiers is shown in Figure 2-2. The two main factors to consider when determining whether the results of a risk assessment are sufficient for decision making are: (1) the results of a comparison of the risk estimate with the risk level of concern; and (2) the level of confidence in the risk estimate.

In Tier 1, comparison of risk estimates with risk levels of concern is relatively straightforward, since the outcome of a point estimate risk assessment is a single estimate of risk that either will exceed or not exceed the risk level of concern. Evaluating confidence in the Tier 1 risk estimates is more difficult because quantitative measures of uncertainty often are not easily obtained from a point estimate analysis. Uncertainty arises from two main

EXHIBIT 2-4

TYPICAL ELEMENTS OF TIER 1 RISK ASSESSMENT

Analysis Tool - point estimate risk assessment

Variability Modeling - semi-quantitative, using central tendency exposure (CTE) and reasonable maximum exposure (RME) estimates as input variables

Uncertainty Modeling - semi-quantitative using confidence limits on certain point estimates (e.g., concentration term)

Sensitivity Analysis - point estimate calculation of percentage contribution of exposure pathways, for both CTE and RME risk. Systematically vary one input variable at a time across a plausible range and rank inputs based on sensitivity ratios or sensitivity scores.

Risk-Based Decision-Making Output - point estimate of risk—*Does the point estimate exceed the risk level of concern?*

sources: (1) uncertainty in the inputs to the risk equations that stems from lack of knowledge (data gaps), and (2) uncertainty in the accuracy of the point estimate that stems from the mathematical simplifications that are inherent in point estimate computations.

There are usually many sources of uncertainty in the values used to calculate risk. One of the most familiar (but not always the most significant) is uncertainty in environmental concentration values of contaminants. This source of uncertainty is usually accounted for by calculating a 95% upper confidence limit (95% UCL) for the mean concentration in the exposure equation (U.S. EPA, 1992b). Chapter 5, Appendix C, and Appendix D provide more complete discussions of policies and methods for quantifying uncertainty in the exposure point concentration. Uncertainties in other variables in the risk equations (intake rates, exposure frequency and duration, toxicity factors, etc.) may also be significant, and are often addressed by choosing inputs that are more likely to yield an overestimate than an underestimate of risk. These sources of uncertainty are usually addressed qualitatively, by providing a discussion of the likely direction and magnitude of the error that may be associated with the use of the specific inputs (U.S. EPA, 1989). Stakeholders can provide useful information about uncertain variables and sources for site-specific data. This is an important reason to ensure that stakeholders are given the opportunity to review the risk assessment and be involved in the process.

Decision Alternatives

The evaluation of the point estimate risk assessment will yield one of two outcomes: (1) sufficient for risk management decisions; or (2) insufficient for risk management decisions. If the risk manager views the results of the point estimate risk assessment as sufficient for risk management decision making, the risk manager can exit the tiered approach and complete the RI/FS process (Figure 2-2). Depending on site-specific information, the results may support a decision for “no further action” or for a “remedial action.” A “no further action” decision may result when the risk estimate is clearly below the level of concern (e.g., the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) risk range of $1\text{E-}04$ to $1\text{E-}06$) and confidence in the risk estimate is high. A decision for remedial action may result when a national standard (e.g., maximum contaminant levels (MCLs) applied to groundwater) may be exceeded, or when the risk is clearly above the level of concern (e.g., the NCP risk range of $1\text{E-}04$ to $1\text{E-}06$) and confidence in the risk estimate is high. The decision for a specific remedial action involves consideration of the NCP’s nine criteria for remedial decisions (U.S. EPA, 1990) and other site-specific factors.

An alternative conclusion would be that the results of the point estimate risk assessment are not sufficient for risk management decision making. For example, results may not be sufficient when the risk estimate is within the NCP risk range of $1\text{E-}04$ to $1\text{E-}06$ and confidence in the risk estimate is low. In this case, the risk manager should not exit the tiered approach. Instead, appropriate steps should be taken to increase the confidence that a management decision is protective. These steps may include discussing the point estimate sensitivity analysis, identifying data gaps, communicating with stakeholders (e.g., to obtain site-specific information), discussing the potential value of conducting a PRA (or a more advanced probabilistic analysis), work planning, and additional data collection (see Figure 2-2).

A sensitivity analysis can be a valuable component of the evaluation of a risk assessment. Sensitivity analysis can identify important variables and pathways that may be targets for further analysis and data collection. The type of information provided by a sensitivity analysis will vary with each tier of a PRA. Several methods are available at each tier, and the results of the analysis can vary greatly depending on the methods used. A comprehensive discussion of these methods is presented in Appendix A and briefly summarized here. Sensitivity analysis in Tier 1 will usually involve relatively

simple methods and will not involve Monte Carlo simulation. A typical approach would be to calculate the relative contributions of individual exposure pathways to the point estimate of risk. A more complex approach involves selecting values from a plausible range for a specific input variable to the exposure or risk equation and to use these values (i.e., low-end estimate and high-end estimate) to calculate corresponding point estimates of risk. The sensitivity of the risk estimate to each variable is then evaluated by calculating a sensitivity ratio, which is simply the percentage change in the risk estimate divided by the percentage change in the input variable value (see Appendix A, Section A.2.1.3, Sensitivity Ratios).

The sensitivity ratio (SR) approach is typically applied to one variable at a time because jointly varying point estimates for multiple variables can be cumbersome (see Chapter 3, Table 3-2 for an example of two jointly varied inputs). Information provided by the SR approach is generally limited to bounding estimates of risk based on small deviations and/or plausible ranges of point estimates for inputs. Because the point estimate approach does not generate a distribution of risk, SRs cannot provide quantitative information about the relative contributions of input variables to the variance in risk or the uncertainty in selected percentile of the risk distribution. This limitation of the SR approach may be particularly important if the ranking of input variables may change depending on the percentile range that is evaluated. For example, in a probabilistic analysis, the soil ingestion rate variable may contribute most to the variability in risk across the entire risk distribution, but the exposure duration may be the driver in the high-end (> 90th percentile) of the risk distribution, where the RME risk is defined. In addition, for standard product-quotient risk equations, the SR approach also has difficulty distinguishing the relative importance of exposure variables in the risk equation. Appendix A presents a hypothetical example to illustrate why this happens for the common risk equations. An improvement over the SR approach, called Sensitivity Score, involves weighting each ratio by the variance or coefficient of variation of the input variable when this information is available. In general, the most informative sensitivity analysis will involve Monte Carlo techniques (see Appendix A, Table A-1). Potential strengths and weaknesses of sensitivity analysis methods may be an important factor in deciding whether or not to conduct a probabilistic analysis in Tier 2.

Once data gaps have been identified, steps may be taken to gather additional data and revise the point estimates of risk based on these data. As with any data collection effort, the data quality objectives (DQO) process should be followed to obtain samples appropriate for the risk assessment and sufficient to support the remedial decision (U.S. EPA, 1992a, 1993, 1994, 2000a). The deliberation and decision cycle (Figure 2-2) should then be reiterated to determine if the refined risk assessment is sufficient to support risk management decisions. The collection of additional data may also provide a compelling reason to consider moving to Tier 2 and conducting a PRA. If, during the PRA discussions, it is determined that information from a PRA may influence the risk management decisions, PRA may be warranted. This iterative process of collecting data, recalculating point estimates, and reconsidering the potential value of PRA may continue until sufficient data are available to support risk management decisions, or data collection efforts are not possible due to resource constraints. For example, soil ingestion rate data may be limited to a few studies with small sample sizes, but a new soil ingestion study may be prohibitively expensive, time consuming, or difficult to conduct in a manner that will reduce the uncertainty in the risk estimate. Uncertainty due to data quantity is not necessarily a reason to exit the tiered process at Tier 1.

In cases where there is uncertainty in selecting a probability distribution because of small sample sizes, it may be informative to develop a preliminary probability distribution such as a triangular or uniform (see Appendix B, Section B.2.0). These preliminary distributions will contribute to the variability in the risk estimate, and can therefore be included in the probabilistic sensitivity analysis. Results of Monte Carlo simulations that include one or more preliminary distributions may lead to several

alternative decisions. If the sensitivity analysis suggests that the risk estimate is relatively insensitive to the variable described with the distribution, then the uncertainty associated with the choice of a distribution should not affect the risk management decision process using the tiered approach (e.g., choice of RME percentile, derivation of a PRG). In other words, the choice would be to continue with the tiered process. If, however, the variables described by preliminary distribution are important sources of variability or uncertainty in the risk estimate, then this information should be presented in the scientific management decision point (see Figure 2-2). The uncertainty may be sufficiently important in the risk management decision to warrant additional data collection efforts. Conversely, it may be necessary to exit the tiered process if the uncertainty cannot be reduced. Although the tiered process may be stopped at this point, it can still be informative to present the results from the PRA. For example, information about uncertainty may affect the choice of the percentile used to characterize the RME risk. In addition, it may be appropriate to weight the results of the point estimate analysis more heavily in the risk management decision when uncertainty in the PRA is high. Further guidance on appropriate choices for distributions based on the information available to characterize variability is given in Appendix B.

PRA also may be warranted if it would be beneficial to know where on the risk distribution the point estimate lies. An example of this would be a risk estimate that is within the NCP risk range of 1E-04 to 1E-06. The assessment may be sufficient to support risk management decisions if it could be shown that the point estimate of risk lies sufficiently high in the risk distribution. For example, a “no further action” decision may be strengthened if the point estimate is at the 99th percentile of the risk distribution, if risks in lower percentiles of the RME risk range are below the NCP risk range, and if there is high confidence in the risk result. This type of evaluation can be conducted using PRA techniques.

Even if the RME point estimate of risk exceeds the risk level of concern, and PRA is not needed to confirm this result, information from a PRA can be helpful in determining a strategy for achieving a protective preliminary remediation goal (PRG). A detailed discussion of the use of PRA in setting remediation action levels is given in Chapter 5. The advantages and disadvantages of the point estimate approach and PRA are presented in Chapter 1 (Exhibits 1-5 and 1-6).

2.3.3 TIER 2

Tier 2 of the tiered approach to risk assessment will generally consist of a simple probabilistic approach such as one-dimensional Monte Carlo analysis (1-D MCA). A 1-D MCA is a statistical technique that may combine point estimates and probability distributions to yield a probability distribution that characterizes variability or uncertainty in risks within a population (see Chapter 1). Guidance for selecting and fitting distributions is presented in Appendix B. Typical elements of a Tier 2 risk assessment, as they relate to higher and lower tiers are presented in Exhibit 2-5. A more detailed discussion of these elements can be found in Chapters 3 and 4, and Appendix A (Sensitivity Analysis).

While most of the Tier 2 assessments are expected to use 1-D MCA to characterize variability in risk, sometimes a 1-D MCA of uncertainty may be of interest. For example, as suggested in Exhibit 2-5, a probability distribution for uncertainty in the arithmetic mean or median (i.e., 50th percentile) for selected input variables may be specified in a 1-D MCA to yield a probability distribution for uncertainty for the central tendency risk estimate. However, as most Tier 2 assessments are expected to combine input distributions for variability, this guidance focuses on 1-D MCA for characterizing variability in the risk estimate.

Decision Alternatives

Generally, the three main questions to consider when determining whether the results of a 1-D MCA are sufficient for risk management decisions are: (1) What is the RME risk range and how does it compare to the level of concern?; (2) Where does the point estimate risk lie on the risk distribution?; and (3) What is the level of confidence in the risk estimate? In Tier 2, similar to the point

estimate approach, the level of confidence in a single 1-D MCA risk distribution is generally addressed in a qualitative or semi-quantitative way. As discussed in Chapter 1 (Section 1.2.4) and Chapter 3 (Section 3.4.1), one should avoid developing input distributions to a PRA model that yield a single risk distribution that intermingles, or represents both variability and uncertainty. In Tier 2, the preferred approach for characterizing uncertainty in the risk estimate is to perform multiple 1-D MCA simulations (of variability), which uses a different point estimate for uncertainty for one or more parameters, combined with probability distributions for variability for one or more variables. Chapter 3 (see Table 3-2 and Figures 3-3 and 3-4) presents an example of iterative 1-D MCA simulations using combinations of point estimates characterizing uncertainty for two variables. More advanced PRA techniques such as two-dimensional Monte Carlo analysis (2-D MCA), in which distributions for variability and uncertainty are propagated separately through an exposure model, can be undertaken in Tier 3 (Appendix D).

In order to use a PRA to determine if risks are unacceptable and to develop preliminary remediation goals (PRGs) that are protective of the RME individual (see Chapter 5), a single point from the RME risk range should be selected (e.g., 95th percentile). In general, this can be accomplished by selecting an estimate within the RME risk range based on the level of confidence in the output of the 1-D MCA. Uncertainty in risk estimates may be quantified or reduced by considering site-specific factors, biological data, and toxicity data. Stakeholders can provide useful information about uncertain variables and sources for site-specific data. More detailed guidance for choosing a percentile value within the RME range is provided in Chapter 7.

EXHIBIT 2-5

TYPICAL ELEMENTS OF TIER 2 RISK ASSESSMENT

Analysis Tool - 1-D MCA

Variability Modeling - full characterization of variability in risk using PDFs or PMFs for input variables

Uncertainty Modeling - semi-quantitative estimate of uncertainty using iterative 1-D MCA simulations of variability, or a single 1-D MCA of uncertainty in the CTE risk

Sensitivity Analysis - varying multiple variables with probability distributions gives a quantitative ranking (e.g., correlation coefficient) of the relative contributions of exposure pathways and variables to CTE or RME risk

Risk-Based Decision-Making Output - risk distribution for variability: *Does the risk level of concern fall within an acceptable range on the risk distribution (i.e., RME range)?* Also, risk distribution for uncertainty: *What is the 90% confidence interval for the CTE risk?*

The evaluation of the risk assessment in a 1-D MCA in Tier 2 will yield one of two outcomes: (1) sufficient for risk management decisions; or (2) insufficient for risk management decisions. If determined to be sufficient, the risk manager can exit the tiered approach and complete the RI/FS process. The results of a 1-D MCA may support a decision for “no further action” or for a “remedial action.” A “no further action” decision may result when the RME risk range (or a specified point in the RME risk range) is clearly below the level of concern (e.g., Hazard Index=1) and confidence in the risk distribution is high. A decision for remedial action may result when a national standard (e.g., MCLs applied to groundwater) may be exceeded, or when the RME risk range (or a specified point in the RME risk range) is clearly above the level of concern and confidence in the risk distribution is high. The decision for a specific remedial action involves consideration of the NCP’s nine evaluation criteria for remedial decisions (U.S. EPA, 1990; see Chapter 1) and other site-specific factors.

An alternative conclusion at the end of a Tier 2 analysis would be that the results of the 1-D MCA are not sufficient for risk management decisions. There are several factors that might support this conclusion:

- (1) The RME risk range is close to the NCP risk range and confidence in the risk distribution is low. In this case, the risk manager might decide to not exit the tiered approach, and instead continue taking appropriate steps to increase the confidence in the risk estimate.
- (2) Uncertainty is high and it is believed that more than one variable is a major contributor to the uncertainty in the risk estimate. It can be difficult to explore uncertainty in more than one variable using 1-D MCA simulations of variability, even using iterative approaches discussed in Chapter 3 (Section 3.4.1).
- (3) Results of the point estimate risk assessment differ significantly from the results of the 1-D MCA. While the RME risk estimates are not expected to be identical, typically the RME point estimate will correspond with a percentile value within the RME range (i.e., 90th to 99.9th percentile) of the risk distribution. If the RME point estimates fall outside this range, further steps may be warranted to evaluate the choices for input variables—both the RME point estimates, and the probability distributions and parameters (including truncation limits) for the 1-D MCA.

The deliberation/decision cycle (Figure 2-2) between Tier 2 and Tier 3 is similar to the cycle between Tier 1 and 2 and includes discussing the Tier 2 probabilistic sensitivity analysis, identifying data gaps, communicating with stakeholders (e.g., to obtain site-specific information), discussing the potential value of further analysis with probabilistic methods, work planning, and additional data collection. As with the Tier 1 assessment, additional data collection should follow the DQO process (U.S. EPA, 1992a, 1993, 1994, 2000a) and point estimates of risk should be revisited with the new data. The deliberation/decision cycle is an iterative process in which the level and complexity of the analysis increases until the scope of the analysis satisfies decision-making needs. This iterative process should continue until sufficient data are available to support risk management decisions. As in all tiers, stakeholder involvement should be encouraged. Once a 1-D MCA for variability or uncertainty is completed and is available for review and interpretation, a stakeholder meeting should be convened. Interested stakeholders should be given the opportunity to review the 1-D MCA and provide comments. Communication issues specific to PRA are discussed in Chapter 6 (Risk Communication).

In addition to identifying data gaps, consideration for a refined 1-D MCA or more advanced PRA techniques may begin as a means of determining what benefits they may confer to the decision-making

process. If, during further discussions of PRA, it is determined that information from a more advanced PRA may influence the risk management decision, the use of an advanced PRA may be warranted. If additional data have been collected, the point estimate and 1-D MCA should be refined. Specifically, an advanced PRA may be warranted if it would be beneficial to characterize uncertainty in more than one variable at a time. A 2-D MCA can simultaneously characterize variability and uncertainty in multiple variables and parameter estimates. The decision to employ such advanced methods should be balanced with considerations of resource constraints and the feasibility of reducing uncertainty in a given variable. A detailed discussion of advanced PRA methods, including 2-D MCA, is provided in Appendix D.

2.3.4 TIER 3

Tier 3 of the tiered approach to risk assessment consists of advanced PRA methods, such as 2-D MCA, Microexposure Event Analysis (MEE), geostatistical analysis of concentration data, and Bayesian statistics. Typical elements of a Tier 3 risk assessment are presented in Exhibit 2-6. A more detailed discussion of these elements is given in Appendix D. As in other tiers, Tier 3 includes an iterative process of deliberation and decision making in which the level and complexity of the analysis increases until the scope of the analysis satisfies decision-making needs. As in all tiers, stakeholder involvement is encouraged.

Generally, the various elements of the deliberation/decision cycle for Tier 3 are the same as those for Tier 1 and 2 (Figure 2-2). An advanced PRA would be conducted and made available for review to the risk manager and stakeholders. The risk manager must determine if the results of the advanced PRA are sufficient for risk management decision making. Issues to consider when making this determination are similar to those identified for evaluating point estimate risk results and 1-D MCA results, and focus on evaluating the sources and magnitude of uncertainty in relation to the established risk level of concern. If the results are sufficient for risk management decisions, the risk manager may exit the tiered approach and complete the RI/FS process. If the results are not found to be sufficient for risk management decisions, data gaps should be identified and if additional data

are collected, all stages of the risk assessment, including the advanced PRA, the 1-D MCA, and the point estimate risk assessment, should be refined. Alternatively, additional advanced PRA methods may be explored. Refer to Appendix D for a discussion of more advanced PRA techniques. Overall, analysis should continue within Tier 3 until sufficiently informed risk management decisions can be made.

EXHIBIT 2-6

TYPICAL ELEMENTS OF TIER 3 RISK ASSESSMENT

Analysis Tool - 2-D MCA, MEE, geostatistics, and Bayesian statistics

Variability Modeling - full characterization using PDFs or PMFs for input variables

Uncertainty Modeling - quantitative, segregating uncertainty from variability, and associated with multiple variables simultaneously

Sensitivity Analysis - varying parameters of probability distributions to identify and rank order parameter uncertainty with the same sensitivity analysis methods used for Tier 2 (see Appendix A). Also, explore alternative choices of probability distributions and sources of model uncertainty.

Risk-based Decision-Making Criteria - risk distribution for variability with confidence limits—*Does the risk level of concern fall within an acceptable range on the risk distribution (i.e., RME range), and with an acceptable level of uncertainty?*

2.3.5 FLEXIBILITY IN DEFINING TIERS

The assignment of specific analytical tools to Tiers 1, 2, and 3 (Figure 2-1 and Exhibits 2-4 through 2-6) results in generalizations that may not be applicable to all site assessments. Upon completion of the deliberation phase between Tier 1 and Tier 2, the conclusion may be that analytical tools in Tier 3 would be applicable and beneficial for addressing decision making issues. For example, geospatial modeling may be beneficial for improving estimates of uncertainty in the exposure point concentration or in designing field sampling plans to further reduce uncertainty. An improved estimate of the 95% UCL from geospatial analysis (shown in Exhibit 2-6 as a Tier 3 analytical tool) would then be integrated into a Tier 2 assessment, or the complete distribution for uncertainty in the mean concentration could be incorporated into a 2-D MCA in Tier 3. Flexibility in defining the level of complexity of the analysis used in a given tier is essential to accommodating the wide range of risk assessment issues likely to be encountered. An important benefit gained from use of the tiered approach is to ensure a deliberative process in the advancement of the assessment to higher levels of complexity. It is far more important that a deliberative process take place and be documented, than it is to constrain a set of analytical tools to a specific tier.

REFERENCES FOR CHAPTER 2

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CHAPTER 3

USING PROBABILISTIC ANALYSIS IN HUMAN HEALTH ASSESSMENT

3.0 INTRODUCTION

This chapter outlines how probabilistic analysis may be applied to human health risk assessments in the Environmental Protection Agency's (EPA) Superfund program. The paradigm for human health risk assessment as described in EPA's *Risk Assessment Guidance for Superfund* (U.S. EPA, 1989), includes data collection/evaluation in addition to exposure and toxicity assessment and risk characterization. Although the strategies and methods used in collecting and analyzing data can significantly impact the uncertainty in a risk estimate, they are issues relevant to risk assessment in general, and are addressed in other guidance documents, such as EPA's *Guidance for Data Useability in Risk Assessment* (U.S. EPA, 1992b). RAGS Volume 3: Part A focuses on a tiered approach for incorporating quantitative information on variability and uncertainty into risk management decisions.

3.1 CHARACTERIZING VARIABILITY IN EXPOSURE VARIABLES

Exhibit 3-1 gives the general equation used for calculating exposure, often expressed as an average daily intake. In a point estimate approach, single values (typically a mixture of average and high-end values) are input into the equation. In probabilistic risk assessment (PRA), the only difference is that a probability distribution, rather than single value, is specified for one or more variables. A Monte Carlo simulation is executed by repeatedly selecting random values from each of these distributions and calculating the corresponding exposure and risk. For the majority of PRAs, it is expected that probability distributions will be used to characterize inter-individual variability, which refers to true heterogeneity or diversity in a population. Thus, variability in daily intake, for example, can be characterized by combining multiple sources of variability in exposure, such as ingestion rate, exposure frequency, exposure duration, and body weight. Variability in chemical concentrations (Chapter 5 and Appendix C) and the toxicity term in ecological risk assessment (Chapter 4) may also be considered in risk calculations.

EXHIBIT 3-1
GENERAL EQUATION FOR EXPOSURE

$$I = \frac{C \times CR \times EF \times ED}{BW \times AT} \quad \text{Eq. 3-1}$$

where,

I	=	daily intake
C	=	contaminant concentration
CR	=	contact rate (ingestion, inhalation, dermal contact)
EF	=	exposure frequency
ED	=	exposure duration
BW	=	body weight
AT	=	averaging time

EXHIBIT 3-2

DEFINITIONS FOR CHAPTER 3

95% UCL for mean - The one-sided 95% upper confidence limit for a population mean; if a sample of size (n) was repeatedly drawn from the population, the 95% UCL will equal or exceed the true population mean 95% of the time. It is a measure of uncertainty in the mean, not to be confused with the 95th percentile (see below), which is a measure of variability. As sample size increases, the difference between the UCL for the mean and the true mean decreases, while the 95th percentile of the distribution remains relatively unchanged.

95th percentile - The number in a distribution that is greater than 95% of the other values of the distribution, and less than 5% of the values. When estimated from a sample, this quantity may be equal to an observed value, or interpolated from among two values.

Arithmetic Mean (AM) - A number equal to the average value of a population or sample. Usually obtained by summing all the values in the sample and dividing by the number of values (i.e., sample size).

Assessment Endpoint - The specific expression of the population or ecosystem that is to be protected. It can be characterized both qualitatively and quantitatively in the risk assessment.

Central Tendency Exposure (CTE) - A risk descriptor representing the average or typical individual in the population, usually considered to be the arithmetic mean or median of the risk distribution.

Credible Interval - A range of values that represent plausible bounds on a population parameter. Credible intervals may describe a parameter of an input variable (e.g., mean ingestion rate) or output variable (e.g., 95th percentile risk). The term is introduced as an alternative to the term confidence interval when the methods used to quantify uncertainty are not based entirely on statistical principles such as sampling distributions or Bayesian approaches. For example, multiple estimates of an arithmetic mean may be available from different studies reported in the literature—using professional judgment, these estimates may support a decision to describe a range of possible values for the arithmetic mean.

CTE Risk - The estimated risk corresponding to the central tendency exposure.

Cumulative Distribution Function (CDF) - Obtained by integrating the PDF or PMF, gives the cumulative probability of occurrence for a random independent variable. Each value c of the function is the probability that a random observation x will be less than or equal to c .

Exposure Point Concentration (EPC) - The average chemical concentration to which receptors are exposed within an exposure unit. Estimates of the EPC represent the concentration term used in exposure assessment.

Frequency Distribution/Histogram - A graphic (plot) summarizing the frequency of the values observed or measured from a population. It conveys the range of values and the count (or proportion of the sample) that was observed across that range.

High-end Risk - A risk descriptor representing the high-end, or upper tail of the risk distribution, usually considered to be equal to or greater than the 90th percentile.

Low-end Risk - A risk descriptor representing the low-end, or lower tail of the risk distribution, such as the 5th or 25th percentile.

Parameter - A value that characterizes the distribution of a random variable. Parameters commonly characterize the location, scale, shape, or bounds of the distribution. For example, a truncated normal probability distribution may be defined by four parameters: arithmetic mean [location], standard deviation [scale], and min and max [bounds]. It is important to distinguish between a variable (e.g., ingestion rate) and a parameter (e.g., arithmetic mean ingestion rate).

Probability Density Function (PDF) - A function representing the probability distribution of a continuous random variable. The density at a point refers to the probability that the variable will have a value in a narrow range about that point.

Probability Mass Function (PMF) - A function representing the probability distribution for a discrete random variable. The mass at a point refers to the probability that the variable will have a value at that point.

Reasonable Maximum Exposure (RME) - The highest exposure that is reasonably expected to occur at a site (U.S. EPA, 1989). The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures.

EXHIBIT 3-2

DEFINITIONS FOR CHAPTER 3—Continued

Sensitivity Analysis - Sensitivity generally refers to the variation in output of a model with respect to changes in the values of the model's input(s). Sensitivity analysis can provide a quantitative ranking of the model inputs based on their relative contributions to model output variability and uncertainty. Common metrics of sensitivity include:

- ▶ Pearson Correlation Coefficient - A statistic r that measures the strength and direction of linear association between the values of two quantitative variables. The square of the coefficient (r^2) is the fraction of the variance of one variable that is explained by the variance of the second variable.
- ▶ Sensitivity Ratio - Ratio of the change in model output per unit change in an input variable; also called *elasticity*.
- ▶ Spearman Rank Order Correlation Coefficient - A “distribution free” or nonparametric statistic r that measures the strength and direction of association between the ranks of the values (not the values themselves) of two quantitative variables. See Pearson (above) for r^2 .

Target Population - The set of all receptors that are potentially at risk. Sometimes referred to as the “population of concern”. A sample population is selected for statistical sampling in order to make inferences regarding the target population (see Appendix B, Section B.3.1, Concepts of Populations and Sampling).

Figure 3-1 shows a hypothetical example of an input distribution for drinking water ingestion rate. Assume that survey data for drinking water ingestion rates were compiled in order to select and fit a probability distribution. One of the first steps in exploring the data set may be to plot a frequency distribution. In the graph, the height of the bars (the y-axis) represents the relative frequency of ingestion rates in the population and the spread of the bars (the x-axis) is the varying amounts of water ingested (L/day). Since ingestion rate is a continuous random variable, the probability distribution can also be represented graphically with a probability density function (PDF). Assume that the following parameters are estimated from the sample: arithmetic mean=1.36, standard deviation=0.36, geometric mean=1.31, and geometric standard deviation=1.30. These parameter estimates may be used to define a variety of probability distributions, including a 2-parameter lognormal distribution. The fit of the lognormal distribution can be evaluated by visual inspection using the PDF given by Figure 3-1, or by a lognormal probability plot (see Appendix B).

The y-axis for a PDF is referred to as the *probability density*, where the density at a point on the x-axis represents the probability that a variable will have a value within a narrow range about the point. This type of graph shows, for example, that there is a greater area under the curve (greater probability density) in the 1-2 L/day range than 0-1 L/day or 2-3 L/day. That is, most people reported consuming 1-2 L/day of drinking water. By selecting a lognormal distribution to characterize inter-individual variability, we can state more precisely that 1 L/day corresponds to the 15th percentile and 2 L/day corresponds to the 95th percentile, so approximately 80% (i.e., $0.95 - 0.15 = 0.80$) of the population is likely to consume between 1 and 2 L/day of drinking water.

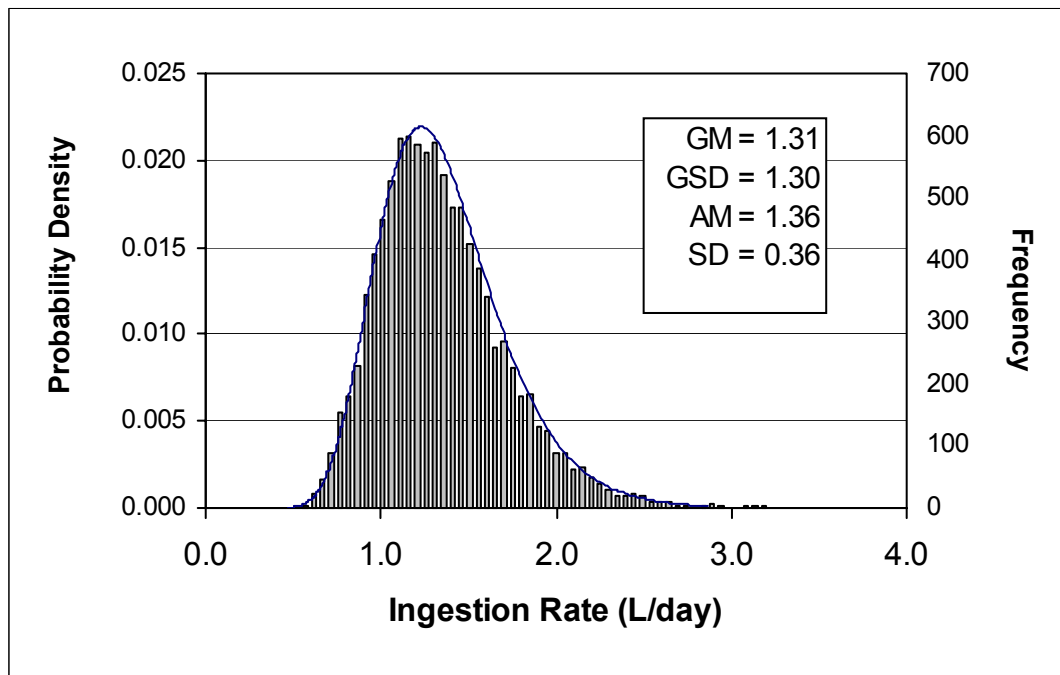


Figure 3-1. Example of a frequency distribution for adult drinking water ingestion rates, overlaid by a graph of the probability density function (PDF) for a lognormal distribution defined by the sample statistics. The distribution represents inter-individual variability in water intakes and is characterized by two parameters. Typically, the geometric mean (GM) and geometric standard deviation (GSD), or the arithmetic mean (AM) and arithmetic standard deviation (SD) are presented to characterize a lognormal distribution.

3.1.1 DEVELOPING DISTRIBUTIONS FOR EXPOSURE VARIABLES

When site-specific data or representative surrogate data are available, a probability distribution can be fit to that data to characterize variability. Appendix B describes how to fit distributions to data, how to assess the quality of the fit and discusses topics such as the sensitivity of the tails of the distribution to various PDFs, and correlations among variables. Many of the issues discussed below regarding the use of site-specific data or surrogate data are relevant to both point estimate risk assessment and PRA.

For the majority of the exposure variables, such as exposure duration, water intake rates, and body weight, site-specific data will not be available. The risk assessor will have to either select a distribution from existing sources, or develop a distribution from published data sets and data summaries. Examples of sources for these distributions and data sets are EPA's *Exposure Factors Handbook* (U.S. EPA, 1997a,b,c), Oregon Department of Environmental Quality's *Guidance for Use of Probabilistic Analysis in Human Health Risk Assessment* (Oregon DEQ, 1998), and the scientific literature. An appropriate PDF should be determined in collaboration with the regional risk assessor. The process by which PDFs are to be selected and evaluated should be described in the workplan. EPA's Superfund program is in the process of developing a ranking methodology to evaluate data representativeness relevant to various exposures scenarios. Following peer review and project completion, the results will be posted on EPA Superfund web page.

☞ At this time, EPA does not recommend generic or default probability distributions for exposure variables.

Regardless of whether a PDF is derived from site-specific measurements or obtained from the open literature, the risk assessor should carefully evaluate the applicability of the distribution to the target population at the site. The distribution selected should be derived from the target population or from a surrogate population that is representative of the target population at the site. For example, a distribution based on homegrown vegetable consumption in an urban population would not be representative for a farming population in the Midwest. If such a distribution were to be used, (and no other data were available), the uncertainty and bias that this PDF would impart to the risk estimate should be communicated to the risk decision makers.

For purposes of risk management decision making, the significance of not having site-specific data should be evaluated in the context of representativeness and sensitivity analysis. If published data are representative of the potentially exposed population, then site-specific data may be unnecessary. For example, body weights of children and adults have been well studied from national surveys and can generally be considered reasonable surrogates for use in site risk assessments. Furthermore, even if a variable is likely to vary among different exposed populations, it may not contribute greatly to the variance or uncertainty in risk estimates. In this case, surrogate data may also be used with confidence in the risk estimate. In addition, the PRA may be simplified by using point estimates instead of probability distributions for the "less sensitive" exposure variables. In part, the decision to use a point estimate in lieu of a probability distribution must balance the benefit of simplifying the analysis and the communication process (see Chapter 6), against the reduction (however small) in the variance of the risk distribution. The utility of sensitivity analysis in identifying the important factors in a risk estimate is discussed further below and in Appendix A.

It is also important to evaluate the sample design and sample size when deciding to apply a distribution to a specific site. Depending on the situation, a very large data set derived from a national

population may be more useful than a site-specific data set derived from a small, incomplete, or poorly designed study. Appendix B provides additional discussion on how to evaluate the data and studies that form the basis for a distribution. Often, the question arises regarding the appropriateness of combining data sets to derive a PDF. Before combining data sets, a careful evaluation should be made of the representativeness of the study populations, and the similarity in study designs and quality. In addition, statistical tests may be used to determine whether or not data sets are compatible with a common probability distribution (Hedges and Olkin, 1985; Stiteler et al., 1993). In general, risk assessors should be reluctant to combine data sets for the purpose of developing a PDF that characterizes variability. Due to the number of potential differences inherent in the study design, alternative data sets may provide a better measure of uncertainty in the probability distribution and parameter estimates, rather than a means of increasing the overall sample size for defining a single probability distribution. For example, if multiple data sets are available, a more informative approach may be to incorporate each data set into the PRA in a separate analysis, as a form of sensitivity analysis on the choice of alternative data sets.

Each probability distribution used in a Monte Carlo Analysis (MCA) should be presented with sufficient detail that the analysis can be reproduced (see Chapter 1, Section 1.4, Condition #2). This information may be presented in tabular and/or graphical summaries. Important information for a summary table would include a description of the distribution type (e.g., lognormal, gamma, etc.), the parameters that define the distribution (e.g., mean and standard deviation, and possibly upper and lower truncation limits for a normal distribution), units, and appropriate references (see Table 3-6, for example). The table should also indicate whether the distribution describes variability or uncertainty. The report should discuss the representativeness of the data and why a particular data set was selected if alternatives were available. Graphical summaries of the distributions may include both PDFs and cumulative distribution functions (CDFs), and should generally be used to document distributions that characterize site-specific data.

3.1.2 CHARACTERIZING RISK USING PRA

Quantitative risk characterization involves evaluating exposure (or intake) estimates against a benchmark of toxicity, such as a cancer slope factor or a noncancer hazard quotient. The general equation used for quantifying cancer risk from ingestion of contaminated soil is shown in Exhibit 3-3, and the equation for noncarcinogenic hazard is shown in Exhibit 3-4. A Hazard Index is equal to the sum of chemical-specific Hazard Quotients.

At this time, this guidance does not propose probabilistic approaches for dose-response in human health assessment and, further, *discourages undertaking such activities on a site-by-site basis*. Such activities require contaminant-specific national consensus development and national policy development (see Chapter 1, Section 1.4.1). Chapter 4 discusses methods for applying probabilistic approaches to ecological dose-response assessment.

The probabilistic calculation of risk involves random sampling from each of the exposure variable distributions. The output of this process is a distribution of risk estimates. When the calculation of risk (or any other model endpoint) is repeated many times using Monte Carlo techniques to sample the variables at random, the resulting distribution of risk estimates can be displayed in a similar fashion. The type of summary graph used to convey the results of a MCA depends on the risk management needs. For example, Chapter 1, Figure 1-3 shows how a PDF for risk might be used to compare the probabilistic estimate of the RME risk (e.g., 95th percentile) with a risk level of concern. This type of summary can also be used to effectively illustrate the relationship between the RME risk determined from point estimate and probabilistic approaches.

EXHIBIT 3-3

EQUATION FOR CANCER RISK

$$Risk = Dose \times CSF$$

Example for Soil Ingestion

$$Risk = \frac{C \times IR \times CF \times EF \times ED}{BW \times AT} \times CSF_{oral}$$

where,

C	=	concentration in soil (mg/kg)	ED	=	exposure duration (years)
IR	=	soil ingestion rate (mg/day)	BW	=	body weight (kg)
CF	=	conversion factor (1E-06 kg/mg)	AT	=	averaging time (days)
EF	=	exposure frequency (days/year)	CSF	=	oral cancer slope factor (mg/kg-day) ⁻¹

EXHIBIT 3-4

EQUATION FOR NONCANCER HAZARD QUOTIENT

$$Hazard\ Quotient = \frac{Dose}{RfD} \text{ or } \frac{Concentration}{RfC}$$

where,

RfD	=	reference dose, oral or dermally adjusted (mg/kg-day)
RfC	=	reference concentration, inhalation (µg/m ³)

In addition, the CDF can be especially informative for illustrating the percentile corresponding to a particular risk level of concern (e.g., cancer risk of 1E-04 or Hazard Index of 1). Figure 3-2 illustrates both the PDF and CDF for risk for a hypothetical scenario. Factors to consider when applying the PDF or CDF are discussed in Chapter 1, Exhibit 1-3. When in doubt about the appropriate type of summary to use, both the PDF and CDF should be provided for all risk distributions. At a minimum, each summary output for risk should highlight the risk descriptors of concern (e.g., 50th, 90th, 95th, and 99.9th percentiles). It can also be informative to include the results of the point estimate analysis—the risks corresponding to the central tendency exposure (CTE) and the reasonable maximum exposure (RME).

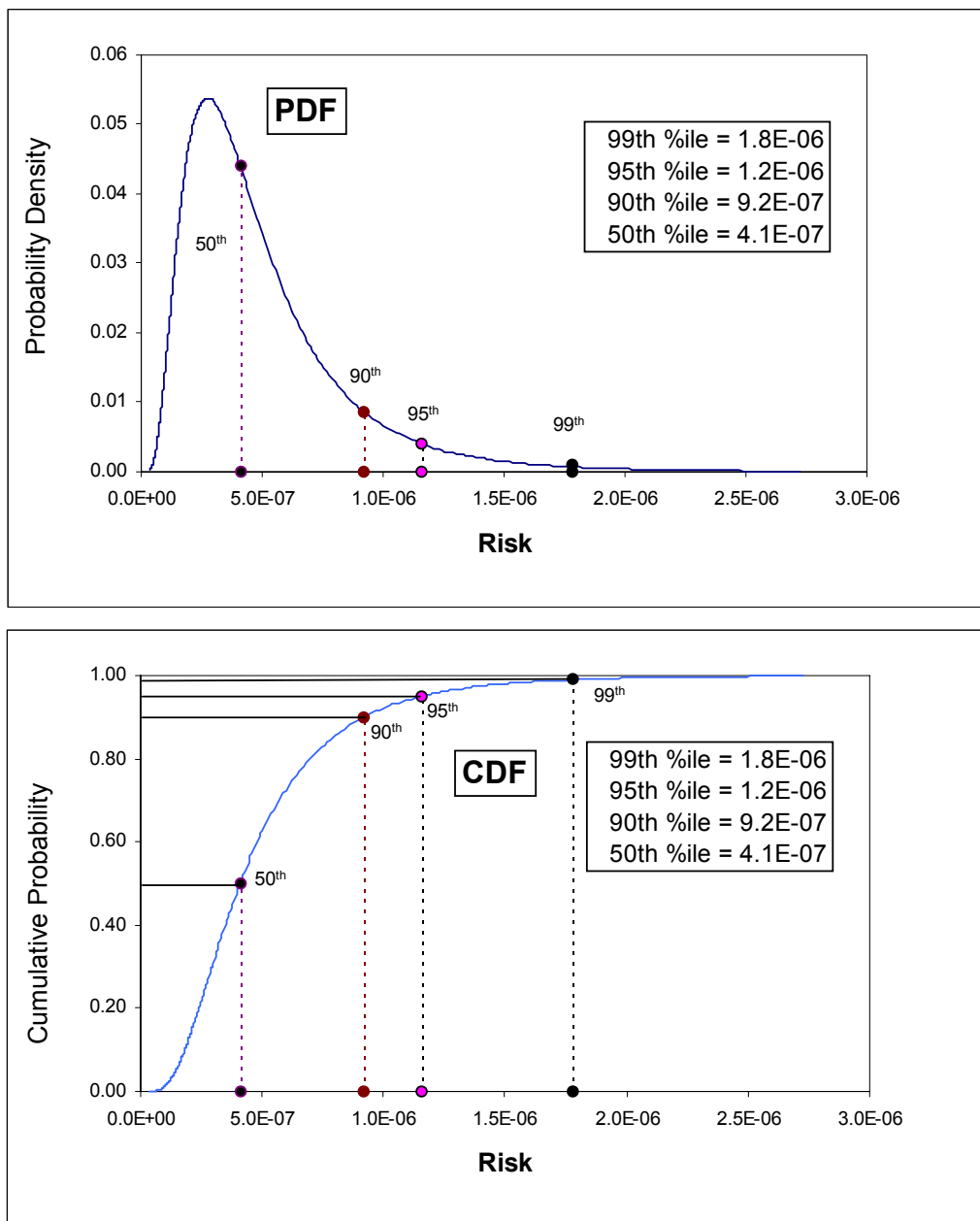


Figure 3-2. Hypothetical PRA results showing a PDF (top panel) and CDF (bottom panel) for cancer risk with selected summary statistics. The CDF rises to a maximum cumulative probability of 1.0. The CDF clearly shows that the level of regulatory concern chosen for this example (1E-06) falls between the 90th and 95th percentiles of the risk distribution.

3.2 ROLE OF THE SENSITIVITY ANALYSIS

Prior to conducting a PRA, it is worthwhile to review several points pertaining to the sensitivity analysis. As shown in Chapter 2 (Figures 2-1 and 2-2), sensitivity analysis can play an important role in decision making at each tier of the tiered process. Beginning with Tier 1, a point estimate for risk should be calculated prior to conducting a PRA. Based on the results of the point estimate, the risk assessor and risk decision makers should determine whether a probabilistic analysis will offer additional benefit. One factor in this decision may be the results of a sensitivity analysis. A primary objective of the sensitivity analysis is to determine which variables and pathways most strongly influence the risk estimate. At many Superfund sites, an estimate of cumulative risk considers contamination in multiple media, moving through multiple pathways and interacting with a number of receptors. Depending on the complexity of the site, and the modeling approaches, a risk assessment may involve one exposure pathway and few variables, or multiple pathways with many variables (e.g., multimedia fate and transport models). However, resources and time are often limited. The sensitivity analysis is invaluable in focusing these limited resources on the most influential variables and pathways.

Several methods for conducting sensitivity analysis are described in Appendix A. It is important to note that when a sensitivity analysis is performed and the major variables are identified, this does not mean that the less influential pathways and variables should be eliminated from the risk assessment. It means that because they are not major contributors to the variability or uncertainty in risk, they can be described with point estimates without affecting the risk management decision. If distributions are readily available for these less influential variables, one may use distributions. The key goal is to provide a comprehensive risk characterization that is scientifically credible and sufficient for risk decision making. The time and effort required to achieve various levels of complexity should be weighed against the value of the information provided to the risk managers.

Additionally, if a variable is specified as influential in the sensitivity analysis, this does not automatically mean that a distribution has to be developed for this variable. If the risk assessor feels that data are simply not sufficient from which to develop a distribution, then a plausible point estimate can be used. The risk assessor should be aware of a possible problem arising from using point estimates in the absence of data adequate to support a distribution. If a variable has the potential to significantly impact the risk outcome, and a very high-end or low-end point estimate is used in the PRA, this has the potential to right-shift or left-shift the final distribution of risk. Even though there might not be enough data to develop a distribution of variability for an influential variable, it would be prudent to communicate the importance of this data gap to the risk decision makers, and perhaps run multiple simulations with several plausible input distributions for that variable. Communication of this uncertainty may persuade the risk decision makers to collect additional data to better define the variable.

3.3 EXPOSURE POINT CONCENTRATION TERM

A brief discussion of the concentration term is provided below. A more complete discussion of the concentration term in PRA is provided in Appendix C. The reader is also referred to Chapter 5 on development of PRGs.

The major source of uncertainty in Superfund risk assessments is often incomplete knowledge of the concentration of one or more chemicals in various exposure media. In any risk assessment, the derivation of the concentration term will reflect assumptions about: (1) properties of the contaminant, (2) the spatial and temporal variability in contamination, (3) the behavior of the receptor, and (4) the time scale of the toxicity of the chemical(s).

Contaminant concentrations contacted by a receptor are likely to vary depending on the spatial variability of contamination and the movements of the receptor. Different individuals may be exposed to different concentrations based on inter-individual variability in activity patterns. If information regarding activity patterns is unavailable, receptors are typically assumed to exhibit random movement such that there is an equal probability of contacting any area within the exposure unit (EU). An EU is defined as the geographical area in which a receptor moves and contacts contaminated medium during the period of the exposure duration. In addition, in Superfund risk assessments, the toxicity criteria are often based on health effects associated with chronic exposure (e.g., lifetime risk of cancer following chronic daily intake over a period of 30 years). Hence, the most appropriate expression for the concentration term, for the majority of risk assessments, is one that characterizes the long-term average exposure point concentration within the EU.

☞ The most appropriate expression of the exposure point concentration term for chronic exposure will characterize the long-term average concentration experienced by a receptor within the exposure unit.

In point estimate risk assessments, the exposure point concentration term is usually calculated as the 95% upper confidence limit (95% UCL) of the arithmetic mean because of the uncertainty associated with estimating the true (i.e., population) mean concentration at a site. If the sampling density is sparse relative to the size of the EU, the uncertainty may be high due to the relatively small number of measurements available to estimate the mean concentration within the EU. The decision to use the upper confidence limit to define the concentration term introduces a measure of protectiveness by reducing the chance of underestimating the mean. Although there will be situations in which modeling variability in concentration will be the appropriate choice (e.g., non-random movement within an EU, acute exposure events, migration of groundwater contaminant plume, migration of fish, etc.), in most cases, characterization of the concentration term will focus on uncertainty. Appendix C provides a more complete discussion on characterizing both variability and uncertainty in the concentration term. Table 3-1 summarizes a number of appropriate methods for characterizing uncertainty in the parameter of an exposure variable, such as the arithmetic mean of the concentration term.

3.4 CHARACTERIZING UNCERTAINTY IN EXPOSURE VARIABLES

Uncertainty is described as a lack of knowledge about factors affecting exposure or risk. To evaluate regulatory options, risk assessors are expected to translate the available evidence, however tentative, into a probability of occurrence of an adverse health effect. Data from a sample or surrogate population are used to develop estimates of exposure and risk in a specific target population (see Section 3.1.4 and Appendix B, Section B.3.1). This extrapolation requires assumptions and inferences that have inherent strengths and limitations, and may bias the outcome of the risk estimate. For example, a common assumption in risk assessments for carcinogens is that a contaminant concentration within the boundaries of a hazardous waste site represents the concentration that a receptor is exposed to throughout the period of exposure, with the corresponding dose averaged over the course of a lifetime. This assumption may be conservative (i.e., result in overestimation of exposure) if it is unlikely that receptors will be exposed at the hazardous waste site for the entire exposure duration. It is incumbent on the risk assessor to clearly present the rationale for the assumptions used in a risk assessment, as well as their implications and limitations.

U.S. EPA guidance, including the *Exposure Assessment Guidelines* (U.S. EPA, 1992a), *Exposure Factors Handbook* (U.S. EPA, 1997a,b,c), and *Guiding Principles for Monte Carlo Analysis* (U.S. EPA, 1997d) have classified uncertainty in exposure assessment into three broad categories:

- (1) *Parameter uncertainty* - uncertainty in values used to estimate variables of a model;
- (2) *Model uncertainty* - uncertainty about a model structure (e.g., exposure equation) or intended use; and
- (3) *Scenario uncertainty* - uncertainty regarding missing or incomplete information to fully define exposure.

Each source of uncertainty is described in detail below, along with strategies for addressing them in PRA.

3.4.1 PARAMETER UNCERTAINTY

Parameter uncertainty may be the most readily recognized source of uncertainty that is quantified in site-specific risk assessments at hazardous waste sites. Parameter uncertainty can occur in each step of the risk assessment process from data collection and evaluation, to the assessment of exposure and toxicity. Sources of parameter uncertainty may include systematic errors or bias in the data collection process, imprecision in the analytical measurements, and extrapolation from surrogate measures to represent the parameter of interest. For example, soil data collected only from the areas of highest contamination, rather than the entire area that a receptor is expected to come into contact, will result in a biased estimate of exposure.

In general, parameter uncertainty can be quantified at any stage of the tiered process, including point estimate analysis (Tier 1), one-dimensional Monte Carlo analysis (1-D MCA) (Tier 2), and two-dimensional Monte Carlo analysis (2-D MCA) (Tier 3). In the point estimate approach, parameter uncertainty may be addressed in a qualitative manner for most variables. For example, the uncertainty section of a point estimate risk assessment document might state that an absorption fraction of 100% was used to represent the amount of contaminant in soil absorbed from the gastrointestinal (GI) tract, and as a result, the risk estimate may overestimate actual risk. In addition, a sensitivity analysis may be performed, wherein one input variable at a time is changed, while leaving the others constant, to examine the effect on the outcome. In the case of absorption from the GI tract, different plausible estimates of the

high-end, or RME absorption fraction might be used as inputs to the risk equation. The differences in the risk estimates would reflect uncertainty in the RME absorption fraction.

Quantitative approaches for characterizing parameter uncertainty in exposure variables in a Monte Carlo simulation are summarized in Table 3-1. If uncertainty in only a few parameter values is of interest, multiple 1-D MCA simulations can yield the same results as a 2-D MCA simulation, but without the time and effort of a 2-D MCA. An example illustrating this concept is given in Table 3-2. With multiple 1-D MCA simulations, variability is characterized in one or more variables using probability distributions for variability (PDFv's), and uncertainty in a parameter is characterized with a series of different point estimates from a probability distribution for uncertainty (PDFu) (e.g., 95% lower confidence limit LCL [95% LCL], sample mean, and 95% UCL). In a 2-D MCA simulation, variability is characterized in one or more variables using PDFv's, and uncertainty in one or more parameters is characterized with PDFu's. With both approaches, the influence of the parameter uncertainty can be presented as a credible interval or confidence interval (CI) around the risk distribution, depending on how the PDFu's are defined. When only a few sources of parameter uncertainty are quantified, multiple 1-D MCA simulations are preferred over a 2-D MCA because the approach is easier to use and communicate. However, if the goal is to explore the effect that many sources of parameter uncertainty may have on the risk estimates simultaneously, a 2-D MCA is preferred. Iterative 1-D MCA simulations with different combinations of confidence limits may be impractical.

Table 3-1. Methods for Characterizing Parameter Uncertainty with Monte Carlo Simulations.

Approach	Example of Model Input	Method	Example of Model Output
Single Point Estimate	<ul style="list-style-type: none"> 95% UCL 	1-D MCA	PDFv ¹ for risk, calculated using the 95% UCL for one parameter.
Multiple Point Estimates	<ul style="list-style-type: none"> 95% LCL sample mean 95% UCL 	1-D MCA	Three PDFv's for risk, representing the 90% CI for each percentile of the risk distribution. ² The 90% CI only accounts for uncertainty in a single parameter (not multiple parameters).
Parametric PDFu ¹	PDFu for the mean based on the sampling distribution, derived from a Student's <i>t</i> -distribution.	2-D MCA	One PDFv for risk with confidence intervals at each percentile of the risk distribution. The CI reflects uncertainty in one or more parameters.
Non-parametric PDFu	PDFu for the mean based on bootstrap resampling methods.	2-D MCA	Same as parametric probability distribution for uncertainty.

¹Probability distribution for uncertainty (PDFu) and probability distribution for variability (PDFv).

²The 95% UCL for the concentration term represents a 1-sided confidence interval (CI), meaning there is a 95% probability that the value is *greater* than or equal to the mean. Similarly, the 95% LCL would represent the 1-sided CI in which there is a 95% probability that the value is *less* than or equal to the mean. Both values are percentiles on the probability distribution for uncertainty (PDFu), also called the sampling distribution for the mean. Together, the 95% LCL and 95% UCL are equal to the 2-sided 90% confidence interval only for cases in which the PDFu is symmetric. For example, the sampling distribution for the arithmetic mean of a sample from a normal distribution with an unknown variance is described with the symmetric Student's *t*-distribution, whereas the PDFu for the mean of a lognormal distribution is asymmetric. In order to compare the results of multiple 1-D MCA simulations and a 2-D MCA simulation, the same methodology should be employed to define the PDFu and the corresponding confidence limits.

It is generally incorrect to combine a PDFu for one parameter (e.g., mean of the concentration term) with one or more PDFv's in other exposure factors when conducting a 1-D MCA for variability.

However, distributions for uncertainty and variability may be appropriately combined in a 2-D MCA. As discussed in Appendix D, with 2-D MCA, a clear distinction should be made between probability distributions that characterize variability (PDFv) and parameter uncertainty (PDFu). A 2-D MCA propagates the uncertainty and variability distributions separately through an exposure model, thereby making it possible to evaluate the effect of each on the risk estimates.

Example: Comparison of Multiple Point Estimates of Uncertainty in 1-D MCA, and Distributions of Uncertainty in 2-D MCA

Table 3-2 illustrates an application of the approaches presented in Table 3-1 for quantifying variability and parameter uncertainty. This is a hypothetical example, and no attempt was made to use standard default assumptions for exposure variables. Two sources of variability are quantified: (1) inter-individual variability in exposure frequency (EF), characterized by a triangular distribution, and (2) inter-individual variability in exposure duration (ED), characterized by a truncated lognormal distribution. In addition, two sources of uncertainty are presented: (1) a point estimate for soil and dust ingestion rate, intended to characterize the RME; and (2) an upper truncation limit of the lognormal distribution for ED, intended to represent a plausible upper bound for the exposed population. Methods for quantifying these sources of uncertainty are discussed below. Additional sources of uncertainty may also have been explored. For example, the choice of a triangular distribution for a PDFv may be provocative for some risk assessors, since there are few cases in which empirical data suggest a random sample is from a triangular distribution. Nevertheless, triangular distributions may be considered rough, or “preliminary” distributions (see Chapter 2 and Appendix B, Section B.2) for cases when the available information supports a plausible range and central tendency.

The choice of distributions is a potential source of uncertainty that can be explored by rerunning simulations with each alternative, plausible choice, and examining the effect on the RME risk. Simulations with preliminary simulations may yield at least three different outcomes. First, this type of sensitivity analysis can help guide efforts to improve characterizations of variability for selected variables that have the greatest affect on the risk estimates. Second, results may provide justification to exit the tiered process without continuing with additional Monte Carlo simulations since further effort would be unlikely to change the risk management decision. Finally, if the major sources of uncertainty can be clearly identified, a subset of the less sensitive variables may be defined by point estimates without significantly reducing the uncertainty in the risk estimates.

Parameter uncertainty can be quantified for both point estimates and PDFv's. In this example, both types of inputs (i.e., point estimates and PDFv's) are presented as sources of parameter uncertainty: the RME point estimate for soil and dust ingestion rate (IRsd), and the upper truncation limit on a PDFv for ED. For IRsd, assume that three different studies provide equally plausible values for the RME: 50, 100, and 200 mg/day. A uniform PDFu is specified to characterize this range of plausible values. For ED, assume that the maximum value reported from a site-specific survey was 26 years, but surrogate data for other populations suggest the maximum may be as long as 40 years. A uniform PDFu is specified to characterize this range of plausible values as well.

In Cases 1-3, the impact of uncertainty in IRsd and ED was evaluated using a series 1-D MCA simulations. Inputs for uncertain parameters associated with IRsd and ED in Case 1, 2, and 3 represent the minimum, central tendency, and maximum values, respectively. Each simulation yields a different risk distribution based on different combinations of point estimates for parameters. Although a PDFu was specified for IRsd, it would have been incorrect to combine the PDFu with the PDFv's for EF and ED in a

1-D MCA because the result would have been a single distribution of risk that co-mingled uncertainty and variability.

In Case 4, a single 2-D MCA simulation was run using the PDFu's for uncertainty and the PDFv's for variability. By propagating variability and uncertainty separately, the 2-D MCA yields a series of distributions of risk, from which credible intervals can be calculated for each percentile of the CDF.

$$Risk = \frac{C \times IR \times CF \times EF \times ED}{BW \times AT} \times CSF_{oral}$$

Table 3-2. Example of 1-D MCA and 2-D MCA.

Variable	Type of Input	1-D MCA			2-D MCA
		Case 1	Case 2	Case 3	Case 4
C (mg/kg)	pt estimate	500	500	500	500
IRsd (mg/day)	pt estimate	50	100	200	see below
	PDFu for pt estimate	--	--	--	uniform (50, 200) ^a
CF (kg/mg)	pt estimate	1E-06	1E-06	1E-06	1E-06
EF (days/year)	PDFv	triangular min = 200 mode = 250 max = 350	triangular min = 200 mode = 250 max = 350	triangular min = 200 mode = 250 max = 350	triangular min = 200 mode = 250 max = 350
ED (years)	PDFv	T-lognormal mean = 9 stdv = 10 max = 26	T-lognormal mean = 9 stdv = 10 max = 33	T-lognormal mean = 9 stdv = 10 max = 40	T-lognormal mean = 9 stdev = 10 max = PDFu (see below)
	PDFu for parameter of PDFv	--	--	--	max ~ uniform (26, 40) ^b
BW (kg)	pt estimate	70	70	70	70
AT (days)	pt estimate	25550	25550	25550	25550
CSF (mg/kg-day) ⁻¹	pt estimate	1E-01	1E-01	1E-01	1E-01

^aUncertainty in the RME point estimate, defined by a uniform distribution with parameters (minimum, maximum).

^bUncertainty in the upper truncation limit of the lognormal distribution, defined by a PDFv with parameters (mean, standard deviation, maximum) and a PDFu for the maximum defined by a uniform distribution with parameters (minimum, maximum).

Monte Carlo Simulation Results

Figures 3-3 and 3-4 illustrate CDFs for risk produced from Monte Carlo simulations using *Crystal Ball*® 2000. The 1-D MCA simulations (Figure 3-3) were run with 10,000 iterations and Latin Hypercube sampling. The 2-D MCA simulation (Figure 3-4) was run with 250 iterations of the outer loop (uncertainty) and 2,000 iterations of the inner loop (variability). Details regarding 2-D MCA simulation are given in Appendix D.

Figure 3-3 shows CDFs for risk based on three simulations of a 1-D MCA simulation. Each simulation used a different combination of plausible estimates of the RME value for IRsd and the upper truncation limit for ED, as discussed above. The results provide a bounding estimate on the risk distribution given these two sources of uncertainty. The 95th percentile risk, highlighted as an example of the RME risk estimate, may range from approximately 7E-06 to 3.5E-05.

Figure 3-4 shows a single CDF for risk, representing the central tendency risk distribution. This CDF was derived by simulating uncertainty in the risk distribution using 2-D MCA. For this example, the 2-D MCA yields 250 simulations of the risk distributions for variability, so that there are 250 plausible estimates of each percentile of the risk distribution. In practice, more than 250 simulations may be needed to adequately quantify uncertainty in the risk distribution. Results of a 2-D MCA can be presented as probability distributions of uncertainty, or box-and-whisker plots of uncertainty at selected percentiles of the risk distributions. Figure 3-4 shows the central tendency (50th percentile) estimate of uncertainty for the entire CDF of risk. In addition, a box-and-whisker plot is shown at the 95th percentile of the CDF. Selected statistics for the box-and-whisker plot are included in a text box on the graphic (i.e., minimum; 5th, 50th, and 95th percentiles, and maximum). The 90% credible interval is given by the 5th and 95th percentiles. For this example, the 90% credible interval for the 95th percentile of the risk distribution is: [9.1E-06, 3.1E-05].

Figures 3-3 and 3-4 demonstrate that the two approaches (i.e., multiple 1-D MCA and 2-D MCA) can yield the same results. However, when there are numerous sources of uncertainty, 2-D MCA offers at least two advantages over multiple 1-D MCA simulations: (1) 2-D MCA allows the multiple sources of uncertainty to be included simultaneously so the approach is more efficient than a series of 1-D MCA simulations; and (2) multiple 1-D MCA simulations yield multiple estimates of the RME risk, but it is not possible to characterize the uncertainty in the RME risk in quantitative terms; a 2-D MCA yields a PDFu for RME risk, which allows for statements regarding the level of certainty that the RME risk is above or below a risk level of concern.

The 95th percentile is a focus of this example because it is a recommended starting point for determining the risk corresponding to the RME. Chapter 7 provides guidance to the risk decision makers on choosing an appropriate percentile (on a distribution of variability) within the RME risk range (90th to 99.9th percentiles). The chapter also includes a qualitative consideration of the uncertainty or confidence surrounding a risk estimate in the decision-making process.

Figure 3-3

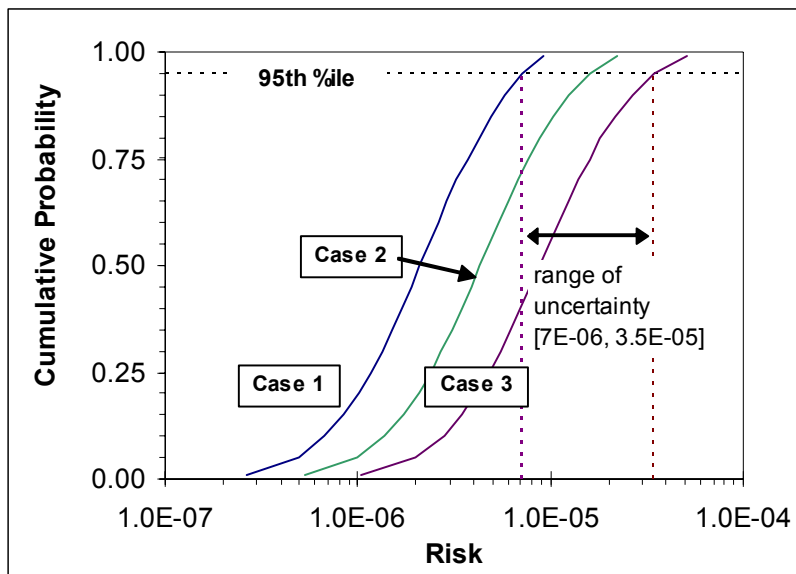
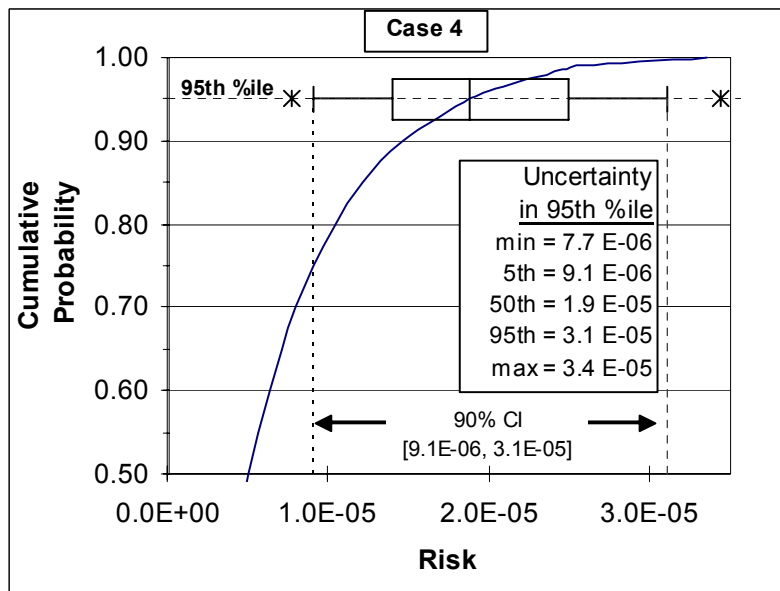


Figure 3-4



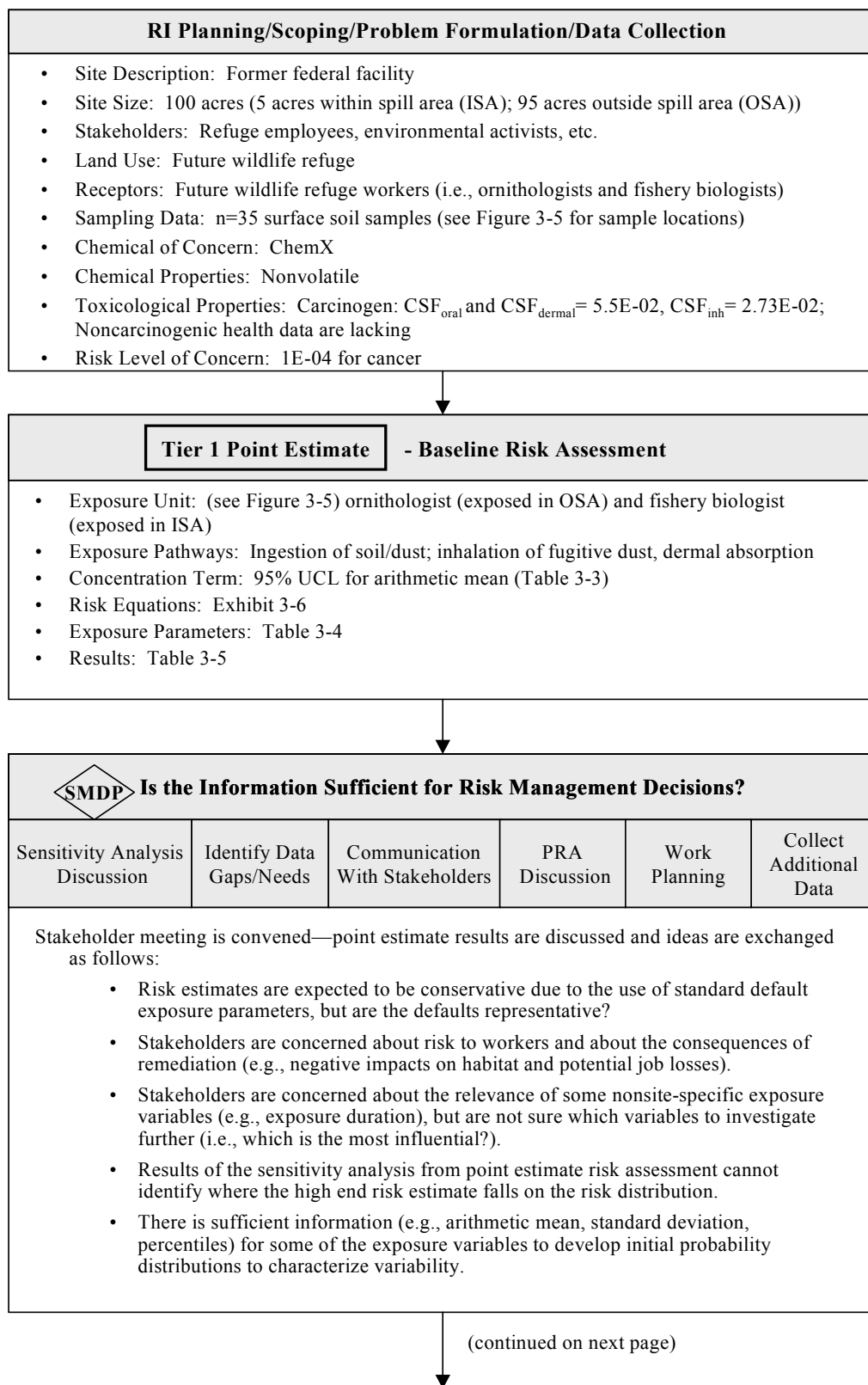
3.4.2 SCENARIO AND MODEL UNCERTAINTY

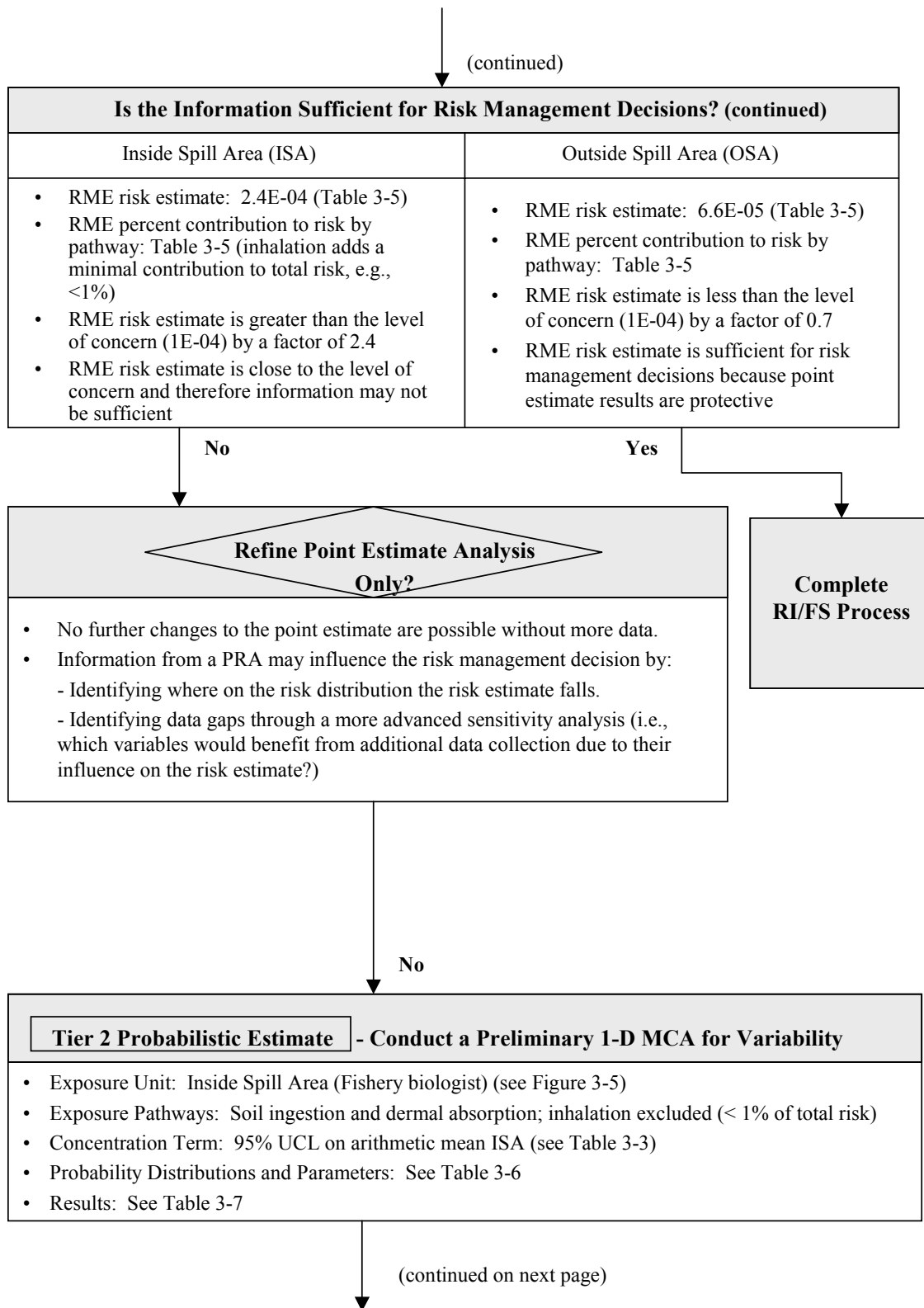
All models are simplified representations of complex biological and physical processes. As such, they, and the scenarios to which they are applied, may introduce a significant source of uncertainty into an exposure and risk estimate. Models may exclude important variables or important pathways of exposure, ignore interactions between inputs, use surrogate variables that are different from the target variables, or they may be designed for specific scenarios and not others. As a result, a model may not adequately represent all aspects of the phenomena it was intended to approximate or it may not be appropriate to predict outcomes for a different type of scenario. For example, a model intended to estimate risk from continuous, steady state exposures to a contaminant may not be appropriate or applicable for estimating risk from acute or subchronic exposure events. In any risk assessment, it is important to understand the original intent of a model, the assumptions being made in a model, what the parameters represent, and how they interact. Based on this knowledge, one can begin to understand how representative and applicable (or inapplicable) a model may be to a given scenario. If multiple models exist that can be applied to a given scenario, it may be useful to compare and contrast results in order to understand the potential implications of the differences. The use of multiple models, or models with varying levels of sophistication, may provide valuable information on the uncertainty introduced into a risk estimate as the result of model or scenario uncertainty. The collection of measured data as a reality check against a given parameter or the predicted model outcome (such as the collection of vegetable and fruit contaminant data to compare against modeled uptake into plants) is also useful in attempting to reduce or at least gain a better understanding of model and scenario uncertainty.

3.5 EXAMPLE OF PRA FOR HUMAN HEALTH

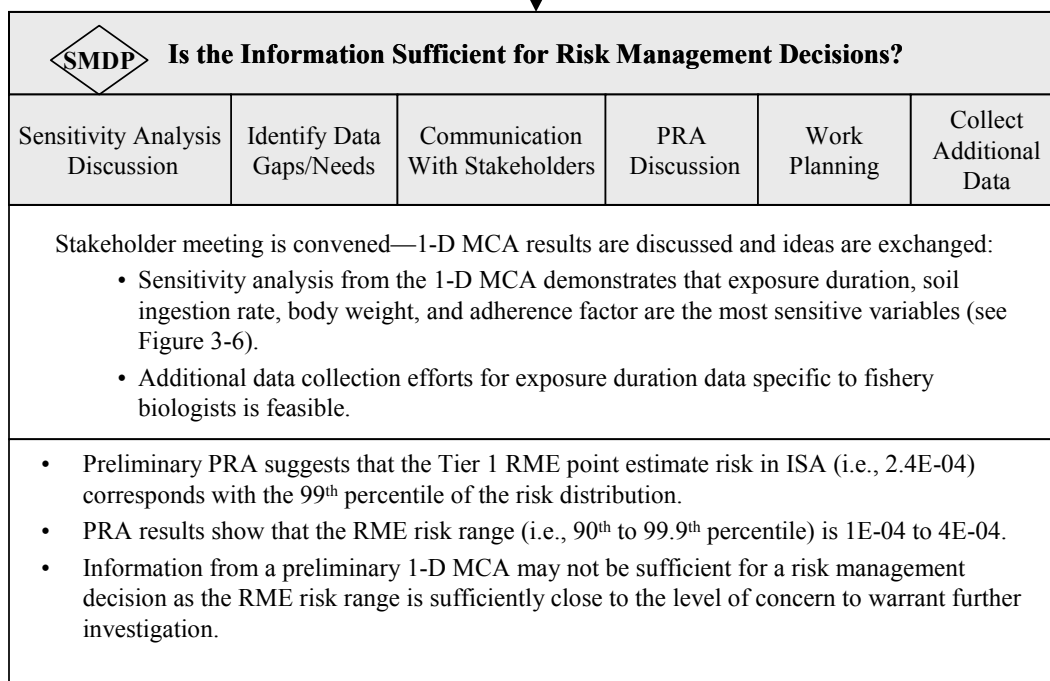
The following hypothetical example provides a conceptual walk-through of the tiered approach for PRA in Superfund risk assessment. The example begins with a baseline human health point estimate risk assessment (Tier 1) and moves to Tier 2, in which multiple iterations of a 1-D MCA are run using default and site-specific assumptions for input distributions. The general concepts associated with the tiered approach are discussed in Chapter 2, and a similar example for ecological risk assessment is given in Chapter 4. The 1-D MCA results are based on simulations with *Crystal Ball*® 2000 using 10,000 iterations and Latin Hypercube sampling. These settings were sufficient to obtain stability (i.e., <1% difference) in the 95% percentile risk estimate. The example is presented in Exhibit 3-5. Tables and figures supporting the example are given immediately following the exhibit.

EXHIBIT 3-5
USING THE TIERED PROCESS FOR PRA
HYPOTHETICAL CASE STUDY FOR HUMAN HEALTH RISK ASSESSMENT

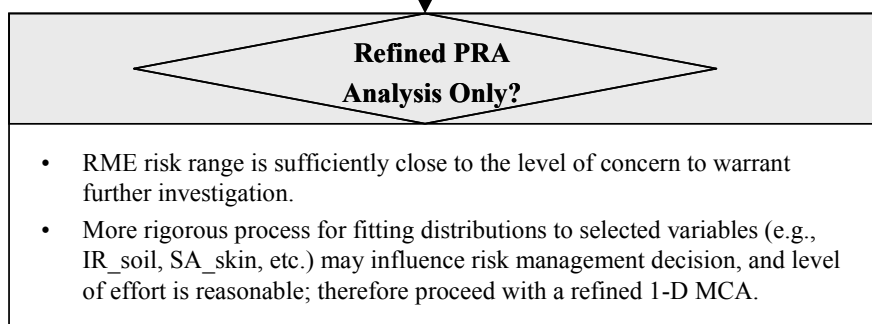




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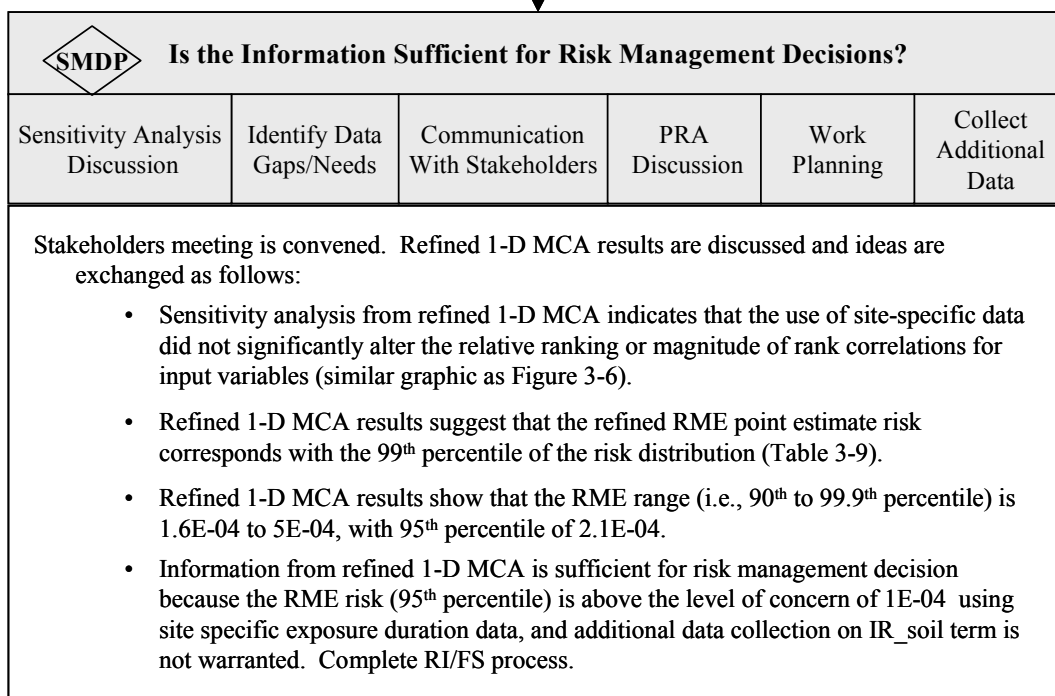
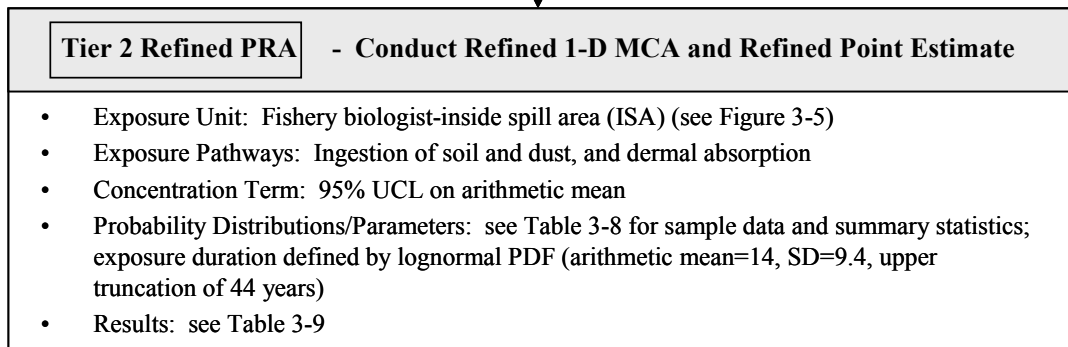
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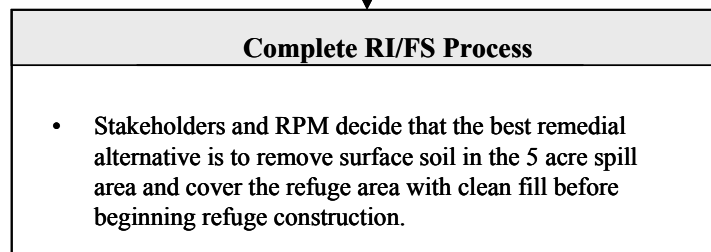
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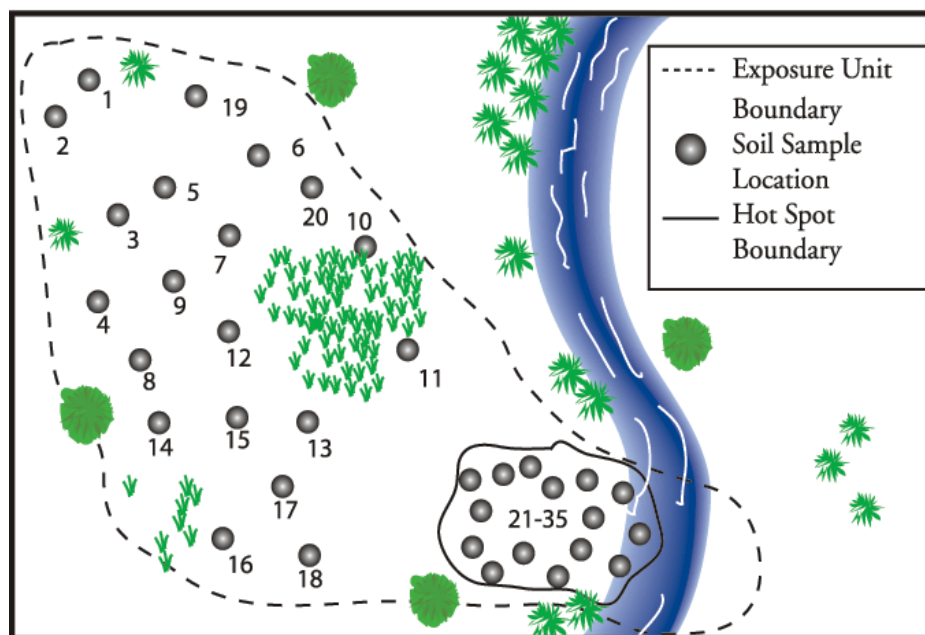


Figure 3-5. Site map for future wildlife refuge showing boundaries for the exposure unit and potential hotspot, as well as sampling locations (n=35). Sample numbers correspond with concentration data given in Table 3-3.

¹The 95% UCL was estimated using the Land method (see Appendix C).

Table 3-3. Concentrations in Surface Soil (mg/kg).

Outside Spill Area (n=20)		Inside Spill Area (n=15)	
1088	305	1934	970
646	2787	402	985
3943	760	4215	743
149	149	1121	158
3704	1088	629	21296
845	837	2293	
488	1295	257	
387	1239	288	
1438	1006	57	
2502	283	228	

Summary Statistics	Outside Spill Area	Inside Spill Area
Mean	1247	2372
Standard Deviation	1121	5348
95% UCL ¹	2303	8444

EXHIBIT 3-6

RISK EQUATIONS

Soil Ingestion

$$\text{Risk} = \frac{Cs \times CF \times IRs \times FI \times EF \times ED}{BW \times AT} \times \text{Oral CSF}$$

Dermal Absorption

$$\text{Risk} = \frac{Cs \times CF \times SA \times AF \times ABS \times EF \times ED}{BW \times AT} \times \text{Dermal-Adjusted CSF}$$

Inhalation of Fugitive Dust

$$\text{Risk} = \frac{Cs \times 1/PEF \times IRa \times ET \times EF \times ED}{BW \times AT} \times \text{Inhalation CSF}$$

Total Risk = Sum of risks from each exposure pathway (soil + dermal + inhalation)

Where:

- Cs = Concentration of ChemX in soil (mg/kg)
- IRs = Soil ingestion rate for receptor (mg/day)
- FI = Fraction ingested from contaminated source (unitless)
- CF = Conversion factor (1E-06 kg/mg)
- SA = Skin surface area available for exposure (cm²/event)
- AF = Soil to skin adherence factor for ChemX (mg/cm²)
- ABS = Absorption factor for ChemX (unitless)
- IRa = Inhalation rate for receptor (m³/hr)
- PEF = Soil-to-air particulate emission factor (kg/m³)
- ET = Exposure time for receptor (hours/day)
- EF = Exposure frequency for receptor (days/year)
- ED = Exposure duration for receptor (years)
- BW = Body weight of receptor (kg)
- AT = Averaging time (years)
- CSF = Cancer slope factor (oral, dermal, inhalation) (mg/kg-day)⁻¹

Table 3-4. Exposure Parameters used in Point Estimate Analysis.

Exposure Variable	CTE Value	RME Value	Units	Reference
IRs	50	100	mg/day	CTE: U.S. EPA, 1997a, p. 4–25 RME: U.S. EPA, 2001
FI	0.5	1	unitless	Site-specific
CF	1E-06	1E-06	kg/mg	Constant
SA	3300	3300	cm ² /event	U.S. EPA, 2001, 50 th percentile value for all adult workers—exposure to face, forearms, and hands
AF	0.1	0.2	mg/cm ²	CTE: U.S. EPA, 1998; Table 3.3, value for gardeners RME: U.S. EPA, 2001
ABS	0.1	0.1	unitless	U.S. EPA, 1998, default for semi-volatile organic compounds (SVOCs)
IRa	1.3	3.3	m ³ /hr	U.S. EPA, 1997a, p. 5–24, outdoor worker hourly average: mean and upper percentile
PEF	1.36E+09	1.36E+09	kg/m ³	U.S. EPA, 2001
ET	8	8	hours/day	Site-specific
EF	200	225	days/year	CTE: Site-specific assumption RME: U.S. EPA, 2001
ED	5	25	years	CTE: U.S. EPA, 1993, p. 6 RME: U.S. EPA, 2001
BW	70	70	kg	U.S. EPA, 1993, p. 7
AT	25550	25550	days	constant

CTE = central tendency exposure; RME = reasonable maximum exposure.

Table 3-5. Point Estimate Risks and Exposure Pathway Contributions.

Risk Estimate by Exposure Pathway	Inside Spill Area (n = 15)		Outside Spill Area (n = 20)	
	CTE	RME	CTE	RME
Soil Ingestion	6.5E-06 (43 %)	1.5E-04 (60 %)	1.7E-06 (43 %)	4.0E-05 (60 %)
Dermal Absorption	8.6E-06 (57 %)	9.6E-05 (40 %)	2.3E-06 (57 %)	2.6E-05 (40 %)
Inhalation	9.9E-10 (< 1 %)	1.4E-08 (< 1 %)	2.7E-10 (< 1 %)	3.8E-09 (< 1 %)
Total Risk	1.5E-05	2.4E-04	4.1E-06	6.6E-05

Example of % contribution: % Soil for RME risk inside spill area = (Soil risk / Total risk) x 100%
= (1.46E-04 / 2.42E-04) x 100% = 60%

Table 3-6. Input Distributions for Exposure Variables used in 1-D MCA for Variability.

Exposure Variable ¹	Distribution Type	Parameters ²	Units	Reference
IR_soil	Triangular	0, 50, 100	mg/day	U.S. EPA, 1993, 2001
SA_skin ³	Lognormal	18150, 37.4	cm ²	U.S. EPA, 1997a, Table 6-4 (Total male/female body surface area)
Absorption Fraction	Uniform	0.1, 0.2	mg/cm ²	U.S. EPA, 2001; minimum truncation limit is professional judgment
IR_air	Lognormal	1.68, 0.72	m ³ /hour	U.S. EPA, 1996, p.5–10
EF	Triangular	200, 225, 250	days	U.S. EPA, 2001; truncation limits are professional judgment
ED	Lognormal ⁴	11.7, 7.0	years	U.S. EPA, 1997b, Table 15-161 and U.S. EPA, 2001 (Mean value is based on average of total median tenure for professional specialty and farming, forestry, and fishing)
	Truncated Lognormal ⁵	14.0, 9.4, 44.0	years	Site-specific survey data, used in refined 1-D MCA
BW	Lognormal	71.75, 14.2	kg	U.S. EPA, 1997a, Tables 7-4 and 7-5; (Combined male/female body weight distributions)

¹All other exposure parameters are inputted as point estimates (see Table 3-4).

²Parameters for lognormal PDF are X ~ Lognormal (arithmetic mean, arithmetic standard deviation) unless otherwise stated. Parameters for triangular PDF are X ~ Triangular (minimum, mode, maximum). Parameters for uniform PDF are X ~ Uniform (minimum, maximum).

³A point estimate of 0.189 was used to adjust the surface area skin (SA_skin) distribution, which is based on total body surface area, to account for skin exposures limited to face, forearms, and hands (U.S. EPA, 1997a, Vol. I).

⁴Parameters for preliminary lognormal PDF for ED were converted from a geometric mean of 10 and a 95th percentile of 25.

⁵Parameters for site-specific lognormal PDF for ED are arithmetic mean, standard deviation, and upper truncation limit.

Table 3-7. 1-D MCA Risk Estimates using Preliminary Inputs.

Cumulative Percentile	Spill Area Risk
50th	5.7E-05
90th	1.3E-04
95th	1.6E-04
99th	2.4E-04
99.9th	3.9E-04

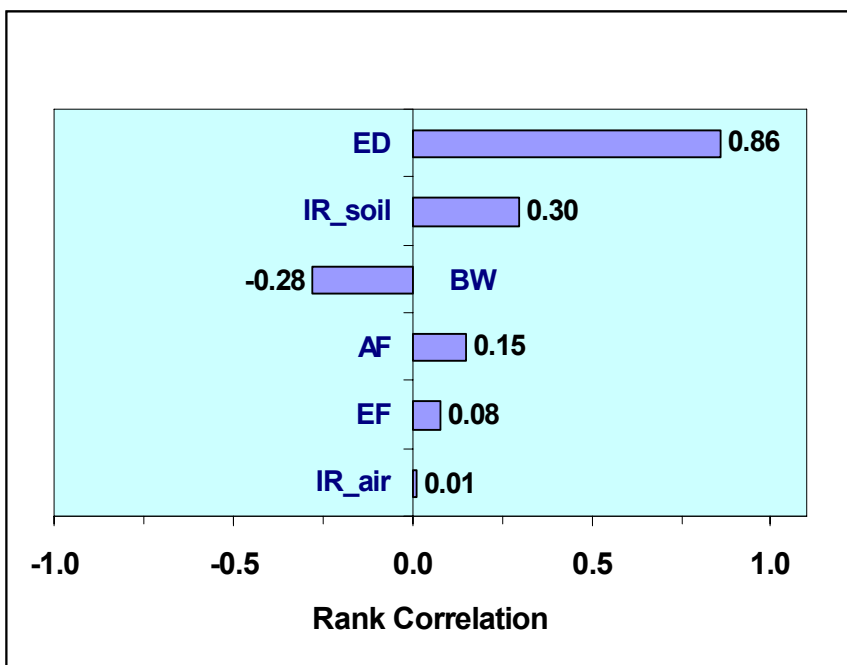


Figure 3-6. Results of sensitivity analysis for preliminary 1-D MCA (Tier 2) showing the Spearman Rank correlations (see Appendix A and B) between input variables and risk estimates.

Table 3-8. Exposure Duration Survey Results.

Survey Results (years)			Summary Statistics	
24.9	20.3	17.2	n	20
8.4	11.7	6.5	min	3.0
3.0	4.7	16.5	max	44.2
6.8	20.9	6.0	arithmetic mean	14.0
18.5	10.6	18.8	standard dev	9.4
9.1	12.7	11.7	median/GM	11.7
7.2	44.2		GSD	1.8

Table 3-9. Refined Point Estimate and 1-D MCA Risk Estimates.

Cumulative Percentile	Spill Area Risk
Refined RME Point Estimate	3.1E-04
50 th	6.7E-05
90 th	1.6E-04
95 th	2.1E-04
99 th	3.2E-04
99.9 th	5.3E-04

REFERENCES FOR CHAPTER 3

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CHAPTER 4

PROBABILISTIC ANALYSIS IN ECOLOGICAL RISK ASSESSMENT

4.1 INTRODUCTION

4.1.1 BASIC APPROACH FOR PERFORMING ECOLOGICAL RISK ASSESSMENTS

Ecological risk assessment (ERA) is defined by the 1997 Environmental Protection Agency's (EPA) *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments (ERAGS)* (U.S. EPA, 1997a) as an evaluation of the "likelihood that adverse ecological effects are occurring or may occur as a result of exposure to one or more stressors". The *ERAGS* document is generally similar to, and consistent with the earlier framework guidance and approach (U.S. EPA, 1992a) which was expanded upon and superseded by the *Guidelines for Ecological Risk Assessment* (U.S. EPA, 1998). The EPA has developed extensive technical and policy guidance on how ERAs should be planned and performed (see Exhibit 4-2). In general, this process has three main elements, as shown in Figure 4-1:

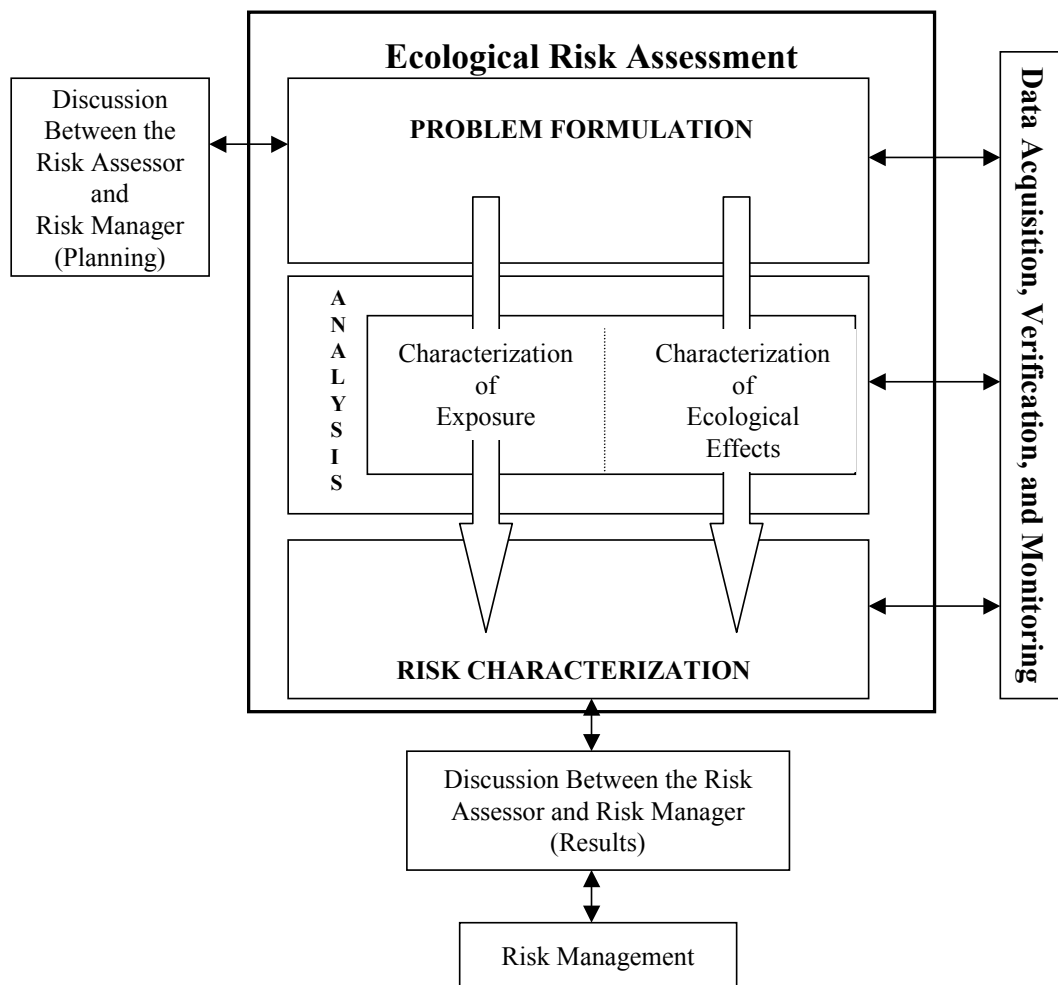


Figure 4-1. Ecological Risk Assessment Framework (U.S. EPA, 1992a)

Problem Formulation provides a foundation for the entire risk assessment. This element includes the specification of risk management goals and assessment endpoints, the development of a site conceptual model with exposure pathways and receptors, and the development of a sampling and analysis plan to collect data on exposures and measures of effects that are needed to support the ERA. In general, problem formulation serves as the foundation of an ERA and often is an iterative process, whereby substantial re-evaluation may occur as new information and data are collected during the site investigations. Collection of data in subsequent iterations is often triggered by identification of major data gaps and uncertainties in the risk characterization that prevent confident decision making by risk managers.

Analysis includes two principal measurement steps that are based upon the problem formulation: Assessment of exposures and assessment of ecological effects. Assessment of exposures includes the identification of stressors at the site that may affect ecological receptors, a characterization of the spatial and/or temporal pattern of the stressors in the environment at the site, and an analysis of the level of contact or co-occurrence between the stressors and the ecological receptors. Assessment of ecological effects includes identification of the types of effects which different stressors may have on ecological receptors, along with a characterization of the relationship between the level of exposure to the stressor and the expected biological or ecological response. This is referred to as the stressor-response relationship.

Risk Characterization combines the exposure characterization and the effects characterization in order to provide a quantitative likelihood or qualitative description of the nature, frequency, and severity of ecological risks attributable to exposure to stressors at a site, as well as an evaluation of the ecological relevance of the effects. Good risk characterizations express results clearly, articulate major assumptions and uncertainties, identify reasonable alternative interpretations, and separate scientific conclusions from policy judgments (U.S. EPA, 1995, 1998).

EXHIBIT 4-1

DEFINITIONS FOR CHAPTER 4

Assessment Endpoint - An explicit expression of an environmental value (ecological resource) that is to be protected, operationally defined by risk managers and risk assessors as valuable attributes of an ecological entity.

Benchmark Dose (BMD) - The dose which causes a specified level of response. The lower confidence limit on the BMD is usually referred to as the BMDL.

Community - An assemblage of populations of different species specified by locales in space and time.

Conceptual Model - A site conceptual model (SCM) in the problem formulation for an ecological risk assessment is a written description and visual representation of predicted relationships between ecological entities and the stressors to which they may be exposed, including sources and pathways of stressors.

Ecological Risk Assessment (ERA) - The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors.

Lines of Evidence - Information derived from different sources or techniques that can be used to characterize the level of risk posed to exposed receptors; weight-of-evidence generally refers to the quantity of science, while strength of evidence generally refers to the quality of science.

Lowest-Observed-Adverse-Effect Level (LOAEL) - The lowest level of a stressor evaluated in a test that caused a statistically significant effect on one or more measurement endpoints linked to undesirable (adverse) biological changes.

Measurement Endpoint (Measure of Effect) - A measurable ecological property that is related to the valued characteristic chosen as the assessment endpoint. Measurement endpoints (also called measures of effect) often are expressed as the statistical or numeric summaries of the observations that make up the measurement.

No-Observed-Adverse-Effect Level (NOAEL) - The highest level of a stressor administered in a test that did not cause a statistically significant effect in any measurement endpoint linked to an undesirable (adverse) biological change.

Population - An aggregate of individuals of a species within a specified location in space and time.

Receptor - The ecological entity (with various levels of organization) exposed to the stressor.

Risk Characterization (ecological) - The third and last phase of ERA that integrates the analyses of exposure to stressors with associated ecological effects to evaluate likelihoods of adverse ecological effects. The ecological relevance of the adverse effects is discussed, including consideration of the types, severity, and magnitudes of the effects, their spatial and temporal patterns, and the likelihood of recovery.

Scientific/Management Decision Point (SMDP) - A time during the ERA when a risk assessor communicates results or plans of the assessment at that stage to a risk manager. The risk manager decides if information is sufficient to proceed with risk management strategies or whether more information is needed to characterize risk.

Species - A group of organisms that actually or potentially interbreed and are reproductively isolated from similar groups; also, a taxonomic grouping of morphologically similar individuals.

Stressor - Any chemical, physical or biological entity that can induce an adverse response in an ecological receptor; Superfund considers all stressors, but focuses on chemical (toxicant) stressors.

Toxicity Reference Value (TRV) - A dose or concentration used to approximate the exposure threshold for a specified effect in a specified receptor. A TRV is often based on a NOAEL or LOAEL from a laboratory-based test in a relevant receptor species.

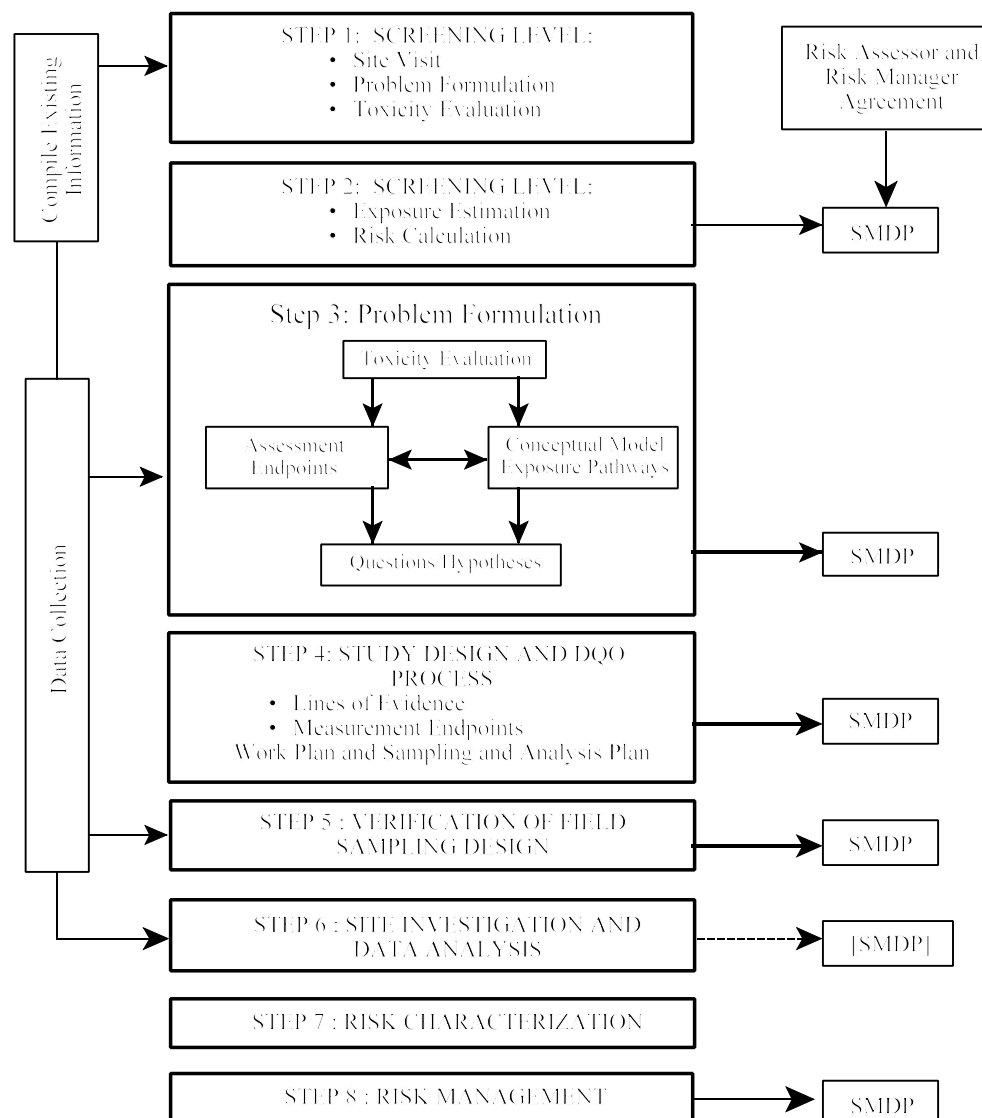
EXHIBIT 4-2

ECOLOGICAL RISK ASSESSMENT GUIDANCE AND POLICY DIRECTIVES

EPA has developed extensive guidance and policies on methods and approaches for performing ERAs, including the following:

- (1) *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments ("ERAGS"), Interim Final* (U.S. EPA, 1997a). This document includes processes and steps specifically selected for use in ERAs at Superfund sites. This document supersedes the 1989 *EPA RAGS, Volume II, Environmental Evaluation Manual, Interim Final* (U.S. EPA, 1989). Supplements to ERAGS include the *EcoUpdates* (U.S. EPA, 1991-present, Intermittent Bulletin Series, 1991 to present), which provide brief recommendations on common issues for Superfund ERAs.
- (2) *Guidelines for Ecological Risk Assessment ("Guidelines")* (U.S. EPA, 1998). This document updates general (nonprogram specific) guidance that expands upon and replaces the earlier *Framework for Ecological Risk Assessment* (U.S. EPA, 1992a). The approaches and methods outlined in the *Guidelines* and in *ERAGS* are generally consistent with each other.
- (3) *Risk Assessment Guidance for Superfund (RAGS): Volume 1—Human Health Evaluation Manual (Part D, Standardized Planning, Reporting, and Review of Superfund Risk Assessments)*, (U.S. EPA, 2001). This guidance specifies formats that are required to present data and results in baseline risk assessments (both human and ecological) at Superfund sites.
- (4) Policy Memorandum: *Guidance on Risk Characterization for Risk Managers and Risk Assessors*, F. Henry Habicht, Deputy Administrator, Feb. 26, 1992 (U.S. EPA, 1992b). This policy requires baseline risk assessments to present ranges of risks based on “central tendency” and “reasonable maximum” (RME) or “high-end” exposures with corresponding risk estimates.
- (5) Policy Memorandum: *Role of the Ecological Risk Assessment in the Baseline Risk Assessment*, Elliott Laws, Assistant Administrator, August 12, 1994 (U.S. EPA, 1994). This policy requires the same high level of effort and quality for ERAs as commonly performed for human health risk assessments at Superfund sites.
- (6) Policy Memorandum: *EPA Risk Characterization Program*, Carol Browner, Administrator, March 21, 1995 (U.S. EPA, 1995). This policy clarifies the presentation of hazards and uncertainty in human and ERAs, calling for clarity, transparency, reasonableness, and consistency.
- (7) Issuance of Final Guidance: *Ecological Risk Assessment and Risk Management Principles for Superfund Sites*. Stephen D. Luftig for Larry D. Reed, October 7, 1999 (U.S. EPA, 1999). This document presents six key principles in ecological risk management and decision making at Superfund sites.

ERA is a key component of the remedial investigation process that EPA uses at Superfund sites. *ERAGS* is a program-specific guidance for Superfund that focuses on chemical stressors released into the environment from hazardous waste sites. This guidance refers to ERA as a “qualitative and/or quantitative appraisal of the actual or potential impacts of contaminants from a hazardous waste site on plants and animals other than humans and domesticated species. An excess risk does not exist unless: (1) the stressor has the ability to cause one or more adverse effects, and (2) the stressor co-occurs with or contacts an ecological component long enough and at a sufficient intensity to elicit the identified adverse effect.” The *ERAGS* document provides guidance on using an eight-step process for completing an ERA for the Superfund Program, as shown in Figure 4-2.



SMDP= Scientific/Management Decision Point

Figure 4-2. Eight-step Ecological Risk Assessment Process for Superfund (U.S. EPA, 1997a).

4.1.2 PREDICTIVE VS OBSERVATIONAL APPROACHES

In general, conclusions about ecological hazards from environmental contamination may be based on information derived from two different techniques: the predictive approach (a comparison of calculated exposures with a set of toxicity reference values), and the observational approach (direct evaluation of the range of potential exposures, coupled with site-specific toxicity testing and population demographic estimates).

Predictive Approach: The core of all Superfund ERAs is the predictive approach, including exposure assessment, toxicity assessment, and risk characterization. The predictive approach is based on a comparison of calculated estimates of chemical exposure of a receptor to one or more Toxicity Reference Values (TRVs) appropriate for that chemical and that receptor. The ratio of exposure at the site to the TRV is referred to as the Hazard Quotient (HQ). The predictive approach has always been used at Superfund sites because it is relatively easy to implement, and because it can be used to evaluate not only current risks, but also risks that might exist in the future if any important changes were to occur in the level of contamination (e.g., due to on-going fate and transport processes), or to changes in land use (a change in land use might alter a number of habitat factors that influence the number and identify of ecological receptors). The predictive approach, however, has the inherent uncertainties of the assumptions in the exposure and toxicity models which are seldom site-specific and thus can lead to either over-protective or under-protective estimates of risk.

Direct Observation: If there is a need to reduce uncertainties in the predictive approach, direct observations of exposure and effects can be collected at Superfund hazardous waste sites. The predictive approach used in ERA does not negate the use of descriptive toxicological approaches or the use of site-specific exposure data, such as toxicity testing or bioaccumulation measurements. Site-specific observations, such as toxicity testing of invertebrates over a gradient of site contaminant exposure levels, may be used to develop site-specific and chemical-specific toxicological relationships. Site-specific measures of exposure or ecosystem characteristics can be used to reduce uncertainty in the exposure assessment and aid in the development of cleanup goals in the Baseline ERA. The direct observation of the exposure and effects on ecological receptors does not however constitute a complete risk assessment. If field or laboratory studies are NOT designed appropriately to elicit stressor-response relationships, direct impacts should not be used as the sole measure of risk because of the difficulty in interpreting and using these results to develop cleanup goals in the ERA. Furthermore, poorly designed toxicological evaluations of environmental media from the site may not allow a definitive identification of the cause of adverse response. For example, receptor abundance and diversity as demographic data reflect many factors (habitat suitability, availability of food, predator-prey relationships among others). If these factors are not properly controlled in the experimental design of the study collecting the observational data, conclusions regarding chemical stressors can be confounded. In addition, direct observation provides information about current risks only and not potential risks should land use or exposure change in the future. Hence, direct observations may be used as a line of evidence in an ERA, but should not be the sole evidence used to characterize the presence or absence of the risks of an adverse effect in the future.

4.1.3 POTENTIAL ADVANTAGES AND LIMITATIONS OF PROBABILISTIC METHODS IN ERA

Probabilistic risk assessment (PRA) is a computational tool that may help increase the strength of the *predictive* evaluation of ecological risks, as well as sometimes helping to better evaluate distributions of observational data for an ERA. The potential advantages of PRA compared to, or possible benefits in augmentation of, the conventional point estimate approach for characterizing variability in exposure or risk are discussed in Chapter 1 and Exhibits 1-6 and 1-7. In brief, point estimate calculations utilize simplifications and assumptions in order to deal with the complex mathematics of combining inputs that are inherently variable. Probabilistic models, in contrast, are designed to combine sets of information on inputs that are expressed as probability distributions. Therefore, PRA generally can yield risk estimates that allow for a more complete characterization of variability and uncertainty, and a potentially more useful sensitivity analysis as compared to estimating sensitivities of inputs from point estimates (see Appendix A). For example, sensitivity analysis can help determine major contributors to exposure factors and sources of uncertainty that could help to design better sampling and analysis plans in later iterations to help fill data gaps and reduce uncertainties for risk characterization.

Because of the inherent differences in the computational approach, as in the case with any additional risk assessment information, PRA may sometimes lead to a different risk assessment outcome and risk management decision than would be derived from the use of point estimate calculations alone. The differences in the decisions stemming from the two approaches will vary from case to case, depending mainly on the form of the exposure or risk model, the attributes of the distributions of the input values, and the quality, quantity, and representativeness of the data on which the input distributions are derived. Sometimes the differences between the two approaches will be quite large, and the information gained from a PRA can play an important role as weight-of-evidence in communicating risks to stakeholders and risk managers.

Even though PRA may have some advantages, it also has limitations and potential for misuse. PRA can not fill basic data gaps and can not eliminate all of the potential concerns associated with those data gaps. That is, if one or more of the input distributions are not well characterized and the distribution(s) must be estimated or assumed, then the results of the PRA approach will share the same uncertainty as the point estimate values. However, given equal states of knowledge, the PRA approach may yield a more complete characterization of the exposure or risk distribution than the point estimate approach.

Of course, any prediction of exposure or risk is based on the use of mathematical models to represent very complex environmental, biological, and ecological systems. No matter how sophisticated the computations, questions will always exist as to whether the calculated values are a good approximation of the truth. Therefore, even when PRA is used as a supplemental tool to point estimations (deterministic) of risks in the ERA process, a weight-of-evidence approach that combines the predictive approach with direct observations will still provide the most appropriate basis for decision making.

A second application of PRA in ERA, besides the characterization and incorporation of distributions of data for ERA, is the characterization of uncertainty in calculated estimates of exposure or risk. In this application, whatever uncertainty may exist in one or more of the input distributions is characterized, and quantitative estimates of the confidence limits around the mean, upper bound, or any other percentile of the output distribution are calculated. This use of PRA is often especially important in risk management decision making, since the range of uncertainty around central tendency exposure (CTE) and reasonable maximum exposure (RME) or other upper bound estimates of exposure or risk can

sometimes be quite large. As stated before, the point estimate approach can also provide estimates of uncertainty, but the PRA approach often provides a more complete characterization of the uncertainty.

4.1.4 FOCUS OF THIS CHAPTER

This chapter focuses on the application of PRA as a tool for predicting ecological risks at Superfund sites. Some of the methods and approaches described in this chapter are similar to those that have been developed by U.S. EPA's Office of Pesticide Programs Committee on Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) Risk Assessment Methods (ECOFRAM, 1999a, 1999b) for use in assessing environmental hazards of pesticide products. However, the methods described in this chapter are specifically designed to be applicable at Superfund sites and to be consistent with other Superfund guidance.

This chapter does not seek to provide guidance on the many basic issues that must be faced in planning and performing any ERA. Prior to considering the use of PRA in an ERA, fundamental concepts will already have been developed, such as a problem formulation with a conceptual site model, selection of representative receptors, definition of exposed populations, definition of risk management objectives and goals, selection of assessment endpoints, calculation of TRVs and development of site sampling plans, etc. Likewise, this chapter does not repeat the presentation of basic statistical and mathematical methods used in PRA, since these are described in other chapters and appendices of this document. In summary:

- ☞ *This chapter focuses on application of PRA techniques to ERA at Superfund sites.*
- ☞ *The reader is assumed to be familiar with the basic methods used in ERA at Superfund sites, and this chapter does not address basic tactical and technical issues in ERA.*
- ☞ *The reader is assumed to be familiar with the basic mathematical principles and techniques of PRA as described in other chapters and appendices of this document.*

4.2 DECIDING IF AND WHEN TO USE PRA IN ECOLOGICAL RISK ASSESSMENT

As shown in Figure 4-2, the ERA process for Superfund includes a number of scientific/management decision points (SMDPs) (U.S. EPA, 1997a). The SMDP is a point of consultation between the risk manager, EPA Regional Biological Technical Assistance Group (BTAG) coordinator, EPA regional ecotoxicologist, and other stakeholders, and is intended to provide an opportunity for re-evaluation of direction and goals of the assessment at critical points in the process. It is during the SMDP discussions that it is important to decide whether or not a PRA is likely to be useful in decision making. If so, the pursuit of distributed data is justified. Within the 8-step process of developing the ERA, PRA could provide insight at several steps. A decision to move forward with distributional analyses should be considered within the BTAG context during the documentation of the outcome of the SMDPs after Step 3 within the process. As a reminder, PRA is NOT intended to be a replacement for point estimate analyses; rather PRA supplements the required presentation of point estimates of risk. It is also emphasized that the use of PRA should never be viewed as or used in an attempt to simply generate an alternative risk estimate or PRG, compared to that which was derived by a point estimate ERA; instead, PRA should be

used to provide insightful information on distributions of various factors (exposure, toxicity, and hazards) which can provide weight-of-evidence evaluations of potential risks in conjunction with a point estimate ERA. There are a number of factors to consider in making these decisions, as discussed below.

4.2.1 TECHNICAL CONSIDERATIONS

The fundamental reason for performing any predictive risk assessment (point estimate or probabilistic) is to provide information to risk managers in order to help support the risk management decision-making process. As noted above, a properly performed PRA may help to yield more description of variability in exposure and risk than can be achieved using the point estimate approach. Therefore, if any of a site's data may be better described and evaluated by distributions, then a PRA can be applied to any part of an ERA or even to the entire ERA for expressing risk characterization in probabilistic terms; again, always in conjunction with the required point estimate ERA. However, when risk estimates derived from the point estimate approach are either far below or far above a level of risk management concern, any such potential improvements in risk characterization are not likely to influence risk management decision making. In these cases, PRA is not likely to be as useful in decision making. Even so, PRA may help in these situations by providing information that may be useful in better deciding where the gradient of excess risks are reduced to acceptable levels. Rather, it is more common for a PRA to be useful when point estimates of risks are close to the decision threshold (such that PRA-based refinements in the risk estimates might be important in making risk management decisions). It is for this reason that PRA may be useful to apply either during the development of the ERA after the screen (Steps 3 to 6, U.S. EPA, 1997a), or after point estimate results from the baseline ERA have been completed (Steps 1 to 7, U.S. EPA, 1997a).

The results of a point estimate risk assessment will normally present the range of risks based on central tendency exposure and reasonable maximum exposure input assumptions and on the no-observed-adverse-effect-level (NOAEL)- and lowest-observed-adverse-effect-level (LOAEL)-based TRVs (U.S. EPA, 1992b, 1997b). The bounds for the highest HQ are derived from the ratio of the RME compared to the NOAEL-based TRV, and the bounds for the lowest HQ are based on the ratio of the CTE compared to the LOAEL-based TRV. These two bounded extreme estimates of risk can be used to screen out cases where PRA is not likely to be as useful. That is, if the risk to the RME receptor is clearly below a level of concern using the NOAEL-based TRV, then risks to the exposed population are likely to be low and PRA analysis is likely not needed. Likewise, if risks to the CTE receptor are clearly above a level of concern using the LOAEL-based TRV, then risks to the exposed population are likely to be of definite concern, and a PRA may not provide as much additional useful information to the risk manager, except in the case where uncertainties remain high and the derivation of an appropriate and realistic clean-up goal may be difficult. If the risks are intermediate between these two bounds (e.g., risks to the CTE receptor are below a level of concern based on the LOAEL-based TRV but are above a level of concern based on the NOAEL-based TRV), then PRA might be helpful in further characterizing the site risks in balance with the point estimates of risks and in supporting decision making or in deciding if additional iterations of analyses would be needed. This concept is illustrated graphically in Figure 4-3.

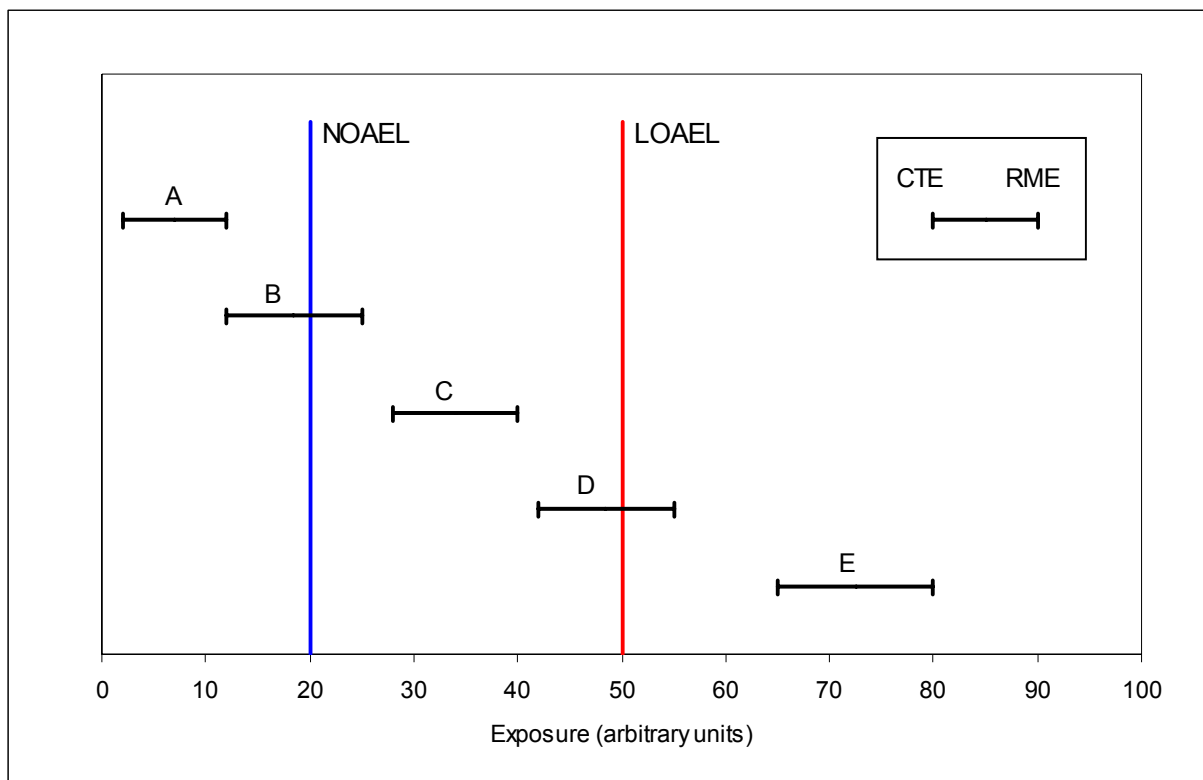


Figure 4-3. Example of cases where use of PRA may be helpful. In cases A and E, the range of risks (CTE to RME) estimated by the point estimate method are either well below (Case A) or well above (Case E) the likely level of concern based on the NOAEL-LOAEL range, and PRA is not likely to alter risk management decisions regarding the potential need for remediation. In cases B, C, and D, the point estimates of risk overlap or fall within the range of potential concern, suggesting that PRA-based risk estimates might be helpful in supporting risk management decisions.

The second main technical reason to consider conducting PRA is that the PRA methodology can help characterize and quantify the degree of variability and uncertainty around any particular estimate of exposure or risk (e.g., the CTE or RME). The purpose of the analysis would be to estimate the uncertainty around an exposure or toxicity or risk estimate, generally with little or no additional data acquisition. The only additional information needed to perform the analysis is an estimate of the uncertainty in the true parameter values of the key variables in the variability model. In some cases, these estimates of uncertainty around parameter values may be developed from statistical analysis of the available data. Alternatively, professional judgment may be used to establish credible bounds on the parameters, especially when relevant data are sparse.

Even in the presence of data gaps, uncertainty analysis using PRA can provide useful information. Indeed, it is when data are limiting or absent that a quantitative probabilistic analysis of uncertainty may be most helpful.

4.2.2 COST AND SCHEDULE CONSIDERATIONS

Performing a PRA can sometimes add time and cost to an ERA. As discussed in Chapter 2, in part, the decision to progress from a point estimate assessment to a PRA reflects a belief that the potential value of the PRA for risk management decision making outweighs the additional time and costs. The tiered process encourages a systematic approach for both the point estimate and probabilistic assessments, whereby the least complex methods are applied first. For example, the initial Tier 2 assessment may be conducted with a set of preliminary probability distributions for variability (PDFv), developed with much the same information and assumptions that were applied to develop point estimates in Tier 1. Parameter values can be estimated by setting the arithmetic mean equal to the CTE point estimate, and the 95th percentile equal to the RME point estimate. The choice of distributions may differ depending on the state of knowledge for a particular variable (see Appendix B). For example, unbounded variables might be characterized with lognormal distributions while bounded distributions are characterized by beta or Johnson Sb distributions. Certain variables may continue to be characterized by point estimates, especially if the sensitivity analysis suggests that the chemical, pathway, and/or exposure variables are relatively minor contributors to total exposure and risk. The decision to collect additional data or explore alternative methods for developing probability distributions can be reexamined in an iterative fashion by evaluating the expected benefits of the added information to the risk management decision-making process. These concepts are presented in greater detail in Chapter 2 (see Figures 2-1 and 2-2).

4.3 PROBLEM FORMULATION

Once a decision has been made to include PRA in an ERA, the first step should be to re-visit the problem formulation step and carefully determine the scope and objectives of the PRA. Typically, a considerable amount of knowledge will have been gained during the screening level and baseline point estimate evaluations, and this knowledge should be used to help focus and narrow the scope of the PRA. That is, the PRA will generally utilize the same basic exposure and risk models used in the point estimate approach, but the PRA will typically evaluate only a sub-set of the scenarios considered. For example, chemicals, pathways, and/or receptors that are found to contribute a negligible level of exposure or risk may usually be omitted from the PRA, while those factors that contribute significantly to an excess level of risk concern in the point estimate approach should generally be retained. As noted previously, when a chemical or pathway is omitted from a PRA analysis, this does not mean that it is eliminated from the overall risk assessment; rather, it may be kept in the assessment as a point estimate.

The next step in problem formulation for a PRA should be to define whether the goal of the analysis is to characterize variability alone, or to characterize both variability and uncertainty. In either case, sensitivity analysis (as summarized in the preceding paragraph, or for more details see Appendix A) should be used to help identify which of the input variables contribute the most to the variability in the outputs (exposure, toxic effects, or risk), and the initial PRA should focus on defining the probability density functions (PDFs) for those input variables. An analysis of uncertainty, if thought to provide additional useful information, may also be included at the initial level, or may be delayed until the initial analysis of variability is completed.

As always, problem formulation should be viewed as an iterative process, and it is reasonable and appropriate that decisions regarding the scope and direction of the PRA should be reassessed (at SMDPs) as information becomes available from the initial evaluations. As stressed above, the fundamental criterion which should be used is whether or not further PRA evaluations are likely to provide additional information to a point estimate ERA that will help strengthen and support the risk management decision-making process.

4.4 MODELING VARIABILITY IN EXPOSURE

There are two main types of descriptors of exposure that may be used in ERA: dose and concentration. For terrestrial receptors such as mammals or birds, exposure is most often described in terms of ingested dose (mg/kg-day). In most cases, this will be based on chemical ingested from drinking water and/or the diet, including incidental soil ingestion, but could also include amounts of chemical taken up across the skin or through inhalation as additional routes of exposure. The exposure levels are most often expressed as doses, since that term tends to normalize the confounding factors of variable daily intake rates and body weights that occur if/when one only evaluates concentrations. For aquatic receptors, the main route of exposure is usually by direct contact and less often by ingestion, so exposure is usually characterized in terms of concentration of contaminants in surface water, pore water and/or sediment. Likewise, exposure of terrestrial plants and terrestrial invertebrates, such as earthworms, is usually described in terms of concentration of contaminants in soil. In some cases, exposure of terrestrial receptors is characterized in terms of specific tissue or whole-body concentrations of contaminants. Examples of calculating and presenting dose-based and concentration-based distributions of exposure are presented below.

4.4.1 CHARACTERIZING VARIABILITY IN DOSE

The general equation used for calculating the **dose** of a contaminant of concern in a specified environmental medium (e.g., water, soil, air, diet, etc.) by a particular member of a population of exposed receptors is:

$$DI_{i,j} = C_i \times IR_{i,j} / BW_j$$

where:

$DI_{i,j}$	=	Average daily intake of chemical due to ingestion of medium "I" by a population member "j" of the exposed population (mg/kg-day)
C_i	=	Concentration of chemical in environmental medium "I" (mg/unit medium)
$IR_{i,j}$	=	Intake rate of medium "I" at the site by population member "j" (units of medium per day)
BW_j	=	Body weight of population member "j" (kg)

Total exposure of a population member "j" is then the sum of the exposures across the different media:

$$DI_{total,j} = \sum DI_{i,j}$$

In this basic equation, $IR_{i,j}$ and BW_j are random variables (i.e., they have different measurable values for different members of the exposed population) that are often correlated. For example, a receptor with a relatively low intake rate can also be expected to have a low body weight. Some studies utilize paired measurements of IR and BW by individual, and present a distribution of the ratio ($IR_{i,j}/BW_j$), referred to as a body weight-normalized intake rate (mg/kg-day). This expression provides an alternative to using a correlation coefficient to relate two input variables (see Appendix B), and can be entered into the dose equation as follows:

$$DI_{i,j} = C_i \times \left(\frac{IR_{i,j}}{BW_j} \right)$$

where the ratio is characterized by a single probability distribution. Because the variability in this ratio is likely to be different than the variability in the ratio of the IR and BW variables treated independently,

accounting for the correlation can affect the distribution of dose and risk. If empirical data for quantifying the ratio are limited but a relationship is expected, plausible ranges of correlations may be explored as a source of uncertainty in the risk estimates.

The concentration term (C_i) may be characterized by a point estimate or a probability distribution, depending on the relationship between the geographic scales of the measurement data and receptor home range (see Appendix C, Section C.3.1). If the home range of the receptor is small compared to the spatial distribution of sampling locations, C_i may be characterized by the probability distribution for variability in measured concentrations. Alternatively, if the home range is large compared with the exposure area evaluated, then a point estimate (e.g., mean or uncertainty in the mean) may be more appropriate.

In the PRA approach, PDFs should be defined for as many of the input variables as reasonable, especially for those variables that are judged (via sensitivity analysis) to contribute the most to the variability in total exposure. The basic principles for selecting the key variables to model as PDFs are presented in Appendix A, and the basic methods used for selecting and fitting distributions are described in detail in Appendix B.

Figure 4-4 shows several examples of graphical formats which may be used to present the estimated distribution of ingested doses in an exposed population. If a single distribution is plotted (top panel), the PDF format is usually the most familiar and useful for risk assessors and managers, but the cumulative distribution function (CDF) format tends to be less cluttered when multiple distributions are shown (e.g., compare the middle graph to the bottom graph). In addition, percentiles can be read directly from a CDF format, but not from a PDF format graph. In all cases, it is very useful to superimpose the CTE and RME point estimate ranges of exposure directly on the same graph as is used to show the distribution of exposures estimated by PRA. This provides a convenient way to compare the results of the two alternative computational methods, and interpret additional information that the PRA can add to the point estimate ERA.

- ☞ *A conventional point estimate, range of exposure (CTE to RME) or toxicity (NOAEL to LOAEL) and corresponding risk ranges should be calculated and presented for comparison with the PRA results.*

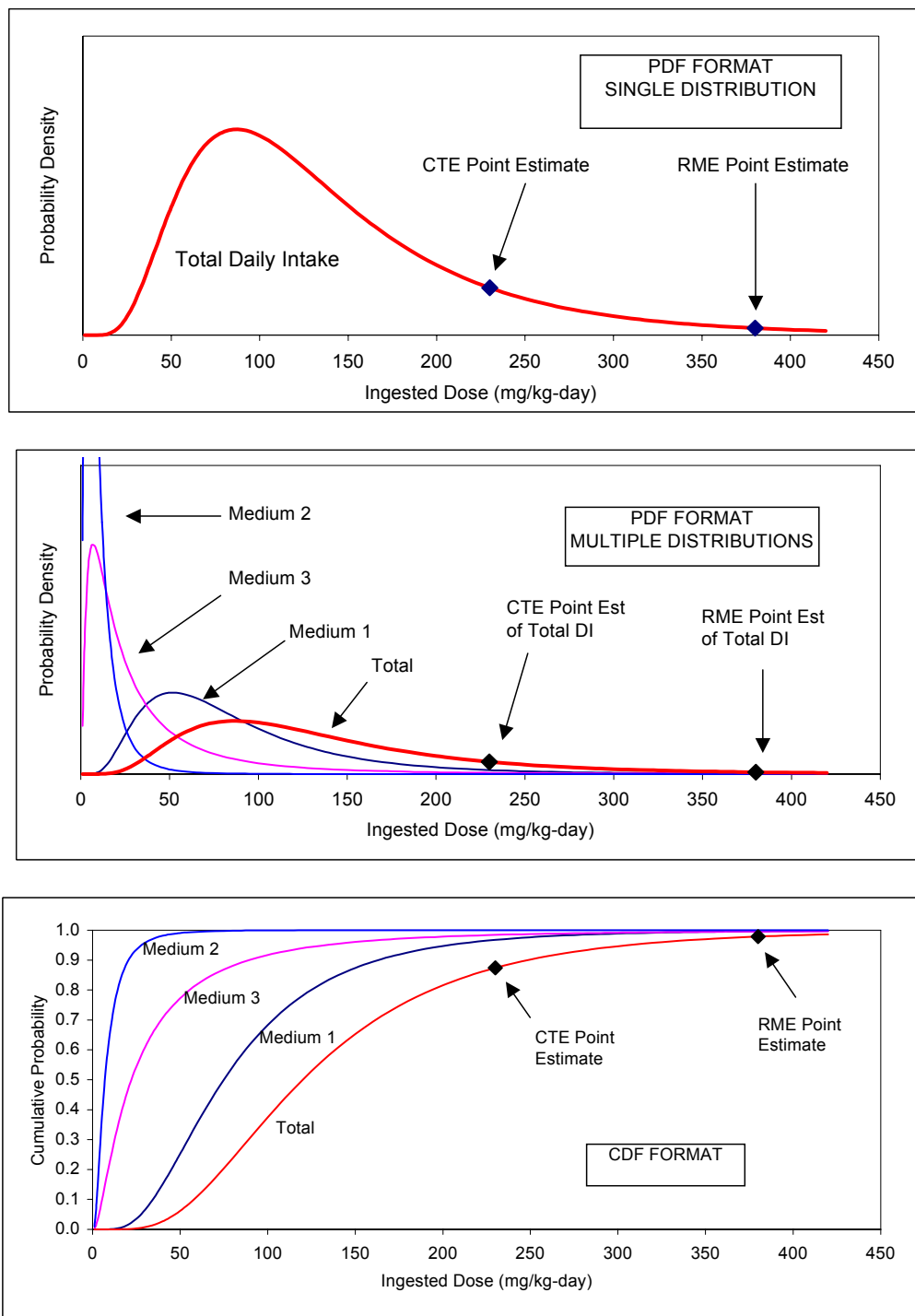


Figure 4-4. Example Graphical Presentations of Dose Distributions.

4.4.2 CHARACTERIZING VARIABILITY IN EXPOSURE CONCENTRATION

As noted above, in some cases the most appropriate descriptor of exposure is concentration (either in an abiotic medium such as water, soil, or sediment, or in the tissues of the receptor), rather than ingested dose. Assuming that the concentration values in the medium of concern are measured rather than modeled, PRA is not required to generate the distribution of concentrations. Rather, the available data may be used to define an appropriate theoretical or empirical distribution function (EDF), as described in Appendix B. If concentrations in the medium are modeled (calculated by PRA) rather than measured, then the exposure distribution may be estimated by using distribution functions (PDFs or CDFs, rather than using point estimates as inputs to the fate and transport model(s) and/or uptake models that predict the concentration levels in the medium of concern. The resulting distribution(s) of concentration may be displayed graphically using the same formats as illustrated in Figure 4-4, except that the x-axis has units of concentration rather than dose. As above, the point estimate ranges of concentration used in the CTE and RME calculations should be plotted on the same graphs to provide a convenient basis for comparing the results of the two approaches and to help interpret the additional information that the PRA can add to the point estimate outputs.

4.5 MODELING VARIABILITY IN TOXICITY

4.5.1 VARIABILITY IN RESPONSE AMONG MEMBERS OF A POPULATION

Data on the toxicity of a chemical usually comes from laboratory studies whereby groups of organisms (laboratory mammals, fish, benthic organisms, plants, earthworms, etc.) are exposed to differing levels of chemical, and one or more responses (endpoints) are measured (survival, growth, reproduction, etc.). These toxicological observations define the exposure-based stressor-response curve that is characteristic for that specific receptor, chemical, and response.

In the point estimate approach, information from the dose/stressor-response curve is generally converted to one or more TRVs, each representing a specific point on the dose-based or concentration-based stressor-response curve. For example, the highest dose or concentration that did not cause a statistically significant change in a toxicologically significant endpoint is defined as either the NOAEL dose or the no-observed-effect concentration (NOEC), while the lowest dose or concentration that did cause a statistically significant effect on a relevant endpoint is the LOAEL dose or the lowest-observed-effect concentration (LOEC). Generally, exposures below NOAEL- or NOEC-based TRVs are interpreted to pose acceptable risk, while exposures above LOAEL- or LOEC-based exposures are judged to pose potentially unacceptable risk. It is essential to note the need for high quality toxicity data to derive reliable and confident TRVs. Strong sampling and study designs, that generate data for site exposure factors and toxicological stressor-response relationships, are of critical importance for producing high quality ERAs by either point estimate or PRA approaches. Shortcomings in either area could be major data gaps or uncertainties that detract from the confidence in the risk characterization of the ERA, and may be a basis for pursuing additional iterations of sampling or studies that are more strongly designed to fill those critical data gaps and reduce uncertainty.

Use of the TRV approach, however, does have some potential limitations. Most important is that the ability of a study to detect an adverse effect depends on both the range of doses tested and the statistical power of the study (i.e., the ability to detect an effect if it occurs). Thus, studies with low power (e.g., those with only a few test animals per dose group) tend to yield NOAEL or NOEC values that are higher than studies with good power (those with many animals per dose group). In addition, the choice of the TRV is restricted to doses or concentrations that were tested, which may or may not be close

to the true threshold for adverse effects, and this uncertainty increases as the interval between doses increases. Finally, it is not always easy to interpret the significance of an exposure that exceeds some particular TRV, since the severity and incidence of response depends on the shape and slope of the exposure response curve (information that is not captured in a point estimate TRV).

One way to resolve some of these stressor-response limitations is to apply uncertainty factors to the NOAEL or NOEC and LOAEL or LOEC, which calculates an adjusted TRV that reduces the study's exposure level of concern to account for those uncertainties, so that there is a lesser chance of overlooking possible adverse exposures (i.e., avoiding a false negative conclusion). Another way to resolve some of the stressor-response limitations is to fit a mathematical equation to the available exposure-response data and describe the entire exposure-response curve. This may be done using any convenient data fitting software, but EPA has developed a software package specifically designed for this type of effort. This software is referred to as the Benchmark Dose Software (BMDS), and is available along with detailed documentation and explanation of the methodology at www.epa.gov/ncea/bmds.htm.

The most appropriate mathematical form of the exposure-response model depends on whether the endpoint measured is discrete and dichotomous (e.g., survival) or continuous (e.g., growth rate). For a dichotomous endpoint, the result of the fitting exercise is a mathematical exposure-response model P that yields the probability of a response in an individual exposed at any specified level of exposure (expressed either as dose or concentration). Exhibit 4-3 shows an example of this process using hypothetical data. Thus, for an individual with an exposure level of " x ", the probability of a response in that individual is simply $P(x)$. In a population of individuals with exposures $x_1, x_2, x_3, \dots, x_i$, the expected number of responses (e.g., deaths) in the exposed population is the sum of the probabilities across all individuals in the population. Stated another way, the average fraction of the population that will experience the response is given by the expected value of P (i.e., the average value of $P(x)$).

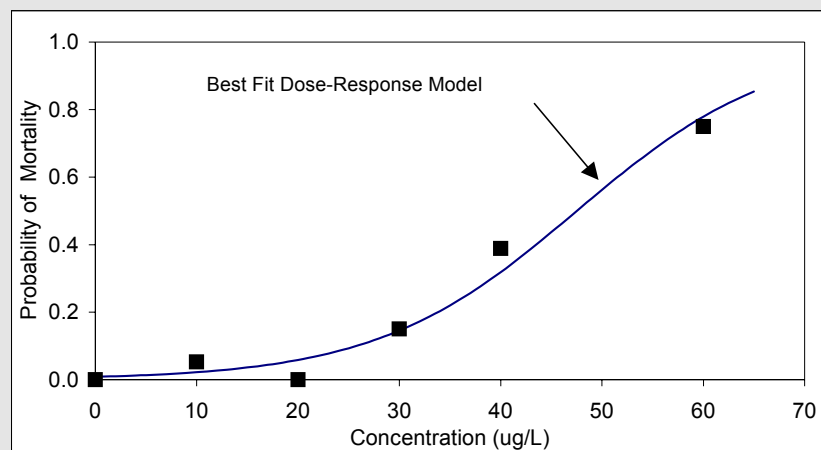
EXHIBIT 4-3

MODELING VARIABILITY IN RESPONSE FOR A DICHOTOMOUS ENDPOINT

The following data are from a hypothetical study of the acute lethality (24 hour) of a chemical using fathead minnows as the test organism:

Concentration ug/L	Number Tested	Survival	
		Dead	Alive
0	20	0	20
10	19	1	18
20	20	0	20
30	20	3	17
40	18	7	11
60	20	15	5

These data were fit to each of the dichotomous models available in BMDS. The best-fit model was the logistic equation. A graph of the best fit curve is shown below.



Basic Equation

$$\text{Probability of mortality (conc)} = 1 / (1 + \exp(-a - b \cdot \text{conc}))$$

Best fit parameters

a	-4.80
b	0.101

Goodness of Fit

P	0.604	P=Chi Square Goodness of Fit test statistic
AIC	79.12	AIC=Akaike's Information Criterion

For a continuous endpoint, the BMDS software yields equations that give the expected mean response $m(x)$ at a specified exposure level, along with the standard deviation $s(x)$ that characterizes how variable the response is among different individuals exposed at that same exposure level. The standard deviation may be modeled either as a constant (homogeneous variance) or a function of the exposure level (heterogeneous variance), with the choice depending on which approach yields the best agreement with the observed variances. In most cases there will not be sufficient data to allow a meaningful analysis of the true shape of the underlying distribution of responses at a given exposure, so the choice of the distributional form of the variability in response will require an assumption. In the absence of any clear evidence to the contrary, it is considered likely that the distribution of responses will not be strongly skewed, and that the distribution may be reasonably well modeled using a normal PDF (truncated as necessary to prohibit selection of biologically impossible or implausible values). Thus, variability in response at dose "x" may generally be modeled as:

$$\text{Response}(x) \sim \text{NORMAL}[m(x), s(x), \text{min}, \text{max}]$$

However, if available data suggest some other distributional form is more appropriate, that form should be used and justified.

Exhibit 4-4 shows an example of this process using hypothetical data. In this case, the mean response was found to be well modeled by the Hill equation, and the standard deviation was found to be best characterized as a constant ($\rho=0$). Thus, given an exposure level "x", the mean response $m(x)$ may be calculated from the model, and this value along with the standard deviation may then be used as parameters for an appropriate type of PDF (e.g., normal) to describe the expected distribution of responses in a population of different individuals exposed at level "x". Section 4.7.2 describes methods that may be used to characterize and quantify the uncertainty associated with this approach.

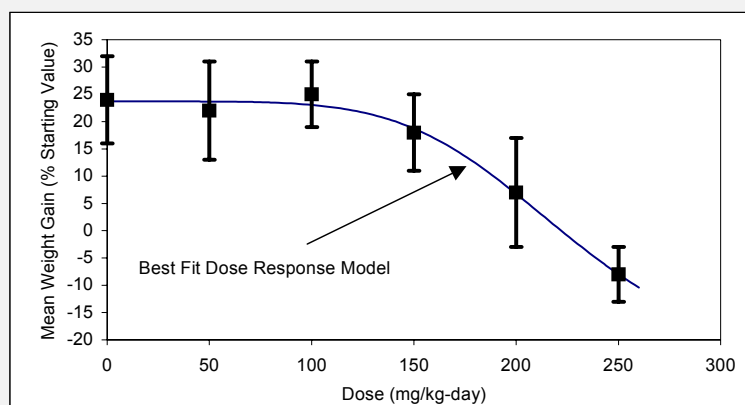
EXHIBIT 4-4

MODELING VARIABILITY IN RESPONSE FOR A CONTINUOUS ENDPOINT

The following data are from a hypothetical study of the effects of a chemical on the growth of laboratory mice. Animals were exposed to the chemical via drinking water for 21 days. The measurement endpoint was weight gain, expressed as a percentage of the starting weight of each animal.

Ingested dose mg/kg-day	Number Tested	Weight Gain (% Starting Value)	
		Mean	Stdev
0	5	24	8
50	5	22	9
100	5	25	6
150	5	18	7
200	5	7	10
250	5	-8	5

These data were fit to each of the continuous models available in BMD5. The best-fit model was the Hill equation with constant variance. A graph of the best fit curve is shown below.



Basic Equations

$$\text{Mean Response}(d) = \text{int} + v \cdot d^n / (k^n + d^n)$$

$$\text{Variance}(d) = \alpha \cdot \text{mean response}(d)^\rho$$

Best fit parameters

int	23.70
v	-51.41
n	5.295
k	228.7
alpha	48.5
rho	0 (constant variance)

Goodness of Fit

P	0.685	P=Chi Square Goodness of Fit test statistic
AIC	154.5	AIC=Akaike's Information Criterion

4.5.2 VARIABILITY IN RESPONSE AMONG SPECIES

In some cases, risk management decisions may also consider community-level effects as well as population-level or sub-populations effects. That is, a stressor might be considered to be below a level of concern for the sustainability of a community if only a small fraction of the total number of exposed species are affected. In this case, toxicological responses may be best characterized by the distribution of toxicity values across species. This is referred to as a Species Sensitivity Distribution (SSD). This type of approach is generally used for communities of aquatic receptors, since all of the different species that make up the community (e.g., all fish, benthic invertebrates, aquatic plants, and amphibians that reside in a stream) will be exposed to approximately the same concentration of contaminant in the water. The process for generating an SSD consists of the following steps:

- (1) Select an appropriate type of endpoint (lethality, growth, reproduction, etc.), and select an appropriate type of point estimate from the exposure-response curve for each species. For example, the TRV might be the LC₅₀ for lethality or the EC₂₀ for growth. The key requirement is that the SSD be composed of TRV endpoints that are all of the same type, not a mixture.
- (2) Collect all reliable values for that type of TRV from the literature for as many relevant species as possible. When more than one value is available for a particular species, either select the value that is judged to be of highest quality and/or highest relevance, or combine the values across studies to derive a single composite TRV for each species. It is important to have only one value per species to maintain equal weighting across species.
- (3) Characterize the distribution of TRVs across species with an appropriate CDF. Note that there is no *a priori* reason to expect that an SSD will be well characterized by a parametric distribution, so both parametric and empirical distributions should be considered.

Once an SSD has been developed, the fraction of species in the exposed community that may be affected at some specified concentration may be determined either from the empirical distribution or from the fitted distribution. Exhibit 4-5 shows examples of this approach. In this hypothetical case, the TRV selected for use was the LC_{low} (in this case, the LC_{low} is defined as all LC values \leq LC₁₀). A total of 13 such values were located. The first graphical presentation is the empirical distribution function, where the Rank Order Statistic (ROS) of each value is plotted as a function of the log of the corresponding value. This may be used directly to estimate the fraction of the species in a community that will be affected by any particular environmental concentration. For example, in this case, it may be seen that a concentration of 10 ug/L would be expected to exceed the LC_{low} for about 33% of the aquatic species for which toxicity data are available. The second graph shows how the data may be characterized by fitting to a continuous distribution. In this case, a lognormal distribution was selected as a matter of convenience, but other distributions may also yield acceptable fits. Based on the best fit lognormal distribution for the SSD data, it is calculated that a concentration of 10 ug/L would be expected to impact about 31% of the exposed species. However, as noted above, there is no special reason to expect that an SSD will be well characterized by a continuous parametric distribution, so some caution should be used in the use of a continuous distribution to fit an SSD, especially when the SSD is based on a limited number of species and when the purpose of the SSD is to estimate percentiles and exposures outside the observed range. The risk assessor should always present an evaluation of the robustness of an SSD to aid in the decision process.

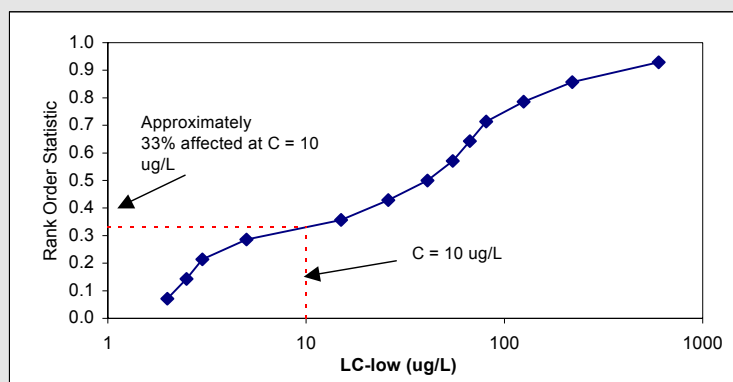
EXHIBIT 4-5

HYPOTHETICAL SPECIES SENSITIVITY DISTRIBUTION

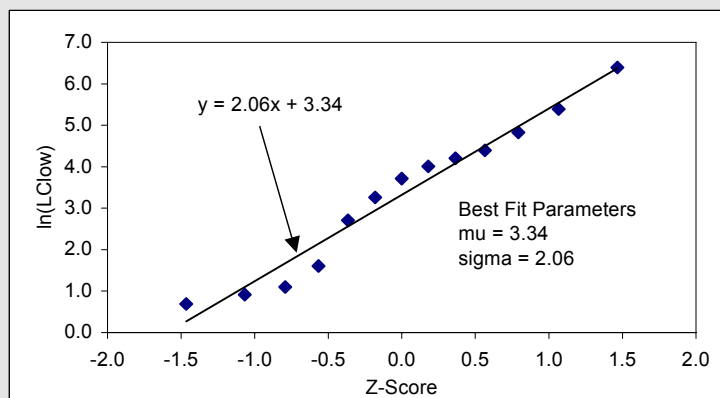
Hypothetical Data

Species	LC _{low}	ln(LC _{low})	Rank	ROS	z-score
a	2	0.693	1	0.07	-1.465
b	2.5	0.916	2	0.14	-1.068
c	3	1.099	3	0.21	-0.792
d	5	1.609	4	0.29	-0.566
e	15	2.708	5	0.36	-0.366
f	26	3.258	6	0.43	-0.180
g	41	3.714	7	0.50	0.000
h	55	4.007	8	0.57	0.180
i	67	4.205	9	0.64	0.366
j	81	4.394	10	0.71	0.566
k	125	4.828	11	0.79	0.792
l	220	5.394	12	0.86	1.068
m	600	6.397	13	0.93	1.465

Example EDF: ROS vs LC_{low} (log-scale)



Example Parametric Fit: (Lognormal)



4.6 MODELING VARIABILITY IN RISK

4.6.1 VARIABILITY IN HAZARD QUOTIENT

As noted above, the most common descriptor of risk used in predictive risk assessments is the Hazard Quotient (HQ). The HQ is the ratio of the exposure for some generalized or typical hypothetical member of the receptor population at a site, compared to an appropriate TRV value that equates to an acceptable level of risk for that receptor and chemical. Usually the HQ approach is not based on a single value, but on a range of values in which different levels of exposure (CTE and RME) are compared to both the NOAEL to LOAEL benchmarks. In general, HQ values below 1 are interpreted as indicating acceptable risk, while HQ values above 1 are interpreted as indicating the potential for adverse effects.

Because exposure varies among different members of an exposed population of receptors, HQ values also vary among members of the exposed population. Several alternative approaches for characterizing this variability by PRA methods are presented below.

Variability Within a Population

Figure 4-5 illustrates the simplest approach for summarizing variability in HQ values among the members of an exposed population. In this format, the TRV values appropriate for a particular exposure are simply superimposed on the graph illustrating the distribution of exposures. This may be done either for a dose-based (as shown in the figure) or for a concentration-based exposure parameter. This format allows an easy evaluation of the fraction of the population above ($HQ > 1$) and below ($HQ < 1$) each TRV, especially when presented in CDF format. However, this format does not allow for a quantitative estimate of the fraction of the population with HQ values above any value other than 1, although a similar calculation and presentation could be made for any multiple of the TRVs, which would directly equate to that multiple of the HQ (e.g., depicting the

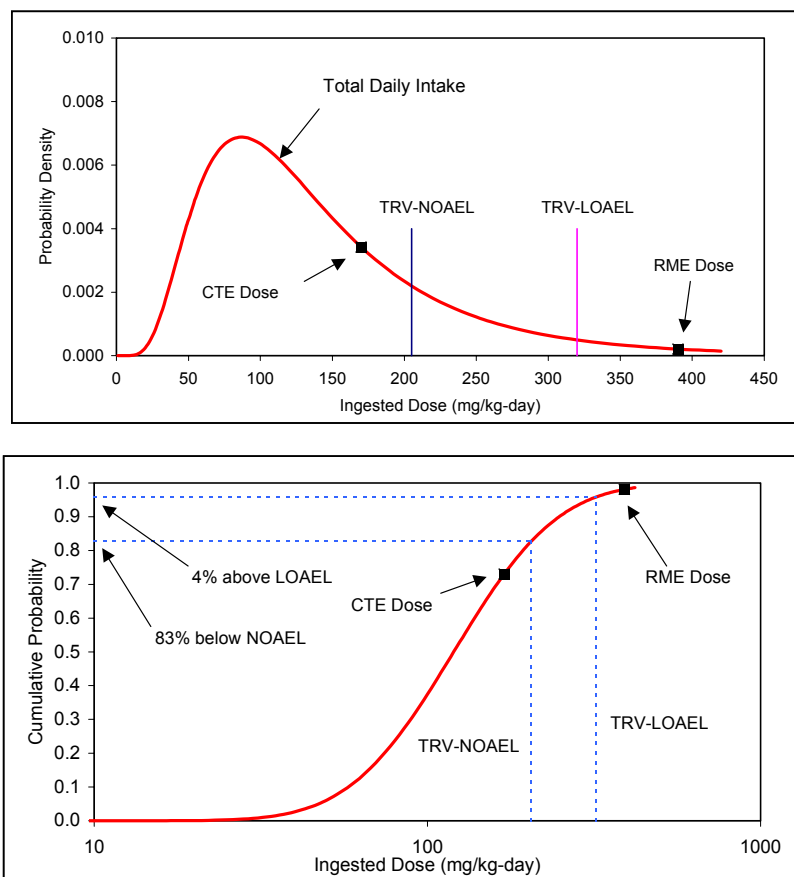


Figure 4-5. Example Comparison of Exposure Distribution to TRV.

results for a value equal to 10-times the TRV would show the fraction of the population with an HQ greater than 10).

More directly, the distribution of HQ values may be calculated by dividing each exposure value by one or all of the TRVs based on the NOAEL, LOAEL, BMDL, etc., as shown in Figure 4-6. Note that dividing a distribution by a constant does not change the shape of the distribution (only its scale), so the shape of the HQ distribution will appear identical to that of the exposure distribution. Figure 4-6 illustrates two HQ distributions; one calculated using the NOAEL-based TRV, the other using the LOAEL-based TRV. In a case such as this where there are two or more HQ distributions, a CDF format is generally easier to evaluate than a PDF format, since overlap between the curves is minimized. The CDF format allows an easy quantitative evaluation of the fraction of the population above and below any particular HQ level. For example, in the case shown in Figure 4-6, it may be seen that 83% of the population is expected to have HQ values below 1 based on the NOAEL-based TRV, while 4% are expected to have HQ values above 1 based on the LOAEL-based TRV. This type of description (percentage of the population with HQ values within a specified range) is very helpful in predicting proportions of a population exposed to specified doses of concern.

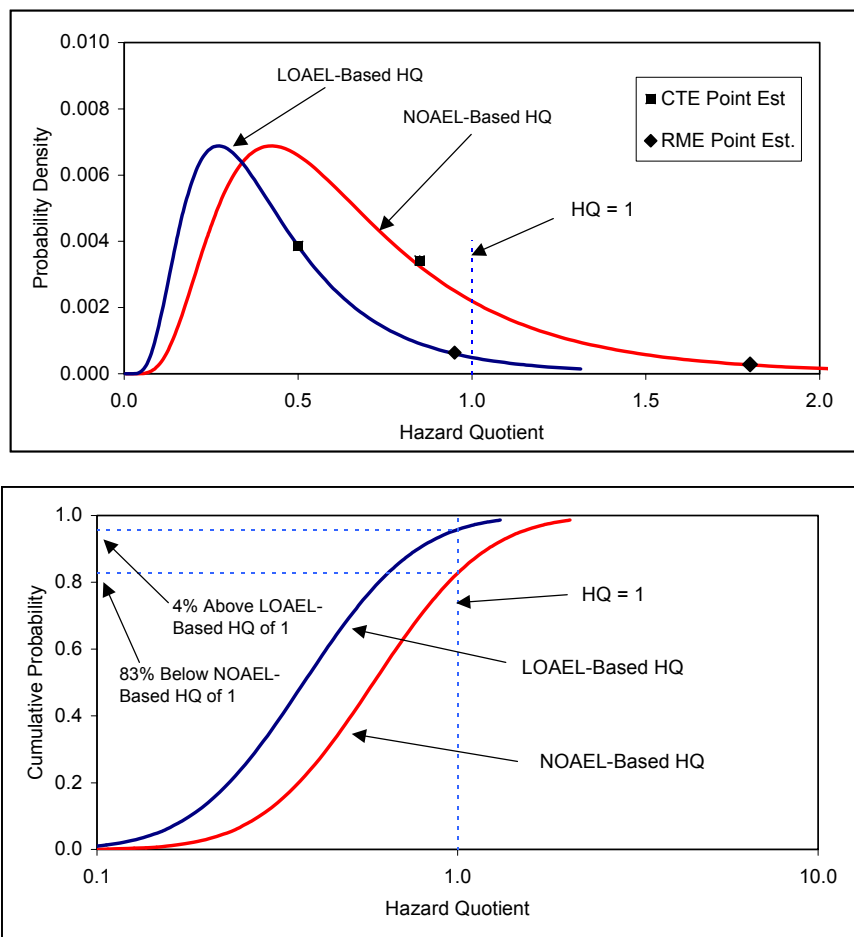


Figure 4-6. Example Distribution of HQ Values.

Variability Between Species

A similar approach may be used for characterizing the variability in risks among different species in a community. Figure 4-7 is an example that compares the distribution of concentration values in a water body (the variability might represent either time or space) to an appropriate SSD of TRVs for different species of aquatic receptors that might reside in that water body. Three different graphical formats are illustrated. In the upper panel, the PDF of concentration is compared to the CDF of the SSD. This format is easy to understand and may be interpreted visually, but is difficult to interpret quantitatively. The middle panel shows that same information, but with both distributions presented in CDF format. This allows for a quantitative evaluation of the fraction of the species that will be above their respective TRVs at any specified part of the exposure distribution. For example, using a simple graphical interpolation process (shown by the dashed lines), it may be seen that the 90th percentile of concentration (21 ug/L) will impact approximately 24% of the exposed species. The bottom panel shows the results when this same process is repeated (mathematically) for each of the concentration percentiles. As seen, hazards to the community of receptor species is quite low until concentration values reach the 80th to 85th percentile, but then rise rapidly. For example, a concentration value equal to the 95th percentile (about 28 ug/L, which will occur approximately 5% of the time) is expected to impact approximately 68% of the exposed species, and the 99th percentile (which will occur about 1% of the time) is expected to impact nearly all of the exposed species.

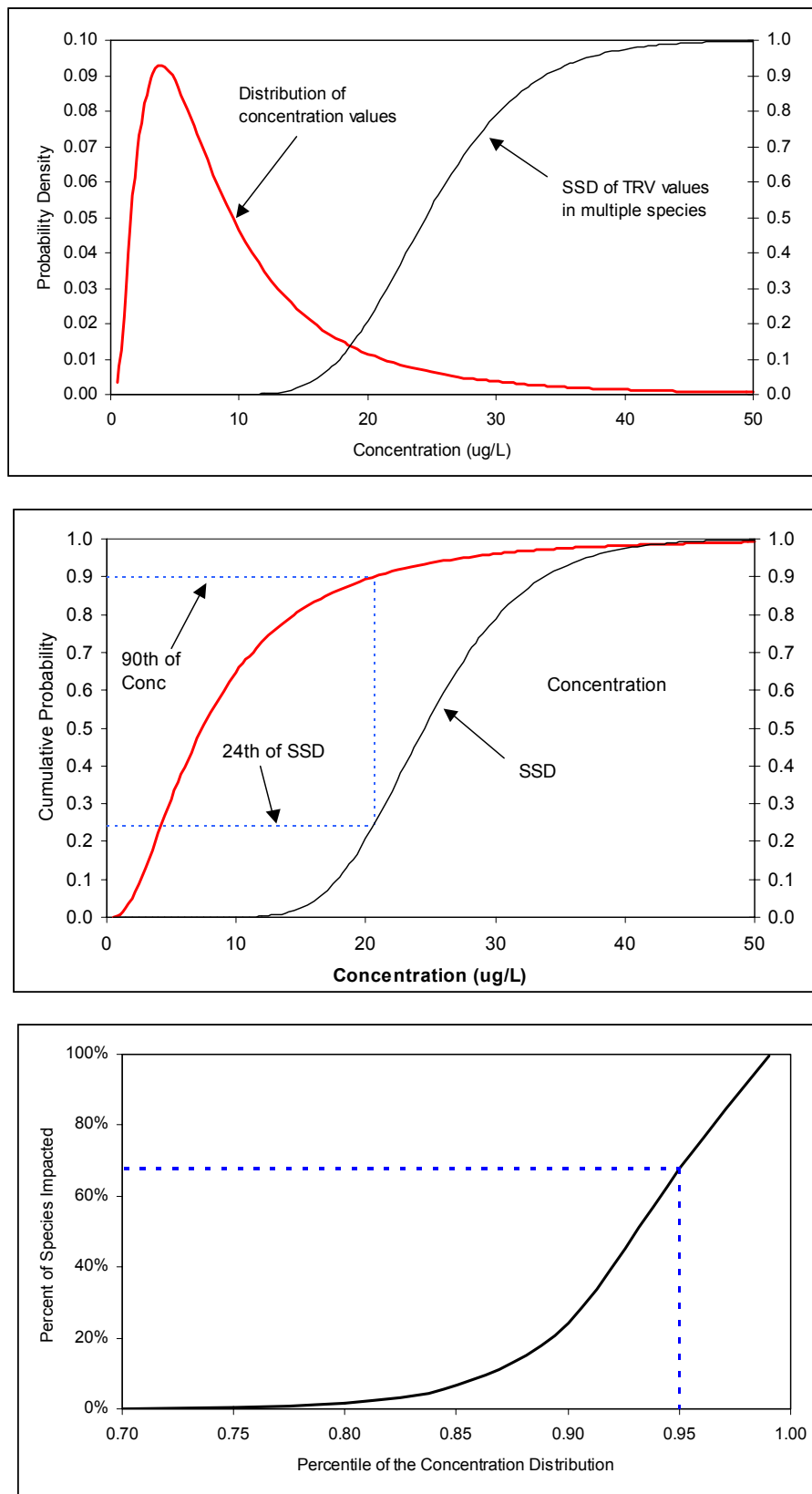


Figure 4-7. Example Presentation of Species Sensitivity Distribution.

4.6.2 VARIABILITY IN RESPONSE

As noted above, HQ and Hazard Index (HI) (where appropriate) values are a convenient way to characterize risk to ecological receptors, but interpreting the biological significance of the ranges of HQ values greater than 1 is not always easy. One of the main advantages to the PRA approach is that distributions of exposure may be combined with exposure-response distributions in order to generate distributions that characterize the frequency and magnitude (severity) of responses in an exposed population. Two examples of this approach are presented below.

Example 1: Dichotomous Response

In this hypothetical example, a toxic chemical is being transported by surface water run-off from a Superfund site into a nearby stream. Because of short-term and seasonal variability in rainfall levels (which influences both run-off rate and stream flow), the concentration of the chemical in the stream has been observed to vary as a function of time. The risk manager at the site wants to know two things: (1) How often will the concentration enter a range that can cause acute lethality in fish?; and (2) When that happens, what percent of the fish population is likely to die? Exhibit 4-6 summarizes the hypothetical concentration data and illustrates the basic approach. In this case, the concentration data are most conveniently modeled as an empirical PDF. Next, assume that the acute concentration-lethality curve is available for the chemical of interest in a relevant indicator species of fish. For convenience, assume the response function is the same as that shown in Exhibit 4-3. Then, the PDF for acute mortality may be generated by repeated sampling from the concentration distribution and calculating the probability of response (acute mortality) for each concentration value selected. Because this is a case where the entire population of fish at the exposure location may be assumed to be exposed to the same concentration in water, the probability of mortality in a single fish is equivalent to the average fraction of the population that is expected to die as a result of the exposure. The resulting PDF is shown in the graph in Exhibit 4-6. As seen, lethality is expected to be low or absent about 95% of the time, but about 5% of the time the concentration may enter a range where acute lethality may occur. The extent of mortality within the exposed population is expected to range from about 20% at the 97th percentile of exposure (i.e., this is expected to occur about 3% of the time), up to about 70% at the 99th percentile of exposure (i.e., this is expected to occur about 1% of the time).

EXHIBIT 4-6

MODELING VARIABILITY IN A DICHOTOMOUS RESPONSE

Scenario

Exposure of a population of fish to concentration values in a stream that vary over time

Hypothetical Concentration Data in Water

Value	Percentile
0.5 (1/2 DL)	0.00
1.1	0.10
2.5	0.25
5.1	0.50
9.2	0.75
15.8	0.90
24.7	0.95
52.6	0.99
83.1 (max)	1.00

Response

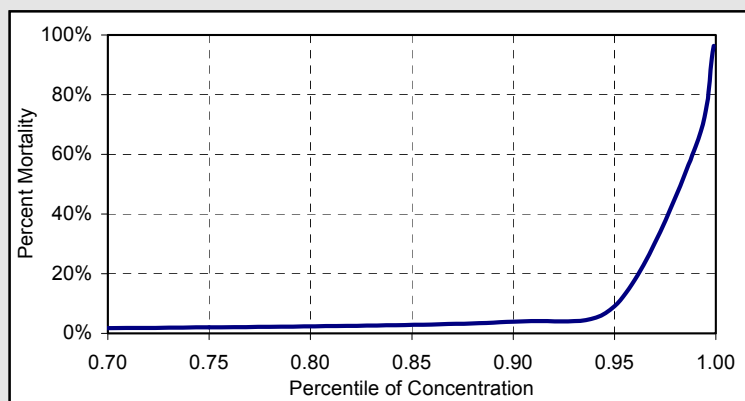
Endpoint = acute mortality
Stressor-response model fit (see Exhibit 4-2)
 $P(c) = 1/(1+\exp(4.8 - 0.1*c))$

PRA Simulation

- Step 1 Draw a concentration at random from the empiric distribution
- Step 2 Calculate the probability of mortality at that dose
Track this as the forecast cell

Example Output

Percentile	% Lethality
0.050	0.9%
0.250	1.0%
0.500	1.4%
0.750	2.0%
0.900	3.9%
0.950	9.1%
0.990	63%
0.999	96%



Example 2: Continuous Response

Exhibit 4-7 provides a hypothetical example of modeling variability in response for a continuous endpoint. In this example, assume that a toxic chemical has been released by a Superfund site and has been transported in low levels by air to a nearby meadow. Among the receptors of potential concern in the meadow are a number of different types of small mammal, and the field mouse has been selected to serve as an indicator species for this group. The goal of the PRA is to characterize the effects of the chemical on the growth of field mice in the meadow. Exposure occurs mainly by ingestion of seeds that have been contaminated by uptake of the chemical from soil, and it has been determined that the variability in average daily intake (DI) of chemical from the diet can be modeled as a lognormal distribution with mean of 104 mg/kg-day, and a standard deviation of 127 mg/kg-day. Assume for convenience that the exposure-response curve for growth inhibition in mice by the chemical is the same as that presented previously in Exhibit 4-4. Given these inputs, the expected distribution of responses is derived as follows:

- Step 1: Draw a random value for the DI of a random member of the population
- Step 2: Calculate the mean response $m(d)$ and the standard deviation of the response $s(d)$ for a group of individuals exposed at that dose (d)
- Step 3: Define the distribution of responses at that dose as $NORMAL[m(d), s(d)]$
- Step 4: Draw a response from that distribution, and track this as the output variable

An example of the output for this example is shown in the two graphs at the bottom of Exhibit 4-7. As seen, mice that are not exposed to the chemical display a range of growth rates ranging from about +10% to +40%. Many of the mice (about 90%) residing in the contaminated field are experiencing a range of growth rates that are only slightly decreased from rates expected for unexposed animals. However, about 10% of the animals have weight gains that are markedly less than for unexposed animals, ranging from about +5% to -30% (i.e., a net weight loss of 30% compared to the starting weight).

It should be noted that the response distribution calculated in this way is what would be expected for a large population of exposed receptors. If the actual exposed population is small, then the actual response distribution may vary somewhat compared to the typical response shown in Exhibit 4-7. In cases where it is important to evaluate this variability about the expected average pattern of response, this may be done by running repeated Monte Carlo simulations using a number of trials (iterations) within each simulation that is equal to the expected size of the exposed population. Each simulation will thus represent a possible response distribution in the exposed population, and the range of responses across different populations may be evaluated by comparing the multiple simulations. As noted above, the magnitude of the variability between populations is expected to be small if the population size (number of trials) is large, although this depends on the characteristics of the exposure and response functions.

EXHIBIT 4-7

MODELING VARIABILITY IN A CONTINUOUS RESPONSE

Scenario

Exposure of a population of field mice to a chemical ingested via the food chain

Example Inputs

Exposure

Distribution of Average DI LN(104,127)

Response (see Exhibit 4-3)

Endpoint = Growth (% increase in 21 days)

Stressor-response model fit

$$\text{Mean response(dose)} = 23.7 - 51.4 \cdot \text{dose}^n / (228.7^{5.29} + \text{dose}^{5.29})$$

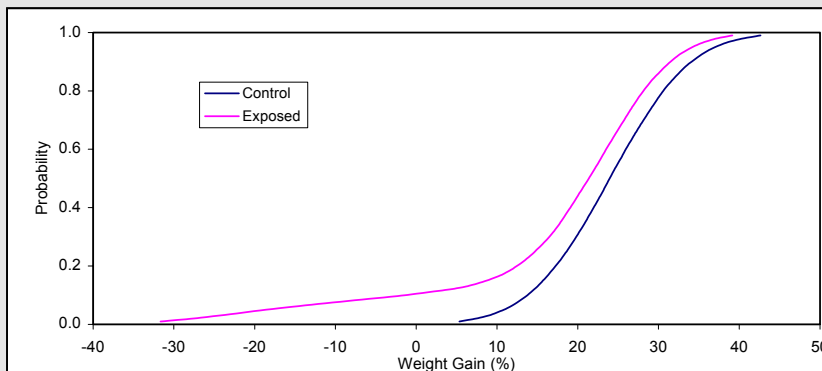
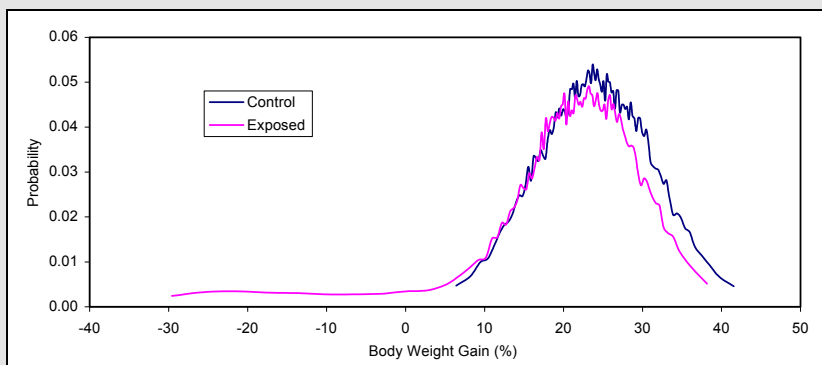
Stdev (dose) = 7.0 (constant)

PRA Simulation

- Step 1 Draw a dose at random from the lognormal distribution of dose
- Step 2 Calculate the mean response [m(d)] and standard deviation of the response (s(d) at that dose
- Step 3 Define the PDF for response at dose d: NORMAL(m(d), s(d))
- Step 4 Draw a response at random from this PDF
Track this as the forecast cell

Example Output

Percentile	Control	Exposed
0.05	10.9	-18.6
0.25	18.6	14.7
0.50	24.0	21.4
0.75	29.3	26.9
0.90	34.1	31.5
0.95	37.0	34.2
0.99	42.6	39.1



4.6.3 JOINT PROBABILITY CURVES

In this approach, if data are available to characterize the probability of a particular exposure occurring, and an exposure-response curve is available, these may be combined to yield a curve (referred to as a Joint Probability Curve) that shows the probability that a response greater than some specified magnitude will occur. An example is shown in Figure 4-8. The upper panel shows a hypothetical cumulative exposure probability distribution (plotted on the primary y-axis) along with the exposure-response curve (plotted on the secondary y-axis). The steps needed to generate the Joint Probability Curve are as follows:

Step 1: Select an exposure level " x " and record the probability (P_x) of exceeding that exposure. For example, in Figure 4-8, at an exposure of 12 units, the cumulative probability of exposure is 84%. Thus, the probability of exceeding that exposure is 16%.

Step 2: Find the expected response at that same exposure (R_x). In this case, the response at an exposure of 12 is 2.2.

Step 3: Plot a data point at R_x on the x-axis and P_x on the y-axis.

Step 4: Repeat this process for many different exposure levels, being sure to draw samples that adequately cover all parts of the probability scale.

The lower panel of Figure 4-8 shows the results obtained using the hypothetical data in the upper panel. The advantage of this format is that it gives a clear visual display of both the probability and magnitude (severity, extent) of response. Further, the area to the left of the curve is a relative index of the population-level or community-level risk, and comparison of this area across different risk scenarios is helpful in comparing different risk scenarios (both in risk characterization and risk management). However, this approach is based on the mean response at a dose, and does not account for variability in response between multiple individuals all exposed at that dose. Employing a two-dimensional Monte Carlo analysis (2-D MCA) procedure could help to display this variability in response between the individuals at a given dose.

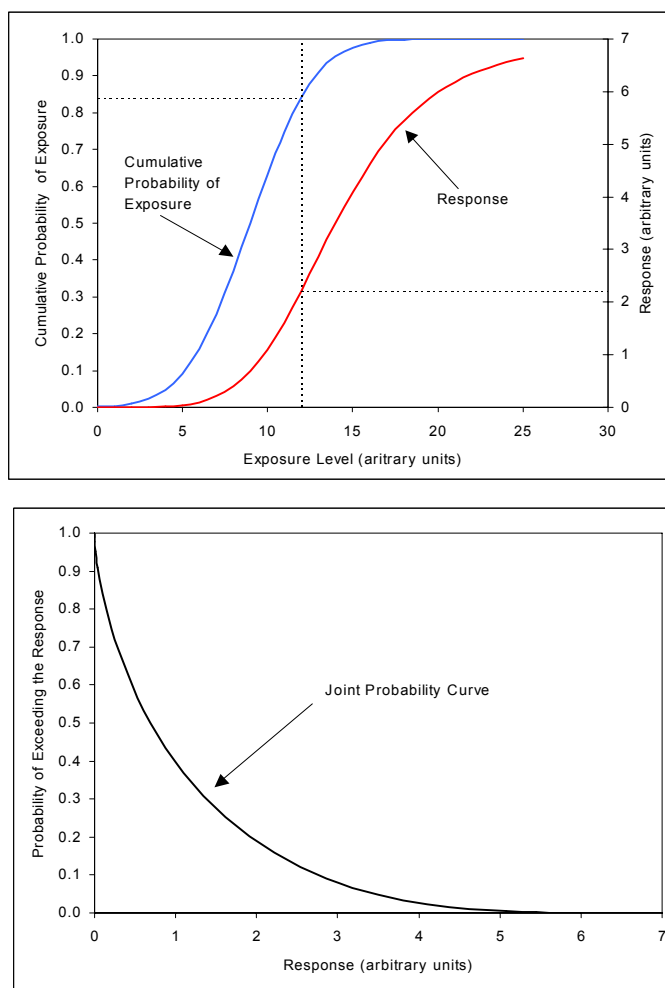


Figure 4-8. Example Joint Probability Curve.

Note that unless 2-D MCA is used, this approach does not require Monte Carlo modeling. Rather, the calculations can usually be performed in a spreadsheet format using built-in spreadsheet functions.

4.7 MODELING UNCERTAINTY IN ECOLOGICAL RISK ASSESSMENTS

As emphasized above, one of the greatest potential benefits of the PRA approach is the ability to combine estimates of uncertainty associated with different components of the exposure and risk models in order to describe the overall uncertainty in final exposure or risk estimates. Some basic options for characterizing and presenting uncertainty in exposure, toxicity, HQ, and response are presented below.

4.7.1 UNCERTAINTY IN EXPOSURE

Most estimates of dose-based exposure for terrestrial receptors (birds, mammals) are based on calculated estimates of chemical intake using simple or complex food web models, sometimes coupled with environmental fate and transport models that can link risk to a receptor with a source of contamination. In cases where receptors are exposed mainly by direct contact rather than ingestion (e.g., fish, soil invertebrates, etc.), concentration-based (as opposed to dose-based) descriptors of exposures may be derived using mathematical fate and transport models. The basic principles for modeling uncertainty in ecological exposure models (either dose-based or concentration-based) are the same as discussed in Appendix D. In brief, probability distribution functions of uncertainty (PDFu's) are used to characterize the uncertainty in the parameters of the probability distribution functions of variability (PDFv's) for some or all variables in the exposure model. Then, a 2-D MCA is used to derive quantitative estimates of the uncertainty around each percentile of the variability distribution of exposure. Figure 4-9 illustrates the type of tabular and graphic outputs that this approach generates.

Variability Percentile	Uncertainty Percentiles		
	5th	Mean	95th
0.05	0.4	1.1	2.0
0.10	0.7	1.6	2.8
0.15	0.9	2.1	3.5
0.20	1.2	2.6	4.2
0.25	1.5	3.1	5.0
0.30	1.8	3.7	5.9
0.35	2.1	4.3	6.7
0.40	2.6	5.0	7.6
0.45	3.0	5.8	8.7
0.50	3.6	6.6	9.9
0.55	4.2	7.7	11.3
0.60	5.0	8.8	12.9
0.65	5.9	10.3	14.8
0.70	7.2	12.1	17.2
0.75	8.8	14.4	20.3
0.80	10.9	17.5	24.1
0.85	14.5	22.0	30.1
0.90	20.1	29.6	39.4
0.95	32.9	46.5	60.0

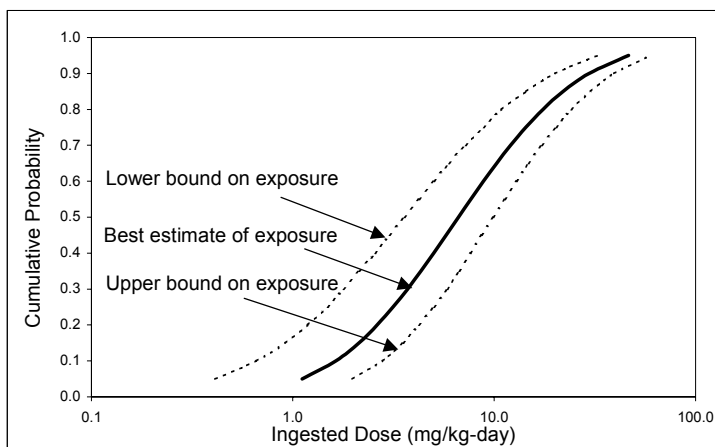


Figure 4-9. Example Presentation of Uncertainty in Exposure.

If exposure is based on measured rather than calculated values by PRA (e.g., measured concentrations in an abiotic medium,

measured concentrations in receptor tissues), uncertainty in the empirical or best-fit continuous distribution through the data can be quantified using the statistical methods detailed in Appendix B.

As discussed in Chapter 1, it is important to understand that there are many sources of uncertainty and that this approach to uncertainty analysis focuses mainly on parameter uncertainty and uncertainty in the true shape of input variable distributions. It does not capture other sources of uncertainty relating to the fundamental adequacy of the exposure and risk models used to describe the behavior of complex biological systems or of sampling and analytical errors and uncertainties, so the uncertainty estimates should always be interpreted in this light as being somewhat incomplete.

4.7.2 UNCERTAINTY IN TOXICITY

Toxicity information used for ERAs is often a source of uncertainty in the risk assessment process. This uncertainty may arise from multiple areas and may include both quantitative uncertainty in the dose-response data (involving toxicokinetics and study designs) and qualitative uncertainty in the relevance of the data (involving toxicodynamics). Methods for characterizing the quantitative uncertainty in both point estimates of toxicity (TRVs) and in full exposure-response curves are outlined below.

Uncertainty in TRVs

TRVs for a chemical are point estimates of exposure levels that do not cause an unacceptable effect in an exposed receptor population. Ideally, all TRVs would be based on NOAEL and LOAEL values derived from studies in which the receptor, endpoint, exposure route and duration were all matched to the assessment endpoints defined for the site. However, such exact matches are seldom available. Therefore, it is often necessary to extrapolate available toxicity data across route, duration, endpoint and/or species, leading to uncertainty in the most appropriate value to use as the NOAEL or LOAEL. There are no default methods for developing TRVs on a site. However, some options include the use of allometric dose scaling models, physiologically-based biokinetic models, benchmark dose estimates or other approaches based mainly on policy and/or professional judgment. Guidelines for dealing with the uncertainty in components of the TRV derivation by uses of PRA are provided below.

Uncertainty in NOAELs and LOAELs

Uncertainty in the NOAEL or LOAEL for a chemical has two components: (1) uncertainty within a study; and (2) uncertainty between studies, under exact specified conditions of exposure.

Assuming that a single study has been selected to provide the NOAEL and/or LOAEL values to be used in deriving a TRV for a chemical, it is customary to define the NOAEL as the highest exposure that did not cause a statistically significant effect, and the LOAEL is the lowest exposure that did cause a statistically significant effect. As noted earlier (see Section 4.5.1), this approach has a number of limitations, and there may be substantial uncertainty as to whether the observed NOAEL and LOAEL values actually bracket the true threshold effect level. One way to quantify uncertainty in the exposure levels that cause some specified level of adverse effect is through the use of exposure-response curve-fitting software such as EPA's BMDS package. In this approach, the risk assessor selects some level of effect that is judged to be below a level of concern, and another level of effect that would be of concern. The choice of these response levels is a matter of judgment, and depends on the nature and severity of the endpoint being evaluated. A specified level of effect is referred to as a Benchmark Response (BMR), and the exposure that causes that response is referred to as the Benchmark Dose (BMD). Given information on the number of test organisms in each test group and on the variability of the response in those

organisms, the BMD software uses maximum likelihood methods to derive the 5% lower confidence bound on the exposure that causes the BMR. This is referred to as the BMDL. This uncertainty bound may be used to quantify the uncertainty in the BMD, and hence to characterize this source of uncertainty in the TRV. The simplest method for approximating the uncertainty distribution around the BMD is to assume the distribution is approximately normal, with mean equal to the BMD and standard deviation (standard error) given by:

$$\text{Stdev} = (\text{BMD} - \text{BMDL}) / 1.645$$

For advanced analyses, a more accurate characterization of the uncertainty distribution around the BMD may be derived by Monte Carlo simulation. In this approach, each model parameter is assumed to be normally distributed, with mean and standard error values provided by the BMDS output. Monte Carlo simulation is then used to select alternative model parameter sets, being sure to account for the covariance between parameters (the covariance matrix is also provided by the BMDS output). For each parameter data set, the BMD is calculated, and the distribution of BMD values across many iterations is a better approximation of the uncertainty in the BMD.

Uncertainty in the effect level (NOAEL or LOAEL) for a chemical may also arise because there is more than one study available for the chemical, and the studies do not yield equal estimates of the effect level. It is important to note that the process of reviewing available toxicity studies, choosing the most relevant endpoint for use in deriving a TRV, and identifying the most relevant study is a process requiring basic toxicological expertise (not probability or statistics), and this process must be completed both for point estimate and probabilistic risk assessments. In general, studies based on different receptors, endpoints, exposure routes and/or durations are not equally relevant for evaluating a particular assessment endpoint in a particular indicator species. However, in some cases, multiple studies of the same endpoint in the same species will be available. In such a case, assuming that all the studies are judged to be equally reliable, the best estimate of the LC50 may be derived by calculating the geometric mean of the available alternative values (after adjustment to constant hardness). Uncertainty around the best estimate may then be based on the observed inter-study variability, using the basic principles for choosing PDFu's as described in Appendix B.

Uncertainty in Extrapolation of TRVs

In general, extrapolation of TRVs across species or endpoints is not desirable, since the magnitude and direction of any potential error is generally not known. Sometimes, extrapolations between species are attempted based on allometric scaling models that seek to adjust toxicity values accounting for differences in body weight. Alternatively, physiologically-based pharmacokinetic (PBPK) models that seek to account for differences in a number of other physiological variables (metabolism rate, organ size, blood flow, etc.) can be used. However, the validity of these models is often not well established. In those cases where these models are used, and where the uncertainty in the model is judged to warrant quantitative evaluation, the primary source of the model should be consulted in order to derive an estimate of the uncertainty in the quality of the extrapolation and in the parameters of the model. As noted earlier, PRA may capture uncertainty associated with model input parameters, but does not usually capture all sources of uncertainty in the model. In particular, most models of this sort are designed to extrapolate only the average response as a function of dose, and are not intended to extrapolate variability between individuals at a specified dose. When no mathematical model is available to support quantitative extrapolation across species, exposure duration or endpoint, professional judgment and/or policy may be used to select extrapolation factors to account for the uncertainty.

The risk assessor should ensure that the risk manager understands the uncertainty associated with any model selected and applied, and that the results of the calculations (point estimate or PRA) are conditional upon the model selected.

Uncertainty in Parameters of the Dose-Response Models

When toxicological exposure-response data are fit to mathematical equations, the fitting software will usually provide quantitative information on the uncertainty in the best estimates for each of the model parameters. For example, in the dichotomous model illustrated in Exhibit 4-3, the output from the BMDS software included the following information on the uncertainty in the parameters of the best-fit logistic equation:

Parameter	Best Est	Std Error (SE)
a	-4.80	0.83
b	0.101	0.019

Because the uncertainty in the best estimate of each model parameter is asymptotically normally, uncertainty in the parameters may be modeled as:

$$\text{PDFu (parameter i)} = \text{NORMAL}(\text{best estimate of parameter i, SE of parameter i})$$

Note that the parameters of the model are generally not independent, and generally should not be treated as such. Thus, when modeling the uncertainty in the parameters of the best-fit exposure-response model, the PDFv's for the parameters should be correlated according to the correlation matrix or the variance-covariance matrix, as provided by the modeling software.

4.7.4 UNCERTAINTY IN RESPONSE

If the risk characterization phase of the risk assessment focuses on an estimation of the distribution of responses rather than the distribution of HQ values, the uncertainty in the distribution of responses can be evaluated using two-dimensional Monte Carlo techniques using PDFu's for the parameters of the exposure and exposure-response models derived as described above. The same graphical output may be used for this presentation as was illustrated in Figure 4-9, except that the x-axis is response rather than HQ. This format is illustrated in Figure 4-10 for a dichotomous endpoint (e.g., acute lethality). In this example, the average probability of response among the members of the exposed population (shown in the graph by the black diamond symbols) is 8.2%, with a confidence bound around the mean of 4.9 to 12.8%. This is equivalent to concluding that about 8.2% of the population is expected to suffer acute lethality, but the true fraction dying could range from 4.9 to 12.8%.

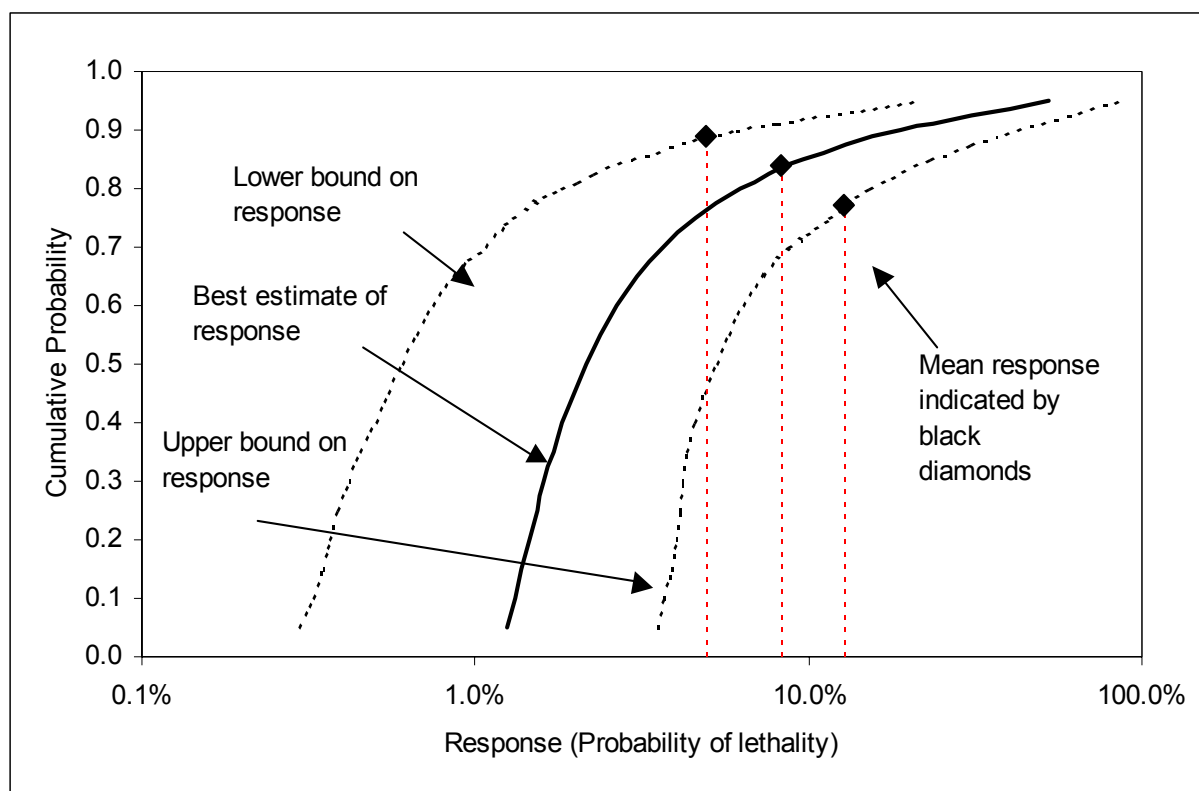


Figure 4-10. Example Presentation of Uncertainty in Response.

4.7.3 UNCERTAINTY IN HAZARD QUOTIENT

Once the uncertainty in exposure and/or toxicity distributions has been characterized as described above, there are a number of options for presenting the resultant uncertainty in the HQ (or HI, if appropriate and applicable for summing HQs) distributions. Figure 4-11 shows one simple graphical format, where the point estimate of the TRV is superimposed on the uncertainty bounds of the exposure distribution (upper panel), or the uncertainty bounds of the TRV are superimposed on the best estimate of exposure (lower panel). One could also superimpose the range of TRVs over the range of exposures, to capture most of the uncertainty in the HQ. Furthermore, such distributional outputs should always show the point estimate ranges of CTE and RME exposures in respect to the ranges of TRVs, for use in weight-of-evidence to help interpret the PRA and point estimate results. The advantage of this format is that no additional Monte Carlo modeling is needed to derive initial descriptors of uncertainty in risk. For example, in the upper panel it may be seen that the best estimate of the fraction of the population exposed at a level below the TRV is about 83%, but that this is uncertain due to uncertainty in the exposure estimates, and the true percent below the TRV might range from 74 to 90%. Similarly, in the bottom panel, the best estimate of the fraction of the population below the TRV is also about 83%, but due to uncertainty in the TRV the actual value could range from 64 to 91%. Uncertainty could also be presented by showing a combined graph with both ranges of exposure and TRVs, such as described below.

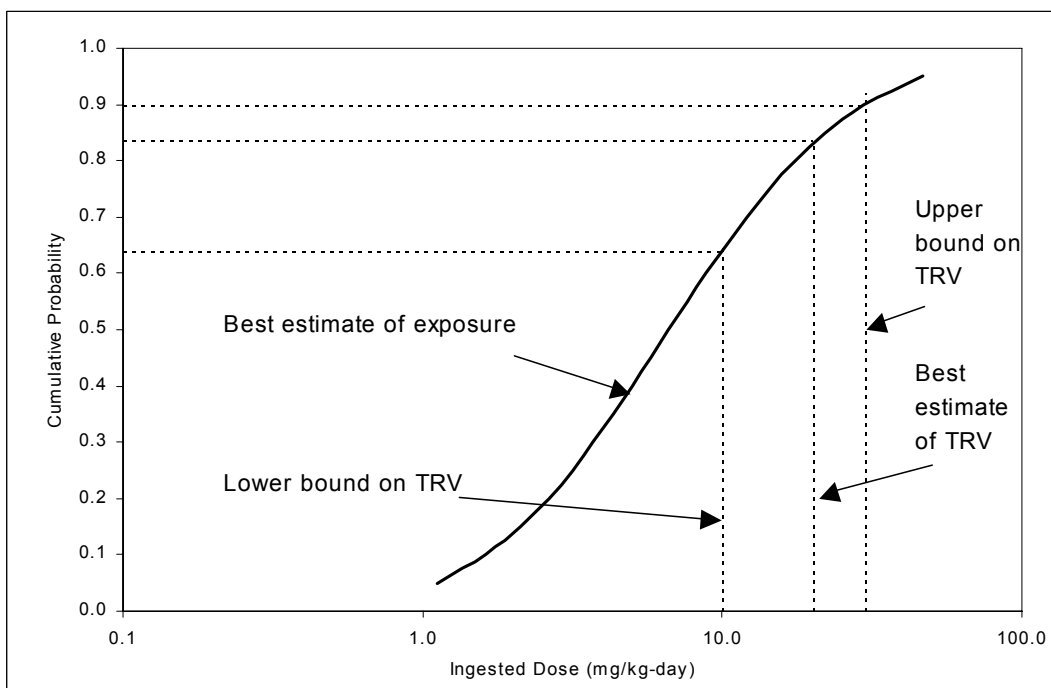
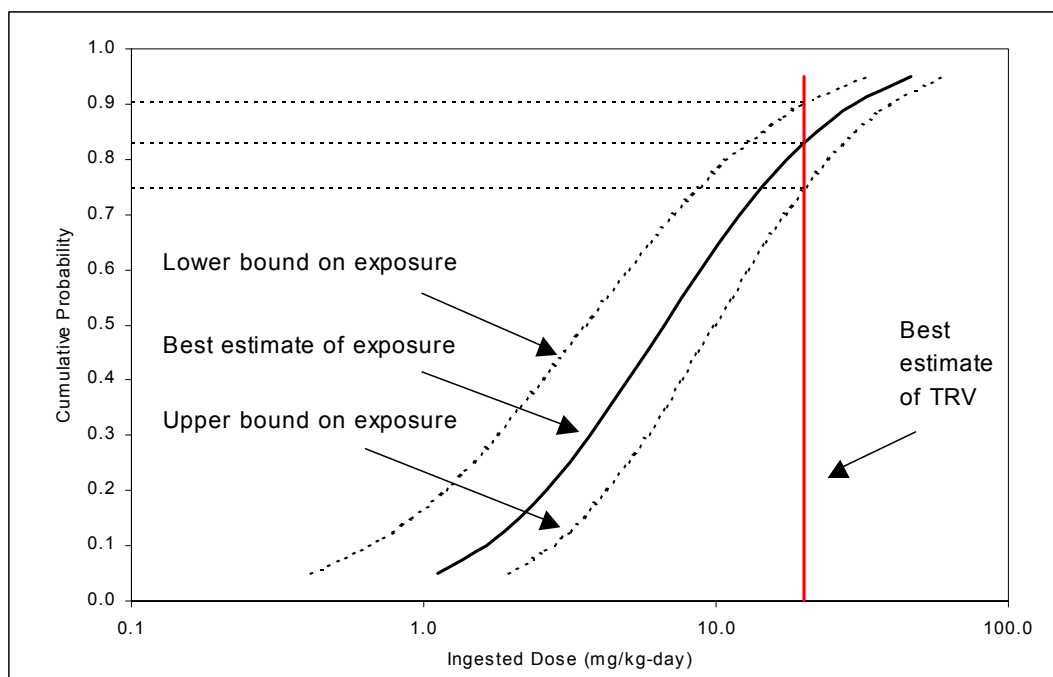


Figure 4-11. Example Presentation of Uncertainty in Exposure and TRV.

A more complete characterization of uncertainty in HQ may be achieved by using PRA to combine the uncertainty in both the exposure and the TRV terms, resulting in the uncertainty bounds on the HQ distribution itself (see Figure 4-12). In this example, it may be seen that 63% of the exposed population is estimated to have an HQ below 1.0, but that this is uncertain due to uncertainty in both the exposure distribution and the TRV, and that the true fraction of the population below a level of concern ($HQ < 1$) could range from 45 to 81%.

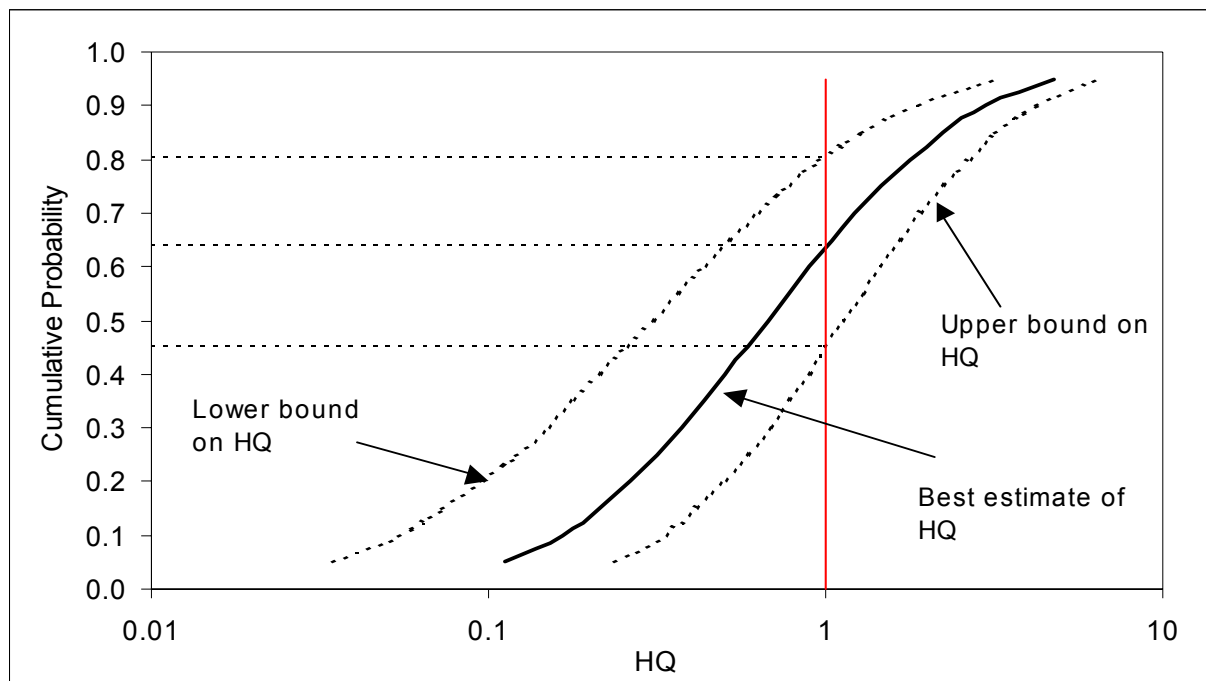


Figure 4-12. Example Presentation of Uncertainty in HQ Estimates.

4.8 INTERPRETING RESULTS OF AN ECOLOGICAL PRA

In some cases, the information contributed by a PRA may provide a more complete characterization of risks to a population of receptors than can be obtained by using point estimate methods. However, whether by PRA or by point estimate or a combination, the results of the risk assessment must be interpreted to reach a risk management decision.

In contrast to the case for human health risk assessments (where default risk-based decision rules are well established), there are no established default decision rules for identifying when risks to ecological receptors are and are not of concern. In the point estimate approach, EPA guidance (U.S. EPA 1992b, 1995) recommends consideration of both the RME and CTE exposure/dose estimates along with TRVs based on both LOAELs and NOAELs (U.S. EPA 1997a) to reach a risk management decision. The same principle applies to probabilistic ERAs.

In some cases, interpretation of an ecological PRA is relatively simple. For example, if the distribution of HQ values calculated using an appropriate NOAEL-based TRV are less than 1.0 for nearly all members of the population, then it is likely that risks are within an acceptable range for the population. Conversely, if the distribution of HQ values calculated using a LOAEL-based TRV are significantly greater than 1.0 for most members of an exposed population, then it is likely that risks are not acceptable

for the population. However, for cases which fall between these bounding conditions (and for cases where one needs to clearly define the boundaries of potential excess risks for a gradient of contamination and exposures), the level of risk or response that is considered acceptable must be defined by the risk assessor and the risk manager on a site-specific and receptor-specific basis. This evaluation should take the following factors into account:

(1) *The Risk Management Goal*

The risk management objective for most Superfund ERAs is defined as population sustainability (U.S. EPA, 1999). In this case, harm to some members of the exposed population may be acceptable, if that harm does not lead to an overall reduction in population viability. This situation (protection of a population rather than protection of individuals) is sometimes equated with use of the CTE (average) receptor as the basis for risk management decision making. That is, if the HQ for the CTE receptor is below a level of concern, it is sometimes assumed that population risks are acceptable.

However, the choice of the CTE receptor as the basis for risk management decision making may not be sufficiently protective in all cases. For the vast majority of wild populations, the proportion of the population that must be protected to ensure population stability will be unknown. At a small number of sites, a population biologist may be able to provide some information. Moreover, the percentile of the CTE receptor in the exposure or risk distribution may vary depending on the shape of the distribution. The proportion of the population experiencing exposure greater than that of the CTE receptor could range from less than 10% up to 50% or even higher. Also, the ecological significance of an adverse effect on some members of a population depends on the nature of the stressors and on the life history and population biology of the receptor species. Because of these complexities, use of the CTE as a decision threshold for nonthreatened or endangered species may be appropriate in a small number of cases, but risk assessors and risk managers should realize that the choice of the CTE receptor requires a species- and endpoint-specific justification and the CTE should not be used as the default basis for a risk management decision. Rather, for the majority of ERAs, the risk management decision should be based on the RME receptor or an upper percentile of the distribution of variability in risk/exposure.

(2) *The Toxicological Basis of the TRV*

The biological significance of a distribution of variability in HQ cannot be interpreted without a proper understanding of the nature of the TRV being used to evaluate the distribution. This includes the nature of the toxicological endpoint underlying the TRV, its relevance to the assessment endpoint, and the shape (steepness) of the dose-response curve. For example, an HQ of 2 based on an EC20 for reduction in reproductive success would likely be interpreted as more significant toxicologically than an HQ of 2 based on the EC20 for an increase in liver weight. Likewise, an HQ of 2 based on an LC_{low} for acute lethality would be more significant if the dose-response curve for lethality were steep than if it were shallow, since it would be easier to cause greater response with smaller increases in exposure to contaminants.

(3) *The Characteristics of the Receptor*

Ultimately, the question which must be assessed is whether an effect of degree "x" occurring in "y" percent of the population is biologically and ecologically significant. This, in turn depends on the attributes of the receptor being evaluated. For example, a reduction of 10% in the reproductive success of a fecund and common species (e.g., the field mouse) might not lead to a significant reduction in population number, while the same effect could be of concern in a species with lower fecundity and/or lower population density (e.g., the moose). Thus, the interpretation of an analysis of variability in exposure and/or effect often requires the input of a trained population biologist with expertise in the receptor of concern.

Because of these issues, there is no default rule for what level of effect is and is not acceptable for an exposed ecological population; except for the case of no potential excess risks where the RME exposures do not exceed the TRV based on a NOAEL, assuming there is reasonable confidence in those exposure and toxicity values. In some cases, mathematical models may be available for predicting the population-level consequences of a given pattern of effects (e.g., see ECOFRAM 1999a for some aquatic population models), but in general the extrapolation from a distribution of individual responses to an estimation of population-level effects is difficult. For this reason, close consultation between the risk manager and the ecological risk assessor is necessary for translating results of an ERA into an appropriate and successful risk management decision.

4.9 GUIDELINES FOR PLANNING AND PERFORMING A PROBABILISTIC ERA

4.9.1 PLANNING AN ECOLOGICAL PRA

Chapter 2 provides a general discussion of the key steps that should be followed when planning a PRA. These guidelines are equally applicable to ecological PRA as to human health PRA. Of the key steps in the process, most important are the following:

Dialogue Among Stakeholders

As discussed in Section 4.2, the decision if and when to perform an ecological PRA is an SMDP shared by risk assessors, risk managers, and stakeholders, including members of the public, representatives from state or county environmental agencies, tribal government representatives, natural resource trustees, private contractors, and potentially responsible parties (PRPs) and their representatives. A scoping meeting should be held after the completion of the baseline risk assessment in order to discuss the potential purpose and objectives of a PRA, and to identify the potential value of the analysis to the risk management process. If it is decided to perform at least an initial PRA evaluation, subsequent meetings of a similar type should occur iteratively in order to assess whether any further effort is warranted.

Preparation of a Workplan

Any PRA beyond the simplest screening level evaluation should always be accompanied by a workplan. The purpose of the workplan is to ensure that all parties agree on the purpose and scope of the effort, and on the specific methods, data, and procedures that will be used in the PRA. Workplans should be developed according to available guidance for workplans for nonprobabilistic ERA (U.S. EPA, 1992b,

1997a) and should consider three elements: (1) the 16 guiding principles of MCA (U.S. EPA, 1997b); (2) the eight guidelines for PRA report submission (U.S. EPA, 1997b); and (3) the tiered approach to ERA (U.S. EPA, 1997a). Development of a workplan for PRA is discussed in greater detail in Chapter 2, and Exhibit 4-8 summarizes the key elements of a proper workplan. The workplan must be submitted to the BTAG coordinator and/or regional ecotoxicologist for review and for approval by the risk manager. The EPA strongly recommends that PRPs who wish to perform PRAs of ecological risk involve the Agency in the development of a workplan in order to minimize chances of significant disagreement, as is required by EPA policy.

EXHIBIT 4-8

EXAMPLE ELEMENTS OF A WORKPLAN FOR ECOLOGICAL PRA

1. Introduction/Overview
 - Conceptual site model
 - Assessment endpoints
 - Indicator species
 - Measures of exposure and effect
2. Description of Exposure and Risk Models
 - Basic exposure models (fate and transport, uptake, food web, intake, etc.)
 - Basic risk models (HQ, dichotomous response, continuous response)
3. Results from a Point Estimate Assessment
 - CTE and RME risk estimates from baseline evaluation
4. Rationale why a PRA will be helpful
 - Goals of the assessment (variability, uncertainty, both)
 - Expected benefit to risk manager
5. Description of the Proposed PRA
 - Exposure scenarios to be evaluated
 - Output variables to be modeled in variability and/or uncertainty space
6. Proposed PDFs, and their basis
 - Method for performing sensitivity analysis and for selecting key variables
 - Data source for characterizing key variables
 - Approach for selecting and parameterizing key variables
 - Proposed list of PDFs for exposure variables (optional but desirable)
 - Method for dealing with the concentration term
 - Method for dealing with correlations
7. Proposed Software and Simulation Approach
 - Commercial or custom
 - Monte Carlo or Latin Hypercube
 - Number of Iterations
 - Method(s) for sensitivity analysis
8. Preliminary Results (optional, but helpful)
 - Results of a screening level evaluation
 - Identification of variables where more effort is needed to improve the distribution function

4.9.2 EVALUATING AN ECOLOGICAL PRA

When an ecological PRA is submitted to EPA for consideration, it will be reviewed in order to determine if it has been performed in accord with sound principles of ERA (U.S. EPA, 1997a, 1998), and with sound principles of PRA (U.S. EPA, 1997b). A general checklist that may be helpful to reviewers is provided in Appendix F, and key features of this checklist are summarized in Exhibit 4-9. Eight specific conditions for acceptance of a PRA submitted to EPA are provided in U.S. EPA (1997b).

At the discretion of EPA risk assessor or risk manager, the PRA report may be submitted for additional EPA internal review and/or an external review process in accord with Agency guidelines for conducting peer reviews (U.S. EPA, 2001). The external peer review may be used in cases where the issues are complex or contentious and the opinions of outside expert peer reviewers can improve the PRA.

EXHIBIT 4-9

CHECKLIST FOR INCLUDING A PRA AS PART OF THE ERA (SEE APPENDIX F)

- All risk assessments should include point estimates prepared according to current Superfund national and regional guidance.
- A workplan must be submitted for review and approval by the appropriate EPA regional project manager (RPM) and/or BTAG coordinator prior to submission of the PRA.
- A tiered approach should be used to determine the level of complexity appropriate for the ERA. The decision to ascend to a higher level of complexity should be made with the risk manager, regional risk assessor and other stakeholders.
- The eight conditions for acceptance presented in the EPA policy on PRA (U.S. EPA, 1997b) should be clearly addressed by each PRA submitted to the Agency.
- Information in the PRA should possess sufficient detail that a reviewer can recreate both the input distributions and all facets of the analysis. This includes copies of published papers, electronic versions of necessary data and other materials deemed appropriate by EPA.

4.10 EXAMPLE OF THE TIERED PROCESS IN ERA

As discussed in detail in Chapter 2, one of the key elements in the risk assessment process is deciding if and when further analysis is warranted. This includes decisions regarding whether to employ PRA calculations to supplement point estimate calculation, and if so, what level of effort to invest in those PRA calculations. The following section presents a relatively simple hypothetical example illustrating how the tiered approach might operate at a site where ecological risk is an important concern.

Problem Formulation

PestCorp is a former chemical manufacturing facility that produced mainly chlorinated pesticides 10 to 20 years ago. Data collected on the PestCorp property indicate that a number of spills or releases of chlorinated pesticides took place when the facility was in operation, and that site soils are broadly contaminated, especially with pesticide X. This contaminated soil has led to impacts on a nearby lake of about 300 acres that receives surface water runoff from the PestCorp site. Samples from the lake reveal low but detectable levels of pesticide X in water, with relatively high values in sediment and in the tissues of a variety of aquatic organisms (crayfish, snails, benthic macroinvertebrates and fish). The concentration values in all media (water, sediment, aquatic organisms) tend to be highest in the part of the lake receiving runoff from the PestCorp property, with a gradient of diminishing values at locations further away from the area where runoff enters the lake.

A BTAG committee formed by EPA to identify potential ecological concerns at the site recognized that many different species could be exposed to the contaminants in the lake, including aquatic receptors residing in the lake (fish, invertebrates, aquatic plants), as well as mammals and birds that frequent the lake for food or water. Because pesticide X is lipophilic and tends to biomagnify in the food web, the BTAG decided that the highest risks would likely occur in higher-level predators such as mammalian omnivores, and selected the racoon as a good indicator species to represent this trophic group. Pathways of exposure that were identified as warranting quantitative evaluation included (a) ingestion of water, (b) ingestion of aquatic food items, and (c) incidental ingestion of sediment while feeding or drinking at the lake. The BTAG determined that the assessment endpoint was protection of mammalian omnivore populations.

Point Estimate Risk Evaluation

A series of iterative screening-level point estimate calculations (Steps 1 to 2 of the 8-step ERAGS process) were performed to investigate whether or not there was a basis for concern at the site. Initial calculations using simplified and conservative inputs (i.e., exposure based on the maximum measured concentration in each medium, an area use factor of 1, and the most conservative available TRVs) indicated that the HQ value for pesticide X could be quite large. Therefore, a refined screening level evaluation was performed in which point estimates of CTE and RME risk were derived using the best information currently available. Key elements of the approach are summarized below:

- The CTE receptor was assumed to be exposed at a location where concentration values were the average for the whole lake, and the RME receptor was assumed to be exposed at a location where concentrations were equal to the 95th percentile of values from the lake.
- Because only limited data were available for measured concentrations of pesticide X in aquatic prey items, the concentration values in aquatic prey were estimated using a linear bioaccumulation model: $C(\text{prey}) = C(\text{sed}) \times \text{BAF}$. The BAF was estimated from the existing data by finding the best fit correlation between the concentration values in sediment and crayfish at 7 locations in the lake: $C(\text{crayfish}) = 5.04 \times C(\text{sed})$ ($R^2 = 0.792$).
- The TRV values were based on a study in mink in which the toxicity endpoint was the percent inhibition of reproductive success.

These inputs and the resulting HQ values are shown in Exhibit 4-10. As seen, estimated risks to the CTE receptor approach or slightly exceed a level of concern (HQ=4.7E-01 to 1.4E+00), and risks to an RME receptor are well above a level of concern (9.1E+00 to 2.7E+01). The chief pathway contributing to the dose and risk is ingestion of contaminant in aquatic food web items (crayfish, fish, amphibians, etc.).

EXHIBIT 4-10

REFINED SCREENING POINT ESTIMATE INPUTS AND RESULTS

Basic model

$$\begin{aligned} HQ &= DI(\text{total}) / TRV \\ DI(\text{total}) &= DI(\text{water}) + DI(\text{food}) + DI(\text{sed}) \\ DI(i) &= C(i) * IR(i) * AUF(i) \end{aligned}$$

Other Assumptions

$$\begin{aligned} C(\text{diet}) &= C(\text{sed}) * BAF \\ IR(\text{sed}) &= IR(\text{diet}) * F(\text{sed}) \\ IR(\text{diet}) &= IR(\text{total}) * F(\text{diet}) \end{aligned}$$

Category	Variable	Variable	Units	Point Est. Values	
				CTE	RME
Inputs	Concentration	Concentration in water	mg/L	0.12	0.38
		Concentration in sediment	mg/kg	24	77
		BAF (sediment to aquatic prey)	--	5	5
		Concentration in aquatic prey	mg/kg	120	385
	Intake Rates	Total water intake rate	L/kg-day	0.082	0.12
		Total food intake rate	kg/kg-day	0.06	0.09
		Fraction of diet that is sed	--	0.03	0.06
		Fraction of diet that is aquatic prey	--	0.15	0.25
	Area Use Factors	Fraction of total water ingested at the lake	--	0.3	0.6
		Fraction of total diet from the lake	--	0.25	0.6
	TRVs	LOAEL-based TRV	mg/kg-day	0.6	0.6
		NOAEL-based TRV	mg/kg-day	0.2	0.2
Results	Daily Intake	Water ingestion	mg/kg-day	3.0E-03	2.7E-02
		Sediment ingestion	mg/kg-day	1.1E-02	2.5E-01
		Aquatic prey ingestion	mg/kg-day	2.7E-01	5.2E+00
		Total	mg/kg-day	2.8E-01	5.5E+00
	HQ (LOAEL-Based)	Water ingestion		4.9E-03	4.6E-02
		Sediment ingestion		1.8E-02	4.2E-01
		Aquatic prey ingestion		4.5E-01	8.7E+00
		Total		4.7E-01	9.1E+00
	HQ (NOAEL-Based)	Water ingestion		1.5E-02	1.4E-01
		Sediment ingestion		5.4E-02	1.2E+00
		Aquatic prey ingestion		1.4E+00	2.6E+01
		Total		1.4E+00	2.7E+01

SMDP 1 at Step 2 of ERAGS

The BTAG considered these results to indicate that inhibition of reproduction was possible in at least some members of the exposed population, but that the fraction of the population that was affected and the degree of impact on the population was difficult to judge from the point estimate calculations. Based on this, a decision was made to conduct a screening level PRA in order to provide some additional information on the magnitude and probability of risk.

Workplan 1

The contractor performing the risk assessment developed a brief workplan that proposed an approach for a screening level PRA. The plan called for a Monte Carlo-based evaluation of variability in exposure and risk among different members of the exposed mammalian omnivore (raccoon) population. In brief, all exposure inputs that were treated as constants in the point estimate approach (i.e., were the same for CTE and RME exposure) were also treated as constants in the PRA evaluation. Because water contributed so little to dose or HQ, this pathway was not evaluated in the PRA, but was accounted for by adding in the point estimate values to the PRA results. All variables that are fractions (i.e., may only assume values between zero and one) were modeled as beta distributions, and all other variables were modeled as lognormal. For screening purposes, the parameters for all distributions were selected so that the mean and 95th percentile values of the PDF's matched the corresponding CTE and RME point estimates. The BTAG reviewed this proposed approach and authorized PRA work to begin.

Screening Level PRA Results

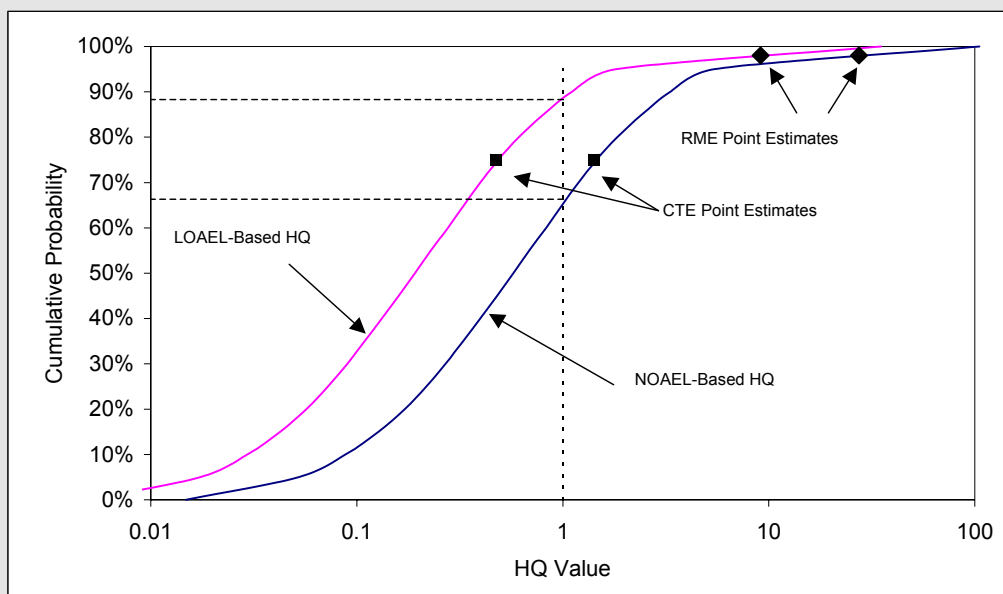
The screening level PRA inputs and the resulting estimates of the variability in HQ are shown in Exhibit 4-11. The CTE and RME point estimates are also shown for comparison. As seen, the PRA distribution of HQ values indicates that about 68% of the individuals in the population are likely to have HQ values below 1E+00, while 32% have HQ values above 1E+00.

Comparison of the CTE point estimates of HQ to the mean HQ values derived by PRA reveals the values are very close. This is expected because both depend on the mean values of the input variables, and the same mean values were used in both sets of calculations. With regard to upper-bound estimates, the RME point estimate values are at the 98th percentile of the PRA HQ distribution, within the target range (90th to 99th) usually considered appropriate. Note, however, that the 98th percentile is about 5-fold higher than the 95th percentile, emphasizing the high sensitivity of the RME HQ values to the precise percentile of the RME.

EXHIBIT 4-11

SCREENING LEVEL PRA CALCULATIONS OF HQ DISTRIBUTION

Data Category	Variable	Units	Screening Level Distribution		
			Type	param 1	param 2
Concentrations	Concentration in water	mg/L	Not evaluated in PRA		
	Concentration in sediment	mg/kg	LN	24	33
	BAF	--	Const	5	
	Concentration in aquatic prey	mg/kg	Calculated		
Intake Rates	Total water intake rate	L/kg-day	Not evaluated in PRA		
	Total food intake rate	kg/kg-day	LN	0.060	0.060
	Fraction of diet that is sed	--	Beta	3.42	110.7
	Fraction of diet that is aquatic prey	--	Beta	6.10	34.6
Area Use Factors	Fraction of total water ingested from lake	--	Not evaluated in PRA		
	Fraction of total diet from the lake	--	Beta	1.20	3.59
TRVs	LOAEL-based TRV	mg/kg-day	Const	0.6	
	NOAEL-based TRV	mg/kg-day	Const	0.2	



TRV Basis	Central Tendency			Upper Bound		
	Mean of PRA	Point Est CTE	Ratio	95th of PRA	Point Est. RME	Ratio
NOAEL	1.44	1.42	0.99	5.4	27.4	5.06
LOAEL	0.48	0.47	0.99	1.80	9.12	5.06

SMDP 2

The BTAG considered these results, and decided that it was very probable that pesticide X was causing an effect in some members of the exposed population, but decided that a final risk management decision would be facilitated by characterizing the distribution of responses (rather than the distribution of HQ values). The BTAG asked the contractor performing the work to develop a proposed approach for characterizing the distribution of responses.

Workplan 2

The contractor obtained a copy of the toxicity report upon which the TRVs were based, and determined that the study did include sufficient dose-response data to support reliable dose-response modeling. The contractor recommended that this be done using EPA's BMDS. The BTAG approved this proposed approach and authorized work to proceed.

PRA Refinement 1

The contractor fit the raw dose-response data (inhibition of reproduction in mink) to a number of alternative models available in BMDS, and found that the dose-response curve could be well characterized by the Hill Equation with nonconstant variance, as follows:

$$\begin{aligned}\text{Mean Response at dose } d (\% \text{ decrease in reproduction}) &= (100 \times d^{2.5}) / (0.9^{2.5} + d^{2.5}) \\ \text{Std. Dev. in Response at dose } d (\%) &= \text{SQRT}[1.6 \cdot (\text{mean response at dose } d)^{1.3}]\end{aligned}$$

Based on this model, the point estimate LOAEL value (0.6 mg/kg-day) corresponds to an effect level of about 27%, and the NOAEL of 0.2 mg/kg-day corresponds to an effect level of about 2%.

Using this exposure-response model in place of the point-estimate TRV values, the refined PRA predicted a distribution of responses in the exposed population as shown in Exhibit 4-12. As seen, approximately 81% of the population was predicted to experience an effect on reproduction smaller than 10%, while 9% were expected to have a reduction of 10 to 30%, 4% a reduction of 30 to 50%, and 6% a reduction of more than 50%. On average across all members of the exposed population, the predicted reduction in reproductive success was about 9%.

EXHIBIT 4-12

SIMULATED DISTRIBUTION OF RESPONSES

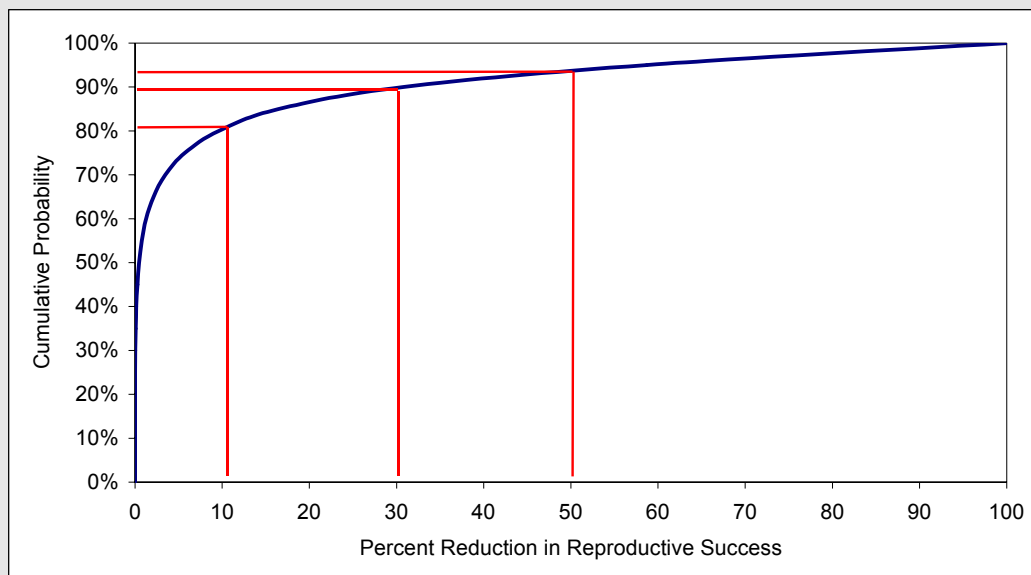
Exposure-Response Model

Resp = Normal(Mean,Stdev)

Mean = $a + b \cdot x^n / (x^n + k^n)$

Stdev = $\alpha \cdot \text{mean}^\rho$

x	Total daily intake
a	0
b	100
k	0.9
n	2.5
alpha	1.6
rho	1.3



Percent Reduction	Percent of Population
0-10%	81%
10-30%	9%
30-50%	4%
>50%	6%

SMDP3

The BTAG debated the likely population-level consequences of this predicted distribution of responses in members of the exposed population. After consulting with a field biologist with experience in the population dynamics of mammals such as racoons, the BTAG decided that the distribution of responses in the exposed population would cause a continued stress on the mammalian omnivore community and that reductions in population number were likely over time. Based on this, the risk manager and the BTAG agreed that remedial action was desirable and that a range of alternative clean-up strategies should be investigated. This was performed using the methods described in Chapter 5 (see Exhibit 5-5).

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CHAPTER 5

PROBABILISTIC RISK ASSESSMENT AND PRELIMINARY REMEDIATION GOALS

5.0 INTRODUCTION

According to the National Contingency Plan (NCP) (U.S. EPA, 1990a, 40CFR §300.430(d)(4)), risk assessment and risk management decision making go hand-in-hand: data from the remedial investigation are used to characterize risk, and results of the baseline risk assessment help to establish acceptable exposure levels for use in developing remedial alternatives. In practice, risk managers may identify two major objectives of risk assessment: (1) to determine if remediation is necessary (i.e., *Is there unacceptable risk at the site?*); and (2) if remediation is necessary, to determine a preliminary remediation goal (PRG) (i.e., *What chemical concentrations would result in a risk estimate that will be adequately protective of human health and the environment?*). The answer to the first question (*is there unacceptable risk?*) depends upon a number of factors, including the measured or estimated concentration levels of contaminants in site media, and takes uncertainty in the measurements into account. In contrast, the answer to the second question (*what is the PRG needed to achieve a specified level of protection?*) does not necessarily depend on any knowledge of the actual level or pattern of site-specific concentration data, and does not necessarily depend on the uncertainty in site concentration data. Thus, while exposure point concentrations (EPCs) and PRGs are closely related to each other, they have important differences (see Section 5.1 for further elaboration on EPCs and PRGs).

Once a risk manager has selected a PRG at a site, determining whether a particular area meets or will meet the PRG requires careful comparison of site data with the PRG, including a consideration of the uncertainty in the site data. For a further discussion on variability and uncertainty in the concentration term, readers are urged to consult Appendix C in this guidance.

EXHIBIT 5-1

SUMMARIES OF SOME KEY TERMS

Preliminary Remediation Goal (PRG) - initially developed chemical concentration for an environmental medium that is expected to be protective of human health and ecosystems. PRGs may be developed based on applicable or relevant and appropriate requirements, or exposure scenarios evaluated prior to or as a result of the baseline risk assessment. (U.S. EPA, 1991a).

Generic PRG - a chemical concentration protective of human health developed prior to the baseline risk assessment that uses default exposure assumptions representing common exposure scenarios, e.g., Region 3 risk-based concentrations (RBCs) or Region 9 PRGs.

Site-specific PRG - site-specific chemical concentration, protective of human health and ecosystems, based on exposure scenarios in the baseline risk assessment. Generally calculated for the various exposure scenarios considered in the baseline risk assessment.

Remediation Goals (RG) - site-specific chemical concentration, protective of human health and ecosystems, chosen by the risk manager as appropriate for a likely land use scenario.

Remediation Action Level (RAL) - the "not-to-exceed" level; a concentration such that remediation of all concentrations above this level in an exposure unit lowers the EPC sufficiently to achieve a target risk level. The RAL will depend on the mean, variance, and sample size of the concentrations within an exposure unit as well as considerations of short-term effects of the chemicals of concern.

Cleanup Level (Final Remediation Level) - chemical concentration chosen by the risk manager after considering both RGs and the nine remedy selection criteria of the NCP (U.S. EPA, 1990a). Also referred to as Final Remediation Levels (U.S. EPA, 1991a), chemical-specific cleanup levels are documented in the Record of Decision (ROD). A cleanup level may differ from a PRG because risk managers may consider details of the site-specific exposure, various uncertainties in the risk estimate, and implementation issues (e.g., the technical feasibility of achieving the PRG).

EXHIBIT 5-2

DEFINITIONS FOR CHAPTER 5

95% UCL for mean - The one-sided 95% upper confidence limit for a population mean; if a sample of size (n) was repeatedly drawn from the population, the 95% UCL will equal or exceed the true population mean 95% of the time. It is a measure of uncertainty in the mean, not to be confused with the 95th percentile (see below), which is a measure of variability. As sample size increases, the difference between the UCL for the mean and the true mean decreases, while the 95th percentile of the distribution remains relatively unchanged.

95th Percentile - The number in a distribution that is greater than 95% of the other values of the distribution, and less than 5% of the values. When estimated from a sample, this quantity may be equal to an observed value, or interpolated from among two values.

Applicable or Relevant and Appropriate Requirements (ARARs) - Federal or state environmental standards; the NCP states that ARARs should be considered in determining remediation goals. ARARs may be selected as site-specific cleanup levels.

Backcalculation - A method of calculating a PRG that involves algebraic rearrangement of the risk equation to solve for concentration as a function of risk, exposure, and toxicity.

Bootstrap Methods - Parametric and non-parametric methods for estimating confidence intervals for a statistic by resampling directly from the data set with replacement.

Coverage - Confidence intervals are expected to enclose a true but unknown parameter according to a specified probability, such as 90% or 95%. This is the expected coverage of the confidence interval, given a specified significance level (α). The difference between the expected coverage and the actual coverage is one metric for evaluating statistical methods that yield different confidence intervals.

Exposure Point Concentration (EPC) - The average chemical concentration to which receptors are exposed within an exposure unit. Estimates of the EPC represent the concentration term used in exposure assessment.

Exposure Unit (EU) - For Superfund risk assessment, the geographical area about which a receptor moves and contacts a contaminated medium during the period of the exposure duration.

Forward Calculation - A method of calculating a risk estimate that involves the standard arrangement of the risk equation to solve for risk as a function of concentration, exposure, and toxicity.

Iterative Reduction (IR) - A method of calculating a PRG that involves successively lowering the concentration term until the calculated risk is acceptable. This method can be applied to any medium.

Iterative Truncation (IT) - A method of calculating a PRG that involves developing an expression for the concentration term in which higher values of concentration are removed or "truncated" to reduce the maximum concentration, and re-calculating risks associated with the reduced concentration. The method may be repeated with consecutively lower truncation limits until risk is acceptable.

Land Method - The conventional method for calculating uncertainty in the mean concentration (e.g., 95% UCL) when the sample data are obtained from a lognormal distribution (U.S. EPA, 1992).

Maximum Detected Concentration (MDC) - The maximum concentration detected in a sample.

True Mean Concentration - The actual average concentration in an exposure unit. Even with extensive sampling, the true mean cannot be known. Only an estimate of the true mean is possible. A greater number of representative samples increases confidence that the estimate of the mean more closely represents the true mean.

Two Office of Solid Waste and Emergency Response (OSWER) guidance documents in preparation: (1) *Draft Guidance on Calculation of Upper Confidence Limits for Exposure Point Concentrations at Superfund Sites* (U.S. EPA, 2001a), and (2) *Draft Guidance on Surface Soil Cleanup at Superfund Sites: Applying Cleanup Levels* (U.S. EPA, 2001b), also address topics related to the calculation of EPCs and comparison of those EPCs to a PRG.

In practice, calculations of risks, given concentration data, are commonly referred to as “forward calculations”, while calculations of PRGs, based on chosen target risk levels, are referred to as “back-calculations”. This terminology reflects the algebraic rearrangement of the standard risk equation needed to solve for the concentration term when point estimates are used to characterize exposure and toxicity input variables. For probabilistic risk assessment (PRA), the process for developing a PRG can be more complex. This chapter presents methods and recommendations for developing site-specific PRGs within the framework of PRA.

Are there different types of PRGs?

Generic PRGs have been developed for some chemicals and exposure media using point estimates based on standard default exposure assumptions (e.g., U.S. EPA, 1991b) and toxicity criteria available in the Integrated Risk Information System (IRIS) or Health Effects Assessment Summary Table(s) (HEAST) or from Environmental Protection Agency’s (EPA’s) National Center for Environmental Assessment. Soil Screening Guidance levels, Region 9’s PRG table and Region 3’s Risk Based Concentrations (RBCs) table are examples of generic point estimate PRGs. Generic PRGs are often used for screening chemicals of potential concern in Data Evaluation and Hazard Identification steps of the risk assessment process.

☞ *There is a clear distinction between generic PRGs, site-specific PRGs, remediation goals (RGs), and cleanup levels. The focus of this chapter is on site-specific PRGs.*

At this time, EPA does not recommend the use of PRA to develop generic PRGs. Until the science and policy decisions associated with the use of default assumptions in PRA have evolved, generic PRGs should only be developed from point estimate methods, as was done in the examples listed above.

As indicated in Exhibit 5-1, site-specific PRGs generally are developed after the baseline risk assessment. However, during the feasibility study or even later in the Superfund process, the methods described in this chapter may be used to modify cleanup levels at the discretion of the risk manager. However, it is generally not appropriate to use PRA for modifying cleanup levels during the feasibility study if PRA was not used in the baseline risk assessment.

☞ *Risk-based PRGs are initial guidelines and do not represent final cleanup levels.*

Only after appropriate analysis in the remedial investigation/feasibility study (RI/FS), consideration of public comments, and issuance of the record of decision (ROD) does a RG become a final cleanup level. A cleanup level may differ from a RG because risk managers may consider various uncertainties in the risk estimate. While the two main criteria for determining a cleanup level are: (1) protection of human health and the environment, and (2) compliance with applicable or relevant and appropriate requirements (ARARs), a cleanup level may differ from the RG because of modifying criteria, such as feasibility, permanence, state and community acceptance, and cost effectiveness. These and other factors are reflected in the nine evaluation criteria outlined in the NCP (U.S. EPA, 1990a; 40CFR §300.430(e)(9)(iii)) (see Chapter 1, Exhibit 1-2).

This chapter and Appendix C provide a comprehensive description of the issues associated with developing site-specific PRGs with both point estimate and probabilistic approaches, including the use of geostatistics. Because methods for calculating a 95% upper confidence limit for the mean (95% UCL) are discussed fully in the *Draft Guidance on Calculation of Upper Confidence Limits for Exposure Point Concentrations at Superfund Sites* (U.S. EPA, 2001a) and *Draft Guidance on Surface Soil Cleanup at Superfund Sites: Applying Cleanup Levels* (U.S. EPA, 2001b), they are covered only briefly in this guidance. In general, this chapter, Appendix C, and the Superfund guidance under development should be consulted by risk assessors when developing site-specific PRGs.

5.1 GENERAL CONCEPTS REGARDING EPCs AND PRGs

PRGs developed from point estimate risk assessments and PRAs will be discussed in this section to compare and contrast the two approaches. The PRG is a special case of the concentration term (or EPC) in the risk equation. The intent of the EPC is to represent the average chemical concentration in an environmental medium in an exposure unit (EU) (i.e., the area throughout which a receptor moves for the duration of exposure). The EPC should be determined for individual EUs within a site. Because an EPC is calculated from a sample, there is uncertainty that the sample mean equals the true mean concentration within the EU; therefore, to account for associated uncertainty, the 95% upper confidence limit for the mean (95% UCL) is generally used for Superfund risk assessments (U.S. EPA, 1992). For both point estimate and probabilistic approaches, the PRG is an assumed value of the EPC that yields a risk estimate that is at or below an acceptable risk level.

☞ The EPC usually represents the average concentration within the EU estimated from a sample; the PRG usually represents the average concentration within the EU that corresponds to an acceptable level of risk.

The PRG may be thought of as a goal for the post-remediation EPC (see Section 5.1.2). Specifically, after remediation is completed, the average concentration (or the 95% UCL used as a measure of uncertainty in the average) for the EU should be sufficiently low to be protective of human health and the ecosystem. While the methods used to calculate the pre- and post-remediation EPC may differ, the interpretation of the EPC remains constant. For example, if the 95% UCL is used to represent the EPC before remediation, then the EPC following remediation (e.g., the PRG) should also represent a 95% UCL (Bowers et al., 1996).

Risk assessors may consider both variability and uncertainty in the development of an EPC. The calculation of a 95% UCL generally requires knowledge of not only chemical concentration measurements within the EU but also the receptor's behavior. Relevant information may include the variability in concentrations in the given sample, the sampling locations, and variability in the movement and activity patterns of receptors within the EU. A discussion of spatial and temporal variability associated with characterizing contamination in different exposure media is presented in Appendix C, and important sources of uncertainty in the EPC are discussed in Section 5.1.1.

For all risk assessments, chemical concentration measurements should be collected in a manner that is consistent with an understanding of both the source of contamination and the definition of the exposure unit. An investment of time and resources should be made in planning, scoping, and problem formulation. Part of this investment is to follow the Data Quality Objectives (DQO) process to obtain samples appropriate for the risk assessment and sufficient to support the remedial decision (U.S. EPA, 1993, 1994, 2000). Using new methods of sample collection and analysis such as dynamic workplans and real-time analysis may enable risk managers to get the most "bang for the buck" from the resources available for site characterization. Information about these methods and the DQO process is available from EPA's Office of Emergency and Remedial Response (U.S. EPA,

2001c) and Technology Innovation Office (U.S. EPA, 2001d, 2001e). The world wide web address is http://clu-in.org/charl_edu.cfm#syst_plan.

5.1.1 SOURCES OF UNCERTAINTY IN THE EPC

The 95% UCL is generally used as the EPC to represent uncertainty in the mean concentration in both the central tendency exposure (CTE) and reasonable maximum exposure (RME) risk estimates for Superfund (U.S. EPA, 1992). Similarly, in PRA, a probability distribution for uncertainty may be used in a two-dimensional Monte Carlo analysis (2-D MCA) simulation (see Appendix D) to represent a source of uncertainty in the EPC. There are numerous potential sources of uncertainty in the estimate of the true mean concentration within the EU. The sources of uncertainty when the EPC is expressed as either a single number or a distribution are the same and can be grouped into the following four broad categories:

- (1) ***Uncertainty in the sample data.*** A limited number of measurements in the sample are used to make inferences about the EPC and the spatial distribution of concentrations at a site. Uncertainties may arise from many factors, including both sampling variability and measurement error. As the number of samples increases, the uncertainty generally decreases (e.g., more information will be available to characterize the spatial distribution and variation in concentration). In point estimate risk assessments, the 95% UCL is generally used as the EPC to account for the uncertainty in estimating the average concentration within an EU.
- (2) ***Uncertainty about the location of the EU.*** When the size of a receptor's EU is less than the size of the site, the placement of the EU may be a source of uncertainty, especially when the contamination is distributed unevenly across the site and the PRA includes exposure scenarios for future land uses.
- (3) ***Uncertainty in the behavior of the receptor.*** Even in the case of extremely well characterized sites, it remains uncertain whether the receptor will contact the environmental medium in a temporal and/or spatial distribution that can be adequately represented by the environmental samples collected.
- (4) ***Uncertainty in chemical concentrations over time.*** The concentration in a given medium may undergo temporal changes, which may introduce uncertainty in estimates of a long-term average. Examples include the movement or attenuation of a solvent plume in groundwater; aerobic or anaerobic degradation; the change in the average concentration in a fish population due to changes in population dynamics; and the mixing of surface and subsurface soil over time.

A lack of knowledge in all four categories may be considered when selecting approaches to quantify uncertainty in the concentration term. One of the first steps in quantifying uncertainty is to define the EU, or the geographical area in which individual receptors are randomly exposed for a relevant exposure duration. Depending on the receptor's movement and activities, an EU may be as small as a child's play area (e.g., sandbox) or as large as the foraging area of an upper trophic level animal predator (e.g., an entire military base). The relationship between the size of the EU, the movements of the target receptor, and health endpoint of concern (i.e., acute or chronic) may dictate the appropriate use of sample data in developing an EPC. One of the assumptions generally made for the concentration term in Superfund risk assessment is that receptors contact all parts of an EU at random, and that measurements are obtained from a simple (or stratified) random sample. If an individual is randomly exposed within the same EU over a long period of time, the most appropriate metric for the EPC would be the true (but unknown) population mean of the concentrations within the EU (e.g., 95% UCL).

Often, the scale of the EU will be different (smaller or larger) than the scale of the sample data. For example, an ecological receptor population may have a small home range relative to the size of the entire site, or the endpoint of concern may be acute toxicity, requiring an evaluation of a short-term exposure scenario. If the receptors are not expected to contact all parts of the site with equal probability, then the EU may be redefined so that only a subset of the data collected for site characterization are used to estimate the EPC. In addition, the location of the EU may be unspecified within the site because there may be multiple areas that provide suitable habitat for the receptor population. Departing from the assumption of random exposure within one unique geographic area presents an additional challenge to estimating an EPC. In some cases, it may be informative to develop multiple estimates of the EPC in a PRA. By treating the EPC as a random variable, risk assessors can explore the effect of uncertainty in the location of the EU. A variety of modeling approaches are available to calculate an EPC (e.g., arithmetic mean, or 95% UCL) based on the spatial variability in chemical concentrations measured over an area larger than the EU. Methods such as geostatistics (see Section 5.5.2 and Appendix D), Microexposure Event Modeling (MEE) (see Appendix D), and random walk scenarios (Hope, 2000, 2001) may be used to quantify both the spatial and temporal variability in exposure to varying concentrations. Using these methods, risk assessors may redefine the EU to be more representative of the random movement of the receptor during the period of exposure. Because these modeling approaches may be considered more advanced methods for quantifying the EPC, they are generally considered in Tier 3 of the PRA process (see Chapter 2).

5.1.2 PRE- AND POST-REMEDIATION EXPOSURE POINT CONCENTRATIONS

The differences between pre- and post-remediation EPCs are discussed below. In general, both estimates of the EPC are based on the same concepts regarding the exposed population and the definition of the EU. However, the post-remediation EPC will tend to yield lower estimates of (post-remediation) risk and can require more advanced methods for calculating uncertainty (e.g., 95% UCL).

The pre-remediation EPC is determined based on existing site sampling at the time of the remedial investigation, prior to remediation. By contrast, the post-remediation EPC generally is determined based on a prediction of site conditions after remediation. For example, in surface soil, the post-remediation EPC can be determined by substituting the nondetect level (generally, half the laboratory reporting limit) for some of the high concentrations in the sample and recalculating the EPC. The underlying assumption in calculating a post-remediation EPC is that remediation will have sufficiently reduced the chemical concentrations at the site, and the risk existing after remediation is complete will be equal to or less than the target risk level of concern.

The preceding discussion is most applicable to surface soil PRGs. In general, compared with other exposure media (e.g., groundwater, air), surface soil is stationary with relatively constant chemical concentrations within an EU. For other environmental media, more complex approaches may be needed to estimate the post-remediation EPC. Modeling of the remediation process may introduce additional uncertainty not encountered in risk estimates based on the pre-remediation EPC.

5.1.3 REMEDIATION ACTION LEVELS (RALs) AND 95% UCL CALCULATION METHODS

The EPC should incorporate knowledge about the spatial distribution of contamination, the behavior of the receptor, the location of the EU, land use, and other factors. These factors affect both the numerical value of an EPC and uncertainty associated with this estimate. In many cases, it is presumed factors associated with land use will not change after remediation.

The remediation action level (RAL) is the maximum concentration that may be left in place at any location within an EU such that the average concentration (or 95% UCL as a measure of the average) will not

present a risk above levels of concern. This RAL may be considered a “not-to-exceed” threshold or action level for the purposes of site remediation. Using surface soil as an example, areas within the EU that have concentrations greater than the RAL may be excavated and replaced with clean fill (e.g., nondetect surrogate values). To obtain a post-remediation EPC, the 95% UCL is calculated after substituting the surrogate nondetect value for all measurements located within the EU that are greater than the RAL.

When appropriate, the same statistical method of uncertainty should be used to estimate UCLs for both the pre- and post-remediation EPCs. However, in some instances, the method used for calculating the pre-remediation EPC will be inappropriate for calculating the post-remediation EPC, because the distribution of contaminant concentration will have changed. For example, pre-remediation site sampling may suggest that variability in concentrations can be reasonably characterized by a lognormal distribution, which would support the use of the Land method for estimating the 95% UCL. The post-remediation site conditions, however, may reflect a mixture of clean fill and contamination, resulting in a poor fit to a lognormal distribution (see Figure 5-3, Section 5.5.3). In this case, the Land method would not be appropriate. Because of the difference in the statistical distribution of concentration measurements used to estimate the pre-remediation EPC and post-remediation EPC, a non-parametric (i.e., distribution free) method should be considered for calculating uncertainty in the average concentrations in both pre- and post-remediation scenarios. In general, when the method used to calculate the 95% UCL for a post-remediation scenario is different than that of the pre-remediation scenario, the 95% UCL for the pre-remediation scenario should be recalculated with the post-remediation method. Results of this change in methodology can be presented as part of a quantitative uncertainty analysis. Specifically, this recalculation will allow for an evaluation of the effect that a RAL has on the confidence interval for the mean. The discordance between pre- and post-remediation distributions can be expected to increase as the degree of remediation needed to achieve a target risk level of concern increases.

In general, risk assessors should be aware of the practical and statistical issues associated with the various methods of calculating the 95% UCL, and the application of these methods to both the pre- and post-remediation concentration distribution. Different methods can yield very different confidence intervals, some of which are expected to yield more accurate coverage (i.e., likelihood that the confidence interval includes the parameter) depending on characteristics of the underlying distribution of concentrations, such as distribution shape, sample size, and variance (Gilbert, 1987; Hall, 1988). Information about a variety of parametric and non-parametric methods, such as bootstrap resampling, can be found in *The Lognormal Distribution in Environmental Applications* (U.S. EPA, 1997), *Estimating EPCs When the Distribution is Neither Normal nor Lognormal* (Schulz and Griffin, 1999) and a Superfund guidance document currently under development, *Draft Guidance on Calculation of Upper Confidence Limits for Exposure Point Concentrations at Superfund Sites* (U.S. EPA, 2001a).

5.1.4 CONSIDERATION OF RISK FROM ACUTE TOXICITY

Sometimes a risk assessment will need to address more than one health endpoint of concern (e.g., cancer and noncancer). The RAL should be sufficiently low so that it is simultaneously protective of each endpoint of concern. Generally, when acute toxicity is a concern, the long-term average concentration across the entire EU may not be the appropriate metric for assessing risks. For example, a single episode of a child ingesting a handful of soil containing malathion may result in an acute toxic effect to that child. Therefore, the RAL must not only be low enough to reduce the post-remediation EPC to acceptable long-term average levels, but also low enough that acute toxicity will not be an issue. This consideration applies to both point estimate and probabilistic estimates of PRGs.

For consideration of acute toxicity, the risk assessor should consult, as appropriate, with a toxicologist in the development of RALs.

For a small number of chemicals, toxicity values have been determined based on acute effects (e.g., nitrate in drinking water). However, at present, EPA does not have acute toxicity criteria or guidance on acute toxicity applied to the RAL. Hence, consultation with a toxicologist is vital.

5.1.5 CHARACTERIZATION OF UNCERTAINTY IN THE EPC: POINT ESTIMATES AND DISTRIBUTIONS

In point estimate risk assessments, the 95% UCL is typically used to characterize uncertainty in the EPC (U.S. EPA, 1992). In PRA, either a point estimate (e.g., 95% UCL) or a probability distribution may be used to characterize uncertainty in the concentration term. The probability distribution may characterize either variability or uncertainty. The terms probability distribution for variability (PDFv) and probability distribution for uncertainty (PDFu) can be used to distinguish between probability distributions for variability and uncertainty, respectively.

The decision to use a point estimate, PDFv, or PDFu, as the input for the concentration term in a Monte Carlo model will depend on the goals of the Monte Carlo simulation, as determined by the tiered process (see Chapter 2). If the goal is to characterize variability in risk, in general, a one-dimensional Monte Carlo analysis (1-D MCA) will be used and the appropriate input for the concentration term will be a point estimate that characterizes uncertainty in the mean concentration within the EU. As explained in Section 5.1.1, risk assessors will need to consider the relationship between the size of the EU, the movements of the target receptor, and health endpoint of concern (i.e., acute or chronic) to determine how to use the available sample data to define the EPC. A PDFu is typically not an appropriate choice for the concentration term in a 1-D MCA when the goal is to characterize variability in risk. Mixing of a PDFu for the concentration term with PDFv's for other exposure variables in 1-D MCA would yield a single risk distribution from which the relative contributions of variability and uncertainty could not be evaluated. Use of a PDFu for the concentration term may be considered in 2-D MCA simulations (see Appendix D), where the goal may be to characterize both variability and uncertainty in risk.

When the sample size is small and the variance is large, the 95% UCL may exceed the maximum detected concentration (MDC). In such a case, the MDC is generally used to estimate the EPC, although the true mean may still be higher than this maximum value (U.S. EPA, 1992). For poorly characterized sites, there may be considerable uncertainty that site remediation will be sufficient to reduce the 95% UCL to a health-protective level. Poor site characterization may provide an impetus for the risk manager to opt for a more health-protective remedial alternative or to collect additional data.

To ensure that actual cleanup based on a RAL is protective generally requires post-remediation confirmation sampling. This step in the risk management process is emphasized further in Section 5.8 on measurement of attainment.

5.1.6 MULTIPLE CHEMICALS


Developing PRGs for multiple chemicals in one or more environmental media is particularly challenging. When multiple chemicals are present, the total risk level should be considered for regulatory purposes with each chemical contributing a portion of the total risk. This issue is quite complex and usually will affect both the calculation of the risk and development of site-specific PRGs. Chemicals may exhibit different spatial and temporal variability within the EU. Fate and transport characteristics may vary between chemicals as well as between different areas of the site. Co-located sampling, or geostatistical techniques (e.g., co-kriging) may

provide insights regarding relationships in spatial patterns for different chemicals (see Appendices C and D) and the corresponding exposures for receptors.

5.2 WHEN TO USE PRA FOR DEVELOPING PRGs

Because point estimate risk assessments and PRA employ different approaches to characterize variability and uncertainty, the resulting RME risk estimates and calculations of PRGs are often different. The magnitude of the difference can depend on many factors, including the number of input variables described with probability distributions in the PRA, the choice of distributions used to characterize variability or uncertainty (especially for those variables that are highly ranked in a sensitivity analysis), the percentile of the probability distribution that corresponds with RME point estimate for each input variable, and the choice of percentile from the PRA used to represent the RME risk (e.g., 95th percentile). Since the results of a point estimate approach and PRA can be expected to differ, but the magnitude of the difference is not known *a priori*, this can present a challenge in deciding whether or not to conduct a PRA to develop a PRG. The potential advantages and disadvantages of both the point estimate approach and the PRA can be factored into the decision (see Chapter 1, Exhibits 1-6 and 1-7).

In general, PRA may be appropriate for developing site-specific PRGs in cases where PRA has also been used to estimate site-specific risks. As indicated by the tiered approach (see Chapter 2), if the risk manager determines that quantifying variability and uncertainty may enhance risk management decision making, PRA may be warranted. If a PRA is feasible, the risk manager should proceed to Tier 2 and employ PRA to complete the RI/FS process. Usually, embedded in a site-specific PRG are all of the exposure assumptions and toxicity metrics used in the risk assessment. Hence, introducing the use of PRA for PRGs in the feasibility study (or any time after the remedial investigation and baseline risk assessment are complete) would, in effect, undermine the tiered approach.

 *If only point estimates were used in the risk assessment, probabilistic methods should not be used for PRG development.*

If additional data have been collected to conduct PRA, the point estimate risk assessment should be revisited with the new data as well. As discussed in Chapter 2, a point estimate risk assessment (Tier 1) should always accompany a PRA. PRA is intended to enhance risk management decision making, and should not be viewed as a substitute for point estimate approaches. Using the tiered approach, a risk assessor can determine the appropriate level of complexity that is supported by the available information to conduct the risk assessment and to calculate a PRG.

5.3 METHODS FOR DEVELOPING PRGs

Risk assessors may use PRA to quantify sources of uncertainty and variability in the calculation of PRGs as well as risks. Two of the common methods for calculating PRGs in PRA include: (1) backcalculation (see Section 5.4), which is equivalent in concept to the point estimate calculation of a PRG; and (2) iterative forward calculation methods, including iterative reduction and iterative truncation (see Section 5.5). Backcalculation can be used in PRA when the target risk and concentration terms are expressed as point estimates. Iterative methods can be more involved, but unlike backcalculation, there are no constraints on their application to PRA. The two approaches yield the same result when the same assumptions are used in the risk assessment.

5.4 BACKCALCULATION

Traditionally, risk is calculated as a function of multiple exposure variables, including the concentration term, and toxicity value (Equation 5-1). If one or more of the exposure variables is described by a PDF, a Monte Carlo simulation will yield a distribution for risk (see Chapter 1).

Backcalculation methods can be envisioned as setting a target risk level (e.g., RME risk equal to 10^{-6} or Hazard Index equal to 1) and then algebraically reversing the risk equation to solve for the concentration term (Equation 5-2). A Monte Carlo simulation using Equation 5-2 will yield a distribution of concentrations that reflects the combination of distributions from all other exposure variables.

$$\frac{C \times IR \times EF \times ED}{BW \times AT} = Intake$$
$$Intake \times Toxicity = Risk$$
$$C \times V = Risk$$

Equation 5-1

$$C = Risk \times V^{-1}$$

Equation 5-2

where,

<i>Toxicity</i>	=	toxicity term representing either the cancer slope factor (CSF) or reference dose (1/RfD) for the chemical in the exposure medium
<i>C</i>	=	concentration term
<i>V</i>	=	algebraic combination of the toxicity term with all exposure variables except <i>C</i>
<i>IR</i>	=	ingestion or inhalation rate
<i>AT</i>	=	averaging time
<i>BW</i>	=	body weight
<i>ED</i>	=	exposure duration
<i>EF</i>	=	exposure frequency

This calculation produces a distribution of PRGs that represents the same sources of variability as a forward calculation of risk. Each percentile of the PRG distribution (i.e., the α percentile) corresponds to the $1-\alpha$ percentile from the distribution of risk estimates. For example, if the 95th percentile of the distribution of risk estimates was chosen to represent the RME individual, the 5th percentile ($1-0.95=0.05$) would be the corresponding concentration value from the distribution of PRGs (Bowers, 1999). The correspondence between the risk distribution and the PRG distribution is intuitive—just as selecting a higher percentile on the risk distribution is more protective, a lower percentile on the PRG distribution is more protective. The RME range for the risk distribution 90th to 99.9th percentile is analogous to an RME range for the PRG distribution of 0.1st to 10th percentile.

Backcalculation has been a familiar method of developing PRGs and may be appropriate in some situations for the sake of clarity and transparency due to the general understanding of this method among risk assessment practitioners. Once a backcalculation has been performed to determine a PRG, the PRG should be used as the concentration term in a forward calculation to ensure that the risk at the PRG is acceptable.

5.4.1 DIFFICULTIES WITH BACKCALCULATION

There are limitations in the use of backcalculation in PRA (Ferson, 1996). Simple rearrangement of Equation 5-1 does not suffice when the variable (i.e., the concentration or risk term) that is backcalculated is represented by a probability distribution (Burmester et al., 1995; Ferson, 1996). The difficulty for PRA arises because each risk estimate from an MCA that uses the familiar “forward-facing” risk equation represents a combination of random values selected from the input distributions. Therefore, the output can be considered conditional on all of the inputs. Rearranging the risk equation does not maintain the same conditional probabilities; therefore, the distribution for risk estimated as a function of the distribution for concentration in Equation 5-1 does not return the same distribution for concentration when applied in Equation 5-2. While there are techniques that can maintain the dependencies and correlations between exposure factors when the risk equation is rearranged (e.g., deconvolution), they are complex and beyond the scope of this guidance.

Backcalculation methods may also be difficult to implement in situations in which complex fate-and-transport considerations are present. Leaching of soil contamination to groundwater, bioconcentration of chemicals at higher trophic levels, and other multimedia processes that result in exposure via several environmental media are situations in which backcalculation may not be useful. Note that these difficulties are not unique to backcalculation. Uncertainty in fate-and-transport considerations makes any type of PRG determination challenging.

Further, the backcalculation approach only provides information on the EPC that corresponds to a risk level of concern; it does not specify an RAL that would achieve this EPC. For example, when a risk equation is algebraically solved for concentration (see Equation 5-2), a PRG is developed without a corresponding RAL. Thus, there is no information associated with the PRG value to indicate the highest concentration in the EU that must be removed so that the average concentration (or 95% UCL) within the EU is at or below the PRG. Hence, additional efforts are needed. In addition, post-remediation concentrations may need to satisfy more than one regulatory constraint. For example, the average (or 95% UCL) concentration within an EU may need to be less than a concentration associated with chronic toxicity or cancer and simultaneously, the RAL concentration may need to be less than a concentration that might cause acute toxicity.

In spite of these caveats, backcalculation methods may be appropriate for some sites. For example, when the target risk is specified by a single numerical value and the risk manager has chosen a percentile of variability to represent the RME individual, then a backcalculated PRG can be derived from a PRA.

Although backcalculation methods may be appropriate for some sites, risk assessors should be familiar with their limitations. Because of these limitations, this guidance recommends iterative forward calculations as the primary method for calculating PRGs when performing a PRA. Iterative methods avoid difficulties associated with applying MCA to a backcalculation, and can provide more information for the risk manager.

5.5 ITERATIVE METHODS

Iterative methods simply involve calculating risk with the “forward-facing” equation (see Equation 5-1) a number of times (iteratively) using progressively lower values for the concentration term until the risk is sufficiently protective. This iterative method has also been called the “repeated runs” method. Note that iterative methods for calculating a PRG are not uniquely applicable to PRA. Iterative methods also may be used to develop PRGs in point estimate risk assessments.

EPA recommends iterative simulations as a general approach for calculating PRGs from probabilistic risk assessments.

Most often, iterative forward calculations are performed using a systematic trial-and-error method until the percentile of variability in risk chosen to represent the RME individual is at or below acceptable risk levels. Sometimes, a short cut can be used to reduce the number of simulations needed with the trial-and-error method. If successive “guesses” of the EPC are plotted with the corresponding risk estimate, the exact solution can be determined from the best-fit line, thereby significantly reducing the effort required to implement this method. An example is given in Figure 5-1. For many risk equations, the relationship between the EPC and the RME risk will be approximately linear. Nevertheless, the final estimate of the EPC should be checked by running another simulation for risk with this estimate.

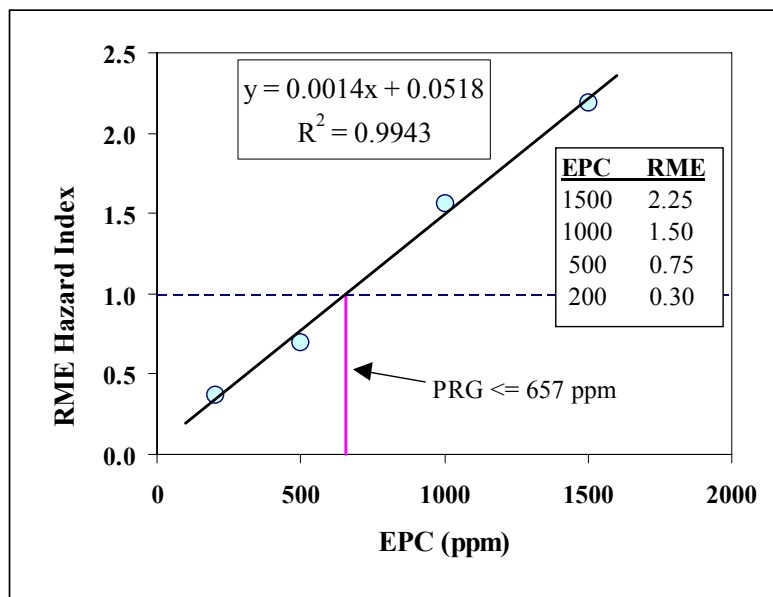


Figure 5-1. A hypothetical example of the use of iterative methods to determine the EPC that corresponds with a target RME Hazard Index (HI) of 1.0. Assume that the EPC is represented by the 95% UCL and the RME HI is the 95th percentile of the output distribution. In this case, four separate Monte Carlo simulations were run with iteratively decreasing values for the EPC. The least-squares, best-fit line to these four data points suggests that a reasonable PRG would be approximately 660 ppm.

A possible and significant advantage of iterative forward calculations over back-calculations is that the method is intuitive and yields a distribution of risks rather than a distribution of PRGs (as with a back-calculation method). The distribution of risks will be more familiar to the public and other stakeholders, and thus, both the method and the resulting output may be easier to communicate to senior level managers and stakeholders (see Chapter 6).

Two general types of iterative methods are described in more detail in Sections 5.5.1 and 5.5.2. The main difference between the methods is in the interpretation of the concentration term that is being reduced. With iterative reduction, the concentration is assumed to be the post-remediation EPC, whereas with iterative truncation, it represents the RAL needed to achieve a post-remediation EPC.

5.5.1 ITERATIVE REDUCTION

Iterative reduction can be applied to any medium. Generally, a point estimate representing the EPC (e.g., 95% UCL) is successively lowered, each time repeating the Monte Carlo simulation of variability in risk. When the EPC is reduced until the endpoint of concern (e.g., RME risk corresponding to the 95th percentile) is at or below an acceptable level of risk, the PRG is set at the corresponding EPC. The goal is to identify the point estimate that corresponds to a target risk level. Note that the PRG is not the same as the RAL. The RAL is the maximum concentration that may be left in place within an EU to achieve the PRG.

The concentration at which the risk is acceptable defines the PRG. Therefore, the PRG bears the same uncertainties as the EPC. For example, assume that a risk assessor examined the carcinogenic effects from

chronic consumption of a chemical in groundwater, then the exposure unit may be determined by the long-term average concentration at any well that potentially draws drinking water from the contaminated groundwater. Uncertainty in the long-term average concentration can reflect a number of factors that contribute to spatial and temporal variability, including the direction of groundwater flow, natural attenuation, and other fate and transport variables. Remediation by a pump-and-treat system for a prolonged period of time may be used to lower the concentrations at the wells. Even though the remediation strategy may be complicated by spatial and temporal variability, iterative reduction can be used to establish a PRG. A remediation strategy may be considered a potential candidate if it can achieve the PRG by reducing the average concentration at each of the well locations. The concept of “hot-spot” removal, or truncation of the highest concentrations first, would not be an option under this scenario (see Section 5.5.2).

5.5.2 ITERATIVE TRUNCATION

Iterative truncation is a method of calculating a PRG that involves developing an expression for the concentration term in which higher values of concentration are removed or “truncated” to reduce the maximum concentration. These higher values are replaced by the surrogate nondetect value. The risk is recalculated for each successive reduction in the highest value. The method is repeated with consecutively lower truncation limits until risk is acceptable.

Iterative truncation is most applicable to surface soil cleanup as the spatial variability over time is minimal compared to other media (e.g., surface water). With each iteration of the risk equation (e.g., Equation 5-1), the highest concentration value is truncated corresponding to a different RAL. In this way a “not-to-exceed” level is specified and the PRG is recalculated the same way in each iteration. The process continues until the risk distribution yields risk estimates at or below the level of concern.

Iterative truncation can be applied to either the empirical distribution function (EDF) for the concentration term, or a fitted distribution for variability in concentrations within the EU. Applied to the EDF, the maximum detected concentration within the EU is replaced with a surrogate value for a nondetect (e.g., half the reporting limit or the background value for some chemicals), and the EPC (e.g., 95% UCL) is recalculated for this altered data set. If this new EPC yields unacceptable risk, then the two highest detected concentrations are replaced by the nondetect value and the EPC is recalculated. In the third iteration, the three highest detections are replaced, and so on, until the target risk level is achieved. Alternatively, the sample data may be fit to a probability distribution for variability, and the process would be repeated with decreasing values in the high-end tail of the continuous distribution.

When the concentration term is a distribution representing uncertainty in the mean concentration, then, similar to the recalculation of the point estimate 95% UCL described above, this distribution of uncertainty in the mean concentration should be determined anew each time a datum is replaced with the nondetect value.

When a distribution of variability in concentration is used for the EPC, for example, in an ecological risk assessment where sampling may be sparse relative to the foraging area of a small home range receptor (see Appendix C), then the distribution developed in an identical way with the high values replaced by the surrogate nondetect value should be used in the iterative determination of a PRG.

The decision to apply iterative truncation should be made after considering a variety of characteristics of the sample data and post-remediation scenario (see Exhibit 5-3). For example, small sample size may result in high uncertainty in the 95% UCL, thereby limiting the use of iterative truncation. Quantitative criteria regarding these factors are not provided in this guidance given that the level of certainty required for decision making will

vary on a case-by-case basis. Use of geostatistical methods (Appendices C and D) may aid in interpreting site data or improving sampling design. Geostatistics is capable of describing the spatial distribution of a contaminant in a quantitative fashion. These methods establish a correspondence between the actual sampling locations and the locations a receptor would be expected to frequent. Additionally, it enables the estimation of concentrations in unsampled locations. Hence, for determination of concentrations at specific locations at a site or within EUs of various sizes and shapes, geostatistics may provide an invaluable tool. Geostatistics has applications both to developing the EPC and PRG and has been recommended and used at some sites for characterization of soil and groundwater contamination (U.S. EPA, 1990b, 1991c).

Although the consideration and use of geostatistics is encouraged, a full consideration of geostatistics is beyond the scope of this guidance. Those interested in greater detail than provided in Appendices C and D are urged to consult the Superfund guidance document currently under development, *Draft Guidance on Surface Soil Cleanup at Superfund Sites: Applying Cleanup Levels* (U.S. EPA, 2001b), for additional discussion of how geostatistics can be used to quantify the concentration term or the PRG.

Generally, iterative truncation methods fail to produce adequate cleanup strategies when site characterization is incomplete. This problem, however, is not specific to PRA. Both point estimate and probabilistic methods are sensitive to poor site characterization.

Risk assessors should realize that application of iterative truncation may result in areas on-site that have concentrations higher than the PRG. This is because the PRG will reflect an average concentration (or 95% UCL) from a distribution of concentrations in which the maximum is truncated at the RAL. For example, Figure 5-3 (see Section 5.5.3) shows how the concentration distribution can be truncated at an RAL, while still leaving behind concentrations greater than the PRG.

5.5.3 EXAMPLE OF ITERATIVE METHODS

The iterative truncation method is easiest to think about with regard to soil cleanup when contaminated soil is removed and replaced with clean fill soil. This replacement would reduce both the mean and 95% UCL. In most cases, risk assessors may assume that the concentrations of chemicals in clean fill soil can be represented by the surrogate nondetect value (e.g., half the detection limit). Alternatively, the fill may be sampled so that the measured concentrations in the fill dirt may be used to calculate the post-remediation

EXHIBIT 5-3

CRITERIA FOR ITERATIVE TRUNCATION

1. **Sample size (n) is sufficient.** Small sample sizes lead to large estimates of uncertainty in the concentration term. Small sample size may cause the risk assessor to overlook some sources of uncertainty.
2. **Concentration distribution is not highly skewed.** A highly skewed distribution may yield unreliable estimates of uncertainty, especially for small sample sizes.
3. **Sampling design yields a representative distribution of measurements within the exposure unit.** Simple random sampling may fail to represent a patchy spatial distribution of contaminants. Similarly, hotspot (e.g., cluster) sampling may fail to represent random movement of receptors. To evaluate potential biases in sampling, analyses with both standard statistical methods and geostatistical methods may be required.
4. **Assumptions about the post-remedial distribution of concentration are reasonable.** If these assumptions are shown to be incorrect by subsequent sampling events, the process for developing a PRG may need to be repeated and additional remedial activities may be required.

concentration term. Generally, metals and other inorganic chemicals will be present in clean fill, albeit at lower concentrations than on site.

A simple example using the 95% UCL as a point estimate for the EPC is given in Exhibit 5-4. In this example, background concentrations of chemical X were very low and hence, the fill was assumed to have a concentration of half the detection limit. The risk management objective is to identify a PRG in which the 95th percentile risk estimate is below 1E-04 and to determine the RAL necessary to achieve this PRG. This example illustrates how iterative truncation is applied to the empirical distribution function, rather than fitting the concentrations to a parametric distribution.

Assume that iterative reduction of the 95% UCL demonstrated that a post-remediation EPC of no greater than 33 mg/kg is needed to achieve a RME risk of 1E-04. What is the RAL that yields this EPC? The risk assessor recognizes that the post-remediation concentration distribution is very often a mixed distribution, consisting of a group of nondetect values and a truncated parametric distribution. Because of the complex nature of mixed distributions (Roeder, 1994), non-parametric methods for calculating the 95% UCL of the arithmetic mean (e.g., bootstrap resampling) were determined to be appropriate (U.S. EPA, 1997; Section 5.1.3).

EXHIBIT 5-4

EXAMPLE OF ITERATIVE METHODS

Scoping and Problem Formulation

Chromium contamination was present at a 12-acre industrial facility. In scoping and problem formulation, all stakeholders agreed that the facility would maintain itself and the current land use would continue into the foreseeable future. Most of the facility area was maintained as green space and as a buffer with the surrounding community. Surrounding the facility to the fence line were lawns and ornamental shrubs tended by landscape workers. These landscape workers were considered to be the high risk group as they would move freely and randomly over the entire area of the facility outside the buildings. Hence, the landscape workers would be exposed to an average concentration over the entire area of the facility outside the buildings. The management of the facility was very cooperative and concerned about their workers. Nonetheless, the facility management did not wish to bear more cost than necessary.

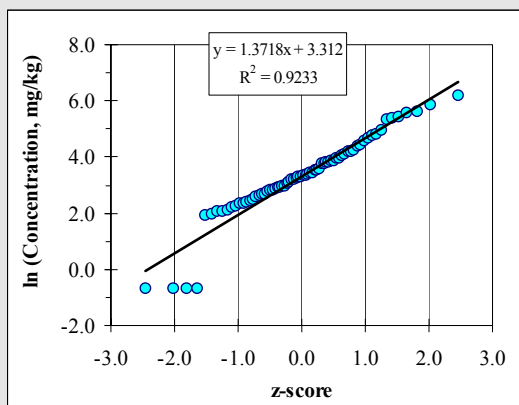
Site Characterization - Soil Sample (n=70)

Seventy surface soil samples were obtained using a sampling grid placed over all 12 acres. Five or six sampling locations were placed in each acre. None of the samples was composited. The grid-based sampling permits a rough estimation of the percentage of the site that would need active remediation. The detection limit for the chromium was 1 mg/kg. Four of the samples were nondetects. Sampling results are shown in Table 5-1. Although the samples from the site appeared to occur in a lognormal distribution (Figure 5-2), the presumed post-remediation distribution would be a mixed distribution, consisting of a truncated lognormal distribution and a group of data at the surrogate nondetect value.

Table 5-1. Soil sample (n=70) (mg/kg).

0.5	9.7	16.2	25.1	34.0	54.1	120.6
0.5	10.6	17.1	25.4	34.0	57.8	122.2
0.5	10.8	17.4	26.4	36.5	60.2	140.7
0.5	11.0	17.9	26.9	43.3	65.7	211.9
6.8	11.8	18.4	27.1	43.3	66.1	224.1
7.2	12.0	18.6	28.2	45.3	71.8	235.6
7.8	13.7	19.7	28.3	46.4	82.7	266.8
8.0	13.9	19.8	30.3	48.2	84.7	284.0
8.2	14.7	22.0	30.9	49.3	98.1	361.2
9.3	15.0	22.8	31.1	52.6	107.7	486.6

Figure 5-2. Lognormal probability plot of soil concentrations, including 4 nondetects.



In this example, a series of iterative truncations showed that removal of all sample results greater than 100 mg/kg (n=11) and replacement of these with the nondetect surrogate of 0.5 mg/kg yielded a 95% UCL of 33 mg/kg and RME risk below 1E-04. Table 5-2 summarizes the results of the calculations for the three conditions: (1) pre-remediation concentrations; (2) post-remediation concentrations using iterative truncation to achieve an RAL of 100 mg/kg; and (3) post-remediation concentrations assuming the 95% UCL calculated is used as the RAL. Note that if the PRG of 33 mg/kg was applied as a “not-to-exceed” level (i.e., RAL), the resulting remediation effort would increase from 15 to 40% of the site, yielding a 95% UCL of 14 mg/kg. While this would be a protective decision, other information was used to support the selection of the second scenario instead. A toxicologist was consulted, who indicated that acute exposure to the workers at levels of 100 mg/kg would not present a health risk. To build additional protectiveness into the remedy, the management also indicated scheduling for the landscape workers would be performed so the areas tended would be rotated among all the workers.

Table 5-2. Pre- and Post-Remediation EPCs (95% UCLs) for Chemical X in Surface Soil Samples.

Remediation Scenario	RAL (mg/kg)	EPC (mg/kg) 95% UCL	Percent of Site to be Remediated
1. Pre-remediation	NA	93	NA
2. Post-remediation using the PRG as the 95% UCL	100	33	15%
3. Post-remediation using the PRG as the RAL (i.e., “not-to-exceed”)	33	14	40%

NA=not applicable for a pre-remediation scenario.

Figure 5-3 shows a conceptual framework for considering the post-remediation distribution as a mixture between a group of nondetects and a distribution of contamination truncated at the RAL. Prior to remediation, the EPC exceeds a level that would be protective of human health and ecosystems. If the high-end soil concentrations are removed and the soil is replaced with clean fill, the resulting distribution will be bimodal, with one peak occurring at the nondetect concentration, and the second occurring near the mean of the post-remediation distribution.

5.5.4 MULTIPLE EXPOSURE UNITS AND ITERATIVE METHODS

When multiple EUs are present at the site, there may be a small number of samples within a given EU and the uncertainty in the concentration term generally will be large. It may be possible to use knowledge of the mechanism of how the contamination occurred along with spatial patterns in the sampling results in other nearby EUs to quantify uncertainty. Geostatistical techniques for estimating the mean concentration may provide useful insights into the importance of accounting for spatial relationships among the sample data. Appendix C also provides a discussion of the situation of multiple EUs within a larger site.

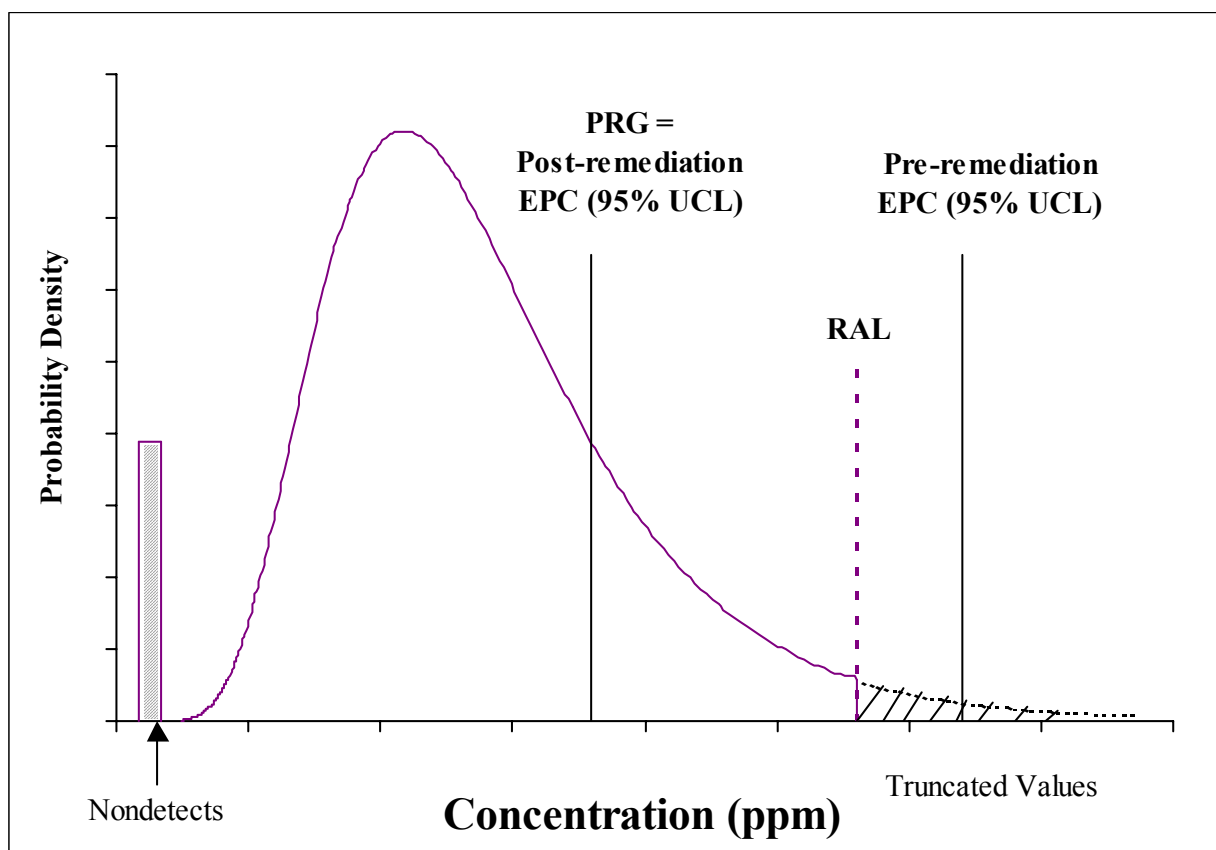


Figure 5-3. Hypothetical example of a mixed, bimodal distribution that represents a combination of the pre-remediation distribution truncated at the remediation action level (RAL) and a uniform distribution representing clean fill at the surrogate nondetect concentration. Shaded portions represent equal areas. In this example, the PRG is defined by the post-remediation EPC (95% UCL).

5.6 PRGs FOR GROUNDWATER

For some chemicals encountered at hazardous waste sites, chemical-specific ARARs may exist, and may be considered as PRGs. ARARs may be selected as site-specific cleanup levels. The maximum contaminant levels of the Safe Drinking Water Act are examples of ARARs.

☞ *For groundwater contamination, ARARs should be applied as RALs if they are protective.*

Of course, for cases in which an ARAR is less protective than a remediation goal determined from a risk assessment, then a risk-based PRG may be developed in accordance with the NCP (U.S. EPA, 1990a).

As an exposure medium, groundwater is the opposite of soil in that groundwater is not static, and receptors are usually exposed at one location (i.e., the well head). Often, a single well can be considered the EU when assessing risks associated with either the residential or industrial/occupational scenarios. The EPC may still reflect the concept of averaging over a long time period (e.g., years) due to potential changes in concentrations in

well water over time. For example, chemical fate and transport modeling may suggest that concentrations are decreasing over time. Similarly, there may be temporal and spatial variability depending on the seasonal fluctuations of the water table. Ideally, the risk assessment would focus on individuals who may be exposed at locations nearest to the center of the contaminant plume, where concentrations are likely to be highest (Freeze and Cherry, 1979; Sposito, et al., 1986).

Because of the uncertainty in the movement of groundwater and the necessity of sampling the medium at fixed locations, identifying a meaningful RAL needed to achieve a given PRG is difficult. In most cases, ARARs will be applicable as RALs or “not-to-exceed” levels.

5.7 PRGs FOR OTHER CONTAMINATED MEDIA

Iterative truncation techniques are generally applied to a static medium, such as soil, rather than dynamic or fluid media such as water and air. This is simply because it is difficult to design a method that will selectively remove high concentrations from a fluid medium. Iterative reduction may be more relevant than iterative truncation when an RAL cannot be developed. These issues are discussed below with respect to sediment, surface water, and fish.

Sediment

Sediment may be transported over time more readily than soils. If it can be assumed that the sediment remains in place, then iterative truncation techniques may be applied. However, at some sites, sediment may be considered a fluid medium. For example, sediment may be resuspended by the movement of water craft, waves, changing tides, or erosion. Similarly, the depth of the contaminated sediment may change over time as new layers of sediment are deposited above more contaminated sediment.

Exhibit 5-5 gives an example of the use of iterative truncation to evaluate alternative RALs for sediment of a lake contaminated by pesticide runoff. In this example, the RAL is related to both the ecological endpoint of concern (i.e., reduction in reproductive success of mammalian omnivores at the lake) and the fraction of areal extent of the lake that would require remediation at that RAL.

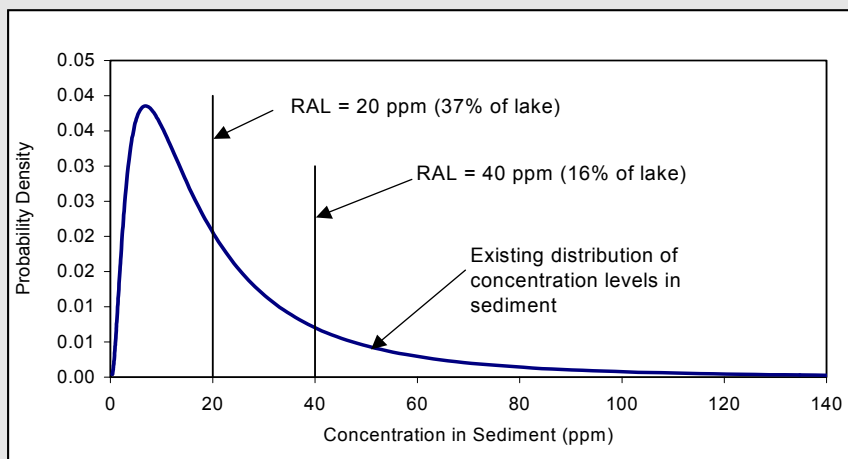
EXHIBIT 5-5

EVALUATION OF ALTERNATIVE RALS USING ITERATIVE TRUNCATION

Risks to a population of mammalian omnivores residing near a lake contaminated with pesticide "X" were judged to be sufficiently high that a reduction in population number over time was expected (see Chapter 4, Exhibit 4-12). The primary reservoir of pesticide X in the lake is sediment. The BTAG committee decided to use the iterative truncation method to estimate the beneficial effects of a series of different Remedial Action Levels (RALs). PRA was used to predict the distribution of responses (percent reduction in population success) and the areal extent of the lake requiring remediation as a function of RAL. The results are summarized below.

RAL in Sediment	Reduction in Reproductive Success			Fraction of Lake
	Mean	90th	95th	
None	8.9%	31%	59%	0%
100 ppm	7.6%	24%	48%	3%
80 ppm	7.0%	27%	45%	5%
60 ppm	5.9%	18%	36%	8%
40 ppm	4.4%	12%	26%	16%
20 ppm	1.9%	4.7%	10%	37%

The BTAG reviewed these results and concluded that while an RAL of 20 ppm would be needed to provide nearly complete protection of the exposed population, an RAL of 40 ppm would provide a good reduction in effect level while tending to minimize the areal extent of the lake that required remediation, which in turn would tend to minimize disturbance of the ecosystem during remediation. Based on this, the risk manager identified 40 ppm as the RAL and initiated a feasibility study to investigate ways of achieving this objective.



Biota (Fish, Aquatic Invertebrates, Plants)

Biota, such as fish, aquatic invertebrates, and plants can serve as bioindicators or indirect estimators of contamination in other exposure media that would be targets for remediation. The concentration of chemicals fish may reflect a combination of exposures via sediment, the water column, and food source (e.g., prey). Therefore, the use of bioindicators to develop PRGs in other media introduces a sources of uncertainty. If there is a high correlation between concentrations in fish and sediment, then sediment concentrations may be considered when developing PRGs to protect the receptor population. The EU, in this case, is the area where the angler population, or ecological predator population, harvests fish. However, in risk assessments that include a fish ingestion exposure pathway, there may be high uncertainty about the true concentration term. Concentrations may be affected by many factors, including changes in the fish population and changes in fish preferences, which may be difficult to address in risk assessments. The choice of fish species consumed by a given individual may also affect the concentration term.

Fish population studies and fate and transport considerations of the contaminants may indicate if and when a fish population will reach a calculated cleanup level. For many sites, it may be difficult to obtain this level of site-specific data due to resource and time constraints.

Although remediation may not immediately reduce contaminant concentrations in biota, the determination of a cleanup level can serve as a target for any future decline in concentrations. In general, iterative reduction methods are applicable for developing PRGs to protect aquatic ecosystems; however, under some conditions iterative truncation may also be used. For example, if contamination is correlated to relatively static sediment, and the home-range of the fish is relatively small (e.g., nonmigratory) then iterative truncation may be applicable.

Surface Water

The development of PRGs for surface water is also difficult with iterative truncation. For fluid media (e.g., groundwater or surface water), iterative reduction can be performed using a range of EPCs to determine a PRG with acceptable risk at the target RME percentile.

5.8 MEASUREMENT OF ATTAINMENT

The NCP (U.S. EPA, 1990a) provides for continued monitoring for groundwater cleanups to ensure attainment of the remedial action objectives. In addition, it is common practice among remedial project managers to conduct confirmation sampling after completing a remedy for soil contamination. However, completion of the remedial action according to this strategy does not necessarily mean that risks within EUs at the site have been reduced to levels specified in the ROD. The degree of uncertainty about whether the remedial action at the site has achieved the cleanup level should determine whether confirmation sampling is warranted. In general, confirmation sampling following cleanup activities is recommended. Sampling after the remedial investigation is complete may show additional areas needing remediation (i.e., where additional contamination exists).

If additional sampling is conducted after the remedial investigation, the concentration term and corresponding estimates of risk should be recalculated. The PRG developed in the remedial investigation may not be health-protective in light of the additional contamination. The same concepts that relate the concentration term to the PRG should be applied in this situation.

Confirmation sampling activities are included in remedial design/remedial action plans to ensure the remedy is successful. In addition, the five-year review presents a second opportunity to ensure that any contamination left on site does not pose an unacceptable risk.

☞ If confirmation sampling indicates an insufficient reduction in risk, a more extensive remediation effort may be needed. Possible reasons for not achieving remedial action objectives can include inadequate site characterization or the discovery of unknown contamination.

For post-remediation sampling, the DQO process should generally be followed. If the post-remediation risk associated with the confirmation sample indicates risk exceeds a level of concern, then additional remediation may be warranted.

5.9 SUMMARY OF RECOMMENDED METHODS

Table 5-3 summarizes the possible methods for developing PRGs for various environmental media. It should be noted that iterative reduction (IR) can be used in all cases, whereas iterative truncation (IT) is limited to situations where the highest concentrations can be identified and removed. Backcalculation may be applicable in all cases, but because of caveats noted in Section 5.4.1, iterative approaches are generally recommended in this document.

Table 5-3. Summary of Potential Methods for PRG Development by Environmental Medium.

Potential Exposure Medium	Back-calculation	Iterative Reduction (IR)	Iterative Truncation (IT)	Explanations for IT
Soil	X	X	X	Applicable if soil is relatively fixed.
Sediment	X	X	X	Applicable if sediment is relatively fixed. In some situations, sediment transport may be a better assumption due to current velocity, tides, resuspension, etc.
Biota (Fish, Aquatic Invertebrates, Plants) - bioindicators of contamination in sediment	X	X	SA	Depends on home-range of fish relative to the scale of the sampling design. If contamination is correlated to relatively static sediment, and the home-range of the fish is relatively small (e.g., non-migratory) then IT may be applicable.
Surface Water	X	X	NA	Not applicable as surface water is a fluid medium.
Groundwater (GW)	X	X	NA	Not applicable as GW is a fluid medium. Generally, ARARs must also be satisfied.
Home-grown produce, milk, livestock, other food items	X	X	SA	Depends on relative contributions of soil uptake (applicable) vs. foliar deposition (not applicable).

X=applicable
NA=not applicable
SA=sometimes applicable

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CHAPTER 6

COMMUNICATING RISKS AND UNCERTAINTIES IN PROBABILISTIC RISK ASSESSMENTS

6.0 INTRODUCTION

The Environmental Protection Agency (EPA) has developed a guidance document, *Risk Assessment Guidance for Superfund: Volume I—Human Health Evaluation Manual, Supplement to Part A: Community Involvement in Superfund Risk Assessments* (U.S. EPA, 1999a) and two videotapes, “*Superfund Risk Assessment and How You Can Help, An Overview*” (10 minutes) (U.S. EPA, 1999b) and “*Superfund Risk Assessment and How You Can Help*” (40 minutes) (U.S. EPA, 2000b), to improve community involvement in the Superfund risk assessment process. The videotapes (available in both English and Spanish) show examples of how regions have involved communities in the risk assessment process at several Superfund sites. The guidance document and videotapes, along with the *Superfund Community Involvement Handbook and Toolkit* (U.S. EPA, 1998), should serve as a primary community involvement resource for risk assessors and remedial project managers (RPMs). The *Handbook and Toolkit* offers the following specific guidance:

- Provides suggestions for how Superfund staff and community members can work together during the early stages of Superfund remedial investigation and feasibility study (RI/FS) and later cleanup
- Identifies where, within the framework of the human health risk assessment methodology, community input can augment and improve EPA’s estimates of exposure and risk.
- Recommends questions the site team (risk assessor, RPM, and community involvement coordinator [CIC]) should ask the community.
- Illustrates why community involvement is valuable during the human health risk assessment at Superfund sites.

This chapter provides guidance and suggestions on how to deal with risk communication issues that arise during a probabilistic risk assessment (PRA). Specifically, the concepts of uncertainty and variability may present additional communication challenges for PRA. For example, whereas discussions of uncertainty for point estimate risk assessments are often qualitative in nature, PRA opens the floor for discussion and presentation of quantitative uncertainty analysis. Concepts associated with quantitative characterizations of uncertainty may be more difficult to communicate and may not be well received due to stakeholder desires for certainty (Slovic et al., 1979). As such, this chapter highlights appropriate stakeholder involvement and principal risk communication skills that are effective for communicating PRA concepts and risk information. Key factors for successful communication of PRA include early and continuous involvement of stakeholders, a well-developed communication plan, good graphics, a working knowledge of the factors that may influence perceptions of risk and uncertainty, and a foundation of trust and credibility.

EXHIBIT 6-1

DEFINITIONS FOR CHAPTER 6

Central Tendency Exposure (CTE) - A risk descriptor representing the average or typical individual in a population, usually considered to be the mean or median of the distribution.

Community Advisory Group (CAG) - A group formed to provide a public forum for community members to present and discuss their needs and concerns related to the Superfund decision-making process. A CAG serves as the focal point for the exchange of information among the local community, EPA, State regulatory agency, and other pertinent Federal agencies involved in the cleanup of a Superfund site.

Community Involvement Coordinator (CIC) - As a member of the CAG and site team, the CIC coordinates communication plans (i.e., the CIP) and addresses site-specific CAG organizational issues.

Community Involvement Plan (CIP) - A plan that identifies community concerns and the preferences of the community for the communication of site-related issues.

Confidence Interval - A range of values that are likely to include a population parameter. Confidence intervals may describe a parameter of an input variable (e.g., mean ingestion rate) or output variable (e.g., 95th percentile risk). When used to characterize uncertainty in a risk estimate, it is assumed that methods used to quantify uncertainty in the model inputs are based on statistical principles such as sampling distributions or Bayesian approaches. For example, given a randomly sampled data set, a 95% confidence interval for the mean can be estimated by deriving a sampling distribution from a Student's t distribution.

Credible Interval - A range of values that represent plausible bounds on a population parameter. Credible intervals may describe a parameter of an input variable (e.g., mean ingestion rate) or output variable (e.g., 95th percentile risk). The term is introduced as an alternative to the term confidence interval when the methods used to quantify uncertainty are not based entirely on statistical principles such as sampling distributions or Bayesian approaches. For example, multiple estimates of an arithmetic mean may be available from different studies reported in the literature—using professional judgment, these estimates may support a decision to describe a range of possible values for the arithmetic mean.

Cumulative Distribution Function (CDF) - Obtained by integrating the PDF, gives the cumulative probability of occurrence for a random independent variable. Each value c of the function is the probability that a random observation x will be less than or equal to c .

Hazard Quotient (HQ) - The ratio of estimated site-specific exposure to a single chemical from a site over a specified period to the estimated daily exposure level, at which no adverse health effects are likely to occur.

Hazardous Substance Research Centers (HSRC) - Research centers providing free technical assistance to communities with environmental contamination programs through two distinct outreach programs: Technical Outreach Services for Communities (TOSC) and Technical Assistance to Brownfields Community (TAB).

Histogram - A graphing technique which groups the data into intervals and displays the count of the observations within each interval. It conveys the range of values and the relative frequency (or proportion of the sample) that was observed across that range.

Monte Carlo Analysis (MCA) or Monte Carlo Simulation - A technique for characterizing the uncertainty and variability in risk estimates by repeatedly sampling the probability distributions of the risk equation inputs and using these inputs to calculate a distribution of risk values. A set of iterations or calculations from Monte Carlo sampling is a simulation. For example, a single iteration for risk from ingestion of water may represent a hypothetical individual who drinks 2 L/day and weighs 65 kg; another iteration may represent a hypothetical individual who drinks 1 L/day and weighs 72 kg.

Parameter - A value that characterizes the distribution of a random variable. Parameters commonly characterize the location, scale, shape, or bounds of the distribution. For example, a truncated normal probability distribution may be defined by four parameters: arithmetic mean [location], standard deviation [scale], and min and max [bounds]. It is important to distinguish between a variable (e.g., ingestion rate) and a parameter (e.g., arithmetic mean ingestion rate).

Percentile - A number in a distribution such that X % of the values are less than the number and $1-X$ % are greater. For example, the 95th percentile is a number in a distribution such that 95% of the values are less than the number and 5% are greater.

EXHIBIT 6-1

DEFINITIONS FOR CHAPTER 6—Continued

Point Estimate Risk Assessment - A risk assessment in which a point estimate of risk is calculated from a set of point estimates for exposure and toxicity. Such point estimates of risk can reflect the CTE or RME, depending on the choice of inputs.

Potentially Responsible Party (PRP) - Individuals, companies, or any other party that is potentially liable for Superfund cleanup costs.

Preliminary Remediation Goal (PRG) - Initially developed chemical concentration for an environmental medium that is expected to be protective of human health and ecosystems. PRGs may be developed based on applicable or relevant and appropriate requirements (ARARs), or exposure scenarios evaluated prior to or as a result of the baseline risk assessment. (U.S. EPA, 1991a, 1991b).

Probabilistic Risk Assessment (PRA) - A risk assessment that yields a probability distribution for risk, generally by assigning a probability distribution to represent variability or uncertainty in one or more inputs to the risk equation.

Probability Density Function (PDF) - A function or graph representing the probability distribution of a continuous random variable. The density at a point refers to the probability that the variable will have a value in a narrow range about that point.

Rank Correlation (Spearman Rank Order Correlation Coefficient) - A “distribution free” or nonparametric statistic r that measures the strength and direction of association between the ranks of the values (not the values themselves) of two quantitative variables.

Reasonable Maximum Exposure (RME) - The highest exposure that is reasonably expected to occur at a site (U.S. EPA, 1989). The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures.

Remedial Investigation/Feasibility Study (RI/FS) - Studies undertaken by EPA to delineate the nature and extent of contamination, to evaluate potential risk, and to develop alternatives for cleanup.

Sensitivity Analysis - Sensitivity generally refers to the variation in output of a model with respect to changes in the values of the model’s input(s). Sensitivity analysis can provide a quantitative ranking of the model inputs based on their relative contributions to model output variability and uncertainty. Common metrics of sensitivity include:

- ▶ Pearson Correlation Coefficient - A statistic r that measures the strength and direction of linear association between the values of two quantitative variables. The square of the coefficient (r^2) is the fraction of the variance of one variable that is explained by the variance of the second variable.
- ▶ Sensitivity Ratio - Ratio of the change in model output per unit change in an input variable; also called *elasticity*.
- ▶ Spearman Rank Order Correlation Coefficient - A “distribution free” or nonparametric statistic r that measures the strength and direction of association between the ranks of the values (not the values themselves) of two quantitative variables. See Pearson (above) for r^2 .

Stakeholder - Any individual or group who has an interest in or may be affected by EPA’s site decision-making process.

Technical Assistance Grant (TAG) - A federal grant that is intended to provide a community with the opportunity to hire independent experts to help evaluate and explain the results of a risk assessment.

Technical Outreach Services for Communities (TOSC) - A service of the HSRC with the aim to provide independent technical information and assistance to help communities with hazardous substance pollution problems.

Uncertainty - Lack of knowledge about specific variables, parameters, models, or other factors. Examples include limited data regarding the concentration of a contaminant in an environmental medium and lack of information on local fish consumption practices. Uncertainty may be reduced through further study.

Variable - A quantity that can assume many values.

Section 6.1 discusses the need for early and continuing stakeholder involvement. Section 6.2 recommends a seven-step process for communicating PRA results to stakeholders, and Sections 6.3 and 6.4 provide guidance on specific techniques for communicating information. The success of risk communication efforts will depend on the extent to which the communication strategy addresses the needs of a diverse audience, with different perceptions of risk and uncertainty (Section 6.5), and the degree of trust and credibility that is established from the outset of the process (Section 6.6). Section 6.7 provides a discussion of risk communication issues that are uniquely relevant to RPMs.

6.1 STAKEHOLDER INVOLVEMENT

Many stakeholders may be interested in a risk assessment (see Exhibit 6-2). It is generally important to *involve and engage interested stakeholders early and continuously* throughout the decision-making process (U.S. EPA, 2001).

Public involvement activities should be tailored to the needs of the community and described in the site communications strategy. The CIC should coordinate these first steps through the development of a Community Involvement Plan (CIP). Coordination between the RPM, risk assessor, and CIC is needed to determine the appropriate points in the RI/FS process to communicate with the community, and plan for the appropriate level of communication. The CIP should identify community concerns and the preferences of the community for the communication of site-related issues. The CIP may be updated during the RI/FS as needed.

EXHIBIT 6-2

STAKEHOLDERS POTENTIALLY INVOLVED IN THE DECISION-MAKING PROCESS FOR PRA

- EPA risk assessors and managers
- Members of the public
- Representatives from state or county environmental or health agencies
- Other federal agencies (e.g., health agencies, Natural Resources Damage Assessment (NRDA), trustees, etc.)
- Tribal government representatives
- Potentially responsible parties (PRPs) and their representatives
- Representatives from federal facilities (e.g., Department of Defense, Department of Energy, etc.)

Examples of outreach activities include giving oral presentations and poster sessions at public meetings, coordinating group meetings or focused workshops, conducting interviews with community members on specific issues, and distributing fact sheets.

Ideally, the public and other interested stakeholders would be involved early in the site-specific decision-making process. If the community has not been previously involved, efforts should be made, in coordination with the CIC, to identify and communicate with the appropriate individuals in the community prior to the Agency's receipt of the PRA workplan. The public and other stakeholders should be given the opportunity to provide input to the workplan for a PRA (see Chapter 2, Section 2.1).

The initial community meeting can serve to establish a rapport between EPA and the community and facilitate the exchange of information needed to support a PRA. This information may include policy decisions associated with both point estimate and probabilistic approaches, as well as technical details regarding the conceptual exposure model and the selection of distributions. A discussion of these topics may increase certainty about the assumptions made in the risk assessment. For example, the community may be able to offer insights regarding site-specific activities and sources of exposure data not readily

available to the risk assessor. This type of discussion should allow for the free exchange of information with the public and sets the stage for future discussions. It is important that an appropriate level of detail be presented at the first meeting. Instead of overloading the audience with information, it is generally better to coordinate several meetings so that complex policy and technical concepts can be broken down into smaller discussion topics.

Following the approval of the PRA workplan, the public and other interested stakeholders should be involved in various stages of the PRA development, including providing and/or reviewing data, reviewing the selected distributions (e.g., selected creel survey) and commenting on PRA documents as appropriate during public comment periods. On-going community involvement may require consideration of EPA's resources including the availability of personnel and contractor support. Other considerations include EPA's compliance with provision in the National Contingency Plan (NCP) for involving the community. The appropriate level of community involvement in the PRA should be based on a number of factors including the nature and extent of contamination at the site, the expressed interests of the community members, the complexity of the PRA, and the role of PRA in site-specific remediation or cleanup decisions.

6.2 COMMUNICATION AND PRESENTATION

Communication is a two-way process that should involve the transfer of information between the Agency and the stakeholders, as well as active listening by the Agency to the stakeholder's ideas and concerns. The goals of risk communication are to present risk information in an understandable manner through an open, honest, frank, and transparent presentation and discussion of risks, including uncertainties. In meeting these goals, it is important that the RPMs and risk assessors be sincere and direct in their presentation of the results of the PRA, accept the public and other interested stakeholders as valuable contributors to the process, and listen to the concerns and ideas that are raised.

One goal of communication should be to respect the stakeholder's concerns. The public and other interested stakeholders should have the opportunity to understand the PRA and its effects on the decision-making process. Technical Assistance Grants (TAGs) may be one way to advance this goal by providing the community the opportunity to hire independent experts to help evaluate and explain the results of the PRA. Alternatively, the RPM and risk assessor may use the tools outlined in Sections 6.3 to 6.6 to present PRA concepts and the results of the PRA to the community in a manner that is easily understood. This may require significant up-front planning, testing, and post-evaluation to identify the appropriate messages to communicate and to determine how well this information was communicated.

The site-specific PRA communication plan should be consistent with the NCP's provisions on community involvement. It is important to recognize that community involvement is part of a regulatory process and that EPA generally will consider all timely public input, but may not implement all of it. Ultimately, EPA must meet the legal requirements of the Superfund law in making decisions regarding remedial actions.

A vast body of literature exists regarding risk communication. Since the early 1980's, a number of researchers have developed models for communicating risk to the public. These models are available in the scientific literature, and a list of supplemental references is provided at the end of this chapter.

6.2.1 COMMUNICATION OF PRA WITH CONCERNED CITIZENS, OTHER STAKEHOLDERS, AND MANAGERS: AN OVERVIEW

Before the decision to conduct a PRA is made, a CIP should be in place. Generally, when a decision is made to conduct a PRA, an important step should be to work with citizens to develop a communication strategy for PRA and its application within the Superfund process (see Chapter 1). The initial introduction of the community to the RI/FS process should include a discussion of the principles of risk assessment. This discussion may be best presented in an informal setting such as a public availability session. Because of the potentially complex nature of PRA and quantitative uncertainty analysis, a small group meeting may be an appropriate forum in which to discuss issues and facilitate an exchange of ideas. If there is interest among a large group of stakeholders, multiple small group sessions may be scheduled. Such meetings may provide the foundation for building trust and credibility (see Section 6.6).

In general, it is important to identify whether a Community Advisory Group (CAG) should be formed. The purpose of a CAG is to provide a public forum for community members to present and discuss their needs and concerns related to the Superfund decision-making process. The CIC is an important member of the team and may coordinate communication plans, hand-out materials, and address site-specific organizational issues.

A number of resources may be available to the community to aid in understanding technical material in a PRA. In addition to the TAG program, which provides funds for qualified citizens' groups affected by a Superfund site to hire independent technical advisors, another program is the Technical Outreach Services for Communities (TOSC), which uses university educational and technical resources to help communities understand the technical issues involved in hazardous waste sites in their communities. This is a no-cost, non-advocate, technical assistance program supported by the Hazardous Substance Research Centers.

The tiered approach for PRA presented in Chapter 2 (Figures 2-1 and 2-2) encourages risk assessors and RPMs to participate in discussions with stakeholders early in the process of developing point estimate and probabilistic approaches. If a decision is made to perform a PRA, a continuing dialogue should be useful to evaluate interim results of the PRA and determine if additional activities are warranted (e.g., data collection, further modeling). These on-going discussions should help assure that RPMs are aware of the details of the PRA analysis and are comfortable with the material that will be shared with the community, other interested stakeholders, and senior managers.

6.2.2 STEPS FOR COMMUNICATION OF THE RESULTS OF THE PRA

The complexity of a PRA will vary depending on the site-specific nature of the assessment performed. For example, PRAs may include an analysis of variability, uncertainty, or both. Some analyses may involve simulations to evaluate temporal variability (e.g., Microexposure Event analysis) and spatial variability (e.g., geostatistics). The challenge for presenters is to *identify the critical information and level of detail to be presented to various audiences that may be involved in the Superfund decision-making process* (e.g., senior risk managers, concerned citizens, congressional staff, and PRPs).

The 7-step process, described below (and summarized in Exhibit 6-3), may be repeated many times during the performance of a PRA. For communication purposes, a PRA normally will involve more interaction with stakeholders than a point estimate risk assessment because PRA concepts and results are often more difficult to communicate.

(1) *Identify the Audience*

The first step should be to identify the audience of potentially interested stakeholders. Strategies for presenting PRA information normally will be tailored to the audience. Participants in the audience may change during the tiered process depending on the complexity of the PRA (see Chapter 2) and the specific site-management decisions being made.

(2) *Identify the Needs of the Audience*

The second step should be to identify the needs of the audience. The relevant information and the appropriate level of detail will vary depending on the audience. For example, some participants may be well informed about PRA concepts and will not need much introductory PRA information. For other audiences, PRA concepts may be new, so it may be beneficial to hold an informal meeting to discuss the general objectives and methods used to conduct a PRA. Once introductory PRA concepts have been discussed and are understood by the audience, more advanced discussions may be warranted on topics such as the sources of data used in the PRA, the most critical variables in the PRA (identified during the sensitivity analysis), the selection of distributions, and the level of characterization of uncertainty (see also Section 6.5). The risk assessor should select the key information for each topic and discuss the significance of this information based on the intended audience.

EXHIBIT 6-3

IMPORTANT STEPS FOR COMMUNICATING PRA RESULTS

- (1) Identify the audience
- (2) Identify the needs of the audience
- (3) Develop a communication plan
- (4) Practice to assure clarity of presentation
- (5) Present information
- (6) Post-meeting review of presentation and community feedback
- (7) Update information as needed for future assessments and presentations

(3) Develop a Communication Plan

The third step should be to develop a plan to communicate significant information to the public in an easily understandable format (Exhibit 6-4). Adequate planning in the presentation of PRA information is essential. A thorough understanding of the design and results of the PRA will help to place the information in proper context and understandable format (U.S. EPA, 1994). Even more importantly, the risk assessors and RPMs should clearly identify the main messages to be presented.

EXHIBIT 6-4

KEY CONSIDERATIONS IN DEVELOPING UNDERSTANDABLE MATERIAL

- Identify main messages
- Place information in appropriate context
- Use clear formats
- Use examples and graphs
- Provide handouts and glossaries
- Present information with minimum jargon

Section 6.4 provides examples of graphics that may be useful in presentations of PRA. Handouts, glossaries, and other materials may complement a presentation and provide information for discussion following the meetings. In addition, examples designed to help demonstrate concepts unique to PRA (e.g., using one probability distribution to describe variability and a second distribution to describe parameter uncertainty) may help facilitate the flow of communication and increase the level of understanding. One useful technique in public meetings is to involve members of the audience to illustrate a concept. For example, the topic of discussion may be the method used to select and fit a probability distribution used to characterize variability in a PRA. To demonstrate this concept, a risk assessor can draw a bell-shaped curve on a flip chart and label the *x-axis*, “number of liters of water consumed per day”, and the *y-axis*, “number of people who consume a specific amount of water in a day”. Next, each meeting participant can be asked to identify their own consumption pattern, perhaps by holding up a 0.5 liter bottle and asking how many such bottles are consumed on an average day. This community-specific information can then be plotted on a new graph in the form of a histogram and the bars can be connected to form a curve or distribution similar to the one first drawn. The resulting distribution (for an example, see Figure 6-1) can then be used to discuss the following PRA concepts in more detail:

- Variability (between individuals)
- Shape of the distribution and plausible range of values
- Central tendency exposure (CTE) and reasonable maximum exposure (RME) estimation
- Uncertainty in the distribution (sample size, potential response bias, differences in activity patterns)
- Uncertainty in a parameter estimate (difference between the 95% upper confidence limit (UCL) for a mean and the 95th percentile)

Using this information as a basis, the risk assessor can compare the results from the community analysis with data from various geographic areas in the U.S. where water consumption patterns may differ. The risk assessor can then lead a discussion with the community regarding the various sources of uncertainty in selecting and fitting exposure distributions, including:

- (a) **Extent of Representation** - Are the available data representative of the target population? For example, would the data on water consumption collected during the meeting be representative for various population groups?

- (b) **Data Quantity** - What sample size is needed to develop a distribution? This discussion will introduce the concept that uncertainty in both point estimates and probability distributions may be reduced by increasing the sample size
- (c) **Data Quality** - Are the data collected using acceptable study protocols? Is the information available from the peer-reviewed literature? An example can be made of the data collected during the meeting to highlight issues associated with survey design, and methods for controlling for potential bias or error. For example, if the survey data were to be used in a risk assessment for a drinking water scenario, the data quality may be improved by repeat sampling over time

Other exposure variables that can be used in this distribution example include: fish consumption rates, chemical concentrations in soil, and fraction of time spent indoors. In general, examples should focus on variables that may be of interest, are easily illustrated, and are unlikely to make participants uncomfortable divulging personal information such as age.

(4) *Practice to Assure Clarity of Presentation*

The fourth step should be to practice the presentation to assure that the information is presented clearly to the intended audience. Staff from communication groups or public information offices within EPA regional offices may help to determine whether or not the presentation addresses the needs of various audiences. Also, practicing the presentation with co-workers who are unfamiliar with the site can help assure that the appropriate messages are being conveyed, and will help the team prepare for potential questions that will arise during the meeting.

(5) *Present Information*

A number of factors should be considered when developing a plan to present the PRA in a meeting. Although the size of the public meeting can sometimes be unpredictable, typically individuals will feel more comfortable asking questions and expressing opinions in small, informal settings. For any audience, it is usually helpful to have general fact sheets on PRA available for distribution. The fact sheets may contain information that describes the PRA process, how information from the PRA will be used at the site, and how the community may comment on the PRA report. The meeting team should usually include the CIC, RPM, Risk Assessor, and additional support as necessary.

Audio-visual materials and equipment should be checked prior to the start of the meeting. For example, overheads should be viewed from the audience seating to assure that information is accessible and readable. Presentations using portable computers can be effective for showing how the results of the PRA may differ with changes in modeling assumptions.

(6) *Post-meeting Review of Presentation and Community Feedback*

At the end of a meeting, it can be helpful to encourage participants to provide feedback regarding effective and ineffective communication techniques. Not only can this information be used to improve presentations offered to similar audiences in the future, it also provides a sense for how well the main messages and specific technical issues were communicated.

(7) *Update Information as Needed for Future Assessments and Presentations*

Shortly after the meeting or briefing, modifications should be made to the materials for future presentations where appropriate. In addition, if information is obtained that is relevant to the risk assessment, this information may be included in a subsequent analysis, and the process would be repeated.

6.3 COMMUNICATING DIFFERENCES BETWEEN POINT ESTIMATE AND PRA

One method for effectively explaining the PRA approach to quantifying variability and uncertainty is to employ comparisons to the more easily understood point estimate methodology. These comparisons can focus on either the inputs or the outputs associated with the two approaches. The communicator may focus on a specific input variable, such as drinking water intake, and explain that with the point estimate methodology, a single average or high-end value (e.g., 2 liters per day for adults) normally is used to quantify exposure, whereas with PRA, a probability distribution (e.g., lognormal) is used to characterize variability in exposure among a population. In addition, the outcomes (e.g., cancer risk estimates) can be compared by showing where the point estimate(s) of risk fall within the distribution of risks generated with PRA.

When communicating results from point estimate and PRA models, an important concept to keep in mind is that both methods yield risk estimates with varying degrees of uncertainty. Continuing with the above example, concepts associated with uncertainty (e.g., representativeness, data quantity, and data quality) can be introduced by asking the audience if their estimate of water consumption on a specific day would be equal to their average daily consumption rate over a 1-year period. This example highlights a common source of uncertainty in exposure data (i.e., using short-term survey data to estimate long-term behavior). Section 6.5 discusses different perceptions of uncertainty.

It is common to accept output from quantitative models without fully understanding or appreciating the corresponding uncertainties and underlying assumptions. One challenge in presenting PRA results is to determine the most effective way to communicate sources of uncertainty without undermining the credibility of the assessment (see Section 6.6). For example, it may be counterintuitive that the more sources of uncertainty that are accounted for in a PRA, the wider the confidence intervals tend to be in the risk estimates (see Section 6.4.2). The audience may question the utility of a method that appears to introduce more complexity in a risk management decision. It may be useful to point out that many sources of uncertainty are present, and methods available to acknowledge and quantify them may differ in point estimate and probabilistic risk assessments.

The basic concepts of PRAs described in Chapter 1 may be used in developing presentations. Exhibits 1-5 and 1-6 in Chapter 1 summarize some of the advantages and disadvantages of point estimates and probabilistic approaches that should be considered when evaluating differences in the risk estimates of the two approaches. For example, point estimates of risk do not specify the proportion of the population that may experience unacceptable risks. In contrast, PRA methods allow statements to be made regarding both the probability of exceeding a target risk, and the level of confidence in the risk estimate.

When summarizing results of PRA, graphs and tables generally should include the results of the point estimates of risk (e.g., CTE and RME). It may be informative to note where on the risk distribution each of the point estimates lies. By understanding the assumptions regarding the inputs and modeling

approaches used to derive point estimates and probabilistic estimates of risk, a communicator will be better prepared to explain the significant differences in risk estimates that may occur. Special emphasis should be given to the model and parameter assumptions that have the most influence on the risk estimates, as determined from the sensitivity analysis (see Appendix A).

6.4 GRAPHICAL PRESENTATION OF PRA RESULTS TO VARIOUS AUDIENCES

Graphics can be an effective tool for communicating concepts in PRA. As the old adage goes, “A picture is worth a thousand words.” A graphic usually can be most easily understood by a diverse audience when it conveys a single message. It is generally a good idea to keep the graphics simple so that the message is clear. In general, each graphic should be developed and modified depending on the type of presentation and the intended audience.

☞ The key to presenting graphics in PRA effectively is to select a relatively small number of appropriate messages, and to find a balance between meaningful information and overwhelming detail.

Points to consider when developing graphics for public meetings, senior staff, and the press are presented below. Certainly, recommendations for presenting clear and informative graphics are applicable to all three forums. Practical recommendations for graphical analysis techniques and tips for successful visual displays of quantitative information are given by Tufte (1983) and Helsel and Hirsch (1993).

6.4.1 PUBLIC MEETING

For a public availability session (or meeting), care should be taken to assure that the graphics are of appropriate size and the lettering is easy to read. For example, a graphic on an 8 ½ x 11 inch sheet of paper, or a font size smaller than 18 pt in a computer presentation, may not be easily seen from the back of a large auditorium. It may be appropriate to present information using large posters, spaced so that the audience may move among them and discuss the posted results with the risk assessor or RPM. Handouts and a glossary of terms may also be used. Using slides with too much text should be avoided, since the information may be difficult to read and understand. Pre-planning and pilot testing the graphics before the presentation may be helpful in assuring that the message is accurately portrayed to the community.

Consistent with EPA’s guidance on risk characterization, the CTE and RME cancer risks and noncancer hazards, and EPA’s decision point should be highlighted on graphics. The discussions accompanying the graph should emphasize that these values represent risks to the average and high-end individuals, respectively, and serve as a point of reference to EPA’s decision point. The distribution of risks should be characterized as representing variability among the population based on differences in exposure. Similarly, graphics that show uncertainty in risk can be described using terms such as “confidence interval”, “credible interval”, or plausible range. The graphics need not highlight all percentiles. Instead, selected percentiles that may inform risk management decisions (such as the 5th, 50th, 90th, 95th, and 99th percentiles) should be the focus. Figure 6-1 shows an example of a PDF for variability in risk with an associated text box for identifying key risk percentiles.

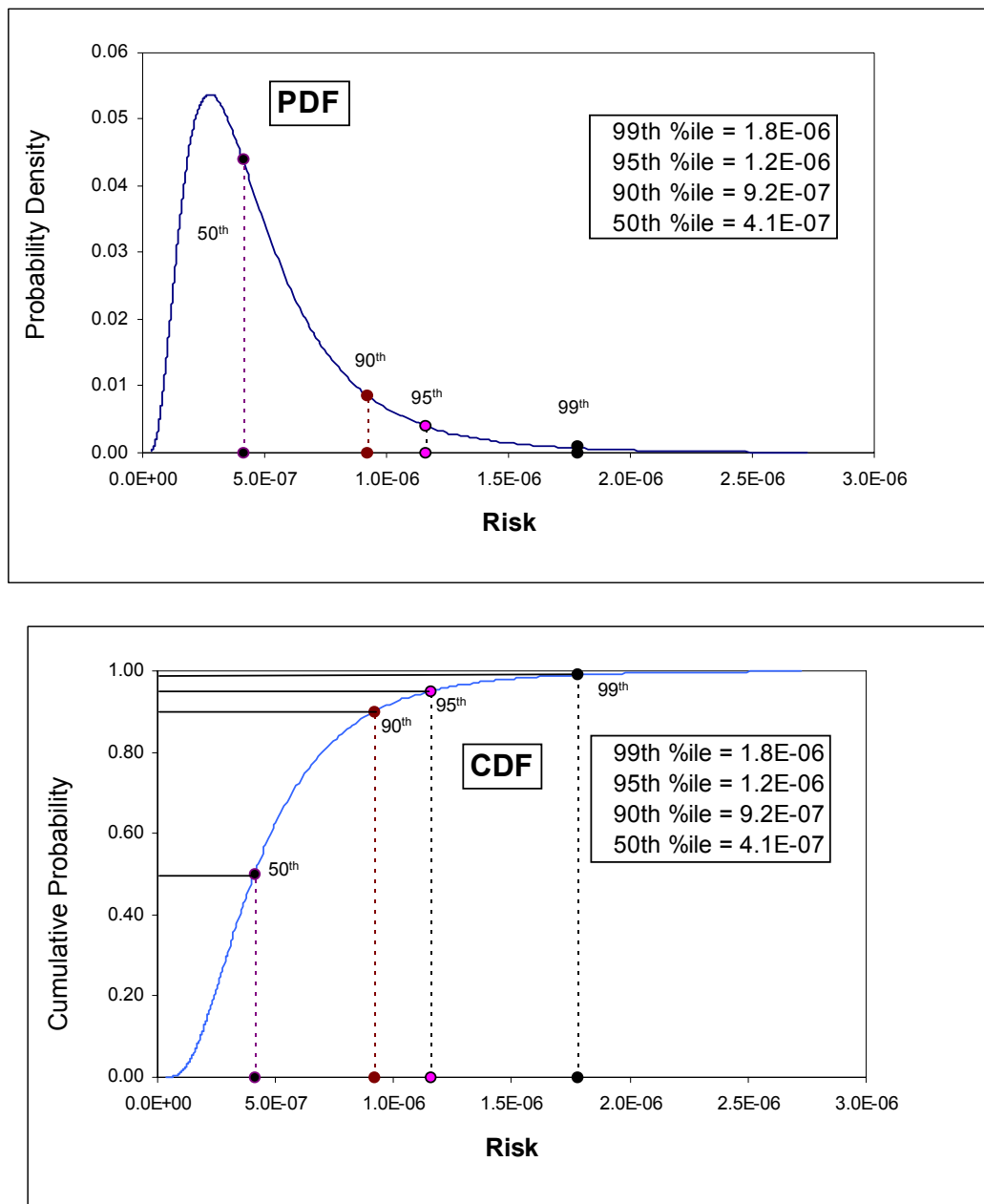


Figure 6-1. Hypothetical PRA results showing a probability density function (PDF) (top panel) for cancer risk with selected summary statistics for central tendency and high-end percentiles. This view of a distribution is useful for illustrating the shape of the distribution (e.g., slightly right-skewed) and explaining the concept of probability as the area under a curve (e.g., most of the area is below 1E-06, but there is a small chance of 2E-06). Although percentiles can also be overlaid on this graphic, a cumulative distribution function (CDF) (bottom panel) may be preferable for explaining the concept of a percentile.

Figure 6-2 gives two examples of graphics that can be used to display results of a sensitivity analysis from a Monte Carlo Analysis (MCA). While both graphics are likely to be understood by non-technical audiences, the pie chart may be more familiar. The pie chart (Figure 6-2A) suggests that the results should sum to 1.0, which may not be true if there are correlations among one or more variables, or if only a subset of the variables are displayed (e.g., those that contribute at least 1%). The available data can be normalized so that the squared correlation coefficients do sum to 100%, and this approach has been adopted by some commercial software available to run Monte Carlo simulations (e.g., *Crystal Ball*® by Decisioneering, www.decisioneering.com). The benefit of showing the squared correlation coefficient (r^2 or *r-square*, also called the coefficient of determination), rather than the correlation coefficient (r) is that *r-square* is proportional to the total variation in risk associated with specified input variable. Therefore, one can use the *r-square* to describe, in quantitative terms, the contribution of the input variable to the total variance in the risk distribution. In this example, exposure duration (ED) contributes approximately two-thirds (64%) to the total variance in risk.

A more technical graphic is the tornado plot (Figure 6-2B). In addition to showing the relative magnitude of the correlations (*r-square*), it illustrates the direction of influence a specific variable has on the final risk estimate. Bars that extend to the right indicate a positive correlation (e.g., high risk estimates correspond with high values for the variable), whereas bars that extend to the left indicate a negative correlation (e.g., high risk estimates correspond with low values for the variable.) In this example, the exposure duration (ED) has the largest positive correlation with risk, while body weight (BW) has the largest negative correlation with risk.

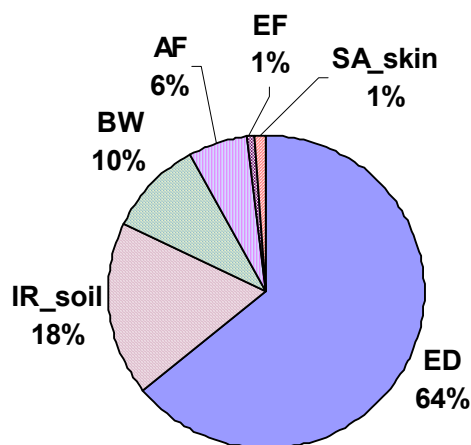
The graphics shown in this chapter are a small fraction of the graphics that might be used to communicate concepts related to PRA. Numerous additional examples are given throughout this guidance document. Table 6-1 provides a summary of cross references to other figures that were developed for this guidance document to convey specific concepts regarding variability and uncertainty.

Table 6-1. Examples of Graphics for Communicating PRA Concepts in this Guidance Document.

General PRA Topic Area	Location	Variability	Uncertainty
Conceptual Diagrams for Fundamental Concepts			
Monte Carlo Analysis	Figure 1-2	X	
Tiered process for PRA	Figure 2-1, 2-2	X	X
PDFs and CDFs			
Input variable(s)	Figure 1-1, 4-4, 4-5, 4-6	X	
Risk distribution with selected percentiles highlighted	Figure 6-1		X
Comparing RME risk (e.g., 95 th percentile) with risk level of concern	Figure 1-3, 4-3, 7-2,	X	
Selecting and Fitting Probability Distributions			
Fitting distributions - frequency distribution overlaid by a PDF	Figure 3-1	X	
Lognormal probability plot	Figure 5-2	X	
Sensitivity Analysis			
Sensitivity analysis - tornado plot of Spearman rank correlations	Figure 3-6, 6-2b	X	
Sensitivity analysis - pie chart	Figure 6-2a	X	
Joint probability curve	Figure 4-8	X	
Variability in toxicity			
Species sensitivity distribution	Figure 4-7	X	
Iterative Simulations			
CDFs from multiple 1-D MCA simulations to convey uncertainty in the risk distribution	Figure 3-3		X
PRG Selection			
Estimation from best-fit line for RME risk and EPC	Figure 5-1	X	
RME risk ranges corresponding to alternative choices of PRG	Figure 7-4	X	
90% credible interval for RME risk (95 th percentile) corresponding to alternative choices of PRG	Figure 7-5		X

Bi-model distribution for concentration showing pre-remediation EPC, post-remediation EPC, remediation action level, and uniform distribution for clean fill	Figure 5-3	X	X
2-D MCA Results			
Illustration of tabular and graphic outputs of a 2-D MCA	Figure 4-9		X
Confidence intervals (or credible intervals) on a risk distribution	Figure 1-4, 4-10, 4-11, 4-12		X
Box-and-whisker plot for results of 2-D MCA	Figure 3-4, 7-3		X
Horizontal box-and-whisker plots with multiple CDFs	Figure 6-3	X	X

A. Pie Chart



B. Tornado Plot

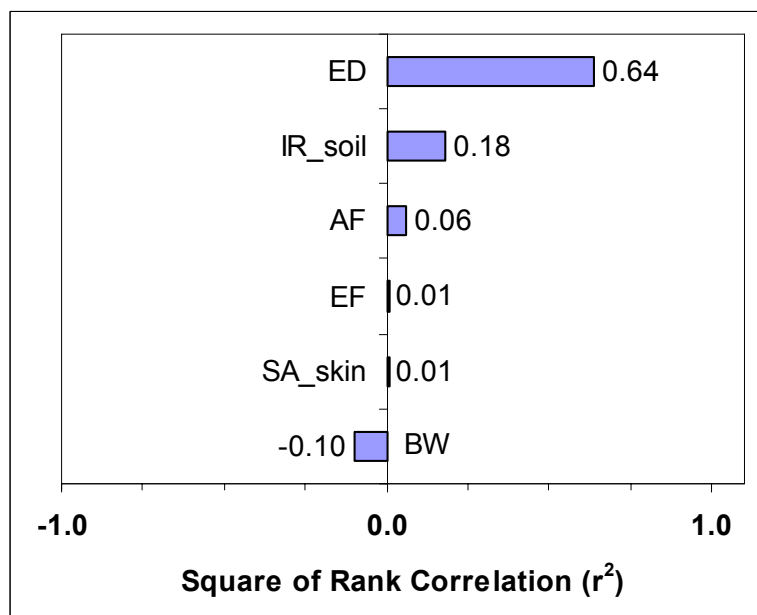


Figure 6-2. Results of a sensitivity analysis shown as a pie chart (A) and tornado plot (B). Both graphics illustrate the concept of the relative contribution to variance for exposure variables that contribute at least 1% to the variance in risk. The pie chart suggests that the sum of the squared rank correlations equals 1.0, which is true only if the results are normalized to 100%. The tornado plot gives both the magnitude and direction (positive or negative) of the correlation. ED=exposure duration, IR_soil=soil ingestion rate, AF=absorption fraction, EF=exposure frequency, SA_skin=surface area of skin, and BW=body weight.

6.4.2 EPA SENIOR STAFF

For communicating PRA with EPA's senior risk managers (e.g., EPA Section Chiefs, EPA Branch Chiefs, or EPA Division Directors), an executive summary or executive briefing package may be appropriate. This presentation should highlight major findings, compare point estimate and probabilistic results, provide sensitivity analysis results, and state uncertainties addressed in the PRA.

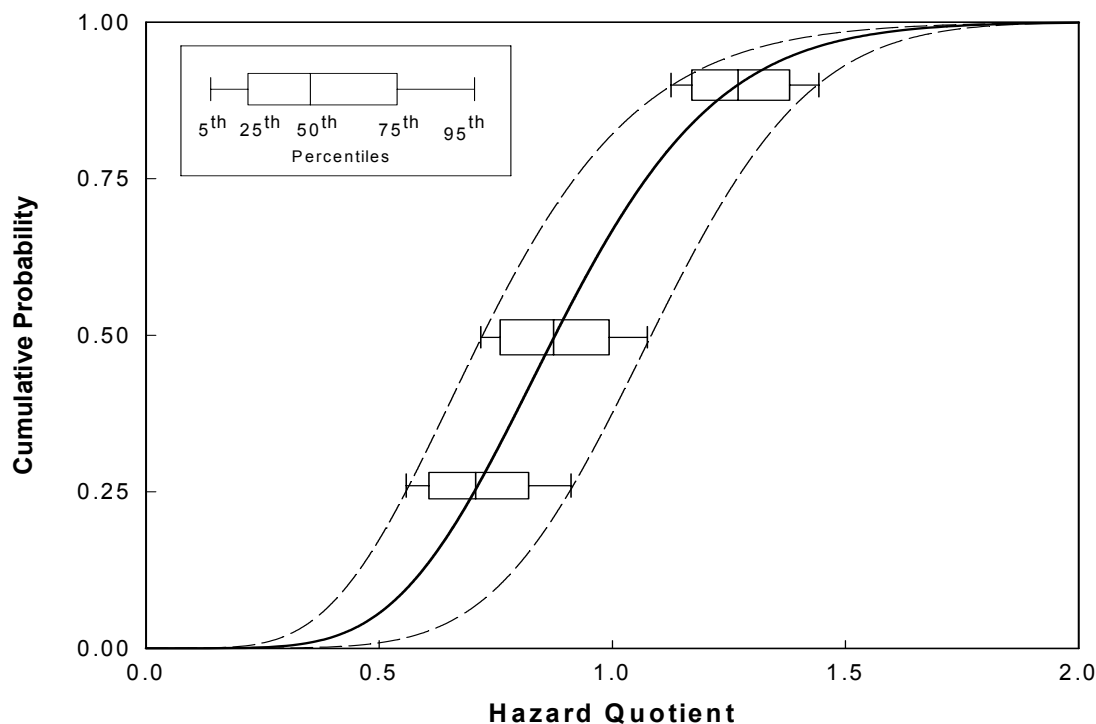


Figure 6-3. The results of a 2-D MCA. The graphic shows a method of presenting variability as a cumulative distribution function and uncertainty as box plots at the 25th, 50th, and 95th percentiles of variability. The CDF of the 50th percentile is represented by the solid line and the CDFs given by the dotted lines represent the 5th and 95th percentiles of uncertainty for each percentile of variability.

EPA senior level risk managers would generally be most interested in the risk estimates at the 50th, 90th, 95th, and 99.9th percentiles (i.e., a CTE risk estimate and the RME risk range). EPA senior managers may also wish to know the uncertainty surrounding each of the percentiles of risk. This uncertainty can be described in a table (e.g., confidence intervals around the 95th percentile risk) or a graphic (e.g., box-and-whisker plots). It is advisable for the risk assessor to have this information on hand during the briefing to respond to questions. Presenting distributions of uncertainty along with distributions of variability can create a very busy figure or table—it is best to keep things simple.

Figure 6-3 shows cumulative distribution functions (CDFs) for the Hazard Quotient (HQ) for a single chemical, representing variability in HQ. One method of displaying uncertainty is to use box-and-whisker plots. In this example, the horizontal box and whiskers represent uncertainty around selected percentile estimates of variability. Specifically, the three box-and-whisker plots correspond to the 25th, 50th, and 95th percentiles of the distribution for variability in HQ. The box shows the 25th and 75th percentiles (i.e., interquartile range) of uncertainty, whereas the whiskers show the 5th and 95th percentiles of uncertainty. In this example, uncertainty in the 95th percentile HQ is quantified by the box-and-whiskers plot in which the 5th percentile of uncertainty is 1.1, the 50th percentile is 1.3, and the 95th percentile is 1.4. This suggests that despite the uncertainty in the estimate of the 95th percentile of variability, an HQ of 1.0 is likely to be exceeded. Sometimes such results are said to describe the 90% *confidence interval* in the 95th percentile HQ. The term “confidence interval” is used loosely in this context to convey information about uncertainty; however, it is not the same as a statistical confidence limit that one might obtain by estimating a population parameter from a sample. An alternative term that may be more appropriate in this case is “credible interval”.

The three curves represent similar information on uncertainty across the complete range of percentiles for variability. The solid line shows the CDF for all of the 50th percentiles of uncertainty, whereas the dotted lines show the 5th and 95th percentiles of uncertainty.

The box-and-whisker plot is simple to produce, conveys information about the symmetry and width of the confidence interval, and is easy to interpret (Tufté, 1983). In general, box-and-whisker plots are useful for summarizing results from two-dimensional Monte Carlo (2-D MCA) simulations. The methods and inferences associated with 2-D MCAs are discussed further in Appendix D. The results of a 2-D Monte Carlo simulation represent a range of possible estimates for the percentile given one or more sources of uncertainty that were included in the simulation. If the target audience for this graphic has a greater understanding of statistics, it may be less confusing if alternative phrases are used to describe the results, such as “credible interval” or “probability band”.

Graphics that show probability density functions for uncertainty (PDFu’s) are generally more meaningful to a technical audience of risk assessors and uncertainty analysts. Alternative graphics may be needed to communicate other sources of uncertainty in risk estimates (e.g., use of alternative probability models for exposure variables, effect of changes in the model time step, application of spatial weighting to concentration data, etc.). Additional information on communicating risks to senior EPA managers is given by Bloom et al. (1993).

The results from the sensitivity analysis may be useful to the senior managers in deciding whether additional sampling is necessary. One issue that may be important to address with risk managers and senior staff is that the width of the credible interval (e.g., 5th to 95th percentiles of uncertainty) will be determined in part by the number of sources of uncertainty that are quantified. As additional sources of uncertainty are quantified and included in the model, the interval around the risk distribution will tend to widen. This situation may appear to be counterintuitive for those managers who expect confidence to increase as uncertainty is quantified. However, by uncovering and quantifying the sources of uncertainty, the benefits in the risk communication and decision-making process should become clear. The results of the sensitivity analysis should help to focus discussions, data collection efforts, and analyses on the more significant sources of uncertainty. In addition, by developing estimates of credible intervals of uncertainty in risk estimates, the decision-making process using the tiered approach may become more transparent.

6.4.3 PRESS RELEASES

For a press briefing presentation, care should be given to identify messages and develop publication quality graphics with clear descriptions that can be provided in press packages. It is usually a good idea to provide the graphics in both color and black and white so that the press can choose the most appropriate presentation style for the story. The RPMs generally should work with the CIC, the press staff in the Communication Division, and senior managers to develop press materials. Adequate time should be left for the preparation of materials and internal Agency review and approval before information is released.

6.5 PERCEPTION OF RISK AND UNCERTAINTY

The purpose of this section is to present current thinking about how people view risk and uncertainty. This section should provide useful information for planning risk communication and addresses the first step in the seven step process (Section 6.2.2), "Identify the Audience."

There are many individual differences in the way people regard the risks and hazards that are present in modern life. These differences have their roots in the differences in perception of risk and uncertainty of the individual human mind (Slovic, 1986). The risk assessor and/or risk communicator should keep in mind the general perceptions about risk held by different groups. Communications should be tailored to the specific audience. This section summarizes some of the criteria used to judge risks in the absence of scientific data and the direction of the potential bias that may be expected by applying these criteria. Additional publications on this issue are identified in the reference section at the end of this chapter.

In the absence of scientific data, the general public evaluates risks using inferences of judgment as described below (Slovic et al., 1979):

- **Availability:** People tend to judge risks as more likely if they are easy to recall.
- **Overconfidence:** People tend to be overconfident about the judgments they make based on the use of heuristics.
- **Desire for Certainty:** People tend to misgauge risk/benefit conflicts in favor of the benefits as a result of a desire for certainty and anxiety about uncertainty.

Slovic et al. (1979) identified nine characteristics of risk that may influence perceptions. These nine dimensions may provide a perspective on whether a health risk is perceived as “more risky” or “less risky”, as described in the table below.

Dimension of Risk	More Risky	Less Risky
Voluntariness	Involuntary	Voluntary
Immediacy of the effect	Delayed	Immediate
Exposed persons’ knowledge about risk	Low	High
Sciences’ knowledge about risk	Low	High
Control over risk	Low	High
Newness	Unfamiliar or New	Familiar
Chronic/Catastrophic	Catastrophic	Chronic
Common/Dread	Dreaded	Common
Severity of the consequences	High	Low

The presentation of uncertainty in a risk estimate can be interpreted with vastly different conclusions depending on the audience and their perceptions. For example, a thorough scientific account of multiple sources of uncertainty presented to a group of interested risk assessors and environmental scientists may be clearly understood. Such a group will likely conclude that the assumptions made in the risk assessment were appropriate and that the results can be used with confidence as a decision support tool. In contrast, a similar scientific presentation given to the community may be misunderstood, and the perceived risk may be greater. Citizens are often more concerned about the potential impact to their personal situation, than to the uncertainty in the risk estimate. Consequently, the community may react negatively to a long, highly scientific presentation on uncertainty. A good rule of thumb is to limit the presentation to no more than 15 minutes.

Focusing heavily on uncertainty may cause citizens to conclude that the risk must be high. They may also conclude that the presenter is incompetent because he or she is not sure of anything, or that the presenter is trying to hide something by cloaking the information in technical jargon, or even that the presenter is intentionally avoiding the public’s issues of concern. To the extent possible, technical jargon during the presentation should be avoided or explained.

A helpful presentation generally should incorporate the following steps: (1) present information about the conclusions that can be drawn from the risk assessment; it is extremely frustrating for decision-makers to receive detailed information on uncertainty without conclusions (Chun, 1996); (2) describe the certainty of the information that supports these conclusions; (3) address the uncertainty and its implications for the conclusions; and (4) present the information without jargon and in a frank and open manner. Section 6.4 provides examples of graphics that may be useful in presentations of PRA.

6.6 TRUST AND CREDIBILITY

The single most important quality a presenter may need to possess in order to communicate to others is a sense of trust and credibility. Trust and credibility are based on working with the community and providing thoughtful, accurate responses to questions and concerns raised by the community. Building trust and credibility is important, whether communicating to a high-level technical audience, a RPM/decision-maker who wishes to have the "big picture," or the public.

Credibility can best be established through a long history of frank and open discussions with the community. In addition, a presenter can gain credibility if he or she has the ability to restate the available information so that it addresses the concerns and interests of an audience. The ability to garner trust and credibility comes from knowing the audience, respecting their opinion, and communicating at an appropriate level (U.S. EPA, 1994).

6.7 COMMUNICATION ISSUES FOR RPMs

Following the RPM's decision to conduct a site-specific PRA, the level of stakeholder involvement in the development and review of the PRA should be evaluated. Establishing the appropriate level of stakeholder involvement may include input from the CIC, risk assessor and appropriate senior managers (e.g., Section Chief, Branch Chief, etc.). The level of stakeholder involvement may vary depending on the site complexity and the interest of the community. As an initial step, it may be appropriate to conduct an exploratory session where letters are sent to various stakeholders (e.g., environmental groups, CAG, etc.) inviting their participation in a general meeting on the topic of PRA. If there is a strong interest among the stakeholders, then a more involved communication plan may be appropriate including, but not limited to the following steps:

- Providing stakeholders with an introduction to the principles of PRA in an informal session (e.g., public availability session).
- Providing a draft Scope of Work (SOW) to interested stakeholders followed shortly thereafter by an availability session to discuss comments on the document.
- Providing a period of time for the stakeholders to review and comment on the selected distributions, including an availability session for discussions with EPA staff where the community may help to identify key site-specific information such as exposure factors and receptor behavior.
- Providing the opportunity for EPA risk assessor to meet with the TAG grantee (if appropriate) and stakeholders to ask questions regarding the SOW.
- Providing a revised SOW including a response to stakeholder comments.

- Providing an overview of the final PRA at a public meeting and providing appropriate supporting PRA documents in the repositories for stakeholder review and comment. This session may be part of the general session regarding the remedial investigation when the risk assessment is discussed. Based on the complexity of the PRA, it may be appropriate to hold a public availability session where the stakeholders (including the TAG grantee), if appropriate, are able to meet with EPA staff to ask questions and offer suggestions regarding the document.
- Providing a response to comments from stakeholders regarding the PRA.

If the level of interest is low, then a less extensive CIP may be appropriate. In this case, fact sheets (in plain language) describing the general principles of PRA to the stakeholders and the key findings of the PRA may be provided (U.S. EPA, 2000a). At public meetings where the risk assessment is discussed, a short discussion of the PRA findings and their significance may be appropriate. The PRA document should be made available in the repositories for review and comment by the stakeholders.

For sites with medium interest, a combination of the activities identified above may be appropriate. For example, it may be appropriate to have a public availability session on the principles of PRA and then make the documents available for review and comment.

The RPM should consider a number of administrative issues in developing the plan for involving the stakeholders in the PRA. Issues to consider include: staff resources, funds for obtaining meeting space, availability of contractor support, significance of PRA in decision making, and the length of time required to complete the RI/FS. To aid in reducing costs, it may be appropriate to combine meetings regarding PRA and point estimate risk assessment based on the close links between the documents.

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CHAPTER 7

ROLE OF THE PRA IN DECISION MAKING

7.0 INTRODUCTION

When deciding whether or not to remediate a hazardous waste site, the risk manager needs to know if an unacceptable risk is present, and if so, what cleanup level to apply to the contaminated media. For this information, the risk manager should turn to the risk assessor for help in interpreting the results of the risk assessment. This chapter provides guidance on how to interpret the results of a probabilistic risk assessment (PRA) to help determine if an unacceptable risk is present, and the criteria to consider when deriving a risk-based preliminary remediation goal (PRG) and a final remedial goal.

7.1 GENERAL PRINCIPLES OF RISK-BASED DECISION MAKING IN SUPERFUND

Under Agency policy, an individual with reasonable maximum exposure (RME) will generally be the principal basis for evaluating potential human health risks at Superfund sites (see *Risk Assessment Guidance for Superfund* (Section 6.1.2 of U.S. EPA, 1989) and the National Contingency Plan's (NCP) Preamble (U.S. EPA, 1990)). The RME is defined as the highest exposure that is reasonably expected to occur at a site, and is intended to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures. In general, where cumulative carcinogenic risk to the RME individual is less than $1\text{E-}04$, and the non-carcinogenic Hazard Index (HI) is less than or equal to 1, remedial action is not warranted under Superfund unless there are adverse environmental impacts, or the applicable or relevant and appropriate requirements (ARARs) are not met. As discussed in Section 7.2.4, the RME receptor is often (although not always) an appropriate basis for evaluation of risks to ecological receptors, as well.

Once a determination of unacceptable risk to humans and/or ecological receptors has been made, the risk managers will typically ask the risk assessor to develop site-specific PRGs. PRGs are generally defined as health-based chemical concentrations in an environmental media for which the risks (cancer or noncancer) to the RME receptor would not exceed some specified target level. For systemic or noncarcinogenic toxicants, the target risk level is generally a HI of unity (1). This is considered to be a threshold concentration to which the human population (including sensitive subgroups) and ecological receptors may be exposed without adverse effect during less-than-lifetime (i.e., chronic, subchronic, or short-term) exposures. For carcinogens, the target risk level used to derive the PRG typically represents a cumulative lifetime cancer risk to an individual of between $1\text{E-}06$ and $1\text{E-}04$ (equivalently expressed as 10^{-6} and 10^{-4}). For carcinogenic risks, less-than-lifetime exposures are converted to equivalent lifetime values (U.S. EPA, 1989). The $1\text{E-}06$ risk level is specified in the NCP as a point of departure for determining remediation goals when ARARs are not available or not sufficiently protective. It is important to remember that risk-based PRGs are initial guidelines and do not represent final cleanup or remediation levels. Remediation levels are finalized after appropriate analysis in the remedial investigation/feasibility study (RI/FS) and record of decision (ROD). A final cleanup level may differ from a PRG based on the risk manager's consideration of various uncertainties in the risk estimate, the technical feasibility of achieving the PRG, and the nine criteria outlined in the NCP (see Chapter 1, Exhibit 1-2).

EXHIBIT 7-1

DEFINITIONS FOR CHAPTER 7

Applicable or Relevant and Appropriate Requirements (ARARs) - Federal or state environmental standards; the NCP states that ARARs should be considered in determining remediation goals. ARARs may be selected as site-specific cleanup levels.

Central Tendency Exposure (CTE) - A risk descriptor representing the average or typical individual in a population, usually considered to be the mean or median of the distribution.

Confidence Interval - A range of values that are likely to include a population parameter. Confidence intervals may describe a parameter of an input variable (e.g., mean ingestion rate) or output variable (e.g., 95th percentile risk). When used to characterize uncertainty in a risk estimate, it is assumed that methods used to quantify uncertainty in the model inputs are based on statistical principles such as sampling distributions or Bayesian approaches. For example, given a randomly sampled data set, a 95% confidence interval for the mean can be estimated by deriving a sampling distribution from a Student's t distribution.

Credible Interval - A range of values that represent plausible bounds on a population parameter. Credible intervals may describe a parameter of an input variable (e.g., mean ingestion rate) or output variable (e.g., 95th percentile risk). The term is introduced as an alternative to the term confidence interval when the methods used to quantify uncertainty are not based entirely on statistical principles such as sampling distributions or Bayesian approaches. For example, multiple estimates of an arithmetic mean may be available from different studies reported in the literature—using professional judgment, these estimates may support a decision to describe a range of possible values for the arithmetic mean.

Hazard Index (HI) - The sum of more than one Hazard Quotient for multiple substances and/or multiple exposure pathways. The HI is calculated separately for chronic, subchronic, and shorter-duration exposures.

Hazard Quotient (HQ) - The ratio of a single substance exposure level over a specified time period (e.g., subchronic) to a reference dose (or concentration) for that substance derived from a similar exposure period.

Preliminary Remediation Goal (PRG) - Initially developed chemical concentration for an environmental medium that is expected to be protective of human health and ecosystems. PRGs may be developed based on applicable or relevant and appropriate requirements, or exposure scenarios evaluated prior to or as a result of the baseline risk assessment. (U.S. EPA, 1991a, 1991b).

Reasonable Maximum Exposure (RME) - The highest exposure that is reasonably expected to occur at a site (U.S. EPA, 1989). The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures.

Remedial Investigation/Feasibility Study (RI/FS) - Studies undertaken by EPA to delineate the nature and extent of contamination, to evaluate potential risk, and to develop alternatives for cleanup.

RME Range - The 90th to 99.9th percentiles of the risk distribution generated from a PRA, within which an RME risk value may be identified. The 95th percentile is generally recommended as the starting point for specifying the RME risk in a Superfund PRA.

RME Risk - The estimated risk corresponding to the reasonable maximum exposure.

7.2 INTERPRETING A RISK DISTRIBUTION

7.2.1 WHAT IS A DISTRIBUTION OF RISK AND WHAT DOES IT LOOK LIKE?

In the traditional point estimate risk assessment approach, risks to the RME individual are characterized as single point values (e.g., HI=2, or cancer risk=1E-05). In the PRA approach, the output of the risk assessment is an estimate of the distribution of risks across all members of the population. An example is shown in Figure 7-1.

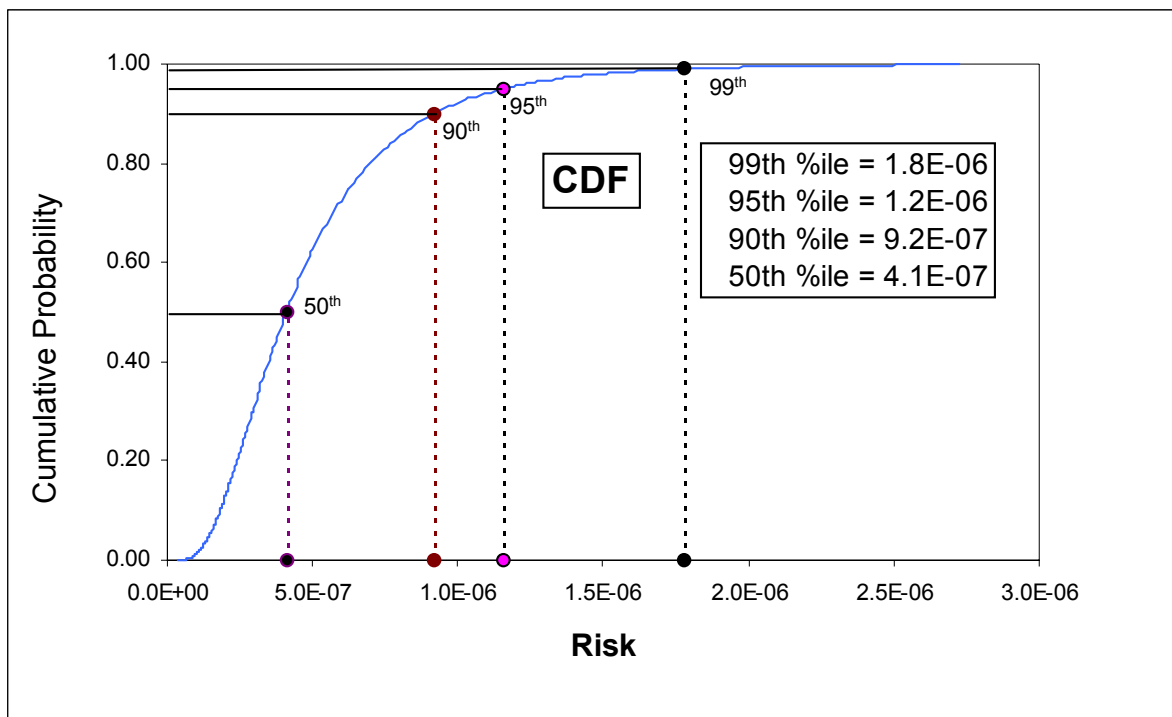


Figure 7-1. Hypothetical PRA results showing a cumulative distribution function (CDF) for lifetime excess cancer risk.

In this example, the x-axis of Figure 7-1 represents the excess lifetime cancer risk level and the y-axis represents the cumulative probability of the cancer risk level within the hypothetical population. The graph also shows various landmarks along the distribution curve such as the 50th percentile, the 90th, 95th, etc. In this illustration, the 95th percentile corresponds to a cancer risk of 1.2E-06.

7.2.2 WHAT IS THE RME RANGE?

Given a risk distribution such as shown in Figure 7-1, what part of the risk distribution should a risk manager be concerned about? As explained above, the risk to the RME receptor is a key factor in making decisions regarding the need for action at a Superfund site. EPA's *Guidelines for Exposure Assessment* (U.S. EPA, 1992) states that the "high-end" (or RME) of exposure for a population occurs between the 90th and 99.9th percentiles, with the 99.9th percentile considered a bounding estimate. Similarly, PRAs developed to support RME risk estimates for Superfund should reflect this approach.

*In this guidance, the 90th to 99.9th percentiles of the risk distribution are collectively referred to as the **recommended RME range**.*

In utilizing PRA results to determine if an unacceptable risk is present and to develop a PRG which is sufficiently protective, risk managers should address two questions:

- (1) What percentile of the risk distribution will be selected to represent the RME receptor?
- (2) How will information on uncertainty in the high-end risk estimates be used in this process?

The risk manager may consider a number of factors in choosing a specific percentile to represent the RME individual. This may include both quantitative information and professional judgment. In particular, risk managers may need to understand what sources of variability and uncertainty are already explicitly accounted for by the modeling approach and inputs (i.e., point estimates and/or probability distributions) used to estimate the risk distribution, and what sources may be present but are not quantified. Approaches for selecting an appropriate percentile in human health and ecological risk assessments are described below.

7.2.3. RELATING THE RISK DISTRIBUTION TO THE RISK MANAGEMENT GOAL FOR HUMAN HEALTH

In most cases, a recommended starting point for risk management decisions regarding the RME is the 95th percentile of the risk distribution. The 95th percentile for the risk distribution is an appropriate description of high-end exposure as identified by the Presidential/Congressional Commission on Risk Assessment and Risk Management (1997).

In human health PRA, a recommended starting point for risk management decisions regarding the RME is the 95th percentile of the risk distribution.

Figure 7-2 illustrates this approach for a site where cancer risks are the risk driver. Assume the risk manager has selected an excess cancer risk of 1E-05 as the risk management goal, and the 95th percentile as the definition of the RME. If line B on the graph represents a 1E-05 probability of cancer, a no-action decision may be warranted because the 95th percentile of the risk distribution is below the cancer risk level of concern. Conversely, if we were to assume that the 95th percentile is above the risk level of concern (i.e., line A on the graph represents 1E-05), remedial action may be warranted.

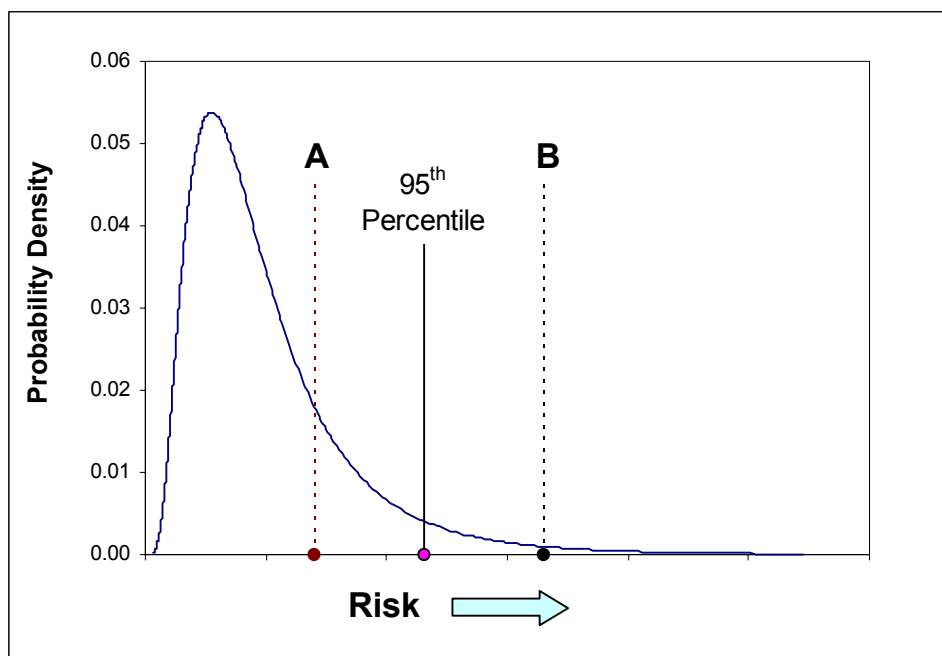


Figure 7-2. Example of a probability distribution for risk illustrating the 95th percentile and two different risk levels of concern (A and B). Assuming the 95th percentile corresponds to the RME, the need for remedial action depends on how the RME risk compares with the risk level of concern. For Case A (RME > level of concern), remedial action may be warranted. For Case B (RME < level of concern), remedial action may be unnecessary.

Although the 95th percentile is recommended as a starting point for defining the RME in the majority of human health risk assessments conducted within the Superfund program, the risk manager may use discretion in selecting a different percentile within the RME range (90th to 99.9th percentiles). In situations where the risk manager believes that a sufficient amount of site-specific information has been collected to indicate that the risk estimates are much more likely to be high (e.g., overestimated due to multiple health protective inputs), the risk manager may choose a lower percentile within the recommended RME risk range (e.g., the 90th) as the most representative of the RME estimate at the site. Conversely, when the risk manager believes that the risk estimates may tend to underestimate true risks, or if there is substantial uncertainty in the accuracy of the risk estimates, the risk manager may choose a percentile higher than the 95th in the recommended RME risk range (e.g., the 98th or the 99th). There are a variety of factors that can be considered when making this decision, such as the qualitative and quantitative uncertainty in the exposure assessment calculations, the uncertainty in the toxicity values, and the presence of biological or measured data (in contrast to modeled data). These factors are discussed below in Section 7.3. It is highly recommended that the risk manager consult with the site risk assessor when applying these factors to determine an appropriate percentile in the RME risk range.

7.2.4 RELATING THE RISK DISTRIBUTION TO THE RISK MANAGEMENT GOAL FOR ECOLOGICAL RISK ASSESSMENT

For ecological risk assessments, the choice of the percentile of the variability distribution for exposure or risk that will be protective depends on the receptor that is being considered as well as the nature of the endpoint used to establish the level of concern. For most species, the risk management objective will generally be to ensure population sustainability, even if some individual members of the population (those at the upper end of the exposure or risk distribution) may experience a higher risk of adverse effects. The risk management goal of population stability does not necessarily correspond to protection of the central tendency receptor at or below the regulatory level of concern.

As indicated in Chapter 4, without knowledge of the proportion of the local population that must survive and reproduce for the population to be stable, the choice of the central tendency exposure (CTE) receptor as the basis of the risk management goal may not be protective. Sustainability of a local population often depends upon the amount of “reserves” within that subpopulation to fill in ecological niches left voided by toxicologically impaired individuals. At a very small number of sites, a population biologist may be able to provide information about the level of effect associated with a decrease in population sustainability. At the majority of sites, the use of the CTE receptor by risk management as the basis for adequate protection of local populations of ecological receptors cannot be supported. Therefore, in the absence of such species-specific (trophic level) information, it is prudent and appropriate to base PRGs and cleanup levels on the upper end of the distribution of variability in the Hazard Quotient (HQ) to provide greater confidence that the receptor population of concern will be protected.

For threatened or endangered species, it will normally be appropriate to provide protection to as high a percentile of the distribution (i.e., the RME receptor) as is practicable (e.g., high-end of the RME range of 90th to 99.9th percentiles), since injury to even a single individual is undesirable.

7.3 FACTORS TO CONSIDER IN CHOOSING THE PERCENTILE FOR THE RME

Risk assessments (both point estimate and PRA) should be based on the best quality data available. A key component of the risk management process is a careful review and evaluation of the potential limitations in the quality and relevance of the data that are used in the risk assessment (i.e., qualitative and quantitative uncertainties) in order to evaluate the strengths and weaknesses of the assessment (U.S. EPA, 1993). Communication between risk managers, risk assessors, and other technical team members is vital at this stage. The main question to be answered is, “How well do the inputs to the risk assessment represent exposure pathways and behaviors at a given site?” The answer to this question can be expressed qualitatively (e.g., high, medium, or low) or quantitatively (e.g., confidence intervals or credible intervals). Some examples of these types of evaluation are illustrated below.

Use of Default Exposure Distributions

When site-specific data are not available, the best available information on some exposure parameters most likely will be from studies at other sites (e.g., in other parts of the country). In both point estimate risk assessment and PRA, the use of surrogate data to support input parameters raises questions about representativeness for both current and future land use scenarios. A specific example of potentially poor representativeness would be the use of national data for estimating the exposure frequency of adult workers when the receptor of concern is a railroad worker. Railroad workers may typically be on the site for only 100 days/year. If the risk assessment were based on the national default assumption of 250 days/year, this choice would give a high bias to the risk estimate.

Another example of a site-specific exposure factor that may vary considerably among different locations is fish ingestion rates. At sites where ingestion of fish contaminated with metals poses a concern, tissue concentrations from fish fillets collected on site are often used to determine the concentration term. However, a cultural practice of people harvesting fish on site may include consuming some of the internal organs of the fish in addition to the fillets. If the metal contaminants selectively accumulate in the internal organs instead of the fillet tissues, use of data only on fillets contaminants would give a low bias to the risk estimate.

Other Factors that Influence Site-Specific Exposures

Exhibits 7-2 and 7-3 list other types of factors that may be important to consider when evaluating the representativeness of an exposure or risk model. Given the source of the available data, the risk assessor should identify potential uncertainties and discuss the likelihood that the values used may under- or overestimate actual site-specific exposures. The risk manager should consider this information in decision making throughout the tiered process for PRA (see Chapter 2).

EXHIBIT 7-2

EXAMPLES OF DEMOGRAPHIC, CULTURAL, AND BEHAVIORAL FACTORS THAT CAN AFFECT EXPOSURE

- Subsistence fishing, hunting, or ingestion of home-grown produce
- Exposures to cultural foods or medicines that contain contaminants
- Preparation of foods in containers that contain contaminants that may leach out into food or beverage
- Hobbies and other personal practices resulting in exposure to contaminants
- Age of the population (e.g., children may have greater exposure and susceptibility than adults (U.S. EPA, 1995b, 1996)

EXHIBIT 7-3

EXAMPLES OF PHYSICAL OR GEOGRAPHICAL FACTORS THAT CAN AFFECT EXPOSURE

- Geographical features that limit or enhance accessibility (e.g., slopes, valleys, mountains)
- Land use, including where exposure occurs within the exposure unit, and the current or future manner in which the receptor contacts the contaminated media
- Availability of contaminated medium for exposure (e.g., grass vs. bare soil)
- Depth of contamination (e.g., surface soil is of greatest concern for direct contact)
- Bioavailability of contaminant from media or water (e.g., physiochemical factors that enhance or reduce absorption)
- Water quality and distribution systems, including water hardness and use of lead-soldered pipes
- Temporary barriers (e.g., fences, ground cover, and concrete) that affect current (but not necessarily future) exposures

For example, the features of a potentially exposed population and the physical and geographical factors at a site can increase or decrease exposure to contaminated media. These factors should be considered in defining exposure pathways and characterizing exposure variables in the risk assessment. Such site-specific information may support a decision to evaluate the entire RME range (90th to 99.9th percentile) before selecting the percentile that represents RME risk. A departure from the 95th percentile would depend on whether or not qualitative or quantitative factors suggest an increased or decreased exposure, and hence, risk. In practice, multiple and sometimes competing factors may need to be balanced in order to determine an appropriate percentile for the RME risk (see hypothetical example in Section 7.5).

Subpopulations may be at increased risk from chemical exposures due to increased sensitivity, behavior patterns that result in high exposures, and/or current or past exposures from other sources. Environmental health threats to children are a particular concern (U.S. EPA, 1995b, 1996). Once identified, a subgroup can be treated as a population in itself, and characterized in the same way as the larger population using similar descriptors for population and individual risk (U.S. EPA, 1995a). This principle applies to both point estimate risk assessments and PRA.

Use of Biological Data

Biological monitoring data and/or other biomarker data can be useful sources of information for evaluating uncertainty in an exposure or risk assessment. These data can provide an indication of the magnitude of current or past exposures and the degree to which the exposures are correlated with contaminated site media. Examples of biological data that are useful in human health assessments include lead in blood, trichloroethylene and its metabolites in blood or urine, arsenic or methyl parathion metabolites in urine, and polychlorinated biphenyls (PCBs) or dioxins in blood or fat tissue. Tissue burdens of contaminants are also widely useful as biomarkers of exposure in ecological risk assessments. Just as air or groundwater monitoring data can provide increased (or decreased) confidence in the results of predictive air or groundwater models, biomarkers can be used in a similar manner to evaluate how much confidence should be placed in predictive exposure assessment models. Biological data can be subject to the same shortcomings as other exposure data in terms of data quality and representativeness. The design and performance of the biological data collection effort generally should be carefully evaluated for these factors (e.g., low, medium, and high quality or confidence; low or high bias, etc.) before using the results in the risk decision. Currently, collection of biological monitoring data is limited at Superfund sites and requires coordination with appropriate agencies outside of EPA.

Issues Related to Toxicity Factors

A variety of factors may affect the magnitude of adverse responses expected to occur in similarly exposed individuals such as age, physiological status, nutritional status, and genotype. In general, these sources of inter-individual variability, and related uncertainties, are taken into account in the derivation of toxicity values (e.g., reference concentration (RfC), reference dose (RfD), and carcinogenic slope factor (CSF)) used in human health risk assessments. Thus, human health toxicity values usually are derived to be health-protective for the most sensitive populations.

☞ Sources of variability or uncertainty are often accounted for in the derivation of toxicity values. The level of protectiveness afforded by the toxicity value may be an important factor in deciding on the appropriate RME risk percentile to use.

Risk managers, in collaboration with risk assessors, should carefully consider whether the toxicity value is representative of the population of concern. For example, the toxicity value may be based on oral exposures to drinking water, whereas exposure to a site population being evaluated may be via soil ingestion. Similarly, the toxicity value could be based on effects in a healthy worker population, whereas the site population encompasses all ages and a range of individual health conditions. Uncertainty in toxicity values may reflect insufficient data to evaluate developmental toxicity concerns or to account for *in utero* exposures. Also, it may be unclear whether the population of concern has similar characteristics to the sensitive population accounted for in the derivation of the toxicity value. This determination may require coordination with a toxicologist to review the basis for the derivation of the toxicity values in question. Even then, in most cases, the determination will be very difficult, because our understanding of human variability in toxicologic responses is very limited for many chemicals. When data are insufficient to support a more quantitative representation of these sources of inter-individual variability an uncertainty factor may be used in the derivation of non-cancer human health toxicity values (RfD, RfC).

Some of the same factors that should be considered when employing toxicity values to estimate risk are also relevant to the use of toxicokinetic and toxicodynamic modeling in risk assessment. For example, a toxicity assessment for methylmercury used a technique called benchmark dose modeling (BMD) to relate the levels in maternal blood to adverse developmental effects, based on data from a large epidemiology study of Faroes Islanders (Grandjean et al., 1997; Budtz-Jørgensen et al., 2000). The RfD determined is well-supported by the other large human studies from the Seychelles (Davidson et al., 1995, 1998) and New Zealand (Kjellstrom et al., 1986, 1989) as well as a physiologically-based pharmacokinetic (PBPK) model based on the Seychelles data (Clewett et al., 1999). The RfD obtained with benchmark dose modeling (BMD) was 1E-04 mg/kg-day. The PBPK model incorporated variability in toxicokinetics to obtain a range of acceptable intakes of methylmercury between 1E-04 and 3E-04 mg/kg-day. Although the PBPK model was not used in the derivation of the benchmark dose value, it was used to support the choice of uncertainty factors in the derivation of the RfD.

At the time this guidance was finalized, the understanding of this type of toxicity information (i.e., human variability) was not well developed. Although such information was not used to characterize variability in human health risks, the estimates of variability from the PBPK model did provide additional information on uncertainty. For decision makers, the toxicity data and the choice of the endpoint (e.g, neurodevelopmental effects in the case of methylmercury) can guide qualitative risk management choices regarding the percentile representing the RME (within the 90th to 99.9th percentile range) and/or the appropriate level of confidence in the RME estimate. Exhibit 7-4 lists some of the issues to consider when evaluating the uncertainty in a toxicity value.

EXHIBIT 7-4

EXAMPLES OF TOXICITY CONSIDERATIONS

- How severe is the effect?
- Is the effect reversible?
- How steep is the slope of the dose-response curve at low dose?
- Is the contaminant persistent in the environment or in receptors?
- Does the contaminant bioconcentrate as it moves through the food chain?
- How bioavailable is the contaminant?

Use of Quantitative Uncertainty Estimates

PRA methods such as a two-dimensional Monte Carlo analysis (2-D MCA) may be used to quantify the uncertainty or confidence surrounding risk estimates, and this information may be helpful in selecting the RME risk percentile. Figure 7-3 provides hypothetical results of a 2-D MCA where a credible interval has been quantified for a 95th percentile of variability in noncancer HI. In exposure units (EU) 1 and 3, the credible intervals for the 95th percentile are fairly narrow, which suggests a high degree of confidence that the risks in EU1 are negligible and that the risks in EU3 are unacceptable. Conversely, the relatively wide credible intervals in EU2 and EU4 give less confidence in the results, but suggest that the 95th percentiles likely exceed a target HI of 1 in both cases. Further efforts to reduce or characterize uncertainties may affect the risk management decision in these two areas.

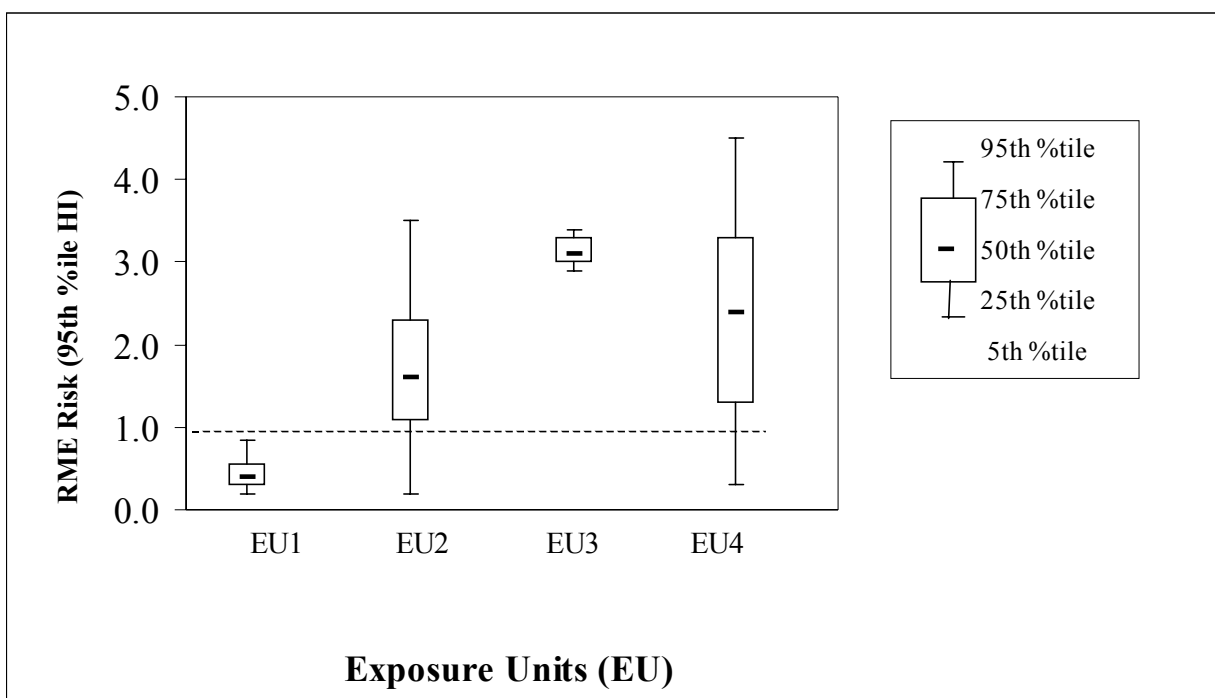


Figure 7-3. Box and whisker plots characterizing uncertainty in the RME risk estimates (95th percentile of the Hazard Index) at four locations. The box represents the inter-quartile range (25th to 75th percentiles) while the whiskers represent the 90% credible interval (5th to 95th percentiles).

Summary: Multiple Criteria Form the Basis of the Remedial Decision

Final risk management decisions should be based on a weighted consideration of all of the relevant factors that influence confidence in the risk distribution. For example, a risk manager may be presented with a risk assessment for a heavy metal in residential soil in which the distribution of cancer risk estimates in the RME range (i.e., 90th to 99.9th percentiles) overlaps the risk range of concern (1E-06 to 1E-04). The risk manager then should proceed with the site technical team to evaluate the data available to define inputs for the risk assessment, as well as the site-specific factors, and the available biological monitoring data. Assume that several factors that are likely to increase the confidence in the risk estimates were noted: (1) the soil collection and analysis effort was well-designed; (2) the predominant chemical and physical forms of the metal in the soil are characterized by relatively low bioavailability; (3) all of the yards in the residential neighborhood are covered with grass lawns, a feature generally expected to reduce direct exposure to soil; and (4) biomonitoring data from the site are all within normal physiological ranges, suggesting little, if any, excess contaminant exposure occurred at the site. In addition, generic national data were used in the absence of site-specific information on two input variables that ranked highest in the sensitivity analysis, thereby reducing confidence in the risk estimates. In this example, the consideration of these factors collectively suggests that the results of the risk assessment are likely biased towards an overestimate of risk, and this information may be used in a risk management selection of a percentile of the risk distribution to represent the RME receptor (e.g., less than or equal to the 95th percentile).

7.4 UNCERTAINTY ASSOCIATED WITH THE USE OF THE 99.9TH PERCENTILE

As previously stated, this guidance adopts the 90th to 99.9th percentiles of the risk distribution as the recommended RME risk range for decision-making purposes, consistent with EPA's *Guidelines for Exposure Assessment* (U.S. EPA, 1992). A cautionary note should be added about the selection of the higher percentiles within that range, especially the 99.9th percentile. The extreme percentiles ("tails") of an input distribution are understandably the most uncertain part of a PDF, since the number of data values in these ranges are less abundant than in the center of the range. This uncertainty in the tails of the input distributions leads in turn to greater uncertainty in the tails of the calculated exposure or risk distribution, and the magnitude of this uncertainty increases rapidly at the very high percentiles. In many cases, estimates at the extreme tails, such as the 99.9th percentile, may be neither accurate nor plausible. For that reason, great care should be taken when evaluating an RME risk in the upper percentiles of the risk range.

7.5 MOVING FROM A PRG TO A REMEDIAL GOAL

As discussed above, where an unacceptable risk is identified, the risk assessor is typically asked to develop site-specific PRGs (see Chapter 5 for discussion on derivation of PRGs). PRGs may be developed using a probabilistic approach much in the same manner as they are developed using a point estimate approach. The target risk level should be set for a specified percentile (corresponding to the RME receptor), and the concentration in contaminated media which corresponds with that target risk level should be calculated. It is important to understand that the PRG is an early step, not the last step, in the selection of a final cleanup level. During the RI/FS, the risk manager should evaluate the remedial alternatives using the nine criteria described in the NCP (U.S. EPA, 1990) (Chapter 1, Exhibit 1-2). Achieving a target level of protection for human and/or ecological receptors is one of the primary factors, but this objective should be balanced by criteria such as feasibility, permanence, state and community acceptance, and cost. Indeed, there may be times when a purely risk-based PRG may be impracticable as a final cleanup goal. In cases such as this, it is important to remember that the RME is not a single, fixed percentile on the risk distribution, but instead represents the portion of the risk distribution curve between

the 90th and 99.9th percentiles. Depending on the specific exposure and toxicity information available at a site, a PRG developed using the 90th percentile of risk may be sufficient to protect the reasonably maximum exposed individual. Alternatively, at some sites, the risk manager may feel that a PRG developed using even the 95th percentile of risk is not sufficiently protective of the RME individual and thus may choose to develop a PRG using a higher percentile.

☞ Selection of final remediation or cleanup levels during the RI/FS and ROD may be an iterative process, and may consider a range of factors in addition to the initial PRG estimate.

For example, at a former nuclear energy site, a PRG of 200 picocuries/gram (pCi/g) was developed for plutonium in soil based on a one-dimensional Monte Carlo analysis (1-D MCA) and the recommended starting point of the 95th percentile for the RME individual. At this particular site, the surrounding communities were strongly opposed to this PRG as a cleanup level. They felt it was not adequately protective, and as a result, limited progress occurred in remediating the site over the years. The communities pointed out to the risk manager that many of the exposure assumptions used in the PRA were not site-specific, and some members of the community felt that exposures occurred more often (i.e., with higher frequency) and for a longer period of time (i.e., for a greater duration) than were assumed. Based on the exposure parameters recommended by the community, the PRG would have been 75 pCi/g. At this point, the risk manager could have chosen to either go back and collect sufficient site-specific demographic and exposure data to refine the risk calculations and the PRG derivation, or evaluate the feasibility of a PRG associated with higher percentiles on the risk distribution curve (e.g., 99th percentile). In this particular example, the risk manager compared the costs associated with the cleanup that would be required to satisfy the community concerns with the costs associated with collection of additional data and recalculation of the risk and PRG. The risk manager decided that the additional cost of cleanup was manageable and expected that the PRG based on the 99th percentile would be accepted by the community. In addition, remedial activity could begin quickly without more investigation. When the risk manager presented these findings to the community, the citizens quickly agreed with this approach and remediation activities moved forward.

How does Variability and Uncertainty in Risk Relate to the Choice of a PRG?

An effective approach for communicating the results of a probabilistic analysis to risk managers is to develop graphics that relate variability and uncertainty in risk to the choice of a PRG. Two graphics are illustrated in Figures 7-4 and 7-5, based on the concept of iterative simulations presented in Chapter 5 (Section 5.5). Continuing the PRG example discussed above, assume that multiple 1-D MCA simulations are run with PRGs for plutonium ranging from 25 pCi/g to 250 pCi/g in increments of 25 pCi/g. As the concentration term is changed to correspond with a PRG, each Monte Carlo simulation yields a different distribution of risk. Figure 7-4 focuses on the RME range of percentiles from the risk distribution (i.e., 90th - 99.9th percentiles). A risk manager might use this graphic to evaluate how the PRG could change based on the choice of the percentile used to represent the RME. A hypothetical risk level of concern of 1E-05 corresponds with the 90th percentile at a PRG of approximately 125 pCi/g, whereas 1E-05 intersects the 95th percentile line at a PRG of approximately 75 pCi/g. Therefore, when variability in risk is the focus of the decision, the difference between an RME set at the 95th percentile instead of the 90th percentile is 50 pCi/g.

Figure 7-5 presents information on uncertainty, rather than variability. This graphic could be used to summarize results of a 2-D MCA (see Appendix D), or a series of 1-D MCA simulations (see

Chapter 3, Section 3.4) applied to the same range of PRGs evaluated in Figure 7-4. In this case, the results yield a 90% credible interval (CI) for the risk distribution. Figure 7-5 highlights the 90% CI for the 95th percentile, assuming that a risk manager selects the 95th percentile to represent the RME risk, and she is interested in the uncertainty in the risk estimates. Using the same hypothetical risk level of concern (1E-05), the 90% upper CI for the 95th percentile corresponds with 1E-05 at a PRG of approximately 25 pCi/g. The risk manager may need to consider the cost and feasibility of achieving a PRG as low as 25 pCi/g. In addition, the 90% lower CI corresponds to a PRG of 250 pCi/g. The risk manager may determine that this range of uncertainty (i.e., an order of magnitude) is too wide to set a PRG, and that further steps are needed to reduce identify the major sources (i.e., sensitivity analysis).

Variations on Figures 7-4 and 7-5 can be developed to focus on different percentiles of the risk range. This information, together with the results of the sensitivity analysis which highlights the major sources of variability and uncertainty, should help to guide the selection of final remediation or cleanup levels, or continued data collection and analysis following the tiered process for PRA.

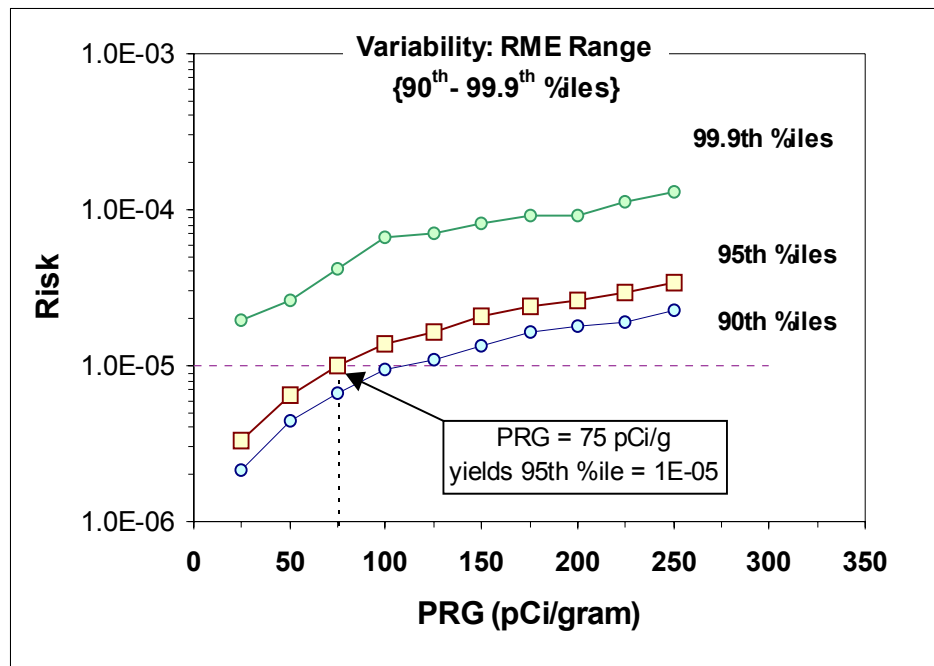


Figure 7-4. Example of graphic showing variability in risk (i.e., RME range, or 90th to 99.9th percentiles) associated with different choices of PRG for plutonium in soil (pCi/g). The hypothetical risk level of concern (1E-05) corresponds to a 90th percentile risk at a PRG of ~ 100 pCi/g, and a 95th percentile at a PRG of ~ 75 pCi/g. In this example, all of the 99.9th percentiles exceed 1E-05, leaving no choices for PRG at the high end of the RME range.

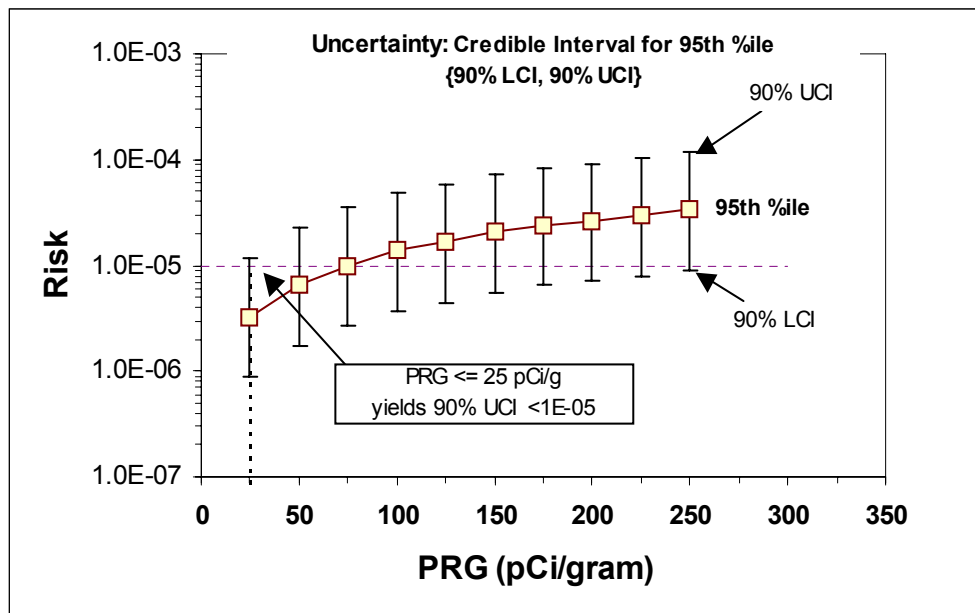


Figure 7-5. Example of graphic showing uncertainty in 95th percentile risk associated with the same choices of PRGs given in Figure 7-4. Uncertainty is given by the 90% upper and lower credible interval (CI). The hypothetical risk level of concern (1E-05) corresponds with the 90% upper CI at a PRG of ~ 25 pCi/g, and the 90% lower CI at a PRG of ~ 250 pCi/g.

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