United States
Environmental Protection
Agency

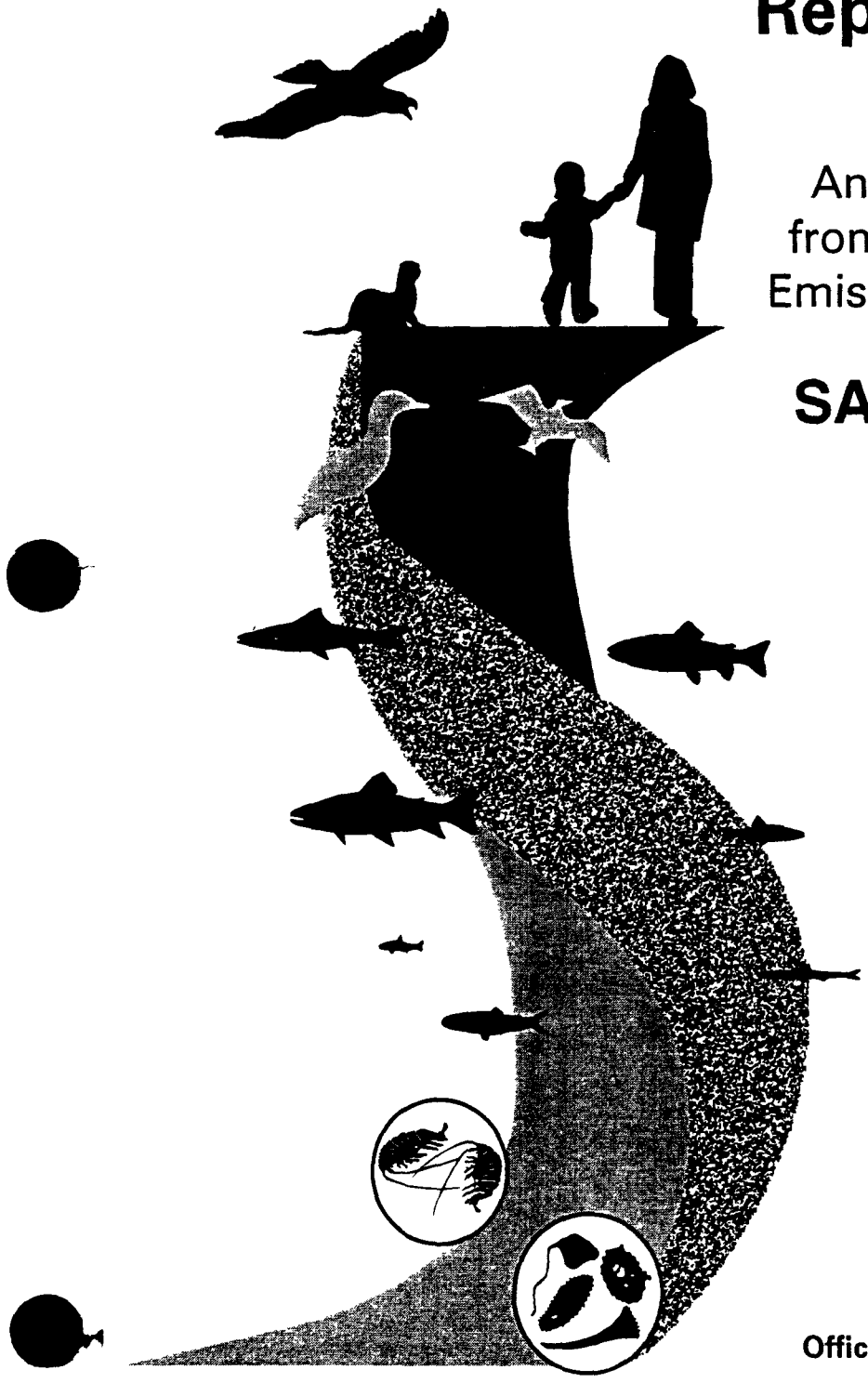
EPA-452/R-96-001c
April 1996

Air

Mercury Study Report to Congress

Volume III: An Assessment of Exposure from Anthropogenic Mercury Emissions in the United States

SAB REVIEW DRAFT



EPA

Office of Air Quality Planning & Standards
and
Office of Research and Development

MERCURY STUDY REPORT TO CONGRESS

VOLUME III:

AN ASSESSMENT OF EXPOSURE FROM ANTHROPOGENIC MERCURY EMISSIONS IN THE UNITED STATES

SAB REVIEW DRAFT

June 1996

U.S. Environmental Protection Agency
Region 5, Library (PL-12J)
77 West Jackson Boulevard, 12th Floor
Chicago, IL 60604-3590

Office of Air Quality Planning and Standards
and
Office of Research and Development

U.S. Environmental Protection Agency

TABLE OF CONTENTS

Page

U.S. EPA AUTHORS v
SCIENTIFIC PEER REVIEWERS vi
WORK GROUP AND U.S. EPA/ORD REVIEWERS vii
LIST OF TABLES viii
LIST OF FIGURES xiii
LIST OF SYMBOLS, UNITS AND ACRONYMS xv

EXECUTIVE SUMMARY ES-1

1. INTRODUCTION 1-1
   1.1 Long Range Atmospheric Transport Modeling 1-8
   1.2 Local Atmospheric Transport Modeling 1-8
   1.3 Modeling of Exposure Through Terrestrial and Aquatic Fate and Transport Models 1-8
   1.4 Exposure Modeling Rationale 1-9
   1.5 Factors Important in Estimation of Mercury Exposure 1-10
   1.6 Estimated Human Exposure through the Consumption of Fish 1-11
   1.7 Definition of Terms 1-12

2. CHEMICAL PROPERTIES AND MEASURED ENVIRONMENTAL CONCENTRATIONS OF MERCURY 2-1
   2.1 Chemical Properties of Mercury 2-1
   2.2 Analytic Measurement Methods 2-1
   2.3 Mercury in the Environment 2-2
      2.3.1 Emissions of Mercury 2-2
      2.3.2 Deposition of Mercury 2-4
      2.3.3 Mercury in Soil 2-7
      2.3.4 Plant and Animal Uptake of Mercury 2-7
      2.3.5 Mercury in the Freshwater Ecosystem 2-8
      2.3.6 Summary 2-9
   2.4 Measurement Data 2-10
      2.4.1 Mercury Air Concentrations 2-10
      2.4.2 Mercury Concentrations in Precipitation 2-12
      2.4.3 Mercury Deposition Rates 2-12
      2.4.4 Mercury Concentrations in Water 2-16
      2.4.5 Mercury Concentrations in Soil 2-18
      2.4.6 Mercury Concentrations in Biota 2-19
   2.5 Measurement Data from Remote Locations 2-34
      2.5.1 Elevated Atmospheric Mercury Concentrations over Remote Locations 2-34
      2.5.2 Elevated Soil Mercury Concentrations in Locations Remote from Emission Sources 2-34
      2.5.3 Elevated Mercury Concentrations in Aquatic Sediments and Fish from Remote Water Bodies 2-35
   2.6 Measurement Data Near Anthropogenic Sources of Concern 2-35
      2.6.1 Municipal Waste Combustors 2-35

TABLE OF CONTENTS (continued)

Page

2.6.2 Chlor-Alkali Plants 2-37
2.6.3 Coal-Fired Utilities 2-37
2.6.4 Mercury Mines 2-38
2.6.5 Mercury Near Multiple Local Sources 2-38
2.6.6 Conclusion of Mercury Measurements Data 2-39
3. INFORMATION ON EXPOSURE TO MERCURY 3-1
3.1 Nonoccupational Exposures to Mercury 3-1
3.1.1 Dietary Mercury 3-1
3.1.1.1 Mercury In Food Sources Other Than Fish 3-1
3.1.1.2 Mercury from Fish 3-4
3.1.1.3 Other Estimates of Human Mercury Intake from Fish 3-6
3.1.2 Dental Amalgams 3-11
3.2 Occupational Exposures to Mercury 3-12
3.3 Estimated Wildlife Exposure to Mercury 3-13

4. MODELING THE FATE OF MERCURY RELEASED TO THE ATMOSPHERE FROM COMBUSTION AND INDUSTRIAL SOURCES 4-1
4.1 Description of Models 4-1
4.1.1 Estimating Impacts from Regional Anthropogenic Sources of Mercury . . . 4-1
4.1.2 Estimating Impacts from Local Anthropogenic Sources of Mercury 4-2
4.1.3 Estimating Environmental Concentrations 4-2
4.1.4 Method of Estimation of MethylMercury Concentration in Freshwater Fish 4-4
4.2 Description of Modeling Scenarios 4-4
4.2.1 Description of Hypothetical Sites and Watersheds 4-5
4.2.2 Description of Hypothetical Exposure Scenarios 4-5
4.2.2.1 Summary of Exposure Parameter Values 4-7
4.2.2.2 Description of Hypothetical Rural and Urban Exposure Scenarios 4-10
4.2.2.3 Description of Hypothetical Human Exposure Scenarios for Individuals Using Water Bodies 4-10
4.2.2.4 Description of Hypothetical Exposure Scenarios for Piscivorous Birds and Mammals using Water Bodies 4-12
4.3 Indirect Exposure Modeling Results Using Measured Mercury Air Concentrations and Deposition Rates 4-13
4.3.1 Mercury Concentrations Predicted for Water Bodies 4-13
4.3.2 Mercury Concentrations Predicted in Biota by IEM2 Modeling Using Measured Concentrations 4-16
4.3.2.1 Concentrations in Fish 4-16
4.3.2.2 Concentrations in Other Biota 4-18
4.3.3 Results for Hypothetical Exposure Scenarios 4-20
4.3.3.2 Predicted Methylmercury Intakes for Wildlife Receptors 4-26
4.4 Summary of IEM2 Model Results 4-26

TABLE OF CONTENTS (continued)

Page

5. LONG RANGE TRANSPORT ANALYSIS 5-1
5.1 Description of the Analytic Approach 5-1
5.1.1 Objectives 5-1
5.1.2 Description of the Long-range Transport Model Used 5-1
5.1.3 Description of the Mercury Emissions Data Used 5-2
5.2 An Interpretive Analysis of the Results 5-9
5.2.1 Mass Balances of Mercury within the Long-range Model Domain 5-9
5.2.2 Qualitative Description of Mercury Concentration Results 5-11
5.2.3 Description of Mercury Wet Deposition Simulation Results 5-14
5.2.4 Qualitative Description of Mercury Dry Deposition Results 5-25
5.2.5 Qualitative Description of Total Mercury Deposition Results 5-28
5.3 General Data Interpretations of the RELMAP Modeling 5-32
5.4 Potential Impacts of Long Range Transport 5-35
5.4.1 Description of Hypothetical Sites and Watersheds 5-36
5.4.2 Description of Hypothetical Exposure Scenarios 5-36
5.4.3 Results of IEM2 Modeling in the Long Range Transport Analysis 5-41
5.4.4 Summary of Potential Impacts of Long Range Transport 5-43

6. LOCAL IMPACT ANALYSIS 6-1
6.1 Description of Approach 6-1
6.1.1 Rationale and Utility of Model Plant Approach 6-1
6.1.2 Phase and Oxidation State of Emitted Mercury 6-2
6.1.3 Modeling the Deposition of Mercury 6-3
6.1.4 Development and Description of Model Plants 6-4
6.1.5 Hypothetical Locations of Model Plants 6-6
6.1.6 Hypothetical Exposure Scenarios and Location of Receptors Relative to Local Source 6-6
6.2 Results of Local Scale Modeling 6-6
6.2.1 Air Concentrations 6-7
6.2.2 Deposition Rates 6-10
6.2.3 Media Concentrations 6-13
6.2.4 Human Intake 6-30
6.2.5 Exposure for Wildlife Receptors 6-30
6.2.6 Mass Balances within the Local-Scale Domain 6-39
6.2.7 Summary of Local Impact Analysis Results 6-43
6.2.8 Additional Analysis Based on Emission Guidelines for Existing MWCs and New Source Performance Standards 6-46
6.3 Results of Combining Local and Regional Models 6-50
6.3.1 Air Concentrations, Deposition, and Water Concentrations 6-51
6.3.2 Human Exposure 6-51
6.3.3 Issues Related to Combining the Modeled Exposure Estimates with Estimates of Exposure from Other Sources 6-68
6.3.4 Wildlife Exposure 6-68
6.3.5 Combining the Results of Local and Regional Models: Summary Conclusions 6-71
6.4 Uncertainty and Sensitivity Analyses 6-71

TABLE OF CONTENTS (continued)

		Page
6.4.1	Dry Deposition	6-72
6.4.2	Wet Deposition	6-73
6.4.3	Effect of Terrain on Results of Local Scale Modeling	6-74

TABLE OF CONTENTS (continued)

		Page
7.	CONCLUSIONS	7-1
8.	RESEARCH NEEDS	8-1
9.	REFERENCES	9-1
APPENDIX A	PARAMETER JUSTIFICATIONS SCENARIO INDEPENDENT PARAMETERS	A-1
APPENDIX B	PARAMETER JUSTIFICATIONS SCENARIO-DEPENDENT PARAMETERS	B-1
APPENDIX C	MERCURY PARTITION COEFFICIENT CALIBRATIONS	C-1
APPENDIX D	DESCRIPTION OF EXPOSURE MODELS	D-1
APPENDIX E	CHEMICAL PROPERTIES OF MERCURY	E-1
APPENDIX F	DESCRIPTION OF MODEL PLANTS	F-1
APPENDIX G	SUMMARY OF PREDICTED CONCENTRATIONS FOR ALL FACILITIES	G-1
APPENDIX H	ESTIMATION OF HUMAN METHYLMERCURY EXPOSURE TO THE GENERAL UNITED STATES POPULATION AND IDENTIFIED SUBPOPULATIONS THROUGH THE CONSUMPTION OF FISH	H-1

U.S. EPA AUTHORS

Principal Authors:

Glenn E. Rice
National Center for Environmental Assessment-Cincinnati
Office of Research and Development
Cincinnati, OH

Robert B. Ambrose, Jr., P.E.
Ecosystems Research Division
National Exposure Research Laboratory
Athens, GA

O. Russell Bullock
Atmospheric Sciences Modeling Division
Air Resources Laboratory
National Oceanic and Atmospheric Administration
Research Triangle Park, NC
on assignment to the
U.S. EPA National Exposure Research Laboratory

Contributing Authors:

William G. Benjey, Ph.D.
Atmospheric Sciences Modeling Division
Air Resources Laboratory
National Oceanic and Atmospheric Administration
Research Triangle Park, NC
on assignment to the
U.S. EPA National Exposure Research Laboratory

Terry Clark, Ph.D.[a]
Atmospheric Sciences Modeling Division
Air Resources Laboratory
National Oceanic and Atmospheric Administration
Research Triangle Park, NC

David H. Cleverly
National Center for Environmental Assessment
Office of Research and Development
Washington, DC

Stanley Durkee
Office of Research and Science Integration
Washington, DC

Martha H. Keating
Office of Air Quality Planning and Standards
Research Triangle Park, NC

James D.Kilgroe, Ph.D.
National Environmental Research Laboratory
Office of Research and Development
Research Triangle Park, NC

Kathryn R. Mahaffey, Ph.D.
National Center for Environmental Assessment-Cincinnati
Office of Research and Development
Cincinnati, OH

John W. Nichols, Ph.D.
Mid-Continent Ecology Division
Office of Research and Development
Duluth, MN

Rita Schoeny, Ph.D.
National Center for Environmental Assessment-Cincinnati
Office of Research and Development
Cincinnati, OH

Jeff Swartout
National Center for Environmental Assessment-Cincinnati
Office of Research and Development
Cincinnati, OH

[a] Deceased

SCIENTIFIC PEER REVIEWERS

Brian J. Allee, Ph.D.
Harza Northwest, Incorporated

Thomas D. Atkeson, Ph.D.
Florida Department of Environmental Protection

Steven M. Bartell, Ph.D.
SENES Oak Ridge, Inc.

Mike Bolger, Ph.D.
U.S. Food and Drug Administration

James P. Butler, Ph.D.
University of Chicago
Argonne National Laboratory

Rick Canady, Ph.D.
Agency for Toxic Substances and Disease Registry

Rufus Chaney, Ph.D.
U.S. Department of Agriculture

Tim Eder
Great Lakes Natural Resource Center
National Wildlife Federation for the States of Michigan and Ohio

William F. Fitzgerald, Ph.D.
University of Connecticut
Avery Point

Robert Goyer, Ph.D.
National Institute of Environmental Health Sciences

George Gray, Ph.D.
Harvard School of Public Health

Terry Haines, Ph.D.
National Biological Service

Joann L. Held
New Jersey Department of Environmental Protection & Energy

Gerald J. Keeler, Ph.D.
University of Michigan
Ann Arbor

Leonard Levin, Ph.D.
Electric Power Research Institute

Malcom Meaburn, Ph.D.
National Oceanic and Atmospheric Administration
U.S. Department of Commerce

Paul Mushak, Ph.D.
PB Associates

Jozef M. Pacyna, Ph.D.
Norwegian Institute for Air Research

Ruth Patterson, Ph.D.
Cancer Prevention Research Program
Fred Gutchinson Cancer Research Center

Donald Porcella, Ph.D.
Electric Power Research Institute

Charles Schmidt
U.S. Department of Energy

Pamela Shubat, Ph.D.
Minnesota Department of Health

Alan H. Stern, Dr.P.H.
New Jersey Department of Environmental Protection & Energy

Edward B. Swain, Ph.D.
Minnesota Pollution Control Agency

M. Anthony Verity, M.D.
University of California
Los Angeles

WORK GROUP AND U.S. EPA/ORD REVIEWERS

Core Work Group Reviewers:

Dan Axelrad, U.S. EPA
Office of Policy, Planning and Evaluation

Angela Bandemehr, U.S. EPA
Region 5

Jim Darr, U.S. EPA
Office of Pollution Prevention and Toxic Substances

Thomas Gentile, State of New York
Department of Environmental Conservation

Arnie Kuzmack, U.S. EPA
Office of Water

David Layland, U.S. EPA
Office of Solid Waste and Emergency Response

Karen Levy, U.S. EPA
Office of Policy Analysis and Review

Steve Levy, U.S. EPA
Office of Solid Waste and Emergency Response

Lorraine Randecker, U.S. EPA
Office of Pollution Prevention and Toxic Substances

Joy Taylor, State of Michigan
Department of Natural Resources

U.S. EPA/ORD Reviewers:

Robert Beliles, Ph.D., D.A.B.T.
National Center for Environmental Assessment
Washington, DC

Eletha Brady-Roberts
National Center for Environmental Assessment
Cincinnati, OH

Annie M. Jarabek
National Center for Environmental Assessment
Research Triangle Park, NC

Matthew Lorber
National Center for Environmental Assessment
Washington, DC

Susan Braen Norton
National Center for Environmental Assessment
Washington, DC

Terry Harvey, D.V.M.
National Center for Environmental Assessment
Cincinnati, OH

LIST OF TABLES

		Page
1-1	Factors Potentially Important in Estimating Mercury Exposure and How They are Addressed in This Study	1-11
2-1	Annual Estimates of Mercury Release by Various Combustion and Manufacturing Source Classes (U.S. EPA, 1996a)	2-5
2-2	Summary of Measured Mercury Concentration in Air (U.S. EPA, 1993)	2-11
2-3	Measured Vapor- and Particulate-Phase Atmospheric Mercury Concentrations	2-11
2-4	Measured Mercury Concentrations in Precipitation	2-13
2-5	Measured Mercury Concentrations in Rain Which Include Methylmercury Estimates (ng/L)	2-14
2-6	Mercury Wet Deposition Rates (ug/m^2/yr)	2-15
2-7	Estimated Mercury Total Deposition Rates	2-16
2-8	Measured Mercury Concentrations in Surface Fresh Water (ng/L)	2-17
2-9	Measured Mercury Concentrations in Ground/Drinking Water (ng/L)	2-17
2-10	Measured Mercury Concentrations in Ocean Water (ng/L)	2-18
2-11	Measured Mercury Concentration in Soil	2-18
2-12	Measured Mercury Concentrations in Aquatic Sediment	2-19
2-13	Freshwater Fish Mercury Concentrations from Nationwide Studies	2-22
2-14	Measured Mercury Concentrations Freshwater Sportfish (Total Mercury, µg/g wet wt.)	2-24
2-15	Measured Mercury Concentrations in Saltwater Commercial Fish (µg/g wet wt.)	2-26
2-16	Mercury Concentrations in Marine Finfish	2-27
2-17	Mercury Concentrations in Marine Shellfish	2-28
2-18	Mercury Concentrations in Marine Molluscan Cephalopods	2-29
2-19	Measured Mercury Concentration in Meats	2-30
2-20	Measured Mercury Concentrations in Garden Produce/Crops	2-31
2-21	Mean Background Total Mercury Levels for Plants in the Netherlands (Wiersma et al., 1986)	2-31
2-22	Range of Mercury Concentrations in Selected Grain Products	2-32
2-23	Measured Total Mercury Concentrations in Human Hair and Blood	2-32
2-24	Measured Mercury Concentrations in Breast Milk	2-33
2-25	Measured Total Mercury Concentrations in Piscivorus Wildlife	2-34
2-26	Mercury Concentrations in the Atmosphere and Mercury Measured in Rainwater Collected in Broward County, FL	2-39
2-27	Mercury Concentrations Measured at Two Sites in the Atmosphere Over Detroit, MI	2-39
3-1	Estimated Average Adult Daily Intake (and retention) of Mercury Compounds by the General Public (ug/day) as Reported by WHO 1990	3-1
3-2	Reported Total Adult Mercury Intake Rates (ug/day)	3-2
3-3	Total Mercury Levels in Various Food Groups from Podrebarac (1984)	3-3
3-4	Fish and Shellfish Consumption Rates and Predicted Methylmercury Doses for Respondents of the 1989-1991 CSFII Survey	3-7
3-5	Average Serving Size (gms) for Seafood from USDA Handbook #1 Used to Calculate Fish Intake by FDA (1978)	3-8
3-6	Fish Species and Number of Persons Using the Species of Fish (Adapted from Rupp et al. 1980)	3-9
3-7	Fish Consumption from the NPD 1973/1974 Survey (Modified from Rupp et al. 1980)	3-10
3-8	Distribution of Fish Consumption for Females By Age* Consumption Category (grams/day) (from SRI, 1980)	3-10

LIST OF TABLES (continued)

		Page
3-9	Estimates of Mercury Exposure Through the Consumption of Fish (Cramer 1994)	3-11
3-10	Occupational Standards for Airborne Mercury Exposure	3-13
3-11	Assumed Fish Consumption Rates and Body Weights for Piscivorous Wildlife	3-14
3-12	Estimates of Current U.S. Methylmercury Exposure from Fish Consumption by Piscivorous Birds and Mammals	3-15
4-1	Models Used to Predict Mercury Air Concentrations, Deposition Fluxes, and Environmental Concentrations	4-1
4-2	Primary Modifications Made to COMPDEP for Exposure Assessment	4-2
4-3	Summary of Human Exposure Scenarios	4-6
4-4	Potential Dependency of Exposure Parameters	4-7
4-5	Default Values of Scenario-Independent Exposure Parameters	4-8
4-6	Values for Scenario-Dependent Exposure Parameters	4-9
4-7	Fish Consumption Rates for Columbia River Tribes	4-11
4-8	Fish Consumption Rates for Columbia River Tribes: Adults	4-11
4-9	Fish Consumption Rates used in this Study	4-12
4-10	Fish Consumption Rates for Piscivorous Birds and Mammals (from U.S. EPA, 1993)	4-12
4-11	Values Assumed for Demonstration Application of Water body and Exposure Models	4-13
4-12	Comparison of Watershed Erosion Characteristics for the Hypothetical Watersheds	4-15
4-13	Predicted Surface Water and Benthic Sediment Concentrations for the Hypothetical Water Bodies Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-16
4-14	Predicted Methylmercury Concentrations in Fish (ug/g) for the Hypothetical Water Bodies Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-17
4-15	Percentiles of Predicted Methylmercury Concentrations in Fish (ug/g) based on a Total Mercury Dissolved Water Concentration of 0.7 ng/L	4-18
4-16	Predicted Dry Weight Plant Concentrations Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-19
4-17	Predicted Dry Weight Livestock Mercury Concentrations Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-21
4-18	Predicted Human Exposure Rates for Hypothetical Receptors Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-22
4-19	Divalent Mercury Exposure for Hypothetical Receptors Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-23
4-20	Methylmercury Exposure for Hypothetical Receptors Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-24
4-21	Predicted Methylmercury Intake for Wildlife Receptors Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-27
5-1	Emission Speciation Profiles for the Point Source Types Defined	5-3
5-2	Mercury Emissions Inventory Totaled by Source Type Using Base-case Emission Speciation Profiles (metric tons per year)	5-4
5-3	Modeled Mercury Mass Budget in Metric Tons for 1989 Using the Base-Case	

LIST OF TABLES (continued)

		Page
	Emission Speciation Profiles	5-9
5-4	Modeled Mercury Mass Budget in Metric Tons for 1989 Using the Alternate Emission Speciation Profiles	5-10
5-5	Percentile Analysis of RELMAP Simulated Concentration Results for the Continental U.S. Using the Base-Case Emissions Speciation	5-16
5-6	Percentile Analysis of RELMAP Simulated Wet Deposition for the Continental U.S. Using the Base-Case Emissions Speciation	5-25
5-7	Percentile Analysis of RELMAP Simulated Dry Deposition for the Continental U.S. Using the Base-Case Emission Speciation	5-26
5-8	Percentile Analysis of RELMAP Simulated Total Depositions for the Continental U.S. Using the Base-Case and Alternate Emission Speciations	5-32
5-9	Summary of Human Exposure Scenarios Considered	5-37
5-10	Default Values of Scenario-Independent Exposure Parameters	5-38
5-11	Default Values for Scenario-Dependent Exposure Parameters	5-39
5-12	Fish Consumption Rates used in this Study (Columbia River Inter-Tribal Commission, 1994)	5-40
5-13	Fish Consumption Rates for Piscivorous Birds and Mammals (from U.S. EPA, 1993)	5-41
5-14	Predicted Media Concentrations using RELMAP Results Only in Conjunction with IEM2	5-41
5-15	Summary of Receptor Intakes using RELMAP Eastern Site Values	5-45
5-16	Summary of Receptor Intakes using RELMAP Western Site Values	5-46
5-17	Predicted Intakes of Wildlife Receptors based on RELMAP Results	5-47
6-1	Representative Particle Sizes and Size Distribution Assumed for Divalent Mercury Particulate Emissions	6-3
6-2	Process Parameters for Model Plants	6-5
6-3	Comparison of Assumed Deposition Parameters for Elemental and Divalent Mercury	6-11
6-4	Breakdown of Total Mercury Deposition Rates Predicted by COMPDEP	6-12
6-5	Predicted Human Mercury Intakes for Agricultural Scenarios in Eastern Site based on COMPDEP Results Alone	6-31
6-6	Predicted Human Mercury Intakes for Agricultural Scenarios in Western Site based on COMPDEP Results Alone	6-32
6-7	Predicted Human Mercury Intakes for Urban Scenarios in Eastern Site based on COMPDEP Results Alone	6-33
6-8	Predicted Human Mercury Intakes for Urban Scenarios in Western Site based on COMPDEP Results Alone	6-34
6-9	Predicted Human Mercury Intakes for Fish Ingestion Scenarios in Eastern Site based on COMPDEP Results Alone	6-35
6-10	Predicted Human Mercury Intakes for Fish Ingestion Scenarios in Western Site based on COMPDEP Results Alone	6-36
6-11	Predicted Intakes for Wildlife Receptors for the Eastern Site based on COMPDEP Results	6-37
6-12	Predicted Intakes for Wildlife Receptors for the Western Site Based on COMPDEP Results	6-38
6-13	Mass Balance of Mercury Emissions for each Facility in a Humid Site	6-40
6-14	Mass Balance of Mercury Emissions for each Facility in an Arid Site	6-41
6-15	Precipitation Frequencies at the Humid and Arid Sites	6-42

LIST OF TABLES (continued)

		Page
6-16	Area-Averaged Mercury Deposition Rates for each Facility in a Humid Site	6-44
6-17	Area-Averaged Mercury Deposition Rates for each Facility in an Arid Site	6-45
6-18	Process Parameters for Municipal Waste Combustor Model Plants After Imposition of MACT Standards	6-48
6-19	Mercury Concentrations Predicted in Media and Biota As a Result of Mercury Emissions From Municipal Waste Combustors After Imposition of the MACT Standards in the Hypothetical Western Site	6-49
6-20	Mercury Concentrations Predicted in Media and Biota As a Result of Mercury Emissions From Municipal Waste Combustors After Imposition of the MACT Standards in the Hypothetical Eastern Site	6-50
6-21	Combination of Local and Regional Impacts: Contribution of Regional Sources to Key Output at Eastern Site	6-52
6-22	Combination of Local and Regional Impacts: Contribution of Regional Sources to Key Output at Western Site	6-53
6-23	Combination of Local and Regional Impacts: Predicted Mercury Inhalation Intake at Eastern Site	6-54
6-24	Combination of Local and Regional Impacts: Predicted Mercury Inhalation Intake at Western Site	6-55
6-25	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Subsistence Farmer at Eastern Site	6-56
6-26	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Subsistence Farmer at Western Site	6-57
6-27	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Home Gardener at Eastern Site	6-58
6-28	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Home Gardener at Western Site	6-59
6-29	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban Average Resident at Eastern Site	6-60
6-30	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban Average Resident at Western Site	6-61
6-31	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban High-End Scenario at Eastern Site	6-62
6-32	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban High-End Scenario at Western Site	6-63
6-33	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for High-End Fisher at Eastern Site	6-64
6-34	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for High-End Fisher at Western Site	6-65
6-35	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Recreational Angler at Eastern Site	6-66
6-36	Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Recreational Angler at Western Site	6-67
6-37	Combination of Local and Regional Impacts: Predicted Methylmercury Intake for Wildlife Receptors at Eastern Site	6-69
6-38	Combination of Local and Regional Impacts: Predicted Methylmercury Intake for Wildlife Receptors at Western Site	6-70

LIST OF TABLES (continued)

		Page
6-39	Comparison of Results using Different Assumptions Regarding Dry Deposition of Mercury Vapor	6-73
6-40	Illustration of Effect of Receptor Height on Dry Deposition at a Distance of 2.5 km from the Source: Ratio of Predicted Value with Value for Receptor in Simple Terrain	6-76
6-41	Illustration of Effect of Receptor Height on Dry Deposition at a Distance of 25 km from the Source: Ratio of Predicted Value with Value for Receptor in Simple Terrain	6-76

LIST OF FIGURES

		Page
1-1	Fate and Transport Models Used and Exposure Routes Considered to Examine Exposure Predictions Using Measured Environmental Concentrations	1-3
1-2	Fate, Transport and Exposure Modeling Conducted in the Long Range Transport Analysis	1-4
1-3	Fate, Transport and Exposure Modeling Conducted in the Local Impact Analysis	1-5
1-4	Fate, Transport and Exposure Modeling Conducted in the Combined COMPDEP and RELMAP Local Impact Analysis	1-6
2-1	The Mercury Circle	2-3
2-2	Mercury Fish Consumption Advisories of the U.S.	2-20
4-1	Overview of the IEM2 Watershed Modules	4-3
4-2	Predicted Steady-state Watershed Dynamics for Eastern and Western Sites Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g	4-14
5-1	Hg(0) Emissions from All Anthropogenic Sources (Base)	5-5
5-2	Hg^{2+} Emissions from All Anthropogenic Sources (Base)	5-6
5-3	Hg(p) Emissions from All Anthropogenic Sources (Base)	5-7
5-4	Hg(p) Emissions from All Anthropogenic Sources (Alternate)	5-8
5-5	Average Hg(0) Concentration (Base) Excluding Background	5-12
5-6	Average Hg^{2+} Concentration (Base)	5-13
5-7	Average (Hg(p) Concentration (Base)	5-15
5-8	Hg(0) Wet Deposition (Base) Excluding Background	5-17
5-9	Hg(0) Wet Deposition (Background Only)	5-18
5-10	Hg^{2+} Wet Deposition (Base)	5-20
5-11	Hg(p) Wet Deposition (Base)	5-21
5-12	Total Hg Wet Deposition (Base)	5-22
5-13	Total Hg Wet Deposition (Alternate)	5-24
5-14	Hg^{2+} Dry Deposition (Base)	5-27
5-15	Hg(p) Dry Deposition (Base)	5-29
5-16	Total Hg Dry Deposition (Base)	5-30
5-17	Total Hg Dry Deposition (Alternate)	5-31
5-18	Total Hg Wet+Dry Deposition (Base)	5-33
5-19	Total Hg Wet+Dry Deposition (Altenate)	5-34
6-1	Configuration of Hypothetical Water Body and Watershed Relative to Local Source	6-7
6-2	Predicted Total Mercury Air Concentration (ng/m^3) at Eastern Site, COMPDEP Only	6-8
6-3	Predicted Total Mercury Air Concentration (ng/m^3) at Western Site, COMPDEP Only	6-9
6-4	Predicted Total Mercury Deposition Rate (ug/m^2/yr) at Eastern Site, COMPDEP Only	6-14
6-5	Predicted Total Mercury Deposition Rate (ug/m^2/yr) at Western Site, COMPDEP Only	6-15
6-6	Predicted Untilled Soil Mercury Concentration (ng/g) at Eastern Site, COMPDEP Only	6-16
6-7	Predicted Untilled Soil Mercury Concentration (ng/g) at Western Site, COMPDEP Only	6-17
6-8	Predicted Leafy Vegetable Mercury Concentration (μg/g) at the Eastern Site, COMPDEP Only	6-18
6-9	Predicted Leafy Vegetable Mercury Concentration (μg/g) at the Western Site, COMPDEP Only	6-19
6-10	Predicted Fruit Mercury Concentration (μg/g) at the Eastern Site, COMPDEP Only	6-20
6-11	Predicted Fruit Mercury Concentration (μg/g) at the Western Site, COMPDEP Only	6-21
6-12	Predicted Beef Mercury Concentration (μg/g) at the Eastern Site, COMPDEP Only	6-22

LIST OF FIGURES (continued)

		Page
6-13	Predicted Beef Mercury Concentration (µg/g) at the Western Site, COMPDEP Only	6-23
6-14	Predicted Total Mercury Surface Water Concentration (ng/l) at the Eastern Site, COMPDEP Only	6-24
6-15	Predicted Total Mercury Surface Water Concentration (ng/l at the Western Site, COMPDEP Only	6-25
6-16	Predicted Trophic Level 3 Fish Mercury Concentration (ug/g) at the Eastern Site, COMPDEP Only	6-26
6-17	Predicted Trophic Level 3 Fish Mercury Concentration (ug/g) at the Western Site, COMPDEP Only	6-27
6-18	Predicted Trophic Level 4 Fish Mercury Concentration (ug/g) at the Eastern Site, COMPDEP Only	6-28
6-19	Predicted Trophic Level 4 Fish Mercury Concentration (ug/g) at the Western Site, COMPDEP Only	6-28

LIST OF SYMBOLS, UNITS AND ACRONYMS

ANC	Acid Neutralizing Capacity
BAF	Bioaccumulation Factor
BCF	Bioconcentration Factor
BW	Body Weight
CAA	Clean Air Act as Amended in 1990
CAP	Chlor-Alkali Plants
CMWI	Continuous MWI Model Plant
COMPDEP	COMPlex terrain and DEPosition air dispersion model
EMEP	European Monitoring and Evaluation Programme
EPA	U.S. Environmental Protection Agency
EPRI	Electric Power Research Institute
FWS	U.S. Fish and Wildlife Service
Hg	Mercury
Hg^o	Elemental mercury
Hg^{2+}	Divalent or mercuric mercury
Hg_2^{2+}	Mercurous mercury
Hg(II)	Divalent or mercuric mercury
$HgCl_2$	Mercuric chloride
HgI	Mercury iodide
Ht	Height
IED	Indirect Exposure Document (U.S. EPA, 1990. Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions, EPA 600/6-90/003)
IEM2	Modified version of U.S. EPA's Indirect Exposure Methodology
IMWI	Intermittent Medical Waste Incinerator Model Plant
LCUB	Large Coal-Fired Utility Boiler
LMWC	Large Municipal Waste Combustor
MCUB	Medium Coal-Fired Utility Boiler
MHg	Methylmercury
Mg	Megagram
MMHg	MonoMethylmercury
MOUB	Median Oil-Fired Utility Boiler
MW	MegaWatt
MWC	Municipal Waste Combustors
MWI	Medical Waste Incinerators
NIEHS	National Institute of Environmental Health and Safety
NJDEPE	New Jersey Department of Environmental Protection
OAQPS	Office of Air Quality Planning and Standards
PBL	Planetary Boundary Layer
PCS	Primary Copper Smelter
PLS	Primary Lead Smelter
RELMAP	Regional Lagrangian Model of Air Pollution
RHG	Rural Home Gardener
RSF	Rural Subsistence Farmer
SCUB	Small Coal-Fired Utility Boiler
SMWC	Small Municipal Waste Combustor
SWMC	Solid Waste Management Council
ug	Microgram ($1x10^{-6}$ gram)
UR	Urban Resident

LIST OF SYMBOLS, UNITS AND ACRONYMS (continued)

USFDA	United States Food and Drug Administration
USGS	United States Geological Survey

EXECUTIVE SUMMARY

Section 112(n)(1)(B) of the Clean Air Act (CAA), as amended in 1990, requires the U.S. Environmental Protection Agency (EPA) to submit a study on atmospheric mercury emissions to Congress. The sources of emissions that must be studied include electric utility steam generating units, municipal waste combustion units and other sources, including area sources. Congress directed that the Mercury Study evaluate many aspects of mercury emissions, including the rate and mass of emissions, health and environmental effects, technologies to control such emissions, and the costs of such controls.

In response to this mandate, U.S. EPA has prepared a seven-volume Mercury Study Report to Congress. This document is the exposure assessment (Volume III) of the Mercury Study Report to Congress. The exposure assessment is one component of the risk assessment of U.S. anthropogenic mercury emissions. This exposure assessment considers both inhalation and ingestion exposure routes. For mercury emitted to the atmosphere, ingestion is an indirect route of exposure that results from mercury deposition onto soil, water bodies and plants and uptake through the food chain. The analyses in this volume are integrated with information relating to human and wildlife health impacts of mercury in the Risk Characterization Volume (Volume VI) of the Report.

Exposure Assessment Approach

This assessment addresses atmospheric mercury emissions from selected, major anthropogenic combustion and manufacturing source categories: municipal waste combustors (MWCs), medical waste incinerators (MWIs), coal- and oil-fired utility boilers, chlor-alkali plants, primary lead smelters and primary copper smelters. It does not address all anthropogenic emission sources nor does it address emissions from natural sources.

There are no extant monitoring data that conclusively demonstrate a relationship between the individual anthropogenic sources above and increased mercury concentrations in environmental media or biota. Available mercury monitoring data around these sources of interest are, however, extremely limited. No comprehensive database describing environmental concentrations has been developed. To determine if there is a connection between the above sources and increased environmental levels, the exposure assessment in this Report utilized exposure modeling techniques to address many major scientific uncertainties.

The exposure assessment in this Report is considered to be a qualitative study based partly on quantitative analyses; it is considered qualitative because of inherent uncertainties. The exposure assessment draws upon the available scientific information and develops two quantitative analyses: a long range transport analysis, and a local impact analysis. It was intended that these two types of analyses would provide a more complete estimate of the nation-wide impact of anthropogenic emission sources than either analysis could provide individually.

The exposure assessment draws upon the available scientific information and presents quantitative modeling analyses which examine the following: (1) the long range transport of mercury from emission sources through the atmosphere; (2) the transport of mercury from emission sources through the local atmosphere; (3) the aquatic and terrestrial fate and transport of mercury at hypothetical sites; and (4) finally, the resulting exposures to hypothetical humans and animals that inhabit these sites.

Long Range Transport Analysis

The long range transport modeling was undertaken to estimate the regional and national impacts of mercury emissions. It estimates the long range atmospheric transport of mercury and the impact of mercury across the continental U.S. The bases of this modeling were assumptions concerning the atmospheric chemistry of emitted elemental mercury (Petersen et al., 1995) and the numerous studies linking increased mercury levels in air, soil, sediments and biota at remote sites to distant anthropogenic mercury release followed by long range transport. Details of several studies which demonstrate the long range transport of mercury are presented in Volume III. These studies provide ample evidence to justify an assessment of long range mercury transport.

The long range transport of mercury was modeled using site-specific, anthropogenic emission source data (presented in Volume II of this Report) to generate mean, annual atmospheric mercury concentrations and deposition values across the continental U.S. The Regional Lagrangian Model of Air Pollution (RELMAP) atmospheric model was utilized to model annual mercury emissions from multiple mercury emission sources. Assumptions were made concerning the form and species of mercury emitted from each source class. The results of the RELMAP modeling were utilized in these ways. First, the predicted atmospheric mercury concentrations and deposition rates were used to identify patterns across the U.S. Secondly, the continental U.S. was divided into Western and Eastern halves along 90 degrees west longitude, and the 50th and 90th percentiles of the predicted atmospheric concentrations and deposition rates were then used as inputs in the indirect exposure models to examine the impacts of long range transport of emissions. Finally, RELMAP results for remote Eastern locations that were predicted to have high deposition rates were also used as inputs to the indirect exposure models to predict fish concentrations at lakes remote from emission sources.

Exposure Assessment of Local Deposition of Mercury

An analysis of the local atmospheric transport of mercury released from anthropogenic emission sources was undertaken to estimate the impacts of mercury from selected, individual sources. Model plants were developed; these are defined as hypothetical facilities that represent actual emissions from existing industrial processes and combustion sources. The modal plants were situated in hypothetical locations intended to simulate a site in either the Western or Eastern U.S. This approach was selected because some environmental monitoring studies suggest that measured mercury levels in environmental media and biota may be elevated in areas around stationary industrial and combustion sources known to emit mercury. These measured data are detailed in Chapter 2 of this Volume.

Atmospheric concentrations and deposition rates were used as inputs to a series of terrestrial and aquatic models described in U.S. EPA's (1990) Methodology for Assessing Health Risks to Indirect Exposure from Combustor Emissions and a 1994 Addendum. The results of these terrestrial and aquatic models were used to predict mercury exposure to hypothetical humans through inhalation, consumption of drinking water and ingestion of soil, farm products (e.g., beef product and vegetables) and fish. These models were also used to predict mercury exposure in hypothetical piscivorous (i.e., fish-eating) birds and mammals through their consumption of fish.

National Assessment of Mercury Exposure from Fish Consumption

A current assessment of U.S. general population methylmercury exposure through the consumption ~~of fish~~ RATES is provided in Chapter 3 and in Appendix H. This assessment was conducted to provide an estimate of mercury exposure through the consumption of fish to the general U.S.

population. It is not a site-specific assessment but rather a national assessment. This assessment utilizes data from the 1989 - 1991 Continuing Surveys of Food Intake by Individuals (CSFII 89-91) to estimate a range of fish consumption rates among U.S. fish eaters. Only individuals who reported fish consumption were considered. For each fish-eater, the CSFII 89-91 study identified the number of fish meals, the quantities and species of fish consumed and the self-reported body weights of the consumers. The constitution of the survey population was weighted to reflect the actual U.S. population.

IN A THREE-DAY PERIOD

FDA (1995)

These estimates of fish consumption rates were combined with fish species-specific mean values for measured methylmercury concentrations. The fish methylmercury concentration data were obtained from the National Marine Fisheries Service, Bahnick et al., (1994) and Lowe et al., (1985). Through the application of specific fish preparation factors (USDA, 1995), estimates of the range of methylmercury exposure from the consumption of fish were prepared for the fish-consuming segment of the U.S. population. Per body weight estimates of methylmercury exposure were determined by dividing the total daily methylmercury exposure from this pathway by the self-reported body weights. The results of this analysis show that children on a per kilogram body weight basis have higher average exposure rates to methylmercury through the consumption of fish than adults.

Results of smaller sources on "high-end" fish consumers are also included.

Conclusions

The following conclusions are presented in approximate order of degree of certainty in the conclusion, based on the quality of the underlying database. The conclusions progress from those with greater certainty to those with lesser certainty.

- The present study in conjunction with available scientific knowledge, supports a plausible link between mercury emissions from anthropogenic combustion and industrial sources and mercury concentrations in air, soil, water and sediments. The critical variables contributing to this linkage are these:

  a) the species of mercury that are emitted from the sources, with elemental mercury (Hg^0) mostly contributing to concentrations in ambient air and divalent mercury (Hg^{2+}) mostly contributing to concentrations in soil, water and sediments;

  b) the overall amount of mercury emitted from a combustion source; and

  c) the climate conditions.

- The present study, in conjunction with available scientific knowledge, supports a plausible link between mercury emissions from anthropogenic combustion and industrial sources and methylmercury concentrations in freshwater fish. The critical variables contributing to this linkage are the following:

  a) the species of mercury that are emitted, with emitted divalent mercury mostly depositing into local watershed areas and, to a lesser extent the atmospheric conversion of elemental mercury to divalent species which are deposited over greater distances;

b) the overall amount of mercury emitted from a source:

c) the extent of mercury methylation in the water body: and

d) the climate conditions.

- There is a lack of adequate mercury measurement data near the anthropogenic atmospheric mercury sources considered in this Report. The lack of such measured data preclude a comparison of the modeling results with measured data around these sources. These data include measured mercury deposition rates as well as measured concentrations in the atmosphere, soils, water bodies and biota.

- From the RELMAP analysis of mercury deposition and on a comparative basis, a facility located in a humid climate has a higher annual rate of mercury deposition than a facility located in an arid climate. The critical variables are the estimated washout ratios of elemental and divalent mercury, as well as the annual amount of precipitation. Precipitation removes various forms of mercury from the atmosphere and deposits mercury to the surface of the earth.

- On a national scale, an apportionment between sources of mercury and mercury in environmental media and biota cannot be described in quantitative terms with the current scientific understanding of the environmental fate and transport of this pollutant.

- Consumption of fish is the dominant pathway of exposure to methylmercury for fish-consuming humans and wildlife. There is a great deal of variability among individuals in these populations with respect to food sources and fish consumption rates. As a result, there is a great deal of variability in exposure to methylmercury in these populations. The anthropogenic contribution to the total amount of methylmercury in fish is, in part, the result of anthropogenic mercury releases from industrial and combustion sources increasing mercury body burdens in fish. As a consequence of human and wildlife consumption of the affected fish, there is an incremental increase in exposure to methylmercury.

- Due to differences in fish consumption rates per body weight and differences in body weights among species, it is likely that piscivorous birds and mammals have much higher environmental exposures to methylmercury than humans through the consumption of contaminated fish. This is true even in the case of fish consumption by humans who consume above average amounts of fish. The critical variables contributing to these outcomes are these:

  a) the fish consumption rate;

  b) the body weight of the individual in relation to the fish consumption rate; and

  c) the rate of biomagnification between trophic levels within the aquatic food-chain.

- The results of the assessment of current exposure of the U.S. population from fish consumption as described in Appendix H indicate that exposure to methylmercury

from contaminated fish results in an incremental increase in mercury exposure for most U.S. fish-consumers. Methylmercury exposure rates on a per body weight basis among fish-consuming children are predicted to be higher than for fish-consuming adults. The exposure rates among fish-consuming children under the age of 15 are estimated to average between 0.12 and 0.16 micrograms of methylmercury per kilogram of body weight per day. The exposure rates among fish-consuming adults are estimated to average between 0.07 and 0.08 micrograms of methylmercury per kilogram of body weight per day. Human adult fish consumption rates vary from 0 to greater than 300 grams per day.

- From the modeling analysis and a review of field measurement studies, it is concluded that mercury deposition appears to be ubiquitous across the continental U.S., and at, or above, detection limits when measured with current analytic methods.

- Based on the RELMAP modeling analysis and a review of recent measurement data published in peer-reviewed scientific literature, there is predicted to be a wide range of mercury deposition rates across the continental U.S. The highest predicted rates (i.e., above 90th percentile) are more than 50 times higher than the lowest predicted rates (i.e., below the 10th percentile). Three principal factors contribute to these modeled and observed deposition patterns:

    a) emission source locations;

    b) amount of divalent and particulate mercury emitted or formed in the atmosphere; and

    c) climate and meteorology.

- Based on the modeling analysis of the transport and deposition of stationary point source and area source air emissions of mercury from the continental U.S., it is concluded that the following geographical areas have the highest annual rate of deposition of mercury in all forms (above the levels predicted at the 90th percentile):

    a) The southern Great Lakes and Ohio River Valley.

    b) The Northeast and southern New England.

    c) Scattered areas in the South with the most elevated deposition occurring in the Miami and Tampa areas.

  Measured deposition estimates are limited, but are available for certain geographic regions. The data that are available corroborate the RELMAP modeling results for specific areas.

- Based on modeling analysis of the transport and deposition of stationary point source and area source air emissions of mercury from the continental U.S., it is concluded that the following geographical areas have the lowest annual rate of deposition of *mercury in all forms* (below the levels predicted at the 10th percentile).

a) The less populated areas of the Great Basin, including southern Idaho, southeastern Oregon, most of southern and western Utah, most of Nevada, and portions of western New Mexico, and

b) Western Texas other than near El Paso, and most of northeastern Montana.

- Based on limited monitoring data, the RELMAP model predictions of atmospheric mercury concentrations and wet deposition across the U.S. are comparable with typically measured data.

- EPA concludes that the selected major anthropogenic sources as modeled and parameterized for this assessment, can be ranked by predicted deposition rate at 2.5 Km in flat terrain, on a relative basis from high to low, as follows:

  Municipal waste combustors
  Chlor-alkali plants
  Lead smelters
  Copper smelters
  Medical waste incinerators
  Utility boilers

  The critical variables impacting the ranking are these:

  a) estimated amounts of divalent and particulate mercury emitted; and

  b) parameters that influence the plume height, primarily the stack height and stack exit gas velocity.

  This ranking may be sensitive to differences in the distance from the source (distances other than 2.5 Km) and the topography of the terrain.

- From the analysis of deposition and on a comparative basis, the deposition of divalent mercury close to an emission source is greater for receptors in elevated terrain (i.e., terrain above the elevation of the stack base) than from receptors located in flat terrain (i.e., terrain equal to the elevation of the stack base). The critical variables are parameters that influence the plume height, primarily the stack height and stack exit gas velocity.

- In terms of methylmercury intake on a per body weight basis, the five wildlife species considered in this analysis can be ranked from high to low as follows:

  -- Kingfisher
  -- River Otter
  -- Mink, Osprey
  -- Bald eagle

  Methylmercury exposures for the most exposed wildlife species (the kingfisher) may be up to two orders of magnitude higher than human exposures from contaminated freshwater fish (on a kilogram fish consumed per body weight basis). This assumes

that the fish within different trophic levels of a given lake are contaminated with the same concentrations of methylmercury.

- Modeling estimates of the transport and deposition of stationary point source and area source air emissions of mercury from the continental U.S. have revealed the following partial mass balance.

  -- Of the total amount of elemental mercury vapor that is emitted, about 1 percent (1.2 metric tons/yr) may be atmospherically transformed into divalent mercury by tropospheric ozone and adsorbed to particulate soot in the air and subsequently deposited in rainfall and snowfall to the surface of the continental U.S. The vast majority of emitted elemental mercury does not readily deposit and is transported outside the U.S. or vertically diffused to the free atmosphere to become part of the global cycle.

  -- Nearly all of the elemental mercury vapor emitted from other sources around the globe also enters the global cycle and can be deposited slowly to the U.S. Nearly 30 times as much elemental mercury vapor is deposited from these other sources than from stationary point sources and area sources within the continental U.S.

  -- Of the total amount of divalent mercury vapor that is emitted, about 68 percent (62.6 metric tons/year) deposits to the surface through wet or dry processes within the continental U.S. The remaining 32 percent is transported outside the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle.

  -- Of the total amount of particulate mercury that is emitted, about 36 percent (14.1 metric tons/year) deposits to the surface through wet or dry processes within the continental U.S. The remaining 64 percent is transported outside the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle.

- Assuming these deposition efficiencies are correct (namely; elemental mercury - 1%, divalent mercury vapor - 68%, and particulate mercury - 36%) the relative source contributions to the total anthropogenic mercury that is deposited to the continental U.S. are ranked as follows:

  -- Medical waste incineration 36% (28 Megagrams (Mg) of 78 Mg)
  -- Municipal waste combustion 31% (24 Mg of 78 Mg)
  -- Coal-fired electric utility boilers 17% (13 Mg of 78 Mg)
  -- Industrial and residential fossil fuel use 10% (8 Mg of 78 Mg)
  -- Chlor-alkali factories 2% (1 Mg of 78 Mg)
  -- Non-ferrous metal smelting 1% (1 Mg of 78 Mg)
  -- Oil-fired electric utility boilers 1% (1 Mg of 78 Mg)

- Based on the local scale atmospheric modeling results in flat terrain, at least 75% of the emitted mercury from each facility is predicted to be transported more than 50 km from the facility.

- The models used in the exposure analysis indicate that, except for utility boilers and intermittent medical waste incinerators, deposition within 10 Km of a facility is generally dominated by emissions from the local source rather than from emissions transported from regional mercury emissions sources.

There are many uncertainties in the exposure assessment. Major uncertainties include the following:

- Comprehensive emission data for various anthropogenic and natural sources are not available. This reflects the current developmental nature of emission speciation methods, resulting in few data on the various species of mercury and proportions of vapor and solid forms emitted. Both elemental and divalent mercury species as well as gaseous and particulate forms are known to be emitted from point and area sources.

- Atmospheric chemistry data are incomplete. Some atmospheric reactions of mercury, such as the oxidation of elemental mercury to divalent mercury in cloud water droplets have been reported. Other chemical reactions in the atmosphere that may reduce divalent species to elemental mercury have not been reported.

- There is inadequate information on the atmospheric processes that affect wet and dry deposition of mercury. Atmospheric particulate forms and divalent species of mercury are thought to wet and dry deposit more rapidly than elemental mercury; however, the relative rates of deposition are uncertain.

- There is no validated air pollution model that estimates wet and dry deposition of vapor-phase compounds close to the emission source.

- There is some uncertainty regarding the revolatilization of deposited mercury.

- There is a lack of information concerning the movement of mercury from watershed soils to water bodies.

- There are not conclusive data on the amount of and rates of mercury methylation in different types of water bodies.

- There is a lack of data on the transfer of mercury between environmental compartments and biologic compartments; for example, the link between the amount of mercury in the water body and the levels in fish appears to vary from water body to water body.

To improve the quantitative exposure assessment component of the risk assessment for mercury and mercury compounds, U.S. EPA would need more and better mercury emissions data and measured mercury data near sources of concern, as well as a better quantitative understanding of mercury chemistry in the emissions plume, the atmosphere, soils, water bodies and biota. Specific needs include the following:

Mercury in the Atmosphere

- aqueous oxidation-reduction kinetics in atmospheric water droplets;

- physical adsorption and condensation of divalent mercury gas to ambient particulate matter
- photolytic reduction of particle-bound divalent mercury by sunlight
- convincing evidence that gas-phase oxidation of mercury is insignificant

Mercury in Soils and Water Bodies

- uptake and release kinetics of mercury from terrestrial plants
- biogeochemical mercury transport and transformation kinetics in benthic sediments
- methylation and demethylation kinetics in water bodies
- sorption coefficients to soils, suspended solids and benthic solids
- complexation to organic matter in water bodies

Information Leading to an Improved Quantitative Understanding of Aquatic Bioaccumulation Processes and Kinetics

- uptake kinetics by aquatic plants and phytoplankton
- partitioning and binding behavior of mercury species within organisms
- metabolic transformations of mercury, and the effect on uptake, internal distribution, and excretion

Information that will facilitate the development of a dynamic, linked terrestrial-aquatic mass balance modeling framework that includes realistic mercury chemistry and the aquatic food web as an integral component

- More measurements of methylmercury concentrations in fish for better identification of the range in fish species,
- Surveys of fish consumption among potential high-end fish consumers which examine specific biomarkers indicating mercury exposure (e.g., blood mercury concentrations and hair mercury concentrations), and
- A pharmacokinetic-based understanding of mercury partitioning in human, (adults, children, and fetuses) as well as wildlife; particularly the interactions of different forms of mercury and different uptake routes.

1. INTRODUCTION

Section 112(n)(1)(B) of the Clean Air Act (CAA), as amended in 1990, requires the U.S. Environmental Protection Agency (EPA) to submit a study on atmospheric mercury emissions to Congress. The sources of emissions that must be studied include electric utility steam generating units, municipal waste combustion units, and other sources, including area sources. Congress directed that the Mercury Study evaluate many aspects of mercury emissions, including the rate and mass of emissions, health and environmental effects, technologies to control such emissions, and the costs of such controls.

In response to this mandate, EPA has prepared a seven-volume Mercury Study Report to Congress. The seven volumes are as follows:

I. Executive Summary
II. An Inventory of Anthropogenic Mercury Emissions in the United States
III. An Assessment of Exposure from Anthropogenic Mercury Emissions in the United States
IV. Health Effects of Mercury and Mercury Compounds
V. An Ecological Assessment of Anthropogenic Mercury Emissions in the United States
VI. Characterization of Human Health and Wildlife Risks from Anthropogenic Mercury Emissions in the United States
VII. An Evaluation of Mercury Control Technologies and Costs

This document is the exposure assessment Volume III of U.S. EPA's Report to Congress on Mercury. The exposure assessment is one element of the human health and ecological risk assessment of U.S. anthropogenic mercury (Hg) emissions. The exposure assessment considers both inhalation and ingestion exposure routes. For atmospheric mercury emissions, ingestion is an indirect route of exposure that results from mercury deposition onto soil, water bodies and plants and uptake through the food chain. The information in this document is integrated with information relating to human and wildlife health impacts of mercury in Volume VI of the report.

This assessment addresses the atmospheric fate and transport as well as the exposures that result from atmospheric mercury emissions from selected, major anthropogenic combustion and manufacturing sources: municipal waste combustors (MWC), medical waste incinerators (MWI), coal- and oil-fired utility boilers, chlor-alkali plants (CAP), primary lead smelters and primary copper smelters. This volume also estimates current exposures to the general U.S. population that result from mercury concentrations in freshwater and marine fish. This volume does not address all anthropogenic emission sources, nor does it address emissions from natural sources.

Volume III is composed of nine chapters and eight appendices. The Introduction is followed by Chapter 2, which briefly describes chemical properties of mercury, the mercury cycle, analytic mercury measurement methods and measured mercury concentrations in environmental media (i.e., air, rain water, soil and surface waters and benthic sediments) and biota (i.e., plants and animals). Chapter 3 describes estimates of mercury exposure to general human populations and occupationally exposed subpopulations. Chapter 3 also presents a summarization of Appendix H, which describes current U.S. exposures through consumption of fish and concludes with a general exposure assessment to wildlife through consumption of contaminated fish. The fish methylmercury concentrations were developed using measured data and the human fish consumption rates were also developed using measured data.

The fate, transport and exposure modeling of mercury is conducted in Chapters 4, 5 and 6. Chapter 4 describes the long range atmospheric transport model (RELMAP), the local scale atmospheric transport model (COMPDEP) and the aquatic and terrestrial fate, transport, and exposure models (IEM2) that were utilized in the modeling analysis; these models are described in detail in Appendix D. Parameters which describe hypothetical watersheds and lakes in both the Western and Eastern U.S. are also detailed in Chapter 4. Hypothetical human and animal occupants of the watershed are also described. Chapter 4 concludes with aquatic and terrestrial fate, transport and exposure modeling using typically measured mercury air concentration, deposition rate and soil concentration. See Figure 1-1.

The primary variables in the modeling conducted in Chapters 4, 5 and 6 are the source of the mercury concentrations in the atmosphere and soil and the source of the mercury deposition rate. Chapter 4 utilizes typically measured atmospheric and soil concentrations and a measured deposition rate as inputs to the aquatic and terrestrial fate, transport, and exposure models (IEM2) at the hypothetical Western and Eastern U.S. sites. Chapter 5 utilizes the 50th and 90th percentiles of the atmospheric mercury concentrations and the deposition rates that were predicted by the RELMAP model for the Eastern and Western halves of the U.S. as inputs to the aquatic and terrestrial fate, transport, and exposure models (IEM2) at the hypothetical Western and Eastern U.S. sites. Additionally, the environmental fate of mercury at a site in the Eastern U.S. that is distant from anthropogenic emissions sources is also modeled in Chapter 5. Figure 1-2 describes the fate transport and exposure modeling conducted in Chapter 4.

Two separate modeling analyses are described in Chapter 6. In the first analysis, model plants were developed to represent major anthropogenic combustion and manufacturing sources: municipal waste combustors (MWC), medical waste incinerators (MWI), coal- and oil-fired utility boilers, chlor-alkali plants (CAP), primary lead smelters and primary copper smelters. The atmospheric fate and transport of the mercury emissions from these representative model plants was modeled on a local scale by the COMPDEP model. The predicted mercury air concentrations and deposition rates that result from individual model plants at 2.5, 10, and 25 kilometers were used as inputs to the aquatic and terrestrial fate, transport, and exposure models (IEM2) at the hypothetical Western and Eastern U.S. sites. Figure 1-3 presents the fate, transport and exposure modeling conducted around hypothetical local sites.

The second modeling analysis in Chapter 6 combines the predictions of the COMPDEP model for the area around the individual model plants at the hypothetical Western and Eastern locations with either the 50th or 90th percentile predictions of the RELMAP model for the Western and Eastern sites. These combined model predictions are used as inputs to the aquatic and terrestrial fate, transport, and exposure models (IEM2) at the hypothetical Western and Eastern U.S. sites. This was conducted to assess the total impact from anthropogenic sources. The results of the assessment of current U.S. exposure due to fish consumption, as presented in Appendix H, and occupational exposure estimates in Chapter 3 are qualitatively integrated with the predicted human exposures in this analysis. Figure 1-4 presents a conceptualization of the integration of long-range atmospheric transport with local atmospheric transport around sources of concern. The terrestrial, aquatic and exposure modeling are also presented.

Chapter 7 describes the conclusions of this Volume. Information needed for better assessment of exposure to emitted mercury and to current concentrations in media and biota is described in Chapter 8. Chapter 9 lists all references cited in this volume.

Figure
Fate and Transport Models Used and Exposure Routes Considered to Examine Exposure Predictions Using Measured Environmental Concentrations

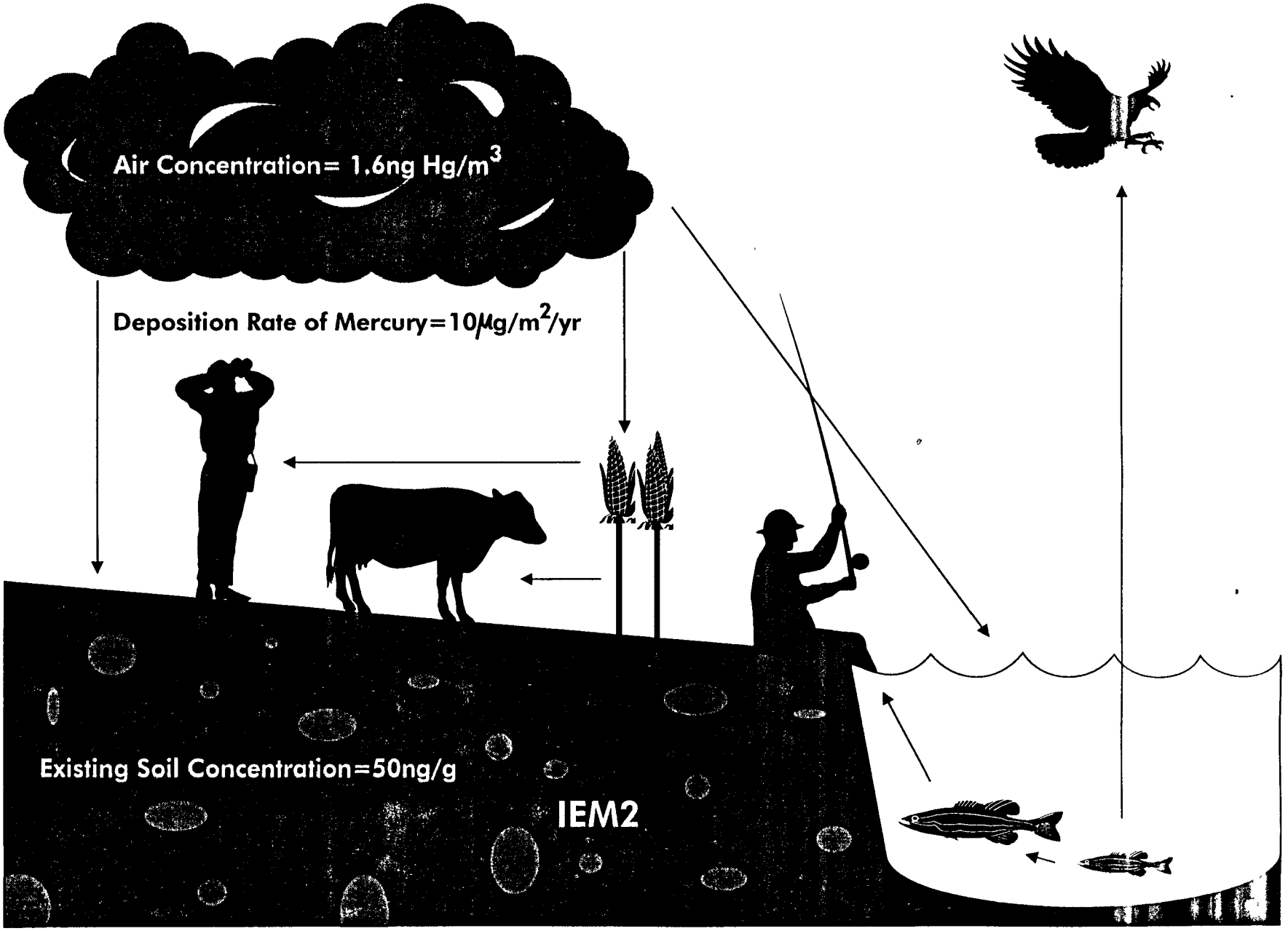
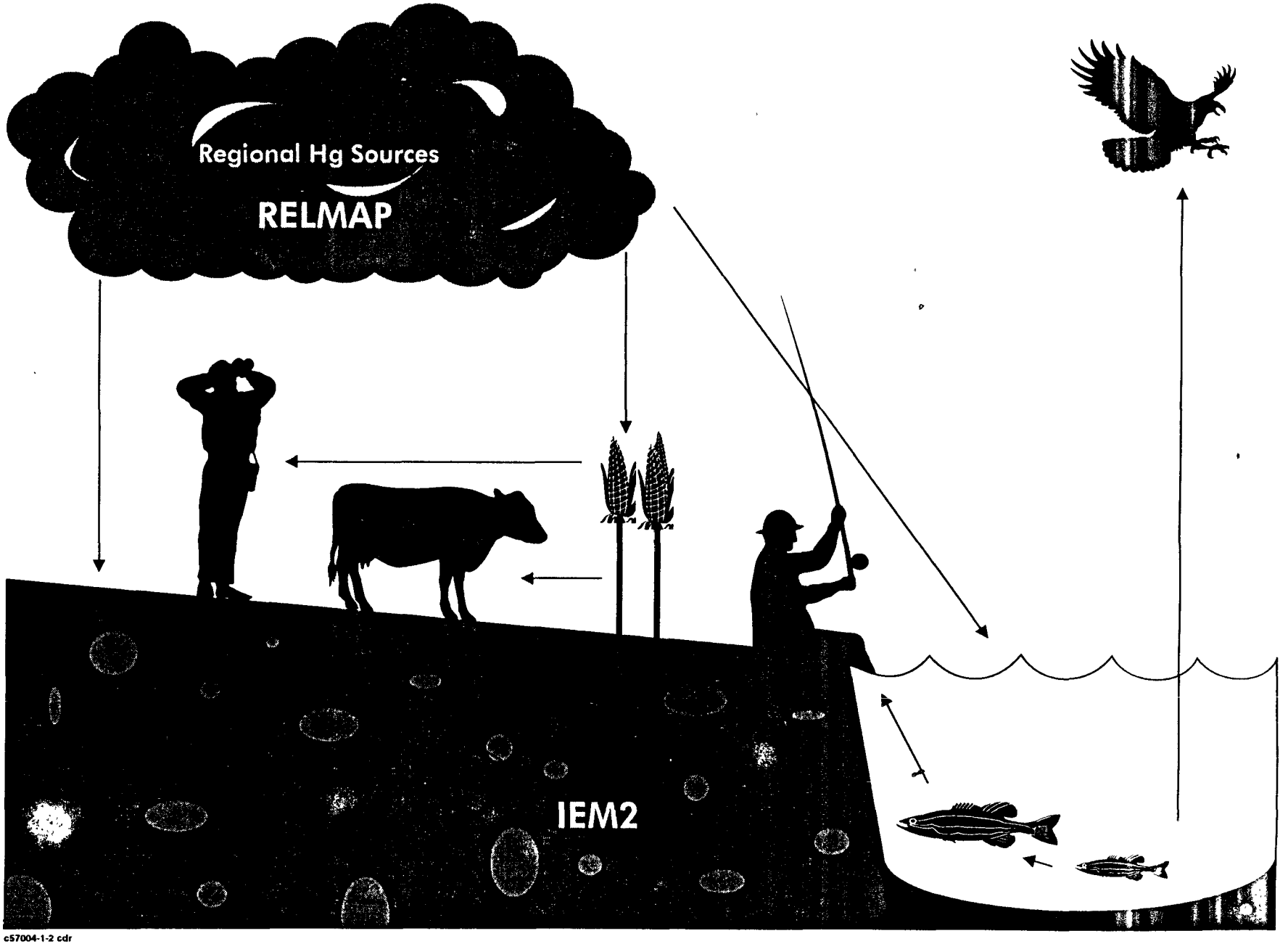
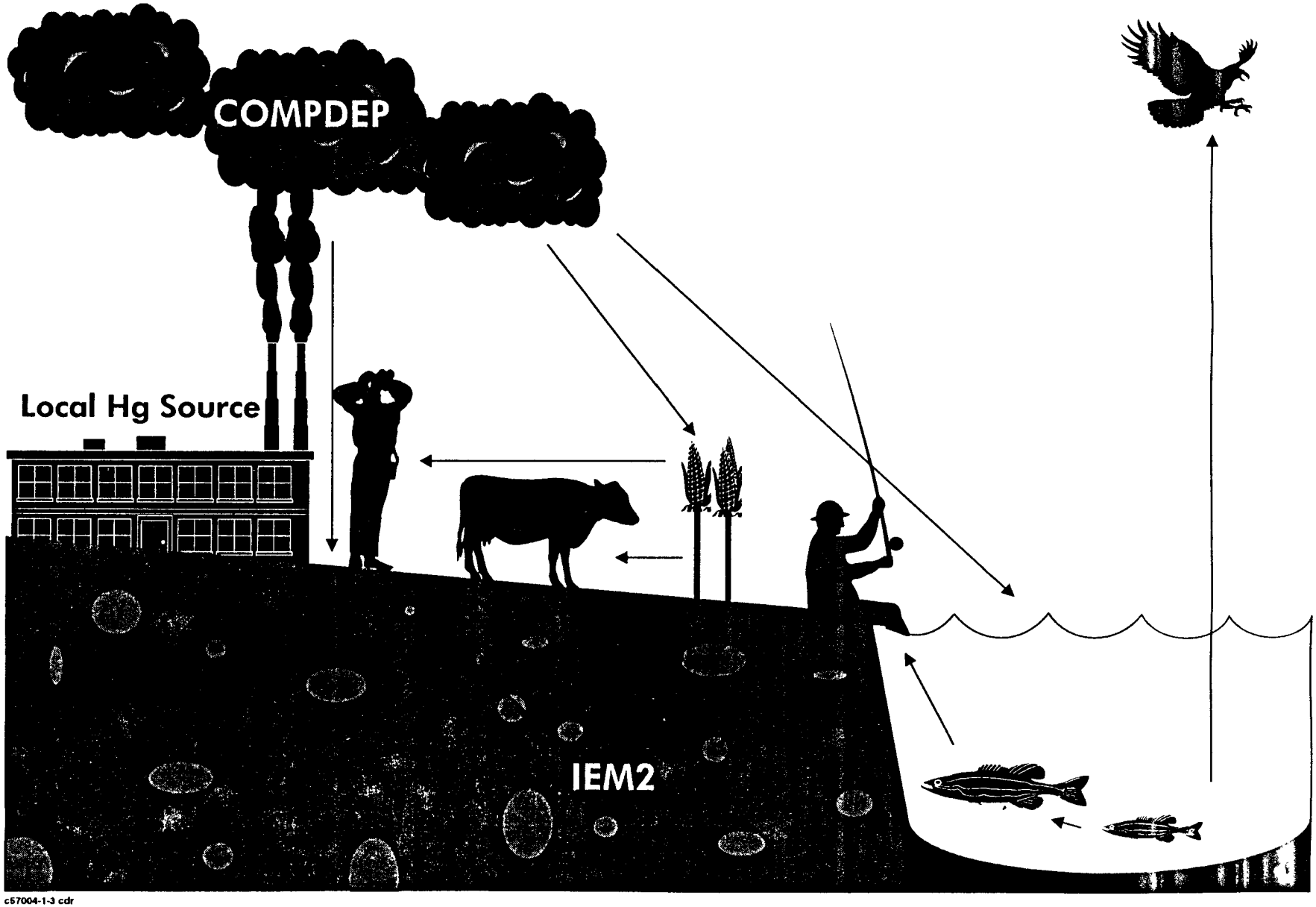


Figure 1-2
Fate, Transport and Exposure Modeling Conducted in the Long Range Transport Analysis



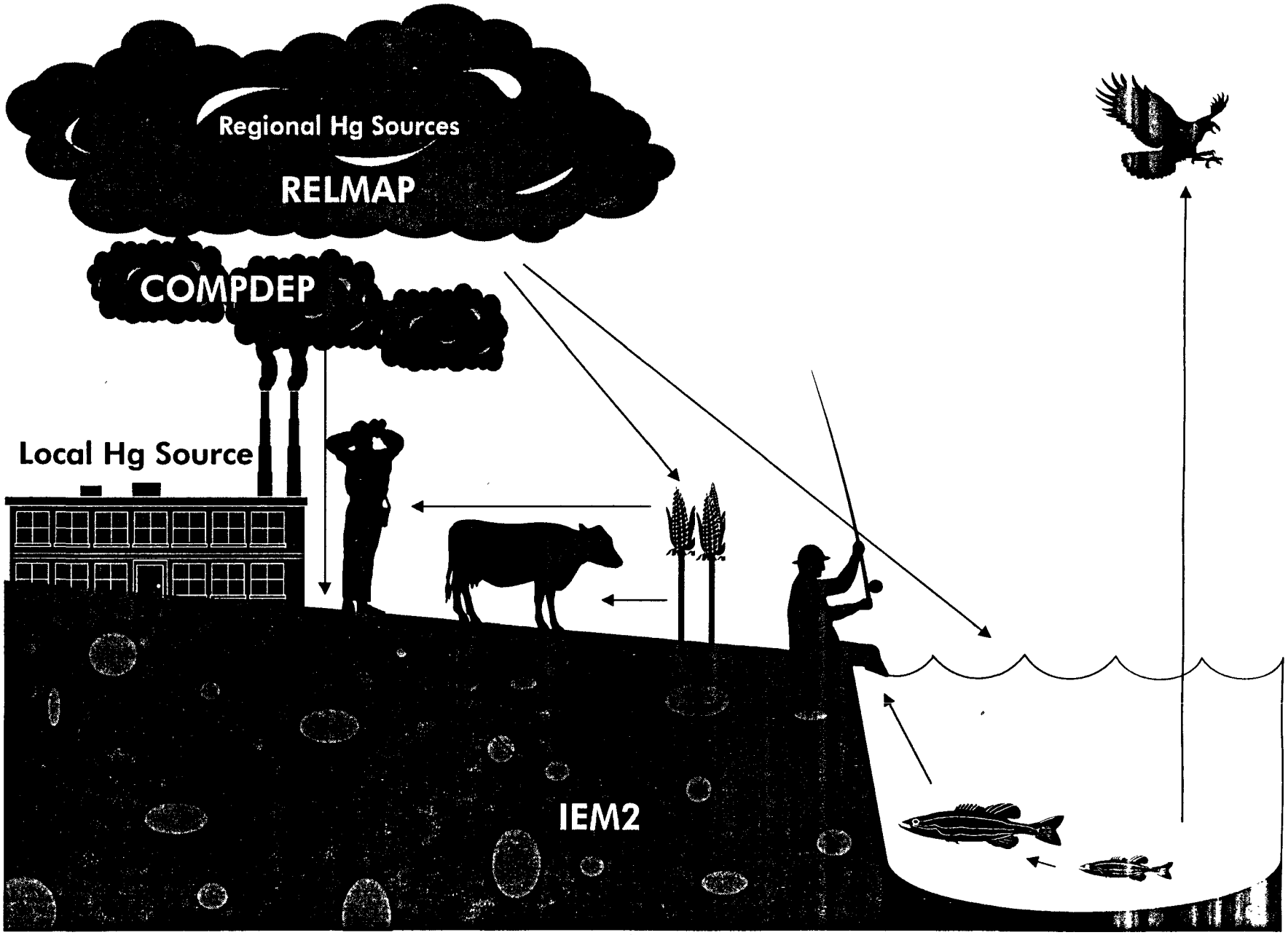
c57004-1-2 cdr

Figure 1
Fate, Transport and Exposure Modeling Conducted in the Local Impact Analysis



c57004-1-3 cdr

Figure 1-4
Fate, Transport and Exposure Modeling Conducted in the Combined COMPDEP and RELMAP Local Impact Analysis



c57004-1-4 cdr

There are eight appendices to volume III (A-H): Site-Independent Parameter Justifications for Local Impact Modeling; Site Dependent Parameter Justifications for Local Impact Modeling; Calibration of Mercury Partition Coefficients; Technical Modeling Details; Chemical Properties of Mercury; Model Plant Descriptions; Listing of Mercury Intakes for All Receptors at All Facilities and Estimated Methylmercury Exposure to The General United States Population through The Consumption of Fish.

Extant mercury monitoring data for particular sources indicate that there is a relationship between emissions and increased mercury in environmental media. Available mercury monitoring data around these sources are extremely limited, however, and no comprehensive data base describing environmental concentrations has been developed. To determine if there is a connection between the above sources and increased environmental mercury concentrations, U.S. EPA utilized current modeling techniques to address many major scientific uncertainties. Because of these and other major uncertainties, the modeling component of this report is essentially a qualitative study based partly on quantitative analyses. These uncertainties include the following:

- Comprehensive emission data for various anthropogenic and natural sources are not available. This reflects the current developmental nature of emission speciation methods, resulting in few data on the various species and proportions of mercury in vapor and solid forms emitted. Both elemental and divalent mercury species as well as gaseous and particulate forms are known to be emitted from point sources.

- Atmospheric chemistry data are incomplete. Some atmospheric reactions of mercury, such as the oxidation of elemental mercury to divalent mercury in cloud water droplets have been reported. Other chemical reactions in the atmosphere which may reduce divalent species to elemental mercury have not been reported.

- There is inadequate information on the atmospheric processes which affect wet and dry deposition of mercury. Atmospheric particulate forms and divalent species of mercury are thought to wet and dry deposit more rapidly than elemental mercury; however, the relative rates of deposition are uncertain.

- There is no validated local air pollution model which estimates wet and dry deposition of vapor-phase compounds.

- There is some uncertainty regarding the revolatilization of deposited mercury.

- There is a lack of information concerning the movement of mercury from watershed soils to water bodies.

- There are no conclusive data concerning the amount of and rates of mercury methylation in different types of water bodies.

- There is a lack of data on the transfer of mercury between environmental compartments and biologic compartments; for example, the link between the amount of mercury in the water body and the levels in fish appears to vary from water body to water body.

The exposure assessment draws upon the available scientific information and presents quantitative modeling analyses which examine (1) the long range transport of mercury through the atmosphere, (2)

the transport of mercury through the local atmosphere, (3) the aquatic and terrestrial fate and transport of mercury at hypothetical sites, and (4) finally, the resulting exposures to hypothetical humans and animals that inhabit these sites. It was intended that these analyses would provide a more complete estimate of the impact of anthropogenic emission sources than an individual analysis.

1.1 Long Range Atmospheric Transport Modeling

The long range transport modeling was undertaken to estimate the regional and national impacts of mercury emissions. It focusses on the long range atmospheric transport of mercury and estimates the impact of mercury across the continental U.S. This type of modeling was conducted based on the atmospheric chemistry of emitted elemental mercury (Petersen et al., 1995) and the numerous studies linking increased mercury concentrations in air, soil, sediments, and biota at remote sites to distant anthropogenic mercury release followed by long-range transport. Details of several studies which demonstrate the long range transport of mercury are presented in Chapter 2. These provide ample evidence to justify this assessment of long-range mercury transport.

The long range transport of mercury was modeled using site-specific, anthropogenic emission source data (Presented in Volume II of this Report) to generate mean, annual atmospheric mercury concentrations and deposition values across the continental U.S. The Regional Lagrangian Model of Air Pollution (RELMAP) atmospheric model was utilized to model cumulative mercury emissions from multiple mercury emission sources. Assumptions were made concerning the form and species of mercury emitted from each source class. The results of the RELMAP modeling were utilized in three ways. The predicted atmospheric mercury concentrations and deposition rates were used to identify patterns across the continental U.S. Secondly, the continental U.S. was divided into Western and Eastern halves along the line of 90° west longitude. The 50th and 90th percentile of the predicted atmospheric concentrations and deposition rates were then used as inputs to the indirect exposure models to examine the impacts of the long-range transport of emissions. Finally, RELMAP results from remote locations in the eastern half of the U.S., which were predicted to have high deposition rates were also used as inputs to the indirect exposure models. Fish mercury concentrations at remote lakes were then predicted.

1.2 Local Atmospheric Transport Modeling

The local atmospheric transport of mercury released from anthropogenic emission sources was undertaken to estimate the impacts of mercury from selected, individual sources. Model plants, defined as hypothetical facilities which were developed to represent actual emissions from existing industrial processes and combustion sources, were located in hypothetical locations intended to simulate a site in either the Western or Eastern U.S. This approach was selected because some environmental monitoring studies suggest that measured mercury levels in environmental media and biota may be elevated in areas around stationary industrial and combustion sources known to emit mercury. These are detailed in Volume III.

1.3 Modeling of Exposure Through Terrestrial and Aquatic Fate and Transport Models

Atmospheric concentrations and deposition rates were used as inputs to a series of terrestrial and aquatic models described in U.S. EPA's (1990) Methodology for Assessing Health Risks to Indirect Exposure from Combustor Emissions and a 1994 Addendum. The results of these terrestrial and aquatic models were used to predict mercury exposure to hypothetical humans through inhalation, consumption of drinking water and ingestion of soil, farm products (e.g., beef product and vegetables)

and fish. These models were also used to predict mercury exposure in hypothetical piscivorous (i.e., fish-eating) birds and mammals through consumption of fish.

1.4 Exposure Modeling Rationale

This section explains the decision to estimate mercury exposure based on the results of fate and transport modeling of stack emissions from anthropogenic sources rather than attempting an assessment based on monitoring data.

Exposure to mercury for the purpose of this assessment may be broadly defined as chemical contact with the outer boundary of an organism (also called a receptor). An organism's contact with mercury may occur through several different exposure routes including dermal, inhalation and oral. The assessment of mercury exposure is complicated by the physical and chemical properties of this naturally occurring element; factors include the different physical forms manifested in the environment, the different oxidative states exhibited and the duality of its environmental behavior as both a metallic and an organic compound. In addition the uncertain accuracy of analytical techniques, particularly at low environmental concentrations, and problems with contamination during environmental sampling complicate an assessment of exposure.

Mercury is generally present as a low-level contaminant in combustion materials; for example, coal, medical wastes or municipal solid wastes. Unlike dioxin it is not created during the combustion process but is released by it. It is difficult to control mercury emissions from the source because at temperatures typical of many combustion and manufacturing processes, mercury is emitted in a gaseous form rather than a particulate form.

Anthropogenic mercury emissions are not the only source of mercury to the atmosphere. Mercury, under certain conditions, may be introduced into the atmosphere through volatilization from natural sources such as lakes and soils; for example, some areas in the western U.S. appear to have naturally elevated mercury levels. Consequently, it is difficult to trace the source(s) of the mercury in environmental media and biota and estimate the impact of any one source type.

Existing environmental concentrations are a potential source of mercury exposure to both humans and animal species. These existing environmental concentrations, often referred to as background mercury concentrations, were generally not modeled in this effort. One of these existing sources of exposure, fish methylmercury concentrations, is utilized in Appendix H along with fish consumption data to estimate a distribution of U.S. population exposures to methylmercury that result from the consumption of fish. It is generally difficult to attribute methylmercury in fish to specific sources. There are clearly other types of assessments in which the knowledge of other background mercury sources is of critical importance; e.g., a site-specific risk assessment. For a site-specific assessment knowledge of total mercury exposure is critical and existing mercury levels in environmental media and biota should be quantified and included in the assessment. An assessment of total mercury exposure is not the aim of this modeling analysis; therefore, the inclusion of background exposures would only complicate the interpretation of the results. Similarly, an assessment of exposure to mercury in combustion ash is also not the aim of this document and this potential exposure source is not addressed.

Mercury has always been present at varying levels in environmental media and biota, and all mercury is, in a sense, naturally occurring; that is mercury is not a substance of human origin. Anthropogenic activities are thought to redistribute mercury from its original matrix through the atmosphere to other environmental media. Numerous studies indicate that the amount of mercury

being deposited from the atmosphere has increased since the onset of the industrial age (Nater and Grigal, 1992; Johansson et al., 1991; Swain et al., 1992). Some of the deposited mercury arises from natural sources and some from anthropogenic activities. Separating the "naturally occurring" background component from the "anthropogenic" component of background for the entire U.S. is impossible at this point in time. One could attempt to model both the current "naturally occurring" and the "anthropogenic" background levels in soil, water bodies and biota by adding an additional mercury load to these media before modeling the anthropogenic sources. Another approach to modeling background would involve the addition of a mercury load to the atmosphere only; this step would be followed by the modeling of the atmospheric transport and the deposition of the "added" mercury as well as the subsequent accumulation in soil, water bodies and biota. Again, this was not attempted for this modeling effort.

Many different yet valid approaches may be used to obtain estimates of mercury exposure. These include the following: direct measurement of mercury concentrations in source emissions (e.g., stack monitoring data), in environmental media (e.g., air, soil or water monitoring) and biota (e.g., fish and flora); direct measurement of mercury concentrations at the expected points of receptor contact (e.g., house or office air monitoring data, measurement of receptor food sources or drinking water); and direct measurements of mercury concentrations in the tissues of human and wildlife receptors (e.g., hair samples, feather samples, muscle samples and leaf samples).

It was decided to model the emissions data from the stacks of combustion sources and industrial processes rather than use the existing measurement data alone. Extant measured mercury data alone were judged insufficient to assess adequately the impact of anthropogenic mercury releases on human and wildlife exposures, the primary goal of the study. The discomfort with the data arose from the lack of extensive measurement data near U.S. anthropogenic sources of concern. It is likely that this data will be available in the near future.

This assessment utilizes the results of measured mercury emissions from selected anthropogenic sources to estimate exposure. The emissions inventory used in this assessment is found in Volume II of the Report to Congress. Using a series of fate, transport and exposure models and hypothetical constructs, the mercury concentrations in environmental media and pertinent biota were estimated. Ultimately mercury contact with human and wildlife receptors was estimated. In Chapter 4 of this document an effort was made to estimate the amount of contact with mercury as well as the oxidative state and form of mercury contacted. No attempt was made to estimate an internal dose for either the animal or human receptors.

There is a great deal of uncertainty in the modeling approach selected to estimate exposure. There is uncertainty in both the predicted fate and transport of this metal and the ultimate estimates of exposure. This uncertainty can be divided into modeling uncertainty and parameter uncertainty. Parameter uncertainty can be further subdivided into uncertainty and variability depending upon the degree to which a particular parameter is understood. Research needs are identified toward reducing these key uncertainties and are presented in Chapter 6.

1.5 Factors Important in Estimation of Mercury Exposure

Factors important in the estimation of mercury exposures in this study are listed in Table 1-1. This Table briefly describes the possible effects of these factors on the fate, transport and exposure to mercury and the means by which these were addressed. More details are provided in subsequent sections describing the modeling analyses.

Table 1-1
Factors Potentially Important in Estimating Mercury Exposure and How They are Addressed in This Study

Factor	Importance and Possible Effect on Mercury Exposure	Means of Addressing in this Study
Type of anthropogenic source of mercury	Different combustion and industrial process sources are anticipated to have different local scale impacts due to physical source characteristics (e.g., stack height), the method of waste generation (e.g., incineration or mass burn) or mercury control devices and their effectiveness.	Six main source categories, with a total of 11 different source types, selected based on their estimated annual mercury emissions or potential to be localized point sources of concern.
Mercury emission rates from stack	Increased emissions will result in a greater chance of adverse impacts on environment.	Emissions of model plants based on emissions inventory.
Mercury species emitted from stack	More soluble species will tend to deposit closer to the source.	Two species considered to be emitted from source: elemental and divalent mercury
Form of mercury emitted from stack	Transport properties can be highly dependent on form.	Both vapor and particle-bound fractions considered.
Deposition differences between vapor and particulate-bound mercury	Vapor-phase forms may deposit significantly faster than particulate-bound forms.	Deposition (wet and dry) of vapor-phase forms calculated separately from particulate-bound deposition.
Transformations of mercury after emission from source	Relatively nontoxic forms emitted from source may be transformed into more toxic compounds.	Equilibrium fractions estimated in all environmental media for three mercury species: elemental mercury, divalent species, and methylmercury.
Facility locations	Effects of meteorology and terrain may be significant.	Both a humid and less humid site considered. Effect of terrain on results addressed separately.
Type of human activity patterns	Some populations may be more highly exposed to various forms of mercury.	Three main types addressed: a rural scenario, an urban scenario, and a fishing scenario. Both an "average" and "high-end" scenario considered within each human activity pattern.
Location relative to local mercury source	Receptors located downwind are more likely to have higher exposures. Influence of distance depends on source type.	Three distances in downwind direction considered.
Contribution from non-local sources of mercury	Important to keep predicted impacts of local sources in perspective.	Results of local mercury source are combined with estimate of impact from non-local sources from RELMAP.
Uncertainty	Reduces confidence in ability to estimate exposure accurately.	Probabilistic capabilities possible for any combination of sources and scenarios. In the current study, limited uncertainty analyses conducted for major aspects of atmospheric transport modeling .

1.6 Estimated Human Exposure through the Consumption of Fish

The assessment of human mercury exposure through the consumption of fish as described in appendix H utilizes data from the 1989 - 1991 Continuing Surveys of Food Intake by Individuals (CSFII 89-91) to estimate a range of fish consumption rates among fish eaters. For each fish-eater, the 3-day CSFII 89-91 study identified the number of fish meals, the quantities and species of fish consumed and the self-reported body weights of the consumers. The constitution of the survey population was weighted to reflect the actual U.S. population.

These estimates of fish consumption rates were combined with species-specific statistical means of chemically-analyzed methylmercury concentrations in fish. The fish methylmercury concentration data were obtained from the National Marine Fisheries Service, (Bahnick et al., (1994), FDA 1992 and Lowe et al., (1985)). Through the application of specific fish preparation factors (USDA, 1995), estimates of the range of methylmercury exposure from the consumption of fish were prepared for the fish-consuming segment of the U.S. population. Per body weight estimates of methylmercury exposures were determined by dividing the total daily methylmercury exposure from this pathway by the self-reported body weights. Per body weight estimates of exposure are utilized in the modeling in Chapter 3.

1.7 Definition of Terms

Definitions for the following terms related to the fate and transport of mercury were largely adapted from the Expert Panel on Mercury Atmospheric Processes (1994), and EPA (1975, 1976).

Anthropogenic Mercury Emissions

The mobilization or release of mercury by human activity that results in a mass transfer of mercury to the atmosphere.

Bioaccumulation Factor

The equilibrium concentration of a chemical in a biological medium divided by the equilibrium concentration of a chemical in an environmental medium. While similar to a bioconcentration factor, a bioaccumulation factor is designed not only to predict chemical uptake through direct contact with or uptake from an environmental medium, but also to account for any food chain pathways that may in some manner connect the environmental medium to the biological medium of interest.

Bioavailability

The state of being capable of being absorbed and available to interact with the metabolic processes of an organism. Bioavailability is typically a function of chemical properties, the physical state of the material to which an organism is exposed, and the ability of the individual organism to physiologically take up the chemical.

Bioconcentration Factor

The equilibrium concentration of a chemical in a biological medium divided by the equilibrium concentration of a chemical in an environmental medium. The parameter is typically used to predict chemical uptake through contact with or uptake from an environmental medium.

Biotransfer Factor

The equilibrium concentration of a chemical in animal tissue divided by the daily intake of the chemical.

Contact Rate

The frequency of an exposure. Generally expressed as the product of an amount of a medium per event and the number of events per a given unit of time.

Current Background Mercury Concentrations

Concentrations of mercury in the abiotic and biotic components of the environment that have resulted from natural mercury concentrations and anthropogenic activities.

Erosion

The removal of soil particles by wind and water. Water erosion is usually characterized by one or more of the following types of erosion: raindrop erosion, sheet erosion, rill erosion, gully erosion, and streambank erosion. Raindrops start soil erosion by detaching soil particles. They aggravate soil erosion by compacting the soil surface and reducing its ability to infiltrate water. Sheet erosion is the removal of a thin layer of soil resulting from sheet flow of water. It has a high transport capability. Rill erosion is on steeper slopes where channels with depths of up to one foot are formed. Gully erosion represents an advanced form of soil erosion from concentrated storm runoff. Streambank erosion is the erosion of soil from stream channels, both on the banks and on the stream beds.

Exposure

Contact of a chemical, physical or biological agent with the outer boundary of an organism. Exposure is quantified as the concentration of the agent in the medium in contact, integrated over the time duration of the contact.

Exposure Scenario

A set of facts, assumptions, and inferences about how exposure takes place that aids the exposure assessor in evaluating estimating, or quantifying exposures.

Natural Background Mercury Concentrations

Concentrations of mercury in the abiotic and biotic components of the environment that resulted from natural mercury concentrations. These concentrations existed prior to the onset of anthropogenic activities.

Natural Mercury Emissions

The mobilization or release of mercury from environmental sources by natural biotic or abiotic activities that results in a mass transfer of mercury to the atmosphere.

Pathway

The physical course a chemical or pollutant takes from the source to the exposed organism.

Re-emitted Mercury

Mass transfer of mercury to the atmosphere by biotic and geological processes drawing on a pool of mercury that was deposited to the earth's surface after initial mobilization by either anthropogenic or natural activities.

Mercury Dry Deposition

Mass transfers of gaseous, aerosol or particulate mercury species from the atmosphere to the earth's surface (either aquatic or terrestrial, including vegetation) in the absence of precipitation.

Mercury Wet Deposition

Mass transfers of dissolved gaseous or suspended particulate mercury species from the atmosphere to the earth's surface (either aquatic or terrestrial) by precipitation.

Local Scale

A relative term, used to describe the area within which emissions travel within one diurnal cycle (generally 100 Km from source but for this analysis 50 Km from the source). Local influences are characterized by measurable pollutant concentration gradients with relatively

large fluctuations in air concentrations caused by meteorological factors such as wind direction.

Regional Scale

A relative term, used to describe the area within which emissions travel in more than one diurnal cycle (generally 100 to 2000 Km from a source). The regional scale describes areas sufficiently remote or distant from large emission sources so that concentration fields are rather homogeneous, lacking measurable gradients.

Runoff

That portion of the precipitation that appears in surface streams. Surface runoff (or overland flow) is water that travels over the ground surface. Subsurface runoff (interflow, storm seepage) is water that has infiltrated the surface soil and moved laterally through the upper soil horizons. Groundwater runoff is water that has infiltrated the surface soil, percolated to the general groundwater table, and then moved laterally to the water body.

2. CHEMICAL PROPERTIES AND MEASURED ENVIRONMENTAL CONCENTRATIONS OF MERCURY

Many studies have examined the environmental behaviors of various mercury species. Other studies have been conducted that attempt to quantify the amount of mercury and the species of mercury present in environmental media and biota. This section describes the chemical properties of mercury and analytical measurement methods and then discusses the global mercury cycle. In the discussion of the mercury cycle the estimated impacts of both anthropogenic and natural mercury atmospheric emissions are described. This conceptualization is followed by a presentation of measured mercury data in environmental media and biota and by a discussion of efforts to quantify mercury levels in the environmental media and biota around anthropogenic sources.

2.1 Chemical Properties of Mercury

Elemental mercury is a heavy, silvery-white liquid metal at typical ambient temperatures and pressures. The vapor pressure of mercury metal is strongly dependent upon temperature, and it vaporizes readily under ambient conditions. Its saturation vapor pressure of 14 mg/m^3 greatly exceeds the average permissible concentrations for occupational (0.05 mg/m^3) or continuous environmental exposure (0.015 mg/m^3) (Nriagu, 1979; WHO, 1976). Elemental mercury partitions strongly to air in the environment and is not found in nature as a pure, confined liquid. Most of the mercury encountered in the atmosphere is elemental mercury vapor.

Mercury can exist in three oxidation states: Hg^0 (metallic), Hg_2^{2+} (mercurous), and Hg^{2+} (mercuric-Hg(II)). The properties and chemical behavior of mercury strongly depend on the oxidation state. Mercurous and mercuric mercury can form numerous inorganic and organic chemical compounds; however, mercurous mercury is rarely stable under ordinary environmental conditions. Mercury is unusual among metals because it tends to form covalent rather than ionic bonds. Most of the mercury encountered in water/soil/sediments/biota (all environmental media except the atmosphere) is in the form of inorganic mercuric salts and organomercurics. Organomercurics are defined by the presence of a covalent C-Hg bond. The presence of a covalent C-Hg bond differentiates organomercurics from inorganic mercury compounds that merely associate with the organic material in the environment but do not have the C-Hg bond. The compounds most likely to be found under environmental conditions are these: the mercuric salts $HgCl_2$, $Hg(OH)_2$ and HgS; the methylmercury compounds, methylmercuric chloride (CH_3HgCl) and methylmercuric hydroxide (CH_3HgOH); and, in small fractions, other organomercurics (i.e., dimethylmercury and phenylmercury).

Mercury compounds in the aqueous phase often remain as undisassociated molecules, and the reported solubility values reflect this. Solubility values for mercury compounds which do not disassociate are not based on the ionic product. Most organomercurics are not soluble and do not react with weak acids or bases due to the low affinity of the mercury for oxygen bonded to carbon. CH_3HgOH, however, is highly soluble due to the strong hydrogen bonding capability of the hydroxide group. The mercuric salts vary widely in solubility. For example $HgCl_2$ is readily soluble in water, and HgS is as unreactive as the organomercurics due to the high affinity of mercury for sulfur. A detailed discussion of mercury chemistry can be found in Nriagu (1979) and Mason et al. (1994).

2.2 Analytic Measurement Methods

A number of methods can be employed to determine mercury concentrations in environmental media. The concentrations of total mercury, elemental mercury, organic mercury compounds (especially methylmercury) and chemical properties of various mercuric compounds can be measured, although speciation among mercuric compounds is not usually attempted. Recent, significant

improvements and standardizations in analytical methodologies enable reliable data on the concentration of methylmercury, elemental mercury and the mercuric fraction to be separated from the total mercury in environmental media. It is possible to speciate the mercuric fraction further into reactive, non-reactive and particle-bound components. It is generally not possible to determine which mercuric species is present in environmental media (e.g., HgS or $HgCl_2$).

One of the significant advances in mercury analytical methods over the past decade or so has been in the accurate detection of mercury at low levels (less than 1 ug/g). Over the past two decades mercury determinations have progressed from detection of µg levels of total mercury to picogram levels of particular mercury species (Mitra, 1986 and Hovart et al., 1993a and 1993b). Typical detection limits for data used or presented in this study are on the order of 1 - 2 ng/L for water samples (Sorensen et al., 1994), 0.1 ng/g for biota (Cappon, 1987; Bloom, 1992) and 0.1 ng/m^3 for atmospheric samples (Lindberg et al., 1992). Mercury contamination of samples has been shown to be a significant problem in past studies. The use of ultra-clean sampling techniques is critical for the more precise measurements required for detection of low levels of mercury.

2.3 Mercury in the Environment

As a naturally occurring element, mercury is present throughout the environment in both environmental media and biota. Nriagu (1979) estimated the global distribution of mercury and concluded that by far the largest repository is ocean sediments. Nriagu estimated that the ocean sediments may contain about 10^{17} g of mercury, mainly as HgS. Nriagu also estimated that ocean waters contain around 10^{13} g, soils and freshwater sediments 10^{13} g, the biosphere 10^{11} g (mostly in land biota), the atmosphere 10^{8} g and freshwater on the order of 10^{7} g. This budget excludes "unavailable" mercury in mines and other subterranean repositories.

Mercury is emitted by both anthropogenic and natural processes. Due to its chemical properties, environmental mercury is thought to move through various environmental compartments, possibly changing form and species during this process. Like other elements such as nitrogen, these movements are conceptualized as a cycle.

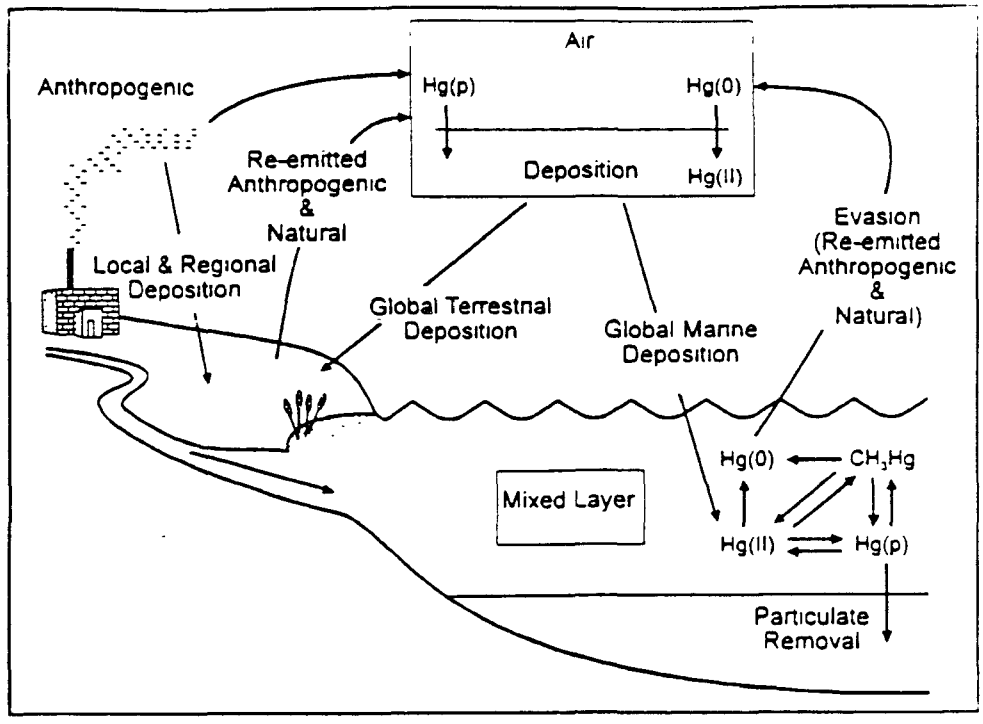
The mercury cycle has been studied and described in several recent reports (Swedish EPA, 1991; Mitra, 1986; Fitzgerald and Clarkson, 1991), and its understanding continues to undergo refinement. The movement and distribution of mercury in the environment can be confidently described only in general terms. There has been increasing consensus on many, but not all, of the detailed behaviors of mercury in the environment (Brosset and Lord, 1991). The depiction of the mercury cycle in Figure 2-1 attempts to illustrate mercury release by both natural and anthropogenic sources into environmental media: air, soil, and water. The figure illustrates the various transport and transformation processes that are expected to occur and includes a number of infinite and/or indefinite loops.

In sections 2.3.1 through 2.3.6 information about the mercury cycle is summarized as it directly relates to the present study: anthropogenic source release to the atmosphere and the resulting exposure to humans and wildlife from the inhalation and ingestion pathways. It is important to note that it is not possible to know exactly what will happen to the stack-released mercury, but enough is known about the speciation and cycling of mercury in the environment at this time to propose a plausible scenario.

2.3.1 Emissions of Mercury

Mercury is emitted to the atmosphere through both naturally occurring and anthropogenic processes. Natural processes include volatilization of mercury in marine and aquatic environments,

Figure 2-1
The Mercury Cycle -



volatilization from vegetation, degassing of geologic materials (e.g., soils) and volcanic emissions. The natural emissions are thought to be primarily in the elemental mercury form. Conceptually, the current natural emissions can arise from two components: mercury present as part of the pre-industrial equilibrium and revolatilized anthropogenic emissions.

Several authors have used a number of different techniques to estimate the pre-industrial mercury concentrations in environmental media. It is difficult to separate current mercury concentrations by origin (i.e., anthropogenic or natural) because of the continuous cycling of the element in the environment. For example, stack releases of elemental mercury may be oxidized and deposit as divalent mercury far from the source; the deposited mercury may be reduced and re-emitted as elemental mercury only to be deposited again continents away. Not surprisingly, there is a broad range of estimates and a great deal of uncertainty with each. When the estimates are combined, they indicate that between 40 and 75% of the current atmospheric mercury concentrations are the result of anthropogenic releases. The Expert Panel on Mercury Atmospheric Processes (1994) concluded that pre-industrial atmospheric concentrations constitute approximately one-third of the current atmospheric concentrations. They estimated that anthropogenic emissions may currently account for 50 - 75% of the total annual input to the global atmosphere (Expert Panel on Mercury Atmospheric Processes, 1994). The estimates of the panel are corroborated by Lindqvist et al., (1991), who estimated that 60% of the current atmospheric concentrations are the result of anthropogenic emissions and Porcella (1994), who estimated that this fraction was 50%. Hovert et al., (1993b) assessed the anthropogenic

fraction as constituting 40 to 50% of the current total. This overall range appears to be in agreement with the several fold increase noted in inferred deposition rates (Swain et al., 1992; Engstrom et al., 1994; Benoit et al., 1994). The percentage of current total atmospheric mercury which is of anthropogenic origin may be much higher near mercury emissions sources.

Anthropogenic mercury releases are thought to be dominated on the national scale by industrial processes and combustion sources that release mercury into the atmosphere. Stack emissions are thought to include both vapor and particulate forms as well as divalent and elemental mercury species in various fractions. The analytic methods for mercury speciation of exit gasses and emission plumes are being refined, and there is still controversy in this field. Chemical reactions occurring in the emission plume are also possible. The speciation of mercury emissions is thought to depend on the fuel used (e.g., coal, oil, municipal waste), flue gas cleaning and operating temperature. The exit stream is thought to range from almost all divalent mercury to nearly all elemental mercury; the elemental mercury is primarily in the gas phase although exit streams containing soot can bind up some fraction of elemental mercury. The divalent fraction is split between gaseous and particle bound phases (Lindqvist et al., 1991, Chapter 4). Much of this divalent mercury is thought to be $HgCl_2$ (Michigan Environmental Science Board, 1993).

An emission factor-based approach was used to develop the nationwide emission estimates for the source categories presented in Table 2-1. The emission factors presented are estimates based on ratios of mass mercury emissions to measures of source activities and nation-wide source activity levels. Details of the emission factor approach are described in Volume II of this Report to Congress. The reader should note that the data presented in this table are estimates; uncertainties occur in the measurement techniques, emission factors, estimates of pollutant control efficiency and nation-wide source class activity levels. The estimates may also be based on limited information for a particular source class, thereby increasing the uncertainty in the estimate further.

Some anthropogenic processes no longer used still result in significant environmental releases from historically contaminated areas which continue to release mercury to surface water runoff, groundwater and the atmosphere. It is estimated that the mercury content of typical lakes and rivers has been increased by a factor of two to four since the onset of the industrial age (Nriagu, 1979). More recently, researchers in Sweden estimate that mercury concentrations in soil, water and lake sediments have increased by a factor of four to seven in southern Sweden and two to three in northern Sweden in the 20th century (Swedish EPA 1991). It is estimated that present day mercury deposition is two to five times greater now than in preindustrial times (Lindqvist et al., 1991).

2.3.2 Deposition of Mercury

The divalent species emitted, either in the vapor or particulate phase, are thought to be subject to much faster atmospheric removal than elemental mercury (Lindberg et al., 1991, Shannon and Voldner, 1994). Particulate bound divalent mercury is assumed to dry deposit (this is defined as deposition in the absence of precipitation). The deposition velocity is dependent on atmospheric conditions and particle size. Particulate mercury is also assumed to be subject to wet deposition due to scavenging by precipitation. The gaseous divalent mercury emitted is also expected to be scavenged readily by precipitation. Divalent mercury species have much lower Henry's law constants than elemental mercury, and thus are assumed to partition strongly to the water phase. Dry deposition of gas phase divalent mercury is thought to be significant due to its reactivity with surface material. Overall, gas phase divalent mercury is more rapidly and effectively removed by both dry and wet deposition than particulate divalent mercury (Lindberg et al., 1992; Petersen et al., 1995; Shannon and Voldner, 1994), a result of the reactivity and water solubility of gaseous divalent mercury.

Table 2-1
Annual Estimates of Mercury Release by Various Combustion and Manufacturing Source Classes (U.S. EPA, 1996a)

Source	Annual Mercury Emission Rate
Combustion Sources - Total	196,400 kg/yr (216 tons/yr)
Electric utilities	
Oil- and Gas-fired	2600 kg/yr (4 tons/yr)
Coal-fired	44,600 kg/yr (49 tons/yr)
Incinerators	
Municipal waste combustors	57,700 kg/yr (63.5 tons/yr)
Medical waste incinerators	58,800 kg/yr (64.7 tons/yr)
Commercial/Industrial boilers	26,400 kg/yr (29 tons/yr)
Chlor-alkali production	5,900 kg/yr (6.5 tons/yr)
Primary lead smelting	8,200 kg/yr (9 tons/yr)
Primary copper smelting	600 kg/yr (0.7 tons/yr)
Other combustion sources	5,100 kg/yr (5.5 tons/yr)
Other sources	19,000 kg/yr (20.8 tons/yr)

In contrast, elemental mercury vapor is not thought to be susceptible to any major process resulting in direct deposition to the earth's surface. Elemental mercury is thought to have a strong tendency to remain airborne. On non-assimilating surfaces elemental mercury deposition appears negligible (Lindberg et al., 1992), and though elemental mercury can be formed in soil and water due to the reduction of divalent mercury species by various mechanisms, this elemental mercury is expected to volatilize into the atmosphere (Expert Panel on Mercury Atmospheric Processes 1994). In fact, it has been suggested that *in-situ* production and afflux of elemental mercury could provide a buffering role in aqueous systems, as this would limit the amount of divalent mercury available for methylation (Fitzgerald, 1994). Water does contain an amount of dissolved gaseous elemental mercury (Fitzgerald et al., 1991), but it is minor in comparison to the total mercury content in freshwater (dissolved + particulate).

There appears to be a potential for deposition of elemental mercury via plant-leaf uptake. Lindberg et. al. (1992) indicated that forest canopies could accumulate elemental mercury vapor, via gas exchange at the leaf surface followed by mercury assimilation in the leaf interior during the daylight hours. This process causes a downward flux of elemental mercury from the atmosphere, resulting in a deposition velocity. Recent evidence (Hanson et al., 1994) indicates that this does occur but only at air concentrations of elemental mercury well above background for a typical forest area. At more common mercury levels, the forest appears to act as a source of elemental mercury to the atmosphere, with the measured mercury flux in the upward direction. Lindberg et. al. (1991) noted

this may be explained by the volatilization of elemental mercury from the canopy/soil system, most likely the soil. Hanson et al. (1994) stated that "dry foliar surfaces in terrestrial forest landscapes may not be a net sink for atmospheric elemental mercury, but rather as a dynamic exchange surface that can function as a source or sink dependent on current mercury vapor concentrations, leaf temperatures, surface condition (wet versus dry) and level of atmospheric oxidants." Similarly, Mosbaek et al. (1988) convincingly showed that most of the mercury in leafy plants is due to air-leaf transfer, but that for a given period of time the amount of elemental mercury released from the plant-soil system greatly exceeds the amount collected from the air by the plants. It is also likely that many plants accumulate airborne mercury to certain concentrations, after which net deposition of elemental mercury does not occur. Overall, dry deposition of elemental mercury does not appear to be a significant pathway for removal of atmospheric mercury, although approximately 95% or more of atmospheric mercury is elemental mercury (Fitzgerald, 1994).

There is a pathway however, by which elemental mercury vapor released into the atmosphere may (eventually) result in increased methylmercury concentrations in fish. Reactions occur in the atmosphere in the aqueous phase (cloud droplets) that both oxidize elemental mercury to divalent mercury and reduce the divalent mercury to elemental mercury. The most important reactions are the oxidation of elemental mercury with ozone, reduction of divalent mercury by sulfite (SO_3^{-2}) ions or production of particulate divalent mercury by complexation with soot:

$Hg^0(g) \rightarrow Hg^0(aq)$
$Hg^0(aq) + O_3(aq) \rightarrow Hg(II)(aq)$
$Hg(II)(aq) + \text{soot/possible evaporation} \rightarrow Hg(II)(p)$
$Hg(II)(aq) + SO_3^{-2}(aq) \rightarrow Hg^0(aq)$
(g) = gas phase molecule
(aq) = aqueous phase molecule
(p) = particulate phase molecule

The Hg(II) produced from oxidation of Hg^0 by ozone can be reduced back to Hg^0 by sulfite; however, the oxidation of Hg^0 by ozone is a much faster reaction than the reduction of Hg(II) by sulfite. Thus, a steady state concentration of Hg(II)(aq) is built up in the atmosphere and can be expressed as a function of the concentrations of $Hg^0(g)$, $O_3(g)$, H^+ (representing acids) and $SO_2(g)$ (Lindqvist et al., 1991, Chapter 6). Note that H^+ and $SO_2(g)$, although not apparent in the listed atmospheric reactions, control the formation of sulfite.

The Hg(II)(aq) produced would then be susceptible to atmospheric removal via wet deposition. The third reaction, however, may transform most of the Hg(II)(aq) into the particulate form, due to the much greater amounts of soot than mercury in the atmosphere. The soot concentration will not be limiting compared to the concentration of Hg(II)(aq), and S atoms in the soot matrix will bond readily to the Hg(II)(aq). The resulting Hg(II)(p) can then be removed from the atmosphere by wet deposition (if the particle is still associated with the cloud droplet) or dry deposition (following cloud droplet evaporation). This transformation of Hg^0 to Hg(II)(p) demonstrates a possible mechanism by which natural and anthropogenic sources of Hg^0 vapor can result in mercury deposition to land and water. The deposition can occur far from the source due to the overall slow rate of conversion. It has been suggested that this mechanism is important in a global sense for Hg pollution, while wet deposition of anthropogenic particulate Hg(II) is the most important locally (Fitzgerald, 1994; Lindqvist et al., 1991, Chapter 6), with gaseous Hg(II) expected to deposit at a faster rate after release than Hg(II)(p). Overall, an atmospheric residence time of 1/2 - 2 years for elemental mercury to as little as hours for some Hg(II) species (Lindqvist and Rodhe, 1985) is expected. This behavior is observed in the modeling results presented in this effort as well. It is possible that dry deposition of Hg^0 can occur from ozone mediated oxidation of elemental mercury taking place on wet surfaces, but this is not expected to be comparable in magnitude to the cloud droplet mediated processes (Lindberg, 1994).

2.3.3 Mercury in Soil

Once deposited, the Hg(II) species are subject to a wide array of chemical and biological reactions. Soil conditions (e.g., pH, temperature and soil humic content) are typically favorable for the formation of inorganic Hg(II) compounds such as $HgCl_2$, $Hg(OH)_2$ and inorganic Hg(II) compounds complexed with organic anions (Schuster 1991). Although inorganic Hg(II) compounds are quite soluble (and, thus, theoretically mobile) they form complexes with soil organic matter (mainly fulvic and humic acids) and mineral colloids; the former is the dominating process. This is due largely to the affinity of Hg(II) and its inorganic compounds for sulfur-containing functional groups. This complexing behavior greatly limits the mobility of mercury in soil. Much of the mercury in soil is bound to bulk organic matter and is susceptible to elution in runoff only by being attached to suspended soil or humus. Some Hg(II), however, will be absorbed onto dissolvable organic ligands and other forms of dissolved organic carbon (DOC) and may then partition to runoff in the dissolved phase. Currently, the atmospheric input of mercury to soil is thought to exceed greatly the amount leached from soil, and the amount of mercury partitioning to runoff is considered to be a small fraction of the amount of mercury stored in soil. The results of Appendix C, which detail the calibration of soil-water partition coefficients in the watershed model, are consistent with these observations. The affinity of mercury species for soil results in soil acting as a large reservoir for anthropogenic mercury emissions (Meili et al., 1991 and Swedish EPA 1991). For example, note the mercury budget proposed by Meili et al., 1991. Even if anthropogenic emissions were to stop entirely, leaching of mercury from soil would not be expected to diminish for many years (Swedish EPA, 1991). Hg^0 can be formed in soil by reduction of Hg(II) compounds/complexes mediated by humic substances (Nriagu, 1979). This Hg^0 will vaporize eventually and re-enter the atmosphere. Methylmercury can be formed by various microbial processes acting on Hg(II) substances. Approximately 1-3% of the total mercury in surface soil is methylmercury, and as is the case for Hg(II) species, it will be bound largely to organic matter. The other 97-99% of total soil mercury can be considered largely Hg(II) complexes, although a small fraction of Hg in typical soil will be Hg^0 (Revis et al., 1990). The methylmercury percentage exceeded 3% (Cappon, 1987) in garden soil with high organic content under slightly acidic conditions. Contaminated sediments may also contain higher methylmercury percentages compared to ambient conditions (Wilken and Hintelmann, 1991; Parks et al., 1989).

2.3.4 Plant and Animal Uptake of Mercury

The Hg(II) and methylmercury complexes in soil are available theoretically for plant uptake and translocation, potentially resulting in transfer through the terrestrial food chain. In reality plant uptake from ordinary soils, especially to above-ground parts of plants, appears to be insignificant (Schuster, 1991; Lindqvist et al., 1991, Chapter 9). Mosbaek et al. (1988) determined (by spiking soil with Hg^{203}) that the atmospheric contribution of the total mercury content of the leafy parts of plants is on the order of 90-95% and for roots 30-60%. The concentrations of mercury in leafy vegetables generally exceeds that of legumes and fruits (Cappon 1981, 1987), where it is not clear whether the mercury content results from air and/or soil uptake. Most plant uptake studies do not explicitly measure both the surrounding soil and air concentrations as performed in Mosbaek et al., 1988. Even when this is performed there is no way to determine whence the mercury in the plant originated. Speciation data do not provide much information; apparently any Hg^0 absorbed from the air is readily converted to Hg(II) in the plant interior, since even leafy vegetables do not appear to contain any Hg^0 (Cappon, 1987). Plants also have some mercury methylation ability (Fortmann et al., 1978), so the percentage of methylmercury in plants may not originate from root uptake. Studies which report plant uptake from soil have typically been conducted on heavily polluted soils near Chlor-alkali plants (Lenka et al., 1992; Temple and Linzon 1977; Lindberg et al., 1979), where the formation of Cl^- complexes can increase Hg(II) movement somewhat. Overall, mercury concentrations in plants, even those whose main uptake appears to be from the air, are small (see ambient mercury concentrations

tables). Accordingly, livestock typically accumulates little mercury from foraging or silage/grain consumption, and mercury content in meat is low (see tables in the ambient mercury concentrations section).

2.3.5 Mercury in the Freshwater Ecosystem

There are a number of pathways by which mercury can enter the freshwater environment: Hg(II) and methylmercury from atmospheric deposition (wet and dry) can enter water bodies directly; Hg(II) and methylmercury can be transported to water bodies in runoff (bound to suspended soil/humus or attached to dissolved organic carbon); or Hg(II) and methylmercury can leach into the water body from groundwater flow in the upper soil layers. Once in the freshwater system, the same complexation and transformation processes that occur to mercury species in soil will occur in aquatic medium along with additional processes due to the aqueous environment. Mercury concentrations are typically reported for particular segments of the water environment; the most common of which are the water column (further partitioned as dissolved or attached to suspended material), the underlying sediment (further divided into surface sediments and deep sediments); and biota (particularly fish). Discussion of several detailed studies on the movement of mercury between soil/water/sediment and how modeling results compare to this data are presented in Appendix C.

Partition coefficients have been calculated for the relative affinity of Hg(II) and methylmercury for sediment or soil over water. Values of the partition coefficient K_d (concentration of Hg in dry sediment, soil or suspended matter divided by the dissolved concentration in water) on the order of 10-100,000 ml/g soil, 100,000 ml/g sediment and 100,000+ ml/g suspended material are typically found for Hg(II) and methylmercury (Appendices A and C), indicating a strong preference for Hg(II) and methylmercury to remain bound up to soil, bottom sediment or suspended matter (increasing affinity in that order). Of course, a river or lake freshwater system often has more water than sediment, and a significant amount of Hg(II) entering a water system may partition to the water column, especially if there is a high concentration of suspended material in the water column. It is often unclear whether the mercury in sediment will be $HgCl_2$ or $Hg(OH)_2$ organic complexes, which can be considered more susceptible to methylation, or will be the more unreactive HgS and HgO forms.

Most of the mercury in the water column (Hg(II) and methylmercury) will be bound to organic matter, either to dissolved organic carbon (DOC; consisting of fulvic and humic acids, carbohydrates, carboxylic acids, amino acids and hydrocarbons; (Lindqvist et al., 1991, (Chapter 2)) or to suspended particulate matter. In most cases, studies that refer to the dissolved mercury in water include mercury complexes with DOC. Studies indicate that about 25%-60% of Hg(II) and methylmercury organic complexes are particle-bound in the water column. The rest is in the dissolved, bound-to-DOC phase (Nriagu, 1979; Bloom et al., 1991; NAS 1977). Typically, total mercury and methylmercury concentrations are positively correlated with DOC concentrations in lake waters (Driscoll et al., 1994; Mierle and Ingram, 1991). Hg^0 is produced in freshwater by humic acid reduction of Hg(II) or demethylation of methylmercury. An amount will remain in the dissolved gaseous state while most will volatilize. As noted previously, Hg^0 constitutes very little of the total mercury in the water column but may provide a significant pathway for the evolution of mercury out of the water body via Hg(II) or methylmercury -> Hg^0 -> volatilization. For many lakes, however, sedimentation of the Hg(II) and methylmercury bound to particulate matter is expected to be the dominant process for removal of mercury from the water column (Sorensen et al., 1990; Fitzgerald et al., 1991).

Generally, no more than 25% of the total mercury in a water column exists as a methylmercury complex; typically, less than 10% is observed (see Appendix A). This is a result of methylation of Hg(II) which is thought to occur in the bottom sediment and the water column by microbial action and abiotic processes. An equilibrium is soon established between Hg(II) and methylmercury in freshwater systems; in a number of sediment-water systems, it has been found that

methylmercury concentrations in waters were independent of water column residence time or time in contact with sediments (Parks et al., 1989). Methylmercury in the water column which is lost through demethylation, exported downstream or taken up by biota is thought to be replaced by additional methylation of Hg(II) compounds to sustain equilibrium.

Once entering a water body, mercury can remain in the water column, be lost from the lake through drainage water, revolatilize into the atmosphere, settle into the sediment or be taken up by aquatic biota. After entry, the movements of mercury through any specific water body may be unique. Only mercury in the water column, the sediment and other aquatic biota appears to be available to aquatic organisms for uptake.

Methylation appears to be a key step in the entrance of mercury into the food chain (Sorrenson et al., 1990). The biotransformation of inorganic mercury species to methylated organic species in water bodies can occur in the sediment (Winfrey and Rudd, 1990) and the water column (Xun et al., 1987). Abiotic processes (e.g., humic and fulvic acids in solution) also appear to methylate the mercuric ion (Nagase et al., 1982). All mercury compounds entering an aquatic ecosystem are not methylated and demethylation reactions (Xun et al., 1987) as well as volatilization of dimethylmercury decrease the amount of methylmercury available in the aquatic environment. It is clear that there is a large degree of scientific uncertainty and variability among waterbodies concerning the processes that methylate mercury.

Bacterial methylation rates appear to increase under aerobic conditions, high temperatures (NJDEPE, 1993) and low pH (Xun et al., 1987; Winfrey and Rudd, 1990). Increased quantities of the mercuric species, the proper biologic community, and adequate suspended soil load and sedimentation rate are also important factors (NJDEPE, 1993). Anthropogenic acidification of lakes appear to increase methylation rates as well (Winfrey and Rudd, 1990).

Methylmercury is very bioavailable and accumulates in fish through the aquatic food web; nearly 100% of the mercury found in fish muscle tissue is methylated (Bloom et al., 1991). Methylmercury appears to be primarily passed to planktivorous and piscivorous fish via their diets. Larger, longer-lived fish species at the upper end of the food web typically have the highest concentrations of methylmercury in a given waterbody. A relationship exists between methylmercury content in fish and lake pH, with higher methylmercury content in fish tissue typically found in more acidic lakes (Winfrey and Rudd, 1990; Driscoll et al., 1994). The mechanisms for this behavior are unclear. Most of the total methylmercury production ends up in biota, particularly fish (Swedish EPA, 1991). In fact, bioconcentration factors (BCFs) for accumulation of methylmercury in fish (compared with the water methylmercury concentration) are on the order of 10^5 - 10^6 (Bloom, 1991; Appendix A). Overall, methylmercury production and accumulation in the freshwater ecosystem places this pollutant into a position to be ingested by fish-eating organisms.

2.3.6 Summary

Mercury released into the atmosphere from natural and anthropogenic sources deposits mainly as Hg(II), from either direct deposition of emitted Hg(II) or from conversion of emitted elemental Hg^0 to Hg(II) through ozone-mediated reduction. The former process may result in elevated deposition rates around atmospheric emission sources and the latter process results in regional/global transport followed by deposition. Measurements indicate that wet deposition of mercury is likely to be greater than dry deposition. There is still a great deal of uncertainty with respect to the amount of dry deposition of mercury. Once deposited, mercury appears to bind tightly to certain soil components. The deposited Hg(II) may revolatilize through reduction and be released back to the atmosphere as Hg^0. Soil Hg(II) may also be methylated to form methylmercury; these two forms may remain in the soil or be transported through the watershed to a waterbody via runoff and leaching. Mercury enters

the water body through direct deposition on the watershed and mercury in waterbodies has been measured in both the water column and the sediments. Hg(II) in the waterbody may also be methylated to form methylmercury; both Hg(II) and methylmercury may be reduced to form Hg^0 which is reintroduced to the atmosphere. An equilibrium is quickly established between Hg(II) and methylmercury in soil, water and sediment. Typical percentages for the fraction of mercury which is methylmercury are 2% in soil or sediment and 10% or less in water column (nearly all the remainder is Hg(II)).

The consumption of fish by humans and wildlife is the main exposure pathway of concern for mercury; the terrestrial pathway is not expected to be significant in comparison. Most plants do not appear to accumulate high concentrations of mercury from either air or soil; livestock also do not appear to accumulate high concentrations of mercury from soil or plant consumption. On the other hand, fish can bioaccumulate high concentrations of mercury in their muscle tissues; most of the bioaccumulated mercury is in the form of methylmercury. This bioaccumulation of methylmercury in fish muscle tissue occurs in waterbodies that are remote from emission sources and seemingly pristine as well as in waterbodies that are less isolated. Methylmercury appears to be efficiently passed through the aquatic food web to the highest trophic level consumers in the community (e.g., piscivorous fish). At this point it can be contacted by fish-consuming wildlife and humans through ingestion. Methylmercury appears to pass from the gastrointestinal tract into the bloodstream more efficiently than the divalent species.

2.4 Measurement Data

Based on the current understanding of the mercury cycle, mercury is thought to be transported primarily through the atmosphere and distributed to other compartments of the environment. The primary source of mercury in terrestrial, aquatic and oceanic environments appears to be the wet or dry deposition of atmospheric mercury. Once deposited, the mercury may be revolatilized back to the atmosphere, incorporated into the medium of deposit or transferred to other abiotic or biotic components of these environments.

Elemental mercury vapor is the most common form of mercury in the atmosphere and divalent mercury the most common in soils, sediments and the water column. The most common form in most biota is Hg(II); the exception is fish in which the most common form is methylmercury.

2.4.1 Mercury Air Concentrations

As noted in section 2.3.1 anthropogenic emissions are currently thought to account for between 40-75% of the total annual input to the global atmosphere (Expert Panel on Mercury Atmospheric Processes, 1994; Hovert et al., 1993b). Current air concentrations are thought to be 2 - 3 times pre-industrial levels. This is in agreement with the several fold increase noted in inferred deposition rates (Swain et al., 1992; Engstrom et al., 1994; Benoit et al., 1994).

As shown in Tables 2-2 and 2-3, measured U.S. atmospheric mercury concentrations are generally very low. The dominant form in the atmosphere is vapor-phase elemental mercury, although close to emission sources, higher concentrations of the divalent form may be present. Small fractions of particulate mercury and methylmercury may also be measured in ambient air. In rural areas, airborne particulate mercury is typically 4% or less of the total (particulate + gas phase) mercury in air (U.S. EPA, 1993; WHO, 1990). Particulate mercury comprises a greater fraction of the total in urban areas U.S. EPA (1993), and will consist primarily of bound Hg(II) compounds.

Table 2-2
Summary of Measured Mercury Concentration in Air (U.S. EPA, 1993)

Total Atmospheric Mercury (ng/m^3)	%Hg(II)	% Methylmercury
Rural areas: 1 - 4 Urban areas: 10 - 170	1-25%[a]	0-21%[b]

[a] Higher fractions in urban areas
[b] Generally % methylmercury on low end of this range

Table 2-3
Measured Vapor- and Particulate-Phase Atmospheric Mercury Concentrations

Site	Vapor-Phase Mercury Conc. in ng/m^3 Mean (Range)	Particulate-Phase Mercury Conc. in ng/m^3 Mean (Range)	Reference
Chicago, IL	8.7 (1.8-62.7)	0.098 (0.022-0.52)	Keeler et al., (1994)
Lake Michigan	2.3 (1.3-4.9)	0.028 (0.009-0.054)	Keeler et al., (1994)
South Haven, MI	2.0 (1.8-4.3)	0.019 (0.009-0.029) 0.022 (max 0.086)	Keeler et al., (1994) Keeler et al., (1995)
Ann Arbor, MI	2.0 (max 4.4)	0.10 (max 0.21) 0.022 (max 0.077)	Keeler et al., (1994) Keeler et al., (1995)
Detroit, MI Site A	>40.8 (max >74)	0.34 (max 1.09) 0.094 (0.022-0.23)	Keeler et al., (1994) Keeler et al., (1995)
Detroit, MI Site B	3.7 (max 8.5)	0.30 (max 1.23)	Keeler et al., (1994)
Pellston, MI		0.011 (max 0.032)	Keeler et al., (1995)
Underhill Center, VT	2.0 (1.2-4.2)	0.011 (0.001-0.043)	Burke et al., (1995)
Broward County, FL[a] Background Site near Atlantic Ocean (Site 1)	1.8	0.034	Dvonch et al., (1995)
Broward County, FL Inland (Site 2)	3.3	0.051	Dvonch et al., (1995)
Broward County, FL Inland (Site 3)	2.8	0.049	Dvonch et al., (1995)
Little Rock Lake, WI	1.6 (1.0-2.5)	0.022 (0.007-0.062)	Fitzgerald et al., (1991)
Long Island Sound, Avery Pt., CT[b]	(1.4-5.3): 95-100% elemental; 0-1% methylmercury	0.062 (0.005-0.18)	Particulate: Fitzgerald et al., (1991) Vapor: Bloom and Fitzgerald et al., (1988)
Crab Lake, WI	1.7	Winter 0.006 Summer 0.014	Lamborg et al., (in press)

[a] Diurnal variations were also noted; elevated concentrations were measured at night. For example at site 2 the average nighttime vapor-phase concentration was 4.5 ng/m^3. This was attributed to little vertical mixing and lower mixing heights that occur in this area at night.

[b] 99% of Total Gaseous Mercury is Hg^0. During 1 month (October) the mean methylmercury concentration was measured to be 12 pg/m^3 with a range of 4-38 pg/m^3; 0.7% of the total gaseous mercury was methylmercury. During November it was measured as <10 pg/m^3 and from December through August it was measured below the detection limit (<5 pg/m^3).

There is a substantial body of recent data pertaining to the atmospheric concentrations and deposition rates of atmospheric mercury collected at specific sites across the U.S. Most of the collected deposition data are from sites located some distance from large emission sources. The data have been collected by several different groups of researchers. These data are briefly summarized here.

Keeler et al., (1994) measured vapor- and particulate-phase atmospheric mercury concentrations from a site in Chicago, IL, two sites in Detroit, MI and a Lake Michigan site. The mean values are presented along with the range of measurement data. The collection period for these sites was generally less than one month; for example, the Detroit data were collected during a 10-day period.

Keeler et al., (1995) reported the results of several short-term atmospheric particulate mercury measurements in Detroit, MI and longer-term (1-year) particulate measurements at rural sites in Michigan and Vermont. In the Detroit measurements the particulates sampled were divided into two categories: fine (<2.5 μm) and coarse (>2.5 μm). The average size of the fine particles was 0.68 μm, and the average size of the coarse particles was 3.78 μm. Most (mean=88%) of the particulate mercury at the Detroit, MI site was measured on fine particles; the range for individual samples was 60-100% of total particulate.

Fitzgerald et al., (1991) reported measured mercury concentrations at Little Rock Lake, WI from May of 1988 through September of 1989 and particulate mercury concentrations at Long Island Sound (Avery Point, CT).

2.4.2 Mercury Concentrations in Precipitation

Mercury concentrations in precipitation are shown in Table 2-4. Total mercury concentrations in rainwater are typically higher than in surface water. This is thought to be the result of efficient scavenging of divalent mercury by rain droplets and the oxidation of elemental mercury to divalent mercury, ~~while~~ M mercury in surface waters can be lost by revolatilization from the water body and sequestration in the sediment. ALSO

Total mercury concentrations in precipitation are generally less than 100 ng/L in areas not directly influenced by an emissions source, including suburban and urban locations. Levels much higher (greater than 1000 ng/L) however, have been reported for precipitation downwind of anthropogenic mercury sources (NJDEPE 1993; see also "Measured Mercury Levels from Point Sources" section below). Areas downwind of mercury sources also show the greatest variability in precipitation concentrations. Mercury concentrations do not vary much among different precipitation types (snow, rain, and ice; NJDEPE 1993, Fitzgerald et al., 1991). Mercury precipitation concentrations show a seasonal pattern, with average concentrations several times higher during the summer than during the winter months, even in areas with a warm climate (Pollman et al., 1994). Current average precipitation mercury levels are on the order of 2-4 times greater than pre-industrial levels, based on information on the increases in mercury deposition rates (Swain et al., 1992; Expert Panel on Mercury Atmospheric Processes, 1994). The concentration of methylmercury in rain is GENERALLY minor, and its origins are uncertain.

2.4.3 Mercury Deposition Rates

Environmental mercury is widely thought to be transported primarily through the atmosphere. The primary source of mercury in terrestrial, aquatic and oceanic environments appears to be the wet or dry deposition of atmospheric mercury. Once deposited, the mercury may be revolatilized back to the atmosphere, incorporated into the medium of deposit or transferred to abiotic or biotic components of these environments.

Table 2-4
Measured Mercury Concentrations in Precipitation

Site	Mean Mercury Concentration in precipitation, ng/L Mean (Range)	Reference
Ely, MN	20 in 1988 51 in 1989 13 in 1990	1988-89 data: Glass et al., (1992) 1990 data: Sorensen et al., (1992)
Duluth, MN	23 in 1988 11 in 1989 13 in 1990	1988-89 data: Glass et al., (1992) 1990 data: Sorensen et al., (1992)
Marcell, MN	18 in 1988 18 in 1989	Glass et al., (1992)
Bethel, MN	13 in 1990	Sorensen et al., (1992)
Cavalier, ND	19 in 1990	Sorensen et al., (1992)
International Falls, MN	9 in 1990	Sorensen et al., (1992)
Lamberton, MN	15 in 1990	Sorensen et al., (1992)
Raco, MN	10 in 1990	Sorensen et al., (1992)
Little Rock Lake, WI	11 (3.2-15) in rain 6 in snow	Fitzgerald et al., (1991)
Crab Lake, WI	7.9 in rain 3.3 in snow	Lamborg et al., (in press)
Underhill Center, VT[a]	8.3	Burke et al., (1995)
Broward County, FL Background Site near Atlantic Ocean (Site 1)	Total: 35 (15-56) Reactive: 1.0 (0.5-1.4)	Dvonch et al., (1995)
Broward County, FL Inland (Site 2)	Total: 40 (15-73) Reactive: 1.9 (0.8-3.3)	Dvonch et al., (1995)
Broward County, FL Inland (Site 3)	Total: 46 (14-130) Reactive: 2.0 (1.0-3.2)	Dvonch et al., (1995)
Broward County, FL 300 m from MWC (Site 4)	Total: 57 (43-81) Reactive: 2.5 (1.7-3.7)	Dvonch et al., (1995)

[a] Both the concentrations of mercury in precipitation and the amount of precipitation deposited/event increased in spring and summer. Most (66%) of the mercury in the spring and fall precipitation samples (only ones tested) was dissolved. The mean concentration of reactive mercury was 1.0 ng/L. Higher particulate concentrations were observed in the winter.

Table 2-5
Measured Mercury Concentrations in Rain Which Include Methylmercury Estimates (ng/L)

Study Description	Total Mercury (ng/L)	Methyl-mercury (ng/L)	% Methyl-mercury	Reference
Swedish rain: 9 samples and 4 sites.	7.5-89.8	0.04-0.59	0.1-3.7	Lee and Iverfeldt (1991)
6 Samples at Little Rock Lake, WI.	3.5-15	0.06-0.22	0.4-6.3	Fitzgerald et al. (1991)

N.B. The difference between Total mercury and methylmercury can be considered Hg(II) species (Brosset 1981; U.S. EPA 1988). This is assumed for all water samples.

Intensive, site-specific studies of environmental mercury fluxes have been done at only a handful of U.S. sites. Watras et al., (1994) summarize the collected data and present a conceptualization of mercury fluxes between abiotic and biotic components of the environment in 7 Northern Wisconsin seepage Lakes, including Little Rock Lake. Most of the mercury was thought to enter the lakes through atmospheric deposition with wet deposition of mercury contributing the most to the total. The total amount deposited was approximately 10 $\mu g/m^2/yr$. Most of the mercury deposited was thought to deposit into the sediment or volatilize back into the atmosphere. There was a net production of methylmercury in the lakes with most of the produced methylmercury being stored in the tissues of fish. The behavior of mercury at most U.S. sites is not characterized to the same degree as at Little Rock Lake, WI. It should be noted that Little Rock Lake is a rather remote seepage lake and that atmospheric mercury may behave differently closer to emission sources. Mercury may also behave differently in different types of watersheds and waterbodies.

Measured wet deposition rates are given in Table 2-6. Similar measurements of dry deposition are rare due to limitations of analytical methods. In particular, dry deposition of divalent mercury vapor has not been measured to date. This is a major source of uncertainly because its high reactivity implies that it may be efficiently removed from the atmosphere via dry deposition.

Burke et al., (1995) measured mercury concentrations on a precipitation event basis for one year at a rural site in Vermont. Underhill Center, VT is located near Lake Champlain and was 200 Km away from a major urban or industrial area.

Dvonch et al., (1995) conducted a 4-location, 20 day mercury study in Broward County, FL. Broward county contains the city of Ft. Lauderdale as well as an oil-fired utility boiler and a municipal waste combustion facility. Daily measurements of atmospheric particulate and vapor-phase mercury were collected at 3 of the 4 sites, and daily precipitation samples were collected at all sites.

Hoyer et al., (1995) conducted a 2-year study of mercury concentrations in precipitation (by event) at 3 rural sites (Pellston, South Haven, and Dexter) in the state of Michigan.

Several authors have estimated mercury total deposition (wet and dry) rates by sample coring of various media. For example, Engstrom et al., (1994) used lake core sediments to estimate a current deposition rate of 12.5 $\mu g/m^2/yr$ and a preindustrial (natural) deposition rate 3.7 $\mu g/m^2/yr$ for remote lakes located in Minnesota and northern Wisconsin. Benoit et al., (1994) analyzed mercury

Table 2-6
Mercury Wet Deposition Rates (ug/m^2/yr)

Site	Wet Mercury Deposition Rates (ug/m^2/yr), Means	Reference
Ely MN	17 in 1988 42 in 1989 6.7 in 1990	1988-89 data. Glass et al., (1992) 1990 data: Sorensen et al., (1992)
Duluth, MN	20 in 1988 6.5 in 1989 9.3 in 1990	1988-89 data: Glass et al., (1992) 1990 data: Sorensen et al., (1992)
Marcell, MN	17 in 1988 14 in 1989	Glass et al., (1992)
Bethel, MN	13 in 1990	Sorensen et al., (1992)
Cavalier, ND	6.1 in 1990	Sorensen et al., (1992)
International Falls, MN	5.5 in 1990	Sorensen et al., (1992)
Lamberton, MN	9.3 in 1990	Sorensen et al., (1992)
Raco, MN	8.9 in 1990	Sorensen et al., (1992)
Little Rock Lake, WI	4.5 from rain 2.3 from snow	Fitzgerald et al., (1991)
Crab Lake, WI	4.4 from rain 0.8 from snow	Lamborg et al., (in press)
Nothern MN	10-15	Sorensen et al., (1990)
Pellston, MI	5.8 in year 1 5.5 in year 2 0.07 ug/m^2 (max 0.51) per rainfall event	Hoyer et al., (1995)
South Haven, MI	9.5 in year 1 13 in year 2 0.12 ug/m^2 (max 0.85) per rainfall event	Hoyer et al., (1995)
Dexter, MI	8.7 in year 1 9.1 in year 2 0.10 ug/m^2 (max 0.98) per rainfall event	Hoyer et al., (1995
Underhill Center, VT	9.3 0.07 ug/m^2 per rainfall event	Burke et al., (1995)

concentrations in a peat bog at a Minnesota site. The estimated pre-1900 deposition rate at this site was 7.0 μg/m^2/yr, and the current mean deposition rate was estimated to be 24.5 μg/m^2/yr. Estimates of total deposition are given in Table 2-7.

Table 2-7
Estimated Mercury Total Deposition Rates

Site	Estimate of Pre-industrial Annual Deposition Rates $\mu g/m^2/yr$	Estimate of Current Annual Deposition Rates $\mu g/m^2/yr$	Reference
Minnesota and northern Wisconsin	3.7	12.5	Swain et al. (1992); Engstrom et al., (1994) Lake core sediments
Minnesota	7.0	24.5	Benoit et al., (1994) Peat bog core sampling
Little Rock Lake, WI[a]		10	Fitzgerald et al., (1991)
Crab Lake, WI[a]		7.0 (86% estimated to deposit in summer)	Lamborg et al., (in press)

[a] Data includes previously tabled values of wet deposition plus particulate deposition. Fitzgerald et al., 1991 did not collect particulate size data. Assuming a particulate deposition velocity of 0.5 cm/s, a yearly average particulate deposition flux of 3.5+/- 3 $ug/m^2/yr$ was estimated. Lamborg et al., (in press) noted the smaller particle sizes in the winter and assumed a deposition velocity 0.1 cm/s for the average winter concentrations (7 pg/m^3) and a deposition velocity of 0.5 cm/s for average summer concentrations (26 pg/m^3).

2.4.4 Mercury Concentrations in Water

Tables 2-8 through 2-10 show measured data in surface water, groundwater and ocean water. There is a great deal of variability in these data, some of which may be due to the seasonality of the water concentrations.

Total mercury levels in lakes and streams generally are lower than mercury levels found in precipitation, with levels typically well under 20 ng/L (NJDEPE 1993). Elevated levels may be found in lakes and streams thought to be impacted by anthropogenic mercury sources but not to the extent that precipitation levels appear to be. Total lake water mercury concentrations tend to increase with lower pH and higher humic content (U.S. EPA, 1993). Present-day mercury levels in freshwater are thought to be 2 - 7 times greater than pre-industrial levels (Swedish EPA, 1991). Methylmercury percentages are higher that those in precipitation, ranging from 5 - 20%, with levels around 10% being the most common. Mercury levels continue to increase in many lakes (Swedish EPA, 1991).

It is important to note that much of the data on mercury in drinking water and ground water report levels as below detection limits (U.S. EPA, 1988), although the detection limit was a somewhat dated 100 ng/L. Lindqvist and Rodhe (1985) report that the concentration range for mercury in drinking water is the same as in rain, with an average estimate for total mercury of 25 ng/L. It seems reasonable to assume similar speciation as no speciation data could be found. Dooley (1992) states that mercury concentrations in pristine wells are likely to be below that of unpolluted surface waters.

Table 2-10 shows published values for mercury concentrations in ocean water. Limited speciation data are available. Hovert et al. (1993b) reported that 2.8% of total mercury was methylmercury, which is not much different from the speciation in fresh water. Total mercury concentrations in ocean and sea water vary from undetectable to over 1000 ng/l (Nriagu, 1979).

Table 2-8
Measured Mercury Concentrations in Surface Fresh Water (ng/L)

Study Description	Total Mercury (ng/L)	Methyl-mercury (ng/L)	% Methyl-mercury	Reference
Swedish lakes: 8 sites, 2-4 samples each.	1.35-15	0.04-0.8	1.0-12	Lee and Iverfeldt (1991)
Swedish mires: 8 sites, 4 samples each.	2.9-12	0.08-0.73	2-14	Westling (1991)
Lake Cresent, WA	0.163	<0.004	<2.5	Bloom and Watras (1989)
Swedish runoff: 7 sites, 3 samples each.	2-12	0.04-0.64	1-6	Lee and Iverfeldt (1991)
Little Rock Lake: reference basin.	1.0-1.2	0.045-0.06	mean of 5	Watras and Bloom (1992)
Lake Michigan (total)	7.2 microlayer 8.0 at 0.3m 6.3 at 10m			Cleckner et al. (1995)
Lake Champlain (filtered)	3.4 microlayer 3.2 at 0.3m 2.2 at 15m			Cleckner et al. (1995)
Lakes Rivers and Streams	0.04 - 74 1 - 7	NA	NA	NJDEPE (1993)

Table 2-9
Measured Mercury Concentrations in Ground/Drinking Water (ng/L)

Study Description	Total Mercury	Reference
Southern New Jersey domestic wells	Up to and exceeding 2000	Dooley (1992)
Drinking/Tap water in U.S.	0.3-25	NJDEPE (1993)
Washington State well	0.3	Bloom (1989)

Table 2-10
Measured Mercury Concentrations in Ocean Water (ng/L)

Study Description	Total Mercury (ng/L)	Reference
Review on concentrations of dissolved mercury: Open ocean	0.5 - 3.0	WHO (1989)
Review on concentrations of dissolved mercury: Coastal sea water	2 - 15	WHO (1989)
Hg along the Italian coast	Dissolved: 1.7-12.2 Particulate: 0.3 - 80	Seritti et al. (1982)
Puget Sound near-shore sea water	0.72	Hovert et al. (1993b)

2.4.5 Mercury Concentrations in Soil

Table 2-11 presents the reported concentrations in soil. The relatively high concentrations illustrate the strong partitioning of mercury to soils. Based on the soil data presented, it can be inferred that soil, while not as important as the atmosphere, is a significant reservoir for environmental mercury. The concentrations are presented as total mercury and methylmercury. Most of the soil mercury is thought to be Hg(II).

Table 2-11
Measured Mercury Concentration in Soil

Study Description	Total Mercury (ng/g dry weight)	Methyl-mercury (ng/g dry weight)	% Methyl-mercury	Reference
Discovery Park, Seattle, WA	29 - 133	0.3-1.3	0.6-1.5	Lindqvist et al. (1991)
Wallace Falls, Cascades (WA)	155 - 244	1.0-2.6	0.5-1.2	Lindqvist et al. (1991)
Control Soil, New York State	117	4.9	4.2	Cappon (1981)
Compost, New York State	213	7.3	3.3	Cappon (1987)
Garden soil, New York State	406	22.9	5.3	Cappon (1987)
Typical U.S. Soils	8 - 117	NA	NA	NJDEPE (1993)

N.B. As in water samples the fraction of Hg^0, if present at all, will be very small compared to Hg(II) (Revis et al., 1990), and the difference between Total mercury and methylmercury can be considered to be Hg(II) to be species.

Soil mercury levels are usually less than 200 ng/g in the top soil layer, but values exceeding this level are not uncommon, especially in areas affected by anthropogenic activities (see section 2.6). Soil mercury levels vary greatly with depth, with nearly all the mercury found in the top 20 cm of

soil. Mercury levels are also positively correlated with the percentage of organic matter in soil (Nriagu 1979). Top soil mercury concentrations are estimated to be a factor of 4-6 (Swedish EPA. 1991) higher now as compared to pre-industrial concentrations. Methylmercury percentages in soil are typically on the order of a few percent. Soil mercury levels are continuing to rise (Fitzgerald 1994). and most (up to 95%) of the anthropogenic mercury released over the past 100 years resides in surface soil (Fitzgerald, 1994; Expert Panel on Mercury Atmospheric Processes, 1994). Mercury from soil provides in most cases (depending on watershed characteristics) the main source of mercury to water bodies and fish. Mercury is very slowly removed from soil, and long after anthropogenic emissions are reduced, soil and water concentrations can be expected to remain elevated.

Sediment mercury levels are typically higher than soil levels, and concentrations exceeding 200 ng/g are not unusual (see Table 2-12). Sediment mercury levels follow the same trends as soil in regards to depth, humic matter, and historical increases, and methylmercury percentage. There is some evidence suggesting that the methylmercury percentage increases with increasing total mercury contamination (Parks et al., 1989).

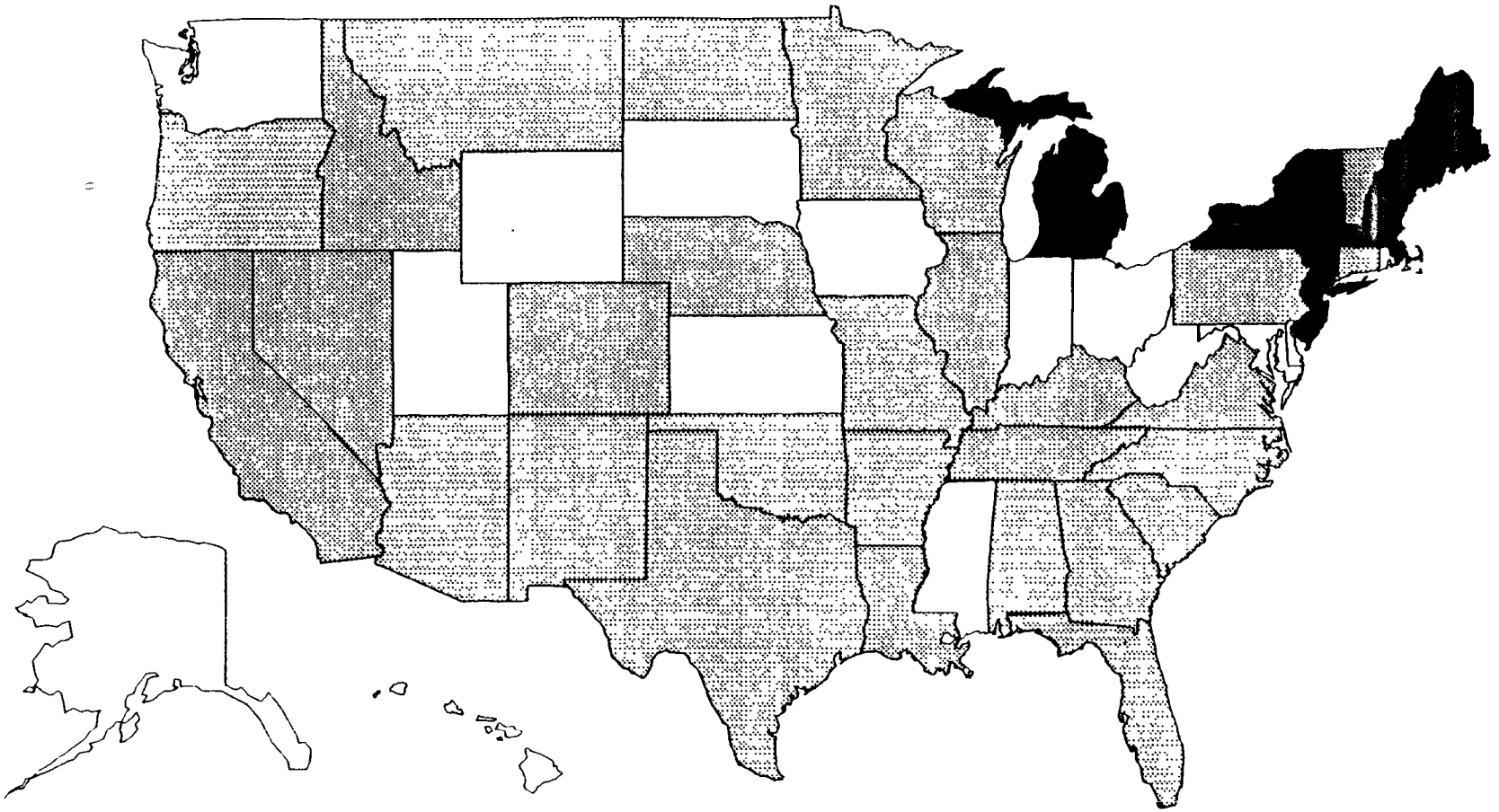
Table 2-12
Measured Mercury Concentrations in Aquatic Sediment

Study Description	Total Mercury (ng/g dry weight)	Reference
80 MN Lakes	34-753; mean 174	Sorensen et al. (1990)
North Central WI Lakes	90-190	Rada et al. (1989)
Little Rock Lake, WI	10-170	Wiener et al. (1990)
U.S. Lake sediment mean ranges	70-310	NJDEPE (1993)

2.4.6 Mercury Concentrations in Biota

Elevated mercury concentrations in fish have been measured across the U.S. As seen in Figure 2-2, 35 states have at least one waterbody under mercury advisory, including six states with statewide mercury advisories. There are differences in the action levels for advisories from state to state. Fish mercury concentrations are the single greatest concern in regards to the effects of mercury pollution. Fish in lakes seemingly far removed from anthropogenic sources have been found to have mercury levels of concern to human health. Mercury levels in fish vary greatly, often showing little correlation to proximity to mercury emission sources. In Sweden, fish mercury concentrations in 1 kg pike have risen from 0.05 - 0.3 µg/g to 0.5 - 1.0 µg/g in southern and central Sweden over the last 100 years. Fish mercury concentrations in most cases strongly correlate with pH (lower pH resulting in higher methylmercury concentrations). Other lake characteristics have been found to correlate with fish mercury levels, but not as strongly as pH, with some factors showing a positive correlation in some lakes and a negative correlation in others (U.S. EPA, 1993).

Figure 2-2
Mercury Fish Consumption Advisories of the U.S.



STATES WITH AT LEAST ONE WATERBODY UNDER MERCURY ADVISORY

STATES WITH STATEWIDE MERCURY ADVISORIES

STATES WITH NO MERCURY ADVISORIES

SOURCE: USEPA FISH ADVISORY DATABASE.

It has been so well established that most (>95%) of the total mercury content of fresh and saltwater fish is methylmercury (Bloom, 1992) that currently some researchers no longer speciate fish samples (NJDEPE 1994). Thus, only total mercury concentrations are reported here. Approximately 90% of the mercury in shrimp, mussels and copepods from IAEA standards contain other forms of mercury (only about 10% of total mercury is methylmercury), but rather about 90% of the mercury total concentration is ethylmercury (Bloom, 1992). (It should be noted that ethylmercury exposure was not assessed in this document.)

The data from two studies national in scope are summarized in Table 2-13. Lowe et al. (1985) reported mercury concentrations in fish from the National Contaminant Biomonitoring Program. The fresh-water fish data were collected between 1978-1981 at 112 stations located across the United States. Mercury was measured by a flameless cold vapor technique, and the detection limit was 0.01 μg/g wet weight. Most of the sampled fish were taken from rivers (93 of the 112 sample sites were rivers); the other 19 sites included larger lakes, canals, and streams. Fish weights and lengths were consistently recorded. A wide variety of types of fishes were sampled; most commonly carp, large mouth bass, and white sucker. The geometric mean mercury concentration of all sampled fish was 0.11 μg/g wet weight; the minimum and maximum concentrations reported were 0.01 and 0.77 μg/g wet weight, respectively. The highest reported mercury concentrations (0.77 μg/g wet weight) occurred in the northern squawfish of the Columbia River.

"A National Study of Chemical Residues in Fish" was conducted by U.S. EPA (1992) and also reported by Bahnick et al. (1994). In this study mercury concentrations in fish tissue were analyzed. Five bottom feeders (e.g., carp) and five game fish (e.g., bass) were sampled at each of the 314 sampling sites in the U.S. The sites were selected based on proximity to either point or non-point pollution sources. Thirty-five "remote" sites among the 314 were included to provide background pollutant concentrations. The study primarily targeted sites that were expected to be impacted by increased dioxin levels. The point sources proximate to sites of fish collection included the following: pulp and paper mills, Superfund sites, publicly owned treatment works, and other industrial sites. Data describing fish age, weight, and sex were not consistently collected. Whole body mercury concentrations were determined for bottom feeders, and mercury concentrations in fillets were analyzed for the game fish. Total mercury levels were analyzed using flameless atomic absorption; the reported detection limits were 0.05 μg/g early in the study and 0.0013 μg/g as analytical technique improved later in the analysis. Mercury was detected in fish at 92% of the sample sites. The maximum mercury level detected was 1.8 μg/g, and the mean across all fish and all sites was 0.26 μg/g. The highest measurements occurred in walleye, large mouth bass, and carp. The mercury concentrations in fish around publicly owned treatment works were highest of all point source data; the median value measured were 0.61 μg/g. Paper mills were located near many of the sites where mercury-laden fish were detected.

Both the studies reported by Lowe et al. (1985) and by Bahnick et al. (1994) appear to be systematic, national collections of fish pollutant concentration data. Clearly, higher mercury concentrations in fish have been detected in other analyses, and the values obtained in these studies should be interpreted as a rough approximation of the mean concentrations in fresh-water finfishes. As indicated in the range of data presented, wide variations are expected in these data.

The mean mercury concentrations in all fish sampled differ by approximately a factor of 2 for each study. The mean mercury concentration reported by Lowe et al. was 0.11 μg/g, whereas the mean mercury concentration reported by Bahnick et al. was 0.26 μg/g. This is difference which can be extended to the highest reported mean concentrations in fish species. Note that the average mercury concentrations in bass and walleye reported by Bahnick's data are higher than the northern squawfish, which is the species with the highest mean concentration of mercury identified by Lowe et al. (1985).

Table 2-13
Freshwater Fish Mercury Concentrations from Nationwide Studies

Species	Mean Mercury Concentration µg/g (fresh weight)	
	Lowe et al., (1985)	U.S.EPA (1992c) and Bahnick et al., (1994)
Bass	0.157	0.38[1]
Bloater	0.093	
Bluegill	0.033	
Smallmouth Buffalo	0.096	
Carp, Common	0.093	0.11
Catfish	0.088[2]	0.16[3]
Crappie (black, white)	0.114	0.22
Fresh-water Drum	0.117	
Northern Squawfish	0.33	
Northern Pike	0.127	0.31
Perch (white and yellow)	0.11	
Sauger	0.23	
Sucker	0.114[4]	0.167[5]
Trout (brown, lake, rainbow)	0.149	0.14[6]
Walleye	0.100	0.52
Mean of all measured fish	0.11	0.26

[1] Average concentration found in white, largemouth and smallmouth bass.
[2] Channel, largemouth, rock, striped, white catfish.
[3] Channel and flathead catfish.
[4] Bridgelip, carpsucker, klamath, largescale, longnose, rivercarpsucker, tahoe sucker.
[5] Mean of average concentrations found in white, redhorse and spotter sucker.
[6] Brown trout only.

The bases for these differences in methylmercury concentrations are not immediately obvious. The trophic positions of the species sampled, the sizes of the fish, or ages of fish sampled could significantly increase or decrease the reported mean mercury concentration. Older and larger fish, which occupy higher trophic positions in the aquatic food chain, would, all other factors being equal, be expected to have higher mercury concentrations. The sources of the fish will also influence fish mercury concentrations. Most of the fish obtained by Lowe et al. (1985) were from rivers. The fate and transport of mercury in river systems is less well characterized than in small lakes. Most of the data collected by Bahnick et al. (1994) were collected with a bias toward more contaminated/industrialized sites, although not sites specifically contaminated with mercury. It could be that there is more mercury available to the aquatic food chains at the sites reported by Bahnick et al. (1994). Finally, the increase in the more recent data as reported in Bahnick et al., 1994 could be the result of temporal increases in mercury concentrations.

Table 2-14 summarizes measured mercury concentrations in freshwater sportfish as reported by a number of researchers, and Table 2-15 summarizes available data on measured mercury concentrations in saltwater commercial fish. Due to the importance of fish mercury levels, discussions of several of the mercury studies referenced in the table are summarized here.

ACTION LEVEL

The New Jersey Department of Environmental Protection and Energy collected individual fish samples throughout the state (NJDEPE 1994). Generally larger fish were sampled from New Jersey rivers, lakes and reservoirs known to be contaminated with mercury or at risk for mercury contamination. Samples were prepared as skin-off fillets, and clean protocols were used throughout the analysis. Mercury levels in fish exceeded the FDA ~~Criterion~~ of 1.0 ug/g (wet weight) in 50 of the 313 sampled fish and at 15 of the 55 sample locations. It is noted that the FDA ~~criterion~~ is applicable to fish sold through interstate commerce in the United States under the Food, Drug and Cosmetic Act (21 U.S.C. 301). Levels of greater than 0.5 ug/g (wet weight) occurred in 108 of the 313 fish. The highest reported concentration occurred in a largemouth bass taken from the Atlantic City Reservoir at a concentration of 8.94 ug/g. The mercury levels in all six of the largemouth bass sampled from this site were elevated. At the Atlantic City site the range of mercury concentrations was 3.05 to 8.94 and the mean was 4.5 ug/g. The overall study range for largemouth bass was 0.05 to 8.94 ug/g. High levels were also noted in chain pickerel particularly those obtained from a series of low pH waterbodies. The range of mercury concentrations reported for chain pickerel was 0.09 to 2.82 ug/g. Levels of greater than 1 ug/g were also reported in yellow bullheads (maximum reported 1.47 ug/g). Acidity of these waterbodies was also measured, and reported in Table 2-14 are the ranges of mean fish mercury concentrations for 9 pH categories.

Simonin et al. (1994) collected yellow perch from 12 drainage lakes located in Adirondack Park, New York State, during the fall of 1987. The age of the fish was determined from acetate impressions of the scales, and filets (including the skin and ribs) were analyzed for total mercury. Lake water samples were taken late in the summer of 1987 and included analysis of pH, dissolved inorganic and organic carbon (DIC and DOC), conductance, color, acid neutralizing capacity (ANC) and a number of metals and ligands. A total of 372 fish were collected, with 7 to 53 fish taken per lake. Fish ranged from 2+ to 11+ years of age, with 4+ year old fish being the most common; fish of this age were used in making comparisons among lakes. It was found that air-equilibrated pH was the best predictor of mercury concentrations, with lower lake pH resulting in higher mercury levels in perch. This was clear despite large variations in mercury concentrations from the same lake. Perch mercury concentrations from the highest pH lake (considering all ages) ranged from 0.07 - 0.27 μg/g wet wt., the corresponding range for the lowest pH lake was 0.63 - 2.28 μg/g. Other variables that were highly correlated ($p < 0.0001$) with fish mercury levels included ANC, DIC, Ca, conductivity, Mg and field pH. Variables less strongly correlated ($p < 0.05$) include DOC, Na, SO_4, lake area and watershed area. Variables not correlated with 4+ year old yellow perch include color, total phosphorus, Al, Cl-, lake depth, ratio of watershed area to lake area, ratio of watershed area to lake volume, fish length and fish weight. For a given lake, fish age was most strongly correlated with mercury concentrations; older fish had the highest concentrations. Fish length and weight were also significantly correlated.

In general, the mercury levels in freshwater fish appear to be higher than the levels in saltwater fish. Several authors report mercury levels that are higher than 1 ug/g (1 μg/g) in the muscle of freshwater fish: NJDEPE (1994); Wren, et al. (1991); Lathrop et al. (1989); MacCrimmon et al. (1983); Lange et al. (1993); Glass et al. (1990); Sorensen et al. (1990); U.S. EPA (1992a), U.S. EPA (1992); Simonin et al. (1994); and Florida DER (1990). Several of these larger studies are described in greater detail.

Table 2-14
Measured Mercury Concentrations Freshwater Sportfish (Total Mercury, ug/g wet wt.)

Study	Pike/Pickerel	Walleye	Bass	Bottom Feeders	Panfish	Trout	Reference
12 Adirondack Lakes					2 year old. 0 23 4 year old 0 36 6 year old 0 41 8 year old. 0 46 10+ year old. 1 65		Simonin et al (1994)
16 New York Lakes					Yellow Perch 0.01 - 0 64 Punkin Seed: 0 01 - 0 19		Mills et al , (1994)
42 New Jersey Lakes and Rivers, averages of 9 pH categories	0.15-1.45		0.15-1.16	0 07-0 72	0 10-0 32	0 05-0.64	NJDLPE (1994)
Historical trends in mean fish concentrations in NE Minnesota Lakes	1930s (museum): 0.08-0 29 1980s: 0.12-0 37	1930s. 0 08-0 20 1980s: 0 07-0.73					Swain and Helwig (1989)
Mean concentrations in 65 northern MN lakes	0.14-1 52	0.13-1 75					Sorensen et al (1990)
Ashtabula River, OH, Means			LMouth: 0 15 Smouth: 0 12	Carp. 0.05	Bluegill 0 07-0 17		U S EPA (1992a)
Saginaw River, MI, Means		Fillets: .12		Carp 0 07			U S EPA (1992b)
Northern Michigan Lakes	0.10-1.64		0 11-1 00				Gloss et al (1990)
Large fish above dams in 3 Michigan rivers	0 11 - 0 28	< 0 05 to 0 72		< 0 05 to 0 23	< 0 05 to 0 73	0 20 - 0 45	Giesy et al , (1994)
Mean Concentrations (and ranges) in Fish from the St. Louis River, MN	0.28 (0 25-0 31)	0 29 (0 20-0 37)	S. Mouth. 0 25 (0 12-0 43) Rock: 0.35 (0 14-0 61)	Channel Cat 0 39 (0 20-0 74) Red H Sucker: 0 41 (0 32-0 55) W Sucker 0.27 (0 12- 0 37)			Sorensen et al (1991)
Historical trends in Onondaga lake Smallmouth Bass, Syracuse, NY	1970 1 96 1972 1 26 1974 0 81	1975 1 09 1977 0 87 1978 0 68	1979 0 68 1981. 1 23 1983. 1.08	1984 1 04 1985 1 20 1986 1 05	1987 1 75 1988 1 43 1989 1 71		Sloan et al (1990)
38 Wisconsin Lakes		0 16-1.74					Lathrop et al (1989)
34 Northern Wisconsin Lakes		0 19 - 0 999					Gerstenberger et al (1993)
53 Florida lakes			0 04-1 90				Lange et al (1993)
Florida Surface Waters			Lakes 0 07 - 0 85 Streams 0 22 - 2 37				Florida DER (1990)

Table 2-14 (continued)
Measured Mercury Concentrations Freshwater Sportfish (Total Mercury, µg/g wet wt.)

Study	Pike/Pickerel	Walleye	Bass	Bottom Feeders	Panfish	Trout	Reference
Mean Concentrations (and ranges) in Maine Predatory Fishes	0 92 (0 58-1.22)		L.Mouth. 0 57 (0.26-0 95) SMouth: 0 67 (0 31-1 12)		Yellow Perch. 0.28 (0 18-0 81)	Brook 0 30 (0 05-0 79) Brown 0 29 (0 12-0 45)	Stafford (1991)
St. Lawrence River Drainage at Massena, NY.	0.21-0.97	0 24-0 93	SMouth: 0 37-0 71 Rock: 0 34-0.76	W Sucker. 0 12-0 55 B Bullhead 0 08-0 32	Punkinseed: 0 10-0 35 Y Perch: 0 18-0 59	Rainbow 0 12-0 13	New York DEC (1990)
9 Canadian Shield Lakes						0 24-3 44	MacCrimmon et al (1983)
Ontario Lakes	0 07-1 28	0 09-3 24					Wren et al (1991)

Table 2-15
Measured Mercury Concentrations in Saltwater Commercial Fish (ug/g wet wt.)

Fish	Mean Hg-tot			References
	U.S. EPA (1992c)	Cramer (1992)	USDOC (1978)	
Cod	0.03		0.13	
Canned Tuna		0.17	0.24	
Fish Sticks			0.21	
Shrimp		0.18	0.46	
Crabs/Lobsters		0.03-0.08	0.25	
Salmon	0.05-0.32	.005		
Flounder	0.03	0.06		
Clams	0.02		0.05	
Boston Mackerel (2 samples)	0.03-0.05			NJDEPE (1994)
Porgy (3 samples)	0.08-0.14			NJDEPE (1994)
Spot (5 samples)	0.02-0.06			NJDEPE (1994)
Scallops	0.05			NOAA (1978)

Ocean fish are an important source of mercury exposure. Although these fish appear to have lower mercury concentrations, humans typically consume higher quantities of these types of fish. Wildlife, depending on location, also may typically consume ocean fish species.

Table 2-16
Mercury Concentrations in Marine Finfish

Fish	Mercury Concentration (µg/g, wet weight)	Source of Data
Anchovy[1]	0 047	NMFS
Barracuda, Pacific[2]	0.177	NMFS
Cod[3]	0.121	NMFS
Croaker, Atlantic	0.125	NMFS
Eel, American	0.213	NMFS
Flounder[4]	0.092	NMFS
Haddock	0.089	NMFS
Hake[5]	0.145	NMFS
Halibut[6]	0.25	NMFS
Herring[7]	0.013	NMFS
Kingfish[8]	0.10	NMFS
Mackerel[9]	0.081	NMFS
Mullet[10]	0.009	NMFS
Ocean Perch[11]	0.116	NMFS
Pollack	0.15	NMFS
Pompano	0.104	NMFS
Porgy	0.522	NMFS
Ray	0.176	NMFS
Salmon[12]	0.035	NMFS
Sardines[13]	0.1	NMFS
Sea Bass	0.135	NMFS
Shark[14]	1.327	NMFS
Skate[15]	0.176	NMFS
Smelt, Rainbow	0.1	NMFS
Snapper[16]	0.25	NMFS
Sturgeon[17]	0.235	NMFS
Swordfish	0.95	FDA Compliance Testing
Tuna[18]	0.206	NMFS
Whiting (silver hake)	0.041	NMFS

[1] This is the average of NMFS mean mercury concentrations for both striped anchovy (0.082 µg/g) and northern anchovy (0.010 µg/g).
[2] USDA data base specified the consumption of the Pacific Barracuda and not the Atlantic Barracuda.
[3] The mercury content for cod is the average of the mean concentrations in Atlantic Cod (0.114 µg/g and the Pacific Cod (0.127 µg/g).
[4] The mercury content for flounder is the average of the mean concentrations measured in 9 types of flounder:Gulf (0.147 µg/g), summer (0.127 µg/g), southern (0.078 µg/g), four-spot (0.090 µg/g), windowpane (0.151 µg/g), arrowtooth (0.020 µg/g), witch (0.083 µg/g), yellowtail (0.067 µg/g), and winter (0.066 µg/g).

[5] The mercury content for Hake is the average of the mean concentrations measured in 6 types of Hake: silver (0.041 µg/g), Pacific (0.091 µg/g), spotted (0.042 µg/g), red (0.076 µg/g), white (0.112 µg/g), and blue (0.405 µg/g).
[6] The mercury content for Halibut is the average of the mean concentrations measured in 3 types of Halibut: Greenland, Atlantic, and Pacific.

[7] The mercury content for Herring is the average of the mean concentrations measured in 4 types of Herring: blueback (0.0 µg/g), Atlantic (0.012 µg/g), Pacific (0.030 µg/g), and round (0.008 µg/g).
[8] The mercury content for Kingfish is the average of the mean concentrations measured in 3 types of Kingfish: Southern, Gulf, and Northern.

[9] The mercury content for Mackerel is the average of the mean concentrations measured in 3 types of Mackerel: jack (0.138 µg/g), chub (0.081 µg/g), and Atlantic (0.025 µg/g).
[10] The mercury content for Mullet is the average of the mean concentrations measured in 2 types of Mullet: striped (0.011 µg/g) and silver (0.007 µg/g).
[11] The mercury content for Ocean Perch is the average of the mean concentrations measured in 2 types of Ocean Perch: Pacific (0.083 µg/g) and Redfish (0.149 µg/g)
[12] The mercury content for Salmon is the average of the mean concentrations measured in 5 types of Salmon: pink (0.019 µg/g), chum (0.030 µg/g), coho (0.038 µg/g), sockeye (0.027 µg/g), and chinook (0.063 µg/g).
[13] Sardines were estimated from mercury concentrations in small Atlantic Herring.
[14] The mercury content for Shark is the average of the mean concentrations measured in 9 types of Shark: spiny dogfish (0.607 µg/g), (unclassified) dogfish (0.477 µg/g), smooth dogfish (0.991 µg/g), scalloped hammerhead (2.088 µg/g), smooth hammerhead (2.663 µg/g), shortfin mako (2.539 µg/g), blacktip shark (0.703 µg/g), sandbar shark (1.397 µg/g), and thresher shark (0.481 µg/g).
[15] The mercury content for skate is the average of the mean concentrations measured in 3 types of skate: thorny skate (0.200 µg/g), little skate 0.135 µg/g) and the winter skate (0.193 µg/g).
[16] The mercury content for snapper is the average of the mean concentrations measured in types of snapper:
[17] The mercury content for sturgeon is the average of the mean concentrations measured in 2 types of sturgeon:green sturgeon (0.218 µg/g) and white sturgeon (0.251 µg/g).
[18] The mercury content for tuna is the average of the mean concentrations measured in 3 types of tuna: albacore tuna (0.264 µg/g), skipjack tuna (0.136 µg/g) and yellowfin tuna (0.218 µg/g)

Table 2-17
Mercury Concentrations in Marine Shellfish

Shellfish	Mercury Concentration (µg/g, wet weight)	Source of Data
Abalone[1]	0.016	NMFS
Clam[2]	0.023	NMFS
Crab[3]	0.117	NMFS
Lobster[4]	0.232	NMFS
Oysters[5]	0.023	NMFS
Scallop[6]	0.042	NMFS
Shrimp[7]	0.047	NMFS

[1] The mercury content for abalone is the average of the mean concentrations measured in 2 types of abalone: green abalone (0.011 µg/g) and red abalone (0.021 µg/g).
[2] The mercury content for clam is the average of the mean concentrations measured in 4 types of clam: hard (or quahog) clam (0.034 µg/g), Pacific littleneck clam (0 µg/g), soft clam (0.027 µg/g), and geoduck clam (0.032 µg/g).
[3] The mercury content for crab is the average of the mean concentrations measured in 5 types of crab: blue crab (0.140 µg/g), dungeness crab (0.183 µg/g), king crab (0.070 µg/g), tanner crab (C.opilio) (0.088 µg/g), and tanner crab (C.bairdi) (0.102 µg/g).
[4] The mercury content for lobster is the average of the mean concentrations measured in 3 types of lobster: spiny (Atlantic) lobster (0.108 µg/g), spiny (Pacific) lobster (0.210 µg/g) and northern (American) lobster (0.378 µg/g).
[5] The mercury content for oyster is the average of the mean concentrations measured in 2 types of oyster: eastern oyster (0.022 ug/g) and Pacific (giant) oyster (0.023 µg/g).
[6] The mercury content for scallop is the average of the mean concentrations measured in 4 types of scallop : sea (smooth) scallop (0.101 µg/g), Atlantic Bay scallop (0.038 µg/g), calico scallop (0.026 µg/g), and pink scallop (0.004 µg/g).
[7] The mercury content for shrimp is the average of the mean concentrations measured in 7 types of shrimp : royal red shrimp (0.074 µg/g), white shrimp (0.054 µg/g), brown shrimp (0.048 µg/g), ocean shrimp (0.053 µg/g), pink shrimp (0.031 µg/g), pink northern shrimp (0.024 µg/g) and Alaska (sidestripe) shrimp (0.042 µg/g).

Table 2-18
Mercury Concentrations in Marine Molluscan Cephalopods

Mercury Concentrations in Marine Molluscan Cephalopods		
Cephalopod	**Mercury Concentration** (μg/g wet wt.)	**Source of Data**
Octopus	0.029	NMFS
Squid[1]	0.026	NMFS

[1] The mercury content for squid is the average of the mean concentrations measured in 3 types of squid: Atlantic longfinned squid (0.025 μg/g), short-finned squid (0.034 μg/g), and Pacific squid (0.018 μg/g)

By comparing the mercury concentration in fish with concentrations in other biota (Tables 2-19 through 2-22), it is noted that fish appear to have the highest concentrations of methylmercury in the environment.

The little recent data available on mercury in meat products show concentrations to be very low (near the detection limits) for both Hg(II) and methylmercury. It is not thought that meat consumption is a major concern with regards to mercury exposure, especially in comparison to concentration in fish tissues. Surprisingly few data however, are available on meat mercury levels.

Plant mercury levels are generally very low and of little concern, as with meats. Levels tend to be highest in leafy vegetables, and plants grown in mercury contaminated conditions (in air and/or soil) do accumulate more mercury than plants in background areas. There are no other noticeable trends in plant concentrations, with mercury levels varying widely among plants and studies. For further information, see appendix A: plant BCFs.

Tables 2-23 and 2-24 show measured mercury concentrations in human hair, blood, and breast milk. Mercury levels in breast milk do not seem to show a clear trend with exposure; blood and hair levels appear to be a better indicator of mercury exposure. Not surprisingly, the highest blood and hair levels are found in those individuals who consume above average amounts of fish.

It is important to keep in mind that the assembled data do not represent an exhaustive review of the literature, nor have these studies been critically evaluated; rather, they represent data appearing in the public literature.

Table 2-19
Measured Mercury Concentration in Meats

Study Description	Total Mercury (ng/g wet weight)	Approx. Total Mercury (ng/g dry weight)[1]	% Methyl-mercury	Reference
6 Saginaw River, MI "roaster" ducks	48	124.7	NA	U.S. EPA (1992b)
Japan background levels				
Chicken	12	31.2	NA	Shitara and Yasumasa (1976)
Beef	5	13.0	NA	Shitara and Yasumasa (1976)
Pork	21	54.5	NA	Shitara and Yasumasa (1976)
Wild Deer (Northern Wisconsin)	5-14	13 -36	11-57 %	Bloom and Kuhn (1994)
Beef				
Raw	< 1	< 2.6	> 10%	Bloom and Kuhn (1994)
Lunch Meat	21	54.5	4%	Bloom and Kuhn (1994)
Frank	<1	< 2.6	> 60%	Bloom and Kuhn (1994)
Beef Muscle - Control group	2-3	5.2 - 7.8	NA	Vreman et al. (1986)*
Beef Muscle - Exposed group	1-4	2.6 - 10.4	NA	Vreman et al. (1986)*
Beef Liver - Control group	3000 - 7000	7800 - 18000	NA	Vreman et al. (1986)*
Beef Liver - Exposed group	9000 - 26000	23400- 67000	NA	Vreman et al. (1986)*
Pork (raw and sausage)	< 1	< 2.6	0-70%	Bloom and Kuhn (1994)
Chicken (raw and lunch meat)	< 1 to 29	< 2.6 to 75.4	20-67%	Bloom and Kuhn (1994)
Turkey (lunch meat)	< 1	< 2.6	>20%	Bloom and Kuhn (1994)

* See Appendix A for a more complete discussion of this study.
[1] Based on an assumed water content of 0.615, which is average for beef (Baes et al., 1984)

Table 2-20
Measured Mercury Concentrations in Garden Produce/Crops

Study Description	Total Mercury (ng/g dry weight)	Methyl-mercury	% Methyl-mercury	Reference
NY Garden conditions: Leafy vegetables	64-139	9.5-30	15-23	Cappon (1987)
NY Garden conditions: Tuberous plants	11-36	0.3-6.6	11-36	Cappon (1987)
NY Garden conditions: Cole	50-64	8.8-12	18	Cappon (1987)
NY Garden conditions: Fruiting vegetables	2.9-27	0-2.4	0-9.1	Cappon (1987)
NY Garden conditions: Beans	4.3	0	0	Cappon (1987)
Herbs: Garden samples from Belgium background	130[a]			Temmerman et al. (1986)

N.B. No Hg^0 was detected in plants (Cappon, 1987).
[a]Conversion to dry wt. assuming 90% water by wt.

Table 2-21
Mean Background Total Mercury Levels for Plants in the Netherlands
(Wiersma et al., 1986)

Plant	Total Mercury Concentration (ng/g wet weight)	Approximate Water Content (from Baes et al., 1984)	Total Mercury Concentration (ng/g dry weight)
Lettuce, greenhouse	2	0.948	38.5
Tomato, greenhouse	1.3	0.941	22.0
Cucumber, greenhouse	0.3	0.961	7.7
Spinach	5	0.927	68.5
Carrot	2	0.882	16.9
Potato	3	0.778	13.5
Wheat	5	0.125	5.7
Barley	6	0.111	6.7
Oats	8	0.083	8.7
Apples	1	0.841	6.3

Table 2-22
Range of Mercury Concentrations in Selected Grain Products

Grain product	Range (ng/g wet weight)	Range (ng/g dry weight)[1]	Reference
Wheat	< 0.1 - 30	< 0.1 - 34	Wiersma et al., (1986)
Barley	1 - 30	1.1 - 34	
Oats	<0.1 - 20	< 0.1 - 22	
Maize	1.5 - 6.5	1.7 - 7.3	Szymczak and Grajeta (1992)

[1] Calculated assuming water content of 0.112 (Baes et al., 1984).

Table 2-23
Measured Total Mercury Concentrations in Human Hair and Blood

Study Description	Concentration (ug/g wet weight)		Reference
	Hair	Blood	
81 exposed pregnant women in the Iraq outbreak; maximum concentration during the pregnancy	Mean: 20; 41% under 10		Marsh et al. (1987)
Mothers of 234 Cree Indian Children from N. Quebec (high fish consumptions)	Mean: 6.0		McKeown-Eyssen et al. (1983)
6 Swedes who ate large amounts of fish		0.006-0.8[a]	Swedish Expert Group (1971)
Swedish women and newborns exposed to mercury from fish consumption		0.002[A] 0.006-0.01 for commercial fish consumers; 0.012-0.072 for coastal/lake fish eaters. Newborns had avr. 47% higher levels.	Skerfving (1988)
34 Tokyo women and newborns	Women: 1.2-7.3; mean 3.3 Newborns: 2.0-7.9; mean 4.3	Women: 0.007- 0.054; mean 0.025 Newborns: 0.012-0.048; mean 0.026	Fujita and Takabatake (1977)

[a] Linear with estimated methylmercury consumption
[A] Subject was a vegetarian (i.e., no fish consumption).

Table 2-24
Measured Mercury Concentrations in Breast Milk

Study Description	Concentration (ng/g wet weight)			Reference
	Total Mercury	Methylmercury	% Methylmercury	
15 Swedish women exposed to methylmercury in Fish	0.2-6.3	0.2-1.2	20 (mean)	Skerving (1988)
29 women from Slovenia	1.2-37.4	NA	NA	Kosta et al. (1983)
130 women from Italy	0-17.5 wet wt. (Median <0.5)	NA	NA	Clemente et al. (1982)
34 women from Japan	0.4-9.8	NA	NA	Fujita and Takabatake (1977)

Table 2-25
Measured Total Mercury Concentrations in Piscivorus Wildlife

	Concentration (ug/g wet weight)		
Study Description	Liver	Muscle	Reference
Levels in Mink from NY: Statewide mean and range of means for 8 areas.	2.2 (0.94-2.87)		Foley et al. (1988)
Levels in Otter from NY: Statewide mean and range of means for 4 areas.	1.8 (1.31-2.28)		Foley et al. (1988)
Maine Bald Eagles	0.7 - 19.8		Welch (1994)
Mean levels in Otters from the Georgia lower coastal plain	7.53	4.42	Halbrook et al. (1994)
Mean levels in Otters from the Georgia piedmont		1.48	Halbrook et al. (1994)

2.5 Measurement Data from Remote Locations

The Long Range Transport Analysis (Chapter 5) focusses on the long range atmospheric transport of mercury and estimates its impact at remote sites. This type of analysis was selected based on the atmospheric chemistry of emitted elemental mercury (Petersen et al., 1995) and the numerous studies linking increased mercury levels in air, soil, sediments, and biota at remote sites to distant anthropogenic mercury release followed by long-range transport. Details of several of the many studies which demonstrate the long range transport of mercury follow. These provide evidence to support this assessment of long-range mercury transport.

2.5.1 Elevated Atmospheric Mercury Concentrations over Remote Locations

Olmez et al. (1994) correlated elevated atmospheric levels of particulate mercury at rural U.S. sites to long range transport from distant sources. Briefly, Olmez et al. (1994) collected ambient particulates of two sizes (< 2.5μm and between 2.5μm and 10μm) for two years at five rural sites in New York State and measured levels of numerous pollutants. Using a pollutant fingerprinting technique, the collected data were evaluated to identify the pollutant sources. Mercury was considered to be a tracer pollutant for mixed industry and coal combustion. There were no local anthropogenic mercury sources at these sites. At the five sites the average sub- 2.5μm particulate mercury concentrations ranged from 0.051 to 0.089 ng/m^3, and the 90th percentile particulate mercury levels ranged from 0.21 to 0.10 ng/m^3. The highest values reported were 0.63 ng/m^3. Elevated mercury levels were attributed to long-range transport from industrial sources in Canada as well as parts of New York State and occasionally the midwest U.S. The authors noted that only 1-10% of the total mercury in remote areas is generally thought to be found on particles. Preliminary vapor-phase analysis (on samples collected for months) indicated that the mercury attached to these small particulates accounted for only 1.8% of the total mercury at these rural sites.

Glass et al. (1991) reported that mercury released from distant sources (up to 2500 km distant) contribute to mercury levels in rain water deposited on remote sites in northern Minnesota.

2.5.2 Elevated Soil Mercury Concentrations in Locations Remote from Emission Sources

Increased concentrations of mercury have been reported in both remote U.S. (Nater and Grigal, 1992) and Swedish soils (as reviewed in Johansson et al:, 1991 and by the Swedish Environmental Protection Agency, 1991). These elevated concentrations have been correlated with regional transport and deposition of mercury to soil. Nater and Grigal (1992) found an increasing mercury gradient from west to east in soils across the upper midwest U.S. This increase was also found to correlate with increasing regional industrialization. Briefly, soils were sampled in 155 different forest stands representing five types of forested stands. Mercury levels were measured in three layers: the surface detritus, surface soil (0-25 cm) and deep mineral soil (75-100 cm). Increases were observed along the west-east gradient in the upper two layers. The highest values reported for the detritus layer and the surface soil layer were >150 ng Hg/g detritus and >200 ng Hg/g soil, respectively. Differences in the ability of various soil types to bind mercury was discounted as a possible reason for the range of mercury values. The authors felt that their results implicated regional source contributions. Data summarized in Johansson et al. (1991) and the Swedish Environmental Protection Agency (1991) indicates that mercury levels in remote soils of southern Sweden are elevated when compared to those in the north. The increase observed in the soils of southern Sweden is related to emissions from regional Swedish industry and East European industry (Hakanson et al., 1990).

2.5.3 Elevated Mercury Concentrations in Aquatic Sediments and Fish from Remote Water Bodies

Elevated mercury levels in remote water body bed sediments have been widely reported and well characterized in many different parts of the world. These elevated levels are related to increased levels of atmospheric mercury which have been linked to anthropogenic activities. For example Swain et al. (1992) showed that, based on the vertical distribution of mercury in sediment, mercury deposition from the atmosphere over Wisconsin and Minnesota had increased from approximately 3.7 to 12.5 $\mu g/m^2$ since 1850 causing increases in sediment levels. For similar data from remote Wisconsin lakes, remote lakes in Ontario (Canada) and from remote Scandinavian bogs see Rada et al. (1989), Evans (1986), and Jensen and Jensen (1991), respectively. Some of the sediment analysis data for Sweden is presented in the report on mercury by the Swedish Environmental Protection Agency (1991).

The regional and widespread nature of mercury pollution was first identified when elevated levels of mercury in fish were discovered. These elevated levels in fish were evidence of the efficient transfer of mercury from prey to predator through the aquatic food chain (Watras and Bloom, 1992). In fact, the bioaccumulative nature of the mercury in fish has generated much of the interest in the measurement of mercury in other environmental media. It should be noted that the data of Hakanson et al. (1990) indicate that mercury levels in Swedish piscivorous fish continue to increase.

Elevated mercury concentrations in fish, particularly higher trophic level fish (e.g., northern pike) have been measured at sites distant from anthropogenic sources in Sweden (Hakanson et al., 1988; Swedish Environmental Protection Agency, 1991) and across the U.S. (e.g., Grieb et al., 1990; Sorensen et al., 1990 and Weiner et al., 1990). The report by Cunningham et al. (1994) illustrates the widespread nature of mercury fish advisories across the U.S.

2.6 Measurement Data Near Anthropogenic Sources of Concern

Measured mercury levels in environmental media around a single anthropogenic source are briefly summarized in this section. These data are not derived from a comprehensive study for mercury around the sources of interest. Despite the obvious needs for such an effort, such a study does not appear to exist. The quality of the following studies has not been assessed in this Report. The data do not appear to be directly comparable among themselves because of differences in analytic techniques and collection methods used. Finally, some of these studies are dated and may not reflect current mercury emissions from the sources described below.

Because these data do not conclusively demonstrate or refute a connection between anthropogenic mercury emissions and elevated environmental levels, a modeling exercise was undertaken to examine further this possible connection. This exercise is described in Chapters 3 and 4 of this document. The conclusions are discussed in Section 5.2 and 5.3. Materials in Appendices A-G support the modeling effort.

2.6.1 Municipal Waste Combustors

Bache et al. (1991) measured mercury concentrations in grasses located upwind and downwind from a modular mass-burn municipal waste combustor located in a rural area. The facility reportedly had no pollution control equipment and had been operating for about seven years when the grasses were sampled. Mercury levels were measured in air-dried grass samples by the flameless atomic absorption method developed by Hatch and Ott (1968). The sensitivity and detection limit of the method were not reported. Mercury levels in grass located downwind (along the prevailing wind direction) from the stack decreased with distance beginning at 100 m and continuing through 900 m.

The highest value recorded downwind of the facility was 0.2 μg mercury/g grass (dry weight) at 100 m. The highest reported value upwind (225 meters in the opposite direction from the prevailing wind direction) of the facility was 0.11 μg/g (dry weight). All other upwind values including measurements closer to the facility were 0.05 μg/g or less.

In response to a Congressional mandate, U.S. EPA assessed the "environmental impact of municipal waste incineration facilities" (U.S. EPA, 1991). Background levels of mercury were measured in air, soil, water and biota in the area around an MWC in Vermont. The facility, which had a 50 m stack, was not yet operational when the initial set of measurements were made. Pollution control equipment included an electrostatic precipitator (ESP) and a wet scrubber. After the facility had begun operating, pollutant levels were again measured. After the start-up of operations mercury emissions were measured at approximately 2×10^{-4} g/s. Mercury levels above the analytical detection limits or above background levels were not observed in this analysis. Problems were noted with some of the analytical equipment used for ambient air monitoring. The MWC was also not operational during some of the time after start-up, and there was a short time (10 months) between operation start-up and environmental measurement data collection.

Greenberg et al. (1992) measured mercury levels in rainwater near a rural New Jersey municipal resource recovery facility (MWC). The measurement protocols developed by Glass et al. (1990) were employed in the analysis. The 2-stack MWC had a 400-ton/day capacity, and pollution control included a dry fabric filter. The maximum allowable mercury emissions were 0.05 pounds/hour/stack (22.7 grams/hour/stack). During one collection period, state-mandated stack testing indicated that the facility was emitting mercury at levels slightly lower than the maximum allowable emission rates. Rain water was collected and analyzed on three separate 2-day time periods; the facility was not operating during one collection period. Collection sites were generally located in the prevailing wind directions. Mercury concentrations in rain water appeared to be elevated near the facility in the prevailing wind directions when compared with measurements taken when the facility was not operating and with measurements at more remote sites (>2 km). Mercury concentrations in rain water measured up to 2 km from the facility while it was not operating exhibited a range of 26 - 62 ng mercury/L rainwater (26-62 ppt). Mercury measurements at sites 3 - 5 km downwind did not exceed 63 ng mercury/L rain water. During facility operation the highest measured mercury concentration was 606 ng/L. The measurement was taken 2 km in the prevailing wind direction. Several other measurements of greater than 100 ng mercury/L rain water were also collected within 2 km of the facility.

Carpi et al. (1994) measured mercury levels in moss and grass samples around a MWC in rural New Jersey (same facility as Greenberg et al., 1992 studied). Pollution control equipment on the MWC reportedly included a spray dryer and a fabric filter. Samples were collected at sites up to 5 km from the source and mercury levels measured by a cold vapor atomic absorption spectroscopy method described in U.S. EPA (1991). Statistically significant elevations in mercury concentrations were measured in moss samples located within 1.7 km of the facility with the highest mercury measured levels exceeding 240 parts per billion (ppb). Oven-dried moss samples had lower levels of mercury than those samples that were not oven-dried. This was attributed to the loss of volatile mercury species during drying. The decrease in total mercury was most notable in moss samples at more distant sites (beyond 2 km from the facility). The authors felt that this might indicate the uptake and retention of different species during drying. The results of the analysis of grass samples were not presented. They were termed "inconclusive" in that they did not appear to exhibit point source influence.

2.6.2 Chlor-Alkali Plants

Temple and Linzon (1977) sampled the mercury content of foliage, soil, fresh fruits, vegetables and snow around a large chlor-alkali plant in an urban-residential area. This facility produced 160 tons of chlorine/day, resulting in approximately 0.8 kg/day of mercury emissions. Resulting mercury concentrations were compared to background levels from an urban area 16 km to the west. Mercury levels averaged 15 μg/g (300 times the background level of 0.05 μg/g) in maple foliage up to 260 m downwind, and concentrations 10 times background were found 1.8 km downwind. Mercury levels in soil averaged 3 μg/g (75 times the background level of 0.04 μg/g) within 300 m of the plant, and soil concentrations averaged 6 times background 1.8 km downwind. The mercury levels in snow ranged from 0.9-16 μg/L within 500 m of the plant dropping to 0.10 μg/l 3 km downwind. The background level was found to average 0.03 μg/L. Leafy crops were found to accumulate the highest mercury among garden produce. One lettuce sample contained 99 ng/g (wet w.) of mercury (background: <0.6 ng/g), and a sample of beet greens contained 37 ng/g (wet w.) (background: 3 ng/g). Tomatoes and cucumbers within 400 m averaged 2 and 4.5 ng/g (wet w.) of mercury. Background levels in each case measured 1 ng/g.

In one of the earliest reports which measured mercury levels around an industrial emission source, Jernelov and Wallin (1973) found elevated levels of mercury in the snow around five chlor-alkali facilities in Sweden. As distance from the facility increased, the amount of mercury detected decreased. They linked the elevated levels to source emissions.

Tamura et al. (1985) measured mercury concentrations in plant leaves and humus from areas with and without mercury emission sources in Japan. Data on total mercury concentrations were determined by cold flameless atomic absorption. Mercury concentrations were determined at four sites within 2 km of a currently operating chlor-alkali electrolysis plant. This facility was estimated to release 10-20 kg of mercury per year. Mercury concentrations at the four sites near this area ranged from 0.04-0.71 μg/g in woody plant leaves, 0.05-0.59 μg/g in herbaceous plants, and 0.11-2.74 μg/g in humus. In contrast, mercury levels for identical species of plants in the uncontaminated area (three sites) ranged from 0.02-0.07 μg/g in woody plant leaves, 0.02-0.08 μg/g in herbs, and 0.02-0.59 μg/g in humus. Values are typically on the order of 5-10 times less than mercury levels from the contaminated area, showing significant mercury contamination of plant biota can result from local point sources.

2.6.3 Coal-Fired Utilities

Crockett and Kinnison (1979) sampled the arid soils around a 2,150 megawatt (MW) coal-fired power plant in New Mexico in 1974. The four stack (two stacks 76 m high and two 91 m high) facility had been operational since 1963 with an estimated mercury release rate of 850 kg/year. The rainfall in the area averaged 15-20 cm/year. Although a mercury distribution pattern was noted, soil mercury levels near the facility did not differ significantly from background. Given the high amounts of mercury released by the facility and the insignificant amounts detected, the authors speculated that much of the mercury emitted was transported over a large area, rather than depositing locally.

Anderson and Smith (1977) measured mercury levels in environmental media and biota around a 200 MW coal-fired power plant in Illinois. The facility used two 152 m high smokestacks and was equipped with an electrostatic precipitator. Commercial operations at the facility had been ongoing for 6 years when sampling was conducted (from 1973 through 1974). Elevated levels of mercury detected in atmospheric particulate samples collected 4.8 and 9.6 km downwind of the facility were not statistically significant when compared with samples collected 4.8 km upwind of the site. Elevated mercury levels detected in samples from the upper 2 cm of downwind agricultural soils (sample mean

0.022 ug/g mercury) were statistically significantly elevated when compared with upwind samples (0.015 ug/g mercury). Core sediment sampling from a nearby lakebed showed statistically significant elevations in sediment mercury concentrations after plant operations began (sample mean 0.049 ug/g mercury) when compared with sediment deposits prior to operation (0.037 ug/g mercury). No increases were observed in mercury levels in fish from the nearby lake when compared with fish from remote lakes. Mercury levels in local duck muscle samples and aquatic plant samples were also reported but not compared to background or data from remote areas.

2.6.4 Mercury Mines

Lindberg et al. (1979) compared soil concentrations and plant uptake of mercury in samples taken one Km west of a mine/smelter operation in Almaden, Spain to levels found in control soils (20 Km east of the smelter). The most significant mercury release from the Almaden complex was from the ore roaster via a 30 m high stack; however, estimates of annual mercury releases were unavailable. Mine soils contained 97 μg/g of mercury compared to the control soil level of 2.3 μg/g, a 40 fold increase. Alfalfa was grown on these soils under controlled conditions. Comparing plant mercury concentrations (grown under conditions of no fertilizer or lime treatment), the above ground parts of alfalfa contained 1.4 and 2.3 μg/g of mercury in the control and mine soils, respectively. The roots of alfalfa contained 0.53 and 9.8 μg/g of mercury in the control and mine soils, respectively. The control levels in this experiment were found to exceed the worldwide average for grass crops by about 10 times; perhaps not surprising, since the control soil mercury content is also quite high. Nevertheless, additional mercury from the mine was found to elevate mercury content in surrounding soil and plant material significantly.

2.6.5 Mercury Near Multiple Local Sources

There are two recent reports of atmospheric mercury measurements in the vicinity of multiple anthropogenic emissions sources. Both are of studies are of short duration but show elevated mercury concentrations in the local atmosphere or locally collected rain.

Dvonch et al., (1994) conducted a 4-site, 20 day mercury study during August and September of 1993 in Broward County, FL. This county contains the city of Ft. Lauderdale as well as an oil-fired utility boiler and a municipal waste combustion facility. One of the sample collection sites (site 4) was located 300 m southwest of the municipal waste combustion facility. Daily measurements of atmospheric particulate and vapor-phase mercury were collected at 3 of the 4 sites; (daily atmospheric concentrations were not collected at the site near the municipal waste combustor (site 4)), and daily precipitation samples were collected at all sites. The average vapor and particulate phase atmospheric mercury concentrations were higher at the inland sites than at the site near the Atlantic Ocean, which was considered by the authors to represent background site. Diurnal variations were also noted; elevated concentrations were measured at night. For example at site 2, an inland site, the average nighttime vapor-phase concentration was 4.5 ng/m^3. This was attributed to little vertical mixing and lower mixing heights that occur in this area at night. Particulate mercury comprised less than 5% of the total (vaporous + particulate) atmospheric mercury. Mercury concentrations in precipitation samples at the 4 sites were variable; the highest mean concentrations were measured at the inland sites. Given the high levels of precipitation in this area of the U.S. and short collection period, it is not appropriate to extend these analysis beyond the time frame measured. These mercury concentrations are nonetheless elevated.

Table 2-26
Mercury Concentrations in the Atmosphere and Mercury Measured in Rainwater Collected in Broward County, FL

Site Description	Avg. Vapor-phase Mercury Conc., ng/m^3	Avg. Particulate Mercury Conc. pg/m^3	Avg. Total Mercury conc. in rain, ng/L (Range)	Avg. Reactive Mercury conc. in rain, ng/L (Range)
Background Near Atlantic Ocean (Site 1)	1.8	34	35 (15-56)	1.0 (0.5-1.4)
Inland (Site 2)	3.3	51	40 (15-73)	1.9 (0.8-3.3)
Inland (Site 3)	2.8	49	46 (14-130)	2.0 (1.0-3.2)
Inland (Site 4), 300 m from MWC	-	-	57 (43-81)	2.5 (1.7-3.7)

Keeler et., al. (1994) and Lamborg et al., (1994) reported results of a 10-day atmospheric mercury measurement at 2 sites (labeled as sites A and B) in Detroit, MI. There is a large MWC 9 Km from site A and a sludge combustor 5 Km from site B. It should be noted that other mercury emission sources such as coal-fired utility boiler and steel manufacturing occur in the city as well. The vapor-phase mercury concentration encountered at site B during the first days of the experiment exceeded the capacity of the measurement device. Subsequent analyses indicated that the concentrations of mercury encountered were significantly higher than other reported U.S. observations.

Table 2-27
Mercury Concentrations Measured at Two Sites in the Atmosphere Over Detroit, MI

Site	Mean Vapor-Phase Mercury Concentrations in, ng/m^3 (Maximum Measured Value)	Mean Particulate-Phase Mercury Concentrations in pg/m^3, (Maximum Measure Value)
Detroit, MI Site A	>40.8, (>74)	341 (1086)
Detroit, MI Site B	3.7, (8.5)	297 (1230)

2.6.6 Conclusion of Mercury Measurements Data

These data collectively indicate that mercury concentrations near these anthropogenic sources are generally elevated when compared with data collected at greater distances from the sources. The data can not be used in a systematic manner to estimate potential exposure especially for fish ingestion. There is a lack of data on mercury concentrations in fish near these sources; this is an important data gap. As a result the fate and transport of mercury emissions was modeled in this assessment.

3. INFORMATION ON EXPOSURE TO MERCURY

Chapters 4, 5, and 6 of Volume III present modeled mercury exposure to specified hypothetical individuals. Chapter 3 presents estimates of mercury exposure to humans from four different sources: food consumption in the general population, fish consumption, exposure through release from dental amalgams and occupational exposures. Finally, nationwide estimates of mercury exposure for piscivorous animals are also presented based on the animal's fish consumption rate and national averages of mercury concentrations in fish.

3.1 Nonoccupational Exposures to Mercury

3.1.1 Dietary Mercury

Food is the major source of total mercury intake by humans who are not occupationally exposed. There are several potential sources of exposure for inorganic mercury (elemental or divalent mercury); however, for the methylated species, food intake is the only significant source of exposure to the general human population (Stern, 1993; Swedish EPA, 1991; WHO, 1990). Methylmercury exposure primarily results from the ingestion of contaminated fish. Total mercury concentrations in meat and cereals often measures hundreds of times less than mercury in fish (Swedish EPA, 1991). In most non-fish foodstuffs mercury concentrations are typically near detection limits and are comprised of mainly inorganic species (WHO, 1990). In contrast, most of the mercury in fish is methylated.

3.1.1.1 Mercury In Food Sources Other Than Fish

The World Health Organization (WHO, 1990) estimated the quantities and species of mercury ingested through the fish and non-fish components of the adult diet. These estimates of the amount of mercury ingested and retained are presented in Table 3-1. The average daily intake of total mercury from fish and fish products was estimated to be 3 ug/day (the fish ingestion rates used to derive this value were not provided in the Report). WHO (1990) assumed that 80% of the total mercury in fish was methylmercury and the remaining 20% was inorganic divalent mercury; more recent estimates indicate that 95-100% of the total mercury in fish is methylmercury. The total mercury intake from non-fish foodstuffs was calculated as the difference between the estimated total dietary mercury intake and the estimated total mercury intake from fish. WHO (1990) assumed that 95% of the methylmercury and 7% of the divalent inorganic mercury ingested would be retained. WHO (1990) estimated total dietary mercury intake by averaging the estimates of the FDA market basket survey of 1984-1986 (3.5 ug/day for a 70 kg adult) and the mean (9.8 ug/day) of the results of two Belgian studies (Fouassin and Fondu, 1978 and Buchet et al., 1983). Data on these and other estimates of total dietary intake of mercury are summarized in Table 3-2.

Table 3-1
Estimated Average Adult Daily Intake (and retention) of Mercury Compounds by the General Public (ug/day) as Reported by WHO 1990

	Elemental Mercury ug/day	Divalent Mercury ug/day	Methylmercury ug/day
Fish	0	0.60 (0.042)	2.4 (2.3)
Non-Fish	0	3.6 (0.25)	0

Table 3-2
Reported Total Adult Mercury Intake Rates (ug/day)

Year	Amount	Population	Reference
1977	6.3	USA	Podrebarac (1984)
1978	3.4	USA	Podrebarac (1984)
1977	13	Belgium	Fouassin and Fondu (1978)
1982	6.5	Belgium	Buchet et al., (1983)
1984-1986	3.5	USA	Shibko (1988)

WHO, 1990 acknowledges some uncertainty in these estimates; the total estimated amount of dietary inorganic mercury (4.3 ug/day) was thought to be the least reliable estimate in the data presented in Table 3-1. The 1990 analysis by the WHO indicates that comparable amounts of inorganic and methylmercury are ingested through the dietary pathway. Since gut uptake from ingestion of the organic form is expected to be much greater than inorganic species [95% for methylmercury vs. 7% for inorganic (EPA 1988, WHO, 1990)], fish ingestion of methylmercury will dominate the body burden of total mercury from the dietary pathway.

The WHO (1990) analysis does not indicate the specific foods from which the non-fish dietary intake of mercury is derived. Podrebarac (1984) analyzed the mercury concentrations in non-fish foodstuffs; that is total mercury in food samples collected in market basket surveys conducted by the FDA from October 1977 - September 1978. Twenty samples from each food group were examined and mercury was detected in at least one sample from each food group with the exception of beverages. Podrebarac's results for total mercury concentrations in various food groups are summarized in Table 3-3. It should be noted that, in contrast to measurements of methylmercury, a greater confidence is generally associated with the results of analytic measurement techniques for assessing total mercury using the chemical methods available during the 1970s.

In terms of both average mercury concentrations and overall number of samples in which Mercury was detected, Podrebarac's grouping of meat, fish and poultry clearly dominates the food groups considered for this analysis. A delineation of the meat, fish and poultry group into individual food types was not given, and a determination of how much mercury was recorded in marine fish products is not possible. The concentrations in all plant foods are quite low, which is expected given the relatively low accumulation of mercury by plants. Based on these results, Podrebarac (1984) calculated the total adult dietary intake of mercury for the years 1977 and 1978 to be 6.3 and 3.4 ug/day, respectively.

The species of mercury ingested in non-fish foods is generally assumed to be inorganic, divalent mercury; for example, in the WHO 1990 analysis the total mercury in the non-fish foodstuffs is assumed to be primarily inorganic divalent mercury. It should be noted that there are reports which indicate that up to a third of the total mercury in some garden plants may be methylated (Cappon 1981, 1987).

Table 3-3
Total Mercury Levels in Various Food Groups from Podrebarac (1984)

Food Group	Number of Samples with detectable Mercury	Mean Mercury concentration based on all 20 samples (ug/g wet weight)	Range of concentrations in samples with detectable Mercury (ug/g wet weight)
Dairy Products	1	0.0001	0.002
Meat, Fish and Poultry	16	0.0091	0.004-0.027
Grains and Cereal	5	0.0014	0.003-0.008
Potatoes	2	0.0004	0.004-0.005
Leafy Vegetables	3	0.0006	0.003-0.005
Legume Vegetables	3	0.0011	0.005-0.009
Root Vegetables	3	0.0007	0.004-0.005
Garden Fruits	1	0.0001	0.002
Fruits	2	0.0002	0.001-0.004
Oils, Fats and Shortening	5	0.0014	0.004-0.009
Sugar and adjuncts	1	0.0001	0.002

Richardson et al. (1995) estimated the daily mercury intake of Canadians based on average estimated consumption rates and the central tendencies of measured mercury concentrations in environmental media and biota. They assumed an urban setting for adults, children, and infants. Several key assumptions pertaining to the species of mercury were also employed in the assessment: 1) the species of mercury in fish was methylmercury; 2) the species of mercury in all other commercial foods and soil was Hg^{2+}; 3) the species of mercury in drinking water was 75% Hg^{2+} and 25% methylmercury; 4) the species of mercury in indoor air was assumed to be 100% Hg^{0}; mercury in the outdoor air was assumed to be 75% Hg^{0}, 20% "organic mercury" in the vapor phase and 5% Hg^{2+} bound to respirable particles; and 5) the species of mercury emitted from dental amalgams was Hg^{0}. Food exposures were predicted to be the most significant route of exposure to both methylmercury and Hg^{2+} and dental amalgams were the most significant route of exposure for total mercury. The specified daily fish consumption rates used did not appear to be presented in Richardson et al. (1995). The methylmercury concentration used were these: canned tuna was 0.195 ug/g (range <0.01-0.97); in other commercial fish was 0.137 ug/g (range 0.02-1.4); in shellfish was 0.024 ug/g (range <0.01-1.4); and in non-commercial fish was 0.38 ug/g (range 0.01-13.0). Given the assumptions employed in this estimate, amalgams accounted for 17-42% of the total absorbed mercury. The daily adult (20+years of age) intake of total mercury via all exposure routes was estimated to be 7.737 ug or 0.11 ug/kg bw/day; 36% of the intake was due to mercury released from dental amalgams and 27% to fish consumption. The daily child (5-11 years of age) intake to total mercury via all exposure routes was estimated to be 3.891 ug (36% of the intake was due to fish consumption and 13% from dental amalgams).

3.1.1.2 Mercury from Fish

As described previously, methylmercury is the primary form of mercury in fish. The purpose of this section is to estimate the magnitude of these exposures to methylmercury in both the general fish-consuming U.S. population and in specific fish-consuming subpopulations (e.g., children and women of child-bearing age). This analysis is briefly described here; a more detailed description of this analysis is provided in Appendix H of this volume.

Estimates of fish consumption rely on dietary survey data that can be obtained using a variety of dietary survey techniques. Critical elements in any survey aimed at determining intake of methylmercury from fish are these:

- Species of fish or shellfish consumed;
- Concentration of methylmercury in the fish; and
- Quantity of fish consumed.

The duration of fish consumption is also of importance; however, the time period of consumption that is relevant when conducting an assessment of risk depends on the health endpoint of concern. To illustrate, acute effects of certain fish contaminants (such as paralytic shell fish toxin or cigutara toxin) may result from eating as little as one meal of contaminated fish. By contrast, if one is interested in the benefits of consuming unsaturated fatty acids (e.g., omega 3 fatty acid) on prevalence of cardiovascular disease, decades of exposure for a group of persons is typically required to establish whether or not an effect would occur. For a health endpoint such as developmental deficits associated with a particular period during gestation (e.g., adverse effects of maternal consumption of methylmercury from fish on the developing fetal nervous system), short-term consumption patterns during the critical weeks or months of gestation are considered the relevant period for the health endpoint.

Survey methods can broadly be classified into longitudinal methods or cross-sectional surveys. Typically long-term or longitudinal estimates of intake can be used to reflect patterns for individuals (e.g., dietary histories); or longitudinal estimates of moderate duration (e.g., month-long periods) for individuals or groups. Cross-sectional data are used to give a "snap shot" in time and are typically used to provide information on the distribution of intakes for groups within the population of interest. Cross-sectional data typically are for 24-hour or 3-day sampling periods and may rely on recall of foods consumed following questioning by a trained interviewer, or may rely on written records of foods consumed. Additional discussion of these issues are found on Appendix H to Volume III.

During the past decade reviewers of dietary survey methodology (for example, the Food and Nutrition Board of the National Research Council/National Academy of Sciences; the Life Sciences Research Office of the Federation of American Societies of Experimental Biology) have evaluated various dietary survey techniques with regard to their suitability for estimating exposure to contaminants and intake of nutrients. The Food and Nutrition Board of the National Research Council/National Academy of Sciences in their 1986 publication on Nutrient Adequacy Assessment Using Food Consumption Surveys noted that dietary intake of an individual is not constant from day to day, but varies both in amount and in type of foods eaten (intraindividual variation). Variations between persons in their usual food intake averaged over time is referred to as interindividual variation. Among North American populations, the intraindividual (within person day-to-day) variation is usually regarded to be as large as or greater than the interindividual (person to person) variations. Having evaluated a number of data sets the Academy's Subcommittee concluded that three

days of observation may be more than is required for the derivation of the distribution of usual intakes.

Major sources of data on dietary intake of fish used in preparing this report to Congress are the cross-sectional data from the USDA Continuing Surveys of Food Intake by Individuals conducted in the years 1989 through 1991 (CSFII 89/91) and the longer-term data on fish consumption based on recorded fish consumption for variable numbers of periods of one-month duration during the years 1973/1974 from the National Purchase Diary (NPD 73/74) conducted by the Market Research Corporation.

Human mercury intake from fish was estimated by combining data on mercury concentrations in fish species, expressed as micrograms of mercury per gram fresh-weight of fish tissue, with the reported quantities and types of fish species consumed by fish eaters or "users" in the USDA's Continuing Surveys of Food Intake by Individuals (CSFII 89/91). The dietary assessment methodology consisted of an assessment of three consecutive days of food intake, measured through one 24-hour-recall and two 1-day food records. For this analysis, the sample was limited to those individuals who provided records or recalls of three days of dietary intake. Respondents were drawn from stratified area probability samples of non-institutionalized United States households. Survey respondents were surveyed across all four seasons of the year and all seven days of the week. Respondents were also asked to report their body weights, and these data were utilized to estimate fish consumption on a per body weight basis.

The CSFII 89/91 data are cross-sectional data based on a three-day sampling period. When appropriately weighted the data can be used to estimate the food consumption patterns for the general United States population for the period 1989/1991. The survey was designed to represent all seasons of the year and all days of the week. Because of the food consumption records rely on standard coding of food intake and records of types of fish represented by a particular dietary records it is possible to estimate how much of particular types of fish were consumed for the population as a whole and for subpopulations of interest. The portion size consumed by individuals is recorded, as is the person's individual indication of their body weight.

The CSFII 89/91 data on fish consumption have been used to estimate fish and methylmercury intake by various population subgroups. These calculations rely on values for the methylmercury concentrations in food supplied to the U.S. EPA by the National Marine Fisheries Service. These data are presented in detail in Appendix H to Volume III.

For nationally representative weighted samples of individuals, 30.9% reported consumption of fish and/or combinations of fish, shellfish, or seafood with vegetables or starches in a 3-day period. Of individuals reporting fish consumption, approximately 98% consumed fish only once, and about 2% consumed fish in two or more meals during the 3-day survey period. For less-frequently consumed foods, estimates of per capita consumption rates overestimate the consumption rate among the general population but underestimate the consumption rate among the portion of the population which actually consumes the food item. As a consequence, fish consumption estimates are based on a "per consumer" basis.

The fish consumption rates used from CSFII 89/91 reflect the consumption of approximately 250 individual "fish only" food codes, and approximately 165 "mixed dish-fish" food codes present in the 1994 version of the USDA food composition tables. The USDA recipe file was searched for food codes containing fish or shellfish. The recipe was then scanned to determine fish codes that were present in the recipe reported as consumed by the survey respondent. The percent of the recipe that

was fish by weight was determined by dividing the weight of the fish/shellfish in the dish by the total weight of the dish. Consumption of fish-only and mixed-fish-dishes was summed across the three available days of dietary intake data. This sum was then divided by three to create average fish consumption per day estimates.

Data describing methylmercury concentrations in marine fish were predominantly based on the National Marine Fisheries' Service (NMFS) data base, the largest publicly available data base on mercury concentrations in marine fish. Data reported by Bahnick et al. (1994) and Lowe et al., 1985 were used to estimate average mercury concentrations in fresh-water finfish from across the U.S. (These data are in Chapter 2 and in Appendix H of this volume).

Table 3-4 shows the percentiles of the fish ingestion rates and corresponding methylmercury exposures for the entire fish-eating population and several fish-eating subpopulations. When the methylmercury intake is expressed on a per kilogram self-reported body weight basis, the exposure of children aged 14 years and younger is approximately two-to-three times that of the adult. This is the result of the higher intake of food on a per weight basis among children. The methylmercury intake of adult males and females is comparable. The maximum intakes on a per kilogram body weight basis are also provided for each group considered. Note that the intake for the maximum respondent in each group of adults is at least 4 times that of the intake for the individual at the 95th percentile.

Methylmercury intakes calculated in this Volume have been developed for nationally-based rather than site-specific estimates. The CSFII/89-91 from USDA was designed to represent the United States population. The concentrations of methylmercury in marine fish and shellfish were taken from a data base that is national in scope and the data on fresh-water finfish were from a large study that sampled fish at a number of sites throughout the United States. The applicability of these data to site-specific assessments must be judged on a case-by-case basis.

The purpose of the estimates of methylmercury intake from fish is to describe current methylmercury intake from ingestion of fish. There is no attempt to attribute the methylmercury concentrations in dietary fish to any anthropogenic or other source. Because of the magnitude of anthropogenic, ambient mercury contamination, the estimates of methylmercury from fish do not provide a "background" value. "Background" values imply an exposure against which the increments of anthropogenic activity could be added. This is not the situation due to release of substantial quantities into the environment.

Conclusions on methylmercury consumption from fish ingestion must include consideration of variability and uncertainty in these estimates. Uncertainty arises through both the method used to estimate fish consumption and the assumed methylmercury concentrations in the fish consumed. See Appendix H for a detailed discussion of the uncertainty.

3.1.1.3 Other Estimates of Human Mercury Intake from Fish

Several other studies estimate the amount of mercury ingested as a result of fish consumption (WHO, 1990; Cramer, 1994; Tollefson and Cordle, 1986; Hall et al., 1978; Lipfert et al. 1994). Of all the possible exposures to background mercury that may occur, marine seafood ingestion is the most important for the general population (WHO, 1990). Marine seafood consumption is likely to be the only source of methylmercury besides freshwater fish consumption for the general U.S. population.

Estimates of fish consumption in the 1970s were determined by the NPD Research Inc., a market research and consulting firm that specializes in the analysis of consumer purchasing behavior

Table 3-4
Fish and Shellfish Consumption Rates and
Predicted Methylmercury Doses for Respondents of the 1989-1991 CSFII Survey

Gender	%	Age (yr)											
		< 15			15 - 45			>45			Total		
		N	Fish g/day	MHg[a] ug/kg/day	N	Fish g/day	MHg[a] ug/kg/day	N	Fish g/day	MHg[a] ug/kg/day	N	Fish g/day	MHg[a] ug/kg/day
Males	Min	380	1.9	0.0	646	1.9	0	556	2.5	0	1582	1.9	0
	5th		4.5	0.03		9.0	0.01		10.2	0.01		7.2	0.01
	25th		14.2	0.09		21.8	0.04		25.6	0.04		20.4	0.05
	50th		23.1	0.17		41.6	0.08		41.5	0.08		37.1	0.10
	75th		43.3	0.29		70.0	0.16		69.0	0.15		62.3	0.18
	95th		87.2	0.63		140.0	0.34		142.0	0.35		125.3	0.46
	Max		139.4	1.51		312.0	1.98		388.9	1.57		388.9	1.98
Females	Min	340	1.0	0.0	864	1.3	0	828	1.9	0	2032	1.0	0.0
	5th		4.7	0.02		7.0	0.01		7.4	0.01		6.2	0.01
	25th		13.5	0.08		18.7	0.04		18.9	0.04		18.6	0.04
	50th		23.9	0.17		30.6	0.08		31.8	0.08		29.0	0.09
	75th		37.9	0.34		56.0	0.16		56.0	0.14		53.2	0.18
	95th		74.9	0.85		112.9	0.33		108.0	0.32		107.0	0.44
	Max		154.0	1.69		461.0	2.76		250.2	1.67		461.0	2.76

[a] MHg = methylmercury

as recorded in monthly diaries. That survey was funded by the Tuna Research Institute (TRI) as part of a study of tuna consumption. Later, the National Marine Fisheries Service (NMFS) received permission from TRI to obtain the data (SRI International Contract Report to U.S. EPA, 1980).

The NPD 73/74 data are based on a sample of 7,662 families (25,165 individuals) out of 9,590 families sampled between September 1973 and August 1974. Data recorded in the survey reflect the marketing nature of the survey design and have limitations with regard to quantities of fish consumed on a body weight basis. To illustrate, the fish consumption was based on questionnaires completed by the female head of the household in which she recorded the date of any meal containing fish, the type of fish (species), the packaging of the fish (canned, frozen, fresh, dried, or smoked, or eaten out), whether fresh fish was recreationally caught or commercially purchased, the amount of fish prepared for the meal, the number of servings consumed by each family member and any guests, and the amount of fish not consumed during the meal. Meals eaten both at home and away from home were recorded.

Use of these data to estimate intake of fish or mercury on a body weight basis are limited by the following data gaps.

1. This survey did not include data on the quantity of fish represented by a serving and information to calculate actual fish consumption from entries described as breaded fish or fish mixed with other ingredients. Portion size was estimated by using average portion size for seafood from USDA Handbook #11, Table 10, page 40-41. The average serving sizes from this USDA source are shown in Table 3-5.

Table 3-5
Average Serving Size (gms) for Seafood from USDA Handbook #1
Used to Calculate Fish Intake by FDA (1978)

Age Group (years)	Male Subjects (gms)	Female Subjects (gms)
0-1	20	20
1-5	66	66
6-11	95	95
12-17	131	100
18-54	158	125
55-75	159	130
Over 75	180	139

2. There may have been systematic under-recording of fish intake; Crispin-Smith et al. noted that typical intakes declined 30% between the first survey period and the last survey period among persons who completed four survey diaries (Crispin-Smith et al., 1985).

3. There have been changes in the quantities and types of fish consumed between 1973/1974 and present. To illustrate, the U.S. Department of Agriculture indicated (Putnam, 1991) that on average, fish consumption increased 27% between 1970 to 1974 and 1990. Whether or not this increase applies to the highest percentiles of fish consumption (e.g., 95th or 99th percentile) was not described in the publication by USDA.

4. An analyses of these data using the sample weights to project estimates for the general United States population was prepared by SRI International under U.S. EPA Contract 68-01-3887 in 1980. U.S. EPA was subsequently informed that the sample weights were no longer available. Consequently additional analyses with these data in a manner than can be projected to the general population appears to be no longer possible.

5. Body weights of the individuals surveyed do not appear in published materials. If body weights of the individuals participating in this survey were recorded these data do not appear to have been used in subsequent analyses.

Data on fish consumption from the NPD 73/74 survey have been published by Rupp et al. (1980) and analyzed by U.S. EPA's contractor SRI International (1980). These data indicate that when a month-long survey period is used, 94% of the surveyed population consumed fish. The species of fish most commonly consumed are shown in Table 3-6.

Table 3-6
Fish Species and Number of Persons Using the Species of Fish
(Adapted from Rupp et al. 1980)

Category	Number of Individuals Consuming Fish Based on 24,652 Replies*
Tuna, light	16,817
Shrimp	5,808
Flounders	3,327
Not reported (or identified)	3,117
Perch (Marine)	2,519
Salmon	2,454
Clams	2,242
Cod	1,491
Pollock	1,466

* More than one species of fish may be eaten by an individual.

Rupp et al. also estimated quantities of fish and shellfish consumed by 12-18 year-old teenagers and by adults 18 to 98 years of age. These data are shown in Table 3-7. The distribution of fish consumption for age groups that included women of child-bearing age are shown in Table 3-8.

Table 3-7
Fish Consumption from the NPD 1973/1974 Survey
(Modified from Rupp et al. 1980)

Age Group	50th Percentile	90th Percentile	99th Percentile	Maximum
Teenagers Aged 12-18 years	1.88 kg/year	8.66 kg/year	25.03 kg/year or 69 grams/day	62.12 kg/year
Adults Aged 18 to 98 Years	2.66 kg/year	14.53 kg/year	40.93 kg/year or 112 grams/day	167.20 kg/year

Table 3-8
Distribution of Fish Consumption for Females By Age*
Consumption Category (grams/day) (from SRI, 1980)

Age (years)	47.6-60.0	60.1-122.5	Over 122.5
10-19	0.2	0.4	0.0
20-29	0.9	0.9	0.0
30-39	1.9	1.7	0.1
40-49	3.4	2.1	0.2

* The percentage of females in an age bracket who consume, on average, a specified amount (grams) of fish per day. The calculations in this table were based upon the respondents to the NPD survey who consumed fish in the month of the survey. The NPD Research estimates that these respondents represent, on a weighted basis, 94.0% of the population of U.S. residents (from Table 6, SRI Report, 1980).

Using the data in the 1977-78 USDA food consumption survey, Tollefson and Cordle (1986) estimated that 86% of the total mercury in a non-angler's diet is derived from 4 basic groups of seafood items: canned tuna, shrimp, fish sticks, and cod/haddock fillets. A mercury intake rate of about 2 ug/day was estimated from the consumption of these products. Hall et al., 1978 calculated a per capita mercury intake from seafood of 2.6 ug/day. WHO, 1990 estimated the average daily intake of total mercury from fish and fish products to be 3 ug/day; the bulk of mercury ingestion was attributed to off-the-supermarket-shelf seafood. Using estimates of total seafood consumption rates for the adult Great Lakes states population that consumes freshwater fish, Lipfert et al., (1994) calculated a mean mercury dose of 4.5 ug/day, based on a total seafood consumption rate of 24.7 g/day. About 64% of the mean total dose was derived from the consumption of freshwater finfish.

Cramer (1994) reported that the U.S. FDA had evaluated exposure to methylmercury through fish consumption in 3 different ways. In the first approach the 1988 Market Research Corporation of America (MRCA) 14-day fish consumption data was combined with an estimated average fish methylmercury concentration of 0.3 ug/g to estimate human methylmercury exposure. The results are summarized in Table 3-9. The second approach utilized Monte Carlo techniques to estimate

distributions of the 1982-1987 MRCA fish consumption data and mercury concentra-tions in fishery products. Based on this analysis, the 50th percentile intake estimate was 4.4 ug mercury/day and the 90th percentile intake estimate was 10 ug mercury/day. The third approach combined per capita fish consumption rates with the mercury concentrations in the top 10 consumed fish species and estimated an intake rate of 1.6 ug mercury/day.

Table 3-9
Estimates of Mercury Exposure Through the Consumption of Fish (Cramer 1994)

Estimates assuming fish concentration of 0.3 ug/g		
	Daily Mercury Intake (ug/person/day)	
Age (yr)	50th percentile	90th percentile
2-5	5	10
18-44	11	22
All Ages	10	19
Estimates based on 1982-87 MRCA data, Fish Mercury data and Monte Carlo analysis		
All Ages	4.4	10

For purposes of comparison, the mercury exposure estimates developed by Cramer (1994) were modified. The 50th and 90th percentile estimates of adult (Ages 18-44) exposure were divided by 70 Kg (assumed human body weight) to roughly estimate predicted exposure on a μg/Kg Bw/day basis. The results of this division are 0.16 and 0.31 μg/Kg Bw/day for the 50th and 90th percentiles, respectively.

3.1.2 Dental Amalgams

Dental amalgams have been the most commonly used restorative material in dentistry. A typical amalgam consists of approximately 50% mercury by weight. The mercury in the amalgam is continuously released over time as elemental mercury vapor (Begerow et al., 1994). Research indicates that this pathway contributes to the total mercury body burden, with mercury levels in some body fluids correlating with the amount and surface area of fillings for non-occupationally exposed individuals (Langworth et al., 1991; Olstad et al., 1987; Snapp et al., 1989). For the average individual an intake of 2-20 ug/day of elemental mercury vapor is estimated from this pathway (Begerow et al., 1994). Additionally, during and immediately following removal or installation of dental amalgams supplementary exposures of 1-5 ug/day for several days can be expected (Geurtsen 1990).

Approximately 80% of the elemental mercury vapor released by dental amalgams is expected to be re-absorbed by the lungs (Begerow et al., 1994). In contrast, dietary inorganic mercury absorption via the gastrointestinal tract is known the be about 7%. The contribution to the body burden of inorganic mercury is, thus, greater from dental amalgams than from the diet or any other source. The inorganic mercury is excreted in urine, and methylmercury is mainly excreted in feces. Since urinary mercury levels will only result from inorganic mercury intake, which occurs almost

exclusively from dietary and dental pathways for members of the general public, it is a reasonable biomonitor of inorganic mercury exposure. Urinary mercury concentrations from individuals with dental amalgams generally range from 1-5 ug/day, while for persons without these fillings it is generally less than 1 ug/day (Zander et al., 1990). It can be inferred that the difference represents mercury that originated in dental amalgams.

Begerow et al., (1994) studied the effects of dental amalgams on inhalation intake of elemental mercury and the resulting body burden of mercury from this pathway. The mercury levels in urine of 17 people aged 28-55 years were monitored before and at varying times after removal of all dental amalgam fillings (number of fillings was between 4-24 per person). Before amalgam removal, urinary mercury concentrations averaged 1.44 ug/g creatinine. In the immediate post-removal phase (up to 6 days), concentrations increased by an average of 30%, peaking at 3 days post-removal. After this phase mercury concentrations in urine decreased continuously and by twelve months had dropped to an average of 0.36 ug/g creatinine. This represents a four-fold decrease from pre-removal steady-state urinary mercury levels.

3.2 Occupational Exposures to Mercury

Industries in which occupational exposure to mercury may occur include chemical and drug synthesis, hospitals, laboratories, dental practices, instrument manufacture, and battery manufacture (National Institute for Occupational Safety and Health, (NIOSH) 1977). Jobs and processes involving mercury exposure include manufacture of measuring instruments (barometers, thermometers, etc.), mercury arc lamps, mercury switches, fluorescent lamps, mercury broilers, mirrors, electric rectifiers, electrolysis cathodes, pulp and paper, zinc carbon and mercury cell batteries, dental amalgams, antifouling paints, explosives, photographs, disinfectants, and fur processing. Occupational mercury exposure can also result from the synthesis and use of metallic mercury, mercury salts, mercury catalysts (in making urethane and epoxy resins), mercury fulminate, Millon's reagent, chlorine and caustic soda, pharmaceuticals, and antimicrobial agents (Occupational Safety and Health Administration (OSHA) 1989).

OSHA (1975) estimated that approximately 150,000 US workers are exposed to mercury in at least 56 occupations (OSHA 1975). More recently, Campbell et al., (1992) reported that about 70,000 workers are annually exposed to mercury. Inorganic mercury accounts for nearly all occupational exposures, with airborne elemental mercury vapor the main pathway of concern in most industries, in particular those with the greatest number of mercury exposures. Occupational exposure to methylmercury appears to be insignificant. Table 3-10 summarizes workplace standards for airborne mercury (vapor + particulate).

A number of studies have been reported that monitored workers' exposure to mercury (Gonzalez-Fernandez et al., 1984; Ehrenberg et al., 1991; Cardenas et al., 1993; Kishi et al., 1993, 1994; Yang et al., 1994). Some studies have reported employees working in areas which contain extremely high air mercury concentrations: 0.2 to over 1.0 mg/m^3 of mercury. Such workplaces include lamp sock manufacturers in Taiwan (Yang et al., 1994), mercury mines in Japan (Kishi et al., 1993,1994), a small thermometer and scientific glass manufacturer in the US (Ehrenberg et al., 1991), and a factory producing mercury glass bubble relays (Gonzalez-Fernandez et al., 1984). High mercury levels have been reported in blood and urine samples collected from these employees (reportedly over 100 ug/l in blood and over 200 - 300 ug/l or 100 - 150 ug/g creatinine for urine). At exposures near or over 1.0 mg/m3, workers show clear signs of toxic mercury exposure (fatigue, memory impairment, irritability, tremors, and mental deterioration). The chronic problems include neurobehavioral deficits

Table 3-10
Occupational Standards for Airborne Mercury Exposure

Concentration Standard (mg/m^3)	Standard Type	Mercury Species	Reference
0.10	STEL	inorganic	CFR (1989)
0.01	TWA	organic	CFR (1989)
0.03	STEL	alkyl	CFR (1989)
0.05	TWA	all besides alkyl	ACGIH (1986)
0.01	TWA	alkyl	ACGIH (1986)
0.03	STEL	alkyl	ACGIH (1986)
0.10	TWA	aryl and inorganic	ACGIH (1986)
0.05	TWA	all besides alkyl	NIOSH (1977)

Abbreviations:
ACGIH - American Conference of Governmental Industrial Hygienists
CFR - Code of Federal Regulations
STEL - Short term exposure limit (15 minutes)
TWA - Time weighted average (8 hour workday)

that persist long after blood and urine mercury levels have returned to normal; many workers required hospitalization and/or drug treatments. With the exception of mercury mines, workplaces producing these mercury levels are typically small and specialized, often employing only a few workers who were exposed to high mercury concentrations.

Many other studies have monitored employees' work areas and reported measured mercury air concentrations of 0.02 - 0.2 mg/m^3; these levels are generally in excess of present occupational standards (see Table 3-10). These mercury levels were most often reported at chlor-alkali plants (Ellingsen et al., 1993; Dangwal 1993; Barregard et al., 1992; Barregard et al., 1991; Cardenas et al., 1993). Employees at these facilities had elevated bodily mercury levels of approximately 10 - 100 ug/l for urine and about 30 ug/l in blood. At these lower levels, chronic problems persisting after retirement included visual response and peripheral sensory nerve effects.

Exposures to mercury levels under 0.02 mg/m^3 typically result in blood and urine levels statistically higher than the general population, but health effects are usually not observed.

3.3 Estimated Wildlife Exposure to Mercury

In this section the potential exposure to mercury for selected wildlife species is estimated. This is performed using national estimates of mercury concentrations in fish as well as consumption rates of freshwater fish by the wildlife species considered. The mercury concentrations in fish are from "A National Study of Chemical Residues in Fish" as conducted by U.S. EPA (1992) and as reported in Bahnick et al., (1994). Exposure was estimated for five piscivorous wildlife species: bald

eagle, osprey, kingfisher, river otter and mink. The ratio of the total daily fish ingestion rate to total body weight and the trophic level of the fish consumed contribute greatly to the estimated methylmercury exposure of the wildlife species. The basis for these assumptions are described in Volume V, An Ecological Assessment for Anthropogenic Mercury Emissions in the United States of this Report and are summarized in Table 3-11.

Table 3-11
Assumed Fish Consumption Rates and
Body Weights for Piscivorous Wildlife

Animal	Body Weight (kg)	Total Ingestion Rate (g/day)	% of Diet from Trophic Level 3 Fish	% of Diet from Trophic Level 4 Fish	% of Diet from Non-fish sources
Bald Eagle	4.6	500	74	18	8
Osprey	1.5	300	100	0	0
Kingfisher	0.15	75	100	0	0
River Otter	7.4	1220	80	20	0
Mink	0.8	178	90	0	10

This assessment of current wildlife exposure considers only the freshwater fish ingestion exposure route for five piscivorous wildlife species; other food sources such as amphibians, reptiles, and insects and other sources of mercury such as the animal's drinking water were not considered in this assessment. The form of mercury in contaminated fish was assumed to be monomethylmercury. The assessment provides a national methylmercury exposure estimate for piscivorous wildlife based on central tendancy estimates of methylmercury concentrations measured in fish. Regional or site-specific assessments would require local measurement data. In the absence of such information, assumptions and default values may require notification to be relevant to the local exposure scenario. For example, an assessment specific to regions bordering the Great Lakes assumed that Bald Eagles consume mercury-contaminated herring gulls as part of their non-fish diet (U.S. EPA 1995).

Two approaches could have been utilized to estimate the concentrations of methylmercury in the collected fish consumed by the wildlife. The mean methylmercury concentration for all sampled freshwater fish (0.26 ug/g) (Bahnick et al., 1994) could have been input for all fish consumed by wildlife, regardless of trophic level. Alternatively, mean methylmercury concentrations could have been calculated for the freshwater fish species collected by Bahnick et al., (1994) which represented aquatic trophic levels 3 and 4.

A modification of the second approach was selected as the most appropriate for this study. The fish species collected by Bahnick et al., (1994) can be categorized into 3 groups: 1) those that do not fit the 4-tier trophic level model used in this document for piscivore exposure; 2) those that are considered to be mixture of trophic levels 3 and 4; or 3) those that represent trophic level 4. The Bahnick data do not adequately describe the mercury concentrations in trophic level 3 fish, typically thought to be 30 - 40 g fish. As a result, the mean of the trophic level 4 species was divided by the trophic level 3 to trophic level 4 predator-prey factor to estimate the methylmercury concentrations in

trophic level 3 fish. (See Vol. V for further discussion of the predator-prey factor.) The mean of average concentrations measured in the following 3 trophic level 4 fish species, bass (0.38 ug/g), walleye (0.52 ug/g), and northern pike (0.31 ug/g), is 0.4 ug/g methylmercury. A mean concentration of 0.08 ug/g methylmercury was estimated for trophic level 3 fish; this is the dividend of 0.4 ug/g and the predator-prey factor of approximately 5.

Using the fish consumption data and the estimates of trophic level 3 and 4 fish concentrations derived from the data of Bahnick et al., (1994), estimates of current methylmercury intake from fish consumption are presented in Table 3-12. The data are presented in units of ug per kilogram of body weight per day.

Table 3-12
Estimates of Current U.S. Methylmercury Exposure from Fish Consumption by Piscivorous Birds and Mammals

Animal	Estimated Current Daily Methylmercury Exposure (ug/kg/day)
Bald Eagle	14
Osprey	16
Kingfisher	40
River Otter	24
Mink	16

4. MODELING THE FATE OF MERCURY RELEASED TO THE ATMOSPHERE FROM COMBUSTION AND INDUSTRIAL SOURCES

In this section the models and modeling scenarios used to predict the environmental transport of and exposure to mercury are briefly described. Measured mercury concentrations in the atmosphere and in soil were used along with measured mercury deposition rates as inputs to these models. The measured values were used as inputs to illustrate the predictions of the models. A limited comparison of model outputs with measured environmental concentrations was used to demonstrate the reasonableness of the modeling predictions. The human and wildlife exposures to mercury that were predicted to result from the modeled concentrations were also presented to demonstrate the impacts of the exposure assessment assumptions.

4.1 Description of Models

The extant measured mercury data alone were judged insufficient for a national assessment of mercury exposure for humans and wildlife. Thus, the decision was made to model the mercury emissions data from the stacks of combustion sources. In this study, there were three major types of modeling efforts: (1) modeling of mercury atmospheric transport on a regional basis; (2) modeling of mercury atmospheric transport on a local scale (within 50 km of source); and (3) modeling of mercury fate and transport through soils and water bodies into biota, as well as the resulting exposures to human and selected wildlife species. The models used for these aspects of this study are described in Table 4-1.

Table 4-1
Models Used to Predict Mercury Air Concentrations, Deposition Fluxes, and Environmental Concentrations

Model	Description
RELMAP	Predicts average annual atmospheric mercury concentration and wet and dry deposition flux for each 40 km^2 grid in the U.S. due to all anthropocentric sources of mercury in the U.S.
COMPDEP	Predicts average concentration and deposition fluxes within 50 km of emission source.
IEM2	Predicts environmental concentrations based on air concentrations and deposition rates to watershed and water body.

4.1.1 Estimating Impacts from Regional Anthropogenic Sources of Mercury

The impact of mercury emissions from stationary, anthropogenic U.S. sources is not entirely limited to the local area around the facility. To account for impacts of mercury emitted from many of these other non-local sources on the area around a specific source, the long-range transport of mercury from all selected sources has been modeled using the RELMAP (Regional Lagrangian Model of Air Pollution) model (the model and justification for parameter values used are described in Appendix D).

The RELMAP model was used to predict the average annual atmospheric mercury concentration and the wet and dry deposition flux for each 40 km^2 grid in the continental U.S. The emission, transport, and fate of airborne mercury over the continental U.S. was modeled using meteorological data for the year of 1989. Over 10,000 mercury emitting cells within the U.S. were addressed: the emission data were those presented in Volume II, *Inventory of Anthropogenic Mercury Emissions*.

The RELMAP model was originally developed to estimate concentrations of sulfur and sulfur compounds in the atmosphere and rainwater in the eastern U.S. The primary modification of RELMAP for this study was the handling of three species of mercury (elemental, divalent, and particulate) and carbon soot (or total carbon aerosol). Carbon soot was included as a modeled pollutant because carbon soot concentrations are important in the modeling estimates of the wet deposition of elemental mercury (Iverfeldt, 1991; Brosset and Lord, 1991; Lindqvist et al., 1991). RELMAP is more fully described in Chapter 5 and Appendix D.

4.1.2 Estimating Impacts from Local Anthropogenic Sources of Mercury

The mercury concentrations in the atmosphere and deposition rates that occur as a result of mercury emissions from a single source were predicted using a version of COMPDEP (COMPlex terrain and DEPosition air dispersion model) specifically modified to address the atmospheric transport of mercury. A detailed description of this model, including justification for parameter values as well as the modifications made for this study, are given in Appendix D. Table 4-2 describes the three primary modifications to COMPDEP.

Table 4-2
Primary Modifications Made to COMPDEP for Exposure Assessment

Modification	Rationale
User can specify vapor-particle ratio for each pollutant	The transport properties of the two phases can be quite different
User can specify stability-class dependent dry deposition velocities for vapor phase pollutants	Algorithms in COMPDEP (v. 93340) used to estimate dry deposition velocities are only applicable for particles
User can specify pollutant-dependent washout ratio that is used to estimate wet deposition of vapor phase pollutants. Scavenging coefficient is then calculated using hourly precipitation rate and mixing height.	COMPDEP (v. 93340) uses user-specified scavenging coefficients. Washout ratios were necessary for model consistency with RELMAP modeling.

COMPDEP uses hourly meteorological data to estimate hourly air concentrations and deposition fluxes within 50 km of a point source. For each hour, general plume characteristics are estimated based on the source parameters, including the gas exit velocity, temperature, stack diameter, stack height, wind speed at stack top, atmospheric stability conditions for that hour.

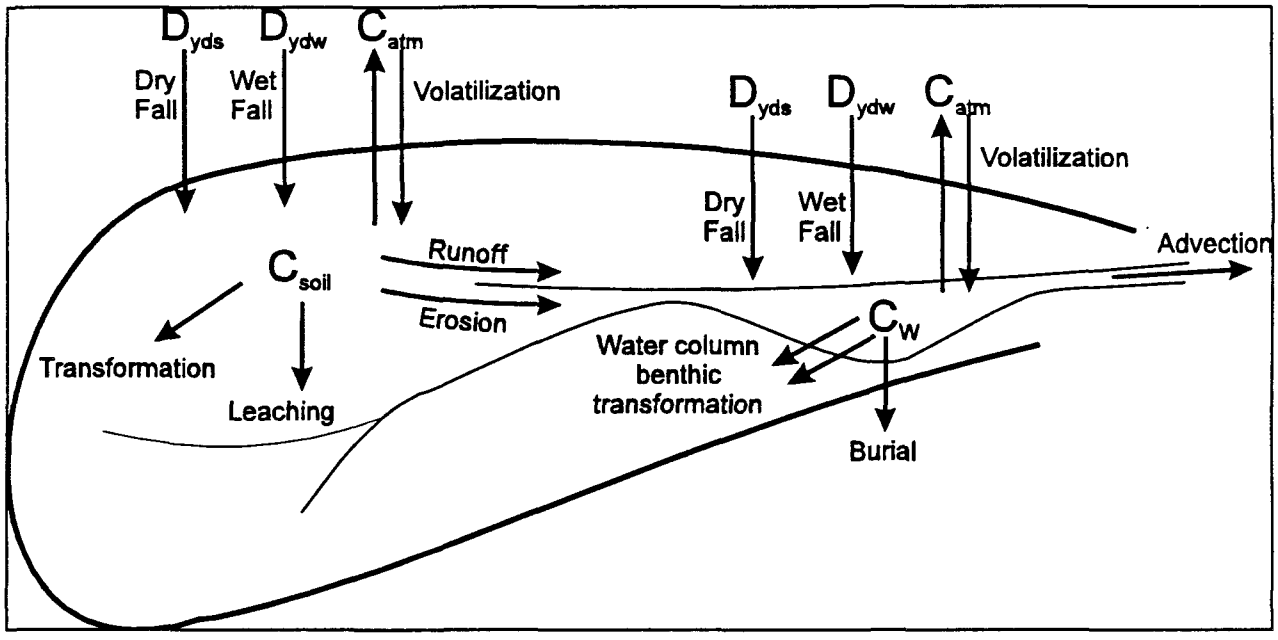
4.1.3 Estimating Environmental Concentrations

Atmospheric mercury concentrations and deposition rates estimated from RELMAP and COMPDEP were used as inputs to the calculations of mercury in watershed soils and surface waters. The soil and water concentrations, in turn, were used as inputs to the calculations of concentrations in the associated biota and fish, which humans and other animals may consume. Relevant sections of the

methodology used, called IEM2, are described below; a more complete description is given in Appendix D. This methodology is derived from the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions (U.S. EPA, 1990) as updated in an Addendum (U.S. EPA 1994, external review draft).

IEM2 uses the deposition of atmospheric chemicals to perform mass balances on a watershed soil element and a surface water element, as illustrated in Figure 4-1. The mass balances were performed for total mercury, which was assumed to speciate into three components: elemental mercury, divalent mercury, and methylmercury. The fraction of mercury in each of these components was specified for the soil and the surface water models. Total pollutant inputs and chemical properties were given for the individual mercury components, and the overall mercury transport and loss rates were calculated by the methodology.

Figure 4-1
Overview of the IEM2 Watershed Modules



Definitions for Figure 4-1

C_{soil}	chemical concentration in upper soil	mg/L
C_w	chemical concentration in water body	mg/L
C_{atm}	vapor phase chemical concentration in air	$\mu g/m^3$
D_{yds}	average dry deposition to watershed	mg/yr
D_{yws}	average wet deposition to watershed	mg/yr

IEM2 first performs a terrestrial mass balance to obtain mercury concentrations in watershed soils. Soil concentrations are used along with vapor concentrations and deposition rates to calculate concentrations in various food plants. These are used in conjunction with soil concentrations to calculate mercury concentrations in animals that consume the plants or soil. IEM2 next performs an aquatic mass balance driven using the direct atmospheric deposition in addition to runoff and erosion

loads from watershed soils. Methylmercury concentrations in fish were derived from total dissolved water concentrations using bioaccumulation factors (BAFs). Derivation of the BAFs are described in Volume V.

IEM2 was developed to handle individual chemicals, or chemicals linked by kinetic transformation reactions. The kinetic transformation rates affecting mercury components in soil, water, and sediments -- oxidation, reduction, methylation, and demethylation -- were considered too uncertain to implement in IEM2 for this study. The IEM2 methodology was expanded to handle multiple chemical components in a steady-state relationship. The fraction of each chemical component in the soil and water column was specified. The methodology predicts the total chemical concentration in the watershed soils and the water body based on pollutant inputs and dissipation rates specified for each of the components.

The model tracks the buildup of watershed soil concentrations over a period of years given a steady depositional load and long-term average hydrological behavior. Its calculations of average water body concentrations are less reliable for unsteady environments, such as streams, than for more steady environments, such as lakes.

4.1.4 Method of Estimation of MethylMercury Concentration in Freshwater Fish

To predict mercury concentrations in fish, a bioaccumulation factor (BAF) approach was used in this effort instead of the bioconcentration factor approach described in U.S. EPA (1990). Uptake of pollutants from water alone is generally expressed as a fish:water bioconcentration factor (BCF). BCF is the ratio of fish tissue concentration to steady-state water concentration. A bioaccumulation factor measures the total pollutant uptake rate from water, diet and sediments and is generally derived from field studies.

The BAF selected was that used in Volume V of this Report to Congress. It is based on a modification of the concept described in the 1993 U.S. EPA Great Lakes Water Quality Initiative, which had previously developed a BAF for mercury of 130,440, based on total measured mercury (all species) in water. An estimate of uncertainty and variability of the BAF was also developed and is described in Volume V.

The current approach utilized data collected on total dissolved mercury and the total mercury concentration in trophic level 3 fish (e.g., gizzard shad) to estimate a methylmercury BAF for trophic level 3 fish. A predator-prey factor was then applied to estimate methylmercury concentrations in trophic level 4 fish (i.e., piscivorous fish such as bass or pike). For more details concerning the structure of the aquatic food chain assumed to occur in the lakes of this assessment, please refer to Volume V of this Report and Appendix A of Volume V. The values used in the deterministic assessment for the BAF are 66,200 L/kg and 335,000 L/kg for trophic levels 3 and 4 fish, respectively.

4.2 Description of Modeling Scenarios

In this analysis the fate, transport and exposure to mercury were modeled at two different hypothetical sites: one configured to simulate a site in the eastern U.S. and one in the western U.S. Three different settings were overlayed on each site: rural, urban, and lacustrine. Each setting was associated with different exposure scenarios. Hypothetical home gardeners and subsistence farmers were assumed to occupy the rural setting. Three different hypothetical urban dwellers were assumed to occupy the urban setting. Hypothetical recreational anglers, high-end local fish consumers and piscivorous wildlife were assumed to occupy the lacustrine setting.

4.2.1 Description of Hypothetical Sites and Watersheds

Two generic sites were considered: a humid site east of 90 degrees west longitude, and a less humid site west of 90 degrees west longitude; these correspond to sites 1 and 6 in Appendix B. The primary differences between the two hypothetical locations are the assumed erosion characteristics for the watershed and the amount of dilution flow from the water body. The eastern site was defined to have steeper terrain in the watershed than the western site.

A circular lake was modeled with a diameter of 1.78 km and average depth of 5 m, with a 2 cm benthic sediment depth. As suggested in EPA (1990), a 15 to 1 ratio was assumed for the watershed area to surface water area; therefore, the watershed area was 37.3 km^2.

The type of water body modeled in this study received mercury from both direct deposition and from runoff/erosion. This kind of lake is sometimes called a drainage lake, in contrast to a seepage lake, which receives mercury primarily from direct deposition alone.

4.2.2 Description of Hypothetical Exposure Scenarios

For the analyses that were conducted for this report, the fate of deposited mercury was examined in three types of settings: rural (agricultural); lacustrine (or water body); and urban. These three settings were selected because they encompass a variety of settings and because each is expected to provide a "high-end" mercury concentration in environmental media of concern for human or wildlife species exposure; for example, elevated mercury concentrations are expected in the waters of lakes near mercury emission sources.

In general, exposure scenarios are real or hypothetical situations that define the source of contamination, the potential receptor populations, the potential pathway(s) of exposure and the variables that affect the exposure pathways. For this study, these exposure scenarios included the total amount of food derived from affected areas and the extent of mercury contamination of these food sources. For an exposure assessment which is meant to represent a broad base of potential exposures, it is not practical to model many different types of farms, gardens, etc. As for the rest of the study, a limited number of representative, generalized types of activities have been modeled.

Human exposure to environmental mercury is the result of mercury concentrations at specific human exposure points (e.g., ingested fish). For each location and setting, mercury exposure was estimated for individuals representing several specific subpopulations expected to have both typical and higher exposure levels. The individuals representing the subpopulations were defined to model average and high-end exposures in three settings: rural, urban, and lacustrine. In this section each subpopulation is discussed. A more detailed description of the values chosen for parameters of the exposure assessment is given in Appendix A. Table 4-3 summarizes the hypothetical scenarios considered as well as the exposure pathways considered in each scenario.

Table 4-3
Summary of Human Exposure Scenarios

	Location											
	Rural			Urban			Lacustrine			Remote Lakes[a]		
	Subsistence Farmer		Home Gardener	Resident	Worker/High-end		High End Fisherman		Rec. angler	High End Fisherman		Rec. angler
	Adult	Child	Adult	Adult	Adult	Pica Child	Adult	Child	Adult	Adult	Child	Adult
Air inhalation	X	X	X	X	X	X	X	X	X	X	X	X
Soil ingestion	X	X	X	X	X	X	X	X	X	X	X	
Animal ingestion	X	X										
Vegetable ingestion	X	X	X		X	X	X			X		
Local fish ingestion							X	X	X	X	X	X
Local water ingestion	X	X					X	X	X			

Notes:

[a] Lakes located greater than 50 km from a mercury emission source

Blank = Pathway not considered.

X = Pathway considered.

4.2.2.1 Summary of Exposure Parameter Values

To a large degree, there are only a few parameters that vary across these scenarios. Table 4-4 categorizes exposure parameters as invariant or variant with each scenario. A complete list of the values used and rationale for these values is given in Appendix A.

Table 4-4
Potential Dependency of Exposure Parameters

Parameters Constant Across Scenarios	Parameters that Potentially Change Across Scenarios
Body weight	Fish ingestion rates
Exposure duration	Contact fractions for vegetables, animal products, and water
Inhalation rate	Contact time for inhalation
Animal and vegetable consumption rates	Child soil ingestion rates
Adult soil ingestion rates	
Drinking water ingestion rates	

Table 4-5 shows the default values for the scenario-independent parameters for both the child and adult receptors, and Table 4-6 shows the default values for the scenario-dependent exposure parameters. The technical bases for these values are in Appendices A and B. The hypothetical scenarios are discussed in more detail in the following sections.

Consumption rates, bioconcentration factors, and biotransfer factors may be derived based on tissue (plant, animal, and dairy) weights on either a wet or dry basis. Wet weight and dry weight are related by this formula:

$$\text{Dry Weight} = \text{Wet Weight} / (1 - \text{moisture content})$$

It is critical that parameters used together are consistent based on either dry weight or wet weight. Many plants are nearly 90% water, and a mix of wet and dry weight modeling parameters can result in a ten-fold error. The fish BAF and fish consumption rates in this Report were calculated using wet weight values. Consumption rates, plant bioaccumulation factors, and animal biotransfer factors were all based upon dry weights of tissues.

Animal and plant consumption rates as well as inhalation rates are constant across exposure scenarios. The contact fraction changes generally across the exposure scenarios. The contact fraction represents the fraction of locally-grown or affected food consumed. Typically, in exposure assessments the higher the contact fraction the greater the exposure.

Table 4-5
Default Values of Scenario-Independent Exposure Parameters

Parameter	Default Value[a]	
	Adult	Child
Body weight (kg)	70	17
Exposure duration (years)	30	18
Inhalation rate (m^3/day)	20	16
Vegetable consumption rates (g DW/kg BW/day)[b]		
Leafy vegetables	0.028	0.008
Grains and cereals	1.87	3.77
Legumes	0.381	0.666
Potatoes	0.17	0.274
Root vegetables	0.024	0.036
Fruits	0.57	0.223
Fruiting vegetables	0.064	0.12
Animal Product Consumption rates (g DW/kg BW/day)		
Beef (excluding liver)	0.341	0.553
Beef liver	0.066	0.025
Dairy	0.599	2.04
Pork	0.169	0.236
Poultry	0.111	0.214
Eggs	0.073	0.093
Lamb	0.057	0.061
Soil Ingestion rates (g/day)	0.1	Scenario-dependent
Water ingestion rate (L/day)	2	1

[a] See Appendix A for details regarding these parameter values.
[b] DW= dry weight; BW = bodyweight.

Table 4-6
Values for Scenario-Dependent Exposure Parameters[a]

Parameter	Rural Subsistence Farmer		Rural Home Gardener	Urban Scenarios			High End Fisher		Recreational Angler
	Adult	Child	Adult	Adult Resident	Home Gardener	Pica Child	Adult	Child	Adult
Fish Ingestion rates (g/day)	NA[c]	NA	NA	NA	NA	NA	60	20	30
Soil Ingestion Rate (g/day)	0.1	0.2	0.1	0.1	0 1	7.5	0.1	0.2	0 1
Contact time for inhalation (hr/day)	24	24	24	16	24	24	24	24	24
Contact fractions (unitless)									
Animal products	1	1	NA	NA	NA	NA	NA	NA	NA
Leafy vegetables	1	1	0.058	NA	0.026	NA	0.058	0.058	NA
Grains and cereals	1	1	0.667	NA	0.195	NA	0.667	0.667	NA
Legumes	1	1	0.8	NA	0.5	NA	0.8	0.8	NA
Potatoes	1	1	0.225	NA	0.031	NA	0.225	0.225	NA
Fruits	1	1	0.233	NA	0.076	NA	0.233	0.233	NA
Fruiting vegetables	1	1	0.623	NA	0.317	NA	0.623	0.623	NA
Root vegetables	1	1	0.268	NA	0.073	NA	0.268	0.268	NA
Drinking water[b]	1	1	NA	NA	NA	NA	1	1	1

[a] See Appendix A for more details regarding these values.
[b] The source of the contaminated drinking water is different for the subsistence farmer and high end fisher scenarios
[c] NA - Not Considered to be Applicable to this assessment. For example, urban residents were assumed to eat no locally caught fish. Any fish ingested by this subpopulation was considered to be contaminated by mercury from outside the modeling domain and, thus, not considered.

4.2.2.2 Description of Hypothetical Rural and Urban Exposure Scenarios

Both a high-end and average rural scenario were evaluated. The high-end scenario consisted of a subsistence farmer and child who consumed elevated levels of locally grown food products. It was assumed that each farm was located on a square plot of land with an area 40,000 m^2 (approximately 10 acres). The subsistence farmer was assumed to raise livestock and to consume home-grown animal tissue and animal products, including chickens and eggs as well as beef and dairy cattle. All chicken feed was assumed to be derived from non-local sources. For bovines, 100% of the hay and corn used for feed was assumed to be from the local area. It was also assumed that the subsistence farmer collected rainwater in cisterns for drinking.

In the urban high end scenario, it was assumed that the person had a small garden similar in size to that of the average rural scenario. To address the fact that home-grown fruits and vegetables generally make up a smaller portion of the diet in urban areas, the contact fractions were based on weight ratios of home-grown to total fruits and vegetables consumed for city households. These fractions can be up to 10 times smaller than the values for rural households, depending on food plant type (see Table 4-6 and Appendix A). Exposure duration for inhalation was 24 hours per day. The high-end urban scenario included a pica child.

An average urban scenario consisted of an adult who worked outside of local area. The exposure duration for inhalation, therefore, was only 16 hours a day compared to the 24 hours a day for the rural and high-end urban scenarios. The only other pathway considered for this scenario was ingestion of average levels of soil.

4.2.2.3 Description of Hypothetical Human Exposure Scenarios for Individuals Using Water Bodies

The fish ingestion pathway was the dominant source of methylmercury intake in exposure scenarios wherein the fish ingestion pathway was considered appropriate. For this assessment, three human fish consumption scenarios were considered for the hypothetical lakes: (1) an adult high-end fish consumer scenario, in which an individual was assumed to ingest large amounts of locally-caught fish as well as home-grown garden produce (plant ingestion parameters identical to the rural home gardener scenario), consume drinking water from the affected water body and inhale the air; (2) a child of a high-end local fish consumer, assumed to ingest local fish, garden produce, and soil as well as inhale the affected air; and (3) a recreational angler scenario, in which the exposure pathways evaluated were fish ingestion, inhalation, and soil ingestion. These consumption scenarios were thought to represent identified fish-consuming subpopulations in the U.S.

Fish for human consumption from local water bodies can be derived from many sources including self-caught, gifts, and grocery and restaurant purchases. For the purposes of this study, all fish consumed were assumed to originate from the hypothetical lakes, which were considered to represent several small lakes that might be present in the type of hypothetical locations considered. No commercial distribution of locally caught fish was assumed; exposure to locally-caught fish was modeled for the three fish-consuming subpopulations described above.

Fish consumption rates for the three fish-consuming subpopulations were derived from the Columbia River Inter-Tribal Fish Commission Report (1994). Other estimates of human fish consumption rates are reported in Chapter 3 and Appendix H of this document; these estimates highlight the broad variability in consumption rates. The Columbia River Inter-Tribal Fish Commission Report (1994) estimated fish consumption rates for members of four tribes inhabiting the

Columbia River Basin. The estimated fish consumption rates were based on interviews with 513 adult tribe members who lived on or near the reservation. The participants had been selected from patient registrations lists provided by the Indian Health Service. Adults interviewed provided information on fish consumption for themselves and for 204 children under 5 years of age.

Fish consumption rates for tribal members are shown in Tables 4-7 and 4-8. The values used in this study are shown in Table 4-9. The values listed below reflect an annual average, but monthly variations were also reported. For example, the average daily consumption rate during the two highest intake months was 107.8 grams/day, and the daily consumption rate during the two lowest consumption months was 30.7 grams/day. Fish were consumed by over 90% of the surveyed population with only 9% of the respondents reporting no fish consumption. The maximum daily consumption rate for fish reported by one member of this group was 972 grams/day. Since most of the population consisted of fish consumers ("users" in Chapter 3 and Appendix H), utilization of per capita estimates was considered appropriate.

Table 4-7
Fish Consumption Rates for Columbia River Tribes

Fish Consumption by Columbia River Tribes[a]	
Subpopulation	**Mean Daily Fish Consumption (g/day)**
Total Adult Population, aged 18 years and older	59
Children, aged 5 years and younger	20
Adult Females	56
Adult Males	63

[a] Columbia River Inter-Tribal Commission, 1994.

Table 4-8
Fish Consumption Rates for Columbia River Tribes: Adults

Daily Fish Consumption Rates Among Adults Fish Consumption by Columbia River Tribes[a]	
Percentile	**grams/day**
50th	29-32
90th	97-130
95th	170
99th	389

[a] Columbia River Inter-Tribal Commission, 1994.

Table 4-9
Fish Consumption Rates used in this Study

Subpopulation	Fish Consumption Rate (g/day)[a]
High-end Adult	60
High-end Child	20
Recreational Angler	30

[a] Columbia River Inter-Tribal Commission, 1994.

4.2.2.4 Description of Hypothetical Exposure Scenarios for Piscivorous Birds and Mammals using Water Bodies

Piscivorous birds and mammals were assumed to inhabit areas adjacent to the hypothetical lakes considered. As modeled, the piscivorous receptors were exposed to mercury through consumption of lake fish. The five wildlife species assumed to inhabit the hypothetical water body were selected because they were considered likely to be exposed to methylmercury through fish consumption. They were not selected because they were more sensitive to methylmercury exposure than other wildlife. Fish-consuming species were the only groups considered in this assessment because mercury bioaccumulation is associated primarily with aquatic ecosystems. All five wildlife species were assumed to consume fish from trophic levels 3 and 4 and to inhabit the aquatic environment modeled for a lifetime. Mercury concentrations in food sources other than fish and migratory behaviors were not considered. Table 4-10 lists the species considered, the assumed animal body weights, fish consumption rates and the trophic level of the fish consumed (U.S. EPA, 1995 and Volume V of this Report). Dermal, inhalation, and drinking water exposures for ecological receptors were not modeled in this assessment, and mercury concentrations in other potential food sources such as amphibians, reptiles, and insects were not considered.

Table 4-10
Fish Consumption Rates for Piscivorous Birds and Mammals (from U.S. EPA, 1993)

Animal	Body Weight (kg)	Total Ingestion Rate (g/day)	% of Diet from Trophic Level 3 Fish	% of Diet from Trophic Level 4 Fish	% of Diet from Non-aquatic sources
Bald Eagle	4.6	500	74	18	8
Osprey	1.5	300	100	0	0
Kingfisher	0.15	75	100	0	0
River Otter	7.4	1220	80	20	0
Mink	0.8	178	90	0	10

4.3 Indirect Exposure Modeling Results Using Measured Mercury Air Concentrations and Deposition Rates

The purpose of this section was to illustrate the modeling results using measured mercury air and soil concentrations and atmospheric deposition rates. A limited comparison of model outputs with measured concentrations was included to demonstrate the reasonableness of the modeling predictions. The human and wildlife exposures to mercury that were predicted to result from the modeled concentrations are also presented. This demonstrated the impacts of the exposure assessment assumptions used for the hypothetical populations inhabiting the watershed and water body. It also provided a forum to discuss the more general features of the exposure assumptions.

A total atmospheric mercury concentration of 1.6 ng/m^3 was used as an input for the modeling exercise. A total mercury deposition rate of 10 μg/m^2/yr was assumed to deposit onto the watershed and water body; 10 μg/m^2/yr is the sum of 3.5 μg/m^2/yr, a typical particulate dry deposition rate, and 6.5 μg/m^2/yr, a typical wet deposition rate. Note that there was no input to the models from dry deposition of vapor-phase mercury. The watershed soils were assumed to have mercury concentrations of 50 ng/g. Table 4-11 lists the input values.

These modeling inputs for the air and soil concentrations as well as the deposition rates are supported by a substantial body of recent, peer-reviewed reports of mercury measurements at sites in the U.S. Brief descriptions of some reported atmospheric and soil mercury concentration measurements were presented in Chapter 2 of this Volume. The input values selected were from the low-end of the reported ranges. For example, most of the collected deposition data were from sites located some distance from large emission sources. The values used in this modeling exercise were thought to be representative of remote sites that are influenced by distant emission sources and the natural background concentrations.

Table 4-11
Values Assumed for Demonstration Application of Water body and Exposure Models

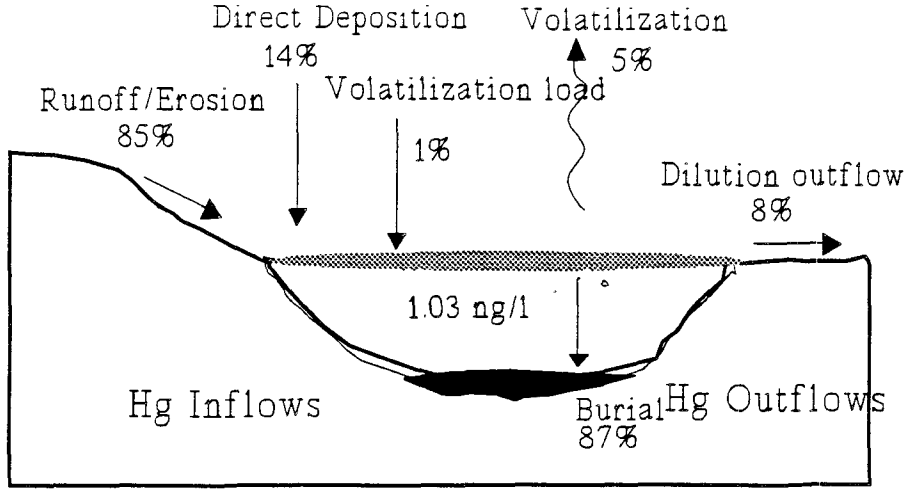
Parameter	Value
Total Mercury Air Concentration (ng/m^3)	1.6
Total Mercury deposition rate to watershed and water body (ug/m^2/yr)	10
Total Mercury Soil Concentration in watershed soil (ng/g)	50

4.3.1 Mercury Concentrations Predicted for Water Bodies

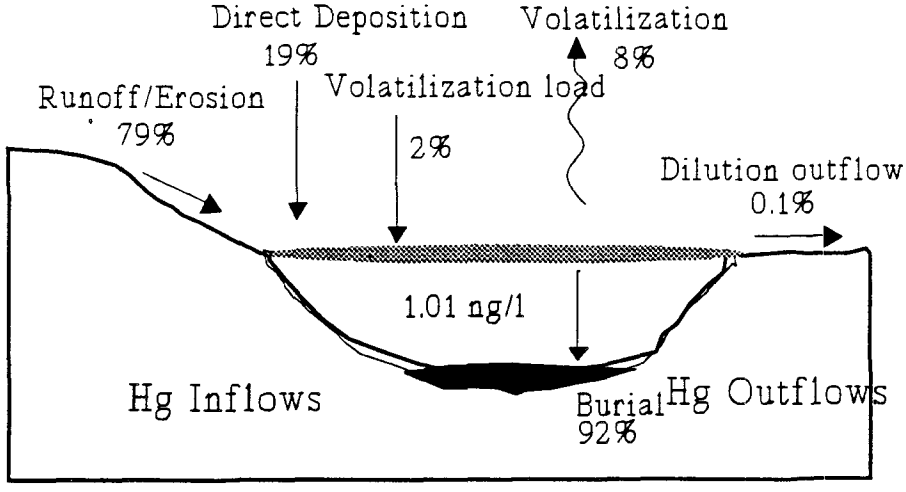
Using an assumed deposition rate of 10 ug/m^2/yr, the total load to the water body from direct deposition was 22 g/yr for both the western and eastern sites. Other loads to the water body include erosion and runoff from the watershed. These were calculated using the assumed soil concentration of 50 ng/g, and load due to gaseous diffusion, which is calculated based on the assumed air concentration of 1.6 ng/m^3. The total load to the water body for the eastern and western sites is 185 g/yr and 128 g/yr, respectively.

Figure 4-2 shows the predicted steady-state breakdowns of the total mercury influxes and outfluxes for the water body for the two hypothetical watersheds and water bodies considered. At

Figure 4-2
Predicted Steady-state Watershed Dynamics for Eastern and Western Sites Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g



Steady-state Watershed/Waterbody Dynamics for the Eastern Site (total mercury inflows of 185 g/yr).



Steady-state Watershed/Waterbody Dynamics for the Western Site (total mercury inflows of 128g/yr).

steady-state, it is assumed that the influxes of mercury are equal to the outfluxes. There are three types of loss processes considered: sedimentation/burial, in which the mercury is lost to the water body bed; volatilization from the water body; and advective loss of mercury from the water column due to dilution flow through the water body.

There is a large difference between the assumed dilution flow for the two water bodies. The dilution flow m^3/yr was calculated using the values given in U.S. EPA (1985) in conjunction with the catchment area (see Appendix B). The value calculated for the dilution flow in the eastern site was $1.4x10^7$ m^3/yr, while the value calculated for the western site was $1.4x10^5$ m^3/yr.

Table 4-12 shows the erosion parameters assumed, as well as the calculated erosion characteristics of each water body. Details regarding the selection of these parameters are provided in Appendix B. The steeper terrain of the eastern site as embodied in the higher erosivity and topographic factors is balanced by the lower cover fraction indicative of the type of forest watershed assumed. The western site, while flatter, is not assumed to have the same type of erosion-reducing cover as the eastern setting.

Table 4-12
Comparison of Watershed Erosion Characteristics for the Hypothetical Watersheds

Parameter	Eastern Site	Western Site
Erosivity factor (/yr)	200	53
Erodibility factor (tons/acre)	0.3	0.28
Topographic factor (unitless)	2.5	0.4
Cover management factor (unitless)	0.006	0.1
Predicted soil loss from watershed (tons/acre/yr)	0.9	0.6
Sediment delivery ratio	0.2	0.2
Amount of eroded soil reaching water body from watershed per unit area; tons/acre/yr ($kg/km^2/yr$)	0.18 (40815)	0.12 (27210)
Pollutant Enrichment factor (unitless)	2	2
Total mass of soil reaching water body per year (kg/yr)[a]	$1.50x10^6$	$9.88x10^5$
Flux of mercury to water body from erosion (g/yr)[b]	152	100

[a] Based on watershed area of 37.3 km^2.
[b] Based on soil concentration of 50 ng/g in watershed soil.

Based on the assumed average soil concentration of 50 ng/g in the watershed soil and the total watershed area, the amount of transport of mercury to the water body from the watershed was approximately four to six times that from direct deposition alone for the two water bodies. This type of predicted behavior was in contrast to "seepage" lakes, which receive most of their total load of mercury from the atmosphere alone. Despite the differences in parameterizations, the predicted total mercury water concentrations were similar at both sites: 1.03 ng/l at the eastern site and 1.01 ng/l at the western site.

Sedimentation plays a large role as a mercury loss process for the lakes at both sites. This was predicted due in part to the large benthic sediment partition coefficients used, which were based on calibrations as described in Appendix C.

Table 4-13
Predicted Surface Water and Benthic Sediment Concentrations for the Hypothetical Water Bodies Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

	Eastern Setting	Western Setting
Total Mercury Water Concentration (ng/l)	1.03	1.01
Percent of Mercury dissolved	71	77
Predicted suspended sediment concentration (mg/l)	3.20	2.17
Total Mercury Benthic Sediment Concentration (ng/g)	111	119

Volatilization was predicted to be of similar importance at both sites, although the exact percentages may vary depending on the air concentrations and deposition rates.

4.3.2 Mercury Concentrations Predicted in Biota by IEM2 Modeling Using Measured Concentrations

4.3.2.1 Concentrations in Fish

In this section the predicted water concentration for the two sites was used to estimate the methylmercury concentrations in trophic level 3 and 4 fish. This was done by multiplying the total dissolved mercury concentration in the water body by the trophic level 3 or 4 fish bioaccumulation factors.

Although the total mercury concentration in water was slightly higher for the eastern setting than for the western setting, the dissolved concentration was higher for the western setting. The fraction dissolved depended on the suspended sediment partition coefficients for the mercury species and the total suspended sediment in the water body. Because more sediment was predicted to be transported into the eastern water body, more of the mercury was in the suspended sediment, resulting in a lower fraction of dissolved mercury.

The methylmercury concentration in the contaminated fish was determined by multiplying the total dissolved mercury concentration in water by a BAF (derivation is described in Volume V). The

higher the dissolved mercury concentrations in the local waters, the proportionally greater the mercury concentration in the fish will be. The concentrations of methylmercury in fish were also influenced by fish diet. In the four-tier trophic food chain model used in this Report, fish were assumed to feed at two levels. Trophic level 3 fish were assumed to feed on plankton which are predicted to be contaminated with comparatively low levels of methylmercury. Trophic level 4 fish were assumed to feed on trophic level 3 fish, which have higher methylmercury concentrations than the plankton. The BAF of 66,200 L/Kg for trophic level 3 fish was estimated using several sets of data on measured mercury concentrations in fish and water. The BAF for trophic level 4 of 335,000 L/Kg) was estimated by applying a predator-prey factor (of approximately 5) to the bioaccumulation factor estimated for trophic level 3 fish.

With this approach, it takes very little dissolved mercury in a water body to result in elevated predicted methylmercury concentrations in fish. This reflects the observed data, as the BAF was derived from field studies. By using the median of the distribution for the BAF for the trophic level 4 fish of 335,000 L/kg, a methylmercury fish concentration of at least 1 ug/g (ug/g) was predicted whenever the dissolved mercury water concentration exceeded 3 ng/l. Interpreted probabilistically, this implies that approximately half of the water bodies with a dissolved water concentration above 3 ng/l would be expected to have fish concentrations exceeding 1 ug/g.

The values predicted using the predicted water concentrations for the two sites are shown in Table 4-14.

Table 4-14
Predicted Methylmercury Concentrations in Fish (ug/g) for the Hypothetical Water Bodies Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

	Predicted Fish Concentration (ug/g)	
	Eastern Site	Western Site
Trophic Level 3 Fish	0.048	0.051
Trophic Level 4 Fish	0.243	0.261

The next table shows the percentiles of the methylmercury concentration in fish based on a dissolved water concentration of 0.7 ng/l. This shows the predicted wide variability in fish methylymercury concentrations that may occur as the result of a relatively low water concentration.

Table 4-15
Percentiles of Predicted Methylmercury Concentrations in Fish (ug/g) based on a Total Mercury Dissolved Water Concentration of 0.7 ng/L

		Percentile of Distribution				
	Geometric Mean	5th	25th	50th	75th	95th
Trophic 3 BAF (L/kg)	6.62e+04	6.40e+03	2.54e+04	6.62e+04	1.72e+05	6.84e+05
Predicted Fish Concentration (ug/g)	0.046	0.004	0.018	0.046	0.121	0.479
Trophic 4 BAF (L/kg)	3.35e+05	2.27e+04	1.11e+05	3.36e+05	1.00e+06	4.70e+06
Predicted Fish Concentration (ug/g)	0.235	0.016	0.078	0.235	0.700	3.290

These results show how there is a likelihood that the fish concentration exceeds 1 ug/g using the dissolved water concentrations of only 0.7 ng/l.

4.3.2.2 Concentrations in Other Biota

In this section the predicted soil and water concentrations are used to estimate environmental concentrations. Because the air concentration, deposition rate, and soil concentration are the same for both sites, the biota concentrations are identical; only a single set of results are presented.

Green Plant Concentrations. Table 4-16 shows the predicted plant concentrations based on the assumed air concentration of 1.6 ng/m^3, deposition rate of 10 ug/m^2/yr, and soil concentration of 50 ng/g. Three routes by which plants can take up mercury are addressed here: root uptake, whereby the plant is assumed to take up mercury from the soil; direct deposition, whereby the mercury deposited on the plant shoot from atmospheric deposition transfers to the plant; and air-to-plant transfer, whereby the mercury in the air is transported through the stomata into the plant.

The mercury in potatoes and root vegetables results solely from root uptake since no air-to-plant uptake was assumed to occur for these plants (Appendix A). For leafy vegetables, all the mercury was predicted to be from air uptake since no root uptake was assumed to occur. For grains, legumes, fruits and fruiting vegetables the bulk of mercury was also modeled to result from air uptake of elemental mercury and transformation to other species; note, however, that the air and soil biotransfer factors were calculated based on a conservtive premise that each could acount for all the mercury measured in a green plant. This was done because the soil concentrations used for this demonstration are several times lower than the soil concentrations from the Cappon (1981 and 1987) studies from which the soil BCFs were derived. For more details pertaining to the plant-soil BCF please see Appendix A of this volume.

Hanson et al. (1994) stated that "dry foliar surfaces in terrestrial forest landscapes may not be a net sink for atmospheric elemental mercury, but rather as a dynamic exchange surface that can function as a source or sink dependent on current mercury vapor concentrations, leaf temperatures, surface condition (wet versus dry) and level of atmospheric oxidants." Similarly, Mosbaek et al.

Table 4-16
Predicted Dry Weight Plant Concentrations Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

Plant	Total Mercury Concentration (ug/g dry weight)	% MHg[b]	Contributions to Concentration (%)				Contributions to Concentration (%)			
			Hg^{2+a} Concentration (ug/g dry weight)	Root Uptake	Direct Deposition	Air-to-plant transfer	MHg[b] Concentration (ug/g dry weight)	Root Uptake	Direct Deposition	Air-to-plant transfer
Grain	2.02E-03	7.6%	1.87E-03	24.4%	0.0%	75.6%	1.54E-04	12.4%	0.0%	87.6%
Legumes	2.33E-03	7.1%	2.16E-03	34.1%	0.5%	65.4%	1.66E-04	18.7%	0.0%	81.3%
Potatoes	5.10E-03	3.9%	4.90E-03	100.0%	0.0%	0.0%	2.00E-04	100.0%	0.0%	0.0%
Root Vegetables	1.86E-03	5.3%	1.77E-03	100.0%	0.0%	0.0%	9.91E-05	100.0%	0.0%	0.0%
Fruits	3.22E-02	5.1%	3.05E-02	2.9%	0.2%	96.9%	1.64E-03	1.5%	0.0%	98.5%
Fruiting Vegetables	3.22E-02	5.1%	3.05E-02	2.9%	0.2%	96.9%	1.64E-03	1.5%	0.0%	98.5%
Leafy Vegetables	3.10E-02	21.7%	2.43E-02	0.0%	0.5%	99.5%	6.72E-03	0.0%	0.0%	100.0%
Forage	3.11E-02	21.6%	2.44E-02	0.0%	0.9%	99.1%	6.72E-03	0.0%	0.0%	100.0%
Silage	3.10E-02	21.7%	2.43E-02	0.0%	0.3%	99.7%	6.72E-03	0.0%	0.0%	100.0%

[a] Hg^{2+} = divalent mercury
[b] MHg = methylmercury

(1988) showed that most of the mercury in leafy plants is attributable to air-leaf transfer, but that for a given period of time the amount of elemental mercury released from the plant-soil system greatly exceeds the amount collected from the air by the plants. It is also likely that many plants accumulate airborne mercury to certain concentrations, after which net deposition of elemental mercury does not occur. This is also a function of the large area of uncertainty in deriving soil-to-plant and air-to-plant BCFs for mercury due to the wide variation in values among different studies. This is described in appendix A, sections A.2.2, A.2.2.1 and A.2.2.2. In fact, a single air-to-plant BCF of 23,000, to be split into divalent mercury and methylmercury air-to-plant BCFs according to the mercury speciation in plants determined by Cappon (1981,1987), was chosen as the default value *for all plant types* assumed to be able to accumulate mercury from the atmosphere. For grain and legumes the air-to-plant transfer factor of 23,000 was reduced by a factor of 20. This accounted for the observation that mercury concentrations in the portions of the plants typically consumed by humans are 20 times lower than the mercury concentrations in the plant tissues for which the BCF was derived (John, 1972; Cappon, 1981; and Somu et al., 1985). See Appendix A. Although similar phenomena may occur in fruits and fruiting vegetables, no data were available to modify the derived air-to-plant transfer factor.

In general, using the air concentration, wet deposition and soil concentration discussed above, the plant uptake of mercury is predicted to be dominated by either root uptake or air-to-plant transfer. For areas in which the deposition rate is significantly higher, direct deposition may be a more important pathway. Similarly, the root uptake pathway may be more important in areas with higher soil concentrations.

Mercury Concentrations in Animal Products. The concentrations in animal products were calculated by multiplying the total daily intake of a particular species of mercury by a transfer factor that can depend on the animal species and tissue. The animals considered may be exposed to mercury via four possible pathways: ingestion of grain, forage, silage, or soil. The contribution from these pathways depends on both the predicted concentration in the plant or soil and the ingestion rate for a particular pathway.

Table 4-17 shows the predicted animal concentrations of divalent and methylmercury.

For beef and dairy products, most of the intake of mercury is from forage and silage because these plants are assumed to make up over 80% of their total diet (see Appendix A). The predicted concentration for beef liver is slightly higher than that for beef due to a higher transfer factor for beef liver. For pork and poultry products, more grain is assumed to be ingested than forage or silage, resulting in most of the exposure to mercury through consumption of grain.

4.3.3 Results for Hypothetical Exposure Scenarios

Based on the predicted concentrations in biota and using the hypothetical exposure scenarios described in the previous sections, the predicted human intake rates for each scenario are shown in Table 4-18. Tables 4-19 and 4-20 show each pathway's contribution to the total exposure rate.

In general, exposure to mercury is dominated by indirect exposure for any scenario that includes an ingestion pathway other than soil. Furthermore, exposure tends to be dominated by either divalent or methylmercury species rather than elemental mercury. For the agricultural scenarios, divalent mercury is the dominant exposure. For the scenarios that include fish ingestion, methylmercury dominates exposure.

Table 4-17
Predicted Dry Weight Livestock Mercury Concentrations Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

Animal Product	Total Hg Concentration (ug/g dry weight)	% MHg	Hg^{2+b} Concentration (ug/g dry weight)	Contributions to Concentration (%)				MHg^b Concentration (ug/g dry weight)	Contributions to Concentration (%)			
				Grain	Forage	Silage	Soil		Grain	Forage	Silage	Soil
Beef	7.47E-03	20.5%	5.94E-03	0.6%	72.4%	20.4%	6.5%	1.53E-03	0.2%	77.3%	22.0%	0.5%
Beef Liver	1.87E-02	20.5%	1.48E-02	0.6%	72.4%	20.4%	6.5%	3.83E-03	0.2%	77.3%	22.0%	0.5%
Dairy	9.42E-03	20.6%	7.48E-03	1.3%	71.8%	21.4%	5.4%	1.94E-03	0.4%	76.3%	22.9%	0.4%
Pork	6.25E-06	19.2%	5.05E-06	14.4%	0.0%	81.2%	4.3%	1.20E-06	5.0%	0.0%	94.6%	0.4%
Poultry/eggs	6.80E-05	3.5%	6.56E-05	25.0%	0.0%	0.0%	75.0%	2.36E-06	57.4%	0.0%	0.0%	42.6%
Lamb	3.31E-03	20.2%	2.64E-03	0.0%	91.5%	0.0%	8.5%	6.70E-04	0.0%	99.3%	0.0%	0.7%

[a] Hg^{2+} = divalent mercury
[b] MHg = methylmercury

Table 4-18

Predicted Human Exposure Rates for Hypothetical Receptors Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

	Total Mercury Intake (mg/kg/day)	Inhalation (mg/kg BW/day)	Ingestion (mg/kg BW/day)		
			Total	% Hg^{++a}	% MHg[b]
Rural Subsistence Farmer					
Child	4.86E-05	1.51E-06	4.71E-05	87%	13%
Adult	3.73E-05	4.57E-07	3.68E-05	90%	10%
Rural Home Gardener					
Child	1.28E-05	1.51E-06	1.13E-05	94%	6%
Adult	9.57E-06	4.57E-07	9.11E-06	94%	6%
Urban Average					
Child	2.10E-06	1.51E-06	5.97E-07	98%	2%
Adult	3.77E-07	3.05E-07	7.24E-08	98%	2%
Urban High End					
Child	2.39E-05	1.51E-06	2.24E-05	98%	2%
Adult	3.81E-06	4.57E-07	3.35E-06	94%	6%
Subsistence Fisher					
Child	2.99E-04	1.51E-06	2.97E-04	4%	96%
Adult	2.18E-04	4.57E-07	2.17E-04	4%	96%
Recreational Angler					
Adult	1.05E-04	4.57E-07	1.04E-04	0%	100%

[a] Hg^{++} = divalent mercury
[b] MHg=methylmercury

Table 4-19
Divalent Mercury Exposure for Hypothetical Receptors
Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

	Divalent Mercury Ingestion (mg/kg/day)																			
	Total (mg/kg/day)	Fish	Total Plant	Grain	Legumes	Potatoes	Root Vegetables	Fruits	Fruiting Vegetables	Leafy Vegetables	Total Animal	Beef	Beef Liver	Dairy	Pork	Poultry	Eggs	Lamb	Soil	Water
Rural Subsistence Farmer																				
Child	4.08E-05	0.0%	50.4%	17.3%	3.5%	3.3%	0.2%	16.7%	9.0%	0.5%	46.8%	8.0%	0.9%	37.4%	0.0%	0.0%	0.0%	0.4%	1.4%	1.4%
Adult	3.32E-05	0.0%	75.9%	10.5%	2.5%	2.5%	0.1%	52.4%	5.9%	2.1%	23.0%	6.1%	2.9%	13.5%	0.0%	0.0%	0.0%	0.5%	0.2%	0.8%
Rural Home Gardener																				
Child	1.06E-05	0.0%	94.5%	44.2%	10.8%	2.8%	0.2%	14.9%	21.5%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	5.5%	0.0%
Adult	8.57E-06	0.0%	99.2%	27.2%	7.7%	2.2%	0.1%	47.3%	14.2%	0.5%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.8%	0.0%
Urban Average																				
Child	5.85E-07	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%
Adult	7.10E-08	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%
Urban High End																				
Child	2.19E-05	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%
Adult	3.15E-06	0.0%	97.7%	21.6%	13.0%	0.8%	0.1%	42.0%	19.7%	0.6%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	2.3%	0.0%
Subsistence Fisherperson																				
Child	1.07E-05	0.0%	94.1%	44.0%	10.8%	2.8%	0.2%	14.9%	21.4%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	5.5%	0.5%
Adult	8.59E-06	0.0%	98.9%	27.1%	7.7%	2.2%	0.1%	47.2%	14.2%	0.5%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.8%	0.3%
Recreational Angler																				
Adult	9.54E-08	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	74.4%	25.6%

Table 4-20
Methylmercury Exposure for Hypothetical Receptors
Using Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

		Methylmercury Ingestion (mg/kg/day)																			
		Total (mg/kg/day)	Fish	Total Plant	Grain	Legumes	Potatoes	Root Vegetables	Fruits	Fruiting Vegetables	Leafy Vegetables	Total Animal	Beef	Beef Liver	Dairy	Pork	Poultry	Eggs	Lamb	Soil	Water
Rural Subsistence Farmer																					
	Child	6.32E-06	0.0%	21.6%	9.2%	1.7%	0.9%	0.1%	5.8%	3.1%	0.9%	78.2%	13.4%	1.5%	62.6%	0.0%	0.0%	0.0%	0.6%	0.2%	0.1%
	Adult	3.59E-06	0.0%	44.9%	8.0%	1.8%	0.9%	0.1%	26.0%	2.9%	5.3%	55.0%	14.5%	7.0%	32.3%	0.0%	0.0%	0.0%	1.1%	0.0%	0.1%
Rural Home Gardener																					
	Child	7.10E-07	0.0%	98.3%	54.4%	12.4%	1.7%	0.1%	12.0%	17.2%	0.4%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	1.7%	0.0%
	Adult	5.45E-07	0.0%	99.7%	35.1%	9.3%	1.4%	0.1%	39.9%	12.0%	2.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.3%	0.0%
Urban Average																					
	Child	1.19E-08	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%
	Adult	1.45E-09	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%
Urban High End																					
	Child	4.47E-07	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	100.0%	0.0%
	Adult	1.99E-07	0.0%	99.3%	28.1%	15.8%	0.5%	0.1%	35.6%	16.7%	2.5%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.7%	0.0%
Subsistence Fisherperson																					
	Child	2.86E-04	99.7%	0.2%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
	Adult	2.09E-04	99.7%	0.3%	0.1%	0.0%	0.0%	0.0%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
Recreational Angler																					
	Adult	1.04E-04	100.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%

Rural Scenarios

For the rural scenarios considered, exposure to divalent mercury accounted for approximately 90% of the total mercury exposure. The primary exposure pathway is from plant products which account for 70-90% of the total mercury exposure. Most of the exposure through plant products is from consumption of fruits. Most of the mercury predicted to occur in fruits is the results of air-to-plant transfer. There is a great deal of uncertainty regarding the magnitude of air to plant transfer of mercury (see Appendix A).

The rural subsistence farmer receptors are predicted to have about twice as much exposure to mercury as the rural home gardener. Thus, although there are many differences between the two rural scenarios as parameterized here, the end results are not substantially different because exposure is dominated by plant pathways, and the differences between scenarios are not as significant for these pathways.

Exposure to mercury from milk (dairy) dominates exposure from animal products for the high end rural scenario considered (total of seven types of animal products are assumed to be consumed).

Urban Scenarios

The results for the urban scenarios are independent of the water body results. For the urban average scenario, the only exposure pathways considered are inhalation and ingestion of soil. For the urban high end scenario, the ingestion of home grown produce is considered as well, although with lower contact fractions than for the rural home gardener scenario.

For the urban average scenarios, exposure to mercury from the inhalation route dominated exposure. The urban high-end scenario included a small garden to the urban average scenario, with the result that similar contributions to the total divalent mercury and methylmercury exposures occurred as for the rural home gardeners. The urban high-end adult receptor had a predicted mercury exposure of about one-third that of the rural home gardener. The high end urban child scenario consisted of a pica child assumed to ingest 7.5 grams of soil per day. The exposure rate is then proportional to the assumed soil concentration, which in this case is 50 ng/g.

Fish Ingestion Scenarios

It was assumed that the high-end fish consumer eats fish from the affected freshwater lake on a daily basis; that is, seasonal consumption rate variation was not addressed. This individual is the most exposed person to methylmercury in this assessment, and was predicted to be exposed to approximately twice the level of methylmercury to which the recreational angler is exposed to. On a gram per bodyweight basis, the high-end fish-consuming child was the maximally exposed subpopulation. This is based on the consumption rate and the bodyweight, and is consistent with the data presented in Chapter 3 and Appendix H.

For the fish ingestion scenarios, intake of mercury was mainly the methylmercury species. Although intake of methylmercury via plants and soil is considered in the subsistence fisher scenario, it accounts for less than 0.5% of the total methylmercury intake. For the high end scenario, however, in which home-grown vegetables are consumed, divalent mercury primarily from green plant consumption was predicted to account for about 5% of the total mercury intake. The recreational angler was assumed to be exposed to mercury via fish, soil and water consumption. Exposure via soil and water however, accounted for less than 0.1% of the total mercury intake.

4.3.3.2 Predicted Methylmercury Intakes for Wildlife Receptors

The exposure of three birds and two mammals to methylmercury through ingestion of methylmercury-contaminated fish was estimated, and the results are shown in Table 4-21. The exposure of these wildlife receptors to mercury may occur through other routes as well; e.g., the ingestion of mercury-contaminated drinking water and food sources other than fish, inhalation of atmospheric mercury, and dermal uptake through soil and water. Fish consumption is thought to be the dominant mercury exposure pathway for piscivorous accounting for most exposure to mercury, because methylmercury bioaccumulates to such a great extent in their food source, fish. Consequently, an analysis of these wildlife receptors' methylmercury contact rate based only on the daily ingestion rate of fish is logical and appropriate. The piscivorous bird's or mammal's estimated methylmercury contact rate from fish consumption were based on two important factors: the methylmercury concentration in the fish and the daily amount of fish eaten.

The ratio of grams fish consumed per day to piscivore body weight is important in estimating methylmercury exposure on a g/Kg Bw/day basis. The greater this ratio the higher the resulting mercury exposure, assuming mercury concentrations in fish consumed are constant.

By using the relationship for mercury described by the four-tier trophic food chain model (i.e., the BAFs specific for fish in trophic levels 3 and 4), the estimates of the animal's daily fish consumption rates from each trophic level and the body weight of the animal, the rates of methylmercury exposure (in mg/Kg BW/day) for the animals in this hypothetical environment can be ranked. This ranking is independent of the actual fish methylmercury concentrations as long as the predator-prey factor is the same. The piscivore exposure ranking from highest to lowest is the following:

Kingfisher > River Otter > Osprey,Mink > Bald Eagle

The ranking demonstrates the importance of three factors: the trophic level which the piscivore consumes, the daily consumption rate and the ratio of daily fish consumption rate to body weight. Despite consuming a comparatively small amount of the trophic level 3 fish, the kingfisher ranks first as having the highest mercury exposure per individual. This is attributable to the high food consumption per bodyweight for this species.

4.4 Summary of IEM2 Model Results

In this chapter the watershed, water body, food chain and exposure models of IEM2 were run using a mercury air concentration of 1.6 ng/m^3, a mercury deposition rate of 10 ug/m^2/yr, and watershed mercury soil concentration of 50 ng/g. These values had been measured in the environment.

Environmental Media Modeling

The predicted surface water and benthic sediment mercury concentration at both the eastern and western site were within the range of measured mercury concentrations for these media. At both sites, most (about 80%) of the input of mercury to the water body was predicted to be from erosion and runoff rather than direct deposition. Similarly, burial of the mercury in the water body bed was predicted to be the dominant loss process of mercury, accounting for more than 85% of the mercury outflow at both sites. Volatilization of mercury from the water body was predicted to be slightly larger at the western site due to the differences in climate.

Table 4-21
Predicted Methylmercury Intake for Wildlife Receptors sing
Mercury Air Concentration of 1.6 ng/m^3, Deposition Rate of 10 ug/m^2/yr, and Soil Concentration of 50 ng/g

				East	West			Total MHg[a] Intake (mg/kg/day)	
	Trophic Level 3 Fish Concentration (ug/g)			0.048	0.051				
	Trophic Level 4 Fish Concentration (ug/g)			0.243	0.261				
Animal	Bodyweight kg	Total Ingestion Rate kg/day	Non-aquatic ingestion rate g/day	Trophic 3 Fish Ingestion rate		Trophic 4 Fish Ingestion rate		Eastern Setting	Western Setting
				g/day	g/kgBW/day	g/day	g/kgBW/day		
Bald Eagle	4.6	0.5	50	370	80.43	90	19.57	8.62e-03	9.21e-03
Osprey	1.5	0.3	0	300	200.00	0	0.00	9.60e-03	1.02e-02
Kingfisher	0.15	0.075	0	75	500.00	0	0.00	2.40e-02	2.55e-02
River Otter	7.4	1.22		976	131.89	244	32.97	1.43e-02	1.53e-02
Mink	0.8	0.178	18	160	200.00	0	0.00	9.60e-03	1.02e-02

[a] MHg=methylmercury

Biota Modeling

The predicted concentrations and speciations of mercury in green plants were within the range of measured concentrations. The mercury concentrations in green plants were the result of direct deposition onto exposed foliar surfaces, air-to-plant transfer and soil-to-plant transfer. As modeled in this assessment, most of the mercury predicted to occur in green plants was the result of air-to-plant transfer for those types of green plants for which this route of transfer was deemed appropriate. The plants for which air-to-plant transfer was not considered were potatoes and root vegetables. The contribution of air-to-plant transfer in the overall plant burden of mercury is uncertain. Of the types of green plants considered, this uncertainty is largest for fruits and fruiting vegetables. The predicted concentrations for these plant types were at or slightly above the upper end of the range of observed values.

The predicted mercury concentrations in all animal products except fish were low. This was the result of generally low concentrations in plants and small plant-to-animal and soil-to-animal biotransfer factors. The dominant exposure pathway for animals was predicted to occur through ingestion of plant products.

Mercury concentrations in fish were predicted to be the highest of the biota considered. Mercury concentrations in fish were the product of the bioaccumulation factor and the dissolved concentration of mercury in surface water. The predicted mercury concentrations in fish were within the range of reported values. The predicted concentrations of mercury in fish at the two sites were consistent with the mean of reported values in Bahnick et al. (1992). There was a great deal of uncertainty and variability associated with uptake of mercury by fish.

Human and Wildlife Exposure Modeling

Human exposure to mercury was predicted to be dominated by indirect routes of exposure except for the hypothetical average urban dweller. This individual was assumed to be exposed to mercury from inhalation and soil ingestion only. Furthermore, exposure tends to be dominated by either divalent or methylmercury species. For the agricultural scenario, divalent mercury dominates mercury exposure, and for scenarios that include fish ingestion, methylmerucry dominates mercury exposure.

For those hypothetical individuals exposed through consumption of both green plants and animal products, mercury exposure through consumption of green plants was greater than through consumption of animal products. This was the result of low plant-to-animal and soil-to-animal biotransfer factors when compared to the air-to-plant biotransfer factors. Of the green plants considered, exposure to mercury from fruits and fruiting vegetables was the largest. There is uncertainty in this result due to uncertainty in the role air-to-plant transfer plays for uptake of mercury for these plant products. Of the animal products considered, exposure to mercury was predicted to be the largest from dairy products.

Those hypothetical humans who were assumed to consume fish had the highest exposures. This was a result of the bioaccumulation factor for fish. Methylmercury was the primary species to which these individuals were exposed. On a per body weight basis, children were predicted to be more exposed than adults.

The piscivorous animals with the highest fish ingestion rate (per body weight) generally had the highest methylmercury intakes, except in cases where a piscivorous species was assumed to prey more heavily on trophic level 4 fish.

5. LONG RANGE TRANSPORT ANALYSIS

5.1 Description of the Analytic Approach

5.1.1 Objectives

The goal of this analysis was to model the emission, transport, and fate of airborne mercury over the continental U.S. using the meteorologic data for the year of 1989. The results of the simulation were intended to be used to answer a number of fundamental questions. Probably the most general question was "How much mercury is emitted to the air annually over the United States, and how much of that is then deposited back to U.S. soils and water bodies?" A second question was that of the contribution by source category to the total amount of mercury emitted and the amount deposited within the U.S. In order to answer the questions about the source relative depositions, information on chemical and physical forms of the mercury emissions from the various source categories was needed since these characteristics determine the rate and location of the wet and dry deposition processes for mercury.

The intent of the analysis was to determine which geographical areas of the United States having the highest and lowest amounts of deposition from sources using the overall results of the long-range transport modeling effort nation-wide. This analysis was expected to contribute understanding of the key variables, such as source location, chemical/physical form of emission, or meteorology, that might contribute to the outcomes. These long-range modeling efforts were also intended to be used for comparison with local impact modeling results, essentially to estimate the effects of hypothetical new local sources in relation to the estimated effects from long-range transport.

5.1.2 Description of the Long-range Transport Model Used

To estimate the nationwide concentration and deposition patterns for airborne mercury, the Regional Lagrangian Model of Air Pollution (RELMAP) was adapted to simulate the emission, transport and diffusion, and wet and dry deposition of elemental mercury vapor (Hg^0), divalent oxidized mercury gases (Hg^{2+}), and particle-bound mercury (Hg_P). The RELMAP was originally designed to simulate sulfur dioxide (SO_2) and sulfate (SO_4^{2-}). An evaluation of the model was performed for those pollutant species (Eder et al., 1986). The RELMAP provides a continual accounting of pollutant mass balances during the simulation process so that the user is assured that no pollutant mass is being created or destroyed by systematic numerical errors.

The RELMAP simulates long-range pollutant transport and diffusion in terms of individual pollutant puffs generated at regular time intervals at their source location with each puff moving independently within a predefined horizontal area. The puff motion is determined by predefined wind fields, with the wind flow vectors defined on a latitude-longitude grid covering the modeled area. Pollutant deposition to the surface by precipitation (wet deposition) is estimated using observed hourly precipitation data that are also analyzed on this latitude-longitude grid. For the RELMAP mercury simulations, the modeling grid resolution was set to ½ degree longitude by ⅓ degree latitude to approximate a 40-km square. Pollutant concentrations and depositions to the Earth's surface were calculated for each rectangular shaped area, or grid cell, defined by the array of grid points in the latitude-longitude grid.

For large modeling domains, such as that in this study for the lower 48 United States, the pollutant emissions from the many individual sources within each grid cell were totaled, and a single pollutant puff was generated at the cell center to represent all sources in that cell. These puffs were

generated on a regular time interval, in this case three hours, and the mass of the pollutant in the puff was the total of all emissions in the grid cell during that three-hour model time step. The puff was moved horizontally by the wind, horizontal diffusion was modeled by puff expansion, and the pollutant mass in the puff was distributed vertically in four layers between the Earth's surface and the top of the Planetary Boundary Layer (PBL). The PBL is the lowest portion of the atmosphere and is the layer most directly affected by physical and chemical interactions with the Earth's surface. As the puff moves, it is acted upon by precipitation (wet deposition) and dry deposition processes which remove some of the pollutant mass and deposit it to the surface. There may also be chemical processes within the puff which create or destroy particular pollutants. Due to the long atmospheric lifetime of mercury emissions compared to the sulfur compounds the RELMAP was originally designed to simulate, a PBL venting process was included to simulate the gradual leakage of pollutants into the free atmosphere above. The venting, dry deposition, wet deposition, and chemical parameterizations used in this modeling effort are described in detail in Appendix D. When the pollutant mass in a puff was depleted to a predefined limit, the puff was dropped from the model and any remaining pollutant was ignored for the remainder of the simulation. Also, if the puff was moved out of the horizontal model domain by the wind it was dropped from the simulation and its remaining pollutant load was ignored.

U.S. EPA's modifications to the RELMAP for atmospheric mercury simulation were heavily based on recent Lagrangian model developments in Europe (Petersen et al., 1995). The mercury version of RELMAP was developed to handle the three types of mercury mentioned above, as well as carbon soot. Recent experimental work indicates that ozone and carbon soot are both important in the wet deposition of Hg^0. Carbon soot, or total carbon aerosol, was included as a modeled pollutant in the mercury version of RELMAP to provide necessary information for the Hg^0 wet deposition parameterization. The Agency was able to obtain observed O_3 air concentration data from its Aerometric Information Retrieval System (AIRS) and from the ACIDMODES field study. Thus, O_3 was not included as an explicitly modeled pollutant. Methylmercury was not included in the mercury version of RELMAP because it is not yet known if it has a primary natural or anthropogenic source or if it is produced in the atmosphere.

For the RELMAP mercury modeling study, each of the source types modeled was assumed to emit mercury in a particular distribution of chemical and physical forms. These form distributions, or speciation factors, define the estimated fraction of mercury emitted as Hg^0, Hg^{2+}, or Hg_p, these were adopted from Petersen et al. (1995). Since there remains considerable uncertainty about Hg^{2+} attachment to particles, an alternate emission speciation was also simulated to measure the sensitivity of the RELMAP results to this uncertainty. The point source type definitions and their mercury emission speciation factors for the base and alternate scenarios are described in detail in Appendix F.

Global oceanic and terrestrial emissions were accounted for by using a background atmospheric concentration of elemental mercury gas of 1.6 ng/m^3. The wet and dry deposition parameterizations described in Appendix D were used to simulate the deposition of Hg^0 given a constant concentration of 1.6 ng/m^3 throughout the entire three-dimensional model domain to estimate the deposition of elemental mercury vapor from this global atmospheric reservoir.

5.1.3 Description of the Mercury Emissions Data Used

The atmospheric mercury emission inventory used for this modeling study is described in detail in Volume II of this report. Data from this inventory were used to compile estimates of the mercury emissions from 7 major stationary source types (point sources) and from a group of minor source types for which individual emission site locations were not available (area sources). The 8

emission source types resolved for input to the RELMAP mercury model were; electric-utility fossil-fueled boilers, non-utility fossil-fueled boilers, municipal solid waste combustion, medical waste incineration, chlor-alkali factories, non-ferrous metal smelting, all other point sources, and area sources. For each of the 7 point source types, an estimate of the mercury emission speciation profile was made to define the most likely chemical and physical forms of the emissions. In addition to these base-case emission profiles, alternate profiles were defined to allow model sensitivity tests to be performed.

The base-case and alternate mercury emission speciation profiles for the 7 point source types are shown in Table 5-1. The area sources were modeled as emissions of elemental mercury vapor only. The area source data used for input to the RELMAP model simulation included estimates of the emission of mercury from latex paints, which has not been included in the

Table 5-1
Emission Speciation Profiles for the Point Source Types Defined

Point Source Type	Base-Case Speciation (%)			Alternate Speciation (%)		
	Hg^0	Hg^{2+}	$Hg_p 0^c$	Hg^{0a}	Hg^{2+b}	$Hg_p{}^c$
Electric Utility Boilers	50	30	20	50	0	50
Non-utility Fossil Fuel Combustion						
Municipal Waste Combustion	20	60	20	20	0	80
Medical Waste Incineration						
Non-ferrous Metal Smelting	85	10	5	85	0	15
Chlor-alkali Plants	70	30	0	70	0	30
Other Point Sources	80	10	10	80	0	20

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor-phase Mercury
[c] Hg_p = Particle Bound/Mercury

assessment of emissions described in emissions inventory in Volume II. These latex paint emission estimates totaled 4 metric tons per year for the lower 48 states. The analyses differs because the inventory reflects that mercury has been removed from paint since the early 90's, however, emissions may still be occurring. A second discrepancy is that the long-range transport analysis modeled 3.25 tons of Hg from oil-fired utility boilers. A more recent analysis of the mercury content of residue oil necessitated the revision of the estimate in the inventory to 0.25 tons/yr from oil-fired utility boilers. The compiled nationwide patterns of elemental, divalent, and particle-bound mercury emissions from all point and area source types using the base-case emission speciation profiles are illustrated in Figures 5-1, 5-2 and 5-3, respectively. Figure 5-4 shows the pattern of particle-bound mercury emissions using the alternate speciations profiles.

Table 5-2 shows the results of applying the base-case emission speciation profiles to the eight mercury emission source types resolved by the RELMAP mercury model. These emissions are for the lower 48 states only and the total mercury mass figures differ slightly from the national totals shown in Volume II. The RELMAP analysis indicates that of the total anthropogenic emissions from the lower 48 states, 26% is from medical waste incineration, 22% is from municipal waste combustion, 22% is from electric utility boilers, 13% is from fossil fuel combustion other than that by electric utilities, 4% is from non-ferrous metal smelting, and 3% is from chlor-alkali factories. As a whole, large-scale fossil fuel combustion represents about 35% of the total anthropogenic mercury emissions to the atmosphere in the lower 48 states. The atmospheric emissions from all other point source types represent 7% of the total emission and area sources represent 3% of all anthropogenic emissions in the lower 48 states. Because of differences in the behavior of the various chemical and physical forms of mercury in the atmosphere, it is necessary to determine their relative contributions to the total mass of mercury emitted. Table 5-2 indicates that, based on the base-case emission speciation profiles, about 41% of all anthropogenic mercury emitted to the air from the lower 48 states is in the form of Hg^0 vapor, 41% is in the form of Hg^{2+} vapor, and 17% is emitted as particle-bound mercury.

Table 5-2
Mercury Emissions Inventory Totaled by Source Type Using Base-case Emission Speciation Profiles (metric tons per year)

Source Type	Hg^{0a}	Hg^{2+b}	$Hg_p{}^c$	Total Mercury
Medical Waste Incineration	11.7	35.1	11.7	58.6
Municipal Waste Combustion	10.0	29.9	10.0	49.8
Electric Utility Boilers	24.3	14.6	9.7	48.5
Non-Utility Fossil Fuel	14.3	8.6	5.7	28.5
Non-Ferrous Smelting	7.4	0.9	0.4	8.7
Chlor-alkali Factories	4.6	1.9	0.0	6.5
Other Point Sources	13.0	1.6	1.6	16.2
Area Sources	6.9	0.0	0.0	6.9
Total	92.0	92.6	39.1	223.8

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor-phase Mercury
[c] Hg_p = Particle-Bound/Mercury

Figure 5-1

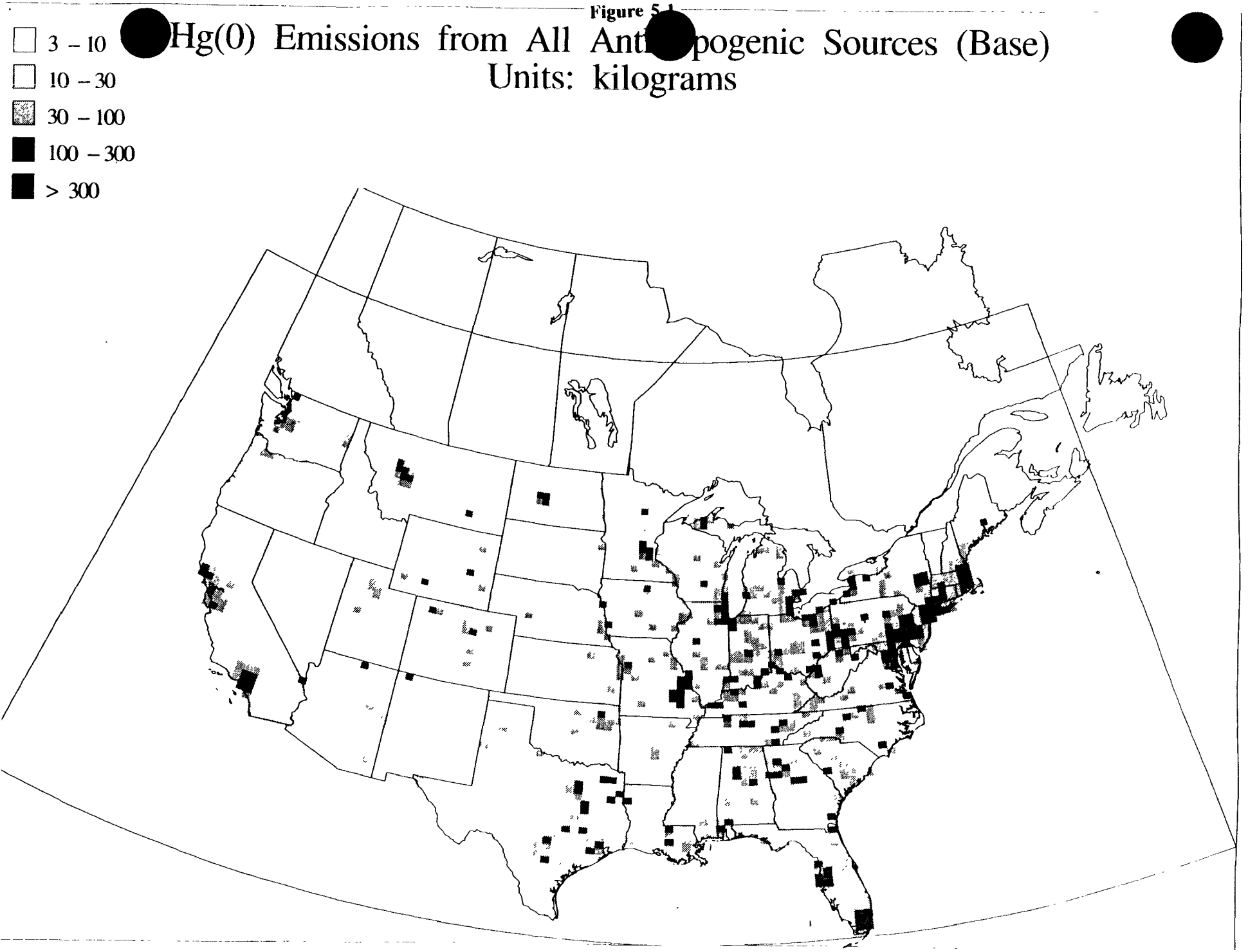

Hg(0) Emissions from All Anthropogenic Sources (Base)
Units: kilograms
3 – 10
10 – 30
30 – 100
100 – 300
> 300

Figure 5-2

Hg(++) Emissions from All Anthropogenic Sources (Base)
Units: kilograms

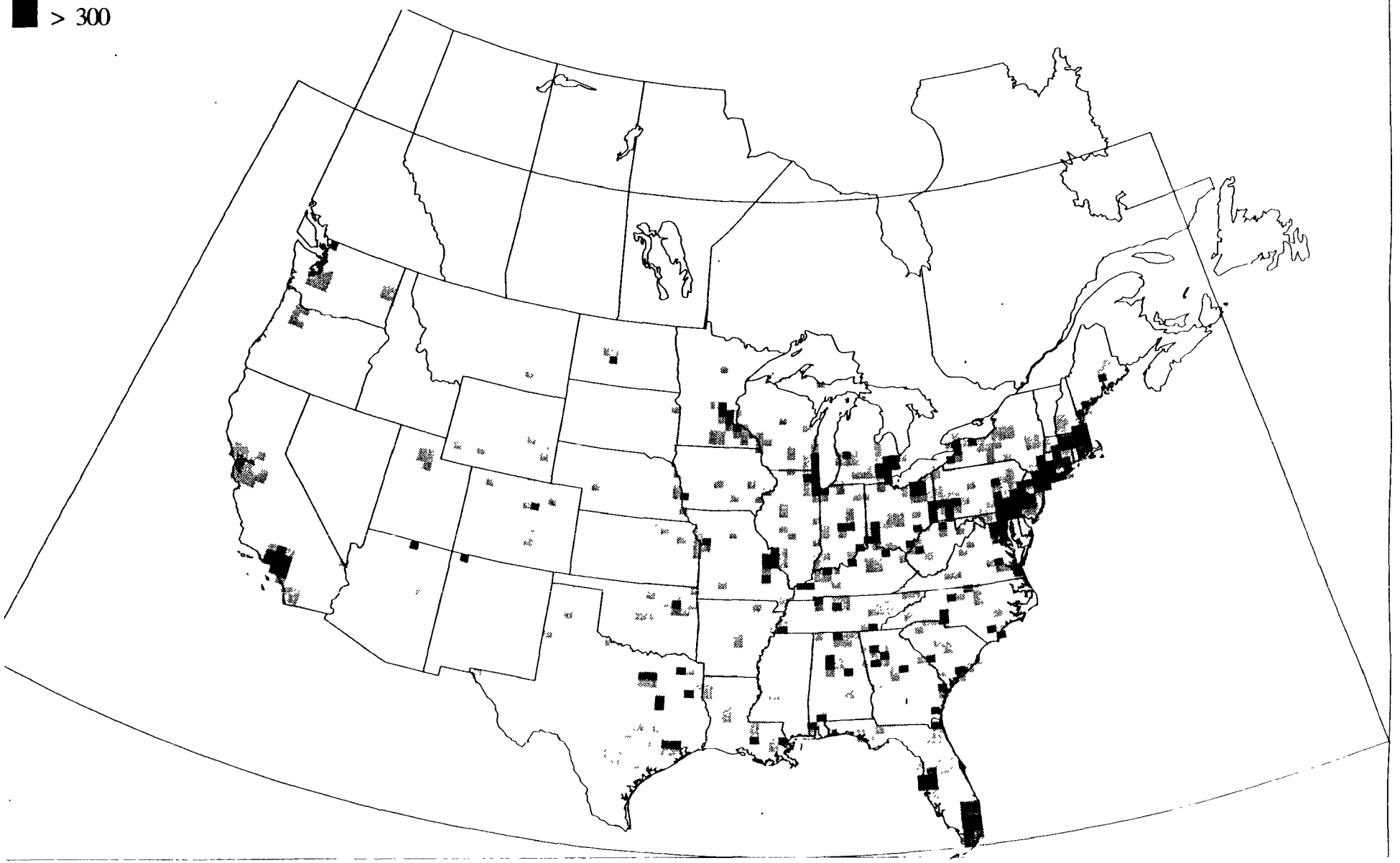


Figure 5-3

Hg(p) Emissions from All Anthropogenic Sources (Base)
Units: kilograms

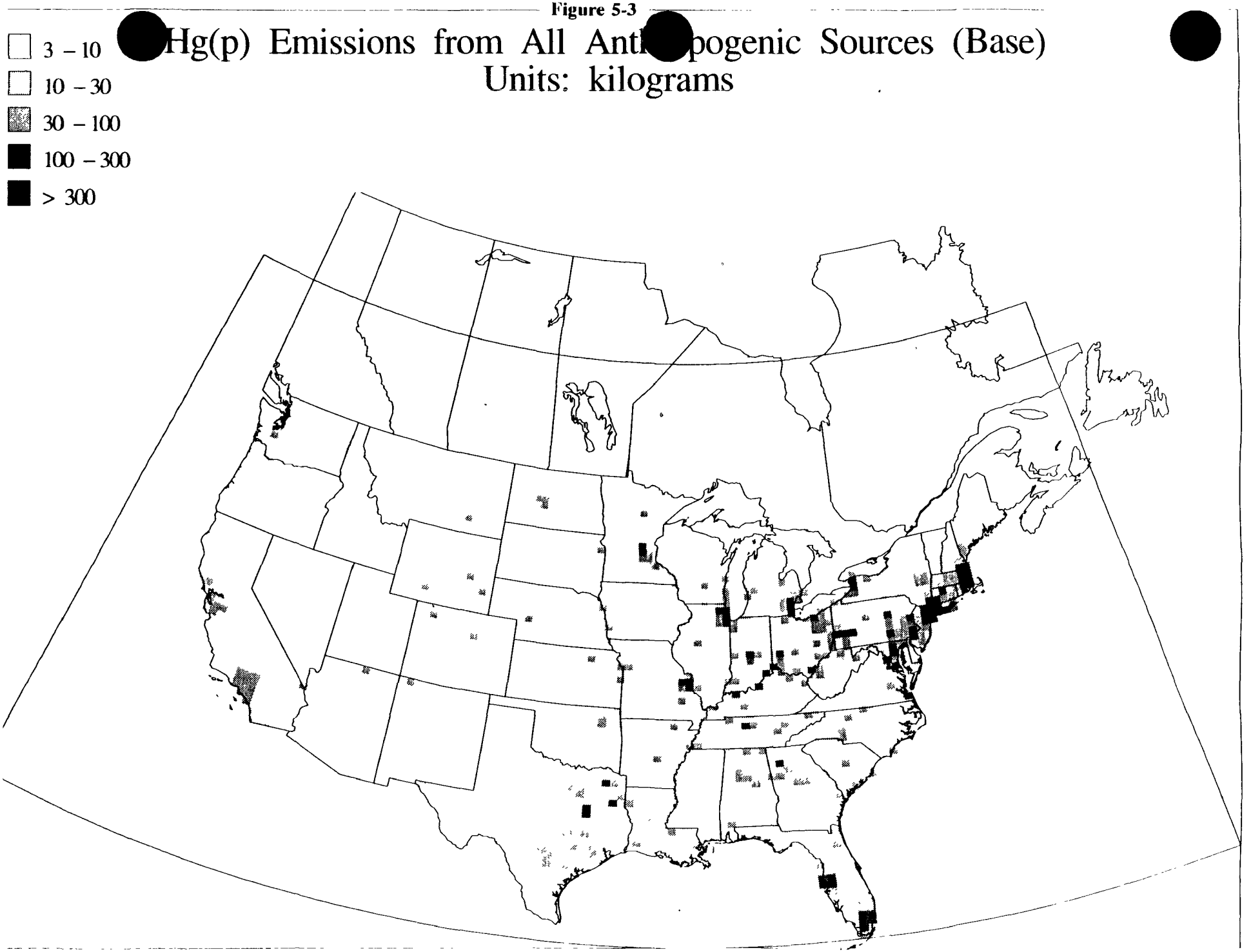
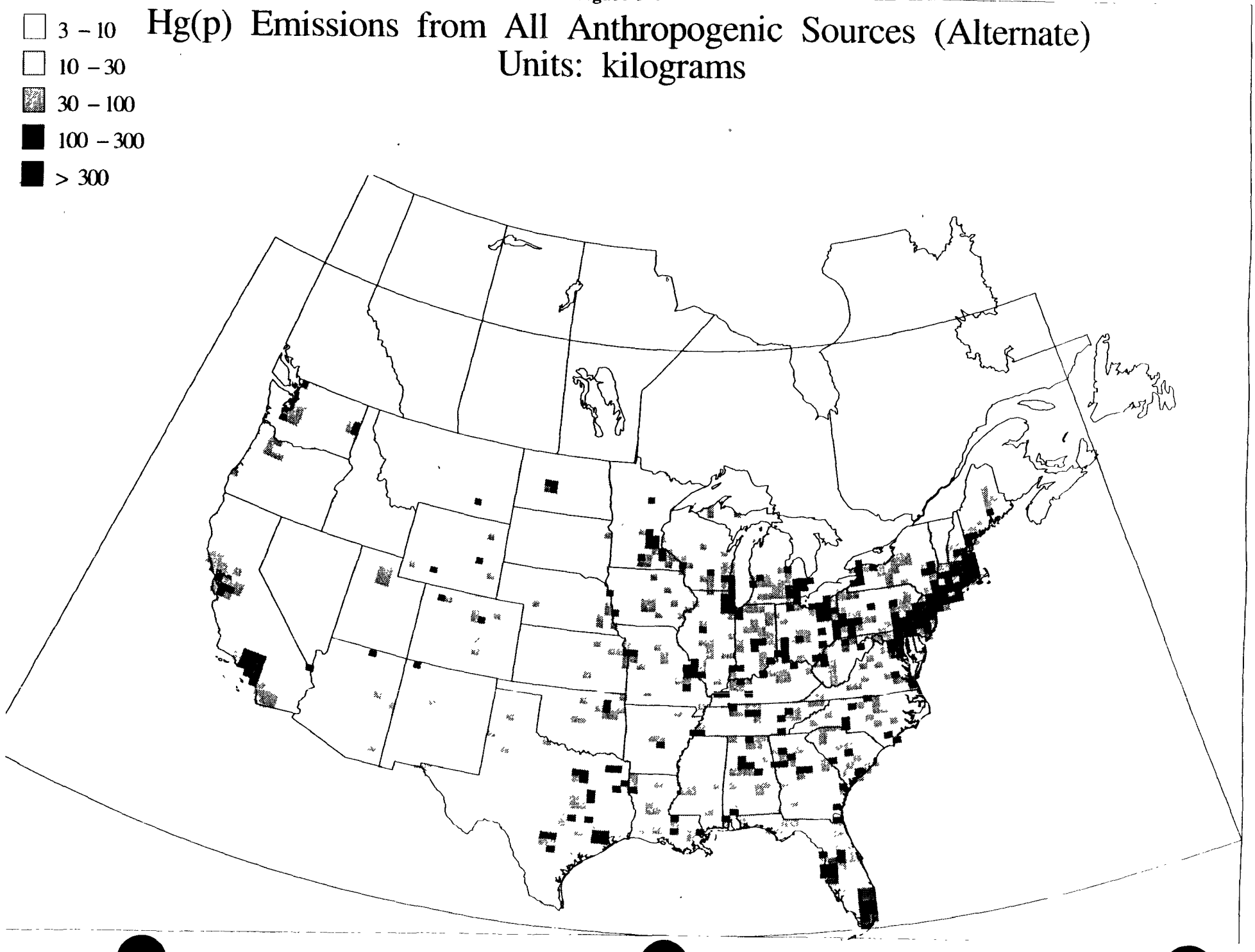


Figure 5-4


Hg(p) Emissions from All Anthropogenic Sources (Alternate)
Units: kilograms
3 – 10
10 – 30
30 – 100
100 – 300
> 300

5.2 An Interpretive Analysis of the Results

5.2.1 Mass Balances of Mercury within the Long-range Model Domain

The general mass balance of elemental mercury gas, divalent mercury gas, and particle-bound mercury from the RELMAP simulation results using the base-case emission speciation profiles are shown in Table 5-3. The mass-balance accounting for the simulation using the meteorologic data from the year 1989 shows a total of 223.8 metric tons of mercury emitted to the atmosphere from anthropogenic sources. This simulated emission total differs from the national totals indicated in Volume II since the states of Alaska and Hawaii are not within the model domain and latex paint emissions are not considered. Using the base-case emission speciation profiles, the simulation indicates that 77.9 metric tons of anthropogenic mercury emissions are deposited within the model domain and 0.6 metric tons remain in the air within the model domain at the end of the simulation. The remainder, about 145.3 metric tons, is transported outside the model domain and probably diffuses into the global atmospheric reservoir. The simulation also indicates that 33.0 metric tons of mercury is deposited within the model domain from this global atmospheric reservoir, suggesting that about four times as much mercury is being added to the global reservoir as is being deposited from it. The total amount of mercury deposited in the model domain annually from U.S. anthropogenic emissions and from the global background concentration is estimated to be 111.0 metric tons, or about one-half of the total atmospheric emissions from anthropogenic sources in the lower 48 United States.

Table 5-3
Modeled Mercury Mass Budget in Metric Tons for 1989
Using the Base-Case Emission Speciation Profiles

Source/Fate	Hg^{0a}	Hg^{2+b}	$Hg_p{}^c$	Total Mercury
Total U.S. anthropogenic emissions	92.0	92.6	39.1	223.8
Mass advected from model domain	90.4	29.9	25.0	145.3
Dry deposited anthropogenic emissions	0.0	39.0	0.6	39.6
Wet deposited anthropogenic emissions	1.2	23.6	13.4	38.3
Remaining in air at end of simulation	0.4	0.1	0.1	0.6
Total deposited anthropogenic emissions	1.2	62.6	14.1	77.9
Deposition from background Hg^0	33.0	0.0	0.0	33.0
Mercury deposited from all sources	34.2	62.6	14.1	111.0
(All figures rounded to the nearest tenth of a metric ton)				

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor-phase Mercury
[c] Hg_p = Particle-Bound/Mercury

As shown in Table 5-4, the alternate case emission speciation profiles result in a noticeably different mass balance. By assuming that all of the Hg^{2+} emitted becomes attached to ambient particulate matter, the total deposition of anthropogenic Hg to the surface is reduced by

Table 5-4
Modeled Mercury Mass Budget in Metric Tons for 1989
Using the Alternate Emission Speciation Profiles

Source/Fate	Hg^{0a}	Hg^{2+b}	Hg_p[c]	Total Mercury
Total U.S. anthropogenic emissions	92.0	0.0	131.7	223.8
Mass advected from model domain	90.4	0.0	84.5	174.9
Dry deposited anthropogenic emissions	0.0	0.0	2.1	2.1
Wet deposited anthropogenic emissions	1.2	0.0	44.9	46.1
Remaining in air at end of simulation	0.4	0.0	0.2	0.6
Total deposited anthropogenic emissions	1.2	0.0	47.0	48.2
Deposition from background Hg^0	33.0	0.0	0.0	33.0
Mercury deposited from all sources	34.2	0.0	47.0	81.3
(All figures rounded to the nearest tenth of a metric ton)				

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor-phase Mercury
[c] Hg_p = Particle-Bound/Mercury

about 40%. This is primarily due to the fact that the dry deposition velocity of particulate matter in the size range typical for continental air masses with moderate urban influences (~0.3 micron diameter) is much smaller than the dry deposition velocity assumed for Hg^{2+} vapor. Less efficient wet scavenging of particulate versus gaseous Hg^{2+} also contributes to the lower total deposition using the alternate emission speciation profiles. While the assumption of total Hg^{2+} attachment to particles is only intended as a bounding exercise, the results show the importance of an accurate determination of the mass of particulate Hg emitted and that formed during transport.

Of the total anthropogenic mercury mass deposited to the surface in the model domain, 80% is estimated by the RELMAP to come from Hg^{2+} emissions, 18% from Hg_p emissions and 2% from Hg^0 emissions when the base-case emission speciation profiles are used. When the deposition of Hg^0 from the global background is considered in addition to anthropogenic sources in the lower 48 states, the species fractions become 56% Hg^{2+}, 31% Hg^0 and 13% Hg_p. The vast majority of mercury already in the global atmosphere is in the form of Hg^0 and, in general, the anthropogenic Hg^0 emissions do not greatly increase the existing Hg^0 concentration. Although Hg^0 is removed from the atmosphere very slowly, the global background reservoir is large and extraction of mercury from it is significant in terms of the total deposition. It should be noted here that dry deposition of Hg^0 is significant only at very high concentrations and has not been included in the RELMAP simulations. Wet deposition is the only major pathway for removal of Hg^0 from the atmosphere. This removal pathway simulated by the RELMAP involves oxidation of mercury by ozone in an aqueous solution; thus, the Hg^0 that is extracted from the atmosphere by the modeled precipitation process would actually be deposited primarily in the form of Hg^{2+}.

Results from the RELMAP simulation show that of the 92.0 metric tons of anthropogenic Hg^0 emitted in the lower 48 states, only 1.2 tons (1.3%) is deposited within the model domain, while of the 92.6 metric tons of Hg^{2+} emitted, about 62.6 tons (67.6%) is deposited. Ninety-eight percent of the deposited anthropogenic mercury was emitted in the form of Hg^{2+} or Hg_p. Thus, a strong argument

can be made that the combined Hg^{2+} and Hg_p component of anthropogenic mercury emissions can be used as an indicator of eventual deposition of those emissions to the lower 48 states and surrounding areas. The emission inventory and base-case chemical/physical speciations profiles indicate that of all combined Hg^{2+} and Hg_p emissions, about 36% is from medical waste incineration, 30% is from municipal waste combustion, 18% is from electric utility boilers, 11% is from combustion of fossil fuel other than by electric utilities, 1% is from chlor-alkali factories, 1% is from non-ferrous metal smelting, and 2% is from all other sources.

5.2.2 Qualitative Description of Mercury Concentration Results

Annual average surface-level concentration fields for elemental mercury, divalent mercury, and particulate mercury have been obtained from the RELMAP simulation using the meteorologic data for the year 1989. Figure 5-5 shows the annual average elemental mercury (Hg^0) concentration at ground level from anthropogenic sources obtained by using the estimates of Petersen et al. (1995) for the source-based emission speciation profiles (base case). It shows that anthropogenic Hg^0 concentrations remain less than 0.1 ng/m^3 over most of the investigation area. The areas where the average anthropogenic Hg^0 concentrations exceed 0.1 ng/m^3 are mostly confined to the highly industrialized regions of the eastern Mid-west and the North-east. Compared to the estimated average global background concentration of 1.6 ng/m^3, this 0.1 ng/m^3 elevation of Hg^0 concentration by anthropogenic emissions is rather small.

Figure 5-6 shows annual average divalent mercury vapor (Hg^{2+}) air concentrations, also using the base case emissions. These values are significantly lower than for anthropogenic Hg^0, and there are some new areas of maximum concentration. The higher concentration areas have values from 0.05 to just over 0.1 ng/m^3 and are mostly confined to Florida, the Midwest and the Northeast corridor. The background atmospheric mercury loading is assumed to be completely in the elemental form, so there is no background contribution to the Hg^{2+} concentrations. In most areas, the anthropogenic component of the Hg^0 concentrations shown in Figure 5-5 are 3 to 5 times higher than the Hg^{2+} concentrations shown in Figure 5-6. For the base-case emission speciation, Hg^{2+} vapor is a minor component of the total mercury emissions from some source types, but it is a significant part of the total mercury emissions from waste incineration (60%) and fossil fuel combustion (30%). Since the total Hg^{2+} emissions are about equal to those for Hg^0, these much lower average annual Hg^{2+} concentrations cannot be attributed to the emissions. The lower simulated air concentrations of Hg^{2+} vapor are due to more rapid depletion of atmospheric mercury from wet and dry deposition.

The RELMAP Hg^0 and Hg^{2+} air concentration results taken together with the assumed background Hg^0 concentration of 1.6 ng/m^3 agree well with observations of vapor-phase Hg air concentration in Minnesota by Fitzgerald et al. (1991), in Vermont by Burke et al. (1994) and in Wisconsin by Lambourg et al. (In press). These works showed that annual average vapor-phase Hg concentrations were near the levels found over other remote locations in the northern hemisphere, from 1.6 to 2.0 ng/m^3. Measurements taken for a two-week period at three sites in Broward County, Florida, (Dvonch et al., 1995) show slightly elevated vapor-phase Hg air concentrations for two of those sites downwind of industrial activities. These two sites had average vapor-phase Hg air concentrations of 3.3 and 2.8 ng/m^3. The RELMAP simulation results for the Fort Lauderdale area show only about a 0.2 ng/m^3 elevation of the annual average vapor-phase Hg (Hg^0 plus Hg^{2+}) concentration over the 1.6 ng/m^3 background value assumed. The measurements of Dvonch et al. (1995), however, did not extend for a significant portion of

Figure 5-5

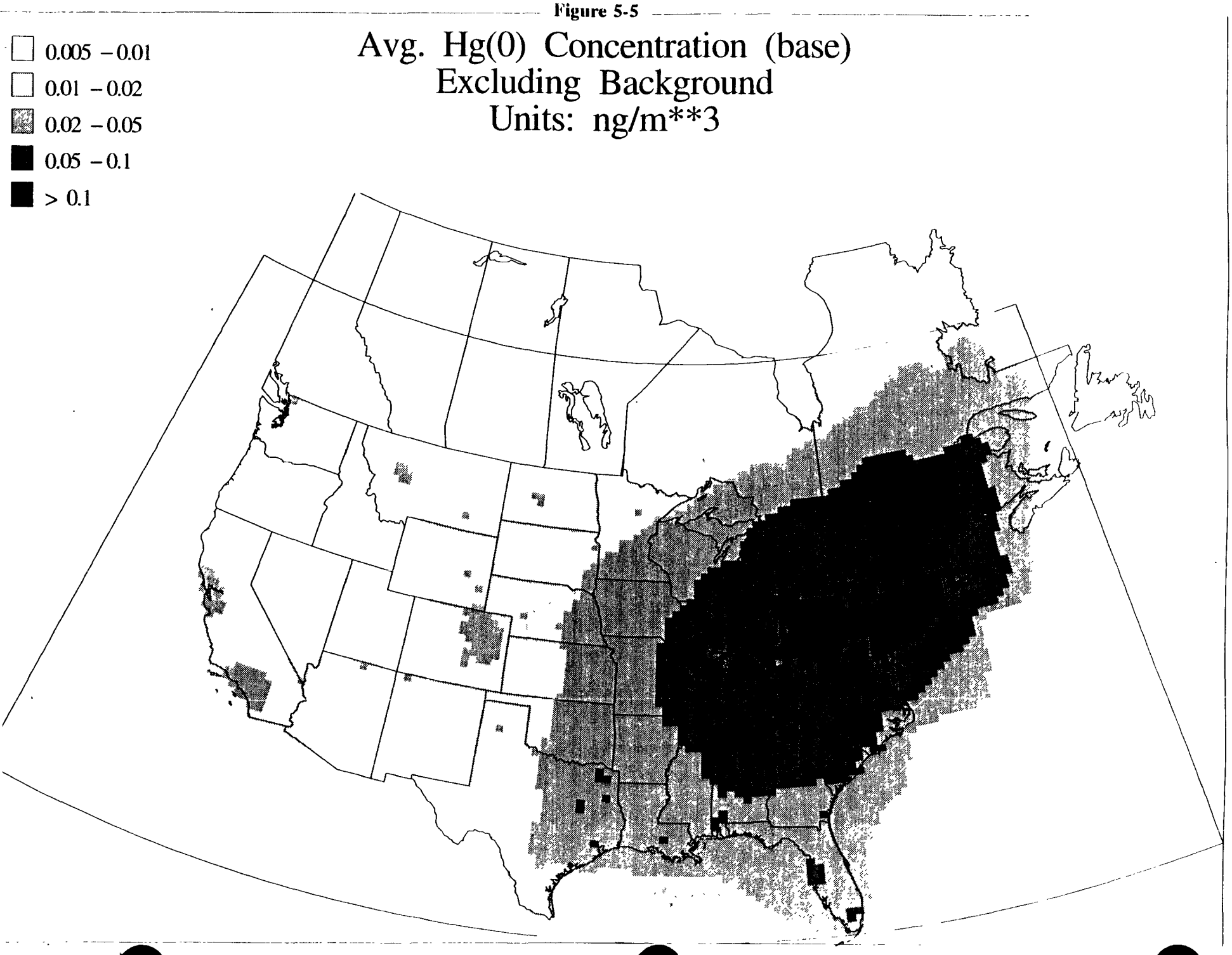
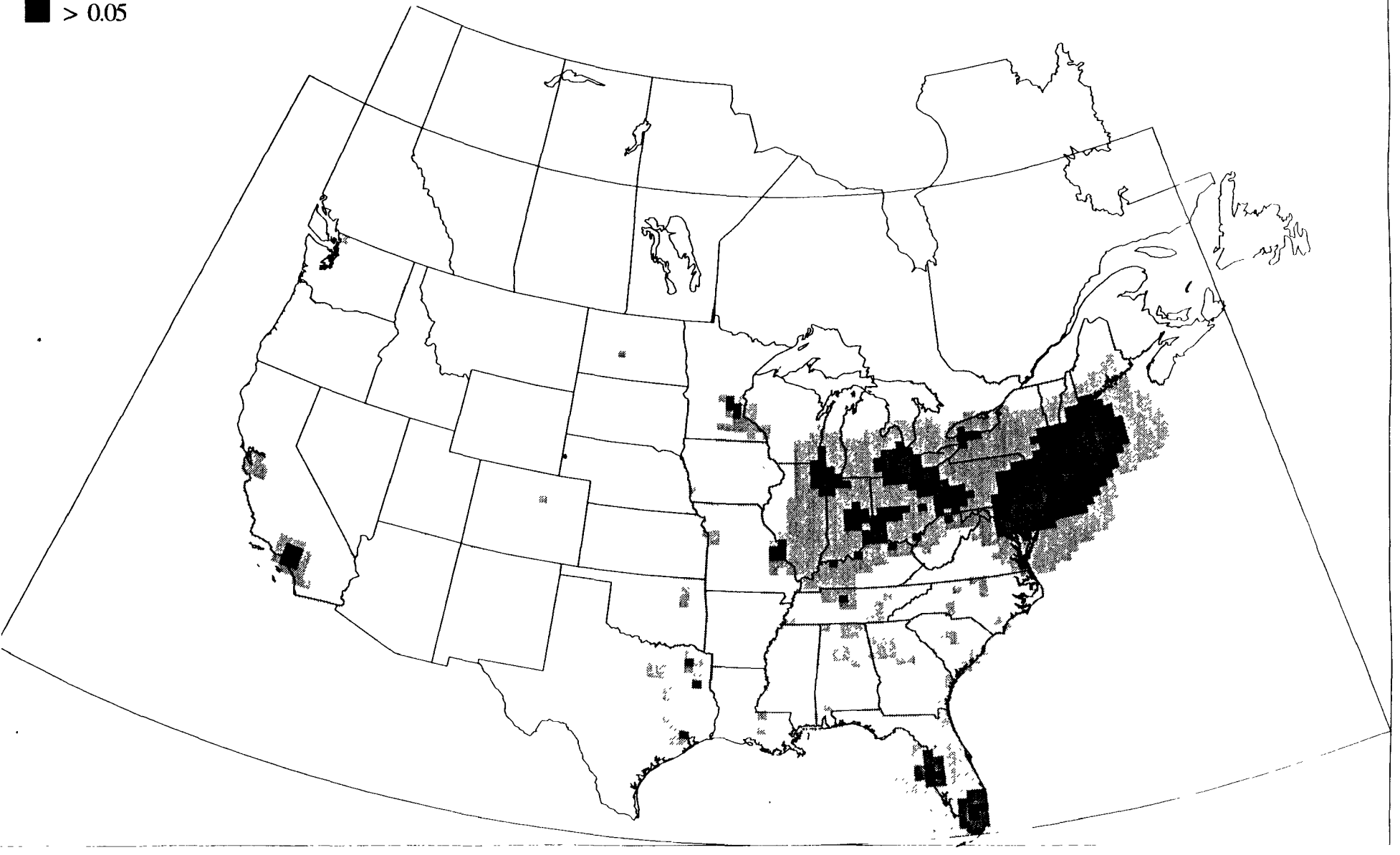

Avg. Hg(0) Concentration (base)
Excluding Background
Units: ng/m**3
0.005 – 0.01
0.01 – 0.02
0.02 – 0.05
0.05 – 0.1
> 0.1

Figure 5-6

Avg. Hg(++) Concentration (base)
Units: ng/m**3



the year and there was no discrimination between Hg^0 and Hg^{2+} forms. The third site for their observations had an average vapor-phase air concentration of 1.8 ng/m^3, which is what the RELMAP simulation suggests. A more comprehensive air monitoring program is required before an evaluation of the RELMAP results in Florida can be performed.

Particulate mercury (Hg_p) emissions are thought to be a small fraction of the total for most source types. For the base-case emission speciation, 20% is the largest particulate fraction of mercury emissions for any source type. Figure 5-7 shows that the simulated annual average Hg_p concentrations were even lower than those for Hg^{2+} vapor. The maximum annual average values are around 50-100 pg/m^3 (0.05-0.1 ng/m^3) in the urban centers of the Northeast. Keeler et al. (1994) found instantaneous Hg_p concentrations in urban Detroit during March of 1992 of over 1 ng/m^3 and average concentrations over an 18-day period of 94 pg/m^3. Given the 40-km horizontal scale of the RELMAP computational grid, however, one cannot expect the simulation to reflect these extreme local-scale measurement results. The RELMAP simulation suggests an annual average Hg_p concentration in the Detroit area of about 50 pg/m^3. Dvonch et al. (1995) found average Hg_p concentrations in Broward County, Florida, of between 34 and 51 pg/m^3 at three sites from 25 August to 7 September of 1993. The RELMAP simulation results agree well with these observations around the city of Fort Lauderdale. Keeler et al. (1994) found annual average Hg_p air concentrations of 10.5 pg/m^3 in Pellston, Michigan, 22.4 pg/m^3 in South Haven, Michigan, and 21.9 pg/m^3 in Ann Arbor, Michigan, from April 1993 to April 1994, and 11.2 pg/m^3 in Underhill, Vermont, for the year of 1993. The RELMAP simulation results agree quite well with these observations.

Table 5-5 shows a percentile analysis of the simulated concentration results from the RELMAP grid cells within the lower 48 United States. This table shows that the Hg^0 concentrations never exceeded the assumed background level of 1.6 ng/m^3 by a large relative amount. It also shows that Hg^{2+} and to a lesser degree Hg_p air concentrations were highly elevated in only a few grid cells. There is an order of magnitude difference in the Hg^{2+} concentrations at the 90th percentile level and the maximum level, with a factor of 5 difference for Hg_p.

For the alternate emission speciation tests, the Hg^{2+} vapor emission fraction was redistributed to the Hg_p fraction. This was done to simulate the complete attachment of Hg^{2+} vapor to ambient particulate matter. The annual average Hg^0 and Hg_p concentration fields from this test (not shown) indicate that, as one would expect, there was no change to the Hg^0 results, but the Hg_p concentrations were increased nearly to the level of anthropogenic Hg^0, with maximum concentrations over 100 pg/m^3 (0.1 ng/m^3) over the larger urban areas of Florida and the Midwest and over nearly all of the Washington, D.C. to Boston corridor.

5.2.3 Description of Mercury Wet Deposition Simulation Results

Figure 5-8 shows the total simulated wet deposition of Hg^0 from anthropogenic sources using the meteorologic data for the year 1989 using the base emission speciation factors. Figure 5-9 shows the total simulated wet deposition of Hg^0 assuming only a non-depleting global background concentration of 1.6 ng/m^3. Both of these wet deposition results are influenced by ozone and soot concentrations due to the chemical transformations modeled by the RELMAP. Emission patterns influence the primary anthropogenic Hg^0 wet deposition pattern, and it is obvious that total annual precipitation is a strong factor in wet deposition from the global background concentration with heaviest wet deposition in areas with the highest annual

Figure 5-

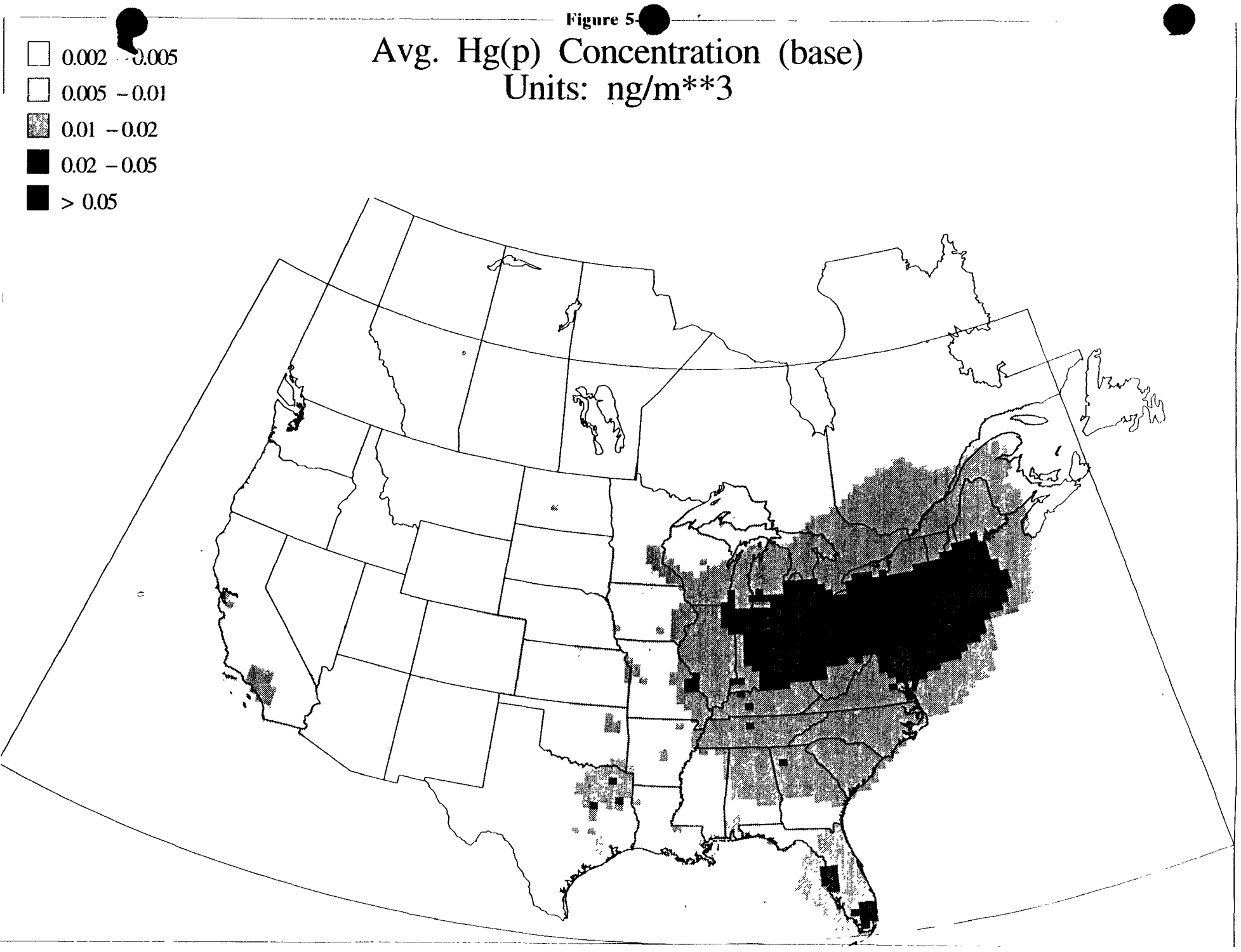

Avg. Hg(p) Concentration (base)
Units: ng/m**3
0.002 - 0.005
0.005 - 0.01
0.01 - 0.02
0.02 - 0.05
> 0.05

Table 5-5
Percentile Analysis of RELMAP Simulated Concentration Results for the Continental U. S. Using the Base-Case Emissions Speciation

Variable	Min	10th	50th	90th	Max
Full Area					
Hg^{0a} concentration (ng/m^3)	1.602	1.606	1.619	1.681	1.899
Hg^{2+b} concentration (pg/m^3)	0.265	0.804	3.368	14.72	149.1
Hg_p^c concentration (pg/m^3)	0.391	1.426	5.183	19.18	99.33
Total mercury (ng/m^3)	1.603	1.608	1.627	1.715	2.114
East of 90°W longitude					
Hg^{0a} concentration (ng/m^3)	1.615	1.638	1.665	1.705	1.899
Hg^{2+b} concentration (pg/m^3)	1.486	4.745	9.966	25.25	149.1
Hg_p^c concentration (pg/m^3)	4.058	8.815	14.88	28.01	99.33
Total mercury (ng/m^3)	1.622	1.655	1.691	1.755	2.114
West of 90°W longitude					
Hg^{0a} concentration (ng/m^3)	1.602	1.605	1.612	1.636	1.818
Hg^{2+b} concentration (pg/m^3)	0.265	0.687	1.909	6.303	29.80
Hg_p^c concentration (pg/m^3)	0.391	1.243	3.327	8.565	28.73
Total mercury (ng/m^3)	1.603	1.607	1.618	1.651	1.860

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor-phase Mercury
[c] Hg_p = Particle-Bound/Mercury

Figure 5-8

Hg(0) Wet Deposition (base) Excluding Background Units: ug/m**2

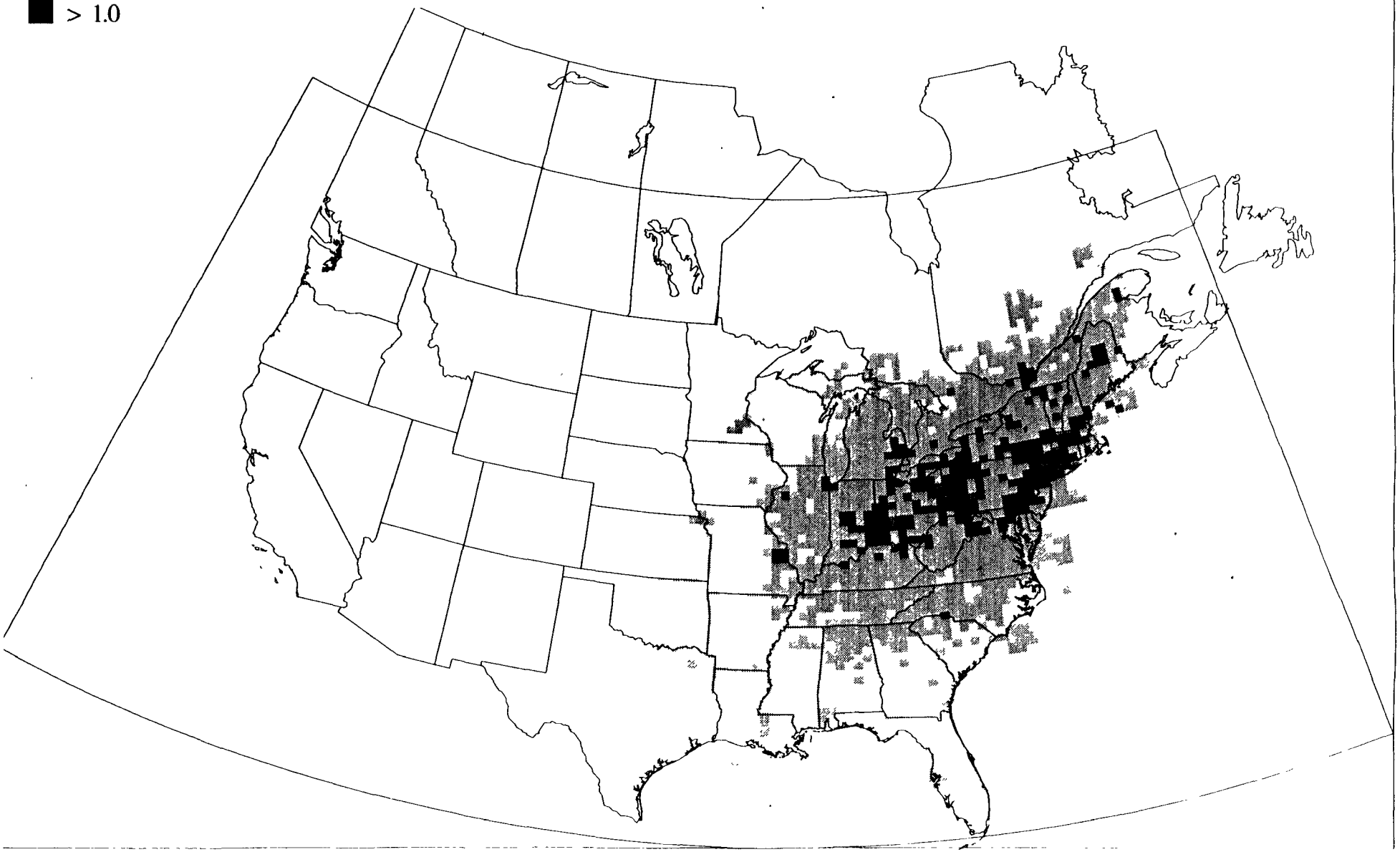
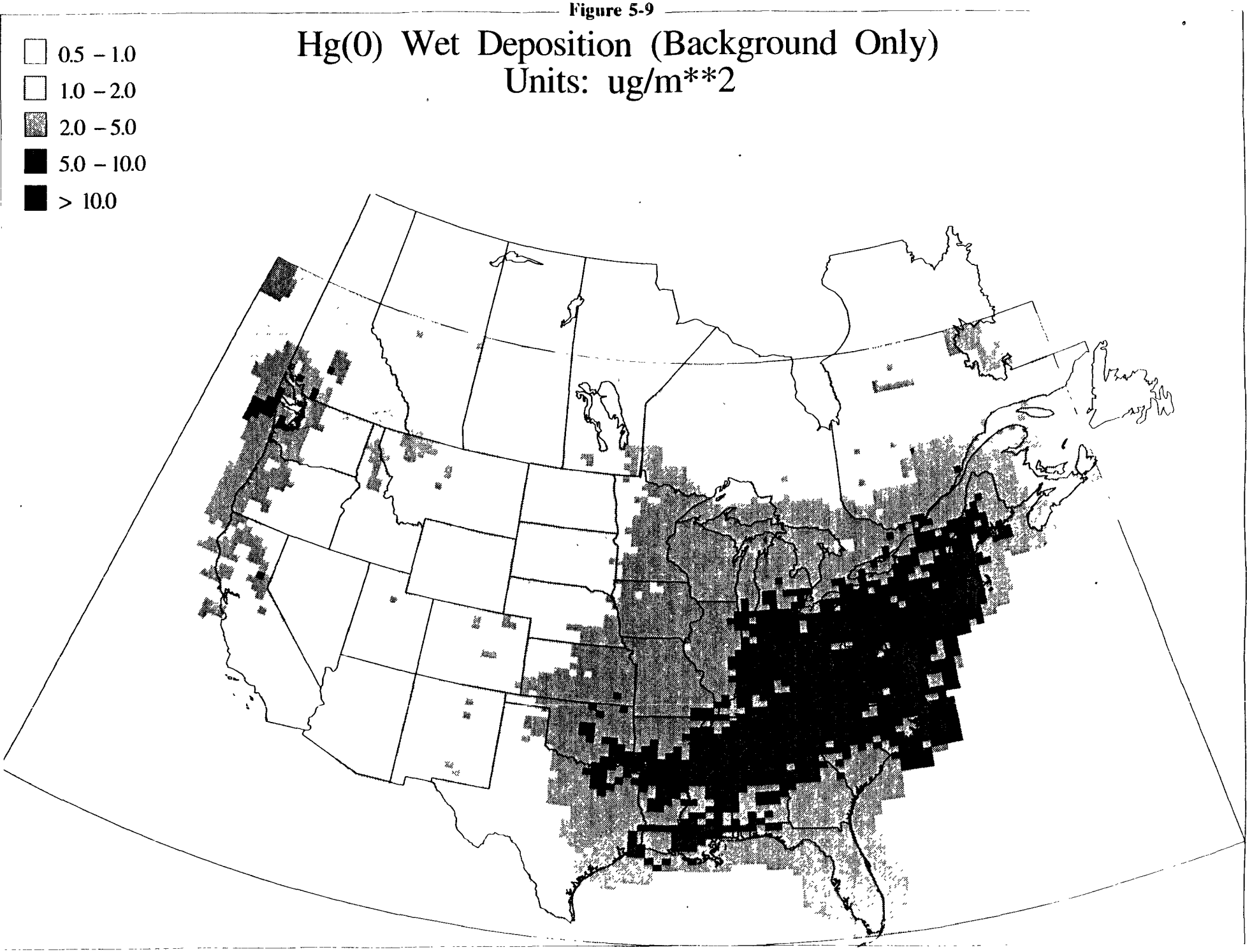


Figure 5-9


Hg(0) Wet Deposition (Background Only)
Units: ug/m**2
0.5 – 1.0
1.0 – 2.0
2.0 – 5.0
5.0 – 10.0
> 10.0

precipitation. It is widely accepted that deposition of measurable quantities of mercury occurs on continental and global scales, and the RELMAP simulation shows areas of Hg^0 wet deposition occurring in remote areas. The base-case wet deposition results for Hg^{2+} vapor shown in Figure 5-10 show high deposition areas that are much more local to the emission source areas. There are many model cells in urban areas with wet deposition totals of Hg^{2+} vapor over 20 $\mu g/m^2$ while most of the cells in the non-urban areas have wet depositions of less than 5 $\mu g/m^2$. This indicates the Hg^{2+} vapor wet deposits more on the local scale and not on regional or global scales and that its wet removal of Hg^{2+} from the atmosphere is much more rapid than for Hg^0. This is an expected result due to the higher water solubility of most mercuric salts compared to mercury in the elemental form. Figure 5-11 shows that for the base emission speciation the maximum simulated wet deposition of Hg_P is about half of that for Hg^{2+} vapor. This is partly due to differences in the total mass of Hg_P emitted compared to Hg^{2+}, but it is also due to the less efficient wet scavenging that is assumed for Hg_P versus Hg^{2+}. The areas of Hg_P wet deposition are also more widely distributed than for Hg^{2+} due to the slower wet scavenging of Hg_P and, thus, a greater opportunity for long-range transport.

The total wet deposition of mercury emitted in all three forms is shown in Figure 5-12. This illustration shows significant wet deposition of mercury over most of the eastern half of the U.S. For the simulated meteorological year of 1989, nearly the entire eastern half of the nation has a wet deposition total of over 5 $\mu g/m^2$ and values exceed 20 $\mu g/m^2$ over much of the urban northeast U.S. In fact, the largest simulated wet deposition exceeded 100 $\mu g/m^2$ in the grid cell containing New York City. Figure 5-12 was not designed to highlight these maximum wet deposition results because at this time such extreme wet deposition rates for total mercury cannot be substantiated by observations. In the RELMAP simulation the most impacted areas are subjected to wet deposition of mercury mainly from emissions of Hg^{2+} vapor. It is likely that the RELMAP model for mercury may still be significantly incomplete, and that other chemical and/or physical transformations may occur which moderate the wet deposition of Hg^{2+} vapor and possibly Hg_P.

There exist only limited data with which to compare the RELMAP simulation results. Measurements of mercury wet deposition at three locations in northeastern Minnesota during 1989 by Glass et al. (1991) indicated annual wet deposition rates of 6.5 $\mu g/m^2$ at Duluth, 13.5 $\mu g/m^2$ at Marcell and 41.9 $\mu g/m^2$ at Ely. A later study by Sorensen et al. (1994) measuring annual wet deposition of mercury during 1990, 1991 and 1992 at Ely, Duluth and seven other sites in Minnesota, upper Michigan and northeastern North Dakota found all annual wet deposition totals to be within the range of 3.8 to 9.7 $\mu g/m^2$, bringing into question the Ely observation of 41.9 $\mu g/m^2$ in 1989 by Glass et al. (1991). Measurements by Fitzgerald et al. (1991) at Little Rock Lake, in northern Wisconsin, of mercury in snow during February and March, 1989, and in rain from May to August, 1989, have been used to estimate annual mercury depositions in rain and snow of 4.5 and 2.3 $\mu g/m^2$, respectively. This suggests a total annual mercury wet deposition of 6.8 $\mu g/m^2$ at Little Rock Lake. Measurements at Presque Isle, also in northern Wisconsin, from 1993 to 1994 by Lambourg et al. (In press) suggested a wet deposition rate for total mercury of 5.2 $\mu g/m^2/yr$, somewhat less than the measurements by Fitzgerald et al. (1991). The extremely heavy rainfall during the summer of 1993 in the mid-west states to the south and west of Presque Isle may be responsible for the lower wet deposition. The RELMAP simulation results using the meteorologic data for 1989 indicate 2 to 10 $\mu g/m^2$ wet deposition of total mercury over most of the area represented by these studies; the major exception was the Minneapolis-Saint Paul metropolitan area where the RELMAP indicates over 20 $\mu g/m^2$.

Figure 5-10

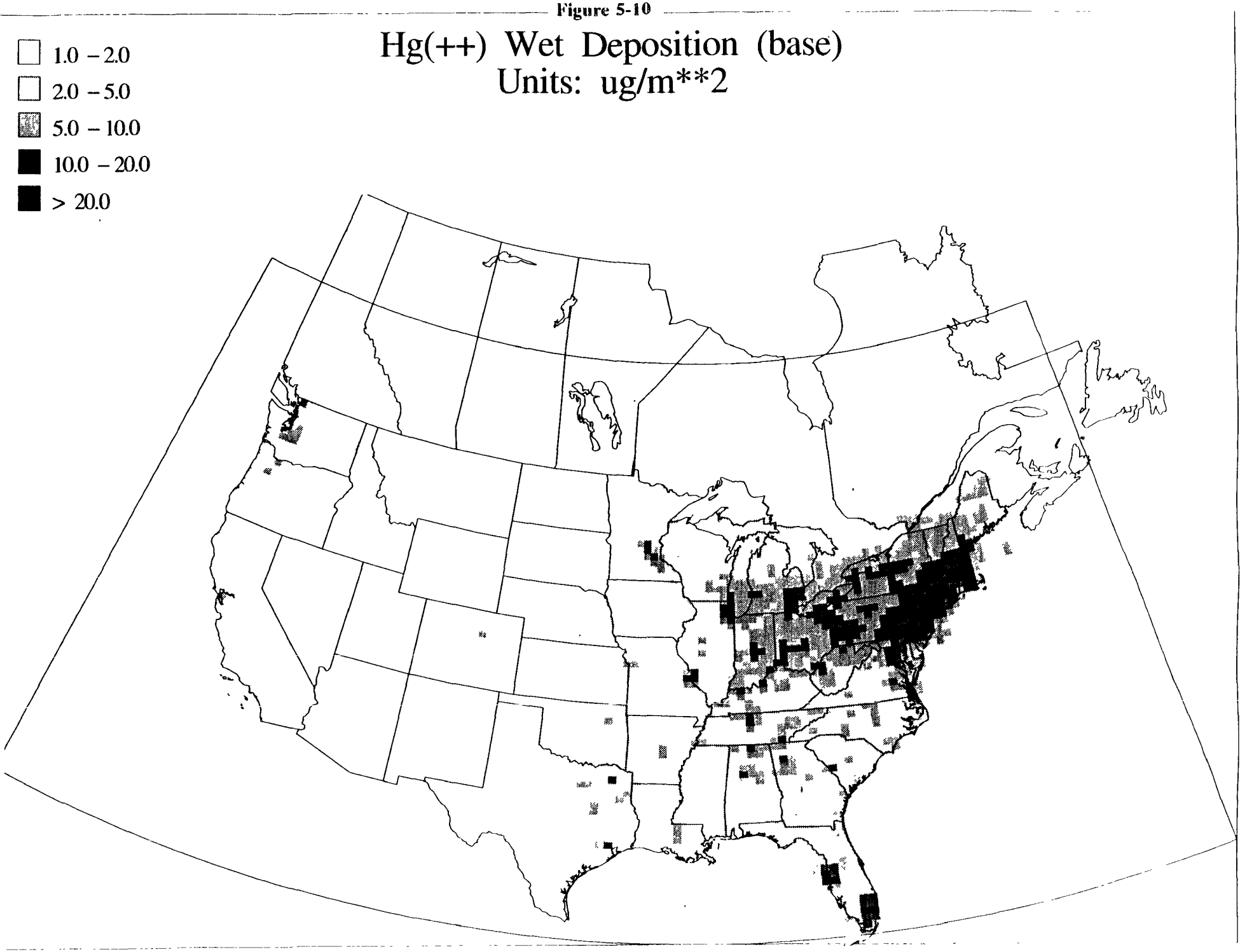

Hg(++) Wet Deposition (base)
Units: ug/m**2
1.0 – 2.0
2.0 – 5.0
5.0 – 10.0
10.0 – 20.0
> 20.0

Figure 5-11

Hg(p) Wet Deposition (base)
Units: ug/m**2

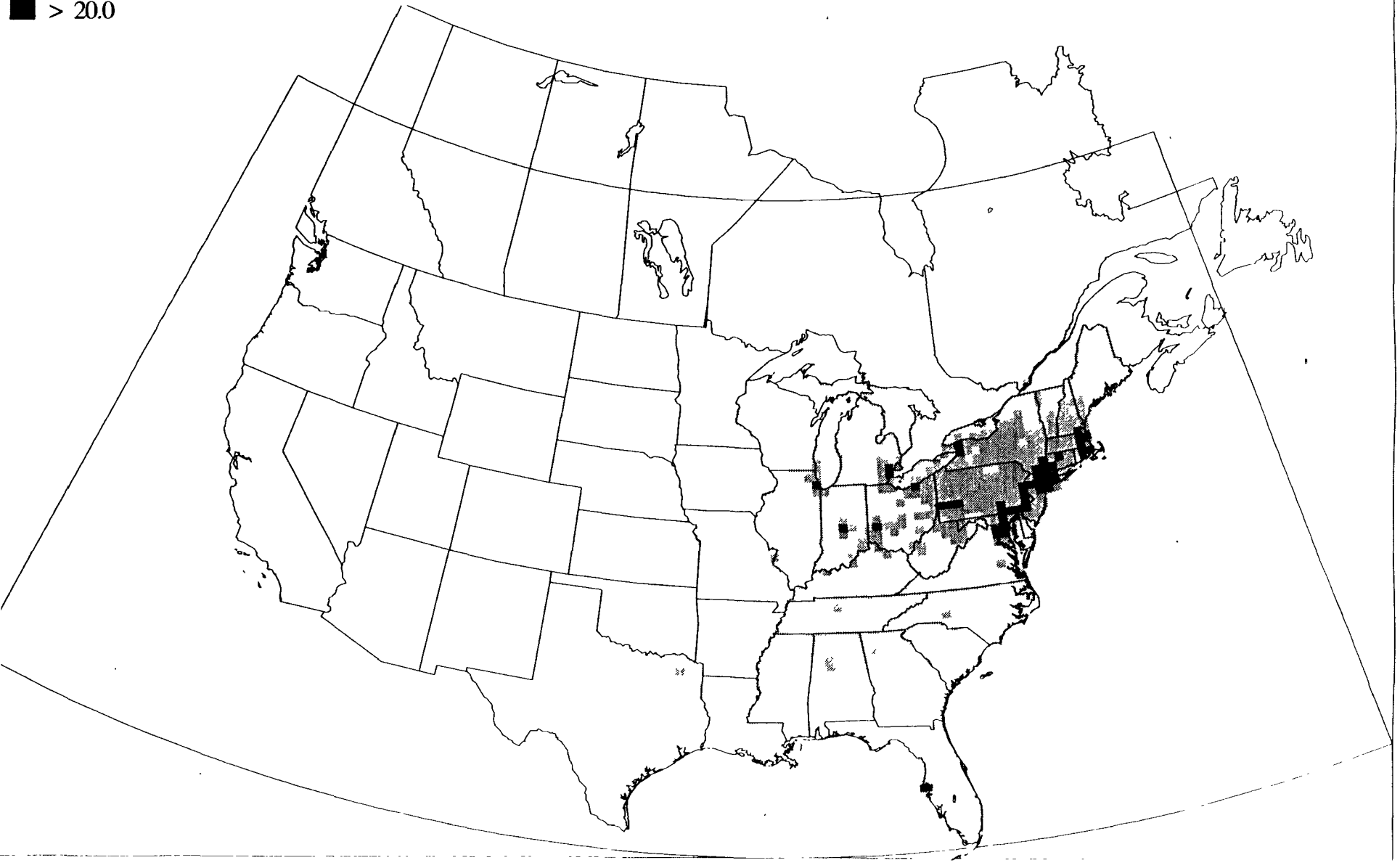
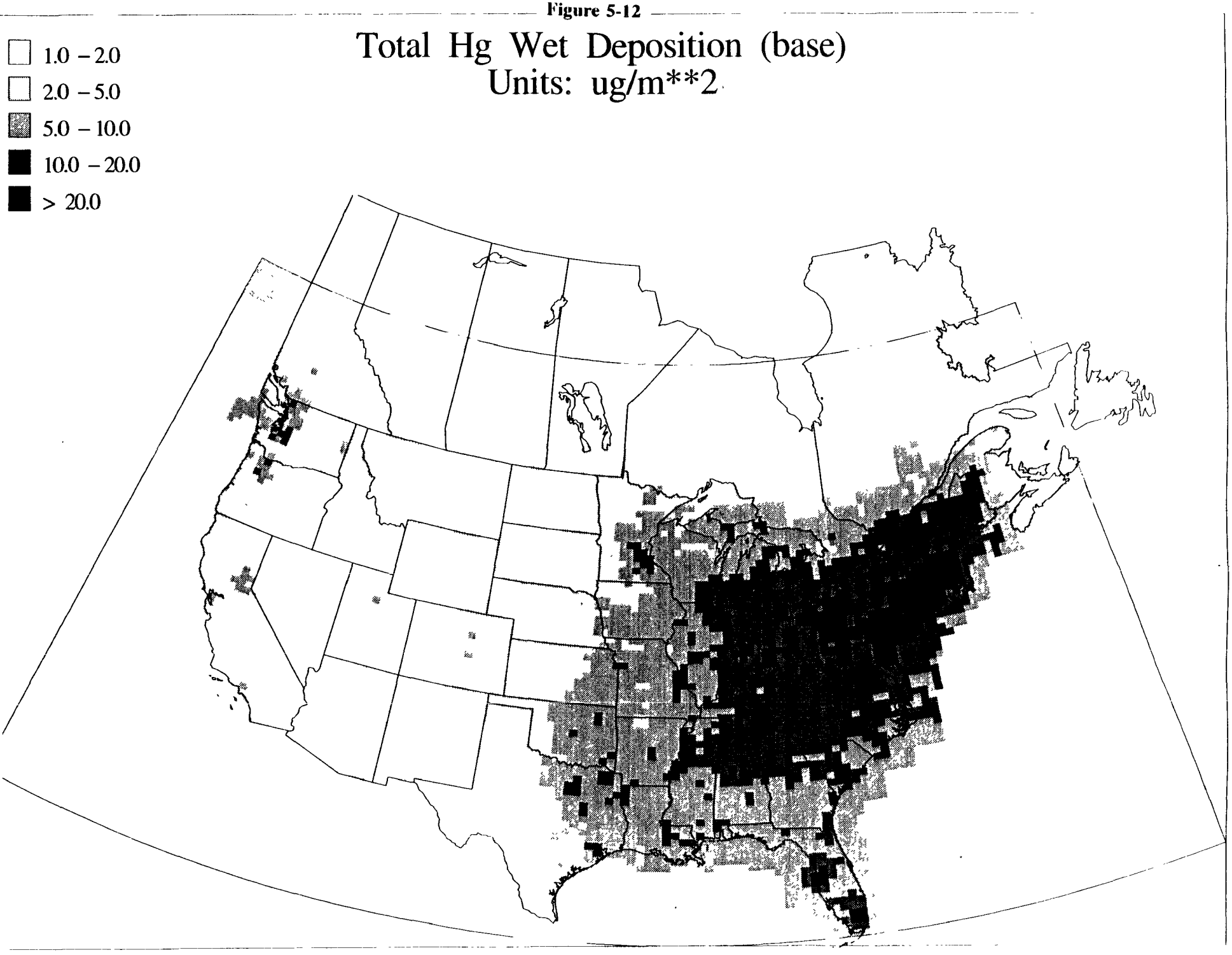


Figure 5-12


Total Hg Wet Deposition (base)
Units: ug/m**2
1.0 – 2.0
2.0 – 5.0
5.0 – 10.0
10.0 – 20.0
> 20.0

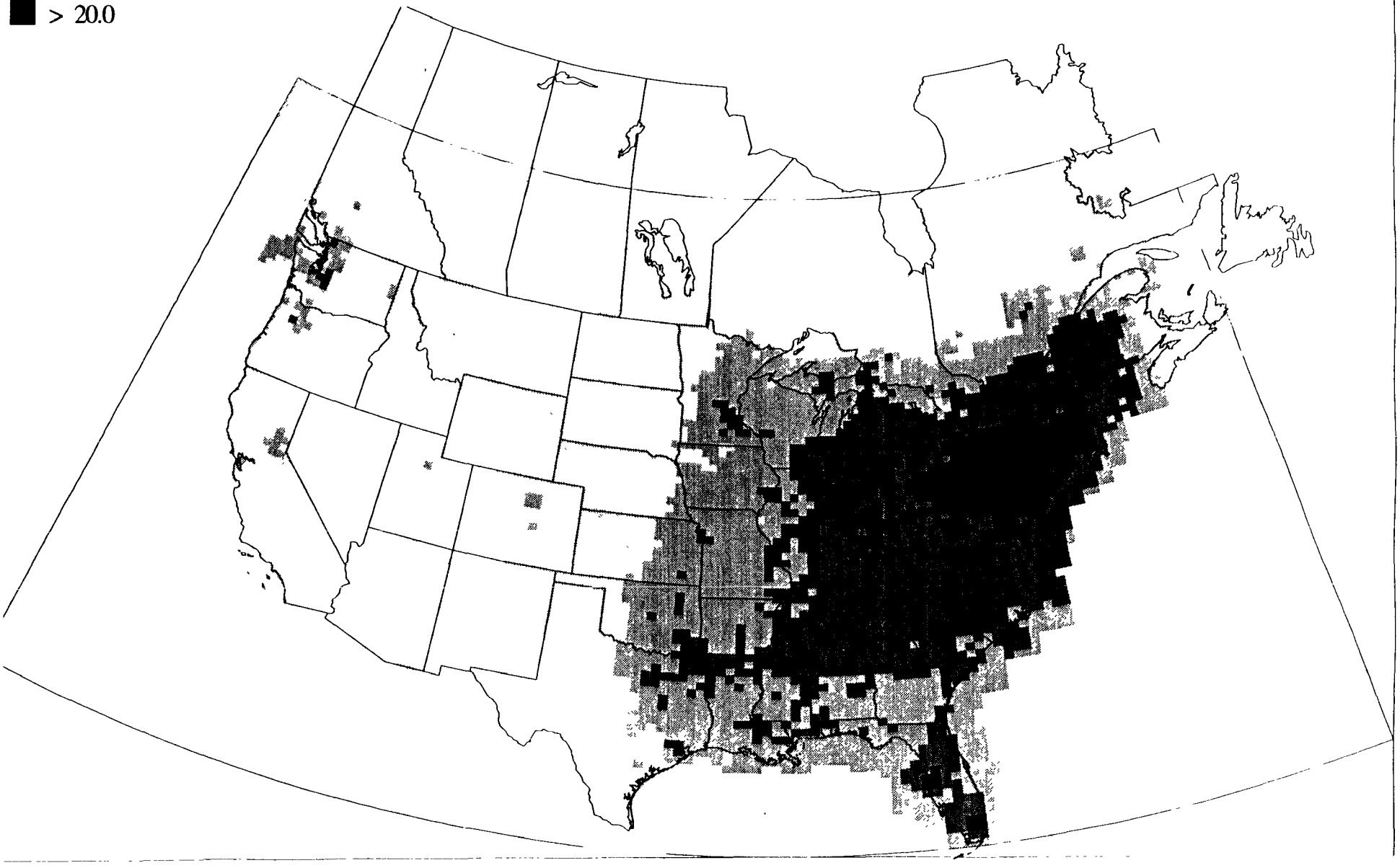
There were also some mercury wet deposition measurement programs conducted during the early 1990's in somewhat less remote sites in Michigan and Vermont. Observations by Hoyer et al. (1995) during two years of event precipitation sampling at three sites in Michigan show evidence for a north-to-south gradient in mercury wet deposition. From March 1992 to March 1993, the total mercury wet deposition observed at South Haven, in southwest Michigan, was 9.45 $\mu g/m^2$. At Pellston, in the northern part of the lower peninsula of Michigan, the wet deposition was 5.79 $\mu g/m^2$. At Dexter, in southeast Michigan about 100 km west of Detroit, the wet deposition was 8.66 $\mu g/m^2$. From March 1993 to March 1994, wet deposition at South Haven was 12.67 $\mu g/m^2$, significantly higher than for the previous year, while measurements at Pellston and Dexter remained about constant at 5.54 and 9.11 $\mu g/m^2$, respectively. Hoyer et al. (1995) attribute the higher second-year wet deposition at South Haven to an increased precipitation rate and cite the measurements by Burke et al (1995) at Underhill, Vermont, as further evidence of the importance of precipitation amount. From December 1992 to December 1993, the average volume-weighted mercury concentration at Underhill (8.3 ng/L) was similar to that observed at Pellston (7.9 ng/L). However, with more precipitation during that period the total mercury wet deposition at Underhill was 9.26 $\mu g/m^2$, significantly higher than at Pellston. The RELMAP simulation results show 5-10 $\mu g/m^2$ wet deposition of total mercury at the Pellston site which agrees well with the 1992 to 1994 observations there. At Underhill, the RELMAP simulation indicates 10-20 $\mu g/m^2$ wet deposition for 1989 which is slightly larger than the observation in 1993. At the South Haven and Dexter sites, the RELMAP appears to be estimating nearly 20 $\mu g/m^2$ wet deposition of total mercury for 1989 which is significantly larger than the measurements of 1992 to 1994.

The very large total mercury wet deposition values (>50 $\mu g/m^2$) from the RELMAP simulation for some of the larger urban centers in the Ohio Valley and Northeast regions cannot be evaluated thoroughly due to a lack of long-term precipitation event sampling at those locations. A study by Dvonch et al. (1995) describes precipitation event sampling from 19 August to 7 September of 1993 at 4 sites in Broward County, Florida, in and around the city of Fort Lauderdale. During the 20-day sampling period, total mercury mean concentrations in precipitation were 35, 57, 40 and 46 ng/L at the 4 sites. Given the average annual precipitation of 150 cm per year typical of that area, the resulting annual wet deposition estimates at these 4 sites would 52.5, 85.5, 60 and 69 $\mu g/m^2$. Since most of the annual rainfall in Broward County occurs in warm tropical conditions of the March to October wet season, this extrapolation from 20 days during the wet season to an annual estimate is inappropriate. Additional urban measurement studies are required to allow any credible evaluation of RELMAP wet deposition results in heavily populated, industrialized area.

Figure 5-13 shows the wet deposition of total mercury from the alternate emission speciation and offers some measure of the sensitivity of the RELMAP simulation results to the emission speciation estimates used. This figure shows basically the same large-scale pattern as for the base-case emissions, but in general the amount of wet deposition is increased. Since the wet scavenging ratio for Hg_P is less than one-third of that for Hg^{2+}, it would be expected to be reduced when using the alternate emission speciation which reallocate all Hg^{2+} vapor emissions to the Hg_P form. This sensitivity is the result of an interaction of the wet deposition processes with those for dry deposition. As discussed in the next sub-section of this report, the alternate emission speciation profiles result in greatly reduced dry deposition compared to the base-case. This results in very little depletion of pollutant puffs by dry deposition, and provides a longer opportunity for wet deposition to occur. The slower dry deposition of Hg_P versus Hg^{2+} vapor more than offsets the effect of lower wet scavenging ratios for Hg_P versus Hg^{2+} vapor.

Figure 5-13

Total Hg Wet Deposition (alternate)
Units: ug/m**2



The percentile analysis of the wet deposition simulation results in Table 5-6 show that for the base-case emission speciation only 10 percent of the land area of the continental U.S. has an annual wet deposition of total Hg of more than about 16 $\mu g/m^2$, and 50 percent of the land area has less than 3.4 $\mu g/m^2$. However, due to rapid wet deposition of Hg^{2+} and Hg_P there are select areas where wet deposition may be significantly higher. In the eastern U.S., east of 90 degrees west longitude, the 50th and 90th percentile levels for total Hg wet deposition are considerably higher than those for the entire continental U.S., about 12 and 25 $\mu g/m^2$, respectively.

Table 5-6
Percentile Analysis of RELMAP Simulated Wet Deposition for the Continental U. S. Using the Base-Case Emissions Speciation

Variable	Min	10th	50th	90th	Max
Full Area					
Hg^{0a} wet dep. ($\mu g/m^2/yr$)	0.022	0.590	2.143	6.306	10.66
Hg^{2+b} wet dep. ($\mu g/m^2/yr$)	0.002	0.087	0.749	6.217	125.1
Hg_P^c wet dep. ($\mu g/m^2/yr$)	0.001	0.067	0.502	3.618	37.72
Total mercury ($\mu g/m^2/yr$)	0.025	0.792	3.365	15.85	173.5
East of 90°W longitude					
Hg^{0a} wet dep. ($\mu g/m^2/yr$)	0.540	3.099	5.382	7.406	10.66
Hg^{2+b} wet dep. ($\mu g/m^2/yr$)	0.242	1.837	4.269	12.40	125.1
Hg_P^c wet dep. ($\mu g/m^2/yr$)	0.191	1.252	2.607	6.482	37.72
Total mercury ($\mu g/m^2/yr$)	0.979	6.846	12.40	25.42	173.5
West of 90°W longitude					
Hg^{0a} wet dep. ($\mu g/m^2/yr$)	0.022	0.512	1.337	3.995	7,854
Hg^{2+b} wet dep. ($\mu g/m^2/yr$)	0.002	0.067	0.313	1.767	13.93
Hg_P^c wet dep. ($\mu g/m^2/yr$)	0.001	0.050	0.253	1.146	6.550
Total mercury ($\mu g/m^2/yr$)	0.025	0.686	1.993	6.936	23.87

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor-phase Mercury
[c] Hg_p = Particle-Bound/Mercury

5.2.4 Qualitative Description of Mercury Dry Deposition Results

As described in the section on the RELMAP mercury model parameterizations, it was assumed that Hg^0 was not effectively dry deposited due to its high vapor pressure and very low water solubility at normal atmospheric temperatures. Therefore, only Hg^{2+} vapor and Hg_P were dry deposited using the base emission speciation, and only Hg_P using the alternate emission speciation. The percentile analysis of the simulated dry deposition using the base-case emission speciation profiles shown in Table 5-7 indicates the strong local dry deposition of Hg^{2+} vapor as

Table 5-7
Percentile Analysis of RELMAP Simulated Dry Deposition for the Continental U. S. Using the Base-Case Emission Speciation

Variable	Min	10th	50th	90th	Max
Full Area					
Hg^{2+a} dry dep. ($\mu g/m^2/yr$)	0.113	0.412	1.641	8.500	153.5
Hg_P[b] dry dep. ($\mu g/m^2/yr$)	0.002	0.010	0.035	0.130	0.749
Total mercury ($\mu g/m^2/yr$)	0.117	0.425	1.669	8.629	154.2
East of 90°W longitude					
Hg^{2+a} dry dep. ($\mu g/m^2/yr$)	0.434	2.649	6.263	15.53	153.5
Hg_P[b] dry dep. ($\mu g/m^2/yr$)	0.017	0.049	0.104	0.189	0.749
Total mercury ($\mu g/m^2/yr$)	0.451	2.699	6.373	15.73	154.2
West of 90°W longitude					
Hg^{2+a} dry dep. ($\mu g/m^2/yr$)	0.113	0.342	0.923	3.614	29.85
Hg_P[b] dry dep. ($\mu g/m^2/yr$)	0.002	0.009	0.024	0.063	0.236
Total mercury ($\mu g/m^2/yr$)	0.117	0.352	0.948	3.679	30.03

[a] Hg^{2+} = Divalent Vapor-phase Mercury
[b] Hg_p = Particle-Bound/Mercury

parameterized in the RELMAP mercury model. There is considerable uncertainty regarding the dry deposition velocity of Hg^{2+} and in the extremely high local depositions indicated from the simulation.

Figure 5-14 shows the simulated annual dry deposition totals for Hg^{2+} using the base emission speciation. Dry deposition of Hg^{2+} appears to occur primarily on the local scale, within one or two grid cells from the source (40-80 km), much like the wet deposition. The magnitude of the dry deposition of Hg^{2+} is similar to that for wet deposition, with urban areas showing values in excess of 20 $\mu g/m^2$. As was the case for wet deposition, dry deposition of Hg^{2+} vapor in heavily populated urban centers is very intense, exceeding 100 $\mu g/m^2$ in the model grid cell containing New York City. Again, it must be stressed that dry deposition of Hg^{2+} vapor is not well understood. The simulation used nitric acid vapor data as a surrogate for Hg^{2+} vapor based on similar water solubilities. The Agency has been unable to find observations of the dry deposition of Hg^{2+} vapor with which to compare to the RELMAP simulation results. Dry deposition rates for vapor-phase Hg have been estimated from vertical eddy flux calculations at a single site (Lindberg et al., 1992), but these calculations estimate the combined effects of both Hg^0 and Hg^{2+} vapors. The relatively high solubility and reactivity of Hg^{2+} compounds suggests that dry deposition of total vapor-phase mercury may be strongly driven by the fractional of Hg^{2+} in the total vapor-phase mercury concentration.

Figure 5-14

Hg(++) Dry Deposition (base)
Units: ug/m**2

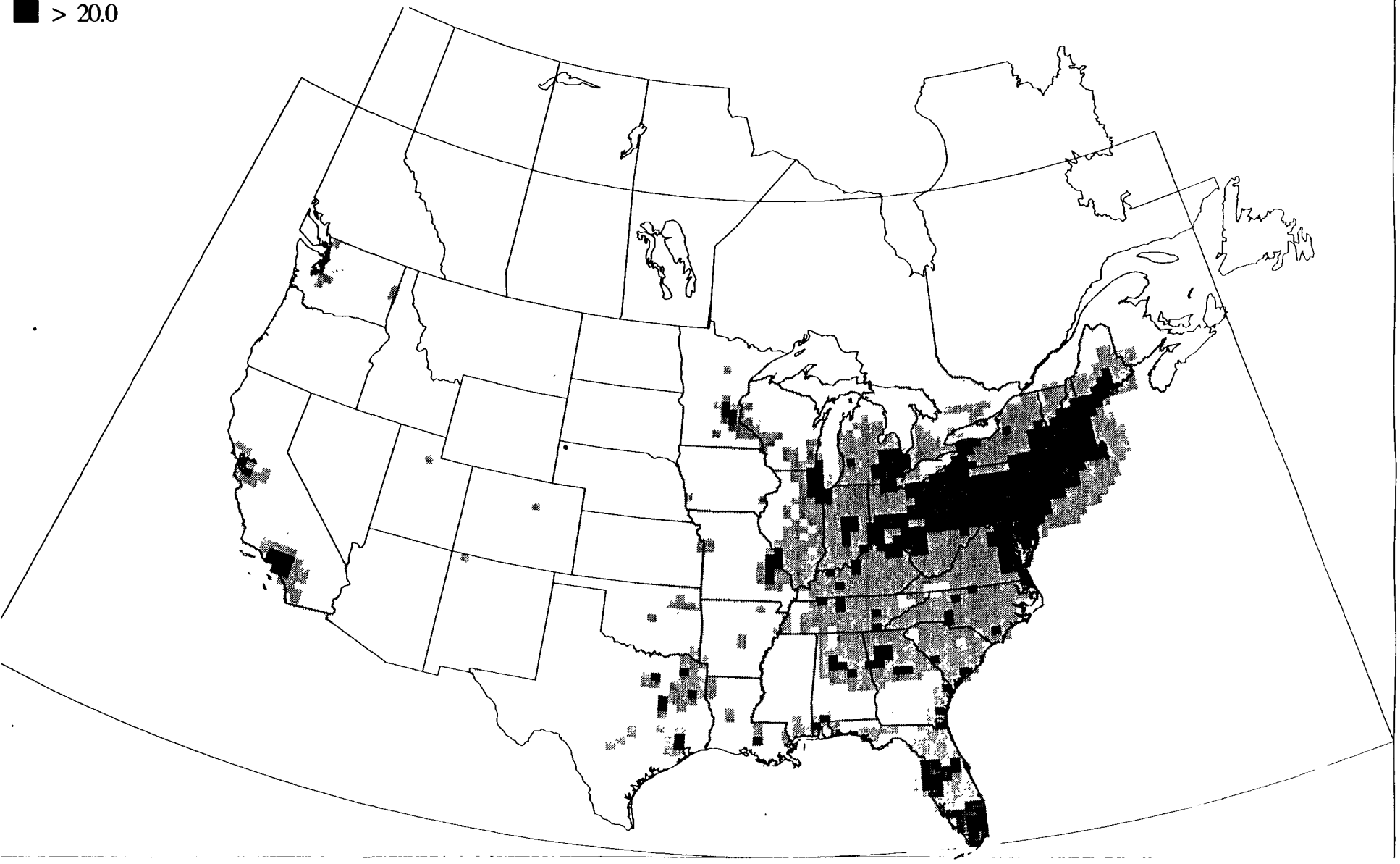


Figure 5-15 shows the simulated annual dry deposition totals for Hg_P using the base emission speciation. As described in Appendix D, the dry deposition velocity estimates for Hg_P have been made based on the assumption that the particulate mass is concentrated around a 0.3 μm diameter size. The patterns show less intense local dry deposition of Hg_P than for Hg^{2+}, but the dry deposition still appears to occur primarily within a few hundred km of the source areas. This slower dry deposition combined with relatively smaller quantities of Hg_P emission result in maximum dry deposition values of only around 0.5 $\mu g/m^2$. In urban areas where larger particle sizes are more prevalent, these estimates of Hg_P dry deposition are probably too low, but the RELMAP could treat only one particle size. Since the focus of this modeling was on the regional scale, 0.3 μm was chosen as the most appropriate diameter size.

Figures 5-16 and 5-17 show the simulated annual dry deposition for all forms of mercury using the base-case and alternate emission speciations, respectively. A comparison of these figures clearly demonstrates the sensitivity of the modeling results to changes in the emission speciation profiles. Dry deposition is not a major pathway for removal of the atmospheric mercury burden when the alternate emission speciation profiles are employed. This result also indicates that dry deposition would be much less important if significant transfer of Hg^{2+} to Hg_P is occurring through particle adsorption or condensation. Thus, it is very important that our understanding of the physical transformations of Hg in the atmosphere be complete and accurate.

5.2.5 Qualitative Description of Total Mercury Deposition Results

Since both wet and dry deposition of mercury can affect human and ecosystem health, an analysis of the simulated total deposition of all forms of mercury has been performed for both the base-case and alternate emission speciations. Table 5-8 shows a percentile analysis of total deposition of mercury in all modeled forms. The strong bias toward mercury deposition in the eastern U.S. is immediately obvious. Also obvious is the order of magnitude difference between the 90th percentile level and the maximum values in the nationwide and eastern U.S. analyses. The extremely high simulated deposition totals over heavily populated urban centers cannot be substantiated by observations at this time. Due to the high degree of uncertainty regarding the emission speciations and possible rapid chemical and physical transformations immediately after emission, it is recommended that these maximum simulated deposition values should be considered highly uncertain until further research is conducted to reduce these uncertainties.

Figure 5-18 shows the base-case total deposition of mercury to the Earth's surface from the RELMAP simulation. These results show deposition totals of over 20 $\mu g/m^2$ throughout most of the Northeast and Ohio Valley regions and in various urban areas nationwide, with values over 50 $\mu g/m^2$ for the northeast corridor and at other major urban centers. Figure 5-19 shows the total mercury deposition results using the alternate emission speciations. For the alternate case, the areas with simulated deposition greater than 20 $\mu g/m^2$ are noticeably reduced, and the alternate case results show few areas with annual mercury deposition totals over 50 $\mu g/m^2$. The sensitivity of the model deposition results to uncertainties in the emission speciation profiles appears to be greatest in areas of high mercury deposition.

Figure 5-1

Hg(p) Dry Deposition (base)
Units: ug/m**2

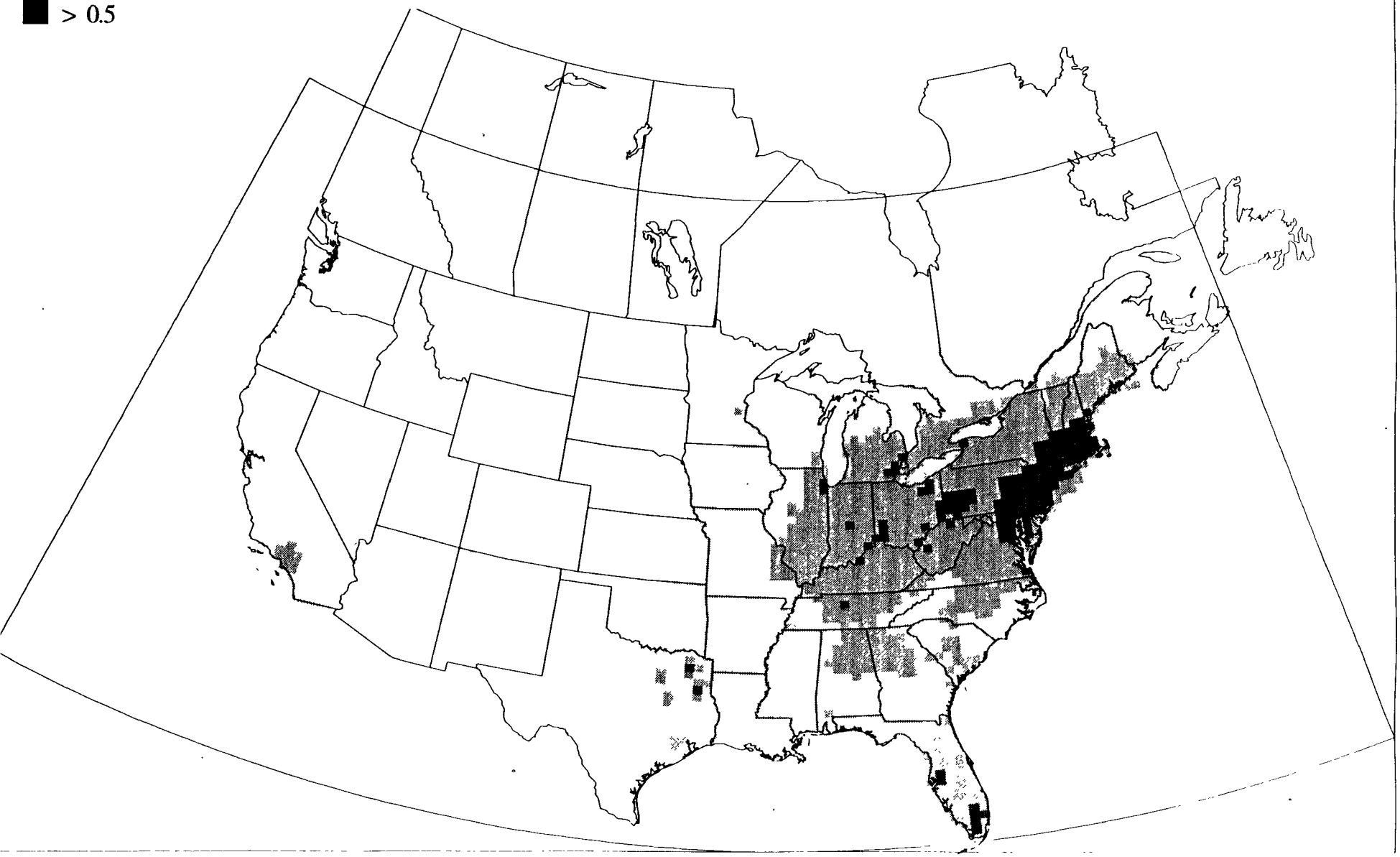


Figure 5-16

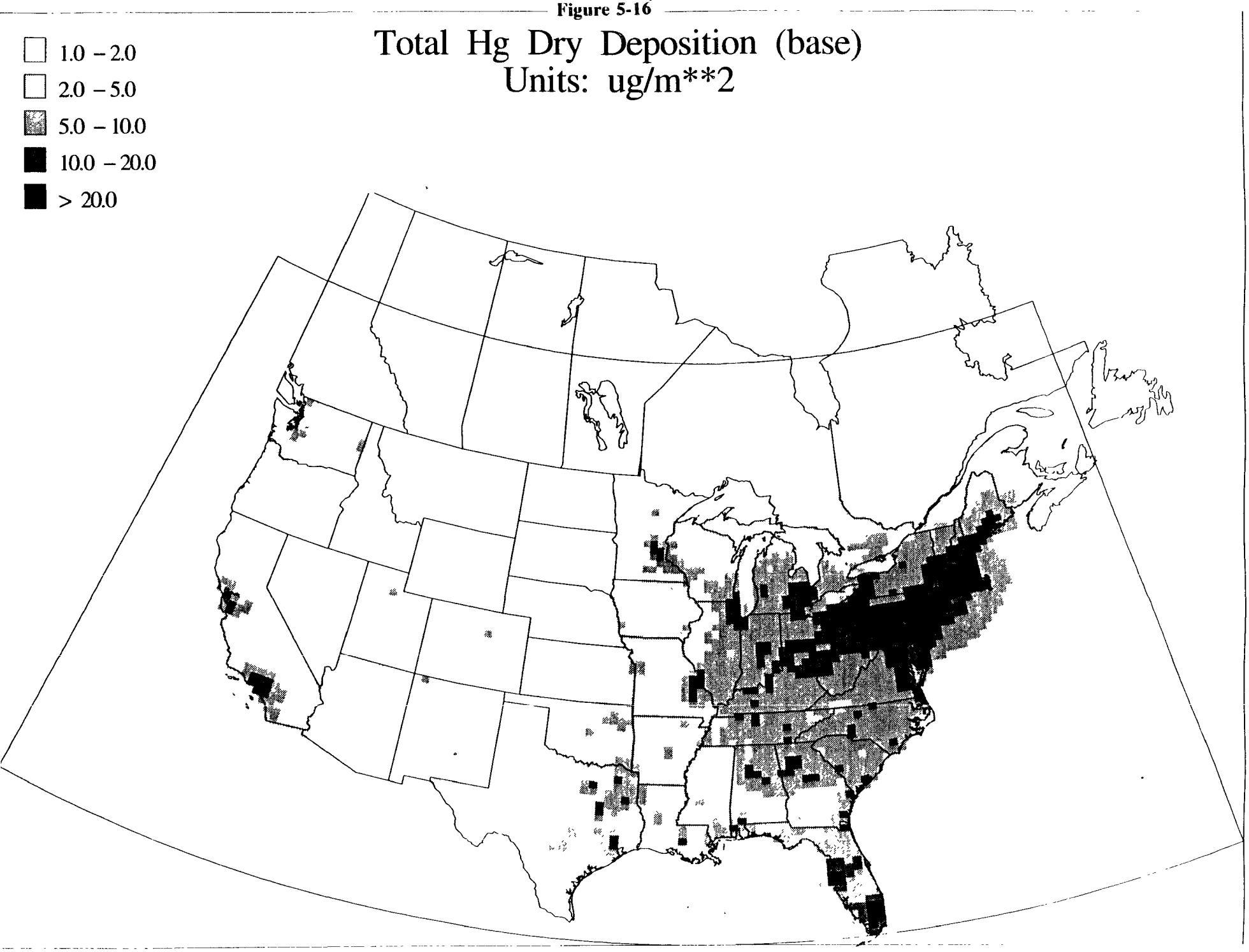

Total Hg Dry Deposition (base)
Units: ug/m**2
1.0 – 2.0
2.0 – 5.0
5.0 – 10.0
10.0 – 20.0
> 20.0

Figure 5-17

Total Hg Dry Deposition (alternate)
Units: ug/m**2

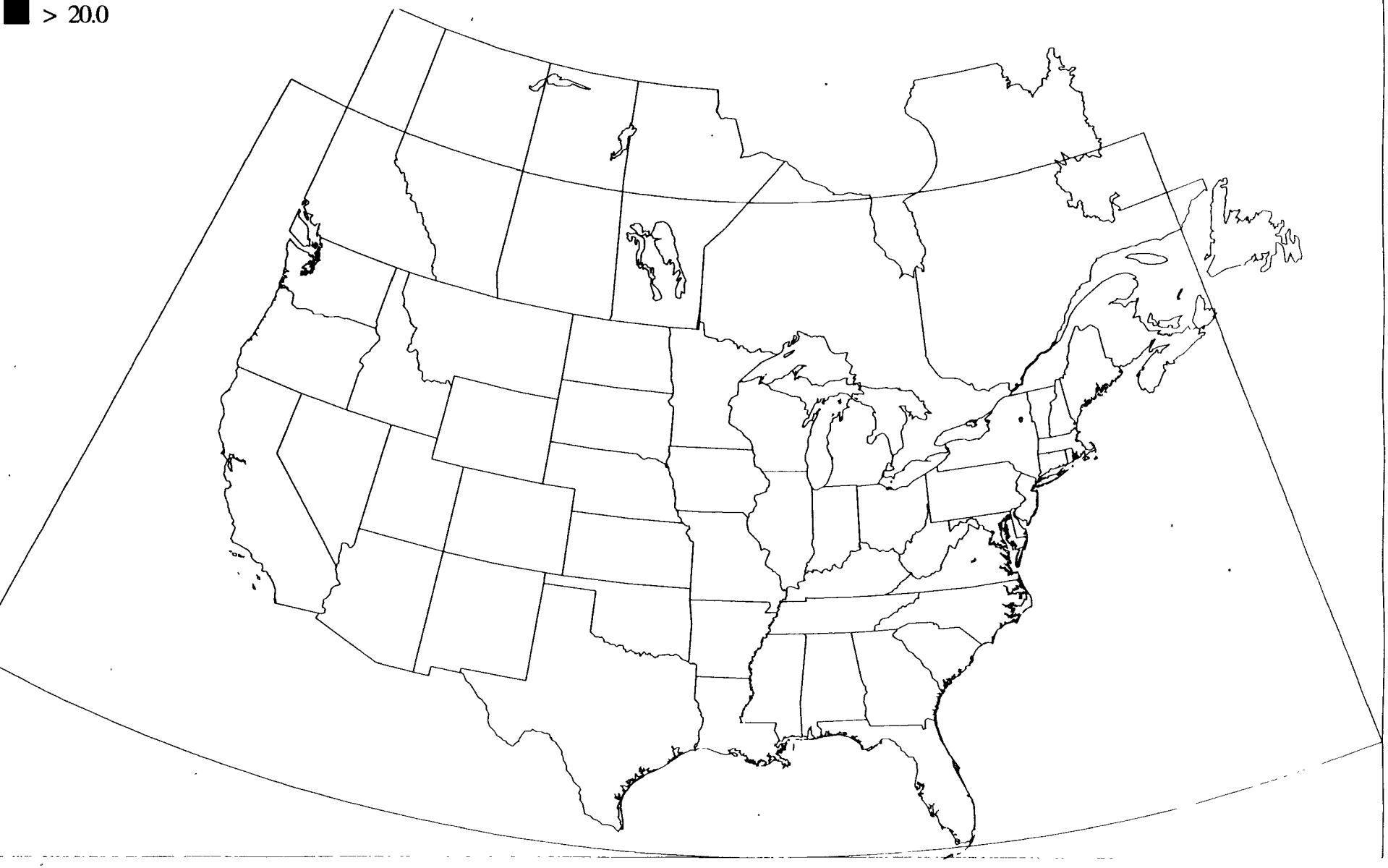


Table 5-8
Percentile Analysis of RELMAP Simulated Total Depositions for the Continental U. S. Using the Base-Case and Alternate Emission Speciations

Variable	Min ($\mu g/m^2/yr$)	10th ($\mu g/m^2/yr$)	50th ($\mu g/m^2/yr$)	90th ($\mu g/m^2/yr$)	Max ($\mu g/m^2/yr$)
Full Area					
Base-Case	0.538	1.470	5.133	24.59	327.6
Alternate	0.122	0.922	3.796	18.12	151.0
East of 90°W longitude					
Base-Case	1.717	9.837	18.88	39.97	327.6
Alternate	1.256	8.252	14.65	28.67	151.0
West of 90°W longitude					
Base-Case	0.538	1.251	2.941	10.58	49.21
Alternate	0.122	0.800	2.303	7.916	23.35

5.3 General Data Interpretations of the RELMAP Modeling

At this time there is significant uncertainty regarding the chemical and physical forms of emissions and their chemical and physical transformations in the atmosphere. This long-range modeling effort did not provide for a complete new model development and evaluation; the modeling effort has relied heavily on the assumptions of Petersen et al. (1995) regarding emission speciation and chemical and physical pathways for mercury deposition. The model parameterizations for the Lagrangian puff model were developed under the European Monitoring and Evaluation Programme (EMEP). These mercury modeling results were compared to measurements of Hg^0 and Hg_p air concentration and wet deposition in northern Europe. The comparison showed their model results agreed with measurements to within a factor of 2 in nearly all cases. While the climate of northern Europe may be quite different from that of some locations in North America, it has been assumed that the predominant chemical and physical mechanisms for mercury transport, transformation and deposition should be the same for both regions.

The wet deposition results from the RELMAP simulation of atmospheric mercury seem to agree with actual measurements within a factor of 2 in most cases. The RELMAP estimates of more than 100 $\mu g/m^2$ wet deposition using the meteorologic data for the year 1989 in the grid cell containing New York City seems a bit too high, but there are very few measurements which can be compared to these results. The RELMAP dry deposition results indicate that the importance of dry versus wet deposition processes may be dependent on the fraction of emitted mercury that eventually becomes particle-bound before deposition. Very few direct measurements of the dry deposition of gaseous and particulate Hg have been made to date. Vertical concentration

Figure 5-18

Total Hg Wet+Dry Deposition (base)
Units: ug/m**2

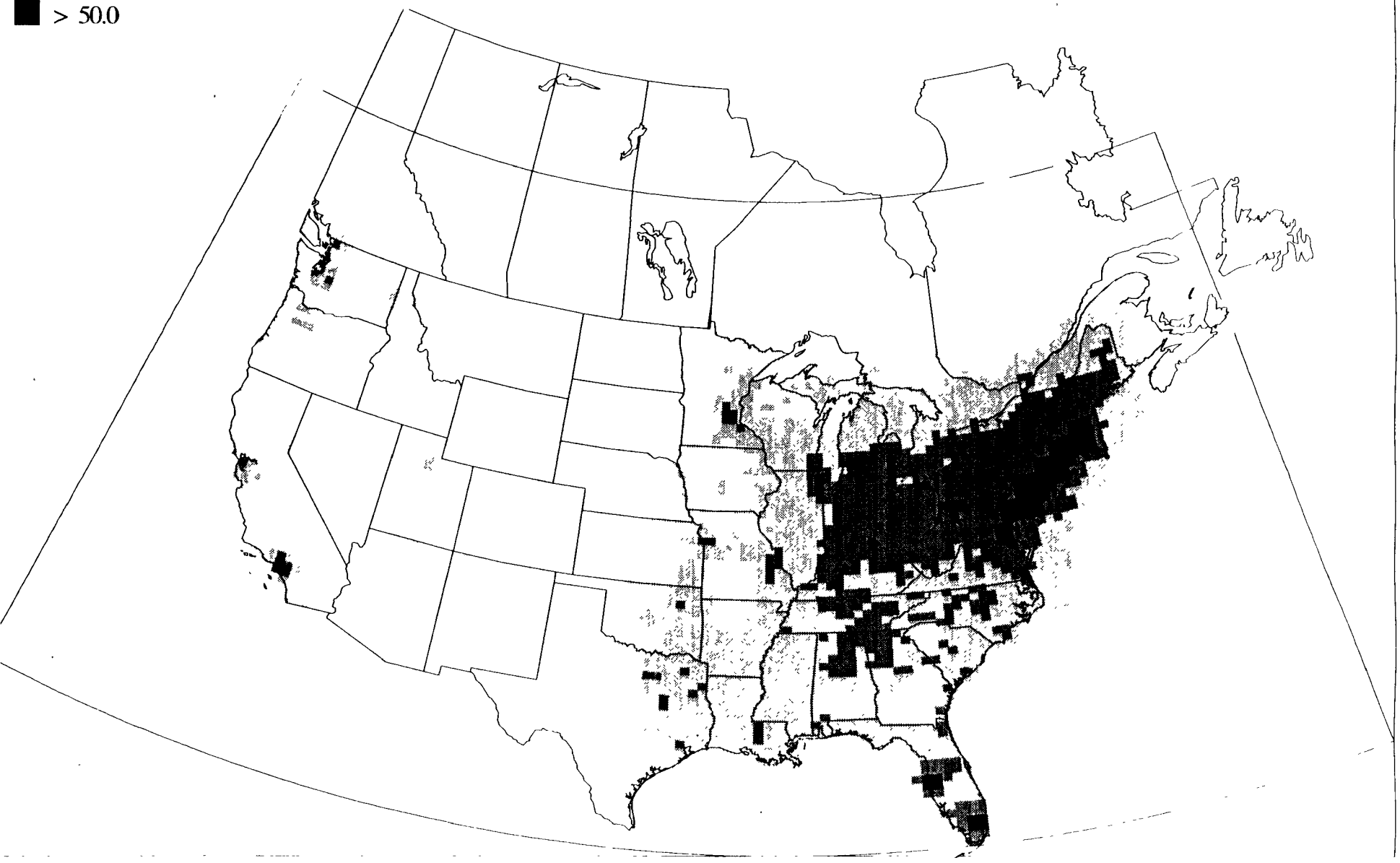
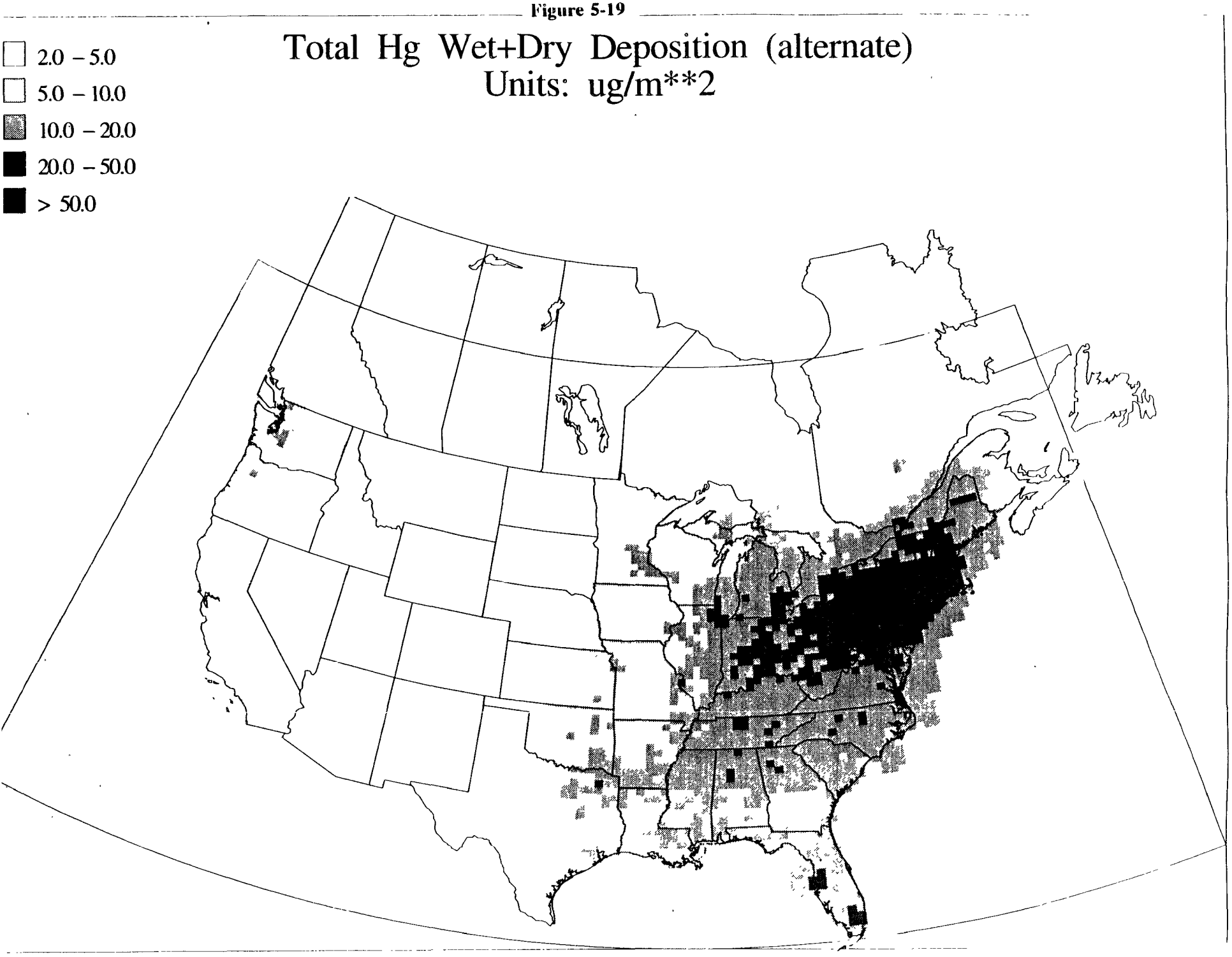


Figure 5-19


Total Hg Wet+Dry Deposition (alternate)
Units: ug/m**2
2.0 – 5.0
5.0 – 10.0
10.0 – 20.0
20.0 – 50.0
> 50.0

gradients and eddy flux correlations have been used to estimate the dry flux of total gaseous mercury by Lindberg et al. (1992), but no discrimination was made between Hg^0 and Hg^{2+} forms.

Many of the measurement studies performed up until the 1980's are now suspected of having been subject to laboratory contamination. It is only recently that, by employing ultra-clean laboratory techniques, mercury measurement studies have been able to assess accurately atmospheric concentrations and deposition quantities of mercury in near-background conditions. Even now, it is very difficult to obtain an accurate assessment of the chemical forms of mercury in typical ambient air samples. The RELMAP air concentration results seem quite plausible, with the vast majority of atmospheric mercury estimated to be in the elemental vapor form, but the precise concentrations of Hg^{2+} and Hg_p cannot be simulated with much confidence until a more complete understanding is established of all pertinent chemical and physical processes in the atmosphere.

There are some limitations of the RELMAP and other Lagrangian puff models that may negatively affect the accuracy of atmospheric mercury modeling. The simulated pollutant puff must move as an integral volume, and differences in wind direction or speed at various heights above the surface are not treated. The pollutant puff is currently simulated with a predefined vertical top, through which turbulent exchanges of air and pollutants are set at any arbitrary value. For pollutants such as Hg^0 that remain in the atmosphere for a long time, significant transfer of mass between the PBL and the rest of the atmosphere is inevitable. These exchanges can be attributed not only to turbulent processes but also larger-scale vertical atmospheric motions, both rising and sinking. Finally, Lagrangian puff models have no straightforward way to treat the horizontal boundary flux of pollutant into the model domain. Hg^0 vapor is known to be transported in the atmosphere on a global scale, but adequate methods are unavailable to model its transport from other parts of the earth into the model domain. The U.S. EPA is working to develop a general purpose air-quality model employing an Eulerian reference frame which should prove more suitable for mercury transport and deposition simulations; completion of this model is still a year or more away. A more rigorous discussion of these and other scientific caveats is provided in Appendix D.

5.4 Potential Impacts of Long Range Transport

In this section the results of the regional air modeling are used in conjunction with the IEM2 model and the hypothetical exposure scenarios to predict environmental concentrations and exposure rates for humans and piscivorous wildlife. The 50th and 90th percentiles of the mercury air concentrations and annual deposition rates that are predicted by RELMAP for the western and eastern halves of the U.S. were used as inputs to the modeling. The potential impact of predicted atmospheric mercury concentrations and deposition rates associated with areas of the eastern U.S. considered to be remote from anthropogenic influences was also included in the long range transport analysis. See Figure 1-2.

The sites and the exposure scenarios utilized in the long range transport analysis have been discussed in Chapter 4 of this Volume and are briefly discussed here. The IEM2 model is described in Chapter 4 of this Volume and in Appendix D; only the inputs to and the results of the IEM2 modeling are described here.

5.4.1 Description of Hypothetical Sites and Watersheds

Two generic sites ~~are~~ WERE considered: a humid site east of 90 degrees west longitude, and a more arid site west of 90 degrees west longitude (these are described in Appendix B). The primary differences between the two sites as parameterized were the assumed erosion characteristics for the watershed and the amount of dilution flow from the water body. The eastern site had generally steeper terrain in the watershed than for the other site. A circular drainage lake with a diameter of 1.78 km and average depth of 5 m, with a 2 cm benthic sediment depth was modeled at both sites. The watershed area was 37.3 km^2.

In the long range transport analysis both the eastern and western sites were assumed to be impacted by mercury from only the RELMAP modeling. The 50th and 90th percentiles of the mercury air concentrations and annual deposition rates for the eastern and western halves of the U.S. were used as inputs to the IEM2 modeling for both sites. An additional eastern site with the same characteristics as previously described was assumed to be remote and only impacted by long range transport. The impacts of a separate atmospheric mercury concentration and annual deposition rate were modeled for this site.

5.4.2 Description of Hypothetical Exposure Scenarios

For the analyses that were conducted for this report, three types of settings were considered: rural (agricultural), lake, and urban. These three settings were selected because they encompass a wide variety and each is expected to provide a "high-end" mercury concentration in environmental media of concern for either human or wildlife species exposure (e.g., elevated mercury concentrations are expected in the waters of the lake setting).

Exposure to environmental mercury is the result of mercury concentrations at specific exposure points (e.g., ingested fish). For each setting in each location of the long range transport analysis, individuals representing several specific subpopulations were considered. For a more detailed description of the parameter values chosen for the exposure assessment see Appendix A. Table 5-9 summarizes the hypothetical scenarios considered as well as the exposure pathways considered in each scenario.

Table 5-10 shows the default values for the scenario-independent parameters for both the child and adult receptors, and Table 5-11 shows the default values for the scenario-dependent exposure parameters. The technical bases for these values are in Appendices A and B.

Both high-end and average rural scenarios were evaluated. The high-end scenario consisted of a subsistence farmer and child who consumed elevated levels of locally-grown food products. It was assumed that each farm was located on approximately 10 acres. The subsistence farmer was assumed to raise livestock and to consume home-grown animal tissue and animal products, including chickens and eggs as well as beef and dairy cattle. All chicken feed was assumed to be derived from non-local sources. For bovine consumption of contaminated feed, 100% of the hay and corn used for feed was assumed to be from the affected area. It was also assumed that the subsistence farmer collected rainwater in cisterns for drinking. The average rural home gardener was assumed to consume garden fruits and vegetables. There was no consumption of locally-raised animals or locally-collected water.

In the urban high end scenario, it was assumed that the person had a small garden similar in size to that of the average rural scenario. To address the fact that home-grown fruits and

Table 5-9
Summary of Human Exposure Scenarios Considered

	Location											
	Rural			Urban			Lacustrine			Remote Lakes[a]		
	Subsistence Farmer		Home Gardener	Resident	Worker/High-end		High End fisher		Rec. angler	High End fisher		Rec. angler
	Adult	Child	Adult	Adult	Adult	Pica Child	Adult	Child	Adult	Adult	Child	Adult
Air inhalation	X	X	X	X	X	X	X	X	X	X	X	X
Soil Ingestion	X	X	X	X	X	X	X	X	X	X	X	
Animal ingestion	X	X										
Vegetable ingestion	X	X	X		X	X	X			X		
Fish ingestion							X	X	X	X	X	X
Water ingestion	X	X					X	X	X			

Notes:

[a] Lakes located greater than 50 km from a mercury emission source.

Blank = Pathway not considered.

X = Pathway considered.

Table 5-10
Default Values of Scenario-Independent Exposure Parameters
All human consumption rates except for soil and water are reported as dry weight.

Parameter	Default Value[a]	
	Adult	Child
Body weight (kg)	70	17
Exposure duration (years)	30	18
Inhalation rate (m^3/day)	20	16
Vegetable consumption rates (g/kg BW/day)		
Leafy vegetables	0.028	0.008
Grains and cereals	1.87	3.77
Legumes	0.381	0.666
Potatoes	0.17	0.274
Fruits	0.57	0.223
Fruiting vegetables	0.064	0.12
Animal Product Consumption rates (g/kg BW/day)		
Beef (excluding liver)	0.341	0.553
Beef liver	0.066	0.025
Dairy	0.599	2.04
Pork	0.169	0.236
Poultry	0.111	0.214
Eggs	0.093	0.073
Lamb	0.057	0.061
Soil Ingestion rates (g/day)	0.1	Scenario-dependent
Water ingestion rate (L/day)	2	1

[a] See Appendix A for details regarding these parameter values.

Table 5-11
Default Values for Scenario-Dependent Exposure Parameters[a]

Parameter	Rural Subsistence Farmer		Rural Home Gardener	Urban Scenarios			High End Fisherperson		Recreational Angler
	Adult	Child	Adult	Adult Resident	Home Gardener	Pica Child	Adult	Child	Adult
Fish Ingestion rates (g/day)	NA[c]	NA	NA	NA	NA	NA	60	20	30
Soil Ingestion Rate (g/day)	0.1	0.2	0.1	0.1	0.1	7.5	0.1	0.2	0.1
Contact time for inhalation (hr/day)	24	24	24	16	24	24	24	24	24
Contact fractions (unitless)									
Animal products	1	1	NA	NA	NA	NA	NA	NA	NA
Leafy vegetables	1	1	0.058	NA	0.026	NA	0.058	0.058	NA
Grains and cereals	1	1	0.667	NA	0.195	NA	0.667	0.667	NA
Legumes	1	1	0.8	NA	0.5	NA	0.8	0.8	NA
Potatoes	1	1	0.225	NA	0.031	NA	0.225	0.225	NA
Fruits	1	1	0.233	NA	0.076	NA	0.233	0 233	NA
Fruiting vegetables	1	1	0.623	NA	0 317	NA	0.623	0.623	NA
Root vegetables	1	1	0.268	NA	0.073	NA	0.268	0.268	NA
Drinking water[b]	1	1	NA	NA	NA	NA	1	1	1

[a] See Appendix A for more details regarding these values.

[b] The source of the contaminated drinking water is different for the subsistence farmer and fisherperson scenarios.

[c] NA - Not Considered to be Applicable to this assessment. For example, urban residents were assumed to eat no locally caught fish. Any fish ingested by this subpopulation was considered to be contaminated by mercury from outside the modeling domain and thus not considered.

vegetables generally make up a smaller portion of the diet in urban areas, the contact fractions were based on weight ratios of home-grown to total fruits and vegetables consumed for city households. These fractions can be up to 10 times smaller than the values for rural households, depending on food plant type (see Table 5-11 and Appendix A). Exposure duration for inhalation was 24 hours per day. Also considered as part of the high-end urban scenario was a pica child. The average urban scenario consisted of an adult who worked outside of the immediate area. The exposure duration for inhalation, therefore, was only 16 hours a day compared to the 24 hours a day for the rural and high-end urban scenarios. The only other pathway considered for this scenario was ingestion of average levels of soil.

Three human fish consumption scenarios were associated with the hypothetical lake: (1) an adult high-end fish consumer scenario, in which an individual was assumed to ingest large amounts of locally-caught fish as well as home-grown garden produce (plant ingestion parameters identical to the rural home gardener scenario) and consume drinking water from the affected lake; (2) a child of a high-end local fish consumer, assumed to ingest local fish, garden produce, and soil as well as inhale the air; and (3) a recreational angler scenario, in which the exposure pathways evaluated were fish ingestion, inhalation, and soil ingestion. These consumption scenarios were thought to represent identified fish-consuming subpopulations in the U.S. No commercial distribution of locally caught fish was assumed Fish consumption rates for the three fish consuming subpopulations were derived from the Columbia River Inter-Tribal Fish Commission Report (1994) (See Table 5-12).

Table 5-12
Fish Consumption Rates used in this Study
(Columbia River Inter-Tribal Commission, 1994)

Subpopulation	Fish Consumption Rate (g/day)
High-end Adult	60
High-end Child	20
Recreational Angler	30

Five piscivorous species of birds and mammals are assumed to inhabit areas around the hypothetical lakes and be exposed to mercury through consumption of fish from the lakes. The piscivorous wildlife exposure scenarios are described in Chapter 4 of this Volume. Table 5-13 lists the assumed animal body weights, fish consumption rates and the trophic level of the fish consumed (U.S. EPA, 1993). Dermal, inhalation, and drinking water exposures for the wildlife species were not modeled in this assessment. Other food sources were also not considered in this assessment.

Table 5-13
Fish Consumption Rates for Piscivorous Birds and Mammals (from U.S. EPA, 1993)

Animal	Body Weight (kg)	Total Ingestion Rate (g/day)	% of Diet from Trophic Level 3 Fish	% of Diet from Trophic Level 4 Fish	% of Diet from Non-aquatic sources
Bald Eagle	4.6	500	74	18	8
Osprey	1.5	300	100	0	0
Kingfisher	0.15	75	100	0	0
River Otter	7.4	1220	80	20	0
Mink	.0.8	178	90	0	10

5.4.3 Results of IEM2 Modeling in the Long Range Transport Analysis

For each of the sites described above the IEM2 models were run using the atmospheric mercury concentrations and the annual deposition rates predicted by RELMAP as inputs. The predicted mercury concentrations in soil, water and fish are presented in Table 5-14, and the predicted intakes are presented in subsequent tables.

The predicted concentrations in soil, water and fish were in the range of measured values reported in Chapter 2 of this volume. The predicted soil concentrations at the "two" eastern sites and the western sites range from 5.8 to 88 ng/g. The predicted concentrations are the result of

Table 5-14
Predicted Media Concentrations using RELMAP Results Only in Conjunction with IEM2

	Total Mercury Air Concentration (ng/m3)	Total Mercury Deposition Rate (µg/m2/yr)	Total Mercury Untilled Soil Concentration (ng/g)	Total Mercury Surface Water Concentration (ng/l)	Predicted Methylmercury Fish Concentration (ug/g)	
					Trophic Level 3	Trophic Level 4
Remote Site in East	1.7e+00	4.5e+01	8.8e+01	2.2e+00	1.0e-01	5.1e-01
Eastern Site						
50th Percentile	1.7e+00	1.9e+01	3.7e+01	9.1e-01	4.3e-02	2.2e-01
90th Percentile	1.8e+00	4.0e+01	7.8e+01	1.9e+00	9.0e-02	4.5e-01
Western Site						
50th Percentile	1.6e+00	2.9e+00	5.8e+00	1.7e-01	8.5e-03	4.3e-02
90th Percentile	1.7e+00	1.1e+01	2.1e+01	5.6e-01	2.9e-02	1.4e-01

the modeled mercury deposition rates based on the RELMAP simulation that used meteorologic data from the year 1989. Sites subjected to higher deposition rates, such as the remote eastern site and the

90th percentile site in the east have higher deposition rates than the other sites. The predicted soil concentration in the 50th percentile eastern site was higher than the 90th percentile western site: again. this is a direct result of deposition estimated in the RELMAP simulation.

The predicted concentrations in the surface waters were primarily the result of transport from the watershed soils and direct deposition onto the water body. The predicted mercury concentrations in the water bodies. which range from 0.17 to 2.2 ng/L, were comparable to mercury concentrations measured in surface waters by several authors; these are reviewed in Chapter 2 of this Volume. Again, as evidenced in Table 5-14, the higher the deposition rate the greater the predicted surface water concentrations. Similarly, the concentration of dissolved mercury in the water body depends in this modeling effort largely on the total amount of mercury deposited. As a result, the predicted methylmercury concentrations in fish, which are the product of the total dissolved mercury concentration in the water body and the methylmercury bioaccumulation factor, are heavily influenced by the total annual deposition rate. The highest concentrations are predicted in trophic level 4 fish in the remote east and 90th percentile eastern sites. The predicted methylmercury concentrations in fish, which range from approximately 0.03 ug/g to 0.5 ug/g, are clearly in the range of measured concentrations which extends up to approximately 9 ug/g. The average fish methylmercury concentrations reported in Bahnick et al., 1994 and Lowe et al., 1985 are 0.26 and 0.11 ug/g, respectively. Other studies have shown higher concentrations in fish in the eastern (particularly the northeastern) U.S. (Simonin et al., 1994; Mills et al, 1994 and NJDEPE, 1994).

The mercury concentrations predicted in soils, plants and animal products describe a range of mercury concentrations in items potentially ingested by humans based on the results of the RELMAP model. The mercury intakes presented in Chapter 3 provide a basis of comparison. The results predicted by the IEM2 model for human ingestion of mercury are comparable to estimates presented in recent reports, given the differences in consumption rates. No data exist to compare predicted wildlife mercury exposures. Note that the consumption rates and contact fractions of the various hypothetical scenarios are not designed to represent the general population, and all pathways not considered were assumed to result in no exposure to mercury.

Using the predicted air concentrations at the Western and two Eastern locations, the inhalation route was never predicted to be the dominant pathway of total mercury exposure, except for the urban average scenario. Because this scenario assumes no plant or animal consumption and exposure through dermal routes was not considered in this Report, the only mercury exposure routes are soil ingestion and inhalation. Low predicted concentrations in soil and low soil ingestion rates result in the prevalence of the inhalation route. The insignificance of exposure through the inhalation route when compared to ingestion routes was described previously by the WHO (WHO, 1990).

The results for the hypothetical adult rural subsistence farmer scenario are comparable to the general population exposure estimates presented in Chapter 3 since most sources of food for consumption are assumed to be derived from a mercury-impacted area. For the Remote Site analysis, "impacted sources" represent mercury concentrations in foods that may be comparable to concentrations in commercial foodstuffs. For the scenario, daily mercury ingestion levels for all locations were about 3.0 ug/day for adults, assuming a bodyweight of 70 kg. As noted in Chapter 4, most (approximately 90%) of this mercury was predicted to be the divalent species. For the agricultural scenarios, predicted mercury intake was dominated by consumption of plant products, and the air-to-plant pathway is the primary pathway that plants are predicted to take up mercury from the environment. As a result there is little difference between the results across sites because the air concentrations are similar.

The hypothetical high-end fisher has a high fish consumption rate but has lower contact fractions for food ingestion than hypothetical subsistence farmers. For purposes of this comparison to previous general population exposure estimates, the mercury intake from non-fish foodstuffs was underestimated. Total mercury ingestion rates for the high-end fishers across the three eastern locations vary from about 15 to 35 ug/day. These estimates are two to three times higher than the estimates presented in Chapter 3 for the general population. Predicted mercury ingestion rates for the hypothetical recreational angler range from about 6.5-15 ug/day. The difference between the total mercury ingestion rates predicted for the hypothetical recreational angler and the hypothetical high-end fisher is primarily due to the lower fish ingestion rate of the hypothetical recreational angler.

5.4.4 Summary of Potential Impacts of Long Range Transport

Air Modeling

The predicted air concentrations used for the two sites are at or slightly above the air concentration of 1.6 ng/m3 used as a representative air concentration in Chapter 4. The predicted deposition rates are higher for the eastern site, due to a combination of different climate conditions and the distribution of mercury emissions sources in the two geographic areas.

Environmental Media Modeling

The predicted soil and surface water body concentrations of mercury were most heavily influenced by the predicted total mercury deposition rate. The predicted soil and surface water body mercury concentrations were all within the range of measured mercury concentrations for these media. Differences between the hypothetical watersheds of the eastern and western sites account for the variation in water body loading from mercury deposited onto the watershed.

Biota Modeling

The mercury concentrations in green plants were the result of direct deposition and air-to-plant transfer onto exposed foliar surfaces, as well as soil-to-plant transfer. As modeled in this assessment, small amounts of mercury in green plants were predicted to result from direct deposition and soil-to-plant transfer. Most of the mercury predicted to occur in green plants was the result of air-to-plant transfer for those types of green plants for which this route of transfer was deemed appropriate. The predicted concentrations and speciation of mercury in green plants are within the range of measured concentrations. The predicted speciation of mercury in green plants was consistent with the reported values.

The mercury concentrations in all animal products except fish were predicted to be low. This was the result of generally low concentrations in plants and small plant-to-animal and soil-to-animal biotransfer factors. Few data are available for comparison.

Mercury concentrations in fish were predicted to be the highest of the biota considered. Mercury concentrations in fish were the product of the bioaccumulation factor and the dissolved concentration of mercury in surface water. The predicted mercury concentrations in fish were within the range of reported values. There is a great deal of uncertainty and variability associated with the uptake of mercury by fish.

Human and Wildlife Exposure Modeling

Human exposure to anthropogenic mercury was predicted to be dominated by indirect routes of exposure except for the hypothetical average urban dweller as indicated in Table 5-15 and 5-16. This individual was assumed to be exposed to emitted mercury from inhalation and soil ingestion only. For all other exposure scenarios, except those including fish consumption, the divalent species was predicted to be the primary species to which humans were exposed.

Those hypothetical humans who were assumed to consume fish had the highest exposures. This was a result of the bioaccumulation factor into fish. Methylmercury was the primary species to which these individuals were exposed. On a per body weight basis, children were predicted to be more exposed than adults.

The animals with the highest fish ingestion rate (per body weight) generally had the highest methylmercury intakes, except in cases where a piscivorous species was assumed to prey more heavily on trophic level 4 fish as indicated in Table 5-17.

Table 5-15
Summary of Receptor Intakes using RELMAP Eastern Site Values

	RelMap Eastern 50th percentile		RelMap Eastern 90th percentile		RelMap Remote	
	Inhalation (mg/kg BW/day)	Ingestion (mg/kg BW/day)	Inhalation (mg/kg BW/day)	Ingestion (mg/kg BW/day)	Inhalation (mg/kg BW/day)	Ingestion (mg/kg BW/day)
Rural Subsistence Farmer						
Child	1.6E-06	4.6E-05	1.7E-06	5.0E-05	1.6E-06	5.0E-05
Adult	4.8E-07	3.6E-05	5.0E-07	3.9E-05	4.9E-07	3.8E-05
Rural Home Gardener						
Child	1.6E-06	9.7E-06	1.7E-06	1.1E-05	1.6E-06	1.1E-05
Adult	4.8E-07	8.4E-06	5.0E-07	8.9E-06	4.9E-07	8.6E-06
Urban Average						
Child	1.6E-06	4.7E-07	1.7E-06	1.0E-06	1.6E-06	1.1E-06
Adult	3.2E-07	5.7E-08	3.3E-07	1.2E-07	3.2E-07	1.4E-07
Urban High End						
Child	1.6E-06	1.8E-05	1.7E-06	3.7E-05	1.6E-06	4.2E-05
Adult	4.8E-07	3.1E-06	5.0E-07	3.3E-06	4.9E-07	3.2E-06
Subsistence Fisherperson						
Child	1.6E-06	2.6E-04	1.7E-06	5.4E-04	1.6E-06	6.1E-04
Adult	4.8E-07	1.9E-04	5.0E-07	4.0E-04	4.9E-07	4.4E-04
Recreational Angler						
Adult	4.8E-07	9.3E-05	5.0E-07	1.9E-04	4.9E-07	2.2E-04

Table 5-16
Summary of Receptor Intakes using RELMAP Western Site Values

7/20/95, 5:39:15 PM		RelMap Western 50th percentile		RELMAP Western 90th Percentile	
		Inhalation (mg/kg BW/day)	Ingestion (mg/kg BW/day)	Inhalation (mg/kg BW/day)	Ingestion (mg/kg BW/day)
Rural Subsistence Farmer					
	Child	1.5E-06	4.2E-05	1.6E-06	4.6E-05
	Adult	4.6E-07	3.4E-05	4.7E-07	3.6E-05
Rural Home Gardener					
	Child	1.5E-06	8.9E-06	1.6E-06	9.3E-06
	Adult	4.6E-07	7.9E-06	4.7E-07	8.2E-06
Urban Average					
	Child	1.5E-06	7.3E-08	1.6E-06	2.7E-07
	Adult	3.1E-07	8.9E-09	3.1E-07	3.2E-08
Urban High End					
	Child	1.5E-06	2.7E-06	1.6E-06	9.9E-06
	Adult	4.6E-07	2.9E-06	4.7E-07	3.0E-06
Subsistence Fisherperson					
	Child	1.5E-06	5.9E-05	1.6E-06	1.8E-04
	Adult	4.6E-07	4.5E-05	4.7E-07	1.3E-04
Recreational Angler					
	Adult	4.6E-07	1.9E-05	4.7E-07	6.2E-05

Table 5-17
Predicted Intakes of Wildlife Receptors based on RELMAP Results

	Predicted Fish Concentrations (ug/g)		Predicted Intakes (mg/kg/day)				
	Trophic 3	Trophic 4	Bald Eagle	Osprey	Kingfisher	River Otter	Mink
RELMAP Remote	1.0e-01	5.1e-01	1.8e-02	2.0e-02	5.0e-02	3.0e-02	2.0e-02
Eastern Site							
RELMAP 50th	4.3e-02	2.2e-01	7.7e-03	8.5e-03	2.1e-02	1.3e-02	8.5e-03
RELMAP 90th	9.0e-02	4.5e-01	1.6e-02	1.8e-02	4.5e-02	2.7e-02	1.8e-02
Western Site							
RELMAP 50th	8.5e-03	4.3e-02	1.5e-03	1.7e-03	4.3e-03	2.5e-03	1.7e-03
RELMAP 90th	2.9e-02	1.5e-01	5.1e-03	5.7e-03	1.4e-02	8.5e-03	5.7e-03

6. LOCAL IMPACT ANALYSIS

6.1 Description of Approach

The results of two separate analyses are presented in Chapter 6. The first analysis shows the predictions of mercury fate, transport, and exposure that result from individual emission sources (see Figure 1-3). These model predictions are confined to the 50 km area around an individual anthropogenic mercury emission source, which is defined here to be the local area around the emissions source. In the second analysis the regional mercury modeling results RELMAP from Chapter 5 are combined with the results of the first analysis and the materials presented in Appendix H to estimate complete mercury exposure. See Figure 1-4.

6.1.1 Rationale and Utility of Model Plant Approach

Mercury is generally present as a low-level contaminant in combustion materials (e.g., coal or municipal solid waste) and industrial material (for more information on mercury in emissions refer to Volume II of this Report). During combustion and high-temperature industrial processes, mercury is volatilized from these materials. Because of its high volatility, it is difficult to remove mercury from the post-combustion air stream. As a consequence, mercury is released to the atmosphere. As noted previously, anthropogenic mercury emissions are not the only source of mercury to the atmosphere. Mercury may be introduced into the atmosphere through volatilization from natural sources such as lakes and soils. Consequently, it is difficult to trace the source(s) of mercury concentrations in environmental media and biota. For this reason it is also difficult to gain an understanding of contribution to those concentrations.

For the purposes of the analysis in this Chapter, exposure to mercury is defined as chemical contact with the outer boundary of an organism (also called a receptor). An organism's contact with mercury may occur through several different exposure routes including dermal, inhalation, and oral. Assessment of mercury exposure is complicated by the physical and chemical properties of this element. These include the different physical forms of mercury which are manifested in the environment, the different oxidative states it exhibits, and the duality of its environmental behavior as both an inorganic and an organic compound. In addition, the uncertain accuracy of analytical techniques, particularly at low environmental concentrations, and problems with contamination during environmental sampling further complicate the assessment of mercury exposure.

For this assessment it was not possible to model the emission impact of every mercury emission source in each selected industrial and combustion class. Consequently, the actual mercury emission data and facility characteristics for any specific source were not modeled. Instead, a model plant approach, as described in Appendix F, was utilized to develop facilities which represent actual sources. Model plants were developed to represent six source categories; namely municipal waste combustors, coal and oil-fired boilers, medical waste incinerators, chlor-alkali plants, primary copper smelters and primary lead smelters. The model plants were designed to characterize the mercury emission rates as well as the atmospheric release processes exhibited by actual facilities in the source class. The modeled facilities were not designed to exhibit extreme sources (e.g., the facility with the highest mercury emission rate) but rather to serve as a representative of the industrial/combustion source class.

This assessment took as its starting point the results of measured mercury emissions from selected anthropogenic sources. Using a series of fate, transport and exposure models and hypothetical constructs, mercury concentrations in environmental media, pertinent biota and ultimately mercury

contact with human and wildlife receptors were predicted. An effort was made to estimate the amount of receptor contact with mercury as well as the oxidative state and form of mercury contacted. No attempt was made to estimate an internal dose for either wildlife or human receptors.

In taking the model plant approach, it was realized that there would be a great deal of uncertainty surrounding the predicted fate and transport of mercury as well as the ultimate estimates of exposure. The uncertainty can be divided into modeling uncertainty and parameter uncertainty. Parameter uncertainty can be further subdivided into uncertainty and variability depending on the level to which a particular model parameter is understood. A limited quantitative analysis of uncertainty is presented. It is also hoped that the direction of future research can be influenced toward reducing the identified uncertainties which significantly impact key results.

6.1.2 Phase and Oxidation State of Emitted Mercury

The literature describes several forms of mercury detected in the emissions from the selected sources. Primarily, these include elemental mercury (Hg^0) and inorganic mercuric (Hg^{2+}). Generally, only total mercury has been measured in emission analyses. The reports of methylmercury in emissions are imprecise. It is believed that, if methylmercury is emitted from industrial processes and combustion sources, the quantities emitted are much smaller than emissions of Hg^0 and Hg^{2+}. Only Hg^0 and Hg^{2+} were considered in this air dispersion modeling.

The two types of mercury species considered in the emissions are expected to behave quite differently once emitted from the stack. Hg^0, due to its high vapor pressure and low water solubility, is not expected to deposit close to the facility. In contrast, Hg^{2+}, because of differences in these properties, is expected to deposit in greater quantities closer to the emission sources.

At the point of stack emission and during atmospheric transport, the contaminant is partitioned between two physical phases: vapor and particle-bound. The mechanisms of transport of these two phases are quite different. Particle-bound contaminants can be removed from the atmosphere by both wet deposition (precipitation scavenging) and dry deposition (gravitational settling, Brownian diffusion). Vapor phase contaminants may also be depleted by these processes, although historically their main impacts were considered to be through absorption into plant tissues (air-to-leaf transfer) and human exposure occurred through inhalation.

For the present analysis, the vapor/particle (V/P) ratio in the local atmosphere was assumed to be equal to the V/P ratio as it would exist in stack emissions. It is recognized that this is a simplification of reality, as the ratio when emitted from the stack is likely to change as the distance from the stack increases. The air concentration used for inhalation was taken as the sum of the vapor and particle air concentrations.

The particle size distribution may differ from one combustion process to another, depending on the type of furnace and design of combustion chamber, composition of feed/fuel, particulate matter removal efficiency and design of air pollution control equipment, and amount of air in excess of stoichiometric amounts that is used to sustain the temperature of combustion. The particle size distribution used is an estimate of the distribution within an ambient air aerosol mass and not at stack tip. Based on this assumption, an aerosol particle distribution based on data collected by Whitby (1978) was used. This distribution was split between two modes: accumulation and coarse particles. The geometric mean diameter of several hundred measurements indicates that the accumulation mode dominates particle size, and a representative particle diameter for this mode is 0.3 microns. The coarse particles are formed largely from mechanical processes that suspend dust, sea spray and soil particles

in the air. A representative diameter for coarse particles is 5.7 microns. The fraction of particle emissions assigned to each particle class was approximated based on the determination of the density of surface area of each representative particle size relative to total surface area of the aerosol mass. Using this method, approximately 93% and 7% of the total surface area was estimated to be in the 0.3 and 5.7 micron diameter particles, respectively.

Table 6-1
Representative Particle Sizes and Size Distribution Assumed for Divalent Mercury Particulate Emissions

Representative Particle Size (microns)*	Assumed Fraction of Particle Emissions in Size Category
0.3	0.93
5.7	0.07

* These values are based on the geometric means of aerosol particle distribution measurements as described in Whitby (1978).

The speciation estimates for the model plants were made from thermal-chemical modeling of mercury compounds in flue gas, from the interpretation of bench and pilot scale combustor experiments and from interpretation of available field test results. The amount of uncertainty surrounding the emission rates data vary for each source. There was also uncertainty with respect to the species of mercury emitted.

Although the speciation may change with distance from the local source, for this analysis it was assumed that there were no plume reactions that significantly modified the speciation at the local source. Because of the differences in deposition characteristics of the two forms of mercury considered, the assumption of no plume chemistry was a particularly important source of uncertainty.

6.1.3 Modeling the Deposition of Mercury

Once emitted from a source, the mercury may be deposited to the ground via two main processes: wet and dry deposition. Wet deposition refers to the mass transfer of dissolved gaseous or suspended particulate mercury species from the atmosphere to the earth's surface by precipitation, while dry deposition refers to such mass transfer in the absence of precipitation.

The deposition properties of the two species of mercury addressed in stack emissions, elemental and divalent mercury, are considered to be quite different. Due to its higher solubility, divalent mercury vapor was thought to deposit much more rapidly than elemental mercury. At this time, however, no conclusive data exist to support accurate quantification of the deposition rate of divalent mercury vapor. In this analysis, nitric acid vapor was used as a surrogate for Hg^{2+} vapor based on their similar solubility in water. Whether a pollutant is in the vapor form or particle-bound is also important for estimating deposition, and each is treated separately.

Dry deposition was estimated by multiplying the predicted air concentration at ground level by a deposition velocity. For particles, the dry deposition velocity was estimated using the CARB algorithms (CARB 1986) that represent empirical relationships for transfer resistances as a function of particle size, density, surface area, and friction velocity. For the vapor phase fraction, the dry deposition velocities were specified for each atmospheric stability class. (Atmospheric stability is a measure of the amount of turbulance in the atmosphere: stable conditions inhibit dispersion and unstable conditions enhance dispersion. See Appendix D.) For this analysis, elemental mercury vapor was assumed to not dry deposit or to deposit at negligible rates. Divalent mercury vapor was assumed to dry deposit in a manner similar to nitric acid, for which deposition velocities were available (See Appendix D for more information). For stability classes A-C, a dry deposition velocity of 1 cm/s was assumed for divalent mercury vapor, while for classes D-F a dry deposition velocity of 0.3 cm/s was assumed. The lower value was assumed for classes D-F is made because it has been observed that dry deposition is lower during nighttime conditions, and classes D-F occur predominantly at night (N.B. Classes A-C by definition can occur only during daytime conditions).

Wet deposition was estimated according to the method of Slinn (1984) and later modified by PEI and Cramer (1986). The wet deposition rate was characterized by a scavenging coefficient that can depend on precipitation intensity and particle size. For particles, the scavenging coefficients were from PEI and Cramer (1986). For the vapor phase fraction, a scavenging coefficient was also used, but it was calculated using estimates for the washout ratio, which is the ratio of the concentration of the chemical in surface-level precipitation to the concentration in surface-level air. Because of its higher solubility, divalent mercury vapor was assumed to be washed out at higher rates than elemental mercury vapor. The washout ratio for divalent mercury vapor was selected based on an assumed similarity between scavenging for divalent mercury and gaseous nitric acid. This is based on Petersen (1995), and the value used for the washout ratio for divalent vapor was 1.6×10^6 (see Appendix D of this volume for more details).

6.1.4 Development and Description of Model Plants

Model plants representing six source classes were developed to represent a range of mercury emissions sources. The source categories were selected for the exposure assessment based on their estimated annual mercury emissions as a class or their potential to be localized point sources of concern. The categories selected were these:

- municipal waste combustors (MWCs),
- medical waste incinerators (MWIs),
- utility boilers,
- chlor-alkali plants (CAP),
- primary copper smelters (PCS), and
- primary lead smelters (PLS).

Parameters for each model plant were selected after evaluation of the characteristics of a given source category and current knowledge of mercury emissions from that source category. Important variables for the mercury risk assessment included mercury emission rates, mercury speciation and mercury transport/deposition rates. Important model plant parameters included stack height, stack diameter, stack volumetric flow rate, stack gas temperature, plant capacity factor (relative average operating hours per year), stack mercury concentration, and mercury speciation. Emission estimates were assumed to represent typical emission levels emitted from existing sources. Table 6-2 shows the process parameters assumed for each model plant considered in this analysis (for details regarding these values, see Appendix F).

Table 6-2
Process Parameters for Model Plants

Model Plant	Plant Size	Capacity (% of year)	Stack Height (ft)	Stack Diameter (ft)	Hg Emission Rate (kg/yr)	Speciation Percent ($Hg^0/Hg^{2+}/Hg^P$)	Exit Velocity (m/sec)	Exit Temp. (°F)
Large Municipal Waste Combustors	2,250 tons/day	90%	230	9.5	1330	20/60/20	21.9	285
Small Municipal Waste Combustors	200 tons/day	90%	140	5	170	20/60/20	21.9	375
Continuous Medical Waste Incinerators	1500 lb/hr capacity (1000 lb/hr actual)	90%	40	2.7	80	20/60/20	7.3	1500
Intermittent Medical Waste Incinerators	200 lb/hr capacity (133 lb/hr actual)	20%	40	1.2	2.4	20/60/20	7.3	1500
Large Coal-fired Utility Boiler	975 Megawatts	65%	732	27	230	50/30/20	31.1	273
Medium Coal-fired Utility Boiler	375 Megawatts	65%	465	18	90	50/30/20	26.7	275
Small Coal-fired Utility Boiler	100 Megawatts	65%	266	12	10	50/30/20	6.6	295
Medium Oil-fired Utility Boiler	285 Megawatts	65%	290	14	2	50/30/20	20.7	322
Chlor-alkali plant	300 tons chlorine/day	90%	10	0.5	380	70/30/0	0.1	Ambient
Primary Copper Smelter	180 tons Cu/day	90%	505	15	536	85/10/5	6	430
Primary Lead Smelter	304 tons lead/day	90%	350	20	2680	85/10/5	2.8	347

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor Phase Mercury
[c] Hg_p = Particle-Bound Mercury

6.1.5 Hypothetical Locations of Model Plants

There are a variety of geographic aspects that can influence the impacts of mercury emissions from an anthropogenic source. These aspects include factors that affect the environmental chemistry of a pollutant and the physics of plume dispersion. Environmental chemistry can include factors such as the amount of wet deposition in a given area. Factors affecting plume dispersion include terrain, wind direction and average wind speed.

Because wet deposition may be an important factor leading to mercury exposures, especially for the more soluble species emitted, the meteorology of a location was used as a selection criterion. Two different types of meteorology were deemed necessary to characterize the environmental fate and transport of mercury: an arid/semi-arid site and a humid site. The humidity of an area was based on total yearly rainfall. (See Appendix B).

Terrain features refer to the variability of the receptor height with respect to a local source. Broadly speaking, there were two main types of terrain used in the modeling: simple, and complex. Simple terrain is defined as a study area that is relatively level and well below stack top (rather, the effective stack height). Complex terrain referred to terrain that is not simple, such as source located in a valley or a source located near a hill. This included receptors that are above or below the top of the stack of the source. Complex terrain can effect concentrations, plume trajectory, and deposition. Due to the complicated nature of plume flow in complex terrain, it is probably not possible to predict impacts in complex terrain as accurately as for simple terrain (for a description of the air model used and how it addressed complex terrain, see Appendix D). In view of the wide range of uncertainty inherent in accurately modeling the deposition of the mercury species considered, the impacts posed by complex terrain were not incorporated in the local scale analysis. However, a limited exercise investigating the extent consideration of complex terrain may affect the air modeling results was performed.

6.1.6 Hypothetical Exposure Scenarios and Location of Receptors Relative to Local Source

Three types local plant settings were utilized: urban, rural and lacustrine. These have been described in section 4.2.2.

There are conceivably an infinite number of possible receptor locations (relative to the source) which could have been used for this analysis. For the purpose of this analysis, a watershed/water body configuration was assumed to be located at 2.5 km, 10 km, and 25 km from the local source, as shown in Figure 6-1.

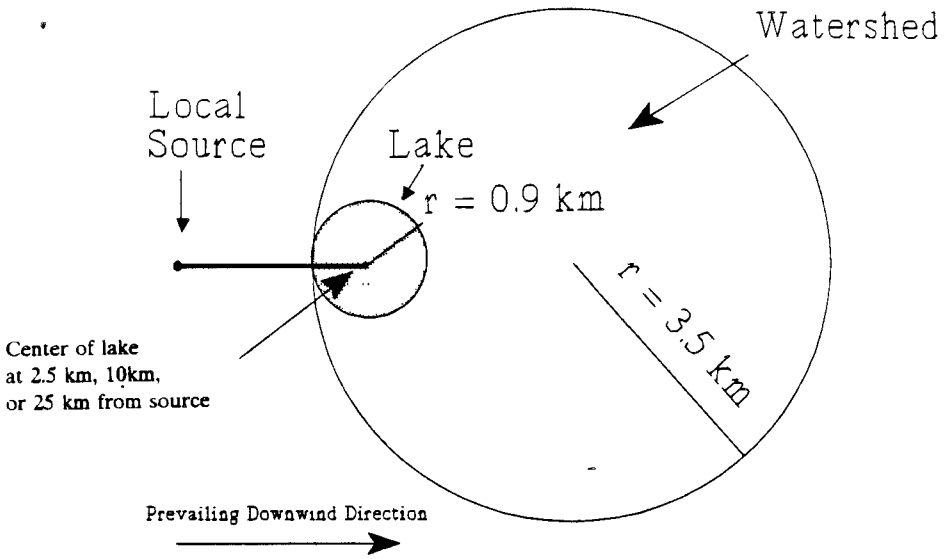
Area-averaged deposition rates and air concentrations were computed for both the water body and watershed. The watershed value was used to estimate soil and vegetation concentrations.

6.2 Results of Local Scale Modeling

In this section the possible effects of a local source on receptors within 50 km are considered. The air modeling alone is one of the most uncertain and controversial aspects of the entire analysis. There are at present no validated models for estimating the deposition of mercury close to an anthropogenic source.

For each hypothetical model plant, COMPDEP (the local atmospheric dispersion and deposition model used in this assessment) was run using weather data from 1989, the same year used

Figure 6-1
Configuration of Hypothetical Water Body and Watershed Relative to Local Source



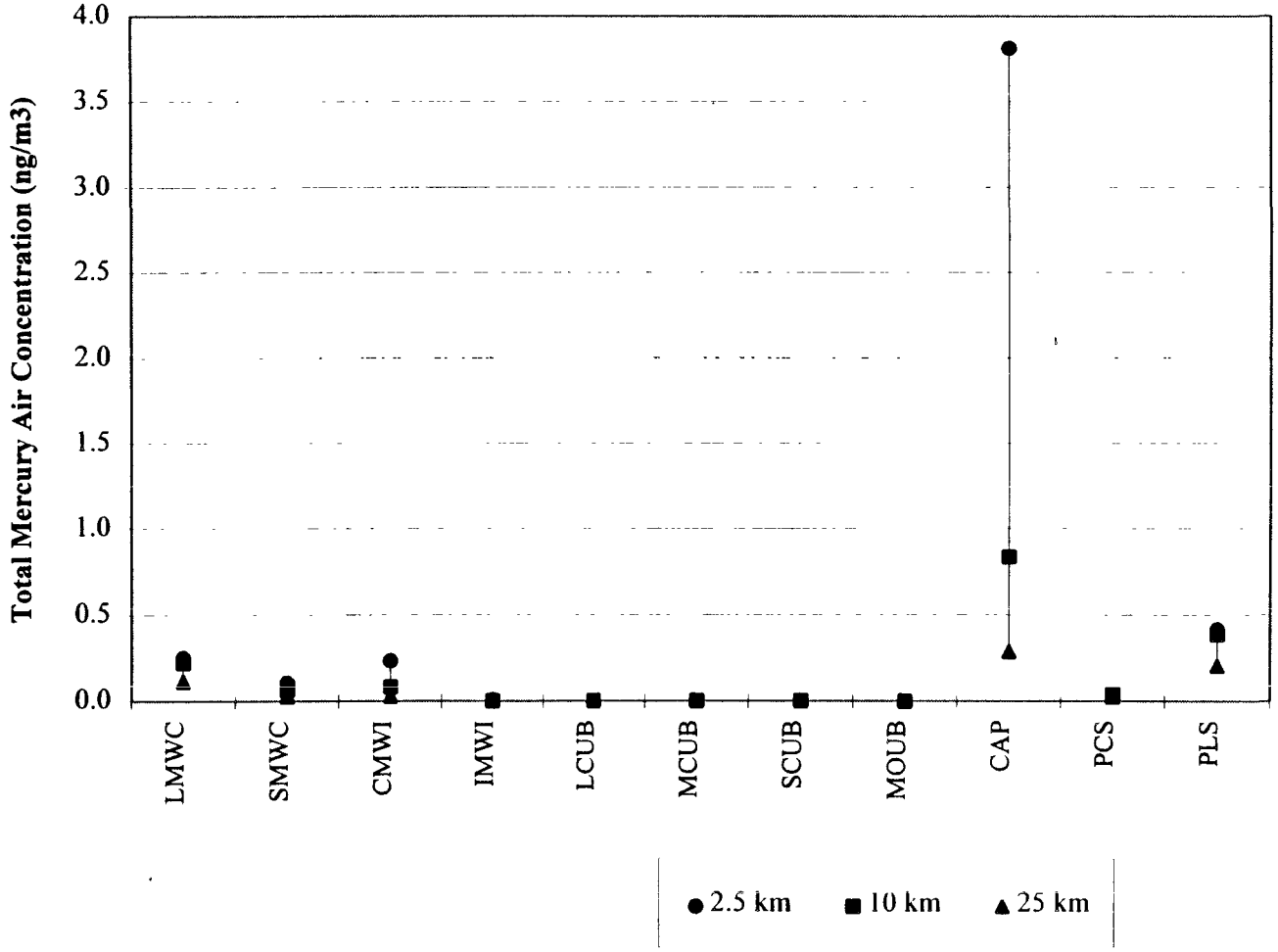
for the RELMAP analysis. The predicted values for air concentration and deposition rate were then used as inputs for the IEM2 model. Separate area-averaged values were used for both the water body and watershed. These values were assumed to be representative for 30 years, the assumed typical lifetime of a facility. During this 30 year period, the mercury concentration in soil was allowed to build-up, taking into account basic loss processes such as leaching, runoff and erosion. The calculated values at the end of the 30 year period were then used as input to the water portion of the IEM2 model, which calculated steady-state water concentrations based on the 30 year values. The only difference between the results presented here and the results in Section 4.3 are the air concentrations and deposition rates used.

6.2.1 Air Concentrations

In analyzing the air concentrations predicted by the COMPDEP model, it is important to observe that in a typical year the predicted air concentration due to the local source at *any* receptor is zero a rather substantial fraction of the time. There are two basic reasons for this. First, in order for there to be a non-zero air concentration for a given hour the receptor must be in the downwind direction. This means that the wind must be blowing in a direction within 90 degrees of the receptor itself. For most sites this only occurs about 50% of the time for the direction with the highest frequency. Second, even if the receptor is downwind, the predicted air concentration will be significant only if the wind is blowing in a direction within about 10 degrees of the receptor's direction relative to the facility. For most sites this occurs for the prevailing downwind direction only about 10 to 15 percent of the time. Because the air concentrations are averaged over the year, this results in (usually) low average air concentrations.

Figures 6-2 and 6-3 show the predicted air concentrations (area-averaged over the watershed) for all facilities for both the Eastern and Western site. The results are virtually identical for both sites, and none of the predicted air concentrations exceed 4 ng/m^3.

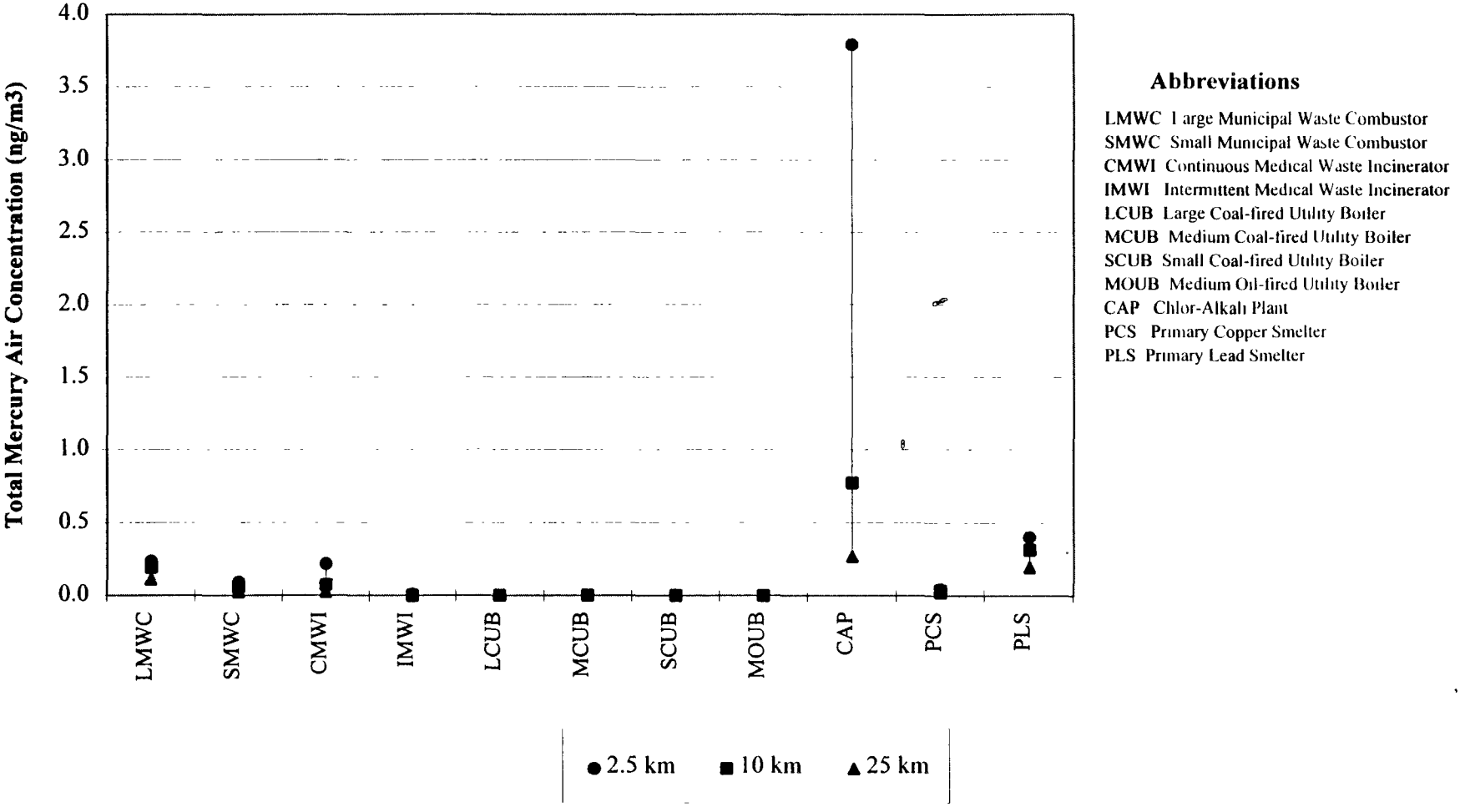
Figure 6-2
Predicted Total Mercury Air Concentration (ng/m^3) at Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-3
Predicted Total Mercury Air Concentration (ng/m^3) at Western Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

The differences in predicted air concentrations across source classes depend primarily on three key parameters: the total mercury emission rate, the stack height and the exit velocity of the plume from the source. The sensitivity of the air model to the emission rate is to be expected because the predicted air concentrations are linear with the total mercury emission rate, and the model plants are assumed to have a wide range of emission rates (from 2 kg/yr up to over 1330 kg/yr; see Table 6-6). Both the stack height and exit velocity were used in calculating the effective stack height, which is the height to which the plume rises from the stack top. The importance of the effective stack height on air concentrations is well known and is demonstrated in this assessment by the predicted air concentrations for the chlor-alkali plant, which clearly dominates the values as a whole. See Figures 6-2 and 6-3. This is due to an assumed combination of a low stack height (10 feet) and slow stack gas exit velocity (0.1 m/s) (this results in a low effective stack height), and a comparatively high assumed total mercury emissions rate of about 280 kg/yr. The low stack parameters result in predicted low plumes that are not as vertically dispersed at the receptor when compared with the facilities with higher stacks, thereby enhancing air concentrations.

In general, the predicted average concentrations of atmosphere mercury were quite low from the model plants. The only source class for which significantly elevated air concentrations were predicted is the chlor-alkali plant. This is due to a very low stack height coupled with a high assumed mercury emission rate. The low stack height resulted in predicted plumes that were close to the receptors considered, and so there was less predicted dispersion of the plume compared to the other facilities.

6.2.2 Deposition Rates

In contrast to the predicted air concentrations, the annual deposition rates are *cumulative*; they represent the sum of any deposition that occurs during the year, and hence are not affected by long periods of little deposition. Furthermore, the COMPDEP model predicts that significant deposition events occur infrequently, and it is these relatively rare events that are responsible for the majority of the annual deposition rate.

Because dry deposition is calculated by multiplying the predicted air concentration for the hour by the deposition velocity, significant dry deposition events only occur when, for the reasons discussed above, there is a "spike" of predicted high air concentration for a given hour. Annual dry deposition tended to be dominated by these peak values when the wind is blowing within a few degrees of the receptor's direction.

For the Eastern site which had a higher rate of precipitation than the Western site, wet deposition can dominate the total deposition for receptors close to the source. Single wet deposition events were predicted to deposit 300 times more mercury than a high dry deposition event. These high wet deposition events based on the modeling are even rarer than elevated dry deposition events because not only must the wind direction be within a few degrees of the receptor's direction, but precipitation must be occurring as well.

Table 6-3 shows the deposition rate parameters for the emitted pollutants addressed in this study.

Table 6-3
Comparison of Assumed Deposition Parameters for Elemental and Divalent Mercury

Parameter	Elemental Mercury Vapor	Divalent Mercury Vapor	Divalent Mercury Particulate
Dry Deposition Velocity (cm/s)	0	0.3 - 1.0	0.2 - 1.4[b]
Washout Ratio (unitless)	1.60×10^{4}	1.60×10^{6}	NA
Wet Deposition Scavenging coefficient (/s)[a]	3×10^{-6} to 1×10^{-5}	3×10^{-4} to 1×10^{-3}	2.2×10^{-4} (light precipitation) to 1.46×10^{-3} (heavy precipitation)

[a] For elemental and divalent vapor, this is calculated by $L = W P / L$, where W is the washout ratio, P is the representative precipitation intensity for the hour, and L is the predicted mixing height for the hour. Due to the dependence on mixing height, the upper end values of the ranges shown routinely occur even for light precipitation.
[b] Based on particle density of 1.8 g/cm^3, particle diameter of 2 μm, surface roughness of 0.3 m, and ambient air temperature of 295 K.

Although such wet deposition events may be rare for a fixed receptor, anytime precipitation occurs a significant deposition event will occur in the downwind direction for that hour. Because of this, the wet deposition rates may be more uniform with respect to direction than one may expect.

In general, only the divalent form of mercury is predicted to have significant deposition rates. This is because the assumed atmospheric removal (wet and dry deposition) of divalent mercury is significantly larger for divalent mercury than for elemental mercury. For dry deposition of divalent vapor, the deposition velocities were either 0.3 cm/s or 1 cm/s, depending on the stability class for a given hour, while it was assumed that elemental mercury vapor does not dry deposit. Although wet deposition of elemental mercury vapor is addressed, the difference in solubilities of the two forms of mercury resulted in the divalent vapor being predicted to wet deposit at a higher rate than elemental vapor.

Table 6-4 shows a breakdown of the total deposition rate in terms of the amount that was predicted to occur via wet deposition and fraction of total deposition that is particulate.

There are several general features noted here. With the current assumptions regarding mercury deposition and receptor terrain, wet deposition dominated the predicted deposition rates. For most of the facilities, wet deposition accounted for most of the total deposition, and the contribution of wet deposition decreases with distance from the source. As expected, wet deposition played a larger role at the more humid site in the east.

The predicted dry deposition rates depend ultimately on the predicted air concentrations and the speciation of the mercury in the local atmosphere. For this reason, dry deposition accounted for most of the total deposition for the facility with the highest predicted air concentrations, the chlor-alkali plant. In complex terrain, dry deposition can play a larger but uncertain role than in the results presented here. This is discussed in more detail below.

Although for some facilities particulates comprised more than 50% of the total divalent emissions, it played a smaller role in terms of total local deposition. This indicates the faster removal from the atmosphere of the vapor form.

Table 6-4
Breakdown of Total Mercury Deposition Rates Predicted by COMPDEP

Facility[a]	Emission Speciation			Eastern Site						Western Site					
				2.5 km		10 km		25 km		2.5 km		10 km		25 km	
	% Elemental Mercury Vapor	% Divalent Mercury Vapor	% Particulate	% Wet	% Particulate	% Wet	% Particulate	% Wet	% Particulate	% Wet	% Particulate	% Wet	% Particulate	% Wet	% Particulate
LMWC	20%	60%	20%	93.2%	9.7%	71.6%	16.9%	34.8%	21.2%	81.4%	12.4%	51.6%	15.0%	27.5%	13.1%
SMWC	20%	60%	20%	82.1%	9.6%	53.5%	15.7%	22.6%	17.1%	60.6%	11.4%	34.7%	12.8%	18.5%	11.1%
CMWI	20%	60%	20%	53.9%	9.7%	30.3%	13.5%	12.2%	13.8%	27.8%	9.2%	18.0%	10.3%	10.6%	9.6%
IMWI	20%	60%	20%	44.6%	9.3%	27.5%	13.0%	11.7%	13.6%	21.1%	8.6%	16.3%	10.0%	10.3%	9.5%
LCUB	50%	30%	20%	99.7%	21.5%	94.3%	27.3%	80.1%	37.1%	98.7%	30.0%	89.3%	31.2%	77.3%	32.0%
MCUB	50%	30%	20%	99.0%	18.2%	90.8%	28.0%	68.5%	39.8%	97.0%	24.7%	82.5%	30.2%	61.5%	30.8%
SCUB	50%	30%	20%	91.3%	16.8%	71.7%	27.8%	44.5%	32.6%	76.1%	20.7%	51.8%	24.5%	31.3%	21.9%
MOUB	50%	30%	20%	97.4%	17.7%	83.8%	28.6%	55.1%	36.9%	92.3%	23.7%	69.3%	28.0%	43.6%	25.7%
CAP	70%	30%	0%	30.6%	0.0%	20.4%	0.0%	11.7%	0.0%	12.1%	0.0%	10.8%	0.0%	7.3%	0.0%
PCS	85%	10%	5%	96.9%	12.4%	86.3%	19.0%	66.8%	22.7%	89.8%	16.6%	70.0%	19.8%	50.0%	18.0%
PLS	85%	10%	5%	94.4%	12.3%	79.7%	19.0%	58.9%	21.4%	83.2%	15.9%	61.5%	19.0%	41.9%	16.8%

[a] The following abbreviations were used for the model plants: LMWC = Large Municipal Waste Combustion; SMWC = Small Municipal Waste Combustor, LMWI = Continuous Medical Waste Incinerators, IMWI - Intermediate Medical Waste Incinerator, LCUB = Large Coal-fired Utility Boiler, MCUB = Medium Coal-fired Utility Boiler, SCUB = Small Coal-fired Utility Boiler, MCUB = Medium Oil-fired Utility Boiler, CAP = Chlor-Alkali Plant, PCS = Primary Copper Smelter, and PLS = Primary Lead Smelter.

[b] $Hg0$ = Elemental Mercury Vapor

[c] Hg^{2+} = Divalent Mercury Vapor

Figures 6-4 and 6-5 show the predicted total mercury deposition rates for all facilities at both sites. Although there is uncertainty in the deposition rates, especially near the source, there are no data currently available to support or refute the predicted deposition rates at 2.5 km.

6.2.3 Media Concentrations

Soil. The predicted soil concentrations are directly proportional to total deposition. A comparison between Figure 6-4 (total deposition) and Figures 6-6 and 6-7 (soil concentrations) does not show any differences, save for a change of scale and units. Mercury that deposits was predicted to speciate as 98% Hg^{2+} and 2% methylmercury (see appendix A) and to mix evenly throughout the untilled (1 cm) or tilled (20 cm) depths. The predicted tilled soil concentration was therefore, 20 times less than the untilled concentration. Any factor changing total deposition behavior has the identical effect on soil concentrations.

Green Plants. Both the soil and air concentrations (root and foliar uptake) were used to calculate the mercury concentrations in grains, beans, fruits, and fruiting vegetables. Only atmospheric uptake was assumed to occur for leafy vegetables and only root uptake for potatoes and root vegetables. Additionally, unprotected plants (non-grains, non-rooting plants) we assumed to accumulate mercury though direct deposition onto the edible portion of the plant.

Shown in Figures 6-8 and 6-9 are the predicted mercury concentrations in leafy vegetables, and shown in Figures 6-10 and 6-11 are values for fruits. Leafy vegetables accumulate slightly more mercury than fruiting vegetables; most of the difference arises because of the higher interception fraction of leafy vegetables (resulting in more efficient direct deposition: Appendix A). Since air uptake dominated plant mercury accumulation for both types of plants, the trends in these figures follow those for predicted air concentrations very closely.

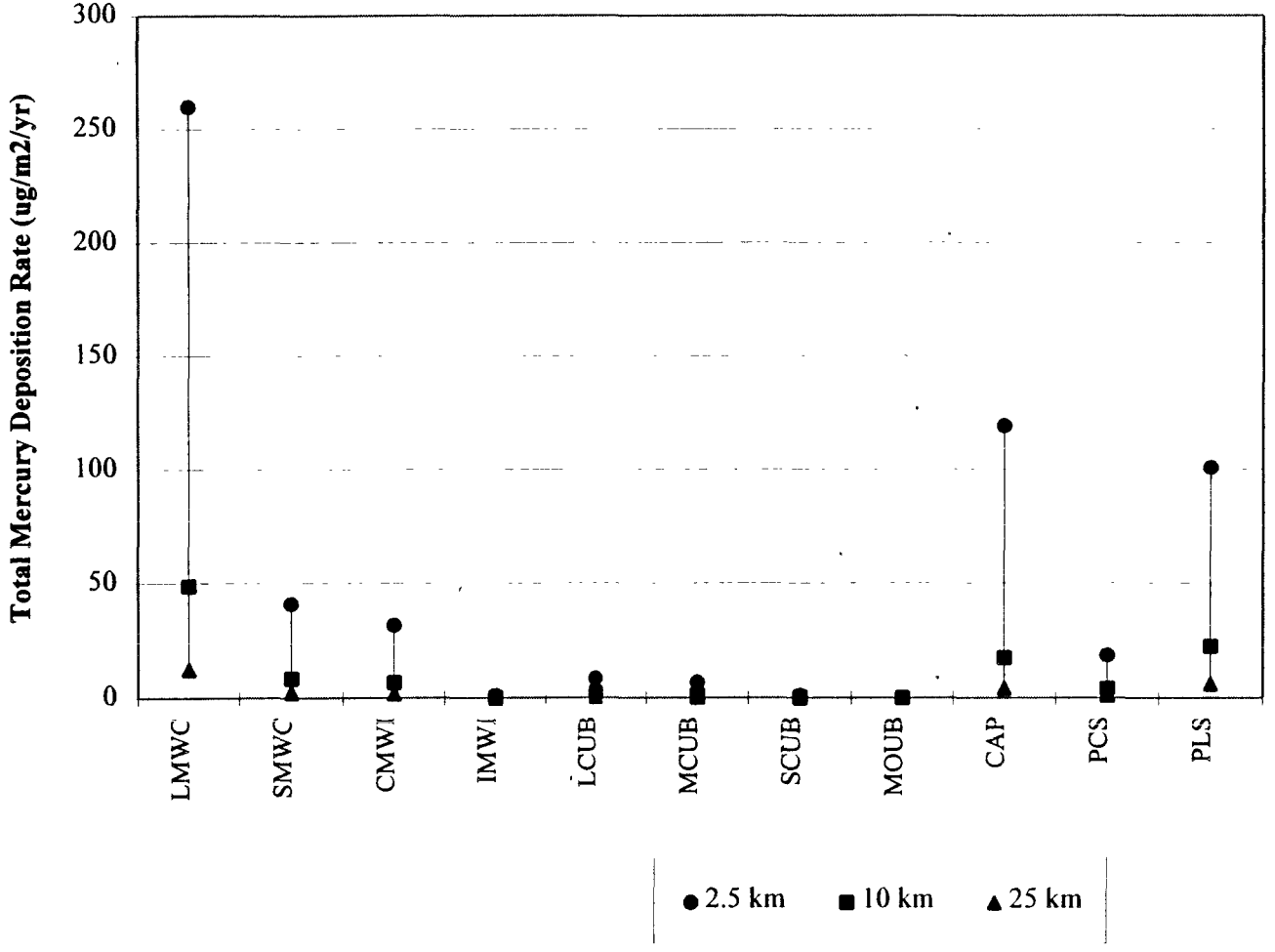
Mercury concentrations in grains and legumes were much lower than the concentrations predicted in leafy vegetables but followed the same overall trends among model plants and distance from the source. The lower concentrations were a result of smaller interception fractions for grains and legumes and the reduction in the air-to-plant biotransfer factor that accounted for lower concentrations in the edible parts of grains and legumes. Fruits were modeled with the same uptake parameters as fruiting vegetables, and had the same mercury concentrations. Potatoes and rooting vegetables had lower mercury concentrations than plant types which also accumulated mercury from the air, due to the low tilled soil concentrations. For all plants, Hg^{2+} always accounted for the bulk of the plant mercury. (See Chapter 4; section 4.3.2.2.)

Beef. As with the plant concentrations, the predicted beef concentrations (Figures 6-12 and 6-13) were rather low, mainly due to a low transfer coefficient (see Appendix A). Beef were assumed to get mercury from both vegetation and soil they consume.

Surface Water. Surface water concentrations, as with soil concentrations, followed total deposition patterns; however, a comparison between total deposition and surface water concentrations indicates noticeable differences in trends. The deposition rate was the most critical factor in determining surface water concentrations.

Fish. The fish concentrations were estimated by multiplying the predicted total mercury dissolved surface water concentration by the trophic-level-specific bioaccumulation factor (BAF). The bioaccumulation factor related total dissolved mercury in water to methylmercury concentrations in fish tissue. There is considerable uncertainty in the BAF, but the perdicted concentrations of mercury

Figure 6-4
Predicted Total Mercury Deposition Rate (ug/m^2/yr) at Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-5
Predicted Total Mercury Deposition Rate (ug/m^2/yr) at Western Site, COMPDEP Only

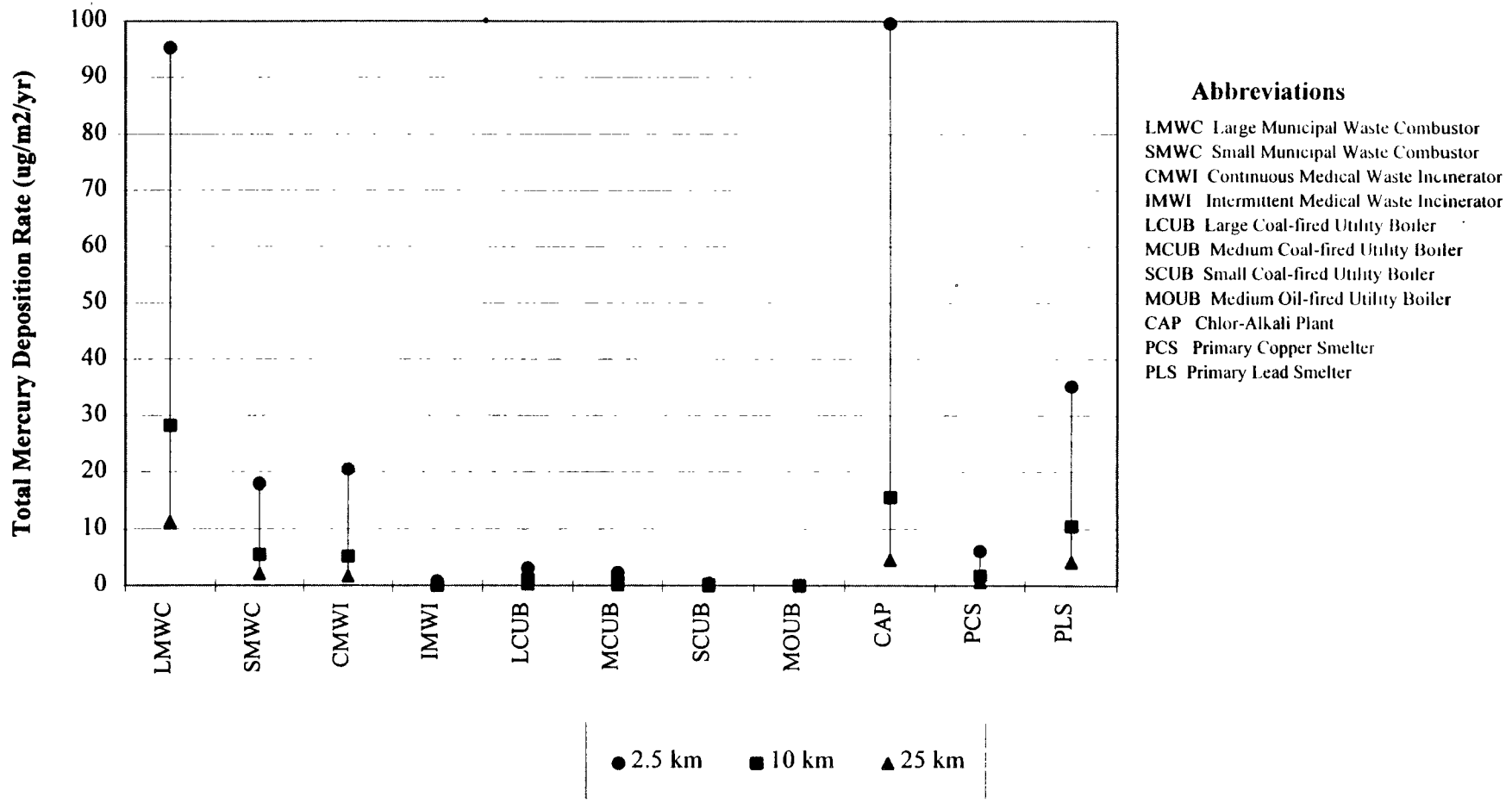
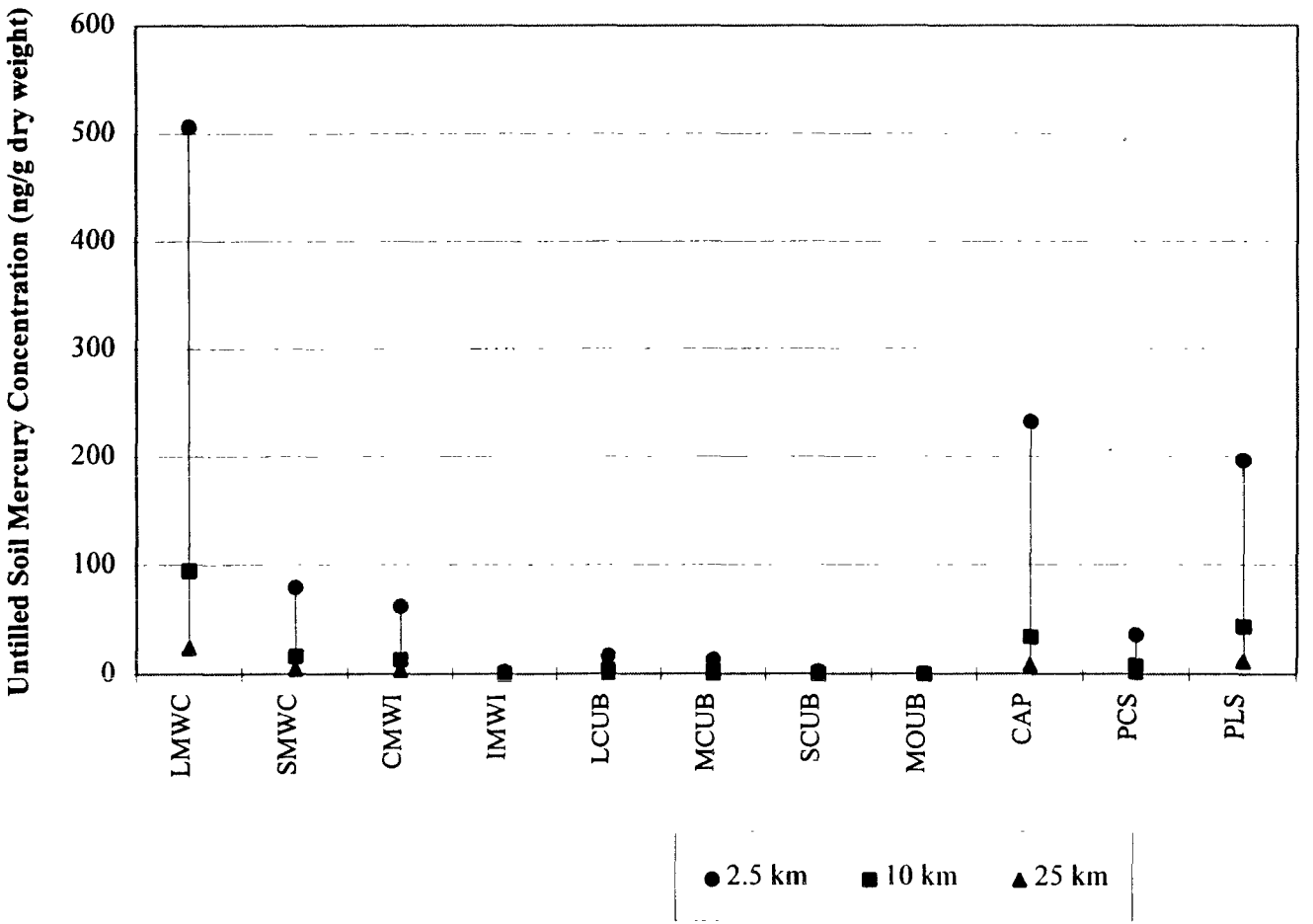


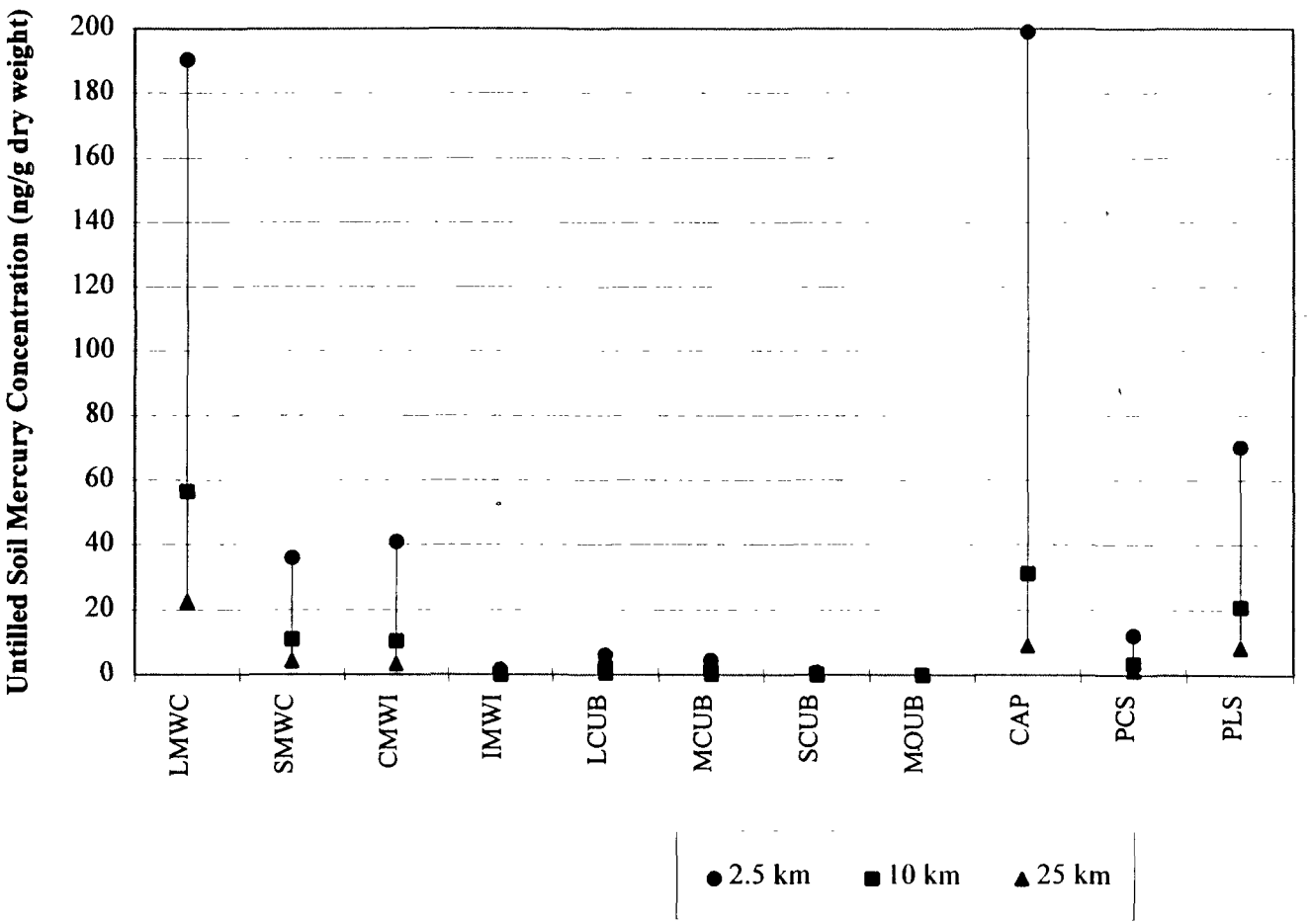
Figure 6-6
Predicted Untilled Soil Mercury Concentration (ng/g)
at Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

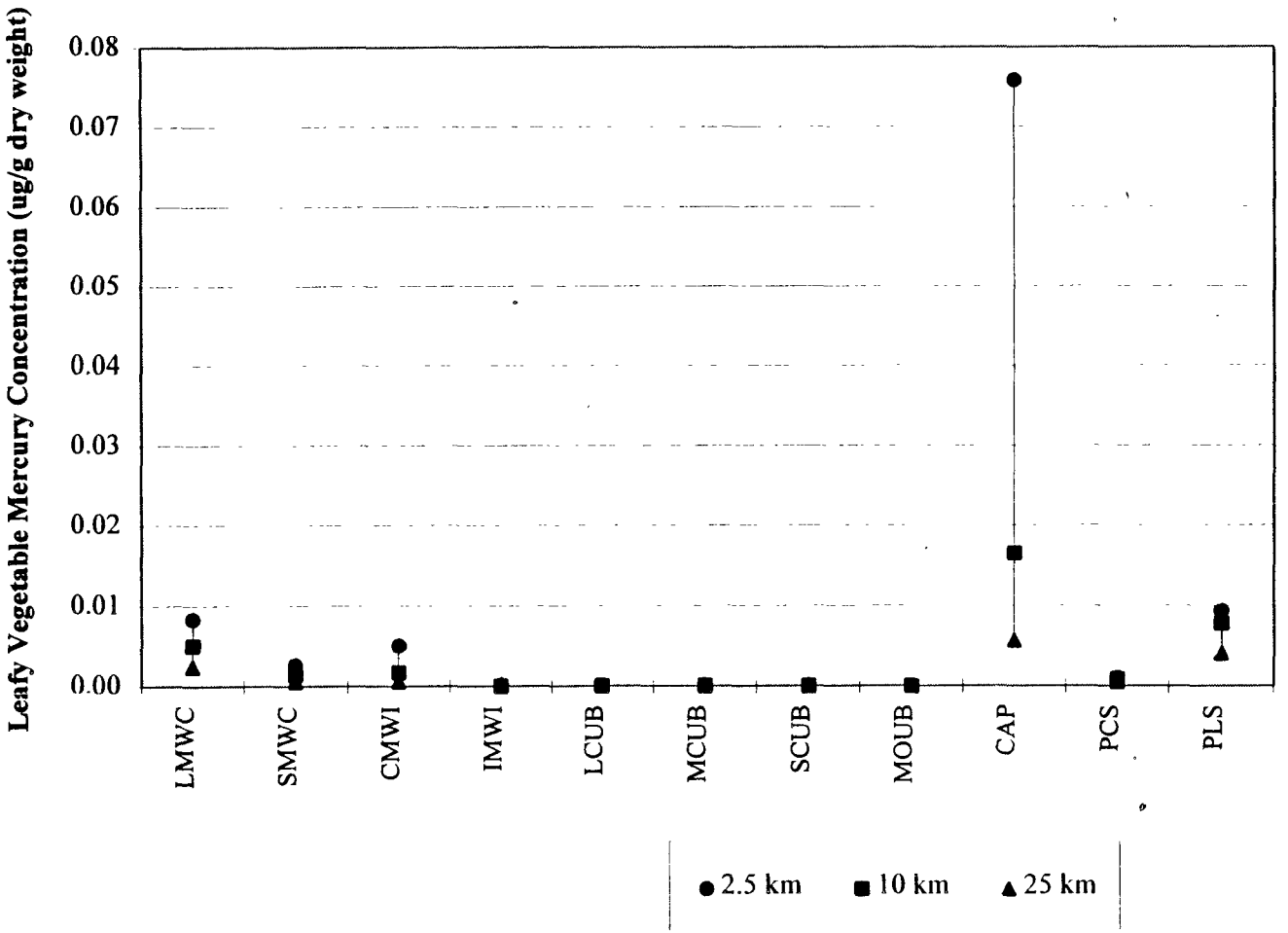
Figure 6-7
Predicted Untilled Soil Mercury Concentration (ng/g)
at Western Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-8
Predicted Leafy Vegetable Mercury Concentration (μg/g)
at the Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-9
Predicted Leafy Vegetable Mercury Concentration (μg/g)
at the Western Site, COMPDEP Only

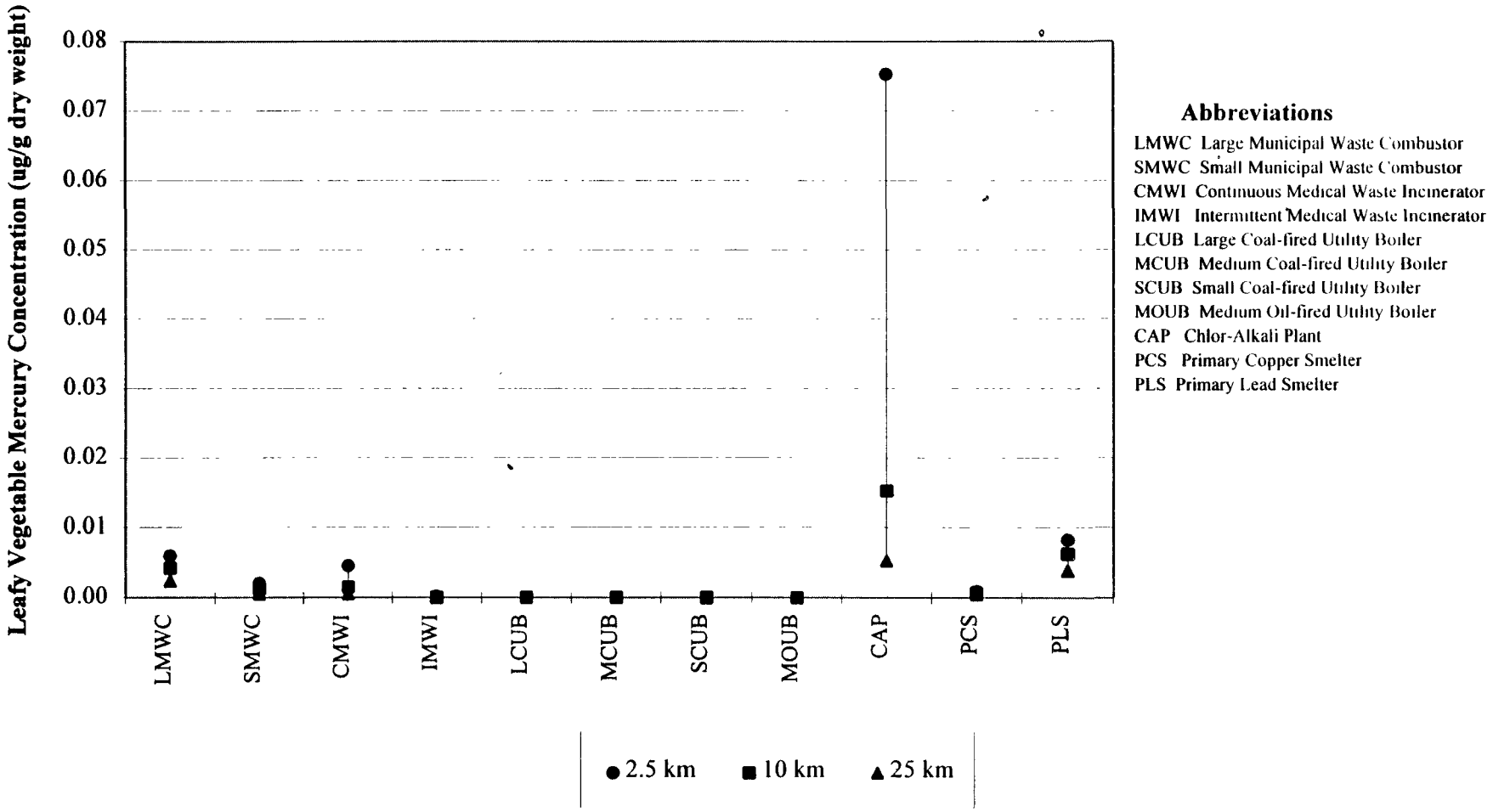
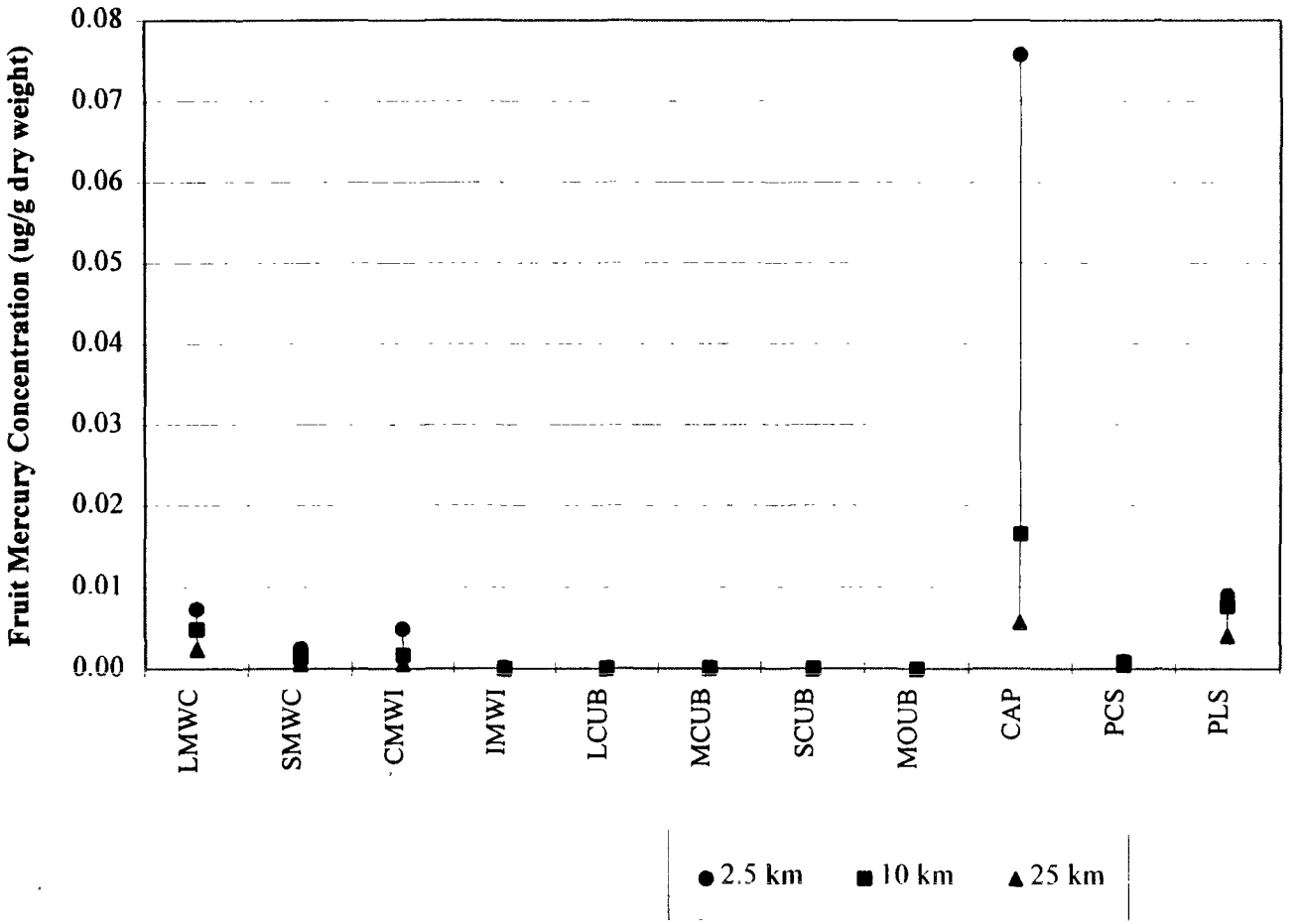


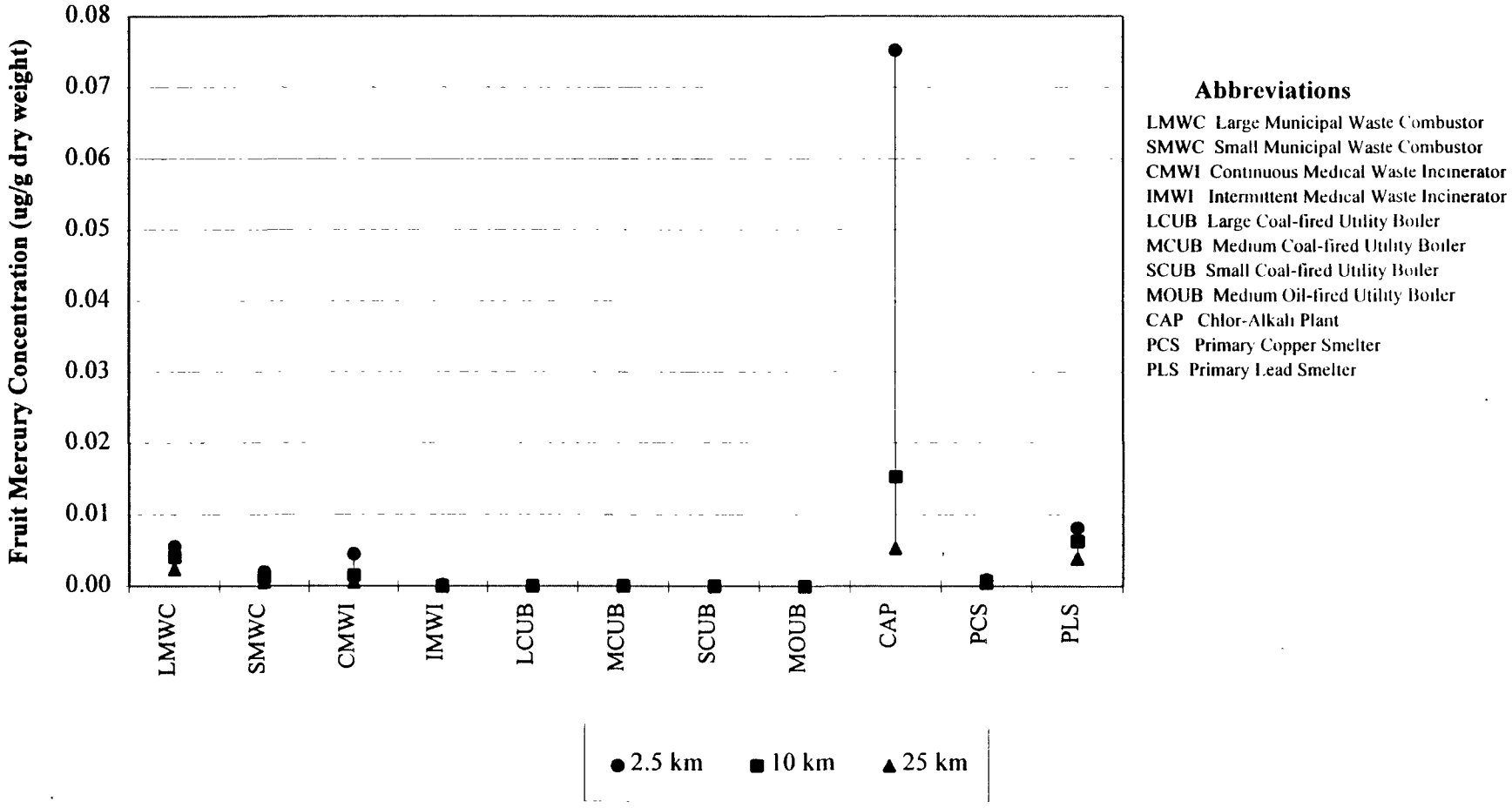
Figure 6-10
Predicted Fruit Mercury Concentration (μg/g)
at the Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

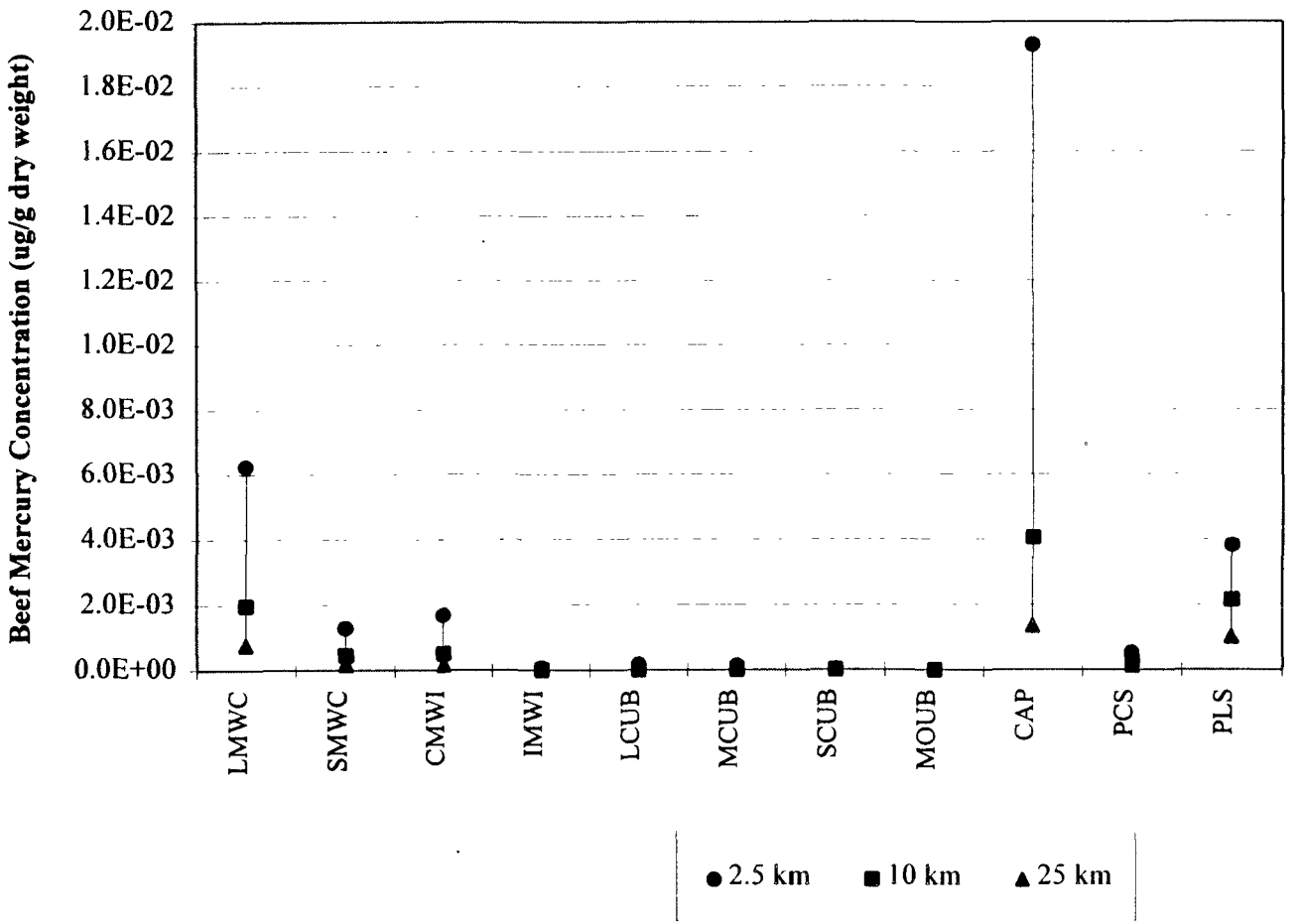
Figure 6-11
Predicted Fruit Mercury Concentration (μg/g)
at the Western Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-12
Predicted Beef Mercury Concentration (μg/g)
at the Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-13
Predicted Beef Mercury Concentration (μg/g)
at the Western Site, COMPDEP Only

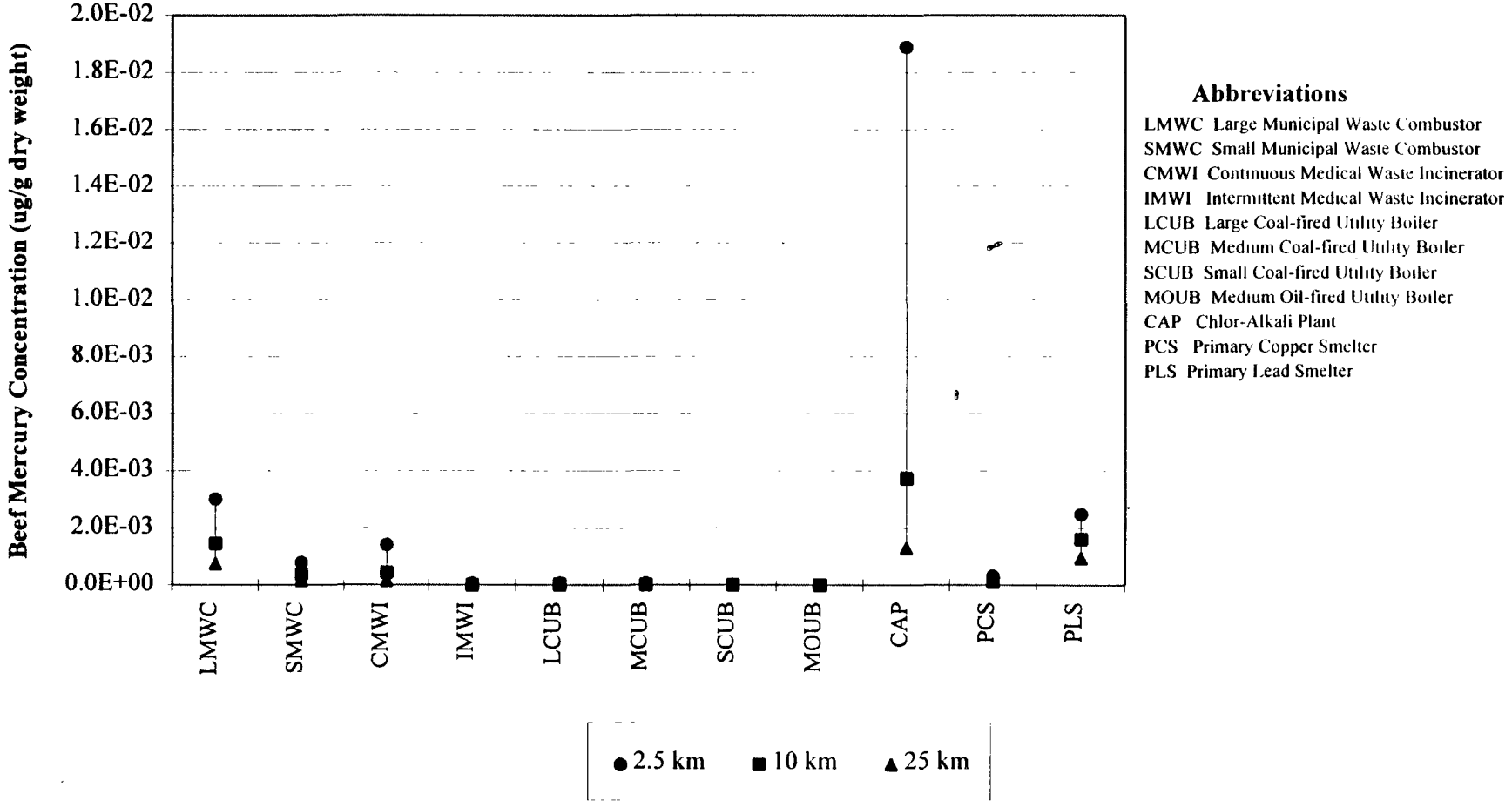
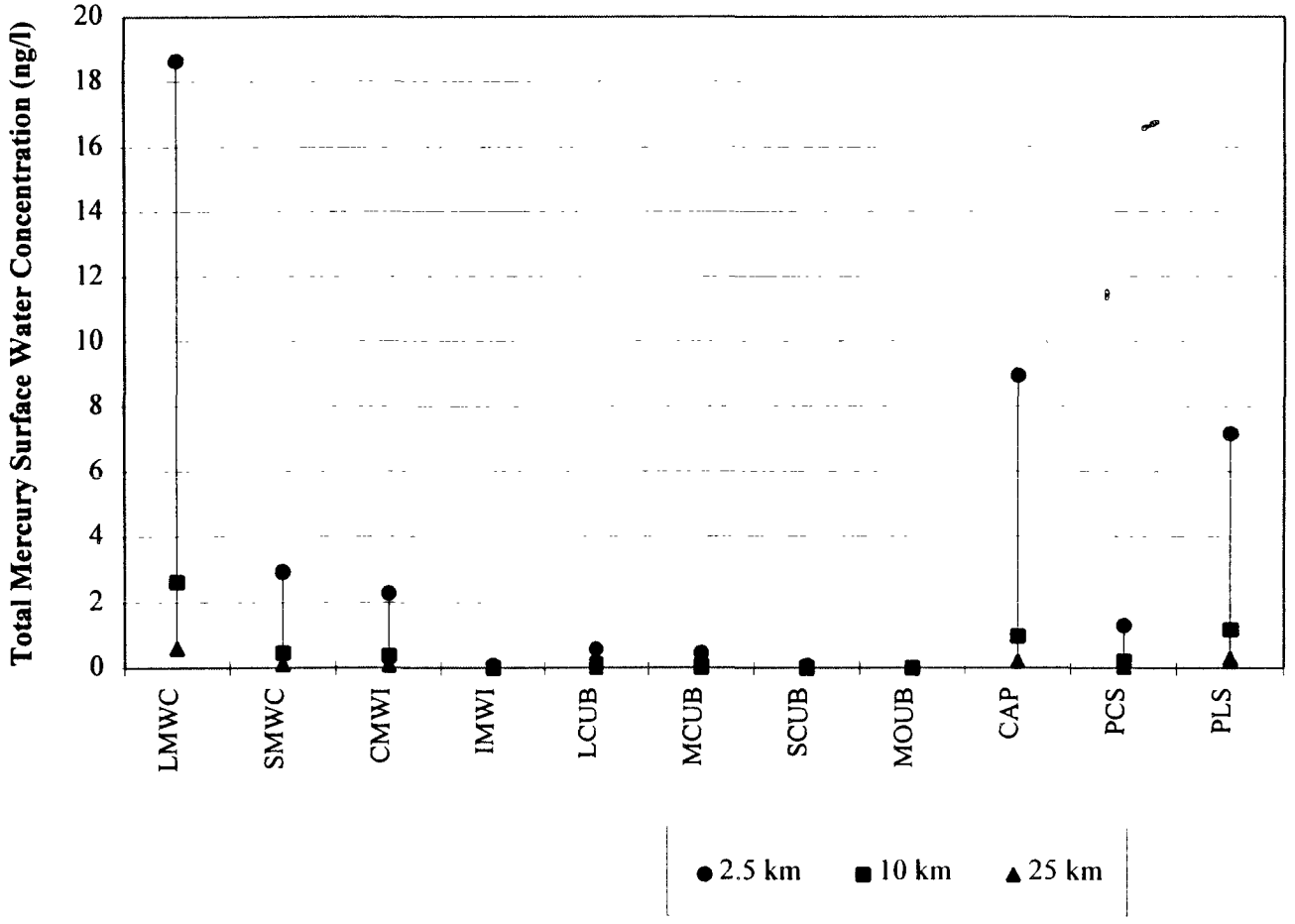


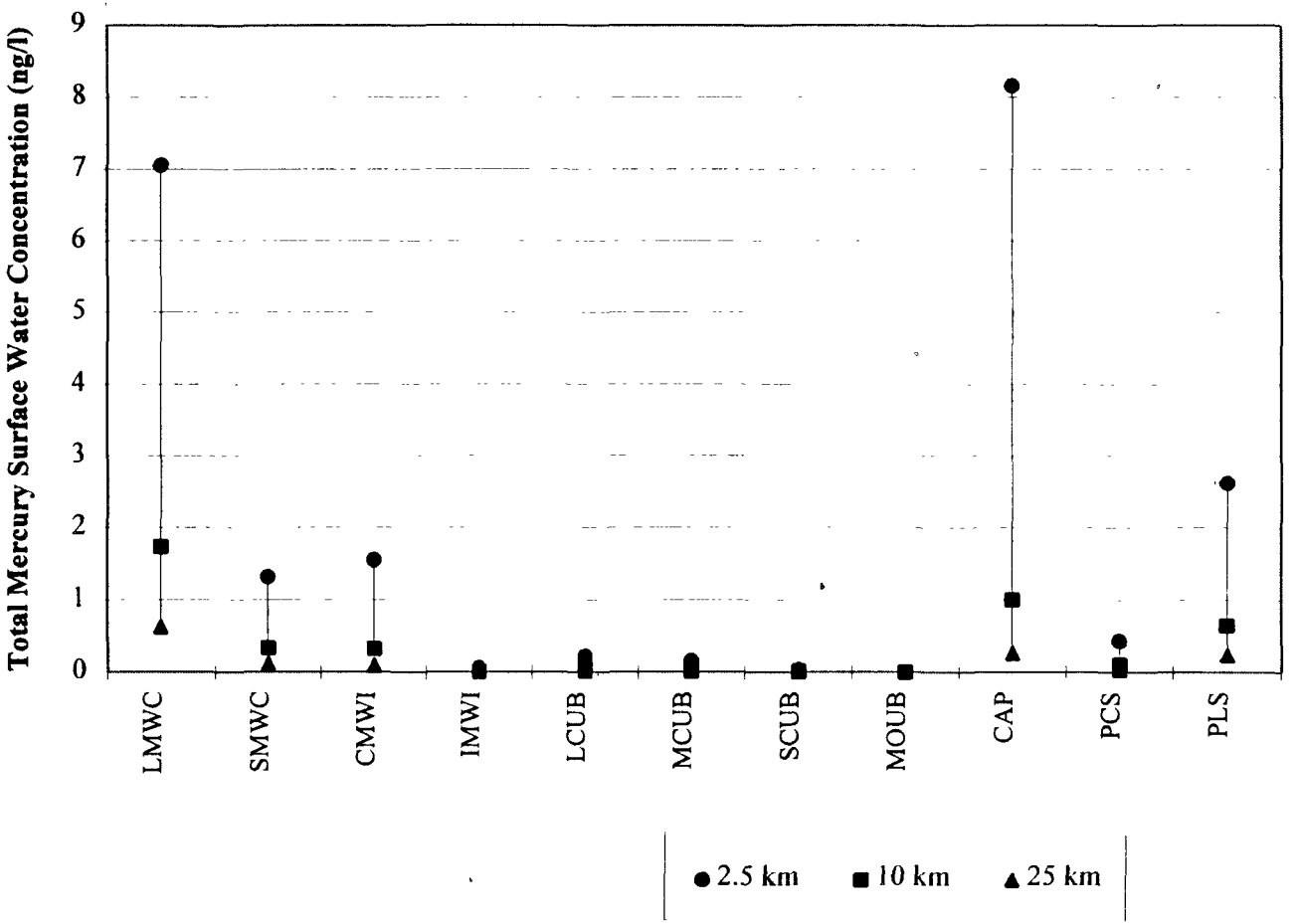
Figure 6-14
Predicted Total Mercury Surface Water Concentration (ng/l)
at the Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

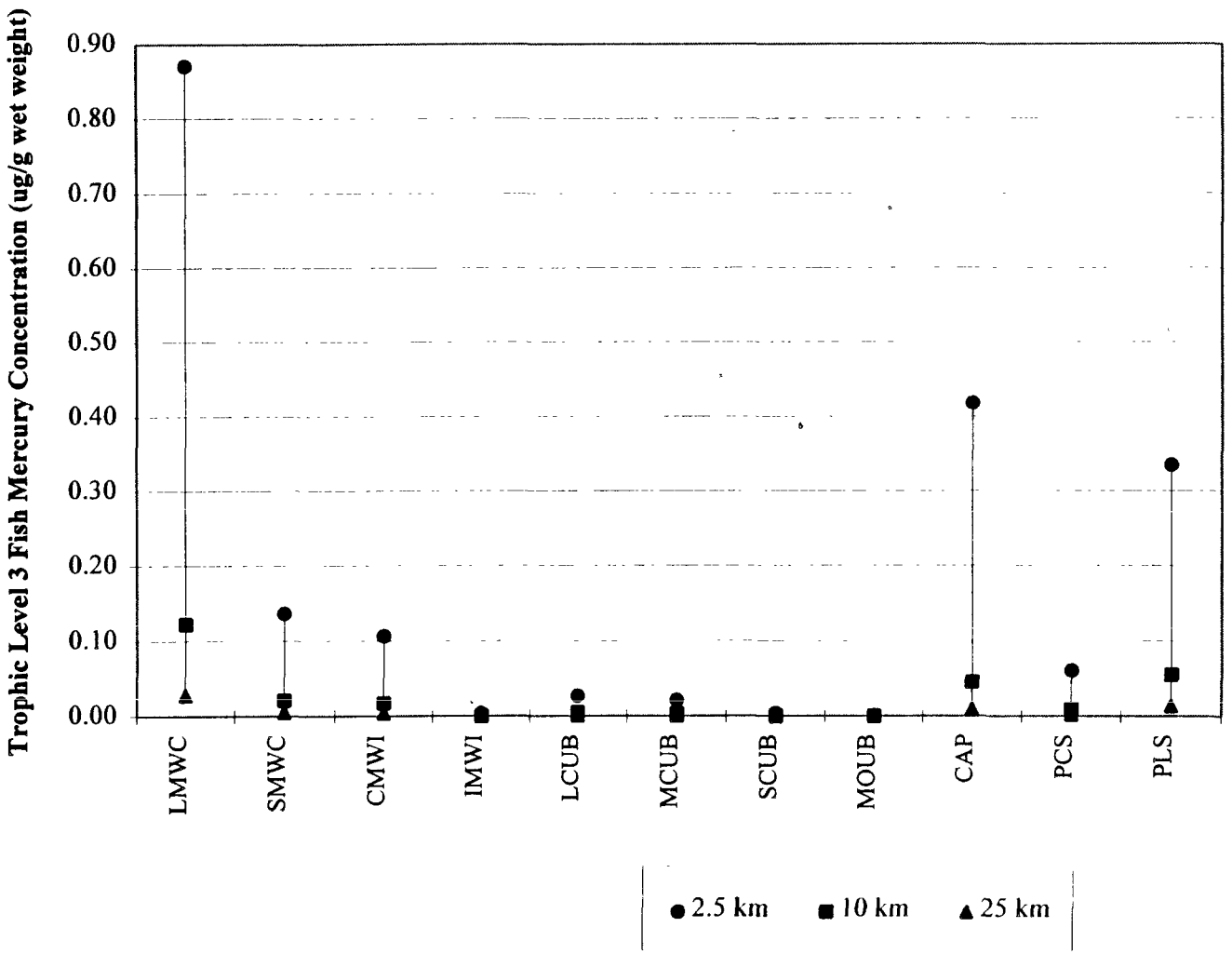
Figure 6-15
Predicted Total Mercury Surface Water Concentration (ng/l at the Western Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-16
Predicted Trophic Level 3 Fish Mercury Concentration (ug/g)
at the Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-17
Predicted Trophic Level 3 Fish Mercury Concentration (ug/g)
at the Western Site, COMPDEP Only

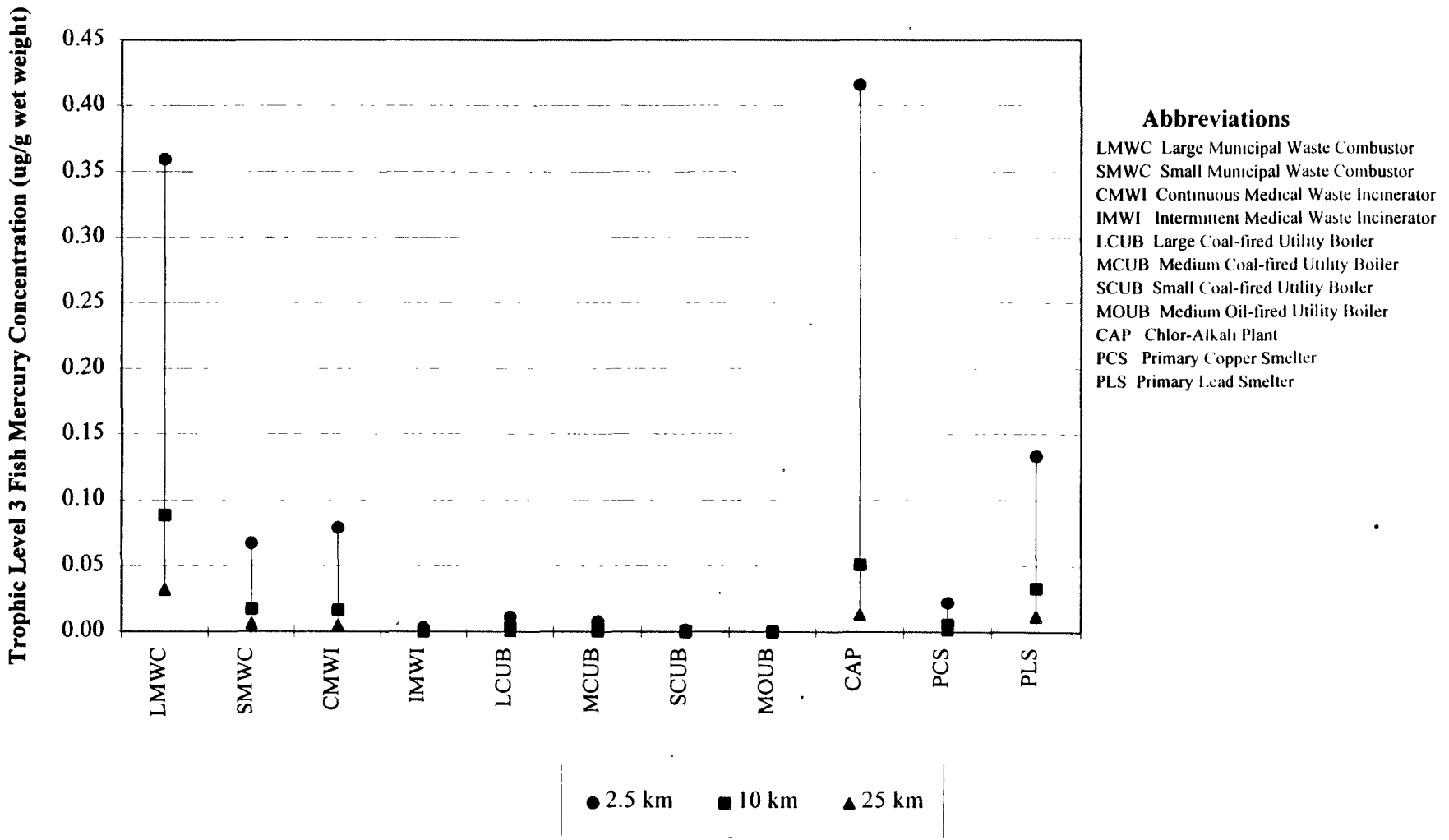
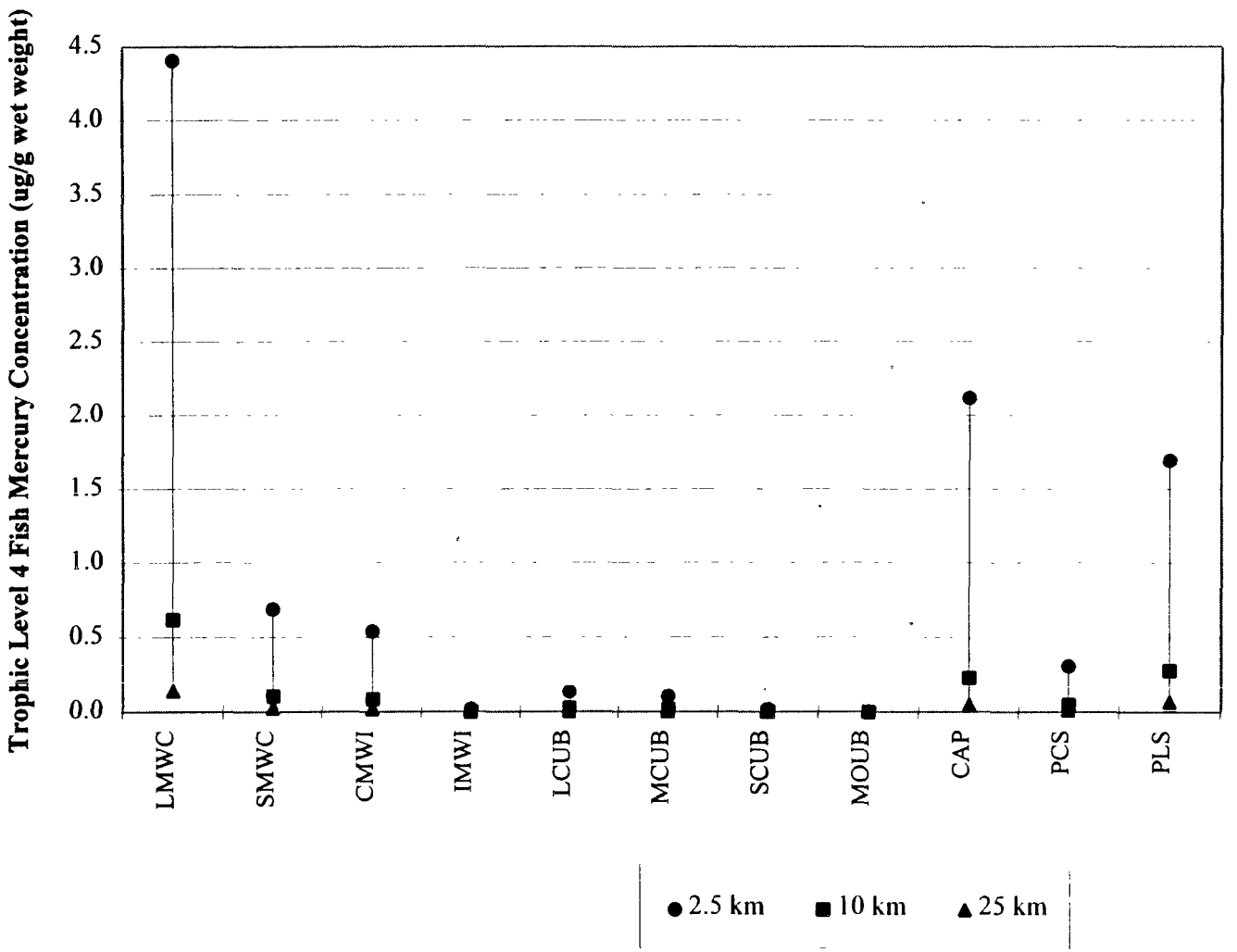


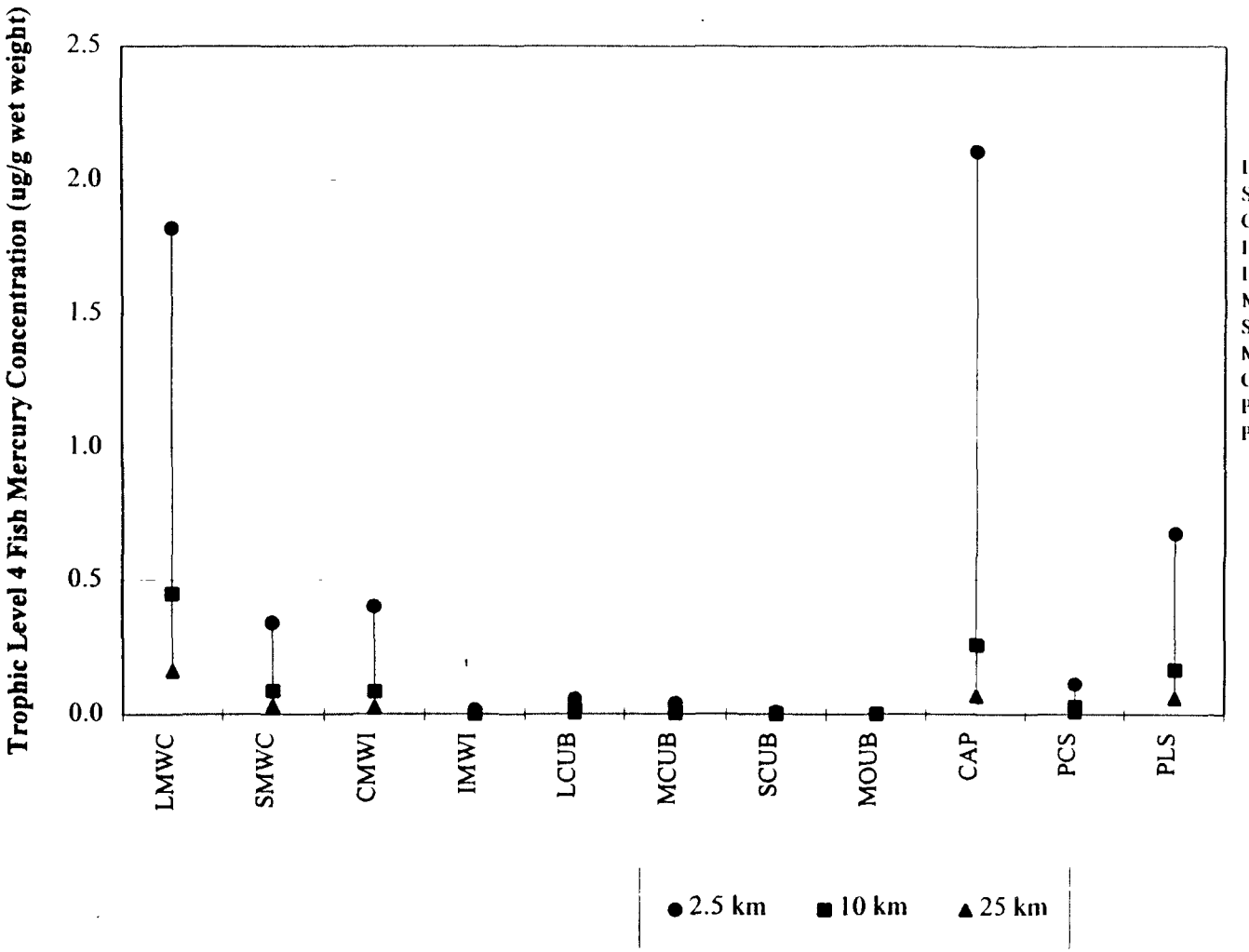
Figure 6-18
Predicted Trophic Level 4 Fish Mercury Concentration (ug/g)
at the Eastern Site, COMPDEP Only



Abbreviations

LMWC Large Municipal Waste Combustor
SMWC Small Municipal Waste Combustor
CMWI Continuous Medical Waste Incinerator
IMWI Intermittent Medical Waste Incinerator
LCUB Large Coal-fired Utility Boiler
MCUB Medium Coal-fired Utility Boiler
SCUB Small Coal-fired Utility Boiler
MOUB Medium Oil-fired Utility Boiler
CAP Chlor-Alkali Plant
PCS Primary Copper Smelter
PLS Primary Lead Smelter

Figure 6-19
Predicted Trophic Level 4 Fish Mercury Concentration (ug/g)
at the Western Site, COMPDEP Only



in fish presented in this section are within the range of observed values (see Chapter 2 of this Volume). The value used for the trophic level 4 fish bioaccumulation factor (BAF_4) was 335,000 L/kg, and the value for trophic level 3 fish (BAF_3) was 66,200 L/kg (see Volume V of this Report for more information on the derivation of these values). The bioaccumulation factor used for trophic level 4 fish resulted in fish concentrations of 1 μg/g (1 ug/g) when total dissolved mercury surface water concentrations were approximately 3 ng/L.

6.2.4 Human Intake

Tables 6-5 through 6-10 show the predicted human intakes for each scenario and model plant at both sites. The predicted mercury ingestion is the total from all ingestion exposure pathways considered for each scenario.

Comparison with the discussion of the ingestion results for the remote eastern site (Section 5.4) is useful in understanding the magnitude of the effects of mercury ingestion from point sources. For the adult rural subsistence farmer in the eastern site, increases of approximately 0.0 - 0.7 ug/day are predicted, depending on the model plant type and distance from the source. Most of this increase in intake is due to ingestion of plants. Increases are even less for rural home gardeners and urban high end receptors, due to the lower rates of ingestion of impacted plants. All western site increases in ingestion for these farming/gardening scenarios are lower than the eastern site values.

Additional mercury intake for high end fishers at the eastern site over all model plant types range from about 0.2-270 ug/day at 2.5 km, 0.0-37.1 ug/day at 10 km, and 0.0-8.4 ug/day at 25 km. It must be noted that there are no comparable measurement data. The values for the Recreational fishers are about half as large because of the assumed daily fish consumption rate. The ranges for the western site are slightly lower.

This analysis shows that increases in mercury ingestion (as methylmercury from fish) may result from local mercury sources impacting nearby lakes. This is a consequence of elevated local deposition affecting the lake mercury concentrations and subsequent bioaccumulation in fish. In contrast, for farming/gardening scenarios most of the mercury ingestion is from plants that are predicted to accumulate mercury mainly from the atmosphere. Air concentrations are generally not as affected by a local mercury source as are deposition rates.

6.2.5 Exposure for Wildlife Receptors

Tables 6-11 and 6-12 show the predicted methylmercury exposure rates for the wildlife receptors considered for each site.

Table 6-5
Predicted Human Mercury Intakes for Agricultural Scenarios in Eastern Site based on COMPDEP Results Alone

7/20/95, 4 54 41 PM	Predicted Mercury Intake for Rural Subsistence Farmer (mg/kg/day)											
COMPDEP in Eastern Site	2 5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2.4e-07	4.4e-05	7 2e-08	2 2e-05	2 1e-07	1 2e-05	6 3e-08	7 4e-06	1 1e-07	4 4e-06	3 3e-08	3 0e-06
Small Municipal Waste Combustor	1.0e-07	8 4e-06	3 0e-08	4 7e-06	6 0e-08	2.7e-06	1 8e-08	1 8e-06	2 6e-08	9 8e-07	8 0e-09	7 0e-07
Continuous Medical Waste Incinerator	2.2e-07	1 0e-05	6 7e-08	6 6e-06	7 7e-08	2 8e-06	2 3e-08	2 0e-06	2 7e-08	9 4e-07	8 3e-09	6 9e-07
Intermittent Medical Waste Incinerator	9.3e-09	3 9e-07	2 8e-09	2.6e-07	2 6e-09	9 5e-08	8 0e-10	6 8e-08	8 8e-10	3 0e-08	2 7e-10	2 2e-08
Large Coal-fired Utility Boiler	1.2e-09	1 3e-06	3 7e-10	5 8e-07	1 5e-09	3 4e-07	4.5e-10	1 6e-07	1 2e-09	9 6e-08	3 7e-10	5 5e-08
Medium Coal-fired Utility Boiler	2.1e-09	1.1e-06	6 5e-10	4 8e-07	2.5e-09	2 6e-07	7 7e-10	1 4e-07	2 1e-09	9 9e-08	6 2e-10	6 4e-08
Small Coal-fired Utility Boiler	2 4e-09	2.3e-07	7.3e-10	1.2e-07	1 9e-09	8 3e-08	5 7e-10	5 5e-08	9 1e-10	3.2e-08	2 8e-10	2 3e-08
Medium Oil-fired Utility Boiler	1 4e-10	3 3e-08	4 2e-11	1 6e-08	1 6e-10	9 8e-09	4 9e-11	5 9e-09	1 1e-10	4 1e-09	3 2e-11	2 9e-09
Chlor-alkali plant	3 6e-06	1 1e-04	1.1e-06	8.5e-05	7 9e-07	2.3e-05	2 4e-07	1 8e-05	2.7e-07	7 8e-06	8 3e-08	6 2e-06
Primary Copper Smelter	3.6e-08	3 7e-06	1 1e-08	2 0e-06	3 9e-08	1.6e-06	1 2e-08	1 1e-06	2 5e-08	8 2e-07	7 6e-09	6 2e-07
Primary Lead Smelter	3 9e-07	2 5e-05	1 2e-07	1 5e-05	3 6e-07	1 3e-05	1.1e-07	9 4e-06	1 9e-07	6 0e-06	5 9e-08	4 6e-06
7/20/95, 4:54 52 PM	Predicted Mercury Intake for Rural Home Gardener (mg/kg/day)											
COMPDEP in Eastern Site	2.5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2.4e-07	8 9e-06	7 2e-08	3 0e-06	2 1e-07	2.6e-06	6.3e-08	1 4e-06	1 1e-07	9 9e-07	3 3e-08	6 6e-07
Small Municipal Waste Combustor	1 0e-07	1.8e-06	3.0e-08	8 1e-07	6 0e-08	5.9e-07	1 8e-08	3 8e-07	2 6e-08	2 2e-07	8 0e-09	1 5e-07
Continuous Medical Waste Incinerator	2 2e-07	2.2e-06	6 7e-08	1.4e-06	7 7e-08	6 4e-07	2 3e-08	4 5e-07	2 7e-08	2 1e-07	8 3e-09	1 6e-07
Intermittent Medical Waste Incinerator	9.3e-09	8 6e-08	2 8e-09	5.6e-08	2 6e-09	2 1e-08	8.0e-10	1 5e-08	8 8e-10	6.7e-09	2 7e-10	5 0e-09
Large Coal-fired Utility Boiler	1 2e-09	2 5e-07	3.7e-10	6.5e-08	1.5e-09	6 7e-08	4.5e-10	2 2e-08	1 2e-09	2 0e-08	3 7e-10	9 6e-09
Medium Coal-fired Utility Boiler	2.1e-09	2 1e-07	6 5e-10	5.8e-08	2 5e-09	5.3e-08	7 7e-10	2 2e-08	2 1e-09	2 1e-08	6 2e-10	1 3e-08
Small Coal-fired Utility Boiler	2.4e-09	4.7e-08	7 3e-10	2.1e-08	1.9e-09	1 8e-08	5 7e-10	1 2e-08	9 1e-10	7.1e-09	2 8e-10	5 2e-09
Medium Oil-fired Utility Boiler	1 4e-10	6.5e-09	4 2e-11	2 1e-09	1 6e-10	2 1e-09	4 9e-11	1 1e-09	1 1e-10	9.0e-10	3 2e-11	6 2e-10
Chlor-alkali plant	3.6e-06	2.4e-05	1.1e-06	2 0e-05	7 9e-07	5 1e-06	2 4e-07	4 2e-06	2 7e-07	1 7e-06	8 3e-08	1 5e-06
Primary Copper Smelter	3.6e-08	7.4e-07	1 1e-08	3 2e-07	3 9e-08	3.4e-07	1 2e-08	2 3e-07	2 5e-08	1.8e-07	7 6e-09	1 4e-07
Primary Lead Smelter	3 9e-07	5.2e-06	1 2e-07	2 7e-06	3.6e-07	2.7e-06	1 1e-07	2 0e-06	1 9e-07	1 3e-06	5 9e-08	1 1e-06

Table 6-6
Predicted Human Mercury Intakes for Agricultural Scenarios in Western Site based on COMPDEP Results Alone

7/20/95, 5:35:51 PM	Predicted Mercury Intake for Rural Subsistence Farmer (mg/kg/day)											
COMPDEP in Western Site	2.5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2.2e-07	3.4e-05	6.7e-08	1.8e-05	1.8e-07	1.1e-05	5.5e-08	6.8e-06	1.1e-07	4.8e-06	3.3e-08	3.2e-06
Small Municipal Waste Combustor	8.8e-08	6.7e-06	2.7e-08	3.9e-06	5.2e-08	2.4e-06	1.6e-08	1.6e-06	2.5e-08	9.7e-07	7.5e-09	6.8e-07
Continuous Medical Waste Incinerator	2.1e-07	9.0e-06	6.2e-08	6.1e-06	6.8e-08	2.6e-06	2.1e-08	1.8e-06	2.5e-08	8.9e-07	7.5e-09	6.4e-07
Intermittent Medical Waste Incinerator	8.8e-09	3.6e-07	2.7e-09	2.5e-07	2.3e-09	8.6e-08	7.0e-10	6.1e-08	7.8e-10	2.7e-08	2.4e-10	2.0e-08
Large Coal-fired Utility Boiler	1.9e-09	1.2e-06	5.8e-10	5.6e-07	1.7e-09	4.0e-07	5.2e-10	2.0e-07	1.4e-09	1.4e-07	4.2e-10	7.8e-08
Medium Coal-fired Utility Boiler	2.4e-09	8.5e-07	7.3e-10	4.2e-07	2.2e-09	2.4e-07	6.6e-10	1.3e-07	1.9e-09	1.0e-07	5.7e-10	6.6e-08
Small Coal-fired Utility Boiler	2.2e-09	1.8e-07	6.7e-10	1.0e-07	1.6e-09	7.2e-08	4.8e-10	4.8e-08	9.0e-10	3.3e-08	2.7e-10	2.4e-08
Medium Oil-fired Utility Boiler	1.4e-10	2.4e-08	4.2e-11	1.3e-08	1.4e-10	8.6e-09	4.1e-11	5.3e-09	9.9e-11	4.1e-09	3.0e-11	2.9e-09
Chlor-alkali plant	3.6e-06	1.1e-04	1.1e-06	8.5e-05	7.3e-07	2.2e-05	2.2e-07	1.7e-05	2.5e-07	7.4e-06	7.7e-08	5.9e-06
Primary Copper Smelter	4.2e-08	3.1e-06	1.3e-08	1.8e-06	3.3e-08	1.4e-06	1.0e-08	9.4e-07	2.3e-08	7.8e-07	7.1e-09	5.9e-07
Primary Lead Smelter	3.8e-07	2.1e-05	1.1e-07	1.3e-05	3.0e-07	1.1e-05	9.0e-08	7.8e-06	1.8e-07	5.9e-06	5.6e-08	4.5e-06
7/20/95, 5:36:01 PM	Predicted Mercury Intake for Rural Home Gardener (mg/kg/day)											
COMPDEP in Western Site	2.5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2.2e-07	4.1e-06	6.7e-08	1.8e-06	1.8e-07	1.9e-06	5.5e-08	1.2e-06	1.1e-07	9.6e-07	3.3e-08	6.5e-07
Small Municipal Waste Combustor	8.8e-08	1.0e-06	2.7e-08	5.9e-07	5.2e-08	4.6e-07	1.6e-08	3.1e-07	2.5e-08	2.1e-07	7.5e-09	1.4e-07
Continuous Medical Waste Incinerator	2.1e-07	1.8e-06	6.2e-08	1.2e-06	6.8e-08	5.5e-07	2.1e-08	4.0e-07	2.5e-08	1.9e-07	7.5e-09	1.4e-07
Intermittent Medical Waste Incinerator	8.8e-09	7.4e-08	2.7e-09	5.2e-08	2.3e-09	1.8e-08	7.0e-10	1.3e-08	7.8e-10	6.0e-09	2.4e-10	4.5e-09
Large Coal-fired Utility Boiler	1.9e-09	1.0e-07	5.8e-10	3.2e-08	1.7e-09	4.2e-08	5.2e-10	1.7e-08	1.4e-09	1.8e-08	4.2e-10	9.7e-09
Medium Coal-fired Utility Boiler	2.4e-09	8.1e-08	7.3e-10	2.9e-08	2.2e-09	3.1e-08	6.6e-10	1.6e-08	1.9e-09	1.7e-08	5.7e-10	1.1e-08
Small Coal-fired Utility Boiler	2.2e-09	2.5e-08	6.7e-10	1.5e-08	1.6e-09	1.3e-08	4.8e-10	9.2e-09	9.0e-10	6.7e-09	2.7e-10	5.1e-09
Medium Oil-fired Utility Boiler	1.4e-10	2.7e-09	4.2e-11	1.2e-09	1.4e-10	1.3e-09	4.1e-11	8.4e-10	9.9e-11	7.9e-10	3.0e-11	5.7e-10
Chlor-alkali plant	3.6e-06	2.4e-05	1.1e-06	1.9e-05	7.3e-07	4.7e-06	2.2e-07	3.9e-06	2.5e-07	1.6e-06	7.7e-08	1.4e-06
Primary Copper Smelter	4.2e-08	4.2e-07	1.3e-08	2.6e-07	3.3e-08	2.4e-07	1.0e-08	1.8e-07	2.3e-08	1.5e-07	7.1e-09	1.3e-07
Primary Lead Smelter	3.8e-07	3.2e-06	1.1e-07	2.2e-06	3.0e-07	2.0e-06	9.0e-08	1.6e-06	1.8e-07	1.2e-06	5.6e-08	9.9e-07

Table 6-7
Predicted Human Mercury Intakes for Urban Scenarios in Eastern Site based on COMPDEP Results Alone

7/20/95, 4 54 55 PM	Predicted Mercury Intake for Urban Average (mg/kg/day)											
COMPDEP in Eastern Site	2 5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2.4e-07	6 5e-06	4 8e-08	7 9e-07	2 1e-07	1 2e-06	4 2e-08	1 5e-07	1.1e-07	3 1e-07	2 2e-08	3 8e-08
Small Municipal Waste Combustor	1.0e-07	1 0e-06	2 0e-08	1.2e-07	6 0e-08	2 1e-07	1 2e-08	2 6e-08	2 6e-08	6 0e-08	5.3e-09	7 2e-09
Continuous Medical Waste Incinerator	2.2e-07	8 0e-07	4 5e-08	9 7e-08	7 7e-08	1.7e-07	1 6e-08	2 0e-08	2 7e-08	4 6e-08	5 6e-09	5 6e-09
Intermittent Medical Waste Incinerator	9.3e-09	2 8e-08	1 9e-09	3 4e-09	2 6e-09	5 4e-09	5 3e-10	6.5e-10	8 8e-10	1 4e-09	1 8e-10	1 7e-10
Large Coal-fired Utility Boiler	1.2e-09	2.1e-07	2 5e-10	2 6e-08	1 5e-09	5 1e-08	3 0e-10	6 1e-09	1 2e-09	1 1e-08	2 5e-10	1 4e-09
Medium Coal-fired Utility Boiler	2 1e-09	1 7e-07	4 3e-10	2 1e-08	2 5e-09	3 3e-08	5.1e-10	4 0e-09	2 1e-09	8 1e-09	4 2e-10	9 9e-10
Small Coal-fired Utility Boiler	2 4e-09	2 9e-08	4 9e-10	3 5e-09	1 9e-09	5.9e-09	3 8e-10	7.2e-10	9 1e-10	1 6e-09	1 8e-10	1 9e-10
Medium Oil-fired Utility Boiler	1.4e-10	4.9e-09	2.8e-11	6 0e-10	1 6e-10	9 7e-10	3 3e-11	1 2e-10	1 1e-10	2 5e-10	2 1e-11	3 0e-11
Chlor-alkali plant	3.6e-06	3.0e-06	7 3e-07	3 6e-07	7 9e-07	4 4e-07	1 6e-07	5 3e-08	2 7e-07	1 1e-07	5 5e-08	1 3e-08
Primary Copper Smelter	3 6e-08	4 7e-07	7 3e-09	5.6e-08	3 9e-08	9 9e-08	7.9e-09	1.2e-08	2.5e-08	2.6e-08	5 1e-09	3 2e-09
Primary Lead Smelter	3 9e-07	2 5e-06	7 9e-08	3 1e-07	3 6e-07	5 6e-07	7 3e-08	6 8e-08	1.9e-07	1 5e-07	3 9e-08	1 9e-08

7/20/95, 4:54 59 PM	Predicted Mercury Intake for Urban High End (mg/kg/day)											
COMPDEP in Eastern Site	2.5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2 4e-07	2 4e-04	7 2e-08	1.6e-06	2.1e-07	4.6e-05	6.3e-08	6 3e-07	1 1e-07	1 2e-05	3 3e-08	2 7e-07
Small Municipal Waste Combustor	1.0e-07	3 8e-05	3 0e-08	3 8e-07	6 0e-08	7.9e-06	1 8e-08	1 6e-07	2 6e-08	2 2e-06	8 0e-09	6 2e-08
Continuous Medical Waste Incinerator	2.2e-07	3.0e-05	6 7e-08	5.7e-07	7 7e-08	6 3e-06	2.3e-08	1.8e-07	2 7e-08	1 7e-06	8 3e-09	6 2e-08
Intermittent Medical Waste Incinerator	9.3e-09	1.0e-06	2 8e-09	2 3e-08	2 6e-09	2.0e-07	8 0e-10	6 0e-09	8 8e-10	5 3e-08	2 7e-10	1 9e-09
Large Coal-fired Utility Boiler	1.2e-09	8.0e-06	3 7e-10	4.1e-08	1.5e-09	1.9e-06	4 5e-10	1 2e-08	1.2e-09	4 2e-07	3 7e-10	4.4e-09
Medium Coal-fired Utility Boiler	2.1e-09	6 3e-06	6.5e-10	3 5e-08	2.5e-09	1.2e-06	7.7e-10	1 1e-08	2.1e-09	3.0e-07	6 2e-10	5 4e-09
Small Coal-fired Utility Boiler	2 4e-09	1 1e-06	7 3e-10	9 8e-09	1.9e-09	2.2e-07	5 7e-10	4 7e-09	9 1e-10	5 8e-08	2 8e-10	2 0e-09
Medium Oil-fired Utility Boiler	1 4e-10	1 8e-07	4 2e-11	1 2e-09	1 6e-10	3.6e-08	4 9e-11	4.9e-10	1 1e-10	9 2e-09	3.2e-11	2 5e-10
Chlor-alkali plant	3 6e-06	1 1e-04	1.1e-06	7 4e-06	7.9e-07	1.6e-05	2.4e-07	1 6e-06	2 7e-07	4 0e-06	8.3e-08	5 4e-07
Primary Copper Smelter	3.6e-08	1 7e-05	1 1e-08	1 5e-07	3.9e-08	3.7e-06	1 2e-08	9.3e-08	2 5e-08	9 9e-07	7 6e-09	5 3e-08
Primary Lead Smelter	3.9e-07	9 4e-05	1 2e-07	1 2e-06	3.6e-07	2.1e-05	1 1e-07	7 9e-07	1 9e-07	5 7e-06	5 9e-08	4 0e-07

Table 6-8
Predicted Human Mercury Intakes for Urban Scenarios in Western Site based on COMPDEP Results Alone

7/20/95, 5 36 05 PM	Predicted Mercury Intake for Urban Average (mg/kg/day)											
COMPDEP in Western Site	2 5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2 2e-07	2 4e-06	4 5e-08	2 9e-07	1.8e-07	7 1e-07	3 7e-08	8 6e-08	1 1e-07	2 8e-07	2.2e-08	3 4e-08
Small Municipal Waste Combustor	8 8e-08	4 5e-07	1 8e-08	5 5e-08	5.2e-08	1.4e-07	1.1e-08	1 7e-08	2 5e-08	5 3e-08	5 0e-09	6 5e-09
Continuous Medical Waste Incinerator	2 1e-07	5.2e-07	4 2e-08	6 3e-08	6.8e-08	1 3e-07	1.4e-08	1 6e-08	2 5e-08	4 3e-08	5 0e-09	5 2e-09
Intermittent Medical Waste Incinerator	8.8e-09	1.9e-08	1.8e-09	2 4e-09	2 3e-09	4.2e-09	4 7e-10	5 1e-10	7 8e-10	1.3e-09	1.6e-10	1 6e-10
Large Coal-fired Utility Boiler	1.9e-09	7 9e-08	3.9e-10	9.5e-09	1 7e-09	2.7e-08	3.5e-10	3 3e-09	1 4e-09	8.9e-09	2.8e-10	1 1e-09
Medium Coal-fired Utility Boiler	2 4e-09	5 7e-08	4 9e-10	6 9e-09	2 2e-09	1 5e-08	4 4e-10	1 9e-09	1 9e-09	5 4e-09	3 8e-10	6 6e-10
Small Coal-fired Utility Boiler	2 2e-09	1 1e-08	4 4e-10	1.3e-09	1 6e-09	3.2e-09	3.2e-10	3 9e-10	9 0e-10	1 3e-09	1 8e-10	1 5e-10
Medium Oil-fired Utility Boiler	1 4e-10	1 6e-09	2 8e-11	1 9e-10	1 4e-10	4 7e-10	2.7e-11	5 7e-11	9 9e-11	1 8e-10	2.0e-11	2 2e-11
Chlor-alkali plant	3.6e-06	2 5e-06	7 2e-07	3.0e-07	7 3e-07	3.9e-07	1 5e-07	4 8e-08	2 5e-07	1.1e-07	5.2e-08	1 4e-08
Primary Copper Smelter	4 2e-08	1 5e-07	8 4e-09	1 9e-08	3 3e-08	4 3e-08	6 6e-09	5 3e-09	2 3e-08	1.7e-08	4 7e-09	2 1e-09
Primary Lead Smelter	3 8e-07	8 9e-07	7 7e-08	1 1e-07	3 0e-07	2 6e-07	6 0e-08	3 2e-08	1 8e-07	1.1e-07	3 7e-08	1 3e-08

7/20/95, 5:36 08 PM	Predicted Mercury Intake for Urban High End (mg/kg/day)											
COMPDEP in Western Site	2.5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2 2e-07	8 9e-05	6 7e-08	8 7e-07	1.8e-07	2.6e-05	5.5e-08	4 8e-07	1 1e-07	1.1e-05	3 3e-08	2 6e-07
Small Municipal Waste Combustor	8 8e-08	1.7e-05	2.7e-08	2 5e-07	5 2e-08	5 1e-06	1 6e-08	1 3e-07	2 5e-08	2.0e-06	7.5e-09	5.7e-08
Continuous Medical Waste Incinerator	2 1e-07	1 9e-05	6 2e-08	4 9e-07	6 8e-08	4.8e-06	2.1e-08	1 6e-07	2 5e-08	1 6e-06	7 5e-09	5 6e-08
Intermittent Medical Waste Incinerator	8.8e-09	7 2e-07	2 7e-09	2 1e-08	2.3e-09	1.6e-07	7.0e-10	5 2e-09	7 8e-10	4 8e-08	2.4e-10	1 7e-09
Large Coal-fired Utility Boiler	1.9e-09	2.9e-06	5 8e-10	1 8e-08	1 7e-09	1 0e-06	5.2e-10	8.2e-09	1 4e-09	3 3e-07	4 2e-10	4 3e-09
Medium Coal-fired Utility Boiler	2 4e-09	2.1e-06	7 3e-10	1 5e-08	2 2e-09	5.7e-07	6 6e-10	7 0e-09	1 9e-09	2.0e-07	5 7e-10	4 6e-09
Small Coal-fired Utility Boiler	2.2e-09	4.0e-07	6 7e-10	6.2e-09	1.6e-09	1.2e-07	4.8e-10	3 6e-09	9 0e-10	4.7e-08	2 7e-10	2 0e-09
Medium Oil-fired Utility Boiler	1 4e-10	5.9e-08	4.2e-11	5 6e-10	1 4e-10	1.7e-08	4 1e-11	3 5e-10	9 9e-11	6.7e-09	3.0e-11	2 2e-10
Chlor-alkali plant	3.6e-06	9.3e-05	1.1e-06	7 3e-06	7 3e-07	1.5e-05	2.2e-07	1.5e-06	2 5e-07	4 2e-06	7.7e-08	5 1e-07
Primary Copper Smelter	4 2e-08	5 7e-06	1 3e-08	1.1e-07	3 3e-08	1 6e-06	1.0e-08	7.1e-08	2.3e-08	6.4e-07	7.1e-09	4 8e-08
Primary Lead Smelter	3 8e-07	3 3e-05	1 1e-07	8 8e-07	3.0e-07	9.8e-06	9.0e-08	6 2e-07	1 8e-07	3.9e-06	5 6e-08	3 7e-07

Table 6-9
Predicted Human Mercury Intakes for Fish Ingestion Scenarios in Eastern Site based on COMPDEP Results Alone

7/20/95, 4 55 02 PM	Predicted Mercury Intake for High End Fish Ingestion (mg/kg/day)											
COMPDEP in Eastern Site	2.5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2.4e-07	5 2e-03	7.2e-08	3.8e-03	2.1e-07	7 3e-04	6 3e-08	5 3e-04	1 1e-07	1 7e-04	3 3e-08	1 2e-04
Small Municipal Waste Combustor	1.0e-07	8 2e-04	3 0e-08	5 9e-04	6.0e-08	1 3e-04	1.8e-08	9 1e-05	2 6e-08	3 2e-05	8.0e-09	2 3e-05
Continuous Medical Waste Incinerator	2 2e-07	6 4e-04	6 7e-08	4 6e-04	7.7e-08	1 0e-04	2.3e-08	7 4e-05	2 7e-08	2 5e-05	8 3e-09	1 8e-05
Intermittent Medical Waste Incinerator	9.3e-09	2 2e-05	2 8e-09	1 6e-05	2.6e-09	3.3e-06	8 0e-10	2 4e-06	8 8e-10	7 8e-07	2 7e-10	5 7e-07
Large Coal-fired Utility Boiler	1.2e-09	1.6e-04	3.7e-10	1 2e-04	1.5e-09	3.0e-05	4 5e-10	2 2e-05	1 2e-09	5.9e-06	3 7e-10	4 3e-06
Medium Coal-fired Utility Boiler	2.1e-09	1.3e-04	6 5e-10	9 1e-05	2.5e-09	2 0e-05	7 7e-10	1 5e-05	2 1e-09	4.4e-06	6 2e-10	3 2e-06
Small Coal-fired Utility Boiler	2 4e-09	2 2e-05	7.3e-10	1 6e-05	1.9e-09	3.5e-06	5 7e-10	2 6e-06	9 1e-10	8 5e-07	2.8e-10	6 2e-07
Medium Oil-fired Utility Boiler	1.4e-10	3 9e-06	4.2e-11	2 9e-06	1.6e-10	5 8e-07	4 9e-11	4 2e-07	1 1e-10	1 3e-07	3 2e-11	9 7e-08
Chlor-alkali plant	3 6e-06	2 5e-03	1 1e-06	1.8e-03	7.9e-07	2 7e-04	2 4e-07	2 0e-04	2 7e-07	6 0e-05	8 3e-08	4 4e-05
Primary Copper Smelter	3.6e-08	3 6e-04	1 1e-08	2 6e-04	3.9e-08	5 9e-05	1 2e-08	4 3e-05	2 5e-08	1 4e-05	7 6e-09	1 0e-05
Primary Lead Smelter	3 9e-07	2.0e-03	1 2e-07	1 5e-03	3 6e-07	3 3e-04	1 1e-07	2 4e-04	1 9e-07	8 3e-05	5 9e-08	6 1e-05

7/20/95, 4:55 06 PM	Predicted Mercury Intake for Recreational Angler (mg/kg/day)											
COMPDEP in Eastern Site	2.5 km				10 km				25 km			
			Adult				Adult				Adult	
Facility[a]			Inhalation	Ingestion			Inhalation	Ingestion			Inhalation	Ingestion
Large Municipal Waste Combustor			7.2e-08	1 9e-03			6.3e-08	2 7e-04			3.3e-08	6 1e-05
Small Municipal Waste Combustor			3 0e-08	3 0e-04			1.8e-08	4 6e-05			8.0e-09	1 2e-05
Continuous Medical Waste Incinerator			6.7e-08	2.3e-04			2.3e-08	3 7e-05			8 3e-09	9 2e-06
Intermittent Medical Waste Incinerator			2 8e-09	8 1e-06			8.0e-10	1 2e-06			2.7e-10	2 8e-07
Large Coal-fired Utility Boiler			3.7e-10	5.8e-05			4.5e-10	1 1e-05			3 7e-10	2 2e-06
Medium Coal-fired Utility Boiler			6.5e-10	4.6e-05			7 7e-10	7.5e-06			6.2e-10	1 6e-06
Small Coal-fired Utility Boiler			7 3e-10	8.2e-06			5.7e-10	1 3e-06			2 8e-10	3 1e-07
Medium Oil-fired Utility Boiler			4.2e-11	1 4e-06			4.9e-11	2 1e-07			3.2e-11	4 8e-08
Chlor-alkali plant			1 1e-06	9 1e-04			2 4e-07	9.8e-05			8.3e-08	2 1e-05
Primary Copper Smelter			1 1e-08	1.3e-04			1.2e-08	2.1e-05			7 6e-09	5 2e-06
Primary Lead Smelter			1 2e-07	7.3e-04			1.1e-07	1 2e-04			5 9e-08	3 0e-05

Table 6-10
Predicted Human Mercury Intakes for Fish Ingestion Scenarios in Western Site based on COMPDEP Results Alone

7/20/95, 5:36:11 PM	Predicted Mercury Intake for High End Fish Ingestion (mg/kg/day)											
COMPDEP in Western Site	2.5 km				10 km				25 km			
	Child		Adult		Child		Adult		Child		Adult	
Facility[a]	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion	Inhalation	Ingestion
Large Municipal Waste Combustor	2.2e-07	2.1e-03	6.7e-08	1.6e-03	1.8e-07	5.3e-04	5.5e-08	3.9e-04	1.1e-07	1.9e-04	3.3e-08	1.4e-04
Small Municipal Waste Combustor	8.8e-08	4.0e-04	2.7e-08	2.9e-04	5.2e-08	1.0e-04	1.6e-08	7.5e-05	2.5e-08	3.6e-05	7.5e-09	2.6e-05
Continuous Medical Waste Incinerator	2.1e-07	4.7e-04	6.2e-08	3.4e-04	6.8e-08	9.9e-05	2.1e-08	7.2e-05	2.5e-08	2.9e-05	7.5e-09	2.1e-05
Intermittent Medical Waste Incinerator	8.8e-09	1.8e-05	2.7e-09	1.3e-05	2.3e-09	3.2e-06	7.0e-10	2.3e-06	7.8e-10	8.8e-07	2.4e-10	6.4e-07
Large Coal-fired Utility Boiler	1.9e-09	6.7e-05	5.8e-10	4.9e-05	1.7e-09	2.0e-05	5.2e-10	1.5e-05	1.4e-09	6.3e-06	4.2e-10	4.6e-06
Medium Coal-fired Utility Boiler	2.4e-09	4.8e-05	7.3e-10	3.5e-05	2.2e-09	1.2e-05	6.6e-10	8.7e-06	1.9e-09	3.8e-06	5.7e-10	2.8e-06
Small Coal-fired Utility Boiler	2.2e-09	9.6e-06	6.7e-10	7.0e-06	1.6e-09	2.4e-06	4.8e-10	1.8e-06	9.0e-10	8.6e-07	2.7e-10	6.3e-07
Medium Oil-fired Utility Boiler	1.4e-10	1.4e-06	4.2e-11	1.0e-06	1.4e-10	3.5e-07	4.1e-11	2.6e-07	9.9e-11	1.2e-07	3.0e-11	9.0e-08
Chlor-alkali plant	3.6e-06	2.5e-03	1.1e-06	1.8e-03	7.3e-07	3.1e-04	2.2e-07	2.3e-04	2.5e-07	8.1e-05	7.7e-08	5.9e-05
Primary Copper Smelter	4.2e-08	1.3e-04	1.3e-08	9.7e-05	3.3e-08	3.3e-05	1.0e-08	2.4e-05	2.3e-08	1.2e-05	7.1e-09	8.6e-06
Primary Lead Smelter	3.8e-07	8.0e-04	1.1e-07	5.8e-04	3.0e-07	2.0e-04	9.0e-08	1.5e-04	1.8e-07	7.3e-05	5.6e-08	5.3e-05

7/20/95, 5:36:15 PM	Predicted Mercury Intake for Recreational Angler (mg/kg/day)											
COMPDEP in Western Site	2.5 km				10 km				25 km			
			Adult				Adult				Adult	
Facility[a]			Inhalation	Ingestion			Inhalation	Ingestion			Inhalation	Ingestion
Large Municipal Waste Combustor			6.7e-08	7.8e-04			5.5e-08	1.9e-04			3.3e-08	7.0e-05
Small Municipal Waste Combustor			2.7e-08	1.5e-04			1.6e-08	3.8e-05			7.5e-09	1.3e-05
Continuous Medical Waste Incinerator			6.2e-08	1.7e-04			2.1e-08	3.6e-05			7.5e-09	1.1e-05
Intermittent Medical Waste Incinerator			2.7e-09	6.7e-06			7.0e-10	1.2e-06			2.4e-10	3.2e-07
Large Coal-fired Utility Boiler			5.8e-10	2.5e-05			5.2e-10	7.4e-06			4.2e-10	2.3e-06
Medium Coal-fired Utility Boiler			7.3e-10	1.7e-05			6.6e-10	4.4e-06			5.7e-10	1.4e-06
Small Coal-fired Utility Boiler			6.7e-10	3.5e-06			4.8e-10	8.9e-07			2.7e-10	3.1e-07
Medium Oil-fired Utility Boiler			4.2e-11	5.1e-07			4.1e-11	1.3e-07			3.0e-11	4.5e-08
Chlor-alkali plant			1.1e-06	9.1e-04			2.2e-07	1.1e-04			7.7e-08	2.9e-05
Primary Copper Smelter			1.3e-08	4.9e-05			1.0e-08	1.2e-05			7.1e-09	4.3e-06
Primary Lead Smelter			1.1e-07	2.9e-04			9.0e-08	7.2e-05			5.6e-08	2.6e-05

Table 6-11
Predicted Intakes for Wildlife Receptors for the Eastern Site based on COMPDEP Results

Percent of surface water concentration dissolved 70 5%		Methylmercury Fish Concentrations (ug/g)		Predicted Methylmercury Intakes (mg/kg/day)				
Facility	Mercury Surface Water Concentration (ng/l)	Trophic 3	Trophic 4	Bald Eagle	Osprey	Kingfisher	River Otter	Mink
2 5 km								
Large Municipal Waste Combustor	1 9E+01	8.7E-01	4.4E+00	1.6E-01	1.7E-01	4.4E-01	2.6E-01	1.7E-01
Small Municipal Waste Combustor	2.9E+00	1 4E-01	6 9E-01	2 5E-02	2.7E-02	6.8E-02	4 1E-02	2 7E-02
Continuous Medical Waste Incinerator	2.3E+00	1.1E-01	5 4E-01	1.9E-02	2.1E-02	5 3E-02	3 2E-02	2 1E-02
Intermittent MWI	8 0E-02	3 7E-03	1.9E-02	6.7E-04	7 5E-04	1.9E-03	1 1E-03	7.5E-04
Large Coal-fired Utility Boiler	5.7E-01	2.7E-02	1.3E-01	4.8E-03	5 3E-03	1.3E-02	8.0E-03	5 3E-03
Medium Coal-fired Utility Boiler	4 5E-01	2 1E-02	1.1E-01	3.8E-03	4.2E-03	1.1E-02	6 3E-03	4 2E-03
Small Coal-fired Utility Boiler	8 1E-02	3.8E-03	1.9E-02	6.8E-04	7 5E-04	1.9E-03	1 1E-03	7.5E-04
Medium Oil-fired Utility Boiler	1 4E-02	6 6E-04	3 3E-03	1 2E-04	1.3E-04	3 3E-04	2 0E-04	1 3E-04
Chlor-alkali plant	9.0E+00	4.2E-01	2.1E+00	7.5E-02	8.4E-02	2.1E-01	1 3E-01	8 4E-02
Primary Copper Smelter	1 3E+00	6 1E-02	3 1E-01	1.1E-02	1 2E-02	3 0E-02	1.8E-02	1.2E-02
Primary Lead Smelter	7.2E+00	3.4E-01	1 7E+00	6 0E-02	6.7E-02	1.7E-01	1.0E-01	6 7E-02
10 km								
Large Municipal Waste Combustor	2.6E+00	1 2E-01	6 2E-01	2.2E-02	2.5E-02	6.1E-02	3 7E-02	2.5E-02
Small Municipal Waste Combustor	4 5E-01	2.1E-02	1 1E-01	3 8E-03	4.2E-03	1.1E-02	6 3E-03	4 2E-03
Continuous Medical Waste Incinerator	3.6E-01	1.7E-02	8 6E-02	3.0E-03	3 4E-03	8.5E-03	5.1E-03	3 4E-03
Intermittent MWI	1 2E-02	5.4E-04	2.8E-03	9.8E-05	1.1E-04	2.7E-04	1.6E-04	1.1E-04
Large Coal-fired Utility Boiler	1.1E-01	5.0E-03	2.5E-02	8.9E-04	9 9E-04	2.5E-03	1.5E-03	1.0E-03
Medium Coal-fired Utility Boiler	7 3E-02	3 4E-03	1.7E-02	6.2E-04	6.9E-04	1.7E-03	1.0E-03	6.9E-04
Small Coal-fired Utility Boiler	1.3E-02	5.9E-04	3.0E-03	1.1E-04	1.2E-04	2.9E-04	1.8E-04	1.2E-04
Medium Oil-fired Utility Boiler	2 1E-03	9.8E-05	4.9E-04	1.8E-05	2.0E-05	4.9E-05	2 9E-05	2.0E-05
Chlor-alkali plant	9.7E-01	4.5E-02	2.3E-01	8.1E-03	9 0E-03	2.3E-02	1.3E-02	9 0E-03
Primary Copper Smelter	2 1E-01	9 8E-03	4.9E-02	1 8E-03	2.0E-03	4.9E-03	2.9E-03	2 0E-03
Primary Lead Smelter	1.2E+00	5.5E-02	2.8E-01	9 9E-03	1.1E-02	2.7E-02	1.6E-02	1.1E-02
25 km								
Large Municipal Waste Combustor	6 0E-01	2.8E-02	1 4E-01	5 1E-03	5 6E-03	1 4E-02	8 4E-03	5 6E-03
Small Municipal Waste Combustor	1 2E-01	5 4E-03	2 7E-02	9 7E-04	1 1E-03	2 7E-03	1.6E-03	1.1E-03
Continuous Medical Waste Incinerator	9 0E-02	4.2E-03	2 1E-02	7 6E-04	8 4E-04	2.1E-03	1.3E-03	8 4E-04
Intermittent MWI	2.8E-03	1 3E-04	6.5E-04	2.3E-05	2.6E-05	6 5E-05	3 9E-05	2 6E-05
Large Coal-fired Utility Boiler	2.1E-02	9.9E-04	5 0E-03	1.8E-04	2.0E-04	5.0E-04	3 0E-04	2.0E-04
Medium Coal-fired Utility Boiler	1.6E-02	7 3E-04	3.7E-03	1 3E-04	1.5E-04	3.7E-04	2.2E-04	1 5E-04
Small Coal-fired Utility Boiler	3.0E-03	1.4E-04	7.1E-04	2.5E-05	2.8E-05	7.0E-05	4.2E-05	2 8E-05
Medium Oil-fired Utility Boiler	4.7E-04	2.2E-05	1.1E-04	4 0E-06	4.4E-06	1 1E-05	6.6E-06	4 4E-06
Chlor-alkali plant	2.1E-01	9.8E-03	4.9E-02	1.8E-03	2.0E-03	4.9E-03	2 9E-03	2.0E-03
Primary Copper Smelter	5.1E-02	2.4E-03	1.2E-02	4.3E-04	4 8E-04	1.2E-03	7 1E-04	4.8E-04
Primary Lead Smelter	2 9E-01	1 4E-02	7 0E-02	2.5E-03	2.8E-03	6 9E-03	4.1E-03	2 8E-03

Table 6-12
Predicted Intakes for Wildlife Receptors for the Western Site Based on COMPDEP Results

Percent of surface water concentration dissolved: 77%		Methylmercury Fish Concentrations (ug/g)		Predicted Methylmercury Intakes (mg/kg/day)				
Plant	Mercury Surface Water Concentration (ng/l)	Trophic 3	Trophic 4	Bald Eagle	Osprey	Kingfisher	River Otter	Mink
2.5 km								
Large Municipal Waste Combustor	7.1E+00	3.6E-01	1.8E+00	6.5E-02	7.2E-02	1.8E-01	1.1E-01	7.2E-02
Small Municipal Waste Combustor	1.3E+00	6.7E-02	3.4E-01	1.2E-02	1.3E-02	3.4E-02	2.0E-02	1.3E-02
Continuous Medical Waste Incinerator	1.6E+00	7.9E-02	4.0E-01	1.4E-02	1.6E-02	4.0E-02	2.4E-02	1.6E-02
Intermittent Medical Waste Incinerator	6.0E-02	3.1E-03	1.6E-02	5.5E-04	6.1E-04	1.5E-03	9.2E-04	6.1E-04
Large Coal-fired Utility Boiler	2.2E-01	1.1E-02	5.7E-02	2.0E-03	2.3E-03	5.7E-03	3.4E-03	2.3E-03
Medium Coal-fired Utility Boiler	1.6E-01	8.0E-03	4.0E-02	1.4E-03	1.6E-03	4.0E-03	2.4E-03	1.6E-03
Small Coal-fired Utility Boiler	3.2E-02	1.6E-03	8.2E-03	2.9E-04	3.2E-04	8.1E-04	4.8E-04	3.2E-04
Medium Oil-fired Utility Boiler	4.6E-03	2.3E-04	1.2E-03	4.2E-05	4.6E-05	1.2E-04	6.9E-05	4.7E-05
Chlor-alkali plant	8.2E+00	4.2E-01	2.1E+00	7.5E-02	8.3E-02	2.1E-01	1.2E-01	8.3E-02
Primary Copper Smelter	4.4E-01	2.2E-02	1.1E-01	4.0E-03	4.5E-03	1.1E-02	6.7E-03	4.5E-03
Primary Lead Smelter	2.6E+00	1.3E-01	6.8E-01	2.4E-02	2.7E-02	6.7E-02	4.0E-02	2.7E-02
10 km								
Large Municipal Waste Combustor	1.7E+00	8.9E-02	4.5E-01	1.6E-02	1.8E-02	4.4E-02	2.6E-02	1.8E-02
Small Municipal Waste Combustor	3.4E-01	1.7E-02	8.7E-02	3.1E-03	3.4E-03	8.6E-03	5.1E-03	3.4E-03
Continuous Medical Waste Incinerator	3.2E-01	1.6E-02	8.3E-02	3.0E-03	3.3E-03	8.2E-03	4.9E-03	3.3E-03
Intermittent Medical Waste Incinerator	1.0E-02	5.3E-04	2.7E-03	9.6E-05	1.1E-04	2.7E-04	1.6E-04	1.1E-04
Large Coal-fired Utility Boiler	6.6E-02	3.4E-03	1.7E-02	6.1E-04	6.8E-04	1.7E-03	1.0E-03	6.8E-04
Medium Coal-fired Utility Boiler	3.9E-02	2.0E-03	1.0E-02	3.6E-04	4.0E-04	1.0E-03	6.0E-04	4.0E-04
Small Coal-fired Utility Boiler	8.0E-03	4.1E-04	2.1E-03	7.3E-05	8.1E-05	2.0E-04	1.2E-04	8.1E-05
Medium Oil-fired Utility Boiler	1.2E-03	5.9E-05	3.0E-04	1.1E-05	1.2E-05	2.9E-05	1.8E-05	1.2E-05
Chlor-alkali plant	1.0E+00	5.1E-02	2.6E-01	9.2E-03	1.0E-02	2.6E-02	1.5E-02	1.0E-02
Primary Copper Smelter	1.1E-01	5.5E-03	2.8E-02	9.9E-04	1.1E-03	2.8E-03	1.6E-03	1.1E-03
Primary Lead Smelter	6.5E-01	3.3E-02	1.7E-01	5.9E-03	6.6E-03	1.7E-02	9.9E-03	6.6E-03
25 km								
Large Municipal Waste Combustor	6.3E-01	3.2E-02	1.6E-01	5.8E-03	6.4E-03	1.6E-02	9.6E-03	6.4E-03
Small Municipal Waste Combustor	1.2E-01	6.0E-03	3.0E-02	1.1E-03	1.2E-03	3.0E-03	1.8E-03	1.2E-03
Continuous Medical Waste Incinerator	9.5E-02	4.8E-03	2.4E-02	8.7E-04	9.7E-04	2.4E-03	1.4E-03	9.7E-04
Intermittent Medical Waste Incinerator	2.9E-03	1.5E-04	7.4E-04	2.6E-05	2.9E-05	7.3E-05	4.4E-05	2.9E-05
Large Coal-fired Utility Boiler	2.1E-02	1.1E-03	5.3E-03	1.9E-04	2.1E-04	5.3E-04	3.1E-04	2.1E-04
Medium Coal-fired Utility Boiler	1.2E-02	6.3E-04	3.2E-03	1.1E-04	1.3E-04	3.2E-04	1.9E-04	1.3E-04
Small Coal-fired Utility Boiler	2.8E-03	1.4E-04	7.3E-04	2.6E-05	2.9E-05	7.2E-05	4.3E-05	2.9E-05
Medium Oil-fired Utility Boiler	4.0E-04	2.1E-05	1.0E-04	3.7E-06	4.1E-06	1.0E-05	6.2E-06	4.1E-06
Chlor-alkali plant	2.6E-01	1.3E-02	6.8E-02	2.4E-03	2.7E-03	6.7E-03	4.0E-03	2.7E-03
Primary Copper Smelter	3.8E-02	2.0E-03	9.9E-03	3.5E-04	3.9E-04	9.8E-04	5.9E-04	3.9E-04
Primary Lead Smelter	2.4E-01	1.2E-02	6.1E-02	2.1E-03	2.4E-03	6.0E-03	3.6E-03	2.4E-03

Animals with the highest ingestion rate (per body weight) generally had the highest methylmercury intakes, except in cases where a species prefers preying on trophic 4 fish. For example, the river otter eats less on a per weight basis than the osprey but eats more highly contaminated fish, and receives a higher mercury exposure as a result.

6.2.6 Mass Balances within the Local-Scale Domain

In this section the fraction of the mercury emitted from each hypothetical facility that is predicted to deposit within 50 km is estimated. The area-averaged wet and dry deposition rates are also estimated based on the fraction from the single source that is predicted to deposit within 50 km.

Tables 6-13 and 6-14 show the results for all facilities at both sites. These results were obtained by using a total of 480 receptors for each facility and site. The receptors were placed in 16 directions around the facility and 30 distances, from 0.5 km to 50 km. The difference between the lower bound and upper bound estimates is due to the method of interpolation between receptors.

In general, 1-25% of the total mercury emitted is predicted to deposit within 50 km at the humid site in flat terrain, while 0.5-18% is predicted to deposit at the arid site. This implies that at least 75% of the total mercury emissions is transported more than 50 km from any of the sources considered, and is consistent with the RELMAP results that predict that mercury may be transported across large distances.

The differences between the results for the two sites are due primarily to the differences in the frequency and intensity of precipitation, although one can expect other differences due to the frequency of each particular stability class during the year. As shown in Table 6-15, at the humid site, precipitation occurs about 12% of the year, with about 5% of this precipitation of moderate intensity (0.11 to 0.30 in/hr). At the arid site, precipitation occurs about 3% of the year, with about 2% of the precipitation of moderate intensity.

Wet deposition is predicted to be an effective removal mechanism within 50 km of the source for some facilities. For the humid site in the year considered, precipitation occurs for about 12.5% of the year. This value is to be compared to the percent of total emissions wet deposited within 50 km. For some facilities, 6-10% of the total emissions is predicted to wet deposit within 50 km at the humid site, implying that 50-80% of the plume is being washed out within 50 km during periods of precipitation.

Wet deposition is predicted to dominate total deposition except for those facilities with low stacks: the medical waste incinerators and the chlor-alkali plant. This is because the predicted air concentrations are higher, and hence more dry deposition is predicted to occur. In general, the fraction of total emissions that is predicted to wet deposit within 50 km is positively correlated with the fraction of emissions that are divalent mercury. This is due to the assumed higher solubility of the divalent species compared to that of elemental mercury.

The percentage of mercury deposited within 50 km depends on two main factors: facility characteristics that influence effective stack height (stack height plus plume rise) and the fraction of mercury emissions that is divalent mercury. In most cases, the effective stack height affects only the air concentrations, and hence dry deposition. It does not affect predicted wet deposition because the precipitation is assumed to originate above and pass through the entire plume.

Table 6-13
Mass Balance of Mercury Emissions for each Facility in a Humid Site

Humid Site Facility	Stack Height (m)	Emissions Speciation (Hg0/Hg2/Hg Part)	Yearly Hg Emission (kg/yr)	Percent of Hg Emissions Deposited within 50 km					
				Total Deposition		Dry Deposition		Wet Deposition	
				Lower Bound	Upper Bound	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Large MWC	71	20/60/20	1.33E+03	8.9%	12.9%	2.9%	3.7%	6.1%	9.2%
Small MWC	43	20/60/20	1.68E+02	11.2%	16.2%	4.8%	6.5%	6.4%	9.8%
Continuous MWI	12	20/60/20	8.19E+01	16.1%	23.4%	9.5%	13.3%	6.6%	10.0%
Intermittent MWI	12	20/60/20	2.42E+00	17.4%	25.4%	10.8%	15.3%	6.6%	10.0%
Large Coal-fired Utility Boiler	225	50/30/20	2.31E+02	2.3%	3.4%	0.2%	0.3%	2.1%	3.2%
Medium Coal-fired Utility Boiler	143	50/30/20	8.88E+01	3.6%	5.3%	0.6%	0.7%	3.0%	4.6%
Small Coal-fired Utility Boiler	82	50/30/20	9.70E+00	5.8%	8.4%	1.7%	2.2%	4.1%	6.2%
Medium Oil-fired Utility Boiler	89	50/30/20	2.00E+00	4.6%	6.7%	1.0%	1.3%	3.6%	5.5%
Chlor-alkali plant	3	70/30/0	3.79E+02	10.2%	15.1%	6.8%	10.1%	3.4%	5.0%
Primary Copper Smelter	155	85/10/5	5.36E+02	1.9%	2.7%	0.3%	0.4%	1.6%	2.3%
Primary Lead Smelter	108	85/10/5	2.68E+03	2.1%	3.0%	0.4%	0.6%	1.7%	2.5%

Table 6-14
Mass Balance of Mercury Emissions for each Facility in an Arid Site

Arid Site Facility	Stack Height (m)	Emissions Speciation ($Hg0/Hg^{++}/Hg^{++}$ part)	Yearly Hg Emission (kg/yr)	Percent of Hg Emissions Deposited within 50 km					
				Total Deposition		Dry Deposition		Wet Deposition	
				Lower Bound	Upper Bound	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Large MWC	71	20/60/20	1.33E+03	4.0%	5.5%	3.3%	4.3%	0.8%	1.2%
Small MWC	43	20/60/20	1.68E+02	6.0%	8.3%	5.2%	7.0%	0.8%	1.3%
Continuous MWI	12	20/60/20	8.19E+01	11.0%	15.7%	10.1%	14.3%	0.9%	1.4%
Intermittent MWI	12	20/60/20	2.42E+00	12.4%	17.8%	11.5%	16.5%	0.9%	1.4%
Large Coal-fired Utility Boiler	225	50/30/20	2.31E+02	0.7%	1.0%	0.3%	0.4%	0.3%	0.5%
Medium Coal-fired Utility Boiler	143	50/30/20	8.88E+01	1.2%	1.6%	0.7%	0.9%	0.4%	0.7%
Small Coal-fired Utility Boiler	82	50/30/20	9.70E+00	2.4%	3.3%	1.9%	2.5%	0.5%	0.8%
Medium Oil-fired Utility Boiler	89	50/30/20	2.00E+00	1.7%	2 2%	1.2%	1.5%	0.5%	0.7%
Chlor-alkali plant	3	70/30/0	3.79E+02	7.9%	11.7%	7.4%	11.0%	0.4%	0.7%
Primary Copper Smelter	155	85/10/5	5.36E+02	0.5%	0.7%	0.4%	0.5%	0.2%	0.3%
Primary Lead Smelter	108	85/10/5	2.68E+03	0.7%	0.9%	0.5%	0.6%	0.2%	0.3%

Table 6-15
Precipitation Frequencies at the Humid and Arid Sites

Site	Hours of Precipitation (% of year)	Breakdown of Precipitation Hours (% of precipitation)		
		Trace to 0.1 in/hr	0.11 to 0.3 in/hr	> 0.3 in/hr
Humid Site	1098 (12.5%)	1044 (95%)	50 (4.5%)	4 (0.4%)
Arid Site	220 (2.5%)	214 (97%)	5 (2.3%)	1 (0.5%)

Exceptions occur when the effective stack height is above the mixing height, in which case no deposition is predicted to occur, or when the receptor is located above the effective stack height (see Appendix D for details on how terrain is addressed). The former can happen routinely for facilities with extremely high stacks and large exit gas velocities (e.g., large coal-fired utility boiler).

The differences between the results for the LMWC and SMWC are primarily due to differences in the parameters used to estimate the effective stack height (stack height plus plume rise): stack height, stack diameter, and exit temperature (please see Table 6-6 on page 6-6-5 of this volume for a summary of all model plant characteristics). The effective stack height is used to estimate dry deposition. The lower plumes predicted for the SMWC result in higher air concentrations, and hence higher predicted dry deposition. About twice as much of the emitted mercury is predicted to dry deposit for the SMWC than for the LMWC. This difference is roughly the same as the ratio of the stack heights (LMWC stack is about twice as high as that of the SMWC). Wet deposition is only affected by differences in wind speed at stack top, and in this case the ultimate effects are minimal. The wind speed at stack top is extrapolated from the height at which it was measured using wind profile exponents (see Appendix D).

In addition to the total mercury emission rate, which does not affect the fraction deposited within 50 km, the only difference between the CMWI and the IMWI is the assumed stack diameter: 2.8 m for the CMWI and 1.2 m for the IMWI. The stack diameter is used to calculate air concentrations and dry deposition rates. The predicted wet deposition rates do not depend on the stack diameter and so there is no difference between the predicted wet deposition for the two facilities. There are differences in the predicted dry deposition, and these differences must be due solely to the difference in stack diameter. Because the assumed exit temperature of the facilities is so high (1500 F), buoyancy forces are always predicted to dominate the plume's rise from the stack. The buoyancy flux is estimated using the method of Briggs (1975), and this estimate is positively correlated with stack diameter (see Appendix D). The larger diameter for the CMWI results in a larger buoyancy flux, which results in slightly higher plumes than that for the IMWI. This results in lower air concentrations and dry deposition rates for the CMWI than that of the IWMI.

Differences between the results for the utility boilers are due primarily to the difference in stack heights. This is also the reason for the differences between the results for the PCS and PLS. For all utility boilers, less than 10% of the total mercury emitted is predicted to deposit within 50 km. Again, this is a reflection of the high effective stack heights associated with this source class.

The deposition rates averaged over the entire 50 km radius region surrounding each facility are given in Tables 6-16 and 6-17. These values are comparable to or well below typically reported deposition rates (see Section 2).

6.2.7 Summary of Local Impact Analysis Results

Air Modeling

The predicted average atmospheric mercury concentrations that result from the emissions of the model plants were generally low. Only mercury emissions from the chlor-alkali model plant markedly elevated predicted air concentrations at the receptors considered. Using the COMPDEP model and assuming flat terrain at the two hypothetical sites, 75% or more of the emitted mercury was predicted to be transported beyond the 50 Km modeling domain of the local impact analysis. The percentage of emitted mercury predicted to deposit within 50 Km of the local source ranged from 0.5% to 25% for all model plants at both sites. Deposition of mercury emitted to the atmosphere from the model plants occurred through both wet and dry mechanisms. Removal of atmospheric mercury through precipitation was, in general, predicted to dominate total mercury deposition near the model plants. The wet deposition of mercury was predicted to be greater at the more humid eastern site than at the more arid western site. The predicted dry deposition rate depended on the predicted atmospheric concentrations of vapor-phase divalent mercury and, to a lesser extent, particle-bound divalent mercury. As the distance of the receptor from the stack of the model plant increased, the percentage of total deposition which resulted from dry deposition was predicted to increase. At this time no suitable data exist for comparing the predicted deposition patterns around the model plants.

Environmental Media Modeling

The predicted soil and surface water body concentrations of mercury were most heavily influenced by the predicted total mercury deposition rate from a local source. The predicted soil and surface water body mercury concentrations were all within the range of measured mercury concentrations for these media. Differences between the hypothetical watersheds of the eastern and western sites account for the variation in water body loading from mercury deposited onto the watershed.

Biota Modeling

The mercury concentrations in green plants were the result of direct deposition and air-to-plant transfer onto exposed foliar surfaces, and soil-to-plant transfer. As modeled in this assessment, small amounts of mercury in green plants were predicted to result from direct deposition and soil-to-plant transfer. Most of the mercury predicted to occur in green plants was the result of air-to-plant transfer for those types of green plants for which this route of transfer was deemed appropriate. Since air-to-plant transfer is primarily the product of the concentration of the pollutant in the air and the air-to-plant biotransfer factor, model plants with the highest predicted atmospheric mercury concentrations have the highest predicted mercury concentrations in green plants. Although the data are not extensive particularly near many of these anthropogenic sources, the predicted concentrations in green plants are within the range of measured concentrations. The predicted speciation of mercury in green plants was consistent with the reported values.

Table 6-16
Area-Averaged Mercury Deposition Rates for each Facility in a Humid Site

Humid Site		Area-Averaged Values within 50 km (ug/m2/yr)					
	Yearly Hg Emission (kg/yr)	Total Deposition Rate		Dry Deposition Rate		Wet Deposition Rate	
Facility		Lower Bound	Upper Bound	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Large MWC	1.3e+03	1.5e+01	2.2e+01	4.9e+00	6.3e+00	1.0e+01	1.6e+01
Small MWC	1.7e+02	2.4e+00	3.5e+00	1.0e+00	1.4e+00	1.4e+00	2.1e+00
Continuous MWI	8.2e+01	1.7e+00	2.4e+00	9.9e-01	1.4e+00	6.9e-01	1.1e+00
Intermittent MWI	2.4e+00	5.4e-02	7.8e-02	3.3e-02	4.7e-02	2.0e-02	3.1e-02
Large Coal-fired Utility Boiler	2.3e+02	6.8e-01	1.0e+00	6.3e-02	7.4e-02	6.1e-01	9.3e-01
Medium Coal-fired Utility Boiler	8.9e+01	4.0e-01	6.0e-01	6.5e-02	8.0e-02	3.4e-01	5.2e-01
Small Coal-fired Utility Boiler	9.7e+00	7.2e-02	1.0e-01	2.1e-02	2.8e-02	5.1e-02	7.6e-02
Medium Oil-fired Utility Boiler	2.0e+00	1.2e-02	1.7e-02	2.6e-03	3.2e-03	9.2e-03	1.4e-02
Chlor-alkali plant	3.8e+02	4.9e+00	7.3e+00	3.3e+00	4.9e+00	1.6e+00	2.4e+00
Primary Copper Smelter	5.4e+02	1.3e+00	1.8e+00	2.1e-01	2.7e-01	1.1e+00	1.6e+00
Primary Lead Smelter	2.7e+03	7.3e+00	1.0e+01	1.5e+00	1.9e+00	5.8e+00	8.4e+00

Table 6-17
Area-Averaged Mercury Deposition Rates for each Facility in an Arid Site

Arid Site Facility	Yearly Hg Emission (kg/yr)	Area-Averaged Values within 50 km (ug/m2/yr)					
		Total Deposition Rate		Dry Deposition Rate		Wet Deposition Rate	
		Lower Bound	Upper Bound	Lower Bound	Upper Bound	Lower Bound	Upper Bound
Large MWC	1.3e+03	6.9e+00	9.3e+00	5.5e+00	7.3e+00	1.3e+00	2.0e+00
Small MWC	1.7e+02	1.3e+00	1.8e+00	1.1e+00	1.5e+00	1.8e-01	2.7e-01
Continuous MWI	8.2e+01	1.2e+00	1.6e+00	1.1e+00	1.5e+00	9.2e-02	1.4e-01
Intermittent MWI	2.4e+00	3.8e-02	5.5e-02	3.5e-02	5.1e-02	2.7e-03	4.2e-03
Large Coal-fired Utility Boiler	2.3e+02	2.0e-01	2.8e-01	1.0e-01	1.3e-01	1.0e-01	1.5e-01
Medium Coal-fired Utility Boiler	8.9e+01	1.3e-01	1.8e-01	8.1e-02	1.0e-01	5.0e-02	7.6e-02
Small Coal-fired Utility Boiler	9.7e+00	3.0e-02	4.0e-02	2.3e-02	3.1e-02	6.4e-03	9.6e-03
Medium Oil-fired Utility Boiler	2.0e+00	4.2e-03	5.7e-03	3.0e-03	3.9e-03	1.2e-03	1.8e-03
Chlor-alkali plant	3.8e+02	3.8e+00	5.7e+00	3.6e+00	5.3e+00	2.2e-01	3.2e-01
Primary Copper Smelter	5.4e+02	3.7e-01	5.1e-01	2.5e-01	3.3e-01	1.2e-01	1.8e-01
Primary Lead Smelter	2.7e+03	2.4e+00	3.2e+00	1.7e+00	2.2e+00	6.9e-01	1.0e+00

The mercury concentrations in all animal products except fish were predicted to be low. This was the result of generally low concentrations in plants and small plant-to-animal and soil-to-animal biotransfer factors. The predicted concentrations and speciation of mercury in animal products was consistent with the reported values.

Mercury concentrations in fish were predicted to be the highest of the biota considered. Mercury concentrations in fish were the product of the bioaccumulation factor and the dissolved concentration of mercury in surface water. The predicted mercury concentrations in fish were within the range of reported values. There is a great deal of uncertainty and variability associated with the uptake of mercury by fish.

Human and Wildlife Exposure Modeling

Human exposure to anthropogenic mercury was predicted to be dominated by indirect routes of exposure except for the hypothetical average urban dweller. This individual was assumed to be exposed to emitted mercury from inhalation and soil ingestion only. For all other exposure scenarios, except those including fish consumption, the divalent species was predicted to be the primary species to which humans were exposed. For those hypothetical individuals exposed through consumption of both green plants and animal products, mercury exposure through consumption of green plants was greater than through consumption of animal products. This was the result of low plant-to-animal and soil-to-animal biotransfer factors when compared to the air-to-plant biotransfer factors.

Those hypothetical humans who were assumed to consume fish had the highest exposures. This was a result of the bioaccumulation factor into fish. Methylmercury was the primary species to which these individuals were exposed. On a per body weight basis, children were predicted to be more exposed than adults.

The animals with the highest fish ingestion rate (per body weight) generally had the highest methylmercury intakes, except in cases where a piscivorous species was assumed to prey more heavily on trophic level 4 fish. For example, the river otter was assumed to consume less fish on a per weight basis than the osprey but was assumed to consume larger fish from a higher trophic level which had higher methylmercury body burdens. As a result the otter was predicted to have a higher mercury exposure.

6.2.8 Additional Analysis Based on Emission Guidelines for Existing MWCs and New Source Performance Standards

The exposure analysis for MWCs presented in the local impact analysis was based on the model plant parameters described in Appendix F. The emission rate was 490 ug/dscm for the large MWC and 700 ug/dscm for the small MWC model plant. The emission rate for the large MWC was based on a combination spray dryer/fabric filter achieving 30 percent mercury control. The small MWC model plant was assumed to not have any appreciable mercury control and was modeled at an emission rate of 700 ug/dscm. Since this analysis was performed, the U.S. EPA has finalized emission guidelines for existing MWCs and New Source Performance Standards for new facilities (October 31, 1995). The final rules require new and existing MWCs that combust more than 39 tons of waste per day, to reduce their mercury emissions to no more than 80 ug/dscm. To reflect this regulation, an additional analysis was performed which evaluated the large and small MWC model plants at this emission rate.

additional analysis was performed which evaluated the large and small MWC model plants at this emission rate.

To achieve this emission reduction it is likely that most facilities will use activated carbon injection as a control measure. As explained in Appendix F, activated carbon injection effectively captures Hg^{2+} with the result that the percentage of Hg^0 as a fraction of total mercury increases. In addition, the fraction of mercury associated with particulate matter (Hg (PM)) also decreases. To address this change in the mercury speciation profile, three emission scenarios were modeled. These are illustrated in Table 6-18.

In the three emission speciation profiles the percentage of elemental mercury released ranges from 30 to 90 percent. The remaining mercury emissions are assumed to be divalent and in the vapor-phase. The velocity and temperature of the exit gas assumed for the model plants were unchanged from the local impact analysis.

As described in the previous sections, the two types of mercury species considered in the emissions are expected to behave quite differently once emitted from the stack. Elemental mercury is not expected to deposit close to the facility and vapor phase divalent mercury is expected to deposit in greater quantities closer to the emission sources. Because of the assumptions made pertaining to the atmospheric chemistry of mercury (i.e., no dry atmospheric chemistry of mercury in the local atmosphere) and atmospheric partitioning of mercury (i.e., the vapor/particle (V/P) ratio in the local atmosphere was assumed to be equal to the V/P ratio as it would exist in stack emissions), the mandated decreases in total mercury emitted and the resulting decreases in the amount of divalent mercury emitted are predicted to result in decreased concentrations in environmental media and biota. These predicted decreases are presented in Tables 6-19 and 6-20.

If the elemental mercury emissions comprise 30% of he total mercury emitted, then the legislated decreases are predicted to result in a decrease of at least 80% of the total mercury air concentration and total mercury deposition rate at 2.5 Km downwind from the plant, when compared to the predictions from the model plants developed for the previous section. The corresponding predicted concentrations in fish are also decreased at least 80% at this distance. If the elemental mercury emissions comprise 90% of the total mercury emitted, then the legislated decreases are predicted to result in a decrease of at least 90% of the total mercury concentration in the air and the total mercury deposition rate when compared to the predictions from the model plants developed for the previous section. The corresponding predicted concentrations in fish are also decreased at least 90%. That the decreases in the amount of mercury emitted and in the amount of divalent mercury emitted result in decreased local deposition, is most evident in emission scenario C, in which 90% of the emissions are assumed to be elemental.

Table 6-18
Process Parameters for Municipal Waste Combustor Model Plants After Imposition of MACT Standards

Model Plant	Plant Size	Capacity (% of year)	Stack Height (ft)	Stack Diameter (ft)	Hg Emission Rate (kg/yr)	Speciation Percent (Hg^0/Hg^{2+})	Exit Velocity (m/sec)	Exit Temp. (°F)
Large Municipal Waste Combustors	2,250 tons/day	90%	230	9.5	220	a) 30/70 b) 60/40 c) 90/10	21.9	285
Small Municipal Waste Combustors	200 tons/day	90%	140	5	20	a) 30/70 b) 60/40 c) 90/10	21.9	375

[a] Hg^0 = Elemental Mercury
[b] Hg^{2+} = Divalent Vapor Phase Mercury

Table 6-19
Mercury Concentrations Predicted in Media and Biota As a Result of Mercury Emissions From Municipal Waste Combustors After Imposition of the MACT Standards in the Hypothetical Western Site

Plant	Distance (Km)	Species Hg^0/Hg^{++}	Watershed Air Concentration (ug/m^3)	Watershed Total Deposition (ug/m^2 /yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/L)	Tier 3 Fish (ug/g)	Tier 4 Fish (ug/g)
LMWC	2.5	30/70	0.04	16.1	32.2	1.2	0.06	0.31
		60/40	0.04	9.3	18.6	0.7	0.04	0.18
		90/10	0.04	2.5	5.1	0.2	0.01	0.05
SMWC		30/70	0.01	2.2	4.4	0.2	0.008	0.04
		60/40	0.01	1.3	2.6	0.1	0.005	0.02
		90/10	0.01	0.3	0.7	0.02	0.001	0.007
LMWC	10	30/70	0.03	4.7	9.3	0.3	0.02	0.07
		60/40	0.03	2.7	5.4	0.2	0.009	0.04
		90/10	0.03	0.8	1.6	0.05	0.002	0.01
SMWC		30/70	0.07	0.7	1.3	0.05	0.002	0.01
		60/40	0.07	0.4	0.8	0.04	0.001	0.006
		90/10	0.07	0.1	0.2	0.02	0.0003	0.002
LMWC	25	30/70	0.02	1.9	3.8	0.1	0.005	0.03
		60/40	0.02	1.1	2.2	0.06	0.003	0.02
		90/10	0.02	0.3	0.6	0.02	0.0009	0.005
SMWC		30/70	0.03	0.3	0.5	0.02	0.0007	0.004
		60/40	0.03	0.2	0.3	0.01	0.0004	0.002
		90/10	0.03	0.04	0.09	0.002	0.0001	0.0006

LMWC = Large Municipal Waste Combustor
SMWC = Small Municipal Waste Combustor
Hg^o = Elemental Mercury
Hg^{++} = Divalent Mercury

Table 6-20
Mercury Concentrations Predicted in Media and Biota As a Result of Mercury Emissions From Municipal Waste Combustors After Imposition of the MACT Standards in the Hypothetical Eastern Site

Plant	Distance (Km)	Species Hg^{o}/Hg^{++}	Watershed Air Concentration (ug/m^3)	Watershed Total Deposition (ug/m^2 /yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/L)	Tier 3 Fish (ug/g)	Tier 4 Fish (ug/g)
LMWC	2.5	30/70	0.04	44.3	86.3	3.2	0.15	0.76
		60/40	0.04	25.9	50.4	1.9	0.09	0.44
		90/10	0.04	7.4	14.5	0.5	0.03	0.13
SMWC		30/70	0.01	5.0	9.8	0.4	0.02	0.09
		60/40	0.01	2.9	5.7	0.2	0.01	0.05
		90/10	0.01	0.8	1.7	0.1	0.003	0.01
LMWC	10	30/70	0.04	7.6	14.9	0.4	0.02	0.1
		60/40	0.04	4.6	9.0	0.2	0.01	0.06
		90/10	0.04	1.6	3.0	0.1	0.004	0.02
SMWC		30/70	0.01	1.0	1.9	0.1	0.002	0.01
		60/40	0.01	0.6	1.1	0.03	0.001	0.007
		90/10	0.01	0.2	0.4	0.01	0.0005	0.002
LMWC	25	30/70	0.02	1.9	3.7	0.1	0.004	0.02
		60/40	0.02	1.2	2.3	0.1	0.003	0.01
		90/10	0.02	0.4	0.9	0.02	0.001	0.01
SMWC		30/70	0.003	0.3	0.5	0.01	0.0006	0.003
		60/40	0.004	0.2	0.3	0.01	0.0004	0.002
		90/10	0.004	0.1	0.1	0.003	0.0001	0.0007

6.3 Results of Combining Local and Regional Models

In this section the results are presented for combining the local and regional impacts of anthropogenic sources. For both the eastern and western sites, the 50th and 90th percentile of the predicted air concentrations and deposition rates by the regional air model are used in conjunction with the air concentrations and deposition rates predicted by the local scale model for each plant to obtain estimates of environmental concentrations and possible exposure for both human and wildlife species.

6.3.1 Air Concentrations, Deposition, and Water Concentrations

In this section the air concentrations, deposition rates, and water concentrations predicted using the local and regional scale model results are presented. The predicted concentrations for other media are given in appendix G.

Tables 6-21 and 6-22 show the predicted air concentration, deposition rates and surface water concentrations for both sites and each facility, using both the 50th and 90th percentiles of the RELMAP values. The predicted air concentrations are typically dominated by the regional values, even for the watersheds relatively close to the facility. The only exception to this is the chlor-alkali plant, for which larger air concentrations are predicted (this is due to the low stack height and assumed stack gas exit velocity).

Except for the utility boilers, the predicted deposition due to the local source was in general larger than the regional contribution at the closest watershed considered. The importance of the regional contribution increased as a function of distance from the source, with the regional sources predicted to dominate total deposition for all facilities at 25 km in the Eastern site. The patterns for the predicted surface water concentrations, and hence fish concentrations, are similar to the deposition results. However, the contribution of the regional sources is slightly smaller than for deposition because the deposition rate to the water body itself is also considered in the local analysis. This rate is larger than that for the watershed because the water body is closer to the source, and so the relative contribution of the regional source is smaller.

6.3.2 Human Exposure

Tables 6-23 through 6-36 show the predicted human intake for each exposure scenario and site. The regional contribution for inhalation intake is similar to the contribution to air concentrations described above.

The intake for the agricultural exposure scenarios, the rural subsistence farmer and the home gardener, was dominated in general by the regional sources. The total intakes were in fact comparable to the predicted intakes using a typical air concentration of 1.6 ng/m^3 as described in Chapter 4. The regional sources are predicted to dominate exposure because these sources dominate the air concentrations, and the plant and animal concentrations are driven by the predicted air concentrations due to the air-to-plant transfer coefficients used (see section 4.3 for detailed assessment of the contribution to plant and animal concentrations from the various routes).

For the agricultural scenarios, most of the mercury intake was divalent mercury, and the dominant pathway was the consumption of fruits and fruiting vegetables. The mercury in potatoes and root vegetables results solely from root uptake since no air uptake was assumed to occur for these plants (Appendix A). For leafy vegetables, all the mercury is predicted to be from air uptake since no root uptake was assumed to occur. For grains, legumes, fruits and fruiting vegetables the bulk of mercury was also modeled to result from air uptake of atmospheric elemental mercury and transformation to other species. Fish consumption was the dominant mercury exposure pathway for those hypothetical individuals modeled to consume fish.

Table 6-21
Combination of Local and Regional Impacts: Contribution of Regional Sources to Key Output at Eastern Site

Eastern Site	Mercury Air Concentration (ng/m3)				Mercury Deposition Rate (ug/m2/yr)				Mercury Surface Water Concentration (ng/l)			
Facility	50th	% RELMAP	90th	% RELMAP	50th	% RELMAP	90th	% RELMAP	50th	% RELMAP	90th	% RELMAP
2 5 km												
Large Municipal Waste Combustor	1.9E+0	87.09%	2 0E+0	87 51%	2.8E+2	6.78%	3.0E+2	13 34%	2 0E+1	4 66%	2 1E+1	9 31%
Small Municipal Waste Combustor	1.8E+0	94 09%	1 9E+0	94.29%	6.0E+1	31 59%	8 1E+1	49 42%	3 8E+0	23 75%	4 8E+0	39 55%
Continuous Medical Waste Incinerator	1 9E+0	87.86%	2 0E+0	88 25%	5 1E+1	37.18%	7.2E+1	55 61%	3 2E+0	28 50%	4 2E+0	45.57%
Intermittent Medical Waste Incinerator	1 7E+0	99.42%	1 8E+0	99 44%	2 0E+1	94.47%	4 1E+1	97.31%	9.9E-1	91.93%	2 0E+0	95 99%
Large Coal-fired Utility Boiler	1 7E+0	99.92%	1 8E+0	99.93%	2.7E+1	68 93%	4 9E+1	82 44%	1.5E+0	61.47%	2 5E+0	77 02%
Medium Coal-fired Utility Boiler	1.7E+0	99.87%	1.8E+0	99 87%	2 6E+1	73 65%	4.7E+1	85.54%	1 4E+0	66 92%	2 4E+0	80 95%
Small Coal-fired Utility Boiler	1.7E+0	99.85%	1.8E+0	99.85%	2 0E+1	94.30%	4 1E+1	97.22%	9 9E-1	91 86%	2 0E+0	95 95%
Medium Oil-fired Utility Boiler	1.7E+0	99 99%	1.8E+0	99 99%	1 9E+1	98.97%	4 0E+1	99 51%	9 3E-1	98 48%	1 9E+0	99 27%
Chlor-alkali plant	5.5E+0	30 73%	5.6E+0	31 53%	1 4E+2	13 67%	1 6E+2	25 10%	9 9E+0	9.23%	1 1E+1	17 59%
Primary Copper Smelter	1.7E+0	97.79%	1 8E+0	97 87%	3 8E+1	50 39%	5 9E+1	68 25%	2 2E+0	41 14%	3 2E+0	59 48%
Primary Lead Smelter	2.1E+0	80.26%	2 2E+0	80 85%	1 2E+2	15 77%	1 4E+2	28 38%	8.1E+0	11 27%	9 1E+0	21 05%
10 km												
Large Municipal Waste Combustor	1.9E+0	88 41%	2 0E+0	88 78%	6.8E+1	27 94%	8 9E+1	45 08%	3 5E+0	25.78%	4 5E+0	42 18%
Small Municipal Waste Combustor	1 8E+0	96 35%	1 8E+0	96.48%	2 7E+1	69 15%	4.8E+1	82 59%	1 4E+0	66 95%	2 4E+0	80 97%
Continuous Medical Waste Incinerator	1 8E+0	95.38%	1 8E+0	95 54%	2.6E+1	73.83%	4 7E+1	85 65%	1 3E+0	71.57%	2 3E+0	84 09%
Intermittent Medical Waste Incinerator	1.7E+0	99.84%	1 8E+0	99 84%	1 9E+1	98.88%	4 0E+1	99 47%	9 2E-1	98 74%	1 9E+0	99 39%
Large Coal-fired Utility Boiler	1.7E+0	99 91%	1 8E+0	99.91%	2.1E+1	90 33%	4.2E+1	95.18%	1 0E+0	89 54%	2 0E+0	94 73%
Medium Coal-fired Utility Boiler	1.7E+0	99.84%	1.8E+0	99 85%	2 0E+1	93 48%	4.1E+1	96 81%	9 8E-1	92 54%	2 0E+0	96 30%
Small Coal-fired Utility Boiler	1.7E+0	99.88%	1.8E+0	99.89%	1 9E+1	98 76%	4.0E+1	99.41%	9 2E-1	98.64%	1 9E+0	99.35%
Medium Oil-fired Utility Boiler	1.7E+0	99.99%	1 8E+0	99 99%	1.9E+1	99.79%	4.0E+1	99 90%	9 1E-1	99.77%	1 9E+0	99 89%
Chlor-alkali plant	2.5E+0	66.91%	2.6E+0	67 73%	3 6E+1	51 86%	5 8E+1	69 51%	1.9E+0	48.51%	2 9E+0	66 43%
Primary Copper Smelter	1 7E+0	97.60%	1 8E+0	97 69%	2.3E+1	82.65%	4 4E+1	90 97%	1 1E+0	81 32%	2 1E+0	90 14%
Primary Lead Smelter	2 1E+0	81.45%	2 1E+0	82 00%	4.1E+1	45 62%	6 3E+1	63.97%	2 1E+0	43.63%	3 1E+0	61 91%
25 km												
Large Municipal Waste Combustor	1.8E+0	93 62%	1.9E+0	93 84%	3.1E+1	60.17%	5.3E+1	76.17%	1.5E+0	60 15%	2 5E+0	76 02%
Small Municipal Waste Combustor	1.7E+0	98 38%	1.8E+0	98 44%	2.1E+1	88.80%	4 2E+1	94 38%	1 0E+0	88 77%	2 0E+0	94 32%
Continuous Medical Waste Incinerator	1.7E+0	98 30%	1.8E+0	98.36%	2 1E+1	91 04%	4.2E+1	95 56%	1 0E+0	90 98%	2 0E+0	95 49%
Intermittent Medical Waste Incinerator	1.7E+0	99 94%	1 8E+0	99 95%	1 9E+1	99 70%	4 0E+1	99 86%	9 1E-1	99 70%	1 9E+0	99 86%
Large Coal-fired Utility Boiler	1.7E+0	99.92%	1 8E+0	99 93%	1 9E+1	97 68%	4.0E+1	98 89%	9 3E-1	97.72%	1 9E+0	98 90%
Medium Coal-fired Utility Boiler	1.7E+0	99.87%	1 8E+0	99 88%	1 9E+1	98.31%	4 0E+1	99 19%	9 3E-1	98 30%	1 9E+0	99 18%
Small Coal-fired Utility Boiler	1.7E+0	99.94%	1 8E+0	99 94%	1 9E+1	99 67%	4.0E+1	99 84%	9.1E-1	99.67%	1 9E+0	99 84%
Medium Oil-fired Utility Boiler	1.7E+0	99.99%	1 8E+0	99 99%	1.9E+1	99.95%	4 0E+1	99.98%	9 1E-1	99.95%	1 9E+0	99 98%
Chlor-alkali plant	2.0E+0	85 38%	2 0E+0	85 83%	2.3E+1	81.62%	4 4E+1	90 39%	1 1E+0	81 33%	2 1E+0	90 15%
Primary Copper Smelter	1.7E+0	98.44%	1 8E+0	98 50%	2 0E+1	94.71%	4 1E+1	97 43%	9 6E-1	94 71%	2 0E+0	97 41%
Primary Lead Smelter	1.9E+0	89.16%	2 0E+0	89 51%	2 5E+1	75.56%	4 6E+1	86 74%	1 2E+0	75 57%	2 2E+0	86 66%

Table 6-22
Combination of Local and Regional Impacts: Contribution of Regional Sources to Key Output at Western Site

Western Site	Mercury Air Concentration (ng/m3)				Mercury Deposition Rate (ug/m2/yr)				Mercury Surface Water Concentration (ng/l)			
Facility	50th	% REL.MAP	90th	% REL MAP	50th	% REL.MAP	90th	% REL.MAP	50th	% REL.MAP	90th	% REL MAP
2.5 km												
Large Municipal Waste Combustor	1.9E+0	87.29%	1.9E+0	87.52%	9.8E+1	2.95%	1.1E+2	10.01%	7.2E+0	2.31%	7.6E+0	7.35%
Small Municipal Waste Combustor	1.7E+0	94.54%	1.7E+0	94.64%	2.1E+1	13.88%	2.9E+1	37.08%	1.5E+0	11.19%	1.9E+0	29.74%
Continuous Medical Waste Incinerator	1.8E+0	88.12%	1.9E+0	88.33%	2.3E+1	12.40%	3.1E+1	34.09%	1.7E+0	9.69%	2.1E+0	26.49%
Intermittent Medical Waste Incinerator	1.6E+0	99.43%	1.7E+0	99.44%	3.7E+0	78.94%	1.1E+1	93.20%	2.3E-1	73.45%	6.2E-1	90.28%
Large Coal-fired Utility Boiler	1.6E+0	99.87%	1.7E+0	99.88%	6.0E+0	48.14%	1.4E+1	77.24%	3.9E-1	42.88%	7.8E-1	71.60%
Medium Coal-fired Utility Boiler	1.6E+0	99.84%	1.7E+0	99.84%	5.2E+0	56.18%	1.3E+1	82.41%	3.2E-1	51.54%	7.2E-1	78.12%
Small Coal-fired Utility Boiler	1.6E+0	99.86%	1.7E+0	99.86%	3.3E+0	87.11%	1.1E+1	96.11%	2.0E-1	84.05%	5.9E-1	94.65%
Medium Oil-fired Utility Boiler	1.6E+0	99.99%	1.7E+0	99.99%	3.0E+0	97.86%	1.1E+1	99.41%	1.7E-1	97.34%	5.6E-1	99.19%
Chlor-alkali plant	5.4E+0	29.90%	5.4E+0	30.33%	1.0E+2	2.83%	1.1E+2	9.62%	8.3E+0	2.00%	8.7E+0	6.42%
Primary Copper Smelter	1.7E+0	97.33%	1.7E+0	97.38%	9.0E+0	32.14%	1.7E+1	63.39%	6.0E-1	27.60%	1.0E+0	56.14%
Primary Lead Smelter	2.0E+0	80.09%	2.1E+0	80.41%	3.8E+1	7.61%	4.6E+1	23.13%	2.8E+0	5.98%	3.2E+0	17.60%
10 km												
Large Municipal Waste Combustor	1.8E+0	89.37%	1.8E+0	89.56%	3.1E+1	9.30%	3.9E+1	27.26%	1.9E+0	8.75%	2.3E+0	24.37%
Small Municipal Waste Combustor	1.7E+0	96.70%	1.7E+0	96.76%	8.4E+0	34.54%	1.6E+1	65.86%	5.0E-1	33.07%	9.0E-1	62.39%
Continuous Medical Waste Incinerator	1.7E+0	95.70%	1.7E+0	95.78%	8.1E+0	35.82%	1.6E+1	67.10%	4.9E-1	34.05%	8.8E-1	63.42%
Intermittent Medical Waste Incinerator	1.6E+0	99.85%	1.7E+0	99.85%	3.1E+0	94.55%	1.1E+1	98.45%	1.8E-1	94.09%	5.7E-1	98.16%
Large Coal-fired Utility Boiler	1.6E+0	99.89%	1.7E+0	99.89%	4.0E+0	72.92%	1.2E+1	90.78%	2.3E-1	71.52%	6.4E-1	89.40%
Medium Coal-fired Utility Boiler	1.6E+0	99.86%	1.7E+0	99.86%	3.5E+0	82.62%	1.1E+1	94.56%	2.1E-1	80.94%	6.0E-1	93.45%
Small Coal-fired Utility Boiler	1.6E+0	99.90%	1.7E+0	99.90%	3.0E+0	95.74%	1.1E+1	98.80%	1.7E-1	95.44%	5.7E-1	98.60%
Medium Oil-fired Utility Boiler	1.6E+0	99.99%	1.7E+0	99.99%	2.9E+0	99.36%	1.1E+1	99.82%	1.7E-1	99.31%	5.6E-1	99.79%
Chlor-alkali plant	2.4E+0	67.69%	2.4E+0	68.13%	1.8E+1	15.68%	2.6E+1	40.47%	1.2E+0	14.23%	1.6E+0	35.78%
Primary Copper Smelter	1.7E+0	97.89%	1.7E+0	97.93%	4.6E+0	62.74%	1.2E+1	86.02%	2.7E-1	60.60%	6.7E-1	83.78%
Primary Lead Smelter	1.9E+0	83.68%	2.0E+0	83.95%	1.3E+1	21.64%	2.1E+1	50.23%	8.2E-1	20.41%	1.2E+0	46.28%
25 km												
Large Municipal Waste Combustor	1.7E+0	93.41%	1.8E+0	93.53%	1.4E+1	20.45%	2.2E+1	48.45%	8.0E-1	20.90%	1.2E+0	47.02%
Small Municipal Waste Combustor	1.6E+0	98.41%	1.7E+0	98.44%	5.0E+0	57.83%	1.3E+1	83.37%	2.8E-1	58.51%	6.8E-1	82.57%
Continuous Medical Waste Incinerator	1.6E+0	98.39%	1.7E+0	98.42%	4.6E+0	63.14%	1.2E+1	86.23%	2.6E-1	63.71%	6.5E-1	85.50%
Intermittent Medical Waste Incinerator	1.6E+0	99.95%	1.7E+0	99.95%	3.0E+0	98.27%	1.1E+1	99.52%	1.7E-1	98.31%	5.6E-1	99.49%
Large Coal-fired Utility Boiler	1.6E+0	99.91%	1.7E+0	99.91%	3.3E+0	89.17%	1.1E+1	96.79%	1.9E-1	88.98%	5.8E-1	96.44%
Medium Coal-fired Utility Boiler	1.6E+0	99.88%	1.7E+0	99.88%	3.1E+0	93.05%	1.1E+1	98.00%	1.8E-1	93.06%	5.7E-1	97.83%
Small Coal-fired Utility Boiler	1.6E+0	99.94%	1.7E+0	99.94%	3.0E+0	98.29%	1.1E+1	99.53%	1.7E-1	98.33%	5.6E-1	99.50%
Medium Oil-fired Utility Boiler	1.6E+0	99.99%	1.7E+0	99.99%	2.9E+0	99.75%	1.1E+1	99.93%	1.7E-1	99.76%	5.6E-1	99.93%
Chlor-alkali plant	1.9E+0	85.68%	1.9E+0	85.92%	7.5E+0	38.91%	1.5E+1	69.96%	4.3E-1	38.80%	8.2E-1	68.05%
Primary Copper Smelter	1.6E+0	98.49%	1.7E+0	98.52%	3.6E+0	80.94%	1.1E+1	93.95%	2.1E-1	81.24%	6.0E-1	93.57%
Primary Lead Smelter	1.8E+0	89.17%	1.8E+0	89.37%	7.1E+0	40.93%	1.5E+1	71.69%	4.0E-1	41.49%	7.9E-1	70.42%

Table 6-23
Combination of Local and Regional Impacts: Predicted Mercury Inhalation Intake at Eastern Site

Eastern Site	Predicted Inhalation Intake for Full Time Resident (mg/kg/day)												Predicted Inhalation Intake for Part Time Resident (mg/kg/day)					
	2.5 km				10 km				25 km				2.5 km		10 km		25 km	
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult		Adult		Adult		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	1.8E-6	87%	5.5E-7	87%	1.8E-6	88%	5.5E-7	88%	1.7E-6	94%	5.2E-7	94%	3.7E-7	87%	3.6E-7	88%	3.4E-7	94%
Small Municipal Waste Combustor	1.7E-6	94%	5.1E-7	94%	1.7E-6	96%	5.0E-7	96%	1.6E-6	98%	4.9E-7	98%	3.4E-7	94%	3.3E-7	96%	3.3E-7	98%
Continuous Medical Waste Incinerator	1.8E-6	88%	5.5E-7	88%	1.7E-6	95%	5.1E-7	95%	1.6E-6	98%	4.9E-7	98%	3.7E-7	88%	3.4E-7	95%	3.3E-7	98%
Intermittent Medical Waste Incinerator	1.6E-6	99%	4.9E-7	99%	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	3.2E-7	99%	3.2E-7	100%	3.2E-7	100%
Large Coal-fired Utility Boiler	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	3.2E-7	100%	3.2E-7	100%	3.2E-7	100%
Medium Coal-fired Utility Boiler	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	3.2E-7	100%	3.2E-7	100%	3.2E-7	100%
Small Coal-fired Utility Boiler	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	3.2E-7	100%	3.2E-7	100%	3.2E-7	100%
Medium Oil-fired Utility Boiler	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	1.6E-6	100%	4.8E-7	100%	3.2E-7	100%	3.2E-7	100%	3.2E-7	100%
Chlor-alkali plant	5.2E-6	31%	1.6E-6	31%	2.4E-6	67%	7.2E-7	67%	1.9E-6	85%	5.7E-7	85%	1.0E-6	31%	4.8E-7	67%	3.8E-7	85%
Primary Copper Smelter	1.6E-6	98%	4.9E-7	98%	1.6E-6	98%	5.0E-7	98%	1.6E-6	98%	4.9E-7	98%	3.3E-7	98%	3.3E-7	98%	3.3E-7	98%
Primary Lead Smelter	2.0E-6	80%	6.0E-7	80%	2.0E-6	81%	5.9E-7	81%	1.8E-6	89%	5.4E-7	89%	4.0E-7	80%	4.0E-7	81%	3.6E-7	89%

Eastern Site	Predicted Inhalation Intake for Full Time Resident (mg/kg/day)												Predicted Inhalation Intake for Part Time Resident (mg/kg/day)					
	2.5 km				10 km				25 km				2.5 km		10 km		25 km	
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult		Adult		Adult		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	1.9E-6	88%	5.7E-7	88%	1.9E-6	89%	5.6E-7	89%	1.8E-6	94%	5.3E-7	94%	3.8E-7	88%	3.8E-7	89%	3.6E-7	94%
Small Municipal Waste Combustor	1.8E-6	94%	5.3E-7	94%	1.7E-6	96%	5.2E-7	96%	1.7E-6	98%	5.1E-7	98%	3.5E-7	94%	3.5E-7	96%	3.4E-7	98%
Continuous Medical Waste Incinerator	1.9E-6	88%	5.7E-7	88%	1.7E-6	96%	5.2E-7	96%	1.7E-6	98%	5.1E-7	98%	3.8E-7	88%	3.5E-7	96%	3.4E-7	98%
Intermittent Medical Waste Incinerator	1.7E-6	99%	5.0E-7	99%	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	3.4E-7	99%	3.3E-7	100%	3.3E-7	100%
Large Coal-fired Utility Boiler	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	3.3E-7	100%	3.3E-7	100%	3.3E-7	100%
Medium Coal-fired Utility Boiler	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	3.3E-7	100%	3.3E-7	100%	3.3E-7	100%
Small Coal-fired Utility Boiler	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	3.3E-7	100%	3.3E-7	100%	3.3E-7	100%
Medium Oil-fired Utility Boiler	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	1.7E-6	100%	5.0E-7	100%	3.3E-7	100%	3.3E-7	100%	3.3E-7	100%
Chlor-alkali plant	5.2E-6	32%	1.6E-6	32%	2.4E-6	68%	7.4E-7	68%	1.9E-6	86%	5.8E-7	86%	1.1E-6	32%	4.9E-7	68%	3.9E-7	86%
Primary Copper Smelter	1.7E-6	98%	5.1E-7	98%	1.7E-6	98%	5.1E-7	98%	1.7E-6	98%	5.1E-7	98%	3.4E-7	98%	3.4E-7	98%	3.4E-7	98%
Primary Lead Smelter	2.0E-6	81%	6.2E-7	81%	2.0E-6	82%	6.1E-7	82%	1.8E-6	90%	5.6E-7	90%	4.1E-7	81%	4.1E-7	82%	3.7E-7	90%

Table 6-24
Combination of Local and Regional Impacts: Predicted Mercury Inhalation Intake at Western Site

Western Site	Predicted Inhalation Intake for Full Time Resident (mg/kg/day)												Predicted Inhalation Intake for Part Time Resident (mg/kg/day)					
	2.5 km				10 km				25 km				2.5 km		10 km		25 km	
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult		Adult		Adult		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	1.7E-6	87%	5.3E-7	87%	1.7E-6	89%	5.2E-7	89%	1.6E-6	93%	4.9E-7	93%	3.5E-7	87%	3.4E-7	89%	3.3E-7	94%
Small Municipal Waste Combustor	1.6E-6	95%	4.9E-7	95%	1.6E-6	97%	4.8E-7	97%	1.5E-6	98%	4.7E-7	98%	3.3E-7	95%	3.2E-7	97%	3.1E-7	98%
Continuous Medical Waste Incinerator	1.7E-6	88%	5.2E-7	88%	1.6E-6	96%	4.8E-7	96%	1.5E-6	98%	4.7E-7	98%	3.5E-7	88%	3.2E-7	96%	3.1E-7	98%
Intermittent Medical Waste Incinerator	1.5E-6	99%	4.6E-7	99%	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	3.1E-7	99%	3.1E-7	100%	3.1E-7	100%
Large Coal-fired Utility Boiler	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Medium Coal-fired Utility Boiler	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Small Coal-fired Utility Boiler	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Medium Oil-fired Utility Boiler	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	1.5E-6	100%	4.6E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Chlor-alkali plant	5.1E-6	30%	1.5E-6	30%	2.2E-6	68%	6.8E-7	68%	1.8E-6	86%	5.4E-7	86%	1.0E-6	30%	4.6E-7	68%	3.6E-7	86%
Primary Copper Smelter	1.6E-6	97%	4.7E-7	97%	1.6E-6	98%	4.7E-7	98%	1.5E-6	98%	4.7E-7	98%	3.2E-7	97%	3.1E-7	98%	3.1E-7	98%
Primary Lead Smelter	1.9E-6	80%	5.8E-7	80%	1.8E-6	84%	5.5E-7	84%	1.7E-6	89%	5.2E-7	89%	3.8E-7	80%	3.7E-7	84%	3.5E-7	89%

Western Site	Predicted Inhalation Intake for Full Time Resident (mg/kg/day)												Predicted Inhalation Intake for Part Time Resident (mg/kg/day)					
	2.5 km				10 km				25 km				2.5 km		10 km		25 km	
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult		Adult		Adult		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	1.8E-6	88%	5.4E-7	88%	1.7E-6	90%	5.3E-7	90%	1.7E-6	94%	5.0E-7	94%	3.6E-7	88%	3.5E-7	90%	3.4E-7	94%
Small Municipal Waste Combustor	1.6E-6	95%	5.0E-7	95%	1.6E-6	97%	4.9E-7	97%	1.6E-6	98%	4.8E-7	98%	3.3E-7	95%	3.2E-7	97%	3.2E-7	98%
Continuous Medical Waste Incinerator	1.8E-6	88%	5.3E-7	88%	1.6E-6	96%	4.9E-7	96%	1.6E-6	98%	4.8E-7	98%	3.6E-7	88%	3.3E-7	96%	3.2E-7	98%
Intermittent Medical Waste Incinerator	1.6E-6	99%	4.7E-7	99%	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	3.2E-7	99%	3.1E-7	100%	3.1E-7	100%
Large Coal-fired Utility Boiler	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Medium Coal-fired Utility Boiler	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Small Coal-fired Utility Boiler	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Medium Oil-fired Utility Boiler	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	1.6E-6	100%	4.7E-7	100%	3.1E-7	100%	3.1E-7	100%	3.1E-7	100%
Chlor-alkali plant	5.1E-6	30%	1.6E-6	30%	2.3E-6	68%	6.9E-7	68%	1.8E-6	86%	5.5E-7	86%	1.0E-6	30%	4.6E-7	68%	3.7E-7	86%
Primary Copper Smelter	1.6E-6	97%	4.8E-7	97%	1.6E-6	98%	4.8E-7	98%	1.6E-6	99%	4.8E-7	99%	3.2E-7	97%	3.2E-7	98%	3.2E-7	99%
Primary Lead Smelter	1.9E-6	80%	5.9E-7	80%	1.9E-6	84%	5.6E-7	84%	1.7E-6	89%	5.3E-7	89%	3.9E-7	80%	3.7E-7	84%	3.5E-7	89%

Table 6-25
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Subsistence Farmer at Eastern Site

Eastern Site	Predicted Intake for Rural Subsistence Farmer (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	9.0E-5	51%	5.8E-5	63%	5.8E-5	79%	4.4E-5	83%	5.0E-5	91%	3.9E-5	92%
Small Municipal Waste Combustor	5.4E-5	84%	4.1E-5	89%	4.8E-5	94%	3.8E-5	95%	4.7E-5	98%	3.7E-5	98%
Continuous Medical Waste Incinerator	5.6E-5	82%	4.3E-5	85%	4.9E-5	94%	3.8E-5	95%	4.7E-5	98%	3.7E-5	98%
Intermittent Medical Waste Incinerator	4.6E-5	99%	3.7E-5	99%	4.6E-5	100%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%
Large Coal-fired Utility Boiler	4.7E-5	97%	3.7E-5	98%	4.6E-5	99%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%
Medium Coal-fired Utility Boiler	4.7E-5	98%	3.7E-5	99%	4.6E-5	99%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%
Small Coal-fired Utility Boiler	4.6E-5	100%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%
Medium Oil-fired Utility Boiler	4.6E-5	100%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%
Chlor-alkali plant	1.6E-4	29%	1.2E-4	30%	6.9E-5	66%	5.5E-5	67%	5.3E-5	85%	4.3E-5	85%
Primary Copper Smelter	4.9E-5	92%	3.8E-5	95%	4.7E-5	97%	3.7E-5	97%	4.6E-5	98%	3.7E-5	98%
Primary Lead Smelter	7.1E-5	64%	5.1E-5	71%	5.9E-5	78%	4.6E-5	80%	5.2E-5	88%	4.1E-5	89%

Eastern Site	Predicted Intake for Rural Subsistence Farmer (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	9.4E-5	53%	6.1E-5	64%	6.3E-5	81%	4.6E-5	84%	5.5E-5	92%	4.2E-5	93%
Small Municipal Waste Combustor	5.9E-5	86%	4.4E-5	89%	5.3E-5	95%	4.1E-5	96%	5.1E-5	98%	4.0E-5	98%
Continuous Medical Waste Incinerator	6.0E-5	83%	4.6E-5	86%	5.3E-5	95%	4.1E-5	95%	5.1E-5	98%	4.0E-5	98%
Intermittent Medical Waste Incinerator	5.1E-5	99%	3.9E-5	99%	5.1E-5	100%	3.9E-5	100%	5.0E-5	100%	3.9E-5	100%
Large Coal-fired Utility Boiler	5.2E-5	97%	4.0E-5	99%	5.1E-5	99%	3.9E-5	100%	5.1E-5	100%	3.9E-5	100%
Medium Coal-fired Utility Boiler	5.1E-5	98%	4.0E-5	99%	5.1E-5	99%	3.9E-5	100%	5.1E-5	100%	3.9E-5	100%
Small Coal-fired Utility Boiler	5.1E-5	100%	3.9E-5	100%	5.1E-5	100%	3.9E-5	100%	5.0E-5	100%	3.9E-5	100%
Medium Oil-fired Utility Boiler	5.0E-5	100%	3.9E-5	100%	5.0E-5	100%	3.9E-5	100%	5.0E-5	100%	3.9E-5	100%
Chlor-alkali plant	1.6E-4	31%	1.2E-4	31%	7.4E-5	69%	5.7E-5	68%	5.8E-5	87%	4.5E-5	86%
Primary Copper Smelter	5.4E-5	93%	4.1E-5	95%	5.2E-5	97%	4.0E-5	97%	5.1E-5	98%	4.0E-5	98%
Primary Lead Smelter	7.6E-5	67%	5.4E-5	72%	6.3E-5	80%	4.8E-5	81%	5.6E-5	89%	4.4E-5	89%

Table 6-26
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Subsistence Farmer at Western Site

Western Site	Predicted Intake for Rural Subsistence Farmer (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	7 6E-5	56%	5 2E-5	66%	5 3E-5	79%	4 1E-5	83%	4.7E-5	90%	3.7E-5	91%
Small Municipal Waste Combustor	4.9E-5	86%	3 8E-5	90%	4.4E-5	95%	3 6E-5	96%	4 3E-5	98%	3 5E-5	98%
Continuous Medical Waste Incinerator	5 1E-5	82%	4 0E-5	85%	4 5E-5	94%	3 6E-5	95%	4 4E-5	98%	3 5E-5	98%
Intermittent Medical Waste Incinerator	4 2E-5	99%	3 4E-5	99%	4.2E-5	100%	3 4E-5	100%	4 2E-5	100%	3 4E-5	100%
Large Coal-fired Utility Boiler	4 3E-5	97%	3.5E-5	98%	4.2E-5	99%	3.4E-5	99%	4 2E-5	100%	3 4E-5	100%
Medium Coal-fired Utility Boiler	4 3E-5	98%	3 4E-5	99%	4 2E-5	99%	3 4E-5	100%	4 2E-5	100%	3 4E-5	100%
Small Coal-fired Utility Boiler	4 2E-5	100%	3 4E-5	100%	4 2E-5	100%	3.4E-5	100%	4 2E-5	100%	3.4E-5	100%
Medium Oil-fired Utility Boiler	4 2E-5	100%	3 4E-5	100%	4 2E-5	100%	3 4E-5	100%	4 2E-5	100%	3.4E-5	100%
Chlor-alkali plant	1.5E-4	28%	1 2E-4	29%	6.4E-5	66%	5.1E-5	67%	4 9E-5	85%	4 0E-5	85%
Primary Copper Smelter	4.5E-5	93%	3 6E-5	95%	4.3E-5	97%	3 5E-5	97%	4.3E-5	98%	3 5E-5	98%
Primary Lead Smelter	6.3E-5	67%	4.7E-5	72%	5.3E-5	80%	4 2E-5	81%	4 8E-5	88%	3 9E-5	88%

Western Site	Predicted Intake for Rural Subsistence Farmer (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	7.9E-5	58%	5 4E-5	67%	5.7E-5	81%	4 3E-5	84%	5.1E-5	91%	3 9E-5	92%
Small Municipal Waste Combustor	5 2E-5	87%	4 0E-5	90%	4 8E-5	95%	3 8E-5	96%	4.7E-5	98%	3.7E-5	98%
Continuous Medical Waste Incinerator	5.5E-5	84%	4 2E-5	86%	4 8E-5	95%	3.8E-5	95%	4.7E-5	98%	3 7E-5	98%
Intermittent Medical Waste Incinerator	4.6E-5	99%	3 6E-5	99%	4.6E-5	100%	3 6E-5	100%	4 6E-5	100%	3 6E-5	100%
Large Coal-fired Utility Boiler	4.7E-5	98%	3.7E-5	98%	4 6E-5	99%	3.6E-5	99%	4 6E-5	100%	3.6E-5	100%
Medium Coal-fired Utility Boiler	4 7E-5	98%	3 7E-5	99%	4.6E-5	99%	3 6E-5	100%	4 6E-5	100%	3 6E-5	100%
Small Coal-fired Utility Boiler	4.6E-5	100%	3 6E-5	100%	4.6E-5	100%	3.6E-5	100%	4.6E-5	100%	3.6E-5	100%
Medium Oil-fired Utility Boiler	4 6E-5	100%	3 6E-5	100%	4.6E-5	100%	3 6E-5	100%	4 6E-5	100%	3.6E-5	100%
Chlor-alkali plant	1 6E-4	30%	1.2E-4	30%	6.7E-5	68%	5.3E-5	68%	5 3E-5	86%	4 2E-5	86%
Primary Copper Smelter	4 9E-5	94%	3 8E-5	95%	4 7E-5	97%	3.7E-5	97%	4 7E-5	98%	3 7E-5	98%
Primary Lead Smelter	6 7E-5	69%	4 9E-5	73%	5 6E-5	81%	4 4E-5	82%	5 2E-5	89%	4 1E-5	89%

Table 6-27
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Home Gardener at Eastern Site

Eastern Site	Predicted Intake for Rural Home Gardener (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	1.9E-5	52%	1.1E-5	73%	1.2E-5	79%	9.9E-6	85%	1.1E-5	91%	9.1E-6	93%
Small Municipal Waste Combustor	1.1E-5	85%	9.2E-6	91%	1.0E-5	94%	8.8E-6	96%	9.9E-6	98%	8.6E-6	98%
Continuous Medical Waste Incinerator	1.2E-5	82%	9.8E-6	86%	1.0E-5	94%	8.9E-6	95%	9.9E-6	98%	8.6E-6	98%
Intermittent Medical Waste Incinerator	9.8E-6	99%	8.5E-6	99%	9.7E-6	100%	8.4E-6	100%	9.7E-6	100%	8.4E-6	100%
Large Coal-fired Utility Boiler	1.0E-5	97%	8.5E-6	99%	9.8E-6	99%	8.4E-6	100%	9.7E-6	100%	8.4E-6	100%
Medium Coal-fired Utility Boiler	9.9E-6	98%	8.5E-6	99%	9.8E-6	99%	8.4E-6	100%	9.7E-6	100%	8.4E-6	100%
Small Coal-fired Utility Boiler	9.8E-6	100%	8.4E-6	100%	9.7E-6	100%	8.4E-6	100%	9.7E-6	100%	8.4E-6	100%
Medium Oil-fired Utility Boiler	9.7E-6	100%	8.4E-6	100%	9.7E-6	100%	8.4E-6	100%	9.7E-6	100%	8.4E-6	100%
Chlor-alkali plant	3.4E-5	29%	2.8E-5	30%	1.5E-5	66%	1.3E-5	67%	1.1E-5	85%	9.9E-6	85%
Primary Copper Smelter	1.0E-5	93%	8.7E-6	96%	1.0E-5	97%	8.6E-6	97%	9.9E-6	98%	8.6E-6	98%
Primary Lead Smelter	1.5E-5	65%	1.1E-5	75%	1.2E-5	78%	1.0E-5	80%	1.1E-5	88%	9.5E-6	89%

Eastern Site	Predicted Intake for Rural Home Gardener (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.0E-5	55%	1.2E-5	74%	1.3E-5	80%	1.0E-5	86%	1.2E-5	91%	9.5E-6	93%
Small Municipal Waste Combustor	1.2E-5	86%	9.7E-6	92%	1.1E-5	95%	9.2E-6	96%	1.1E-5	98%	9.0E-6	98%
Continuous Medical Waste Incinerator	1.3E-5	83%	1.0E-5	87%	1.1E-5	94%	9.3E-6	95%	1.1E-5	98%	9.0E-6	98%
Intermittent Medical Waste Incinerator	1.1E-5	99%	8.9E-6	99%	1.1E-5	100%	8.9E-6	100%	1.1E-5	100%	8.9E-6	100%
Large Coal-fired Utility Boiler	1.1E-5	98%	8.9E-6	99%	1.1E-5	99%	8.9E-6	100%	1.1E-5	100%	8.9E-6	100%
Medium Coal-fired Utility Boiler	1.1E-5	98%	8.9E-6	99%	1.1E-5	100%	8.9E-6	100%	1.1E-5	100%	8.9E-6	100%
Small Coal-fired Utility Boiler	1.1E-5	100%	8.9E-6	100%	1.1E-5	100%	8.9E-6	100%	1.1E-5	100%	8.9E-6	100%
Medium Oil-fired Utility Boiler	1.1E-5	100%	8.9E-6	100%	1.1E-5	100%	8.9E-6	100%	1.1E-5	100%	8.9E-6	100%
Chlor-alkali plant	3.5E-5	31%	2.8E-5	31%	1.6E-5	68%	1.3E-5	68%	1.2E-5	86%	1.0E-5	86%
Primary Copper Smelter	1.1E-5	93%	9.2E-6	97%	1.1E-5	97%	9.1E-6	97%	1.1E-5	98%	9.0E-6	98%
Primary Lead Smelter	1.6E-5	67%	1.2E-5	76%	1.3E-5	80%	1.1E-5	81%	1.2E-5	89%	9.9E-6	89%

Table 6-28
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Rural Home Gardener at Western Site

Western Site	Predicted Intake for Rural Home Gardener (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	1.3E-5	68%	9.8E-6	81%	1.1E-5	82%	9.1E-6	87%	9.8E-6	90%	8.6E-6	92%
Small Municipal Waste Combustor	9.9E-6	89%	8.5E-6	93%	9.3E-6	95%	8.3E-6	96%	9.1E-6	98%	8.1E-6	98%
Continuous Medical Waste Incinerator	1.1E-5	83%	9.2E-6	87%	9.4E-6	94%	8.3E-6	95%	9.1E-6	98%	8.1E-6	98%
Intermittent Medical Waste Incinerator	8.9E-6	99%	8.0E-6	99%	8.9E-6	100%	8.0E-6	100%	8.9E-6	100%	7.9E-6	100%
Large Coal-fired Utility Boiler	9.0E-6	99%	8.0E-6	100%	8.9E-6	100%	8.0E-6	100%	8.9E-6	100%	8.0E-6	100%
Medium Coal-fired Utility Boiler	8.9E-6	99%	8.0E-6	100%	8.9E-6	100%	8.0E-6	100%	8.9E-6	100%	8.0E-6	100%
Small Coal-fired Utility Boiler	8.9E-6	100%	8.0E-6	100%	8.9E-6	100%	8.0E-6	100%	8.9E-6	100%	8.0E-6	100%
Medium Oil-fired Utility Boiler	8.9E-6	100%	7.9E-6	100%	8.9E-6	100%	7.9E-6	100%	8.9E-6	100%	7.9E-6	100%
Chlor-alkali plant	3.2E-5	27%	2.7E-5	29%	1.4E-5	66%	1.2E-5	67%	1.0E-5	85%	9.3E-6	85%
Primary Copper Smelter	9.3E-6	95%	8.2E-6	97%	9.1E-6	97%	8.1E-6	98%	9.0E-6	98%	8.1E-6	98%
Primary Lead Smelter	1.2E-5	73%	1.0E-5	78%	1.1E-5	81%	9.6E-6	83%	1.0E-5	88%	8.9E-6	89%

Western Site	Predicted Intake for Rural Home Gardener (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	1.3E-5	69%	1.0E-5	82%	1.1E-5	83%	9.3E-6	88%	1.0E-5	91%	8.8E-6	93%
Small Municipal Waste Combustor	1.0E-5	90%	8.8E-6	93%	9.7E-6	95%	8.5E-6	96%	9.5E-6	98%	8.3E-6	98%
Continuous Medical Waste Incinerator	1.1E-5	84%	9.4E-6	87%	9.8E-6	94%	8.6E-6	95%	9.5E-6	98%	8.3E-6	98%
Intermittent Medical Waste Incinerator	9.3E-6	99%	8.2E-6	99%	9.3E-6	100%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%
Large Coal-fired Utility Boiler	9.4E-6	99%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%
Medium Coal-fired Utility Boiler	9.3E-6	99%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%
Small Coal-fired Utility Boiler	9.3E-6	100%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%
Medium Oil-fired Utility Boiler	9.3E-6	100%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%	9.3E-6	100%	8.2E-6	100%
Chlor-alkali plant	3.3E-5	28%	2.8E-5	30%	1.4E-5	67%	1.2E-5	68%	1.1E-5	85%	9.5E-6	86%
Primary Copper Smelter	9.7E-6	96%	8.4E-6	97%	9.5E-6	97%	8.3E-6	98%	9.4E-6	98%	8.3E-6	98%
Primary Lead Smelter	1.2E-5	74%	1.0E-5	79%	1.1E-5	82%	9.8E-6	83%	1.0E-5	89%	9.2E-6	89%

Table 6-29
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban Average Resident at Eastern Site

Eastern Site	Predicted Intake for Urban Average (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	7.0E-6	7%	8.5E-7	7%	1.7E-6	28%	2.1E-7	28%	7.8E-7	60%	9.5E-8	60%
Small Municipal Waste Combustor	1.5E-6	32%	1.8E-7	32%	6.8E-7	69%	8.3E-8	69%	5.3E-7	89%	6.5E-8	89%
Continuous Medical Waste Incinerator	1.3E-6	37%	1.5E-7	37%	6.4E-7	74%	7.8E-8	74%	5.2E-7	91%	6.3E-8	91%
Intermittent Medical Waste Incinerator	5.0E-7	94%	6.1E-8	94%	4.8E-7	99%	5.8E-8	99%	4.7E-7	100%	5.8E-8	100%
Large Coal-fired Utility Boiler	6.9E-7	69%	8.3E-8	69%	5.2E-7	90%	6.3E-8	90%	4.8E-7	98%	5.9E-8	98%
Medium Coal-fired Utility Boiler	6.4E-7	74%	7.8E-8	74%	5.1E-7	93%	6.1E-8	93%	4.8E-7	98%	5.8E-8	98%
Small Coal-fired Utility Boiler	5.0E-7	94%	6.1E-8	94%	4.8E-7	99%	5.8E-8	99%	4.7E-7	100%	5.8E-8	100%
Medium Oil-fired Utility Boiler	4.8E-7	99%	5.8E-8	99%	4.7E-7	100%	5.7E-8	100%	4.7E-7	100%	5.7E-8	100%
Chlor-alkali plant	3.5E-6	14%	4.2E-7	14%	9.1E-7	52%	1.1E-7	52%	5.8E-7	82%	7.0E-8	82%
Primary Copper Smelter	9.4E-7	50%	1.1E-7	50%	5.7E-7	83%	6.9E-8	83%	5.0E-7	95%	6.1E-8	95%
Primary Lead Smelter	3.0E-6	16%	3.6E-7	16%	1.0E-6	46%	1.3E-7	46%	6.3E-7	76%	7.6E-8	76%

Eastern Site	Predicted Intake for Urban Average (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	7.5E-6	13%	9.1E-7	13%	2.2E-6	45%	2.7E-7	45%	1.3E-6	76%	1.6E-7	76%
Small Municipal Waste Combustor	2.0E-6	49%	2.5E-7	49%	1.2E-6	83%	1.5E-7	83%	1.1E-6	94%	1.3E-7	94%
Continuous Medical Waste Incinerator	1.8E-6	56%	2.2E-7	56%	1.2E-6	86%	1.4E-7	86%	1.0E-6	96%	1.3E-7	96%
Intermittent Medical Waste Incinerator	1.0E-6	97%	1.2E-7	97%	1.0E-6	99%	1.2E-7	99%	1.0E-6	100%	1.2E-7	100%
Large Coal-fired Utility Boiler	1.2E-6	82%	1.5E-7	82%	1.1E-6	95%	1.3E-7	95%	1.0E-6	99%	1.2E-7	99%
Medium Coal-fired Utility Boiler	1.2E-6	86%	1.4E-7	86%	1.0E-6	97%	1.3E-7	97%	1.0E-6	99%	1.2E-7	99%
Small Coal-fired Utility Boiler	1.0E-6	97%	1.2E-7	97%	1.0E-6	99%	1.2E-7	99%	1.0E-6	100%	1.2E-7	100%
Medium Oil-fired Utility Boiler	1.0E-6	100%	1.2E-7	100%	1.0E-6	100%	1.2E-7	100%	1.0E-6	100%	1.2E-7	100%
Chlor-alkali plant	4.0E-6	25%	4.8E-7	25%	1.4E-6	70%	1.7E-7	70%	1.1E-6	90%	1.3E-7	90%
Primary Copper Smelter	1.5E-6	68%	1.8E-7	68%	1.1E-6	91%	1.3E-7	91%	1.0E-6	97%	1.2E-7	97%
Primary Lead Smelter	3.5E-6	28%	4.3E-7	28%	1.6E-6	64%	1.9E-7	64%	1.2E-6	87%	1.4E-7	87%

Table 6-30
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban Average Resident at Western Site

Western Site	Predicted Intake for Urban Average (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.5E-6	3%	3.0E-7	3%	7.8E-7	9%	9.5E-8	9%	3.6E-7	20%	4.3E-8	20%
Small Municipal Waste Combustor	5.3E-7	14%	6.4E-8	14%	2.1E-7	35%	2.6E-8	35%	1.3E-7	58%	1.5E-8	58%
Continuous Medical Waste Incinerator	5.9E-7	12%	7.1E-8	12%	2.0E-7	36%	2.5E-8	36%	1.2E-7	63%	1.4E-8	63%
Intermittent Medical Waste Incinerator	9.2E-8	79%	1.1E-8	79%	7.7E-8	95%	9.4E-9	95%	7.3E-8	98%	9.0E-9	98%
Large Coal-fired Utility Boiler	1.5E-7	48%	1.8E-8	48%	1.0E-7	73%	1.2E-8	73%	8.2E-8	89%	9.9E-9	89%
Medium Coal-fired Utility Boiler	1.3E-7	56%	1.6E-8	56%	8.8E-8	83%	1.1E-8	83%	7.8E-8	93%	9.5E-9	93%
Small Coal-fired Utility Boiler	8.4E-8	87%	1.0E-8	87%	7.6E-8	96%	9.2E-9	96%	7.4E-8	98%	9.0E-9	98%
Medium Oil-fired Utility Boiler	7.4E-8	98%	9.0E-9	98%	7.3E-8	99%	8.9E-9	99%	7.3E-8	100%	8.9E-9	100%
Chlor-alkali plant	2.6E-6	3%	3.1E-7	3%	4.6E-7	16%	5.6E-8	16%	1.9E-7	39%	2.3E-8	39%
Primary Copper Smelter	2.3E-7	32%	2.8E-8	32%	1.2E-7	63%	1.4E-8	63%	9.0E-8	81%	1.1E-8	81%
Primary Lead Smelter	9.6E-7	8%	1.2E-7	8%	3.4E-7	22%	4.1E-8	22%	1.8E-7	41%	2.2E-8	41%

Western Site	Predicted Intake for Urban Average (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.7E-6	10%	3.2E-7	10%	9.8E-7	27%	1.2E-7	27%	5.5E-7	48%	6.7E-8	48%
Small Municipal Waste Combustor	7.2E-7	37%	8.7E-8	37%	4.0E-7	66%	4.9E-8	66%	3.2E-7	83%	3.9E-8	83%
Continuous Medical Waste Incinerator	7.8E-7	34%	9.5E-8	34%	4.0E-7	67%	4.8E-8	67%	3.1E-7	86%	3.8E-8	86%
Intermittent Medical Waste Incinerator	2.9E-7	93%	3.5E-8	93%	2.7E-7	98%	3.3E-8	98%	2.7E-7	100%	3.3E-8	100%
Large Coal-fired Utility Boiler	3.4E-7	77%	4.2E-8	77%	2.9E-7	91%	3.6E-8	91%	2.8E-7	97%	3.3E-8	97%
Medium Coal-fired Utility Boiler	3.2E-7	82%	3.9E-8	82%	2.8E-7	95%	3.4E-8	95%	2.7E-7	98%	3.3E-8	98%
Small Coal-fired Utility Boiler	2.8E-7	96%	3.4E-8	96%	2.7E-7	99%	3.3E-8	99%	2.7E-7	100%	3.3E-8	100%
Medium Oil-fired Utility Boiler	2.7E-7	99%	3.3E-8	99%	2.7E-7	100%	3.2E-8	100%	2.7E-7	100%	3.2E-8	100%
Chlor-alkali plant	2.8E-6	10%	3.4E-7	10%	6.6E-7	40%	8.0E-8	40%	3.8E-7	70%	4.6E-8	70%
Primary Copper Smelter	4.2E-7	63%	5.1E-8	63%	3.1E-7	86%	3.8E-8	86%	2.8E-7	94%	3.4E-8	94%
Primary Lead Smelter	1.2E-6	23%	1.4E-7	23%	5.3E-7	50%	6.4E-8	50%	3.7E-7	72%	4.5E-8	72%

Table 6-31
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban High-End Scenario at Eastern Site

Eastern Site	Predicted Intake for Urban High End (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.6E-4	7%	4.8E-6	65%	6.3E-5	28%	3.7E-6	83%	2.9E-5	60%	3.4E-6	92%
Small Municipal Waste Combustor	5.6E-5	32%	3.5E-6	89%	2.6E-5	69%	3.3E-6	95%	2.0E-5	89%	3.2E-6	98%
Continuous Medical Waste Incinerator	4.8E-5	37%	3.7E-6	85%	2.4E-5	74%	3.3E-6	95%	1.9E-5	91%	3.2E-6	98%
Intermittent Medical Waste Incinerator	1.9E-5	94%	3.1E-6	99%	1.8E-5	99%	3.1E-6	100%	1.8E-5	100%	3.1E-6	100%
Large Coal-fired Utility Boiler	2.6E-5	69%	3.2E-6	99%	2.0E-5	90%	3.1E-6	100%	1.8E-5	98%	3.1E-6	100%
Medium Coal-fired Utility Boiler	2.4E-5	74%	3.1E-6	99%	1.9E-5	93%	3.1E-6	100%	1.8E-5	98%	3.1E-6	100%
Small Coal-fired Utility Boiler	1.9E-5	94%	3.1E-6	100%	1.8E-5	99%	3.1E-6	100%	1.8E-5	100%	3.1E-6	100%
Medium Oil-fired Utility Boiler	1.8E-5	99%	3.1E-6	100%	1.8E-5	100%	3.1E-6	100%	1.8E-5	100%	3.1E-6	100%
Chlor-alkali plant	1.3E-4	14%	1.1E-5	30%	3.4E-5	52%	4.7E-6	66%	2.2E-5	82%	3.7E-6	85%
Primary Copper Smelter	3.5E-5	50%	3.3E-6	95%	2.1E-5	83%	3.2E-6	97%	1.9E-5	95%	3.2E-6	98%
Primary Lead Smelter	1.1E-4	16%	4.3E-6	72%	3.9E-5	46%	3.9E-6	80%	2.3E-5	76%	3.5E-6	89%

Eastern Site	Predicted Intake for Urban High End (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.8E-4	13%	5.0E-6	67%	8.3E-5	45%	4.0E-6	84%	4.9E-5	76%	3.6E-6	93%
Small Municipal Waste Combustor	7.6E-5	49%	3.7E-6	90%	4.5E-5	83%	3.5E-6	96%	4.0E-5	94%	3.4E-6	98%
Continuous Medical Waste Incinerator	6.7E-5	56%	3.9E-6	85%	4.4E-5	86%	3.5E-6	95%	3.9E-5	96%	3.4E-6	98%
Intermittent Medical Waste Incinerator	3.8E-5	97%	3.3E-6	99%	3.8E-5	99%	3.3E-6	100%	3.7E-5	100%	3.3E-6	100%
Large Coal-fired Utility Boiler	4.5E-5	82%	3.4E-6	99%	3.9E-5	95%	3.3E-6	100%	3.8E-5	99%	3.3E-6	100%
Medium Coal-fired Utility Boiler	4.4E-5	86%	3.4E-6	99%	3.9E-5	97%	3.3E-6	100%	3.8E-5	99%	3.3E-6	100%
Small Coal-fired Utility Boiler	3.8E-5	97%	3.3E-6	100%	3.8E-5	99%	3.3E-6	100%	3.7E-5	100%	3.3E-6	100%
Medium Oil-fired Utility Boiler	3.8E-5	100%	3.3E-6	100%	3.7E-5	100%	3.3E-6	100%	3.7E-5	100%	3.3E-6	100%
Chlor-alkali plant	1.5E-4	25%	1.1E-5	31%	5.4E-5	70%	4.9E-6	68%	4.1E-5	90%	3.9E-6	86%
Primary Copper Smelter	5.5E-5	68%	3.5E-6	96%	4.1E-5	91%	3.4E-6	97%	3.8E-5	97%	3.4E-6	98%
Primary Lead Smelter	1.3E-4	28%	4.5E-6	73%	5.8E-5	64%	4.1E-6	81%	4.3E-5	87%	3.7E-6	89%

Table 6-32
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Urban High-End Scenario at Western Site

Western Site	Predicted Intake for Urban High End (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	9.2E-5	3%	3.8E-6	77%	2.9E-5	9%	3.4E-6	86%	1.3E-5	20%	3.2E-6	92%
Small Municipal Waste Combustor	1.9E-5	14%	3.2E-6	92%	7.8E-6	35%	3.0E-6	96%	4.7E-6	58%	3.0E-6	98%
Continuous Medical Waste Incinerator	2.2E-5	12%	3.4E-6	86%	7.5E-6	36%	3.1E-6	95%	4.3E-6	63%	3.0E-6	98%
Intermittent Medical Waste Incinerator	3.4E-6	79%	2.9E-6	99%	2.9E-6	95%	2.9E-6	100%	2.8E-6	98%	2.9E-6	100%
Large Coal-fired Utility Boiler	5.6E-6	48%	2.9E-6	99%	3.7E-6	73%	2.9E-6	100%	3.0E-6	89%	2.9E-6	100%
Medium Coal-fired Utility Boiler	4.8E-6	56%	2.9E-6	99%	3.3E-6	83%	2.9E-6	100%	2.9E-6	93%	2.9E-6	100%
Small Coal-fired Utility Boiler	3.1E-6	87%	2.9E-6	100%	2.8E-6	96%	2.9E-6	100%	2.7E-6	98%	2.9E-6	100%
Medium Oil-fired Utility Boiler	2.8E-6	98%	2.9E-6	100%	2.7E-6	99%	2.9E-6	100%	2.7E-6	100%	2.9E-6	100%
Chlor-alkali plant	9.6E-5	3%	1.0E-5	29%	1.7E-5	16%	4.4E-6	67%	6.9E-6	39%	3.4E-6	85%
Primary Copper Smelter	8.4E-6	32%	3.0E-6	96%	4.3E-6	63%	3.0E-6	98%	3.3E-6	81%	3.0E-6	98%
Primary Lead Smelter	3.6E-5	8%	3.8E-6	77%	1.2E-5	22%	3.5E-6	83%	6.6E-6	41%	3.4E-6	89%

Western Site	Predicted Intake for Urban High End (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	9.9E-5	10%	3.9E-6	78%	3.6E-5	27%	3.5E-6	86%	2.0E-5	48%	3.3E-6	92%
Small Municipal Waste Combustor	2.7E-5	37%	3.3E-6	92%	1.5E-5	66%	3.1E-6	96%	1.2E-5	83%	3.1E-6	98%
Continuous Medical Waste Incinerator	2.9E-5	34%	3.5E-6	86%	1.5E-5	67%	3.2E-6	95%	1.1E-5	86%	3.1E-6	98%
Intermittent Medical Waste Incinerator	1.1E-5	93%	3.0E-6	99%	1.0E-5	98%	3.0E-6	100%	9.9E-6	100%	3.0E-6	100%
Large Coal-fired Utility Boiler	1.3E-5	77%	3.0E-6	99%	1.1E-5	91%	3.0E-6	100%	1.0E-5	97%	3.0E-6	100%
Medium Coal-fired Utility Boiler	1.2E-5	82%	3.0E-6	100%	1.0E-5	95%	3.0E-6	100%	1.0E-5	98%	3.0E-6	100%
Small Coal-fired Utility Boiler	1.0E-5	96%	3.0E-6	100%	1.0E-5	99%	3.0E-6	100%	9.9E-6	100%	3.0E-6	100%
Medium Oil-fired Utility Boiler	9.9E-6	99%	3.0E-6	100%	9.9E-6	100%	3.0E-6	100%	9.9E-6	100%	3.0E-6	100%
Chlor-alkali plant	1.0E-4	10%	1.0E-5	29%	2.4E-5	40%	4.5E-6	67%	1.4E-5	70%	3.5E-6	86%
Primary Copper Smelter	1.6E-5	63%	3.1E-6	97%	1.1E-5	86%	3.1E-6	98%	1.1E-5	94%	3.1E-6	98%
Primary Lead Smelter	4.3E-5	23%	3.9E-6	77%	2.0E-5	50%	3.6E-6	83%	1.4E-5	72%	3.4E-6	89%

Table 6-33
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for High-End Fisher at Eastern Site

Eastern Site	Predicted Intake for High End Fish Ingestion (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	5.5E-3	5%	4.0E-3	5%	1.0E-3	26%	7.3E-4	27%	4.3E-4	61%	3.2E-4	61%
Small Municipal Waste Combustor	1.1E-3	24%	7.9E-4	25%	3.9E-4	68%	2.8E-4	68%	3.0E-4	89%	2.2E-4	89%
Continuous Medical Waste Incinerator	9.0E-4	29%	6.6E-4	29%	3.6E-4	72%	2.7E-4	72%	2.9E-4	91%	2.1E-4	91%
Intermittent Medical Waste Incinerator	2.9E-4	92%	2.1E-4	92%	2.7E-4	99%	2.0E-4	99%	2.6E-4	100%	1.9E-4	100%
Large Coal-fired Utility Boiler	4.2E-4	62%	3.1E-4	63%	2.9E-4	90%	2.1E-4	90%	2.7E-4	98%	2.0E-4	98%
Medium Coal-fired Utility Boiler	3.9E-4	68%	2.8E-4	68%	2.8E-4	93%	2.1E-4	93%	2.7E-4	98%	2.0E-4	98%
Small Coal-fired Utility Boiler	2.9E-4	92%	2.1E-4	92%	2.7E-4	99%	2.0E-4	99%	2.6E-4	100%	1.9E-4	100%
Medium Oil-fired Utility Boiler	2.7E-4	99%	2.0E-4	99%	2.6E-4	100%	1.9E-4	100%	2.6E-4	100%	1.9E-4	100%
Chlor-alkali plant	2.8E-3	9%	2.0E-3	10%	5.4E-4	49%	3.9E-4	49%	3.2E-4	81%	2.4E-4	81%
Primary Copper Smelter	6.3E-4	42%	4.6E-4	42%	3.2E-4	82%	2.4E-4	82%	2.8E-4	95%	2.0E-4	95%
Primary Lead Smelter	2.3E-3	12%	1.6E-3	12%	5.9E-4	44%	4.3E-4	45%	3.5E-4	76%	2.5E-4	76%

Eastern Site	Predicted Intake for High End Fish Ingestion (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	5.7E-3	9%	4.2E-3	9%	1.3E-3	43%	9.3E-4	43%	7.1E-4	76%	5.2E-4	76%
Small Municipal Waste Combustor	1.4E-3	40%	9.9E-4	40%	6.7E-4	81%	4.9E-4	81%	5.8E-4	94%	4.2E-4	94%
Continuous Medical Waste Incinerator	1.2E-3	46%	8.6E-4	46%	6.4E-4	84%	4.7E-4	84%	5.7E-4	96%	4.2E-4	96%
Intermittent Medical Waste Incinerator	5.7E-4	96%	4.1E-4	96%	5.5E-4	99%	4.0E-4	99%	5.4E-4	100%	4.0E-4	100%
Large Coal-fired Utility Boiler	7.0E-4	77%	5.1E-4	77%	5.7E-4	95%	4.2E-4	95%	5.5E-4	99%	4.0E-4	99%
Medium Coal-fired Utility Boiler	6.7E-4	81%	4.9E-4	81%	5.6E-4	96%	4.1E-4	96%	5.5E-4	99%	4.0E-4	99%
Small Coal-fired Utility Boiler	5.7E-4	96%	4.1E-4	96%	5.5E-4	99%	4.0E-4	99%	5.4E-4	100%	4.0E-4	100%
Medium Oil-fired Utility Boiler	5.5E-4	99%	4.0E-4	99%	5.4E-4	100%	4.0E-4	100%	5.4E-4	100%	4.0E-4	100%
Chlor-alkali plant	3.1E-3	18%	2.2E-3	18%	8.2E-4	66%	6.0E-4	66%	6.0E-4	90%	4.4E-4	90%
Primary Copper Smelter	9.1E-4	60%	6.6E-4	60%	6.0E-4	90%	4.4E-4	90%	5.6E-4	97%	4.1E-4	97%
Primary Lead Smelter	2.5E-3	21%	1.9E-3	21%	8.7E-4	62%	6.4E-4	62%	6.3E-4	87%	4.6E-4	87%

Table 6-34
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for High-End Fisher at Western Site

Western Site	Predicted Intake for High End Fish Ingestion (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 50th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.2E-3	3%	1.6E-3	3%	5.9E-4	10%	4.3E-4	10%	2.5E-4	24%	1.8E-4	24%
Small Municipal Waste Combustor	4.6E-4	13%	3.4E-4	13%	1.6E-4	37%	1.2E-4	37%	9.6E-5	62%	7.1E-5	63%
Continuous Medical Waste Incinerator	5.3E-4	11%	3.9E-4	12%	1.6E-4	38%	1.2E-4	38%	8.8E-5	67%	6.6E-5	68%
Intermittent Medical Waste Incinerator	7.8E-5	76%	5.8E-5	77%	6.3E-5	95%	4.7E-5	95%	6.0E-5	99%	4.5E-5	99%
Large Coal-fired Utility Boiler	1.3E-4	47%	9.4E-5	48%	8.0E-5	75%	5.9E-5	75%	6.6E-5	90%	4.9E-5	91%
Medium Coal-fired Utility Boiler	1.1E-4	56%	7.9E-5	56%	7.1E-5	83%	5.3E-5	84%	6.3E-5	94%	4.8E-5	94%
Small Coal-fired Utility Boiler	6.9E-5	86%	5.2E-5	86%	6.2E-5	96%	4.7E-5	96%	6.0E-5	99%	4.5E-5	99%
Medium Oil-fired Utility Boiler	6.1E-5	98%	4.6E-5	98%	6.0E-5	99%	4.5E-5	99%	6.0E-5	100%	4.5E-5	100%
Chlor-alkali plant	2.6E-3	2%	1.9E-3	2%	3.7E-4	16%	2.7E-4	17%	1.4E-4	42%	1.0E-4	43%
Primary Copper Smelter	1.9E-4	31%	1.4E-4	32%	9.3E-5	64%	6.9E-5	65%	7.1E-5	83%	5.3E-5	84%
Primary Lead Smelter	8.6E-4	7%	6.3E-4	7%	2.6E-4	23%	1.9E-4	24%	1.3E-4	45%	9.8E-5	46%

Western Site	Predicted Intake for High End Fish Ingestion (mg/kg/day)											
	2.5 km				10 km				25 km			
COMPDEP + RELMAP 90th	Child		Adult		Child		Adult		Child		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.3E-3	8%	1.7E-3	8%	7.1E-4	25%	5.2E-4	26%	3.7E-4	48%	2.7E-4	48%
Small Municipal Waste Combustor	5.8E-4	31%	4.2E-4	31%	2.8E-4	64%	2.1E-4	64%	2.2E-4	83%	1.6E-4	83%
Continuous Medical Waste Incinerator	6.5E-4	27%	4.8E-4	28%	2.8E-4	65%	2.0E-4	65%	2.1E-4	86%	1.5E-4	86%
Intermittent Medical Waste Incinerator	2.0E-4	91%	1.5E-4	91%	1.8E-4	98%	1.3E-4	98%	1.8E-4	100%	1.3E-4	100%
Large Coal-fired Utility Boiler	2.5E-4	73%	1.8E-4	73%	2.0E-4	90%	1.5E-4	90%	1.9E-4	97%	1.4E-4	97%
Medium Coal-fired Utility Boiler	2.3E-4	79%	1.7E-4	79%	1.9E-4	94%	1.4E-4	94%	1.8E-4	98%	1.3E-4	98%
Small Coal-fired Utility Boiler	1.9E-4	95%	1.4E-4	95%	1.8E-4	99%	1.3E-4	99%	1.8E-4	100%	1.3E-4	100%
Medium Oil-fired Utility Boiler	1.8E-4	99%	1.3E-4	99%	1.8E-4	100%	1.3E-4	100%	1.8E-4	100%	1.3E-4	100%
Chlor-alkali plant	2.7E-3	7%	2.0E-3	7%	4.9E-4	37%	3.6E-4	37%	2.6E-4	69%	1.9E-4	69%
Primary Copper Smelter	3.1E-4	57%	2.3E-4	58%	2.1E-4	84%	1.6E-4	85%	1.9E-4	94%	1.4E-4	94%
Primary Lead Smelter	9.8E-4	18%	7.1E-4	18%	3.8E-4	47%	2.8E-4	48%	2.5E-4	71%	1.8E-4	71%

Table 6-35
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Recreational Angler at Eastern Site

Eastern Site	Predicted Intake for Recreational Angler (mg/kg/day)					
	2.5 km		10 km		25 km	
COMPDEP + RELMAP 50th	Adult		Adult		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.0E-3	5%	3.6E-4	26%	1.5E-4	60%
Small Municipal Waste Combustor	3.9E-4	24%	1.4E-4	67%	1.0E-4	89%
Continuous Medical Waste Incinerator	3.2E-4	29%	1.3E-4	72%	1.0E-4	91%
Intermittent Medical Waste Incinerator	1.0E-4	92%	9.4E-5	99%	9.3E-5	100%
Large Coal-fired Utility Boiler	1.5E-4	61%	1.0E-4	90%	9.5E-5	98%
Medium Coal-fired Utility Boiler	1.4E-4	67%	1.0E-4	93%	9.4E-5	98%
Small Coal-fired Utility Boiler	1.0E-4	92%	9.4E-5	99%	9.3E-5	100%
Medium Oil-fired Utility Boiler	9.4E-5	98%	[illegible]	100%	9.3E-5	100%
Chlor-alkali plant	1.0E-3	9%	1.9E-4	49%	1.1E-4	81%
Primary Copper Smelter	2.2E-4	41%	1.1E-4	81%	9.8E-5	95%
Primary Lead Smelter	8.2E-4	11%	2.1E-4	44%	1.2E-4	76%

Eastern Site	Predicted Intake for Recreational Angler (mg/kg/day)					
	2.5 km		10 km		25 km	
COMPDEP + RELMAP 90th	Adult		Adult		Adult	
Facility	Value	RELMAP	Value	RELMAP	Value	RELMAP
Large Municipal Waste Combustor	2.1E-3	9%	4.6E-4	42%	2.6E-4	76%
Small Municipal Waste Combustor	4.9E-4	40%	2.4E-4	81%	2.1E-4	94%
Continuous Medical Waste Incinerator	4.3E-4	46%	2.3E-4	84%	2.0E-4	95%
Intermittent Medical Waste Incinerator	2.0E-4	96%	2.0E-4	99%	1.9E-4	100%
Large Coal-fired Utility Boiler	2.5E-4	77%	2.1E-4	95%	2.0E-4	99%
Medium Coal-fired Utility Boiler	2.4E-4	81%	2.0E-4	96%	2.0E-4	99%
Small Coal-fired Utility Boiler	2.0E-4	96%	2.0E-4	99%	1.9E-4	100%
Medium Oil-fired Utility Boiler	2.0E-4	99%	1.9E-4	100%	1.9E-4	100%
Chlor-alkali plant	1.1E-3	18%	2.9E-4	66%	2.2E-4	90%
Primary Copper Smelter	3.3E-4	60%	2.2E-4	90%	2.0E-4	97%
Primary Lead Smelter	9.2E-4	21%	3.1E-4	62%	2.2E-4	87%

Table 6-36
Combination of Local and Regional Impacts: Predicted Mercury Ingestion Intake for Recreational Angler at Western Site

Western Site	Predicted Intake for Recreational Angler (mg/kg/day)								
	2.5 km			10 km			25 km		
COMPDEP + RELMAP 50th	Adult			Adult			Adult		
Facility	Value	RELMAP		Value	RELMAP		Value	RELMAP	
Large Municipal Waste Combustor	8.0E-4	2%		2.1E-4	9%		8.9E-5	21%	
Small Municipal Waste Combustor	1.7E-4	11%		5.6E-5	33%		3.2E-5	59%	
Continuous Medical Waste Incinerator	1.9E-4	10%		5.4E-5	34%		2.9E-5	64%	
Intermittent Medical Waste Incinerator	2.5E-5	73%		2.0E-5	94%		1.9E-5	98%	
Large Coal-fired Utility Boiler	4.3E-5	43%		2.6E-5	72%		2.1E-5	89%	
Medium Coal-fired Utility Boiler	3.6E-5	52%		2.3E-5	81%		2.0E-5	93%	
Small Coal-fired Utility Boiler	2.2E-5	84%		1.9E-5	95%		1.9E-5	98%	
Medium Oil-fired Utility Boiler	1.9E-5	97%		1.9E-5	99%		1.9E-5	100%	
Chlor-alkali plant	9.3E-4	2%		1.3E-4	14%		4.8E-5	39%	
Primary Copper Smelter	6.7E-5	28%		3.1E-5	61%		2.3E-5	81%	
Primary Lead Smelter	3.1E-4	6%		9.1E-5	20%		4.5E-5	41%	

Western Site	Predicted Intake for Recreational Angler (mg/kg/day)								
	2.5 km			10 km			25 km		
COMPDEP + RELMAP 90th	Adult			Adult			Adult		
Facility	Value	RELMAP		Value	RELMAP		Value	RELMAP	
Large Municipal Waste Combustor	8.5E-4	7%		2.6E-4	24%		1.3E-4	47%	
Small Municipal Waste Combustor	2.1E-4	30%		1.0E-4	62%		7.5E-5	83%	
Continuous Medical Waste Incinerator	2.3E-4	27%		9.8E-5	63%		7.3E-5	86%	
Intermittent Medical Waste Incinerator	6.9E-5	90%		6.3E-5	98%		6.3E-5	99%	
Large Coal-fired Utility Boiler	8.7E-5	72%		7.0E-5	89%		6.5E-5	96%	
Medium Coal-fired Utility Boiler	8.0E-5	78%		6.7E-5	93%		6.4E-5	98%	
Small Coal-fired Utility Boiler	6.6E-5	95%		6.3E-5	99%		6.3E-5	99%	
Medium Oil-fired Utility Boiler	6.3E-5	99%		6.2E-5	100%		6.2E-5	100%	
Chlor-alkali plant	9.7E-4	6%		1.7E-4	36%		9.2E-5	68%	
Primary Copper Smelter	1.1E-4	56%		7.4E-5	84%		6.7E-5	94%	
Primary Lead Smelter	3.5E-4	18%		1.3E-4	46%		8.8E-5	70%	

6.3.3 Issues Related to Combining the Modeled Exposure Estimates with Estimates of Exposure from Other Sources

In the modeling effort exposure estimates for six different hypothetical adult humans were modeled. The mercury exposures were estimated using the predictions of the RELMAP, COMPDEP and IEM2 modeling have been quantified. Atmospheric emissions of anthropogenic origin are not the only sources of mercury. As noted in Chapter 3 of this Volume, individuals can be exposed to mercury from sources such as occupational activities and fish consumption.

Occupational mercury may be an important source of exposure in some subpopulations. The expected exposure route would be primarily inhalation, although some workers could be expected to experience some degree of dermal contact. In this analysis this source of exposure may apply to any hypothetical adult considered with the exception of the subsistence farmer. Mercury from occupational exposures was not combined with the modeling results because of the hypothetical nature of both the humans and the sites that were modeled. For a site-specific assessment, it may be appropriate to consider occupational mercury exposure for members of the population.

In the modeling effort several hypothetical humans were assumed to not consume locally-caught fish. The hypothetical individuals included these: a subsistence farmer and child, rural home gardener, urban dwellers and pica child. Methylmercury exposure from pathways other than fish consumption, if considered, would be an incremental increase over the intake received through fish consumption. This estimated intake of methylmercury received through fish consumption by the U.S. population is thought to be a range of exposures. For a site-specific assessment, inclusion of this range of methylmercury exposure from fish consumption must be considered.

In the modeling effort several hypothetical individuals were assumed to consume high levels of locally caught fish. These individuals include the following: an angler, who is assumed to consume 60 grams of local fish/day; a child, who is assumed to consume 20 grams of local fish/day; and a recreational angler who is assumed to consume 30 grams fish/day.

Only the existing background mercury concentrations in the atmosphere were modeled in Chapters 4, 5, and 6. No attempt was made to model background mercury concentrations in other media or biota in these Chapters. Clearly, there are geographic areas with higher concentrations of mercury in media or biota. It may be appropriate to consider existing environmental mercury concentrations for some risk assessment efforts.

6.3.4 Wildlife Exposure

Tables 6-37 and 6-38 show the predicted methylmercury intake for the wildlife receptors considered. The relative contributions of regional and local sources are identical to those for the surface water concentrations, but are repeated here for completeness.

IT IS REASONABLE TO ASSUME THAT SOME EXPOSURE TO METHYLMERCURY OCCURS THROUGH CONSUMPTION OF MARINE FISH. THESE EXPOSURES (ESTIMATED IN APPENDIX H) ARE APPLICABLE TO INDIVIDUALS IN THESE GROUPS WHO CONSUME FISH

Table 6-37
Combination of Local and Regional Impacts: Predicted Methylmercury Intake for Wildlife Receptors at Eastern Site

Facility	% Regional using RELMAP 50th	% Regional using RELMAP 90th	Predicted Methylmercury Intakes (mg/kg/day) Local + RELMAP 50th					Local + RELMAP 90th				
			Bald Eagle	Osprey	Kingfisher	River Otter	Mink	Bald Eagle	Osprey	Kingfisher	River Otter	Mink
2 5 km												
Large Municipal Waste Combustor	4.7%	9 3%	1 6E-1	1 8E-1	4 6E-1	2.7E-1	1 8E-1	1 7E-1	1 9E-1	4 8E-1	2 9E-1	1 9E-1
Small Municipal Waste Combustor	23 8%	39.5%	3 2E-2	3.6E-2	9.0E-2	5.4E-2	3 6E-2	4 1E-2	4 5E-2	1 1E-1	6 8E-2	4 5E-2
Continuous Medical Waste Incinerator	28.5%	45.6%	2.7E-2	3 0E-2	7 5E-2	4 5E-2	3 0E-2	3 5E-2	3 9E-2	9 8E-2	5 9E-2	3 9E-2
Intermittent Medical Waste Incinerator	91.9%	96 0%	8.3E-3	9.3E-3	2 3E-2	1 4E-2	9 3E-3	1.7E-2	1 9E-2	4.7E-2	2 8E-2	1 9E-2
Large Coal-fired Utility Boiler	61.5%	77 0%	1.2E-2	1 4E-2	3 5E-2	2 1E-2	1 4E-2	2 1E-2	2.3E-2	5 8E-2	3 5E-2	2 3E-2
Medium Coal-fired Utility Boiler	66.9%	80 9%	1.1E-2	1 3E-2	3 2E-2	1 9E-2	1 3E-2	2 0E-2	2 2E-2	5 5E-2	3 3E-2	2 2E-2
Small Coal-fired Utility Boiler	91.9%	96.0%	8 3E-3	9 3E-3	2 3E-2	1.4E-2	9 3E-3	1 7E-2	1 9E-2	4.7E-2	2 8E-2	1 9E-2
Medium Oil-fired Utility Boiler	98 5%	99.3%	7 8E-3	8.6E-3	2 2E-2	1 3E-2	8.7E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Chlor-alkali plant	9 2%	17.6%	8 3E-2	9 2E-2	2 3E-1	1 4E-1	9 2E-2	9 1E-2	1 0E-1	2 5E-1	1 5E-1	1 0E-1
Primary Copper Smelter	41.1%	59.5%	1 9E-2	2 1E-2	5.2E-2	3 1E-2	2 1E-2	2 7E-2	3 0E-2	7 5E-2	4 5E-2	3 0E-2
Primary Lead Smelter	11.3%	21.1%	6.8E-2	7 6E-2	1 9E-1	1.1E-1	7 6E-2	7 6E-2	8 5E-2	2 1E-1	1 3E-1	8 5E-2
10 km												
Large Municipal Waste Combustor	25.8%	42.2%	3 0E-2	3 3E-2	8 3E-2	4.9E-2	3 3E-2	3 8E-2	4 2E-2	1 1E-1	6 3E-2	4 2E-2
Small Municipal Waste Combustor	67 0%	81.0%	1.1E-2	1 3E-2	3.2E-2	1.9E-2	1.3E-2	2 0E-2	2 2E-2	5 5E-2	3 3E-2	2 2E-2
Continuous Medical Waste Incinerator	71 6%	84.1%	1 1E-2	1 2E-2	3 0E-2	1 8E-2	1 2E-2	1 9E-2	2 1E-2	5.3E-2	3 2E-2	2 1E-2
Intermittent Medical Waste Incinerator	98 7%	99.4%	7 7E-3	8.6E-3	2 2E-2	1 3E-2	8 6E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Large Coal-fired Utility Boiler	89.5%	94.7%	8 5E-3	9 5E-3	2.4E-2	1 4E-2	9 5E-3	1 7E-2	1 9E-2	4 7E-2	2 8E-2	1 9E-2
Medium Coal-fired Utility Boiler	92.5%	96.3%	8 3E-3	9 2E-3	2 3E-2	1.4E-2	9 2E-3	1 7E-2	1 9E-2	4 6E-2	2 8E-2	1 9E-2
Small Coal-fired Utility Boiler	98 6%	99.3%	7 7E-3	8 6E-3	2 2E-2	1 3E-2	8 6E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Medium Oil-fired Utility Boiler	99 8%	99.9%	7.7E-3	8 5E-3	2 1E-2	1 3E-2	8 5E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Chlor-alkali plant	48 5%	66.4%	1 6E-2	1 8E-2	4 4E-2	2.6E-2	1 8E-2	2 4E-2	2 7E-2	6 7E-2	4 0E-2	2 7E-2
Primary Copper Smelter	81.3%	90.1%	9 4E-3	1 0E-2	2.6E-2	1 6E-2	1 0E-2	1 8E-2	2.0E-2	5 0E-2	3 0E-2	2 0E-2
Primary Lead Smelter	43 6%	61.9%	1 8E-2	2 0E-2	4 9E-2	2 9E-2	2.0E-2	2 6E-2	2.9E-2	7 2E-2	4 3E-2	2 9E-2
25 km												
Large Municipal Waste Combustor	60.2%	76.0%	1 3E-2	1 4E-2	3 5E-2	2 1E-2	1 4E-2	2 1E-2	2 4E-2	5 9E-2	3 5E-2	2 4E-2
Small Municipal Waste Combustor	88 8%	94.3%	8 6E-3	9 6E-3	2 4E-2	1 4E-2	9 6E-3	1 7E-2	1 9E-2	4 7E-2	2 8E-2	1 9E-2
Continuous Medical Waste Incinerator	91 0%	95.5%	8 4E-3	9.4E-3	2 3E-2	1 4E-2	9 4E-3	1 7E-2	1 9E-2	4 7E-2	2 8E-2	1 9E-2
Intermittent Medical Waste Incinerator	99 7%	99.9%	7.7E-3	8 5E-3	2 1E-2	1 3E-2	8 5E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Large Coal-fired Utility Boiler	97.7%	98.9%	7 8E-3	8 7E-3	2 2E-2	1 3E-2	8 7E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Medium Coal-fired Utility Boiler	98 3%	99.2%	7.8E-3	8 7E-3	2 2E-2	1 3E-2	8 7E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Small Coal-fired Utility Boiler	99 7%	99.8%	7 7E-3	8 5E-3	2 1E-2	1 3E-2	8 5E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Medium Oil-fired Utility Boiler	99 9%	100 0%	7 6E-3	8 5E-3	2 1E-2	1 3E-2	8 5E-3	1 6E-2	1 8E-2	4 5E-2	2 7E-2	1 8E-2
Chlor-alkali plant	81 3%	90.1%	9 4E-3	1 0E-2	2 6E-2	1 6E-2	1 0E-2	1 8E-2	2 0E-2	5 0E-2	3 0E-2	2 0E-2
Primary Copper Smelter	94 7%	97.4%	8 1E-3	9 0E-3	2 2E-2	1 3E-2	9 0E-3	1 6E-2	1 8E-2	4 6E-2	2 7E-2	1 8E-2
Primary Lead Smelter	75 6%	86.7%	1 0E-2	1 1E-2	2 8E-2	1 7E-2	1 1E-2	1 9E-2	2 1E-2	5 2E-2	3 1E-2	2 1E-2

Table 6-38
Combination of Local and Regional Impacts: Predicted Methylmercury Intake for Wildlife Receptors at Western Site

Facility	% Regional using RELMAP 50th	% Regional using RELMAP 90th	Predicted Methylmercury Intakes (mg/kg/day) Local + RELMAP 50th: Bald Eagle	Osprey	Kingfisher	River Otter	Mink	Local + RELMAP 90th: Bald Eagle	Osprey	Kingfisher	River Otter	Mink
2.5 km												
Large Municipal Waste Combustor	2.3%	7.3%	6.6E-2	7.4E-2	1.8E-1	1.1E-1	7.4E-2	7.0E-2	7.8E-2	1.9E-1	1.2E-1	7.8E-2
Small Municipal Waste Combustor	11.2%	29.7%	1.4E-2	1.5E-2	3.8E-2	2.3E-2	1.5E-2	1.7E-2	1.9E-2	4.8E-2	2.9E-2	1.9E-2
Continuous Medical Waste Incinerator	9.7%	26.5%	1.6E-2	1.8E-2	4.4E-2	2.6E-2	1.8E-2	1.9E-2	2.2E-2	5.4E-2	3.2E-2	2.2E-2
Intermittent Medical Waste Incinerator	73.4%	90.3%	2.1E-3	2.3E-3	5.8E-3	3.5E-3	2.3E-3	5.7E-3	6.3E-3	1.6E-2	9.4E-3	6.3E-3
Large Coal-fired Utility Boiler	42.9%	71.6%	3.6E-3	4.0E-3	9.9E-3	5.9E-3	4.0E-3	7.1E-3	8.0E-3	2.0E-2	1.2E-2	8.0E-3
Medium Coal-fired Utility Boiler	51.5%	78.1%	3.0E-3	3.3E-3	8.2E-3	4.9E-3	3.3E-3	6.6E-3	7.3E-3	1.8E-2	1.1E-2	7.3E-3
Small Coal-fired Utility Boiler	84.0%	94.7%	1.8E-3	2.0E-3	5.1E-3	3.0E-3	2.0E-3	5.4E-3	6.0E-3	1.5E-2	9.0E-3	6.0E-3
Medium Oil-fired Utility Boiler	97.3%	99.2%	1.6E-3	1.7E-3	4.4E-3	2.6E-3	1.7E-3	5.2E-3	5.8E-3	1.4E-2	8.6E-3	5.8E-3
Chlor-alkali plant	2.0%	6.4%	7.6E-2	8.5E-2	2.1E-1	1.3E-1	8.5E-2	8.0E-2	8.9E-2	2.2E-1	1.3E-1	8.9E-2
Primary Copper Smelter	27.6%	56.1%	5.5E-3	6.2E-3	1.5E-2	9.2E-3	6.2E-3	9.1E-3	1.0E-2	2.5E-2	1.5E-2	1.0E-2
Primary Lead Smelter	6.0%	17.6%	2.5E-2	2.8E-2	7.1E-2	4.2E-2	2.8E-2	2.9E-2	3.2E-2	8.1E-2	4.8E-2	3.2E-2
10 km												
Large Municipal Waste Combustor	8.8%	24.4%	1.7E-2	1.9E-2	4.9E-2	2.9E-2	1.9E-2	2.1E-2	2.3E-2	5.9E-2	3.5E-2	2.3E-2
Small Municipal Waste Combustor	33.1%	62.4%	4.6E-3	5.1E-3	1.3E-2	7.7E-3	5.1E-3	8.2E-3	9.1E-3	2.1E-2	1.4E-2	9.2E-3
Continuous Medical Waste Incinerator	34.0%	63.4%	4.5E-3	5.0E-3	1.2E-2	7.5E-3	5.0E-3	8.1E-3	9.0E-3	2.2E-2	1.3E-2	9.0E-3
Intermittent Medical Waste Incinerator	94.1%	98.2%	1.6E-3	1.8E-3	4.5E-3	2.7E-3	1.8E-3	5.2E-3	5.8E-3	1.5E-2	8.7E-3	5.8E-3
Large Coal-fired Utility Boiler	71.5%	89.4%	2.1E-3	2.4E-3	5.9E-3	3.5E-3	2.4E-3	5.7E-3	6.4E-3	1.6E-2	9.5E-3	6.4E-3
Medium Coal-fired Utility Boiler	80.9%	93.4%	1.9E-3	2.1E-3	5.2E-3	3.1E-3	2.1E-3	5.5E-3	6.1E-3	1.5E-2	9.1E-3	6.1E-3
Small Coal-fired Utility Boiler	95.4%	98.6%	1.6E-3	1.8E-3	4.5E-3	2.7E-3	1.8E-3	5.2E-3	5.8E-3	1.4E-2	8.6E-3	5.8E-3
Medium Oil-fired Utility Boiler	99.3%	99.8%	1.5E-3	1.7E-3	4.3E-3	2.6E-3	1.7E-3	5.1E-3	5.7E-3	1.4E-2	8.5E-3	5.7E-3
Chlor-alkali plant	14.2%	35.8%	1.1E-2	1.2E-2	3.0E-2	1.8E-2	1.2E-2	1.4E-2	1.6E-2	4.0E-2	2.4E-2	1.6E-2
Primary Copper Smelter	60.6%	83.8%	2.5E-3	2.8E-3	7.0E-3	4.2E-3	2.8E-3	6.1E-3	6.8E-3	1.7E-2	1.0E-2	6.8E-3
Primary Lead Smelter	20.4%	46.3%	7.5E-3	8.3E-3	2.1E-2	1.2E-2	8.3E-3	1.1E-2	1.2E-2	3.1E-2	1.8E-2	1.2E-2
25 km												
Large Municipal Waste Combustor	20.9%	47.0%	7.3E-3	8.1E-3	2.0E-2	1.2E-2	8.1E-3	1.1E-2	1.2E-2	3.0E-2	1.8E-2	1.2E-2
Small Municipal Waste Combustor	58.5%	82.6%	2.6E-3	2.9E-3	7.3E-3	4.3E-3	2.9E-3	6.2E-3	6.9E-3	1.7E-2	1.0E-2	6.9E-3
Continuous Medical Waste Incinerator	63.7%	85.5%	2.4E-3	2.7E-3	6.7E-3	4.0E-3	2.7E-3	6.0E-3	6.7E-3	1.7E-2	1.0E-2	6.7E-3
Intermittent Medical Waste Incinerator	98.3%	99.5%	1.6E-3	1.7E-3	4.3E-3	2.6E-3	1.7E-3	5.1E-3	5.7E-3	1.4E-2	8.6E-3	5.7E-3
Large Coal-fired Utility Boiler	89.0%	96.4%	1.7E-3	1.9E-3	4.8E-3	2.9E-3	1.9E-3	5.3E-3	5.9E-3	1.5E-2	8.8E-3	5.9E-3
Medium Coal-fired Utility Boiler	93.1%	97.8%	1.6E-3	1.8E-3	4.6E-3	2.7E-3	1.8E-3	5.2E-3	5.8E-3	1.5E-2	8.7E-3	5.8E-3
Small Coal-fired Utility Boiler	98.3%	99.5%	1.6E-3	1.7E-3	4.3E-3	2.6E-3	1.7E-3	5.1E-3	5.7E-3	1.4E-2	8.6E-3	5.7E-3
Medium Oil-fired Utility Boiler	99.8%	99.9%	1.5E-3	1.7E-3	4.3E-3	2.5E-3	1.7E-3	5.1E-3	5.7E-3	1.4E-2	8.5E-3	5.7E-3
Chlor-alkali plant	38.8%	68.0%	3.9E-3	4.4E-3	1.1E-2	6.5E-3	4.4E-3	7.5E-3	8.4E-3	2.1E-2	1.3E-2	8.4E-3
Primary Copper Smelter	81.2%	93.6%	1.9E-3	2.1E-3	5.2E-3	3.1E-3	2.1E-3	5.5E-3	6.1E-3	1.5E-2	9.1E-3	6.1E-3
Primary Lead Smelter	41.5%	70.4%	3.7E-3	4.1E-3	1.0E-2	6.1E-3	4.1E-3	7.3E-3	8.1E-3	2.0E-2	1.2E-2	8.1E-3

6.3.5 Combining the Results of Local and Regional Models: Summary Conclusions

Except for the chlor-alkali plant, the mercury air concentrations that were predicted to result from the emissions of the model plants were generally dominated by the regional sources. The larger contribution to the air concentration from the chlor-alkali plant was due to the predicted lower effective stack heights for this source. For the two hypothetical sites considered, the regional contribution was estimated using either the 50th or 90th percentile air concentration and deposition rate as predicted by RELMAP. The regional contribution was larger at the hypothetical eastern U.S. site than the western site.

Except for the utility boilers, mercury deposition rates at the closest receptor considered (2.5 Km from the local source) were predicted to be dominated by the emissions from the local source (i.e., the model plant), with the impact of the regional contribution increasing as a function of distance from the source. The predicted deposition rates were lower for the utility boilers due to lower mercury emission rates associated with the model plants developed to categorize this source class and the predicted high effective stack heights, which disperse the mercury to a greater degree than the other source classes.

The predicted green plant concentrations followed the air concentration patterns, and so the relative contributions of the regional and local sources were similar to that for the air concentrations. The predicted concentrations of mercury in soil, surface water and fish followed deposition patterns, and the relative contributions of the regional and local sources were similar to that for deposition.

For the agricultural scenarios, the ingestion of plants was the primary route of exposure to mercury, and hence the contributions of regional and local sources were similar to that for the air concentrations. For the human fish ingestion scenarios and piscivorous wildlife exposure scenarios, the predicted intakes follow deposition patterns. Since deposition from local sources generally dominated total deposition close to the model plants, exposure through fish consumption followed local deposition patterns. As the predicted deposition from the local source decreased with increasing distance and regional deposition as a fraction of total deposition increased, the impact deposition of mercury predicted to result from regional source emissions increased.

Because of the hypothetical nature of both the individual humans and the sites that were considered, estimates of exposures to mercury resulting from the consumption of non-local fish, from occupation or from background sources, other than the atmosphere, were not added to the exposure estimates developed in Chapter 6 of this volume. These sources of mercury exposure may be significant and for a site-specific assessment it may be appropriate to consider these for members of the population.

Piscivorous wildlife may also be exposed to mercury from other sources. For example, they may be exposed through the non-fish portion of their diet or through consumption of drinking water. It may be appropriate to consider these sources for a specific site assessment.

6.4 Uncertainty and Sensitivity Analyses

As has been noted previously, the behavior of atmospheric mercury close to the point of release has not been studied extensively. This alone results in a significant degree of uncertainty implicit in the preceding modeling exercises. In this section, several of these assumptions along with other possible behaviors are examined to illustrate the implications of these potential properties of atmospheric mercury in the near-field.

6.4.1 Dry Deposition

In this study, divalent mercury vapor was assumed to deposit in a manner similar to nitric acid, and elemental mercury was assumed to dry deposit at a negligible rate. In particular, a dry deposition velocity of 1 cm/s during daytime conditions and 0.3 cm/s during nighttime conditions was assumed for divalent mercury vapor, and a dry deposition velocity of 0 cm/s was used for elemental mercury vapor. It is noted that under certain conditions dry deposition velocities of nitric acid have been estimated to be as high as 4 cm/s (see Appendix D).

In this section the results using this assumption are compared to the results using the assumption that all mercury vapor dry deposits with a deposition velocity of 0.06 cm/s, a rate based on the work of Lindberg et al., (1991).

Lindberg et al., (1991) calculated dry deposition velocities for total mercury vapor (assumed to be all Hg^0) to a forest canopy in eastern Tennessee. It was assumed that the dry deposition of mercury vapor to plant canopies is based exclusively on the leaf's physiology and biochemistry. Dominant processes for mercury uptake are gas exchange at the leaf's surface via stomata followed by mercury assimilation at the gas-liquid interface within the leaf mesophyll. Lindberg et al., (1991) modified the "big leaf" aerodynamic resistance (resistance the leaf provides to transport of atmospheric pollutants to the leaf interior) model of Hicks et al., (1987) to infer mean hourly dry deposition velocities for mercury vapor. Detailed hourly meteorological data and canopy measurements were used in this model to calculate the mean total resistance and its reciprocal, the dry deposition velocity. Weekly mean Hg^0 dry deposition velocities ranged from 0.006 in winter to 0.12 cm/s in summer, from which an average value of 0.06 cm/s is taken. This value is probably a low-end value for mercury vapor. Lindberg's group reported in Hanson et al., (1994) that dry deposition to the forest canopy may only occur when mercury concentrations exceeded a "compensation concentration" around 15 ng/m^3. This means overall deposition may not occur under typical ambient concentrations of gaseous mercury, and illustrates the uncertainties in calculation and measurement of dry deposition velocities for mercury.

The alternate deposition rate used in the present analysis based on Lindberg et al., (1991) was chosen more to provide a lower-end bound for the dry deposition modeling rather than represent an accurate approximation to the actual mercury deposition process near an anthropogenic source. Additionally, Lindberg et al., (1991) recognize that Hg^{2+} is thought to deposit much more readily than Hg^0 and the dry deposition velocities reported do not necessarily apply to Hg^{2+}.

When a single dry deposition velocity for all vapor-phase mercury emissions is used, the vapor-phase dry deposition flux is the result of the mass of vapor-phase mercury emissions and the stack characteristics. The assumption of a single dry deposition velocity for vapor-phase mercury ties the total dry deposition rate more closely to the mass of mercury vapor emitted from the stack compared to the method that estimates different deposition velocities for vapor phase Hg^0 and Hg^{2+}.

Table 6-39 shows the results for select facilities using both assumptions.

Table 6-39
Comparison of Results using Different Assumptions Regarding Dry Deposition of Mercury Vapor

Plant	Base Emissions Speciation (% of Total Hg Emissions)			Predicted Dry Deposition Rate (ug/m^2/yr)					
				2.5 km Watershed		10 km Watershed		25 km Watershed	
	Hg0 Vapor	Hg^{2+} Vapor	Hg^{2+} Particulate	#1[A]	#2[B]	#1[A]	#2[B]	#1[A]	#2[B]
Large Municipal Waste Combustor	20	60	20	5.59	26.28	4.79	17.16	2.45	8.21
Small Municipal Waste Combustor	20	60	20	2.25	8.82	1.34	4.44	0.59	1.85
Continuous Medical Waste Incinerator	20	60	20	4.78	15.36	1.73	4.89	0.64	1.63
Intermittent Medical Waste Incinerator	20	60	20	0.20	0.62	0.06	0.16	0.02	0.05
Large Coal-fired Utility Boiler	50	30	20	0.03	0.13	0.04	0.14	0.03	0.08
Chlor-alkali Plant	70	30	0	70.78	82.70	15.13	14.08	4.85	3.68
Primary Copper Smelter	85	10	5	0.73	0.92	0.75	0.64	0.47	0.34
Primary Lead Smelter	85	10	5	7.66	7.93	6.91	5.07	3.65	2.41

[a] Assuming dry deposition velocity of 0.06 cm/s for all mercury vapor.
[b] Assuming dry deposition velocity of 0 cm/s for elemental mercury vapor, 1 cm/s for divalent mercury vapor during daytime, and 0.3 cm/s during nighttime.

The difference in estimated deposition rates between the two methods is the largest for those facilities assumed to emit the highest levels of divalent mercury vapor. This is the result of changes in the assumption of dry deposition velocity associated with this fraction of the total mercury emissions. Some increases in the overall estimated deposition rates are also observed using the first assumption, although these are considerably smaller than the reductions associated with the assumption of the same deposition velocity for both vapor-phase species emitted. Note the increases observed in the estimated dry deposition rates at both 10 and 25 km for the chlor-alkali plant and the smelters. For both sources this reflects the deposition of Hg^0, which was assumed to have a deposition velocity of 0 cm/s in the modeling exercise. Note that for both sources Hg^0 is assumed to be a large fraction of the total mercury emitted.

Since the calculated dry deposition velocity associated with the particle sizes assumed is generally larger than the assumed vapor-phase deposition velocity of 0.06 cm/s using the first assumption, the particulate deposition rate is significant for those facilities assumed to emit higher particulate levels.

If the vapor-phase divalent mercury dry deposition velocity is higher than the estimates used for daytime and nighttime conditions, then the model would predict a linear increase in the deposition rate associated with that fraction of the emissions that are divalent mercury vapor.

6.4.2 Wet Deposition

In the local impact analysis, wet deposition of particulate mercury is estimated by calculating a scavenging coefficient that depends on particle size and precipitation intensity, while wet deposition of vapor is estimated by converting a washout ratio to a scavenging coefficient (see Appendix D). The washout ratio is the ratio of the concentration in surface-level precipitation to the concentration in surface level air (Slinn 1984). Because most facilities are assumed to emit primarily vapor-phase

mercury, in this section the possible impacts of uncertainty in wet deposition of vapor are briefly discussed.

Due to its higher solubility, divalent mercury is thought to wet deposit at much higher rates than that of elemental mercury vapor. Determination of washout ratios for divalent mercury vapor is precluded by the limitations of current analytical measurement techniques: it has not been possible to obtain measurements of divalent vapor air concentrations, and hence there are no reported values in the peer-reviewed literature. For this reason, the washout ratio used for divalent mercury vapor is based on an assumed similarity between divalent mercury and nitric acid, for which washout ratios are available (Petersen 1995). In particular, a value of $1.6x10^6$ is used. Comparisons of concentrations in precipitation calculated using this value agree quite well for nitric acid. However, the applicability of this value for divalent mercury vapor is uncertain, as there may be other processes specific to mercury that would result in a smaller washout ratio. It seems that a lower bound for this value is on the order of 10^4, which is based on measured values for elemental mercury (Appendix D).

Because the washout ratio is used to calculate a scavenging coefficient, the effect of the uncertainty in the washout ratio is not strictly linear (see Appendix D) using three different washout ratios. A larger washout ratio results in a larger scavenging coefficient, which results in more of the plume being depleted closer to the source. Thus, at larger distances from the source, the predicted wet deposition may be higher using a smaller washout ratio. In the example here, this happens at about 22 km when comparing $1.6x10^6$ and $1.6x10^5$, and at about 40 km when comparing $1.6x10^6$ and $1.6x10^4$.

Thus, these results imply that the uncertainty in the washout ratio will primarily affect predictions of deposition close to the facility. These predictions are of course the most critical, and at present cannot be validated due to a lack of available measured data near the facilities of concern.

6.4.3 Effect of Terrain on Results of Local Scale Modeling

In the previous sections, mercury stack emissions from hypothetical model plants, characteristic of industrial source categories, were evaluated with the COMPDEP air dispersion and deposition model. These model plants were placed in simple terrain, and in either dry or humid climate conditions. In reality, many of these emission sources may actually be located in rolling topography, which may ultimately affect the predicted media concentrations near the facility. In this study EPA chose to ignore quantitatively the possible ambient air and deposition impacts posed by terrain. This was done to simplify the air modeling analysis in view of the wide range of uncertainty inherent in accurately modeling the deposition of the various mercury species. Moreover, EPA currently does not have refined air dispersion and deposition models capable of accurately modeling plume behavior in elevated terrain. The uncertainty lies in the predicted magnitude of the result. However, because elevated terrain can cause an increase in the predicted ambient air and deposition impacts in comparison to flat terrain, EPA has undertaken a limited modeling exercise to investigate to what extent consideration of elevated terrain may affect the air modeling results. The results of this exercise are summarized in this section.

In general, terrain refers to the height of a receptor with respect to a local source. However, it is actually the height of the receptor with respect to the effective stack height (stack height plus plume rise) that is important in the calculations. There are three types of terrain: simple, intermediate, and complex. Simple terrain consists of receptors located at the same elevation as the stack base. Complex terrain refers to receptors at or above the effective stack height, and intermediate terrain refers to receptors between these two extremes.

For calculating air concentrations and dry deposition rates, non-simple terrain is addressed by the COMPDEP model in a standard manner by reducing the effective stack height and, in stable conditions for receptors above stack top, by scaling the predicted values by a height-dependent correction factor. The first method reflects the fact that the receptor is closer to the center of the plume, thereby resulting in higher concentrations and rates. The specific amount that the effective stack height is reduced is determined based on the methods described in Briggs (1973) and Egan (1975), and are discussed in Appendix D. The effective stack height used is always at least half of the effective stack height calculated independent of terrain (see section D.2.1.6 of Appendix D). This results in a steady increase as a function of receptor height up to half of the effective stack height, after which the calculated value is essentially constant. It is not actually constant because in stable conditions (atmospheric stabilities E and F) for receptors located above the effective stack height the calculated values are scaled by an additional "correction factor" that further reduces the predicted values. Although most of the dry deposition occurs during unstable and neutral conditions, this additional reduction can result in the predicted deposition diminishing slightly for receptors that are above the effective stack height.

The general approach taken in this exercise was to compute the dry deposition as a function of receptor height above ground level. This was done for each facility at distances of 2.5 and 25 km from the source. Facilities from each source class were included because the effect of terrain on the predicted deposition rates can differ substantially depending on the source characteristics that affect the plume height.

Tables 6-40 and 6-41 show the effects of receptor height on the predicted dry deposition rate for all facilities at the two different distances from the source. These tables show the dimensionless ratio of the predicted value at a given height and the predicted value for a receptor at the same elevation as the stack base.

The magnitude of the ratios is different across facilities because the effective stack height depends on the source characteristics, and the distance between the receptor and the center of the plume depends on the calculated effective stack height. For facilities with lower stacks (e.g., waste incinerators, chlor-alkali plant), there is little change for the receptor heights considered because the receptors are so much higher than the effective stack height, and the slight diminishing effect discussed above can even be observed. At 2.5 km, these results show that the effect of receptor height can be substantial for receptors close to the source: there can be an increase of an order of magnitude in the predicted dry deposition rates. At 25 km, the difference as a function of receptor height is not as extreme because more dispersion has occurred: the vertical change in air concentrations is not as great as it is for closer receptors, thereby resulting in less deposition.

Table 6-40
Illustration of Effect of Receptor Height on Dry Deposition at a Distance of 2.5 km from the Source: Ratio of Predicted Value with Value for Receptor in Simple Terrain

	Receptor Height (m)					
Plant	25	50	75	100	125	150
Large MWC	1.4	2.1	4.4	6.6	9.9	11.1
Continuous MWI	2.2	2.2	2.0	2.0	1.9	1.8
Large Coal-fired Utility Boiler	1.1	1.2	1.4	1.7	2.0	2.5
Small Coal-fired Utility Boiler	1.4	2.2	3.6	6.7	9.0	9.0
Medium Oil-fired Utility Boiler	1.3	1.9	3.0	5.8	9.8	16.6
Chlor-alkali plant	1.0	1.0	1.0	0.9	0.9	0.8
Primary Lead Smelter	1.3	1.9	2.8	4.7	8.4	11.7

Table 6-41
Illustration of Effect of Receptor Height on Dry Deposition at a Distance of 25 km from the Source: Ratio of Predicted Value with Value for Receptor in Simple Terrain

	Receptor Height (m)					
Plant	25	50	75	100	125	150
Large MWC	1.2	1.4	2.6	2.8	2.9	2.6
Continuous MWI	1.7	1.3	1.0	1.0	0.9	0.9
Large Coal-fired Utility Boiler	1.1	1.2	1.3	1.5	1.7	2.0
Small Coal-fired Utility Boiler	1.2	1.4	1.6	2.7	2.8	2.3
Medium Oil-fired Utility Boiler	1.2	1.4	1.7	3.1	3.3	3.5
Chlor-alkali plant	1.1	1.0	1.0	1.0	0.9	0.9
Primary Lead Smelter	1.1	1.3	1.6	1.8	3.1	3.2

The predicted wet deposition rates (not shown) are not significantly impacted by change in receptor height. This is because the only height-dependent aspect of the method of wet deposition calculation is when the receptor is above the effective stack height, in which case sector-averaging is performed (see Appendix D).

The results of this exercise show that the deposition flux of divalent mercury in elevated terrain can increase up to 15 times over simple terrain at a receptor 2.5 km from the source, and up to 4 times at 25 km.

7. CONCLUSIONS

The following conclusions are presented in approximate order of degree of certainty in the conclusion, based on the quality of the underlying database. The conclusions progress from those with greater certainty to those with lesser certainty.

- The present study in conjunction with available scientific knowledge, supports a plausible link between mercury emissions from anthropogenic combustion and industrial sources and mercury concentrations in air, soil, water and sediments. The critical variables contributing to this linkage are these:

  a) the species of mercury that are emitted from the sources, with elemental mercury (Hg^0) mostly contributing to concentrations in ambient air and divalent mercury (Hg^{2+}) mostly contributing to concentrations in soil, water and sediments;

  b) the overall amount of mercury emitted from a combustion source; and

  c) the climate conditions.

- The present study, in conjunction with available scientific knowledge, supports a plausible link between mercury emissions from anthropogenic combustion and industrial sources and methylmercury concentrations in freshwater fish. The critical variables contributing to this linkage are the following:

  a) the species of mercury that are emitted, with emitted divalent mercury mostly depositing into local watershed areas and, to a lesser extent the atmospheric conversion of elemental mercury to divalent species which are deposited over greater distances;

  b) the overall amount of mercury emitted from a source;

  c) the extent of mercury methylation in the water body; and

  d) the climate conditions.

- There is a lack of adequate mercury measurement data near the anthropogenic atmospheric mercury sources considered in this report. The lack of such measured data preclude a comparison of the modeling results with measured data around these sources. These data include measured mercury deposition rates as well as measured concentrations in the atmosphere, soils, water bodies and biota.

- From the RELMAP analysis of mercury deposition and on a comparative basis, a facility located in a humid climate has a higher annual rate of mercury deposition than a facility located in an arid climate. The critical variables are the estimated washout ratios of elemental and divalent mercury, as well as the annual amount of precipitation. Precipitation removes various forms of mercury from the atmosphere and deposits mercury to the surface of the earth.

- On a national scale, an apportionment between sources of mercury and mercury in environmental media and biota cannot be described in quantitative terms with the current scientific understanding of the environmental fate and transport of this pollutant.

- Consumption of fish is the dominant pathway of exposure to methylmercury for fish-consuming humans and wildlife. There is a great deal of variability among individuals in these populations with respect to food sources and fish consumption rates. As a result, there is a great deal of variability in exposure to methylmercury in these populations. The anthropogenic contribution to the total amount of methylmercury in fish is, in part, the result of anthropogenic mercury releases from industrial and combustion sources increasing mercury body burdens in fish. As a consequence of human and wildlife consumption of the affected fish, there is an incremental increase in exposure to methylmercury.

- Due to differences in fish consumption rates per body weight and differences in body weights among species, it is likely that piscivorous birds and mammals have much higher environmental exposures to methylmercury than humans through the consumption of contaminated fish. This is true even in the case of fish consumption by humans who consume above average amounts of fish. The critical variables contributing to these outcomes are these:

  a) the fish consumption rate;

  b) the body weight of the individual in relation to the fish consumption rate; and

  c) the rate of biomagnification between trophic levels within the aquatic food-chain.

- The results of the assessment of current exposure of the U.S. population from fish consumption as described in Appendix H indicate that exposure to methylmercury from contaminated fish results in an incremental increase in most U.S. fish-consumers. Methylmercury exposure rates on a per body weight basis among fish-consuming children are predicted to be higher than for fish-consuming adults. The exposure rates among fish-consuming children under the age of 15 are estimated to average between 0.12 and 0.16 micrograms of methylmercury per kilogram of body weight per day. The exposure rates among fish-consuming adults are estimated to average between 0.07 and 0.08 micrograms of methylmercury per kilogram of body weight per day. Human adult fish consumption rates vary from 0 to greater than 300 grams per day.

- From the modeling analysis and a review of field measurement studies, it is concluded that mercury deposition appears to be ubiquitous across the continental U.S., and at, or above, detection limits when measured with current analytic methods.

- Based on the RELMAP modeling analysis and a review of recent measurement data published in peer-reviewed scientific literature, there is predicted to be a wide range of mercury deposition rates across the continental U.S. The highest predicted rates (i.e., above 90th percentile) are more than 50 times higher than the lowest predicted rates (i.e., below the 10th percentile). Three principal factors contribute to these modeled and observed deposition patterns:

  a) emission source locations;

  b) amount of divalent and particulate mercury emitted or formed in the atmosphere; and

  c) climate and meteorology.

- Based on the modeling analysis of the transport and deposition of stationary point source and area source air emissions of mercury from the continental U.S., it is concluded that the following geographical areas have the highest annual rate of deposition of mercury in all forms (above the levels predicted at the 90th percentile):

    a) The southern Great Lakes and Ohio River Valley.

    b) The Northeast and southern New England.

    c) Scattered areas in the South with the most elevated deposition occurring in the Miami and Tampa areas.

  Measured deposition estimates are limited, but are available for certain geographic regions. The data that are available corroborate the RELMAP modeling results for specific areas.

- Based on modeling analysis of the transport and deposition of stationary point source and area source air emissions of mercury from the continental U.S., it is concluded that the following geographical areas have the lowest annual rate of deposition of *mercury in all forms* (below the levels predicted at the 10th percentile).

    a) The less populated areas of the Great Basin, including southern Idaho, southeastern Oregon, most of southern and western Utah, most of Nevada, and portions of western New Mexico, and

    b) Western Texas other than near El Paso, and most of northeastern Montana.

- Based on limited monitoring data, the RELMAP model predictions of atmospheric mercury concentrations and wet deposition across the U.S. are comparable with typically measured data.

- EPA concludes that the selected major anthropogenic sources as modeled and parameterized for this assessment, can be ranked by predicted deposition rate at 2.5 Km in flat terrain, on a relative basis from high to low, as follows:

  Municipal waste combustors
  Chlor-alkali plants
  Lead smelters
  Copper smelters
  Medical waste incinerators
  Utility boilers

  The critical variables impacting the ranking are these:

  a) estimated amounts of divalent and particulate mercury emitted; and

  b) parameters that influence the plume height, primarily the stack height and stack exit gas velocity.

  This ranking may be sensitive to differences in the distance from the source (distances other than 2.5 Km) and the topography of the terrain.

- From the analysis of deposition and on a comparative basis, the deposition of divalent mercury close to an emission source is greater for receptors in elevated terrain (i.e., terrain above the elevation of the stack base) than from receptors located in flat terrain (i.e., terrain below the elevation of the stack base). The critical variables are parameters that influence the plume height, primarily the stack height and stack exit gas velocity.

- In terms of methylmercury intake on a per body weight basis, the five wildlife species considered in this analysis can be ranked from high to low as follows:

  -- Kingfisher
  -- River Otter
  -- Mink, Osprey
  -- Bald eagle

  Methylmercury exposures for the most exposed wildlife species (the kingfisher) may be up to two orders of magnitude higher than human exposures from contaminated freshwater fish (on a kilogram fish consumed per body weight basis). This assumes that the fish within different trophic levels of a given lake are contaminated with the same concentrations of methylmercury.

- Modeling estimates of the transport and deposition of stationary point source and area source air emissions of mercury from the continental U.S. have revealed the following partial mass balance.

  -- Of the total amount of elemental mercury vapor that is emitted, about 1 percent (1.2 metric tons/yr) may be atmospherically transformed into divalent mercury by tropospheric ozone and adsorbed to particulate soot in the air and subsequently deposited in rainfall and snowfall to the surface of the continental U.S. The vast majority of emitted elemental mercury does not readily deposit and is transported outside the U.S. or vertically diffused to the free atmosphere to become part of the global cycle.

  -- Nearly all of the elemental mercury vapor emitted from other sources around the globe also enters the global cycle and can be deposited slowly to the U.S. Nearly 30 times as much elemental mercury vapor is deposited from these other sources than from stationary point sources and area sources within the continental U.S.

  -- Of the total amount of divalent mercury vapor that is emitted, about 68 percent (62.6 metric tons/year deposits to the surface through wet or dry processes within the continental U.S. The remaining 32 percent is transported outside the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle.

  -- Of the total amount of particulate mercury that is emitted, about 36 percent (14.1 metric tons/year deposits to the surface through wet or dry processes within the continental U.S. The remaining 64 percent is transported outside the U.S. or is vertically diffused to the free atmosphere to become part of the global cycle.

- Assuming these deposition efficiencies are correct (namely; elemental mercury - 1%, divalent mercury vapor - 68%, and particulate mercury - 36%) the relative source contributions to the total anthropogenic mercury that is deposited to the continental U.S. are ranked as follows:

-- Medical waste incineration 36% (28 Megagrams (Mg) of 78 Mg)
-- Municipal waste combustion 31% (24 Mg of 78 Mg)
-- Coal-fired electric utility boilers 17% (13 Mg of 78 Mg)
-- Industrial and residential fossil fuel use 10% (8 Mg of 78 Mg)
-- Chlor-alkali factories 2% (1 Mg of 78 Mg)
-- Non-ferrous metal smelting 1% (1 Mg of 78 Mg)
-- Oil-fired electric utility boilers 1% (1 Mg of 78 Mg)

- Based on the local scale atmospheric modeling results in flat terrain, at least 75% of the emitted mercury from each facility is predicted to be transported more than 50 km from the facility.

- The models used in the exposure analysis indicate that, except for utility boilers and intermittent medical waste incinerators, deposition within 10 Km of a facility is generally dominated by emissions from the local source rather than from emissions transported from regional mercury emissions sources.

To improve the quantitative exposure assessment component of the risk assessment for mercury and mercury compounds, U.S. EPA would need more and better mercury emissions data and measured mercury data near sources of concern, as well as a better quantitative understanding of mercury chemistry in the emissions plume, the atmosphere, soils, water bodies and biota. Specific needs include these.

Mercury in the Atmosphere

- aqueous oxidation-reduction kinetics in atmospheric water droplets;
- physical adsorption and condensation of divalent mercury gas to ambient particulate matter
- photolytic reduction of particle-bound divalent mercury by sunlight
- convincing evidence that gas-phase oxidation of mercury is insignificant

Mercury in Soils and Water Bodies

- uptake and release kinetics of mercury from terrestrial plants
- biogeochemical mercury transport and transformation kinetics in benthic sediments
- methylation and demethylation kinetics in water bodies
- sorption coefficients to soils, suspended solids and benthic solids
- complexation to organic matter in water bodies

Information Leading to an Improved Quantitative Understanding of Aquatic Bioaccumulation Processes and Kinetics

- uptake kinetics by aquatic plants and phytoplankton

- partitioning and binding behavior of mercury species within organisms

- metabolic transformations of mercury, and the effect on uptake, internal distribution, and excretion

Information that will facilitate the development of a dynamic, linked terrestrial-aquatic mass balance modeling framework that includes realistic mercury chemistry and the aquatic food web as an integral component

- More measurements of methylmercury concentrations in fish for better identification of the range in fish species,

- Surveys of fish consumption among potential high-end fish consumers which examine specific biomarkers indicating mercury exposure (e.g., blood mercury concentrations and hair mercury concentrations), and

- A pharmacokinetic-based understanding of mercury partitioning in humans and wildlife particularly the interactions of different forms of mercury and different uptake routes.

8. RESEARCH NEEDS

During the development of the mercury exposure assessment, many areas of uncertainty and significant data gaps were identified. Many of these have been identified in the document, and several are presented in the following list.

1. Improved analytical techniques for measuring speciated mercury air emissions as well as total mercury emissions from major point sources. Laboratory evidence suggests that divalent mercury gas emissions will wet and dry deposit much more readily than elemental mercury gas. Particle-bound mercury is also likely to deposit relatively quickly. Current stack sampling methods do not provide sound information about the fraction of mercury emissions that are in oxidized form. While filters are used to determine particulate mercury fractions, high temperature stack samples may not be indicative of the fraction of mercury that is bound to particles after dilution and cooling in the first few seconds after emission to the atmosphere. Methods for determination of the chemical and physical forms of mercury air emissions after dilution and cooling need to be developed and used to characterize all known major point sources.

2. Evaluated Local and Regional Atmospheric Fate and Transport Models are needed. These models should treat all important chemical and physical transformations which take place in the atmosphere. The development of these models will require comprehensive field investigations to determine the important atmospheric transformation pathways (e.g., aqueous cloud chemistry, gas-phase chemistry, particle attachment, photolytic reduction) for various climatic regions. The evaluation of these models will require long-term national (possibly international) monitoring networks to quantify the actual air concentrations and surface deposition rates for the various chemical and physical forms of mercury.

3. Better understanding of mercury transport from watershed to water body including the soil chemistry of mercury, the temporal aspects of the soil equilibrium, the impact of low levels of volatile mercury species in surface soils and water bodies on total mercury concentrations and equilibrium.

4. Better understanding of foliar uptake of mercury and plant/mercury chemistry. (The most important questions: do plants convert elemental or divalent mercury into forms of mercury that are more readily bioaccumulated? Do plants then emit these different forms to the air?) A better understanding of the condensation point for mercury is needed.

5. Better understanding of mercury movement from plant into soil (detritus). May need to refine the models used to account for movement of mercury in leaf litter to soil.

6. The impact of anthropogenic mercury on the "natural," existing mercury levels and species formed in soil, water, and sediments needs better understanding. How does the addition of anthropogenic mercury affect "natural" soil and water mercury cycles? Natural emission sources need to be studied better and their impacts better evaluated.

7. Improved understanding of mercury flux in water bodies and impact of plant and animal biomass are needed. Unlike many other pollutants, most of the methylmercury in a water body appears to be in the biological compartment. The sedimentation rate as well as benthic sediment:water partition coefficient require field evaluation. Important to consider rivers and other larger water bodies in these flux analyses.

8. The BAF contains a substantial level of uncertainty. A more appropriate BAF can probably be developed when the data base upon which the estimate is based is enlarged; i.e., need data from more than four studies. The availability of more data would enable the possible development of lake-type adjustment factors for the mercury BAF possibly based on color, acidification susceptibility, etc., or species-specific BAF adjustment factors for freshwater species most commonly consumed. Also need a time analysis of fish mercury uptake which could lead to the development of a dynamic fish model. A mercury BAF for saltwater fish is needed.

9. Better estimates of fish consumption rates for high-end consumers (subsistence) as well as recreational anglers are needed. Fish species-specific consumption rates are also needed.

10. Need to improve the biotransfer factors for mercury from soil and plants to beef.

9. REFERENCES - *SEE ATTACHED FOR ADDITIONAL REFERENCES

Akagi H., D.C. Mortimer, and D.R. Miller (1979). Mercury Methylation and Partition in Aquatic Systems. *Bull. Environ. Contam. Toxicol.* 23:372-376.

Agency for Toxic Substances and Disease Registry (1989). Toxicological Profile for Mercury.

American Conference of Governmental and Industrial Hygienists (1986). TLVs: threshold limit values and biological exposure indices for 1986-1987. Cincinnatti, OH: American Conference of Governmental and Industrial Hygienists, Inc.; pp. 253-256.

Anderson, W. L. and K.E. Smith (1977). Dynamic of mercury at coal-fired utility power plant and adjacent cooling lake. *Environ. Sci and Technol.* 11:75.

Anderson, Sue, Life Sciences Research Office *

Bache, E., W. Gutenmann, M. Rutzke, G. Chu, D. Elfving and D. Lisk (1991). Concentrations of Metals in Grasses in the Vicinity of a Municipal Refuse Incinerator. *Arch. Environ. Contam. Toxicol.* 20:538-542.

Baes, C., R. Sharp, A. Sjoreen and R. Shor (1984). A Review and Analysis of parameters for assessing transport of environmentally released radionuclides through agriculture. Prepared under contract No. DE-AC05-84OR21400. U.S. Department of Energy, Washington, D.C.

Bahnick, D., C.Sauer, B. Butterworth and D. Kuehl (1994). A National Study of Mercury Contamination of Fish. *Chemosphere* 29(3):537-546.

Barregard, L., G. Sallsten, A Schutz, R. Attewell, S. Skerfving, and B. Jarvholm (1992). Kinetics of Mercury in Blood and Urine after Brief Occupational Exposure. *Arch. Environ. Health* 47(3):176-184.

Barregard, L., B. Hogstedt, A Schutz, A. Karlsson, G. Sallsten, G. Thiringer (1991). Effects of Occupational Exposure to Mercury Vapor on Lymphocyte Micronuclei. *Scand. J. Work Environ. Health* 17(4):263-268.

Begerow, J., D. Zander, I. Freier, and L. Dunemann (1994). Long-Term Mercury Excretion in Urine after removal of Amalgam Fillings. *Arch. Occup. Environ. Health* 66:209-212.

Benoit, J.M., W.F. Fitzgerald and A.W.H. Damman. 1994. Historical Atmospheric Mercury Deposition in the Mid-Continental U.S. as Recorded in an Ombrotrophic Peat Bog. Pp. 187-202 in Watras, C.J. and J.W. Huckabee eds. Mercury Pollution Integration and Synthesis.

Bloom, N. and W. F. Fitzgerald (1988). Determination of Volatile Mercury Species at the Picogram level by Low-Temperature Gas Chromatography with Cold-Vapor Atomic Fluorescence Detection. *Analytica Chimica Acta*, 208:151-161.

Bloom, N. S. (1989). Determination of picogram levels of methyl-mercury by aqueous phase ethylation, followed by cryogenic gas chromatography with cold-vapor atomic fluorescence detection. *Can. J. Fisher. Aq. Sci.* 46:1131-1140.

Bloom, N. S. and C. J. Watras (1989). Observations of Methylmercury in Precipitation. *The Sci. Tot. Environ.* 87/88:191-207.

Bloom, N.S., C.J. Watras, and J.P. Hurley (1991). Impact of Acidification on the Methylmercury Cycle of Remote Seepage Lakes. *Water, Air, and Soil Poll.* 56: 477-491.

Bloom, N. S., C. J. Watras, and J. P. Hurley (1991). Impact of Acidification on the Methylmercury Cycle of Remote Seepage Lakes. *Water, Air and Soil Poll.* 56:477-491.

Bloom, N. S. (1992). On the Chemical Form of Mercury in Edible Fish and Marine Invertebrate Tissue. *Can. J. Fisher. Aq. Sci.* 49:1010-1017.

Bloom, N.S. and E. Kuhn (1994). Mercury speciation in meat products, personal communication, October 1, 1994.

Bowers, J.R., J.R. Bjorkland and C.S. Cheney, 1979: Industrial Source Complex (ISC) Dispersion Model User's Guide. Volume I, EPA-450/4-79-030, U.S. Environmental Protection Agency,Research Triangle Park, North Carolina 27711.

Briggs, G.A. (1969). *Plume Rise*, AEC Critical Review Series, TID - 25075, National Technical Information Service, Springfield, VA., 81pp.

Briggs, G.A. (1972). Discussion on Chimney Plumes in Neutral and Stable Surroundings. *Atmos. Environ.* 6:507-510.

Briggs, G.A. (1973). *Diffusion Estimation for Small Emissions*, ATDL Contribution File No. 79. Atmospheric Turbulence and Diffusion Laboratory.

Briggs, G.A. (1973). *Diffusion Estimates for Small Emissions*, Atmospheric Turbulence and Diffusion Laboratory, Contribution No. 70 (Draft), Oak Ridge, Tennessee.

Briggs, G.A. (1975). Plume Rise Predications, in *Lectures on Air Pollution and Environmental Impact Analysis*, Americal Meteorological Society, Boston, Massachusetts.

Brosset, C. (1981). The Mercury Cycle. *Water, Air and Soil Poll.* 16:253-255.

Brosset, C. and E. Lord (1991). Mercury in Precipitation and Ambient Air: A new Scenario. *Water, Air and Soil Poll.* 56:493-506.

Burke et al., 1994 and 1995 appears in chapters 2, 5 and 9 both citations should read Burke et al., (*in press*). In Chapter 9 keep the reference as is just replace 1995 with *in press*.

Buchet J. P., R. Lauwerys, A. Vandevoorde, and J. M. Pycke (1983). Oral Daily Intake of Cadmium, Lead, Manganese, Chromium, Mercury, Calcium, Zinc and Arsenic in Belgium. A Duplicate Metal Study. *Food Chem. Tox.* 21:19-24.

Burke, J., M. Hoyer, G. Keeler and T. Scherbatskoy. 1995. Wet Deposition of Mercury and Ambient Mercury Concentrations as a Site in the Lake Champlain Basin. Accepted for Publication in *Water, Air and Soil Pollution.*

Campbell, D., M. Gonzales, and J. B. Sullivan (1992). Mercury. in Hazardous Materials Toxicology. Clinical Principles of Environmental Health. J. B. Sullivan and G. R. Krieger, Eds. Williams and Wilkins, Baltimore, MD. p. 824-833.

Cappon, C.J. (1981). Mercury and Selenium Content and Chemical Form in Vegetable Crops Grown on Sludge-Amended Soil. *Arch. Environm. Contam. Toxicol.* 10: 673-689.

Cappon, C. J. (1987). Uptake and Speciation of Mercury and Selenium in Vegetable Crops Grown on Compost-Treated Soil. *Water, Air and Soil Poll.* 34:353-361.

Carpi, A., L. Weinstein and D. Ditz (1994). Bioaccumulation of Mercury by Sphagnum Moss near a Municipal Solid Waste Incinerator. *Air and Waste* 44:669-672.

CARB (1986). Subroutines for calculating dry deposition velocities using Sehmel's curves. Prepared by Bart Croes, California Air Resources Board.

Cardenas, A., H. Roels, A. M. Bernard, R. Barbon, J. P. Buchet, R. R. Lauwerys, J. Rosello, G. Hotter, A. Mutti, I. Franchini, L. M. Fels, H. Stolte, M. E. De-Broe, G. D. Nuyts, S. A. Taylor, and R. G. Price (1993). Markers of Early Renal Changes Induced by Industrial Pollutants. I. Applications to Workers Exposed to Mercury Vapour. *Brit. J. Indus. Med.* 50(1):17-27.

Catalano, J.A., D.B. Turner, and J.H. Novak (1987). *User's Guide for RAM -- Second Edition.* EPA/600/8-87/046, U.S. Environmental Protection Agency, Research Triangle Park, North Carolina 27711.

Chapman and Hall (1984). *Dictionary of Organometallic Compounds.* Vol. 1, p. 1030.

Cleckner, L. B., E. S. Esseks, P. G. Meier, and G. J. Keeler. 1995. Mercury Concentrations in Two "Great Waters". Accepted for Publication in *Water, Air and Soil Pollution.*

Clemente, G. F., G. Ingrao, and G. P. Santaroni (1982). The Concentration of Some Trace Elements in Human Milk form Italy. *The Sci. Tot. Environ.* 24:255-265.

Code of Federal Regulations (1989). Air contaminants - permissible exposure limits. C.F.R. 29:1910:1000.

Columbia River Iner-Trival Fish Commission (1994). A Fish Consumption Survey of the Umtilla, Nez Perce, Yakama and Warm Springs Tribes of the Columbia River Basin. Technical Report 94-3. October, 1994.

CRC Handbook of Chemistry and Physics. 69th ed. 1988.

Cramer, G.M. (1992), Interoffice memorandum, April 21, 1992.

Cramer, G. M. (1994). Exposure of U. S. Consumers to Methylmercury from Fish. Presented at the DOE/FDA/EPA Workshop on Methylmercury and Human Health, Bethesda, MD March 22-23 1994.

Cramer, H.E. 1957; A Practical Method for Estimating the Dispersal of Atmospheric Contaminants, in *Proceedings of the First National Conference on Applied Meteorology*, Sec. C, pp. C-33-C-35, American Meteorological Society, Hartford, Conn.

Crispin-Smith, J., Turner, M.D., March D.O. Project III. Hair methylmercury levels in women of childbearing age.

Crockett, A. and R. Kinnison (1979). Mercury residues in soil around a coal-fired power-plant. *Envir. Sci. Technol.* 13:712-715.

Cunningham, P. S. Smith, J. Tippett and A. Greene. 1994. A National Fish Consumption Advisory Data Base: A Step Toward Consistency. Fisheries. 19(5):14-23.

Dangwal, S. K. (1993). Evaluation and Control of Mercury Vapor Exposure in the Cell House of Chlor - Alkali Plants. *Environ. Res.* 60(2):254-258.

Dooley, J. H. (1992). Natural Sources of Mercury in the Kirkwood-Cohansey Aquifer System of the New Jersey Coastal Plain. New Jersey Geological Survey, Report 27.

Driscoll, C. T., C. Yan, C. L. Schofield, R. Munson, and J. Holsapple (1994). The Mercury Cycle and Fish in the Adirondack Lakes. ES+T.

Dvonch et al., 1994 and 1995 appears in chapters 2, 5 and 9 both citations should read Dvonch et al., (*in press*). In Chapter 9 keep the reference as is just replace 1995 with *in press*.

Dvonch, J.T., A.F. Vette, G.J. Keeler, G. Evans and R. Stevens. 1995. An intensive Multi-site Pilot Study Investigating Atmospheric Mercury in Broward County, Florida. Accepted for Publication in Water, Air and Soil Pollution.

Eder, B. K., D. H. Coventry, T. L. Clark, and C. E. Bollinger, 1986. RELMAP: A regional Lagrangian model of air pollution - users guide. Project Report, EPA/600/8-86/013, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Egan, B.A. (1975). Turbulent Diffusion in Complex Terrain, in *Lectures on Air Pollution and Environmental Impacts Analysis*, pp.112-135, D. Haugen (Ed.), American Meteorological Society, Boston, Mass.

Expert Panel on Mercury Atmospheric Processes (1994). *Mercury Atmospheric Processes: A Synthesis Report.* Report No. TR-104214.

Ehrenberg, R. L., R. L. Vogt, A. B. Smith, J. Brondum, W. S. Brightwell, P. J. Hudson, K. P. McManus, W. H. Hannon, and F. C. Phipps (1991). Effects of Elemental Mercury Exposure at a Thermometer Plant. *Amer. J. Indus. Med.* 19(4):495-507.

Ellingsen, D. G., T. Moreland, A. Andersen, and H. Kjuus (1993). Relation between Exposure Related Indices and Neurological and Neurophysiological Effects in Workers Previously Exposed to Mercury Vapour. *Brit. J. Indus. Med.* 50(8):736-744.

Engelmann, R.J. (1968). The Calculation of Precipitation Scavenging, in *Meteorology and Atomic Energy 1968*, D.H. Slade, editor. U.S. Atomic Energy Commission.

Engstrom D.R., E.B. Swain, T.A. Henning, M.E. Brigham and P.L. Brezonick. 1994. Atmospheric Mercury Deposition to Lakes and Watersheds: A Quantitative Reconstruction from Multiple Sediment Cores. Pp. 33-66 in L.A. Baker (ed). Environmental Chemistry of Lakes and Reservoirs. American Chemical Society.

Evans, R.D. (1986). Sources of mercury contamination in the sediments of small headwater lakes in south-central Ontario, Canada. Arch. Environ. Contam. Toxicol. 15:505-512.

Fay, J.M., M. Escudier, and D.P. Hoult (1969). *A Correlation of Field Observations of Plume Rise.* Fluid Mechanics Laboratory Publication No. 69-4, Massachusetts Institute of Technology.

FDA Compliance Testing as described in the NMFS data base.

Fitzgerald, W. F. and T. W. Clarkson (1991). Mercury and Monomethylmercury: Present and Future Concerns. *Envir. Health Perspec.* 96:159-166.

Fitzgerald, W. F., R. P. Mason and G. M. Vandal (1991). Atmospheric Cycling and Air-Water Exchange of Mercury over Mid-Continental Lacustrine Regions. *Water, Air and Soil Poll.* 56:745-767.

Fitzgerald, W. F. (1994). Global Biogeochemical Cycling of Mercury. Presented at the DOE/FDA/EPA Workshop on Methylmercury and Human Health, Bethesda, MD March 22-23 1994.

Fitzgerald, E.G., S. Hwang, K. Brix, B. Bush, K. Cook, and P. Worsick. 1995. Fish PCB Concentrations and Consumption Patterns Among Mohawk Women at Akswsanse. J. Expsoure Analysis and Environmental Epidemiology: 5(1):1-19.

Fiore, B.J., H.A. Anderson, L.P. Hanrahan, L.J. Olson and W.C. Sonzogni. 1989. Sport Fish Consumption and Body Burden Levels of Chlorinated Hydrocarbons: A Study of Wisconsin Anglers. Arch. Environ. Health. 44(2):82-88.

Florida Department of Environmental Regulation (1990). Mercury, Largemouth Bass and Water Quality: A Preliminary Report.

Foley, R. E., S. L. Jackling, R. L. Sloan, and M. K. Brown (1988). Organochlorine and Mercury Residues in Wild Mink and Otter: Comparison with Fish. *Environ. Toxicol. Chem.* 7:363-374.

Fortmann, L. C., D. D. Gay, and K. O. Wirtz (1978). Ethylmercury: Formation in Plant Tissues and Relation to Methylmercury Formation. U.S. EPA Ecological Research Series, EPA-600/3-78-037.

Fouassin, A. and M. Fondu (1978). Evaluation de la teneur moyenne en mercure de la ration alimentaire en Belgique. *Arch. Belg. Med. Soc. Hyg. Med Trav. Med Leg.* 36:481-490.

Fujita, M. and E. Takabatake (1977). Mercury Levels in Human Maternal and Neonatal Blood, Hair and Milk. *Bull. Environ. Contam. Toxicol.* 18:205-209.

Fukuzaki, N. R. Tamuro, Y. Hirano, and Y. Mizushima (1986). Mercury Emission from a Cement Factory and its Influence on the Environment. *Atmos. Environ.* 20(12):2291-2299.

Gerstsenberger, S., J. Pratt-Shelley, M. Beattie and J. Dellinger. 1993. Mercury Concentrations of Walleye (Stizostedion vitreum vitreum) in 34 Northern Wisconsin Lakes. Bull. Environ. Contam. Toxicol. 50:612-617.

Geurtsen W. (1990). Amalgam in der Diskussion - Zur frage der Amalgamtoxizitat. *Phillip J.* 3:121-128.

Giesy, J., D. Verbrugge, R. Othout, W. Bowerman, M. Mora, et. al., (1994). Contaminants in Fishes from Great Lakes-influenced Sections and above dams of three Michigan Rivers. I. Concentrations of organochlorine insecticides, polychlorinated biphenyls, dioxin equivalents and mercury. Arch. Environ. Contam. Toxicol. 27:202-212.

Gifford, F.A. (1976). Turbulent Diffusion Typing Schemes - A Review, *Nucl. Saf.*, 17:68-86.

Glass, G.E., E.N. Leonard, W.H. Chan and D.B. Orr. 1986. Airborne mercury in precipitation in the Lake Superior Region. J. Great Lakes Res. 12(1) 37-51.

Glass, G.E., J.A. Sorensen, K.W. Schmidt and G.R. Rapp. 1990. New Source Identification of Mercury Contamination in the Great Lakes. Environ. Sci. Technol. 24 (7):1059-1069.

Glass, G., J. Sorensen, K. Schmidt, G. Rapp, D. Yap, and D. Fraser. 1991. Mercury Deposition and sources for the Upper Great Lakes Region. *Water, Air and Soil Pollution.* 56:235-249.

Glass, G.E., J.A. Sorensen, K.W. Schmidt, G.R. Rapp, D. Yap and D. Fraser. 1992. Mercury Sources and Distribution in Minnesota's Aquatic Resources: Deposition. Part 1 of Chapter 4 (Mercury Washout from Precipitation: Atmospheric Sources) in Mercury in the St. Louis River, Mississippi River, Crane Lake and Sand Point Lake: Cycling, Distribution and Sources. Report to the Legislative Commission on Minnesota Resources. April, 1992. Water Quality Division Minnesota Pollution Control Agency St. Paul, MN.

Gloss, S. P., T. M. Grieb, C. T. Driscoll, C. L. Scholfield, J. P. Baker, D. H. Landers, and D. B. Porcella (1990). Mercury levels in fish from the Upper Peninsula of Michigan (ELS Subregion 2B) in Relation to Lake Acidity. USEPA Corvallis Env. Res. Lab. Corvallis.

Gmelins Handbuch, Der Anorganischen Chemie (34), Queck Silber, 1967 Verlag Chemie. GMBH Weinheim/Burger.

Gonzalez-Fernandez, E., J. Mena, M. Diaz-Gonzalez, and J. M. Martinez-Gil-De-Arana (1984). A Long-Term Study of Environmental, Blood and Urine Mercury Levels and Clinical Findings in Workers Manufacturing Mercury Relays. *Indus. Health* 22(2):97-106.

Greenberg, A., I Wojtenko, H. Chen, S. Krivanek, J. Butler, J. Held, P. Weis and N. Reiss (1992). Mercury in Air and Rainwater in the Vicinity of a Municipal Resource Recovery Facility in Western New Jersey. Presented at International Symposium on Measurement of Toxic and Related Pollutants, Durham, NC. May 8.

Grieb, T., C. Driscoll, S. Gloss, C. Schofield, G. Bowie and D. Porcella. 1990. Factors Affecting Mercury Accumulation in Fish in the Upper Michigan Peninsula. *Environ. Tox. Chem.* 9:919-930.

Hakanson, A. T. Andersson and A. Nilsson. 1990. Mercury in Fish in Swedish Lakes-Linkages to Domestic and European Sources. *Water, Air and Soil Poll.* 50:171-191.

Hakanson, L. A. Nilsson and T. Anderson. 1988. Mercury in Fish in Swedish Lakes. *Environmental Pollution.* 49:145-162.

Halbach, S. (1985). The Octanal/Water distribution of Mercury Compounds. *Arch. Toxicol.* 57:139-141.

Halbrook, R. S., J. H. Jenkins, P. B. Bush, and N. D. Seabolt (1994). Sublethal Concentrations of Mercury in River Otters: Monitoring Environmental Contamination. *Arch. Environ. Contam. Toxicol.* 27:306-310.

Hall, R. A., E. G. Zook, and G. M. Meaburn (1978). National Marine Fisheries Survey of Trace Elements in the Fishery Resource, NOAA Technical Report NMFS SSRF-721. U.S. Department of Commerce, Washington, DC.

Hanna, S.R., G.A. Briggs, and R.P. Hosker, Jr. (1982). *Handbook on Atmospheric Diffusion*, DOE/TIC-11223.

Hanson, P.J., S.E. Lindberg, K.H. Kim, J.G. Owens and T.A. Tabberer (1994). Air/Surface Exchange of Mercury Vapor in the Forest Canopy I. Laboratory Studies of Foliar Hg Vapor Exchange. 3rd International Conference on Mercury as a Global Pollutant. Whistler, BC, Canada (July 10-14, 1994)

Hatch, W. and W. Ott. (1968). Determination of submicrogram quantities of mercury by atomic absorption spectrophotometry. Anal. Chem. 40:2085 - 2087.

Hicks, B.B., D.D. Baldocchi, T.P Meyers, R.P. Hosker, Jr., and D.R. Matt. (1987). A preliminary multiple resistance routine for deriving dry deposition velocities from measured quantities. Water, Air, and Soil Pollution 36: 311-330.

Holzworth, G.C. (1972). Mixing Heights, Wind Speeds and Potential for Urban Air Pollution Throughout the Contiguous United States. Publication No. AP-101, U.S. Environmental Protection Agency, Research Triangle Park, North Carolina 27711.

Horvat, M., N. S. Bloom, L. Liang, (1993a). Comparison of distillation with other current isolation methods for the determination of methylmercury compounds in low level environmental samples. Part I: Sediments. *Analytica Chimica Acta*, 281:135-152.

Horvat, M., L. Liang, N. S. Bloom (1993b). Comparison of distillation with other current isolation methods for the determination of methylmercury compounds in low level environmental samples. Part II: Water. *Analytica Chimica Acta*, 282:153-168.

Hoyer, M., J. Burke and G. Keeler. 1995. Atmospheric Sources, Transport and Deposition of Mercury in Michigan: Two Years of Event Precipitation. Accepted for Publication in *Water, Air and Soil Pollution.*

Hubert, A.H. and W.H. Snyder (1976). Building Wake Effects on Short Stack Effluents, in *Third Symposium on Atmospheric Turbulence, Diffusion, and Air Quality*, Raleigh, NC, Oct. 19-22, pp. 235-242, American Meteorological Society, Bostaon, Mass.

Huber, A.H. (1977). Incorporating Building/Terrain Wake Effects on Stack Effluents, in *Preprints of Joint Conference on Applications of Air Pollution Meteorology*, Salt Lake City, Nov. 29-Dec. 2, ppp. 353-356, American Meteorological Society, Boston, Mass.

Humphrey, H. E. B. (1983). in PCB's: Human and Environmental Hazards D'Itri and Kamrin, eds. Ann Arbor Science Publications, Ann Arbor, MI.

International Atomic Energy Agency (IAEA). *Evaluating the Reliability of Predictions Made Using Environmental Transfer Models.* IAEA Safety Series 100. Vienna, Austria.

Irwin, J. 1979. Estimating Plume Dispersion: A Recommended Generalized Scheme, in *Proceedings of the Fourth Symposium on Turbulence, Diffusion, and Air Pollution,* Jan. 15-18, 1979, Reno, Nev., pp. 62-69. American Meteorological Society, Boston, Mass.

Iverfeldt, Å. and I. Persson (1985). The Solvation Thermodynamics of Methylmercury(II) Species Derived from Measurements of the Heat of Solution and the Henry's Law Constant. *Inorganica Chimica Acta,* 103:113-119.

Iverfeldt, Å., 1991. Occupance and turnover of atmospheric mercury over the nordic countries. *Water, Air and Soil Pollution* 56:251-265.

Jensen, A. and A. Jensen. 1991. Historical Deposition Rates of Mercury in Scandinavia Estimated by Dating and Measurement in Cores of Peat Bogs. Water, Air and Soil Poll. 56:769-777.

Johansson, K., M. Aastrup, A. Andersson, L. Bringmark and A Iverfeldt. 1991. Mercury in Swedish Forest Soils and Waters-Assessment of Critical Load. Water, Air and Soil Pollution.56:267-281.

John, M.K. (1972). Mercury Uptake from Soil by Various Plant Species. *Bull. Environ. Contam. Toxicol.* 8(2): 77-80.

Jernelov, A. and T. Wallin (1973). Air-borne Mercury Fallout on Snow around five Swedish Chlor-alkali Plants. *Atmos. Environ.* 7:209-214.

Keeler, G., M. Hoyer, and C. Lamborg. 1994. Measurements of Atmospheric Mercury in the Great Lakes Basin. Pp. 231-241 in Watras, C.J. and J.W. Huckabee eds. Mercury Pollution Integration and Synthesis.

Keeler, G., G. Glinsorn and N. Pirrone. 1995. Particulate Mercury in the Atmosphere: Its Significance, Transport, Transformation and Sources. Accepted for Publication in *Water, Air and Soil Pollution.*

Kishi, R., R. Doi, Y. Fukuchi, H. Satoh, T. Satoh, A. Ono, F. Moriwaka, K. Tashiro, N. Takahata, H. Sasatani, H. Shriakashi, T. Kamada, and K. Nakagawa (1994). Residual Neurobehavioural Effects Associated with Chronic Exposure to Mercury Vapor. *Occup. Environ. Med.* 51(1):35-41.

Kishi, R., R. Doi, Y. Fukuchi, H. Satoh, T. Satoh, A. Ono, F. Moriwaka, K. Tashiro, N. Takahata (1993). Subjective Symptoms and Neurobehavioural Performances of Ex-Mercury Miners at an average of 18 Years after the Cessation of Chronic Exposure to Mercury Vapor. *Environ. Res.* 62(2):289-302.

Kosta, L., A. R. Byrne, and M. Dermelj (1983). Trace elements in some human milk samples by radiochemical neutron activation analysis. *The Sci. Tot. Environ.* 29:261-268.

Kudo, A., H.Nagase, and Y. Ose. 1982. Proportion of Methylmercury to the Total Mercury in river waters of Canada and Japan. Water Res. 16:1011-1015.

Lambourg, C., W. Fitzgerald, G. Vandal, and K. Rolfhus. Atmospheric Mercury in Northern Wisconsin: Sources and Species. (In press).

Lambourg, C.H., M.E. Hoyer, G.J. Keeler, I. Olmez and X. Huang. 1994. Particulate-Phase Mercury in the Atmosphere: Collection/Analysis Method Development and Applications. Pp. 251-259 in Watras, C.J. and J.W. Huckabee eds. Mercury Pollution Integration and Synthesis.

Lange, T. R., H. R. Royals, and L. L. Conner (1993). Influence of Water Chemistry on Mercury Concentrations in Largemouth Bass from Florida Lakes. *Trans. Amer. Fish. Soc.* 122:74-84.

Langsworth, S., C. G. Elinder, C. J. Gothe, and O. Vesterburg (1991). Biological Monitoring of Environmental and Occupational Exposure to Mercury. *Arch. Occup. Environ. Health* 63:161-167.

Lathrop, R. C., K. C. Noonan, P. M. Guenther, T. L. Grasino, and P. W. Rasmussen (1989). Mercury Levels in Walleyes form Wisconsin Lakes of different Water and Sediment Chemistry Characteristics. Tech. Bull. No. 163. DNR, State of Wisconsin, Madison.

Lenka, M., K. K. Panda, and B. B. Panda (1992) Monitoring and Assessment of Mercury Pollution in the Vicinity of a Chloralkali Plant. IV. Bioconcentration of Mercury in *In Situ* Aquatic and Terrestrial Plants at Ganjam, India. *Arch. Environ. Contam. Toxicol.* 22:195-202.

Lee, Y. and A. Iverfeldt (1991). Measurement of Methylmercury and Mercury in Run-off, Lake and Rain Waters. *Water, Air and Soil Poll.* 56:309-321.

Lindberg, S. E., D. R. Jackson, J. W. Huckabee, S. A. Janzen, M. J. Levin, and J. R. Lund (1979). Atmospheric Emission and Plant Uptake of Mercury from Agricultural Soils near the Almaden Mercury Mine. *J. Environ. Qual.* 8(4):572-578.

Lindberg, S. E., R. R. Turner, T. P. Meyers, G. E. Taylor, and W. H. Schroeder (1991). Atmospheric Concentrations and Deposition of Hg to a Deciduous Forest at Walker Branch Watershed, Tennessee, USA. *Water, Air and Soil Poll.* 56:577-594.

Lindberg, S. E., T. P. Meyers, G. E. Taylor, R. R. Turner, and W. H. Schroeder (1992). Atmosphere-Surface Exchange of Mercury to a Forest: Results of Modelling and Gradient Approaches. *J. of Geophy. Res.* 97(D2):2519-2528.

Lindqvist, O. and H. Rodhe (1985). Atmospheric Mercury-a review. *Tellus.* 37B:136-159.

Lindqvist, O., K. Johansson, M. Aastrup, A. Andersson, L. Bringmark, G. Hovsenius, L. Hakanson, A. Iverfeldt, M. Meili, and B. Timm (1991). Mercury in the Swedish Environment - Recent Research on Causes, Consequences and Corrective Methods. *Water, Air and Soil Poll.* 55:(all chapters)

Lipfert, F.W., P.D. Moskowitz, V.M. Fthenakis, M.P. DePhillips, J. Viren and L. Saroff. 1994. Assessment of Mercury Health Risks to Adults from Coal Combustion. Brookhaven National Laboratory. Upton, NY.

Lowe TP, May TW, Brumbaugh WG, and Kane DA (1985) National Contaminant Biomonitoring Program: Concentrations of seven elements in fresh-water fish, 1978-1981. *Arch. Environ. Contamin. Toxicol.* 14: 363-388.

Mason, R.P., Fitzgerald, W.F., and Morel, M.M., 1994. The Biogeochemical Cycling of Elemental Mercury: Anthropogenic Influences. Geochem. Cosmocheim. Acta, (in press).

MacCrimmon, H. R., C. D. Wren, and B. L. Gots (1983). Mercury Uptake by Lake Trout, *Salveiinus namaycush*, relative to age, growth and diet in Tadenac Lake with comparative data from other Precambrian Sheild lakes. *Can. J. Fisher. Aq. Sci.* 40:114-120.

McKeown-Eyssen, G.E., J. Ruedy, and A. Neims (1983). Methylmercury Exposure in Northern Quebec. II. Neurologic Findings in Children. *Amer. J. Epidem.* 118:470-479.

Market Research Corporation of America (1988). 14 Day Survey (5 Year Census, 1982-1987) Northbrook, IL.

Marsh, D.O., T.M. Clarkson, C. Cox, G.J. Myers, L. AminZaki, and S. Al-Tikriti (1987). Fetal Methylmercury Poisoning: Relationship Between Concentration in Single Strands of Maternal Hair and Child Effects. *Arch. Neurol.* 44:1017-1022.

Mason, R.P., W.F. Fitzgerald and F.M.M. Morel. 1994. The Biogeochemical Cycling of Elemental Mercury: Anthropogenic Influences. *Geochimica et Cosmochimicia Acta.* 58(15):3191-3198.

Meger, (1989) found that the mercury levels in sediment core samples from 2 Minnesota lakes had increased significantly since 1880.

Meili, M., A. Iverfeldt and L. Hakanson (1991)., Mercury in the Surface Water of Swedish Forest Lakes - Concentrations, Speciation and Controlling Factors, *Water, Air, and Soil Pollution* 56: 439-453.

Michigan Environmental Science Board (1993). Mercury in Michigan's Environment: Environmental and Human Health Concerns. Report to Gov. John Engler.

Mierle, G. and R. Ingram (1991). The Role of Humic Substances in the Mobilization of Mercury from Watersheds. *Water, Air and Soil Poll. 56*:349-357.

Mills, E.L., W.H. Gutenmann, and D.J. Lisk. 1994. Mercury content of small pan fish from New York State Waters, *Chemosphere*, Vol. 29, No. 6, pp. 1357-1359.

Mitra, S. (1986). Mercury in the Ecosystem. Trans Tech Publications Ltd. Switzerland.

MMES (Martin Marietta Energy Systems) material safety data sheets, VAX VTX online.

Mosbaek, H., J. C. Tjell, and T. Sevel (1988). Plant Uptake of Mercury in Background Areas. *Chemosphere* 17(6):1227-1236.

Nagase, H., Y. Ose, T. Sato, and T. Ishikawa. (1982). Methylation of Mercury by Humic Substances in an Aquatic Environment. Sci. Total Environ. 32:147-156.

NAS (National Academy of Science) (1977). An Assessment of Mercury in the Environment. Safe Drinking Water Committee, National Research Council.

Nater, E. and D. Grigal. 1992. Regional Trends in Mercury Distribution across the Great Lakes States, north central USA. Nature 358:139-141.

National Institute for Occupational Safety and Health (1977). A Recommended Standard for Occupational Exposure to Inorganic Mercury.

National Marine Fisheries Service (NMFS) (1993). Fisheries of the United States, 1992. Current Fishery Statistics No. 9200, National Oceanic and Atmospheric Administration, U. S. Department of Commerce, U. S. Government Printing Office, Washington, DC.

New Jersey Department of Environmental Protection and Energy (1993). Final Report on Municipal Solid Waste Incineration. Volume II: Environmental and Health Issues.

New Jersey Department of Environmental Protection and Energy Division of Science and Research (1994). Preliminary Assessment of Total Mercury Concentrations in Fishes from Rivers, Lakes and Reservoirs of New Jersey. 93-15F.

New York State Department of Environmental Conservation (1990). Chemical Contaminants in Fish from the St. Lawrence River Drainage on Lands of the Mohawk Nation at Akwesasne and Near the General Motors Corporation /Central Foundry Division Massena, New York Plant. Technical Report 90-1.

NMFS (National Marine Fisheries Service). The current publically-available National Marine Fisheries Service Data base was supplied to U.S. EPA via fax from Malcolm Meaburn (Charleston Laboratory/Southeast Fisheries Science Center/National Marine Fisheries Service/National Oceanic and Atmospheric Administration/U.S. Dept. Of Commerce) to Kathryn Mahaffey (Environmental Criteria and Assessment Office-Cincinnati,OH/Office of Health and Environmental Assessment/Office of Research and Development/ U.S. Environmental Protection Agency). February 23, 1995.

NOAA (1978) as described in the NMFS data base.

Nriagu, J. O. (1979). The Biogeochemistry of Mercury in the Environment. Elsevier/North Holland. Biomedical Press: New York.

Occupational Safety and Health Administration, Job Health Hazards Series (1975). Mercury. OSHA report 2234.

Occupational Safety and Health Administration (1989). Industrial Exposure and Control Technologies for OSHA Regulated Hazardous Substances: Mercury, Aryl and Inorganic Compounds. p 1207-1213.

Olmez, I., G. Keeler and P. Hopke. 1994. Interim Data Interpretation Report of MIT/ALSC Data Set (MIT Report No. MITNRL-060).

Olstad, M. L., R. I. Hooland, N. Wandel, and P. A. Hengsten (1987). Correlation between Amalgam Restorations and Mercury Concentrations in Urine. *J. Dent. Res.* 66:1179-1182.

Overcamp, T.J. (1977). *Modelling of Air Quality for Industrial Pollution Control*, Appendix from a Continuing Education Shortcourse taught at Clemson University on December 7, 1977.

Parks, J. W., A. Lutz, and J. A. Sutton (1989). Water Column Methylmercury in the Wabigoon/English River-Lake System: Factors Controlling Concentrations, Speciation, and Net Production. *Can. J. Fisher. Aq. Sci.* 46:2184-2202.

Pasquill, F. (1961). The Estimation of Dispersion of Windborne Material, *Meteorol. Mag.*, 90.33-49.

Pasquill, F. (1974). *Atmospheric Diffusion, the Dispersion of Windborne Material from Industrial and other Sources*, Ellis Horwood, New York.

Pasquill, F. (1976). *Atmospheric Dispersion Parameters in Gaussian Plume Modelling: Part II. Possible Requirements for Change in Turner Workbook Values*, Report EPA-600/4-760306, U.S. Environmental Protection Agency.

PEI Associates, Inc and H.E. Cramer Company, Inc. (1986). Air quality modeling analysis of municipal waste combustors. Prepared for Monitoring and Data Analysis Division, Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina 27711.

Pennington, J. A. T. (1983). Revision of the Total Diet Study Food List and Diets. *J. Amer. Dietetic Assoc.* 82:166-173.

Petersen, G., Å. Iverfeldt and J. Munthe, 1995. Atmospheric mercury species over Central and Northern Europe. Model calculations and comparison with observations from the Nordic Air and Precipitation Network for 1987 and 1988. Atmospheric Environment 29:47-68.

Pierce, T.E. and D.B. Turner (1980). User's Guide for MPTER: A multiple point Gaussian dispersion algorithm with optional terrain adjustment. EPA-600/8-80-016. U.S. EPA., Research Triangle Park, N.C.

Poderbarac, D. S. (1984). Pesticide, Metal, and Other Chemical Residues in Adult Total Diet Samples. (XIV). October 1977 - September 1978. *J. Assoc. Off. Anal. Chem.* 67:176-185.

Pollman, C. D., G. A. Gill, W. M. Landing, D. A. Bare, J. Guentzel, D. Porcella, E. Zillioux, and T. Atkeson. (1994). Overview of the Flordia Atmospheric Mercury Study (FAMS).

Porcella, D.B. (1994). Mercury in the Environment: Biogeochemistry. Pp. 3-19 in Watras, C.J. and J.W. Huckabee eds. Mercury Pollution Integration and Synthesis.

Putnam, J.J. 1991. Food Consumption, 1970-1990. Food Review 14(3):2-12. July-September.

Rada, R., J Wiener, M. Winfrey, and D. Powell. 1989. Recent increases in atmospheric deposition of mercury to north-central Wisconsin Lakes inferred from sediment Analysis. Arch. Environ. Contam. Toxicol. 18:175-181.

Randerson, D. (1984) D. Randerson, editor, *Atmospheric Science and Power Production*, DOE/TIC-27601).

Rao, K.S. and L. Satterfield (1980). A study of the probable environmental impact of fugitive coal dust emissions at the Ravenswood Power Plant, New York. ATDL Contribution 80/26, NOAA, Oak Ridge, TN.

Rao, K.S. (1981). *Analytical Solutions of a Gradient-Transfer Model for Plume Deposition and Sedimentation*, National Oceanic and Atmospheric Administration, Environmental Research Laboratories, Technical Memorandum ERL ARL-109.

Revis, N. W., T. R. Osborne, G. Holdsworth, and C. Hadden (1990). Mercury in Soil: A Method for Assessing Acceptable Limits. *Arch. Environ. Contam. Toxicol.*, 19:221-226.

Richardson, M., M. Mitchell, S. Coad and R. Raphael. 1995. Exposure to Mercury in Canada: A Multimedia Analysis. Water, Air and Soil Pol. 80:21-30.

Schuster, E (1991). The Behavior of Mercury in the Soil with special emphasis on Complexation and Adsorption processes- A Review of the Literature. *Water, Air and Soil Poll.* 56:667-680.

Schroeder, W. H. and R. A. Jackson (1987). Environmental Measurements with an Atmospheric Mercury Monitor having Specific Capabilities. *Chemosphere*, 16:183-199.

Seafood Business (September/October 1993). p. 35

Sehmel, G.A. (1984). Deposition and Resuspension, in *Atmospheric Science and Power Production*, D. Randerson (Ed.), DOE/TIC-27601.

Seritti, A., A. Petrosino, E. Morelli, R. Ferrara, and C. Barghigiani (1982). The biogeochemical cycling of mercury in the Mediterranean. Part I: Particulate and dissolved forms of mercury in the northern Tyrrhenian Sea. *Environ. Tech. Lett.* 3:251-256.

Shannon, J. D., and E. C. Voldner (1994). Modelling Atmospheric Concentrations and Deposition of Mercury to the Great Lakes. Presented at the DOE/FDA/EPA Workshop on Methylmercury and Human Health, Bethesda, MD March 22-23 1994.

Sherlock, R.H. and E.A. Stalker (1941). A Study of Flow Phenomena in the Wake of Smoke Stacks, *Engineering Research Bulletin* 29, University of Michigan, Ann Arbor.

Shitara, K., and A. Yasumasa (1976). *Hokkaidoritzu Eigel kenkyushoho Hokkaidorita* 26:73-78.

Siegel, S. M., B. Z. Siegel, C. Barghigani, K. Aratani, P. Penny, and D. Penny (1987). A Contribution to the Environmental Biology of Mercury Accumulation in Plants. *Water, Air and Soil Poll.* 33:65-72.

Simonin, H. A., S. P. Gloss, C. T. Driscoll, C. L. Schofield, W. A. Kretser, R. W. Karcher, and J. Symula (1994). Mercury in Yellow Perch from Adirondack Drainage Lakes (New York, U.S.), soon to be published in: (1994) Watras, C. J. and J. W. Huckabee [eds], Mercury Pollution: Integration and Synthesis, Lewis Publishers, Boca Raton, FL (in press).

Skerving, S. (1988). Mercury in Women Exposed to Methylmercury Though Fish Consumption and in Their Newborn Babies and Breast Milk. *Bull. Environ. Contam. Toxicol.* 41:475-482.

Slemr, F. and E. Langer. 1992. Increase in Global Atmospheric Concentrations of Mercury inferred from Measurements over the Atlantic Ocean. *Nature* 335:434-437.

Slinn, W.G.N. (1984). Precipitation Scavenging, in *Atmospheric Science and Power Production*, D. Randerson, ed. DOE/TIC-27601.

Sloan, R. (1990). Trends in Mercury Concentrations of the Fish of Onondaga Lake. in Proceedings of the Onondaga Lake Remediation Conference, Bolton Landing, NY, February 5-8 1990.

Smith, M.E. (1951). The Forecasting of Micrometeorological Variables, *Meteorol. Monogr.*, 4:50-55.

Snapp. K. R., D. B. Boyer. L. C. Peterson, and C. W. Svare (1989). The Contribution of Dental Amalgams to Mercury in Blood. *J. Dent. Res.* 68:780-785.

Sorensen, J., G. Glass, K. Schmidt, J. Huber and G. Rapp (1990). Airborne Mercury Deposition and Watershed Characteristics in Relation to Mercury Concentrations in Water, Sediments, Plankton and Fish of Eighty Northern Minnesota Lakes. *Environ. Sci. Technol.* 24:1716-1727.

Sorensen, J., G. Glass, K. Schmidt, and G. Rapp (1991). Mercury Concentrations in Fish from the St. Louis River Near the Fond Du Lac-Indian Reservation (Below the Cloquet and White Pine Rivers). First Year Report on the St. Louis River Water Resources Project.

Sorensen, J.A., G.E. Glass and K.W. Schmidt (1992). Regional Patterns of Mercury Wet Deposition and Major Ions. Part 2 of Chapter 4 in Mercury in the St. Louis River, Mississippi River, Crane Lake and Sand Point Lake: Cycling, Distribution and Sources. Report to the Legislative Commission on Minnesota Resources. April, 1992. Water Quality Division Minnesota Pollution Control Agency St. Paul, MN.

Sorensen, J.A., G.E. Glass and K.W. Schmidt (1994). Regional Patterns of Mercury Wet Deposition. *Environ. Sci. Tech.* 28:2025-2032.

Stafford, C. 1994. Mercury Contamination in Maine Predatory Fishes. Masters Thesis Submitted to The Graduate School, University of Maine.

Stafford, C. P. (1994). Mercury Concentrations in Maine Predatory Fishes. Masters Thesis, University of Maine.

Stern, A. (1993). Re-evaluation of the Reference Dose for Methylmercury and Assessment of Current Exposure Levels. Risk Analysis 13, 3:355-363.

Subcommittee on Criteria for Dietary Evaluation, Coordinating Committee on Evaluation of Food Consumption Surveys, Food and Nutrition Board, Commmission on Life Sciences, National Research Council, National Academy Press, Washington, DC. 1986.

Sumo, E., B.R. Singh, A.R. Selmer-Olsen, and K. Steenburg (1985). Uptake of ^{203}Hg-labeled Mercury compounds by Wheat and Beans Grown on an oxisol. *Plant and Soil* 85: 347-355.

Swain, E. B., and D. D. Helwig (1989). Mercury in Fish from Northeastern Minnesota Lakes: Historical Trends, Environmental Correlates, and Potential Sources. *Journal of the Minnesota Acadamy of Science* 55:103-109.

Swain, E. B., D. A. Engstrom, M. E. Brigham, T. A. Henning, and P. L. Brezonik. (1992). Increasing Rates of Atmospheric Mercury Deposition in Midcontinental North America. *Science* 257:784-787.

Swedish EPA (1991). Mercury in the Environment: Problems and Remedial Measures in Sweden. ISBN 91-620-1105-7.

Swedish Expert Group (1971). Methylmercury in Fish. A toxicological-epidemiological evaluation of risks. *Nord. Hyd. Tidskr.* Supp. 4:1-364.

Szymczak, J. and H. Grajeta.(1992). Mercury Concentrations in Soil and Plant Material, *Pol. J. Food Nutr. Sci.*, Vol 1/42, No.2, pp.31-39.

Tamura, R., M. Fukuzaki, Y. Hirano, and Y. Mitzushima (1985). Evaluation of Mercury Contamination using Plant Leaves and Humus as Indicators. *Chemosphere* 14(11/12):1687-1693.

Temmerman, L. R., R. Vandeputte, and M. Guns (1986). Biological Monitoring and Accumulation of Airborne Mercury in Vegetables. *Environ. Poll.*, 41:139-151.

Temple, P. J. and S. N. Linzon (1977). Contamination of Vegetation, Soil, Snow and Garden Crops by Atmospheric Deposition of Mercury from a Chlor-Alkali Plant, in (1977) D. D. Hemphill [ed] Trace Substances in Environmental Health - XI, Univ Missouri, Columbia. p. 389-398.

Tollefson, L., and F. Cordle (1986). Methylmercury in Fish: A review of Residual Levels, Fish Consumption and Regulatory Action in the United States. *Envir. Health Perspect.* 68:203-208.

Turner , D.B. (1967). *Workbook of Atmospheric Dispersion Estimates*, Public Health Service 999-AP-26, Robert A. Taft Sanitary Engineering Center, Cincinnati, Ohio.

Turner, D.B. (1970). *Workbook of Atmospheric Dispersion Estimates.* Public Health Service Publication No. 999-AP-26, U.S. EPA, Research Triangle Park, N.C.

U.S.D.A. 1995. Adjustments to U.S.D.A. recipe files. Fax from Betty Perloff (Agricultural Research Service/U.S.D.A.) to Kate Mahaffey (Environmental Criteria and Assessment Office-Cincinnati,OH/Office of Health and Environmental Assessment/Office of Research and Development/U.S. Environmental Protection Agency). March 13, 1995.

U.S. Department of Commerce (1978). Report on the chance of U.S. seafood consumers exceeding the current acceptable daily of mercury and on recommended regulatory controls. National Oceanic and Atmospheric Administration, Washington, DC.

U.S. EPA (1975). Control of Water Pollution from Cropland: Volume I, A Manual for Guideline Development. EPA-600/2-75-026a. Office of Research and Development.

U.S. EPA (1976). Control of Water Pollution from Cropland: Volume I, An Overview. EPA-600/2-75-026b. Office of Research and Development.

U.S. EPA (1985). *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water (Part 1).* Washington, D.C. EPA/600/6-85/002-A.

U.S. EPA (1988). Drinking Water Criteria Document for Inorganic Mercury. ECAO-CIN-025

U.S. EPA (1989). Assessing Human Health Risks from Chemically Contaminated Fish and Shellfish: A Guidance Manual. EPA/500/3-89-00.

U.S. EPA (1990). Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emission. Interim Final. EPA/600/6-90/003.

U.S. EPA (1991). Feasibility of Environmental Monitoring and Exposure Assessment for a Municipal Waste Combustor: Rutland, Vermont Pilot Study. EPA/600/8-91/007.

U.S. EPA (1992a). Assessment and Remediation of Contaminated Sediments (ARCS) Program. EPA 905-R92-007.

U.S. EPA (1992b). Assessment and Remediation of Contaminated Sediments (ARCS) Program. EPA 905-R92-008.

U.S. EPA (1992c). National Study of Chemical Residues in Fish. EPA 823-R-92-008a.

U.S. EPA (1992). *User's Guide for the Industrial Source Complex (ISC2) Dispersion Models, Volume II - Description of Model Algorithms*, EPA-450/4-92-008b, U.S. Environmental Protection Agency, Research Triangle Park, North Carolina 27711.

U.S. EPA. 1993. Great Lakes Water Quality Initiative Criteria Documents for the protection of Wildlife (proposed). DDT, Mercury; 2,3,7,8-TCDD, PCBs. Office of Water, Office of Science and Technology, Washington, D.C.

U.S. EPA (1993). Summary Review of Health Effects Associated with Mercuric Chloride. Office of Health and Environmental Assessment, Washington, D.C. EPA/600/R-92/1993. September 1993.

U.S. EPA (1993a). Summary Review of Health Effects Associated with Mercuric Chloride. EPA/600/R-92/199.

U.S. EPA (1993b). Wildlife Exposure Factors Handbook. Prepared by ORD/OHEA. EPA/600/R-93/187a & b.

U.S. EPA (1993c). Wildlife Criteria Portions of the Proposed Water Quality Guidance for the Great Lakes System. (Proposed) Office of Water and Office of Science and Technology. EPA-822-R-93-006.

U.S. EPA (1996). Mercury Study Report to Congress. Volume II: Inventory of Anthropogenic Mercury Emissions in the United States. SAB REVIEW DRAFT.

Vreman, K., N.J. van der Veen, E.J. van der Molen and W.G. de Ruig (1986). Transfer of cadmium, lead, mercury and arsenic from feed into milk and various tissues of dairy cows: chemical and pathological data. *Netherlands Journal of Agricultural Science* 34:129-144.

Wark, K. and C.F. Warner (1981). *Air Pollution Its Origin and Control*, Harper Collins, 1981.

Watras, C. J. and N. S. Bloom (1992). Mercury and Methylmercury in Individual Zooplankton: Implications for bioaccumulation. *Limnol. Oceanagr.*, 37(6):1313-1318.

Welch, L. J. (1994). Contaminant Burdens and Reproductive Rates of Bald Eagles Breeding in Maine. Masters Thesis, University of Maine.

Wells, A. E. (1917). Results of Recent Investigations of the Smelter Smoke Problem, *Ind. Eng. Chem.*, 9:640-646.

Whitby, K. (1978). The physical characteristics of sulfer aerosols. Atmosph. Env. 12:135-159.

Wiener, J., W. Fitzgerald, C. Watras and R. Rada. 1990. Partitioning and Bioavailability of Mercury in an Experimentally Acidified Wisconsin Lake. Environ. Toxicol. Chem. 9:909-918.

Wiersma, D., B. J. van Goor, and N. G. van der Veen (1986). Cadmium, Lead, Mercury, and Arsenic Concentrations in Crops and Corresponding Soils in the Netherlands. *J. Agric. Food Chem.*, 34:1067-1074.

Wilken, R. D. and H. Hintelmann (1991). Mercury and Methylmercury in Sediments and Suspended particles from the River Elbe, North Germany. *Water, Air and Soil Poll.* 56:427-437.

Winfrey, M. R. and J. W. M. Rudd (1990). Environmental Factors Affecting the Formation of Methylmercury in Low pH Lakes. *Environ. Toxicol. and Chem.*, 9:853-869.

World Health Organization (1976). Environmental Health Criteria I, Mercury. Geneva.

World Health Organization (1989). Environmental Health Criteria 86: Mercury Environmental Aspects. Geneva.

World Health Organization (1990). Environmental Health Criteria 101: Methylmercury. Geneva.

Wren, C. D., W. A. Scheinder, D. L. Wales, B. M. Muncaster, and I. M. Gray (1991). Relation Between Mercury Concentrations in Walleye (*Sitzostedion vitreum vitreum*) and Northern Pike (*Esox lucius*) in Ontario Lakes and Influence of Environmental Factors. *Can. J. Fisher. Aq. Sci.* 44:750-757.

Xun, L., N. Campbell and J.W. Rudd. (1987). Measurements of Specific Rates of Net Methyl Mercury Production in the Water Column and Surface Sediments of Acidified and Circumneutral Lakes. Can. J. Fish Aquat. Sci. 44:750-757.

Yang, Y. J., C. C. Huang, T. S. Shih, and S. S. Yang (1994). Chronic elemental Mercury Intoxication: Clinical and Field Studies in Lampsocket Manufacturers. *Occup. Environ. Med.* 51(4):267-270.

Zander D., U. Ewers, I. Freier, S. Westerweller, E. Jermann, and A. Brockhaus (1990). Untersuchungen zu Quecksilberbelastung der Bevolkerung II. Quecksilberfreisetzung aus Amalgamfullungen. *Zbl. Hyg.* 190:325-334.

Zeller, K.F. (1984). The Environmental Impact Statement, in *Atmospheric Science and Power Production*, D. Randerson (Ed.), DOE/TIC-27601.

APPENDIX A

PARAMETER JUSTIFICATIONS
SCENARIO INDEPENDENT PARAMETERS

TABLE OF CONTENTS

DISTRIBUTION NOTATION iii

A. SCENARIO INDEPENDENT PARAMETERS A-1
A.1 Chemical Independent Parameters A-1
A.1.1 Basic Constants A-1
A.1.2 Receptor Parameters A-1
A.1.2.1 Body Weight A-2
A.1.2.2 Exposure Duration A-2
A.1.3 Agricultural Parameters A-3
A.1.3.1 Interception Fraction A-3
A.1.3.2 Length of Plant Exposure A-4
A.1.3.3 Plant Yield A-5
A.1.3.4 Plant Ingestion by Animals A-6
A.1.3.5 Soil Ingestion by Animals A-7
A.1.4 Exposure Parameters A-7
A.1.4.1 Inhalation Rate A-8
A.1.4.2 Consumption Rates A-9
A.1.4.3 Soil Ingestion Rate A-10
A.1.4.4 Groundwater Ingestion Rate A-11
A.1.4.5 Fish Ingestion Rate A-12
A.1.4.6 Contact Fractions A-14
A.2 Chemical Dependent Parameters A-15
A.2.1 Basic Chemical Properties A-15
A.2.1.1 Molecular Weight A-15
A.2.1.2 Henry's Law Constant A-15
A.2.1.3 Soil-Water Partition Coefficient A-16
A.2.1.4 Sediment-to-Water Partition Coefficient A-16
A.2.1.5 Suspended Sediment-Water Partition Coefficient A-17
A.2.1.6 Soil and Water Loss Degradation Constants A-18
A.2.1.7 Equilibrium Fraction for Chemical in Soil A-18
A.2.1.8 Equilibrium Fraction for Chemical in Water A-19
A.2.2 Biotransfer Factors A-20
A.2.2.1 Plant-Soil BCF A-21
A.2.2.2 Air-Plant BCF A-24
A.2.2.3 Animal BTF A-28
A.2.2.4 Fish Bioaccumulation Factor A-32
A.2.2.5 Plant Surface Loss Coefficient A-33
A.2.2.6 Fraction of Wet Deposition Adhering A-33
A.3 References A-35

LIST OF TABLES

A-1 Chemical Independent Constants A-1
A-2 Ranges of Values for Suspended Sediment-to-Water Partition Coefficient A-17
A-3 Reported Values for Fraction of Total Mercury that is Methylmercury in Water A-20
A-4 Soil-to-Plant Transfer Coefficients for Mercury (from Cappon, 1987 and Cappon, 1981) A-22
A-5 Other Values for Soil-to-Plant Transfer Coefficients for Hg^{2+} A-23
A-6 Relative Concentration of Mercury in Different Parts of Edible Plants A-26
A-7 Mercury Speciation in Various Plants (Cappon, 1981, 1987) A-27
A-8 Mercury Concentrations in Specific Beef Tissue Media Per Test Group and Dose (from Vreman et al, 1986) A-29
A-9 Animal Biotransfer Factors Derived from Vreman et al. (1986) A-30
A-10 Mercury Concentrations and Resulting BTFs in Lamb Muscle Tissue Per Test Group and Dose (from van der Veen and Vreman 1986) A-31
A-11 Values From Hoffman et al. (1992) and the Values of *Fw* Estimated Using Those Values A-34

DISTRIBUTION NOTATION

A comprehensive uncertainty analysis was not conducted as part of this study. Initially, preliminary parameter probability distributions were developed. These are listed in Appendicies A and B. These were not utilized in the generation of quantative exposure estimates. They are provided as a matter of interest for the reader.

Unless noted otherwise in the text, distribution notations are presented as follows.

Distribution	Description
Log (A,B)	Lognormal distribution with mean A and standard deviation B
Log*(A,B)	Lognormal distribution, but A and B are mean and standard deviation of underlying normal distribution.
Norm (A,B)	Normal distribution with mean A and standard deviation B
U (A,B)	Uniform distribution over the range (A,B)
T (A,B,C)	Triangular distribution over the range (A,C) with mode of B

A. SCENARIO INDEPENDENT PARAMETERS

This appendix describes the scenario-independent parameters used in the exposure modeling for the Mercury Study Report to Congress. Scenario independent parameters are variables whose values are independent of a particular site and are constant among various site-specific situations. Examples of scenario independent parameters are air density, the average height of an adult, or the average crop yield of a particular food item. These scenario independent parameters may be either chemical independent or chemical dependent. The following sections present the chemical independent and chemical dependent parameters used in this study.

A.1 Chemical Independent Parameters

Chemical independent parameters are variables that remain constant despite the specific contaminant being evaluated. The chemical independent variables used in this study are described in the following sections.

A.1.1 Basic Constants

Table A-1 lists the chemical independent constants used in the study, their definitions, and values.

Table A-1
Chemical Independent Constants

Parameter	Description	Value
R	ideal gas constant	8.21E-5 m^3-atm/mole-K
pa	air density	1.19E-3 g/cm^3
ua	viscosity of air	1.84E-4 g/cm-second
Psed	solids density	2.7 kg/L
Cdrag	drag coefficient	1.1E-3
κ	Von Karman's coefficient	7.40E-1
λ_2	boundary thickness	4.0

A.1.2 Receptor Parameters

Receptor parameters are variables that reflect information about potential receptors modeled in the study. These parameters include body weight, exposure duration, and other characteristics of potential receptors.

A.1.2.1 Body Weight

Parameter: BWa, BWc

Definition: Body weights (or masses) of individual human receptors

Units: kg

Receptor	Default Value (kg)
Child	17
Adult	70

Technical Basis:

The default values for children and adults are those assumed in U.S. EPA, 1990.

A.1.2.2 Exposure Duration

Parameter: ED

Definition: Length of time that exposure occurs.

Units: years

Receptor	Default Value (years)	Distribution	Range (years)
Child	18	U(1,18)	1-18
Adult	30	U(7,70)	7-70

Technical Basis:

The 18-year exposure duration for the child is based on U.S. EPA guidance for this study. For adults, the 30-year duration is the assumed lifetime of the facility (U.S. EPA, 1990). It should be noted for noncarcinogenic chemicals the exposure duration is not used in the calculations. The range and distribution are arbitrary to determine the relative sensitivity of this variable, when appropriate.

A.1.3 Agricultural Parameters

A.1.3.1 Interception Fraction

Parameter: RPi

Definition: The fraction of the total deposition within a unit area that is initially intercepted by vegetation.

Units: unitless

Crop	Default Value	Distribution	Range
Leafy vegetables	0.15	Log (0.16, 0.10)	0.08 - 0.38
Legume vegetables	0.008	Log(0.008, 0.004)	0.005 - 0.01
Fruiting vegetables	0.05	Log(0.05, 0.05)	0.004 - 0.08
Rooting vegetables	0	N/A	N/A
Grains and cereals	0	N/A	N/A
Forage	0.47	Norm(0.47, 0.3)	0.02 - 0.89
Silage	0.44	Log (0.44, 0.3)	
Fruits	0.05	Log (0.05, 0.05)	0.004 - 0.08
Potatoes	0	N/A	N/A

Technical Basis:

For leafy vegetables, Baes et al. (1984) obtained an average interception fraction of 0.15 where it was emphasized that this value represents a theoretical average over the United States. This value was calculated assuming a logistic growth pattern for leafy vegetables and taking into account a distribution of field spacings (for details see Baes et al., p.68). The associated distribution and ranges shown in the previous table were calculated based on Baes's analyses by Belcher and Travis (1989).

For legumes and fruits, Belcher and Travis (1989) used the exposed produce equation that relates the interception fraction to the standing crop biomass (also called productivity) and crop biomass values from Shor et al. (1982) to obtain the range of values given in the previous table. The values for fruiting vegetables are assumed to be the same as for fruits.

The distribution for forage is based on the work of Hoffman and Baes (1979), who determined that the values are normally distributed with the parameters presented in the previous table.

The value for silage was calculated in Baes et al. (1984) and is based essentially on sorghum and corn plantings (Knott, 1957; Rutledge, 1979).

Potatoes, root vegetables and grains are assumed to equal zero since the edible portion of the plant is protected from direct deposition (grains have a protective husk).

A.1.3.2 Length of Plant Exposure

Parameter: TPi

Definition: The amount of time that the edible part of an exposed plant is exposed to direct deposition.

Units: years

Plant Type	Default Value (years)	Distribution	Range (years)
Leafy vegetables	0.157	U(0.082,0.247)	0.082- 0.247
Legume vegetables	0.123	U(0.082,0.247)	0.082- 0.247
Fruiting vegetables	0.123	U(0.082,0.247)	0.082 - 0.247
Forage	0.123	U(0.082,0.247)	0.082 - 0.247
Silage	0.123	U(0.082,0.247)	0.082 - 0.247
Fruits	0.123	U(0.082,0.247)	0.082 - 0.247

Technical Basis:

Bounding estimates were obtained by assuming an average time between successive harvests of 30 and 90 days. This range is based on the values in Baes et al. (1984) of 60 to 90 days and the reported values by the South Coast Air Quality Management District (SCAQMD) (1988) of 45 days for tomatoes and 30-85 days for lettuce.

The default value for leafy vegetables is the midpoint of the range for lettuce. The values for legumes, fruits and fruiting vegetables are based on the value of 45 days for tomatoes. The value for forage and silage is the average time between successive hay harvests and successive grazings by cattle (Baes et al., 1984).

A.1.3.3 Plant Yield

Parameter: YPi

Definition: Yield of the *i*th plant per unit area.

Units: kg (dry weight)/m^2

Type of Crop	Default Value (kg (dry weight)/m^2)	Range (kg (dry weight)/m^2)	Distribution
Leafy vegetables	0.177	0.091 - 0.353	Log (0.177, 0.086)
Legume vegetables	0.104	0.077 - 0.130	Log (0.104, 0.038)
Fruiting vegetables	0.107	0.012 - 0.253	Log(0.107, 0.093)
Rooting vegetables	0.334	0.090 - 0.434	Log(0.334, 0.142)
Grains and cereals	0.3	0.14 - 0.45	Log (0.30, 0.09)
Forage	0.31	0.02- 0.75	0.84482993969
Fruits	0.107	0.012 - 0.253	Log(0.107, 0.093)
Potatoes	0.48	0.405 - 0.555	Log (0.48, 0.106)
Silage	0.84	0.3- 1.34	Log(0.84,0.26)

Technical Basis:

The distributions and ranges shown for all but the silage values are those used in Belcher and Travis (1989). The distributions selected were chosen based on a probability plot for leafy vegetables with data in Shor et al. (1982). The default values are the means of the distributions. Silage was not considered in Belcher and Travis (1989), but the same method by which the default values and distributions were calculated there were replicated using data from Shor et al. (1982) for the purpose of this assessment.

A.1.3.4 Plant Ingestion by Animals

Parameter: QPij

Definition: The daily consumption of plants by livestock.

Units: kg dry weight/day

Livestock Consumption of Plants		Default Value (kg dry weight/day)	Distribution	Range (kg dry weight/day)
Beef/Beef Liver				
	grain	0.97	U(0.5,6.5)	0.5-6.5
	forage	8.80	U(2.0,9.0)	2.0-9.0
	silage	2.50	U(1,5)	1.0-5.0
Dairy				
	grain	2.60	U(0.5,6.5)	0.5 - 6.5
	forage	11.0	U(7,15)	7.0-15.0
	silage	3.30	U(1,5)	1.0-5.0
Pork				
	grain	3.0	U(2,4)	2.0-4.0
	silage	1.3	U(0.5,3)	0.5-3.0
Sheep (lamb)				
	forage	1.1	U(0,2)	0.0 - 2.0
Poultry/Eggs				
	grain	0.08	U(0.04,0.10)	0.04-0.10

Technical Basis:

With the exception of the beef liver, egg and lamb-forage values, the default values are from U.S. EPA (1990). The value for beef liver is assumed to be the same as for cattle, and the value for eggs is assumed to be the same as for poultry. The value for lamb-forage is from the National Academy of Sciences (NAS,1987).

The ranges shown are based on a combination of the ranges determined by Belcher and Travis (1989), the U.S. EPA (1990) values, and the objective of capturing all of the most likely values.

Although lognormal distributions were chosen in Belcher and Travis (1989), this was not based on the actual distribution of the available data; that is, no probability plots were done. For that reason, uniform distributions are suggested here.

A.1.3.5 Soil Ingestion by Animals

Parameter: QSj

Definition: Quantity of soil ingested daily by the a specific animal.

Units: kg/day

Livestock	Default Value (kg/day)	Range (kg/day)
Beef/beef liver	0.39	0.1 - 0.72
Dairy	0.41	0.1 - 0.72
Pork	0.034	0.0 - 0.0688
Sheep (lamb)	0.05	0.01 - 0.15
Poultry/eggs	0.009	0.006 - 0.012

Technical Basis:

The values for beef cattle and dairy cattle are from McKone and Ryan (1989). The value for beef liver is assumed to be the same as for beef. The value for pork is the mean of the distributions used in Belcher and Travis (1989) and are based on values in Fries (1987). The sheep value is from Fries (1982). The value for poultry is the mean of the distribution used in the Hanford Environmental Dose Reconstruction Project (HEDR, 1992) and is based on values for free-ranging chickens. The range is that used in HEDR (1992).

For beef, dairy and pork, the ranges are from Belcher and Travis (1989).

The range for sheep is based on the values reported in Fries (1982). The lower end of the range is for sheep that are fed in a lot, in which case they eat little soil. The upper end is based on sheep grazing on poor pasture land.

A.1.4 Exposure Parameters

Exposure parameters are variables that directly affect an individual's dose or intake of a contaminant. Such parameters include inhalation and ingestion rates of air, water and crops and the surface area of skin for the purposes of dermal contact scenarios.

A.1.4.1 Inhalation Rate

Parameter: INH

Definition: Rate of inhalation of air containing contaminants.

Units: m^3/day

Receptor	Default Value (m^3/day)	Distribution
Infant	5.14	T(1.7,5.14,15.4)
Child	16	T(2.9,16,53.9)
Adult	20	T(6,20,60)

Technical Basis:

The default value for infants is the central value of the distribution used for 1 year olds in Hanford Environmental Dose Reconstruction Project (HEDR) (1992) and is from Roy and Courtay (1991). The default value for children is based on U.S. EPA (1990). The default value for adults is that recommended in U.S. EPA (1991), which states that this value represents a reasonable upper bound for individuals that spend a majority of time at home.

The range for infants is that used for 1 year olds in HEDR (1992) and was determined by scaling the value 5.14 by 0.3 and 3.0, respectively. The range for children is the smallest range containing the values used for 5-, 10-, and 15-year-old children in HEDR (1992). The range for the adult was obtained by scaling the default value by the same numbers used for infants of 0.3 and 3.0 (we note that HEDR, 1992 used a slightly higher central value of 22 m^3/day).

To prevent a bias towards upper-end inhalation rates, triangular distributions were considered more appropriate than more arbitrary uniform distributions, with a most likely value equal to the default value.

A.1.4.2 Consumption Rates

Parameter: CPi, CAj

Definition: Consumption rate of food product per kg of body weight per day.

Units: g dry weight/kg BW/day

Food Type	Child (gDW/kgBW/day)	Adult (g DW/kg BW/day)
Leafy Vegetables	0.008	0.0281
Grains and cereals	3.77	1.87
Legumes	0.666	0.381
Potatoes	0.274	0.170
Fruits	0.223	0.570
Fruiting vegetables	0.120	0.064
Rooting Vegetables	0.036	0.024
Beef, excluding liver	0.553	0.341
Beef liver[a]	0.025	0.066
Dairy (milk)	2.04	0.599
Pork	0.236	0.169
Poultry	0.214	0.111
Eggs	0.093	0.073
Lamb[a]	0.061	0.057

[a] Only the 95-100 percentile of the data from TAS (1991) was nonzero.

Technical Basis:

All of the values reported above are given on a gram dry weight per kg of body weight per day basis. With the exception of the ingestion rates for adults for leafy vegetables and fruits, the values are either the 50-55 percentile (or the 95-100 percentile if the median was zero) of the data from Technical Assessment Systems, Inc. (TAS). The values for the percentiles were reported in g DW/kg of body weight per day.

TAS conducted this analysis of food consumption habits of the total population and five population subgroups in the United States. The data used were the results of the Nationwide Food Consumption Survey (NFCS) of 1987-88 conducted by the United States Department of Agriculture. The information in the NFCS was collected during home visits by trained interviewers using one-day interviewer-recorded recall and a two-day self-administered record. A stratified area-probability sample of households was drawn in the 48 contiguous states from April 1987 to 1988. More than 10,000 individuals provided information for the basic survey.

Each individual's intake of food was averaged across the 3 days of the original NFCS survey, and food consumption for each food group was determined for each individual. Percentiles were then computed for six population subgroups:

- U.S. population
- males ≥ 13 years
- females ≥ 13 years
- children 1-6 years
- children 7-12 years
- infants < 1 year.

The values for children in the previous table are based on the data for children between 7 and 12 year of age, while the adult values are for males older than 12 years of age. The males older than 12 years of age were chosen to represent the adult since rates for females are lower; this is recoganized to be somewhat conservative. The United States population rates include the rates of children which were considered inappropriate for the hypothetical adult receptors modeled in this analysis.

The values for leafy vegetables and fruits for adults are from (USU.S. EPA 1989).

A.1.4.3 Soil Ingestion Rate

Parameter: Cs

Definition: Amount of soil ingested daily.

Units: g/day

Receptor	Default Value (g/day)	Distribution	Range (g/day)
Pica Child	7.5	U(5,10)	5-10
Child	0.2	U(0.016,0.2)	0.016-0.2
Adult	0.1	U(0.016,0.1)	0.016-0.1

Technical Basis:

Soil ingestion may occur inadvertently through hand-to-mouth contact or intentionally in the case of a child who engages in pica. The default values for adults and non-pica children are those suggested for use in U.S. EPA (1989). More recent studies have found that these values are rather conservative. For example, Calabrese and Stanek (1991) found that average soil intake by children was found to range from 0.016 to 0.055 g/day. This range, in conjunction with the suggested U.S. EPA values, was used to obtain the ranges shown.

Several studies suggest that a pica child may ingest up to 5 to 10 g/day (LaGoy, 1987, U.S. EPA, 1989). This range was selected, and the midpoint was chosen as the default value.

A.1.4.4 Groundwater Ingestion Rate

Parameter: Cw

Definition: The amount of water consumed each day.

Units: L/day

Receptor	Default Values (L/day)	Distribution
Child	1.0	Log*(0.378; 0.079)
Adult	2.0	Log*(0.1; 0.007)

Technical Basis:

The default values for children and adult are those also suggested in U.S. EPA (1989) and were first published by the Safe Drinking Water Committee of the National Academy of Sciences (NAS, 1977).

The distributions are those computed in Roseberry and Burmaster (1992). In that paper, lognormal distributions were fit to data collected in a national survey for both total water intake and tap water intake by children and adults. These data were originally gathered in the 1977-1978 Nationwide Food Consumption Survey of the United States Department of Agriculture and were analyzed by Ershow and Cantor (1989).

In Roseberry and Burmaster (1992), distributions were fit to the intake rates for humans ages 0-1 year, 1-11 years, 11-20 years, 20-65 years and older than 65 years. The distribution for children ages 1-11 was chosen for the child's distribution given in the previous table and the distribution for adults ages 20-65 was used for the adult. For the purpose of the present analysis, the tap water intake was deemed more appropriate than total water intake. The total water intake included water intrinsic in foods that are accounted for in the agricultural pathways, while the tap water intake was the sum of water consumed directly as a beverage and water added to foods and beverages during preparation.

The minima and maxima were selected as the 2.5 and 97.5 percentiles, respectively.

A.1.4.5 Fish Ingestion Rate

Parameter: Cf

Definition: Quantity of locally - caught fish ingested per day.

Units: g/day

Receptor	Default Value (g/day)
High End Fisher	60
Child of high end fisher	20
Recreational Angler	30

Technical Basis:

Because of the bioaccumulation of methylmercury in fish, the fish ingestion rate is an important parameter for modeling mercury exposure. Fish consumption rates are difficult to determine for a general population study because individual fish ingestion rates vary widely across the United States. This animal protein source may be readily consumed or avoided on a seasonal, social, economic or demographic basis. Ideally, for an actual site, specific surveys identifying the type, source, and quantity of fish consumed by area residents would be used. Within the context of this study, it is not possible to characterize this variability completely (Please see Appendix H for a more complete discussion of reported fish consumption rate variability).

For this part of the assessment, individuals in three broad groups of exposed populations will be considered: high end fishers, recreational anglers and the general population. For the general population, no commercial distribution of locally caught fish was assumed. All consumers of locally-caught fish were assumed to be recreational anglers or subsistence fishers.

In U.S. EPA's 1989 Exposure Factors Handbook, fish consumption data from Puffer (1981) and Pierce et al. (1981) are suggested as most appropriate for fish consumption of recreational anglers from large water bodies. The median of this subpopulation is 30 g/day with a 90th percentile of 140 g/day (340 meals/year). The median was used as the surrogate value for recreational anglers.

For subsistence fishers, human fish consumption data were obtained from the report of the Columbia River Inter-Tribal Fish Commission (1994), which estimated fish consumption rates for members of four tribes inhabiting the Columbia River Basin. The estimated fish consumption rates were based on interviews with 513 adult tribe members who lived on or near the reservation. The participants had been selected from patient registration lists provided by the Indian Health Service. Adults interviewed provided information on fish consumption for themselves and for 204 children under 5 years of age.

During the study fish were consumed by over 90% of the population with only 9% of the respondents reporting no fish consumption. Monthly variations in consumption rates were reported. The average daily consumption rate during the two highest intake months was 107.8 grams/day, and the daily consumption rate during the two lowest consumption months was 30.7 grams/day. Members who were aged 60 years and older had an average daily consumption rate of 74.4 grams/day. During

the past two decades, a decrease in fish consumption was generally noted among respondents in this survey. The maximum daily consumption rate for fish reported for this group was 972 grams/day.

The mean daily fish consumption rate for the total adult population (aged 18 years and older) was reported to be 59 grams/day. The mean daily fish consumption rate for the adult females surveyed was 56 g/day and the mean daily fish consumption rate for the adult males surveyed was 63 grams. A value of 60 grams of fish per day was selected for the subsistence angler modeled in this report.

Other fish consumption rate studies for specific subpopulations (i.e., anglers and subsistence consumers) have been conducted. These studies are briefly described in Appendix H. These studies demonstrate the wide range of fish consumption rates exhibited across the U.S. population. They also tend to corroborate the estimates to be used in this analysis. These analyses also illustrate the difficulty in determining average and high-end consumption rates for subpopulations considered to be more likely to consume more fish.

In the lacustrine scenarios of this assessment, all fish were assumed to originate from the lakes, which are considered to represent several small lakes that may be present in a hypothetical location.

The effects of fish preparation for food on extant mercury levels in fish have also been evaluated (Morgan et al., 1994). Total mercury levels in walleye were found to be constant before and after preparation; however, mercury concentrations in the cooked fish were increased 1.3 to 2.0 times when compared to mercury levels in the raw fish. It was suggested that this increase was probably due to water and fat loss during cooking and fish skin removal. A preparation factor adjustment was noted but not implemented in this analysis because human consumption levels were measured on uncooked fish. (For more information see Appendix H.)

A.1.4.6 Contact Fractions

Parameter: FPi, Faj

Definition: that fraction of the food type grown or raised on contaminated land

Units: Unitless

Food	Subsistence Farmer	Rural Home Gardener/ Subsistence Fisher	Urban Gardener	Comment
Grains	1	0.667	0.195	Values are for corn from Table 2-7 in U.S. EPA (1989)
Legumes	1	0.8	0.5	Values are for peas from Table 2-7 in U.S. EPA (1989).
Potatoes	1	0.225	0.031	Values are for total fresh potatoes from Table 2-7 in U.S. EPA (1989).
Root Vegetables	1	0.268	0.073	Values are for carrots from Table 2-7 in U.S. EPA (1989).
Fruits	1	0.233	0.076	Values are for Total non-citrus fruit from Table 2-7 in U.S. EPA (1989).
Fruiting Vegetables	1	0.623	0.317	Values are for tomatoes from Table 2-7 in U.S. EPA (1989).
Leafy Vegetables	1	0.058	0.026	Values are for lettuce from U.S. EPA (1989).
Beef	1	0	0	
Beef liver	1	0	0	
Dairy	1	0	0	
Pork	1	0	0	
Poultry	1	0	0	
Eggs	1	0	0	
Lamb	1	0	0	

Technical Basis:

The values for the subsistence farmer are consistent with the assumptions regarding this scenario. The values for the gardeners are from U.S. EPA (1989), per U.S. EPA guidance. Because it is assumed that only the subsistence farmers will consume contaminated animal products, the contact fractions for gardeners is 0 for consumption of local animal products.

A.2 Chemical Dependent Parameters

Chemical dependent parameters are variables that change depending on the specific contaminant being evaluated. The chemical dependent variables used in this study are described in the following sections.

A.2.1 Basic Chemical Properties

The following sections list the chemical properties used in the study, their definitions, and values.

A.2.1.1 Molecular Weight

Parameter: Mw

Definition: The mass in grams of one mole of molecules of a compound.

Units: g/mole

Chemical	Default Value (g/mole)
Hg^0, Hg^{2+}	201
Methylmercury	216
Methyl mercuric chloride	251
Mercuric chloride	272

A.2.1.2 Henry's Law Constant

Parameter: H

Definition: Provides a measure of the extent of chemical partitioning between air and water at equilibrium.

Units: atm-m^3/mole

Chemical	Default Value (atm-m^3/mole)
Hg^0	$7.1x10^{-3}$
Hg^{2+} ($HgCl_2$)	$7.1x10^{-10}$
Methylmercury	$4.7x10^{-7}$

Technical Basis:

The higher the Henry's Law Constant, the more likely a chemical is to volatilize than to remain in the water. The value for Hg^0 is from Iverfeldt and Persson (1985), while the other values are from Lindquist and Rodhe (1985).

A.2.1.3 Soil-Water Partition Coefficient

Parameter: Kd

Definition: Equilibrium concentration in dry soil divided by concentration in water.

Units: mL/g

Chemical	Default Value (mL/g)
Hg^{2+}	53,700
Methylmercury	53,700

Technical Basis:

The values in the previous table are the geometric mean of calibrated values (see Appendix C of Volume III).

A.2.1.4 Sediment-to-Water Partition Coefficient

Parameter: Kdb

Definition: Equilibrium concentration in dry sediment divided by concentration in water.

Units: mL/g

Chemical	Default Value (mL/g)
Hg^{2+}	157,000
Methylmercury	157,000

Technical Basis:

The values in the previous table are the geometric mean of calibrated values (see Appendix C of Volume III).

A.2.1.5 Suspended Sediment-Water Partition Coefficient

Parameter: Kdw

Definition: Suspended sediment-water partition coefficient.

Units: L/kg

Chemical	Default Value (L/kg)	Range
Hg^{2+}	95000	1340-188,000
Methylmercury	650000	320,000 - 1,000,000

Technical Basis:

For divalent mercury, data were available from three studies, and are shown in Table A-2. The default value is the midpoint of the range.

Table A-2
Ranges of Values for Suspended Sediment-to-Water Partition Coefficient

Range (L/kg)	Reference
1380-188,000	Moore and Ramamodoray (1984)
118,000	Glass et al. (1990)
86,800-113,000	Robinson and Shuman (1989)

For methylmercury, the only data found that specifically address suspended material are those in Bloom et al. (1991). In particular, they report that "Regardless of pH, for over three orders of magnitude, the log K_d for seston [suspended matter] was in the range of 5.5 to 6.0." The range listed in the previous table corresponds to this range. The midpoint of the observed range is used as the default value.

A.2.1.6 Soil and Water Loss Degradation Constants

Parameter: ksg and kwg

Definition: Soil and water body loss of the contaminant due to biotic and abiotic degradation and aqueous hydrolysis, respectively.

Units: /yr

Chemical	Default Value (year)	Range
Hg^0	0.0	N/A
Hg^{2+}	0.0	N/A
Methylmercury	0.0	N/A

Technical Basis:

Data indicate that equilibrium is established between different species of mercury rather than a degradation/breakdown process. Parks et al., (1989) found that "In water, methylmercury and inorganic appear to be in quasi-equilibrium, as the methylmercury/total mercury ratio in river water is independent of contact time with sediments, the atmosphere, and the theoretical residence time of waters." For this reason, it appears reasonable simply to assume no net loss with time if any mercury species occurs in either soil or water.

A.2.1.7 Equilibrium Fraction for Chemical in Soil

Parameter: fspecs

Definition: For all chemicals tied together in soil equilibrium, the fraction which is chemical *i* is given by fspec.

Units: unitless

Chemical	Default Value	Distribution
Hg^0	0	None
Hg^{2+}	0.98	T(0.9,0.98,0.9998)
Methylmercury	0.02	1-%Hg^{2+}

Technical Basis:

Akagi et al. (1979) reported methylmercury fractions of .02, .072 and .089 for sand, silt/woodchips, and woodchip sediments as compared to total mercury. Wilken and Hintelmann (1991) reported that 0.10 of the total mercury in sediments from the River Elbe in Germany is methylated, although they pointed out that others had reported maximums of 0.01 and 0.02. Hildebrand et al. (1980) found methylmercury fractions of .0002 - .0005 in sediments from the Holston River, VA.

The measurements in the previous table did not distinguish between Hg^{2+} and Hg^0 in the remaining fractions, leaving the partitioning of these species in soil uncertain. It is known that Hg^0 can be formed from reduction of Hg^{2+} in the soil environment, a fraction of which will volatilize and a fraction of which can be bound to organic matter. Both processes depend strongly on soil conditions (Nriagu, 1979). At the redox potential normally found in soils, however, Hg^{2+} complexes are expected to be predominant than Hg^0.

Cappon (1984) found that percent of methylmercury over total mercury for nonamended soils is 2.6%. This is an upper bound on values from unpublished data reported by several authors (Lindqvist et al., 1991).

A.2.1.8 Equilibrium Fraction for Chemical in Water

Parameter: fspecw

Definition: For all chemicals tied together in water equilibrium, the fraction which is chemical *i* is given by fspecw.

Units: unitless

Chemical	Default Value	Distribution	Range
Hg^0	0.02	NA	0.007 - 0.04
Hg^{2+}	0.83	1 - (Methyl+Hg^0 %)[a]	.31 - .96
Methylmercury	0.15	Log(0.14,1.0)	.03 - .65

[a] The distribution is 1 minus the methylmercury concentration and elemental mercury concentration dissolved in the water.

Technical Basis:

The default value given for methylmercury is that suggested in U.S. EPA (1993). In well oxygenated water, the remaining fraction (i.e., non-methylated) will be mainly Hg^{2+} complexes (Nriaga, 1979). There will be a small fraction of total mercury in water that will be Hg^0 due to reduction of Hg^{2+} by humic acid and microorganisms (Nriaga, 1979; Alberts et al., 1974).

Fitzgerald et al. (1991) measured the concentration of total dissolved gaseous mercury in various lake waters and found in all cases that it consisted mainly of elemental mercury (> 97%). Much of these measurements were taken at both basins of Little Rock Lake, WI, from which total mercury concentrations for the acid-treatment and reference basins are known from the work of Watras and Bloom (1992). Comparing the concentrations within each basin gives a possible range for the percent mercury in water that is Hg^0 of 0.7 - 4%, the midpoint of which (2%) we use as the default equilibrium percentage of mercury in the water column that is elemental mercury.

There are a wealth of data on the Methylmercury/Total mercury in the water column. Table A-3 lists the values found reported in the literature. These values were used to determine the range given previously for methylmercury. The range for Hg^{2+} is then given by subtracting the contributions from methylmercury and elemental mercury from the total. (For more details see Volume V of this Report.)

Table A-3
Reported Values for Fraction of Total Mercury that is Methylmercury in Water

Values	Reference
0.26, 0.11, 0.07, 0.07, 0.15	Bloom et al. (1991)
0.01, 0.022, 0.019, 0.054, 0.055, 0.052, 0.049, 0.064	Parks et al. (1989)
0.32, 0.48, 0.57	Akagi et al. (1979)
0.12,0.05	Watras and Bloom (1992)
$\leq$0.025	Bloom and Watras (1989)
0.04-0.05	Lee et al. (1990)
0.26-0.46	Kudo et al. (1982)
0.01-0.89	Gill and Bruland (1990)
0.036-0.273	Bloom and Effler (1990)
0.036-0.053	Lee and Hultberg (1990)

A.2.2 Biotransfer Factors

Biotransfer factors reflect the extent of chemical partitioning between a biological medium (plants, meats or fish) and an external medium (air, soil or water). The following sections describe the BCFs used in this study.

It is necessary to note the uncertainty inherent in determining BCFs for mercury species with regard to plant uptake. In general, there seems to be no consensus in the literature on plant bioconcentration factors for mercury, as values for each crop vary widely among studies. Further, in many studies the mercury speciation is not determined. In deriving BCFs for plant absorption of mercury species fromthe air and soil, it was, therefore sometimes necessary to make assumptions about certain behaviors ofmercury based on whatever information was at hand, as opposed to established scientific knowledg, which was lacking. These assumptions are described in each Technical Basis section that follows, but it is useful at this time to identify some of the general uncertainties regarding plant uptake of mercury.

(1) Plants both absorb and release mercury to the environment. Hanson et al. (1994) demonstrates clearly that at ambient air concentrations forest foliage usually acts as a source of elemental mercury to the atmosphere; deposition (plant absorption) only occurs above a "compensation concentration" at air mercury levels well above background. It is not yet known from where the mercury released by the plants originates (air uptake during periods of high mercury air concentrations, root uptake, Hg(II) absorption, etc.). Similarly, Mosbaek (1988) found that for a given period of time more elemental mercury was released from a plant-soil system than was absorbed by the plant. These cases, however, in no way indicate that mercury is not bioconcentrated in plants; the above behaviors are consistent with mercury being collected by plants only to certain levels, after which any mercury absorbed is simply released.

(2) It is usually not known from where the mercury that is found in plants originated (air vs. soil). Only one study determined the fractions of total mercury in plants which came from air and soil (Mosbaek, 1988); in this study, soil was isotopicly labelled with ^{203}Hg. After some time the specific activity in the plant was compared to that in the soil to ascertain how much of the mercury in the plant came from the soil. Although the experiment worked well, isotopic equilibrium in the soil was never achieved, and the number of plants studied was limited.

(3) The speciation of mercury in plants is often not known. If it is known, it is still very unclear as to how the speciation occurred. The plant speciation may be simply a result of direct uptake of different mercury species from the environment (but from air or soil?). It has been shown, however, that a few plants have the ability to change the species of mercury initially taken up from the environment (Fortmann et al., 1978). Such behavior may have to be accounted for regarding plant uptake of mercury.

A.2.2.1 Plant-Soil BCF

Parameter: BRi

Definition: The ratio of the contaminant concentration in plants (based on dry weight) to that in the soil.

Units: Unitless

Crop	Hg^{2+}		Methylmercury	
	Default Value	**Distribution**	**Default Value**	**Distribution**
Leafy vegetables	0	None	0	None
Legume vegetables	0.015	U(0.00026, 0.157)	0.031	U(0.0, 0.090)
Fruiting vegetables	0.018	U(0.007,0.059)	0.024	U(0.0,0.11)
Rooting vegetables	0.036	U(0.011, 0.073)	0.099	U(0.013,0.29)
Grains and cereals	0.0093	U(0.0024,0.057)	0.019[a]	U(0.0048,0.11)[a]
Forage	0	None	0	None
Fruits	0.018	U(0.007-0.059)	0.024	U(0.0,0.11)
Potatoes	0.1	U(0.05,0.2)	0.2[a]	U(0.1,0.4)[a]
Silage	0	None	0	None

[a] Hg^{2+} values multiplied by 2

Technical Basis:

Mosbaek (1988) convincingly showed that for leafy, above-ground parts of plants virtually all of the mercury uptake was from air; therefore, for leafy vegetables, forage and silage no root uptake

was modeled.

Values in Cappon (1987) and Cappon (1981) were the only data located which measured methylmercury concentrations in plants, and methylmercury plant-soil BCF's were determined for rooting vegetables, fruiting vegetables, and legumes. Values were determined for crops grown on compost (Cappon 1987) and sludge-treated soils (Cappon 1981), and those values considering edible portions of plants are shown in Table A-4.

Table A-4
Soil-to-Plant Transfer Coefficients for Mercury
(from Cappon, 1987 and Cappon, 1981)

Crop	1987 Values		1981 Values	
	Hg^{2+}	Methylmercury	Hg^{2+}	Methylmercury
	Rooting Vegetables			
Beet	.055	.227	.017	.11
Carrot	.026	.118	.014	.048
Onion, Yellow	.073	.288	.053	.042
Onion, Spanish	-	-	.047	.030
Red Radish	.056	.092	.018	.066
White Radish	-	-	.011	.060
Turnip	.026	.013	-	-
	Fruiting Vegetables			
Cucumber, slicing	-	-	.015	0
Cucumber, pickle	.007	0	.015	.006
Pepper	.019	.022	.016	.042
Zucchini	.021	0	.014	.018
Summer Squash	-	-	.007	0
Acorn Squash	-	-	.016	.012
Spaghetti Squash	-	-	.016	.024
Pumpkin	-	-	.008	.006
Tomato	.059	.105	.020	.072
	Legumes			
Green Bush Beans	.011	0	.014	.020
Yellow Bush Beans	-	-	.017	.015
Lima Beans	-	-	.017	.090

It has been shown, however, that mercury taken up into plants from the environment can be transformed into other mercury species, especially to organomercuric forms such as methylmercury (Fortmann et al., 1978). The methylmercury in plants, therefore, may not have been directly absorbed from the environment. For the purposes of this study, considering root uptake, methylmercury concentrations in plants were treated as though they originated from the soil. It is also important to

note that air-to-plant transfer may have occurred, but the Cappon (1981, 1987) study was not designed to measure air-uptake.

Table A-5 shows additional soil-to-plant transfer coefficients for Hg^{2+} species (it was assumed that all the mercury in the soil is Hg^{2+}, which at worst would result in an error of a few percent in the Hg^{2+} soil-to-plant transfer coefficients) determined from a number of studies. Temple and Linzon (1977) sampled garden produce in the vicinity of a chlor-alkali plant. Lenka et al. (1992) also measured mercury concentrations in soil and plants near a chlor-alkali plant. Somu et al. (1985) determined mercury uptake in wheat and beans grown on $HgCl_2$ contaminated soil. John (1972) determined mercury concentrations in plants grown on soil artificially contaminated with $HgCl_2$. Wiersma et al. (1986) measured soil and plant total mercury concentrations from major growing areas in the Netherlands. Belcher and Travis (1989) compiled data from EPA (1985). Mosbaek (1988) studied plant concentrations from soil and air uptake under background conditions. For studies reporting wet weight plant concentrations, wet weight to dry weight conversion factors in Baes et al. (1984) were used to convert to dry weight based concentrations.

Table A-5
Other Values for Soil-to-Plant Transfer Coefficients for Hg^{2+}

Crop	Values	References
Legume vegetables	0.157-1.79, 0.00026-0.0003, 0.0005, 0.003-0.03	Lenka et al. (1992), Somu et al. (1985), John (1972), Belcher and Travis (1989).
Fruiting vegetables	0.013-0.33, 0.127-1.36, 0.0078-0.028	Temple and Linzon (1977), Lenka et al. (1992), Belcher and Travis (1989).
Rooting vegetables	0.09-0.33, 0.090-0.149, 0.0065-0.013, 0.05-0.2, 1.6-1.9	Temple and Linzon (1977), Lenka et al. (1992), John (1972), Belcher and Travis (1989), Mosbaek (1988)
Grains and cereals	0.0024-0.0093, 0.0033, 0.00038-0.057	Somu et al. (1985), John (1972), Belcher and Travis (1989).
Fruits	0.0078-0.028	Belcher and Travis (1989).
Potatoes	0.05-0.2	Belcher and Travis (1989).

When possible, default values were chosen based on experiments under reasonable or background conditions, as opposed to experiments where the soil was "spiked" with large amounts of mercury or measurements were taken from severely polluted areas. This is actually a conservative approach; although plants from mercury polluted areas will have greater contaminate levels, the efficiency of accumulation (quantified in the transfer coefficients) tends to decrease with increasing contaminate concentrations. Values from Cappon (1987) and Cappon (1981) were used when possible, since these experiments were conducted under reasonable garden conditions, edible portions of plants were analyzed separately, and different mercury species were measured. Cappon (1981) analyzed plants grown in control soil (total mercury soil content of 120 ng/g with 4.2% methylmercury) in

addition to the sludged soil (330 ng/g with 5.1% methylmercury, which is comparable to the 1987 soil levels of 430 ng/g with 5.3% methylmercury). The control soil data were not used since the methylmercury levels were often undetectable. Note that the compost and sludge-amended soils, although elevated in mercury, are nonetheless at reasonable concentrations. For fruiting vegetables, rooting vegetables and legumes values from Cappon (1987) and values derived from the edible potions of plants grown on sludged soil from Cappon (1981) were pooled and averaged; the results were used as the defaults for these plant types.

Default Hg^{2+} values for grains and cereals are from Somu (1985); the methylmercury values were assumed to be twice as great in accordance with the overall average trend noted in plants from the pooled Cappon data. The default values for fruits were assumed to be the same as for fruiting vegetables. The default Hg^{2+} value for potatoes was taken from Belcher and Travis (1989); the methylmercury value for potatoes was assumed to be twice the Hg^{2+} value.

A.2.2.2 Air-Plant BCF

Parameter: BI

Definition: The ratio of the contaminant concentration in plants (based on dry weight) to that in the air.

Units: Unitless

Crop	Hg^{2+a}		Methylmercury[a]	
	Default Value	Distribution	Default Value	Distribution
Leafy vegetables	18000	U[12000,24000]	5000	U[3300,6800]
Legume vegetables	1050	U[700,1400]	100	U[65,130]
Fruiting vegetables	22000	U[14000,29000]	1200	U[780,1600]
Rooting vegetables	0	NA	0	NA
Grains and cereals	1050	U[700,1400]	100	U[65,130]
Forage	18000	U[12000,24000]	5000	U[3300,6800]
Fruits	22000	U[14000,29000]	1200	U[780,1600]
Potatoes	0	NA	0	NA
Silage	18000	U[12000,24000]	5000	U[3300,6800]

[a] Based on elemental mercury air concentration, and speciation of divalent and methylmercury species based on Cappon (1981,1987).

Technical Basis:

Mosbaek (1988) determined that mercury concentration in the above-ground, leafy parts of plants is almost entirely the result of air-to-plant transfer of mercury. Cappon (1987,1981), however, found only divalent and methylmercury in these types of plants. Fitzgerald (1986) noted that up to 99% of the total airborne mercury is Hg^0 vapor (Fitzgerald, 1986). It was assumed that any

atmospheric elemental mercury taken up by the plant is converted into Hg^{2+} and methylmercury in the plant tissue. This is not unreasonable: it has been shown that mercury taken up into plants from the environment can be transformed into other mercury species (Fortmann et al., 1978).

A strong correlation between mercury soil concentration and concentration in rooting vegetables has been established (John, 1972; Lenka et al., 1992; Lindberg et al., 1979), and the Mosbaek study (1988) demonstrated that much of the mercury in rooting vegetables was from the soil. As a result, air-to-plant uptake of mercury was not modeled for rooting vegetables and potatoes.

For grains, fruits, legumes and fruiting vegetables, little correlation between mercury plant concentrations and either air or soil concentrations has been found; however, non-negligible concentrations of mercury species in these plants are routinely observed. For this reason, both air-to-plant and soil-to-plant uptake was modeled for these plants. Using a conservative approach, the transfer factors for each accumulation pathway were calculated as if all of the mercury in the plant came only from that pathway. This has the effect of possibly double-counting the amount of mercury in the plant tissue. There is a great deal of uncertainty due to the lack of applicable data.

The range of air-plant bioconcentration factors based on Mosbaek et al. (1988) was found to be 15,000 - 31,000, based on total mercury concentration in the plant tissue. Mosbeak et al. (1988) determined average mercury concentrations due to air uptake in lettuce, radish tops, and grass. Concentrations were converted to dry weight according to Baes et al. (1984), and the overall range of air-plant bioconcentration factors based on total mercury in the plant tissue was found to be 15,000 - 31,000. Air to plant bioconcentration factors can be derived from other studies only indirectly (by making a reasonable estimate of the air concentration and assuming all the mercury in plant tissue comes from air), and the values arrived at for various plant species generally fall into the previous range. Due to the limited data, it was decided to use the midpoint of the Mosbeak et al. (1988) bioconcentration values (23,000) as the starting default for all plant species assumed to accumulate mercury from the air.

This approach was adjusted for the consideration of portions of grains and legumes that are not directly exposed to the atmosphere. Although atmospherically absorbed mercury can translocate throughout different portions of the plant, data indicate internal portions of grains and legumes (the edible portions) do not appear to accumulate mercury to the same degree as plant leaves or vines. Somu et al. (1985), John (1972), and Cappon (1981) determined mercury concentrations from different portions of the same plants. Table A-6 below shows the relative concentrations of total mercury found in plant parts from the portions of these studies representative of noncontaminated conditions.

Table A-6
Relative Concentration of Mercury in Different Parts of Edible Plants

Legumes	Beans (Somu et al. 1985)	Peas (John 1972)	Beans (Cappon 1981)
vines		1.0	
stalks	1.0		
pods		0.045	1.0
seeds	0.060	0.0091	0.028 - 0.089
Grains	**Wheat (Somu et al. 1985)**	**Oats (John 1972)**	
leaves		1.0	
stalks	1.0	0.063	
husks		0.61	
grain	0.14	0.051	

A clear trend of decreasing mercury concentrations is seen proceeding from leafy to seed portions of the plants. Based on these data, it was decided to decrease the default air-to-plant biconcentration factor of 23,000 by a factor of 20 (to 1200) to account for the decreasing accumulation of airborne mercury for the edible portions of these plants as compared to the leafy portions (for which the biconcentration factor of 23,000 is applicable). Airborne mercury uptake by fruits may also be overestimated with the default bioconcentration factor. However, no data are available to explore this possibility.

The product of the bioconcentration factors and the atmospheric mercury concentration is the total mercury in the plant tissue resulting from accumulation of airborne elemental mercury. Plant-specific speciation estimates from Cappon (1981,1987) were used to partition the total mercury bioconcentration factor (and corresponding range) in order to model the relative fractions of methylmercury and Hg^{2+} found in the plant; these are shown in Table A-7; note that the rest of the mercury was found to be divalent mercury.

Thus, for leafy, fruiting and legume vegetables, the default values for the bioconcentration of methylmercury based on the elemental mercury concentration in air were assumed to be 23,000 or 1200 multiplied by the average methylmercury percentages in Table A-6; the Hg^{2+} values were derived similarly (Hg^{2+} fraction x 23,000). The values for fruits were assumed to be the same as for fruiting vegetables. The values for forage and silage were assumed the same as for leafy vegetables, and the values for grains were assumed to be the same as for legumes (beans).

Table A-7
Mercury Speciation in Various Plants

Plant Type	% Methylmercury Cappon (1981)	% Methylmercury Cappon (1987)
	Leafy vegetables	
Head lettuce	8.8	21.4
Leaf lettuce	16.5	18
Spinach	19.8	23.1
Swiss chard, Fordhook	30.2	14.8
Swiss chard, Ruby Red	28.6	-
Broccoli[a]	33.1	17.8
Late Cabbage	28.8	-
Red Cabbage	22.4	-
Savoy King Cabbage	25.2	-
Jersey Wakefield Cabbage[a]	-	18
Cauliflower	21.2	-
Collards	22.8	-
Average	21.8	
	Legume vegetables	
Green Bush Beans	0	7.2
Yellow Bush Beans	-	4.3
Lima Beans	-	22.4
Average	8.5	
	Fruiting vegetables	
Cucumber, slicing	0	-
Cucumber, pickle	2.1	0
Pepper	12.5	6.1
Zucchini	6.7	0
Summer Squash	0	-
Acorn Squash	4.1	-
Spaghetti Squash	7.4	-
Pumpkin	4.0	-
Tomato	16.0	9.1
Average	5.2	

[a] These were classified as "cole" in Cappon (1987).

A.2.2.3 Animal BTF

Parameter: BAj

Definition: The equilibrium concentration of a pollutant in an animal divided by the average daily intake of the pollutant.

Units: day/kg DW

Livestock	Default Value (day/kg DW)	Distribution
beef	0.02	U(0.0008,0.04)
beef liver	0.05	U(0.02,0.1)
dairy	0.02	U(0.003,0.09)
pork	0.00013	U(0.00005,0.00026)
poultry	0.11	U(0.094,0.13)
eggs	0.11	U(0.094,0.13)
lamb	0.09	U(0.009,0.3)

Technical Basis:

Biotransfer factors measure pollutant transfer from the environment to animal tissues and products. They are defined as the ratio of pollutant concentration in animal tissue to the daily pollutant intake of an animal. The biotransfer factors for mercury to cattle tissues were estimated based on data found in Vreman et al. (1986), and biotransfer factors for mercury to lamb were based on data found in van der Veen and Vreman (1986).

The data collected from Vreman et al. (1986) and van der Veen and Vreman (1986) are not from single pollutant and single route ingestion studies; rather, the animals in these studies were generally dosed with elevated levels of several metals in a single wafer. This is not the ideal set of studies for assessing the transfer of mercury primarily from ingested grass and soil. These studies, however are multiple dose and long-term experiments which should provide data more representative of the desired equilibrium situation than a single, very large dose experiment.

In two experiments, Vreman et al. (1986) measured transfer of mercury from diet to tissues and milk of dairy cattle. In the first experiment 12 lactating cows/group were placed on pasture in 2 groups for 3 months. The control group was fed uncontaminated wafers and, based on mercury levels in the pasture grass, were estimated to ingest 0.2 mg mercury/day. The exposed group received wafers treated with a solution of mercury acetate, lead, cadmium and arsenic pentoxide; the daily mercury ingestion rate for the exposed group was 1.7 mg/day. During the experiment mercury levels in milk were measured. After three months on test, four cows/group were slaughtered, and mercury levels were measured in liver, kidney and muscle samples. In the second study, lactating cows were kept indoors and divided into 4 groups of 8 for up to 28 months. In addition to the control group, the diets of 3 other groups were supplemented with the following: wafers containing the same metals (1.7 mg mercury/day), sludge delivering dietary levels of 3.1 mg mercury/day, and sludge delivering dietary levels of 1.2mg mercury/day. Two cows from each group were slaughtered at study termination

(except for the group receiving 3.1 mg mercury/day from sludge in which only one cow was sacrificed). Mean milk mercury concentrations in the groups were reported, and mercury levels in the slaughtered cows were measured in liver, kidney and muscle samples.

Shown in Table A-8 are data from Vreman et al. (1986) that are relevant to deriving beef and dairy biotransfer factors. The tissue mercury concentrations presented are in wet weight.

Table A-8
Mercury Concentrations in Specific Beef Tissue
Media Per Test Group and Dose (from Vreman et al, 1986)

Test Group	Dose (mg mercury/day)	Mercury in Milk (ug/Kg WW[A])	Mercury in Muscle (ug/Kg WW[A])	Mercury in Liver (ug/Kg WW[A])
Pasture Control	0.2	2.3	3	7
Pasture Treated	1.7	0.9	4	10
Indoor Control	0.2	<0.5	2	3
Indoor Wafer	1.7	0.6	2	26
Indoor High-Level Sludge	3.1	2.4	1	14
Indoor Low-Level Sludge	1.2	1.3	2	9

[A] Wet weight

The data in Table A-8 can be easily converted into milk, beef and liver biotransfer factors by converting the tissue concentrations to dry weight and dividing the tissue concentrations by the daily intake of mercury (after converting the intake from mg/day to ug/day). The moisture content of the above tissues are reported in Baes et al. (1984): 0.87 for whole milk, 0.615 for beef and 0.70 for liver. The biotransfer factors derived are shown in Table A-9.

Table A-9
Animal Biotransfer Factors Derived from Vreman et al. (1986)

	Biotransfer Factor (day/kg DW)		
Test Group	Dairy	Beef	Beef Liver
Pasture Control	0.09	0.04	0.1
Pasture Treated	0.004	0.006	0.02
Indoor Control	0.02	0.03	0.05
Indoor Wafer	0.003	0.003	0.05
Indoor High-Level Sludge	0.006	0.0008	0.02
Indoor Low-Level Sludge	0.008	0.004	0.03

Using the number of animals sampled for each value in Table A-9, weighted averages for the Dairy, Beef and Beef Liver Biotransfer factors can be derived. These are chosen as the default values, with the ranges taken from Table A-9.

In a experiment very similar to Vreman et al. (1986), van der Veen and Vreman (1986) measured transfer of mercury from diet to tissues of 10 week old fattening lambs. Two groups of 8 lambs were placed on pasture for 3 months. The control group was fed uncontaminated feed concentrate and based on mercury levels in the pasture grass and uncontaminated feed were estimated to ingest <0.02 mg mercury/Kg dry feed-day. The exposed group received feed concentrate treated with a solution of mercury acetate, lead, cadmium and arsenic pentoxide; the daily mercury ingestion rate for the exposed group was 0.08 mg/Kg dry feed. Another four groups of 8 lambs were kept indoors and were fed hay and feed concentrate. A control group was fed uncontaminated feed concentrate, and were estimated to ingest <0.02 mg mercury/Kg dry feed-day. The 3 other groups were fed feed concentrate contaminated with, respectively, a soluble solution of the metals, harbor sludge and sewage sludge. Daily mercury ingestion rates for these groups ranged from 0.14 - 0.27 mg/Kg dry feed. After three months all lambs were slaughtered and mercury levels were measured in liver, kidney, brain and muscle samples.

Shown in Table A-10 are data from van der Veen and Vreman et al. (1986) and the biotransfer factors derived from these data.

Table A-10
Mercury Concentrations and resulting BTFs in Lamb Muscle Tissue
Per Test Group and Dose (from van der Veen and Vreman 1986)

Test Group	Dose (mg mercury/Kg dry feed-day)	Feed Amount (Kg DW/day)	mercury in Muscle (ug/Kg WW)	Muscle Dry %	BTF[A] (day/Kg DW)
Pasture Control	<0.02	1.36	1	32.3	0.2
Pasture Treated	0.08	1.36	3	32.8	0.08
Indoor Control	<0.02	1.3	2	30.5	0.3
Indoor Wafer	0.14	1.28	1	29.5	0.02
Indoor High-Level Sludge	0.27	1.39	1	30.5	0.009
Indoor Low-Level Sludge	0.17	1.38	1	29.1	0.02

[A] Biotransfer Factor (BTF)

To calculate the biotransfer factors listed from the data in Table A-10, the daily mercury intake was calculated from the mercury concentration in dry feed and daily intake of dry feed. van der Veen and Vreman (1986) reported the dry weight fractions of the muscle samples, and the mercury concentration in muscle was calculated on a dry weight basis. The biotransfer factor for each group of lambs was then determined. The average over all groups was chosen as the default value, with the ranges taken from Table A-10.

In U.S. EPA (1993b), uptake slopes were developed for a number of pollutants found in sludge including mercury. For pork and poultry, U.S. EPA (1993b) reviewed the literature on concentrations of metals in meat from studies in which livestock were fed known concentrations of the metals in feed. These values were used to obtain the default values (after converting wet-weight values to dry-weight).

A.2.2.4 Fish Bioaccumulation Factor

Parameter: Tier 3 Fish BAF (BAF_3)
Tier 4 Fish BAF (BAF_4)

Definition: The concentration of the methylmercury in fish divided by the concentration of total dissolved mercury in water .

Units: L/kg

Fish Type	Default Value (L/kg)	Percentiles (L/kg) 5th Percentile	Median	95th Percentile
Trophic Level 3 Fish	66,200	6,400	662,000	684,000
Trophic Level 4 Fish	335,000	22,700	335,000	4,700,000

Technical Basis:

For a more complete discussion, the reader is referred to Volume V of this Report. The methylmercury value is most important since virtually 100% of mercury in fish tissue is methylmercury.

The BAFs for methylmercury is defined as the ratio of the methylmercury concentration in fish flesh divided by the concentration of total dissolved mercury (organic plus inorganic forms) in the water column. As virtually 100% of the mercury in fish flesh is in the methyl form, the definition of the BAF used here is equivalent to the definition of a total mercury BAF as found in the literature. The BAF represents the accumulation of mercury in fish of a specific trophic level from both water intake and predation on contaminated organisms, the latter being the dominant pathway. In this report BAFs for methylmercury are estimated for trophic level 3 (forage fish) and trophic level 4 (piscivorous fish) designated as BAF_3 and BAF_4, respectively. The BAFs are intended to be representative of the random selection of a fish from a random lake in a random geographical location.

The BAFs were estimated by probabilistic Monte Carlo simulation methods as described in Appendix A to Volume V of this report. Distributions were constructed from a limited number of available studies for BAF_3 and a predator-prey factor for trophic level 4 (PPF_4). PPF_4 represents the bioaccumulation of mercury for piscivorous trophic level 4 fish feeding on trophic level 3 fish. BAF_4 is the product of BAF_3 and PPF_4. Five studies were available for the estimation of BAF_3 with values ranging from 10,000 to 350,000. PPF_4 is based on 12 studies with values ranging from 1.2 to 15.5. A sensitivity analysis shows that BAF_3 has the greatest effect on the variance of the BAF_4 output, contributing 75% of the variance. A major source of variability in the BAF estimates is the dependence of PPF_4 and to some extent, BAF_3, on the age (and consequently the size) of the fish. Because fish accumulate mercury throughout their lives, the predator-prey and bioaccumulation factors increase with age, particularly for trophic level 4 fish. There is uncertainty as to whether a single BAF value is appropriate for derivation of water concentration when the fish-size range of the fish-consuming populations is known. For example, kingfishers feed on smaller fish while human recreational anglers primarily consume large fish. Because of the large variance in the BAF distributions and the lack of distinction between uncertainty and variability, the current

recommendation is to apply BAFs derived from valid data collected at the site of concern. Otherwise, it is recommended that the mean values of the BAF distributions, rather than upper or lower percentiles, be used for exposure assessment.

A.2.2.5 Plant Surface Loss Coefficient

Parameter: kp

Definition: A measure of the loss of contaminants deposited on plant surfaces over time as a result of environmental processes.

Units: /yr

Chemical	Default Value (per year)	Distribution	Range
Hg^0	40.41	Log(40.41,17.39)	28.11 - 52.7
Hg^{2+}	40.41	Log(40.41,17.39)	28.11-52.7
Methylmercury	40.41	Log(40.41,17.39)	28.11-52.7

Technical Basis:

The values in the previous table were taken from Belcher and Travis (1989), although no speciation was provided. The values for all species were assumed to be the same. The default value is the mean of the lognormal distribution used in Belcher and Travis (1989). The choice of a lognormal distribution was based on the work of Miller and Hoffman (1983).

A.2.2.6 Fraction of Wet Deposition Adhering

Parameter: Fw

Definition: Fraction of wet deposition that adheres to plant (i.e., is not washed off).

Units: unitless

Default Value	Distribution	Range
0.6	T(0.1,0.6,0.8)	0.1-0.8

Technical Basis:

The unitless parameter *Fw* represents the fraction of the pollutant in wet deposition that adheres to the plant, is not washed off by precipitation and is used to estimate plant pollutant levels. A value of 1 is the most conservative; this implies that all of the pollutant which deposits onto the plant via wet deposition will adhere to the plant. U.S. EPA (1990) originally used a value of 0.02, which significantly diminishes the impact of this pathway. A more recent study by Hoffman et al.

(1992) suggests an answer between these extremes for both dissolved pollutants and suspended particulates in simulated rain drops.

Hoffman et al. (1992) attempted to quantify the amount of radiolabeled beryllium (Be) and Iodine (I) as well as particles of sizes 3, 9, and 25 μm that adhered to three plant types (fescue, clover, and a typical weeded plot). The radiolabeled pollutants were dissolved or suspended in water, which was then showered upon the different types of vegetation to simulate precipitation. Two precipitation intensities were modeled in the experiment: moderate (1-4 cm/hour) and high (4-12 cm/hour). Due to experimental complications, total deposition and pollutant retention upon the vegetation were estimated by the authors; these estimates were termed the interception fraction in the Hoffman report. For example, in the experiment. Beryllium in the form of $BeCl_2$ was dissolved in the water and then showered upon the vegetation. For the moderate and high intensity precipitation events simulated, the mean interception fractions were estimated to be 0.28 and 0.15, respectively.

The 1993 Addendum to the Indirection Exposure Methodology (U.S. EPA, 1993a) models deposition and retention as the product of the interception fraction (Rp_i) and *Fw*. In terms of the U.S. EPA model, the Hoffman report estimates the product $Rp_i x Fw$. To obtain estimates for *Fw*, the values reported in Hoffman et al. (1992) were divided by the interception fraction for forage used in this assessment (0.47; Baes et al., 1984). This provides estimates of 0.60 and 0.32 for *Fw* for the moderate and high precipitation intensities, respectively (see Table A-11).

Table A-10 shows the Hoffman et al. (1992) estimates for the interception and adhesion of dissolved pollutants and suspended particles in simulated moderate and high intensity precipitation. Based on the Hoffman estimates and the assumption of an interception fraction for forage of 0.47, the *Fw* for the two pollutants and three particle sizes were estimated for the precipitation intensities studied, and the means were calculated. No attempt has been made to adjust the final estimate for frequency of the two precipitation intensities; however, since moderate precipitation intensities are more common, the unadjusted means are probably an underestimate.

Table A-11
Values From Hoffman et al. (1992)
and the Values of *Fw* Estimated Using Those Values

Compound	Rp_i x Fw for Moderate Intensity	Rp_i x Fw for High Intensity	Fw Estimate for Moderate Intensity	Fw Estimate for High Intensity	Fw Mean
I	0.08	0.05	0.17	0.11	0.14
Beryllium	0.28	0.15	0.60	0.32	0.46
3 μm	0.30	0.24	0.64	0.51	0.58
9 μm	0.33	0.26	0.70	0.55	0.63
25 μm	0.37	0.31	0.79	0.66	0.72

The *Fw* estimated for beryllium was used as a surrogate for mercury. Be^{2+}, as a cation, is assumed to behave in a manner similar to Hg^{2+} during deposition. Because the moderate intensity is expected to be more common than the heavy intensity, an *Fw* of 0.60 is assumed to be a reasonable estimate of *Fw* for divalent mercury. This value is higher than the range of 0.1-0.3 presented in

McKone and Ryan (1989). For beryllium, Hoffman noted the appearance of a strong attraction between the cation and the plant surface, which was assumed to be negatively charged. Beryllium is believed to adsorb to cation exchange sites in the leaf cuticle. Once dried on the plant surface, beryllium was not easily removed by subsequent precipitation events. Divalent mercury is assumed to exhibit a similar behavior. The range of 0.1-0.8 was used to estimate the sensitivity of this parameter.

The adjusted Hoffman data indicate that the greater the intensity of the precipitation, the smaller the *Fw* estimate for both dissolved pollutants and suspended particles. This is intuitively appealing given the understanding of the physical process. Hoffman et al. (1992) noted that the intensity and amount of rainfall had approximately the same impact on the estimated values. It should also be noted that the data indicate that the value of *Fw* for pollutants that deposit as anions (e.g., I) may be significantly lower than cations.

A.3 References

Akagi H., D.C. Mortimer, and D.R. Miller (1979). Mercury Methylation and Partition in Aquatic Systems. *Bull. Environ. Contam. Toxicol.* 23: 372-376.

Alberts, J.J., J.E. Schindler, and R.W. Miller. (1974). Elemental mercury evolution mediated by humic acid.

Albeuts, J.J., J.E. Schindler, and R.W. Miller (1974). Elemental Mercury Evolution Mediated by Humic Acid. *Science* 184: 895-897.

Anderson, A., N. Browne, S. Duletsky, J. Ramig, and T. Warn (1985). *Development of Statistical Distributions or Ranges of Standard Factors Used in Exposure Assessments.* EPA 600/8-85-101, U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, D.C.

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 under contract No. DE-AC05-84OR21400. U.S. Department of Energy, Washington, D.C.

Belcher, G.D. and C.C. Travis (1989). *Modeling Support for the Rura and Municipal Waste Combustion Projects: Final Report on Sensitivity and Uncertainty Analysis for the Terrestrial Food Chain Model.* Prepared for the U.S. EPA.

Bloom, N.S. and C.J. Watras (1989). Observations of Methylmercury in Precipitation. *The Sci. Tot. Environ.* 87/88: 199-207.

Bloom, N. and S.W. Effler. (1990). Water, Air, and Soil Poll. 53:251-265.

Bloom, N.S., C.J. Watras, and J.P. Hurley (1991). Impact of Acidification on the Methylmercury Cycle of Remote Seepage Lakes. *Water, Air, and Soil Poll.* 56: 477-491.

Bousset, C. (1981). The Mercury Cycle. *Water, Air and Soil Pollution* 16: 253-255.

Butcher, B., B. Davidoff, M.C. Amacher, C. Hinz, I.K. Iskandar, and H.M. Selim (1989). Correlation of Freundlich K_d and retention parameters with soils and elements. *Soil Science* 148: 370-379.

Calabrese, E.J. and E.J. Stanek, III (1991). A Guide to Interpreting Soil Ingestion Studies. II. Qualitative and Quantitative Evidence of Soil Ingestion. *Regul. Toxicol. Pharmacol.* 13:278-292.

Cappon, C.J. (1981). Mercury and Selenium Content and Chemical Form in Vegetable Crops Grown on Sludge-Amended Soil. *Arch. Environm. Contam. Toxicol.* 10: 673-689.

Cappon, C. (1984). Content and Chemical Form of mercury and selenium in soil, sludge and fertilizer materials. Water, Air, Soil Pollut 22:95-104.

Cappon, C.J. (1987). Uptake and Speciation of Mercury and Selenium in Vegetable CropsGrown on Compost-Treated Soil. *Water, Air, Soil Poll.* 34: 353-361.

Carsel, R.F., C.N. Smith, L.A. Mulkey, J.D. Dean, and P. Jowise (1984). User's Manual for the Pesticide Root Zone Model (PRZM) Release 1. U.S. EPA, Athens, GA. EPA-600/3-84-109.

Ershow, A.G. and K.P. Cantor (1989). *Total Water and Tapwater Intake in the United States: Population-based Estimates of Quantities and Sources.* Life Sciences Research Office, Federation of American Societies for Experimental Biology. Bethesda, Maryland.

Fitzgerald, W. 1986. Cycling of mercury between the atmosphere and oceans, in: The Role of Air-Sea Exchange in Geochemical Cycling, NATO Advanced Science Institutes Series, P. Buat-Menard (Ed.), D. Reidel publishers, Dordrecht, pp 363-408.

Fitzgerald, W.F., R.P. Mason, and G.M. Vandal (1991). Atmospheric Cycling and Air-Water Exchange of Mercury over Mid-Continental Lacustrine Regions. *Water, Air, and Soil Pollution* 56: 745-767.

Fortmann, L. C., D. D. Gay, and K. O. Wirtz (1978). Ethylmercury: Formation in Plant Tissues and Relation to Methylmercury Formation. U.S. EPA Ecological Research Series, EPA-600/3-78-037.

Fries, G.F. 1982. Potential Polychlorinated Biphenyl Residues in Animal Products from Application of Contaminated Sewage Sludge to Land. J. Environ. Quality. 11: 14-20.

Fries, G.F. (1987). Assessment of Potential Residues in Food Derived from Animals Exposed to TCDD-Contaminated Soil. *Chemosphere* 16: 2123-2128.

Gill, G. And K. Bruland (1990). Mercury Speciation in surface freshwaters systems in California and other areas. Sci. Total Environ. 24:1392.

Glass, G.E., J.A. Sorenson, K.W. Schmidt, and G.R. Rapp (1990). New Source Identification of Mercury Contamination in the Great Lakes. Environmental Science and Technology 24:1059-1069.

Hanford Environmental Dose Reconstruction Project (1992). *Parameters Used in Environmental Pathways (DESCARTES) and Radiological Dose (CIDER) Modules of the Hanford Environmental Dose Reconstruction Integrated Codes (HEDRICF) for the Air Pathway,* PNWD-2023 HEDR, Pacific Northwest Laboratories, September 1992.

Hattemer-Frey, H. and C.C. Travis (1989). An Overview of Food Chain Impacts from Municipal Waste Combustion. Prepared for U.S. EPA.

Hawley, J.K. (1985). Assessment of Health Risk from Exposure to Contaminated Soil. *Risk Anal.* **5**(4): 289-302.

Hildebrand, S.G., S.E. Lindberg, R.R. Turner, J.W. Huckabee (1980). *Biogeochemistry of Mercury in a River Reservoir System: Impact of an Inactive Chloralkali Plant on the Holston River.* ORNL/TM-6141.

Hildebrand, S.G., R.H. Strand, and J.W. Huckabee (1980). Mercury accumulation in fish and invertebrates of the North Fork Holston river, Virginia and Tennessee. J. Environ. Quan. 9:393-400.

Hoffman, F.O. and D.F. Baes (1979). *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionucleotides.* ORNL/NUREG/TM-882.

Hoffman, F.O., K.M. Thiessen, M.L. Frank and B.G. Blaylock (1992). Quantification of the interception and initial retention of radioactive contaminants deposited on pasture grass by simulated rain. *Atmospheric Environment,* 26A (18): 3313-3321.

Iverfeldt, A. and J. Persson (1985). The solvation thermodynamics of methylmercury (II) species derived from measurements of the heat of solvation and the Henry's Law constant. *Inorganic Chimica Acta,* 103: 113-119.

John, M.K. (1972). Mercury Uptake from Soil by Various Plant Species. *Bull. Environ. Contam. Toxicol.* 8(2): 77-80.

Knott, J.E. (1957). *Handbook for Vegetable Growers.* J. Wiley and Sons, Inc., New York.

LaGoy, L.K. (1987). Estimated Soil Ingestion Rates for Use in Risk Assessment. *Risk Anal.* 7(3): 355-359.

Lee, Y., and H. Hultburg (1990). Methylmercury in some Swedish Surface Waters. *Environ. Toxicol. Chem.* 9: 833-841.

Lenka, M., K. K. Panda, and B. B. Panda (1992) Monitoring and Assessment of Mercury Pollution in the Vicinity of a Chloralkali Plant. IV. Bioconcentration of Mercury in *In Situ* Aquatic and Terrestrial Plants at Ganjam, India. *Arch. Environ. Contam. Toxicol.* 22:195-202.

Lenka, M., K.K. Panda, and B.B. Panda (1992a). Monitoring and Assessment of Mercury Pollution in the Vicinity of a Chloralkali Plant. II. Plant Availability, Tissue Concentration and Genotoxicity of Mercury from Agricultural Soil Contaminated with Solid Waste Assessed in Barley. *Environ. Poll.* 0269-7491/92. pp. 33-42.

Lenka, M., K.K. Panda, and B.B. Panda (1992b). Monitoring and Assessment of Mercury Pollution in the Vicinity of a Chlor-alkali Plant. IV. Bioconcentration of Mercury in In Situ Aquatic and Terrestrial Plants at Ganjam, India. *Arch. Environ. Contam. Toxicol.* 22:195-202.

Lepow, M.L., L. Bruckman, M. Gillette, S. Markowitz, R. Robino, and J. Kapish (1975). Investigations into Sources of Lead in the Environment of Urban Children. *Environ. Res.* 10: 415-426.

Lindberg, S. E., D. R. Jackson, J. W. Huckabee, S. A. Janzen, M. J. Levin, and J. R. Lund (1979). Atmospheric Emission and Plant Uptake of Mercury from Agricultural Soils near the Almaden Mercury Mine. *J. Environ. Qual.* 8(4):572-578.

Lindquist, O. and H. Rodhe (1985). Atmospheric Mercury: A Review. *Tellus* 37B: 136-159.

Lindqvist, O., K. Johansson, M. Aastrup, A. Andersson, L. Bringmark, G. Hovsenius, L. Hakanson, A. Iverfeldt, M. Meili, and B. Timm (1991). Mercury in the Swedish Environment - Recent Research on Causes, Consequences and Corrective Methods. *Water, Air and Soil Poll.* 55:(all chapters)

Lyman, W.J., W.F. Rheel, and D.H. Rosenblatt (1982). *Handbook of Chemical Property Estimation Methods.* New York: McGraw-Hill.

McKone, T.E. and P.B. Ryan (1989). Human Exposure to Chemicals Through Food Chains: An Uncertainty Analysis. *Environ. Sci. Technol.* 23(9): 1154-1163.

Miller, C.W. and F.O. Hoffman (1981). An Examination of the Environmental Half-time for Radionuclides Deposited on Vegetation. *Health Phys.* 45:731-744.

Miller, C. And F. Hoffman. (1983). An examination of the environmental half-time for radionuclides deposited on vegetation. Health Physics 45:731.

Miskimmin, B.M. (1991). Effect of Natural Levels of Dissolved Organic Carbon (DOC) on Methylmercury Formation and Sediment-Water Partitioning. *Bull. Environ. Contam. Toxicol.* 47: 743-750.

Moore, J.W. and S. Ramamoorthy (1984). *Heavy Metal in Natural Waters - Applied Monitoring in Impact Assessment.* New York, Springer-Verlag.

Morgan, J., M. Berry and R. Graves. (1994) Effects of Native American Cooking Practices on Total Mercury Concentrations in Walleye. Abstract presented at ISEE/ISEA Joint Conference. September 18-21, 1994.

Mosbaek, H., J. C. Tjell, and T. Sevel (1988). Plant Uptake of Mercury in Background Areas. *Chemosphere* 17(6):1227-1236.

NAS (National Academy of Science) (1977). Safe Drinking Water Committee. An Assessment of Mercury in the Environment. National Research Council.

NAS (National Academy of Sciences). 1987. Predicting Feed Intake of Food-Producing Animals. National Research Council, Committee on Animal Nutrition, Washington, DC.

Nriagu, J.O. (1979). *The Biogeochemistry of Mercury in the Environment.* Elsevier/North Holland. Biomedical Press: New York.

Parks, J.W., A. Luitz, and J.A. Sutton (1989). Water Column Methylmercury in the Wabigoon/English River-lake System: Factors Controlling Concentrations, Speciation, and Net Production. *Can. J. Fish. Aquat. Sci.* 46: 2184-2202.

Pierce, R., D. Noviello and S. Rogers. 1981. Commencement Bay Seafood Consumption Report. Preliminary Report. Tacoma, WA: Tacoma-Pierce County Health Department. As cited in U.S. EPA (1991). Exposure Factors Handbook. EPA/600/8-89/043.

Puffer, H. 1981. Consumption rates of potentially hazardous marine fish caught in the metropolitan Los Angeles area. EPA Grant #R807 120010.As cited in U.S. EPA (1991). Exposure Factors Handbook. EPA/600/8-89/043.

Robinson, K.G. and M.S. Shuman (1989). Determinination of mercury in surface waters using an optimized cold vapor spectrophotometric technique. *International Journal of Environmental Chemistry* 36: 111-123.

Roels, H.A. , J.P. Buchet, and R.R. Lauwerys (1980). Exposure to Lead by the Oral and Pulmonary Routes of Children Living in the Vicinity of a Primary Lead Smelter. *Environ. Res.* 22: 81-94.

Roseberry, A.M. and D.E. Burmaster (1992). Lognormal Distributions for Water Intake by Children and Adults. *Risk Analysis* 12: 65-72.

Roy, M. and C. Courtay (1991). Daily Activities and Breathing Parameters for Use in Respiratory Tract Dosimetry. *Radiation Protection Dosimetry* 35 (3): 179-186.

Rutledge, A.D. (1979). "Vegetable Garden Guide." Publication 447 (Revised) University of Tennessee Agricultural Extension Service, The University of Tennessee.

Sanemasa, I. (1975). Solubility of Elemental Mercury Vapor in Water. *Bull. Chem. Soc. Jpn.* 48(6): 1795-1798.

Schroeder, W., G. Yarwood and H. Niki (1991). Transformation Processes involving Mercury Species in the Atmosphere-Results from a Literature Survey. *Water, Air and Soil Pollution* 56: 653-666.

Shor, R.W. , C.F. Baes , and R.D. Sharp (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.* Oak Ridge National Laboratory, ORNL-5786.

South Coast Air Quality Management District (SCAQMD) 1988. Multi-pathway Health Risk Assessment Input parameters. Guidance Document.

Sumo, E., B.R. Singh, A.R. Selmer-Olsen, and K. Steenburg (1985). Uptake of ^{203}Hg-labeled Mercury compounds by Wheat and Beans Grown on an oxisol. *Plant and Soil* 85: 347-355.

Technical Assessment Systems (TAS) (1991). Distribution of Intake of Foods in Sixteen Food Groupings: grains and cereals; potatoes including sweet potatoes and yams; leafy vegetables, excluding brassica; brassica vegetables; legume vegetables; fruiting vegetables, including

cucurbits; root vegetables; other vegetables; beef excluding liver; beefliver; sheep; pork; poultry; eggs; milk; and fruits, excluding cucurbits. Prepared for U.S. EPA.

Temple, P. J. and S. N. Linzon (1977). Contamination of Vegetation, Soil, Snow and Garden Crops by Atmospheric Deposition of Mercury from a Chlor-Alkali Plant, in (1977) D. D. Hemphill [ed] Trace Substances in Environmental Health - XI, Univ Missouri, Columbia. p. 389-398.

Turner, R. (1994). Personal communication with Ralph Turner, Environmental Sciences Division, Oak Ridge National Laboratory.

U.S. Department of Agriculture (USDA) (1978). *Handbook No. 537: Predicting Rainfall Erosion Losses.* U.S. Government Printing Office, Washington, D.C.

U.S. Department of Commerce. (1977). 1974 Census of Agriculture. Bureau of the Census, Agriculture Division, Washington, D.C.

U.S. EPA (1984). *Health Assessment Document for Mercury.* Prepared by the Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, Cincinnati, OH for the Office of Emergency and Remedial Response, Washington, D.C.

U.S. EPA (1985). *Environmental profiles and hazard indices for constituents of municipal sludge: Mercury.* Washington, D.C.: Office of Water Regulations and Standards.

U.S. EPA (1989). *Exposure Factors Handbook.* Office of Health and Environmental Assessment, Exposure Assessment Group, Washington, D.C. EPA 600/8-89-043. NTIS PB90-106774.

U.S. EPA (1990). *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Office of Health and Environmental Assessment, Washington, D.C. EPA/600/6-90/003.

U.S. EPA (1993a). *Assessment of Mercury Occurence in Pristine Freshwater Ecosystems*, prepared by ABT Associates, Inc. Draft as of September 1993.

U.S. EPA. 1993b. Technical Support document or Land Application of Sewage Sludge. Federal Register. 40 CFR Part 257 et al. Standards for the Use or Disposal of Sewage sludge; Final Rules. February 19.

van der Veen, N.G. and K. Vreman (1986). Transfer of cadmium, lead, mercury and arsenic from feed into various organs and tissues of fattening lambs. *Netherlands Journal of Agricultural Science* 34: 145-153.

Vreman, K., N.J. van der Veen, E.J. van der Molen and W.G. de Ruig (1986). Transfer of cadmium, lead, mercury and arsenic from feed into milk and various tissues of dairy cows: chemical and pathological data. *Netherlands Journal of Agricultural Science* 34: 129-144.

Watras, C.J. and N.S. Bloom (1992). Mercury and Methylmercury in Individual Zooplankton: Implications for Bioaccumulation. *Limnol. Oceanogr.*37(6): 1313-1318.

Wiessman, D., B.J. vanGoor, amd N.G. van der Veen (1986). Cadmium, Lead, Mercury, and Arsenic Concentrations in Coops amd Corresponding Soils in the Netherlands. *J. Agric. Food Chem.* 34: 1067-1074.

Wiersma, D., B. J. van Goor, and N. G. van der Veen (1986). Cadmium, Lead, Mercury, and Arsenic Concentrations in Crops and Corresponding Soils in the Netherlands. *J. Agric. Food Chem.*, 34:1067-1074.

Wilken, R.D. and H. Hintelmann (1991). Mercury and Methylmercury in Sediments and Suspended Particles from the River Elbe, North Germany. *Water, Air, and Soil Poll.* 56: 427-437.

APPENDIX B

PARAMETER JUSTIFICATIONS
SCENARIO-DEPENDENT PARAMETERS

TABLE OF CONTENTS

			Page
B.	SCENARIO DEPENDENT PARAMETERS		B-1
	B.1	Time of Concentration	B-1
	B.2	Average Air Temperature	B-2
	B.3	Watershed Area	B-2
	B.4	Average Annual Precipitation	B-3
	B.5	Average Annual Irrigation	B-3
	B.6	Average Annual Runoff	B-4
	B.7	Average Annual Evapotranspiration	B-4
	B.8	Wind Speed	B-5
	B.9	Soil Density	B-5
	B.10	Mixing Depth in Watershed Area	B-6
	B.11	Mixing Depth for Soil Tillage	B-6
	B.12	Soil Volumetric Water Content	B-7
	B.13	Soil Erosivity Factor	B-8
	B.14	Soil Erodability Factor	B-8
	B.15	Topographic Factor	B-9
	B.16	Cover Management Factor	B-10
	B.17	Sediment Delivery Ratio to Water Body	B-11
	B.18	Pollutant Enrichment Factor	B-11
	B.19	Water Body Surface Area	B-12
	B.20	Water Body Volume	B-12
	B.21	Long-Term Dilution Flow	B-13
	B.22	Suspended Solids Deposition Rate	B-14
	B.23	Benthic Sediment Concentration	B-14
	B.24	Upper Benthic Sediment Depth	B-15
	B.25	Aquatic Plant Biomass	B-15
	B.26	Total Fish Biomass	B-16
	B.27	References	B-16

LIST OF TABLES

		Page
B-1	Water Content Per Soil Type	B-7
B-2	Representative Soil Types For Each Site	B-8
B-3	Cover Factor Values of Undisturbed Forest Land (from WQAM, 1985; original citation Wischmeier and Smith, 1978)	B-10
B-4	Long-Term Dilution Flow in In/Yr	B-13

DISTRIBUTION NOTATION

A comprehensive uncertainty analysis was not conducted as part of this study. Initially, preliminary parameter probability distributions were developed. These are listed in Appendices A and B. These parameter probability distributions were not utilized to generate quantitative exposure estimates. They are provided as a matter of interest for the reader.

Unless noted otherwise in the text, distribution notations are presented as follows.

Distribution	Description
Log (A,B)	Lognormal distribution with mean A and standard deviation B
Log* (A,B)	Lognormal distribution, but A and B are mean and standard deviation of underlying normal distribution.
Norm (A,B)	Normal distribution with mean A and standard deviation B
U (A,B)	Uniform distribution over the range (A,B)
T (A,B,C)	Triangular distribution over the range (A,C) with mode of B

B. SCENARIO DEPENDENT PARAMETERS

This appendix describes the scenario dependent parameters used in the exposure modelling for the Mercury Study Report to Congress. Scenario dependent parameters are variables whose values are dependent on a particular site and may differ among various site-specific situations. For this assessment, three settings are being evaluated: (1) rural, (2) lacustrine, and (3) urban. The receptors differ for each of these scenarios, as do the parameters. These scenario dependent parameters may be either chemical independent or chemical dependent. The following sections present the chemical independent and chemical dependent parameters used in this assessment.

Chemical independent parameters are variables that remain constant despite the specific contaminant being evaluated. The chemical independent variables used in this assessment are described in the following sections.

Site physical data include information such as the environmental setting, vegetative cover, presence of surface water or groundwater, area of source and meteorological and climatological data. These parameters are described in the following sections.

B.1 Time of Concentration

Parameter: Tc

Definition: Number of years that the air concentration at the above level persists; equal to the facility lifetime for calculations from anthropogenic sources

Units: yrs

Scenario	Default Value(s) (years)	Distribution
All	30	None

Technical Basis:

The time of concentration is the same as the assumed facility lifetime. The generic value is 30 years. It is noted that this assumption is made only for estimation of soil concentrations. The water concentrations are calculated assuming steady-state has been attained, with the flux due to runoff/erosion based on the 30-year soil concentrations.

B.2 Average Air Temperature

Parameter: Ta

Definition: Average air temperature of microscale area

Units: °C

Location	Default Value (Years value is based upon) (°C)	Distribution
Eastern Location	11.9 (25)	U (8,16)
Western Location	13.4 (47)	U (9,17)

Technical Basis:

The values for local airports are reported in the section "U.S. Local Climatological Data Summaries for 288 Primary Stations throughout the U.S." on CDROM by WeatherDisc Associates (1992). The distributions are arbitrary to explore the sensitivity of this parameter.

B.3 Watershed Area

Parameter: WAI

Definition: Area of contamination which drains into a water body

Units: Km^2

Location	Default Value (Km^2)
Eastern Location	37.3
Western Location	37.3

Technical Basis:

The values for the fish ingestion pathways are based on hypothetical watershed/waterbody surface area ratio of 15 and a lake diameter of 1.78 km. This parameter was used only to calculate the erosion and runoff load to the water body.

B.4 Average Annual Precipitation

Parameter: P

Definition: Average annual precipitation

Units: cm/yr

Location	Default Value (cm/yr)	Distribution
Eastern Location	102	T(82,102,122)
Western Location	21	T(1,21,41)

Technical Basis:

All values are for local airports as reported in the section "U.S. Local Climatological Data Summaries for 288 Primary Stations throughout the U.S." on CDRom by WeatherDisc Associates (1992). These were considered the "best estimates" of a triangular distribution, with a range of 20 in/yr above and below the mode.

B.5 Average Annual Irrigation

Parameter: I

Definition: Average annual irrigation of plants

Units: cm/yr

Location	Default Value (cm/yr)	Distribution
Eastern Location	12.5	U(0,25)
Western Location	57.5	U(50,65)

Technical Basis:

The ranges were approximated from Figure 4.25 in Baes et al. (1984). The tentative default values are the midpoint of this range. It was assumed that both the farmer and home gardener will irrigate the same amount if they are in the same area of the country (i.e., irrigation rate does not depend on size of plot).

B.6 Average Annual Runoff

Parameter: Ro

Definition: Average annual runoff

Units: cm/yr

Location	Default Value (cm/yr)	Distribution
Eastern Location	18	U(9,27)
Western Location	1	U(0,2)

Technical Basis:

The default values for the eastern location are from Geraghty et al. (1973). The total runoff values given in that report include groundwater recharge, direct runoff, and shallow interflow. Following U.S. EPA (1993), this number was reduced by one-half to represent surface runoff. Because of the difficulty of hydrologic modelling in the western location, the PRZM-2 model (Carsel, 1984) was used to estimate the runoff for this area. The estimated value was 1 cm/yr. The distributions are arbitrary to determine the sensitivity of this parameter.

B.7 Average Annual Evapotranspiration

Parameter: Ev

Definition: Average annual loss of water due to evaporation

Units: cm/yr

Location	Default Value (cm/yr)	Distribution
Eastern Location	65	U(60,70)
Western Location	13	U(8,18)

Technical Basis:

For the eastern location, the ranges are based on estimates from isopleths given in Figure 4.24 in Baes et al. (1984). The values presented there were estimated based on local data (average temperature and precipitation) as well as the maximum possible sunshine for the area. The default value is the midpoint of this range. For the western location, the model PRZM-2 was used to estimate the values given previously.

B.8 Wind Speed

Parameter: W

Definition: Wind speed

Units: m/s

Location	Default Value (m/s)	Distribution
Eastern Location	4.3	U(1,7)
Western Location	4.0	U(1,7)

Technical Basis:

All values were collected for local airports and reported in the section "U.S. Local Climatological Data Summaries for 288 Primary Stations throughout the U.S." on CDROM by WeatherDisc Associates (1992). The primary use of this parameter is for estimating volatilization from soil and water bodies. The distributions are arbitrary to explore the sensitivity of this parameter.

B.9 Soil Density

Parameter: BD

Definition: Soil density

Units: g/cm^3

Location	Default Value (g/cm^3)	Distribution	Range
All Sites	1.4	Log(1.4,0.15)	0.93-1.84

Technical Basis:

The distribution is from Belcher and Travis (1989) and is based on a probability plot using data from Hoffman and Baes (1979). There is little variation in the parameter, despite the fact that more than 200 data points were used. The default value is the mean of the distribution.

B.10 Mixing Depth in Watershed Area

Parameter: Zd

Definition: The depth that contaminants are incorporated into soil (no tillage)

Units: cm

Location	Default Value (cm)	Distribution
All Sites	1.0	U(0.5,5)

Technical Basis:

The default value is based on U.S. EPA (1990). The distribution is arbitrary to determine the relative sensitivity of the parameter.

B.11 Mixing Depth for Soil Tillage

Parameter: Ztill

Definition: The depth that contaminants are incorporated into tilled soil

Units: cm

Location	Default Value (cm)	Distribution
All Sites	20	U(10,30)

Technical Basis:

The default value is based on U.S. EPA (1990). The distribution is arbitrary to determine the sensitivity of this parameter.

B.12 Soil Volumetric Water Content

Parameter: Theta,O

Definition: Amount of water that a given volume of soil can hold

Units: ml/cm^3

Location	Default Value (ml/cm^3)	Distribution
Eastern Location	0.30	U(0.15,0.42)
Western Location	0.36	U(0.15,0.42)

Technical Basis:

Values for water content can range from 0.003 to 0.40 ml/cm^3 depending on the type of soil (Hoffman and Baes, 1979). Table B-1 demonstrates the dependency of values on the hydrologic soil type. These values were derived from the PATRIOT software system (Imhoff et al., 1994), which can be obtained from the Center for Exposure Assessment Modeling at the U.S. Environmental Protection Agency, Athens, Georgia.

Table B-1
Water Content Per Soil Type

Soil Type	Water Content
A	0.15
B	0.22
C	0.30
D	0.42

Representative soil types for both sites are shown in Table B-2 and were determined from Carsel (1984). The soil types were used in conjunction with the previous table to determine the default value for the soil water content, with the value for the western location being the average of the values for types C and D.

Table B-2
Representative Soil Types For Each Site

Location	Soil Type
Eastern Location	C
Western Location	C/D

The distribution for all sites is a uniform distribution over the range over all soil types.

B.13 Soil Erosivity Factor

Parameter: R

Definition: Quantifies local rainfall's ability to cause erosion

Units: kg/km^2-yr

Location	Default Value (kg/km^2-yr)	Distribution
Eastern Location	200	U(100,300)
Western Location	53	U(30,75)

Technical Basis:

The ranges were determined based on an isopleth map for the region in USDA (1978). The upper and lower bounds were determined from this map by finding extremes within a 300-mile radius.

B.14 Soil Erodability Factor

Parameter: K

Definition: Quantifies soil's susceptibility to erosion

Units: tons/acre

Location	Default Value (tons/acre)	Distribution
Eastern Location	0.30	U(0.12,0.48)
Western Location	0.28	U(0.08,0.48)

Technical Basis:

Based on similar soil near the eastern location (loamy sand, loam, and silt loam) and using Table A2-2 in U.S. EPA (1989), a range of 0.12 to 0.48 was obtained. A similar analyses has not been performed for the other sites, but the ranges listed in the previous table are apparently the maximum range possible based on Table A2-2 in U.S. EPA (1990); therefore, these ranges encompass all likely values and can be used for sensitivity analyses. The default values are the midpoint of these ranges.

B.15 Topographic Factor

Parameter: LS

Definition: Provides a measure of the length and steepness of the land slope

Units: unitless

Location	Default Value	Distribution
Eastern Location	2.5	U(0.25,5)
Western Location	0.4	U(0.1,1.2)

Technical Basis:

The length and steepness of the land slope substantially affect the rate of soil erosion. Table A2-3 in U.S. EPA (1989) contains LS values for various slopes and slope lengths and was used in conjunction with United States Geological Survey (USGS) maps to obtain the ranges given in the previous table. A 1:24000 map was available for the humid/east/complex I site while only a 1:250000 USGS map was available for all other sites. The default value was chosen as representative of the most common slope and length in the area.

B.16 Cover Management Factor

Parameter: C

Definition: The ratio of soil loss from land cropped under local conditions to the corresponding loss from clean tilled fallow

Units: unitless

Location	Default Value
Eastern Location	0.006
Western Location	0.1

Technical Basis:

The lower end of the range for areas having forests (0.001) is the lower of two values suggested for woodlands in U.S. EPA (1988). For those areas lacking forests (i.e., western site), the value of 0.1 given for grass in U.S. EPA (1993) was used.

For the watershed, it was decided to use a cover fraction representative of undisturbed grass or forested areas, although high-end values were used. It was noted that the cover fraction can vary by several orders of magnitude, depending on the land use type and soil type. Table B-3 shows estimates of cover factor values for undisturbed forest land (Wischmeier and Smith, 1978).

Table B-3
Cover Factor Values of Undisturbed Forest Land
(from WQAM, 1985; original citation Wischmeier and Smith, 1978)

Percent of Area Covered by Canopy of Trees and Undergrowth	Percent of Area Covered by Duff (litter) at least 5 cm deep	Cover Management Factor Value
75-100	90-100	0.0001-0.001
45-70	75-85	0.002-0.004
20-40	40-70	0.003-0.009

Based on the above values and the objectives of this exposure assessment, it was decided that the high-end values (of those above) would be appropriate; a nominal value of 0.006 (the midpoint of the high-end range) was chosen.

B.17 Sediment Delivery Ratio to Water Body

Parameter: Sdel

Definition: Sediment delivery ratio to water body

Units: unitless

Location	Default Value	Distribution
Both Locations	0.2	U(0.14,0.23)

Technical Basis:

The sediment delivery ratio is the fraction of soil eroded from the watershed that reaches the water body. It can be calculated based on the watershed surface area using an approach proposed by Vanoni (1975):

$$Sdel = a\ WA_L^{-b}$$

where WA_L is watershed area in m^2, b is an empirical slope coefficient (-0.125) and a is an empirical intercept coefficient that varies with watershed area. A graph of the sediment delivery ratio as a function of watershed area is given in the Water Quality Assessment Manual (Mills et al. 1985, pp. 177,178).

B.18 Pollutant Enrichment Factor

Parameter: EF

Definition: The pollutant enrichment factor accounts for the fact that the lighter particles susceptible to erosion tend to have a greater concentration of pollutants attached per mass than what the average soil concentration may suggest.

Units: unitless

Location	Default Value	Distribution
Both Locations	2	U(1.5,2.6)

Technical Basis:

Enrichment refers to the fact that erosion favors the lighter soil particles, which have higher surface area to volume ratios and are higher in organic matter content. Concentrations of hydrophobic pollutants would be expected to be higher in eroded soil as compared to in-situ soil. While enrichment is best ascertained with sampling or site-specific expertise, generally it has been assigned values in the range of 1 to 5 for organic matter, phosphorus, and other soil-bound constituents of concern. Mullins et al. (1993, p.6-22) describe the following equation for calculating enrichment ratio for storm events:

$$EF = 2 + 0.2 \ln(X_e/A_w)$$

where X_e is the mass of soil eroded, in metric tons (1 metric ton = 1000 kg), and A_w is watershed area, in hectares (1 hectare = 10,000 m^2). Experience suggests that typical values range from 1.5 to 2.0, reflecting erosion events from 0.08 to 1.0 tonnes per hectare. A very large erosion event of 20 tonnes per hectare would have a predicted enrichment ratio of 2.6. The default value assumed here is 2.

B.19 Water Body Surface Area

Parameter: Waw

Definition: Water body surface area

Units: km^2

Location	Default Value	Distribution
Both Locations	2.49	U(1.5,3)

Technical Basis:

For the purpose of this assessment, it was assumed that the hypothetical water body has a diameter of 1.78 km, from which the default surface area is calculated.

B.20 Water Body Volume

Parameter: Vw

Definition: Water body volume

Units: m^3

Location	Default Value	Distribution
Both Locations	1.24×10^7	Constant

Technical Basis:

For the purpose of this assessment, it was assumed that the hypothetical water body has a diameter of 1.78 km and mean depth of 5 m. The corresponding volume assuming a disk of height 5 m and radius 0.89 km is then given by 1.24×10^7 m^3 (using the formula $volume = \pi r^2 h$).

B.21 Long-Term Dilution Flow

Parameter: Q

Definition: Long term dilution flow

Units: m^3/yr

Location	Default Value (m^3/yr)
Eastern Location	$1.44x10^7$
Western Location	$1.44x10^5$

Technical Basis:

The long-term dilution flow can be estimated from Tables in U.S. EPA (1985). The values in in/yr are given in Table B-4. These were multiplied by the watershed area of $3.3x10^7$ m^2 to obtain the default values.

Table B-4
Long-Term Dilution Flow In In/Yr

Location	Value (in/yr)
Eastern Location	15
Western Location	0.15

B.22 Suspended Solids Deposition Rate

Parameter: Ssdep

Definition: Suspended solids deposition rate

Units: m/day

Scenario	Default Value (m/day)
Both Locations	0.5

Technical Basis:

Stokes equation can be used to calculate the terminal velocity of a sediment particle settling through the water column, as described in Ambrose et. al. (1988):

$$Vs = \frac{8.64g}{18\ E6}\ (FEp - FEw)\ dp^2$$

where:

Vs is Stokes velocity for a particle with diameter dp and density =FEp, m/day, g is acceleration of gravity =3D 981 cm/sec2, = E6 is absolute viscosity of water =3D 0.01 poise (g/cm3-sec) at 20 =F8C, = FEp is density of the solid, g/cm3, = FEw is density of water, 1.0 g/cm3, and dp is particle diameter, mm.

Values of Vs for a range of particle sizes and densities are provided in Table 3.1. Deposition velocities should be set at or below the Stoke's velocity corresponding to the median suspended particle size, keeping in mind that pollutants tend to sorb more to the smaller silts and clays than to large silt and sand particles = 20. The deposition velocity here represents net deposition over time and so will be smaller for systems experiencing periodic scour = 20. The value chosen here is an order of magnitude below the Stoke's velocity calculated for medium silt particles.

B.23 Benthic Sediment Concentration

Parameter: BS

Definition: Benthic sediment concentration

Units: kg/L

Scenario	Default Value (kg/L)
Both Locations	1 kg/L

Technical Basis:

Benthic sediment concentration is related to the densities of sediment particles, water, and the bulk sediment:

$$Cs = \frac{FEp\ (FEb - FEw)}{(FEp - FEw)}$$

where

FEp is the particle density in g/cm3, FEw is the water density in g/cm3, and =FEB is the sediment bulk density in g/cm3.

Typical particle densities in sediments range between 2.6 and 2.7 k/cm3, and at 20 degrees Celcius water density is close to 1.0 g/cm3. For these properties, a bulk density value of 1.6 g/cm3 corresponds to a sediment concentration of 1.0 g/cm3 (or kg/L) and a porosity of 0.65, which represents consolidated benthic sediment. An analysis of 1680 measured bulk densities in marine sediments exhibited a range from 1.25 to 1.8 g/cm3 and an average particle density of 2.7 (Richards et al.,1974). Some waterbodies contain an upper unconsolidated layer of sediment with bulk densities of 1.1 to 1.3, which correspond to porosities of 0.94 to 0.82 and sediment concentrations of 0.16 to 0.48 g/cm3. In this study, we represent pollutant storage in consolidated beds.

B.24 Upper Benthic Sediment Depth

Parameter: Db

Definition: Benthic sediment concentration
Units: m

Scenario	Default Value (m)	Distribution
Both Locations	0.02	U(0.01,0.03)

Technical Basis:

The total benthic sediment depth can vary from essentially zero in rocky streams to hundreds of meters in oceans. In the lake environments being modeled here, the total benthic sediment depth usually exceeds a few centimeters. Here we are modeling only the upper layer that is in partial contact with the water column through physical mixing and bioturbation. Although bioturbation can descend to tens or even hundreds of centimeters, only the top few centimeters would be in significant contact with the water column. Because this model assumes chemical equilibrium between the upper sediment layer and the water column, a shallow depth of 2 cm was chosen.

B.25 Aquatic Plant Biomass

2 mg/L

Technical Basis:

Aquatic biomass can include phytoplankton and, in shallow areas, benthic algae and rooted aquatic plants. Phytoplankton biomass, as measured by chlorophyll a, can range from less than 1 E 6g/L in

oligotrophic lakes to higher than 200 E 6g/L during blooms = in eutrophic lakes. Given a typical carbon to chlorophyll ratio of 30 (Ambrose et al., 1988), and carbon to biomass ratio of approximately 0.4 (Bowie et al., 1985), the range of aquatic phytoplankton biomass is from 0.08 to '15 mg/L. A yearly average chlorophyll a value of 25 E6g/L gives an estimated biomass of about
2 mg/L, which was used in this study.

B.26 Total Fish Biomass

Bioenergetics typically dictate that biomass declines each trophic level by a factor of 10. Trophic level 3 fish supported by 2 mg/L aquatic biomass, then, would be supported at about 0.02 mg/L. Additionally, trophic level 4 fish and fish supported by external energy sources (such as insects) can be present. A total fish biomass is estimated to be 0.05 mg/L.

B.27 References

Baes, C.F. and R.D. Sharp. 1983. A proposal for estimation of soil leaching constants for use in assessment models. *J. Environ. Qual.* 12: 17-28.

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.* Oak Ridge National Laboratory, ORNL-5786.

Belcher, G.D. and C.C. Travis (1989). *Modelling Support for the Rural and Municipal Waste Combustion Projects: Final Report on Sensitivity and Uncertainty Analysis for the Terrestrial Food Chain Model.* Prepared for the U.S. EPA.

Bowie, G.L. et al. 1985. Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling (Second Edition). EPA/600/3-85/040. U.S. EPA, Athens, GA.

Carsel, R.F., C.N. Smith, L.A. Mulkey, J.D. Dean, and P. Jowise (1984). User's Manual for the Pesticide Root Zone Model (PRZM) Release 1. U.S. EPA, Athens, GA. EPA-600/3-84-109.

Geraghty, J. J., D. W. Miller, F. V. Der Leenden, and F. L. Troise, Water Atlas of the United States. A Water Information Center Publication, Port Washington, N.Y. (1973).

Hoffman, F.O. and D.F. Baes (1979). *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionucleotides.* ORNL/NUREG/TM-882.

Imhoff, J.C., P.R. Hummel, J.L. Kittle, and R.F. Carsel. 1994. PATRIOT - A Methodology and Decision Support System for Evaluating the Leaching Potential of Pesticides. EPA/600/S-93/010. U.S. Environmental Protection Agency, Athens, Georgia.

Mills, W.B., et al. 1985. Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water. Part 1. EPA/600/6-85/002a. U.S. Environmental Protection Agency, Athens, Georgia.

Mullins, J.A., R.F. Carsel, J.E. Scarbrough, and A.M. Ivery. 1993. PRZM-2, A Model for Predicting Pesticide Fate in the Crop Root and Unsaturated Soil Zones: Users Manual for Release 2.0. EPA/600/R-93/046. U.S. Environmental Protection Agency, Athens, Georgia.

National Climatic Data Center. 1986. *Local Climatological Data - Annual Summary with Comparative Data - Cincinnati, (Greater Cincinnati Airport) Ohio.* ISSN 0198-3911, Asheville, North Carolina.

Richards, A.F., T.J. Hirst, and J.M. Parks. 1974. Bulk Density-Water Content Relationship in Marine Silts and Clays. Journal of Sedimentary Petrology, Vol.44, No.4, p 1004-1009.

U.S. Department of Agriculture. (1978). *Handbook No. 537: Predicting Rainfall Erosion Losses.* U.S. Government Printing Office, Washington, D.C.

U.S. EPA. 1988. *Superfund Exposure Assessment Manual.* Office of Remedial and Emergency Response, Washington, D.C. EPA/540/1086/060.

U.S. EPA. 1985. *Water Quality Assessment: A Screening Procedure for Toxic and Conventional Pollutants in Surface and Ground Water (Part 1).* Washington, D.C. EPA/600/6-85/002-A.

U.S. EPA. 1989. *Development of Risk Assessment Methodology for Land Application and Distribution and Marketing of Municipal Sludge.* Office of Health and Environmental Assessment, Washington, D.C. EPA/600/6-89/001.

U.S. EPA. 1990. *Methodology for Estimating Health Risks from Indirect Exposure to Combustor Emissions.* Office of Health and Environmental Assessment, Washington, D.C. EPA/600/6-690/003.

U.S. EPA. 1993. *Indirect Exposure Assessment Working Group Recommendations, DRAFT pending review.*

Vanoni, V.A., 1975. *Sedimentation Engineering.* American Society of Civil Engineers, New York, NY. pp. 460-463.

WeatherDisc Associates. 1992. *U.S. Local Climatological Data Summaries for 288 Primary Stations throughout the U.S.*, on CDROM.

Wischmeier, W. and D. Smith. (1978). Predicting Rainfall and Erosion Losses: A Guide to Conservation Planning. U.S. Department of Agriculture, Agriculture Handbook No. 537.

APPENDIX C

MERCURY PARTITION COEFFICIENT CALIBRATIONS

TABLE OF CONTENTS

C.1 Introduction C-1

C.2 Background C-1
C.2.1 Parameters and Coefficients C-1
C.2.2 Mercury C-2

C.3 Calibrations C-4
C.3.1 Description of Calibration Approach C-4
C.3.2 Parameters Constant for All Calibrations C-4
C.3.3 Calibration Results C-6
C.3.3.1 Swedish Composite Lake C-6
C.3.3.2 Fen at Tivedan National Park, Sweden C-9
C.3.3.3 Composite Minnesota Lake C-15

C.4 Limitations and Uncertainties C-21

C.5 Conclusions C-23

C.6 References C-25

LIST OF TABLES

C-1 Parameters Constant for All Calibrations C-5
C-2 Target Values for Swedish Lake Calibration C-6
C-3 Site-Specific Parameter Values Used in Swedish Composite Lake Calibrations (Excluding Partition Coefficients) C-7
C-4 Summary of Composite Swedish Lake Calibrations C-10
C-5 Soil Layers and mercury Concentrations Reported in Aastrup et al. 1991 . . . . . C-11
C-6 Soilwater Mercury Concentrations Reported in Aastrup et al., 1991 . . . . . C-12
C-7 Model Parameter Values Estimated from Aastrup et al., 1991 . . . . . C-13
C-8 Calibrated Values for Soil-Water Partition Coefficients for Combined Soil Layers and Mor Layer Alone C-14
C-9 Parameter Values for 80 Minnesota Lakes Reported in Sorensen et al., 1990 . . . . . C-16
C-10 Parameter Values Used in Composite Minnesota Lake Calibrations (Excluding Partition Coefficients) C-17
C-11 Results for Composite Minnesota Lake Calibrations C-20
C-12 Summary of Calibration Results C-24

C.1 Introduction

For an assessment of mercury exposure, an accurate modelling of watershed chemistry is critical. One of the important parameters in this watershed chemistry model is the soil-water partition coefficient (K_d) and the benthic sediment-water partition coefficient (see Appendix F for a more complete model description). The method by which literature values were determined did not account completely for the watershed transport of mercury. As a consequence, a calibration effort was undertaken in which mercury watershed transport was assessed at specific sites and modeled in Addendum to the Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions (IEM2). To estimate a general effective K_d value for mercury, the model was calibrated at three sites. The geometric mean of the generated estimates was selected as the final value.

As noted by Dooley (1992), there is a difference of about three orders of magnitude between the reported K_d values for mercury in soil-water systems and those in water-suspended solid systems. Dooley indicated that this difference is not as large as other values in the literature suggest. In this appendix evidence is presented in support of this hypothesis by means of a series of calibrations of a watershed model. The calibrated partition coefficients are about one order of magnitude lower than the reported partition coefficients in water suspended solid systems.

C.2 Background

C.2.1 Parameters and Coefficients

The parameters used to address mobility properties are among the most important in multimedia chemical fate and transport modeling. In many models, it is assumed that the total chemical mass is partitioned among several different compartments. A common assumption is that partitioning is linear; that is, the fraction in one compartment is directly proportional to the fraction in another compartment.

For soil-water systems, the constant of proportionality is called the partition or distribution coefficient and is usually denoted by K_d, with units of (mg/kg)/(mg/L) or L/kg. The partition coefficient is the ratio of the concentration sorbed onto soil particles to that dissolved in soil water at equilibrium; that is no net changes of amount of chemical in soil and water components. The adsorptive properties of a chemical can depend on a variety of environmental factors; e.g., pH of soil, amount of organic matter in the soil or water, percent of sand, silt or clay in soil, other chemicals present and even the magnitude of the chemical concentration in the water itself. Because of the complicated nature of the sorptive process, it is not surprising that reported values for the linear partition coefficient can vary over many orders of magnitude for a given chemical.

More complicated methods than linear partitioning exist for addressing sorption. The nonlinear Freundlich equation is an example of a more complex model in which the soil concentration is assumed to be proportional to some power of the water concentration. The particular power, usually denoted *n* and called the Freundlich exponent, affords a wider range of data fitting capabilities, but as with the simpler approach and as noted by Buchter et al., (1989), it does not provide much information about the actual processes involved.

For long-term (years) estimates, the simple linear approach is perhaps most applicable. The equilibrium assumptions necessary are more likely to be achieved over a long period of time, and the variation that could be observed and expected for short-term simulations are more likely to be adequately characterized by a single representative value. It is more appropriate to call such a partition coefficient an "effective" partition coefficient to reflect its strong empirical nature.

It is this kind of estimate that is appropriate for the methodology described in the Draft Addendum to the Methodology for Assessing Health Risks from Combustor Emissions (U.S. EPA 1993). The IEM2 model also requires partition coefficients for the suspended sediment-water and benthic sediment-water. The soil-water K_d is critical in determining the movement of mercury from land to water bodies, while the other coefficients partition the mercury once it arrives in the water body.

C.2.2 Mercury

Mercury (Hg) can exist in three oxidation states: Hg^0 (metallic or elemental), Hg_2^{2+} (mercurous), and Hg^{2+} (mercuric). The properties and behavior of mercury depend strongly on the oxidation state. Mercurous and mercuric mercury can form numerous inorganic and organic compounds; however, mercurous mercury is rarely stable under ordinary environmental conditions. Most of the mercury encountered in all environmental media except the atmosphere is in the form of inorganic mercuric salts and organomercurics. Organomercurics are defined by the presence of a covalent C-Hg bond. The compounds most likely to be found under environmental conditions are these: the mercuric salts $HgCl_2$, $Hg(OH)_2$ and HgS; the methylmercury (MHg) compounds CH_3HgCl and CH_3HgOH; and, in small fractions, other organomercurics (e.g., dimethylmercury, phenylmercury and ethylmercury).

A number of methods can be used to determine mercury concentrations in environmental media. The concentrations of total mercury, elemental mercury, organic mercury compounds (especially methylmercury) and information on various Hg^{2+} compounds can be measured, although speciation among Hg^{2+} compounds is not usually attempted. Recently, significant improvements and standardizations in analytical methodologies enable reliable data on the concentration of methylmercury, elemental mercury and the Hg^{2+} fraction to be readily separated from the total mercury in environmental media. It is possible to further speciate the Hg^{2+} fraction into reactive, non-reactive and particle-bound compounds, but it is not generally possible to determine which Hg^{2+} species is present (e.g., HgS or $HgCl_2$).

Most of the mercury in soil is thought to be in the form of Hg^{2+} species. Soil conditions are typically favorable for the formation of inorganic Hg^{2+} compounds such as $HgCl_2$, $Hg(OH)_2$ and inorganic Hg^{2+} compounds complexed with organic anions (Schuster 1991). Although inorganic Hg^{2+} compounds are quite soluble and thus theoretically mobile, they form complexes with soil organic matter (mainly fulvic and humic acids) and mineral colloids, with the former being the dominating process. This is due largely to the affinity of Hg^{2+} and its inorganic compounds for sulfur containing functional groups. This complexing behavior greatly limits the mobility of mercury in soil. Much of the mercury in soil is bound to bulk organic matter and is susceptible to elution in runoff only by being attached to suspended soil or humus. However, some Hg^{2+} will be absorbed onto dissolvable organic ligands and other forms of dissolved organic carbon (DOC) and may then partition to runoff in the dissolved phase. Hg^0 can be formed in soil by reduction of Hg^{2+} compounds/complexes mediated by humic substances (Nriagu 1979). This Hg^0 will eventually vaporize and re-enter the atmosphere. methylmercury can be formed by various microbial processes acting on Hg^{2+} substances. Generally, approximately 1-3% of the total mercury in surface soil is methylmercury, and as is the case for Hg^{2+} species, it will be largely bound to organic matter. The other 97-99% of total soil mercury can be considered largely Hg^{2+} complexes, although a small fraction of mercury in typical soil will be Hg^0 (Revis et al. 1990). The methylmercury percentage can exceed 3% (Cappon 1987) in garden soil with high organic content under slightly acidic conditions. Contaminated sediments may also contain higher methylmercury percentages compared to soils (Wilken and Hintelmann 1991; Parks et al. 1989).

Values for soil-water partition coefficients for mercury are rarely reported in the literature, regardless of species. Reported values for mercury range from 10 ml/g (Baes et al., 1984) to 408 ml/g

(Rai and Zachara, 1984). For a Freundlich model, the partition coefficients range from 19-299 ml/g, with the Freundlich exponent ranging from 0.5 to 2.2 (Buchter et al., 1989). Although there is considerable variability in these results, they suggest that typical values in soil-water systems are between 10 and 500 ml/g and are certainly less than 1000 ml/g. These values are based on laboratory experiments under conditions typically not representative of ambient mercury concentrations.

Values derived from measurement under real-world conditions are naturally most appropriate. A determination of the soil-water partition coefficient requires a measurement of speciated soil mercury concentration and the speciated soil water dissolved phase mercury concentration. Measurements of the speciated soil concentrations are typically reported in the literature, but speciated soil water dissolved phase mercury concentration are considerably harder to find.

Data on the benthic sediment-water K_d that are based on measurements under realistic conditions are scarce as well. Wiener et al., (1990) studied mercury partitioning at Little Rock Lake, a clear water seepage lake in north-central Wisconsin. The mercury concentrations in the surficial sediment ranged from 10 to about 170 ng/g (dry weight). Assuming that the reactive mercury values reported represent dissolved Hg^{2+}, the dissolved water concentrations range from 0.29 to 0.59 ng/L. Using these values results in a range for the benthic sediment partition coefficients for this site from 16950 ml/g to 586200 ml/g. There appears to be at least as much uncertainty in the benthic K_d as the soil-water K_d.

In contrast, a number of values for the suspended-sediment K_d have been determined. These are for the most part based on measured data under realistic conditions, unlike the values for the soil-water and benthic-sediment partition coefficients (Appendix A). These values range from 10^3 ml/g (Moore and Ramamodoray 1984) to 10^6 ml/g (Bloom et al., 1991).

Because of the need for realistic partition coefficients in the exposure assessment, several calibrations were performed and are described here. Studies were found that include data on the movement and partitioning of mercury in and around watersheds. The type of information available varies among these studies and can include soil, sediment, surface water, soil water, and runoff water mercury concentrations, as well as lake outflow, lake inflow (runoff and erosion) and sedimentation rates for mercury. The main purposes of these calibrations are twofold: 1) to determine values for the soil-water partition coefficients and the benthic sediment-water partition coefficients that result in mercury to water transport and partitioning behavior that are in reasonable agreement with available mercury transport data; and 2) to confirm that the IEM2 model is capable of correctly predicting the complex process of mercury movement and partitioning in the soil and water environments with the use of realistic parameters. This is one of the most critical aspects of mercury behavior addressed in the exposure assessment.

The modelling results were not as sensitive to the suspended-sediment partition coefficients as the benthic sediment-water coefficients in predicting mercury behavior in the lakes considered in the calibrations, due to the clarity of the water bodies considered. There is also enough reliable information on the suspended-sediment partition coefficients to believe that the mid-point from measured values should reasonably predict mercury partitioning in this study (see Appendix A). Thus, the suspended-sediment partition coefficients were not used in the calibration process.

It is stressed that these calibrations are intended to be only semi-quantitative, with the degree of accuracy of the calibration determined qualitatively; the calibrated results are only to be "consistent with" available data. There are doubtless other possible calibrations. This problem is an intractable aspect of almost any calibration and is discussed in section C.4.

C.3 Calibrations

C.3.1 Description of Calibration Approach

For the model application described in Volume III, the models were run in a "forward" fashion: the input parameters were specified, and the output values (e.g., media concentrations) were obtained. In the calibration effort described here, this process was reversed; the input parameters were modified so that certain output values were within specific ranges. The values for the partition coefficients that yielded acceptable output values were then used as representative values in the main report when the models were run in the "forward" fashion.

The watershed model used, IEM2, uses atmospheric chemical loads and perform to mass balances on a watershed soil element and a surface water element. This mass balance tracks all mercury specified in the background soil concentrations and the mercury deposition rates. The mass balances are performed for total mercury, which is assumed to speciate into three components: Hg^0, Hg^{2+}, and methylmercury. The fraction of mercury in each of these components is specified for the soil and the surface water elements. Loadings and chemical properties are given for the individual mercury components, and the overall mercury transport and loss rates are calculated by the model. IEM2 first performs a terrestrial mass balance to obtain mercury concentrations in watershed soils. IEM2 next performs an aquatic mass balance driven by direct atmospheric deposition along with runoff and erosion fluxes (i.e., amount of the chemical transported from soil element to surface water element per unit time) from watershed soils. The water body output values of the IEM2 model are calculated based on the assumption that steady-state conditions (i.e., fluxes out of surface water element are equal to fluxes into element so that concentrations are independent of time) have been achieved.

There are two main assumptions made in these calibrations. The first is that the measured surface water concentrations are due to the estimated (or reported) fluxes to the water body. Other processes of mercury influx and outflux not considered here are assumed negligible. If these are significant, it could significantly modify the necessary calibrated values. Second, it is assumed that conditions are approximately at steady-state.

The calibrations were generally performed in three steps, depending on the particular data available. First the soil-water partition coefficients were adjusted until the soil-water concentration was within the target soil-water concentration range. Then the runoff/erosion parameters were adjusted until the fluxes to the water body were within the target range of values. Finally, the benthic sediment partition coefficients were adjusted until the water concentrations and benthic sediment concentrations were both within acceptable range (increasing the benthic sediment partition coefficient reduces the water concentration and increases the benthic sediment concentration).

C.3.2 Parameters Constant for All Calibrations

Table C-1 shows the values for parameters that were the same for all calibrations. For a more complete description of these parameters and the rationale for the values chosen, see Appendix A to Volume III.

Table C-1
Parameters Constant for All Calibrations

Chemical-Dependent Parameters	Hg^0	Hg^{2+}	Methylmercury
Molecular weight (g/mole)	201	201	216
Henry's Constant (atm-m^3/mole)	$7.3x10^{-03}$	$7.3x10^{-10}$	$4.7x10^{-07}$
Soil-water partition coefficient (ml/g)	130[a]	Calibrated	Calibrated
Benthic-sediment partition coefficient	130[a]	Calibrated	Calibrated
Suspended-sediment partition coefficient	1[a]	$9.50x10^4$ [b]	$6.50x10^5$ [b]
Equilibrium fraction in soil	0	0.98	0.02
Equilibrium fraction in water	0.02	0.83	0.15

Constants	Value	Comment
Ideal gas constant (m-atm/mole-K)	$8.21x10^{-05}$	Used for volatilization from soil and surface water
Air density (g/cm^3)	$1.19x10^{-03}$	Used for water body calculations
Solids density (kg/L=g/cm^3)	2.70	Used to estimate speciation in waterbody and concentration in benthos
Drag coefficient	$1.10x10^{-03}$	Used for water body volatilization calculations
Von Karman's Coefficient	$7.40x10^{-01}$	Used for water body volatilization calculations
Dimensionless boundary thickness	4.00	Used for water body volatilization calculations

Run Options	Value	Comment
Water body type	1	Stagnant ponds, lakes
Suspended solids options switch	0	Use given sediment deposition rate to calculate suspended solids concentration
Equilibrium speciation option	1	Species are tied together in equilibrium

[a] Because it is assumed that the equilibrium fraction of elemental mercury in soil is 0 (see Appendix A), the soil-water partition coefficient does not affect the calculations. Similarly, due to the low assumed equilibrium fraction in surface water the other partition coefficients for elemental mercury does not significantly affect calculations and so it is not varied from the value shown here.

[b] The suspended sediment partition coefficients were not as influential on the results used in the calibrations here, based on the initial sensitivity analyses. For this reason, they were assigned the values given in Appendix A. These values are based on these studies: Moore and Ramamodoray (1984), and Robinson and Shuman (1989).

C.3.3 Calibration Results

C.3.3.1 Swedish Composite Lake

In a series of papers, Meili investigated the mercury cycle through Swedish boreal forest watersheds and lakes (Meili 1991a; Meili 1991b; Meili et al. 1991). The data in these papers, which consist of a combination of summary values for Swedish lakes and predicted values, were used to construct a model Swedish lake, from which many of the necessary IEM2 parameters could be approximated. The output values in Meili (1991a, 1991b) that were used as target values in the calibrations are shown in Table C-2. Table C-3 shows the input values, excluding the partition coefficients, used in the IEM2 model for the calibrations.

Table C-2
Target Values for Swedish Lake Calibration

Output Parameter	Value	Comment
Mercury Concentration in lake (ng/L)	2-3	For clearwater lake; Table I of Meili 1991b (page 723)
Runoff load of mercury to lake, $ug/m^2/yr$ or g/yr[a]	4-8 Central Sweden, 6-11 Southern Sweden	
Mercury Concentration in Runoff (ng/L)	3.7	Based on mercury/Carbon ratio of 0.25 ug/g in runoff water and organic carbon concentration of 15 g/m^3 (Table I in Meili 1991b)
Outflow of mercury from lake, $ug/m^2/yr$ or g/yr[a]	2-5 Central Sweden, 3-7 Southern Sweden	
Sedimentation of mercury in lake, $ug/m^2/yr$ or g/yr[a]	7-20 Central Sweden, 10-30 Southern Sweden	
Ratio of runoff load and direct deposition	0.6	For clearwater lakes (p.724 of Meili 1991b)
Surface sediment concentration, ng/g	150-460	Range from Table I in Meili et al., (1991), p.441 for characteristics of 25 study lakes in 1986
Mass balance of Loss Processes	% Sedimentation	% Outflow
Central Sweden	80	20
Southern Sweden	79	21

[a] Fluxes are given per unit lake surface area (Meili 1991b, p.722). Using the assumed surface area of 1 km^2 for a typical lake gives an estimate of the flux in grams per year.

Table C-3
Site-Specific Parameter Values Used in Swedish Composite Lake Calibrations (Excluding Partition Coefficients)

Parameter	Value	Comment
Mercury deposition rate (ug/m^2/yr)	12.5	Average of values for Central and Southern Sweden (Range 7-25)
Irrigation Rate (cm/yr)	0	Assumed negligible
Time of Concentration (years)	1	Fluxes estimated in Meili et al. are presented on a yearly basis
Average air temperature (C)	4	Based on mean air temperature if 1-7 (Meili 1991a, p.337)
Average water temperature	4	Assumed to be the same as average air temperature
Average annual precipitation (cm/yr)	75	Average of central and southern Sweden (Table I in Meili 1991b, p.723)
Average annual runoff (cm/yr)	40	Annual area-specific runoff water discharge for central and southern Sweden (Table I in Meili 1991b, p.723)
Average annual evapotranspiration (cm/yr)	17.5	Rough estimate (this is only used for estimating leaching loss rate)
Average wind speed (m/s)	3.5	Rough estimate (this is only used for volatilization)
Soil Erosion Parameters		
Practice Support Factor	1	Cover management factor is used to incorporate this.
Sediment delivery ratio (unitless)	0.2	Typical value
Pollutant enrichment factor (unitless)	2	Typical value
Erosivity factor (/yr)	175	Rough estimate based on U.S. values (values range from 100 to 300)
Erodability factor (tons/acre)	0.28	Typical value (U.S. values range from 0.28 to 0.3)
Topographic factor (unitless)	2.5	Value for a location in the upper northeast, which is assumed to be roughly similar to the Swedish area
Cover management factor (unitless)	0.006	Forest value
Watershed Parameters		
Watershed area (km^2)	4	Mean of lakes in Meili et. al (1991a,1991b)
Bulk soil density (g/cm^3)	1.4	Typical value.
Watershed soil mixing depth (cm)	1	Value suggested for non-tilled soil (U.S. EPA 1993)
Soil moisture content (unitless)	0.3	Typical value

Table C-3
Site-Specific Parameter Values Used in Swedish Composite Lake Calibrations (Excluding Partition Coefficients)
(continued)

Parameter	Value	Comment
Background soil concentration (ng/g)	100	Average of mean reported in Jensen and Jensen (1991) for Swedish soils of 76 ng/g and weighted average for soil layers in Swedish soil of 125 ng/g (Aastrup et al., 1991)
Water body parameters		
Sediment deposition rate (m/day)	0.5	Rough mean of typical values (U.S. EPA 1994)
Benthic sediment concentration (kg/L)	1	Rough mean of typical values(U.S. EPA 1994)
Upper benthic depth (m)	0.02	Typical value (U.S. EPA 1994)
Water body surface area (km^2)	1	Value for clearwater lake in Meili (1991b), Table II, p.723
Water column volume (m^3)	$6x10^6$	Based on area of 1 km^2, depth of 6 m (Table I, Meili 1991b, p.723)
Long term dilution flow (m^3/yr)	$2x10^6$	Based on volume of lake and mean water residence time of 3 years (Table I, p.723 of Meili 1991b)

Because there is considerable uncertainty about the degree of volatilization from the surface water body, two separate calibrations were performed. In the first, volatilization from the surface water body was considered as a loss process. In the second, no volatilization was assumed. The latter is consistent with assumptions in Meili et al. (1991b), where volatilization was not considered due to uncertainty. The calibration of the soil-water partition coefficients is the same in both calibrations because it is not affected byloss processes from the surface water body.

The first step was to calibrate the soil-water partition coefficients so that the predicted total mercury soil-water concentration was within range of the target values (3.75-5 ng/l). Then the erosion parameters were modified so that the fluxes to the water body from runoff and erosion agreed. In the IEM2 model, the various erosion parameters (sediment delivery ratio, pollutant enrichment factor, erosivity factor, erodability factor, topographic factor, cover management factor) are multiplied together to obtain an estimate of the annual amount of soil erosion; thus, there are many different possible combinations of various values for these parameters that can yield the same general erosion rate. Finally, the benthic sediment partition coefficients were modified so that the predicted surface water and benthic sediment concentrations were consistent with the values reported in Meili et al. 1991. The results of both calibrations are shown in the Table C-4.

Although there is general agreement with the target outfluxes, the high benthic sediment concentration suggests that the assumption of no significant volatilization from the surface water may not be true, unless there are processes not addressed here that serve to prevent the predicted high benthic sediment concentrations from occurring.

In summary, using the available data on 88 lakes in Sweden, the IEM2 watershed model was calibrated using these available data. The calibrated values of the benthic-sediment partition coefficients depend on the significance of the volatilization pathway from the water body. The calibrated benthic-sediment partition coefficients were found to agree with the overall range reported in Wiener et al. (1990).

C.3.3.2 Fen at Tivedan National Park, Sweden

Aastrup et al., (1991) describe mercury transport within a small segment (6%) of a larger watershed area. The results presented in this paper were used for calibrating the parameters involved in estimating the runoff of mercury from the watershed.

The study area was described as a minicatchment watershed consisting of a small forested catchment in the Tivedan National Park located in southern Sweden. The mercury budget was estimated for a till formation on a slope, making up a funnel-shaped minicatchment of 0.014 km^2. This area drains into a fen (low land covered wholly or partly with water unless artificially drained). Elevation in the minicatchment ranges from approximately 1.5 m at the point farthest from the fen to 0 m at the fen itself. The catchment was divided into three areas: an upper level with shallow soils, an intermediate area, and a waterlogged area.

Table C-4
Summary of Composite Swedish Lake Calibrations

Calibration Parameters	Volatilization from Surface Water Assumed	No Volatilization from Surface Water Assumed	Target Calibration Value
Soil-Water partition coefficients for Hg^{2+} and methylmercury (ml/g)	22500	22500	NA
Topographic factor (unitless)	0.8	0.8	NA
Pollutant enrichment factor (unitless)	1.0	1.0	NA
Sediment delivery ratio (unitless)	0.10	0.10	NA
Soil erosivity factor (kg/km^2-yr)	125	125	NA
Benthic Sediment Partition Coefficients for Hg^{2+} and methylmercury (ml/g)	155000	350000	NA
Output Parameters			
Total mercury Soil Concentration after one year	101	101	100
Total mercury Soil-Water Concentration (ng/l)	4.47	4.47	3.75 Central 5 Southern
Total Load to water body from runoff/erosion (g/yr)	8.53	8.53	4-8 Central 6-11 Southern
% Erosion	18	18	NA
% Runoff	82	82	NA
Steady-state Total Water Column Concentration (ng/l)	3.07	3.00	2-3
Steady-state outflows from water body (g/yr)			
Volatilization	8.72 (40%)	0 (0%)	NA
Sedimentation	6.79 (32%)	15 (71 %)	7-20 Central 10-30 Southern
Dilution flow	6.12 (28%)	6 (29%)	2-5 Central 3-7 Southern
Concentration in Benthos (ng/g)	460	1020	150-460

Mercury concentrations were reported for different soil layers . It was noted that 41% of the total mercury estimated (8.8 kg/km^2) was found in the highest humic layer. Table C-5 shows the total mercury found in each soil layer analyzed.

Table C-5
Soil Layers and mercury Concentrations Reported in Aastrup et al. 1991

Soil Layer	Assumed Thickness (cm)	Mercury Content (ng/g)	Total Mercury (g)
Mor	8	250	50
E-Horizon	6	27	9
upper B-horizon	6	58	34
lower B-horizon	NA	23	23
C-horizon	NA	6	8
Total		NA	124

Also reported were estimates of mercury concentrations in soilwater and groundwater. There was a large amount of variation in the measured values. The values included in Table C-6 below generally have a standard deviation as large as the mean, even for those with as many as 30 samples.

The soilwater mercury concentration calculated in the IEM2 model is assumed to be the dissolved mercury, and hence does not consist of any particulate-bound mercury. For the purpose of the calibration effort, it was decided that this calculated quantity would be bounded above by the values for Hg-II (sum of reactive and unreactive mercury) reported in Aastrup et al. and shown in Table C-6. These values, estimated in accordance with the standard Swedish sampling program (Chapter 2 in Lindquist et al., 1991), are the sum of the dissolved Hg^{2+} plus some reactive particle associations and some humic matter associations which may fall under particulate mercury (Hg-IIa and Hg-IIb). The difference between total mercury concentration (Hg-tot) and Hg-II was assumed to be Hg^{2+} strongly bound to particulates (i.e., not dissolved). For more details regarding the mercury notation of the Swedish Sampling Program see Section 2 of Appendix E of this report.

The fluxes out of the minicatchment area were estimated from Figure 4 (page 165) in Aastrup et al. (as well as Figure 2 in Johansson et al., 1991). The flux to the fen from the top 20 cm of soil is 2.6 g/km^2/yr. Because the fluxes are normalized to the minicatchment area, multiplying by the area (0.014 km^2) yields a total flux of 0.0364 g/yr. Similarly, for the top 8 cm (called the mor layer, which consists of humic matter distinct from mineral soil), the flux is 1.5 g/km^2/yr, corresponding to a total flux of 0.02 g/yr.

Two separate calibrations were performed. In the first, the top 20 cm was treated as a single layer, while in the second only the mor layer was used. The parameters, their values, and the rationale for their selection are shown in Table C-7. Table C-8 shows the results for the both calibrations.

Table C-6
Soilwater Mercury Concentrations Reported in Aastrup et al., 1991

Soil Layer	Total Mercury (ng/L)				Hg-II[a] (ng/L)				Hg-IIa[b] (ng/L)			
	Mean	Std.Dev.	Minimum	Maximum	Mean	Std.Dev.	Minimum	Maximum	Mean	Std.Dev.	Minimum	Maximum
0-8 cm	11.9	12.7	1.9	39.6	4.1	3.7	0.9	10.3	1.4	1.1	0.5	3.4
8-20 cm	13.6	13.7	1.9	48.4	4.3	4.8	0.5	20.2	1.5	1.1	0	4.2
20-50 cm	15.1	12.9	0.6	52.1	4.1	4.6	0.1	19.5	1.3	0.8	0	3.8
50-100 cm	9.9	8.8	0.7	39.2	3.3	4.9	0	27.6	1.1	0.9	0	3.8
100-165 cm	6.8	3.7	1.5	12.5	0.9	0.5	0.3	1.7	0.5	0.4	0	1.4

[a] Hg-II - Reactive and non-reactive mercury fraction of total mercury (see Appendix E)
[b] Hg-IIa - Reactive, acid labile, inorganic fraction of total mercury (see Appendix E)

Table C-7
Model Parameter Values Estimated from Aastrup et al., 1991

Parameter	Value	Comment
Mercury Deposition Rate (ug/m^2/yr)	20	Value assumed in Aastrup et al., (1991)
Approximate Estimated background soil concentration (ng/g) in soil layer of consideration	125 for Combined mor, E and upper B 250 for mor alone	Weighted average soil concentration for mor, E-horizon, and upper B-horizon soil layers.
Time period of consideration (yr)	1.3	16 months, which was the length of the study period in Aastrup et al., (1991)
Mixing Depth (cm)	20 cm for combined mor, E and upper B 8 cm for mor alone	Depths reported in Aastrup et al., (1991).
Estimated precipitation (cm/yr)	120	Estimated from Figure 3, page 163 in Aastrup et al., (1991)
Estimated runoff (cm/yr)	58.7 for Mor, E and upper B 37.7 for mor alone	Figure 3, page 163 in Aastrup et al., (1991).
Evapotranspiration (cm/yr)	34 for mor, E and upper B 38.5 for mor alone	Value set so that precipitation minus runoff minus evapotranspiration is equal to estimated leach rate from soil layer considered (Figure 3 in Aastrup et al., 1991). Leach rate is 26.7 cm/yr from top 20 cm, 42.8 cm/yr from top 8 cm (mor layer).
Average Air Temperature (C)	4	Estimate of average Swedish temperatures (Meilie 1991a)
Average Wind speed (m/s)	3.5	Rough estimate
Bulk Soil Density (g/cm^3)	1.4	There is usually little variation in this parameter across the U.S.; Swedish soils assumed to be similar
Soil Moisture Content	0.30	Rough midpoint of typical values; there is usually little variation across U.S. soils.
Erosivity factor (/yr)	175	Rough average of typical values.
Erodability factor (tons/acre)	0.28	There was little variation for U.S. Sites (0.28-0.30).
Topographic factor (unitless)	0 19	Computed using method of Arnold et al., (1990) using slope of area
Cover management factor (unitless)	0.006	Typical forest value

Table C-8
Calibrated Values for Soil-Water Partition Coefficients for Combined Soil Layers and Mor Layer Alone

Parameter	Combined Mor, E and Upper B Horizon Soil Layers		Mor Soil Layer Alone	
	Value	Target Calibration Value	Value	Target Calibration Value
Soil-Water Partition Coefficient (ml/g) for Hg^{2+} and methylmercury	38000	NA	250000	NA
Total Mercury Soil Concentration (ng/g)	125	125	250	250
Total Mercury Soil-Water Concentration (ng/L)	3.29	0.5 - 20.2 (mean 4.2)	1	0.9 - 10.3 (mean 4.1)
Total Load to Water body from Catchment Region Considered (g/yr)	0.036	0.036	0.02	0.02
% Erosion	24.4	NA	77	NA
% Runoff	75.6	NA	23	NA

Despite the low values used for the default soil erosion parameters, the loss due to erosion becomes significant when the effective partition coefficient is increased. Whether this was actually true at the Tividen site is not certain. Nevertheless, the calibration for the mor layer alone provides an upper bound for the partition coefficient, as the predicted soil-water concentration lies at the lower end of the observed range.

C.3.3.3 Composite Minnesota Lake

These calibrations are based on the Minnesota lake characterization presented in Sorensen et al., (1990) based on 80 lake watersheds. Mercury concentrations in precipitation, lake water and sediment were measured and analyzed along with watershed data for 80 lake watersheds in the study region of northeastern Minnesota. The summary values are shown in Table C-9. Median values were used when possible because a number of large watersheds/ waterbodies biased the mean values. The values used in the IEM2 model are shown in Table C-10.

The median value for evaporation reported in Sorensen et al., (1990) was 47.6; however, this results in a negative net leach rate since leach rate is proportional to *Precipitation + Irrigation - Runoff - Evapotranspiration*. For this reason, the evapotranspiration was set so that leach rate is 0. This has little practical affect on predicted values because the background soil concentration is not subject to these loss processes.

Although no values are reported for the soil-water concentrations in Sorensen et al., (1990), the values reported in Aastrup et al., (1991) for Swedish soil indicate that they may be approximated by the surface water concentration. The target value for this effort was the median of the surface water concentrations reported in Sorensen et al. of 2.3 ng/l. After the soilwater concentrations were calibrated, the topographic factor, used in estimating soil erosion, was set so that the indirect/direct ratio was consistent with that estimated from Sorensen et al., (1990).

As in the calibrations performed for the composite Swedish lake, two separate calibrations were performed, one with and one without volatilization from the surface water body. The results of the calibrations are shown in Table C-11. These two calibrations provide a range for the benthic sediment concentrations.

Despite the uncertainties introduced by using a composite lake, the results are in general agreement with previous calibrations. As for the composite Swedish lake, the benthic-sediment partition coefficients had to be substantially increased if volatilization from the surface water body was not considered as a loss process.

Table C-9
Parameter Values for 80 Minnesota Lakes Reported in Sorensen et al., 1990

Parameter	Median	Range/Comment
Lake concentration (ng/L)	2.30	0.90-7.00
Total organic carbon as C in surface water (mg/L)	6.76	3.53-14.3
Annual Direct deposition onto lake (ug/m^2/yr)	12.7	10.4-15.4
Deposition immediate ug/m^2 (calculated deposition falling directly on lake surface plus calculated runoff from immediate watershed (assuming 100% mercury transport to lake)	24.8	14.8-58.4
Indirect/Direct Deposition to Water Body	0.95 (calculated using median values, but not necessarily the median)	This is a value calculated here by assuming that the difference between the "deposition immediate" and the direct deposition onto the lake is the flux from the immediate watershed (which turns out to be 12.1 ug/m^2/yr). Dividing this by the direct deposition gives this ratio.
Median surface sediment concentration (ng/g)	154	34-753
Lake surface area (Ha)	328	24-89400
Immediate watershed area (Ha)	650	55-168000
Site elevation (m)	432	388-587
Topographic high immediate (m)	476	378-664
Annual precipitation (m/yr)	0.665	0.560-0.762
Annual evaporation, land (m/yr)	0.476	0.446-0.506
Annual runoff (m/yr)	0.196	0.103-0.315
Lake renewal time (yr)	49.1	5.85-202
Total renewal time (yr)	2.18	0.01-45.4
Mean depth (m)	5.70	1.08-29.0
Lake volume (m^3)	$1.5x10^7$	$4.35x10^5$ - $5.47x10^9$
% Forest	83	46.2-94.7
% Water	16.2	4.10-38.5

Table C-10
Parameter Values Used in Composite Minnesota Lake Calibrations (Excluding Partition Coefficients)

Parameter	Value	Comment
Mercury deposition rate ($ug/m^2/yr$)	12.7	Median of values reported in Table III of Sorensen et al, (1990)
Irrigation Rate (cm/yr)	0	Assume it is negligible
Time of Concentration (years)	1	Used only in calculating soil concentrations.
Average air temperature (C)	5	Mean for Minnesota
Average water temperature	5	Assumed to be the same as average air temperature
Average annual precipitation (cm/yr)	66.5	Median of values reported in Table III of Sorensen et. al., (1990)
Average annual runoff (cm/yr)	19.6	Median of values reported in Table III of Sorensen et. al., (1990
Average annual evapotranspiration (cm/yr)	46.9	Set to precipitation minus runoff
Average wind speed (m/s)	3.5[a]	Rough estimate; this is only used for volatilization.
Soil Erosion Parameters		
Practice Support Factor	1	Cover management factor is used to incorporate this.
Sediment delivery ratio (unitless)	0.2	Typical value
Pollutant enrichment factor (unitless)	2	Typical value
Erosivity factor (/yr)	175	Rough estimate based on U.S. values; values range from 100 to 300

Table C-10
Parameter Values Used in Composite Minnesota Lake Calibrations (Excluding Partition Coefficients)
(continued)

Parameter	Value	Comment
Erodability factor (tons/acre)	0.28	Typical value (U.S. values range from 0.28 to 0.3)
Topographic factor (unitless)	2.45	This is actually a calibrated value so that indirect/direct ratio matched a value consistent with the values reported in Sorensen et al., (1990).
Cover management factor (unitless)	0.01	Forest value
Watershed Parameters		
Watershed area (km^2)	6.5	Based on median value of 650 hectares reported in Table III in Sorensen et al., (1990).
Bulk soil density (g/cm^3)	1.4	Typical value
Watershed soil mixing depth (cm)	1	Standard value. Variations in the mixing depth will contribute to the overall uncertainty, but the value is deemed appropriate given the qualitative nature of the current calibration efforts.
Soil moisture content (unitless)	0.3	Typical value (usually relatively little variation)
Background soil concentration (ng/g)	87	Average of values reported in Sorensen et al., 1990 (p.1724); values ranged from 12 to 220 ng/g.
Water body parameters		
Sediment deposition rate (m/day)	0.5	Rough mean of typical values (U.S. EPA 1994)
Benthic sediment concentration (kg/L)	1	Rough mean of typical values (U.S. EPA 1994)
Upper benthic depth (m)	0.02	Typical value (U.S. EPA 1994)

Table C-10
Parameter Values Used in Composite Minnesota Lake Calibrations (Excluding Partition Coefficients)
(continued)

Parameter	Value	Comment
Water body surface area (km^2)	3.28	Based on median value of 328 hectares reported in Table III in Sorensen et al. (1990).
Water column volume (m^3)	$1.5x10^7$	Median value reported in Table III in Sorensen et al. (1990).
Long term dilution flow (m^3/yr)	$6.88x10^6$	Based on volume of lake and median total renewal time of 2.18 years (Table III of Sorensen et al. 1990).

[a] The wind speed was also set to zero for a calibration run in which no volatilization occurs.

Table C-11
Results for Composite Minnesota Lake Calibrations

Parameter	Volatilization from Surface Water Assumed	No Volatilization from Surface Water Assumed	Target Calibration Value
Effective Soil-Water Partition Coefficient (ml/g) for Hg^{2+} and methylmercury	38200	38200	NA
Effective Benthic Sediment Partition Coefficient (ml/g) for Hg^{2+} and Methylmercury	100500	149600	NA
Total Mercury Soil Concentration after one year (ng/g)	87.8	87.8	87
Total Mercury Soilwater Concentration (ng/L)	2.3	2.3	2.30
Indirect/Direct Ratio	0.95	0.95	0.95
Total Load to Water body from Catchment Region Considered (g/yr)	39.7	39.7	NA
% Erosion	92.36	92.36	NA
% Runoff	7.64	7.64	NA
Total Water column concentration (ng/L)	2.3	2.3	2.3
Steady-state outflows (g/yr)			
Volatilization	21.4 (26%)	0	
Sedimentation	44.0 (54%)	65.4 (80%)	
Dilution	15.7 (20%)	15.7 (20%)	
Total Mercury Concentration in Benthos (ng/g)	214	318	154 (Range 34-753)

C.4 Limitations and Uncertainties

The calibrated partition coefficients derived here are intended to represent long-term retention properties of the watershed systems for which they were derived. An obvious limitation with any calibration effort is that there may be other calibrations that also give the same qualitative agreement but have significantly different values for the calibrated parameters. The likelihood of such alternative calibrations is increased when large data gaps exist. Because the calibrations were performed in a sequence of steps, these possibilities are discussed in turn for each step in the calibration process used here.

The soil-to-water partition coefficient calibrations seem the most defensible because there is relatively little involved in calculating soilwater concentrations. Furthermore, only a relatively simple argument is required in order to suggest that the typically reported values for the mercury partition coefficients in soil-water systems are questionable. In the IEM2 model, the total concentration of each mercury component in soil is assumed to reach equilibrium between its particulate and aqueous phases. The concentration of species *i* dissolved in pore water is given by the following equation:

$$C_{ds,i} = \frac{Sc_i \; BD}{\theta_s + Kd_{s,i} \; BD} \tag{1}$$

The concentration in particulate phase is defined in equation (2).

$$C_{ps,i} = C_{ds,i} \; Kd_{s,i} = \frac{Sc_i \; Kd_{s,i} \; BD}{\theta_s + Kd_{s,i} \; BD} \tag{2}$$

where:

Sc_i	=	total soil concentration of component "i" (µg/g)
θ_s	=	volumetric soil water content (L_{water}/L)
$Kd_{s,i}$	=	soil/water partition coefficient for component "i" (L/kg = ml/g)
BD	=	soil bulk density (g/cm^3)
$C_{st,i}$	=	total soil concentration of component "i" (mg/L)
$C_{ds,i}$	=	concentration of "i" dissolved in pore water (mg/L_{water})
$C_{ps,i}$	=	concentration of "i" in particulate phase (mg/kg)

The total soil concentration in ug/g is given by this equation.

$$SC_i = \left(\frac{\theta_s}{BD} + Kd_{s,i} \right) C_{ds,i} \tag{3}$$

The value for the partition coefficient to achieve a given soil-water concentration is , thus,

$$Kd_{s,i} = \frac{SC_i}{C_{ds,i}} - \frac{\theta_s}{BD} \tag{4}$$

If the mercury soil-water concentrations reported in Meili et al., (1991) are accurate, then these values indicate that the mercury in the soil-water represents only a small fraction of the total mercury per volume. For example, if a typical total soil concentration of 100 ng/g (0.10 ug/g) were completely dissolved in a liter of water, and assuming a typical soil density of 1.4 g/cm^3, the resulting water

concentration would be 100000 ng/L (100 ug/L). This is to be compared to the reported soil-water concentrations in the range of 1-10 ng/L. Thus, at most about 0.01% can be dissolved to achieve the values observed, and the rest must be bound to particulates in the soil matrix. Even assuming a volumetric soil water content of 1, using equation (4) above the partition coefficient must be about 10^4 ml/g in order to have a dissolved water concentration of 10 ng/L. Achieving soil-water concentrations of 2 ng/L requires a partition coefficient of slightly less than $5x10^4$ ml/g.

Because adequate speciation estimates were not available, there is uncertainty in the values for the partition coefficients for methylmercury. For the purpose of this effort they were assumed to be the same as for Hg^{2+}. In sediment, values between about 2% and 9% methylmercury have been reported (Agaki et al., 1979) for sand, silt/woodchips and woodchip sediments. Cappon (1984) found that percent methylmercury for nonamended soils was about 2.6% (this is an upper bound on values from unpublished data reported by several authors as cited in *Water, Air and Soil Pollution* 1991). If the speciation in soil-water is similar to that sorbed onto soil particles, then the partition coefficients for methylmercury would be similar to those for Hg^{2+}. Although there is considerable variability in the percent of total mercury that is methylmercury in surface waters, the latest estimates (Bloom et al., 1991; Watras and Bloom 1992) range from 5% to 25% methylmercury. If the fraction in soil water is slightly larger than that sorbed onto soil particles, as the data would indicate, then the required calibrated partition coefficient for methylmercury would be correspondingly lower than that for Hg^{2+} derived above. This is because the fraction dissolved for methylmercury would be higher than that for Hg^{2+}. However, for the purpose of this calibration effort, it was felt that the data were not adequate to justify separate calibrations for both Hg^{2+} and methylmercury. The result of this assumption is that the amount of methylmercury in the flux to the water body from runoff may be underestimated and amount in soil erosion overestimated.

Calculation of the flux to the water body boils down to determining the a set of adequate erosion/runoff parameters. The total load due to runoff and erosion, denoted here by $L_{E/R,i}$ (g/yr), is given by equation (5).

$$L_{R/E,i} = WA_L \left(R\, C_{ds,i}\, 10^{-2} + X_e\, SD\, ER\, C_{ps,i}\, 10^{-3} \right) \quad (5)$$

where:

$L_{E/R,i}$	=	load to water body from surface runoff and soil erosion for component *i* (g/yr)
WA_L	=	watershed surface area (m^2)
R	=	average annual runoff (cm/yr)
$C_{ds,i}$	=	concentration of "i" dissolved in pore water (mg/L_{water})
X_e	=	unit soil loss ($kg/m^2/yr$)
SD	=	sediment delivery ratio (unitless)
ER	=	contaminant enrichment ratio (unitless)
$C_{ps,i}$	=	concentration of "i" in particulate phase (mg/kg)

The unit soil loss rate is given by equation (6).

$$X_e = R_\, K\, LS\, C\, PS\, \frac{907.18}{0.00407} \quad (6)$$

where

R_	=	soil erosivity factor ($kg/km^2/yr$)

K	=	soil erodability factor (tons/acre)
LS	=	topographic factor (unitless)
C	=	cover management factor (unitless)
PS	=	support practice factor (unitless)

Values for the site-specific average annual runoff were available for the sites considered. Values for the soil erosion parameters (R_, K, LS, C, PS, SD, ER) were not available. This was further complicated by the composite nature of the sites considered for the calibration efforts. For this reason, the erosion parameters were calibrated so that the ultimate total fluxes to the water body were consistent with measured data.

The calibrated benthic sediment partition coefficients are consistent with values reported elsewhere (e.g., Wiener et al., 1990). Because there is apparently considerable uncertainty as to the degree of volatilization from surface water bodies, two different calibrations were performed, when possible. In the first the volatilization was assumed to contribute to the total loss rate from the surface water using a generic set of the relevant parameters (e.g., equilibrium speciation of the mercury species in water,wind speed). Another calibration was also performed assuming that volatilization was negligible. The differences in the calibrated partition coefficients are well within the range of the usual variation of the partition coefficients themselves. Calibration without volatilization requires that sedimentation play a larger role as a loss process. The higher benthic sediment partition coefficients needed to achieve this effect, while resulting in high benthic sediment concentrations, provide upper bounds on the partition coefficients for the site under consideration.

C.5 Conclusions

Calibrations of the IEM2 model were performed using three data sets, with the partition coefficients serving as the primary calibration parameters. Due to uncertainty as to the exact degree of volatilization from the surface water body, two separate calibrations were performed with and without volatilization. The calibrated partition coefficients are shown in Table C-12.

The significance of volatilization as a mercury loss process from water bodies is currently unclear. The results derived here show that the assumption that volatilization is negligible, while it can be modelled by increasing the benthic sediment partition coefficients, results in benthic sediment concentrations that are near or above the upper end of the measured values. This suggests that volatilization may in fact be nonnegligible.

Despite acknowledged uncertainties and limitations, the results derived here support the use of soil-to-water partition coefficients that are higher than previously published values, when the equilibrium assumption is considered appropriate. The effective partition coefficients (soil and benthic) determined from these calibrations are similar to values found for suspended-sediment partition coefficients, for which much reliable data are available. Additionally, it was confirmed that the IEM2 model could predict mercury movement and partitioning in the soil and water environments with the use of realistic modeling parameters. Thus, we have established that

Table C-12
Summary of Calibration Results

	Composite Swedish Lake		Fen in Tivedan Park, Sweden		Composite Minnesota Lake		Geometric Mean of Calibrated Values
	Volatilization Included	Volatilization not Included	Combined Soil Layers	Top layer alone	Volatilization Included	Volatilization not Included	
Soil-Water Partition Coefficients for Hg^{2+} and Methylmercury (ml/g)	$2.3x10^4$	$2.3x10^4$	$3.8x10^4$	$2.5\ x10^5$	$3.8x10^4$	$3.8x10^4$	$5.37x10^4$
Benthic Sediment Partition Coefficients for Hg^{2+} and Methylmercury (ml/g)	$1.5x10^5$	$2.7x10^5$	N/A	N/A	$1.0x10^5$	$1.5x10^5$	$1.57x10^5$

applying this part of the IEM2 model using partition coefficients representative of those in Table C-12 can be expected to result in reasonable predictions of mercury movement and behavior in and out of watersheds.

That soil-water partition coefficients larger than previously published values would be necessary is consistent with the growing concern that watershed soils may be serving as a significant repository for mercury. This repository can potentially act as a source of mercury to water bodies long after enhanced mercury deposition has occurred.

C.6 References

Aastrup, M., J. Johnson, E. Bringmark, I. Bringmark, and A. Iverfeldt (1991). Occurrence and Transport of Mercury within a Small Catchment Area, *Water, Air, and Soil Pollution* 56: 155-167.

Akagi H., D.C. Mortimer, and D.R. Miller (1979). Mercury Methylation and Partition in Aquatic Systems. *Bull. Environ. Contam. Toxicol.* 23:372-376.

Arnold, J.G., J. Williams, A. Nicks, and N. Sammons, 1990. SWRBB: A Basin-Scale Simulation Model for Soil and Water Resources Management, Texas A & M University Press, College Station, Texas.

Baes et al., (1984) is cited in Appendix B Reference Section. It should also be cited in Appendix C Reference Section.

Bloom, N.S., C.J. Watras, and J.P. Hurley (1991). Impact of Acidification on the Methylmercury Cycle of Remote Seepage Lakes. *Water, Air, and Soil Poll.* 56:477-491.

Buchter, B., B. Davidoff, M.C. Amacher, C. Hinz, I.K. Iskandar, and H.M. Selim (1989). Correlation of Freundlich K_d and retention parameters with soils and elements. *Soil Science* 148:370-379.

Cappon, C. (1984). Content and Chemical Form of mercury and selenium in soil, sludge and fertilizer materials. Water, Air, Soil Pollut 22:95-104.

Cappon, C.J. (1987). Uptake and Speciation of Mercury and Selenium in Vegetable CropsGrown on Compost-Treated Soil. *Water, Air, Soil Poll.* 34:353-361.

Dooley, J. H. (1992). Natural Sources of Mercury in the Kirkwood-Cohansey Aquifer System of the New Jersey Coastal Plain. New Jersey Geological Survey, Report 27.

Jensen, A. and Jensen, A. Historical deposition rates of mercury in Scandavia estimated by dating and measurement of mercury in cores of peat bogs, *Water, Air, and Soil Pollution* 56:759-777.

Johansson, K., M. Aastrup, A. Andersson, L. Bringmark, and A. Iverfeldt (1991). Mercury in Swedish Forest Soils and Waters - Assessment of Critical Load, *Water, Air, and Soil Pollution* 56:267-281.

Lindqvist, O., K. Johansson, M. Aastrup, A. Andersson, L. Bringmark, G. Hovsenius, L. Hakanson, A. Iverfeldt, M. Meili, and B. Timm (1991). Mercury in the Swedish Environment - Recent Research on Causes, Consequences and Corrective Methods. *Water, Air and Soil Poll.* 55:(all chapters)

Meili, M. (1991a). The Coupling of Mercury and Organic Matter in the Biogeochemical Cycle - Towards a Mechanistic Model for the Boreal Forest Zone, *Water, Air, and Soil Pollution* 56:333-347.

Meili, M., A. Iverfeldt and L. Hakanson (1991). Mercury in the Surface Water of Swedish Forest Lakes - Concentrations, Speciation and Controlling Factors, *Water, Air, and Soil Pollution* 56:439-453.

Meili, M. (1991b). Fluxes, Pools, and Turnover of Mercury in Swedish Forest Lakes, *Water, Air, and Soil Pollution* 56:719-717.

Moore, J.W. and S. Ramomodoray (1984). *Heavy Metal in Natural Waters - Applied Monitoring in Impact Assessment.* New York, Springer-Varlag.

Nriagu, J.O. (1979). The Biogeochemistry of Mercury in the Environment. Elsevier/North Holland. Biomedical Press: New York.

Parks, J.W., A. Lutz, and J.A. Sutton (1989). Water Column Methylmercury in the Wabigoon/ English River-Lake System: Factors Controlling Concentrations, Speciation, and Net Production. *Can. J. Fisher. Aq. Sci.* 46:2184-2202.

Rai, D. and J.M. Zachara (1984). Chemical attenuations, coefficients and constants in leachate migration, v.1, A Critical Review. EA-3356, v.1, Research Pro.

Revis, N.W., T.R. Osborne, G. Holdsworth, and C. Hadden (1990). Mercury in Soil: A Method for Assessing Acceptable Limits. *Arch. Environ. Contam. Toxicol.* 19:221-226.

Robinson, K.G. and M.S. Shuman (1989). Determination of mercury in surface waters using an optimized cold vapor spectrophotometric technique. *International Journal of Environmental Chemistry* 36:111-123.

Sorensen, J., G.E. Glass, K.W. Schmidt, J.K Huber, and G.R. Rapp, Jr. (1990). Airborne Mercury Deposition and Watershed Characteristics in Relation to Mercury Concentrations in Water, Sediment, Plankton, and Fish of Eighty Northern Minnesota Lakes, *Environ. Sci. Technol.* 24:1716-1727.

U.S. EPA. 1990. Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions. Interim final. EPA/600/690/003.

U.S. EPA. 1993. Draft addendum to Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions. EPA/600-93/003 November 1993.

U.S. EPA. 1994. Personal communication with R.A. Ambrose, U.S. EPA, Athens, GA.

Watras, C.J. and N.S. Bloom (1992). Mercury and Methylmercury in Individual Zooplankton: Implications for Bioaccumulation. *Limnol. Oceanogr.* 37(6):1313-1318.

Wiener, J.G., W.F. Fitzgerald, C.J. Watras, and R.G. Rada (1990). Partitioning and Bioavailibility of Mercury in an Experimentally Acidified Wisconsin Lake, *Environmental Toxicology and Chemistry*, Vol.9, pp. 909-918.

Wilken, R.D. and H. Hintelmann (1991). Mercury and Methylmercury in Sediments and Suspended particles from the River Elbe, North Germany. *Water, Air and Soil Poll.* 56:427-437.

APPENDIX D

DESCRIPTION OF EXPOSURE MODELS

TABLE OF CONTENTS

Page

D.1 Description of RELMAP Mercury Modelling D-1
D.1.1 History and Background Information D-1
D.1.2 RELMAP Modeling Strategy for Atmospheric Mercury D-1
D.1.3 Model Parameterizations D-5
D.1.4 Discussion of RELMAP Modeling Uncertainties D-10

D.2 Description of COMPDEP Air Dispersion Model D-11
D.2.1 Description of the COMPDEP Air Quality Model D-11
D.2.2 Application of the COMPDEP Model for the Exposure Assessment D-30

D.3 Description of the IEM2 Indirect Exposure Methodology D-38
D.3.1 The Terrestrial Equations D-39
D.3.2 The Aquatic Equations D-44
D.3.3 Derivation of Select Equations D-60
D.3.4 Summary of Notation D-64

D.4 References D-67

LIST OF TABLES

		Page
D-1	Emission Speciation Profiles for the Point Source Types Defined	D-3
D-2	Dry Deposition Velocity (cm/s) for Divalent Mercury (Hg^{2+})	D-8
D-3	Wind Speeds Used for Each Pasquill Stability Category in the CARB Subroutine Calculations	D-9
D-4	Roughness Length Used for Each Land-Use Category in the CARB Subroutine Calculations	D-9
D-5	Classes of Atmospheric Stability and Associated Vertical Temperature Distribution	D-12
D-6	Pasquill Turbulence Types and Corresponding Atmospheric Conditions (from Gifford, 1976)	D-13
D-7	Meteorological Conditions Defining Pasquill Turbulence Types	D-13
D-8	Wind Profile Exponents Used In Scoping Study	D-14
D-9	The Three Main Cases for Determining Air Concentration With Plume Depletion Effects	D-19
D-10	Parameters Used to Calculate Horizontal Dispersion Parameter s_y in COMPDEP (Turner, 1970)	D-20
D-11	Parameters Used to Calculate Vertical Dispersion Parameter s_z in COMPDEP (Turner, 1970)	D-21
D-12	Example of Precipitation Scavenging Coefficients (per second) in COMPDEP	D-24
D-13	Terrain Adjustment Factors Used in Calculating Terrain-Dependent Effective Stack Height	D-25
D-14	Coefficients Used to Calculate Lateral Virtual Distances for Pasquill Dispersion Rates	D-28
D-15	Precipitation Intensities Considered by COMPDEP	D-29
D-16	Description of Meteorological Files Used to Make Input Files for COMP	D-32
D-17	Divalent Mercury Vapor Seasonally-Averaged Deposition Velocities (cm/s)	D-33
D-18	Air Modeling Parameter Values Used In the Exposure Assessment: Generic Parameters	D-35
D-19	Model Plant Parameter Values Used in COMPDEP	D-36
D-20	Model Plant Mercury Speciation of Emissions	D-37

LIST OF FIGURES

		Page
D-1	Overview of the IEM2 Watershed Modules	D-38
D-2	Overview of the IEM2 Soils Processes	D-40
D-3	Overview of the IEM2 Water Body Processes	D-47
D-4	IEM2 Steady State Sediment Balance in Water Bodies	D-48

D.1 Description of RELMAP Mercury Modelling

D.1.1 History and Background Information

During the mid-1970's, SRI International developed a Lagrangian puff air pollution model called the European Regional Model of Air Pollution (EURMAP) for the Federal Environment Office of the Federal Republic of Germany (Johnson et al. 1978). This regional model simulated monthly sulfur dioxide (SO_2) and sulfate (SO_4^{2-}) concentrations, wet and dry deposition patterns, and generated matrices of international exchanges of sulfur for 13 countries of western and central Europe. In the late-1970's, the U.S. EPA sponsored SRI International to adapt and apply EURMAP to eastern North America. The adapted version of this model, called Eastern North American Model of Air Pollution (ENAMAP), also calculated monthly SO_2 and SO_4^{2-} concentrations, wet and dry deposition patterns, and generated matrices of interregional exchanges of sulfur for a user-defined configuration of regions (Bhumralker et al., 1980; Johnson, 1983). In the early-1980's, U.S. EPA modified and improved the ENAMAP model to increase its flexibility and scientific credibility.

By 1985, simple parameterizations of processes involving fine (diameters < 2.5 μm) and coarse (2.5 μm < diameters < 10.0 μm) particulate matter were incorporated into the model. This version of the model, renamed the Regional Lagrangian Model of Air Pollution (RELMAP), is capable of simulating concentrations and wet and dry deposition patterns of SO_2, SO_4^{2-} and fine and coarse particulate matter and can also generate source-receptor matrices for user defined regions. In addition to the main model program, the complete RELMAP modeling system includes 19 preprocessing programs that prepare gridded meteorological and emissions data for use in the main program. The RELMAP code was developed using FORTRAN on a Sperry-UNIVAC 1100/82 computing system. It has since been migrated and adapted to operate on other computing systems. Currently, the RELMAP is operated by U.S. EPA's Atmospheric Research and Exposure Assessment Laboratory on DEC VAX and CRAY computing systems, and a test version has recently been installed on a DEC 3000 AXP (Alpha) workstation. The simulations for the Mercury Study Report to Congress were performed on a CRAY Y-MP supercomputer at the National Environmental Supercomputing Center. A complete scientific specification of the RELMAP as used at U.S. EPA for atmospheric sulfur modeling is provided in *RELMAP: A Regional Lagrangian Model of Air Pollution - User's Guide* (Eder et al., 1986). Section D.1.2.1 discusses the modifications made to the original sulfur version of RELMAP to enable the simulation of atmospheric mercury.

D.1.2 RELMAP Modeling Strategy for Atmospheric Mercury

D.1.2.1 Introduction

Previous versions of RELMAP have been described by Eder et al. (1986) and Clark et al. (1992). The goal of the current effort was to model the emission, transport, and fate of airborne mercury over the continental U.S. for the year of 1989. Modifications to the RELMAP for atmospheric mercury simulation were heavily based on recent Lagrangian model developments in Europe (Petersen et al., 1995). The mercury version of RELMAP was developed to handle three species of mercury: elemental (Hg^0), divalent (the mercuric ion, Hg^{2+}) and particulate mercury (Hg_{part}), and also carbon soot. Recent experimental work indicates that ozone (Munthe, 1992) and carbon soot (Iverfeldt, 1991; Brosset and Lord, 1991; Lindqvist et al., 1991) are both important in determining the wet deposition of Hg^0. Carbon soot, or total carbon aerosol, was included as a modeled pollutant in the mercury version of RELMAP to provide necessary information for the Hg^0 wet deposition parameterization. Observed O_3 air concentration data were obtained from the Agency's Aerometric Information Retrieval System (AIRS) data base, and it was not necessary to include O_3 as an explicitly modeled pollutant. Observed O_3 air concentration data were objectively interpolated in time and space for each 3-hour timestep of the model simulation to produce grids of O_3 air

concentration. A minimum O_3 air concentration value of 20 ppb was imposed. Methylmercury was not included in the mercury version of RELMAP because it is not yet known if it has a primary natural or anthropogenic source, or if it is produced in the atmosphere.

RELMAP may be run in either of two modes. In the field mode, wet deposition, dry deposition, and air concentrations are computed at user-defined time intervals. In the source-receptor mode, RELMAP also computes the contribution of each source cell to the deposition and concentration at each receptor cell. For mercury, only the field mode of RELMAP operation was used. With over 10,000 model cells in the high-resolution receptor grid and a significant fraction of these cells also emitting mercury, the data accounting task of a source-receptor run for all mercury sources could not be performed with the computing resources and time available.

Unless specified otherwise in the following sections, the modeling concepts and parameterizations described by Eder et al. (1986) were preserved for the RELMAP mercury modeling study.

D.1.2.2 Physical Model Structure

Because of the long atmospheric residence time of mercury, long range transport of the majority of mercury emitted was expected. RELMAP simulations were originally limited to the area bounded by 25 and 55 degrees north latitude and 60 and 105 degrees west longitude and had a minimum spatial resolution of 1 degree in both latitude and longitude. For the Mercury Study Report to Congress, the western limit of the RELMAP modeling domain was moved out to 130 degrees west longitude, and the modeling grid resolution was reduced to ½ degree longitude by ⅓ degree latitude (approximately 40 km square) to provide high-resolution coverage over the entire continental U.S.

Since the descriptive document by Eder et al. (1986) was produced, the original 3-layer puff structure of the RELMAP has been replaced by a 4-layer structure. The following model layer definitions were used for the RELMAP mercury simulations:

Layer 1 top	-	30 to 50 meters above the surface (season-dependent)
Layer 2 top	-	200 meters above the surface
Layer 3 top	-	700 meters above the surface
Layer 4 top	-	700 to 1500 meters above the surface (month-dependent)

D.1.2.3 Mercury Emissions

Area source emissions were introduced into the model in the lowest layer. Point source emissions were introduced into model layer 2 to account for the effective stack height of the point source type in question. Effective stack height is the actual stack height plus the estimated plume rise. The layer of emission is inconsequential during the daytime when complete vertical mixing is imposed throughout the 4 layers. At night, since there is no vertical mixing, area source emissions to layer 1 are subject to dry deposition while point source emissions to layer 2 are not. Large industrial emission sources and sources with very hot stack emissions tend to have a larger plume rise, and their effective stack heights might actually be larger than the top of layer 2. Since, however the layers of the pollutant puffs remain vertically aligned during advection, the only significant process effected by the layer of emission is nighttime dry deposition.

Mercury emissions data were grouped into seven different point-source types and a general area-source type. The area source emissions data describe those sources that are too small to be accounted for individually in pollutant emission surveys. For the RELMAP mercury modeling study, area sources were assumed to emit mercury entirely in the form of Hg^0 gas, while the seven point

source types were each assigned particular mercury speciation profiles. These speciation profiles defined the estimated fraction of mercury emitted as Hg^0, Hg^{2+}, or Hg_{part}. Since there remains considerable uncertainty as to the actual speciation factors for each point source type, an alternate emission speciation was simulated in addition to the base speciation in order to test the sensitivity of the RELMAP results to the speciation profiles used. The base-case and alternate speciation profiles used for this study are shown in Table D-1. The total (non-speciated) mercury emissions inventory used is that described in Volume II of this Report. Gridded fields of total Hg^0, Hg^{2+} and Hg_{part} point source emission rates and Hg^0 area source emission rates were produced and used as input to the RELMAP model simulation.

Table D-1
Emission Speciation Profiles for the Point Source Types Defined

Point Source Type	Base-Case Speciation (%)			Alternate Speciation (%)		
	Hg^0 [a]	Hg^{2+} [b]	Hg_p [c]	Hg^0 [a]	Hg^{2+} [b]	Hg_p [c]
Electric Utility Boilers	50	30	20	50	0	50
Non-utility Fossil Fuel Combustion						
Municipal Waste Combustion	20	60	20	20	0	80
Medical Waste Incineration						
Non-ferrous Metal Smelting	85	10	5	85	0	15
Chlor-alkali Factories	70	30	0	70	0	30
Other Point Sources	80	10	10	80	0	20

[a] Hg^o symbolizes elemental mercury
[b] Hg^{2+} symbolizes divalent mercury
[c] Hg_p symbolizes particle bound mercury

Global-scale natural and recycled anthropogenic emissions were accounted for by assuming an ambient atmospheric concentration of Hg^0 gas of 1.6 ng/m^3. This use of a constant background concentration to account for global-scale natural and anthropogenic emissions is the same technique used by Petersen et al. (1995). The deposition parameterizations described in section D.1.3.1 were used to simulate the scavenging of Hg^0 from this constant ambient concentration throughout the entire 3-dimensional model domain. The result was used as an estimate of the deposition of mercury from all natural sources and anthropogenic sources outside the model domain.

D.1.2.4 Carbon Aerosol Emissions

Penner et al. (1993) concluded that total carbon air concentrations are highly correlated with sulfur dioxide (SO_2) air concentrations from minor sources. They concluded that the emissions of total carbon and SO_2 from minor point sources are correlated as well, since both pollutants result from the combustion of fossil fuel. Their data indicate a 35% proportionality constant for total carbon air concentrations versus SO_2 air concentrations. The RELMAP mercury model estimated total carbon aerosol emissions using this 35% proportionality constant and SO_2 emissions data for minor sources obtained by the National Acidic Precipitation Assessment Program (NAPAP) for the year 1988. Much of these SO_2 emissions data had been previously analyzed for use by the Regional Acid Deposition Model (RADM). For the portion of the RELMAP mercury model domain not covered by the RADM domain, state by state totals of SO_2 emissions were apportioned to the county level on the basis of weekday vehicle-miles-traveled data since recent air measurement studies have indicated that aerosol elemental carbon can be attributed mainly to transportation source types (Keeler et al., 1990). The county level data were then apportioned by area to the individual RELMAP grid cells. Total carbon soot was assumed to be emitted into the lowest layer of the model.

D.1.2.5 Ozone Concentration

Ozone concentration data were obtained from U.S. EPA's Aerometric Information Retrieval System (AIRS) and the Acidmodes experimental air sampling network. AIRS and Acidmodes data were available hourly. Any observations of ozone concentration below 20 ppb were treated as missing. For each RELMAP grid cell, the ozone concentrations were computed for the two mid-day time steps by using the mean concentration value during two corresponding time periods (1000-1300 and 1400-1600 local time). The mean of these two mid-day values was used to estimate the ozone concentration for the time steps after 1600 local time and before 1000 local time the next morning. This previous-day average was used at night since ground-level ozone data are not valid for the levels aloft, where the wet removal of elemental mercury was assumed to be occurring. Finally, an objective interpolation scheme was used to produce complete ozone concentration grids for each time step, with a minimum value of 20 ppb imposed.

D.1.2.6 Lagrangian Transport and Deposition

In the model, each pollutant puff begins with an initial mass equal to the total emission rate of all sources in the source cell multiplied by the model time-step length. For mercury, as for most other pollutants, emission rates for each source cell were defined from input data, and a time step of three hours was used. The initial horizontal area of each puff was set to 1200 km^2, instead of the standard initial size of 2500 km^2, in order to accommodate the finer grid resolution used for the mercury modeling study; however, the standard horizontal expansion rate of 339 km^2 per hour was not changed. Although each puff was defined with four separate vertical layers, each layer of an individual puff was advected through the model cell array by the same wind velocity field. Thus, the layers of each puff always remained vertically stacked. Wind field initialization data for a National Weather Service prognostic model, the Nested Grid Model (NGM), were obtained from the NOAA Atmospheric Research Laboratory for the entire year of 1989. Wind analyses for the σ_p=0.897 vertical level of the NGM were used to define the translation of the puffs across the model grid, except during the months of January, February, and December, when the σ_p=0.943 vertical level was used to reflect a more shallow mixed layer. σ_p is a pressure-based vertical coordinate equal to $(p-p_{top})/(p_{surface}-p_{top})$.

Pollutant mass was removed from each puff by the processes of wet deposition, dry deposition, diffusive air exchange between the surface-based mixed layer and the free atmosphere and, in the case of reactive species, chemical transformation. The model parameterizations for these processes are discussed in Section D.1.3. Precipitation data for the entire year of 1989, obtained from the National

Climatic Data Center, were used to estimate the wet removal of all pollutant species modeled. Wet and dry deposition mass totals were accumulated and average surface-level concentrations were calculated on a monthly basis for each model cell designated as a receptor. Except for cells in the far southwest and eastern corners of the model domain where there were no wind data, all cells were designated as receptors for the mercury simulation. When the mass of pollutant on a puff declines through deposition, vertical diffusion or transformation to a user-defined minimum value, or when a puff moves out of the model grid, the puff and its pollutant load is no longer tracked. The amount of pollutant in the terminated puff is taken into account in monthly mass balance calculations so that the integrity of the model simulation is assured. Output data from the model includes monthly wet and dry deposition totals and monthly average air concentration for each modeled pollutant, in every receptor cell.

D.1.3 Model Parameterizations

D.1.3.1 Chemical Transformation and Wet Deposition

The simplest type of pollutant to model with RELMAP is the inert type. To model inert pollutants, one can simply omit chemical transformation calculations for them, and not be concerned with chemical interactions with the other chemical species in the model. In the mercury version of RELMAP, particulate mercury and total carbon were each modeled explicitly as inert pollutant species. Reactive pollutants are normally handled by a chemical transformation algorithm. RELMAP was originally developed to simulate sulfur deposition, and the algorithm for transformation of sulfur dioxide to sulfate was independent of wet deposition. For gaseous mercury, however, the situation is more complex. Since there are no gaseous chemical reactions of mercury in the atmosphere which appear to be significant (Petersen et al., 1995), for this modeling study mercury was assumed to be reactive only in the aqueous medium. Elemental mercury has a very low solubility in water, while oxidized forms of mercury and particle bound mercury readily find their way into the aqueous medium through dissolution and particle scavenging, respectively. Worldwide observations of atmospheric mercury, however, indicate that particulate mercury is generally a minor constituent of the total mercury loading (Iverfeldt, 1991) and that gaseous elemental mercury (Hg^0) is, by far, the major component. Swedish measurements of large north-to-south gradients of mercury concentration in rainwater without corresponding gradients of atmospheric mercury concentration suggest the presence of physical and chemical interactions with other pollutants in the precipitation scavenging process (Iverfeldt, 1991). Aqueous chemical reactions incorporated into the mercury version of RELMAP were based on research efforts in Sweden (Iverfeldt and Lindqvist, 1986; Lindqvist et al., 1991; Munthe et al., 1991; Munthe and McElroy, 1992; Munthe, 1992) and Canada (Schroeder and Jackson, 1987; Schroeder et al., 1991).

Unlike other pollutants that have been modeled with RELMAP, mercury has wet deposition and chemical transformation processes that are interdependent. A combined transformation/wet-removal scheme proposed by Petersen et al. (1995) was used. In this scheme, the following aqueous chemical processes were modeled when and where precipitation is present.

1) oxidation of dissolved Hg^0 by ozone yielding Hg^{2+}
2) catalytic reduction of this Hg^{2+} by sulfite ions
3) adsorption of Hg^{2+} onto carbon soot particles suspended in the aqueous medium

$$W(Hg^0) = \frac{k_1}{k_2} \cdot \frac{1}{H_{Hg}} \cdot [O_3]_{aq} \cdot (1 + K_3 \cdot \frac{c_{soot}}{r})$$

Petersen et al. (1995) shows that these three simultaneous reactions can be considered in the formulation of a scavenging ratio for elemental mercury gas as follows:
where,

k_1 is the second order rate constant for the aqueous oxidation of Hg^0 by O_3 equal to . 4.7 x 10^7 $M^{-1}s^{-1}$,

k_2 is the first order rate constant for the aqueous reduction of Hg^{2+} by sulfite ions . . . equal to 4.0 x 10^{-4} s^{-1},

H_{Hg} is the dimensionless Henry's Law coefficient for Hg^0 (0.18 in winter, 0.22 in . . . spring and autumn, and 0.25 in summer as calculated from Sanemasa (1975)),

$[O_3]_{aq}$ is the aqueous concentration of ozone,

K_3 is a model specific adsorption equilibrium constant (5.0 x 10^{-6} m^4g^{-1}),

c_{soot} is the total carbon soot aqueous concentration, and

r is the assumed mean radius of soot particles (5.0 x 10^{-7} m).

$[O_3]_{aq}$ is obtained from this equation.

$$[O_3]_{aq} = \frac{[O_3]_{gas}}{H_{O3}}$$

where H_{O3} is the dimensionless Henry's Law coefficient for ozone (0.448 in winter, 0.382 in spring and autumn, and 0.317 in summer as calculated from Seinfeld (1986)). c_{soot} is obtained from the simulated atmospheric concentration of total carbon aerosol using a scavenging ratio of 5.0 x 10^5.

The model used by Petersen et al. (1995) defined one-layer cylindrical puffs, and the Hg^0 scavenging layer was defined as the entire vertical extent of the model. The RELMAP defines 4-layer puffs to allow special treatment of surface-layer and nocturnal inversion-layer processes. It was believed that, due to the low solubility of Hg^0 in water, the scavenging process outlined above would only take place effectively in the cloud regime, where the water droplet surface-area to volume ratio is high, and not in falling raindrops. Thus the Hg^0 wet scavenging process was applied only in the top two layers on RELMAP, which extends from 200 meters above the surface to the model top.

For the modeling study described in Petersen et al. (1995), the wet deposition of Hg^{2+} was treated separately from that of Hg^0. Obviously, any Hg^{2+} dissolved into the water droplet directly from the air could affect the reduction-oxidation balance between the total concentration of Hg^0 and Hg^{2+} in the droplet. Since the solubility and scavenging ratio for Hg^{2+} is much larger than that for Hg^0, and since air concentrations of Hg^0 are typically larger than those of Hg^{2+}, separate treatment of Hg^{2+} wet deposition was deemed acceptable. Thus, process 2 above was only considered as a moderating factor for the oxidation of dissolved Hg^0.

In the exposure analysis in Volume III, there was no attempt to develop a new interacting chemical mechanism for simultaneous Hg^0 and Hg^{2+} wet deposition. Although Hg^{2+} was recognized as a reactive species in aqueous phase redox reactions, it was, in essence, modeled as an inert species just like particulate mercury and total carbon soot. With the rapid rate at which the aqueous Hg^{2+} reduction reaction is believed to occur in the presence of sulfite, it is possible that an interactive cloud-water chemical mechanism might produce significant conversion of scavenged Hg^{2+} to Hg^0, with possible release of that Hg^0 into the gaseous medium.

Wet deposition of Hg^{2+}, particulate mercury, and total carbon soot in the mercury version of RELMAP were modeled with the same scavenging ratios used by Petersen et al. (1995). The gaseous nitric acid scavenging ratio of 1.6 x 10^{-6} has been applied for Hg^{2+} since the water solubilities of these two pollutant species are similar. For particulate mercury, a scavenging ratio of 5.0 x 10^{-5} was used,

based on experiences in long-range modeling of lead in northern Europe. As previously mentioned, a scavenging ratio of 5.0×10^{-5} was also used for total carbon soot. These scavenging ratios for Hg^{2+}, particulate mercury, and total carbon soot were applied to all four layers of the RELMAP in the calculation of pollutant mass scavenging by precipitation.

D.1.3.2 Dry Deposition

Recent experimental data indicate that elemental mercury vapor does not exhibit a net dry depositional flux to vegetation until the atmospheric concentration exceeds a rather high compensation point of around 10 ng/m^3 (Hanson et al., 1994). This compensation point is apparently dependent on the surface or vegetation type and represents a balance between emission from humic soils and dry deposition to leaf surfaces (Lindberg et al., 1992). Since the emission of mercury from soils was accounted for with a global-scale ambient concentration and not an actual emission of Hg^0, for consistency, there was no explicit simulation of the dry deposition of Hg^0.

For Hg^{2+} during daylight hours, a dry deposition velocity table previously developed based on HNO_3 data (Walcek et al., 1985; Wesely, 1986) was used. The dry deposition characteristics of HNO_3 and Hg^{2+} should be similar since their water solubilities are similar. This dry deposition velocity data, shown in Table D-2, provided season-dependent values for 11 land-use types under six different Pasquill stability categories. Based on the predominant land-use type and climatological Pasquill stability estimate of each RELMAP grid cell, and the season for the month being modeled, the dry deposition velocity values shown in Table D-2 were used for the daytime only. For nighttime, a value of 0.3 cm/s was used for all grid cells since the RELMAP does not have the capability of applying land-use dependent dry deposition at night. Since the nighttime dry deposition was applied only to the lowest layer of the model and no vertical mixing is assumed for nighttime hours, all Hg^{2+} modeled to be quickly depleted from the lowest model layer by larger dry deposition velocities.

For Hg_{part}, Petersen et al. (1995) used a dry deposition velocity of 0.2 cm/s at all times and locations. Lindberg et al. (1991) suggests that the dry deposition of Hg_{part} seems to be dependent on foliar activity. During the daylight hours of spring, summer, and autumn, a dry deposition velocity of 0.11 cm/s was used for Hg_{part}, except for model cells with predominant surface characteristics of water, barren, and rocky terrain where 0.02 cm/s was used. At night and at all hours during the winter, all cells used 0.02 cm/s as the dry deposition velocity for Hg_{part}. Lindberg et al. (1991) suggested a value of 0.003 cm/s for non-vegetated land, but since the RELMAP can not model land-use dependent dry deposition at night, the value of 0.02 cm/s was used for these cells by necessity.

Table D-2
Dry Deposition Velocity (cm/s) for Divalent Mercury (Hg^{2+})

Season	Land-Use Category	Pasquill Stability Category					
		A	B	C	D	E	F
Winter	Urban	4.83	4.80	4.61	4.30	2.79	0.36
	Agricultural	1.32	1.30	1.20	1.05	0.46	0.15
	Range	1.89	1.86	1.73	1.52	0.73	0.19
	Deciduous Forest	3.61	3.57	3.34	3.02	1.68	0.29
	Coniferous Forest	3.61	3.57	3.34	3.02	1.68	0.29
	Mixed Forest/Wetland	3.49	3.46	3.27	2.99	1.77	0.29
	Water	1.09	1.07	0.98	0.85	0.38	0.13
	Barren Land	1.16	1.14	1.06	0.92	0.39	0.31
	Non-forested Wetland	2.02	2.00	1.89	1.70	0.96	0.21
	Mixed Agricultural/Range	1.62	1.60	1.48	1.30	0.60	0.17
	Rocky Open Areas	1.98	1.95	1.81	1.58	0.73	0.20
Spring	Urban	4.59	4.54	4.35	4.05	2.49	0.36
	Agricultural	1.60	1.56	1.46	1.28	0.53	0.18
	Range	1.49	1.46	1.36	1.19	0.48	0.17
	Deciduous Forest	3.42	3.36	3.13	2.81	1.42	0.29
	Coniferous Forest	3.42	3.36	3.13	2.81	1.42	0.29
	Mixed Forest/Wetland	3.28	3.23	3.05	2.78	1.55	0.29
	Water	0.98	0.96	0.89	0.77	0.31	0.13
	Barren Land	1.05	1.04	0.97	0.85	0.30	0.13
	Non-forested Wetland	1.85	1.82	1.73	1.56	0.84	0.21
	Mixed Agricultural/Range	1.60	1.56	1.46	1.28	0.53	0.18
	Rocky Open Areas	1.84	1.81	1.67	1.46	0.58	0.20
Summer	Urban	4.47	4.41	4.12	3.73	2.07	0.36
	Agricultural	2.29	2.25	2.04	1.76	0.72	0.24
	Range	1.67	1.64	1.48	1.26	0.41	0.19
	Deciduous Forest	3.32	3.26	2.95	2.57	1.04	0.29
	Coniferous Forest	3.32	3.26	2.95	2.57	1.04	0.29
	Mixed Forest/Wetland	3.17	3.12	2.86	2.53	1.27	0.29
	Water	0.92	0.90	0.81	0.69	0.22	0.13
	Barren Land	0.98	0.98	0.89	0.76	0.23	0.13
	Non-forested Wetland	1.91	1.88	1.73	1.52	0.77	0.22
	Mixed Agricultural/Range	1.90	1.87	1.69	1.44	0.52	0.21
	Rocky Open Areas	1.95	1.91	1.71	1.46	0.42	0.21
Autumn	Urban	4.64	4.59	4.35	4.05	2.49	0.36
	Agricultural	2.02	1.98	1.81	1.60	0.73	0.21
	Range	1.78	1.74	1.59	1.40	0.60	0.19
	Deciduous Forest	3.46	3.40	3.13	2.81	1.42	0.29
	Coniferous Forest	3.46	3.40	3.13	2.81	1.42	0.29
	Mixed Forest/Wetland	3.32	3.27	3.05	2.78	1.55	0.29
	Water	1.00	0.98	0.89	0.77	0.31	0.13
	Barren Land	1.07	1.06	0.97	0.85	0.30	0.13
	Non-forested Wetland	1.88	1.86	1.73	1.56	0.84	0.21
	Mixed Agricultural/Range	1.93	1.90	1.74	1.53	0.68	0.20
	Rocky Open Areas	1.97	1.94	1.76	1.54	0.63	0.20

For total carbon soot, daytime dry deposition velocities were calculated using a FORTRAN subroutine developed by the California Air Resources Board (CARB, 1987). A particle density of 1.0 g/cm^3 and radius of 0.5 μm was assumed. Table D-3 shows the wind speed (u) used for each Pasquill stability category in the calculation of deposition velocity from the CARB subroutine, while Table D-4 shows the roughness length (z_0) used for each land-use category. For nighttime, a dry deposition velocity of 0.07 cm/s was used for all seasons and land-use types.

Table D-3
Wind Speeds Used for Each Pasquill Stability Category in the CARB Subroutine Calculations

Stability Category	Wind Speed (m/s)
A	10.0
B	5.0
C	5.0
D	2.5
E	2.5
F	1.0

Table D-4
Roughness Length Used for Each Land-Use Category in the CARB Subroutine Calculations

Land-Use Category	Roughness Length (meters)	
	autumn-winter	spring-summer
Urban	0.5	0.5
Agricultural	0.15	0.05
Range	0.12	0.1
Deciduous Forest	0.5	0.5
Coniferous Forest	0.5	0.5
Mixed Forest/Wetland	0.4	0.4
Water	10^{-6}	10^{-6}
Barren Land	0.1	0.1
Non-forested Wetland	0.2	0.2
Mixed Agricultural/Range	0.135	0.075
Rocky Open Areas	0.1	0.1

The RELMAP assumes instantaneous vertical mixing of all pollutants through the entire depth of the model. For grid cells with significant emission rates, this results in an underestimation of the ground-level concentration and therefore an underestimation of the dry deposition rate for mercury species emitted near the ground. To remedy this, the model used a local dry deposition factor for Hg_{part} in a similar manner as Petersen et al. (1995). This local deposition factor was 0.5, meaning that one-half of the Hg_{part} emissions from a grid cell were assumed to dry deposit within that grid cell by processes not otherwise simulated by the dry deposition parameterization. There was no application of a local deposition factor for Hg^{2+} since the majority of its emission was assumed to be from elevated point sources.

D.1.3.3 Vertical Exchange of Mass with the Free Atmosphere

Due to the long atmospheric lifetime of mercury, the RELMAP was adapted to allow a treatment of the exchange of mass between the surface-based mixed layer and the free atmosphere above. As a first approximation, a pollutant depletion rate of 5 percent per 3-hour timestep was chosen to represent this diffusive mass exchange. When compounded over a 24-hour period, this depletion rate removes 33.6% of an inert, non-depositing pollutant. Since all three forms of the modeling mercury deposit to the surface to some degree, their effective diffusion rate out of the top of the model is less than 33.6% per day.

D.1.4 Discussion of RELMAP Modeling Uncertainties

D.1.4.1 Vertical Model Domain

The RELMAP model top is defined to be the maximum vertical extent of the convectively driven mixed layer. Monthly values defined from mixed-layer-height climatology are rough estimates of a meteorological phenomenon that may not exist in many situations. Although a surface-based mixed layer may be well defined, pollutants that persist in the atmosphere for long periods of time are certain to mix to some degree into the free atmosphere above the mixed-layer top. Chlorofluorocarbon (CFC) compounds are an extreme example of this possibility. Elemental mercury deposits relatively slowly through precipitation processes due to its low water solubility, and its dry deposition appears to be minimal since it is in vapor form under normal atmospheric conditions. In fact, pollutant mass balance accounting information from the RELMAP mercury simulation indicated that approximately 75% of all elemental mercury emitted was transported out of the model domain before it was wet or dry deposited. Elemental mercury appears to be quite persistent in the atmosphere.

Since the RELMAP does not simulate the flux of air or pollutant through the height of the mixed layer, which is fixed for each monthly simulation, the use of horizontally divergent/convergent wind fields to define the motion of the pollutant puffs can sometimes result in unrealistic instantaneous concentration fields. Horizontally convergent winds will tend to concentrate puffs at the point of convergence, resulting in high modeled concentrations when the effects of the puffs are summed together. Ordinarily, horizontal convergence in the surface-based mixed layer would push the mixed-layer top higher into the atmosphere as a result of the incompressible nature of air in general atmospheric motion. This higher mixed-layer top would compensate for the greater pollutant mass loading per unit area from the converging puffs, keeping the resulting pollutant concentration more constant. The RELMAP was not designed to provide instantaneous realizations of pollutant concentration fields. Rather, it was designed for seasonal and annual simulations where the total effects of convergent and divergent wind fields can balance one another. There does exist some uncertainty, however, as to whether this balance actually occurs in all situations.

D.1.4.2 Aqueous Chemistry

The aqueous reduction-oxidation chemistry mechanism in the mercury version of RELMAP was applied only to the Hg^0 dissolved from the ambient air into the water droplet. Where significant concentrations of Hg^{2+} from emissions exist in the ambient air, this Hg^{2+} could be dissolved into the water droplet along with the Hg^0 and inhibit the scavenging of Hg^0. The RELMAP results described above indicate that Hg^{2+} air concentrations are certainly lower than those for Hg^0 at the length scales of the RELMAP grid cells; however, the magnitude of the effect of ambient Hg^{2+} on the Hg^0 oxidation scavenging is not yet well understood.

Another source of modeling uncertainty in aqueous chemistry relates to the fact that the aqueous chemical mechanisms were invoked only when and where precipitation was known to occur,

and precipitation fields were only defined over land areas where precipitation observations were available. Significant wet transformation and removal of mercury may occur over oceanic areas were precipitation observations are not available, and it is possible that significant aqueous chemistry is occurring in non-precipitating clouds.

D.1.4.3 Transport and Diffusion

Since the RELMAP simulates transport and diffusion only in the surface-based mixed layer and vertical wind shear is small when the mixed-layer is well defined, under ideal conditions transport and diffusion are handled adequately. When the surface-based mixed layer is not well defined, vertical gradients in the speed and/or direction of the wind may be present which cannot be represented by the motion of individual Lagrangian puffs whose layers remain vertically stacked. There are two techniques that might be used to represent vertical wind shear in the RELMAP: puff splitting and wind-shear-dependent puff expansion. Due to computational limits and scheduling constraints, these were not attempted. The most complete solution to the problem of vertical wind shear is the use of a Eulerian reference frame for numerical modeling. The Atmospheric Characterization and Modeling Division of U.S. EPA's Atmospheric Research and Exposure Assessment Laboratory has proposed the development of a Toxics Linear Chemistry Model (TLCM) using the Eulerian reference frame of the Regional Acid Deposition Model (RADM). The TLCM could be operational within two years.

D.1.4.4 Boundary fluxes of pollutants

Due to the fact that RELMAP simulates atmospheric pollutant loading as the combined effect of a population of discrete Lagrangian puffs, and the fact that elemental mercury gas has a long residence time in the atmosphere, natural mercury emissions from the oceans and land surfaces could not be explicitly modeled. Given the general west-to-east wind flow at the latitudes of the continental U.S., simulated puffs of natural mercury emissions could be emitted from all grid cells, but their effects would be artificially concentrated in the eastern sections of the model domain. The only puffs that could impact the western areas would be those originating from the far western grid cells, while the eastern areas could be impacted by puffs from all parts of the model domain. The use of a Eulerian-type model would allow the definition of boundary fluxes of pollutant based on larger-scale model results or assumed background concentration levels.

D.2 Description of COMPDEP Air Dispersion Model

D.2.1 Description of the COMPDEP Air Quality Model

General references for this section are Overcamp (1977), Rao (1981) and U.S. EPA (1992).

The COMPDEP model uses hourly meteorological data to estimate air concentrations and deposition fluxes from a point source. In this section a summary description of the model is presented. In Section D.2.2.1, specific modifications made for this assessment are discussed.

D.2.1.1 Atmospheric Stability and System Used in COMPDEP

After pollutants are emitted from a source, they are diluted with ambient air. The degree of dispersion is a function of wind speed and the level of turbulence. In general, higher wind speeds or turbulence result in lower air concentrations. The amount of turbulence is quantified in terms of the atmospheric stability. A stable atmosphere is one that suppresses vertical motions, hence mitigating turbulence, while an unstable atmosphere is one that enhances turbulence. Atmospheric turbulence *per se* is difficult and expensive to measure (Randerson, 1984), and it is usually estimated from other more

easily measurable quantities. In particular, the stability of the atmosphere is typically characterized by the vertical temperature profile of the atmosphere.

For an isolated parcel of air in which no heat is transferred in or out (adiabatic), one can show using the first law of thermodynamic and the hydrostatic equation (see, e.g., Hannat et al. 1982, pp. 2-3) that there is a 1 degree (C) decrease for every 100 m increase in altitude, and a 1 degree increase for every 100 m decrease. This is called the adiabatic lapse rate.

The atmosphere is not adiabatic as it is both heated and cooled. This results in temperature profiles that differ from the adiabatic profile, and it is this difference that is ultimately responsible for a given atmospheric stability. The three broad classes of stability and their associated temperature profiles are summarized in Table D-5.

Table D-5
Classes of Atmospheric Stability and
Associated Vertical Temperature Distribution

Vertical Temperature Profile	General Result	Class of Stability
Increases with height, or decreases less rapidly than adiabatic lapse rate	Vertical motions inhibited	Stable
Nearly identical with adiabatic rate	No significant buoyant forces	Neutral
Decreases with height faster than adiabatic lapse rate	Vertical motions enhanced	Unstable

It should be noted that any time the temperature increases with altitude, the atmospheric condition is termed an inversion. Because of the associated stability, inversions tend to decrease (Wark and Warner, 1981).

The most widely used scheme of atmospheric stability classification, and that used in COMPDEP, was developed by Pasquill (1961) and modified by Gifford (1961). There are six stability classes, denoted with the letters A through F. In general, classes A through C indicate unstable conditions, D is roughly neutral, and classes E and F represent stable conditions. Table D-6, from Hanna et al., (1982), originally from Gifford (1976), shows the criteria for the different classes.

Table D-6
Pasquill Turbulence Types and
Corresponding Atmospheric Conditions
(from Gifford, 1976)

Pasquill Turbulence Type	Atmospheric Stability Conditions
A	Extremely unstable
B	Moderately unstable
C	Slightly unstable
D	Neutral[a]
E	Slightly stable
F	Moderately stable

[a] Applicable to heavy overcast day or night.

The meteorological conditions that are used to determine the stability class are shown in Table D-7.

Table D-7
Meteorological Conditions Defining Pasquill Turbulence Types
(from Gifford, 1976)

	Daytime Solar Radiation			Nighttime conditions	
Surface wind speed (m/sec)	Strong	Moderate	Slight	Mostly Overcast	Mostly Clear
<2	A	A-B	B		
2-3	A-B	B	C	E	F
3-4	B	B-C	C	D	E
4-6	C	C-D	D	D	D
>6	C	D	D	D	D

From this table, it can be seen that extremely unstable conditions (class A) occur during the day with light winds and moderate to strong solar radiation (necessary conditions for the formation of an unstable temperature profile; (Overcamp, 1977)). Conversely, extremely stable conditions can occur only at night with clear skies and light winds. Hanna et al. (1982) note that some have filled in the blank in Table D-7 with a "G" class, but this has not received wide acceptance (Hanna et al., 1982).

Table D-8
Wind Profile Exponents Used In The Assessment

Stability Category	Wind Profile Exponent
A	0.07
B	0.07
C	0.10
D	0.15
E	0.35
F	0.55

Other stability classification schemes exist. For example, M.E. Smith (1951) proposed a classification scheme that is based on wind direction, and Cramer (1957) advocated a method based on observed wind fluctuations at a height of 10m (often referred to the Brookhaven National Laboratory, or BNL, stability classes). In addition, Irwin (1979) proposed a method of allowing for a continuum of stability, as opposed to a discrete approach such as the Pasquill method. Use of the Pasquill letter classes is common due to their ease of use, and because they have produced satisfactory results.

For this assessment, the stability classes for each hour were estimated with the RAMMET program (Catalano et al., 1987), using hourly surface meteorological data.

Estimation of wind speed is important because higher wind speeds result in greater dispersion and hence reduced concentrations of pollutants. Frictional forces cause the surface wind speed, which is usually the value available, to be lower than the speed at the stack top. A power law wind speed profile is typically used to calculate the change of wind speed with height, and takes the following form:

$$\frac{u_s}{u_{ref}} = \left(\frac{z_s}{z_{ref}}\right)^p$$

where, u_{ref} = wind speed at the reference height
z_{ref} = reference height
u_s = wind speed at the release height
z_s = release height
p = wind speed profile exponent (dependent on atmospheric stability and is between 0 and 1)

In general, a reference anemometer height of 10 m, the standard height for measurement of wind speed and direction by the National Weather Service (Overcamp, 1977), is used.

The wind profile exponents used in COMPDEP are given in Table D-8. These are the default values for rural settings in U.S. EPA (1992) and are based on Irwin (1979). Although default values for urban settings are available in U.S. EPA (1992) as well, due to limitations of the COMPDEP

model it was decided that their use was not warranted for this assessment. The default values for urban settings were about twice as high as the rural ones for classes A, B, and C, which resulted in higher wind speeds at the stack top.

Directional shear with height is not included, which means that the direction of flow is assumed to be the same at all heights over the region. The taller the effective height of a source, the larger the expected error in direction of plume transport (Pierce and Turner, 1980).

D.2.1.2 Plume Rise

A general principle, borne out by the analytic solutions of the diffusion algorithms and known since at least 1917 (Wells, 1917), is that the maximum ground level concentration is inversely proportional to the height of release (Randerson, 1984). Due to the buoyant properties of the stack gases and the velocity of the stack gases emitted, the height of release from a modeling perspective is usually higher than the actual physical height of the stack. This height is called the effective stack height and is the sum of the physical stack height and the rise of the plume.

Due to the sensitivity of the maximum concentrations to the effective stack height, and because the maximum downwind concentration has been historically the output of interest for regulatory agencies, numerous methods exist for estimating plume rise in a variety of conditions. Overcamp (1977) noted that over 50 different plume rise formulas had been published by 1977, and Pasquill (1974) observed that there are many rival formulae from which to choose.

The method used in COMPDEP is based on Briggs (1969, 1972, 1975) and Bowers et al. (1979). With this approach, it must be determined whether thermal buoyancy or vertical momentum is dominating the plume's motion. Estimates of the buoyancy flux (F_b, units of m^4/s^3) and momentum flux (F_m, units of m^4/s^2) are based on Briggs (1975):

$$F_b = g\, v_s\, d_s^2 \left(\frac{T_s - T_a}{4T_s}\right)$$

$$F_m = v_s^2\, d_s^2\, \frac{T_a}{4T_s}$$

where, g = acceleration due to gravity (9.80616 m/s^2)
v_s = stack gas exit velocity (m/s)
d_s = stack diameter (m)
T_s = stack gas temperature (K)
T_a = ambient air temperature (K).

If the stack gas temperature is less than or equal to the ambient air temperature, it is assumed that plume rise is dominated by momentum, in which case the effective stack height is given by this formula:

where, h_s = actual physical stack height (m)
s = stability parameter (Briggs, 1970; Hanna et al., 1982) and is only used in calculations for stable conditions (classes E and F):

where the constants 0.02 and 0.035 are default approximations of the derivative of the ambient potential temperature with respect to height.

$$h_e = h_s + \begin{cases} 3d_s \dfrac{v_s}{u_s} & \textit{Unstable or neutral} \\ \min\left\{ 1.5 \left(\dfrac{F_m}{u_s \sqrt{s}} \right)^{1/3}, 3d_s \dfrac{v_s}{u_s} \right\} & \textit{Stable} \end{cases}$$

$$s = \frac{g}{T_a} \begin{cases} 0.020 & \textit{StabilityClassE} \\ 0.035 & \textit{StabilityClassF} \end{cases}$$

If the stack gas temperature is greater than the ambient air temperature, then the determination of which force is dominating is made by calculating a critical crossover temperature difference DT_c above which it is assumed that buoyancy dominates. This critical value depends on the stack gas temperature, atmospheric stability, and the magnitude of the buoyant flux itself in a chain of empirical formulas (Briggs, 1969, 1972, 1975; Bowers et al., 1979) as follows:

$$\Delta T_c = \begin{cases} 0.0297 T_s \dfrac{v_s^{1/3}}{d_s^{2/3}} & \textit{Unstable or neutral,} F_b < 55 \\ 0.00575 T_s \dfrac{v_s^{2/3}}{d_s^{1/3}} & \textit{Unstable or neutral,} F_b \geq 55 \\ 0.019582 T_s v_s \sqrt{s} & \textit{Stable} \end{cases}$$

If the difference between the stack gas temperature and the ambient air is less than the critical temperature, then it is assumed that momentum dominates, and the equations above are used. Otherwise, the effective stack height is given by these equations:

$$h_e = h_s + \begin{cases} 21.425 \dfrac{F_b^{3/4}}{u_s} & \textit{Unstable or neutral,} F_b < 55 \\ 38.71 \dfrac{F_b^{3/5}}{u_s} & \textit{Unstable or neutral,} F_b \geq 55 \\ 2.6 \left(\dfrac{F_b}{u_s s} \right)^{1/3} & \textit{Stable} \end{cases}$$

Past a certain distance the plume is assumed to stop rising. This distance is called the distance to final rise (x_f) and is calculated in a similar method as for the plume rise. The calculation is dependent on which force dominates, the atmospheric stability class, and the magnitude of the buoyant flux. It is estimated by the following:

$$x_f = \begin{cases} 49\, F_b^{5/8} & \textit{Buoyancy rise, Unstable or neutral, } F_b<55 \\ 119\, F_b^{2/5} & \textit{Buoyancy rise, Unstable or neutral, } F_b\geq 55 \\ 2.0715\dfrac{u_s}{\sqrt{s}} & \textit{Buoyancy rise, Stable} \\ 0 & \textit{Momentum rise, all stability classes} \end{cases}$$

The estimated distance-dependent plume rise is the minimum of the effective stack height for final rise and the height based on that for buoyancy-dominated conditions (Briggs, 1972, p. 1030). The distance-dependent plume effective stack height $h_e(x)$ is this:

$$h_e(x) = \min\left\{ h_e\,,\ h_s + 1.60\left(F_b^{1/3}\,\frac{x^{2/3}}{u_s}\right)\right\}$$

This is sometimes referred to as the "2/3 law" of plume rise (Briggs, 1970) and follows from the assumptions that buoyancy is conserved and that the initial plume momentum is negligible for a very buoyant plume in unstratified surroundings. It is claimed (Hanna et al., 1982, p.14; Fay et al., 1969) that the constant 1.60, based on the best fit to data in Table II of Briggs (1970), can be expected to be accurate within 40% with variations due to downwash or local terrain effects.

D.2.1.3 Estimation of Air Concentration Accounting for Plume Depletion

The method used is that developed in Rao (1981). All estimations of concentration and deposition originate from the steady-state form of the atmospheric advection-diffusion equation:

$$U\,C_x = K_y\,C_{yy} + K_z\,C_{zz} + W\,C_z$$

where, $C(x,y,z)$ = pollutant concentration at (x,y,z)
- x = downwind distance
- y = horizontal crosswind distance
- z = vertical distance
- U = the constant average wind speed for the hour
- W = the gravitational settling velocity (cm/g)
- K_y = the eddy diffusivity in the crosswind direction
- K_z = the eddy diffusivity in the vertical directions

For a continuous point source of strength Q located at (0,0,H), the assumed boundary conditions are defined by this equation:

$$C(0,y,z) = \frac{Q}{U}\,\delta(y)\,\delta(z-H)$$

$$C(\infty,y,z) = C(x,\pm\infty,z) = C(x,y,\infty) = 0$$

$$\left[K_z C_z + WC\right]_{z=0} = \left[V_d C\right]_{z=0}$$

where, U = wind speed (m/s)
V_d = depositional velocity (cm/s)
H = height above the ground

The first condition is the limiting condition of the mass continuity equation at the source, with d being the Dirac delta "function". This condition is implicated by the assumption that the source is coming from an infinitely small point located at height H above the ground.

The second condition (actually three separate boundary conditions) is equivalent to the assumption that for all times the concentration of the pollutant is zero infinitely far away from the source in all directions.

The final condition is the one that accounts for possible depletion of the plume. It is the mathematical formulation of the assumption that at ground level (z=0) the sum of the turbulent transfer of pollutant down the concentration gradient ($K_z\ C_z$) and the downward settling flux due to the particles' weight ($W\ C$) is balanced by the net flux of material to the surface resulting from an exchange between the atmosphere and the surface (Rao and Satterfield, 1981). The deposition velocity V_d is the parameter that is assumed to characterize the interaction between the diffusing pollutant and the surface. If the deposition velocity is 0, then the lower boundary acts as a perfect reflector. If it is infinite, it acts as a perfect sink. If the deposition velocity is equal to the settling velocity, then the net deposition due to vertical diffusion is zero. For gases and small particles, the settling velocity is approximately 0, while for particles the settling and deposition velocities are estimated using the CARB algorithms (CARB, 1986) that represent empirical relationships for transfer resistances as a function of particle size, density, surface roughness, and friction velocity.

It is not difficult to derive an analytic expression for the solution of the advection-diffusion equation satisfying the boundary conditions above, with the solution involving nothing more complicated than exponential and error functions.

The eddy diffusion coefficients are expressed in terms of the standard deviations of the crosswind and vertical Gaussian concentration distributions (s_y and s_z, respectively), for which extensive empirical data exist. In particular, for Fickian diffusion (Rao, 1981) the relationships are these:

$$K_y = \sigma_y^2(x)\,\frac{U}{2x} \qquad , \qquad K_z = \sigma_z^2(x)\,\frac{U}{2x}$$

In practice the dependence of the standard deviations on the downwind distance is not usually explicitly noted. Also, as is standard in the atmospheric dispersion literature, the partial differential equation is solved as if the eddy diffusion coefficients do not depend on the downwind distance x. In fact, the solution to the advection-diffusion equation would be different were this dependence

considered. with the magnitude of difference between the two solutions depending on how "nonconstant" the standard deviations are with respect to x (i.e., on the magnitude of the derivative of the eddy diffusion coefficients with respect to x).

It is assumed that the plume is allowed to travel in a potentially vertically bounded layer called the mixing layer (sometimes called the Ekman layer; Pasquill, 1974). The height of this layer is called the mixing height, denoted here by L. If the effective stack height exceeds the mixing height, then the plume is assumed to fully penetrate the elevated inversion and the ground level concentration is set to zero. The mixing height is estimated based on twice-daily mixing heights using the RAMMET program, which uses the Holzworth (1972) procedures. These mixing heights are considered representative in rural areas only during periods of instability or neutral stability (stability classes A-D). The applicability of the Holzworth method to rural areas with stable atmospheric conditions is considered questionable, because the minimum mixing heights include the heat island effect for urban areas. In this case, unlimited vertical mixing is assumed.

Depending on the atmospheric stability class and mixing layer depth, the air concentration was estimated in three different ways, all of which are derived from the analytic solution of the original advection-diffusion equation above. The methods are summarized in Table D-9.

Table D-9
The Three Main Cases for Determining Air Concentration With Plume Depletion Effects

Condition	Criteria of Determination	Method of Solution
Stable or unlimited mixing	Pasquill classes E or F or unstable/neutral and L>5000 m	Analytic solution used.
Unstable/neutral, non-uniform mixing	Pasquill classes A-D, s_z <1.6 L and L<5000 m	Multiple eddy reflections from both the ground and stable layer aloft (plume is "trapped")
Unstable/neutral, limited and uniform mixing	Pasquill classes A-D, s_z >1.6 L and L<5000 m	Non-uniform vertical term integrated with limited mixing from height 0 to infinity

Only the vertical diffusion field was modified by deposition, and for deposition velocities on the order of a few centimeters per second, the shape of the vertical concentration profile was modified only slightly (Rao, 1980).

D.2.1.4 Estimation of the Atmospheric Dispersion Parameters

The dispersion parameters s_y and s_z were estimated using equations that approximately fit the Pasquill-Gifford curves (Turner, 1970) These equations approximately fit the Pasquill-Gifford curves (Turner, 1970) and were based on a rural setting:

$$\sigma_y = 465.11628\ x\ \tan(\theta(x))$$

where x is the downwind distance (in km), and

$$\theta(x) = 0.017453293(\ c\ -\ d\ \ln x).$$

The parameters c and d depend on the stability class and are given in Table D-10

Table D-10
Parameters Used to Calculate Horizontal Dispersion Parameter s_y in COMPDEP (Turner, 1970)

Pasquill Stability Class	c	d
A	24.1670	2.5334
B	18.3330	1.8096
C	12.5000	1.0857
D	8.3330	0.72382
E	6.2500	0.54287
F	4.1667	0.36191

The vertical dispersion parameter is estimated by

$$\sigma_z = a\ x^b$$

where the parameters a and b depend on the stability class, and are given in Table D-11.

Table D-11
Parameters Used to Calculate Vertical Dispersion
Parameter s_z in COMPDEP (Turner, 1970)

Pasquill Stability Class	x (km)	a	b
A[a]	<0.10	122.800	0.94470
	0.10 - 0.15	158.080	1.05420
	0.16 - 0.20	170.220	1.09320
	0.21 - 0.25	179.520	1.1262
	0.26 - 0.30	217.410	1.26440
	0.31 - 0.40	258.890	1.40940
	0.41 - 0.50	346.750	1.72830
	0.51 - 3.11	453.850	2.1160
	>3.11	[b]	[b]
B[a]	<0.20	90.673	0.93198
	0.21 - 0.40	98.483	0.98332
	>0.40	109.300	1.09710
C[a]	All	61.141	0.91465
D[a]	<0.30	34.459	0.86974
	0.31 - 1.00	32.093	0.81066
	1.01 - 3.00	32.093	0.64403
	3.01 - 10.00	33.504	0.60486
	10.01 - 30.00	36.650	0.56589
	>30.00	44.053	0.51179
E[a]	<0.10	24.260	0.83660
	0.11 - 0.30	23.331	0.81956
	0.31 - 1.00	21.628	0.75660
	1.01 - 2.00	21.628	0.63077
	2.01 - 4.00	22.534	0.57154
	4.01 - 10.00	24.703	0.50527

Table D-11 (continued)
Parameters Used to Calculate Vertical Dispersion
Parameter s_z in COMPDEP (Turner, 1970)

Pasquill Stability Class	x (km)	a	b
A[a]	<0.10	122.800	0.94470
	10.01 - 20.00	26.970	0.46713
	20.01 - 40.00	35.420	0.37615
	>40.00	47.618	0.29592
F[a]	<0.20	15.209	0.81558
	0.21 - 0.70	14.457	0.78407
	0.71 - 1.00	13.953	0.68465
	1.01 - 2.00	13.953	0.63227
	2.01 - 3.00	14.823	0.54503
	3.01 - 7.00	16.187	0.46490
	7.01 - 15.00	17.386	0.41507
	15.01 - 30.00	22.651	0.32681
	30.01 - 60.00	27.074	0.27436
	> 60.00	34.219	0.21716

[a] If the calculated value of s_z exceeds 5000 m, then it is set to 5000 m.
[b] s_z is set to 5000 m.

D.2.1.5 Deposition Processes

COMPDEP addresses both wet and dry deposition, taking into account the fraction of an hour during which precipitation occurs.

The air concentrations are calculated accounting for plume depletion from dry deposition. The dry deposition rate, in g/m^2/time, is given by the product of the deposition velocity, the air concentration and the fraction of the hour during which precipitation does not occur. For particles, the settling and deposition velocities were estimated using the CARB algorithms (CARB, 1986) that represent empirical relationships for transfer resistances as a function of particle size, density, surface roughness and friction velocity. In general, the deposition velocity has values that can range from zero up to 180 cm/s (Sehmel, 1984).

COMPDEP calculates the annual wet deposition flux according to the method developed by Slinn (1984) and later modified by PEI and Cramer (1986). The scavenging process consists of

repeated exposures of particles and gases to cloud or precipitation elements with some chance of collection on the element for each exposure (Engelmann, 1968). This has been addressed historically as a first order decay process with decay constant L, called the scavenging coefficient (units of inverse time). The concentration at any distance x downwind is then given by $C(x,y,z)\ exp(-L\ t)$, where $C(x,y,z)$ is the (steady-state) concentration without scavenging and t is the time since precipitation began (Hanna et al., 1982). For the purpose of modelling, t was replaced with x/u_s (the travel time to the receptor); thus, the decay is essentially accounting for previous scavenging upwind of the receptor.

The wet deposition flux *Dyw* at a given location is

$$Dyw = \Lambda \int_0^z C(x,y,z)\ e^{-\Lambda t} dz$$

where z is the height from which the precipitation falls. Because it was assumed that the effects of dry deposition and gravitational settling are negligible compared with precipitation scavenging, the concentration used was that without deposition effects (deposition and settling velocities set to zero). To facilitate evaluation of the integral, it was extended to infinity as an approximation. The deposition flux for a given hour then reduces to this equation:

$$Dyw = f\,\Lambda\,Q\,R_{DW}\ e^{-\Lambda \frac{x}{u_s}}$$

where, f = the fraction of the hour that precipitation occurs
R_{DW} = the integrated vertical relative concentration (unit source strength) without depletion effects

R_{DW} can be calculated by these equations:

$$R_{DW} = \begin{cases} \dfrac{\exp\left(-0.5\ y^2/\sigma_y^2\right)}{u_s \sigma_y \sqrt{2\pi}} & \textit{Simple or intermediate terrain} \\ \dfrac{1}{2\pi u_s x/16} & \textit{Complex terrain (sector-averaged)} \end{cases}$$

As noted in PEI and Cramer (1986), there are several assumptions in deriving the equation for wet deposition.

1) The intensity of precipitation is constant over the entire path between the source and receptor.

2) The precipitation originates at a level above the top of the plume so that hydrometeors (i.e., products formed by condensation of water vapor) pass vertically through the entire plume.

3) The time duration of the precipitation over the entire path between the source and the receptor point is such that exactly f (f is defined as the fraction of the hour in which precipitation occurs) of the hourly emission is subject to a constant intensity for the

entire travel time required to traverse the distance between the source and the receptor. The remaining fraction is subject only to dry deposition processes.

In COMPDEP, the scavenging coefficient may be intensity- and particle-size dependent, in which case the total wet deposition is the sum of the contributions of each category particle size category. For particles, example scavenging coefficients are from PEI and Cramer (1986) and are shown in Table D-12. Only a small fraction of the pollutants of concern for this exposure assessment are particulate or particulate-bound.

Table D-12
Example of Precipitation Scavenging Coefficients (per second) in COMPDEP

Precipitation Intensity	Particle Size Category (mM)		
	Less than 2	2 to 10	Greater than 10
Heavy	1.46E-03	4.64E-03	9.69E-03
Moderate	5.60E-04	8.93E-04	9.69E-03
Light	2.20E-04	1.80E-04	9.69E-03

Estimation of the scavenging coefficients for vapor phase pollutants are discussed in Section D.2.2.1.

D.2.1.6 Treatment of Terrain

The "COMP" in the name COMPDEP refers to the capability of the model to estimate concentrations and deposition at receptor locations at or above stack top. This can be done in three ways: (1) the effective stack height may be modified based on the receptor height; (2) the concentrations may be reduced by a height-dependent correction factor for receptors above the stack top; and (3) sector-averaging is used for receptors above stack top.

The method of adjusting the effective stack height H_t is based on models developed by Briggs (1973) and Egan (1975). With this method the amount of reduction depends on the receptor height and empirical terrain adjustment factors.

$$H_t = \max\{H - r_h \cdot (1 - ter),\ H \cdot ter,\ H_{min}\}$$

where, H = the effective stack height calculated without considering terrain
r_h = the height of the receptor above the stack base
ter = stability-class dependent terrain adjustment factor
H_{min} = the minimum distance between the plume centerline and ground.

Following standard practice, $H_{min} = 10\ m$.

The terrain adjustment factors used in the exposure assessment are consistent with the method of Egan (1975) and are shown in Table D-13.

Table D-13
Terrain Adjustment Factors Used in
Calculating Terrain-Dependent Effective Stack Height

Pasquill Stability Class	Terrain Adjustment Factor (unitless)
A	0.5
B	0.5
C	0.5
D	0.5
E	0
F	0

By choosing these terrain factors in neutral and unstable conditions the effective stack height is reduced by $r_h/2$ or $H/2$, whichever is smaller. It should be noted that Briggs (1973) suggests that the stack height be reduced by r_h or $H/2$, whichever is smaller. Briggs' method will result in slightly higher ground-level concentrations for the surface of small hills (Hanna et al., 1982). It should also be noted that the reduction by $H/2$ is based on potential flow theory and wind-tunnel experiments (Hanna et al., 1982). Both Egan's and Briggs' methods assume terrain factors of zero for stable conditions, in which case it is assumed that the plume maintains a constant elevation (and so the effective height is reduced by the receptor height).

For receptors whose ground level elevation exceeds the effective stack height, the concentrations and deposition rates are multiplied by a "correction" factor *corr*, given by this:

$$corr = \begin{cases} (400 - Diff)/400 & 0 < Diff < 400m \text{ and Stable conditions} \\ 0 & Diff \geq 400m \text{ and Stable conditions} \\ 1 & \text{Unstable or neutral conditions} \end{cases}$$

where *Diff* is the difference between receptor ground level elevation and the effective stack height. Thus, in stable conditions (Pasquill stability classes E or F) the concentrations and deposition rates were assumed to be zero if the receptor is 400 m above the effective stack height. Although the exact origin of this method is not clear, it is used in the Valley dispersion model (one of the precursors of COMPDEP), where it is observed that the application of the correction factor in stable conditions should not be inferred to represent pollutant decay or penetration into the terrain. This is an empirical scheme intended as a general representation of the blocking of air flow by significant terrain features. Furthermore, the concentrations calculated for the leeward side of a substantial hill will not reflect this attenuation upwind, and so such concentrations should be considered suspect.

By setting the correction factor to one in unstable or neutral atmospheric stability conditions, the plume was assumed to parallel the terrain feature at the terrain-dependent effect stack height as calculated above.

For terrain above the effective stack height, sector averaging was used to calculate the air concentration and was subsequently used for deposition. This assumes that there is no crosswind

variation in concentration within an angular sector equal to the resolution of the wind direction data (22.5 degrees for this assessment). It should be noted that there was no technical basis for using sector averaging for terrain above stack height rather than point estimates for the horizontal dispersion parameters. This decision was made consequent to personal communication with Donna Schwede (7-20-94).

D.2.1.7 Downwash

Usually, emissions from an industrial source will rise due to a combination of their initial vertical momentum and buoyancy. Under certain peculiar conditions, however, they may be trapped in either the wake of the stack or nearby building, resulting in increased concentrations. Two types of these phenomena are addressed in COMPDEP: stack-tip downwash and building downwash (building wake effects).

Stack-tip Downwash. In practice, it has been observed that if the stack exit velocity is low relative to the wind speed, then the stack emissions may be pulled into the low pressure cavity in the wake of the stack. The emissions are pulled down and may not rise further, resulting in higher ground level concentrations than if plume rise had occurred. In COMPDEP, this phenomenon was assumed to occur when the ratio of the stack exit velocity to the wind speed at stack top was below 1.5. This value, which has survived without modification for 25 years, is that recommended by Briggs (1969) and is based on wind tunnel experiments by Sherlock and Stalker (1941). It should be noted that very buoyant sources may accelerate fast enough to avoid any significant downwash (Overcamp, 1977; Briggs, 1969).

If stack-tip downwash occurs, then the physical stack height used is that of Briggs (1974):

$$h_s' = h_s + 2d_s\left(\frac{v_s}{u_s} - 1.5\right)$$

From the above equation it can be seen that the maximum amount the stack height will be reduced by this method is three times the diameter of the stack.

Building Downwash. Building downwash can occur when the stack emittants are captured in the wake of a nearby building. A long-standing rule-of-thumb is that building effects should not occur if the stack height is at least 2.5 times the height of any adjacent building. Because this was considered overly restrictive (from a design perspective) for tall, thin buildings, Briggs (1973) proposed a modification of this rule in which building downwash was assumed not to occur if the stack height was greater than the sum of the building height and 1.5 times the minimum of the building height and width.

In the use of COMPDEP for the mercury assessment, building downwash was considered if the plume height, calculated from the sum of the stack height and the distance-dependent plume rise at a distance of two building heights, was greater than either (a) 2.5 times the building height, or (b) the sum of the building height and 1.5 times the building width. If wake effects were predicted, then the effective stack height was reduced by reducing the estimated plume rise. First a distance-dependent plume rise is estimated based on momentum-dominated conditions (Bowers et al., 1979).

The effective stack height was then set to the maximum of this value and the distance-dependent buoyancy-dominated effective stack height:

$$\Delta h_p = \begin{cases} \max\left[\dfrac{3v_s d_s}{u_s}, \left(\dfrac{3F_m x}{\beta_j^2 u_s^2}\right)^{1/3}\right] & \textit{Unstable or neutral} \\ \max\left[\dfrac{3v_s d_s}{u_s}, \left(3F_m \dfrac{\sin(x\sqrt{s}/u_s)}{\beta_j^2 u_s \sqrt{s}}\right)^{1/3}\right] & \textit{Stable} \end{cases}$$

$$h_e = h_s + \max\left\{\Delta h_p\ ,\ 1.60\left(F_b^{1/3}\frac{x^{2/3}}{u_s}\right)\right\}$$

The dispersion parameters were also modified based on the dimensions of the building. These modifications are based on Huber and Snyder building downwash procedures (Huber, 1977; Huber, and Snyder, 1976) and are principally based on the results of wind-tunnel experiments using a model building with a crosswind double that of the building height. Because the atmospheric turbulence simulated in the wind-tunnel experiments was intermediate (between a slightly unstable Pasquill C category and neutral D), the data upon which the formulas were based reflect a specific stability, building shape, and orientation with respect to the mean wind direction (U.S. EPA 1992, p.1-20).

The basic idea was to estimate modified lateral and vertical dispersion parameters, and then use the minimum of these and the dispersion parameters estimated without wake effects. In general, the ratio of the building width to building height plays a key role.

Setting

$$\sigma_y' = \begin{cases} 0.35h_1 + 0.067(x-3h_2) & \textit{if } 3h_m \le x \le 10h_m \\ \sigma_y(x+x_y) & \textit{if } x \ge 10h_m \end{cases}$$

where h_m is the minimum of the building width and height, and h_1 and h_2 are given by

$$h_1 = \begin{cases} h_b & \textit{if } w_b/h_b > 5 \\ w_b & \textit{else} \end{cases}$$

and

$$h_2 = \begin{cases} w_b & \textit{if } w_b/h_b < 1 \\ h_b & \textit{else} \end{cases}$$

and x_y is the lateral virtual distance, given by

where the coefficients p and q depend on the stability class, and the coefficients c_w and c_h are given by

$$x_y = \left(\frac{c_w w_b + c_h h_b}{p} \right)^{1/q} - h_m$$

$$c_w = \begin{cases} 0.85 & \text{if } w_b/h_b < 1 \\ 0.35 & \text{if } 1 \le w_b/h_b \le 5 \\ 0 & \text{if } w_b/h_b > 5 \end{cases}$$

and

$$c_h = \begin{cases} 0 & \text{if } w_b/h_b < 1 \\ 0.5 & \text{if } 1 \le w_b/h_b \le 5 \\ 0.85 & \text{if } w_b/h_b > 5 \end{cases}$$

The virtual source location was calculated by requiring that $s_y'(10\ h_1) = .35\ h_1 + 0.5\ h_2$ (Randerson, 1984; p.303).

Table D-14 presents the coefficients used to calculate lateral virtual distances.

Table D-14
Coefficients Used to Calculate Lateral Virtual Distances for Pasquill Dispersion Rates

Pasquill Stability Class	p	q
A	209.14	0.890
B	154.46	0.902
C	103.26	0.917
D	68.26	0.919
E	51.06	0.921
F	33.92	0.919

The vertical dispersion term is modified similarly.

$$\sigma_z' = \begin{cases} 0.7h_m + 0.067(x-3h_m) & if\ 3h_m \le x \le 10h_m \\ \sigma_z(x+x_z) & if\ x \ge 10h_m \end{cases}$$

where h_m is the minimum of the building width and height, and x_z is the vertical virtual distance, given by

$$x_z = \left(\frac{1.2h_m}{a}\right)^{1/b} - h_m$$

where the coefficients *a* and *b* are given in Table D-11 above. The virtual source location x_z is calculated by requiring that $s_z'(10\ h_m) = 1.2\ h_m$ (Randerson, 1984; p.303)., and is added in order to account for the enhanced initial plume growth caused by the building wake (U.S. EPA, 1992, p.1-24)

D.2.1.8 Buoyancy-Induced Dispersion

It has been observed that the initial dispersion of plumes may be augmented by turbulent motion of the plume and turbulent entrainment of ambient air. This is addressed by increasing the calculated standard deviations in the crosswind and vertical directions using the method of Pasquill (1976)

$$\sigma_{ze} = \left(\sigma_z^2 + \left(\frac{\Delta h}{3.5}\right)^2\right)^{1/2}$$

$$\sigma_{ye} = \left(\sigma_y^2 + \left(\frac{\Delta h}{3.5}\right)^2\right)^{1/2}$$

where,

s_{ye} = the effective standard deviation of lateral concentration distributions (m) for buoyancy-induced dispersion

σ_{ze} = the effective standard deviation of vertical concentration distributions (m) for buoyancy-induced dispersion

s_y = the standard deviation of lateral concentration distributions (m) without buoyancy effects

s_z = the standard deviation of vertical concentration distributions (m) without buoyancy effects,

D*h* = the estimated plume rise (m).

D.2.1.9 Meteorological Data

COMPDEP uses hourly meteorological data to estimate hourly concentrations and deposition rates. If wet deposition is not to be modeled, then the only data file required is one containing hourly values for average wind speed, wind direction, Pasquill stability class, mixing height, and ambient air temperature.

If wet deposition is to be modeled, then a data file containing a summary of the hourly precipitation intensities and fraction of hour for which precipitation occurred is also required. COMPDEP only considers four precipitation intensity classes. These are summarized in Table D-15.

Table D-15
Precipitation Intensities Considered by COMPDEP

Intensity Class	Precipitation Rate (in/hr)
0	0
1	trace to 0.10
2	0.11 to 0.30
3	greater than 0.30

D.2.2 Application of the COMPDEP Model for the Exposure Assessment

To estimate local mercury concentrations in environmental media, hypothetical sources (model plants) were designed using available information. Each model plant/control scenario/ emission speciation estimate was placed in the hypothetical locations. In this section modifications made to the COMPDEP model for this exposure assessment, as well as parameter values used are discussed.

D.2.2.1 Modifications of COMPDEP for the Exposure Assessment

Several modifications were made to COMPDEP in order to address more effectively the atmospheric deposition of mercury species. These modifications were necessary, because mercury exists primarily in the vapor phase, and the previous version of COMPDEP (version 93340) could not estimate deposition for vapor.

Specification of Vapor phase/Particle-Bound Phase Ratio. This modification consisted of adjusting COMPDEP to allow the user to specify the fractions of the emissions of a particular pollutant that are in vapor phase and particle-bound phase. The modification was necessary because the transport properties of the two phases can be quite different and the forms of mercury considered in this report are primarily in the vapor phase.

Dry Deposition of Vapor Phase Contaminants. The mercury species assumed to be emitted (elemental and divalent mercury) are predominantly in the vapor phase; however, the algorithms in COMPDEP for calculating deposition velocities can only be used for particles. For this reason, COMPDEP was modified so that, for the vapor phase fraction of a pollutant, the user can specify atmospheric stability class-dependent deposition velocities. These were used in the concentration algorithms with plume depletion, with the gravitational settling velocity set to zero as is recommended for gases (Rao, 1981). Section D.2.2.3 gives a description of the deposition velocities used in the exposure assessment.

Wet Deposition of Vapor Phase Contaminants. To estimate wet deposition COMPDEP requires a scavenging coefficient for each pollutant. This scavenging coefficient can depend on the

precipitation intensity and particle size. No values are given for gases. for which the scavenging coefficient will depend strongly on the chemical properties of the pollutant in the vapor phase (e.g., solubility).

Modifications were made to COMPDEP to enable the user to specify a unitless washout ratio W for the vapor phase portion of the pollutant. The washout ratio is the ratio of the concentration in air to the concentration in precipitation.

Connection Between the Washout Ratio and Scavenging Coefficient. By definition, the washout ratio W is the ratio of the concentration in surface-level precipitation to the concentration in surface level air (Slinn, 1984). Let C_w and C_a denote the concentrations in surface-level precipitation and surface-level air, respectively (units of g/m^3). Then

$$W = \frac{C_w}{C_a}$$

Using the scavenging coefficient to estimate air concentration during periods of precipitation, the concentration in air is given by this equation:

$$C_a = C\,(x,y,z_s)\; e^{-\Lambda t}$$

where, z_s = height of the receptor (m)
x = downwind distance to the receptor
y = crosswind distances to the receptor (m),
t = travel time (seconds) to the receptor
L = scavenging coefficient (units of inverse seconds).

The concentration in precipitation can be approximated by the wet deposition flux divided by the precipitation rate for the time period:

$$C_w = \frac{\Lambda \int_{z_s}^{H} C\,(x,y,z)\; e^{-\Lambda t}\, dz}{P}$$

where, H = height from which the precipitation falls (m) and
P = precipitation rate (m/s).

The washout ratio is then given by these formulae:

$$W = \frac{\Lambda \int_{z_s}^{H} C\,(x,y,z)\, e^{-\Lambda t}\, dz}{P\, C\,(x,y,z_s)\, e^{-\Lambda t}} = \frac{\Lambda \int_{z_s}^{H} C\,(x,y,z)\, dz}{P\, C\,(x,y,z_s)} = \frac{(\Lambda\, H)(1/H) \int_{z_s}^{H} C\,(x,y,z)\, dz}{P\, C\,(x,y,z_s)} = \frac{(\Lambda\, H)\, \overline{C(x,y)}}{P\, C\,(x,y,z_s)}$$

where Overline{$C(x,y)$} is the vertically averaged air concentration at (x,y). Assuming that Overline{$C(x,y)$} is approximately equal to $C(x,y,z_s)$, and that the height H is the mixing height H_L, the equation reduces to

$$W = \frac{\Lambda\, H_L}{P}$$

and conversely

$$\Lambda = \frac{W\, P}{H_L}$$

Because only the intensity classes (0-3) of the precipitation were assumed to be in the precipitation data file (see Section 7.2.8), the user must specify the representative precipitation rate for each intensity class.

For a discussion of the washout ratios used in this study see D.2.2.3.

D.1.2.2 Meteorological Data and Receptor Locations Relative to Local Source

For both of the locations, meteorological data were obtained for one year (1989). The types of data files and their use are described in Table D-16.

Table D-16
Description of Meteorological Files Used to Make Input Files for COMPDEP

Data File	Use
Hourly surface observations (CD144)	Used by RAMMET program to create meteorological data file. Used by ORNL precipitation preprocessor to create precipitation data file for wet deposition calculations.
Mixing Height Data file	Used by RAMMET to create meteorological data file.

D.2.2.3 Vapor Deposition Parameters

Dry Deposition Velocities. The dry deposition velocities for divalent vapor were based on those used by the RELMAP model and were estimated based on assumed similar deposition properties of nitric acid. Deposition velocities depend on the season, land use, time of day, and stability class. Table D-17 shows the seasonally-averaged deposition velocities as a function of land-use. Dry deposition rate is proportional to the dry deposition velocities.

Table D-17
Divalent Mercury Vapor Seasonally-Averaged Deposition Velocities (cm/s)

Land-use	Pasquill Stability Class					
	A	B	C	D	E	F
Urban	4.63	4.59	4.36	4.03	2.46	0.36
Agricultural	1.81	1.77	1.63	1.42	0.61	0.20
Range	1.71	1.68	1.54	1.34	0.56	0.19
Deciduous Forest	3.45	3.40	3.14	2.80	1.39	0.29
Coniferous Forest	3.45	3.40	3.14	2.80	1.39	0.29
Mixed forest/wetland	3.32	3.27	3.06	2.77	1.54	0.29
Water	1.00	0.98	0.89	0.77	0.31	0.13
Barren Land	1.07	1.06	0.97	0.85	0.31	0.18
Non-forested wetland	1.92	1.89	1.77	1.59	0.85	0.21
Mixed agric./rangeland	1.76	1.73	1.59	1.39	0.58	0.19
Rocky open areas	1.94	1.90	1.74	1.51	0.59	0.20

In order to address the fact that deposition is lower during nighttime conditions, it was assumed that the deposition velocity for divalent vapor was 0.3 cm/s for stability classes D-F, which occur primarily at night, and 1 cm/s for stability classes A-C.

Washout Ratios. For this assessment, it was assumed that both elemental and divalent mercury species would be deposited via wet deposition. Because of its higher solubility, the divalent form would be washed out at significantly higher rates. The washout ratio is a function of the concentration, total carbon, and ozone air distribution. For divalent vapor, a washout ratio of 1.6×10^6 was used, while for the elemental phase a value of 1.6×10^4 was used; this was roughly the average of the washout ratio for elemental vapor for both locations (see Section D.1).

In order to calculate the scavenging coefficient, the precipitation rate for the hour is required. In the precipitation data file, the precipitation rate is classified as either none, light (trace to 0.1 in/hr), moderate (0.11 to 0.3 in/hr), or heavy (greater than 0.3 in/hr). Thus, a representative precipitation rate is required and is specified by the version of COMPDEP modified for this assessment. For the light and moderate categories, the midpoint of the range was used, while for the heavy category the representative rate was assumed equal to 0.3 in/hr.

An informal examination of the scavenging coefficients computed for divalent vapor (using the representative rates discussed above) showed that they were in the same range as the upper end

scavenging coefficients for particles as estimated by PEI and Cramer (1986). Table D-16 presents divalent mercury vapor deposition velocities. It was assumed that the elemental mercury vapor has a dry deposition velocity of zero.

D.2.2.4 Other Parameters Used in Air Modeling

This section presents the values for all parameters not already discussed that were used in the air modeling for all of the model plants. These are given in Tables D-18 to D-20.

Table D-18
Air Modeling Parameter Values Used In the Exposure Assessment: Generic Parameters

Parameter	Value Used in Exposure Assessment
Particle Density (g/cm^3)	1.8
Surface Roughness Length (m)[a]	0.30
Anemometer Height (m)	10
Wind Speed Profile Exponents	
Stability Class A	0.07
Stability Class B	0.07
Stability Class C	0.10
Stability Class D	0.15
Stability Class E	0.35
Stability Class F	0.55
Terrain Adjustment Factors	
Stability Class A	0.5
Stability Class B	0.5
Stability Class C	0.5
Stability Class D	0.5
Stability Class E	0
Stability Class F	0
Distance Limit for Plume Centerline (m)	10
Model Run Options	
Terrain Adjustment	Yes
Stack-tip Downwash	No
Building Wake Effects	No
Transitional Plume Rise	Yes
Buoyancy-induced dispersion	Yes
Calms Processing Option	No

[a] This is used to estimate deposition velocities for particles.

Table D-19
Model Plant Parameter Values Used in COMPDEP

Plant Type	Stack Height (ft)	Stack Diameter (ft)	Stack Exit Temperature (F)	Stack Exit Velocity (m/s)	Total Mercury Emission Rate (kg/yr)
Large Municipal Waste Combustor	230	10	285	21.9	1330
Small Municipal Waste Combustor	140	5	375	21.9	170
Continuous Medical Waste Incinerator	40	3	1500	7.3	80
Intermittent Medical Waste Incinerator	40	1	1500	7.3	2.4
Large Coal-fired Utility Boiler	732	27	273	31.1	230
Medium Coal-fired Utility Boiler	465	18	275	26.7	90
Small Coal-fired Utility Boiler	266	12	295	6.6	10
Medium Oil-fired Utility Boiler	290	14	322	20.7	2
Chlor-alkali Plant	10	1	Ambient[a]	0.1	380
Primary Copper Smelter	505	15	430	6	536
Primary Lead Smelter	350	20	347	2.80	2680

[a] Average annual temperature was used (see Appendix B).

Table D-20
Model Plant Mercury Speciation of Emissions

Plant Type	Speciation of Emissions			Speciation of Emissions: "Error-Bounding Estimate"		
	% Elemental Mercury Vapor	% Divalent Mercury Vapor	% Divalent Particulate	% Elemental Mercury Vapor	% Divalent Mercury Vapor	% Divalent Particulate
Large Municipal Waste Combustor[a]	20	60	20	57.50	42.50	0.0
Small Municipal Waste Combustor	20	60	20	57.50	42.50	0.0
Continuous Medical Waste Incinerator	20	60	20	70.00	30.00	0.0
Intermittent Medical Waste Incinerator	20	60	20	70.00	30.00	0.0
Large Coal-fired Utility Boiler	50	30	20	85.00	15.00	0.0
Medium Coal-fired Utility Boiler	50	30	20	85.00	15.00	0.0
Small Coal-fired Utility Boiler	50	30	20	85.00	15.00	0.0
Medium Oil-fired Utility Boiler	*50*	*30*	*20*	*85.00*	*15.00*	*0.0*
Chlor-alkali Plant	70	30	0	85.00	15.00	0.0
Primary Copper Smelter	85	10	5	95.00	5.00	0.0
Primary Lead Smelter	85	10	5	95.00	5.00	0.0

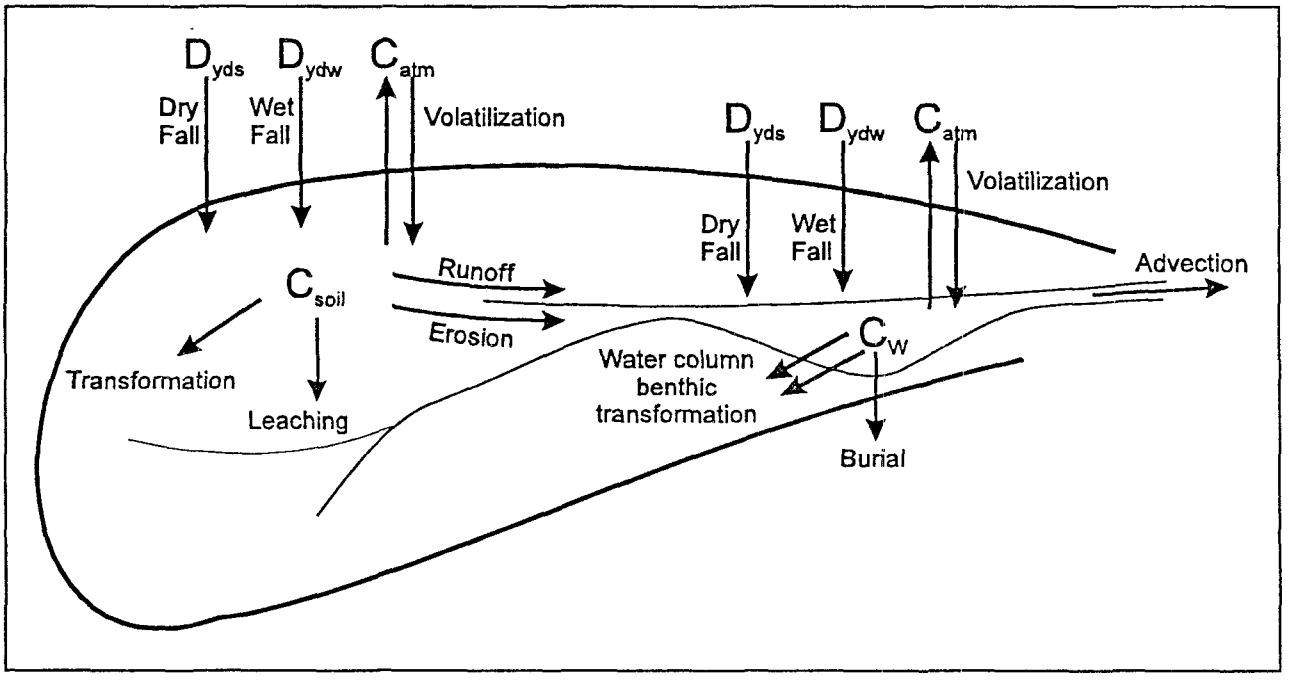
[a] For the control scenario in which emissions are 90% of base, the assumed speciation is the same as for the utility boilers.

D.3 Description of the IEM2 Indirect Exposure Methodology

Atmospheric mercury concentrations and deposition rates estimated from RELMAP and COMPDEP drive the calculations of mercury in watershed soils and surface waters. The soil and water concentrations, in turn, drive calculations of concentrations in the associated biota and fish, which humans and other animals are assumed to consume. These "indirect" exposure calculations were modified from the Indirect Exposure Document (IED; U.S. EPA, 1990) as updated in an Addendum (in preparation). Relevant sections of the updated methodology, IEM2, are described below. The equations were implemented in a spreadsheet and parameterized for several hypothetical scenarios as described in Volume III of this report.

IEM2 uses atmospheric chemical loadings to perform mass balances on a watershed soil element and a surface water element, as illustrated in Figure D-1. The mass balances were performed for total mercury, which was assumed to speciate into three components: Hg^0, Hg(II), and methylmercury. The fraction of mercury in each of these components was specified for the soil and the surface water elements. Loadings and chemical properties were given for the individual mercury components, and the overall mercury transport and loss rates are calculated by the methodology.

Figure D-1
Overview of the IEM2 Watershed Modules



Definitions for Figure D-1

C_{soil}	chemical concentration in upper soil	mg/L
C_w	chemical concentration in water body	mg/L
C_{atm}	vapor phase chemical concentration in air	$\mu g/m^3$
D_{yds}	average dry deposition to watershed	mg/yr
D_{yws}	average wet deposition to watershed	mg/yr

IEM2 first performs a terrestrial mass balance to obtain mercury concentrations in watershed soils. Soil concentrations were used along with vapor concentrations and deposition rates to calculate concentrations in various food plants. These were used, in turn, to calculate concentrations in animals. IEM2 next performs an aquatic mass balance driven by direct atmospheric deposition along with runoff and erosion loads from wateshed soils. Methylmercury concentrations in fish were derived from total dissolved water concentrations using bioaccumulation factors (BAF).

IEM2 was developed to handle individual chemicals, or chemicals linked by kinetic transformation reactions. The kinetic transformation rates affecting mercury components in soil, water, and sediments -- oxidation, reduction, methylation, and demethylation -- were considered too uncertain to implement in this study. For this study, the methodology was expanded to handle multiple chemical components in a steady-state relationship. The fraction of each chemical component in the soil and water column was specified by the user. The methodology predicts the total chemical concentration in watershed soils and the water body based on loading and dissipation rates specified for each of the components.

The nature of this methodology is basically steady with respect to time and homogeneous with respect to space. While it tracks the buildup of watershed soil concentrations over the years given a steady depositional load and long-term average hydrological behavior, it does not respond to unsteady loading or meteorological events. There are, thus, limitations on the analysis and interpretations imposed by these simplifications. The methodology cannot be used to predict the response to reduction or elimination of loadings. The model's calculations of average water body concentrations are less reliable for unsteady environments, such as streams, than for more steady environments, such as lakes.

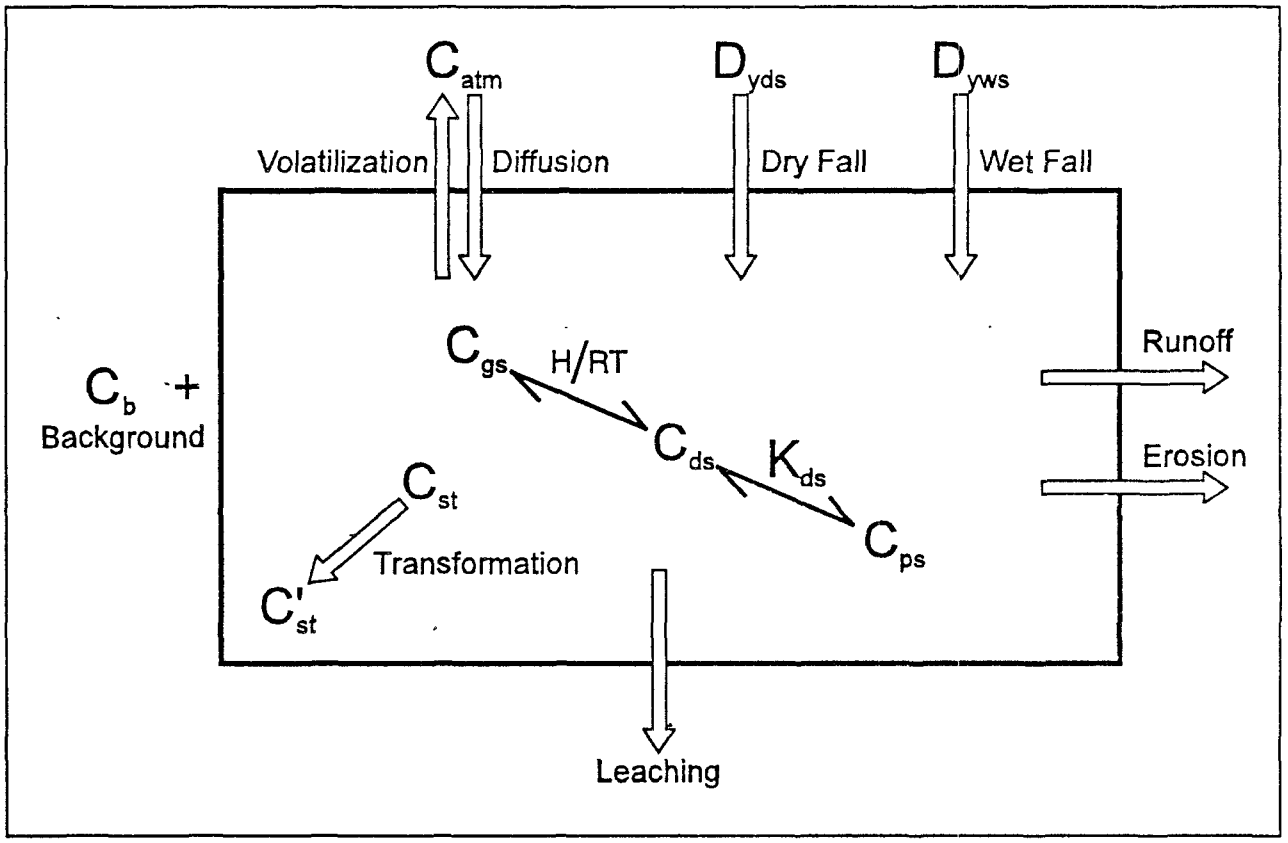
D.3.1 The Terrestrial Equations

The IEM2 framework for estimating watershed soil impacts from stack emissions calculates surface soil concentrations, including dissolved and sorbed phases, as illustrated in Figure D-2. The model accounts for three routes of contaminant entry into the soil: deposition of particle-bound contaminant through dryfall; deposition through wetfall; and diffusion of vapor phase contaminant into the soil surface. The model also accounts for five dissipation processes that remove contaminants from the surface soils: decay of total contaminants (sorbed + dissolved) within the soil horizon; volatilization (diffusion of gas phase out of the soil surface); runoff of dissolved phase from the soil surface; leaching of the dissolved phase through the soil horizon; and erosion of particulate phase from the soil surface. Key assumptions in the watershed soil impact algorithm were these:

- Soil concentrations within a depositional area are assumed to be uniform within the area, and can be estimated by the following key parameters: dry and wet contaminant deposition rates, a wind-driven gaseous exchange rate with the atmosphere, a soil dissipation rate, a soil bulk density, and a soil mixing depth.

- The partitioning of the contaminant within the soil/water matrices can be described by partition coefficients.

Figure D-2
Overview of the IEM2 Soils Processes



Definitions for Figure D-2

C_{atm}	vapor phase chemical concentration in air	µg/m^3
D_{yds}	average dry deposition to watershed	mg/yr
D_{yws}	average wet deposition to watershed	mg/yr
$C_{st,}$	total chemical concentration in soil	mg/L
C_{st}	reaction product concentration in soil	mg/L
C_b	background chemical concentration in soil	mg/L
C_{gs}	chemical concentration in soil gas	µg/m^3
C_{ds}	chemical concentration in soil water	mg/L
C_{ps}	chemical concentration on soil particles	µg/g
H	Henry's Law constant	atm-m^3/mole
R	universal gas constant	atm-m^3/mole-°K
T	temperature	°K
K_{ds}	soil/water partition coefficient	L/kg

D.3.1.1 Chemical Mass Balance in Watershed Soils

A mass balance equation can be written for total mercury in watershed soils, balancing areal deposition fluxes with chemical loss processes:

$$Sc = \frac{L_W}{ks\ Z\ BD}\ (1 - e^{-ks\ Tc})\ 100 + C_{sb}$$

where:

Sc	=	average watershed soil concentration after time period of deposition (μg pollutant/g soil)
L_W	=	yearly average load of pollutant to watershed on an areal basic (g pollutant/m^2-yr)
ks	=	total chemical loss rate constant from soil (yr^{-1})
Tc	=	total time period over which deposition has occurred (yr)
Z	=	representative watershed mixing depth to which deposited pollutant is incorporated (cm)
BD	=	representative watershed soil bulk density (g/cm^3)
100	=	units conversion factor (μg-m^2/g-cm^2)
C_{sb}	=	background "natural" soil concentration (μg pollutant/g soil)

The first term in the equation represents the steady-state concentration achieved after a sufficient period of constant loading. The exponential term gives the fraction of the steady-state response achieved after Tc years of loading. The final term gives the natural background concentration found in soils. The background soil concentration of mercury was assumed to be negligible in this study. The major terms in this equation are discussed in sections below.

D.3.1.2 Equilibrium Speciation in Watershed Soils

Total mercury in the soil was assumed to be distributed among three components -- Hg^0, Hg(II), and methylmercury. The steady-state fraction of the total in each component is specified by the user, so that:

$$Sc_{Hg^0} = Sc \cdot f_{s1}$$

$$Sc_{Hg(II)} = Sc \cdot f_{s2}$$

$$Sc_{MeHg} = Sc \cdot f_{s3}$$

where:

Sc	=	soil concentration of total mercury (μg pollutant/g soil)
Sc_{Hg0}	=	soil concentration of elemental mercury (μg pollutant/g soil)
$Sc_{Hg(II)}$	=	soil concentration of divalent mercury (μg pollutant/g soil)
Sc_{MeHg}	=	soil concentration of methylmercury (μg pollutant /g soil)
f_{s1}	=	fraction of soil concentration that is elemental mercury
f_{s2}	=	fraction of soil concentration that is divalent mercury
f_{s3}	=	fraction of soil concentration that is methylmercury

The total concentration of each mercury component in soil was assumed to reach equilibrium between its particulate and aqueous phases according to the following equations:

$$C_{st,i} = Sc_i\ BD$$

$$C_{ps,i} = C_{ds,i}\ Kd_{s,i}$$

$$C_{ps,i} = \frac{Sc_i\ Kd_{s,i}\ BD}{\theta_s + Kd_{s,i}\ BD}$$

$$C_{ds,i} = \frac{Sc_i\ BD}{\theta_s + Kd_{s,i}\ BD}$$

Where:

Sc_i	=	total soil concentration of component "i" (μg/g)
θ_s	=	volumetric soil water content (L_{water}/L)
$Kd_{s,i}$	=	soil/water partition coefficient for component "j" (L/kg)
BD	=	soil bulk density (g/cm^3)
$C_{st,i}$	=	total soil concentration of component "i" (mg/L)
$C_{ds,i}$	=	concentration of "i" dissolved in pore water (mg/L)
$C_{ps,i}$	=	concentration of "i" in particulate phase (mg/kg)

A derivation of the soil equilibrium equations is given in Section 3.3.2.

D.3.1.3 Loads to Watershed Soils

The total pollutant load term L_W in the mass balance equation is the sum of the loadings for each component "i." Component loadings include wetfall and dryfall fluxes, atmospheric diffusion fluxes and internal transformation loads:

$$L_{W,i} = Dydw_i + Dyww_i + L_{Is,i} + L_{DIF,i} \quad (48)$$

where:

$Dydw_i$	=	yearly average dry depositional flux of component "i" (g/m^2-yr)
$Dyww_i$	=	yearly average wet depositional flux of component "i" (g/m^2-yr)
$L_{Is,i}$	=	internal transformation load of component "i" per areal basis (g/m^2-yr)
$L_{DIF,i}$	=	atmospheric diffusion flux of component "i" to soil (g/m^2-yr)

Internal transformation loads are set to 0 in the equilibrium component approach. Wet and dry depositional fluxes were determined by measurement or by air modeling and were specified as input to this model. The load due to vapor diffusion is given as the following:

$$L_{DIF,i} = 0.31536\ K_{t,i}\ C_{atm,i}$$

where:

$L_{DIF,i}$	=	atmospheric diffusion flux of component "i" to soil (g/m^2-yr)

$K_{t,i}$ = gas phase mass transfer coefficient for component "i" (cm/s; see Eq [4-6], IED)

$C_{atm,i}$ = gas phase atmospheric concentration for component "i" ($\mu g/m^3$)

D.3.1.4 Loss Processes in Watershed Soils

The total chemical loss rate constant ks in the soil mass balance equation is the weighted sum of the chemical loss rate constants for each component "i":

$$ks = \sum_i ks_i \cdot f_{si}$$

where:

ks_i = total soil loss constant for component "i" (yr^{-1})

f_{si} = fraction of soil concentration that is component "i" (i.e., elemental, divalent, and methylmercury)

The total chemical loss rate constant for component "i" is due to several physical and chemical processes:

$$ks_i = ksl_i + kse_i + ksr_i + ksg_i + ksv_i$$

where:

ks_i = soil loss constant for component "i" due to all processes (yr^{-1})

ksl_i = soil loss constant due to leaching (yr^{-1})

kse_i = soil loss constant due to erosion (yr^{-1})

ksr_i = soil loss constant due to runoff (yr^{-1})

ksg_i = soil loss constant due to chemical transformation/ degradation (yr^{-1})

ksv_i = soil loss constant due to volatilization (yr^{-1})

The degradation constant, ksg_i, is set to 0 in the equilibrium component approach. The other four constants are given by these equations:

$$ksl_i = \frac{P + I - Ro - EV}{\theta_s Z}\left(\frac{1}{1.0 + Kd_{s,i}\, BD/\theta_s}\right)$$

$$kse_i = \frac{0.1\, X_e\, SD\, ER}{BD\, Z}\left(\frac{Kd_{s,i}\, BD}{\theta_s + Kd_{s,i}\, BD}\right)$$

$$ksr_i = \frac{Ro}{\theta_s Z}\left(\frac{1}{1 + Kd_{s,i}\, BD/\theta_s}\right)$$

$$ksv_i = Ke_i\, Kt_i$$

where:

P	=	average annual precipitation (cm/yr)
I	=	average annual irrigation (cm/yr)
Ro	=	average annual runoff (cm/yr)
Ev	=	average annual evapotranspiration (cm/yr)
θ_s	=	volumetric water content (dimensionless; cm^3/cm^3)
Z	=	watershed mixing zone depth (cm)
BD	=	soil bulk density (g/cm^3)
SD	=	sediment delivery ratio
ER	=	contaminant enrichment ratio
$Kd_{s,i}$	=	soil-water partition coefficient for component "i" (cm^3/g)
X_e	=	unit soil loss (kg/m^2-yr; see Eq [9-3], IED; Wischmeier and Smith, 1978)
Ke_i	=	equilibrium coefficient for component "i" (s/cm-yr); see Eq [4-5], IED; Travis, et al., 1983)
Kt_i	=	gas phase mass transfer coefficient for component "i" (cm/s; see Eq [4-6], IED; Travis, et al., 1983)
0.1	=	units conversion factor

Sc is the concentration resulting from contaminated particles depositing on and mixing with surface soils. For mercury components, where $Kd_{s,i}$ values are large, Sc_i was essentially equal to the sorbed concentration, $C_{ps,i}$, and the dissolved phase concentration, $C_{ds,i}$, was small. Mercury components depositing as particles were assumed to reequilibrate in the soil/soil water system (see the state equations above). In the listing of state equations, the reequilibrated sorbed phase concentration, $C_{ps,i}$, and the dissolved phase concentration, $C_{ds,i}$, were used to estimate loads to the water body due to soil erosion and surface runoff, respectively.

D.3.2 The Aquatic Equations

The following framework for estimating surface water impacts from stack emissions estimates water column as well as bed sediment concentrations. Water column concentrations included dissolved, sorbed to suspended sediments and total (sorbed plus dissolved, or total contaminant divided by total water volume). This framework also provides three concentrations for the bed sediments: dissolved in pore water, sorbed to bed sediments, and total. The model accounts for five routes of contaminant entry into the water body: erosion of chemical sorbed to soil particles; runoff of

dissolved chemical in runoff water: deposituon of particle-bound contaminant through wetfall and dryfall; and diffusion of vapor phase contaminants into the water body. The model also accounts for four dissipation processes that remove contaminants from the water column and/or bed sediment reservoirs: decay of total contaminants (sorbed + dissolved) within the water column; decay of total contaminants (sorbed + dissolved) within the bed sediment; volatilization of dissolved phase out of the water column; and removal of total contaminant via "burial" from the surficial bed sediment layer. This burial rate constant is a function of the deposition of sediments from the water column to the bed; it accounts for the fact that much of the soil eroding into a water body annually becomes bottom sediment rather than suspended sediment. The impact to the water body was assumed to be uniform. This tends to be more realistic for smaller water bodies as compared to large river systems. Key assumptions in the surface water impact algorithm are the following.

- The partitioning of the contaminant within the sediment/water matrices - suspended solids in the water column, and bed sediments in the benthos of the water body - can be described by partition coefficients.

- One route of entry into the surface water body is direct deposition. A second route of entry is contaminant dissolved in annual surface runoff. This is estimated as a function of the contaminant dissolved in soil water and annual water runoff. A third route of entry is via soil erosion. A sorbed concentration of contaminant in soil, together with an annual soil erosion estimate, a sediment delivery ratio and an enrichment ratio, can be used to describe the delivery of contaminant to the water body via soil erosion. A sediment delivery ratio serves to reduce the total potential amount of soil erosion (where the total potential equals a unit erosion rate in kg/m^2 multiplied by the watershed area, in m^2) reaching the water body recognizing that most of the erosion within a watershed during a year deposits prior to reaching the water body. The enrichment ratio accounts for the fact that eroding soils tend to be lighter in texture, be more abundant in surface area, and have higher organic carbon. All these characteristics lead to concentrations in eroded soils that tend to be higher in concentration as compared to in *situ* soils. A fourth and final route of entry is via diffusion in the gaseous phase. The dissolved concentration in a water body is driven toward equilibrium with the vapor phase concentration above the water body. At equilibrium, gaseous diffusion into the water body is matched by volatilization out of the water body. Gaseous diffusion is estimated with a transfer rate (determined internally given user inputs) and a vapor phase air concentration. This air concentration is specified by the user and is an output of the atmospheric transport model.

- For the surface water solution algorithm, it is assumed that equilibrium is maintained between contaminants within the water column and contaminants in surficial bed sediments. Equilibrium is established when the dissolved phase concentration in the water column is equal to the dissolved phase concentration within the surficial bed sediments. This condition is imposed by the water body equations.

- A rate of contaminant "burial" in bed sediments is estimated as a function of the rate at which sediments deposit from the water column onto the surficial sediment layer. This burial represents a permanent sink, recognizing that a portion of the eroded soil and sorbed contaminant becomes bottom sediment while the remainder becomes suspended sediment. This solution assumes that there will be a net depositional loss, even though resuspension and redeposition of sediments is ongoing, particularly with moving water bodies. For cases where the net deposition rate is zero, there will be no burial loss calculated.

- Separate water column and benthic decay rate constants allow for the consideration of decay mechanisms that remove contaminants from the water body, optionally linking them through internal loading to a reaction product. For the equilibrium component approach to mercury, decay constants are set to 0.

Figure D-3 displays the framework for this analysis, with a listing of the ten concentrations that were part of the solution algorithm. In the following sections, the mass balance equations and the equilibrium state equations that link the concentrations are developed.

D.3.2.1 Chemical Mass Balance in the Water Body

Taking Figure D-4 as a control volume for the water body, it can be seen that a steady-state mass balance equation can be written that balances chemical loadings with outflow and loss:

$$C_{wtot} = \frac{L_T}{Vf_x f_{water} d_z/d_w + k_{wt} V_z}$$

where:

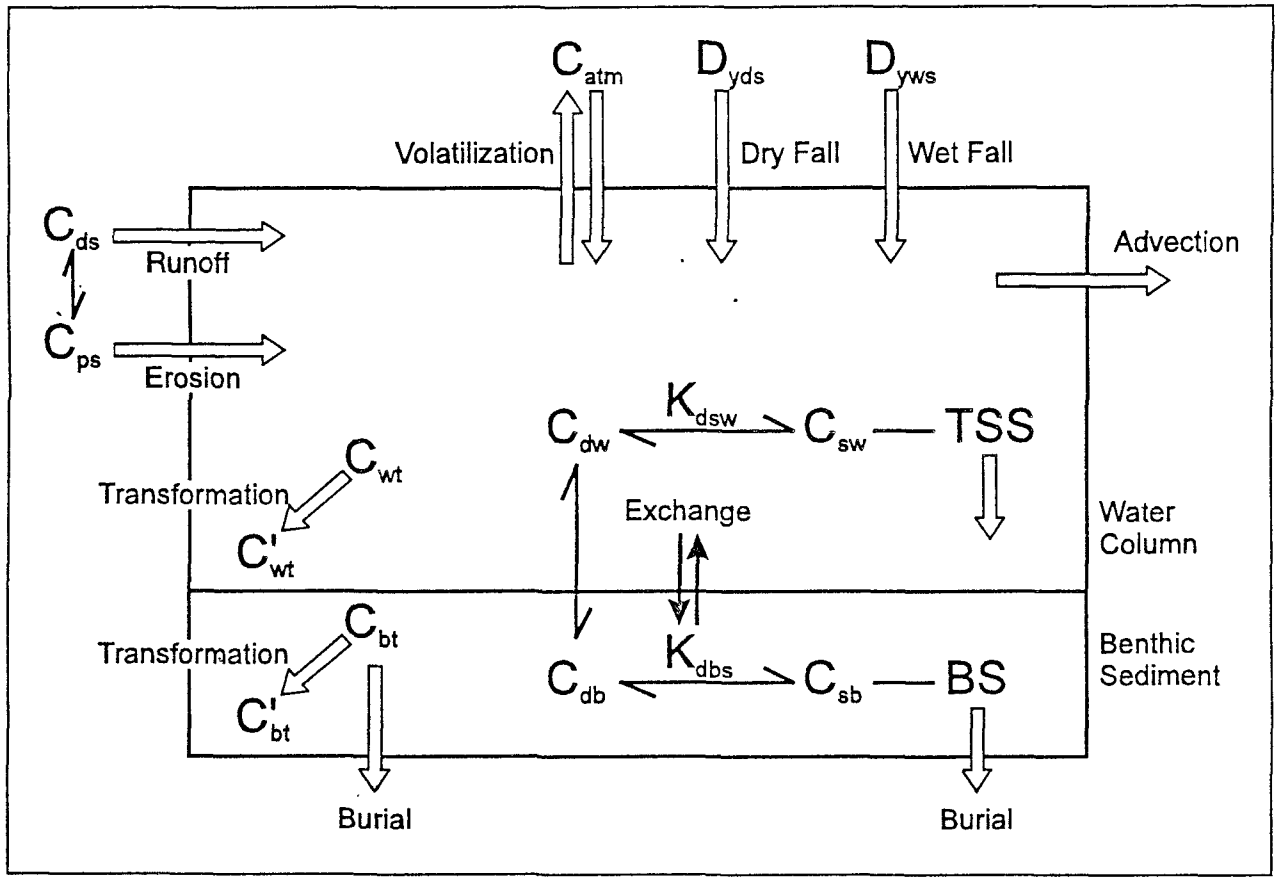
C_{wtot} = total water body concentration, including water column and benthic sediment (mg/L)
L_T = total chemical load into water body, including deposition, runoff, erosion, atmospheric diffusion, and internal chemical transformation (g/yr)
Vf_x = average volumetric flow rate through water body (m^3/yr)
V_z = total volume of water body or water body segment being considered, including water column and benthic sediment (m^3)
k_{wt} = total first order dissipation rate constant, including water column and benthic degradation, volatilization, and burial (yr^{-1})
f_{water} = fraction of total water body contaminant concentration that occurs in the water column
d_w = depth of the water column (m)
d_z = total depth of water body, d_w+d_b (m)

The first term in the denominator accounts for the advective flow of chemical from the water column, while the second term accounts for loss processes from the bed and water column. This mass balance equation is derived in Section D.3.3.3. The terms in this equation are discussed in sections below.

D.3.2.2 Sediment Mass Balance in the Water Body

Before calculating chemical fate, a mass balance equation for sediments in the water body must be solved. Taking Figure D-4 as a control volume for the water body, it can be seen that a steady-state mass balance equation can be written that balances sediment loadings with outflow and loss:

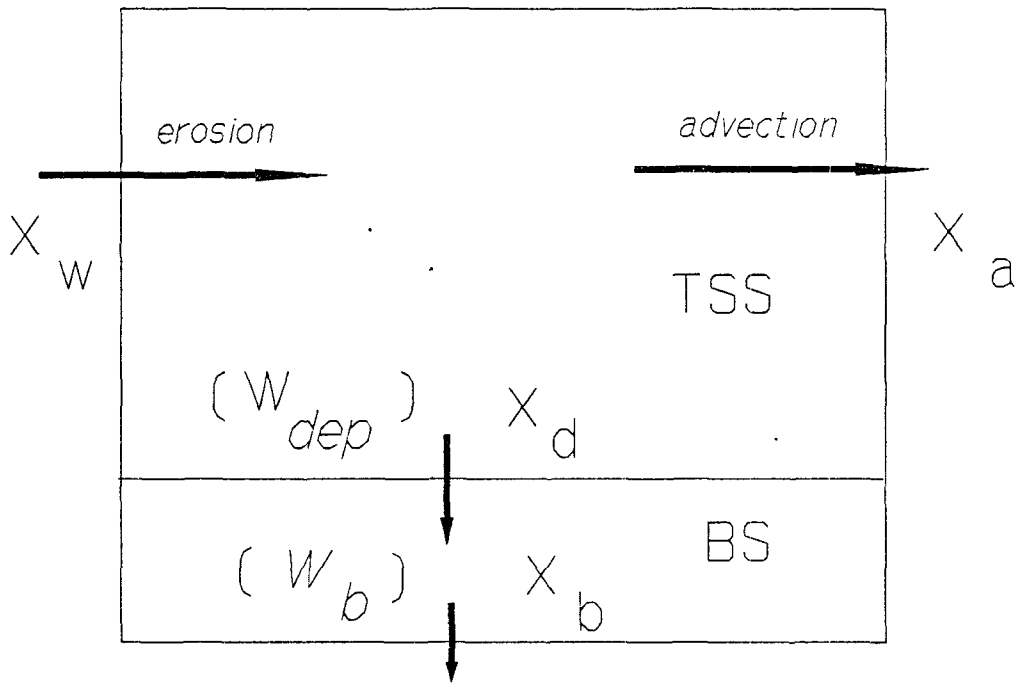
Figure D-3
Overview of the IEM2 Water Body Processes



Definitions for Figure D-3

C_{ds}	concentration dissolved in soil water		mg/L
C_{ps}	concentration sorbed to soil		mg/kg
D_{yds}	yearly dry deposition to surface water		mg/yr
D_{yws}	yearly wet deposition to surface water		mg/yr
C_{atm}	vapor phase atmospheric concentration		$\mu g/m^3$
C_{wt}	total concentration in water column		mg/L
C_{wtot}	total water concentration in surface water system, including water column plus benthic sediment (not shown in figure)		mg/L
C_{dw}	dissolved phase water concentration		mg/L
C_{sw}	sorbed phase water concentration	mg/kg	
C_{bt}	total concentration in bottom sediment		mg/L
C_{db}	concentration dissolved in bed sediment pore water	mg/L	
C_{sb}	concentration sorbed to bottom sediments	mg/kg	

Figure D-4
IEM2 Steady State Sediment Balance in Water Bodies



Definitions for Figure D-4

X_w	soil erosion input from depositional area	g/yr
X_a	advective loss from water body	g/yr
X_d	deposition onto bottom sediment	g/yr
X_b	burial below bottom sediment layer	g/yr
TSS	suspended solids concentration	mg/L
BS	bottom sediments concentration	g/L
W_{dep}	rate of deposition onto bed sediment	m/yr
W_b	rate of burial	m/yr

$$TSS = \frac{X_e \; WA_L \; SD \; 10^3}{Vf_x + W_{dep} \; WA_w}$$

where:

TSS	=	suspended solids concentration (mg/L)
X_e	=	unit soil erosion flux, calculated in the soils section from the USLE equation (kg/m^2-yr)
WA_L	=	watershed surface area (m^2)
SD	=	watershed sediment delivery ratio (unitless)
Vf_x	=	average volumetric flow rate through water body (m^3/yr)
W_{dep}	=	suspended solids deposition rate (m/yr)
WA_w	=	water body surface area (m^2)
10^3	=	units conversion factor

The first term in the denominator accounts for the advective flow of sediment from the water column, while the second term accounts for depositional loss from the water column. This mass balance equation is derived in Section 3.3.3. The terms in this equation are discussed in sections below.

In the second part of the solids balance, the mass deposited to the bed, X_d, was set equal to the mass buried, X_b. Solving for the burial rate gives the following:

$$W_b = W_{dep} \frac{TSS \; 10^{-6}}{BS}$$

where:

W_b	=	burial rate (m/yr)
W_{dep}	=	deposition rate (m/yr)
TSS	=	suspended solids concentration (mg/L)
BS	=	benthic solids concentration (kg/L)
10^{-6}	=	conversion factor (kg/mg)

Finally, the benthic porosity, the volume of water per volume of benthic space, was calculated from the benthic solids concentration and sediment density:

$$\theta_{bs} = 1 - BS/\rho_s$$

where:

θ_{bs}	=	benthic porosity (L/L)
BS	=	benthic solids concentration (kg/L)
ρ_s	=	solids density, 2.65 kg/L

For input benthic solids concentrations between 0.5 and 1.5 kg/L, benthic porosity ranged between 0.8 and 0.4.

The suspended solids, benthic solids, and benthic porosity were used in the chemical equilibrium speciation equations. The burial rate was used in the chemical burial equation. These equations are developed below.

D.3.2.3 Equilibrium Speciation in Water Body

Total mercury in the water body is assumed to be distributed among three components -- Hg^0, Hg(II), and methylmercury. The steady-state fraction of the total in each component in the water column is specified by the user using the following relationship:

$$C_{wt,Hg^0} = C_{wt} \cdot f_{w1}$$

$$C_{wt,Hg(II)} = C_{wt} \cdot f_{w2}$$

$$C_{wt,MeHg} = C_{wt} \cdot f_{w3}$$

where:

C_{wt}	=	water column concentration of total mercury (µg/L)
$C_{wt,Hg0}$	=	water column concentration of elemental mercury (µg/L)
$C_{wt,Hg(II)}$	=	water column concentration of divalent mercury (µg/L)
$C_{wt,MeHg}$	=	water column concentration of methylmercury (µg/L)
f_{w1}	=	fraction of water column concentration that is elemental mercury
f_{w2}	=	fraction of water column concentration that is divalent mercury
f_{w3}	=	fraction of water column concentration that is methylmercury

The total concentration of each mercury component in the water body, $C_{wtot,i}$, was assumed to reach equilibrium between the benthic and water column compartments and between its particulate and aqueous phases within each compartment. $C_{wtot,i}$ gives the mass per volume of the entire water body, including both water column and benthic sediment. The water column concentration, $C_{wt,i}$, was based on the water column volume; the benthic concentration, $C_{bt,i}$, was based on the benthic volume. The equilibrium relationships are given by the following equations, which are derived in Section 3.3.4.

Surface Water System --

$$C_{wtot,i} = C_{wt,i} \cdot d_w/d_z + C_{bt,i} \cdot d_b/d_z$$

$$= \textit{see mass balance equation}$$

$$f_{water,i} = \frac{(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}) \cdot d_w/d_z}{(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}) \cdot d_w/d_z + (\theta_{bs} + Kd_{bs,i} \cdot BS) \cdot d_b/d_z}$$

$$f_{benth,i} = \frac{(\theta_{bs} + Kd_{bs,i} \cdot BS) \cdot d_b/d_z}{(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}) \cdot d_w/d_z + (\theta_{bs} + Kd_{bs,i} \cdot BS) \cdot d_b/d_z}$$

Water Column --

$$C_{wt,i} = f_{water,i} \cdot C_{wtot,i} \cdot d_z/d_w$$

$$C_{dw,i} = C_{wt,i} \cdot f_{dw,i} = C_{wt,i}\left[\frac{1}{1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}}\right]$$

$$C_{sw,i} = Kd_{sw,i} \cdot C_{dw,i}$$

Bed Sediment --

$$C_{bt,i} = f_{benth,i} \cdot C_{wtot,i} \cdot d_z / d_b$$

$$C_{db,i} = C_{bt,i} \cdot f_{db,i} / \theta_{bs} = C_{bt,i}\left[\frac{1}{\theta_{bs} + Kd_{bs,i} \cdot BS}\right]$$

$$C_{sb,i} = Kd_{bs,i} \cdot C_{db,i}$$

Note that by substituting the relationship between $C_{wt,i}$ and $C_{wtot,i}$ into the expression for $C_{bt,i}$, one can obtain benthic concentrations as a function of water column concentrations:

$$C_{bt,i} = C_{wt,i} \cdot \frac{\theta_{bs} + Kd_{bs,i} \cdot BS}{1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}}$$

$$C_{db,i} = C_{dw,i}$$

$$C_{sb,i} = C_{ps,i} \cdot Kd_{bs,i} / Kd_{sw,i}$$

where:

θ_{bs}	=	bed sediment porosity (L_{water}/L)
$Kd_{sw,i}$	=	suspended sediment/surface water partition coefficient for component "i" (L/kg)
$Kd_{bs,i}$	=	bottom sediment/pore water partition coefficient for component "i" (L/kg)
TSS	=	total suspended solids (mg/L)
BS	=	bed sediment concentration (g/cm^3)
d_w	=	depth of the water column (m)
d_b	=	depth of the upper benthic layer (m)
d_z	=	total depth of water body, d_w+d_b (m)
$f_{water,i}$	=	fraction of total water body component "i" concentration that occurs in the water column
$f_{benth,i}$	=	fraction of total water body component "i" concentration that occurs in the bed sediment
$f_{dw,i}$	=	fraction of water column component "i" concentration that is dissolved
$f_{db,i}$	=	fraction of bed sediment component "i" concentration that is dissolved

D.3.2.4 Loads To The Water Body

The total chemical load term L_T in the mass balance equation is the sum of the loadings for each component "i." Component loadings included wet and dry deposition, impervious and pervious runoff, erosion, atmospheric diffusion, and internal transformation:

$$L_{T,i} = L_{Dep,i} + L_{RI,i} + L_{R,i} + L_{E,i} + L_{Dif,i} + L_{I,i}$$

where:

$L_{T,i}$	=	total component "i" load to the water body (g/yr)
$L_{Dep,i}$	=	deposition of particle bound component "i" (g/yr)
$L_{RI,i}$	=	runoff load from impervious surfaces (g/yr)
$L_{R,i}$	=	runoff load from pervious surfaces (g/yr)
$L_{E,i}$	=	soil erosion load (g/yr)
$L_{Dif,i}$	=	diffusion of vapor phase component "i" (g/yr)
$L_{I,i}$	=	internal transformation load, equal to 0 for equilibrium mercury chemistry (g/yr)

The runoff and erosion loads required estimation of average contaminant concentration in watershed soils that comprise the depositional area. These concentrations were developed in terrestrial sections above.

Load due to direct deposition -- The load to surface waters via direct deposition is solved as follows:

$$L_{Dep,i} = (Dyds_i + Dyws_i)\ WA_w$$

where:

$L_{Dep,i}$	=	direct component "i" deposition load (g/yr)
$Dyds_i$	=	yearly dry deposition rate of component "i" onto surface water body (g pollutant/m^2-yr)
$Dyws_i$	=	yearly wet deposition rate of component "i" onto surface water body (g pollutant/m^2-yr)
WA_w	=	water body area (m^2)

Load due to impervious surface runoff -- A fraction of the wet and dry chemical deposition in the watershed will be to impervious surfaces. Dry deposition may accumulate and be washed off during rain events. If the impervious surface includes gutters, the pollutant load will be transported to surface waters, bypassing the watershed soils. The average load from such impervious surfaces is given by this equation:

$$L_{RI,i} = (D_{yww,i} + D_{ydw,i})\ WA_I$$

where:

$L_{RI,i}$	=	impervious surface runoff load for component "i" (g/yr)
WA_I	=	impervious watershed area receiving pollutant deposition (m^2)
$D_{yww,i}$	=	yearly wet deposition flux of component "i" onto the watershed (g/m^2-yr)
$D_{ydw,i}$	=	yearly dry deposition flux of component "i" onto the watershed (g/m^2-yr)

Load due to pervious surface runoff -- Most of the chemical deposition to a watershed will be to pervious soil surfaces. These loads are accounted for in the soil mass balance equation. During periodic runoff events, dissolved chemical concentrations in the soil are transported to surface waters as given by this equation:

$$L_{R,i} = Ro\ (WA_L - WA_I) \left[\frac{Sc_i\ BD}{\theta_s + Kd_{s,i}\ BD} \right] 10^{-2}$$

where:

$L_{R,i}$	=	pervious surface runoff load for component "i" (g/yr)
Ro	=	average annual runoff (cm/yr)
Sc_i	=	component "i" concentration in watershed soils (µg/g)
BD	=	soil bulk density (g/cm^3)
θ_s	=	volumetric soil water content (cm^3/cm^3)
$Kd_{s,i}$	=	soil-water partition coefficient for component "i" (L/kg or cm^3/g)
WA_L	=	total watershed area receiving pollutant deposition (m^2)
WA_I	=	impervious watershed area receiving pollutant deposition (m^2)
10^{-2}	=	units conversion factor (g^2/kg-µg)

Load due to soil erosion -- During periodic erosion events, particulate chemical concentrations in the soil are transported to surface waters as described by this relationship:

$$L_{E,i} = X_e\ (WA_L - WA_I)\ SD\ ER \left[\frac{Sc_i\ Kd_{s,i}\ BD}{\theta_s + Kd_{s,i}\ BD} \right] 10^{-3}$$

where:

$L_{E,i}$	=	soil erosion load for component "i" (g/yr)
X_e	=	unit soil loss (kg/m^2-yr)
Sc_i	=	component "i" concentration in watershed soils (µg/g)
BD	=	soil bulk density (g/cm^3)
θ_s	=	volumetric soil water content (cm^3/cm^3)
$Kd_{s,i}$	=	soil-water partition coefficient for component "i" (L/kg or cm^3/g)
WA_L	=	total watershed area receiving pollutant deposition (m^2)
WA_I	=	impervious watershed area receiving pollutant deposition (m^2)
SD	=	watershed sediment delivery ratio (unitless)
ER	=	soil enrichment ratio (unitless)
10^{-3}	=	units conversion factor (g-cm^2/µg-m^2)

Load due to gaseous diffusion -- The change in the total water concentration over time due to volatilization is given by this:

$$\frac{\partial C_{wtot,i}}{\partial t}\Big|_{volat} = -\frac{K_{v,i}}{D}\left(f_{water,i} f_{dw,i} C_{wtot,i} - \frac{C_{atm,i}\ 10^{-6}}{H_i' RT_k} \right)$$

where:

$C_{wtot,i}$	=	total water body component "i" concentration (mg/L)

$K_{v,i}$	=	overall component "i" transfer rate (m/yr)
D	=	depth of water body (m)
$f_{water,i}$	=	fraction of total water body component "i" concentration that occurs in the water column
$f_{dw,i}$	=	fraction of water column component "i" concentration that is dissolved
$C_{atm,i}$	=	component "i" vapor phase air concentration over water body ($\mu g/m^3$)
H_i	=	component "i" Henry's Constant ($atm\text{-}m^3/mole$)
R	=	universal gas constant, $8.206*10^{-5}$ $atm\text{-}m^3/mole\text{-}K$
T_k	=	water body temperature (°K)
10^{-6}	=	units conversion factor

This treatment of volatilization is based on the well-known two-film theory (Whitman, 1923), as implemented in standard chemical fate models (Burns, et al., 1982, Ambrose, et al., 1988). The right side of the volatilization equation contains two terms. The first term constitutes a first order loss rate of aqueous contaminant, which is covered in more detail below. The second term in the volatilization equation describes diffusion of gas-phase contaminant from the atmosphere into the water body. Because this term is independent of water body contaminant concentration, it can be treated as an external load. As formulated above, this term has units of mg/L-yr. It must be converted to loading units by multiplying by the water column volume, V. Noting that V/D is equal to the surface water area WA_w, we see that the atmospheric diffusion load is given as this:

$$L_{Dif,i} = \frac{K_{v,i}\, C_{atm,i}\, WA_w\, 10^{-6}}{H_i/RT_k}$$

where:

$L_{Dif,i}$	=	diffusion of vapor phase component "i" (g/yr)
$K_{v,i}$	=	the overall component "i" transfer rate (m/yr)
WA_w	=	surface water body area (m^2)
$C_{atm,i}$	=	component "i" vapor phase air concentration over water body ($\mu g/m^3$)
H_i	=	component "i" Henry's Constant ($atm\text{-}m^3/mole$)
R	=	universal gas constant (8.206×10^{-5} $atm\text{-}m^3/mole\text{-}°K$)
T_k	=	water body temperature (°K)
10^{-6}	=	units conversion factor

D.3.2.5 Advective Flow From The Water Body

The first term in the denominator of the chemical mass balance equation accounts for advective flow from the water body. It is the product of the average annual volumetric flow rate, Vf_x; the fraction of the chemical in the water body that is present in the water column, f_{water}; and the adjustment factor d_z/d_w, which normalizes the outflowing chemical concentration to a water column volume basis. An impacted water body derives its annual flow from its watershed or effective drainage area. Flow and watershed area, then, are related, and compatible values should be specified by the user. Given the area of drainage, one way to estimate annual flow volume is to multiply total drainage area (in length squared units) by a unit surface water runoff (in length per time). The *Water Atlas of the United States* (Geraghty et al., 1973) provides maps with isolines of annual average surface water runoff, which is defined as all flow contributions to surface water bodies, including direct runoff, shallow interflow, and groundwater recharge. The values ranged from 5 to 40 in/yr in various parts of the United States.

D.3.2.6 Chemical Dissipation Within The Water Body

The second term in the denominator of the chemical mass balance equation accounts for dissipation within the water body. It is the product of the water body volume, V_t, and the total first order dissipation rate constant, k_{wt}. The water body volume, in units of m^3, together with the annual flow rate, in m^3/yr, determines the average residence time of a pollutant traveling through the water body. The residence time for Lake Erie is about 10 years, for example, while for the larger Lake Superior it is estimated to be 200 years. For a swiftly moving river, on the other hand, the residence time can be on the order of hours (1 hour = 0.00011 yr). Larger volumes and residence times allow the internal dissipation processes to have a larger effect on pollutant concentration, while smaller volumes and residence times lessen the effect. It is necessary to specify reasonable volumes for the type of surface water body being represented. In addition, compatible values for related water body parameters, such as surface area, WA_w must be used. The water body volume divided by the surface area gives the average depth, which can vary from a fraction of a meter for small streams to a few meters for shallow reservoirs to tens of meters for deep lakes.

The total dissipation rate constant, k_{wt}, applies to the total water body concentration, C_{wtot}, and is the weighted sum of the chemical loss rate constants for each component "i":

$$k_{wt} = \sum_i k_{wt,i} \cdot f_{wi}$$

where:

ks_i = total soil loss constant for component "i" (yr^{-1})
f_{si} = fraction of soil concentration that is component "i" (i.e., elemental, divalent, and methylmercury)

The total chemical loss rate constant for component "i" includes processes affecting any of the chemical phases -- dissolved or sorbed in the water column or benthic sediments. Volatilization, water column and benthic degradation, and burial are considered in this relationship:

$$k_{wt,i} = f_{water,i}\, k_{gw,i} + f_{benth,i}\, k_{gb,i} + f_{water,i}\, k_{v,i} + f_{benth,i}\, k_{b,i}$$

where:

$k_{wt,i}$ = overall total water body dissipation rate constant for component "i" (yr^{-1})
$k_{gw,i}$ = water column degradation or transformation rate constant for component "i" (yr^{-1})
$k_{gb,i}$ = benthic degradation or transformation rate constant for component "i" (yr^{-1})
$k_{v,i}$ = water column volatilization rate constant (yr^{-1})
$k_{b,i}$ = benthic burial rate constant for component "i" (yr^{-1})
$f_{water,i}$ = fraction of total water body component "i" concentration that occurs in the water column
$f_{benth,i}$ = fraction of total water body component "i" concentration that occurs in the benthic sediment

These processes are described below.

Chemical/Biological Degradation -- Contaminants can be degraded and transformed by a number of processes in the water column or in the benthic sediment. Mercury components are subject to oxidation, reduction, and methylation. In the equilibrium approach taken here, the transformation rates were set to 0 and the fraction of total chemical in each component was specified directly.

Volatilization -- Volatile chemicals can move between the water column and the overlying air, as described by Equation (3-41). The right side of this equation contains two terms. The second term describes diffusion into the water from the atmosphere and was treated as an external load. The first term, $(K_{v,i}f_{water,i}f_{dw,i}C_{wtot,i}/D)$, constitutes a first order loss rate of aqueous contaminant. This term includes the quantity $f_{water,i}f_{dw,i}C_{wtot,i}$, which is equal to the water column dissolved phase concentration $C_{dw,i}$ and which is subject to volatilization loss. The rate constant for volatilization from the water column, $k_{v,i}$, is given as this:

$$k_{v,i} = \frac{K_{v,i}\, f_{dw,i}}{D}$$

where:

$k_{v,i}$	=	water column volatilization loss rate constant for component "i" (yr^{-1})
$K_{v,i}$	=	overall transfer rate, or conductivity for component "i" (m/yr)
$f_{dw,i}$	=	fraction of component "i" in the water column that is dissolved
D	=	water body depth (m)

The overall transfer rate, $K_{v,i}$ or conductivity, was determined by the two-layer resistance model (Whitman, 1923; or see Burns, et al., 1982 or Ambrose, et al., 1988). The two-resistance method assumes that two "stagnant films" are bounded on either side by well mixed compartments. Concentration differences serve as the driving force for the water layer diffusion. Pressure differences drive the diffusion for the air layer. From mass balance considerations, it is obvious that the same mass must pass through both films; thus, the two resistances combine in series, so that the conductivity is the reciprocal of the total resistance:

$$K_{v,i} = (R_{L,i} + R_{G,i})^{-1} = \left(K_{L,i}^{-1} + \left(K_{G,i} \frac{H_i}{R\, T_K} \right)^{-1} \right)^{-1}$$

where:

$R_{L,i}$	=	liquid phase resistance (year/m)
$K_{L.i}$	=	liquid phase transfer coefficient (m/year)
$R_{G,i}$	=	gas phase resistance (year/m)
$K_{G,i}$	=	gas phase transfer coefficient (m/year)
R	=	universal gas constant (atm-m^3/mole-°K)
H_i	=	Henry's law constant for component "i" (atm-m^3/mole)
T_k	=	water body temperature (°K)

The value of $K_{v,i}$, the conductivity, depends on the intensity of turbulence in a water body and in the overlying atmosphere. As the Henry's Law coefficient increases, the conductivity tends to be increasingly influenced by the intensity of turbulence in water. As the Henry's Law coefficient decreases, the value of the conductivity tends to be increasingly influenced by the intensity of atmospheric turbulence.

Because Henry's Law coefficient generally increases with increasing vapor pressure of a compound and generally decreases with increasing solubility of a compound, highly volatile low solubility compounds are most likely to exhibit mass transfer limitations in water, and relatively nonvolatile high solubility compounds are more likely to exhibit mass transfer limitations in the air. Volatilization is usually of relatively less magnitude in lakes and reservoirs than in rivers and streams.

The estimated volatilization rate constant was for a nominal temperature of 20 °C. It is adjusted for the actual water temperature using the equation:

$$K_{v,i,T} = K_{v,i,20}\ \theta^{(T-20)}$$

where:

θ	=	temperature correction factor, set to 1.026.
T	=	water body temperature (°C)

There have been a variety of methods proposed to compute the liquid ($K_{L,i}$) and gas phase ($K_{G,i}$) transfer coefficients. The particular method that was used in the exposure assessment is the O'Connor-Dobbins (1958) method.

The liquid and gas film transfer coefficients computed under this option vary with the type of water body. The type of water body was specified as one of the surface water constants and can either be a flowing stream, river or estuary, or a stagnant pond or lake. The primary difference is that in a flowing water body, the turbulence is primarily a function of the stream velocity, while for stagnant water bodies, wind shear may dominate. The formulations used to compute the transfer coefficients vary with the water body type, as shown below.

Flowing Stream or River -- For a flowing system, the transfer coefficients are controlled by flow-induced turbulence. For these systems, the liquid film transfer coefficient (K_L) was computed using the O'Connor-Dobbins (1958) formula:

$$K_{L,i} = \left(\frac{10^{-4}\ D_{w,i}\ u}{d_w}\right)^{1/2} (3.15x10^7)$$

where:

$K_{L,i}$	=	liquid phase transfer coefficient for component "i" (m/year)
u	=	current velocity (m/s)
$D_{w,i}$	=	diffusivity of the component "i" in water (cm^2/s)
d_w	=	water depth (m)
10^{-4}	=	units conversion factor
$3.15x10^7$	=	units conversion factor

The gas transfer coefficient (K_G) was assumed constant at 36500 m/yr for flowing systems.

Quiescent Lake or Pond -- For a stagnant system, the transfer coefficients are controlled by wind-induced turbulence. For stagnant systems, the liquid film transfer coefficient (K_L) was computed using the O'Connor (1983) equations:

$$K_{L,i} = u^* \left(\frac{\rho_a}{\rho_w}\right)^{0.5} \left(\frac{k^{0.33}}{\lambda_2}\right) Sc_{w,i}^{-0.67} (3.15x10^7)$$

where:

$$K_{G,i} = u^* \left(\frac{k^{0.33}}{\lambda_2} \right) Sc_{a,i}^{-0.67} (3.15x10^7)$$

$$u^* = C_d^{0.5} W$$

$$Sc_{a,i} = \frac{\mu_a}{\rho_a D_{a,i}} = \frac{\nu_a}{D_{a,i}}$$

$$D_{a,i} = \frac{1.9}{MW_i^{2/3}}$$

$$\nu_a = (1.32 + 0.009 T_a) \times 10^{-1}$$

$$Sc_{w,i} = \frac{\mu_w}{\rho_w D_{w,i}}$$

$$D_{w,i} = \frac{22 \times 10^{-5}}{MW_i^{2/3}}$$

$$\rho_w = 1 - 8.8 \times 10^{-5} T_w$$

$$\log(\mu_w) = \left[\frac{1301}{998.333 + 8.1855(T_w - 20) + 0.00585(T_w - 20)^2} \right] - 3.0233$$

and:

u^*	=	shear velocity (m/s)
C_d	=	drag coefficient (= 0.0011)
W	=	wind velocity, 10 m above water surface (m/s)
ρ_a	=	density of air corresponding to the water temperature (g/cm ^{3})
ρ_w	=	density of water corresponding to the water temperature (g/cm ^{3})
k	=	von Karman's constant (= 0.4)
λ_2	=	dimensionless viscous sublayer thickness (= 4)
$Sc_{a,i}$	=	air Schmidt number for component "i" (dimensionless)

$Sc_{w,i}$	=	water Schmidt number for component "i" (dimensionless)
$D_{a,i}$	=	diffusivity of component "i" in air (cm^2/sec)
$D_{w,i}$	=	diffusivity of component "i" in water (cm^2/sec)
μ_a	=	viscosity of air corresponding to the air temperature (g/cm-s)
μ_w	=	viscosity of water corresponding to the water temperature (g/cm-s)
ν_a	=	dynamic viscosity of air (cm^2/sec)
MW_i	=	molecular weight of component "i"
T_a	=	air temperature (°C)
T_w	=	water temperature (°C)
3.15×10^7	=	units conversion factor

Deposition and Burial -- The benthic burial rate, W_b, was determined as a function of user input variables as part of the sediment balance. This burial rate is used to determine the mass loss of contaminant from the benthic sediment layer. As seen in Figure D.3, the burial loss rate was applied to the total benthic contaminant concentration, C_{bt}. The water body contaminant burial loss rate was solved by equating the mass loss rate of total water body chemical with mass loss rate of benthic chemical:

$$C_{wtot,i}\, V_t\, k_{b,i} = C_{bt,i}\, V_b\, \frac{W_b}{d_b}$$

where:

$C_{wtot,i}$	=	total water body component "i" concentration, including water column and benthic sediment (mg/L)
V_t	=	total volume of water body or water body segment being considered, including water column and benthic sediment (m^3)
$k_{b,i}$	=	first order burial rate constant for total component "i" (yr^{-1})
$C_{bt,i}$	=	total benthic component "i" concentration (mg/L)
V_b	=	volume of upper benthic sediment layer (m^3)
d_b	=	depth of the upper benthic sediment layer (m)
W_b	=	benthic burial rate (m/yr)

From the equilibrium state equations, it is seen that the total benthic contaminant concentration, $C_{bt,i}$, can be expressed as a function of the total water body concentration, $C_{wtot,i}$. Solving for the total chemical burial rate gives this:

$$k_{b,i} = f_{benth,i} \frac{W_{b,i}}{d_b}$$

where:

$f_{benth,i}$	=	fraction of total water body component "i" concentration that occurs in the bed sediment
d_b	=	depth of the upper benthic sediment layer (m)
W_b	=	burial rate (m/yr)

D.3.3 Derivation of Select Equations

In this section, some of the equations in Sections 3.1 and 3.2 are derived. A summary of notation is given in Section 3.4.

D.3.3.1 Terrestrial Mass Balance Relationships

Taking Figure D-2 as a control volume for the watershed soil, a mass balance equation can be written. Contaminant concentration is expressed per unit mass of soil. The change in concentration per unit time is equal to the loading rate per unit soil mass minus the total loss rate:

$$\frac{dSc}{dt} = \frac{L_W \cdot A_s}{BD \cdot V_s} - ks \cdot Sc$$

$$= \frac{100 L_W}{BD \cdot Z} - ks \cdot Sc$$

This is a first order, ordinary differential equation with this general solution:

$$Sc = \frac{100 L_W}{ks \cdot BD \cdot Z} + C_X e^{-ks \cdot t}$$

where C_X is an unknown constant. Applying the condition Sc=0 at t=0 gives the following:

$$Sc = \frac{100 L_W}{ks \cdot BD \cdot Z} \left(1 - e^{-ks \cdot t}\right)$$

If a stable natural background concentration exists in the soil that is independent of the loading and loss processes, then this C_{sb} can be added to the above solution to obtain the overall concentration.

D.3.3.2 Terrestrial Equilibrium Relationships

Within the soil, the dissolved chemical concentration equilibrium with the sorbed concentration is defined by a partition coefficient:

$$C_{ps,i} = C_{ds,i} \cdot Kd_{s,i}$$

Assuming that the soil gas phase is negligible on a mass basis, the total soil chemical concentration is composed of dissolved chemical plus sorbed chemical:

$$C_{st,i} = C_{ds,i}\theta_s + C_{ps,i} \cdot BD \quad (104)$$

Substituting the partitioning relationship between dissolved and sorbed chemical, one gets the total soil chemical concentration:

$$C_{st,i} = C_{ds,i}\left(\theta_s + Kd_{s,i} \cdot BD\right)$$

The fraction of chemical dissolved in the soil water is the following:

$$f_{ds,i} = \frac{C_{ds,i} \cdot \theta_s}{C_{st,i}} = \frac{\theta_s}{\theta_s + Kd_{s,i} \cdot BD}$$

The fraction of chemical sorbed in the soil is $1\text{-}f_{ds,i}$:

$$f_{ps,i} = \frac{Kd_{s,i} \cdot BD}{\theta_s + Kd_{s,i} \cdot BD}$$

The concentration of chemical dissolved in the soil water is derived as follows:

$$C_{ds,i} = \frac{f_{ds,i} \cdot Sc_i \cdot BD}{\theta_s} = \frac{Sc_i \cdot BD}{\theta_s + Kd_{s,i} \cdot BD}$$

and the concentration of chemical sorbed to the soil is this:

$$C_{ps,i} = f_{ps,i} \cdot Sc_i = \frac{Sc_i \cdot Kd_{s,i} \cdot BD}{\theta_s + Kd_{s,i} \cdot BD}$$

D.3.3.3 Surface Water Mass Balance Relationships

Taking Figure D-3 as a control volume for the water body, at steady-state total chemical loading equals the sum of chemical outflow and chemical loss. The chemical outflow is the product of the volumetric flow rate and the water column concentration:

$$Outflow = Vf_x \cdot C_{wt}$$

The fraction of the total water body chemical concentration that is in the water column is defined in this way:

$$f_{water} = \frac{C_{wt} \cdot d_w / d_z}{C_{wtot}}$$

The outflow can, thus, be given in terms of the total water body concentration:

$$Outflow = Vf_x \cdot f_{water} \cdot C_{wtot} \cdot d_z / d_w$$

The chemical loss can also be given in terms of the total water body concentration:

$$Loss = k_{wt} \cdot C_{wtot} \cdot V_z$$

Equating loading to outflow plus loss, and solving for the total water body concentration gives this:

$$C_{wtot} = \frac{L_T}{Vf_x\, f_{water}\, d_s/d_w + k_{wt}\, V_z}$$

In a similar manner, the mass balance equation for sediment in the water body can be derived. Taking Figure D-9 as a control volume, the soil eroding into the water body, X_w, equals the sum of the amount depositing into the upper bed, X_d, and the advective loss from the water column, X_a. X_w is the product of the areal soil erosion flux, the watershed surface area, and the watershed sediment delivery ratio, with a factor converting kg to mg:

$$X_w = X_e \cdot WA_L \cdot SD \cdot 10^3$$

X_d is the product of the suspended solids concentration and the deposition rate:

$$X_d = W_{dep} \cdot WA_w \cdot TSS$$

X_a is the product of the suspended solids concentration and the volumetric flow rate:

$$X_a = Vf_x \cdot TSS$$

Substituting these relationships into the solids mass balance and solving for the suspended solids concentration in the water column gives this equation:

$$TSS = \frac{X_e\ WA_L\ SD\ 10^3}{Vf_x + W_{dep}\ WA_w}$$

D.3.3.4 Surface Water Equilibrium Relationships

Within the water column, the dissolved chemical concentration equilibrium with the sorbed concentration is defined by a partition coefficient:

$$C_{sw,i} = C_{dw,i} \cdot Kd_{sw,i}$$

The total chemical concentration in the water column is composed of dissolved chemical plus sorbed chemical:

$$C_{wt,i} = C_{dw,i} + C_{sw,i} \cdot TSS \cdot 10^{-6} \qquad (120)$$

Substituting the partitioning relationship between dissolved and sorbed chemical, the total chemical concentration is calculated as this:

$$C_{wt,i} = C_{dw,i} \left(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}\right)$$

The fraction of chemical dissolved in the water column becomes the following:

$$f_{dw,i} = \frac{C_{dw,i}}{C_{wt,i}} = \frac{1}{1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}}$$

Within the bed sediment, the dissolved chemical concentration equilibrium with the sorbed concentration is defined by a partition coefficient:

$$C_{db,i} = C_{sb,i} \cdot Kd_{bs,i}$$

The total benthic chemical concentration is composed of dissolved chemical plus sorbed chemical:

$$C_{bt,i} = C_{db,i}\theta_{bs} + C_{sb,i} \cdot BS$$

Substituting the partitioning relationship between dissolved and sorbed chemical, one calculates the total benthic chemical concentration:

$$C_{bt,i} = C_{db,i} \left(\theta_{bs} + Kd_{bs,i} \cdot BS\right)$$

The fraction of chemical dissolved in the benthic pore water is the following:

$$f_{db,i} = \frac{C_{db,i} \cdot \theta_{bs}}{C_{bt,i}} = \frac{\theta_{bs}}{\theta_{bs} + Kd_{bs,i} \cdot BS}$$

The total chemical concentration on a total water body basis is the sum of the water column and the benthic concentrations, weighted by the respective volumes is given by these relationships:

$$C_{wtot,i} = C_{wt,i} \cdot \frac{V_w}{V_z} + C_{bt,i} \cdot \frac{V_b}{V_z} = C_{wt,i} \cdot \frac{d_w}{d_z} + C_{bt,i} \cdot \frac{d_b}{d_z}$$

Substituting in the relationships for $C_{wt,i}$ and $C_{bt,i}$, and noting that at equilibrium $C_{dw,i} = C_{db,i}$, one obtains the total water body concentration as a function of the dissolved water column concentration:

$$C_{wtot,i} = C_{dw,i} \left[(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}) \cdot d_w/d_z + (\theta_{bs} + Kd_{bs,i} BS) \cdot d_b/d_z\right]$$

The fraction of the chemical that is in the water column is defined as this:

$$f_{water,i} = \frac{C_{wt,i} \cdot d_w/d_z}{C_{wtot,i}}$$

Substituting in the relationships for $C_{wt,i}$ and $C_{wtot,i}$, one obtains expressions for $f_{water,i}$ and $f_{benth,i}$ as functions of environmental and chemical properties:

$$f_{water,i} = \frac{(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}) \cdot d_w/d_z}{(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}) \cdot d_w/d_z + (\theta_{bs} + Kd_{bs,i} \cdot BS) \cdot d_b/d_z}$$

$$f_{benth,i} = \frac{(\theta_{bs} + Kd_{bs,i} \cdot BS) \cdot d_b / d_z}{(1 + Kd_{sw,i} \cdot TSS \cdot 10^{-6}) \cdot d_w / d_z + (\theta_{bs} + Kd_{bs,i} \cdot BS) \cdot d_b / d_z}$$

D.3.4 Summary of Notation

A_s = surface area of watershed soil element (m^2)

BD = representative watershed soil bulk density (g/cm^3)
BS = benthic solids concentration (kg/L)

$C_{atm,i}$ = component "i" vapor phase air concentration over watershed ($\mu g/m^3$)
$C_{bt,i}$ = total benthic component "i" concentration (mg/L)
C_d = drag coefficient (= 0.0011)
$C_{ds,i}$ = concentration of "i" dissolved in pore water (mg/L)
$C_{ps,i}$ = concentration of "i" in particulate phase (mg/kg)
C_{sb} = background "natural" soil concentration (µg pollutant/g soil)
$C_{st,i}$ = total soil concentration of component "i" (mg/L)
C_{wt} = water column concentration of total mercury (µg/L)
$C_{wt,Hg0}$ = water column concentration of elemental mercury (µg/L)
$C_{wt,Hg(II)}$ = water column concentration of divalent mercury (µg/L)
$C_{wt,MeHg}$ = water column concentration of methylmercury (µg/L)
$C_{wtot,i}$ = total water body component "i" concentration, including water column and benthic sediment (mg/L)
C_{wtot} = total water body concentration, including water column and benthic sediment (mg/L)

d_b = depth of the upper benthic sediment layer (m)
d_w = depth of the water column (m)
d_z = total depth of water body, d_w+d_b (m)
D = depth of water body (m)
$D_{a,i}$ = diffusivity of component "i" in air (cm^2/sec)
$D_{w,i}$ = diffusivity of component "i" in water (cm^2/sec)
$Dyds_i$ = yearly dry deposition rate of component "i" onto surface water body (g pollutant/m^2-yr)
$Dydw_i$ = yearly average dry depositional flux of component "i" onto watershed (g/m^2-yr)
$Dyws_i$ = yearly wet deposition rate of component "i" onto surface water body (g pollutant/m^2-yr)
$Dyww_i$ = yearly average wet depositional flux of component "i" onto watershed (g/m^2-yr)

ER = soil enrichment ratio (unitless)
Ev = average annual evapotranspiration (cm/yr)

$f_{benth,i}$ = fraction of total water body component "i" concentration that occurs in the benthic sediment
$f_{db,i}$ = fraction of bed sediment component "i" concentration that is dissolved
$f_{ds,i}$ = fraction of soil component "i" concentration that is dissolved
$f_{dw,i}$ = fraction of water column component "i" concentration that is dissolved
$f_{ps,i}$ = fraction of soil component "i" concentration that is sorbed

f_{s1} = fraction of soil concentration that is component "i" (i.e., elemental, divalent, and methylmercury)

f_{s1} = fraction of soil concentration that is elemental mercury

f_{s2} = fraction of soil concentration that is divalent mercury

f_{s3} = fraction of soil concentration that is methylmercury

$f_{water,i}$ = fraction of total water body component "i" concentration that occurs in the water column

f_{w1} = fraction of water column concentration that is elemental mercury

f_{w2} = fraction of water column concentration that is divalent mercury

f_{w3} = fraction of water column concentration that is methylmercury

H_i = Henry's law constant for component "i" (atm-m^3/mole)

I = average annual irrigation (cm/yr)

$k_{b,i}$ = benthic burial rate constant for component "i" (yr^{-1})

$k_{gb,i}$ = benthic degradation or transformation rate constant for component "i" (yr^{-1})

$k_{gw,i}$ = water column degradation or transformation rate constant for component "i" (yr^{-1})

$k_{v,i}$ = water column volatilization loss rate constant for component "i" (yr^{-1})

k_{wt} = total first order dissipation rate constant, including water column and benthic degradation, volatilization, and burial (yr^{-1})

$k_{wt,i}$ = overall total water body dissipation rate constant for component "i" (yr^{-1})

ks = total chemical loss rate constant from soil (yr^{-1})

ks_i = soil loss constant for component "i" due to all processes (yr^{-1})

ksl_i = soil loss constant due to leaching (yr^{-1})

kse_i = soil loss constant due to erosion (yr^{-1})

ksr_i = soil loss constant due to runoff (yr^{-1})

ksg_i = soil loss constant due to chemical transformation/ degradation (yr^{-1})

ksv_1 = soil loss constant due to volatilization (yr^{-1})

$K_{G,i}$ = gas phase transfer coefficient (m/year)

$K_{L,i}$ = liquid phase transfer coefficient for component "i" (m/year)

$K_{t,i}$ = gas phase mass transfer coefficient for component "i" (cm/s; see Eq [4-6], IED)

$K_{v,i}$ = overall transfer rate, or conductivity for component "i" (m/yr)

$Kd_{bs,i}$ = bottom sediment/pore water partition coefficient for component "i" (L/kg)

$Kd_{s,i}$ = soil-water partition coefficient for component "i" (L/kg or cm^3/g)

$Kd_{sw,i}$ = suspended sediment/surface water partition coefficient for component "i" (L/kg)

Ke_i = equilibrium coefficient for component "i" (s/cm/yr); see Eq [4-5], IED)

Kt_i = gas phase mass transfer coefficient for component "i" (cm/s; see Eq [4-6], IED)

$L_{Dep,i}$ = deposition of particle bound component "i" (g/yr)

$L_{DIF,i}$ = atmospheric diffusion flux of component "i" to soil (g/m^2-yr)

$L_{Dif,i}$ = diffusion of vapor phase component "i" (g/yr)

$L_{E,i}$ = soil erosion load for component "i" (g/yr)

$L_{I,i}$ = internal transformation load, equal to 0 for equilibrium mercury chemistry (g/yr)

$L_{Is,i}$ = internal transformation load of component "i" per areal basis (g/m^2-yr)

$L_{R,i}$ = pervious surface runoff load for component "i" (g/yr)
$L_{RI,i}$ = runoff load from impervious surfaces (g/yr)
L_T = total chemical load into water body, including deposition, runoff, erosion, atmospheric diffusion, and internal chemical transformation (g/yr)
$L_{T,i}$ = total component "i" load to the water body (g/yr)
L_W = yearly average load of pollutant to watershed on an areal basis(g pollutant/m^2-yr)

MW_i = molecular weight of component "i"

P = average annual precipitation (cm/yr)

R = universal gas constant (8.206×10^{-5} atm-m^3/mole-°K)
$R_{G,i}$ = gas phase resistance (year/m)
$R_{L,i}$ = liquid phase resistance (year/m)
Ro = average annual runoff (cm/yr)

Sc = average watershed soil concentration after time period of deposition (µg pollutant/g soil)
$Sc_{a,i}$ = air Schmidt number for component "i" (dimensionless)
Sc_i = total component "i" concentration in watershed soils (µg/g)
Sc_{Hg0} = soil concentration of elemental mercury (µg pollutant/g soil)
$Sc_{Hg(II)}$ = soil concentration of divalent mercury (µg pollutant/g soil)
Sc_{MeHg} = soil concentration of methylmercury (µg pollutant /g soil)
$Sc_{w,i}$ = water Schmidt number for component "i" (dimensionless)
SD = watershed sediment delivery ratio (unitless)

T_a = air temperature (°C)
T_k = water body temperature (°K)
T_w = water temperature (°C)
Tc = total time period over which deposition has occurred (yr)
TSS = suspended solids concentration (mg/L)

u = current velocity (m/s)
u^* = shear velocity (m/s)

V_b = volume of upper benthic sediment layer (m^3)
V_s = volume of watershed soil layer (m^3)
V_w = volume of water column (m^3)
V_z = total volume of water body or water body segment being considered, including water column and benthic sediment (m^3)
Vf_x = average volumetric flow rate through water body (m^3/yr)

W = wind velocity, 10 m above water surface (m/s)
W_b = benthic burial rate (m/yr)
W_{dep} = suspended solids deposition rate (m/yr)
WA_I = impervious watershed area receiving pollutant deposition (m^2)
WA_L = total watershed area receiving pollutant deposition (m^2)
WA_w = water body surface area (m^2)

X_e = unit soil erosion flux, calculated in the soils section from the Universal Soil Loss Equation (USLE); see Eq [9-3], IED (kg/m^2-yr)

Z	=	representative watershed mixing depth to which deposited pollutant is incorporated (cm)
k	=	von Karman's constant (= 0.4)
λ_2	=	dimensionless viscous sublayer thickness (= 4)
μ_a	=	viscosity of air corresponding to the air temperature (g/cm-s)
μ_w	=	viscosity of water corresponding to the water temperature (g/cm-s)
ρ_a	=	density of air corresponding to the water temperature (g/cm^3)
ρ_s	=	solids density, 2.65 kg/L
ρ_w	=	density of water corresponding to the water temperature (g/cm^3)
ν_a	=	dynamic viscosity of air (cm^2/sec)
θ	=	temperature correction factor, set to 1.026.
θ_s	=	volumetric soil water content (L_{water}/L)
θ_{bs}	=	bed sediment porosity (L_{water}/L)

D.4 References

Ambrose, R.B. et al. 1988. WASP4, A Hydrodynamic and Water Quality Model--Model Theory, User's Manual, and Programmer's Guide. U.S. Environmental Protection Agency, Athens, GA. EPA/600/3-87-039.

Bhumralkar, C. M., R. L. Mancuso, D. E. Wolf, R. H. Thuillier, K. D. Nitz, and W. B. Johnson, 1980. Adaptation and application of a long-term air pollution model ENAMAP-1 to Eastern North America. Final Report, EPA-600/4-80/039, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Bowers, J.R., J.R. Bjorkland and C.S. Cheney, 1979: Industrial Source Complex (ISC) Dispersion Model User's Guide. Volume I, EPA-450/4-79-030, U.S. EPA, Research Triangle Park, North Carolina 27711.

Briggs, G.A. (1969). *Plume Rise*, AEC Critical Review Series, TID - 25075, National Technical Information Service, Springfield, VA., 81 pp.

Briggs, G.A. (1970). Some recent analyses of plume rise observation, Paper presented at the Second International Clean Air Congress of the International Union of Air Pollution Prevention Associations, Washington, D.C., December 6-11, 1970.

Briggs, G.A. (1972). Discussion on Chimney Plumes in Neutral and Stable Surroundings. *Atmos. Environ.* 6:507-510.

Briggs, G.A. (1973). *Diffusion Estimation for Small Emissions,* Atmospheric Turbulance and Diffusion Laboratory, Contribution File No. 70 (Draft), Oak Ridge, Tennessee.

Briggs, G.A. (1973). *Diffusion Estimation for Small Emissions,* Atmospheric Turbulance and Diffusion Laboratory, Contribution File No. 79, Oak Ridge, Tennessee.

Briggs, G.A. 1974. Diffusion Estimation for Small Emissions. In ERL, ARL USAEC Report ATDL-106. U.S. Atomic Energy Commission, Oak Ridge, Tennessee.

Briggs, G.A. (1975). Plume Rise predications, in *Lectures on Air Pollution and Environmental Impact Analysis,* American Meteorological Society, Boston, Massachusetts.

Burns, L.A., D.M. Cline, and R.R. Lassiter. 1982. Exposure Analysis Modeling System (EXAMS): User Manual and System Documentation. U.S. Environmental Protection Agency, Athens, GA. EPA-600/3-82-023.

Burke, J., M. Hoyer, G. Keeler and T. Scherbatskoy (1995). Wet deposition of mercury and ambient mercury concentrations at a site in the Lake Champlain basin. Accepted for publication in *Water, Air and Soil Pollution* volume for "Mercury as a Global Pollutant" Conference Proceedings.

Brosset, C. and E. Lord, 1991. mercury in precipitation and ambient air. A new scenario. *Water, Air and Soil Pollution* 56:493-506.

CARB (1986). Subroutines for Calculating Dry Deposition Velocities Using Sehmel's Curves. Prepared by Bart Croes, California Air Resources Board.

CARB, 1987. Deposition rate calculations for air toxics source assessments. Report dated September 16, 1987, by California Air Resources Board, 6 pp.

Catalano, J.A., D.B. Turner, and J.H. Novak. 1987. *User's Guide for RAM -- Second Edition.* EPA/600/8-87/046, U.S. EPA, Research Triangle Park, North Carolina 27711.

Clark, T. L., P. Blakely, and G. Mapp, 1992. Model calculations of the annual atmospheric deposition of toxic metals to Lake Michigan. 85th Annual Meeting of the Air and Water Management Assoc., Kansas City, MO, June 23-37.

Cramer, H.E. 1957. A Practical Method for Estimating the Dispersal of Atmospheric Contaminants, in *Proceedings of the First National Conference on Applied Meteorology,* Sec. C, pp. C-33-C-35, American Meteorological Society, Hartford, Conn.

Dvonch, J. T., A. F. Vette, G. J. Keeler, G. Evans and R. Stevens (1995). An intensive multi-site pilot study investigating atmospheric mercury in Broward County, Florida. Accepted for publication in *Water, Air and Soil Pollution* volume for "Mercury as a Global Pollutant" Conference Proceedings.

Eder, B. K., D. H. Coventry, T. L. Clark, and C. E. Bollinger, 1986. RELMAP: A regional Lagrangian model of air pollution - users guide. Project Report, EPA/600/8-86/013, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Egan, B.A. (1975). Turbulent Diffusion in Complex Terrain, in *Lectures on Air Pollution and Environmental Impacts Analysis,* pp. 112-135, D.Haugen (Ed.), American Meteorological Society, Boston, Mass.

Engelmann, R.J. (1968). The Calculation of Precipitation Scavenging, in *Meteorology and Atomic Energy 1968,* D.H. Slade, editir. U.S. Atomic Energy Commission.

Fay, J.A., M. Escudier, and D.P. Hoult (1969). A Correlation of Field Observations of Plume Rise, Fluid Mechanics Laboratory Publication No. 69-4, Massachusetts Institute of Technology.

Geraghty, J.J., D.W. Miller, F.V. Der Leenden, and F.L. Troise, *Water Atlas of the United States,* A Water Information Ceter Publication, Port Washington, N.Y. (1973).

Gifford. F.A. 1961. Use of routine meteorological observations for estimating atmospheric dispersion. Nuclear Safety, 2, No. 4, 47-51.

Gifford, 1967.

Gifford, F.A. (1976). Turbulent Diffusion Typing Schemes - A Review, *Nucl.Saf.*, 17:68-86.

Hanna, S.R., G.A. Briggs, and R.P. Hosker, Jr. (1982). Handbook on Atmospheric Diffusion, DOE/TIC-11223.

Hanson, P. J., S. E. Lindberg, K. H. Kim, J. G. Owens, and T. A. Tabberer, 1994. Air/surface exchange of mercury vapor in the forest canopy: I. Laboratory studies of foliar mercury vapor exchange. International Conference on Mercury as a Global Pollutant, July 10-14, Whistler, British Columbia, Canada.

Holzworth,G.C. (1972). Mixing Heights, Wind Speeds and Potential for Urban Air Pollution Throughout the Contiguous United States. Publication No.AP-101, U.S. EPA, Research Triangle Park, North Carolina 27711.

Hoyer, M., J. Burke and G. Keeler (1995). Atmospheric sources, transport and deposition on mercury in Michigan: Two years of event precipitation. Accepted for publication in *Water, Air and Soil Pollution* volume for "Mercury as a Global Pollutant" Conference Proceedings.

Hubert, A.H. and W.H. Snyder (1976). Building Waste Effects on Short Stack Effluents, in *Third Symposium on Atmospheric Turbulence, Diffusion, and Air Quality,* Raliegh, NC, Oct. 19-22, pp. 235-242, American Meteorological Society, Bostan, Mass.

Huber, A.H. (1977). Incorporating Building/Terrain Wake Effects on Stack Effluents, in *Preprints of Joint Conference on Applications of Air Pollution Meteorology,* Salt Lake City, Nov. 29-Dec.2, pp. 353-356, American Meteorological Society, Boston, Mass.

Irwin, J. 1979. Estimating Plume Dispersion: A Recommended Generalized Scheme, in *Proceedings of the Fourth Symposium on Turbulence, Diffusion, and Air Pollution,* Jan. 15-18, 1979, Reno, Nev., pp. 62-69, American Meteorological Society, Boston, Mass.

Iverfeldt, Å., 1991. Occupance and turnover of atmospheric mercury over the nordic countries. *Water, Air and Soil Pollution* 56:251-265.

Iverfeldt, Å., and O. Lindqvist, 1986. Atmospheric oxidation of elemental mercury by ozone in the aqueous phase. *Atmospheric Environment* 20:1567-1573.

Johnson, W. B., 1983. Interregional exchanges of air pollution: Model types and applications. *Journal of the Air Pollution Control Association* 33: 563-574.

Johnson, W. B., D. E. Wolf, and R. L. Mancuso, 1978. Long-term regional patterns and transfrontier exchanges of airborne sulfur pollution in Europe. *Atmospheric Environment* 12: 511-527.

Keeler, G. J., S. M. Japar, W. W. Brachaczek, R. A. Gorse, Jr., J. M. Norbeck and W. R. Pierson, 1990. The sources of aerosol elemental carbon at Allegheny Mountain. *Atmospheric Environment* 24A:2795-2805.

Keeler, G., G. Glinsorn and N. Pirrone (1995). Particulate mercury in the atmosphere: Its significance, transport, transformation and sources. Accepted for publication in *Water, Air and Soil Pollution* volume for "Mercury as a Global Pollutant" Conference Proceedings.

Lamborg, C. H., W. F. Fitzgerald, G. M. Vandal and K. R. Rolfhus (1995). Atmospheric mercury in northern Wisconsin: Sources and species. Accepted for publication in *Water, Air and Soil Pollution* volume for "Mercury as a Global Pollutant" Conference Proceedings.

Lindberg, S. E., R. R. Turner, T. P. Meyers, G. E. Taylor, Jr. and W. H. Schroeder, 1991. Atmospheric concentrations and deposition of mercury to a deciduous forest at Walker Branch Watershed, Tennessee, USA. *Water, Air and Soil Pollution* 56:577-594.

Lindberg, S. E., T. P. Meyers, G. E. Taylor, Jr., R. R. Turner and W. H. Schroeder, 1992. Atmosphere-surface exchange of mercury in a forest: Results of modeling and gradient approaches. *Journal of Geophysical Research* 97:2519-2528.

Lindqvist, 0., K. Johansson, M. Aastrup, A. Andersson, L. Briŋgmark, G. Hovsenius, L. Håkanson, Å. Iverfeldt, M. Meili, and B. Timm, 1991. Mercury in the Swedish environment. Transformation and deposition processes. *Water, Air and Soil Pollution* 55:49-63.

Munthe, J., Z. F. Xiao and O. Lindqvist, 1991. The aqueous reduction of divalent mercury by sulfite. *Water, Air and Soil Pollution,* 56:621-630.

Munthe, J. and W. McElroy, 1992. Some aqueous reactions of potential importance in the atmospheric chemistry of mercury. *Atmospheric Environment* 26A: 553-557.

Munthe, J., 1992. The aqueous oxidation of elemental mercury by ozone. *Atmospheric Environment* 26A: 1461-1468.

O'Connor, D.J. and W.E. Dobbins. 1958. Mechanism of Reaeration in Natural Streams, ASCE Transactions, pp. 641-684, Paper No. 2934.

O'Connor, D.J. 1983. Wind Effects on Gas-Liquid Transfer Coefficients. Journal of Environmental Engineering, Volume 109, Number 9, pp. 731-752.

Overcamp, T.J. (1977). *Modelling of Air Quality for Industrial Pollution Control,* Appendix from a Continuing Education Short course taught at Clemson University on December 7, 1977.

Pasquill, F. (1961). The Estimation of Dispersion of Windborne Material, *Meteorol.Mag.,* 90:33-49.

Pasquill, F. (1974). *Atmospheric Diffusion, the Dispersion of Windborne Material from Industrial and other Sources,* Ellis Horwood, New York.

Pasquill, F. (1976). *Atmospheric Dispersion Parameters in Gaussian Plume Modeling: Part II. Possible Requirements for Change in Turner Woeknook Values,* Report EPA-600/4-760306, U.S. EPA.

PEI Associates, Inc, and H.E. Cramer Company, Inc. (1986). Air quality modeling analysis of municipal waste combustors. Prepared for Monitoring and Data Analysis Division, Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina 27711.

Penner, J. E., H. Eddleman and T. Novakov, 1993. Towards the development of a global inventory for black carbon emissions. *Atmospheric Environment* 27A:1277-1295.

Petersen, G., Å. Iverfeldt and J. Munthe, 1995. Atmospheric mercury species over Central and Northern Europe. Model calculations and comparison with observations from the Nordic Air and Precipitation Network for 1987 and 1988. *Atmospheric Environment* 29:47-68.

Pierce, T.E. and D.B. Turner (1980). User's Guide for MPTER: A multiple point Gaussian dispersion algorithm with optional terrain adjustment. EPA-600/8-80-016. U.S. E.P.A., Research Triangle Park, N.C.

Randerson,D. (1984). D. Randerson, editor, *Atmospheric Science and Power Production,* DOE/TIC-27601.

Rao, K.S. Analytical solutions of a gradient-transfer model for plume deposition and sedimentation. NOAA-TM-ERL ARL-109. U.S. National Oceanic and Atmospheric Administration, Silver Spring, MD.

Rao, K.S. and L. Satterfield (1980). A study of the probable environmental impact of fugitive coal dust emissions at the Ravenswood Power Plant, New York. ATDL Contribution 80/26, NOAA, Oak Ridge, TN.

Sanemasa, I., 1975. The solubility of elemental mercury vapor in water. *Bulletin of the Chemical Society of Japan,* 48:1795-1798.

Schroeder, W. H. and R. A. Jackson, 1987. Environmental measurements with an atmospheric mercury monitor having speciation capabilities. *Chemosphere* 16:183-199.

Sehmel,G.A. (1984). Deposition and Resuspension, in *Atmospheric Science and Power Production,* D. Randerson (Ed.), DOE/TIC-27601.

Seinfeld, J. H., 1986. *Atmospheric Chemistry and Physics of Air Pollution*, John Wiley and Sons, New York, p. 198-200.

Sherlock,R.H. and E.A. Stalker (1941). A Study of Flow Phenomena in the Wake of Smoke Stack, *Engineering Research Bulletin* 29, University of Michigan, Ann Arbor.

Smith, M.E. (1951). The Forecasting of Micrometerological Variables, *Meteorol. Monogr.,* 4:50-55.

Slinn, W.G.N. (1984). Precipitation Scavenging, in Atmospheric Science and Power Production, D. Randerson, ed. DOE/TIC-27601.

Sorenson, J. A., G. E. Glass and K. W. Schmidt (1994). Regional patterns of wet mercury deposition. *Environ. Sci. Technol.,* 28, 2025-2032.

Travis, C.C., C.F. Baes, III, L.W. Barnthouse, et al. 1983. *Exposure Assessment Methodology and Reference Environments for Synfuel Risk Analysis.* Oak Ridge National Laboratory. ORNL/TM-8672. Prepared for U.S. Environmental Protection Agency, Office of Research and Development.

Turner, D.B. (1967). *Workbook of Atmospheric Dispersion Estimates*, Public Health Service 999-AP-26, Robert A. Taft Sanitary Engineering Center, Cincinnati, Ohio.

Turner, D.B. (1970). *Workbook of Atmospheric Dispersion Estimates.* Public Health Service Publication No. 999-AP-26, U.S. E.P.A., Research Triangle Park, N.C.

U.S. EPA. 1990. *Methodology for Assessing Health Risks Associated with Indirect Exposure to Combustor Emissions.* Interim Final. Office of Health and Environmental Assessment. Cincinnati, Ohio. EPA/600/6-90/003.

U.S. EPA (1992). *User's Guide for the Industrial Source Complex (ISC2) Dispersion Models, Volume II - Description of Model Algorithms,* EPA-450/4-92-008b, Research Triangle Park, North Carolina 27711.

Walcek, C. J., R. A. Brost, J. S. Chang and M. L. Wesely, 1985. SO_2, sulfate and HNO_3 deposition velocities computed using regional land use and meteorological data. *Atmospheric Environment* 20:949-964.

Wark, K. and C.F. Warner (1981). *Air Pollution Its Origin and Control,* Harper Collins.

Wells, A.E. (1917). Results of Recent Investigations of the Smelter Smoke Problem, *Ind. Eng. Chem,* 9:640-646.

Whitman, R.G. 1923. A Preliminary Experimental Confirmation of the Two-Film Theory of Gas Absorption. Chem. Metallurg. Eng. 29:146-148.

Wischmeier, W.H. and D.D. Smith. 1978. *Predicting Rainfall Erosion Losses -- A Guide to Conservation Planning.* USDA Handbook No. 537.

Wesely, M. L., 1986. On the parameterization of dry deposition of acidifying substances for regional models. Internal Report (Nov. 1986): Interagency Agreement DW89930060-01 to the U.S. Department of Energy, U.S. Environmental Protection Agency, Atmospheric Sciences Research Laboratory, Research Triangle Park, NC, 23 pp.

Yarwood, G. and H. Niki, 1990. A critical review of available information on transformation pathways for mercury species in the atmospheric environment. Prepared for Atmospheric Environment Service, Environment Canada, 4905 Dufferin Street, Downsview, Ontario M3H 5T4 and Supply and Services Canada, Science Procurement Contract No. KM171-9-0129, Scientific Liason Officer: Dr. W. H. Schroeder.

Zeller, K.F. (1984). The Environmental Impact Statement, in *Atmospheric Science and Power Production,* D. Randerson (Ed.), DOE/TIC-27601.

APPENDIX E

CHEMICAL PROPERTIES OF MERCURY

TABLE OF CONTENTS

		Page
E.1	Chemical Properties of Mercury	E-1
E.2	Analytic Measurement Methods	E-3
	E.2.1 Methods of N. S. Bloom and W. F. Fitzgerald	E-5
	E.2.2 Swedish Sampling Program	E-6
	E.2.3 Other Methods	E-6
	E.2.4 Detection Limits	E-7
E.3	References	E-7

E.1 Chemical Properties of Mercury

Elemental mercury metal is a heavy, silvery-white liquid at typical ambient temperatures and pressures. The vapor pressure of mercury metal is strongly dependent upon temperature, and it vaporizes readily under ambient conditions; its saturation vapor pressure of 14 mg/m^3 greatly exceeds the average permissible concentrations for occupational (0.05 mg/m^3) or continuous environmental (0.015 mg/m^3) exposure (Nriagu, 1979; WHO, 1976). Elemental mercury partitions strongly to air in the environment and is not found as the pure, confined liquid. Most of the mercury encountered in the atmosphere is elemental mercury vapor.

Mercury can exist in three oxidation states, Hg^0 (metallic), Hg_2^{2+}, (mercurous) and Hg^{2+} (mercuric). The properties and behavior of mercury strongly depend on the oxidation state. Mercurous and mercuric mercury can form numerous inorganic and organic chemical compounds; however, mercurous mercury is rarely stable under ordinary environmental conditions. Mercury is unusual among metals because it tends to form covalent rather than ionic bonds. Most of the mercury encountered in water/soil/sediments/biota (all environmental media except the atmosphere) is in the form of inorganic mercuric salts and organomercurics. Organomercurics are defined by the presence of a covalent C-Hg bond. This is thought to differ from the common behavior of inorganic mercury compounds associating with organic material in the environment. The compounds most likely to be found under environmental conditions are these: the mercuric salts $HgCl_2$, $Hg(OH)_2$ and HgS; the methylmercury compounds CH_3HgCl and CH_3HgOH; and, in small fractions, other organomercurics (i.e., dimethylmercury and phenylmercury).

In the aqueous phase, mercury compounds often remain as undisassociated molecules, and the solubility values reported below reflect this. Solubility values for mercury compounds which do not disassociate are not based on the ionic product. Most organomercurics are not soluble and do not react with weak acids or bases, due to the low affinity of the mercury for oxygen bonded to carbon. CH_3HgOH, however, is highly soluble due to the strong hydrogen bonding capability of the hydroxide group. The mercuric salts vary widely in solubility. For example $HgCl_2$ is readily soluble in water, and HgS is as unreactive as the organomercurics, due to the high affinity of mercury for sulfur. A detailed discussion of mercury chemistry can be found in Nriagu (1979) and Fitzgerald (1994).

Table E-1
Chemical Properties of Elemental Mercury

Property	Value	Units	References
Chemical Name	Elemental Mercury		
Chemical formula	Hg^0		
CAS Registry Number	7439-97-6		
Molecular Weight	200.59	g/mole	
Melting Point (at 760 torr)	-39	^0C	MMES (1994)
Boiling Point (at 760 torr)	357	^0C	MMES (1994)
Density as a Dry Gas at 25°C	7.0	Density of Air	MMES (1994)
Specific Gravity at 25°C	13.59	g/cm^3	MMES (1994)
Vapor Pressure at 25°C	2.0×10^{-3}	mm mercury	MMES (1994)

Table E-1 (continued)

Property	Value	Units	References
Physical Form at 25°C	Silver-white liquid		
Solubility in Water at 25°C	6×10^{-5}	g/L	Schuster (1991)
Henry's Law Constant at 25°C	7.1×10^{-3}	atm-m^3/mole	Lindqvist and Rodhe (1985)

Table E-2
Chemical Properties of Mercuric Mercury

Property	Value	Value	Value	Units	References
Chemical Name/Form	Mercuric Sulphide (Cinnabar)	Mercuric Chloride	Mercuric Hydroxide		
Chemical formula	HgS	$HgCl_2$	$Hg(OH)_2$		
CAS Registry Number	1344-48-5	7487-94-7			
Molecular Weight	232.68	271.52	234.60	g/mole	
Melting Point (at 760 torr)	583.5[a]	277		^{0}C	MMES, MMES (1994)
Boiling Point (at 760 torr)	NA	302		^{0}C	MMES
Density as a Dry Gas at 25°C	8.0	9.44		Density of Air	MMES, Gmelins, 1967
Specific Gravity at 25°C	Red: 8.10 Black: 7.73	5.44		g/cm^3	MMES (1994)
Vapor Pressure at 25°C		1.2×10^{-4}		mm mercury	U.S. EPA, 1993
Physical Form at 25°C	Red or Black powder	White crystal/ powder	White powder		
Solubility in Water at 20°C	2 X 10-9	69	0.126	g/L	Schuster, 1991 U.S EPA, 1993 Schuster, 1991
Henry's Law Constant at 25°C		7.1×10^{-10}	7.8×10^{-8}	atm-m^3/mole	Lindqvist and Rodhe, 1985 Lindqvist and Rodhe, 1985
Log of Octanol:Water coefficient		-0.215			Halbach, 1985

Table E-3
Chemical Properties of Methylmercury

Property	Value	Value	Units	References
Chemical Name	Methylmercury Chloride	Methylmercury hydroxide		
Chemical formula	CH_3HgCl	CH_3HgOH		
CAS Registry Number	115-09-3			
Molecular Weight	251.08	232.64	g/mole	
Melting Point (at 760 torr)	170			Chapman and Hall (1984)
Vapor Pressure at 25°C	8.5×10^{-3}		mm mercury	WHO (1990)
Physical Form at 25°C	White powder		^0C	
Solubility in Water at 20°C	Hg^0 < Sol < $HgCl_2$ (WHO, 1990)	>232 (due to existence of 1M stock solution, but at higher pH)	^0C	WHO (1990),Iverfeldt and Persson (1985)
Henry's Law Constant at 25°C	4.7×10^{-7}	6.1×10^{-9}		Iverfeldt and Persson (1985),Lindqvist and Rodhe (1985)
Log of Octanol:Water coefficient	0.405		Density of Air	Halbach (1985)

E.2 Analytic Measurement Methods

Mercury contamination of samples has been shown to be a serious problem in past studies. The development of ultra-clean techniques has been critical for the more precise measurements required for detection of low levels of mercury.A number of methods can be employed to determine mercury concentrations in environmental media. Data on the concentrations of total mercury, elemental mercury, organic mercury compounds (especially methylmercury) and information on various Hg(II) compounds can be collected, although exact speciation among Hg(II) compounds is not usually attempted. Methods that accurately and reliably measure the total mercury concentration in environmental media have been established for some time. Recently significant improvements/standardizations in analytical methodologies enable reliable data on the concentration of methylmercury, elemental mercury and the Hg(II) fraction to be readily separated from the total mercury in environmental media. It is possible further to speciate the Hg(II) fraction into reactive, non-reactive and particle-bound components. It is generally not possible to determine which Hg(II)species is present (e.g., HgS or $HgCl_2$). For example, it has been suggested that much of the mercury in sediments from East Fork Poplar Creek in Oak Ridge, Tennessee is HgS (Revis et al., 1990).

Summarized below are the analytical methods used to determine the speciation of mercury outlined in the above table, principally methods employed by N.S. Bloom and W.F. Fitzgerald in the U.S. and methods used in the Swedish sampling program. This information is included in this report to support the assertions made in Table E-4 and to illustrate the difficulties and limitations of species-selective mercury determinations.

Table E-4

Comparison of U.S. and Swedish Methods for Obtaining Concentrations of Different Species/ Types of Mercury from Environmental Samples (an X indicates this reaction of mercury is often obtained in the recent literature)

Fractions of total mercury that are determined in various media	U.S.	Sweden
Air Samples		
Total gaseous Hg^0	X	X
methylmercury	X	
dimethylmercury	X	
particle bound mercury	X	X
Water/sediment/biota/precipitation		
Total mercury	X	X
methylmercury	X	
Hg^0	X	
dimethylmercury	X	
"reactive" or "labile" mercury (Hg_R) [a]	X	X
Total Hg(II), reactive + non-reactive		X
Hg(II) non-reactive [b]		X
Hg(II) inert [c]		X

[a] In the U.S. this is considered mainly if not exclusively Hg(II) species, and can more specifically be thought of as environmentally active Hg(II) (Fitzgerald et al., 1991). It includes inorganic Hg(II) (Hg^{2+}, $Hg(OH)_2$, $HgCl_2$...), simple organic Hg(II) associations, and labile (acid reactive) Hg(II) particulate associations. It is suggested that the determination of the amount of Hg_R is particularly important, since it represents the fraction of mercury which is susceptible to various biological and chemical reactions, including reduction to Hg^0 and production of methylmercury. The source of Hg_R in rain (at least in rain distant from anthropogenic emissions) is most probably due to atmospheric or gas/solution oxidation process which depends on the stable concentration of Hg^0 in the atmosphere (Fitzgerald et al., 1991; Lindqvist et al., 1991, chapter 6; Petersen et al., 1995).

The fraction of mercury not accounted for in U.S. speciation (that is not methylated mercury, Hg^0, or Hg_R) is mercury that is bound to particulates, some of which will be bound strongly to organic substances. This fraction is also made up of Hg(II) but is far less reactive that Hg_R. In Sweden this fraction is considered to be mainly inorganic Hg(II) compounds, such as Hg^{2+}, HgX_2, HgX_3-, HgX_4--, with X as Cl-, OH-, etc; also HgO on aerosol particles, and Hg^{2+} complexes with organic acids. It is denoted as Hg-IIa.

[b] This fraction, denoted Hg-IIb, includes methylmercury and other organomercuric compounds, $Hg(CN)_2$ and HgS, and Hg^{2+} bound to humic mater.

[c] Denoted Hg-IIc, this fraction is mainly mercury strongly bound to particulates.

These are some of the latest, highest yield methods used to determine the amounts and speciation of mercury in the environment. These methods show how much mercury information can be determined from environmental media with the latest methods that have been proven over several experiments and are commonly available to researchers: experimental or non-established analytical routines are not included. See Mitra (1986) for a review of methods for total mercury analysis including a number of out-dated routines.

E.2.1 Methods of N. S. Bloom and W. F. Fitzgerald

Water, Sediment or Soil, Seston and Biota (Bloom and Watras, 1989). A water sample is first buffered to pH 4.9, and then ethylated with sodium tetraethylborate to convert the mercury species into compounds which can be readily volatilized. Sediment, seston and biological samples are first digested with 25% KOH in methanol and then diluted and buffered to a pH range of 4 to 6 prior to ethylation. The ethylation results in the formation of $CH_3CH_2HgCH_3$ from reactive monomethylmercury and $(CH_3CH_2)_2Hg$ from reactive inorganic mercury. $(CH_3)_2Hg$ and Hg^0 contained in the sample are unaffected and may be purged and analyzed with the diakyl reaction products (in practice they are purged from unethylated samples for better precision). The volatile ethyl analogs are then purged onto a graphitic carbon column (Carbotrap^R). The carbon column is purged with N_2 to remove all water, and the organomercurials on the carbon trap are thermally desorbed into a gas chromatograph. The mercury species elute at different times from the column, where the gas is pyrolized at 750^{0}C to release the mercury as Hg^0. The amount of each eluted species is then determined by cold vapor atomic fluorescence spectroscopy.

To determine the total amount of methylmercury in a sample, it is necessary to account for organocomplexed and colloidal methylmercury, not just reactive CH_3Hg. A water sample treated with acidic saturated KCl and extracted with CH_2Cl_2 will convert these unreactive methylmercury species to reactive CH_3HgCl, which is then back-extracted into high purity water by solvent evaporation. The aqueous phase is then analyzed for methylmercury as above, starting with the ethylation step. Recently (Hovart et al., 1993b), it was found that distilling the water sample after pretreatment with KCl (to bring the Cl- concentration to 0.08%) and acidification H_2SO_4 at a rate of 6-8 ml/hr resulted in a sample (when ethylated as above) that resulted in higher and more consistent recoveries, lower detection limits and greater ease of use (no organic solvents needed). Distillation provided similar benefits for analyzing sediment samples (Hovart et al., 1993a).

This procedure can determine the total concentration of CH_3Hg, the fraction of CH_3Hg that is particle bound, the reactive CH_3Hg fraction, total mercury content and the fraction of total mercury which is particulate bound.

Two other procedures are used to determine mercury speciation further. First, to determine the amount of "reactive" mercury (Hg_R) in a sample it is acidified to pH 1, then reduce with $SnCl_2$. This reduces the reactive portion of the mercury present to Hg^0. The sample is aerated, and the Hg^0 is collected on gold.

To determine the total mercury present, the sample is first oxidized by BrCl, followed by reduction of the BrCl with NH_2OH-HCl. This destroys many strong organo-metal associations and decomposes methylmercury, making all this mercury available for $SnCl_2$ reduction as above.

Air. The amount of Hg^0 in air is readily measured by pumping ambient air though a gold trap, since gold and mercury readily form amalgams. Bloom and Fitzgerald (1988) uses gold coated quartz sand as a trap. The gold trap is then heated, and the amount of Hg^0 vaporized is determined by atomic fluorescence spectroscopy.

Air samples are also collected by drawing air though a Carbotrap^R, which is then subjected to the protocol described earlier for Carbotraps^R derived from purged ethylated water samples (water purge, followed by gas chromatography and atomic fluorescence detection). This allows determination of the fraction of mono and dimethylmercury in air.

Particulate mercury is collected by drawing an atmospheric sample though a quartz wool plug. The plug is pryrolized, and the resulting mercury vapor is trapped on gold. The gold trap is then analyzed as above.

E.2.2 Swedish Sampling Program

The Swedish government has undertaken a large, multidisciplinary research program to study mercury in its environment, and much of the recent mercury literature is from this program. The concentration/speciation protocol most commonly employed in this program is outlined below (see work by Brosset and Lord, (1991); Lindqvist et al, (1991); and Westling, (1991)). Air sampling was not reviewed, since the Bloom methods can speciate more directly (Lindqvist et al.,1991, Chapter 1).

Water/Precipitation. Water samples are speciated using differing oxidation/reduction procedures which determine reactivity of the extracted mercury fraction. The following notation is used to divide up the mercury found in the samples.

Hg-tot (total mercury): The sample is oxidized with BrCl, followed by reduction of the BrCl with NH_2OH-HCl, and then $SnCl_2$ reduction. This is very similar to the Bloom method described above. The volatilized Hg^0 was purged from solution and concentrated on a gold trap. Upon desorption from the trap the amount of mercury is analyzed with a direct current plasma-atomic emission spectrometer.

Hg-II (reactive + non-reactive): The sample is treated with $NaBH_4$, followed by the same collection procedure as for Hg-tot.

Hg-IIa (reactive, acid labile, inorganic): Reduction with $SnCl_2$ in HCl or H_2SO_4.

Hg-IIb (non-reactive) = Hg-II - Hg-IIa

Hg-IIc (inert) = Hg-tot - Hg-II

E.2.3 Other Methods

These are some of the methods employed to pretreat biological samples, making them applicable to the same mercury analysis protocols as the U.S. and Swedish methods for water samples.

Plants/Soils. Typically, these samples are digested in Nitric acid or HNO_3-H_2O_2 followed by procedures similar to the above for water samples (Siegel et al., 1987).

Fish. The sample is digested in nitric acid with microwave heating under pressure, followed by treatment with potassium permanganate. After standing overnight, the excess potassium permanganate is reduced with sodium chloride-hydroxylamine hydrochloride (New Jersey, 1994). The sample can then be analyzed as in water/precipitation methods described above.

E.2.4 Detection Limits

One of the significant advances in mercury analytical methods over the past decade or so has been the ability to measure mercury concentrations accurately at much lower levels than were previously reported. Over the past two decades mercury determinations have progressed from detection of μg levels of total mercury to pg levels of particular mercury species (based on information in Mitra, 1986 and Hovart et al., 1993a and 1993b). Typical detection limits for data used or presented in this study are on the order of 1 ng/L for water samples (Sorensen et al., 1994), 0.1 ng/g for biota (Cappon, 1987; Bloom, 1992) and 0.1 ng/m^3 for atmospheric samples (Lindberg et al., 1992).

E.3 References

Bloom, N. and W. F. Fitzgerald (1988). Determination of Volatile Mercury Species at the Picogram level by Low-Temperature Gas Chromatography with Cold-Vapor Atomic Fluorescence Detection. *Analytica Chimica Acta*, 208:151-161.

Bloom, N. S. (1992). On the Chemical Form of Mercury in Edible Fish and Marine Invertebrate Tissue. *Can. J. Fisher. Aq. Sci.* 49:1010-1017.

Brosset, C. (1981). The Mercury Cycle. *Water, Air and Soil Poll.* 16:253-255.

Brosset, C. and E. Lord (1991). Mercury in Precipitation and Ambient Air: A new Scenario. *Water, Air and Soil Poll.* 56:493-506.

Cappon, C. J. (1987). Uptake and Speciation of Mercury and Selenium in Vegetable Crops Grown on Compost-Treated Soil. *Water, Air and Soil Poll.* 34:353-361.

Chapman and Hall (1984). *Dictionary of Organometallic Compounds.* Vol. 1, p. 1030.

Fitzgerald, W. F., R. P. Mason and G. M. Vandal (1991). Atmospheric Cycling and Air-Water Exchange of Mercury over Mid-Continental Lacustrine Regions. *Water, Air and Soil Poll.* 56:745-767.

Fitzgerald, W. F. (1994). Global Biogeochemical Cycling of Mercury. Presented at the DOE/FDA/EPA Workshop on Methylmercury and Human Health, Bethesda, MD March 22-23 1994.

Gmelins Handbuch, Der Anorganischen Chemie (34), Queck Silber, 1967 Verlag Chemie. GMBH Weinheim/Burger.

Halbach, S. (1985). The Octanal/Water distribution of Mercury Compounds. *Arch. Toxicol.* 57:139-141.

Hovart, M., L. Liang, N. S. Bloom (1993a). Comparison of distillation with other current isolation methods for the determination of methylmercury compounds in low level environmental samples. Part I: Sediments. *Analytica Chimica Acta*, 281:135-152.

Hovart, M., L. Liang, N. S. Bloom (1993b). Comparison of distillation with other current isolation methods for the determination of methylmercury compounds in low level environmental samples. Part II: Water. *Analytica Chimica Acta*, 282:153-168.

Iverfeldt, A. and I. Persson (1985). The Solvation Thermodynamics of Methylmercury(II) Species Derived from Measurements of the Heat of Solution and the Henry's Law Constant. *Inorganica Chimica Acta*, 103:113-119.

Lindberg, S. E., T. P. Meyers, G. E. Taylor, R. R. Turner, and W. H. Schroeder (1992). Atmosphere-Surface Exchange of Mercury to a Forest: Results of Modeling and Gradient Approaches. *J. of Geophy. Res.* 97(D2):2519-2528.

Lindqvist, O. and H. Rodhe (1985). Atmospheric Mercury-a review. *Tellus.* 37B:136-159.

Lindqvist, O., K. Johansson, M. Aastrup, A. Andersson, L. Bringmark, G. Hovsenius, L. Hakanson, A. Iverfeldt, M. Meili, and B. Timm (1991). Mercury in the Swedish Environment - Recent Research on Causes, Consequences and Corrective Methods. *Water, Air and Soil Poll.* 55:(all chapters)

Mitra, S. (1986). *Mercury in the Ecosystem.* Trans Tech Publications Ltd. Switzerland.

Martin Marietta Energy Systems (MMES), 1994. Material Safety Data Sheets for methylmercuric chloride, mercuric sulfide, metallic metrcury, and mercuric chloride. MMES, Inc. P.O. Box 2008, M.S. 6291, Oak Ridge, TN 37831.

New Jersey Department of Environmental Protection and Energy Division of Science and Research (1994). Preliminary Assessment of Total Mercury Concentrations in Fishes from Rivers, Lakes and Reservoirs of New Jersey. 93-15F.

Nriagu, J. O. (1979). The Biogeochemistry of Mercury in the Environment. Elsevier/North Holland. Biomedical Press: New York.

Westling, O. (1991). Mercury in Runoff from drained and undrained Peatlands in Sweden. *Water, Air and Soil Poll.* 56:419-426.

Petersen, G., Å. Iverfeldt and J. Munthe, 1995. Atmospheric mercury species over Central and Northern Europe. Model calculations and comparison with observations from the Nordic Air and Precipitation Network for 1987 and 1988. *Atmospheric Environment* 29:47-68.

Revis, N. W., T. R. Osborne, G. Holdsworth, and C. Hadden (1990). Mercury in Soil: A Method for Assessing Acceptable Limits. *Arch. Environ. Contam. Toxicol.*, 19:221-226.

Schuster, E (1991). The Behavior of Mercury in the Soil with special emphasis on Complexation and Adsorption processes - A Review of the Literature. *Water, Air and Soil Poll.* 56:667-680.

Sorensen, J.A., G.E. Glass and K.W. Schmidt. (1994). Regional Patterns of Mercury Wet Deposition. *Environ. Sci. Tech.* 28:2025-2032.

U.S. EPA (1993). Summary Review of Health Effects Associated with Mercuric Chloride. EPA/600/R-92/199

Westling, O. (1991). Mercury in Runoff from drained and undrained Peatlands in Sweden. *Water, Air and Soil Poll.* 56:419-426.

World Health Organization (1990). Environmental Health Criteria 101: Methylmercury. Geneva.

APPENDIX F

DESCRIPTION OF MODEL PLANTS

TABLE OF CONTENTS

F.1 Introduction F-1

F.2 Relationship of Plant Process Conditions to Emissions F-1
F.2.1 Municipal Waste Combustors F-4
F.2.1.1 Description of Source Category F-4
F.2.1.2 Summary of Available Data on Emissions and Controls F-5
F.2.1.3 Selection of MWC Model Plant Parameters F-7
F.2.2 Medical Waste Incinerators F-8
F.2.2.1 Description of Source Category F-8
F.2.2.2 Summary of Available Data on Emissions and Controls F-8
F.2.2.3 Selection of MWI Model Plant Parameters F-8
F.2.3 Utility Boilers F-9
F.2.3.1 Description of Source Category F-9
F.2.3.2 Summary of Available Data on Emissions and Controls F-9
F.2.3.3 Selection of Model Plant Parameters F-11
F.2.4 Chlor-Alkali Production F-11
F.2.4.1 Description of Source Category F-11
F.2.4.2 Summary of Available Data on Emissions and Control F-12
F.2.4.3 Selection of Model Plant Parameters F-12
F.2.5 Primary Copper Smelters F-12
F.2.5.1 Description of Source Category F-12
F.2.5.2 Summary of Available Data on Emissions and Control F-13
F.2.5.3 Selection of Model Plant Parameters F-13
F.2.6 Primary Lead Smelters F-13
F.2.6.1 Description of Source Category F-13
F.2.6.2 Summary of Available Data on Emissions and Controls F-14
F.2.6.3 Selection of Model Plant Parameters F-14
F.3 References F-15

LIST OF TABLES

Table F-1 Process Parameters for Model Plants F-2
Table F-2 Mercury Removal Efficiencies F-10

LIST OF FIGURES

Figure F-1 Distribution of Mercury in EPA Method 29 Sampling Train, Camden County and Stanislaus County Carbon Injection Projects F-6

F.1 Introduction

Model plants representing six source categories were developed to represent a range of mercury emission parameters. The source categories were selected for the local impact analysis based on their estimated annual mercury emissions or their potential to be localized point sources of concern. The categories were municipal waste combustors (MWCs), medical waste incinerators (MWIs), utility boilers, chlor-alkali plants, primary copper smelters, and primary lead smelters.

Descriptive characteristics, or parameters for each model plant were selected after evaluating the characteristics of the entire source class. Important variables for mercury risk assessments include the following: mercury emission rates, mercury speciation, and mercury transport/deposition rates. Important model plant parameters included stack height, stack diameter, stack volumetric flow rate, stack gas temperature, plant capacity factor (relative average operating hours per year), stack concentration, and mercury speciation.

Table F-1 summarizes the model plant parameters modeled in this analysis. These parameters represent operating conditions associated with each source type's current mercury control. That is, mercury emissions reductions being achieved by the air pollution control devices presently in place were considered for each source category. These parameters are not meant to represent a "worst-case" emissions scenario; they are believed to be representative of the full range of sources (of a given category) across the United States. The amount of uncertainty surrounding the emission rates varies for each model plant. This uncertainty is reflected in Chapter 6 -- Research Needs -- of this ~~report~~ VOLUME 8

F.2 Relationship of Plant Process Conditions to Emissions

Mercury speciation estimates for elemental mercury (Hg^0) and divalent mercury (Hg^{2+}) were made using literature results of thermal-chemical modeling of mercury compounds in flue gas, interpretation of bench and pilot scale combustor experiments and interpretation of available field test results.

The amount and speciation of mercury emitted from high temperature process depends on the composition of the feed material, amount of mercury in the process feed material, process operating conditions, and process flue gas cleaning techniques (Lindqvist and Schager, 1990).

"The inorganic mercury compounds that are considered important in high temperature processes are HgS(s), HgO(s,g), $HgCl_2$(s,g), Hg_2Cl_2(s) and $HgSO_4$(s)." Some organic compounds, such as methylmercury, CH_3HgCH_3, and CH_3HgCl may also occur (Hall et al., 1991). Thermochemical calculations indicate that at combustion temperatures above 700°C nearly all mercury is vaporized to form gaseous Hg^0. As the flue gas cools, changing equilibrium conditions favor oxidized forms of mercury. When there are significant levels of HCl, Cl_2, O_2 and SO_2 in the flue gas, all of the above oxidized inorganic forms of mercury will tend to occur. In flue gas from coal and peat combustion at temperatures below 200°C, the dominant equilibrium species are HgO and Hg^0. For combustion wastes containing relatively high levels of chlorine, $HgCl_2$ will be the dominant mercury compound (Hall et al., 1990; Hall 1991, Lindqvist and Schager, 1990).

Table F-1
Process Parameters for Model Plants

Model Plant	Plant size	Capacity (% of year)	Stack Height (ft)	Stack Diameter (ft)	Mercury Emission Rate (kg/yr)	Exist Velocity (m/sec)	Exit Temp. (°F)
Large MWC	2,250 tons/day	90%	230	9.5	1,330	21.9	285
Small MWC	200 tons/day	90%	140	5	170	21.9	375
Continuous MWI	1,500 lb/hr capacity (1,000 lb/hr actual)	90%	40	2.7	80	7.3	1500
Intermittent MWI	200 lb/hr capacity (133 lb/hr actual)	20%	40	1 2	2.4	7.3	1500
Large Coal-fired Utility Boiler	975 Megawatts	65%	732	27	230	31.1	273
Medium Coal-fired Utility Boiler	975 Megawatts	65%	465	18	90	26.7	275
Small Coal-fired Utiity Boiler	100 Megawatts	65%	266	12	10	6.6	295
Medium Oil-fired Utility Boiler	285 Megawatts	65%	290	14	2	20.7	322
Chlor-alkali plant	300 tons chlorine/ day	90%	10	0.5	380	0.1	Ambient
Primary Copper Smelter	180 tons copper/day	90%	505	15	536.0	6	430
Primary Lead Smelter	304 tons lead/day	90%	350	20	2680	2.8	347

Table F-1
Process Parameters for Model Plants
(continued)

Model Plant	Base-Case Speciation (%)			Alternate Speciation (%)		
	Hg^0	Hg^{2+}	Hg_p	Hg^0	Hg^{2+}	Hg_p
Large MWC	20	60	20	20	0	80
Small MWC	20	60	20	20	0	80
Continuous MWI	20	60	20	20	0	80
Intermittent MWI	20	60	20	20	0	80
Large Coal-Fired Utility Boiler	50	30	20	50	0	50
Medium Coal-Fired Utility Boiler	50	30	20	50	0	50
Small Coal-Fired Utility Boiler	50	30	20	50	0	50
Medium Oil-Fired Utility Boiler	50	30	20	50	0	50
Chlor-alkali plant	70	30	0	70	0	30
Primary Copper Smelter	85	10	5	85	0	15
Primary Lead Smelter	85	10	5	85	0	15

Experimental evidence shows that the speciation of mercury is more complicated than indicated by thermochemical equilibrium calculations. For example SO_2, soot, activated carbon, CaO and iron may promote low temperature reactions that reduce oxidized forms of mercury to Hg^0 (Hall et al., 1990; Hall 1991). The presence of trace gases and particulate in flue gas promote mercury reactions and provide surfaces for physical and chemical adsorption. Reaction kinetics can also be expected to play an important role under the changing thermodynamics conditions that exist in high temperature flue gas streams. Also, some gases such as HCl may not always be or reaction with mercury because of mixing limitations.

While thermochemical chemical calculations provide information on likely mercury compounds in flue gas, their presence and relative magnitude have not been confirmed with experimental data. This shortcoming is primarily the result of difficulties in the sampling and analysis for mercury compounds. One study of mercury speciation on a pilot combustor and a full scale municipal waste incinerator found that the flue gas contained mainly mercury chlorides, and that Hg^0 mercury was present in insignificant amounts (Metzger and Braun, 1987). Another experimental study of mercury sampling methods concluded that total mercury and Hg^0 can be adequately measured, that the results of the different sampling methods tested for ionic mercury differed significantly and that additional efforts must be devoted the development of mercury speciation methodologies (Lindqvist and Schager, 1990).

In a third study, conducted in the U.S., measurements with a sampling train designed to provide information on mercury speciation tentatively indicated the presence of methylmercury in the exhaust of coal and municipal waste combustors (Bloom, 1993). It was later reported that the methylmercury reported in that study were the result of artifacts associated with the laboratory analytical procedures. Based on these studies it is concluded that the results of tests providing information on total mercury, Hg^0 and Hg^{2+} are probably valid, but results of tests for methylmercury and other compounds must be considered suspect until sampling and analysis protocols for those compounds are validated.

The capture of mercury in flue gas cleaning devices depends on the mercury form [e.g, speciation and phase (gas, liquid or solid)] and the control devices employed. Most metals condense to form solid particles as the flue gases are cooled so that the metals can be collected as particulate matter (PM). However, mercury specie such as Hg^0 and $HgCl_2$ are vapors at flue gas temperatures and are difficult to control. Some mercury compounds such as $HgCl_2$ are soluble in water and can be controlled by wet scrubbers. Some specie such as HgCl2 can be adsorbed onto activated carbon and fly ash carbon for subsequent collection as PM. Reagents can be used to produce mercury compounds that condense for collection as PM. Reagents can also be used to produce soluble mercury compounds for scrubber collection.

F.2.1 Municipal Waste Combustors

F.2.1.1 Description of Source Category

There are three major types of municipal waste combustors (MWC's): mass burn combustors, refuse-derived fuel (RDF) combustors and modular combustors. There are number of sub-categories of these three major types, plus some other types of MWCs, such as fluidized bed combustors. These other types of MWCs constitute a minor fraction of the total MWC population.

As of December 1991, there were over 160 MWC plants in the U.S. with aggregate capacities ranging from greater than 40 tons/day to 3000 tons/day (Kiser, 1993). Most large facilities contain

from two to four mass burn or refuse-derived fuel (RDF) combustors. Approximately 50 percent of the MWC capacity in the U.S. now employ spray dryers and fabric filters (SD/FFs) or SD and electrostatic precipitators (SD/ESPs) for emission control. The remaining large facilities do not have acid gas control equipment and generally use good combustion practice and electrostatic precipitators (ESPS) for emissions control. Few U.S. facilities use wet scrubbers. A number of facilities are planning to use activated carbon to control mercury emissions as mandated by siting permits. At least one MWC using activated carbon is in commercial operation.

F.2.1.2 Summary of Available Data on Emissions and Controls

Uncontrolled mercury emissions from MWC's range from less than 200 ug/dscm to more than 1500 ug/dscm depending on the mercury content of wastes being burned. Average uncontrolled flue gas concentrations in mass burn combustors are estimated to be in the range of 600 to 700 ug/dscm (U.S. EPA, 1993; White et al., 1992; Nebel et al., 1992; White et al., 1993). Uncontrolled flue gas concentrations in RDF combustors are somewhat lower since some mercury contained in batteries and other items is removed in the process that produces RDF.

For combustion sources, the degree of mercury control depends on the flue gas composition, the amount of fly ash carbon and the flue gas cleaning techniques employed. Welldesigned and operated mass burn combustors have little carbon in their fly ash; even when equipped with SD/FFs or SD/ESPs they exhibit mercury control levels that typically range from 0 to 50 percent (Nebel et al., 1992; White et al.,1992). When powdered activated carbon is injected into the flue gas upstream of the spray dryer in mass burn combustors, control levels exceeding 90 percent can be achieved for both SD/FF and SD/ESP systems (Brown and Felsvang, 1992; White et al., 1992; Nebel et al., 1992; White et al., 1993). However, SD/ESP systems require from 2 to 3 times more carbon than SD/FF systems (Kilgroe et al., 1993).

The RDF combustors contain a relatively high amounts of carbon in the fly ash and exhibit control efficiencies of approximately 80 percent when equipped with SD/ESPs and above 90 percent when equipped with SD/FFs (White et al., 1992). Injection of powdered activated carbon can also be used to augment mercury control in RDF combustion facilities.

Little information is available concerning the performance of flue gas cleaning techniques for controlling mercury in modular MWCs. It is expected that the performance of flue gas cleaning devices on modular units will be similar to the performance of comparable equipment installed on conventional mass burn combustors.

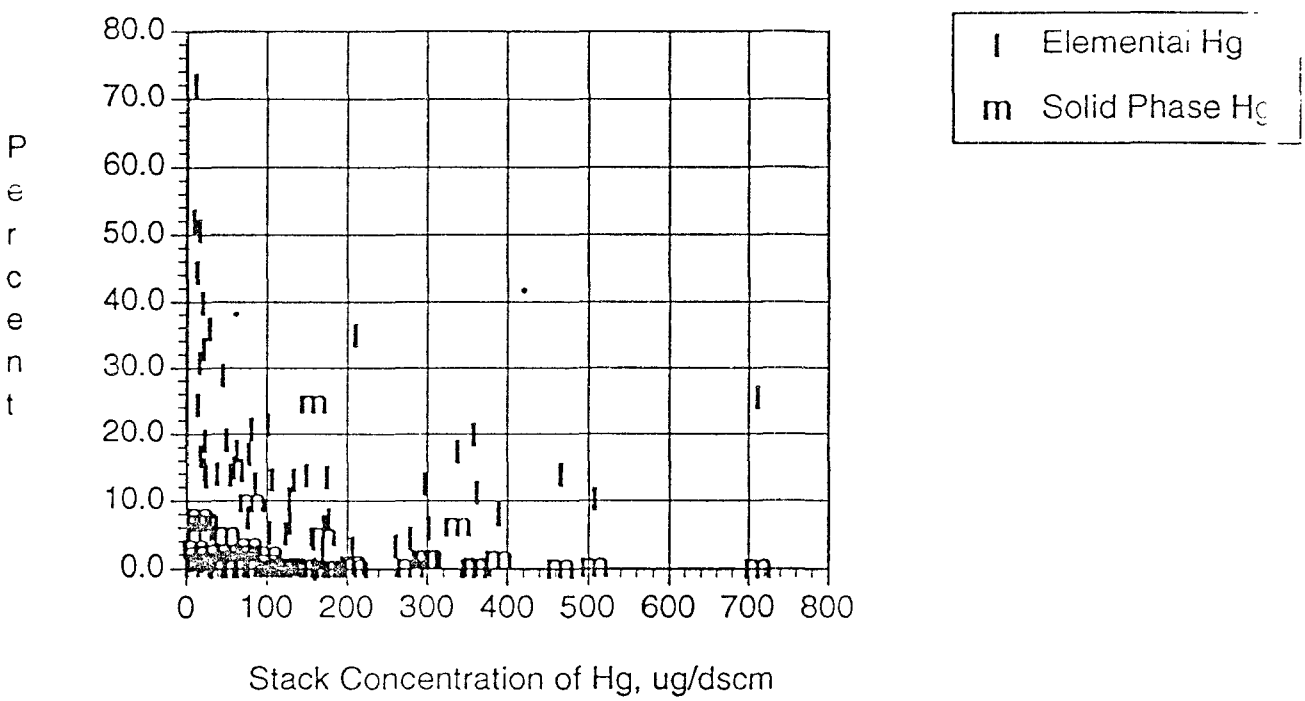
Electrostatic precipitators and wet scrubbers are commonly used to control emissions from European MWCs. Some European plants have installed activated carbon beds downstream of the primary air pollution control devices to act as polishing filters for control of metals, dioxins and acid gases. The use of activated carbon filter beds in combination with conventional control equipment have demonstrated mercury reductions exceeding 99 percent and mercury outlet concentrations of less than 1 ug/dscm (Hartenstein, 1993).

In conducting risk assessments, it is important to estimate the form and speciation of mercury emitted in the flue gas. Several studies estimate the speciation of mercury in MWC flue gases. Metzger and Braun (1987) estimate that nearly all mercury emitted from MWCs at flue gas cleaning temperatures is in the form of mercury chlorides. Lindqvist and Scahger (1990) estimate that the speciation of mercury emissions from European waste incinerators consist of 10 percent Hg^0, 85 percent Hg^{2+}, and 5 percent mercury associated with PM (Hg (PM)). Pacyna (1991) estimates that

mercury emissions from European waste incinerators consist of 10 percent Hg^0, 85 percent Hg^{2+}, and 5 percent Hg(PM).

There is currently no validated U. S. EPA method for determining the speciation of mercury in stack gas. Information on the chemical behavior of mercury and the distribution of mercury in EPA's multi-metal sampling train (Method 29) can be used to estimate the form and speciation of mercury in MWC stack gas. Mercury found in the probe and filter can be assumed to have been vapor-phase mercury adsorbed onto PM or be a solid-phase compound. Both phases are associated with PM. Mercuric chloride is soluble in water and mercury found in the $KMnO_4/H_2SO_4$ impingers is probably Hg^0. The distribution of multi-metal train samples collected during the activated carbon injection tests at the Camden County MWC and Stanislaus County tests is shown in Figure F-1 (Nebel et al., 1992; White et al., 1993).

Figure F-1
Distribution of Mercury in EPA Method 29 Sampling Train,
Camden County and Stanislaus County Carbon Injection Projects



Tests showing mercury stack concentrations of greater than 100 µg/dscm represent either low carbon injection feed rates or no carbon injection. For these tests, Hg^0 ranged from 2 to 26 percent of total mercury. As carbon injection rates and mercury capture increased, the percentage of Hg^0 as a fraction of total mercury increased. This implies that Hg^{2+} is more easily captured by activated carbon than Hg^0. For mercury stack concentrations less than 50 µg/dscm, the fraction of Hg^0 ranged from approximately 14 to 72 percent. The fraction of Hg(PM) was generally below detection limits for most tests. It exceeded 10 percent for only one test and was below 10 percent for all other tests where it was detected.

At low levels of control, the stack concentration of mercury is probably 15 to 30 percent $Hg^0(v)$ and the rest is $Hg^{2+}(v)$. At high levels of control, $Hg^{2+}(v)$ is selectively removed, increasing the relative concentration of $Hg^0(v)$, and the relative concentration of $Hg^0(v)$ may be 50 percent or higher.

For this analysis the speciation profiles for these course types were derived from Petersen et al., 1995. The profiles are shown below for each model plant.

F.2.1.3 Selection of MWC Model Plant Parameters

In this analysis, the range of MWC plant conditions are represented by a large 2250 tons/day model plant and a small 200 tons/day model plant. The large model plant consists of 3 conventional 750 tons/day mass burn combustors each equipped with a SD/FF. The small MWC model plant was assumed to consist of two conventional 100 tons/day mass burn combustors each equipped with a system for dry sorbent injection followed by an electrostatic precipitator (DSI/ESP).

Large MWC Model Plant

Average uncontrolled mercury emissions for mass burn combustors are typically in the range of 600 to 700 ug/dscm. (U.S. EPA, 1993; White et al., 1992; Nebel et al., 1992; White et al., 1993). Control efficiencies of units equipped with SD/FF may range from 0 to 50 percent with typical control levels near 30 percent. It was assumed for the baseline scenario that 30 percent control is achieved by the flue gas cleaning equipment (SD/FF) and that the uncontrolled emissions average 490 ug/dscm. Stack emissions were assumed to consist of mercury consisting of 60 percent Hg^{2+} 20 percent Hg^0 and 20 percent particulate mercury.

Small MWC Model Plant

The small MWC model plant was assumed to consist of two state-of-the-art 100 tons/day mass burn combustors equipped with DSI/ESPs. The waste composition and behavior of mercury in the combustor was assumed to be similar to that observed in large mass burn combustors. As a result, for the small MWC model plant it was assumed that no control is achieved by the flue gas cleaning equipment and that the uncontrolled emissions average 700 ug/dscm. Stack emissions were assumed to consist of mercury consisting of 60 percent Hg^{2+} and 20 percent Hg^0, and 20 percent particulate mercury.

F.2.2 Medical Waste Incinerators

F.2.2.1 Description of Source Category

Medical waste incinerators (MWIs) are small incinerators that burn from 1 ton/day (0.9 Mg/day) to 60 tons/day (55 Mg/day) of infectious and noninfectious wastes. These wastes are generated by various facilities including hospitals, clinics, medical and dental offices, veterinary clinics, nursing homes, medical laboratories, medical and veterinary schools and research laboratories and funeral homes.

Approximately 3,700 MWIs currently operate throughout the country; geographic distribution is relatively even. Of these 3,700 units, about 3,000 are hospital incinerators, about 150 are commercial units, and the remaining units are distributed among veterinary facilities, nursing homes, laboratories, and other miscellaneous facilities (U.S. EPA, 1994).

The primary function of MWIs is to render the waste biologically innocuous and to reduce the volume and mass of solids that must be landfilled (by combusting the organic material contained within the waste). Currently, three major types of MWI operate in the United States: continuous, intermittent, and batch. All three have two chambers that operate on a similar principle. Waste is fed to a primary chamber, where it is heated and volatilized. The volatiles and combustion gases are then sent to a secondary chamber, where combustion of the volatiles is completed by adding air and heat. All mercury in the waste is assumed to be volatilized during the combustion process and emitted with the combustion stack gases.

F.2.2.2 Summary of Available Data on Emissions and Controls

A number of air pollution control systems are used to control PM and acid gas emissions from MWI stacks. Most of these systems fall into the general classes of either wet or dry systems. Wet systems typically comprise a wet scrubber, designed for PM control (venturi scrubber or rotary atomizing scrubber), in series with a packed-bed scrubber for acid gas removal and a high efficiency mist elimination system. Most dry systems use a fabric filter for PM removal, but ESP's have been used on some of the larger MWIs. All of these systems have limited success in controlling mercury emissions. Recent EPA studies indicate that sorbent injection/fabric filtration systems can achieve improved mercury control by adding activated carbon to the sorbent material (U.S. EPA, 1994).

Mercury speciation data for MWI facilities were derived from Petersen et al., 1995. Stack emissions were assumed to consist of vapor-phase mercury consisting of 60 percent Hg^{2+} and 20 percent Hg^{0}, and 20 percent Hg(PM).

F.2.2.3 Selection of MWI Model Plant Parameters

To represent the MWI source category, two model plants were devised: one continuous facility and one facility that is operated intermittently.

Continuous MWI Facility

The continuous MWI model plant was assumed to consist of one starved air modular incinerator with a design capacity of 1500 lb/hr. The unit was assumed to operate at an average capacity of 1000 lb/hr for 7,889 hours/yr. The incinerator was assumed to be equipped with a DSI/ESP with no control of mercury assumed to occur across this system (Lerner, 1992). Mercury in

the waste feed was assumed to result in flue gas emissions of 3000 ug/dscm (U.S. EPA. 1991: U.S. EPA, 1992b). Stack emissions were assumed to consist of vapor-phase mercury consisting of 60 percent Hg^{2+} and 20 percent Hg^0, and 20 percent Hg(PM).

Intermittent MWI

The intermittent MWI model plant was assumed to consist of one state-of-the-art starved air modular incinerator with a design capacity of 200 lb/hr. The unit was assumed to operate at an average capacity of 133 lb/hr, 5 days per week. An annual capacity factor of 20 percent (1753 hr/year) was assumed. No flue gas cleaning equipment was assumed. Mercury in the waste feed was assumed to result in uncontrolled flue gas emissions of 3000 ug/dscm. Stack emissions were assumed to consist of vapor-phase mercury consisting of 60 percent Hg^{2+} and 20 percent Hg^0, and 20 percent Hg(PM).

F.2.3 Utility Boilers

F.2.3.1 Description of Source Category

Utility boilers are large boilers used by public and private utilities to generate electricity. There are approximately 1800 utility boilers in the U.S. which burn coal, oil and natural gas. In 1990, utility boilers consumed fossil fuel at an annual level of 21 x 10^{15} British thermal units (Btu). About 80 percent of this total energy consumption resulted from coal combustion, 6 percent from oil and petroleum fuels and 14 percent from natural gas consumption. Ninety-five percent of the coal burned is bituminous and subbituminous; lignite accounts for 4 percent. Mercury emission estimates were not calculated for natural gas combustion because reliable test data necessary to calculate an emission factor do not exist. Given these factors, the indirect exposure analysis focused only on coal-fired units burning bituminous coal and residual oil-fired units.

F.2.3.2 Summary of Available Data on Emissions and Controls

About 80 percent of coal-fired utility boilers use ESPs for PM control. Scrubbers (or flue gas desulfurization units (FGDs)) are the most commonly used device for sulfur dioxide (SO_2) control. Spray dryer absorption (SDA), or dry scrubbing, followed by a PM control device may also be used. Mechanical collectors are used infrequently. Coal washing, which separates coal and impurities from crushed and screened coal by differences in specific gravity, is done routinely to meet customer specifications for heating value, ash and sulfur content. Advanced coal cleaning techniques may reduce the concentration of mercury contained in the mineral and organic phases of the coal, but the reliability and feasibility of these emerging techniques are unknown at this time.

Carbon filter beds are being used successfully in Europe for control of heavy metals, organic compounds and acid gases (Hartenstein, 1993). Five full-scale applications of carbon beds are currently in use for utilities, with future applications planned for hazardous waste incinerators and MWCs. Activated carbon injection has been used successfully in the U.S. for mercury removal from the stack gas of MWCs and MWIs. This technology has been tested on a pilot-scale basis in the U.S.. Table F-2 summarizes the control efficiencies for various control technologies for utility boilers, based on pilot-scale test data.

Table F-2
Mercury Removal Efficiencies

Control Technique	Range of Removal Efficiency (percent)	Median Removal Efficiency (percent)	Reference
Carbon bed	Unknown	99	Hartenstein (1993)
Fabric Filter + AC (Low temp. + Low C injection rate)	76-99	98	Volume VII, App. A.
Fabric Filter + AC (High temp. + Low C injection rate)	14-47	29	Volume VII. App. A
Fabric Filter + AC (Low temp. + High C injection rate)	95-99	98	Volume VII, App. A
Fabric Filter + AC High temp. + High C injection rate)	69-91	73	Volume VII, App. A
SDA/ESP + AC	75-91	86	Volume VII. App. A
SDA/FF + AC	50- >99	NA	Volume VII, App. A
Fabric filter	0 - 51	29	Volume II
Scrubber (FGd)	18 - 84	85	Volume II
Dry scubber (SDa)	23 - 83	67	Volume II
ESP	0 - 22	10	Volume II
Mechanical collector	0	0	Volume II
Coal washing	-200 - 64	21 (average removal)	Volume II
Advanced coal washing	Unknown	-	Volume VII

Mercury emissions of mercury from utility boilers can vary depending on the mercury content of the fuel and the control technique used. Based on emissions test data (as described in the mercury emissions inventory, documented in a separate report), a mercury emission rate of 10 ug/dscm was chosen to represent emissions from a coal-fired utility with PM control. Two ug/dscm was chosen as the emission rate for mercury emissions from an uncontrolled residual oil-fired utility. This emission rate is a worse-case estimate for an oil-fired plant. This high estimate was selected for the modeling because the impacts from oil-fired boiler were expected to be very small even using the worst-case.

As discussed above, the chemical specie of mercury being emitted affects both the removal efficiency of the control device and the deposition of mercury from the atmosphere. Based on Petersen et al., 1995, it was assumed for the local impact analysis that the mercury emitted from the utility model plants (both coal- and oil-fired) consisted of 50 percent HgO and 30 percent Hg^{2+}, and 20 percent Hg(PM).

F.2.3.3 Selection of Model Plant Parameters

The source of data for selecting the model plant sizes was the Utility Data Institute (UDI)/Edison Electric Institute (EEI) Power Statistics Database (1991 edition). This database provided information on fuel use, boiler sizes and stack parameters. The database had information on 1708 units, of which 795 were bituminous coal-fired and about 225 were oil-fired. The remainder were primarily fired with natural gas although there were some boilers burning lignite and anthracite coals. Given the predominance of bituminous coal-fired units, those were the units chosen for the indirect exposure analysis as well as one residual oil-fired unit.

The 795 coal-fired units were divided into 3 size classifications (by megawatt (MW)) according to 33rd percentiles. The size classes had approximately the same number of units in each. The "large" group which consisted of units greater than or equal to 575 MW had 262 units. The "medium" group which consisted of units between 199 MW and 575 MW had 256 units. The "small" group consisting of units greater than 25 MW but less than 200 MW had 277 units. The model plant parameters were chosen by evaluating each group separately and taking the average value for each parameter from each group (e.g., for representative MW, stack height and stack diameter).

Based on this analysis, three boiler sizes were chosen as the basis for the coal-fired model plants: 975 MW, 375 MW, and 100 MW. The same type of analysis for the oil-fired units led to the selection of a 285 MW residual-oil fired unit as a representative model plant size for the oilfired units.

Coal-Fired Utility Model Plants

All of the coal-fired utility model plants had a capacity factor of 65 percent, and were equipped with a cold-side ESP. The inlet mercury level (i.e., the amount of mercury entering the emission control device) was assumed to be 10 μ/dscm (4.4 gr/million dscf). For emissions, it was assumed that no mercury control across the ESP was achieved and that the mercury emissions were 10 ug/dscm (4.4 gr/million dscf).

Oil-Fired Utility Model Plant

The oil-fired utility model plant was a 285 MW boiler firing No. 6 fuel oil containing 1 percent sulfur and 300 ppm chlorine. It was assumed to have a capacity factor of 65 percent, and was not equipped with any particulate matter control device. The inlet mercury level associated with this model plant was assumed to be 2 μ/dscm (1 gr/million dscf). It was assumed that no mercury control was achieved and that the mercury emissions were 2 ug/dscm (1 gr/million dscf).

F.2.4 Chlor-Alkali Production

F.2.4.1 Description of Source Category

Chlor-alkali production using the mercury cell process, (which is the only chlor-alkali process that uses mercury), accounted for 17 percent of all U.S. chlorine production in 1988 (U.S. EPA, 1993). The three primary sources of mercury air emissions from chlor-alkali plants are the byproduct hydrogen stream, the end box ventilation air and the cell room ventilation air. The byproduct hydrogen stream from the decomposer is saturated with mercury vapor and may also contain fine droplets of liquid mercury. The quantity of mercury emitted in the end box ventilation air depends on the degree of mercury saturation and the volumetric flow rate of the air. The amount of mercury in the cell room ventilation air is variable and comes from many sources, including end box sampling,

removal of mercury butter from end boxes, maintenance operations, mercury spills, equipment leaks and cell failure (U.S. EPA, 1984).

F.2.4.2 Summary of Available Data on Emissions and Control

The most recent source of mercury emission data is from Clean Air Act section 114 survey questionnaires of the chlor-alkali industry (as referenced in section 4.2.1 of Volume II of this report). The industry survey data were the basis for the nationwide estimate for mercury emissions of 5.9 Mg (6.5 tons). A previous report by the U.S. EPA was used to develop plant- specific parameters (U.S. EPA, 1973).

The control techniques that are typically used to reduce the level of mercury in the hydrogen streams and in the ventilation stream from the end boxes are the following: gas stream cooling, mist eliminators, scrubbers and adsorption on activated carbon or molecular sieves. Mercury emissions from the cell room air circulation are not subject to specific emission control measures. Concentrations are maintained at acceptable worker exposure levels through good housekeeping practices and equipment maintenance procedures (U.S. EPA, 1984).

Speciated emissions data for chlor-alkali plants are extremely limited. For this analysis was assumed that the emitted mercury was in the vapor phase and consisted of 70 percent Hg^0 and 30 percent Hg^{2+} (Peterson et al., 1995).

F.2.4.3 Selection of Model Plant Parameters

For the indirect exposure analysis, one chlor-alkali model plant, which produces 273 Mg (300 tons) of chlorine per day, was devised. This model plant represented the mid-range size of chlor-alkali plants in operation (U.S. EPA, 1984). The model plant had individual flow rates from the hydrogen and end-box streams of 4,080 dscm/hr (144,000 dscf/hr) each at 21 percent 02 (combined to equal 8,160 dscm/hr [288,000 dscf/hr]) (U.S. EPA, 1973). A 90 percent capacity factor (operation for 7889 hr/yr) was assumed.

The typical emissions control scenario for both the hydrogen and end-box streams was assumed to consist of a heat exchanger to coot the effluent gas, followed by a knockout drum to separate the condensed mercury from the hydrogen and end-box streams. A mercury level of 1,000 g/day (2.2 lb/day) was assumed for the purpose of indirect exposure analysis to be consistent with the federally-mandated mercury standard for the hydrogen and end-box streams at all chlor-alkali plants (U.S. EPA, 1984).

F.2.5 Primary Copper Smelters

F.2.5.1 Description of Source Category

Copper is recovered from a sulfide ore principally by pyrometallurgical smelting methods. The ore contains significant quantities of arsenic, cadmium, lead, antimony and mercury. A conventional copper smelting process sequentially involves roasting ore concentrates to produce calcine, smelting of roasted or unroasted ore concentrates to produce matte, converting matte to produce blister copper and fire refining the blister copper in an anode furnace. After fire refining, the 99.5 percent pure copper is cast into "anodes" and sent to an electrolytic refinery for further impurity removal (Buonicore and Davis, 1992).

F.2.5.2 Summary of Available Data on Emissions and Control

Mercury emissions data for primary copper smelting facilities are very limited. Only one emission test report for the Copper Range Company located in White Pine, Michigan, containing metals analysis results was available (TRC Environmental, 1992). This facility operates a reverberatory furnace with an ESP to control PM. The exhaust stream from the convertor (which is uncontrolled) is mixed with the exhaust from the ESP outlet and is routed through the main stack and discharged into the atmosphere.

Because the feed mix varies from facility to facility and because the Copper Range Company is the only facility in the United States that operates a reverberatory, the emissions data from the Copper Range Company may not be representative of industry practice. Because data were available for this plant, the operating parameters for the White Pine, Michigan facility were selected for the indirect exposure analysis.

Speciated emissions data for primary copper smelters are extremely limited. For this analysis it was assumed that the emitted mercury consisted of 85 percent Hg^0, 10 percent Hg^{2+} and 5 percent Hg(PM) (Peterson et al., 1995).

F.2.5.3 Selection of Model Plant Parameters

Copper smelters use high efficiency air pollution control options to control PM and SO_2 emissions from roasters, smelting furnaces, and convertors. Electrostatic precipitators are the most common PM control device at copper smelters. Control of SO_2 emissions is achieved by absorption to sulfuric acid in the sulfuric acid plants, which are common to all copper smelters. For the emissions scenario, it was assumed that no mercury emissions control occurred with these emission control devices. Using the test data for the Copper Range facility described above, a mercury emission factor of 0.068 kg/hour (0.15 lb/hour) was calculated (U.S. EPA, 1993). A 90 percent capacity factor was assumed. This corresponds to a 236 Mg/day (260 tons/day) copper production capacity.

The mercury emissions for both of the primary lead smelter off-gas streams were assumed to be uncontrolled resulting in an outlet mercury concentration of 1,000 ug/scm (436 gr/million scf at 19 percent O_2 for each stream.

F.2.6 Primary Lead Smelters

F.2.6.1 Description of Source Category

Lead is recovered from a sulfide ore, primarily galena (lead sulfide--PbS), which also contains small amounts of copper, iron, zinc and other trace elements such as mercury. Recovery of lead from the lead ore in primary lead smelters consists of three main steps: sintering, reduction, and refining. Sintering is carried out in a sintering machine, which is a continuous steel pallet conveyor belt. Each pallet consists of perforated grates, beneath which are wind boxes connected to fans to provide a draft through the moving sinter charge. The sintering reactions take place at about 1000°C (1832°F) during which lead sulfide is converted to lead oxide. Since mercury and its compounds vaporize below this temperature, most of the mercury present in the ore can be expected to be emitted during sintering either as Hg^0 or as HgO.

Reduction of the sintered lead is carried out in a blast furnace at a temperature of 1600°C (2920°F). In the blast furnace, the sinter is reduced to lead. The heat for the reaction is supplied by

the combustion of coke. Slag, consisting of impurities, flows from the furnace and is either land deposited or is further processed to recover zinc. The impurities include arsenic, antimony, copper and other metal sulfides, iron and silicates. Lead bullion, which is the primary product, undergoes a preliminary treatment to remove impurities, such as copper, sulfur, arsenic, antimony and nickel. Residual mercury can be expected to be emitted during the reduction step. Further refining of the lead bullion is carried out in cast iron kettles. Refined lead, which is 99.99 to 99.999 percent pure, is cast into pigs for shipment (U.S. EPA, 1988a).

F.2.6.2 Summary of Available Data on Emissions and Controls

Primary lead smelters use high-efficiency emission control systems to reduce the levels of PM and SO_2 from the blast furnace and sintering machines. Centrifugal collectors (cyclones) are used in conjunction with FFs or ESPs for PM control. Control of SO_2 emissions is achieved by absorption to form sulfuric acid in the sulfuric acid plants, which are commonly part of lead smelting plants. The blast furnace and the sintering machine operate at very high temperatures (in excess of 1000°C [1832°F]), and as a result, mercury would be emitted from these sources in vapor form. Therefore, particulate control devices would have little effect on mercury emissions from the sintering machine and blast furnace.

Speciated emissions data for primary lead smelters are extremely limited. For this analysis it was assumed that the emitted mercury consisted of 85 percent Hg^0, 10 percent Hg^{2+} and 5 percent Hg(PM) (Peterson et al., 1995).

F.2.6.3 Selection of Model Plant Parameters

The lead smelter model plant consisted of a sintering machine, which feeds to a blast furnace and a dross furnace. The sintering machine is an updraft machine, which does not employ recirculation of the weak gas. Consequently, there are two off-gas streams, one strong SO_2 stream and a weak SO_2 stream. The strong stream is fed to a single-stage acid plant at a flow rate of 31,600 scm/hr (1,120,000 scf/hr);[1] the weak stream, with a flow rate of 138,000 scm/hr (4,860.000 scf/hr)[2] is sent to a FF for PM control and then emitted to the atmosphere. The off-gases from the dross furnace are fed to a fabric filter, at a flow rate of 72,200 scm/hr (2.550,000 scf/hr)[3] for PM control and then emitted to the atmosphere. The lead production capacity for this facility was assumed to be 90,900 Mg/yr (100,000 tons/yr) from concentrate containing 55 percent lead and 16 percent sulfur. The plant is assumed to have a 90 percent capacity factor. This corresponds to a daily capacity of 276 Mg/yr (304 tpd) (U.S. EPA, 1974).

The mercury emissions for both of the primary lead smelter off-gas streams were assumed to be uncontrolled resulting in an outlet mercury concentration of 1,000 ug/scm (436 gr/million scf) at 19 percent O_2 for each stream.

F.3 References

Bloom, Nicolas S., Eric M. Prestbo, Vesna L. Miklavcic, "Flue Gas Mercury Emissions and Speciation from Fossil Fuel Combustion," Second International Conference on Managing Hazardous Air Pollutants, Washington, D.C. July 1993.

Brna, T. G. Toxic Metal Emissions from MWCs and their Control, In Proceedings: 1991 International Conference on Municipal Waste Combustion, Volume 3, EPA-600/R-92-209c (NTIS PB93-124196). pp 23-39, November 1992.

Brna, T.G., J.D. Kilgroe, and C.A. Miller, Reducing Mercury Emission from Municipal Waste Combustion with Carbon Injection into Flue Gas, ECO World '92 Conference, Washington, DC, June 1992.

Brown, B. and K.S. Felsvang, Control of Mercury and Dioxin Emissions from United States and European Municipal Solid Waste Incinerators by Spray Dryer Absorption Systems, In Proceedings, 1991 International Conference on Municipal Waste Combustion, Volume 3, EPA600/R-92-209c (NTIS PB93-124196), pp 287-317, November 1992.

Buonicore, A.J., and W.T. Davis, , 1992, eds. Air Pollution Engineering Manual. Van Nostrand Reinhold, New York. 1992.

Edlund. H., 1993, Boliden Contech, telefax to K. Nebel, Radian Corporation. August 17, 1993.

Hall, B. Reactions of Mercury with Flue Gas Components, Statens Energiverk, National Energy Administration, Sweden, (STEV-FBT-91-18), 1991.

Hall, B., 0. Lindqvist, and D. Ljungstrom, Mercury Chemistry in Simulated Flue Gases Related to Waste Incineration Conditions, Environ. Sci. and Tech., 24 (1990), 108-111.

Hartenstein, H.U., 1993. Activated Carbon Filters for Flue Gas Polishing of MWI's. Presented at the International Conference on Municipal Waste Combustion. Williamsburg, Virginia, March 1993.

Hartenstein, H. U. Activated Carbon Filters for Flue Gas Polishing of MWI'S, In Proceedings 1993 Conference on Municipal Waste Combustion, Air & Waste Management Association (VIP32), Pittsburgh, PA, 1993, pp 87-105.

Kilgroe, J. D. et al., Camden County MWC Carbon Injection Test Results, Presented at the 1993 International Conference on Municipal Waste Combustion, Williamsburg, VA, March 30 to April 2, 1993.

Kiser, J. V. L. The IWSA Municipal Waste Combustion Directory: 1993 Update of U. S. Plants, Integrated Waste Services Association, Washington, DC, 1993.

Lerner, B.J., Beco Engineering Company, 1992. Mercury Emissions Control in Medical Waste Incineration. Presented at the 86th Annual Meeting, Air and Waste Management Association, Denver, Colorado. June 1992.

Lindqvist, O. and P. Schager, Continuous Measurements of Mercury in Flue Gases from Waste Incinerators and Combustion Plants, VDI Berichte, (NR. 838), 1990, pp. 401-421.

Metzger, M. and H. Braun, In-situ Mercury Speciation in Flue Gas by Liquid and Solid Sorption Systems, Chemosphere, 16 (1987), 821-832.

Midwest Research Institute, 1992. Medical Waste Incinerators -Background Information for Proposed Standards and Guidelines: Control Technology Performance Report for New and Existing Facilities. Draft Report. July 1992.

Nebel, K.L. et al., Emission Test Report: OMSS Field Test on Carbon Injection for Mercury Control, EPA-600/R-92-192 (NTIS PB93-105518), Air and Energy Engineering Research Laboratory, Research Triangle Park, NC, September 1992.

Pacyna, J. M. Anthropogenic Mercury Emission in Europe, Water, Air and Soil Pollution, 56 (1991), 51-61.

Peterson, G., A. Iverfeldt, J. Munthe, 1995. Atmospheric Mercury Species Over Central and Northern Europe, Model Calculations and Comparison with Observations from the Nordic Air and Precipitation Network for 1987 and 1988. GKSS Research Centre, Institue of Physics, Max-Planck-Str. 1, D-21502, Geesthacht, Germany.

TRC Environmental Corporation, 1992. Emission Characterization Program. Prepared for Copper Range Company, White Pine, Michigan. October 15, 1992.

U.S. Environmental Protection Agency, 1994. Medical Waste Incinerators - Background Information for Proposed Standards and Guidelines: Industry Profile Report for New and Existing Facilities. EPA-453/R-94-042a. Office of Air Quality Planning and Standards, Research Triangle Park, NC. July 1994.

U.S. Environmental Protection Agency, 1993. Locating and Estimating Air Emissions form Sources of Mercury and Mercury Compounds. EPA 454/R-93-023. Office of Air Quality Planning and Standards, Research Triangle Park, NC. September 1993.

U.S. Environmental Protection Agency, 1992a. Medical Waste Incinerators - Background Paper for New and Existing Facilities, Draft. U.S., Environmental Protection Agency, Research Triangle Park, NC. June 1992.

U.S. Environmental Protection Agency, 1992b. Medical Waste Incineration Emission Test Report -- Borgess Medical Center, Kalamazoo, Michigan. September 1992.

U.S. Environmental Protection Agency, 1991. Medical Waste Incineration Emission Test Report -- Morristown Memorial Hospital, Morristown, New Jersey. EMB Report 91-MWI-8. December 1991.

U.S. Environmental Protection Agency, 1989. Municipal Waste Combustors, Background Information for Proposed Standards, Post-Combustion Technology Performance, Volume 3, EPA450/-3-89-27c (NTIS PB90-154865), Research Triangle Park, NC, August 1989.

U.S. Environmental Protection Agency, 1988a. Compilation of Air Pollution Emission Factors, A.P-42, Fourth Edition, Supplement B. Office of Air Quality Planning and Standards, Research Triangle Park, NC.

U.S. Environmental Protection Agency, 1984. Review of National Emissions Standards for Mercury. EPA 450/-3-84-014. Office of Air Quality Planning and Standards, Research Triangle Park, NC. December 1984.

U.S. Environmental Protection Agency. 1974. Background Information for New Source Performance Standards: Primary Copper, Zinc and Lead Smelters. EPA-450/2-74-002a. Office of Air Quality Planning and Standards, Research Triangle Park, NC. October 1974.

U.S. Environmental Protection Agency, 1973. Control Techniques for Mercury Emissions from Extraction and Chlor-Alkali Plants. Office of Air Quality Planning and Standards, Research Triangle Park, NC. February 1973.

White, D. M. et al., Field Test of Carbon Injection for Mercury Control, Camden County Municipal Waste Combustor, EPA-600/R-93-181 (NTIS PB94-101540), September 1993.

White, D.M., K.L. Nebel. and M.G. Johnston, Municipal Waste Combustors: A Survey of Mercury Emissions and Applicable Control Technologies , In Proceedings, 1991 International Conference on Municipal Waste Combustion, Volume 3, EPA-600/R-92-209c (NTIS PB93124196), pp 247-257, November 1992.

White, D.M., et al., Parametric Evaluation of Activated Carbon Injection for Control of Mercury Emissions from a Municipal Waste Combustor, Paper No. 92-40.06, 1992 Annual Meeting, Air & Waste Management Association, Kansas City, MO, June 1992.

APPENDIX G

SUMMARY OF PREDICTED CONCENTRATIONS FOR ALL FACILITIES

Tables G-1 and G-2 provide summary results for precited concentrations in various media for all facilities.

Table G-1
Summary of Predicted Concentrations for Eastern Site

Eastern Site

Concentrations in Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ug/g)	Surface Water Concentration (ug/l)	Fruits	% MeHg	Leafy Veg	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Tier 3 Fish	Tier 4 Fish
Typical Deposition/Air Conc/Soil Conc	1.6E+0	1.0E+1	5.1E+1	1.0E+0	3.2E-2	5.1%	3.1E-2	21.7%	7.5E-3	20.5%	9.4E-3	20.6%	6.8E-5	3.5%	6.3E-6	19.2%	4.8E-2	2.4E-1
RELMAP 50th	1.7E+0	1.9E+1	3.7E+1	9.1E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.8E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1
RELMAP 90th	1.8E+0	4.0E+1	7.8E+1	1.9E+0	3.5E-2	5.1%	3.4E-2	21.4%	8.5E-3	19.8%	1.1E-2	20.0%	9.2E-5	3.1%	6.8E-6	19.3%	8.9E-2	4.5E-1
RELMAP Remote	1.7E+0	4.5E+1	8.8E+1	2.2E+0	3.4E-2	5.1%	3.3E-2	21.4%	8.3E-3	19.6%	1.0E-2	19.8%	1.0E-4	3.0%	6.7E-6	19.1%	1.0E-1	5.1E-1
2.5 km																		
Large Municipal Waste Combustor	2.5E-1	2.6E+2	5.1E+2	1.9E+1	7.3E-3	3.6%	8.3E-3	12.6%	6.3E-3	5.1%	7.1E-3	5.5%	5.1E-4	2.0%	3.6E-6	6.5%	8.7E-1	4.4E+0
Small Municipal Waste Combustor	1.1E-1	4.1E+1	8.0E+1	2.9E+0	2.5E-3	4.4%	2.6E-3	16.9%	1.3E-3	8.8%	1.5E-3	9.4%	8.0E-5	2.1%	8.2E-7	10.6%	1.4E-1	6.9E-1
Continuous Medical Waste Incinerator	2.3E-1	3.2E+1	6.2E+1	2.3E+0	4.9E-3	4.8%	5.0E-3	19.5%	1.7E-3	13.7%	2.0E-3	14.3%	6.4E-5	2.2%	1.2E-6	15.0%	1.1E-1	5.4E-1
Intermittent Medical Waste Incinerator	9.8E-3	1.1E+0	2.2E+0	8.0E-2	2.0E-4	4.9%	2.1E-4	19.8%	6.7E-5	14.6%	8.1E-5	15.1%	2.2E-6	2.3%	4.8E-8	15.7%	3.7E-3	1.9E-2
Large Coal-fired Utility Boiler	1.3E-3	8.5E+0	1.7E+1	5.7E-1	1.0E-4	1.7%	1.3E-4	4.1%	1.7E-4	2.2%	1.9E-4	2.3%	1.7E-5	2.0%	9.3E-8	2.8%	2.7E-2	1.3E-1
Medium Coal-fired Utility Boiler	2.3E-3	6.8E+0	1.3E+1	4.5E-1	1.0E-4	2.5%	1.3E-4	7.3%	1.4E-4	3.0%	1.6E-4	3.1%	1.3E-5	2.0%	7.8E-8	3.8%	2.1E-2	1.1E-1
Small Coal-fired Utility Boiler	2.6E-3	1.1E+0	2.2E+0	8.1E-2	6.1E-5	4.4%	6.5E-5	16.6%	3.4E-5	8.2%	3.9E-5	8.8%	2.2E-6	2.1%	2.1E-8	10.0%	3.8E-3	1.9E-2
Medium Oil-fired Utility Boiler	1.5E-4	2.0E-1	3.8E-1	1.4E-2	4.7E-6	3.4%	5.4E-6	11.5%	4.5E-6	4.4%	5.1E-6	4.8%	3.8E-7	2.0%	2.6E-9	5.7%	6.6E-4	3.3E-3
Chlor-alkali plant	3.8E+0	1.2E+2	2.3E+2	9.0E+0	7.6E-2	5.1%	7.6E-2	21.1%	1.9E-2	19.0%	2.4E-2	19.2%	2.6E-4	2.8%	1.5E-5	18.8%	4.2E-1	2.1E+0
Primary Copper Smelter	3.8E-2	1.9E+1	3.6E+1	1.3E+0	9.1E-4	4.3%	9.8E-4	16.4%	5.3E-4	7.9%	6.2E-4	8.5%	3.6E-5	2.1%	3.3E-7	9.6%	6.1E-2	3.1E-1
Primary Lead Smelter	4.2E-1	1.0E+2	2.0E+2	7.2E+0	9.0E-3	4.7%	9.4E-3	18.6%	3.8E-3	11.2%	4.5E-3	11.8%	2.0E-4	2.1%	2.6E-6	12.8%	3.4E-1	1.7E+0
10 km																		
Large Municipal Waste Combustor	2.2E-1	4.9E+1	9.5E+1	2.6E+0	4.8E-3	4.7%	5.0E-3	18.5%	2.0E-3	11.4%	2.3E-3	12.1%	9.6E-5	2.1%	1.3E-6	13.1%	1.2E-1	6.2E-1
Small Municipal Waste Combustor	6.4E-2	8.4E+0	1.6E+1	4.5E-1	1.3E-3	4.8%	1.4E-3	19.5%	4.6E-4	13.8%	5.5E-4	14.4%	1.7E-5	2.2%	3.2E-7	15.2%	2.1E-2	1.1E-1
Continuous Medical Waste Incinerator	8.2E-2	6.7E+0	1.3E+1	3.6E-1	1.7E-3	4.9%	1.7E-3	20.1%	5.0E-4	15.9%	6.2E-4	16.3%	1.4E-5	2.3%	3.7E-7	16.7%	1.7E-2	8.6E-2
Intermittent Medical Waste Incinerator	2.8E-3	2.1E-1	4.2E-1	1.2E-2	5.7E-5	5.0%	5.8E-5	20.2%	1.7E-5	16.1%	2.1E-5	16.5%	4.4E-7	2.4%	1.2E-8	16.9%	5.4E-4	2.8E-3
Large Coal-fired Utility Boiler	1.6E-3	2.0E+0	3.9E+0	1.1E-1	5.0E-5	3.5%	5.7E-5	11.7%	4.7E-5	4.5%	5.3E-5	4.9%	3.9E-6	2.0%	2.7E-8	5.8%	5.0E-3	2.5E-2
Medium Coal-fired Utility Boiler	2.7E-3	1.9E+0	2.6E+0	7.3E-2	6.5E-5	4.3%	7.0E-5	16.1%	3.8E-5	7.8%	4.4E-5	8.3%	2.6E-6	2.1%	2.4E-8	9.5%	3.4E-3	1.7E-2
Small Coal-fired Utility Boiler	2.0E-3	2.4E-1	4.6E-1	1.3E-2	4.2E-5	4.9%	4.2E-5	19.9%	1.4E-5	14.5%	1.7E-5	15.1%	4.8E-7	2.2%	9.8E-9	15.6%	5.9E-4	3.0E-3
Medium Oil-fired Utility Boiler	1.7E-4	3.9E-2	7.6E-2	2.1E-3	3.7E-6	4.7%	3.9E-6	18.6%	1.5E-6	11.4%	1.8E-6	12.0%	7.7E-8	2.1%	1.0E-9	13.0%	9.8E-5	4.9E-4
Chlor-alkali plant	8.4E-1	1.8E+1	3.4E+1	9.7E-1	1.7E-2	5.1%	1.6E-2	21.3%	4.1E-3	19.8%	5.1E-3	19.9%	4.1E-5	3.2%	3.2E-6	19.3%	4.5E-2	2.3E-1
Primary Copper Smelter	4.2E-2	4.0E+0	7.7E+0	2.1E-1	8.5E-4	5.0%	8.6E-4	20.3%	2.6E-4	15.6%	3.2E-4	16.1%	8.0E-6	2.3%	1.9E-7	16.1%	9.8E-3	4.9E-2
Primary Lead Smelter	3.9E-1	2.3E+1	4.4E+1	1.2E+0	7.7E-3	5.0%	7.8E-3	20.8%	2.1E-3	17.4%	2.7E-3	17.8%	4.7E-5	2.5%	1.6E-6	17.7%	5.5E-2	2.8E-1
25 km																		
Large Municipal Waste Combustor	1.2E-1	1.3E+1	2.4E+1	6.0E-1	2.4E-3	4.9%	2.5E-3	19.7%	7.7E-4	14.7%	9.4E-4	15.2%	2.5E-5	2.3%	5.6E-7	15.8%	2.8E-2	1.4E-1
Small Municipal Waste Combustor	2.8E-2	2.4E+0	4.6E+0	1.2E-1	5.7E-4	4.9%	5.8E-4	20.0%	1.7E-4	15.6%	2.1E-4	16.1%	4.9E-6	2.3%	1.3E-7	16.6%	5.4E-3	2.7E-2
Continuous Medical Waste Incinerator	2.9E-2	1.9E+0	3.6E+0	9.0E-2	5.9E-4	5.0%	6.0E-4	20.4%	1.7E-4	16.8%	2.1E-4	17.2%	3.9E-6	2.4%	1.3E-7	17.3%	4.2E-3	2.1E-2
Intermittent Medical Waste Incinerator	9.8E-4	5.7E-2	1.1E-1	2.8E-3	1.9E-5	5.0%	1.9E-5	20.5%	5.3E-6	16.9%	6.6E-6	17.3%	1.2E-7	2.5%	4.0E-9	17.5%	1.3E-4	6.5E-4
Large Coal-fired Utility Boiler	1.5E-3	4.5E-1	8.8E-1	2.1E-2	3.0E-5	4.5%	3.2E-5	17.4%	1.5E-5	9.4%	1.7E-5	10.0%	8.8E-7	2.1%	9.5E-9	11.1%	9.9E-4	5.0E-3
Medium Coal-fired Utility Boiler	2.2E-3	3.2E-1	6.3E-1	1.6E-2	4.6E-5	4.8%	4.7E-5	19.5%	1.6E-5	13.4%	2.0E-5	14.0%	6.5E-7	2.2%	1.1E-8	14.8%	7.3E-4	3.7E-3
Small Coal-fired Utility Boiler	9.7E-4	6.2E-2	1.2E-1	3.0E-3	2.0E-5	5.0%	2.0E-5	20.6%	5.6E-6	16.9%	6.9E-6	17.3%	1.3E-7	2.4%	4.2E-9	17.4%	1.4E-4	7.1E-4
Medium Oil-fired Utility Boiler	1.1E-4	9.8E-3	1.9E-2	4.7E-4	2.3E-6	5.0%	2.3E-6	20.2%	7.0E-7	15.8%	8.5E-7	16.2%	2.0E-8	2.3%	5.1E-10	16.6%	2.2E-5	1.1E-4

Table G-1 (continued)
Summary of Predicted Concentrations for Eastern Site

Eastern Site

Concentrations in Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Tier 3 Fish	Tier 4 Fish
Chlor-alkali plant	2.9E-1	4.3E+0	8.3E+0	2.1E-1	5.7E-3	5.1%	5.7E-3	21.4%	1.4E-3	20.3%	1.7E-3	20.4%	1.1E-5	3.5%	1.1E-6	19.6%	9.8E-3	4.9E-2
Primary Copper Smelter	2.7E-2	1.1E+0	2.1E+0	5.1E-2	5.3E-4	5.1%	5.3E-4	21.1%	1.4E-4	18.5%	1.7E-4	18.8%	2.3E-6	2.7%	1.1E-7	18.5%	2.4E-3	1.2E-2
Primary Lead Smelter	2.1E-1	6.1E+0	1.2E+1	2.9E-1	4.1E-3	5.1%	4.1E-3	21.2%	1.0E-3	19.2%	1.3E-3	19.4%	1.4E-5	2.9%	8.1E-7	18.9%	1.4E-2	7.0E-2
2.5 km																		
Large Municipal Waste Combustor and RelMap Eastern 50th percentile	1.9E+0	2.8E+2	5.4E+2	2.0E+1	4.0E-2	4.9%	4.1E-2	19.8%	1.4E-2	13.8%	1.7E-2	14.4%	5.6E-4	2.2%	1.0E-5	15.0%	9.1E-1	4.6E+0
Small Municipal Waste Combustor and RelMap Eastern 50th percentile	1.8E+0	6.0E+1	1.2E+2	3.8E+0	3.6E-2	5.1%	3.6E-2	21.2%	9.1E-3	19.0%	1.1E-2	19.3%	1.3E-4	2.8%	7.2E-6	18.8%	1.8E-1	9.1E-1
Continuous Medical Waste Incinerator and RelMap Eastern 50th percentile	1.9E+0	5.1E+1	9.9E+1	3.2E+0	3.8E-2	5.1%	3.8E-2	21.3%	9.5E-3	19.5%	1.2E-2	19.7%	1.1E-4	3.0%	7.5E-6	19.1%	1.5E-1	7.6E-1
Intermittent Medical Waste Incinerator and RelMap Eastern 50th percentile	1.7E+0	2.0E+1	3.9E+1	9.9E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.9E-3	20.7%	9.9E-3	20.7%	5.3E-5	3.8%	6.4E-6	19.8%	4.6E-2	2.3E-1
Large Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	2.7E+1	5.3E+1.	1.5E+0	3.3E-2	5.1%	3.3E-2	21.5%	8.0E-3	20.3%	1.0E-2	20.4%	6.8E-5	3.4%	6.4E-6	19.6%	6.9E-2	3.5E-1
Medium Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	2.6E+1	5.0E+1	1.4E+0	3.3E-2	5.1%	3.3E-2	21.5%	7.9E-3	20.4%	1.0E-2	20.5%	6.4E-5	3.5%	6.4E-6	19.6%	6.4E-2	3.2E-1
Small Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	2.0E+1	3.9E+1	9.9E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.9E-3	20.7%	5.3E-5	3.8%	6.4E-6	19.8%	4.6E-2	2.3E-1
Medium Oil-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.3E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.8E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1
Chlor-alkali plant and RelMap Eastern 50th percentile	5.5E+0	1.4E+2	2.7E+2	9.9E+0	1.1E-1	5.1%	1.1E-1	21.3%	2.7E-2	19.5%	3.4E-2	19.7%	3.1E-4	3.0%	2.2E-5	19.1%	4.6E-1	2.3E+0
Primary Copper Smelter and RelMap Eastern 50th percentile	1.7E+0	3.8E+1	7.3E+1	2.2E+0	3.4E-2	5.1%	3.4E-2	21.4%	8.3E-3	19.9%	1.0E-2	20.1%	8.7E-5	3.1%	6.7E-6	19.3%	1.0E-1	5.2E-1
Primary Lead Smelter and RelMap Eastern 50th percentile	2.1E+0	1.2E+2	2.3E+2	8.1E+0	4.2E-2	5.1%	4.2E-2	20.9%	1.2E-2	17.6%	1.4E-2	18.0%	2.5E-4	2.5%	8.9E-6	17.8%	3.8E-1	1.9E+0
10 km																		
Large Municipal Waste Combustor and RelMap Eastern 50th percentile	1.9E+0	6.8E+1	1.3E+2	3.5E+0	3.8E-2	5.1%	3.8E-2	21.2%	9.8E-3	18.9%	1.2E-2	19.1%	1.5E-4	2.7%	7.7E-6	18.7%	1.7E-1	8.4E-1
Small Municipal Waste Combustor and RelMap Eastern 50th percentile	1.8E+0	2.7E+1	5.3E+1	1.4E+0	3.4E-2	5.1%	3.4E-2	21.5%	8.2E-3	20.3%	1.0E-2	20.5%	6.8E-5	3.5%	6.7E-6	19.6%	6.4E-2	3.2E-1
Continuous Medical Waste Incinerator and RelMap Eastern 50th percentile	1.8E+0	2.6E+1	5.0E+1	1.3E+0	3.5E-2	5.1%	3.5E-2	21.5%	8.3E-3	20.4%	1.0E-2	20.5%	6.5E-5	3.6%	6.7E-6	19.6%	5.9E-2	3.0E-1
Intermittent Medical Waste Incinerator and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.2E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.9E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.4E-2	2.2E-1
Large Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	2.1E+1	4.1E+1	1.0E+0	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.6%	9.9E-3	20.7%	5.5E-5	3.8%	6.4E-6	19.8%	4.8E-2	2.4E-1
Medium Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	2.0E+1	3.9E+1	9.8E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.9E-3	20.7%	5.4E-5	3.8%	6.4E-6	19.8%	4.6E-2	2.3E-1
Small Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.2E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.9E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1
Medium Oil-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.1E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.8E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1

Table G-1 (continued)
Summary of Predicted Concentrations for Eastern Site

Eastern Site

Concentrations in Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg.	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Tier 3 Fish	Tier 4 Fish
Chlor-alkali plant and RelMap Eastern 50th percentile	2.5E+0	3.6E+1	7.1E+1	1.9E+0	5.0E-2	5.1%	4.9E-2	21.5%	1.2E-2	20.4%	1.5E-2	20.5%	9.2E-5	3.6%	9.6E-6	19.6%	8.8E-2	4.4E-1
Primary Copper Smelter and RelMap Eastern 50th percentile	1.7E+0	2.3E+1	4.5E+1	1.1E+0	3.4E-2	5.1%	3.4E-2	21.6%	8.0E-3	20.6%	1.0E-2	20.6%	5.9E-5	3.7%	6.5E-6	19.7%	5.2E-2	2.6E-1
Primary Lead Smelter and RelMap Eastern 50th percentile	2.1E+0	4.1E+1	8.1E+1	2.1E+0	4.1E-2	5.1%	4.1E-2	21.4%	9.9E-3	20.0%	1.2E-2	20.2%	9.8E-5	3.2%	8.0E-6	19.4%	9.8E-2	4.9E-1
25 km																		
Large Municipal Waste Combustor and RelMap Eastern 50th percentile	1.8E+0	3.1E+1	6.1E+1	1.5E+0	3.6E-2	5.1%	3.5E-2	21.5%	8.6E-3	20.2%	1.1E-2	20.3%	7.6E-5	3.4%	6.9E-6	19.5%	7.1E-2	3.6E-1
Small Municipal Waste Combustor and RelMap Eastern 50th percentile	1.7E+0	2.1E+1	4.1E+1	1.0E+0	3.4E-2	5.1%	3.4E-2	21.6%	8.0E-3	20.6%	1.0E-2	20.7%	5.6E-5	3.8%	6.5E-6	19.8%	4.8E-2	2.4E-1
Continuous Medical Waste Incinerator and RelMap Eastern 50th percentile	1.7E+0	2.1E+1	4.0E+1	1.0E+0	3.4E-2	5.1%	3.4E-2	21.6%	8.0E-3	20.6%	1.0E-2	20.7%	5.5E-5	3.8%	6.5E-6	19.8%	4.7E-2	2.4E-1
Intermittent Medical Waste Incinerator and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.1E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.8E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1
Large Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.8E+1	9.3E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.9E-3	20.8%	5.2E-5	3.9%	6.4E-6	19.8%	4.4E-2	2.2E-1
Medium Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.3E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.9E-3	20.8%	5.2E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1
Small Coal-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.1E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.8E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1
Medium Oil-fired Utility Boiler and RelMap Eastern 50th percentile	1.7E+0	1.9E+1	3.7E+1	9.1E-1	3.3E-2	5.1%	3.3E-2	21.6%	7.8E-3	20.7%	9.8E-3	20.8%	5.1E-5	3.9%	6.4E-6	19.8%	4.3E-2	2.2E-1
Chlor-alkali plant and RelMap Eastern 50th percentile	2.0E+0	2.5E+1	4.5E+1	1.1E+0	3.9E-2	5.1%	3.9E-2	21.6%	9.2E-3	20.7%	1.2E-2	20.7%	6.2E-5	3.8%	7.5E-6	19.8%	5.2E-2	2.6E-1
Primary Copper Smelter and RelMap Eastern 50th percentile	1.7E+0	2.0E+1	3.9E+1	9.6E-1	3.4E-2	5.1%	3.3E-2	21.6%	7.9E-3	20.7%	1.0E-2	20.8%	5.3E-5	3.8%	6.5E-6	19.8%	4.5E-2	2.4E-1
Primary Lead Smelter and RelMap Eastern 50th percentile	1.9E+0	2.5E+1	4.9E+1	1.2E+0	3.7E-2	5.1%	3.7E-2	21.5%	8.8E-3	20.6%	1.1E-2	20.6%	6.5E-5	3.7%	7.2E-6	19.7%	5.6E-2	2.8E-1
2.5 km																		
Large Municipal Waste Combustor and RelMap Eastern 90th percentile	2.0E+0	3.0E+2	5.8E+2	2.1E+1	4.2E-2	4.9%	4.3E-2	19.7%	1.5E-2	13.6%	1.8E-2	14.2%	6.0E-4	2.2%	1.0E-5	14.8%	9.6E-1	4.9E+0
Small Municipal Waste Combustor and RelMap Eastern 90th percentile	1.9E+0	8.1E+1	1.6E+2	4.8E+0	3.7E-2	5.1%	3.7E-2	21.1%	9.8E-3	18.4%	1.2E-2	18.7%	1.7E-4	2.6%	7.6E-6	18.4%	2.3E-1	1.1E+0
Continuous Medical Waste Incinerator and RelMap Eastern 90th percentile	2.0E+0	7.2E+1	1.4E+2	4.2E+0	3.9E-2	5.1%	3.9E-2	21.2%	1.0E-2	18.8%	1.3E-2	19.1%	1.6E-4	2.7%	8.0E-6	18.6%	2.0E-1	9.9E-1
Intermittent Medical Waste Incinerator and RelMap Eastern 90th percentile	1.8E+0	4.1E+1	8.0E+1	2.0E+0	3.5E-2	5.1%	3.5E-2	21.4%	8.5E-3	19.8%	1.1E-2	20.0%	9.5E-5	3.1%	6.9E-6	19.2%	9.3E-2	4.7E-1
Large Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1.8E+0	4.9E+1	9.5E+1	2.5E+0	3.5E-2	5.1%	3.5E-2	21.4%	8.7E-3	19.5%	1.1E-2	19.7%	1.1E-4	2.9%	6.9E-6	19.0%	1.2E-1	5.9E-1
Medium Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1.8E+0	4.7E+1	9.1E+1	2.4E+0	3.5E-2	5.1%	3.5E-2	21.4%	8.6E-3	19.6%	1.1E-2	19.8%	1.1E-4	3.0%	6.9E-6	19.1%	1.1E-1	5.6E-1

Table G-1 (continued)
Summary of Predicted Concentrations for Eastern Site

Eastern Site

Concentrations in Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ug/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Tier 3 Fish	Tier 4 Fish
Small Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1 8E+0	4 1E+1	8 0E+1	2 0E+0	3.5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19 8%	1 1E-2	20 0%	9.5E-5	3.1%	6 8E-6	19 2%	9 3E-2	4 7E-1
Medium Oil-fired Utility Boiler and RelMap Eastern 90th percentile	1 8E+0	4 0E+1	7 8E+1	1 9E+0	3 5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19 8%	1 1E-2	20 0%	9.3E-5	3 1%	6 8E-6	19 1%	9 0E-2	4 6E-1
Chlor-alkali plant and RelMap Eastern 90th percentile	5 6E+0	1 6E+2	3 1E+2	1 1E+1	1 1E-1	5 1%	1 1E-1	21 2%	2 8E-2	19 2%	3 5E-2	19 5%	3 6E-4	2 9%	2 2E-5	18 9%	5 1E-1	2 6E+0
Primary Copper Smelter and RelMap Eastern 90th percentile	1 8E+0	5 9E+1	1 1E+2	3 2E+0	3 5E-2	5 1%	3 5E-2	21 3%	9 0E-3	19 1%	1 1E-2	19 4%	1 3E-4	2 8%	7 1E-6	18 8%	1 5E-1	7 6E-1
Primary Lead Smelter and RelMap Eastern 90th percentile	2 2E+0	1 4E+2	2.7E+2	9 1E+0	4 4E-2	5 0%	4 4E-2	20 8%	1 2E-2	17 1%	1 5E-2	17 6%	2 9E-4	2 4%	9 4E-6	17 5%	4 2E-1	2 1E+0
10 km																		
Large Municipal Waste Combustor and RelMap Eastern 90th percentile	2 0E+0	8 9E+1	1 7E+2	4 5E+0	3 9E-2	5 1%	3 9E-2	21 0%	1 0E-2	18 2%	1 3E-2	18 6%	1 9E-4	2 6%	8 1E-6	18 3%	2 1E-1	1 1E+0
Small Municipal Waste Combustor and RelMap Eastern 90th percentile	1 8E+0	4 8E+1	9 4E+1	2 4E+0	3 6E-2	5 1%	3 6E-2	21 3%	8 9E-3	19 5%	1 1E-2	19 7%	1 1E-4	3 0%	7 1E-6	19 1%	1 1E-1	5 6E-1
Continuous Medical Waste Incinerator and RelMap Eastern 90th percentile	1.8E+0	4.7E+1	9 1E+1	2 3E+0	3 6E-2	5.1%	3.6E-2	21.4%	9 0E-3	19 6%	1 1E-2	19 8%	1 1E-4	3 0%	7 2E-6	19 1%	1 1E-1	5 4E-1
Intermittent Medical Waste Incinerator and RelMap Eastern 90th percentile	1 8E+0	4 0E+1	7 8E+1	1 9E+0	3 5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19 8%	1 1E-2	20 0%	9 3E-5	3 1%	6 8E-6	19 1%	9 0E-2	4 5E-1
Large Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1 8E+0	4 2E+1	8 2E+1	2 0E+0	3 5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19.7%	1 1E-2	19.9%	9 6E-5	3 0%	6 8E-6	19 2%	9 4E-2	4 8E-1
Medium Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1 8E+0	4 1E+1	8 1E+1	2 0E+0	3 5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19 8%	1 1E-2	20 0%	9 5E-5	3 1%	6 8E-6	19 2%	9 3E-2	4 7E-1
Small Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1 8E+0	4 0E+1	7 8E+1	1 9E+0	3 5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19 8%	1 1E-2	20 0%	9 3E-5	3 1%	6 8E-6	19 1%	9 0E-2	4 6E-1
Medium Oil-fired Utility Boiler and RelMap Eastern 90th percentile	1 8E+0	4 0E+1	7 8E+1	1 9E+0	3 5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19 8%	1.1E-2	20 0%	9 2E-5	3 1%	6 8E-6	19 1%	8 9E-2	4 5E-1
Chlor-alkali plant and RelMap Eastern 90th percentile	2 6E+0	5 8E+1	1 1E+2	2 9E+0	5 1E-2	5 1%	5 1E-2	21 4%	1 3E 2	19 8%	1 6E-2	20 0%	1 3E-4	3 1%	1 0E-5	19 1%	1 3E-1	6 8E-1
Primary Copper Smelter and RelMap Eastern 90th percentile	1 8E+0	4 4E+1	8 6E+1	2 1E+0	3 5E-2	5 1%	3 5E-2	21 4%	8 7E-3	19 7%	1 1E-2	19 9%	1 0E-4	3 0%	7 0E-6	19 2%	9 9E-2	5 0E-1
Primary Lead Smelter and RelMap Eastern 90th percentile	2 1E+0	6 3E+1	1 2E+2	3 1E+0	4 2E-2	5 1%	4 2E-2	21 3%	1 1E-2	19 3%	1 3E-2	19 6%	1 4E-4	2 9%	8 4E-6	19 0%	1 4E-1	7 3E-1
25 km																		
Large Municipal Waste Combustor and RelMap Eastern 90th percentile	1 9E+0	5 3E+1	1 0E+2	2 5E+0	3 7E-2	5 1%	3 7E-2	21 3%	9 3E-3	19 4%	1 2E-2	19 6%	1 2E-4	2 9%	7 4E-6	19 0%	1 2E-1	5 9E-1
Small Municipal Waste Combustor and RelMap Eastern 90th percentile	1 8E+0	4 2E+1	8 3E+1	2 0E+0	3 5E-2	5 1%	3 5E-2	21 4%	8 7E-3	19 7%	1 1E-2	19 9%	9 7E-5	3 1%	6 9E-6	19 2%	9 5E-2	4 8E-1
Continuous Medical Waste Incinerator and RelMap Eastern 90th percentile	1 8E+0	4 2E+1	8 2E+1	2 0E+0	3 5E-2	5 1%	3 5E-2	21 4%	8 6E-3	19 8%	1 1E-2	19 9%	9 6E-5	3 1%	6 9E-6	19 2%	9 4E-2	4 7E-1
Intermittent Medical Waste Incinerator and RelMap Eastern 90th percentile	1 8E+0	4 0E+1	7 8E+1	1 9E+0	3 5E-2	5 1%	3 4E-2	21 4%	8 5E-3	19 8%	1 1E-2	20 0%	9 3E-5	3 1%	6 8E-6	19 3%	8 9E-2	4 5E-1

Table G-1 (continued)
Summary of Predicted Concentrations for Eastern Site

Eastern Site

Concentrations in Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ug/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Tier 3 Fish	Tier 4 Fish
Large Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1.8E+0	4.0E+1	7.9E+1	1.9E+0	3.5E-2	5.1%	3.4E-2	21.4%	8.5E-3	19.8%	1.1E-2	20.0%	9.3E-5	3.1%	6.8E-6	19.3%	9.0E-2	4.6E-1
Medium Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1.8E+0	4.0E+1	7.9E+1	1.9E+0	3.5E-2	5.1%	3.4E-2	21.4%	8.5E-3	19.8%	1.1E-2	20.0%	9.3E-5	3.1%	6.8E-6	19.3%	9.0E-2	4.6E-1
Small Coal-fired Utility Boiler and RelMap Eastern 90th percentile	1.8E+0	4.0E+1	7.8E+1	1.9E+0	3.5E-2	5.1%	3.4E-2	21.4%	8.5E-3	19.8%	1.1E-2	20.0%	9.3E-5	3.1%	6.8E-6	19.3%	9.0E-2	4.5E-1
Medium Oil-fired Utility Boiler and RelMap Eastern 90th percentile	1.8E+0	4.0E+1	7.8E+1	1.9E+0	3.5E-2	5.1%	3.4E-2	21.4%	8.5E-3	19.8%	1.1E-2	20.0%	9.2E-5	3.1%	6.8E-6	19.3%	8.9E-2	4.5E-1
Chlor-alkali plant and RelMap Eastern 90th percentile	2.0E+0	4.4E+1	8.6E+1	2.1E+0	4.0E-2	5.1%	4.0E-2	21.4%	9.8E-3	19.9%	1.2E-2	20.1%	1.0E-4	3.1%	7.9E-6	19.3%	9.9E-2	5.0E-1
Primary Copper Smelter and RelMap Eastern 90th percentile	1.8E+0	4.1E+1	8.0E+1	2.0E+0	3.5E-2	5.1%	3.5E-2	21.4%	8.6E-3	19.8%	1.1E-2	20.0%	9.5E-5	3.1%	6.9E-6	19.3%	9.2E-2	4.6E-1
Primary Lead Smelter and RelMap Eastern 90th percentile	2.0E+0	4.6E+1	9.0E+1	2.2E+0	3.9E-2	5.1%	3.8E-2	21.4%	9.5E-3	19.8%	1.2E-2	19.9%	1.1E-4	3.1%	7.6E-6	19.2%	1.0E-1	5.2E-1

Table G-2
Summary of Predicted Concentrations for Western Site

Western Site

Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Troph 3 Fish	Troph 4 Fish
RELMAP 50th	1.6E+0	2.9E+0	5.8E+0	1.7E-1	3.2E-02	5.2%	3.1E-02	21.7%	7.2E-03	21.5%	9.1E-03	21.5%	2.0E-05	6.7%	5.9E-06	20.3%	8.5E-03	4.3E-02
RELMAP 90th	1.7E+0	1.1E+1	2.1E+1	5.6E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.4E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.1%	2.9E-02	1.4E-01
2.5 km																		
Large Municipal Waste Combustor	2.4E-1	9.5E+1	1.9E+2	7.1E+0	5.6E-03	4.4%	6.0E-03	16.5%	3.0E-03	8.4%	3.5E-03	9.0%	1.9E-04	2.1%	1.9E-06	10.2%	3.6E-01	1.8E+00
Small Municipal Waste Combustor	9.3E-2	1.8E+1	3.6E+1	1.3E+0	2.0E-03	4.7%	2.1E-03	18.6%	7.9E-04	11.9%	9.5E-04	12.5%	3.7E-05	2.2%	5.4E-07	13.6%	6.7E-02	3.4E-01
Continuous Medical Waste Incinerator	2.2E-1	2.0E+1	4.1E+1	1.6E+0	4.5E-03	4.9%	4.6E-03	19.9%	1.4E-03	15.2%	1.7E-03	15.7%	4.3E-05	2.3%	1.0E-06	16.2%	7.9E-02	4.0E-01
Intermittent Medical Waste Incinerator	9.3E-3	7.7E-1	1.5E+0	6.0E-2	1.9E-04	4.9%	2.0E-04	20.1%	5.8E-05	15.7%	7.1E-05	16.2%	1.6E-06	2.3%	4.3E-08	16.6%	3.1E-03	1.6E-02
Large Coal-fired Utility Boiler	2.0E-3	3.1E+0	6.2E+0	2.2E-1	6.8E-05	3.2%	8.0E-05	10.6%	7.2E-05	4.1%	8.0E-05	4.3%	6.2E-06	2.0%	4.0E-08	5.2%	1.1E-02	5.7E-02
Medium Coal-fired Utility Boiler	2.6E-3	2.3E+0	4.5E+0	1.6E-1	7.1E-05	3.8%	8.0E-05	13.5%	5.7E-05	5.5%	6.5E-05	5.9%	4.5E-06	2.0%	3.3E-08	7.0%	8.0E-03	4.0E-02
Small Coal-fired Utility Boiler	2.3E-3	4.3E-1	8.6E-1	3.2E-2	5.0E-05	4.8%	5.2E-05	19.0%	1.9E-05	12.2%	2.3E-05	12.9%	8.7E-07	2.2%	1.3E-08	13.8%	1.6E-03	8.2E-03
Medium Oil-fired Utility Boiler	1.5E-4	6.3E-2	1.3E-1	4.6E-3	3.5E-06	4.4%	3.8E-06	16.6%	1.9E-06	8.3%	2.3E-06	8.9%	1.3E-07	2.1%	1.2E-09	10.1%	2.3E-04	1.2E-03
Chlor-alkali plant	3.8E+0	1.0E+2	2.0E+2	8.2E+0	7.5E-02	5.1%	7.5E-02	21.2%	1.9E-02	19.3%	2.4E-02	19.5%	2.3E-04	2.9%	1.5E-05	19.0%	4.2E-01	2.1E+00
Primary Copper Smelter	4.4E-2	6.1E+0	1.2E+1	4.4E-1	9.2E-04	4.9%	9.4E-04	19.7%	3.2E-04	13.8%	3.9E-04	14.4%	1.3E-05	2.2%	2.3E-07	15.0%	2.2E-02	1.1E-01
Primary Lead Smelter	4.0E-1	3.5E+1	7.0E+1	2.6E+0	8.2E-03	5.0%	8.3E-03	20.4%	2.5E-03	15.8%	3.0E-03	16.3%	7.3E-05	2.3%	1.8E-06	16.6%	1.3E-01	6.8E-01
10 km																		
Large Municipal Waste Combustor	1.9E-1	2.8E+1	5.7E+1	1.7E+0	4.1E-03	4.8%	4.2E-03	19.2%	1.5E-03	13.2%	1.8E-03	13.8%	5.8E-05	2.2%	1.0E-06	14.7%	8.9E-02	4.5E-01
Small Municipal Waste Combustor	5.5E-2	5.5E+0	1.1E+1	3.4E-1	1.1E-03	4.9%	1.2E-03	19.8%	3.6E-04	14.9%	4.4E-04	15.5%	1.1E-05	2.3%	2.6E-07	16.0%	1.7E-02	8.7E-02
Continuous Medical Waste Incinerator	7.3E-2	5.2E+0	1.0E+1	3.2E-1	1.5E-03	5.0%	1.5E-03	20.3%	4.4E-04	16.3%	5.4E-04	16.7%	1.1E-05	2.4%	3.2E-07	17.0%	1.6E-02	8.3E-02
Intermittent Medical Waste Incinerator	2.5E-3	1.7E-1	3.3E-1	1.0E-2	5.0E-05	5.0%	5.1E-05	20.3%	1.5E-05	16.5%	1.8E-05	16.9%	3.5E-07	2.4%	1.1E-08	17.1%	5.3E-04	2.7E-03
Large Coal-fired Utility Boiler	1.8E-3	1.1E+0	2.2E+0	6.6E-2	4.6E-05	4.1%	5.0E-05	15.3%	3.0E-05	6.9%	3.4E-05	7.4%	2.2E-06	2.1%	1.8E-08	8.6%	3.4E-03	1.7E-02
Medium Coal-fired Utility Boiler	2.3E-3	6.1E-1	1.2E+0	3.9E-2	5.1E-05	4.6%	5.3E-05	18.1%	2.3E-05	10.5%	2.7E-05	11.1%	1.2E-06	2.1%	1.5E-08	12.2%	2.0E-03	1.0E-02
Small Coal-fired Utility Boiler	1.7E-3	1.3E-1	2.6E-1	8.0E-3	3.4E-05	5.0%	3.5E-05	20.4%	1.0E-05	16.2%	1.2E-05	16.6%	2.7E-07	2.4%	7.6E-09	16.9%	4.1E-04	2.1E-03
Medium Oil-fired Utility Boiler	1.4E-4	1.9E-2	3.7E-2	1.2E-3	3.0E-06	4.9%	3.1E-06	19.7%	1.0E-06	14.0%	1.2E-06	14.6%	3.8E-08	2.2%	7.3E-10	15.2%	5.9E-05	3.0E-04
Chlor-alkali plant	7.7E-1	1.6E+1	3.1E+1	1.0E+0	1.5E-02	5.1%	1.5E-02	21.3%	3.7E-03	19.8%	4.7E-03	19.9%	3.8E-05	3.2%	3.0E-06	19.3%	5.1E-02	2.6E-01
Primary Copper Smelter	3.5E-2	1.7E+0	3.4E+0	1.1E-1	7.0E-04	5.1%	7.0E-04	20.9%	1.9E-04	17.8%	2.3E-04	18.2%	3.7E-06	2.5%	1.5E-07	18.0%	5.5E-03	2.8E-02
Primary Lead Smelter	3.2E-1	1.1E+1	2.1E+1	6.5E-1	6.3E-03	5.1%	6.3E-03	21.1%	1.6E-03	18.9%	2.0E-03	19.1%	2.4E-05	2.8%	1.3E-06	18.7%	3.3E-02	1.7E-01
25 km																		
Large Municipal Waste Combustor	1.1E-1	1.1E+1	2.3E+1	6.3E-1	2.4E-03	4.9%	2.4E-03	19.8%	7.5E-04	14.9%	9.1E-04	15.5%	2.3E-05	2.3%	5.4E-07	16.0%	3.2E-02	1.6E-01
Small Municipal Waste Combustor	2.6E-2	2.1E+0	4.2E+0	1.2E-1	5.4E-04	4.9%	5.5E-04	20.1%	1.6E-04	15.8%	2.0E-04	16.2%	4.4E-06	2.3%	1.2E-07	16.7%	6.0E-03	3.0E-02
Continuous Medical Waste Incinerator	2.6E-2	1.7E+0	3.4E+0	9.5E-2	5.4E-04	5.0%	5.4E-04	20.4%	1.5E-04	16.7%	1.9E-04	17.1%	3.6E-06	2.4%	1.2E-07	17.3%	4.8E-03	2.4E-02
Intermittent Medical Waste Incinerator	8.3E-4	5.1E-2	1.0E-1	2.9E-3	1.7E-05	5.0%	1.7E-05	20.4%	4.8E-06	16.8%	5.9E-06	17.2%	1.1E-07	2.4%	3.6E-09	17.4%	1.5E-04	7.4E-04
Large Coal-fired Utility Boiler	1.5E-3	3.5E-1	7.0E-1	2.1E-2	3.2E-05	4.7%	3.4E-05	18.1%	1.4E-05	10.9%	1.6E-05	11.6%	7.1E-07	2.1%	9.1E-09	12.6%	1.1E-03	5.3E-03
Medium Coal-fired Utility Boiler	2.0E-3	2.2E-1	4.3E-1	1.2E-2	4.1E-05	4.9%	4.2E-05	19.9%	1.3E-05	14.7%	1.6E-05	15.3%	4.5E-07	2.3%	9.7E-09	15.8%	6.3E-04	3.2E-03
Small Coal-fired Utility Boiler	9.5E-4	5.0E-2	1.0E-1	2.8E-3	1.9E-05	5.0%	1.9E-05	20.7%	5.3E-06	17.4%	6.5E-06	17.8%	1.1E-07	2.5%	4.0E-09	17.8%	1.4E-04	7.3E-04
Medium Oil-fired Utility Boiler	1.1E-4	7.2E-3	1.4E-2	4.0E-4	2.1E-06	5.0%	2.2E-06	20.5%	6.2E-07	16.6%	7.6E-07	17.0%	1.5E-08	2.4%	4.6E-10	17.2%	2.1E-05	1.0E-04
Chlor-alkali plant	2.7E-1	4.6E+0	9.1E+0	2.6E-1	5.3E-03	5.1%	5.3E-03	21.4%	1.3E-03	20.1%	1.6E-03	20.2%	1.1E-05	3.4%	1.0E-06	19.5%	1.3E-02	6.8E-02
Primary Copper Smelter	2.5E-2	6.8E-1	1.4E+0	3.8E-2	4.9E-04	5.1%	4.9E-04	21.2%	1.2E-04	19.3%	1.5E-04	19.5%	1.6E-06	2.9%	9.8E-08	19.0%	2.0E-03	9.9E-03
Primary Lead Smelter	2.0E-1	4.2E+0	8.4E+0	2.4E-1	3.9E-03	5.1%	3.9E-03	21.3%	9.5E-04	19.8%	1.2E-03	19.9%	1.0E-05	3.1%	7.6E-07	19.3%	1.2E-02	6.1E-02

Table G-2 (continued)
Summary of Predicted Concentrations for Western Site

Western Site

Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Trophic 3 Fish	Trophic 4 Fish
2 5 km																		
Large Municipal Waste Combustor and RelMap Western 50th percentile	1.9E+0	9.8E+1	2 0E+2	7 2E+0	3 7E-02	5 0%	3 7E-02	20.9%	1 0E-02	17 6%	1 3E-02	18 0%	2 1E-04	2 5%	7 8E-06	17 9%	3 7E-01	1 9E+00
Small Municipal Waste Combustor and RelMap Western 50th percentile	1.7E+0	2.1E+1	4 2E+1	1 5E+0	3 4E-02	5.1%	3.3E-02	21 5%	7 9E-03	20 6%	1 0E-02	20 6%	5 6E-05	3 7%	6 5E-06	19 7%	7 6E-02	3 8E-01
Continuous Medical Waste Incinerator and RelMap Western 50th percentile	1 8E+0	2.3E+1	4 7E+1	1.7E+0	3 6E-02	5 1%	3 6E-02	21 5%	8 6E-03	20 5%	1 1E-02	20 6%	6 2E-05	3.7%	6 9E-06	19 7%	8 8E-02	4 4E-01
Intermittent Medical Waste Incinerator and RelMap Western 50th percentile	1.6E+0	3.7E+0	7.3E+0	2 3E-1	3.2E-02	5 2%	3 2E-02	21 7%	7.2E-03	21.5%	9.2E-03	21.4%	2.1E-05	6.4%	6.0E-06	20 1%	1.2E-02	5 9E-02
Large Coal-fired Utility Boiler and RelMap Western 50th percentile	1 6E+0	6.0E+0	1.2E+1	3 9E-1	3 2E-02	5 2%	3 1E-02	21 7%	7 2E-03	21 3%	9 2E-03	21 3%	2.6E-05	5 6%	6 0E-06	20 2%	2 0E-02	1 0E-01
Medium Coal-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	5.2E+0	1.0E+1	3 2E-1	3 2E-02	5 2%	3 1E-02	21 7%	7 2E-03	21 4%	9 2E-03	21 4%	2 4E-05	5 8%	6 0E-06	20 2%	1 6E-02	8 3E-02
Small Coal-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	3.3E+0	6 7E+0	2 0E-1	3.2E-02	5.2%	3 1E-02	21 7%	7 2E-03	21 5%	9 1E-03	21 5%	2 0E-05	6 5%	5 9E-06	20 1%	1 0E-02	5 1E-02
Medium Oil-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	3 0E+0	5 9E+0	1 7E-1	3 2E-02	5.2%	3.1E-02	21 7%	7 2E-03	21 5%	9 1E-03	21.5%	2 0E-05	6 7%	5 9E-06	20 1%	8 7E-03	4 4E-02
Chlor-alkali plant and RelMap Western 50th percentile	5.4E+0	1.0E+2	2 0E+2	8 3E+0	1 1E-01	5 1%	1 1E-01	21 3%	2 6E-02	19.9%	3 3E-02	20 0%	2 5E-04	3.2%	2 1E-05	19 4%	4 2E-01	2 1E+00
Primary Copper Smelter and RelMap Western 50th percentile	1.7E+0	9 0E+0	1 8E+1	6 0E-1	3 2E-02	5 2%	3.2E-02	21 7%	7 5E-03	21 2%	9.5E-03	21 2%	3 2E-05	5.0%	6 2E-06	20 1%	3 1E-02	1 6E-01
Primary Lead Smelter and RelMap Western 50th percentile	2 0E+0	3 8E+1	7 6E+1	2 8E+0	4 0E-02	5.1%	4 0E-02	21 4%	9 6E-03	20 0%	1 2E-02	20 2%	9 3E-05	3.2%	7 8E-06	19 4%	1 4E-01	7 2E-01
10 km																		
Large Municipal Waste Combustor and RelMap Western 50th percentile	1.8E+0	3.1E+1	6 2E+1	1 9E+0	3 6E-02	5 1%	3 6E-02	21 4%	8 6E-03	20 1%	1.1E-02	20 2%	7 7E-05	3 3%	6 9E-06	19 5%	9 7E-02	4 9E-01
Small Municipal Waste Combustor and RelMap Western 50th percentile	1.7E+0	8 1E+0	1 7E+1	5 0E-1	3 3E-02	5 2%	3 2E-02	21 6%	7 5E-03	21 2%	9 5E-03	21 2%	3 1E-05	5 1%	6 2E-06	20 1%	2 6E-02	1 3E-01
Continuous Medical Waste Incinerator and RelMap Western 50th percentile	1 7E+0	8.1E+0	1 6E+1	4 9E-1	3 3E-02	5 2%	3 3E-02	21 6%	7 6E-03	21 2%	9 6E-03	21 2%	3 0E-05	5 2%	6 2E-06	20 1%	2 5E-02	1 3E-01
Intermittent Medical Waste Incinerator and RelMap Western 50th percentile	1 6E+0	3 1E+0	6 1E+0	1 8E-1	3 2E-02	5 2%	3 1E-02	21 7%	7 2E-03	21 5%	9 1E-03	21 5%	2 0E-05	6 6%	5 9E-06	20 1%	9 0E-03	4 6E-02
Large Coal-fired Utility Boiler and RelMap Western 50th percentile	1 6E+0	4 0E+0	7 9E+0	2 3E-1	3 2E-02	5 2%	3 1E-02	21 7%	7 2E-03	21 5%	9 1E-03	21 4%	2 2E-05	6 3%	5 9E-06	20 1%	1 2E-02	6 0E-02
Medium Coal-fired Utility Boiler and RelMap Western 50th percentile	1 6E+0	3.5E+0	7 0E+0	2 1E-1	3 2E-02	5 2%	3 1E-02	21 7%	7 2E-03	21 5%	9.1E-03	21 5%	2 1E-05	6.5%	5.9E-06	20 1%	1 0E-02	5 3E-02
Small Coal-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	3 0E+0	6 1E+0	1 7E-1	3 2E-02	5 2%	3 1E-02	21 7%	7 2E-03	21 5%	9 1E-03	21 5%	2 0E-05	6 7%	5 9E-06	20 1%	8 9E-03	4 5E-02
Medium Oil-fired Utility Boiler and RelMap Western 50th percentile	1 6E+0	2 9E+0	5 8E+0	1 7E-1	3 2E-02	5 2%	3 1E-02	21 7%	7 2E-03	21 5%	9 1E-03	21 5%	2 0E-05	6 7%	5 9E-06	20 1%	8 6E-03	4 3E-02
Chlor-alkali plant and RelMap Western 50th percentile	2 4E+0	1.8E+1	3 7E+1	1 2E+0	4 7E-02	5 1%	4 7E-02	21 6%	1 1E-02	20 9%	1 4E 02	21 0%	5 7E-05	4 4%	8 9E-06	20 0%	6 0E-02	3 0E-01

Table G-2 (continued)
Summary of Predicted Concentrations for Western Site

Western Site

Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg.	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Trophic 3 Fish	Trophic 4 Fish
Primary Copper Smelter and RelMap Western 50th percentile	1.7E+0	4.6E+0	9.2E+0	2.7E-1	3.2E-02	5.2%	3.2E-02	21.7%	7.3E-03	21.4%	9.3E-03	21.4%	2.3E-05	6.1%	6.1E-06	20.2%	1.4E-02	7.1E-02
Primary Lead Smelter and RelMap Western 50th percentile	1.9E+0	1.3E+1	2.7E+1	8.2E-1	3.8E-02	5.2%	3.8E-02	21.6%	8.8E-03	21.0%	1.1E-02	21.1%	4.3E-05	4.6%	7.2E-06	20.0%	4.2E-02	2.1E-01
25 km																		
Large Municipal Waste Combustor and RelMap Western 50th percentile	1.7E+0	1.4E+1	2.8E+1	8.0E-1	3.4E-02	5.1%	3.4E-02	21.6%	7.9E-03	20.9%	1.0E-02	20.9%	4.3E-05	4.3%	6.5E-06	19.9%	4.1E-02	2.1E-01
Small Municipal Waste Combustor and RelMap Western 50th percentile	1.6E+0	5.0E+0	1.0E+1	2.8E-1	3.2E-02	5.2%	3.2E-02	21.7%	7.3E-03	21.4%	9.3E-03	21.4%	2.4E-05	5.9%	6.0E-06	20.2%	1.5E-02	7.3E-02
Continuous Medical Waste Incinerator and RelMap Western 50th percentile	1.6E+0	4.6E+0	9.2E+0	2.6E-1	3.2E-02	5.2%	3.2E-02	21.7%	7.3E-03	21.4%	9.3E-03	21.4%	2.3E-05	6.1%	6.0E-06	20.2%	1.3E-02	6.7E-02
Intermittent Medical Waste Incinerator and RelMap Western 50th percentile	1.6E+0	3.0E+0	5.9E+0	1.7E-1	3.2E-02	5.2%	3.1E-02	21.7%	7.2E-03	21.5%	9.1E-03	21.5%	2.0E-05	6.7%	5.9E-06	20.3%	8.6E-03	4.4E-02
Large Coal-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	3.3E+0	6.5E+0	1.9E-1	3.2E-02	5.2%	3.1E-02	21.7%	7.2E-03	21.5%	9.1E-03	21.5%	2.0E-05	6.6%	5.9E-06	20.3%	9.5E-03	4.8E-02
Medium Coal-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	3.1E+0	6.2E+0	1.8E-1	3.2E-02	5.2%	3.1E-02	21.7%	7.2E-03	21.5%	9.1E-03	21.5%	2.0E-05	6.6%	5.9E-06	20.3%	9.1E-03	4.6E-02
Small Coal-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	3.0E+0	5.9E+0	1.7E-1	3.2E-02	5.2%	3.1E-02	21.7%	7.2E-03	21.5%	9.1E-03	21.5%	2.0E-05	6.7%	5.9E-06	20.3%	8.6E-03	4.4E-02
Medium Oil-fired Utility Boiler and RelMap Western 50th percentile	1.6E+0	2.9E+0	5.8E+0	1.7E-1	3.2E-02	5.2%	3.1E-02	21.7%	7.2E-03	21.5%	9.1E-03	21.5%	2.0E-05	6.7%	5.9E-06	20.3%	8.5E-03	4.3E-02
Chlor-alkali plant and RelMap Western 50th percentile	1.9E+0	7.5E+0	1.5E+1	4.3E-1	3.7E-02	5.2%	3.7E-02	21.7%	8.4E-03	21.3%	1.1E-02	21.3%	3.1E-05	5.5%	7.0E-06	20.2%	2.2E-02	1.1E-01
Primary Copper Smelter and RelMap Western 50th percentile	1.6E+0	3.6E+0	7.2E+0	2.1E-1	3.2E-02	5.2%	3.2E-02	21.7%	7.3E-03	21.5%	9.2E-03	21.4%	2.1E-05	6.4%	6.0E-06	20.3%	1.0E-02	5.3E-02
Primary Lead Smelter and RelMap Western 50th percentile	1.8E+0	7.1E+0	1.4E+1	4.0E-1	3.5E-02	5.2%	3.5E-02	21.7%	8.1E-03	21.3%	1.0E-02	21.3%	3.0E-05	5.5%	6.7E-06	20.2%	2.0E-02	1.0E-01
2.5 km																		
Large Municipal Waste Combustor and RelMap Western 90th percentile	1.9E+0	1.1E+2	2.1E+2	7.6E+0	3.8E-02	5.0%	3.8E-02	20.8%	1.0E-02	17.5%	1.3E-02	17.8%	2.3E-04	2.5%	8.0E-06	17.7%	3.9E-01	2.0E+00
Small Municipal Waste Combustor and RelMap Western 90th percentile	1.7E+0	2.9E+1	5.7E+1	1.9E+0	3.4E-02	5.1%	3.4E-02	21.5%	8.2E-03	20.2%	1.0E-02	20.3%	7.2E-05	3.4%	6.7E-06	19.5%	9.6E-02	4.9E-01
Continuous Medical Waste Incinerator and RelMap Western 90th percentile	1.9E+0	3.1E+1	6.2E+1	2.1E+0	3.7E-02	5.1%	3.7E-02	21.4%	8.9E-03	20.2%	1.1E-02	20.3%	7.8E-05	3.4%	7.2E-06	19.5%	1.1E-01	5.5E-01
Intermittent Medical Waste Incinerator and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.3E+1	6.2E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.7E-05	4.6%	6.2E-06	20.0%	3.2E-02	1.6E-01
Large Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.4E+1	2.7E+1	7.8E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.0%	9.5E-03	21.0%	4.1E-05	4.3%	6.2E-06	20.0%	4.0E-02	2.0E-01
Medium Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.3E+1	2.6E+1	7.2E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.0%	9.5E-03	21.0%	4.0E-05	4.4%	6.2E-06	20.0%	3.7E-02	1.8E-01
Small Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.2E+1	5.9E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.6E-05	4.6%	6.1E-06	20.0%	3.0E-02	1.5E-01

Table G-2 (continued)
Summary of Predicted Concentrations for Western Site

Western Site

Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg.	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Trophic 3 Fish	Trophic 4 Fish
Medium Oil-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.1E+1	5.6E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.4E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.1%	2.9E-02	1.5E-01
Chlor-alkali plant and RelMap Western 90th percentile	5.4E+0	1.1E+2	2.2E+2	8.7E+0	1.1E-01	5.1%	1.1E-01	21.3%	2.6E-02	19.8%	3.3E-02	19.9%	2.7E-04	3.2%	2.1E-05	19.3%	4.4E-01	2.2E+00
Primary Copper Smelter and RelMap Western 90th percentile	1.7E+0	1.7E+1	3.3E+1	1.0E+0	3.3E-02	5.2%	3.3E-02	21.6%	7.8E-03	20.8%	9.8E-03	20.9%	4.8E-05	4.0%	6.4E-06	19.9%	5.1E-02	2.6E-01
Primary Lead Smelter and RelMap Western 90th percentile	2.1E+0	4.6E+1	9.2E+1	3.2E+0	4.0E-02	5.1%	4.0E-02	21.4%	9.9E-03	19.8%	1.2E-02	19.9%	1.1E-04	3.1%	8.0E-06	19.2%	1.6E-01	8.2E-01
10 km																		
Large Municipal Waste Combustor and RelMap Western 90th percentile	1.8E+0	3.9E+1	7.8E+1	2.3E+0	3.6E-02	5.1%	3.6E-02	21.4%	8.9E-03	19.8%	1.1E-02	20.0%	9.3E-05	3.1%	7.1E-06	19.9%	1.2E-01	5.9E-01
Small Municipal Waste Combustor and RelMap Western 90th percentile	1.7E+0	1.6E+1	3.2E+1	9.0E-1	3.3E-02	5.2%	3.3E-02	21.6%	7.8E-03	20.8%	9.9E-03	20.9%	4.6E-05	4.1%	6.4E-06	19.9%	4.6E-02	2.3E-01
Continuous Medical Waste Incinerator and RelMap Western 90th percentile	1.7E+0	1.6E+1	3.2E+1	8.8E-1	3.4E-02	5.2%	3.4E-02	21.6%	7.9E-03	20.8%	1.0E-02	20.9%	4.6E-05	4.1%	6.5E-06	19.9%	4.5E-02	2.3E-01
Intermittent Medical Waste Incinerator and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.2E+1	5.7E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.0%	2.9E-02	1.5E-01
Large Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.2E+1	2.3E+1	6.3E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.7E-05	4.5%	6.1E-06	20.0%	3.2E-02	1.6E-01
Medium Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.2E+1	6.0E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.6E-05	4.6%	6.1E-06	20.0%	3.1E-02	1.5E-01
Small Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.1E+1	5.7E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.0%	2.9E-02	1.5E-01
Medium Oil-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.1E+1	5.6E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.4E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.1%	2.9E-02	1.4E-01
Chlor-alkali plant and RelMap Western 90th percentile	2.4E+0	2.6E+1	5.2E+1	1.6E+0	4.8E-02	5.1%	4.7E-02	21.5%	1.1E-02	20.7%	1.4E-02	20.7%	7.3E-05	3.9%	9.1E-06	19.8%	8.0E-02	4.0E-01
Primary Copper Smelter and RelMap Western 90th percentile	1.7E+0	1.2E+1	2.5E+1	6.7E-1	3.3E-02	5.2%	3.3E-02	21.6%	7.6E-03	21.0%	9.7E-03	21.1%	3.9E-05	4.5%	6.3E-06	20.0%	3.4E-02	1.7E-01
Primary Lead Smelter and RelMap Western 90th percentile	2.0E+0	2.1E+1	4.2E+1	1.2E+0	3.9E-02	5.1%	3.8E-02	21.6%	9.1E-03	20.7%	1.1E-02	20.8%	5.9E-05	3.9%	7.4E-06	19.8%	6.2E-02	3.1E-01
25 km																		
Large Municipal Waste Combustor and RelMap Western 90th percentile	1.8E+0	2.2E+1	4.4E+1	1.2E+0	3.5E-02	5.1%	3.4E-02	21.5%	8.2E-03	20.5%	1.0E-02	20.6%	5.8E-05	3.7%	6.7E-06	19.7%	6.1E-02	3.1E-01
Small Municipal Waste Combustor and RelMap Western 90th percentile	1.7E+0	1.3E+1	2.5E+1	6.8E-1	3.3E-02	5.2%	3.3E-02	21.6%	7.6E-03	21.0%	9.6E-03	21.0%	4.0E-05	4.4%	6.2E-06	20.0%	3.5E-02	1.7E-01
Continuous Medical Waste Incinerator and RelMap Western 90th percentile	1.7E+0	1.2E+1	2.5E+1	6.5E-1	3.3E-02	5.2%	3.3E-02	21.6%	7.6E-03	21.0%	9.6E-03	21.0%	3.9E-05	4.5%	6.2E-06	20.0%	3.3E-02	1.7E-01
Intermittent Medical Waste Incinerator and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.1E+1	5.6E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.4E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.1%	2.9E-02	1.5E-01
Large Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.2E+1	5.8E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.6E-05	4.6%	6.1E-06	20.0%	3.0E-02	1.5E-01

Table G-2 (continued)
Summary of Predicted Concentrations for Western Site

Western Site

Biota (fish are ug/g wet weight, others are ug/g dry weight)

Plant	Watershed Air Concentration (ng/m3)	Watershed Total Deposition (ug/m2/yr)	Untilled Soil Concentration (ng/g)	Surface Water Concentration (ng/l)	Fruits	% MeHg	Leafy Veg	% MeHg	Beef	% MeHg	Milk	% MeHg	Poultry	% MeHg	Pork	% MeHg	Trophic 3 Fish	Trophic 4 Fish
Medium Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.2E+1	5.7E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.5E-03	21.1%	3.6E-05	4.7%	6.1E-06	20.0%	2.9E-02	1.5E-01
Small Coal-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.1E+1	5.6E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.4E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.1%	2.9E-02	1.5E-01
Medium Oil-fired Utility Boiler and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.1E+1	5.6E-1	3.2E-02	5.2%	3.2E-02	21.6%	7.5E-03	21.1%	9.4E-03	21.1%	3.5E-05	4.7%	6.1E-06	20.1%	2.9E-02	1.4E-01
Chlor-alkali plant and RelMap Western 90th percentile	1.9E+0	1.5E+1	3.0E+1	8.2E-1	3.8E-02	5.2%	3.7E-02	21.6%	8.7E-03	21.0%	1.1E-02	21.0%	4.6E-05	4.4%	7.2E-06	20.0%	4.2E-02	2.1E-01
Primary Copper Smelter and RelMap Western 90th percentile	1.7E+0	1.1E+1	2.3E+1	6.0E-1	3.3E-02	5.2%	3.3E-02	21.6%	7.6E-03	21.1%	9.6E-03	21.1%	3.7E-05	4.6%	6.2E-06	20.0%	3.0E-02	1.5E-01
Primary Lead Smelter and RelMap Western 90th percentile	1.8E+0	1.5E+1	3.0E+1	7.9E-1	3.6E-02	5.2%	3.6E-02	21.6%	8.4E-03	21.0%	1.1E-02	21.0%	4.5E-05	4.3%	6.9E-06	20.0%	4.1E-02	2.0E-01

APPENDIX H

ESTIMATION OF HUMAN METHYLMERCURY EXPOSURE TO THE GENERAL UNITED STATES POPULATION AND IDENTIFIED SUBPOPULATIONS THROUGH THE CONSUMPTION OF FISH

TABLE OF CONTENTS

Page

1 FISH CONSUMPTION AMONG THE GENERAL UNITED STATES POPULATION H-1
1.1 Patterns Of Fish Consumption H-1
1.1.1 1973 and 1974 National Purchase Diary Data H-4
1.1.2 Nationwide Food Consumption Survey of 1977-78 . . . . . H-7
1.1.3 CSFII/1989-1991 . . . . . H-8
1.2 Frequency of Consumption of Fish Based on Surveys of Individuals H-9
1.3 Consumption Rates Among Subpopulations with Potentially Higher Rates of Fish Consumption than the General Population H-12
1.3.1 Introduction H-12
1.3.2 U.S. Angling Population Size Estimate and Behaviors H-16
1.3.3 U.S. Angler Surveys H-17
1.3.4 Indigenous Populations of the United States H-21
1.4 Summary of Alaska and Hawaiian Fish Consumption Dat H-27
1.5 Mercury Concentrations In Fish H-30
1.5.1 Mercury Concentrations In Marine Fish H-30
1.5.2 Mercury Concentrations In Fresh-water Fish H-40

2. CALCULATION OF MERCURY CONCENTRATIONS IN FISH DISHES H-43
2.1 Additional Application Of Mean Mercury Concentrations In Fresh-water Fish . . H-44

3. INTAKE OF METHYLMERCURY FROM FISH/FISH DISHES H-44
3.1 Intakes "per User" and "per Capita" H-44
3.2 Fish Intake by Age and/or Gender Grouping of Subpopulations H-44
3.3 Types of Fish Consumed H-45
3.4 Methylmercury Consumption H-45

4. CONCLUSIONS ON METHYLMERCURY INTAKE FROM FISH H-64

5. ACKNOWLEDGMENTS H-67

6. REFERENCES H-67

7. ADDENDUM . . . . . H.A

LIST OF TABLES

		Page
H-1	Average Serving Size (gms) for Seafood from USDA Handbook # 11 Used to Calculate Fish Intake by FDA (1978)	H-5
H-2	Fish Species and Number of Persons Using the Species of Fish. (Adapted from Rupp et al., 1980)	H-6
H-3	Fish Consumption from the NPD 1973/1974 Survey (modified from Rupp et al., 1980)	H-6
H-4	Distribution of Fish Consumption for Females by Age* Consumption Category (grams/day) (from SRI (1980))	H-7
H-5	Number of Individuals in the 1989-1991 CSFII With Three-Days of Dietary Records	H-8
H-6	Respondents Reporting Consumption of All Fish and Shellfish in the 1989-1991 CSFII Survey Based on 3-Days Diet Records	H-8
H-7	Consumption of Fish and Shellfish (gms/day), and Self-Reported Body Weight (Kg) in Respondents of the 1989-1991 CSFII Survey. "Per Capita" Data for All Survey Respondents	H-11
H-8	Consumption of Fish and Shellfish (gms/day), and Self-Reported Body Weight (Kg) in Respondents of the 1989-1991 CSFII Survey	H-11
H-9	Compilation of the 11 Angler Consumption Studies	H-14
H-10	Fish Consumption by Native United States Populations	H-15
H-11	The Median Recreationally Caught Fish Consumption Rate Estimates by Ethnic Group (Puffer, 1981)	H-17
H-12	The Freshwater Fish Consumption Estimates of Turcotte (1983)	H-18
H-13	The Daily Intake of Sportfish and Total Fish for the Fish-consuming Portion of the Population Studied by Fiore et al., (1989)	H-18
H-14	Fish Consumption Rate Data for Groups Identified in Hovinga et al., 1992, as Eaters and Controls	H-20
H-15	Fish Consumption Rates for Maine Anglers	H-20
H-16	Fish Consumption Rates of Florida Anglers Who Receive Foodstamps	H-21
H-17	Fish Consumption by Columbia River Tribes, Columbia River Inter-Tribal Commission, 1994	H-24
H-18	Fish Consumption by Columbia River Tribes Columbia River Inter-Tribal Commission, 1994 Daily Fish Consumption Rates Among Adults	H-24
H-19	Fish Consumption (g/day) by the Tulalip and Squaxin Island Tribes (Toy et al. 1995)	H-25
H-20	Local Fish Meals Consumed By Time Period for the Mohawk and Comparison Nursing Mothers (Source: Fitzgerald et al., 1995)	H-26
H-21	Species Composition of Hawaii's Retail Seafood Trade, 1981 (purchases) as described by Higuchi and Pooley, 1985	H-29
H-22	Mercury Concentrations in Marine Finfish	H-31
H-23	Mercury Concentrations in Marine Shellfish	H-33
H-24	Mercury Concentrations in Marine Molluscan Cephalopods	H-33
H-25	Summary of Mercury Concentrations in Fish Species Micrograms Mercury per Gram Fresh Weight (ug Hg/g)	H-34
H-26	Fresh-water Fish Mercury Concentrations from Lowe et al., (1985)	H-41
H-27	U.S. EPA (1992) and Bahnick et al. (1994)	H-43
H-28	Consumption of Freshwater Fish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Fish methylmercury concentrations based on Bahnick et al., (1994)[a]	H-46

LIST OF TABLES (continued)

H-29 Consumption of All Fish & Shellfish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Bahnick et al estimates for fresh-water fish Methylmercury Concentrations H-47

H-30 Consumption of Fish & Shellfish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Lowe et al. estimates for fresh-water fish Methylmercury Concentrations H-48

H-31 Respondents Reporting Consumption of Marine Finfish (Excluding tuna, swordfish, barracuda, and shark) in the 1989-1991 CSFII Survey Based on 3-Days Diet Records . . H-49

H-32 Consumption (gms/day) of Marine Finfish (Excluding Tuna, Shark, Barracuda, and Swordfish) and Self-Reported Body Weight (Kg) among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only H-49

H-33 Respondents Reporting Consumption of Tuna in the 1989-1991 CSFII Survey Based on 3-Days Diet Records H-49

H-34 Consumption (gms/day) of Tuna Fish and Self-Reported Body Weight (Kg) among Respondents in the 1989-1991 CSFII Survey. Data for "Users" Only H-50

H-35 Respondents Reporting Consumption of Marine Shellfish in the 1989-1991 CSFII Survey Based on 3-Days Diet Records H-50

H-36 Consumption (gms/day) of Shellfish and Self-Reported Body Weights (Kg) of Respondents in the 1989-1991 CSFII Survey. Data for "Users" Only. H-50

H-37 Respondents Reporting Consumption of Shark, Barracuda, and/or Swordfish in the 1989-1991 CSFII Survey Based on 3-Days Dietary Records H-51

H-38 Consumption (gms/day) of Swordfish, Barracuda, and Shark and Self-Reported Body Weight (Kg) among Respondents in the 1989-1991 CSFII Survey. Data for "Users" Only H-51

H-39 Respondents Reporting Consumption of Fresh-water Fish in the 1989-1991 CSFII Survey Based on 3-Days Dietary Records H-51

H-40 Consumption (gms/day) of Fresh-water Fish and Self-Reported Body Weight (Kg) among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only H-52

H-41 Consumption of Freshwater Fish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Fish methylmercury concentrations based on Lowe et al., (1994) H-53

LIST OF FIGURES

	Page
H-1	H-13
H-2	H-54
H-3	H-55
H-4	H-56
H-5	H-57
H-6	H-58
H-7	H-59
H-8	H-60
H-9	H-61
H-10	H-62
H-11	H-63

1. FISH CONSUMPTION AMONG THE GENERAL UNITED STATES POPULATION

Consumption of fish is highly variable across the United States population unlike consumption of other dietary components, such as bread or starch dishes, that are almost ubiquitously consumed. Both marine and freshwater fish bioaccumulate methylmercury in their muscle tissues. Ingestion of methylmercury-contaminated fish tissue by humans results in exposure to this pollutant. This appendix presents an estimate of the magnitude of these exposures in the both the general fish-consuming U.S. population and in specific fish-consuming subpopulations (e.g., children and women of child-bearing age). This is not an estimate of background exposure to methylmercury for the general population but rather an estimate for only that part of the U.S. population which consumes fish. Use of a national data base differentiates data in this Appendix from a site-specific assessment. In this appendix estimates of methylmercury concentrations in freshwater fish are based on nation-wide survey results. Data presented in this Appendix differ from site-specific assessments in which consumption of contaminated local freshwater fish are included. Concentrations in fish are variable among water bodies and regions; thus, to do an assessment for a particular site or population, local measurements are required.

Inclusion of fish in the diet varies with geographic location, season of the year, ethnicity, and personal food preferences. Analysis of dietary survey data (Crochetti and Guthrie, 1982) showed fish to be among the most infrequently consumed food groups. Data on fish consumption have been calculated typically as either "per capita" or "per user". The former term is obtained by dividing the supply of fish across an entire population to establish a "per capita" consumption rate. The latter term divides the supply of fish across only the portion of the population that consumes fish, providing "per user" rates of consumption.

Identifying differences in fish consumption rates for population groups can be achieved through analysis of dietary survey data for the general United States population and specified subpopulations; e.g., some Native American tribes, recreational anglers, women of child-bearing age, and children. The United States Department of Agriculture (USDA) has conducted a series of nationally based dietary surveys including the 1977-1978 Nationwide Food Consumption Survey and the *Continuing Surveys of Food Intake by Individuals* over the period 1989 through 1991 (CSFII/89-91). Analyses of fish consumption patterns among the general United States population and selected age/gender groupings are described below. Fish consumption rate data from specific Native American tribes and angling populations are identified and used to corroborate the nation-wide fish-consumption data.

Fish bioaccumulate methylmercury through the freshwater aquatic and marine food-chains. Mercury-contaminated phytoplankton and zooplankton are consumed by planktivorous fish (referred to in other parts of this Volume as trophic level 3 fish). Methylmercury is thought to bioaccumulate in this group as well as in the piscivorous fish. As a result high fish concentrations of methylmercury indicative of bioaccumulation have been well documented. Consumption of these methylmercury-contaminated fish results in exposure to piscivorous human populations.

1.1 Patterns Of Fish Consumption

Despite the lower consumption frequency compared with staple foods such as grain products, dietary intake of fish can be estimated from dietary survey data. An initial question in how to estimate fish consumption is choice of dietary assessment methods. Available techniques include long-term dietary histories and questionnaires to identify typical food intake or short-term dietary recall techniques. The first consideration is to obtain dietary information that reflects typical fish

consumption. A true estimate of methylmercury intake from fish is complicated by changes in fish intake over time, differences in species of fish consumed and variation in the methylmercury concentration in a species of fish. Temporal variation in dietary patterns is an issue to consider in evaluation of short-term recall/record data. For epidemiological studies that seek to understand the relationship of long-term dietary patterns to chronic disease, typical food intake is the relevant parameter to evaluate (Willett, 1990). Because methylmercury is a developmental toxin that may produce adverse effects following a comparatively brief exposure period (i.e., a few months rather than decades), comparatively short-term dietary patterns can have importance. Consequently estimation of recent patterns of methylmercury consumption from fish is the relevant exposure for the health endpoint of concern.

In this Appendix the purpose is to describe the distribution of fish intakes for either the general population or for subpopulations defined by age or gender; e.g., women of child-bearing age. In the analysis of fish consumption data to estimate methylmercury intakes, the purpose is not to estimate fish consumption by an individual and relate it to an individual outcome. Dietary questionnaires or dietary histories may identify broad patterns of fish consumption, but these techniques provide less specific recollection of foods consumed such as the species of fish eaten. Likewise estimates of the quantity of fish consumed become less precise as the eating event becomes more remote in time. Selection of dietary survey methods to describe fish intakes by the subpopulation of interest requires a balancing of specificity of information collected with the generalizability of short-term dietary patterns to longer-term food intakes.

After selection of the appropriate period of fish intake to evaluate, a second major area of concern is variation in methylmercury concentration of the fish consumed. A central feature of food intakes among subjects with free-choice of foods is day-to-day variability superimposed on an underlying pattern of food intake (Willett, 1990, pg. 35). In epidemiology studies an individual's true intake of a food such as fish could be considered as the mean intake for a large number of days. Collectively the true intakes of these individuals define a frequency distribution for the study population as a whole (Willett, 1990, pg. 35). It is rarely possible to measure a large number of days of dietary intake for individual subjects; consequently a sample of one or several days is used to represent the true intake (Willett, 1990, pg. 35). The effect of this sampling is to increase the standard deviation artificially or, stated another way, broaden the tails of the distribution (Willett, 1990, pg. 35). This results in estimates of intake that are both larger and smaller than the true long-term averages for any subject. Overall, authorities in nutritional epidemiology (among other see Willett, 1990; pg. 50), conclude that "measurements of dietary intake based on a single or small number of 24-hour recalls per subject may provide a reasonable (unbiased) estimate of the mean of a group, but the standard deviation will be greatly overestimated."

Assessment of recent dietary intakes can be achieved through dietary records for various periods (typically 7-day records or 3-day records) or dietary recall (typically 24-hour recalls or 3-day recalls) (among other see Witschi, 1990). Research is currently in progress to estimate usual intake distributions that take into account properties exhibited by intake data for foods that are not consumed on a daily basis (among other see Nusser and Guenther, 1995). This remains, however, a data gap to be filled by additional research.

Sources of error in short-term recalls and records affect all dietary survey methodologies. These include errors made by the respondent or recorder of dietary information; the interviewer or reviewer. Information used to calculate the intake of the chemical of interest is another source of error. The chemical may be either nutritionally required (nutrients) or of interest because it is both

potentially toxic and not nutritionally required (toxicants). The third source of error in dietary assessments is the data base used to calculate intakes of the chemical from the food consumed.

The ability of the subject to remember the food consumed and in what quantities it was consumed is central to these methods (among many other see Witschi, 1990). In an analysis of data from the Health and Nutrition Evaluation Survey, the largest source of error was uncertainty of subjects about foods consumed on the recall day (Youland and Engle, 1976). Fish consumption appears to be more accurately remembered than most other food groups. Karvetti and Knuts (1985) observed the actual intake of 140 subjects and later interviewed them by 24-hour recall. They found that fish was omitted from the dietary recall less than 5% of the time and erroneously recalled approximately 7% of the time. The validity of 24-hour recalls for fish consumption was greater than all other food groups. Interviewer and reviewer errors can be reasonably predicted to be consistent for a given survey and unlikely to affect reporting of fish consumption selectively.

Estimates of Fish Intake for Populations

Data on fish consumption have been calculated typically as either "per capita" or "per user". The former term is obtained by dividing the supply of fish across an entire population to establish a "per capita" consumption rate. The latter term divides the supply of fish across only the portion of the population that consumes fish; i.e., "per user" rates of consumption.

Survey methods can broadly be classified into longitudinal methods or cross-sectional surveys. Typically long-term or longitudinal estimates of intake can be used to reflect patterns for individuals (e.g., dietary histories); or longitudinal estimates of moderate duration (e.g., month-long periods) for individuals or groups. Cross-sectional data are used to give a "snap shot" in time and are typically used to provide information on the distribution of intakes for groups within the population of interest. Cross-sectional data typically are for 24-hour or 3-day sampling periods and may rely on recall of foods consumed following questioning by a trained interviewer, or may rely on written records of foods consumed.

During the past decade reviewers of dietary survey methodology (for example, the Food and Nutrition Board of the National Research Council/National Academy of Sciences; the Life Sciences Research Office of the Federation of American Societies of Experimental Biology) have evaluated various dietary survey techniques with regard to their suitability for estimating exposure to contaminants and intake of nutrients. The Food and Nutrition Board of the National Research Council/National Academy of Sciences in their 1986 publication on Nutrient Adequacy Assessment Using Food Consumption Surveys noted that dietary intake of an individual is not constant from day to day, but varies both in amount and in type of foods eaten (intraindividual variation). Variations between persons in their usual food intake averaged over time is referred to as interindividual variation. Among North American populations, the intraindividual (within person day-to-day) variation is usually regarded to be as large as or greater than the interindividual (person to person) variations. Having evaluated a number of data sets the Academy's Subcommittee concluded that 3 days of observation may be more than is required for the derivation of the distribution of usual intakes.

Major sources of data on dietary intake of fish used in preparing this Report to Congress are the cross-sectional data from the USDA Continuing Surveys of Food Intake by Individuals conducted in the years 1989 through 1991 (CSFII 89/91) and the longer-term data on fish consumption based on recorded fish consumption for variable numbers of periods of one-month duration during the years

1973/1974 from the National Purchase Diary (NPD 73/74) conducted by the Market Research Corporation.

Currently a fish consumption rate of 6.5 grams per day is the default value used by some part of the U.S. EPA in the calculation of human health criteria. This value is based on data from the National Purchase Diary Survey conducted in the United States during the period 1973 and 1974. The overall fish consumption rate from this survey for fish-eaters was 14.3 g/day (U.S. EPA, 1989). This value was used in setting Ambient Water Quality Criteria (U.S. EPA 1980). This rate is a per capita rate averaged over the entire U.S. population including fish-eaters and nonfish-eaters. According to the NPD tabulation, fish consumers represented 94.0% of the entire U.S. population (SRI, 1980).

Identifying differences in fish consumption rates for population groups can be achieved through analysis of dietary survey data for the general United States population and specified subpopulations: e.g., some tribes of Native Americans, recreational anglers. The United States Department of Agriculture (USDA) has conducted a series of nationally based dietary surveys including the 1977-1978 Nationwide Food Consumption Survey and the Continuing Surveys of Food Intake by Individuals over the period 1989 through 1991 (CSFII 89-91). Analyses of fish consumption patterns among the general United States population are described below.

1.1.1 1973 and 1974 National Purchase Diary Data

The National Purchase Diary 1973/74 (NPD 73/74) data are based on a sample of 7,662 families (25,165 individuals) out of 9,590 families sampled between September 1973 and August 1974 (SRI International Contract Report to U.S. EPA, 1980; Rupp et al., 1980). The fish consumption was based on questionnaires completed by the female head of the household in which she recorded the date of any meal containing fish, the type of fish (species), the packaging of the fish (canned, frozen, fresh, dried, or smoked, or eaten out), whether fresh fish was recreationally caught or commercially purchased, the amount of fish prepared for the meal, the number of servings consumed by each family member and any guests, and the amount of fish not consumed during the meal. Meals eaten both at home and away from home were recorded. Ninety-four percent of the respondents reported consuming seafood during the sample period.

Use of these data to estimate intake of fish or mercury on a body weight basis are limited by the following data gaps.

1. This survey did not include data on the quantity of fish represented by a serving and information to calculate actual fish consumption from entries described as breaded fish or fish mixed with other ingredients. Portion size was estimated by using average portion size for seafood from USDA Handbook # 11, Table 10, page 40-41. The average serving sizes from this USDA source are shown in Table **H-1**.

Table H-1
Average Serving Size (gms) for Seafood from
USDA Handbook # 11 Used to Calculate
Fish Intake by FDA (1978)

Age Group (years)	Male Subjects (gms)	Female Subjects (gms)
0-1	20	20
1-5	66	66
6-11	95	95
12-17	131	100
18-54	158	125
55-75	159	130
Over 75	180	139

2. There may have been systematic under-recording of fish intake as Crispin-Smith et al. noted that typical intakes declined 30% between the first survey period and the last survey period among persons who completed four survey diaries (Crispin-Smith et al., 1985).

3. There have been changes in the quantities and types of fish consumed between 1973/1974 and present. The United States Department of Agriculture indicated (Putnam, 1991) that on average fish consumption increased 27% between 1970 to 1974 and 1990. This increase is also noted by the National Academy of Sciences in Seafood Safety (1991). Whether or not this increase applies to the highest percentiles of fish consumption (e.g., 95th or 99th percentile) was not described in the publication by USDA.

   Changes in the types of fish consumed have been noted. For example, Heuter et al. (1995) note that currently there is a much greater U.S. consumption of shark when compared to past decades.

4. Although an analysis of these data using the sample weights to project these data for the general United States population was prepared by SRI International under U.S. EPA Contract 68-01-3887 in 1980, U.S. EPA was subsequently informed that the sample weights were not longer available. Consequently additional analyses with these data in a manner than can be projected to the general population appears to no longer be possible.

5. Body weights of the individuals surveyed do not appear in published materials. If body weights of the individuals participating in this survey were recorded these data do not appear to have been used in subsequent analyses.

Data on fish consumption from the NPD 73/74 survey have been published by Rupp et al., 1980 and analyzed by U.S. EPA's contractor SRI International (1980). These data indicate that when a month-long survey period is used 94% of the surveyed population consumed fish. The species of fish most commonly consumed are shown in Table H-2.

Table H-2
Fish Species and Number of Persons Using the Species of Fish.
(Adapted from Rupp et al., 1980)

Category	Number of Individuals Consuming Fish Based on 24,652 Replies*
Tuna, light	16,817
Shrimp	5,808
Flounders	3,327
Not reported (or identified)	3,117
Perch (Marine)	2,519
Salmon	2,454
Clams	2,242
Cod	1,492
Pollock	1,466

* More than one species of fish may be eaten by an individual.

Rupp et al. also estimated quantities of fish and shellfish consumed by 12-18 year-old teenagers and by adults 18 to 98 years of age. These data are shown in Table H-3. The distribution of fish consumption for age groups that included women of child-bearing ages are shown in Table H-4.

Table H-3
Fish Consumption from the NPD 1973/1974 Survey
(modified from Rupp et al., 1980)

Age Group	50th Percentile	90th Percentile	99th Percentile	Maximum
Teenagers aged 12-18 Years	1.88 kg/year	8.66 kg/year	25.03 kg/year or 69 grams/day	62.12 kg/year
Adults aged 18 to 98 Years	2.66 kg/year	14.53 kg/year	40.93 kg/year or 112 grams/day	167.20 kg/year

Table H-4
Distribution of Fish Consumption for Females by Age*
Consumption Category (grams/day) (from SRI (1980))

Age (years)	47.6-60.0	60.1-122.5	over 122.5
10-19	0.2	0.4	0.0
20-29	0.9	0.9	0.0
30-39	1.9	1.7	0.1
40-49	3.4	2.1	0.2

* The percentage of females in an age bracket who consume, on average, a specified amount (grams) of fish per day. The calculations in this table were based upon the respondents to the NPD survey who consumed fish in the month of the survey. The NPD Research estimates that these respondents represent, on a weighted basis, 94.0% of the population of U.S. residents (from Table 6, SRI Report, 1980).

1.1.2 Nationwide Food Consumption Survey of 1977-78

Fish consumption is not evenly divided across the United States population. Analysis of patterns of fish consumption have been performed on data obtained from dietary surveys of nationally representative populations. For example, Crochetti and Guthrie (1982) analyzed the food consumption patterns of persons who participated in the Nationwide Food Consumption Survey of 1977/78. Populations specifically excluded from this analysis were children under four years of age, pregnant and nursing women, vegetarians, individuals categorized by race as "other" (i.e., not "white" and not "black"), individuals not related to other members of the household in which they lived, and individuals with incomplete records. After these exclusions, the study population consisted on 24,085 individual dietary records for a three-day period.

Persons reporting consumption of fish, shellfish, and seafood at least once in their 3-day dietary record were categorized as fish consumers. Combinations of fish, shellfish, or seafood with vegetables and/or starches (e.g., rice, pasta) or fish sandwiches were categorized as consumers of fish "combinations". Among the overall population, 25.0% of respondents reported consumption of fish with an additional 9.6% reporting consumption of fish "combinations" in the 3-day period for a total of 34.6% reporting consumption of fish and/or fish combinations. Frequency of consumption was comparable for male and female respondents with 24.1% of men and 25.7% of women reporting consumption of fish in their 3-day dietary records. Fish "combinations" were reported as dietary items by 11.2% of women and 9.9% of men. Both these food categories were consumed typically as mid-day and evening meals, rather than as breakfast or as snacks. For persons who listed fish in their 3-day dietary records, 89.7% listed fish in one meal only with 10.1% of respondents consuming fish in two meals and 0.1% consuming fish in three meals. For dishes that combined fish and other foods (i.e., fish "combinations"), among persons who reported eating fish combinations, 93.4% reported this food in one meal only with 6.5% of individuals consuming two meals containing fish "combinations."

There appears to be little difference between men and women in their likelihood of consuming fish based on patterns observed in this national survey (Crochetti and Guthrie, 1982). Based on this analysis, allocation of fish consumption on a "per capita" basis does not adequately reflect the fish consumption patterns of the general population of the United States. While "per capita" estimates

resulted in an overestimate of fish consumption for the approximately 65% of the United States population who did not report consuming fish. these types of estimates by their nature substantially underestimated fish consumption rates by persons who consume fish. This pattern of underestimation is important in an assessment of impact of infrequently consumed foods such as fish.

1.1.3 CSFII/1989-1991

The second set of nation-wide data (CSFII/89-91) are presented in Table H-5. Based on analysis of 11,706 respondents who supplied 3-days of dietary record in the USDA CSFII of 1989-1991, the frequency of fish consumption within the 3-day period was determined. Analyses of these dietary records indicate that 30.9% of respondents consumed fish, either alone or as part of a dish that contained fish. See Table H-6 for age/gender analyses of the fish-consuming population. Most respondents eating fish consumed one fish meal within the 3-day period. Two percent (2%) of respondents reported consuming fish two or more times during the 3-day period, and 0.5% of these fish-eating respondents reported fish consumption three or more times during the 3-day study period. Among persons who reported eating fish within the 3-day period of the survey, 44.1% reported eating marine finfish (other than or in addition to tuna, shark, barracuda, and swordfish). Marine finfish were more frequently consumed than fresh-water fish. Of the 1593 people who reported eating finfish, 492 (30.9%) identified these as fresh-water fish.

Table H-5
Number of Individuals in the 1989-1991 CSFII With Three-Days of Dietary Records

Gender	Aged 14 Years or Younger	Aged 15 through 44 Years	Aged 45 Years or Older	Total for All Age Groups
Males	1497 (51.7%)	2131 (42.9%)	1537 (40.0%)	5,165 (44.1%)
Females	1396 (48.3%)	2837 (57.1%)	2308 (60.0%)	6,541 (55.9%)
Total	2893 (24.7%)	4968 (42.4%)	3845 (32.8%)	11,706

Table H-6
Respondents Reporting Consumption of All Fish and Shellfish in the 1989-1991 CSFII Survey Based on 3-Days Diet Records
(Data weighted to be representative of the U.S. population.)

Gender	Aged 14 Years or Younger	Aged 15 through 44 Years	Aged 45 Years or Older	Total
Males	380 (52.8%)	646 (42.8%)	556 (39.3%)	1582 (43.8%)
Females	340 (47.2%)	864 (57.2%)	828 (58.5%)	2032 (56.2%)
Total	720 (19.9%)	1510 (41.8%)	1415 (39.2%)	3614

1.2 Frequency of Consumption of Fish Based on Surveys of Individuals

In the USDA 1989, 1990, and 1991 Continuing Surveys of Food Intake by Individuals (CSFII 89-91), food consumption data were obtained from nationally representative samples of individuals. These surveys included women of child-bearing age; that is, 15 through 44 years of age. Data from the CSFII for the period including 1989 and 1991 were used to calculate fish intake by the general population and women of child-bearing age. This subpopulation included pregnant women which are a subpopulation of interest in the Mercury Study: Report to Congress because of the potential developmental toxicity to the fetus accompanies ingestion of methylmercury. Analysis of *Vital and Health Statistics* data from 1990 indicated that 9.5% of women in this age group can be predicted to be pregnant in a given year. The size of this population has been estimated using the methodology described in the Addendum to this appendix, entitled "Estimated National and Regional Populations of United States Women of Child-Bearing Age."

The data described in this section were obtained from nationally representative samples of individuals and were weighted to reflect the U.S. population using the sampling weights provided by USDA. The basic survey was designed to provide a multistage stratified area probability sample representative of the 48 conterminous states. Weighting for the 1989, 1990 and 1991 data sets was done in two stages. In the first phase a fundamental sampling weight (the inverse of the probability of selection) was computed and the responding weight (the inverse of the probability of selection) was computed for each responding household. This fundamental sampling weight was then adjusted to account for non-response at the area segment level. The second phase of computations used the weights produced in the first phase as the starting point of a reweighting process that used regression techniques to calibrate the sample to match characteristics thought to be correlated with eating behavior.

The weights used in this analysis reflect CSFII individuals providing intakes for three days. Weights for the 3-day individual intake sample were constructed separately for each of the three gender-age groups: males ages 20 and over, females ages 20 and over and persons aged less than 20 years. Characteristics used in weight construction included day of the week, month of the year, region, urbanization, income as a percent of poverty, food stamp use, home ownership, household composition, race, ethnicity and age of the individual. The individual's employment status for the previous week was used for persons ages 20 and older, and the employment status of the female head of household was used for individuals less than 20 years of age. The end result of this dual weighting process was to provide consumption estimates which are representative of the U.S. population.

Respondents were drawn from stratified area probability samples of noninstitutionalized United States households. Survey respondents were surveyed across all four seasons of the year, and data were obtained across all seven days of the week. The dietary assessment methodology consisted of assessment of three consecutive days of food intake, measured through one 24-hour-recall and two 1-day food records. For this analysis, the sample was limited to those individuals who provided records or recalls of three days of dietary intake.

For purposes of interpretability, it should be noted that assessment of fish consumption patterns by recall/record assessment methods will probably differ from assessments based on food frequency methods. In order to be designated a consumer or "user" of fish for purposes of the present analysis, an individual would need to have reported consumption of one or more fish/shellfish products at some time during the three days in which dietary intake was assessed. Since fish is not a frequently consumed food for the majority of individuals, this dietary assessment method will likely underestimate the extent of fish consumption, since some individuals will be missed who normally

consume fish, but who did not consume fish during the three days of assessment. In contrast, such users would be picked up by a food frequency questionnaire. The recall/record dietary assessment method does have the advantage, however, of providing more precise estimates of the quantities of fish consumed that would be obtained with a food frequency record.

The information that follows comes from the CSFII/89-91 and was provided under contract to U.S. EPA by Dr. Pamela Haines of the Department of Nutrition of the University of North Carolina School of Public Health. Data are presented for following groups of individuals surveyed by USDA in the CSFII: data for the total population, data grouped by gender, and for data grouped by age-gender categories for the age groups 14 years or younger, 15 through 44 years, and 45 years and older (Table H-2).

Fish consumption was defined to reflect consumption of approximately 250 individual "Fish only" food codes and approximately 165 "Mixed dish-fish" food codes present in the 1994 version of the USDA food composition tables. The USDA maintains a data base (called the "Recipe File") that describes all food ingredients that are part of a particular food. Through consultation with Dr. Betty Perloff, an USDA expert in the USDA recipe file, and Dr. Jacob Exler, an USDA expert in food composition, the USDA recipe file was searched for food codes containing fish or shellfish. The recipe was then scanned to determine fish codes that were present in the recipe reported as consumed by the survey respondent. The percent of the recipe that was fish by weight was determined by dividing the weight of the fish/shellfish in the dish by the total weight of the dish.

As with most dietary assessment studies, multiple days of intake were averaged to reflect usual dietary intake better. Intakes reported over the three-day period were summed and then divided by three to provide consumption estimates on a per person, per day basis.

Fish consumption was defined within the following categories.

1. *Fish and Shellfish*, all types reflected consumption of any fish food code.
2. *Marine Finfish*, included fish not further specified (e.g., tuna) and processed fish sticks, as well as anchovy, cod, croaker, eel, flounder, haddock, hake, herring, mackerel, mullet, ocean perch, pompano, porgy, ray, salmon, sardines, sea bass, skate, smelt, sturgeon, whiting.
3. *Marine Shellfish* included abalone, clams, crab, crayfish, lobster, mussels, oysters, scallops, shrimp and snails.
4. *Tuna*, contained only tuna.
5. *Shark, Barracuda, and Swordfish* contained just these three species of fish.
6. *Fresh-water Fish* contained carp, catfish, perch, pike, trout and bass.

The analysis was stratified to reflect "per capita" (Table H-7), as well as "per user" (Table H-8), consumption patterns. A "consumer" of *Fish and Shellfish*, all types was one who consumed any of the included fish only or mixed-fish dish foods. A *Marine Finfish* consumer was one who consumed any of the species of fish included within the marine finfish category, and so on for each category. The percent of the population or subpopulation consuming fish was listed for the entire population, as well as gender specific values, and age-gender category specific values.

Table H-7
Consumption of Fish and Shellfish (gms/day), and Self-Reported Body Weight (Kg) in Respondents of the 1989-1991 CSFII Survey.
"Per Capita" Data for All Survey Respondents
(Data are weighted to be representative of the U.S. population.)

Gender	Aged 14 Years or Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	Kg_{bw}	Mean	SD	kg_{bw}
Males	8.6	19.8	26	18.6	34.5	73	20.2	36.4	90	16.7	32.5	68
Females	7.9	18.0	24	14.0	28.4	63	17.5	30.1	67	13.8	27.4	58

Table H-8
Consumption of Fish and Shellfish (gms/day), and Self-Reported Body Weight (Kg) in Respondents of the 1989-1991 CSFII Survey
(Data for "Users" Only. Data are weighted to be representative of the U.S. population.)

Gender	Aged 14 Years or Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	Kg_{bw}	Mean	SD	kg_{bw}
Males	31.8	26.6	27.7	53.7	39.3	80.2	51.4	42.0	82.9	48.8	39.1	58.6
Females	29.2	24.0	23.7	41.4	35.4	63.0	42.4	33.7	68.1	39.7	33.4	53.9

Consumption of fish-only and mixed-fish-dishes was summed across the three available days of dietary intake data. This sum was then divided by three to create average per day fish consumption figures. In the tables that describe fish intake, information is presented on sample size, percent of the population who consumed any product within the specified fish category, the mean grams consumed per day and the mean grams consumed per kilogram body weight (based on self-reported body weights), standard deviation, minimum, maximum, and the population intake levels at the 5th, 25th, 50th (median), 75th, and 95th percentiles of the intake distribution for each age-gender category. The means and standard deviations were determined using a SAS program. Survey sample weights were applied. Analysis with SAS does not take design effects into account, so the estimates of variance may differ from those obtained if SUDAAN or such packages had been used. It should be noted, however, that the point estimates of consumption (grams per consumer per day, grams per consumer per kilogram of body weight) will be exactly the same between the two statistical analysis packages. Thus, the point estimates reported are accurate and appropriate for interpretation on a national level.

Data were obtained for 11,706 individuals reporting 3-days of diet in the 1989-1991 CSFII survey. Analyses were based on data weighted through statistical procedures (as described previously) to be representative of the United States population. The total group of respondents reporting consumption of finfish and/or shellfish during the 3-day period were grouped as a subpopulation who consumed fish, as can be observed in Table H-6. Fish and shellfish (total fish consumption) were reported to be eaten by 3614 persons (30.9%) of the 11,706 of the survey respondents (see Tables H-5

and H-6). The subpopulation considered to be of greatest interest in this Mercury Study Report to Congress were women of child-bearing age (15 through 44 year-old females). Among this group of women ages 15 through 44 years, 864 women of the 2837 surveyed (30.5%) reported consuming fish (see Tables H-5 and H-6). Within this group, 334 women reported consumption of finfish during the 3-day survey period.

Consumption of fish and shellfish varied by species of fish. Overall, marine finfish (not including tuna, swordfish, barracuda, and shark) and tuna were consumed by more individuals and in greater quantity than were shellfish. Tuna fish was the most frequently consumed fish product, and separate tables are provided that identify quantity of tuna fish consumed. Two other categories of finfish were identified: fresh-water fish and a category comprised of swordfish, barracuda, and shark. Fresh-water fish were of interest because U.S. EPA's analysis of the fate and transport of ambient, anthropogenic mercury emissions from sources of concern in this report indicates that fish may bioaccumulate emitted mercury. Swordfish, barracuda, and shark were also identified as a separate category. These are predatory, highly migratory species that spend much of their lives at the high end of marine food web. These fish are large and accumulate higher concentrations of mercury than do lower trophic level, smaller fish.

1.3 Consumption Rates Among Subpopulations with Potentially Higher Rates of Fish Consumption than the General Population

1.3.1 Introduction

The purpose of this section is to document fish consumption rates among U.S. subpopulations thought to have higher rates of fish consumption. These subpopulations include residents of the States of Alaska and Hawaii and Native American Tribes; these groups were selected for analysis because of potentially elevated fish consumption rates rather than because they were thought to have a high innate sensitivity to methylmercury. The presented estimates are the results of fish consumption surveys conducted on the specific populations. The surveys use several different techniques and illustrate a broad range of consumption rates among these subpopulations. In several studies the fish consumption rates of the subpopulations corroborate the high-end (90th percentile and above) fish consumption estimates of the NPD survey (Rupp et al. 1980), the Nationwide Food Consumption Survey of 1977/78 and the Continuing Surveys of Food Intake by Individuals over the period 1989 through 1991 (CSFII/89-91). The consumption rates of fish-consumers are highlighted in the surveyed populations rather than estimates for the entire population, which would include non-consumers. Summary tables (Tables H-9 and H-10) listing daily fish consumption estimates are provided at the end of this section; selected values are shown graphically in Figure H-1.

Fish consumption data for Alaskans and Hawaiian islanders are included because these populations are not included in the nationwide surveys of the contiguous U.S. Members of these populations may exhibit elevated fish consumption rates as part of their ethnic heritage.

Analytic and survey methods to estimate the fish consumption rates of the respondents are described for each population. This Appendix does not constitute an exhaustive review of the methods employed. An attempt was made to characterize the population surveyed. Additionally, to characterize the entire range of fish consumption rates in the surveyed populations, the

Figure H-1
Distribution of Fish Consumption
Rates Among Various Populations

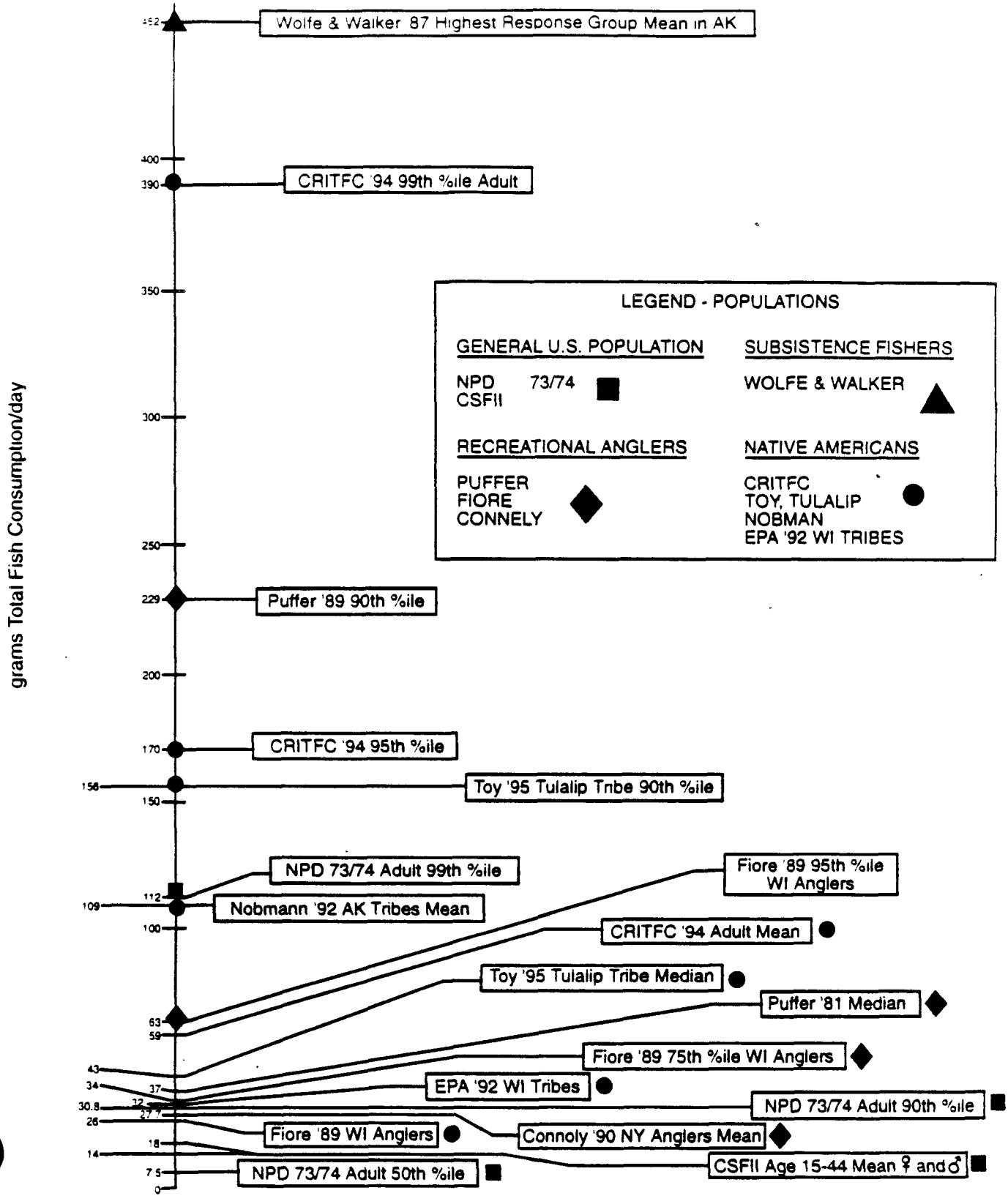


Table H-9
Compilation of the 11 Angler Consumption Studies

Source	Population	Percentile	Daily Fish Consumption g/day	Notes
Soldat, 1970	Columbia River Anglers	Mean	1.8	Estimate of average finfish consumption from river
Puffer 1981 as cited in U.S. EPA, 1990	Los Angeles area coastal anglers	Median 90th Percentile Ethnic Subpopulation Medians African-American Caucasian Mexican-American Oriental/Samoan	37 224.8 24.2 46 33 70.6	Estimates for anglers and family members who consume their catch. Consumption rate includes ingestion of both finfish and shellfish.
Pierce et al., 1981 as cited in EPA, 1990	Commencement Bay in Tacoma, WA	50th Percentile 90th Percentile Maximum Reported	23 54 381	Finfish only
Fiore et al., 1989	Licensed WI Anglers	Mean 75th Percentile 95th Percentile	12.3 15.5 37.3	Fish-Eaters, Daily Sportfish Intake
		Mean 75th Percentile 95th Percentile	26.1 34.2 63.4	Fish-Eaters, Total Fish Intake
West et al., 1989	Licensed MI Anglers	Mean Mean for Minorities Maximum Reported	19.2 21.7 >200	Daily Sportfish Intake
West et al., 1993	Licensed MI Anglers	Mean Mean for Minority Populations	14.5 43.1	Daily Sportfish intake
Turcotte, 1983	GA anglers	Child Teenager Average Angler Maximum Angler	10 23 31 58	Estimates of Freshwater Fish Intake from the Savannah River
Hovinga et al., 1992 and 1993	Caucasians living along Lake Michigan	Maximum Reported	132	Re-examination of Previously Identified High-End Fish Consuming Population
Ebert et al., 1993	ME anglers licensed to fish inland waters	Mean 50th Percentile 75th Percentile 90th Percentile 95th Percentile	6.4 2.0 5.8 13 26	Sportfish Intake
Sekerke et al, 1994	FL residents receiving foodstamps	Male Mean Female Mean	60 40	Total Home Fish Consumption

Table H-10
Fish Consumption by Native United States Populations

Source	Population	Percentile	Fish-Meals Consumed or Fish Consumption in Grams	Notes
Nobmann et al., 1992	351 Alaska Native Adults (Eskimos, Indians, Aleuts)	Mean	109 grams of fish and shell fish per day.	
U.S. EPA, 1992b	Wisconsin Tribes, 11 Native American Indian Tribes	Mean	31.5 grams of fish per day	
Peterson et al.	323 Chippewa Adults > 18 years of age.	Mean = 1.7 fish meals/week. (1.9 and 1.5 fish meals/week for male and for female respondents, respectively). 0.26% of males and 0.15% of females reported eating 3 or more fish-meals per week. 50% of respondents ate one or less fish meals per week. 21% of respondents ate three or more fish meals per week. 2% of respondents ate fish-meals each day.		
U.S. EPA, 1992	Wisconsin Tribes, 11 Native American Tribes	Mean	31.5 grams	
Toy et al., 1995	Tulalip and Squaxin Island Tribes, 263 adult subjects.	50th percentile: Finfish, 22.4 grams/day Total fish consumed, 42.6 grams/day. 90th Percentile: Finfish, 87.5 grams/day Total fish, 155.9 grams/day.		Report contains data for anadromous fish, pelagic, bottom and shell fish. Data are based on an average body weight of 70 kg/day.
Fitzgerald et al., 1995	97 Nursing Mohawk women	24.7% ate 1-9 local fish meals/year during pregnancy 10.3% ate >9 local fish meals/year during pregnancy 41.2% ate 1-9 local fish meals/year one year prior to pregnancy 15.4% ate >9 local fish meals/year one year prior to pregnancy		Study conducted from 1986-1992 in area where fish are contaminated with PCB

consumption rates of both average and high-end consumers as well as other specific angler subpopulations (e.g., fish consumption by angler race or age) are presented.

The sources of consumed fish are also identified in the summaries. Fish consumed by humans can be derived from many sources, these include self-caught, gift, as well as grocery and restaurant purchases. Some studies describe only the consumption rates for self-caught fish or freshwater fish, others estimate total fish consumption, and some delineate each source of fish. Humans also consume fish from many different types of water bodies (U.S. FWS, 1988). When described by the reporting authors, these are also identified.

Assumptions concerning fish consumption made by the study authors are also identified. Humans generally do not eat the entire fish; however, the species and body parts of fish which are consumed may be highly variable among angler populations (for example, see Toy et al. 1995). Anglers do not eat their entire catch, and, some species of fish are typically not eaten by specific angling subpopulations. For example, Ebert et al., (1993) noted that some types and parts of harvested fish are used as bait, fed to pets or simply discarded. Study authors account for the differences between catch weight and number in a variety of different ways. Typically, a consumption factor was applied. These assumptions impact the author's consumption rate estimates.

Data from angler and indigenous populations are useful in that they corroborate the ranges identified in the 3-day fish consumption data. The data are not utilized in this Report as the basis of a site-specific assessment. In a site-specific assessment the fish consumption rates among a surveyed population would be combined with specific measurements of methylmercury concentrations in the local fish actually consumed to estimate the human contact rate. Ideally, some follow-up analysis such as concentrations in human blood or hair would ensue.

1.3.2 U.S. Angling Population Size Estimate and Behaviors

Many citizens catch and consume fish from U.S. waters. The U.S. Fish and Wildlife Service (1988) reported that in 1985 26% of the U.S. population fished; over 46 million people in the U.S. spent time fishing during 1985. Within the U.S. population fishing rates ranged from a low of 17% for the population in the Middle Atlantic states up to 36% in the West North Central States. These angling subpopulations included both licensed and non-licensed fishers, hook and line anglers as well as those who utilized special angling techniques (e.g., bow and arrows, spears or ice-fishing).

U.S. Fish and Wildlife Service (1988) also noted the harvest and consumption of fish from water bodies where fishing is prohibited. This disregard or ignorance of fish advisories is corroborated in other U.S. angler surveys. For example, Fiore et al., (1989) noted that 72% of the respondents in a Wisconsin angler survey were familiar with the State of Wisconsin Fish Consumption Health Advisory, and 57% of the respondents reported changing their fishing or fish consumption habits based on the advisory. West et al., 1989 noted that 87.3% of respondents were "aware or generally aware" of Michigan State's fish consumption advisories. Finally, Connelly et al., 1990 reported that 82% of respondents knew about the New York State fish health advisories. They also noted a specific example in which angler consumption exceeded an advisory. The State of New York State recommends the consumption of no more than 12 fish meals/year of contaminated Lake Ontario fish species; yet, 15% of the anglers, who fished this Lake, reported eating more than 12 fish meals of the contaminated species from the Lake in that year.

1.3.3 U.S. Angler Surveys

Anglers of the Columbia River, Washington

Soldat (1970) measured fishing activity along the Columbia River during the daylight hours of one calendar year (1967-68). The average angler in the sampled population made 4.7 fishing trips per year and caught an average of 1 fish per trip. Assuming 200 g of fish consumed per meal, Soldat estimated an average of 0.7 fish meals were harvested per trip; this results in an average of 3.3 Columbia River fish meals/year. The product of 3.3 meals/year and 200 g/meal is 660 g/year; an estimate of 1.8 g/day results. While not reporting the high-end harvesting or consumption rates, Soldat reported that approximately 15% of the 1400 anglers interviewed caught 90% of the fish.

Los Angeles, California Anglers

The results of studies from Puffer (1981) and Pierce et al. (1981) are described in U.S. EPA's 1989. Puffer (1981) conducted 1,059 interviews with anglers in the coastal Los Angeles area for an entire year. Consumption rates were estimated for anglers who ate their catch. These estimates were based on angling frequency and the assumption of equal fish consumption among all fish-eating family members. The median consumption rate for fish and shellfish was 37 g/day. The 90th percentile was 224.8 g/day. Table H-11 notes the higher consumption rate estimates among Orientals and Samoans.

Table H-11
The Median Recreationally Caught Fish Consumption Rate Estimates by Ethnic Group (Puffer, 1981)

Ethnic Group	Median Consumption Rate (g/day)
African-American	24.2
Caucasian	46
Mexican-American	33
Oriental/Samoan	70.6
Total	**37**

Anglers of the Commencement Bay Area in Tacoma, Washington

Pierce et al., (1981), as reported in the U.S. EPA's 1990 Exposure Factors Handbook, conducted a total of 509 interviews in the summer and fall around Commencement Bay in Tacoma, WA. They assumed that 49% of the live fish weight was edible and that 98% of the total catch was eaten. The estimated 50th percentile consumption rate was 23 g/day and the estimated 90th percentile consumption rate 54 g/day. The maximum estimated consumption rate was 381 g/day based on daily angling.

Anglers of the Savannah River in Georgia

Turcotte (1983) estimated fish consumption from the Savannah River based on total harvest, population studies and a Georgia fishery survey (Table H-12). The angler survey data, which included the number of fishing trips per year as well as the number and weights of fish harvested per trip, were used to estimate the average consumption rate in the angler population. Several techniques including the use of the angler survey data were used to estimate the maximum fish consumption in the angler population. Estimates of average fish consumption for children and teens was also provided.

Table H-12
The Freshwater Fish Consumption Estimates of Turcotte (1983)

Georgia Subpopulation	Estimated Freshwater Fish Consumption Rate (g/day)
Child	10
Teen-ager	23
Average Angler	31
Maximum Angler	58

Wisconsin Anglers

Fiore et al., (1989) surveyed the fishing and fish consumption habits of 801 licensed Wisconsin anglers. The respondents were divided into 2 groups: fish eaters and non-eaters. The fish eaters group was further subdivided into 4 groups: those who consumed 0 - 1.8 Kg fish/yr, 1.9 - 4.5 Kg fish/yr, 4.6 - 10.9 Kg fish/yr and 10.9 <.Kg fish/yr. Using an assumption of 8 oz. (227 grams) fish consumed/meal, the authors estimated that the mean number of sport fish meals/year for all respondents (including non-eaters) was 18. The mean number of other fish meals/year including non-eaters was 24. The total number of fish meals/year was 41 for fish eaters and non-eaters combined and 42 for fish eaters only. Recreational anglers were found to consume both commercial fish as well as sport fish. The estimated daily consumption rates of the eaters-only are presented in Table H-13.

Table H-13
The Daily Intake of Sportfish and Total Fish for the Fish-consuming Portion of the Population Studied by Fiore et al., (1989)

Percentile	Daily Sport-Fish Intake	Daily Total Fish Intake
Mean	12.3 g/day	26.1 g/day
75th	15.5 g/day	34.2 g/day
95th	37.3 g/day	63.4 g/day

Michigan Anglers

West et al., 1989 used a mail survey to conduct a 7-day fish consumption recall study for licensed Michigan anglers. The respondents numbered 1104, and the response rate was 47.3%. The mean fish consumption rate for anglers and other fish-eating members of their households was 18.3 g/day, and the standard deviation was 26.8 g. Because the study was conducted from January through June, an off-season for some forms of angling in Michigan, higher rates of fish consumption would be expected during the summer and fall months. A full-year's mean fish consumption rate of 19.2 g/day was estimated from seasonal data. The mean fish consumption rate for minorities was estimated to be 21.7 g/day. The highest consumption rates reported were over 200 g/day; this occurred in 0.1% of the population surveyed. Overall, fish consumption rates increased with angler age and lower education levels. Lower income and education level groups were found to be the only group which consumed bottom-feeders.

New York State Anglers

Connelly et al., (1990) reported the results of a statewide survey of New York anglers. The 10,314 respondents (62.4% response rate) reported a mean of 20.5 days spent fishing/year. Of the respondents, 84% fished the inland waters of New York State, and 42% reported fishing in the Great Lakes. An overall mean of 45.2 fish meals per year was determined for New York anglers. The authors assumed an average meal size of 8 oz. (227 g) of fish and estimated a yearly consumption rate of 10.1 Kg fish (27.7 g fish/day). Unlike the Michigan angler study (West et al., 1989), the overall mean number of fish meals consumed increased with education level of the angler. Fish consumption also increased with increasing income; respondents earning more than $50,000/year consumed a mean of 54.3 meals per year, and those with some post-graduate education consumed a mean of 56.2 meals per year. The highest reported regional mean consumption rates (58.8 meals/year) occurred in the Suffolk and Nassau Regions of New York State.

Anglers of Lake Michigan

As part of a larger effort, Hovinga et al. (1992 and 1993) re-examined 115 eaters of Great Lakes Fish and 127 controls, who consumed smaller quantities of fish, originally identified in a 1982 effort. Both current (1989) as well as 1982 consumption rates of Great Lakes sportfish were estimated. All of the participants in the study were Caucasian and resided in 11 communities along Lake Michigan. The population was divided into eaters (defined as individuals consuming 10.9 kg (30 g/day) or greater) and controls (defined as individuals consuming no more than 2.72 kg/yr). The consumption rates for the groups are reported in Table H-14.

Table H-14
Fish Consumption Rate Data for Groups Identified in
Hovinga et al., 1992, as Eaters and Controls

Groups	1982 Meals/Yr Mean (Range)	1982 Consumption Rates (Kg/Yr) Mean (Range)	1989 Meals/Yr Mean (Range)	1989 Consumption Rates (Kg/Yr) Mean (Range)
Eaters	53.5 (24-132)	17.64 Kg/Yr (10.9-52.6)	38 (0-108)	9.8 (0-48)
Controls	-	-	4.1 (0-52)	0.73 (0-8.8)

Anglers of Inland Waters in the State of Maine

Ebert et al., (1993) examined freshwater fish consumption rates of 1,612 anglers licensed to fish the inland (fresh) waters of Maine. They only analyzed fish caught and eaten by the anglers. Anglers were asked to recall the number, species and average length of fish eaten in the previous year: the actual fish consumption rates were estimated based on an estimate of edible portion of the fish. The 78% of respondents who fished in the previous year and 7% who did not fish but did consume freshwater fish were combined for the analysis. Anglers who practiced ice-fishing as well as fish caught in both standing and flowing waters were included. Twenty-three percent of the anglers consumed no freshwater fish. If the authors assumed that the fish were shared evenly among all fish consumers in the angler's family, a mean consumption rate of 3.7 g/day was estimated for each consumer. Table H-15 provides the fish consumption rates for Maine anglers.

Table H-15
Fish Consumption Rates for Maine Anglers

Percentile	All Anglers	Fish-consuming anglers
Mean	5.0	6.4
50th (median)	1.1	2.0
75th	4.2	5.8
90th	11	13
95th	21	26

Florida Anglers Who Receive Foodstamps

As part of a larger effort the Florida Department of Environmental Regulation attempted to identify fish consumption rates of anglers who were thought to consume higher rates of fish. Face-to-face interviews were conducted at 5 Florida Foodstamp Distribution Centers. The selected Foodstamp Distribution Centers were located in counties either thought to have a high likelihood of subsistence anglers or where pollutant concentrations in fish were known. Interviews with twenty-five household's

primary seafood preparer were conducted at each center per quarter for an entire year. A total of 500 interviews was collected. The interviewed were asked to recall fish consumption within the last 7 days. Specifically, the respondents were asked to recall the species, sources and quantities of fish consumed. Note that the respondents were only asked to recall fish meals prepared at home (actual consumption rates may have been higher if the respondents consumed seafood elsewhere) and that the sources of fish were from both salt and freshwaters. The results of the survey conducted by Sekerke et al., 1994 are in Table H-16.

Table H-16
Fish Consumption Rates of Florida Anglers Who Receive Foodstamps

Respondent	No.	Average Finfish Consumption	Average Shellfish Consumption
Adult Males	366	60 g/day	50 g/day
Adult Females	596	40 g/day	30 g/day

Summary of Angler Surveys

The results of the 10 fish consumption surveys are compiled in Table H-9. These results illustrate the range of fish consumption rates identified in angler consumption surveys. There is a broad range of fish consumption rates reported for angling populations. The range extends from 2 g/day to greater than 200 g/day. The variability is the result of differences in the study designs and purposes as well as differences in the populations surveyed.

1.3.4 Indigenous Populations of the United States

The tribes and ethnic groups who comprise the indigenous populations of the United States show wide variability in fish consumption patterns. Although some tribes, such as the Navajo, consume minimal amounts of fish as part of their traditional culture, other native groups, (such as the Eskimos, Indians, and Aleuts of Alaska or the tribes of Puget Sound) traditionally consume high quantities of fish and fish products. The United States' indigenous populations are widely distributed geographically. For example, a U.S. EPA Report (1992b) identified 281 Federal Indian reservations that cover 54 million acres in the United States. Treaty rights to graze livestock, hunt, and fish are held by native peoples for an additional 100 to 125 million acres. There are an estimated two million American Indians in the United States (U.S. EPA, 1992b). Forty-five percent of these two million native people live on or near reservations and trust lands. High-end fish consuming groups include Alaska natives who number between 85,000 and 86,000 people (Nobmann et al., 1992).

Fish products consumed by indigenous populations may rely on preparation methods that differ from ones typically encountered in the diet of the general United States population. By way of illustration, food intake data obtained from Alaskan natives were used to calculate nutrient intakes using a computer and software program. These computerized databases had been developed by the United States Veterans Administration for patients in the national Veteran's Administration hospital system. Nobmann et al. (1992) found they needed to add data for 210 dietary items consumed by Alaskan Natives to the 2400 food items in the VA files.

In the mid-1990s data on fish consumption by indigenous populations of the United States were reported for Alaska Natives (Nobmann et al., 1992), Wisconsin Tribes (U.S. EPA, 1992), the Columbia River Tribes (Columbia River Inter-Tribal Fish Commission, 1994) and selected Puget Sound Tribes (Toy et al. 1995). Findings from these studies can be used to assess differences in fish consumption between these indigenous groups and the general United States population.

Alaskan Natives

Dietary analyses on seasonal food intakes of 351 Alaska Native adults from 11 communities were performed during 1987-1988 (Nobmann et al., 1992). Alaska Natives include Eskimos, Indians and Aleuts. There is no main agricultural crop in Alaska which, combined with a short growing season, results in limited availability of edible plants. Alaska Natives have traditionally relied on a diet of fish, sea mammals, game and a few native plants (seaweed, willow leaves, and sourdock) and berries (such as, blueberries and salmonberries). Although consumption of significant amounts of commercially produced foods occurs, use of subsistence foods continues. .

The survey sample of 351 adults, aged 21 - 60 years, was drawn from 11 communities. Information was obtained using 24-hour dietary recalls during five seasons over an 18-month period. Fish were consumed much more frequently by Alaska Natives than by the general United States population. Fish ranked as the fourth most frequently consumed food by Alaska Natives compared with the 39th most frequently consumed food by participants in the nationally representative Second National Health and Nutrition Assessment Survey (NHANES II). The mean daily intake of fish and shellfish for Alaska Natives was 109 grams/day contrasted with an intake of 17 grams per day for the general United States population described in NHANES II. Among Alaska Natives fish was consumed more frequently in the summer and fall months.

Despite a degree of acculturation in the area of foods, native foods were still eaten frequently by Alaskan Native peoples based on results of the 1987-1988 survey. Diets that include major quantities of fish (especially salmon) and sea mammals retain a major place in the lives of Alaskan Native peoples. The consumption of traditional preparations of salmon and other fish continues: this includes fermented foods such as salmon heads and eggs, other fish and their eggs, seal, beaver, caribou and whale.

Alaskans from Subsistence Economies

Wolfe and Walker (1987) described the productivity and geographic distribution of subsistence economies in Alaska during the 1980s. Based on a sample of 98 communities, the economic contributions of harvests of fish, land mammals, marine mammals and other wild resources were analyzed. Noncommercial fishing and hunting play a major role in the economic and social lives of persons living in these communities. Harvest sizes in these communities were established by detailed retrospective interviews with harvesters from a sample of households within each community. Harvests were estimated for a 12 month period. Data were collected in pounds of dressed weight per capita per year. Although it varies by community and wildlife species, generally "dressed weight" is approximately 70 to 75% of the round weight for fish and 20 to 60% of round weight for marine animals. Dressed weight is the portion of the kill brought into the kitchen for use, including bones for particular species. The category "fish" contains species including salmon, whitefish, herring, char, halibut, and pike. "Land mammals" included species such as moose, caribou, deer, black bear, snowshoe and tundra hare, beaver and porcupines. "Marine mammals" consisted of seal, walrus and whale. "Other" contained birds, marine invertebrates, and certain plant products such as berries.

Substantial community-to-community variability in the harvesting of fish, land mammals, marine mammals and other wild resources were noted (Wolfe and Walker, 1985). Units are pounds "dressed weight" per capita per year. The median harvest was 252 pounds with the highest value approximately 1500 pounds. Wild harvests (quantities of fish,land mammals and marine mammals) in 46% of the sampled Alaskan communities exceeded the western United States consumption of meat, fish, and poultry. These communities have been grouped by general ecological zones which correspond to historic/cultural areas. Arctic-Subarctic Coast, Aleutian-Pacific Coast, Subarctic Interior, Northwest Coast and contemporary urban population centers. The Arctic-Subarctic Coast displayed the greatest subsistence harvests of the five ecological zones (610 pounds per capita), due primarily to the relatively greater harvests of fish and marine animals. For all regions the fishing output is greater than the hunting; fishing comprises 57 - 68% of total subsistence output. Above 60° north latitude fishing predominates other wildlife harvests, except for the extreme Arctic coastal sea mammal-caribou hunting communities. Resource harvests of fish ("dressed weight" on a per capita basis) by ecological zone (and cultural area) were these: Arctic-Subarctic Coast (Inupiaq-Yup'ik), 363 pounds/year or 452 grams/day; Aleutian-Pacific Coast (Aleut-Sugpiaq), 251 pounds/year or 312 grams/day; Subarctic Interior (Athapaskan), 256 pounds/year or 318 grams/day; Northwest Coast (Tingit-Haida), 122 pounds/year or 152 grams/day; and Other (Anchorage, Fairbanks, Juneau, Matanuska-Susitna Borough, and Southern Cook Inlet), 28 pounds/year or 35 grams/day.

Wisconsin Tribes

The U.S. EPA's 1992 document entitled *Tribes at Risk* (The Wisconsin Tribes Comparative Risk Project) reported an average total daily fish intake for Native Americans living in Wisconsin of 35 grams/day. The average daily intake of locally harvested fish was 31.5 grams.

Peterson et al. (1995) surveyed 323 Chippewa adults over 18 years of age living on the Chippewa reservation in Wisconsin. The survey was conducted by interview and included questions about season, species and source of fish consumed. The survey was carried out in May. Fish consumption was found to be seasonal with the highest fish consumption occurring in April and May. Fish species typically consumed were walleye and northern pike, muskellunge and bass. During the months in which the Chippewa ate the most fish, 50% of respondents reported eating one or fewer fish meals per week, 21% reported eating three or more fish meals per week, and 2% reported daily fish consumption. The mean number of fish meals per week during the peak consumption period was 1.7 meals; this is approximately 42% higher than the 1.2 fish meals per week that respondents reported as their usual fish consumption. Higher levels of fish consumption were reported by males (1.9 meals per week) than by females (1.5 meals per week). Among male respondents 0.26% ate 3 or more fish meals per week, whereas 0.15% of female respondents ate 3 or more meals of fish per week. Unemployed persons typically had higher fish consumption rates.

Columbia River Tribes

The Columbia River Inter-Tribal Fish Commission (1994) estimated fish consumption rates for members of four tribes inhabiting the Columbia River Basin. The estimated fish consumption rates were based on interviews with 513 adult tribe members who lived on or near the reservation. The participants had been selected from patient registration lists provided by the Indian Health Service. Adults interviewed provided information on fish consumption for themselves and for 204 children under 5 years of age.

Fish consumption rates are shown in Tables H-17 and H-18. The values reflect an annual average, but monthly variations were also reported. Fish were consumed by over 90% of the

population with only 9% of the respondents reporting no fish consumption. The average daily consumption rate during the two highest intake months was 107.8 grams/day, and the daily consumption rate during the two lowest consumption months was 30.7 grams/day. Members who were aged 60 years and older had an average daily consumption rate of 74.4 grams/day. During the past two decades, a decrease in fish consumption was generally noted among respondents in this survey. The maximum daily consumption rate for fish reported for this group was 972 grams/day.

Table H-17
Fish Consumption by Columbia River Tribes,
Columbia River Inter-Tribal Commission, 1994

Subpopulation	Mean Daily Fish Consumption (g/day)
Total Adult Population, aged 18 years and older	59
Children, aged 5 years and younger	20
Adult Females	56
Adult Males	63

Table H-18
Fish Consumption by Columbia River Tribes
Columbia River Inter-Tribal Commission, 1994
Daily Fish Consumption Rates Among Adults

Percentile	Amount (g/day)
50th	29-32
90th	97-130
95th	170
99th	389

Tribes of Puget Sound

A study of fish consumption among the Tulalip and Squaxin Island Tribes of Puget Sound was completed in November of 1994 (Toy et al. 1995). The Tulalip and Squaxin Island Tribes live predominantly on reservations near Puget Sound, Washington. Both tribes rely on commercial fishing as an important part of tribal income. Subsistence fishing and shell-fishing are significant parts of tribal members economies and diets.

The study was conducted between February and April in 1994. Fish consumption practices were assessed by questionnaire and interview using dietary recall methods, food models and a food frequency questionnaire. The food frequency questionnaire was aimed as identifying seasonal variability. Questions in the interview included food preparation methods and obtained information on

the parts of the fish consumed. Fish consumed were categorized into anadromous fish (King salmon, sockeye salmon coho salmon, chum salmon, pink salmon, steelhead salmon, salmon unidentified and smelt), pelagic fish (cod, pollock, sable fish, spiny dogfish, rockfish, greenling, herring and perch), bottom fish (halibut, sole/flounder and sturgeon), and shell fish (manila clams, little clams, horse clams, butter clams, cockles, oysters, mussels, shrimp, dungeness crab, red rock crab, scallops, squid, sea urchin, sea cucumbers and moon snails)

Among consumers of anadromous fish, local waters (i.e., Puget Sound) supplied a mean of 80% of the fish consumed. Respondents from the Tulalip Tribes purchased a mean of approximately two-thirds of fish from grocery stores or restaurants, while among the Squaxin Island Tribe, the source of fish was about 50% self-caught and 50% purchased from grocery stores or restaurants. For bottom fish, members of both tribes caught about half of the fish they consumed. Anadromous fish were much more likely to be consumed with the skin attached. Most other fish were consumed minus the skin. Approximately 10% of the respondents consumed parts of the fish other than muscle: i.e., head, bones, eggs.

Data on fish consumption were obtained for 263 members from the Tulalip and Squaxin Island tribes. The mean consumption rate for women of both tribes was between 10-and-12-times higher than the default rate of 6.5 grams/day used by some parts of the United States government to estimate fish intake. Among male members of both tribes, the consumption rate was approximately 14-times higher than the default rate. The 50th percentile consumption rate for finfish for both tribes combined was 32 grams/kg body weight/day. Male members of the Tulalip and Squaxin Island tribes had average body weights of 189 pounds and 204 pounds, respectively. Female members of the Tulalip and Squaxin Island tribes weighed on average 166 pounds and 150 pounds, respectively. If an average body weight is assumed to be 70 kg, the daily fish consumption rate for both tribes for adults was 73 grams per day with a 90th percentile value of 156 grams per day for total fish. Fish consumption data for selected categories of fish are shown in Table H-19.

Table H-19
Fish Consumption (g/day) by the Tulalip and Squaxin Island Tribes
(Toy et al. 1995)

Type of Fish	5th Percentile	50th Percentile	90th Percentile	95th Percentile	Mean	SE	95th Percent CI
Anadromous	.0087	.2281	1.2026	1.9127	.4600	0345	3925, 0.5275
Pelagic	.0000	.0068	1026	2248	.0390	.0046	0300, 0 0480
Bottom	.0000	.0152	1095	2408	0482	0060	0364, 0 4375
Shell Fish	.0000	.1795	1.0743	1.4475	.3701	.0343	.3027, 0.4375
Other Fish	.0000	.0000	.0489	.1488	.0210	.0029	.0152, 0 0268
Total Finfish	.0200	.3200	.1350	2.1800	.5745	.0458	.4847, 0.6643
Total All Fish	.0495	.6081	2.2267	3.2292	1.0151	.0865	.8456, 1.1846

During the survey period, 21 of the 263 tribal members surveyed reported fish consumption rates greater than three standard deviations from the mean consumption rate. For example, six subjects reported consumptions of 5.85, 6.26, 9.85, 11.0, 22.6 and 11.2 grams of finfish and shell fish/kg body weight/day. If a 70-kg body weight is assumed these consumption rates correspond to 410, 438, 690, 770 and 1582 grams per day.

Mohawk Tribe

A study of fish consumption among 97 nursing Mohawk women in rural New York State was conducted from 1986 to 1992 (Fitzgerald et al., 1995). Fish consumption advisories had been issued in the area due to PCB contamination of the local water body. Using food frequency history and a long-term dietary history, the women were asked about their consumption of locally caught fish during three specific periods of time: during pregnancy, the year prior to pregnancy, and more than a year before pregnancy. For comparison, the study also surveyed fish consumption rates among 154 nursing (primarily caucasian) women from neighboring counties. The socioeconomic status of the women of the control group were similar to that of the Mohawk women. The fish in these counties had background PCB concentrations.

The results (See Table H-20) showed that the Mohawk women had a higher prevalence of consuming locally caught fish than the comparison group in the two intervals assessed prior to the pregnancy; the prevalence of local fish consumption during pregnancy for the two groups was comparable. A decrease in local fish consumption rates was also noted over time; these may be related to the issuance of advisories.

Table H-20
Local Fish Meals Consumed By Time Period for the Mohawk and Comparison Nursing Mothers (Source: Fitzgerald et al., 1995)

Fish Meals/ Year	During Pregnancy		1 Year Before Pregnancy		>1 Year Before Pregnancy	
	Mohawk	Control	Mohawk	Control	Mohawk	Control
0	64.9%	70.8%	43.3%	64.3%	20.6%	60.4%
1-9	24.7%	15.6%	41.2%	20.1%	43.3%	22.7%
10-19	5.2%	4.5%	4.1%	3.9%	6.2%	5.2%
>19	5.1%	9.1%	11.3%	11.7%	29.9%	11.7%

Summary of Native American Angler Surveys

Table H-10 summarizes the reported consumption rates of Native Americans detailed here. Table H-10 presents the range of fish consumption rates observed in the U.S. subpopulation.

1.4 Summary of Alaska and Hawaiian Fish Consumption Data

Alaska

The CSFII analyses of food intake by the U S Department of Agriculture include the 48 contiguous states but have not included Alaska or Hawaii. A number of investigators have published data on fish consumption in Alaska by members of native populations (e.g., Intuits, Eskimos) and persons living in isolated surroundings. These reports focus on nutritional/health benefits of high levels of fish consumption, food habits of native populations, and/or effects of bioaccumulation of chemicals in the aquatic food web.

General Population

After contacting professionals from the Alaskan health departments and representatives of the United States Centers for Disease Control in Anchorage, the authors of this report have not identified general population data on fish consumption among Alaskan residents who are not part of native population groups, subsistence fishers/hunters, or persons living in remote sites. Patterns of fish consumption among urban residents (e.g., Juneau, Nome, Anchorage) appear not to be documented in the published literature.

Non-urban Alaskan Populations

Native people living in the Arctic rely on traditional or "country" foods for cultural and economic reasons. The purpose of the current discussion is not to assess the comparative risks and benefits of these foods. The risks and benefits of these food consumption habits have been compared by many investigators and health professionals (among others see Wormworth, 1995; Kinloch et al., 1992; Bjerregaard, 1995).

Diets of Native Alaskans differ from the general population and rely more extensively on fish and marine mammals. These are population groups that are characterized by patterns of food consumption that reflect availability of locally available foods and include food preparation techniques that differ from those usually identified in nutrient data bases. For example, Nobmann et al. (1992) surveyed a population of Alaska Natives that included Eskimos (53%), Indians (34%), and Aleuts (13%). The distribution of study participants was proportional to the distribution of Alaska Natives reported in the 1980 Census. The 1990 Census identified an overall population of 85,698 persons as Alaska Natives.

Quantitative information on dietary intakes of Native Alaskan populations are few. Estimates can be derived from harvest survey data, but these have limitations because not all harvested animals are consumed nor are all edible portions consumed. Other edible portions may be fed to domestic animals (e.g., sled dogs). Substantial variability in intake of foods including ringed seal, bearded seal, muktuk (beluga skin with an underlying thin layer of fat) and walrus has been reported (Ayotte et al., 1995).

Nobmann et al. (1992) indicated that Alaska Natives have traditionally subsisted on fish; sea mammals; game; a few plants such as seaweed, willow leaves, and sourdock, and berries such as blueberries and salmonberries rather than on a plant-based diet. In preparing a nutrient analysis of the food consumed in eleven communities that represented different ethnic and socioeconomic regions of Alaska, these investigators added nutrient values for 210 foods consumed by Alaska Natives in addition to the 2400 foods present in the Veteran's Administration's nutrient data base. Nobmann et

al (1992) found fish were an important part of the diet. The mean daily intake of fish and shellfish of Alaska Natives was 109 grams/day. Fish consumption was more frequent in the summer and fall and game meat was eaten more often in the winter.

Traditional Native foods were still consumed in 1987-1988. Among participants in this study, 42% reported consuming traditional fermented foods at some time in their lives. Twenty-three percent of the surveyed population ate them in the fall of 1988; the season when these foods were most frequently eaten. Fermented foods of aquatic or marine origin that were reported to be consumed included salmon heads and eggs, other fish and their eggs, seal, beaver, and whale.

Consumption of marine mammals was reported among Yupik Eskimos living in either a coastal or river village of southwest Alaska (Parkinson et al., 1994). Concentrations of plasma omega-3 fatty acids were elevated (between 6.8 and 13 times) among the Yupic-speaking Eskimos living in two separate villages compared with non-Native control subjects (Parkinson et al., 1994). Concentrations of omega-3 fatty acids in plasma phospholipid has been shown to be a valid surrogate of fish consumption (Silverman et al., 1990). Among coastal-village participants the concentrations of eicosapentaonoic and docosahexaenoic acids reflected higher consumption of marine fish and marine mammals and the use of seal oil in food preparation. Among river village Natives, the increase reflected higher consumption of salmon.

Hawaiian Islands

As indicated above, the CSFII 89/91 did not include the Hawaiian Islands. To the knowledge of the authors of the Mercury Study Report to Congress, data describing fish consumption by the general Hawaiian population that estimate Island-wide levels of consumption have not been reported. However, reports on commercial utilization of seafoods (Hawaii Seafood, 198X; Higuchi and Pooley, 1985; Hudgins, 1980) and analysis of epidemiology data (Wilkens and Hankin, personal communication, 1996) provide a basis to describe general patterns of consumption. Overall, seafood consumption in Hawaii is much higher than in the contiguous United States. On a per capita basis the United States as a whole consumed 5.45 kg and 5.91 (12 and 13 pounds) of seafood in 1973 and 1977, respectively (Hudgins, 1980). By contrast Hawaiian per capita consumption for all fish products was 11.14 kg (24.5 pounds) in 1972 and 8.77 kg (19.3 pounds) in 1974.

The most popular species of fish and shellfish consumed were moderately comparable between Hawaii and the contiguous 48 states. The methods of food preparation differed, however, with raw fish being far more commonly consumed in Hawaii (Hawaii Seafood, 198X). Sampled at the retail trade level the most commonly purchased fish were: tuna, mahimahi, and shellfish [see also Table H-21 based on data in Higuchi and Pooley (1985)]. A survey of seafood consumption by families was identified. In 1987 the Department of Business and Economic Development (State of Hawaii, 1987) conducted a survey of 400 residents selected on a random digit dialing basis of a population representing 80% of total state seafood consumption. All data were collected in July and August, 1987 and would not reflect any seasonal differences in fish/shellfish consumption. The respondents were asked to describe seafood consumption by their families. Shrimp was the most popular seafood with mahimahi or dolphin fish as the second most popular (Hawaii Seafood, 1988). Reports on fish consumption in Hawaii separate various species of tuna: ahi (Hawaiian Yellowfin tuna, Bigeye tuna & Albacore tuna), aku (Hawaiian Skipjack tuna), and tuna. In 1987 nearly 66% of the 400 families surveyed had seafood at least once a week and 30% twice a week. Only 4% did not report consuming seafood during the previous week based on a telephone survey.

Wilkens and Hankin (personal communication, 28 February 1996) analyzed fish intake from 1856 control subjects from Oahu who participated in research studies conducted by the Epidemiology Program of the Cancer Research Center of Hawai'i, University of Hawai'i at Manoa. These subjects were asked about consumption over a one-year period prior to the interview. Within this group the most commonly consumed fish was tuna [canned with tuna species undesignated (70.8 % of subjects reporting consumption)]; shrimp (47.7% of subjects); tuna (yellowfin fresh designated aku, ahi with 42.2% of subjects reporting consumption); mahimahi [(or dolphin) with 32.5% of respondents reporting consumption]; and canned sardines (with 29.1% of subjects reporting consumpuon).

Table H-21
Species Composition of Hawaii's Retail Seafood Trade, 1981 (purchases) as described by Higuchi and Pooley, 1985

Fish/Shellfish	Pounds Purchased	Percent of Total Purchases
Tuna	11,600,000	20.9
Ahi (Hawaiian Yellowfin, Bigeye & Albacore Tuna)	(5,400,000)	
Billfish (including Swordfish) and Shark	5,900,000	11.3
Mahimahi and ono (wahoo)	9,900,000	17.7
Akule (Hawaiian Big Eye Scad) and opelu	4,00,000	6.9
Bottom fish	2,600,000	7.0
Reef fish	3,500,000	5.3
Shellfish Shrimp Lobster	8,200,000 (4,200,000) (900,000)	15.5
Other species Salmon/trout Snapper Frozen filets Frozen Sticks/blocks	8,300,000 (1,500,000) (1,800,000) (2,300,000) (1,400,000)	15.4
Total	54,000,000	100.0

1.5 Mercury Concentrations In Fish

1.5.1 Mercury Concentrations In Marine Fish

Human mercury intake from fish was estimated by combining data on mercury concentration in fish species with the reported quantities and types of fish species reported as consumed by 'users' in CSFII/89-91. The mercury concentrations in the consumed fish reported by the CSFII/89-91 were estimated using data on mercury concentration in fish expressed as micrograms of mercury per gram fresh-weight of fish tissue.

The CSFII/89-91 is one of the United States Department of Agriculture's (USDA's) food consumption surveys. The food items reported by individuals interviewed are identified by 7-digit food codes. The USDA has developed a recipe file identifying the primary components that make up the food or dish reported "as eaten" by a survey respondent. The total weight of a fish-containing food is typically not 100% fish. The food code specifies a preparation method and gives additional ingredients used in preparation of the dish. For example, in the Recipe File "Fish, floured or breaded, fried" contains 84% fish, by weight. Fish dishes contained a wide range of fish: from approximately 5% for a frozen "shrimp chow mein dinner with egg roll and peppers" to 100% for fish consumed raw, such as raw shark.

Data describing methylmercury concentrations in marine fish were predominantly based on the National Marine Fisheries' Service (NMFS) data base, the largest publicly available data base on mercury concentrations in marine fish. In the early 1970s, the NMFS conducted testing for total mercury on over 200 seafood species of commercial and recreational interest (Hall et al., 1978). The determination of mercury in fish was based on flameless (cold vapor) atomic absorption spectrophotometry following chemical digestion of the fish sample. These methods are described in Hall et al. (1978). Data supplied by NMFS give the mercury concentration in fresh weight of fish muscle of numerous marine fish, shellfish, and other molluscan species shown in Tables H-22, H-23 and H-24.

Although the NMFS data were initially compiled beginning in the 1970s, comparisons of the mercury concentrations identified in the National Marine Fisheries Service's data base with compliance samples obtained by the United States Food and Drug Administration indicate that the NMFS data are appropriate to use in estimating intake of mercury from fish at the national level of data aggregation. Cramer (1994) of the Office of Seafood of the Center for Food Safety and Applied Nutrition of the US Food and Drug Administration reported on "Exposure of U.S. Consumers to Methylmercury from Fish". He noted that recent information from National Marine Fisheries Service (NMFS) indicated that the fish mercury concentrations reported in the 1978 report do not appear to have changed significantly. The US FDA continues to monitor methylmercury concentration in seafood. Cramer (1994) observed that results of recent US FDA surveys indicate results parallel to FDA's and NMFS's earlier findings. To illustrate, Cramer estimated the mean methylmercury content of the 1973 samples of canned tuna at 0.21 ppm mercury, whereas a recently completed survey of 245 samples of canned tuna was 0.17 ppm mercury. These data are considered to be comparable, although the small decrease reported between these two studies may reflect increased use in canned tuna of tuna species with slightly lower average methylmercury concentrations. The National Academy of Sciences' National Research Council's Subcommittee on Seafood Safety (1991) also assessed the applicability of the NMFS' 1970's data base to current estimates of mercury concentrations in fish. This subcommittee also concluded that the 1978 data base differed little in mercury concentration from FDA compliance samples estimating mercury concentrations in fish.

Table H-22
Mercury Concentrations in Marine Finfish

Fish	Mercury Concentration (ug/g, wet weight)	Source of Data
Anchovy[1]	0.047	NMFS
Barracuda, Pacific[2]	0 177	NMFS
Cod[3]	0.121	NMFS
Croaker, Atlantic	0 125	NMFS
Eel, American	0.213	NMFS
Flounder[4]	0.092	NMFS
Haddock	0.089	NMFS
Hake[5]	0.145	NMFS
Halibut[6]	0 25	NMFS
Herring[7]	0 013	NMFS
Kingfish[8]	0.10	NMFS
Mackerel[9]	0 081	NMFS
Mullet[10]	0.009	NMFS
Ocean Perch[11]	0.116	NMFS
Pollack	0.15	NMFS
Pompano	0.104	NMFS
Porgy	0.522	NMFS
Ray	0.176	NMFS
Salmon[12]	0.035	NMFS
Sardines[13]	0 1	NMFS
Sea Bass	0.135	NMFS
Shark[14]	1.327	NMFS
Skate[15]	0.176	NMFS
Smelt, Rainbow	0.1	NMFS
Snapper[16]	0.25	NMFS
Sturgeon[17]	0.235	NMFS
Swordfish	0 95	FDA Compliance Testing
Tuna[18]	0.206	NMFS
Whiting (silver hake)	0.041	NMFS

Table H-22 Footnotes

[1] This is the average of NMFS mean mercury concentrations for both striped anchovy (0.082 µg/g) and northern anchovy (0.010 µg/g)

[2] USDA data base specified the consumption of the Pacific Barracuda and not the Atlantic Barracuda.

[3] The mercury content for cod is the average of the mean concentrations in Atlantic Cod (0.114 µg/g and the Pacific Cod (0.127 µg/g)

[4] The mercury content for flounder is the average of the mean concentrations measured in 9 types of flounder:Gulf (0.147 µg/g), summer (0.127 µg/g), southern (0.078 µg/g), four-spot (0.090 µg/g), windowpane (0.151 µg/g), arrowtooth (0.020 µg/g), witch (0.083 µg/g), yellowtail (0.067 µg/g), and winter (0.066 µg/g).

[5] The mercury content for Hake is the average of the mean concentrations measured in 6 types of Hake. silver (0.041 µg/g), Pacific (0.091 µg/g), spotted (0.042 µg/g), red (0.076 µg/g), white (0.112 µg/g), and blue (0.405 µg/g).

[6] The mercury content for Halibut is the average of the mean concentrations measured in 3 types of Halibut: Greenland, Atlantic, and Pacific.

[7] The mercury content for Herring is the average of the mean concentrations measured in 4 types of Herring: blueback (0.0 µg/g), Atlantic (0.012 µg/g), Pacific (0.030 µg/g), and round (0.008 µg/g).

[8] The mercury content for Kingfish is the average of the mean concentrations measured in 3 types of Kingfish: Southern, Gulf, and Northern.

[9] The mercury content for Mackerel is the average of the mean concentrations measured in 3 types of Mackerel. jack (0.138 µg/g), chub (0.081 µg/g), and Atlantic (0.025 µg/g)

[10] The mercury content for Mullet is the average of the mean concentrations measured in 2 types of Mullet: striped (0.011 µg/g) and silver (0.007 µg/g).

[11] The mercury content for Ocean Perch is the average of the mean concentrations measured in 2 types of Ocean Perch: Pacific (0.083 µg/g) and Redfish (0.149 µg/g)

[12] The mercury content for Salmon is the average of the mean concentrations measured in 5 types of Salmon: pink (0.019 µg/g), chum (0.030 µg/g), coho (0.038 µg/g), sockeye (0.027 µg/g), and chinook (0.063 µg/g).

[13] Sardines were estimated from mercury concentrations in small Atlantic Herring.

[14] The mercury content for Shark is the average of the mean concentrations measured in 9 types of Shark: spiny dogfish (0.607 µg/g), (unclassified) dogfish (0.477 µg/g), smooth dogfish (0.991 µg/g), scalloped hammerhead (2.088 µg/g), smooth hammerhead (2.663 µg/g), shortfin mako (2.539 µg/g), blackup shark (0.703 µg/g), sandbar shark (1.397 µg/g), and thresher shark (0.481 µg/g).

[15] The mercury content for skate is the average of the mean concentrations measured in 3 types of skate: thorny skate (0.200 µg/g), little skate 0.135 µg/g) and the winter skate (0.193 µg/g).

[16] The mercury content for snapper is the average of the mean concentrations measured in types of snapper:

[17] The mercury content for sturgeon is the average of the mean concentrations measured in 2 types of sturgeon.green sturgeon (0.218 µg/g) and white sturgeon (0.251 µg/g).

[18] The mercury content for tuna is the average of the mean concentrations measured in 3 types of tuna: albacore tuna (0.264 µg/g), skipjack tuna (0.136 µg/g) and yellowfin tuna (0.218 µg/g)

Table H-23
Mercury Concentrations in Marine Shellfish

Shellfish	Mercury Concentration (µg/g, wet weight)	Source of Data
Abalone[1]	0.016	NMFS
Clam[2]	0.023	NMFS
Crab[3]	0.117	NMFS
Lobster[4]	0.232	NMFS
Oysters[5]	0.023	NMFS
Scallop[6]	0.042	NMFS
Shrimp[7]	0.047	NMFS

[1] The mercury content for abalone is the average of the mean concentrations measured in 2 types of abalone: green abalone (0.011 µg/g) and red abalone (0.021 µg/g).
[2] The mercury content for clam is the average of the mean concentrations measured in 4 types of clam: hard (or quahog) clam (0.034 µg/g), Pacific littleneck clam (0 µg/g), soft clam (0.027 µg/g), and geoduck clam (0.032 µg/g)
[3] The mercury content for crab is the average of the mean concentrations measured in 5 types of crab: blue crab (0.140 µg/g), dungeness crab (0.183 µg/g), king crab (0.070 µg/g), tanner crab (C.opilio) (0.088 µg/g), and tanner crab (C.bairdi) (0.102 µg/g).
[4] The mercury content for lobster is the average of the mean concentrations measured in 3 types of lobster: spiny (Atlantic) lobster (0.108 µg/g), spiny (Pacific) lobster (0.210 µg/g) and northern (American) lobster (0.378 µg/g).
[5] The mercury content for oyster is the average of the mean concentrations measured in 2 types of oyster: eastern oyster (0.022 µg/g) and Pacific (giant) oyster (0.023 µg/g).
[6] The mercury content for scallop is the average of the mean concentrations measured in 4 types of scallop: sea (smooth) scallop (0.101 µg/g), Atlantic Bay scallop (0.038 µg/g), calico scallop (0.026 µg/g), and pink scallop (0.004 µg/g).
[7] The mercury content for shrimp is the average of the mean concentrations measured in 7 types of shrimp: royal red shrimp (0.074 µg/g), white shrimp (0.054 µg/g), brown shrimp (0.048 µg/g), ocean shrimp (0.053 µg/g), pink shrimp (0.031 µg/g), pink northern shrimp (0.024 µg/g) and Alaska (sidestripe) shrimp (0.042 µg/g).

Table H-24
Mercury Concentrations in Marine Molluscan Cephalopods

Cephalopod	Mercury Concentration (µg/g wet wt.)	Source of Data
Octopus	0.029	NMFS
Squid[1]	0.026	NMFS

[1] The mercury content for squid is the average of the mean concentrations measured in 3 types of squid: Atlantic longfinned squid (0.025 µg/g), short-finned squid (0.034 µg/g), and Pacific squid (0.018 µg/g)

Table H-25
Summary of Mercury Concentrations in Fish Species
Micrograms Mercury per Gram Fresh Weight (ug Hg/g)

Data Used by USEPA Mercury Study Report to Congress In Review		Data Used by US FDA Report on the Chance of U.S. Seafood Consumers Exceeding "The Current Daily Intake for Mercury and Recommended Regulatory Controls" 1978			Data Used by Stern et al. 1996	
Fish Species	**Average (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**	**Maximum (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**
Abalone	0.016	Abalone	0.018	0.120	Not Reported (NR)	
Anchovies	0 047	Anchovies	0.039	0.210	NR	
Bass, Freshwater	Avgs.= 0.157 (Lowe et al., 1985) and 0.38 (Bahnick et al., 1994)	Bass, Striped	0.752	2.000	Bass, freshwater	0.41
Bass, Sea	Not Reported	Bass, Sea	0.157	0 575	Sea Bass	0 25
Bluefish	Not Reported	Bluefish	0.370	1.255	Bluefish	0.35
Bluegills	0.033	Bluegills	0.259	1.010	NR	
Bonito	Not Reported	Bonito (below 3197)	0.302	0.470	NR	
Bonito	Not Reported	Bonito (above 3197)	0.382	0.740	NR	
Butterfish	Not Reported	Butterfish	0.021	0.190	Butterfish	0 05
Carp, Common	0.093	Carp	0.181	0.540	Catfish, freshwater	0.15
Catfish (channel,large mouth, rock, striped, white)	0.088	Catfish (freshwater)	0.146	0.380	Clams	0.05
Catfish (Marine)	Not Reported	Catfish (Marine)	0.475	1.200	Cod/Scrod	0.15
Clams	0.023	Clams	0.049	0.260	See crab.	
Cod	0.121	Cod	0.125	0.590	Crab	0.15

Table H-25 (continued)
Summary of Mercury Concentrations in Fish Species
Micrograms Mercury per Gram Fresh Weight (µg Hg/g)

Data Used by USEPA Mercury Study Report to Congress In Review		Data Used by US FDA Report on the Chance of U.S. Seafood Consumers Exceeding "The Current Daily Intake for Mercury and Recommended Regulatory Controls" 1978			Data Used by Stern et al. 1996	
Fish Species	**Average (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**	**Maximum (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**
Crab, King	0.070: Calculations based on 5 species of crab combined at 0 117	Crab, King	0.070	0 240	NR	
Crab	0 117	Crab, other than H1	0.140	0 610	NR	
Crappie (black, white)	0.114	Crappie	0.262	1.390	NR	
Croaker	0.125	Croaker	0.124	0 810	NR	
Dolphin	Not Reported	Dolphin	0.144	0.530	Dolphin (Mahi-mahi)	0.25
Drums, Freshwater	0.117	Drums	0.150	0.800	NR	
Flounders	0.092	Flounders	0.096	0.880	Flounder	0.10
Groupers		Groupers	0.595	2.450	NR	
Haddock	0.089	Haddock	0.109	0.368	Haddock	0 05
Hake	0.145	Hake	0.100	1 100	Hake	0 10
Halibut	0.250	Halibut 4	0 187	1 000	Halibut	0 25
Halibut	0.250	Halibut 3	0 284	1 260	Halibut	0 25
Halibut	0.250	Halibut 2H	0.440	1.460	Halibut	0.25
Halibut	0.250	Halibut 25	0.534	1.430	Halibut	0.25
Herring	0.013	Herring	0 023	0.260	Herring	0.05
Kingfish	0.100	Kingfish	0.078	0.330	Kingfish	0.05
Lobster	0.232	Lobster, Northern 11	0.339	1 603	Lobster	0.25
Lobster	0.232	Lobster Northern 10	0.509	2.310	Lobster	0.25

Table H-25 (continued)
Summary of Mercury Concentrations in Fish Species
Micrograms Mercury per Gram Fresh Weight (µg Hg/g)

Data Used by USEPA Mercury Study Report to Congress In Review		Data Used by US FDA Report on the Chance of U.S. Seafood Consumers Exceeding "The Current Daily Intake for Mercury and Recommended Regulatory Controls" 1978			Data Used by Stern et al. 1996	
Fish Species	**Average (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**	**Maximum (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**
Lobster Spiny	0.232; Includes spiny (Pacific) lobster=0.210	Lobster.Spiny	0.113	0.370	Lobster	0 25
Mackerel	0.081: Averaged Chub = 0.081. Atlantic= 0.025: Jack=0.138	Mackerel. Atlantic	0.048	0 190	Mackerel	0 28
Mackerel	0.081	Mackerel. Jack	0.267	0.510	Mackerel	0.28
Mackerel	0.081	Mackerel. King (Gulf)	0.823	2.730	Mackerel	0.28
Mackerel	0 081	Mackerel. King (other)	1.128	2.900	Mackerel	0.28
Mackerel	0.081	Mackerel. Spanish 16	0.542	2.470	Mackerel	0.28
Mackerel	0.081	Mackerel. Spanish 10	0.825	1.605	Mackerel	0 28
Mullet	0.009	Mullet	0.016	0.280	Mullet	0.05
Oysters	0.023	Oysters	0.027	0.460	NR	
Perch. White and Yellow	0.110	Perch. Freshwater	0.290	0.880	Perch	0.18
Perch. Ocean	0.116	Perch. Marine	0.133	0.590	NR	
Pike. Northern	0.310 0.127	Pike	0.810	1.710	NR	
Pollock	0.150	Pollock	0.141	0.960	NR	
Pompano	0.104	Pompano	0.104	8.420	NR	
Rockfish	Not Reported	Rockfish	0.340	0.930	NR	

Table H-25 (continued)
Summary of Mercury Concentrations in Fish Species
Micrograms Mercury per Gram Fresh Weight (µg Hg/g)

Data Used by USEPA Mercury Study Report to Congress In Review		Data Used by US FDA Report on the Chance of U.S. Seafood Consumers Exceeding "The Current Daily Intake for Mercury and Recommended Regulatory Controls" 1978			Data Used by Stern et al. 1996	
Fish Species	**Average (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**	**Maximum (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**
Sablefish	Not Reported	Sablefish	0.201	0.700	NR	
Salmon	0.035	Salmon	0.040	0.210	Salmon	0 05
Scallops	0.042	Scallops	0 058	0 220	NR	
Scup	Not Reported	Scup	0 106	0.520	NR	
Sharks	1 327	Sharks	1 244	4 528	Shark	1 11
Shrimp	0 047	Shrimp	0 040	0.440	Shrimp	0 11
Smelt	0.100	Smelt	0.016	0.058	Smelts	0 05
Snapper	0.25	Snapper.Red	0 454	2.170	Snapper	0 31
Snapper	0 25	Snapper. Other	0.362	1.840	Snapper	0.31
Snook	Not Reported	Snook	0.701	1 640	NR	
Spot	Not Reported	Spot	0.041	0.180	Spotfish	0 05
Squid	0.026	Squid and Octopi	0.031	0.400	Squid	0.05
Octopi	0.029	Squid and Octopi	0.031	0.400	NR	
Sunfish	Not Reported	Sunfish	0.312	1.200	NR	
Swordfish	0.95	Swordfish	1.218	2.720	Swordfish	0.93
Tillefish	Not Reported	Tillefish	1.607	3.730	NR	
Trout,	0.149	Trout, Freshwater	0.417	1.220	Trout	0.05
Trout	0.149	Trout, Marine	0.212	1.190	Trout	0.05

Table H-25 (continued)
Summary of Mercury Concentrations in Fish Species
Micrograms Mercury per Gram Fresh Weight (µg Hg/g)

Data Used by USEPA Mercury Study Report to Congress In Review		Data Used by US FDA Report on the Chance of U.S. Seafood Consumers Exceeding "The Current Daily Intake for Mercury and Recommended Regulatory Controls" 1978			Data Used by Stern et al. 1996	
Fish Species	**Average (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**	**Maximum (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**
Tuna	0.206; Averaged: Tuna, light skipjack=0.136Tuna.light yellow=0.218; Albacore=0.264	Tuna. Light Skipjack	0.144	0.385	Tuna. fresh	0.17
Tuna	0.206	Tuna. Light Yellow	0.271	0.870	Tuna. fresh	0 17
Tuna	0.206	Tuna. White	0.350	0.904	Tuna. fresh	0 17
Whitefish	Not Reported	Whitefish	0.054	0.230	Whitefish	0 04
Other finfish		Other finfish	0 287	1.020	Finfish. other	0.17
Other shellfish	Not Reported				Shellfish. other	0.12
Fish Species (Freshwater) Not Reported by FDA, 1978						
Bloater	0.0.93					
Smallmouth Buffalo	0.096					
Northern Squawfish	0.33					
Sauger	0.23					
Sucker	0.114 (Lowe et al., 1985; 0.167 (Bahnick et al., 1994).					

Table H-25 (continued)
Summary of Mercury Concentrations in Fish Species
Micrograms Mercury per Gram Fresh Weight (µg Hg/g)

Data Used by USEPA Mercury Study Report to Congress In Review		Data Used by US FDA Report on the Chance of U.S. Seafood Consumers Exceeding "The Current Daily Intake for Mercury and Recommended Regulatory Controls" 1978			Data Used by Stern et al. 1996	
Fish Species	**Average (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**	**Maximum (µg Hg/g)**	**Fish Species**	**Average (µg Hg/g)**
Walleye	0.100 (Lowe et al., 1985) and 0.52 (Bahnick et al., 1994).					
Trout (brown, lake, rainbow)	0.149 (Lowe et al., 1985) and 0.14 (Bahnick et al., 1994 for brown trout).					
Fish Species Reported by the State of New Jersey and Not Reported by EPA or FDA						
Blowfish						0.05
Orange roughy						0.5
Sole						0.12
Weakfish						0.15
Porgy						0.55
Blackfish						0.25
Whiting						0.05
Turbot						0.10
Sardines						0.05
Tilapia						0.05

Central tendency estimates of seafood mercury concentrations were utilized in the report. This seems appropriate since commercial seafood is widely distributed across the U.S. (Seafood Safety, 1991). The source of a particular fish purchase is generally not noted by the consumer (e.g., canned tuna). As a result, a randomness and averaging may be achieved. Additionally, only common names of commercial seafood were utilized; specific species which could be considered to be that type of fish were included in the central tendency estimate. Again, typical consumers were assumed to generally not be aware of the species of fish they were consuming, rather just the type.

As noted above, there are other estimates of mercury concentrations in seafood. After the analysis of mercury exposure from seafood was completed for this Report, two other databases were obtained: U.S. FDA and Stern et al., 1996. These data are presented in Table H-22 for comparison with those data used for this analysis.

1.5.2 Mercury Concentrations In Fresh-water Fish

Estimation of average mercury concentrations in fresh-water finfish from across the U.S. required a compilation of measurements of fish mercury concentrations from randomly selected U.S. water bodies. A large number of sources of mercury concentrations in fish were not used in this part of the assessment. Mercury concentrations in fish have been analyzed for a number of years in many local or regional water bodies in the U.S.; several of these studies are detailed in this Report, and the results of these as well as other studies are listed in Tables 2-2 through 2-9 in Chapter 2 of Volume III. Data described in this body of literature are a collection of individual studies which characterize mercury concentrations in fish from specific geographic regions such as individual water bodies or in individual states. Many of the studies were initiated because of a problem, perceived or otherwise, with mercury concentrations in the fish or the water body. Thus, the sample presented by a compilation of these data may be biased toward the high-end of the distribution of mercury concentrations in fresh-water fish. Additionally, the methods varied from study to study, and there is no way of determining the consistency of the reported data from study to study.

Two studies, more national in scope, are thought to provide a more complete picture of mercury concentrations in U.S. fresh-water finfish populations: "National Contaminant Biomonitoring Program: Concentrations of Seven Elements in Fresh-water Fish, 1978-1981" by Lowe et al. (1985) and "A National Study of Chemical Residues in Fish" conducted by U.S. EPA (1992) and also reported in Bahnick et al. (1994).

Lowe et al. (1985) reported mercury concentrations in fish from the National Contaminant Biomonitoring Program. The fresh-water fish data were collected between 1978-1981 at 112 stations located across the United States. Mercury was measured by a flameless cold vapor technique, and the detection limit was 0.01 µg/g wet weight. Most of the sampled fish were taken from rivers (93 of the 112 sample sites were rivers); the other 19 sites included larger lakes, canals, and streams. Fish weights and lengths were consistently recorded. A wide variety of types of fishes were sampled: most commonly carp, large mouth bass and white sucker. The geometric mean mercury concentration of all sampled fish was 0.11 µg/g wet weight; the minimum and maximum concentrations reported were 0.01 and 0.77 µg/g wet weight, respectively. The highest reported mercury concentrations (0.77 µg/g wet weight) occurred in the northern squawfish of the Columbia River. See Table H-26 for mean mercury concentrations by fish species.

Table H-26
Fresh-water Fish Mercury Concentrations from Lowe et al., (1985)

Species	Mean Mercury Concentration µg/g (fresh weight)
Bass	0.157
Bloater	0.093
Bluegill	0.033
Smallmouth Buffalo	0.096
Carp, Common	0.093
Catfish (channel, largemouth, rock, striped, white)	0 088
Crappie (black, white)	0 114
Fresh-water Drum	0 117
Northern Squawfish	0.33
Northern Pike	0.127
Perch (white and yellow)	0.11
Sauger	0.23
Sucker (bridgelip, carpsucker, klamath, largescale, longnose, rivercarpsucker, tahoe)	0.114
Trout (brown, lake, rainbow)	0.149
Walleye	0.100
Mean of all measured fish	0.11

"A National Study of Chemical Residues in Fish" was conducted by U.S. EPA (1992) and also reported by Bahnick et al. (1994). In this study mercury concentrations in fish tissue were analyzed. Five bottom feeders (e.g., carp) and five game fish (e.g., bass) were sampled at each of the 314 sampling sites in the U.S. The sites were selected based on proximity to either point or non-point pollution sources. Thirty-five "remote" sites among the 314 were included to provide background pollutant concentrations. The study primarily targeted sites that were expected to be impacted by increased dioxin levels. The point sources proximate to sites of fish collection included these: pulp and paper mills, Superfund sites, publicly owned treatment works and other industrial sites. Data describing fish age, weight, and sex were not consistently collected. Whole body mercury concentrations were determined for bottom feeders and mercury concentrations in fillets were analyzed for the game fish. Total mercury levels were analyzed using flameless atomic absorption; the reported detection limits were 0.05 µg/g early in the study and 0.0013 µg/g as analytical technique improved later in the analysis. Mercury was detected in fish at 92% of the sample sites. The maximum mercury level detected was 1.8 µg/g, and the mean across all fish and all sites was 0.26 µg/g. The highest measurements occurred in walleye, large mouth bass and carp. The mercury concentrations in fish around publicly owned treatment works were highest of all point source data; the median value measured were 0.61 µg/g. Paper mills were located near many of the sites where mercury-laden fish

was detected. Table H-27 contains the mean mercury concentrations of the species collected by Bahnick et al. (1994).

Both the studies reported by Lowe et al. (1985) and by Bahnick et al. (1994) appear to be systematic, national collections of fish pollutant concentration data. Clearly, higher mercury concentrations in fish have been detected in other analyses, and the values obtained in these studies should be interpreted as a rough approximation of the mean concentrations in fresh-water finfishes. As indicated in the range of data presented in Tables H-26 and H-27, as well as the aforementioned Tables in Chapter 2, wide variations are expected in data on mercury concentrations in freshwater fish.

The mean mercury concentrations in all fish sampled vary by a factor of two between the studies. The mean mercury concentration reported by Lowe et al. was 0.11 μg/g, whereas the mean mercury concentration reported by Bahnick et al. was 0.26 μg/g. This difference can be extended to the highest reported mean concentrations in fish species. Note that the average mercury concentrations in bass and walleye reported by Bahnick's data are higher than the northern squawfish, which is the species with the highest mean concentration of mercury identified by Lowe et al. (1985).

The bases for these differences in methylmercury concentrations are not immediately obvious. The trophic positions of the species sampled, the sizes of the fish, or ages of fish sampled could significantly increase or decrease the reported mean mercury concentration. Older and larger fish, which occupy higher trophic positions in the aquatic food chain, would, all other factors being equal, be expected to have higher mercury concentrations. The sources of the fish also influence fish mercury concentrations. Most of the fish obtained by Lowe et al. (1985) were from rivers. The fate and transport of mercury in river systems is less well characterized than in small lakes. Most of the data collected by Bahnick et al. (1994) were collected with a bias toward more contaminated/industrialized sites, although not sites specifically contaminated with mercury. It could be that there is more mercury available to the aquatic food chains at the sites reported by Bahnick et al. (1994). Finally, the increase in the more recent data as reported in Bahnick et al., 1994 could be the result of temporal increases in mercury concentrations.

There is a degree of uncertainty in the mercury concentrations selected for this assessment. This uncertainty reflects both the adequacy of the sampling protocol for this application and the known variability in fish body burden. The variability in these data is as broad as the range of reported concentrations, which extends from non-detect (below 0.01 μg/g wet weight) up to 9 μg/g wet weight. Where possible, when specific fresh-water fish species are described in the USDA 3-day consumption studies, the mean methylmercury concentration for that particular species was derived in two separate calculations based on the data on methylmercury concentration in the fish reported by Lowe et al. (1985) and by Bahnick et al. (1994).

Data for mean mercury concentration in fresh-water fish from Bahnick et al. (1994) were combined with the U.S. consumption rates for fresh-water fish from CSFII/89-91 to estimate methylmercury intakes for the population. The concentrations in the fish utilized are shown in Table H-27. The exposure estimates for fresh-water fin fish consumption are found in Table H-28. Bahnick et al. (1994) fresh-water fish concentration data were utilized, along with data on mercury concentrations in marine fish and shellfish (Tables H-22, H-23, H-24) to calculate total exposure, for general U.S. population, to mercury through consumption of fish and shellfish (shown in Table H-28).

Some species of fresh-water fishes were not sampled by Bahnick et al. (1994), and some respondents in the USDA CSFII/89-91 survey did not identify the type of fresh-water fish consumed. In these situations, it was assumed that the fish consumed contained 0.26 μg methylmercury/g, which

Table H-27
U.S. EPA (1992) and Bahnick et al. (1994)

Freshwater Fish	Average Mercury Concentration (µg/g, wet weight)
Carp	0.11
Sucker[1]	0.167
Catfish, Channel and Flathead	0.16
Bass[2]	0.38
Walleye	0.52
Northern Pike	0.31
Crappie	0.22
Brown Trout	0.14
Mean All Fish Sampled	0.26

[1] The value presented is the mean of the average concentrations found in 3 types of Sucker fish (White, Redhorse and Spotter).

[2] The value presented is the mean of the average concentrations found in 3 types of Bass (White, Largemouth and Smallmouth).

is the average of all sampled fish Bahnick et al. (1994). It is important to note that the freshwater fish data are for wild populations not farm-raised fish.

2. CALCULATION OF MERCURY CONCENTRATIONS IN FISH DISHES

To estimate the mercury intake from fish and fish dishes reported as consumed by respondents in the CSFII 1989/1991 survey several steps were taken. Using the Recipe File available from USDA, the fish species for a particular reported food was identified. The average mercury concentration in fish tissue on a fresh (or wet) weight basis was identified using the NMFS data or the data reported by Bahnick et al. (1994). The food intake of the U.S. population includes a large number of components of aquatic origin. A few of these appear not to have been analyzed for mercury concentrations. Methylmercury concentration data were not available for some infrequently consumed food items: e.g., turtle, roe or jelly fish. Data on the quantity of fish present in commercially prepared soups were also not available and were excluded from the analysis.

Physical changes occur to a food when it is processed and/or cooked. The NMFS and Bahnick et al. (1994) data bases were used to estimate mercury intake report mercury concentrations on a µg mercury per gram of fresh tissue basis. Earlier research (Bloom, 1992) indicated that over 90% of mercury present in fish and shellfish is chemically speciated as methylmercury which is bound to protein in fish tissue. Morgan et al., (1994) indicated that over 90% of mercury present in fish and shellfish is chemically speciated as methylmercury. Consequently the quantity of methylmercury present in the fish tissue in the raw state will remain in the cooked or processed fish. In cooking or processing raw fish, there is typically a reduction in the percent moisture in the food. Thus, mercury concentration data were recalculated to reflect the loss of moisture during food processing, as well as

retention of methylmercury in the remaining lowered-moisture content fish tissues. Standard estimates of cooking/processing-related weight reductions were provided by Dr. Betty Perloff and Dr. Jacob Exler, experts in the USDA recipe file and in USDA's food composition. Percent moisture lost for baked or broiled fish was 25%. Fried fish products lose weight through loss of moisture but add weight from fat added during frying for a total weight loss of minus 12%. The percent moisture in fish that were dried, pickled or smoked was identified for individual fish species (e.g., herring, cod, trout, etc.) from USDA handbooks of food composition. Information on the percent moisture in the raw, and in the dried, smoked or pickled fish was obtained. The methylmercury concentration in the fish was recalculated to reflect the increased methylmercury concentration of the fish as the percent moisture decreased in the drying, pickling or smoking process.

2.1 Additional Application Of Mean Mercury Concentrations In Fresh-water Fish

The mean mercury concentrations for all fish from Lowe et al. (1985) and Bahnick et al. (1994) were combined with the freshwater fish consumption data to estimate a range of exposure from total fish consumption. Given the human fish consumption rates and the differences between the mercury concentrations in the two data sets, it is important to use data from both studies of mercury exposures to assess mean concentrations in fish. For purposes of comparison both sets of data were utilized to illustrate the predicted methylmercury exposure. For this comparison, the average mercury concentrations for fish in the Lowe and the Bahnick data were analyzed separately by combining the freshwater fish data with the data in Tables H-22 through H-24. The bodyweight data were obtained from Table H-36 (Consumption of Fresh-water fish and Self-Reported Bodyweight among Respondents of the 1989-1991 CSFII 89-91) and the fresh-water fish consumption rates were obtained from Table H-36. Exposure to methylmercury based on an assumption of 0.11 μg methylmercury/g fish tissue (wet weight) (Lowe et al., 1985). These values are estimated on a body weight basis. Tables H-26 and H-27 were developed using the mean data on mercury concentrations for all fishes sampled for these two studies.

3. INTAKE OF METHYLMERCURY FROM FISH/FISH DISHES

3.1 Intakes "per User" and "per Capita"

Estimated intake of methylmercury from fish can be made by summing the grams of fish consumed and dividing by the number of individuals surveyed to obtain a "per capita" intake of fish. "Per capita" consumption of total fish is 13.5 grams per day with an overall freshwater fish mean intake of 2.2 grams per day (standard deviation of 13.7 grams per day) for all 11,706 individuals surveyed. Overall, 30.9% of the surveyed population reported eating fish at least once during the three-day survey period. The extreme variability of this estimate (standard deviation approximately six-times the mean fish intake) indicates the high degree of heterogeneity of the population. A fundamental determinant of this heterogeneity is the subset of the population who consumes fish on a sufficiently regular basis that fish appears in the individuals diets at least once within the three-day period surveyed. In order to predict the impact of consumption of a chemical contaminant almost exclusively contained in fish, the choice was made to report data on a "per user" basis.

3.2 Fish Intake by Age and/or Gender Grouping of Subpopulations

The age and gender distribution of respondents to the CSFII/89-91 survey are shown in Table H-5. Tests of statistical significance comparing distribution of fish consumption by age and gender have not been conducted. Generally the percent of the population ages birth through 14 years who eat fish is smaller (19.0% of persons ages 14 years or younger) than the representation of this group in the

overall survey population (24.7%). The percent of adults ages 45 and older who reported fish consumption (39.2) is higher than their proportion in the overall surveyed population (32.8%).

The percent of males and females who report eating fish is comparable to their distribution in the surveyed population. Males overall and within each age category consumed more fish expressed either as total grams per day or per kilogram self-reported body weight. Average self reported body weights are shown in Tables H-7, H-8, H-32, H-34, H-36, H-38 and H-40, and the fish consumption data for all age and gender distribution are shown in Tables H-5, H-6, H-31, H-33, H-35, H-37 and H-39. The data specific to this survey rather than default values of 70 kilograms have been used in reporting fish and methylmercury intake data for CSFII/89-91.

Although children ages birth through 14 years consumed smaller total amounts of fish than did older survey respondents, children were exposed to just over twice as much methylmercury relative to body size when exposure is expressed on a per kilogram body weight basis. These data are presented in Table H-7. Fish consumption among adult women of child-bearing age are shown in Table H-7.

Estimates of methylmercury intake from fish have been calculated for children ages birth through 14 years. These data are presented in Tables H-28, H-29, H-30 and H-41. Subpopulations within the general population may be of concern because of their vulnerability to the toxicity of methylmercury or because of increased likelihood of elevated exposure to methylmercury. Particular emphasis has been given to the frequency of fish consumed by women of child-bearing age as methylmercury is a developmental toxicant. Children ages birth through 14 years have methylmercury exposures between two and three times adult levels when methylmercury intake is expressed on a per kilogram body weight basis.

3.3 Types of Fish Consumed

To interpret the data presented in Tables H-28 through H-41, it is essential to recognize that the data reported for a particular category of fish (e.g., marine finfish, fresh-water finfish, tuna fish, etc.) are average intakes per day for persons who report consumption of a particular type of fish. Because an individual may consume more than one type of fish during the three-day survey period, this individual may appear in more than one category of fish consumer. For this reason the number of consumers of individual categories of fish (e.g., tuna, shellfish) sum to more than the number of consumers of fish and shellfish reported in Table H-6.

In CSFII/89-91, marine finfish were consumed to a far greater extent than fresh-water fish. For example, marine finfish (excluding tuna, swordfish, barracuda, and shark) were consumed by 1595 respondents, tuna by 1444 respondent, swordfish/barracuda/shark by 21 respondents for a total of 2060 reported consumptions; this is in contrast to 492 respondents reporting consumption of fresh-water finfish. For individual species of fish, tuna was the most commonly consumed fish. Shellfish were consumed more often than fresh-water finfish, but to a far smaller extent than marine finfish.

3.4 Methylmercury Consumption

Quantities of methylmercury consumed in fish depend upon both the quantity of fish consumed and the methylmercury concentration of the fish. Although they are infrequently consumed, swordfish, barracuda and shark have a much higher methylmercury concentration than other marine finfish, fresh-water finfish or shellfish. By contrast most shellfish contain low concentrations of methylmercury resulting in far lower methylmercury exposures than would occur if finfish species were chosen.

Table H-28

Consumption of Freshwater Fish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Fish methylmercury concentrations based on Bahnick et al., (1994)*

Gender	Percentage	Aged 14 Years of Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
		N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$
Males	Min	60	4 0	0 04	80	7 6	0 03	82	3 1	0 01	222	3 1	0 01
	5th		8 8	0 07		19 0	0 06		18 1	0 04		12 7	0 05
	25th		17.2	0 15		42 9	0 11		23 4	0 09		26 0	0 10
	50th		26 2	0 19		58 5	0 16		43 0	0 11		44 7	0 16
	75th		37 5	0 24		77 1	0 24		63 7	0 17		72 4	0 21
	95th		57 0	0 55		140 6	0 50		147 6	0 35		134 0	0 46
	97 5th		78 7	0 68		224 4	0 52		172 9	0 45		172 9	0 55
	99th		86 4	0 77		224 4	0 55		388 9	0 71		224 4	0 71
	Max		96 5	0 77		247 8	0 64		388 9	0 71		388 9	0 77
Females	Min	46	1 0	0 02	109	8 8	0 02	115	2 0	0 01	270	1 0	0 01
	5th		2 5	0 05		18 1	0 05		13 8	0 04		10 7	0 04
	25th		13 8	0 12		23 4	0 08		22 4	0 07		21 4	0 08
	50th		14 1	0 16		37 5	0 11		34 0	0 12		31 2	0 13
	75th		37 5	0 19		56 9	0 19		67 5	0 2		56 2	0 19
	95th		43 7	0 64		178 3	0 68		102 0	0 39		118 9	0 55
	97 5th		63 3	0 65		178 3	0 68		172 9	0 55		172 9	0 65
	99th		71 0	0 87		213 2	0 71		172 9	0 55		178 3	0 68
	Max		71 0	0 87		217 8	0 73		222 1	0 55		222 1	0 87

* Data are weighted to be representative of the U S population

Table H-29

Consumption of All Fish & Shellfish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Bahnick et al. estimates for fresh-water fish Methylmercury Concentrations*

Gender	Percentage	Aged 14 Years of Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
		N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$
Males	Min	380	1 9	< 0 01	646	1 9	< 0 01	556	2 5	< 0 01	1582	1 9	0 01
	5th		3 5	0 01		9 3	0 01		9 3	0 01		7 0	0 01
	25th		14 2	0 09		24 6	0 04		22 7	0 03		20 4	0 04
	50th		23 0	0 16		45 5	0 08		40 3	0 07		37 8	0 09
	75th		45 0	0 25		73 3	0 16		64 5	0 13		65 3	0 17
	95th		80 6	0 84		124 5	0 33		127 2	0 38		120 8	0 15
	97 5th		104 5	0 94		153 3	0 46		172 8	0 71		149 2	0 64
	99th		132 4	1 33		177 9	0 60		203 1	0 95		177 2	0 91
	Max		139 4	1 51		312 0	1 98		388 9	1 57		388 9	1 98
Females	Min	340	1 0	< 0 01	864	1 3	< 0 01	828	1 9	< 0 01	2032	1 0	0 01
	5th		4 7	0 02		7 0	0 01		7 2	0 01		7 0	0 01
	25th		14 0	0 07		18 6	0 04		18 9	0 04		18 6	0 04
	50th		23 0	0 15		28 6	0 08		31 8	0 07		28 6	0 09
	75th		37 5	0 28		55 8	0 16		56 0	0 14		52 3	0 17
	95th		67 4	0 81		110 8	0 35		107 8	0 34		106 7	0 44
	97 5		112 1	0 90		132 8	0 48		134 1	0 55		127 4	0 60
	99th		113 8	1 29		174 6	0 74		159 0	0 63		161 3	0 90
	Max		153 5	1 69		461 0	2 76		250 2	1 67		461 0	2 76

* Data weighted to reflect U S population

Table H-30

Consumption of All Fish & Shellfish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Lowe et al. estimates for fresh-water fish Methylmercury Concentrations*

Gender	Percentage	Aged 14 Years of Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
		N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$
Males	Min	380	1 9	< 0 01	646	1 9	< 0 01	556	2 5	< 0 01	1582	1 9	0 01
	5th		3 5	0 01		9 3	0 01		9 3	0 01		7 0	0 01
	25th		14 2	0 08		24 6	0 04		22 7	0 03		20 4	0 04
	50th		23 0	0 15		45 5	0 08		40 3	0 07		37 8	0 08
	75th		45 0	0 25		73 3	0 14		64 5	0 12		65 3	0 16
	95th		80 6	0 84		124 5	0 31		127 2	0 33		120 8	0 10
	97 5th		104 5	0 94		153 3	0 40		172 8	0 67		149 2	0 62
	99th		132 4	1 33		177 9	0 60		203 1	0 97		177 2	0 94
	Max		139 4	1 51		312 0	1 98		388 9	1 57		388 9	1 98
Females	Min	340	1 0	< 0 01	864	1 3	< 0 01	828	1 9	< 0 01	2032	1 0	< 0 01
	5th		4 7	0 02		7 0	0 01		7 2	0 01		7 0	0 01
	25th		14 0	0 07		18 6	0 04		18 9	0 03		18 6	0 04
	50th		23 0	0 12		28 6	0 08		31 8	0 07		28 6	0 08
	75th		37 5	0 27		55 8	0 15		56 0	0 13		52 3	0 16
	95th		67 4	0 68		110 8	0 32		107 8	0 32		106 7	0 41
	97 5		112 1	0 90		132 8	0 47		134 1	0 43		127 4	0 55
	99th		113 8	1 29		174 6	0 69		159 0	0 61		161 3	0 90
	Max		153 5	1 69		461 0	2 76		250 2	1 67		461 0	2 76

* Data weighted to reflect U S population

Table H-31
Respondents Reporting Consumption of Marine Finfish (Excluding tuna, swordfish, barracuda, and shark) in the 1989-1991 CSFII Survey Based on 3-Days Diet Records

Gender	Aged 14 Years or Younger	Aged 15 through 44 Years	Aged 45 Years or Older	Total
Males	164 (53.4%)	268 (44.5%)	271 (39.6%)	703 (44.1%)
Females	143 (46.6%)	334 (55.5%)	413 (60.4%)	890 (55.9%)
Total	307 (19.2%)	602 (37.7%)	684 (42.9%)	1593

Table H-32
Consumption (gms/day) of Marine Finfish (Excluding Tuna, Shark, Barracuda, and Swordfish) and Self-Reported Body Weight (Kg) among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only

Gender	Aged 14 Years or Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}
Males	34.4	22.3	28	46.0	28.3	80	46.7	31.8	82.9	44.1	29.0	83.6
Females	26.8	22.1	23	37.9	27.6	63	40.4	28.0	67.2	36.8	27.3	51.1

[a] Data weighted to reflect U.S. population.

Table H-33
Respondents Reporting Consumption of Tuna in the 1989-1991 CSFII Survey Based on 3-Days Diet Records

Gender	Aged 14 Years of Younger	Aged 15 through 44 Years	Aged 45 Years or Older	Total
Males	155 (50.0%)	255 (39.6%)	184 (37.6%)	594 (41.1%)
Females	155 (50.0%)	389 (60.4%)	306 (62.4%)	850 (58.9%)
Total	310 (21.5%)	644 (44.6%)	490 (33.9%)	1444

Table H-34
Consumption (gms/day) of Tuna Fish and Self-Reported Body Weight (Kg) among Respondents in the 1989-1991 CSFII Survey. Data for "Users" Only

Gender	Aged 14 years or Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}
Males	24.1	28.3	26.7	37.9	29.6	88	31.3	21.1	85	33.2	27.5	60.4
Females	24.4	21.3	23.7	28.3	20.8	57	24.7	16.3	59	26.4	19.5	48.6

[a] Data weighted to reflect U.S. population.

Table H-35
Respondents Reporting Consumption of Marine Shellfish in the 1989-1991 CSFII Survey Based on 3-Days Diet Records

Gender	Aged 14 Years or Younger	Aged 15 through 44 Years	Aged 45 Years or Older	Total
Males	41 (47.1%)	138 (44.8%)	116 (47.2%)	295 (46.0%)
Females	46 (52.9%)	170 (55.2%)	130 (52.8%)	346 (54.0%)
Total	87 (13.6%)	308 (48.0%)	246 (38.4%)	641

[a] Data weighted for U.S. population.

Table H-36
Consumption (gms/day) of Shellfish and Self-Reported Body Weights (Kg) of Respondents in the 1989-1991 CSFII Survey. Data for "Users" Only.[a]

Gender	Aged 14 Years or Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}
Males	20.7	23.9	32	38.7	38.6	87	40.8	33.6	80	42.7	36.4	72
Females	18.4	19.6	22	38.1	39.6	68	38.7	33.7	63	35.9	36.1	47

[a] Data weighted to reflect U.S. population.

Table H-37
Respondents Reporting Consumption of Shark, Barracuda, and/or Swordfish in the 1989-1991 CSFII Survey Based on 3-Days Dietary Records

Gender	Aged 14 Years or Younger	Aged 15 through 44 Years	Aged 45 Years or older	Total
Males	0	3	7	10
Females	0	7	4	11
Total	0	10	11	21

Table H-38
Consumption (gms/day) of Swordfish, Barracuda, and Shark and Self-Reported Body Weight (Kg) among Respondents in the 1989-1991 CSFII Survey. Data for "Users" Only

Gender	Aged 15 through 44 Years			Aged 45 Years or Older			Total		
	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}
Males	44.5	21.7	87.8	56.2	22.1	84.5	53.5	22.6	85.6
Females	76.3	39.7	57.2	51.9	22.4	59.1	61.5	23.7	57.8

[a] Data weighted to reflect U.S. population.

Table H-39
Respondents Reporting Consumption of Fresh-water Fish in the 1989-1991 CSFII Survey Based on 3-Days Dietary Records

Gender	Aged 14 Years or Younger	Aged 15 through 44 Years	Aged 45 Years or Older	Total
Males	60 (56.6%)	80 (42.3%)	82 (41.6%)	222 (45.1%)
Females	46 (43.4%)	109 (57.7%)	115 (58.4%)	270 (54.9%)
Total	106 (21.5%)	189 (38.4%)	197 (40.0%)	492

[a] Data weighted to reflect U.S. population.

Table H-40
Consumption (gms/day) of Fresh-water Fish and Self-Reported Body Weight (Kg) among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only

Gender	Aged 14 Years or Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}	Mean	SD	kg_{bw}
Males	30 4	17 2	29 0	70 4	45 0	77 4	58 7	54 5	83 2	55 6	45 4	64 5
Females	22.2	14.7	21 2	49 6	40.4	64 2	48 1	36 3	70 9	43 8	36 6	54.6

[a] Data weighted to reflect U.S. population.

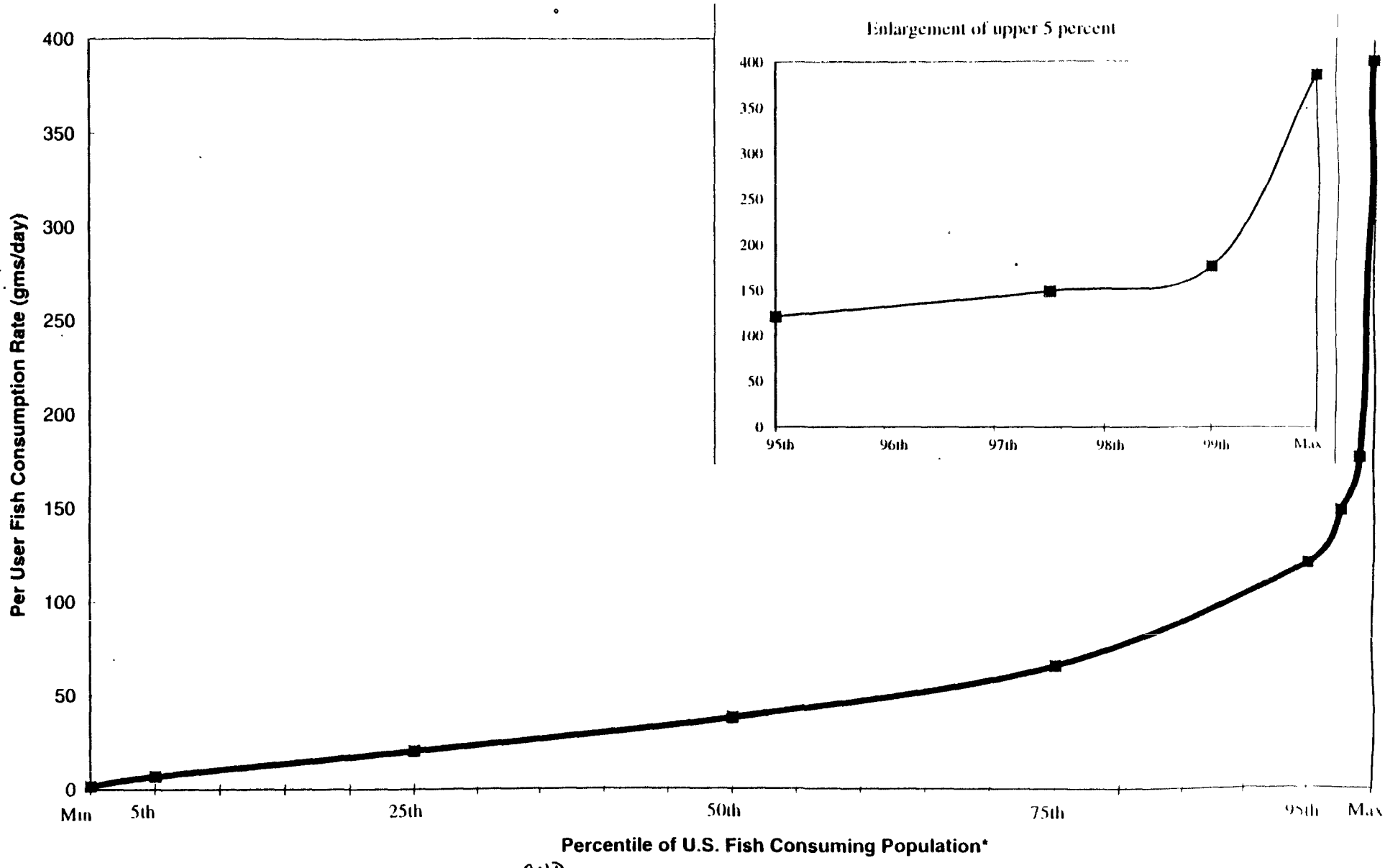
Table H-41

Consumption of Freshwater Fish (gms/day) and Methylmercury per Kg body weight from Fish among Respondents of the 1989-1991 CSFII Survey. Data for "Users" Only. Fish methylmercury concentrations based on Lowe et al., (1994)

Gender	Percentage	Aged 14 Years of Younger			Aged 15 through 44 Years			Aged 45 Years or Older			Total		
		N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$	N	Fish gms/day	Methyl-mercury $\mu g/kg_{bw}$
Males	Min	60	4 01	0 02	80	7 6	0 01	82	3 1	0 01	222	3 1	0 010
	5th		8 8	0 05		19 0	0 03		18 1	0 02		12 7	0 03
	25th		17 2	0 08		42 8	0 07		23 4	0 04		26 0	0 06
	50th		26 2	0 11		58 5	0 11		43 0	0 07		41 7	0 090
	75th		37 5	0 23		77 1	0 17		63 7	0 13		72 4	0 17
	95th		57 0	0 30		140 6	0 27		147 6	0 26		134 0	0 30
	97 5th		78 7	0 30		224 4	0 27		172 9	0 39		172 9	0 33
	99th		86 4	0 42		224 4	0 33		388 9	0 40		224 4	0 42
	Max		96 5	0 53		247 8	0 66		388 9	0 44		388 9	0 66
Females	Min	46	1 0	0 01	109	8 8	0 02	115	2 0	<0 01	270	1 0	0 01
	5th		2 5	0 02		18 1	0 03		13 8	0 02		10 7	0 02
	25th		13 8	0 06		23 4	0 04		22 4	0 04		21 4	0 05
	50th		14 1	0 07		37 5	0 09		34 0	0 07		31 2	0 07
	75th		37 5	0 16		57 9	0 13		67 5	0 13		56 2	0 13
	95th		43 7	0 33		178 3	0 29		102 0	0 25		118 9	0 29
	97 5th		63 3	0 36		178 3	0 30		172 9	0 30		172 9	0 30
	99th		71 0	1 06		213 2	0 40		172 9	0 31		178 3	0 36
	Max		71 0	1 06		217 8	0 69		222 1	0 33		222 1	1 06

* Data are weighted to be representative of the U S population

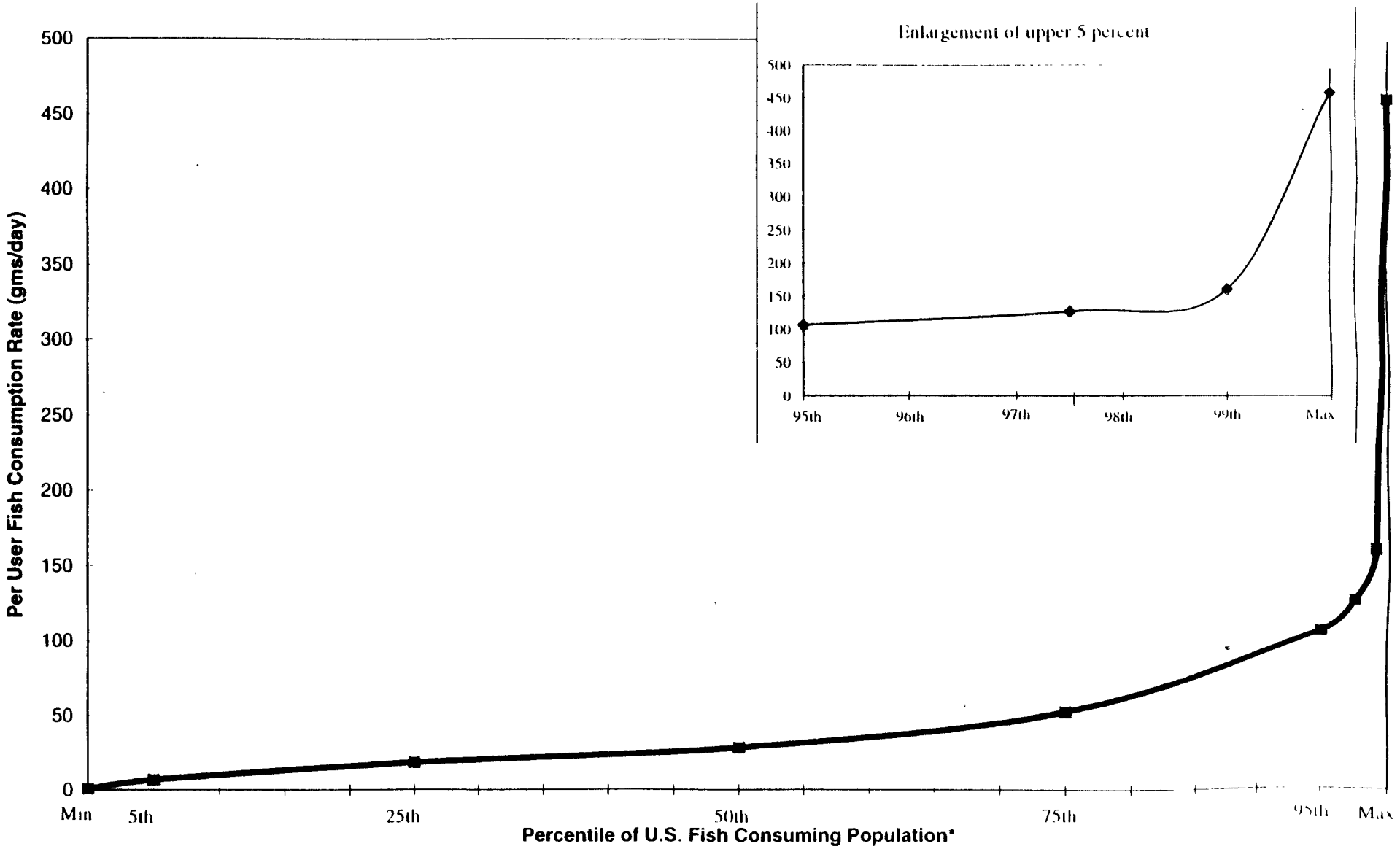
Figure H-2
Fish and Shellfish Consumption Rates: General U.S. Male Population Who Reported Consuming Fish and Shellfish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91 ~~provided by~~ AND Bahnick et al., and Lowe et al

Note: The percentile indicates the fraction of the surveyed population that consumed ≤ the amount of fish (gms/day) shown on the Y axis. For example, at the 25th percentile, 25 percent of the surveyed population consumed ≤ 20.4 gms of fish per day

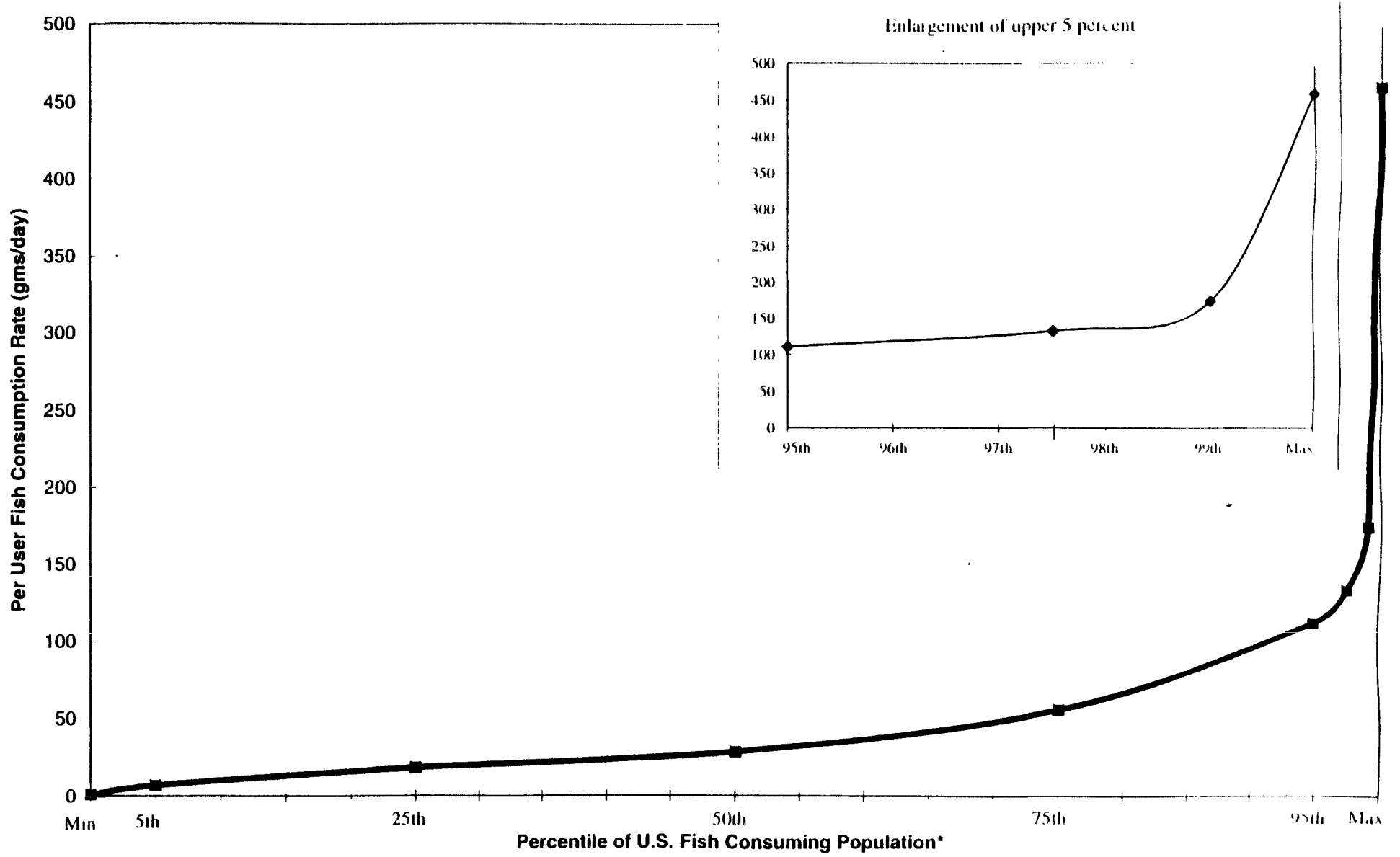
Figure H-3
Fish and Shellfish Consumption Rates: General U.S. Female Population Who Reported Consuming Fish and Shellfish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91 ~~provided by~~ AND Bahnick et al., and Lowe et al

Note: The percentile indicates the fraction of the surveyed population that consumed ≤ the amount of fish (gms/day) shown on the Y-axis. For example, at the 25th percentile, 25 percent of the surveyed population consumed ≤ 18.6 gms of fish per day

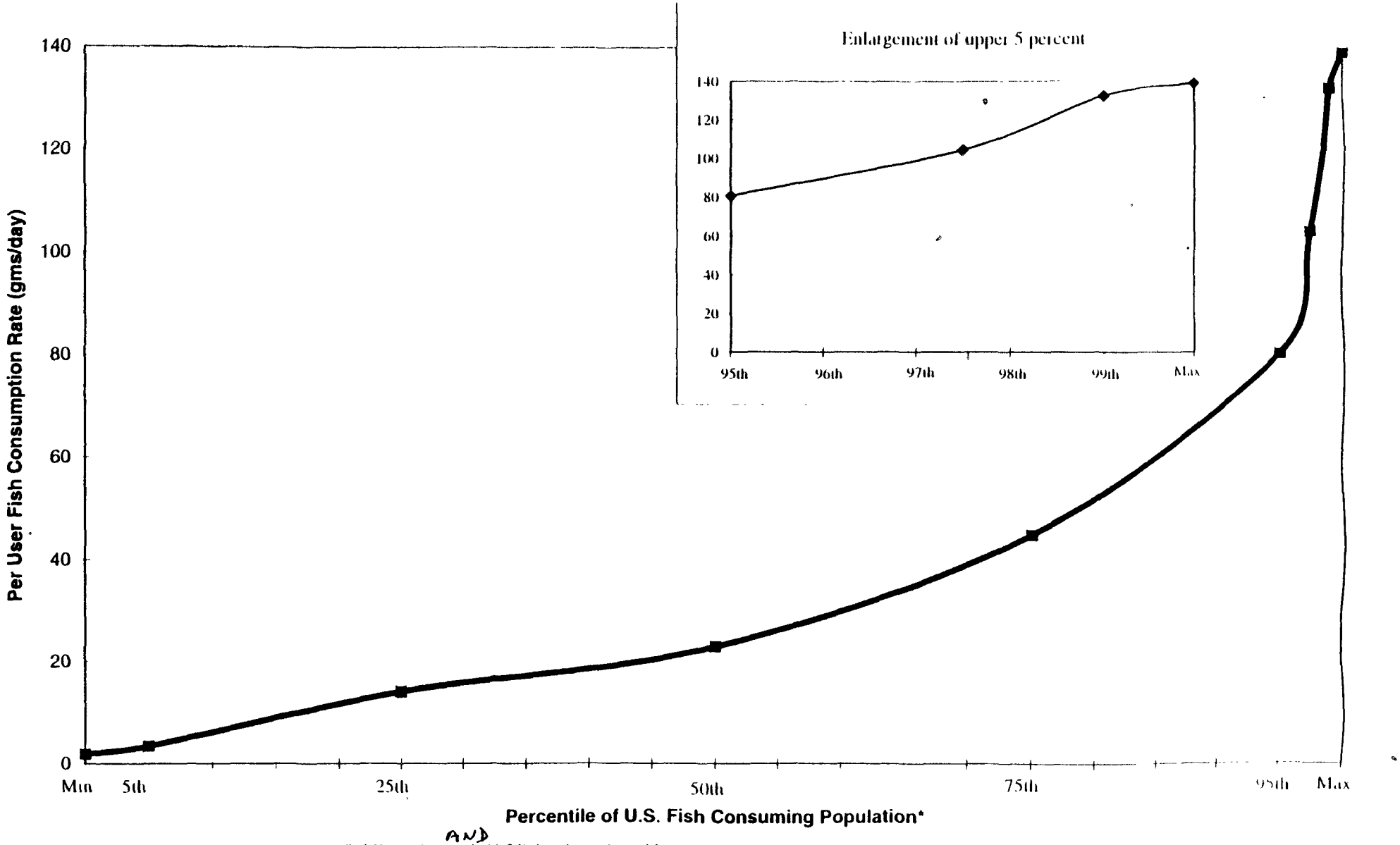
Figure H-4
Fish and Shellfish Consumption Rates: U.S. Women of Child-Bearing Age Who Reported Consuming Fish and Shellfish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91 ~~provided by~~ Bahnick et. al , and Lowe et al

Note: The percentile indicates the fraction of the surveyed population that consumed ≤ the amount of fish (gms/day)shown on the Y axis For example, at the 25th percentile, 25 percent of the surveyed population consumed ≤ 18 6 gms of fish per day

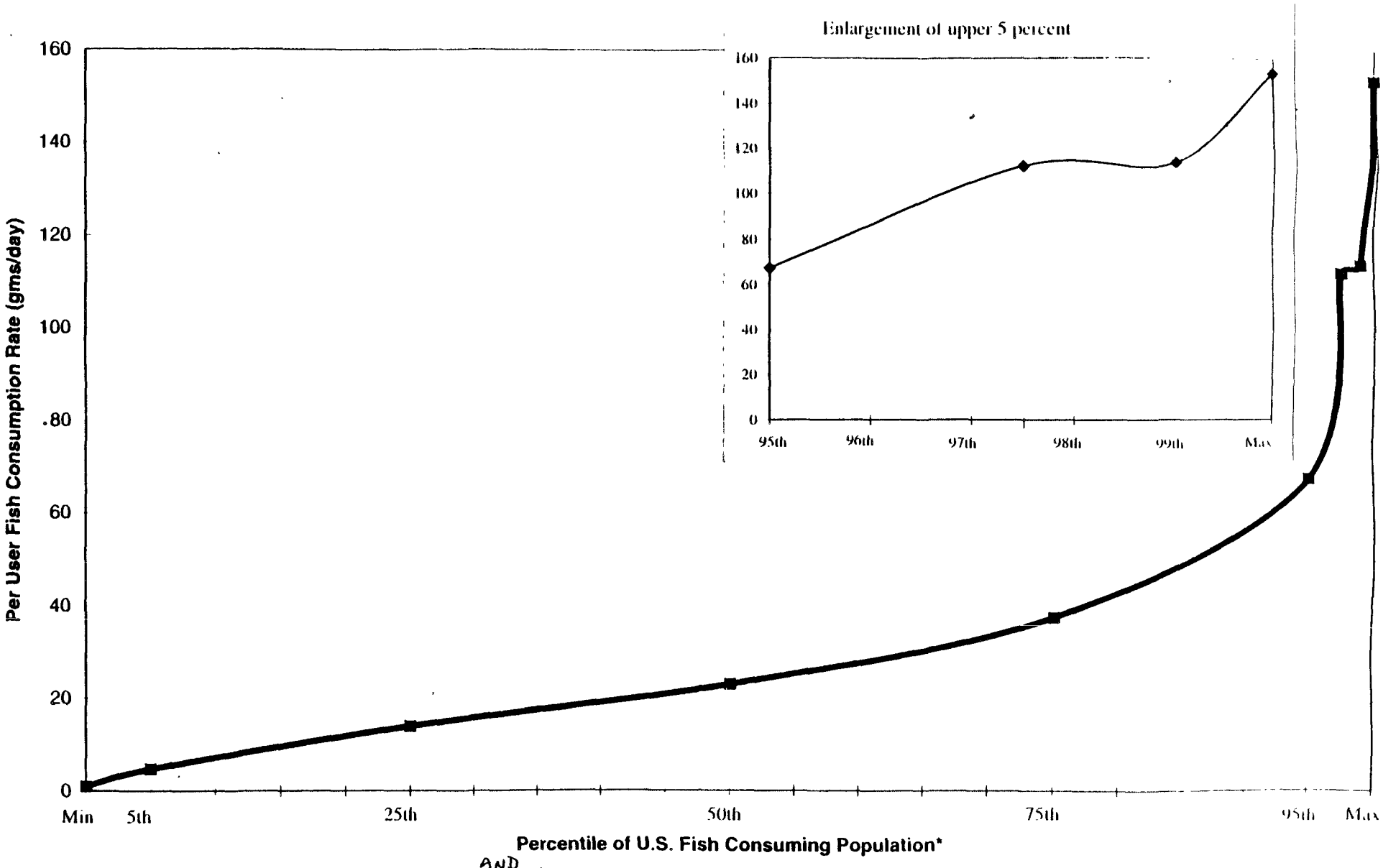
Figure H-5
Fish and Shellfish Consumption Rates: U.S. Male Children Who Reported Consuming Fish and Shellfish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91 ~~provided by~~ AND Bahnick et al., and Lowe et al.

Note: The percentile indicates the fraction of the surveyed population that consumed ≤ the amount of fish (gms/day) shown on the Y-axis. For example, at the 25th percentile, 25 percent of the surveyed population consumed ≤ 14.2 gms of fish per day.

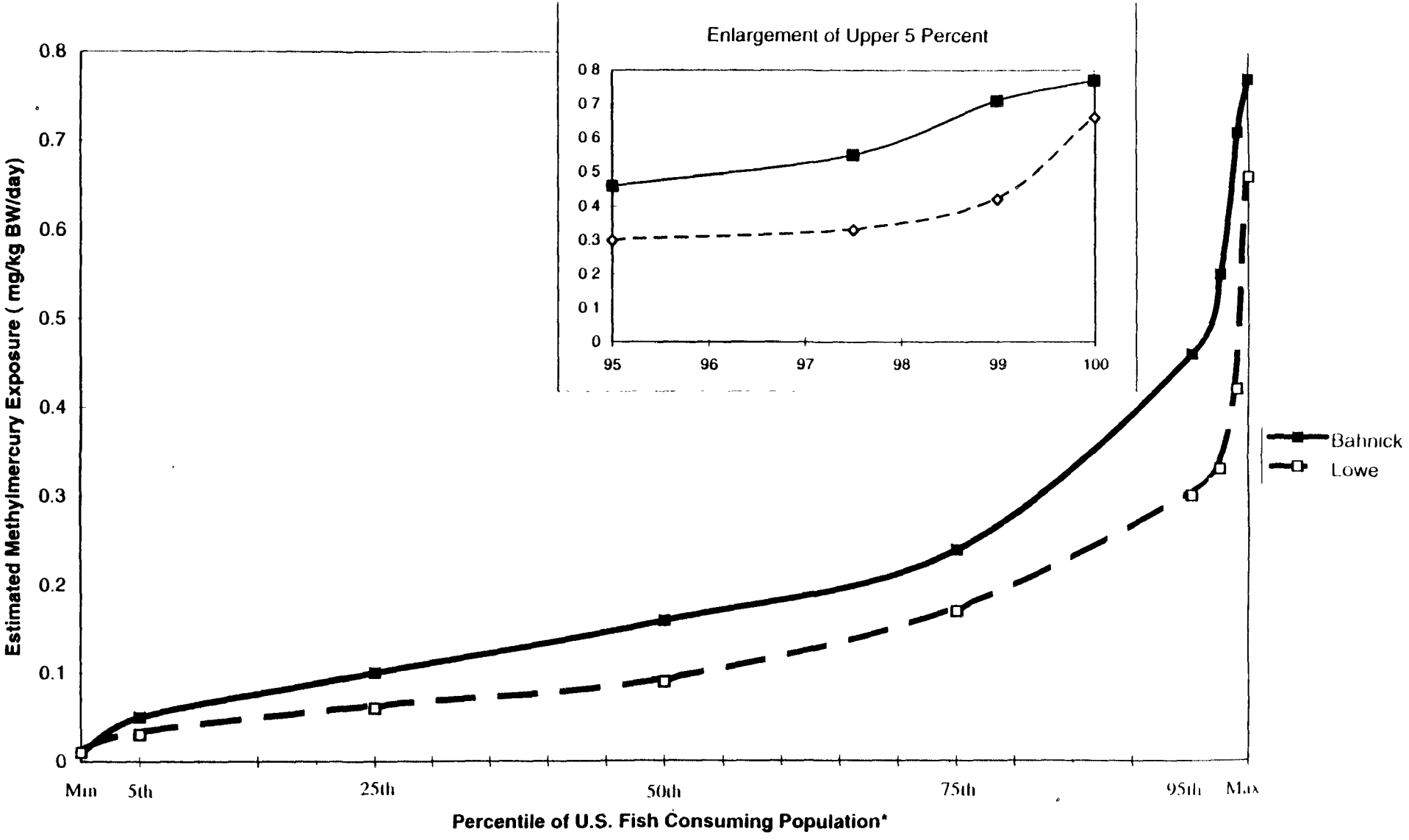
Figure H-6
Fish and Shellfish Consumption Rates: U.S. Female Children Who Reported Consuming Fish and Shellfish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91 ~~provided by~~ AND by Bahnick et al., and Lowe et al

Note: The percentile indicates the fraction of the surveyed population that consumed ≤ the amount of fish (gms/day) shown on the Y axis. For example, at the 25th percentile, 25 percent of the surveyed population consumed ≤ 14.0 gms of fish per day.

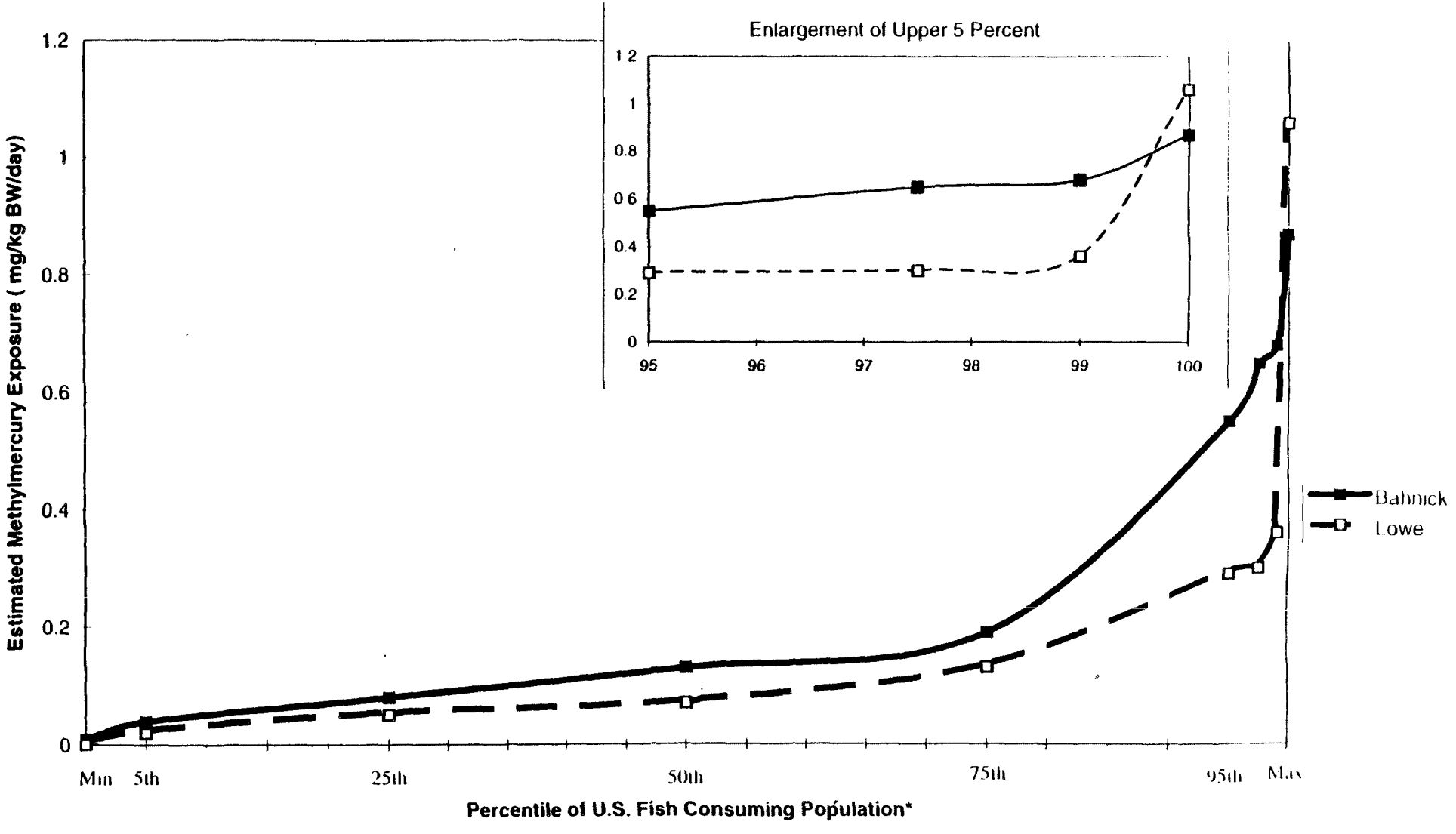
Figure H-7
Estimated Exposure to Methylmercury from Freshwater Fish: General U.S. Male Population Who Reported Consuming Freshwater Fish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91, and mercury concentrations provided by National Marine Fisheries Service, Bahnick et al., and Lowe et al.

Note: The percentile indicates the fraction of the surveyed population with an estimated methylmercury exposure ≤ the value shown on the Y axis. For example, at the 25th percentile, 25 percent of the surveyed population had an estimated methylmercury exposure ≤ 0.1 µg/kg BW/day.

Figure H-8
Estimated Exposure to Methylmercury from Freshwater Fish: General U.S. Female Population Who Reported Consuming Freshwater Fish in a Cross-Sectional Survey With a Three-Day Sampling Period

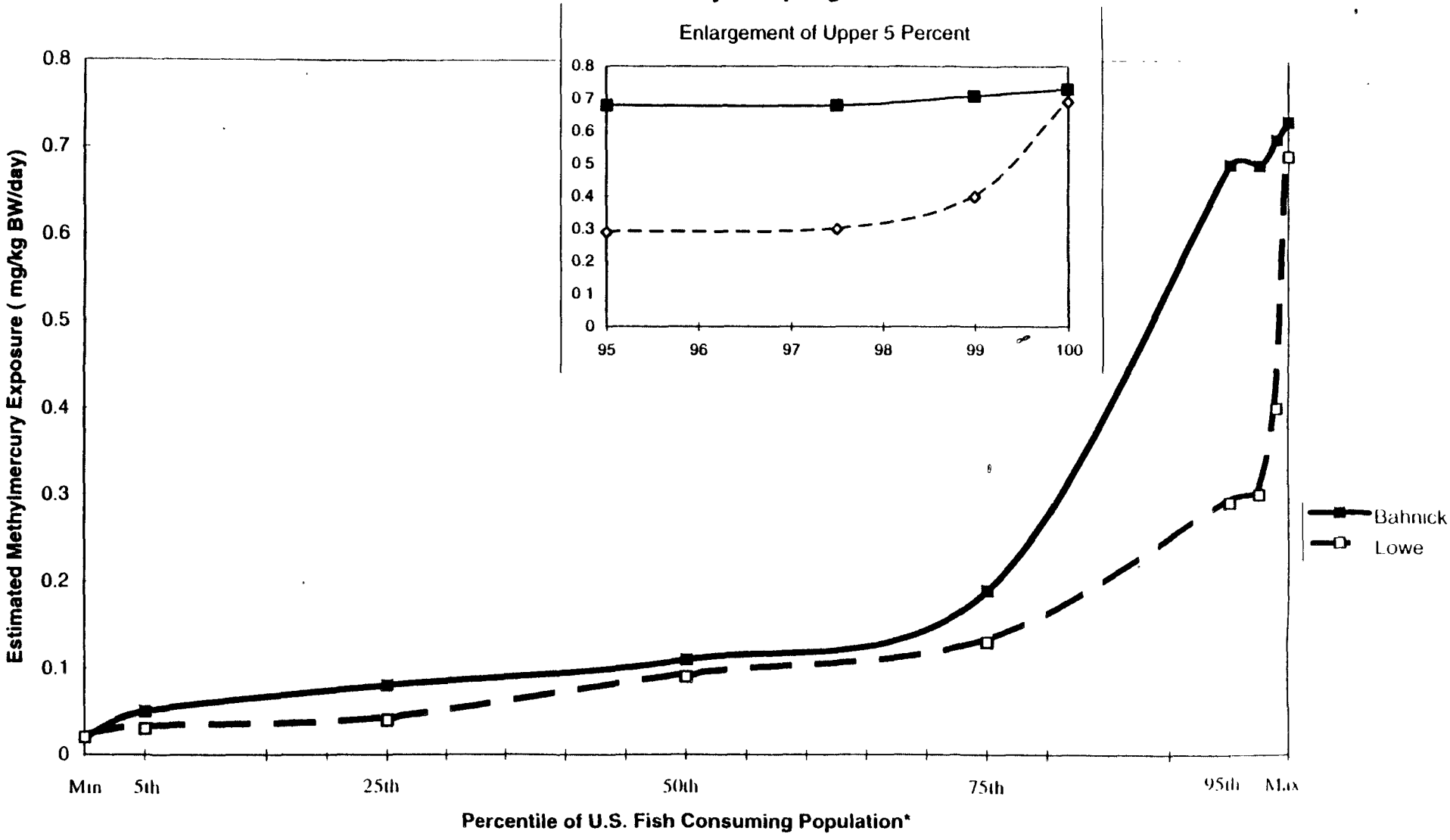


* Estimated from fish consumption data in CSFII 89/91, and mercury concentrations provided by National Marine Fisheries Service. Bahnick et al., and Lowe et al

Note. The percentile indicates the fraction of the surveyed population with an estimated methylmercury exposure ≤ the value shown on the Y axis

For example, at the 25th percentile, 25 percent of the surveyed population had an estimated methylmercury exposure ≤ 0.1 µg/kg BW/day

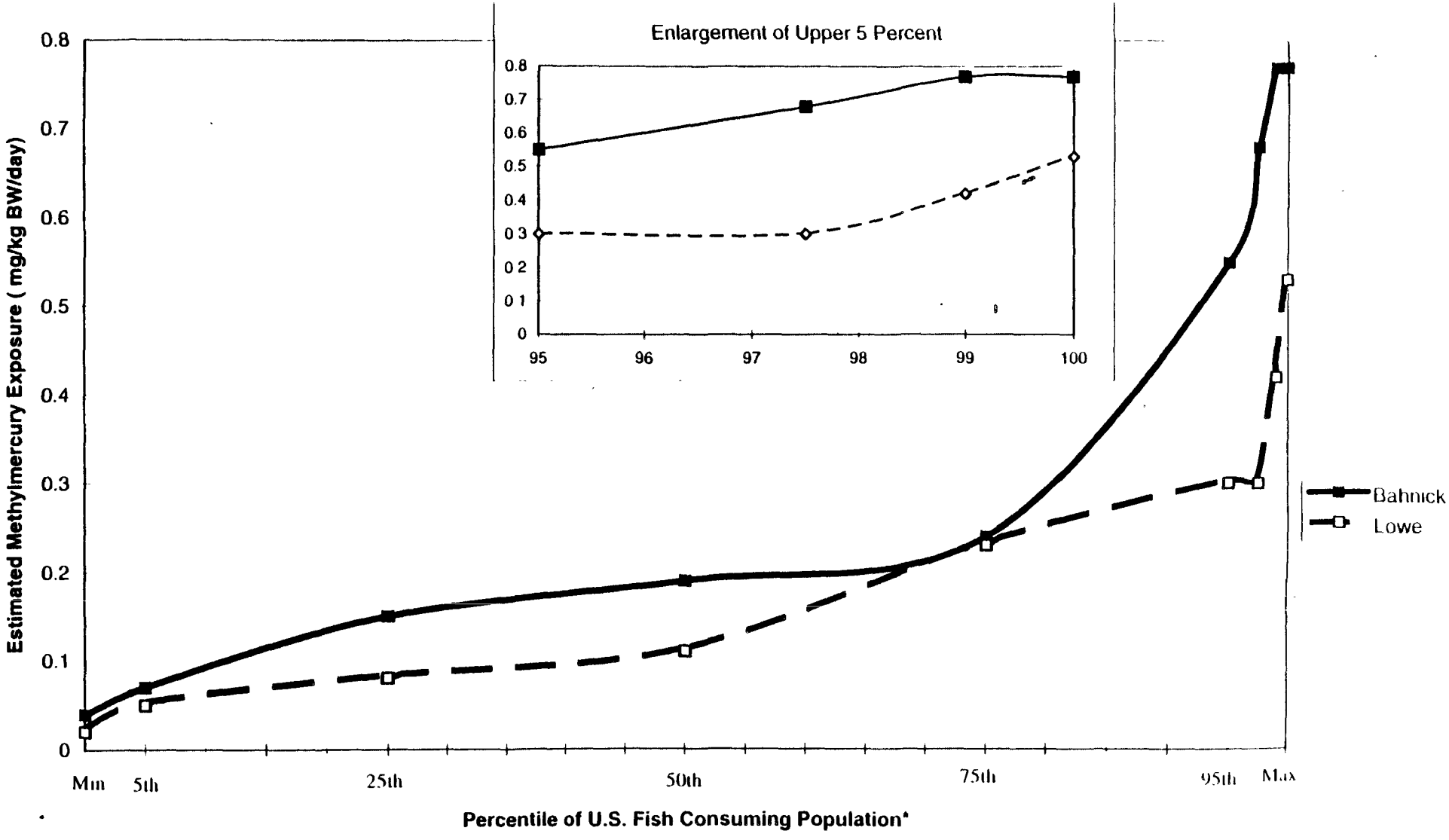
Figure H-9
Estimated Exposure to Methylmercury from Freshwater Fish: General U.S. Women Who Reported Consuming Freshwater Fish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91, and mercury concentrations provided by National Marine Fisheries Service, Bahnick et al., and Lowe et al.

Note: The percentile indicates the fraction of the surveyed population with an estimated methylmercury exposure ≤ the value shown on the Y axis. For example, at the 25th percentile, 25 percent of the surveyed population had an estimated methylmercury exposure ≤ 0.1 µg/kg BW/day.

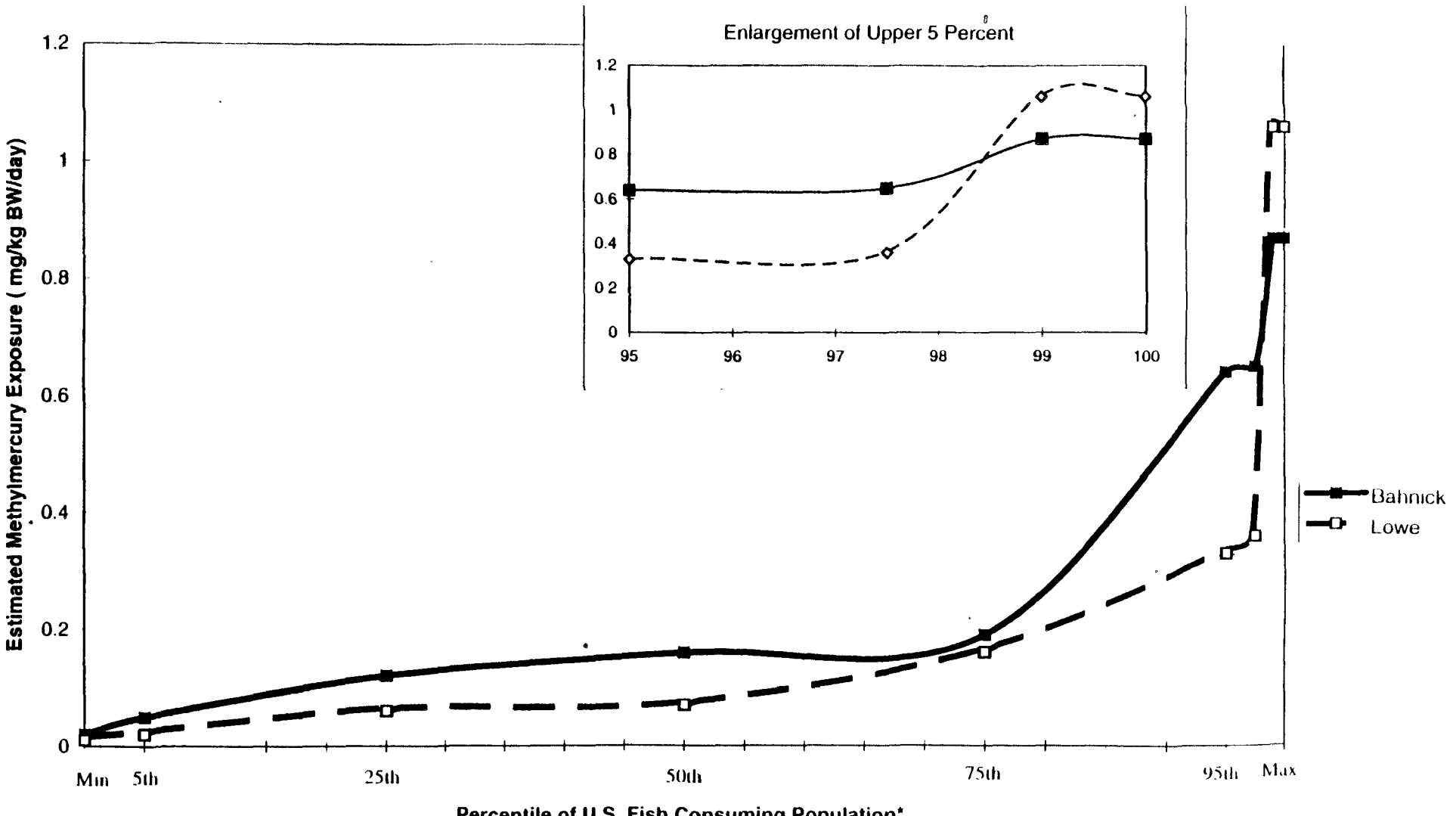
Figure H-10
Estimated Exposure to Methylmercury from Freshwater Fish: General U.S. Male Children Who Reported Consuming Freshwater Fish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91, and mercury concentrations provided by National Marine Fisheries Service, Bahnick et al., and Lowe et al.

Note: The percentile indicates the fraction of the surveyed population with an estimated methylmercury exposure ≤ the value shown on the Y axis. For example, at the 25th percentile, 25 percent of the surveyed population had an estimated methylmercury exposure ≤ 0.1 µg/kg BW/day.

Figure H-11
Estimated Exposure to Methylmercury from Freshwater Fish: General U.S. Female Children Who Reported Consuming Freshwater Fish in a Cross-Sectional Survey With a Three-Day Sampling Period



* Estimated from fish consumption data in CSFII 89/91, and mercury concentrations provided by National Marine Fisheries Service, Bahnick et al., and Lowe et al.

Note: The percentile indicates the fraction of the surveyed population with an estimated methylmercury exposure ≤ the value shown on the Y axis. For example, at the 25th percentile, 25 percent of the surveyed population had an estimated methylmercury exposure ≤ 0.1 µg/kg BW/day.

4. CONCLUSIONS ON METHYLMERCURY INTAKE FROM FISH

Methylmercury intakes calculated in Appendix H to Volume III have been developed for a nationally based rather than site-specific estimates. The CSFII/89-91 from USDA was designed to represent the United States population. The concentrations of methylmercury in marine fish and shellfish were from a data base that is national in scope. Data on fresh-water finfish were taken from two large studies that sampled fish at a number of sites throughout the United States. The extent of applicability of these data to site-specific assessments must rest with the professional judgments of the assessor. Because of the magnitude of anthropogenic, ambient mercury contamination, the estimates of methylmercury from fish do not provide a value that reflects methylmercury from nonindustrial sources. "Background" values imply an exposure against which the increments of anthropogenic activity could be added. This is not the situation due to release of substantial quantities into the environment.

Consumption of fish is much higher when expressed "per user" rather than "per capita." A U.S. fish consumption rate of 6.5 grams per day is the default value that has been used in the calculation of human health-related criteria for mercury in water; for example, the Ambient Water Quality Criteria (U.S. EPA 1980). This value is based on data from the National Purchase Diary Survey conducted in the United States during the period 1973 and 1974. This survey estimated a non-marine fish consumption rate for the United States population. The overall fish consumption rate from this survey was 14.3 g/day (U.S. EPA 1990, Exposure Factor Handbook). This rate is a per capita rate averaged over the entire population including fish-eaters and nonfish-eaters.

Conclusions on methylmercury consumption from fish must include consideration of variability and uncertainty in these estimates. Because the vast majority of methylmercury intake occurs via fish, the first consideration is how accurately fish consumption patterns over a three-day period mirror long-term fish consumption patterns. Data from CSFII 89/91 yield "per capita" fish consumption data that are comparable to those reported in other surveys. (Data indicate that 30.9% of respondents reported consuming at least one fish dish during the three-day period.) This does not mean that all of the remaining 69.1% of people avoid fish consistently, rather that fish did not appear as a dietary item during the three-day survey period. If the survey were continued for a longer period, a higher percent of individuals would report consuming fish. It is uncertain how much this percent would increase. Because the CSFII/89-91 is conducted over all seasons of the year seasonality is not expected to be an issue in generalizability of the three-day fish consumption data.

An additional consideration in evaluating the usefulness of three-day exposure data to the question of methylmercury intake from fish concerns the ability of a respondent to specify the species of fish consumed. Longer survey periods could have variable influences on species specificity questions. If the longer survey period is achieved by food history questionnaire, it is likely that the ability to specify species of fish consumed would be reduced. If the longer survey period is achieved by written dietary record, typically the diet is simplified to reduce recording burden on the respondent. Either of these outcomes (difficulty in remembering species of fish consumed, or diet simplification in response to recording burden) would reduce the accuracy of species identification and increase the uncertainty in estimating methylmercury intake from fish.

A second area of variability and uncertainty includes differences in rates of food consumption based on age group of the respondent. If data on methylmercury intake are expressed on a "per kilogram body weight" basis, the higher energy and protein requirements of children are associated with a higher intake of fish (with accompanying methylmercury) relative to body weight. This difference is largely a reflection of the higher caloric requirements of growing children. For example,

the recommended caloric intake for a child is approximately as high as that of an adult female. The average body weight reported for children in CSFII 89/91 was 25 kilograms in contrast with the adult for women of 63 kg. It should be noted that self-reported body weights were utilized in these calculations rather than default values.

When methylmercury intake is expressed on a per kilogram self-reported body weight basis children's exposure is approximately two-to-three times that of the adult. Because of limitations in the methylmercury toxicology data base, the relevance of these differences in exposure to maintaining children's health is uncertain. There are no data on which to judge whether or not children differ from adults in susceptibility to methylmercury toxicity. The extent of the effects on young children of postnatal exposure to methylmercury has not been evaluated.

A third major source of variability and uncertainty in estimating methylmercury exposure from fish includes the type of fish consumed. The data from National Marine Fisheries Service indicate substantial variation in the mercury concentrations in fish both within an across species. In these data analyses the mean value was used in the calculations. Although the data were gathered across approximately two decades, comparison of these values with FDA compliance data has yielded the broad general assessment that the concentrations in the NMFS data bases are not different from current mercury concentration in fish. This may reflect the large reservoir of mercury that is present is present in environmental media, especially sediments, that continues to supply mercury to the aquatic food web.

For species such as tuna a large body of data on mercury concentrations exists resulting in a broadly based estimate of central tendency. For other species of fish, for example ray, only a small number of samples have been analyzed for mercury concentrations. Consequently confidence in the variability of methylmercury intake from some fish species (e.g., ray) is much lower than for other species (e.g., tuna). For fresh-water fish, in general, two distinctly different data sets were identified: Lowe et al. and Bahnick et al. Consequently calculations were carried out using both of these data sets. Many factors may contribute to the divergence among these data. It is uncertain which or whether either data set best represents methylmercury intake from fresh-water fish for the general United States population.

The last area of uncertainty and variability to be addressed concerns the highest consumers of fish. Among CSFII 89/91 respondents, 97% of fish consumers reported eating only one fish meal during the three-day period. The variability in fish intake reflected different portion sizes of fish consumed. Differences in body weight and species of fish selected would then determine the range of intakes of methylmercury on a per kilogram body weight basis. The remaining 3% of the fish-consuming population ingested more than one fish meal in the three-day period. Whether or not this group of respondents represent a subpopulation who are the very highest consumers of fish is uncertain. Based on reported high levels of fish consumption in studies of anglers and Native American populations, it is certain that such high consumers of fish are part of specialized subpopulations. In addition to being part of groups defined by their cultural or geographic membership in fish-catching subpopulation, individuals may also consume high levels of fish due to health considerations (e.g., efforts to reduce likelihood of cardiovascular disease) or reduce caloric intake (e.g., weight watchers). Whatever the reason, there is a portion of the United States general population and some subpopulations who frequently consume high quantities of fish.

Issues dealing with confidence in data on the methylmercury concentration of fish consumed include these:

- In a number of situations individuals cannot identify with accuracy the species of fish consumed. The USDA Recipe File Data Base has "default" fish species specified if the respondent does not identify the fish species consumed. There is no way, however to estimate the magnitude of uncertainty encountered by misidentification of fish species by the survey respondents.

- The data base used to estimate methylmercury concentrations in marine fish and shellfish was provided by the National Marine Fisheries Service. This data base has been gathered over approximately the past 20 years and covers a wide number of species of marine fish and shellfish. The number of fish samples for each species varies but typically exceeds 20 fish per species.

- The analytical quality of the data base has been evaluated by comparison of these data with compliance samples run for the Food and Drug Administration. The National Academy of Sciences' Report on "Seafood Safety" and the Food and Drug Administration have found this data base from NMFS to be consistent with 1990s data on methylmercury concentrations in fish.

- The methylmercury concentrations in fresh-water fish come from two publications, each giving data that represent fresh-water fish from a number of locations. These data were gathered between the early 1980s and early 1990s. These surveys by U.S. EPA (1992)/Bahnick et al. (1994) and Lowe et al. (1985) report different mean mercury concentrations: 0.260 ppm mercury (wet weight) and 0.114 ppm mercury (wet weight), respectively. The extent to which either of these data sets represents nationally based data on fresh-water fish methylmercury concentrations remains uncertain. Utilizing either the Bahnick et al. (1994) or Lowe et al. (1985) data set, exposures above the methylmercury reference dose (R/D) of 0.1 μg/Kg body weight/day occur among the subpopulation relevant to developmental effects of methylmercury (i.e., women of child-bearing age), when these concentration data are combined with the fish consumption data. Using the higher estimate of fish methylmercury concentration based on the Bahnick et al. (1994) data results in the prediction of a larger number in the vulnerable population being exposed above this level. Even when the Lowe, et al. (1985) data are used, the 75th percentile of this group is exposed to levels over 0.1 μg/kg body weight/day. It should be noted that this is not a site-specific assessment.

Because methylmercury is a developmental toxin, a subpopulation of interest is women of child-bearing age. In this analysis of methylmercury intake, dietary intakes of women aged 15 through 44 years were used to approximate the diet of the pregnant woman. From data on Vital and Health Statistics, it has been determined that 9.5% of women of reproductive age can be anticipated to be pregnant within a given year. Generally food intake increases during pregnancy (Naismith, 1980). Information on dietary patterns of pregnant women has been assessed (among other see Bowen, 1992; Greeley et al., 1992). Most of these analyses have focussed on intake of nutrients rather than contaminants. It is uncertain whether or not pregnancy would modify quantities and frequency of fish consumed beyond any increase that may result from increased energy (i.e., caloric) intake that typically accompanies pregnancy.

5. ACKNOWLEDGMENTS

U S. EPA acknowledges the assistance of Dr Malcomb Meaburn from the National Marine Fisheries Laboratory in Charleston. South Carolina for advice on use of the NMFS data base. The assistance of Dr Betty Perloff and Dr Jacob Exler of the United States Department of Agriculture in interpreting the USDA Recipe File and USDA's information on fish composition is acknowledged

The assistance of David Reisman. David Oberlin. and Dr. Bruce Peirano of the National Center for Environmental Assessment - Cincinnati in compiling these data is also acknowledged.

6. REFERENCES

Anderson, Sue, Life Sciences Research Office.

Ayotte P., Dewailly E., Bruneau S., Careau H., and Vezina A. (1995). Arctic air pollution and human health: what effects should be expected? Science of the Total Environment 160/161:529-537.

Bahnick, D., C. Sauer, B. Butterworth and D. Kuehl. 1994. A National Study of Mercury Contamination of Fish. Chemosphere 29(3):537-546.

Bloom, N. S. (1992). On the Chemical Form of Mercury in Edible Fish and Marine Invertebrate Tissue. Can. J. Fisher. Aq. Sci. 49:1010-1017.

Bowen, D.J. (1992). Taste and food preference changes across the course of pregnancy. Appetite 19: 233-242.

Bjerregaard, P. (1995). Health and environment in Greenland and other circumpolar areas. The Science of the Total Environment. 160/161: 521-527.

Columbia River Inter-Tribal Fish Commission (1994) A Fish Consumption Survey of the Umtilla, Nez Perce, Yakama and Warm Springs Tribes of the Columbia River Basin. Technical Report 94-3. October, 1994.

Connelly, N.A., T.L. Brown and B.A. Knuth. New York Statewide Angler Survey 1988. 1990. New York State Department of Environmental Conservation. NYDEC, Albany.

Cramer GM (1994) Exposure of U.S. consumers to methylmercury from fish. pps. 103-118. In: DOE/FDA/EPA Workshop on Methylmercury and Human Health Eds: Moskowitz PD, Saroff L, Bolger M, Cicmanec J, and Durkee J. Conference Number 9403156. Published through: Biomedical and Environmental Assessment Group, Brookhaven National Laboratory, Upton, New York.

Crispin-Smith J, Turner MD, Marsh DO. Project III. Hair methylmercury levels in women of childbearing age.

Crochetti AF, Guthrie HA (1982) Eating Behavior and Associated Nutrient Quality of Diets. Final Report for US Department of Agriculture Contract 53-22U4-9-192, The Human Nutrition Center, Science and Education Administration US Department of Agriculture. Anarem Systems Research Corporation, 31 Union Square West, New York, NY 10003.

Ebert, E.S., N.W. Harrington, K.J. Boyle, J.W. Knight and R.E. Keenan. 1993. Estimating Consumption of Freshwater Fish among Maine Anglers. North American Journal of Fisheries Management. 13:737-745.

FDA Compliance Testing as described in the NMFS data base.

Fiore, B.J., H.A. Anderson, L.P. Hanrahan, L.J. Olson and W.C. Sonzogni. 1989. Sport Fish Consumption and Body Burden Levels of Chlorinated Hydrocarbons: A Study of Wisconsin Anglers. Arch. Environ. Health. 44(2):82-88.

Fitzgerald E.F., S. Hwang, K. Brix, B. Bush, K. Cook, and P. Worsick. 1995. Fish PCB Concentrations and Consumption Patterns Among Mohawk Women at Akwesanse. J. Exposure Analysis and Environmental Epidemiology: 5(1):1-19.

Greeley S, Storbakken L, and Magel R (1992) Use of a modified food frequency questionnaire during pregnancy. J. Amer. Col. Nutr. 11: 728-734.

Hall RA, Zook EG, and Meaburn GM (1978) National Marine Fisheries Service Survey of Trace Elements in the Fishery Resource. NOAA Technical Report NMFS SSRF-721, National Technical Information Service No. PB 283 851, March. U.S. Government Printing Office, Washington, DC cited in Seafood Safety. Chapter 6. "Chemical Health Risk Assessment - Critique of Existing Practices and Suggestions for Improvement", pp. 187 - 281. National Academy of Sciences Press, Washington, DC, 1991.

Higuchi, W.K. and Pooley, S.G. 1985. Hawaii's Retail Seafood Volume. Administrative Report H-85-6. Southwest Fisheries Center. Honolulu Laboratory, national Marine Fisheries Service. P.O. Box 3830, Honolulu, HI 96812.

Hudgins, L.L. 1980. Per capita annual utilization and consumption of fish and shellfish in Hawaii, 1970-77. Marine Fisheries Review [February; No Volume Cited]; pgs. 16-20.

Hovinga, M.E., M. Sowers and H.B. Humphrey. 1993. Environmental Exposure and Lifestyle Predictors of Lead, Cadmium, PCB and DDT Levels in Great Lakes Fish Eaters. Archives of Environmental Health 48(2):98-104.

Hovinga, M.E., M. Sowers and H.B. Humphrey. 1992. Historical Changes in Serum PCB and DDT Levels in an Environmentally-Exposed Cohort. Arch. Environ. Contam. Toxicol. 22: 362-366.

Karvetti, R. and Knuts, L. 1985. Validity of the 24-hour recall. J. Amer. Dietet. Assoc. 85: 1437-1442.

Kinloch, D., Kuhnlein, H., and Muir, D.C. 1992. Inuit foods and diet: a preliminary assessment of benefits and risks. The Science of the total Environment 122: 247-278.

Lowe TP, May TW, Brumbaugh WG, and Kane DA (1985) National Contaminant Biomonitoring Program: Concentrations of seven elements in fresh-water fish, 1978-1981. Arch. Environ. Contamin. Toxicol. 14: 363-388.

Morgan, J N , M R. Berry Jr , and R L. Graves. 1994 Effects of Native American cooking practices on total mercury concentrations in walleye Presented at ISEE/ISEA Joint Conference September 18-21. 1994

Naismith DJ (1980) Maternal nutrition and the outcome of pregnancy - a critical appraisal. Proc. Nutrition Society 39. 1-11.

National Research Council/National Academy of Sciences (Committee on Evaluation of the Safety of Fishery Products). 1991. Seafood Safety. Ed: Ahmed FE. National Academy Press. Washington. DC.

NMFS (National Marine Fisheries Service). The current publically-available National Marine Fisheries Service Data base was supplied to U.S. EPA via fax from Malcolm Meaburn (Charleston Laboratory/Southeast Fisheries Science Center/National Marine Fisheries Service/National Oceanic and Atmospheric Administration/U.S. Dept. Of Commerce) to Kathryn Mahaffey (Environmental Criteria and Assessment Office-Cincinnati.OH/Office-of Health and Environmental Assessment/Office of Research and Development/U.S. Environmental Protection Agency). February 23, 1995.

NOAA (1978) as described in the NMFS data base.

Nobmann ED, Byers T, Lanier AP, Hankin JH, and Jackson MY (1992) The diet of Alaska Native adults: 1987-1988. American Journal of Clinical Nutrition 55: 1024-1032:

Nusser SM, and Guenther PM (1995) Estimating usual food intake distributions. Paper presented at the Second International Dietary Assessment Conference, January 22-24, 1995, Boston, Mass.

Parkinson, A.J., Cruz, A.L., Heyward, W.L., Bulkow, L.R., Hall, D., Barstaed, L., and Connor, W E. 1994. Elevated concentrations of plasma omega-3 polyunsaturated fatty acids among Alaskan Eskimos. Amer. J. Clin. Nutr. 59:383-388.

Peterson DE, Kanarek MS, Keykendall MA, Diedrich JM, Anderson HA, Remington PL, and Sheffy TB (1995) Fish consumption patterns and blood mercury levels in Wisconsin Chippewa Indians. Archives of Environmental Health 49(11): 53-58.

Pierce, R., D. Noviello and S. Rogers. 1981. Commencement Bay Seafood Consumption Report. Preliminary Report. Tacoma, WA: Tacoma-Pierce County Health Department. As cited in U.S. EPA (1989). Exposure Factors Handbook. EPA/600/8-89/043.

Puffer, H. 1981. Consumption rates of potentially hazardous marine fish caught in the metropolitan Los Angeles area. EPA Grant #R807 120010.As cited in U.S. EPA (1989). Exposure Factors Handbook. EPA/600/8-89/043.

Putnam, J.J. 1991. Food Consumption, 1970-1990. Food Review 14(3):2-12. July-September.

Richardson, M., M. Mitchell, S.Coad and R. Raphael. 1995. Exposure to Mercury in Canada: A Multimedia Analysis. Water, Air and Soil Pol. 80:21-30.

Sekerke, H.J., R.L. Denger, C.M. Adams and S.D. Moss. 1994. Seafood Consumption Survey in Florida. Abstract presented at the Society of Toxicology 33rd Annual Meeting (Vol. 14 No. 1. March, 1994.) Abstract No. 563.

Silverman, D L., Reis, G J., Sacks, F M., Boucher, T.M., and Pasternak, R.C. 1990. Usefulness of plasma phospholipid N-3 fatty acid levels in predicting dietary fish intake in patients with coronary artery disease. Amer. J Cardiol 66, 800-862

Soldat, J K. 1970. A statistical study of the habits of fishermen utilizing the Columbia River below Hanford. Pages 302-308 in W C. Rienig, editor Environmental Surveillance in the vicinity of nuclear facilities. C.C. Thomas, Springfield Illinois.

State of Hawaii, Department of Business and Economic Development. 1988. Hawaii Seafood Consumption. A Survey of Seafood Consumption in Hawaii. Department of Business and Economic Development. Honolulu, Hawaii.

Subcommittee on Criteria for Dietary Evaluation, Coordinating Committee on Evaluation of Food Consumption Surveys, Food and Nutrition Board, Commission on Life Sciences, National Research Council, National Academy Press, Washington, DC. 1986.

Toy, K.A., Gawne-Mittelstaedt G.D., Polissar N.L., and Liao S., 1995. A fish consumption survey of the Tulalip and Squaxin Island tribes of Puget Sound. Seattle, Washington.

Turcotte, S. 1983. Georgia Fishery Study: Implications for Dose Calculations. Technical Division Savannah River Laboratory. DPST--83--319-Rev.1. Memorandum to H.P. Olson.

U.S. EPA. 1980. Ambient Water Quality Criteria Documents (various). U.S. Environmental Protection Agency, Office of Water Regulations and Standards. EPA 440/5-80 Series.

U.S. EPA. (1992). A National Study of Chemical Residues in Fish. (EPA 823-R-92-008a and b.) Office of Water Regulations and Standards, U.S. EPA. Washington DC Vols. 1 and 2. September 1992.

U.S. EPA. (1992). Tribes at Risk. The Wisconsin Tribes Comparative Risk Project. Office of Policy Planning and Evaluation and Region 5. EPA 230-R-92-017. October, 1992.

U.S. EPA. (1992b). Tribes at Risk. The Wisconsin Tribes Comparative Risk Project. Office of Policy Planning and Evaluation and Region 5. EPA 230-R-92-017. October, 1992.

U.S. EPA. (1992b). Environmental Risk in Indian Country. U.S. EPA Washington, DC. Report No. EPA 171-R-92-014. PB 92-182393.

Welch, H.E., Bergmann, M.A., Siferd, T.D., Martin, K.A., Curtis, M.F., Crawford, R.E., Conover, R.J., and Hop, H. 1992. Energy flow through the marine ecosystem of the Lancaster Sound Region, Arctic Canada. Arctic 45: 343-357.

West, P.C., J.M. Fly, R. Marans and F. Larkin. 1989. Michigan Sport Anglers Fish Consumption Survey. A Report to Michigan Toxic Substance Control Commission by University of Michigan Ann Arbor, MI.

Wilkins, L. and Hankin, J. Personal Communication. 28 February 1996. University of Hawaii at Manoa. Cancer Research Center of Hawai'i, Epidemiology Program. 1236 Lauhain Street, Honolulu, Hawaii.

Willett W 1990. "Nature of Variation in Diet" pp. 34-51 In. Nutritional Epidemiology Ed Willett. W ·Nutritional Epidemiology Monographs in Epidemiology and Biostatistics. Vol 15 Oxford University Press. New York/Oxford.

Witschi JC 1990. "Short-Term Dietary Recall and Recording Methods". pp. 52-68. In Nutritional Epidemiology Monographs in Epidemiology and Biostatistics. Volume 15 Oxford University Press New York/Oxford.

Wolfe. R. and R. Walker. 1987. Subsistence Economies in Alaska: Productivity, geography and developmental impacts. Arctic Anthropol. 24:56-81.

Wormworth, J. 1995. Toxins and tradition: The impact of food-chain contamination on the Inuit of Northern Quebec. Canadian Medical Association Journal 152(8):1237-1240.

Youland DM, and Engle A (1976) Practices and problems in HANES: Dietary data methodology. J. Amer. Dietet. Assoc. 68: 22-25.

ADDENDUM TO APPENDIX H

ESTIMATED NATIONAL AND REGIONAL POPULATIONS OF WOMEN OF CHILD-BEARING AGE: UNITED STATES, 1990

Estimated National and Regional Populations of Women of Child-Bearing Age: United States, 1990

Because methylmercury is a developmental toxin, the subpopulation judged of particular concern in this Mercury Study Report to Congress was women of child-bearing age. Estimates of the size of the population of women of reproductive age, number of live births, number of fetal deaths, and number of legal abortions can be used to predict the percent of the population and number of women of reproductive age who are pregnant in a given year. This methodology has been previously used in the Agency for Toxic Substances and Disease Registry's (ATSDR's) Report to Congress on The Nature and Extent of Lead Poisoning in Children in the United States (Mushak and Crocetti, 1990).

The estimates of number of women of child-bearing age calculated for this Mercury Study: Report to Congress were prepared by Dr. A.M. Crocetti under purchase order from OAQPS. The techniques used by Dr. Crocetti parallel those used to prepared the 1984 estimates for ATSDR. To estimate the size of this population on a national basis Vital and Health Statistics data for number of live births (National Center for Health Statistics of the United States, 1990; Volume I, Natality, Table 1-60, pages 134-140), and fetal deaths (National Center for Health Statistics of the United States, 1990; Volume II, Mortality; Table 3-10, pages 16, 18, and 20). Fetal wastage, that is, spontaneous abortions prior to 20 weeks of gestation were not considered since no systematically collected, nationally based data exist.

The estimate of number of women of child-bearing age includes some proportion of women who will never experience pregnancy. However, substitution of the number of pregnancies in a given year provides some measure of assessing the size of the surrogate population at risk. Estimates of the size of the population were based on "Estimates of Resident Population of the United States Regions and Divisions by Age and Sex" (Byerly, 1993). The Census data for 1990 were grouped by age and gender. The sizes of these populations are shown in Table H-1.

Women ages 15 through 44 are the age group of greatest interest in identifying a subpopulation of concern for the effects of a developmental toxin such as methylmercury. This population consisted of 58,222,000 women living within the contiguous United States. This population was chosen rather than for the total United States (population 58,620,000 women ages 15 through 44 years) because the dietary survey information from CSFII/89-91 did not include Hawaii and Alaska. Based on estimates of fish consumption data for Alaska by Nobmann et al. (1992) the quantities of fish eaten by Alaskans exceeds those of the contiguous US population. It is also estimated that residents of the Hawaiian Islands also have fish consumption patterns that differ from those of the contiguous United States.

The number of pregnancies per year was estimated by combining the number of live births, number of fetal deaths (past 20 weeks of gestation) and the number of legal abortions. The legal abortion data were based on information published by Koonin et al. (1993) in Morbidity and Mortality Weekly Report. These totals are presented in Table H-2. As noted in this table, the total of legal abortions includes those with unknown age which were not included in the body of each table entry. There were 2,929 such cases for the United States in 1990 or 0.2% of all legal abortions. Another complication in the legal abortion data was for the age group 45 and older. The available data provide abortion data for 40 years and older only. To estimate the size of the population older than 45 years, the number of legal abortions for women age 40 years and older were allocated by using the proportions of Live Births and Fetal Deaths for the two age groups 40-44 and 45 and older.

It was estimated that within the contiguous United States 9.5% of women ages 15 to 44 years were pregnant in a given year. The total number of live births reported in 1990 for this age group was 4,112,579 with 30,974 reported fetal deaths and 1,407,830 reported legal abortions. The estimated number of total pregnancies for women ages 15 to 44 years was 5,551,383 in a population of 58,222,000 women.

REFERENCES

Byerly, E.R. (1993) State Population Estimates by Age and Sex: 1980-1992, U.S. Bureau of the Census, Current Population Reports P25-1106, U.S. Government Printing Office, Washington, DC.

Koonin, L.M., Smith, J.C., and Ramick, M. (1993) Division of Reproductive Health, National Center for Chronic Disease Prevention and Health Promotion: Abortion Surveillance - United States, 1990: Morbidity Mortality Weekly Report, Vol 42/No. SS-6, pps. 29-57, December 17.

Mushak, P., and Crocetti, A.M. (1988). The Nature and Extent of Lead Poisoning in Children in the United States: A Report to Congress. Agency for Toxic Substances and Disease Registry, United States Public Health Service, United States Department of Health and Human Services.

National Center for Health Statistics of the United States (1990) Volume I. Natality: Table 1-60: pages 134-140.

National Center for Health Statistics of the United States (1990) Volume II. Mortality; Table 3-10, pages 16, 18, and 20.

Nobmann, E.D., Byers, T., Lanier, A.P., Hankin, J.H., and Jackson, M.Y. (1992) The diet of Alaska Native adults: 1987-1988. Amer. J. Clin. Nutr. 55: 1024-1032.

Table H-1

Resident Population of the United States and Divisions, April 1, 1990
Census by Gender and Age; in Thousands, including Armed Forces Residing in Region

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	**Total**	**< 15 Years of Age**	**15-44 Years of Age**	**> 44 Years of Age**
United States	248.710	53.853	117.610	77.248
Male	121.239	27.570	58.989	34.680
Female	127.471	26.284	58.620	42.567
% Female	51.3	48.8	49 8	55.1

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	**Total**	**< 15 Years of Age**	**15-44 Years of Age**	**> 44 Years of Age**
Contiguous United States	247.052	53.462	116.772	76.817
Male	120.385	27.369	58.548	34.467
Female	126.667	26.094	58.222	42.348
% Female	51.3	48.8	49.9	55.1

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	**Total**	**< 15 Years of Age**	**15-44 Years of Age**	**> 44 Years of Age**
New England	13.207	2.590	6.379	4.239
Male	6.380	1.327	3.174	1.878
Female	6.827	1.264	3.202	2.361
% Female	51.7	48.8	50.2	55.7

Table H-1 (continued)

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age: in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	Total	< 15 Years of Age	15-44 Years of Age	> 45 Years of Age
Middle Atlantic States	37.602	7.471	17.495	12.638
Male	18.056	3.824	8.676	5.554
Female	19.547	3.645	8.818	7.083
% Female	52	49	50	56

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	Total	< 15 Years of Age	15-44 Years of Age	> 44 Years of Age
E North Central	42.009	9.233	19.596	13.180
Male	20.373	4.728	9.744	5.899
Female	21.636	4.505	9.851	7.279
% Female	51.5	48.8	50.3	55.2

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	Total	< 15 Years of Age	15-44 Years of Age	> 44 Years of Age
West North Central	17,660	3,967	8,017	5.676
Male	8,599	2.032	4.020	2.546
Female	9.061	1.935	3.997	3.129
% Female	51.3	48.8	49.9	55.1

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	Total	< 15 Years of Age	15-44 Years of Age	> 44 Years of Age
South Atlantic	43.567	8.864	20.579	14.122
Male	21.129	4.531	10.279	6.321
Female	22.438	4.333	10.301	7.804
% Female	51.5	48.9	50.1	55.3

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	Total	< 15 Years of Age	15-44 Years of Age	> 44 Years of Age
East South Central	15.176	3.316	7.037	4.823
Male	7.301	1.698	3.472	2.132
Female	7.875	1.618	3.565	2.692
% Female	51.9	48 8	50.7	55.8

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	Total	< 15 Years of Age	15-44 Years of Age	> 44 Years of Age
West South Central	26,703	6,366	12,687	7,651
Male	13.061	3.256	6.359	3.445
Female	13.641	3.110	6.328	4.204
% Female	51.1	48.9	49.9	54.9

Resident Population of the United States and Divisions. April 1. 1990 Census by Gender and Age: in Thousands. including Armed Forces Residing in Region.				
Division/ Gender	**Total**	**< 15 Years of Age**	**15-44 Years of Age**	**> 44 Years of Age**
Mountain States	13.659	3.313	6.435	3.910
Male	6.779	1.696	3.259	1.825
Female	6.880	1.616	3.176	2.087
% Female	50.4	48.8	49.4	53.4

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	**Total**	**< 15 Years of Age**	**15-44 Years of Age**	**> 44 Years of Age**
West North Central	17.660	3,967	8.017	5.676
Male	8.599	2.032	4.020	2.546
Female	9.061	1.935	3.997	3.129
% Female	51.3	48.8	49.9	55.1

Resident Population of the United States and Divisions, April 1, 1990 Census by Gender and Age; in Thousands, including Armed Forces Residing in Region.				
Division/ Gender	**Total**	**< 15 Years of Age**	**15-44 Years of Age**	**> 44 Years of Age**
Pacific (5 States including Alaska and Hawaii)	39,127	8,734	19,394	11,011
Male	19,562	4,476	10,004	5,083
Female	19,565	4,258	9,379	5,929
% Female	50.0	48.8	48.4	53.8