United States
Environmental Protection
Agency

Office of Research and
Development
Washington, DC 20460

EPA/600/6-88/005B
August 1992
Workshop Review Draft

EPA

Estimating Exposure to Dioxin-Like Compounds

Review Draft (Do Not Cite or Quote)

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U.S. Environmental Protection Agency
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CONTENTS

Tables viii
Figures xii
Foreword xiii
Preface xiv
Authors and Reviewers xv

1. INTRODUCTION 1-1

2. PHYSICAL AND CHEMICAL PROPERTIES 2-1

2.1. INTRODUCTION 2-1
2.2. GENERAL INFORMATION 2-2
2.3. PHYSICAL/CHEMICAL PROPERTIES - CHLORINATED COMPOUNDS 2-4
2.3.1. Water Solubility 2-5
2.3.2. Vapor Pressure 2-6
2.3.3. Henry's Law Constant 2-7
2.3.4. Octanol/Water Partition Coefficient 2-8
2.3.5. Photo Quantum Yields 2-9
2.4. PHYSICAL/CHEMICAL PROPERTIES - BROMINATED COMPOUNDS 2-10
2.5. FATE - CHLORINATED COMPOUNDS 2-10
2.5.1. Polychlorinated Dibenzo-p-dioxins (PCDDs) 2-10
2.5.2. Polychlorinated Dibenzofurans (PCDFs) 2-16
2.5.3. Coplanar PCBs 2-18
2.6. FATE - BROMINATED COMPOUNDS 2-20

3. ENVIRONMENTAL LEVELS OF PCDD, PCDF, AND PCB CONGENERS . . . 3-1

3.1. INTRODUCTION 3-1
3.2. CONCENTRATIONS IN SOIL 3-2
3.3. CONCENTRATIONS IN WATER 3-5
3.4. CONCENTRATIONS IN SEDIMENT 3-7
3.5. CONCENTRATIONS IN FISH AND SHELLFISH 3-12
3.6. CONCENTRATIONS IN FOOD PRODUCTS 3-18
3.7. CONCENTRATIONS IN AIR 3-19
3.8. HISTORICAL TRENDS 3-20
3.9. ASSESSMENT OF BACKGROUND EXPOSURES 3-21
3.9.1. Procedure for Estimating Background Exposures 3-21
3.9.2. Geographic Comparisons 3-21

4. ESTIMATING EXPOSURES AND RISKS 4-1

4.1. INTRODUCTION 4-1
4.2. EXPOSURE EQUATION 4-1
4.3. RISK EQUATION 4-2
4.4. PROCEDURE FOR ESTIMATING EXPOSURE 4-5

CONTENTS (continued)

5. ESTIMATING EXPOSURE MEDIA CONCENTRATIONS 5-1

5.1. INTRODUCTION 5-1
5.2. CONSIDERATIONS FOR SOLUTION ALGORITHMS 5-7
5.3. ALGORITHMS FOR THE "ON-SITE SOIL" SOURCE CATEGORY 5-8
5.3.1. Surface Water and Sediment Contamination 5-9
5.3.2. Vapor-Phase Air Concentrations 5-20
5.3.3. Particulate-Phase Air Concentrations 5-24
5.3.4. Biota Concentrations 5-27
5.3.4.1. Fish Concentrations 5-27
5.3.4.2. Vegetation Concentrations 5-38
5.3.4.3. Beef and Milk Concentrations 5-48
5.4. ALGORITHMS FOR THE "OFF-SITE" SOURCE CATEGORY 5-53
5.4.1. Exposure Site Soil Concentrations 5-56
5.4.2. Vapor-Phase Transport 5-62
5.5. ALGORITHMS FOR THE INCINERATOR STACK EMISSION SOURCE CATEGORY 5-65
5.5.1. Steady-State Soil Concentrations 5-67
5.6. ALGORITHMS FOR THE ASH DISPOSAL IN LANDFILLS SOURCE CATEGORY 5-68
5.6.1. Landfill Size 5-69
5.6.2. Fugitive Particulate Emissions at the Landfill 5-70
5.6.2.1. Vehicular Traffic Over Landfill Roadways 5-70
5.6.2.2. Fugitive Emissions From Trucks 5-75
5.6.2.3. Emissions From Unloading 5-77
5.6.2.4. Emissions From Spreading and Compacting 5-79
5.6.2.5. Total Particulate Emissions From Ash Landfills 5-80

6. MUNICIPAL SOLID WASTE INCINERATION 6-1

6.1. INTRODUCTION 6-1
6.2. OVERVIEW OF PRINCIPAL MUNICIPAL INCINERATION TECHNOLOGIES IN THE UNITED STATES 6-2
6.3. THE THEORETICAL MECHANISMS GIVING RISE TO THE AIR EMISSIONS OF DIOXIN-LIKE COMPOUNDS DURING MSW INCINERATION 6-4
6.4. HYPOTHETICAL MSW INCINERATOR FOR PURPOSES OF EXPOSURE ANALYSIS 6-10
6.5. DERIVATION OF MASS EMISSION FACTORS FOR THE HYPOTHETICAL MASS BURN, HEAT RECOVERY MSW INCINERATOR 6-12

CONTENTS (continued)

6.5.1. Congener-Specific Stack Gas Emission Factors for Dioxin-Like Compounds 6-12
6.5.2. Estimation of Emission of Dioxin-Like Compounds in MSW Incineration Ash 6-25
6.6. AIR DISPERSION MODELING OF THE STACK GAS EMISSIONS OF DIOXIN-LIKE COMPOUNDS FROM THE HYPOTHETICAL MSW INCINERATOR 6-28
6.6.1. Estimation of Distribution in the Stack Emissions to the Hypothetical MSW Incinerator 6-34
6.7. RESULTS OF THE AIR DISPERSION MODELING OF CONGENER-SPECIFIC EMISSIONS FROM THE HYPOTHETICAL MSW INCINERATOR IN TAMPA, FLORIDA 6-41

7. EXPOSURE SCENARIO DEVELOPMENT 7-1

7.1. INTRODUCTION 7-1
7.2. STRATEGY FOR DEVISING EXPOSURE SCENARIOS 7-1
7.3. EXPOSURE PATHWAYS 7-3
7.3.1. Soil Ingestion 7-5
7.3.2. Soil Dermal Contact 7-6
7.3.3. Vapor and Dust Inhalation 7-7
7.3.4. Water Ingestion 7-8
7.3.5. Beef and Dairy Product Ingestion 7-8
7.3.6. Fish Ingestion 7-10
7.3.7. Fruits and Vegetables 7-11

8. PHARMACOKINETICS 8-1

8.1. INTRODUCTION 8-1
8.2. DAILY BACKGROUND LEVELS 8-2
8.2.1. Basis for Calculation 8-2
8.2.2. Daily Intakes 8-6
8.3. COMPARTMENTAL MODELING 8-11
8.3.1. Pharmacokinetic Model 8-12
8.3.2. Model Utilization 8-14
8.3.3. Determining Liver Concentrations From Fat Levels 8-18
8.4. BIOAVAILABILITY AND TISSUE DISTRIBUTION 8-20
8.4.1. Bioavailability 8-21
8.4.1.1 Bioavailability Data 8-21
8.4.1.2. Summary of Bioavailability 8-33
8.4.2. Distribution 8-35
8.4.3. Determination of Daily Intake Dose From Exposure Concentrations 8-36

CONTENTS (continued)

9. DEMONSTRATION OF METHODOLOGY 9-1

9.1. INTRODUCTION 9-1
9.2. STRATEGY FOR DEVISING EXPOSURE SCENARIOS FOR DEMONSTRATION PURPOSES 9-2
9.3. EXAMPLE EXPOSURE SCENARIOS 9-8
9.4. EXAMPLE COMPOUNDS 9-11
9.5. SOURCE TERMS 9-12
9.6. RESULTS 9-16
9.6.1. Observations Concerning Exposure Media Concentrations 9-16
9.6.2. Observations Concerning LADD Exposure Estimates 9-27

10. UNCERTAINTY 10-1

10.1. INTRODUCTION 10-1
10.2. UNCERTAINTIES IN SPECIFIC METHODS APPLIED 10-3
10.2.1. Environmental Chemistry of Dioxin-Like Compounds 10-3
10.2.2. Lifetime, Body Weights, and Exposure Durations 10-6
10.2.3. Soil Erosion Algorithm 10-7
10.2.3.1. Comparison with Literature Data 10-8
10.2.3.2. Key Parameters and Assumptions of the Erosion Algorithm 10-13
10.2.4. Surface Water Suspended and Bottom Sediments 10-18
10.2.4.1. Comparison with Literature Data 10-18
10.2.4.2. An Alternate Approach 10-21
10.2.4.3. Sensitivity to Key Parameters and Assumptions 10-22
10.2.5. Soil Ingestion Exposure 10-26
10.2.6. Soil Dermal Contact Pathway 10-28
10.2.7. Water Ingestion 10-31
10.2.7.1. Exposure Parameters 10-31
10.2.7.2. Comparison of Estimated Water Concentrations With Observed Data 10-34
10.2.7.3. An Alternate Modeling Approach for Estimating Water Concentrations 10-35
10.2.7.4. Sensitivity to Model Parameters 10-38
10.2.8. Fish Ingestion Exposure 10-39
10.2.8.1. Fish Ingestion Rates 10-39
10.2.8.2. Comparison of Estimated Fish Concentrations With Literature Values 10-41
10.2.8.3. Alternate Modeling Approaches 10-45
10.2.8.4. Sensitivity to Key Model Parameters 10-49
10.2.9. Vapor Phase Inhalation Exposures 10-51
10.2.9.1. Comparison of Model Results With Measured Concentrations 10-52

CONTENTS (continued)

10.2.9.2. Other Modeling Approaches and Considerations for Vapor-Phase Emissions 10-55
10.2.9.4. Uncertainty of Key Model Parameters 10-59
10.2.10. Particulate Phase Inhalation 10-62
10.2.10.1. Wind Erosion Modeling 10-63
10.2.10.2. Fugitive Ash Emissions 10-65
10.2.11. Uncertainty in Fruit and Vegetable Ingestion 10-66
10.2.11.1. Fruit and Vegetable Ingestion Rates 10-67
10.2.11.2. Comparing Predicted Versus Measured Plant Concentrations 10-70
10.2.11.3. Uncertainty Evaluation of the Vegetation Concentration Algorithm 10-75
10.2.12. Beef and Milk Ingestion 10-82
10.2.12.1. Beef and Milk Ingestion Rates 10-83
10.2.12.2. Comparison of Modeled Beef and Milk Concentrations With Concentrations Found . 10-85
10.2.12.3. Alternate Modeling Approaches for Estimating Beef and Milk Concentrations . . . 10-88
10.2.12.4. Uncertainty of Model Parameters Estimating Beef and Milk Concentrations 10-95
10.3. UNCERTAINTY AND VARIABILITY IN ANALYSIS OF DIOXIN-LIKE COMPOUNDS EMITTED FROM MUNICIPAL SOLID WASTE INCINERATION 10-98
10.3.1. The Probability of Emission of Dioxin-Like Compounds . . . 10-99
10.3.2. The Quantity of Release of Dioxin-Like Compounds from the Stack of the Incinerator 10-101
10.3.3. The Representativeness of the Hypothetical Incinerator in Comparison to Actual Facilities 10-105
10.3.4. Air Dispersion Analysis of Incinerator Emissions 10-109
10.3.5. Comparison of Predicted Ambient Concentrations to Ambient Measurements 10-112

11. CONCLUSIONS AND RECOMMENDATIONS 11-1

11.1. CONCLUSIONS 11-1
11.2. RECOMMENDATIONS 11-7

APPENDIX A: ENVIRONMENTAL CHEMISTRY A-1

APPENDIX B: ENVIRONMENTAL CONCENTRATIONS B-1

APPENDIX C: SPREADSHEET ANALYSIS AND MODEL PARAMETERS C-1

APPENDIX D: MODELING THE IMPACT OF EFFLUENT DISCHARGES INTO SURFACE WATERS D-1

TABLES

1-1 Toxicity equivalency factors (TEF) 1-4

1-2 Dioxin-Like PCBs 1-5

1-3 Nomenclature for dioxin-like compounds 1-6

2-1 Possible number of positional CDD (or BDD) and CDF (or BDF) congeners 2-3

3-1 Regional comparisons of toxic equivalent concentrations 3-23

3-2 Background exposures 3-24

5-1 Summary and fate and transport solution algorithms for estimating exposure media concentrations for the four source categories 5-3

5-2 Available Biota to Sediment Accumulation Factors, BSAF, for dioxin-like compounds 5-33

5-3 Summary of Biota Sediment Accumulation Factors, BSAFs, for PCBs. 5-36

6-1 Concentration of PCDD/PCDF (ng/g) on municipal incinerator fly ash at varying temperatures 6-7

6-2 Capacity distribution (TDP) of existing and projected mass burn, heat recovery MSW incinerators in the United States 6-11

6-3 Modeling parameters for the hypothetical MSW incinerator 6-12

6-4 Estimation of PCDD congener-specific emission factors (grams/metric ton of MSW burned) for representative mass burn heat recovery incinerators 6-16

6-5 Estimation of PCDF congener-specific emission factors (grams/metric ton of MSW burned) for representative mass burn heat recovery incinerators 6-18

6-6 Coplanar polychlorinated biphenyl congeners having dioxin-like biological activity; number of possible isomers; ratio of toxic isomer to total isomers in homologue group 6-22

6-7 Emission factors (grams/metric ton) of homologue groups of polychlorinated biphenyls from emission tests of MSW incinerators . . 6-23

TABLES (continued)

6-8 Estimation of congener-specific emission factors of coplanar PCBs . . 6-24

6-9 Emission factors (grams/metric ton) of PCDDs/PCDFs for collected fly ash in the ESP control scenario 6-29

6-10 Emission factors (grams/metric ton of MSW incinerated) of coplanar PCBs for collected fly ash in the ESP control scenario . . . 6-29

6-11 Emission factors (grams/metric ton of MSW incinerated) of PCDDs/PCDFs for collected fly ash in the dry-scrubber, fabric filter control scenario 6-30

6-12 Emission factors (grams/metric ton of MSW incinerated) of coplanar PCBs for collected fly ash in the dry-scrubber, fabric filter control scenario 6-30

6-13 Comparison of particle-size distribution in particulate emissions resulting from ESP controls with particulate emissions from fabric filter controls 6-35

6-14 Congener-specific emission factors (grams/sec) as input to the ISC air dispersion model for both ESP and DSFF control scenarios 6-39

6-15 Congener-specific emission factors by particle size (grams/sec) for both the ESP and dry scrubber-fabric filter (DSFF) control scenarios to the hypothetical MSW incinerator 6-40

6-16 Predicted annual average ambient air concentrations (grams/m^3) of specific dioxin-like congeners at specified distances from the hypothetical MSW incinerator--ESP controls 6-44

6-17 Predicted annual average air concentrations (grams/m^3) of specific dioxin-like congeners at specified distances from the hypothetical MSW incinerator--DSFF controls 6-45

6-18 Wet+dry surface deposition (grams/m^2-yr) of specific congeners at specified distances in the vicinity of the hypothetical MSW incinerator--ESP controls 6-47

6-19 Wet+dry surface deposition (grams/m^2-yr) of specific congeners at specified distances in the vicinity of the hypothetical MSW incinerator--DSFF controls 6-48

7-1 Summary of exposure pathway parameters 7-13

TABLES (continued)

8-1	Calculated Daily Intakes for 2,3,7,8-TCDD	8-7
8-2	Half life calculations	8-11
8-3	Model determined daily intakes	8-18
9-1	Summary of key source terms for the six exposure scenarios and the three example compounds	9-13
9-2	Exposure media concentrations estimated for all scenarios and pathways	9-17
9-3	Lifetime average daily dose (LADD) estimates for all scenarios and exposure pathways (all results in mg/kg-day)	9-20
9-4	Percent contribution of the different exposure pathways within each exposure scenario	9-29
9-5	Exposures to low soil concentrations of 2,3,7,8-TCDD assuming lifetime exposure durations and unlimited contact with impacted media, compared with exposures assuming limited durations and limited contact	9-31
9-6	Comparison of exposure pathway contributions to total daily exposure as estimated in example Scenario #2 and in Travis and Hattemer-Frey (1991)	9-34
10-1	Uncertainties associated with the lifetime, body weight, and exposure duration parameters	10-8
10-2	Summary of uncertainties associated with the soil delivery algorithm	10-9
10-3	Summary of off-site soil contamination from Tier 1 and 2 sites of the National Dioxin Study	10-12
10-4	Summary of uncertainties associated with the surface water sediment algorithms	10-19
10-5	Uncertainties associated with the soil ingestion pathway	10-29
10-6	Uncertainties associated with the dermal exposure pathway	10-32
10-7	Uncertainties associated with the water ingestion pathway	10-33

TABLES (continued)

10-8 Uncertainties associated with the fish ingestion pathway 10-40

10-9 Uncertainties and sensitivities associated with estimating vapor-phase air concentrations from contaminated soils 10-53

10-10 Uncertainties and sensitivities associated with estimating particulate-phase air concentrations from contaminated soils 10-64

10-11 Uncertainties associated with vegetable and fruit ingestion exposure algorithms 10-68

10-12 Summary of plant concentration vs. soil concentration data for 2,3,7,8-TCDD 10-72

10-13 Contribution of above ground vegetation concentrations from air-to-leaf transfers and particulate depositions 10-78

10-14 Uncertainties associated with beef and milk ingestion exposure algorithms 10-84

10-15 Estimated relative importance of soil, pasture grass, and fodder ingestion to the estimation of beef and milk concentration when soil and incinerator emissions are the source of 2,3,7,8-TCDD contamination 10-92

10-16 Basic statistical analysis of emissions of dioxin used in the hypothetical MSW incinerator scenarios 10-104

10-17 Comparison between measured ambient air concentrations with predicted ambient air impacts of the hypothetical MSW incinerator (values as pg/m^3) 10-114

FIGURES

4-1 Roadmap for assessing exposure and risk to dioxin and dioxin-like compounds 4-6

5-1 Watershed delivery ratio, SD_w, as a function of watershed size 5-17

6-1 Principal physical processes used by the ISC dispersion model to estimate the atmospheric transport and deposition of pollutants discharged from the smokestack 6-31

6-2 Dispersion gradient of the ground-level concentration with distance from the incinerator--ESP control scenario 6-43

6-3 Dispersion gradient of the ground-level concentration with distance from the incinerator--DSFF control scenario 6-43

8-1 Sample Calculation of Daily Intake for 2,3,7,8-TCDD 8-7

8-2 Model estimates of elimination of 2,3,7,8-TCDD from fat 8-15

8-3 Accumulation of TCDD in fat with 0.44 pg/kg/day dose - human 8-16

8-4 Accumulation of TCDD in fat with 0.30 pg/kg/day dose - human 8-17

10-1 Comparison of relative distributions of MSW stack emissions of PCB congeners as derived by Tiernan 10-102

10-2 Percent distribution of existing incinerators 10-107

10-3 Percent distribution of incinerators by 1995 10-107

10-4 Capacity distribution of existing and planned mass burn incinerators 10-108

10-5 Scatter plot of stack height relative to capacity 10-108

FOREWORD

The Exposure Assessment Group (EAG) within the Office of Health and Environmental Assessment of EPA's Office of Research and Development has three main functions: (1) to conduct exposure assessments, (2) to review assessments and related documents, and (3) to develop guidelines for exposure assessments. The activities under each of these functions are supported by and respond to the needs of the various program offices. In relation to the third function, EAG sponsors projects aimed at developing or refining techniques used in exposure assessments.

The purpose of this document is to present procedures for conducting site-specific exposure assessments to dioxin-like compounds. It serves as a final version of the 1988 draft document titled "Estimating Exposure to 2,3,7,8-TCDD." This effort represents a substantial expansion in scope to include all compounds that exhibit dioxin-like toxicity. The types of sites covered in this document include incinerators, landfills, and other areas involving contaminated soils. The procedures identify possible exposure pathways associated with these sites, present fate models to estimate media concentrations at the point of exposure, and suggest ways to estimate contact rates and resulting exposure levels. Hypothetical examples are used to illustrate the procedures.

The document is intended to be used as a companion to the health reassessment of dioxin-like compounds that the Agency is publishing concurrently. It is hoped that these two documents will improve the accuracy and validity of risk assessments involving this important family of compounds.

Michael A. Callahan
Director
Exposure Assessment Group

PREFACE

The Exposure Assessment Group of the Office of Health and Environmental Assessment has prepared this guidance document for general use throughout the agency. The purpose of this document is to present procedures for conducting site-specific exposure assessments to dioxin-like compounds. It serves as a final version of the 1988 draft document titled "Estimating Exposure to 2,3,7,8-TCDD" (U.S. EPA, 1988).

AUTHORS AND REVIEWERS

The Exposure Assessment Group (EAG) within EPA's Office of Health and Environmental Assessment was responsible for the preparation of this document. General support was provided by VERSAR Inc. under EPA Contract Number 68-DO 0101. Matthew Lorber of EAG served as EPA task manager (as well as contributing author) providing overall direction and coordination of the production effort as well as technical assistance and guidance.

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1. INTRODUCTION

The primary purpose of this document is to present procedures for conducting site-specific exposure assessments to dioxin-like compounds. In addition, information is also provided on the levels of these compounds found in various media, identification of the possible associated sources, and estimation of the resulting exposure levels.

This document serves as a final version of the 1988 draft document entitled "Estimating Exposure to 2,3,7,8-TCDD" (EPA, 1988). However, due to the expansion in scope to cover all dioxin-like compounds and numerous updates, it appears significantly different than the original draft. The draft document was reviewed by the Science Advisory Board in 1988 and their comments have been addressed in this assessment. Additional peer reviews of earlier drafts of this report were conducted in 1992.

Dioxin-like compounds are defined to include those compounds with nonzero Toxicity Equivalency Factor (TEF) values as defined in the 1989 International scheme. As shown in Table 1-1, this TEF scheme assigns nonzero values to all chlorinated dibenzodioxins (CDDs) and chlorinated dibenzofurans (CDFs) with chlorines substituted in the 2,3,7,8 positions. Additionally, the analogous brominated compounds (BDDs and BDFs) and certain polychlorinated biphenyls (PCBs, see Table 1-2) have recently been identified as having dioxin-like toxicity (EPA, 1992a) and thus are also included in the definition of dioxin-like compounds. The nomenclature adapted here for purposes of describing these compounds is summarized in Table 1-3.

The types of sites covered in this document include incinerators, landfills and other areas involving contaminated soils. The procedures identify possible exposure pathways associated with these sites, present fate models to estimate media concentrations at the point of exposure and identify ways to estimate contact rates and resulting exposure levels. Hypothetical examples are used to illustrate the procedures. A brief discussion is presented on procedures specific to effluent discharges to water bodies. For further details on assessing these sources, the reader is directed to EPA's recent document titled "Analysis of Populations at Risk from the Consumption of Dioxin-Contaminated Fish Caught Near Bleached Pulp and Paper Mills" (EPA, 1992b).

The end products of the exposure assessment procedures presented in this document are estimates of potential dose expressed in mg of toxicity equivalents/kg-day.

Pharmacokinetic models are presented to predict dioxin levels in blood, adipose tissue or other body compartments. The procedures for converting these dose estimates to risk estimates are provided in a companion document (EPA, 1992a) on health assessment which EPA is publishing concurrently and which addresses the same compounds.

The scope of each chapter is summarized below.

Chapter 2, Physical and Chemical Properties, summarizes the information on physical and chemical properties of the dioxin-like compounds.

Chapter 3, Occurrence, summarizes the levels of dioxin-like compounds found in various media.

Chapter 4, Estimating Exposure and Risks, presents overall framework for conducting exposure assessments. Procedures for using the Toxicity Equivalency Factors in exposure assessments are discussed here.

Chapter 5, Estimating Exposure Media Concentrations, provides procedures for estimating concentrations of the dioxin-like compounds in exposure media (soil, air, water, biota) resulting from soil contamination and nearby incinerators.

Chapter 6, Municipal Solid Waste Incineration, provides procedures to estimate the emission rates of dioxin-like compounds from municipal waste incinerators, including stack and fly ash emissions. It also demonstrates the use of the Industrial Source Complex model on a hypothetical incinerator and lists the associated atmospheric dispersion and deposition estimates from that model exercise.

Chapter 7, Exposure Scenario Development, provides procedures for identifying exposure pathways, estimating contact rates and resulting exposure levels. Approaches for defining exposure scenarios are presented.

Chapter 8, Pharmacokinetics, summarizes information about uptake and distribution of dioxin-like compounds in the body and presents pharmacokinetic models to predict blood levels resulting from exposure. Additionally, information on blood levels is used to back calculate associated exposure levels.

Chapter 9, Demonstration of Methodology, develops hypothetical scenarios and generates exposure estimates to demonstrate the methodologies of this document.

Chapter 10, Uncertainty, discusses the sources and possible magnitude of uncertainty in the exposure assessment procedures. Uncertainty and variability of fate and transport, and exposure parameters, are discussed. Modeled exposure media

concentrations are compared with concentrations that have been found in the literature, and alternate modeling approaches are demonstrated and compared with modeling approaches discussed in Chapter 5 and demonstrated in Chapter 9.

Chapter 11, Conclusions and Recommendations, presents conclusions concerning exposure to dioxin-like arrived at through the analyses conducted throughout the document and recommends areas for further research.

Appendix A, Tables of Chemical Properties, summarizes congener specific data on a variety of chemical properties

Appendix B, Tables of Media Levels, summarizes congener specific data on a levels of congeners in various media.

Appendix C, Spreadsheet Analysis and Model Parameters, explains use of spreadsheets designed to accompany this report and lists model parameters.

Appendix D, Modeling the Impact of Effluent Discharges Into Surface Waters, discusses procedures for modeling the fate and transport of dioxin-like-compounds discharged via pipes to water bodies.

Table 1-1. Toxicity equivalency factors (TEF)

Compound	TEF
Mono-, Di-, and Tri-CDDs	0
2,3,7,8-TCDD	1
Other TCDDs	0
2,3,7,8-PeCDD	0.5
Other PeCDDs	0
2,3,7,8-HxCDD	0.1
Other HcCDDs	0
2,3,7,8-HpCDD	0.01
Other HpCDDs	0
OCDD	0.001
Mono-, Di-, and Tri-CDFs	0
2,3,7,8-TCDF	0.1
Other TCDFs	0
1,2,3,7,8-PeCDF	0.05
2,3,4,7,8-PeCDF	0.5
Other PeCDFs	0
2,3,7,8-HxCDF	0.1
Other HcCDFs	0
2,3,7,8-HpCDF	0.01
Other HpCDFs	0
OCDF	0.001

Source: EPA, 1989.

Table 1-2. Dioxin-Like PCBs

IUPAC No.	Congener
77	3,3′,4,4′-tetra PCB
81	3,4,4′,5-tetra PCB
105	2,3,3′,4,4′-penta PCB
114	2,3,4,4′,5-penta PCB
118	2,3′,4,4′,5-penta PCB
126	3,3′,4,4′,5-penta PCB
156	2,3,3′,4,4′,5-hexa PCB
157	2,3,3′,4,4′,5′-hexa PCB
167	2,3′,4,4′,5,5′-hexa PCB
169	3,3′,4,4′,5,5′-hexa PCB
189	2,3,3′,4,4′,5,5′-hepta PCB

Table 1-3. Nomenclature for dioxin-like compounds

Term/Symbol	Definition
Congener	Any one particular member of the same chemical family; e.g., there are 75 congeners of chlorinated dibenzo-p-dioxins.
Homologue	Group of structurally related chemicals that have the same degree of chlorination. For example, there are eight homologues of CDDs, monochlorinated through octochlorinated.
Isomer	Substances that belong to the same homologous class. For example, there are 22 isomers that constitute the homologues of TCDDs.
Specific Congener	Denoted by unique chemical notation. For example, 2,4,8,9-tetrachlorodibenzofuran is referred to as 2,4,8,9-TCDF.
D	Symbol for homologous class: dibenzo-p-dioxin
F	Symbol for homologous class: dibenzofuran
M	Symbol for mono, i.e., one halogen substitution
D	Symbol for di, i.e., two halogen substitution
Tr	Symbol for tri, i.e., three halogen substitution
T	Symbol for tetra, i.e., four halogen substitution
Pe	Symbol for penta, i.e., five halogen substitution
Hx	Symbol for hexa, i.e., six halogen substitution
Hp	Symbol for hepta, i.e., seven halogen substitution
O	Symbol for octo, i.e., eight halogen substitution
CDD	Chlorinated dibenzo-p-dioxins, halogens substituted in any position
CDF	Chlorinated dibenzofurans, halogens substituted in any position
PCB	Polychlorinated biphenyls
2378	Halogen substitutions in the 2,3,7,8 positions

Source: EPA, 1989.

REFERENCES FOR CHAPTER 1

U.S. Environmental Protection Agency. (1988) Estimating exposure to 2,3,7,8-TCDD. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, DC; EPA/600/6-88/00SA.

U.S. Environmental Protection Agency. (1989) Interim procedures for estimating risks associated with exposures to mixtures of chlorinated dibenzo-p-dioxins and -dibenzofurans (CDDs and CDFs) and 1989 update. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC; EPA/625/3-89/016.

U.S. Environmental Protection Agency. (1992a) Health reassessment of dioxin-like compounds. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, DC.

U.S. Environmental Protection Agency. (1992b) Analysis of populations at risk from the consumption of dioxin-contaminated fish caught near bleached pulp and paper mills. U.S. Environmental Protection Agency, Office of Technology Transfer, Washington, DC.

2. PHYSICAL AND CHEMICAL PROPERTIES AND FATE

2.1. INTRODUCTION

This chapter summarizes available information regarding the physical and chemical properties and fate of the CDDs, CDFs, BDDs, BDFs, and coplanar PCBs, with an emphasis on the subset of these chemicals defined as dioxin-like in Chapter 1. Physical/chemical properties addressed in this chapter include melting point, water solubility, vapor pressure, Henry's Law constant, octanol/water partition coefficient and photochemical quantum yield. Fate and transport processes addressed in this chapter include photolysis, oxidation, hydrolysis, biodegradation, volatilization, and sorption. Biologically-mediated transport properties (i.e., bioconcentration, plant uptake, etc.) are covered in Chapter 5.

Knowledge of basic physical and chemical properties is essential to understanding and modeling environmental transport and fate as well as pharmacokinetic and toxicologic behavior. The most essential parameters for the dioxin and dioxin-like compounds appear to be water solubility (WS), vapor pressure (VP), octanol/water partition coefficient (Kow) and photochemical quantum yield. The ratio of VP to WS (VP/WS) yields the Henry's Law constant (H_c) for dilute solutions of organic compounds, an index of partitioning for a compound between the atmospheric and the aqueous phase (Mackay et al., 1982).

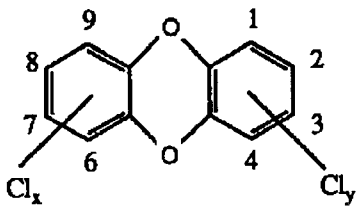
To maximize and optimize the identification of information on the physical/chemical properties of these compounds, a thorough search of the recent literature was conducted. A computer literature search was conducted in the on-line Chemical Abstracts (CA) database maintained by the Scientific Technical Network (STN), printed abstracts were obtained and screened, selected hard-copy of literature were retrieved and critically evaluated, and the appropriate value for each physical/ chemical property for each congener was selected. These property values are summarized in Tables A-1 and A-2 in Appendix A. Table A-1 lists the property values for the dioxin-like compounds, and Table A-2 lists the property values for all CDDs, CDFs, and coplanar PCBs. Where technically feasible, estimation procedures have been used to provide values where measured data are not available. Where data could not be found and estimates are not appropriate, the field is left blank and a homologue group average is presented as the property for that homologue group.

Brief summaries of the scientific literature on the physical-chemical properties and fate of the polychlorinated and polybrominated dibenzodioxins, dibenzofurans, and biphenyls are provided in the following sections. The data presented have been chosen based on the adequacy of the experimental methods, the reliability of the estimation procedures, and the credibility of the obtained results. No detailed discussions relative to earlier reported data and scientific findings are included here, although such discussions are usually offered in the original papers.

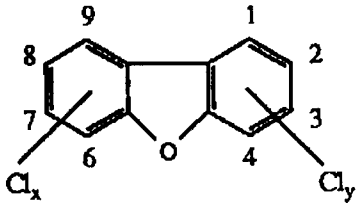
2.2. GENERAL INFORMATION

Polychlorinated dibenzodioxins (CDDs), polychlorinated dibenzofurans (CDFs), and polychlorinated biphenyls (PCBs) are chemically classified as halogenated aromatic hydrocarbons. CDDs and CDFs are formed as by-products through a variety of chemical reactions (e.g., dimerization of chlorophenols and phenoxy-acids to form chlorodioxins). Both compound classes have a triple-ring structure that consists of two benzene rings connected by a third ring. For dibenzodioxins, the connection of the benzene rings is through a pair of oxygen atoms as opposed to one oxygen atom for the dibenzofurans (see structures below). PCBs are a class of compounds formed by the chlorination of a biphenyl molecule.

The chlorinated and brominated dibenzodioxins and dibenzofurans are tricyclic aromatic compounds with similar physical and chemical properties, and both classes are quite similar structurally. There are 75 possible different positional congeners of CDDs and 135 different congeners of CDFs. Likewise, there are 75 possible different positional congeners of BDDs and 135 different congeners of BDFs (see Table 2-1). The basic structure and numbering of each chemical class is shown below.



PCDDs



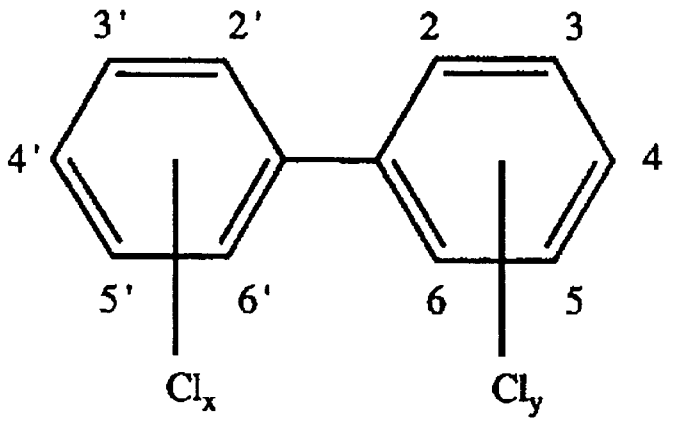
PCDFs

X = 1 to 4, Y = 1 to 4, X + Y $\geq$ 1

Table 2-1. Possible number of positional CDD (or BDD) and CDF (or BDF) congeners

	Number of Congeners		
Halogen substitution	CDDs (or BDDs)	CDFs (or BDFs)	PCBs
Mono	2	4	3
Di	10	16	12
Tri	14	28	24
Tetra	22	38	42
Penta	14	28	46
Hexa	10	16	42
Hepta	2	4	24
Octa	1	1	12
Nona	0	0	3
Deca	0	0	1

There are 209 possible PCB congeners (see Table 2-1). The physical/chemical properties of each congener vary according to the degree and position of chlorine substitution. The list of coplanar PCBs can be found in Table 1-2. PCBs assume a coplanar structure when the two benzene rings rotate into a position where the two rings are in the same plane. The PCBs assume a dioxin-like structure when the substituent chlorines occupy the 3, 3', 4, 4', 5, or 5' positions, or possibly, one of the 2 or 2' positions, and the structure is not hindered from assuming the preferred planar configuration. The basic structure and numbering of each chemical class is shown below.



X = 1 to 5, Y = 1 to 5, X + Y $\geq$ 1

2.3. PHYSICAL/CHEMICAL PROPERTIES - CHLORINATED COMPOUNDS

Limited research has been carried out to determine physical and chemical properties of PCDFs and PCDDs. The congeners having 2,3,7,8-chlorination have received the most attention with 2,3,7,8,-TCDD being the most intensely studied compound. Of the large number of possible isomers, very few are available commercially and preparation and synthesis can be both time consuming and difficult. In addition, many of these isomers have not been prepared in pure form. The isomers that have been prepared may not be available in sufficient quantities for testing. Another factor to be considered is the high toxicity of these compounds, which necessitates extreme precautions to prevent adverse effects if their syntheses are to be attempted.

2.3.1. Water Solubility

Very few measured water solubility values are available in the literature. Marple et al. (1986a) reported the water solubility of 2,3,7,8-TCDD as 19.3 ± 3.7 parts per trillion (nanograms per liter, ng/L) at 22°C (Table A-1). Marple et al. (1986a) used a procedure of equilibrating thin films of resublimed 2,3,7,8-TCDD with a small volume of water followed by gas chromatography (GC) analysis with 63Ni electron capture detection. Other water solubility values for 2,3,7,8-TCDD have been reported in the literature and are summarized in U.S. EPA (1990). Values ranging from 7.9 ng/L to 483 ng/L are reported in U.S. EPA (1990) with 19.3 ng/L selected as the recommended value.

Friesen et al. (1985) used a HPLC generator column to measure the water solubilities of a series of chlorinated dioxins (1,2,3,7- and 1,3,6,8-TCDD; 1,2,3,4,7-PCDD; 1,2,3,4,7,8-HxCDD; 1,2,3,4,6,7,8-HpCDD; and 1,2,3,4,6,7,8,9-OCDD) and reported water solubilities ranging from 430 ng/L to 0.4 ng/L for the 1,2,3,7-TCDD and 1,2,3,4,6,7,8,9-OCDD isomers, respectively (Table A-1). Friesen et al. (1990) used a gas chromatography/mass spectrometry detection (GC/MSD) generator column technique to measure the water solubilities of a series of chlorinated furans (2,3,7,8-TCDF; 2,3,4,7,8-PeCDF; 1,2,3,6,7,8- and 1,2,3,4,7,8-HxCDF; and 1,2,3,4,6,7,8-HpCDF) and again reported a decrease in water solubility with an increase in the number of chlorine substituents. The reported water solubility values ranged from 1.37×10^{-9} mol/L (419 ng/L) for the 2,3,7,8-TCDF isomer to 3.30×10^{-12} mol/L (1.35 ng/L) for the 1,2,3,4,6,7,8-HpCDF isomer (Table A-1).

The reported water solubility values for the PCB compounds are higher than those for the CDD and CDF compounds (Table A-1). The reported values range from 36.1 ng/L for 3,3',4,4',5,5'-HxCB to 2,920 ng/L for 3,4,4',5-TeCB. However, only one of the values listed in Table A-1 is a measured value, 569 ng/L for 3,3',4,4'-TeCB, and the other values are only estimates reported in the literature. The measured water solubility value was reported by Dickhut et al. (1986) based on a modified generator column method.

For those compounds without reported measured water solubility values, estimations were calculated by the homologue-average method. For example, for the tetra-chlorinated dioxins, values reported in the literature were averaged to yield an estimated water solubility value for the tetra-chlorinated dioxin homologue group. A similar procedure was used to develop the average value for each of the other

polychlorinated dioxin, and furan homologue groups. Estimating the water solubility values from measured log Kow values by using the estimation procedure of Lyman et al. (1982) did not yield satisfactory results; the estimated water solubilities for 2,3,7,8-TCDD and 1,3,6,8-TCDD were at least two orders of magnitude greater from the measured values in Tables A-1 and A-2. Compounds which have water solubility values in the ranges reported for these chlorinated compounds are considered to be very poorly soluble in water.

2.3.2. Vapor Pressure

Very few measured vapor pressure values are available in the literature for the CDDs and CDFs. The only measured vapor pressures available are for 2,3,7,8-TCDD, 1,2,3,4-TCDD, and 1,3,6,8-TCDD (Tables A-1 and A-2).

Similar to water solubility, U.S. EPA (1990) presented a variety of vapor pressure data for 2,3,7,8-TCDD, and selected a recommended value of 7.4×10^{-10} mm Hg at 25°C. This value was reported by Podoll et al. (1986) who used radiolabeled 2,3,7,8-TCDD and a gas saturation technique with combustion to $^{14}CO_2$. Rordorf (1987, 1989) reported a higher vapor pressure value for 2,3,7,8-TCDD, 1.49×10^{-9}mm Hg. SRC (1991) reported this same value by extrapolating the vapor pressures measured by Schroy et al. (1985b) at four higher temperatures, 30°, 55°, 62°, and 71°C. The value recommended in U.S. EPA (1990) is reported in Table A-1. Webster et al. (1985) used a gas saturation method to determine the vapor pressure for 1,3,6,8-TCDD, 4.03×10^{-6} mm Hg, but this value appears to be out of line with the other reported values, so it is not used in Table A-1.

Rordorf (1987, 1989) reported an experimental vapor pressure value for 1,2,3,4-TCDD, 4.8×10^{-8} mm Hg, 1,2,3,4,6,7,8,9-OCDD, 8.25×10^{-13}mm Hg, and 1,2,3,4,6,7,8,9-OCDF, 3.75×10^{-12} mm Hg (Table A-1). Rordorf (1987, 1989) used a gas flow method in a saturation oven, with integrated gas chromatographic analysis, to measure vapor pressure values for 10 PCDDs and 4 CDFs. Rordorf (1987, 1989) also used a vapor pressure correlation method to predict the vapor pressures of 15 other CDDs and 55 CDFs based on the measured vapor pressures for the 10 PCDDs, 4 CDFs, and the deduced boiling point and enthalpy data for the larger series of CDDs and CDFs. Measured boiling point and enthalpy data are in good agreement with the deduced data used in the correlation method. Of the CDDs studied by Rordorf (1987, 1989), only two of the ten,

1,2,3,4-TCDD and 2,3,7,8-TCDD, were in the dioxin-like compound group of chemicals studied in this report. The other CDDs with measured values were monochloro-, dichloro-, and trichloro-dibenzo-p-dioxins.

The vapor pressure values reported for the PCBs are higher than those reported for the CDD and CDF compounds (Table A-1), although all of the PCB and CDF values and most of the CDD values are calculated-estimated values reported in the literature. The values reported in Tables A-2 and A-3 are an average of the OV-101 RI and Dexsil 410 RI correlation methods because both methods were determined to be equally valid. As with the other groups, the vapor pressures of the PCBs decrease with an increase in the number of chlorine substituents. The highest value for PCBs in Table A-2 is 1.77×10^{-5} mm Hg for 3,4,4',5-TeCB and the lowest value reported is 3.04×10^{-7} mm Hg for a heptachloro-PCB.

Vapor pressure values for CDDs and CDFs that were not found in the literature were also calculated by the homologue-average method using the literature-reported values within a homologue group. For example, the literature values for the TCDDs were averaged to obtain an estimated vapor pressure assumed to apply to the TCDD congeners that did not have literature values. A similar procedure was used to develop a homologue-average for each of the other homologue groups. Compounds with vapor pressures in the ranges reported for these compounds are considered to have very low vapor pressures. However, volatilization is considered to be one of the important transport properties for chlorinated dioxins, furans, and biphenyls.

2.3.3. Henry's Law Constant

Measured data for Henry's Law constant have been reported for only two compounds: 1,3,6,8-TCDD at 6.81×10^{-5} atm-m^3/mol (Webster et al. 1985), and 3,3',4,4'-PCB at 9.4×10^{-5} atm-m^3/mol (Dunnivant and Elzerman 1988) (Tables A-1 and A-2). Both values were determined by the gas purging technique. All other values reported in the literature for CDDs, CDFs, and PCBs were calculated by the vapor pressure/water solubility ratio technique.

Group-average Henry's Law constants are estimated for each homologue group based on the reported data for that group. The Henry's Law constant values for the PCBs are similar to those for the CDDs and CDFs, except for the value estimated for

2,3,3',4,4',5,5'-HpCB (3.0×10^{-3} atm -m^3/mol) which is higher than most of the other reported values and homologue-averages.

Lyman et al. (1982) offers guidelines, though not specific to these compounds, for comparing the degree to which organic compounds volatilize from water. These guidelines suggest that volatilization of polycyclic aromatic hydrocarbons and halogenated aromatics (which includes all the dioxin-like compounds) from water represents a significant transfer mechanism from the aqueous to the atmospheric phase.

2.3.4. Octanol/Water Partition Coefficient

Marple et al. (1986b) reported the octanol/water partition coefficient of 2,3,7,8-TCDD as 4.24 ($\pm$ 2.73) x 10^6 at 22 $\pm$ 1°C, yielding a log Kow of 6.64 (Table A-1). Two similar experimental techniques were used, but the more reliable method involved equilibration of water-saturated octanol, containing the 2,3,7,8-TCDD, with octanol-saturated water, over 6 to 31 days. U.S. EPA (1990) reported that the available low Kow data ranged from 6.15 to approximately 8.5, and the 6.64 value reported by Marple et al. (1988b) was the value recommended in that report.

Burkhard and Kuehl (1986) used reverse-phase High Pressure Liquid Chromatography (HPLC) and Liquid Chromatography/Mass Spectrometry (LCMS) detection to determine octanol/water partition coefficients for 2,3,7,8-TCDD and a series of seven other tetrachlorinated planar molecules, including three other TCDD isomers (1,2,3,4-TCDD; 1,3,7,9-TCDD; 1,3,6,8-TCDD), 2,3,7,8-TCDF and 3,3',4,4'-tetrachlorobiphenyl. The log Kow values for the four TCDD isomers ranged from 7.02 to 7.20, the log Kow for the 2,3,7,8-TCDF was 5.82, and the log Kow for 3,3',4,4'-TCB was 5.81.

Burkhard and Kuehl (1986) also reevaluated data on 13 CDDs and CDFs previously reported by Sarna et al. (1984) under similar experimental techniques. In the reevaluation, Burkhard and Kuehl (1986) used experimental rather than estimated log Kow values in correlations with gas chromatographic retention times. This approach yielded log octanol-water partition coefficients ranging from about 4.0 for the non-chlorinated parent molecules to about 8.78 for the octa-chlorinated compounds, much lower than the values originally reported by Sarna et al. (1984).

Sijm et al. (1989) used a slow stirring method to obtain log Kow values for 73 CDD and CDF congeners. Values ranged from 6.10 to 7.92.

The measured and literature-estimated log Kow values for the PCBs are similar to those reported for the CDDs and CDFs (Table A-1). The values range from 6.2 (measured) for 3,3',4,4'-TeCB to 7.72 (literature-estimate) for 2,3,3',4,4',5,5'-HpCB. The log Kow values increase with an increase in the number of chlorine substituents. The log Kow for the 3,3',4,4'-TeCB was measured by Hawker and Connell (1988) using the generator column technique against the linear relationship of relative retention time on a nonselective gas chromatograph stationary phase. Log Kow values for the HxCBs were measured by Risley et al. (1990) using a high-performance liquid chromatographic (HPLC) system. The values reported in Tables A-2 and A-3 are an average of the two HPLC techniques because both methods were determined to be equally valid.

Partition coefficient values that were not found in the literature were calculated by averaging the literature values within homologue groups, as was done for vapor pressure and water solubility. Partition coefficients in the ranges of these reported values indicate that the substances tend to adsorb strongly to organic components in the soil and may bioconcentrate in those organisms exposed to the compounds.

Literature values for the hexachlorodibenzofurans could not be found.

2.3.5. Photo Quantum Yields

Photo quantum yields have been reported for only eight compounds: 1,2,3,7-TCDD at 5.42×10^{-4} (Choudhry and Webster, 1989)

1,3,6,8-TCDD at 2.17×10^{-3} (Choudhry and Webster, 1989)

2,3,7,8-TCDD at 2.2×10^{-3} (Dulin et al. 1986)

1,2,3,4,7-PeCDD at 9.8×10^{-5} (Choudhry and Webster, 1987)

1,2,3,4,7,8-HxCDD at 1.1×10^{-4} (Choudhry and Webster, 1987)

1,2,3,4,6,7,8-HpCDD at 1.53×10^{-5} (Choudhry and Webster, 1987)

1,2,3,4,6,7,8,9-OCDD at 2.26×10^{-5} (Choudhry and Webster, 1987)

1,2,4,7,8-PeCDF at 1.3×10^{-2} (Choudhry et al. 1990)

1,2,3,4,7,8-HxCDF at 6.96×10^{-4} (Choudhry et al. 1990)

All of the quantum yields were measured in a water-acetonitrile solution at 313 nm. No values were found for the PCBs.

Homologue group averages were not calculated because photo quantum yields are very sensitive to chlorine position and the solvent system used in the experiments. Different water to acetonitrile volume ratios were used in these experiments.

The photo quantum yields can be used to estimate the photolysis half-life of these compounds. This is further discussed in Section 2.4.

2.4. PHYSICAL CHEMICAL PROPERTIES - BROMINATED COMPOUNDS

Information on the physical and chemical properties of the polybrominated dioxins and furans is very limited. Until very recently, only estimated values were available, but in early 1992 Webster et al. (personal communication) will publish measured results for testing with brominated compounds. The data for these brominated compounds are listed in Table A-1.

2.5 FATE - CHLORINATED COMPOUNDS

2.5.1 Polychlorinated Dibenzo-p-dioxins (PCDDs)

With the exception of 2,3,7,8-TCDD, the environmental behavior of PCDDs has been studied very little. To what extent these studies with 2,3,7,8-TCDD apply to other PCDDs and, conversely, to what extent the results of studies on other PCDDs apply to 2,3,7,8-TCDD is uncertain in many instances. In general, PCDDs are extremely stable compounds under normal environmental conditions, are relatively immobile in the environment, and are primarily associated with particulate and organic materials. The only environmentally significant path for destruction of PCDDs is photodegradation.

Upon deposition of PCDDs onto surfaces, there can be a relatively high initial loss due to photodegradation and/or volatilization. Once PCDDs adsorb onto or move into soils or sediments, however, they are apparently strongly sorbed. Erosion and aquatic transport of sediment appears to be the dominant physical transport mechanism for sorbed PCDDs. Some studies have shown that there may be slow rates of vapor phase transport out of soils.

If discharged to water, PCDDs are expected to preferentially sorb to solids. Volatilization may also be a significant transport mechanism for non-sorbed PCDDs even though PCDDs have negligible vapor pressures. In the atmosphere, PCDDs are expected

to also preferentially sorb to airborne particulate matter, particularly the more chlorinated congeners. Removal mechanisms include dry and wet deposition and photodegradation.

- **Photodegradation**

The photochemistry of PCDDs has been reviewed by Miller and Zepp (1987), Choudry and Webster (1987), and Esposito et al. (1980). This section summarizes the key findings of those reviews and the results of recent environmentally significant studies.

Photodegradation appears to be the most significant natural degradation mechanism for PCDDs. PCDDs absorb electromagnetic radiation above 290 nm wavelength (i.e., the lower bound of the sun's radiation reaching the earth's surface) and, therefore, can be expected to be subject to photolysis by sunlight (EPRI 1983). It is expected that all PCDDs can be dechlorinated photolytically in the presence of a suitable hydrogen donor (Crosby et al. 1973, Crosby 1978). The major products of photolysis are lower chlorinated PCDDs. Numerous studies have demonstrated that photolysis is slow in water and on dry surfaces but is dramatically increased when organic solvents are present.

Dulin et al. (1986) studied the photolysis of 2,3,7,8-TCDD in various solutions under sunlight and artificial light. Using the results obtained in a water:acetonitrile solution (1:1, v/v) under sunlight conditions, Dulin et al. (1986) calculated the half-life of 2,3,7,8-TCDD in surface water in summer at 40 degrees north latitude to be 4.6 days. The quantum yield for photodegradation of 2,3,7,8-TCDD in water was three times greater under artificial light at 313 nm than under sunlight, and the artificial light photolysis quantum yield for hexane, a good hydrogen donor, was 20 times greater than for the water:acetonitrile solution, a poor hydrogen donor.

Podoll et al. (1986) used the Dulin et al. (1986) quantum yield data for the water:acetonitrile solution to calculate seasonal half-life values for dissolved 2,3,7,8-TCDD at 40 degrees north latitude in clear near-surface water. The seasonal values for half-lives were calculated to be 21 hours in summer, 51 hours in fall, 118 hours in winter, and 27 hours in spring. The authors noted, as do the authors of the studies described below, that sorption of 2,3,7,8-TCDD to suspended particulate would have the effect of slowing the photolysis rates.

Choudry and Webster (1989) studied the photolytic behavior under 313 nm light of a series of PCDDs in a water:acetonitrile solution (2:3, v/v). Assuming that the quantum

yields observed in these studies are the same as would be observed in natural waters, Choudry and Webster (1989) estimated the mid-summer half-life values at 40 degrees north latitude in clear near-surface water to be as follows: 1,2,3,7-TCDD (1.8 days); 1,3,6,8-TCDD (0.31 days); 1,2,3,4,7-PeCDD (15 days); 1,2,3,4,7,8-HxCDD (6.3 days); 1,2,3,4,6,7,8-HpCDD (47 days); and 1,2,3,4,6,7,8,9-OCDD (18 days). In addition, the authors also experimentally determined the sunlight photolysis half-life of 1,3,6,8-TCDD in pond water to be 3.5 days.

A recent study by Friesen et al. (1990) examined the photolytic behavior of 1,2,3,4,7-PeCDD and 1,2,3,4,6,7,8-HpCDD in water:acetonitrile (2:3, v/v) and in pond water under sunlight conditions at 50 degrees north latitude. The observed half-lives of these two compounds in the acetonitrile solution were 12 and 37 days, respectively, and were 0.94 and 2.5 days in pond water, respectively. Based on these results, the authors conclude that an indirect photolytic mechanism may dominate the photodegradation of PCDDs in natural waters.

Substantial research on the environmental persistence of 2,3,7,8-TCDD has been done as part of the decontamination of the area around the ICMESA chemical plant in Seveso, Italy, which was contaminated when a trichlorophenol reaction vessel overheated in 1976, blowing out the safety devices and spraying dioxin contaminated material into the environment. The levels of dioxin in the soil decreased substantially during the first six months following the accident, reaching a steady state of 1/5 to 1/11 of the initial levels (diDomenico et al. 1982). An experiment was conducted at this site to determine the effectiveness of photolysis in decontaminating surface deposits on foliage. Test plots were sprayed with olive oil to act as a hydrogen donor and the levels of dioxin on grass were found to be reduced by over 80 percent within nine days (Crosby 1981). The 2,3,7,8-TCDD in contaminated soil was also found to be photolabile in sunlight when the soil was suspended in an aqueous solution of a surfactant. The destruction of 8 ug/ml of 2,3,7,8-TCDD in 0.02 M hexadecylpyridinium chloride could be accomplished in 4 hours (Botre et al. 1978).

Photolysis in the gas phase also appears to be a degradative mechanism for PCDDs. However, because of the low volatility of PCDDs few studies have been attempted to measure actual rates of photodegradation. Also, if airborne PCDDs are predominantly sorbed to particulates, then vapor phase photolysis measurements may be unimportant.

In studies conducted at Stanford Research Institute, the half-life of gaseous 2,3,7,8-TCDD in sunlight was found to be 5 to 24 days, and may be the result of indirect photolysis through the attack of hydroxyl radicals on the dioxin. This mode of destruction would be similar to the attack of hydroxyl radicals on PCBs (Leifer et al. 1983).

Mill et al. (1987) reported preliminary photolysis experiments with vapor phase 2,3,7,8-TCDD. The half-life for vapor phase 2,3,7,8-TCDD in simulated sun was several hours. The photolysis of 2,3,7,8-TCDD sorbed onto small diameter fly ash particulates suspended in air was also measured. The results indicated that fly ash appears to confer photo-stability on 2,3,7,8-TCDD. There was little (8 percent) to no loss observed on the two fly ash samples after 40 hours of illumination.

Orth et al. (1989) conducted photolysis experiments with vapor-phase 2,3,7,8-TCDD under illumination with a light source and filters to achieve radiation in the UV region from 250 nm to 340 nm. Carrier gases included air, helium, and an isobutane/helium mixture. The rate constants in helium and air were very similar, 5.4×10^{-3} sec^{-1} and 5.9×10^{-3} sec^{-1}, which corresponds to a quantum yield in air of 0.033 ± 0.046. No products could be observed in the mass spectrometer, so Orth et al. (1989) postulated that the product might be sorbing to the surface of the photolysis cell and being lost from potential analysis. Further studies were suggested to study product sorption to surfaces and to determine any wave length dependence of the photoinduced loss across the absorption band studied.

Podoll et al. (1986) estimated the photolysis rate of 2,3,7,8-TCDD vapors in the atmosphere. Based on the quantum yield for photolysis in hexane, the half-life in summer sunlight at 40 degrees north latitude was calculated to be 58 minutes, but Podoll et al. (1986) stated this estimate is an upper limit.

- **Oxidation**

Stehl (1973) has suggested that 2,3,7,8-TCDD is probably stable to oxidation in the ambient environment. The reaction rates of hydroxyl (OH) radicals with PCDDs have not been measured. The low vapor pressures of these compounds makes direct measurements very difficult with currently available techniques. However, Podall et al. (1986) estimated the half-life of 2,3,7,8-TCDD vapor via OH oxidation in the atmosphere

to be 200 hours. Atkinson (1987) estimated the atmospheric lifetime for 2,3,7,8-TCDD due to the OH radical reaction to be about 3 days.

- **Hydrolysis**

There is no available evidence indicating that hydrolysis would be an operative environmental process for degradation of PCDDs (EPRI 1983, Miller and Zepp 1987).

- **Volatilization and Sorption**

Based on their very low vapor pressures and high K_{oc} values, PCDDs would be expected to sorb to rather than volatilize from soils, sediments, and other solids; however, due to the stability and persistence of PCDDs via other transformation and transport pathways, volatilization should not be ignored as a transport mechanism. Freeman and Schroy (1985) point out that low volatility chemicals may bind strongly with dry soil but, once a molecular microlayer of water covers the soil particles, the chemical should become more volatile; in addition, water vaporization may enhance the rate of chemical vaporization from a soil column.

Observations from the Seveso incident indicate that when 2,3,7,8-TCDD is deposited on the soil surface, both volatilization and photodegradation are initially rapid (DiDomenico et al. 1982). Once it has been washed into the soil, however, 2,3,7,8-TCDD is unaffected by photolysis, and volatilization is greatly reduced (Plimmer 1978).

Concerning the volatility of 2,3,7,8-TCDD from water, Podoll et al. (1986) utilized data for the molecular diffusivity in air and in water to calculate volatilization half-lives. The calculated volatilization half-life was about 32 days for ponds and lakes and about 15 days for rivers.

Palausky et al. (1986) injected 2,3,7,8-TCDD dissolved in various organic solvents into soil columns to determine the extent of vapor phase diffusion in soil and the effects of carrier medium on the extent of migration. No noticeable changes in 2,3,7,8-TCDD concentration profiles were observed after 30 days incubation at temperatures ranging from 0° to 20°; however, a measurable change was observed at 40°C. The extent of migration was related to solubility of 2,3,7,8-TCDD in the solvent, but a direct correlation between solubility and extent of migration was not found.

Puri et al. (1989) of the same lab as Palausky reported continuing research which indicates that 2,3,7,8-TCDD is highly sorptive, but that organic co-contaminants such as waste oil and surfactants may act to enhance the translocation of 2,3,7,8-TCDD. Also, the distribution coefficient of 2,3,7,8-TCDD in soils is dependent on the total organic matter, with higher sorption at higher soil organic levels, but the degree of sorption is also dependent on the types of organic matter present and their stability and solubility in the presence of the various co-contaminants.

Kapila et al. (1989) of the same lab as Palausky and Puri tested the soil migration of 2,3,7,8-TCDD in waste crankcase oil. Field experiments showed that 2,3,7,8-TCDD moved downward in the soil column, and this movement was attributed to application of water to simulate rainfall. The water application resulted in displacement of the waste oil components and 2,3,7,8-TCDD from the macropore spaces in the soil column. The formation of colloidal suspensions and the presence of 2,3,7,8-TCDD in these suspensions may play a significant role in the downward movement of 2,3,7,8-TCDD. Since the amount of 2,3,7,8-TCDD recovered from each column after 12 months was "approximately the same as the amount initially applied," it was inferred by Kapila et al. (1989) that there was little loss of 2,3,7,8-TCDD from the soil surface via volatilization or photolysis during the one-year period.

Muir et al. (1985) studied the fate of radiolabeled 1,3,6,8-TCDD in sandy loam soil under field conditions and in silty-clay pond and lake sediments under laboratory conditions. Under field conditions, the time for disappearance of a surface-applied dilute solution of 1,3,6,8-TCDD was in the 130 to 400 day range. The authors attributed the losses to volatilization and/or erosion because no downward movement in the soil and no biodegradation could be detected. In sediment, 80 percent of the intact chemical was still present after 675 days.

- **Biotransformation and Biodegradation**

Investigations on the biodegradability of PCDDs have focused on the microbial degradation of 2,3,7,8-TCDD. Arthur and Frea (1989) provide a comprehensive review of studies conducted during the 1970s and 1980s. Arthur and Frea (1989) conclude that 2,3,7,8-TCDD is recalcitrant to microbial degradation. Several of the major studies conducted during this period are discussed below.

Matsumura and Benezet (1973) tested approximately 100 strains of micro-organisms which had previously shown the ability to degrade persistent pesticides, and only five strains showed any ability to degrade 2,3,7,8-TCDD, based on autoradiographs of thin-layer chromatograms. Although it is possible that the less chlorinated dioxins are more susceptible to biodegradation, microbial action on 2,3,7,8-TCDD is very slow under optimum conditions (Hutter and Philippi 1982). Long-term incubations of radiolabeled 2,3,7,8-TCDD yielded no radioactivity in carbon dioxide traps after one year, and analyses of the cultures showed that at most, 1 to 2 percent of a potential metabolite (assumed to be an hydroxylated derivative of 2,3,7,8-TCDD) could be detected. Camoni et al. (1982) added organic compost to contaminated soil from the Seveso area in an attempt to enrich the soil and enhance the 2,3,7,8-TCDD biodegradation rate, but the soil amendment had no clear effect on degradation.

Bumpus et al. (1985) tested the white rot fungus, Phanerochaete chrysosporium, which secretes a unique H202-dependent extracellular lignin-degrading enzyme system capable of generating carbon-centered free radicals. Lignin is resistant to attack by all microorganisms except some species of fungi and a relatively small number of species of bacteria. Radiolabeled 2,3,7,8-TCDD was oxidized to labeled C02 by nitrogen-deficient, ligninolytic cultures of P. chrysosporium, and since the label was restricted to the ring, it was concluded that the strain was able to degrade halogenated aromatic rings. In 10 ml cultures containing 1,250 pmol of substrate, 27.9 pmol of 2,3,7,8-TCDD were converted to labeled-CO2 during the 30-day incubation period, thus only about 2 percent of the starting material was converted.

2.5.2. Polychlorinated Dibenzofurans (PCDFs)

There is little information on the environmental transport and fate of PCDFs. However, the available information on the physical/chemical properties of PCDFs indicate that PCDFs should resemble PCDDs in environmental behavior. In general, PCDFs are expected to be extremely stable compounds under normal environmental conditions and to be strongly sorbed to soils, sediments and particulate matter. Erosion and aquatic transport of sediment is expected to be the main transport mechanism; the potential for significant leaching and volatilization are minimal. The only environmentally significant path for destruction of PCDFs is apparently photodegradation.

- **Photodegradation**

PCDFs absorb electromagnetic radiation at wavelengths above 290 nm and should be capable, therefore, of undergoing photolysis when subjected to sunlight (EPRI 1983). The potential for photodegradation of PCDFs in the environment appears to be similar to the photodegradation potential of PCDDs; in the presence of a hydrogen donor and sunlight, PCDFs will dechlorinate. The major photoproducts are lower chlorinated PCDFs.

Crosby et al. (1973) report that polychlorinated dibenzofurans undergo photolytic dechlorination in the presence of a hydrogen donor, with more highly chlorinated congeners being more stable. In contrast, Hutzinger (1973) and Buser (1976) report that the more highly chlorinated congeners undergo photodegradation at a rate similar to that of lower chlorinated PCDFs. Hutzinger (1973) found that both 2,8-DCDF and OCDF photolyze rapidly in methanol and hexane.

Buser (1988) studied the photolytic decomposition rates of 2,3,7,8-TCDF, 1,2,3,4-TCDF, and 1,2,7,8-TCDF. Studies were performed in dilute isooctane solutions and as solid phases on quartz surfaces under sunlight and artificial laboratory illumination (fluorescent lights). When the solutions were illuminated with sunlight, the estimated half-lives were 220 minutes for a solution containing 3 ng/ul of 2,3,7,8-TCDF, 180 minutes for a solution containing 2 ng/ul of 1,2,3,4-TCDF, and 600 minutes for a solution containing 0.3 ng/ul of 1,2,7,8-TCDF. For the same solutions illuminated with artificial light, the half lives were greater than 28 days. When the same solutions dispersed and dried as thin films in quartz vials and exposed to sunlight, the estimated half-lives were reported to be 120 hours, 95 hours, and 35 hours, respectively.

- **Oxidation**

Although there is no information available on the oxidation of PCDFs under environmentally relevant conditions, PCDFs are expected to be stable to oxidation (EPRI 1983).

- **Hydrolysis**

No information is available indicating that hydrolysis would be an operative environmental process for degradation of PCDFs (EPRI 1983).

- **Volatilization and Sorption**

Based on their very low vapor pressures and high K_{oc} values, PCDDs would be expected to sorb to rather than volatilize from soils, sediments, and other solids; however, due to the stability and persistence of PCDDs via other transformation and transport pathways, volatilization should not be ignored as a transport mechanism.

- **Biotransformation and Biodegradation**

No information was found relating specifically to the biodegradability of PCDFs. However, structurally similar PCDDs are considered to be essentially nonbiodegradable in the environment (EPRI 1983) and, like other persistent compounds, PCDFs are found in soils, sediments, and biota. Therefore, PCDFs are expected to behave in a like manner and remain persistent to attack by microorganisms and other types of biotransformation processes.

2.5.3 Coplanar PCBs

Little information exists on the environmental transport and fate of the specific coplanar PCBs. However, the available information on the physical/chemical properties of coplanar PCBs coupled with the body of information available on the widespread occurrence and persistence of PCBs in the environment indicates that these coplanar PCBs are likely to be strongly sorbed by soils and sediments, and to be thermally and chemically stable. Photodegradation of the more highly chlorinated congeners followed by slow aerobic biodegradation is believed to be the principal path for destruction of PCBs.

The following sections present brief summaries of readily available information on the fate of the 11 coplanar PCBs. Because few studies have been performed, much of the available information was based on estimation techniques.

- **Photodegradation**

Leifer et al. (1983) summarized available data on photolysis of PCB isomers. Based on the available data, Leifer et al. (1983) concluded that all PCBs, especially the more highly chlorinated congeners and those that contain two or more chlorines in the ortho position, photodechlorinate. In general, as the chlorine content increases, the photolysis rate increases and the half-life decreases. The products of photolysis are predominantly

lower chlorinated PCBs. In addition, Choudry and Webster (1987) report that the photolysis of PCBs in certain solvent systems yields PCDFs.

- **Oxidation**

PCBs are extremely resistant to oxidation as evidenced by the high temperatures required to achieve thermal destruction. However, reaction in the atmosphere with OH radicals may be a significant degradation mechanism. Leifer et al. (1983) estimated that atmospheric transformation occurs reasonably fast for those PCB congeners containing either a small number of chlorines or which have all or most of the chlorines on one ring. For the 11 coplanar PCBs, the predicted half-lives are: for the TeCBs, 11 to 20 days; for the PeCBs, 12 to 31 days; for the HxCBs, 32 to 62 days; and for the HpCBs, 94 days.

- **Hydrolysis**

There are no experimental data published on the hydrolysis of PCBs under environmental conditions (Leifer et al. 1983). However, it is expected that the attachment of chlorines directly to the aromatic ring in PCBs confers hydrolytic stability (Leifer et al. 1983).

- **Volatilization and Sorption**

Based on their very low vapor pressures and high K_{oc} values, PCDDs would be expected to sorb to rather than volatilize from soils, sediments, and other solids; however, due to the stability and persistence of PCDDs via other transformation and transport pathways, volatilization should not be ignored as a transport mechanism. Freeman and Schroy (1985) point out that low volatility chemicals may bind strongly with dry soil but, once a molecular microlayer of water covers the soil particles, the chemical should become more volatile; in addition, water vaporization may enhance the rate of chemical vaporization from a soil column.

- **Biotransformation and Biodegradation**

Leifer et al. (1983) summarized the available information on the degradation of PCBs by microorganisms. The following general findings were made. There are numerous aerobic microorganisms in the environment that are capable of degrading most PCBs and

that such organisms are widely distributed in the environment. In general, the rate of aerobic biodegradation decreases with increasing chlorination. For example, the half-lives resulting from biodegradation for TeCBs in fresh surface water and soil are 7 to 60+ days and 12 to 30 days, respectively. For PeCBs and higher chlorinated PCBs, the half-lives in fresh surface water and soil are likely to exceed one year. Also, PCBs with all or most chlorines on one ring and PCBs with fewer than two chlorines in the ortho position tend to degrade more rapidly. Finally, there is no evidence for biodegradation of PCBs under anaerobic conditions.

2.6 FATE - BROMINATED COMPOUNDS

Although there are no available published studies documenting measured fate rate constants, relatively few studies with measured physical/chemical property data, and few relevant environmental monitoring studies, it is possible to estimate the environmental transport and transformation processes for major PBDDs, PBDFs, and PBBs using structure activity and property estimation methods. Mill (1989) performed such an assessment and much of what is reported in this section is a summary of that review paper.

Mill (1989) concluded that the estimated physical/chemical properties of these compounds indicate that will behave in a similar fashion to their chlorinated analogs. In general, these chemicals are expected to be stable under normal environmental conditions, relatively immobile in the environment, and primarily associated with particulate and organic materials. The only environmentally significant path for destruction is photodegradation. If discharged to the atmosphere, any vapor phase compounds will probably be rapidly photolyzed.

Upon deposition onto surfaces, there can be an initial loss due to photodegradation and/or volatilization. Once sorbed onto soils or sediments, however, they are expected to be strongly sorbed with erosion and aquatic transport of sediment the dominant physical transport mechanism. If discharged to water, they are expected to preferentially sorb to solids. Volatilization may also be a significant transport mechanism for non-sorbed chemicals even though they have negligible estimate vapor pressures.

- **Photodegradation**

Photolysis in the atmosphere appears to be a major pathway for loss of PBDDs and PBDFs based on recent studies by Buser. Buser (1988) studied the photolytic decomposition rates of the following compounds in dilute isooctane solutions and as solid phases on quartz surfaces under sunlight and artificial laboratory illumination: 1,2,3,4-TBDD; 2,3,7,8-TBDD; 2,3,7,8-TBDF; and mono- and dibrominated 2,3,7,8-TCDD and 2,3,7,8-TCDF. Under natural sunlight, estimated half-lives were very short, on the order of minutes. Solid phase photolysis was significantly slower, in the range of 7 to 35 hours. The major photolytic pathway was reductive dehalogenation with the formation of lower halogenated or unsubstituted dibenzo-p-dioxins and dibenzofurans.

Mill (1989) used the results obtained by Buser (1988) together with assumptions to overcome the lack of quantum yield data from Buser (1988) to estimate the photolysis half-lives of the three brominated-only compounds tested by Buser (1988). Mill (1989) estimated the following half-lives in water (top one meter) and for vapor in air (first kilometer above surface) for clear-sky conditions in mid-summer at 40 degrees north latitude:

Compound	Half-life in water (hrs)	Half-life in air (min)
1,2,3,4-TBDD	7	<1
2,3,7,8-TBDD	2	0.3
2,3,7,8-TBDF	1.7	0.2

- **Oxidation**

The reaction rates of OH radicals with PBDDs, PBDFs, and PBBs have not been measured. The low vapor pressures of these compounds makes direct measurements very difficult with the current techniques. However, Mill (1989), using a structure activity relationship developed by Atkinson (1987), has estimated the half-lives of OH oxidation for the tetra- through octa- PBDDs and PBDFs. The estimated half-lives listed below indicate that OH oxidation is probably too slow to compete with photolysis.

No. of Br	PBDD Half-life (hrs)	PBDF Half-life (hrs)
4	50	420
5	50	430
6	100	960
7	200	1900
8	770	3800

- **Hydrolysis**

There is no available evidence indicating that hydrolysis would be a significant degradation process for these compounds.

- **Volatilization and Sorption**

Little information exists on the environmental transport of PBDDs, PBDFs, and PBBs. However, the available information on the physical/chemical properties of these compounds and their chlorinated analogs coupled with the body of information available on the widespread occurrence and persistence of the chlorinated analogs in the environment indicates that these compounds are likely to be strongly sorbed by soils and sediments and to be resistant to leaching and volatilization.

- **Biotransformation and Biodegradation**

Although there are no data available concerning the biodegradability of the brominated analogs of PCDDs, PCDFs, and PCBs, it is expected that these brominated analogs, especially the more halogenated congeners, will be recalcitrant to biodegradation.

REFERENCES FOR CHAPTER 2

Abramowitz, R.; Yalkowsky, S.H. (1990) Estimation of aqueous solubility and melting point for PCB congeners. Chemosphere 21(10/11): 1221-1229.

Arthur, M.F., Frea, J.I. (1989) 2,3,7,8-Tetrachlorodibenzo-p-dioxin: aspects of its important properties and its potential biodegradation. J. Environ. Qual. 18:1-11.

Atkinson, R. (1987) Estimation of OH radical reaction rate constants and atmospheric lifetimes for polychlorobiphenyls, dibenzo-p-dioxins, and dibenzofurans. Environ. Sci. Technol. 21:305-307.

Botre, C., Memoli, A., Al-Haique, F. (1978) TCDD solubilization and photodecomposition in aqueous solutions. Environ. Sci. Technol. 12:335-336.

Bumpus, J.A., Tien, M., Wright, D., Aust, S.D. (1985) Oxidation of persistent environmental pollutants by a white rot fungus. Science 228:1434-1436.

Burkhard, L.P.; Kuehl, D.W. (1986) N-octanol/water partition coefficients by reverse phase liquid chromatography/mass spectrometry for eight tetrachlorinated planar molecules. Chemosphere 15(2): 163-167.

Buser, H.R. (1976) Preparation of qualitative standard mixtures of polychlorinated dibenzo-p-dioxins and dibenzofurans by ultraviolet and gamma irradiation of the octachlorocompounds. J. Chromatogr. 192:303-307.

Buser, H.R. (1988) Rapid photolytic decomposition of brominated and brominated/chlorinated dibenzodioxins and dibenzofurans. Chemosphere 17:889-903.

Camoni, I., Dimuccio, A., Pontecorvo, D., Taggi, F., Vergori, I. (1982) Laboratory investigation for the microbial degradation of 2,3,7,8-tetrachlorodibenzo-p-dioxin in soil by addition of organic compost. Pergammon Ser. Environ. Sci. 5:95-103.

Choudhry, G.G., Webster, G.R.B. (1987) Environmental photochemistry of polychlorinated dibenzofurans (PCDFs) and dibenzo-p-dioxins (PCDDs): A review. Toxicol. Environ. Chem. 14:43-61.

Choudhry, G.G., Webster, G.R.B. (1989) Environmental photochemistry of PCDDs. 2. Quantum yields of direct phototransformation of 1,2,3,7-tetra-, 1,3,6,8-tetra-, 1,2,3,4,6,7,8-hepta-, and 1,2,3,4,6,7,8,9-octachlorodibenzo-p-dioxin in aqueous acetonitrile and their sunlight half-lives. J. Agric. Food Chem. 37:254-261.

Choudhry, G.G.; Foga, M.; Webster, G.R.B.; Muir, D.C.G.; Friesen, K. (1990) Quantum yields of the direct phototransformation of 1,2,4,7,8-penta- and 1,2,3,4,7,8-hexa chlorodibenzofuran in aqueous acenitrile and their sunlight half-lives. Toxicology and Environmental Chemistry 26:181-195.

Crosby, D.G., Moilanen, K.W., Wong, A.S. (1973) Environmental generation and degradation of dibenzodioxins and dibenzofurans. Environ. Health Perspect., Exp. Issue. 5:259-266.

Crosby, D.G. (1978): Conquering the monster - the photochemical destruction of chlorodioxins. In: Disposal and decontamination of pesticides. M.V.Kennedy, Ed. ACS Symposium Series 73:1-12.

Crosby, D.G. (1981) Methods of photochemical degradation of halogenated dioxins in view of environmental reclamation. Paper presented on "Human health aspects of accidental exposure to dioxins. Strategy for environmental reclamation and community protection," Bethesda, MD, October 5-7, 1982.

Davis, C.A.: University of California, Department of Environmental Toxicology.

Dickhut, R.M.; Andren, A.W.; Armstrong, D.E. (1986) Aqueous solubilities of six polychlorinated biphenyl congeners at four temperatures. Environ. Sci. Technol. 20(8): 807-810.

DiDomenico, A., Viviano, G., Zapponi, G. (1982) Environmental persistence of 2,3,7,8-TCDD at Seveso. In: Chlorinated dioxins and related compounds, impact on the environment. O. Hutzinger et al., Eds. Elmsford, NY: Pergamon Press, pp. 105-113. (1983).

Dulin, D., Drossman, H., Mill, T. (1986) Products and quantum yields for photolysis of chloroaromatics in water. Environ. Sci. Technol. 20:72-77.

Dunnivant, F.M.; Elzerman, A.W. (1988) Aqueous solubility and henry's law constant data for PCB congeners for evaluation of quantitative structure-property relationships (QSPRs). Chemosphere 17(3): 525-541.

EPRI (1983) Electric Power Research Institute. State-of-the-art review; PCDDs and PCDFs in utility fluid. Palo Alto, CA: EPRI-CS-3308.

Esposito, M.P., Teirnan, T.D., Dryden, F.E. (1980) Dioxins. U.S. EPA, IERL, Office of Research and Development, Cincinnati, OH. EPA-600/2-80-197.

Foreman, W.T.; Bidleman, T.F. (1985) Vapor pressure estimates of individual polychlorinated biphenyls and commercial fluids using gas chromatographic retention data. J. Chromatog. 330: 203-216.

Freeman, R.A., Schroy, J.M. (1985) Environmental mobility of TCDD. Chemosphere 14:873-876.

Friesen, K.J.; Vilk, J.; Muir, D.C.G. (1990) Aqueous solubilities of selected 2,3,7,8-substituted polychlorinated dibenzofurans (PCDFs). Chemosphere 20(1-2): 27-32.

Friesen, K.J.; Sarna, L.P.; Webster, G.R.B. (1985) Aqueous solubility of polychlorinated dibenzo-p-dioxins determined by high pressure liquid chromatography chemosphere. 14(9): 1267-1274.

Friesen, J.K., Muir, D.C.G., Webster, G.R.B. (1990) Evidence of sensitized photolysis of polychlorinated dibenzo-p-dioxins in natural waters under sunlight conditions. Environ. Sci. Technol. 24(11):1739-1744.

Hawker, D.W.; Connell, D.W. (1988) Octanol-water partition coefficients of polychlorinated biphenyl congeners. Environ. Sci. Technol. 22(4): 382-387.

Hutter, R., Philippi, M. (1982) Studies in microbial metabolism of TCDD under laboratory conditions. in: Chlorinated dioxins and related compounds, impact on the environment. O. Hutzinger et al. eds. Elmsford, NY: Pergammon Press. pp. 87-93 as cited in Vuceta et al. (1983).

Hutzinger, O. (1973) Photochemical degradation of di and octachlorodibenzofuran. Environ. Health. Perspect. Exp. Issue. 5:253-256.

Kapila, S., Yanders, A.F., Orazio, C.E., Meadows, J.E., Cerlesi, S., Clevenger, T.E. (1989) Field and laboratory studies on the movement and fate of tetrachlorodibenzo-p-dioxin in soil. Chemosphere 18:1297-1304.

Leifer, A., Brink, R.H., Thom, G.C., Partymiller, K.G. (1983) Environmental transport and transformation of polychlorinated biphenyls. Washington, D.C.: U.S. Environmental Protection Agency, Office of Toxic Substances. EPA-560/5-83-025.

Lyman, W.J.; Reehl, W.F.; Rosenblatt, D.H. (1982) Handbook of chemical property estimation methods. New York, NY: McGraw-Hill.

Mackay, D.; Bobra, A.; Chan, D.W.; Shiu, W.Y. (1982) Vapor pressure correlations for low-volatility environmental chemicals. Environ. Sci. Technol. 16(10): 645-649.

Marple, L.; Brunck, R.; Throop, L. (1986a) Water solubility of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environ. Sci. Technol. 20(2): 180-182.

Marple, L.; Berridge, B.; Throop, L. (1986b) Measurement of the water-octanol partition coefficient of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environ. Sci. Technol. 20(4): 397-399.

Matsumura, F., Benezet, J.H. (1973) Studies on the bioaccumulation and microbial degradation of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environ. Health Perspect. Sept. 253-258.

Mill, T., Rossi, M., McMillen, D., Coville, M., Leung, D., Spang, J. (1987) Photolysis of tetrachlorodioxin and PCBs under atmospheric conditions. Internal report prepared by SRI International for USEPA, Office of Health and Environmental Assessment, Washington, D.C.

Mill, T. (1989) Environmental fate of polybrominated dibenzodioxins and dibenzofurans. Office of Toxic Substances, Washington, D.C.

Miller, G.C., Zepp, R.G. (1987). 2,3,7,8-Tetrchlorodibenzo-p-dioxin: environmental chemistry. In: Exner, J.H. ed. Solving hazardous waste problems -- learning from dioxins. Washington, D.C.: American Chemical Society.

Muir, C.G., Yarechewski, A.L., Corbet, R.L., Webster, G.R.B., Smith, A.E. (1985) Laboratory and field studies on the fate of 1,3,6,8 - tetrachlorodibenzo-p-dioxin in soil and sediments. J. Agric. Food Chem. 33:518-523.

Orth, R.G., Ritchie, C., Hileman, F. (1989) Measurement of the photoinduced loss of vapor phase TCDD. Chemosphere 18:1275-1282.

Palausky, J., Kapila, S., Manahan, S.E., Yanders, A.F., Malhotra, R.K., Clevenger, T.E. (1986) Studies on vapor phase transport and role of dispersing medium on mobility of 2,3,7,8-TCDD in soil. Chemosphere 15:1387-1396.

Plimmer, J.R. (1978) Photolysis of TCDD and trifluralin on silica and soil. Bull. Environ. Contam.Toxicol. 20:87-92.

Podoll, R.T., Jaber, H.M., Mill, T. (1986) Tetrachlorodibenzodioxin: Rates of volatilization and photolysis in the environment. Environ. Sci. Technol. 20:490-492.

Podoll, R.T.; Jaber, H.M.; Mill, T. (1986) Tetrachlorodibenzodioxin rates of volatilization and photolysis in the environment. Envrion. Sci. Technol. 20(5): 490-492.

Puri, R.K., Clevenger, T.E., Kapila, S., Yanders, A.F., Malhotra, R.K. (1989) Studies of parameters affecting translocation of tetrachlorodibenzo-p-dioxin in soil. Chemosphere 18:1291-1296.

Risby, T.H.; Hsu, T.B.; Sehnert, S.S.; Bhan, P. (1990) Physiochemical parameters of individual hexachlorobiphenyl congeners. Environ. Sci. Technol. 24(11): 1680-1687.

Rordorf, B.F. (1989) Prediction of vapor pressures, boiling points and enthalpies of fusion for twenty-nine halogenated dibenzo-p-dioxins and fifty-five dibenzofurans by a vapor pressure correlation method. Chemosphere 18(1-6): 783-788.

Rordorf, B.F. (1987) Prediction of vapor pressures, boiling points, and enthalpies of fusion for twenty-nine halogenated dibenzo-p-dioxins. Thermochimica Acta 112: 117-122.

Sabljić, A.; Güsten, H. (1989) Predicting Henry's Law constants for polychlorinated biphenyls. Chemosphere 19(10/11): 1503-1511.

Sarna, L.P.; Hodge, P.E.; Webster, G.R.B. (1984) Octanol-water partition coefficients of chlorinated dioxins and dibenzofurans by reversed-phase HPLC using several C_{18} columns. Chemosphere 13(9): 975-983.

Shiu, W.Y.; Doucette, W.; Gobas, F.A.P.C.; Andren, A.; Mackay, D. (1988) Physical-chemical properties of chlorinated dibenzo-p-dioxins. Environ. Sci. Technol. 22(6): 651-658.

Sijm, D.T.H.M.; Wever, H.; de Vries, P.J.; Opperhuizen, A. (1989) Octan-1-ol/water partition coefficients of polychlorinated dibenzo-p-dioxins and dibenzofurans: experimental values determined with a stirring method. Chemosphere 19(1-6): 263-266.

SRC. (1991) Syracuse Research Corporation calculated values from chemfate database literature search.

Stehl, R.H. (1973) The stability of pentachlorophenol and chlorinated dioxins to sunlight, heat, and combustion. In: Chlorodioxins: origin and fate. Blair, E.H., ed. Adv. Chem. Ser. 120:119-125.

U.S. Environmental Protection Agency. (1990) Background document to the integrated risk assessment for dioxins and furans from chlorine bleaching in pulp and paper mills. Office of Toxic Substances, Washington, DC; EPA 560/5-90-014.

Webster, G.R.B.; Friesen, K.J.; Sarna, L.P.; Muir, D.C.G. (1985) Environmental fate modelling of chlorodioxins: determination of physical constants. Chemosphere 14(6/7):

3. ENVIRONMENTAL LEVELS OF PCDD, PCDF, AND PCB CONGENERS

3.1. INTRODUCTION

Polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and polychlorinated biphenyls (PCBs) have been found throughout the world in practically all media including air, soil, water, sediment, fish and shellfish, such other food products as meat, milk, and vegetation. PCDDs and PCDFs are contaminants that can be released to the environment as by-products resulting from the manufacture of such chlorinated compounds as polychlorinated phenols, PCBs, phenoxy herbicides, hexachlorobenzene, and chlorodiphenyl ethers. Although the manufacture of most chlorinated phenolic intermediates and products, including PCBs, was terminated in the late 1970s, continued use and disposal of those compounds can result in releases of PCDDs, PCDFs, and PCBs to the environment. Significant releases may also result from the combustion of municipal and chemical wastes, and burning such non-chlorinated organic substances as polystyrene, cellulose, lignin, and coal in the presence of a chlorine donor. Dioxins can also be released through the condensation of naturally occurring phenolic compounds such as that resulting from chlorination of municipal water or the contribution of household bleaches flushed into sewer systems. In addition, paper mills that use chlorine bleaching processes have been identified recently as a source of both PCDDs and PCDFs. Consequently, PCDDs, PCDFs, and PCBs become available for human exposure via the pathways mentioned previously.

There are 75 PCDD congeners and 135 PCDF congeners (based on the positioning of between 1 and 8 chlorine atoms in these compounds). Of the 210 PCDD/PCDF congeners, those that pose the greatest human health risk are the 2,3,7,8-substituted compounds, and the congeners of greatest concern are 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) and 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8-TCDF). With regard to PCBs, some congeners exhibit dioxin-like attributes in their biologic processes and toxic effects. For the purposes of this report, particular attention will be paid to the distribution of 2,3,7,8-TCDD and 2,3,7,8-TCDF congeners, and total Cl_4 to Cl_8 homologues in the environment. In addition, distribution of the following PCB congeners in the various media will be discussed:

IUPAC No.	Congener
77	3,3',4,4'-tetra PCB
81	3,4,4',5-tetra PCB
105	2,3,3',4,4'-penta PCB
114	2,3,4,4',5-penta PCB
118	2,3',4,4',5-penta PCB
126	3,3',4,4',5-penta PCB
156	2,3,3',4,4',5-hexa PCB
157	2,3,3',4,4',5'-hexa PCB
167	2,3',4,4',5,5'-hexa PCB
169	3,3',4,4',5,5'-hexa PCB
189	2,3,3',4,4',5,5'-hepta PCB

The sections that follow present ranges of the aforementioned compounds in air, soil, water, sediment, fish and shellfish, and food throughout the world. This literature summary is not all inclusive, but is meant to present the reader with a general idea of worldwide values reported in the literature.

3.2. CONCENTRATIONS IN SOIL

Tables B-1 and B-2 (Appendix B) contain summaries of data from the published literature regarding concentrations of PCDDs and PCDFs in soil. Data on coplanar PCB congener soil concentrations were not found in the literature; the PCB soil concentration data that were found in the literature were reported as either total PCB concentrations or concentrations of Aroclor PCB mixtures.

Soil samples from rural and semi-urban sites in England, Wales, and lowland Scotland showed a general increase in concentration from the tetra (Cl_4) to the octa (Cl_8) homologues of PCDD, whereas the PCDF levels showed very little variation between the homologue groups (Creaser et al., 1989). Concentrations of 2,3,7,8-TCDD at those sites ranged from <0.5 to 2.1 parts per trillion (ppt). The median values for the Cl_4 to Cl_8 CDD homologues were 6.0, 4.6, 31, 55, and 143 ppt, respectively. The median values for the Cl_4 to Cl_8 CDF homologues, on the other hand, were 16, 17, 32, 15, and 15 ppt. Evaluation of soil data from urban sites in the same geographical area showed that the mean levels for the PCDD and PCDF homologues were significantly greater ($p<0.01$) than those for rural and semi-urban background soils (Creaser et al., 1990). Concentrations of 2,3,7,8-TCDD at those sites ranged from <0.5 to 4.2 ppt. The median values for the Cl_4

to Cl_8 CDD homologues were 40, 63, 141, 256, and 469 ppt, respectively. The median values for the Cl_4 to Cl_8 CDF homologues were 140, 103, 103, 81, and 40 ppt. Although the same argument was held as for the previous study that the median values for the data set probably better reflect the typical background concentrations, the elevated levels of the lower congeners, together with higher overall concentrations, are indicative that local sources and short range transport mechanisms are major contributors of PCDDs and PCDFs to urban soils.

Analysis of four sites in Hamburg, Germany, contaminated by an organochlorine pesticide manufacturing company showed patterns of PCDD and PCDF homologue distribution that are similar to the urban and industrial sites examined in England, Wales, and Scotland (Sievers and Friesel, 1989). The study indicated that PCDDs and PCDFs showed a regular increase from the Cl_4 to Cl_8 homologues (although individual data points were not presented). The maximum concentrations of 2,3,7,8-TCDD ranged from 900 ppt to 874,000 ppt. The very high concentrations of 2,3,7,8-TCDD at the sites were attributed to an admixture of wastes from 2,4,5-T production.

Soil sampled in 1987 from the vicinity of a sewage sludge incinerator was compared with soil from rural and urban sites in Ontario, Canada (Pearson et al., 1990). Soil in the vicinity of the incinerator showed a general increase in concentration from the Cl_4 to Cl_8 CDD homologues, whereas only the Cl_8 CDF homologue was detected (mean concentration 43 ppt). Rural woodlot soil samples only contained the Cl_8 CDD homologue (mean concentration of 30 ppt). Soil samples from undisturbed urban parkland settings revealed only Cl_7 and Cl_8 CDD homologues, but all CDF homologues (Cl_4 to Cl_8) were present. Those samples showed an increase in concentration from the Cl_7 to Cl_8 CDD homologues and Cl_5 to Cl_8 CDF homologues. The Cl_4 homologue had the highest mean value (29 ppt) of all the CDF homologues. Resampling of one of the urban sites in 1988, however, showed high variability in the concentrations of PCDD and PCDF.

Data were collected on PCDD and PCDF homologues in soil samples from industrial, urban, and rural sites in Ontario and some U.S. Midwestern states (Birmingham, 1990). The levels of PCDD/PCDF in rural soils were primarily nondetected (ND), although the Cl_7 and Cl_8 CDD homologues were found in a few samples. In urban soils, the Cl_4 to Cl_8 homologues were measured for both CDDs and CDFs. The Cl_7 and Cl_8 CDD homologues dominated the homologue profile, and were two orders of magnitude greater than in the

rural soils. These soils also contained measurable quantities of the Cl_4 and Cl_5 CDD homologues. Industrial soils did not contain any Cl_4 or Cl_5 CDD homologues, but they contained the highest levels of the Cl_4, Cl_7, and Cl_8 CDF homologues. In another study, soils from industrialized areas of a group of cities from Midwestern and Mid-Atlantic states (MI, IL, OH, TN, PA, NY, WV, VA) were analyzed for levels of 2,3,7,8-TCDD (Nestrick et al., 1986). Many of the samples were taken within one mile of major steel, automotive or chemical manufacturing facilities, or municipal solid waste incinerators. Concentrations of 2,3,7,8-TCDD measured in this study ranged from ND to 9.4 ppt.

EPA conducted a 2-year nationwide study to investigate the national extent of 2,3,7,8-TCDD contamination (USEPA, 1987). The results of this large study were summarized broadly in the primary reference (i.e., the number and types of samples per site, and range of detection). Moreover, sampling for five of the seven "tiers" of the study had detection limits in soil, sediment, and water of 1 part per billion (ppb). Only Tier 5 (sites where pesticides derived from 2,4,5-TCP have been or are being used for commercial purposes) and Tier 7 (ambient sampling for fish and soil) had detection limits of 1 ppt. Subsequently, the data from this study are not included in the tables, but some observations from this study with regard to soil contamination are discussed below.

The 100 Tier 1 and 2 sites were investigated by the Superfund program as sites already on or expected to be on the NPL list. Soil concentrations found in most of the sites were in the ppb range, although in a few sites where concentrated 2,4,5-TCP production wastes were stored or disposed of, concentrations were as high as 2,000 parts per million (ppm). Off-site soil contamination of concern was confirmed in 7 of the 100 Tier 1 and 2 sites, with soil concentrations in the ppb range. Eleven of 64 Tier 3 sites (facilities and associated disposal sites where 2,4,5-TCP and its derivatives were formulated into pesticide products) were found to have soil concentrations exceeding 1 ppb, and in 7 of 11 sites where contamination was found, only one or two soil samples were above 1 ppb. Fifteen of 26 Tier 5 sites (areas where 2,4,5-TCP and pesticide derivatives were or are being used) had concentrations above 1ppt, and one of those had a single detection of 6 ppb. Two-thirds of all detections at the Tier 5 sites were below 5 ppt. Three of 18 Tier 6 sites (organic chemical and pesticide manufacturing facilities where improper quality control on production processes could have resulted in 2,3,7,8-TCDD being introduced into the waste streams) had soil concentrations that exceeded the

detection limit of 1 ppb, although these levels were limited to one or two samples per site. Seventeen of the 221 urban soil sites and 1 of the 138 rural sites from Tier 7 (background sites not expected to have contamination) had soil concentrations exceeding 1 ppt. The highest concentration detected (11.2 ppt) was found in an urban sample. The results from Tier 7 are consistent with the other studies discussed above regarding soil concentrations of 2,3,7,8-TCDD in non-industrial settings.

Some general observations for PCDD and PCDF levels in soils are possible from the data presented in the various soil studies discussed above:

- Generalizations about the prevalence of specific congeners within a homologue group are not possible.
- As the degree of chlorination increases, the concentrations increase. Concentrations of the Cl_7 and Cl_8 CDD and CDF homologues generally are higher than the Cl_4, Cl_5, and Cl_6 homologues.
- Concentrations associated with industrial sites clearly are the highest, with concentrations in the hundreds to thousands of parts per trillion. Concentrations in settings identified as urban are higher than those in areas identified as rural.

3.3. CONCENTRATIONS IN WATER

Tables B-3 and B-4 (Appendix B) contain summaries of data from the published literature regarding concentrations of PCDDs and PCDFs in water. Data on coplanar PCB congener water concentrations were not found in the literature.

PCDDs in surface water samples collected from the Eman River in southern Sweden generally increased in concentration from the Cl_4 to the Cl_8 homologues, whereas the PCDF levels showed very little variation between the homologue groups (Rappe et al., 1989b). In general, however, the levels of PCDFs were higher than the level of PCDDs. Concentrations of 2,3,7,8-TCDF were 0.022 parts per quadrillion (ppq) in Jarnsjon and 0.026 ppq in Fliseryd. The filtered water, before chlorination and distribution as drinking water had no detectable Cl_4, Cl_5, or Cl_6 homologues for CDDs or CDFs, but the Cl_7 and Cl_8 homologues for CDDs were detected at 120 and 170 ppq, respectively. However, the concentration of the Cl_8 CDD homologue was close to the level found in the blank.

A survey of 49 drinking water supplies in Ontario, including supplies in the vicinity of chemical industries and pulp and paper mills was initiated in 1983 (Jobb et al., 1990). As of February 1989, 4,347 results were received for 399 raw and treated water samples. The Cl_8 CDD homologue was detected in 36 of 37 positive results, and ranged from 9 to 175 ppq in raw samples and 19 to 46 ppq in treated samples. These low concentrations primarily were found in raw water located downstream of industrialized areas in the St. Clair/Detroit River system. The 2,3,7,8-TCDD congener was not detected in any sample. Since PCDDs and PCDFs are hydrophobic compounds and consequently have a tendency to sorb onto particulate matter in water, conventional water treatment processes can be effective in removing the contaminants along with the particulates. This is substantiated by the fact that 33 of the 37 positive results were raw water samples. Because of the relatively low levels of PCDDs detected in the samples, however, it is difficult to ascertain whether the PCDDs were particulate-associated or dissolved.

A survey of 20 community water systems throughout New York State was conducted in 1986 (Meyer et al., 1989). The sampling sites were representative of the major surface source waters in New York and included sources receiving industrial discharges or known to contain dioxin-contaminated fish as well as waters in more remote areas. The Cl_4 CDF homologue was detected in the finished water at the Lockport (duplicate samples had concentrations of 2.1 and 2.6 ppq). Except for a trace of the Cl_8 CDF homologue detected at one location, no other PCDDs/PCDFs were detected in finished water at any of the other 19 community water systems surveyed. Raw water sampled at the Lockport facility contained concentrations of the Cl_4 CDD homologue (1.7 ppq) as well as Cl_4 to Cl_8 CDF homologues (18, 27, 85, 210, and 230 ppq, respectively). As can be seen from the data, the PCDF homologue group concentrations increased with increasing chlorine number.

Some general observations for PCDD and PCDF levels are possible from the data presented in the various water studies above:

- Raw water samples generally have higher concentrations of PCDDs/PCDFs than finished water samples.
- The concentration of PCDDs and PCDFs in surface water generally increase from the Cl_4 to the Cl_8 homologues.

3.4. CONCENTRATIONS IN SEDIMENT

Tables B-5 through B-7 (Appendix B) contain summaries of data from the published literature regarding concentrations of PCDDs, PCDFs, and coplanar PCB congeners in sediment.

Sediment samples from the vicinity of a magnesium production plant were analyzed for PCDDs and PCDFs (Oehme et al., 1989). The concentration distribution of PCDD and PCDF congeners was rather homogeneous except for a slight decrease at a sampling station further downstream of the plant. However, the deeper sediments (4-6 and 11-13 cm depth) at that site had somewhat higher levels. Another sampling station even further downstream had concentrations that were a factor of 4 to 10 lower, thereby indicating substantial transport of PCDDs and PCDFs. The Cl_4 CDF congener profiles were the same as those for magnesium production. In addition, the Cl_5 CDF congener profiles were very similar to those found in the waste water. Trapped sediments from the archipelago of Stockholm, Sweden displayed PCDD and PCDF homologue distribution patterns that were very similar to those exhibited in total air and air particulates (Rappe and Kjeller, 1987). The Cl_7 and Cl_8 CDDs and Cl_7 CDF were the dominant homologues in the sediment. Bottom surface sediment samples collected from the Baltic Sea showed interesting PCDD and PCDF distribution patterns (Rappe et al.,1989a). The background samples, one between the Swedish and Soviet coasts and the other between the Swedish and Finnish coasts, contained similar levels and distribution profiles. The study indicated that the pattern of the Cl_4 CDF congeners at these sites were typical of the "incineration pattern" (i.e., patterns resulting from MSW incineration, car exhausts, steel mills, etc.) which also had been found in samples of air and air particulates. However, sediment samples collected at a distance of 4 to 30 km from a pulp mill revealed a congener distribution pattern typical of bleaching mills. The Cl_4 CDF homologue in the sediment 4 km from the pulp mill contained only two major congeners, and even the sediment collected 30 km from the mill displayed the same pattern. Surface sediments collected from 18 lake areas in central Finland were analyzed for PCDDs, PCDFs, and PCBs. Although 2,3,7,8-TCDD was not detected in any of the samples, two other Cl_4 CDD congeners that are common and abundant in pulp mill effluents were detected. In addition, Cl_6 to Cl_8 CDD congeners as well as Cl_7 and Cl_8 CDF congeners that are not linked to pulp mills also were detected. The study suggested that these may have resulted from combustion operations in the

more densely populated and industrialized areas in south Finland. Coplanar PCBs were detected at low background levels. The most toxic PCB congener (IUPAC No. 126) was found only in one area that is located nearest to a PCB leakage point. The concentration of PCB 126 at that site was 110 ppt. Evaluation of sediments in Hamburg Harbor in Germany revealed high concentrations of Cl_4 to Cl_8 CDD homologues (mean concentrations of 564, 1112, 2744, 4040, and 7560 ppt, respectively) and Cl_4 to Cl_8 CDF homologues (mean concentrations of 526, 2980, 4106, 2358, and 2712 ppt) (Gotz et al., 1990). The average concentration of 2,3,7,8-TCDD was 375.3 ppt. The high concentrations of 2,3,7,8-TCDD, especially in the Moorfleeter Canal and the Auserer Vering Canal was attributed to discharges from an organochlorine pesticide manufacturing plant. The patterns of 2,3,7,8-TCDD and the other Cl_4 CDD congeners is characteristic of the patterns resulting from the production of 2,4,5-T and 2,4,6-Trichlorophenol. In addition, the pattern of the Cl_7 CDF congeners can be linked to emissions from thermal processes employed by chemical industries in the production of chlorinated organic chemicals. The high concentrations of Cl_7 and Cl_8 CDD and CDF homologues may also be the result of other industrial combustion processes in the Hamburg area.

In sediment samples collected from estuaries adjacent to an industrial site in Newark, New Jersey, where chlorinated phenols had been produced, the level of the Cl_8 CDD homologue was many times higher than that of 2,3,7,8-TCDD. The study indicated that there probably is a significant regional source (i.e., combustion and use of a common wood preservative, pentachlorophenol) for the Cl_8 CDD homologue, depleted in 2,3,7,8-TCDD relative to the local industrial source. A high correlation was found between 2,3,7,8-TCDD and 2,3,7,8-TCDF ($R^2=0.87$), which suggests that the industrial site was a major source of 2,3,7,8-TCDF to the natural waters of the study area. An interesting note is that the bottom section (108-111 cm) of one of the sediment cores contained 2,3,7,8-TCDD at a concentration of 21,000 ppt, the highest concentration measured in the study. This value was consistent with deposition of that sample during the mid to later stages of active 2,4,5-T production at the site from the late 1950s to early 1960s. PCDD and PCDF in Hudson River sediment samples primarily comprised the higher chlorinated Cl_6 to Cl_8 CDD and CDF homologues (Petty et al., 1982). Concentrations of the Cl_7 and Cl_8 CDD homologues ranged from 5 to 15 ppb, and the Cl_8 CDD homologue in most instances

accounted for more than half of the total PCDD residue. Likewise, the Cl_7 and Cl_8 CDF homologues occurred at the highest levels (ca. 1 ppb).

Surface sediment samples were collected from several estuaries in the United States (Norwood et al., 1989). The sampling sites included Black Rock Harbor in Bridgeport, Connecticut (an industrialized urban estuary); central Long Island Sound (a relatively clean reference site); Narragansett Bay, Rhode Island (where chemical industries may have contributed to the input); New Bedford Harbor, Massachusetts (a section of which is a National Superfund Site because of PCB contamination); and Eagle Harbor, Washington (the site of a creosote wood treatment facility). The sediments in New Bedford Harbor were reported to be more heavily contaminated with PCDFs, especially with regard to the Cl_6 congeners which were greater by a factor of 40 (although individual data points were not presented). Sediments from Eagle Harbor, on the other hand, practically were devoid of PCDFs, and showed a large increase in the Cl_7 and Cl_8 CDD congeners closer to the treatment facility. Whereas Narragansett Bay and Black Rock Harbor were quite similar in both concentration and distribution of PCDDs and PCDFs, Black Rock Harbor contained slightly greater levels of the Cl_4 to Cl_6 CDD and CDF congeners. Sediment from Long Island Sound was cleaner and had a distribution of PCDFs between that of Narragansett Bay and Black Rock Harbor. Sediment with the least contamination was collected in New Bedford Harbor, up-river from the PCB facilities, and the highest Cl_8 CDD congener concentration (1400 ppt) was detected in Eagle Harbor.

Sediment samples from Siskiwit Lake, on Isle Royale, Lake Superior were examined to evaluate the atmospheric input of PCDDs and PCDFs to the lake (Czuczwa et al., 1984). The water level in Siskiwit Lake is 17 meters higher than that in Lake Superior, and in addition, there are no anthropogenic inputs in the drainage basin of Siskiwit Lake. Consequently, the atmosphere is the only source of anthropogenic chemicals in that lake. The Cl_8 CDD homologue was most predominant, and the Cl_7 CDD and CDF homologues also were abundant. The study indicated that the considerable decrease in concentration of all PCDD and PCDF between 6 and 8 cm of the sediment core depth was the result of the "1940 horizon."

Surficial sediments collected from Jackfish Bay on the north shore of Lake Superior contained moderate concentrations of the Cl_4 CDF and Cl_8 CDD homologues, with trace concentrations of other congeners (Sherman et al., 1990). The magnitude of the Cl_8 CDD

homologue was similar to that found in the Siskiwit Lake sediment samples. The Cl_4 CDF and Cl_8 CDD profile for a sediment core collected from Moberly Bay was similar to the surficial sediment pattern. Congeners from those homologues predominated at all depths where detectable concentrations occurred. In addition, low concentrations of the Cl_7 CDD, and Cl_5 and Cl_7 CDF homologues were detected. The concentration profile of the Cl_4 CDF homologue showed a relatively high value which dropped abruptly to non-detectable (<60 ppt) below a depth of 10 cm. This abrupt change corresponded to a section date 1973 which reflects an operational change at the pulp mill, resulting in the formation of Cl_4 CDF congeners in the mill effluent some time after 1973.

A survey of surficial harbor sediments collected near a wood preserving plant in Thunder Bay, Ontario, Canada, on the north shore of Lake Superior, found PCDDs and PCDFs, the highest concentrations of which occurred at stations closest to the plant dock, and lower concentrations at locations further from the source (Mckee et al., 1990). No PCDDs or PCDFs were detected below the surficial layer. Cl_4 and Cl_5 CDD homologues were below analytical detection limits in all samples. However, the concentrations of the Cl_6 to Cl_8 CDD homologues increased with the degree of chlorination. The maximum concentrations of the Cl_6 to Cl_8 CDD homologues ranged from 5,700 ppt for the Cl_6 homologue to 980,000 ppt for the Cl_8 homologue. As with the PCDD homologue distribution profile, the Cl_6 to Cl_8 CDF homologues increased with the degree of chlorination.

Bottom surficial sediments (0-3 cm) were collected from the sedimentation basins of Lake Ontario to assess the levels of the various PCB congeners there (Oliver and Niimi, 1988). Concentrations of PCB congeners 118, 105, and 156 in the sediment were 15, 10, and 2.1 ppb, respectively. A baseline assessment of PCDDs and PCDFs was performed on the Elk River, a semi-rural area located about 25 miles northwest of Minneapolis-St. Paul, Minnesota (Reed et al., 1990). Sediment samples were collected from Lake Orono, a reservoir on the Elk River, and from an abandoned gravel pit. Although none of the sediment samples contained 2,3,7,8-TCDD, the gravel pit sediments contained measurable concentrations of the Cl_4 CDF homologue. Only one of the Lake Orono samples contained measurable concentrations of 2,3,7,8-TCDF (0.31 ppt) and total Cl_4 CDF (0.54 ppt) homologue. The gravel pit samples also contained Cl_6 to Cl_8 CDD and Cl_5 to Cl_8 CDF homologues. Lake Orono samples contained Cl_7 and Cl_8 CDD and Cl_7 CDF

homologues. The Cl_7 CDD homologue ranged from 7.3 ppt in the lake inlet to 110 ppt in the gravel pit and the lake, near the dam. The Cl_8 CDD homologue ranged from 450 ppt in the gravel pit to 600 ppt in the middle of Lake Orono. The sediment profiles were reminiscent of combustion source influences.

The Sheboygan River, a Wisconsin tributary to Lake Michigan, is polluted with PCBs from the mouth to about 14 miles upstream (Sonzogni et al., 1991). That portion of the river is a Superfund site as well as one of the Great Lakes "Areas of Concern." Sediment cores were collected at Rochester Park, near the original source of the PCBs, about 14 miles upstream from the mouth. The PCB congeners 118, 105, and 77 were detected in all samples, and ranged from about 5 to 1500 ppb. The remaining toxic PCB congeners, 81, 114, 167, 126, and 169 were detected less frequently and ranged from non-detectable to slightly over 100 ppb. PCB congener 118 appears to be the most common toxic congener in environmental samples, and was found in the Sheboygan River sediments in the highest weight percent. The eight toxic PCBs presented in this study were present in relatively low concentrations compared to total PCBs or other more abundant congeners. The concentrations of these congeners were also low relative to the congeners with which they co-elute.

Sediments collected from Waukegan Harbor in Lake Michigan contained PCB congeners 77 and 105 (Huckins et al., 1988). The percentage of PCB 77 (0.16 +/- 0.15) in the samples varied by 1.4 orders of magnitude, with concentrations ranging from 13 to 27,500 ppb. The percentage of PCB 105 averaged 0.66 +/- 0.37, with concentrations ranging from 102 to 131,000 ppb. In another Lake Michigan study, sediment samples collected from Green Bay contained concentrations of the toxic PCB congeners 81, 77, 118, 114, 105, 126, 167, 156, 157, 169, and 189 (Smith et al., 1990). PCB 118 and 105 were the dominant congeners, with concentrations of 11 and 5.8 ppb, respectively.

Some general observations for PCDD and PCDF levels are possible from the data presented in the various sediment studies above:

- The PCDD and PCDF homologue distribution patterns in sediment generally follow those exhibited by the contaminant source.

- The concentration of Cl_6 to Cl_8 PCDD and PCDF homologues in sediment are usually the result of industrial combustion processes and generally increase with the degree of chlorination, but decrease uniformly with distance from the source.

3.5. CONCENTRATIONS IN FISH AND SHELLFISH

Tables B-8 through B-10 (Appendix B) contain summaries of data from the published literature regarding concentrations of PCDDs, PCDFs, and coplanar PCB congeners in fish and shellfish. PCB congener data were found only for North American species.

Evaluation of fish in the Baltic Sea (Gulf of Bothnia) and Northern Atlantic Ocean in the vicinity of Sweden revealed that concentrations of PCDDs and PCDFs in herring from the Atlantic Ocean were lower than those in the Gulf of Bothnia (Rappe et al., 1989b). Detectable levels of 2,3,7,8-TCDD in salmon were found in both wild homing (4.6 to 19 ppt) and hatchery-reared (0.2 to 0.3 ppt) varieties in the Gulf of Bothnia. In addition, concentrations of the same representative congeners of the Cl_5 to Cl_8 CDD and CDF homologues found in herring were found in both varieties of salmon. Levels of those congeners in the wild salmon, however, were five to ten times higher than the herring levels, while the levels in the hatched salmon essentially were the same as in the herring samples. Perch collected at a distance of 1-6 km from a pulp mill in the southern part of the Gulf of Bothnia contained 2,3,7,8-TCDD and 2,3,7,8-TCDF, the levels of which were higher in the samples collected closer to the pulp mill. Those two isomers have been identified in bleaching effluents from pulp mills as well as in bleached pulp. Arctic char collected from Lake Vattern, a popular fishing lake in southern Sweden, contained levels of 2,3,7,8-TCDD (6.5 to 25 ppt), 2,3,7,8-TCDF (21 to 75 ppt), and representative congeners of the Cl_5 CDD and CDF homologues. There was a good correlation between the weight of the fish and the levels of PCDDs and PCDFs. The main general pollution sources of the long, deep, narrow lake are two pulp mills.

Fish (cod, haddock, pole flounder, plaice, flounder, and eel), mussels, and edible shrimps from a fjord area contaminated by wastewater from a magnesium factory in Norway were analyzed for PCDDs and PCDFs (Oehme et al., 1989). Magnesium production results in the formation of substantial amounts of PCDDs and PCDFs as

byproducts. The pattern of Cl_4 and Cl_5 CDD and CDF homologues released in wastewater during the magnesium production process is very characteristic, and is dominated by congeners with chlorine in the positions 1,2,3,7, and/or 8. Fish and shellfish differ in their ability to bioconcentrate PCDD and PCDF congeners. For example, fish generally only concentrate the most toxic 2,3,7,8-substituted congeners, whereas shellfish can usually concentrate most of the congeners. Nearly all congeners were present in the shrimp and mussel samples. Whereas these organisms displayed the very characteristic Cl_5 CDF congener pattern of the magnesium production process, some deviations were found in the Cl_4 CDF congener distribution within those species. For fish, the concentrations of PCDDs and PCDFs are dependent on the exposure level, fat content, living habit, and how stationary the species is. The highest PCDD and PCDF levels were found in comparatively fat bottom fish collected close to the source. Cod and haddock, leaner, non-stationary fishes, had much lower concentrations, even in the vicinity of the magnesium production factory. An interesting note is that the main stream of the fjord follows the west coast, and subsequently, cod and eel samples collected along the west coast of the fjord had considerably higher levels of PCDDs and PCDFs than those collected from the eastern fjord entrance. Similarly, the level of 2,3,7,8-TCDD in mussels decreased by one order of magnitude from the vicinity of the magnesium production factory to the outer region of the fjord system.

Brown trout, grayling, barbel, carp, and chub collected in the Neckar River in southwest Germany contained much higher levels of 2,3,7,8-TCDF than in eels collected from the same river and the Rhine River (Frommberger, 1991). In addition, eels from both rivers showed very similar patterns for PCDD and PCDF congener distribution, whereas the patterns of PCDD and PCDF distribution generally showed some degree of difference among the other fishes collected from the Neckar River. Perch and bream collected from various locations in the vicinity of Hamburg Harbor, however, showed similar patterns in the distribution of the Cl_4 to Cl_8 CDD and CDF homologues (Gotz et al., 1990). In general, however, the levels of PCDFs were higher than the level of PCDDs in these fishes, especially with regard to the Cl_4 to Cl_6 CDF homologues. Pooled samples of eels collected at six different localities in the Netherlands contained low levels of PCDDs and PCDFs, the major congeners of which were 2,3,7,8-chlorine substituted (Van den Berg et al., 1987). The concentrations of the various congeners identified in the eel samples ranged from 0.1

to 9.1 ppt. The sample with the highest concentration of 2,3,7,8-TCDD (9.1 ppt) was collected from Broekervaart in a location that was not far from a chemical waste dump that contained high concentrations of the same congener.

Samples of striped bass, blue crabs, and lobsters collected from Newark Bay and the New York Bight all contained high levels (up to 6200 ppt) of 2,3,7,8-chlorine substituted tetra- and penta-CDDs and CDFs (Rappe et al., 1991). The levels of 2,3,7,8-TCDD were higher than any other New Jersey samples, and the highest sample in this study may be the highest level of 2,3,7,8-TCDD ever reported for aquatic animals. The crustaceans resembled one another in congener pattern. Specifically, they all contained both a large number and large amounts of PCDD and PCDF congeners in addition to the 2,3,7,8-chlorine substituted compounds. The striped bass samples, on the other hand, primarily contained only the 2,3,7,8-chlorine substituted congeners.

Carp, catfish, striped bass, large mouth bass and lake trout were collected from sites in the Hudson River and the Great Lakes Basin that were contaminated with industrial chemicals or contained known or suspected levels of PCBs (Gardner and White, 1990). The congener 2,3,7,8-TCDF was detected in 12 fish at levels which ranged from 3 to 93 ppt. A 2,3,7,8-chlorine substituted penta-CDF was detected in 14 fish at levels ranging from 4 to 113 ppt. An interesting observation in this study was that 2,4,6-chlorine substituted CDFs were detected in four of the fish samples, suggesting that those fish may have been exposed to chlorinated phenols. The study indicated that the 2,4,6-chlorine substituted CDFs occurred in the fish at levels similar to those of the 2,3,7,8-chlorine substituted CDFs, but with less frequency.

Samples of lake trout or walleye collected from each of the Great Lakes and Lake St. Clair were analyzed for PCDDs and PCDFs (De Vault et al., 1989). PCDF and PCDD concentrations in lake trout were substantially different from each lake and between sites in Lake Michigan, probably reflecting differences in types and amounts of loadings to the lakes.

In all of the sampling sites except Lake Ontario, 2,3,7,8-TCDF was the dominant furan congener in lake trout and ranged from 14.8 ppt in Lake Superior to 42.3 ppt in Lake Michigan. In Lake Ontario, the dominant congener in lake trout was a 2,3,7,8-chlorine substituted penta-CDF. The distribution of PCDF congeners in the Lake Erie walleye were very similar to that of the lake trout from Lake Superior. With regard to dioxins, the

concentrations of 2,3,7,8-TCDD ranged from 1 ppt in Lake Superior to 48.9 ppt in Lake Ontario. With the exception of Lake Ontario, the dominant dioxin congener was a 2,3,7,8-chlorine substituted penta-CDD. A 2,3,7,8-chlorine substituted hexa-CDD also contributed significantly to the total PCDD concentrations. As with PCDFs, the distribution of PCDD congeners in the Lake Erie walleye was very similar to that of the lake trout from Lake Superior.

In another study, PCDDs and PCDFs were measured in four species of salmonids (lake trout, coho salmon, rainbow trout, and brown trout) that were collected from Lake Ontario (Niimi and Oliver, 1989a). Levels of 2,3,7,8-TCDD in whole fish ranged from 6 to 20 ppt, and the CL_6 CDD homologue was most dominant in all fishes. High levels of the CL_8 CDD homologue also were detected in lake trout and coho salmon, but not in rainbow trout or brown trout. Although the total PCDF levels were about 25 percent lower than the total PCDD concentrations, the levels of 2,3,7,8-TCDF which was the dominant component of the CL_4 CDF homologue was the same range as 2,3,7,8-TCDD (6 to 20 ppt). However, the study suggested that whereas collection sites can influence chemical levels and congener composition, comparisons of chemical levels and congener frequencies may not be suitable because of differences resulting from localized factors. The study also indicated that the importance of the various PCDD and PCDF congeners can differ with location (i.e., the same species of fish collected at different locations in a study area may reveal that the most common congener is different at each site).

Channel catfish, carp, yellow perch, small mouth bass, sucker, and lake trout were collected from the Tittabawassee, Grand, and Saginaw Rivers, Lake Michigan, and Saginaw Bay, and analyzed for 2,3,7,8-TCDD residues (Harless and Lewis, 1980). Detectable quantities were found in 26 of 36 samples, and ranged from 4 to 695 ppt. The bottom feeders (catfish and carp) contained the highest concentrations, and the lowest levels were found in bass, perch, and sucker. Because the Grand and Saginaw Rivers are not connected to the Tittabawassee River, the presence of 2,3,7,8-TCDD in fish from those rivers suggests that several separate bodies of water are contaminated with TCDD residues.

Travis and Hattemer-Frey (1991) evaluated the data generated as part of the National Dioxin Study regarding 2,3,7,8-TCDD concentrations in fish. The fish were collected from 304 urban sites in the vicinity of population centers or areas with known

commercial fishing activity, including sites from the Great Lakes Region. Data from that study indicated that concentrations of 2,3,7,8-TCDD in whole fish from urban sites ranged from non-detectable to 85 ppt. In addition, only 29 percent of the fillets from urban sites had detectable levels of 2,3,7,8-TCDD, with a geometric mean concentration of 0.3 ppt. Whole fish samples from the Great Lakes Region had higher 2,3,7,8-TCDD levels than fish from urban areas (e.g., 80% vs 35% detectable levels). In the Great Lakes Region, 2,3,7,8-TCDD concentrations in whole fish samples ranged from nondetectable to 24 ppt, with a geometric mean of 3.8 ppt. These levels were ten times higher than the concentration in whole fish from urban areas. Likewise, the mean concentration of 2,3,7,8-TCDD in Great Lakes Region fish fillets (2.3 ppt) was about seven times higher than the levels in the fillets from urban areas (0.3 ppt). As with the whole fish samples, fish fillet samples from the Great Lakes Region had higher 2,3,7,8-TCDD levels than fillets from background urban areas (e.g., 67% vs 29% detectable levels). Comparable levels of 2,3,7,8-TCDD were detected in whole bottom feeders and predators from the Great Lakes Region.

Samples from all trophic levels in the Lake Ontario ecosystem were analyzed for PCB congeners (Oliver and Niimi, 1988). Analysis revealed that the PCB concentration increased from water to lower organisms to small fish to salmonids, demonstrating the classical biomagnification process. In addition, the chlorine content of the PCBs increased at the higher trophic levels. PCBs with the highest chlorine content (57%) were found in sculpin, small bottom-living fish that feed on benthic invertebrates. The CL_3 and CL_4 PCB homologues comprised a much higher percentage of the PCBs in the lower trophic levels than in salmonids and small fish. The percentage of CL_5 and CL_8 PCB homologues in all samples was fairly uniform, but the CL_6 and CL_7 PCB homologues comprised a much larger fraction of the PCBs in the small fish and salmonids than in the lower trophic levels.

A study regarding the distribution of PCBs in Lake Ontario salmonids (brown trout, lake trout, rainbow trout, and coho salmon) showed that the CL_5 and CL_6 PCB homologues were dominant in all species (Niimi and Oliver, 1989b). The 10 most common PCB congeners represented about 52 percent of the total content and did not appear to be influenced by species or total concentration. The homologues observed averaged approximately 56 percent chlorine by weight in whole fish and muscle. The analysis of the

chlorine content suggested that the more persistent congeners tend to behave as a homogeneous mixture instead of as individual congeners.

A large quantity of fish data were collected as part of EPA's National Bioaccumulation Study during the period of 1986 to 1989. The report associated with this study has been through several reviews, is currently in final Agency clearance procedures and is expected to be released in 1993. Although the report has not yet been released, the raw data is public information (Personal communication from Richard Healy, US EPA Office of Water in Washington, DC, July 22, 1992). Based on this raw data, several summaries were prepared and are presented here. Tables B-8 and B-9 include the dioxin and furan data collected from a wide variety of sites across the U.S. including 314 sites thought to be influenced by point or nonpoint sources, and 34 sites identified as relatively free of influence from point and nonpoint sources. This latter group of sites can be characterized as background per the definition used in this document. Finally, Table B-10 includes similar data for the various PCB congener groups from 362 sites. Since, the specific PCB congeners could not be identified, it is not known what percentage of these concentrations represent the PCBs identified as dioxin-like. Twenty of these sites were identified as background sites. The total PCB, all 209 congeners, mean concentration for these background sites was 46,900 ppt. Since the dioxin-like PCBs consist of only 11 of the 209 possible PCB congeners, it may be that they are a small percentage of the total. However, only congener specific analysis can ultimately confirm this.

Some general observations for PCDD and PCDF levels are possible from the data presented in the various fish and shellfish studies above:

- Fish and shellfish differ in their ability to bioconcentrate PCDD and PCDF congeners. Fish generally concentrate the most toxic 2,3,7,8-substituted congeners, but shellfish can usually concentrate most of the congeners.
- For fish, the concentrations of PCDDs and PCDFs are dependent on the exposure level, fat content, living habit, and the degree of movement of the species. Comparatively fat bottom fish collected close to the contaminant source generally have the highest PCDD/PCDF levels, whereas leaner, non-

stationary fishes have much lower concentrations, even in the vicinity of the contaminant source.

- The National Dioxin Study indicated that the levels of 2,3,7,8-TCDD in fish from the Great Lakes Region were higher than those from urban areas. Comparable levels were detected in whole bottom feeders and predators from the Great Lakes Region.
- With regard to PCBs, concentrations increase from water to lower organisms to small fish to salmonids, and the chlorine content of the PCBs increase at the higher trophic levels.

3.6. CONCENTRATIONS IN FOOD PRODUCTS

Tables B-11 and B-12 (Appendix B) contain summaries of data from the recent published literature regarding concentrations of PCDDs and PCDFs in food products. Most of the selected studies investigated "background" levels of PCDDs and PCDFs rather than studies targeted at areas of known contamination. Data on relevant coplanar PCB congener concentrations in food products were not found in the literature.

The studies summarized in Tables B-11 and B-12 primarily examined PCDD and PCDF levels in products of animal origin (i.e., fish, meat, eggs, and dairy products). Because of their lipophilic nature, PCDDs and PCDFs are expected to accumulate in these food groups. The data in the tables indicate that PCDDs and PCDFs are found at levels ranging from the intermediate ppq up to the low ppt range. As expected, the highest levels reported are those measured in fat. The highest reported congener concentrations are for the Cl_7 and Cl_8 CDDs. In general, for the less-chlorinated homologue groups (i.e., Cl_4 - Cl_6), the CDF levels measured were larger than the CDD levels but were still within an order of magnitude. The situation is reversed for the Cl_7 and Cl_8 homologue groups.

Few data have been reported on levels of PCDDs and PCDFs in foods of vegetative origin. Beck et al. (1989) found no PCDDs or PCDFs in samples of vegetable oil, cauliflower, lettuce, cherries, and apples bought in stores in Berlin. The detection limit was approximately 0.01 ppt for each isomer on a whole-weight basis. Analyses of three food baskets collected from stores in Stockholm showed PCDD and PCDF levels close to or below detection limits (0.1 to 0.5 pg/g detection limits) (de Wit et al., 1990). The only congeners found were 2,3,7,8-TCDF (0.1 to 0.4 pg/g) and Cl_8 CDD (1.0 to 1.3 pg/g).

It is generally assumed that consumption of food containing trace levels of PCDDs and PCDFs is responsible for the majority of human exposure to these compounds. Travis and Hattemer-Frey (1991), using a fugacity model, estimated that the food chain, especially meat and dairy products, accounts for 99 percent of human exposure to 2,3,7,8-TCDD. Despite these assumptions and conclusions, few data on actual levels of PCDDs, PCDFs, and dioxin-like compounds in food products have been published. Those data that have been published have generally resulted from studies of a specific food product(s) in a specific location(s) rather than from survey studies designed to allow estimation of daily intake of the chemicals for a population. The detection limits used in large survey studies, such as the FDA foodbasket survey, have generally been too high to detect the low levels of these chemicals likely to be present. Analysis of trace levels of these chemicals has in the past been hindered by lack of sensitive analytical detection methods, extraction difficulties from the high-lipid content food products in which these chemicals are most often found, and the presence of potentially interfering other organochlorine compounds. As these difficulties have been and continue to be overcome, the results of more narrow scope studies, as well as broad scope surveys, are likely to be reported. For example, relatively extensive multiyear surveys of the levels of PCDDs and PCDFs in food are being undertaken in the United Kingdom and Sweden (de Wit et al., 1990 and Startin et al., 1990).

3.7. CONCENTRATIONS IN AIR

This section summarizes measurements of dioxin-like congeners in ambient air. The purpose of sampling and monitoring the ambient air was to provide an approximate indication of the background concentration of PCDD and PCDF compounds in urban environs. Analysis of ambient air is extremely difficult because of the low analytical detection limits required to detect specific congeners. Minimum limits of analytical detection are in the range of 0.05 to 0.4 picograms of the specific congener per cubic meter of air. These limits of detection in ambient air samples were not achieved until the mid 1980's. In general, an air sampling apparatus consisting of a quartz-fiber filter and polyurethane foam (PUF) is used for collection of samples. The quartz-fiber filter collects airborne particulate, and the PUF collects vapors. The quartz-fiber filter is usually spiked with a carbon-labelled isotope of the specific congener prior to operation to determine the

collection and retention efficiency for the PCDDs and PCDFS. To obtain subparts-per-trillion levels of analytical detection requires sampling relatively large volumes of air over a 24-hour period, e.g., 350 to 450 cubic meters of ambient air.

Tables B-13 and B-14 (Appendix B) provide a general overview of the limited ambient air measurements that have been made in selected cities in the U.S. and Europe. It is interesting to note that there appears to be good agreement with respect to the magnitude of specific congeners of PCDDs and PCDFs in urbanized areas in the U.S. and Europe, and specific dioxin-like congeners are quantified from 1/10 to 1/100 picograms per cubic meter of air sample. Most of these measurements tend to be very close to the current analytical detection limit. This increases the probability that congeners indicated in Table 3-13 as Not Detected (ND) may actually be present, but will not be observed until further advances are made in analytical methods. The value in parentheses after the designation ND on the Table was the reported detection limit (pg/M3). The designation NA in Table 3-13 means the investigator did not analyze that specific congener in the sample.

3.8 HISTORICAL TRENDS

Small amounts of dioxin-like compounds may be formed during natural fires suggesting that these compounds may have always been present in the environment. However, it is generally believed that much more of these compounds have been produced in association with industrial processes and therefore environmental levels are likely to be higher in modern times.

Rappe (1991) reports testing of archived soils and plants collected in southeast England between 1846 and 1986. PCDDs and PCDFs were found in all samples and showed generally increasing levels of dioxins. Rappe further notes that the congener pattern is typical of those for combustion sources until about 1950 when the pattern becomes more dominated by hepta- and octa-CDDs corresponding to increases in production of chlorinated compounds.

Schecter (1991) analyzed ancient liver tissues (estimated to be 100 to 400 years old) recovered from frozen bodies of Native American (Eskimo) women. He found that the dioxin levels were much lower than those commonly found in livers of people currently living in industrial areas.

Travis and Hattemer-Frey (1991) used the Fugacity Food Chain (FFC) model to predict the contribution of municipal solid waste incinerators, motor vehicles, hospital waste incinerators, residential wood burning, and pulp and paper mill effluents, to the U.S. total environmental input of 2,3,7,8-TCDD. It was estimated that the total input from all five sources combined accounted for only 11 percent of the total TCDD found in the different media in the U.S. The authors concluded that this low value indicated: (1) that the source term used in the FFC modeling exercise for 2,3,7,8-TCDD may have been too high; (2) some unidentified major source(s) of 2,3,7,8-TCDD exist; or (3) there are multiple environmental sources of TCDD, and no one source dominates the total input.

Rappe (1991) compared the known emission sources of PCDD and PCDF inputs from Sweden with an estimated aerial deposition rate into the Swedish environment. The aerial deposition rate was found to be 10-20 times higher than the total emissions sources originating from Sweden. In an earlier publication, Rappe (1987) characterized sources of PCDD and PCDF sources as well as environmental and human samples. He concluded that there was a poor correlation between the isomeric distribution found in emissions from known sources and human and environmental samples. He speculated that the pattern in human and environmental samples was the result of a combination of sources, coupled with environmental and biological degradation.

3.9 ASSESSMENT OF BACKGROUND EXPOSURES

The occurrence data were analyzed to evaluate background levels and differences between contaminant levels in Europe and North America.

3.9.1. Procedures for Estimating Background Exposures

The term: "background" as applied to exposure can be confusing since it can be used to represent very different concepts and is commonly not defined by the users. The two most common uses are 1) the level of exposure that would occur in an area without known sources of the contaminant of concern or 2) the average level of exposure occurring in an area whether sources are present or not. For purposes of this document we will define background as suggested in the first definition above and use "average" (with a specification of the area of concern) to describe the second concept. The data collected in rural areas far from known contaminant sources are probably the most useful

for describing background since the fewest sources of dioxin-like compounds are likely to be present.

Using the above definition, background data was considered to be those collected in areas defined as rural, pristine or residential unless known sources were also reported in the vicinity. Tables B-15 through B-27 in Appendix B show the geometric and arithmetic averages of these background data for each media. The geometric averages are consistently lower than the arithmetic levels. This results from the fact that the data spans several orders of magnitude with the distribution skewed toward the lower end (due to the large number of nondetects). If the data were collected in a truly random fashion, and human contact occurred randomly, then an arithmetic average would be more appropriate for estimating average exposures. However, most of these data cannot be considered random and since the geometric average should provide a better indication of the central tendency (due to the skewed distribution), it was selected as the best way to estimate a level representative of typical exposure levels.

3.9.2. Geographic Comparisons

Another issue associated with background levels of the dioxins, concerns how the levels differ between various geographic areas, particularly North America and Europe. As shown in Table 3-1, the available data were divided into these areas and averaged. Considering the similar levels of industrialization and lifestyle between Europe and North America, one may expect similar environmental levels of the dioxin-like compounds. However, the geographic averages shown on Table 3-1, indicate some discrepancies. It should be emphasized that it is not really known how representative these averages are of such large areas. The North America data were collected from approximately 20 different sites (all media) and the European data were collected from about 18 sites (all media), but almost all were located in the United Kingdom. Thus, the significance of these discrepancies is largely unknown due to the relatively sparse database and uncertainties regarding representativeness.

The exposures associated with these levels can be estimated by multiplying the levels by typical contact rates. Table 3-2 shows the results of this analysis. Bearing in mind the limitations of the occurrence database noted above, some observations from this table include:

Table 3-1. Regional Comparisons of Toxic Equivalent Concentrations

Media		Worldwide[1]	North America[1]	Europe[1]
Soil, ppt:	CDD (TEq)	4.12	5.24	0.64
	CDF (TEq)	2.91	2.45	2.93
	Total CDD + CDF (TEq)	7.03	7.69	3.57
Sediment, ppt:	CDD (TEq)	16.2	3.24	25.0
	CDF (TEq)	1.18	0.03	3.13
	Total CDD + CDF (TEq)	17.4	3.27	28.1
	PCB (ng/kg)[2]	7.42-14,881	250-14,881	6.1-250
Fish, ppt:	CDD (TEq)	0.90	0.94	0.19
	CDF (TEq)	0.46	0.43	0.87
	Total CDD + CDF (TEq)	1.36	1.37	1.06
Air, pg/m^3:	CDD (TEq)	0.018	0.019	0.0059
	CDF (TEq)	0.031	0.032	0.012
	Total CDD + CDF (TEq)	0.049	0.051	0.018
Water, ppq:	CDD (TEq)	0.86	0.86	NDA
	CDF (TEq)	7.40	7.40	NDA
	Total CDD + CDF (TEq)	8.26	8.26	NDA
Dairy, ppt[3]:	CDD (TEq)	0.0427	NDA	0.0427
	CDF (TEq)	0.0568	NDA	0.0568
	Total CDD + CDF (TEq)	0.0995	NDA	0.0995
Beef, ppt[3]:	CDD (TEq)	0.18	NDA	0.18
	CDF (TEq)	0.19	NDA	0.19
	Total CDD + CDF (TEq)	0.37	NDA	0.37

Footnotes

NDA = No data available.

[1] Values are the geometric mean TEqs from Tables B-15 to B-27.

[2] Values are the ranges of the geometric means across all PCBs from Table B-20.

[3] Values calculated for beef and dairy only from Beck et al. (1989), Furst et al. (1990) and Schecter et al. (1990), and expressed in table on whole product basis - fat basis given by authors converted to whole product basis by multiplication by fat content: beef fat by 0.22, and dairy fat by 0.035.

Table 3-2. Background Exposures

Media	Worldwide				North America				Europe			
	Concentration TEq[1]	Contact rate[2]	Daily intake mg/day	% of total	Concentration TEq[1]	Contact rate[2]	Daily intake mg/day	% of total	Concentration TEq[1]	Contact rate[2]	Daily intake mg/day	% of total
Soil ingestion	7.03×10^{-12}	100 mg/day	7.03×10^{-10}	0.79	7.69×10^{-12}	100 mg/day	7.69×10^{-10}	3.43	3.57×10^{-12}	100 mg/day	3.57×10^{-10}	0.48
Fish ingestion	1.36×10^{-9}	6.5 g/day	8.84×10^{-9}	9.92	1.37×10^{-9}	6.5 g/day	8.91×10^{-9}	39.7	1.06×10^{-9}	6.5 g/day	6.89×10^{-9}	9.25
Inhalation	4.9×10^{-11}	23 m^3/day	1.13×10^{-9}	1.27	5.1×10^{-11}	23 m^3/day	1.17×10^{-9}	5.21	1.8×10^{-11}	23 m^3/day	4.14×10^{-10}	0.57
Water ingestion	8.26×10^{-9}	1.4 L/day	1.16×10^{-8}	13.0	8.26×10^{-9}	1.4 L/day	1.16×10^{-8}	51.7	NDA	1.4 L/day	NA	NA
Dairy ingestion	9.95×10^{-11}	300 g/day	2.98×10^{-8}	33.5	NDA	300 g/day	NA	NA	9.95×10^{-11}	300 g/day	2.98×10^{-8}	40.0
Beef ingestion	3.70×10^{-10}	100 g/day	3.70×10^{-8}	41.5	NDA	100 g/day	NA	NA	3.70×10^{-10}	100 g/day	2.94×10^{-8}	49.7
	Total		8.91×10^{-8}		Total		2.24×10^{-8}		Total		7.45×10^{-8}	

Footnotes: NA = Not applicable, NDA = No Data Available.
[1] Values from Table 3-1, and units were converted for calculation of daily intake rate.
[2] Values from Exposure Factors Handbook (USEPA, 1989)

- Exposure via beef or dairy ingestion in North America could not be estimated due to lack of occurrence data. Since these pathways accounts for about two thirds of the total exposure in Europe, the North America exposure estimates may be low by this amount.
- Exposure via fish ingestion appears much higher in North America than Europe.
- The total background exposures to TEq shown in Table 3-2 were 74, 89, and 22 pg/day for Europe, Worldwide, and North America, respectively. Because of the database weaknesses noted above, these discrepancies cannot be considered be significant. It is probably more appropriate to conclude that permutations of the data tested in this exercise indicates a range, perhaps, of 20-90 pg/day TEq exposure. Other qualifying considerations include: 1) the addition of the pathways presumes that individuals are exposed by all pathways, 2) on the other hand, several exposures such as those from other meat products, are not considered, and 3) the fraction absorbed into the body is assumed to be the same for all pathways. Thus, these estimates must be considered highly uncertain. Similar analyses have been done by other investigators. Furst et al. (1991) estimated average exposures to the dioxin-like compounds as 70 to 200 pg of TEq/day. Travis and Hattemer-Frey (1991), estimated typical exposures to 2,3,7,8-TCDD only may be in the range of 35 pg/day.
- This exercise, and those undertaken by others, suggest that over 90% of the exposure is associated with ingestion of fish, beef and dairy products, with the remainder due to air inhalation, soil ingestion, and water ingestion.
- The Worldwide exposure estimate of 89 pg/day was examined further to determine how much of the exposure was due to 2,3,7,8-TCDD and how much was due to other congeners. It was found that 14 pg/day was due to 2,3,7,8-TCDD, which is 16 percent of the total exposure. Again, this is an uncertain result. It was driven by the result for fish, dairy and beef ingestion, which was 13 pg/day 2,3,7,8-TCDD. Fish, dairy and beef ingestion contributed approximately equal amounts to the 2,3,7,8-TCDD exposure. Examination of the fish data from which this estimate is derived (Table B-22 in Appendix B) shows that there were very few congener specific analyses for fish except for 2,3,7,8-TCDD and 2,3,7,8-TCDF.

Examination of body burden data may provide a more accurate way to estimate typical exposures. However, these data may not represent true background levels as

defined here since the sampled individuals may have lived in urban or residential areas where dioxins sources were present. Levels of these compounds found in human tissue/blood appear similar in Europe and North America. Schecter (1991) compared levels of dioxin-like compounds found in blood among people from U.S. and Germany. Although mean levels of individual compounds differed by as much as a factor of two, the total PCDD/Fs averaged 42 ppt of TEq in Germany and 41 ppt of TEq in the U.S. Schecter (1991) also reports tissue levels in various countries:

Country	Mean Tissue Level (ppt of TEq)
USA	24
Germany	69
China	18
Japan	38
Canada	36
S. Vietnam	30
N. Vietnam	4

The tissue data shows more variation between countries but also involved much fewer samples, reducing confidence in the accuracy of the mean. In Chapter 8, the level of 2,3,7,8-TCDD found in human adipose tissue is reported to average about 5.0 to 6.7 ppt in the U.S. based on data from a variety of studies. This data was used to estimate the associated exposure levels using a simple pharmacokinetic model which back calculated the dose needed to achieve these tissue levels under the assumption of steady state. This model requires an estimate of the elimination rate constant. Based on available data this was assumed to be about 5.8 to 7 years which yielded a background dose rate of about 10 to 31 pg/day. This estimate agrees very well with the background exposure estimates (to 2,3,7,8-TCDD only) of 35 pg/day by Travis and Hattemer-Frey (1991) and 25 pg/day by Furst et al. (1991), both derived using typical media levels and contact rates. Further discussion on body burden data and associated exposures is presented in Chapter 8.

REFERENCES FOR CHAPTER 3

Beck, H.; Eckart, K.; Kellert, M.; Mathar, W.; Ruhl, C.S.; Wittkowski, R. (1987) Levels of PCDFs and PCDDs in samples of human origin and food in the Federal Republic of Germany. Chemosphere 16 (8/9): 1977-1982.

Beck, H.; Eckart, K.; Mathar, W.; Wittkowski, R. (1989) PCDD and PCDF body burden from food intake in the Federal Republic of Germany. Chemosphere 18 (1-6): 417-424.

Birmingham, B. (1990) Analysis of PCDD and PCDF patterns in soil samples: use in the estimation of the risk of exposure. Chemosphere 20(7-9): 807-814.

Bopp, R.F.; Gross, M.L.; Tong, H.; Simpson, M.J.; Monson, S.J.; Deck, B.L.; Moser, F.C. (1991) A major incident of dioxin contamination: sediments of New Jersey estuaries. Environ. Sci. Technol. 25: 951-956.

Branson, D.R.; Takahashi, I.T.; Parker, W.M.; Blau, G.E. (1985) Bioconcentration kinetics of 2,3,7,8-tetrachlorodibenzo-p-dioxin in rainbow trout. Environ. Toxicol. Chem. 4(6):779-788.

Broman, D.; Naf, C.; Rolff, C.; Zebuhr, Y. (1990) Analysis of polychlorinated dibenzo-p-dioxins (PCDD) and polychlorinated dibenzofurans (PCDF) in soil and digested sewage sludge from Stockholm, Sweden. Chemosphere 21: 1213-1220.

CDEP. (1988) Measurement of selected polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans in ambient air in the vicinity of Wallingford, Connecticut. Connecticut Department of Environmental Protection, Air Compliance Unit, Hartford, CT. Project report by ERT, Concord, MA., Project # 7265-001-004. July 8,1988.

Creaser, C.S.; Fernandes, A.R.; Al-Haddad,A.; Harrad, S.J.; Homer, R.B.; Skett, P.W.; Cox, E.A. (1989) Survey of background levels of PCDDs and PCDFs in UK soils. Chemosphere 18(1-6): 767-776.

Creaser, C.S.; Fernandes, A.R.; Harrad, S.J.; Cox, E.A. (1990) Levels and sources of PCDDs and PCDFs in urban British soils. Chemosphere 21: 931-938.

Czuczwa, J.M.; McVeety, B.D.; Hites, R.A. (1984) Polychlorinated dibenzo-p-dioxins and dibenzofurans in sediments from Siskiwit Lake, Isle Royale. Science 226: 568-569.

DeVault, D.; Dunn, W.; Bergquist, P.A.; Wiberg, K.; Rappe, C. (1989) Polychlorinated dibenzofurans and polychlorinated dibenzo-p-dioxins in Great Lakes fish: a baseline and interlake comparison. Environ. Toxicol. Chem. 8: 1013-1022.

deWit et al. (1990) Results from the first year of the Swedish dioxin survey. Chemosphere 2 (10-12): 1473-1480.

Edgerton, S.A.; Czuczwa, J.M.; Rench, J.D.; Hodanbosi, R.F.; Koval, P.J. (1989) Ambient air concentrations of polychlorinated dibenzo-p-dioxins and dibenzofurans in Ohio: sources and health risk assessment. Chemosphere 18(9/10):1713-1730.

Frommberger, R. (1991) Polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans in fish from south-west Germany: river Rhine and Neckar. Chemosphere 22(1-2): 29-38.

Furst, P.; Furst, C.; Groebel, W. (1990) Levels of PCDDs and PCDFs in food-stuffs from the Federal Republic of Germany. Chemosphere 20 (7-9): 787-792.

Gardner, A.M.; White, K.D. (1990) Polychlorinated dibenzofurans in the edible portion of selected fish. Chemosphere 21(1-2): 215-222.

Gotz, R.; Schumacher, E. (1990) Polychlorierte dibenzo-p-dioxine (PCDDs) und polychlorierte dibenzofurane (PCDFs) in sedimenten und fischen Aus Dem Hamburger Hafen. Chemosphere 20(1-2): 51-73.

Harless, R.L.; Lewis, R.G. (1982) Quantitative determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin residues by gas chromatography/mass spectrometry. In: Hutzinger, O.; Frei, R.W.; Merian, E.; Pocchiari, F., eds. Chlorinated dioxins and related compounds: impact on the environment, pergamon series on environmental science. Vol. 5. Oxford: Pergamon Press. pp. 25-35.

Harless, R.L.; McDaniel, D.D.; Dupuy, A.E. (1990) Sampling and analysis for polychlorinated dibenzo-p-dioxins and dibenzofurans in ambient air. Proceedings of the Tenth International Symposium on Chlorinated Dioxins and Related Compounds, Bayreuth, Germany, September 10-14, 1990.

Harless, R.L.; Lewis, R.G. (1991) Evaluation of a sampling and analysis method for determination of polyhalogenated dibenzo-p-dioxins and dibenzofurans in ambient air. Presented at the IIth International Symposium on Chlorinated Dioxins and Related Compounds, Research Triangle Park, NC, September 23-27, 1991.

Huckins, J.N.; Schwartz, T.R.; Petty, J.D.; Smith, L.M. (1988) Determination, fate, and potential significance of PCBs in fish and sediment samples with emphasis on selected AHH-Inducing congeners. Chemosphere 17(10): 1995-2016.

Hunt, G.T.; Maisel, B. (1990) Atmospheric PCDDs/PCDFs in wintertime in a northeastern U.S. urban coastal environment. Chemosphere 20: 14551 462.

Jobb, B.; Uza, M.; Hunsinger, R.; Roberts, K.; Tosine, H.; Clement, R.; Bobbie, B.; LeBel, G.; Williams, D.; Lau, B. (1990) A survey of drinking water supplies in the province of Ontario for dioxins and furans. Chemosphere 20(10-12): 1553-1558.

Kitunen, V.H.; Salkinoja-Salonen, M.S. (1990) Soil contamination at abandoned sawmill areas. Chemosphere 20: 1671-1677.

Kjeller, L.O.; Kulp, S.E.; Bergetk, S.; Bostrom, M.; Bergquist, P.A.: Rappe, C. ; Jonsson, B.; de Wit, C.; Jansson, B.; Olsson, M. (1990) Levels and possible sources of PCDD/PCDF in sediment and pike samples from Swedish lakes and rivers. (Part one). Chemosphere 20 (10-12): 1489-1496.

Koistinen, J.; Passivirta, J.; Sarkka, J. (1990) Organic chlorine compounds in Lake Sediments. IV. Dioxins, furans and related chloroaromatic compounds. Chemosphere 21(12): 1371-1379.

Maisel, G.E.; Hunt, G.T. (1990) Background concentrations of PCDDs/PCDFs in ambient air. A comparison of toxic equivalency factor (TEF) models. Chemosphere 20: 771-778.

McKee, P.; Burt, A.; McCurvin, D.; Hollinger, D.; Clement, R.; Sutherland, D.; Neaves, W. (1990) Levels of dioxins, furans and other organic contaminants in harbour sediments near a wood preserving plant using pentachlorophenol and creosote. Chemosphere 20(10-12): 1679-1685.

Meyer, C.; O'Keefe, D.; Hilker, D.; Rafferty, L.; Wilson, L.; Connor, S.; Aldous, K.; Markussen, K.; Slade, K. (1989) A survey of twenty community water systems in New York State for PCDDs and PCDFs. Chemosphere 19(1-6): 21-26.

Naf, C; Broman, D.; Ishaq, R.; Zebuhr, Y. (1990) PCDDs and PCDFs in water, sludge and air samples from various levels in a waste water treatment plant with respect to composition changes and total flux. Chemosphere 20: 1503-1510.

Nestrick, T.J.; Lamparski, L.L.; Frawley, N.N.; Hummel, R.A.; Kocher, C.W.; Mahle, N.H.; McCoy, J.W.; Miller, D.L.; Peters, T.L.; Pillepich, J.L.; Smith, W.E.; Tobey, S.W. (1986) Perspectives of a large scale environmental survey for chlorinated dioxins: overview and soil data. Chemosphere 15: 1453-1460.

Niimi, A.J.; Oliver, B.G. (1989a) Assessment of relative toxicity of chlorinated dibenzo-p-dioxins, dibenzofurans, and biphenyls in Lake Ontario salmonids to mammalian systems using toxic equivalent factors (TEF). Chemosphere 18(7-8): 1413-1423.

Niimi, A.J.; Oliver, B.G. (1989b) Distribution of polychlorinated biphenyl congeners and other halocarbons in whole fish and muscle among Lake Ontario salmonids. Environ. Sci. Technol. 23: 83-88.

Norwood, C.B.; Hackett, M.; Powell, R.J.; Butterworth, B.C.; Williamson, K.J.; Naumann, S.M. (1989) Polychlorinated dibenzo-p-dioxins and dibenzofurans in selected estuarine sediments. Chemosphere 18(1-6): 553-560.

Oehme, M.; Mane, S.; Brevik, E.M.; Knutzen, J. (1989) Determination of polychlorinated dibenzofuran (PCDF) and dibenzo-p-dioxin (PCDD) levels and isomer patterns in fish, crustacea, mussel and sediment samples from a fjord region polluted by Mg-production. Fresenius Z. Anal. Chem. 335: 987-997.

Oliver, B.G.; Nilmi, A.J. (1988) Trophodynamic analysis of polychlorinated biphenyl congeners and other chlorinated hydrocarbons in the Lake Ontario ecosystem. Environ. Sci. Technol. 22: 388-397.

Pearson, R.G.; McLaughlin, D.L.; McIlveen, W.D. (1990) Concentrations of PCDD and PCDF in Ontario soils from the vicinity of refuse and sewage sludge incinerators and remote rural and urban locations. Chemosphere 20: 1543-1548.

Petty, J.D.; Smith, C.M.; Bergquist, P.A.; Johnson, J.L.; Stalling, D.L.; Rappe, C. (1982) Chlorinated dioxins, dibenzofurans total environment, [Proc. Symp.], meeting date 1982, 203-7. Edited by Choudhary, Keith, Rappe, and Butterworth. Boston, Massachusetts.

Rappe, C. (1987) Global distribution of polychlorinated dioxins and dibenzofurans, in Exner, J.H., ed., Solving Hazardous Waste Problems Learning from Dioxins. ACS Symposium Series 338. developed from a symposium sponsored by the Division of Environmental Chemistry at the 191st meeting the American Chemical Society, New York, New York, April 13-18, 1986.

Rappe, C. (1991) Sources of human exposure to PCDDs and PCDFs. In: Biological Basis for Risk Assessment of Dioxins and Related Compounds. Banbury Report #35. Edited by M. Gallo, R. Scheuplein and K. Van der Heijden. Cold Spring Harbor Laboratory Press. Plainview, NY.

Rappe, C.; Nygren M.; Lindstrom G. (1987) Polychlorinated dibenzofurans and dibenzo-p-dioxins and other chlorinated contaminants in cow milk from various locations in Switzerland. Environ. Sci. Technol. 21 (10): 964-970.

Rappe, C.; Kjeller, L.O.; Andersson, R. (1989b) Analysis of PCDDs and PCDFs in sludge and water samples. Chemosphere 19(1-6): 13-20.

Rappe, C.; Kjeller, L.O. (1987) PCDDs and PCDFs in environmental samples, air, particulates, sediments, and soil. Chemosphere 16(8/9): 1775-1780.

Rappe, C.; Bergqvist, P.A.; Kjeller, L.A. (1989a) Levels, trends and patterns of PCDDs and PCDFs in Scandinavian environmental samples. Chemosphere 18(1-6): 651-658.

Rappe, C.; Bergquist, P.A.; Kjeller, L.O.; Swanson, S.; Belton, T.; Ruppel, B.; Lockwood, K.; Kahn, P.C. (1991) Levels and patterns of PCDD and PCDF contamination in fish, crabs, and lobsters from Newark Bay and the New York Bight. Chemosphere 22(3-4): 239-266.

Rappe, C.; Bergquist, P.A.; Hansson, M.; Kjeller, L.O.; Lindstöm, G.; Marklund, S.; Nygren, M. (1984) Chemistry and analysis of polychlorinated dioxins and dibenzofurans in biological samples. In: Banbury Report 18: Biological Mechanisms of Dioxin Action. Cold Spring Harbor Laboratory. pp. 17-25.

Reed, L.W.; Hunt, G.T.; Maisel, B.E.; Hoyt, M.; Keefe, D.; Hackney, P. (1990) Baseline assessment of PCDDs/PCDFs in the vicinity of the Elk River, Minnesota generating station. Chemosphere 21(1-2): 159-171.

Ryan, J.J.; Lizotte, R.; Sakuma, T.; Mori, B. (1985) Chlorinated dibenzo-p-dioxins, chlorinated dibenzofurans, and pentachlorophenol in Canadian chicken and pork samples. J. Agric. Food Chem. 33: 1021-1026.

Schecter, A. (1991) Dioxins and related compounds in humans and in the environment. In: Biological Basis for Risk Assessment of Dioxins and Related Compounds. Banbury Report #35. Edited by M. Gallo, R. Scheuplein and K. Van der Heijden. Cold Spring Harbor Laboratory Press. Plainview, NY.

Schecter et al. (1990) Levels of chlorinated dioxins, dibenzofurans and other chlorinated xenobiotics in food from the Soviet Union and the South of Vietnam. Chemosphere 20 (7-9): 799-806.

Sherman, R.K.; Clement, R.E.; Tashiro, C. (1990) The distribution of polychlorinated dibenzo-p-dioxins and dibenzofurans in Jackfish Bay, Lake Superior, in relation to a Kraft Pulp Mill effluent. Chemosphere 20(10-12): 1641-1648.

Sievers, S.; Friesel, P. (1989) Soil contamination patterns of chlorinated organic compounds: looking for the source. Chemosphere 19(1-6): 691-698.

Smith, R.M.; O'Keefe, P.W.; Hilker, D.R.; Aldous, K.M.; Mo, S.H.; Stelle, R.M. (1989) Ambient air and incinerator testing for chlorinated dibenzofurans and dioxins by low resolution mass spectrometry. Chemosphere 18: 585-592.

Smith, L.M.; Schwartz, T.R.; Feltz, K. (1990) Determination and occurence of AHH-Active polychlorinated biphenyls, 2,3,7,8-tetrachloro-p-dioxin and 2,3,7,8-tetrachlorodibenzofuran in Lake Michigan sediment and biota. The question of their relative toxicological significance. Chemosphere 21(9): 1063-1085.

Sonzongni, W.; Maack, L.; Gibson, T.; Lawrence, J. (1991) Toxic polychlorinated biphenyl congeners in Sheboygan River (USA) sediments. Bull. Environ. Contam. Toxicol. 47: 398-405.

Stalling, D.L.; Smith, L.M.; Petty, JU.D.; Hogan, J.W.; Johnson, J.L.; Rappe, C.; Buser, H.R. (1983) Residues of polychlorinated dibenzo-p-dioxins and dibenzofurans in laurentian Great Lakes fish. In: Tucher, R.E.; Young, A.L.; Gray, A.P.; eds. Human and environmental risks of chlorinated dioxins and related compounds. Arlington, Virginia: Plenum Press. pp. 221-240.

Startin, J.R.; Rose, M.; Wright, C.; Parker, I.; Gilbert, J. (1990) Surveillance of British foods for PCDDs and PCDFs. Chemosphere 20 (7-9): 793-798.

Stenhouse, I.A.; Badsha, K.S. (1990) PCB, PCDD, and PCDF concentrations in soils from the Kirk Sandall/Edenthorpe/Barnby Dun area. Chemosphere 21: 563-573.

Travis, C.C.; Hattemer-Frey, H.A. (1991) Human exposure to dioxin. Sci. Total Environ. 104: 97-127.

U.S. EPA. (1992) EPA National bioaccumulation study. Personal communication between John Schaum, U.S. EPA Office of Health and Environmental Assessment and Richard Healy, U.S. EPA Office of Water. July 22, 1992.

U.S. EPA. (1990) Risk assessment for 2378-TCDD and 2,3,7,8-TCDF contaminated receiving waters from U.S. chlorine-bleaching pulp and paper mills. U.S. EPA Office of Water Regulations and Standards. PB #90272873. August 1990.

U.S. EPA. (1987) National dioxin study. Office of Solid Waste and Emergency Response. EPA/530-SW-87-025. August 1987.

U.S. EPA. (1985) Soil screening survey at four midwestern sites. Region V. Enviornmental Services Division, Eastern District Office, Westlake, Ohio, EPA-905/4-805-005, June 1985.

Van den berg, M. (1987) Presence of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans in fish-eating birds and fish from the Netherlands. Arch. Environ. Contam. Toxicol. 16: 149-158.

4. ESTIMATING EXPOSURES AND RISKS

4.1. INTRODUCTION

In this chapter, the framework for assessing exposure and risk to 2,3,7,8-TCDD and related dioxin-like compounds will be described. Section 4.2 introduces the exposure equation and its key terms. Section 4.3 describes how risk is estimated given estimates of exposure. It also discusses the use of toxicity equivalency factors. Section 4.4 provides the overview of the procedures used in this document, and provides a roadmap for finding pertinent information in other chapters of the document.

4.2. EXPOSURE EQUATION

This document describes procedures for conducting exposure assessments to estimate either potential or internal dose. A potential dose is defined as a daily amount of contaminant inhaled, ingested, or otherwise coming in contact with outer surfaces of the body, averaged over an individual's body weight and lifetime. An internal dose is defined as the amount of the potential dose which is absorbed into the body (EPA, 1991). Section 4.3 below discusses the relevancy of this distinction for dioxin-like compounds.

The general equation used to estimate potential dose normalized over bodyweight and lifetime is as follows:

$$\text{Lifetime Average Daily Dose (LADD)} = (\text{exposure media concentration} \times \text{contact rate} \times \text{contact fraction} \times \text{exposure duration}) / (\text{body weight} \times \text{lifetime}) \quad (4\text{-}1)$$

This procedure is used to estimate dose in the form needed to assess cancer risks. Each of the terms in this exposure equation is discussed briefly below:

- **Exposure media concentrations:** These include the concentrations in soil for dermal contact and soil ingestion exposure pathways, in vapor and particulate phase in air for inhalation exposure pathways, in water for a water ingestion pathway, and in food products such as fish, fruits and vegetables, and beef and milk, for food ingestion pathways. This document provides procedures for estimating exposures associated with all these pathways.

- **Contact rate:** These include the ingestion rates, inhalation rates, and soil contact rates for the exposure pathways. These quantities are generally the total amount of food ingested, air inhaled, etc. Only a portion of this material may be contaminated. The next term, the contact fraction, which is 1.0 or less, reduces the total contact rate to the rate specific to the contaminated media.
- **Contact fraction:** As noted, this term describes the distribution of total contact between contaminated and uncontaminated media. For example, a contact fraction of 0.8 for inhalation means that 80% of the air inhaled over the exposure period contains dioxin-like compounds in vapor form or sorbed to air-borne particulates.
- **Exposure duration:** This is the overall time period of exposure. Values such as 9 years and 20 years are used in the example scenarios described in Chapter 9. The value of 9 years corresponds to the average time spent at one residence (EPA, 1989), and was used as an exposure duration for a non-farming family living in a rural setting for the example scenarios in Chapter 9. Twenty years was used as the exposure duration for farming families in a rural setting. Another exposure duration demonstrated in Chapter 10 is one associated with a childhood pattern of soil ingestion. The exposure duration in this case is 5 years.
- **Body weight:** For all the pathways, the human adult body weight of 70 kg is assumed. This value represents the United States population average. The body weight for child soil ingestion is 17 kg (EPA, 1989).
- **Lifetime:** Following traditional assumptions, the average adult lifetime assumed throughout this document is 70 years. Even though actuarial data indicate that the United States average lifetime now exceeds 70 years, this convention is used to be consistent with other Agency assessments of exposure and risk.

4.3. RISK EQUATION

Although estimation of risk is technically beyond the scope of an exposure assessment, the exposure assessor needs some background understanding in this area.

The primary source of information on the health risks of the dioxin-related compounds is the Health Reassessment that EPA is publishing concurrently with this document. However some general considerations for using exposure estimates in support of cancer risk assessments are summarized here. The usual procedure used to calculate an upper-limit incremental cancer risk is as follows:

$$R = 1 - e^{-q_1^* d} \approx q_1^* d \qquad (4\text{-}2)$$

when $q_1^* d < 10^{-3}$ and where q_1^* is the 95% upper confidence limit of the linearized cancer slope factor of the dose-response function (expressed in inverse units of the dose quantity, typically kg-day/mg) and d is the dose (typically in mg/kg-day). The dose is generally equal to the potential dose given above in Equation (4-1). The slope factor, q_1^*, for 2,3,7,8-TCDD has been previously estimated by EPA as 0.156 kg-d/ng. The derivation of this factor is described in EPA (1984) and further background is provided in EPA (1981). The Agency is currently reevaluating this slope factor and the reader should consult the companion Health Reassessment for the current policy.

EPA derived the 1984 slope factor for 2,3,7,8-TCDD from animal feeding studies on the basis of potential (i.e., administered) dose. Thus, for purposes of consistency, when using this slope factor to estimate risk to humans, the exposure assessor should provide the dose estimate as a potential dose. This point raises issues specific to the various pathways.

The absorption which occurred during the animal experiments EPA used to derive the 1984 slope factor for 2,3,7,8-TCDD was estimated to be 55% (Farland, 1987). The review of literature on bioavailability in Chapter 8 of this assessment indicates that the gut absorption of 2,3,7,8-TCDD in humans when the vehicle is soil is 20-40% of potential dose. Fries and Marrow (1975) found that 50-60% of the 2,3,7,8-TCDD was absorbed by rats from feed. Rose, et al. (1976) estimated that 86% of 2,3,7,8-TCDD in a mixture of acetone and corn oil fed by gavage to rats was absorbed. EPA (1984), using animal data and information on fate of particles in the respiratory system, estimated that the fraction of 2,3,7,8-TCDD absorbed into the body ranges from 0.25 to 0.29. What this discussion indicates is that the absorption for human ingestion and inhalation pathways might range

from 20-80% of potential dose, which compares to 55% found in the laboratory experiments. If no adjustment were made to potential dose estimates, then human risk estimates might be overestimated (when absorption is in the 20% range) or underestimated (in the 80% range). This discrepancy is not felt to be large enough or certain enough to warrant an absorption correction factor.

The rate of absorption of vapor-phase 2,3,7,8-TCDD into the lungs has not been studied, but it seems reasonable to assume that the absorption in the vapor phase should exceed that of absorption from bound 2,3,7,8-TCDD on particulates, probably above 50%. There is also an unknown uncertainty introduced when assuming that the q^{1*} developed from a feeding study can be used for an inhalation pathway. Thus, it is unclear what adjustment is needed to account for differences between the feeding study and a vapor-phase inhalation exposure. Accordingly, it is recommended that assessors not attempt any such adjustments at this point, but fully acknowledge the uncertainty.

Estimating risks associated with dermal exposure introduces several issues to consider. First, use of an oral dose-response function may not be applicable to the dermal route. Second, dermal absorption of dioxin-like compounds from soil appears to be much lower than that which occurred in the dose-response feeding studies. EPA (1992) indicates that 0.1 - 3% of 2,3,7,8-TCDD may be dermally absorbed from soil, which is significantly less than the 55% absorption found in the laboratory feeding experiments. It is assumed for this assessment that an absorption fraction of 0.03 (3%) applies to 2,3,7,8-TCDD as well as the other dioxin-like compounds. Specifically, this assessment estimates the total amount of compounds applied to skin and then reduces it by 97% to estimate the absorbed dose. This is the only pathway in which an absorption fraction is used to adjust a dose (see Chapter 7 for more information on the procedures to estimate exposure from all pathways). Because of this adjustment, an additional adjustment to the risk equation, Equation (4-2) above, is needed when estimating risk from dermal exposure in a manner consistent with other exposure pathways: the slope factor should be multiplied by (100%) / (55%), or about 2, to convert it to an absorbed basis. Finally, the assessor should acknowledge that considerable uncertainty is introduced by applying an oral based dose-response function to dermal exposure.

Another set of issues facing the exposure/risk assessor is how to estimate exposure to mixtures of dioxin-like compounds with differing slope factors. EPA (1989)

has proposed a procedure to address this issue, which is to adjust the risk estimate using a "toxicity equivalency factor", commonly referred to as TEF. The TEF for a congener of interest is the cancer potency of that congener divided by the cancer potency of 2,3,7,8-TCDD. As shown in Table 2-1, the TEF for 2,3,7,8-TCDD is 1 and all other dioxin-like compounds have TEFs less than 1. The combined risk resulting from exposure to a mixture of dioxin-like compounds can be computed using the TEFs and assuming that the risks are additive:

$$Risk = 1 - e^{-q_1^* \sum (TEF_i \, d_i)} \tag{4-3}$$

where q_1^* is the cancer slope factor for 2,3,7,8-TCDD (kg-day/mg), TEF_i is the toxicity equivalency factor for dioxin-like compound i, d_i is the potential dose for dioxin-like compound i (mg/kg-day), and n is the total number of dioxin-like compounds to which an individual is exposed.

4.4. PROCEDURE FOR ESTIMATING EXPOSURE

Section 4.2 described the exposure equation as it applies to dioxin and dioxin-like compounds. Before making exposure estimates, the assessor needs to gain a more complete understanding of the exposure setting and be prepared to estimate exposure media concentrations. The purpose of this section is to provide guidance for the procedures followed in this assessment to define such settings and estimate exposure media concentrations. The approach used here is termed the exposure scenario approach. Brief descriptions of the steps and associated document chapters are presented below and summarized in Figure 4-1.

Step 1. Identify Source

Two principal sources are addressed in this document. The first, identified as "soil", is called a source in that the starting point of the assessment is soil contamination. Of course, the ultimate source for soil contamination is some unidentified cause for the soil to become contaminated. For exposure and risk assessment purposes, the cause for contamination is not relevant except to assume that the cause is not ongoing and that the impact of the "initial" levels is what is being evaluated. The soil source is further

STEPS	DOCUMENT CHAPTERS
Step 1. Identify Sources	
A. Soil On-site, off-site	Chapter 3
B. Incinerators Stack emissions, fly ash disposal	Chapter 6
Step 2. Estimate Release Rates	
A. Volatilization, erosion, etc.	Chapter 5
B. Stack emission rates	Chapter 6
Step 3. Estimate Exposure Point Concentrations	
A. Food chain, dilution, etc.	Chapter 5
B. Dispersion, deposition, etc.	Chapter 6
Step 4. Characterize Exposed Individuals and Exposure Patterns	
A. Contact rates, exposure durations	Chapter 7
Step 5. Put It Together in Terms of Exposure Scenarios	
A. Scenario concept expanded	Chapter 7
B. Demonstration with scenarios	Chapter 9
Step 6. Estimate Exposure and Risk	
A. Equations and background	Chapter 4
B. Results for example scenarios	Chapter 9
Step 7. Assess Uncertainty	Chapter 10

Figure 4-1. Roadmap for assessing exposure and risk to dioxin and dioxin-like compounds.

characterized as off-site or on-site. Off-site implies that the soil contamination is located some distance from the site of exposure. The site of exposure could be a residence or farm, and the site of contamination could be a landfill, for example. On-site implies that the soil contamination is on the site of exposure. The second principal source is "incinerators." Unlike the soil source, the contamination is assumed to be on-going. The two subcategories under this principal source are stack emissions and fly ash. Stack emissions in particulate form are assumed to deposit onto the soil of the site of exposure, and emissions in vapor form result in air-borne concentrations for inhalation exposures. For fly ash, the exposed individuals are those who are assumed to reside near a landfill where the ash is being disposed of in an ongoing process. The procedures for calculating exposures and risks for those individuals are similar to the off-site soil source category. It is noted that individuals working at the incinerator or involved in the disposal of fly ash are also exposed. The procedures in this document only apply to residents who are not associated with the incinerator or its operations.

Step 2. Estimate Release Rates

Estimating the release of contaminants from the initial source is the first step towards estimating the concentration in the exposure media. Releases from soil contamination include volatilization, erosion, transfer to biota, and so on. Chapter 5 on estimating exposure media concentrations describes fate and transport modeling procedures for estimating soil releases. Emissions from incinerator stacks are the release rate pertinent to that source. Releases from off-site soil contamination areas include suspension of particulates via wind erosion, and soil erosion. Additional releases from a fly ash disposal areas include fugitive dust emissions from landfill operations. Background on incinerators including assumptions made for the methodology demonstration in Chapter 9 are provided in Chapter 6.

Step 3. Estimate Exposure Point Concentrations

Contaminants released from soils or emitted from stacks move through the environment to points where human exposure may occur. Contaminated soil that is near but not at the site of exposure is assumed to slowly erode and contaminate the exposure site soil, but to a level lower than the initial source strength. The only time when the

source concentrations equal the exposure concentrations is for the soil pathways, soil ingestion and dermal contact, when the soil contamination is on-site. Chapter 6 describes the use of the Industrial Source Complex (ISC) Model used to estimate dispersion of stack plumes to arrive at air-borne concentrations at the site of exposure as well as deposition rates of stack emitted particulates onto exposure site soil. Chapter 5 describes how soil concentrations are estimated given particulate deposition rates, and also how release rates from soil initially contaminated translate to exposure point concentrations.

Step 4. Characterize Exposed Individuals and Exposure Patterns

The patterns of exposure are described in Chapter 7. Exposed individuals in the scenarios of this assessment are individuals who are exposed in their home environments. They are adult residents who also recreationally fish, have a home garden, farm, and are children ages 2-6 for the soil ingestion pathway. Their exposure is characterized in terms of exposure pathways, which have generally been alluded to in discussions above - inhalation, ingestion, and soil dermal contact. Each pathway has the set of parameters including contact rates, contact fractions, body weights, and lifetime. These parameters were defined earlier in Section 4.4.

Step 5. Put It Together in Terms of Exposure Scenarios

A common framework for assessing exposure is with the use of "settings" and "scenarios." Settings are the physical aspects of an exposure area and the scenario characterizes the behavior of the population in the setting and determines the severity of the exposure. A wide range of exposures are possible depending on behavior pattern assumptions. An exposure scenario framework offers the opportunity to vary any number of assumptions and parameters to demonstrate the impact of changes to exposure and risk estimates. Exposure estimates for six example scenarios are demonstrated in Chapter 9.

Step 6. Estimate Exposure and Risk

Section 4.2 described the basic equation that estimates exposure for every assumed pathway in an exposure scenario. Chapter 9 demonstrates the methodology on six example scenarios, which includes the generation of exposure estimates for ten different exposure pathways and three different dioxin-like compounds per scenario.

Step 7. Assess Uncertainty

Chapter 10 provides a comprehensive discussion on possible sources of uncertainty associated with this methodology. These uncertainties should be considered when applying this procedures to a particular site.

REFERENCES FOR CHAPTER 4

Farland, W.H. (1987) Memorandum titled, "Absorption fraction when calculating upper-limit risks due to dioxin exposure", dated September 2, 1987, to Michael Callahan, Exposure Assessment Group, Washington, DC. from William Farland, U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, D.C.

Fries, G.F.; Marrow, G.S. (1975) Retention and excretion of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) by rats. J. Agric. Food Chem. 23: 265-269.

Rose, J.Q.; Ramsey, J.C.; Wentzler, T.H. (1976) The fate of 2,3,7,8-tertrachloridbenzo-p-dioxin following single and repeated oral doses to the rat. Toxicol. Appl. Pharmacol. 36: 209-226.

U.S. Environmental Protection Agency. (1981) Risk assessment on (2,4,5-trichlorophenoxy)acetic acid (2,4,5-T), (2,4,5-trichlorophenoxy)propionic acid (silvex), and the 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Office of Health and Environmental Assessment, Washington, DC; EPA-600/6-81-003. NTIS PB81-234825.

U.S. Environmental Protection Agency. (1984) Risk analysis of TCDD contaminated soil. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, DC; EPA/600/8-84-031.

U.S. Environmental Protection Agency. (1989) Interim procedures for estimating risks associated with exposures to mixtures of chlorinated dibenzo-p-dioxins and -dibenzofurans (CDDs and CDFs) and 1989 update. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC; EPA/625/3-89/016.

U.S. Environmental Protection Agency. (1991) Guidelines for exposure assessment. SAB Draft Final. August 8, 1991. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, DC; OHEA-E-451.

U.S. Environmental Protection Agency. (1992) Health reassessment of dioxin-like compounds. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, DC.

U.S. Environmental Protection Agency. (1992) Dermal exposure assessment: principles and applications. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, DC; EPA\600\8-91\011B.

5. ESTIMATING EXPOSURE MEDIA CONCENTRATIONS

5.1. INTRODUCTION

The purpose of this chapter is to describe the algorithms used to determine exposure media concentrations of the dioxin-like compounds. Discussion of the algorithms are structured around four "source categories." These categories roughly translate to beginning points, or origins, of contamination. The source categories are also the basis for the example scenarios described in Chapter 9.

Section 5.3 describes the algorithms used for the first source category, on-site soil, where the contaminants occur in surface soils, and this contamination source and subsequent exposure occur at the same site. The second source category, described in Section 5.4, is termed off-site soil. The contaminated soil is remote from the exposure, such as in a landfill impacting a nearby residence. Section 5.5 describes algorithms to determine exposure media concentrations resulting from incinerator stack emissions, the third source category. The Industrial Source Complex (ISC) Model (EPA, 1986) was used in the example of this source category to generate two key quantities: air-borne contaminant concentrations at a site of exposure, and particulate deposition rates. Chapter 6 discusses incineration as a source of dioxin-like compounds and describes this ISC application. The ISC model is the only complex fate and transport model used in this document. This exception was made because the ISC model is widely available and relatively easy to use. Section 5.5 describes how ISC-modeled deposition rates translate to soil, vegetative, and water concentrations. Section 5.6 concludes the chapter with a discussion of algorithms specific to the fourth source category, incinerator ash disposal in a landfill. The contaminated ash is spread onto the surface of an active landfill, so the fate and transport algorithms associated with this category are assumed to be the same as those for the second category, off-site soil. Section 5.6 shows how landfill size (or the size of the landfill portion dedicated to ash disposal) can be estimated given an amount of ash disposed. Section 5.6 also describes algorithms to estimate fugitive ash emissions that result from ash handling operations.

Algorithms are presented which estimate exposure media concentrations for: 1) surface water impacts: sediment and dissolved phase concentrations, 2) air including the vapor phase and in particulate form, and 3) biota including beef, milk, fruit and vegetables,

and fish. The soil at the site of exposure is the exposure media for the soil ingestion and dermal exposure pathways. Table 5-1 summarizes the solution algorithms used to estimate exposure media concentrations for each of the four source categories.

The purpose of this chapter is twofold. First, it provides the fate and transport algorithms for estimating these exposure media concentrations. Second, it provides information about all the model parameters and justification for the default values selected for the demonstration of methodologies in Chapter 9. The four source categories outlined in this chapter are demonstrated in Chapter 9 in what are termed exposure scenarios. An exposure scenario includes a physical description of an area where exposure occurs, including features such as the proximity to the soil contamination or incinerator, the presence of nearby water bodies used for fishing or drinking, and so on. Exposure scenario definition also includes assumptions on the behavior of individuals and the pathways of exposures. One set of scenarios was crafted to represent "central" exposures, and the other set, "high end" exposures. The central exposure scenario is modeled as a residence in a rural setting, and the high end a farm in the same rural setting. One way a residence is central in contrast to a farm is that residents are assumed to reside a shorter time in the residence, 9 years, in contrast to the family residing at the farm, 20 years. For purposes of exposure estimation, residence time is equivalent to exposure duration.

The introductory portion of Chapter 9 provides a complete description of the example exposure scenarios, and other chapters in this assessment discuss issues on exposure estimation. This introductory portion is offered only so that references below to the example scenarios in Chapter 9 are understood.

It has been noted in earlier chapters that the assessment procedures described in this document are site specific. This is perhaps most true for the algorithms estimating exposure media concentrations. The values selected to demonstrate the methodologies are not being promoted as general default values. Brief discussions and references are provided for all model parameters so that users of the methodology can make alternate parameter selections to better suit their purposes.

Table 5-1. Summary and fate and transport solution algorithms for estimating exposure media concentrations for the four source categories.

Source category		Fate and transport solution algorithms	Exposure media
On-site soil	a)	Suspended and bottom sediment concentrations estimated as a function of soil erosion from contaminated site and from watershed; water concentrations estimated by partitioning from suspended sediment.	Surface water
	b)	Surface water bottom sediment concentrations (as estimated above) multiplied by a Biota to Sediment Accumulation Factor, or BSAF	Fish
	c)	No dissipation of initial soil concentrations	Soil
	d)	Volatilization assuming vapor phase diffusion; near-field dispersion estimates air concentration	Air: vapor
	e)	Wind erosion estimates emissions of contaminates associated with suspended particulates; same near-field dispersion model as used for vapor-phase diffusion	Air: particulate
	f)	Below ground vegetation assume soil to plant transfer; above ground vegetation assumes air-to-plant vapor phase transfer and particulate deposition	Fruits and vegetables
	g)	Biotransfer to cattle meat and milk as a linear function of modeled concentrations in total dry matter intake including pasture grass, fodder, and soil ingested by cattle; different assumptions on proportion of noted dry matter intakes made for cattle raised for beef vs. raised for dairy; pasture grass and fodder estimated using above ground vegetation algorithms as noted in "f)" above	Meat and milk

(continued on the following page)

Table 5-1. (continued)

Source category		Fate and transport solution algorithms	Exposure media
Off-site soil	a)	Same algorithm as on-site source category	Surface water
	b)	Same algorithm as on-site source category	Fish
	c)	Off-site soil concentrations remain constant; erosion onto exposure site (residence or farm, e.g.) delivers soil-borne contaminants; steady-state contaminant concentrations on exposure site soil estimated using Universal Soil Loss Equation for erosion estimates and assuming further dissipation of contaminants; exposure site soil concentrations further reduced for tillage-related activities of vegetable gardening and crop growing.	Soil
	d)	Volatilization assuming vapor phase diffusion as in on-site source category; far-field dispersion model replaces near-field dispersion model of on-site source category to estimate exposure site vapor phase concentrations	Air: vapor
	e)	Particulates suspended via wind erosion as in on-site source category; far-field dispersion model estimates exposure site air-borne particulate phase concentrations	Air: particulate
	f)	Same algorithm as on-site soil source category using modeled media concentration; one difference is that exposure site soil concentrations further diluted by tillage for below ground vegetables	Fruits and vegetables
	g)	Same algorithm as on-site source category; air concentrations for vapor phase transfers and particulates for deposition (for grass and fodder) now originate at off-site contamination	Meat and milk

(continued on the following page)

Table 5-1. (continued)

Source category		Fate and transport solution algorithms	Exposure media
Stack emission	a)	Watershed average steady state soil concentration estimated using ISC modeled deposition rates and residue dissipation as in algorithm c) below; suspended and bottom sediment concentrations assumed equal to watershed soil concentrations; surface water concentrations then estimated assuming partitioning as in other source categories.	Surface water
	b)	Same algorithm as on-site source category	Fish
	c)	ISC model estimates particulate deposition rates; steady-state concentrations in soil estimated assuming further dissipation of deposited particles	Soil
	d)	ISC model estimates exposure site vapor-phase concentrations	Air: vapor
	e)	No suspended particulates assumed to result from stack emissions; wind erosion assumed not to occur as in other source categories	Not modeled
	f)	Same basic algorithm as in on-site source category using: ISC modeled deposition rates of particle-bound contaminants, ISC modeled vapor-phase concentrations, and soil concentrations diluted by tillage	Fruits and vegetables
	g)	Same algorithm as for on-site soil with pasture grass and fodder concentrations now a function of ISC modeled contaminant concentrations	Meat and milk
Ash disposal in a landfill	a)	Same algorithm as on-site source category	Surface water
	b)	Same algorithm as on-site source category	Fish

(continued on the following page)

Table 5-1. (continued)

Source category		Fate and transport solution algorithms	Exposure media
Ash Landfill (continued)	c)	Landfill soil concentrations assumed equal to 1/2 contaminant concentrations in disposed ash; total size of landfill, 27 hectares, estimated assuming 30 years of ash disposal; "active" portion over a duration of exposure assumed to 1/4 of total size of 109 ha required for 30-year lifetime; exposure site soil concentrations estimated using erosion algorithm as in off-site source category with similar dilution for the tillage activity of vegetable gardening	Soil
	d)	Same algorithm as off-site source category	Air: vapor
	e)	Emissions of suspended particulate include: wind erosion as in on-site and off-site source categories; vehicular resuspension of contaminated roadway dust; emissions off trucks delivering ash; unloading; and spreading and compacting; far-field dispersion models exposure site air-borne particulate phase concentrations given emission fluxes	Air: particulate
	f)	Same algorithm as off-site source category	Fruits and vegetables
	g)	Same algorithm as off-site source category	Meat and milk

5.2. CONSIDERATIONS FOR SOLUTION ALGORITHMS

Algorithms selected for estimation of exposure media concentrations are best described as screening level algorithms. They are approximations of complex environmental processes. Simple assumptions are often made in order to arrive at the desired result, which is long-term average exposure media concentrations. It cannot be overemphasized that measured concentrations of dioxin-like compounds in exposure media provide more appropriate information for estimating exposure than the estimations made with the algorithms of this assessment, or even estimations that could be made with the most sophisticated modeling approaches. In Chapter 10 on Uncertainty, the exposure media concentrations predicted were compared with measured field concentrations reported in the literature. Although this was not a validation exercise, the comparisons generally showed that predictions were similar to measurements.

One important assumption made for all but one of the algorithms is that the source strength remains constant throughout the period of exposure; no dissipation via environmental degradation or otherwise is assumed. The tendency for the dioxin-like compounds to resist degradation and tightly sorb to soils and sediments supports the part of the assumption concerning degradation. However, initial concentrations in soil or sediment will dissipate over time due to the processes which move the chemical in the environment, such as runoff, volatilization, or leaching from sediments into overlying surface waters. The one algorithm where this assumption was not made is the algorithm estimating volatilization flux (and subsequently vapor phase concentrations, inhalation exposures and risks). Contaminants available for volatilization are assumed to originate from deeper soil depths over time reflecting the dissipation, although not necessarily degradation, of surface residues. Because residues originate from deeper in the soil, volatilization flux decreases over time.

Assumptions about the most likely or reasonable route between the beginning point of contamination, on-site or off-site soil for example, and the exposure media endpoint needed to be made for some of the algorithms. Often it was possible that exposure media could become contaminated by more than one route. For example, volatilization was modeled to occur for the on-site soil source category, resulting in vapor-phase air concentrations to which an individual living at the site was then exposed. For the off-site soil source category, vapor phase concentrations are also estimated for an exposed

individual who is residing at some distance from the contaminated site. The soil at this individual's residence is assumed to become contaminated over time due to erosion from the nearby contaminated site. Two approaches are then available to estimate vapor-phase contaminant concentrations. One is to assume that volatilization occurs at the contamination site, and vapors are then transported to the exposure site. Another is to assume that the exposure site soil (and soil between the contaminated site and the exposure site, for that matter), now assumed to contain a thin layer of contaminated soil, emits contaminated vapors to which the individual is exposed. Should emissions to which individuals are exposed be assumed to originate at the contaminated site or at the exposure site, or is it most appropriate to assume emissions occur from the contaminated site and all nearby impacted soil? Emission of vapors from the off-site contaminated soil, followed by air transport to the exposed individual, was the strategy selected for this assessment. This assumption was made because it may be more plausible to assume that the deeper and more uniform soil contamination located off-site is more likely to be an on-going source of vapor emissions in comparison to the thin layer of contaminated eroded soil which does deposit on the site of exposure. This and other alternate solution strategies are discussed in Chapter 10 on uncertainty.

Chapter 10 also provides other technical discussions that are not included in this chapter, as a way of evaluating some of the algorithms described here. Where possible, model predictions are compared with observed data. For example, algorithms to estimate vegetative concentrations of contaminants are presented in this chapter. Chapter 10 evaluates the algorithm by comparing results with literature studies which measured soil contamination and resulting plant contamination.

5.3. ALGORITHMS FOR THE "ON-SITE SOIL" SOURCE CATEGORY

As earlier noted, the contamination and exposure occur at the same site for this source category. The contamination is assumed to originate at the soil surface. As such, the soil itself is the exposure media for the dermal contact and soil ingestion pathways. Sections 5.3.1 through 5.3.4 describe the algorithms for estimating concentrations of the dioxin-like compounds in: bottom sediment, suspended solids, and in the dissolved phase in the water column of surface water bodies (5.3.1), in the air in the vapor phase (5.3.2) and particulate phase (5.3.3), and in biota including fish (5.3.4.1), home-grown vegetables

and fruit (5.3.4.2), and home-produced beef and milk (5.3.4.3).

5.3.1. Surface Water and Sediment Contamination

Contaminated soil can erode into and, over time, result in the contamination of sediments of nearby surface water bodies. If soil sources are expected to supply contaminants to the surface water body over time, then concentrations in suspended and bottom sediments can be estimated as a function of concentrations in eroding soil. For purposes of this assessment, it will be assumed that equilibrium is achieved between concentrations in the dissolved phase in the water column, concentrations in the sorbed phase in the water column, and concentrations in bottom sediments. Concentrations in bottom sediment are desired because fish concentrations are estimated as a function of bottom sediment concentrations (see Section 5.3.4.1). Concentrations in suspended solids are desired because they are used to estimate bottom sediment concentrations, and dissolved phase concentrations are needed for estimating drinking water exposures.

The principal assumption for this approach is that dioxin-like compounds enter the surface water body, either a standing water body such as a lake or a moving water body such as a river, only through surface erosion. Other key assumptions and simplifications include: 1) suspended solids are simply a reservoir into which dioxin-like compounds sorb; more complex approaches consider sorption onto more than one reservoir of suspended materials including suspended particulates and dissolved organic matter, 2) the sorption of dioxin-like compounds onto suspended solids and bottom sediments is principally a function of organic carbon, 3) the water body is completely mixed; results are described as average for the water, 4) contaminants are not subject to dissipation processes such as photolysis or volatilization, and 5) suspended solids concentrations are assumed to be constant; also, no deposition/suspension or other sediment transfer and transport processes enter into the solution.

Concentrations of dioxin-like compounds entering the water body via erosion are assumed to be a simple linear function of concentrations of dioxin-like compounds in surface soils from all or a portion of a watershed draining into the surface water body:

$$C_{swb} = \frac{(C_s \ SL_s \ A_s \ SD_s) + (C_w \ SL_w \ A_w \ SD_w)}{(SL_s \ A_s \ SD_s) + (SL_w \ A_w \ SD_w)} \qquad (5-1)$$

where:

C_{swb}	=	concentration on soil entering water body, mg/kg
C_s	=	contaminated site soil concentration of dioxin-like compound, mg/kg
SL_s	=	unit soil loss from contaminated site area, kg/ha-yr
A_s	=	area of contaminated site, ha
SD_s	=	sediment delivery ratio for soil eroding from contaminated site to water body, unitless
C_w	=	average concentration of dioxin-like compound in effective area of watershed not including contaminated site, mg/kg
SL_w	=	average unit soil loss for land area within watershed not including contaminated site, kg/ha-yr
A_w	=	effective area of watershed; the area contributing sediment which mixes with the sediment originating from As, ha
SD_w	=	sediment delivery ratio for watershed, unitless

The solution for estimating water column sorbed phase, water column dissolved phase, and bottom sediment concentrations then requires estimating total water body concentration, defined as the mass of contaminant in the water column divided by the volume of water. This is equal to:

$$C_{tot} = C_{swb} \ TSS \ 10^{-6} \qquad (5-2)$$

where:

C_{tot}	=	total water column concentration, sorbed + dissolved, mg/kg
C_{swb}	=	concentration on soil entering water body, mg/kg
TSS	=	total suspended sediment concentration, mg/L

10^{-6} = converts C_{swb} in mg/kg to mg/mg

Dissolved phase water column concentrations are then estimated using an approach developed by Mills, et al. (1985) and others. The equation is given as Equation (5-3a) below. The concentration on suspended solids can then be estimated given dissolved phase concentrations and a partition coefficient, as in Equation (5-3b). For estimating bottom sediment concentrations, it will be assumed that the carbon normalized concentrations on suspended solids equals the carbon normalized concentration on bottom sediments. This is given below as Equation (5-3c):

$$C_{wat} = \frac{C_{tot}}{1 + (Kd_{ssed} \; TSS \; 10^{-6})} \qquad (5\text{-}3a)$$

$$C_{ssed} = Kd_{ssed} \, C_{wat} \qquad (5\text{-}3b)$$

$$C_{sed} = C_{ssed} \frac{OC_{sed}}{OC_{ssed}} \qquad (5\text{-}3c)$$

where:

C_{wat}	=	dissolved-phase water concentration of contaminant, mg/L
C_{tot}	=	total water column concentration, sorbed + dissolved, mg/kg
Kd_{ssed}	=	soil-water partition coefficient for contaminant in suspended sediment, L/kg
TSS	=	total suspended solids in water column, mg/L
C_{ssed}	=	concentration on suspended sediments, mg/kg
C_{sed}	=	concentration on bottom sediments, mg/kg
OC_{sed}	=	fraction organic carbon on bottom sediments, unitless
OC_{ssed}	=	fraction organic carbon on suspended sediments, unitless
10^{-6}	=	converts mg/kg to mg/kg

Note that this solution is applicable to both a pond/lake or a stream/river setting. The differences in the two water systems can be expressed in the parameters, effective watershed area, Aw, organic carbon contents of suspended solids and bottom sediments, OC_{ssed} and OC_{sed}, and total suspended sediment, TSS. Guidance on each of these parameters is now given:

- **C_s and C_w:** These are concentrations of dioxin-like compounds in the contaminated site soil, C_s, and the average within the effective area of the watershed, C_w. The contaminated site concentrations drive the concentrations assumed for most exposures, and is a principal user input (for the on-site source category, the contaminated site is also the site of exposure). The simplest assumption for C_w is that it is 0.0. However, examination of soil data from around the world shows that, where researchers have measured concentration in what they described as "background" or "rural" settings, soil concentrations of PCDDs and PCDFs are in the non-detect to low ng/kg (ppt) range (see Section 9.5). Example Scenarios 1 and 2 in Chapter 9 demonstrate the on-site source category. For these example scenarios, C_w and C_s are both initialized at 10^{-6} mg/kg (1 ppt) for all example compounds representing low concentrations that might be possible for basin-wide areas.

- **SL_s and SL_w:** These are the unit soil loss, in kg/ha, from the exposure site and the average from the effective land area draining into the surface water body. In the simplest case, the unit losses can be considered equal, in which case the SL_s and SL_w terms drop out of Equation (5-1). In the most complicated solution, the effective drainage area can be broken up into "source areas", where each source area can be unique in terms of the erosion potential, concentration of contaminant, and so on. The total contribution equals the sum of contributions from each source area, as: $\Sigma C_j*SL_j*A_j*SD_j$ for the numerator in Equation (5-1) and $\Sigma SL_j*A_j*SD_j$ for the denominator, for j number of source areas not including the exposure site. For direct input into Equation (5-1), the terms Cj, SL_j, A_j, and SD_j, should be determined and C_w, SL_w, and SD_w should be estimated as weighted averages over all source areas, A_j. The effective drainage area, A_w, would be the sum of all source areas, A_j.

For the example scenarios in Chapter 9 demonstrating the on-site source

categories, SL_s and SL_w are assumed equal, and hence drop out of Equation (5-1). However, soil loss estimates for modeling surface water and off-site impacts will be required for other source categories. The following is offered as general guidance and background for estimation of unit soil losses in this assessment.

The unit soil loss is commonly estimated using the Universal Soil Loss Equation. This empirical equation estimates the amount of soil eroding from the edge of a field (Wischmeier and Smith, 1965):

$$SL = R \ K \ LS \ C \ P \qquad (5\text{-}4)$$

where:

SL	=	average annual soil loss, Eng. tons/acre-year
R	=	rainfall/runoff erosivity index, t-ft/ac-yr
K	=	soil erodibility factor, t/ac-(unit of RF)
LS	=	topographical factor, unitless
C	=	cover and management practice, unitless
P	=	supporting practices factor, unitless.

Several references are available to evaluate USLE factors for agricultural and non-agricultural settings (EPA, 1977; USDA, 1974; Wischmeier, 1972; Novotny and Chesters, 1981). For this assessment, values for four of these terms, the R, K, LS, and P factors, were determined from a survey of over 70 sanitary landfills in the United States (SAIC, 1986). The fifth term, the cropping factor C, is discussed independent of that survey.

- **Rainfall/erosivity index, R:** The R term represents the influence of precipitation on erosion, and is derived from data on the frequency and intensity of storms. This value is typically derived on a storm-by-storm basis, although it has been compiled regionally for average annual values (EPA, 1977). This value is one such annual average value. Annual values range from < 50 for the arid western United States to > 300 for the Southeast. The value from the landfill survey of 155 is typical of rainfall patterns seen in much of the midwestern United States.

- **Soil erodibility, K:** The soil erodibility factor reflects the influence of soil properties on erosion, with values ranging from <0.05 for non-erodible sandy soils to >0.50 for highly erodible silty soils. The landfill survey value of 0.23 represents mid-range erodibility typical of, for example, sandy loam with 2% organic matter.

- **Length-slope factor, LS:** The topographic factor reflects the influence of slope steepness and length of the field in the direction of the erosion. Steeper slopes and longer lengths lead to higher LS values, with a range of 0.1 for slopes <1.0% and lengths <100 ft to >2.0 for slopes generally >10%. A landfill survey value of 1.5 can correspond to a variety of combinations, such as a 8% slope over a 100-foot length or a 4% slope over a 1,000-foot length.

- **Support practice factor, P:** The support practice factor reflects the use of surface conditioning, dikes, or other methods to control runoff/erosion. P can be no greater than 1.0. The landfill survey value of 1.0 reflects a compacted surface without control structures.

The final term in the USLE is the cover and management practice factor, C, which primarily reflects how vegetative cover and cropping practices, such as planting across slope rather than up and down slope, influences erosion. C values can be no greater than 1.0, with this value appropriate for bare soils. A C value of 1.0 is an appropriate choice for active landfills. For an inactive landfill with grass cover or any area with dense vegetative cover, a value of 0.1 is appropriate.

The landfill survey data (SAIC, 1986) allowed for averages for the R, K, LS, and P values, as noted above. The survey data could be used in two ways. These average values could be multiplied together, which yields (R)(K)(LS)(P) = 53 t/ac-yr. Alternatively, the factors for each landfill in the SAIC survey could be multiplied together and then averaged, which yields (R)(K)(LS)(P) = 62 t/ac-yr. Since there may be some dependency among the factors, the latter method may be more appropriate. A unit erosion rate from bare, contaminated sites will be assumed to be 62 t/ac-yr (converted to kg/ha-yr) in this assessment. The average erosion rate for other areas in the watershed, SL_W, used in this assessment will be 6 t/ac-yr. The factor of ten difference assumes that vegetation

reduces erosion by a factor of ten on the average in the watershed as compared to bare contaminated sites.

It should be noted that more sophisticated models are available for estimating erosion rates (i.e., CREAMS as described in Knisel, 1980), and should be considered in actual site-specific assessments.

- **A_s and A_w:** These are the area terms, including the area of the exposure site and the effective drainage area of the watershed, both in ha. The scenarios demonstrated in Chapter 9 have assumed 0.4 ha (1 acre roughly) for exposure sites described as rural residences and 4 ha (10 acres) for farms. The total area impacting a river system has been termed a watershed. For purposes of this assessment, an "effective" drainage area can be less than the total area of a watershed. Instead, it equals only the portion of the watershed impacting the river system upgradient of the point where water is extracted for drinking or fish are recreationally caught for fishing. For example, say a contaminated site is at the top of a watershed, and water was extracted and fish were caught also at the top of the watershed. Then sediments and hence water will principally be impacted by that contaminated area and a small top portion of the larger watershed; the "effective" drainage area would equal a small part of the total watershed including the contaminated site. If, on the other hand, water were extracted for drinking and fish for consumption at the bottom of the watershed, the "effective" drainage area would equal the entire watershed. If water were extracted upgradient of the contaminated site, than the sediment would not be impacted by the contaminated site. For a standing water body such as a lake, and given an assumption of complete mixing, the effective area equals all areas contributing sediment, including the contaminated site.

A useful data source for this term and the suspended sediment term below, for specific sites in the United States, is Appendix F in Mills, et al. (1985). This appendix includes a compilation of data from river and reservoir sediment deposition surveys, including drainage area, water body volumes, and rates of sediment deposition (mass/area-time). Water bodies in this data base are located in the 48 conterminous states. An estimate of suspended sediment concentrations can be made using the water volume and the sediment deposition rates from this data, and an assumption on sediment deposition velocity. The specific weight of sediments in the water body, also supplied in

this appendix, can be used to estimate sediment deposition velocity.

A survey of 70 landfills describe above (for estimation of unit soil loss terms) revealed that the landfill were within an average watershed size of 4,000 hectares (10,000 acres, 15.6 mi^2). This will be the assumed size of all watersheds for the example Scenarios in Chapter 9. Furthermore, it will be assumed that water is extracted at the bottom of the watershed for drinking, and this will also be where fish are recreationally caught for consumption. With this assumption, the placement of the contaminated site in relation to the point where water is extracted does not become relevant. The effective drainage area for the example scenarios in Chapter 9 will be 4000 hectares.

- **SD_s and SD_w:** These are the sediment delivery ratios applied to the exposure site and the watershed as a whole. Such a ratio is required because not all the soil which erodes from an area deposits into the receiving water body. The following delivery ratio was proposed for construction sites (EPA, 1977):

$$SD_s = (3.28 \ DL)^{-0.22} \qquad (5\text{-}5)$$

where:

SD_s	=	sediment delivery ratio from site of interest, unitless
DL	=	distance from site to receiving water body, m
3.28	=	converts m to ft (empirical equation was developed for units of ft).

Note that the sediment delivery empirical equation simplifies all land features pertinent to erosion to a function only of length. The equation was developed to estimate sediment loads from construction sites to nearby surface water bodies, and from distances up to 250 m (800 ft, roughly). Without specific information on the sites from which it was developed, it is assumed that the land area between the construction sites and the receiving water body is "average" and this relationship can be used for applications other than construction sites.

As noted in previous bullets, the example scenarios demonstrating the on-site source category assumed $C_s = C_w$, and $SL_s = SL_w$. With these assumptions, derivation for SD_s and SD_w are irrelevant since they cancel in Equation (5-1). However, sediment

delivery ratios are required for the other example scenarios. In those scenarios, the impacted water body was assumed to be 150 meters away from the site of contamination, or site of exposure. This distance translates to a delivery ratio of 0.26. Site-specific conditions could result in a larger (steeper slope, e.g.) or smaller proportion of the eroded soil being delivered to the water body than would be estimated with this equation.

Figure 5.1 shows a watershed delivery ratio as a function of watershed size (figure from Vanoni, 1975). As seen, the ratio decreases as land area increases. The total watershed size assumed for the example scenarios in Chapter 9 was 4,000 hectares, or 40 km^2. From Figure 5.1, this translates to a watershed delivery ratio, SD_w, of 0.15.

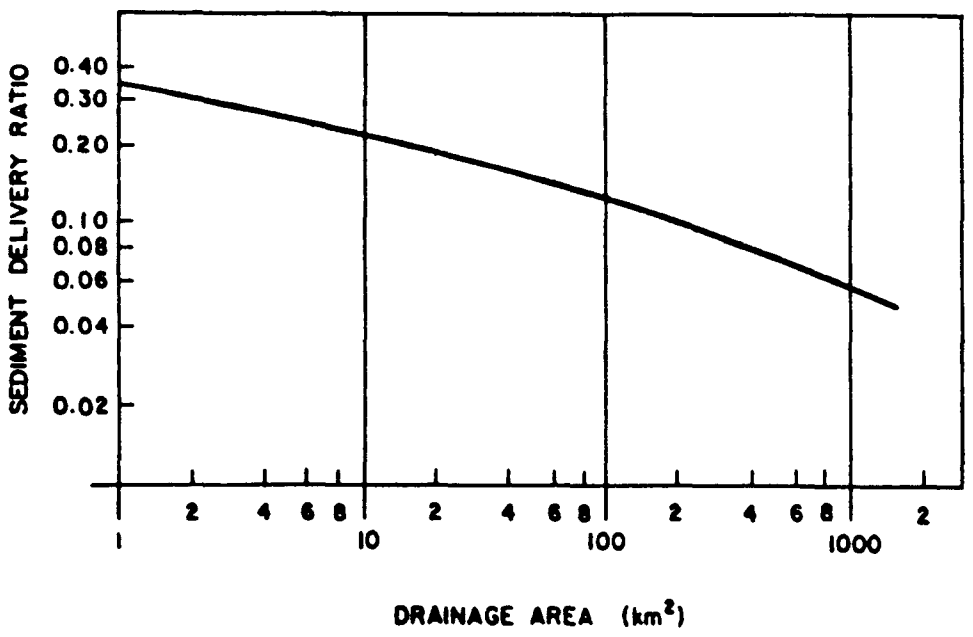


Figure 5.1. Watershed delivery ratio, SD_w, as a function of watershed size (figure from Vanoni, 1975).

- TSS: This is the total suspended sediment in the water body. This value will be lower for standing water bodies such as ponds or lakes as compared to streams or rivers. The more turbulent flow in rivers will suspend sediments to a greater degree than a

relatively calm lake. A complex modeling exercise evaluating the impact of 2,3,7,8-TCDD to Lake Ontario assumed a suspended sediment concentration of 1.2 mg/L (EPA, 1990b). For use in pond or lake settings, an assumption of a suspended sediment concentration of 1-2 mg/L is reasonable. All example scenarios in Chapter 9 assume that the 4,000 ha watershed drains into a river suitable for supporting fish for consumption and water for drinking purposes. General guidance offered for the potential for pollution problems in rivers and streams as a function of suspended sediment concentration are: 10 mg/L or less - no problem, 100 mg/L or less - potential problem, and greater than 100 mg/L - probable problem. A cutoff concentration for protection of aquatic life is 80 mg/L (Mills, et al., 1985). The value assumed for TSS for all example scenarios in Chapter 9 is 10 mg/L, indicating no turbidity problems and a river fully supportive of fish for consumption.

- **Kd_{ssed}:** This adsorption partition coefficient describes the partitioning between suspended sediment and the water column. For numerous applications for organic contaminants, particularly for estimating the partitioning between soil and soil water, this partition coefficient has been estimated as a function of the organic carbon partition coefficient and the fraction organic carbon in the partitioning media:

$$Kd_{ssed} = Koc \quad OC_{ssed} \qquad (5-6)$$

where:

Kd_{ssed}	=	partition coefficient between suspended sediment and water, L/kg
Koc	=	organic carbon partition coefficient for contaminants, L/kg or cm^3/gm
OC_{ssed}	=	fraction organic carbon content of suspended sediment, unitless.

The organic carbon partition coefficient, Koc, can be a measured value or it can be estimated. Schroy, et al. (1985) listed an organic solids/water partition coefficient of 468,000 for 2,3,7,8-TCDD. Information in Jackson, et al. (1986), imply that this is a very low partition coefficient for 2,3,7,8-TCDD. They obtained soil samples contaminated with 2,3,7,8-TCDD from 8 sites in the Times Beach area of Missouri, and 2 from industrial sites in New Jersey. These contaminated soils had 2,3,7,8-TCDD concentrations ranging from

8 to 26,000 μg/kg (ppb), and organic carbon contents ranging from 0.015 to 0.08. They determined soil water partition coefficients, Kd_s, for these soil samples, and using the organic carbon fraction data, estimated Kocs for 2,3,7,8-TCDD. The mean Koc from these ten samples was roughly 24,500,000. EPA (1990) evaluated the Koc for sorption of 2,3,7,8-TCDD onto Lake Ontario sediments. They concluded that log Koc was greater than 6.3 (Koc = 2,000,000), but less than 7.3 (Koc = 20,000,000).

In the absence of measured values, the Koc can be estimated from a chemical's octanol water partition coefficient, Kow. Empirical equations relating Kow to Koc are listed in Lyman, et al. (1982). Of six different equations listed in that reference, the following derived by Karickhoff, et al. (1979) is used to estimate the Koc for the example compounds in Chapter 9:

$$\log Koc = \log (Kow) - 0.21 \qquad (5\text{-}7)$$

where:

Koc = organic carbon partition coefficient, L/kg

Kow = octanol water partition coefficient, unitless

This equation was empirically developed from a limited number of hydrophobic contaminants (n=10, R^2 = 1.00). It implies that Koc is very similar to Kow for strongly sorbed compounds such as the dioxin-like compounds. Using the log Kow of 6.64 given in this assessment for 2,3,7,8-TCDD in Karickhoff's relationship estimates a Koc of roughly 2,700,000.

- **OC_{sed}, OC_{ssed}:** The organic carbon content of solids and sediments of water bodies are generally higher than organic carbon contents of the surrounding lands. Furthermore, organic carbon contents of suspended organic materials and solids are typically greater than those of bottom sediments. A significant sink for strongly hydrophobic contaminants such as the dioxin-like compounds is thought to be suspended, or non-settling, organic material. In modeling 2,3,7,8-TCDD in Lake Ontario (EPA, 1990b) using the WASP4 model, a compartment separate from suspended solids termed "non-settling organic matter" served as a permanent sink. For purposes of this assessment, a

single reservoir of suspended materials onto which incoming TCDD sorbs is principally characterized by OC_{ssed}, and the values selected for OC_{sed} and OC_{ssed} should reflect the relative partitioning behavior of suspended and bottom materials. As noted above, these water body carbon contents are also related to the organic carbon contents of surrounding soils. The model parameter, OC_{sl}, is the soil organic carbon fraction and is required for modeling of soil contamination by dioxin-like compounds. Foth (1978) lists the organic nitrogen content of several soil types ranging from sand and sandy loam to clay. The range from that list is 0.0002 - 0.0024 on a fractional basis. Assuming a carbon to nitrogen ratio of 10 (Brady, 1984; who notes that C:N ratios vary from 8 to 15, with the typical range of 10 to 12), organic carbon ranges from 0.002 to 0.024. A soil organic carbon fraction, OC_{sl}, is assumed to be 0.01 for all example settings in Chapter 9, which is in the middle of this range. The organic carbon content of bottom sediments, OC_{sed} will be higher at 0.02. Bottom sediments originate as erosion from surrounding land, but also include decay of organic materials within water bodies. The organic carbon content of suspended materials can approach 0.20, but OC_{ssed} will be assumed to be 0.05 for the example settings in Chapter 9.

5.3.2. Vapor-Phase Air Concentrations

The algorithms for estimating vapor-phase concentrations of contaminants were presented and derived in Hwang, et al. (1986). These procedures were developed for soil surface and subsurface contamination with polychlorinated biphenyls, PCBs. The models are based on the assumptions that: 1) PCBs move through the soil primarily by vapor phase diffusion, i.e., leaching is not considered, 2) PCB vapor in the soil matrix reaches a local equilibrium with pore air, 3) degradation processes for PCBs were not considered , and 4) the PCB contamination occurs at the surface and extends down infinitely. These assumptions are similar to the general types of assumptions that have been made for all the algorithms estimating exposure media concentrations in this assessment. The procedures in that PCB assessment were also used for this assessment. Details of the derivation are presented in Hwang, et al. (1986).

The average flux rate over an exposure duration of ED can be estimated as:

$$FLUX = \frac{(2)\ (E_{slp})\ (D_{ea})\ (C_s)\ (H)\ (41)\ (10^{-6})}{Kd_s\ [\ (\pi)\ (I)\ (ED)\ ]^{0.5}} \qquad (5\text{-}8)$$

where:

FLUX	=	average volatilization flux rate of contaminant from soil, g/cm^2-s
E_{slp}	=	soil pore porosity, unitless
D_{ea}	=	effective diffusivity of contaminant in air, cm^2/s
C_s	=	contaminant concentration in soil, ppm or mg/kg
H	=	Henry's Constant of contaminant, atm m^3/mol
Kd_s	=	soil/water partition coefficient, cm^3/g
ED	=	exposure duration, s
I	=	interim undefined term for calculation, cm^2/s
	=	$\dfrac{D_{ea}\ E_{slp}}{E_{slp} + P_{soil}\ (1-E_{slp})\ [Kd_s/(41\ H)]}$
P_{soil}	=	particle bulk density of soil, g/cm^3.

The effective diffusivity, D_{ea}, is solved as a function of contaminant diffusivity in air, and soil pore porosity:

$$D_{ea} = D_c\ E_{slp}^{0.33} \qquad (5\text{-}9)$$

where:

D_{ea}	=	effective diffusivity of contaminant in air, cm^2/s
D_c	=	molecular diffusivity of contaminant in air, cm^2/s
E_{slp}	=	soil pore porosity, unitless.

The soil adsorption partition coefficient, Kd_s, is given as:

$$Kd_s = Koc\ OC_{sl} \qquad (5\text{-}10)$$

where:

Koc = contaminant organic partition coefficient, cm^3/g

OC_{sl} = fraction organic carbon in soil, unitless.

Porosity, E_{slp}, is defined as the pore space in soils occupied by air and water, and for sandy surface soils show a range of 0.35-0.50. Medium to fine-textured soils (loams, clays, etc.) show a higher range of 0.40-0.60 (Brady, 1984). Soil porosities in the example settings were 0.50. Particle bulk density is defined as the mass of a volume of soil solids. This contrasts the more common parameter, bulk density, which is the mass of a unit of dry soil, which includes both pores and solids. Particle bulk density, Psoil, has a narrow range of 2.60 to 2.75, and for general calculation purposes, Brady (1984) recommends a value of 2.65 for average mineral surface soils, the value used for the example settings. Background on Koc and OC_{sl} were given in Section 5.3.1. above.

It is noted in Hwang, et al. (1986) that this procedure would tend to overestimate emissions and resulting exposures in situations involving small spills which would not involve deep contamination. It is also noted that the average flux rate is inversely proportional to the square root of the duration of exposure - the longer the duration of exposure, the lower will be the average flux rate. Whereas this solution assumes an unlimited reservoir of contaminant, it is an unsteady state solution (unlike other solution strategies) and is essentially an average flux rate over an amount of time defined by the exposure duration. Inherent in the solution was the consideration that residues dissipate by volatilization at the surface layers, resulting in contaminants diffusing upwards from deeper soil layers over time. With this longer path of diffusion, volatilized amounts decrease, and hence the average flux over time also decreases.

Vapor-phase concentrations along the center (y=0.0) of an area source can be estimated from (Hwang, 1987):

$$C_{va} = \frac{(2/\pi)^{0.5} \; FLUX \; a \; 10^{10} \; erf(e)}{u_m \, S_z} \left(e^{-0.5\,(Z/S_z)^2}\right) \qquad (5\text{-}11)$$

where:

C_{va}	=	vapor-phase concentration of contaminant in air, $\mu g/m^3$
FLUX	=	Average volatilization flux rate of contaminant from soil, g/cm^2-s
a	=	side length parallel to the wind direction, m
U_m	=	mean annual wind speed, m/s
S_z	=	vertical dispersion coefficient in air, m
z	=	height of the exposed individual, m
erf	=	error function
e	=	error function term, unitless
	=	$b/(2*(2*S_y)^{.5})$
b	=	side length perpendicular to the wind direction, m
S_y	=	horizontal dispersion coefficient in air, m
10^{10}	=	converts g/cm^2-m to $\mu g/m^3$.

This was the model used to estimate on-site vapor-phase concentrations. The dispersion terms, S_z and S_y can be estimated using site-specific wind rose data. In the absence of data, these terms can be estimated assuming the most common stability class, D, as:

$$S_y = 0.1414 \; X^{0.894} \qquad (5\text{-}12)$$

$$S_z = 0.222 \; X^{0.725} \qquad (5\text{-}13)$$

where:

$S_{y,z}$	=	horizontal and vertical dispersion coefficient, m
X	=	distance upwind of the contaminated site, m.

Mean annual windspeeds, U_m, vary from between 2.8 and 6.3 m/s (EPA, 1985b). An assumption of 4.0 m/s in the absence of site-specific average wind speeds might be acceptable. Simple assumptions can be made to assign values to the length terms above:

a, b, z, and x. Assuming a square-shaped contaminated site, a equals b which equals the square root of the area of the site. A common assumption for z, the height of the exposed individual, is 2 m. The x term can be assumed equal to a side length (a or b), or can equal the side length plus the distance to the exposed individual if the contamination is not on-site and dispersion is modeled as "near field." For the residence and farm setting examples in Chapter 9, where the contamination was on-site, the x term was equal to a side length. Other assumptions and parameter values as described here were also made for the example settings in Chapter 9.

5.3.3. Particulate-Phase Air Concentrations

The method for determining the flux of soil particles due to wind erosion for on-site conditions was developed in EPA (1985b) based on Gillette's (1981) field measurements of highly erodible soils. A key assumption for this solution is that the soil surface is assumed to be exposed to the wind, uncrusted, and to consist of finely divided particles. This creates a condition defined by EPA (1985b) as an "unlimited reservoir" and results in maximum dust emissions due to wind only. This wind erosion flux is given as (EPA, 1985b):

$$E_e = 0.036 \quad (1-V) \quad (U_m/U_t)^3 \quad F(x) \qquad (5\text{-}14)$$

where:

E_e	=	total dust flux of <10 um particle due to wind erosion, g/m^2-hr
V	=	fraction of vegetation cover, unitless
U_m	=	mean annual wind speed, m/s
U_t	=	threshold wind speed, m/s
F(x)	=	a function specific to this model.

EPA (1985b) provides details allowing for the application of this equation under a variety of circumstances. The following is offered as guidance specific to on-site conditions:

- **Fraction of vegetation cover, V:** For a "residence" or "farm", grass or crops are likely to substantially cover the soil, and V can range from 0.5 (minimal coverage) to 0.9 (more lush coverage). For the residence example settings, V was set at 0.9 which assumes a continual grass cover over the contaminated soil. The V for the farm settings was instead 0.5. The area of contamination for the example farm settings was larger than the residence setting, 10 acres to 1 acre. The land where crops were grown was also contaminated; the 0.5 value for V assumes that the cropland is totally or partially bare at some times - perhaps during spring land preparation and fall harvest.
- **Mean annual wind speed, U_m:** EPA (1985b) lists the mean annual wind speeds at 10 meter height for the 60 major cities in the U.S. These values range from 2.8 to 6.3 m/s, with an average of approximately 4 m/s. This average value was assumed for all example settings.
- **Threshold wind speed, U_t:** This is the wind velocity at a height of 7 m above the ground needed to initiate soil erosion. It depends on nature of surface crust, moisture content, size distribution of particles, and presence of non-erodible elements. It can be estimated on the basis of the following procedure (EPA, 1985b):

Step 1. Determine the Threshold Friction Velocity

This is the wind speed measured at the surface needed to initiate soil erosion. EPA (1985b) shows how it can be determined as a function of soil aggregate size distribution. However, for the "unlimited reservoir" approach for which Equation (5-14) was developed, soil particles are assumed to be fine at 1.5 mm or less as an average. This translates to a threshold friction velocity of 75 cm/s and less. A value of 50 cm/s might be reasonably assumed to be representative of these types of surfaces, and was assumed for this assessment.

Step 2. Estimate the "Roughness Height"

EPA (1985b) graphically shows the roughness height for a range of possible conditions. Included in this range are a roughness height of 0.1 cm for natural snow, 1.0 cm for a plowed field, 2.0-4.0 cm for grassland, 4.0 cm for a wheat field or for suburban residential dwellings, and up to 1000 cm for high rise buildings. The assumption made for

the residence and farm example settings was 4.0 cm, following the information given for a wheat field or a suburban residence.

Step 3. Estimate Ratio of Threshold Wind Speed at 7 m to Friction Velocity

A chart provided by EPA (1985b) shows this ratio as a function of roughness height. For a roughness height of 4.0 cm, this ratio is seen to be 13.

Step 4. Estimate Threshold Wind Speed

This is simply the product of the ratio given in step 3 above and the friction velocity. Using values given above, 50 cm/sec * 13 = 6.5 m/sec.

Finally, the model-specific function, F(x), is determined by first calculating the dimensionless ratio x, where $x = 0.886\ U_t/U_m$ (where U_t is the erosion threshold wind speed (m/s) and U_m is the mean wind speed (m/s)) and finding F(x) from a chart of F(x) versus x, as provided in EPA (1985b). For $U_t = 6.5$ and $U_m = 4.0$, x = 1.44 and F(x) = 1.05.

The unit dust flux is easily converted to a total contaminant flux by multiplying by soil concentration and area:

$$WE = (2.8 \times 10^{-13})\ C_s\ E_e\ A_{sc} \qquad (5\text{-}15)$$

where:

WE	=	contaminant wind erosion emission rate, g/s
E_e	=	total dust flux of <10 um particle due to wind erosion, g/m^2-hr
C_s	=	contaminant concentration in soil, ppb or ng/g
A_{sc}	=	area of contaminated site, m^2
$2.8*10^{-13}$	=	converts ng/hr to g/sec.

The next step in estimate particulate-phase contaminant concentration is to estimate the dispersion term. The model that is used is the same as the one used to estimate on-site vapor phase dispersion given in Equation (5-11) above. The following two

changes obtain the correct "FLUX" term for use in Equation (5-11):

1) Instead of WE, a total flux term presented here in units (g/sec) consistent with other particulate flux terms discussed in this chapter, an appropriate "FLUX" for Equation (5-11) is a unit flux term: C_s*E_e (ng/m^2-hr). Since the algorithm for E_e was developed for 10 μm size particles, the multiplication of E_e by C_s assumes that the concentration of contaminant on 10 μm size particulates is the same as that for the soil overall.

2) C_s*E_e is still not in the right units for Equation (5-11). The conversion term of Equation (5-11), 10^{10}, should instead be, .00028.

Substituting C_s*E_e for FLUX, and .00028 for 10^{10} in Equation (5-11) will allow for the estimate of C_{pa}, the particulate phase concentration of contaminant in air, in units of μg/m^3.

5.3.4. Biota Concentrations

This section summarizes the algorithms to estimate contaminant concentrations in fish, vegetation (including vegetables for human consumption and pasture grass or fodder grown on contaminated soil for beef cattle consumption), beef, and milk. As will be shown, all algorithms are simple empirical equations which relate an environmental media concentration to a biota concentration, using a "biotransfer" or "bioaccumulation" factor.

5.3.4.1. *Fish Concentrations*

The procedure and supportive data for the algorithm to estimate fish tissue concentrations can be found in Cook, et al. (1991). The information in that reference focuses on 2,3,7,8-TCDD, although there is discussion on the related compounds covered in this assessment including other CDDs, CDFs, and PCBs. As noted in that reference, these compounds share a high degree of hydrophobicity that increases as the degree of chlorination increases. Cook, et al. (1991) note that this corresponds in general to an increase in lipophilicity and an increase in ability to bind to organic carbon in sediments and to dissolved organic matter in water. However, these tendencies are not paralleled by an increase in bioaccumulation. Only the 2,3,7,8-chlorine-substituted congeners are substantially bioaccumulated by fish, although large quantities of other PCDD and PCDF congeners are found in sediments. This pattern of bioaccumulation results because of higher rates of metabolism of PCDDs and PCDFs in fish as compared to the

2,3,7,8-chlorine-substituted congeners (Cook, et al., 1991, with references to Muir et al., 1986; Gobas, 1990). While the highly chlorinated 2,3,7,8-substituted congeners are very slowly accumulated, they have very slow elimination rates.

2,3,7,8-TCDD and other planar polyhalogenated aromatic hydrocarbons often have not been detected in water from aquatic ecosystems even when biota are highly contaminated. Because surface layers of bottom sediments are a good indicator of the relative amount of chemical in the system over the time scale involved for bioaccumulation of super-hydrophobic chemicals, a term known as the Biota to Sediment Accumulation Factor, or BSAF, has been offered as a measure of site-specific bioaccumulation potential. This term was recently proposed to replace equivalent terms which were known as the Bioavailability Index, or BI (Kuehl, et al., 1987; Cook, et al., 1991, EPA, 1990b), the Accumulation Factor, AF (Lake, et al., 1990) and the Biota to Sediment Factor, or BSF (Parkerton, 1992; Parkerton, 1991, Thomann, et al, 1992). The BSAF is defined as:

$$BSAF = \frac{C_{lipid}}{C_{oc}} \qquad (5\text{-}16)$$

where:

BSAF	=	biota to sediment accumulation factor, unitless
C_{lipid}	=	concentration of contaminant in lipid of fish, mg/kg,
C_{oc}	=	concentration of contaminant in bottom sediment organic carbon, mg/kg

The surface water algorithms estimate concentration of contaminant in bottom sediments (see Section 5.3.1 above). This concentration, C_{sed}, can be converted to an organic carbon basis as a function of OC_{sed}:

$$C_{oc} = \frac{C_{sed}}{OC_{sed}} \qquad (5\text{-}17)$$

where:

C_{oc} = concentration of contaminant in bottom sediment organic carbon, mg/kg;

C_{sed} = concentration of contaminant in bottom sediment, mg/kg;

OC_{sed} = fraction organic carbon in bottom sediment, unitless

The organic carbon content of bottom sediments was assumed to 0.02, and the concentration of contaminants on bottom sediments was modeled as a function of the concentration of contaminants on incoming erosion and a modeled concentration on suspended materials (see Section 5.3.1).

Since the accumulation of contaminant is assumed to occur only in fish lipid, a correction term to estimate the whole fish tissue concentrations is needed since fish consumption in g/day refers to whole fish consumption. The correction term is simply f_{lipid}, and so whole fish concentrations are simply $C_{lipid} * f_{lipid}$.

The BSAF was developed as a measure of bioaccumulation potential rather than as a predictor, as it is being used here. It is uncertain as to whether measured BSAFs are generally applicable to other water bodies. Efforts are underway to evaluate the general applicability of BSAFs (P. Cook, Duluth Environmental Research Laboratory, US EPA, 6201 Congdon Boulevard, Duluth, MN 55804, personal communication). Using the BSAF approach as a predictive tool greatly underplays the complexity of the processes transferring contaminants from aquatic ecosystems to aquatic organisms. Following are some of the key issues to consider:

1) **Resident vs. Migratory Species:** Parkerton (1991) applied a bioenergetics-based bioaccumulation model in an attempt to duplicate BSAFs for 2,3,7,8-TCDD found for carp and blue crabs in the Passaic River, New Jersey. He showed nearly a ten-fold difference in 2,3,7,8-TCDD BSAF calculated from data for resident species as compared to migratory species in the Passaic River. This would be expected for fish which also reside part of the time in relatively clean water bodies; migration would enable depuration of residues from fish. The possibility that migration patterns might explain some of the results for fish concentrations of 2,3,7,8-TCDD in the Lake Ontario bioaccumulation study was also raised (EPA, 1990b). That assessment also discussed a

related issue of concern - to consider lakewide average sediment concentrations or concentrations near where sampled fish were captured in calculating the BSAF. Even within a large lake, more sedentary populations of fish may be impacted by localized contamination.

2) **Past history of contamination:** If contamination of surface water bodies with hydrophobic compounds like the dioxin-like compounds has occurred principally in the past, then it can be expected that most of the contamination occurs in or near the bottom sediment layer and not within the water column. Furthermore, if inputs to water bodies are declining or low in comparison to past loadings, then sediments would be undergoing depuration - residue levels would be declining, and the system may not be equilibrium. EPA (1990b) noted that very low BAF*s (defined as a fish to sediment ratio not including the sediment organic carbon and fish lipid considerations of BSAFs) and BSAFs for 2,3,7,8-TCDD in Lake Ontario contrasts higher BAF*s for other hydrophobic compounds such as DDE or PCBs. An explanation offered is that loadings to the Lake may be declining, such that there is a substantial disequilibrium between sediments, water, fish, and their prey. One parameter required in the bioenergetics model Parkerton (1991) used (referred to in the above bullet) was a ratio of contaminant concentration in bottom sediment to that in suspended sediment, rs/rw. He found closer agreement between measured and predicted BSAFs with this ratio equal to 10 in contrast to 1, the only two values tested; a ratio of ten means that much more contaminant is in the sediment compared to the water column. BSAFs found with this ratio equal to 1 were roughly 4 times as high as measured BSAFs, and BSAFs found with rs/rw equal to 10 were twice as high as measured. A related result of his modeling exercise was that, at best fit between modeled and measured BSAFs where the rs/rw was 10, dietary exposures explained over 50% of the BSAFs for carp and 85% in blue crabs, in contrast to water column exposures. He speculates that prey organisms consist of benthic animals which ingest contaminated sediment. Finally, he notes that 2,3,7,8-TCDD contamination in Passaic river largely occurred as a result of historical loadings. The picture that emerges from Parkerton's modeling is as follows: sediments are serving as an internal source of contaminants due to past historical loadings despite a reduction of contaminant levels in the water column. As a result, the bioaccumulation of these compounds in carp and blue crabs appears to be mediated by trophic transfer via the benthic foodweb. Finally, in both the Lake Ontario

and Passaic River studies, concentrations of 2,3,7,8-TCDD were higher in deeper bottom sediments as compared to surficial bottom sediments - this implies historical loadings and possibly depuration of surficial residues.

This issue is non-trivial for the methodology of this assessment, since an assumption for deriving suspended and bottom sediment concentrations is that the contamination is ongoing, and that the hypothetical water body may be closer to a state of equilibrium as compared to situations where contamination was principally in the past. The BSAF assumed for 2,3,7,8-TCDD was based mainly on the data from EPA (1990b) on Lake Ontario and from Parkerton (1991) from data in Passaic River. If 2,3,7,8-TCDD contamination in these water bodies occurred mainly in past, and not in the present, then measured BSAFs might be lower than those that might be measured in water bodies where contamination is ongoing.

3) **Variations among fish species:** Feeding habits, age, migratory patterns, and lipid contents (including lipid content of edible vs. inedible fish tissues) are just a few of the interacting factors which determine a site-specific BSAF as a function of fish species. The demonstration of this approach in Chapter 9 assigns a single BSAF to each of the three example contaminants. Although not unlike other simplifications of this assessment, such approaches are recognized as oversimplifications.

4) **Study designs to obtain BSAFs:** Although there is some evidence that BSAFs specific to a contaminant may be applicable to other aquatic settings, data to evaluate such a hypothesis is still sparse. Even data sets that do exist need to be carefully evaluated before deriving BSAFs. Such an evaluation should consider data associated with the sediment as well as data associated with the fish species. Critical factors for sediment sampling include location, number, depth of sampling, variability, availability of organic carbon fraction information, and so on. Similar issues are pertinent for fish sampling and analysis.

Following now are guidance for the terms required for estimating fish tissue concentrations.

- **BSAF:** Table 5.2 summarizes available biota sediment accumulation factors for dioxin and furan congeners. Only four sets of data with such estimates were available in the literature. The data from the Wisconsin River (Kuehl, et al. 1987) and that from 1 lake in Sweden (Kjeller, et al. 1990) both show decreasing BSAF with increasing chlorination. The BSAF of 2.94 for 2,3,7,8-TCDD determined from a lake in Sweden should be questioned since it is more than an order of magnitude different than any of the other data. Causes for this discrepancy could be manifold. Some observations from Kjeller, et al. (1990) might shed some light on this result. Although sediment data was from three water bodies, 8 of the 9 Pike samples (pike samples were composites of 2-5 fish from one location in the water body) were from one of the water bodies. This is why only data from the one water body was summarized in Table 5.2. This water body, Lake Vanern, was clearly the most contaminated of the three water bodies studied. A paper mill was located at the northern part of this lake and the authors concluded that discharges from this mill impacted the lake. The average of 2,3,7,8-TCDD and 2,3,7,8-TCDF organic carbon normalized concentrations for five sediment samples from this lake was 297 pg/g; the analogous average concentration for 10 samples taken from another lake, Lake Vattern (5 samples), and a river, Dala (5 samples), was 65 pg/g. A similar disparity between Lake Vanern and the other water bodies is found with the penta-CDD/CDF concentrations: 205 pg/g vs. 108 pg/g, with similar comparisons for the hexa-, hepta, and octa-CDD/CDF. The sediment and corresponding pike sample nearest this mill had the highest concentrations reported - pike samples were given as 3000 and 833 pg/g lipid normalized 2,3,7,8-TCDF and 2,3,7,8-TCDF (a composite from 5 pike taken at this sampling station), respectively, and sediment was 1800 and 244 pg/g organic carbon normalized for 2,3,7,8-TCDF and 2,3,7,8-TCDD. Note the BSAF for 2,3,7,8-TCDD implied from this data point is 3.41. Another consideration for high BSAFs might be source of contamination. Speculation from the Lake Ontario and Passaic River field data was that contamination principally occurred in the past, whereas in the Swedish data, contamination appears to have been ongoing at the time of sampling. This might be one indication that BSAFs for aquatic systems where contamination is ongoing might be greater than from systems where the contamination is primarily historical.

The Swedish data also illustrates some of the complexities of interpreting literature data. First, the sediment data was expressed concentrations normalized to "sediment

Table 5-2. Available Biota to Sediment Accumulation Factors, BSAF, for dioxin-like compounds.

Reference/Congener	Fish Species	Water Body	# Sed. samples # Fish samples	BSAF	Comments
Kuehl, et al., 1987	Carp	Wisconsin River	1/1		Laboratory flow through experiment using Wisconsin River sediment and Lake Superior water; BSAFs determined from one "representative" sediment sample and one "composited" fish sample - no other details provided; sed. organic carbon and fish lipid determined analytically
2,3,7,8-TCDD				0.27	
2,3,7,8-TCDF				0.06	
1,2,3,7,8-PeCDD				0.06	
1,2,3,6,7,8- & 1,2,3,4,7,8-HxCdd				0.035	
1,2,3,6,7,8-HxCDF				0.037	
1,2,3,4,6,7,8-HpCDD				0.0048	
1,2,3,4,6,7,8-HpCDF				0.0033	
US EPA, 1990b		Lake Ontario			Comprehensive field study on bioaccumulation of 2,3,7,8-TCDD in Lake Ontario; BSAFs are lakewide and fish averages; report evaluates matching fish with sediment data from where fish were caught
2,3,7,8-TCDD	Brown Trout		55/81	0.03	
	Lake Trout		55/81	0.07	
	Smallmouth Bass		55/14	0.05	
	White Perch		55/38	0.20	
	Yellow Perch		55/77	0.03	
Kjeller, et al. (1990)	Pike	Lake Vanern in Sweden			Results presented at right derived from data in Kjeller, et. al (1990); data includes sediment samples from four sites in Lake Vanern and 6 composited (2-5 fish in composite) pike associated with the four sites; pike concentrations were expressed on a lipid basis; Lake Vanern is near a paper mill
2,3,7,8-TCDD			4/6	2.94	
1,2,3,7,8-PeCDD				1.03	
1,2,3,4,7,8-HxCDD				0.17	
1,2,3,6,7,8-HxCDD				0.086	
1,2,3,7,8,9-HxCDD				0.018	
OCDD				0.002	

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Table 5.2. (cont'd)

Reference/Congener	Fish Species	Water Body	# Sed. samples # Fish samples	BSAF	Comments
Kjeller, et al. (1990) (cont'd)					
	Pike	Lake			
2,3,7,8-TCDF		Vanern	4/6	1.40	
12348/12378-PeCDF		in Sweden		0.25	
2,3,4,7,8-HxCDF				0.71	
123479/123478-HxCDF				0.036	
1,2,3,6,7,8-HxCDF				0.065	
1,2,3,7,8,9-HxCDF				0.27	
2,3,4,6,7,8-HxCDF				0.047	
1,2,3,4,6,7,8-HpCDF				0.0009	
1,2,3,4,7,8,9-HpCDF				0.023	
1,2,3,4,6,7,8-HpCDF				0.006	
OCDD				0.0001	
Parkerton, T.F.		Passaic River			7 "resident" fish species was most represented by carp; "migratory" species were eel and striped bass; TCDD contamination attributed to historical industrial input, particularly a 2,4,5-T plant in operation in 1940s to 60s
2,3,7,8-TCDD	Resident fish		61/11	0.081	
	Migratory fish		61/15	0.009	
	Blue Crab		61/14	0.055	

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contents of organic material" (sic). This was interpreted as organic matter normalized, not organic carbon normalized. Parkerton (1991) assumed that organic carbon was 45% of organic matter to derive BSAFs when organic carbon data was unavailable; following this lead, organic matter normalized concentrations in Kjeller, et al. (1990) were divided by 0.45 to arrive at organic carbon normalized concentrations. Also, there was not an exact match in "sites" between sediment samples and fish samples; these sites were physical locations within the large lake where samples were taken. There were five sites where sediment samples were taken, and five sites where composited pike samples were taken in Lake Vanern. However, one of the sediment and one of the pike samples were from unique sites; only four sites had both sediment and pike samples. The results in Table 5.2. were derived using average sediment and pike concentrations from only these four sites. Another way to have derived BSAFs would be to average all lake sediment and pike concentrations; since there may be some relationship between sediment and pike concentrations based on lake location, it was decided to include only the four sites with both fish and sediment samples. Finally, there were two sets of results listed for 1,2,3,4,6,7,8-HpCDF as though there were two unique sets of analyses for the same congener; this is why there are two entries for this congener in Table 5.2.

Excluding the Swedish data, there are nine reported BSAFs in Table 5.2 for 2,3,7,8-TCDD. These range from 0.009 to 0.27, with an average of 0.09. A BSAF of 0.09 will be assumed for 2,3,7,8-TCDD in the example scenarios in Chapter 9. Although there is indications of declining BSAFs with increasing chlorination, there is probably not sufficient grounds to assign a BSAF for the second example compound, 2,3,4,7,8-PCDF, significantly different from that of 2,3,7,8-TCDD. The BSAF for this example furan will also be 0.09.

EPA (1990b) estimates BSAFs for PCBs and other selected chemicals (DDE, HCB, etc.) for Lake Ontario from several data sets. Parkerton (1992) summarizes BSAFs for PCBs and other compounds from other water bodies using other data sets. A selected summary by water body taken from these two sources for PCBs is given in Table 5.3.

Two trends are apparent. First, the BSAFs for PCBs appear to exceed those of the dioxin and furan congeners by an order of magnitude and more. Second, and from limited data, it would appear that BSAFs increase from dichloro- through hexa- or perhaps hepta-chloro PCBs, and then decrease thereafter. An assignment of a BSAF for

Table 5.3. Summary of Biota Sediment Accumulation Factor, BSAFs, for PCBs (summaries developed by EPA (1990b) and Parkerton (1992)).

PCB congener	Water body/Species	BSAF	Comments
PCB	Lake Ontario trout, salmon, perch, bass	4.06, 0.77, 0.77, 0.52, 0.52, 0.86, 0.86, 3.35, 3.35, 1.42 0.58	Compiled in EPA (1990) from several data sources, years of study, fish species, and so on. Summary in this table includes all uniquely derived BSAFs for PCBs from fish species noted; PCBs not further identified except BSAF=0.58 identified for perch and Aroclor 1254.
	Siskiwit Lake lake trout, whitefish		Compiled by Parkerton (1992) from data in Swackhammer, et al. (1988) and Swackhammer and Hites (1988); Parkerton presents data for individual congeners - summary at right aggregates by chlorination and including both fish species; only one data point presented for heptachloro-PCB
trichloro-PCB		0.45-2.6	
tetrachloro-PCB		0.71-1.3	
pentachloro-PCB		3.4-9.4	
hexachloro-PCB		2.9-20.8	
heptachloro-PCB		12.5	
octachloro-PCB		2.2-12.7	

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Table 5.3. (cont'd)

PCB congener	Water body/Species	BSAF	Comments
	New Bedford Harbor flounder, lobster, crab		Compiled by Parkerton (1992) from data in BOS (1990); summary at right is the range of values across noted species
dichloro-PCB		0.11-0.59	
trichloro-PCB		0.26-0.65	
tetrachloro-PCB		0.65-1.02	
pentachloro-PCB		1.05-2.08	
hexachloro-PCB		1.29-4.00	
heptachloro-PCB		0.84-2.74	
octachloro-PCB		0.23-1.17	
nonachloro-PCB		0.02-0.38	
	Rio de La Plata Argentina Three species of marine fish		Determined by Parkerton (1992) from Colombo, et al. (1990) on total PCBs; reference also has data on PCB IUPAC congeners 5-8, 14, 19, 28-31, 52, 101, 110, 138, 153, 180
Total PCB		4.40	

2,3,3',4,4',5,5'-HPCB is not apparent from the data summary below. The data point from Siskiwit for the single heptachloro-PCB, which was 2,2',3,4',5,5',6-HPCB, was estimated by Parkerton (1991) as 12.5. The BSAF for flounder from New Bedford Harbor estimated by Parkerton (1991) was 0.84, with BSAFs for lobster and crab as 1.29 and 2.74, respectively. A value of 2.00 is assigned to 2,3,3',4,4',5,5'-HPCB for the example Scenarios in Chapter 9.

Finally, it should be noted that these assignments are based on data on vertebrate rather than invertebrate aquatic species. It is generally recognized that invertebrates do not possess the enzymatic capability to metabolize hydrophobic compounds as effectively

as higher chordates. As a result, invertebrate species such as mussels, clams, oysters, shrimp, crabs and lobsters may have BSAF values much higher than those observed for fish. Parkerton (1991) and Parkerton, et al. (1992) reviewed the literature to estimate BSAFs of 1 to 5 for species including grass shrimp, sandworms, deposit feeding clams, and blue mussel for PCDD/PCDFs and PCBs.

- f_{lipid}: Lipid contents of edible fish species have not been compiled, although such a compilation would clearly be useful if applying a BSAF in an assessment mode such as is done here. BSAFs are developed on the basis of whole fish lipid content, so estimates of whole fish concentrations should assign f_{lipid} on a whole fish rather than on a tissue basis. Parkerton (1992) cautions, however, that lipid contents of edible portions of fish may be lower than lipid contents of some of the fish portions that were sampled and used to develop BSAFs. Non-edible high lipid content portions include, for example, liver and hepatopancreas. Parkerton (1992) develops the parameter, β, which is defined as the ratio of the lipid content of the edible portion and the sampled tissue. To demonstrate the impact of this ratio, Parkerton used data from Niimi and Oliver (1989) which included PCB and other halocarbon compound concentration in whole fish and fillets of fish taken from the Great Lakes. The β (defined here as the ratio of lipid in fillet to lipid of whole fish) forthese fish, which included brown trout, lake trout, rainbow trout, and coho salmon, ranged from 0.22 to 0.51. The ratio of fillet to whole fish contaminant concentrations ranged from 0.20 to 0.54. Cook, et al. (1990) assumed a lipid content of 0.07 for fish in discussions of BSAF and related methodologies for estimating bioaccumulation of 2,3,7,8-TCDD in aquatic ecosystems. This assessment will also assume a f_{lipid} of 0.07. Different lipid contents have been reported for the same fish, so generalizations are difficult to make at this point. EPA (1990b) lists percent lipid contents for Lake Ontario fish including brown trout: 14.3%, lake trout: 21.1%, coho salmon: 6.45%, yellow perch: 5.2%, and white perch: 17.1%. Kuehl, et al. (1987) lists a range of percent lipid for carp taken at different days during a study of between 13.0 and 18.7%..

5.3.4.2. *Vegetation Concentrations*

Vegetation concentrations are required for the estimation of exposure to homegrown fruits and vegetables, and also for the beef and dairy food chain algorithms.

Three principal assumptions are made to estimate vegetative concentrations:

- Outer surfaces of bulky below ground vegetation are impacted by soils which contain dioxin-like compounds. Inner portions are largely unimpacted.
- Translocation of dioxin-like compounds from roots to above ground portions of plants are negligible compared to other mechanisms which impact above ground portions of plants (this assumption is examined in Chapter 10, Section 10.2.11.3). As such, translocation into above ground portions will be assumed to be zero.
- Similar to the assumption concerning transport of contaminants from outer to inner portions of below ground vegetation, it will be assumed that outer and not inner portions of above ground bulky vegetation are impacted.

Concentration of contaminants in below ground vegetation is only required for vegetables (carrots, potatoes, e.g.) grown underground. The basis for the below ground algorithm is the experiments of Briggs, et al. (1982) on uptake of contaminants into barley roots from growth solution, and their elaboration of a Root Concentration Factor. The below ground concentration is given by:

$$C_{bgv} = \frac{C_s \quad RCF \quad VG_{bg}}{Kd_s} \qquad (5\text{-}18)$$

where:

C_{bgv}	=	fresh weight concentration of below ground vegetables, mg/kg
C_s	=	contaminant concentration in soil, ppm or mg/kg
Kd_s	=	soil-water partition coefficient, L/kg
	=	$Koc*OC_{sl}$
Koc	=	contaminant organic partition coefficient, L/kg
OC_{sl}	=	fraction organic carbon in soil, unitless.
RCF	=	root concentration factor equaling the ratio of the contaminant concentration in roots (fresh weight basis) and the concentration in soil water, unitless
VG_{bg}	=	empirical correction factor for below ground vegetation which

accounts for the differences in the barley roots for which the RCF was derived and bulky below ground vegetables, unitless

Two processes, air-borne vapor phase absorption and air-borne particle deposition, are assumed to contribute to above ground vegetation concentrations:

$$C_{abv} = C_{vpa} + C_{ppa} \qquad (5\text{-}19)$$

where:

C_{abv} = concentration in above-ground vegetation, expressed on a dry weight basis, mg/kg or ppm

C_{vpa} = contribution of concentration due to vapor-phase absorption or airborne contaminants, mg/kg or ppm

C_{ppa} = contribution of concentration due to settling of contaminated particulates onto plant matter, mg/kg or ppm.

The basis for a vapor-phase bioconcentration factor for various airborne contaminants, including 1,2,3,4-TCDD, from the atmosphere to vegetation was developed by Bacci, et al. (1990, 1992). These authors conducted laboratory growth chamber experiments on vapor-phase transfer from air to azalea leaves and developed a model to predict the vapor-phase bioconcentration for 14 organic compounds. The outcome of their work is a air-to-plant bioconcentration factor, which is termed B_{vpa} in this assessment. The algorithm estimating plant concentrations as a function of vapor-phase air concentrations is:

$$C_{vpa} = \frac{B_{vpa} \; C_{va} \; VG_{ag}}{1000 \; d_a} \qquad (5\text{-}20)$$

where:

C_{vpa} = contribution concentration due to vapor-phase absorption or airborne contaminants, mg/kg or ppm

B_{vpa} = air-to-leaf biotransfer factor, unitless [(μg contaminant/kg plant dry)/(μg contaminant/kg air)]

C_{va} = vapor-phase concentration of contaminant in air, $\mu g/m^3$

VG_{ag} = empirical correction factor which reduces vegetative concentrations considering that B_{vpa} was developed for transfer of air-borne contaminants into leaves rather than into bulky above ground vegetation

d_a = density of air, kg/m^3, 1.19

1/1000 = converts resulting concentration from μg/kg to mg/kg.

Several exposure efforts for 2,3,7,8-TCDD (Fries and Paustenbach, 1990; Stevens and Gerbec, 1988; Connett and Webster, 1987; Travis and Hattermeyer-Frey, 1991), have modeled the accumulation of residues in vegetative matter (grass, fodder, vegetables) resulting from deposition of contaminated particulates. Key components of their approach, as well as the one for this assessment, include:

- Vegetative concentrations result from particulate deposition onto plant surfaces.
- Vegetative dry matter yield is the reservoir for depositing contaminants; this reservoir varies according to crop.
- Not all particulate deposition reaches the plant, some goes directly to the ground surface; the "interception fraction", less than 1.0, reduces the total deposition rate.
- Weathering processes, such as wind or rainfall, remove residues and this process is reasonably modeled as a first-order exponential loss with an associated weathering dissipation rate. All the above references have justified a dissipation rate derived from a half-life of 14 days (based principally on field measurements described in Baes, et al. (1984)); this is the value used for all dioxin-like compounds in this assessment as well.

- Vegetative concentrations may not reach steady state because of harvesting or grazing, but a steady state algorithm is used.

The steady state solution for plant concentrations attributed to particle deposition is:

$$C_{ppa} = \frac{F \quad I_j}{1000 \quad k_w \; Y_j} \qquad (5\text{-}21)$$

where:

C_{ppa}	=	Vegetative concentration due to settling of contaminated particulates onto plant matter, mg/kg or ppm
F	=	unit contaminant deposition rate, $\mu g/m^2$-yr
I_j	=	fraction of contaminant deposition intercepted by crop j, unitless
k_w	=	first-order weathering dissipation constant, 1/yr
Y_j	=	dry matter yield of crop j, kg/m^2
1/1000	=	converts $\mu g/kg$ to mg/kg.

Following is brief guidance on assignment of values to the terms in Equations (5-18) to (5-21).

- **C_s and Kd_s:** This is the soil concentration and soil/water partition coefficient, respectively. The soil concentration is specified for the on-site source category. For the two source categories where soil contamination is distinct from the site of exposure, the soil concentration at the site of exposure is estimated. As discussed in Section 5.4.1 below, two soil concentrations including one for a no-till and one for a tilled situation, are estimated. For estimating vegetable concentration, the tilled concentration is required. The soil partition coefficient is a function of the contaminant organic carbon partition coefficient, Koc, and the soil organic carbon fraction, OC_{sl}, as discussed above in Section 5.3.1. Division of C_s by Kd_s results in the equilibrium soluble phase concentration

of the contaminant, in mg/L.

- **RCF:** Briggs, et al. (1982) conducted experiments measuring the uptake of several compounds into barley roots from growth solution. He developed the following relationship for lipophilic compounds tested (lipophilic compounds were identified as those tested that had log Kow 2.0 and higher; n=7, r=0.981):

$$\log RCF = 0.77 \log (Kow) - 1.52 \qquad (5\text{-}22)$$

where:

RCF = root concentration factor equaling the ratio of the contaminant concentration in roots (fresh weight basis) and the concentration in soil water, unitless

Kow = contaminant octanol water partition coefficient, unitless

Since his experiments were conducted in growth solution, the RCF is most appropriately applied to soil water in field settings. This is why the C_s was divided by Kd in Equation (5-18).

- **VG_{bg}:** This correction factor and the one used to correct for air-to-leaf transfer of contaminants, VG_{ag}, are based on a similar hypothesis. That hypothesis for VG_{bg} is that the uptake of lipophilic compounds into the roots of this experiments is due to sorption onto root solids. High root concentrations were not due to translocation to within portions of the root hairs. Direct use of the RCF for estimating concentrations in bulky below ground vegetation would greatly overestimate concentrations since an assumption (stated above) is that there is insignificant translocation to inner parts of below ground bulky vegetation for the dioxin-like compounds. Concentrations in outer portions of edible below ground vegetation would mirror concentrations found in barley roots, by this hypothesis.

VG_{bg} can be estimated by assuming that the outer portion, or skin, of below ground

vegetables would contain concentrations that can be predicted directly using the RCF, but that the inner portions would effectively be free of residue. The correction factor can be estimated as the ratio of the mass of the outer portion to mass of the entire vegetable:

$$VG_{bg} = \frac{MASS_{skin}}{MASS_{vegetable}} \qquad (5\text{-}23)$$

where:

VG_{bg}	=	below ground vegetation correction factor, unitless
$MASS_{skin}$	=	mass of a thin (skin) layer of below ground vegetables
$MASS_{vegetable}$	=	mass of the entire vegetable

Simplifying assumptions are now made to demonstrate this ratio for a carrot and a potato. First, it will be assumed that the density of the skin and of the vegetable as a whole are the same, so the above can become a skin to whole vegetable volume ratio. The thickness of the skin should be assumed to be same as the thickness of the barley root for which the RCF was developed. Without that information given in Briggs, et al. (1982), what will instead be assumed is that the skin thickness is equal to 0.03 cm. This was the thickness of a leaf from broad-leaved trees assumed by Riederer (1990) in modeling the atmospheric transfer of contaminants to trees. The shape of a carrot can be assumed to be a cone. The volume of a cone is given as $(\pi/3)r^2l$, where r is a radius of the base and l is length. Assuming a carrot base radius of 1 cm and a length of 15 cm, the volume is 16 cm^3. The curved surface area of a cone is given as: $\pi(r^2 + l^2)^{1/2}$, which equals 47 cm^2, given the r and l assumptions. The volume of the cone surface area is 47 cm^2 * 0.03 cm, or 1.41 cm^3. The skin to whole plant ratio for this carrot is 0.09 (1.41/16). A similar exercise is done for a potato, assuming a spherical shape with a radius of 3 cm. The volume is given as $4/3\pi^3$, or 113 cm^3. The surface area of a sphere is $4\pi^2$, or 113 cm^2, and the volume of this surface area is 3.39 cm^3. The skin to whole plant ratio for the potato is 0.03.

This exercise indicates upper bounds for such an empirical parameter. For exposure assessments, other factors which reduce vegetative concentrations should also be

considered and will be considered in this empirical correction factor in this assessment. Additional reductions in concentration result from peeling, cooking, or cleaning, for example. Wipf, et al. (1982) found that 67% of unwashed carrot residues of 2,3,7,8-TCDD came out in wash water, and 29% was in the peels. A peeled, washed carrot correction factor might instead be, 0.09*0.04, or 0.004 (0.09 from above; 0.04 = 100% - 67% - 29%). A 96% reduction in the estimated VG_{bg} for the potato (the potato is cleaned and the skin is not eaten; additional reductions possibly when cooking the potato) would equal 0.001. In a site-specific application, the type of vegetation, preparation, and so on, should be considered. The VG_{bg} for underground vegetables for this assessment is assumed to be 0.01. This is less than the estimates of 0.09 and 0.03 for the carrot and potato above, but greater than it might be if based on this discussion on cleaning, washing, peeling, and so on.

- **C_{va}:** The vapor-phase concentration of contaminant in air used in this equation, C_{va}, is estimated using procedures described in Section 5.3.2 above.

- **B_{vpa}:** The model developed by Bacci, et al. (1990, 1992) relates the vapor-phase bioconcentration factor from air to azalea leaves, B_{vpa}, to the chemical octanol-water and air-water partition coefficients, Kow and Kaw. The air-water partition coefficient, Kaw, is a dimensionless form of Henry's Law constant, H, derived by dividing H by the product of the ideal gas constant, R, and the temperature, T. After substituting H/RT for Kaw, Bacci's algorithm for B_{vpa} can be expressed as:

$$B_{vpa} = \frac{Kow^{1.065} \quad R \quad T}{101160 \quad H} \qquad (5\text{-}24)$$

where:

B_{vpa} = air-to-leaf biotransfer factor, unitless [(μg contaminant/kg plant dry)/(μg contaminant/kg air)]

Kow = contaminant octanol water partition coefficient, unitless

H	=	contaminant Henry's Constant, atm-m^3/mol.
R	=	ideal gas constant, 8.205 x 10^{-5} atm-m^3/mol-deg K
T	=	temperature, 298.1 K
101160	=	empirical constant

Substituting R and T into Equation (5-24) simplifies the right hand side to: $2.42*10^{-6}$ $Kow^{1.065}/H$

- **VG_{ag}:** The same discussion for this correction factor for below ground vegetation applies here. Fruits such as apples, pears, plums, figs, peaches, and so on, can be approximated by spheres, and upper bound estimates of correction factors would be less than 0.05. Cooking and cleaning further reduces residues. The VG_{ag} for unspecified above ground fruits and vegetables in this assessment is assumed to be 0.01. Like VG_{bg}, this value is assigned considering that it should be less than estimated just based on surface volume to whole fruit volume ratios.

Two other VG_{ag} values are required for this assessment. One is for pasture grass and the other for fodder consumed by cattle. Both are required to estimate concentrations in these vegetations consumed by cattle in order to estimate beef and milk concentrations. A VG_{ag} value of 1.0 was used to estimate pasture grass concentrations since there appears to be a direct analogy to exposed azalea leaves. Hay can be considered to consist mostly of leafy vegetation, and a VG_{ag} of 1.0 might be appropriate. Corn grown for fodder would include substantially protected grain although cattle also consume some of the outer portions of the vegetation. Given these considerations, fodder was assigned a VG_{ag} of 0.5 in this assessment.

- **F:** Section 5.3.3 above describes the algorithm estimating the air-borne concentration of contaminant in the particulate phase, C_{pa}, in units of $\mu g/m^3$. This term needs to be multiplied by the deposition velocity, Vd, in units of m/yr, to arrive at a unit deposition rate in appropriate units. The value of Vd of 315,000 m/yr was adopted for this assessment from Seinfeld (1986) who listed a gravitational deposition velocity of 1 cm/sec for 10 μm size particles. This size is appropriate since airborne concentrations of contaminants in the particulate are estimated with parameter assignments corresponding

to inhalable, or 10 μm, size particulates.

- **kw:** Fries and Paustenbach (1990) note that this approach may overestimate concentrations because crops can be harvested or pastures grazed before the plant concentrations reach steady state, and that a kw based on a weathering half-life of 14 days may be too long given experimental results of Baes, et al. (1984) which showed a range of 2-34 days, and a median value of 10 days. Stevens and Gerbec (1987) considered harvest intervals by including the exponential term, $(1-e^{-kt})$, and assigning values of t based on harvest intervals of different crops. This assessment uses a kw of 18.02 yr^{-1}, which is equivalent to a half-life of 14 days.

- **I_j and Y_j:** Interception values and crop yields were determined in the aforementioned assessments based on geographic-specific crop yield data provided in Baes, et al., (1984) and the following types of crop-specific relationships estimating interception fraction based on yield, also presented in Baes, et al., (1984):

corn silage: $I = 1 - e^{-0.768Y}$

hay/grasses: $I = 1 - e^{-2.88Y}$

lettuce: $I = 1 - e^{-0.068Y}$

Judgments by Fries and Paustenbach (1990) on high, medium, and low yields of silage, hay, and pasture grass, and the use of the first two interception equations above (the first for silage, and the second for hay and grass), can give some guidance on interception fractions and yields for these crops:

	Yield (kg/m^2)	Intercept Fraction
corn silage	0.30 (low)	0.20
	0.90 (med)	0.50
	1.35 (high)	0.64
hay	0.25 (low)	0.51
	0.45 (med)	0.73
	1.30 (high)	0.98
grass	0.05 (low)	0.13
	0.15 (med)	0.35
	0.35 (high)	0.64

This information can be used for cattle intake of vegetation, and the resulting beef and milk concentrations. The medium values for grass and hay were used for pasture grass and fodder assumptions in the example setting in Chapter 9. Use of the hay yields and interceptions assumes that the cattle fodder grown on-site is hay and not corn silage.

Stevens and Gerbec (1988), using yields obtained from the Minnesota State Agricultural Office, derived the following yield and interception estimates, respectively, for vegetables for human consumption in their assessment: lettuce - 8.6,0.72; tomatoes - 12.0,0.55; and beans - 2.7,0.18. Average yields and interception fractions from their exercise: 7.8 kg/m^2 and 0.48, were used in the example setting in Chapter 9. These vegetable yields are fresh weight, so they need to be converted to a dry weight basis in order to estimate a C_{ppa} appropriate for use in Equation (5-19). Since vegetables are generally 80 - >90% water, a fresh to dry weight conversion factor of 0.15 was used, resulting in an average vegetable dry matter yield of 1.17 (7.8 * 0.15). This was used in the example settings in Chapter 9.

When calculating concentrations in below ground fruit and vegetables using Equation (5-18), C_{bgv} is on a fresh weight basis since the RCF developed by Briggs, et al. (1982) is on a fresh weight basis, and no correction for estimating exposures is necessary. However, C_{abv} as estimated in Equation (5-19) is on a dry weight basis, and should be multiplied by a dry weight to fresh weight conversion factor when applied to above ground fruits and vegetables. A reasonable estimate for this parameter for fruits and vegetables is 0.15 (which assumes 85% water), which was used in this assessment. When using Equation (5-19) to estimate C_{abv} for the beef and milk food chain algorithm, a conversion to fresh weight is not required, however, since the algorithms were developed assuming dry weight concentrations.

5.3.4.3. *Beef and Milk Concentrations*

The algorithm to estimate the concentration of contaminant in beef and/or milk was based on methods developed by Fries and Paustenbach (1990). Their key solution assumptions were: 1) contaminants bioconcentrate equally in the fat portions of beef and milk, 2) bioavailability, which, as they define it, is the fraction of ingested contaminant which is absorbed into the body, depends on the vehicle of ingestion - dioxin in corn oil

has a bioavailability in the range of 0.7 to 0.8, in rodent feed it has an estimate of 0.5, while in soil it has a range of 0.3 to 0.4, 3) differing concentrations of contaminant found in beef and milk fat are not due to differences in how the contaminant bioconcentrates in beef or milk fat, but rather to differences in the diets of cattle raised for beef and cattle raised for dairy products.

The concentration in the fat of cattle products is given as:

$$C_{fat} = (F\ DF_s\ B_s\ AC_s) + (F\ DF_g\ AC_g) + (F\ DF_f\ AC_f) \qquad (5\text{-}25)$$

where:

C_{fat}	=	concentration in beef fat or milk fat, mg/kg
F	=	bioconcentration ratio of contaminant as determined from cattle vegetative intake (pasture grass or fodder), unitless
DF_s	=	fraction of cattle diet that is soil, unitless
B_s	=	bioavailability of contaminant on the soil vehicle relative to the vegetative vehicle, unitless
AC_s	=	average contaminant soil concentration, mg/kg
DF_g	=	fraction of cattle diet that is pasture grass, unitless
AC_g	=	average concentration of contaminant on pasture grass, mg/kg
DF_f	=	fraction of cattle diet that is fodder, unitless
AC_f	=	average concentration of contaminant in fodder, mg/kg.

The following is offered as brief guidance to these terms and also the justification for the values selected in the example Scenarios in Chapter 9.

- **Bioconcentration ratio, F:** Fries and Paustenbach (1990) reviewed the literature on steady-state concentrations of 2,3,7,8-TCDD in the fat of meat and milk from cows ingesting contaminated feed ("feed" was not specifically defined in Fries and Paustenbach (1990); the word used in this discussion is "fodder" which is cattle feed grown on-site and impacted by soil contamination). They calculated an F of between 4

and 6, and assumed a value of 5.0 in their example applications. This value is used in the example settings for 2,3,7,8-TCDD. It is not used for the other example compounds, however. Fries and Paustenbach (1990) concluded that the bioconcentration for PCDDs, PCDFs, and PCBs were comparable to TCDD based on a literature review. They found PCB BCFs of 3.1, 4.4, and 4.8 at 20, 40, and 60 days in a particular study, and a factor of 5.7 for 1,2,3,6,7,8-CDD at 70 days. They also note, however, that bioconcentration ratios and PCDDs and PCDFs in body and milk fat of cattle decreased significantly as chlorination increased. The bioconcentration ratio of 1,2,3,4,6,7,8-CDD in beef in cows at 70 days and heifers (two data points for heifers) at 160 days was 0.4 and 0.2 (and 0.3), respectively. They note a bioconcentration ratio of OCDD in cows at 70 days and heifers at 160 days of 0.1 and 0.05. Finally, they note a bioconcentration ratio of 0.1 for OCDF in beef fat of heifers at 260 days. The second example compound in Chapter 9 was 2,3,4,7,8-PCDF. This should have a slightly lower bioconcentration ratio as 2,3,7,8-TCDD, since it is more chlorinated. However, there was no data found in the literature for this compound. The F value of 2,3,4,7,8-PCDF was set at 3.0. The third example compound was 2,3,3',4,4',5,5'-PCB, a heptachloro-PCB, might have a bioconcentration ratio less than 1.0, consistent with the above summary. It was given a value of 0.5.

- **Soil bioavailability, B_s:** This parameter reduces the bioconcentration ratio, F, considering that soil is a less efficient vehicle of transfer compared to fodder. Fries and Paustenbach (1990) reviewed several studies on the oral bioavailability of TCDD in soil in the diet of rats. Most studies used corn oil as the positive control, since there is a high absorption of TCDD, with 70-83% of the dose absorbed. Their literature review showed that the bioavailability of TCDD in soil was between 0.4 and 0.5 that in corn oil, or 0.3 to 0.4 overall. The literature implied a range of 50 to 60% of TCDD in standard rat feed is absorbed, and although few studies were available, a similar 50% absorption rate of TCDD in cattle feed was noted. They concluded, therefore, that the rat data was a reasonable surrogate for cattle. This would lead to a B_s of 0.6 to 0.7 (0.3-0.4 divided by 0.5-0.6) implying that absorption of TCDD when soil is the vehicle is 60 to 70% of what it would be if fodder were the vehicle. The soil bioavailability term used for all example compounds in Chapter 9 is 0.65.

- **Soil diet fraction, DF_s:** Fries and Paustenbach (1990) report that soil intake by cattle feeding on pasture varies between 2 and 18% of total dry matter intake, depending on whether the grazing area is lush or not. The soil diet fraction would be lower for cattle which are barn-fed with minimal opportunity for contaminated soil intake. Cattle raised for milk are rarely pastured, so one possible assumption for estimating milk fat concentrations would be a DF_s of 0.0. Fries and Paustenbach (1990) assumed between 0 and 2% of the dry matter intake by lactating cattle was soil in various sensitivity tests. Since cattle raised for beef are commonly pastured, a conservative assumption would be a high DF_s of 0.15 (15%), although a more reasonable assumption which would consider grazing in lush conditions and/or a portion of diet in fodder or supplemental feed leads to DF_s less than 0.10. Fries and Paustenbach (1990) assumed DF_s of between 0 and 0.08 for beef cattle in various sensitivity tests. The example settings in Chapter 9 assume 0.02 (2%) for lactating cattle, and 0.08 (8%) for beef cattle.

- **Fodder and grass diet fractions, DF_f and DF_g:** The sum of the three diet fractions, $DF_s + DF_f + DF_g$ must equal 1.0. Setting DF_s equal to 0.02 (2%) for lactating cattle assumed that they are pastured to some extent or could be taking in residues of soil sticking to home grown fodder. Assuming lactating cattle graze a small amount of time, the DF_g for lactating cattle will be 0.08 (8%), and therefore the DF_f will be 0.90. Beef cattle are often fattened prior to slaughter by being fed fodder, so that some of their lifetime intake of dry matter is fodder. Assuming that most of their lifetime dry matter is pasture grass, beef cattle DF_f will be 0.08, and DF_g will be 0.90.

- **Average contaminant soil concentration, AC_s:** The simplest assumption for AC_s would be that it equals the initial level of contamination, C_s. However, this would be too high if the cattle also graze in uncontaminated areas. Where cattle have random access to all portions of a grazing area with contaminated and uncontaminated portions, a ratio of the spatial average of the contaminated area to the total area should be multiplied by C_s to estimate AC_s. If cattle spend more time in certain areas, these areas should be weighted proportionally higher. Different assumptions might also be in order when using Equation (5-25) to estimate milk fat as compared to beef fat concentrations. Lactating cattle, if pastured, might graze on different areas than beef cattle. After determining a

spatial average based on current conditions, a second consideration might be given to temporal changes. If soil levels are expected to change over time (due to changes in source strength or other factors) then the concentrations should be averaged over the exposure duration as well. The example scenarios in Chapter 9 where beef and milk exposures were estimated were termed "farms". The methodologies in this chapter were used to estimate the average soil concentration over the entire farm property. Assuming the cattle are raised on the farm property, than 100% of their intake of soil comes from the farm. This means that the average soil concentration, AC_s in Equation (5-25), is equal to the level of contamination given as the initial level, or determined as average for the farm based on fate and transport algorithms.

- **Average fodder and pasture grass concentration, AC_f and AC_g:** The concentration of contaminant in pasture grass or fodder is equal to C_{abv} as calculated in Equation (5-19). As detailed in Section 5.3.4.2. above, pasture grass or fodder grown on-site can be impacted by air-to-plant vapor phase transfer and particulate deposition. Refinements noted above include the empirical parameter VG_{ag} equals 1.00 when applying the air-to-leaf transfer algorithm to pasture grass and 0.50 when applied to cattle fodder. A refinement noted here, and like AC_s above, is that an assumption needs to be made about the fraction of fodder or fraction of pasture grass that is impacted by contamination. Part of the fodder diet could come from outside sources and not be contaminated, and part of the grazing area could be far from a localized area of soil contamination, making it less impacted by contaminated particulates or vapors. The simplest assumption is that the entire vegetative diet of the cattle includes pasture grass and fodder impacted by the contaminated soil, in which case AC_f and AC_g would equal C_{abv}. For the sake of simplicity and consistency, the assumption made for AC_s was also made for AC_f and AC_g in the example Scenarios in Chapter 9. That is, the grass and fodder intakes of beef and dairy cattle originate within the farm property and concentrations in grass and fodder are a function of the soil concentrations within the farm property; AC_f and AC_g are equal to C_{abv} as calculated in Equation (5-19). For site-specific situations, AC_f and AC_g should be estimated as C_{abv} reduced according to assumptions on quality of cattle fodder, and impacts of air-borne contaminants on grazing land and cattle fodder grown at the site where cattle are raised.

There is one final but critical note on solving for beef and milk concentrations given a solution for C_{fat} as in Equation (5-25). Human daily ingestion amounts are typically expressed in whole product rather than the fat portion of product. Whole milk is 4% fat, meaning that the C_{fat} needs to be multiplied by 0.04 to get whole milk concentration. Similarly, beef is generally 22% fat, meaning that the C_{fat} needs to be multiplied by 0.22 to get whole beef concentration. However, the ingestion rates in this assessment for beef and milk were developed on a fat basis, so no adjustment is necessary (see Chapter 8).

5.4. ALGORITHMS FOR THE "OFF-SITE" SOURCE CATEGORY

As noted in Section 5.1, the contaminated soil is remote from the site of exposure for the "off-site" source category. A common example is an industrial site with soil contamination or a landfill with contaminated soil. The example settings in Chapter 9 include an industrial site with bare, contaminated soil and an active ash landfill. Since many of the parameters in the algorithms discussed below are specific to particular off-site soil contamination sites, guidance in this section as well as Section 5.6 will be specific to the example settings in Chapter 9.

Table 5-1 summarized the fate and transport algorithms for this source category and as seen, several of the algorithms estimating exposure media concentrations are the same or very nearly the same as in the on-site source category. Following now are bullet summaries for similarities and small refinements to these algorithms. Sections below describe algorithms that are unique for the off-site source category.

- **Surface water impacts:** Equation (5-1) is applied to estimate concentrations in suspended and bottom sediments for this source category. Section 5.3.1. describes the parameters required for these equations. In applying this algorithm for the example Scenario demonstrating the off-site source category in Chapter 9, Example Scenario 3, the important assumption was made that the average concentration of contaminants in the watershed was very low compared to the concentration on the contaminated soil - hence, C_w was set to 0.0. The impact to the surface water body was just from the contaminated site. The unit erosion rate from the contaminated site, SL_s, is assumed to be 62 t/ac-yr. The unit erosion rate from other areas of the

watershed, SL_w, is assumed to be 6 t/ac-yr. Derivation of these terms is given above in Section 5.3.1. The contaminated site is assumed to be 150 meters away from the water body, and SD_s is estimated therefore as 0.26. Water is assumed to be extracted for drinking at the bottom of the watershed. Therefore, the effective drainage area is 4000 ha. From Figure 5.1, it is seen the sediment delivery ratio associated with this drainage area is approximately 0.15, which is assumed for the example scenario in Chapter 9. The contaminated site for the example scenario is 4 ha.

- **Vapor-phase air concentrations:** The volatilization of contaminants from soil can be estimated similarly to the way described in Section 5.3.2 for the on-site source category with no clean cover. Section 5.4.2 below describes an approximation which reduces this flux when there is a clean cover, such as in a capped landfill. The example scenario in Chapter 9 does not assume a clean cover. Also, section 5.4.2 describes a dispersion model which transports contaminants through the air to the exposure site. The far-field dispersion model described in Section 5.4.2 differs from the near-field dispersion model presented in Section 5.3.2.
- **Particulate-phase air concentrations:** The same model for particulate flux due to wind erosion is used for the off-site source category as described in Section 5.3.3. This flux is considered only when there is no clean cover. However, two parameters might be different than described above (Section 5.3.3, Equation (5-14)) if the model is applied to off-site soil contamination when the soil is bare. One is the vegetative cover, V, which might be more appropriately assigned a 0.0 implying no ground cover for an active landfill or an industrial site. The other is the threshold wind speed, U_t. The different assumption would be in roughness height, assumed 4 cm for a residence or farm setting, but perhaps more appropriately assumed to be 1.0 cm for bare soil. This value is appropriate for a tilled field (EPA, 1985b). With this change, a Ut is calculated as 8.25 m/sec, and F(t) is calculated as 0.5. Note that V and U_t might be the same as a residential or farm setting if the off-site soil contamination had a grass cover, and contamination originated at the surface - i.e., no layer of clean soil. Once a flux has been

calculated, the far-field dispersion model used in vapor-phase transport (Section 5.4.2. below) is used also for estimating air-borne particulate-phase contaminant concentrations at the exposure site. It is noted that if a landfill were to have a clean cover of uncontaminated soil, than particulate emissions would not occur from the landfill. The example ash landfill in Chapter 9 was "active", meaning that the soil was bare and contaminated at the surface.

- **Biota concentrations:** The basic strategy for estimating biota concentrations - as a linear function of environmental media concentrations (bottom sediment concentrations, soil, air) and based on bioaccumulation or biotransfer factors (along with diet fractions, etc.) - remains the same. Section 5.4.1. describes how exposure site soil concentrations are estimated from concentrations at the off-site source. Exposure site soil then becomes a "source" for plant, beef, and milk contaminant concentrations. Similarly, air-borne particle and vapor-phase contaminants originating from the off-site become sources for pasture grass and fodder concentrations, which are above ground vegetations. As described below, exposure site soil concentrations are a function both of the amount of soil estimated to erode from the off-site contamination, and of a mixing depth which is different for "tilled" vs. "untilled" situations. The soil concentration used for cattle ingestion of soil is assumed to be untilled. The soil concentration used to estimate concentrations in underground vegetables is assumed to be tilled. The algorithm to estimate fish tissue concentrations as a function of bottom sediment concentrations remains the same for this source category.

The soil ingestion and dermal exposure pathways are still a function of exposure site soil concentrations; i.e., no assumption of direct contact with the off-site contamination is made. Also, both of these direct soil exposure pathways used the untilled soil concentrations.

Section 5.4.1. discusses how exposure site soil concentrations are calculated from off-site concentrations. Section 5.4.2. describes adjustments to the volatilization flux algorithm and the far-field dispersion model which transports vapor and particulate-phase

residues to the nearby exposure site.

5.4.1. Exposure Site Soil Concentrations

The key assumptions for the solution strategy estimating exposure site soil concentrations resulting from an off-site soil contamination source are: 1) the exposure site soil becomes contaminated due to erosion of contaminated soil from the source to the exposure site, 2) the amount of soil at the exposure site does not increase, which means that soil delivered to the site via erosion is matched by an equal amount which leaves the site, and 3) not only does soil erode off the contaminated site en route to the exposure site, but soil between the contaminated site and the exposure site also erodes to the exposure site.

The second and third assumption translate to:

$$D_1 + D_2 = R \qquad (5\text{-}26)$$

where:

D_1	=	mass of soil delivered from off-site contaminated source, kg
D_2	=	mass of soil delivered from land area between contaminated source and exposure site, kg
R	=	mass of soil removed from exposure site, kg.

The mass balance equation for exposure site soil concentrations can now be qualitatively stated as (with "ΔC" used as shorthand for change in exposure site soil concentration over time):

		(the incremental addition to C resulting from the change in erosion of contaminated soil) *
ΔC	=	(the incremental substraction of C resulting from removal of now contaminated soil from the exposure site) *
		(the incremental substraction of C resulting from dissipation of residues at the exposure site)

This can be expressed mathematically as:

$$\frac{dC}{dt} = \frac{D_1 C_o}{M} - \frac{RC}{M} - kC \qquad (5\text{-}27)$$

where:

C	=	the exposure site soil concentration, mg/kg
D_1	=	mass of soil delivered from off-site contaminated source, kg/yr
C_o	=	concentration of contaminant at contaminated site, mg/kg
M	=	mass of soil at exposure site into which contaminant mixes, kg
R	=	mass of soil removed from exposure site, kg/yr
k	=	first order dissipation rate constant, 1/yr.

Assuming that the contaminant concentration at the exposure site, C, is initially 0, Equation (5-27) can be solved to yield:

$$C = \frac{D_1 C_o}{R + kM} \left[1 - e^{-(\frac{R}{M} + k)t} \right] \qquad (5\text{-}28)$$

which computes C as a function of time, t (in years since k is in years). This can be solved for various increments of time starting from a time when the exposure or contaminated site initially became contaminated, or it can be simplistically assumed that the contamination has existed at the contaminated site for a reasonably large amount of time such that the exponential term approaches zero. This can be alternately stated that the assumption is made that the system has reached steady state over a long period of time. Either way, the exponential term drops out, and C is estimated as:

$$C = \frac{D_1 C_o}{R + kM} \qquad (5\text{-}29)$$

Equation (5-29) was used to estimate exposure site concentrations resulting from off-site contamination for the example scenario in Chapter 9. Guidance for estimation of these terms including justification for their values as selected in the example settings are:

- **k:** For the on-site source category, and for contaminated soil at the off-site contaminated location, the assumption is made that residues do not degrade or dissipate to the point of reducing the concentration of the "initial" soil levels. This was partly based on information indicating generally low rates of biological or chemical degradation for the dioxin-like compounds of this assessment, coupled with the assumption that on-site and off-site contamination was sufficiently deep implying a reservoir of contaminant that would remain available during a period of exposure. These assumptions are less likely to be valid for residues which have migrated over the surface to deposit on the exposure site. The deposition is likely to result in only a thin layer of contaminated soil. Though very small, surface-related dissipation mechanisms such as photolysis, volatilization, or degradation, might reduce surface soil contaminant concentrations. For these reasons, a "dissipation" rate constant is assumed to apply to delivered contaminant, where the precise mechanisms of dissipation are not specified, but could include transport (volatilization, erosion) and degradation (photolysis, biodegradation) mechanisms. The studies on 2,3,7,8-TCDD described in Young (1983) imply a dissipation half-life of 10 years. Fries and Paustenbach (1990) suggested the use of a half-life of at least 10 years, and used a 15 year half-life in their example scenarios on the impact of air-borne deposition of 2,3,7,8-TCDD originating from stack emissions. This assessment uses a dissipation half-life of 10 years for all of the three example compounds in Chapter 9. This half-life translates to a first-order dissipation rate constant of 0.0693 yr^{-1}.

- **M:** The delivered contaminant mixes to a shallow depth at the exposure site. The mixing depth depends on activities which disturb the surface, such as construction, plowing, vehicle traffic, movement of cattle or other animals, burrowing

action of animals, other biological activity, normal leaching, and raindrop splash. Mixing depths for fallout plutonium have been found to be 20 cm on cultivated land and 5 cm on uncultivated forest and rangeland (Foster and Hakonson, 1987). Fries and Paustenbach (1990) suggested a depth of 15 cm for agricultural tillage, but assumed values of 1 and 2 cm for various sensitivity tests. However, they did not need to make a distinction between tilled and untilled situation because vegetation (pasture grass and forage for estimating beef and milk fat concentration; above ground fruits and vegetables for human consumption) was assumed to be impacted only by particulate deposition and not root uptake. In another assessment on indirect impacts from incinerator emissions, EPA (1990) estimated vegetation concentrations as a function of particulate depositions, root uptake, and air-to-leaf transfer from the vapor phase. Different mixing depths for untilled and tilled concentration estimation was required. For root uptake estimation for vegetable and other crops, the estimated soil concentrations assuming a tillage mixing depth of 20 cm. For soil concentrations in untilled situations, they assumed a mixing depth of 1 cm. The methodology of this assessment uses 1 cm for the untilled and 20 cm for the tilled conditions. Soil concentrations for calculation of concentrations of underground vegetables will be a function of a 20-cm depth. This assumption is made because tilling gardens is assumed to distribute surface residues to the 20-cm depth. Soil concentrations for dermal contact, soil ingestion, and pasture grass and soil intake for cattle grazing will assume a depth that has been used for the untilled situation, that of 1 cm. These activities are assumed to occur on soil which has not been tilled. These assumptions will be used for both of the situations where the source of contamination is off-site and residues get delivered to the exposure site: the off-site as well as the stack emission source categories.

Given the area of the exposure site, the mass of soil into which the eroded contaminant is mixed can be calculated as:

$$M = A_{es} \quad BD \quad d \qquad (5\text{-}30)$$

where:

M	=	mass of soil for contaminant mixing per unit depth, kg/m
A_{es}	=	area of exposure site, m^2
BD	=	soil bulk density, kg/m^3
d	=	depth of mixing, m

- **D_1 and D_2:** The first step in deriving both these amounts of soil is to use the Universal Soil Loss Equation (USLE). This approach was described above. Justification was given for an assumption of unit soil loss from the contaminated site of 62 t/ac-yr, based on USLE calculations and survey data from 70 landfills nationally. D_1 equals this unit loss times the area of contamination times a sediment delivery ratio. The example scenario in Chapter 9 assumed that the exposure site was 150 meters from the contaminated site, and using Equation (5-5), the sediment delivery ratio is 0.26. The unit loss assumed for the area between the contaminated site and the exposure site is 6 t/ac-yr. Since this area is adjacent to the exposure site, there is no sediment delivery, and D_2 equals this unit loss times the area between the contaminated and exposure sites.

D_2 and D_2 can now be expressed as:

$$D_1 = 0.224 \; SL_1 \; SD_1 \; A_{LS} \qquad (5\text{-}31a)$$

$$D_2 = 0.224 \; SL_2 \; SD_2 \; A_{BLE} \qquad (5\text{-}31b)$$

where:

$D_{1,2}$	=	mass of soil delivered from off-site contaminated source, D_1, and from the land area between contaminated source and exposure site, D_2, kg/yr,
$SL_{1,2}$	=	average annual unit soil loss, Eng. tons/acre-year, equal to 62 t/ac-yr for SL_1 and 6 t/ac-yr for SL_2
$SD_{1,2}$	=	sediment delivery ratio, unitless, 0.26 for SD_1 (with distance = 150 meters) and 1.00 for SD_2,

A_{LS}/A_{BLE} = land area of contaminated site, A_{LS}, and of area between contaminated site and exposure site, A_{BLE}, m^2

.224 = converts t/ac-yr to kg/m^2-yr.

An adjustment is made to the sediment delivery ratio, SD_1, considering the size discrepancies between the contaminated site and the exposure site. For example, if the contaminated site is larger than the exposure site, then the amount of eroded soil delivered 150 meters downgradient would not all mix with soil at the exposure site. On the other hand, if the contaminated site were smaller than the exposure site, than the full amount of eroded soil delivered 150 meters downgradient would be contained within the exposure site. A simple correction factor, equaling the ratio of a side length of the exposure site (assumed square-shaped) and a side length of the contaminated site size (also assumed square shaped), is used to adjust the sediment delivery ratio SD1:

$$SD_{1a} = SD_1 \quad CF \qquad (5\text{-}32)$$

where:

SD_{1a} = adjusted sediment delivery ratio corresponding to SD_1, unitless

SD_1 = sediment delivery ratio reducing the amount eroding from the contaminated site to be delivered to the exposure site, unitless

CF = $A_{ES}^{0.5}/A_{LS}^{0.5}$ if $A_{ES} < A_{LS}$

= 1 if $A_{ES} > A_{LS}$

A_{ES} = area of exposure site, m^2

A_{LS} = area of contaminated site, m^2

Similar considerations are pertinent to the land area between the contaminated and exposure site. Remember that the algorithm assumed that some "clean" (D_2) and some "contaminated" soil (D_1) erodes onto the exposure site, and that a similar amount of soil entering the exposure site (R, which equals $D_1 + D_2$) leaves the exposure site so as to maintain a mass balance. The amount of clean soil eroding from upgradient sources

mixing with exposure site soil can be larger than the amount of contaminated soil if the exposure site is larger than the contaminated site. If the exposure site is smaller than the contaminated, and similar to the solution for SD_{1a} above, then only the small corridor defined by the size of the exposure site contributes clean soil. Either way (i.e., the exposure site is larger or smaller than the contaminated site), the size of the land area contributing clean soil is defined by the size of the exposure site. ABLE can be estimated as the product of the distance between the exposure and contaminated site, and the side length of the exposure site:

$$A_{BLE} = DL \; SL \qquad (5\text{-}33)$$

where:

A_{BLE}	=	land area between contaminated and exposure site, m^2
DL	=	distance from landfill to exposure site, m
SL	=	side length of exposure site, m
	=	$(A_{ES})^{0.5}$
A_{ES}	=	area of exposure site, m^2.

5.4.2. Vapor-phase Transport

In the case of bare soil contamination or soil contamination with a grass cover where the contamination originates at the surface, the contaminant flux can be estimated as given in Section 5.3.2, Equation (5-8). If the contaminated soil is instead covered with contaminant-free soil, a "clean cap" situation, the solution is more complicated. A rigorous approach is detailed in Hwang, et al. (1986). Use of this approach requires a computer to iteratively solve a partial differential equation, expressed in terms of a Fourier series. It can be shown, with these equations, that the vapor emission rate through such a cover will not reach steady state for hundreds of years. For this assessment, a computer program was written, and calculations performed for 2,3,7,8-TCDD contamination with a thickness of contamination of 8 ft, and clean caps ranging from 10 to 25 cm. The results of this exercise suggest that the average emission rate of a 70-year period are 1/4 to 1/5 of what they would be without the cap. Based on this exercise, a

simple assumption can be made that a clean cap will reduce the average emission rate calculated without a clean cap by 80%. Obviously, this assumption should be viewed with caution. The industrial site and the ash landfill example settings in Chapter 9 did not have a clean cap, so this adjustment was not made.

Estimating the dispersion and resulting exposure site concentrations requires a different solution for the off-site as compared to the on-site situation. A simplified solution, given as a virtual point source model, can be found in Turner (1970). This model approximates the dispersion that occurs from an area source by using an imaginary point source. This point is located upwind of the actual source at a distance calculated to create a lateral dispersion at the site equal to its width:

$$C_{air} = \frac{2.03 \quad FLUX \quad A_{sc} \quad FREQ \quad 10^{10}}{VD \quad S_z \quad U_m} \qquad (5\text{-}34)$$

where:

C_{air}	=	concentration of contaminant in air, $\mu g/m^3$
FLUX	=	Average contaminant volatilization flux rate, calculated with or without a clean cap, g/cm^2-s
A_{SC}	=	area of contaminated site, m^2
FREQ	=	frequency wind blows from source to receptor, unitless
VD	=	virtual distance, source center to receptor, m
S_z	=	vertical dispersion coefficient, m
U_m	=	average wind speed, m/s
10^{10}	=	converts g/cm^2 to $\mu g/m^2$.

The term, FREQ, has been added to this equation to appropriately account for changing wind directions, and hence, obtain a more accurate annual average air concentration. The vertical dispersion, S_z, is estimated as an empirical function of the distance from the source center to receptor:

$$S_z = 0.222\ x^{0.725} - 1.7\ ,\ x < 1000\ m \qquad (5\text{-}35a)$$

$$S_z = 1.260\ x^{0.516} - 13.0\ ,\ x > 1000\ m \qquad (5\text{-}35b)$$

where:

S_z = vertical dispersion coefficient, m

X = actual distance from source center to receptor, m.

The virtual distance, VD, is an empirical function of the width of the contaminated area and the actual distance from source center to receptor:

$$VD = 2.514\ a + x \qquad (5\text{-}36)$$

where:

VD = virtual distance, source center to receptor, m

a = width of contaminated area perpendicular to wind direction - defined previously as side length for assumed square-shaped contaminated area, m

X = actual distance from source center to receptor, m.

Prior guidance on windspeed (Section 5.3.2) indicated windspeeds ranged from 2.8 to 6.3 m/sec, and suggested a mid-range of 4.0 m/sec in the absence of better information. Where the wind blows from all directions equally, then it will blow from one compass sector about 15% of the time. On these bases, a windspeed of 4.0 m/sec and a FREQ of 0.15 were used in the example scenarios in Chapter 9. In most places, however, wind direction is much less variable, and the appropriate value is best determined with site

specific information.

5.5. ALGORITHMS FOR THE INCINERATOR STACK EMISSION SOURCE CATEGORY

Contaminants emitted from incinerator stacks are transported in air and deposit on the exposure site and on surrounding land. Chapter 6 describes the application of the Industrial Source Complex (ISC; EPA, 1986) air transport model to obtain vapor-phase air concentrations and deposition rates of particles at a specified distance from an example incinerator. These quantities are assumed to be given for purposes of discussion in this section; further discussion of the ISC model application is given in Chapter 6.

Table 5-1 summarized the fate and transport algorithms for this source category. Estimating soil concentrations based on particulate depositions follows a similar approach as estimating exposure site soil concentrations resulting from erosion of contaminated soil from off-site areas of contamination. Section 5.5.1. describes how soil concentrations are estimated given total (wet plus dry) deposition rates. Following now are bullet summaries for similarities and small refinements to algorithms previously discussed:

- **Surface water impacts:** Stack emission depositions over a watershed will result in an average watershed soil concentration, similar to the term C_w of Equation (5-1). The suspended and bottom sediment concentrations will be equal to this average concentration, which is a simplification of Equation (5-1) when all soil in the watershed is represented by one concentration. Section 5.5.1. shows how soil concentrations are estimated given deposition rates of contaminated particulates. Two key values are required for estimating average watershed soil and sediment concentration given deposition: a representative deposition rate and a representative depth of contaminant mixing. When using the ISC model in a site-specific application to estimate a representative deposition rate, isopleths of particulate deposition rate should be overlain on a watershed map. An average deposition rate can be estimated as an area-weighted average from this overlay. The depth of mixing might consider land use within the watershed. For a watershed predominantly undeveloped, a representative mixing depth of 1 cm, used for the "non-tilled" assumption for estimating mixing depth,

might be appropriate, or if the watershed is predominantly agricultural, a 20-cm mixing depth would be appropriate. For the example scenarios in Chapter 9 demonstrating the incinerator scenarios, the following assumptions were made for deposition rate and mixing depth. Table 6-18 and 6-19 show deposition rates at various distances up to 5 km from the incinerator stack for the example incinerator. Deposition rates at 5 km will be used to estimate the average watershed sediment concentration. The watershed is within a watershed with agricultural and non-agricultural settings. As such, the representative mixing depth will be 10 cm.

- **Particle-phase air-borne concentrations:** There is no air-borne particulate phase exposure. ISC model outputs include air-borne vapor phase concentrations (actually the hypothesis is made that what is called "vapor phase" is actually "aerosol phase" contamination, or contamination on particles less than 1 μm in diameter; see Chapter 6) and deposition rate of contaminants on particulates. Particulates are not assumed to remain air-borne for inhalation exposures. This, of course, is different from the assumption made for on-site and off-site soil source categories, where <10 μm size particulates are suspended due to wind erosion and are available for particulate inhalation exposures. That algorithm was characterized as conservative as the soil was described as containing an unlimited reservoir of particles available for wind erosion. Also, on-site and off-site soil contamination is assumed to be relatively deep, which makes wind erosion a more plausible phenomena for those source categories.
- **Biota concentrations:** The algorithm estimating concentration in fish tissue based on bottom sediment concentrations is the same as in previous source categories. Underground vegetable concentrations are a function of "tilled" soil concentrations. The soil concentration used for cattle soil ingestion is "untilled". Untilled and tilled depths of mixing are 1 and 20 cm, respectively, as in the off-site source category. Air-to-plant contributions are a function of ISC modeled vapor phase concentrations. ISC outputs also include the rate of particulate deposition for that portion of vegetation concentrations. Beef and milk concentrations are again a function of

vegetative and soil concentrations, diet fractions, and bioconcentration and bioavailability factors as described in Section 5.3.5.

5.5.1. Steady-State Soil Concentrations

Chapter 6 describes the use of the ISC Model to estimate the particulate phase deposition rate at the exposure site. This deposition rate, F, includes both dry and wet deposition, and is used to estimate the steady state soil concentrations. The deposition of contaminated particulates from the air is assumed to be somewhat analogous to the process of eroding contaminated soil from an off-site source depositing on an exposure site. Specifically, the following assumptions are also made: 1) only a thin layer of soil becomes contaminated, 2) this layer is either 1 cm or 20 cm deep, depending of surface activities, and 3) surface residues are assumed to dissipate with a half-life of 10 years corresponding to a first order decay rate of 0.0693 yr^{-1}. Considerations of upgradient erosion and exposure site soil removal are not made. Depositions occur over the exposure site and surrounding land area on an on-going basis. It might be said that upgradient soil concentrations are similar to exposure site concentrations at all times. Contributions of upgradient erosion to exposure site soil concentrations are matched by an equal amount of downgradient erosion removal of contaminants. The qualitative mass balance statement (similar to the one made above in Section 5.4.1, with ΔC equalling change in exposure site soil concentrations over time) can now be made as:

ΔC = (the incremental addition to C resulting from the change in deposition of stack emitted particulates) * (the incremental substraction of C resulting from degradation of residues at the exposure site)

This is mathematically stated as:

$$\frac{dC}{dt} = \frac{F}{M} - kC \qquad (5\text{-}37)$$

where:

C	=	the exposure site soil concentration, mg/kg
F	=	deposition rate of contaminant on particles, mg/yr
M	=	mass of soil at exposure site into which contaminant mixes, kg
k	=	first order dissipation rate constant, 1/yr.

The solution to this equation is:

$$C = \frac{F}{kM} \left(1 - e^{-kt} \right) \qquad (5\text{-}38)$$

which computes C as function of time, t. Similar to the assumption made above in Section 5.4.1., the steady state solution for C is simply F/kM. The deposition rates supplied by the ISC model are in units of g/m2-yr, so a conversion to mg/yr requires a multiplication by the land area of the exposure site and a multiplication of 1000 mg/g. Procedures to estimate M are given above in Section 5.4.1. The assumptions made for the mixing zone depth, d, described in Section 5.4.1, as also appropriate for particle deposition impact on soil for the stack emission source category.

5.6. ALGORITHMS FOR THE ASH DISPOSAL IN LANDFILLS SOURCE CATEGORY

The algorithms to estimate exposure media concentrations in this source category are the same as those described in Section 5.4, the off-site source category, with three exceptions. One is that the size, or potential size, of the landfill can be a function of the amount of ash disposed. Simple considerations for estimating this landfill size are described in Section 5.6.1. The second is that the transport, unloading, and spreading and compacting of the ash at the landfill are three processes that contribute to the particulate flux that gets transported by air to the exposure site. The transport process results in two distinct emissions. One is vehicular resuspension of roadway particulates; roads and other surfaces over which vehicles travel at a landfill can become contaminated over time by the contaminated soil of the landfill. Since ash is being transported in trucks to the ash

disposal site, another potential transport related emission occurs directly off the trucks. These fugitive ash emissions into air are modeled used "AP-42" emission factor equations (EPA, 1988), which are described in Section 5.6.2 below. The other component of the total particulate flux, described in Sections 5.3.3 above, is due to wind erosion. Discussions in Section 5.4 about application of the wind erosion algorithm to off-site soil contamination are also relevant for this section. The third difference between this source category and the off-site source category is in the estimation of concentrations of soil/ash in the landfill. Chapter 6 discusses the estimation of the dioxin-like compounds on fly ash. For the example setting, it will be assumed that the ash mixes with landfill soil and that final concentrations are half the concentrations for the ash.

5.6.1. Landfill Size

A size of a contaminated landfill can be assumed, or a simple estimate can be made considering the amount of ash transported to the landfill. An estimate of a landfill size receiving ash from the 2727 TPD incinerator is made in this section.

The first consideration is the amount of ash generated. It is typical to combine bottom and fly ash for disposal, and this is the assumption made for the example landfill in Chapter 9. Chapter 6 justifies a total fly ash generation for a plant of this capacity employing fabric filters with semi-dry alkaline scrubbers to be 30 metric tons/day, or 30000 kg/day, and estimates that bottom ash is ten times this quantity, or 300000 kg/day. Total daily ash generation is therefore 330000 kg/day. Next is required an assumption of ash bulk density. Since ash is comprised of finer particles than soil in general, a bulk density lower than that of soil is appropriate. The average bulk density of ash varies between .9 - 1.3 g/cm^3 (EPA, 1991a; soil bulk densities range from 1.2-1.8 g/cm^3). A reasonable assumption might therefore be an ash bulk density of 1.1 g/cm^3, or 1100 kg/m^3. With a daily amount of ash estimated, a bulk density, and an assumption on the total number of days over which disposal occurs, one can estimate the total landfill volume required. This volume can be converted to an area if an assumption on ultimate depth of disposal is made. The example assessments in Chapter 9 assumed a depth of 3 meters.

A thirty-year landfill life, with continuous ash disposal (365 d/yr, 30 yrs) to a depth of 3 m results in landfill sizes of 110 hectares for a 2727 TPD facility, as follows:

$$\frac{(\ 330{,}000\ kg/d)\quad (365\ d/yr)\quad (30\ yrs)}{(10{,}000\ m^2/ha)\quad (3m)\quad (1{,}100\ kg/m^3)} = 109.5\ hectares$$

It will be assumed that one-fourth of that size, or 27 hectares, is active at all time during the 30 year landfill lifetime. The other 83 hectares are either not filled or completed and covered. Other refinements might be considered. It may be unlikely for a single landfill to receive all the ash generated. Further, it may be unlikely that a 27 hectare area is uncovered at all times. The concentration of the dioxin-like compounds in the ash are described in Chapter 9 and were based on information provided in Chapter 6.

5.6.2. Fugitive Particulate Emissions at the Landfill

Four additional processes contribute to the total flux of particulates at the landfill. These include vehicular resuspension of roadway dust, the emission from trucks transporting the ash to the landfill, the unloading at the landfill, and the spreading and compacting operations.

5.6.2.1. *Vehicular Traffic Over Landfill Roadways*

A particulate flux is introduced in situations involving vehicular traffic over roadways at or near the soil contamination. Particulates on these roadways, paved or unpaved, can become contaminated over time by movement of particulates from the contaminated soil surface - by wind erosion, soil erosion, tire track-off, and so on. While the concentration of contaminants on these particulates is considerably less than the concentration on the contaminated soil itself (because of mixing with clean particulates, ongoing migration away from roadways, and so on), the flux of contaminant can be significant because of much higher particulate emission rates for vehicular traffic as compared to a wind erosion flux.

Estimates of particulate emission rates from paved and unpaved roadways can be estimated with AP-42 emission factor equations (EPA, 1988). The ash landfill example setting in Chapter 9 contains roadways that are unpaved. This is assumed because more emissions occur over unpaved roadways. Use of both emission factor equations for

roadway emissions is demonstrated in EPA (1991a). The equation for unpaved roadways is:

$$E_{up} = 1.7\ k_{unp} \left(\frac{s}{12}\right) \left(\frac{Vs}{48}\right) \left(\frac{W}{2.7}\right)^{0.7} \left(\frac{nw}{4}\right)^{0.5} \left(\frac{365-P}{365}\right) \qquad (5\text{-}39)$$

where:

E_{up}	=	emission flux for unpaved surfaces, kg/VKt (VKt equals vehicle kilometer traveled)
k_{unp}	=	particle size multiplier specific to the unpaved road emission flux equation, unitless
s	=	silt content of unpaved roadway, %
Vs	=	vehicle speed, km/hr
W	=	vehicle weight, kg
nw	=	number of wheels per vehicle, unitless
P	=	number of days with at least 0.254 mm (0.01 inch) precipitation per year, unitless.

Critical information from EPA (1988) includes different values for the particle size multiplier, k_{unp}, depending on the particulate size of concern, and ranges of other parameters (number of vehicle wheels, for example) for which the equations are relevant. Guidance for parameter selection specific to a landfill setting is provided below.

There have been a few key applications of AP-42 roadway emission factor equations (Kellermeyer and Zeimer, 1989; MRI, 1990) to evaluate the potential impact of suspended particulates in landfills. One in particular, conducted by EPA's Office of Solid Waste (MRI, 1990), not only applied the equations but also took a limited number of roadway samples to determine, among other factors: realistic measurements of the parameter s, roadway length measurements and vehicle counts to estimate the number of vehicle kilometers travelled, and other relevant information using a questionnaire survey given to the landfill disposal operator. The portion of their work used in the discussions below was conducted on 46 disposal sites accepting ash from municipal solid waste

(MSW) combustors. A portion of these disposal sites were industrial solid waste (ISW) landfills. Following now is brief guidance on these factors specific to a landfill setting.

- **Particle size multiplier, k_{unp}:** Values for unpaved roads were developed for particle sizes 2.5 μm and less, k_{unp} = 0.095, up to particle sizes 30 μm and less, k_{unp} = 0.80 (EPA, 1988). The value corresponding to particle sizes 10 μm and less, k_{unp} = 0.36, is the appropriate selection if the application is to estimate the flux of inhalable size particulates. If the objective is to estimate total flux for purposes such as upwind deposition onto soil, plants, or water, then the highest k_{unp} value, k_{unp} = 0.80, would be more appropriate. The value used in the example settings of Chapter 9 was 0.36, since the objective was to estimate the flux of particulates which are then transported downwind for inhalation exposures.
- **Silt content of unpaved roadways, s:** MRI (1990) took only two samples from haul routes on unpaved roads in two landfills, and obtained values of 6.7 and 20.1%. The mean of these values, 13.4%, is used in the example settings. EPA (1988) gives an appropriate range of 3.8-15.1% for this parameter.
- **Vehicle speed, Vs:** Based on their observations and questionnaires, MRI (1990) assumed a vehicle speed of 24 km/hr (15 mph) in their assessment, which was also used for this assessment. EPA (1988) gives an appropriate range for this parameter of 21-64 km/hr.
- **Vehicle weight, W:** A range of vehicle weights for vehicles disposing materials in industrial solid waste landfills was given as 14-40 x 10^3 kg in MRI (1990). The midpoint of this range, 27 x 10^3 kg, was used in Chapter 9. EPA (1988) gives an appropriate range for this parameter as 2.7-142 x 10^3 kg.
- **Number of wheels, nw:** MRI (1990) noted a range for this parameter of between 6 and 14 wheels. An average of 10 wheels per vehicle was used in Chapter 9. EPA (1988) gives an appropriate range for this parameter of 4-13 wheels.
- **Number of days with precipitation > 0.254 mm, P:** EPA (1988) presents a

map of the United States showing the number of days with greater than this amount of rain. Regional values range from: 150-170 days for the Northeast, 110-120 days for the Southeast, 70-110 days for the Mid and Southwest, 30-40 days for the Southern West Coast, and 90-150. The example setting in Chapter 9 assumed a value for P of 121, a mid-range value equalling 1/3 of a year.

After determining a value for E_{up}, the next step is to determine the total number of vehicle kilometers traveled per time period. This involves making assumptions for the length of impacted roadway (how much roadway has particulates that are contaminated), and the number of "vehicle passes" over that roadway during a specified time period. Again, based on the study of MRI (1990):

- **Length of impacted roadway:** The median haul route length from 46 landfills accepting MSW ash was 402 meters, although the mean length was 614 meters with the highest noted of 4828 meters. It may be reasonable to assume that only a portion of the haul route has contaminated particulates. One way to estimate an appropriate length would be to multiply the total haul route length (perhaps 400-600 m without better information) by a ratio of the area of contaminated landfill soil to the entire area of the landfill. If the entire landfill has contaminated soil, than the entire length of the haul route would have contaminated particulates. Another consideration is that higher capacity landfills have longer roadways than lower capacity landfills. The example settings in Chapter 9 assumes an impacted haul route of 100 m, with no particular justification.
- **Number of vehicle passes:** MRI (1990) determined that the number of vehicle "transactions", number of vehicles entering the 46 landfill sites in order to conduct business per day, excluding those specifically associated with ash disposal (it was not explained why ash transactions were excluded from their presented data; it was noted that vehicles other than those associated with ash comprised the large majority of transactions in industrial solid waste landfills), on the average was 116, although the median was 26. One observation they made was that the higher capacity landfills, the highest having >1600 vehicle transactions, had paved roads, while those

having lower capacities had unpaved roads. An assumption of 100 vehicle transactions over unpaved haul routes might be a reasonable assumption. A "vehicle pass" estimate could then equal vehicle transactions * 2, which would assume that each vehicle travels over the contaminated roadway going in and coming out. However, if one assumption is that only a portion of the roadway is contaminated (as discussed above), then it would be consistent to assume that not all vehicle transactions occur over the portion which is contaminated. A more reasonable assumption, perhaps, is that half the vehicles drive over impacted roadways; this would equate vehicle passes to vehicle transactions. The example settings in Chapter 9 assume 100 vehicle passes/day, or 50 vehicle transactions occurring per day driving over the impacted roadway twice.

There are two more considerations to note before summarizing the strategy for estimating dust flux from vehicular traffic over landfill roadways. First is how many days per year landfills operate. A continually operating landfill, 365 days/year, is not an unreasonable assumption (MRI, 1990). Second is the consideration of a "control efficiency factor." The AP-42 emission factor equations were developed with no controls in place. It is common for landfills to use dust suppression measures on their roadways, particularly for unpaved roadways where visibility can be an issue. Common means of dust suppression in landfills are wetting and chemical dust suppression (MRI, 1990). An assumption of a 90% reduction in total emissions possible because of dust suppression measures is probably reasonable, given the observations made in MRI (1990). Landfills operating 365 days/year and a control efficiency factor of 0.10 (90% efficiency) were assumed for the example settings of Chapter 9.

In summary, the strategy for estimating roadway vehicular dust emissions is as follows: estimate E_{up} from Equation (5-47), estimate total vehicular kilometers per day considering length of impacted roadway and vehicle passes per day, multiply E_{up} by total kilometers passes per day to get total dust flux per day (a good steady state assumption given 365 operating days per year), and finally reduce this total by a control efficiency factor. The resulting dust flux will be called E_v, and will be in units of kg/day.

The total contaminant flux is then equal to E_v times a concentration of contaminant on the dust, which can be assumed to be a linear function of the contaminant

concentration in the landfill soil:

$$VE = 1.2 \times 10^{-11} \; C_s \; CDF \; E_v \qquad (5\text{-}40)$$

where:

VE	=	total contaminant flux due to vehicular traffic on landfill roadways, g/sec
C_s	=	contaminant concentration on landfill soil, ppb or μg/kg
CDF	=	contaminant dilution factor, unitless
E_v	=	roadway dust emission rate, kg/day
1.2×10^{-11}	=	converts μg/day to g/sec.

Again very little information is available on CDF. However, MRI (1990) did take some measurements which are applicable. Briefly, they took particulate samples from landfill haul routes while at the same time taking samples of incinerator ash being delivered for disposal the same day. Each paired sample (roadway particulate and ash), were measured for four metals: As, Cd, Cr, and Pb. Several paired samples were taken on both paved and unpaved haul routes. Ratios were then generated for roadway particulate metal concentrations over ash metal concentrations. Results were: As - paved and unpaved ratios were similar and consistently near 0.1 (roadside particulate concentrations of As were 10% of ash concentrations of As), Cd - paved and unpaved ratios were similar and ranged between 0.0 and 0.4, Cr - paved ratios ranged from 0.3 to 0.6, while unpaved had a wide range of 0.3 to 2.0, Pb - paved and unpaved ratios were similar between 0.0 and 0.2. For analogous situations - daily deliveries of contaminated ash - one might assume a CDF in the 0.1-0.2 range. A value of 0.1 for CDF was used in the ash landfill example setting in Chapter 9.

5.6.2.2. *Fugitive Emissions from Trucks*

The following equation for estimating emissions from open storage piles has been suggested for use in estimating fugitive emissions from trucks in transit (EPA, 1991a; the

emission factor equation from EPA, 1985a):

$$E_t = 1.9 \left(\frac{s}{1.5}\right) \left(\frac{f}{15}\right) \left(\frac{365-P}{365}\right) \qquad (5\text{-}41)$$

where:

E_t = particulates emitted from trucks in transit, kg/day/hectare

s = silt content material of ash, %

P = number of days with >0.25 mm precipitation per year

f = percentage of time that the unobstructed wind speed exceeds 5.4 m/s.

Brief guidance on these parameters is:

- **Silt content of ash, s:** EPA (1991) lists the particle size distribution of combined ash, and lists 6.7% as less than 63 μm. The USDA describes silt-size particles to be between .002 and .05 mm (.05 mm = 50 μm; Brady, 1984), with clay-size particles less than .002. Therefore, the silt content of ash will be assumed to be 6.7% for the example setting in Chapter 9.
- **Number of days with precipitation > .25 mm per year, P:** Background on this parameter was provided in Section 5.6.2.1 above, and as noted there, an assumption of 121 days was a reasonable mid-continent assumption.
- **Percentage of time unobstructed wind speed exceeds 5.4 m/sec:** Since this speed equals just over 12 miles per hour, it is probably reasonable to assume that trucks transporting the ash travel over this speed 100% of the time. Hence, a value of 100% is assumed for this parameter.

As this equation was developed for stationary piles rather than moving trucks, a few additional steps are required.

Step 1. Estimate daily number of truck loads transported. This requires an estimate of truck capacity, which is then divided into the total load delivered per day. The average volume of a truck delivering ash has been estimated as 40 yd^3 (Wells, et al., 1988). The

example setting in Chapter 9 used a volume of 40 m^3. With a density of 1100 kg/m^3, the average truck can haul 44,000 kg/trip. The 2727 TPD incinerator generating 330,000 kg/day might therefore send out 8 trucks per day.

Step 2. Estimate truck surface area. Wells, et al. (1988) assumed a depth of 1 yd for their estimate of 40 yd^3. The example setting in Chapter 9 had an assumption of 40 m^2 surface area.

Step 3. Estimate total daily emission of ash. This first requires a conversion of E_t units of kg/day/hectare to more reasonable units for this application, which might be $kg/min/m^2$; that conversion factor is 6.94. Then, an assumption of the length of roadway over which this emission occurs needs to be made. For simplicity, Chapter 9 assumed that this emission only occurred within the landfill. Section 5.6.2.1 described the survey of 46 landfills accepting ash from municipal solid waste combustor facilities; the mean haul route was 614 meters, and the setting in Chapter 9 used a length of 600 meters. If the truck travels at 15 miles/hr, or 390 m/min, than it takes 1.5 minutes of travel within the landfill. The emission rate in kg/day for the ash landfill can now be estimated as 8 trips/day * 40 m^2 * 1.5 min * 6.94 $(kg/min/m^2)/(kg/day/ha)$ * E_t kg/day/ha (E_t as calculated with Equation (5-41) above).

Fugitive emissions from trucks carrying ash can be minimized by wetting the ash or the use of truck covers. The application of this equation to a situation where the ash is properly wetted prior to transport would show that no fugitive emissions of dust are expected to occur (i.e., a proper assumption would be that $P=365$, hence $E_t=0$). Another management control is the use of tarpaulins or other truck coverings. The above equation was developed for wind erosion estimation without covering. Therefore, a control efficiency is appropriate to use with this equation. The example setting in Chapter 9 assumed a 90% efficiency; i.e., multiply the final daily emission from the above exercises by 0.10.

5.6.2.3. *Emissions from Unloading*

The unloading operations at the disposal site may result in the release of fugitive dust. The following emission factor equation provides emission factors for kilograms of

particulate emitted per megagram (metric ton, or 1000 kg) of soil loaded and unloaded (EPA, 1988):

$$E_{lu} = 0.0016\ k_{unl}\ \left(\frac{U_m}{2.2}\right)^{1.3}\ \left(\frac{M}{2}\right)^{-1.4} \qquad (5\text{-}42)$$

where:

E_{lu}	=	emission factor for loading and unloading, kg fugitive dust/MT ash
k_{unl}	=	particle size multiplier, dimensionless
U_m	=	wind speed, m/s
M	=	material moisture content, %.

Guidance on these parameters is:

- **Particle size multiplier, k_{unl}:** Values were supplied in EPA (1988) for sizes ranging from <2.5 μm, k_{unl} = 0.11, to sizes <30 μm, k_{unl} = 0.74. The value corresponding to particle sizes 10 μm and less, k_{unl} = 0.35, is the appropriate selection if the application is to estimate the flux of inhalable size particulates, as it is for the example setting in Chapter 9.
- **Wind speed, U_m:** A wind speed of 4 m/sec is used for the wind erosion algorithm (Section 5.4.1), and was justified as being an average value. For the loading and unloading emission factor, the range of wind speed for which is assumed to be generally valid is 0.6 to 6.7 m/s (EPA, 1988).
- **Moisture content, M:** The moisture content of ash after exiting the quench tank at combustor facilities can be as high as 40% (EPA, 1991a), although ash would not be transported with this high a moisture content. Furthermore, fugitive emissions should not be considered with higher moisture content ash. This emission factor equation was developed for a range of moisture contents of 0.25-4.8% (EPA, 1988). The conservative assumption of a moisture content of 0.25% was made for the example setting.

The Elu factor still needs to be multiplied by the number of MT to obtain total emissions in kg/day. The 2727 TPD facility produces 330 MT/day ash, so the multiplication factor is 330.

5.6.2.4. *Emissions from Spreading and Compacting*

Fugitive dust emissions from spreading and compacting ash at disposal sites have been estimated in more than one way, although the different ways found in the literature to estimate emissions from these processes are all based on AP-42 emission factor equations. The differences were due to assumptions as to which processes the spreading and compacting of ash was most analogous to, and used the AP-42 factor developed for that process. MRI (1990) used an AP-42 emission factor developed for dozer moving of overburden in western surface coal mines. Kellermeyer and Ziemer (1989) assumed that the spreading and compaction of ash was analogous to vehicular transport on unpaved surfaces, and used the emission factor for that process. A third possible assumption, and the one used for this assessment, is that the processes of spreading and compacting are analogous to agricultural tillage. That emission factor equation for agricultural tillage is (EPA, 1988):

$$E_{at} = 5.38 \; k_{at} \; s^{0.6} \qquad (5\text{-}43)$$

where:

E_{at} = emission factor for agricultural tillage, kg/ha

k_{at} = particle size multiplier, dimensionless

s = silt content, %.

Guidance for these parameters is:

- **Particle size multiplier, k_{at}:** Values were supplied in EPA (1988) for sizes ranging from <2.5 μm, k_{at} = 0.10, to sizes <30 μm, k_{at} = 0.33, with a final size listed as "total suspended sediment" and given a kat of 1.00. The value corresponding to particle sizes 10 μm and less, k_{at} = 0.21, is the

appropriate selection (and the selection made for the example setting in Chapter 9) if the application is to estimate the flux of inhalable size particulates.

- **Silt content, s:** This equation was developed for silt contents between 1.7 and 88.0%. As noted above, the silt content for the example setting for ash was 6.7%.

A final consideration for application of the agricultural tillage emission factor equation is the number of hectares per day over which spreading and compacting occur. A simple way to estimate this area is to assume a depth of spreading, and calculate area of coverage based on total daily amount spread. The example setting in Chapter 9 assumed a .05 m (2 in) depth of spreading each day. Eight full truckloads of 40 m^3 were emptied each trip. At a depth of 0.05 m, 6400 m^2 (320/.05) or .64 ha were required. The total daily emission from spreading and compacting can now be estimated as: 1 operation/day * .64 ha * E_{at} kg/ha.

5.6.2.5. *Total Particulate Emissions from Ash Landfills*

A few details must be remembered for these emission factor equations. First, all total particulate emissions need to eventually be in appropriate units of g/cm^2-sec, which are unit emissions over an area where these emissions occur. That area is assumed to be the area of the ash landfill, the parameter termed ACS which earlier was shown to be equal to 27 ha.

Emissions off trucks, E_t, emissions from unloading, E_{lu}, and emissions from spreading and compacting, E_{at}, were uniformly converted to kg/day, so these results need to be further multiplied by:

$$Emissions\ (g/cm^2\ day) = \frac{E\ (kg/day)\ (1000\ g/kg)\ (1\ day/8640\ sec)}{A_{sc}\ (ha)\ (10^8\ cm^2/ha)}$$

where:

E	=	emission factors E_t, E_{lu}, E_{at} (kg/ha)
A_{sc}	=	area of soil contamination (ha)

The final emissions then need to be multiplied by a unitless (g/g) concentration of the dioxin-like compound in the ash to get contaminant emission flux appropriate for air transport modeling. Multiplication by a unitless concentration is necessary so that the total contaminant emissions are still in g/cm^2-sec.

A fourth contaminant flux, that due to resuspension of roadway particulates, was described above as VE (Section 5.6.2.1) and was already converted to units of g/sec. This was done in order to facilitate the discussion on the contaminant dilution factor (CDF). To bring this flux in line with the others, it must be divided by the total landfill area in cm^2.

The final contribution to particulate emissions is that due to wind erosion. The wind erosion flux is given as E_e, estimated by Equation (5-14) with refinements for bare, off-site soil contamination as discussed in the introductory portion of Section 5.4. This flux term is solved for in units of g/sec, so it as well must be divided by the total landfill area in cm^2.

Total flux of contaminant on particulates can now be given as $E_e + VE + E_t + E_{lu} + E_{at}$, all in units of g/cm^2-sec. The final step is to set this total equal to flux factor, FLUX, of Equation (5-34). That equation now will estimate far-field dispersion and resulting exposure site air-borne particulate concentrations.

REFERENCES FOR CHAPTER 5

Baes, C.F.; Sharp, R.; Sjoreen, A.; Shor, R. (1984) A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture. Oak Ridge National Laboratory, ORNL-5786.

Bacci, E.; Calamari, D.; Gaggi, C.; Vighi, M. (1990) Bioconcentration of Organic Chemical Vapors in Plant Leaves: Experimental Measurements and Correlation. Environ. Sci. Technol. 24: 885-889.

Bacci, E.; M.J. Cerejeira; C. Gaggi; G. Chemello; D. Calamari; M. Vighi (1992) Chlorinated Dioxins: Volatilization from Soils and Bioconcentration in Plant Leaves. Bulletin of Environmental Contamination and Toxicology (in press).

BOS (1990) Battelle Ocean Services. New Bedford Harbor Modeling Program Final Report. U.S. Environmental Protection Agency, Boston, MA.

Brady, N.C. (1984) The Nature and Properties of Soils. Ninth Edition. New York, NY: Macmillan.

Briggs, G.G., R.H. Bromilow, and A.A. Evans (1982) Relationships between lipophilicity and root uptake and translocation of non-ionised chemicals by barley. Pesticide Science 13: 495-504.

Carslaw, H.S.; Jaeger, J.C. (1959) Conduction of Heat in Solids. Oxford University Press, Oxford, England.

Columbo, J.C., M.F. Khahl, M. Arnac, A.C. Horth (1990). Distribution of Chlorinated Pesticides and Individual Polychlorinated Biphenyls in Biotic and Abiotic Compartments of the rio de la Plata, Argentina, Environmental Science and Technology 24: 498-505.

Connett, P.; Webster, T. (1987) An Estimation of the Relative Human Exposure to 2,3,7,8-TCDD Emissions Via Inhalation and Ingestion of Cow's Milk. Chemosphere 16 (8/9): 2079-2084.

Cook, P.M.; Duehl, D.W.; Walker, M.K.; Peterson, R.E. (1991) Bioaccumulation and Toxicity of TCDD and Related Compounds in Aquatic Ecosystems. p. 143-168 in Gallo, M.A., R.J. Scheuplein, and K.A. Van Der Heijden, eds., Banbury Report 35: Biological Basis for Risk Assessment of Dioxins and Related Compounds. Cold Spring Harbor Laboratory Press 0-87969-235-9/91.

Forrester, K.E. (1988) Comparison of Municipal Solid Waste Leachate to MSW Ash Residual Leachate using a Risk Algorithm. Wheelabrator Environmental Systems, Inc., Danvers, MA.

Foster, G.R.; Hakonson, T.E. (1987) Erosion Losses of Fallout Plutonium. In: Dynamics of Transuranics and Other Radionuclides in Natural Environments. Howard, W.A., and R. Fuller, eds. NTIS-87014456. Springfield, VA.

Foth, H.D. (1978) Fundamentals of Soil Science. 6th ed. New York: John Wiley and Sons.

Gobas, F.A.P.C. (1990) Bioaccumulation of some polychlorinated dibenzo-p-dioxins and octachlorodibenzofuran in the guppy (Poecilia reticulata). Chemosphere 20: 495.

Fries, G.F.; Paustenbach, D.J. (1990) Evaluation of Potential Transmission of 2,3,7,8-Tetrachlorodibenzo-p-dioxin-Contaminated Incinerator Emissions to Humans Via Foods. J. Toxicol. Environ. Health 29: 1-43.

Gillette, D.A. (1981) Production of dust that may be carried great distances. In: T. Pewe, ed. Desert Dust: Origin, Characteristics, and Effect on Man. Geological Society of America Special Paper 186, pp. 11-26.

Hwang, S.T. (1987) Methods for estimating on-site ambient air concentrations at disposal sites. Nucl. Chem. Waste Management 7: 95-98.

Hwang, S.T.; Falco, J.W.; Nauman, C.H. (1986) Development of Advisory Levels for Polychlorinated Biphenyls (PCBs) Cleanup. U.S. Environmental Protection Agency, Exposure Assessment Group, Office of Research and Development. EPA/600/6-86/002.

Jackson, D.R., M.H. Roulier, H.M. Grotta, S.W. Rust, and J.S. Warner (1986) Solubility of 2,3,7,8-TCDD in Contaminated Soils. pp. 185-200 in Rappe, C., G. Choudhary, and L.H. Keith (eds.), Chlorinated Dioxins and Dibenzofurans in Perspective. Lewis Publishers, Inc.

Karickhoff, S.W., D.S. Brown, and T.A. Scott (1979) Soprtion of hydrophobic pollutants on natural sediments. Water Research 13: 241-248.

Kellermeyer, D.A.; Ziemer, S.E. (1989) What are the health risks of ash monofills? Solid Waste and Power III(4): 40-44.

Kjeller, L.O., S.E. Kulp, S. Bergek, M. Bostrom, P.A. Bergquist, C.Rappe, B.Jonsson, D.de Wit, B. Jansson, M. Olson (1990) Levels and Possible Sources of PCDD/PCDF in Sediment and Pike Samples from Swedish Lakes and Rivers (Part One). Chemosphere 20: 1489-1496.

Knisel, W.G., ed. (1980) CREAMS: A Field Scale Model for Chemicals, Runoff, and Erosion from Agricultural Management Systems. US Department of Agriculture, Conservation Research Report No. 26, Tuscon, Arizona.

Kuehl, D.W.; Cook, P.M.; Batterman, A.R.; Lothenbach, D.B.; Butterworth, B.C. (1987) Bioavailability of polychlorinated dibenzo-p-dioxins and dibenzofurans from contaminated Wisconsin river sediment to carp. Chemosphere 16: 667.

Lake, J.L., N.I. Rubinstein, H. Lee II, C.A. Lake, J. Heltshe and S. Pavignano. 1990. Equilibrium partitioning and bioaccumulation of sediment-associated contaminants by infaunal organisms. Environ. Toxicol. Chem. 9: 1095-1106.

Liss, P.S.; Slater, P.G. (1974) Flux of gases across the air-sea interface. Nature 247: 181-184.

Lyman, W.J.; Reehl, W.F.; Rosenblatt, D.H. (1982) Handbook of Chemical Property Estimation Methods. New York: Mcgraw-Hill.

McCrady, J.K., McFarlane, C.; Gander, L.K. The transport and fate of 2,3,7,8-TCDD in soybean and corn. Chemosphere 21: 359-376.

Mills, W.B.; Porcella, D.B.; Ungs, M.J.; Gherini, S.A.; Summers, K.V.; Mok, L.; Rupp, G.L.; Bowie, G.L.; Haith, D.A. (1985) Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water - Parts I and II. EPA/600/6-85/002a&b.

MRI, 1990. Special Management Standards for Municipal Waste Combustion (MWC) Ash. Submitted by: Midwest Research Institute, 425 Volker Boulevard, Kansas City, MO 64110-2299, to US EPA, Municipal Solid Waste Program. EPA Contract No. 68-01-7287. June 29, 1990.

Muir, D.C.G.; Yarechewski, A.L.; Webster, G.R.B. (1986) Bioconcentration for four chlorinated dioxins by rainbow trout and fathead minnows. Environ. Toxicol. Chem. 5: 261.

Niimi,A.J.; B.G. Oliver. (1989) Distribution of Polychlorinated Biphenyl Congeners and Other Halocarbons in Whole Fish and Muscle from Lake Ontario Salmonids. Environmental Science and Technology 23: 83-88.

Novotny, V.; Chesters, G. (1981) Handbook of Nonpoint Pollution, Sources and Management. New York: Van Nostrant Reinhold.

Office of Technology Assessment. (1989) Facing America's Trash: What's Next for Municipal Solid Waste? U.S. Congress, Office of Technology Assessment. Washington, D.C. OTA-0-424.

Parkerton, T.F., J.P. Connolly, R.V. Thomann, C.G. Uchrin. (1992) Do Aquatic Effects or Human Health Endpoints Govern Sediment Quality Criteria for Non-ionic Organic Chemicals? Regulatory Implications of the Equilibrium Partitioning Paradigm. To appear in: Environmental Toxicology and Chemistry.

Parkerton, T.F. (1991) Development of a Generic Bioenergetics-Based Model for Predicting the Bioaccumulation of Persistent Sediment-Associated Contaminants - Final Report, New Jersey Department of Environmental Protection, Trenton, NJ.

Riederer, M. (1990) Estimating Partitioning and Transport of Organic Chemicals in the Foliage/Atmosphere System: Discussion of a Fugacity-Based Model. Environmental Science and Technology 24: 829-837.

SAIC. (1986) Science Applications International Corp. Conversion factors for use in estimating environmental transport of contaminated soil from unregulated disposal sites. Prepared for the U.S. Environmental Protection Agency, Office of Solid Waste, Washington, DC, under EPA contract 68-01-7624.

Schroy, J.M.; Hileman, F.D.; Cheng, S.C. (1985) Physical/chemical properties of 2,3,7,8-TCDD. Chemosphere 14: 877-880.

Shor, R.W.; Baes, F.F.; Sharp, R.D. (1982) Agricultural Production in the United States by County: A Compilation of Information from the 1974 Census of Agriculture for Use in Terrestrial Food-Chain Transport and Assessment Models. ORNL-5768, Oak Ridge National Laboratory, Oak Ridge, Tenn.

Seinfeld, J.H. (1986) Atmospheric Chemistry and Physics of Air Pollution. John Wiley and Sons, New York.

Stevens, J.B.; Gerbec, E.N. (1988) Dioxin in the agricultural food chain. Risk Analysis 8(3): 329-335.

Swackhammer, D.L.; R.A. Hites (1988). Occurrence and Bioaccumulation of Organochlorines in Fishes from Siskiwit Lake, Isle Royale, Lake Superior. Environmental Science and Technology 22: 543-548.

Swackhammer, D.L., B.D. McVeety, R.A. Hites (1988). Deposition and Evaporation of Polychlorinated Congeners to and from Siskiwit Lake, Isle Royale, Lake Superior. Environmental Science and Technology 22: 664-672.

Thomann, R.V., J.P. Connoly and T.F. Parkerton. 1992. An equilibrium model of organic chemical accumulation in aquatic foodwebs with sediment interaction. Environmental Toxicology and Chemistry. In Press.

Travis, C.C.; Arms, A.D. (1988) Bioconcentration of organics in beef, milk, and vegetation. Environmental Science and Technology 22(3): 271-274.

Travis, C.C.; Hattermeyer-Frey, H.A. (1991) Human exposure to dioxin. Sci. Total Environ. 104: 97-127.

Travis, C.C.; Hattermeyer-Frey, H.A. (1988) Uptake of Organics by Aerial Plant Parts: A Call for Research. Chemosphere 17: 277-283.

Turner, D.B. (1970) Workbook of Atmospheric dispersion estimates. PHS publication no. 999-AP-26 (NTIS PB 191482), EPA, Research Triangle Park, North Carolina.

U.S. Department of Agriculture. (1974) United States Department of Agriculture. Universal Soil Loss Equation. Agronomy Technical note no. 32. Portland, Oregon. U.S. Soil Conservation Service. West Technical Service Center.

U.S. Environmental Protection Agency. (1977) United States Environmental Protection Agency. Water Quality Assessment: A Screening Method for Nondesignated 208 Areas. US EPA # EPA-600/9-77-023. J. Falco, Project Officer, US EPA, Athens ERL, Athens, GA 30605. Authors: Zison, S.W., K.F. Have, and W.B. Mills; Tetra Tech, Inc., Lafeyette, California.

U.S. Environmental Protection Agency. (1985a) Compilation of Air Pollutant Emission Factors. Fourth Edition. U.S. Environmental Protection Agency, Office of Air Quality, Planning and Standards. Research Triangle Park, NC.

U.S. Environmental Protection Agency. (1985b) Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites. Environmental Protection Agency, Office of Health and Environmental Assessment, Office of Research and Development. EPA/600/8-85/002, February, 1985.

U.S. Environmental Protection Agency. (1986) Industrial Source Complex (ISC) Dispersion Model User's Guide. Second Edition Vol. 1. Office of Air Quality Planning and Standards, Research Triangle Park, NC, EPA-450/4-86-005a. NTIS PB86-234254.

U.S. Environmental Protection Agency. (1988) Compilation of Air Pollutant Emission Factors. Volume 1. Stationary Point and Area Sources. Fourth Edition. Supplement B. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. Research Triangle Park, NC. September, 1988.

U.S. Environmental Protection Agency. (1990a) Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions. U.S. Environmental Protection Agency, Office of Health and Environmental Assessment. Washington, D.C. EPA/600/6-90/003. January, 1990.

U.S. Environmental Protection Agency. (1990b) Lake Ontario TCDD Bioaccumulation Study Final Report. Cooperative study including US EPA, New York State Department of Environmental Conservation, New York State Department of Health, and Occidental Chemical Corporation. May 1990.

U.S. Environmental Protection Agency. (1991a) Methodology for Assessing Environmental Releases of and Exposure to Municipal Solid Waste Combustor Residuals. Exposure Assessment Group, Office of Health and Environmental Assessment, EPA, Washington, D.C. EPA/600/8-91/031. April, 1991.

U.S. Environmental Protection Agency. (1991b) Methodology for Assessing Exposure and Risk to PCBs in Soils and Sediments - A Screening Approach. Exposure Assessment Group, Office of Health and Environmental Assessment, EPA, Washington, D.C. (August 1991 draft report).

Vanoni, V.A. (ed.). (1975) Sedimentation Engineering. American Society of Civil Engineers, New York, NY.

Wells, R.C.; Machaver, R.J.; Allen, W.R. (1988) Fugitive Particulate Emission Estimates for Incinerator Ash Handling. Camp Dresser & McKee, Inc, Edison, NJ.

Wischmeier, W.H. (1972) Estimating the cover and management factor on undisturbed areas. Proceedings of the USDA Sediment Yield Workshop. Oxford, MS: U.S. Department of Agriculture.

Wischmeier, W.H.; Smith, D.D. (1965) Predicting Rainfall-Erosion Losses from Cropland East of the Rocky Mountains, Agriculture Handbook 282. U.S. Department of Agriculture, Agriculture Research Service.

Young, A.L. (1983) Long term studies on the persistence and movement of TCDD in a national ecosystem. In: Tucker, et al., eds. Human and environmental risks of chlorinated dioxins and related compounds. New York, NY: Plenum Publishing.

6. MUNICIPAL SOLID WASTE INCINERATION

6.1. INTRODUCTION

The combustion of municipal solid waste (MSW) releases potentially harmful pollutants to the air from the smokestack of the incinerator. Of particular concern is the potential human health and environmental effects of these emissions during the period of facility operation. The rapid growth in this stationary combustion source category, coupled with the *de novo* synthesis and stack emissions of PCDDs, PCDFs and related compounds, provides an opportunity to evaluate multiple pathways of human exposure of the emissions. The following chapter provides procedures for estimating the emission rates of dioxin-like compounds from municipal waste incinerators and the associated deposition rates and air concentrations as a function of distance from the incinerator. Once the air concentrations and deposition rates have been established, the reader can then use the procedures provided in other Chapters of this document to determine possible exposure pathways and levels of exposure. These procedures are illustrated using a hypothetical MSW incinerator arbitrarily located in Tampa, Florida. This example exposure scenario is not intended as a regulatory exposure/risk assessment, since it involves a theoretical incinerator, but provides an example of an exposure assessment of the emissions of dioxin-like compounds that can be followed and applied in a similar manner to the evaluation of actual emissions from a planned or operational facility. This Chapter reviews the current MSW incineration technology in the U.S.; discusses the theoretical basis for explaining why dioxin-like compounds are emitted during MSW combustion; describes the hypothetical MSW incinerator selected for analysis of direct and indirect exposures to dioxin-like compounds emitted from the stack; summarizes current emissions data of specific congeners of PCDD, PCDF, and coplanar PCBs appropriate to the hypothetical MSW incinerator: describes the air dispersion and surface deposition modeling of the chemicals; and gives the results appropriate for the analysis of direct and indirect human exposure of ambient air concentrations and surface deposition flux of the dioxin-like congeners.

The per capita generation of municipal solid waste in the United States is approximately 1.8 kilograms (kg) per day (US EPA, 1990). This equates to a 50% increase from the 1.2 kg/person in 1960 (U.S. EPA, 1989). As a result, nearly 164 million

metric tons of MSW are generated each year, and the rate is continuing to increase. Currently the breakdown of methods for disposing of the MSW waste stream in the U.S. is as follows: 73% is landfilled; 13% is recycled; 14% is incinerated (U.S. EPA,1990). This is in marked contrast to the breakdown of 1985: 84% MSW was landfilled; 12% was recycled, and 4% was incinerated (U.S. EPA, 1987a).

Faced with increasing problems of land disposal, many communities are constructing incinerators as the primary method for waste disposal. These systems are designed for the recovery of heat from refuse combustion in the form of steam or hot water that can be used as an energy source to generate electricity, to supplement energy demands of industry, and for use in district heating of residential and commercial properties. Although the combustion of MSW does not eliminate the need for landfilling, since residual ash must be disposed of, it does reduce the volume of waste requiring landfilling by 70 to 90 percent, and therefore extends the operational life of existing landfills. Without a reduction in the volume of waste that ultimately is landfilled, some urban areas will soon reach the design capacity of existing landfills and be compelled to select some method of MSW disposal or face a possible waste disposal crisis.

Significant growth in the population of municipal incinerators has occurred in the U.S. since 1985. In 1985 approximately 43,240 metric tons per day (MT/d) of MSW was incinerated in 99 facilities nationwide (U.S. EPA, 1987a). In 1989, approximately 63,670 MT/d of MSW were disposed of in an estimated 200 municipal incineration facilities (U.S.EPA,1989). This represents a 200 % increase in the number of facilities, and a 47 % increase in tons of MSW incinerated within a span of four years. If trends continue, then it is estimated that by the year 2000, a total of 400 incineration facilities may be incinerating 33% of the MSW waste stream per day (U.S. EPA, 1987a).

6.2. OVERVIEW OF PRINCIPAL MUNICIPAL INCINERATION TECHNOLOGIES IN THE UNITED STATES

Municipal incinerators operating in the U.S. can be classified into four general design categories: mass burn, modular, refuse-derive fuel (RDF), and fluidized-bed (U.S. EPA, 1987a; OTA,1989; Cleverly,1991). The first type is called mass burn, because the waste is combusted without any preprocessing other than removal of items too large to go through the feed system. In a typical mass burn combustor, refuse is placed on a grate

that moves through the combustion chamber. Combustion air in excess of stoichiometric amounts is supplied both below (underfire air) and above (overfire air) the grate. Mass burn incinerators are usually field-erected and can burn 45 to 909 metric tons of MSW per day (MT/d) per incineration unit. Many mass burn facilities have two or more incineration units comprising the incineration facility with a total refuse combustion capacity ranging from 45 to 2727 MT/d.

Modular incinerators also burn waste without preprocessing, but they are typically factory fabricated and shipped in reassembled sections to the intended site. Modular incinerators generally incinerate 4.5 to 91 metric tons of refuse per day per incineration unit. One of the most common types of modular incinerators is the starved air or controlled air type incorporating two combustion chambers. Air is supplied to the primary chamber at sub-stoichiometric levels. The incomplete combustion products entrained in the combustion gases from the primary combustion chamber pass into the secondary combustion chamber where excess air is added, and combustion is completed. Another type of modular combustor, functionally similar to larger mass burn units, uses excess air in the primary chamber; no additional air is added in the secondary chamber. Modular facilities are also comprised of one or more incineration units having a total combustion capacity of 4.5 to 270 MT/d of MSW.

The third general type of MSW incinerator is designed to burn RDF. RDF is a term describing MSW which has been processed into a physical form that theoretically enhances combustibility. RDF is a fuel that has been prepared by shredding, sorting, separating, and trommeling MSW to reduce the noncombustible content (Cleverly, 1991). It may vary in physical form from shredded waste having a uniform size to a finely divided fuel suitable for co-firing with pulverized coal. Some cases of RDF processing involves magnetic separation of ferrous metals and reduction of other noncombustible components of the refuse, such as aluminum and glass products. RDF facilities have a total combustion capacity of 227 to 3636 MT/d.

The fourth type of MSW incineration technology is the fluidized-bed design. In this design, the waste burns in a turbulent bed of non-combustible material, usually sand. The MSW may be fed into the incinerator either as unprocessed waste, or as a form of RDF. There are two basic design concepts to the fluidized-bed technology: a bubbling-bed incineration unit, and a circulating-bed incineration unit (U.S. EPA, 1989). Fluidized-bed

MSW incinerator units are typically designed to burn from 182 to 455 MT/d per unit, and the total incineration facility capacity by combining one or more incineration units is in the range of 184 to 920 MT/d.

Existing MSW incinerators can be further classified as having or not having heat recovery boilers. All RDF and fluidized bed incinerators have boilers to recover heat. Over 66% of existing mass burn incinerators are equipped with heat recovery boilers. Modular units are usually absent heat recovery capability because of size limitations.

There is a distinct distribution of incinerator technologies (U.S. EPA, 1987a). This distribution can be determined as a percentage of the total mass of MSW that is incinerated annually in the U.S. On this basis, the approximate distribution by technology is as follows: 58% mass burn; 24% RDF; 6% modular; 12% fluidized-bed (U.S. EPA, 1989). The majority of existing MSW incinerators currently operating in the U.S. are located in the New England and mid-Atlantic states (U.S. EPA, 1987a). The pattern of distribution seems to follow population density with those areas of the U.S. with the highest population densities having the most numbers of MSW incinerators. With shifting trend in population to the "sunbelt" states, additional numbers of incinerators are being planned or under construction in Florida, Texas, and southern California. The projected growth in incineration facilities indicates that mass burn heat recovery incinerators will continue to dominate the market by the year 2000, accounting for between 60% and 70% of all future facility technologies (U.S. EPA,1987a).

6.3. THE THEORETICAL MECHANISMS GIVING RISE TO THE AIR EMISSION OF DIOXIN-LIKE COMPOUNDS DURING MSW INCINERATION

MSW is used as a source of fuel for the incineration process, and as such, it chemically reacts with oxygen to cause an exothermic reaction to release heat. Flames are really the apparent indication of this exothermic reaction, but the process of combustion is actually a complex reaction, interaction and re-reaction of the basic constituent chemical compounds and elements in the fuel. The molecules found in MSW are complex and generally have too many atoms for combustion to proceed in any predetermined manner (Gardiner,1982). It is difficult to predict how constituents present in the fuel will break apart, reform new compounds, disassociate into free radicals and recombine into new compounds under the influence of oxygen, turbulent mixing and heat. The breakdown of

molecules composing MSW and the formation of combustion products proceed in long sequences of reactions, and each sequence involves a rearrangement of chemical bonds. Heat activates the collision of molecules, and energy increases molecular collisions disassociating molecules into fragmentary atoms that react with other molecules or radicals yielding new intermediate combustion products. The intermediate products collide, become dissociated into free radicals which sets up the process of creating intermediates. The repetition of the same molecular events is termed chain reactions, and the atoms or radicals that cause the chain reactions are termed reaction centers. It may require a relatively small number of chain centers to lead to a large occurrence of chemical reactions. The randomness of these molecular events makes it difficult, if not improbable, to predict chemical reactions and the formation of chemical intermediates during MSW combustion with any absolute certainty. It is possible, however, to identify the central chemical events in the formation of chlorinated dioxins and dibenzofurans through laboratory experiments under simulated incineration conditions.

Various theories have been proposed to explain the presence of PCDDs and PCDFs in municipal solid waste incinerator emissions (Cleverly, 1991; Vogg, 1987; Gullett,1990,1991a,b; Bruce, 1991; Commoner, 1987; Choudhry, 1983; ASME, 1981; U.S. EPA, 1987b). These theories include one or more of the following:(1) these compounds are present as contaminants in bleached paper or other constituents of MSW, and a portion of them survives the thermal stress imposed by the incineration process; (2) they result from the *de novo* synthesis from precursors, such as polychlorinated biphenyls (PCBs), chlorophenols (CPs), and chlorinated benzenes (CBs); and (3) they are synthesized from materials not chemically related to PCDDs and PCDFs such as petroleum products, polynuclear aromatic hydrocarbons (PAH), inorganic chloride ions, and plastics.

It is postulated that PCDDs and PCDFs are created on the reactive surface of fly ash (particulate matter) entrained in the combustion plasma downstream of the furnace zone in regions where the temperature of the combustion offgases have cooled to between 200° and 400° celsius (Vogg,1987; Bruce,1991; Cleverly,1991; Gullet,1990; Commoner,1987). Through a series of laboratory experiments, Vogg and coworkers (1987) have postulated that inorganic chloride ions, such as copper chloride, present in the combustion gas may act as a catalyst to promote surface reactions on particulate matter to convert aromatic compounds to chlorinated dioxins and dibenzofurans. Isotopically

labeled MSW incinerator fly ash was heated in a laboratory furnace purged by an air flow for two hours. The treated fly ash was exposed to increasing temperatures in 50° C increments in a temperature range of 200° to 400° celsius. Table 6-1 summarizes these data.

Vogg (1987) concluded that formation of PCDDs and PCDFs on the surfaces of fly ash during MSW incineration probably occurs in a temperature window defined as a range of 200° to 350° celsius. Vogg (1987) proposed an oxidation reaction pathway giving rise to the formation of PCDDs and PCDFs in the post furnace regions of the incinerator: (1) Hydrogen chloride gas (HCl) is thermolytically formed as a product from the combustion of heterogeneous refuse containing abundant chlorinated organic chemicals and chlorides; (2) oxidation of HCl via the Deacon process, with copper chloride ($CuCl_2$) as a catalyst, yields free radical chlorine, (3) Phenolic compounds (present from combustion of lignin in the waste) entrained in the combustion plasma are substituted on the ring structure by contact with free radical chlorine; (4) the chlorinated precursor to dioxin, e.g. chlorophenol, is oxidized with copper chloride as a catalyst to form PCDDs and PCDFs and chlorine. Gullett and coworkers (1990, 1991) have studied the formation mechanisms through extensive combustion research, and have concluded the PCDDs and PCDFs can be formed from low temperature reactions (350° C) between Cl_2 and phenolic precursors forming a chlorinated precursor followed by oxidation reactions of chlorinated precursors involving copper chloride as a catalyst as in examples (1) and (2).

(1) The initial step in the formation of dioxin is the formation of chlorine from HCl in the presence of oxygen, as follows (Vogg, 1987):

$$2HCl + 1/2\ O_2 \xrightarrow{\Delta} H_2O + Cl_2$$

(2) PCDD formation from dioxin precursor on fly ash surfaces catalyzed by copper chloride (Vogg, 1987: Gullett, 1990):

$$\text{2-chlorophenol} + 1/2\ O_2 \xrightarrow{CuCl_2} \text{dioxin} + Cl_2$$

Table 6-1. Concentration of PCDD/PCDF (ng/g) on municipal incinerator fly ash at varying temperatures

	Temperature (° Celsius)				
	200°	250°	300°	350°	400°
PCDD					
Tetra	15	26	188	220	50
Penta	40	110	517	590	135
Hexa	65	217	1029	550	110
Hepta	100	208	1103	430	60
Octa	90	147	483	200	15
PCDF					
Tetra	122	560	1379	1185	530
Penta	129	367	1256	1010	687
Hexa	61	236	944	680	260
Hepta	48	195	689	428	112
Octa	12	74	171	72	12

Source: Adapted from Vogg et al., 1987.

The major direct source of chlorine available for participating in the formation of PCDDs/PCDFs is gas-phase HCl, which is initially formed as a combustion product from the chlorine and chlorinated organic chemicals contained in the MSW (Vogg, 1987; Bruce, 1991, Cleverly, 1984; Commoner, 1987). MSW contains approximately 0.66 percent (by weight) chlorine (Trinklein, 1982). MSW incinerators are a major stationary combustion source of air emissions of HCl, which averages between 400 to 600 ppm_V in the combustion gas (U.S. EPA,1987c). HCl is converted to chlorine by the Deacon process, and the free radical chlorine directly chlorinates a dioxin precursor along the aromatic ring structure. Oxidation of the chlorinated precursor in the presence of an alkali inorganic chloride ion catalyst (of which copper chloride is the most active catalyst) yields PCDDs and PCDFs as intermediate reaction products. Increasing the free radical chlorine concentration in the combustion gas generally causes an increase in the rate of formation of PCDDs/PCDFs. Formation kinetics are most favored at temperatures between 200° to 350° celsius. Any reduction in chlorine formation from HCl or decreases in combustion

gas temperatures below the window of formation will result in decreases in the rate and magnitude of formation of PCDDs and PCDFs (Bruce,1991, Gullett,1990; Commoner,1987). In the testing of a variety of industrial stationary combustion sources during the National Dioxin Study in 1987, EPA made a series of qualitative observations on the relationship between total chlorides present in the fuel and the magnitude of emissions of PCDDs and PCDFs (U.S.EPA,1987d). In general, combustion units with the highest PCDD emission concentrations contained significant quantities of chlorine in the feed, and, conversely, sites with the lowest PCDD emission concentrations contained only trace quantities of chlorine in the feed.

The evaluation of the mass balance of PCDD and PCDF emissions from various MSW technologies supports the observations made from these experimental data of post-furnace, low-temperature formation. Commoner and coworkers (1989) evaluated the test data of a mass burn MSW incinerator for the concentration of PCDDs and PCDFs at multiple sampling points during the combustion process: (1) at the exit to the furnace; (2) at the entry to the heat exchanger; (3) inlet to the electrostatic precipitator (ESP); (4) exit to the ESP; (5) exit to the smokestack. Lowest, or non-detectable concentrations of PCDDs/ PCDFs were found at sampling point (1), and highest concentrations were measured at sampling point (5). Commoner concluded that dioxins were not substantially formed within the furnace region, but were formed in areas downstream of the combustion zone where the combustion offgases had cooled to less than 400^0 C. This phenomena was independently observed by Environment Canada in an earlier test of a modular MSW incinerator (Hay,1986; Environment Canada,1985). On a mass balance basis, the concentration of PCDDs and PCDFs measured at the smokestack were approximately two orders of magnitude higher as compared to the inlet to the boiler just after exiting the secondary furnace. The temperatures of the combustion gases at these two points of measurement were 130^0 C and 740^0 C at the stack and boiler inlet, respectively (Environment Canada, 1985). For the most part,only Octa-CDD was present in the hot gases exiting the furnace, whereas the all the congeners were present in the stack emissions.

In a series of tests of an RDF facility conducted jointly by U.S. EPA and Environment Canada (U.S. EPA, 1991), approximately 5 milligrams of total PCDD and PCDF per metric ton of MSW were measured in the refuse prior to combustion, but no

PCDDs nor PCDFs were detected at the exit to the furnace prior to the inlet to the economizer (heat exchanger used to extract additional heat from the hot gases). Once heat in the combustion gas was extracted for energy purposes, and the gases were further cooled to within the "window" of temperatures that promote dioxin formation on fly ash surfaces, then the total array of PCDDs and PCDFs could be detected. These series of experiments in which the mass balance of PCDD/PCDF was estimated for the entire combustion process, including the MSW, discounts the likelihood of the first theory of dioxin emissions: that dioxin in the MSW accounts for dioxin emissions at the stack.

The air emission of polychlorinated biphenyls (PCBs) from MSW incinerators is less understood. There are virtually no theories explaining the detection of these compounds in MSW incinerator emissions. However PCBs have been measured in the raw refuse prior to incineration (Choudhry, 1983, U.S. EPA, 1991). The mass balance of total PCB beginning with measurement in the raw refuse and ending with measurement at the stack to an RDF MSW incinerator (U.S. EPA, 1991) can be used to calculate the destruction rated efficiency (DRE) of incineration of the PCB contaminated MSW. Using results from test number 11 at the RDF facility (U.S. EPA, 1991), a computation of DRE can be made with the following equation (Brunner,1984):

$$DRE = 100 \left[\frac{W_{in} - W_{out}}{W_{in}} \right] \qquad (6\text{-}1)$$

where:

W_{in} = mass rate of contaminant fed into the incinerator system.

W_{out} = mass rate of contaminant exiting the incinerator system.

In test eleven, 811 nanograms total PCBs/gram of refuse (ng/g) was measured in the MSW fed into the incineration system, and 9.52 ng/g of total PCB was measured inlet to the pollution control device (e.g., outside the furnace region, but preceding emission control). From this datum a DRE of 98.8 percent can be calculated. This level of combustion efficiency corresponds to typical MSW incineration (Brunner, 1984). Therefore it appears that PCB contamination in the raw MSW that is fed into the incinerator may account for the emission of PCBs from the stack of the MSW incinerator. This is unlike the

mechanism of formation of PCDDs and PDCFs in that the existing empirical data tends not to support the de novo synthesis of PCBs.

PCBs can be thermolytically converted into PCDFs (Choudhrly, 1983; U.S. EPA, 1984). This process occurs at temperatures somewhat lower than typically measured inside the firebox. Laboratory experiments conducted by EPA (U.S. EPA, 1984) indicate that the optimum conditions for PCDF formation from PCBs are near a temperature of 675° C in the presence of 8% oxygen and a residence time of 0.8 seconds. This resulted in a 3 - 4 % conversion efficiency of PCBs to PCDFs. Since 1- 2 % of the PCBs present in the raw refuse may survive the thermal stress imposed in the combustion zone to the incinerator (U.S. EPA, 1991), then it is reasonable to presume that PCBs in the MSW may contribute to the total mass of PCDF emissions released from the stack of the incinerator.

6.4. HYPOTHETICAL MSW INCINERATOR FOR PURPOSES OF EXPOSURE ANALYSIS

For purposes of illustrating the procedures proposed in this chapter, a specific incinerator design and set of environmental conditions must be chosen. Accordingly a hypothetical,but realistic, facility and location were selected. The mass burn, heat recovery incinerator technology was selected to provide an appropriate example. This technology dominates the current and projected population of incinerators in the U.S. The hypothetical incinerator is assumed to have a combined daily combustion capacity of 2727 metric tons per day (3000 tons/d), which represents the upper 75th percentile of the distribution of existing and projected mass burn, heat recovery facilities in the U.S. (US. EPA, 1989). Table 6-2 summarizes the distribution of existing and projected mass burn heat recovery MSW incinerators by size of facility. The projected facilities are to the year 1995 (U.S. EPA, 1989). If all MSW incinerator technologies are counted, e.g., refuse derived fuel, fluidized bed, mass burn nonheat recovery, and modular incinerator, then EPA estimates there will be 272 MSW incineration facilities operating in 1995 (U.S. EPA, 1989).

The EPA selected the stack height, diameter, exit velocity of the gaseous emissions from the stack, and temperature of the exhaust gases based on actual measurements at facilities of this size. The hypothetical facility is located, for modeling purposes, in a wet and humid climate in a geographical area where the MSW incineration industry is expected to grow rapidly over the next 15 years (U.S. EPA,1987). This site is Tampa, Florida. The

Table 6-2. Capacity distribution (TDP) of existing and projected mass burn, heat recovery MSW incinerators in the United States

Mass burn technology	Facility capacity[a]	Number of facilities[b]	Percent of total
MBWW	200	28	38.4
MBREF	500	3	4.1
MBRCWW	500	5	6.8
MBRCWW	1050	3	4.1
MBWW	1080	16	22.0
MBWW	2250	18	24.7
TOTAL		73	

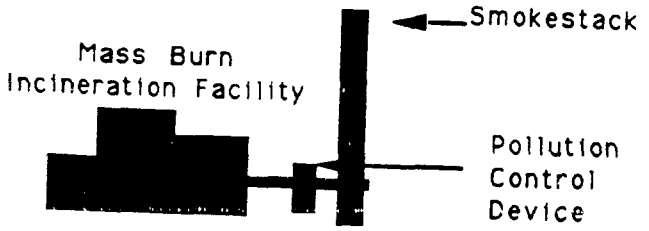
MBWW = mass burn waterwall
MBREF = mass burn refractory
MBRCWW = mass burn rotary combustor, waterwall

[a]Capacity is total tons MSW incinerated each day.
[b]The number of facilities includes only mass burn, heat recovery incinerators. This counts existing facilities, and facilities EPA estimates will be operational by 1994. Although this totals 73 facilities, if all MSW incinerator technologies, both heat and nonheat recovery, are included, the sum is 273 facilities.

Source: Adapted from U.S. EPA, 1989, pp 4-5 and 4-6.

climate conditions were selected to maximize the potential of surface deposition of the emitted dioxin-like compounds with the physical action of washout during periods of precipitation. The site receives in excess of 122 centimeters of rainfall per year,and is classified as a humid subtropical climate. Table 6-3 summarizes the modeling parameters for the hypothetical MSW incinerator.

Table 6-3. Modeling parameters for the hypothetical MSW incinerator



Technology: Mass burn, heat recovery MSW incinerator

Rate of MSW combustion: 2727 metric tons per day (3000 tons/d)

Location: Tampa, Florida

Latitude: 27° 57′

Longitude: 82° 27′

Stack height: 46 meters (m)

Stack diameter: 4.1 m

Exit velocity of stack gases: 11.3 m/s

Pollution control scenarios: (1) electrostatic precipitators (ESPs); (2) dry scrubbers combined with fabric filters (DSFF)

Stack temperature: (1) 470° kelvin with ESPs; (2) 430° kelvin with DSFFs

Building configuration: Height: 35 m; width: 76 m; length: 42 m

6.5 DERIVATION OF MASS EMISSION FACTORS FOR THE HYPOTHETICAL MASS BURN, HEAT RECOVERY MSW INCINERATOR

6.5.1. Congener-Specific Stack Gas Emission Factors for Dioxin-like Compounds

Since 1980, the U.S. EPA has been actively stack testing representative MSW incinerators to measure the magnitude of air emissions of PCDDs and PCDFs. In 1987, the EPA gave advanced notice of proposed rulemaking under the Clean Air Act to establish

numeric emission limits of organic pollutants, including PCDDs and PCDFs from new and exiting MSW incinerators (Cleverly,1989). In support of these regulatory activities, the U.S. EPA initiated combustion research to determine the magnitude of PCDD and PCDFs and other organic emissions from all types of incinerator systems operational in the U.S. and Canada, e.g., mass burn, RDF, and modular. This research was a joint effort with Environment Canada, and included measurements of pollution control technology to determine effectiveness in reducing and controlling PCDD and PCDF emissions to the atmosphere. At the same time regulatory agencies in Europe, Japan, and Scandinavia were undertaking similar activities.

Regardless of the type of MSW incinerator, all MSW incineration systems emit PCDDs and PCDFs. Every known toxic congener of PCDDs and PCDFs, e.g. having chlorine atoms in the lateral 2,3,7,8 positions on the molecule, have been measured in stack gas emissions to MSW incinerators. The emission of PCDDs and PCDFs is variable between technologies, and whether constructed and operational before or after 1987 (U.S.EPA,1987b;1989). This is because the testing of incinerators has yielded information on design characteristics, techniques of incinerator operation and performance, and air pollution control technology that has had the overall benefit of reducing emissions of organic pollutants (OTS,1989). Review of existing emissions data shows that the magnitude in stack emissions of PCDDs and PCDFs between newer and older MSW incinerators, and between facilities equipped with electrostatic precipitators (ESPs) or dry scrubbers in combination with fabric filters (DSFF), indicates up to a 1000-fold difference in the range of emissions of PCDDs/PCDFs (U.S.EPA,1987b; Cleverly,1991; Siebert,1991; OTS,1989).

Statistical analysis of the variability in emissions (using Spearman rank-order techniques) indicated that, as a general trend, emissions of PCDDs and PCDFs appear to be inversely related to furnace temperature, and that low combustion efficiency may promote post-furnace formation (U.S.EPA,1986). However further analysis of the fly ash emissions of PCDDs and PCDFs are not related to furnace conditions alone, and most likely involve a number of complex variables related to function of the incineration process and design, including post-combustion formation outside the firebox region (see section 6.3: The Mechanism of Formation). The compounds found in MSW are complex and variable, and thus mapping any mechanism for formation of specific toxic organic compounds is an

improbable task and is largely theoretical. It is difficult to predict how constituents present in the fuel will break apart, reform new compounds, dissociate into free radicals, or recombine under the influence of oxygen, turbulent mixing and temperature.

Emission factors for specific congeners of dioxin-like compounds are key parameters in estimating the environmental fate and transport and ultimate pathways of human exposure to the pollutants. In the past the EPA converted the concentration of mixtures of PCDD and PCDF congeners into an equivalent concentration of 2,3,7,8-TCDD (Cleverly,1991; U.S.EPA,1989b; Mukerjee,1988) when deriving an emission factor. Therefore when emissions were modeled using standard air dispersion models, the emission to the atmosphere at the point of the stack to the MSW incinerator was the toxic equivalent of 2,3,7,8-TCDD. The physical and chemical properties of 2,3,7,8-TCDD were applied to model the environmental fate of the chemical mixture. In order to increase the level of confidence with respect to predicting pathways to human exposure, the air dispersion modeling of emissions from the hypothetical incinerator is for the specific toxic congeners, and the chemical and physical properties of the individual toxic congeners are employed to estimate the magnitude and pathways of human exposure. Only at the interface to human exposure, e.g., ingestion, inhalation, dermal adsorption, etc., are the individual congeners recombined and converted into the toxic equivalence of 2,3,7,8-TCDD to be factored into quantitative risk assessments.

Emission factors for the analysis of emissions were estimated from test reports of technologies that are most similar in design and operation as the hypothetical MSW incinerator. The inclusion of multiple test reports from multiple facilities increases the confidence that the predicted emissions of the pollutants from the hypothetical facility under study will be in good agreement and fairly representative of emissions from actual facilities of this size. The variability in emissions of PCDDs and PCDFs across MSW technologies and by age of facility indicates that it is important to establish a selection criteria to increase the likelihood that the emission factors are fairly typical. The criteria used here was to use congener-specific emissions data from mass burn heat recovery MSW incinerators that were appropriately designed to today's standards, and that apply the principles of good combustion practice (Brunner, 1984).

Simplified statistical techniques were used to derive a representative emission factor for each dioxin-like congener. After selecting test data of emissions from similar

technologies, it is necessary to place all emission terms into common units of measurement. English units were converted into metric, and the density of the mass of pollutant per unit volume of combustion gas emitted from the stack were corrected to the same measurement of percent carbon dioxide within the combustion gas (12% CO_2). These adjustments were necessary to combine data from a variety of tests of different mass burn incinerators. The next step involved converting the mass emission rate of the specific dioxin-like congener in units of nanograms per normal cubic meter of combustion gas corrected to 12% carbon dioxide into an equivalent emission rate in units of grams per metric ton of refuse (g/T) that is incinerated. This was done for each incinerator by the formula:

$$\frac{g}{T} = \frac{D \quad V_{cg}}{CR} \qquad (6\text{-}2)$$

where:

D = measured mass of dioxin congener per volume of combustion gas, e.g., grams/dry standard cubic meter

V_{cg} = volume of combustion gas/unit of time measured as a release from the stack, e.g., dry standard cubic meters per minute.

CR = MSW charging rate (metric tons) to the incinerator per unit of time equivalent to time in V_{cg}, e.g., metric ton per minute.

Tables 6-4 and 6-5 are the computed mass emission factors (grams/metric ton) for each dioxin-like congener of PCDD and PCDF, respectively, of the mass burn heat recovery MSW incinerators selected to be most similar to the hypothetical facility in this analysis. On the bottom of each table is the computed arithmetic average emission factor derived from inclusion of emissions from ten mass burn facilities. These congener-specific emission factors are used to represent emissions from the hypothetical MSW incinerator, and will be used to derive emissions for air dispersion and deposition modeling of emission impacts.

The estimation of the emission factors of dioxin-like polychlorinated biphenyl (PCB) congeners is made difficult by lack of emissions data from tested MSW incinerators. PCBs

Table 6-4. Estimation of PCDD congener-specific emission factors (grams/metric ton of MSW burned) for representative mass burn heat recovery incinerators

Facility	2,3,7,8-TCDD	Other TCDD	1,2,3,7,8-CDD	Other PCDD	1,2,3,4,7,8-HxCDD	1,2,3,6,7,8-HxCDD
1	2.10E-06	3.16E-05	NA	NA	NA	NA
2	7.85E-06	7.80E-05	6.38E-06	1.44E-04	7.80E-06	1.65E-05
3	9.40E-06	1.67E-04	1.88E-05	1.77E-04	1.05E-05	1.77E-05
4	3.97E-07	5.94E-06	2.40E-06	3.54E-05	1.90E-06	4.68E-06
5	1.94E-06	2.68E-06	2.27E-07	2.34E-06	2.89E-07	1.47E-07
6	1.56E-05	1.10E-03	2.81E-04	2.75E-03	4.99E-05	1.15E-04
7	1.97E-06	4.72E-05	NA	1.24E-04	NA	NA
8	2.78E-07	1.38E-05	6.63E-07	1.08E-05	5.14E-07	1.18E-06
9	2.91E-06	2.69E-05	7.65E-06	2.19E-05	1.57E-06	NA
10	8.15E-07	1.69E-05	1.66E-06	2.30E-05	2.52E-06	4.76E-06
Average	4.33E-06	1.49E-04	3.98E-05	3.65E-04	9.38E-06	2.29E-05

Facility
1. Chicago NW (U.S. EPA, 1983)
2. N. Andover, MA (U.S. EPA, 1988a)
3. Saugus, MA (Knisley, 1986c)
4. Tulsa, OK (Seelinger, 1986b)
5. Marion Co., OR[a] (U.S. EPA, 1988c)
6. Umea, Sweden (Markland, 1985)
7. Zurich, Switzerland (Siebert, 1991)
8. Alexandria, VA (Siebert, 1991)
9. Westchester, NY (Siebert, 1991)
10. Pinellas Co., FL (Entropy, 1987)

[a]Marion County emissions are controlled by dry scrubbers combined with fabric filters. Because the hypothetical facility is equipped with an ESP as baseline conditions, the uncontrolled emissions were used in the derivation of emission factors.
NA = Congener not analyzed.

(continued on the following page)

Table 6-4. (continued)

Facility	1,2,3,7,8,9-HxCDD	Other HxCDD	1,2,3,4,6,7,8-HpCDD	Other HpCDD	OctaCDD
1	NA	8.24E-05	NA	3.83E-05	1.28E-05
2	8.84E-06	1.99E-04	NA	1.88E-04	1.54E-04
3	NA	1.63E-04	NA	1.66E-04	2.06E-04
4	NA	6.45E-05	2.78E-05	5.60E-05	6.08E-05
5	5.68E-08	4.28E-05	1.80E-05	1.92E-05	3.76E-05
6	4.06E-05	7.92E-04	NA	5.62E-04	3.74E-04
7	NA	2.45E-04	NA	2.96E-04	6.04E-04
8	1.71E-06	1.18E-05	7.06E-06	6.61E-06	1.43E-05
9	NA	3.63E-05	NA	5.53E-05	9.23E-05
10	6.65E-06	3.96E-05	3.15E-05	3.34E-05	8.30E-05
Average	1.16E-05	1.68E-04	2.11E-05	1.42E-04	1.64E-04

Facility
1. Chicago NW (U.S. EPA, 1983)
2. N. Andover, MA (U.S. EPA, 1988a)
3. Saugus, MA (Knisley, 1986c)
4. Tulsa, OK (Seelinger, 1986b)
5. Marion Co., OR[a] (U.S. EPA, 1988c)
6. Umea, Sweden (Markland, 1985)
7. Zurich, Switzerland (Siebert, 1991)
8. Alexandria, VA (Siebert, 1991)
9. Westchester, NY (Siebert, 1991)
10. Pinellas Co., FL (Entropy, 1987)

[a]Marion County emissions are controlled by dry scrubbers combined with fabric filters. Because the hypothetical facility is equipped with an ESP as baseline conditions, the uncontrolled emissions were used in the derivation of emission factors.
NA = Congener not analyzed.

Table 6-5. Estimation of PCDF congener-specific emission factors (grams/metric ton of MSW burned) for representative mass burn heat recovery incinerators

Facility	2,3,7,8-TCDF	Other TCDF	1,2,3,7,8-PCDF	2,3,4,7,8-PCDF	Other PCDF
1	NA	4.53E-04	NA	NA	NA
2	1.05E-04	3.10E-04	2.66E-05	5.39E-05	1.74E-04
3	1.29E-04	8.76E-04	3.26E-05	5.64E-05	4.96E-04
4	1.14E-05	2.86E-05	1.77E-06	3.63E-06	1.31E-05
5	2.44E-05	2.20E-05	9.47E-08	1.19E-05	7.62E-06
6	7.80E-05	2.61E-03	2.81E-04	1.90E-04	2.56E-03
7	NA	2.51E-04	NA	NA	2.60E-04
8	1.80E-05	1.15E-04	3.39E-06	5.38E-06	5.20E-05
9	2.30E-05	2.91E-04	6.58E-05	NA	1.17E-04
10	4.05E-06	8.74E-05	2.60E-06	6.05E-06	4.18E-05
Average	4.91E-05	5.04E-04	5.17E-05	4.68E-05	4.13E-04

Facility
1. Chicago NW (U.S. EPA, 1983)
2. N. Andover, MA (U.S. EPA, 1988a)
3. Saugus, MA (Knisley, 1986c)
4. Tulsa, OK (Seelinger, 1986b)
5. Marion Co., OR[a] (U.S. EPA, 1988c)
6. Umea, Sweden (Markland, 1985)
7. Zurich, Switzerland (Siebert, 1991)
8. Alexandria, VA (Siebert, 1991)
9. Westchester, NY (Siebert, 1991)
10. Pinellas Co., FL (Entropy, 1987)

[a]Marion County emissions are controlled by dry scrubbers combined with fabric filters. Because the hypothetical facility is equipped with an ESP as baseline conditions, the uncontrolled emissions were used in the derivation of emission factors.
NA = Congener not analyzed.

(continued on the following page)

Table 6-5. (continued)

Facility	1,2,3,4,7,8-HxCDF	1,2,3,6,7,8-HxCDF	1,2,3,7,8,9-HxCDF	2,3,4,6,7,8-HxCDF	Other HxCDF
1	NA	NA	NA	NA	1.31E-04
2	3.56E-05	1.09E-05	1.03E-04	NA	3.13E-05
3	7.19E-05	4.31E-05	NA	NA	2.69E-04
4	2.14E-06	8.52E-07	3.53E-07	2.29E-06	7.10E-06
5	1.47E-07	1.56E-07	1.89E-07	8.33E-06	NA
6	1.12E-04	1.15E-04	2.50E-05	8.11E-05	6.65E-04
7	NA	NA	NA	NA	1.22E-04
8	5.25E-06	2.42E-06	1.03E-07	2.88E-06	3.75E-06
9	2.09E-05	NA	NA	NA	1.64E-04
10	1.00E-05	5.05E-06	1.50E-07	8.05E-06	2.72E-05
Average	3.23E-05	2.54E-05	2.15E-05	2.05E-05	1.58E-04

Facility
1. Chicago NW (U.S. EPA, 1983)
2. N. Andover, MA (U.S. EPA, 1988a)
3. Saugus, MA (Knisley, 1986c)
4. Tulsa, OK (Seelinger, 1986b)
5. Marion Co., OR[a] (U.S. EPA, 1988c)
6. Umea, Sweden (Markland, 1985)
7. Zurich, Switzerland (Siebert, 1991)
8. Alexandria, VA (Siebert, 1991)
9. Westchester, NY (Siebert, 1991)
10. Pinellas Co., FL (Entropy, 1987)

[a]Marion County emissions are controlled by dry scrubbers combined with fabric filters. Because the hypothetical facility is equipped with an ESP as baseline conditions, the uncontrolled emissions were used in the derivation of emission factors.
NA = Congener not analyzed.

(continued on the following page)

Table 6-5. (continued)

Facility	2,3,4,6,7,8-HpCDF	2,3,4,7,8,9-HpCDF	Other HpCDF	OctaCDF
1	NA	NA	3.76E-05	3.03E-06
2	NA	NA	2.57E-04	2.18E-04
3	NA	NA	2.00E-04	9.79E-05
4	5.72E-06	6.59E-07	9.21E-06	4.16E-07
5	2.84E-07	3.79E-07	3.27E-07	2.98E-07
6	NA	NA	1.06E-03	3.12E-04
7	NA	NA	2.76E-05	1.01E-04
8	6.38E-06	7.20E-07	3.19E-06	2.93E-06
9	9.08E-05	NA	2.09E-05	4.18E-06
10	2.65E-05	1.50E-06	7.60E-06	6.30E-06
Average	2.59E-05	8.14E-07	1.62E-04	7.45E-05

Facility
1. Chicago NW (U.S. EPA, 1983)
2. N. Andover, MA (U.S. EPA, 1988a)
3. Saugus, MA (Knisley, 1986c)
4. Tulsa, OK (Seelinger, 1986b)
5. Marion Co., OR[a] (U.S. EPA, 1988c)
6. Umea, Sweden (Markland, 1985)
7. Zurich, Switzerland (Siebert, 1991)
8. Alexandria, VA (Siebert, 1991)
9. Westchester, NY (Siebert, 1991)
10. Pinellas Co., FL (Entropy, 1987)

[a]Marion County emissions are controlled by dry scrubbers combined with fabric filters. Because the hypothetical facility is equipped with an ESP as baseline conditions, the uncontrolled emissions were used in the derivation of emission factors.
NA = Congener not analyzed.

are a class of toxic air pollutants that have not been routinely monitored in stack emissions to MSW incinerators. The data that are available usually have reported PCBs as the sum total of all PCBs present in the sample without further speciation of toxic congeners or congener groups (U.S. EPA,1987c). The greatest level of detail in any test report is a further breakdown of total PCBs into homologue groups, e.g., mono - deca-chlorobiphenyl.

For purposes of undertaking an air dispersion modeling analysis of the emission of specific dioxin-like congeners from the stack of the hypothetical MSW incinerator, it is necessary to derive a presumption of the distribution of specific congeners using the best available emission data, e.g., homologue groups of PCBs in the emission. Lacking any applicable law of thermodynamics that would permit an estimation of congener-specific emissions based on the characteristics of the MSW fed into the system, and the characteristics of time, temperature and turbulence in the incineration process itself, one can estimate a distribution based on simple assumptions of the probability a specific congener exists within the appropriate homologue group. Until EPA has congener-specific emissions data from testing operational MSW incinerators, the simplifying assumption is that all congeners within a homologue group shall have an equal probability of occurrence within the emission. Therefore the specific toxic coplanar PCB congener listed in Table 6-6 shall have an equal probability of occurrence in the respective homologue group. If equal probability of occurrence is assumed, then the estimation of the emission factor of the specific coplanar PCB congener can be made by establishing a ratio of the congener of concern to the total possible isomers in the chlorinated homologue group. For example, there are 42 possible isomers in the tetrachloro homologue group. Therefore if equal probability of occurrence of each isomer in the homologue group is assumed, then each tetrachlorinated isomer in Table 6-6 has a chance occurrence in the stack gas emissions of 1 in 42, or a ratio of 0.024. These ratios can be multiplied by the homologue-specific emission factor derived from incinerator tests to estimate the emission factor for the specific congener under evaluation. Table 6-7 displays the computation of emission factors by homologue group of PCBs, and Table 6-8 is the estimated emission factor for specific dioxin-like congeners of coplanar PCBs.

It must be noted that the estimated emission factors for specific coplanar congeners of PCBs listed in Table 6-8 are highly uncertain. The following contribute to this uncertainty:

Table 6-6. Coplanar polychlorinated biphenyl congeners having dioxin-like biological activity; number of possible isomers; ratio of toxic isomer to total isomers in homologue group

Congener	Homologue	Number of isomers	Ratio
3,3′,4,4′-TeCB	Tetra-chloro	42	0.024
3,4,4′,5-TeCB	Tetra-chloro	42	0.024
3,3′,4,4′,5-PeCB	Penta-chloro	46	0.022
2,3,3′,4,4′-PeCB	Penta-chloro	46	0.022
2,3,4,4′,5-PeCB	Penta-chloro	46	0.022
2,3′,4,4′,5-PeCB	Penta-chloro	46	0.022
3,3′,4,4′,5,5′-HxCB	Hexa-chloro	42	0.024
2,3,3′,4,4′,5-HxCB	Hexa-chloro	42	0.024
2,3,3′,4,4′,5′-HxCB	Hexa-chloro	42	0.024
2,3′,4,4′,5,5′-HxCB	Hexa-chloro	42	0.024
2,3,3′,4,4′,5,5′-HCB	Hepta-chloro	24	0.042

Table 6-7. Emission factors (grams/metric ton) of homologue groups of polychlorinated biphenyls from emission tests of MSW incinerators

Facility	DiCB	TriCB	TetraCB	PentaCB	HexaCB	HeptaCB	OctaCB
1. Padua, Italy[a]	2.00E-03	1.41E-02	1.37E-02	2.46E-05	4.35E-03	3.09E-03	3.08E-05
2. Chicago, NW[b]	2.76E-04	2.55E-04	9.97E-05	4.16E-05	NA	NA	NA
3. Hartford, Conn[c]	7.08E-05	1.01E-04	6.47E-05	NA	NA	2.95E-05	NA
Average	7.82E-04	4.81E-03	4.63E-03	3.31E-05	4.35E-03	1.56E-03	3.08E-05

[a]Mass burn, heat recovery unit rated at 109 MT/d (Magagni, 1990).
[b]Mass burn, heat recovery unit rated at 363 MT/d (U.S. EPA, 1983).
[c]RDF, heat recovery unit rated at 662 MT/d (U.S. EPA, 1991).

Table 6-8. Estimation of congener-specific emission factors of coplanar PCBs

PCB congener	Homologue emission factor[a] (grams/MT)	Ratio isomer/ homologue[b]	Congener emission factor[c] (grams/MT)
3,3',4,4'-TeCB	4.63×10^{-3}	0.024	1.1×10^{-4}
3,4,4',5-TeCB	4.63×10^{-3}	0.024	1.1×10^{-4}
3,3',4,4',5-PeCB	3.31×10^{-5}	0.022	7.3×10^{-7}
2,3,3',4,4'-PeCB	3.31×10^{-5}	0.022	7.3×10^{-7}
2,3,4,4',5-PeCB	3.31×10^{-5}	0.022	7.3×10^{-7}
2,3',4,4',5-PeCB	3.31×10^{-5}	0.022	7.3×10^{-7}
3,3',4,4',5,5'-HxCB	4.35×10^{-3}	0.024	1.0×10^{-4}
2,3,3',4,4',5-HxCB	4.35×10^{-3}	0.024	1.0×10^{-4}
2,3,3',4,4',5'-HxCB	4.35×10^{-3}	0.024	1.0×10^{-4}
2,3',4,4',5,5'-HxCB	4.35×10^{-3}	0.024	1.0×10^{-4}
2,3,3',4,4',5,5'-HCB	1.56×10^{-3}	0.042	6.6×10^{-5}

[a]Homologue PCB emission factor (grams/metric ton of MSW burned) taken from Table 6-7.
[b]Ratio of dioxin-like coplanar PCB isomer to number of isomers in homologue group taken from Table 6-6.
[c]The homologue group emission factor times the ratio of specific isomer within the homologue group equals the estimated congener-specific coplanar PCB emission factor (grams/metric ton of MSW burned).

- A general lack of emission test reports characterizing the stack gas emission of coplanar PCBs from representative MSW incineration technologies.
- Because of a general lack of emission characterization studies, it was necessary to combine emissions data of unrelated MSW incinerator technologies, e.g., mass burn and RDF facilities, in order to derive a PCB homologue-specific emission factor. It is not known if the magnitude of PCB emissions varies greatly from one technology to another.
- The emission of PCBs seems to be related to the incomplete combustion of PCBs present as contaminants in the raw MSW, and not a post-combustion formation phenomenon. PCBs may or may not be present as contaminants in the MSW fed into the incinerator, and therefore, may or may not be

present in the stack gas emissions.

- Congener-specific emission factors of coplanar PCBs were estimated from homologue emission data using simplifying assumptions. No congener-specific emissions from MSW incinerators could be found in the technical literature.

6.5.2. Estimation of Emission of Dioxin-Like Compounds in MSW Incineration Ash

The previous section estimated the mass rate of emission of specific congeners of PCDDs, PCDFs and PCBs from the stack of the hypothetical MSW incinerator. This section is an estimation of the mass of ash that is produced as a residue from the combustion of MSW on the grate within the incinerator (bottom ash), and the ash that is collected by the particulate matter control device preceding the stack (fly ash). This estimation is appropriate for the analysis of human exposures to dioxin-like compounds after storing, transporting, and disposing of the fly ash residues from the hypothetical mass burn incinerator into a sanitary landfill or ash monofill.

In general there are many factors that may influence the formation of particulate matter known as bottom ash and fly ash from the incineration of MSW. Among these factors are: the heating value of the MSW (Btu/kg), the percent moisture in the refuse, the furnace temperature and combustion efficiency, and the efficiency of particulate matter capture by the pollution control device (Brunner,1984; OTS,1989).

For purposes of exposure analyses of the hypothetical MSW incinerator, the assumptions of Cook (1991) are used. Cook (1991) extensively evaluated ash generated from modern mass burn heat recovery incinerators and created the following generalized mass balance from the data:

- Bottom ash from the combustion grate is produced at a rate of 10 percent of the mass of MSW incinerated per day.
- Fly ash is generated from the particulate matter pollution control device (ESP or Fabric Filter) at a rate of 1 percent of the mass of MSW incinerated per day.
- Fly ash from the addition of a semi-dry acid gas control device is produced at a rate of 0.1 percent of the mass of MSW incinerated per day. Applying these generalized assumptions, the daily mass of ash generated at the

hypothetical MSW incinerator is as follows:

-- With Electrostatic Precipitators (ESPs) for particulate matter control at 99% control efficiency:

1. Bottom ash: 2727 metric tons/day incinerated X 10 %.
   Bottom ash: 272.7 metric tons/day.
2. Fly ash: 2727 metric tons/day incinerated X 1 %.
   Fly ash: 27.3 metric tons/day.

-- With Fabric Filters (99% control efficiency) combined with semi-dry alkaline scrubbers:

1. Bottom ash: 272.7 metric tons/day.
2. Fly ash:
   a. 27.3 metric tons/day from fabric filters.
   b. 2.7 metric tons/day from dry scrubber.
   c. Sum (a) + (b) = 30 metric tons/day.

The PCDDs, PCDFs, and coplanar PCBs are primarily associated with the fly ash that is prevented from escaping into the atmosphere by the particulate matter control device (Chrostowski, 1991; Denison, 1991; OTS,1989). Negligible amounts of PCDDs/PCDFs have been detected on bottom ash from the combustion grate because these chemicals are synthesized outside of the furnace region (OTS, 1989). No data could be found on PCB contamination of MSW bottom ash. It is customary for MSW incineration facilities to combine fly ash with the bottom ash prior to disposal in a sanitary landfill (Chrostowski, 1991). This will have the overall effect of diluting the initial contaminant concentration, because the mass of bottom ash exceeds fly ash by a ratio of 10/1. For purposes of the exposure analysis of ash disposed in landfills, this Chapter will focus only on the fly ash generated at the hypothetical MSW incinerator, and will characterize the distribution of concentration of dioxin-like congeners without dilution through mixing with bottom ash.

The descriptive air dispersion modeling parameters for the hypothetical mass burn MSW incinerator are given in Table 6-3. There are basically two pollution control scenarios to the hypothetical incinerator which will affect the estimated generation of fly ash. The first scenario uses only electrostatic precipitators for particulate matter control, and the second scenario uses dry-alkaline scrubbers combined with fabric filters. Both the electrostatic precipitators and the fabric filters are assumed to have the same level of

efficiency for particulate matter control, e.g. 99%. However it is assumed that ESPs will reduce emissions of PCDDs, PCDFs, and coplanar PCBs by only 20%, which is consistent with reports of testing ESP units (Cleverly,1991; U.S. EPA, 1987b). Dry alkaline scrubbers combined with fabric filters are assumed to reduce and control these pollutants by 95%, which is consistent with EPA tests of the effectiveness of such control devices (U.S EPA, 1987b). The mechanism involved is the dramatic temperature reduction achieved with the dry alkaline scrubbers (e.g., to less than 200° C.), which promotes condensation of vapor phase pollutants onto fly ash surfaces before passing through the fabric filter (U.S. EPA, 1987b). From this it can be seen that the mass emission rate of dioxin-like compounds from the stack to the atmosphere will be lower with the application of dry-alkaline scrubbers combined with fabric filters than with ESPs alone, but the fly ash generation rate and the concentration of dioxin-like chemicals in the fly ash will be greater.

The concentration of dioxin-like compounds in the fly ash can be estimated by the difference in capture and control efficiency between the two pollution control scenarios. What is not emitted from the stack to the atmosphere will be contained in the captured fly ash to the particulate matter control device. Tables 6-4, 6-5, and 6-8 are the estimated congener-specific stack air emission factors for dioxin, dibenzofurans, and coplanar PCBs, respectively. These emission factors were derived from mass burn heat recovery MSW incinerators equipped with ESPs for particulate matter control. To estimate stack emission factors resulting from the application of dry alkaline scrubbers in combination with fabric filters, the emission factors from ESP facilities can be adjusted as follows:

$$EF_c = EF_{esp} \left[\frac{1-CE_c}{1-CE_{esp}} \right] \qquad (6-3)$$

where:

EF_c = emission factor for the additional control with dry alkaline scrubbers combined with fabric filters

EF_{esp} = emission factor for the ESP control scenario from Tables 6-4, 6-5, and 6-8

CE_c = control efficiency (95%) assumed for dry-alkaline scrubbers combined with fabric filters

CE_{esp} = assumed control efficiency (20%) for ESPs.

From this relationship, it can be seen that the stack gas emission factor for dioxin-like congeners is 16 times greater for ESP control scenario than for dry-scrubber coupled with fabric filter (DSFF) control scenario. However, the concentration of dioxin-like congeners in the collected fly ash is 4.75 times greater in the dry-scrubber, fabric filter scenario than in the ESP control scenario. The concentration in the collected fly ash is about 4 times greater in the collected fly ash than what is emitted from the stack under the ESP control scenario, and about 19 times greater in the collected fly ash than in stack emissions under the DSFF scenario. Tables 6-9 and 6-10 are the estimated concentration of chlorinated dioxin, dibenzofuran, and coplanar PCB congeners in the fly ash in the ESP control scenario, and Tables 6-11 and 6-12 are the fly ash concentrations of specific congeners in the DSFF control scenario to the hypothetical MSW incinerator facility.

6.6. AIR DISPERSION MODELING OF THE STACK GAS EMISSIONS OF DIOXIN-LIKE COMPOUNDS FROM THE HYPOTHETICAL MSW INCINERATOR

It has been customary for EPA to use air dispersion models to estimate ambient air concentrations of specific pollutants attributable to smokestack emissions from an industrial source. In the analysis of potential ground-level exposures to dioxin-like compounds in the vicinity of the hypothetical MSW incinerator, the Industrial Source Complex (ISC) air dispersion model was used.

Air dispersion models are mathematical constructs that approximate the physical and chemical processes occurring in the atmosphere that directly influence the dispersion of gaseous and particulate emissions from smokestacks of stationary combustion sources. These models are computer programs encompassing a series of partial differential and algebraic equations to calculate and approximate the dispersion and surface deposition of the emissions to estimate the spacial and temporal ground-level concentrations of pollutants. Concentration isopleths of the pollutants discharged from the stack are computed along the polar azimuth at specified distances downwind of the smokestack. The estimated ground-level concentrations are used to estimate the magnitude of potential exposures to the human receptor.

Kapahi (1991) has reviewed the physical principles involved in most air dispersion models used by EPA. Emphasis is placed on the fact that these models are abstractions of reality that permit the prediction of hypothetical future states of a system that may not

Table 6-9. Emission factors (grams/metric ton) of PCDDs/PCDFs for collected fly ash in the ESP control scenario

Congener	Emission factor	Congener	Emission factor
2,3,7,8-TCDD	1.7×10^{-5}	2,3,4,7,8-PeCDF	2.0×10^{-4}
1,2,3,7,8-PeCDD	1.6×10^{-4}	1,2,3,4,7,8-HxCDF	1.3×10^{-4}
1,2,3,4,7,8-HxCDD	3.8×10^{-5}	1,2,3,6,7,8-HxCDF	1.0×10^{-4}
1,2,3,6,7,8-HxCDD	9.2×10^{-5}	1,2,3,7,8,9-HxCDF	8.6×10^{-5}
1,2,3,7,8,9-HxCDD	4.6×10^{-5}	2,3,4,6,7,8-HxCDF	8.2×10^{-5}
1,2,3,4,6,7,8-HpCDD	8.4×10^{-5}	1,2,3,4,6,7,8-HpCDF	1.0×10^{-4}
OctaCDD	6.6×10^{-4}	1,2,3,4,7,8,9-HpCDF	3.3×10^{-6}
2,3,7,8-TCDF	2.0×10^{-4}	OctaCDF	3.0×10^{-4}
1,2,3,7,8-PeCDF	2.0×10^{-4}		

Table 6-10. Emission factors (grams/metric ton of MSW incinerated) of coplanar PCBs for collected fly ash in the ESP control scenario

Congener	Emission factor
3,3',4,4'-TeCB	4.4×10^{-4}
3,4,4',5-TeCB	4.4×10^{-4}
3,3',4,4',5-PeCB	3.0×10^{-6}
2,3,3',4,4'-PeCB	3.0×10^{-6}
2,3,4,4',5-PeCB	3.0×10^{-6}
2,3',4,4',5-PeCB	3.0×10^{-6}
3,3',4,4',5,5'-HxCB	4.0×10^{-4}
2,3,3',4,4',5-HxCB	4.0×10^{-4}
2,3,3',4,4',5'-HxCB	4.0×10^{-4}
2,3',4,4',5,5-HxCB	4.0×10^{-4}
2,3,3',4,4',5,5'-HCB	2.6×10^{-4}

Table 6-11. Emission factors (grams/metric ton of MSW incinerated) of PCDDs/PCDFs for collected fly ash in the dry-scrubber, fabric filter control scenario

Congener	Emission factor	Congener	Emission factor
2,3,7,8-TCDD	8.0×10^{-5}	2,3,4,7,8-PeCDF	9.5×10^{-4}
1,2,3,7,8-PeCDD	7.6×10^{-4}	1,2,3,4,7,8-HxCDF	6.2×10^{-4}
1,2,3,4,7,8-HxCDD	1.8×10^{-4}	1,2,3,6,7,8-HxCDF	4.8×10^{-4}
1,2,3,6,7,8-HxCDD	4.4×10^{-4}	1,2,3,7,8,9-HxCDF	4.1×10^{-4}
1,2,3,7,8,9-HxCDD	2.2×10^{-4}	2,3,4,6,7,8-HxCDF	4.0×10^{-4}
1,2,3,4,6,7,8-HpCDD	4.0×10^{-4}	1,2,3,4,6,7,8-HpCDF	4.8×10^{-4}
1,2,3,4,6,7,8,9-OctaCDD	3.1×10^{-3}	1,2,3,4,7,8,9-HpCDF	1.6×10^{-5}
2,3,7,8-TCDF	9.5×10^{-4}	1,2,3,4,6,7,8,9-OctaCDF	1.4×10^{-4}
1,2,3,7,8-PeCDF	9.5×10^{-4}		

Table 6-12. Emission factors (grams/metric ton of MSW incinerated) of coplanar PCBs for collected fly ash in the dry-scrubber, fabric filter control scenario

Congener	Emission factor
3,3',4,4'-TeCB	2.1×10^{-3}
3,4,4',5-TeCB	2.1×10^{-3}
3,3',4,4',5-PeCB	1.4×10^{-5}
2,3,3',4,4'-PeCB	1.4×10^{-5}
2,3,4,4',5-PeCB	1.4×10^{-5}
2,3',4,4',5-PeCB	1.4×10^{-5}
3,3',4,4',5,5'-HxCB	2.0×10^{-3}
2,3,3',4,4',5'-HxCB	2.0×10^{-3}
2,3,3',4,4',5-HxCB	2.0×10^{-3}
2,3',4,4',5,5-HxCB	2.0×10^{-3}
2,3,3',4,4',5,5'-HCB	2.6×10^{-4}

exist. The primary utility is to avoid unwanted adverse occurrences through a thorough simulation and analysis of possible impacts over a specified time interval.

In simple terms the ISC air dispersion model uses the basic physical processes of advection, turbulent diffusion, and removal to estimate the atmospheric transport and settling of particles comprising the gaseous and particulate emissions from the stack. These physical principles are illustrated in Figure 6-1. Advection describes the physical movement of the air pollutants by the horizontal movement of wind. The horizontal wind direction and speed are key to estimating the geographical direction and distance the emission plume will disperse, and to approximate the rate of dilution of pollutant concentrations with downwind distance from the emission source. Wind speed and direction can vary seasonally and diurnally, or as a function of passing frontal systems (Kapahi,1991). Wind roses are developed through systematic measurement of hourly wind speed and direction at 10 meters above the surface of the earth that reflect monthly wind direction and intensity along the polar azimuth. These records are crucial to air dispersion analysis.

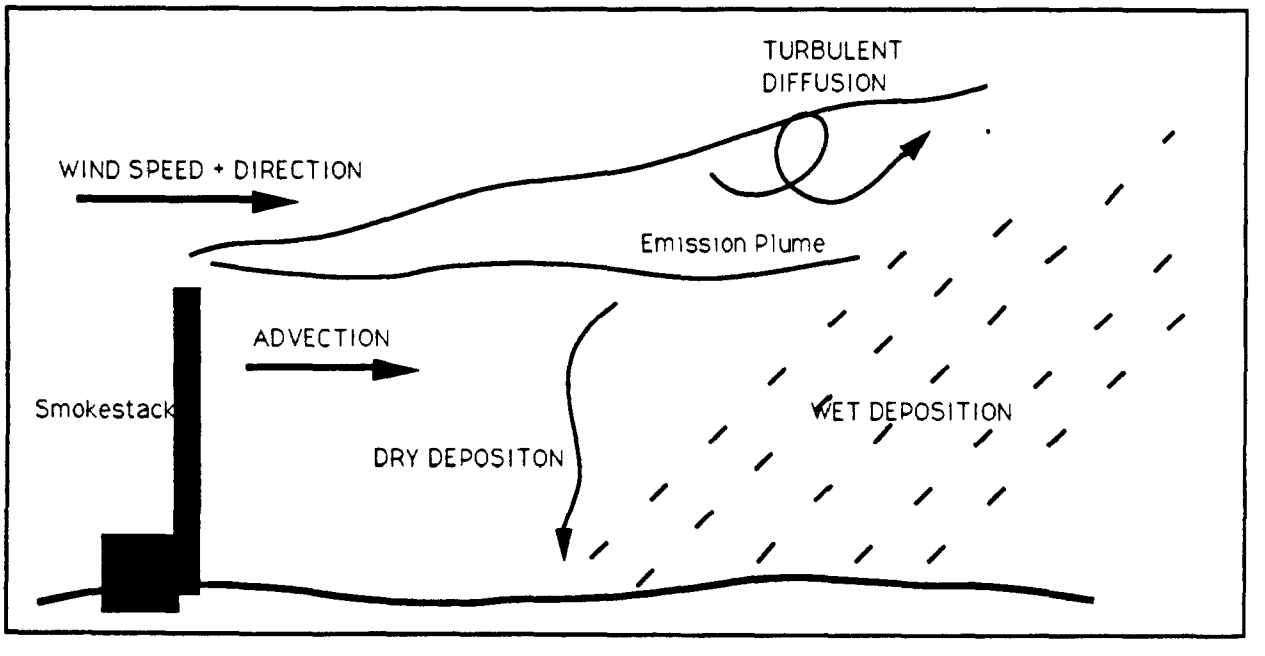


Figure 6-1. Principal physical processes used by the ISC dispersion model to estimate the atmospheric transport and deposition of pollutants discharged from the smokestack.

Turbulent diffusion is simulated by the model to approximate the "spreading" of the emissions plume with distance from the stack (Kapahi, 1991). Turbulence is caused by the randomized variation of wind speed and direction over time, and the stratified temperatures at various altitudes in the lower 500 meters in the atmosphere. Random fluctuations in wind velocities result in rates of momentum, heat, and mass transfer in turbulence that are greater than that of simple molecular transport (Seinfeld,1986).

In the evaluation of the emissions of the hypothetical incinerator, the Industrial Source Complex (ISC) model was used to characterize plume dispersion (U.S. EPA,1986). The ISC uses the generalized Briggs (1975,1977) equation to estimate plume-rise and downwind dispersion as a function of turbulent diffusion. A wind-profile exponent law is used to adjust the observed mean wind speed from the measurement height to the emission height for the plume rise and pollutant concentration calculations. The Pasquill-Gifford curves (Turner, 1970) are used to calculate lateral and vertical plume spread. The ISC contains rural and urban options to account for the effects of terrain, and the aerodynamic wakes and eddies of building and tall structures on turbulent diffusion.

Removal processes that are simulated by the ISC model are dry and wet deposition of particulate matter to the surface of the earth. Dry deposition technically refers to the removal of particulate-bound contaminants by gravitational settling and turbulent diffusion. The rate at which the atmospheric particulate is removed by the physical forces of gravity and atmospheric turbulence is termed the deposition flux (Kapahi,1991), and is mathematically represented by F_d. The relationship between F_d, the concentration of the chemical contaminant,C_o, and the settling velocity of the contaminated particles can be defined by:

$$F_d = V_d \quad C_o \qquad (6-4)$$

where V_d is the settling velocity of the particulates (Kapahi,1991). Particles greater than 30 micrometers (μm) in diameter will be removed from the atmosphere primarily by the force of gravity, whereas particles less than 30 μm will be removed primarily by atmospheric turbulence. The deposition flux for the smaller particles is influenced by many factors, including: the distribution of particles by diameter and density; assumptions of atmospheric turbulence; the friction of the terrain; the height of the stack release of

emissions; and the concentration of the pollutant by particle size. The ISC Short-Term (ISCST) version was used to estimate deposition flux based on the model developed by Dumbauld and coworkers (1976). This model assumes that a fraction of the particulate comes into contact with the ground surface by the combined processes of gravitational settling and atmospheric turbulence. The ISCST is designed to calculate deposition flux for time periods of from one to 24 hours. If used with a year of sequential hourly meteorological data, ISCST can also calculate annual concentration or deposition values. A more detailed description of the default settling velocities by particle size is discussed in the user's guide (U.S. EPA, 1986).

Wet deposition occurs by precipitation (rain,hail,snow) physically washing out the chemically contaminated particulate from the atmosphere. This is termed "scavenging", and the wet deposition flux relates fraction of the time precipitation occurs, the fraction of material removed by precipitation per unit of time by particle size, the strength of the source emission (in grams), the wind speed at the release point to the smokestack, and turbulent diffusion of the emission plume from the source (U.S. EPA,1987b). Based on these relationships, scavenging coefficients were developed for varying types and intensities of precipitation relative to different particle diameters by incorporating the observations of Radke et al. (1980) in a study of scavenging of aerosol particles by precipitation.

The principal assumptions made in computing wet deposition flux are: (1) the intensity of precipitation is constant over the entire path between the source and the receptor; (2) the precipitation originates at a level above the top of the emission plume so that the hydrometers pass vertically through the entire plume; (3) the time duration of the precipitation over the entire path between the source and receptor point is such that exactly f, a fraction between zero and one, of the hourly emission is subject to a constant intensity for the entire travel time required to traverse the distance between the source and receptor, and the remaining fraction (1-f) is subject only to deposition processes. Thus, no dry deposition occurs during hours of steady precipitation, and dry deposition occurs between the periods of precipitation.

The ISC model is commonly referred to as a Gaussian plume model. In the estimation of turbulent diffusion, the spread of the emission plume is viewed as a random process (Kapahi,1991), with key measurement parameters following an approximately

normal distribution according to the Central Limit Theorem of statistics. The Central Limit Theorem states that if random samples of n observations are drawn from a population with finite mean, μ, and a standard deviation, ∂, then when n is large, the sample mean will be approximately normally distributed with mean equal to μ and standard deviation $\partial/\sqrt{n}$. The approximation will become more and more accurate as n becomes large.

6.6.1. Estimation of Distribution in the Stack Emissions to the Hypothetical MSW Incinerator

Certain inferences must be made concerning the distribution of particulate differentiated on the basis of particle diameter (μm) before the ISCST program can predict deposition of the dioxin-like congeners. Unfortunately there exists very few studies of the distribution of particulate matter entrained in the emissions from MSW mass burn incinerators broken down and fractionated by particle diameter. The distribution of particulate matter by particle diameter will differ from one incineration process to another, and is greatly dependent on the percent efficiency of particulate captor by particle diameter from various pollution control devices. Table 6-13, for example, compares the particle diameter distribution between emissions controlled by and ESP verses emissions controlled by a fabric filter.

In comparison between ESP and the fabric filter controlled incinerator, it is apparent that the distribution by particle diameter differs greatly from one control device to another. A major uncertainty exists with respect to particulate distributions resulting from increasing the collection plates and electrostatic fields in advanced ESP units. These systems are capable of controlling the mass of particulate emissions to the same level of performance as fabric filters, e.g., approximately 9.0 milligrams per dry standard cubic meter at 12 percent CO_2 (U.S. EPA,1987c). Although specific mass fractionalization studies are lacking, this implies that modern multi-field ESPs can be designed to result in a similar distribution of particulate as fabric filters in the stack emissions.

For purposes of deposition analysis of particulate contaminated with dioxin-like chemicals that are emitted from the hypothetical MSW incinerator, three particle size categories are generalized from the data: $> 10\ \mu$m, 2 to 10 μm, and $< 2\ \mu$m. This particle size distribution will be kept constant in the analysis of dioxin-like compound emissions resulting from the application of the two distinct control scenarios: ESPs, and

Table 6-13. Comparison of particle-size distribution in particulate emissions resulting from ESP controls with particulate emissions from fabric filter controls

ESP Controlled MSW Incinerator[a]		Fabric Filter Controlled MSW Incinerator[b]	
Mean diameter (μm)[c]	% of particulate[d]	Mean diameter (μm)	% of particulate
> 15.0	12.8	> 12	37.8
12.5	10.5	7.5 - 12.0	0.0
8.1	10.4	5.1 - 7.5	1.0
5.5	7.3	3.5 - 5.1	1.5
3.6	10.3	2.3 - 3.5	3.6
2.0	10.5	1.1 - 2.3	2.0
1.1	8.2	2.3 - 0.7	1.0
0.7	7.6	< 0.47	52.6
< 0.7	22.4		

[a]ESP data from U.S. EPA, 1980.
[b]Fabric filter data adapted from Hahn and Hagenmaier, 1986.
[c]Mean particle diameter is geometric mean.
[d]Percent of particulate is percent of total particulate mass.

dry-alkaline scrubbers combined with fabric filters. Not enough particle mass fractionalization data exists to generalize a distribution pattern for each control device based on the available limited measurements. Therefore the distribution from ESPs in Table 6-13 will be applied to the calculation of mass emission rate of particulate-bound congeners by particle size category.

For purposes of estimating deposition of the dioxin-like compounds emitted from the stack of the hypothetical MSW incinerator, the relationship between the pollutant concentration and particle diameter were assumed. The mass emission rate of the dioxin-like congeners is distributed to the particulate matter distribution by calculating proportion and available surface area for a given particle size (assuming that the particles are perfect spheres, and particle density remains constant)(Hart,1985). The particle weight is proportional to volume if density is held constant. Therefore, the ratio of the surface area to volume is proportional to the ratio of the surface area to weight for a particle with a

given radius (Hart,1985). Multiplying this proportion times the weight fraction of particles of a specific diameter (μm) yields a number that approximates the amount of surface area available for chemical adsorption. The logic is as follows:

(a) Assume aerodynamic spherical particles.

(b) Specific surface area of a spherical particle with radius,r:

$$S = 4\pi \ r^2 \qquad (6\text{-}5)$$

(c) Volume of spherical particle with radius, r:

$$V = \left(\frac{4}{3}\right) \pi \ r^3 \qquad (6\text{-}6)$$

(d) The ratio of surface area to volume is:

$$\frac{S}{V} = \frac{4 \ \pi \ r^2}{(\frac{4}{3}) \ \pi \ r^3} = \frac{3}{r} \qquad (6\text{-}7)$$

If particle density is held constant, then it is assumed that particle weight is proportional to particle volume. Hart (1986) further postulates that the ratio of surface area to volume is proportional to the ratio of surface area to weight for a particle with a given radius. Therefore, the ratio of surface area to volume represents the potential relationship between the surface area and the weight of the particle.

When the calculations are summed for all particle size categories, total surface area is assumed for total particle emissions. Dividing the surface area for each particle category by the total available surface area for all particles gives an estimation of the fraction of total area on any size particle. Multiplication of the emission rate of the dioxin-like congener times the fraction of available surface area will estimate the emission rate of the pollutant per particle size. The fraction of total surface area was computed for the three particle size categories. The fraction of total surface areas for the ranges of particle diameters are summed with each particle size category to represent a single fraction of total surface area for the given particle size category, e.g., 0.03 for > 10 μm; 0.095 for 2

to 10 μm and 0.875 for < 2 μm. Thus 87.5 % of the emission rate of the dioxin-like congener is calculated to be associated with particles less than 2 μm in diameter.

In modeling the wet and dry deposition flux, and the ambient air concentrations of the emitted dioxin-like congeners, it is necessary to assume the physical state of the pollutants at the point of emission from the stack. This assumption is derived by the ratio of vapor-phase to particle-bound emissions of these congeners. While the available literature is weak in this area, inferences can be made from field validation studies of the stack sampling procedure for dioxins (Method 23) using $^{13}C_{12}$ labeled PCDD/PDCF congeners as field spikes (MRI,1989). Sampling *in situ* in the stack of the Chicago NW mass burn heat recovery MSW incineratora while using a dynamic spiking system, demonstrated that most of the isotope was recovered in the filter trap and front half of the sampling train designed to capture particulate, and a lower amount was recovered in the XAD resin designed to capture vaporous compounds. In the particular tests in which the overall percent recovery of the dynamic spike were found to be acceptable, the XAD resin contained only about 10% of the isotope. The remainder, 90%, was associated with particulate. Other field validation studies involving recovery of labeled compounds while sampling in the stack of an operating MSW incinerator have shown wide variation in the distribution of spiked congeners (Hagenmaier,1989), depending on the configuration of the sampling apparatus, and the temperature of the sampling probe. Hagenmaier's analysis of a sampling train analogous to EPA Method 23 produced a distribution of dioxin congeners of 15 % to 30 % in vapor form and 70 % to 85 % associated with particulate matter (Hagenmaier,1989). Thus for purposes of assuming vapor/particulate partitioning of the dioxin-like compounds at the point of emission from the stack of the hypothetical MSW incinerator, a vapor to particle ratio of 20 % vapor and 80 % particles is assumed. Given the low vapor pressures of the dioxin-like chemicals, the analysis of condensate and XAD-2 resin from EPA Method 23 could actually be aerosol particles and not true vapors. Primary aerosols, defined as originating from the stack of a combustion source, usually are characterized as mixtures of particles having diameters less than 1.0 μm (Seinfeld, 1984), and within these dimensions such particles may remain in suspension in the lower atmosphere from a few days to several weeks. Wet deposition may be the principal removal process of aerosol particles, however dry deposition will occur by increasing mass through collisions and adhesion. The settling velocities of aerosol particles have been

estimated in the range of from 10^{-3} to 10^{-5} centimeter per second (Seingeld,1984). At these extremely low settling velocities the aerosol particles will be transported by winds to areas beyond microscale dispersion analysis of ISC. To simplify the analysis EPA has assumed that aerosols of less than 1.0 μm will behave like gases, and are modeled accordingly. Therefore aerosols will be equivalent to vapor for purposes of air dispersion modeling, and will result in an ambient air concentration of the pollutants that will be available for direct inhalation exposure by the human receptor.

Multiplication of the vapor/particulate ratio by the emission factor of each dioxin like congener yields the portion of emission in vapor state and that which is associated with particulate matter emissions. Tables 6-4, 6-5, and 6-7 are the average emission factors (grams/Metric ton) of congeners of PCDDs, PCDFs, and PCBs, respectively. The ISCST model accepts emission factors in units of grams per second of time. The hypothetical MSW incinerator burns 0.03 metric tons per second (MT/sec.). Multiplication of the average emission factors derived for specific congeners by 0.03 MT/sec. gives an emission factor in units of grams per second. Table 6-14 displays the congener-specific emission factors for the ESP and the dry scrubber-fabric filter control scenarios.

To estimate the wet and dry deposition flux of the emitted dioxin-like congener, it is necessary to partition the rate of emission by the generalized particle size categories derived for the particulate matter emissions from the hypothetical MSW incinerator. Three general particle size categories are assumed: > 10 μm; 2 to 10 μm; < 2 μm. For the three general particle size categories, the fraction of total surface area available for chemical adsorption is calculated to be: 0.03 for > 10 μm; 0.095 for 2 to 10 μm; and 0.875 for less than 2 μm. Multiplication of the emission rate of the specific congener associated with particulate for each pollution control scenario (Table 6-14) by the fraction of available surface area gives and estimation of the emission rate of the congener per particle size. The emission rate (grams/sec.) of the specific congeners by particle size is shown in Table 6-15 for both the ESP and the dry-scrubber-fabric filter control scenarios of the hypothetical MSW incinerator. These emission rates by particle size are inputs to the ISCST model for calculations of deposition flux.

Table 6-14. Congener-specific emission factors (grams/sec) as input to the ISC air dispersion model for both ESP and DSFF control scenarios

	ESP control		DSFF control	
Congener	Vapor	Particulate	Vapor	Particulate
2,3,7,8-TCDD	2.62E-08	1.05E-07	1.64E-09	6.54E-09
1,2,3,7,8-PeCDD	2.39E-07	9.55E-07	1.49E-08	5.97E-08
1,2,3,4,7,8-HxCDD	5.63E-08	2.25E-07	3.52E-09	1.41E-08
1,2,3,6,7,8-HxCDD	1.37E-07	5.50E-07	8.59E-09	3.44E-08
1,2,3,7,8,9-HxCDD	6.96E-08	2.78E-07	4.35E-09	1.74E-08
1,2,3,4,6,7,8-HpCDD	1.27E-07	5.06E-07	7.91E-09	3.17E-08
Octa-CDD	9.84E-07	3.94E-06	6.15E-08	2.46E-07
2,3,7,8-TCDF	2.95E-07	1.18E-06	1.84E-08	7.37E-08
1,2,3,7,8-PeCDF	3.10E-07	1.24E-06	1.94E-08	7.76E-08
2,3,4,7,8-PeCDF	2.81E-07	1.12E-06	1.76E-08	7.02E-08
1,2,3,4,7,8-HxCDF	1.94E-07	7.75E-07	1.21E-08	4.85E-08
1,2,3,6,7,8-HxCDF	1.52E-07	6.10E-07	9.53E-09	3.81E-08
1,2,3,7,8,9-HxCDF	1.29E-07	5.16E-07	8.06E-09	3.23E-08
2,3,4,6,7,8-HxCDF	1.23E-07	4.92E-07	7.69E-09	3.08E-08
1,2,3,4,6,7,8-HpCDF	1.55E-07	6.22E-07	9.71E-09	3.89E-08
1,2,3,4,7,8,9-HpCDF	4.88E-09	1.95E-08	3.05E-10	1.22E-09
OctaCDF	4.52E-07	1.81E-06	2.83E-08	1.13E-07
3,3',4,4'-TeCB	6.60E-07	2.64E-06	4.13E-08	1.65E-07
3,4,4',5-TeCB	6.60E-07	2.64E-06	4.13E-08	1.65E-07
3,3',4,4',5-PeCB	6.60E-10	2.64E-09	4.13E-11	1.65E-10
2,3,3',4,4'-PeCB	6.60E-10	2.64E-09	4.13E-11	1.65E-10
2,3,4,4',5-PeCB	6.60E-10	2.64E-09	4.13E-11	1.65E-10
2,3',4,4',5-PeCB	6.60E-10	2.64E-09	4.13E-11	1.65E-10
3,3',4,4',5,5'- HxCB	6.00E-07	2.40E-06	3.75E-08	1.50E-07
2,3,3',4,4',5-HxCB	6.00E-07	2.40E-06	3.75E-08	1.50E-07
2,3,3',4,4',5'-HxCB	6.00E-07	2.40E-06	3.75E-08	1.50E-07
2,3,3',4,4',5-HxCB	6.00E-07	2.40E-06	3.75E-08	1.50E-07
2,3,3',4,4',5,5'-HpCB	9.36E-08	3.74E-07	5.85E-09	2.34E-08

Table 6-15. Congener-specific emission factors by particle size (grams/sec) for both the ESP and dry scrubber-fabric filter (DSFF) control scenarios to the hypothetical MSW incinerator

	ESP scenario			DSFF scenario		
Congener	< 2 μm	2 - 10 μm	> 10 μm	< 2μm	2 - 10 μm	> 10 μm
2,3,7,8-TCDD	9.16E-08	9.94E-09	3.14E-09	5.72E-09	6.21E-10	1.96E-10
1,2,3,7,8-PeCDD	8.36E-07	9.07E-08	2.87E-08	5.22E-08	5.67E-09	1.79E-09
1,2,3,4,7,8-HxCDD	1.97E-07	2.14E-08	6.75E-09	1.23E-08	1.34E-09	4.22E-10
1,2,3,6,7,8-HxCDD	4.81E-07	5.22E-08	1.65E-08	3.01E-08	3.26E-09	1.03E-09
1,2,3,7,8,9-HxCDD	2.44E-07	2.64E-08	8.35E-09	1.52E-08	1.65E-09	5.22E-10
1,2,3,4,6,7,8-HpCDD	4.43E-07	4.81E-08	1.52E-08	2.77E-08	3.01E-09	9.50E-10
OctaCDD	3.44E-06	3.74E-07	1.18E-07	2.15E-07	2.34E-08	7.38E-09
2,3,7,8-TCDF	1.03E-06	1.12E-07	3.54E-08	6.44E-08	7.00E-09	2.21E-09
1,2,3,7,8-PeCDF	1.09E-06	1.18E-07	3.72E-08	6.79E-08	7.37E-09	2.33E-09
2,3,4,7,8-PeCDF	9.83E-07	1.07E-07	3.37E-08	6.14E-08	6.67E-09	2.11E-09
1,2,3,4,7,8-HxCDF	6.78E-07	7.36E-08	2.33E-08	4.24E-08	4.60E-09	1.45E-09
1,2,3,6,7,8-HxCDF	5.33E-07	5.79E-08	1.83E-08	3.33E-08	3.62E-09	1.14E-09
1,2,3,7,8,9-HxCDF	4.52E-07	4.90E-08	1.55E-08	2.82E-08	3.06E-09	9.68E-10
2,3,4,6,7,8-HxCDF	4.31E-07	4.67E-08	1.48E-08	2.69E-08	2.92E-09	9.23E-10
1,2,3,4,6,7,8-HpCDF	5.44E-07	5.91E-08	1.86E-08	3.40E-08	3.69E-09	1.17E-09
1,2,3,4,7,8,9-HpCDF	1.71E-08	1.86E-09	5.86E-10	1.07E-09	1.16E-10	3.66E-11
OctaCDF	1.58E-06	1.72E-07	5.43E-08	9.90E-08	1.07E-08	3.39E-09
3,3',4,4'-TeCB	2.31E-06	2.51E-07	7.92E-08	1.44E-07	1.57E-08	4.95E-09
3,4,4',5-TeCB	2.31E-06	2.51E-07	7.92E-08	1.44E-07	1.57E-08	4.95E-09
3,3',4,4',5-PeCB	2.31E-09	2.51E-10	7.92E-11	1.44E-10	1.57E-11	4.95E-12
2,3,3',4,4'-PeCB	2.31E-09	2.51E-10	7.92E-11	1.44E-10	1.57E-11	4.95E-12
2,3,4,4',5-PeCB	2.31E-09	2.51E-10	7.92E-11	1.44E-10	1.57E-11	4.95E-12
2,3',4,4',5-PeCB	2.31E-09	2.51E-10	7.92E-11	1.44E-10	1.57E-11	4.95E-12
3,3',4,4',5,5'-HxCB	2.10E-06	2.28E-07	7.20E-08	1.31E-07	1.43E-08	4.50E-09
2,3,3',4,4',5'-HxCB	2.10E-06	2.28E-07	7.20E-08	1.31E-07	1.43E-08	4.50E-09
2,3,3',4,4',5'-HxCB	2.10E-06	2.28E-07	7.20E-08	1.31E-07	1.43E-08	4.50E-09
2,3,3',4,4',5-HxCB	2.10E-06	2.28E-07	7.20E-08	1.31E-07	1.43E-08	4.50E-09
2,3,3',4,4',5,5'-HpCB	3.28E-07	3.56E-08	1.12E-08	2.05E-08	2.22E-09	7.02E-10

6.7. RESULTS OF THE AIR DISPERSION MODELING OF CONGENER-SPECIFIC EMISSIONS FROM THE HYPOTHETICAL MSW INCINERATOR IN TAMPA, FLORIDA

The dispersion and deposition computations performed by the ISCST model require data on wind speed, wind direction, and the intensity of atmospheric turbulence. The turbulence intensity is represented by the atmospheric stability class. These data are provided by a permanent data base in the ISCST model known as the Stability Array or STAR data set. Martin and Tikvart (1968) developed the STAR program from routinely collected meteorological data to generate frequencies and percentage frequencies of wind direction by speed classes for each stability category. The specifications of stability categories depending on wind speed and sky cover were established by Pasquill (1961), and later modified by Turner (1964). The program was adopted for use at the National Climatic Center (NCC), where archived records of all national reporting weather stations are kept. The most current version of the STAR data from all STAR stations in the country were used to produce the matrices of STAR frequencies used in the ISCST (Martin,1968). The format of the STAR data includes sixteen wind directions, six wind speed classes, and seven stability categories with categories A,B,C, and Dday in the daytime, and categories Dnight, E, and F in the nighttime. There are data for 312 stations geographically distributed in the U.S.

Meteorological data recorded at the Tampa Airport STAR station was used in the dispersion modeling of stack gas and particulate emissions of dioxin-like congeners from the hypothetical MSW incinerator. Inputs for the model included hourly meteorological data, characteristics and receptor features. Hourly meteorological data requirements are the mean wind speed, the direction towards which the wind is blowing, the wind-profile exponent, the ambient air temperature, the Pasquill stability category, the vertical potential temperature gradient with height, the mixing layer height, and the frequency distribution of hourly precipitation. Source input data requirements included the congener-specific mass emission rate partitioned by vapor and particulate; the physical stack measurements, e.g., diameter, base elevation of the stack, and exit velocity and temperature of the stack gas; dimensions of the incinerator building, and settling parameters for particulate matter for both dry and wet deposition.

The output of the ISCLT model for both surface deposition and ambient air impacts is a concentration array for 160 ground-level receptors around the incinerator, e.g., 10

receptor points along each of the 16 wind directions every 22.5° on the polar azimuth. For the assumed vaporous state of the congener emissions the ambient air, ground-level concentrations are in units of grams per cubic meter of air (g/m3), and for the assumed portion of emissions associated with particulate matter the concentration estimates are in units of grams per square meter of surface area (g/m2). Because these concentrations are expressed as annual averages using annual average meteorological data, the concentration units are g/m3·yr., and g/m2·yr. Ground-level concentrations for both ambient air and surface deposition were estimated at concentric radial distances from the incinerator of 0.2, 0.5, 0.8, 1.0, 2.0, 5.0, 10, 20, 30, and 40 kilometers. Ambient air and surface deposition modeling of emissions for the hypothetical incinerator located in Tampa, Florida were estimated for both pollution control scenarios: (1) ESP; (2) dry scrubber combined with fabric filter (DSFF).

The air dispersion modeling for both the ESP and DSFF control scenario shows that the maximum ambient air and surface deposition receptor concentration is 200 meters east of the incinerator, corresponding to a westerly wind direction. If this point of impact is referenced as 1.0, then the concentration gradient at each concentric radial distance in the easterly direction is a fraction of 1.0. Figures 6-2, and 6-3 display the ratio of concentration beyond the 200 meters maximum ground-level concentration resulting from the ESP and DSFF control scenarios, respectively. For example, at the radial distance of 0.8 km in the ESP scenario, approximately 0.5 (50 %) of the maximum ground-level concentration at 0.2 km is estimated by the ISCST model.

Ambient air concentrations of dioxin-like congeners at ground-level are expressed in units of grams per cubic meter of air (g/m3), and are annual average concentrations. The ambient air concentration results from the emission of specific congeners in the vapor state, and it is assumed that the congener will remain in the vapor state, or be associated with aerosol particles less than 1.0 μm in diameter. The annual average ambient air concentration is directly available for human inhalation exposure. The maximum ground-level ambient air concentration of all modeled congeners is estimated to occur 200 meters in a westerly wind direction from the center of the stack. As Figures 6-2 and 6-3 indicate, most of the ground-level impact associated with the stack emissions from the hypothetical MSW incinerator will occur within 5 kilometers from the facility for both control scenarios. Tables 6-16 and 6-17 display the annual average ambient air concentrations of specific

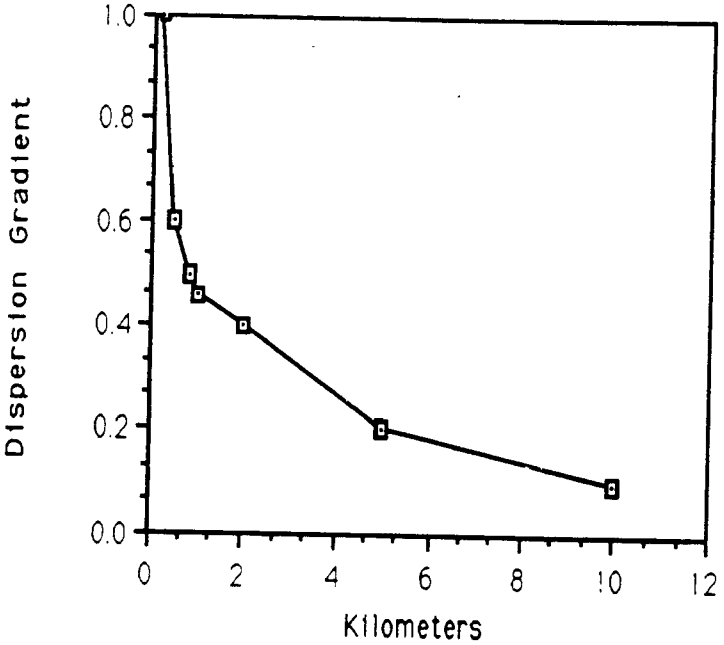


Figure 6-2. Dispersion gradient of the ground-level concentration with distance from the incinerator-ESP control scenario.

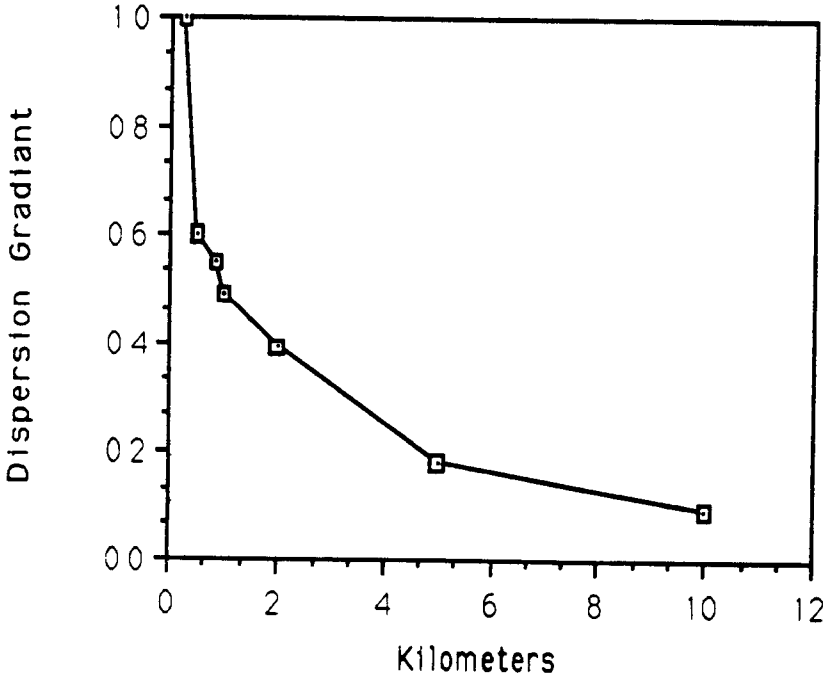


Figure 6-3. Dispersion gradient of the ground-level concentration with distance from the incinerator-DSFF control scenario.

Table 6-16. Predicted annual average ambient air concentrations (grams/m^3) of specific dioxin-like congeners at specified distances from the hypothetical MSW incinerator--ESP controls

	Ambient air concentrations (grams/m^3)--ESP controls					
Congeners	0.2 km	0.5 km	0.8 km	1 km	2 km	5 km
2,3,7,8-TCDD	1.65E-15	9.91E-16	8.26E-16	7.60E-16	6.61E-16	3.30E-16
1,2,3,7,8-PeCDD	1.51E-14	9.05E-15	7.54E-15	6.94E-15	6.03E-15	3.02E-15
1,2,3,4,7,8-HxCDD	3.55E-15	2.13E-15	1.78E-15	1.64E-15	1.42E-15	7.11E-16
1,2,3,6,7,8-HxCDD	8.68E-15	5.21E-15	4.34E-15	3.99E-15	3.47E-15	1.74E-15
1,2,3,7,8,9-HxCDD	4.40E-15	2.64E-15	2.20E-15	2.02E-15	1.76E-15	8.79E-16
1,2,3,4,6,7,8-HpCDD	8.00E-15	4.80E-15	4.00E-15	3.68E-15	3.20E-15	1.60E-15
OctaCDD	6.21E-14	3.73E-14	3.11E-14	2.86E-14	2.49E-14	1.24E-14
2,3,7,8-TCDF	1.86E-14	1.12E-14	9.30E-15	8.56E-15	7.44E-15	3.72E-15
1,2,3,7,8-PeCDF	1.96E-14	1.18E-14	9.80E-15	9.01E-15	7.84E-15	3.92E-15
2,3,4,7,8-PeCDF	1.77E-14	1.06E-14	8.87E-15	8.16E-15	7.09E-15	3.55E-15
1,2,3,4,7,8-HxCDF	1.22E-14	7.34E-15	6.12E-15	5.63E-15	4.90E-15	2.45E-15
1,2,3,6,7,8-HxCDF	9.63E-15	5.78E-15	4.81E-15	4.43E-15	3.85E-15	1.93E-15
1,2,3,7,8,9-HxCDF	8.15E-15	4.89E-15	4.07E-15	3.75E-15	3.26E-15	1.63E-15
2,3,4,6,7,8-HxCDF	7.77E-15	4.66E-15	3.88E-15	3.57E-15	3.11E-15	1.55E-15
1,2,3,4,6,7,8-HpCDF	9.81E-15	5.89E-15	4.91E-15	4.51E-15	3.93E-15	1.96E-15
1,2,3,4,7,8,9-HpCDF	3.08E-16	1.85E-16	1.54E-16	1.42E-16	1.23E-16	6.17E-17
OctaCDF	2.86E-14	1.71E-14	1.43E-14	1.31E-14	1.14E-14	5.71E-15
3,3',4,4'-TeCB	4.17E-14	2.50E-14	2.08E-14	1.92E-14	1.67E-14	8.34E-15
3,4,4',5-TeCB	4.17E-14	2.50E-14	2.08E-14	1.92E-14	1.67E-14	8.34E-15
3,3',4,4',5-PeCB	4.17E-17	2.50E-17	2.08E-17	1.92E-17	1.67E-17	8.34E-18
2,3,3',4,4'-PeCB	4.17E-17	2.50E-17	2.08E-17	1.92E-17	1.67E-17	8.34E-18
2,3,4,4',5-PeCB	4.17E-17	2.50E-17	2.08E-17	1.92E-17	1.67E-17	8.34E-18
2,3',4,4',5-PeCB	4.17E-17	2.50E-17	2.08E-17	1.92E-17	1.67E-17	8.34E-18
3,3',4,4',5,5'-HxCB	3.79E-14	2.27E-14	1.89E-14	1.74E-14	1.52E-14	7.58E-15
2,3,3',4,4',5-HxCB	3.79E-14	2.27E-14	1.89E-14	1.74E-14	1.52E-14	7.58E-15
2,3',4,4',5,5-HxCB	3.79E-14	2.27E-14	1.89E-14	1.74E-14	1.52E-14	7.58E-15
2,3',4,4',5,5'-HxCB	3.79E-14	2.27E-14	1.89E-14	1.74E-14	1.52E-14	7.58E-15
2,3,3',4,4',5,5'-HCB	5.91E-15	3.55E-15	2.96E-15	2.72E-15	2.36E-15	1.18E-15

Table 6-17. Predicted annual average ambient air concentrations (grams/m^3) of specific dioxin-like congeners at specified distances from the hypothetical MSW incinerator--DSFF controls

	Ambient air concentrations (grams/m^3)--DSFF controls					
Congeners	0.2 km	0.5 km	0.8 km	1 km	2 km	5 km
2,3,7,8-TCDD	1.03E-16	6.20E-17	5.16E-17	4.75E-17	4.13E-17	2.07E-17
1,2,3,7,8-PeCDD	9.43E-16	5.66E-16	4.71E-16	4.34E-16	3.77E-16	1.89E-16
1,2,3,4,7,8-HxCDD	2.22E-16	1.33E-16	1.11E-16	1.02E-16	8.89E-17	4.44E-17
1,2,3,6,7,8-HxCDD	5.42E-16	3.25E-16	2.71E-16	2.49E-16	2.17E-16	1.08E-16
1,2,3,7,8,9-HxCDD	2.75E-16	1.65E-16	1.37E-16	1.26E-16	1.10E-16	5.49E-17
1,2,3,4,6,7,8-HpCDD	5.00E-16	3.00E-16	2.50E-16	2.30E-16	2.00E-16	9.99E-17
OctaCDD	3.88E-15	2.33E-15	1.94E-15	1.79E-15	1.55E-15	7.77E-16
2,3,7,8-TCDF	1.16E-15	6.98E-16	5.81E-16	5.35E-16	4.65E-16	2.33E-16
1,2,3,7,8-PeCDF	1.22E-15	7.35E-16	6.12E-16	5.63E-16	4.90E-16	2.45E-16
2,3,4,7,8-PeCDF	1.11E-15	6.65E-16	5.54E-16	5.10E-16	4.43E-16	2.22E-16
1,2,3,4,7,8-HxCDF	7.65E-16	4.59E-16	3.83E-16	3.52E-16	3.06E-16	1.53E-16
1,2,3,6,7,8-HxCDF	6.02E-16	3.61E-16	3.01E-16	2.77E-16	2.41E-16	1.20E-16
1,2,3,7,8,9-HxCDF	5.09E-16	3.06E-16	2.55E-16	2.34E-16	2.04E-16	1.02E-16
2,3,4,6,7,8-HxCDF	4.86E-16	2.91E-16	2.43E-16	2.23E-16	1.94E-16	9.71E-17
1,2,3,4,6,7,8-HpCDF	6.13E-16	3.68E-16	3.07E-16	2.82E-16	2.45E-16	1.23E-16
1,2,3,4,7,8,9-HpCDF	1.93E-17	1.16E-17	9.64E-18	8.87E-18	7.71E-18	3.86E-18
OctaCDF	1.79E-15	1.07E-15	8.93E-16	8.21E-16	7.14E-16	3.57E-16
3,3',4,4'-TeCB	2.61E-15	1.56E-15	1.30E-15	1.20E-15	1.04E-15	5.21E-16
3,4,4',5-TeCB	2.61E-15	1.56E-15	1.30E-15	1.20E-15	1.04E-15	5.21E-16
3,3',4,4',5-PeCB	2.61E-18	1.56E-18	1.30E-18	1.20E-18	1.04E-18	5.21E-19
2,3,3',4,4'-PeCB	2.61E-18	1.56E-18	1.30E-18	1.20E-18	1.04E-18	5.21E-19
2,3,4,4',5-PeCB	2.61E-18	1.56E-18	1.30E-18	1.20E-18	1.04E-18	5.21E-19
2,3',4,4',5-PeCB	2.61E-18	1.56E-18	1.30E-18	1.20E-18	1.04E-18	5.21E-19
3,3',4,4',5,5'-HxCB	2.37E-15	1.42E-15	1.18E-15	1.09E-15	9.47E-16	4.74E-16
2,3,3',4,4',5-HxCB	2.37E-15	1.42E-15	1.18E-15	1.09E-15	9.47E-16	4.74E-16
2,3',4,4',5,5-HxCB	2.37E-15	1.42E-15	1.18E-15	1.09E-15	9.47E-16	4.74E-16
2,3',4,4',5,5'-HxCB	2.37E-15	1.42E-15	1.18E-15	1.09E-15	9.47E-16	4.74E-16
2,3,3',4,4',5,5'-HCB	3.69E-16	2.22E-16	1.85E-16	1.70E-16	1.48E-16	7.39E-17

dioxin-like congeners emitted from the stack under the ESP and DSFF control scenarios.

Surface deposition flux is expressed in units of grams per square meter per year. The deposition flux is the summation of both dry and wet removal processes, and is an annual average concentration. Tables 6-18 and 6-19 display the spacial deposition flux in the vicinity of the incinerator in the ESP and DSFF control scenarios.

Table 6-18. Wet + dry surface deposition (grams/m^2.yr) of specific congeners at specified distances in the vicinity of the hypothetical MSW incinerator--ESP controls

	Surface deposition (grams/m^2·yr)--ESP control scenario					
Congeners	0.2 km	0.5 km	0.8 km	1 km	2 km	5 km
2,3,7,8-TCDD	3.52E-08	2.11E-08	1.76E-08	1.62E-08	1.41E-08	7.04E-09
1,2,3,7,8-PeCDD	3.21E-07	1.93E-07	1.61E-07	1.48E-07	1.29E-07	6.43E-08
1,2,3,4,7,8-HxCDD	7.57E-08	4.54E-08	3.79E-08	3.48E-08	3.03E-08	1.51E-08
1,2,3,6,7,8-HxCDD	1.85E-07	1.11E-07	9.24E-08	8.50E-08	7.39E-08	3.70E-08
1,2,3,7,8,9-HxCDD	9.36E-08	5.62E-08	4.68E-08	4.31E-08	3.75E-08	1.87E-08
1,2,3,4,6,7,8-HpCDD	1.70E-07	1.02E-07	8.52E-08	7.84E-08	6.81E-08	3.41E-08
OctaCDD	1.32E-06	7.94E-07	6.62E-07	6.09E-07	5.30E-07	2.65E-07
2,3,7,8-TCDF	3.96E-07	2.38E-07	1.98E-07	1.82E-07	1.59E-07	7.93E-08
1,2,3,7,8-PeCDF	4.17E-07	2.50E-07	2.09E-07	1.92E-07	1.67E-07	8.35E-08
2,3,4,7,8-PeCDF	3.78E-07	2.27E-07	1.89E-07	1.74E-07	1.51E-07	7.56E-08
1,2,3,4,7,8-HxCDF	2.61E-07	1.56E-07	1.30E-07	1.20E-07	1.04E-07	5.21E-08
1,2,3,6,7,8-HxCDF	2.05E-07	1.23E-07	1.03E-07	9.43E-08	8.20E-08	4.10E-08
1,2,3,7,8,9-HxCDF	1.74E-07	1.04E-07	8.68E-08	7.98E-08	6.94E-08	3.47E-08
2,3,4,6,7,8-HxCDF	1.65E-07	9.93E-08	8.27E-08	7.61E-08	6.62E-08	3.31E-08
1,2,3,4,6,7,8-HpCDF	2.09E-07	1.25E-07	1.05E-07	9.62E-08	8.36E-08	4.18E-08
1,2,3,4,7,8,9-HpCDF	6.57E-09	3.94E-09	3.29E-09	3.02E-09	2.63E-09	1.31E-09
OctaCDF	6.09E-07	3.65E-07	3.04E-07	2.80E-07	2.43E-07	1.22E-07
3,3',4,4'-TeCB	8.88E-07	5.33E-07	4.44E-07	4.08E-07	3.55E-07	1.78E-07
3,4,4',5-TeCB	8.88E-07	5.33E-07	4.44E-07	4.08E-07	3.55E-07	1.78E-07
3,3',4,4',5-PeCB	8.88E-10	5.33E-10	4.44E-10	4.08E-10	3.55E-10	1.78E-10
2,3,3',4,4'-PeCB	8.88E-10	5.33E-10	4.44E-10	4.08E-10	3.55E-10	1.78E-10
2,3,4,4',5-PeCB	8.88E-10	5.33E-10	4.44E-10	4.08E-10	3.55E-10	1.78E-10
2,3',4,4',5-PeCB	8.88E-10	5.33E-10	4.44E-10	4.08E-10	3.55E-10	1.78E-10
3,3',4,4',5,5'-HxCB	8.07E-07	4.84E-07	4.04E-07	3.71E-07	3.23E-07	1.61E-07
2,3,3',4,4',5-HxCB	8.07E-07	4.84E-07	4.04E-07	3.71E-07	3.23E-07	1.61E-07
2,3',4,4',5,5-HxCB	8.07E-07	4.84E-07	4.04E-07	3.71E-07	3.23E-07	1.61E-07
2,3',4,4',5,5'-HxCB	8.07E-07	4.84E-07	4.04E-07	3.71E-07	3.23E-07	1.61E-07
2,3,3',4,4',5,5'-HpCB	1.26E-07	7.56E-08	6.30E-08	5.79E-08	5.04E-08	2.52E-08

Table 6-19. Wet+dry surface deposition (grams/m^2.yr) of specific congeners at specified distances in the vicinity of the hypothetical MSW incinerator--DSFF controls

Congeners	Surface deposition (grams/m^2.yr)--DSFF control scenario					
	0.2 km	0.5 km	0.8 km	1 km	2 km	5 km
2,3,7,8-TCDD	2.20E-09	1.32E-09	1.10E-09	1.01E-09	8.80E-10	4.40E-10
1,2,3,7,8-PeCDD	2.01E-08	1.20E-08	1.00E-08	9.24E-09	8.03E-09	4.02E-09
1,2,3,4,7,8-HxCDD	4.73E-09	2.84E-09	2.37E-09	2.18E-09	1.89E-09	9.47E-10
1,2,3,6,7,8-HxCDD	1.16E-08	6.93E-09	5.78E-09	5.31E-09	4.62E-09	2.31E-09
1,2,3,7,8,9-HxCDD	5.85E-09	3.51E-09	2.93E-09	2.69E-09	2.34E-09	1.17E-09
1,2,3,4,6,7,8-HpCDD	1.06E-08	6.39E-09	5.32E-09	4.90E-09	4.26E-09	2.13E-09
OctaCDD	8.27E-08	4.96E-08	4.14E-08	3.81E-08	3.31E-08	1.65E-08
2,3,7,8-TCDF	2.48E-08	1.49E-08	1.24E-08	1.14E-08	9.91E-09	4.95E-09
1,2,3,7,8-PeCDF	2.61E-08	1.57E-08	1.30E-08	1.20E-08	1.04E-08	5.22E-09
2,3,4,7,8-PeCDF	2.36E-08	1.42E-08	1.18E-08	1.09E-08	9.45E-09	4.72E-09
1,2,3,4,7,8-HxCDF	1.63E-08	9.78E-09	8.15E-09	7.50E-09	6.52E-09	3.26E-09
1,2,3,6,7,8-HxCDF	1.28E-08	7.69E-09	6.41E-09	5.90E-09	5.13E-09	2.56E-09
1,2,3,7,8,9-HxCDF	1.08E-08	6.51E-09	5.42E-09	4.99E-09	4.34E-09	2.17E-09
2,3,4,6,7,8-HxCDF	1.03E-08	6.21E-09	5.17E-09	4.76E-09	4.14E-09	2.07E-09
1,2,3,4,6,7,8-HpCDF	1.31E-08	7.84E-09	6.53E-09	6.01E-09	5.23E-09	2.61E-09
1,2,3,4,7,8,9-HpCDF	4.11E-10	2.46E-10	2.05E-10	1.89E-10	1.64E-10	8.21E-11
OctaCDF	3.80E-08	2.28E-08	1.90E-08	1.75E-08	1.52E-08	7.61E-09
3,3',4,4'-TeCB	5.55E-08	3.33E-08	2.78E-08	2.55E-08	2.22E-08	1.11E-08
3,4,4',5-TeCB	5.55E-08	3.33E-08	2.78E-08	2.55E-08	2.22E-08	1.11E-08
3,3',4,4',5-PeCB	5.55E-11	3.33E-11	2.78E-11	2.55E-11	2.22E-11	1.11E-11
2,3,3',4,4'-PeCB	5.55E-11	3.33E-11	2.78E-11	2.55E-11	2.22E-11	1.11E-11
2,3,4,4',5-PeCB	5.55E-11	3.33E-11	2.78E-11	2.55E-11	2.22E-11	1.11E-11
2,3',4,4',5-PeCB	5.55E-11	3.33E-11	2.78E-11	2.55E-11	2.22E-11	1.11E-11
3,3',4,4',5,5'-HxCB	5.05E-08	3.03E-08	2.52E-08	2.32E-08	2.02E-08	1.01E-08
2,3,3',4,4',5-HxCB	5.05E-08	3.03E-08	2.52E-08	2.32E-08	2.02E-08	1.01E-08
2,3',4,4',5,5-HxCB	5.05E-08	3.03E-08	2.52E-08	2.32E-08	2.02E-08	1.01E-08
2,3',4,4',5,5'-HxCB	5.05E-08	3.03E-08	2.52E-08	2.32E-08	2.02E-08	1.01E-08
2,3,3',4,4',5,5'-HCB	7.87E-09	4.72E-09	3.94E-09	3.62E-09	3.15E-09	1.57E-09

REFERENCES FOR CHAPTER 6

ASME. (1981) Study on state-of-the art of dioxin from combustion sources, Arthur D. Little, Inc. for the American Society of Mechanical Engineers, New York, NY.

Briggs, G.A. (1979) Plume rise. USAEC Critical Review Series. NTIS publication no. TID-25075.

Briggs, G.A. (1975) Plume rise predictions. In: Lectures on air pollution and environmental impact analyses, American Meteorology Society.

Bruce, K.R.; Beach, L.O.; Gullet, B.K. (1991) The role of gas-phase Cl_2 in the formation of PCDD/PCDF during waste combustion. Waste Management 11: 97-102.

Brunner, C. (1984) Incineration systems, selection and design. New York, NY: Van Nostrand Reinhold Co.

Choudhry,G.G.; Hutzinger, O. (1983) Mechanistic aspects of the thermal formation of halogenated organic compounds including polychlorinated dibenzo-p-dioxins. New York, NY: Gordon and Breach.

Chrostowski, P.C.; Sager, S.L. (1991) Management of ash from municipal solid waste combustion.In: Hattermer-Frey, H.A.; Travis,C., eds., Health effects of municipal waste incineration. Boca Raton,FL: CRC Press, pp. 249-264.

Cleverly, D.H. (1984) Chlorinated dibenzo-p-dioxins and furans in incineration of municipal solid waste. Proceedings of workshop on energy from municipal waste research: A technical review of thermochemical systems, February 22-24,1984, Kissimmee, FL. Argonne National Lab, U.S. Department of Energy, pp. 295-319.

Cleverly, D.H.; Morrison, R.M.; Riddle, B.L.; Kellam, R.G. (1989) Regulatory analysis of pollutant emissions, including polychlorinated dibenzo-p-dioxins (CDDs) and dibenzofurans (CDFs), from the stacks of municipal waste combustors. Chemosphere 18: 1143-1153.

Cleverly, D.H.; Morrison, R.M.; Riddle, B.L.; Kellam, R.G. (1991) Regulatory analysis of pollutant emissions, including polychlorinated dibenzo-p-dioxins (CDDs) and dibenzofurans (CDFs), from municipal waste combustors. In: Hattermer-Frey, H.A.; Travis,C., eds., Health effects of municipal waste incineration. Boca Raton, FL: CRC Press, pp. 47-65.

Commoner, B.; Shapiro, K.; Webster, T. (1987) The origin and health risks of PCDD and PCDF. Waste Management Res. 5: 327-346.

Dumbauld, R.K.; Rafferty, J.E.; Cramer, H.E. (1976) Dispersion deposition from aerial spray releases. Preprint volume for the Third Symposium on Atmospheric Diffusion and Air Quality. American Meteorolical Society.

Environment Canada. (1985) The national incineration testing and evaluation program: Two-stage combustion (Prince Edward Island) Environment Canada, Ottawa, Ontario, Canada, Report EPS 3/UP/1, September, 1985.

Entropy. (1987) Stationary source sampling report. Signal Resco Pinnellas County resource recovery facility, St. Petersburg, FL. EEI, REF. #5286-B, Entropy Environmental Inc., Research Triangle Park, NC, September, 1987.

Gardiner, W. (1982) The chemistry of flames. Sci. Am. February: 112.

Gullet, B.K.; Bruce, K.R.; Beach., L.O. (1990a) Formation of chlorinated organics during solid waste combustion. Waste Management Res. 8: 203-214.

Gullett, B.K.; Bruce, K.R., Beach, L.O. (1990b) The effect of metal catalysts on the formation of polychlorinated dibenzo-p-dioxin and polychlorinated dibenzofuran precursors. Chemosphere 20: 1945-1952.

Gullet, B.K.; Bruce, K.R.; Beach. L.O. (1991a) The effect of sulfur compounds on the formation mechanism of PCDD and PCDF in municipal waste combustors. In: Conference papers from the second international conference on municipal waste combustion, April 15-19, 1991, Tampa, FL. Air and Waste Management Association, Pittsburgh, PA. pp.16-34.

Gullet, B.K.; Bruce, K.R.; Beach. L.O.; Drago,A. (1991b) Mechanistic steps in the production of PCDD and PCDF during waste combustion. In: Abstracts of the 11th international symposium on chlorinated dioxins and related compounds, September 23-27, 1991, Research Triangle Park, NC, University of North Carolina, Chapel Hill, NC, p. 298.

Hagemaier, H.; Brunner, H.; Haag, R.; Kraft, M.; Lutzke, K. (1987) Problems associated with the measurement of PCDD and PCDF emissions from waste incineration plants. Waste Management Res. 5: 239-250.

Hahn, J.L. (1986) Air emissions testing at the Wurzburg, West Germany waste-to-energy facility. Presented at the annual meeting of the Air Pollution Control Association; June.

Hart, F.C. (1984) Assessment of potential public health impacts associated with predicted emissions of polychlorinated dibenzodioxins and polychlorinated dibenzofurans from the Brooklyn Navy Yard resource recovery facility. Prepared for the New York City Department of Sanitation, Fred C. Hart, Associates, New York, NY.

Hay, D.J.; Finkelstein, A.; Klicius, R. (1986) The national incineration testing and evaluation program: An assessment of two-stage incineration and pilot scale emission control. Paper presented to U.S. EPA Science Advisory Board, Washington,DC. Environment Canada, Ottawa, Ontario, Canada. June, 1986.

Kapahi, R. (1991) Modeling the dispersion of toxic air pollutants emitted from municipal solid waste incinerators. In: Hattermer-Frey, H.A.; Travis,C., eds., Health effects of municipal waste incineration. Boca Raton, FL: CRC Press, pp. 67-82.

Knisley, D.R.; Jamgochian, C.G.; Gergen, W.P.; Holder, D.J. (1986) Draft emissions test report dioxin/furans and total organic chlorides emission testing. Saugus resource recovery facility, Saugus, MA. Prepared by Radian Corporation for Office of Air Quality Planning and Standards, U.S. EPA, Research Triangle Park, NC, DCN No. 86-223,015-05. October 2, 1986.

Magagni, A.; Boschi, G.; Cocheo, V. (1990) Emissions of a MSW incinerator equipped with a post-combustion chamber, dry scrubber and ESP. Chemosphere 20: 1882-1890.

Marklund, S.; Kjeller, L.O.; Hansson, M.; Tysklind, M.; Rappe, C. (1985) Determination of PCDDs and PCDFs in incineration samples and pyrolytic products, Presented at American Chemical Society National meeting, Chlorinated dioxins and dibenzofurans in the total environment, Miami, Fl,April, 1985.

Martin, D.O.; Tikvart, J.A. (1968) A general atmospheric diffusion model for estimating the effects on air quality of one or more sources. Presented at the 61st annual air pollution control association meeting, St. Paul, MN. June, 1968.

MRI. (1989) Validation of emission test method for PCDDs and PCDFs. Prepared by Midwest Research Institute, Kansas City, MO., for Invironmental Monitoring Systems Laboratory, U.S. EPA, Research Triangle Park, NC. EPA contract # 68-02-4395. February 24, 1989.

Mukerjee, D.; Cleverly, D.H. (1987) Risk from exposure to polychlorinated dibenzo-p-dioxins and dibenzofurans emitted from municipal incinerators. Waste Management Res. 5:269-283.

OTA. (1989) Facing America's trash. What next for municipal solid waste? Office of Technology Assessment,U.S.Congress, Washington, DC.

Pasquill, F. (1961) The estimation of the dispersion of windborne material. Meteorology 90: 33-49.

Radke, L.F.; Hobbs, P.V.; Eltgroth, M.W. (1980) Scavenging of aerosol particles by precipitation. J. Applied Meteorol. 19: 715-722.

Seelinger, R.; Hahn, J.; VonDemfange, H.P.; Zurlinden, R.A. (1986) Environmental test report, Ogden Martin Systems of Tulsa, Inc., Compliance with Tulsa City/County health department and U.S. EPA permits. September 9, 1986.

Seinfeld, J.H. (1986) Atmospheric chemistry and physics of air pollution. New York, NY: John Wiley and Sons.

Siebert, P.C.; Alston-Guiden, D.; Jones, K.H. (1991) An analysis of worldwide resource recovery emissions data and the implications for risk assessment. In: Hattermer-Frey, H.A.; Travis,C., eds., Health effects of municipal waste incineration. Boca Raton, FL:, CRC Press, pp. 2-46.

Turner, D.B. (1964) A diffusion model for an urban area. Atmospheric dispersion estimates. 6th printing. Washington, DC: U.S. Government Printing Office, Publication no. AP-26.

U.S. Environmental Protection Agency. (1980) Environmental assessment of a waste-to-energy process. Braintree municipal incinerator. Office of Research and Development, Washington, DC., EPA-600/7-80-149.

U.S. Environmental Protection Agency. (1983) Comprehensive assessment of the specific compounds present in combustion processes. Volume 1: Pilot study of combustion emission variability. Office of Toxic Substances, Washington, DC, EPA-560/5-83-004, June, 1983.

U.S. Environmental Protection Agency. (1984) Thermal degradation products from dielectric fluids. Office of Toxic Substances, Washington,DC.,EPA-560/5-84-009,September, 1984.

U.S. Environmental Protection Agency. (1986) Industrial source complex (ISC) dispersion model user's guide-second edition. Office of Air Quality Planning and Standards,Research Triangle Park, NC, EPA-450/4-86-005a.

U.S. Environmental Protection Agency. (1987a) Characterization of the municipal waste combustion industry. Office of Solid Waste and Emergency Response, Washington,DC: EPA-530-SW-87-021h. June, 1987.

U.S. Environmental Protection Agency. (1987b) Assessment of health risks associated with municipal waste combustion emissions. Office of Solid Waste and Emergency Response, Office of Air and Radiation, Office of Research and Development, Washington,DC., EPA/530-SW-87-021g, September, 1987.

U.S. Environmental Protection Agency. (1987c) Emission data base for municipal waste combustors.Office of Solid Waste and Emergency Response, Office of Air and Radiation, Office of Research and Development, Washington,DC., EPA/530-SW-87-021b, June, 1987.

U.S. Environmental Protection Agency. (1987d) National dioxin study tier 4-combustion sources. Engineering analysis report. Office of Air Quality Planning and Standards, Research Triangle Park, NC, EPA-450/4-84-014h, September, 1987.

U.S. Environmental Protection Agency. (1988a) Municipal waste combustion multipollutant study. Summary report. Signal Environmental Systems, Inc., North Andover RESCO, North Andover,MA. Office of Air Quality Planning and Standards, Research Triangle Park, NC, EMB Report No. 86-MIN-02A, March, 1988.

U.S. Environmental Protection Agency. (1988b) Municipal waste combustion multipollutant study. Summary report.Marion County solid waste-to-energy facility, Brooks, OR. Office of Air Quality Planning and Standards, Research Triangle Park, NC, EMB Report No. 86-MIN-03A,September, 1988.

U.S. Environmental Protection Agency. (1989) Background paper, municipal waste combustors air emission standards. Office of Air Quality Planning and Standards, Research Triangle Park, NC. April 26, 1989.

U.S. Environmental Protection Agency. (1990) The solid waste dilemma: Solutions for the 90's. Preliminary Draft for focus group comment. Office of Solid Waste Municipal Solid Waste Program, Washington, DC. August, 1990.

U.S. Environmental Protection Agency; Environment Canada. (1991) The environmental characterization of RDF combustion technology. Mid-Connecticut facility, Hartford,CT. Test program and results, volume II., June 1991.

Vogg, H.: Metzger, M.; Steiglitz, L. (1987) Recent findings on the formation and decomposition of PCDD/PCDF in municipal solid waste incineration. Waste Management Res. 5: 285-294.

7. EXPOSURE SCENARIO DEVELOPMENT

7.1. INTRODUCTION

EPA (1991a) states, "In exposure scenario evaluation, the assessor attempts to determine the concentrations of chemicals in a medium or location and link this information with the time that individuals or populations contact the chemical. The set of assumptions about how this contact takes place is an exposure scenario." These assumptions can be made many different ways producing a wide variety of scenarios and associated exposure levels. The number of people exposed at different levels form a distribution of exposures. Ideally assessors would develop this entire distribution to fully describe the exposed population. The necessary information is rarely available however, and instead EPA (1991a) recommends developing a central and high end scenario to provide some idea of the possible range of exposure levels. The purpose of this chapter is to illustrate this procedure as applied to the dioxin-like compounds. In addition this Chapter identifies the exposure pathways which are relevant to these compounds, and provides background and justification for the exposure parameters which were selected for the demonstrations in Chapter 9.

7.2. STRATEGY FOR DEVISING EXPOSURE SCENARIOS

For any physical setting, a wide variety of exposure scenarios are possible. The range of exposure levels results from a number of different factors including individual behavior patterns, proximity of individuals to the source of contamination, the fate characteristics of the contaminant, and others. In order to illustrate the possible range, the assessor should try to characterize a central and high end scenario. The general strategy recommended here for defining these scenarios is to first identify and quantify the source of contamination. Next, the assessor should determine the geographic area that is impacted by this source. The contaminant levels are likely to vary widely over this area. Select locations of interest within this area such as the location of the nearest exposed individual or most heavily populated area. For each of these locations, identify behavior patterns which characterize central and high end exposure patterns. Central scenarios correspond to average or median levels and high end scenarios are defined as levels above the 90th percentile but within the actual range of exposure levels. Statistical data are

rarely available to precisely define such scenarios. Instead judgement is usually required to identify behavior patterns meeting these criteria. For example, most rural areas probably include both farming and nonfarming residents. Farmers who grow or raise much of their own food could be selected to represent the high end scenario and those living in typical residential areas could represent the central scenario. Alternatively, if more detail is desired, central and high end scenarios could be defined for both segments of the population, i.e., farmers and residents. For each scenario, determine relevant exposure pathways and assign values for exposure parameters such as contact rate, exposure duration, and so on, which represent average values for the type of receptor. Finally, compute the associated exposure level. The resulting range of exposure levels for each location can be used to illustrate the possible range of exposures.

Reference will be made in this chapter to the example scenarios found in Chapter 9 Demonstration of Methodology. Four "source categories", categories of contamination sources described in Chapter 5, are demonstrated in Chapter 9. The on-site soil and stack emission sources are assumed to expose a relatively large population in a rural area containing residences and a few farms. For these sources, both central and high end scenarios are defined in the manner outlined above. Specifically, a central scenario is based on typical behavior at a residence and the high end is based on a farm family that raises a portion of its own food. For the other two sources, off-site soil and ash landfill, the principally exposed population is assumed to include individuals residing at one or two farms located near the site. Other individuals residing further from these source categories might be exposed via consumption of fish or water from a nearby impacted water body, trespassing, and so on. For the demonstration scenarios in Chapter 9, only a high end scenario was defined for the off-site and ash landfill source categories.

The methodologies of this document are site-specific; assessors need to make the kinds of assumptions discussed here for their own source and populations of concern. Defining a high end scenario as a farm and a central scenario as a residence was done for demonstration purposes only. On the other hand, the example scenarios in Chapter 9 were carefully crafted to be plausible and meaningful, considering key factors such as source strength, fate and transport parameterization, exposure parameters, and selection of exposure pathways. For example, a beef and milk exposure pathway is demonstrated only for the high end scenarios. Farmers raise cattle for beef and dairy products and are

assumed to obtain a portion of their intake of these products from their own farm, whereas non-farming residents are assumed not to be exposed to contaminated farm products. Key source strength terms, including stack emission rates, concentrations of dioxin-like compounds in ash for the ash landfill source category, and plausible soil concentrations all came an examination of numerous data sources. Introductory sections of Chapter 9 provide a more complete description of the example scenarios.

7.3. EXPOSURE PATHWAYS

As discussed in Chapter 3, the dioxin-like compounds have been found primarily in air, soil, sediment and biota and to a lesser extent in water. Thus, the most likely exposure pathways are:

- Ingestion of soil, water, beef, dairy products, fish, fruit, and vegetables
- Dermal contact with soil
- Inhalation of particulates and vapors.

Exposure is calculated as the potential dose normalized over body weight and lifetime. This value can be computed as the lifetime average daily dose (LADD) for all exposure pathways:

$$\text{LADD} = (\text{exposure media concentration} \times \text{contact rate} \times \text{contact fraction} \times \text{exposure duration}) / (\text{body weight} \times \text{lifetime}). \quad (7\text{-}1)$$

A principal reference for most of the exposure parameters in this equation (not including the exposure media concentration) is the Exposure Factors Handbook (EPA, 1989). Parameters relating to dermal contact are more completely described in EPA (1992). Both references contain comprehensive literature reviews and recommendations for approaches which consider typical and high end exposures. Their recommendations are followed for this assessment. Details beyond the summaries provided here can be found in those references.

Exposure factors are best determined on a site specific basis. However, generic default values can be used as a starting point for developing site specific values or as defaults when the data needed to further refine these estimates are not available. The default values are expressed as a range from a central to high end value. Each term in this

equation is discussed briefly below.

- **Lifetime:** This refers to the expected lifetime of the exposed individuals and is used to normalize the exposure estimate such that it is consistent with the cancer slope factor. EPA (1989) recommends using the approximate United States average of 70 years for all pathways. This value was also adopted for the example scenarios in Chapter 9.
- **Body weight:** This refers to the average body weight of the exposed individuals over the exposure period and is also used to normalize the exposure estimate such that it is expressed in a manner consistent with the cancer slope factor. EPA (1989) recommends using the approximate United States average of 70 kg for all pathways involving adults and 16 kg for children ingesting soil. These values were also adopted for the example scenarios in Chapter 9.
- **Exposure media concentration:** This is the concentration of the contaminant in the media of interest and should represent a temporal average over the time of exposure. Estimation of this term was covered in Chapter 5.
- **Exposure duration:** This is the overall time period that individuals spend in situations that expose them to a contaminated media. Based on EPA (1989), a range of 9 to 30 years is recommended as default values for all residential exposure pathways. This range corresponds to the 50th and 90th percentiles of time a person spends at one residence. In some situations other times may be appropriate. For example farmers may have a tendency to remain in the same house for more years than the non-farmer resident. For the example scenarios presented in Chapter 9, and as noted in Section 7.1. above, 9 years was adopted for residents and 20 years as the estimate for farmers. They are characterized as central and high end estimates of exposure duration.
- **Contact rate:** This is the total rate of contact with the exposure media via ingestion, inhalation, or dermal contact. The values for this quantity vary by pathway as discussed below.
- **Contact fraction:** This is the portion of contacted material that is contaminated. It reduces the total contact rate to reflect only contact with

contaminated media. The contact fractions for the exposure pathways of air inhalation and water ingestion are related to the time individuals spend at home. Other pathways such as fish ingestion or ingestion of home grown foods are not related to time at home. Similarly, contact fractions for individuals exposed at work places relate largely to time spent at the work place. EPA (1989) discusses several time use studies which can be used to make assumptions about time spent at home (and outdoors at the home environment) versus time spent away from home. Generally, these time use studies asked participants to keep 24 hour diaries of all activities. Studies summarized were national in scope, involved large numbers of individuals, cross-sections of populations in terms of age and other factors, and up to 87 categories of activities. Results from different studies consistently indicate that the average adult spends between 68 to 73% of time at the home environment. The values selected for this factor for use in the example scenarios are described below.

The parameter values for contact rate and contact fraction applicable to each of exposure pathways are presented in Sections 7.2.1 to 7.2.7. The default values for all exposure parameters are summarized in Table 7-1 at the end of this chapter.

7.3.1. Soil Ingestion

Soil ingestion occurs commonly among children during activities such as mouthing of toys and other objects, nonsanitary eating habits, and inadvertent hand-to-mouth transfers. In addition to normal soil ingestion activities, some individuals exhibit behavior known as pica which involves intentional soil ingestion. Soil ingestion rates associated with pica are probably much higher. No measured values for pica patterns have been reported in the literature, though EPA (1989) reports that other assessments have assumed values such as 5 and 10 g/day. This document considers only normal soil ingestion among children.

To a lesser extent, soil ingestion also occurs among adults from activities such as hand-to-mouth transfer when eating sandwiches or smoking. However, the data to estimate the adult rate of soil ingestion is essentially unavailable, so adult soil ingestion is not demonstrated in this assessment.

Based on the review of literature, particularly the studies of Binder et al. (1986) and Clausing et al. (1986), the following values for soil ingestion were suggested in EPA (1989): average soil ingestion in the population of young normal children (under the age of 7) is estimated at approximately 0.1 to 0.2 g/d. An upper-range ingestion estimate among children with a higher tendency to ingest soil materials, although not a pica pattern, could be as high as 1 g/d. However, a value of 0.8 g/day is recommended for high end exposure estimates. The values of 0.2 g/d and 0.8 g/d were the values adopted for the central and high end exposure scenarios in Chapter 9.

EPA generally assumes a contact fraction of 1 for soil ingestion in residential settings. This is based on the assumption all soil ingestion by children occurs at home, and, of course, that the soil at the home is contaminated. In situations where the contaminated area is located remote from where children live, and children have some access to these areas (if the areas are parks or playgrounds, e.g.), lower fractions would be appropriate. A value of 1 has been adopted in the example scenarios presented in Chapter 9.

7.3.2. Soil Dermal Contact

The total annual dermal contact, expressed in mg/yr, is the product of three terms: the contact rate per soil contact event, the surface area of contact, and the number of dermal contact events per year. EPA (1992) recommends the following ranges for these terms:

- Contact rate: 0.2 to 1.0 mg/cm^2-event
- Adult surface area: 5000 to 5800 cm^2
- Event frequency: 40 to 350 events/year.

An event frequency and contact rate near the upper end of these ranges may be appropriate for an high end exposure activity pattern such as farming, where the individual more often comes in contact with soil and may be exposed to fugitive dust emissions while they work. An event frequency of 40 and 350, and a contact rate of 0.2 and 1.0 mg/cm^2-event, are assumed for the central and high end exposure scenarios in Chapter 9. However, the exposed surface area of 5000 cm^2 may be reduced for farmers. This area corresponds to 25% of the total body area and apparel such as short sleeves, shoes, socks and short pants. Farmers working in the field are likely to wear long pants at least, if

not also long sleeves. Although EPA (1992) indicates that clothing is not always effective in preventing dermal contact, it seems reasonable that a value of 1000 cm^2 (5% of total body area) representing hands, neck, and face might be more appropriate in a farming scenario. Values of 5000 and 1000 cm^2 were selected for the central and high end scenarios in Chapter 9.

The considerations for contact fraction are similar to those for soil ingestion; i.e., that all contact occurs with contaminated soil at the residence or farm site. Accordingly a value of 1 was selected for the example scenarios presented in Chapter 9.

One further adjustment was made for this exposure pathway. The contact as estimated above is the amount of soil which contacts the body. EPA (1992) indicates that only a small percent of strongly hydrophobic organic compounds such as 2,3,7,8-TCDD are absorbed into the body from soil dermal contact. The "dermal absorption fraction" recommended for 2,3,7,8-TCDD in EPA (1992) was 0.001 (0.1%) to 0.03 (3%). EPA (1992) recommends using the upper end of this range for application to other dioxin-like compounds as a conservative assumption until these compounds have been tested. An absorption fraction of 0.03 was used for the three compounds demonstrated in Chapter 9. The dermal contact exposure pathway was the only one in which such an absorption fraction was used.

7.3.3. Vapor and Dust Inhalation

EPA (1989) describes derivation of the commonly used ventilation rates of 20 and 23 m^3/day. As noted in that reference, these values assume 16 hours of light activity and 8 hours of resting. Other recommendations in that reference are a rate of 30 m^3/day for high end exposures, and to derive specific ventilation rates (the reference gives information to do so) for specific activity patterns. The example scenarios of this assessment all use 20 m^3/day.

An additional assumption needs to be made for the vapor and dust inhalation pathways. This pertains to an assumption concerning the differences in air quality between indoor and outdoor conditions. Algorithms for both particulate and vapor-phase air-borne concentrations of contaminants are specific to outdoor air. Hawley (1985) assumed, based on several other studies in which measurements were made, that the concentration of suspended particulate matter in indoor air is equal to 75% of that outside.

Also, his report stated that most household dust is outdoor dust that is transported into the house, and that only a small percentage is developed from sources within. He then concluded that 80% of the indoor dust is identical in contaminant content to outdoor soil. Refinements to the concentration of contaminants on indoor versus outdoor dust should have a minor effect on exposure estimates. A similar trend is assumed for air-borne vapor phase concentrations. For this reason, differences between indoor and outdoor concentrations are not specifically considered, or equivalently, no distinctions are made for outdoor and indoor air quality.

The contact fraction for this pathway is equal to the fraction of total inhaled air which is contaminated. Thus it relates largely to percent of time spent in the contaminated area. For the example exposure scenarios presented in Chapter 9, the contact fraction corresponds to percent time at home. EPA (1989) suggests a range of 0.75 to 1.0; the lower value will be adopted as the central value used in the residence setting. The value selected for high end scenarios will be 0.90 instead of 1.00, recognizing that 1.00 is more likely a bounding rather than a high end estimate.

7.3.4. Water Ingestion

The water ingestion rate of 2 L/day is traditionally assumed for exposure through drinking water. However, EPA (1989), after review of several literature sources, concludes that 2.0 L/day may be more appropriately described as a 90% value, or a value for high end exposure estimates. That document recommends a rate of 1.4 L/day as representative of average adult drinking water consumption. This is the rate used for central and high end settings in Chapter 9. Like the vapor and dust inhalation pathways, the difference in central and high end tendencies is modeled using the contact fraction. Again, this fraction is based on the time spent at home. The value of 0.75 is used to model the central estimate, for the residence setting, and the value of 0.90 is used to model the high end estimate, for the farm setting.

7.3.5. Beef and Dairy Product Ingestion

If contaminated beef or dairy products from one source are marketed along with uncontaminated products from many sources, only a small percent of the product consumed by an individual may be contaminated. The potential effects of such "market

dilution" of beef and dairy products on human exposure are discussed briefly in EPA (1984a), at more length by Fries (1986), and at much greater length in EPA (1985) for the particular case of cattle production in Missouri. Aspects of the beef industry in this region specifically noted in EPA (1985) as important to estimating exposure were type of activity within the industry (e.g. cow-calf production, "backgrounding" - preparing calves for feedlots, feeding for slaughter), replacement rates as a function of activity, fractions of cattle fed to maturity outside contaminated areas before slaughter, and slaughter categories and rates relative to national figures. EPA (1984a and 1985) concluded that dilution will vary widely between different marketing areas. EPA (1984a and 1985) and Fries (1986) noted that the subpopulations most likely to receive high exposures are beef producers, dairy farmers, and their direct consumers. The residents in the central exposure scenario in Chapter 9 are not assumed to be producers or direct consumers of farm products, and for this reason, the central estimate scenarios do not include a beef and milk pathway. The high end farming scenarios do have these pathways, meaning that the farmers home slaughter for beef and also obtain a portion of their milk from their lactating cattle.

Average consumption rates and fat content data for beef and dairy products are described in EPA (1989). Summary information presented in that reference comes principally from a U.S. Department of Agriculture Nationwide Food Consumption Survey (NFCS) conducted in 1977-1978 (described in EPA, 1984b), with additional information added by Fries (1986). The NFCS covered intake of 3,735 possible food items by 30,770 individuals characterized by age, sex, geographic location, and season of the year. The average beef fat consumption rates listed in EPA (1989) ranges from 14.9 to 26.0 g per 70-kg person/day, with a single high consumption estimate of 30.6 g per 70-kg person/day. Based on this information, EPA (1989) recommends using an average beef fat consumption rate of 22 g/day (this assumes 100 g/day whole beef and 22% fat content). This is an average rate, which may underestimate the amounts eaten by households who home slaughter. Milk fat consumption from all dairy products ranges from 18.8 to 43 g per 70-kg person/day. Considering fresh milk only, milk fat consumption is reported to average 8.9 to 10.7 g per 70-kg person/day, with a single high consumption estimate of 35 g per 70-kg person/day. An average milk fat consumption rate of 10.5 g/day is given in EPA (1989) (this assumes 300 g/day whole milk and 3.5% fat). This may also

underestimate the consumption rate of farming families who consume milk supplied by their own cattle. The rate of beef and milk fat consumption assumed for the high end farming scenarios in Chapter 9 are 22 and 10.5 g/day, respectively.

Consumption rates of beef and milk are expressed in terms of fat ingested per day because dioxin-like compounds tend to partition strongly toward lipids. It is assumed that virtually all of such compounds will be found in the fat portion of milk or beef. Further, the algorithms to estimate concentrations in these food products estimated fat concentrations and not whole product concentrations (see Section 5.3.4.3).

EPA (1989) also reports on another survey of 900 rural farm households (USDA, 1966), including some where the farm's beef and dairy cattle supply a portion of the household's beef and milk. In these situations, the average percent of homegrown beef and milk (dairy products) is 44% and 40%, respectively. Contact fractions of 0.44 and 0.40 were used in this assessment for the high end farming scenarios. Lacking better information, EPA (1989) recommends a contact fraction for beef and dairy of 75% if the intent is to estimate high end estimates for a farmer who uses a portion of his farm's products.

7.3.6. Fish Ingestion

EPA (1989) concludes that consumption rate data from two studies, that of Puffer (1981) and Pierce, et al. (1981) are most appropriate for estimating consumption rates for recreational fishing from large water bodies. The recommended 50th percentile consumption rate, or typical rate, for this subpopulation is 30 g/day, and the 90th percentile rate is 140 g/day. For smaller water bodies, EPA (1989) recommends that site-specific information be obtained via surveys of local fisherman to obtain the most appropriate fish consumption information for site-specific assessments. Alternately, EPA (1989) recommends using judgement regarding how many fish meals per year an individual could obtain from the contaminated waters and assuming meal sizes of 100 to 200 g. Consumption of commercial fish (at restaurants or from markets) raises market dilution issues analogous to those described earlier for beef and milk. For this reason, exposed individuals in both the central and high end scenarios in Chapter 9 are assumed to obtain their contaminated fish intake from a nearby contaminated stream or pond; other fish they may consume is assumed to not be contaminated.

The examples used in this assessment assume that the contaminated waters are small lakes or streams which are occasionally fished on a recreational basis. Further it is assumed that an individual could eat 3 to 10 meals per year from the contaminated waters. Assuming an average meal size of 150 g, this translates to 450 to 1500 g/year or an average of 1.2 to 4.1 g/day. The central estimate for the example scenarios in Chapter 9 will therefore be 1.2 g/day, and the high end will be 4.1 g/day. Since these fish ingestion rates are rates of ingestion of contaminated fish, the contact fraction would be 1.

7.3.7. Fruits and Vegetables

EPA (1989) estimated ingestion rates for individuals who have home gardens and hence grow a portion of their fruit and vegetable intake. Their approach was to review the literature and derive average intake rates for all individuals, whether or not they have a home garden, and considering a variety of different fruits and vegetables. A typical and high end exposed individual had the same total ingestion rates. Their exposure was distinguished by the contact fractions; high end exposed individuals grew a larger proportion of their intake in their home gardens.

The average amounts of fruit and vegetable consumption are 200 and 140 g/day, respectively. These total ingestion rates are further refined considering two factors pertinent to estimation of concentration of dioxin-like compounds: whether the vegetation is grown below (carrots, e.g.) or above ground (tomatoes), and whether the edible portion is protected (citrus) or unprotected (apples). Chapter 5 discusses distinct procedures for estimating vegetative concentrations for below and above ground vegetation. Also, both algorithms assume that inner portions of vegetation are largely unimpacted, whereas outer portions of both above and below ground vegetation are impacted (see Chapter 5 for further detail on these algorithms and assumptions). Therefore, for fruits or vegetables which are protected, it can be assumed that there will be no exposure since the outer portions are not eaten. Results from a food consumption survey, such as that from Pao, et al. (1982), can be used to determine percent of total fruit/vegetable intake which is below/above ground and which is protected/unprotected. Such an exercise was undertaken using data from Pao, et al. (1982) summarized in EPA (1989) to arrive at the following percentages:

	I. Fruits		II. Vegetables	
	Above	Below	Above	Below
Protected	56%	0	25%	1%
Unprotected	44%	0	54%	20%

As seen, it was found that there are no fruit grown underground, and there was a fairly similar proportion of protected and unprotected fruit. Fruits considered protected for this exercise included oranges, grapefruits, and cantaloupe; unprotected fruits included apples, peaches, pears, and strawberries. It is noted that this is clearly not a complete inventory, but only those fruits from the survey of Pao as summarized in EPA (1989). Similarly, these percentages are not being recommended as general values for other site-specific assessments. For this assessment, it will be assumed that a total ingestion rate of unprotected above ground fruit is 88 g/day (0.44*200 g/day), and that there is no ingestion of unprotected under ground fruit. Vegetables above ground and unprotected include: cabbage, cucumbers (including cucumbers as pickles), lettuce, tomatoes, broccoli, spinach, string beans, and squash. Above ground protected vegetables include: corn, lima beans, and peas (several kinds). Below ground unprotected vegetables included potatoes and carrots; mature onions were considered below ground and protected. Assumed for this assessment are ingestion rates of unprotected above ground vegetables of 76 g/day (0.54*140 g/day) and unprotected below ground vegetables of 28 g/day (0.20*140 g/day).

These ingestion rates are defined as total ingestion rates of unprotected above/below ground fruits/vegetables. Only a portion of these are homegrown. Data summarized in EPA (1989) shows that the fraction of vegetables consumed that are homegrown ranges from 0.04 to 0.75, depending on type. The overall average of the data is 0.25, which is recommended as a contact fraction for the average home gardener. The recommendation for the high end exposure was 0.40. These contact fractions were adopted as the central and high fractions for the example scenarios in Chapter 9. Similar data for fruits show a homegrown range of 0.09 to 0.33, with an average of 0.20, which is the central estimate used in Chapter 9. EPA (1989) recommends a high end value of 0.30, which is the value used for the high end exposure scenarios.

Table 7-1. Summary of exposure pathway parameters

Pathway description	Contact rates	Contact fractions	Comments
Soil Ingestion			
Central	200 mg/d	1.0	Only pathway specific to an age-range; 2- to 6-year-old children
High End	800 mg/d	1.0	
Soil Dermal Contact			
Central	0.2 mg/cm^2-event 5000 cm^2 40 events/yr	1.0	Unlike other pathways, daily contact is not assumed; approach instead estimates contact in terms of contact/event * events/yr; both soil pathways distinguish typical and high end contact by contact rate rather than contact fraction; high end based on behavior of farmers; typical pattern based on non-farming adults; a "dermal absorption fraction" reduces the amount contacting the body to an amount absorbed by the body.
High End	1.0 mg/cm^2-event 1000 cm^2 350 events/yr	1.0	
Dermal absorption fraction: 0.03			
Vapor/Dust Inhalation			
Central	20 m^3/day	0.75	Indoor/outdoor air quality assumed equal; contact fraction can be used to reflect a different assumption; central contact fraction is an average time-at-home based on time use surveys
High End	20 m^3/day	0.90	
Water Ingestion			
Central	1.4 L/day	0.75	2.0 L/day evaluated as 90th percentile value in EPA (1989), who instead recommend 1.4 L/day as an average consumption rate; like inhalation pathways, 0.75 contact fraction is an average time-at-home estimate
High End	1.4 L/day	0.90	

(continued on the following page)

Table 7-1. (continued)

Pathway description	Contact rates	Contact fractions	Comments
Beef Fat Ingestion			
Central	NA	NA	Central scenario includes adults who do not home produce their beef supply; 22 g/day beef fat assumes 100 g/day average beef consumption and 22% fat
High End	22 g/day	0.44	
Milk Fat Ingestion			
Central	NA	NA	Like beef ingestion, central scenarios do not include home milk production; 10.5 g/day milk fat assumes 300 g/day whole milk and 3.5% fat
High End	10.5 g/day	0.40	
Fish Ingestion			
Central	1.2 g/day	1.00	Unlike other exposure pathways, contact rate is rate of contaminated fish - hence contact fraction is 1.00; 1.2 g/day based on 3 recreationally caught fish meals/yr; 4.1 g/day based on 10 recreationally caught fish meals/yr
High End	4.1 g/day	1.00	
Fruit Ingestion			
Central		0.20	200 g/day is average total fruit consumption rate; 44% assumed above ground and unprotected - no below ground unprotected fruit; contact fraction of 0.20 based on a range of rates of 0.09 to 0.33 for wide variety of fruits (EPA, 1989)
Above ground unprotected	88 g/day		
Below ground unprotected	0 g/day		
High End		0.30	
Above ground unprotected	88 g/day		
Below ground unprotected	0 g/day		

(continued on the following page)

Table 7-1. (continued)

Pathway description	Contact rates	Contact fractions	Comments
Vegetable Ingestion			
Typical		0.25	140 g/day is average total vegetable consumption rate; 54% assumed above ground and unprotected, 20% assumed below ground and unprotected; central contact fraction of 0.25 based on a range of rates of 0.04 to 0.75 rates for wide variety of vegetables (EPA, 1989)
Above ground unprotected	76 g/day		
Below ground unprotected	28 g/day		
High End		0.40	
Above ground unprotected	76 g/day		
Below ground unprotected	28 g/day		

Exposure Duration: A duration of 9 years was assumed for the "residence" setting based on mobility data showing the average time in one residence was 9 years (EPA, 1989). A duration of 30 years was evaluated as the 90th percentile of time in one residence, and was selected for the "farm" settings, which assumes that farming families live in a given residence longer than a non-farming family. These values apply to all pathways with no variation, except for soil ingestion, which was demonstrated only for children and was at 5 years.

Body Weight/Lifetime: The standard assumptions of a 70 kg adult and 70 years lifetime were assumed for all pathways, except that of soil ingestion. In that case, a body weight of 17 kg was used.

REFERENCES FOR CHAPTER 7

Binder, S.; Sokal, D.; Maughn, D. (1986) The use of tracer elements in estimating the amount of soil ingested by young children. Arch. Environ. Health 41: 341-345.

Clausing, P.; Brunekreff, B.; Van Wijen, J.H. (1987) A method for estimating soil ingestion by children. Int. Arch. Occupational Environ. Health 59: 73-82.

Fries, G.F. (1986) Assessment of potential residues in foods derived from animals exposed to TCDD-contaminated soil. Presented at 6th international symposium on chlorinated dioxins and related compounds; September; Fukuoka, Japan.

Hawley, J.K. (1985) Assessment of health risk from exposure to contaminated soil. Risk Analysis 5(4): 289-302.

Pao, E.M., K.H. Fleming, P.M. Guenther, et al. 1982. Foods commonly eaten by individuals: amount per day and per eating occasion. U.S. Department of of Agriculture. Home Economics Report No. 44.

Pierce, R.S.; Noviello, D.T.; Rogers, S.H. (1981) Commencement Bay seafood consumption report. Preliminary report. Tacoma, WA: Tacoma-Pierce County Health Department.

Puffer, H. (1981) Consumption rates of potentially hazardous marine fish caught in the metropolitan Los Angeles area. EPA Grant #R807 120010.

U.S. Department of Agriculture. (1966) Household food consumption survey 1965-1966. Report 12. Food Consumption of households in the U.S., Seasons and years 1965-1966. United States Department of Agriculture, Washington, D.C. U.S. Government Printing Office.

U.S. Environmental Protection Agency. (1991a) Guidelines For Exposure Assessment SAB Draft Final. OHEA-E-451.

U.S. Environmental Protection Agency. (1992) Dermal Exposure Assessment: Principals and Applications. Exposure Assessment Group, Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency. EPA/600/8-91/011B.

U.S. Environmental Protection Agency. (1989) Exposure Factors Handbook. Exposure Assessment Group, Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency. EPA/600/8-89/043. July, 1989.

U.S. Environmental Protection Agency. (1985) Dioxin Transport From Contaminated Sites to Exposure Locations: A Methodology for Calculating Conversion Factors. Exposure Assessment Group, Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency. EPA-600/8-85-012. NTIS PB85-214310.

U.S. Environmental Protection Agency. (1984a) Risk Analysis of TCDD Contaminated Soil. Exposure Assessment Group, Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency. EPA-600/8-84-031.

U.S. Environmental Protection Agency. (1984b) Stochastic processes applied to risk analysis of TCDD contaminated: a case study. Internal report dated May 31, 1984; Exposure Assessment Group, Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency.

8. PHARMACOKINETICS

8.1. INTRODUCTION

The pharmacokinetic profiles of CDDs and the CDFs are quite complex. A thorough analysis and understanding of these pharmacokinetic data would be very helpful in ensuring that exposure assessments for these compounds are reliable. In addition, such information would be useful in providing enhanced knowledge and understanding for the purposes of risk assessment.

Exposure to 2,3,7,8-TCDD and related compounds results in numerous species and tissue specific toxic and biological responses. Many, if not, all of these responses are mediated by a soluble intracellular protein, the aryl hydrocarbon (Ah) receptor, to which 2,3,7,8-TCDD binds with high affinity. After 2,3,7,8-TCDD and related compounds bind to this Ah receptor the complex undergoes a transformation process involving dissociation of hsp90. The transformed receptor complex is then able to bind with high affinity to a specific DNA sequence referred to as a dioxin responsive enhancer (DRE). The conserved nature of the DRE and Ah receptor is also indicated by the ability of transformed 2,3,7,8-TCDD: Ah receptor complexes from a wide variety of species to bind to the DRE. Studies also indicate a similarity in DNA recognition by Ah receptor from a variety of species suggestive of a functional role of this sequence in 2,3,7,8-TCDD responsiveness (Denison et al., 1991; Gasiewicz and Henry, 1991; Perdew and Hollenbeck, 1990; Andersen and Greenlee, 1991). Thus the definition of "disposition" may have to be extended to include suborgan or subcellular sites in order to more fully describe the congener, species, and train specific pharmacokinetics (dosimetry) of these compounds.

Pharmacokinetic analysis may be used in several ways to aid in the exposure and dose assessment of foreign chemicals. They may, for example, allow for predicting the time and profile of elimination of chemicals from the body. The redistribution of CDDs among the various tissues and organs, which may occur during elimination, can be accounted for and tracked. Effects on disposition which may result from altered physiology, such as sudden weight loss or from lactation, can be incorporated and thus adequately considered in exposure and risk assessments. Lactation is known to be an efficient route for the transfer of many of these chemicals from mother to offspring (Nau et al., 1986; Bowman et al., 1989).

Pharmacokinetic analyses can be used to estimate background exposure levels from body burden data. They can also be used to estimate uptake rates from various food sources, elimination rates and times from the body, and to estimate tissues levels from blood and adipose tissue monitoring. In addition, with the appropriate data on several congeners, estimates can be made for other congeners about which less data are available.

The remainder of this chapter will cover areas of pharmacokinetics pertinent to exposure assessment. Background levels and daily uptake of 2,3,7,8-TCDD will be reviewed and discussed; a method for the calculation of uptake of other congeners from food will be outlined; use of a compartmental model to estimate daily uptake will be demonstrated; a method will be outlined and reviewed for determining internal tissue concentrations from monitored blood and/or adipose tissue; pertinent data on bioavailability will be summarized.

8.2 DAILY BACKGROUND LEVELS

8.2.1 Basis for Calculation

Physiologically based pharmacokinetic (PBPK) models are convenient and useful methods for describing and predicting disposition of foreign chemicals in the body. These models take into account physiologic and biochemical processes such as blood flows, metabolism, and renal clearance, and describe the body according to its normal anatomy. PBPK models can, given adequate data, predict disposition from one exposure scenario to another and even from species to species. One such model was developed for 2,3,7,8-TCDF (King et al., 1983) and is used here with some modifications. The anatomic regions depicted in the King model are the blood, liver, fat, skin, and muscle. The remaining organs of the body are lumped together as the "carcass." Input may be by a variety of routes, but for the purposes of this discussion is considered to occur through the gastrointestinal system by continuous chronic dosing. This is consistent with the common belief that most of these compounds enter the body through the gastrointestinal track as a result of the consumption of products containing animal fat (Schlatter, 1991). The pertinent equations follow.

For the liver:

$$V_L \frac{dc_L}{dt} = Q_L(C_B - \frac{C_L}{R_L}) - KL\frac{C_L}{R_L} + D \qquad (8-1)$$

Where:

V_L	=	Volume of the liver
C_B	=	Concentration of toxin in blood
C_L	=	Concentration of toxin in liver
t	=	Time
Q_L	=	Blood flow to liver
R_L	=	Equilibrium concentration ratio between liver and blood
KL	=	Clearance term (L/time)
D	=	Input or dosing function

The clearance in the liver is considered to be by metabolic processes.

For the fat:

$$V_F \frac{dC_F}{dt} = Q_F(C_B - \frac{C_F}{R_F}) \qquad (8-2)$$

Where: All terms are analogous as those in equation (8-1).

Note that there is no metabolic elimination assumed. The only disappearance of material from the fat is assumed to be diffusion driven and is accounted for in the above mass balance equation. Equations for the skin, carcass, and muscle are analogous to that for fat.

The equation for the blood:

$$V_B \frac{dC_B}{dt} = Q_L\frac{C_L}{R_L} + Q_F\frac{C_F}{R_F} + Q_M\frac{C_M}{R_M} + Q_S\frac{C_S}{R_S} + Q_C\frac{C_C}{R_C} - Q_B C_B \qquad (8-3)$$

Where:

Q	=	Blood Flows
C	=	Concentrations
R	=	Equilibrium Concentration Ratios between tissue and blood

Subscripts:

B, L, F, M, S, C refer to blood, liver, fat, muscle, skin and carcass.

Some assumptions may be made to simplify the model for use to estimate daily background doses. If steady state is assumed then the equations for the individual organs can be summed. The resultant equation for the liver is then:

$$D = \frac{(KL)\,C_L}{R_L} \qquad (8\text{-}4)$$

and at steady state:

$$\frac{C_L}{R_L} = C_B \qquad (8\text{-}5)$$

and

$$C_B = \frac{C_{F,SS}}{R_F} \qquad (8\text{-}6)$$

and hence

$$D = \frac{(KL)\,C_{F,SS}}{R_F} \qquad (8\text{-}7)$$

When clearance is expressed in days, D is the daily intake. Clearance can be approximated from the half life information according to the following equations:

$$KL = k_e V_d \qquad (8\text{-}8)$$

Where:

k_e = first order elimination rate constant
V_d = Volume of distribution

and

$$k_e = \frac{ln2}{t_{1/2}} \qquad (8\text{-}9)$$

Where:

$t_{1/2}$ = biological half life of compound in body

The volume of distribution may be estimated as in King et al., (1983) according to:

$$V_{d,i} = V_i R_i \qquad (8\text{-}10)$$

Where:

$V_{d,i}$ = Volume of distribution of organ, i
V_i = Actual volume of organ, i
R_i = Equilibrium concentration ratio between organ i and blood

With substitution equation (8-7) becomes:

$$D = \left(\frac{ln2}{t_{1/2}}\right)(V_i)(C_{i,ss}) \qquad (8\text{-}11)$$

Where:

V_i = Volume of the organ in which toxin is measured
$C_{i,ss}$ = Steady State concentration of toxin in organ

It is important to note that two major assumptions are in effect when the above formula is used to calculate average daily uptake. First steady state conditions are assumed. Given that the half life of some of these compounds (e.g. 2,3,7,8-TCDD) are at least 5 years it would take well over 15 years to reach 90% of steady state, and over 30 years to reach 99% of steady state levels. Thus, the assumption of steady state is only reasonable if background environmental concentrations are relatively similar and constant throughout the nation. Under such conditions even the normal movement from one geographical location to another would result in relatively constant exposures. Also implicit in this assumption is that bioavailability is relatively constant through the nation. Given that the source of this background exposure is believed to be animal fat (Schlatter et al., 1991) the assumption of steady state for adults might be considered a reasonable assumption. Exceptions are those individuals who may for a portion of their adult lives be consuming foods with unusually high levels of these compounds. It would be expected however that those individuals would have higher than average body burdens, and hence would not be considered to have only average background exposure, but would rather be considered part of a source specific exposure group. Conditions such as sudden weight loss and lactation would also alter the steady state condition. However, for purposes of

calculating daily background exposure levels the sampling of tissues for body burden must be so designed to account for such deviations in the average. In summary, for adults (over 25 years of age) not in a source specific exposure group, the steady state assumption is a reasonable approximation. It should be remembered however, that the longer the biological half life the longer it would take to reach steady state. A compound with a half life of 10 years, for example, would take over 50 years to reach the 90% of steady state value.

The second major assumption is that these compounds are eliminated from the body by monophasic kinetics. Biphasic elimination is very possible for many of these compounds. Data gathered to calculate elimination rates or half lives would only reveal biphasic elimination profiles if gathered several years after the last exposure. Using only the short term half life would result in an underestimated value for half life and an overestimate of daily intake. This is particularly problematic for those compounds with extremely long half lives and for which few data exist. In a later section an approach to calculating half lives for some of these compounds will be presented. Also, the elimination kinetics are assumed to be constant over the entire life of the individual. Sudden weight loss and lactation would, for example, be conditions which violate that assumption. Again, it would be assumed that for calculation of daily intake due to background exposure the body burden data from such individuals would be identified and calculations handled accordingly.

8.2.2 Daily Intakes

Figure 8-1 shows a sample calculation for 2,3,7,8-TCDD using the above procedure. A fat volume of 14 L was chosen, representing 20% of the body weight. Also, for the purposes of this example, 1 ml of tissue was assumed to be equivalent to 1 gm. Table 8-1 shows the estimated daily intake of 2,3,7,8-TCDD at several conditions. The range of daily intakes calculated are in agreement with those reported elsewhere (Fürst et al., 1991; US EPA, 1992).

In order to perform similar calculations for other congeners three pieces of information are necessary. First, concentrations in the adipose tissues must be known. Second, the half lives of the compounds within the body must be known. Third, some understanding of the kinetics and exposure conditions to assure that steady state

$$D = \left[\frac{ln2}{t_{1/2}}\right](V_i)\,(C_{i,ss})\left(\frac{1}{70KG}\right)$$

$$D = \left(\frac{ln2}{5.8years}\right)\left(14L\ \ 1000\ \frac{ml}{L}\right)\left(6.72\frac{pg}{ml}\right)\left(\frac{1}{70KG}\right)\left(\frac{1year}{365days}\right)$$

$$D = 0.44\ \ pg/kg/day$$

Figure 8-1: Sample Calculation of Daily Intake for 2,3,7,8-TCDD

Table 8-1 Calculated Daily Intakes for 2,3,7,8-TCDD.

Half life (yrs)	Fat Vol. (L)	Fat Conc. (ppt)	Calculated Daily Intake (PG/KG/Day)
5.8	14.0	6.72	0.44
7.0	14.0	6.72	0.37
5.8	14.0	5.00	0.33
7.0	14.0	5.00	0.27
5.8	7.0	6.72	0.22
7.0	7.0	6.72	0.18
5.8	7.0	5.00	0.16
7.0	7.0	5.00	0.14

conditions were achieved at the time of monitoring.

Concentrations of various congeners in adipose tissues can be found in several sources (Stanley et al., 1986; Scheter, 1991). Values range from around 2 ppt for 2,3,7,8-TCDF to several hundred ppt for 1,2,3,4,6,7,8,9-OCDD.

Half lives could be determined from elimination data, if available. Methods have

been suggested to determine the half life of such compounds from uptake data relative to 2,3,7,8-TCDD. Schlatter (1991) has proposed one such method. The following has been adapted from that proposed method.

Manipulation of equation (8-11) results in:

$$C_{TCDD} = \frac{D_{TCDD} \quad t_{1/2,TCDD} \quad V}{\ln 2} \qquad (8\text{-}12)$$

Where:

C_{TCDD} = Concentration of TCDD in body
D_{TCDD} = Daily Intake of TCDD
$t_{1/2,TCDD}$ = half life of TCDD in body
V = Volume of body compartment

For some other congener x:

$$C_x = \frac{D_x \quad t_{1/2,x} \quad V}{ln2} \qquad (8\text{-}13)$$

Where:

Symbols are same as for equation (8-12) and subscript x applies to compound x.

Thus the ratio of concentrations of TCDD to x can be described by:

$$\frac{C_{TCDD}}{C_x} = \left[\frac{D_{TCDD} \quad t_{1/2,TCDD} \quad V}{ln2}\right] \left[\frac{ln2}{V \quad D_x \quad t_{1/2,x}}\right] \qquad (8\text{-}14)$$

Which with algebraic manipulation and simplification becomes:

$$t_{1/2,x} = \left[\frac{D_{TCDD} \quad t_{1/2,TCDD}}{C_{TCDD}}\right] \left[\frac{C_x}{D_x}\right] \qquad (8\text{-}15)$$

Assuming intake D, to be mostly from the food, especially animal fat products, they can be related to absorption from these foods according to:

$$D_{TCDD} = (k_{a,TCDD})\,(A_{TCDD}) \qquad (8\text{-}16)$$

Where:

$k_{a,TCDD}$ = Absorption rate constant for TCDD
A_{TCDD} = Concentration of TCDD in animal fat (diet)

and

$$D_x = (k_{a,x}) \ (A_x) \tag{8-17}$$

Where:

$k_{a,x}$ = Absorption rate constant for x
A_x = Concentration of x in animal fat (diet)

As a result the half life for compound x can be described by:

$$t_{1/2,x} = t_{1/2,TCDD} \left[\frac{A_{TCDD}}{C_{TCDD}} \right] \left[\frac{C_x}{A_x} \right] \left[\frac{k_{a,TCDD}}{k_{a,x}} \right] \tag{8-18}$$

When the absorption rate constants for each are equal or when the difference between them is small compared to differences in other parameters (concentration, half lives) equation (8-18) can be further simplified to:

$$t_{1/2,x} = t_{1/2,TCDD} \left[\frac{A_{TCDD}}{C_{TCDD}} \right] \left[\frac{C_x}{A_x} \right] \tag{8-19}$$

Before using the above approach to calculate half lives for some of the other substances of interest it is well to briefly highlight one of the assumptions in this approach. The relationship of half life to elimination as described in equation (8-9) only applies to simple single compartment kinetics. These compounds would not necessarily be expected to behave in such a manner. However, the error introduced by such an assumption is not great if the one phase predominates over the other, or if it is remembered that the calculation applies to one phase only. In fact, as will be discussed in a subsequent section, it is believed that for these chemicals the relationship between half life and elimination as described here is a reasonable approximation.

It should be noted that for some of these substances exposure is expected from other than food sources. For such cases equation (8-18) would be modified to include

these other sources as follows.

Where:

$$t_{1/2,x} = t_{1/2,TCDD} \left[\sum \frac{k_{a,i,TCDD} \quad A_{i,TCDD}}{C_{TCDD}} \right] \left[\sum \frac{C_x}{k_{a,i,x}} \quad A_{i,x} \right] \qquad (8\text{-}20)$$

$k_{a,i,TCDD}$	=	Absorption rate constants for TCDD from each of the i media
$A_{i,TCDD}$	=	Concentration of TCDD in each of the i media
$k_{a,i,x}$	=	Absorption rate constants for x from each of the i media
$A_{i,x}$	=	Concentration of x in each of the i media
Other symbols:		As previously defined

Again, if the differences between the absorption rate constants for TCDD and x are judged to be small then the following variation of equation (8-19) can be used:

$$t_{1/2,x} = t_{1/2,TCDD} \left[\sum \left(\left(\frac{A_{i,TCDD}}{C_{TCDD}} \right) \left(\frac{C_x}{A_{i,x}} \right) \right) \right] \qquad (8\text{-}21)$$

Table 8-2 shows the results of some half lives calculated in this manner.

The half lives calculated using equation (8-19) for the first three compounds in Table 8-2 agree with those calculated by Schlatter (1991). The large difference in the two calculations for OCDD is due to significant differences in absorption rates between the TCDD and the OCDD. Schlatter notes in his paper that for some compounds, including the OCDD, corrections were made of differences in absorption. No explanation was offered on how this was done. In US EPA (1992) results are summarized that indicate a possible several fold greater oral absorption of TCDD over OCDD. This adjustment would result in a calculated half life closer to that calculated by Schlatter.

In summary, this illustrates a method for calculating the half lives of similar behaving compounds. Several pieces of information are necessary: 1) the concentration in the body of 2,3,7,8-TCDD; 2) the half life of 2,3,7,8-TCDD; 3) the concentration of the substance of interest in the body; 4) the concentration of the substance of interest in the media from which exposure occurs; 5) the differential absorption rates for TCDD and the

Table 8-2 Half life calculations.

Chem.	Food-ppt[a]	Body-ppt[b]	$t_{1/2}$[c] years	$t_{1/2}$[d] years
2378-TCDD	0.23	6.72	7.0	6.0
2378-TCDF	0.84	3.9	1.11	1.3
12378-PeCDD	0.7	21.5	7.35	5.0
23478-PeCDF	1.4	36.8	7.94	6.3
OCDD	19.2	653.0	8.14	50.0

[a]Concentrations in food for TCDD, TCDF, and OCDD were obtained as discussed in Chapter 3; those for PCDD, PCDF were taken from Schlatter, 1991

[b]Concentrations in body (adipose tissue) were all taken from Schecter, 1991

[c]Calculated using equation 8-19, except for TCDD

[d]Calculated by Schlatter (1991), except for TCDD

substance of interest.

From the half life information, average daily intakes may be calculated when steady state can be assumed, by using equation (8-12). Caution should be exercised when calculating daily intakes for those compounds with very long half lives such as OCDD in the above example.

8.3 COMPARTMENTAL MODELING

As previously discussed and also discussed elsewhere (US EPA, 1992) PBPK models are very useful for describing and predicting the disposition of chemicals in the body. They are generally designed for predicting some measure of dose at a target site. They are also used to extrapolate from one species to another, between different doses, and between different routes of exposure. Several PBPK models have been published specifically for 2,3,7,8-TCDD (Leung et al., 1990; Leung et al., 1988; Kissel and Robarge, 1988). Also, as previously discussed, a model for 2,3,7,8-TCDF (King et al., 1983) is also

available and from which were derived the equations for estimating daily intake from body concentrations at steady state conditions. Most of these models have been developed to describe in great detail the metabolism and binding of TCDD within the body, for the purpose of estimating target tissue dose.

8.3.1 Pharmacokinetic Model

Assuming a linear relationship between the concentrations in the fat and the body at low exposure concentrations and the near linear elimination profile, a simpler model relating exposure, whole body elimination, and whole body concentration is developed here. As described earlier, after a prolonged exposure most of the body's organs can be assumed to have very similar kinetic profiles and can thus be lumped together. The fat, while sharing such a similar profile, is kept separate because of its important role in storing most of the body burden and because it has been the most typically monitored tissue. The three compartments in this model are blood, fat, and a "body" representing all other tissues. The model is a flow or perfusion limited model with the assumption that the toxin is well stirred or uniformly distributed within each compartment. The pertinent equations for the model follow:

$$\frac{dC_{bo}}{dt} = \frac{Q_{bo}\left[C_B - \frac{C_{bo}}{R_{bo}}\right] - K(C_{bo}) + D}{V_{bo}} \qquad (8\text{-}22)$$

Where:

dC_{bo}/dt	=	Rate of change of concentration of toxin in body (bo)
Q_{bo}	=	Blood flow (Volume/time) to body
C_B	=	Concentration of toxin in blood (B)
R_{bo}	=	Equilibrium concentration ratio of toxin between body and blood
K	=	Clearance (Vol/time) of toxin from body
V_{bo}	=	Volume of body
D	=	Intake of toxin

The integral of the differential equation over time is the actual concentration.

$$\frac{dC_F}{dt} = \frac{Q_F \left[C_B - \frac{C_F}{R_F}\right]}{V_F} \qquad (8\text{-}23)$$

Where:

dC_F/dt	=	Rate of change of concentration of toxin in fat (F)
Q_F	=	Blood flow (Volume/time) to fat
R_F	=	Equilibrium concentration ratio of toxin between fat and blood
V_F	=	Volume of fat

$$\frac{dC_B}{dt} = \frac{Q_F \left[\frac{C_F}{R_F}\right] + Q_{BO} \left[\frac{C_{bo}}{R_{bo}}\right] - C_B(Q_F + Q_{bo})}{V_B} \qquad (8\text{-}24)$$

It should be noted that the description of intake is somewhat different than what is typically found in most PBPK models. D here is actually a dose rate. That is, D is in terms of pg/kg/day coming into the body as a dose, not just a concentration in the food, drinking water, or air. The usual description for gastrointestinal absorption of toxin from food would be

$$D = k_a F_c \qquad (8\text{-}25)$$

Where:

k_a	=	absorption rate constant
F_c	=	Concentration of toxin in food

In this case because of inadequate knowledge regarding the absorption rate constants for many of the congeners a dose rate is used instead. This does not allow for estimation of body burden directly from environmental concentrations (e.g. F_c in equation (8-25)). As more data are collected, more accurate values of the parameters and descriptions of absorption functions can be input into equation (8-22). For the present other approaches can be used to relate concentration in the environmental media to daily intake (see Section 8.4.3).

8.3.2 Model Utilization

Most of the model's parameters are known or can be estimated from experimental data for many of these toxins. For example, the equilibrium concentration ratios between the fat and the body and between the fat and the blood for 2,3,7,8-TCDD are approximately 10 and 100 respectively on a tissue basis. The blood flows and compartment volumes are well known. One parameter that had to be estimated when applying this model to 2,3,7,8-TCDD was the clearance term K. This was done by first allowing the model to simulate elimination as though exposure was suddenly terminated (D becomes 0). Initial concentrations for the tissues were taken as those typically expected in the general population (7.0 ppt in the fat). The value of K was then adjusted until the model predicted a half life of 7 years. Values of clearance for the other compounds in this series would be determined similarly. The necessary information includes the equilibrium concentration distribution ratios, the half lives of the compounds, and some reasonable approximation of steady state fat concentrations.

With an estimated value for the clearance, the model can now be used with various exposure inputs to establish body burdens of toxin. The model can also be used to estimate elimination profiles from the body. In addition, events such as lactation can be incorporated into the model with knowledge about the appropriate parameters.

Figure 8-2 shows the results of the model run describing the elimination of 2,3,7,8-TCDD from fat. The clearance rate was adjusted to predict a half life of 7 years. The same clearance value was used with different starting conditions (concentration of TCDD in tissues) and the model produced a half life of 7 years. Next the model was used with a constant daily intake as an input. Figure 8-3 shows the resulting profile of TCDD in the fat. Figure 8-3 shows the results of a model run using an input of 0.44pg/kg/day. Note that the steady state fat concentrations are approximately 7.0 ppt. The clearance rate used in this model run had been independently determined in the previous run based on reported half life values. Thus, for these conditions this model does an adequate job of predicting tissue levels of TCDD. Figure 8-4 shows a similar profile for a 0.30 pg/kg/day

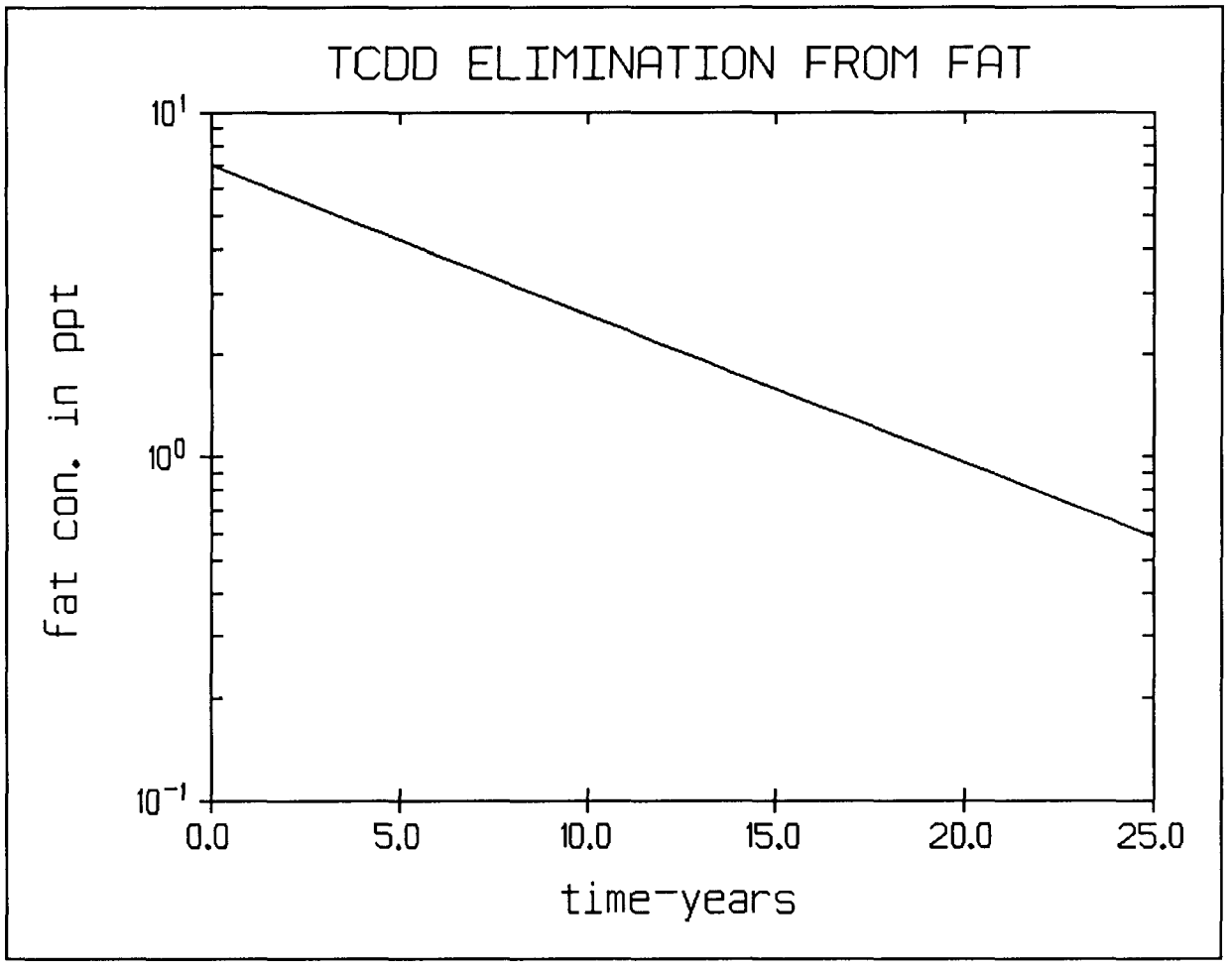


Figure 8-2: Model Estimates of Elimination of 2,3,7,8-TCDD from Fat.

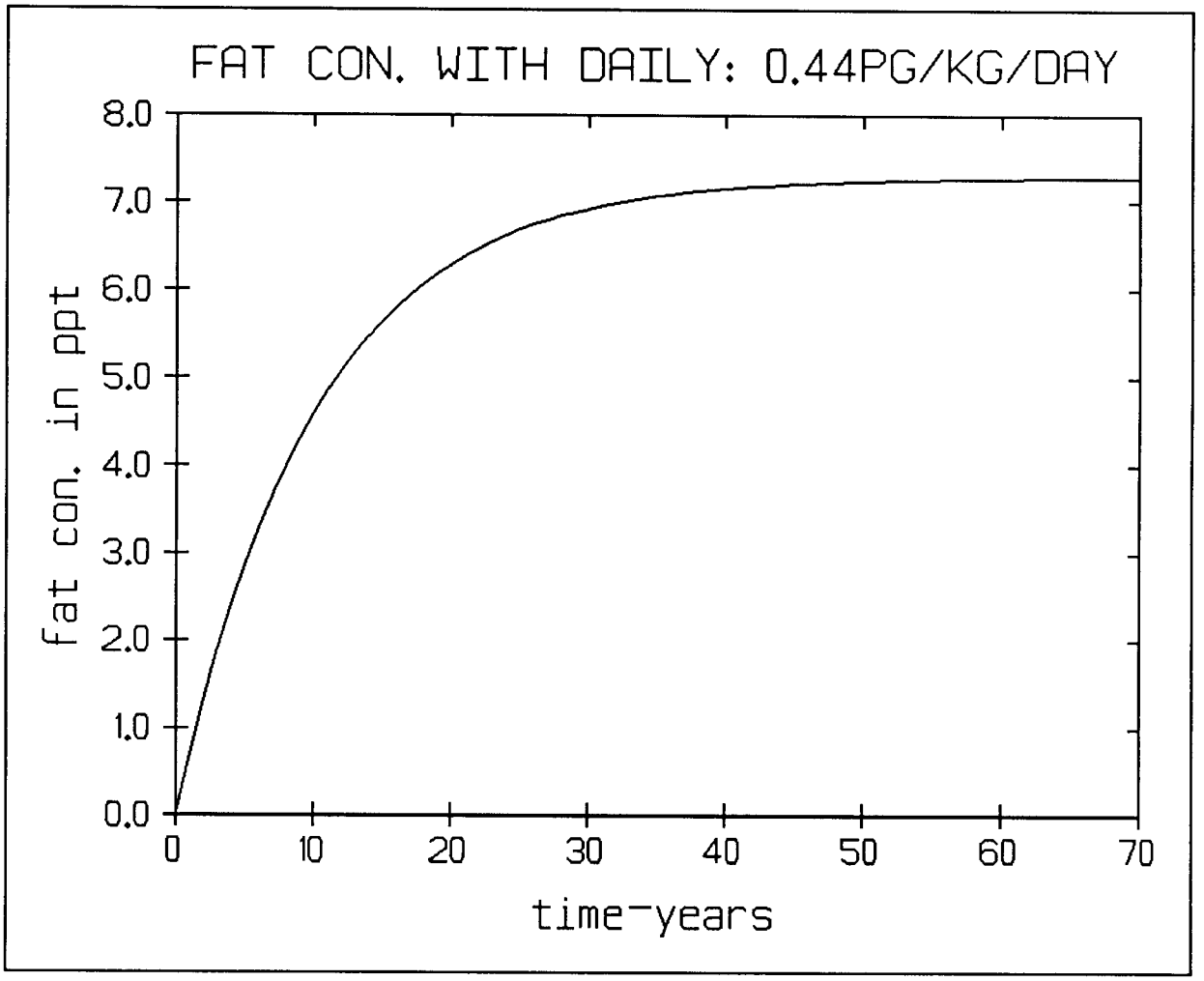


Figure 8-3: Accumulation of TCDD in Fat with 0.44 pg/kg/day dose - Human.

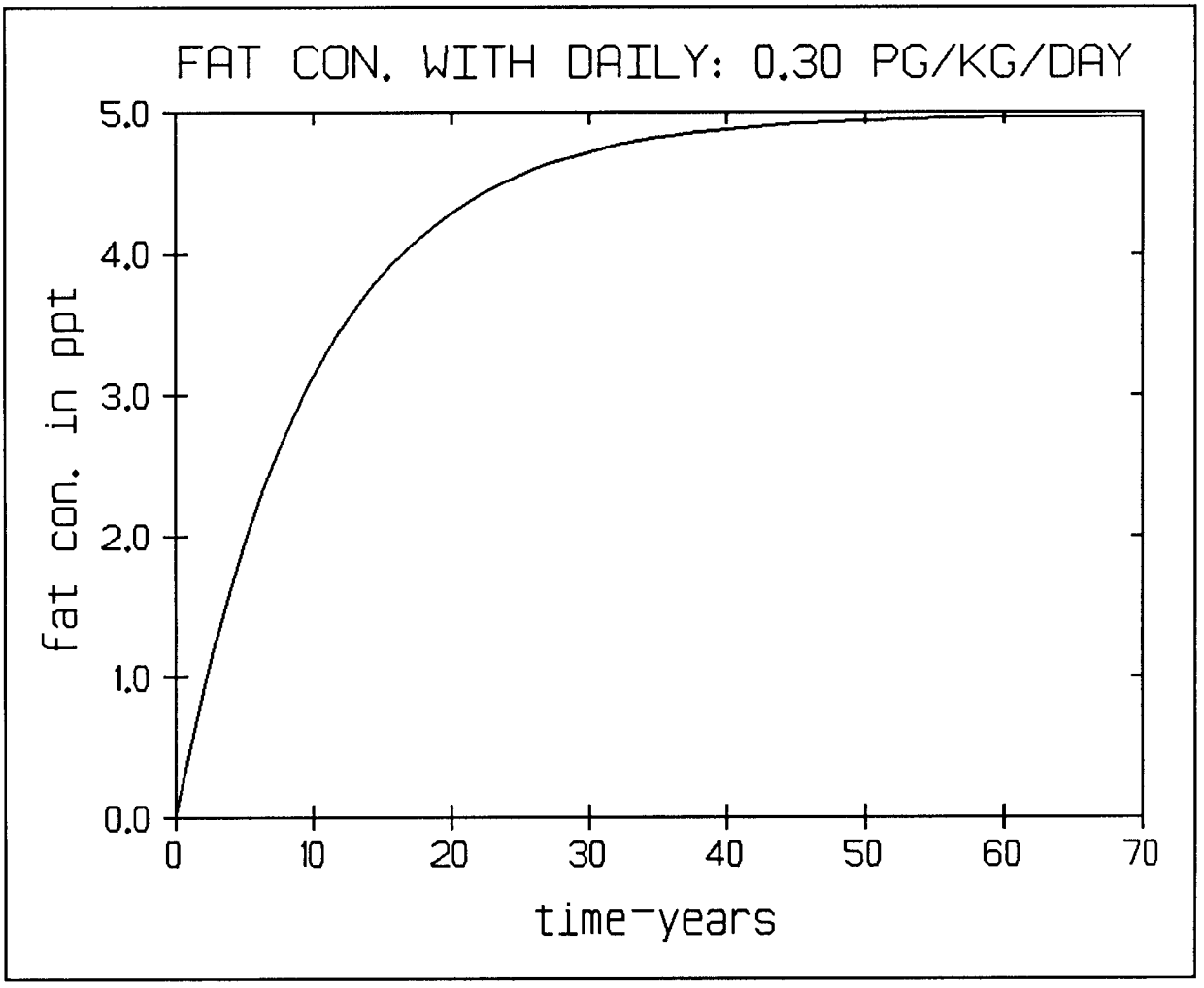


Figure 8-4: Accumulation of TCDD in Fat with 0.30 pg/kg/day dose - Human.

dose. The daily intakes chosen were similar to those calculated by the steady state equations in section 8.1.2 (Table 8-1). Note that the steady state fat levels predicted by the pharmacokinetic model agree closely with those used as a starting point for the steady state calculation using seven years for the half life. This three compartment model appears to provide reasonable approximations of body burden, at least under the circumstances that it has currently been tested. The necessary information for use with other substances in this series include the equilibrium distribution ratios between fat and

blood and between fat and the rest of the body. An estimate of the half life is also needed in order to establish an appropriate value for the clearance term used in the model. Table 8-3 shows the results when the model was adjusted for other substances. The TEF (Toxic Equivalency Factors) based intakes, along with those for 2,3,7,8-TCDD and others, can then added together as so desired to arrive at a TEF based total daily intake for all the TCDDs, TCDFs, and PCBs of interest to a particular assessment.

Table 8-3: Model Determined Daily Intakes

Compound	Half life (yrs)[a]	Fat Con. (ppt)[b]	Intake (pg/kg/day)	Intake-TEF (pg/kg/day)
OCDD	50[c]	1174.0	23.48	0.02
1,2,3,7,8 -PeCDF	8	2.8	0.55	0.03
2,3,4,7,8 -PeCDF	8	13.0	0.93	0.47

[a]Taken from Table 8-2
[b]Take from Schecter (1991)
[c]Value determined by Schlatter (1991)

8.3.3 Determining Liver Concentrations from Fat Levels

As mentioned previously, higher resolution PBPK models are necessary to estimate and predict concentration at cellular and sub-cellular targets (*vidae supra*). At the present time many of the data necessary to develop and apply these models to estimate target tissue or cellular dose in humans are lacking. Andersen and Greenlee (1991) provide an approach to use the equations of a PBPK model (Leung et al., 1990) to predict liver concentrations from monitored fat concentrations. It should also be noted that blood or milk concentrations expressed on a per lipid basis could also be used. Basically the

approach calculates the ratio of fat to liver concentration by dividing the tissue concentration equations of the PBPK model.

$$\frac{C_L}{C_F} = \left[\frac{P_L}{P_F} + \frac{1}{V_L\ P_F}\left(\frac{BM_1}{KB_1 + CVL} + \frac{BM_2(T)}{KB_2 + CVL}\right)\right]\frac{CVL}{CVF} \qquad (8\text{-}26)$$

Where:

C_L, C_F	=	Concentrations of toxin in liver and fat
C_{VL}, C_{VF}	=	Concentrations of toxin in venous blood from liver and fat
P_L, P_F	=	Liver:blood and Fat:blood partition coefficients of toxin
V_L	=	Volume of Liver compartment
BM_1, KB_1	=	Forward and reverse binding constants for binding of toxin with Ah receptor
$BM_2(T)$	=	Forward binding constant with microsomal binding protein - time dependent upon the amount of inducible microsomal binding protein
KB_2	=	Reverse binding constant for microsomal binding protein

Source: Andersen and Greenlee, 1991.

Andersen and Greenlee (1991) go on to further simplify the previous equation for various limiting conditions. For the case of very low doses where $CVL << KB_1$, KB_2 and their is no induction of the cytochrome P450 enzyme (microsomal binding protein) the equation simplifies to:

$$\frac{C_L}{C_F} = \frac{P_L}{P_F} + \frac{1}{P_F\ V_L}\left[\frac{BM_1}{KB_1} + \frac{BM_{2(0)}}{KB_2}\right] \qquad (8\text{-}27)$$

Where:

$BM_{2(0)}$	=	Basal level (non-induced) binding level of P450 (microsomal binding protein)

For the case where $KB_2 << CVL << KB_1$, P450 is maximally induced but binding is less than half saturation equation (8-26) becomes

$$\frac{C_L}{C_F} = \frac{P_L}{P_F} + \frac{1}{P_F\ V_L}\left[\frac{BM_{2(max)}}{KB_2}\right] \qquad (8\text{-}28)$$

Where:

$BM_{2(max)}$ = Maximally induced binding level of P450

For the case where $CVL >> KB_1$, KB_2 equation (8-26) becomes

$$\frac{C_L}{C_F} = \frac{P_L}{P_F} \qquad (8-29)$$

Andersen and Greenlee provide values for each of the above conditions for experimental animals (1991). It can be readily observed from examining equation (8-26) that when binding levels are very small the ratio of concentration between liver and fat is influenced mostly by the ratio of partitioning coefficients.

With this approach and by knowing the necessary parameters equation (8-26) can be used to estimate liver concentrations from fat (including blood lipid and milk lipid) concentrations. Andersen and Greenlee (1991) further suggest that most of the necessary binding parameters can be determined from in-vitro studies. Further, the pharmacokinetic model from which equation (8-26) was derived can be used to estimate and predict cellular concentrations (both free and bound) under various exposure conditions. The equations used by Leung et al. (1990) or other like equations provide estimates of amount bound to intracellular receptors sites, and can thus provide a relationship between multiple binding sites. Denison et al. (1991) suggest that binding to the Ah receptor is only one of several steps necessary for TCDD to have an intracellular toxic effect. As further knowledge becomes available about the mechanism and kinetics of each step the model can be expanded to include these other processes such as DNA enhancing and hormonal modulation. The pharmacokinetic model will therefore become a pharmacodynamic model which will more explicitly link exposure to effect. Also, other tissues can be more specifically described in the model if the mechanism of action data so warrant.

8.4 BIOAVAILABILITY AND TISSUE DISTRIBUTION

In a companion document (US EPA, 1992) there is an extensive discussion on the disposition and pharmacokinetics of these compounds of interest in animals and humans. For detailed descriptions of the pharmacokinetic and bioavailability studies and findings the

reader is referred to that document. Following is a brief summary of the pertinent findings related to bioavailability in humans.

8.4.1 Bioavailability

Following ingestion of a material containing 2,3,7,8-TCDD or other toxic species, the toxic effect of the material is modified by the degree of absorption, principally in the small intestine. In several experimental studies, investigators administered 2,3,7,8-TCDD-containing environmental matrices to experimental animals, and measured parameters relating to bioavailability. These studies included quantification of 2,3,7,8-TCDD in liver and other tissues following treatment; comparison of toxicities of contaminated environmental materials with pure 2,3,7,8-TCDD; and examination of enzyme induction. The results of these different approaches, their limitations, and needs for further research are discussed below.

8.4.1.1 Bioavailability Data

Umbreit et al. (1985, 1986a,b) conducted experiments in guinea pigs, administering 2,3,7,8-TCDD in corn oil, 2,3,7,8-TCDD added to chemically decontaminated soil, or soil from two industrial sites in Newark, New Jersey (a manufacturing site and a salvage site) contaminated with CDDs. 2,3,7,8-TCDD was the principal lower chlorinated isomer (dioxin or furan) present in the soil from the manufacturing site (for which a chemical analysis was presented). Soil from the manufacturing site was found to have 1,500 to 2,500 ppb 2,3,7,8-TCDD under soxhlet extraction; release under ambient temperature manual solvent extraction was much lower, reported as "> 2.5 ppb." The soil from the salvage site was reported as approximately 180 ppb 2,3,7,8-TCDD under soxhlet extraction.

In this study groups of two or four male and two or four female guinea pigs received single gavage doses of the test materials and were observed until death or sacrifice at 60 days. 2,3,7,8-TCDD in corn oil or in recontaminated soil (6 g/kg in both) proved highly toxic, without similar toxicity being observed in animals treated with up to twice this dose of 2,3,7,8-TCDD in the soil from the manufacturing site. The limited data on 2,3,7,8-TCDD levels in the liver showed much higher levels following administration of recontaminated soil versus contaminated soil from the manufacturing site.

Umbreit et al. (1986a) thus demonstrated that gavaged 2,3,7,8-TCDD containing soil from the manufacturing site was substantially less toxic than equivalent doses of 2,3,7,8-TCDD in corn oil. However, quantitative comparison of the effective doses in this study is difficult. Approaches to a quantitative comparison are outlined below.

(1) Guinea pigs receiving 12 ug/kg 2,3,7,8-TCDD in contaminated soil experienced no deaths, while five out of eight guinea pigs receiving 6 ug/kg 2,3,7,8-TCDD in corn oil died, with no groups tested having lower doses in corn oil. Other authors have provided data on the toxic effects of 2,3,7,8-TCDD in corn oil which could aid in the comparison.

McConnell et al. (1984) observed one out of six animals dying at 1 ug/kg and six out of six animals dying at 3 ug/kg. Silkworth et al. (1982) observed three out of six animals dying at 2.5 ug/kg and no deaths out of six at 0.5 ug/kg. Comparing these data directly with the Umbreit et al. results would suggest that the 2,3,7,8-TCDD in the Newark manufacturing site soil was less effective, by a factor of 10 or greater, in producing toxicity than 2,3,7,8-TCDD in corn oil.

(2) Umbreit et al. reported a "slightly reduced" weight gain in guinea pigs receiving 6 ug/kg of 2,3,7,8-TCDD in Newark manufacturing site soil, and a "greater reduction" at the 12 ug/kg dose. No other signs of toxicity were noted in these groups. The animals receiving 6 ug/kg 2,3,7,8-TCDD in corn oil, in contrast, exhibited a marked loss of body weight and showed toxicity and mortality. Silkworth et al. (1982) also provided data on weights of guinea pigs receiving 2,3,7,8-TCDD in corn oil. Those receiving 2.5 ug/kg exhibited a marked reduction in weight gain among three out of six survivors, while those receiving 0.5 ug/kg showed a weight gain comparable to vehicle controls. Comparison of this weight data with that of Umbreit et al. suggests that the 2,3,7,8-TCDD in corn oil was more than 5 times but less than 25 times as potent as 2,3,7,8-TCDD in the Newark soil. This comparison assumes that the effect of the Newark manufacturing site soil on weight gain was due to 2,3,7,8-TCDD as opposed to other compounds in the soil. Numerous other dioxin and furan compounds and other chemicals

have been identified in this soil (Umbreit et al., 1987a). It has not been established that 2,3,7,8-TCDD is the sole or prime source of toxicity in the soil.

(3) Umbreit et al. presented liver concentrations of 2,3,7,8-TCDD after death or sacrifice at 60 days following gavage. Much lower concentrations of 2,3,7,8-TCDD were found in the livers of animals receiving soil from the manufacturing site compared with those receiving the dose in corn oil. There are, however, two factors that limit the conclusions than can be drawn from this comparison.

First, the corn oil group experienced major toxicity and weight loss, particularly complete loss of body fat. These changes may have affected the partitioning of 2,3,7,8-TCDD within the body, leading to a higher concentration in the livers of the animals experiencing toxicity. Second, the animals gavaged with corn oil died early--half were dead by 26 days, while all of the guinea pigs treated with soil survived to 60 days (with the exception of one gavage death). The U.S. EPA (1985c) reported a half-life for 2,3,7,8-TCDD elimination of 30 $\pm$ 6 or 22 to 43 days from two studies in guinea pigs. Additionally, the U.S. EPA (1985c) stated that elimination in the guinea pig may follow zero-order kinetics. Differences in elimination due to differences in periods of survival are likely to have affected the relative quantities of 2,3,7,8-TCDD found in the livers of the test groups.

Perhaps a more appropriate comparison can be made with the four animals receiving 0.32 ug/kg of 2,3,7,8-TCDD in contaminated soil from the Newark salvage site. These animals experienced no reported toxic signs (weight data not presented) and survived the full 60-day experiment. Approximately 6% of the gavage dose was found in the liver of these animals, while only about 0.06% of the gavage dose was found in the livers of guinea pigs in the 12 ug/kg group receiving the Newark manufacturing site soil. This would suggest that the 2,3,7,8-TCDD in the manufacturing site soil was 100 times less bioavailable. However, given the different doses used and the fact that only a single pooled sample was analyzed for 2,3,7,8-TCDD in each group, caution must be used in interpreting this comparison.

The 2,3,7,8-TCDD in soil from the salvage site was substantially bioavailable, based on the single liver tissue analysis. Approximately 6% of the administered dose was

recovered from the livers of these animals at 60 days. This can be compared with data on hamsters given 2,3,7,8-TCDD in corn oil by McConnell et al. (1984), where approximately 8% of the 2,3,7,8-TCDD could be recovered in the 1 ug/kg dose group among survivors at 30 days.

McConnell et al. (1984) treated Hartley guinea pigs (2.5 weeks old) with single gavage doses of either 2,3,7,8-TCDD or dioxin contaminated soil from two sites in Missouri. The 2,3,7,8-TCDD concentrations from the two sites were reported at 700 and 880 ppb respectively; total tetrachlorodibenzofurans (TCDF) concentrations in the soil were 40 to 80 ppb, and polychlorinated biphenyls (PCB) concentrations were 3 to 4 ppm. Taking into account the relative toxicities, the authors concluded that toxicity from the other compounds was likely to be small compared with that from 2,3,7,8-TCDD. Livers were analyzed for 2,3,7,8-TCDD at death or sacrifice at 30 days following treatment. Treatment deaths occurred between 5 and 21 days post-gavage.

Guinea pigs that died exhibited severe loss of body fat, markedly reduced thymus and testicle size, and adrenal hemorrhage. No adverse affects were noted in animals treated with decontaminated soil. For 2,3,7,8-TCDD in corn oil and for both contaminated soils, there were clear dose-responses in mortality. The calculated LD_{50} values for the two soil types were lower than the LD_{50} for 2,3,7,8-TCDD in corn oil by a factor of three to four.

There was a dose-response between the liver concentration of 2,3,7,8-TCDD and the gavage dose; the details of this relationship are complex. Animals dying during the experiment had liver concentrations a factor of 1.4 to 3.2 higher than animals in the same dose groups who survived 30 days. This observation makes quantification of the dose-response relationships difficult (all or most of the animals in the low-dose groups survived the experiment, while all of the animals in the high-dose groups died). When the liver concentrations of 2,3,7,8-TCDD in animals dying early at the middle and high-dose groups are compared, there appears to be a greater-than-linear increase in liver concentration with dose for the Times Beach and Minker Stout soil groups, with a 3.3-fold increase in dose producing a 10- to 13-fold increase in liver concentration.

Liver concentrations of animals in the different dosing groups can best be compared among groups that experienced similar mortality.

(1) Animals in dose groups in which all animals died within 30 days: 2,3,7,8-

TCDD in corn oil, approximately 20% of the administered dose was in the liver. For the soil-treated groups, 13% and 11% of the doses, respectively, were in the liver. Comparison of these data suggest that 2,3,7,8-TCDD was approximately twice as available through corn oil as through soil.

(2) Animals surviving the 30-day experiment (in groups where at least 4 out of 6 survived): For 2,3,7,8-TCDD in corn oil, 7.5% of the administered dose was in the liver. For soil-treated animals < 3.6, 1.3, < 4.2, and 2.0% of the doses, respectively, were in the liver. Comparison here would suggest that 2,3,7,8-TCDD was approximately four times as available through corn oil as through soil.

The authors note that the differences in liver concentrations observed in the study may reflect varying partitioning of the 2,3,7,8-TCDD among internal organs, since dying animals suffered major loss of body weight and fat content. In addition, surviving animals would have had greater opportunity to metabolize and excrete 2,3,7,8-TCDD due to a longer lifetime.

Umbreit et al. (1986a) reported additional chemical analyses of the Times Beach soil. Soxhlet extraction of the Times Beach soil yielded a similar quantity of 2,3,7,8-TCDD to the solvent extraction reported by McConnell et al. (1984). This is in contrast to the Newark manufacturing site soil used in the Umbreit et al. (1987a) experiments, where only a small fraction of soxhlet-extractable 2,3,7,8-TCDD was extractable by the solvent extraction methodology used by McConnell et al. (1984).

McConnell et al. (1984) also reported an experiment in which groups of six Sprague-Dawley rats were given single gavage doses of 2,3,7,8-TCDD in corn oil or dioxin-contaminated soil from the Minker site. Induction of aryl hydrocarbon hydroxylase (AHH) in the rat livers was measured at sacrifice 6 days after dosing. Experimental doses ranged from 0.4 to 5.0 ug/kg 2,3,7,8-TCDD. Measured AHH induction was similar for groups receiving 2,3,7,8-TCDD in corn oil or receiving contaminated soil containing nearly equal doses of 2,3,7,8-TCDD. For example (based on the rate of formation of 3-hydroxybenzo[a]pyrene), AHH activity was measured at 1,269 pmole min^{-1} mg^{-1} for the group receiving 5 ug/kg 2,3,7,8-TCDD in corn oil and at 1,230 pmole min^{-1} mg^{-1} for the group receiving 5.5 ug/kg 2,3,7,8-TCDD in contaminated soil. For the five dose groups, the AHH activity for the soil group ranged from 50% to 110% of the activity in the corn

oil group.

The McConnell et al. rat data indicate that the bioavailability of 2,3,7,8-TCDD from the Minker site soil was at least 50% of that of equivalent doses of 2,3,7,8-TCDD in corn oil.

Lucier et al. (1986) provided additional information on the induction of hepatic enzymes in rats by the 2,3,7,8-TCDD contaminated soil from the Minker site tested by McConnell et al. (1984). AHH induction was similar for the groups of rats receiving 2,3,7,8-TCDD in corn oil and contaminated soil (within a factor of two) over a broader range of doses (0.015 ug/kg to 5 ug/kg) than reported by McConnell et al. In a second enzyme assay using the same animals, UDP glucuronyltransferase activity was found to be slightly higher in groups receiving 2,3,7,8-TCDD in corn oil than groups receiving equal doses in contaminated soil.

Liver concentrations of 2,3,7,8-TCDD for the rats were also reported. For the corn oil vehicle the liver concentrations were 40.8 $\pm$ 6.5 ppb at the 5 ug/kg dose and 7.6 $\pm$ 2.5 ppb at the 1 ug/kg dose. Assuming that the liver comprises 4.0% of body weight, the retention rates for the 5 and 1 ug/kg doses were 33% and 30%, respectively. In rats receiving 2,3,7,8-TCDD in contaminated soil, the 5.5 ug/kg group had liver concentrations of 20.3 $\pm$ 12.9 ppb, and the 1.1 ug/kg group had concentrations of 1.8 $\pm$ 0.3. Thus, retention rates for the 5.5 and 1.1 ug/kg groups are estimated at 14% and 7%, respectively. These data indicate that liver retention in the soil group was 20% to 40% of that in the corn oil vehicle groups.

Umbreit et al (1986b) report additional studies of mortality in guinea pigs treated with soil containing 2,3,7,8-TCDD from Newark (manufacturing site) and Missouri (Times Beach) previously tested by Umbreit et al (1985, 1986a) and McConnell (1984), respectively. Guinea pigs received a single gavage dose of a soil suspension and were observed for 60 days. After autopsy, deaths were classified as whether or not they appeared to be due to TCDD toxicity. Substantial mortality (25% overall) from conditions not attributed to TCDD was observed across all groups.

The data for both the Newark and Missouri sites are similar in trend for the previous data on these sites; and clearly indicate the greater toxicity of the Newark soil for given equal administered doses of 2,3,7,8-TCDD. With larger groups of guinea pig studied, a toxicity-related death was observed in both the 5 and 10 mg/kg dose groups for Newark

soil while no deaths were observed in corresponding dose groups (6 and 12 mg/kg) with fewer animals in Umbreit et al (1986a).

Comparing groups within this study, similar mortality (1 or 2 deaths in 10 to 16 animals) was seen in both the 5 and 10 ug/kg Newark groups and the 1 and 3 ug/kg Missouri groups. These results suggest that the toxicity of these materials, differs by an order of magnitude or less. As noted above the degree to which toxicity from these soils can be attributed to 2,3,7,8-TCDD in the presence of numerous other related toxic compounds is not known. 2,3,7,8-TCDD tissue concentrations were not reported in this work.

In another comparative study Umbreit et al. (1987b) compared the Newark manufacturing site and Times Beach soils in the induction of aryl hydrocarbon hydroxylase (AHH) in rats. While the use of only single dose levels prevents detailed analysis, the two soils proved quite similar in their ability to induce AHH. The explanation for the difference in this finding from those observed in the toxicity studies discussed above is not clear, but may relate to the presence of other toxic and/or AHH inducing compounds.

Umbreit et al. (1987a) report a reproductive toxicity study with soils from the Newark manufacturing site and salvage yard previously studied by Umbreit (1986a). Female mice were treated thrice weekly with soil from these sites, with treatment continuing through fertilization to weaning of pups. The total doses of 2,3,7,8-TCDD received by the mice were 720 ug/kg in manufacturing site soil, and 86 ug/kg in salvage yard soil. A corn oil vehicle group and are contaminated soil group received a total of 225 ug/kg.

Deaths in animals showing "classic signs" of TCDD toxicity were observed in the corn oil and recontaminated soil groups, and indicate appreciable bioavailability of 2,3,7,8-TCDD. Deaths were also observed in animals receiving manufacturing site soil but the authors did not observe "classic signs" of TCDD toxicity. Fewer live pups born and fewer pups surviving until weaning were observed in the manufacturing site soil group compared with those receiving decontaminated soil. TCDD completely blocked reproduction in the corn oil and recontaminated soil groups. The results of this study demonstrate acute and reproductive effects occurred in animals receiving manufacturing site soil. However, these effects were of a lesser magnitude than those seen in animals treated with 2,3,7,8-TCDD in corn oil at a dose three fold lower. The authors note the presence of substantial

quantities of other toxic substances in the manufacturing site soil (chemical analyses presented). No toxic effects were noted in animals treated with salvage site soil, who received a much smaller 2,3,7,8-TCDD dose. The data does not allow a quantitative evaluation of the bioavailability of 2,3,7,8-TCDD.

Kaminski et al. (1985) and Silkworth et al. (1982) reported the results of a series of studies on the toxicity of soot containing dioxin and furan compounds from a fire involving transformer fluid containing PCBs. Hartley guinea pigs (500 to 600 g) received single oral doses of soot in an aqueous vehicle, a soxhlet extract of the soot in the same vehicle, or 2,3,7,8-TCDD in either an aqueous vehicle or corn oil.

The soot was reported to contain 2.8 to 2.9 ppm 2,3,7,8-TCDD and 124 to 273 ppm 2,3,7,8-TCDF. The total polychlorinated dibenzofuran content was estimated at 5,000 ppm. Animal weights and mortality were recorded for 42 days, at which point the survivors were sacrificed and LD_{50} values were calculated. Blood chemistry and a pathologic examination were performed at sacrifice.

Silkworth et al. (1982) noted that the LD_{50}'s for contaminated soot and soot extract were similar at 410 and 327 equivalent ug/kg, indicating that the matrix had only a small effect on toxicity. If expressed in terms of the content of 2,3,7,8-TCDD, the LD_{50} from soot is 2.5 ug/kg, which is a factor of seven below the LD_{50} for 2,3,7,8-TCDD in an aqueous vehicle, suggesting that other compounds contributed to the toxicity of the soot and soot extract.

The authors stated that they adopted an aqueous vehicle in these experiments because it was nontoxic and provided a stable suspension of soot; they regarded this vehicle as more appropriate for modeling of human exposure conditions than an oil vehicle. The data from these experiments also demonstrate that use of an oil vehicle leads to substantially greater 2,3,7,8-TCDD toxicity than does an aqueous vehicle.

Comparison of mortality and weight loss in groups of female guinea pigs receiving 500 ug/kg of soot or the equivalent amount of soot extract suggests that the extract may be somewhat more toxic; however, all six animals died in the 1,000 ug/kg soot group, while four out of five died in the 500 ug/kg extract group. Taken together, these data indicate that the soxhlet extract of soot in an aqueous vehicle was between one and two times as toxic as the soot itself. It is likely that a larger difference in toxicity would have been observed if the soot extract was in an oil vehicle.

Van den Berg et al. (1983) fed small groups of male Wistar rats fly ash from a municipal incinerator (pretreated with HCl) containing dioxins and furans, a soxhlet extract of the fly ash, or a purified extract of the ash that was obtained using column chromatography. 2,3,7,8-TCDD was present as 3.3% of the TCDD isomer group in the fly ash extract. (The authors did not specify whether this reference was to crude or purified extract.) 2,3,7,8-TCDF was present as 17.9% of the tetra-CDF isomer group in the extract. The rats were fed 2 g/d fly ash mixed with diet or the residual from 2 mL/d extract after the extract was mixed with diet and the solvent was evaporated. The animals were exposed to the treated diet for 19 days, and then sacrificed, and the liver tissue was analyzed for the presence of dioxins and furans.

Approximately 1% of the 2,3,7,8-TCDD dose from fly ash was retained in the liver, and approximately 4% of the dose of this isomer from fly ash extract was so retained. The corresponding percentages for 2,3,7,8-TCDF are 0.3 and 1.0. Data on the retention of isomer groups in adipose tissue were presented for the extract-treated groups but not for the fly-ash-treated group. The concentrations of the various isomers in adipose tissue are comparable to, or less than, the concentrations in liver tissue.

The U.S. EPA (1985b) reported a half-life for elimination of 2,3,7,8-TCDD in the rat of 20 days at high dose. If a similar half-life is assumed in this experiment, the quantities of 2,3,7,8-TCDD in the animals at the end of the 19-day feeding experiment would be significantly less than the absorbed dose, but still of the same order of magnitude. However, the recovery percentages in this study are low for both the fly ash and fly ash extract groups in comparison with other studies in which 2,3,7,8-TCDD was administered to rats. Fries and Marrow (1975) fed rats diets containing 7 or 20 ppb of 2,3,7,8-TCDD for a period of up to 42 days. After 14 days of feeding, the rat livers contained an average of 32% of the cumulative administered dose; at 28 days, 21% of the dose; and at 42 days, 18% of the dose. Thus, in the van den Berg et al. study, the liver retention of 2,3,7,8-TCDD for the fly ash extract group is a factor of five to eight below what could be anticipated for the Fries and Marrow data, and the liver retention in the van den Berg group fed soot is a factor of 20 to 30 lower than that seen by Fries and Marrow. Data from Kociba et al. (1976), Rose et al. (1976), and Kociba et al. (1978) lead to similar conclusions to those from the Fries data regarding the fraction of cumulative 2,3,7,8-TCDD dose retained in the rat liver.

An explanation of the low level of recovery for the animals receiving the soxhlet extract of soot is not apparent. It is possible that the presence of multiple compounds affected absorption or metabolism in the rats fed soot and soot extract.

A second approach to the van den Berg et al. data is to compare the ratios of liver concentrations for dioxins in fly-ash-treated animals to the concentrations in extract-treated animals. These ratios, based on measurements in small numbers of animals, indicate a substantial bioavailability of dioxin and furan compounds from the tested fly ash. This availability varied among the different isomers with the value of 0.3 for 2,3,7,8-TCDD, indicating that this isomer was three times as available from fly ash extract as from fly ash.

Van den Berg (1985) fed fly ash (pre-treated with HCl) to Wistar rats, guinea pigs, and Syrian golden hamsters. Fly ash was mixed with standard laboratory diet at 2.5% by weight, and animals were allowed to eat ad libitum. The amount of fly ash consumed by each group of five rodents was determined by the authors. For each species there were three groups of animals each fed fly ash for approximately 32 days (group I), 60 days (group II), or 94 days (group III). Concentrations of dioxin and furan isomer groups in the food were presented, and include 1.4 ng/g TCDD compounds and 2.1 ng/g TCDF compounds.

The authors presented calculated recovery percentages for the cumulative dose of specific isomers in the rodent liver. For 2,3,7,8-TCDD in guinea pigs, 3.7%, 0.9%, and 1.4% of the administered dose was recovered in the liver in groups I, II, and III, respectively. The 32-day (group I) recovery percentage is somewhat higher than seen in the lower dose groups receiving 2,3,7,8-TCDD contaminated soil in McConnell et al. (1984). The value in hamsters was approximately 2% (only reported for group II), and analytical problems prevented this determination in rats. No other TCDD compounds were quantified. Similarly, for 2,3,7,8-TCDF, guinea pigs showed retention of 4.7%, 2.2%, 2.5% of the administered dose in groups I, II, and III, respectively. For both 2,3,7,8-TCDD and 2,3,7,8-TCDF the recovery percentages in guinea pigs at 32 days were approximately a factor of 4 to 15 higher than that observed in the van den Berg et al. (1983) study in rats.

Other TCDD compounds that were present showed comparable or somewhat lower retention, averaging 1% to 2% over the animals groups. No TCDD or TCDF compounds

were detected in hamster liver or analyzed for in rat liver. Higher chlorinated isomers most typically showed retention in the range of 2% to 5% in rat liver and 1% to 3% in guinea pig liver, with the exception of 2,3,4,7,8-PnCDF (9.8%, 8.3%, and 11.3% in the hamster groups). Few other compounds were found in hamster liver, but 2,3,4,7,8-PnCDF was found with a recovery of 5% to 8% and 2,3,4,7,8-HxCDD was found at 3% to 7%.

As with other experiments in which the retention of dioxins in the liver has been determined, these percentages place a lower bound on the bioavailability of the dioxins but, because not all dioxin is localized in the liver, do not permit bioavailability to be estimated without knowledge of the elimination of the administered dose over time and the quantity of dioxins in the remainder of the organism. No positive control group receiving 2,3,7,8-TCDD was included for comparison.

Poiger and Schlatter (1980) conducted several experiments in Sprague-Dawley rats (180 to 220 g) in which liver concentrations of tritium label from 2,3,7,8-TCDD were determined using various doses and vehicles. All experiments consisted of a single gastric intubation of 2,3,7,8-TCDD-containing material, followed by animal sacrifice at predetermined times. The doses used were well below the LD_{50} in the rat (the maximum dose applied was 5 ug/kg), and no deaths or toxic effects were reported.

In a preliminary experiment, rats were treated with 14.7 ng/rat 2,3,7,8-TCDD in ethanol. These data indicate substantial localization of 2,3,7,8-TCDD in the rat liver, with a decrease of a factor of two in the fraction of the dose in the liver between 1 and 4 days. Poiger and Schlatter (1980) conducted all further studies with sacrifice at 24 hours to maximize the recovery of 2,3,7,8-TCDD from the liver.

In a second experiment, the authors administered 2,3,7,8-TCDD doses in ethanol ranging from 15 to 1,070 ng/rat to groups of six rats. They found a graded increase in percentage retained in the liver from 37% $\pm$ 1% at the 15 ng dose to 51% $\pm$ 4% at 280 ng. At the high-dose point, the percentage may have fallen (42% $\pm$ 10% at 1,070 ng).

In a further experiment, 2,3,7,8-TCDD was administered at low dose in a series of vehicles. These data demonstrate that administration of 2,3,7,8-TCDD in soil reduced the retention of the dose in the liver to 66%, or 44% of the retention seen with 2,3,7,8-TCDD in ethanol. The lower value, 44% was obtained for soil that was aged for 8 days at 30-40 ^{o}C following addition of 2,3,7,8-TCDD. This observation is consistent with the findings of other studies reported here that 2,3,7,8-TCDD from environmental soil (naturally aged)

was generally less available than 2,3,7,8-TCDD freshly added to clean samples of these soils. The aqueous suspension of 2,3,7,8-TCDD in activated carbon showed little evidence of bioavailability; this is supported by the authors' measurements showing that 2,3,7,8-TCDD was only slightly extractable from the activated carbon matrix by various solvents. In contrast, 58% to 70% of 2,3,7,8-TCDD could be recovered from soil samples by washing with hexane/acetone (4:1 v/v).

Poiger and Schlatter (1980) also presented results from several skin application experiments with TCDD-containing materials using rats and rabbits (not reviewed here).

Bonaccorsi et al (1984) reported the results of a study of gut absorption of 2,3,7,8-TCDD from soil taken from the Seveso, Italy accident site. Soil containing 81 $\pm$ 8 ppb 2,3,7,8-TCDD from the "highly contaminated" area in Seveso was administered to albino male rabbits (2.6 $\pm$ 0.3 kg) in daily gavage doses for seven days. Additional samples of clean soil were spiked with 2,3,7,8-TCDD in the laboratory to yield 10 and 40 ppb contamination levels and were administered to rabbits following the same protocol. For comparison, rabbits were also treated with 2,3,7,8-TCDD in solution in acetone-vegetable oil (1:6) or alcohol-water (1:1). Rabbits were sacrificed on the day after treatment stopped and liver concentrations of 2,3,7,8-TCDD were measured. The authors did not remark on the presence or absence of toxicity in the treated rabbits. EPA (1985a) reports values for the single dose LD_{50} of 2,3,7,8-TCDD in rabbits of 115 and 275 ug/kg. The total doses received by the rabbits in this study were approximately 54, 107, and 215 ug/kg over seven days. Based on this comparison, there is a likelihood that toxic effects occurred in the Bonaccorsi work, and noted above, toxicity has the potential to affect the tissue concentrations of 2,3,7,8-TCDD. For this reason the most appropriate comparisons among these data are between groups showing similar liver concentrations of 2,3,7,8-TCDD, which may then be inferred to have experienced similar toxic effects.

That this method of comparison is desirable can also be seen from the Bonaccorsi data, where both solvent vehicle groups and the spiked soil groups show an increase of the fraction of the dose in the liver at the higher administered doses. However, it should be mentioned that use of two different solvent vehicles complicates interpretation. Similar liver concentrations of 2,3,7,8-TCDD were seen in the 40 ug/d solvent vehicle and 80 ug/d Seveso soil groups. Comparing the percentage of liver retention in these two groups indicates absorption from Seveso soil was 40% of that from the solvent vehicle. Using

the same approach, comparison of the 80 ug/d solvent vehicle and 160 ug/d Seveso soil groups indicates that absorption from the soil was 41% of that from the solvent.

The same approach can be used to compare absorption from the solvent vehicle and from the spiked soil. In this case the 40 ug/d solvent vehicle group had the liver concentrations closest to either the 40 or 80 ug/d spiked soil groups. Comparison of the percentage of dose in the liver indicates absorption from spiked soil is 68-73% of that from the solvent vehicle. Bonaccorsi et al (1984) work conducted with either aged or non-aged spiked soil but do not present data to allow a comparison of these groups.

Shu et al. (1987, as cited by Leung and Paustenbach, 1987) study of 2,3,7,8-TCDD from the Missouri site tested by McConnell et al. (1984). Their paper "reports an oral bioavailability of approximately 43% in the rat dosed with environmentally contaminated soil from Times Beach, Missouri. This figure did not change significantly over a 500-fold dose range of 2 to 1450 ng 2,3,7,8-TCDD per kg of body weight for soil contaminated with approximately 2, 30 or 60 ppb of 2,3,7,8-TCDD. The data from this study is not now available to the Exposure Assessment Group for review.

8.4.1.2 Summary of Bioavailability

Table 8-4 summarizes data that are pertinent to the bioavailability of 2,3,7,8-TCDD from environmental matrices. Studies of bioavailability, which examined soil samples, soot, and fly ash, have utilized three methodologies: measuring acute toxicity, retention of 2,3,7,8-TCDD in the liver, and induction of hepatic enzymes.

Among the five samples of soil from contaminated sites that have been tested, three have shown substantial bioavailability, e.g., 25% to 50%, when compared with 2,3,7,8-TCDD in corn oil gavage. A fourth soil sample was compared with 2,3,7,8-TCDD administered in a solvent vehicle, and fell in this range. The fifth soil, tested by Umbreit et al. (1986a,b; 1987a,b) showed bioavailability substantially less than the other soils tested. While difficult to gauge quantitatively, dioxin from this soil may be an order of magnitude less available than from the other soils.

Additionally, three samples of soil spiked with 2,3,7,8-TCDD have been tested for bioavailability, including one sample in which the 2,3,7,8-TCDD was incubated with soil at an elevated temperature. The 2,3,7,8-TCDD added to these soil samples proved to be highly available (e.g., 40% to 70%).

In one study, soot from a transformer fire containing dioxins and furans proved similarly toxic to a soxhlet extract of the soot in an aqueous vehicle. However, the soot extract may have proved more toxic if delivered in corn oil, as was 2,3,7,8-TCDD in the soil studies. The availability of 2,3,7,8-TCDD and other dioxins and furans from incinerator fly ash have been addressed by van den Berg et al. in extended feeding studies. In these studies, liver retention of 2,3,7,8-TCDD from either fly ash or fly ash extract proved low, with availability from fly ash being approximately 25% of that from the extract.

The individual studies reviewed have a variety of limitations, as discussed in the preceding text. A notable limitation was that some experiments were conducted at using highly toxic doses of 2,3,7,8-TCDD, so that determination of bioavailability was complicated by wasting and early death of the test animals. It should also be noted that, while the relative retention of 2,3,7,8-TCDD in the liver can serve as an appropriate indication of differences in bioavailability between samples, the percentage of dose found in the liver only places a lower bound on absorption. This is particularly relevant to experiments where animals have been maintained for many weeks after dosing and an undetermined quantity of 2,3,7,8-TCDD has been excreted.

Finally, toxicity data for mixtures for which both toxicity and bioavailability of individual compounds may vary are difficult to interpret quantitatively in terms of bioavailability.

As presented in U.S. EPA (1985c), Rose et al. (1976) determined gut absorption of 2,3,7,8-TCDD in a 1:25 mixture of acetone to corn oil (by volume) in the rat. In both single dose and multiple dose experiments, measured absorption was approximately 85%. Assuming that absorption from pure corn oil is similar to that from this mixture, and assuming that absorption in other species for which data are not available is similar, the 85% factor can be applied to the data presented here to obtain an approximate range for typical 2,3,7,8-TCDD absorption from soil. Using this factor, the estimated relative bioavailability of 2,3,7,8-TCDD from soil is 25% to 50% and, when compared with corn oil, provides an estimate of gut absorption of 20% to 40% of ingested 2,3,7,8-TCDD in soil. This estimate is comparable with the 20% to 26% absorption from 2,3,7,8-TCDD treated soil from the work of Poiger and Schlatter (1980).

Recognizing these limitations, the weight of evidence indicates that 2,3,7,8-TCDD

is often highly available from environmental materials. However, in one tested soil sample the compound was substantially less bioavailable. While the data are too sparse to allow a prediction as to whether a particular environmental sample will prove more or less bioavailable, one important suggestion has emerged. In the two samples that have proved least bioavailable (the Umbreit et al. (1986a) manufacturing site soil sample, and 2,3,7,8-TCDD on activated carbon tested by Poiger and Schlatter (1980)) the 2,3,7,8-TCDD was largely resistant to solvent extraction. This was not the case for more bioavailable materials.

Further research, using short-term experiments in which animals are handled under identical conditions and are fed dioxins in different media, is needed for an improved comparison of absorption between different environmental samples. Acutely toxic doses should be avoided to ensure that tissue concentrations are directly interpretable. Experiments studying both tissue retention and enzyme induction should prove valuable for this research. Whole-body levels of 2,3,7,8-TCDD need to be related to liver concentrations, and the effects of metabolism need to be addressed. The vehicle of administration has been shown to affect acute 2,3,7,8-TCDD toxicity, and vehicle effects need to be considered in designing experiments.

8.4.2 Distribution

Ryan et al. (1985) examined the distribution of 2,3, 7,8-TCDD in two humans at autopsy. On a weight basis, they determined that there were 6 ppt of TCDD in fat, 2 ppt in liver and below levels of detection in kidney and muscle. They reported that on a per lipid basis the levels were similar between tissues. It is important to note that one of these subjects suffered from a fatty liver syndrome, possibly resulting in higher levels in the liver than might normally be found in healthy individuals.

Poiger and Schlatter (1986) estimated that about 90% of the total body burden of 2,3,7,8-TCDD was sequestered in fat. Levels of 2,3,7,8-TCDD averaging 5-10 ppt have been reported for background populations in St. Louis, MO, by Graham et al. (1986), in Atlanta, GA, and Utah by Patterson et al. (1986). These data consistent with the lipid bioconcentrations assumptions made in calculations of daily intakes (*vidae supra*).

Patterson et al. (1987) developed a high resolution gas chromatographic/high resolution mass spectrometric analysis for 2,3,7,8-TCDD in human serum. A high

correlation between adipose tissue and serum concentrations when adjusted for total lipid content. The reader is referred to other documents (US EPA, 1992; Schlatter, 1991; Schecter, 1991) for more details on the distribution and elimination.

8.4.3 Determination of Daily Intake Dose from Exposure Concentrations

As was discussed in Section 8.3.1 it would be most advantageous to know more about the kinetics of absorption in the various animal species and the human. This is necessary for both extrapolation between species in the risk assessment and for determining body burdens from levels in the exposure media. For the time being, until more data become available regarding the kinetic absorption constants a slightly modified approach can be used. Basically, the needed information is the concentration of the toxins in the media, the fraction of toxin absorbed from each of the media, and the amount of media coming into contact with the body. The following equation describes this in more detail.

$$D = \sum f_i \, C_i \, U_i \qquad (8\text{-}30)$$

Where:

D	=	Daily Intake
f_i	=	Fraction absorbed from various i media
C_i	=	Concentration in various i media
U_i	=	Amount of Contact of i media, (gm/time for food, L/time for air, etc.)

This approach should be used with caution. The major assumption which impacts upon equation (8-30) is that the fraction absorbed is constant across various concentrations and doses. As discussed in Section 8.4.2 and in US EPA (1992) this assumption cannot always be considered to be sound. It most probably only applies at low doses and within any one species. It is an approach which, with care, can be used for certain conditions to give estimates until more reliable kinetic absorption data become available.

REFERENCES FOR CHAPTER 8

Andersen, Melvin E.; Greenlee, Willam F. (1991) Biological determinants of TCDD pharmacokinetics and their relation a biologically based risk Assessment in Biological Basis for Risk Assessment of Dioxins and Related Compounds, Michael Gallo, Robert Scheuplein and Kees Van Der Heijden, eds.; Banbury Report 35, Cold Spring Harbor Laboratory Press.

Bonaccorsi, A.; diDomenico, A.; Fanelli, R.; Merli, F.; Motta, R.; Vanzati, R.; Zapponi, G.A. (1984) The influence of soil particles adsorption on 2,3,7,8-tetra-chlorodibenzo-p-dioxin biological uptake in the rabbit. Arch. Toxicolo. Suppl. 7:431-434.

Bowman, R.E.; Schantz, S.L.; Weerasinghe, N.C.A.; Gross, M.L.; Barsotti, D.A. (1989) Chronic dietary intake of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) at 5 or 25 parts per trillion in the monkey: TCDD kinetics and dose-effect estimate of reproductive toxicity Chemosphere 18(1-6):243-252.

Brewster, D.W.; Elwell, M.R.; Birnbaum, L.S. (1988) Toxicity and disposition of 2,3,4,7,8-pentachlorodibenzofuran (4PeCDF) in the rhesus monkey (Macaca mulatta). Toxicology and Applied Pharmacology 93:231-246.

Denison, Machael S.; Phelphs, Cynthia L.; Dehoog, jane; Kim, Hun J.; Bank, Paula A.; Yao, Eveline F. (1991) Species variation in Ah receptor transformation and DNA binding in Biological Basis for Risk Assessment of Dioxins and Related Compounds, Michael Gallo, Robert Scheuplein and Kees Van Der Heijden, eds.; Banbury Report 35, Cold Spring Harbor Laboratory Press.

Fries, G.F.; Marrow, G.S. (1975) Retention and excretion of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) by rats. J. Agric. Food Chem. 23:265-269.

Fürst, Per; Fürst, Christiane; Wilmers, Klaus (1991) Body burden with PCDD and PCDF from food in Biological Basis for Risk Assessment of Dioxins and Related Compounds, Michael Gallo, Robert Scheuplein and Kees Van Der Heijden, eds.; Banbury Report 35, Cold Spring Harbor Laboratory Press.

Gasiewicz, Thomas A.; Henry, Ellen C. (1991) Different forms of the Ah receptor: possible role in species- and tissue-specific responses to TCDD. in Biological Basis for Risk Assessment of Dioxins and Related Compounds, Michael Gallo, Robert Scheuplein and Kees Van Der Heijden, eds.; Banbury Report 35, Cold Spring Harbor Laboratory Press.

Graham, M.; Hileman, F.D.; Orth, R.G.; Wendling, J.M.; Wilson, J.W. (1986) Chlorocarbons in adipose tissue from a Missouri population. Chemosphere 15:1595-1600.

Huetter, R.; Phillipi, M. (1982) Studies on microbial metabolism of TCDD under laboratory conditions. Permamon Ser. Environ. Sci. 5:87-93.

Kaminski, L.S.; DeCapiro, A.P.; Gierthy, J.F.; Silkworth, J.B.; Tumasonis, C. (1985) The rule of environmental matrices and experimental vehicles i chlorinated dibenzodioxin and dibenzofuran toxicity. Chemosphere 14:685-695.

King, F.G.; Dedrick. R.L.; Collins, J.M.; Matthews, H.B.; Birnbaum, L.S. (1983) Physiological model for the pharmacokinetics of 2,3,7,8-tetra-chlorodibenzofuran in several species. Toxicology and Applied Pharmacology 67:390-

Kociba, R.J.; Deeler, P.A.; Park, C.N.; Gehring, P.J. (1976) 2,3,7,8-Tetra-chlorodibenzo-p-dioxin (TCDD): results of 13-week oral toxicity study in rats. Toxicol. Appl. Pharmacol. 35:553-573.

Kociba, R.J.; Deyes, D.G.; Beyer, J.E., et al. (1978) Results of a two year chronic toxicity and oncogenic study of 2,3,7,8-tetrachlorodibenzo-p-dioxin in rats. Toxicol. Appl. Pharmacol. 46(2):279-303.

Leung, H.; Paustenbach, J. (1987) A proposed occupational exposure limit for 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environmental Health and Safety, Syntex, U.S.A. Inc., Palo Alto, CA.

Leung, H-W.; Ku, R.H.; Paustenbach, D.J.; Andersen, M.E. (1988) A physiologically based pharmacokinetic model for 2,3,7,8-tetra-chlorodibenzo-p-dioxin in C57BL/6J and DBA/2J mice. Toxicology Lett. 42: 15-28.

Leung, H-W.; Poland, A.; Paustenbach, D.J.; Murray, F.J. Andersen, M.E. (1990) Pharmacokinetic of [125I]-2-iodo-3,7,8-trichlorodibenzo-p-dioxin in mice: Analysis with a physiological modeling approach. Toxicology and Applied Pharmacology 103: 411-419.

Luceir, G.W.; Rumbaugh, R.C.; McCoy, A.; Haas, R.; Harvan, D.; Albro, P. (1986) Ingestion of soil contaminated with 2,3,7,8-tetrachlordibenzo-p-dioxin (TCDD) alters hepatic enzyme activities in rats. Fund. Appl. Toxicol. 6: 364-371.

McConnell, E.E.; Lucier, G.W.; Rubaugh, R.C.; et al. (1984) Dioxin in soil: Bioavailability after ingestion by rats and guinea pigs.

Nau, H.; Bab, R.; Neuber, D. (1986) Transfer of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) via placenta milk, and postnatal toxicity in mouse. Arch. Toxicol. 59:36-40.

Nessel, C.S.; Amoruso, M.A.; Umbreit, T.H.; Gallo, M.A. (1990) Hepatic aryl hydrocarbon hydroxylase and cytochrome P450 induction following the transpulmonary absorption of TCDD from intratracheally instilled particles. Fund. Appl. Toxicol. 15:500-509.

Patterson, D.G.; Holler, J.S.; Lapez, C.R., Jr. (1986) High resolution gas chromatographic/high resolution mass spectrometric analysis of human adipose tissue for 2,3,7,8-tetrachorodibenzo-p-dioxin. Anal. Chem. 58: 705-713.

Patterson, D.G.; Hampton, L.; Lapeza, C.R., Jr; (1987) High resolution gas chromatographic/high resolution mass spectrometeric analysis of human serum on whole weight and lipid basis for 2,3,7,8-tetrachlorodibenzo-p-dioxin. Anal. Chem. 59:2000-2005.

Perdew, G.H.; Hollenbeck, C.E. (1990) Analysis of photoaffinity labeled aryl hydrocarbon receptor heterogeneity by two dimensional gel electrophoresis. Biochemistry 29: 6210-6214.

Philippe, M.; Drasnobagew, V.; Zeyer, J.; Huetter, R. (1981) Fate of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in microbial cultures and soil under laboratory conditions. FEMS Symp. 12:2210-2233.

Poiger, H.; Schlatter, C. (1980) Influence of solvents and adsorbents on dermal and intestinal absorption of TCDD. Food Cosmet. Toxicol. 18: 477-481.

Poiger, H.; Schlatter, C. (1986) Pharmacokinetics of 2,3,7,8-TCDD in man. Chemosphere 15(9-12): 1489-1494.

Rose, J.Q.; Ramsey, J.C.; Wentzler, Th.H. (1976) The fate of 2,3,7,8-tetrachlorodibenzo-p-dioxin following single and repeated oral doses to the rat. Toxicol. Appl. Pharmacol. 36:209-226.

Ryan, J.J; Schecter, A.; Lizotte, R.; Sun, W.F.; Miller, L. (1985) Tissue distribution of dioxins and furans in humans from the general population. Chemosphere. 14(6/7): 929-932.

Schecter, Arnold (1991) Dioxins and related chemicals in humans and in the environment in Biological Basis for Risk Assessment of Dioxins and Related Compounds, Michael Gallo, Robert Scheuplein and Kees Van Der Heijden, eds.; Banbury Report 35, Cold Spring Harbor Laboratory Press.

Schlatter, Christian (1991) Data on kinetics of PCDDs and PCDFs as a prerequisite for human risk assessment in Biological Basis for Risk Assessment of Dioxins and Related Compounds, Michael Gallo, Robert Scheuplein and Kees Van Der Heijden, eds.; Banbury Report 35, Cold Spring Harbor Laboratory Press.

Shu, J.; Paustenbach, D.; Murray, F.J. (1988) Bioavailability of soil bound TCDD: Oral bioavailability in the rat. Fund. Appl. Toxicol. 10:648-654.

Silkworth, J.; Marti, D.; DeCapri, A.; Rej, R.; O'Keefe, P.; Kaminsk, L. (1982) Acute toxicity in guinea pigs and rabbits of soot from a polychlorinated biphenyl-containing transformer fire. Toxicology and Applied Pharmacology. 65:425-439.

Stanley, J.S.; Boggess, K.; Onstot, J.; Sack, T.; Remmers, J.; Breen, J.; Kutz, F.W.; Robinson, P.; Mack, G. (1986) PCDDs and PCDFs in human adipose tissues from the EPA FY82 NHATS repository. Chemosphere 15:1605-1612.

Umbreit, T.J.; Patel, D.; Gallo, M.A. (1985) Acute toxicity of TCDD contaminated soil from an industrial site. Chemosphere 14:945-947.

Umbreit, T.H.; Hesse, E.J.; Gallo, M.A. (1986a) Bioavailability of dioxin in soil from a 2.4.5.-T manufacturing site. Science 232:497-499.

Umbreit, T.H.; Hesse, E.J.; Gallo, M.A. (1986b) Comparative toxicity of TCDD contaminated soil from Times Beach, Missouri, and Newark, New Jersey. Chemosphere 15(9-12): 2121-2124.

Umbreit, T.H.; Hesse, E.J.; Gallo, M.S. (1987a) Reproductive toxicity of female mice of dioxin-contaminated soils from a 2,4,5-trichlorooxacetic acid manufacturing site. Arch. Environmental Contamination and Toxicology 16: 461-466.

Umbreit, T.H.; Hesse, E.J.; Gallo, M.S. (1987b) Differential bioavailabilty of 2,3,7,8-Tetrachlorodibenzo-p-dioxin from contaminated soils. American Chemical Society Symposium Series No. 338, "Solving hazardous waste problems: Learning from dioxin.

US EPA (1985a) Compilation of air pollutant emission factors, Vol 1. Office of Air Quality Planning and Standards, Research Triangle Park, N.C.

US EPA (1985b) Rapid assessment of exposure to particulate emission from surface contamination sites. Office of Solid Waste and Emergency Response, Washington, DC EPA-600/8-85-002.

US EPA (1985c) Health assessment document for polychlorinated dibenzo-p-dioxins. Office of Air Planning and Standards, Research Triangle Park, NC. EPA-600/8-84-014F.

US EPA (1992) Health Assessment For 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and related compounds. Preliminary Draft.

van den Berg, M.; Ollie, K.; Hutzinger, O. (1983) Uptake and selective retention in rats of orally administered chlorinated dioxins and dibenzofurans from fly-ash and fly-ash extract. Chemosphere 12:537-544.

van den Berg, M.; Vroom, A.; van Greevenbroek, M.; Olie, K.; Hutzinger, O. (1985) Bioavailability of PCDDs and PCDFs adsorbed on fly ash in the rat, guinea pig, and Syrian golden hamster. Chemosphere 14:865-869.

Whitlock, James P., Jr. (1991) Mechanism of dioxin action: Relevance to risk assessment in Biological Basis for Risk Assessment of Dioxins and Related Compounds, Michael Gallo, Robert Scheuplein and Kees Van Der Heijden, eds.; Banbury Report 35, Cold Spring Harbor Laboratory Press.

9. DEMONSTRATION OF METHODOLOGY

9.1. INTRODUCTION

This document has provided methodologies and background information for conducting site-specific exposure assessments to the dioxin-like compounds. Chapter 2 described physical and chemical properties of these compounds, and Chapter 3 described the occurrence of these compounds in environmental and exposure media. Chapter 4 summarized an overall exposure assessment framework, and Chapter 7 provided details on development of exposure scenarios including exposure pathways and associated parameters. Chapter 5 provided methodologies to estimate exposure media concentrations for four sources of contamination, which were termed source categories. Chapter 6 discussed a principal source of dioxin-like compounds in the environment, incinerators. Incinerator stack emissions and disposal of incinerator fly ash were two of the four source categories identified in Chapter 5. Chapter 6 also discussed and demonstrated the use of the Industrial Source Complex (ISC) model to generate air concentrations of contaminants and deposition rates at various distances from a hypothetical incinerator. Chapter 8 on pharmacokinetics summarized information on uptake and distribution of dioxin-like compounds and developed a model to predict blood levels resulting from exposure.

The purpose of this chapter is to put all this information together and demonstrate the site-specific methodologies that have been developed. For this demonstration, exposure scenarios are developed which are associated with the four source categories. These categories were defined in Chapter 5, and are:

- **On-site soil**: The source of contamination is soil and both the source and exposure site are on the same tract of land.
- **Off-site soil**: The source of contamination is soil and this source is located distant and upgradient/upwind from the site of exposure.
- **Incinerator stack emissions**: Exposed individuals reside downwind of the incinerator and are exposed to resulting air-borne vapor phase contaminants originating at the incinerator, and soil on their property is impacted by deposition of contaminated particulates.

Incinerator ash disposal in a landfill: Contaminated ash is spread onto the surface of an active landfill, and exposed individuals reside upgradient/upwind of the

landfill.

The demonstration in this chapter is structured around what are termed exposure scenarios. As defined in Chapter 7, an exposure scenario includes a description of the physical setting of the source of contamination and the site of exposure, behavior of exposed individuals, and exposure pathways. Chapter 7 also described the objective of exposure assessors to determine "central" and "high end" exposure scenarios. This objective was an important one for this demonstration, and the strategy to design such scenarios is detailed in Section 9.2 below. Three dioxin-like compounds are demonstrated for each of the exposure scenarios, including 2,3,7,8-TCDD, 2,3,4,7,8-PCDF, and 2,3,3',4,4',5,5'-HPCB.

Exposure results, expressed in lifetime average daily doses (LADDs in mg/kg-day), are listed separately for each pathway and for each contaminant. These exposure estimates could be summed across all contaminants for a given exposure pathway to estimate the total risk from that pathway. This procedure uses Toxicity Equivalency Factors (TEFs), as described in Chapter 4, Section 4.3. This summation was not done for this demonstration since this document only focuses on exposure and not risk, and also because TEFs for PCBs have not been well established.

Section 9.2 describes the strategy for development of the demonstration exposure scenarios. Section 9.3. gives a complete summary of the example scenarios. Section 9.4 provides some detail on the three example compounds chosen for demonstration. Section 9.5 describes the source strength terms for the scenarios. Section 9.6 summarizes the results for all scenarios, which are exposure media concentrations for all exposure pathways, and exposure estimates which are Lifetime Average Daily Doses (LADD) for all pathways and for all example compounds.

9.2. STRATEGY FOR DEVISING EXPOSURE SCENARIOS FOR DEMONSTRATION PURPOSES

Chapter 4 of this document culminated with Figure 4.1, a roadmap for assessing exposure to dioxin-like compounds. These procedures assess individual exposures to known sources of contamination. Chapter 7 discussed central and high end exposure patterns and developed parameters for each exposure pathway consistent with these patterns. The demonstration in this chapter attempts to merge procedures for estimating

individual exposures to known sources of contamination and current thoughts on devising central and high end exposure scenarios.

An exposure assessor's first task in determining patterns of exposure is to fully characterize the exposed population in relation to the source of contamination. If the extent of contamination can be characterized, then the exposed population would be limited to those within the geographically bounded area. An example of this situation might be an area impacted by incinerator stack emissions. Chapter 6 demonstrated the use of ISC atmospheric dispersion model on a hypothetical but realistic incinerator using meteorological data from Tampa, Florida, and showed that maximum ambient air concentrations and particulate deposition rates occurred east of the hypothetical incinerator, and that the maximum values of these quantities occurred 200 m east. Simulations were performed for as far away as 40 kilometers in all directions, although results were only displayed (for sake of brevity) for up to 5 kilometers in the easterly direction. By overlaying the concentration isopleths onto a population density map, the exposed population can be identified. If the extent of contamination is not as clearly defined, such as extent of impact of nonpoint source pollution (impacts from use of agricultural pesticides, e.g.) or the compound is found ubiquitously without a clearly defined source, then the emphasis shifts from geographical bounding to understanding ambient concentrations, exposure pathways and patterns of behavior in general populations.

After identifying the exposed population, the next task is to develop an understanding of the continuum of exposures. The exposures faced by the 10 percent of the population most exposed has been defined as high end exposures. Those faced by the middle of the continuum are called central exposures. Another important estimate of exposure level is a bounding exposure, which is the one faced by the individual(s) most exposed (EPA, 1991a). Arriving at such an understanding can be more of an art than a science. One consideration is the proximity of individuals within an exposed population to the source of contamination. For the incinerator example discussed above, one might begin an analysis by assuming that bounding or high end exposures occur 200 meters east of the incinerator, the location modeled as having the highest ambient air concentrations and particle deposition rates. Another important consideration is the relative contribution of different exposure pathways to an individual's total exposure. While individuals residing

200 meters east of the incinerator might experience the highest inhalation exposures, they may not experience other exposure pathways associated with contaminated soil on their property - such as consumption of home grown vegetables, dermal contact, or soil ingestion. Families with home gardens and individuals who regularly work in those gardens may reside over a kilometer from the incinerator and possibly be more exposed because of their behavior patterns. Screening tools, such as the spreadsheets developed for this assessment, can be used in an iterative mode to evaluate the interplay of such complex factors. When applied to a real world situation, information should be sought as to the makeup and behavior patterns of an exposed population.

The demonstration in this chapter attempts to be consistent with the goal of quantifying central and high end exposures. However, it is not exhaustive in its analysis, nor should be construed as a case study with generalizable results. Following are bullet summaries of key features of the structure and intent of the demonstrations.

- **Exposed populations:** Exposed individuals are assumed to reside in a rural setting. Exposures occur in the home environment, in contrast to the work environment or other environments away from home (parks, etc.). The presumption is made that the sources of contamination of this assessment can occur in rural settings in the United States. Sources demonstrated include soil with concentrations characteristic of background levels, incinerator stack emissions, ash landfills, and areas of soil contamination with concentrations that have been found in industrial sites (see Section 9.5. below). It is further assumed that the behavior patterns associated with the exposure pathways can exist in rural settings. Several of these behaviors characterized as high end relate to individuals on farms as compared to behaviors characterized as central for individuals not on farms. The exposed population for this demonstration, therefore, consists of rural individuals in farming and non-farming residences. For each of the four source categories demonstrated, the exposed populations can be further defined:
  - On-site soils: The on-site source category is demonstrated with soil concentrations that have been found and characterized in the literature as "background" and "rural", or not associated with an

identified source. For this source, the exposed population includes all individuals within a rural area for which the background concentration can be considered representative.

-- Incinerator stack emissions: As indicated earlier, the populations exposed to incinerator stack emissions can be identified by overlaying results of an atmospheric dispersion modeling exercise (or using measured air concentration data) over a map containing population density information. Such an exercise was not done for this demonstration. Instead, simulated ambient air concentrations and deposition rates were taken from tables in Chapter 6 for two locations, one for use in a central scenario and another for a high end scenario.

-- Off-site soil and ash landfills: The two remaining source categories are both finite areas of soil contamination. The off-site soil source category is demonstrated with a 10-acre site having elevated soil concentrations that have been found in the United States and elsewhere in industrial sites. The other is an ash landfill, whose size was estimated using information on the quantity of ash generated by the hypothetical incinerator. The soil concentrations were estimating based on concentrations estimated to be in the ash. For these source categories, a working hypothesis is made that the population most exposed are those residing very near these sites. Their soil is assumed to become contaminated over time due to the process of erosion; these processes normally do not carry contaminants long distances across land, particularly land developed with residences or where erosion is interrupted with ditches or other surface water bodies. People from the surrounding community can be impacted by visiting or trespassing on the contaminated land, volatilized residues may reach their home environments, they may obtain water and fish from impacted water bodies, and so on. It seems reasonable to assume that those residing near these sites comprise the principally exposed individuals, or equivalently, the individuals experiencing the

high end or bounding exposures associated with these areas of soil contamination.

- **Proximity to sources of contamination:** As noted above, the on-site soil contamination source category was demonstrated using soil concentrations typical of background levels that have been found in rural settings. In this case, proximity to the source of contamination was not an issue. Proximity to an incinerator was identified as an important determinant for identifying the continuum of exposures. Assuming there is a uniform distribution of exposure-related behaviors among exposed populations, i.e., their behavior patterns are not a function of where they live in relation to the incinerator, the most exposed individuals will be those exhibiting high end exposure behavior nearest the incinerator. This was the assumption made for purposes of this demonstration. A set of high end exposure behaviors and pathways were demonstrated for individuals residing 500 meters from the incinerator, and a set of central exposure pathways were demonstrated for individuals residing 2000 meters from the incinerator. The highest exposures would occur at 200 meters, using the model output given in Chapter 6. Since the intent is to characterize the upper 10 percent with the high end scenario, and not the bounding estimate, it was felt that 500 meters might be more reasonable. Figures 6.2 and 6.3 show that ground concentrations at 500 meters are 60% of what they are at 200 meters. These figures also show that concentrations at 2000 meters are 40% of what they are at 200 meters, and at 10000 meters are 10% of what they are at 200 meters. Without rigorous justification, the model output (concentrations and deposition rates) at 500 and 2000 meters was felt to appropriately characterize high end and central exposures. The above bullet justifies a definition of principally exposed individuals as those nearest both source categories of off-site soil contamination, while recognizing that lesser exposures can occur for other individuals in the community. These lesser exposures will not be demonstrated. Instead, the off-site and ash landfill source categories will only be demonstrated with a single, high end scenario. Individuals exposed will be assumed to reside 150 meters downgradient

from the sites of soil contamination.

- **Central and high end exposure patterns:** Chapter 7 described the exposure pathways that are considered in this methodology, and justified assignment of key exposure parameters (contact rates and contact fractions, exposure durations, and so on) as central or high end estimates. That chapter notes that the exposure pathways identified were those that were consistent with the sources of contamination, and consistent with literature which identified predominant media where these compounds were found. The bullet above discussing exposed population indicated that several of the behavior assumptions were specific to individuals on farm, and that these behavior patterns were evaluated as high end. High end behaviors assumed to be different for individuals on farms vs central behaviors for individuals not on farms include: residing on larger tracts of land (10 acres assumed for farmers; 1 acre assumed for non-farmers), ingestion of home produced and impacted beef and milk, tendencies to reside in a single location longer (20 years versus 9 years), tendencies to be present in the home environment longer (90% of the time versus 75% of the time), and patterns of soil dermal contact designed to be plausible for farmers working with soil versus those incidentally contacting soil. Other patterns of behavior modeled as central and high end are not specifically associated with farming and not farming, but are assumed to be plausible for individuals in rural settings. These include home gardening for fruit and vegetables, inhalation exposures, children that ingest soil, and the use of impacted surface water bodies for drinking and fish to be ingested.
- **Plausibility of source strength terms:** The objective to determine plausible levels of source strength contamination was an important one for this demonstration. The source terms are soil concentrations, incinerator ash concentrations, and incinerator stack emissions. Section 9.5 describes the source terms in detail.
- **Appropriate estimation of exposure media concentrations:** The realism of estimated exposure media concentrations is dependent on the appropriateness of the models used for such estimations and the assignment

of parameter values for those models. One way to arrive at a judgement as to the realism of estimated concentrations is to compare predictions with observations. To the extent possible (i.e., given the availability of appropriate data), model predictions of exposure media concentrations are compared with occurrence data in Chapter 10 on Uncertainty. As will be shown, predictions fell within the realm of observed data. Chapter 5 describes the justification of all model parameter values. Many of the parameters are specific to the contaminants. Several contaminant properties were estimated as empirical functions of the contaminant's octanol water partition coefficient, Kow, and others were measured values. For non-contaminant parameters such as soil properties, patterns of cattle ingestion of soil, ash disposal practices, and many others, values were selected with the incentive to be mid-range and plausible.

9.3. EXAMPLE EXPOSURE SCENARIOS

As noted above, all exposures occur in a rural setting. Exposure pathways were those which could be associated with places of residence in contrast to the work place or other places of exposure. The example scenarios are structured so that all the behaviors associated with high end exposures are included in the "high end" scenarios and all the central behaviors are in the scenarios characterized as "central". To summarize, the components which distinguished the high end exposure scenarios in contrast to the central scenarios include:

- Individuals in the central scenarios lived in their homes and were exposed to the source of contamination for only 9 years, in contrast to individuals in the high end scenarios, who were exposed for 20 years (except for the exposure pathway of soil ingestion, where the individuals are assumed to be children ages 2-6, and in both the central and high end scenarios, the exposure duration is 5 years).
- Individuals in the central scenarios lived on properties 1 acre in size, whereas individuals in the high end scenarios lived on properties 10 acres in size.
- Individuals in the high end scenario associated with the stack emission source category lived 500 meters from the incinerator, whereas individuals

in central scenario lived 2000 meters from the incinerator.

- High end individuals obtained a portion of their beef and milk using home produced beef and milk - such individuals are obviously farmers. Central, and non-farming rural individuals, were not exposed to contaminated beef or milk.
- Ninety percent of the inhaled air and ingested water by the high end individuals were assumed to be contaminated, whereas only 75% of these exposures were with impacted media for the central individuals. This is based on time at home versus time away from home assumptions for central vs. high end individuals.
- Although their total intake of fruit and vegetables was assumed to be the same, a larger proportion of the intake of those food products in the high end household was home grown and impacted as compared to the central household.
- The rates of ingestion of soil by children and of recreationally caught and impacted fish were higher for the high end individuals than the central individuals.

These are the distinguishing features for the central and high end exposure scenarios. For the sake of convenience mainly, all the scenarios defined below as high end are called "farms", and all central scenarios are called "residences". The assertion is not being made that all behaviors are likely to occur simultaneously (or in some cases, simply to occur) on a farm or a non-farm residence, although several of the high behavior patterns are specific to farms. In an exhaustive site-specific analysis, one might begin by evaluating all possible pathways, further evaluating pathways of most exposure, and then determining what pathways occur simultaneously for identified individuals in the exposed population. Only then can be the assessor begin to define a continuum of exposures.

The following bullets describe six exposure scenarios that are demonstrated. The numbering scheme and titles will be referenced for the remainder of this chapter:

Exposure Scenarios 1 and 2: On-site Soil Contamination, Residence and Farm

Surface soil on a 4,000 m^2 (1-acre roughly) rural residence (Scenario 1) and on a 40,000 m^2 (10-acre roughly) rural farm (Scenario 2) is contaminated with the three

example contaminants. The concentrations of the contaminants are uniformly set at 1 part per trillion, which was evaluated as reasonable background levels (see Section 9.5 below). Bottom sediment in a nearby small river becomes contaminated. Water and fish in that river are subsequently impacted. Fish are recreationally caught and eaten, and the water is extracted for drinking purposes (perhaps at an upstream water system intake, although such configurations do not impact water concentration predictions; concentrations for exposure are those that are calculated to occur above contaminated bottom sediment).

Exposure Scenario 3: Off-site Soil Contamination, Farm

A 40,000 m^2 rural farm is located 150 m (500 ft roughly) from a 40,000 m^2 area of bare soil contamination; an area that might be typical of abandoned industrial property. The surface soil at this property is contaminated with the three example compounds to the same concentration of 1 part per billion. This is evaluated as reasonable for industrial sites of contamination of dioxin-like compounds, and three orders of magnitude higher than concentrations for Scenarios 1 and 2. As in the above and all scenarios, bottom sediment in a nearby river is impacted, which impacts the drinking water supply and fish which are recreationally caught and consumed by members of this farming household.

Exposure Scenarios 4 and 5: Incinerator Stack Emissions, Residence and Farm

A 4,000 m^2 rural residence (Scenario 4) is located 2000 meters from an incinerator, and a 40,000 m^2 (Scenario 5) rural farm is located 500 meters downwind from an incinerator emitting the three contaminants. The emission rates were developed from a hypothetical but realistic incinerator: incinerator capacity, stack pollution control technology, and rates of contaminant emissions were all developed using current information. The modeling of the transport of these contaminants using the Industrial Source Complex (ISC) model was also realistic, using a weather data set from Tampa, Florida. Details on the incinerator and the ISC model application are found in Chapter 6. A nearby impacted river supplies the drinking water and a portion of the fish diet for both families.

Exposure Scenario 6: Incinerator Ash Disposal in a Landfill Near a Farm

Combined (fly + bottom) ash generated by the hypothetical incinerator is disposed of in 100+ hectare landfill, in which a 270,000 m^2 area (27 hectares, or 67 acres roughly) is assumed to be active and bare at any time. The landfill is 150 m upgradient from a 40,000 m^2 rural farm. The concentration of the contaminants on the ash were also hypothetical but realistic. These concentrations are consistent with the incinerator capacity and pollution control technology of the incinerator designed for Scenarios 4 and 5. Ash mixes with landfill soil and the concentrations at the landfill are assumed to be half of what they are in the delivered ash. A nearby impacted river supplies the families with their drinking water and a portion of their fish diet.

9.4. EXAMPLE COMPOUNDS

For purposes of illustration, one compound was arbitrarily selected from each of the major classes of dioxin-like compounds. They are: 2,3,7,8-tetrachlorodibenzo-p-dioxin, 2,3,4,7,8-pentachlorodibenzofuran, and 2,3,3',4,4',5,5'-heptachloro-PCB. For the remainder of this chapter, these compounds will be abbreviated as 2,3,7,8-TCDD, 2,3,4,7,8-PCDF, and 2,3,3',4,4',5,5'-HPCB.

These compounds demonstrate a range of expected results because of the variability of their key fate and transport parameters. The log octanol water partition coefficients (log Kow) for 2,3,7,8-TCDD, 2,3,4,7,8-PCDF, and 2,3,3',4,4',5,5'-HPCB were 6.64, 6.92, and 7.71, respectively. Whereas the span of reported log Kow ranged from less than 6.00 to greater than 8.00, only a few reported values were at these extremes. Increasing log Kow translates to the following trends: tighter sorption to soils and sediments and less releases into air and water, less accumulation in plants and in cattle products (beef, milk), and more accumulation in fish. The Henry's Constants for the three compounds span the range of reported values, with the value of the PCB compound the highest of all reported at $3.0 * 10^{-3}$. There were few values less than the $4.99 * 10^{-6}$ reported for 2,3,4,7,8-PCDF. Higher Henry's Constants translate to greater amounts of volatilization flux. Further information on the chemical properties and fate parameters can be found in Chapters 2 and 5, respectively.

9.5. SOURCE TERMS

This section describes the source terms for the five settings. Source terms include the areas of contamination and soil concentrations. The source terms for the stack emission scenarios, 5 and 6, strictly speaking are the emission rates of contaminants from the incinerator stacks. These are provided in Chapter 6 and not repeated here. This section does summarize the soil concentrations that result from stack emission depositions. It also summarizes the exposure site soil concentrations that result from erosion of contaminated soil from the nearby soil contamination sites (Scenarios 2, 3, 7, and 8).

Key source terms are summarized in Table 9-1. Following now are discussions on these terms for all scenarios.

Scenarios 1 and 2

The residence in Scenario 1 is 4,000 m^2 and in Scenario 2 is 40,000 m^2 in size. Table 3.1 listed literature values for soil concentrations of the dioxin-like compounds. As noted in that chapter, concentrations of the coplanar PCBs were not found in the literature; soil concentrations assigned for 2,3,3',4,4',5,5'-HPCB will be the same as the other two compounds. These scenarios were designed to demonstrate exposures to low concentrations which might be considered "background" soil concentrations. Soil concentrations of 2,3,7,8-TCDD and 2,3,4,7,8-PCDF described as "background" or "rural" by researchers were found in the non-detect to low ng/kg (ppt) in Illinois, Ohio, and Minnesota in the United States (EPA, 1985; Reed, et al., 1990), and in Sweden (Broman, et al. 1990) and England (Creaser, et al., 1989; Stenhouse and Badsha, 1990). Tier 7 of EPA's National Dioxin Study (EPA, 1987) consisted of "background" sites, or sites that did not have previously known sources of 2,3,7,8-TCDD contamination. The purpose of this tier was to provide a basis for comparison for the other 6 tiers of study, which did include sites of known or suspected 2,3,7,8-TCDD contamination. The results were that 17 of 221 urban sites and only 1 of 138 rural sites had detectable levels of 2,3,7,8-TCDD, with a range of positives of 0.2 to 11.2 ng/kg (ppt). While the value of 1 ng/kg selected for these scenarios may not be a true "background" concentration, the intent in designing Scenarios 1 and 2 was to select a concentration that might be typical of areas where no known identifiable source impacts the soil.

Table 9.1. Summary of key source terms for the six exposure scenarios and the three example compounds

I. Soil Concentrations, μg/kg (ppb)	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB
Source Category 1: On-Site Soil			
Scenario 1. Central	0.001	0.001	0.001
Scenario 2. High End	0.001	0.001	0.001
Source Category 2: Off-Site Soil			
Off-site soil concentration	1.000	1.000	1.000
Scenario 3. High End No-till	0.613	0.613	0.613
Tilled	0.031	0.031	0.031
Source Category 3: Stack Emissions			
Scenario 4. Central No-till	0.0008	0.009	0.003
Tilled	0.00004	0.0005	0.0001
Scenario 5. High End No-till	0.0013	0.014	0.005
Tilled	0.00006	0.0007	0.0002
Source Category 4: Ash Landfill			
Landfill soil concentrations	0.700	8.200	2.200
Scenario 6. High End No-till	0.563	6.590	1.770
Tilled	0.028	0.330	0.088

II. Land Areas, m^2			
Source Category 1: On-site Soil		Source Category 3: Stack Emissions	
Scenario 1. Central	4,000	Scenario 4. Central	4,000
Scenario 2. High End	40,000	Scenario 5. High End	40,000
Source Category 2: Off-site Soil		Source Category 4: Ash Landfill	
Off-site soil	40,000	Landfill size	270,000
Scenario 3. High End	40,000	Scenario 6. High End	40,000

Scenario 3

This scenario was designed to be plausible for properties located near inactive industrial sites with contaminated soil. The selection of 1 μg/kg (ppb) for the three compounds was based on 2,3,7,8-TCDD findings associated with the Dow Chemical site in Midland, MI (EPA, 1985; Nestrick, et al. 1986) as well as the 100 industrial sites

evaluated in the National Dioxin Study (which included the Dow Chemical site; EPA, 1987). In that study, most of the sites studied had soil concentrations in the parts per billion range. The farm size was 40,000 m^2, as in all high end scenarios. Table 9-1 shows these concentrations for the example compounds at the site of contamination, and also for the tilled and untilled condition at the sites of exposure. Exposure site soil is assumed to become contaminated over time due to erosion of soil from the contaminated site. The "tilled" condition distributes the eroded contaminants to a depth of 20 cm and impacts the estimated concentrations on home-grown vegetables and fruit. The "untilled" condition distributes the eroded contaminants only to a depth of 1 cm, and results in a soil concentration for which soil exposure pathways, ingestion and dermal contact, are estimated. Note that there are no differences in concentrations at the exposure site among the three example contaminants. It is assumed that eroded soil contains concentrations identical to the concentrations in soil at the landfill site. Therefore, the same amounts of contaminants erode onto and contaminate the soil on the exposure site properties to the same levels.

Scenarios 4 and 5

Chapter 6 described the application of the ISC model to estimate air-borne concentrations and deposition rates of the contaminants at the residence and farm. Wet plus dry particle deposition rates, in units of g/m^2-yr, were determined for all dioxin-like compounds, for two pollution control technologies, and at various distances from the stack. Results from this exercise can be found on Tables 6-18 and 6-19 in Chapter 6. The properties of Scenarios 4 and 5 are located 500 and 2000 meters, respectively, from the incinerator. Assuming 500 meters is the midpoint of the larger property, and that property is square-shaped at 200 meters per side, then it begins roughly 400 meters from the stack and ends roughly 600 meters away. Strictly speaking, the deposition rate for modeling can be estimated as the average of several rates between 400 and 600 meters. For demonstration purposes, the deposition rates at 500 meters as listed in Tables 6-18 and 6-19 were used. Finally, the emission control technology assumed for the hypothetical incinerator in these scenarios was the fabric filter combined with semi-dry alkaline scrubbers (abbreviated DSFF in Chapter 6). Deposition rates for this technology

are given in Table 6-19, and for the three compounds at 500 meters are: $1.32*10^{-9}$ g/m^2-yr (2,3,7,8-TCDD), $1.42*10^{-8}$ g/m^2-yr (2,3,4,7,8-PCDF), and $4.72*10^{-9}$ g/m^2-yr (2,3,3',4,4',5,5'-HPCB). At 2000 meters, the deposition rates are: $8.08*10^{-10}$ g/m^2-yr (2,3,7,8-TCDD), $9.45*10^{-9}$ g/m^2-yr (2,3,4,7,8-PCDF), and $3.15*10^{-9}$ g/m^2-yr (2,3,3',4,4',5,5'-HPCB). The soil concentrations resulting from these depositions are listed in Table 9-1.

Scenario 6

Concentrations on combined (fly + bottom) ash can be estimated using information provided in Chapter 6. Chapter 6 indicates that negligible amounts of PCDDs and PCDFs have been found on bottom ash. This is consistent with data on fly, bottom, and combined ash concentrations of PCDDs, PCDFs, and PCBs summarized in EPA (1991b). For purposes of this demonstration, it will be assumed that bottom ash contains 1/10 the concentrations assumed for fly ash. Chapter 6 also indicates that the mass of bottom ash exceeds that of fly ash by about a factor of ten, and estimates 273 tons of bottom ash generated daily given a solid waste load of 2727 tons daily. Tables 6-9 through 6-12 list ash emission factors for PCDDs, PCDFs, and PCBs, for the two control technologies. These factors are given in units of gms of contaminant in the fly ash per metric ton waste incinerated. Since the hypothetical incinerator handles 2727 metric tons/day, the total daily contaminant in fly ash is estimated as these emission factors times 2727. Also discussed in Chapter 6 is the estimation of 30 metric tons fly ash generated per day with this load of solid waste, and using the fabric filter combined with semi-dry alkaline scrubbers. Tables 6-11 and 6-12 provide the emission factors for this technology, and now concentrations can be demonstrated. From Table 6-11, the emission factor for 2,3,7,8-TCDD is $8*10^{-5}$ g/MT solid waste, and the ash concentration of 2,3,7,8-TCDD is (using conversion factors 1000 kg/MT and 1000 μg/mg):

$$\frac{(8 \times 10^{-5}\ g/MT)\quad(2727\ MT)\quad(10^{6}\ \mu g/g)}{(30\ MT)\quad(1000\ kg/MT)} = 7\ \mu g/kg$$

The concentrations for 2,3,4,7,8-PCDF and 2,3,3',4,4',5,5'-HPCB are 86 μg/kg and 23

μg/kg on fly ash, respectively. Combined ash concentration can then be estimated given 273 MT bottom ash at 1/10 the concentration just estimated:

$$\frac{[(7\ \mu g/kg)\ (30\ MT)] + [(0.7\ \mu g/kg)\ (273\ MT)]}{30\ MT + 273\ MT} = 1.3\ \mu g/kg$$

for 2,3,7,8-TCDD, and 16.3 and 4.3 μg/kg for 2,3,4,7,8-PCDF and 2,3,3',4,4',5,5'-HPCB, respectively. The final assumption made is that the ash mixes with equal amounts of soil at the landfill, so that landfill concentrations are half these concentrations at 0.7, 8.2 and 2.2 μg/kg. The concentrations of contaminants at the farm for the untilled condition are: 0.56 μg/kg for 2,3,7,8-TCDD, 6.59 μg/kg for 2,3,4,7,8-PCDF, and 1.77 μg/kg for 2,3,3',4,4',5,5'-HPCB.

9.6. RESULTS

The results of this exercise include the exposure media concentrations for all exposure pathways and settings, and the LADD exposure estimates. These two categories of results are summarized in Tables 9-2 and 9-3. Following now are several observations from this exercise. **It is important to understand that these observations are not generalizable comments. Different results would arise from different source strength characteristics, proximity considerations, model parameter values, different models altogether, and so on. Chapter 10 on uncertainty describes many areas of this assessment which should be considered when evaluating the methodology or viewing the results.**

9.6.1. Observations Concerning Exposure Media Concentrations

- ***Soil Concentrations***:

The background soil concentrations at the site of exposure assumed for Scenarios 1 and 2, 0.001 μg/kg (1 ppt) were not very different from concentrations estimated at the sites of exposure 500 and 2000 meters from the hypothetical incinerator, 0.0008-0.014 μg/kg (.8-14 ppt) for Scenarios 4 and 5. Higher soil concentrations at the site of exposure resulted from erosion of contaminated soil originating at the 10-acre abandoned industrial site, Scenario 3, and at the 67-acre ash landfill, Scenario 6. Concentrations at the sites of exposure were 0.613 μg/kg (613 ppt) for Scenario 3 (off-site soil with 1 μg/kg soil

Table 9-2. Exposure media concentrations estimated for all scenarios and pathways

Exposure pathway/ scenario	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB
1. Concentration of contaminants in soil for soil ingestion and dermal contact pathways, μg/kg (ppb)			
#1 On-site soil, central	0.001	0.001	0.001
#2 On-site soil, high end	0.001	0.001	0.001
#3 Off-site soil, high end	0.613	0.613	0.613
#4 Stack emissions, central	0.0008	0.009	0.003
#5 Stack emissions, high end	0.0013	0.014	0.005
#6 Ash landfill, high end	0.563	6.590	1.770
2. Concentration of contaminants in air in vapor phase for vapor inhalation pathway, $\mu g/m^3$			
#1 On-site soil, central	$4*10^{-11}$	$2*10^{-11}$	$1*10^{-10}$
#2 On-site soil, high end	$4*10^{-11}$	$2*10^{-11}$	$1*10^{-10}$
#3 Off-site soil, high end	$4*10^{-9}$	$2*10^{-9}$	$2*10^{-8}$
#4 Stack emissions, central	$4*10^{-11}$	$4*10^{-10}$	$1*10^{-10}$
#5 Stack emissions, high end	$6*10^{-11}$	$7*10^{-10}$	$2*10^{-10}$
#6 Ash landfill, high end	$6*10^{-9}$	$3*10^{-8}$	$8*10^{-8}$
3. Concentration of contaminants in air in particulate phase for particulate inhalation pathway, $\mu g/m^3$			
#1 On-site soil, central	$6*10^{-13}$	$6*10^{-13}$	$6*10^{-13}$
#2 On-site soil, high end	$5*10^{-12}$	$5*10^{-12}$	$5*10^{-12}$
#3 Off-site soil, high end	$2*10^{-10}$	$2*10^{-10}$	$2*10^{-10}$
#6 Ash landfill, high end	$6*10^{-10}$	$6*10^{-9}$	$2*10^{-9}$
4. Concentration of contaminants in water for water ingestion pathway, mg/L			
#1 On-site soil, central	$4*10^{-12}$	$3*10^{-12}$	$6*10^{-13}$
#2 On-site soil, high end	$4*10^{-12}$	$3*10^{-12}$	$6*10^{-13}$
#3 Off-site soil, high end	$7*10^{-11}$	$5*10^{-11}$	$1*10^{-11}$
#4 Stack emissions, central	$9*10^{-14}$	$6*10^{-13}$	$4*10^{-14}$
#5 Stack emissions, high end	$9*10^{-14}$	$6*10^{-13}$	$4*10^{-14}$
#6 Ash landfill, high end	$3*10^{-10}$	$2*10^{-9}$	$1*10^{-10}$

(continued on the following page)

Table 9-2. (continued)

Exposure pathway/ scenario	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB
5. Concentration of contaminants in fish for fish ingestion pathway, mg/kg			
#1 On-site soil, central	$7*10^{-8}$	$9*10^{-8}$	$3*10^{-6}$
#2 On-site soil, high end	$7*10^{-8}$	$9*10^{-8}$	$3*10^{-6}$
#3 Off-site soil, high end	$1*10^{-6}$	$2*10^{-6}$	$4*10^{-5}$
#4 Stack emissions, central	$2*10^{-9}$	$2*10^{-8}$	$2*10^{-7}$
#5 Stack emissions, high end	$2*10^{-9}$	$2*10^{-8}$	$2*10^{-7}$
#6 Ash landfill, high end	$6*10^{-4}$	$8*10^{-5}$	$6*10^{-4}$
6. Concentration of contaminants in below ground vegetables (no below ground fruit was assumed) for their respective pathways, mg/kg fresh weight			
#1 On-site soil, central	$1*10^{-9}$	$1*10^{-9}$	$8*10^{-10}$
#2 On-site soil, high end	$1*10^{-9}$	$1*10^{-9}$	$8*10^{-10}$
#3 Off-site soil, high end	$4*10^{-8}$	$4*10^{-8}$	$3*10^{-8}$
#4 Stack emissions, central	$6*10^{-11}$	$6*10^{-10}$	$1*10^{-10}$
#5 Stack emissions, high end	$9*10^{-11}$	$9*10^{-10}$	$2*10^{-10}$
#6 Ash landfill, high end	$4*10^{-8}$	$4*10^{-7}$	$7*10^{-8}$
7. Concentration of contaminants in above ground fruit and vegetables for their respective pathways, mg/kg fresh weight			
#1 On-site soil, central	$4*10^{-11}$	$9*10^{-11}$	$2*10^{-11}$
#2 On-site soil, high end	$4*10^{-11}$	$9*10^{-11}$	$2*10^{-11}$
#3 Off-site soil, high end	$4*10^{-9}$	$9*10^{-9}$	$3*10^{-9}$
#4 Stack emissions, central	$3*10^{-9}$	$4*10^{-8}$	$2*10^{-8}$
#5 Stack emissions, high end	$5*10^{-9}$	$5*10^{-8}$	$2*10^{-8}$
#6 Ash landfill, high end	$6*10^{-9}$	$2*10^{-7}$	$1*10^{-8}$
8. Concentration of contaminants in beef fat for beef ingestion pathway, mg/kg dry weight			
#2 On-site soil, high end	$4*10^{-7}$	$3*10^{-7}$	$3*10^{-8}$
#3 Off-site soil, high end	$2*10^{-4}$	$1*10^{-4}$	$2*10^{-5}$
#5 Stack emissions, high end	$1*10^{-6}$	$1*10^{-5}$	$4*10^{-7}$
#6 Ash landfill, high end	$2*10^{-4}$	$1*10^{-3}$	$5*10^{-5}$

(continued on the following page)

Table 9-2. (continued)

Exposure pathway/ scenario	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB
9. Concentration of contaminants in milk fat for milk ingestion pathway, mg/kg dry weight			
#2 On-site soil, high end	$1*10^{-7}$	$1*10^{-7}$	$9*10^{-9}$
#3 Off-site soil, high end	$5*10^{-5}$	$3*10^{-5}$	$4*10^{-6}$
#5 Stack emissions, high end	$8*10^{-7}$	$8*10^{-6}$	$3*10^{-7}$
#6 Ash landfill, high end	$5*10^{-5}$	$4*10^{-4}$	$1*10^{-5}$

Table 9-3. Lifetime average daily dose, LADD, estimates for all scenarios and exposure pathways (all results in mg/kg-day)

Scenario description exposure pathway	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB
#1 On-site soil contamination Central exposure scenario			
a. Soil ingestion	$8*10^{-13}$	$8*10^{-13}$	$8*10^{-13}$
b. Soil dermal contact	$4*10^{-14}$	$4*10^{-14}$	$4*10^{-14}$
c. Inhalation-vapor	$1*10^{-15}$	$5*10^{-16}$	$3*10^{-15}$
d. Inhalation-particle	$2*10^{-17}$	$2*10^{-17}$	$2*10^{-17}$
e. Water ingestion	$8*10^{-15}$	$5*10^{-15}$	$1*10^{-15}$
f. Fish ingestion	$2*10^{-13}$	$2*10^{-13}$	$6*10^{-12}$
g. Fruit ingestion	$1*10^{-15}$	$3*10^{-15}$	$5*10^{-16}$
h. Vegetable ingestion	$2*10^{-14}$	$2*10^{-14}$	$1*10^{-14}$
#2 On-site soil contamination High end exposure scenario			
a. Soil ingestion	$3*10^{-12}$	$3*10^{-12}$	$3*10^{-12}$
b. Soil dermal contact	$9*10^{-13}$	$9*10^{-13}$	$9*10^{-13}$
c. Inhalation-vapor	$3*10^{-15}$	$1*10^{-15}$	$7*10^{-15}$
d. Inhalation-particle	$3*10^{-16}$	$3*10^{-16}$	$3*10^{-16}$
e. Water ingestion	$2*10^{-14}$	$1*10^{-14}$	$3*10^{-15}$
f. Fish ingestion	$1*10^{-12}$	$2*10^{-12}$	$4*10^{-11}$
g. Fruit ingestion	$4*10^{-15}$	$1*10^{-14}$	$2*10^{-15}$
h. Vegetable ingestion	$7*10^{-14}$	$7*10^{-14}$	$4*10^{-14}$
i. Beef ingestion	$1*10^{-11}$	$1*10^{-11}$	$1*10^{-12}$
j. Milk ingestion	$2*10^{-12}$	$2*10^{-12}$	$2*10^{-13}$
#3 Off-site soil contamination High end exposure scenario			
a. Soil ingestion	$2*10^{-9}$	$2*10^{-9}$	$2*10^{-9}$
b. Soil dermal contact	$3*10^{-11}$	$3*10^{-11}$	$3*10^{-11}$
c. Inhalation-vapor	$3*10^{-13}$	$1*10^{-13}$	$1*10^{-12}$
d. Inhalation-particle	$2*10^{-14}$	$2*10^{-14}$	$2*10^{-14}$
e. Water ingestion	$4*10^{-13}$	$2*10^{-13}$	$5*10^{-14}$
f. Fish ingestion	$2*10^{-11}$	$3*10^{-11}$	$7*10^{-10}$
g. Fruit ingestion	$4*10^{-13}$	$1*10^{-12}$	$3*10^{-13}$
h. Vegetable ingestion	$3*10^{-12}$	$3*10^{-12}$	$2*10^{-12}$

(continued on the following page)

Table 9-3. (continued)

Scenario description exposure pathway	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB
#3 Off-site soil contamination High end exposure scenario (continued)			
i. Beef ingestion	$7*10^{-9}$	$4*10^{-9}$	$7*10^{-10}$
j. Milk ingestion	$8*10^{-10}$	$6*10^{-10}$	$8*10^{-11}$
#4 Stack emissions Central exposure scenario			
a. Soil ingestion	$7*10^{-13}$	$8*10^{-12}$	$3*10^{-12}$
b. Soil dermal contact	$4*10^{-14}$	$4*10^{-13}$	$1*10^{-13}$
c. Inhalation-vapor	$1*10^{-15}$	$1*10^{-14}$	$4*10^{-15}$
d. Water ingestion	$2*10^{-16}$	$1*10^{-15}$	$9*10^{-17}$
e. Fish ingestion	$3*10^{-15}$	$5*10^{-14}$	$4*10^{-13}$
f. Fruit ingestion	$1*10^{-13}$	$1*10^{-12}$	$5*10^{-13}$
g. Vegetable ingestion	$1*10^{-13}$	$1*10^{-12}$	$6*10^{-13}$
#5 Stack emissions High end exposure scenario			
a. Soil ingestion	$4*10^{-12}$	$5*10^{-11}$	$2*10^{-11}$
b. Soil dermal contact	$5*10^{-14}$	$6*10^{-13}$	$2*10^{-13}$
c. Inhalation-vapor	$5*10^{-15}$	$5*10^{-14}$	$2*10^{-14}$
d. Water ingestion	$5*10^{-16}$	$3*10^{-15}$	$2*10^{-16}$
e. Fish ingestion	$3*10^{-14}$	$3*10^{-13}$	$3*10^{-12}$
f. Fruit ingestion	$5*10^{-13}$	$6*10^{-12}$	$2*10^{-12}$
g. Vegetable ingestion	$6*10^{-13}$	$7*10^{-12}$	$2*10^{-12}$
h. Beef ingestion	$5*10^{-11}$	$5*10^{-10}$	$2*10^{-11}$
i. Milk ingestion	$1*10^{-11}$	$1*10^{-10}$	$4*10^{-12}$
#6 Ash landfill High end exposure scenario			
a. Soil ingestion	$2*10^{-9}$	$2*10^{-8}$	$6*10^{-9}$
b. Soil dermal contact	$2*10^{-11}$	$3*10^{-10}$	$8*10^{-11}$
c. Inhalation-vapor	$5*10^{-13}$	$2*10^{-12}$	$6*10^{-12}$
d. Inhalation-particle	$4*10^{-14}$	$5*10^{-13}$	$1*10^{-13}$

(continued on the following page)

Table 9-3. (continued)

Scenario description exposure pathway	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB
#6 Ash landfill High end exposure scenario (continued)			
e. Water ingestion	$2*10^{-12}$	$1*10^{-11}$	$7*10^{-13}$
f. Fish ingestion	$9*10^{-11}$	$1*10^{-9}$	$1*10^{-8}$
g. Fruit ingestion	$6*10^{-13}$	$2*10^{-12}$	$1*10^{-12}$
h. Vegetable ingestion	$3*10^{-12}$	$4*10^{-11}$	$5*10^{-12}$
i. Beef ingestion	$6*10^{-9}$	$5*10^{-8}$	$2*10^{-9}$
j. Milk ingestion	$8*10^{-10}$	$7*10^{-9}$	$2*10^{-10}$

concentrations) and 0.563 to 6.590 μg/kg (563 to 6590 ppt) for Scenario 6 (ash landfill with 0.7-8.1 μg/kg concentrations). Exposure site soil concentrations for Scenario 3 were 61% of what they were at the off-site area of soil contamination, and for the ash landfill, were 80% of what they were at the ash landfill for Scenario 6. The distance from the off-site location of soil contamination was the same in these two scenarios at 150 meters; the ratios were higher for the landfill scenarios because of significantly higher amounts of contaminated eroded soil at the 67-acre landfill compared to the 10-acre site. The soil concentrations were uniform for all three compounds within Scenarios 1, 2, and 3. For Scenarios 1 and 2, soil concentrations were initialized at 0.001 μg/kg. For Scenario 3, exposure site soil concentrations resulting from erosion were also the same for all three compounds because concentrations at the site of contamination were uniformly set at 1.0 μg/kg. The process of erosion and mixing with exposure site soil was not a function of chemical properties. For the other four scenarios, different exposure site soil concentrations resulted because stacks emitted different quantities of compounds (Scenarios 4 and 5), and because concentrations at the ash landfill were different for each compound (Scenario 6). One final note is that the soil concentrations listed for Scenarios 3 through 6 are those that result from the assumption of a 1-cm mixing depth, and are the concentrations used for dermal and soil ingestion exposure estimates. A second soil concentration, the tilled concentrations, is 20 times less than those listed for Scenarios 3-

6, and is used to estimated concentrations of underground vegetables.

- ***Vapor-Phase Air Concentrations***:

One statement to make up front about vapor-phase air concentrations is that using the descriptor "vapor-phase" does not necessarily mean that the contaminants are expected to remain in a pure vapor state while air-borne. As discussed in Chapter 6, the hypothesis is that contaminants sorb to aerosol-size particles, particles < 1 μm. Because the atmospheric residence time is in the order of days for aerosol-size particles, contaminants sorbed to aerosols are expected to remain air-borne for inhalation exposures. Residues which volatilize from the soil are expected to initially be a vapor phase. However, it has also been theorized that dioxin-like compounds released into the air this way would not remain in vapor phase, but would also sorb to air-borne particulates, aerosol size or otherwise. The term vapor-phase is used more to distinguish this reservoir of air-borne contaminants from the one directly linked to the suspension of particulates. Processes which suspend particulates include wind erosion, ash management at the ash landfill, and the release of stack emissions.

Concentrations of contaminants in the vapor phase range from 10^{-11} to 10^{-8} $\mu g/m^3$. Similar and lower concentrations, 10^{-11} to 10^{-10} $\mu g/m^3$, resulted from the volatilization of background soil concentrations of 0.001 μg/kg, Scenarios 1 and 2, and the transport of vapor-phase contaminants originating from the incinerator stack 500 and 2000 meters away, Scenarios 4 and 5. Like the trend for soil concentrations above, volatilization from off-site and more extensive soil contamination, followed by transport 150 meters away to a site of exposure, resulted in higher exposure media concentrations, 10^{-8} to 10^{-9} $\mu g/m^3$ air concentrations for Scenarios 3 and 6. One interesting trend of note is that the vapor-phase concentrations for the central and high end scenarios of Scenarios 1 and 2 are similar for each compound; i.e., the 2,3,7,8-TCDD concentration for Scenario 1 is the same as the 2,3,7,8-TCDD concentration of Scenario 2 (although they are different within a Scenario for different compounds; that will be discussed shortly). This is, in fact, the result of two inverse trends of the solution algorithm. First, the average volatilization flux (mass/area-time) will always be lower for the high end scenario as compared to the central scenario. This is due to the solution algorithm assumption that residues available for volatilization originate from deeper in the soil profile over time, so that the average flux is

lower for longer periods of volatilization. This is seen in the volatilization flux equation - Equation (5-12), Chapter 5 - which has a time term (ED, or exposure duration) in the denominator. The high end scenarios assume 20 years exposure duration compared to 9 years for the central scenarios. This alone would have resulted in lower air concentrations in the high end as compared to the central scenario. However, the dispersion of volatilized residues is a direct function of the area over which volatilization occurs. This is expressed in terms of a side length, parameter "a" in Equation (5-15), Chapter 5, which is in the numerator. This by itself would have resulted in higher air concentrations at the larger farm site of the high end scenario, 10 acres, as compared to the smaller residence of the central scenario, 1 acre. The two trends cancel each other and vapor phase concentrations for a given compound are similar for both scenarios. However, for different compounds within the same scenario, vapor phase concentrations are different. This difference is due to chemical parameters, principally the Henry's Constant, H. 2,3,3',4,4',5,5'-HPCB had the highest value for H, and it was 2 orders of magnitude higher than the value for 2,3,7,8-TCDD and 3 orders of magnitude higher than the value for 2,3,4,7,8-PCDF. This drove the trend for air concentrations, as 2,3,3',4,4',5,5'-HPCB had the highest air concentrations, followed by 2,3,7,8-TCDD at a concentration 1 order of magnitude lower and 2,3,4,7,8-PCDF at slightly lower than 2,3,7,8-TCDD.

- ***Particulate-Phase Air Concentrations***:

In all cases, particulate-phase air-borne concentrations of contaminants were 1 to 3 orders of magnitude lower than vapor-phase concentrations for the same scenario. Concentrations resulting from wind erosion of background soil concentrations, Scenarios 1 and 2, were 2 to 3 orders of magnitude lower than those which originate 150 meters away at the off-site locations of contaminated soil, Scenario 3 and 6. This was due principally to 3-4 orders of magnitude higher soil concentrations at these off-site soil contamination locations. Another trend is that the concentrations are the same for all three compounds within Scenarios 1-3. This is because the algorithm to estimate air-borne concentrations of particulate phase contaminants is independent of chemical properties. The concentrations are different for the different compounds in the ash landfill scenario, 6, because of different initial concentrations in the landfill soil. The trend discussion above concerning vapor phase concentrations resulting from volatilization and

dispersion is not true for particulate phase estimation. In this case, a steady flux is estimated which is not a function of time. The same dispersion algorithm is used, however, so that the high end concentrations in Scenario 2 are higher than the central concentrations in Scenario 1. One final observation is that concentrations for the ash landfill scenario, 6, are 4 to 10 times higher than those from the off-site scenario, 3. Part of this is due to different initial soil concentrations, but part is also due to more sources of particulate emissions at the ash landfill. Sources of particulate emissions at the ash landfill not assumed for the off-site soil contamination include: vehicular resuspension of contaminated roadway dust, emissions off trucks in transit, unloading, and spreading and compacting. An evaluation of the ash landfill flux contributions from the various sources show that the flux due to these ash management activities was twice the flux due to wind erosion alone. (Chapter 6 discussed the use of the ISC model to estimate exposure site concentrations, and it was noted there that air-borne particulate-phase contaminants are not assumed to result from stack emissions.)

- ***Drinking Water and Fish***:

Concentrations of the example contaminants in water were 10^{-14} to 10^{-9} mg/L (ppm). Concentrations in fish ranged from 10^{-9} to 10^{-4} mg/kg. One trend of note is that fish concentrations of the 2,3,3',4,4',5,5'-HPCB are 1-2 orders of magnitude higher than 2,3,7,8-TCDD or 2,3,4,7,8-PCDF. This is because the key variable estimating fish tissue concentrations, the Biota Sediment Accumulation Factor, BSAF, is 2.0 for the example PCB while it is 0.09 for the example dioxin and furan. Another noteworthy trend is that the water and fish concentrations are the same for each compound for the pair of scenarios, #4 and #5, which demonstrate the stack emission source category. The key difference between the central and high end example scenarios is the distance from the stack to the site of exposure - it is 500 m for the high end and 2000 for the central scenario. This impacts all exposure media estimations except the fish and water estimates. Those two are a function of average watershed impact to the stack emissions, not impact to the site of exposure. All other exposure media concentrations for the stack emission source category are a function of specific impact to a site, which will differ as a function of distance from the stack.

- ***Fruit and Vegetable Concentrations***:

Concentrations in these foods ranged from 10^{-11} to 10^{-7} mg/kg (ppm) expressed on a fresh weight basis. Concentrations in below ground vegetables are found to exceed those in above ground vegetables when the source of contamination is soil - the on-site, off-site, and ash landfill scenarios of this assessment. When the source of contamination is stack emissions, above ground concentrations exceed those of below ground. The main cause for this trend was that particle depositions have a large impact on above ground vegetation for the stack emission source category, whereas particle depositions have a small impact on above ground vegetation for the other three soil source categories (this is further explained in Chapter 10, Section 10.2.11.3). As in the air and soil trends discussed above, off-site soil contamination in the range of 1 μg/kg (1 ppb; example Scenarios #3 and #6) results in higher concentrations than on-site background soil concentrations of 0.001 μg/kg (1 ppt; example Scenarios #1 and #2). Another trend noted for Scenarios 1-3, where the initial soil concentrations were the same among the three compounds, is that transfers from soil to plant are driven by chemical parameters, particularly the octanol water partition coefficient, Kow. 2,3,3',4,4',5,5'-HPCB had the highest Kow, with 2,3,4,7,8-PCDF and 2,3,7,8-TCDD at lower but similar Kow. Higher Kow translates to tighter sorption to soil, and less transfer to plant, either through root uptake or air-to-leaf transfer. This trend translated to the lowest fruit/vegetable concentrations for 2,3,3',4,4',5,5'-HPCB. 2,3,7,8-TCDD and 2,3,4,7,8-PCDF had similar fruit/vegetable concentrations for Scenarios 1-3.

- ***Beef and Milk Concentrations***:

These concentrations were comparable to those of fish and ranged from 10^{-9} to 10^{-3} mg/kg. Milk concentrations were lower than beef concentrations in all cases. This was due to assumptions concerning apportioning of total dry matter intake between contaminated soil, contaminated pasture grass, and home-grown contaminated fodder. Beef cattle were assumed to take in 4 times as much soil as dairy cattle, 8% of their dry matter intake versus 2%. Another observation that can be made is similar to the observation concerning the key biotransfer term for fish, the Biota Sediment Accumulation Factor, BSAF. In that case, it was noted that the BSAF for the PCB example compound was much higher than the BSAF for either the dioxin or the furan - hence higher fish

concentrations result. In this case, however, the PCB example compound had the lowest beef/milk biotransfer factor, F. The F for the dioxin and furan example compound was 5.0 and 3.0, respectively, while the PCB F was 0.5. This resulted in uniformly lower PCB beef and milk concentrations.

9.6.2. Observations Concerning LADD Exposure Estimates

Much of the differences between exposure pathways and scenarios is due to differences in exposure media estimation. Therefore, much of the above discussion is also appropriate for trend analysis of Lifetime Average Daily Dose, LADD, estimates. What will be noted below are unique observations:

- ***General***:

LADD estimates ranged from 10^{-17} mg/kg-day to 10^{-8} mg/kg-day. The highest exposures were associated with the ash landfill and the off-site soil contamination scenarios, Scenarios #3 and #6. As discussed above, these scenarios had the highest exposure media concentrations for all exposure media. LADDs for Scenarios 3 and 6 ranged from 10^{-14} to 10^{-8} mg/kg-day. Exposures associated with stack emissions, Scenarios #4 and #5, and on-site soil contamination with low soil concentration, Scenarios #1 and #2, were fairly similar with a range of 10^{-17} to 10^{-10} mg/kg-day.

- ***"Central" vs. "High End"***:

Differences between analogous "central" and "high end" exposures were, in all cases, near or less than an order of magnitude (inhalation exposure for the central on-site scenario and the inhalation exposure for high end on-site scenario are analogous exposures). This is because the exposure parameters used to distinguish typical and high end exposures, the contact rates, contact fractions, and exposure durations, themselves did not differ significantly. In the stack emission scenario, placing exposed individuals either 500 or 2000 meters away from the incinerator did not significantly impact the results. The high end scenarios were modeled after a rural farm and did have exposures from home grown beef and milk food products. The central scenarios were modeled after a non-farming rural residence, and did not have beef and milk exposure pathways. Since beef and milk exposure pathways were noted as the highest exposures, along with

ingestion of fish (see next bullet), it would be appropriate to conclude that farming families ingesting a portion of their home produced beef and milk are more exposed than non-farming families without these exposures.

- ***Exposure pathway analyses***:

It is inappropriate to compare and rank exposure pathways across all scenarios because the source terms are different. However, relationships between different pathways within each scenario can be discussed. Table 9.4 was constructed by summing the LADDs for all pathways, and then determining the percent contribution by each pathway. Before the summation, LADDs were corrected to account for absorption - all ingestion LADDs assumed 50% absorption and inhalation LADDs assumed 75%. The dermal contact LADD was the only one where absorption was already considered in its estimation: absorbed dose was estimated as 3% of dose contacting the body. Also, this exercise assumes all pathways occur simultaneously, and so on. Table 9.4. was generated only for the 2,3,7,8-TCDD example compound, and the rows are listed generally from the highest to lowest percentage contribution. The following observations are made from that table:

- In high end scenarios which assumed exposure to home grown beef and milk - Scenarios 2, 3, 5, 6 - the beef exposures dominated the results.
- Milk exposures were lower than beef exposures because of less milk fat ingestion (10.5 g/day milk fat vs. 22 g/day beef fat) and lower concentrations in milk as compared to beef (this was discussed above). The general dominance of beef and milk exposures underscore the importance of food chain exposures.
- Soil ingestion exposures dominated the results when beef exposures were not considered, and was generally the second highest exposure when beef was considered. Dermal exposures were non-trivial. These demonstrations also show the importance of direct soil exposures
- Fish exposures were critical when exposure site soil concentrations matched concentrations throughout the watershed, which was true for example Scenarios 1 and 2. Proximity to the site of high soil contamination, the off-site example Scenario 2 and the ash landfill Scenario 2, led to high beef, milk, and soil related exposures which overshadowed fish exposures. The particular incinerator modeled for example Scenarios 4

Table 9.4. Percent contribution of the different exposure pathways within each exposure scenario.*

Exposure Pathway	Scenario # 1	2	3	4	5	6
Meat Ingestion	NA	63	70	NA	73	69
Soil Ingestion	75	15	21	71	6	21
Milk Ingestion	NA	9	8	NA	19	8
Fish Ingestion	14	5	0	1	0	1
Soil Dermal	8	7	1	7	0	1
Vegetable Ingestion	2	4	0	11	1	0
Fruit Ingestion	0	0	0	10	1	0
Water Ingestion	1	0	0	0	0	0
Vapor Inhalation	0	0	0	0	0	0
Particle Inhalation	0	0	0	NA	NA	0

* Assumes exposed individual experiences all relevant pathways and exposures are additive; see text for further explanation.

and 5 apparently had a relatively low impact to the water body in comparison to exposures at the site of residence or farming. This led to marginal water and fish exposures for the incinerator scenarios.

- Fruit and vegetable exposures were noteworthy only in the stack emission source category where individuals had home gardens but did not raise cattle.
- Water ingestion, vapor and particle inhalation exposures were very low in comparison to the other exposures in these scenarios.

The LADD estimates of all example scenarios were derived assuming a limited duration of exposure to the dioxin-like compounds, and also limited contact with exposure media. A pattern of childhood soil ingestion was assumed to occur over a five-year period. The central scenarios assumed a nine-year duration of exposure to the contamination, and the high end scenarios assumed a twenty-year exposure period. The contact with impacted media was only assumed to occur in the home environment - only a portion of an individual's meat, milk, water, and fruit and vegetable ingestion was evaluated. This is only one approach to scenario development; other approaches might consider the quality of exposure media not associated with the home environment. For example, if the bulk of an individual's ingestion of produce comes from local farms, and local farms may be

impacted by an incinerator, then perhaps 90-100% of an individual's fruit and vegetable ingestion, rather than the 20-40% assumed in this assessment, should be considered impacted.

Another issue to consider while interpreting these scenarios is their relation to background exposures. As discussed in Chapter 3, dioxin-like compounds are commonly found throughout the environment, leading to "background" exposures even in situations where known sources are not present. Since these scenarios are developed around defined sources, they could be interpreted as incremental exposures beyond "background" levels. This interpretation is less satisfying for Scenarios 1 and 2 where the ambient soil is assumed to be contaminated at 1 ng/kg (ppt) level which, as discussed in Chapter 3 and earlier in this Chapter in Section 9.5, may be close to a background level. In this sense, Scenarios 1 and 2 are more representative of background exposures than incremental exposures. However, diet fractions were applied indicating that only a portion of the food supply was contaminated. Also, the exposure duration was defined as less than a lifetime. For purposes of a true background exposure analysis, it would be more appropriate to assume 100% for diet fractions and lifetime exposures for example Scenarios 1 and 2.

An exercise was undertaken making the following changes in example Scenarios 1 and 2: 70 years exposure duration was assumed for all pathways except the childhood pattern of soil ingestion (which remained at 5 years), and all contact fractions were set equal to 1.00. All contact rates were unchanged for this exercise. The one contact rate which might be changed is the rate of fish ingestion. For that pathway, the 1.2 (central) and 4.1 (high end) g/day rates were estimated assuming a number of meals and meal sizes that a rural individual would recreationally obtain from a nearby impacted water body. For background impacts, it may be more appropriate to have an average ingestion rate, and assume that the 0.63 pg/g fish concentration estimated for example Scenarios #1 and #2 is typical of all fish ingested. That was not done for the results of this exercise that are displayed in Table 9.5, but was done separately and will be discussed in the bullets below. That table includes LADDs and the percent contribution from each pathway for the original and the amended example Scenarios #1 and #2, and only for 2,3,7,8-TCDD. Observations from this exercise are:

- All exposures increased except soil ingestion. Mostly the increase in LADD were within an order of magnitude. The LADDs are still about an order of magnitude

Table 9.5. Exposures to low soil concentrations of 2,3,7,8-TCDD assuming lifetime exposure durations and unlimited contact with impacted media, compared with exposures assuming limited durations and limited contact.

Pathway	Limited Exposures LADD, mg/kg-day	Percent	Lifetime Exposures LADD, mg/kg-day	Percent
Example Scenario #1 On-site soil contamination Concentration = 1.0 ng/kg Central exposure scenario				
Soil ingestion	$8*10^{-13}$	75	$8*10^{-13}$	24
Soil dermal contact	$4*10^{-14}$	8	$3*10^{-13}$	20
Inhalation-vapor	$1*10^{-15}$	0	$4*10^{-15}$	0
Inhalation-particle	$2*10^{-17}$	0	$2*10^{-16}$	0
Water ingestion	$8*10^{-15}$	1	$9*10^{-14}$	2
Fish ingestion	$2*10^{-13}$	14	$1*10^{-12}$	37
Fruit ingestion	$1*10^{-15}$	0	$2*10^{-14}$	0
Vegetable ingestion	$2*10^{-14}$	2	$1*10^{-12}$	17
Example Scenario #2 On-site soil contamination Concentration = 1 ng/kg High end exposure scenario				
Soil ingestion	$3*10^{-12}$	15	$3*10^{-12}$	3
Soil dermal contact	$9*10^{-13}$	7	$3*10^{-12}$	5
Inhalation-vapor	$3*10^{-15}$	0	$7*10^{-15}$	0
Inhalation-particle	$3*10^{-16}$	0	$1*10^{-15}$	0
Water ingestion	$2*10^{-14}$	0	$4*10^{-14}$	0
Fish ingestion	$1*10^{-12}$	6	$4*10^{-11}$	3
Fruit ingestion	$4*10^{-15}$	0	$3*10^{-14}$	0
Vegetable ingestion	$7*10^{-14}$	0	$6*10^{-13}$	0
Beef ingestion	$1*10^{-11}$	63	$1*10^{-10}$	77
Milk ingestion	$2*10^{-12}$	9	$1*10^{-11}$	12

lower than limited exposures estimated to occur from living near an area of high soil contamination, such as ash landfill, as can be seen from comparing these lifetime results with those in Table 9.3. for Example Scenarios #3 (off-site source category) and #6 (ash landfill source category).

- The increases to vapor phase inhalation exposures were small relative to other increases. This occurs because average volatilization flux decreases as a function of time - average fluxes over long periods of time are lower than average fluxes over short periods of time (see Section 5.3.2 for further explanation). This has a rippling effect on fruit and vegetable concentrations as well as beef and milk concentrations. A trend of decreasing volatilization flux is realistic when highly contaminated soils become depleted over time. This was a key principal of the volatilization flux algorithm. However, this is probably not a realistic trend for ubiquitous concentrations which are likely to be replenished in soil over time. Sensitivity analysis in Chapter 10 (Section 10.2.9.4) shows that increasing exposure duration from 20 to 70 years decreases average flux by a factor of 2. By implication, vapor inhalation, fruit and vegetable ingestion, and beef and milk ingestion, are all underestimated for a lifetime of exposure for example Scenario #2, by around the same factor of 2.

- The relative impact of soil ingestion dropped when assuming lifetime exposures. It is interesting that direct soil impacts, a childhood pattern of soil ingestion and a lifetime of soil dermal contact, still are noteworthy impacts, particularly when not considering beef or milk exposures.

- Fish ingestion exposures became the predominant lifetime exposures when beef and milk were not considered. Interestingly, home grown vegetable exposures also gained in significance when considering a lifetime of exposure, being 17% of total exposures when beef and milk were not considered.

- Different ingestion rates of fish were evaluated for example Scenario #2. The rate of 4.1 g/day was increased to 6.5, 30, and 140 g/day. The 6.5 g/day was used in the Ambient Water Quality Criteria document for 2,3,7,8-TCDD, and was described as an average daily per capita consumption of freshwater and estuarine fish and shellfish (EPA, 1984). The 30 and 140 g/day were recommended as 50th and 90th ingestion rates in EPA (1989) for recreational fisherman in an area where there is a large water body present and widespread contamination is evident, and where site-specific information is

unavailable. Increasing to 6.5, 30, and 140 g/day increased 70-year exposure LADDs to $7*10^{-12}$, $3*10^{-11}$, and $1*10^{-10}$, respectively. The percent of total exposure in this test increased from 3% to 5, 20, and 54%, respectively.

An evaluation of total human exposure, and the relative impact of different exposure pathways for 2,3,7,8-TCDD, was also undertaken by Travis and Hattemer-Frey (1991). Their approach was based on modeling, using their Fugacity Food Chain model. This model requires as input emission rates into air, soil, and water; these emission rates are then transformed into concentrations. Those concentrations are used to estimate food crop concentrations, beef, milk, and fish concentrations. In the application of this model by Travis and Hattemer-Frey (1991), emission rates were calibrated to arrive at air, soil, and water concentrations that were supported by the literature. Resulting concentrations in these three media were: 0.02 pg/m^3 in air partitioned as 0.016 pg/m^3 in particulate form and 0.004 pg/m^3 in vapor form, 0.96 pg/g in soil, and 0.003 pg/L in water. Using the model to estimate exposure media concentrations, they then assumed contact rates as given by Yang and Nelson (1986) to estimate human exposure. Assuming 100% absorption, they concluded that typical human exposure is on the order of 35 pg/day 2,3,7,8-TCDD.

Their analysis will be compared to the analysis of this methodology. The example scenario most like their's was example Scenario #2, which estimated exposures to soil levels of 1.0 pg/g. The assumption used in the above sensitivity exercise of 100% contact fractions rather than partial contact is also the appropriate assumption for comparison. A key difference to keep in mind is that air and water impacts are estimated given soil concentrations in this methodology, whereas air and water impacts are specified with their model. This comparison is given in Table 9.6 below.

While the exposure of 9.5 pg/day estimated in Scenario 2 (assuming 100% contact fractions) is lower than the 34.3 pg/day estimated by Travis and Hattemer-Frey, there are a few critical differences in the two approaches. One was in the contact rates. However, replacing the contact rates used by Travis and Hattemer-Frey with the contact rates used in this methodology would not particularly change the total exposure estimate - it would increase only to 9.9 pg/day using the concentrations estimated in this methodology. The second and more important difference is that air concentrations are derived from soil concentrations in this methodology whereas they are input for the Fugacity Food Chain

Table 9.6. Comparison of exposure pathway contributions to total daily exposure as estimated in example Scenario #2 and in Travis and Hattemer-Frey (1991).

Quantity	Travis & Hattemer-Frey (1991)	Scenario #2[1]
Inhalation		
concentration, pg/m^3	0.02	0.000045
contact rate, m^3/day	20	20
intake, pg/day	0.40	0.0009
Water Ingestion		
concentration, pg/L	0.003	0.004
contact rate, L/day	1.4	1.4
intake, pg/day	0.004	0.006
Soil Ingestion		
concentration, pg/g	0.96	1.0
contact rate, g/day	0.02	0.06
intake, pg/day	0.02	0.06
Fruit and Vegetables		
concentration, pg/g	0.06	0.001 & 0.0004[2]
contact rate, g/day	20	192
intake, pg/day	1.2	0.09
Milk		
concentration, pg/g	0.02	.0035
contact rate, g/day	400	300
intake, pg/day	8.0	1.05
Beef		
concentration, pg/g	0.2	.08
contact rate, g/day	90	100
intake, pg/day	18.2	8.0
Fish		
concentration, pg/g	0.38	0.07
contact rate, g/day	17.6	4.1
intake, pg/day	6.7	0.29
TOTAL INTAKE, pg/day	34.3	9.5

[1] Quantities in this column were transformed from what is otherwise displayed in this document to be analogous to the quantities given in Travis and Hattemer-Frey (1991). Specifically, the following was done: 1) all contact rates were displayed as contact rate (20 m^3 inhalation) * contact fraction (0.90; to get 20.7 m^3 contact rate); 2) rate of child soil ingestion for this scenario, 0.8 g/day, was normalized over a lifetime to obtain 0.06 g/day ingestion, and 3) beef and milk ingestion were expressed in total ingestion rather than in fat ingestion; also beef and milk concentrations were expressed in whole product rather than in terms of fat.

[2] below and above ground vegetative concentrations, respectively.

model. Air concentrations are used for inhalation exposures, to estimate fruit and vegetable concentrations, but most importantly, beef and milk concentrations in a food chain model. The air concentrations used by Travis and Hattemer-Frey, in the order of 10^{-2} pg/m^3, are nearly three orders of magnitude higher than are predicted by the volatilization/dispersion modeling used in this methodology. Section 10.2.9.1. (Chapter 10) discusses the fact that air concentrations of 2,3,7,8-TCDD in the 10^{-2} pg/m^3 range were found in urban environments, that urban sources of air contamination are likely to exceed sources in rural environments, and that had the model predicted such concentrations given low soil concentrations perhaps typical of rural environments, the model should be questioned. In the same vein, it seems unlikely that concentrations typical of urban environments would be impacting beef and dairy cattle operations. In any case, air concentrations used by Travis and Hattemer-Frey were input into the food chain models of this assessment to evaluate the impact to predicted beef and milk concentrations. It was found that whole beef concentrations rose three orders of magnitude to 2.9 pg/g to now exceed the beef concentrations estimated by Travis and Hattemer-Frey by an order of magnitude, and that milk concentrations also rose three orders of magnitude to 0.28 pg/g to similarly exceed estimations of Travis and Hattemer-Frey by an order of magnitude. Finally, above ground concentrations of fruit/vegetables rose three orders of magnitude to 0.02 pg/g. Using these air, fruit/vegetable, beef, and milk concentrations, all other concentrations estimated by this methodology, and all the contact rates as given by Travis and Hattemer-Frey raises the daily intake to 375 pg/day. The principal difference between 375 pg/day estimated this way and the 38.4 pg/day estimated by Travis and Hattemer-Frey is how the models of this assessment take air concentrations and translate them to vegetative, beef, and milk concentrations.

This exercise has demonstrated the complex interplay of fate and transport modeling, food chain modeling, assumptions regarding the physical environment, and assumptions regarding exposure behavior. Chapter 10 on Uncertainty critically evaluates the modeling approaches used in this assessment. Information in that Chapter 10 should be reviewed when evaluating the validity of the approaches demonstrated in this Chapter.

REFERENCES FOR CHAPTER 9

Broman, D.; Naf, C.; Rolff, C.; Zebuhr, Y. (1990) Analysis of Polychlorinated Dibenzo-P-Dioxins (PCDD) and Polychlorinated Dibenzofurans (PCDF) in Soil and Digested Sewage Sludge from Stockholm, Sweden. Chemosphere (21): 1213-1220.

Creaser, C.S.; Fernandes, A.R.; Al-Haddad, A; Harrad, S.J.; Homer, R.B; Skett; P.W; Cox, E.A. (1989) Survey of Background Levels of PCDDs and PCDFs in UK soils. Chemosphere 18: 767-776.

Nestrick, T.J.; Lamparski, L.L.; Frawley, N.N.; Hummel, R.A.; Kocher, C.W.; Mahle, N.H.; McCoy, J.W.; Miller, D.L.; Peters, T.L.; Pillepic, J.L.; Smit, W.E.; Tobey, S.W. (1986) Perspectives of a Large Scale Environmental Survey for Chlorinated Dioxins: Overview and Soil Data. Chemosphere 15: 1453-1460.

Reed, L.W.; Hunt, G.T.; Maisel, B.E.; Hoyt, M.; Keefe, D.; Hacknew, P. (1990) Baseline Assessment of PCDDs/PCDFs in the Vicinity of the Elk River, Minnesota Generating Station. Chemosphere 21: 159-171.

Stenhouse, I.A.; Badsha, K.S. (1990) PCB, PCDD, and PCDF Concentrations in Soils from the Kirk Sandall/Edenthorpe/Barnby Dun Area. Chemosphere 21: 563-573.

Travis, C.C., and H.A. Hattemer-Frey. (1991) Human exposure to dioxin. The Science of the Total Environment 104: 97-127.

U.S. Environmental Protection Agency. (1984) Ambient water quality criteria document for 2,3,7,8-tetrachlorodibenzo-p-dioxin. Office of Water Regulations and Standards, Washington, D.C. EPA-440/5-84-007.

U.S. Environmental Protection Agency. (1985) Soil Screening Survey at Four Midwestern Sites. Region V. Environmental Services Division, Eastern District Office, Westlake, Ohio, EPA-905/4-805-005, June 1985.

U.S. Environmental Protection Agency. (1987) National Dioxin Study. Office of Solid Waste and Emergency Response. EPA/530-SW-87-025. August 1987.

U.S. Environmental Protection Agency. (1989) Exposure Factors Handbook. Exposure Assessment Group. Office of Health and Environmental Assessment, Office of Research and Development, Washington, D.C. EPA/600/8-89/043.

U.S. Environmental Protection Agency. (1991a) Guidelines for Exposure Assessment. Draft final submitted to the Science Advisory Board. August 8, 1991. OHEA-E-451.

U.S. Environmental Protection Agency. (1991b) Methodology for Assessing Environmental Releases of and Exposure to Municipal Solid Waste Combustor Residuals. Exposure Assessment Group, Office of Health and Environmental Assessment, Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC; EPA/600/8-91/031. April 1991.

Yang, Y.Y., and C.R. Nelson. (1986) An estimation of daily food usage factors for assessing radionuclide intake the U.S. population. Health Phys., 50: 245-257.

10. UNCERTAINTY

10.1 INTRODUCTION

This chapter addresses uncertainty in dioxin exposure assessment performed with the methodologies presented in this document. Some discussion of the issues commonly lumped into the term "uncertainty" is needed at the outset. The following questions capture the range of issues involved:

(1) How certain are site specific exposure predictions that can be made with the methods?
(2) How variable are the levels of exposure among different members of the exposed local population?
(3) How variable are exposures <u>between</u> different sites?

The emphasis in this document is in providing the technical tools needed to perform site specific exposure and risk assessments. For the assessor focusing on a particular site, question (1) will be of preeminent importance. Therefore the emphasis of this Chapter is to elucidate those uncertainties inherent to the exposure assessment tools presented in this document. For the reader focusing on a particular site, the scenario calculations presented here will be of secondary importance, because site specific data on concentrations of dioxin-like compounds, land use, and resident populations are, at least in theory, available. Specific issues that will be addressed are:

- The validation and expected uncertainties of the pollutant fate and transport models utilized;
- Uncertainties in physical and chemical coefficients used in calculations; and
- Uncertainties in estimated human contact rates with contaminated media and other human behavioral parameters.

Uncertainties in these site specific input data will, of course, need to be examined by the site assessor and may contribute significantly to the overall uncertainty in site risks.

A site specific assessment will also need to address the variability of risks among

different members of the exposed population. The level of detail with which this can be done depends on the assessors knowledge about the actual or likely activities of these residents. In this document, we illustrate one approach to evaluating this variability. Separate "central" and "high end" scenario calculations are presented to reflect different patterns of human activities at a site. The examples presented focus on two areas. "Central" scenarios are constructed to represent typical behavior patterns for residential exposures. "High end" calculations focus on a farming scenario where individuals raise much food for their own consumption. It should be emphasized that high end calculations could also have been developed for residential exposures by making, for example, higher range assumptions about duration of residence or contact rates with the contaminated media. Indeed, this would be recommended for an assessment where considerable emphasis was placed on residential exposures.

With regard to question (3), this document does not present a detailed consideration of how exposure levels will vary between sites. As the scenario calculations in Chapter 9 are intended to be illustrative rather than predictive, the exposure levels that are obtained here are not intended to be typical of actual site exposures for the pathways assessed. However, some readers will not be performing site specific exposure and risk assessments but will refer to this document to obtain a general understanding of pathways of exposure to dioxin-like compounds. To that end, the example scenarios in Chapter 9 were carefully crafted with regard to source strength, chemical and physical parameters, and so on, to be plausible and meaningful. Some readers might ideally wish information on both the magnitude of actual exposures and the variability of these exposures among sites where dioxin-like compounds are present. However, the analysis presented here can not support so broad a goal. Representative data to address the variation of dioxin risks between sites are generally not available and therefore this document can only provide a partial understanding of this variation through use of sensitivity calculations and distributional data on some of the input parameters. Therefore, this analysis will include discussion of the plausibility of site specific assumptions that are made in the example scenarios to the extent that this is possible. Note, however, that the frequency with which the basic exposure setting used in the scenarios actually occur, is not addressed in this document. The structure of the exposure scenario will be taken as basic to the analysis. For example, the frequency farmers raise dairy cattle near an incinerator is not

addressed. A discussion of observed environmental concentrations of 2,3,7,8-TCDD compared with modeled concentrations will provide a perspective on the validity of models used to make such estimations.

10.2. UNCERTAINTIES IN SPECIFIC METHODS APPLIED

Sections 10.2.1. to 10.2.12 discuss the uncertainties associated with specific assumptions, parameters, and methodologies for estimating individual exposures. Sections 10.2.1 through 10.2.4. are on topics pertinent to more than one exposure pathway, whereas sections 10.2.5. through 10.2.12. are specific to exposure pathways. Much of the discussion refers to the example scenarios described in Chapter 9; familiarity with these examples would be helpful. Also, much of the discussion focuses of 2,3,7,8-TCDD, one of the three example compounds in Chapter 9. Unless otherwise noted, sensitivity and uncertainty discussions below are specific to this compound. Sections 10.2.1. through 10.2.12 contain all or some of the following general subsections and information:

- A discussion of the uncertainties associated with the exposure parameters including ingestion, inhalation, and contact rates;
- A subsection comparing estimations of exposure media concentrations with concentrations found in the literature;
- A subsection evaluating alternate modeling approaches to estimating exposure media concentrations;
- A subsection quantifying the impact of variable and uncertain parameters required to estimate exposure media concentrations;
- Tables summarizing key uncertainties.

10.2.1. Environmental Chemistry of Dioxin-Like Compounds

This assessment assumed that levels of dioxin-like compounds in soil and sediment were constant over the period of exposure, with two exceptions. One circumstance was when contaminated soil eroded from one site and deposited on a site of exposure nearby - the off-site and ash landfill source categories. The other was when stack emitted particulates deposited onto a site of exposure - the stack emission source category. In both these instances, it is assumed that only a relatively thin layer of surface soil would be impacted, and that this thin layer is subject to dissipation processes - erosion,

volatilization, possibly degradation. Data in Young (1983) implied a soil half-life of 10 years for surficial 2,3,7,8-TCDD residues, although the circumstances of the soil contamination were not analogous. Specifically, a 37 ha test area at the site had received an estimated 2.6 kg of 2,3,7,8-TCDD over a two year period. Soil sampling which occurred over 9 years from the last application suggested that less than 1 percent remained at the test area. Although Young hypothesized that photodegradation at the time of application was principally responsible for the dissipation of residues, other mechanisms of dissipation including volatilization, erosion, and biological removal may also have contributed to the loss of residues. Soil sampling over time after application implied a dissipation half-life of 10 years for soil residues of 2,3,7,8-TCDD. This value was assumed to apply to the other dioxin-like compounds, introducing further uncertainty.

Section 10.2.3.2 below discusses the impact of this assumed half-life for erosion of contaminated soil. As will be noted, alternate dissipation half-lives did not have a great impact on estimated soil concentrations.

Section 2.5 in Chapter 2 reviewed the literature on degradation of dioxin-like compounds. As discussed, biological transformations as well as chemical processes (oxidation, hydrolysis, and reduction) do not appear to result in substantial degradation of these compounds. There is evidence of photolysis, particularly when dissolved in solution and when organic solvents are present. Most of these data are specific to 2,3,7,8-TCDD, introducing further uncertainty when applied to the other dioxin-like compounds.

Dissipation of surficial residues could translate to lower soil-related exposures including particulate inhalations, soil ingestion, and soil dermal contact. However, it is not clear that reductions in exposure would, in fact, occur, particularly if the soil is contaminated below the surface. Processes such as wind erosion, soil erosion, or volatilization originating from deeper in the soil profile, could serve, in a sense, to replenish reservoirs at the soil surface. Given very low rates of degradation (for all degradation processes except photolysis), the assumption of no degradation is reasonable with moderate, but unquantifiable uncertainty.

In evaluating an assumption of no degradation, another issue to consider is the duration of exposure. If such a duration is assumed to be very long, then degradation or dissipation of soil residues would be more critical than if the duration were relatively short. Uncertainties associated with the duration of exposure are discussed in Section 10.2.2.

below.

Sections below summarize the impact of key chemical-specific parameters to predictions of exposure media concentrations. The following provides a background and discussion of these parameters.

The most critical contaminant parameter required for the procedures in this assessment is the octanol water partition coefficient, Kow, although none of the fate and transport algorithms directly require a Kow. Rather, empirical parameters are a function of Kow. As reported in Chapter 2, log Kow estimates for dioxin-like compounds range from 6.00 to 8.5, with higher log Kow associated with higher chlorination. However, this is not a certain parameter. Estimates in literature for 2,3,7,8-TCDD, for example, range from 6.15 to 8.5.

The most important chemical-specific parameter estimated from Kow is the organic carbon partition coefficient, Koc. Koc describes the steady state partitioning between soil or sediment organic carbon and water; it impacts the volatilization flux from soils, and the partitioning between suspended sediment and water in the water column. Koc is used to estimate in-situ partitioning using a fraction organic carbon in the soil or sediment, OC_{sl}, OC_{sed}, and OC_{ssed}, as Koc*OC_{sl}, etc. The resulting chemical-specific parameter is termed the soil (or sediment) partition coefficient, Kd_s (or Kd_{sed}, Kd_{ssed}). The empirical equation used to estimate Koc from Kow was derived by Karickhoff (1979). This equation was chosen over others available (Lyman, 1982) because it was derived from laboratory testing of 10 hydrophobic contaminants. Others available would have led to lower estimates of Koc. For example, using a relationship developed by Kenaga and Goring (1980) would estimate a 2,3,7,8-TCDD Koc (given log Kow for 2,3,7,8-TCDD of 6.64) of 97,500. The Koc for 2,3,7,8-TCDD estimated for this assessment using Karickhoff's relationship was 2,700,000. Some data implies that this estimate itself may be low for 2,3,7,8-TCDD. Jackson, et al. (1986) obtained soil samples contaminated with 2,3,7,8-TCDD from 8 sites in the Times Beach area of Missouri, and 2 from industrial sites in New Jersey. These contaminated soils had 2,3,7,8-TCDD concentrations ranging from 8 to 26,000 μg/kg (ppb), and organic carbon fractions ranging from 0.015 to 0.08. They determined Kd_s for these soil samples, and using the OC_{sl} data, estimated Kocs for 2,3,7,8-TCDD. The mean Koc from these ten samples was roughly 24,500,000.

The transfer of contaminants from soils to plants requires assignment to two

empirical parameters: the root concentration factor, RCF, and the air-to-leaf biotransfer parameter, B_{vpa}. Both these parameters are empirical functions of Kow. Section 10.2.11. describes the impact to estimated vegetation concentration if these parameters were estimated assuming a different log Kow for 2,3,7,8-TCDD than was assumed for this assessment.

Another critical contaminant parameter is the Henry's Constant. Table 2-2 (in Chapter 2) provides estimates of Henry's Constants, H, for dioxin-like compounds, most of which were estimated given vapor pressure and water solubility data. As seen, the PCDDs and PCDFs were in the 10^{-6} to 10^{-5} atm-m^3/mol range, while coplanar PCBs were in the 10^{-5} to 10^{-4} range, with one high value at $3x10^{-3}$ atm-m^3/mol. Sections below on air and vegetation concentrations describe the impact of this parameter on media concentrations.

Finally, the contaminant molecular diffusivity in air is required for estimates of volatilization flux from soils. The molecular diffusivity in air is set at 0.05 cm^2/sec for all dioxin-like compounds. Molecular diffusivity is a property of both the chemical and the medium. It represents the propensity of a chemical to move through a medium. It is recognized to be largely a function of molecular weight. The values selected are evaluated as reasonable for all dioxin-like compounds, since the molecular weight for these compounds are similar.

10.2.2. Lifetime, Body Weights, and Exposure Durations

As discussed in EPA (1989), values for lifetime of 70 years and adult body weight of 70 kg are derived from large national studies and are not expected to introduce significant uncertainty into exposure estimates. The assumed child body weight of 17 kg (for ages 2-6) is similarly well founded and not expected to introduce much uncertainty into soil ingestion exposure estimates (Section 10.2.5. discusses further issues in soil ingestion exposure estimation).

Assumptions on exposure durations are the most uncertain of the three parameters discussed here. A value of 9 years assumed for central exposure scenarios was an average derived from census survey data (EPA, 1989) which only asked of respondents the amount of time they lived in their current residences. It is likely to therefore be an underestimate as an average amount of time spent in one residence (i.e., respondents are expected to continue to live at their residence). The estimate of 20 years for the average

residence time of farming families (used to define high end exposure scenarios) was not based on data but rather on judgement that farming families live at their farm site longer than non-farming families.

Exposure durations are also tied to assumptions about source strength over time. Assuming 20 years of exposure to incinerator stack emissions, for example, assumes that the incinerator will be (or has been) in operation for this length of time with the same stack emission controls in place. If the source is contaminated soil, assumptions include whether or not the soil will be removed, the site will be capped, and so on. Another consideration is the dissipation of soil residues. Section 10.2.1. discussed uncertainties with the assumption of degradation of dioxin-like compounds in soil. As noted, this assumption is the only one that can appropriately be made. It is reasonable to assume dioxin concentrations in soil remain relatively constant over the durations of 9 and 20 years, given the data which implies little degradation of dioxin-like compounds in the soil environment. For the example scenarios of this assessment, it was assumed that the source strength remains constant over the duration of exposure. In site specific assessments, which are either based on past or projected exposures, more precise statements should address the strength of the contamination source over time.

Exposure estimates are linearly related to all three exposure parameters - increasing body weight and lifetime decreases exposures in an inverse linear fashion, while increasing exposure durations increase estimates in a direct linear fashion.

Uncertainties associated with body weight, lifetime, and exposure durations are summarized in Table 10.1.

10.2.3. Soil Erosion Algorithm

This section addresses uncertainties associated with overland transport of contaminated soil from one site to a nearby site. Soil erosion and deposition was used for the two source categories evaluating the impact of elevated levels of soil contamination at a site away from the site of exposure, the off-site and ash landfill source categories. Transport to an exposure site is modeled using the Universal Soil Loss Equation coupled with a soil delivery ratio. Two soil concentrations are estimated at the site of exposure. One assumes delivered soil mixes to a depth of 20 cm. This depth is associated with tillage operations including vegetable gardening and farming. The other assumes delivered

Table 10.1. Uncertainties associated with the lifetime, body weight, and exposure duration parameters

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Lifetime	70 yrs	Standard EPA assumption and based on data	Actuary data indicate that lifetime may may be increasing	Not a major source of uncertainty
Body Weight	70 kg adult	Standard EPA assumption and based on data	Not much uncertainty	Not a major source of uncertainty
Exposure duration	9 & 20 yrs	Based on assumptions for central and high end exposure scenarios. 9 years based on data for time spent in one residence; rural farming families assumed to live in one location longer than non-farming families in rural settings.	Can vary for populations in rural settings; also important to consider how long exposure has been occurring for retrospective site-specific assessments or how long exposure may occur for prospective assessments.	Assuming non-farming families are more transient than farming families is probably reasonable, although data is unavailable to verify that assumption. Considering exposure durations to be in the range of 10-20 yrs rather than 70 years is felt to be more appropriate.

Overall: Of these three parameters, the exposure duration is the most uncertain. The estimates of 9 and 20 years were made in this assessment for non-farming residents in rural settings, and farming residents in rural settings. These values were based on assumptions of time living at one residence.

soil only mixes to a depth of 1 cm; this is the no-till assumption. The no-till soil concentration is used to estimate dermal exposures and soil ingestion exposures; tilled soils are used only to estimate concentration in underground home-grown vegetables.

This section will first compare results from the example scenarios with data from the literature. Then key assumptions and parameters of the algorithms will be evaluated for sensitivity and certainty. A summary of the uncertainties using this approach is given in Table 10.2.

10.2.3.1. *Comparison with literature data*

A *contaminant concentration ratio* is defined for purposes of this discussion as the ratio of soil concentration at the site of exposure to the soil concentration at the site of contamination. For example Scenario 3, soil eroded from a 10-acre contaminated site was assumed to partially deposit onto a 10-acre exposure site. The contaminant concentration ratio was 0.61 for the 1-cm depth of mixing at the site of exposure and 0.03 for the 20-cm mixing depth. For Scenario 6, a 67-acre ash landfill eroded 150 meters to a 10-acre

Table 10.2. Summary of uncertainties associated with the soil delivery algorithm.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Quantity of erosion from contaminated area	Used average erosion determined from a survey of 70 landfills: 62 T/ac-yr.	Use of landfill data judged reasonable for example scenarios.	Landfill erosion estimates ranged from 0.6 to 306 T/ac-yr; 10th and 90th percentile were 2 and 163 T/ac-yr.	Reducing erosion to 5 T/ac-yr reduced exposure site concentrations 64-77%; increasing to 106 T/ac-yr increased exp. site concentrations 5-11%.
Delivery of soil to exposure site	Used soil delivery equa- and additional correc- when contaminated site size exceeded exposure site size	Soil delivery equation was empirically developed; with 150 m distance, calculated soil delivery ratios were 0.26 and 0.10.	A reasonable range for delivery 150 meters speculated as 0.10 to 0.40.	If this range is reasonable, than exposure site conc. would be reduced by 1/2 at the low end and increased 1/5 at the high end.
Concurrent deposition of clean soil onto exposure site	"Clean" soil is assumed to erode at 6 T/ac-yr, or 1/10 of 62 T/ac-yr erosion off cont. site; difference is due to vegetation on clean soil; no soil delivery ratio applied to clean soil to clean soil.	The mathematical model assumes clean + cont. soil erodes to exp. site; this amount is matched by an equal amount leaving exp. site since soil does not build up on exp. site.	It is unlikely that clean soil would erode less than 1/10 rate of cont. soil already assumed - it might erode more than 1/10; assuming 1/2 erosion rate decreases exp. site concentrations 40-44%.	Assumptions made for example scenarios are plausible; more erosion of "clean" soil could occur, which would reduce exposure site concentrations by less than 50%.
Rate of dissipation of delivered contaminant	A rate corresponding to to a 10-year half-life assumed.	Dioxin-like compounds resist degradation, although evidence suggests photodegradation for residues exposed to sunlight; for deposition of resi- onto otherwise "clean" soil, dissipation will likely occur; 10-year half-life estimated from field data.	Half-life of 4 years decreases estimates of exposure concentrations up to 22%; assuming non-degradation increases concentrations up to 28%	4 years is likely a rapid rate of degradation, but non-degrada- is also unlikely; thus a degradation assumption is not critical to model predictions.
Steady state assumption	Erosion has occurred for a sufficiently long time such that the system has reached steady state	All methodologies of this assessment essentially assume steady state; bio-transfer factors, average volatilization rates, etc. are based on this assumption.	Testing the analytical equation including the time term showed that concentrations after 5 years were 81% of steady state, and after 10 years were 96% of steady state.	A sound assumption if contamination source has existed for more than 5-10 years; could be critical for evaluation of planned landfill and subsequent short-term impacts.

(continued on next page)

Table 10.2. (cont'd)

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Depth of mixing	1-cm depth assumed for non-tilled concentrations, and 20 cm for tilled concentrations; non-tilled used for soil related exposures - ingestion and dermal exposure; tilled used for underground vegetable concentration estimation.	Evidence for fallout plutonium suggests 5-cm for uncultivated situations and 20-cm for cultivated situations. Other analogous exposure assessments have adopted two mixing depths reflecting tilled and untilled conditions.	The 20-cm depth is judged to be reasonable; the 1-cm depth may be low. Increasing it to 2 and 5 cm decreased exposure site concentrations by 18 and 46%.	Of all parameter and assumptions made for the erosion algorithm, this one might be considered the least certain. However, given the tight sorption to soils and the evidence for plutonium, a depth greater than 5 cm would not be warranted. Hence, the 1-cm assumption might be leading to predictions up to two times too high

Evaluation: Whereas there is some uncertainty with each of the six factors associated with estimating exposure site soil concentrations, reasonable variations on assumptions and parameters from what was assumed for example Scenarios 3 and 6 would result in changes in estimated exposure site concentrations within an order of magnitude and mostly within 50% of estimated concentrations (with parameter changes made individually, not in a Monte Carlo mode). Most of the likely variations would tend to reduce rather than increase exposure site concentrations. The amount of soil eroding off the contaminated site, 62 t/ac-yr, might be high, and the erosion of clean soil, estimated at 6 t/ac-yr, might be low. A mass balance analytical equation assumed clean and contaminated soil mix with exposure site soil to be matched by an equal amount of soil eroding off the exposure site; the assumption that clean soil was less erodible than contaminated soil was reasonable. Assuming steady state for situations where the contamination existed over 5-10 years is justified given that 96% of steady state is reached after 10 years. The 10-year half-life rate of dissipation in soil was based on data on 2,3,7,8-TCDD, and the model is reasonably insensitive to changes in that assumption. The model is sensitive to assumed depth of mixing, particularly for the untilled situation. Concentrations for the untilled situation were used to estimate soil related exposures - dermal and soil ingestion. If the 1-cm depth, which was assumed in this assessment, were increased to 5 cm, soil concentrations and related exposures would decrease by 46%. The 1-cm assumption for untilled situation is not based on data, although others have made the same assumption. This parameter might be considered the most uncertain because of this lack of data.

exposure site. The contaminant concentration ratios were 0.80 for the 1-cm depth and 0.04 for the 20-cm depth. The impact of a larger size contaminated site is noted for Scenario 6: going from a 10-acre contaminated site to a 67-acre contaminated site increased the contaminant concentration ratio from 0.61 to 0.80.

Data to rigorously validate the approach taken in this assessment is unavailable. However, there have been documented evidence of migration of 2,3,7,8-TCDD away from industrial sites with soil contamination of 2,3,7,8-TCDD, resulting in off-site soil contamination. Off-site soil concentrations of concern were identified in 7 of 100 Tier 1 and Tier 2 sites of the National Dioxin Study (EPA, 1987). The study noted that in most cases, 2,3,7,8-TCDD had not migrated off-site. Most, but not all, Tier 1 and 2 sites did

have some off-site soil sampling without detection. It should be noted, however, that soil detection limits for most of these 100 Tier 1 and 2 sites were at 1 ppb; this would have precluded finding concentrations less than 1 ppb in some of the off-site soil sampling, particularly important for many of the sites where on-site detections were in the low ppb range. Summary data from the 7 sites noted above is provided in Table 10.3. Contaminant concentration ratios cannot be evaluated by this summary because of lack of detail provided in the National Dioxin Study.

Further detail on the 1984 sampling at the Dow Chemical site in Midland is provided in Nestrick, et al. (1986). An evaluation of the information in that reference is more informative than the Dow Chemical summary in Table 10.3. The entire site is 607 hectares. On-site sampling included areas identified as chlorophenolic production areas, a waste incinerator area, and "background" areas. Background areas were within the 607 ha site but away from production areas. Two of the on-site areas were further identified as areas with Localized Elevated Levels (LELs). These two areas comprise less than 0.5% of the total site area, but had the three highest occurrences of 2,3,7,8-TCDD at 25, 34, and 52 ppb. Including these three high occurrences in the total of 33 samples taken on-site at sites of concern (i.e., not including the background sites) leads to an average concentration of 4.3 ppb; excluding them leads to an average of 1.0 ppb. The average of 11 background samples (including two ND assumed to be 0.0) was 0.15 ppb. A contaminant concentration ratio of 0.035 is calculated assuming an average concentration for contaminated soil of 4.3 ppb (0.15/4.3 = 0.035), and a ratio of 0.15 is calculated if the average soil contamination concentration is more like 1.0 ppb rather than 4.3 ppb.

This ratio of 0.035 is about 1.5 orders of magnitude lower than the 0.61 and 0.80 ratios estimated assuming the shallow 1-cm depth of contamination, although the ratio of 0.15 is within a factor of five of these modeled ratios. The depth of 20 cm leads to modeled ratios of 0.03-0.04, which is more in line with the Dow contaminant ratios of 0.035 or 0.15. The 1-cm depth ratios are probably more pertinent for comparison since it is unlikely that there were tillage operations (or other soil practices which would distribute residues) in background areas of the 607 ha Dow site.

It appears reasonable that the no-till contaminant ratios of 0.61 and 0.80 are higher than the Dow ratios for several reasons. First, the contaminated areas sampled were those likely to be of concern and comprising only a small percentage of the total 607

Table 10.3. Summary of off-site soil contamination from Tier 1 and 2 sites of the National Dioxin Study.

Site name	On-site # samples/range (ppb)	Off-site # samples/range (ppb)	Comments
Diamond Alkali Newark, NJ	9/60-51,000	537/ND-725	Facility involved in the manufacture of 2,4,5-T; off-site sampling covered a 4000-ft radius including public areas such as a public housing unit, park, streets, and river. Two of 11 samples from a park were positive at 1-3.1 ppb; detection limit was 1 ppb. Other off-site positives were from streets and river sediments.
Brady Metals Newark, NJ	10/1.9-3500	30/1.7-1156	Site directly associated with the Diamond Alkali site summarized above; text did not provide any further detail on off-site soil sampling.
Love Canal Niagara, NY	NA/NA-6.7	20/3-263	Love Canal contamination well documented elsewhere; few details provided in reference for soil sampling programs; it was noted that 3,000 cubic yards of fly ash and BHC cake were taken from Love Canal in 1954 and used as fill at the nearby 93rd Street School, a subsurface sample 3+ ft deep showed a concentration of 6.7 ppb. The off-site summary provided here was from an area identified as Hyde Park.
Vertac Jacksonville, AR	45/<1-1,200	320/<1-33.4	A site manufacturing 2,4,5-T; it is not clear than any of the off-site sampling was for surface soil - summary tables identified it as "various"; text description did not mention off-site soil contamination and indicated that solid and liquid waste were buried on-site in a series of landfills. 2,3,7,8-TCDD was found in fish as far away as 100 miles.
Hooker Chemical Niagara, NY	17/ND-18,600	4/ND-1.1	A site manufacturing 2,4,5-TCP; subsurface soil sampling ranged from ND to 18.6 parts per million; one off-site surface soil detection noted at 1.1 ppb.
Bliss Tank Property Rosati, MO	NA/ND-430	NA/ND-430	No summary text provided in primary reference; tabular summary identified soil sampling as on/off-site soil; non-detects were noted in 13 off-site dust sampling.
Dow Chemical Property Midland, MI	#1: 43/.041-52.0 #2: 106/ND-1500	11/.0006-.45 42/.003-2.03	Site most extensively studied of those in National Dioxin Study; Data identified as #1 was a summary of 1984 data supplied in NDS; #2 was a summary of 1985 data; the 1984 data was further detailed in Nestrick, et al. 1986; see text for further discussions on this site.

hectare site. That might question the representativeness of 4.3 ppb as average soil contamination in impacted areas; the three highest concentrations came from specifically identified LELs comprising only 0.5% of the 607 ha site area. Second, a map provided in Nestrick, et al. (1986) including a distance scale clearly shows that all of the background samples were much further than 150 meters from the contaminated sample points, with several sample points hundreds to over a thousand meters from the contaminated sample points. The contaminant concentration ratios, 0.61 and 0.80, were estimated with a distance of 150 meters. Third, the example scenarios had specific assumptions about erosion which may or may not have been appropriate for application to the Dow site.

Ideally validation of this model would involve direct application at the DOW site and comparison of predicted values to measured values. This was not feasible due to lack of information regarding the DOW site. Instead, this analysis has shown that the model predictions of contaminant concentration ratios differ logically from observed ratios at the DOW site. This comparison raises confidence in the model but cannot be considered a validation exercise.

10.2.3.2. *Key Parameters and Assumptions of the Erosion Algorithm*

The algorithm to estimate exposure site concentration is comprised of six key features:

- the total amount of erosion off the contaminated site;
- a soil delivery ratio;
- the concurrent erosion of "clean" soil - soil between the contaminated and exposure site - which mixes with the eroded contaminated soil upon delivery to the exposure site;
- the rate of dissipation of contaminants which have reached the exposure site;
- the steady state simplification;
- the depth to which contaminants are assumed to mix in at the site of exposure.

Each of these factors will be examined separately.

1. Erosion from the contaminated site

The quantity of material eroded from the contaminated site is estimated using the Universal Soil Loss Equation (USLE). The USLE is a widely used empirical model for

agricultural applications. It estimates the edge-of-field soil loss and has been extensively used to also estimate loss of contaminants bound to soil. The accuracy and precision of USLE estimates are not reviewed for this assessment. However, the USLE was developed using literally hundreds of plot-years of erosion (Wischmieir and Smith, 1965), so that estimates of edge-of-field erosion can be considered reasonable.

USLE parameters used for the example scenarios were derived from a survey of 70 landfills conducted for EPA (SAIC, 1986). The survey, which was based on interviews with site managers and local officials and review of USGS topographic maps, ascertained estimates of the parameters related to rainfall, soil properties, topography, and site management practices. For the 70 landfills, estimated erosion rates ranged from 0.6 to 306 T/ac-yr, with a mean calculated value of 62 T/ac-yr. The 10th to 90th percentile range was 2 to 163 T/ac-yr. The soil loss estimate for the example scenarios, Scenarios 3 (off-site bare soil contamination with soil concentrations based on concentrations that have been found in sites of industrial contamination) and Scenario 6 (active ash landfill with soil concentrations based on estimated ash concentrations), were 62 T/ac-yr. It is noteworthy that sites with low erosion potential, principally because of vegetative cover, may have two orders of magnitude less erosion than 62 T/ac-yr. An assumption of dense vegetative cover would reduce the estimate of 62 T/ac-yr to 6 T/ac-yr.

Sensitivity runs were done for the two scenarios involving off-site soil contamination, Scenarios 3 and 6. Reducing the erosion by a factor of ten to 6 T/ac-yr reduces exposure soil concentrations by 64% for Scenario 6 (ash landfill) and 77% for Scenario 3 (off-site soil contamination). Interestingly, doubling the amount of erosion leaving the contaminated site to 124 T/ac-yr has a marginal impact on exposure site concentrations: they would be increased only by 5% (ash landfill) and 11% (off-site soil contamination). The estimate of 62 T/ac-yr is generally high - it assumes bare soil conditions, but an alternate estimate of 6 T/ac-yr is low for contaminated sites such as industrial sites of landfill. This low erosion estimate is reasonable for land with dense vegetation such as pasture or undeveloped land. In summary, the high estimate of erosion used in the example scenarios may lead to an overestimate of soil concentrations at a nearby site of exposure - perhaps by a factor of 2.

2. Soil Delivery Ratio

The soil delivery ratio is defined as the ratio of soil delivered downgradient at a specified distance and the edge-of-field erosion losses. The algorithm in this assessment uses an empirical equation which is a function of length (Equation (5-5)), corrected by consideration of size discrepancies between contaminated and exposure site. Both example scenarios involving off-site soil contamination, Scenarios 3 and 6, had exposure sites located 150 meters from the site of contamination. The calculated soil delivery ratios for these scenarios were 0.26 (Scenario 3) and 0.10 (Scenario 6).

The soil delivery ratio was varied to evaluate its impact on exposure site soil concentrations. The soil delivery ratio of 0.26 for Scenario 3 was reduced ten-fold to 0.026, five-fold to 0.05, and to 0.10. Exposure site soil concentrations were 77%, 61%, and 38% less with these three reductions as compared to predictions at the 0.26 soil delivery ratio. Increasing the delivery ratio to 0.40 and 0.50 increased exposure site soil concentrations by 16 and 24%. Similar exercises were done for example Scenario 6, the ash landfill. The delivery ratio was reduced ten-fold to 0.01, two-fold to 0.05, and doubled to 0.20. Exposure site concentrations were reduced 63% and 16%, and increased 11% with these changes.

Like the USLE, the soil delivery equation was developed using field data. However, it was developed for erosion from mining sites to nearby surface water bodies. Its use here assumes that delivery of erosion from mining sites is similar enough to delivery of erosion from contaminated soil sites. Given this caveat, estimates from the soil delivery equation might be reasonable enough to rule out sensitivity estimates at 10-fold reductions or twice as much delivery, the extremes tested above. If so, one might conclude that the uncertainty based on this factor is in the range of 1/2 less than estimated concentrations to 1/5 more.

3. Concurrent Deposition of Clean Soil

Currently, the assumptions for example scenarios #3 and #6 is that the contaminated site is bare, and that the land area between the contaminated site and the exposure site has a dense vegetative cover. This translates to a reduction of "C" factor, or cover, from 1.0 for the contaminated sites to 0.1 for the areas between the contaminated sites and the exposure site, and a resulting 1/10th unit edge-of-field soil

reduction for this "clean" soil area compared to the contaminated area. All other USLE parameters are assumed to be the same for both sites for the example scenarios in Chapter 9. This led to a unit erosion of clean soil of 6 T/ac-yr. Like erosion from a contaminated site, this erosion estimate is subject to variation in site-specific assessments. The bare soil assumption applied to the contaminated site might be appropriate for an active landfill, might be an overestimate for sites of industrial contamination (there would likely be at least sparse vegetation), but is unreasonable for land separating a contaminated site and the nearest exposure site.

Another difference between erosion of soil from the contaminated site and from the clean area is that there is no soil delivery ratio applied to erosion of the clean soil. The USLE estimates edge-of-field losses, and the clean area is configured to be adjacent to the exposure site.

Intuitively, the larger the amount of clean soil which mixes with the contaminated soil, the lower will be the estimated exposure site concentrations. This hypothesis was tested with increases to the soil loss estimate applied to the clean soil area. When it was increased to 62 T/ac-yr, the same unit erosion amount as from the contaminated site, amounts of depositing clean soil increased by ten and exposure site soil concentrations were decreased between 48 and 59% for the two scenarios. When the unit soil loss was increased to 31 T/ac-yr, exposure site concentrations decreased slightly less at 40-44%.

The possibility that the land area between the contaminated site and the exposure site is more erodible than the contaminated site is unlikely, principally because of the likely existence of vegetation. It can be concluded that if the land area between the contaminated site and exposure site is more erodible than assumed (but still less erodible than the contaminated site), than exposure site concentrations were overpredicted by no more than 50%.

4. The rate of contaminant dissipation

The rate of dissipation of transported residues was assumed to be .0693 yr^{-1}, which corresponds to a half-life of 10 years. This parameter was changed to 0.1735 yr^{-1} (half-life = 4 yrs), 0.0198 yr^{-1} (35 yrs), and 0.0 yr^{-1} (no degradation). Results indicate only a modest change in exposure site soil concentrations for the example Scenarios #3 and #6. Decreasing the half-life to 4 yrs reduced exposure site concentrations by 12 and

22% for Scenarios 3 and 6. Increasing the half-life to 35 years increased concentrations between 7 and 16%, and assuming no degradation increased concentrations between 11 and 28%.

Indications are that photodegradation may be a relevant dissipation process for dioxin-like compounds. Volatilization is also a process which dissipates residues. However, these compounds are known to resist degradation. The assumption that in areas of elevated soil contamination (industrial sites, landfills, etc.), contamination extends below the surface and residues are assumed not to degrade is thought to be a reasonable one. The assumption that contaminants transported to otherwise residue-free soil to impact a thin surface layer are subject to dissipation processes is also a reasonable one. The assumed half-life of 10 years has been estimated from field data, and variations are shown to not greatly impact exposure site soil concentrations. The sensitivity exercise showed that concentrations might be over or underpredicted by no more than 30%.

5. The steady-state assumption

Section 5.4.1, Chapter 5, described the derivation of the algorithm estimating exposure site concentrations given contaminated site concentrations. The analytical equation, Equation (5-27), had an exponential term including a time variable. Assuming long periods of time results in this exponential term approaching 1.0. The term was, therefore, dropped, to arrive at the steady state version of Equation (5-27), which was Equation (5-28).

Tests were run including the exponential term with t equal to 1, 2, 3, 4, 5, 10, and 15 yrs. Expressing results in terms of the percent of steady state concentration reached at the end of each year, the results for these time intervals are 28, 49, 63, 74, 81, 96, and 99%, respectively. As seen, 81% of steady state is reached after 5 years, and 96% is reached after 10 years. Therefore, if off-site contamination has existed for 5-10 years or more, estimates of exposure site concentration (and related exposures) with a steady state assumption should be reasonable. However, if an assessment is to be done for a site to be newly impacted, such as a planned landfill, than the steady state approach would lead to some overprediction of concentrations and exposures. For the first five years, concentrations would average about 60% of steady state, and for the first 10 years, concentrations would average about 75% of steady state. This would be of most

concern for a childhood pattern of soil ingestion, which would be 40% lower for the first five years of a landfill operation as compared to a steady state assumption. Otherwise, it is seen that the steady state assumption does not greatly impact exposure site concentrations.

6. Depth of mixing

Section 5.4.1. describes the justification for the 1-cm and 20-cm depths of mixing for situations where contaminants from a distant source are transported to a soil surface in off-site locations. Non-tilled depths of 2, 5, and 10 cm were tested. Results indicate that exposure site concentrations were reduced by 18%, 46%, and 66% for these three depths. If a tillage depth of 15-20 cm is judged reasonable, then a 10 cm depth for non-tilled situations should probably be judged too high. If so, this exercise indicates that non-tilled exposure site soil concentrations could be overestimated by less than 50%.

10.2.4. Surface Water Suspended and Bottom Sediments

Concentrations of contaminants in surface water sediments are assumed to be correlated to concentrations in surface soils in the watershed draining into the surface water body. This section will first compare results from the example scenarios with data from the literature. An alternate approach to estimating sediment concentrations from soil concentrations will be evaluated. Then key assumptions and parameters of the algorithms will be evaluated for sensitivity and certainty. Table 10.4 summarizes the uncertainties with this algorithm.

10.2.4.1. *Comparison With Literature Data*

Assuming sediment concentrations are a function of surface soil concentrations is a reasonable approach when the only source of water body contamination is soil contamination. However, the algorithm would not be appropriate if sediments and water were impacted by industrial discharges, which has often been cited as the cause for sediment and water impacts (see Bopp, et al., 1991; Norwood, et al., 1989; e.g.). Sediment concentrations of note have also been found in surface water bodies near urban settings, with car and industrial stack emissions cited as likely causes (Gotz and Schumacher, 1990; Rappe and Kjeller, 1987). Rappe, et al. (1989) collected samples from

Table 10.4. Summary of uncertainties associated with the surface water sediment algorithms.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Site and watershed soil concentrations	Site concentration is site-specific factor; watershed concentration assumed to be 0.0 except for "background" example scenario, where it was 1 ppt	Assuming no residues in watershed is the most appropriate assumption unless a background concentration can be established.	Assuming a background concentration of 1 ppt which is 3 orders of magnitude less than the site contamination of 1 ppb resulted in only a 5% increase in sediment concentrations.	No change in assumption is currently warranted.
Site and watershed size and delivery ratios	Sizes are site-specific and known with certainty; delivery ratios empiri- estimated given distance to water body for site, and watershed size	Approach appropriately considers sizes and distances, and uses developed empirical relationships to estimate delivery ratios	Delivery ratio for site developed from surface mining data and for distances up to 250 m, use for sites further from water body would likely overestimate delivery of contaminated soil	Changes in sizes and distances show direct linear impact to sediment concentrations; watershed size appropriately selected as median value from survey of landfills.
Quantity of erosion from contaminated area and watershed	Like soil erosion algorithm, assumed 62 T/ac-yr for contaminated sites; 6 T/ac-yr assumed for watershed.	Assuming contaminated site is more erodible than average for water- is not inappropriate, although 10 times more erodible might be high	Site-specific estimates would lead to erosion rates that are known with reasonable certainty.	Sediment concentrations are linearly related to estimates of unit soil loss from both the site and the watershed.
Stack emission algorithm	ISC modeled watershed representative deposition rates mixes in represen- tative mixing zone depth to estimate soil concen- tration for watershed; sediment concentration equals this concentration	Approach for stack emissions is consis- with other approaches which assume sediment in water body is a function of soil concen- trations in watershed	Does not consider direct depositions to water body which could be critical if water body is very large or if incinera- is near where water is withdrawn; picking an ISC modeled deposition rate is a site-specific exercise; depth of mixing in watershed is uncertain	Example scenarios picked ISC deposition rate at 5000 meters; exposure sites were at 500 and 2000 m; representative mixing depth for watershed assumed to be 10 cm.

Overall: Data was unavailable on the impact of surface water sediments from erosion of soil contaminated with dioxin-like compounds. Most, if not all, of the literature on sediment impacts attribute causes to direct industrial discharges or, where the source is not a point source, to an "incineration pattern", which links concentrations found in sediments to concentrations found in air and air particulates where sources are speculated to be incinerator, car, or other industrial air emissions. The algorithm estimates sediment concentrations as a linear function of soil concentrations within the watershed. Sensitivity analysis showed mostly a linear impact to sediment predictions with all parameters - some of the impacts are direct and some are inverse. An alternate modeling approach was compared with the approach in this assessment. Both approaches predicted similar soil/sediment concentration ratios (ratio of concentration on contaminated land area within watershed to concentration on bottom sediments). Changes in predicted sediment concentrations from changes in any one parameter would result in concentrations within an order of magnitude of currently modeled concentrations.

the Baltic Sea, which were described as background samples. They note that the pattern of tetra-CDF congener concentrations found in the Baltic Sea were typical of what they termed the "incineration patterns" - air and air particulate concentrations that were attributed to sources such as incineration, car exhausts, steel mills, etc. On the other hand, sediment samples collected between 4 and 30 km downstream from a pulp mill revealed a congener pattern typical of bleaching mills. Different approaches to estimating sediment concentrations as a function of direct industrial discharges or direct air-borne depositions would be required for these sources. The effluent discharge issue is further discussed in Appendix D.

Data was not found linking sediment concentrations to soil concentrations, in an urban or more pertinent to this assessment, a rural setting. Some sampling has occurred in areas described as rural or background. Sediment sampling in Lake Orono in Central Minnesota in such a setting found no tetra- and penta-CDDs, although occurrences of total hexa-CDDs were found in the low ng/kg (ppt) level, occurrences of hepta-CDDs to a high of 110 ppt, and total OCDD concentrations ranged from 490-600 ppt for three samples (Reed, et al., 1990). A report on sampling of several estuaries in Eastern United States included a "reference" or relatively clean site, central Long Island Sound. There were no occurrences of 2,3,7,8-TCDD, although 2,3,7,8-TCDF was found at 15 ppt in this clean site. Other sites had identified industrial source inputs and higher noted concentrations (Norwood, et al., 1990). 2,3,7,8-TCDD was extensively found in sediments of Lake Ontario (EPA, 1990a). The average of samples from all depths of sediment collection from 49 stations including 55 samples was 68 ppt. The average of 30 surficial sediment samples was 110 ppt. A modeling exercise implied that an annual load of 2.1 kg/year into Lake Ontario corresponds to a concentration of 110 ppt. One identified source was the Hyde Park Landfill, located about 2000 feet from the Niagara River, which drains into Lake Ontario. Between 1954 and 1975, an estimated 0.7 to 1.6 tons of 2,3,7,8-TCDD were deposited in the landfill. A principal conclusion from the modeling exercise, however, was that a characterization of historical loadings of 2,3,7,8-TCDD into the lake was not available and would be necessary to evaluate the contributions by the Hyde Park Landfill.

The sediment concentrations of 2,3,7,8-TCDD predicted to occur in the river for the six example Scenarios in Chapter 9 were 0.005 ppt (Scenarios 4 and 5 - stack emission source category), 0.2 ppt (Scenarios 1 and 2 - on-site source category), 4 ppt (Scenario

3 - off-site source category), and 17 ppt (Scenario 6 - ash landfill). The highest concentrations in sediment were predicted to occur from sites of elevated concentrations located 150 meters from the river. 2,3,7,8-TCDD was set at 1 μg/kg (ppb) at a 4 ha site for Example Scenario 3 demonstrating the off-site source category, and was set at 0.7 ppb for the 27 ha site of Example Scenario 6, demonstrating the ash landfill scenario. These elevated concentrations for the small land areas were three orders of magnitude higher than soil concentrations estimated or assumed in the other four Scenarios. One ratio of note for Scenarios 3 and 6 is the ratio of soil concentration at the site of contamination to soil concentration in the sediments. These contaminant ratios equal 0.004 for Scenario 3 and 0.024 for Scenario 6. No literature was found which listed contaminated soil and resulting surface water body sediment concentrations, which would be necessary to evaluate these ratios. The next section, however, does attempt an alternate modeling approach to arrive at analogous ratios.

10.2.4.2. *An Alternate Approach*

As noted, the validity of these results cannot be rigorously evaluated because of a lack of data on impact to surface water bodies by soil erosion of dioxin-like compounds. However, the dilution of contaminated sediments entering a river system can be estimated using an alternate approach.

The average runoff rate for the midwestern U.S. is about 15 inches/year (Linsley, et al., 1982). For a 10,000-acre watershed (4,000 hectares; the watershed size and effective drainage area for Scenarios 3 and 6), this yields a stream flow of about 17.2 ft^3/sec. The sediment yield can be estimated from the stream flow as follows (Linsley, et al., 1982): $Q_s = aQ^n$, where Q_s = sediment flow rate (Eng. T/yr); Q = stream flow rate (ft^3/sec); and a and n are empirical constants, reflecting the vegetative cover in the watershed. Linsley, et al., (1982) recommend using a = 3,500 and n = 0.82 for coniferous forest and tall grass, and a = 19,000 and n = 0.65 for scrub and short grass. Substituting these into the equation above (and Q = 17.2 ft^3/sec) gives an annual sediment flow rate of 36,000 to 121,000 T/yr. Total sediment flows will be comprised of contaminated as well as uncontaminated sediments. A "contaminant concentration ratio" can be calculated by estimating the sediment contributed by the contaminated areas and dividing by this sediment flow rate range; this assumes all other sediment contributions are

uncontaminated. The annual soil loss for Scenarios 3 and 6 was 62 T/ac-yr. The contaminated site area for Scenario 3 was 10 acres (4 ha) and the area for Scenario 6 was 67 ac (27 ha). The total soil erosion contributed by these sites equals: the unit soil loss * area * soil delivery ratio; for Scenario 3, this equals 62*10*0.26, or 161 T/yr, and the total amount contributed by Scenario 6 was 62*67*0.26, or 1080 T/yr. The contaminant concentration ratio is (161 T/yr)/(36,000 to 121,000 T/yr), or a range of 0.001-0.004. This can be compared to the contaminant ratio estimated in the previous section for Scenario 3 at 0.004. For Scenario 6, the range is 0.009-0.03, which compares to 0.024 estimated from above.

This implies that both approaches estimate similar contaminant ratios, although they are both subject to key assumptions. A lower runoff rate (instead of 15 inches/year) would have resulted in lower total sediment yields using the approach given in Linsely, et al. (1982). Also, several assumptions were made to estimate sediment concentrations in the approach of this assessment. The impacts of these assumptions are discussed below. Of particular note is the difference in erosion rates assumed for the contaminated site in comparison to the watershed. Contaminated site erosion was high at 62 T/ac-yr, and watershed rate was low at 6 T/ac-yr. If both rates were assumed to be equal at 31 T/ac-yr, estimated contaminant ratios would drop by a factor of ten: the contaminant concentration ratio for example Scenario 3 would be 0.0004 instead of 0.004. Recalculating the ratio range using the Linsley method above would yield 0.0005-0.002. Now, 0.0004 is slightly lower than the range of 0.0005-0.002. The contaminant concentration ratio for Scenario 6 would be 0.0024, also lower than the analogous range of 0.0045-0.015. In short, both methods are based on assumptions about erodibility and both appear to be consistent with each other.

10.2.4.3. *Sensitivity to Key Parameters and Assumptions*

The parameters estimating sediment concentrations are of three types: site-specific, empirical, and modeled parameters. Site specific parameters include: concentrations at the site of contamination and average for the watershed, C_s and C_w, distance from contaminated site to water body, DIS, and areas of contamination, A_s, and of the watershed, A_w. The empirical parameters include the sediment delivery ratios assigned to the contaminated site and for the watershed, SD_s and SD_w, and the depth of mixing

assumed for incinerator stack depositions in order to estimate watershed average soil impact. The modeled input were the incinerator stack deposition rates assumed to be average for the watershed and the unit soil loss assumed for the contaminated site and the watershed, SL_s and SL_w.

The impact to estimated sediment concentrations in the river to reasonable changes in most of these parameters is minimal. Sensitivity tests described in this section were conducted using example Scenario 3 from Chapter 9, which demonstrated the off-site soil source category. For that demonstration, a 4 ha area was initialized to 1 μg/kg (ppb) within a 4,000 hectare watershed. Background watershed concentrations were set at 0.0 for Scenarios 3.

1. Site and Watershed Contaminant Concentrations

Assuming that only the contaminated site has residues and that all other watershed soils are residue free may be inappropriate and would underestimate surface water impacts. The evaluation of the Dow Site in Midland, MI (Nestrick, et al., 1986; described above in Section 10.2.2.1) showed part per trillion levels of soil concentrations within the city of Midland, which was three orders of magnitude less than concentrations at the site of contamination. When the watershed concentrations in Example Scenario 3 were set at 1 ppt, three orders of magnitude lower than the site concentrations of 1 ppb, river sediment concentrations increased by less than 5%.

Sediment concentrations are roughly linearly related to soil concentrations at the site of contamination. Increasing the concentration from 1 μg/kg (ppb) to 10 ppb increased sediment concentrations by a factor of 9.5; decreasing it to 0.1 ppb has roughly the same order of magnitude reduction.

2. Site Size and Delivery Ratio and Watershed Size and Concurrent Sediment Delivery Ratio

Sediment delivery ratios had an impact on sediment concentrations. When the contaminated site was 50 meters instead of 150 meters from the rivers edge, concentrations increased by roughly 27%. When the sites are set at 500 meters from the rivers edge, the concentrations decreased by 23%. These distances are used to estimate the sediment delivery ratio from a site to the edge of the water body. The range of ratios

estimated with these distances (see Equation (5-5), was 0.20 (at 500 meters) to 0.33 (at 50 meters), which is a rather narrow range. It was noted in Section 5.3.1 that the sediment delivery ratio equation was developed from construction sites and from data up to 250 meters from a water body. The delivery ratio estimated with this equation at 500 meters of 0.20 might be high. The delivery ratio was reduced to 0.05, a factor of 5 reduction from the estimated ratio of 0.26 at 150 meters, and sediment concentrations were also reduced by 5. If the contaminated site drained directly into the river - the delivery ratio equals 1.0, and concentrations would increase by 4.0. In other words, sediment concentrations are linearly related to the sediment delivery ratio applied to the contaminated site. It also seems reasonable to assume, unless the site were adjacent to a water body, that a range of reasonable sediment ratios are in the 0.10 to 0.40 range; the value of 0.26 estimated at 150 meters would appear to be a reasonable midrange value.

The size of the contaminated site was found to be roughly linearly related to sediment concentrations. Increasing the contaminated site from its initial size of 4 ha to 40 ha increased concentrations by a factor of 9; decreasing the size to 0.4 ha decreased concentrations by an order of magnitude.

Sediment delivery ratios applied to a watershed are narrowly defined. As seen in Figure 5.1, the watershed sediment delivery ratio, SD_w, is approximately 0.12 for a 1,000 hectare watershed, and 0.22 for a 10,000 hectare watershed. The value selected for the example scenario for a 4,000 ha watershed area was 0.15. Changes in watershed sediment delivery ratios should accompany changes in assumed watershed size. Tests were run increasing the watershed size to 10,000 ha with an accompanying watershed delivery ratio to 0.12, and reducing the watershed size to 1,000 ha with the appropriate ratio of 0.22. Increasing the watershed size decreased the sediment concentrations by 1/2; and decreasing the watershed size increased sediment concentrations by 2.5.

A small watershed may not have a water body adequate to support fish for consumption or water for drinking exposures. SAIC (1986) reports watershed areas for most of the 70 landfills for which erosion rates were calculated. These watershed sizes vary considerably with 10 of 51 watersheds being smaller than 1,000 acres (400 ha) and 16 of the 51 being 100,000 acres (40,000 ha) or above (median 11,000 acres - 4500 ha; mean 29,000 acres - 12,000 ha). Thus the assumed value of 10,000 acres, or 4,000 ha, falls in the middle of the observed values.

3. Unit Soil Loss Estimates for Site and Watershed

Concentrations are also impacted by assumptions for unit soil loss, including that from the contaminated site, SL_s, and that from the watershed on the average, SL_w. The unit soil loss from the contaminated site was modeled from the landfill data earlier described (SAIC, 1986). The value of 62 T/ac-yr is high - mainly because of the assumption of bare soil. All other parameters were reasonable mid-range values (see Section 5.3.1). Reducing SL_s by 1/2 to 31 T/ac-yr decreased sediment concentrations by 1/2. The unit soil loss assumed for the watershed on the average, on the other hand, was relatively low, at 6 T/ac-yr. The only difference was the assumption of erosion retarding ground cover (grass, crops, etc.). If the average soil loss over the watershed were instead 5 times higher at 31 T/ac-yr, sediment concentrations would be diluted and would be five times lower. As seen, there is a linear relationship between soil loss estimates and estimated sediment concentrations.

4. Stack Emission Deposition Rate and Mixing Depth

Average concentrations over a watershed resulting from stack emissions are a function of an average watershed contaminant deposition rate, a representative watershed mixing zone depth, and an assumed dissipation rate for depositing residues. The impact of the assumption of a dissipation rate corresponding to a 10-year half-life was minimal, as discussed above in Section 10.2.3.2.

The validity of deposition rates modeled with the ISC model is discussed in Section 10.3. A representative deposition rate for the example scenarios was one given in Table 6.19 corresponding to the stack emission controls which were demonstrated, and at a distance of 5,000 meters; the exposure sites of the demonstration were at 500 and 2,000 m from the incinerator. For a 4,000 hectare watershed, depositions modeled to occur at 5,000 meters could be high, low, or appropriate depending on: 1) where water is withdrawn and fish caught for consumption within the watershed, 2) where the incinerator is located within the watershed, and 3) climate, terrain, and all other assumptions that are required for atmospheric transport using the ISC model. For the example scenarios, water was assumed to be withdrawn, and fish caught for consumption, at the bottom of the watershed - this translates to the assumption that all land draining into the river system impacts the sediment quality. If water is withdrawn from the middle of the watershed, the

appropriate assumption would be that all land upgradient of the withdrawal point impacts the sediment quality. Selection of an appropriate deposition rate is a site-specific issue. Selecting the rate at 5000 meters was done for this assessment also because it was felt that the average watershed soil concentration should be lower than concentrations assumed to occur at relatively close exposure sites (500 and 2000 meters). Ideally, this analysis would be done by estimating the deposition in numerous small subsections of the water shed and using an integration approach to estimate the total contaminant load to the stream.

In any case, the average watershed soil and hence sediment concentrations are a linear function of deposition rates. Figure 6-3 shows that deposition rates at 10 km are half of what they are at 5 km, and they are likely to be half again at 20 km. Sediment concentrations using a deposition rate corresponding to 10 km would, therefore, be one-half what they were assuming the rate at 5 km.

The assumed average mixing depth over the watershed was 10 cm. This is midway between the assumption of 1 cm for the non-tilled mixing depth and 20 cm for tilled situations. All example scenarios were described as rural settings. If the rural setting were predominantly agricultural, then an average mixing depth might be closer to 20 cm. However, agricultural experts are currently promoting no-till or minimum till in order to reduce erosion losses. With these practices, a mixing depth due to tillage may not be as high as 20 cm. Also, a rural setting which is not predominantly agricultural may also not call for an average mixing depth of 20 cm. In any case, average watershed concentrations and hence sediment concentrations are inversely related to mixing depth assumptions: doubling it to 20 cm would halve the estimated sediment concentrations, and reducing it to 5 cm would double sediment concentrations.

10.2.5. Soil Ingestion Exposure

This exposure is directly a function of the concentration of contaminants in surface soil layers. For example Scenarios 1 and 2, demonstrating the on-site soil source category, soil at the site of exposure was contaminated to a specified level. For example Scenarios 3 and 6, demonstrating the off-site and ash landfill source categories, erosion onto the site of exposure deposited residues into a thin, no-till, surface layer of 1 cm, and a thicker, 20-cm, till layer of soil. Soil ingestion exposures were based on concentrations

in the 1-cm layer. In Scenarios 4 and 5 demonstrating the stack emission source category, contaminated particles deposited onto the exposure site creating a till and a no-till concentration as well.

Uncertainties associated with the erosion methodologies are discussed in Section 10.2.1 above, and uncertainties associated stack emission and deposition and discussed in Section 10.3. below. Of particular note from these discussions is the uncertainty associated with a 1-cm depth of mixing. As was noted, others have also assumed depths of mixing this shallow for analogous applications. Evidence from radioactive fallout suggests depths no deeper than 5 cm. Increasing the depth to 5 cm would reduce surface concentrations by a factor of five and estimated soil ingestion exposures as well.

Another issue is whether children should be assumed to be exposed to tilled soils, tilled by home gardening, farming, etc., or untilled soils. It is feasible that children would be exposed to tilled soils, which have lower concentrations, during periods of home garden preparation. However, it is more reasonable to assume that they generally play outside in areas that are not mechanically tilled.

The estimated soil ingestion quantity is based on field measurements, using trace elements, of soil ingested by relatively small groups of children over brief periods. Methodological issues in these studies remain to be addressed. In particular, ingestion estimates may have been lower if dietary intake of the trace elements was taken into account. Research is underway to refine soil ingestion estimates obtained through trace element measurements. Given the available data, 0.2 g/day is used as a typical value for soil ingestion in young children. Due to the behavior known as pica, some children are known to be high ingesters of various non-food materials; although no quantitative data on soil ingestion are available for children known to exhibit pica, the use of the high-end estimate of 1.0 g/day may better reflect such behavior.

Because of remaining methodological research needs, no quantitative estimate of the uncertainties in these estimates is made here.

Soil ingestion exposure estimates also depend on the duration of the period over which children are assumed to ingest soil. Data on soil ingestion by age are not available, and the estimate that significant ingestion occurs between ages 2 and 6 is broadly supportable on behavioral grounds.

No measurement data are available on soil ingestion in infants (0-2 yrs. old) or in

older children or adults, and no ingestion is assumed for these groups. While some soil ingestion will occur in these groups, e.g., through contact of soiled hands with food, it is plausible that such ingestion is of a lesser degree than occurs in early childhood. If Hawley's (1985) estimate that an adult ingests an average 0.060 g/d of soil is used, after accounting for differences in exposure duration (9-20 yrs vs. 5 yr) and body weight (70 kg vs. 17 kg), the adult soil ingestion exposure is close to the estimated exposure for children (at 0.2 g/d). The high end example scenarios in Chapter 9 assumed that the exposed family was involved in farming operations. One implication is that individuals on the farm would be working closely with the soil, which may result in some soil or dust ingestion (dust ingestion is distinct from the particulate inhalation exposure pathway). The other implication is that, should this be the case, they would be in contact with tilled soil, whose concentration is 20 times less than the no-till soil for which children are assumed to be exposed.

Considering these uncertainties, the soil ingestion exposure estimates presented for children are plausible. Further consideration may be warranted for considering adult soil ingestion, particularly in farming situations. Uncertainties associated with the soil ingestion pathway are summarized in Table 10.5.

10.2.6. Soil Dermal Contact Pathway

Estimates of dermal exposure to soil rely largely on four factors unique to this pathway: exposed skin area, soil adherence, frequency of soil contact and fraction of contaminant absorbed. The uncertainty in these three terms are discussed below.

The uncertainty in the assumed value for exposed skin area reflects primarily population variability. As reported in EPA (1992), relatively accurate measurements have yielded a good data base on total skin area. Thus the uncertainty in this factor is derived more from the assumptions of how much of the total skin area is exposed. EPA (1992) recommends approaching this issue by determining the coverage of normal apparel in the exposed population and assuming exposure is limited to the uncovered skin. As discussed in EPA (1992), this assumption could lead to underestimates of exposure since studies have shown that some exposure can occur under clothing, especially in the case of vapors or fine particulates. A default assumption of 25% uncovered is recommended corresponding to short sleeved shirt, short pants, shoes, and socks. Thus the key

Table 10.5. Uncertainties associated with the soil ingestion pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Children exposed to surficial residues	Migrating residues deposit in a 1-cm no-till depth and a 20-cm till depth; children are exposed to concentrations estimated at 1-cm	There is no research to refine the 1-cm estimate, although others have assumed similarly shallow depths for analogous applications; evidence suggests a no-till depth no deeper than 5 cm.	Increasing the no-till depth to 5 cm would reduce concentrations and exposures by a factor of 5.	Sections 10.2.1 and 10.3 discuss related uncertainty issues concerning migration of dioxin-like compounds; a 1-cm depth is a conservative estimate that should be further examined.
Child's ingestion rate (2-6 years old).	Ingestion rate assumed to vary from 0.2-0.8 g/d.	The range selected was primarily based on the results of two field studies of soil ingestion in children.	Field study methodology not fully validated. Data from several sources indicate this range of values for small children.	Pica children have been estimated to ingest higher quantities (5 g/d).
Ingestion rate for other ages	Ingestion assumed to occur only during ages 2-6.	Mouthing tendencies strongest and understanding of personal hygiene low during childhood	Hawley estimates inadvertent ingestion may be 60 μg/d for adults, which would lead to an estimated exposure pattern similar to that of children	Adults may inadvertently ingest soil during gardening and yard work; farmers may have a non-trivial soil ingestion pattern

Overall: Soil ingestion for older children and adult was not considered, which may have underestimated exposures by a factor of two. The other major area of uncertainty for this pathway is for the scenarios where the source of contamination is located distant from the site of exposure, including areas of high soil contamination (the off-site and ash landfill source categories) and the stack incinerator. In those cases, depositions onto the site of exposure are distributed within a thin 1-cm layer. The depth of this layer is uncertain - it is more likely too low than high. Increasing it to 5-cm would decrease exposure estimates by a factor of five. Pica soil ingestion patterns were not evaluated in this assessment; the ingestion rates considering this appear reasonable.

uncertainty issue concerns the variability in clothing behavior of the exposed population. In this document the 25% assumption was adopted for residents and 5% was judged more reasonable for farmers who are more likely to wear long pants and long sleeved shirts for field work. Although clothing coverage is likely to vary over the year and with personal habits, these assumptions are judged to be reasonable averages and unlikely to introduce more than a factor of two uncertainty.

The potential for soil adherence probably varies little across the population, but few actual measurements have been made. Thus the uncertainty in these estimates reflect primarily the lack of measurement data rather than population variability. Site variability is probably important as well since soil properties such as moisture content, clay content and

particle size distribution are likely to affect adherence. EPA (1992) reports four studies which estimated soil adherence on hands under both laboratory and field conditions. Data from these studies were analyzed to obtain a central estimate of 0.2 mg/cm^2 and a high end estimate of 1.0 mg/cm^2. The uncertainty in these estimates are derived from unknown efficiencies in the collection methods, relatively small number of subjects, assumption that hand measurements apply to other parts of the body and assumption that child measurements apply to adults as well. The central default value 0.2 mg/cm^2 was adopted here for the residents and the high end value of 1.0 mg/cm^2 was adopted for farmers. The uncertainties in this estimate could combine to produce either under or over estimates and may vary by as much as a factor of 5 on the basis of the ratio of the high end to central estimates.

Exposure frequency to soil reflects largely personal habits and thus the uncertainty is primarily based on population variability. Seasonal and climate conditions can also affect this behavior introducing site variability as well. EPA (1992) suggests a central frequency of 40 days/yr corresponding to someone who does yard work, gardens or plays outdoors on most weekends and a high end estimate of 350 days/yr corresponding to a serious gardener in a warm climate. These recommendations were based on judgement rather than actual survey data. In this document, 40 days/year was selected for the residential scenarios and 350 days/yr for the farmer. The lack of survey data to support these estimates introduces uncertainty, but the values are judged to be reasonable and to create relatively little uncertainty.

The dermal absorption fraction of compounds varies widely across chemicals, whereas skin properties that affect absorption, i.e. thickness and composition vary little across the population. Thus the uncertainty in this factor is derived primarily from measurement error rather than population variability. Soil properties, such as organic carbon content, can also affect the extent of dermal absorption and thus create site variability as well. EPA (1992) reports two studies which measured dermal absorption of 2,3,7,8-TCDD from soil. Testing included human skin in vitro, rat skin in vitro and rat skin in vivo. On the basis of these tests, a range of 0.1 - 3.0% was recommended in EPA (1992). Dermal absorption testing, especially for soils, is a relatively new field and many uncertainty issues are involved. These include extrapolation of animal tests to humans, extrapolation of in vitro to in vivo conditions, and extrapolation of experimental conditions

to expected exposure conditions. Extrapolation of the tests on 2,3,7,8-TCDD to the other dioxin like compounds (which have not been tested) introduces further uncertainties. A dermal absorption fraction of 3.0% was adopted here for application to all the dioxin like compounds. Based on the observed range of values for 2,3,7,8-TCDD this assumption may lead to overestimates of a factor of 30. Considering all possible uncertainties, under estimates are also possible, though judged less likely.

In summary, dermal exposure estimations rely on a number of parameters whose values are not well established. Although it is difficult to estimate the overall uncertainty with this pathway, it is judged to be plus or minus one to two orders of magnitude. A summary of the uncertainties associated with the dermal absorption pathway is given in Table 10.6.

10.2.7 Water Ingestion

Surface water concentrations of dioxin-like compounds are driven by concentrations estimated to occur in suspended sediments. Suspended sediment concentrations are a function of surface soil concentrations of dioxin-like compounds within the watershed, or sub-watershed area, draining into the water body. The uncertainty in estimating sediment concentrations are discussed in Section 10.2.4 above. As water concentrations are directly and linearly related to suspended sediment concentrations, uncertainties in estimating concentrations in suspended sediment are also pertinent to estimating surface water concentrations.

The section begins with evaluating the exposure parameters, which include the water ingestion rate and then contact fraction. Then, the model results are compared with limited water quality data. The next subsection considers an alternate approach for estimating water column concentrations. Finally, the sensitivity and variability of key model parameters are evaluated. Table 10.7 summarizes the uncertainties associated with water ingestion.

10.2.7.1. *Exposure Parameters*

The classically assumed water ingestion rate of 2.0 L/day was examined in EPA (1989). The conclusion was that this estimate is more appropriately described as an upper percentile consumption rate for adults, and recommended 1.4 L/day for use as an average.

Table 10.6. Uncertainties associated with the dermal exposure pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Soil Concentrations	same for on-site source category; used "tilled" concentrations for high end farming scenarios, "non-tilled" for non-farmers in central scenarios for all other source categories	behavior parameters for farmers assume dermal exposure results from farming in tilled soils; for non-farmers, yard work during summer assumes dermal contact occurs in non-tilled soils	tilled concentrations are 20 times less than non-tilled concentrations; much uncertainty associated with difference between tilled and untilled mixing zone depths; non-farmers working in home gardens also are exposed to tilled concentrations	For real sites, monitoring is best way to resolve this uncertainty
Contact rate	0.2 mg/cm^2-event for non-farming residents; 1 mg/cm2-event for farmers	corresponds to central and high end values of measurement data; supported by EPA, 1992	measurement data may have experimental error or not be representative; uncertainty plus/minus factor of 5	soil properties could affect adherence
Contact frequency	40 for residents; 350 for farm families	Based on judgement; supported by EPA (1992)	personal behavior patterns could differ but uncertainty judged to be small	climatic conditions introduce site variability
Surface area	5% (1000 cm^2) for farming adults and 25% (5000 cm^2) for non-farming adults	Based on total body surface area data and clothing assumptions; supported by EPA (1992)	Clothing assumptions based on judgement rather than survey data; uncertainty judged to be plus or minus factor of 2.	Studies have shown that fine particulates can deposit under clothing
Absorption fraction	0.03 for all dioxin-like compounds	Based on EPA (1992) which gave a range of 0.001 to 0.03; range supported by 2 different studies in rat and human	Experimental procedures very uncertain, may over estimate by factor of 30	Soil properties may also affect absorption

Overall: The high uncertainty in estimates of soil adherence and absorption fraction make the overall uncertainty in the exposure estimates highly uncertain, judged to be plus or minus 1 to 2 orders of magnitude.

However, EPA (1989) cautions that data on consumption rate for sensitive subpopulations such as manual laborers are unavailable. As such, the 1.4 L/day rate for individuals in farming families who work the field may be low.

The contact fraction is defined as fraction of total contact with an exposure media that is contact with contaminated media. For drinking water, this translates to the fraction of water ingestion that comes from the contaminated water source. In the example scenarios, it was assumed that the impacted water was a river which supplied water to

Table 10.7. Uncertainties associated with the water ingestion pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Water ingestion rate	1.4 L/day	The classically assumed 2.0 L/day was evaluated in EPA (1989) and found to be high; recommended 1.4 L/day in general assessments	EPA (1989) also noted that information on sensitive subpopulations such as laborers was unavailable; estimate might be low, therefore, for farmers	Not expected to be a critical factor for uncertainty
Contact rate	0.75 & 1.00	0.75 recommended in EPA (1989) for general uses based on time spent at home; recommended value of 1.00 was felt to be too high for high end scenarios	Not expected to be widely variable for rural settings.	Same comment as above
Sediment concentrations		SEE Section 10.2.2.		
Model Parameters	Koc, TSS, foc	organic carbon partition coefficient (Koc) estimated from octanol water partition coefficient (Kow); total suspended solids (TSS) appropriate for river which supports fish; fraction organic carbon (foc) may be low but reasonable	Koc an estimated rather than measured value; selected value for 2,3,7,8-TCDD may be low; other parameters are site-specific; overall, estimations in example scenarios may have led to overpredictions by a factor of 2	Relatively narrow range of possible values for all noted parameters; uncertainty would be evaluated as low

EVALUATION: Data was unavailable in the literature which linked soil contamination with surface water concentrations. Estimated concentrations were in the low to sub pg/L (ppq) range, which was consistent with literature values. Modeling exercise using complex WASP4 model applied to Lake Ontario (EPA, 1990a) was duplicated using simplistic approach of this assessment, and results from both exercises were comparable. Model parameters were judged as reasonable values with narrow range of possible values. Possibly low values of 2,3,7,8-TCDD Koc and foc may have led to overpredictions of concentrations by a factor of 2.

the exposed individuals, perhaps through a public water system. The contact fraction of 0.75 for central scenarios is based on time use surveys which showed roughly this fraction of time spent in and around the home environment on the average. The upper recommended limit in EPA (1989) was 1.00; this was felt to be unrealistic for the example scenarios which involved relatively small sources and consequently the likelihood that contamination would not be widespread. Thus, a farmer would likely obtain some water from outside his home where the water supply was not contaminated. An assumption of

0.90 for farming families was selected for the high end scenarios of this assessment.

In summary, the ingestion rate and contact fractions are justifiable for use in central and high end scenarios. Variations outside noted ranges are not expected to be great and any reasonable variation would have limited impact to long term exposure estimates.

10.2.7.2. *Comparison of Estimated Water Concentrations With Observed Data*

Tables 3-3 and 3-4 (Chapter 3) summarize available data on surface water concentrations of the PCDDs and PCDFs. The results on this table are not directly amenable to comparison because the sources of contamination were unspecified except to note that, in some studies, a portion of the sampling occurred for water bodies known to be impacted by industrial discharges. No studies were found which both measured and found surface soil concentrations, suspended sediment concentrations, and water column concentrations of dioxin-like compounds. Such studies would be necessary to validate the modeling approach in this assessment. The impact to surface water from industrial discharges is not discussed in this assessment. Assessments of exposure to 2,3,7,8-TCDD resulting from paper and pulp mill discharges can be found in EPA (1990b) and EPA (1991).

Nonetheless, this data does indicate that occurrences of PCDDs and PCDFs are generally not-detected or in the low pg/L (ppq) range; detection limits were generally at or near 1 pg/L. The one exception to this is occurrences in tens to hundreds of pg/L range for PCDFs in one of twenty community water systems sampled in New York (Meyer, et al., 1989). Concentrations exceeding 200 pg/L were found in the hepta- and octa-CDFs; concentrations between 2 and 85 pg/L were found in the tetra to hexa-CDFs for this impacted water system.

The highest water concentration estimated in the demonstration scenarios in this assessment was 8 pg/L for the ash landfill demonstration scenario, Scenario 6, for 2,3,4,7,8-PCDF. Soil concentrations of this congener for the 27 ha ash landfill were 8 ng/kg (ppb). Water concentrations for 2,3,7,8-TCDD and 2,3,3',4,4',5,5'-HPCB were both 0.9 pg/L for this scenario, and ash landfill concentrations were 0.7 and 2.0 ppb for each congener, respectively. Water concentrations of the three demonstration congeners were 0.2 (2,3,7,8-TCDD), 0.2 (2,3,4,7,8-PCDF), and 0.07 (2,3,3',4,4',5,5'-HPCB) pg/L range for the example scenario demonstrating off-site soil contamination, Scenario 3; soil

concentrations of the three contaminants were set at 1 ppb for the 4 ha area of off-site contamination. For Scenarios 1 and 2, where watershed soil concentrations were set at 1 ng/kg (ppt), surface water concentrations were in the 10^{-3} pg/L (ppq) range. For Scenarios 4 and 5 demonstrating stack emission depositions and where watershed soil concentrations were estimated to be in the 10^{-2} to 10^{-1} ppt range, surface water concentrations were the in the 10^{-4} to 10^{-3} ppq range.

10.2.7.3. *An Alternate Modeling Approach for Estimating Water Concentrations*

A study to evaluate bioaccumulation of 2,3,7,8-TCDD in fish in Lake Ontario included an extensive modeling exercise (EPA, 1990a). The model used was WASP4 (Ambrose, et al., 1988). This is a substantially more complicated model than used in this assessment. The underlying principal for this model is a conservation of mass. Contaminant input to a water body, described in mass/time units, enters what are termed control volumes, or segments. The contaminant entering partitions between sorbed, bound, and dissolved phases once entering; it is not required to specify whether the contaminant enters via soil erosion, water runoff, surface deposition, or otherwise. Contaminants are, however, assumed to enter via the surface or as part of inflows to the water body, in contrast to ground water recharge. The mass transported into a segment is either transported out of the segment, accumulates in the segment, or is transformed by chemical reaction or biological degradation.

As noted, 2,3,7,8-TCDD input into the Lake Ontario application partitions within the water column into a sorbed compartment, a dissolved compartment, and a bound compartment. This bound compartment is further described as non-settling organic matter. Three analogous compartments receive 2,3,7,8-TCDD in the bottom sediment layer. Several exchanges between the six compartments and contaminant losses within each compartment are modeled. For example, losses from water column compartments include downstream transport, volatilization and photolysis; the loss mechanism from the bottom sediment layer is sedimentation. Exchanges between compartments consider partitioning, diffusion, and sediment settling and resuspension.

This model requires substantial parameterization. Once values were selected for the Lake Ontario application, an evaluation was made on the impact of different levels of 2,3,7,8-TCDD input. Dynamic and steady state results were discussed. Principally

examined for the steady state results were the concentrations of bottom sediment sorbed 2,3,7,8-TCDD and water column dissolved (soluble phase) 2,3,7,8-TCDD. A given level of steady 2,3,7,8-TCDD input, in kg/yr, resulted in a steady state concentration sorbed to bottom sediment and dissolved in the water column.

The premise in both the Lake Ontario steady state application of WASP4 and the water concentration algorithms in this assessment is that contaminants continue to enter water bodies over time unabated. Ground water entry of contaminants is not considered in either approach. Although a direct modeling comparison cannot be done, it is possible to slightly adjust the algorithms of this assessment to evaluate how results from a simple partitioning approach would compare with results from the complex fate and transport approach of the WASP4 steady state application.

Assume a surface water body is initially free of contaminant and at time t equals 1 day, a strongly hydrophobic contaminant, such as the dioxin-like compounds of this assessment, begins to enter a lake. Assuming the contaminant enters via soil particles, as in the approach of this assessment, it will then partition between those soil particles and surrounding water. The soil particles will slowly move toward the bottom of the lake at a rate described by a particle settling velocity. A settling velocity of 1 m/day is assumed in the Lake Ontario simulations. The amount of time it takes to settle to the bottom once entering from the surface equals the lake depth divided by this settling time. The Lake Ontario depth was 86 m. Therefore, it might take 86 days to settle. This, of course, neglects resuspension of settled particulates. With this simplistic framework, a steady state amount coming into the lake after 86 days is matched by an amount depositing onto the lake bottom; the amount of contaminant within the water column has reached steady state. Water concentrations can then be estimated assuming equilibrium partitioning.

Results of sediment and water column steady state concentrations are described for any loading of 2,3,7,8-TCDD in the WASP4 steady state application; those loadings are described in kg/yr. Loadings in kg/yr are easily correlated to a steady state water column amount, given the above analysis. For example, a loading of 1.0 kg/yr could translate to a within water column steady state amount of 0.24 kg (1.0 kg/yr * (86 d)/(365 d/yr)).

This steady water column amount partitions between suspended sediment and surrounding water. First, the total concentration (sorbed + soluble) simply equals:
where:

$$C_{tot} = 1000 \frac{LD}{VOL} \qquad (10\text{-}1)$$

C_{tot}	=	total concentration, mg/L
LD	=	water column steady state amount of contaminant, kg
VOL	=	lake volume, m^3
1000	=	converts kg to mg and m^3 to L

The dissolved phase portion of total is given, as in Equation (5-3), by:

$$C_{wat} = \frac{C_{tot}}{1 + (Kd_{ssed}\ TSS\ 10^{-6})} \qquad (10\text{-}2)$$

where:

C_{wat}	=	soluble phase water concentration, mg/L
C_{tot}	=	total concentration, mg/L
Kd_{ssed}	=	partition coefficient between suspended sediment and surrounding water, L/kg
	=	$Koc*OC_{ssed}$
Koc	=	organic carbon partition coefficient, L/kg
OC_{ssed}	=	fraction organic carbon of suspended sediments
TSS	=	total suspended sediments, mg/L
10^{-6}	=	converts mg/kg to mg/mg

Parameters in this equation for the Lake Ontario WASP4 application include VOL, Koc, OC_{ssed}, and TSS. Lake Ontario volume was given as 1.68×10^{12} m^3, Koc was estimated for the WASP4 application as 3,162,000, OC_{ssed} was estimated at 0.03, and TSS was estimated 1.2 mg/L. For a steady load of 1 kg/yr and a resulting LD of 0.24 kg, the steady state water column 2,3,7,8-TCDD concentration, using the simplistic approach described above, is estimated as 0.13 pg/L (ppq). The steady state water column concentration estimated by WASP4 given the same parameters and a load of 1 kg/yr is roughly 0.20 pg/L. An uncertainty analysis done with these WASP4 results concluded

that 95% confidence limits around this prediction are 0.03 and 0.40 pg/L.

This would seem to imply that the simple partitioning approach used in this assessment compares favorably with the more complex fate and transport modeling assessment using WASP4, for Lake Ontario.

10.2.7.4. *Sensitivity to Model Parameters*

As noted in the introduction to this section, a key quantity of uncertainty is the concentration of contaminants in suspended sediments. An uncertainty discussion on this quantity is given in Section 10.2.2. In summary, that section notes the following: 1) the premise that suspended sediment concentrations are related to concentrations in soil within the watershed is a sound one, 2) most data in the literature including sediment concentrations cited industrial discharges (direct discharges such as from paper or pulp mills, or air emission discharges) as the source of contamination; there were no data linking concentrations found to soil concentrations, and 3) sediment concentration predictions would change by no more than a factor of five, either higher or lower, with changes to single parameters within a range of reasonable values.

Parameters which are required given suspended sediment contaminant concentrations include the concentration of suspended sediment, TSS, the organic carbon partition coefficient, Koc, and the fraction organic carbon in suspended sediments.

All example scenarios assumed that the site of contamination was in a watershed draining into a river. This justified a TSS of 10 mg/L. Suspended sediment concentrations in a standing water body such as a lake or a reservoir would likely be lower, around 2 mg/L. River suspended sediments could be higher, which would be the result of more erosion, turbulence, or bioturbation. The discussion in Section 5.3.1. indicated that a suspended sediment concentration of 10 mg/L or less indicated no problem with sustainability of aquatic life. The rivers of the example scenarios were assumed to support fish for recreational fishing and water for consumption. Water column concentrations are linearly related to suspended sediment concentrations. Reducing the concentration by 5 to 2 mg/L would reduce water concentrations by 5; increasing it to 20 or 30 mg/L would increase water concentrations 2-3 times.

The OC_{ssed} and Koc parameters measure the sorption of contaminant to suspended sediments. Increasing the value of these terms tends to decrease water concentrations.

The organic carbon fraction, OC_{ssed}, was set at 0.01 (1% organic carbon). This was lower than the organic carbon fraction measured for sediments of Lake Ontario in the study evaluating the bioaccumulation of 2,3,7,8-TCDD (EPA, 1990a). Increasing OC_{ssed} to 0.03 decreases concentrations by approximately 30%. Increasing OC_{ssed} to 0.05, which is a reasonable upper estimate for this parameter, reduces concentrations by 46%. The Koc for 2,3,7,8-TCDD was set at 2,700,000. Evaluation of Koc for sediments of Lake Ontario led researchers to conclude that it was greater than $10^{6.3}$ (2,000,000) and may be as high as $10^{7.3}$ (20,000,000). Changing Koc to 24,500,000, evaluated as a high estimate for 2,3,7,8-TCDD Koc, reduces estimated concentrations by 63%. Keeping this high Koc and increasing OC_{ssed} from 0.01 to 0.05 decreases concentrations by an order of magnitude. This analysis shows that selections of Koc and OC_{ssed} for suspended sediments might result in an overprediction of water column soluble phase concentrations, but likely by no more than a factor of 2 (i.e., higher Koc and OC_{ssed} might reduce concentrations by no more than 50%).

10.2.8. Fish Ingestion Exposure

Fish lipid concentrations are estimated as the multiplication of a Biota Sediment Accumulation Factor, BSAF, and an organic carbon normalized concentration of contaminant on bottom sediment. Whole fish tissue concentrations equals this lipid concentration times the lipid content of fish. The first section evaluates the rate of fish ingestion. Then, estimates of fish tissue concentrations made in this assessment are compared with fish tissue concentrations given in the literature. The third section evaluates alternate approaches to estimating fish concentrations, and the final section evaluates the sensitivity and uncertainty in key model parameters. A summary of key uncertainty issues associated with the fish ingestion pathway is given in Table 10.8.

10.2.8.1. *Fish Ingestion Rates*

Although fish consumption surveys are available and are discussed in EPA (1989), this assessment uses a different approach to estimate the consumption of fish from an impacted water body. The approach is recommended for use when site-specific survey or other information is unavailable (EPA, 1989). Briefly, assume a meal size of between 100 and 200 g/meal - this assessment assumed 150 g/meals - and estimate the number of fish

Table 10.8. Uncertainties associated with the fish ingestion pathway.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Fish ingestion rate	1.2 and 4.1 g/day	Based on 3 and 10 meals per year caught from the impacted water, and 150 g/meal	A low estimate compared to surveys of recreational or subsistence fisherman near large water bodies	Example scenarios were for "rural", agricultural settings; higher rates of ingestion appropriate for other purposes was not deemed appropriate for such settings.
Model Parameters	BSAF, f_{lipid}, OC_{sed}, Koc, TSS	Biota Sediment Accumulation Factors, BSAFs, based on field measured values; fish lipid content, f_{lipid}, of 0.07 used by Cook, et al. (1991) for 2,3,7,8-TCDD exposure assessment; fraction organic carbon in sediment, OC_{sed}, may be low but reasonable. TSS reasonable for river setting, and Koc based on literature	BSAFs for dioxin-like compounds field measured to be in 0.01-0.30 except one study measuring 2,3,7,8-TCDD BSAF to be 1.60; f_{lipid} variable even for same species, but values unlikely to exceed 0.20; OC_{sed} no higher than 0.20; TSS lower for lake setting; Koc may be higher for aquatic settings	Relatively narrow range of possible values for all noted parameters; High BSAF from one study and variable BSAF for PCBs suggests further refinement of BSAF assignment; sensitivity showed narrow range for fish tissue concentrations for individual changes in OC_{sed}, TSS, and Koc

EVALUATION: Data was unavailable in the literature which linked soil contamination with sediment and fish concentrations; comparison with literature values of fish concentrations of dioxin-like compounds suggests that concentrations predicted for background and high levels of soil contamination may be slightly low for dioxins, although predictions are clearly within range, and relative increases from background to high soil contamination as noted in the literature are mirrored by the model. Much higher sediment concentrations of PCBs and hence fish concentrations, in comparison to the estimates made in this assessment, were noted. The BSAFs were assigned based on field measurements for dioxins, furans, and PCBs. PCB BSAFs were an order of magnitude and more higher than dioxin and furan BSAFs; one literature reference for dioxin/furan BSAFs is an order of magnitude higher than others - this should be further investigated. Alternate modeling approaches based on water column concentrations show higher but comparable fish concentration estimations. Fish concentrations estimations vary by less than an order of magnitude with changes in model parameters.

meals that may be recreationally caught from the impacted water body. An estimate of 3 meals/year was made for central exposure scenarios, and 10 meals/year was made for high end exposures. Ingestion of contaminated fish is therefore, estimated as 1.2 and 4.1 g/day, respectively (150 g/meal * 3 meals/yr * 1/(365 d/yr) = 1.2 g/d).

Surveys of recreational fisherman near large water indicates that these estimates are low for this subgroup. As noted in Chapter 7, EPA (1989) estimates that a typical rate of ingestion of recreationally caught fish for this subgroup is 30 g/day, with a 90% estimate of 140 g/day. The range of 30-140 g/day may be more appropriate, therefore, if estimating fish ingestion exposure for recreational fisherman near a large, impacted water

body. EPA (1989) also reports on two surveys of per capita fish consumption, on a national basis and including all consumption, not just recreationally caught consumption. The mean and 95% consumption rates determined by Javitz (1980) were 14.3 and 41.7 g/day. The median and 95% consumption rates determined by Pao, et al. (1982) were 37 and 128 g/day. If using either of these estimates in exposure exercises, assumptions on percent of total consumption which is recreationally caught and/or impacted by dioxin-like compounds needs to be made.

A key trend noted for the example scenarios in Chapter 9 is that fish, along with beef and milk ingestion, led to the highest exposure estimates for the dioxin-like compounds. Obtaining site-specific information for fish ingestion is critical for this pathway. The ingestion rates made in this assessment are very likely low by an order of magnitude or more for use to a subgroup of recreational fisherman obtaining fish from a nearby large water body.

10.2.8.2. *Comparison of Estimated Fish Concentration With Literature Values*

This assessment estimated fish concentrations of 2,3,7,8-TCDD for the various source categories to be: 1) ash landfill - 5 ppt, 2) off-site soil contamination - 1 ppt, 3) on-site soil concentration - 0.07 ppt, and 4) stack emissions - 0.002 ppt. This section will examine some available data on fish concentrations in order to compare these results with measured results.

The most appropriate study with which to make comparisons is the National Bioaccumulation Study (see Chapter 3, Section 3.5 for reference and more detail; abbreviated NBS). Fish tissue data on a variety of species and contaminants of concern in aquatic environments and fish from around the country were developed. Most important for current purposes, the sites were carefully characterized in terms of potential sources of fish contamination. There were 353 sites from which fish tissue data was available, of which 347 had data on 2,3,7,8-TCDD. Results from four site categories might be appropriate for comparison with concentrations estimated to occur from low, possibly background, soil concentrations of 2,3,7,8-TCDD. The four categories and number of sites per category were: the USGS water quality network NASQAN - 40 sites; Background (B) - 34 sites, Agricultural (A) - 17 sites, and Publicly Owned Treatment Works (POTW) - 8 sites. The average 2,3,7,8-TCDD concentrations measured for these four categories were:

NASQAN - 1.02 ppt; B - 0.56 ppt; A - 0.75 ppt, and POTW - 0.90 ppt. The "on-site" source category was demonstrated in Chapter 9 using a soil concentration of 2,3,7,8-TCDD that might be typical of background conditions - 1 ng/kg (ppt). The resulting fish tissue concentrations estimated for this soil concentration was 0.07 ppt. Four of the site categories of the NBS might be considered representative of sources characterized as land areas of high soil concentrations of 2,3,7,8-TCDD. These were: Industrial/Urban site (IND/URB) - 105 sites, Refinery/Other Industry (R/I) - 20 sites, Wood Preservers (WP) - 11 sites, and Superfund Sites (NPL) - 7 sites. Average fish tissue concentrations measured for these site categories were: IND/URB - 4.04 ppt, R/I - 4.38 ppt, WP - 1.40 ppt, and NPL - 30.02 ppt. Two of the source categories of this assessment are similarly characterized - land areas with elevated levels of 2,3,7,8-TCDD. These were the off-site category, demonstrated with 1 μg/kg (ppb) concentrations of 2,3,7,8-TCDD, and the ash landfill source category, demonstrated assuming 0.7 ppb 2,3,7,8-TCDD. As noted above, the fish tissue concentrations estimated with these source categories were 1.0 (off-site) and 5.0 ppt (ash landfill). The two remaining site categories of the NBS were Paper Mills Using Chlorine (PPC), and Other Paper Mills (PPNC). Fish tissue concentrations with these sources were 19.02 and 2.17 ppt, respectively. A discussion of effluent discharge modeling is given in Appendix D of this assessment.

In general, the range of fish tissue concentrations measured for (perhaps) background conditions, 0.56 - 1.02 ppt, were somewhat higher than the fish tissue concentration estimating assuming the low (perhaps) background soil concentration of 1 ppt soil concentration, 0.07 ppt. It may also be appropriate to make the same observation for the source categories assuming higher soil concentrations as compared to measured concentrations noted above. In this case, the range of average measured concentrations - 1.4 - 30.02 ppt compares with the modeled 1.0 - 5.0 ppt. Obviously, it would be inappropriate to conclude that the model underestimates fish tissue concentrations since there is no information on the source strengths (or other information for environmental fate modeling) for the measured tissue concentrations. It is probably more appropriate to view this comparison as one that tends to lend some credibility to perhaps both the selection of source strengths and modeling approaches used in this assessment. The difference in measured background concentrations versus those measured for industrial-like sites is mirrored by the model. The actual concentrations predicted are within range of the

measured concentrations. Different assumptions on the source strength terms (soil concentrations, erosion amounts) and different assumptions on water column partitioning (as modeled with partition coefficients, suspended solids concentrations, and fractions of organic carbon in suspended solids and bottom sediments) would increase modeled fish concentrations. The impact of the partition coefficient is discussed in Section 10.2.8.4 below.

Another comprehensive data base of fish concentrations of 2,3,7,8-TCDD is from EPA's National Dioxin Study (1987; abbreviated the NDS). Fish concentrations from that study are also listed and discussed in Kuehl, et al. (1989). Travis and Hattermer-Frey (1991) summarized the fish data from the NDS, and that summary is reiterated in Chapter 3, Section 3.5. Briefly, data collected from 304 urban sites in the vicinity of population centers or areas with known commercial fishing activity, including the Great Lakes Region, showed concentrations to range from non-detected to 85 ng/kg (ppt). The geometric mean concentration was 0.3 ppt, and only 29% had detectable levels of 2,3,7,8-TCDD. The Great Lakes data had more contamination, with 80% detection rate and a geometric mean concentration of 3.8 ppt. The fish tissue concentrations estimated for 2,3,7,8-TCDD for the two categories of off-site soil contamination - the ash landfill at 5 ppt and off-site soil at 1 ppt - are comparable to the two geometric means for the NDS, 0.3 and 3.8 ppt.

While the data from the NDS targeted urban areas where higher fish concentrations can be expected, other data with known point source inputs had higher concentrations of 2,3,7,8-TCDD and related compounds. In evaluating the impact of paper and pulp mills, EPA (1991) used specific data from the NDS. This data was for four species of fish (composites of each species) from the Wisconsin River, and samples were taken approximately two miles from a mill discharging into the river. The estimated average whole body concentration of 2,3,7,8-TCDD for these fish was 33 ppt (estimated because 3 of the 4 fish species were sampled for fillet only; whole fish concentrations estimated at twice the fillet concentration). Pike sampled from Lake Vanern in Sweden had whole fish average concentrations of 41 ppt of 2,3,7,8-TCDD with a high concentration of 150 ppt (Kjeller, et al., 1990).

The National Bioaccumulation Study (NBS) also collected data on 2,3,4,7,8-PCDF, the second example compound demonstrated. Briefly, the range of average fish tissue

concentrations noted for the site categories evaluated as background above is 0.42-0.78 ppt, very similar to the 2,3,7,8-TCDD range of 0.56-1.02 ppt. The modeled fish tissue concentration of 2,3,4,7,8-PCDF for background conditions was 0.09 ppt, slightly higher than the 2,3,7,8-TCDD modeled concentration, 0.07 ppt, because of different partitioning in the water body. The range of 2,3,4,7,8-PCDF average fish concentrations for the NBS sites of elevated soil concentration was 1.86-5.44 ppt, which compares to the modeled 2,3,4,7,8-PCDF concentrations of 2 (off-site source category with initial soil concentrations of 1.0 ppt 2,3,4,7,8-PCDF) and 80 ppt (ash landfill with initial soil concentrations of 8.2 ppt).

The NBS also collected data on PCB concentrations in fish, although the results were expressed in terms of total tetra-, hepta-, and so on. The data indicates concentrations well into the part per billion range for this breakout, and even higher considering total PCBs. The average concentration of total heptachlorobiphenyls over all NBS sites was 96.7 μg/kg (ppb). The average concentration of total PCBs over all sites was estimated as 1897.88 ppb, and the average concentration of total PCBs for background sites was 46.9 ppb. The modeled concentration of the example heptachlorobiphenyl, 2,3,3',4,4',5,5'-HPCB for the background scenario (soil concentration = 1.0 ppt) was 3 ppt. The modeled concentration of 2,3,3',4,4',5,5'-HPCB for the off-site scenario (soil concentration = 1 ppb) and the ash landfill (soil concentration = 2.2 ppb) was 40 and 600 ppt, respectively.

Data from the Great Lakes region indicate that PCB concentrations are significantly higher than PCDD/PCDF concentrations in this area. PCB concentrations from fish in Lake Ontario are in the tens to hundreds of ppb level (Niimi and Oliver, 1989), while 2,3,7,8-TCDD contamination in Lake Ontario was in the tens of ppt level (EPA, 1990a) - a three order of magnitude difference. Other data in Table B.10 (Appendix B), where concentrations were similarly in the tens to hundreds of ppb level were from Lake Michigan (Smith, et al. 1990) and Waukegan Harbor in Illinois (Huckins, et al. 1988). The single data point from that table for 2,3,3',4,4',5,5'-HPCB, the example PCB congener in Chapter 9, was for carp in Lake Michigan, and was 29 ppb (29,000 ppt).

While the modeled PCDD/PCDF fish concentrations seem reasonably in line with measured concentrations, this assessment may have underestimated concentrations of 2,3,3',4,4',5,5'-HPCB. As noted, concentrations for fish in the Great Lakes Region were

in the tens to hundreds of ppb range, while this assessment derived estimates all under 1 ppb. It is inappropriate to make direct comparisons without also comparing source strengths. Concentrations of PCBs in bottom sediments ranged from the low ppb for the tri-PCBs, to the tens of ppb for the tetra through hexa-PCBs, back to the low ppb for the hepta and octa-PCBs, in Lake Ontario (Oliver and Niimi, 1988). Another literature source showing fish concentrations in Waukegan Harbor, IL, in the hundreds of ppb range, had sediment concentrations of specific congeners as low as 5 ppb to as high as 131 ppm. The concentration of 2,3,3',4,4',5,5'-HPCB in bottom sediments never reached 1 ppb. Therefore, one reason PCB concentrations in fish estimated in this assessment are as much as three orders of magnitude lower than noted in the literature is because sediment concentrations estimated for the source categories in this assessment are three orders of magnitude lower. The BSAF for PCBs also was noted to be variable, with values below 1.0 to values over 20.0 (see Section 5.4.2). The BSAF for the example PCB congener in this assessment was 2.0. Higher BSAFs would also increase PCB concentrations estimated for fish.

10.2.8.3. *Alternate Modeling Approaches*

EPA is currently preparing a document titled, "Interim Aquatic Ecological Risk Characterization for 2,3,7,8-TCDD" (P. Cook, personal communication, Duluth Environmental Research Laboratory, US EPA, Duluth, MN). A draft version of an exposure and bioaccumulation chapter was obtained for purposes of this section.

Alternate measures of the potential for accumulation of contaminants in fish based on water column and bottom sediment (i.e., the BSAF approach) contaminant concentrations were discussed in that chapter. One water column measure which has been classically used is termed the Bioconcentration Factor, or BCF. Bioconcentration refers to the net accumulation of a chemical from exposure via water only, and BCFs are most often obtained in laboratory conditions. BCFs are defined as the ratio of the chemical concentration in organism (mass of chemical divided by wet weight of organism tissue) to that in water.

Another water column measure of the potential for a contaminant to accumulate in fish tissue is termed the Bioaccumulation Factor, or BAF. Bioaccumulation refers to the net accumulation of a chemical from exposure via food and sediments as well as water.

Similar to the BCF, BAFs are defined as the ratio of the chemical concentration in the organism to that in the water.

For chemicals that are not strongly hydrophobic (unlike the dioxin-like compounds), the distinction between bioconcentration and bioaccumulation is small. Whereas food intake is generally a few percent of body weight per day, water passing over gills will equal hundreds to thousands times the organism weight per day, depending on species, activity, temperature, and other factors. Given this, the concentration of chemical in food must be 3 or more orders of magnitude greater in food than in water before food can substantially contribute to uptake. Cook (reference above) estimates food intake becomes a critical contributor to the accumulation of contaminants in fish tissue for contaminants with log Kow of 5 and greater.

Since the dioxin-like compounds fall into this category, the remainder of this section will focus on the Bioaccumulation Factor. Cook defines steady-state lipid-based BAFs for total chemical in water and freely dissolved chemical in water (i.e., chemical which is truly in a dissolved phase and not bound to dissolved organic materials) as:

$$ssBAF_l^t = \frac{C_{lipid}}{C_w^t} \qquad (10\text{-}3a)$$

$$ssBAF_l^d = \frac{C_{lipid}}{C_w^d} \qquad (10\text{-}3b)$$

where:

$ssBAF_l^t$ = steady-state lipid-based BAF for total chemical in water, unitless

C_{lipid} = the mass of contaminant in fish lipid tissue divided by the mass of fish lipid tissue, mg/kg

C_w^t = the mass of total contaminant in water divided by the mass of water in the water body, mg/kg (note: 1 L water = 1 kg)

$ssBAF_l^d$ = steady-state lipid-based BAF for freely dissolved chemical in water, unitless

C_w^d = the mass of freely dissolved contaminant in water divided by the mass of water in the water body, mg/kg

Cook then develops these steady-state lipid-based BAFs for lake trout, 2,3,7,8-TCDD, and for Lake Ontario 1987 contamination conditions. The WASP4 model (Ambrose, et al., 1988) was used to model two loading conditions to the lake: steady state loading and no loading of 2,3,7,8-TCDD. The BSAF for lake trout estimated for 1987 conditions is given in EPA (1990a) as 0.07. Details of the Lake Ontario study can be found in EPA (1990a). The BAFs Cook determined will be tested using the models of this assessment.

In order to do this exercise, all critical model parameters used to develop the BAFs for Lake Ontario will be used in the model framework of this assessment. The most critical parameter is the organic carbon partition coefficients, Koc, assumed for 2,3,7,8-TCDD. BAFs were determined assuming Koc of 10^7 and 10^8. Since the models of this assessment assume steady loading into water bodies (via erosion of soil from the watershed), only the BAFs developed under "steady state" loading conditions will be used. Two compartments onto which 2,3,7,8-TCDD sorbs in the water column were simulated using the WASP4 model - these include the suspended solids and what was termed the water column non-settling organic matter (NSOM). The concentration of suspended solids was 1.2 mg/L, and the concentration of the NSOM was 3 mg/L. The framework of this assessment has only one compartment termed suspended solids; the value given this term for this exercise will be 4.2 mg/L, the sum of concentrations of the two compartments of the WASP4 exercise. The other critical parameters are the fraction organic carbon contents of the suspended solids and the bottom sediments, OC_{ssed} and OC_{sed}, respectively. Assigned values to these parameters in the WASP4 exercise (and in this exercise) were 0.12 (12%) and 0.03 (3%), respectively. One final note is that there is no analogous NSOM compartment in this model. This compartment effectively removes contaminant from the water column - it cannot partition into the dissolved phase. EPA (1990a) estimated that the fraction of total water column 2,3,7,8-TCDD to be unavailable by this binding is 6.7% of total water column 2,3,7,8-TCDD. Because the model of this assessment does not have a similar binding compartment, it may generally tend to predict higher dissolved phase concentrations than the WASP4 model.

Since the purpose of this exercise is to evaluate how the modeling approaches of this document perform using the BSAF or the alternate BAF approach, it is not critical which source category is demonstrated. Any source category will estimate a given load to

a water body - the question is how the BSAF approach predicts fish tissue concentrations as compared to BAF approach, whatever load enters the water body. For simplicity, the on-site source category as demonstrated in Chapter 9 will be used. In this scenario, the soil within the watershed is assumed to uniformly be 1.0 ppt.

In summary, the parameters for this exercise including the BAFs are:

Test 1: Koc = 10,000,000; $ssBAF_l^d$ = 4,040,000; $ssBAF_l^t$ = 866,000; BSAF = 0.07

Test 2: Koc = 100,000,000; $ssBAF_l^d$ = 40,400,000; $ssBAF_l^t$ = 1,080,000; BSAF = 0.07

For both tests: soil concentration of 2,3,7,8-TCDD = 1.0 ng/kg (ppt), total suspended solids (TSS) = 4.2 mg/L, the organic carbon content of suspended sediments (OC_{ssed}) = 0.12, and the organic carbon content of bottom sediments (OC_{sed}) = 0.03

The whole fish tissue concentration (i.e., C_{lipid} multiplied by the fraction lipid, f_{lipid}, assumed to be 0.07) for the BSAF approach using Koc = 10^7 was estimated to be 0.03 ppt. Using the $ssBAF_l^t$ and $ssBAF_l^d$, the whole fish tissue concentrations were estimated to be 0.25 and 0.20 ppt, respectively. The test results for Koc = 10^8 were similar; the BSAF estimated fish tissue concentrations to be 0.04 ppt, and the $ssBAF_l^t$ and $ssBAF_{ld}$ predicted concentrations of 0.32 and 0.23 ppt.

While it appears that the water column based approaches estimate fish tissue concentrations within an order of magnitude higher than the sediment approach, it would not be appropriate to conclude that as a generalization. Rather, the predictions are reasonably close, given the uncertainties in the bioaccumulation and water modeling parameters. One important consideration in using the water column based approaches is that the BAFs developed by Cook (or that could be developed otherwise) are based on modeled rather than measured water column concentrations, and measured lake trout tissue concentrations. In that sense, the BAFs were calibrated for Lake Ontario conditions and specific to the WASP4 modeling exercise. The BSAF developed for lake trout for Lake Ontario was developed using measurements of both fish tissue and bottom sediment concentrations.

Both the BSAF and BAF are most appropriately developed using site specific data (coupled with a modeling exercise for BAF). Inasmuch as that can be impractical or difficult for many sites, efforts are underway to determine the general applicability of BSAFs and BAFs determined for one site to other sites. Cook (reference given at

beginning of this section) writes that BAF_ls for different congeners can be roughly estimated as the BAF_l for 2,3,7,8-TCDD multiplied by the ratio of the BSAF for the congener and the BSAF for 2,3,7,8-TCDD. Such an estimate will incorporate differences in uptake, metabolism and chemical partitioning but not differences caused by chemical loss processes such as volatilization and photolysis. This approach for estimating BAF_ls for other congeners does allow for some generality since sediment and fish tissue data for other congeners and water bodies is available. Cook also suggests that the $ssBAF_l^d$ may be the best predictor of 2,3,7,8-TCDD residues in other systems <u>if</u> (sic) the dissolved phase concentrations can be estimated accurately. A comprehensive discussion of the many issues involved in exposure and bioaccumulation will be available in the document cited at the beginning of this section.

10.2.8.4. *Sensitivity to Key Model Parameters*

Fish tissue concentrations are based on organic carbon normalized concentrations in sediments. Strictly speaking, they are a function of the organic carbon normalized concentration on bottom sediments. However, one key assumption made for the solution for bottom sediment concentrations is that the organic carbon normalized concentrations of bottom sediments and suspended solids are equal. Soil eroding into the water body from the watershed is assumed to equilibrate with the water in the water body while suspended. Therefore, since the organic carbon normalized concentrations of bottom sediments and suspended solids are assumed to be equal, fish tissue concentrations are a function of partitioning in the water column.

Three model parameters determine water column partitioning: the total suspended solids concentration, TSS, the organic carbon partition coefficient, Koc, and the fraction organic carbon fraction of suspended solids, OC_{ssed}. Sensitivity analysis exercises on these parameters individually showed a reasonably narrow range of predicted fish concentrations.

The value of Koc used in this assessment for 2,3,7,8-TCDD was 2.7×10^6. Cook (P. Cook, Duluth Environmental Research Laboratory, US EPA, Duluth, MN) suggests that the apparent Koc in aquatic systems may be higher at 10^7 or even 10^8. The 2,3,7,8-TCDD was increased to these two values and the predicted fish concentrations increased 46 and 72% respectively. Some literature places Koc for 2,3,7,8-TCDD lower at 10^6; this

value results in a 42% reduction from the value of Koc used. There is less than an order of magnitude difference in fish tissue concentration predictions between 10^6 and 10^8, indicating little sensitivity to this parameter.

The same trend of little sensitivity was noted for the other two parameters, TSS and OC_{ssed}. An earlier discussion on total suspended solids, TSS, indicated a reasonable range of 2 to 20 mg/L (see section 10.2.7.4); the value in this assessment was described as reasonable for a river setting at 10 mg/L. Changing the value to 2 and up to 20 resulted in a 63% decrease and a 28% increase in estimated fish tissue concentration. Like Koc, there was less than an order of magnitude difference between 2 and 20 mg/L estimates. The organic carbon content of suspended sediments, OC_{ssed}, should be higher than bottom sediment organic carbon content, OC_{sed}; OC_{ssed} was set at 0.05. It will rarely exceed 0.15, and setting it to this value decreased concentrations by 58%.

The reason that fish concentrations did not vary with changes in these parameters is that the Koc is very high to begin with. Changes to any of the three parameters noted above will only result in slightly more or slightly less 2,3,7,8-TCDD partitioning into the water column from suspended solids. More sensitivity would undoubtably be shown with this modeling framework and Koc below 10^5.

The two remaining parameters used to estimate fish tissue concentrations are the Biota Sediment Accumulation Factor, BSAF, and the lipid content of fish, f_{lipid}. Fish concentrations are linearly related to these two parameters, so there is a linear response to predictions with changes to these parameters.

The BSAF was assigned to the example compounds on the basis of literature values. Available values are summarized on Table 5.2 and discussed in Section 5.3.4.1. Available BSAFs for CDDs and CDFs from 3 of 4 studies were consistent and were in the 0.01 to 0.30 range, with some indications that BSAFs decreased with increasing chlorination. One study on lake and river contamination in Sweden, including data on sediment and lipid-based fish tissue concentrations for Pike, allowed for estimation of a BSAF for 2,3,7,8-TCDD to be 1.6 (see further discussion in Section 5.3.4.1. on this data point). Changes in BSAF lead to direct changes in fish tissue concentrations. The value selected for 2,3,7,8-TCDD and 2,3,4,7,8-PCDF was 0.09; increasing this value three-fold, which is near the maximum reported for all but this one data point, would increase fish tissue concentrations three-fold.

Table 5.2. and the discussion in Section 5.3.4.1. conclude that the data on BSAFs for PCBs are more variable. Data from a remote lake in Lake Superior (Swackhamer, et al., 1988) had no BSAFs less than 2.2 for lake trout and whitefish for penta- through octa-PCBs, with one BSAF estimated at 20.8. The one BSAF for hepta-PCBs from this study was 12.5. In contrast, data from New Bedford Harbor showed a lower range for hepta-PCBs at 0.84 to 2.74 for flounder, lobster and crab. Total PCB BSAFs for assorted Lake Ontario data, developed and summarized in EPA (1990a), showed a range of 0.58-4.06. The BSAF selected for 2,3,3',4,4',5,5'-PCB was 2.0. Increasing it by 5, which might mirror the highest found BSAFs for PCBs, would increase fish concentrations five-fold.

Fish lipid concentrations are also variable, even within the same fish species. EPA (1990a) found that fish from Lake Ontario had three times the lipid content as the same species of fish brought into the laboratory from Lake Ontario for microcosm study. A brief summary of some available lipid contents of fish given in Section 5.4.2 showed that f_{lipid} would not be much larger than 0.20. The default value assigned was 0.07. Whole fish concentrations, which are used in exposure estimates, would roughly triple if using 0.20 instead of 0.07 for f_{lipid}.

10.2.9. Vapor Phase Inhalation Exposures

This section will address the uncertainty associated with vapor phase inhalation exposures. Sources addressed in this assessment include incinerator stack emissions and contaminated soils; this section will only address contaminated soils. Uncertainties associated with vapor phase air concentrations resulting from stack emissions are discussed in Section 10.3.

Vapor-phase emissions are estimated with a volatilization flux algorithm. The procedures were developed in Hwang, et al. (1986). A near-field dispersion model estimates air concentrations for the on-site source category - the category addressing soil contamination at the site of exposure. For the off-site and ash landfill source categories, where the site of contamination is located distant from the site of exposure, the same volatilization flux model is used. Exposure site concentrations for these sources are estimated using a far-field dispersion model.

The respiration rate of 20 m^3/day used for inhalation exposures is within the standard range of 20-23 m^3/day. The contact fraction is 0.75 for central scenarios and

0.90 for high end scenarios. Like the water ingestion contact fractions, these were based on time at home surveys. The inhalation rate and contact fractions are not expected to introduce much uncertainty into inhalation exposure estimates. Another exposure parameter critical for the inhalation pathway is exposure durations, which is 9 years for central and 20 years for high end exposures. The uncertainties associated with this parameter are discussed above in Section 10.2.3. However, exposure duration is additionally critical for the inhalation pathway, as estimated volatilization flux is a function of the time during which volatilization is occurring; this is further explained below.

The first section below compares measured air concentrations to concentrations predicted in this assessment. The next section looks at alternate modeling approaches and considerations. The final section evaluates the impact and uncertainty of key factors that estimate vapor phase air concentrations. Uncertainty and model sensitivity issues are summarized in Table 10.9.

10.2.9.1. *Comparison of Model Results with Measured Concentrations*

The volatilization and near-field dispersion models were developed from well established theoretical principals, and were developed as part of an effort to assess the impact of soils contaminated with PCBs (Hwang, et al., 1986). The virtual point source dispersion model for far-field dispersion estimates is also based on well established theory (Turner, 1970). However, these models have not been field validated for soils contaminated with dioxin-like compounds. Ideally, the overall process would be validated using concentrations of dioxin-like compounds in soil and concurrently measured concentrations in air above and downwind of the contaminated soil.

However, some sense of the reasonableness of model values can be made by comparing predictions to measured concentrations in ambient air in urban environments. Reports of such concentrations are summarized in Section 3.7 (Chapter 3) and Table B.13 (Appendix B). Sources other than soil are likely to be the cause of levels measured in urban air environments; sources such as industrial and automobile emissions. Also, the dispersion model is designed for situations where the contaminated soil is surrounded by relatively clean soil. As discussed in Section 10.2.1. above, off-site impacts have been noted in several sites of 2,3,7,8-TCDD contamination. These two points are made in order to establish a basis for comparing urban air concentration to concentrations predicted to

Table 10.9. Uncertainties and sensitivities associated with estimating vapor-phase air concentrations from contaminated soils.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Site-specific physical parameters	porosity, wind speed areas of contamination, distance to receptor, frequencey wind blows to receptor; height of individual	All such parameters assigned reasonable midrange values.	With the exception of distance to receptor, all parameters show no more than 50% differences in estimated air concentrations with changes within a range of expected values;	Physical parameters do not appear to limit certainty of of estimations.
Chemical parameters	Organic carbon partition coefficient, Koc, Henry's Constant, H	Koc empirically estimated from Kow; H estimated from water solubility and vapor pressure	Field data implies Koc could be an order of magnitude higher than assumed in this assessment for 2,3,7,8-TCDD; H consistent with literature values; alternate values tend to decrease estimated air concentrations by up to 75%.	Consideration should be given to Koc selected for 2,3,7,8-TCDD; alternate value may be higher but not lower.
Hwang vs. Jury volatilization model	selected Hwang model solutions; Hwang model includes development of average flux term based on time; both	Both analytical comparable volatilization flux; Hwang model based on tentative assumption of an infinite reser- models require similar data; assumptions and boundary conditions different	Both models estimate be appropriate. voir; but analysis showed this to be tenable for short exposure durations	Either model would
Near field vs. box-model dispersion.	selected near-field dispersion model	required parameters are less uncertain for near-field model	Both models estimated comparable concentrations; box model showed particular sensitivity to area and height of mixing.	Selected model is preferable.
Volatilization or eroded contaminants	contaminants eroding to exposure site assumed not to volatilize and contribute to exposure site air concentrations	Uncertainty in depth of mixing at site of exposure implies that volatilization of delivered contaminated soil could an order of higher to an order of magnitude lower than volatilization originating at the site of contamination.	As noted, the key uncertainty is in the fate of delivered residues	No change in approach is warranted.

(continued on next page)

Table 10.9. (cont'd)

Evaluation: Comparison of model estimations of vapor-phase 2,3,7,8-TCDD and 2,3,4,7,8-PCDF with observed urban air concentrations of these contaminants supports but does not validate the approaches taken. Sensitivity analysis showed estimations to be relatively insensitive to physical parameters but sensitive to Koc and H. Data suggests that Koc may be underestimated with selected empirical approach. Alternate approaches estimate comparable air concentrations, which lends credibility (but not validity) to models. Approaches to estimate particulate phase concentrations are empirical and based on field data. Evaluation of parameters indicates that predictions could change by up to an order of magnitude with reasonable parameter changes.

occur from soils: one would expect urban air concentrations to be at least higher, if not higher by orders of magnitude, than soil emissions.

Tables B.13 and B.14 summarize average PCDD/PCDF congener-specific concentrations in urban air in the United States and in Europe. Results for two example compounds demonstrated in Chapter 9, 2,3,7,8-TCDD and 2,3,4,7,8-PCDF, are examined in this section. Observed concentrations of 2,3,7,8-TCDD were mostly non-detects with detection limits listed at 0.002-0.03 pg/m^3, with two occurrences at 0.01 and 0.05 pg/m^3. Concentrations of 2,3,4,7,8-PCDF were detected in all reported studies. The range of 2,3,4,7,8-PCDF concentrations was 0.01-1.03 pg/m^3. In the two reports where both compounds were detected, 2,3,4,7,8-PCDF was detected at 5 and 10 times higher concentration than 2,3,7,8-TCDD.

The on-site source category was demonstrated using concentrations of 1.0 ng/kg (ppt) for each example compound. This low concentration was assigned based on reports by researchers who measured concentrations of dioxin-like compounds in what they described as "background" and "rural" soils - they found non-detects to concentrations in the low ppt level. Modeled air concentrations of the example compounds 2,3,7,8-TCDD and 2,3,4,7,8-PCDF resulting from this level in soil were in the 10^{-5} pg/m^3 range, or over three orders of magnitude lower than observed urban concentrations (5 orders of magnitude lower than the high noted observation of 1 pg/m^3 2,3,4,7,8-PCDF). Two source categories, the off-site and ash landfill categories, evaluated the impact of elevated soil concentrations in areas that were located distant from the from the site of contamination. The example scenarios demonstrating these source categories had concentrations of this dioxin and furan congener in the ppb range, or three orders of magnitude higher than the on-site source category concentrations. Air concentrations

predicted in these example scenarios were in the 10^{-3} to 10^{-2} pg/m^3 range, which is generally the same order of magnitude as these urban findings.

As noted, this was not meant to be a validation exercise, merely an attempt to put the predicted concentrations into perspective. The fact that concentrations predicted to occur from very low soil concentrations are 3 orders of magnitude lower than measured urban concentrations supports the models approach. It is at least plausible that elevated concentrations in soil would result in air concentrations that are in the same range as found in urban environments. A model result that would have questioned the model validity would have been, for example, that air concentrations resulting from soils of very low concentrations would match urban air concentrations, and that concentrations resulting from soils with elevated concentrations exceed those found in urban environments.

10.2.9.2. *Other Modeling Approaches and Considerations for Vapor-Phase Emissions*

Volatilization flux was modeled using an approach given in Hwang, et al. (1986), developed for PCB flux from soils. Principal assumptions for their derivation were that contamination extended indefinitely, biodegradation or other degradation processes were not considered, residues were in equilibrium between soil and soil air, and vertical movement was through vapor phase diffusion. Their analytical solution was integrated over time and a solution was presented which gave average unit flux as a function of time during which volatilization occurs. PCBs and other dioxin-like compounds resist degradation, although there is evidence of photodegradation, which may influence surficial residues. These compounds sorb tightly to soil, so that an assumption of vertical movement primarily through vapor phase diffusion (rather than in a soluble phase with leaching, runoff, or evaporating water) is a tenable one. Also, presentation of an average flux rate solution made Hwang's approach amenable to spreadsheet analysis, the computer software tool used in this assessment.

An alternate model for estimating volatilization flux was presented in Jury, et al. (1983). It is a generalized analytical solution which assumes equilibrium between the sorbed, soluble, and vapor phases. It incorporates considerations of steady state water fluxes and degradation mechanisms. A depth over which contamination occurs is

specified. A computer code of this model was obtained from the author (William A. Jury, Professor and Chair, Department of Soil and Environmental Sciences, University of California, Riverside, 92521-0424). Tests were run holding all pertinent quantities the same with both models including initial concentrations, organic carbon partition coefficients, Henry's Constant, molecular diffusivity, fraction organic carbon in soil, soil bulk density, porosity, and an assumption of contaminant non-degradation. All of these parameters, the contaminant as well as the physical parameters, were the ones assumed for 2,3,7,8-TCDD and the surface soils of this assessment. In applying Jury's model, the depth of contamination was assumed to be 10 cm. Also, Jury's model allowed for a selection of water flux to be 0.5 cm/day (heavy leaching), -0.5 cm/day (heavy evapotranspiration), or 0.0 cm/day (no water flux). The latter selection of no water flux was chosen. This model comparison test showed that the Hwang model predicted an average flux over 10 years roughly three times higher than the average flux predicted by the Jury model over the same time period. Running both models over 50 years showed similar results. The average flux over that time dropped by about 50% for both models and there was still a three-fold difference in predicted volatilization fluxes. The exact reason for this three-fold difference was not investigated, and could lie in differences in assumed boundary conditions (Hwang, et al. (1986) discusses differences in boundary conditions between his and Jury's models). In any case, it is judged that both models predict comparable volatilization fluxes. The Hwang model might be considered conservative in that it predicts 3 times higher volatilization flux (with 2,3,7,8-TCDD parameters, etc.).

The Jury model also provides other informative results. It provides a mass balance which, for the 50-year test, showed that only 2.6% percent of the original mass within the 10-cm layer had volatilized. By implication, the Hwang model predicts a 7.3% loss by volatilization over that time period. With the other parameters and assumptions - no degradation and tight sorption to soil - the Jury model showed that 97.4% remained in the profile and that only a minute quantity diffused below 10 cm. Also, the Jury model gives a concentration profile over time. After 50 years, it showed that all volatilization loss was contained within the upper 2 cm of soil profile. This implies that the boundary condition assumption for the Hwang model, that contamination extends indefinitely, is not consequential for the dioxin-like compounds.

A near-field dispersion model is used to estimate air concentrations resulting from soil volatilization, for the on-site source category (where contamination and exposure occur at the same site). An alternate approach to estimating on-site dispersion given a volatilization flux is the "box-model" approach. This simple approach can be visualized as follows: air above soil is contained within a structure which has two walls, say a north and south wall, and a ceiling - wind blows through the building in an east-west direction mixing the volatilized flux. This is expressed mathematically (using similar parameter naming as in Equation (5-11), Section 5.3.2 in Chapter 5) as:

$$C_{va} = \frac{FLUX\ AREA\ 10^6}{b\ U_{mix}\ Z} \qquad (10\text{-}4)$$

where:

C_{va}	=	vapor-phase concentration of contaminant in air, $\mu g/m^3$
FLUX	=	average volatilization flux rate of contaminant from soil, g/cm^2-sec
AREA	=	area over which flux occurs, cm^2
b	=	side length perpendicular to wind direction, m
U_{mix}	=	mean annual wind speed corresponding to mixing zone height, m/sec; estimated as 1/2*Um, where Um is average wind speed
z	=	mixing zone height, m
10^6	=	converts g to μg

Before testing the box-model equation, results for the approach used in this assessment are summarized. The key factors impacting air concentration calculations in Scenarios 1 and 2 is the duration of exposure and area over which contamination occurs. In the central scenario, Scenario 1, the area was 4,000 m^2 (1 acre) and in the high end scenario, Scenario 3, the area was 40,000 m^2 (10 acres). The exposure duration was 9 years in Scenario 1 and 20 years in Scenario 2. The volatilization flux was different for both scenarios, but not because of area considerations, but because of exposure duration assumptions; the average flux of 2,3,7,8-TCDD for the high end scenario was 1.2×10^{-21} g/cm^2-sec, whereas the average flux for the central scenario was 1.7×10^{-21} g/cm^2-sec.

The air concentration estimated for both the central and high end scenario was the same at 4.4×10^{-11} $\mu g/m^3$. Larger areas tend to increase air concentration prediction; the larger area of the high end scenario countered the effect of having a lower average volatilization flux; hence similar air concentrations were predicted for the central and high end scenarios.

The values used to evaluate the box model approach were the fluxes, as given above, the mixing zone wind speed, 2 m/sec, which is half the average wind speed assumed in this assessment, the areas noted above, the side length, estimated as the square root of the area, and a mixing zone height estimated initially at 2 m. The box-model air concentration for the central scenario with these parameters is 2.7×10^{-10} $\mu g/m^3$. This is 6 times higher than the concentrations predicted in this assessment. The box-model concentration estimation for the high end scenario, given slightly lower flux as noted above and the larger land area, was 8.5×10^{-10} $\mu g/m^3$, which is over an order of magnitude higher than the concentration estimated for this assessment.

These box-model estimations are higher than the ones made for this assessment. An uncertain parameter for both modeling approaches is the area of soil contamination (see next section which discusses area considerations for the approach used in this assessment). The mixing zone height for the box model is also a parameter of uncertainty. Users of the box model approach have often assumed a conservative 2 m height approximating the height of exposed individuals. However, others have claimed this is far too low a mixing height, suggesting 10 meters or even an atmospheric height closer to 100 meters. Higher mixing zone heights would have brought the box model estimations more in line with estimations made in this assessment. The closest analogous parameter in the dispersion model to the mixing zone height is the height of exposed individual, which is more unambiguously the breathing zone height of 2 m.

One key assumption concerning the exposure site air concentrations resulting from an off-site area of soil contamination should be questioned. The current approach assumes that air-borne contaminates originate at the site of contamination and are transported to the site of exposure. On the other hand, this assessment also assumes that exposure site soil becomes contaminated over time due to erosion. Also, some of the example scenarios have tested the impact of very low, perhaps "background", levels of dioxin-like compounds, which would occur surrounding a site of exposure. It is at least

plausible that volatilization from soils other than the area of elevated contamination would contribute to air-borne contamination, and concentrations to which individuals are exposed to.

This was tested by using the on-site algorithms starting with the concentration estimated to occur at exposure sites resulting from erosion from the nearby off-site contaminated area. Two exposure site concentrations are considered: one, the untilled or higher concentration assumed to occur in a 1-cm soil depth at the exposure site, and a lower tilled concentration relevant for a 20-cm depth. The air concentration estimated to occur from untilled soil is nearly an order of magnitude higher than that estimated to occur from the off-site area and transported; the air concentration estimated to occur from tilled soil is nearly an order of magnitude lower. If the untilled depth were estimated at 5-cm instead of 1-cm, resulting air concentrations would be similar to concentrations estimated to originate from the off-site location.

This might imply that exposure site air concentrations are being underestimated if air concentrations at the site of exposure are assumed to only originate at the site of contamination, and not also at the site of exposure, or even from other areas. This implication, as discussed above, is dependent on the assumption made for a depth of mixing for transported residues in untilled situations. If the exposure site were a farm, which might be dominated by tilled soil (distributing residues to a 20 cm depth, or thereabouts), than this exercise implies that exposure site air concentrations might be dominated by residues which have been transported from a nearby site of contamination. If the exposure site and surrounding land is residential, which might not be dominated by tilled soil, than the procedures might be estimating air concentrations up to an order of magnitude too low.

10.2.9.4. *Uncertainty of Key Model Parameters*

The two chemical parameters which impact air concentrations are the soil organic carbon partition coefficient, Koc, and the Henry's Constant, H. Section 10.2.1 above indicated that Koc for 2,3,7,8-TCDD might be as high as 24,500,000, although it appears unlikely to be much lower than the value assumed in this assessment, 2,700,000. Increasing Koc for 2,3,7,8-TCDD to this higher value would decrease exposure site concentrations by approximately 70% for the various scenarios. Section 10.2.1. above

also notes the H values were in the 10^{-6} to 10^{-5} atm-m^3/mol range for PCDDs and PCDFs. The value selected for 2,3,7,8-TCDD was 1.65 x 10^{-5} atm-m^3/mol. Reducing this value to 1x10^{-6} atm-m^3/mol reduces air concentrations by approximately 75% over the various scenarios. While the Koc and H assignments to 2,3,7,8-TCDD are evaluated as appropriate, alternate selections would tend to decrease predicted air concentrations, perhaps by as much as 75%.

Physical parameters associated with estimating volatilization flux are soil porosity and soil organic carbon content. The value used for soil porosity was 0.50. Porosities for sandy surface soils show a range of 0.35-0.50. Medium to fine-textured soils (loams, clays, etc.) show a range of 0.40-0.60 (Brady, 1984). Varying porosity between 0.3 and 0.6 showed little impact to model results. Reducing the porosity to 0.30 only reduced estimated air concentrations by 17%; increasing to 0.60 increased air concentrations by 1.3%. The soil organic carbon content, OC_{sl}, was 0.01. This might be low and more typical of sandy soils rather than soils with high silt or clay content. Most soils would not have organic carbon contents exceeding 0.05. Increasing the OC_{sl} to 0.05 decreases air concentrations by approximately 55%; increasing OC_{sl} to 0.03 decreases concentrations by 42%.

Physical parameters associated with dispersion modeling for the on-site source category include wind speed, area of contamination, and height of exposed individual. None of these parameters have a significant impact on estimating air concentrations. The wind speed was set at 4.0 m/s, which is mid-range between observed windspeeds around the United States of 2.8 to 6.0 m/s. Decreasing the windspeed to 2.8 m/sec increased air concentrations by 22%, increasing the wind speed to 6.0 m/s decreased air concentrations by 35%. The area was set at 4,000 and 40,000 m^2 for the on-site central example scenario, #1, and the on-site high end scenario, #2. Increasing these areas by a factor of ten increases air concentrations 39 (Scenario #1) and 48% (#2). Decreasing them by a factor of ten reduces air concentrations 22 (#1) and 30% (#2). For both Scenarios #1 and #2, soil concentrations were set at very low levels which researchers described as "background" or "rural". The concept for example Scenarios 1 and 2 was that there may be a low level extensive concentration typical of large areas, such as a background level. If so, than much more area than just the area of the exposure site would be characterized by this low concentrations. Given this framework, the exposure sites should probably be

considered small land areas within larger areas that contribute to emissions impacting exposed individuals. The model's estimations, which limited land area, may be underpredictions, although more appropriate predictions might only be 30-40% higher than model results (i.e., a 10-fold larger land area impacting the breathing areas of exposed individuals is probably too large). Finally, the height of the exposed individual was set at 2.0 meters. The dispersion model includes an exponential term which results in a logarithmic decrease in estimated air concentration as the height parameter increases. The height term is not expected to be variable, and the estimate of 2 meters as the breathing zone height is a reasonable parameter assignment. Regardless, changing this parameter to 1 or 3 m only results in a 5% increase (at 1 m) or decrease (at 3 m) in estimated air concentrations.

Physical parameters associated with dispersion modeling for the source categories where soil contamination is distant from the site of exposure, the off-site and ash landfill source categories, include the frequency wind blows from source to receptor and the distance from source to receptor. The 15% frequency used generally assumes wind blows in all directions equally. Assuming 30% roughly doubles exposure site air concentrations (100% increase), and reducing frequency to 5% reduces air concentrations by 67%. These are more likely for real world settings, where wind normally blows in a prevailing direction, so it may be concluded that different frequencies for specific sites would change predictions by less than 50%. The distance from the sites of soil contamination and the exposure sites was 150 meters for the two source categories. Increasing this distance to 500 meters decreases concentrations by 66%, increasing it to 1000 meters results in nearly an order of magnitude reduction (85% reduction), and increasing the distance to 1500 meters results in an more than an order of magnitude reduction in air concentrations. At 15,000 meters, air concentrations are three orders of magnitude lower than at 150 meters. The virtual point source model used to calculate far field dispersion is not considered reliable for small distances, i.e., less than 100 meters. It is more appropriate to use the near field dispersion model used to estimate on-site air concentrations if an exposed individual is very near a site of contaminated soil. A test was run using the on-site source near-field dispersion model and the concentration (1 ppb) assumed for the off-site example scenario. The air concentration predicted was one order of magnitude higher than calculated for the exposure site 150 meters from the contaminated site.

Another parameter which impacts air concentrations is the duration of exposure, although of course this is not a physical parameter. It is seen in Equation (5-18) that volatilization flux is a function of the inverse square root of the exposure duration (in seconds) - the longer the duration of exposure, the lower will be estimated average flux rate (see Section 5.3.2. for further explanation). The impact of the exposure duration assumption is small, however. Starting with the 20-year duration assumed for high end scenarios, average air concentrations are doubled when duration is decreased to 5 years, and reduced just over one-half when duration is increased to 70 years.

In summary, this section has evaluated the sensitivity of physical and chemical parameters, and the exposure duration, to estimates of air concentrations. It is important to note that all physical parameters discussed above can be known with certainty for a specific site. The same cannot be said for the chemical parameters, Koc and H. As such, these chemical parameters might be judged as the most uncertain. Possible alternate values for these two parameters might decrease air concentrations by as much as 75%. Most physical parameters, when changed within a range of reasonable values, changed concentrations by 50% or less. Parameters with this small an impact include soil porosity, wind speed, area of contamination, height of the exposed individual, frequency wind blows from contaminated source to exposed individual, and exposure duration. Distance from a contaminated site to exposed individuals could have more of a pronounced impact, with differences orders of impact or more with similar large changes in distance from the 150 meters assumed for the example scenarios in Chapter 9.

10.2.10. Particulate Phase Inhalation

This section will summarize the uncertainties associated with estimating particulate-phase inhalation exposures. All exposure parameters for this pathway, including the inhalation rate, contact fractions, and exposure durations, were discussed in the previous section on vapor-phase inhalation exposures. The same discussion is pertinent to this pathway, and will not be further discussed. This section will focus on the algorithms estimating particulate-phase concentrations at the site of exposure.

Dust emissions from contaminated soil occur via wind erosion. An empirical equation estimates the emission due to wind erosion. Other sources of emissions estimated for the ash landfill source category include: suspension of contaminated

roadway dust due to vehicular traffic, fugitive emissions from trucks, emissions from truck unloading, and emissions from spreading and compacting ash at the landfill. The near- and far-field dispersion models as described above are used to estimate dispersion and transport for dust emissions; their uncertainties are discussed above in the section on vapor-phase inhalation exposure. No data was found on air-borne particle-phase concentrations of dioxin-like compounds originating from soil contamination. This section focuses only on sensitivity and uncertainty of model parameters.

A summary of uncertainty issues with this pathway is given in Table 10.10.

10.2.10.1 *Wind Erosion Modeling*

The empirical relationship used to estimate wind erosion assumes that the soil surface is uncrusted and composed of finely divided particulates, a situation where dust emissions will be maximized. The procedures are presented in EPA (1985) and were developed using field measurements presented in Gillette (1981) for highly erodible soils. It is presumed that estimations made with these procedures would correlate with data in Gillette (1981); other means to validate the approach were not presented in EPA (1985), and additional validation is not attempted here.

The model does show some sensitivity to required parameters. The wind speed generally varies from 2.8 to 6.0 m/sec, with 4.0 m/sec used in model estimations. Reducing this speed to 2.8 m/sec reduces air-borne concentrations by about one-half, while increasing wind speed to 6.0 m/sec roughly doubles air-borne concentrations. Other parameters include the fraction of ground covered by vegetation, the threshold wind velocity, and the model specific function, F(x). These are all related, as assumptions of ground cover influence all three. The example scenarios include three sets of assumptions demonstrate the range of predictions with different ground cover assumptions. Scenario 1, the 1-acre residential scenario, assumes grass cover and a 90% ground cover assumption, Scenario 2, the 10-acre farm scenario, assumes 50% ground cover, and Scenario 3, the off-site 10-acre contaminated site, is assumed to be bare with 0% ground cover. There is a 5-fold difference in wind erosion flux from the set of assumptions leading to the most wind erosion flux, those for the farm with 50% cover, to the set of assumptions leading to the least wind erosion, those for the 1-acre residence with 90% ground cover. Interestingly, the wind erosion flux from the off-site area with bare soil

Table 10.10. Uncertainties and sensitivities associated with estimating particulate-phase air concentrations from contaminated soils.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Exposure parameters	See Table 10.7. and Section 10.2.9.1. for discussion of inhalation rate and contact fraction			
Wind Erosion	Empirical equation assumes uncrusted soil surface and finely divided particulates	Equation based on field data and given assumptions, would tend to maximize predictions	Data was unavailable for comparison with model results; approach assumed reasonable since it was based on field data; changes in parameters could lead to as much as order of magnitude change in predictions as compared to estimations in this assessment.	Particulate phase exposures are 2-3 orders of magnitude lower than vapor phase exposures; given estimation sensitivity within an order of magnitude, uncertainty is less an issue for particulate phase concentrations
Fugitive Ash Emissions	Used AP-42 emission factor equations EPA (1988)	Like wind erosion, these are empirical equations based on data	Same comment as made for wind erosion applies here	Same comment as made for wind erosion

Overall: Sensitivity and uncertainty do not appear to be critical issues for particulate phase concentration estimation as exposures from contaminants on particulates are 2 to 3 orders of magnitude lower than exposure to vapor-phase contaminant concentrations, and lower still compared to all other exposures.

contamination was half as much as from the farm with 50% ground cover. This is counterintuitive and would question the procedure.

In summary, the validity and hence certainty of the approach to estimate wind erosion flux cannot be fully evaluated, although it was developed from field data. The underlying assumption that the soil surface is uncrusted and composed of fine particulates is a maximizing assumption. In Chapter 9 demonstrating the methodologies, it was observed that particulate phase exposures were 2 to 3 orders of magnitude lower than vapor-phase exposures, and were lower than all exposures except water and fish exposures from a stream setting. The fact that the approach would tend to maximize dust emissions and yet particulate phase exposures were still significantly lower than vapor phase exposures tends to minimize the need for certainty in this algorithm. Air

concentrations were relatively insensitive to changes in required parameters: alternate parameter selections from those selected for demonstration in Chapter 9 resulted in less than an order of magnitude difference.

10.2.10.2 *Fugitive Ash Emissions*

Fugitive emissions of contaminated ash occur due to ash handling and management at the landfill site. Emission factor equations for these processes are known as AP-42 emission factors and are described in EPA (1988). These equations are empirical and are developed from data. Like the wind erosion algorithm, further validation of these equations was not undertaken for this assessment.

There are numerous parameters and assumptions necessary to apply these equations. All such selections made for the demonstration in Chapter 9 considered the following: 1) particle size multipliers appropriate to estimate emissions of inhalable size particulates, $< 10\ \mu m$, 2) survey information on ash landfill operations described in MRI (1990), and collected with the intent of applying similar AP-42 emission factor equations, 3) mid-range values for AP-42 parameters such as number of truck wheels, number of wheels, number of days with rainfall, and so on, and 4) control efficiencies of 90% (only 10% of potential emissions occur) associated with 2 of 4 sources of emissions for which controls are common and expected - these include emissions resulting from vehicular traffic over unpaved roadways (chemical suppression or roadway wetting) and emissions off trucks in transit (wetting or tarpaulin). The observation was made in Chapter 9 that fugitive ash emissions were equal to the emissions that result from wind erosion of the ash landfill surface.

An in-depth analysis of the several parameters and assumptions is not warranted. They are all site-specific, and guidance to select parameter values is provided in EPA (1988). However, it is easily shown that estimates of fugitive emissions can be larger if a set of plausible alternate assumptions are made. The following are made in a single demonstration: 1) 90% control efficiencies will be reduced to 30% indicating some but little control of emissions, 2) 60 instead of 121 days of rainfall (not atypical of western United States), 3) larger (more wheels) and heavier trucks involved in ash delivery, and 4) slightly windier at 5 m/s instead of 4 m/sec. No assumptions on increased deliveries of ash were made since the 2700 TPD incinerator is relatively large to start with.

This exercise showed wind erosion doubled because of the increase in wind speed, and fugitive emissions increased by an order of magnitude with the several changes. Particulate phase air concentrations at the site of exposure 150 meters away only increased by a factor of five. Examination of the results also showed that emissions off trucks, from unloading of trucks, and from spreading of ash onto the landfill were 4-6 orders of magnitude lower than emissions resulting from vehicular traffic over unpaved roads and wind erosion. Any changes to parameters with those less significant contributors is irrelevant.

As noted above in the wind erosion section, exposures due to particulate phase emissions are 2 to 3 orders of magnitude lower than vapor-phase exposures, and lower than nearly all other exposures. This observation was made including the estimations from the ash landfill. While there are several parameters and assumptions required for fugitive ash emissions, the above analysis showed that only emissions from vehicular traffic over unpaved roadways is significant. It also showed that with a set of plausible assumptions that would lead to more particulate emissions, exposure site particulate phase concentrations would only increased by a factor of five.

10.2.11. Uncertainty in Fruit and Vegetable Ingestion

This section will evaluate the algorithms used to estimate concentrations of dioxin-like compounds in vegetation, including fruits and vegetables as well as pasture grass and fodder. The latter two vegetations are used in the food chain algorithm estimating concentrations in beef and milk. Discussions here are therefore pertinent to the next section below.

The vegetation algorithm separates above and below ground vegetations. Below ground vegetation includes vegetables such as potatoes, carrots, or onions. Above ground vegetation includes other fruits and vegetables, as well as pasture grass and fodder. Below ground vegetation concentration are estimated solely on the basis of soil to root transfers. Above ground vegetations are estimated based on vapor phase transfers and particulate depositions.

The first section below discusses the rates of fruit and vegetable ingestion. Following that is an indepth evaluation of the algorithms estimating vegetation concentrations. First, studies measuring soil and plant concentrations are evaluated and

compared to model behavior. Then, several issues of uncertainty are discussed, including assumptions and model parameters. A summary of uncertainty issues associated with the fruit and vegetable ingestion pathway is given in Table 10.11.

10.2.11.1 *Fruit and Vegetable Ingestion Rates*

Consumption rates of 200 g/day for vegetables and 140 g/day for fruit were derived in EPA (1989) and recommended for general assessment purposes. They include all fruits and vegetables and were derived from two principal sources: Foods Commonly Eaten by Individuals: Amount Per Day and Per Eating Occasion (Pao, et al. 1982), and 2) Food Consumption: Households in the United States, Seasons and Year 1977-1978 (USDA, 1983). Pao, et al. (1982) used the data from the USDA survey, which included interview responses from 37,874 individuals, to estimate total consumption and percentiles of home-grown fruits and vegetables. EPA (1989) identifies two principal sources of uncertainty with Pao's estimates:

- These data are from all consumers, only a small percentage of whom are also home gardeners. Those who home garden may have higher total rates of consumption.
- USDA's survey only included information for 3 days from each respondent: products eaten the day before, the day of, and the day after the interview. Therefore, the results on a per day basis only include information for three days from respondents; what is required for long term exposure assessments is an amount eaten per average day over the course of a long time period, such as a year or a duration of exposure. EPA (1989) did not discuss whether this aspect of uncertainty might render the 200 and 140 g/day estimates over- or underestimates.

These total consumption rates were reduced considering fruit and vegetables which are "protected" and "unprotected". Protected fruit, for example, included citrus and cantaloupe, whereas unprotected fruit included peaches or apples. This distinction was made because evidence indicates very little translocation or residues to within the plant. It was assumed that there would be no exposure when the produce was protected. Again using data from Pao, et al. (1982) as summarized in EPA (1989), it was estimated that 44% of total fruit ingestion was ingestion of unprotected fruit and 74% of total vegetable ingestion was unprotected vegetables.

Table 10.11. Uncertainties associated with vegetable and fruit ingestion exposure algorithms.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Rates of ingestion	fruits: 88 g/day above ground unprotected, 0 g/day below ground unp., 0.20-0.30 home grown; veg: 76 g/day abv grd unp. 28 g/day bel. grd. unp.; 0.25-0.40 home grown.	Protected/unprotected distinction because residues not expected to tranlocate; above/below ground because procedures for soil transfers are different	Much variability expected when using approach for a specific site when actual home gardening can be ascertained.	All parameters assumed are evaluated as reasonable for general exposure to broad categories of fruits and vegetables.
Below ground vegetable concentration	Uses empirical Root Concentration Factor which is function of Kow, VG_{bg}, an empirical correction factor, and soil water concentrations	Separating below ground with above ground vegeta- was critical and supported by the literature; approach based on laboratory experiments with barley roots.	The VG_{bg} empirically describes the difference in barley roots and bulky underground vegetables; although assignment of 0.01 is rationally based, arguments presented could estimate it instead at 0.10 or 0.001; algorithm also a function of Kow, which is uncertain by 2 orders of magnitude; most likely change in Kow decreases concentration by up to an order of magnitude	Comparison with the literature indicates estimates of below ground vegetables may be low by an order of magnitude; however, the trend that below ground vegetables have higher transfers from soil to plant as compared to above ground vegetables was correctly captured.
Air-to-leaf Transfer for Above Ground Vegetation	Uses air-to-leaf factor developed in laboratory conditions for 10 compounds transferred to azalea leaves; empirically corrects for plants like fruit/veg. that have much less transfer to inner parts as compared to leaves	Sound data on which to develop the empirical transfer factor, which is also presented as a function of Kow and H (Bacci, et al., 1990) air to plant transfer algorithm is critical as researchers believe this process to dominate in real world settings for dioxin-like compounds	Empirical correction factor is necessary, but values selected could also be low or high, as above; like algorithm above, transfers critically a function of Kow; most likely alternate Kow would increase concentration estimates by up to an order of magnitude	Like root uptake, this algorithm should be considered for future revisions. The theoretical basis for vapor phase transfers to plants is sound, and the transfer factor is based on sound data. A refined correction factor might consider photodegradation of residues on plant surfaces. surfaces.

(continued on next page)

Table 10.11. (cont'd)

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Particulate Phase Deposition for Above Ground Vegetation	Model for soil contamination solves for airborne particulate phase concentration and applies deposition rate; ISC model estimates deposition rate for stack emission source source category; plant concentrations based on deposition rates, plant mixing volumes, canopy cover and washoff	Algorithm developed for radionuclide impact to agriculture (Baes, et al., 1984) but applied to other contaminants sorbed to airborne particulates; particulate deposition as mechanism of plant contamination speculated to be of concern for 2,3,7,8-TCDD in early literature. Model results show particulate deposition critical for stack emissions, less critical for soil contamination.	Fugitive dust emission algorithms (wind erosion, ash handling) developed from data but not field validated; ISC modeling theory well founded; particulate depositions underestimated, but higher depositions, at most, will double fruit/vegetable concentrations while not impacting grass and fodder	The basic approach given contaminant deposition rates is defensible; model results imply particulate deposition is insignificant for soil contamination but significant for stack emissions of particulates.

Overall: All ingestion parameters assumed are evaluated as reasonable for general exposure to broad categories of fruits and vegetables. However, great variability is expected if using these procedures on a specific site where home gardening practices can be more precisely ascertained. A contaminant concentration ratio was defined as the concentration of contaminant in vegetation divided by the concentration in the soil. A comparison of the modeled ratios with those found in the literature showed that, whereas the modeled ratios tended to be lower for all vegetation (above and below ground fruit and vegetation, grass and cattle fodder) by about an order of magnitude, important trends noted in the literature were duplicated by the model. These are higher below ground vegetable ratios as compared to above ground vegetable ratios, and higher ratios for perennials (grasses, e.g.) as compared to vegetables. A key assumption in the vegetation algorithm, that dioxin-like compounds to not translocate from root to shoot, was verified by two experiments. Vapor-phase contributions to vegetation dominated the contaminated soil source categories - a minor contribution was made by particulate deposition. This assessment likely underestimated vegetative concentrations attributable to particulate phase concentrations, although increasing those contributions by an order of magnitude only as much as doubled vegetable concentrations, while still not impacting grass of fodder concentrations. A critical empirical parameter was the above and below ground correction factors, VG_{ag} and VG_{bg}, both set at 0.01. A different assumption for its derivation might set it at 0.10, which would increase estimated concentrations and perhaps make concentration ratios more in line with literature values. Other experimentally derived empirical factors describing the transfer of compounds from soil to below ground vegetables and air to above ground vegetation were a function of contaminant Kow. An alternate value of log Kow for 2,3,7,8-TCDD would more likely be higher than lower, given a literature range of 6.15 to 8.5, and a selected value of 6.64. Increasing log Kow tends to decrease below ground vegetation, by as much as an order of magnitude, while increasing above ground vegetation by as much as an order of magnitude.

A final distinction was required which divided unprotected fruit/vegetables to those which grow underground and those which grow above ground. Different algorithms were used to transfer soil residues to plants depending on whether they were above or below ground. Using the same data once again, it was estimated that no fruits were grown underground (unprotected or protected), and that 37% of unprotected vegetables were grown underground.

The result of these two distinctions was to estimate total consumptions rates of unprotected fruit as no below ground and 88 g/day above ground consumption; for

unprotected vegetables, total consumption included 76 g/day above ground and 28 g/day below ground.

The overall average fraction of total vegetable and fruit consumption which is homegrown is estimated as 0.25 and 0.20, respectively (EPA, 1989). EPA (1989) recommends 90th percentile assumptions for these parameters of 0.40 (vegetables) and 0.30 (fruit), which were assumed in the high end scenarios of this assessment. EPA (1989) notes a wide range of fraction homegrown for individual vegetables, 0.04-0.75, and fruits, 0.09-0.33.

All these assumptions discussed: total consumption rates, protected or unprotected, above or below ground, and fraction home grown, are probably reasonable for general assessment purposes as long as exposures are to the broad categories of fruits or vegetables, and not for individual fruits or vegetables. For a site specific assessment, there will likely be wide variability on the types of produce grown at home, what percentage of that is unprotected, and so on. Finally, and as is also true for beef and milk exposures, this assessment only considers the impact of home-grown fruits and vegetables. In rural settings, it is plausible that a large percentage of an individual's total fruit and vegetable intake comes from nearby and impacted sources, more than the 20-40% assumed in this assessment. If that is the case, than contact fractions should be set at 1.0, and exposures would increase 2-5 times from what they are estimated as in this assessment.

10.2.11.2. *Comparing Predicted Versus Measured Plant Concentrations*

There have been several studies which have measured plant concentrations for plants grown in soils with known concentrations of 2,3,7,8-TCDD. One quantity that can be estimated from these studies is a *plant:soil contaminant concentration ratio*. This quantity is analogous to the contaminant concentration ratio defined for evaluating the erosion algorithm (see Section 10.2.3 above), and equals the concentration in the plant divided by the concentration in soil in which the plant is growing. Contaminant concentration ratios predicted to have occurred can be compared against those that have been measured in the various studies. It is important to note that this is not a validation exercise. None of the experimental or field conditions for the literature studies were duplicated in this exercise. Comparing contaminant concentration ratios only gives a sense

of how literature and modeled transfers from soil to plant compare. The literature articles measuring soil and resulting plant concentrations of 2,3,7,8-TCDD are summarized in Table 10.12. This table also includes contaminant concentration ratios.

What is common about all these studies is that the soil in which the plants were grown likely represents the source of 2,3,7,8-TCDD. It seems logical, therefore, that results from these studies can be compared with model estimations where soil in which the plants grow is the ultimate source. This is only true for the "on-site" source category. For that category, the soil is contaminated to an unspecified depth with dioxin-like compounds. That soil provides the source for underground plant to root transfers and above ground volatilization and dust emissions for air-to-leaf transfers and particulate depositions. The same is not true for the other source categories, where the source of vapor phase and particulate phase contaminants is either distant incinerator stacks or areas of contaminated soil. Because the stack emission, off-site, and ash landfill source to be distinctly different than the studies summarized in Table 10.12, only the contaminant concentration ratios from the on-site source category will be examined.

The following contaminant concentration ratios were estimated for the two scenarios demonstrating the on-site source category in Chapter 9, Scenarios 1 and 2: below ground vegetables - 1.45×10^{-3}, above ground vegetables/fruit - 3.6×10^{-5}, grass - 2.4×10^{-2}, and fodder 1.2×10^{-2}. Some observations from contaminant concentration ratios found in the literature and these estimated by the model are:

1) The contaminant concentration ratios for above and below ground fruits/vegetables, 10^{-5} and 10^{-3} respectively, are about an order of magnitude lower than found by Wipf, et al. (1982) for experiments conducted outdoors and indoors, on fruits and vegetables, using Seveso contaminated soils. Wipf showed ratios for above ground fruit grown outdoors to be in the range of 0.0009 to 0.0042, which is higher than the model's 10^{-5} above ground ratio. Wipf found no residues at 1 ppb detection limit in the cobs and kernels of corn, although a small amount in the sheaths of corn, leading to a ratio of 0.0 for the edible part of corn and 0.0008 for the sheath. He showed (washed and unwashed) contaminant ratios for carrots in the range of 0.01 to 0.17. These ratios are higher than the 10^{-3} ratio estimated by the model. Wipf's underground ratio range, 0.01-0.17, was two orders of magnitude higher than they found for above ground fruits and vegetables, 0.0009-0.0042. This two order of magnitude difference is similar to the

Table 10.12. Summary of plant concentration vs. soil concentration data for 2,3,7,8-TCDD.

Plant/Soil Concentrations	Contaminant Ratio	Reference and Comments
(9-42 ppt)/ 10 ppb	.0009-.0042	Wipf, et al., 1982; analysis of apples, pears, plums, figs, peaches, and apricots grown in Seveso, Italy year following contamination; apples, pears, and peaches showed >95% of whole fruit concentrations listed here was in the peels; analysis of vegetative samples in less contaminated areas showed non-detections at 1 ppt detection limit; reference was unclear as to whether reported concentrations in fruit was based on fresh or dry weight.
(8-9 ppt)/ 10 ppb	.0008	Wipf, et al., 1982; concentrations listed were those found in sheaths of corn grown year following Seveso contamination; none found in cobs and kernels at 1 ppt detection limit.
(54-167 ppt)/ (1-5 ppb)	.01-.17	Wipf, et al., 1982; results are for greenhouse carrots grown in Seveso contaminated soil; the 54 ppt concentration listed was for carrot peels and inner portions; the 167 ppt listed includes the 54 ppt plus additional residues found in wash water and can be described as "unwashed" concentration; 96% of 167 ppt unwashed concentration includes that found in wash water (67%) and peels (29%).
(0.8-9.2 ppb)/ (2.7-8.3 ppb)	.24-1.73	Coccusi, et al., 1979; results are for carrots, potatoes, narcissus, and onions grown on contaminated soil the spring following the Seveso contamination; aerial plant part ratios were 0.25-0.40 - underground part ratios were 0.23-1.73; residues in contaminated plants were found to dissipate when contaminated plants transplanted to unpolluted soils; results contradict those from Seveso in Reference 1 noted above; results were expressed in fresh plant weight and fresh soil basis.

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Table 10.12. (cont'd)

Plant/Soil Concentrations	Contaminant Ratio	Reference and Comments
(1-63 ppt)/ (12-3300 ppt)	0.003-0.35	Sacchi, et al., 1986; data was for: "aerial parts" (sic) of bean and maize plants, tritiated TCDD amended soil with concentrations ranging as noted, taken at different intervals including 7, 34 and 57 days (one test), 17, 34, and 57 days (another test), 8 and 77 days, and 8 and 49 days, and in tests where soil was and was not amended with peat. Results showed increasing plant concentrations with increasing soil concentrations, but the ratio of plant to soil concentrations was inversely related to increasing soil concentrations (lowest ratios at highest soil concentrations). Soils without peat had higher ratios than soils with peat. Plant concentrations were on a fresh weight basis.
(0.2-4.2 ppt)/ (2-6000 ppt)	.00008-.3	Hulster, A. 1991; potatoes (shoot and tuber results) and lettuce grown outside on soil and in indoor potted plants PCDD and PCDF concentrations in soil ranging from 2.4 to 6071 ng/kg (ppt). Plant:soil ratio trend followed similar trend as of Sacchi above - highest ratios were at the lowest soil concentrations. Plant concentrations given in dry weight basis.
ND (DL=1 ppb)/ 60 ppb	<0.017	Isensee and Jones, 1971; results are for mature oat and soybean tops, and oat grain and the bean of soybean, in soil treated with [^{14}C]TCDD to achieve a concentration of 60 ppb - no residues of TCDD were found; ratios of 0.14 and 0.28 were found for 2,4,-dichlorophenol (DCP) in oat and soybean tops, and 0.20 for 2,7-dichlorodibenzo-p-dioxin (DCDD) in oat tops; trace amounts of DCP and DCDD were found in the bean of soybean.
(10-760 ppt)/ 411 ppt	.02-1.85	Young, 1983; data was perennial grasses and broadleaf plants grown on soils which had received 9 years of 2,4,5-T herbicide applications - the herbicides were contaminated with dioxin - 8 years prior to the plant studies and at the time of the study, had known 2,3,7,8-TCDD soil concentrations at approximately 400 ppt to a total depth of 15 cm over the test area. Plant tops (culm, leaves, and stems) had concentrations 10-75 ppt, or ratios of .02-.18. Plant crown (portion just above the soil surface) had a concentration of 270 ppt, or a ratio of 0.66. Plant roots had concentrations of 710-760, or a ratio greater than 1.7. It was not known whether concentrations are on a fresh or dry weight basis.

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difference in modeled contaminant concentration ratios. This trend of estimating higher ratios for below ground vegetation as compared to above ground vegetation was noted in several of the studies summarized in Table 10.12.

Wipf's contaminant concentration ratios, however, are much different than those of Coccusi, et al. (1979) under similar experimental conditions. Coccusi's ratios ranged from 0.24 to 1.73 for several vegetables. Hulster (1991) tested different soil concentrations of TE (toxic equivalents of 2,3,7,8-TCDD) ranging from 2 to 6000 ppt in potted and outdoor experiments including potatoes and lettuce. Contaminant concentration ratios were lowest at the highest concentrations in the 10^{-5} to 10^{-4} range, consistent with this assessment, but were in the 10^{-2} to 10^{-1} for the lowest concentrations, again much higher than the ratios in this assessment.

2) The trend in Hulster's data of having higher ratios at lower soil concentrations was also found in Sacchi (1986). This trend will not be duplicated with the modeling framework of this assessment, unless the user changes model parameters with changes in soil concentration. It is unclear why such a trend would occur in experimental conditions. One possible explanation for below ground vegetation is that binding sites in the vegetation become limiting when the supply of available contaminant to bind increases. For above ground vegetation, binding site limitations with increases in air concentration may also be the case, or volatilization from soil may be unhindered at low concentrations but inhibited at high concentrations.

3) Intuitively, perennial plants grown in soil contaminated with the dioxin-like compounds ought to have higher concentrations compared to annual agronomic crops grown in similar conditions. These compounds resist degradation in soils, and although they are likely to be less persistent in plants than soils, it seems likely that there would be a positive accumulation of residues in vegetation not removed from the soil over a number of years compared to agronomic crops that are removed each year. Young (1983) sampled perennials in soils previously and extensively treated with 2,4,5-T. At the time of plant sampling, the soil had measured concentrations of 400 ppt of 2,3,7,8-TCDD to a depth of 15 cm (this depth is about the depth to which grass and weed roots extend). Plant tops (culm, leaves, and stems) had ratios of .02-.18, plant crown (portion just above the soil surface) had a ratio of 0.66, and plant roots had a ratio greater than 1.7. This assessment estimated the highest contaminant concentration ratios for grass and fodder at

10^{-2}, which compares favorably to the plant top ratios reported by Young. The difference between grass/fodder and fruits/vegetables in this model is expressed in the empirical parameter, VG_{ag}, which is set at 1.00 for grass and 0.50 for fodder, and 0.01 for fruits/vegetables. Fruit/vegetable concentrations would be similar to grass/fodder if the empirical parameter VG_{ag} was set to the same value for both types of vegetation. The basis for parameter assignment was not on a difference in annual or perennial residue accumulations. Rather, the above-ground VG parameter assignments was based on the differences between the leaves of Bacci's experiments (1990, 1992) used to derive a vapor phase air-to-leaf transfer factor and the above ground fruits/vegetables, grass, and fodder. Similarly, the below ground VG_{bg} assignments were based on differences between roots of young barley plants of Brigg's (1982) experiments and below ground vegetables.

4) The early literature is not consistent in its reporting of fresh weight vs. dry weight, and in some articles, such information is not provided. Also, the early literature was unclear as to the mechanism of plant contamination. Outer portions of below ground vegetation contained concentrations that seem unambiguously due to sorption. However, translocation with transpiring water has been hypothesized as the cause for contamination of above ground portions in two studies (Coccusi, et al. 1979; Sacchi, et al., 1986), while contamination by soil particles has been hypothesized as the cause for contamination of above ground vegetation in two other studies (Young, 1983; Wipf, et al., 1982). Current thought on highly hydrophobic organic contaminants such as the dioxin-like compounds is that sorption to outer portions of below ground vegetation is principally the cause of transfer from soil to plant. Little within plant translocation is expected to occur. Above ground portions of vegetation are thought to be principally impacted by vapor phase transfers. These issues are further discussed in the next section.

10.2.11.3. *Uncertainty Evaluation of the Vegetation Concentration Algorithm*

A key assumption for the algorithm in this assessment is that there is an insignificant translocation of residues taken in by roots to above ground portions of plants. Recent literature on 2,3,7,8-TCDD attributes below ground vegetation concentrations to sorption onto outer plant surfaces, and above-ground concentrations to vapor-phase transfers 2,3,7,8-TCDD into shoot, leaves, and other above-ground plant material.

The specific issue of uptake and translocation via transpiration was investigated

using soybean and corn plants grown hydroponically in carefully constructed growth chambers (McCrady, et al., 1990). Roots and the hydroponic growth solution were separated from the shoots and leaves of these plants using two separate chambers, one inverted over the other. Separate air-flow systems for each chamber included traps for volatile organics. Mass balance on the tritiated TCDD experiments was able to recover 98% in the soybean experiment and 86% for the corn experiment. Most of the recovered material was found in the roots; 75% for soybeans and 67% for corn, with the second highest recovery was on the inside surface of the root chamber, around 15% for both experiments. Recovered TCDD was also found, in order of decreasing percentage, in the growth solution, root chamber air, shoot chamber air, and shoots. The recovery from the shoots was negligible at 0.004% and 0.001% of the total TCDD for the soybean and corn, respectively. McCrady, et al. (1990) concluded that transpiration stream transport of 2,3,7,8-TCDD to plant shoots is an insignificant mechanism of plant contamination, and that volatilization of TCDD is an important transport mechanism that can result in significant quantities of airborne TCDD being absorbed by plant shoots.

Briggs, et al. (1982) provide another way to evaluate the translocation of contaminants from roots to above ground vegetation. Experiments with barley roots in growth solution led to the development of an empirical parameter describing the efficiency of transport of organic chemicals to plant shoots from root uptake. This parameter is called the Transpiration Stream Concentration Factor (TSCF) and is defined as (concentration in transpiration stream)/(concentration in external solution). The empirical formula presented for this factor is:

$$TSCF = 0.784\, e^{-[\log Kow - 1.78]^2 / 2.44} \qquad (10\text{-}5)$$

Given a log Kow for 2,3,7,8-TCDD of 6.64, TSCF is solved for as roughly $5 * 10^{-5}$. Assuming that the concentration of external solution term for the experimental conditions of Briggs' experiments is equivalent to the concentration in soil water in a field situation, then the TSCF for 2,3,7,8-TCDD implies that the transpiration stream water of a plant is over 5 orders of magnitude lower than the soil water concentration. Like McCrady's experiments, this also shows the insignificance of translocation of residues from roots to shoots.

The experiments of Bacci, et al. (1990, 1992) in carefully controlled laboratory conditions demonstrated the transfer of 10 air-borne vapor-phase contaminants, including TCDD, into azalea leaves. His work formed the basis of the air-to-leaf transfer algorithm of this and several related efforts. Although some of the early research speculated that particulate depositions were the cause for above ground concentrations, the current thought is that vapor phase transfers dominate the impact to above ground vegetation.

Model results on the proportion of above ground plant concentrations that are due to air-to-leaf transfer and particulate deposition were examined. Results from this examination for 2,3,7,8-TCDD are summarized in Table 10.13. Results were opposite for soil based contamination source categories, the on-site, off-site, and ash landfill source categories, versus the stack emission source category. For the soil-based categories, it was found that the air-to-leaf transfer of vapor-phase contaminants generally accounted for over 90% of total above ground plant concentrations. However, for the stack emission source category, particulates originating from the stack and depositing onto vegetation at the site of exposure provided the largest contributor to plant concentrations, between 83.8% and 99.0% for the three types of vegetation.

An examination of the vapor-phase and particulate-phase concentrations of 2,3,7,8-TCDD in air provide one explanation for this discrepancy. For the stack emission source category, vapor phase concentrations are one approximately order of magnitude lower than particulate phase concentrations, as predicted by the ISC model. For the other three soil contamination source categories, predictions of vapor phase concentrations are approximately one order of magnitude higher than particulate phase concentrations. Another part of the explanation is the rate of particle deposition. The ISC internally calculates a deposition rate of approximately 0.06 m/sec, which is based on particle size distributions and includes wet deposition. For the soil contamination source categories, a gravitational deposition velocity of 0.01 m/sec was used. In summary, the stack emission source category predicts higher contributions by particulates because of a greater reservoir of air-borne particulate phase contaminants and because of a greater deposition velocity, as compared to the soil contamination source categories.

Table 10.13. Contribution of above ground vegetation concentrations from air-to-leaf transfers and particulate depositions.[1]

Description[2]	Air-to-leaf Transfers	Particulate Deposition
Scenarios 1 & 2: On-site Soil Source Category		
vegetables/fruit	89.2	10.8
pasture grass	99.4	0.6
fodder	98.8	1.2
Scenarios 3 & 6: Off-site and Ash Landfill Source Categories		
vegetables/fruit	93.8	6.2
pasture grass	99.6	0.4
fodder	99.5	0.5
Scenarios 4 & 5: Stack Emission Source Category		
vegetables/fruit	1.0	99.0
pasture grass	16.2	83.8
fodder	12.3	87.7

[1] Results are in percent of total contribution and are specific to 2,3,7,8-TCDD.

[2] Scenarios 1 and 2 demonstrated the "on-site" source category, where soil at the exposed individuals home was contaminated at a level of 1 ng/kg (ppt) 2,2,7,8-TCDD; Scenario 3 demonstrated the "off-site" source category, where soil at a contaminated site 150 meters away was initialized at 1 μg/kg (ppb) 2,3,7,8-TCDD; Scenario 6 demonstrated the "ash landfill" source category, where soil at a nearby landfill where ash was disposed of was initialized at 0.7 μg/kg (ppb) 2,3,7,8-TCDD; Scenarios 4 and 5 demonstrated the "incinerator stack emission" source category - in Scenario 4, the exposed site was 2000 meters from the incinerator, and in Scenario 5, the exposed site was 500 meters from the incinerator.

It is very likely that the soil contamination source categories, and to some extent the stack emission source category, are underestimating the particulate phase contributions, for at least four reasons:

- All algorithms estimating air-borne contaminant concentrations for soil contamination only estimate concentrations of PM-10, or inhalable size particulates, those 10 μm size diameter and less, while the ISC model considers all size particulates emitted from stacks. Larger size air-borne particulates, while not

inhalable, would deposit onto vegetation.

- Dry deposition only is considered for the soil contamination categories, whereas the ISC model includes dry plus wet deposition;
- For the source categories involving soil contamination distant from the site of exposure, the off-site and the ash landfill source category, only these off-site locations provide the source of air-borne particulates. Meanwhile, algorithms are in place estimating exposure site contamination, albeit to thin surface levels. Certainly, the reservoir of air-borne particulates depositing onto vegetation would also include contributions from where the vegetation is located.
- The modeling does not consider the splash effect of rainfall, which would deposit soil onto the lower parts of plants. This would make the most impact for grass and for vegetables near the ground surface such as lettuce.

The precise impact of these factors might be investigated more fully in a later assessment. However, it may turn out that the increase in particulate contributions to vegetation impacts is still low compared to vapor phase transfers for the soil contamination source categories. Tests were run increasing the amount of particulate phase contaminants depositing onto vegetation by an order of magnitude to the on-site and the ash landfill source category. Minimal impact was noted for the on-site source category. The vapor phase/particulate phase contributions to above ground fruits and vegetables, originally 98%/2% (from Table 10.13 above), changed to 84%/16% with an order of magnitude increase in particulate phase contributions. Vegetable concentrations only increased 16%. The impact was less for grass and fodder, with less than a 1% increase in concentrations. The impact was greater for the ash landfill source category. The fruit and vegetable vapor phase/particulate phase breakdown, originally 89%/11%, swung in favor of particulate phase, 46%/54%. Vegetable and fruit concentrations nearly doubled. The impact to grass and fodder was still marginal, with only a 5% increase.

This analysis does raise questions about the impact of particulate depositions onto vegetation, particularly fruits and vegetables. However, a ten-fold increase in particulate depositions would only appear, at most, to double estimated fruit and vegetable concentrations, while not impacting grass and fodder calculations.

Particulate deposition was the significant contributor to plant concentrations when incinerators were the source. Section 10.3 discusses use of the ISC model to model

particulate deposition rates, and use of the model otherwise. While that section notes several areas of variability and uncertainty, the trend for higher particulate phase air-borne contaminants at exposure sites as compared to vapor phase probably holds up under different emission technologies, emission rates, stack heights, topography, climate, and so on. The ISC model is based on accepted atmospheric transport modeling approaches, and is widely accepted and used throughout EPA. The emission rates of dioxin-like compounds used in this assessment are based on data, and limited validation of the model for downwind impacts shows model results to be consistent with observations. The conclusion that particulate deposition is of principal concern for vegetation concentration estimation is probably a reasonable one.

The result that vapor phase transfers are more critical than particulate phase depositions for above ground vegetation is tentative. The uncertainty in that result is not only a function of uncertainties concerning particulate phase contributions to plant contributions, but also the algorithms estimating vapor phase concentrations. Section 10.2.9. above discusses uncertainty issues associated with vapor phase concentrations. Critical for estimating plant concentrations from air-borne concentrations include the concentration, the air-to-leaf transfer factor, B_{vpa}, and the empirical correction factor, VG_{ag}.

The laboratory experiments used to derive the air-to-plant transfer factor, B_{vpa}, are evaluated as sound and appropriate as the starting point for current purposes. However, some correction is necessary when applying that factor for estimating fruits and vegetable concentrations due to this mechanism, since this vegetation is distinctly different than the azalea leaves of the experiments of Bacci, et al. (1990). Leaves are hypothesized as analogous to outer portions of bulky fruits and vegetables. Also, the literature discussed above did find that the large majority of fruit and vegetable 2,3,7,8-TCDD residues in outer portions of the fruit. The empirical parameter, VG_{ag}, was introduced to correct for the difference in azalea leaves and the vegetation pertinent to this assessment. Assignments to this parameter were made based mostly on a surface volume to whole fruit volume ratio. This ratio was shown to be in the range of 0.03 to 0.09. An additional correction was made based on literature showing additional residue loss after peeling or washing. The final VG_{ag} for fruits and vegetables was 0.01. It was assumed to be equal to 1.0 for grass, which is thought to be directly analogous to leaves, and 0.5 for cattle fodder, which

is a combination of leafy hay and corn grain (which includes protected grain and leafy vegetation).

The comparison of literature contaminant concentration ratios with modeled ratios above did show that the methodologies of this assessment may, in fact, be underestimating vegetative concentrations. This could imply that the VG empirical correction parameters, both for above and below ground, may be too low (the below ground VG, VG_{bg}, was also 0.01). The surface area volume was estimated assuming a very thin outer shell of 0.03 cm. Consideration of some within plant translocation would lead to an assumption of a thicker shell, and a higher surface area volume to whole fruit volume ratio. Below and above ground vegetable/fruit concentrations are linearly related to the VG parameters - increasing them to 0.10, possibly a more plausible value, would increase concentrations by a factor of ten.

A refined surface area volume to whole fruit volume ratio may be one consideration for further development of an experimentally based empirical parameter such as VG. Another consideration is the degradation of dioxin-like residues sorbed to vegetation surfaces. Bacci's data was developed for short time intervals, in the order of hours and days, and under laboratory lighting conditions. Data is now being developed which demonstrates photodegradation of dioxin-like compounds in plants in field conditions (McCrady and Maggard, 1992). The use of VG term of 0.01 for vegetables may already account for this empirically, but the VG of 0.50 for fodder and 1.00 for grass may result in predicted concentrations from air-to-leaf transfer being too high.

Finally, one has to consider the uncertainty associated with the empirical transfer factors, including the Root Concentration Factor, RCF, describing the transfer from soil water to below ground vegetables, and the air-to-leaf transfer parameter, B_{vpa}, describing the vapor phase transfer to above ground vegetation. Both these parameters are critically a function of the octanol water partition coefficient, Kow, a parameter itself which is uncertain. The B_{vpa} is additionally a function of Henry's Constant.

The range of log Kow given in Chapter 2 (Section 2.3.4) for 2,3,7,8-TCDD is 6.15 to 8.5. The value chosen for this assessment is 6.64. Equation (5-21) gives the empirical relationship estimating the RCF as a function of log Kow. The RCF for log Kow equalling 6.15, 6.64, and 8.5 are 1642, 3915 (which is the value chosen for this assessment), and 105925. Below ground vegetable concentrations are linearly and directly related to RCF;

this might imply that assuming a higher log Kow would lead to greater below ground vegetable concentrations. However, the equation for estimating such concentrations, Equation (5-18), shows them also to be an inverse function of the adsorption partition coefficient, Kd, which is also estimated as an empirical function of log Kow in this assessment. Increasing Kow also increases Kd, or increases sorption to soil, which tends to reduce concentrations. Plant concentrations are a function of RCF/Kd. Log Kows of 6.15, 6.64, and 8.5 translate to the following RCF/Kd ratios: 0.19, 0.14, and 0.005. Therefore, an assumed log Kow of 8.5 results in over an order of magnitude reduction in below ground vegetation from what was assumed in this assessment, whereas a lower log Kow of 6.15 results in roughly a 36% increase in estimated concentrations.

A similar dual effect is noted when assuming a different Kow to estimate air-to-leaf vapor phase transfers. Equation (5-23) shows the transfer factor, B_{vpa}, to be a function of log Kow. Increasing log Kow to 8.5 over the assumed 6.64 results in a B_{vpa} over 700 times higher. However, increasing log Kow again would imply an increase in Kd, and increase of sorption to soil, and a subsequent reduction in volatilization flux. Air concentrations are decreased by roughly an order of magnitude with a log Kow of 8.5 for 2,3,7,8-TCDD. With roughly two orders of magnitude increase in the transfer factor, but an order of magnitude decrease in air concentrations, the net effect is to increase the vapor transfers to plants by an order of magnitude with an increase in log Kow to 8.5.

Henry's Constants are less variable for the dioxins and furan compounds, as seen in Table 2.2. The value assumed for 2,3,7,8-TCDD, 1.65×10^{-5} atm-m^3/mol, is the highest of listed values, with the lower values being on the order of 1×10^{-6} atm-m^3/mol. Reducing H to this value increases the B_{vpa} parameter by a factor of 16. Again, changing H influences air concentrations - they would decrease by a factor of 4 with the noted reduction. Therefore, reducing H by an order of magnitude has the net effect of increasing the vapor phase transfers by a factor of 4.

10.2.12. Beef and Milk Ingestion

Concentrations in beef and milk are a function of cattle ingestion of contaminated soil, pasture grass, and cattle fodder. Therefore, previous sections are relevant to estimating beef and milk concentrations. Section 10.2.3. on soil erosion describes uncertainties with estimating soil concentrations at an exposure site distant from a site of

soil contamination. Section 10.2.11. above describes uncertainties associated with estimating grass and fodder concentrations.

The first section below describes uncertainties in the ingestion rates of beef and milk. The next section compares model estimations of beef and milk concentrations with those that have been found worldwide. An evaluation of alternate modeling approaches is then made. Finally, uncertainties in the parameters of the algorithm estimating beef and milk concentrations are described. Table 10.14 summarizes uncertainties associated with the beef and milk exposure pathways.

10.2.12.1. *Beef and Milk Ingestion Rates*

Data on rates of milk and beef consumption were taken from surveys summarized in EPA (1989). Whereas the survey data may lead to adequate estimates for per capita consumption of these products, EPA (1989) cautions that farm families who home slaughter or who home produce dairy products may have higher consumption rates. Data is unavailable for these situations. Another consideration for application to real world rural situations is that farming and non-farming families may be obtaining cattle food products from local farms which may also be impacted by dioxin-like compounds. This possibility was not addressed in this assessment.

The fractions of meat or milk intake coming from the farmer's home supplies was determined in a survey of 900 rural farm households (USDA, 1966). The 0.44 (44%) of meat and 0.40 of dairy contact fractions from this survey were, appropriately, proportions of total dietary intake that is home-produced and consumed by farming families. Therefore, more certainty is expected for these contact fractions as compared to ingestion rates.

The trend analysis for the example scenarios in Chapter 9 indicated that the greatest exposures occur for beef, milk, and fish. Therefore, the rate of consumption of impacted beef and milk is critical. The range of beef fat consumption noted in surveys summarized in EPA (1989) is 14.9 to 26.0 g/day, but a single high consumption rate of 30.6 g/day was noted. If this high rate is more typical of home-producing farm families,than the value of 22 g/day selected for this assessment may be 28% low (28% is (30.6-22)/(30.6)*100%). The single high rate of 35 g/day of milk fat is significantly higher than the 8.9-10.7 g/day range noted in EPA (1989) and the 10 g/day ingestion rate

Table 10.14. Uncertainties associated with beef and milk ingestion exposure algorithms.

Assumption/ Method	Approach	Rationale	Uncertainty	Comments
Ingestion rates	22 g/day beef fat 10 g/day milk fat	Literature showed 14.9-26.0 g/day beef fat, and 18.8-43 g/day milk fat; ranges developed from 3 surveys	Shape of distribution of consumption not well defined; study and survey showing 43 g/day milk fat less well documented than other 2 surveys.	Beef and milk home producers may tend to ingest more than average families.
Contact rates	0.44 for beef fat 0.40 for milk fat	Data from USDA survey including percent of annual consumption of beef and milk homegrown.	Likely to be substantial differences between families; some may not home slaughter.	Again, home producers may obtain more than 44 or 40% of of beef and milk from their own supplies.
Beef and milk fat concentration estimates	Developed by Fries and Paustenbach (1990); bioconcentration factor multiplied by proportionally weighted concentrations of soil, grass, and fodder.	Their principle premise was that 2,3,7,8-TCDD bioconcentrates equally in beef and milk fat; their literature survey developed key parameters used here as well.	Uncertainties associated with soil, pasture grass, and fodder carry over into beef and milk fat concentrations; other uncertainties with parameters as noted below.	Section 10.2.12.3 shows how current approach is the same as earlier approaches which used whole beef and milk biotransfer factors and similar models for particle deposition impact to soil and vegetation.
Key parameters and assumptions	Bioconcentration factor, F for 2,3,7,8-TCDD of 5.0; Soil bioavailability Bs of 0.65; and assumptions on proportions of diet in soil, grass, and fodder	F developed by Fries and Paustenbach from data, which also showed lower F for higher chlorinated dioxin-like compounds; also developed Bs and tested different diet assumptions.	Of three noted, assumptions on proportions of diet most variable and having the most impact; hypothetical worst and best scenarios for cattle ingestion tested and shown to result in roughly an order of magnitude range due to these assumptions alone.	Fattening of beef cattle prior to slaughter is not considered and could result in 50% or more reductions in fat concentrations.

Overall: Model predicted beef and milk fat concentrations of 2,3,7,8-TCDD resulting from a 1 ppt soil concentration, chosen to be similar to low, perhaps background, concentrations found by researchers, compared favorably with beef and milk concentrations taken by researchers similarly looking for background concentrations in milk or beef and/or sampling from farms with no known nearby source of possible 2,3,7,8-TCDD contamination. A similar favorable comparison was noted for beef and milk concentrations resulting from incinerator depositions. Comparison with earlier modeling approaches showed that the current approach is the same as earlier approaches, although mathematically formulated differently. Earlier approaches also estimated cattle dose of 2,3,7,8-TCDD from contaminated air (directly) and contaminated ground water - these earlier estimations showed these contributions to be minimal, and they were not considered in this assessment. Finally, earlier assessments considered the practice of fattening beef cattle prior to slaughter by feeding them residue-free grains. These efforts estimated over a 50% reduction in beef concentration due to residue degradation or elimination and/or dilution with increases in body fat. This assessment does not consider this practice. Including a depuration algorithm or other cattle production practices which influence concentrations in fat should be considered in future refinements of the models in this assessment.

for milk fat may be 71% too low.

10.2.12.2. *Comparison of modeled beef and milk concentrations with concentrations found*

The example scenario in Chapter 9 demonstrating the on-site source category (where the soil at the place of residence/farming/exposure is the source of contamination) had soil concentrations initialized at 1 ng/kg (ppt) 2,3,7,8-TCDD. This concentration was chosen because it was similar to concentrations of 2,3,7,8-TCDD found in studies where researchers had measured what they characterized as "rural" or "background" soils. Beef and milk fat concentrations of 2,3,7,8-TCDD estimated with this soil concentration were 0.4 and 0.1 ppt 2,3,7,8-TCDD, respectively. Assuming fat contents for beef and milk of 0.22 and 0.035, respectively, whole beef and milk concentrations are estimated as 0.09 and 0.001 ppt. Beef and milk fat concentrations for an exposure site located 500 meters from a hypothetical incinerator, another of the example scenarios in Chapter 9, were 1.3 and 0.8 ppt. Corresponding whole beef and milk concentrations were 0.3 and 0.03 ppt. The other two source categories were sites of higher soil concentration located near sites of exposure. One, termed the off-site source category, had a 4 hectare site contaminated with 2,3,7,8-TCDD at 1 μg/kg (ppb) located 150 meters from an exposure site. This concentration was selected based on similar 2,3,7,8-TCDD concentrations found in sites of elevated contamination, such as Superfund sites. Soil concentrations at the site of exposure were estimated to be 0.61 ppb, or 610 ppt. Concentrations in beef and milk fat were 170 and 46 ppt, respectively, which corresponds to whole product concentrations of 37 and 1.6 ppt. The second source category with elevated soil concentrations near a site of exposure is the ash landfill category. Initial 2,3,7,8-TCDD concentrations on a 27 hectare landfill site were estimated based on expected ash concentrations. Landfill 2,3,7,8-TCDD concentrations were 700 ppt, and nearby exposure site soil concentrations were 563 ppt. Resulting beef and milk fat concentrations were 161 and 45 ppt, respectively, and whole product concentrations were 35 and 1.6 ppt.

Several studies have measured concentrations of dioxin-like compounds in farm products (dairy, beef, etc.) where it is not known whether the products are impacted by a nearby known source - market basket surveys may be characterized this way - or where samples were obtained on farms chosen specifically because no identifiable source was

nearby. It may be appropriate to compare results from these types of studies to results from the first source category noted above, where soil concentration levels were low and maybe typical of background or ambient levels. In such a survey taken in Germany (Beck, et al. 1989), the fat content of meat and eggs averaged 0.23 ppt, and for dairy products averaged 0.14 ppt 2,3,7,8-TCDD. Another random sample of food products from Germany found average cow's milk fat concentration to average 0.4 ppt for ten samples, ranging up to 1.9 ppt (Furst, 1990). Concentrations of 2,3,7,8-TCDD were not found in 3 beef fat samples, with a detection limit of 0.5 ppt (Furst, 1990). Samples of whole milk from seven farms in England chosen because they were well removed from known sources of dioxin showed a mean concentration of 0.009 ppt (detection limit = 0.004 ppt; Startin, et al., 1990). No 2,3,7,8-TCDD residues were detected in three whole milk samples in Switzerland (detection limit = 0.013 ppt) where no identifiable source of dioxins was nearby (Rappe, et al., 1987).

The example scenario results from the incinerator source showed beef and milk concentrations higher than those for the first example source category, but still near 1.0 ppt. The literature also shows comparable trends for sampling which occurred near incinerators. The study sampling remote farms in England also sampled two farms near incinerators and two farms near industrial centers. Whereas samples from remote farms averaged 0.009 ppt for whole milk, two concentrations near the incinerators were 0.034 and 0.036 ppt 2,3,7,8-TCDD, and the samples near the industrial centers were 0.043 and 0.081 ppt (Startin, et al., 1990). The study noted above sampling milk from locations remote from 2,3,7,8-TCDD sources in Switzerland, and not detecting residues, also sampled three locations that were within 1000 meters of incinerators. Whole milk concentrations near the incinerators were 0.021, 0.038, and 0.049 ppt.

Sampling of beef and milk near areas of elevated soil concentrations, or where cattle were raised on soils with known high concentrations of 2,3,7,8-TCDD, were not found in the literature. There are some studies on other animals indicating high tissue concentrations in areas of high soil contamination of 2,3,7,8-TCDD. Lower, et al. (1989) studied animal tissues for wild animals in the abandoned town of Times Beach, Missouri, and compared their results for similar wild animals tissue concentrations found in Eglin Air Force Base in Florida; Seveso, Italy; and Volgermeerpolder in Holland. With 2,3,7,8-TCDD soil levels in these areas in the hundreds to thousands of ppt, tissue levels for earthworm,

mouse, prairie vole, rabbit, snake, and liver samples from some of these animals, were in the tens to hundreds to thousands of ppt.

There is an episode of beef and dairy cows (as well as lambs and pigs) being raised on lots where the soil was heavily contaminated with polybrominated biphenyls (PBB; details of the extent and cause of contamination are given in Fries and Jacobs, 1986; results cited are from Fries, 1985). Soil concentrations to which dairy and beef cows were exposed were 830 and 350 μg/kg (ppb), respectively. Body fat of the dairy cows had PBB concentrations of 305, 222, and 79 ppb (dairy heifers, primaparous dairy, and multiparous dairy, respectively). Body fat for the beef cows exposed to 350 ppb soil levels were 95 and 137 ppt (cows and calves, respectively). Milk fat concentrations from the primaparous dairy and multiparous dairy cows exposed to 830 ppb soil levels were 48 and 18 ppt.

Fries estimated a quantity which is also useful for purposes of comparison - this quantity is the ratio of concentration in animal fat to concentration in soil. His justification for deriving this ratio is that soil was speculated as the principal source of body burdens of PBB in the data listed above. For the source categories where contaminated soil is the source of dioxin-like compounds, the on-site, off-site, and ash landfill source categories, a similar assumption is warranted. Ratios he derived for body fat of dairy heifers ranged from 0.10 to 0.37, while it was 0.02 and 0.06 for milk fat. For body fat of beef cows, these ratios were 0.27 and 0.39. Fries also measured a ratio of 1.86 for sows and gilts. He attributes much higher sow ratios due to their tendencies to ingest more soil. Analogous ratios can be derived for the contaminated soil source categories, and for beef and milk fat. For the onsite source category with low soil concentrations, beef fat to soil and milk fat to soil ratios were 0.4 and 0.1, respectively. For the off-site and ash landfill source categories, ratios were the same at 0.28 for beef fat and 0.08 for milk fat. These compare favorably with PBB ratios derived by Fries (1985).

The incinerator source category does not have comparable results. The beef and milk fat to soil ratios were 1.00 and 0.61. The reason this occurs is that vegetation is more impacted by emission depositions as compared to soil transfers. The soil concentration resulting from the emission depositions is 1.27 ppt, which compares to the 1.00 ppt assumed for the on-site source category. However, the grass and fodder concentrations resulting from incinerator depositions are 0.20 and 0.14 ppt, respectively,

whereas in the on-site source category, grass and fodder concentrations are 0.02 and 0.01 ppt. With an order of magnitude higher vegetation concentration resulting from incinerator emission depositions, beef and dairy impacts are greater than estimated when soil contamination is the source. It is not appropriate to compare the 1.00 and 0.61 fat to soil ratios with PBB ratios developed by Fries, for this reason.

10.2.12.3. *Alternate Modeling Approaches for Estimating Beef and Milk Concentrations*

Webster and Connett (1990) compared five models which estimated the 2,3,7,8-TCDD content of cow's milk from 2,3,7,8-TCDD air contamination. The five models were described in Michaels (1989), Connett and Webster (1987), Stevens and Gerbec (1988), Travis & Hattermer-Frey (1987), and McKone and Ryan (1989). Ironically, a sixth model by Fries and Paustenbach (1990), noted by Webster and Connett as available but received too late for inclusion in their article, formed the basis for the approach taken in this assessment.

All five models compared by Webster and Connett have the same basic framework. Particulate-bound 2,3,7,8-TCDD deposits onto the ground and vegetation (cattle fodder and pasture grass). Algorithms to estimate resulting vegetation and soil concentrations in these models are the same ones used in this approach, although parameter assignments are different. A daily dosage of 2,3,7,8-TCDD to the cattle is calculated and converted to a concentration in whole milk using a "biotransfer factor". This same structure was used to estimate concentrations in beef, using a beef biotransfer factor different than the milk biotransfer factor. Mathematically, this is expressed as:

$$C_{m,b} = F_{m,b} \; Dose \qquad (10\text{-}6)$$

where:

$C_{m,b}$ = concentration in whole milk/beef, mg/kg

$F_{m,b}$ = milk/beef biotransfer factor, day/kg

= $(BCF_{mf,bf} * f_{m,b})/Q$

$BCF_{mf,bf}$ = experimentally-derived unitless bioconcentration factor defined as the concentration in milk fat/beef fat divided by the concentration in the experimental vehicle (cattle feed, e.g.)

$f_{m,b}$	=	fat content of milk/beef, unitless
Q	=	daily mass intake of cattle in experiment, kg
Dose	=	total daily dose of 2,3,7,8-TCDD, mg/day
	=	$\Sigma (a_j * c_j * Q_j)$
a_j	=	relative bioavailability on intake vehicle j (soil, air, vegetation, etc)
c_j	=	concentration of 2,3,7,8-TCDD in vehicle j, mg/kg (or equivalent units)
Q_j	=	mass of vehicle j intake, kg (or equivalent units)

Further details on the models can be found in their primary references and in Webster and Connett's comparison. Some highlights, including comparisons of the five approaches to the approach taken in this assessment, are:

1) Two of the approaches, that of Stevens and Gerbec (1988), and McKone and Ryan (1989), consider inhalation of contaminated air by cattle to contribute to their daily dose of 2,3,7,8-TCDD. One of the approaches, that of Travis and Hattemer-Frey (1987), considers ingestion of contaminated water by cattle. A later assessment by Travis and Hattemer-Frey (1991) has all the components of their earlier assessment, and adds cattle inhalation exposures. This assessment does not consider cattle inhalation of contaminated air nor ingestion of contaminated water in estimating beef and milk concentrations. However, these intakes were shown to be insignificant when estimated by these researchers. Stevens and Gerbec estimate inhalation contributions to be less than 0.05% (0.0005 in fractional terms) of total daily dose, or an essentially insignificant amount. Travis and Hattemer-Frey (1991) estimate inhalation to contribute between 0.3 and 1.0% to milk and beef concentrations, respectively. McKone and Ryan (1989) did not provide sufficient information to easily determine the relative contribution of inhalation on estimation of cattle beef and milk concentrations by their estimations. Travis and Hattemer-Frey (1987, 1991) estimate water contributions to be less than 0.01% (0.0001) of total daily cattle dose of 2,3,7,8-TCDD.

2) None of the approaches considered vapor phase transfers from air to plant, although Webster and Connett recommended its inclusion in their article. The later assessment by Travis and Hattemer-Frey (1991) on 2,3,7,8-TCDD did include vapor phase transfers into vegetation consumed by cattle. According to results of the example

scenarios in this assessment, these transfers appear to be particularly critical when the source of contamination is soil. This is discussed in Section 10.2.11.3, where vapor phase transfers were shown to contribute over 90% of 2,3,7,8-TCDD contamination of above ground vegetation. Where the source was incinerator emissions, however, particulate depositions dominated plant concentrations, with contributions between 88 and 99% of total above-ground plant concentrations. Three of the assessments evaluated by Webster and Connett were specific to incinerator emissions and estimated plant concentrations based on particulate deposition (Stevens and Gerbec (1988), Connett and Webster (1987), and Michaels (1989)), so an omission of vapor phase transfers in these assessments may not have been critical. However, one of them assumed particle depositions did not impact vegetation (Michaels (1989)), but impacted soil, and calculated plant concentrations based on a root uptake algorithm. Webster and Connett discuss why this particular approach, which neglected particle deposition onto vegetation, greatly underestimated vegetative concentrations.

3) One of the assessments, Stevens and Gerbec (1988), assumed a specific mode of beef cattle production - in particular, one which included placing the cattle on a grain-only diet for fattening prior to slaughter. They assumed the grain was residue free, and therefore, residues in cattle would depurate during the last 130 days of their lives. Assuming a half-life of 2,3,7,8-TCDD in cattle of 115 days, they showed a 54% reduction in beef concentrations due to this practice. Fries and Paustenbach (1990) also identified cattle production practices as critical, and evaluated the impact of a 120-day grain only diet prior to slaughter in their modeling. They note that cattle can gain as much as 60-70% in body weight, so dilution can also result in lower beef concentrations at slaughter. Procedures are not described in this assessment to estimate the reduction of concentrations in beef and milk fat due to depuration or dilution periods, although the procedure is simple. Assuming first order kinetics sufficiently describes reduction in concentrations during a period prior to slaughter, the fractional reduction during such a period is given as, $1 - \exp(-k_d t)$, where k_d is the depuration rate constant, in $days^{-1}$, and t is the depuration period, in days. The rate constant can be estimated from the depuration half-life, HL, as 0.693/HL. The 115 day half-life assumed by Stevens and Gerbec (1988) corresponds to a rate constant of 0.006 day^{-1}, and assuming a 130 day depuration period, the fractional reduction is easily calculated as 0.54 (i.e., $1 - \exp(-k_d t)$). The amount

remaining after 130 days is estimated as the initial amount multiplied by 0.46 (i.e., exp($-k_d t$)).

4) Two of the assessments did not assume any cattle ingestion of contaminated soil, and two of the assessments estimated the contribution to milk concentrations due to ingestion of contaminated soil was minor at 1 and 2%. Only one of the assessments, Travis and Hattermer-Frey (1987), estimated any significant impact due to soil ingestion, attributing 19% of the concentration due to ingestion of contaminated soil. Their later assessment (Travis and Hattemer-Frey (1991)) estimated soil to contribute 29 and 20% of beef and milk concentration estimations, respectively. They estimated this high a contribution by contaminated soil even though they assumed that contaminated soil comprised 1% of the total dry matter intake by cattle. Fries and Paustenbach (1990) recognized the importance of cattle soil ingestion, evaluating scenarios where cattle soil ingestion ranged from 1 to 8% of total cattle dry matter intake. This example scenarios in Chapter 9 assumed that beef cattle ingestion of contaminated soil was 8% of their total dry matter intake, and 2% of a dairy cattle's intake was contaminated soil. The percentage of beef and milk concentrations of 2,3,7,8-TCDD attributed to soil, fodder, and pasture grass, when soil contamination is the source and when incinerator emissions are the source, are given in Table 10.15. As seen, cattle soil ingestion is a significant contributor to beef and milk concentration estimations using procedures and assumptions in this assessment, explaining 11 to 71% of the beef and milk concentrations. Two key explanations for the differences in results in this assessment versus the ones in the literature include: the other assessments assumed less soil ingestion, 0.5% in Stevens and Gerbec (1988) and 1-3% in Travis and Hattemer-Frey (1987) and McKone and Ryan (1989), and differences in soil and plant concentration estimations (due to different mixing depth assumptions for soil concentrations, particle deposition rates for plant concentrations, and so on).

The various approaches used different parameter values for the biotransfer factor, intake quantities, initial air concentrations, and others. The critical focus of the Webster and Connett (1990) comparison, is the milk fat bioconcentration factor, BCF_{mf}. As shown in Equation (10-6), the biotransfer factor, F_m, is estimated using experimental data which yields a milk fat bioconcentration factor, BCF_{mf}. Experiments most relied upon by these modelers are those described in Jensen, et al. (1981), and Jensen and Hummel (1982). A

Table 10.15. Estimated relative importance of soil, pasture grass, and fodder ingestion to the estimation of beef and milk concentrations when soil and incinerator emissions are the source of 2,3,7,8-TCDD contamination.

	Percent of beef and milk fat concentrations due to:		
Description	Soil	Pasture Grass	Fodder
I. Soil Contamination			
Beef	71	29	<1
Milk	51	7	42
II. Incinerator Emissions			
Beef	26	73	1
Milk	11	11	78

key difference in the early modeling approaches is the interpretation of these two and other studies and the resulting assignment of BCF_{mf}, with values ranging from 5 to 25. Webster and Connett (1990) discuss issues of experimental interpretation.

Parameter assignments and assumptions (cattle soil ingestion vs. no ingestion, etc.) obviously all impact estimations and can be a critical source of variation and uncertainty in estimates of beef and milk concentrations. The uncertainty associated with the modeling framework described above was explored by McKone and Ryan (1989) using Monte Carlo techniques. They found that the 90% confidence range for human exposure to 2,3,7,8-TCDD, where the source was air contamination and the human exposure route was through milk, spanned two to three orders of magnitude.

The approach taken by all five researchers centers on the milk biotransfer factor, abbreviated F_m in Webster and Connett (1990) and in units of day/kg. Beef bioaccumulation was modeled in the same way using a beef biotransfer factor, F_b. Travis and Arms (1988) developed this concept to the fullest, taking several data sets from the

literature on a variety of contaminants and animals, to derive empirical formulas for F_b and F_m, which they termed B_b and B_m, as a function of contaminant octanol water partition coefficient, Kow:

$$\log B_b = \log Kow - 7.6 \qquad (10\text{-}7a)$$

$$\log B_m = \log Kow - 8.1 \qquad (10\text{-}7b)$$

Given a log Kow of 6.64 for 2,3,7,8-TCDD (assumed in this assessment), B_b is solved for as 0.110 and B_m is solved for as 0.034. Travis and Hattemer-Frey (1991) use 0.80 and 0.03 for B_b and B_m.

Simple transformations can show how the earlier approaches and the approach of Fries and Paustenbach (1990), the one used in this assessment, are the same. First, the concentration of dioxin-like compounds in the fat of beef and milk is given in this assessment by (also as Equation (5-24) in Chapter 5):

$$C_{fat} = (F\ DF_s\ B_s\ AC_s) + (F\ DF_g\ AC_g) + (F\ DF_f\ AC_f) \qquad (10\text{-}8)$$

where:

C_{fat}	=	concentration in beef fat or milk fat, mg/kg
F	=	bioconcentration ratio of contaminant as determined from cattle vegetative intake (pasture grass or fodder), unitless
DF_s	=	fraction of cattle diet that is soil, unitless
B_s	=	bioavailability of contaminant on the soil vehicle relative to the vegetative vehicle, unitless
AC_s	=	average contaminant soil concentration, mg/kg
DF_g	=	fraction of cattle diet that is pasture grass, unitless
AC_g	=	average concentration of contaminant on pasture grass, mg/kg
DF_f	=	fraction of cattle diet that is fodder, unitless
AC_f	=	average concentration of contaminant in fodder, mg/kg.

Transformation steps are: 1) factor out the F from Equation (10-8) , 2) multiply the top

and bottom of Equation (10-8) by total dry matter intake by cattle, Q; 3) the multiplication of Q by the diet fraction terms in the numerator, DFs, DFg, and Dff, gives the values for soil dry matter intake, Qs, grass - Qg, and fodder - Qf, 4) with F factored out, and Q*DFs replaced by Qs, etc., the parenthetical now reads, (Qs*Bs*ACs + Qg*ACg + Qf*ACf) - this is the "Dose" term defined above in Equation (10-6), 5) finally, multiply the right hand side of Equation (10-8) by fat content, say f_m for milk, which would transform the right and hence left hand side of that equation to whole product concentration. Transformed Equation (10-8) is analogous to Equation (10-6):

$$C = \frac{F\ f_m}{Q}\ [\ (Q_s\ B_s\ AC_s\ )\ +\ (\ Q_g\ AC_g\ )\ +\ (\ Q_f\ AC_f\ )\ ] \qquad (10\text{-}9)$$

Fries and Paustenbach (1990) discuss one critical theoretical assumption not explored in the earlier literature: that 2,3,7,8-TCDD bioaccumulates equally in beef fat and milk fat - that the BCF_{mf} and BCF_{bf} are equal. The differences in observed concentrations in beef and milk are attributed to differences in the diets of cattle raised for beef versus those raised for milk. The key difference Fries and Paustenbach cite is the tendency for beef cattle to graze while lactating cattle are more often barn fed. Grazing cattle intake more contaminated soil than barn fed cattle. Fries and Paustenbach justify their selection of 5.0 for F, the bioconcentration factor, for 2,3,7,8-TCDD, which is also used in this assessment. They also derive F for higher chlorinated dioxin-like compounds from experimental data, noting that the F value is less with higher chlorination. Webster and Connett (1990) made the analogous observation, saying that 2,3,7,8-TCDD equivalents transferred from air to milk less efficiently than 2,3,7,8-TCDD.

Some conclusions from this analysis of earlier efforts for estimating bioconcentration in beef and milk are:

- Although the framework of the earlier approaches looks different than the framework used in this assessment, they are actually the same with a simple mathematical transformation;
- The possible dosage to cattle of 2,3,7,8-TCDD via contaminated air or water was considered in earlier assessments, but was not found to be a significant pathway, and was not considered in this assessment;

- Earlier assessments did not consider soil ingestion to the extent considered in this assessment; the results of this assessment are that 11-71% of the beef and milk concentrations were attributed to contaminated soil ingestion by cattle;
- Earlier assessments did not consider vapor phase transfers to vegetation consumed by cattle; this assessment implies that this transfer is particularly critical for soil contamination, but less critical for incinerator emissions;
- Even though the structure of the analysis has been consistent from the earlier to the current approaches, different assumptions on parameter values greatly impacts modeling results. The critical bioconcentration factor, earlier termed BCF_m (for milk) and termed F in this assessment, was assumed to be 5 to 25 for 2,3,7,8-TCDD in different assessments. This assessment uses an F value of 5 for 2,3,7,8-TCDD. Using Monte Carlo techniques on this model structure for estimating human exposure to milk resulting from air contamination of 2,3,7,8-TCDD, McKone and Ryan (1989) showed a 90% confidence interval spanning 2 to 3 orders of magnitude.

10.2.12.4. *Uncertainty of Model Parameters Estimating Beef and Milk Concentrations*

Earlier sections have discussed the uncertainty of soil and vegetation concentration estimations. Needless to say, any differing assumption or parameter value which impacts soil and/or vegetation concentrations will also impact beef and milk concentrations. Exercises undertaken to evaluate the sensitivity of model estimations to soil and vegetation concentrations were not undertaken once again to evaluate the modeled impact to beef and milk. However, beef and milk concentrations are linearly related to soil and vegetation concentration - see Equation (10-10) above. Using the information in Table 10.15 above, and referring to earlier sections for other sensitivity information, one can note key relationships and estimate impacts. For example, it is easily shown that beef and milk concentrations are linearly related to soil concentrations when soil is the source of contamination. Specifically, vapor-phase air concentrations are linearly related to soil concentrations, and vegetation concentrations are principally a function of vapor phase

transfers for the soil contamination source categories. Therefore, doubling soil concentrations will double the portion of the impact to beef and milk concentrations due to soil ingestion, and will also result in a doubling of vegetation concentrations; the net impact would appear to be roughly a doubling of beef and milk concentrations with a doubling in soil concentration. A similar analysis shows that particle deposition rates and concentrations are the principal quantities for beef and milk concentrations for the stack emission source category. Because of the greater impact of vegetation to beef and milk concentrations when incinerator emissions are the source (see Table 10.13), assumptions and quantities involved in plant concentration estimation for depositing particles become critical.

This section will focus on the three remaining areas of variability and uncertainty. One is the parameter describing the relative bioavailability of soil in relation to plants, Bs. The second is the bioconcentration factor, termed F. The third is the dietary assumptions of beef and dairy cattle - the proportion of diet in soil, pasture grass, and cattle fodder, as well as the assumption that cattle fodder and pasture grass are grown on the soil that is contaminated and is therefore impacted. The basis for assignments of all parameters in these areas was provided in Fries and Paustenbach (1990). Their justifications will be noted, but further details are provided in that and supporting references.

The literature survey done by Fries and Paustenbach (1990) showed that soil was likely to be a less efficient vehicle for internal absorption than cattle feed, but not by much. They hypothesized that absorption of 2,3,7,8-TCDD with cattle feed as the vehicle was approximately 50%, and that absorption with soil as the vehicle was 30-40%. This provided the basis for the assignment of 0.65 for Bs. It is unlikely that Bs would vary greatly. Tests were run for Bs equal to 0.5 and 0.8. The response to estimated beef and milk fat concentrations to these changes, when soil was the source of contamination, was minimal. Beef concentrations increased and decreased 16% with an increase to 0.8 and a decrease to 0.5. Milk concentrations increased and decreased 10% with the same changes. There was an even smaller impact for the incinerator source with these changes in Bs. Beef concentrations increased and decreased by 6%, and milk concentrations were impacted by only 2%.

The bioconcentration factor, F, applicable to both beef and milk fat, was justified by Fries and Paustenbach (1990) to be in the range of 4 to 6 for 2,3,7,8-TCDD. They note

that earlier estimates in the literature based on experiments placing F in the range of 12 to above 20 assumed that 100% of the delivered dose was absorbed. Evidence suggests that percent absorption is more appropriately 50%. They also note that some of an earlier estimates by Jensen, et al. (1981) were based on a 28-day feeding trial and then projecting to steady state. Sixty-day studies of cattle ingestion of PCBs (Fries, 1982) estimated F at about 5, and other studies running 160 days and involving cattle ingestion of hexa-, hepta-, and octa-CDDs, estimated F at 2 and below (Parker, et al., 1980).

The analysis by Fries and Paustenbach (1990) suggests that there is not much variation in F for 2,3,7,9-TCDD. There may be an argument that an F for milk should be lower than an F for beef since lactation is a form of residue elimination. However, the data is not available to support this position, and Fries and Paustenbach (1990) argue that the principle variation noted in milk and beef fat concentration is due to diet. In any case, milk and beef fat concentrations are linearly related to F, as seen in Equation (5-24) above; doubling F to 10 would double concentrations of 2,3,7,8-TCDD, and so on.

The impact of different dietary assumptions can be evaluated with variations in the parameters describing the proportion of diet in soil, grass, and fodder, and also the parameters describing the proportion of contaminated land to which the cattle are exposed. The latter set of parameters were used to determine the average concentrations on soil, grass, and fodder. For the example scenarios, it was assumed that all the soil, grass, and fodder were impacted - all soil ingested had concentrations initially specified or solved for, etc.

Rather than define several alternate assumptions, a set of extreme assumptions were defined and tested. Currently, for estimating beef concentrations, the proportion of dietary intakes are 8% soil, 90% pasture grass, and 2% fodder, all of which are impacted by 2,3,7,8-TCDD. A worst-case scenario might have cattle ingesting 15% soil, 83% pasture grass, and 2% fodder (Fries and Paustenbach (1990) suggest a bounding estimate for cattle soil ingestion of 18%). A reasonable best-case scenario might have beef cattle ingest 2% soil, 65% pasture grass, and 33% residue-free grain or other fodder. These scenarios were tested for source categories involving contaminated soil and for the stack emission source category. For the soil source categories, the worst case assumptions increased beef concentrations by 60% and the best case assumptions decreased beef concentrations by 61%. For the emission source category, the worst case assumption

increased beef concentrations by 17% and the best case reduced concentrations by 40%.

The current assumptions for lactating cattle were 2% soil, 8% pasture grass, and 90% fodder, all of which were impacted. A worst-case scenario might have dairy cattle pastured more with 4% soil intake, 25% pasture grass intake, and 71% fodder, all impacted. A best-case scenario might have less soil ingestion at 1%, some pasturing with 9% pasture grass, and 90% residue-free fodder. For the soil source categories, the worst case scenario increased milk concentrations by 61%, and a decrease of 66% was noted for the best case scenario. For the stack emission scenario, the worst case increased concentrations by 16% and the best case decreased concentrations by 83%.

This analysis shows that there can be nearly an order of magnitude difference in predicted fat concentrations between the highest and lowest estimates due to dietary assumptions, with the assumptions for the example scenarios being near the middle of this possible range. Assuming unimpacted fodder can evaluate the impact of a lower lifetime intake of contaminant. However, the practice of fattening beef cattle with hypothesized residue-free grain prior to slaughter was not evaluated in this exercise. Fries and Paustenbach (1990) speculated that lower fat concentrations would result from a 60-70% weight gain and dilution effects during this period, while Stevens and Gerbec (1988) assumed elimination would reduce body fat concentrations by roughly 50%. Either way, this assessment may have overestimated beef fat concentrations by a factor of two if the practice of fattening prior to slaughter occurs for a specific site.

10.3. UNCERTAINTY AND VARIABILITY IN ANALYSIS OF DIOXIN-LIKE COMPOUNDS EMITTED FROM MUNICIPAL SOLID WASTE INCINERATION

Chapter 6 described the basis for evaluating the environmental fate and transport of dioxin-like compounds that may be discharged to the atmosphere from the stack of a municipal solid waste incinerator. The procedures were applied to a hypothetical MSW incinerator that was arbitrarily located in Tampa, Florida. Uncertainty could be analyzed using Monte Carlo Simulation techniques to generate probability distributions of input parameters used in the modeling of the emissions from the incinerator. If it is known how the data is distributed, e.g., normal distribution, log-normal distribution, etc., the random values of these parameters could be generated. Key to this analysis is a relatively large set of measured values. With regard to the modeling of emissions of the dioxin-like

compounds, not enough discrete measurements of congener-specific emissions under varying conditions and operations have been ascertained. The limiting factor has always been the analytical costs involved. Therefore, this section is not a true uncertainty analysis, rather it addresses basic variability in the key data inputs and outputs. Because the algorithms supporting the air dispersion and deposition analysis are linear, variability can be estimated if only one dependent variable to the analysis is changed, and the other variables are held constant. Variability can be addressed in terms of the spread of modeled input values and their effect on the spread of the modeled output values of the estimated environmental impacts. A second issue is the estimated extent or magnitude of change.

The principle components of the analysis of estimating spacial and temporal ground-level concentrations of the stack emissions of dioxin-like compounds from the hypothetical incinerator are the following:

(1) The probability of emission of dioxin-like compounds from the stack of the incinerator.

(2) The quantity or magnitude of release of dioxin-like compounds from the stack of the incinerator.

(3) The representativeness of the hypothetical incinerator in comparison to actual facilities.

(4) The estimation of dilution, dispersion, wet and dry surface deposition of the dioxin-like compounds using the Industrial Source Complex Model.

(5) Comparison of ambient measurements with the predicted ambient air concentration of dioxin-like compounds.

Each of these elements will be discussed separately.

10.3.1. The Probability of Emission Of Dioxin-Like Compounds

The emission of PCDD and PCDFs from the stacks of municipal solid waste combustors has been well characterized through stack gas sampling and laboratory analysis. It is certain that all incineration systems that only incinerate municipal solid waste will emit chlorinated congeners of dibenzo-para-dioxins and dibenzofurans (Rappe,1991; Cleverly,1991; U.S.EPA,1987,1989; Siebert, 1991; OTA,1989). There are no instances whereby dioxin has not been analytically determined to be present in the

stack gas emissions of an operating system. This is true regardless of incinerator design, the type of incinerator technology, the geographical location of the facility, whether processed or unprocessed refuse is incinerated, whether the incinerator is or is not equipped with an air pollution control device, or by the type of air pollution control system that is employed. Therefore the probability that any incinerator burning municipal solid waste will emit to the atmosphere PCDDs and PCDFs is absolute. In other words the probability of occurrence in the stack emissions can be expressed as Pr = 1, or 1 chance in 1.

The emission of congeners of dioxin-like polychlorinated biphenyls is highly uncertain. There exists only limited data of PCB emissions from MSW incinerators, none of which have been speciated on a congener-specific level. The emission of dioxin-like congeners of PCBs was inferred from the limited data of homologue groups of PCBs by applying an assumption that the specific congener has a probability of occurrence equal to the ratio of the individual congener to all congeners existing within the homologue group. There is no way of addressing the reliability of this estimation procedure without congener-specific PCB emissions data. However, if it is the case, as the author has assumed in Chapter 6, that PCB emissions are a consequence of PCB contamination present in the MSW that is burned in the incinerator, and not a product of chemical synthesis, then the emission database can be used in infer a probability or likelihood that PCBs will be emitted from any MSW incinerator. A probability can in implied from recent test reports of the detection of total PCBs in the raw municipal solid waste taken as a sample prior to combustion in an incinerator (U.S. EPA; Environment Canada, 1991). A total of 13 separate samples were taken corresponding to 13 discrete days of the month. No PCBs were measured (level of detection: 0.1 ng/g) in the MSW in 5 of the 13 sampling days. From this limited data a probability of 5 chances out of 13 of detecting measurable PCBs on any day of random sampling can be implied for this particular incinerator, or Pr of 0.39. Again this is only a relative probability based on a small sample that may be specific only to the conditions of the MSW charged to that particular incinerator. There simply is no additional data that could be found in the literature to increase confidence in the estimation. What may be important is the fact that the emission of PCBs may not be a continuous process during MSW combustion. This observation can be sustained by comparing the probability of release of PCB from the stack of the incinerator with the

MSW prior to combustion. The stack tests showed the exact frequency of detection over the same sampling interval, e.g., PCBs were detected in the flue gas 5 out of 13 test samples, or a Pr of 0.39. The stack release of PCBs will be highly dependent on contamination in the raw refuse and the failure of the incineration system to sustain 100 % thermal destruction of the molecules.

Tiernan (1992; personal communication from T. Tiernan, Professor of Chemistry and Director of the Toxic Contaminant Research Program, Wright State University, Dayton Ohio, July, 1992) has reviewed the "reasonableness" of the PCB homologue emissions data as derived in Table 6.7 of Chapter 6. The assumption made to derive that table was that PCB stack emissions are a function of the incomplete thermal destruction of total PCBs present as a contaminant in the MSW. Working with this premise, Tiernan assumed an equal mixture of Aroclors 1016, 1242, 1248, 1254, and 1260 in the MSW as comprising the total PCBs present. By summing the congeners present in this equal distribution of PCB Aroclors for dichloro- through octachloro-biphenyl homologue groups, a normalized distribution can be constructed. This distribution can be compared to the predicted distribution derived in Table 6.7. This comparison is given in Figure 10.1. As can be seen, the relative distribution of congeners derived by Tiernan closely approximates the relative distribution of emission factors indicated in Table 6.7. The exception seems to be PentaCB. This may lend further support to the theory that PCB contamination in the MSW is responsible for PCB air emission from the stack of the MSW incinerator.

10.3.2. The Quantity of Release of Dioxin-Like Compounds From the Stack of the Incinerator

Emission factors of dioxin-like compounds were estimated from test reports of technologies that are most similar in design and operation as the hypothetical MSW incinerator. The use of multiple test reports from multiple facilities of the same basic technology should increase the confidence that the predicted emissions of the contaminants will be in good agreement and fairly representative of emissions from actual facilities of this size, e.g., 2727 metric tons of MSW combusted per operating day. Simplified statistical techniques were used to derive a representative emission factor for each dioxin-like congener. It was assumed that the emissions data for 10 individual incinerators is normally distributed. It was assumed that only state-of-the art incineration

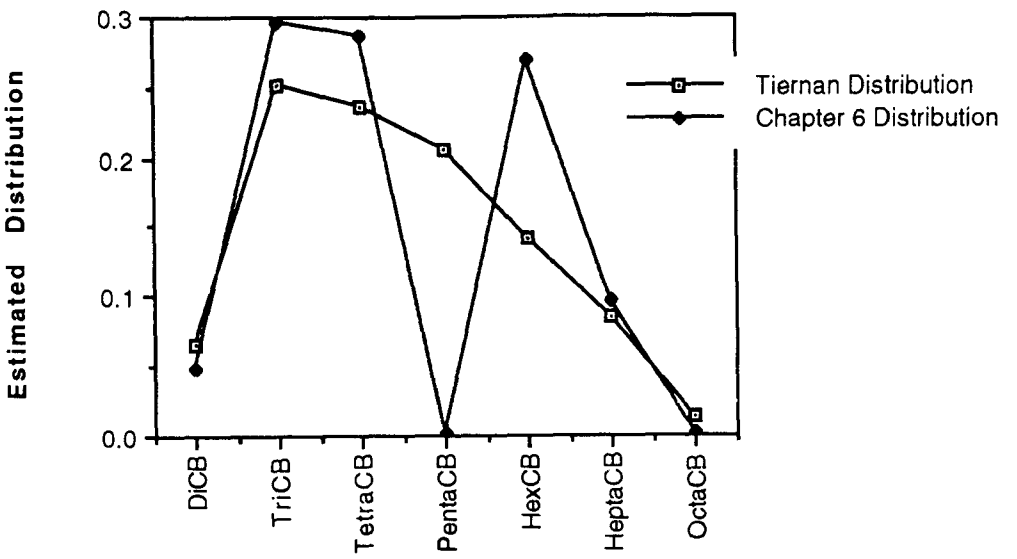

Figure 10.1. Comparison of relative distributions of MSW stack emissions of PCB congeners as derived by Tiernan (see text, Section 10.3.1) and as given in Table 6.7, Chapter 6.

systems would be included in the universe of emissions for purposes of this analysis. Itwas assumed that only data derived under ideal test conditions would be used. No data was included if the test report showed malfunctioning of the incinerator during stack testing. Thus only mass burn heat recovery systems of modern features design to delimit the formation of organic compounds were selected. Only tests reporting the identification of individual congeners of dioxin-like compounds were used, when available, to minimize the necessity for extrapolating unknown values. These selection criteria may present a bias toward underestimating the magnitude of potential emissions in the hypothetical case.

When assuming a normal distribution of the data, the arithmetic mean becomes a measure of central tendency, that is, a measure of the center of the distribution of the finite data set. Statistical measures of the dispersion of the data about the point of central tendency are the standard deviation (s), and the mean absolute percentage deviation (MAPD). In a normal distribution of data, (s) is equal to the positive square root of the variance of the measurements. The Empirical Rule states that given a distribution of measurements that is approximately bell-shaped, the mean plus or minus one standard deviation will contain approximately 68% of the measurements. The MAPD is defined as the absolute deviation of each measurement divided by the mean (times 100%). This gives the percentage deviation (or difference) of every emission measurement from the mean.

Subsequent addition of these percentages divided by the number of measurements, in this case 10 facilities, yields the mean absolute percentage deviation (MAPD).

To investigate the variability of the magnitude of emissions, statistical analysis will focus on three congeners: 2,3,7,8-TCDD (TCDD), 2,3,7,8-TCDF (TCDF), and OctaCDD (OCDD). Table 10-16 is a summary of the statistical analysis of dispersion of the emission measurements for these three specific congeners. The mean mass emission factor for TCDD, OCDD, and TCDF derived from the testing of 10 facilities chosen for this analysis is 4.33 μg/metric ton (Mt) of MSW burned, 150 μg/Mt, and 49.1 μg/Mt, respectively. The range of the measurements is from 0.28 to 15.6 μg/Mt for TCDD; 12.8 to 92.3 μg/Mt for OCDD, and 4.06 to 129 μg/Mt for TCDF. The MAPD is 68.36 % for TCDD, 97.86 % for OCDD, and 83.83 % for TCDF. The standard deviation for the set of measurements is 5.03 μg/Mt for TCDD, and 195 μg/Mt for OCDD, and 47.9 for TCDF.

It appears from this data set that the emission of TCDD, OCDD and TCDF is highly variable. This is the case even when efforts were made to segregate emissions data by design type (mass burn energy recovery), and whether or not combustion engineering principles employed represent state-of-the-art. Although there is an absolute certainty that any incineration system burning MSW will emit dioxin from the stack to the atmosphere, it may be that the magnitude of concentration, hence the magnitude of emission, is a random event. This may be in keeping with the theories supporting a *de novo* synthesis on the surface of particulate matter in regions of the incinerator where the flue gases have cooled to between 200° and 350° C. The rate limiting factors are: relative range of temperature; amount of chlorine in the MSW; the amount of HCl produced and required for acting as a chlorine donor;the presence and amount of phenolic precursor compounds, and the amount of inorganic ions present to act as a catalyst in the chemical synthesis of dioxin. Of these factors, only temperature can be controlled to reduce variability. The other rate limiting factors are random variables, because the physical/chemical composition of MSW burned in the incinerator is constantly changing with time. MSW is heterogenous in composition. If it is true that the magnitude of concentration of the stack emission of dioxin is random and highly variable, then segregating emissions on the basis of "good" incinerators versus "bad" incinerators (for the purpose of deriving a representative emission factor) solely on the basis of the magnitude of concentration of dioxin may be a weak argument, and that a more inclusive data set may be more representative of the

Table 10.16. Basic Statistical Analyses of Emissions of Dioxin Used in the Hypothetical MSW Incinerator Scenario.

Facility	TCDD (μg/Mt)	Ab. Dev[1] (μg/Mt)	Deviat[2] (%)	OCDD (μg/Mt)	Ab Dev.[1] (μg/Mt)	Deviat. (%)
1	2.10	2.23	51.50	12.8	137	91.33
2	7.85	3.52	81.29	15.4	135	90.00
3	9.40	5.08	117.32	206	56	37.33
4	0.39	3.93	90.76	60.80	89.2	59.47
5	1.94	2.38	54.97	37.6	112	74.67
6	15.6	1.13	26.10	374	224	149.33
7	1.97	2.36	54.50	604	454	302.67
8	0.28	4.05	93.53	14.3	136	90.67
9	2.91	1.41	32.56	92.3	57.7	38.47
10	0.82	3.51	81.06	83.0	67.0	44.67
Mean[a]	4.33			150		
St.Dev[b]	5.03			195		
MAPD[c]		68.36%				97.86

Facility	TCDF (μg/Mt)	Ab. Dev[1] (μg/Mt)	Deviat[2] (%)
1	NA	NA	NA
2	105	55.9	113.85
3	129	79.9	162.73
4	11.4	37.7	76.78
5	24.4	24.7	50.31
6	78.0	28.9	58.86
7	NA	NA	NA
8	18.0	31.1	63.34
9	23.0	26.1	53.16
10	4.06	45.0	91.65
Mean[a]	49.1		
St.Dev[b]	47.9		
MAPD[c]			83.83%

Key:

[a] arithmetic mean
[b] standard deviation
[c] mean absolute percentage deviation
[1] absolute deviation from the mean
[2] percentage deviation from the mean

wide range of potential variation in incinerator emissions. If a more inclusive set of emissions from existing databases on mass burn, heat recovery incinerators (equipped with electrostatic precipitators) are included in the universe of emission factors, the range of potential average emissions of 2,3,7,8-TCDD would span roughly four orders of magnitude in relative concentration, e.g., 0.2 μg/Mt to 2000 μg/Mt (U.S.EPA, 1987). No such variation was noted in the data set in this analysis. Emissions from incinerators equipped with dry-scrubbers combined with fabric filters are from 1 % to 10 % of emissions from incinerators equipped with electrostatic precipitators (U.S. EPA, 1989; U.S. EPA,1987). This provides additional information to support a hypothesis that the formation and subsequent emission of chlorinated dioxins and dibenzofurans from the incineration of MSW is absolutely certain, but the magnitude of concentration of emission may be random and highly variable over some discrete time interval. The hypothesis warrants further testing through additional combustion engineering research. It may be that emission factors that show small variability, high correlations with plant operations (such as temperature of the flame in the combustion zone, and % carbon monoxide monitored in stack emissions), were derived as a result of committing a type I error, e.g, the researcher wishes to select critical measurements so that the variance is small. Critical data may be excluded by some subjective criteria that, in itself, becomes difficult to measure the overall effect on the distribution of data. These criteria may include one or more of the following: (1) eliminating all facilities without heat recovery; (2) eliminating facilities that experience anomalies in operation during test periods; (3) eliminating facilities on the basis of some arbitrary capacity cutoff; (4) eliminating facilities on the subjective basis that the emission appears to be too great in magnitude.

10.3.3. The Representativeness of the Hypothetical Incinerator In Comparison to Actual Facilities

A third element to investigating uncertainty in the analysis of the emission of dioxin-like compounds is the "representativeness" of the hypothetical incinerator. This can be addressed by examining the distribution of design type of existing and projected facilities; by examining the distribution of plant capacity, e.g., metric tons incinerated over 24 hours; by comparisons to the distribution of stack height relative to plant capacity.

In 1989, the year when figures were last available, there were approximately 200

municipal solid waste incineration facilities operating in the U.S. (U.S.EPA,1989). Figures 10-2 and 10-3 are an approximate distribution (broken down by percent mass of total MSW combusted) of existing incineration technologies operating in the U.S, and technologies projected to be operational in the year 1995. From these distributions it is apparent that the mass burn technology currently dominates, and will continue to dominate, the type of technology incinerating most of the MSW in the U.S. Therefore the selection of the mass burn technology for the hypothetical incinerator is representative of actual and projected technologies.

A second issue relevant to the question of how representative the hypothetical incinerator is of existing technology is the combustion capacity. The hypothetical incinerator has a combustion capacity of 2727 Mt per day. Figure 10-4 is a histogram of the distribution of mass burn MSW technologies by combustion capacity. The capacity of 2727 metric tons per day is within the upper 75th percentile in the distribution of mass burn MSW incinerators that exist and are planned out to the year 1995. This means that 3 out of every 4 of the mass burn incinerators are smaller in capacity than the capacity chosen for the hypothetical case.

A third area involving representativeness is the height of the stack assumed for the hypothetical case. The stack height that is assumed is 46 meters. Figure 10-5 relates facility capacity to stack height of 117 existing MSW incinerators operational in the U.S. Although there is a poor correlation between capacity and stack height, e.g., $r^2 = .37$, the distribution gives some reference to the selection made. The scatter plot of 117 existing MSW incinerators in the U.S. shows that for the capacity of 2727 Mt/day, the stack height of 46 meters is fairly representative of actual facilities, and is a reasonable assumption.

The final criteria addressing "representativeness" of the hypothetical MSW incinerator is the type of pollution control devices assumed for control of dioxin emissions. The hypothetical facility has two pollution control scenarios: electrostatic precipitators (ESPs), and dry scrubber combined with a fabric filter (DSFF). The distribution of air pollution control technologies of the existing MSW incinerators is as follows: 43.67% are controlled by ESPs; 20.25% are controlled by DSFF; 5.7% are controlled by dry scrubber combined with ESPs; and 13.29% have no form of pollution control (EPA, 1992a). Therefore, about 64% of all existing MSW incinerators are controlled by either ESPs or

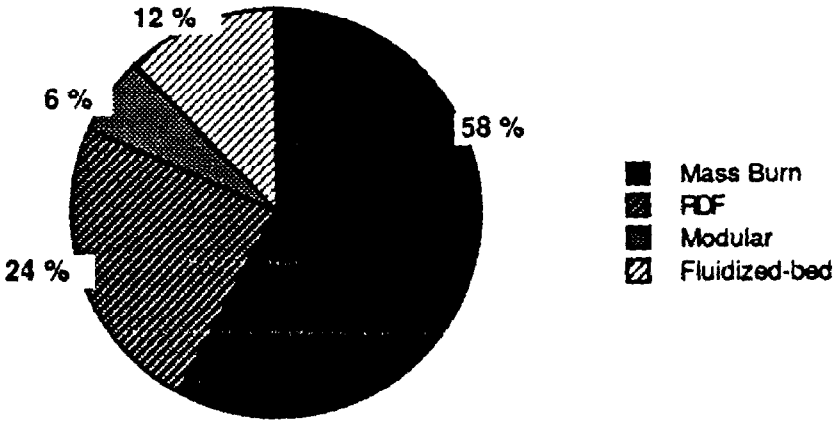

Figure 10-2. Percent distribution of existing incinerators.

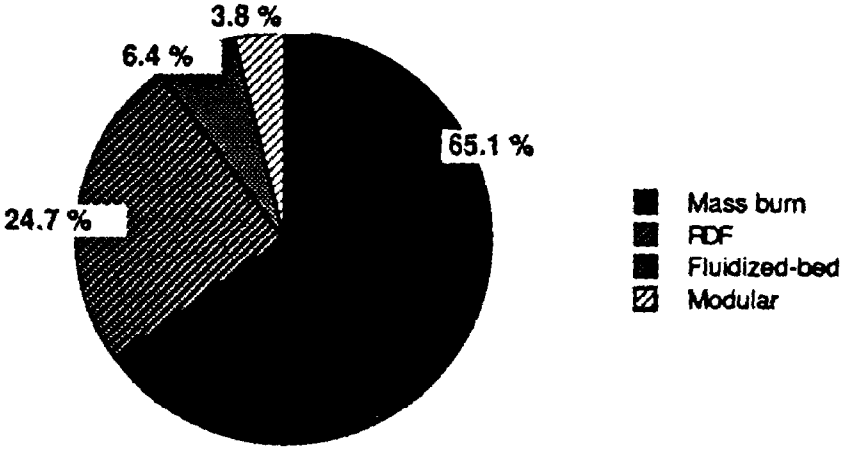

Figure 10-3. Percent distribution of incinerators by 1995.

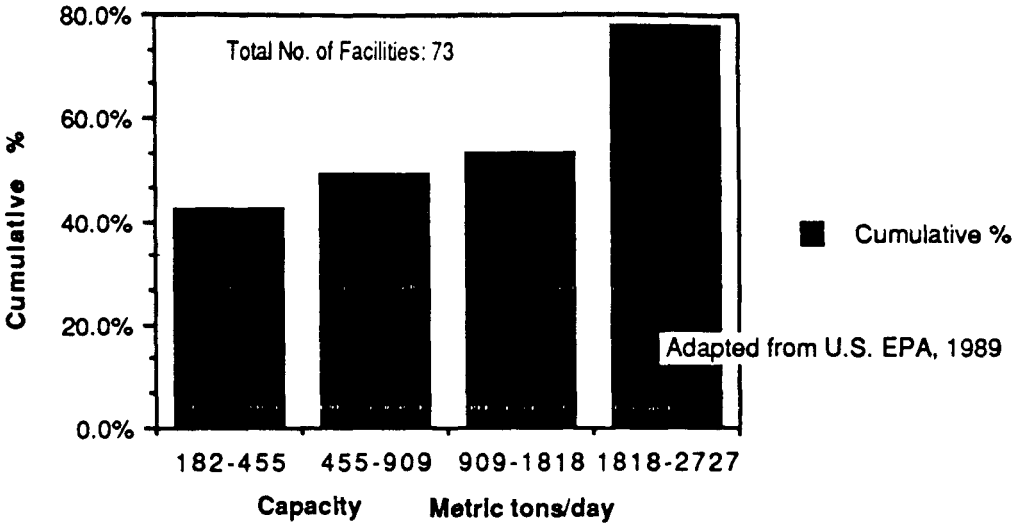

Figure 10-4. Capacity distribution of existing and planned mass burn incinerators.

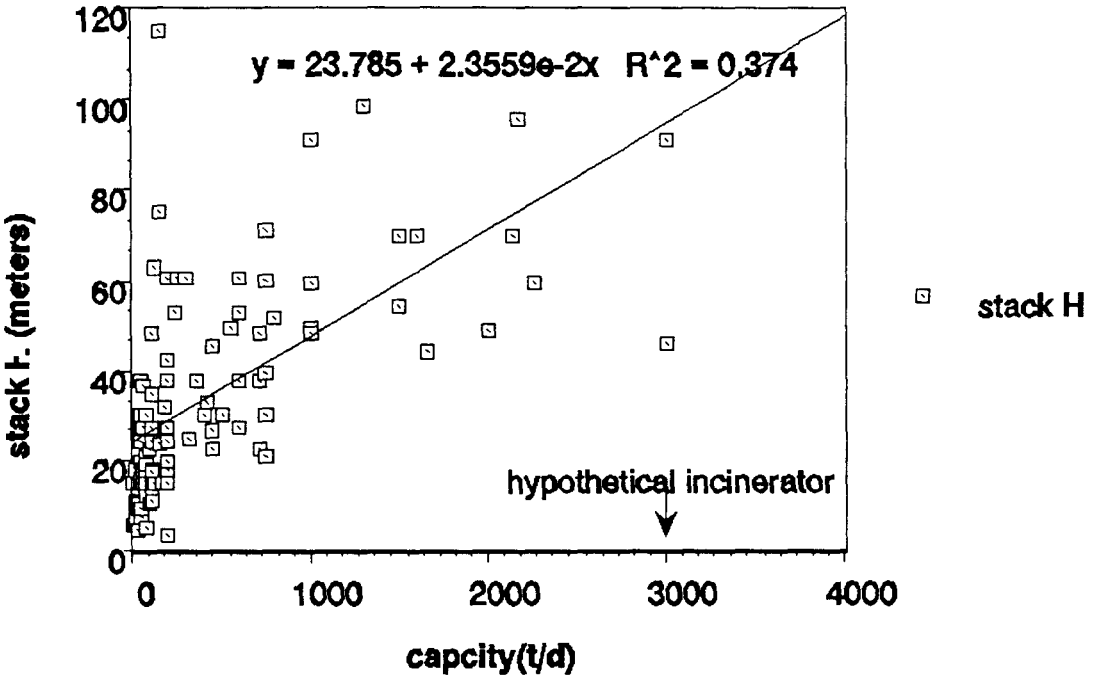

Figure 10.5. Scatter plot of stack height relative to capacity for 117 MSW incinerators operation in the United States (EPA, 1987).

DSFF. Based on this distribution, the control scenarios assumed for the hypothetical case seem reasonable.

10.3.4. Air Dispersion Analysis of Incinerator Emissions

Air dispersion analysis was performed using the Industrial Source Complex Model. The model is intended to give approximate estimations of atmospheric dispersion and wet and dry deposition flux, and does not give absolute values. The model is used in the context of predicting future states based on known characteristics of geographical location, local meteorological conditions, temporal rates of emissions, and the physical description of the facility.

The ISC is a Gaussian plume model. Downwind concentrations of the dioxin-like chemicals are calculated as a function of stack height, the mass emission rate, the wind speed, and general atmospheric conditions. The Gaussian model assumes that the emission concentrations predicted by the model will fit a normal distribution. The principal assumptions in the Gaussian model are (Kapahi, 1991):

- The air concentration of the chemical at a fixed distance from the source is directly proportional to the emission rate from the source;
- The air concentration of a given chemical is inversely proportional to the wind speed corresponding to the effective height of release of the chemical into the air;
- The predicted ground-level concentration of the chemical approaches zero at large distances from the initial point of release.
- The model is steady-state.
- The model assumes constant wind speed, wind direction, and atmospheric stability over time and space for a given time period.

In general the stochastic features of the model have been shown to predict annual average ambient air concentrations of a chemical emission from an industrial source to within a factor of one-order of magnitude of measured values, and in some cases, within a factor of 3 to 4 -fold of field measurements (Cohrssen ,1989). This modeling error spans both sides of the predicted concentration, that is the actual concentration may be plus or minus this amount of the predicted value. The most sensitive aspects to variability in modeled predictions of ambient air impacts, if emissions are held constant, are stack height (height of the release), and terrain (flat verses complex topography). To investigate

modeling variability, the EPA placed a prototype hypothetical hazardous waste incinerator in flat terrain and elevated terrain in geographical areas around the U.S. (U.S. EPA, 1991a). Then the stack height was varied at these particular locations. Numerous runs of the ISC model were made at twelve specific sites to compare and contrast the influence of stack height and terrain on predicted ambient air concentrations of various mass emission rates of specific inorganic pollutants. A series of tables were developed from this sensitivity analysis from which the numerical estimation of the variability as a function of stack height and terrain can be inferred. When the hypothetical hazardous waste incinerator was modeled in flat terrain, e.g., topography within a distance of 5 km is not above the height of the stack, and the stack height was varied from 4 meters to 120 meters, the variability in the predicted ambient air concentration spanned two orders of magnitude (100). The lower stack height resulted in a predicted ambient air concentration that was 100 times greater than the concentration predicted using the tallest stack height. When the hypothetical hazardous waste incinerator was located in complex terrain over the same range of physical stack heights, the variability in estimated groundlevel concentration of the subject pollutant spanned two orders of magnitude (100-fold). In the latter case the stack height was computed as the terrain-adjusted stack height by subtracting from the physical stack height the influence of terrain on plume rise. From the limited sensitivity analysis of hazardous waste incinerators it can be assumed that the predictions of spacial ground-level ambient air concentrations of dioxin-like chemicals could differ from values in Tables 6-16 and 6-17 by two-orders of magnitude in consideration of changes in stack height or changes in terrain. For example, Table 6-16 shows that the maximum annual average ambient air concentration of 2,3,7,8-TCDD predicted near the hypothetical MSW incinerator is 1.65 femtograms/cubic meter (fg/m^3) for the stack height of 46 meters, and assuming flat terrain. If only the stack height is varied from 20 meters to 120 meters, and all other modelling parameters are held constant, then the predicted ambient air concentration would be approximately 10 times greater and 10 times less than the estimated concentration, respectively. The uncertainty is broader when considering the influence of topography on predictability of the ground-level concentrations from the model. If only terrain elevation is varied at a distance of 5 km from the hypothetical MSW incinerator from zero elevation to 46 meters, e.g., the height of the stack, then the predicted ambient air concentration of 2,3,7,8-TCDD would be approximately ten times

greater. The tables derived in the hazardous waste incineration analysis have a limitation of elevation of terrain to the height of the stack.

The most uncertain aspect to the modeling is the estimation of dry and wet deposition flux of dioxin-like compounds on the vicinity of the hypothetical MSW incinerator. Contributing most to this uncertainty seems to be the settling velocities and scavenging coefficients estimated for specific particle size diameters (Cohrssen, 1989; Doran, 1985). Seinfeld (1986) found that particles over 20 microns in diameter settle primarily by gravity, whereas smaller particles deposit primarily by atmospheric turbulence and molecular diffusion. Considerable, but non-quantifiable, uncertainty exists with respect to deposition velocities of particles 0.1 to 1.0 microns in diameter (Seinfeld,1986). The uncertainty is difficult to define. The wide variation of predicted deposition velocities as a function of particle size, atmospheric turbulence and terrain adds to this uncertainty (Sehmel, 1980). However, Gaussian plume dispersion models have been field validated for their ability to spatially predict dry deposition flux over some specified distance (Doran, 1985). In a series of field experiments conducted by Pacific Northwest Laboratory (Doran, 1985) zinc sulfide was used as a depositing tracer gas, and sulfur hexafluoride was used as a non-depositing tracer gas to compare and contrast modeling results with field measurements of dry deposition and atmospheric diffusion of the gases. The tracer was released from a height of 2 meters, and all releases were made under relatively stable atmospheric conditions. Five sampling stations were located downwind of the release from 100 to 3200 meters. The results of these experiments showed good agreement with the predicted verses the measured deposition of the tracer ZnS. The overall correlation coefficient between predicted and measured deposition concentration was found to be 0.82 (Doran, 1985), but the models marginally over-predicted deposition flux near the source of release, and under-predicted deposition flux at 3200 meters.

Travis and Yambert (1991) have evaluated the uncertainty in modeling the dry deposition flux of particulates using four standard Gaussian plume dispersion models. Since deposition flux is dependent on deposition velocity for a given particle mass and diameter, comparisons were made between model-generated deposition velocities and measured values found in the open literature for particles ranging from 0.01 to 30 microns in diameter. It was found that measured deposition velocities for a given particle size in the scientific literature exhibit variability spanning roughly two orders of magnitude. The

analysis of the mean predicted deposition velocities to mean measured values showed that most measured data exceeded the predicted data for all four models. Moreover, the models underestimated the mean deposition velocities for particles in the range of diameters from 0.05 to 1.0 microns.

Similar uncertainty probably exists with regard to scavenging of various diameter particles by various intensity of rainfall. Seinfeld (1986) has calculated scavenging coefficients in terms of the removal efficiency of particles of a given size by rain droplets having a given momentum. Seinfeld (1986) found that the scavenging coefficient of a given particle diameter corresponding to a given rainfall intensity can be calculated based on physical laws, but there is a complete absence of research data to verify these calculations. Hence it is not possible to address the accuracy nor uncertainty of the wet deposition flux estimated in Tables 6-18 and 6-19.

10.3.5 Comparison of predicted ambient concentrations to ambient measurements

In the analysis of the environmental impact associated with the emissions from the hypothetical MSW incinerator, predictions have been made with respect to the spacial and temporal distribution of ambient air concentrations of specific dioxin-like compounds (Tables 6-16, 6-17). Comparing these predictions to ambient air measurements taken downwind of an operating MSW incinerator provides a qualitative check of the relative correctness of the estimated concentrations. There are certain caveats to such a comparison:

- Measured concentrations are in different geographical settings from the hypothetical case, and therefore reflect different meteorological conditions and topography.
- Measured concentrations most likely reflect stack emissions of dioxin-like compounds from the actual incinerator that significantly differ from the hypothetical case. No stack measurements were provided by the investigator in the literature cited.
- Measured values are a "snapshot" in time. Therefore comparisons are between annual average predicted concentrations and less than annual average monitored concentrations.
- Measured values may reflect differences in analytical sensitivities; ambient monitoring instrumentation; and statistical reliability of placement of the sampling network

with respect to measuring the true ambient air impact.

Given these limitations in the database, average ambient air concentrations of the dioxin-like congeners measured in the vicinity of four operational MSW incinerators were summarized and compared with the predicted annual average ambient air impact from the hypothetical incinerator. Table 10-17 is a summary of this data. The range of measurements specific to each congener is reported in the Table. The qualitative "reality" check of the adequacy of the modeling of air emissions from the hypothetical case involves a comparison of the range of measured data to see if the predicted values fall within the range. As can be determined from the analysis, most of the predicted values do fall within the range of measured values.

Table 10-17. Comparison between measured ambient air concentrations with predicted ambient air impacts of the hypothetical MSW incinerator (values as pg/m^3)

Congeners	Reference # 1	2	3	4	Range	Modeled	Within Range
2,3,7,8-TCDD	0.004	0.238	0.020	0.015	0.004 to 0.238	0.002	Yes
1,2,3,7,8-PeCDD	0.011	0.010	0.220	0.015	0.011 to 0.220	0.015	Yes*
1,2,3,4,7,8-HxCDD	0.025	0.005	0.019	0.243	0.005 to 0.190	0.004	Yes*
1,2,3,6,7,8-HxCDD	0.023	0.013	0.710	0.390	0.013 to 0.710	0.010	Yes*
1,2,3,7,8,9-HxCDD	0.011	0.015	0.360	0.063	0.011 to 0.063	0.004	No
1,2,3,4,6,7,8-HpCDD	0.040	0.150	NA	2.000	0.041 to 2.000	0.008	No
OctaCDD	0.330	1.590	7.400	2.950	0.330 to 7.400	0.062	No
2,3,7,8-TCDF	0.006	0.031	0.380	1.470	0.006 to 1.470	0.020	No
1,2,3,7,8-PeCDF	0.010	0.019	0.420	0.247	0.010 to 0.420	0.020	Yes
2,3,4,7,8-PeCDF	0.010	0.029	0.430	0.687	0.010 to 0.687	0.020	Yes
1,2,3,4,7,8-HxCDF	0.030	0.050	0.270	0.110	0.030 to 0.270	0.012	Yes*
1,2,3,6,7,8-HxCDF	0.030	0.022	0.240	0.453	0.030 to 0.453	0.010	Yes*
1,2,3,7,8,9-HxCDF	0.020	0.005	0.020	0.043	0.005 to 0.043	0.010	Yes
2,3,4,6,7,8-HxCDF	0.030	0.010	0.210	0.763	0.010 to 0.763	0.008	Yes*
1,2,3,4,6,7,8-HpCDF	0.060	0.135	NA	2.080	0.060 to 2.080	0.010	No
1,2,3,4,7,8,9-HpCDF	0.040	0.010	NA	NA	0.010 to 0.040	0.0003	No
OctaCDF	0.010	0.059	0.780	0.748	0.010 to 0.780	0.030	Yes

References: 1. Conn. DEP,1988; 2. Conn. DEP,1989; 3.Rappe, 1987; 4. Smith, 1989.

Notes: a. All values are average values. b. Non detects were treated as measured values.

NA = Not Available.

* Predicted value judged to be close to the lower bound of the range.

REFERENCES FOR CHAPTER 10

Ambrose, R.B., T.A. Wood, J.P. Connolly, and R.W. Schanz. 1988. WASP4, A Hydrodynamic and Water Quality Model -- Model Theory, User's Manual, and Programmer's Guide. Athens Environmental Research Laboratory, Office of Research and Development, US EPA. EPA/600/3-87/039.

Bacci, E., D. Calamari, C. Gaggi, M. Vighi. 1990. Bioconcentration of organic chemical vapors in plant leaves: experimental measurements and correlation. Environmental Science and Technology 24: 885-889.

Bacci, E., M.J. Cerejeira, C. Gaggi, G. Chemello, D. Calamari, and M. Vighi. 1992. Chlorinated dioxins: volatilization from soils and bioconcentration in plant leaves. In Press in Bulletin of Environmental Contamination and Toxicology.

Baes, C.F., R.D. Sharp, A.L. Sjoreen, and R.W. Shor. 1984. A review and analysis of parameters for assessing transport of environmentally released radionuclides through agriculture. Prepared for the U.S. Department of Energy under contract no. DE-AC05-840R21400.

Beck, H., K. Eckart, W. Mather, and R. Wittkowski. 1989. PCDD and PCDF body burden from food intake in the Federal Republic of Germany. Chemosphere 18: 417-424.

Bopp, R.F., M.L. Gross, H. Tong, M.J. Simpson, S.J. Monson, B.L. Deck, and F.C. Moser. 1991. A major incident of dioxin contamination: sediments of New Jersey estuaries. Environmental Science and Technology 25: 951-956.

Brady, N.C. 1984. The Nature and Properties of Soils. Ninth Edition. Macmillan Publishing Company. New York, NY 10022.

Briggs, G.G., R.H. Bromilow, and A.A. Evans. 1982. Relationships between lipophilicity and root uptake and translocation of non-ionised chemicals by barley. Pesticide Science 13:495-504.

Cleverly, D.H.; Morrison, R.M.; Riddle, B.L.;Kellam, R.G. (1991). Regulatory analysis of pollutant emissions, including polychlorinated dibenzo-p-dioxins (CDDs) and dibenzofurans (CDFs), from municipal waste combustors. In: Hattermer-Frey, H.A.; Travis,C., eds., Health effects of municipal waste incineration. Boca Raton,FL., CRC Press, pp.47-65.

Coccusi, S., F. DiGerolamo, A. Verderio, A. Cavallaro, G. Colli, A. Gorni, G. Invernizzi, and L. Luciani. 1979. Absorption and translocation of tetra-chlorodibenzo-p-dioxin by plants from polluted soil. Experientia 35(4): 482-484.

Cohrssen, J., Covello, V. 1989. Risk analysis: A guide to principles and methods for analyzing health and environmental risks. U.S. Council on Environmental Quality, Office of the President. National Technical Information Service #PB 89-137772.

Conn DEP. 1988. Ambient background monitoring for selected PCDDs/PCDFs in the vicinity of the Mid-Connecticut Resource Recovery Facility. Connecticut Department of Environmental Protection, Air Compliance Unit, Hartford, CT. Project report by Combustion Engineering,Inc,Windsor,CT., Document # 1790-002/6350-001.Final Report. September,1988.

Conn DEP. 1989. Ambient background monitoring for selected PCDDs/PCDFs in the vicinity of the Bristol Resource Recovery Facility.Connecticut Department of Environmental Protection,Air Compliance Unit, Hartford, CT. Project report by Ogden Martin Systems,Fairfield,NJ. Document # 5160-003-400.Final Report. February 1989.

Connett, P. and T. Webster. 1987. An estimation of the relative human exposure to 2,3,7,8-TCDD emissions via inhalation and ingestion of cow's milk. Chemosphere 15: 2079-2084.

Connolly, J.P. and C.J. Pedersen. 1987. A thermodynaic-based evaluation of organic chemical accumulation in aquatic organisms. Environmental Science and Technology 22:99-103.

Cook, P.M., M.K. Walker, D. W. Kuehl, and R.E. Peterson. 1991. p. 143-168 in Gallo, M.A., R.J. Scheuplein, and K.A. Van Der Heijden, eds., Banbury Report 35: Biological Basis for Risk Assessment of Dioxins and Related Compounds. Cold Spring Harbor Laboratory Press, Plainview, New York 11803.

Doran, J.C., Horst, T.W. (1985). An evaluation of gaussian plume-depletion models with dual-tracer field measurements. Atmos. Environ. 19:939-951.

Foster, G.R., and T.E. Hakonson. 1987. Erosion Losses of Fallout Plutonium. In Howard, W.A., and R. Fuller, eds. Dynamics of Transuranics and Other Radionuclides in Natural Environments. NTIS-87014456. Springfield, VA.

Fries, G.F. 1985. Bioavailability of soil-borne polybrominated biphenyls ingested by farm animals. Journal of Toxicology and Environmental Health 16: 565-579.

Fries, G.F., and L.W. Jacobs. 1986. Evaluation of Residual Polybrominated Biphenyl Contamination Present on Michigan Farms in 1978. Research Report 477 from the Michigan State University Agricultural Experiment Station, East Lansing, Michigan. December, 1986.

Fries, G.F. and D.J. Paustenbach. 1990. Evaluation of potential transmission of 2,3,7,8-tetrachlorodibenzo-p-dioxin-contaminated incinerator emissions to humans via foods. Journal of Toxicology and Environmental Health 29: 1-43.

Furst, P., C. Furst, and W. Groebel. 1990. Levels of PCDs and PCDFs in food-stuffs from the Federal Republic of Germany. Chemosphere 20: 787-792.

Gillette, D.A. 1981. Production of dust that may be carried greate distances. In: Desert Dust: Origin, Characterists, and Effect on Man. edited by Troy Pewe. Geological Society of America Special Paper 186, pp. 11-26.

Gotz, R., and E. Schumacher. 1990. Polychloriete dibenzo-p-dioxine (PCDDs) und plychlorierte dibnezofurane (PCDFs) in sedimenten und fischen Aus Dem Hamburger Hafen. Chemosphere 20: 51-73.

Hawley, J.K. 1985. Assessment of health risk from exposure to contaminated soil. Risk Analysis 5(4): 289-302.

Huckins, J.N. T.R. Schwartz, J.D. Petty, L.M. Smith. 1988. Determination, fate, and potential significance of PCBs in fish and sediment samples with emphasis on selected AHH-Inducing congeners. Chemosphere 17: 1995-2016.

Hulster, A. 1991. Transfer of PCDD/PCDF from Contaminated Soils into Two Vegetable Crops. Unpublished data presented in a poster at Dioxin 91.

Hwang, S.T., J.W. Falco, and C.H. Nauman. 1986. Development of Advisory Levels for Polychlorinated Biphenyls (PCBs) Cleanup. Exposure Assessment Group, Office of Research and Development, U.S. Environmental Protection Agency. EPA/600/6-86/002.

Isensee, A.R.; Jones, G.E. 1971. Absorption and translocation of root and foliage applied 2,4,-dichlorophenol, 2,7-dichlorodibenzo-p-dioxin and 2,3,7,8-tetrachlorodibenzo-p-dioxin. J. Agric. Food Chem. 19(6): 1210-1214.

Jackson, D.R., M.H. Roulier, H.M. Grotta, S.W. Rust, and J.S. Warner. 1986. Solubility of 2,3,7,8-TCDD in Contaminated Soils. pp. 185-200 in Rappe, C., G. Choudhary, and L.H. Keith, eds., Chlorinated Dioxins and Dibenzofurans in Perspective. Lewis Publishers, Inc.

Javitz, H. 1980. Seafood Consumption Data Analysis. SRI International. Final report prepared for EPA Office of Water Regulations and Standards. EPA Contract 68-01-3887.

Jensen, D.J. and R.A. Hummel. 1982. Secretion of TCDD in milk and cream following the feeding of TCDD to lactating cows. Bulletin of Environmental Contamination and Toxicology 29: 440-446.

Jensen, D.J., R.A. Hummel, N.H. Mahle, C.W. Kocker, and H.S. Higgins. 1981. A Residue study on beef cattle consuming 2,3,7,8-tetrachlorodibenzo-p-dioxin. Journal of Agriculture and Food Chemistry 29: 265-268.

Kapahi, R. 1991. Modeling the dispersion of toxic air pollutants emitted from municipal solid waste incinerators.In: Hattermer-Frey, H.A.; Travis,C., eds., Health effects of municipal waste incineration. Boca Raton,FL., CRC Press, pp.67-82.

Karickhoff, S.W., D.S. Brown, and T.A. Scott. 1979. Sorption of hydrophobic pollutants on natural sediments. Water Research 13: 241-248.

Kenaga, E.E. and C.A.I. Goring. 1980. Relationship between water solubility, soil-sorption, octanol-water partitioning, and bioconcentration of chemicals in biota. In Eaton, J.SC., Parrish, P.R., and Hendrics, A.C. (eds). Aquatic Toxicology. ASTM STP 707. Philadelphia, PA: American Society for Testing and Materials.

Kjeller, L.O., S.E. Kulp, S. Bergek, M. Bostrom, P.A. Bergquist, C. Rappe, B. Jonsoson, D.De Wit, B. Jansson, and M. Olsson. 1990. Levels and possible sources of PCDD/PCDF in sediment and pike samples from Swedish lakes and rivers (Part One). Chemosphere 20: 1489-1496.

Kuehl, D.W., B.C. Butterworth, A. McBride, S. Kroner, and D. Bahnick. 1989. Contamination of fish by 2,3,7,8-tetrachloridibenzo-p-dioxin: a survey of fish from major watersheds in the United States. Chemosphere 18: 1997-2017.

Linsley, R.K., M.A. Kohler, and J.L.H. Paulus. 1982. Hydrology for Engineers. McGraw-Hill, Inc., New York, NY.

Lower, W.R., A.F. Yanders, C.E. Orazio, R.K. Puri, J. Hancock, and S. Kapila. 1989. A survey of 2,3,7,8-tetrachlorodibenzo-p-dioxin residues in selected animal species from times Beach, Missouri. Chemosphere 18: 1079-1088.

Lyman, W.J., W.F. Reehl, and D.H. Rosenblatt. 1982. Handbook of Chemical Property Estimation Methods. Mcgraw-Hill, Inc., New York, NY.

McCrady, J.K, and S.P. Maggard. 1992. The uptake and photodegradation of 2,3,7,8-TCDD sorbed to grass foliage. Draft, to be submitted for publication.

McCrady, J.K., C. McFarlane, and L.K. Gander. 1990. The transport and fate of 2,3,7,8-TCDD in soybean and corn. Chemosphere 21:359-376.

McKone, T.E. and P.B. Ryan. 1989. Human exposures to chemicals through food chains: an uncertainty analysis. Environmental Science and Technology 23: 1154-1163.

Meyer, C., D. O'Keefe, D. Hilker, L. Rafferty, L. Wilson, S. Conner, K. Aldous, K. Markussen, and K. Slade. 1989. A survey of twenty community water systems in New York State for PCDDs and PCDFs. Chemosphere 19:21-26.

Michaels, R. 1989. Health Risk Assessment for the Planned St. Lawrence County, New York, Resource Recovery Facility. RAMTRAC, Long Island City, New York.

MRI, 1990. Special Management Standards for Municipal Waste Combustion (MWC) Ash. Submitted by: Midwest Research Institute, 425 Volker Boulevard, Kansas City, MO 64110-2299, to US EPA, Municipal sSolid Waste Program. EPA Contract No. 68-01-7287. June 29, 1990.

Nestrick, T.J., L.L. Lamparski, N.N. Frawley, R.A. Hummel, C.W. Kocher, N.H. Mahle, J.W. McCoy, D.L. Miller, T.L. Peters, J.L. Pillepich, W.E. Smith, and S.W. Tobey. 1986. Perspectives of a large scale environmental survey for chlorinated dioxins: overview and soil data. Chemosphere 15: 1453-1460.

Niimi, A.J., and B.G. Oliver. 1989. Distribution of polychlorinated biphenyl congeners and other halocarbons in whole fish and muscle from Lake Ontario salmonids. Environmental Science and Technology 23: 83-88.

Norwood, C.B, M. Hackett, R.J. Powell, B.C. Butterworth, K.J. Williamson, and S. M. Naumann. 1989. Polychlorinated dibenzo-p-dioxins and dibenzofurans in selected estuarine sediments. Chemosphere 18: 553-560.

Oliver, B.G., and A.J. Niimi. 1988. Trophodynamic analysis of polychlorinated biphenyl congeners and other chlorinated hydrocarbons in the Lake Ontario ecosystem. Environmental Science and Technology 22: 388-397.

OTA. (1989). Facing America's trash. What next for municipal solid waste? Office of Technology Assessment,U.S.Congress, Washington, DC.

Pao, E.M., K.H. Fleming, P.M. Guenther, et al. 1982. Foods commonly eaten by individuals: amount per day and per eating occassions. U.S. Department of Agriculture. home Economics Report No. 44.

Parker, C.E., W.A. Jones, H.B. Matthews, E.E. McConnell, and J. Hass. 1980. The chronic toxicity of technical and analytical pentachlorophenol in cattle. II. Chemical analysis of tissues. Toxic. Appl. Pharmacol. 55: 359-369.

Parkerton, T.F. 1991. Development of a Generic Bioenergetics-Based Model for Predicting the Bioaccumulation of Persistent Sediment-Associated Contaminants - Final Report, New Jersey Department of Environmental Protection, Trenton, NJ.

Poiger, H. and C. Schlatter. 1980. Influence of solvents and adsorbents on dermal and intestinal adsorption of TCDD. Food Cosmet. Toxicol 18: 477-481.

Rappe, C. 1991. Introduction, profiles, and patterns. In: Procedings of the 11th international symposium on chlorinated dioxins and related compounds, Research Triangle Park, NC, September 23-27, 1991, pg 17.

Rappe, C., P.A. Bergqvist, and L.A. Kjeller. 1989. Levels, trends and patterns of PCDDs and PCDFs in Schandinavian environmental samples. Chemosphere 18: 651-658.

Rappe, C., and L.O. Kjeller. 1987. PCDDs and PCDFs in environmental samples, air, particulates, sediments, and soil. Chemosphere 16: 1775-1780.

Rappe, C., M. Nygren, G. Lindstrom, H.R. Buser, O. Blaser, and C. Wuthrich. 1987. Polychlorinated Dibenzofurands and Dibenzo-p-dioxins and other chlorinated contaminants in cow milk from various locations in Switzerland. Environmental Science and Technology. 21: 964-970.

Reed, L.W., G.T. Hunt, B.E. Maisel, M. Hoyt, D. Keefe, and P. Hackney. 1990. Baseline assessment of PCDDs/PCDFs in the vicinity of the Elk River, Minnesota generating station. Chemosphere 21: 159-171.

Sacchi, G.A., P. Vigano, G. Fortunati, S.M. Coccusi. 1986. Accumulation of 2,3,7,8-TCDD from soil and nutrient solution by bean and maize plants. Experientia 42: 586-588.

SAIC. 1986. Science Applications International Corp. Conversion factors for use in estimating environmental transport of contaminated soil from unregulated disposal sites. Prepared for the U.S. Environmental Protection Agency, Office of Solid Waste, Washington, D.C., under EPA contract 68-01-7624.

Sehmel, G.A. 1980. Paritcle and gas dry deposition. A review. Atmos. Environ., vol. 14: 983-1012.

Seinfeld, J.H. 1986. Atmospheric chemistry and physics of air pollution. New York,NY., John Wiley and Sons.

Siebert, P.C.; Alston-Guiden, D.; Jones, K.H. (1991). An analysis of worldwide resource recovery emissions data and the implications for risk assessment.In: Hattermer-Frey, H.A.; Travis,C., eds., Health effects of municipal waste incineration. Boca Raton,FL., CRC Press, pp.2-46.

Smith, L.M., T.R. Schwartz, and K. Feltz. 1990. Determination and occurrence of AHH-Active polychlorinated biphenyls, 2,3,7,8-tetrachloro-p-dioxin and 2,3,7,8-tetrachlodibenzofuran in Lake Michigan sediment and biota. The question of their relative toxicological significance. Chemosphere 21: 1063-1085.

Startin, J.R., M. Rose, C. Wright, I. Parker, and J. Gilbert. 1990. Surveillance of British foods for PCDDs and PCDFs. Chemosphere 20: 793-798.

Stevens, J.B. and E.N. Gerbec. Dioxin in the agricultural food chain. Risk analysis 8(3): 329-335.

Swackhammer, D.L., B.D. McVeety, and R.A. Hites. 1988. Deposition and evaporation of polychlorinated congeners to and from Siskiwit Lake, Isle Royale, Lake Superior. Environmental Science and Technology 22: 664-672.

Thomann, R.V. 1989. Bioaccumulation model of organic chemical distribution in aquatic food chains. Environmental Science and Technology 23: 699-707.

Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. An equillibrium model of organic chemical accumulation in aquatic foodwebs with sediment interaction. In Press, to appear in Environmental Toxicology and Chemistry.

Travis, C.C. and A.D. Arms. 1988. Bioconcentration of Organics in Beef, Milk, and Vegetation. Environmental Science and Technolology, Vol 22(3): 271-274.

Travis, C.C. and H.A. Hattemer-Frey. 1987. Human exposure to 2,3,7,8-TCDD. Chemosphere 16: 2331-2342.

Travis, C.C. and H.A. Hattemer-Frey. 1991. Human exposure to dioxin. The Science of the Total Environment 104: 97-127.

Travis, C., Yambert, M. (1991). Accounting for dry depostion in incinerator risk assessments. In:C.C. Travis, editor, Municipal waste incineration risk assessment, Plenum Press, New York, NY., pp 53-63.

Turner, D.B. 1970. Workbook of Atmospheric Dispersion Estimates. PHS publication no. 999-AP-26 (NTIS PB 191482), Research Triangle Park, EPA, North Carolina.

USDA. 1983. Food Consumption: Households in the United States, Seasons and Year 1977-1978. U.S. Department of Agriculture.

USDA. 1966. Household food consumption survey 1965-1966. Report 12. Food Consumption of Households in the U.S., Seasons and Years 1965-1966. United States Department of Agriculture, Washington, D.C. U.S. Government Printing Office.

U.S. EPA. 1985. Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites. Office of Health and Environmental Assessment, Office of Research and Development, US EPA. EPA/600/8-85/002.

U.S. EPA. 1987. National Dioxin Study Report to Congress. Office of Solid Waste and Emergency Response, U.S. Environmental Protection Agency, EPA 530-SW087-025. August, 1987.

U.S. EPA. 1987a. Emission data base for municipal waste combustors. Office of Solid Waste and Emergency Response, Office of Air and Radiation, Office of Research and Development, Washington, DC., EPA/530-SW-87-021b, June, 1987.

U.S. EPA. 1987b. Characterization of the municipal waste combustion industry.Office of Solid Waste and Emergency Response, Washington,DC., EPA-530-SW-87-021h. June, 1987.

U.S. EPA. 1988. Compilation of Air Pollution Emission Factors. Volume 1. Stationary Point and Area Sources. Fourth Edition. Supplement B. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. Research Triangle Park, NC. Sepetember, 1988.

U.S. EPA. 1989. Exposure Factors Handbook. Exposure Assessment Group. Office of Health and Environmental Assessment, Office of Research and Development. U.S. Environmental Protection Agency. EPA/600/8-89/043. July, 1989.

U.S. EPA. 1989a. Background paper, municipal waste combustors air emission standards. Office of Air Quality Planning and Standards, Research Triangle Park, NC. April 26, 1989.

U.S. EPA. 1990a. Lake Ontario TCDD Bioaccumulation Study Final Report. Cooperative study including US EPA, New York State Department of Environmental Conservation, New York State Department of Health, and Occidental Chemical Corporation. May, 1990.

U.S. EPA. 1990b. Integrated Risk Assessment for Dioxins and Furans from Chlorine Bleaching in Pulp and Paper Mills. Office of Pollution Prevention and Toxics (formerly Office of Toxic Substances), U.S. EPA. EPA 560/5-90-011.

U.S. EPA. 1990c. Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions. Office of Health and Environmental Assessment, Office of Research and Development, US Environmental Protection Agency. EPA/600/6-90/003.

U.S. EPA. 1991. A Methodology for Estimating Population Exposures from the Consumption of Chemically Contaminated Fish. Prepared by Tetra Tech, Inc., Fairfax VA. for Office of Policy, Planning, and Evaluation and Office of Research and Development, US EPA. EPA/600/9-91/017.

U.S. EPA. 1991a. Burning of hazardous waste in boilers and industiral furnaces, Final Rule. Federal Register, volume 56, number 35. U.S. Government Printing Office, Wash. DC., February 21, 1991, pp 7134 - 7240.

U.S. EPA. 1992. Dermal Exposure Assessment: Principals and Applications. Exposure Assessment Group, Office of Health and Environmental Assessment, Office of Research and Development, US EPA. EPA/600/8-91/011A [DRAFT].

U.S. EPA. 1992a. Economic impact and preliminary regulatory impact analysis for proposed MACT-based emission standards and guidelines for municipal waste combustors. Office of Quality Planning and Standards, Research Triangle Park, NC. EPA-450/3-91-029. January. 1992

U.S.EPA; Environment Canada. 1991. The environmental characterization of RDF combustion technology. Mid-Connecticut facility, Hartford,CT. Test program and results, volume II., June 1991.

Webster, T. and P. Connett. 1990. The use of bioconcentration factors in estimating the 2,3,7,8-TCDD content of cow's milk. Chemosphere 20: 779-786.

Wipf, H.K., E. Homberger, N. Neuner, U.B. Ranalder, W. Vetter, and J.P. Vuilleumier. 1982. TCDD levels in soil and plant samples from the seveso area. In: Chlorinated Dioxins and Related Compounds: Impact on the Environment. Eds. Hutzinger, O., et al., Pergamon, NY.

Wischmeier, W.H., and D.D. Smith. 1965. Predicting Rainfall-Erosion Losses from Cropland East of the Rocky Mounts. Agriculture Handbook 282. U.S. Department of Agriculture, Agricultural Research Service.

Young, A.L. 1983. Long term studies on the persistence and Movement of TCDD in a natural ecosystem. In: Human and Environmental Risks of Chloronated Dioxins and Related Compounds, Eds. Tucker, R.E., A.L. Young, A.P. Gray, Plenum Press, New York, NY.

11. CONCLUSIONS AND RECOMMENDATIONS

This chapter presents the conclusions regarding how people are exposed to the dioxin-like compounds and recommendations for further research to resolve uncertainties in these estimates.

11.1. CONCLUSIONS

The following are principal findings and conclusions of this assessment:

1. The dioxin-like compounds are commonly found in soils, sediments and biota throughout the world. Concentrations in nonindustrial rural areas are generally lower than in urban or industrial areas. The higher chlorinated hepta- and octa- dioxin and furan congeners were generally more frequently found and at higher concentrations than the tetra- through hexa- congeners in environmental and exposure media. Selected literature references containing occurrence information on dioxin-like compounds were reviewed in an attempt to estimate background concentrations of these compounds in exposure media. The estimated background exposure (assuming all pathways are equally additive) to all dioxin-like compounds was estimated by multiplying the average media levels by typical contact rates. This analysis suggests a background exposure level in the range of 20-90 pg of TEq/day using data for world-wide sources. This estimate is highly qualified considering factors such as: sparseness of world-wide data, judgements as to whether data used represents "background" exposures, the assumption that all exposures considered to derive this estimate occur simultaneously contrasting the fact that several exposures, particularly ingestion of meat products other than beef, were not considered, and so on. Data on tissue levels suggest that body burden levels among industrialized nations are reasonably similar. Chapter 3, Section 3.8 contains more details on this exercise.

2. Typical exposure levels were also estimated by applying pharmacokinetic models to body burden data. Using this approach, exposure levels to 2,3,7,8-TCDD are estimated to be about 20 to 40 pg/day. This is consistent with the analysis for exposure to total

dioxin-like compounds as described above. More details of this analysis are presented in Chapter 8.

3. Procedures to estimate individual exposures to four categories of dioxin-like contamination sources were described and demonstrated in this assessment. These categories include: 1) on-site soils - the soil contamination and the exposure occur at the same site, 2) off-site soil - the contaminated soil is located distant from the site of exposure, 3) incinerator stack emissions - individuals near incinerator stacks are directly exposed via inhalation of impacted air and indirectly exposed as a result of deposition of contaminated particulates onto soil and vegetation, and 4) ash landfill - similar to the off-site soil category except that source material is incinerator ash rather than soil. Exposure assessments associated with these four source categories were demonstrated in Chapter 9. The example assessments involved development of scenarios which were carefully crafted to be plausible and meaningful. However, such assessments are based on numerous site-specific assumptions and therefore may not be generalizable to other sites. For example, demonstration of the incinerator stack emission source category was based on realistic emission rates, but only for a specific technology, air transport modeling was accomplished using the Industrial Source Complex (ISC) model using climate data from a specific location, and exposed individuals resided specific distances from the incinerator. All source categories were demonstrated on three example compounds: a dioxin (2,3,7,8-TCDD), a furan (2,3,4,7,8-PCDF), and a PCB (2,3,3',4,4',5,5'-HPCB). Finally, the example scenarios were crafted to be consistent with current exposure assessment guidelines which recommend assessment of "central" and "high-end" exposures. The following observations are offered based on the demonstration of these source categories in Chapter 9 - again, the assertion is not being made that these observations are fully generalizable conclusions:

- The highest human exposures estimated in these examples were associated with ingestion of farm products: ingestion of beef and dairy products. Fish ingestion was of principal concern when not overshadowed by ingestion of these farm products. The apparent reasons for the prevalence of food chain exposures include the tendency of dioxin-like compounds to bioaccumulate in food products of

high fat content, coupled with the ingestion rates of these food products. Lesser but still noteworthy exposures occur through contact with soils including soil ingestion and soil dermal contact. Other exposure pathways considered, including inhalation (of dioxin-like compounds in a vapor phase and associated with dust particles), fruit and vegetable ingestion, and water ingestion, were found to be of lesser consequence in comparison to these other exposure pathways.

- Estimates of central exposures were generally less than an order of magnitude lower than high-end exposure estimates. This trend was the result of exposure parameter assignments and assumptions; the differences between a central and a high end pattern of exposure was expressed primarily in terms of exposure parameters - exposure durations, intake rates, etc. For example, exposure estimates for the soil ingestion pathway for the central scenario demonstrated for the on-site source category was one-fourth the exposure for soil ingestion for the high end on-site source category demonstration scenario. The childhood pattern of soil ingestion was modeled assuming 5 years of ingestion between the ages of 2 and 6 for both the central and high end scenarios. Children were assumed to ingest 0.8 g/day soil in the high end scenarios and 0.2 g/day in the central scenarios, explaining the factor of four difference. One key difference between central and high end example scenarios was that central exposures were assumed to occur within a "residential" setting and high end within a "farm" setting. Those in the farm setting were assumed to obtain a portion of their beef and milk intake from home-produced and impacted cattle, whereas individuals in the residential setting were assumed not to be exposed to impacted beef and milk. In that sense, individuals in farm settings were evaluated as generally more exposed because of these two exposure pathways. This strategy for defining central and high end exposures is just one among many acceptable alternatives and should not be interpreted as a general recommendation for future assessments of exposure to dioxin-like (or other) compounds.

- The on-site source category was demonstrated assuming soil levels of 1 ng/kg (ppt) which was characterized as typical of soil levels found by researchers

specifically targeting "background" or "rural" areas. The stack emission source category was demonstrated using emission rates specific to a fabric filter combined with semi-dry alkaline scrubbers (abbreviated DSFF in Chapter 6) emission control technology. This was characterized in Chapter 6 as a high level of emission control in use for a relatively small number of incinerators currently operating, but expected to be more prevalent for new incinerators. Exposure media concentrations and resulting exposures assuming these soil concentrations and incinerator emission source strengths were found to be comparable. The off-site soil source category was demonstrated assuming 1 μg/kg (ppb) concentrations in a contaminated area located 150 meters from a farm (the example scenario was described as a high end scenario). This soil concentration was characterized as typical of soils of known industrial contamination of the dioxin-like compounds. The ash landfill soil concentrations were developed given estimated ash concentrations of the example compounds based on the same emission control technology. Resulting landfill concentrations were also in the low ppb range, and exposed individuals resided 150 meters from the landfill. Exposures estimated for individuals living near these areas of relatively high soil concentrations were 2 to 3 orders of magnitude higher as compared to the on-site and stack emission exposure estimates. Like other scenario observations, these results are not offered as generalizations. Key qualifiers include: they are specific to the source strengths assumed and are specific to that source alone (i.e., total exposures might include impacts from more than one source), proximity to the source strengths, realism of estimated exposure media concentrations, and so on.

Further observations from the demonstration of source categories are given in Chapter 9, Section 9.6.

4. In contrast to a few years ago, consensus is emerging as to causes and extent of vegetative contamination by the dioxin-like compounds. It now appears that vegetation is impacted as a result of three processes: sorption to outer portions of below ground parts, "air-to-leaf" transfers of vapor-phase contaminants to outer above ground plant surfaces, and deposition and sorption of air-borne particulate-phase contaminants onto

outer portions of above ground vegetation. Translocation from outer to inner portions appears to be minimal, and translocation from root to shoot appears to be essentially nonexistent. This document presents unique approaches for estimating vegetative impacts from these routes. Also, estimates of fruit and vegetable ingestion exposures considered below vs. above ground and protected vs. unprotected fruits. The models estimating vegetative concentrations were examined in Chapter 10. Some observations from that examination are now listed. It is noted that these observations are specific to the models and the model parameters used for demonstration purposes. These observations are not field verified and might change with a different set of parameters:

- When the source of contamination is soil, 90% and more of the estimated above ground vegetative concentrations are explained by volatilization followed by air-to-leaf transfers. In contrast, when incinerator emissions are the source, above ground vegetative concentrations are dominated by particulate depositions.

- When the source of contamination is soil, below ground vegetation concentrations exceed those of above ground vegetation by 1-2 orders of magnitude. When the source of contamination is incinerator emissions, the reverse is true: above ground vegetation concentrations exceed below ground concentrations by roughly 2 orders of magnitude.

The analysis of the vegetative concentration models is given in Chapter 10, Section 10.2.11.

5. With some modifications, procedures in this assessment to estimate beef and milk concentrations are similar to procedures developed during the 1980s. One principal difference was that most earlier assessments did not include air-to-leaf transfers in their estimates of pasture grass and cattle fodder concentrations. The other principal difference was that earlier assessments did not consider soil ingestion by cattle to the extent considered in this assessment. Analysis showed that critical components of the beef and milk models include the assumptions of soil ingestion by cattle, and estimates of pasture grass and cattle fodder concentrations of the dioxin-like compounds. A dichotomy in

model results between the soil source categories and the incinerator emission categories, similar to the dichotomies noted above, was also observed in predictions of beef and milk concentrations in the demonstration of source categories. For the soil source categories, ingestion of soil explained 51-71% of estimated beef and milk concentrations, the remainder due to ingestion of impacted pasture grass and cattle fodder. For the incinerator source category, soil ingestion explained only 11-26% of estimated concentrations, the remainder due to pasture grass and cattle fodder ingestion. The analysis of the beef and milk concentration algorithms are given in Chapter 10, Section 10.2.12.

6. There are two basic approaches for estimating concentrations in fish. One is based on water concentrations (soluble phase or total concentrations) and one is based on bottom sediment concentrations. Estimates of fish concentrations have classically been made based on water concentrations and a "bioconcentration factor" (BCF), and this approach has been used in assessment of exposures to 2,3,7,8-TCDD. Bioconcentration refers to the net accumulation of a chemical from exposure via water only, and BCFs are most often obtained in laboratory conditions. Another measure of the potential for chemicals to accumulate in fish tissue is termed the "bioaccumulation factor" (BAF). Bioaccumulation refers to the net accumulation of a chemical from exposure via food and sediments as well as water. Fish exposure to strongly hydrophobic contaminants such as the dioxin-like compounds is not via passive movement through the gills but rather through the food chain. This argument supports the use of a BAF over a BCF if estimating fish tissue concentrations based on water column concentrations. This assessment presents the BAF procedure (Chapter 10, Section 10.2.8.3). The focus of this assessment, however, is on an approach to estimating fish tissue concentrations based on bottom sediment concentrations. This procedure uses an organic carbon-based, lipid-based "biota sediment accumulation factor" (BSAF), which is defined as the ratio of contaminant concentrations in the lipid of fish to the organic-carbon based concentrations in bottom sediments. In theory, the BAF and BSAF are analogous indicators of bioaccumulation, and should therefore lead to similar estimations of fish tissue concentrations if used for that purpose. One key advantage of the BSAF approach is that bottom sediment concentrations of the dioxin-like compounds are generally at levels amenable to analytical measurement, and a relatively large amount of data exists on concentrations of these

compounds in bottom sediments. In contrast, water concentrations of hydrophobic contaminants are very low so as to preclude analytical measurements; development of BAFs require computer model simulations to estimate the necessary water column concentrations. A limited comparative modeling test was conducted on the BAF and BSAF approaches. General conclusions cannot be made from this exercise due to the uncertainties in the input parameters, the uncertainties in the modeling of sediment and water concentrations, and so on. However, the exercise showed that the BSAF and BAF approaches arrived at comparable concentrations; within a factor of ten with the BAF approach generally predicting higher than the BSAF approach. Details of this sensitivity exercise can be found in Chapter 10, Section 10.2.8.3.

11.2. RECOMMENDATIONS

Further research is recommended in the following areas. These are not listed in any specific priority order.

1. Throughout this document the lack of congener-specific data is cited as a major source of uncertainty. For example, congener-specific data is lacking for basic chemical properties such as octanol-water partition coefficients, degradation rates, and vapor pressures. Also this data is lacking for estimation of incinerator emission factors, metabolic rate constants, and bioavailability and biotransfer factors. Thus, gathering more data on the congener-specific properties of these compounds is a high priority for further research.

2. The use of pharmacokinetics in body burden analysis has shown great potential for estimating exposure levels. In order to reduce the uncertainty in these procedures, increased collection of biological samples and improvements in PK model structure and input parameters are recommended. In addition, further research should be conducted on the application of these procedures to estimating target organ dose, absorbed dose, lactational/placental transfers, and effects on offspring.

3. There are key uncertainties in estimating fish tissue concentrations using the BSAF approach, as there are using the BAF approach. Because of their affinity for organic carbon, the fate and transport of dioxin-like compounds in water bodies is likely to be more a function of sediment-related processes rather than water-related processes. Key sediment processes in water bodies include: sorption/desorption, importance and prevalence of dissolved organic materials in the water column, deposition/suspension/resuspension, downstream sediment transport, and so on. Although some preliminary information on sediment modeling in surface water bodies is presented in Appendix D, much more development is needed, especially for evaluating pipe discharges. Further development is needed for the BSAF approach for estimating fish tissue concentrations. A key issue that has been identified is whether BSAFs that have been developed for one species and water body are generalizable to another species and another water body. This question will be difficult to answer because of the several uncertainties associated with BSAF development that were discussed in Chapter 5: fish migratory patterns, variability in fish lipid content and other differences within and between species, study design with regard to fish and sediment sampling, ecosystem differences, and so on. However, after careful examination of existing data sets and considering key differences between species (invertebrates vs. vertebrates, fresh water vs. salt water, bottom feeders vs. water column feeders, etc.), it may be possible to develop a workable system for BSAF assignment based on key considerations.

4. Considering that beef and dairy exposures are identified as critical exposures in this assessment, more information is needed on several of the components of the model to estimate beef and milk concentrations. Such information includes: cattle soil ingestion rates, pasture grass concentrations and mechanisms of transfer from the air/soil to pasture grass, inventories of cattle production practices (such as fattening prior to slaughter) and the impact of these practices to cattle food product concentrations, and models and data to further develop the bioconcentration factor (termed F in this assessment and translates concentrations on cattle dry matter intake to concentrations in beef and milk fat).

5. This assessment did not evaluate all possible exposure pathways. Other potential pathways include ingestion of other farm products such as eggs, chicken, and

pork. The occurrence of these compounds in ground water is expected to be minimal based on strong sorption to soils. However, they have been found in ground water below and near sites of industrial contamination. Co-occurrence with other organic compounds, co-occurrence with solvents, and transport associated with oils have been cited as causes of enhanced mobility in these settings. Exposures through mother's milk or placental transfers (as noted above) need to be evaluated and compared with other exposure pathways. Exposure resulting from bleached paper products, such as occurrences of 2,3,7,8-TCDD leached from milk containers, has also been cited as a route of exposure. Further evaluation of these pathways is recommended.

6. Recent assessments of exposure to dioxin-like compounds have concluded that an inventory of currently identified sources of contamination falls well short of explaining environmental concentrations that have been measured. Researchers have claimed that key sources of dioxin-like compounds in the environment have yet to be identified. This issue wasn't explored in this assessment, but is certainly worthy of careful consideration. Complete source identification and comparative contributions by different sources should be explored in future assessments.

7. Chapter 10 demonstrates the breadth of variability and uncertainty in the models and model parameters described in this assessment; some of the more important areas of uncertainty have been noted in previous recommendations. Reducing uncertainty in all areas will increase the reliability of estimates using procedures in this assessment and similar ones. Areas not discussed above include: emission rates of dioxin-like compounds from different stack emission technologies, vapor/particulate partitioning in air from stack emitted and soil volatilized residues, overland transport of residues from one site to nearby land areas and to surface water, and the impact of photodegradation on air-borne residues and concentrations in biota and surface soils.

APPENDIX A. ENVIRONMENTAL CHEMISTRY

The tables in this appendix are discussed in Chapter 2. References listed at the end of each table are included in the reference list at the end of Chapter 2. Following are the tables included in this Appendix:

A-1 Physical and chemical properties for the dioxin-like compounds

A-2 Physical and chemical properties for the dioxin, furan, and PCB congeners

TABLE A-1. P-CHEM PROPERTIES for the DIOXIN-LIKE CONGENERS

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow} K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
Tetrachlorodibenzo-p-dioxins (MW=321.98)														
2.3.7.8-TCDD 1746-01-6	305-306	9	1.90E-05	25	1.2	7.40E-10	25	2,3	(1.65E-05)	2,3	6.64	2,4	2.2E-03[b]	22
Homologue Group Average			(3.50E-04)	25	20	(1.36E-08)	25	20	(4.06E-05)	20	(6.52)	20		
Pentachlorodibenzo-p-dioxins (MW=356.42)														
1,2,3,7,8-PeCDD 40321-76-4	240-241	9				(4.35E-10)	25	9			6.64	10		
Homologue Group Average			(1.20E-04)	20	20	(5.60E-10)	25	20	(2.19E-06)	20	(6.57)	20		
Hexachlorodibenzo-p-dioxins (MW=390.87)														
1,2,3,4,7.8-HxCDD 39227-28-6	273-275	9	4.40E-06	20	6	(3.82E-11)	25	9	(4.47E-06)	19	(7.79)	8	1.1E-04[c]	23
1,2,3,6,7,8-HxCDD 57653-85-7	285-286	9				(3.60E-11)	25	9						
1,2,3,7,8,9-HxCDD 19408-74-3	243-244	9				(4.88E-11)	25	9						
Homologue Group Average			(4.40E-06)	20	20	(4.35E-11)	25	20	(4.47E-06)	20	(7.25)	20		
Heptachlorodibenzo-p-dioxins (MW=425.31)														
1,2,3,4,6.7,8-HpCDD 35822-46-9	264-265	9	2.40E-06	20	6	(5.62E-12)	25	9	(1.31E-06)	19	(8.20)	8	1.53E-05[c]	23
Homologue Group Average			(2.40E-06)	20	20	(5.62E-12)	25	20	(1.31E-06)	20	(8.20)	20		
Octachlorodibenzo-p-dioxins (MW=460.76)														
1,2,3,4,6,7.8.9-OCDD 3268-87-9	325-326	6	4.00E-07	20	6	8.25E-13	25	9	(6.74E-06)	5	(8.60)	8	2.26E-05[c]	23

TABLE A-1 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
Tetrachlorodibenzofurans (MW=305.98)														
2,3,7,8-TCDF 51207-31-9	227-228	21	4.19E-04	22.7	11	(1.50E-08)	25	21	(1.44E-05)	19	6.53	10		
Homologue Group Average			(4.19E-04)	22.7	20	(2.49E-08)	25	20	(1.44E-05)	20	(6.21)	20		
Pentachlorodibenzofurans (MW=340.42)														
1,2,3,7.8-PeCDF 57117-41-6	225-227	21				(1.72E-09)	25	21			6.79	10		
2,3,4,7,8-PeCDF 57117-31-4	196-196.5	21	2.36E-04	22.7	11	(2.63E-09)	25	21	(4.99E-06)	19	6.92	10		
Homologue Group Average			(2.36E-04)	22.7	20	(2.57E-09)	25	20	(4.99E-06)	20	(6.44)	20		
Hexachlorodibenzofurans (MW=374.87)														
1,2,3,4,7,8-HxCDF 70648-26-9	225.5-226.5	21	8.25E-06	22.7	11	(2.40E-10)	25	21	(1.43E-05)	19			6.96E-04[b]	25
1.2,3,6,7,8-HxCDF 57117-44-9	232-234	21	1.77E-05	22.7	11	(2.18E-10)	25	21	(6.08E-06)	19				
1,2,3,7,8,9-HxCDF 72918-21-9	246-249	21												
2,3,4,6,7,8-HxCDF 60851-34-5	239-240	21				(1.95E-10)	25	21						
Homologue Group Average			(1.30E-05)	22.7	20	(2.83E-10)	25	20	(1.02E-05)	20				
Heptachlorodibenzofurans (MW=409.31)														
1,2,3,4,6.7,8-HpCDF 67562-39-4	236-237	21	1.35E-06	22.7	11	(3.53E-11)	25	21	(1.41E-05)	19	7.92	10		

TABLE A-1 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2,3,4,7,8,9-HpCDF 55673-89-7	221-223	21				(4.65E-11)	25	21						
Homologue Group Average			(1.35E-06)	22.7	20	(4.65E-11)	25	20	(1.41E-05)	20	(7.92)	20		
Octachlorodibenzofurans (MW=444.76)														
1,2,3,4.6,7,8.9-OCDF 39001-02-0	258-260	21	(1.16E-06)	25	11	3.75E-12	25	21	(1.89E-06)	19	(8.78)	8		
Tetrachloro-PCB (MW=291.99)														
3,3',4,4'-TeCB 32598-13-3	453	17	5.69E-04	25	12	(1.34E-05)	25	18	9.40E-05	13	6.21	15		
3,4,4',5-TeCB 70362-50-4	(410)	17	(2.92E-03)	25	17	(1.77E-05)	25	18	(1.48E-04)	16	(6.36)	15		
Pentachloro-PCB (MW=326.44)														
2,3,3',4,4'-PeCB 32598-14-4	(398)	17	(2.06E-03)	25	17	(5.86E-06)	25	18	(6.00E-05)	16	(6.65)	15		
2.3,4,4',5-PeCB 74472-37-0	(392)	17	(2.59E-03)	25	17	(9.00E-06)	25	18	(1.14E-04)	16	(6.65)	15		
2,3',4,4',5-PeCB 31508-00-6	378	17	(2.06E-03)	25	17	(8.50E-06)	25	18	(1.16E-04)	16	(6.74)	15		
3,3',4.4',5-PeCB 57465-28-8	(398)	17	(1.03E-03)	25	17	(2.90E-06)	25	18	(5.40E-05)	16	(6.89)	15		
Hexachloro-PCB (MW=360.88)														
2,3,3'.4,4',5-HxCB 38380-08-4	(414)	17	(3.61E-04)	25	17	(1.39E-06)	25	18	(2.20E-05)	16	7.13	14		
2,3,3',4,4'.5'-HxCB 69782-90-7	(414)	17	(3.61E-04)	25	17	(1.23E-06)	25	18	(6.60E-05)	16	7.20	14		

TABLE A-1 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
2,3',4,4',5,5'-HxCB 52663-72-6	(408)	17	(3.61E-04)	25	17	(1.95E-06)	25	18	(1.23E-04)	16	7.26	14		
3,3',4,4',5,5'-HxCB 32774-16-6	(485)	17	(3.61E-05)	25	17	(1.52E-06)	25	20	(5.90E-05)	16	7.47	14		
Heptachloro-PCB (MW=396.33)														
2,3,3',4,4',5,5'-HpCB 39635-31-9	(431)	17	(6.28E-05)	25	17	(3.04E-07)	25	18	3.00E-3	19	(7.71)	15		

Footnote References

[a] Values are presented as they appeared in the referenced article. Values in ( ) are calculated.
[b] Tested in solution of water:acetonitrile (1:1 v/v) at 313 nm.
[c] Tested in solution of water:acetonitrile (2:3 v/v) at 313 nm.

1. Marple et al. (1986a)
2. USEPA (1990)
3. Podoll et al. (1986)
4. Marple et al. (1986b)
5. Shiu et al. (1988)
6. Friesen et al. (1985)
8. Burkhard and Kuehl (1986)
9. Rordorf (1987)
10. Sijm et al. (1989)
11. Friesen et al. (1990)
12. Dickhut et al. (1986)
13. Dunnivant and Elzerman (1988)
14. Risby et al. (1990)
15. Hawker and Connell (1988)
16. Sabljic and Gusten (1989)
17. Abramowitz and Yalkowsky (1990)
18. Foreman and Bidleman (1985)
19. Calculated by the VP/WS ratio technique
20. Average of all literature values (measured and calculated) within a homologue group
21. Rordorf (1989)
22. Dulin et al. (1986)
23. Choudhry and Webster (1987)

TABLE A-2. P-CHEM PROPERTIES for the DIOXIN, FURAN, and PCB CONGENERS

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
Tetrachlorodibenzo-p-dioxins (MW=321.98)														
1,2,3,4-TCDD 30746-58-8			6.30E-04	25	5	4.80E-08	25	9	(3.72E-05)	5	6.60	5		
1,2,3,6-TCDD 71669-25-5											6.86	10		
1,2,3,7-TCDD 67028-18-6	172-175	6	4.30E-04	20	6	(7.50E-09)	25	9	(7.39E-06)	19	(6.91)	8	5.42E-04[c]	24
1.2,3.8-TCDD 53555-02-5											6.48	10		
1,2,3.9-TCDD 71669-26-6											6.39	10		
1,2,4,6-TCDD 71669-27-7											6.10	10		
1.2,4,7-TCDD 71669-28-8											6.25	10		
1,2,4,8-TCDD 71669-29-9											6.25	10		
1.2,4,9-TCDD 71665-99-1											6.10	10		
1,2,6,7-TCDD 40581-90-6														
1.2.6,8-TCDD 67323-56-2											6.43	10		
1,2,6,9-TCDD 40581-91-7														

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2.7,8-TCDD 34816-53-0											6.38	10		
1.2,7,9-TCDD 71669-23-3											6.86	10		
1,2,8,9-TCDD 62470-54-6														
1,3,6,8-TCDD 33423-92-6	219-220	6	3.20E-04	20	6	(5.25E-09)	25	9	6.81E-05	7	7.20	8	2.17E-03[c]	24
1,3,6,9-TCDD 71669-24-4											6.25	10		
1,3,7,8-TCDD 50585-46-1	193.5-195	9				(6.30E-09)	25	9			6.30	10		
1,3,7,9-TCDD 62470-53-5											7.06	8		
1,4,6,9-TCDD 40581-93-9											6.38	10		
1,4,7,8-TCDD 40581-94-0											6.39	10		
2,3,7,8-TCDD 1746-01-6	305-306	9	1.90E-05	22	1.2	7.40E-10	25	2,3	(1.65E-05)	2,3	6.64	2,4	2.2E-03[b]	22
Homologue Group Average			(3.50E-04)	25	20	(1.36E-08)	25	20	(4.06E-05)	20	(6.52)	20		
Pentachlorodibenzo-p-dioxins (MW=356.42)														
1,2,3,4,6-PeCDD 67028-19-7											6.30	10		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2,3,4,7-PeCDD 39227-61-7	195-196	9	1.20E-04	20	6	(6.60E-10)	25	9	(2.58E-06)	19	(7.44)	8	9.8E-05[c]	23
1,2,3,6,7-PeCDD 71925-15-0											6.74	10		
1,2,3,6,8-PeCDD 71925-16-1											6.53	10		
1,2,3,6,9-PeCDD 82291-34-7											6.24	10		
1,2,3,7,8-PeCDD 40321-76-4	240-241	9				(4.35E-10)	25	9			6.64	10		
1,2,3,7,9-PeCDD 71925-17-2											6.40	10		
1,2,3,8,9-PeCDD 71925-18-3														
1,2,4,6,7-PeCDD 82291-35-8														
1,2,4,6,8-PeCDD 71998-76-0														
1,2,4,6,9-PeCDD 82291-36-9											6.60	10		
1,2,4,7,8-PeCDD 58802-08-7	206	9				(5.85E-10)	25	9			6.20	10		
1,2,4,7,9-PeCDD 82291-37-0														

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2,4,8,9-PeCDD 82291-38-1														
Homologue Group Average			(1.20E-04)	20	20	(5.60E-10)	25	20	(2.19E-06)	20	(6.57)	20		
Hexachlorodibenzo-p-dioxins (MW=390.87)														
1,2,3,4,6,7-HxCDD 58200-66-1														
1,2,3,4,6,8-HxCDD 58200-67-2											6.85	10		
1,2,3,4,6,9-HxCDD 58200-68-3														
1,2,3,4,7,8-HxCDD 39227-28-6	273-275	9	4.40E-06	20	6	(3.82E-11)	25	9	(4.47E-06)	19	(7.79)	8	1.1E-04[c]	23
1,2,3,6,7,8-HxCDD 57653-85-7	285-286	9				(3.60E-11)	25	9						
1,2,3,6,7,9-HxCDD 64461-98-9											7.59	10		
1,2,3,6,8,9-HxCDD 58200-69-4											7.59	10		
1,2,3,7,8,9-HxCDD 19408-74-3	243-244	9				(4.88E-11)	25	9						
1,2,4,6,7,9-HxCDD 39227-62-8	238-240	9				(5.10E-11)	25	9			6.85	10		
1,2,4,6,8,9-HxCDD 58802-09-8											6.85	10		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
Homologue Group Average			(4.40E-06)	20	20	(4.35E-11)	25	20	(4.47E-06)	20	(7.25)	20		
Heptachlorodibenzo-p-dioxins (MW=425.31)														
1,2,3,4,6,7,8-HpCDD 35822-46-9	264-265	9	2.40E-06	20	6	(5.62E-12)	25	9	(1.31E-06)	19	(8.20)	8	1.53E-05[c]	23
1,2,3,4,6,7,9-HpCDD 58200-70-7														
Homologue Group Average			(2.40E-06)	20	20	(5.62E-12)	25	20	(1.31E-06)	20	(8.20)	20		
Octachlorodibenzo-p-dioxin (MW=460.76)														
1,2,3,4,6,7,8,9-OCDD 3268-87-9	325-326	6	4.00E-07	20	6	8.25E-13	25	9	(6.74E-06)	5	(8.60)	8	2.26E-05[c]	23
Tetrachlorodibenzofurans (MW=305.98)														
1,2,3,4-TCDF 30402-14-3	168.5-169	21				(3.06E-08)	25	21			6.17	10		
1,2,3,6-TCDF 83704-21-6											6.15	10		
1,2,3,7-TCDF 83704-22-7	167.5-168	21				(3.15E-08)	25	21						
1,2,3,8-TCDF 62615-08-1	197-198	21				(2.10E-08)	25	21			6.15	10		
1,2,3,9-TCDF 83704-23-8											6.06	10		
1,2,4,6-TCDF 71998-73-7														

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2,4,7-TCDF 83719-40-8														
1,2,4,8-TCDF 64126-87-0	191-193	21				(2.25E-08)	25	21			6.31	10		
1,2,4,9-TCDF 83704-24-9														
1,2,6,7-TCDF 83704-25-0	199-200	21				(2.02E-08)	25	21			6.25	10		
1,2,6,8-TCDF 83710-07-0														
1,2,6,9-TCDF 70648-18-9														
1,2,7,8-TCDF 58802-20-3	210-211	21				(1.80E-08)	25	21			6.23	10		
1,2,7,9-TCDF 83704-26-1	184-185	21				(2.48E-08)	25	21			6.25	10		
1,2,8,9-TCDF 70648-22-5														
1,3,4,6-TCDF 83704-27-2											6.31	10		
1,3,4,7-TCDF 70648-16-7											6.23	10		
1,3,4,8-TCDF 92341-04-3											6.13	10		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,3,4.9-TCDF 83704-28-3											5.89	10		
1,3,6,7-TCDF 57117-36-9	176.5-177	21				(2.78E-08)	25	21						
1.3,6.8-TCDF 71998-72-6	177-178	21				(2.70E-08)	25	21			6.37	10		
1.3,6,9-TCDF 83690-98-6														
1,3,7,8-TCDF 57117-35-8											6.34	10		
1.3,7,9-TCDF 64560-17-4	206.5-207.5	21				(1.88E-08)	25	21			6.34	10		
1,4,6,7-TCDF 66794-59-0	180-181	21				(2.62E-08)	25	21			6.15	10		
1,4.6,8-TCDF 82911-58-8														
1,4,6,9-TCDF 70648-19-0											5.60	10		
1,4,7,8-TCDF 83704-29-4														
1,6,7,8-TCDF 83704-33-0											6.17	10		
2.3.4,6-TCDF 83704-30-7	153-154	21				(3.98E-08)	25	21			6.11	10		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
2,3,4,7-TCDF 83704-31-8	172.5-173	21				(2.92E-08)	25	21			6.06	10		
2,3,4,8-TCDF 83704-32-9	176-177	21				(2.78E-08)	25	21						
2,3,6,7-TCDF 57117-39-2	195-196	21				(2.10E-08)	25	21			6.31	10		
2,3,6,8-TCDF 57117-37-0	202-203	21				(1.95E-08)	25	21			6.73	10		
2,3,7,8-TCDF 51207-31-9	227-228	21	4.19E-04	22.7	11	(1.50E-08)	25	21	(1.44E-05)	19	6.53	10		
2,4,6,7-TCDF 57117-38-1	164-165.5	21				(3.30E-08)	25	21			6.25	10		
2,4,6,8-TCDF 58802-19-0	198-200	21				(2.02E-08)	25	21			6.17	10		
3,4,6,7-TCDF 57117-40-5														
Homologue Group Average			(4.19E-04)	22.7	20	(2.49E-08)	25	20	(1.44E-05)	20	(6.21)	20		
Pentachlorodibenzofurans (MW=340.42)														
1,2,3,4,6-PeCDF 83704-47-6	194-195	21				(2.70E-09)	25	21						
1,2,3,4,7-PeCDF 83704-48-7											6.53	10		
1,2,3,4,8-PeCDF 67517-48-0	177-178	21				(3.60E-09)	25	21			6.79	10		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2,3,4,9-PeCDF 83704-49-8														
1,2,3,6,7-PeCDF 57117-42-7	205-207	21				(2.25E-09)	25	21			6.26	10		
1,2,3,6,8-PeCDF 83704-51-2											6.33	10		
1,2,3,6,9-PeCDF 83704-52-3														
1,2,3,7,8-PeCDF 57117-41-6	225-227	21				(1.72E-09)	25	21			6.79	10		
1,2,3,7,9-PeCDF 83704-53-4														
1,2,3,8,9-PeCDF 83704-54-5														
1,2,4,6,7-PeCDF 83704-50-1	178.5-179.5	21				(3.52E-09)	25	21			6.27	10		
1,2,4,6,8-PeCDF 69698-57-3	204-205	21				(2.32E-09)	25	21			6.34	10		
1,2,4,6,9-PeCDF 70648-24-7											6.59	10		
1,2,4,7,8-PeCDF 58802-15-6	236-238	21				(1.50E-09)	25	21			6.26	10	1.30E-02[b]	25
1,2,4,7,9-PeCDF 71998-74-8	196-197	21				(2.63E-09)	25	21			6.19	10		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2,4,8,9-PeCDF 70648-23-6														
1.2,6,7,8-PeCDF 69433-00-7	220-221	21				(1.88E-09)	25	21			6.42	10		
1,2,6,7,9-PeCDF 70872-82-1											6.51	10		
1,3,4,6,7-PeCDF 83704-36-3	195-195.5	21				(2.70E-09)	25	21			6.19	10		
1,3,4,6,8-PeCDF 83704-55-6											6.24	10		
1,3,4,6,9-PeCDF 70648-15-6											6.34	10		
1,3,4,7,8-PeCDF 58802-16-7	168-170	21				(4.28E-09)	25	21						
1,3,4,7,9-PeCDF 70648-20-3											6.33	10		
1,3,6,7,8-PeCDF 70648-21-4														
1,4,6.7,8-PeCDF 83704-35-2											6.53	10		
2,3,4,6,7-PeCDF 57117-43-8	201.5-202	21				(2.40E-09)	25	21			6.47	10		
2,3,4,6,8-PeCDF 67481-22-5	219-220	21				(1.88E-09)	25	21			6.59	10		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
2,3,4,7,8-PeCDF 57117-31-4	196-196.5	21	2.36E-04	22.7	11	(2.63E-09)	25	21	(4.99E-06)	19	6.92	10		
Homologue Group Average			(2.36E-04)	22.7	20	(2.57E-09)	25	20	(4.99E-06)	20	(6.44)	20		
Hexachlorodibenzofurans (MW=374.87)														
1,2,3,4,6,7-HxCDF 79060-60-9	227-228	21				(2.40E-10)	25	21						
1,2,3,4,6,8-HxCDF 69698-60-8	233.5-234	21				(2.18E-10)	25	21						
1,2,3,4,6,9-HxCDF 91538-83-9	196-197	21				(4.13E-10)	25	21						
1,2,3,4,7,8-HxCDF 70648-26-9	225.5-226.5	21	8.25E-06	22.7	11	(2.40E-10)	25	21	(1.43E-05)	19			6.96E-04[b]	25
1,2,3,4,7,9-HxCDF 91538-84-0	216-217	21				(2.85E-10)	25	21						
1,2,3,4,8,9-HxCDF 92341-07-6														
1,2,3,6,7,8-HxCDF 57117-44-9	232-234	21	1.77E-05	22.7	11	(2.18E-10)	25	21	(6.08E-06)	19				
1,2,3,6,7,9-HxCDF 92341-06-5														
1,2,3,6,8,9-HxCDF 75198-38-8	206-207	21				(3.38E-10)	25	21						
1,2,3,7,8,9-HxCDF 72918-21-9	246-249	21				(1.80E-10)	25	21						

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
1,2,4,6,7,8-HxCDF 67562-40-7	221-222	21				(2.63E-10)	25	21						
1,2,4,6,7.9-HxCDF 75627-02-0	180-181	21				(5.70E-10)	25	21						
1,2,4,6,8,9-HxCDF 69698-59-5	246-248	21				(1.80E-10)	25	21						
1.3.4,6,7,8-HxCDF 71998-75-9	229-230	21				(2.33E-10)	25	21						
1,3,4.6,7,9-HxCDF 92341-05-4														
2,3.4,6,7,8-HxCDF 60851-34-5	239-240	21				(1.95E-10)	25	21						
Homologue Group Average			(1.30E-05)	22.7	20	(2.83E-10)	25	20	(1.02E-05)	20				
Heptachlorodibenzofurans (MW=409.31)														
1,2,3,4,6,7,8-HpCDF 67562-39-4	236-237	21	1.35E-06	22.7	11	(3.53E-11)	25	21	(1.41E-05)	19	7.92	10		
1,2,3,4,6,7,9-HpCDF 70648-25-8														
1,2,3,4,6,8,9-HpCDF 69698-58-4	211-212	21				(5.78E-11)	25	21						
1,2,3,4.7.8,9-HpCDF 55673-89-7	221-223	21				(4.65E-11)	25	21						
Homologue Group Average			(1.35E-06)	22.7	20	(4.65E-11)	25	20	(1.41E-05)	20	(7.92)	20		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
Octachlorodibenzofurans (MW=444.76)														
1,2,3,4,6,7.8,9-OCDF 39001-02-0	258-260	21	(1.16E-06)	25	11	3.75E-12	25	21	(1.89E-06)	19	(8.78)	8		
Tetrachloro-PCB (MW=291.99)														
3.3',4,4'-TeCB 32598-13-3	453	17	5.69E-04	25	12	(1.34E-05)	25	18	9.40E-05	13	6.21	15		
3.4,4',5-TeCB 70362-50-4	(410)	17	(2.92E-03)	25	17	(1.77E-05)	25	18	(1.48E-04)	16	(6.36)	15		
Pentachloro-PCB (MW=326.44)														
2,3,3',4,4'-PeCB 32598-14-4	(398)	17	(2.06E-03)	25	17	(5.86E-06)	25	18	(6.00E-05)	16	(6.65)	15		
2,3,4,4',5-PeCB 74472-37-0	(392)	17	(2.59E-03)	25	17	(9.00E-06)	25	18	(1.14E-04)	16	(6.65)	15		
2,3',4,4',5-PeCB 31508-00-6	378	17	(2.06E-03)	25	17	(8.50E-06)	25	18	(1.16E-04)	16	(6.74)	15		
3,3',4,4',5-PeCB 57465-28-8	(398)	17	(1.03E-03)	25	17	(2.90E-06)	25	18	(5.40E-05)	16	(6.89)	15		
Hexachloro-PCB (MW=360.88)														
2.3.3',4,4',5-HxCB 38380-08-4	(414)	17	(3.61E-04)	25	17	(1.39E-06)	25	18	(2.20E-05)	16	7.13	14		
2,3,3',4,4',5'-HxCB 69782-90-7	(414)	17	(3.61E-04)	25	17	(1.23E-06)	25	18	(6.60E-05)	16	7.20	14		
2,3',4,4',5,5'-HxCB 52663-72-6	(408)	17	(3.61E-04)	25	17	(1.95E-06)	25	18	(1.23E-04)	16	7.26	14		

TABLE A-2 (continued)

Chemical CAS No.	Melting Point C°[a]	Melting Point Ref.	Water Solubility mg/l[a]	WS Temp.	WS Ref.	Vapor Pressure mm Hg[a]	VP Temp.	VP Ref.	Henry's Constant atm-m³/mol[a]	Henry's Constant Ref.	Log K_{ow}[a]	Log K_{ow} Ref.	Photo Quantum Yield[a]	Photo Quantum Yield Ref.
3,3',4,4',5,5'-HxCB 32774-16-6	(485)	17	(3.61E-05)	25	17	(1.52E-06)	25	20	(5.90E-05)	16	7.47	14		
Heptachloro-PCB (MW=396.33)														
2,3,3',4,4',5,5'-HpCB 39635-31-9	(431)	17	(6.28E-05)	25	17	(3.04E-07)	25	18	3.00E-3	19	(7.71)	15		

Footnote References

[a] Values are presented as they appeared in the referenced article. Values in ( ) are calculated.
[b] Tested in solution of water:acetonitrile (1:1 v/v) at 313 nm.
[c] Tested in solution of water:acetonitrile (2:3 v/v) at 313 nm.

1. Marple et al. (1986a)
2. USEPA (1990)
3. Podoll et al. (1986)
4. Marple et al. (1986b)
5. Shiu et al. (1988)
6. Friesen et al. (1985)
7. Webster et al. (1985)
8. Burkhard and Kuehl (1986)
9. Rordorf (1987)
10. Sijm et al. (1989)
11. Friesen et al. (1990)
12. Dickhut et al. (1986)
13. Dunnivant and Elzerman (1988)
14. Risby et al. (1990)
15. Hawker and Connell (1988)
16. Sabljic and Gusten (1989)
17. Abramowitz and Yalkowsky (1990)
18. Foreman and Bidleman (1985)
19. Calculated by the VP/WS ratio technique
20. Average of all literature values (measured and calculated) within a homologue group
21. Rordorf (1989)
22. Dulin et al. (1986)
23. Choudhry and Webster (1987)
24. Choudhry and Webster (1989)
25. Choudhry et al. (1990)

APPENDIX B. ENVIRONMENTAL CONCENTRATIONS

The tables in this appendix are discussed in Chapter 3. References listed at the end of each table are included in the reference list at the end of Chapter 3. Following are the tables included in this Appendix:

Table B-1.	Environmental Levels of Dioxins in Soil (ppt).
Table B-2.	Environmental Levels of Dibenzofurans in Soil (ppt).
Table B-3.	Environmental Levels of Dioxins in Water (ppq).
Table B-4.	Environmental Levels of Dibenzofurans in Water (ppq).
Table B-5.	Environmental Levels of Dioxins in Sediments (ppt).
Table B-6.	Environmental Levels of Dibenzofurans in Sediments (ppt).
Table B-7.	Environmental Levels of PCBs in Sediment (ppt).
Table B-8.	Environmental Levels of Dioxins in Fish (ppt).
Table B-9.	Environmental Levels of Dibenzofurans in Fish (ppt).
Table B-10.	Environmental Levels of PCBs in Fish (ppt).
Table B-11.	Levels of Dioxins in Food Products (ppt).
Table B-12.	Levels of Dibenzofurans in Food Products (ppt).
Table B-13.	Environmental Levels of Dioxins in Air (pg/m^3).
Table B-14.	Environmental Levels of Dibenzofurans in Air (pg/m^3).
Table B-15.	Mean Background Environmental Levels of Dioxins in Soil (ppt).
Table B-16.	Mean Background Environmental Levels of Dibenzofurans in Soil (ppt).
Table B-17.	Mean Background Levels of Dioxins in Water (ppq).
Table B-18.	Mean Background Levels of Dibenzofurans in Water (ppq).
Table B-19.	Mean Background Environmental Levels of Dioxins in Sediments (ppt).
Table B-20.	Mean Background Environmental Levels of Dibenzofurans in Sediment (ppt).
Table B-21.	Mean Background Environmental Levels of PCBs in Sediment (ppt).

Table B-22. Mean Background Environmental Levels of Dioxins in Finfish (ppt).

Table B-23. Mean Background Levels of Dibenzofurans in Finfish (ppt).

Table B-24. Mean Background Levels of Dioxins in Food Products (ppt).

Table B-25. Mean Background Levels of Dibenzofurans in Food Products (ppt).

Table B-26. Mean Background Environmental Levels of Dioxins in Air (pg/m^3).

Table B-27. Mean Background Environmental Levels of Dibenzofurans in Air (pg/m^3).

Table B-1. Environmental Levels of Dioxins in Soil (ppt)

Chemical	Number samples	Number positive samples	Concentration range (ppt)	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
Tetrachlorodibenzo-p-dioxins (MW=321.98)									
2,3,7,8-TCDD	77	NR	<0.5-2.1	<0.5	British Isles	Background	NR	11	
	3	0	<0.2-<2.0	<1.43	Various parts of Europe	Rural	NR	10	
	2	2	2.4-0.84	1.62	Various parts of Europe	Industrial	NR	10	
	23	23	10-36000	2133	Midland, MI	Industrial		1	
	62	59	ND-270	55	Midland, MI	Residential		1	
	13	1	ND-2	<1	Henry, IL	Residential		1	
	22	6	ND-5	1	Middletown, OH	Residential		1	
	4	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	2	Finland	Industrial		2	
	33	33	41-52000	4300	Midland, MI	Industrial		3	
	11	9	ND-590	145	Midland, MI	Inddustrial		3	
	20	13	ND-9.4	2	US	Industrial		3	Urban area
	8	8	0.6-3.1	2	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
	12	12	1-7	3	England	Residential		7	Rural
	19	6	ND-4.2	1	England	Urban		8	
TCDDs	1	1	3.2	NA	Various parts of Europe	Rural		10	
	77	NR	<0.5-69	9.4	British Isles	Background	NR	11	
	NR	NR	200	NR	Muggenburger st. Hamburg,Germany	Industrial	1985	12	Maximum contents reported
	NR	NR	2.8	NR	Kirchsteinbek, Hamburg,Germany	Industrial	1985	12	Maximum contents reported
	NR	NR	0.9	NR	Ochsenwerder Landscheideweg, Hamburg,Germany	Contaminated site	1985	12	Maximum contents reported
	NR	NR	874	NR	Moorefleeter Brack Hamburg,Germany	Contaminated site	1985	12	Maximum contents reported
	47	11	ND-430	40.3	Ontario and U.S. Midwestern States.	Urban		13	

Table B-1. Environmental Levels of Dioxins in Soils (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range (ppt)	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
TCDDs (continued)	2	2	11.2-55.5	33.35	Various parts of Europe	Industrial		10	
	1	1	320	NA	Midland, MI	Industrial		1	
	7	5	ND-290	109	Midland, MI	Residential		1	
	5	0	ND	NA	Middletown, OH	Residential		1	
	3	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	89	Finland	Industrial		2	
	11	NR	ND-7	<1	Canada			4	Near Incinerator
	12	NR	ND-430	69	Canada			4	Near Incinerator
	53	0	ND	NA	Canada	Rural		4	
	29	NR	ND-1200	69	Canada	Urban		4	
	8	8	37-217	98	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Agriculture		6	
	12	12	17-120	42	England	Residential		7	Rural
	19	19	9-160	65	England	Urban		8	
Pentachlorodibenzo-p-dioxins (MW=356.42)									
1,2,3,7,8-PeCDD	NR	NR	NR	15	Finland	Industrial		2	
	8	8	2.6-18.3	10	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
	19	7	ND-11	2	England	Urban		8	
	77	NR	<0.5-2.4	<0.5	British Isles	Background	NR	11	
PeCDDs	1	1	4.6	NA	various parts of Europe	Rural		10	
	2	2	220-270	245	various parts of Europe	Industrial		10	
	77	NR	<0.5-46	6.6	British Isles	Background	NR	11	
	47	7	ND-580	38.45	Ontario and U.S. Midwestern States.	Urban		13	
	1	1	240	NA	Midland, MI	Industrial		1	
	6	2	ND-120	37	Midland, MI	Residential		1	
	5	0	ND	NA	Middletown, OH	Residential		1	

Table B-1. Environmental Levels of Dioxins in Soils (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range (ppt)	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
PeCDDs (continued)	3	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	900	Finland	Industrial		2	
	11	NR	ND-580	53	Canada			4	Near Incinerator
	12	NR	ND-540	81	Canada			4	Near Incinerator
	53	0	ND	ND	Canada	Rural		4	
	29	NR	ND-130	23	Canada	Urban		4	
	8	8	46-476	159	Sweden	Urban		5	Near Stockholm
	4	1	ND-38	10	Elk River, MN	Rural		6	Agriculture
	12	12	4-50	20	England	Residential		7	Rural
	19	19	6-190	69	England	Urban		8	
Hexachlorodibenzo-p-dioxins (MW=390.87)									
1,2,3,4,7,8-HxCDD	NR	NR	NR	<2	Finland	Industrial		2	
	8	8	4.3-8.0	6	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
1,2,3,6,7,8-HxCDD	NR	NR	NR	2100	Finland	Industrial		2	
	8	8	3.3-32.2	12	Sweden	Urban		5	Near Stockholm
	4	1	ND-14	4	Elk River, MN	Rural		6	AGriculture
1,2,3,7,8,9-HxCDD	NR	NR	NR	700	Finland	Industrial		2	
	8	8	1.6-16.6	8	Sweden	Urban		5	Near Stockholm
	4	2	ND-9.9	9	Elk River, MN	Rural		6	Agriculture
HxCDDs	1	1	4.7	NA	Various parts of Europe	Rural		10	
	2	2	200-330	265	Various parts of Europe	Industrial		10	
	77	NR	2.8-165	38	British Isles	Background	NR	11	
	47	13	ND-410	38.12	Ontario and U.S. Midwestern States.	Urban		13	
	20	8	ND-240	44.1	Canada and U.S.A.	Industrial		13	
	1	1	4000	NA	Midland, MI	Industrial		1	
	7	5	ND-410	172	Midland, MI	Residential		1	

Table B-1. Environmental Levels of Dioxins in Soils (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range (ppt)	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
HxCDDs (continued)	5	1	ND-72	14	Middletown, OH	Residential		1	
	3	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	7200	Finland	Industrial		2	
	11	NA	ND-170	15	Canada			4	Near Incinerator
	12	NA	ND-70	9	Canada			4	Near Incinerator
	53	0	ND	NA	Canada	Rural		4	
	29	0	ND	NA	Canada	Urban		4	
	8	8	43-349	156	Sweden	Urban		5	Near Stockholm
	4	4	12-99	48	Elk River, MN	Rural		6	Agriculture
	12	12	8-43	23	England	Residential		7	Rural
	19	19	23-340	154	England	Urban		8	
Heptachlorodibenzo-p-dioxins (MW=425.31)									
1,2,3,4,6,7,8-HpCDD	NR	NR	NR	4700	Finland	Industrial		2	
	8	8	43-492	144	Sweden	Urban		5	Near Stockholm
	4	4	37-360	194	Elk River, MN	Rural		6	Agriculture
1,2,3,4,6,7,9-HpCDD	NR	NR	NR	7100	Finland	Industrial		2	
HpCDDs	3	1	<10-17	NA	Various parts of Europe	Rural		10	
	2	2	370-1600	985	Various parts of Europe	Industrial		10	
	77	NR	7.5-234	66	British Isles	Background	NR	11	
	30	3	ND-91	5.4	Ontario and U.S. Midwestern States.	Rural		13	
	47	25	ND-2400	211.8	Ontario and U.S. Midwestern States.	Urban		13	
	20	19	ND-5000	1196.9	Canada and U.S.A.	Industrial		13	
	1	1	75000	NA	Midland, MI	Industrial		1	
	7	7	150-2400	930	Midland, MI	Residential		1	
	5	5	23-200	113	Middletown, OH	Residential		1	
	3	3	25-91	54	MN	Pristine		1	
	11	NR	ND-390	90	Canada			4	Near Incinerator
	12	NR	ND-300	43	Canada			4	Near Incinerator
	53	0	ND	NA	Canada	Rural		4	
	29	NR	ND-1100	93	Canada	Urban		4	
	8	8	83-904	277	Sweden	Urban		5	Near Stockholm

Table B-1. Environmental Levels of Dioxins in Soils (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range (ppt)	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
HpCDDs (continued)	4	4	62-640	346	Elk River, MN	Rural		6	Agriculture
	12	12	20-130	64	England	Residential		7	Rural
	19	19	77-5500	817	England	Urban		8	
Octachlorodibenzo-p-dioxin (MW=460.76)									
OCDD	1	1	14	NA	Various parts of Europe	Rural		10	
	2	2	140-160	160	Various parts of Europe	Industrial		10	
	77	NR	29-832	191	British Isles	Background	NR	11	
	30	17	44-810	67.33	Ontario and U.S. Midwestern States.	Rural		13	
	47	38	ND-12000	1599.1	Ontario and U.S. Midwestern States.	Urban		13	
	20	20	15-26000	3442.25	Canada and U.S.A.	Industrial		13	
	1	1	375000	NA	Midland, MI	Industrial		1	
	7	7	330-12000	4473	Midland, MI	Residential		1	
	5	5	170-10600	2418	Middletown, OH	Residential		1	
	3	3	25-91	54	MN	Pristine		1	
	NR	NR	NR	6200	Finland	Industrial		2	
	11	NR	ND-3500	663	Canada			4	Near Incinerator
	12	NR	ND-1500	570	Canada			4	Near Incinerator
	53	NR	ND-100	31	Canada	Rural		4	
	29	NR	ND-16000	2464	Canada	Urban		4	
	8	8	113-2659	687	Sweden	Urban		5	Near Stockholm
	4	4	340-3300	1655	Elk River, MN	Rural		6	Agriculture
	12	12	20-150	58	England	Residential		7	Rural
	19	19	176-99000	9980	England	Urban		8	

Footnote References

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = Not applicable;
ND = Non-detect;
NR = Not reported.
Detection limits varied by study and as was different for different compounds, but generally were 1 to 5 ng/kg (ppt). Descriptions provided were those given by reference or surmised from study description when not given.

Table B-1. Environmental Levels of Dioxins in Soils (ppt) (continued)

Sources:

1. EPA (1985)
2. Kitunen and Salkinoja-Salonen (1990).
3. Nestrick, et al. (1986).
4. Pearson, et al. (1990).
5. Broman, et al. (1990).
6. Reed, et al. (1990).
7. Stenhouse and Badsha (1990).
8. Creaser, et al. (1990)
10. Rappe and Kjeller (1987)
11. Creaser, et al. (1989)
12. Sievers and Friesel (1989)
13. Birmingham (1990)

Table B-2. Environmental Levels of Dibenzofurans in Soil (ppt)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Sampley ear	Ref. no.	Comments
Tetrachlorodibenzofurans (MW=305.98)									
2,3,7,8-TCDF	2	2	27-450	238	Midland, MI	Industrial		1	
	8	3	ND-15	5	Midland, MI	Residential		1	
	5	2	ND-6	2	Middletown, OH	Residential		1	
	3	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	<2	Finland	Industrial		2	
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
	12	12	3-50	17	England	Residential		7	Rural
2,3,4,8/2,3,7,8-TCDF	8	8	8.4-57.5	22	Sweden	Urban		5	Near Stockholm
TCDFs	3	3	7.7-11	9.3	Various parts of Europe	Rural		10	
	2	2	320-370	345	Various parts of Europe	Industrial		10	
	77	NR	<0.5-237	25	British Isles	Background		11	
	47	13	ND-120	42.92	Ontario and U.S. Midwestern states	Urban		13	
	20	3	ND-1850	1016.66	Ontario and U.S. Midwestern states	Industrial		13	
	1	0	ND	NA	Midland, MI	Residential		1	
	1	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	300	Finland	Industrial		2	
	11	NR	ND-71	10	Canada			4	Near Incinerator
	12	0	ND	NA	Canada			4	Near Incinerator
	53	1	ND-280	NA	Canada	Rural		4	
	29	NR	ND-120	15	Canada	Urban		4	
	8	8	109-454	237	Sweden	Urban		5	Near Stockholm
	4	1	ND-1.2	<1	Elk River, MN	Rural		6	Agriculture
	12	12	20-300	102	England	Residential		7	Rural
	19	19	29-950	232	England	Urban		8	

Table B-2. Environmental Levels of Dibenzofurans in Soil (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Sampley ear	Ref. no.	Comments
Pentachlorodibenzofurans (MW=340.42)									
1,2,3,7,8-PeCDF	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
	12	12	1-10	4	England	Residential		7	Rural
2,3,4,7,8-PeCDF	NR	NR	NR	580	Finland	Industrial		2	
	8	8	3.1-26.5	13	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
	12	12	1-5	2	England	Residential		7	Rural
1,2,3,7,8/1,2,3,4,8-PeCDF	NR	NR	NR	82	Finland	Industrial		2	
	8	8	7.4-32.1	16	Sweden	Urban		5	Near Stockholm
PeCDFs	3	3	6.7-14	11.23	Various parts of Europe	Rural		10	
	77	NR	<0.5-185	23	British Isles	Background		11	
	47	4	ND-110	41.5	Ontario and U.S. Midwestern states	Urban		13	
	20	5	ND-285	86	Ontario and U.S. Midwestern states	Industrial		13	
	2	2	200-450	325	Various parts of Europe	Industrial		10	
	1	1	900	NA	Midland, MI	Industrial		1	
	8	2	ND-110	19	Midland, MI	Residential		1	
	5	0	ND	NA	Middletown, OH	Residential		1	
	NR	NR	NR	27000	Finland	Industrial		2	
	11	0	ND	NA	Canada			4	Near Incinerator
	12	0	ND	NA	Canada			4	Near Incinerator
	53	0	ND	NA	Canada	Rural		4	
	29	NR	ND-160	35	Canada	Urban		4	
	8	8	36-457	149	Sweden	Urban		5	Near Stockholm
	4	3	18-45	35	Elk River, MN	Rural		6	Agriculture
	12	12	6-70	31	England	Residential		7	Rural

Table B-2. Environmental Levels of Dibenzofurans in Soil (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Sampley ear	Ref. no.	Comments
PeCDFs (continued)	19	19	19-830	189	England	Urban		8	
Hexachlorodibenzofurans (MW=374.87)									
1,2,3,4,7,8/1,2,3,4,7,9-HxCDF	NR	NR	NR	920	Finland	Industrial		2	
	8	8	7.8-29.1	16	Sweden	Urban		5	Near Stockholm
1,2,3,4,7,8-HxCDF	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
1,2,3,6,7,8-HxCDF	NR	NR	NR	<2	Finland	Industrial		2	
	8	8	7.7-28.9	14	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
1,2,3,7,8,9-HxCDF	NR	NR	NR	<2	Finland	Industrial		2	
	8	8	0.5-3.8	1	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
2,3,4,6,7,8-HxCDF	NR	NR	NR	<2	Finland	Industrial		2	
	4	1	ND-7.1	2	Elk River, MN	Rural		6	Agriculture
HxCDF	3	3	11-16	13	Various parts of Europe	Rural		10	
	77	NR	4.3-212	41	British Isles	Background		11	
	47	6	ND-260	94.83	Ontario and U.S. Midwestern states	Urban		13	
	20	7	ND-420	177.85	Ontario and U.S. Midwestern states	Industrial		13	
	2	2	270-1900	1085	Various parts of Europe	Industrial		10	
	1	1	15400	NA	Midland, MI	Industrial		1	
	8	3	64-260	62	Midland, MI	Residential		1	
	5	0	ND	NA	Middletown, OH	Residential		1	
	3	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	7200	Finland	Industrial		2	
	11	0	ND	NA	Canada			4	Near Incinerator
	12	0	ND	NA	Canada			4	Near Incinerator

Table B-2. Environmental Levels of Dibenzofurans in Soil (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Sampley ear	Ref. no.	Comments
HxCDF (continued)	53	0	ND	NA	Canada	Rural		4	
	29	NR	ND-120	9	Canada	Urban		4	
	8	8	53-308	145	Sweden	Urban		5	Near Stockholm
	4	4	7-150	66	Elk River, MN	Rural		6	Agriculture
	12	12	6-50	24	England	Residential		7	Rural
	19	19	17-660	156	England	Urban		8	
Heptachlorodibenzofurans (MW=409.31)									
1,2,3,4,6,7,8-HpCDF	NR	NR	NR	4700	Finland	Industrial		2	
	8	8	31-134	73	Sweden	Urban		5	Near Stockholm
	4	4	11-80	47	Elk River, MN	Rural		6	Agriculture
1,2,3,4,7,8,9-HpCDF	NR	NR	NR	<5	Finland	Industrial		2	
	8	8	1.0-6.3	3	Sweden	Urban		5	Near Stockholm
	4	0	ND	NA	Elk River, MN	Rural		6	Agriculture
HpCDFs	3	3	14-22	18	Various parts of Europe	Rural		10	
	2	2	260-4500	2380	Various parts of Europe	Industrial		10	
	77	NR	1.5-138	26	British Isles	Background		11	
	47	10	ND-120	283.1	Ontario and U.S. Midwestern states	Urban		13	
	20	15	ND-3750	732.66	Ontario and U.S. Midwestern states	Industrial		13	
	1	1	75000	NA	Midland, MI	Industrial		1	
	8	6	ND-820	300	Midland, MI	Residential		1	
	5	1	ND-43	9	Middletown, OH	Residential		1	
	3	0	ND	NA	MN	Pristine		1	
	11	NR	ND-180	30	Canada			4	Near Incinerator
	12	0	ND	NA	Canada			4	Near Incinerator
	53	0	ND	NA	Canada	Rural		4	

Table B-2. Environmental Levels of Dibenzofurans in Soil (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Sampley ear	Ref. no.	Comments
HpCDFs (continued)	29	NR	ND-410	29	Canada	Urban		4	
	8	8	31-187	81	Sweden	Urban		5	Near Stockholm
	4	4	30-260	100	Elk River, MN	Rural		6	Agriculture
	12	12	4-59	20	England	Residential		7	Rural
	19	19	16-458	152	England	Urban		8	
Octachlorodibenzofurans (MW=444.76)									
OCDF	1	1	5.7	NA	Various parts of Europe	Rural		10	
	77	NR	<2.0-144	27	British Isles	Background		11	
	47	15	ND-	184.8	Canada and USA	Urban		13	
	20	15	ND-5200	842.53	Canada and USA	Industrial		13	
	2	2	68-71	69.5	Various parts of Europe	Industrial		10	
	1	1	8600	NA	Midland, MI	Industrial		1	
	8	6	ND-660	240	Midland, MI	Residential		1	
	5	1	ND-50	10	Middletown, OH	Residential		1	
	3	0	ND	NA	MN	Pristine		1	
	NR	NR	NR	3000	Finland	Industrial		2	
	11	NR	ND-33	4	Canada			4	Near Incinerator
	12	NR	ND-230	43	Canada			4	Near Incinerator
	53	0	ND	NA	Canada	Rural		4	
	29	NR	ND-600	50	Canada	Urban		4	
	8	8	2.9-19.0	8	Sweden	Urban		5	Near Stockholm
	4	3	60-270	113	Elk River, MN	Rural		6	Agriculture
	12	12	10-90	30	England	Residential		7	Rural
	19	19	7-1100	196	England	Urban		8	

Table B-2. Environmental Levels of Dibenzofurans in Soil (ppt) (continued)

Footnote References

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = not available;
NR = not reported;
ND = Non-detect.
Detection limits varied by study and as was different for different compounds, but generally were 1 to 5 ng/kg (ppt). Descriptions provided were those given by reference or surmised from study description when not given.

Sources:

1. EPA (1985)
2. Kitunen and Salkinoja-Salonen (1990.
3. Nestrick, et al. (1986).
4. Pearson, et al. (1990).
5. Broman, et al. (1990).
6. Reed, et al. (1990).
7. Stenhouse and Badsha (1990).
8. Creaser, et al. (1990)
10. Rappe and Kjeller (1987)
11. Creaser, et al. (1989)
12. Sievers and Friesel (1989)
13. Birmingham (1990)

Table B-3. Environmental Levels of Dioxins in Water (ppq)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Samp. year	Ref. no.	Comments
Tetrachlorodibenzo-p-dioxins (MW=321.98)									
2,3,7,8-TCDD	1	0	ND(0.7)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	0	ND(.02-.024)	NA	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
TCDDs	185	1	ND-40	0.23	Ontario, Canada	NR	83-89	1	raw surface drinking water
	22	0	ND(.4-2.6)	NA	New York State	NR	86-88	2	treated surface drinking water
	1	1	1.7	1.7	Lockport, New York	NR	88	2	raw surface drinking water
	2	2	.05-.084	.067	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Pentachlorodibenzo-p-dioxins (MW=356.42)									
1,2,3,7,8-PeCDD	1	0	ND(1.0)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	0	ND(.025-.039)	NA	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
PeCDDs	1	0	ND(1.0)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	22	0	ND(1.2-7.4)	NA	New York State	NR	86-88	2	treated surface drinking water
	2	2	.067-.12	.094	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Hexachlorodibenzo-p-dioxins (MW=390.87)									
1,2,3,4,7,8-HxCDD	1	0	ND(1.8)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	1	ND-.054	.027	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
1,2,3,6,7,8-HxCDD	1	0	ND(1.5)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	1	ND-.12	.06	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
1,2,3,7,8,9-HxCDD	1	0	ND(1.5)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	1	ND-.075	.038	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
HxCDDs	1	0	ND(1.5)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	22	0	ND(.4-4.7)	NA	New York State	NR	86-88	2	treated surface drinking water

Table B-3. Environmental Levels of Dioxins in Water (ppq) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Samp. year	Ref. no.	Comments
HxCDDs (continued)	2	2	.13-.67	.4	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Heptachlorodibenzo-p-dioxins (MW=425.31)									
1,2,3,4,6,7,8-HpCDD	1	0	ND(2.8)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	2	.15-.30	.22	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
HpCDDs	1	0	ND(2.8)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	22	0	ND(.4-6.8)	NA	New York State	NR	86-88	2	treated surface drinking water
	2	2	.17-.64	.40	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Octachlorodibenzo-p-dioxin (MW=460.76)									
1,2,3,4,6,7,8,9-OCDD	185	32	ND-175	8.45	Ontario, Canada	NR	83-89	1	raw surface drinking water
	214	4	ND-46	0.63	Ontario, Canada	NR	83-89	1	treated surface drinking water

Footnote References

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = not applicable;
ND = non-detected (limit of detection);
NR = not reported.
Descriptions provided were those given by reference or surmised from study description when not given.

Sources:
1. Jobb, et al. (1990)
2. Meyer, et al. (1989)
3. Rappe, et al. (1989b)

Table B-4. Environmental Levels of Dibenzofurans in Water (ppq)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Samp. year	Ref. no.	Comments
Tetrachlorodibenzofurans (MW=305.98)									
2,3,7,8-TCDF	22	1	ND-1.2	0.05	New York State	NR	86-88	2	treated surface drinking water
	2	2	.022-.026	.024	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
	1	0	ND(0.7)	NA	Lockport, New York	NR	88	2	raw surface drinking water
TCDFs	22	1	ND-2.6	0.12	New York State	NR	86-88	2	treated surface drinking water
	1	1	18	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	2	.21-.23	0.22	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Pentachlorodibenzofurans (MW=340.42)									
1,2,3,7,8-PeCDF	1	1	2.0	NA	Lockport, New York	NR	88	2	raw surface drinking water
2,3,4,7,8-PeCDF	2	2	.014-.019	.016	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
	1	0	ND(1.0)	NA	Lockport, New York	NR	88	2	raw surface drinking water
1,2,3,4,8/1,2,3,7,8-PeCDF	2	2	.013-.025	.019	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
PeCDFs	1	1	27	NA	Lockport, New York	NR	88	2	raw surface drinking water
	22	0	ND(0.3-4.0)	NA	New York State	NR	86-88	2	treated surface drinking water
	2	2	.13-.21	.17	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Hexachlorodibenzofurans (MW=374.87)									
1,2,3,4,7,8-HxCDF	1	1	39	NA	Lockport, New York	NR	88	2	raw surface drinking water
1,2,3,6,7,8-HxCDF	1	1	9.2	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	2	.019-.025	.022	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
2,3,4,6,7,8-HxCDF	1	0	ND(1.3)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	1	ND-.027	.014	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
1,2,3,7,8,9-HxCDF	1	0	ND(1.2)	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	1	ND-.022	.011	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water

Table B-4. Environmental Levels of Dibenzofurans in Water (ppq) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Location	Location description	Samp. year	Ref. no.	Comments
1,2,3,4,7,9/1,2,3,4,7,8-HxCDF	2	2	.021-.026	.024	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
HxCDFs	1	1	85	NA	Lockport, New York	NR	88	2	raw surface drinking water
	22	0	ND(.3-4.4)	NA	New York State	NR	86-88	2	treated surface drinking water
	2	2	.17-.19	.18	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Heptachlorodibenzofurans (MW=409.31)									
1,2,3,4,6,7,8-HpCDF	1	1	210	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	2	.083-.13	.11	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
1,2,3,4,7,8,9-HpCDF	2	2	.03-.058	.044	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
HpCDFs	1	1	210	NA	Lockport, New York	NR	88	2	raw surface drinking water
	22	0	ND(.8-6.6)	NA	New York State	NR	86-88	2	treated surface drinking water
	2	2	.18-.35	.26	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water
Octachlorodibenzofurans (MW=444.76)									
1,2,3,4,6,7,8,9-OCDF	22	2	ND-0.8	.07	New York State	NR	86-88	2	treated surface drinking water
	1	1	230	NA	Lockport, New York	NR	88	2	raw surface drinking water
	2	2	.15-.36	.26	Eman river, Sweden	PCB contaminated	NR	3	raw surface drinking water

Footnote References

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = not applicable;
ND = non-detected (limit of detection);
NR = not reported.
Descriptions provided were those given by reference or surmised from study description when not given.

Sources: 2. Meyer, et al. (1989)
3. Rappe, et al. (1989b)

Table B-5. Environmental Levels of Dioxins in Sediments (ppt)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
Tetrachlorodibenzo-p-dioxins (MW=321.98)										
2,3,7,8-TCDD	18	0	N/A	N/A	Dry	South Central Finland	Various	88/89	1	AB
	9	8	ND-730	236	Dry	Newark, NJ	Industrial	85/86	2	0-2", AC
	4	4	75-2500	1769	Dry	Newark, NJ	Industrial	85/86	2	2-4", AC
	2	2	190-1200	695	Dry	Newark, NJ	Industrial	85/86	2	4-8", AC
	1	1	N/A	680	Dry	Newark, NJ	Industrial	85/86	2	12-16", AC
	1	1	N/A	150	Dry	Newark, NJ	Industrial	85/86	2	20-24", AC
	2	2	660-1100	880	Dry	Newark, NJ	Industrial	85/86	2	24-28", AC
	3	1	ND-7600	367	Dry	Newark, NJ	Industrial	85/86	2	28-32", AC
	3	3	390-2900	1227	Dry	Newark, NJ	Industrial	85/86	2	32-36", AC
	1	1	N/A	93	Dry	Newark, NJ	Industrial	85/86	2	40-44", AC
	2	1	ND-7600	3800	Dry	Newark, NJ	Industrial	85/86	2	48-52", AC
	1	1	N/A	21000	Dry	Newark, NJ	Industrial	85/86	2	108-111", AC
	1	0	N/A	ND	Dry	Long Island Sound	Reference Site	NR	4	A
	12	6	ND-57	13	Dry	New England	Industrial	NR	4	A/3 sites
	4	0	N/A	ND	Dry	Seattle, WA	Industrial	NR	4	A
	4	0	N/A	ND	NR	Central Minnesota	Rural	NR	5	A
	2	2	1.0-1.4	1.2	Dry	Baltic Sea	Reference Site	NR	13	A
	3	2	ND-2.4	1.5	NR	Stockholm Sweden	Various	NR	12	A
	4	4	1.9-26	9.6	Dry	Iggesund Sweden	Industrial	NR	13	A/papermill
	4	1	ND(.5-2)-3	1.19	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	1.2-32	12.05	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm

Table B-5. Environmental Levels of Dioxins in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
2,3,7,8-TCDD (continued)	5	3	ND(4-11)-110	28.08	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
	4	4	0.03-0.11	0.06	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	0-1 cm, E
	1	1	N/A	0.04	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	4-6 cm, E
	1	1	N/A	0.01	Dry	Fjord between Demark, Sweden, and Norway	Industrial	87	14	9-13 cm, E
	5	5	3.4-1,500	375	NR	Hamburg Germany	Urban	NR	15	
TCDDs	18	14	ND-1,400	372	Dry	South Central Finland	Various	88/89	1	AB
	1	1	N/A	26	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	0-0.5 cm, AD
	1	1	N/A	12	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	5-6 cm, AD
	1	0	N/A	ND	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	8-9 cm, AD
	4	0	N/A	ND	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	0	N/A	ND	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	7	ND-44	17	NR	NY/Mass	NR	NR	10	A
	3	3	21-69	38	NR	Stockholm Sweden	Various	NR	12	A
	4	4	21-66	45	Dry	Iggesund Sweden	Industrial	NR	13	A/paper mill
	2	2	19-35	27	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	1.4-6.7	4.2	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	0-2 cm, E
	1	1	N/A	13	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	4-6 cm, E

Table B-5. Environmental Levels of Dioxins in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments*
TCDDs (continued)	1	1	N/A	5.0	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	9-13 cm, E
	5	5	80-1,700	564	NR	Hamburg Germany	Urban	NR	15	
Pentachlorodibenzo-p-dioxins (MW=356.42)										
1,2,3,7,8-PeCDD	4	3	ND(2)-25	7.68	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	7.4-95	44.1	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	4	ND(14)-100	49.6	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
PeCDDs	1	1	N/A	12	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	0-0.5 cm, AD
	1	1	N/A	11	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	5-6 cm, AD
	1	0	N/A	ND	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	8-9 cm, AD
	4	0	N/A	ND	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	0	N/A	ND	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	7	ND-235	50	NR	NY/Mass	NR	NR	10	A
	3	3	86-230	138	NR	Stockholm Sweden	Various	NR	12	A
	4	4	52-500	209	Dry	Iggesund Sweden	Industrial	NR	13	A/paper mill
	2	2	52-100	76	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	1.7-41	19	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	0-2 cm, E
	1	1	N/A	6.6	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	4-6 cm, E
	1	1	N/A	15	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	9-13 cm, E

Table B-5. Environmental Levels of Dioxins in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
PeCDDs (continued)	5	5	260-2,700	1,112	NR	Hamburg Germany	Urban	NR	15	
Hexachlorodibenzo-p-dioxins(MW=390.87)										
1,2,3,4,7,8-HxCDD	4	3	ND(6)-19	6.68	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	6.1-33	20.5	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	14-94	36.0	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
1,2,3,6,7,8-HxCDD	4	4	4.9-120	36.38	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	21-450	188	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	36-600	236	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
1,2,3,7,8,9-HxCDD	4	4	3.9-51	18.1	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	18-200	95.5	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	18-330	149	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
HxCDDs	1	1	N/A	10	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	0-0.5 cm, AD
	1	1	N/A	8	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	5-6 cm, AD
	1	0	N/A	ND	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	8-9 cm, AD
	4	2	ND-14	5.2	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	6	ND-5,700	1,157	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	10	ND-1,335	399	NR	NY/Mass	NR	NR	10	A
	3	3	16-49	28	NR	Stockholm Sweden	Various	NR	12	A
	2	2	120-170	145	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	130-1,900	608	Dry	Iggesund Sweden	Industrial	NR	13	A/paper mill

Table B-5. Environmental Levels of Dioxins in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
HxCDDs (continued)	4	4	2.3-27.0	14	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	0-2 cm, E
	1	1	N/A	16	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	4-6 cm, E
	1	1	N/A	14	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	9-13 cm, E
	5	5	580-7,500	2,744	NR	Hamburg Germany	Urban	NR	15	
Heptachlorodibenzo-p-dioxins (MW=425.31)										
HpCDDs	4	4	7.3-110	71	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	20	ND-320,000	51,680	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	11	ND-18,950	4,168	NR	NY/Mass	NR	NR	10	A
	3	3	880-5,700	2,233	NR	Stockholm Sweden	Various	NR	12	A
	2	2	79-210	145	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	90-340	190	Dry	Iggesund Sweden	Industrial	NR	13	A/paper mill
	4	4	2.2-19	12.4	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	0-2 cm, E
	1	1	N/A	7.2	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	4-6 cm, E
	1	1	N/A	7.1	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	9-13 cm, E
	5	5	1,300-8,600	4,040	NR	Hamburg Germany	Urban	NR	15	
Octachlorodibenzo-p-dioxin (MW=460.76)										
OCDD	18	3	ND-42	6.1	Dry	South Central Finland	Various	88/89	1	AB
	9	9	3,100-14,000	8,100	Dry	Newark, NJ	Industrial	85/86	2	0-2", AC
	4	4	5,300-23,000	14,100	Dry	Newark, NJ	Industrial	85/86	2	2-4", AC

Table B-5. Environmental Levels of Dioxins in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
OCDD (continued)	2	2	10,000-31,000	20,500	Dry	Newark, NJ	Industrial	85/86	2	4-8", AC
	1	1	N/A	7,500	Dry	Newark, NJ	Industrial	85/86	2	12-16", AC
	1	1	N/A	5,900	Dry	Newark, NJ	Industrial	85/86	2	20-24", AC
	2	2	11,000-19,000	15,000	Dry	Newark, NJ	Industrial	85/86	2	24-28", AC
	2	2	5,500-22,000	13,800	Dry	Newark, NJ	Industrial	85/86	2	28-32", AC
	3	3	5,600-42,000	17,800	Dry	Newark, NJ	Industrial	85/86	2	32-36", AC
	1	1	N/A	5,500	Dry	Newark, NJ	Industrial	85/86	2	40-44", AC
	2	2	4,400-24,000	14,200	Dry	Newark, NJ	Industrial	85/86	2	48-52", AC
	1	1	N/A	38,000	Dry	Newark, NJ	Industrial	85/86	2	108-111", AC
	1	1	N/A	560	Dry	Siskiwit Lake, Isle Royale, Lake Superior	Pristine	NR	3	0-0.5 cm, AD
	1	1	N/A	390	Dry	Siskiwit Lake, Isle Royale, Lake Superior	Pristine	NR	3	5-6 cm, AD
	1	1	N/A	54	Dry	Siskiwit Lake, Isle Royale, Lake Superior	Pristine	NR	3	8-9 cm, AD
	4	4	450-600	518	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	20	ND-980,000	141,420	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	12	1,990-15,500	8,201	NR	NY/Mass	NR	NR	10	A
	7	7	12-250	145	Dry	Jackfish Bay, Lake Superior	Various	NR	11	Papermill, atmospheric contamination
	3	3	260-3,100	1,290	NR	Stockholm Sweden	Various	NR	12	A
	2	2	89-250	170	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	96-330	194	Dry	Iggesund Sweden	Industrial	NR	13	A/paper mill

Table B-5. Environmental Levels of Dioxins in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
OCDD (continued)	4	4	180-16,830	5195	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	800-2200	1775	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	1320-6090	3040	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
	4	4	3.6-45	24	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	0-2 cm, E
	1	1	N/A	10	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	4-6 cm, E
	1	1	N/A	6.9	Dry	Fjord between Denmark, Sweden, and Norway	Industrial	87	14	9-13 cm, E
	5	5	2,800-15,000	7,560	NR	Hamburg Germany	Urban	NR	15	

[a]Key A LOD (Ref 1) = 20 to 50 ppt, LOD (Ref 2) = 22 to 60 ppt, LOD (Ref 3) - 0.4 ppt, LOD (Ref 4) = 0.7 to 12.0 ppt, LOD (Ref 5) = 0.61 to 4.1 ppt, LOD (Ref 8) = 10 to 500 ppt, LOD (Ref 10) = 3 ppt, LOD (Ref 12) = 1 to 20 ppt, LOD (Ref 13) = 1 to 6.6 ppt.

B Dry surface sediments from 18 lakes.

C Industry produced 2,4,5-Trichlorophenate (2,4,5-T precursor).

D No anthropogenic inputs into drainage basin--only atmospheric sources into lake.

E Mg-Production Facility.

Table B-5. Environmental Levels of Dioxins in Sediments (ppt) (continued)

Notes

NR = Not Reported
N/A = Not Applicable
ND = Not Detected
ppt = Parts per trillion

Sources:

1. Koistinen et al. (1990)
2. Bopp et al. (1991)
3. Czuczwa et al. (1984)
4. Norwood et al. (1989)
5. Reed et al. (1990)
6. Sonzongi et al. (1991)
7. Oliver and Nilmi (1988)
8. McKee et al. (1990)
9. Huckins et al. (1988)
10. Petty et al. (1982)
11. Sherman et al. (1990)
12. Rappe and Kjeller (1987)
13. Rappe et al. (1989a)
14. Oehme et al. (1989)
15. Gotz and Schumacher (1990)
16. Smith et al. (1990)
17. Kjeller et al. (1990)

Table B-6. Environmental Levels of Dibenzofurans in Sediment (ppt)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Weight basis	Location	Location description	Sample year	Ref. no.	Comments[a]
Tetrachlorodibenzofurans (MW=305.98)										
2,3,7,8-TCDF	18	0	N/A	ND	Dry	South Central Finland	Various	88/89	1	AB
	9	9	24-490	232	Dry	Newark, NJ	Industrial	85/86	2	0-2", AC
	4	4	150-1,400	855	Dry	Newark, NJ	Industrial	85/86	2	2-4", AC
	2	2	580-1,200	890	Dry	Newark, NJ	Industrial	85/86	2	4-8", AC
	1	1	N/A	370	Dry	Newark, NJ	Industrial	85/86	2	12-16", AC
	1	1	N/A	300	Dry	Newark, NJ	Industrial	85/86	2	20-24", AC
	2	2	300-390	345	Dry	Newark, NJ	Industrial	85/86	2	24-28", AC
	3	2	N/D-530	243	Dry	Newark, NJ	Industrial	85/86	2	28-32", AC
	3	3	190-730	370	Dry	Newark, NJ	Industrial	85/86	2	32-36", AC
	1	1	140	140	Dry	Newark, NJ	Industrial	85/86	2	40-44", AC
	2	2	80-3,100	1,590	Dry	Newark, NJ	Industrial	85/86	2	48-52", AC
	1	1	4,500	4,500	Dry	Newark, NJ	Industrial	85/86	2	108-111", AC
	1	1	N/A	15	Dry	Long Island Sound	Reference Site	NR	4	A
	12	12	8.8-1,400	566	Dry	New England	Industrial	NR	4	A/3 sites
	4	0	N/A	ND	Dry	Seattle, WA	Industrial	NR	4	A
	4	1	ND-0.31	0.08	NR	Central Minnesota	Rural	NR	5	A/4 sites
	2	2	8.3-14	11	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	11-210	72	Dry	Iggesund Sweden	Industrial	NR	13	A/papermill
	4	4	4.9-170	46.5	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	41-320	175.5	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	54-810	241.2	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm

Table B-6. Environmental Levels of Dibenzofurans in Sediment (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Weight basis	Location	Location description	Sample year	Ref. no.	Comments[a]
2,3,7,8-TCDF (continued)	4	4	0.7-9.6	5.2	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	0-2cm, E
	1	1	N/A	13	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	4-6cm, E
	1	1	N/A	5.2	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	9-13cm, E
TCDFs	1	1	N/A	15	Dry	Siskiwit Lake, Isle Royale, Lake Superior	Pristine	NR	3	0-0.5cm, AD
	1	1	N/A	18	Dry	Siskiwit Lake, Isle Royale, Lake Superior	Pristine	NR	3	5-6cm, AD
	1	0	N/A	ND	Dry	Siskiwit Lake, Isle Royale, Lake Superior	Pristine	NR	3	8-9cm, AD
	4	2	ND-0.54	0.21	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	0	N/A	ND	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	11	ND-200	58	NR	NY/MASS	NR	NR	10	A
	7	7	2.4-6,223	1,260	Dry	Jackfish Bay, Lake Superior	Various	NR	11	Papermill/ atmospheric contamination
	3	3	120-290	187	NR	Stockholm Sweden	Various	NR	12	A
	2	2	87-130	109	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	79-360	180	Dry	Iggesund Sweden	Industrial	NR	13	A/papermill
	4	4	6.7-54	30	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	0-2cm, E
	1	1	N/A	63	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	4-6cm, E
	1	1	N/A	23	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	9-13cm, E

Table B-6. Environmental Levels of Dibenzofurans in Sediment (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Weight basis	Location	Location description	Sample year	Ref. no.	Comments[a]
TCDFs (continued)	5	5	170-1070	526	NR	Hamburg Germany	Urban	NR	15	
Pentachlorodibenzofurans (MW=340.42)										
1,2,3,7,8-PeCDF	4	4	5.6-110	34.35	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	27-120	74.3	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	50-300	120	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
2,3,4,7,8-PeCDF	4	4	7.8-99	34.7	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	25-110	74.2	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	36-250	108	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
PeCDFs	1	1	N/A	5.0	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	0-0.5cm AD
	1	1	N/A	2.0	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	5-6cm AD
	1	0	N/A	ND	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	8-9cm AD
	4	2	ND-25	7.4	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	0	N/A	ND	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	9	ND-193	64	NR	NY/MASS	NR	NR	10	A
	3	3	130-260	177	NR	Stockholm Sweden	Various	NR	12	A
	2	2	66-125	96	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	48-58	55	Dry	Iggesund Sweden	Industrial	NR	13	A/papermill
	4	4	7.7-81	47	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	0-2cm, E
	1	1	N/A	24	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	4-6cm, E

Table B-6. Environmental Levels of Dibenzofurans in Sediment (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Weight basis	Location	Location description	Sample year	Ref. no.	Comments[a]
PeCDFs (continued)	1	1	N/A	44	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	9-13cm, E
	5	5	1,300-5,200	2,980	NR	Hamburg Germany	Urban	NR	15	
Hexachlorodibenzofurans (MW=374.87)										
1,2,3,4,7,8-HxCDF	4	4	9.3-120	44.3	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	28-170	89.5	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	32-460	163	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
1,2,3,6,7,8-HxCDF	4	4	3.7-73	26.7	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	15-110	64.7	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	25-140	73.4	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
1,2,3,7,8,9-HxCDF	4	3	ND(2)-25	9.38	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	2	ND(.9-3)-4.4	1.58	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	2	ND(5-14)-14	5.68	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
2,3,4,6,7,8-HxCDF	4	4	1.8-78	26.2	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	32-130	77.7	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	36-110	70.8	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
HxCDFs	1	1	N/A	2.0	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	0-0.5cm, AD
	1	1	N/A	2.0	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	5-6cm, AD
	1	0	N/A	ND	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	8-9cm, AD
	4	1	ND-12	3.0	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	17	ND-6,500	1,339	NR	Ontario Canada	Industrial	88	8	A/25 sites

Table B-6. Environmental Levels of Dibenzofurans in Sediment (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Weight basis	Location	Location description	Sample year	Ref. no.	Comments[a]
HxCDFs (continued)	12	10	ND-377	133	NR	NY/MASS	NR	NR	10	A
	3	3	92-250	187	NR	Stockholm Sweden	Various	NR	12	A
	2	2	78-150	114	Dry	Baltic Sea	Reference Site	NR	13	A
	4	4	59-150	104	Dry	Iggesund Sweden	Industrial	NR	13	A/papermill
	4	4	23-283	148	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	0-2cm, E
	1	1	N/A	166	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	4-6cm, E
	1	1	N/A	131	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	9-13cm, E
	5	5	930-8,600	4,106	NR	Hamburg Germany	Urban	NR	15	
Heptachlorodibenzofurans (MW=409.31)										
1,2,3,4,6,7,8-HpCDF	8	8	59-2180	558	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	12	12	130-1030	584	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	10	10	21-8400	1764	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
1,2,3,4,7,8,9-HpCDF	4	3	ND(4)-42	14.25	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	4.3-91	33.2	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	4	ND(21)-260	78.3	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm
HpCDFs	4	3	ND-30	16	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	20	ND-53,000	11,715	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	10	ND-2,436	1,039	NR	NY/MASS	NR	NR	10	A
	3	3	190-1,500	997	NR	Stockholm Sweden	Various	NR	12	A
	2	2	79-180	130	Dry	Baltic Sea	Reference Site	NR	13	A

Table B-6. Environmental Levels of Dibenzofurans in Sediment (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Weight basis	Location	Location description	Sample year	Ref. no.	Comments[a]
HpCDFs (continued)	4	4	11-410	178	Dry	Iggesund Sweden	Industrial	NR	13	A/papermill
	4	4	20-158	100	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	0-2cm, E
	1	1	N/A	43	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	4-6cm, E
	1	1	N/A	192	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	9-13cm, E
	5	5	560-4,300	2,358	NR	Hamburg Germany	Urban	NR	15	
Octachlorodibenzofurans (MW=444.76)										
OCDF	18	3	ND-160	14	Dry	South Central Finland	Various	88/89	1	AB
	1	1	N/A	4	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	0-0.5cm, AD
	1	1	N/A	3.2	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	5-6cm, AD
	1	1	N/A	1.1	Dry	Siskiwit Lake, Isle Royale, Lake Michigan	Pristine	NR	3	8-9cm, AD
	4	1	ND-23	5.8	NR	Central Minnesota	Rural	NR	5	A/4 sites
	25	21	ND-400,000	34,912	NR	Ontario Canada	Industrial	88	8	A/25 sites
	12	11	ND-1,010	460	NR	NY/MASS	NR	NR	10	A
	3	1	ND-39	14	NR	Stockholm Sweden	Various	NR	12	A
	2	1	ND-3.8	1.9	Dry	Baltic Sea	Reference Site	NR	13	A
	4	2	ND-15	5	Dry	Iggesund Sweden	Industrial	NR	13	A/Papermill
	4	4	150-4250	1212	NR	Dala River, Sweden	Industrial	88	17	0-1 cm
	6	6	170-1310	602	NR	Lake Vattern, Sweden	Industrial	88	17	0-1 cm
	5	5	230-79,250	19,356	NR	Lake Vanern, Sweden	Industrial	88	17	0-1 cm

Table B-6. Environmental Levels of Dibenzofurans in Sediment (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Weight basis	Location	Location description	Sample year	Ref. no.	Comments[a]
OCDF (continued)	4	4	58-151	96	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	0-2cm, E
	1	1	N/A	43	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	4-6cm, E
	1	1	N/A	192	Dry	Fjord between Sweden, Norway and Denmark	Industrial	87	14	9-13cm, E
	5	5	660-5,200	2,712	NR	Hamburg Germany	Urban	NR	15	

[a]Key A LOD (Ref. 1) = 20 to 50 ppt, LOD (Ref. 2) = 32 ppt, LOD (Ref. 3) = 0.4 ppt, LOD (Ref. 4) = 0.7 to 12.0 ppt, LOD (Ref. 5) = 0.61 to 4.1 ppt, LOD (Ref. 8) = 10 to 700 ppt, LOD (Ref. 10) = 3 ppt, LOD (ref. 12) = 1 to 20 ppt, LOD (Ref. 13) = 1 to 6.6 ppt.
B Dry surface sediments from 18 lakes.
C Industry produced 2,4,5 trichlorophenate (2,4,5T precursor).
D No anthropogenic inputs into drainage basin -- only atmospheric sources into lake.
E Mg-production facility.

Notes

NR = Not Reported
N/A = Not Applicable
ND = Not Detected
ppt = Parts per trillion

Sources:

1. Koistinen et al. (1990)
2. Bopp et al. (1991)
3. Czuczwa et al. (1984)
4. Norwood et al. (1989)
5. Reed et al. (1990)
6. Sonzongi et al. (1991)
7. Oliver and Nilmi (1988)
8. McKee et al. (1990)
9. Huckins et al. (1988)
10. Petty et al. (1982)
11. Sherman et al. (1990)
12. Rappe and Kjeller (1987)
13. Rappe et al. (1989a)
14. Oehme et al. (1989)
15. Gotz and Schumacher (1990)
16. Smith et al. (1990)
17. Kjeller et al. (1990)

Table B-7. Environmental Levels of PCBs in Sediment (ppt)

IUPAC number	Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
Tetrachloro-PCB (MW=291.99)											
77	3,3',4,4'-TeCB	18	13	N/D-550	137.7	Dry	South Central Finland	Various	88/89	1	AB
		8	8	500-360,000	64,250	Dry	Eastern Wisconsin	Industrial	88/89	6	AD
		5	5	5,000-27.5M	9.47M	NR	Waukegan, Illinois	Urban	78	9	5 sites
		NR	NR	NR	ND	Dry	Green Bay, Lake Michigan	Various	NR	16	A
81	3,4,4',5-TeCB	8	3	ND-90,000	26,880	Dry	Eastern Wisconsin	Industrial	88/89	6	AD
		NR	NR	NR	ND	Dry	Green Bay, Lake Michigan	Various	NR	16	A
Pentachloro-PCB (MW=326.44)											
126	3,3',4,4',5-PeCDD	18	1	N/D-110	6.1	Dry	South Central Finland	Various	88/89	1	AB
		8	2	ND-10,000	2,380	Dry	Eastern Wisconsin	Industrial	88/89	6	AD
		NR	NR	NR	ND	Dry	Green Bay, Lake Michigan	Various	NR	16	A
105	2,3,3',4,4'-PeCDD	10	10	52-120	96.4	Dry	South Central Finland	Various	88/89	1	ABC
		8	8	6,000-490,000	85,120	Dry	Eastern Wisconsin	Industrial	88/89	6	AD
		38	NR	NR	10,000	Dry	Lake Ontario	Various	81	7	Bottom Sediment
		5	5	102,000-131M	35.14M	NR	Waukegan, Illinois	Urban	78	9	5 Locations
		NR	NR	NR	5,800	Dry	Green Bay, Lake Michigan	Various	NR	16	A
114	2,3,4,4',5-PeCDD	8	1	ND-110,000	13,750	Dry	Eastern Wisconsin	Industrial	88/89	6	AD

Table B-7. Environmental Levels of PCBs in Sediment (ppt) (continued)

IUPAC number	Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments[a]
114	2,3,4,4',5-PeCDD (continued)	NR	NR	NR	1,000	Dry	Green Bay, Lake Michigan	Various	NR	16	A
118	2,3',4,4',5-PeCDD	8	8	86,000-1.48M	351,620	Dry	Eastern Wisconsin	Industrial	88/89	6	AD
		38	NR	NR	15,000	Dry	Lake Ontario	Various	81	7	Bottom Sediment
		NR	NR	NR	11,000	Dry	Green Bay, Lake Michigan	Various	NR	16	A
Hexachloro-PCB (MW=360.88)											
156	2,3,3',4,4',5-HxCDB	38	NR	NR	2,100	Dry	Lake Ontario	Various	81	7	Bottom Sediment
		NR	NR	NR	1,700	Dry	Green Bay, Lake Michigan	Various	NR	16	A
167	2,3',4,4',5,5'-HxCDB	8	2	ND-80,000	15,500	Dry	Eastern Wisconsin	Industrial	88/89	6	AD
		NR	NR	NR	ND	Dry	Green Bay, Lake Michigan	Various	NR	16	A
169	3,3',4,4',5,5'-HxCDB	18	0	N/A	ND	Dry	South Central Finland	Various	88/89	1	AB
		8	3	ND-19,000	4,800	Dry	Eastern Wisconsin	Industrial	88/89	6	AD
		NR	NR	NR	ND	Dry	Green Bay, Lake Michigan	Various	NR	16	A
Heptachloro-PCB (MW=396.33)											
189	2,3,3',4,4',5,5'HpCDB	NR	NR	NR	ND	Dry	Green Bay, Lake Michigan	Various	NR	16	A

[a]Key

A LOD (Ref. 1)=20 to 50 ppt, LOD (Ref. 6)=1,000 ppt, LOD (Ref. 16)=500 ppt.
B Dry Surface Sediments from 18 lakes.
C All collected samples not analyzed.
D Superfund/Michigan "Area of Concern" Site.

Table B-7. Environmental Levels of PCBs in Sediment (ppt) (continued)

Notes

NR = Not Reported
N/A = Not Applicable
ND = Not Detected
ppt = Parts per trillion

Sources:

1. Koistinen et al. (1990)
6. Sonzongi et al. (1991)
7. Oliver and Nilmi (1988)
9. Huckins et al. (1988)
16. Smith et al. (1990)

Table B-8. Environmental Levels of Dioxins in Fish (ppt)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
Tetrachlorodibenzo-p-dioxins (MW=321.98)												
2,3,7,8-TCDD	Eel	Liver	6	6	1.2-9.1	3.32	NR	various, Netherlands	NR	NR	1	one sample near dump site
	Eel	Fillet	5	5	2.4-3.3	3.04	Fat	Rhine river, Germany	NR	88	2	up & downstream Basal
	Trout	Fillet	1	1	1.4	NA	Fat	Neckar river, Germany	Urban	88	2	composite sample
	Grayling	Fillet	1	1	3.8	NA	Fat	Neckar river, Germany	Urban	88	2	
	Barbel	Fillet	1	1	5.1	NA	Fat	Neckar river, Germany	Urban	88	2	
	Carp	Fillet	1	1	2.5	NA	Fat	Neckar river, Germany	Urban	88	2	
	Chub	Fillet	1	0	ND(2.3)	NA	Fat	Neckar river, Germany	Urban	88	2	composite sample
	Eel	Fillet	5	5	0.9-1.5	1.3	Fat	Neckar river, Germany	Urban	87-88	2	
	Bream		14	14	1.4-94.4	18.0	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		3	3	1.8-8.1	5.9	Fresh	Hamburg, Germany	Urban	84	4	
	Herring	Whole	1	0	ND(0.1)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 5-10 fish
	Herring	Whole	2	0	ND(0.1)	NA	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 5-10 fish
	Herring	Whole	2	0	ND(0.1)	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 5-10 fish
	Salmon	Muscle	2	2	4.6-19.0	11.8	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	wild salmon
	Salmon	Muscle	2	2	0.2-0.3	0.25	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	hatched salmon
	Perch	NR	3	3	2.6-19	11.5	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	caught near pulp mill
	Artic Char	NR	5	5	6.5-25	14.3	Fresh	Lake Vattern, Sweden	NR	NR	5	
	Pike	Muscle	8	8	40-833	186	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	78	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Ch. Catfish	Fillets	8	8	28-695	157	NR	various, Michigan	NR	78	7	
	Carp	Fillets	14	10	20-153	55	NR	various, Michigan	NR	78	7	

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,7,8-TCDD (continued)	Y. Perch	Fillets	6	3	10-20	13	NR	various, Michigan	NR	78	7	
	Sm. M. Basss	Fillets	2	2	7-8	8	NR	various, Michigan	NR	78	7	
	Sucker	Fillets	4	3	4-21	10	NR	various, Michigan	NR	78	7	
	Lake Trout	Fillets	2	0	ND(5.0)	NA	NR	various, Michigan	NR	78	7	
	Carp	Whole	3	0	ND(6.6)	NA	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	0	ND(6.6)	NA	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	2	2	3-28	16	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Blue Crab	Meat	2	2	105.7-116.1	110.9	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	4.7-6.3	5.5	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	83.9-733.9	408.9	Wet	Newark Bay, NJ	Urban	NR	10	
	Lake Trout	Whole	1	1	1.0	NA	Wet	Lake Superior	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	8.6	NA	Wet	Lake Huron	NR	84	11	mean 5 samples
	Lake Trout	Whole	3	3	3.5-5.8	4.4	Wet	Lake Michigan	NR	84	11	range 3 sample sites
	Walleye	Whole	1	1	1.8	NA	Wet	Lake Erie	NR	84	11	mean 5 samples
	Walleye	Whole	1	1	6.6	NA	Wet	Lake St. Clair	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	48.9	NA	Wet	Lake Ontario	NR	84	11	mean 5 samples
	Lake Trout	Whole	10	10	3.0-8.7	4.2	Wet	Lake Michigan	NR	84	11	
	Br. Trout	Whole	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	1	5	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	14	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	5	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	18	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,7,8-TCDD (continued)	Lake Trout	Fillets	1	1	8	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	1	20	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Cod	Fillets	4	0	ND(1.0)	NA	Fresh	various, Sweden	Industrial	88	17	
	Haddock	Fillets	1	0	ND(0.2)	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	P. Flounder	Fillets	1	0	ND(0.2)	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Plaice	Fillets	1	0	ND(0.5)	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Flounder	Fillets	1	0	ND(0.5)	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Eel	Fillets	4	1	ND-1.4	0.35	Fresh	various, Sweden	Industrial	87-88	17	
	Mussel	Muscle	3	0	ND(0.5)	NA	Fresh	Grenlandsfjord,Sweden	Industrial	87	17	
	Shrimp	Muscle	2	0	ND(2.0)	NA	Fresh	Grenlandsfjord,Sweden	Industrial	88	17	
	Cod	Fillets	6	NR	ND-3.8	NR	Fresh	Frierfjord, Sweden	Industrial	87	17	only conc. range given
	Various[c]	Mixed[d]	314	220	NR	6.84	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
	Various[c]	Mixed[d]	34	34	0.06-2.26	0.56	Wet	Various, US	Background[e]	86-89	20	samples composite 3-5 fish
TCDDs	Bream		13	13	2.5-102.4	17.6	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	9.0-10.5	9.8	Fresh	Hamburg, Germany	Urban	84	4	
	Carp	Whole	3	1	ND-3.9	1.3	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	0	ND(6.6)	NA	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	2	117.6-149.5	133.6	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	6.6-8.3	7.4	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	85.4-733.9	409.6	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	1	11	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
TCDDs (continued)	Br. Trout	Fillets	1	1	9	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	29	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	11	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	32	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	1	12	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	1	22	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	1	9	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
Pentachlorodibenzo-p-dioxins (MW=356.42)												
1,2,3,7,8-PeCDD	Carp	Whole	2	2	2-11	6	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Pike	Muscle	8	8	70-250	129	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	39	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	1	0.6	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 5-10 fish
	Herring	Whole	2	2	1.1-2.8	1.95	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 5-10 fish
	Herring	Whole	2	2	2.0-4.7	3.35	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 5-10 fish
	Various[c]	Mixed[d]	314	170	NR	2.38	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
	Various[c]	Mixed[d]	34	34	0.15-2.67	0.77	Wet	Various, US	Background[e]	86-89	20	samples composite 3-5 fish
PeCDDs	Bream		13	13	3.2-27.8	12.1	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	9.8-29.8	19.8	Fresh	Hamburg, Germany	Urban	84	4	
	Carp	Whole	3	3	3.5-4.5	3.9	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	0	ND(6.6)	NA	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	2	17.2-18.0	17.6	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	10.0-11.0	10.5	Wet	New York Bight	Dump Site	NR	10	former sewage sludge

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
PeCDDs (continued)	Str. Bass	Fillets	2	2	5.2-10.6	7.9	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	1	8	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	31	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	15	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	39	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	1	9	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	1	5	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
Hexachlorodibenzo-p-dioxins (MW=390.87)												
1,2,3,4,7,8-HxCDD	Carp	Whole	2	2	3-5	4	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Pike	Muscle	8	8	6.7-22	12.8	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	11	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	0	ND(0.2)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 5-10 fish
	Herring	Whole	2	2	0.2-0.3	0.25	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 5-10 fish
	Herring	Whole	2	0	ND(0.2)	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 5-10 fish
	Various[c]	Mixed[d]	314	100	NR	1.67	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
1,2,3,6,7,8-HxCDD	Pike	Muscle	8	8	30-100	50.5	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	22	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	0	ND(0.2)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 5-10 fish
	Herring	Whole	2	1	ND-2.4	1.25	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 5-10 fish

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
1,2,3,6,7,8-HxCDD	Herring	Whole	2	2	2.2-8.1	5.15	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 5-10 fish
(continued)	Various[c]	Mixed[d]	314	217	NR	4.29	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
1,2,3,7,8,9-HxCDD	Pike	Muscle	8	0	ND(3-11)	NA	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	0	ND(6)	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	0	ND(0.2)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 5-10 fish
	Herring	Whole	2	0	ND(0.2)	NA	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 5-10 fish
	Herring	Whole	2	0	ND(0.2)	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 5-10 fish
	Various[c]	Mixed[d]	314	119	NR	1.15	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
HxCDDs	Bream		13	13	4.3-46.4	17.8	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	18.6-21.5	20.0	Fresh	Hamburg, Germany	Urban	84	4	
	Carp	Whole	3	3	2.3-11	6.9	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	1	ND-3.0	1.0	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	2	0.3-1.5	0.9	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	3.0-3.4	3.2	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	0.6-0.7	0.65	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	1	20	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	1	25	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	67	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	37	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	114	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	1	27	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	1	16	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
HxCDDs (continued)	Coho Salmon	Fillets	1	1	22	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Various[c]	Mixed[d]	34	NR	ND-3.57	0.39	Wet	Various, US	Background[e]	86-89	20	samples composite 3-5 fish
Heptachlorodibenzo-p-dioxins(MW=425.31)												
1,2,3,4,6,7,8-HpCDD	Carp	Whole	2	2	3-4	3.5	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Herring	Whole	1	0	ND(0.2)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 5-10 fish
	Herring	Whole	2	1	ND-0.6	0.35	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 5-10 fish
	Herring	Whole	2	0	ND(0.2)	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 5-10 fish
	Various[c]	Mixed[d]	314	279	NR	10.5	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
HpCDDs	Bream		13	13	1.5-14.4	6.7	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	3.5-7.1	5.3	Fresh	Hamburg, Germany	Urban	84	4	
	Carp	Whole	3	3	15-22	19.3	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	2	ND-11	7.0	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	0	ND(1.1)	NA	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	1	ND-8.5	4.25	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	4.0-11.4	7.7	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	1	7	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	1	9	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	12	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	16	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	1	30	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
HpCDDs (continued)	Coho Salmon	Fillets	1	1	50	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
Octachlorodibenzo-p-dioxin (MW=460.76)												
1,2,3,4,6,7,8,9-OCDD	Eel	Fillet	5	5	28-60	44.4	Fat	Rhine river, Germany	NR	88	2	up & downstream Basal
	Trout	Fillet	1	0	ND(5.0)	NA	Fat	Neckar river, Germany	Urban	88	2	composite sample
	Grayling	Fillet	1	1	47	NA	Fat	Neckar river, Germany	Urban	88	2	
	Barbel	Fillet	1	1	9.0	NA	Fat	Neckar river, Germany	Urban	88	2	
	Carp	Fillet	1	1	23	NA	Fat	Neckar river, Germany	Urban	88	2	
	Chub	Fillet	1	1	15	NA	Fat	Neckar river, Germany	Urban	88	2	composite sample
	Eel	Fillet	5	5	25-40	30	Fat	Neckar river, Germany	Urban	87-88	2	
	Bream		14	14	1.4-5.1	2.5	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		3	3	2.3-10.5	5.2	Fresh	Hamburg, Germany	Urban	84	4	
	Herring	Whole	1	1	1.1	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 5-10 fish
	Herring	Whole	2	1	ND-0.7	0.4	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 5-10 fish
	Herring	Whole	2	1	ND-0.3	0.2	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 5-10 fish
	Salmon	Muscle	2	1	ND-1.5	0.75	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	wild salmon
	Salmon	Muscle	2	2	0.8-1.9	1.35	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	hatched salmon
	Perch	NR	3	3	0.6-0.8	0.73	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	caught near pulp mill
	Pike	Muscle	8	8	10-17	14.75	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	22	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Carp	Whole	3	3	56-62	59	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	3	35-43	39	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	2	2	3-5	4	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
1,2,3,4,6,7,8,9-OCDD	Blue Crab	Meat	2	2	34.3-78.8	56.6	Wet	Passaic river, NJ	Urban	NR	10	
(continued)	Lobster	Meat	2	2	6.3-10.9	8.6	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	5.1-49.5	27.3	Wet	Newark Bay, NJ	Urban	NR	10	
	Lake Trout	Whole	1	1	1.0	NA	Wet	Lake Superior	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	0.7	NA	Wet	Lake Huron	NR	84	11	mean 5 samples
	Lake Trout	Whole	3	3	1.1-2.5	1.8	Wet	Lake Michigan	NR	84	11	range 3 sample sites
	Walleye	Whole	1	1	2.8	NA	Wet	Lake Erie	NR	84	11	mean 5 samples
	Walleye	Whole	1	1	1.8	NA	Wet	Lake St. Clair	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	1.2	NA	Wet	Lake Ontario	NR	84	11	mean 5 samples
	Lake Trout	Whole	10	10	0.8-3.7	1.6	Wet	Lake Michigan	NR	84	11	
	Br. Trout	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	1	11	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	89	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	1	28	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	1	160	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	1	280	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Cod	Fillets	4	3	ND-11	4.95	Fresh	various, Sweden	Industrial	88	17	
	Haddock	Fillets	1	0	ND(3.6)	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	P. Flounder	Fillets	1	1	3.4	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Plaice	Fillets	1	1	424	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples

Table B-8. Environmental Levels of Dioxins in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
1,2,3,4,6,7,8,9-OCDD	Flounder	Fillets	1	1	2.4	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
(continued)	Eel	Fillets	4	3	ND-770	204.2	Fresh	various, Sweden	Industrial	87-88	17	
	Mussel	Muscle	3	3	12-140	62.0	Fresh	Grenlandsfjord,Sweden	Industrial	87	17	
	Shrimp	Muscle	2	1	ND-18	9.0	Fresh	Grenlandsfjord,Sweden	Industrial	88	17	
	Cod	Fillets	6	NR	0.63-2.2	NR	Fresh	Frierfjord, Sweden	Industrial	87	17	only conc. range given

Footnote References

[a] Ch. = Channel; Y. = Yellow; Sm. M. = Small Mouth; Str. = Striped; Br. = Brown; Rb. = Rainbow; P. = Pole.
[b] Various, Netherlands = samples taken from six locations around Ijsselmeer Lake; Various, Michigan = samples taken from Tittabawassee River, Grand River, Saginaw River, Saginaw Bay, and Lake Michigan; Various, Sweden = samples taken from Grenlandsfjord and Frierfjord, US = samples taken from 314 sites across the US, including industrial and background sites.
[c] Species were taken from both bottom feeders and open water feeders, and then composited.
[d] Whole fish samples and fillet samples were combined during analysis.
[e] The 34 background sites are a subset of the 314 US sites.

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = not applicable;
ND = non-detected (limit of detection);
NR = not reported.
Descriptions provided were those given by reference or surmised from study description when not given.

Sources:
1. Van den Berg, et al. (1987)
2. Frommberger (1991)
4. Gotz, et al. (1990)
5. Rappe, et al. (1989)
7. Harless and Lewis (1982)
8. Reed, et al. (1990)
10. Rappe, et al. (1991)
11. Vault, et al. (1989)
12. Niimi and Oliver (1989a)
17. Oehme, et al. (1989)
18. Stalling, et al. (1983)
19. Kjeller, et al. (1990)
20. USEPA (1992)

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
Tetrachlorodibenzofurans (MW=305.98)												
2,3,7,8-TCDF	Eel	Liver	6	0	ND	NA	NR	various, Netherlands	NR	NR	1	one sample near dump site
	Eel	Fillets	5	5	2.1-12	6.98	Fat	Rhine river, Germany	NR	88	2	up & downstream Basal
	Br. Trout	Fillets	1	1	45	NA	Fat	Neckar river, Germany	Urban	88	2	Composite sample
	Grayling	Fillets	1	1	108	NA	Fat	Neckar river, Germany	Urban	88	2	
	Barbel	Fillets	1	1	57	NA	Fat	Neckar river, Germany	Urban	88	2	
	Carp	Fillets	1	1	58	NA	Fat	Neckar river, Germany	Urban	88	2	
	Chub	Fillets	1	1	128	NA	Fat	Neckar river, Germany	Urban	88	2	Composite sample
	Eel	Fillets	5	5	0.9-2.0	1.35	Fat	Neckar river, Germany	Urban	87-88	2	
	Herring	Adipose	2	2	3-4	3.5	Fat	South Baltic Sea	NR	NR	3	
	Herring	Whole	1	1	1.7	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	2	5.3-5.5	5.4	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	2	3.0-6.2	4.6	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Salmon	Muscle	2	2	28-35	31.5	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	wild salmon
	Salmon	Muscle	2	2	7.8-9.0	8.4	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	hatched salmon
	Perch	NR	3	3	2.1-8.7	5.4	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	caught near pulp mill
	Artic Char	NR	5	5	21-75	55	Fresh	Lake Vattern, Sweden	NR	NR	5	
	Pike	Muscle	8	8	330-3000	774	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	430	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Carp	Whole	2	2	11	NA	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Carp	Whole	3	3	1.8-3.0	2.6	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	3	1.0-1.3	1.1	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,7,8-TCDF	Carp	Fillets	1	1	49	NA	NR	NR	NR	78	9	contamimated site
(continued)	Catfish	Fillets	1	1	6	NA	NR	Saginaw river	NR	84	9	contaminated site
	Smk. Chub	Fillets	1	1	3	NA	NR	NR	NR	85	9	contaminated site
	Str. Bass	Fillets	5	5	7-93	28.0	NR	Hudson river	NR	85	9	contaminated site
	Lg. M. Bass	Fillets	1	1	10	NA	NR	Hudson river	NR	85	9	contaminated site
	Lake Trout	Fillets	3	3	11-56	31.7	NR	Lake Superior	NR	85	9	contaminted site
	Blue Crab	Meat	2	2	11.0-15.5	13.25	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	3.5-4.1	3.8	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	51.9-85.5	68.7	Wet	Newark Bay, NJ	Urban	NR	10	
	Lake Trout	Whole	1	1	14.8	NA	Wet	Lake Superior	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	22.8	NA	Wet	Lake Huron	NR	84	11	mean 5 samples
	Lake Trout	Whole	3	3	34.8-42.3	39.5	Wet	Lake Michigan	NR	84	11	range 3 sample sites
	Walleye	Whole	1	1	11.3	NA	Wet	Lake Erie	NR	84	11	mean 5 samples
	Walleye	Whole	1	1	24.8	NA	Wet	Lake St. Clair	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	18.5	NA	Wet	Lake Ontario	NR	84	11	mean 5 samples
	Lake Trout	Whole	10	10	27.0-56.0	38.4	Wet	Lake Michigan	NR	84	11	
	Br. Trout	Whole	1	1	11	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	1	8	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	19	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	15	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,7,8-TCDF (continued)	Coho Salmon	Whole	1	1	20	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Cod	Fillets	4	4	0.2-1.4	0.62	Fresh	various, Sweden	Industrial	88	17	
	Haddock	Fillets	1	1	0.75	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	P. Flounder	Fillets	1	1	0.28	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Plaice	Fillets	1	1	1.4	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Flounder	Fillets	1	1	1.2	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Eel	Fillets	4	3	ND-68	17.1	Fresh	various, Sweden	Industrial	88	17	
	Mussel	Muscle	3	3	16.1-61	32.4	Fresh	Grenlandsfjord,Sweden	Industrial	87	17	
	Shrimp	Muscle	2	2	6.1-37	21.6	Fresh	Grenlandsfjord,Sweden	Industrial	88	17	
	Cod	Fillets	6	NR	0.49-14.3	NR	Fresh	Frierfjord, Sweden	Industrial	87	17	only conc. range given
	Various[c]	Mixed[d]	314	279	NR	13.6	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
	Various[c]	Mixed[d]	34	34	0.1-13.73	1.61	Wet	Various, US	Background[e]	86-89	20	samples composite 3-5 fish
TCDFs	Bream		13	13	7.8-86.5	44.3	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	10.7-41.1	25.9	Fresh	Hamburg, Germany	Urban	84	4	
	Y. Perch	Whole	1	1	1060	NA	NR	Housatonic river, MA	NR	NR	6	
	Carp	Whole	3	3	2.4-4.0	3.1	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	3	1.0-1.3	1.2	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	2	132.9-164.5	148.7	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	23.0-31.2	27.1	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	77.2-107.7	92.4	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	1	11	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
TCDFs (continued	Br. Trout	Fillets	1	1	8	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	19	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	18	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	1	20	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	1	6	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
TCDFs other than 2,3,7,8-TCDF	Eel	Fillets	5	5	4.6-13	8.8	Fat	Rhine river, Germany	NR	88	2	up & downstream Basal
	Grayling	Fillets	1	1	142	NA	Fat	Neckar river, Germany	Urban	88	2	
	Barbel	Fillets	1	1	77	NA	Fat	Neckar river, Germany	Urban	88	2	
	Carp	Fillets	1	1	14	NA	Fat	Neckar river, Germany	Urban	88	2	
	Chub	Fillets	5	5	17	NA	Fat	Neckar river, Germany	Urban	88	2	Composite sample
	Eel	Fillets	2	2	1.5	NA	Fat	Neckar river, Germany	Urban	88	2	composite sample
Pentachlorodibenzofurans (MW=340.42)												
1,2,3,7,8-PeCDF	Carp	Whole	2	2	1-5	3	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Pike	Muscle	8	8	43-140	73.2	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	39	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	1	0.4	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	2	1.4-2.5	1.95	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.8-0.9	0.85	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Various[c]	Mixed[d]	314	151	NR	1.71	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
	Various[c]	Mixed[d]	34	34	0.1-1.90	0.43	Wet	Various, US	Background[e]	86-89	20	samples composite 3-5 fish

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,4,7,8-PeCDF	Carp	Whole	2	2	4-11	7.5	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Pike	Muscle	8	8	120-290	189	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	110	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	1	3.0	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	2	6.8-19.0	12.9	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	2	8.8-8.9	8.85	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Various[c]	Mixed[d]	314	201	NR	3.06	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
	Various[c]	Mixed[d]	34	34	0.1-1.39	0.5	Wet	Various, US	Background[e]	86-89	20	samples composite 3-5 fish
PeCDFs	Bream		13	13	13.9-114	65.4	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	62.3-153.3	107.8	Fresh	Hamburg, Germany	Urban	84	4	
	Y. Perch	Whole	1	1	640	NA	NR	Housatonic river, MA	NR	NR	6	
	Carp	Whole	3	3	15-45	26.3	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	2	ND-13	9.0	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	2	89.7-94.1	91.9	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	29.8-37.3	33.6	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	34.3-82.6	58.4	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	1	3	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	8	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	7	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	39	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	1	8	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
PeCDFs (continued)	Coho Salmon	Whole	1	1	13	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
PeCDFs other than 1,2,3,7,8-PeCDF and 2,3,4,7,8-PeCDF	Grayling	Fillets	1	1	22	NA	Fat	Neckar river, Germany	Urban	88	2	
	Barbel	Fillets	1	1	21	NA	Fat	Neckar river, Germany	Urban	88	2	
	Carp	Fillets	1	1	17	NA	Fat	Neckar river, Germany	Urban	88	2	
Hexachlorodibenzofurans (MW=374.87)												
1,2,3,4,7,8-HxCDF	Carp	Whole	2	2	2-5	3.5	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Pike	Muscle	8	8	10-33	14.4	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	11	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	1	0.2	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.4-0.7	0.55	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.3	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Various[c]	Mixed[d]	314	132	NR	2.35	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
1,2,3,6,7,8-HxCDF	Pike	Muscle	8	8	5.6-22	11.9	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	1	5.6	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	1	0.1	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.4-0.8	0.6	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.2	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Various[c]	Mixed[d]	314	66	NR	1.74	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
1,2,3,7,8,9-HxCDF	Pike	Muscle	8	0	ND(3-6)	NA	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	0	ND(6)	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Various[c]	Mixed[d]	314	3	NR	1.22	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,4,6,7,8-HxCDF	Pike	Muscle	8	7	ND(3)-17	7.96	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	0	ND(6)	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	0	ND(0.2)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.4-0.8	0.6	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.3	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Various[c]	Mixed[d]	314	100	NR	1.24	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
HxCDFs	Bream		13	13	5.7-47.4	25.2	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	21.9-53.8	37.8	Fresh	Hamburg, Germany	Urban	84	4	
	Y. Perch	Whole	1	1	440	NA	NR	Housatonic river, MA	NR	NR	6	
	Carp	Whole	3	2	ND-24	10.0	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	3	2.7-5.1	3.5	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	2	9.3-9.4	9.35	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	2	7.7-7.9	7.8	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	2.0-4.4	3.2	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	1	2	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	8	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	1	2	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	16	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
HxCDFs (continued)	Various[c]	Mixed[d]	34	NR	ND-2.59	0.22	Wet	Various, US	Background[e]	86-89	20	samples composite 3-5 fish
non-2,3,7,8-HxCDFs	Barbel	Fillets	1	1	2.1	NA	Fat	Neckar river, Germany	Urban	88	2	
Heptachlorodibenzofurans (MW=409.31)												
1,2,3,4,6,7,8-HpCDF	Carp	Whole	2	2	3-4	3.5	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Pike	Muscle	16	1	ND(3-7)-17	3.41	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	2	0	ND(6-11)	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Herring	Whole	1	0	ND(0.2)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	2	0.8-1.2	1.0	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	1	ND-0.9	0.5	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Various[c]	Mixed[d]	314	170	NR	1.91	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
1,2,3,4,7,8,9-HpCDF	Pike	Muscle	8	0	ND(3-11)	NA	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	0	ND(6)	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Various[c]	Mixed[d]	314	13	NR	1.24	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
HpCDFs	Bream		13	13	1.8-6.0	3.6	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		2	2	5.0-10.1	7.6	Fresh	Hamburg, Germany	Urban	84	4	
	Y. Perch	Whole	1	0	ND(5.0)	NA	NR	Housatonic river, MA	NR	NR	6	
	Carp	Whole	3	1	ND-14	4.7	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	0	ND(6.6)	NA	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station
	Blue Crab	Meat	2	2	2.8-3.5	3.15	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	0	ND(0.9)	NA	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	2	1.3-2.4	1.8	Wet	Newark Bay, NJ	Urban	NR	10	
	Br. Trout	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
HpCDFs (continued)	Br. Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	1	1	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	1	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
Octachlorodibenzofurans (MW=444.76)												
1,2,3,4,6,7,8,9-OCDF	Bream		14	10	ND-3.1	1.2	Fresh	Hamburg, Germany	Urban	84	4	
	Perch		3	3	1.1-8.3	3.7	Fresh	Hamburg, Germany	Urban	84	4	
	Herring	Whole	1	0	ND(0.2)	NA	Fresh	Atlantic Coast, Sweden	Pristine	NR	5	composite 2-5 fish
	Herring	Whole	2	1	ND-0.3	0.2	Fresh	Baltic Sea, Sweden	Urban	NR	5	composite 2-5 fish
	Herring	Whole	2	0	ND(0.2)	NA	Fresh	Gulf of Bothnia, Sweden	Industrial	NR	5	composite 2-5 fish
	Salmon	Muscle	2	0	ND(2.0)	NA	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	wild salmon
	Salmon	Muscle	2	0	ND(0.5)	NA	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	hatched salmon
	Perch	NR	3	1	ND-1.7	0.57	Fresh	Gulf of Bothnia, Sweden	NR	NR	5	caught near pulp mill
	Pike	Muscle	8	0	ND(3-11)	NA	Fat	Lake Vanern, Sweden	Industrial	88	19	samples composite 2-5 fish
	Pike	Muscle	1	0	ND(11)	NA	Fat	Hedesunda Bay, Sweden	Industrial	88	19	composite 5 fish
	Carp	Whole	2	2	4-8	6	NR	Lake Huron	NR	NR	18	samples composite 3-5 fish
	Y. Perch	Whole	1	0	ND(5.0)	NA	NR	Housatonic river, MA	NR	NR	6	
	Carp	Whole	3	0	ND(6.6)	NA	NR	Mississippi river, MN	Industrial	NR	8	Elk river power station
	Carp	Whole	3	3	ND(6.6)	NA	NR	Lake Orono, MN	Industrial	NR	8	Elk river power station

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
1,2,3,4,6,7,8,9-OCDF (continued)	Blue Crab	Meat	2	0	ND(8.3)	NA	Wet	Passaic river, NJ	Urban	NR	10	
	Lobster	Meat	2	0	ND(8.4)	NA	Wet	New York Bight	Dump Site	NR	10	former sewage sludge
	Str. Bass	Fillets	2	0	ND(3.1)	NA	Wet	Newark Bay, NJ	Urban	NR	10	
	Lake Trout	Whole	1	1	0.4	NA	Wet	Lake Superior	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	0.1	NA	Wet	Lake Huron	NR	84	11	mean 5 samples
	Lake Trout	Whole	3	3	0.3-1.0	0.85	Wet	Lake Michigan	NR	84	11	range 3 sample sites
	Walleye	Whole	1	1	0.9	NA	Wet	Lake Erie	NR	84	11	mean 5 samples
	Walleye	Whole	1	1	0.4	NA	Wet	Lake St. Clair	NR	84	11	mean 5 samples
	Lake Trout	Whole	1	1	0.4	NA	Wet	Lake Ontario	NR	84	11	mean 5 samples
	Lake Trout	Whole	10	10	0.1-3.3	0.65	Wet	Lake Michigan	NR	84	11	
	Br. Trout	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Br. Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Rb. Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Whole	1	1	2	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Lake Trout	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Whole	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Coho Salmon	Fillets	1	0	ND(7)	NA	NR	Lake Ontario	NR	NR	12	composite 3 samples
	Cod	Fillets	4	4	3.4-9.6	6.3	Fresh	various, Sweden	Industrial	88	17	
	Haddock	Fillets	1	1	4.3	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	P. Flounder	Fillets	1	1	4.7	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
	Plaice	Fillets	1	1	41	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples

Table B-9. Environmental Levels of Dibenzofurans in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
1,2,3,4,6,7,8,9-OCDF	Flounder	Fillets	1	1	5.6	NA	Fresh	various, Sweden	Industrial	88	17	composite 10 samples
(continued)	Eel	Fillets	4	4	31-581	205.2	Fresh	various, Sweden	Industrial	88	17	
	Mussel	Muscle	3	3	13.4-933	339.1	Fresh	Grenlandsfjord,Sweden	Industrial	87	17	
	Shrimp	Muscle	2	2	2.3-41	21.6	Fresh	Grenlandsfjord,Sweden	Industrial	88	17	
	Cod	Fillets	6	NR	ND-21	NR	Fresh	Frierfjord, Sweden	Industrial	87	17	only conc. range given

Footnote References

[a] Br. = Brown; Smk. = Smoked; Str. = Striped; Lg. M. = Large Mouth; Rb. = Rainbow; P. = Pole; Y. = Yellow.

[b] Various, Netherlands = samples taken from six locations around IJsselmeer Lake; various, Sweden = samples taken from Grenlandsfjord and Frierfjord, US = samples taken from 314 sites across the US, including industrial and background sites.

[c] Species were taken from both bottom feeders and open water feeders, and then composited.

[d] Whole fish samples and fillet samples were combined during analysis.

[e] The 34 background sites are a subset of the 314 US sites.

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = not applicable
ND = non-detected (limit of detection)
NR = not reported
Descriptions provided were those given by reference or surmised from study description when not given.

Sources:
1. Van den Berg, et al. (1987)
2. Frommberger (1991)
3. Rappe, et al. (1984)
4. Gotz, et al. (1990)
5. Rappe, et al. (1989)
8. Reed, et al. (1990)
9. Gardner and White (1990)
10. Rappe, et al. (1991)
11. Vault, et al. (1989)
12. Niimi and Oliver (1989a)
17. Oehme, et al. (1989)
18. Stalling, et al. (1983)
19. Kjeller, et al. (1990)
20. USEPA (1992)

Table B-10. Environmental Levels of PCBs in Fish (ppt)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
Tetrachloro-PCB (MW=291.99)												
3,3',4,4'-TeCB	Blk. Bullhead	Whole	1	1	89,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 6 samples
	Lg. M. Bass	Whole	1	1	86,000	NA	NR	Waukegan Harbor, IL	NR	78	13	
	Blk. Crappie	Whole	1	1	43,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 3 samples
	Wh. Sucker	Whole	1	1	50,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 6 samples
	Coho Salmon	Whole	1	1	2,000	NA	NR	Waukegan Harbor, IL	NR	78	13	
	Wh. Crappie	Whole	1	1	24,000	NA	NR	Waukegan Harbor, IL	NR	78	13	
	Y. Perch	Whole	1	1	23,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 5 samples
	Br. Trout	Whole	1	1	5,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Br. Trout	Fillets	1	1	2,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Whole	1	1	18,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Fillets	1	1	8,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Rb. Trout	Whole	2	2	6000-11,000	8,500	NR	Lake Ontario	NR	NR	15	
	Rb. Trout	Fillets	2	1	ND-4,000	2,000	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Whole	2	2	8000-10,000	9,000	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Fillets	2	2	3,000-5,000	4,000	NR	Lake Ontario	NR	NR	15	
	Carp	Whole	1	0	ND(5,000)	NA	NR	Saginaw Bay	NR	NR	16	
3,4,4',5-TeCB	Br. Trout	Whole	1	1	24,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Br. Trout	Fillets	1	1	10,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Whole	1	1	90,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Fillets	1	1	38,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Rb. Trout	Whole	2	2	13,000-30,000	21,500	NR	Lake Ontario	NR	NR	15	

Table B-10. Environmental Levels of PCBs in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
3,4,4',5-TeCB (continued)	Rb. Trout	Fillets	2	1	ND-9,000	4,500	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Whole	2	2	2,000-26,000	14,000	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Fillets	2	2	8,000-14,000	11,000	NR	Lake Ontario	NR	NR	15	
	Carp	Whole	1	1	17,000	NA	NR	Saginaw Bay	NR	NR	16	
TeCBs	Various[c]	Mixed[d]	362	263	NR	696,240	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
Pentachloro-PCB (MW=326.44)												
3,3',4,4',5-PeCB	Carp	Whole	1	0	ND(5,000)	NA	NR	Saginaw Bay	NR	NR	16	
2,3,3',4,4'-PeCB	Blk. Bullhead	Whole	1	1	352,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 6 samples
	Lg. M. Bass	Whole	1	1	290,000	NA	NR	Waukegan Harbor, IL	NR	78	13	
	Blk. Crappie	Whole	1	1	114,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 3 samples
	Wh. Sucker	Whole	1	1	483,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 6 samples
	Coho Salmon	Whole	1	1	45,000	NA	NR	Waukegan Harbor, IL	NR	78	13	
	Wh. Crappie	Whole	1	1	242,000	NA	NR	Waukegan Harbor, IL	NR	78	13	
	Y. Perch	Whole	1	1	80,000	NA	NR	Waukegan Harbor, IL	NR	78	13	composite 5 samples
	Small Smelt	Whole	1	1	15,000	NA	Wet	Port Credit, Lake Ontario	NR	86	14	composite 48 samples
	Large Smelt	Whole	1	1	38,000	NA	Wet	Vineland, Lake Ontario	NR	82	14	composite 20 samples
	Salmonids	Whole	1	1	110,000	NA	Wet	Lake Ontario	NR	81-82	14	composite 60 samples
	Br. Trout	Whole	1	1	55,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Br. Trout	Fillets	1	1	24,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Whole	1	1	253,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Fillets	1	1	101,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Rb. Trout	Whole	2	2	34,000-138,000	86,000	NR	Lake Ontario	NR	NR	15	

Table B-10. Environmental Levels of PCBs in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,4',4,4'-PeCB (continued)	Rb. Trout	Fillets	2	2	6,000-50,000	28,000	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Whole	2	2	48,000-121,000	84,500	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Fillets	2	2	19,000-56,000	37,500	NR	Lake Ontario	NR	NR	15	
	Carp	Whole	1	1	427,000	NA	NR	Saginaw Bay	NR	NR	16	
2,3,4,4',5-PeCB	Carp	Whole	1	1	57,000	NA	NR	Saginaw Bay	NR	NR	16	
2,3',4,4',5-PeCB	Small Smelt	Whole	1	1	37,000	NA	Wet	Port Credit, Lake Ontario	NR	86	14	composite 48 samples
	Large Smelt	Whole	1	1	87,000	NA	Wet	Vineland, Lake Ontario	NR	82	14	composite 20 samples
	Salmonids	Whole	1	1	250,000	NA	Wet	Lake Ontario	NR	81-82	14	composite 60 samples
	Br. Trout	Whole	1	1	133,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Br. Trout	Fillets	1	1	60,000	NA	NR	Vineland, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Whole	1	1	634,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Lake Trout	Fillets	1	1	242,000	NA	NR	Port Credit, Lake Ontario	NR	NR	15	mean 10 samples
	Rb. Trout	Whole	2	2	80,000-310,000	195,000	NR	Lake Ontario	NR	NR	15	
	Rb. Trout	Fillets	2	2	16,000-115,000	65,500	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Whole	2	2	100,000-271,000	185,500	NR	Lake Ontario	NR	NR	15	
	Coho Salmon	Fillets	2	2	39,000-136,000	87,500	NR	Lake Ontario	NR	NR	15	
	Carp	Whole	1	1	1.35×10^6	NA	NR	Saginaw Bay	NR	NR	16	
PeCBs	Various[c]	Mixed[d]	362	314	NR	564,700	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
Hexachloro-PCB (MW=360.88)												
3,3',4,4',5,5'-HxCB	Carp	Whole	1	0	ND(5,000)	NA	NR	Saginaw Bay	NR	NR	16	
2,3,3',4,4',5-HxCB	Small Smelt	Whole	1	1	2,700	NA	Wet	Port Credit, Lake Ontario	NR	86	14	composite 48 samples
	Large Smelt	Whole	1	1	6,100	NA	Wet	Vineland, Lake Ontario	NR	82	14	composite 20 samples

Table B-10. Environmental Levels of PCBs in Fish (ppt) (continued)

Chemical	Fish species[a]	Tissue	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location[b]	Location description	Samp. year	Ref. no.	Comments
2,3,3',4,4',5-HxCB	Salmonids	Whole	1	1	34,000	NA	Wet	Lake Ontario	NR	81-82	14	composite 60 samples
(continued)	Carp	Whole	1	1	79,000	NA	NR	Saginaw Bay	NR	NR	16	
2,3,3',4,4',5'-HxCB	Carp	Whole	1	1	76,000	NA	NR	Saginaw Bay	NR	NR	16	
2,3',4,4',5,5'-HxCB	Carp	Whole	1	1	77,000	NA	NR	Saginaw Bay	NR	NR	16	
HxCBs	Various[c]	Mixed[d]	362	321	NR	355,930	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish
Heptachloro-PCB (MW=396.33)												
2,3,3',4,4',5,5'-HpCB	Carp	Whole	1	1	29,000	NA	NR	Saginaw Bay	NR	NR	16	
HpCBs	Various[c]	Mixed[d]	362	250	NR	96,700	Wet	Various, US	Various	86-89	20	samples composite 3-5 fish

Footnote References

[a] Blk. = Black; Lg. M. = Large Mouth; Wh. = White; Y. = Yellow; Br. = Brown; Rb. = Rainbow

[b] US = samples taken from 362 sites across the US, including industrial and background sites.

[c] Species were taken from both bottom feeders and open water feeders, and then composited.

[d] Whole fish samples and fillet samples were combined for analysis.

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = not applicable
ND = non-detected (limit of detection)
NR = not reported
Descriptions provided were those given by reference or surmised from study description when not given.

Sources:
13. Huckins, et al. (1988)
14. Oliver and Niimi (1988)
15. Niimi and Oliver (1989b)
16. Smith, et al. (1990)
20. USEPA (1991)

Table B-11. Levels of Dioxins in Food Products (ppt)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments
Tetrachlorodibenzo-p-dioxins (MW=321.98)										
2,3,7,8-TCDD	3	0	ND	--	Fresh	Sweden		NR	1	Food basket; LOD=0.1-0.4
	1	1	1.9	--	Fresh	Sweden		1988	1	Fish, cooked
	1	1	1.4	--	Fresh	Sweden		1988	1	Fish, raw
	2	0	ND	--	Whole Milk	Switzerland	Background	NR	2	Milk; LOD = 0.012/0.013
	4	3	ND-0.049	0.027	Whole Milk	Switzerland	Near Incinerators	NR	2	Milk; LOD = 0.013
	2	2	0.08-0.2	0.14	Fat	Germany		NR	3	Dairy Products
	5	5	0.01-0.6	0.23	Fat	Germany		NR	3	Meat and Eggs
	3	3	2.8-23	10	Fat	Germany		NR	3	Fish
	8	5	ND-0.33	0.2[a]	Fat	Germany		NR	4	Milk; LOD Not reported
	25	NR	ND-1.9	NC	Fat	Germany		NR	5	Dairy Products; LOD = 0.5
	16	0	ND	--	Fat	Germany		NR	5	Meat; LOD = 0.5
	8	7	ND-0.19	0.12	Wet	Norwich, UK		NR	7	Fish; LOD = 0.16
	7	NR	ND-0.013	0.009[b]	Whole Milk	England/Wales	Rural	1989	7	Milk; LOD = 0.004
	4	0	ND	--	Wet	USSR		1988, 1989	8	Dairy Products; LOD=0.03-0.53
	5	0	ND	--	Wet	USSR		1988, 1989	8	Meat; LOD=0.03-0.72
	3	0	ND	--	Wet	Vietnam		NR	8	Meat; LOD=0.34-0.99
	3	0	ND	--	Lipid	Vietnam		NR	8	Meat; LOD = 1
Hexachlorodibenzo-p-dioxins (MW = 390.87)										
HxCDDs	26	13	ND-67	27	Fat	Canada		1980	6	Chicken; LOD = 5-20
	3	3	0.81-4.95	2.8	Wet	Vietnam		NR	8	Meat; Total for 2,3,7,8 substitutes
	3	3	2.4-5.2	3.4	Lipid	Vietnam		NR	8	Meat; Total for 2,3,7,8 substitutes

Table B-11. Levels of Dioxins in Food Products (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Conc. mean	Wt. basis	Location	Location description	Sample year	Ref. no.	Comments
Heptachlorodibenzo-p-dioxins(MW = 425.31)										
HpCDDs	26	16	ND-142	52	Fat	Canada		1980	6	Chicken; LOD = 5-20
Octachlorodibenzo-p-dioxin(MW=460.76)										
OCDD	3	3	1.0-2.1	1.47	Fresh	Sweden		NR	1	Food basket
	1	1	0.72	--	Fresh	Sweden		1988	1	Fish, cooked
	1	1	0.34	--	Fresh	Sweden		1988	1	Fish, raw
	2	2	≤0.12 - ≤0.16	0.14[c]	Whole Milk	Switzerland	Background	NR	2	Milk
	4	4	≤0.16 - ≤0.59	0.32[c]	Whole Milk	Switzerland	Near Incinerators	NR	2	Milk
	2	2	3.4-10	6.7	Fat	Germany		NR	3	Dairy Products
	5	5	12-68	35	Fat	Germany		NR	3	Meat and Eggs
	3	3	11-83	38	Fat	Germany		NR	3	Fish
	8	0	ND	--	Fat	Germany		NR	4	Milk; LOD = 10
	25	25	2.0-35.0	10.9	Fat	Germany		NR	5	Dairy Products
	16	16	3.1-122	19.2	Fat	Germany		NR	5	Meat
	26	12	ND-238	90	Fat	Canada		1980	6	Chicken; LOD - 10-20
	8	8	1.36-4.81	2.62	Wet	Norwich, UK		NR	7	Fish
	7	7	0.215-0.323	0.230	Whole Milk	England/Wales	Rural	1989	7	Milk
	4	4	0.66-9.01	4.6	Wet	USSR		1988, 1989	8	Dairy Products
	5	5	0.63-11.52	4.5	Wet	USSR		1988, 1989	8	Meat
	3	3	9.46-28	21	Wet	Vietnam		NR	8	Meat
	3	3	24-30	27	Lipid	Vietnam		NR	8	Meat

Table B-11. Levels of Dioxins in Food Products (ppt) (continued)

[a]For ND values 1/2 LOD was used in calculating the mean.
[b]For ND values the detection limit was used in calculating the mean.
[c] Signal to noise ratio S/N $\leq$ 3 (values reported as "$\leq$" are used as real values in calculating mean).
[d] Signal to noise ratio S/N=5/1 (values reported as "$\leq$" are used as real values in calculating mean)

NC = value could not be calculated
NR = not reported
NA = not applicable

Sources:
1. de Wit et al. (1990)
2. Rappe et al. (1987)
3. Beck et al. (1989)
4. Beck et al. (1987)
5. Furst et al. (1990)
6. Ryan et al. (1985)
7. Startin et al. (1990)
8. Schecter et al. (1990)

Table B-12. Levels of Dibenzofurans in Food Products (ppt)

Chemical	Number Samples	Number Positive Samples	Conc. Range	Conc. Mean	Wt. Basis	Location	Location Description	Samp. Year	Ref. No.	Comments
Tetrachlorodibenzofurans (MW=305.98)										
2,3,7,8-TCDF	3	3	0.1-0.4	2.3	Fresh	Sweden		NR	1	Food basket
	1	1	10	--	Fresh	Sweden		1988	1	Fish, cooked
	1	1	7.6	--	Fresh	Sweden		1988	1	Fish, raw
	2	2	≤0.021-≤0.028	0.025[c]	Whole milk	Switzerland	Background	NR	2	Milk
	4	4	≤0.022-≤0.035	0.039[c]	Whole milk	Switzerland	Near Incinerators	NR	2	Milk
	2	2	0.15-0.7	0.43	Fat	Germany		NR	3	Dairy Products
	5	5	0.11-2.1	0.84	Fat	Germany		NR	3	Meat and Eggs
	3	3	57-98	78	Fat	Germany		NR	3	Fish
	8	8	≤0.1-1.4	0.7[d]	Fat	Germany		NR	4	Milk
	25	NR	ND-10.0	NC	Fat	Germany		NR	5	Dairy products; LOD-0.3/0.5
	16	NR	ND-4.0	NC	Fat	Germany		NR	5	Meat; LOD=0.3
	8	8	0.14-2.47	1.21	Wet	Norwich, UK		NR	7	Fish
	7	NR	ND-0.011	0.008[b]	Whole	England/Wales	Rural	1989	7	Milk; LOD-0.006
	4	4	0.021-0.285	0.16	Wet	USSR		1988, 1989	8	Dairy products
	5	3	ND-0.288	0.11	Wet	USSR		1988, 1989	8	Meat; LOD-0.17
	3	3	0.34-1.9	0.91	Wet	Vietnam		NR	8	Meat
	3	3	0.5-2	1.2	Lipid	Vietnam		NR	8	Meat
Hexachlorodibenzofurans (MW=374.87)										
HxCDF	4	4	0.024-2.597	1.06	Wet	USSR		1988,1989	8	Dairy products; Total for 2,3,7,8-substitutes
	5	4	ND-0.72	0.28	Wet	USSR		1988,1989	8	Meat products; Total for 2,3,7,8-substitutes; LOD=0.17
	3	3	0.51-1.39	0.98	Wet	Vietnam		NR	8	Meat products; Total for 2,3,7,8-substitutes

Table B-12. Levels of Dibenzofurans in Food Products (ppt) (continued)

Chemical	Number Samples	Number Positive Samples	Conc. Range	Conc. Mean	Wt. Basis	Location	Location Description	Samp. Year	Ref. No.	Comments
HxCDFs (continued)	3	3	1.1-1.5	1.3	Lipid	Vietnam		NR	8	Meat products; Total for 2,3,7,8-substitutes
Octachlorodibenzofurans (MW=444.76)										
OCDF	3	0	ND	--	Fresh	Sweden		NR	1	Food basket; LOD=0.4-2.1
	1	0	ND	--	Fresh	Sweden		1988	1	Fish, cooked; LOD=0.07
	1	0	ND	--	Fresh	Sweden		1988	1	Fish, raw; LOD=0.07
	2	1	ND-≤0.20	0.1[c]	Whole milk	Switzerland	Background	NR	2	Milk; LOD=0.09
	4	1	ND-≤0.52	0.13[c]	Whole milk	Switzerland	Near Incinerator	NR	2	Milk; LOD=0.13-0.21
	2	2	0.25-1	0.63	Fat	Germany		NR	3	Dairy Products
	5	5	0.2-0.6	0.34	Fat	Germany		NR	3	Meat and Eggs
	3	3	0.3-2.1	1.27	Fat	Germany		NR	3	Fish
	8	0	ND	--	Fat	Germany		NR	4	Milk; LOD=1.0
	25	NR	ND-4.3	NC	Fat	Germany		NR	5	Dairy products; LOD=0.3
	16	NR	ND-5.0	NC	Fat	Germany		NR	5	Meat; LOD=0.3
	8	7	ND-0.26	0.18	Wet	Norwich, UK		NR	7	Fish; LOD=0.30
	7	7	0.023-0.071	0.041	Whole	England/Wales	Rural	1989	7	Milk
	4	0	ND	--	Wet	USSR		1988, 1989	8	Dairy products; LOD=0.02-0.26
	5	1	ND-0.06	0.06	Wet	USSR		1988, 1989	8	Meat; LOD=0.02-0.36
	3	2	ND-0.57	0.34	Wet	Vietnam		NR	8	Meat; LOD=0.3
	3	2	ND-0.6	0.3	Lipid	Vietnam		NR	8	Meat; LOD=0.3

Table B-12. Levels of Dibenzofurans in Food Products (ppt) (continued)

[a]For ND values 1/2 LOD was used in calculating the mean.
[b]For ND values the detection limit was used in calculating the mean.
[c]Signal to noise ratio S/N ≤ 3 (values reported as "$\leq$" are used as real values in calculating mean).
[d]Signal to noise ratio S/N 5/1 (values reported as "$\leq$" are used as real values in calculating mean).

NC = value culd not be calculated.
NR = not reported.
NA = not applicable.

Sources:
1. de Wit et al. (1990)
2. Rappe et al. (1987)
3. Beck et al. (1989)
4. Beck et al. (1987)
5. Furst et al. (1990)
6. Ryan et al. (1985)
7. Startin et al. (1990)
8. Schecter et al. (1990)

Table B-13. Environmental Levels of Dioxins in Air (pg/m³)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
Tetrachlorodibenzo-p-dioxins (MW=321.98)									
2,3,7,8-TCDD	3	0	ND(.01-.02)	NA	Niagra Falls, NY	Urban	86-87	1	
	3	0	ND(.01-.02)	NA	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	ND(0.01)	NA	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.0095)	NA	Los Angeles, CA	Urban	87	3	
	7	2	ND-0.05	0.01	Bridgeport, CT	Urban	87-88	4	
	28	2	ND-0.004	0.002	Wallingford, CT	Urban	88	5	
	NR	NR	ND(0.02)	NA	Rutland, VT	Urban	NR	6	
	NR	NR	ND(0.03)	NA	Durham, NC	Urban	NR	6	
	1	1	0.0004	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.02-0.06	0.04	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.02-0.08	0.05	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	0	ND(.012-0.2)	NA	Akron, OH	Industrial	87	9	near incinerators
	2	0	ND(0.24-0.82)	NA	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.15)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	0	ND(0.058)	NA	Waldo, OH	Rural	87	9	background site
TCDDs	16	3	ND-0.18	0.04	Niagra Falls, NY	Urban	86-87	1	
	16	12	ND-10.12	0.99	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	7	6	ND-0.54	0.20	Bridgeport, CT	Urban	87-88	4	
	27	20	ND-0.07	0.03	Wallingford, CT	Urban	88	5	
	1	1	0.05	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.10-0.22	0.16	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.21-1.5	0.86	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	1	ND-0.18	0.12	Akron, OH	Industrial	87	9	near incinerators
	2	0	ND(0.24-0.82)	NA	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.15)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	0	ND(0.058)	NA	Waldo, OH	Rural	87	9	background site

Table B-13. Environmental Levels of Dioxins in Air (pg/m^3) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
Pentachlorodibenzo-p-dioxins (MW=356.42)									
1,2,3,7,8-PeCDD	3	0	ND(.01-.02)	NA	Niagra Falls, NY	Urban	86-87	1	
	3	1	ND-0.49	0.17	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	ND(0.02)	NA	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.039)	NA	Los Angeles, CA	Urban	87	3	
	7	4	ND-0.07	0.02	Bridgeport, CT	Urban	87-88	4	
	28	9	ND-0.02	0.006	Wallingford, CT	Urban	88	5	
	NR	NR	ND(0.03)	NA	Rutland, VT	Urban	NR	6	
	NR	NR	ND(0.01)	NA	Durham, NC	Urban	NR	6	
	1	1	0.006	NA	Stockholm, Sweden	Urban	89	7	
	2	1	ND-0.28	0.14	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.22-0.60	0.41	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	0	ND(.034-0.27)	NA	Akron, OH	Industrial	87	9	near incinerators
	2	0	ND(.047-.06)	NA	Columbus, OH	Industrial	87	9	near incinerators
	1	1	ND(0.082)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	0	ND(0.033)	NA	Waldo, OH	Rural	87	9	background site
PeCDDs	16	1	ND-0.05	0.02	Niagra Falls, NY	Urban	86-87	1	
	16	11	ND-11.16	1.04	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	7	7	0.01-0.66	0.24	Bridgeport, CT	Urban	87-88	4	
	28	21	ND-0.15	0.05	Wallingford, CT	Urban	88	5	
	1	1	0.11	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.07-1.3	0.68	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	2.4-5.0	3.70	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	1	ND-0.10	0.097	Akron, OH	Industrial	87	9	near incinerators
	2	0	ND(0.47-.06)	NA	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.082)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	0	ND(0.033)	NA	Waldo, OH	Rural	87	9	background site

Table B-13. Environmental Levels of Dioxins in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
Hexachlorodibenzo-p-dioxins (MW=390.87)									
1,2,3,4,7,8-HxCDD	3	0	ND(.01-.02)	NA	Niagra Falls, NY	Urban	86-87	1	
	3	3	.04-.64	0.24	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.01	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.076)	NA	Los Angeles, CA	Urban	87	3	
	7	4	ND-0.08	0.03	Bridgeport, CT	Urban	87-88	4	
	28	8	ND-0.03	0.01	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.05	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.01	Durham, NC	Urban	NR	6	
	1	1	0.004	NA	Stockholm, Sweden	Urban	89	7	
	2	0	ND(0.08-0.17)	NA	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.19-1.0	0.60	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	3	.032-.055	0.041	Akron, OH	Industrial	87	9	near incinerators
	2	0	ND(.028-.039)	NA	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.032)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.031	NA	Waldo, OH	Rural	87	9	background site
1,2,3,6,7,8-HxCDD	3	1	ND-.03	0.02	Niagra Falls, NY	Urban	86-87	1	
	3	3	.05-1.06	0.39	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.02	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.083)	NA	Los Angeles, CA	Urban	87	3	
	7	5	ND-0.13	0.04	Bridgeport, CT	Urban	87-88	4	
	28	22	ND-0.06	0.02	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.07	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.01	Durham, NC	Urban	NR	6	
	1	1	0.008	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.23-0.66	0.44	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.71-2.2	1.46	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	3	.052-.053	0.053	Akron, OH	Industrial	87	9	near incinerators

Table B-13. Environmental Levels of Dioxins in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
1,2,3,6,7,8-HxCDD (continued)	2	1	ND-0.078	0.046	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.032)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.025	NA	Waldo, OH	Rural	87	9	background site
1,2,3,7,8,9-HxCDD	3	1	ND-.03	0.02	Niagra Falls, NY	Urban	86-87	1	
	3	2	ND-0.11	0.06	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.02	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.086)	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.25	0.08	Bridgeport, CT	Urban	87-88	4	
	28	18	ND-0.07	0.03	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.05	Rutland, VT	Urban	NR	6	
	NR	NR	ND(0.01)	NA	Durham, NC	Urban	NR	6	
	1	0	ND(0.001)	NA	Stockholm, Sweden	Urban	89	7	
	2	0	ND(0.08-0.17)	NA	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.36-5.2	2.78	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	3	.017-.050	0.031	Akron, OH	Industrial	87	9	near incinerators
	2	1	ND-.064	0.039	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.032)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.025	NA	Waldo, OH	Rural	87	9	background site
HxCDDs	16	11	ND-0.23	0.08	Niagra Falls, NY	Urban	86-87	1	
	16	12	ND-12.16	1.69	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	7	6	ND-2.17	0.72	Bridgeport, CT	Urban	87-88	4	
	28	27	ND-0.68	0.26	Wallingford, CT	Urban	88	5	
	1	1	0.10	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.74-2.7	1.72	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	5.3-24	14.6	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	3	0.6-0.63	0.62	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.43-0.78	0.60	Columbus, OH	Industrial	87	9	near incinerators
	1	1	0.15	NA	Columbus, OH	Urban	87	9	next to interstate highway

Table B-13. Environmental Levels of Dioxins in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
HxCDDs (continued)	1	1	0.33	NA	Waldo, OH	Rural	87	9	background site
Heptachlorodibenzo-p-dioxins (MW=425.31)									
1,2,3,4,6,7,8-HpCDD	3	3	0.34-0.51	0.41	Niagra Falls, NY	Urban	86-87	1	
	3	2	ND-5.43	2.0	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.11	Greenbay, WI	Urban	NR	2	
	1	1	0.25	NA	Los Angeles, CA	Urban	87	3	
	7	7	0.02-1.07	0.48	Bridgeport, CT	Urban	87-88	4	
	28	23	ND-0.73	0.29	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.41	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.04	Durham, NC	Urban	NR	6	
	1	1	0.10	NA	Stockholm, Sweden	Urban	89	7	
	3	3	0.52-0.57	0.54	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.26-0.52	0.39	Columbus, OH	Industrial	87	9	near incinerators
	1	1	0.32	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.24	NA	Waldo, OH	Rural	87	9	background site
HpCDDs	16	14	ND-0.86	0.44	Niagra Falls, NY	Urban	86-87	1	
	15	15	0.24-9.78	2.60	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	7	7	0.02-2.19	1.02	Bridgeport, CT	Urban	87-88	4	
	28	26	ND-1.48	0.61	Wallingford, CT	Urban	88	5	
	1	1	0.20	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.6-3.4	2.0	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	5.3-15	10.2	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	3	1.00-1.10	1.07	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.41-1.00	0.70	Columbus, OH	Industrial	87	9	near incinerators
	1	1	0.56	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.48	NA	Waldo, OH	Rural	87	9	background site
Octachlorodibenzo-p-dioxin (MW=460.76)									
OCDD	16	15	ND-5.79	1.14	Niagra Falls, NY	Urban	86-87	1	
	14	14	0.39-8.88	2.94	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.30	Greenbay, WI	Urban	NR	2	
	1	1	1.9	NA	Los Angeles, CA	Urban	87	3	
	7	7	0.17-5.55	2.10	Bridgeport, CT	Urban	87-88	4	
	16	15	ND-29.5	5.53	Wallingford, CT	Urban	88	5	

Table B-13. Environmental Levels of Dioxins in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
OCDD (continued)	NR	NR	NR	1.10	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.13	Durham, NC	Urban	NR	6	
	1	1	0.23	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.37-6.4	3.38	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	7.4-40	23.7	Hamburg, Germany	Urban	NR	8	downwind incinerator & industrial complex
	3	3	1.00-1.20	1.13	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.51-1.10	0.80	Columbus, OH	Industrial	87	9	near incinerators
	1	1	0.96	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.50	NA	Waldo, OH	Rural	87	9	background site

Footnote References

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = Not applicable;
ND = Non-detect;
NR = Not reported.

Sources:

1. Smith, et al. (1989).
2. Harless, et al. (1990).
3. Maisel and Hunt (1990).
4. Hunt and Maisel (1990).
5. CDEP (1988).
6. Harless and Lewis (1991).
7. Naf, et al. (1990).
8. Rappe and Kjeller (1987).
9. Edgerton, et al. (1989).

Table B-14. Environmental Levels of Dibenzofurans in Air (pg/m³)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
Tetrachlorodibenzofurans (MW=305.98)									
2,3,7,8-TCDF	3	3	0.04-0.14	0.09	Niagra Falls, NY	Urban	86-87	1	
	3	3	.28-3.81	1.47	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.03	Greenbay, WI	Urban	NR	2	
	1	1	0.02	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-.20	0.08	Bridgeport, CT	Urban	87-88	4	
	28	22	ND-0.10	0.04	Wallingford, CT	Urban	88	5	
	NR	NR	ND(0.1)	NA	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.03	Durham, NC	Urban	NR	6	
	2	2	0.04-0.72	0.38	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.18-0.38	0.28	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	0.19-0.20	0.20	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.32-0.49	0.40	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.13)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.13	NA	Waldo, OH	Rural	87	9	background site
TCDFs	16	10	ND-0.66	0.23	Niagra Falls, NY	Urban	86-87	1	
	16	16	0.18-17.45	3.25	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	7	6	ND-2.29	0.86	Bridgeport, CT	Urban	87-88	4	
	28	26	ND-0.86	0.38	Wallingford, CT	Urban	88	5	
	1	1	0.33	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.36-6.2	3.28	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	3.3-4.9	4.1	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	0.99-1.50	1.23	Akron, OH	Industrial	87	9	near incinerators
	2	2	1.90-3.80	2.85	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.13)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.89	NA	Waldo, OH	Rural	87	9	background site
Pentachlorodibenzofurans (MW=340.42)									
1,2,3,7,8-PeCDF	3	0	ND(0.01)	NA	Niagra Falls, NY	Urban	86-87	1	

Table B-14. Environmental Levels of Dibenzofurans in Air (pg/m^3) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
1,2,3,7,8-PeCDF (continued)	3	3	0.03-0.61	0.25	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.05	Greenbay, WI	Urban	NR	2	
	1	1	0.08	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.10	0.03	Bridgeport, CT	Urban	87-88	4	
	28	9	ND-0.02	0.01	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.03	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.01	Durham, NC	Urban	NR	6	
	3	3	.026-.033	0.029	Akron, OH	Industrial	87	9	near incinerators
	2	2	.032-.057	0.044	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.036)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.021	NA	Waldo, OH	Rural	87	9	background site
2,3,4,7,8-PeCDF	3	0	ND(.01-.02)	NA	Niagra Falls, NY	Urban	86-87	1	
	3	2	ND-1.92	0.68	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.04	Greenbay, WI	Urban	NR	2	
	1	1	0.08	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.16	0.05	Bridgeport, CT	Urban	87-88	4	
	28	16	ND-0.04	0.02	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.20	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.01	Durham, NC	Urban	NR	6	
	1	1	0.02	NA	Stockholm, Sweden	Urban	89	7	
	1	1	0.04	NA	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.43-1.2	0.82	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	.032-.042	0.036	Akron, OH	Industrial	87	9	near incinerators
	2	1	ND-.089	0.050	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.036)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	0	ND(0.033)	NA	Waldo, OH	Rural	87	9	background site
PeCDFs	16	8	ND-0.41	0.12	Niagra Falls, NY	Urban	86-87	1	
	16	15	ND-12.4	2.08	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex

Table B-14. Environmental Levels of Dibenzofurans in Air (pg/m^3) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
PeCDFs (continued)	7	6	ND-1.77	0.57	Bridgeport, CT	Urban	87-88	4	
	28	28	0.04-0.71	0.29	Wallingford, CT	Urban	88	5	
	1	1	0.17	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.51-4.1	2.30	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	5-10	7.5	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	0.53-0.66	0.59	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.69-1.30	1.00	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.036)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.50	NA	Waldo, OH	Rural	87	9	background site
Hexachlorodibenzofurans (MW=374.87)									
1,2,3,4,7,8-HxCDF	3	1	ND-0.06	0.02	Niagra Falls, NY	Urban	86-87	1	
	3	2	ND-0.22	0.11	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.03	Greenbay, WI	Urban	NR	2	
	1	1	0.15	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.41	0.11	Bridgeport, CT	Urban	87-88	4	
	28	21	ND-0.13	0.05	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.04	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.02	Durham, NC	Urban	NR	6	
	3	3	.053-0.10	0.083	Akron, OH	Industrial	87	9	near incinerators
	2	2	.060-0.27	0.16	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.034)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.098	NA	Waldo, OH	Rural	87	9	background site
1,2,3,6,7,8-HxCDF	3	1	ND-0.02	0.01	Niagra Falls, NY	Urban	86-87	1	
	3	3	0.05-1.17	0.45	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.03	Greenbay, WI	Urban	NR	2	
	1	1	0.25	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.15	0.04	Bridgeport, CT	Urban	87-88	4	
	28	17	ND-0.07	0.03	Wallingford, CT	Urban	88	5	

Table B-14. Environmental Levels of Dibenzofurans in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
1,2,3,6,7,8-HxCDF (continued)	NR	NR	NR	0.03	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.02	Durham, NC	Urban	NR	6	
	1	1	0.008	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.03-0.15	0.09	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.24-1.4	0.82	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	.048-.092	0.065	Akron, OH	Industrial	87	9	near incinerators
	2	2	.092-0.19	0.14	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.034)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.014	NA	Waldo, OH	Rural	87	9	background site
1,2,3,7,8,9-HxCDF	3	0	ND(.01-.02)	NA	Niagra Falls, NY	Urban	86-87	1	
	3	1	ND-0.1	0.04	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.01	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.08)	NA	Los Angeles, CA	Urban	87	3	
	7	2	ND-0.02	0.01	Bridgeport, CT	Urban	87-88	4	
	28	1	ND-0.003	0.003	Wallingford, CT	Urban	88	5	
	NR	NR	ND(0.03)	NA	Rutland, VT	Urban	NR	6	
	NR	NR	ND(0.01)	NA	Durham, NC	Urban	NR	6	
	1	1	0.0008	NA	Stockholm, Sweden	Urban	89	7	
	2	0	ND(.01-.05)	NA	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	1	ND-0.33	0.17	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	.020-.039	0.032	Akron, OH	Industrial	87	9	near incinerators
	2	2	.038-0.12	0.079	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.034)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.097	NA	Waldo, OH	Rural	87	9	background site
2,3,4,6,7,8-HxCDF	3	1	ND-0.04	0.02	Niagra Falls, NY	Urban	86-87	1	
	3	2	ND-2.17	0.76	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	ND(.01)	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.08)	NA	Los Angeles, CA	Urban	87	3	

Table B-14. Environmental Levels of Dibenzofurans in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
2,3,4,6,7,8-HxCDF (continued)	7	6	ND-0.30	0.09	Bridgeport, CT	Urban	87-88	4	
	28	19	ND-0.10	0.04	Wallingford, CT	Urban	88	5	
	NR	NR	ND(0.03)	NA	Rutland, VT	Urban	NR	6	
	NR	NR	ND(0.01	NA	Durham, NC	Urban	NR	6	
	1	1	0.005	NA	Stockholm, Sweden	Urban	89	7	
	2	1	ND-0.05	0.03	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.21-0.8	0.50	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	0	ND(.005-.036)	NA	Akron, OH	Industrial	87	9	near incinerators
	2	0	ND(.012-.028)	NA	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.034)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	0	ND(.008)	NA	Waldo, OH	Rural	87	9	background site
HxCDFs	16	11	ND-0.58	0.14	Niagra Falls, NY	Urban	86-87	1	
	16	15	ND-10.23	1.96	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	7	6	ND-2.15	0.58	Bridgeport, CT	Urban	87-88	4	
	28	28	0.03-1.57	0.49	Wallingford, CT	Urban	88	5	
	1	1	0.08	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.18-1.1	0.64	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	2.2-9.5	5.85	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	0.56-0.70	0.62	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.37-1.20	0.78	Columbus, OH	Industrial	87	9	near incinerators
	1	1	0.10	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.51	NA	Waldo, OH	Rural	87	9	background site
Heptachlorodibenzofurans (MW=409.31)									
1,2,3,4,6,7,8-HpCDF	3	1	ND-0.15	0.05	Niagra Falls, NY	Urban	86-87	1	
	3	3	0.26-5.43	2.08	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.08	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.2)	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.54	0.21	Bridgeport, CT	Urban	87-88	4	

Table B-14. Environmental Levels of Dibenzofurans in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
1,2,3,4,6,7,8-HpCDF (continued)	28	22	ND-0.80	0.26	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.12	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.02	Durham, NC	Urban	NR	6	
	1	1	0.09	NA	Stockholm, Sweden	Urban	89	7	
	3	3	0.22-0.25	0.24	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.20-0.47	0.34	Columbus, OH	Industrial	87	9	near incinerators
	1	1	0.087	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.22	NA	Waldo, OH	Rural	87	9	background site
1,2,3,4,7,8,9-HpCDF	NR	NR	NR	0.01	Greenbay, WI	Urban	NR	2	
	1	0	ND(0.02)	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.07	0.03	Bridgeport, CT	Urban	87-88	4	
	28	12	ND-0.30	0.03	Wallingford, CT	Urban	88	5	
	NR	NR	ND(0.01)	NA	Rutland, VT	Urban	NR	6	
	NR	NR	ND(0.01)	NA	Durham, NC	Urban	NR	6	
	1	0	ND(0.001)	NA	Stockholm, Sweden	Urban	89	7	
	3	1	ND-0.031	0.020	Akron, OH	Industrial	87	9	near incinerators
	2	0	ND(.015-.028)	NA	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.013)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.019	NA	Waldo, OH	Rural	87	9	background site
HpCDFs	16	12	ND-0.43	0.13	Niagra Falls, NY	Urban	86-87	1	
	16	12	ND-8.76	1.67	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	7	6	ND-1.0	0.37	Bridgeport, CT	Urban	87-88	4	
	28	26	ND-1.58	0.47	Wallingford, CT	Urban	88	5	
	1	1	0.11	NA	Stockholm, Sweden	Urban	89	7	
	2	2	0.1-1.2	0.65	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	2-5	3.5	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	0.37-0.39	0.38	Akron, OH	Industrial	87	9	near incinerators
	2	2	0.26-0.64	0.45	Columbus, OH	Industrial	87	9	near incinerators

Table B-14. Environmental Levels of Dibenzofurans in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Conc. mean	Location	Location description	Sample year	Ref. no.	Comments
HpCDFs (continued)	1	1	0.15	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.29	NA	Waldo, OH	Rural	87	9	background site
Octachlorodibenzofurans (MW=444.76)									
OCDFs	16	11	ND-0.22	0.09	Niagra Falls, NY	Urban	86-87	1	
	15	8	ND-3.38	0.62	Niagra Falls, NY	Industrial	86-87	1	downwind industrial complex
	NR	NR	NR	0.07	Greenbay, WI	Urban	NR	2	
	1	1	0.06	NA	Los Angeles, CA	Urban	87	3	
	7	6	ND-0.56	0.21	Bridgeport, CT	Urban	87-88	4	
	28	18	ND-0.70	0.21	Wallingford, CT	Urban	88	5	
	NR	NR	NR	0.14	Rutland, VT	Urban	NR	6	
	NR	NR	NR	0.03	Durham, NC	Urban	NR	6	
	1	1	0.02	NA	Stockholm, Sweden	Urban	89	7	
	2	0	ND(.11-1.0)	NA	Hamburg, Germany	Urban	NR	8	urban air & inside traffic tunnel
	2	2	0.78-7.0	3.89	Hamburg, Germany	Industrial	NR	8	downwind incinerator & industrial complex
	3	3	0.17-0.19	0.18	Akron, OH	Industrial	87	9	near incinerators
	2	1	ND-0.21	0.18	Columbus, OH	Industrial	87	9	near incinerators
	1	0	ND(0.16)	NA	Columbus, OH	Urban	87	9	next to interstate highway
	1	1	0.077	NA	Waldo, OH	Rural	87	9	background site

Footnote References

NOTES: Summary statistics provided in or derived from references; means from references included non-detects counted as 0.0; when reference did not compute mean, it was computed with same strategy;
NA = not available;
NR = not reported;
ND = Non-detect.

Sources:

1. Smith, et al. (1989).
2. Harless, et al. (1990).
3. Maisel and Hunt (1990).
4. Hunt and Maisel (1990).
5. CDEP (1988).
6. Harless and Lewis (1991).
7. Naf, et al. (1990).
8. Rappe and Kjeller (1987).
9. Edgerton, et al. (1989).

Table B-15. Mean Background Environmental Levels of Dioxins in Soil (ppt)

Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no.
Tetrachlorodibenzo-p-dioxins (MW=321.98)							
2,3,7,8-TCDD	135	19	ND-7	0.71	0.46	World Wide	1,6,7,10,11
	43	7	ND-5	0.88	0.81	North America	1,6
	92	12	ND-7	0.62	0.36	Europe	7,10,11
TCDDs	155	13	ND-120	17.4	12.4	World Wide	1,4,6,7,10,11
	65	0	ND(1.0-55)	22.6	14.2	North America	1,4,6
	90	13	ND-120	13.6	11.3	Europe	7,10,11
Pentachlorodibenzo-p-dioxins (MW=356.42)							
1,2,3,7,8-PeCDD	81	0,NR	ND-2.4	0.33	0.28	World Wide	6,11
	4	0	ND(3.75)	1.88	1.88	North America	6
	77	NR	ND-2.4	0.25	0.25	Europe	11
PeCDDs	155	14	ND-50	14.5	10.3	World Wide	1,4,6,7,10,11
	65	1	ND-38	23.1	15.8	North America	1,4,6
	90	13	ND-50	8.36	7.62	Europe	7,10,11
Hexachlorodibenzo-p-dioxins (MW=390.87)							
1,2,3,4,7,8-HxCDD	4	0	ND(3.75)	1.88	1.88	World Wide	6
	4	0	ND(3.75)	1.88	1.88	North America	6
	ND	ND	ND	ND	ND	Europe	ND
1,2,3,6,7,8-HxCDD	4	1	ND-14	4	4	World Wide	6
	4	1	ND-14	4	4	North America	6
	ND	ND	ND	ND	ND	Europe	ND
1,2,3,7,8,9-HxCDD	4	2	ND-9.9	9	9	World Wide	6
	4	2	ND-9.9	9	9	North America	6
	ND	ND	ND	ND	ND	Europe	ND
HxCDDs	155	18	ND-165	31.8	28.9	World Wide	1,4,6,7,10,11
	65	5	ND-99	26.5	22.5	North America	1,4,6
	90	13	ND-165	35.6	34.7	Europe	7,10,11

Table B-15. Mean Background Environmental Levels of Dioxins in Soil (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no.
Heptachlorodibenzo-p-dioxins (MW=425.31)							
1,2,3,4,6,7,8-HpCDD	4	4	37-360	194	194	World Wide	6
	4	4	37-360	194	194	North America	6
	ND	ND	ND	ND	ND	Europe	ND
HpCDDs	187	28	ND-640	51.4	34.9	World Wide	1,4,6,7,10,11,13
	95	15	ND-640	39.2	20.1	North America	1,4,6,13
	92	13	ND-234	63.9	61.6	Europe	7,10,11
Octachlorodibenzo-p-dioxin (MW=460.76)							
OCDD	185	42	ND-10,600	205	96.1	World Wide	1,4,6,7,10,11,13
	95	29	ND-10,600	237	59.9	North America	1,4,6,13
	90	13	ND-832	171	158	Europe	7,10,11
TEQs for Dibenzodioxins				4.51	4.12	World Wide	
				5.48	5.24	North America	
				0.92	0.64	Europe	

Footnote References

[a] Means were taken from pristine, reference, residential, rural, and agricultural sites. Industrial, urban, and dump sites were not used because they were assumed to be contaminated.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)

Sources:

1. EPA (1985)
4. Pearson, et al. (1990).
6. Reed, et al. (1990).
7. Stenhouse and Badsha (1990).
10. Rappe and Kjeller (1987).
11. Creaser, et al. (1989).

Table B-16. Mean Background Environmental Levels of Dibenzofurans in Soil (ppt)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric Mean[a]	Location	Ref. no(s).
Tetrachlorodibenzofurans (MW=305.98)							
2,3,7,8-TCDF	24	14	ND-50	9.29	4.85	World Wide	1,6,7
	12	2	ND-6	1.59	1.39	North America	1,6
	12	12	3-50	17	17	Europe	7
TCDFs	150	17	ND-300	42.2	33.8	World Wide	1,4,6,7,10,11
	58	2	ND-280	54.2	43.0	North America	1,4,6
	92	15	ND-300	34.5	29.1	Europe	7,10,11
Pentachlorodibenzofurans (MW=340.42)							
1,2,3,7,8-PeCDF	16	12	ND-10	3.47	3.31	World Wide	6,7
	4	0	ND(3.75)	1.88	1.88	North America	6
	12	12	1-10	4	4	Europe	7
2,3,4,7,8-PeCDF	16	12	ND-5	1.97	1.97	World Wide	6,7
	4	0	ND(3.75)	1.88	1.88	North America	6
	12	12	1-5	2	2	Europe	7
PeCDFs	154	18	ND-185	24.5	22.0	World Wide	1,4,6,7,10,11
	62	3	ND-45	25.8	20.2	North America	1,4,6
	92	15	ND-185	23.7	23.4	Europe	7,10,11
Hexachlorodibenzofurans (MW=374.87)							
1,2,3,4,7,8-HxCDF	4	0	ND(3.75)	1.88	1.88	World Wide	6
	4	0	ND(3.75)	1.88	1.88	North America	6
1,2,3,6,7,8-HxCDF	4	0	ND(3.75)	1.88	1.88	World Wide	6
	4	0	ND(3.75)	1.88	1.88	North America	6
	ND	ND	ND	ND	ND	Europe	ND
1,2,3,7,8,9-HxCDF	4	0	ND(3.75)	1.88	1.88	World Wide	6
	4	0	ND(3.75)	1.88	1.88	North America	6
	ND	ND	ND	ND	ND	Europe	ND
2,3,4,6,7,8-HxCDF	4	1	ND-7.1	2	2	World Wide	6
	4	1	ND-7.1	2	2	North America	6
HxCDF	157	19	ND-212	33.2	27.2	World Wide	1,4,6,7,10,11
	65	4	ND-150	26.5	17.7	North America	1,4,6

Table B-16. Mean Background Environmental Levels of Dibenzofurans in Soil (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric Mean[a]	Location	Ref. no(s).
	92	15	ND-212	37.9	36.8	Europe	7,10,11
Heptachlorodibenzofurans (MW=409.31)							
1,2,3,4,6,7,8-HpCDF	4	4	11-80	47	47	World Wide	6
	4	4	11-80	47	47	North America	6
	ND	ND	ND	ND	ND	Europe	ND
1,2,3,4,7,8,9-HpCDF	4	0	ND(3.75)	1.88	1.88	World Wide	6
	4	0	ND(3.75)	1.88	1.88	North America	6
	ND	ND	ND	ND	ND	Europe	ND
HpCDFs	157	20	ND-260	26.8	23.9	World Wide	1,4,6,7,10,11
	65	5	ND-260	29.3	22.7	North America	1,4,6
	92	15	ND-138	25.0	24.8	Europe	7,10,11
Octachlorodibenzofurans (MW=444.76)							
OCDFs	155	17	ND-270	28.4	25.2	World Wide	1,4,6,7,10,11
	65	4	ND-270	30.2	23.1	North America	1,4,6
	90	13	ND-144	27.2	26.9	Europe	7,10,11
TEQs for Dibenzofurans				3.37	2.91	World Wide	
				2.48	2.45	North America	
				2.93	2.93	Europe	

Footnote References

[a] Means were taken from pristine, reference, residential, rural, and agricultural sites. Industrial, urban, and dump sites were not used because they were assumed to be contaminated.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)

Sources:

1. EPA (1985)
4. Pearson, et al. (1990).
6. Reed, et al. (1990).
7. Stenhouse and Badsha (1990).
10. Rappe and Kjeller (1987)
11. Creaser, et al. (1989).

Table B-17. Mean Background Levels of Dioxins in Water (ppq)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
Tetrachlorodibenzo-p-dioxins (MW=321.98)							
2,3,7,8-TCDD	1	0	ND(0.7)	0.35	0.35	World Wide	2
	1	0	ND(0.7)	0.35	0.35	North America	2
TCDDs	208	2	ND-40	0.37	0.28	World Wide	1,2
	208	2	ND-40	0.37	0.37	North America	1,2
Pentachlorodibenzo-p-dioxins (MW=356.42)							
1,2,3,7,8-PeCDD	1	0	ND(1.0)	0.5	0.5	World Wide	2
	1	0	ND(1.0)	0.5	0.5	North America	2
PeCDDs	23	0	ND(1.0-7.4)	4.13	3.92	World Wide	2
	23	0	ND(1.0-7.4)	4.13	3.92	North America	2
Hexachlorodibenzo-p-dioxins (MW=390.87)							
1,2,3,4,7,8-HxCDD	1	0	ND(1.8)	0.9	0.9	World Wide	2
	1	0	ND(1.8)	0.9	0.9	North America	2
1,2,3,6,7,8-HxCDD	1	0	ND(1.5)	0.75	0.75	World Wide	2
	1	0	ND(1.5)	0.75	0.75	North America	2
1,2,3,7,8,9-HxCDD	1	0	ND(1.5)	0.75	0.75	World Wide	2
	1	0	ND(1.5)	0.75	0.75	North America	2
HxCDDs	23	0	ND(0.4-4.7)	2.47	2.55	World Wide	2
	23	0	ND(0.4-4.7)	2.47	2.55	North America	2

Table B-17. Mean Background Levels of Dioxins in Water (ppq) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
Heptachlorodibenzo-p-dioxins (MW=425.31)							
1,2,3,4,6,7,8-HpCDD	1	0	ND(2.8)	1.4	1.4	World Wide	2
	1	0	ND(2.8)	1.4	1.4	North America	2
HpCDDs	23	0	ND(0.4-6.8)	3.50	3.46	World Wide	2
	23	0	ND(0.4-6.8)	3.5	3.46	North America	2
Octachlorodibenzo-p-dioxin (MW=460.76)							
1,2,3,4,6,7,8,9-OCDD	399	36	ND-175	4.26	2.10	World Wide	1
	399	36	ND-175	4.26	2.10	North America	1
TEQs for Dibenzodioxins				0.86	0.86	World Wide	
				0.86	0.86	North America	

Footnote References

[a] Means were taken from non-contaminated sites.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)

Sources:
1. Jobb, et al. (1990)
2. Meyer, et al. (1989)

Table B-18. Mean Background Levels of Dibenzofurans in Water (ppq)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean*	Geometric mean*	Location	Ref. no(s).
Tetrachlorodibenzofurans (MW=305.98)							
2,3,7,8-TCDF	23	1	ND-1.2	0.06	0.05	World Wide	2
	23	1	ND-1.2	0.06	0.05	North America	2
TCDFs	23	2	ND-18	0.9	0.15	World Wide	2
	23	2	ND-18	0.9	0.15	North America	2
Pentachlorodibenzofurans (MW=340.42)							
1,2,3,7,8-PeCDF	1	1	2.0	2.0	2.0	World Wide	2
	1	1	2.0	2.0	2.0	North America	2
2,3,4,7,8-PeCDF	1	0	ND(1.0)	0.5	0.5	World Wide	2
	1	0	ND(1.0)	0.5	0.5	North America	2
PeCDFs	23	1	ND-27	3.18	2.35	World Wide	2
	23	1	ND-27	3.18	2.35	North America	2
Hexachlorodibenzofurans (MW=374.87)							
1,2,3,4,7,8-HxCDF	1	1	39	39	39	World Wide	2
	1	1	39	39	39	North America	2
1,2,3,6,7,8-HxCDF	1	1	9.2	9.2	9.2	World Wide	2
	1	1	9.2	9.2	9.2	North America	2
2,3,4,6,7,8-HxCDF	1	0	ND(1.3)	0.65	0.65	World Wide	2
	1	0	ND(1.3)	0.65	0.65	North America	2
1,2,3,7,8,9-HxCDF	1	0	ND(1.2)	0.6	0.6	World Wide	2
	1	0	ND(1.2)	0.6	0.6	North America	2

Table B-18. Mean Background Levels of Dibenzofurans in Water (ppq) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
HxCDFs	23	1	ND-85	5.94	2.75	World Wide	2
	23	1	ND-85	5.94	2.75	North America	2
Heptachlorodibenzofurans (MW=409.31)							
1,2,3,4,6,7,8-HpCDF	1	1	210	210	210	World Wide	2
	1	1	210	210	210	North America	2
HpCDFs	23	1	ND-210	12.67	4.41	World Wide	2
	23	1	ND-210	12.67	4.41	North America	2
Octachlorodibenzofurans (MW=444.76)							
1,2,3,4,6,7,8,9-OCDF	23	3	ND-230	10.67	0.9	World Wide	2
	23	3	ND-230	10.67	0.9	North America	2
TEQs for Dibenzofurans				7.41	7.40	World Wide	
				7.41	7.40	North America	

Footnote References

[a] Means were taken from non-contaminated sites.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)

Sources: 2. Meyer, et al. (1989)

Table B-19. Mean Background Environmental Levels of Dioxins in Sediments (ppt)

Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a] (ppt)	Geometric mean[a] (ppt)	Location	Ref. no(s).
Tetrachlorodibenzo-p-dioxins (MW=321.98)							
2,3,7,8-TCDD	25	2	ND-35	25.9	16.2	World Wide	1,4,5,13
	5	0	ND(4.7-12.7)	3.16	2.88	North America	4,5
	20	2	ND-35	31.6	25.0	Europe	1,13
TCDDs	27	18	ND-1400	252	87.4	World Wide	1,3,5,13
	7	2	ND-26	6.81	2.95	North America	3,5
	20	16	ND-1400	338	286	Europe	1,13
Pentachlorodibenzo-p-dioxins (MW=356.42)							
PeCDDs	9	4	ND-100	20.5	5.52	World Wide	3,5,13
	7	2	ND-12	4.66	2.61	North America	3,5
	2	2	52-100	76	76	Europe	13
Hexachlorodibenzo-p-dioxins (MW=390.87)							
HxCDDs	9	6	ND-170	36.6	8.56	World Wide	3,5,13
	7	4	ND-14	5.57	3.81	North America	3,5
	2	2	120-170	145	145	Europe	13
Heptachlorodibenzo-p-dioxins (MW=425.31)							
HpCDDs	6	6	7.3-210	95.7	90.1	World Wide	5,13
	4	4	7.3-110	71	71	North America	5
	2	2	79-210	145	145	Europe	13
Octachlorodibenzo-p-dioxin (MW=460.76)							
OCDD	27	12	ND-600	131	22.5	World Wide	1,3,5,13

Table B-19. Mean Background Environmental Levels of Dioxins in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a] (ppt)	Geometric mean[a] (ppt)	Location	Ref. no(s).
OCDD (continued)	7	7	54-600	439	364	North America	3,5
	20	5	ND-250	22.5	8.51	Europe	1,13
TEQs for Dibenzodioxins				26.0	16.2	World Wide	
				3.60	3.24	North America	
				31.6	25.0	Europe	

Footnote References

[a] Means were taken from pristine, reference, residential, rural, and various location sites. Industrial, urban, and dump sites were not used because they were assumed to be contaminated.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)
ppt = Parts per trillion

Sources:

1. Koistinen et al. (1990)
3. Czuczwa et al. (1984)
4. Norwood et al. (1989)
5. Reed et al. (1990)
13. Rappe et al. (1989a)

Table B-20. Mean Background Environmental Levels of Dibenzofurans in Sediments (ppt)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
Tetrachlorodibenzofurans (MW=305.98)							
2,3,7,8-TCDF	25	4	ND-35	26.7	11.7	World Wide	1,4,5,13
	5	2	ND-15	3.06	0.23	North America	4,5
	20	2	ND-35	32.6	31.2	Europe	1,13
TCDFs	9	6	ND-130	28.0	2.21	World Wide	3,5,13
	7	4	ND-18	4.86	0.72	North America	3,5
	2	2	87-130	109	109	Europe	13
Pentachlorodibenzofurans (MW=340.42)							
PeCDFs	9	6	ND-125	25.4	7.25	World Wide	3,5,13
	7	4	ND-25	5.26	3.47	North America	3,5
	2	2	66-125	96	96	Europe	13
Hexachlorodibenzofurans (MW=374.87)							
HxCDFs	9	5	ND-150	27.1	4.55	World Wide	3,5,13
	7	3	ND-12	2.31	1.81	North America	3,5
	2	2	78-150	114	114	Europe	13
Heptachlorodibenzofurans (MW=409.31)							
HpCDFs	6	5	ND-180	54.0	32.2	World Wide	5,13
	4	3	ND-30	16	16	North America	5
	2	2	79-180	130	130	Europe	13
Octachlorodibenzofurans (MW=444.76)							
OCDF	27	8	ND-160	10.6	8.72	World Wide	1,3,5,13

Table B-20. Mean Background Environmental Levels of Dibenzofurans in Sediments (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
OCDF (continued)	7	4	ND-23	4.50	3.98	North America	3,5
	20	4	ND-160	12.8	11.5	Europe	1,13
TEQs for Dibenzofurans				2.68	1.18	World Wide	
				0.31	0.03	North America	
				3.27	3.13	Europe	

Footnote References

[a] Means were taken from pristine, reference, residential, rural, and various location sites. Industrial, urban, and dump sites were not used because they were assumed to be contaminated.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)
ppt = Parts per trillion

Sources:

1. Koistinen et al. (1990)
3. Czuczwa et al. (1984)
4. Norwood et al. (1989)
5. Reed et al. (1990)
13. Rappe et al. (1989a)

Table B-21. Mean Background Environmental Levels of PCBs in Sediment (ppt)

IUPAC number	Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
Tetrachloro-PCB (MW=291.99)								
77	3,3'4,4'-TeCB	19	13	ND-550	144	142	World Wide	1,16
		NR	NR	ND(500)	250	250	North America	16
		18	13	ND-550	137.7	137.7	Europe	1
81	3,4,4'5-TeCB	NR	NR	ND(500)	250	250	World Wide	16
		NR	NR	ND(500)	250	250	North America	16
Pentachloro-PCB (MW=326.44)								
126	3,3',4,4',5-PeCB	19	1	ND-110	18.9	7.42	World Wide	1,16
		NR	NR	ND(500)	250	250	North America	16
		18	1	ND-110	6.1	6.1	Europe	1
105	2,3,3',4,4'-PeCB	49	10	ND-10,000	7893	3835	World Wide	1,7,16
		39	NR	NR	9892	9861	North America	7,16
		10	10	52-120	96.4	96.4	Europe	1
114	2,3,4,4',5-PeCB	NR	NR	NR	1000	1000	World Wide	16
		NR	NR	NR	1000	1000	North America	16
118	2,3,4,4',5-PeCB			11,000-15,000	14,897	14,881	World Wide	7,16
		38	NR	11,000-15,000	14,897	14,881	North America	7,16
Hexachloro-PCB (MW=360.88)								
156	2,3,3',4,4',5-HxCB			1700-2100	2090	2089	World Wide	7,16
		38	NR	1700-2100	2090	2089	North America	7,16
167	2,3',4,4',5,5'-HxCB	R	NR	ND(500)	250	250	World Wide	16
		NR	NR	ND(500)	250	250	North America	16
169	3,3',4,4',5,5'-HxCB	18	0	ND(36-500)	30.21	20.67	World Wide	1,16

Table B-21. Mean Background Environmental Levels of PCBs in Sediments (ppt) (continued)

IUPAC number	Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
169	3,3',4,4',5,5'-HxCB	NR	NR	ND(500)	250	250	North America	16
	(continued)	18	0	ND(36)	18	18	Europe	1
Heptachloro-PCB (MW=396.33)								
189	2,3,3',4,4',5,5'-HpCB	NR	NR	ND(500)	250	250	World Wide	16
		NR	NR	ND(500)	250	250	Europe	16

Footnote References

[a] Means were taken from various location sites. Industrial and urban sites were not used because they were assumed to be contaminated.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

NR = Not Reported
ND = Not Detected (detection limit)
ppt = Parts per trillion

Sources:

1. Koistinen et al. (1990)
7. Oliver and Nilmi (1988)
9. Huckins et al. (1988)
16. Smith et al. (1990)

Table B-22. Mean Background Environmental Levels of Dioxins in Finfish (ppt)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
Tetrachlorodibenzo-p-dioxins (MW=321.98)							
2,3,7,8-TCDD	35	34	ND-2.26	0.54	0.51	World Wide	5,20
	34	34	0.06-2.26	0.56	0.56	North America	20
	1	0	ND(0.1)	0.025	0.025	Europe	5
Pentachlorodibenzo-p-dioxins (MW=356.42)							
1,2,3,7,8-PeCDD	35	35	0.15-2.67	0.76	0.75	World Wide	5,20
	34	34	0.15-2.67	0.77	0.77	North America	20
	1	1	0.3	0.3	0.3	Europe	5
Hexachlorodibenzo-p-dioxins (MW=390.87)							
1,2,3,4,7,8-HxCDD	1	0	ND(0.2)	0.05	0.05	World Wide	5
	1	0	ND(0.2)	0.05	0.05	Europe	5
1,2,3,6,7,8-HxCDD	1	0	ND(0.2)	0.05	0.05	World Wide	5
	1	0	ND(0.2)	0.05	0.05	Europe	5
1,2,3,7,8,9-HxCDD	1	0	ND(0.2)	0.05	0.05	World Wide	5
	1	0	ND(0.2)	0.05	0.05	Europe	5
HxCDDs	34	NR	ND-3.57	0.39	0.39	World Wide	20
	34	NR	ND-3.57	0.39	0.39	North America	20
Heptachlorodibenzo-p-dioxins (MW=425.31)							
1,2,3,4,6,7,8-HpCDD	1	0	ND(0.2)	0.05	0.05	World Wide	5
	1	0	ND(0.2)	0.05	0.05	Europe	5
Octachlorodibenzo-p-dioxin (MW=460.76)							
1,2,3,4,6,7,8,9-OCDD	1	1	0.55	0.55	0.55	World Wide	5
	1	1	0.55	0.55	0.55	Europe	5

Table B-22. Mean Background Environmental Levels of Dioxins in Finfish (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
TEQs for Dibenzodioxins				0.94	0.90	World Wide	
				0.94	0.94	North America	
				0.19	0.19	Europe	

Footnote References

[a] Whole fish concentrations were divided in half to obtain the mean concentrations (USEPA, 1990; Branson et al., 1985).

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection);

Sources: 5. Rappe, et al. (1989)
20. USEPA (1992)

Table B-23. Mean Background Environmental Levels of Dibenzofurans in Finfish (ppt)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
Tetrachlorodibenzofurans (MW=305.98)							
2,3,7,8-TCDF	35	35	0.1-13.73	1.59	1.58	World Wide	5,20
	34	34	0.1-13.73	1.61	1.61	North America	20
	1	1	0.85	0.85	0.85	Europe	5
Pentachlorodibenzofurans (MW=340.42)							
1,2,3,7,8-PeCDF	35	35	0.1-1.9	0.42	0.42	World Wide	5,20
	34	34	0.1-1.9	0.43	0.43	North America	20
	1	1	0.2	0.2	0.2	Europe	5
2,3,4,7,8-PeCDF	35	35	0.1-1.5	0.53	0.52	World Wide	5,20
	34	34	0.1-1.39	0.50	0.50	North America	20
	1	1	1.5	1.5	1.5	Europe	5
Hexachlorodibenzofurans (MW=374.87)							
1,2,3,4,7,8-HxCDF	1	1	0.1	0.1	0.1	World Wide	5
	1	1	0.1	0.1	0.1	Europe	5
1,2,3,6,7,8-HxCDF	1	1	0.05	0.05	0.05	World Wide	5
	1	1	0.05	0.05	0.05	Europe	5
2,3,4,6,7,8-HxCDF	1	0	ND(0.2)	0.05	0.05	World Wide	5
	1	0	ND(0.2)	0.05	0.05	Europe	5
HxCDFs	34	NR	ND-2.59	0.22	0.22	World Wide	20
	34	NR	ND-2.59	0.22	0.22	North America	20

Table B-23. Mean Background Environmental Levels of Dibenzofurans in Finfish (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).
Heptachlorodibenzofurans (MW=409.31)							
1,2,3,4,6,7,8-HpCDF	1	0	ND(0.2)	0.05	0.05	World Wide	5
	1	0	ND(0.2)	0.05	0.05	Europe	5
Octachlorodibenzofurans (MW=444.76)							
1,2,3,4,6,7,8,9-OCDF	1	0	ND(0.2)	0.05	0.05	World Wide	5
	1	0	ND(0.2)	0.05	0.05	Europe	5
TEQs for Dibenzofurans				0.47	0.46	World Wide	
				0.43	0.43	North America	
				0.87	0.87	Europe	

Footnote References

[a] Whole fish concentrations were divided in half to obtain the mean concentrations (USEPA, 1990; Branson et al., 1985).

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)

Sources:
5. Rappe, et al. (1989)
20. USEPA (1992)

Table B-24. Mean Background Levels of Dioxins in Food Products (ppt)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).	Comments
Tetrachlorodibenzo-p-dioxins (MW=321.98)								
2,3,7,8-TCDD	3	0	ND(0.1-0.4)	0.125	0.125	World Wide	1	Food basket
	13	12	ND-23	2.64	0.50	World Wide	1,3,7	Fish
	31	2	ND-1.9	0.92	0.80	World Wide	3,5,8	Dairy products
	17	5	ND-0.33	0.099	0.040	World Wide	2,4,7	Milk
	32	11	ND-5.2	0.77	0.44	World Wide	3,5,8	Meat
Hexachlorodibenzo-p-dioxins (MW = 390.87)								
HxCDDs	26	13	ND-67	27	27	Canada	6	Chicken
	6	6	0.81-5.2	3.1	3.09	World Wide	8	Meat
Heptachlorodibenzo-p-dioxins (MW = 425.31)								
HpCDDs	26	16	ND-142	52	52	Canada	6	Chicken
Octachlorodibenzo-p-dioxin (MW=460.76)								
OCDD	3	3	1.0-2.1	1.47	1.47	World Wide	1	Food basket
	13	13	0.34-83	10.46	3.76	World Wide	1,3,7	Fish
	31	31	0.66-35	9.82	9.45	World Wide	3,5,8	Dairy products
	17	9	ND-0.323	2.46	0.92	World Wide	2,4,7	Milk
	26	12	ND-238	90	90	Canada	6	Chicken
	32	32	0.63-122	20.27	17.5	World Wide	3,5,8	Meat
TEQs for Dibenzodioxins				0.13	0.13	World Wide		Food basket
				2.65	0.50	World Wide		Fish
				0.93	0.81	World Wide		Dairy products
				0.10	0.04	World Wide		Milk

Table B-24. Mean Background Levels of Dioxins in Food Products (ppt) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no(s).	Comments
TEQs for Dibenzofurans (continued)				0.09	0.09	World Wide		Chicken
				0.79	0.46	World Wide		Meat

Footnote References

[a] Means were taken from all sites except near incinerators.
World Wide locations do not include North America

Notes: One-half the limit of detection was used in calculating the means where applicable.

ND = not detected (limit of detection)

Sources:
1. de Wit et al. (1990)
2. Rappe et al. (1987)
3. Beck et al. (1989)
4. Beck et al. (1987)
5. Furst et al. (1990)
6. Ryan et al. (1985)
7. Startin et al. (1990)
8. Schecter et al. (1990)

Table B-25. Mean Background Levels of Dibenzofurans in Food Products (ppt)

Chemical	Number Samples	Number Positive Samples	Conc. Range	Arithmetic mean	Geometric mean[a]	Location	Ref. no(s).	Comments
Tetrachlorodibenzofurans (MW=305.98)								
2,3,7,8-TCDF	3	3	0.1-0.4	2.3	2.3	World Wide	1	Food basket
	13	13	0.14-98	20.1	4.29	World Wide	1,3,7	Fish
	31	6	ND-10.0	4.24	2.83	World Wide	3,5,8	Dairy products
	17	10	ND-1.4	0.34	0.075	World Wide	2,4,7	Milk
	32	11	ND-4.0	1.45	1.03	World Wide	3,5,8	Meat
Hexachlorodibenzofurans (MW=374.87)								
HxCDFs	4	47	0.024-2.60	1.06	1.06	World Wide	8	Dairy Products
	11	10	ND-1.5	0.75	0.60	World Wide	8	Meat
Octachlorodibenzofurans (MW=444.76)								
1,2,3,4,6,7,8,9-OCDF	3	0	ND(4.50)	2.25	2.25	World Wide	1	Food basket
	13	10	ND-2.1	0.41	0.23	World Wide	1,3,7	Fish
	31	2	ND-4.3	1.91	1.47	World Wide	3,5,8	Dairy products
	17	8	ND-0.2	0.26	0.15	World Wide	2,4,7	Milk
	32	10	ND-5.0	1.45	0.72	World Wide	3,5,8	Meat
TEQs for Dibenzofurans				0.23	0.23	World Wide		Food basket
				2.01	0.43	World Wide		Fish
				0.43	0.43	World Wide		Dairy products
				0.03	0.008	World Wide		Milk
				0.15	0.10	World Wide		Meat

Table B-25. Mean Background Levels of Dibenzofurans in Food Products (ppt) (continued)

Footnote References

[a] Means were taken from all sites except near incinerators.
World Wide locations do not include North America

Notes: One-half the limit of detection was used in calculating the means where applicable.

ND = not detected (limit of detection)

Sources:
1. de Wit et al. (1990)
2. Rappe et al. (1987)
3. Beck et al. (1989)
4. Beck et al. (1987)
5. Furst et al. (1990)
7. Startin et al. (1990)
8. Schecter et al. (1990)

Table B-26. Mean Background Environmental Levels of Dioxins in Air (pg/m³)

Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no.
Tetrachlorodibenzo-p-dioxins (MW=321.98)							
2,3,7,8-TCDD	41	5	ND-0.05	0.005	0.003	World Wide	1,3,4,5,7,9
	40	4	ND-0.05	0.005	0.003	North America	1,3,4,5,9
	1	1	0.0004	0.0004	0.0004	Europe	7
TCDDs	52	30	ND-0.54	0.06	0.04	World Wide	1,4,5,7,9
	51	29	ND-0.54	0.06	0.04	North America	1,4,5,9
	1	1	0.05	0.05	0.05	Europe	7
Pentachlorodibenzo-p-dioxins (MW=356.42)							
1,2,3,7,8-PeCDD	41	14	ND-0.07	0.01	0.008	World Wide	1,3,4,5,7,9
	40	13	ND-0.07	0.01	0.008	North America	1,3,4,5,9
	1	1	0.006	0.006	0.006	Europe	7
PeCDDs	53	30	ND-0.66	0.07	0.05	World Wide	1,4,5,7,9
	52	29	ND-0.66	0.07	0.05	North America	1,4,5,9
	1	1	0.11	0.11	0.11	Europe	7
Hexachlorodibenzo-p-dioxins (MW=390.87)							
1,2,3,4,7,8-HxCDD	41	14	ND-0.08	0.01	0.01	World Wide	1,3,4,5,7,9
	40	13	ND-0.08	0.02	0.01	North America	1,3,4,5,9
	1	1	0.004	0.004	0.004	Europe	7
1,2,3,6,7,8-HxCDD	41	30	ND-0.13	0.02	0.02	World Wide	1,3,4,5,7,9
	40	29	ND-0.13	0.02	0.02	North America	1,3,4,5,9
	1	1	0.008	0.008	0.008	Europe	7
1,2,3,7,8,9-HxCDD	41	26	ND-0.25	0.04	0.03	World Wide	1,3,4,5,7,9
	40	26	ND-0.25	0.04	0.03	North America	1,3,4,5,9
	1	0	ND(0.001)	0.0005	0.005	Europe	7
HxCDDs	53	46	ND-2.17	0.26	0.21	World Wide	1,4,5,7,9
	52	45	ND-2.17	0.27	0.21	North America	1,4,5,9
	1	1	0.1	0.1	0.1	Europe	7

Table B-26. Mean Background Environmental Levels of Dioxins in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Concentration range	Arithmetic mean[a]	Geometric mean[a]	Location	Ref. no.
Heptachlorodibenzo-p-dioxins (MW=425.31)							
1,2,3,4,6,7,8-HpCDD	41	36	ND-1.07	0.32	0.31	World Wide	1,3,4,5,7,9
	40	35	ND-1.07	0.33	0.32	North America	1,3,4,5,9
	1	1	0.1	0.1	0.1	Europe	7
HpCDDs	53	49	ND-2.19	0.60	0.58	World Wide	1,4,5,7,9
	52	48	ND-2.19	0.61	0.59	North America	1,4,5,9
	1	1	0.2	0.2	0.2	Europe	7
Octachlorodibenzo-p-dioxin (MW=460.76)							
OCDD	42	40	ND-29.5	2.95	2.20	World Wide	1,3,4,5,7,9
	41	39	ND-29.5	3.02	2.33	North America	1,3,4,5,9
	1	1	0.23	0.23	0.23	Europe	7
TEQs for Dibenzodioxins				0.023	0.018	World Wide	
				0.024	0.019	North America	
				0.0059	0.0059	Europe	

Footnote References

[a] Means were taken from urban sites, and pristine sites.
Industrial, sites were not used because they were assumed to be contaminated.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)

Sources:

1. Smith, et al. (1989).
3. Maisel and Hunt (1990).
4. Hunt and Maisel (1990).
5. CDEP (1988).
7. Naf, et al. (1990).
9. Edgerton, et al. (1989).

Table B-27. Mean Background Environmental Levels of Dibenzofurans in Air (pg/m³)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric Mean[a]	Location	Ref. no(s).
Tetrachlorodibenzofurans (MW=305.98)							
2,3,7,8-TCDF	40	33	ND-0.2	0.05	0.05	World Wide	1,3,4,5,9
	40	33	ND-0.2	0.05	0.05	North America	1,3,4,5,9
TCDFs	53	44	ND-2.29	0.41	0.37	World Wide	1,4,5,7,9
	52	43	ND-2.29	0.41	0.37	North America	1,4,5,9
	1	1	0.33	0.33	0.33	Europe	7
Pentachlorodibenzofurans (MW=340.42)							
1,2,3,7,8-PeCDF	40	17	ND-0.10	0.02	0.01	World Wide	1,3,4,5,9
	40	17	ND-0.10	0.02	0.01	North America	1,3,4,5,9
2,3,4,7,8-PeCDF	41	24	ND-0.16	0.03	0.02	World Wide	1,3,4,5,7,9
	40	23	ND-0.16	0.03	0.02	North America	1,3,4,5,9
	1	1	0.02	0.02	0.02	Europe	7
PeCDFs	53	44	ND-1.77	0.28	0.24	World Wide	1,4,5,7,9
	52	43	ND-1.77	0.28	0.24	North America	1,4,5,9
	1	1	0.17	0.17	0.17	Europe	7
Hexachlorodibenzofurans (MW=374.87)							
1,2,3,4,7,8-HxCDF	40	30	ND-0.41	0.06	0.06	World Wide	1,3,4,5,9
	40	30	ND-0.41	0.06	0.06	North America	1,3,4,5,9
1,2,3,6,7,8-HxCDF	41	27	ND-0.25	0.03	0.03	World Wide	1,3,4,5,7,9
	40	26	ND-0.25	0.03	0.03	North America	1,3,4,5,9
	1	1	0.008	0.008	0.008	Europe	7
1,2,3,7,8,9-HxCDF	41	5	ND-0.097	0.008	0.004	World Wide	1,3,4,5,7,9
	40	4	ND-0.097	0.009	0.005	North America	1,3,4,5,9
	1	1	0.0008	0.0008	0.0008	Europe	7
2,3,4,6,7,8-HxCDF	41	27	ND-0.30	0.05	0.04	World Wide	1,3,4,5,7,9
	40	26	ND-0.30	0.05	0.04	North America	1,3,4,5,9
	1	1	0.005	0.005	0.005	Europe	7
HxCDFs	53	47	ND-2.15	0.40	0.33	World Wide	1,4,5,7,9
	52	46	ND-2.15	0.39	0.34	North America	1,4,5,9
	1	1	0.08	0.08	0.08	Europe	7

Table B-27. Mean Background Environmental Levels Dibenzofurans in Air (pg/m³) (continued)

Chemical	Number samples	Number positive samples	Conc. range	Arithmetic mean[a]	Geometric Mean[a]	Location	Ref. no(s).
Heptachlorodibenzofurans (MW=409.31)							
1,2,3,4,6,7,8-HpCDF	41	31	ND-0.80	0.23	0.21	World Wide	1,3,4,5,7,9
	40	30	ND-0.80	0.23	0.22	North America	1,3,4,5,9
	1	1	0.09	0.09	0.09	Europe	7
1,2,3,4,7,8,9-HpCDF	38	19	ND-0.30	0.03	0.03	World Wide	3,4,5,7,9
	37	19	ND-0.30	0.03	0.03	North America	3,4,5,9
	1	0	ND(0.001)	0.0005	0.0005	Europe	7
HpCDFs	53	46	ND-1.58	0.34	0.30	World Wide	1,4,5,7,9
	52	45	ND-1.58	0.35	0.30	North America	1,4,5,9
	1	1	0.11	0.11	0.11	Europe	7
Octachlorodibenzofurans (MW=444.76)							
OCDFs	54	38	ND-0.70	0.17	0.15	World Wide	1,3,4,5,7,9
	53	37	ND-0.70	0.17	0.16	North America	1,3,4,5,9
	1	1	0.02	0.02	0.02	Europe	7
TEQs for Dibenzofurans				0.039	0.031	World Wide	
				0.039	0.032	North America	
				0.012	0.012	Europe	

Footnote References

[a] Means were taken from urban and pristine sites.
Industrial, sites were not used because they were assumed to be contaminated.

NOTES: One-half the limit of detection was used in calculating the means where applicable.

ND = non-detected (limit of detection)

Sources:

1. Smith, et al. (1989).
3. Maisel and Hunt (1990).
4. Hunt and Maisel (1990).
5. CDEP (1988).
7. Naf, et al. (1990).
9. Edgerton, et al. (1989).

APPENDIX C. SPREADSHEET ANALYSIS AND MODEL PARAMETERS

All the results given for the example settings in Chapter 9, with the exception of results associated with the Industrial Source Complex model (ISC model; see Chapter 6), were generated using spreadsheets. The purpose of this appendix is to describe the structure of the spreadsheets and to list all parameters and their values.

C.1. INTRODUCTION

Four separate spreadsheets were developed for this document. Each one corresponds to one of four source categories that are described in Chapter 5. The four source categories were termed on-site soil, off-site soil, incinerator stack emissions, and incinerator ash disposal in a landfill. These categories roughly translate to beginning points, or origins, of contamination. All spreadsheets estimate exposure, i.e., lifetime average daily doses, LADD in mg/kg-day, for all exposure pathways considered in this methodology. What is unique, therefore, for each spreadsheet are the methods used to estimate exposure media concentrations. Each spreadsheet was used to model "central" and "high end" exposure scenarios associated with each source category. This meant that generating results for the high end scenario required editing several parameters after generating results for the central scenario. All spreadsheets also had both a pond and a stream as possible sources of drinking water and fish consumption.

Section C.2 describes the structure of the spreadsheets, and Section C.3 provides tables which list all parameters, their values, and sections of the document where further discussion of the parameters appear.

C.2. SPREADSHEET STRUCTURE

Each spreadsheet has three principal sections. The first section describes the results of the assessment, which are the LADDs for each exposure pathway. The second section lists all the parameters: those associated with estimating exposure media concentrations, those associated with patterns of exposure, and those specific to chemicals. The third section contains the equations which solve for exposure media concentrations using methodologies given in Chapter 5. Example portions of these three sections for the on-site source category are given in Figure C-1.

ON-SITE SOURCE CATEGORY	Dioxin	Furan	PCB
I. RESULTS			
A. LADD in mg/kg-day			
1. Soil ingestion	8E-13	8E-13	8E-13
2. Soil dermal			
Residence scenario	4E-14	4E-14	4E-14
Farm scenario	4E-13	4E-13	4E-13
3. Vegetable ingestion	2E-14	2E-14	2E-14
II. PARAMETERS			
A. FATE AND TRANSPORT PARAMETERS			
1. Site characteristics			
* Cs: Soil concentration, mg/kg	1.00E-06	1.00E-06	1.00E-06
* ASC: Area of contamination, m^2	4.00E+04	4.00E+04	4.00E+04
B. EXPOSURE AND RISK PARAMETERS			
1. Durations and body weights			
* EDa: Exposure duration, adult, yrs	2.00E+01	2.00E+01	2.00E+01
C. CHEMICAL PARAMETERS			
* Koc: Org. carbon part. coef., L/kg	2.69E+06	5.13E+06	3.163E+07
III. EXPOSURE MEDIA CONCENTRATIONS			
A. Vapor and particulate air conc.			
* DEA: Effective diffusivity, cm^2/sec	3.97E-02	3.97E-02	3.97E-02
FLUX: Average emission flux, g/cm^2-sec	1.71E-21	6.86E-22	3.91E-21
Ambient on-site concentration, $\mu g/m^3$	4.38E-11	1.75E-11	9.96E-11

Figure C-1. Portions of the on-site source category spreadsheet.

What is also seen in Figure C-1 are features of the columnar structure of the spreadsheet. The first column, column A, is wide and describes the contents in columns B, C, and D. Columns B through D contain parameters and quantities specific to the three example compounds of the assessment. Values in Column B not tied to a specific chemical, such as the parameters associated with estimating exposure media concentrations, the exposure parameters, and others, are also repeated in Columns C and D. In this way, each of Columns B, C, and D contain the complete information to uniquely estimate exposures for each of the example compounds. Each column could also be used to develop distinct scenarios with this structural strategy.

Finally, Figure C-1 also shows the numeric format of the spreadsheets. All numbers are in scientific notation. Dose results give only one significant figure while all other quantities give three significant figures. Giving more than one significant figure for dose estimates would give the inappropriate impression of accurracy.

C.3. PARAMETER VALUES

The second section of each spreadsheet contains all parameters needed to generate exposure media concentrations and exposure estimates. That section is further subdivided into three categories of parameters: exposure media concentration parameters, exposure parameters, and chemical parameters. The exposure media parameters were identified and discussed in Chapter 3. Table C-1 lists the parameter values for each parameter and example setting. Chapter 3 describes the equations in which these parameter appears, and also the justification for the selection of the values shown in this table. The final column in Table C-1 lists the section in Chapter 3 where this information can be found. Table C-2 lists the parameters for the three example compounds demonstrated in Chapter 9. As discussed in that chapter, the example compounds include 2,3,7,8-TCDD, 2,3,4,7,8-PCDF, and 2,3,3',4,4',5,5'-PCB. Background information on the values selected for these compounds can be found in Chapters 2, 5, 6, and 9. Specific sections in these chapters are noted in Table C-2.

Table C-3 lists all the exposure-related parameters for all exposure pathways. Information on these parameters can be found principally in Chapter 7, and sections in that Chapter are noted on Table C-3. These parameters are independent of the example scenarios, with one exception. The exposure duration of 20 years was specific to a

"farmer" who was the impacted individual in the high end scenarios. The "residence" setting exposed individual was called an adult, and has an exposure duration of 9 years. This assumption is noted on Table C-3.

Table C-1. Exposure media concentration parameters for the four source categories

Description[b]	Source category[a] 1	2	3	4	Chapter 5 sections
1. Contaminated site characteristics					
ASC: area, m^2	NA	40000	NA	270000	4.1, 6.1
Psoil: particle bulk den, g/cm	NA	2.65	NA	2.65	3.2
OCsl: organic carbon fraction	NA	0.01	NA	0.01	3.1
ESLP: soil porosity	NA	0.50	NA	0.50	3.2
2. Exposure site parameters					
AES: area, "central", m^2	4000	4000	4000	4000	3.1, 4.1.,
AES: area, "high end", m^2	40000	40000	40000	40000	& Chap 9
Bsoil: soil bulk den, g/m^3	NA	1500	1500	1500	4.1
Psoil: particle bulk den, g/cm^3	2.65	NA	NA	NA	3.2
dt: tillage mixing depth, m	NA	0.20	0.20	0.20	4.1, 5.0
dnot: no-till mixing depth, m	NA	0.01	0.01	0.01	4.1, 5.0
OCsl: organic carbon fraction	0.01	0.01	0.01	0.01	3.1
ESLP: soil porosity	0.50	NA	NA	NA	3.2
Um: wind speed, m/sec	4.00	4.00	NA	4.00	3.2, 6.2
DL: distance to soil cont., m	NA	150	NA	150	4.1

(continued on the following page)

Table C-1. (continued)

Description[b]	Source category[a] 1	2	3	4	Chapter 5 sections
3. Soil and Sediment Delivery Parameters					
SLs: Soil loss, cont. site, kg/ha-yr	118800	118800	NA	118800	3.1
Cw: Watershed cont. conc., all chem, ppm	1E(-6)	0	NA	0	3.1, 4.0
OCssed: Fraction org. car., sus. sed	0.05	0.05	0.05	0.05	3.1
OCsed: Fraction org. car., bot. sed	0.02	0.02	0.02	0.02	3.1
Aw: Watershed drainage area, ha:	4000	4000	NA	4000	3.1
SDw: Watershed sed. delivery ratio	0.15	0.15	NA	0.15	3.1
SLw: Soil loss, watershed, kg/ha-yr	11880	11880	NA	11880	3.1
MXdep: Watershed mixing depth, m	NA	NA	0.10	NA	3.1
TSS: Total suspended sediment, mg/L	10	10	10	10	3.1
DL: Distance, cont. site to water, m	150	150	NA	150	3.1
4. Air modeling parameters					
z: height of individual, m	2	2	NA	2	3.2
V: fraction of veg. cover, central	0.90	0.00	NA	0.00	3.3, 4.0
V: fraction of veg. cover, high	0.50	0.00	NA	0.00	3.3, 4.0
Ut: threshold wind speed, m/sec	6.50	8.25	NA	8.25	3.3, 4.0
F(x): model-specific parameter	1.05	0.50	NA	0.50	3.3, 4.0
FREQ: frequency wind blows to site	NA	0.15	NA	0.15	4.2

(continued on the following page)

Table C-1. (continued)

Description[b]	Source category[a] 1	2	3	4	Chapter 5 sections
4. Air modeling parameters (continued)					
kunp: particle size multiplier for unpaved roads	NA	NA	NA	0.36	6.2.1
s: silt cont. of unpaved roads, %	NA	NA	NA	13.4	6.2.1
W: mean vehicle wt., kg	NA	NA	NA	27000	6.2.1
VS: mean vehicle speed, km/hr	NA	NA	NA	24	6.2.1
nw: number of wheels	NA	NA	NA	10	6.2.1
Prec: # of days/yr precip>.254 mm	NA	NA	NA	121	6.2.1
Limp: length of impacted roadway, m	NA	NA	NA	100	6.2.1
Nvp: number of vehicle passes/day	NA	NA	NA	50	6.2.1
Nwd: number of working days/yr	NA	NA	NA	365	6.2.1
CEFR: roadway control eff. factor	NA	NA	NA	0.10	6.2.1
CDF: contaminant dilution factor	NA	NA	NA	0.10	6.2.1
ASH: ash generated, kg/d	NA	NA	NA	330000	6.2.2
NDTL: daily truck loads	NA	NA	NA	8	6.2.2
fe: time wind speed > 5.4 m/sec, %	NA	NA	NA	100	6.2.2
TSA: truck surface area, m^2	NA	NA	NA	40	6.2.2
LOR: road length of truck emiss, m	NA	NA	NA	600	6.2.2
Sa: silt content of ash, %	NA	NA	NA	6.7	6.2.2
CEFT: truck control eff. factor	NA	NA	NA	0.10	6.2.2
kunl: particle size multiplier	NA	NA	NA	0.35	6.2.3
Ms: ash moisture content, %	NA	NA	NA	0.25	6.2.3

(continued on the following page)

Table C-1. (continued)

Description[b]	Source category[a] 1	2	3	4	Chapter 5 sections
4. Air modeling parameters (continued)					
ksc: particle size multiplier for spreading/compacting	NA	NA	NA	0.21	6.2.4
Dspr: depth of spread, m	NA	NA	NA	0.05	6.2.4
TVL: truck volume, m^3	NA	NA	NA	40	6.2.4
5. Bioconcentration and biotransfer parameters					
f_{lipid}: fish lipid fraction	0.07	0.07	0.07	0.07	3.4.1
FDW: fresh to dry wt conversion	0.15	0.15	0.15	0.15	3.4.2
Vp: particle deposition vel, m/yr	315360	315360	NA	315360	3.4.2
Ygr: yield of grass, kg/m^2 dry	0.15	0.15	0.15	0.15	3.4.2
INTG: grass intercept fraction	0.35	0.35	0.35	0.35	3.4.2
Yfod: cattle fodder yield, kg/m^2	0.45	0.45	0.45	0.45	3.4.2
INTfod: fodder intercept fraction	0.73	0.73	0.73	0.73	3.4.2
Yveg: vegetable yield, kg/m^2 dry	1.17	1.17	1.17	1.17	3.4.2
INTveg: vegetable intercept frac.	0.48	0.48	0.48	0.48	3.4.2
VGbg: below ground veg. correc. fact.	0.01	0.01	0.01	0.01	3.4.2

(continued on the following page)

Table C-1. (continued)

Description[b]	Source category[a] 1	2	3	4	Chapter 5 sections
5. Bioconcentration and biotransfer parameters (cont'd)					
VGveg: veg/fuit air-to-leaf corr.	0.01	0.01	0.01	0.01	3.4.2
VGgr: grass air-to-leaf correction	1.00	1.00	1.00	1.00	3.4.2
VGfod: fodder air-to-leaf corr.	0.50	0.50	0.50	0.50	3.4.2
SDFb: beef cattle soil diet frac.	0.08	0.08	.08	0.08	3.4.3
SDFd: dairy cattle soil diet frac.	0.02	0.02	0.02	0.02	3.4.3
FDFb: beef cattle fodder diet frac.	0.02	0.02	0.02	0.02	3.4.3
FDFd: dairy cattle fodder diet frac.	0.90	0.90	0.90	0.90	3.4.3
GDFb: beef cattle grass diet frac.	0.90	0.90	0.90	0.90	3.4.3
GDFd: dairy cattle grass diet frac.	0.08	0.08	0.08	0.08	3.4.3
BCGRA: fraction beef cattle grazing on contaminated land	1.00	1.00	1.00	1.00	3.4.3
DCGRA: fraction dairy cattle grazing on contaminated land	1.00	1.00	1.00	1.00	3.4.3
BCFOD: fraction beef cattle fodder grown on contaminated land	1.00	1.00	1.00	1.00	3.4.3
DCFOD: fraction dairy cattle fodder grown on contaminated land	1.00	1.00	1.00	1.00	3.4.3
Bs: soil bioavailability of cont.	0.65	0.65	0.65	0.65	3.4.3

(continued on the following page)

Table C-1. (continued)

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[a]Source Categories: 1 = on-site soil, 2 = off-site soil, 3 = incinerator stack emissions, and 4 = incinerator ash disposed of in landfill.
[b]The parameter names are the ones on the spreadsheets. Most often they match the names given in the referenced section in Chapter 5. In some instances, however, they do not match exactly or the discussions in Chapter did not assign specific parameter names--for example, the fraction of cattle grazing which occurs on contaminated land was not a named parameter in Chapter 5.
NA = not applicable; parameter was not required for noted spreadsheet.

Table C-2. Chemical parameters for the three example compounds for all settings

Description	2,3,7,8-TCDD	2,3,4,7,8-PCDF	2,3,3',4,4',5,5'-HPCB	Chapter section/table
I. Physical/chemical properties				
MW: molecular weight, g/m	322	340	396	Table A-1
log Kow[a]: log octanol water partition coeff, L/kg	6.64	6.92	7.71	2.2
H: Henry's Constant, atm-m^3/$mole^{-5}$	1.65×10^{-5}	$4.99 * 10^{-6}$	$1.00 * 10^{-3}$	2.2
Da: molecular diffusivity in air, cm^2/s	.05	.05	.05	5.3.2
II. Estimated parameters				
Koc: organic carbon partition coeff., L/kg	2691500	5128600	31622800	5.3.1
Bva: Air-to-Leaf transfer factor	640223	4033804	124112	5.3.4.2
F: beef/milk bioconcentration factor	5.0	3.0	0.5	5.3.4.3
BSAF: lipid-based biota sediment acc. factor	0.09	0.09	2.00	5.3.4.1
k: dissipation rate constant for eroding or depositing contaminants, yr^{-1}	0.0693	0.0693	0.0693	5.4.1
kw: first-order plant wash-off constant, yr^{-1}	18.02	18.02	18.02	5.3.4.2
RCF: Root bioconcentration factor	3916	6432	26103	5.3.4.2

[a]log Kow not strictly needed for any model estimations, but it was used in the estimation of several parameters, and it is otherwise a critical compound characteristic.

(continued on the following page)

Table C-3. Summary of exposure pathway parameters

Pathway description	Contact rates	Contact fractions	Chapter section
Soil ingestion			
Central	200 mg/d	1.0	7.2.1
High end	800 mg/d	1.0	
Soil dermal contact			
Central	0.2 mg/cm^2-event 5000 cm^2, 40 events/yr	1.0	7.2.2
High end	1.0 mg/cm^2-event 1000 cm^2, 350 events/yr	1.0	
Dermal Absorption	0.03		
Vapor/dust inhalation			
Central	20 m^3/day	0.75	7.2.3
High end	20 m^3/day	0.90	
Water ingestion			
Central	1.4 L/day	0.75	7.2.4
High end	1.4 L/day	0.90	
Beef fat ingestion			
Central	NA	NA	7.2.5
High end	22 g/day	0.44	
Milk fat ingestion			
Central	NA	NA	7.2.5
High end	10.5 g/day	0.40	
Fish ingestion			
Central	1.2 g/day	1.00	7.2.6
High end	4.1 g/day	1.00	

(continued on the following page)

Table C-3. (continued)

Pathway description	Contact rates	Contact fractions	Chapter section
Fruit ingestion			
Central		0.20	7.2.7
Above Ground	88 g/day		
Below Ground	0 g/day		
High end		0.30	
Above Ground	88 g/day		
Below Ground	0 g/day		
Vegetable ingestion			
Central		0.25	7.2.7
Above Ground	76 g/day		
Below Ground	28 g/day		
High end		0.40	
Above Ground	76 g/day		
Below Ground	28 g/day		

Exposure Duration: A duration of 9 years was assumed for central exposures, and a duration of 20 years was selected for the high end exposures. The soil ingestion pathway was demonstrated only for children and the exposure duration was 5 years. See Section 7.2

Body Weight/Lifetime: The standard assumptions of a 70 kg adult and 70 years lifetime were assumed for all pathways, except that of soil ingestion. In that case, a child body weight of 17 kg was used. See Section 7.2

APPENDIX D. MODELING THE IMPACT OF EFFLUENT DISCHARGES INTO SURFACE WATERS

Dioxin-like compounds have been found in the effluent discharges from various industries such as pulp and paper mills where chlorine bleaching processes are used. This potential source for releasing dioxins to the environment was not considered earlier in this document since it has been addressed in other recent Agency publications. However, in response to review comments, a brief discussion is included here summarizing possible approaches to evaluating this source.

In 1988, EPA and the paper mill industry cooperated on a study which examined discharges from five pulp and paper mills (EPA, 1988). Among other findings, that study concluded that: the principal dioxin-like compounds in effluent discharges were 2,3,7,8-TCDF and 2,3,7,8-TCDD, that 2,3,7,8-TCDF concentrations exceeded those of 2,3,7,8-TCDD by 2 to 18 times, and that once these compounds are formed, they are not altered in further processing or wastewater treatment (EPA, 1988). A follow-up study (called the "104-mill study") collected data on 2,3,7,8-TCDD and 2,3,7,8-TCDF effluent discharges from 87 kraft pulp mills and 17 sulfite pulp mills in the United States. Further details on this 104-mill EPA/Paper Industry Cooperative Dioxin Study can be found in EPA (1990a; 1990b).

EPA (1990b) used data from the 104-mill study to evaluate the impact of 2,3,7,8-TCDD and 2,3,7,8-TCDF discharges to receiving water, fish, and to humans through consumption of drinking water and fish. Two approaches were used to estimate the impact to water and fish. One was to use a simple dilution model to estimate total (soluble plus sorbed) water concentrations after the effluent mixes with the receiving water (i.e., at some point near but not directly at the point of effluent discharge). These total water concentrations were assumed to be bioavailable to fish, and also were used as concentrations for drinking water exposures. This approach was identified as representing the upper bound for bioaccumulation in fish, since the bioconcentration factors used were based on dissolved contaminant concentrations, not total concentrations. The second was to use the EXAMS II model which allowed for partitioning of these hydrophobic contaminants between suspended and benthic sediments, and dissolved in water. The same bioconcentration factors were used to estimate fish tissue concentration, but they

were applied to the dissolved phase concentration in the water column. EPA (1990b) describes all other assumptions and results of this exercise.

EPA (1992) also used the EXAMS II model in a document which describes a methodology to evaluate the impact of pulp and paper mill discharges on receiving water bodies, fish, and humans that consume the fish. The use of the EXAMS II modeling in this methodology document was different than the exercises described above. One key objective of this methodology was to identify a "Geographic Area of Potential Population Exposure", or GAE. Identification of a GAE begins with analysis of fish tissue data. This can be existing data downstream from an effluent discharge, or data specifically collected in order to evaluate the impact of the effluent discharges. Given fish tissue concentrations, a BCF can be used to backcalculate a water concentration that corresponds to this fish concentration. Then, the EXAMS II model can be used to estimate how far from the discharge point these water concentrations would occur. This distance is a principal point of definition of the GAE. This distance would vary depending on the fish concentration used - it could be the mean, median, or minimum concentration found in the fish tissue sampling program. Other site-specific characteristics of the water body in question, such as the distance fisherman will travel to fish in the water body, further define the GAE.

The approaches described above model dioxin transport in the water phase and estimate fish tissue concentrations from water concentrations using a BCF. An alternative method is to model the transport in the sediment phase and use a Bioavailability Index or BI to estimate fish concentrations.

Such models would require as input effluent discharge characterizations, water body characterizations, and would predict downstream sediment impacts. The input data pertaining to many specific water bodies is currently available through data bases such as STORET, a water quality data base maintained by EPA; GAGE, a water quantity data base supported by USGS and accessible through STORET; and the REACH file within GAGE, which gives flow data for specific reaches within rivers. Effluent discharge data for 2,3,7,8-TCDD and 2,3,7,8-TCDF are summarized for the 104-mill study in Appendix C of EPA (1990b). Although qualified as only a 5-day temporal sample from the 104 mills, this does provide for all mills: total suspended sediment concentrations in effluent discharges (mg/L), effluent flow rate (MGD), 2,3,7,8-TCDD and 2,3,7,8-TCDF total concentrations

(ppq total concentration) and total load (kg/hr), and receiving water body harmonic mean flow and 7Q10 low flow (m^3/hr).

This data could be used in a simplistic sediment dilution model similar to that used in this assessment to estimate bottom and suspended sediment concentrations resulting from erosion of contaminated and uncontaminated land within a watershed.

Specifically, the effluent data allows one to estimate the input of contaminated sediment to the river by this effluent stream. River flows and information about the watershed draining into the water body allow one to estimate the input of residue-free eroded sediment, or sediment with a known or estimated concentration of 2,3,7,8-TCDD or 2,3,7,8-TCDF. The following summarizes an example of a sediment dilution approach in a four-step procedure. This example is based on the first entry in Table C.1 (Appendix C of EPA (1990b)):

- **Estimate the concentration of TCDD on effluent suspended sediment discharges:** First, a reasonable assumption to make is that all the discharged 2,3,7,8-TCDD is associated with suspended sediment in the effluent discharge. The effluent is discharged at a rate of 23 MGD, it has a total suspended sediments concentration of 248.9 mg/L, and a total 2,3,7,8-TCDD concentration of 0.0068 pg/L (ppt, 6.8 ppq). The TCDD load to the river is also listed, but can be calculated from the concentration and discharge rate, as 2.5×10^{-8} kg/hr. A concentration on that sediment can be estimated by first estimating the rate of discharge of sediments, which for this example is 903 kg/hr, and then inverting and multiplying by the 2,3,7,8-TCDD discharge rate. The resulting concentration, (2.5×10^{-8} kg TCDD/hr)/(903 kg sus. sed/hr), equals 28 ng/kg (ppt).

- **Estimate the total sediment load contributed by the effluent discharge:** A total sediment load to the river in this example equals 7.9×10^6 kg/yr (903 kg/hr * 24 hr/day * 365 day/yr). The simple sediment dilution model assumes that this will mix with all other sediment inputs to the river which occur downstream of the effluent discharge point.

- **Estimate the total sediment load contributed by other sources to the water body:** The example scenarios in Chapter 9 assumed a rural setting and that erosion from the watershed contributed mixtures of residue-free and contaminated soils. Further, the

examples assumed that the contaminated land area was at the top of watershed; hence, the contributions from the contaminated site to the river mixed with the contributions from the watershed as a whole. The same assumption will be used for this example. Site-specific information on the watershed size and erodibility can be obtained, but assumptions will be made for this example. First, the size of the receiving water body can be used to roughly estimate the watershed size, if that information is unavailable, as it is for this example. The harmonic mean flow of the receiving water body is given as 184716 m^3/hr, which converts to 1811 ft^3/sec. An average runoff rate over a land area can be translated to a stream flow rate. It was shown in Chapter 10, Section 10.2.4.2, that an average runoff rate of 15 inches/year over a 10,000 acre watershed - the size of the watershed in the example scenarios - translates to a stream flow of 17.2 ft^3/sec. The much higher flow rate of 1811 ft^3/sec backcalculates to a watershed size just over 1,000,000 acres (assuming 15 in/year runoff). Next, an erosion rate is needed, and a rate of 6 T/ac-yr will be assumed for this example - this was also the average watershed erosion rate used in the example scenarios. Finally, a sediment delivery ratio will reduce the total erosion to an amount which reaches the river. From Figure 5.1, a drainage area of 1,000,000 (which equals just over 4000 km^2) corresponds to a delivery ratio of about 0.04. The total sediment contributed by overland erosion from a 1,000,000 acre watershed is: (1,000,000 ac) * (6 T/ac-yr) * 0.04 * 906 kg/T = 2.2×10^8 kg/yr.

- **Determine the sediment concentrations using a simple dilution model:** This can be estimated using a dilution ratio approach similar to the approach given in Chapter 5, Section 5.3.1:

$$C_{sed} = \frac{(C_{eff}\ SED_{eff}) + (C_w\ SED_w)}{SED_{eff} + SED_w} \qquad \text{(D-1)}$$

where:

C_{sed} = bottom and suspended sediment concentration, ppt

C_{eff} = effluent sediment concentration, ppt

SED_{eff} = sediment contribution by effluent discharge, kg/yr

C_w = average concentration of dioxin-like compound in soil eroding into

water body, ppt

SED_w = sediment contribution from watershed, kg/yr

From the example discussed above, C_{eff} = 28 ppt 2,3,7,8-TCDD, SED_{eff} = 7.9×10^6 kg/yr, C_w can be assumed to be 0.0 which would estimate the increment of sediment concentration due to the discharge, and SED_w = 2.2×10^8 kg/yr. C_{sed} is solved for as 0.97, or 1 ppt. The dilution from pipe sediment concentrations to average river sediment concentrations within the example watershed is 3.5% (1 ppt/28 ppt expressed in percent). If a background watershed soil concentration were assumed to be 1 ppt, then the sediment in the river is estimated as 2.0 ppt; i.e., the effluent discharge effectively doubles the background sediment concentrations within the water body.

Conceptually this approach mirrors the one presented earlier in this document for estimating the impact to a nearby water body from an area of contaminated land. However, it could be argued that an effluent discharge, as a point source modeling problem, is amenable to a more sophisticated approach as compared to nonpoint erosion modeling. For effluent discharges, perfect mixing throughout a river system is clearly an oversimplification - to a lesser extent, the same could be said for erosion from a relatively small tract of contaminated land within a larger watershed. One empirical approach would be to estimate the average concentration with this or a similar dilution approach, but then assume a sediment concentration distribution around this average as a function of distance from the pipe. Continuing the example above, instead of an average 3.5% dilution ratio throughout the river, the dilution would be higher near the pipe discharge at 10 or 20% and lower at great distances, say 1% or less. The shape of such a distribution is unknown at the current time.

Currently available models such as EXAMS II can be used to estimate bottom sediment concentrations resulting from pipe discharges, although these equilibrium partitioning models can also introduce uncertainties. Hanna, et al. (1992), using relationships in Karichoff and Morris (1985), estimate that the time to equilibrium for dioxin in a suspended sediment/water system is several months. Therefore, dioxin-like compounds associated with suspended sediment in effluent discharges would likely reach bottom sediments before they would equilibrate and partition partly into the water column into a dissolved phase. The EXAMS II model assumes instantaneous equilibrium once

introduced into the water. It is not clear how much error is introduced with this assumption of instantaneous equilibrium. Hanna, et al. (1992) also note that an important parameter in an idealized "pipe-to-sediment" model is the size distribution of suspended sediment particles in effluents. If they are predominantly large, for example, they may not travel far from the point of discharge. Hanna, et al. (1992) further state that a generalized pollutant fate and transport model "must include: convection/dispersion in receiving water; particle sorption/desorption in sediment and water column; settling, resuspension, and burial of particulates; diffusive exchange between water column, active sediments, and deep sediments; pollutant sources via wet/dry deposition, point and nonpoint sources; and sinks, e.g., volatilization, and chemical/biochemical reactions". Finally, Hanna, et al. (1992) note that the fate and transport model could use a mixing zone analysis similar to that of Holley and Jirka (1986). The problem with the analysis of Holley and Jirka (1986) is that it is specific to totally soluble contaminants and would require empirical or theoretical corrections for transport and fate of contaminants sorbed to sediments.

In summary, it appears that sophisticated "pipe-to-sediment" models have not yet been well developed but may offer conceptual advantages over the classical water phase based approaches.

REFERENCES

Hanna, L.M., E.J. Bouwer, and D.W. Smith. 1992. An Approach to Estimating Effluent Limits for Hydrophobic Compounds Through Transport and Fate Considerations. Summary of presentation made at a conference titled, National Conference on Bioavailability of Dioxin, PCBs and Metals in Aquatic Ecosystems. Sponsored by BCM Engineers, Inc. and Rifkin and Associates, May 14-15, Loews L'enfant Plaza Hotel, Washington, D.C. Authors are from BCM Engineers, Inc., One Plymouth Meeting, Plymouth Meeting, PA 19462.

Holley, E.R. and G.H. Jirka. 1986. "Mixing in Rivers". Technical Report E-86-11, US Army Corp of Engineers, US Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Karickhoff, S.W. and K.R. Morris. 1985. Sorption Dynamics of Hydrophobic Pollutants in Sediment Suspensions. Environmental Toxicology and Chemistry 4: 469-479.

U.S. EPA. 1988. U.S. EPA/Paper Industry Cooperative Dioxin Screening Study. Office of Water Regulations and Standards, U.S. Environmental Protection Agency, Washington, D.C. EPA-440/1-88-025. March, 1988.

U.S. EPA. 1990a. Integrated Risk Assessment for Dioxins and Furans from Chloring Bleaching in Pulp and Pepr Mills. Office of Toxic Substances, U.S. Environmental Protection Agency, Washington, D.C. EPA 560/5-90-011.

U.S. EPA. 1990b. Risk Assessment for 2378-TCDD and 2378-TCDF Contaminated Receiving Waters from U.S. Chlorine-Bleaching Pulp and Paper Mills. Prepared by Tetra Tech, Inc., 10306 Eaton Place, Suite 340, Fairfax, VA 22030., Contract #68-C9-0013.

U.S. EPA. 1992. A Methodology for Estimating Population Exposures from the Consumption of Chemically Contaminated Fish. EPA/600/9-91/017 (Draft final as of 1/13/92).

☆ U.S. GOVERNMENT PRINTING OFFICE: 1992—650 938/60904