RI/FS-EIS DOCUMENT: DOE/EIS-0185D
BASELINE ASSESSMENT: DOE/OR/21548-091

Baseline Assessment for the Chemical Plant Area of the Weldon Spring Site

November 1992

U.S. Department of Energy
Oak Ridge Field Office
Weldon Spring Site Remedial Action Project

RI/FS-EIS DOCUMENT: DOE/EIS-0185D
BASELINE ASSESSMENT: DOE/OR/21548-091

Baseline Assessment for the Chemical Plant Area of the Weldon Spring Site

November 1992

prepared by

U.S. Department of Energy, Oak Ridge Field Office, Weldon Spring Site Remedial Action Project

with technical assistance from

Environmental Assessment and Information Sciences Division, Argonne National Laboratory, under Contract W-31-109-Eng-38

Documents Comprising the Draft
Remedial Investigation/Feasibility Study-Environmental Impact Statement
for the Weldon Spring Site Remedial Action Project

DOE/EIS-0185D

Baseline Assessment for the Chemical Plant Area of the Weldon Spring Site, DOE/OR/21548-091, U.S. Department of Energy, Oak Ridge Field Office, Oak Ridge, Tennessee, November 1992.

Feasibility Study for Remedial Action at the Chemical Plant Area of the Weldon Spring Site, DOE/OR/21548-148, Volumes I-II, U.S. Department of Energy, Oak Ridge Field Office, Oak Ridge, Tennessee, November 1992.

Proposed Plan for Remedial Action at the Chemical Plant Area of the Weldon Spring Site, DOE/OR/21548-160, U.S. Department of Energy, Oak Ridge Field Office, Oak Ridge, Tennessee, November 1992.

Remedial Investigation for the Chemical Plant Area of the Weldon Spring Site, DOE/OR/21548-074, Volumes I-II, U.S. Department of Energy, Oak Ridge Field Office, Oak Ridge, Tennessee, November 1992.

Addendum to the Remedial Investigation for the Chemical Plant Area of the Weldon Spring Site, DOE/OR/21548-272, U.S. Department of Energy, Oak Ridge Field Office, Oak Ridge, Tennessee, November 1992.

Reproduced directly from the best available copy.

Available to DOE and DOE contractors from the Office of Scientific and Technical Information, P.O. Box 62, Oak Ridge, TN 37831; prices available from (615) 576-8401.

Available to the public from the National Technical Information Service, U.S. Department of Commerce, 5285 Port Royal Road, Springfield, VA 22161.

CONTENTS

FIGURES xi

TABLES xv

NOTATION xxii

ENGLISH/METRIC AND METRIC/ENGLISH EQUIVALENTS xxiv

1 INTRODUCTION 1-1

1.1 Site Overview 1-1
1.1.1 Environmental Compliance Process 1-1
1.1.2 Site Response Actions 1-5
1.1.3 General Site Problem 1-8
1.1.4 Objectives of the Baseline Assessment 1-9
1.2 Site Background 1-10
1.2.1 Site Description 1-10
1.2.2 General History 1-10
1.2.3 Summary of Site Contamination 1-14
1.3 Scope of the Baseline Assessment 1-15
1.3.1 Staged Approach for the Baseline Assessment 1-16
1.3.2 Time Period 1-16
1.3.3 Institutional Controls and Exposure Scenarios 1-17
1.4 Report Organization 1-19

2 IDENTIFICATION OF CONTAMINANTS OF CONCERN 2-1

2.1 Site Operational History 2-1
2.1.1 Ordnance Works Operations 2-1
2.1.2 Uranium Processing 2-2
2.2 Site Conceptual Model 2-5
2.2.1 On-Site Exposure Pathways 2-6
2.2.2 Off-Site Exposure Pathways 2-7
2.3 Data Collection 2-9
2.3.1 Sampling Locations and Media 2-9
2.3.2 Analytical Parameters and Methods 2-10
2.3.2.1 Radiological Analyses 2-10
2.3.2.2 Chemical Analyses 2-13
2.3.3 Quality Assurance/Quality Control 2-14
2.4 Data Presentation and Evaluation 2-15
2.4.1 Soil 2-17
2.4.1.1 Radioactive Contaminants 2-17
2.4.1.2 Chemical Contaminants 2-18
2.4.2 Surface Water, Sediment, and Sludge 2-23
2.4.2.1 Radioactive Contaminants 2-23
2.4.2.2 Chemical Contaminants 2-24

CONTENTS (Cont.)

2.4.3 Groundwater 2-25
2.4.3.1 Radioactive Contaminants 2-26
2.4.3.2 Chemical Contaminants 2-26
2.4.4 Air 2-27
2.4.4.1 Radioactive Contaminants 2-27
2.4.4.2 Chemical Contaminants 2-27
2.4.5 Biota 2-28
2.4.5.1 Radioactive Contaminants 2-28
2.4.5.2 Chemical Contaminants 2-30
2.4.6 Building Air and Residues 2-31
2.4.6.1 Radioactive Contaminants 2-31
2.4.6.2 Chemical Contaminants 2-32
2.5 Final List of Contaminants of Concern 2-33

3 HUMAN EXPOSURE ASSESSMENT 3-1

3.1 Characterization of Exposure Setting 3-1
3.1.1 Environmental Setting 3-1
3.1.1.1 Climate and Meteorology 3-1
3.1.1.2 Topography and Surface Hydrology 3-2
3.1.1.3 Soil and Geology 3-2
3.1.1.4 Groundwater 3-3
3.1.1.5 Vegetation and Wildlife 3-4
3.1.2 Land Use and Demography 3-4
3.2 Identification of Exposure Pathways 3-6
3.2.1 Source Areas 3-7
3.2.2 Contaminant Fate and Transport 3-9
3.2.2.1 Atmospheric Conditions 3-10
3.2.2.2 Surface Water Flow 3-11
3.2.2.3 Groundwater Flow 3-13
3.2.2.4 Soil Geochemistry 3-14
3.2.2.5 Contaminant Persistence 3-14
3.2.2.6 Vegetation and Wildlife 3-15
3.2.2.7 Primary Release Mechanisms and Transport Pathways 3-16
3.2.3 Exposure Points, Receptors, and Exposure Routes 3-17
3.2.4 Complete Exposure Pathways 3-19
3.3 Estimation of Exposure Point Concentrations 3-21
3.3.1 Surface Soil 3-23
3.3.1.1 Radioactive Contaminants 3-24
3.3.1.2 Chemical Contaminants 3-25
3.3.2 Surface Water 3-27
3.3.2.1 Radioactive Contaminants 3-27
3.3.2.2 Chemical Contaminants 3-28
3.3.3 Sediment and Sludge 3-29
3.3.3.1 Radioactive Contaminants 3-29
3.3.3.2 Chemical Contaminants 3-29

CONTENTS (Cont.)

3.3.4 Air 3-30
3.3.5 Biota 3-32
3.3.5.1 Fish 3-32
3.3.5.2 Game 3-33
3.3.6 Building Air and Contaminated Dust and Residues 3-36
3.3.6.1 Radioactive Contaminants 3-37
3.3.6.2 Chemical Contaminants 3-37
3.4 Estimation of Contaminant Intakes 3-38
3.4.1 Introduction 3-38
3.4.2 Scenario-Specific Assumptions and Intake Parameters 3-39
3.4.2.1 Exposure Time, Exposure Frequency, and Exposure Duration 3-40
3.4.2.2 Inhalation Rates 3-41
3.4.2.3 Soil, Sediment, Sludge, and Residue Ingestion Rates 3-41
3.4.2.4 Water Ingestion Rates 3-42
3.4.2.5 Fish and Game Ingestion Rates 3-42
3.4.2.6 Body Weight 3-43
3.4.2.7 Averaging Time 3-43
3.4.3 Equations and Intake Estimates for Exposures to Soil 3-43
3.4.3.1 External Gamma Irradiation 3-44
3.4.3.2 Ingestion 3-45
3.4.4 Equations and Intake Estimates for Exposures to Surface Water 3-46
3.4.5 Equations and Intake Estimates for Exposure to Sediment and Sludge 3-47
3.4.5.1 External Gamma Irradiation 3-47
3.4.5.2 Ingestion 3-48
3.4.6 Equations and Intake Estimates for Inhalation of Airborne Contaminants 3-48
3.4.7 Equations and Intake Estimates for Ingestion of Biota 3-50
3.4.8 Equations and Intake Estimates for Exposures to Air and to Contaminated Residues and Dust in Buildings 3-51
3.4.8.1 External Gamma Irradiation 3-51
3.4.8.2 Inhalation 3-52
3.4.8.3 Ingestion 3-52
3.4.9 Multiple Exposure Pathways 3-53

4 TOXICITY ASSESSMENT 4-1

4.1 Risk Factors for the Assessment of Radiological Effects 4-1
4.2 Toxicity Values for the Assessment of Noncarcinogenic Effects of Chemical Contaminants 4-5
4.3 Toxicity Values for the Assessment of Carcinogenic Effects of Chemical Contaminants 4-11

CONTENTS (Cont.)

4.4 Summary Toxicity Profiles 4-11
4.4.1 Radionuclides 4-14
4.4.1.1 Actinium 4-17
4.4.1.2 Lead 4-17
4.4.1.3 Protactinium 4-18
4.4.1.4 Radium 4-18
4.4.1.5 Radon 4-19
4.4.1.6 Thorium 4-20
4.4.1.7 Uranium 4-20
4.4.2 Metals 4-21
4.4.2.1 Antimony 4-21
4.4.2.2 Arsenic 4-21
4.4.2.3 Barium 4-22
4.4.2.4 Beryllium 4-23
4.4.2.5 Cadmium 4-24
4.4.2.6 Chromium 4-25
4.4.2.7 Cobalt 4-26
4.4.2.8 Copper 4-26
4.4.2.9 Lead 4-27
4.4.2.10 Lithium 4-28
4.4.2.11 Manganese 4-28
4.4.2.12 Mercury 4-29
4.4.2.13 Molybdenum 4-30
4.4.2.14 Nickel 4-30
4.4.2.15 Selenium 4-31
4.4.2.16 Silver 4-32
4.4.2.17 Thallium 4-32
4.4.2.18 Uranium 4-33
4.4.2.19 Vanadium 4-34
4.4.2.20 Zinc 4-34
4.4.3 Inorganic Anions 4-35
4.4.3.1 Fluoride 4-35
4.4.3.2 Nitrate and Nitrite 4-35
4.4.4 Asbestos 4-36
4.4.5 Polycyclic Aromatic Hydrocarbons 4-37
4.4.6 Polychlorinated Biphenyls 4-38
4.4.7 Nitroaromatic Compounds 4-39
4.4.7.1 1,3-Dinitrobenzene 4-39
4.4.7.2 2,4-Dinitrotoluene and 2,6-Dinitrotoluene 4-39
4.4.7.3 Nitrobenzene 4-40
4.4.7.4 1,3,5-Trinitrobenzene 4-41
4.4.7.5 2,4,6-Trinitrotoluene 4-41

CONTENTS (Cont.)

5 HEALTH RISK CHARACTERIZATION 5-1

5.1 Risk Characterization Methodology 5-1
5.1.1 Radiological Risks 5-1
5.1.2 Chemical Risks and Hazard Quotients 5-2
5.1.2.1 Carcinogenic Risks 5-2
5.1.2.2 Hazard Quotients and Hazard Indexes 5-3
5.2 Risk Estimates for On-Site Exposure Points 5-4
5.2.1 Soil 5-4
5.2.1.1 Radiological Risks 5-5
5.2.1.2 Chemical Carcinogenic Risks and Hazard Indexes 5-5
5.2.2 Surface Water and Sludge 5-6
5.2.2.1 Radiological Risks 5-7
5.2.2.2 Chemical Risks and Hazard Indexes 5-7
5.2.3 Air 5-8
5.2.3.1 Radiological Risks 5-8
5.2.3.2 Chemical Risks and Hazard Indexes 5-9
5.2.4 Game 5-9
5.2.4.1 Radiological Risks 5-9
5.2.4.2 Chemical Risks and Hazard Indexes 5-10
5.2.5 Building Air and Residues 5-10
5.2.5.1 Radiological Risks 5-10
5.2.5.2 Chemical Risks and Hazard Indexes 5-11
5.2.6 Multiple Exposure Pathways 5-11
5.2.6.1 Trespasser 5-13
5.2.6.2 Future Recreational Visitor 5-15
5.3 Risk Estimates for Off-Site Exposure Points 5-16
5.3.1 Surface Water and Sediment at Lakes 34, 35, and 36 5-16
5.3.1.1 Radiological Risks 5-17
5.3.1.2 Chemical Risks and Hazard Indexes 5-17
5.3.2 Fish at Lakes 34, 35, and 36 5-17
5.3.2.1 Radiological Risks 5-17
5.3.2.2 Chemical Health Effects 5-17
5.3.3 Surface Water at Burgermeister Spring 5-18
5.3.3.1 Radiological Risks 5-18
5.3.3.2 Chemical Risks and Hazard Indexes 5-18
5.3.4 Surface Water and Sediment in the Southeast Drainage 5-18
5.3.4.1 Radiological Risks 5-19
5.3.4.2 Chemical Risks and Hazard Indexes 5-19
5.3.5 Soil at Vicinity Properties 5-20
5.3.5.1 Radiological Risks 5-20
5.3.5.2 Chemical Risks 5-20
5.3.6 Multiple Exposure Pathways 5-20
5.4 Multiple Exposure Pathways for On-Site and Off-Site Locations 5-21
5.4.1 Trespasser and Recreational Visitor 5-21
5.4.2 Sportsman 5-22

CONTENTS (Cont.)

5.5 Assessment of the Dermal Pathway 5-24
5.5.1 Health Assessment 5-24
5.5.2 Interpretation of Results 5-28
5.6 Uncertainty in the Assessment Process 5-29
5.6.1 Contaminants of Concern 5-29
5.6.1.1 Characterization Data 5-30
5.6.1.2 Selection Process 5-32
5.6.2 Exposure Assessment 5-33
5.6.2.1 Exposure Point Concentrations 5-33
5.6.2.2 Scenario Assumptions and Intake Parameters 5-35
5.6.2.3 Exposure Pathways 5-36
5.6.3 Toxicity Assessment 5-36
5.6.4 Risk Characterization 5-38
5.7 Summary of the Health Risk Characterization 5-40

6 SUMMARY OF THE HUMAN HEALTH ASSESSMENT 6-1

6.1 Contaminants of Concern 6-1
6.2 Exposure Assessment 6-1
6.2.1 Contaminant Fate and Transport 6-1
6.2.2 Potential Receptors and Routes of Exposure 6-2
6.2.3 Exposure Point Concentrations 6-4
6.3 Toxicity Assessment 6-4
6.4 Risk Characterization 6-5
6.5 Conclusions 6-10

7 ECOLOGICAL RESOURCES AND CONTAMINANTS OF CONCERN 7-1

7.1 Ecological Resources 7-1
7.1.1 On-Site Biota and Habitats 7-1
7.1.1.1 Terrestrial Resources 7-1
7.1.1.2 Aquatic Resources 7-2
7.1.2 Off-Site Biota and Habitats 7-4
7.1.2.1 Terrestrial Resources 7-4
7.1.2.2 Aquatic Resources 7-6
7.1.3 High-Quality Natural Communities 7-7
7.1.3.1 Unique Natural Areas 7-7
7.1.3.2 Wetlands and Riparian Corridors 7-9
7.1.4 Threatened and Endangered Species 7-9
7.2 Overview of Habitat Contamination 7-12
7.2.1 On-Site Terrestrial Habitats 7-12
7.2.2 On-Site Aquatic Habitats 7-14
7.2.3 Off-Site Terrestrial Habitats 7-14
7.2.4 Off-Site Aquatic Habitats 7-14
7.3 Conceptual Biouptake and Food Chain Transfer of Contaminants 7-15

CONTENTS (Cont.)

7.4 Potential Ecological Receptors 7-18
7.4.1 Previous Investigations 7-18
7.4.2 Ecological Receptors 7-20
7.4.3 Threatened and Endangered Species 7-20
7.5 Contaminants of Concern for the Ecological Risk Assessment 7-23
7.6 Ecological Risk Assessment 7-24
7.6.1 Radiological Risks 7-24
7.6.2 Chemical Risks 7-25
7.6.2.1 Soil 7-25
7.6.2.2 On-Site Surface Water 7-28
7.6.2.3 Off-Site Surface Water 7-32
7.6.2.4 Waterfowl 7-32
7.6.3 Biomagnification 7-33
7.7 Uncertainties 7-33
7.8 Summary and Conclusions 7-35

8 ASSESSMENT OF IMPACTS TO OTHER ENVIRONMENTAL RESOURCES 8-1

9 REFERENCES 9-1

10 LIST OF CONTRIBUTORS 10-1

APPENDIX A: Standard Analyte List and Detection Limits A-1

APPENDIX B: Groundwater Quality B-1

B.1 Metals B-5
B.1.1 Antimony B-6
B.1.2 Barium B-6
B.1.3 Chromium B-6
B.1.4 Lead B-7
B.1.5 Lithium B-7
B.1.6 Manganese B-7
B.1.7 Mercury B-8
B.1.8 Molybdenum B-8
B.1.9 Nickel B-8
B.1.10 Thallium B-8
B.2 Radionuclides B-9
B.3 Inorganic Anions B-9
B.3.1 Fluoride B-9
B.3.2 Nitrate B-9
B.3.3 Sulfate B-10
B.4 Nitroaromatic Compounds B-10
B.5 References B-79

APPENDIX C: Biotic Resources of St. Charles County and the Busch Wildlife Complex C-1

CONTENTS (Cont.)

APPENDIX D: Literature Review of Ecological Impacts on Select Biota D-1

APPENDIX E: Literature Review for Contaminant Mobility and Availability, Plant Uptake, and Animal Biotransfer E-1

E.1 Contaminant Mobility and Availability E-4
E.1.1 Radionuclides E-4
E.1.1.1 Actinium and Protactinium E-4
E.1.1.2 Radium E-5
E.1.1.3 Thorium E-5
E.1.1.4 Uranium E-6
E.1.2 Metals E-7
E.1.2.1 Antimony E-7
E.1.2.2 Arsenic E-7
E.1.2.3 Barium E-8
E.1.2.4 Beryllium E-8
E.1.2.5 Cadmium E-9
E.1.2.6 Chromium E-9
E.1.2.7 Cobalt E-10
E.1.2.8 Copper E-11
E.1.2.9 Lead E-11
E.1.2.10 Lithium E-12
E.1.2.11 Manganese E-12
E.1.2.12 Mercury E-13
E.1.2.13 Molybdenum E-13
E.1.2.14 Nickel E-14
E.1.2.15 Selenium E-14
E.1.2.16 Silver E-15
E.1.2.17 Thallium E-15
E.1.2.18 Vanadium E-16
E.1.2.19 Zinc E-16
E.1.3 Inorganic Anions E-16
E.1.3.1 Fluoride E-16
E.1.3.2 Nitrate and Nitrite E-17
E.1.3.3 Sulfate E-17
E.1.4 Polycyclic Aromatic Hydrocarbons E-17
E.1.5 Polychlorinated Biphenyls E-18
E.1.6 Nitroaromatic Compounds E-18
E.1.7 Distribution Coefficients E-19
E.2 Plant Uptake and Animal Biotransfer E-22
E.2.1 Radionuclides E-22
E.2.1.1 Actinium and Protactinium E-22
E.2.1.2 Radium E-23
E.2.1.3 Thorium E-24
E.2.1.4 Uranium E-24

CONTENTS (Cont.)

E.2.2 Metals E-25
E.2.2.1 Antimony E-25
E.2.2.2 Arsenic E-25
E.2.2.3 Barium E-26
E.2.2.4 Beryllium E-27
E.2.2.5 Cadmium E-27
E.2.2.6 Chromium E-28
E.2.2.7 Cobalt E-28
E.2.2.8 Copper E-29
E.2.2.9 Lead E-29
E.2.2.10 Lithium E-30
E.2.2.11 Manganese E-30
E.2.2.12 Mercury E-31
E.2.2.13 Molybdenum E-31
E.2.2.14 Nickel E-32
E.2.2.15 Selenium E-32
E.2.2.16 Silver E-33
E.2.2.17 Thallium E-33
E.2.2.18 Vanadium E-34
E.2.2.19 Zinc E-35
E.2.3 Inorganic Anions E-35
E.2.3.1 Fluoride E-35
E.2.3.2 Nitrate and Nitrite E-36
E.2.4 Polycyclic Aromatic Hydrocarbons E-36
E.2.5 Polychlorinated Biphenyls E-36
E.2.6 Nitroaromatic Compounds E-36
E.2.7 Selected Uptake and Biotransfer Factors E-37
E.3 References E-40

FIGURES

1.1 Location of the Weldon Spring Site 1-2

1.2 Components of Site Remediation 1-3

1.3 Primary Documents of the Environmental Compliance Process for the Proposed Remedial Action at the Chemical Plant Area 1-5

1.4 Major Environmental Compliance Activities and Related Documents for the Weldon Spring Site Remedial Action Project 1-6

1.5 General Layout of the Chemical Plant Area 1-11

1.6 Surface Features near the Weldon Spring Site 1-12

FIGURES (Cont.)

1.7 Location of Contaminated Vicinity Properties in the Area of the Weldon Spring Site 1-13

2.1 Uranium-238 Radioactive Decay Series 2-3

2.2 Thorium-232 Radioactive Decay Series 2-4

2.3 Uranium-235 Radioactive Decay Series 2-5

2.4 Site Conceptual Model 2-6

5.1 Relative Contaminant Contribution to the Pathway-Specific Chemical Carcinogenic Risks for the Worker 5-44

5.2 Relative Contaminant Contribution to the Pathway-Specific Hazard Indexes for the Worker 5-44

5.3 Location-Specific Radiological Risks for the Worker 5-45

5.4 Location-Specific Radiological Risks for the Trespasser 5-46

5.5 Location-Specific Radiological Risks for the Future Recreational Visitor 5-47

5.6 Location-Specific Chemical Carcinogenic Risks for the Worker 5-48

5.7 Location-Specific Chemical Carcinogenic Risks for the Recreational Visitor 5-49

5.8 Location-Specific Hazard Indexes for the Worker 5-50

5.9 Relative Pathway Contribution to the Overall Radiological Risk for Scenario T12 5-51

5.10 Relative Pathway Contribution to the Overall Chemical Carcinogenic Risk for Scenario T12 5-52

5.11 Relative Pathway Contribution to the Overall Hazard Index for Scenario T12 5-53

5.12 Estimated Radiological Risks for Ingestion of Surface Water from the Southeast Drainage 5-54

5.13 Estimated Radiological Risks from Exposure to Sediment in the Southeast Drainage 5-55

FIGURES (Cont.)

7.1 Wetland and Riparian Areas at the Busch Wildlife Complex 7-8

7.2 Terrestrial and Aquatic Habitats at the Weldon Spring Site 7-13

7.3 Schematic of Interrelationships among Contaminants, the Physico-chemical Environment, and Biota that Can Influence Bioavailability and Biouptake 7-16

7.4 Conceptual Contaminant Food Chain Pathways for Terrestrial Biota: Vegetation to Avifauna Consumers 7-17

7.5 Conceptual Contaminant Food Chain Transfer to the Bald Eagle, Peregrine Falcon, and Cooper's Hawk 7-21

B.1 Location and Completion Data for Site Monitoring Wells B-12

B.2 Estimated Surface of the Shallow Groundwater Table B-13

B.3 Estimated Surface of the Deep Groundwater Table B-14

B.4 Groundwater Weathered and Weathered/Competent Well Locations at the Weldon Spring Site B-15

B.5 Groundwater Competent Well Locations at the Weldon Spring Site B-16

B.6 Average Concentrations of Antimony in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-17

B.7 Average Concentrations of Barium in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-18

B.8 Average Concentrations of Barium in the Competent Zone, 1987-1990 . . . . . B-19

B.9 Average Concentrations of Chromium in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-20

B.10 Average Concentrations of Chromium in the Competent Zone, 1987-1990 . . . . . B-21

B.11 Average Concentrations of Lead in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-22

B.12 Average Concentrations of Lead in the Competent Zone, 1987-1990 . . . . . B-23

B.13 Average Concentrations of Lithium in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-24

FIGURES (Cont.)

B.14 Average Concentrations of Manganese in the Weathered and Weathered/ Competent Zones, 1987-1990 . . . . . B-25

B.15 Average Concentrations of Manganese in the Competent Zone, 1987-1990 . . . . . B-26

B.16 Average Concentrations of Mercury in the Weathered and Weathered/ Competent Zones, 1987-1990 . . . . . B-27

B.17 Average Concentrations of Mercury in the Competent Zone, 1987-1990 . . . . . B-28

B.18 Average Concentrations of Molybdenum in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-29

B.19 Average Concentrations of Molybdenum in the Competent Zone, 1987-1990 . . . . . B-30

B.20 Average Concentrations of Nickel in the Weathered and Weathered/ Competent Zones, 1987-1990 . . . . . B-31

B.21 Average Concentrations of Thallium in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-32

B.22 Average Concentrations of Thallium in the Competent Zone, 1987-1990 . . . . . B-33

B.23 Average Concentrations of Uranium in the Weathered and Weathered/ Competent Zones, 1987-1990 . . . . . B-34

B.24 Average Concentrations of Uranium in the Competent Zone, 1987-1990 . . . . . B-35

B.25 Average Concentrations of Fluoride in the Weathered and Weathered/ Competent Zones, 1987-1990 . . . . . B-36

B.26 Average Concentrations of Fluoride in the Competent Zone, 1987-1990 . . . . . B-37

B.27 Average Concentrations of Nitrate in the Weathered and Weathered/ Competent Zones, 1987-1990 . . . . . B-38

B.28 Average Concentrations of Nitrate in the Competent Zone, 1987-1990 . . . . . B-39

B.29 Average Concentrations of Sulfate in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-40

B.30 Average Concentrations of Sulfate in the Competent Zone, 1987-1990 . . . . . B-41

FIGURES (Cont.)

B.31 Average Concentrations of 1,3-Dinitrobenzene in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-42

B.32 Average Concentrations of 2,4-Dinitrotoluene in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-43

B.33 Average Concentrations of 2,4-Dinitrotoluene in the Competent Zone, 1987-1990 . . . . . B-44

B.34 Average Concentrations of 2,6-Dinitrotoluene in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-45

B.35 Average Concentrations of 2,6-Dinitrotoluene in the Competent Zone, 1987-1990 . . . . . B-46

B.36 Average Concentrations of Nitrobenzene in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-47

B.37 Average Concentrations of Nitrobenzene in the Competent Zone, 1987-1990 . . . B-48

B.38 Average Concentrations of 1,3,5-Trinitrobenzene in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-49

B.39 Average Concentrations of 1,3,5-Trinitrobenzene in the Competent Zone, 1987-1990 . . . . . B-50

B.40 Average Concentrations of 2,4,6-Trinitrotoluene in the Weathered and Weathered/Competent Zones, 1987-1990 . . . . . B-51

B.41 Average Concentrations of 2,4,6-Trinitrotoluene in the Competent Zone, 1987-1990 . . . . . B-52

TABLES

2.1 Principal Contaminant Characterization Studies Conducted for the Weldon Spring Site 2-34

2.2 Principal and Associated Radionuclides in the Uranium-238, Thorium-232, and Uranium-235 Decay Series 2-38

2.3 Activity Concentration Ratios Estimated for the Major Radionuclides 2-39

2.4 Characterization Results for Radionuclides, Metals, and Inorganic Anions in Site Soil 2-40

TABLES (Cont.)

2.5 Characterization Results for Organic Compounds in Site Soil 2-42

2.6 Soil Contaminants Deleted from the Risk Assessment 2-44

2.7 Characterization Results for Raffinate Pit Surface Water and Sludge 2-46

2.8 Characterization Results for NPDES Outfalls 2-49

2.9 Characterization Results for Surface Water in the Busch Wildlife Area 2-51

2.10 Characterization Results for Surface Water Discharges and Springs in the Southeast Drainage 2-54

2.11 Characterization Results for Background Surface Water Locations 2-56

2.12 Characterization Results for Sediment in the Southeast Drainage 2-58

2.13 Characterization Results for Sediment in the Busch Wildlife Area 2-60

2.14 Characterization Results for Groundwater 2-62

2.15 Uranium-238 Concentrations in Fish from Lakes 34, 35, 36, and 37 2-64

2.16 Radioactive Contaminants of Concern for the Human Health Assessment 2-65

2.17 Chemical Contaminants of Concern for the Human Health Assessment 2-66

3.1 Primary Exposure Pathways Evaluated for On-Site Locations 3-54

3.2 Primary Exposure Pathways Evaluated for Off-Site Locations 3-55

3.3 Exposure Scenario Assumptions and Intake Parameters 3-56

3.4 Intake Assumptions for Game Ingestion 3-56

3.5 Estimated Doses from Exposure to External Gamma Radiation from Site Soil 3-57

3.6 Estimated Doses for a Recreational Visitor from Exposure to Radionuclides in Vicinity Property Soil 3-58

3.7 Estimated Doses from Incidental Ingestion of Radionuclides in Site Soil 3-59

3.8 Estimated Average Daily Intake of Chemicals from Incidental Ingestion of Site Soil 3-60

TABLES (Cont.)

3.9 Estimated Doses from Ingestion of Radionuclides in Raffinate Pit Surface Water 3-62

3.10 Estimated Doses for a Recreational Visitor from Ingestion of Radionuclides in Surface Water at Off-Site Locations 3-62

3.11 Estimated Average Daily Intake of Chemicals from Ingestion of Raffinate Pit Surface Water 3-63

3.12 Estimated Average Daily Intake of Chemicals for a Recreational Visitor from Ingestion of Surface Water from Lakes 34, 35, and 36 3-64

3.13 Estimated Average Daily Intake of Chemicals for a Recreational Visitor from Ingestion of Surface Water at Off-Site Locations 3-66

3.14 Estimated Doses from Incidental Ingestion of Radionuclides in Raffinate Pit Sludge 3-68

3.15 Estimated Doses for a Recreational Visitor from Incidental Ingestion of Radionuclides in Sediment at Off-Site Locations 3-68

3.16 Estimated Average Daily Intake of Chemicals from Incidental Ingestion of Raffinate Pit Sludge 3-69

3.17 Estimated Average Daily Intake of Chemicals for a Recreational Visitor from Incidental Ingestion of Sediment from Off-Site Locations 3-71

3.18 Estimated Doses from Inhalation of Airborne Radioactive Particulates On-Site 3-72

3.19 Estimated Average Daily Intake of Chemicals from Inhalation of Airborne Particulates On-Site 3-73

3.20 Estimated Doses from Ingestion of Game 3-75

3.21 Estimated Average Daily Intake of Chemicals from Ingestion of Game 3-76

3.22 Estimated Doses from Exposure to Radionuclides in Building 403 3-78

3.23 Estimated Average Daily Intake of PCBs from Building 408 3-79

4.1 Dose Conversion Factors for the External Gamma Irradiation, Ingestion, and Inhalation Pathways at the Weldon Spring Site 4-3

4.2 Toxicity Information for Noncarcinogenic Effects: Oral Exposure 4-7

4.3 Toxicity Information for Noncarcinogenic Effects: Inhalation Exposure 4-10

TABLES (Cont.)

4.4 Toxicity Information for Carcinogenic Effects: Oral Exposure 4-12

4.5 Toxicity Information for Carcinogenic Effects: Inhalation Exposure 4-13

4.6 References for Chemical Toxicity Information 4-15

5.1 Contaminant Groupings for Segregation of the Hazard Index 5-56

5.2 Estimated Radiological Risks for Exposure to Site Soil 5-57

5.3 Estimated Chemical Carcinogenic Risks for Incidental Ingestion of Site Soil 5-58

5.4 Estimated Hazard Quotients for Incidental Ingestion of Site Soil 5-59

5.5 Estimated Radiological Risks for Incidental Ingestion of Raffinate Pit Surface Water and Sludge 5-61

5.6 Estimated Chemical Carcinogenic Risks for Incidental Ingestion of Raffinate Pit Surface Water and Sludge 5-61

5.7 Estimated Hazard Quotients for Incidental Ingestion of Raffinate Pit Surface Water and Sludge 5-62

5.8 Estimated Radiological Risks for Inhalation of Airborne Particulates On-Site 5-63

5.9 Estimated Chemical Carcinogenic Risks for Inhalation of Airborne Particulates On-Site 5-64

5.10 Estimated Hazard Quotients for Inhalation of Airborne Particulates On-Site 5-64

5.11 Estimated Radiological Risks for Ingestion of Game 5-65

5.12 Estimated Chemical Carcinogenic Risks for Ingestion of Game 5-66

5.13 Estimated Hazard Quotients for Ingestion of Game 5-67

5.14 Estimated Radiological Risks for Exposures in the Buildings 5-69

5.15 Estimated Chemical Carcinogenic Risks and Hazard Quotients for Exposures in the Buildings 5-69

5.16 Total Radiological Risks, Chemical Carcinogenic Risks, and Hazard Indexes for the On-Site Worker 5-70

TABLES (Cont.)

5.17 Total Radiological Risks for the On-Site Trespasser from Single and Multiple Exposure Pathways 5-70

5.18 Total Radiological Risks for the On-Site Recreational Visitor from Single and Multiple Exposure Pathways 5-71

5.19 Total Chemical Carcinogenic Risks for the On-Site Trespasser from Single and Multiple Exposure Pathways 5-71

5.20 Total Hazard Indexes for the Trespasser from Single and Multiple Exposure Pathways 5-72

5.21 Total Chemical Carcinogenic Risks for the On-Site Recreational Visitor from Single and Multiple Exposure Pathways 5-72

5.22 Total Hazard Indexes for the On-Site Recreational Visitor from Single and Multiple Exposure Pathways 5-73

5.23 Estimated Radiological Risks for Exposure for the Recreational Visitor from Exposure to Surface Water and Sediment at Off-Site Locations 5-74

5.24 Estimated Chemical Carcinogenic Risks for the Recreational Visitor from Ingestion of Surface Water at Off-Site Locations 5-75

5.25 Estimated Hazard Quotients for the Recreational Visitor from Ingestion of Surface Water at Off-Site Locations 5-76

5.26 Estimated Chemical Carcinogenic Risks and Hazard Indexes for the Recreational Visitor from Incidental Ingestion of Sediment at Off-Site Locations 5-77

5.27 Estimated Radiological Risks for Ingestion of Fish 5-78

5.28 Estimated Radiological Risks to the Recreational Visitor from Exposure to Surface Soil at Vicinity Properties 5-79

5.29 Total Radiological Risks, Chemical Carcinogenic Risks, and Hazard Indexes for the Off-Site Recreational Visitor 5-80

5.30 Estimated Chemical Carcinogenic Risks and Hazard Quotients for Dermal Exposures 5-81

5.31 Estimated Carcinogenic Risks and Hazard Indexes for the Swimmer 5-82

TABLES (Cont.)

6.1 Contaminants of Concern for the Human Health Assessment 6-2

6.2 Estimated Carcinogenic Risks for On-Site Receptors 6-6

6.3 Estimated Hazard Indexes for On-Site Receptors 6-6

6.4 Total On-Site Risks and Hazard Indexes 6-7

6.5 Estimated Carcinogenic Risks and Hazard Indexes for the Recreational Visitor at Off-Site Areas 6-7

7.1 Vegetation Observed at the Weldon Spring Site 7-2

7.2 Mammals, Fish, Reptiles, and Amphibians Observed at the Weldon Spring Site 7-3

7.3 Birds Observed at the Weldon Spring Site, 1991 7-3

7.4 Vegetative Cover at the Busch Wildlife Complex 7-5

7.5 Threatened, Endangered, or Special Concern Species Reported from St. Charles County 7-10

7.6 Potential Risks to Terrestrial Biota from the Chemical Soil Contaminants at the Weldon Spring Chemical Plant 7-26

7.7 Potential Risks to Aquatic Biota from the Chemical Contaminants Reported from On-Site and Off-Site Surface Waters in the Weldon Spring Area 7-29

A.1 Detection Limits for Inorganic Analytes A-3

A.2 Detection Limits for CLP Organic Compounds and Nitroaromatic Compounds A-4

B.1 Monitoring Well Construction Data B-53

B.2 Concentrations of Metals in Groundwater Samples, 1987-1990 ..... B-55

B.3 Concentrations of Inorganic Anions in Groundwater Samples, 1987-1990 ..... B-68

B.4 Concentrations of Nitroaromatic Compounds in Groundwater Samples, 1987-1990 ..... B-72

C.1 Mammals, Reptiles, and Amphibians Reported from St. Charles County C-3

C.2 Birds Reported from St. Charles County C-4

TABLES (Cont.)

C.3 Fish Occurring in St. Charles County C-6

C.4 Multiple Species Fishing Waters of the Busch Wildlife Complex C-7

D.1 Tissue Concentrations of Radionuclides Reported for Select Species of Plants D-3

D.2 Tissue Concentrations of Radionuclides Reported for Select Species of Fish D-5

D.3 Lethal and Sublethal Effects of Metals on Select Biota D-7

D.4 Lethal and Sublethal Effects of Inorganic Anions on Select Biota D-16

D.5 Lethal and Sublethal Effects of Polycyclic Aromatic Hydrocarbons and Polychlorinated Biphenyls on Select Biota D-17

D.6 Lethal and Sublethal Effects of Nitroaromatic Compounds on Select Biota D-18

E.1 Estimated K_d Values for Contaminants in Site Soil E-20

E.2 Estimated Plant Uptake and Animal Biotransfer Factors E-38

NOTATION

The following is a list of the acronyms, initialisms, and abbreviations (including units of measure) used in this document. Acronyms used in figures and tables only are defined in the respective figures and tables.

ACRONYMS, INITIALISMS, AND ABBREVIATIONS

AEC	U.S. Atomic Energy Commission
ALARA	as low as reasonably achievable
ATSDR	Agency for Toxic Substances and Disease Registry
BA	baseline assessment
beta-BHC	β-hexachlorocyclohexane
CEDE	committed effective dose equivalent
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980, as amended
CFR	Code of Federal Regulations
CLP	Contract Laboratory Program
DCF	dose conversion factor
DDE	dichlorodiphenyldichloroethylene
DNB	dinitrobenzene
DNT	dinitrotoluene
DOE	U.S. Department of Energy
ECAO	Environmental Criteria and Assessment Office
EC_{50}	concentration at which a specified effect is observed in 50% of the test population
EDE	effective dose equivalent
EE/CA	engineering evaluation/cost analysis
EIS	environmental impact statement
EPA	U.S. Environmental Protection Agency
FS	feasibility study
HEAST	*Health Effects Assessment Summary Tables*
ICRP	International Commission on Radiological Protection
IRIS	*Integrated Risk Information System*
LC_{50}	concentration lethal to 50% of the test population
LOAEL	lowest-observed-adverse-effect level
MCL	maximum contaminant level
MCLG	maximum contaminant level goal
MRL	minimal risk level
MSA	material staging area
MSL	(above) mean sea level
NB	nitrobenzene
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NEPA	National Environmental Policy Act of 1969
NESHAPs	National Emission Standard for Hazardous Air Pollutants
NOAEL	no-observed-adverse-effect level
NOEL	no-observed-effect level
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List

PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCM	phase contrast microscopy
PP	proposed plan
PQL	practical quantitation level
QA	quality assurance
QAPP	Quality Assurance Program Plan
QC	quality control
RDA	recommended dietary allowance
RETA	Ryckman/Edgerley/Tomlinson & Associates
RfC	reference concentration
RfD	reference dose
RI	remedial investigation
RME	reasonable maximum exposure
ROD	record of decision
SMCL	secondary maximum contaminant level
tDNT	technical grade dinitrotoluene
TEM	transmission electron microscopy
TNB	trinitrobenzene
TNT	trinitrotoluene
TSP	total suspended particulates
UL_{95}	95% upper confidence limit of the arithmetic average
USATHAMA	U.S. Army Toxic and Hazardous Materials Agency
USGS	U.S. Geological Survey

UNITS OF MEASURE

Bq	becquerel
cm	centimeter
cm^2	square centimeter
cm^3	cubic centimeter
d	day
dL	decaliter
ft	foot
g	gram
h	hour
ha	hectare
in.	inch
kg	kilogram
km	kilometer
L	liter
lb	pound
µg	microgram
µm	micrometer
µR	microroentgen
m	meter
m^2	square meter
m^3	cubic meter
mBq	millibecquerel
MeV	megaelectronvolt
mg	milligram
mi	mile
min	minute
mL	milliliter
mmol	millimole
mo	month
mol	mole
mph	mile per hour
mR	milliroentgen
mrem	millirem
ng	nanogram
oz	ounce
pCi	picocurie
ppb	part per billion
ppm	part per million
ppt	part per trillion
rad	radiation absorbed dose
rem	roentgen equivalent man
s	second
t	metric ton
wk	week
WL	working level
WLM	working-level month
yr	year

ENGLISH/METRIC AND METRIC/ENGLISH EQUIVALENTS

In this document, units of measure are presented with the metric equivalent first, followed by the measured English unit in parentheses. In cases where the measurement was originally made in metric units, the values were not converted back to English units; in tables, the data are generally in English or metric units only. The following table lists the appropriate equivalents for English and metric units.

Multiply	By	To Obtain
English/Metric Equivalents		
acres	0.4047	hectares (ha)
cubic feet (ft^3)	0.02832	cubic meters (m^3)
cubic yards (yd^3)	0.7646	cubic meters (m^3)
degrees Fahrenheit (°F) -32	0.5555	degrees Celsius (°C)
feet (ft)	0.3048	meters (m)
gallons (gal)	3.785	liters (L)
gallons (gal)	0.003785	cubic meters (m^3)
inches (in.)	2.540	centimeters (cm)
miles (mi)	1.609	kilometers (km)
pounds (lb)	0.4536	kilograms (kg)
short tons (tons)	907.2	kilograms (kg)
short tons (tons)	0.0972	metric tons (t)
square feet (ft^2)	0.09290	square meters (m^2)
square yards (yd^2)	0.8361	square meters (m^2)
square miles (mi^2)	2.590	square kilometers (km^2)
yards (yd)	0.9144	meters (m)
Metric/English Equivalents		
centimeters (cm)	0.3937	inches (in.)
cubic meters (m^3)	35.31	cubic feet (ft^3)
cubic meters (m^3)	1.308	cubic yards (yd^3)
cubic meters (m^3)	264.2	gallons (gal)
degrees Celsius (°C) +17.78	1.8	degrees Fahrenheit (°F)
hectares (ha)	2.471	acres
kilograms (kg)	2.205	pounds (lb)
kilograms (kg)	0.001102	short tons (tons)
kilometers (km)	0.6214	miles (mi)
liters (L)	0.2642	gallons (gal)
meters (m)	3.281	feet (ft)
meters (m)	1.094	yards (yd)
metric tons (t)	1.102	short tons (tons)
square kilometers (km^2)	0.3861	square miles (mi^2)
square meters (m^2)	10.76	square feet (ft^2)
square meters (m^2)	1.196	square yards (yd^2)

1 INTRODUCTION

1.1 SITE OVERVIEW

The Weldon Spring site is located in St. Charles County, Missouri, about 48 km (30 mi) west of St. Louis (Figure 1.1). The site became contaminated as a result of processing and disposal activities that took place from the 1940s through the 1960s, and it is listed on the National Priorities List (NPL) of the U.S. Environmental Protection Agency (EPA). The U.S. Department of Energy (DOE) is responsible for cleanup activities at the site under its Environmental Restoration and Waste Management Program. The major goals of this program are to eliminate potential hazards to human health and the environment that are associated with contaminated DOE sites and to make surplus property available for other uses, to the extent possible.

The Weldon Spring site consists of two noncontiguous areas: an 88-ha (217-acre) chemical plant area and a 3.6-ha (9-acre) limestone quarry. The chemical plant area is about 3.2 km (2 mi) southwest of the junction of Missouri (State) Route 94 and U.S. Route 40/61, and the quarry is about 6.4 km (4 mi) south-southwest of the chemical plant area. Both locations are accessible from State Route 94 and are fenced and closed to the public. Explosives were produced at the chemical plant during the 1940s, and uranium and thorium materials were processed during the 1950s and 1960s. Some waste slurries and solids were disposed of at the plant, and various solid materials (including process wastes and decontamination debris) were disposed of in the quarry between 1942 and 1969.

Cleanup of the Weldon Spring site consists of several integrated components (Figure 1.2). This baseline assessment (BA) addresses potential human health and environmental impacts associated with the chemical plant area of the site under both current and hypothetical future conditions in the absence of cleanup. The assessment provides a framework for developing risk information that can be used to support cleanup decisions for the chemical plant area, and it serves to focus the selection of appropriate cleanup remedies. Other components of the integrated site remediation process are described in Section 1.1.2.

1.1.1 Environmental Compliance Process

The assessment of baseline health risks and environmental impacts for a contaminated site is an important element of the remedial investigation/feasibility study (RI/FS) process developed by the EPA (1988a, 1989i, 1989j, 1990c). This process addresses the cleanup of hazardous waste sites under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), as amended. Four primary evaluation documents comprise the RI/FS for the chemical plant area of the Weldon Spring site:

- The RI — which presents site characterization results and addresses the nature and extent of contamination (DOE 1992c);

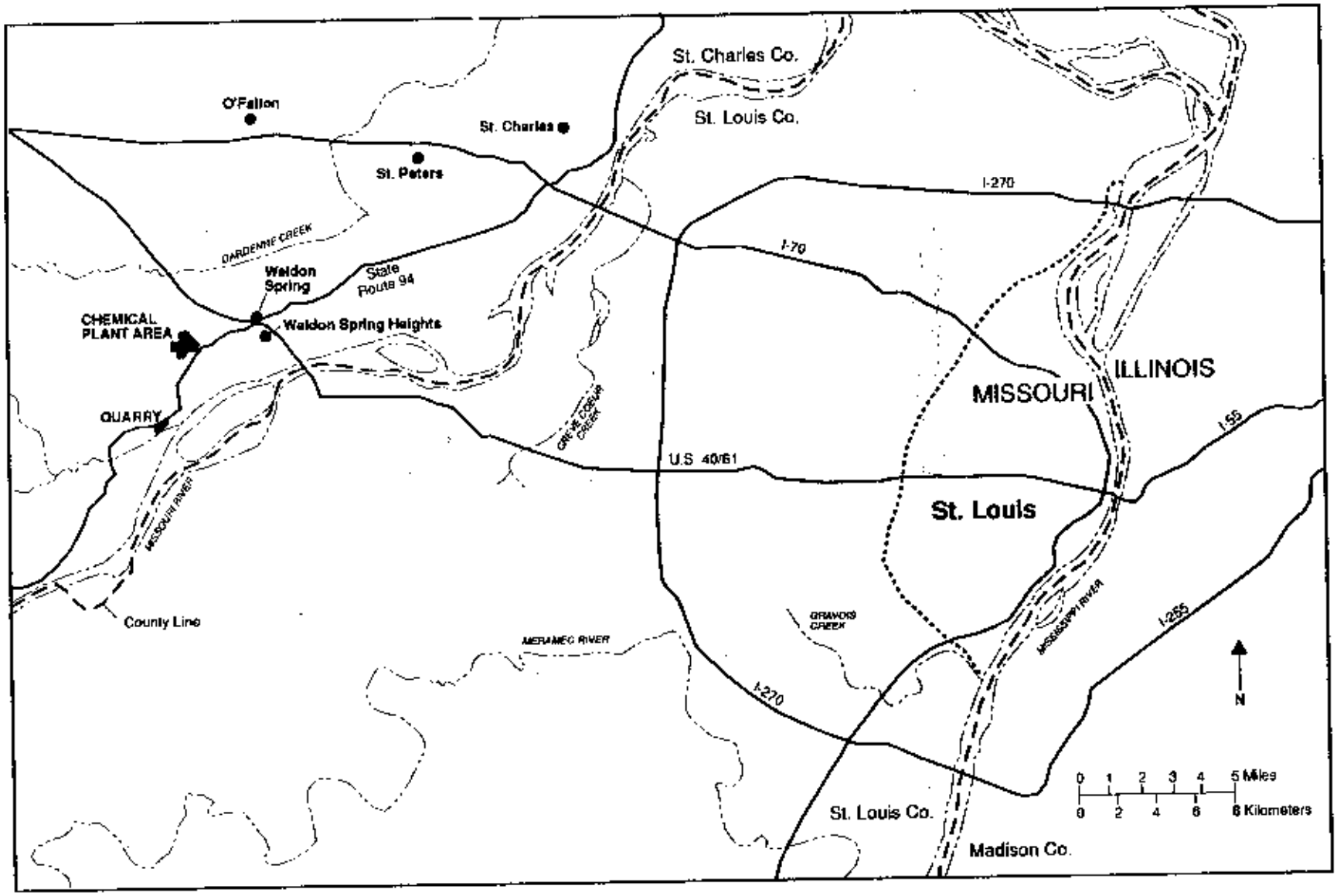
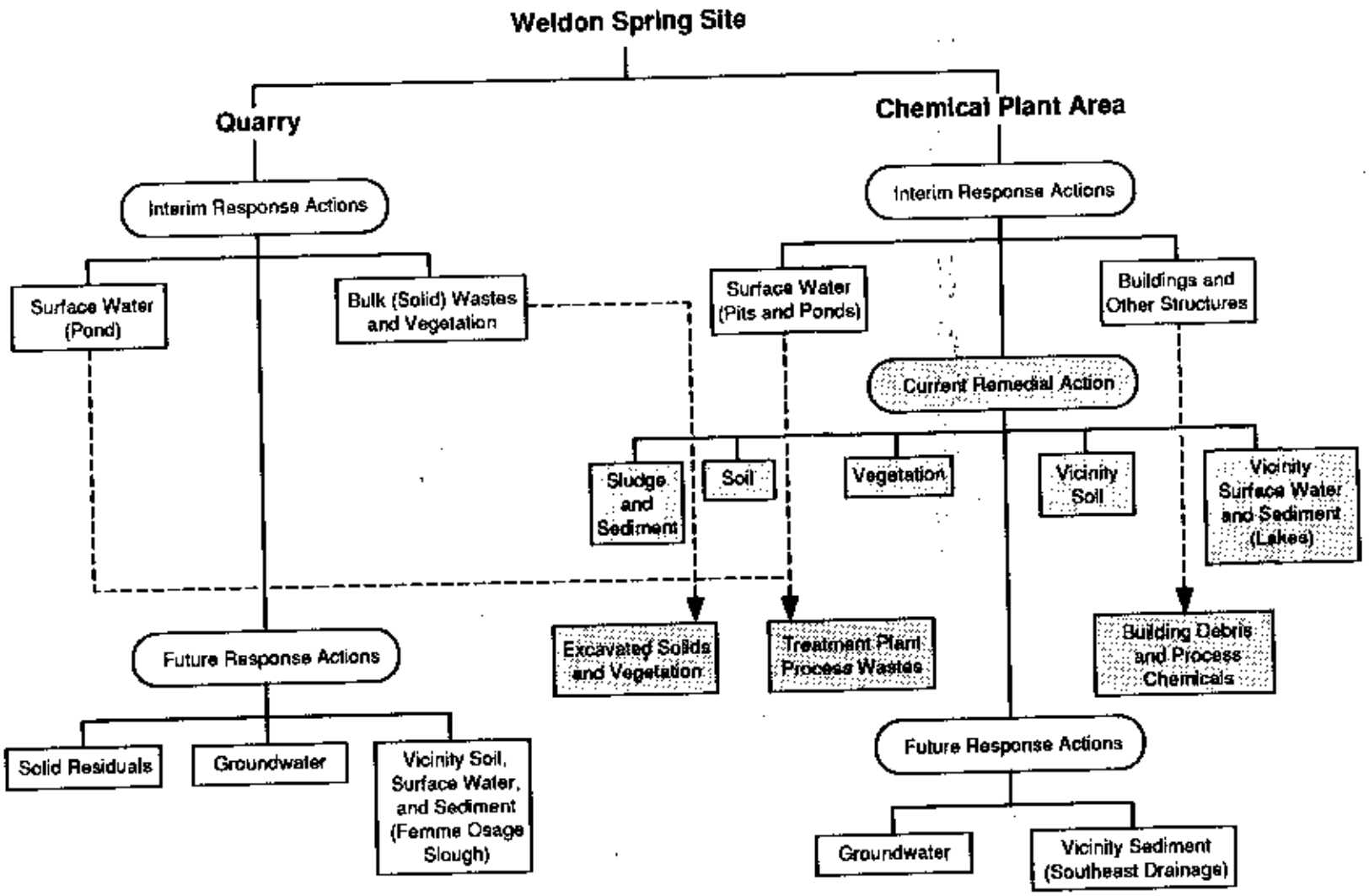


FIGURE 1.1 Location of the Weldon Spring Site



Note: The boxes represent contaminated media addressed by the project's cleanup actions for the chemical plant area and the quarry, and they are connected by solid lines to the appropriate phase of site cleanup. Dashed lines identify waste stored at the chemical plant area as a result of the interim actions. The media for which specific treatment and disposal decisions will be made as part of the current remedial action are indicated by shading.

FIGURE 1.2 Components of Site Remediation (disposition of contaminated material that could result from future response actions is addressed in the current remedial action)

- This BA — which uses information from the RI to estimate human health and environmental impacts that could occur if no cleanup action were taken and includes both the baseline human health evaluation and the baseline ecological assessment;

- The FS — which develops and evaluates cleanup alternatives, considering the results of the BA and the various response actions that might be appropriate for the contaminated locations and media at the site (DOE 1992a); and

- The proposed plan (PP) — which summarizes the analysis of final alternatives from the FS and identifies the preferred remedial action alternative (DOE 1992b).

The decision-making process for cleanup of the Weldon Spring site integrates the requirements of two major environmental laws: (1) the National Environmental Policy Act (NEPA), which addresses major federal actions that may significantly affect the quality of the human environment, and (2) CERCLA, which addresses the cleanup of NPL sites. The documents developed for site cleanup under the RI/FS process of CERCLA have also been written to incorporate NEPA values.

For the major response action currently being considered for the chemical plant area, the level of analysis in the RI/FS under CERCLA is consistent with that provided in an environmental impact statement (EIS) under NEPA. The document package is therefore referred to as an RI/FS-EIS. The appropriate NEPA elements of an EIS are integrated into the CERCLA documentation for this action as follows: the RI generally presents the affected environment, the BA evaluates the no-action alternative, the FS evaluates the action alternatives, and the PP discusses the preferred alternative.

Under CERCLA, the baseline human health evaluation is presented in a "baseline risk assessment." However, to address all components for an assessment of the no-action alternative under NEPA, the site baseline assessment was extended to address environmental resources such as archaeological and cultural resources for which terminology is more appropriately impact than risk. For this reason, the term "baseline assessment" was used for this document because it contains baseline analyses beyond those typical of a CERCLA baseline risk assessment.

The activities and environmental compliance documents for the Weldon Spring site are developed in coordination with EPA Region VII and the state of Missouri. The documents are also made available to the public, and public involvement is an important factor in the decision-making process for site remediation. The RI, BA, FS, and PP will be used to develop the upcoming record of decision (ROD) for cleanup of the chemical plant area. Responses to public comments on these four documents will be incorporated into both the ROD and a responsiveness summary (RS). The RS — together with the RI, BA, FS, and PP — will comprise the final RI/FS-EIS for the chemical plant area. These documents will be included with the ROD in the administrative record for this action (Figure 1.3). Following the ROD, remedial design and

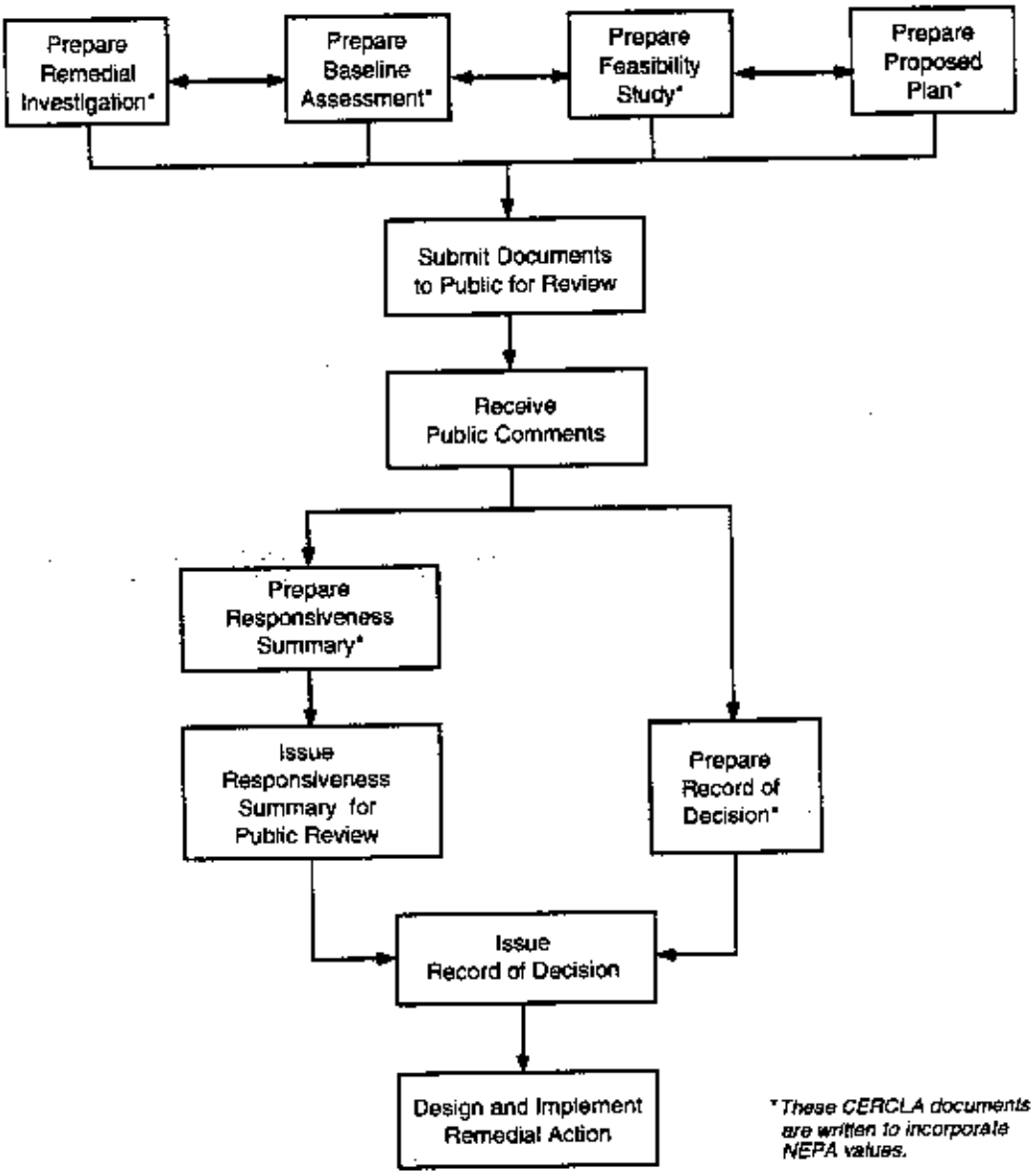
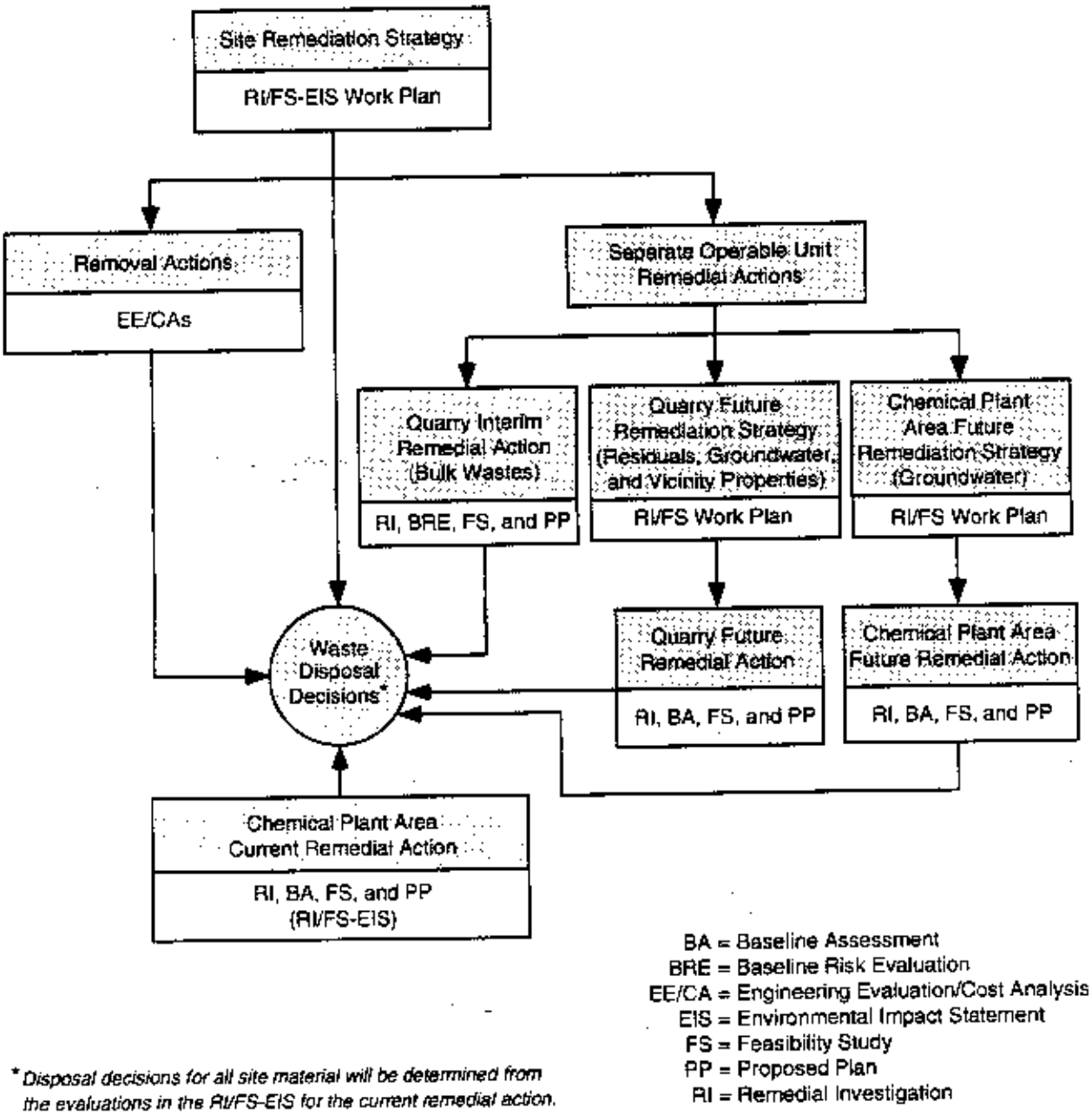


FIGURE 1.3 Primary Documents of the Environmental Compliance Process for the Proposed Remedial Action at the Chemical Plant Area

remedial action activities will be implemented at the chemical plant area. Public involvement in the proposed action will continue during the post-ROD period, e.g., through public briefings and informational meetings.

1.1.2 Site Response Actions

An overview of the environmental compliance activities and documents for the integrated response actions at the Weldon Spring site is presented in Figure 1.4. Decisions for several interim response actions — both removal actions and interim remedial actions — have already been finalized. These actions include (1) removing and treating solid (bulk) waste from



BA = Baseline Assessment
BRE = Baseline Risk Evaluation
EE/CA = Engineering Evaluation/Cost Analysis
EIS = Environmental Impact Statement
FS = Feasibility Study
PP = Proposed Plan
RI = Remedial Investigation

** Disposal decisions for all site material will be determined from the evaluations in the RI/FS-EIS for the current remedial action.*

FIGURE 1.4 Major Environmental Compliance Activities and Related Documents for the Weldon Spring Site Remedial Action Project

the quarry, (2) removing and treating contaminated water from the quarry pond, (3) dismantling all of the chemical plant structures, and (4) removing and treating contaminated surface water from the chemical plant area (hereafter referred to as the site). These actions are in various stages of design and implementation. A number of other small-scope actions have already been implemented to respond to potential worker health and safety concerns and to mitigate contaminant releases. These interim actions included (1) dismantling overhead pipes and removing related asbestos, (2) dismantling inactive power lines and poles, (3) removing inactive electrical transformers that contained polychlorinated biphenyls (PCBs), (4) decontaminating and dismantling several buildings, and (5) constructing a surface runoff diversion system around a former dump area.

The RI/FS-EIS package for the site, which includes this BA, addresses the second and most comprehensive stage of site remediation. The contaminated materials for which decisions will be made at this stage include (1) soil, sediment, and sludge on-site and (2) soil and sediment at several off-site locations. Neither groundwater nor the Southeast Drainage are included in decisions for the current remedial action.

Groundwater is being addressed as a separate operable unit of site remediation because current data are limited for conducting a risk assessment in support of final decisions. During the next several years, additional characterization data will be collected, and a separate RI/FS package will be prepared in support of the decision for groundwater. Nevertheless, to focus future characterization efforts and provide an integrated picture of the affected environment at the site for this stage of the documentation process, current groundwater data have been presented in the RI and key information is summarized in this BA (Appendix B). In addition, preliminary risk estimates associated with the hypothetical ingestion of this groundwater have been made on the basis of the limited data currently available and are presented in Appendix E of the FS. These estimates will be refined in the RI/FS to be prepared for the groundwater operable unit after additional information becomes available.

The Southeast Drainage is a natural 2.4-km (1.5-mi) channel that carries surface runoff to the Missouri River from a small portion of the site and from an extensive off-site area. This drainage will be addressed as a future removal action for the project for two reasons. First, conditions in the Southeast Drainage are expected to change as a result of the cleanup decision that will be determined from the RI/FS-EIS. For example, decreased contaminant levels in surface runoff from the site would be expected to result in improved water quality in the drainage. Second, similar to groundwater, additional sampling is needed to further characterize the drainage for purposes of conducting a risk assessment in support of final decisions. Nevertheless, preliminary risk estimates for the Southeast Drainage are presented in this BA to address potential current and short-term future uses of the wildlife area and to focus upcoming characterization efforts. These estimates are based on the limited data available; after additional information is collected, these preliminary estimates will be refined in the engineering evaluation/cost analysis (EE/CA) report to be prepared within the next several years to support the cleanup decision for this drainage.

A baseline risk assessment is prepared to address a contaminated site under baseline conditions, and this BA addresses the Weldon Spring site as it existed in early 1992. This assessment reflects conditions resulting from interim actions that have already been completed (such as construction of a surface runoff diversion system at Ash Pond), but it does not reflect conditions that will result from actions that have been finalized but not yet fully implemented. That is, the following conditions were assumed for this BA, even though these conditions are expected to change within the next few years as a result of implementing recent decisions that were made for interim cleanup activities: (1) the bulk wastes from the quarry are not yet in short-term storage on-site, (2) water has not yet been removed and treated from the four raffinate pits and two ponds on-site, and (3) site structures have not all been dismantled. The objective of this BA is to support the current stage of site remediation, so simplifying assumptions have been made to address the components for which decisions have already been made, such as buildings and surface water impoundments. That is, because this BA need not reconfirm decisions that have already been made to remove those contaminated components, screening-level calculations are presented for those exposures. Conditions that will exist after decisions have been implemented for the recently approved interim actions are evaluated in Appendix E of the FS (DOE 1992a).

1.1.3 General Site Problem

The Weldon Spring site is radioactively and chemically contaminated as a result of past processing activities. The radioactive contaminants are associated with the uranium-238, thorium-232, and uranium-235 decay series, and the chemical contaminants are associated with both ordnance works and uranium and thorium ore processing operations (see Section 1.2.2). Detailed information on the nature and extent of contamination at the site is presented in the RI report (DOE 1992c); a brief overview of this information is presented in Section 1.2.3 and Chapter 2 of this BA.

The site is fenced, and access to contaminated areas is restricted. Thus, under current conditions, no on-site exposures of the general public occur. Also, no significant adverse impacts to vegetation and wildlife have been reported. The site has been extensively characterized, and an environmental monitoring program is under way both within and beyond the site boundary. Results of the site characterization and monitoring programs indicate the following general conditions:

- Concentrations of airborne contaminants such as radon gas are at background levels at the perimeter fence.
- On-site soil at former dump areas and other scattered locations contains contaminants at concentrations above background levels, including radionuclides, some heavy metals, and PCBs.
- Off-site soil at vicinity properties contains radionuclides at above background levels, but the concentrations are generally low.

- Off-site surface water, including a spring and three lakes in the wildlife area north of the site and springs in the Southeast Drainage, contains uranium and some inorganic anions such as nitrate at above background levels as a result of surface runoff, but the concentrations are generally low.

- Off-site sediment in the wildlife area lakes and the Southeast Drainage contains radionuclides at above background levels as a result of surface runoff; concentrations in the lakes are generally low but are higher at several locations in the Southeast Drainage.

- Groundwater at certain locations in the shallow (Burlington-Keokuk) aquifer contains contaminants at above background levels — including nitrate, sulfate, nitroaromatic compounds, uranium, and other metals; current field data indicate that site-specific contaminants are present in groundwater beyond the site boundary but the extent of contamination is limited to the general site vicinity.

1.1.4 Objectives of the Baseline Assessment

The specific objectives of this BA are as follows:

- Estimate the magnitude of potential health risks and environmental impacts that would be associated with the site if no remedial action were taken;

- Identify the areas, environmental media, and contaminants that pose the primary health and environmental concerns;

- Identify the areas, environmental media, and contaminants that pose little or no threat to human health or the environment;

- Identify any existing data gaps so that additional information can be collected to support cleanup decisions; and

- Provide a baseline for comparing the protectiveness of cleanup alternatives in the FS relative to potential human health and environmental impacts.

The results of this BA have been used to support various analyses in the FS. These analyses include (1) determining specific areas and environmental media for which cleanup is appropriate, as indicated by estimated current and future risks; (2) identifying locations and environmental media for which "no action" might be the most appropriate response; and (3) developing preliminary soil cleanup criteria to address those contaminants that, if left at the site, might pose significant adverse impacts to human health or the environment.

1.2 SITE BACKGROUND

The background information presented in Sections 1.2.1 through 1.2.3 represents a general overview of the site, its history, and existing contamination; much more detailed information is presented in the RI and in Chapters 2 and 3 and Appendix B of this document. Because this BA focuses on the potential health and environmental impacts associated with the chemical plant area and its vicinity, the site overview is limited to that area; separate documentation will be prepared for the final quarry action within the next several years.

1.2.1 Site Description

The Weldon Spring site contains about 40 buildings and support structures that are in various stages of dismantlement, four raffinate pits, two ponds (Ash Pond and Frog Pond), two former dump areas (North Dump and South Dump), and a former coal storage area (Figure 1.5). Much of the land surface around the buildings is paved or covered with gravel; the remainder of the site contains a variety of grasses and scattered small shrubs and trees. Except for the northern quadrant, much of the vegetated area is mowed, and little undisturbed and/or natural habitat exists.

The site is bordered by the August A. Busch Memorial Wildlife Area to the north, the Weldon Spring Wildlife Area to the south and east, and the U.S. Army Reserve and National Guard Training Area to the west (Figure 1.6). Several small areas on the Army property and in the wildlife areas are contaminated as a result of previous chemical plant activities; these areas are termed vicinity properties (Figure 1.7). The contamination on these vicinity properties is described in detail in Section 5.2.3 of the RI report.

Francis Howell High School is located about 1 km (0.6 mi) east of the site on State Route 94. The two closest communities are Weldon Spring and Weldon Spring Heights; they are located about 3.2 km (2 mi) east of the site and have a combined population of about 850. The largest city in the county is St. Charles; it is located about 24 km (15 mi) northeast of the site and has a population of about 40,000.

1.2.2 General History

In 1941, the U.S. Department of the Army obtained 7,000 ha (17,000 acres) of land in St. Charles County to construct an ordnance works facility. The Army produced trinitrotoluene (TNT) and dinitrotoluene (DNT) explosives at this facility from 1941 to 1944 and again from 1945 to 1946. In 1955, 83 ha (205 acres) of the former ordnance works property was transferred to the U.S. Atomic Energy Commission (AEC, a predecessor of DOE) for construction and operation of the Weldon Spring Uranium Feed Materials Plant, now referred to as the chemical plant; an additional 6 ha (15 acres) was later transferred for expansion of waste storage capacity. Considerable explosives decontamination and site regrading was performed prior to construction of the chemical plant. Uranium and some thorium ore concentrates were processed at the plant from 1957 to 1966; the types of materials used for processing activities are summarized in Chapter 2, and much more detailed information is presented in Appendix D of the RI.

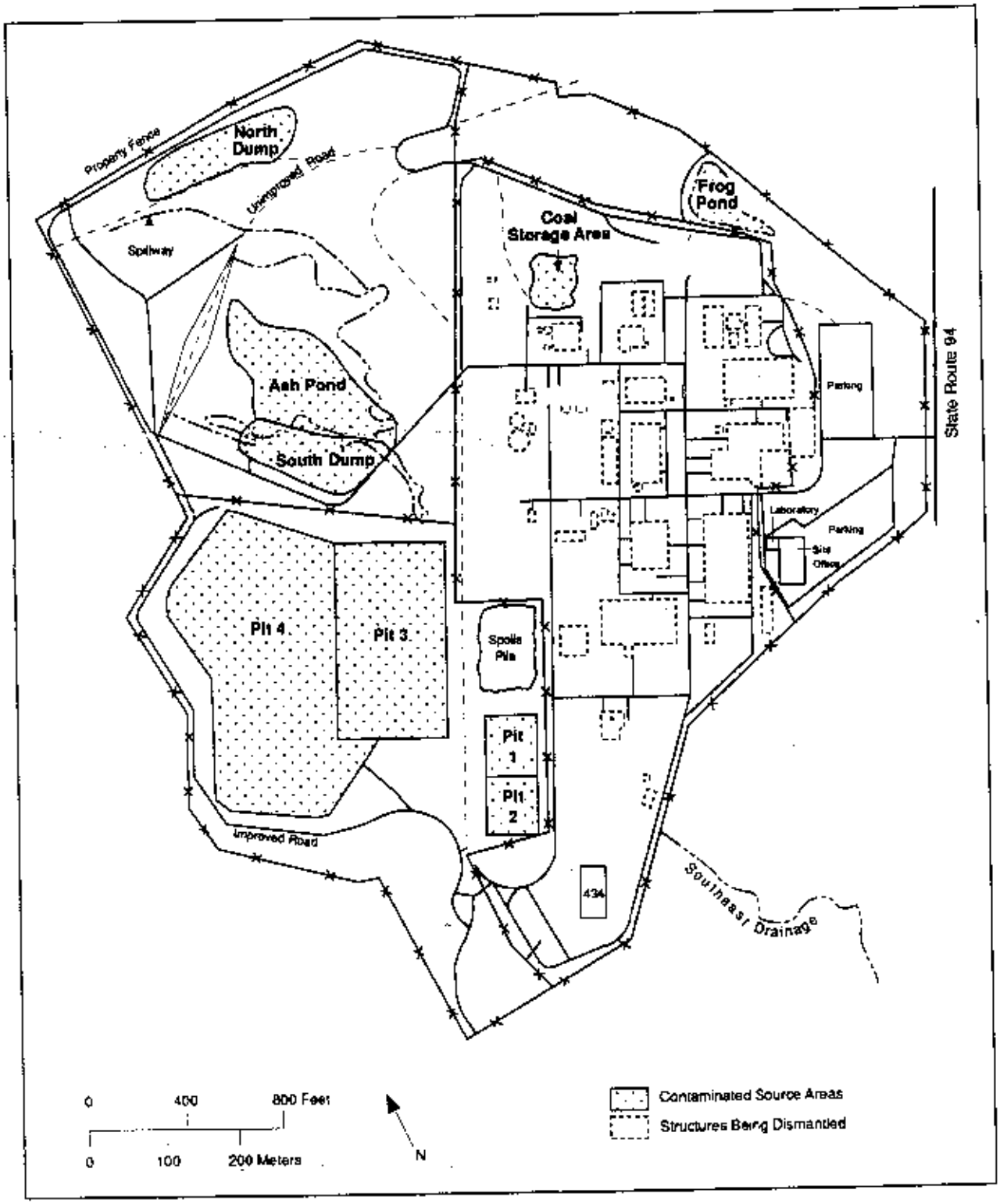


FIGURE 1.5 General Layout of the Chemical Plant Area

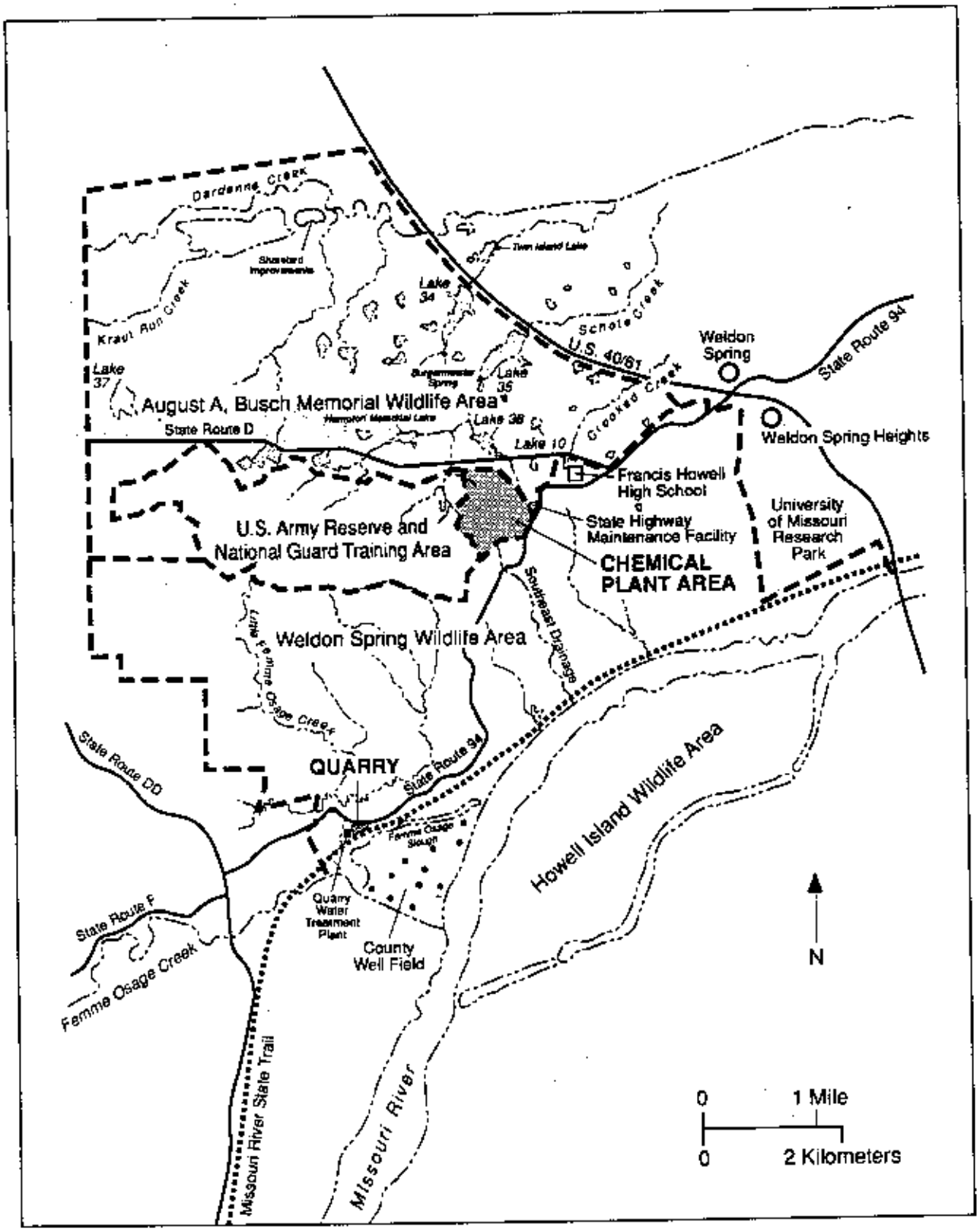


FIGURE 1.6 Surface Features near the Weldon Spring Site

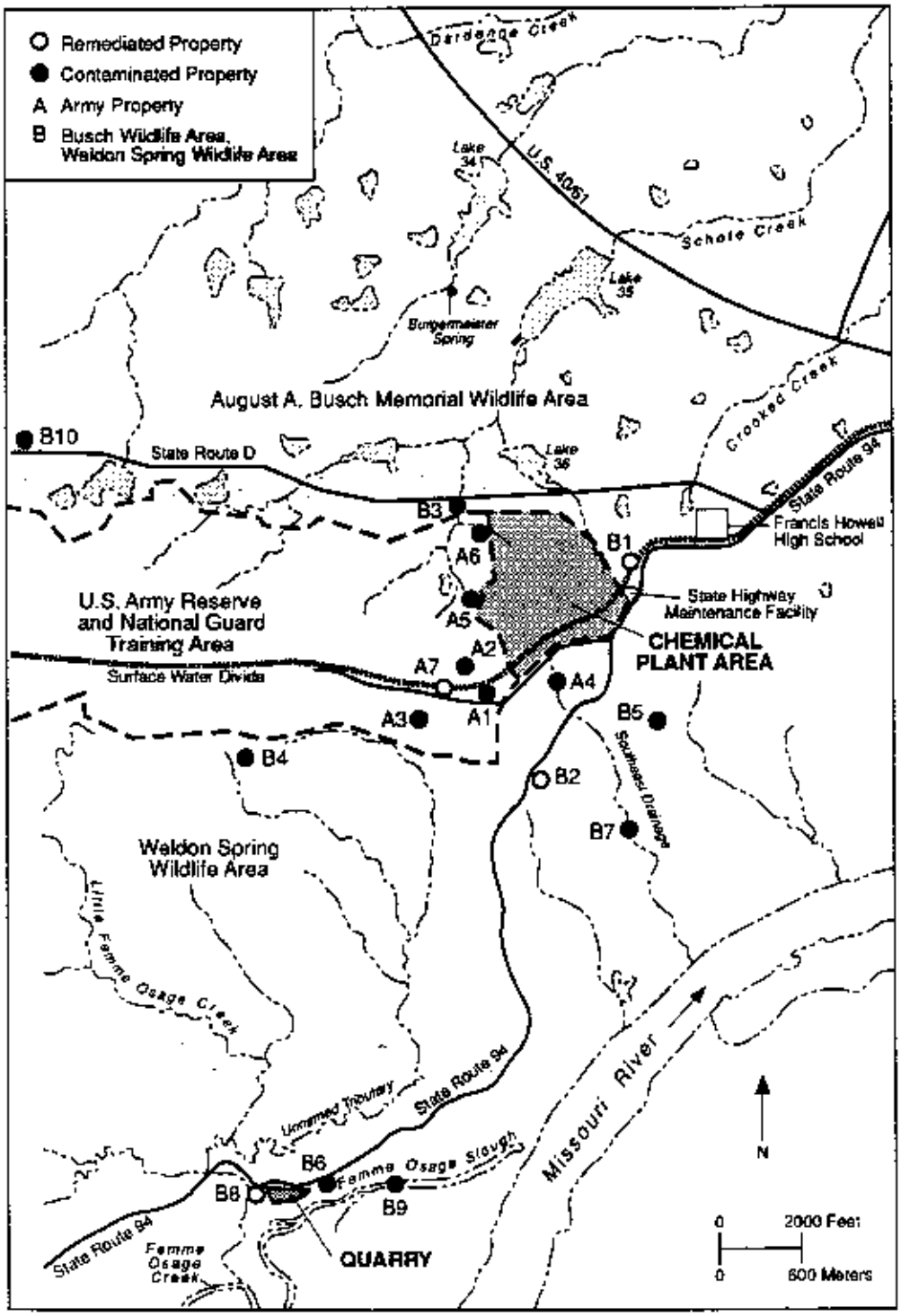


FIGURE 1.7 Location of Contaminated Vicinity Properties in the Area of the Weldon Spring Site

Several chemical and radioactive waste streams were generated during plant operations. These streams were piped to the raffinate pits where the solids settled out and the supernatant liquids were decanted to the plant process sewer; this sewer drained off-site into the Southeast Drainage, which empties into the Missouri River (Figure 1.6). Localized areas on-site were also used to dispose of solid waste; most solid waste was disposed of in the quarry.

After the AEC closed plant operations, the Army reacquired the chemical plant area in 1967 and initiated decontamination and dismantling operations to convert the facility for herbicide production. Much of the resultant debris was placed in the quarry; some debris was also placed in one of the raffinate pits. The extensive decontamination effort and associated costs required to meet extant radiological contamination limits, combined with reduced requirements for the herbicide, resulted in cancellation of the herbicide project in 1969 prior to any production. In 1984, the Army repaired several buildings, conducted additional decontamination, and isolated some contaminated equipment.

The Army retained responsibility for the land and facilities at the chemical plant until 1985, at which time DOE assumed custody. During that same year, the EPA proposed to list the quarry on its NPL; this listing occurred in 1987 (EPA 1987d). In 1988, the EPA proposed to expand the listing to include the chemical plant area; this listing occurred in 1989 (EPA 1989h). The balance of the former Weldon Spring Ordnance Works property, which is adjacent to the DOE portion and for which the Army has responsibility, was proposed for NPL listing in 1989; this listing occurred in 1990 (EPA 1990d).

1.2.3 Summary of Site Contamination

An extensive characterization program has been conducted over the past several years to determine the nature and extent of site contamination; sample collection and analysis has generated more than 120,000 data records. Additional information continues to be collected through the ongoing site environmental monitoring program. Sampling results for soil, surface water, groundwater, and air are described in the RI and summarized in Chapter 2 of this document. Several general sources of contamination on-site have been identified from the results of this sampling effort (Figure 1.5). Information on the types and concentrations of contaminants in various media at these and other areas impacted by the site is presented in the RI. General information is further described in Chapter 2 and is summarized as follows:

- Surface water impoundments — four raffinate pits, Frog Pond, and Ash Pond — contain radionuclides, primarily thorium and uranium; metals such as arsenic and chromium; and inorganic anions such as nitrate and sulfate;

- Some chemical plant buildings and support structures are contaminated with asbestos, PCBs, and radionuclides (including radon gas);

- Former dump areas, the North Dump and South Dump, contain radionuclides such as uranium and thorium and metals such as arsenic and lead; and

- The former coal storage area contains metals such as arsenic and selenium, inorganic anions such as sulfate, and polycyclic aromatic hydrocarbons (PAHs).

In addition to these discrete areas, site soil contains generally low levels of radionuclides such as uranium, thorium, and radium; some heavy metals such as arsenic and lead; and inorganic ions such as sulfate. Groundwater in the shallow aquifer contains nitrate, sulfate, nitroaromatic compounds, uranium, and some heavy metals (see Appendix B).

Several off-site locations are also radioactively contaminated as a result of releases from the site. Low levels of radioactivity (primarily uranium) are present in several small areas of soil; in surface water and sediment from Lakes 34, 35, and 36 of the Busch Wildlife Area; and in Burgermeister Spring and springs in the Southeast Drainage. Some higher levels of radionuclides (e.g., uranium, thorium, and radium) are present in sediment at certain locations in the Southeast Drainage because of past operational discharges.

1.3 SCOPE OF THE BASELINE ASSESSMENT

A BA is prepared to evaluate potential health risks and environmental impacts associated with a contaminated site, assuming that no cleanup actions are taken beyond those that have already occurred. Therefore, this assessment of the Weldon Spring site assumes the status quo, i.e., conditions that exist in early 1992, without considering interim actions that have been finalized but not yet implemented. Such actions include the on-site storage of bulk waste excavated from the quarry, treatment of surface water from the raffinate pits and ponds, and dismantlement of site structures with on-site storage of debris (see Section 1.1.2 of this document and Chapter 1 of the FS).

To the extent possible, this assessment follows EPA's recent guidance for both human health evaluation (EPA 1989j, 1991b) and environmental evaluation (EPA 1989i) at hazardous waste sites. The appropriate NEPA elements of a no-action assessment have also been incorporated into this BA, in accordance with the integrated CERCLA/NEPA process (Section 1.1.1). The basic approaches to risk assessment are similar under CERCLA and NEPA; however, certain assumptions differ, such as the time period prior to a response and the retention or loss of institutional controls. In addition, for CERCLA documents, risks are estimated for reasonable maximum exposures that reflect site-specific consideration of likely future land use. In previous NEPA documents prepared for DOE sites, exposures have often been estimated on the basis of bounding-case assumptions (e.g., for a resident intruder), which are typically very conservative. The level of detail and stepwise quantification of risk calculations presented under CERCLA can also be more extensive than that typically presented under NEPA. Finally, until recently, the level of environmental evaluation under CERCLA was typically less standardized than that developed under NEPA. However, the EPA is in the

process of defining a more prescriptive approach to ecological assessment (EPA 1991d). Because the CERCLA/NEPA process is already under way for the Weldon Spring site, this BA incorporates recent EPA guidance to the extent practicable. The evolving EPA guidance for NPL sites will be used in planning and implementing additional ecological studies at the Weldon Spring site and in preparing future risk assessments.

To integrate the CERCLA and NEPA processes for the Weldon Spring site, this BA incorporates assumptions and methodologies most appropriate to site-specific conditions. The following discussion outlines the rationale for defining the scope of this BA.

1.3.1 Staged Approach for the Baseline Assessment

A sequenced approach is being followed to clean up the Weldon Spring site (Section 1.1.2), which means that baseline conditions will undergo a series of changes. To parallel the cleanup strategy, the BA process for the chemical plant area is actually a three-stage process that reflects site conditions as the sequenced cleanup progresses. This BA document is the first-stage assessment; it addresses existing conditions, and it focuses on short-term impacts because current conditions will soon be changing such that the long-term assessment would no longer reflect extant site conditions.

The second-stage assessment (the "rebaseline" health assessment) is presented in Appendix E of the FS; it addresses conditions as they will exist at the site in the near future after the approved interim actions have been implemented. These actions include the on-site storage of bulk waste excavated from the quarry, the removal and treatment of contaminated surface water, and the dismantlement and on-site storage of site structures. Hence, the reassessment of baseline conditions in the FS evaluates potential impacts associated with no further actions beyond those that have already been finalized for the site.

The third-stage assessment will be presented in the BA prepared for the groundwater operable unit. The groundwater BA will be prepared within the next several years and will address the new baseline conditions at the site as they will exist after implementing the cleanup decisions resulting from the current RI/FS-EIS.

1.3.2 Time Period

The scope of the "future condition" scenario in a BA typically extends into the long term at a Superfund site for which no party has undertaken cleanup responsibility. This scope is based on the need to identify potential risks associated with future land-use conditions and the uncertainty regarding the time of initiation of remedial action at such sites. For cases in which timing is unclear, the BA serves to conservatively bound potential impacts if no action is taken, in order to justify the use of enforcement actions or the expenditure of EPA funds for site cleanup. However, conditions at the Weldon Spring site differ considerably from those at such a Superfund site. That is, DOE has (1) assumed responsibility (including fiscal allocations) for site cleanup, (2) made a commitment to pursue a timely response, and (3) demonstrated

considerable progress with cleanup activities to date. Further, because of the phased cleanup approach for the Weldon Spring site, the assessment of baseline conditions at the chemical plant area is being conducted in three stages. The first stage emphasizes the short term because conditions will change within the next several years as interim actions are implemented, such as taking down the buildings and pumping water out of the raffinate pits. Therefore, the time period considered reasonable for the short-term on-site scenarios in this BA begins now and extends over the next 10 years; that for the off-site scenarios extends over the next 30 years.

To estimate possible long-term impacts at a site at which institutional controls are in place, it is typically assumed that these controls are lost in the somewhat distant future, e.g., after 100 years (see Section 1.3.3). However, because the first-stage baseline conditions at the Weldon Spring site are already beginning to change, it must be hypothetically assumed that such controls are lost much earlier in order to present a long-term analysis that is not inconsistent with the time frame over which the interim actions will be initiated; by this approach, impacts associated with no action can be more appropriately evaluated.

Thus, the long-term component of this BA does not present a range of potential impacts that could occur if no cleanup actions were initiated for a very long time because certain actions are already well into the design and preparation stage, so the possibility for a variety of exposure scenarios is highly unlikely. Rather, under the staged approach, the long-term future assessment that considers several exposure scenarios (including an on-site resident) is addressed in the other sequenced risk assessments for the project — i.e., in both the rebaseline assessment presented in the FS for the current response action and in the BA that will be prepared for the groundwater operable unit. The time period for the long-term assessment in this BA is assumed to begin essentially now, as for the short-term assessments, and to extend over the next 30 years. The more extensive long-term assessments — in the second-stage assessment presented in the FS and the third-stage assessment to be prepared for the groundwater operable unit — will be more typical; that is, the time period for these assessments will begin and extend from at least 100 years in the future to, for example, 200 years and beyond.

1.3.3 Institutional Controls and Exposure Scenarios

Under the CERCLA process, the BA typically considers impacts that could occur in the absence of remedial action at a site under both current conditions, which include access controls such as fences and security personnel, and projected future conditions, for which no institutional controls are assumed. However, under the NEPA process, the impact assessment for the no-action alternative typically addresses the status quo at a site, which may assume the retention of institutional controls such as access restrictions and monitoring for 100 years. Although loss of institutional controls at the Weldon Spring site is in fact implausible, this loss has been hypothetically assumed for the current BA to address that element of the CERCLA baseline assessment process. The future loss of institutional controls assumed for the second- and third-stage assessments for the site might be somewhat more plausible because of the extended future time periods addressed in those assessments. However, DOE intends to maintain institutional controls at the Weldon Spring site for as long as the waste remains there.

The exposure assessments presented in Chapter 3 (human health assessment) and Chapter 7 (ecological assessment) of this BA describe the potential receptors and locations that were selected to assess current baseline impacts on humans and other biota associated with the Weldon Spring site. The potential receptors and receptor locations were determined from a reasonable area of impact, considering site-specific environmental conditions, the results of site characterization and environmental monitoring, and the nature of potential exposure pathways. The rationale for this determination is provided in the following discussion.

Discrete receptors and locations considered in the human exposure assessment were selected on the basis of current land-use conditions, in accordance with the time period evaluated for this BA. The selection of potential receptors also considered the future hypothetical loss of institutional controls. Most of the land surrounding the Weldon Spring site is wildlife area, so local land use is primarily recreational. On the basis of this information, potential human exposure scenarios considered reasonable for the general public are an off-site recreational visitor, an on-site trespasser, and an on-site recreational visitor. The off-site recreational visitor and on-site trespasser scenarios represent current land-use conditions with maintenance of institutional controls, whereas the on-site recreational visitor scenario represents hypothetical future conditions under which land use would remain the same but institutional controls would be lost, i.e., access restrictions such as fences, locked gates, and security guards would no longer exist. Off-site receptor locations include the Southeast Drainage, three lakes in the adjacent wildlife area, and localized areas of soil; on-site locations include the chemical plant buildings, surface water impoundments, and sitewide soil.

Residential use of the chemical plant area is unlikely in the short term because it is inconsistent with (1) the nature of the site — including the presence of surface water, debris at the impoundments and other scattered locations, and deteriorating buildings; and (2) the surrounding land use and reasonable projections thereof — considering federal (Army) and state ownership and the state's plans for continued recreational use of the wildlife areas. Hence, a recreational scenario rather than a residential scenario has been considered in this risk assessment. This approach is consistent with EPA guidance on incorporating site-specific considerations into baseline human health evaluations (EPA 1989j). However, the residential scenario is addressed in the second-stage rebaseline assessment presented in Appendix E of the FS because that scenario might occur in the extended long-term future if controls were assumed to be lost at the site.

Similar to human receptors and locations, the potential ecological receptors and locations considered for the environmental evaluation of this BA were determined from site-specific information, including (1) the results of site characterization and environmental monitoring studies, (2) the nature of potential exposure pathways, and (3) state management of the adjacent wildlife areas. The receptors considered in this assessment include herbaceous and woody vegetation, invertebrates, mammals, and birds. Receptor locations include contaminated surface water impoundments and old-field, upland, and bottomland forest habitats (see Chapter 7).

1.4 REPORT ORGANIZATION

This BA serves as the combined baseline (i.e., no-action) assessment of potential health and environmental impacts associated with current conditions at the chemical plant area of the Weldon Spring site. As described in EPA guidance, a baseline risk assessment is typically prepared in four basic steps (EPA 1989j). The organization of this BA is consistent with those four steps.

The first step — data compilation and analysis — consists of gathering and evaluating site data. Historical information is considered as part of this step, as is information on the concentration, toxicity, mobility, and persistence of the various chemical and radioactive contaminants present at the site. The objective of the first step is to identify for detailed assessment the "contaminants of concern," i.e., the significant contributors to potential impacts.

The second step — the exposure assessment — involves identifying the contaminant sources, the possible mechanisms of contaminant release, and their environmental fate. Potential receptors are then identified for each exposure pathway, and exposure point concentrations are determined from measured and/or modeled values. Finally, pathway-specific intakes are calculated for use in the subsequent risk calculations.

The third step — the toxicity assessment — consists of identifying the types of adverse effects associated with an exposure and evaluating the extent and quality of available toxicological data. This step provides information for identifying potential hazards at the site. Toxicity values are compiled (i.e., dose conversion factors and risk estimators for the radioactive contaminants and reference doses and slope factors for the chemical contaminants), and the uncertainties in these values are discussed in order to understand the limitations in subsequent calculations.

The fourth step — risk characterization — combines the results of the exposure and toxicity assessments to present a quantitative and qualitative prediction of risks for the site. For the human health assessment, the potential noncarcinogenic health effects from chemicals are assessed by comparing the estimated average daily exposures (intakes) with reference doses established by the EPA; a reference dose is the chemical-specific average daily dose to which an individual can be exposed without likely adverse health effects. Potential carcinogenic risks for both radiological and chemical exposures are characterized in terms of the increased likelihood of cancer; this characterization is based on toxicity values and risk factors that provide a relationship between dose or intake and the likelihood of cancer induction.

In contrast with the human health assessment, standard conversion factors relating exposure to impact are not available for the ecological assessment. Hence, the characterization of ecological impacts is more qualitative than quantitative. Nevertheless, data available in the literature from field and laboratory studies have been used in this BA to evaluate contaminants of concern and possible ecological impacts associated with the site in the absence of cleanup.

The four basic steps of the risk assessment are reflected in the organization of this BA; the elements addressed for the human health assessment are provided in parallel for the

ecological assessment, as appropriate, to provide a consistent discussion of potential baseline impacts. The report is organized as follows:

- Chapter 1 presents general information on the site background and current environmental conditions.
- Chapter 2 reviews the data collection effort and identifies contaminants of concern for the human health evaluation.
- Chapter 3 describes the human exposure assessment, including contaminant fate and transport, potential receptors, exposure routes, and estimated contaminant intakes.
- Chapter 4 provides human toxicity information for the contaminants of concern.
- Chapter 5 presents the methodology for and results of the health risk characterization.
- Chapter 6 summarizes the contaminants of concern, exposure and toxicity assessments, and risk characterization for the human health assessment.
- Chapter 7 presents baseline information on the ecological resources at the site; identifies ecological contaminants of concern, potential receptors, and exposure routes; summarizes ecological toxicity information; and presents the impact characterization for the baseline assessment.
- Chapter 8 describes potential impacts associated with other environmental resources in the absence of site cleanup.
- Chapter 9 is a list of the references cited in the BA.
- Chapter 10 provides information regarding technical contributions to this BA.
- Appendixes A through E provide supporting information for chemical detection limits, groundwater, area biota, ecological toxicity, and uptake and assimilation factors for vegetation and wildlife.

2 IDENTIFICATION OF CONTAMINANTS OF CONCERN

Information relevant to the collection and evaluation of data for the human health risk assessment is summarized in this chapter. These data were used as the basis for selecting the radioactive and chemical contaminants of potential concern for this assessment. The selection process included a review of previous operations at the site that contributed to current levels of contamination, as described in Section 2.1. On the basis of this information, a conceptual model of potential pathways of human exposure at the site was developed, and this model is outlined in Section 2.2. General issues relevant to the design of the contaminant characterization studies are discussed in Section 2.3, and the results of the contaminant characterization studies are summarized and evaluated in Section 2.4. These results and the site operational history were considered in selecting the contaminants of concern.

2.1 SITE OPERATIONAL HISTORY

A general history of the Weldon Spring site is presented in Section 1.2.2. A brief summary of the operations and chemicals associated with the ordnance works and the uranium feed materials plant is presented in Sections 2.1.1 and 2.1.2, respectively. A detailed description of these processes is presented in Appendix D of the RI report (DOE 1992c) and in the chemical soil investigation sampling plan (MK-Ferguson Company and Jacobs Engineering Group 1988b).

2.1.1 Ordnance Works Operations

The Weldon Spring Ordnance Works consisted of 18 TNT and 2 DNT explosives manufacturing plants situated within the current boundaries of the chemical plant area and the adjoining U.S. Army Reserve and National Guard Army Training Area. Three TNT plants and part of a fourth were located within the current boundaries of the site. The former locations of the TNT plants, roads, and process lines relative to current site features are shown on Plate 1 of the RI report.

The primary chemicals used to manufacture TNT and DNT were toluene, nitric acid, sulfuric acid, and sellite (sodium sulfite). Toluene was reacted with nitric and sulfuric acids to produce TNT. By-products of the reaction were then removed from the reaction mixture by treatment with sellite. The spent sellite solution, which contained extracted sulfonate derivatives (called red water), was the major waste stream of the production process. Another waste stream produced during one of the purification steps was essentially a dilute nitrator (called yellow water). The processing operations involved millions of gallons of water per day for washing and mixing chemicals. Water and wastewater lines served each batch plant, and the red and yellow wastewaters were carried from the batch plants to various locations where the lines terminated. Prior to construction of wastewater treatment plants in 1943, the wastewaters were discharged via wastewater lines into seven lagoons for temporary storage. After 1943, the lines discharged into wastewater treatment plants where the wastewaters were evaporated into sludge. The

sludge was then incinerated in open burning grounds, one of which may have been located within the current site boundary.

Surface water and groundwater investigations conducted by the U.S. Geological Survey (USGS) during the early 1940s indicated that ordnance works wastewaters had also been discharged into ditches and streams and that wastewater lines and catchment tanks frequently overflowed. These observations indicated a potential for soil contamination along former wastewater lines, surface drainages, production areas, waste lagoons, and wastewater catchment tanks.

Structures and materials from the ordnance works operations were decontaminated and dismantled following cessation of operations at the end of World War II and again prior to construction of the uranium feed materials plant. More than 200 buildings were destroyed, and approximately 80 t (90 tons) of nitroaromatic compounds and residues were burned. The results of the RI characterization studies indicate that these activities removed the major source areas associated with the ordnance works. Considerable regrading was also performed prior to construction of the uranium plant, and this was taken into consideration during development of the soil sampling plans.

2.1.2 Uranium Processing

Uranium ore concentrates and recycled scrap were processed at the uranium feed materials plant to produce uranium metal; intermediate forms in the chemical processing operation included uranium dioxide, uranium trioxide, and uranium tetrafluoride. An average of 14,000 t (16,000 tons) of uranium material was processed each year. In addition, a small amount of thorium ore concentrate was processed at the plant. Constituents of potential concern associated with the uranium feed materials plant include radionuclides of the uranium-238, thorium-232, and uranium-235 decay series (Figures 2.1, 2.2, and 2.3); metals associated with uranium and thorium ores; and process chemicals. Metals associated with uranium and thorium ores include arsenic, cobalt, copper, lead, manganese, molybdenum, nickel, selenium, vanadium, and zinc (Dreesen et al. 1982; Stokinger 1981). The process chemicals used at the plant to manufacture uranium metal included nitric, sulfuric, and hydrofluoric acids; hexane; tributylphosphate; magnesium; sodium carbonate; and sodium hydroxide.

Plant operations generated several chemical and radioactive waste streams, including raffinates from the refinery operations and magnesium fluoride slurry (washed slag) from the uranium recovery process. These streams were piped to the raffinate pits, where the solids settled out and the supernatant liquids were decanted to the plant process sewer, this sewer drained off-site to the Missouri River through the Southeast Drainage. The solids remaining in the pits consist of (1) silica and insoluble metals and oxides associated with the uranium ore feed material, (2) hydroxides and other precipitates formed from lime neutralization of the raffinate, and (3) washed slag residue from uranium production (Peterson et al. 1988).

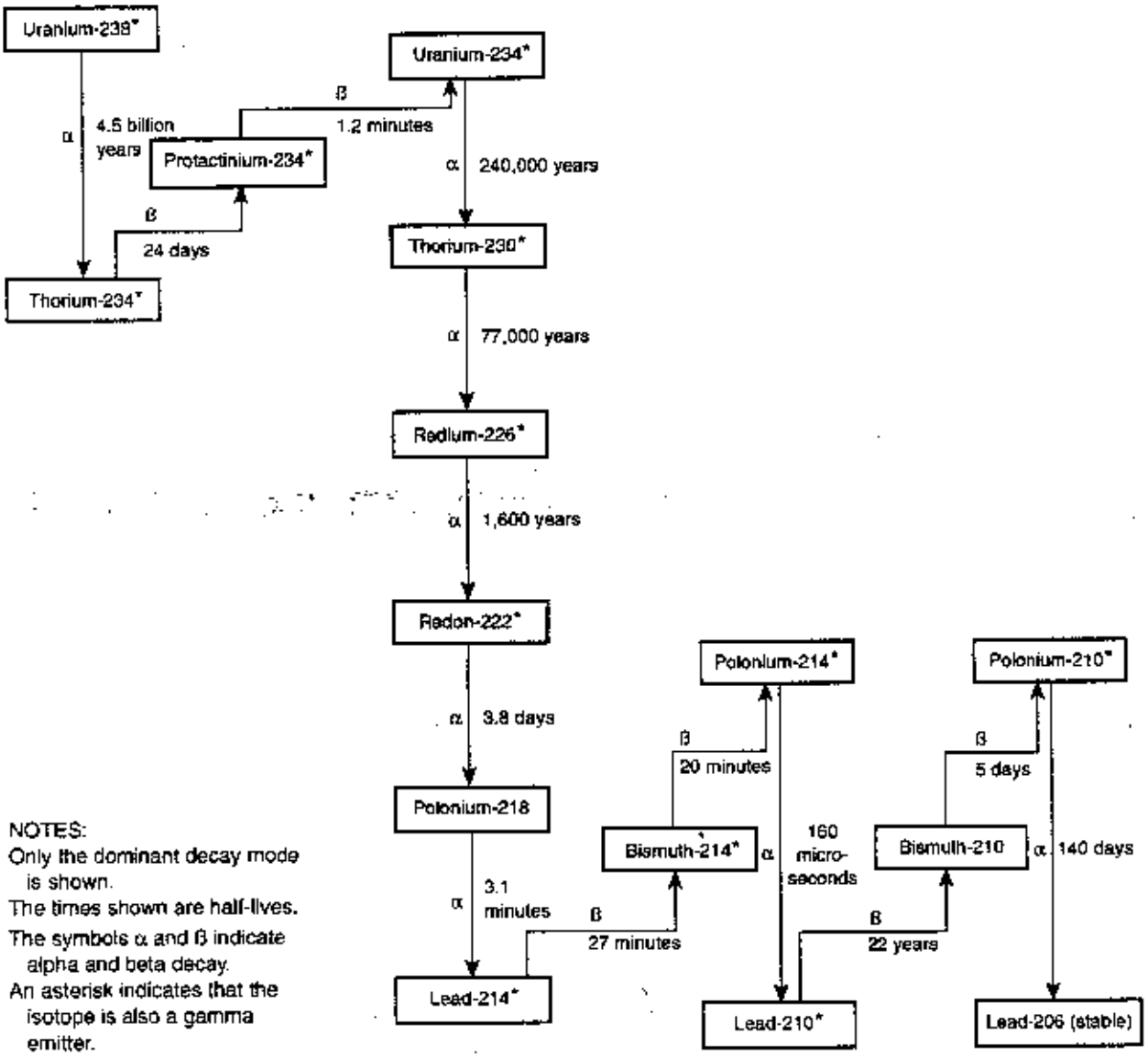
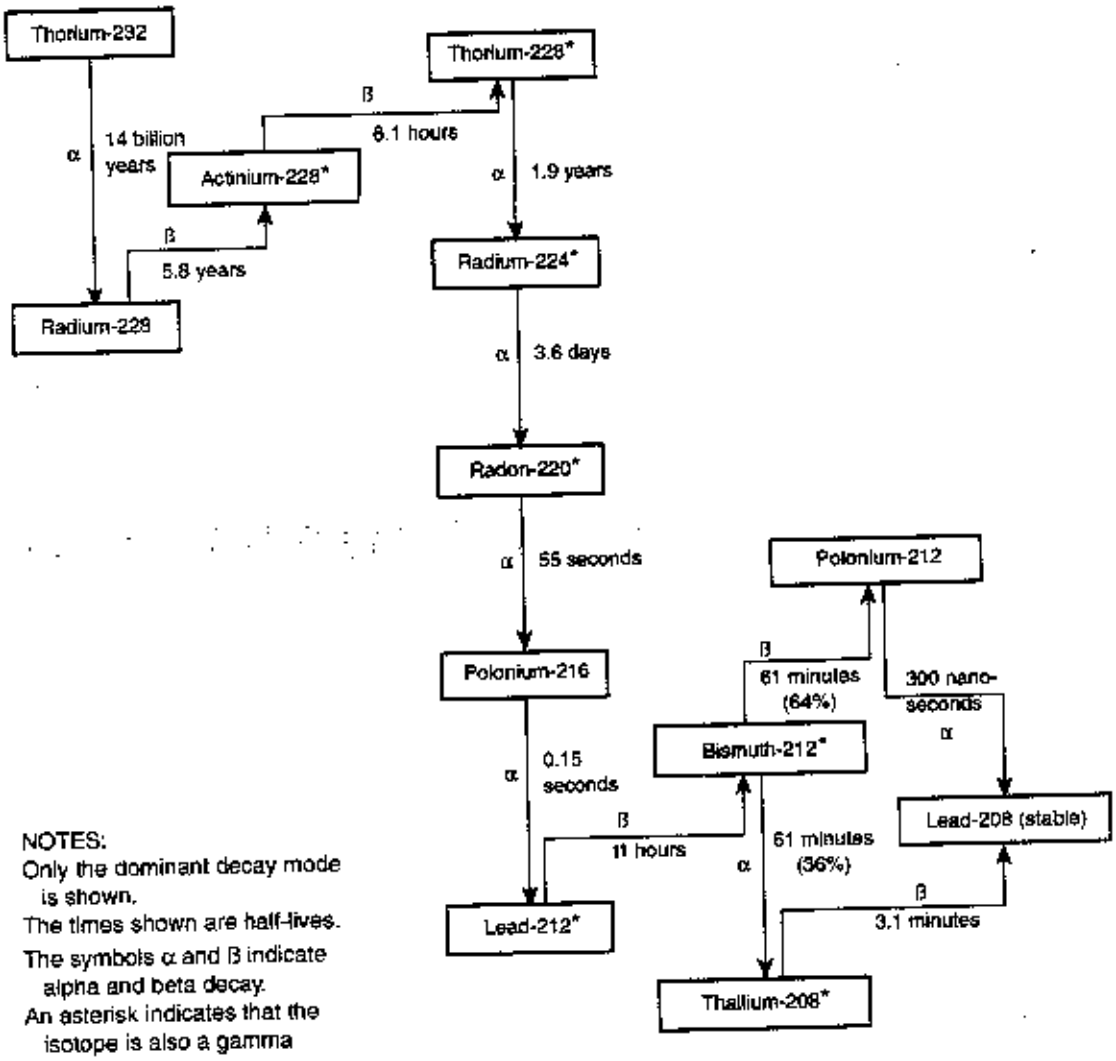


FIGURE 2.1 Uranium-238 Radioactive Decay Series



NOTES:
Only the dominant decay mode is shown.
The times shown are half-lives.
The symbols α and β indicate alpha and beta decay.
An asterisk indicates that the isotope is also a gamma emitter.

FIGURE 2.2 Thorium-232 Radioactive Decay Series

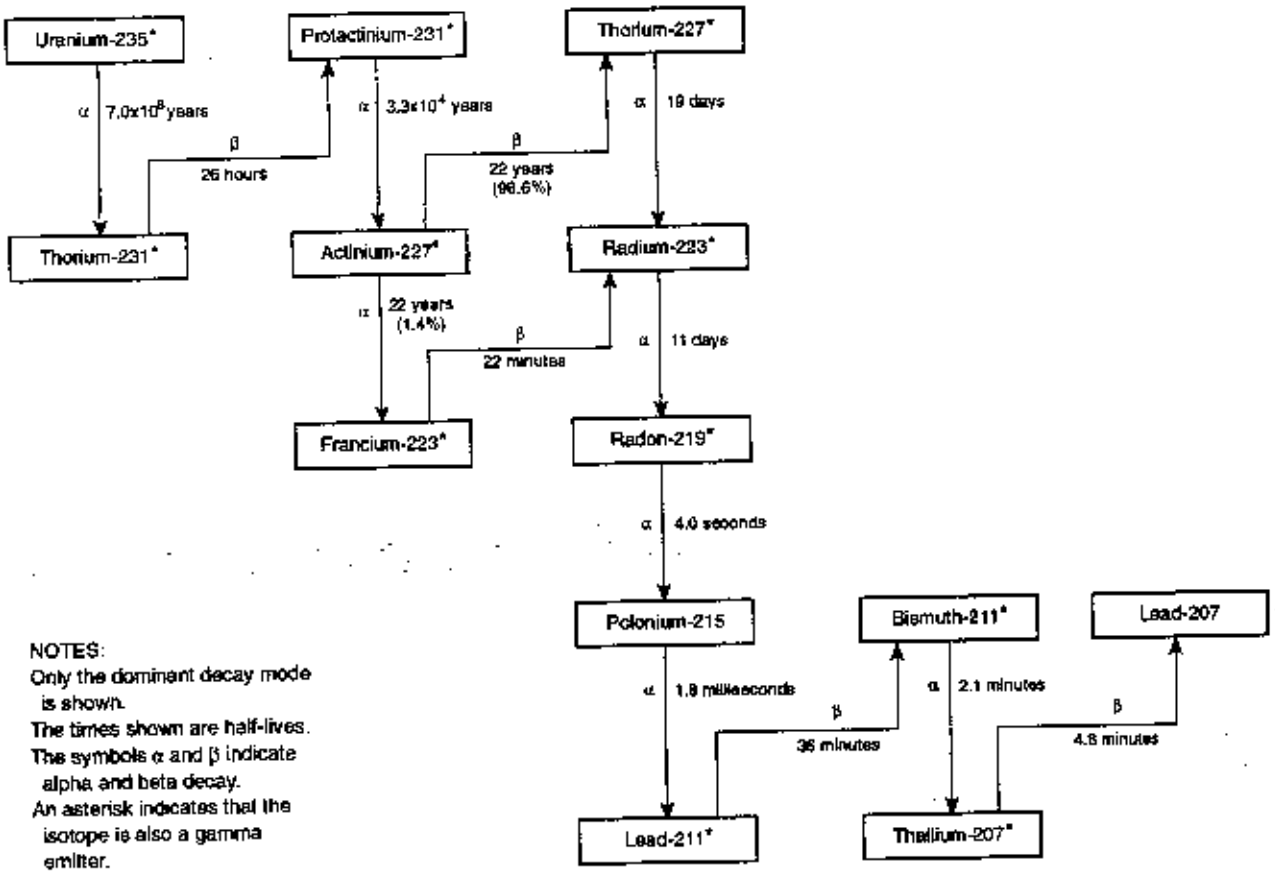
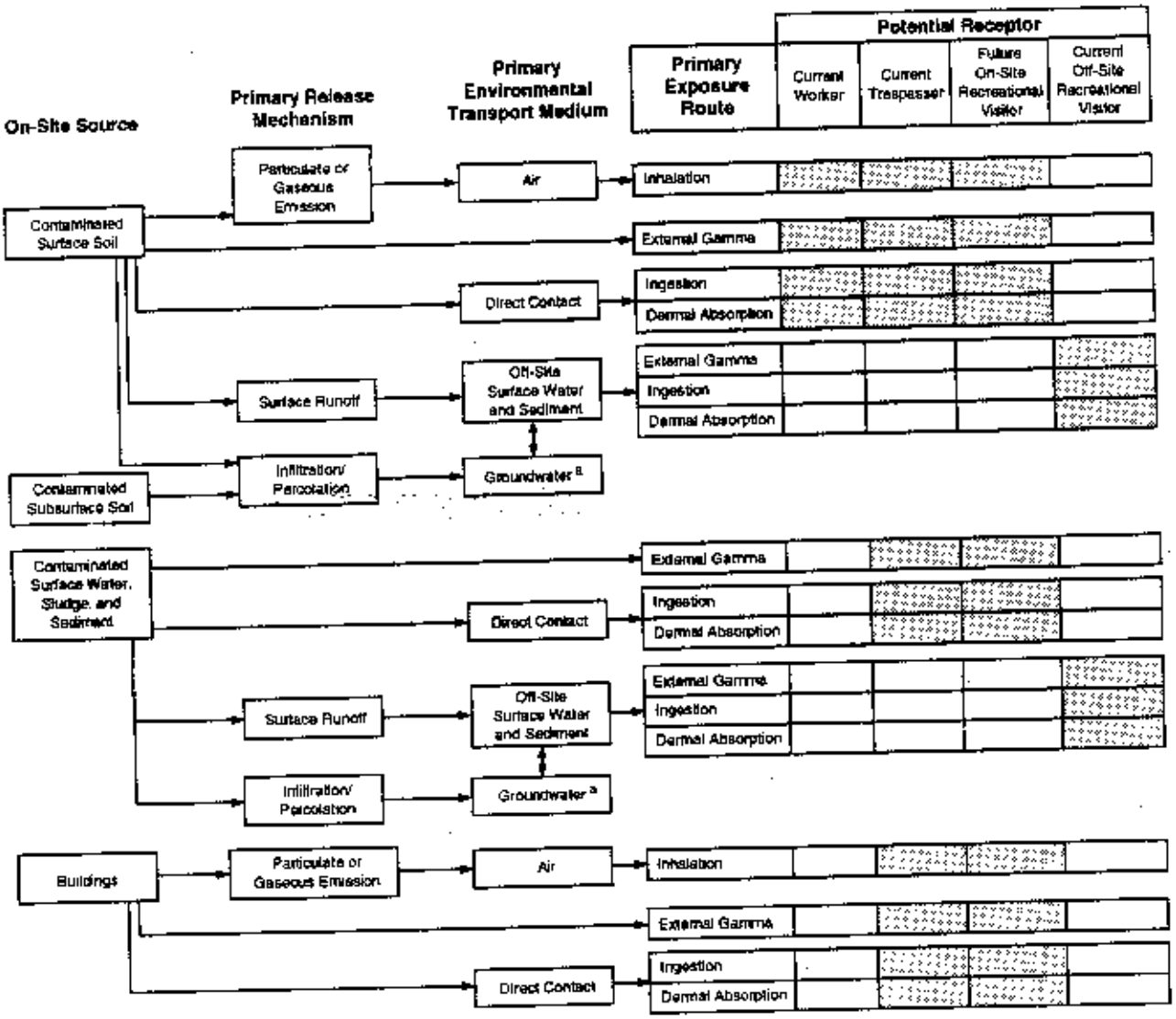


FIGURE 2.3 Uranium-235 Radioactive Decay Series

2.2 SITE CONCEPTUAL MODEL

During the initial phases of the RI/FS-EIS process, a conceptual model of the fate and transport of contaminants associated with the site was developed to determine the potential exposure of human receptors. Characterization studies developed for the RI were based, in part, on this model. The model considers exposures that could result from receptor contact with contaminant sources within the site boundary and contaminants transported to off-site areas. The site conceptual model is diagrammed in Figure 2.4 (shaded boxes represent the relevant exposure routes for each receptor). Components of this model are described in Sections 2.2.1 and 2.2.2.

Vicinity property soil is not depicted separately in Figure 2.4 because it does not represent a primary contaminant source; related exposures for the off-site receptor are those associated with external gamma irradiation and incidental ingestion. Exposure to groundwater is not quantified in this BA because the potential receptors considered in this assessment (Figure 2.4) would not be exposed. A preliminary evaluation of this pathway is presented in the rebaseline assessment of the FS (Appendix E), in which exposures are evaluated for a hypothetical future resident. A more detailed evaluation will be presented in upcoming



[a] The groundwater pathway is not addressed in this BA (see text).

FIGURE 2.4 Site Conceptual Model

documentation for the groundwater operable unit, using additional data that will be collected during the next several years.

2.2.1 On-Site Exposure Pathways

Human exposure could result from contact with contaminated source areas and environmental media within the site boundary. The principal source areas and media identified at the site are (1) chemical plant buildings and support structures; (2) surface water and sludge from the four raffinate pits; (3) surface water and sediment from Frog Pond; (4) surface water, sediment, and soil from Ash Pond; and (5) contaminated soil at the North Dump, the South Dump, the coal storage area, and other scattered locations. The main contaminants present at these source areas are radionuclides, metals, inorganic anions, asbestos-containing material,

PAHs, PCBs, and nitroaromatic compounds. Under current conditions, access to these areas by the general public is restricted. The site is surrounded by a 2-m (7-ft) chain link fence topped by barbed wire, and security guards are present 24 hours per day. Hence, any exposures to members of the general public would be limited to those of an occasional trespasser who might gain access to the site.

Exposures incurred by such a trespasser would be intermittent, and the primary exposures associated with environmental media would be expected to occur through external gamma irradiation and through direct contact with contaminated soil and surface water, resulting in incidental ingestion of soil and water and dermal exposure. Other potential exposures associated with the chemical plant buildings are external gamma irradiation; inhalation of radon, airborne particulates, and asbestos; and direct contact with contaminated residues, resulting in incidental ingestion and dermal exposure. Under future conditions, assuming that access to the site would no longer be restricted, the frequency of exposures to members of the general public might increase but the specific routes of exposure would remain the same.

For workers, limited exposures could occur even though routine maintenance operations at the site would be conducted in accordance with site health and safety plans. Potential routes of worker exposure are external gamma irradiation; direct contact with contaminated soil, resulting in incidental ingestion and dermal exposure; and inhalation of airborne particulates.

2.2.2 Off-Site Exposure Pathways

Evaluation of contaminated sources and potential release mechanisms indicates that four main pathways could result in contaminant transport to off-site areas:

- Surface water transport to off-site drainage areas,
- Surface water loss to groundwater (via losing streams),
- Groundwater discharge to surface water, and
- Leaching from surface and/or subsurface material to groundwater.

The site is divided into three general drainage systems: (1) Ash Pond and the raffinate pit area drain to the northwest, (2) Frog Pond and related streams drain to the north, and (3) the southeastern corner of the chemical plant area drains to the southeast (Figure 1.6). The first two drainage systems are hydrologically connected to springs, streams, and lakes in the Busch Wildlife Area; dye-tracing studies indicate that some surface water lost to groundwater beyond the northwestern site boundary from the first drainage system reemerges at Burgermeister Spring (Missouri Department of Natural Resources 1989). The third system is hydrologically connected to springs and streams in the Weldon Spring Wildlife Area. These connections have resulted in contaminant transport via surface water to off-site areas. These wildlife areas were sampled as part of the site characterization effort for the RI, and the following locations were identified as having been impacted by contaminant releases from the site: Lakes 34, 35, and 36 and

Burgermeister Spring in the Busch Wildlife Area; and the Southeast Drainage, which runs through the Weldon Spring Wildlife Area to the Missouri River. Other off-site areas, such as other surface drainages, have been impacted to a lesser degree by releases from the site (see Section 4.4 of the RI).

Concentrations of uranium elevated above background levels have been measured in surface water and sediment from Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage. Elevated concentrations of thorium and radium have also been measured in the limited samples of sediment and soil from the Southeast Drainage. Elevated concentrations of fluoride, nitrate, and nitroaromatic compounds were detected infrequently in Burgermeister Spring and in the intermittent flows in the Southeast Drainage. The site can be identified as the source of uranium, thorium, radium, fluoride, and nitrate contamination, but no current on-site source of nitroaromatic compounds has been identified that could be impacting the off-site drainage areas (see Section 3.2.2.2). Because the Weldon Spring site is adjacent to an Army NPL site formerly used for ordnance operations (Section 1.2.2), sources of nitroaromatic compounds other than the Weldon Spring site are present in the area. For example, old pipe remnants from previous Army operations have been found in the Southeast Drainage. Other contaminants associated with the site have not been measured at significantly elevated concentrations at off-site locations.

The Busch and Weldon Spring wildlife areas are used for hiking, hunting, and fishing. Contamination from the site is restricted to a few lakes and springs, and potential exposures are associated mainly with recreational use of these lakes and possible exposures to the springs. Wading and swimming are not permitted at the lakes; nevertheless, swimming has been evaluated in this BA to address the possibility of "trespassers" who would violate posted restrictions. Recreational visitors to the wildlife areas could potentially be exposed to contaminants by the following routes of exposure: (1) external gamma irradiation, (2) ingestion of surface water, (3) incidental ingestion of lake and stream sediments, (4) dermal absorption of contaminants while swimming, and (5) ingestion of biota (e.g., fish and game). These areas are expected to be maintained as wildlife areas into the reasonably foreseeable future; therefore, the land use would continue to be recreational, and potential future exposures are considered to be the same as under current conditions.

Groundwater in the weathered limestone at the Weldon Spring site moves off-site by Darcian diffuse flow, with superimposed conduit flow in the weathered portion of the bedrock. Elevated concentrations of certain metals, inorganic anions, and radionuclides have been detected in on-site wells, but currently available data suggest that the distribution of these contaminants may be localized. Groundwater in the weathered portion of the bedrock that has been impacted at the site is not currently used as a drinking water source. Available data indicate that the raffinate pits may constitute a major source of groundwater contamination (Appendix B). For example, elevated levels of uranium have been detected in wells adjacent to the pits and just beyond the site boundary. On the basis of current data, contamination of groundwater with nitroaromatic compounds appears to extend beyond the general site area as a result of previous activities at the original ordnance works.

In addition to the exposure pathways already identified for off-site receptors, exposures could occur via inhalation if contaminated airborne particulates and gases (e.g., radon) were released from the site. However, the airborne levels of fugitive dust and radon gas are expected to be low because the wind speeds at the site are low relative to speeds expected to cause wind erosion (Section 3.1.1.1) and because most of the site is vegetated, most of the radium-contaminated material is covered by water, and much of the surrounding area is heavily wooded. Data from extensive air monitoring support the expectation of low levels of fugitive dust and radon; radionuclide levels at the site boundary are at background levels, indicating that the air pathway is not a significant exposure route to off-site receptors under current site conditions (MK-Ferguson Company and Jacobs Engineering Group 1991; DOE 1992c).

2.3 DATA COLLECTION

Comprehensive sampling plans were developed for the site to provide data for the risk assessment and other components of the RI/FS-EIS process; these plans were developed on the basis of a site conceptual model and the extensive historical information available on past processes associated with the ordnance works and uranium feed materials plant (Section 2.1). The purpose of the sampling effort was to characterize the areas and media identified in the site conceptual model as being contaminated or potentially contaminated by site releases and to identify the parameters necessary for modeling the fate and transport of contaminants. The principal contaminant characterization studies considered in this BA are summarized in Table 2.1.* A comprehensive summary of all contaminant characterization studies is presented in Chapters 3 and 5 of the RI report.

2.3.1 Sampling Locations and Media

Sampling locations and media were identified on the basis of historical site records, characterization studies, and the site conceptual model (Section 2.2). The media and locations sampled included (1) surface and subsurface soil on-site and surface soil at vicinity properties off-site; (2) surface water and sludge from the raffinate pits on-site; (3) surface water and sediment from Frog Pond and Ash Pond on-site, and from the Busch lakes, Burgermeister Spring, and the Southeast Drainage and other nearby drainages off-site; (4) groundwater at and near the site; (5) air at and near the site; (6) air, residues, and asbestos-containing material within the chemical plant buildings on-site; and (7) vegetation, animals, and waterfowl on-site and fish from the off-site lakes. Detailed information on the sampling programs is presented in the media-specific sampling plans, including sampling methods and maps of the sampling locations (MK-Ferguson Company 1988a, 1988c; MK-Ferguson Company and Jacobs Engineering Group 1987d, 1987e, 1988b, 1988c); this information is summarized in Section 2.2.1 of the RI report (DOE 1992c). Area- and media-specific sample collection strategies are described in Section 2.4 of this BA.

*For readability, all tables are presented in sequence at the end of the text in this chapter.

Background samples were generally collected as part of the various media-specific sampling programs. Local background concentrations of radionuclides in soil in the immediate vicinity of the Weldon Spring site were reported in two separate studies (Marutzky et al. 1988; MK-Ferguson Company and Jacobs Engineering Group 1988e). Samples were collected from four locations within an 8-km (5-mi) radius of the site during the first study, and six additional locations were sampled within the same radius during the second study. Samples for determining background concentrations of metals and inorganic anions in soil were collected from a nearby location in the Weldon Spring Wildlife Area. These locations were selected because they are close to the site, have soil characteristics similar to those of site soil, and are not considered to have been contaminated by site releases. Background concentrations of radionuclides and metals in soil are also available from the literature (see Section 2.4.1).

Background samples for radioactive and chemical contaminants in surface water were collected from Lake 10 in the Busch Wildlife Area and from an upstream location of Dardenne Creek. Sediment samples were collected from Lake 37. Although these locations are not within drainage areas impacted by the site, the sampling locations at Lake 37 and Dardenne Creek may have been influenced by chemical releases from the adjacent Army property. Hence, for the chemical contaminants, the concentrations at these locations cannot be considered true background values; however, the results are presented for comparative purposes. Limited data are also available from the USGS (see Section 2.4.2).

Groundwater locations were not sampled for background concentrations as part of the RI characterization studies. However, limited data are available from the USGS (see Section 2.4.3).

2.3.2 Analytical Parameters and Methods

2.3.2.1 Radiological Analyses

The majority of sites under EPA's Superfund program do not contain radioactive contaminants, and analyses for radionuclides are not currently included in EPA's Contract Laboratory Program (CLP), an analytical program developed to identify chemical analytical methods of known quality for analyzing samples in support of the Superfund effort. However, the procedures used for radiological analyses at the Weldon Spring site are generally accepted by professionals in the environmental radiation field. Specific analytical methods for each medium are listed in Table 2.3-1 of the RI report. Because of the high degree of resolution and sensitivity of standard instruments and methods for radiological analyses, radionuclides can be detected at levels below those naturally present in the environment.

The radionuclides expected at the site are those associated with the uranium-238, thorium-232, and uranium-235 decay series (Figures 2.1, 2.2, and 2.3). These decay series include a large number of radionuclides, so related decay and toxicity information was used in combination with site-specific processing information to focus the data evaluation process.

The radiological risks associated with radionuclides of the uranium-235 decay series are expected to be much lower than those of the uranium-238 decay series at the Weldon Spring site because the abundance of uranium-235 in uranium ore is low (the activity ratio of uranium-235 to uranium-238 in the ore is 0.046 to 1) and because only a small amount of slightly enriched uranium was processed at the site. In nature, the radionuclides in these three decay series are in a state of secular equilibrium in which the activities of all radionuclides within each series are equal. Two conditions are necessary for secular equilibrium to occur: (1) the parent radionuclide must have a half-life much greater than that of any other radionuclide in the series and (2) a sufficiently long period of time must have elapsed (e.g., more than seven half-lives of the decay product having the longest half-life) to allow for ingrowth of the decay products.

Under secular equilibrium, the activity of the parent radionuclide undergoes no appreciable changes during many half-lives of its decay products. Both conditions for secular equilibrium are met for the uranium-238, thorium-232, and uranium-235 decay series in naturally occurring ores. However, this natural state is altered during the processing of uranium and thorium ores. The rate at which equilibrium conditions are reestablished depends on the half-lives of the decay products. Radionuclides with half-lives of less than 1 year will reestablish equilibrium conditions with their longer-lived parent radionuclides within several years. Thus, because operations at the Weldon Spring site ceased more than 20 years ago, it can be assumed that all radionuclides with half-lives of less than 1 year have reestablished equilibrium conditions.

This assumption is consistent with the presentation in Gilbert et al. (1989), in which the term "principal" radionuclide refers to those radionuclides in the various decay series with half-lives of more than 1 year. The decay products of any principal radionuclide in a series down to, but not including, the next principal radionuclide in the decay series are termed "associated" radionuclides and consist of radionuclides with half-lives of less than 1 year. The principal radionuclide is assumed to be in secular equilibrium with its associated radionuclides at the point of exposure. The principal and associated radionuclides for the uranium-238, thorium-232, and uranium-235 decay series are given in Table 2.2.

Because the Weldon Spring site processed mostly natural uranium, in which the activities of uranium-238 and uranium-234 are equal, the radiological doses associated with exposure to uranium-234 can be combined with those for uranium-238. Also, because the half-life of thorium-228 (1.9 years) is very short relative to the time since plant closure, it can be assumed that this radionuclide is in equilibrium with radium-228. Therefore, the radiological doses associated with exposure to thorium-228 can be combined with those for radium-228.

An isotope of radon gas — i.e., radon-222, radon-220, and radon-219 — is included in each of the three decay series. As a gas, radon tends to escape from contaminated media, thereby decreasing the activity concentrations of subsequent decay products in the decay series. Because the reduction is generally small, it has not been incorporated into this assessment. The risks associated with radon-219 would be very low compared with those from radon-222 and radon-220 because of its very short half-life (4.0 seconds); therefore, these risks have not been

quantified. In addition, exposure to radon-220 is a concern in only a few process buildings at the site. Under all other conditions, radon exposure would be dominated by radon-222.

On the basis of these data, the primary radioactive contaminants evaluated for the various media at the Weldon Spring site are uranium-238, uranium-235, thorium-232, thorium-230, protactinium-231, actinium-227, radium-228, radium-226, and lead-210. Separate analyses were performed for radon-222 and for radon-220 in the chemical plant buildings. Samples were not analyzed for all of these radionuclides in all media; for example, soil samples were analyzed only for radium-228 of the thorium-232 decay series radionuclides, and samples from all media were analyzed for very few radionuclides in the uranium-235 decay series. To account for the possible presence of radionuclides for which no analysis was performed during the characterization effort, a radiological source term analysis was performed for soil and sludge at the site. The results of this source term analysis, expressed as activity concentration ratios normalized to uranium-238, are presented in Table 2.3; additional information on the data and procedures used to obtain these ratios is provided in a report of MK-Ferguson Company and Jacobs Engineering Group (1992b).

The activity concentration ratios for the raffinate pit sludge, site soil, and quarry soil vary widely (Table 2.3), as expected from the history of plant operations and the various sources of contamination. The raffinate pit sludge resulted from the processing of uranium and thorium ore concentrates. This sludge contains high concentrations of thorium-230, which was present in the ore concentrate delivered to the site for processing, and lesser amounts of other radionuclides. Most of the actinium-227 and protactinium-231 at the site is contained in the raffinate pit sludge. The major radioactive contaminant in site soil is uranium-238, which is consistent with the operations conducted at the chemical plant (uranium processing) and the likely mechanisms of soil contamination (airborne emissions and spills). Some soil in the quarry contains high concentrations of thorium-230, radium-228, radium-226, and lead-210, which is also consistent with the historical use of the quarry for disposal of solid waste largely associated with uranium processing activities at the chemical plant and at the former Destrehan Street plant in downtown St. Louis. These contaminants are typically elevated in waste from uranium processing activities.

Results of the source term analysis were used to estimate radionuclide activity concentrations if isotope-specific data were not available, but they were not used in place of any measured values. Use of this analysis to derive values for radionuclides that might be present on-site represents a conservative approach for purposes of the risk assessment. That is, because the source term analysis was developed from data known to be biased high (e.g., data for the South Dump), applying these data to areas not expected to be similarly contaminated (e.g., nonsource areas of sitewide surface soil) is likely to result in the assumption of artificially elevated radionuclide concentrations in those areas. In fact, this was confirmed by comparing values derived for certain soil locations with other available data such as radionuclide and external gamma measurements for those locations. (In some cases, derived values were more than twice as high as other data indicated.)

2.3.2.2 Chemical Analyses

Environmental samples were analyzed for compounds on the target analyte list and the target compound list, as appropriate to the medium and sampling location. The target analyte list consists of 23 metals and cyanide; the target compound list consists of 34 volatile organic compounds, 65 semivolatile organic compounds, and 27 pesticides and PCBs. The analytes on the two lists are presented in Appendix A. For most analytes, the analytical methods are those specified for routine analytical services under the CLP. Data were generated with the same protocols that are specified for routine analytical services (i.e., analytical parameters, sample handling, quantitation limits, and quality control requirements).

In reviewing the chemicals used during past operations at the Weldon Spring site, additional contaminants that are not present on the target lists were identified as potential site contaminants. These were nitroaromatic compounds associated with the ordnance works operations, certain metals (molybdenum and lithium) associated with the ores and uranium feed materials operations, inorganic anions (fluoride, nitrate, and sulfate) associated with the use of acids during both operations, and asbestos associated with the building construction material.

The TNT and DNT production processes were similar to those used at other plants operated by the Army during the same period. On the basis of discussions with the Army concerning contaminants identified at similar sites, the following nitroaromatic compounds were selected for analysis: 1,3-dinitrobenzene (DNB); 2,4-DNT; 2,6-DNT; nitrobenzene (NB), trinitrobenzene (TNB), and TNT. Samples were analyzed for these contaminants according to quality control procedures developed by the U.S. Army Toxic and Hazardous Materials Agency (USATHAMA). In general, the USATHAMA analyses were conducted at those locations for which information on past usage suggested possible contamination with nitroaromatic compounds.

Three of the nitroaromatic compounds (2,4-DNT, 2,6-DNT, and NB) are on the target compound list, so data were available from both the CLP and USATHAMA analyses for certain soil samples. Although the results were generally in good agreement, the detection frequency was higher for the USATHAMA analyses because of its greater specificity for nitroaromatic compounds. For this reason, the data from the USATHAMA analyses were used for the calculations in this BA. The majority of water samples were collected as part of the ongoing environmental monitoring program. These samples were not routinely analyzed for CLP organic compounds because previous characterization studies indicated that such compounds (except for the nitroaromatic compounds) were not present. Hence, water samples were analyzed according to the USATHAMA methodology for nitroaromatic compounds. Finally, asbestos fiber analyses were included because asbestos-containing material was found in buildings and pipes throughout the chemical plant. Samples of this material were analyzed by phase contrast microscopy and/or transmission electron microscopy.

2.3.3 Quality Assurance/Quality Control

The quality assurance/quality control (QA/QC) requirements for the Weldon Spring site that were applied to the data used in this BA are presented in detail in the *Quality Assurance Program Plan* (QAPP) (MK-Ferguson Company and Jacobs Engineering Group, undated) and the *Remedial Investigation Quality Assurance Program Plan* (RI QAPP) (MK-Ferguson Company 1988b). The QA requirements of the QAPP comply with the applicable requirements of the American National Standards Institute/American Society of Mechanical Engineers Nuclear Quality Assurance for a nuclear facility (ANSI/ASME NQA-1) (ANSI 1986), DOE Order 5700.6B, and DOE-OR Order 5700.6A. The RI QAPP specifically addresses the QA/QC requirements for Superfund sites as outlined in Title 40, Code of Federal Regulations, Part 300.430 (40 CFR 300.430).

The RI QAPP defines goals for the level of QA effort and data requirements in terms of precision, accuracy, representativeness, completeness, and comparability. Specific QC procedures defined in existing documents were incorporated into the plan by reference. The overall QA objective is to develop and ensure implementation of procedures for field sampling, chain of custody, laboratory analysis, and reporting to provide technically sound and legally defensible data.

Environmental media, existing structures, and other material at the site were sampled in accordance with project procedures and sampling plans to ensure that adequate characterization data were acquired in a controlled, systematic, and technically sound manner. Procedural requirements for data collection and analyses and database management were defined in the site's QA/QC program and in the sampling plans prepared to support each investigation.

For the radiological and chemical analyses, field QC samples — i.e., field blanks, trip blanks, duplicate samples (split samples analyzed by the same laboratory), and replicate samples (split samples analyzed by different laboratories) — were collected and processed in a manner identical to that used for the actual characterization samples. The QC samples for chemical analyses were handled in accordance with CLP protocol. Similar protocols were also followed for radiological characterization, including some sample splitting and archiving to permit possible future analysis.

The quality of the laboratory analytical results was evaluated in accordance with the site's data validation program. This program includes routine audits of laboratories; verification of laboratory reporting; technical review of laboratory procedures, results, and calculations; collection, analysis, and evaluation of duplicate and replicate samples; and, if necessary, collection of check samples to confirm previous findings. The results of this effort are described in a data validation report (MK-Ferguson Company and Jacobs Engineering Group 1992a).

2.4 DATA PRESENTATION AND EVALUATION

Radiological and chemical characterization data were evaluated to address possible data gaps and verification and validation issues; the data were then screened for purposes of this risk assessment in accordance with the general approaches recommended by the EPA in the *Human Health Evaluation Manual* (EPA 1989j). The objective of the evaluation was to identify radioactive and chemical contaminants of potential concern at the site for detailed consideration in the risk assessment. The following steps were performed for each of the areas and environmental media sampled:

- Contaminant characterization data were assembled by location and environmental medium.
- Data quality was evaluated with respect to analytical methods, detection limits, QC samples, and blanks.
- Contaminant concentrations were compared with background concentrations, where appropriate.
- Chemical contaminants classified as essential human nutrients were screened on the basis of their concentration and potential toxicity.
- A concentration-toxicity screen was applied to determine whether the potential list of chemical contaminants should be further screened on the basis of the relative contribution of individual contaminants to site risks.

Under CLP protocol, various qualifiers and codes are generally provided with the results reported for chemical analyses. These codes identify specific concerns about data quality and indicate uncertainties about the identity and/or reported concentration of a chemical. Although most site samples were analyzed according to CLP analytical protocols for chemical contaminants, such qualifiers were not systematically provided with the results reported by the laboratories conducting the analyses because the qualifiers had not been requested with the original data. However, some (non-CLP) qualifiers did accompany certain data, and other qualifiers were developed as part of the site's verification/validation program (MK-Ferguson Company and Jacobs Engineering Group 1992a). This information was used for assessing data quality to determine the appropriateness of data for use in this risk assessment.

The EPA has established sample quantitation limits for chemicals that a CLP laboratory must be able to routinely achieve, referred to as the contract-required detection limits. The actual detection limits reported by a laboratory may in fact be lower or higher than the contract-required detection limits. For the site characterization data, the actual detection limits were corrected on an analysis-specific basis to account for dilution; size of the sample aliquots; and — for soil, sediment, and sludge — percent moisture content of the sample. Taking these adjustments into consideration, the quantitation limits met, or were lower than, the contract-required detection limits with few exceptions.

For radionuclides, the detection limit is a function of the background count rate and the counting system. With standard techniques, the detection limit is sufficiently low to ensure the detection of radionuclides at activity concentrations below the levels naturally present in the environment. The EPA has not established a CLP protocol for radiological sampling and analysis programs or a corresponding system of laboratory and data validation qualifiers. Therefore, the procedures used to obtain and report radiological data for the site could not be evaluated against a standard protocol. However, the procedures used at the site are standard industry methods that have been used at many other sites and are generally accepted by professionals in the environmental radiation field. Hence, the quality of the radiological data is considered acceptable for purposes of this risk assessment.

Although limitations in data reporting precluded a complete evaluation of the quality of the characterization data in accordance with EPA guidance, the data for all media sampled are considered, at a minimum, to be equivalent to the Level III analytical level defined by the EPA (1987a). The EPA considers an analytical level of III or higher to be appropriate for use in a risk assessment.

The analyses presented in this document are generally based on data available as of June 1990. Data from the radiological and chemical characterization studies conducted for or by the current Project Management Contractor were entered into a computerized database, which served as the primary source for all data summaries and analyses. Earlier studies to characterize site contamination focused primarily on radioactive contamination. Thus, considerable historical data (not entered into the database) were available for the radiological characterization and risk assessment, and these data were also used, as appropriate, for the analyses in this document. For the chemical characterization and risk assessment, only the more recent data were used.* Some limited data received beyond the indicated date have also been considered, e.g., for cases in which data were not previously available. Such data include measurements for the game ingestion pathway, check samples taken to confirm previous results, and data from the radiological source term analysis for soil and sludge.

The results of the contaminant characterization studies for the site are summarized in Sections 2.4.1 through 2.4.6 according to environmental medium or area. The evaluation criteria specific to each medium or area are discussed, and contaminants of concern are identified using the screening procedure described above. The final list of contaminants of concern is presented in Section 2.5.

---

*The entries in the database (e.g., sample identifiers and concentrations) were verified subsequent to preparation of the individual characterization reports (MK-Ferguson Company and Jacobs Engineering Group 1992a). Because the data had not been verified prior to preparation of the earlier reports, some discrepancies may exist between the values reported in this BA and those previously reported.

2.4.1 Soil

2.4.1.1 Radioactive Contaminants

Several extensive soil investigations have been conducted to characterize the radioactive contamination at the site that resulted from past chemical plant operations (see Section 3.2.2.1 of the RI and Table 2.1). Preliminary surveys confirmed the presence, but not the extent, of radioactive contamination in site soil. A comprehensive sampling program was subsequently conducted to characterize the nature and extent of radioactive contamination within the site boundary.

A sampling program initiated in 1987 included measuring exposure rates at the ground surface and at 1 m (3 ft) above the surface to delineate areas of elevated external gamma radiation (Marutzky et al. 1988). At locations where external gamma exposure rates were elevated above background, uranium-238, radium-226, and radium-228 were measured in situ. Where in-situ measurements were elevated above background, soil samples were collected for laboratory analysis. These samples were taken from more than 300 locations at 15-cm (6-in.) intervals to a depth of 0.5 m (1.5 ft), as indicated by contaminant levels. Where radioactive contamination extended beyond 0.5 m (1.5 ft), subsurface samples were collected at 0.3-m (1-ft) intervals to a minimum depth of 1.5 m (5 ft). These samples were analyzed for total uranium and/or thorium-230; a few samples were also analyzed for radium-226 and radium-228.

A follow-on study was conducted to characterize areas for which additional information was needed; areas where the preliminary data required verification were also resampled in this study, the results of which are reported in the RI. A total of 169 locations were sampled; surface soil was collected at 138 locations, and subsurface soil was collected at the other 31 locations. The majority of boreholes did not exceed a depth of 1.8 m (6 ft); however, in areas where contaminated material was believed to be buried, the boreholes extended to as deep as 3.6 m (12 ft). Soil samples were analyzed for uranium-238, radium-226, and radium-228; however, not all parameters were measured at all locations. A few select samples were analyzed for thorium-230.

Radioactive contamination at vicinity properties has also been characterized in several studies (see Section 3.2.2.2 of the RI and Table 2.1); the locations of these properties are shown in Figure 1.7. Data from the various studies include gamma exposure rates, surface beta-gamma dose rates, and concentrations of specific radionuclides in surface and subsurface soil samples. Four of these vicinity properties have already been remediated as part of previous cleanup actions at the site (A7, B1, B2, and B8); these properties are also shown in Figure 1.7.

External gamma exposure rates have been monitored quarterly since 1982 as part of the environmental monitoring program. Monitoring stations are located along the site perimeter, at the Francis Howell High School, at the guard house of the U.S. Army Reserve Training Area, and at the headquarters of the Busch Wildlife Area. Background gamma exposure rates are measured at four off-site locations near the site but beyond its area of influence. Results from

the environmental monitoring program indicate that gamma exposure rates at the site perimeter and nearby receptor locations are not statistically different from background levels.

Characterization results are presented in Table 2.4. The average measured background concentration of uranium-238, thorium-230, radium-228, and radium-226 is 1.2 pCi/g (Marutzky et al. 1988). The site characterization data indicate that uranium-238 is generally distributed at low levels across surface soil, but a few discrete areas of relatively high concentrations occur at the North Dump, at the South Dump, and around the process buildings. Elevated levels of radium-226 and radium-228 have also been detected in a few scattered areas around the process buildings, and elevated levels of thorium-230 have been detected in scattered locations around the raffinate pits and in the South Dump. Off-site soil contamination is limited to several small, isolated areas, which are referred to as soil vicinity properties. A detailed discussion of radioactive soil contamination is presented in Section 5.2.2 of the RI report.

Overall, evaluation of the available data indicates that the site has been adequately characterized for purposes of this risk assessment. The radiological database for surface soil is extensive. Although soil samples were explicitly analyzed for only four radionuclides, a radiological source term analysis was conducted from which the relative concentrations of radionuclides in the three radioactive decay series at the site could be determined. The results of this analysis were used to derive the concentrations of those radionuclides not directly analyzed (Section 2.3.2.1). One limitation in the data available for soil results from the inability to characterize subsurface radioactive contamination beneath existing features such as the raffinate pits, buildings, process sewers, and paved roads. This data gap will be addressed by sampling the subsurface soil at the affected areas as the confining features are removed. The same contaminants are expected to be found in this follow-on effort that were identified in the extensive characterization studies conducted to date, so the impact on the overall risk assessment is expected to be minor. These areas would be addressed in accordance with the cleanup criteria developed in Chapter 2 of the FS and finalized in the ROD.

The primary radionuclides detected in one or more soil samples, and those for which elevated values were derived from analytical values, were retained as contaminants of concern for this assessment. These contaminants are uranium-238, uranium-235, thorium-232, thorium-230, protactinium-231, actinium-227, radium-228, radium-226, and lead-210.

2.4.1.2 Chemical Contaminants

The site has been extensively sampled to determine the nature and extent of chemical contamination in soil. The three main studies conducted are (1) a Phase I study, which provided preliminary characterization data for the development of a comprehensive soil sampling investigation (MK-Ferguson Company and Jacobs Engineering Group 1988d); (2) a focused study, which investigated soil contamination in areas impacted by interim remedial actions and construction activities (MK-Ferguson Company 1989a); and (3) a Phase II study, which more fully characterized chemical contamination of site soil (MK-Ferguson Company 1989b). The Phase II study, the most comprehensive of the three, included two biased sampling programs and one unbiased program. The biased programs were conducted to investigate areas

potentially contaminated by ordnance works operations and uranium processing operations. The unbiased program systematically investigated all areas of the site. Off-site locations were also sampled to provide background concentrations for metals and inorganic ions. Information for these three studies is presented in Table 2.1.

The results of the three site characterization studies have been combined to provide a comprehensive overview of the nature of soil contamination at the site. The results for metals and inorganic anions are summarized in Table 2.4, and those for organic compounds are summarized in Table 2.5. The background concentrations for metals and inorganic anions are based on soil samples from the Weldon Spring Wildlife Area (Section 2.3.1). Background concentrations for contaminants listed in Table 2.5 are not available because these compounds are not naturally present in soil; hence, background concentrations were assumed to be equal to zero. Although not naturally present in soil, some of the organic compounds detected in site soil (particularly the PAHs, PCBs, and phthalate esters) are widely distributed in the environment as a result of anthropogenic sources. However, the background concentrations of the organic compounds detected at the Weldon Spring site were assumed to be equal to zero for this assessment.

Results from the characterization studies indicate that the main chemical contaminants in site soil are metals and inorganic anions. Nitroaromatic compounds are present in soil at discrete areas associated with former ordnance works operations, and low levels of PAHs are present in an area previously used for coal storage and at a concrete pad adjacent to two chemical plant buildings. Areas adjacent to transformers and around the chemical plant buildings are contaminated with low levels of PCBs. Although asbestos-containing material is present throughout the chemical plant, e.g., in building material and on overhead piping, asbestos fibers were not detected in surface or subsurface soil. The soil chemical characterization results are discussed in detail in Section 5.2.1 of the RI report, including figures that show contaminated areas and contaminant locations.

The data available for chemical contamination in site soil are adequate for purposes of this risk assessment. Data limitations for chemical contaminants are the same as those described for radioactive contaminants.

Data evaluation considerations specific to soil include sample detection limits and the presence of contaminants in field and/or laboratory blanks. A review of the analytical results indicates that the sample quantitation limits are acceptable for all compounds. This conclusion is based on a preliminary screen of health risks for compounds assumed to be present at their detection limits. That is, if a contaminant were present in soil at or below its detection limit (in which case it would be present but reported as not detected), no significant health effects would be expected for the receptors considered in this BA.

Blank samples* provide a means of assessing contamination that has been introduced in the field or laboratory. Consistent with EPA guidance, the volatile and semivolatile organic compounds detected in blank samples can be classified as either (1) common laboratory contaminants (i.e., acetone, methylene chloride, toluene, and the phthalate esters) or (2) contaminants that are not common laboratory contaminants (i.e., all other compounds on the target compound list). In accordance with EPA guidance, common laboratory contaminants were considered to be present in samples if their concentrations exceeded 10 times the maximum amount detected in any blank; all other contaminants were considered present only if their concentrations in the samples exceeded 5 times the amount detected in the blanks (EPA 1989j). A review of the data for volatile and semivolatile organic compounds indicates that acetone and methylene chloride can be deleted as contaminants of concern on the basis of these criteria. However, only limited data are available relative to the concentrations of these compounds in blank samples; thus, prior to their deletion as contaminants of concern, they were also evaluated for their potential contribution to site risks through a concentration-toxicity screen in the final step of the contaminant screening procedure.

For the screening analysis of metals in soil samples, the concentrations were first compared with local background concentrations. For discussion purposes, the upper background limit or upper bound concentration — i.e., the concentration equal to the arithmetic mean plus two standard deviations — was used to provide a reference level to which the soil data could be compared (Table 2.4). For all metals and inorganic anions, the maximum concentration in on-site soil samples is higher than the upper bound. Mean concentrations of contaminants were not compared with background levels because of the possibility of masking contamination by averaging over a large area and large number of samples. Therefore, no contaminants were screened from the analysis on the basis of a comparison with local background concentrations.

Several of the contaminants present at elevated concentrations are essential human nutrients — including calcium, chloride, iron, magnesium, potassium, and sodium. Potential intakes of these elements resulting from ingestion of soil were estimated and compared to the recommended dietary allowances (RDAs) established by the National Research Council (1989). Intakes were estimated for young children (the most sensitive receptors) and were based on the maximum concentrations detected in surface soil and an assumed soil ingestion rate of 200 mg/d. For calcium, chloride, magnesium, potassium, and sodium, the estimated intakes are less than 6% of the RDAs for children who are 1 to 6 years of age. The estimated intake of 17 mg/d for iron is higher than the RDA of 10 mg/d, but it is considerably lower than the dose of 500 mg/d associated with iron intoxication (Friberg et al. 1986) or doses of 50 to 100 mg/d that result in accumulation of iron in body tissues; it is also well within the range of normal daily intakes of 9 to 35 mg. Oral absorption of iron is homeostatically regulated, for which the major controlling factor appears to be the adequacy of iron stores in the body (Friberg et al. 1986). For magnesium, the estimated intake of 40 mg/d is only about half the RDA of

*Blank samples are analytical QC samples that are handled and analyzed with the same procedures as site samples. Blanks are used to measure contamination introduced into a sample either (1) in the field while the samples are being collected and/or transported or (2) in the laboratory during sample preparation or analysis.

90 mg/d for children and considerably below therapeutic doses (magnesium salts are used as cathartics, anticonvulsants, and antacids). For example, the dose to a child from ingesting the recommended one-third to one teaspoon of milk of magnesia (magnesium hydroxide) four times daily would be about 300 to 900 mg/d. These comparisons indicate that the potential health impacts associated with calcium, chloride, iron, magnesium, potassium, and sodium would be insignificant, so they were screened from further consideration in this BA.

Cobalt, copper, manganese, molybdenum, nickel, selenium, and zinc are also essential human nutrients. Except for cobalt, EPA toxicity values are available for these elements, and they were included in the concentration-toxicity screen in the final step of the contaminant screening procedure. Cobalt was also retained through this step of the screening.

Aluminum and sulfate were screened from further consideration as potential contaminants of concern. They are both present in plant and animal food products and exhibit very low human toxicity. Concentrations of aluminum at the site are within the range commonly found in soil. The human intake of aluminum from food, water, and atmospheric sources averages 3 to 160 mg/d, and the intake from antacids containing aluminum ranges from 200 to 3,000 mg/d (Friberg et al. 1986; Seiler and Sigel 1988). The estimated intake of aluminum from site soil, 32 mg/d, is considerably below these levels. The major observed health effect of sulfate is its laxative action. Such effects would not be observed as a result of the estimated intake of sulfate from site soil, 0.6 mg/d. No other contaminants were eliminated at this stage of the screening procedure.

As the next step of the screening procedure, a concentration-toxicity screen was applied to determine which of the remaining 71 contaminants had the potential to contribute significantly to site risks; this concentration-toxicity screen is based on methodology developed by the EPA (1989j). A chemical score for soil was calculated with the following algorithm:

$$R = \Sigma (C_i \times T_i) \tag{2.1}$$

where:

R = total risk factor for soil;

C_i = concentration of contaminant i in soil; and

T_i = toxicity value for contaminant i (1/reference dose for noncarcinogens or the slope factor for carcinogens).

Any set of units can be used for these variables as long as they are consistent for all chemicals. The total risk factor for soil was calculated from the maximum contaminant concentrations shown in Tables 2.4 and 2.5 and EPA toxicity values, i.e., reference doses and slope factors (EPA 1991a, 1991c). The derivation and use of reference doses and slope factors are described in Chapter 4. This algorithm was used with oral and inhalation reference doses or slope factors to calculate separate risk factors for carcinogenic and noncarcinogenic effects, resulting in a total of four separate screens.

The concentration-toxicity screen was applied to 66 of the 71 remaining compounds (all those for which a toxicity value was available). In general, those contaminants contributing less than 1% to the route-specific risk factors for soil were deleted as potential contaminants of concern. However, additional factors were also taken into consideration before deleting a contaminant, including background concentrations (where applicable), detection frequency, and presence in other environmental media at the site.

Although benzidine accounts for more than 1% of the total risk factor for soil (based on its carcinogenicity via the oral route), it was not retained as a contaminant of concern because of three factors. First, it was reported for only one sample, i.e., the detection frequency was very low. Second, it was reported only as a tentatively identified compound whose presence was not confirmed. Finally, benzidine is not expected to be present from knowledge of site history. Although screened from further consideration in this BA, additional samples will be taken at the same location to confirm that this compound is not present.

A few contaminants — cobalt, lead, zirconium, chloroethane, and dibenzofuran — could not be included in the concentration-toxicity screen because EPA toxicity values are not available. Cobalt and lead were retained as contaminants of concern because they are present in more than 80% of the samples tested and some concentrations were elevated above background. In contrast, chloroethane and dibenzofuran were detected in only one and two samples, respectively, and were therefore screened on the basis of very low detection frequency. Similarly, a measured value for zirconium was reported for only one sample. Because zirconium is not on the target analyte list, it was not routinely analyzed. Also, site history indicates that zirconium would not be present at elevated concentrations.

In summary, the following contaminants were deleted on the basis of the above considerations: (1) all volatile organic compounds, (2) all semivolatile organic compounds except the PAHs, and (3) all pesticides. Except for the compounds considered to be common field or laboratory contaminants (i.e., acetone, methylene chloride, toluene, and the phthalate esters), these compounds were detected in less than 3% of the samples analyzed.

For presentational purposes, the individual PCBs detected at the site — Aroclor 1248, Aroclor 1254, and Aroclor 1260 — were considered together because their respective toxicity values for the human health assessment are identical. Also, chromium was assumed to be present in both the trivalent (chromium III) and hexavalent (chromium VI) forms. On the basis of site history, the few cation-specific measurements that are available, and data from the literature, the predominant form of this metal (90%) was assumed to be chromium III. This distinction is significant because the trivalent and hexavalent forms differ considerably with respect to their toxicological properties (see Section 4.4.2.6). The uncertainty that this assumption introduces into the analysis is discussed in Section 5.6.

The final list of contaminants of concern for soil consists of 46 metals, inorganic anions, PAHs, PCBs, and nitroaromatic compounds. These compounds are listed in Section 2.5, following the evaluation of contaminants of concern for the remaining media at the site. The contaminant screening process for soil is summarized in Table 2.6, which lists all compounds deleted as contaminants of concern together with a brief description of the screening rationale.

2.4.2 Surface Water, Sediment, and Sludge

2.4.2.1 Radioactive Contaminants

Several investigations have evaluated the nature and extent of radioactive contamination in surface water and in associated sludge or sediment. Surface water has been sampled at six on-site impoundments — the four raffinate pits, Frog Pond, and Ash Pond — and at five National Pollutant Discharge Elimination System (NPDES) outfalls at or near the site perimeter. Off-site surface water bodies and drainages potentially impacted by releases from the site have also been sampled, i.e., Lakes 34, 35, and 36 and Burgermeister Spring in the Busch Wildlife Area; and the Southeast Drainage and other drainages nearby.

Characterization results for the raffinate pit surface water and the NPDES outfalls (excluding the two outfalls within the Southeast Drainage) are summarized in Tables 2.7 and 2.8, respectively. Characterization results for raffinate pit sludge are also presented in Table 2.7. Uranium and thorium-230 are the major contaminants in surface water at the raffinate pits, and uranium is the primary contaminant in surface water at the NPDES outfalls. The concentrations of uranium are elevated compared with the other radionuclides because of uranium's higher solubility; the elevated concentration of thorium-230 in the raffinate pit water is related to the very high concentration of this radionuclide in the raffinate pit sludge (Table 2.7). Characterization results for Frog Pond and Ash Pond are presented in Section 5.3 of the RI; contaminant concentrations in these two water bodies were considerably lower than those in the raffinate pits.

Results for surface water and sediment (including shoreline soil) in the Busch Wildlife Area and the Southeast Drainage are summarized in Tables 2.9 through 2.13. Uranium is the major radioactive contaminant in surface water at these locations, in sediment at Burgermeister Spring, and in sediment and shoreline soil at the Busch Wildlife Area lakes. The limited data available for sediment in the Southeast Drainage indicate that thorium-230 concentrations are elevated at several locations and that concentrations of uranium and radium are lower but still elevated. Results for other drainages are not presented because contaminant levels were either considerably lower than the levels for which data are presented or at background concentrations. Data for these drainage areas are presented in the *Phase I Spring and Seep Report* (MK-Ferguson Company and Jacobs Engineering Group 1989c).

The characterization data indicate that concentrations of radionuclides in surface water both on-site and at certain nearby locations — Lakes 34, 35, and 36 in the Busch Wildlife Area and the Southeast Drainage in the Weldon Spring Wildlife Area — are elevated above background levels (as measured in Lake 10 and Dardenne Creek). Uranium is generally the major radioactive contaminant. Similarly, uranium concentrations in the sediment of certain off-site lakes are elevated above background (as measured in Lake 37).

Except for the Southeast Drainage, available data are generally considered adequate for the purposes of a risk assessment. Because of data limitations and expectations of changing conditions, the drainage is outside the scope of the current stage of site remediation and

documentation. Nevertheless, the data currently available for this location have been used to develop preliminary risk estimates to focus upcoming characterization efforts (Section 1.1.2). Only a limited number of samples were collected along the 2.4-km (1.5-mi) Southeast Drainage, and only five surface samples were analyzed for thorium-230. All of the analytical results are biased high because the samples were taken from areas of elevated radioactivity, which were identified during an initial gamma walkover scan (Boerner 1986; Deming 1986). Therefore, the related risk estimates are very conservative. These preliminary estimates will be revised in upcoming documentation within the next several years, after additional information becomes available for the drainage, to support final cleanup decisions for the project (Section 1.1.2).

Compared with the data limitations for the Southeast Drainage, those for the raffinate pit sludge and site soil are much less significant, primarily because the radiological source term analysis (Table 2.3) permits consideration of all radionuclides in the uranium-238, thorium-232, and uranium-235 decay series that might be present at the site. One minor data limitation exists for the raffinate pit sludge because, although analytical procedures for soil were adapted to permit sludge analysis, interference by thorium-230 precluded the determination of thorium-232 in pits 1, 2, and 3. However, this data gap is insignificant relative to the size of the database for the pits, and it does not appreciably affect the characterization results for all sludge.

The primary radionuclides detected in one or more samples of surface water, sediment, or sludge — and those for which values were derived from measured values — were retained as contaminants of concern for this assessment. These radionuclides are uranium-238, uranium-235, thorium-232, thorium-230, protactinium-231, actinium-227, radium-228, radium-226, and lead-210.

2.4.2.2 Chemical Contaminants

The nature and extent of chemical contamination in surface water and in associated sludge or sediment have been evaluated in conjunction with the radiological studies (Section 2.4.2.1 and Table 2.1). Chemical characterization results are summarized together with the radiological results in Tables 2.7 through 2.13. Data limitations for chemical contaminants in the Southeast Drainage are the same as those described for radioactive contaminants.

Results from the chemical characterization studies indicate that the raffinate pits are the most highly contaminated of the on-site surface water impoundments. The main contaminants detected in surface water and sludge were inorganic anions and certain metals. The principal anions detected were nitrate, sulfate, and fluoride. Metals present in sludge at high concentrations (relative to background concentrations in site soil) included arsenic, calcium, lead, magnesium, nickel, and vanadium. Low concentrations of several volatile and semivolatile organic compounds were detected in a small number of sludge samples; in general, the presence of these compounds is believed to have resulted from laboratory contamination of samples because similar concentrations of contaminants (e.g., solvents and phthalate esters) were present in the laboratory blanks. Certain sludge samples also contained PCBs and the pesticides dichlorodiphenyldichloroethylene (DDE) and β-hexachlorocyclohexane (beta-BHC). Although concentrations of benzene ranging from 6 to 88 μg/kg were initially reported for over half the

sludge samples from raffinate pits 1 and 2, this compound was not detected at all in follow-on check samples (MK-Ferguson Company and Jacobs Engineering Group 1992a). All samples analyzed had levels of oil and grease of 0.12 to 0.94%.

Contaminant concentrations in off-site surface water and sediment were generally less than the detection limits or did not appear to be above typical levels. Nitroaromatic compounds were detected infrequently in surface water, mainly in the Southeast Drainage and Burgermeister Spring.

Similar to radioactive contamination, the data for chemical contamination are generally considered adequate for purposes of this risk assessment. Data evaluation considerations specific to surface water, sediment, and sludge include (1) sample quantitation limits for these media, (2) limitations in the available surface water data, (3) the presence of contaminants in field and/or laboratory blanks, and (4) limitations in data available to determine background concentrations. The main limitation in the available data was the number of samples collected to characterize contaminant levels in the Busch Wildlife Area. Only one to three samples were taken at Dardenne Creek and Lake 10 (background locations) and at Lakes 34, 35, and 36. To supplement these data, results from the nearest sampling locations upstream of Lakes 34 and 35 were also used to represent the concentrations in the lakes. This constitutes a conservative approach because contaminant concentrations upstream of the lakes should be higher than the lake concentrations as a result of mixing and dilution in the lakes. Nearby upstream samples were not available for the other lakes.

For surface water, background concentrations were not compared with characterization data as part of the contaminant selection process because of the very limited data from background locations, both in terms of the number of locations sampled and the number of sampling events. In general, compounds detected in one or more surface water, sediment, or sludge samples were retained as contaminants of concern; the exceptions were the essential nutrients and other food components previously deleted for soil (aluminum, calcium, iron, magnesium, potassium, sodium, chloride, and sulfate). In addition, the phthalates, the volatile organic compounds, and the pesticides DDE and beta-BHC — all of which were detected only in raffinate pit sludge — were deleted on the basis of a concentration-toxicity screen for this medium. The compounds retained included metals, inorganic anions, PCBs, and nitroaromatic compounds (see Section 2.5).

2.4.3 Groundwater

Groundwater is outside the scope of the current stage of assessment and remediation because of data limitations. Nevertheless, preliminary risk estimates have been developed to focus upcoming sampling efforts (Section 1.1.2). Characterization results for groundwater are discussed in Appendix B, and the preliminary risk estimates are presented in Appendix E of the FS. These preliminary estimates will be revised in documentation to be prepared within the next several years, after additional information becomes available, to support final cleanup decisions for the project. Summary data for the groundwater sampling activities to date and general

information for radioactive and chemical contaminants are presented in Sections 2.4.3.1 and 2.4.3.2.

2.4.3.1 Radioactive Contaminants

Groundwater has been sampled on-site and at several off-site locations near the site boundary to assess the nature and extent of radioactive contamination. The two main studies conducted were (1) a Phase I study, which provided information on the quality of groundwater and supported site characterization activities (MK-Ferguson Company 1987) and (2) a Phase II study, which assessed the horizontal and vertical extent of groundwater contamination (MK-Ferguson Company and Jacobs Engineering Group 1989d). In addition to these studies, groundwater has been routinely monitored for total uranium since 1987 as part of the environmental monitoring program (MK-Ferguson Company and Jacobs Engineering Group 1988a, 1989a, 1990a, 1991). Groundwater has also been sampled for radium-226, thorium-232, thorium-230, gross alpha, and gross beta during this time. Currently, the groundwater monitoring network consists of 59 wells within and around the site boundary. The wells are completed in three zones within the upper Burlington-Keokuk Limestone: 13 wells in the weathered limestone, 20 wells in the competent (unweathered) limestone, and 26 wells across the weathered/competent contact. No on-site wells have been completed in the St. Peter Sandstone.

Results of the Phase I and Phase II studies and the environmental monitoring program indicate that uranium is the only radioactive contaminant present in groundwater at concentrations elevated above background levels; these results are summarized in Table 2.14. Elevated concentrations of uranium are present in both the weathered and competent zones of the limestone formation, but the vertical extent of contamination has not yet been fully characterized. Although a representative background activity has not been measured at an off-site location as part of site characterization, data are available from the USGS for nearby locations. These data indicate that background levels of total uranium range from 0.5 to 2.9 pCi/L (MK-Ferguson Company and Jacobs Engineering Group 1989d).

2.4.3.2 Chemical Contaminants

Chemical contamination in groundwater has been investigated in the same manner as radioactive contamination (Section 2.4.3.1 and Table 2.1). Contaminants evaluated in the Phase I study included metals, inorganic anions, volatile and semivolatile organic compounds, PCBs, pesticides, and nitroaromatic compounds. On the basis of the Phase I results, the Phase II study was limited to an analysis of metals, inorganic anions, and nitroaromatic compounds. As part of the environmental monitoring program, groundwater has also been monitored since 1987 for inorganic anions and nitroaromatic compounds.

The combined results of the Phase I, Phase II, and environmental monitoring studies provide an overview of the nature of groundwater contamination as currently understood at the site; these results are summarized in Table 2.14. The background concentrations presented in

the table are based on data from the USGS (MK-Ferguson Company and Jacobs Engineering Group 1989d; Kleeschulte and Cross 1990). Background concentrations are not listed for the organic contaminants because these compounds are not naturally present in groundwater; hence, the background concentration is assumed to be equal to zero.

Results from the characterization studies indicate that the main chemical contaminants in groundwater are inorganic anions, nitroaromatic compounds, and certain metals. Some chemical classes — volatile organic compounds, semivolatile organic compounds (PAHs), PCBs, and pesticides — were not detected in groundwater and are therefore not listed in Table 2.14.

2.4.4 Air

2.4.4.1 Radioactive Contaminants

Radioactive contamination in air was assessed with data collected through the ongoing environmental monitoring program (MK-Ferguson Company and Jacobs Engineering Group 1988a, 1989a, 1990a, 1991). An air monitoring program was established in 1981 to measure radon gas, and monitoring for airborne radioactive particulates was initiated in 1987. Monitors are located around the site perimeter to ensure that airborne contaminants would be detected if released from the site; additional detectors are in place at the Francis Howell High School, the guard house at the U.S. Army Reserve Training Area, and the headquarters at the Busch Wildlife Area. In 1990, quarterly radon concentrations at these locations ranged from 0.1 to 0.7 pCi/L, as did background radon concentrations. Historically, radon concentrations at the various monitoring locations have not been statistically different from background levels. Similarly, the concentrations of airborne radioactive particulates have not differed statistically from background levels.

2.4.4.2 Chemical Contaminants

Air has been monitored for asbestos at the site perimeter and at outdoor locations at Francis Howell High School since 1988 as part of the environmental monitoring program; it has also been monitored at other times during asbestos abatement operations. Detailed data on these monitoring efforts, including locations and results, are presented in the annual environmental monitoring reports (MK-Ferguson Company and Jacobs Engineering Group 1989a, 1990a, 1991, 1992c) and in the RI report. Filters from the air monitors were analyzed by phase contrast microscopy (PCM), with results reported as fibers per cubic centimeter of air. Selected samples were also analyzed by transmission electron microscopy (TEM), with results reported as structures per cubic centimeter of air. Analyses by PCM provide total airborne concentrations of fibers having a size and shape characteristic of asbestos fibers but are not specific to asbestos; analyses by TEM provide actual asbestos fiber concentrations.

During 1988 to 1991, 226 asbestos samples were collected at the site perimeter and 274 samples at the high school. The PCM measurements at the site perimeter ranged from less

than detectable concentrations (i.e., <0.0001 to <0.001 fibers/cm^3) to 0.038 fibers/cm^3. Concentrations at the high school ranged from less than detectable concentrations (i.e., <0.0001 to <0.001 fibers/cm^3) to 0.017 fibers/cm^3. Three samples from the site perimeter and 15 samples from the high school were also analyzed by TEM. Measurements at the site perimeter were all less than detectable concentrations (i.e., <0.001 to <0.005 structures/cm^3), and those at the high school ranged from less than detectable concentrations (i.e., <0.001 to <0.002 structures/cm^3) to 0.060 structures/cm^3. (As discussed in the 1989 environmental monitoring report, the high maximum concentration measured at the high school probably resulted from sample treatment prior to analysis.) The only data evaluation consideration specific to these measurements is associated with the analytical method for asbestos fiber determinations. Because nonasbestos particles are included in the PCM analyses, the reported results generally overestimate the actual airborne asbestos concentrations (see Section 5.6 of the RI report).

The primary purpose of the asbestos monitoring effort was to establish baseline levels for comparison with levels during asbestos abatement activities. Although these baseline measurements showed no significant trends over the monitoring period, true background asbestos levels are not available for comparison. Most of the friable asbestos on exterior structures at the site has been removed and is currently stored in double bags within an on-site building, but asbestos-containing material is found inside the buildings and in exterior siding. Therefore, asbestos is retained as a contaminant of concern for the buildings.

Air particulate monitors have been used to detect radioactive particulates rather than total suspended particulates (TSP). Hence, data are not available for ambient particulate or nonradioactive contaminant concentrations. However, TSP have been measured at other locations in Missouri, and air concentrations were derived from these data for the analysis in this BA (see Section 3.3.4).

2.4.5 Biota

2.4.5.1 Radioactive Contaminants

Biouptake studies have been conducted both on-site and off-site to assess the potential contribution of fish and game ingestion to human exposure. The earliest study sampled terrestrial vegetation, aquatic vegetation, invertebrates, and fish (Ryckman/Edgerley/Tomlinson & Associates [RETA] 1977). Samples were analyzed for total uranium, uranium-238, uranium-235, thorium-232, thorium-230, and radium-226. Low levels of contaminants were detected in many of the organisms examined, but little evidence existed of biomagnification in upper trophic levels. Subsequent fish uptake studies by the Missouri Department of Conservation in Lakes 34, 35, and 36 identified low levels of radionuclides in tissue samples (deRoos 1984). The results of these early studies are discussed in greater detail in the RI report (DOE 1992c).

The Project Management Contractor also studied fish and small mammals. Fish were collected in off-site lakes, in cooperation with the state of Missouri (MK-Ferguson Company and

Jacobs Engineering Group 1989e), and composite samples were analyzed for total uranium, thorium-232, thorium-230, and radium-226. Several small mammals were collected from the immediate area of the chemical plant, and composite samples were split into bone and tissue and analyzed separately for the same parameters as the fish samples. No radioactive contaminants were detected above the detection limits. However, the statistical significance of the radiological study results is weak because of the small sample size and the compositing necessary to achieve adequate tissue volumes for analysis. More recent fish studies, which used lower detection limits, did indicate some uranium uptake; the results of these studies are shown in Table 2.15.

One adult deer, collected from the material staging area (MSA), was also analyzed for radionuclides. Samples were split into bone, internal organs, and flesh and analyzed for uranium-238, uranium-235, uranium-234, thorium-228, thorium-230, thorium-232, radium-228, and radium-226. Of the two flesh samples analyzed, concentrations of uranium-235, thorium-228, thorium-232, and radium-228 were consistently less than the detection limit; concentrations of uranium-238, uranium-234, and radium-226 were less than the detection limit in one of two samples; and thorium-230 was detected at above the detection limit in both samples. The following concentrations were determined for flesh samples in which the analyte was detected (wet weights): 0.002 pCi/g uranium-238 and uranium-234, 0.003 pCi/g thorium-230, and 0.001 pCi/g radium-226. Similar to the small mammal studies, the statistical significance of these data is weak because of the small sample size and the lack of information regarding the amount of time the animal spent on-site. In addition, no control sample is available for comparison.

Similarly, recent analyses of waterfowl collected from the raffinate pits indicated some elevated concentrations of radionuclides. Samples of flesh from three wood ducks, one mallard, and one goose were analyzed for uranium-238, uranium-235, uranium-234, thorium-232, thorium-230, thorium-228, thorium-227, radium-228, and radium-226. The following average concentrations were determined for those waterfowl in which the analyte was detected (wet weights): 0.004 pCi/g uranium-238; 0.016 pCi/g thorium-230, with two samples less than the detection limit; 0.27 pCi/g thorium-227; 0.027 pCi/g radium-228, with three samples less than the detection limit; and 0.002 pCi/g radium-226, with three samples less than the detection limit. Standard deviations associated with these measurements are 0.002 pCi/g for uranium-238, 0.020 pCi/g for thorium-230, 0.313 pCi/g for thorium-227; 0.013 pCi/g for radium-228; and 0.0007 pCi/g for radium-226. Because of the limited data collected, the standard deviations are high. In addition, no samples were taken to establish background values.

Data evaluation considerations specific to these studies include (1) sample size and representativeness, (2) detection limits, and (3) reporting. The first consideration reflects the difficulties inherent in conducting comprehensive biotic studies. For example, although a study design may identify a specific number of individuals of different species that should be collected to provide an adequate sample size, it is often difficult to collect the desired number and type of samples. Comparing the results from the different studies is difficult not only because species, tissue types, and sample sizes differed but because detection limits and reported values (e.g., on a wet-weight versus dry-weight basis) were inconsistent. Therefore, the results of the studies were considered separately.

Results of the biotic studies conducted to date were considered for the fish and game ingestion pathway analyses in this BA. In addition, these data are being used to focus upcoming biotic studies planned for the site. For this pathway, the contaminants of concern are those associated with the environmental media that contribute to biotic uptake, i.e., surface water and site soil. Hence, the contaminants of concern for the biotic pathway are those identified in Sections 2.4.1 and 2.4.2.

2.4.5.2 Chemical Contaminants

The uptake of chemical contaminants by fish was evaluated in conjunction with the radiological uptake studies. In studies conducted by the state of Missouri, chlordane, arsenic, cadmium, lead, and mercury were detected in fish flesh from Lake 35 and/or Lake 36 and all but chlordane were detected in samples from Lake 37 (the control location for the site). In subsequent studies by the Project Management Contractor, composite samples were analyzed for PCBs and metals (not all samples were analyzed for each parameter), but PCBs were not detected above the detection limit and levels of metals were not elevated; the results of these biotic studies are discussed in greater detail in Section 5.5.5 of the RI report.

Data evaluation considerations for the chemical contaminants include those identified for the radioactive contaminants. In addition, the analytes were limited and, in some cases, little or no information was reported for levels in the media to which the fish were exposed. Hence, potential health impacts from chemical exposure via fish ingestion are difficult to assess. Nevertheless, to address this potential impact, the Missouri Department of Health and the Agency for Toxic Substances and Disease Registry (ATSDR) have prepared health assessments for fish consumption from the lakes on the basis of study results.

The ATSDR assessment addressed potential exposure to lead, arsenic, and mercury via fish ingestion from Lakes 34, 35, 36, and 37 (Mellard 1989). An intake rate of 280 g/wk was assumed, which represents an average daily rate of about 40 g; for comparison, this rate was considered appropriate for persons fishing for either sport or subsistence and is about twice that estimated for a typical American. The ATSDR study considered the following data: concentrations of lead in fish composites taken from Lakes 34 and 36 were 4.0 and 4.2 μg/g; concentrations of arsenic in fish composites taken from Lakes 35 and 37 were 13.9 and 13.6 μg/g; and concentrations of mercury in fish composites from Lakes 34, 35, 36, and 37 were 0.27, 0.24, 0.16, and 0.23 μg/g, respectively. As indicated by these values, the levels of metals in fish from Lake 37, which is not impacted by the Weldon Spring site, are similar to those in fish from the other lakes. Recognizing that the drainage areas for these lakes are extensive and that the concentrations of these metals in site soil are generally within the range of local background levels, it is reasonable to consider that other factors contribute to the elevated levels of metals and other nonradioactive constituents in these lakes (e.g., the natural occurrence of these elements in soil and the local use of pesticides, herbicides, and fertilizers).

The ATSDR assessment concluded that the estimated intake level of mercury is below levels that may cause adverse health effects, whereas that of arsenic exceeds the intake level established on the basis of drinking water ingestion. For lead, the estimated intake level has the

potential for increasing blood lead levels in regularly exposed populations above those levels that could result from other dietary intake. This was noted by comparing estimates from fish ingestion with estimates from drinking water ingestion, conservatively assuming the same absorption efficiency. The overall conclusion of the ATSDR study was that the heavy metals in the composite fish samples do not present a public health threat to area residents who have occasional fish meals from locally caught fish, although sport and subsistence fishermen may be at increased risk to adverse health effects if a major portion of their diet came from the indicated lakes (Mellard 1989).

The separate assessment by the Missouri Department of Health addressed consumption of fish from Lakes 35, 36, and 37 (Schmidt 1990). The contaminants measured in fish from these lakes included arsenic, cadmium, lead, and mercury. This assessment incorporated data for use of the lakes and assumed that a person fishing for subsistence would generally concentrate on catfish and, at the study lakes, would (1) fish the lake with the highest contaminant levels in such fish (Lake 36) at twice the average rate observed, (2) succeed in catching the legal limit, and (3) consume all of the daily catch. The assessment concluded that consuming fish from the lakes does not present a significant increased risk of average health effects from the contaminants evaluated (see also Section 3.4.2.5).

From the considerations discussed in this section, it was determined that the potential is low for significant human exposure to chemicals via the fish ingestion pathway. Hence, this potential exposure was not quantified further in this BA.

2.4.6 Building Air and Residues

2.4.6.1 Radioactive Contaminants

The chemical plant consists of about 40 buildings and miscellaneous structures. All buildings have been surveyed to varying degrees for radioactive contamination. Preliminary studies by the U.S. Department of the Army and by RETA were conducted soon after chemical plant closure to determine the general nature of contamination and the decontamination effort necessary to release buildings for use without radiological restrictions (RETA 1978). A more comprehensive study of buildings in which uranium and thorium were processed was conducted in 1986 by Bechtel National (1986). Further characterization studies were conducted by the Project Management Contractor in 1988 and 1989 (MK-Ferguson Company and Jacobs Engineering Group 1990c).

The results of these studies indicate that the process buildings, including equipment inside these buildings, are generally contaminated to levels exceeding DOE surface contamination guidelines for use without radiological restrictions (DOE Order 5400.5; see Appendix G of the FS). The primary radioactive contaminants in most of these buildings are uranium-238 and thorium-232. Average concentrations of these radionuclides in bulk samples collected from the buildings ranged from <1 to 12,000 pCi/g; the maximum value of 12,000 pCi/g for uranium-238 occurred in Building 403. Significant levels of radon-220 have also been measured

in three process buildings (Buildings 106, 108, and 403). The maximum concentration measured for radon-220 decay products was 2.5 WL in Building 403; the average concentration of radon-220 in this building was 0.61 WL.

The nonprocess buildings are also contaminated to varying degrees as a result of past storage activities and tracking from the process buildings. The primary radioactive contaminants in most of these buildings are also uranium-238 and thorium-232. A detailed description of contamination in individual buildings is provided in the *Buildings Radiological Characterization Report* (MK-Ferguson Company and Jacobs Engineering Group 1990c). Additional information regarding the radiological characterization results for these buildings is provided in the EE/CA for the chemical plant structures (Peterson and MacDonell 1991).

The quality of radiological data available for the buildings was considered adequate for purposes of this risk assessment because the decision to dismantle these buildings has already been made (Section 1.1.2). Hence, available data are adequate for the related screening-level assessment.

2.4.6.2 Chemical Contaminants

During a preliminary survey in 1986, the chemical plant buildings were sampled to identify asbestos-containing material (MK-Ferguson Company and Jacobs Engineering Group 1986; metaTRACE 1987). Results of the survey indicate that this material is present as corrugated siding and in pipe, steam valve, and heating duct insulation. The asbestos content ranged from 15 to 75%, depending on the material. Visual inspection of building interiors indicated that a variety of other probable asbestos-containing material was present.

In subsequent chemical characterization studies, 27 nonprocess buildings were sampled and surveyed primarily for asbestos-containing material and PCBs (MK-Ferguson Company and Jacobs Engineering Group 1990b). Because the preliminary survey showed that the majority of pipe, boiler, and vessel insulation contained asbestos, only insulating and building materials with unknown asbestos content were sampled. Additional contaminated material was identified in that study, with the asbestos content ranging from about 5 to 75%.

The buildings were sampled for PCBs from 1987 through early 1989 by either swipe sampling of surfaces or bulk sampling of residues, dust, and ballast from electrical lighting fixtures. Results indicated that surface concentrations ranged from about 2 to more than 29,000 $\mu g/100\ cm^2$. However, almost 40% of the swipe samples were below the detection limit of 1 $\mu g/100\ cm^2$. Bulk sample concentrations ranged from about 40 to 13,000 mg/kg, with 15% below the detection limits of 2 to 5 mg/kg.

Data characterizing other chemical contamination in the buildings are very limited. In one study, no combustible gases were detected in a survey of a few locations in Buildings 201, 301, 403, and 404 (Bechtel National 1986).

The quality of data available for chemical contamination in the buildings was considered adequate for purposes of this risk assessment. Data evaluation considerations specific to the buildings and structures include the analytical method used for asbestos fiber determination (Section 2.4.4.2) and the limitations in available chemical characterization data. Although a comprehensive survey of the site was completed for asbestos-containing material, nonprocess buildings were characterized only for possible PCB contamination. Nevertheless, these data are believed to be adequate for the related risk assessment because the decision to dismantle these buildings has already been made (Section 1.1.2).

2.5 FINAL LIST OF CONTAMINANTS OF CONCERN

Radiological investigations at the chemical plant area and associated vicinity properties indicate that the radioactive contaminants of concern are those associated with the uranium-238, thorium-232, and uranium-235 decay series. The risks associated with the radionuclides in these series can be determined from measured and derived concentrations of uranium-238, uranium-235, thorium-232, thorium-230, protactinium-231, actinium-227, radium-228, radium-226, lead-210, radon-222, and radon-220. Hence, these represent the radioactive contaminants of concern. The radiological effects associated with the other radionuclides in these three decay series are accounted for in the dose conversion factors used for the listed radionuclides. The risks associated with external gamma exposure were also considered in the health risk assessment.

The chemical contaminants of concern were selected for each medium in a stepwise process. For soil, a preliminary list was developed comprising all contaminants detected in one or more samples. Contaminants were then screened from this list by evaluating the soil characterization data with respect to background concentrations, data quality, detection frequencies, and contribution to overall risks at the site. A more limited screening process was applied to the other media of concern. In general, contaminants detected in one or more samples were retained as contaminants of concern for each medium. The exceptions were (1) elements identified as essential nutrients and common food components and (2) common laboratory contaminants that were also present in blank samples. In addition, a limited number of organic contaminants detected only in sludge were screened because of their low contribution to the overall risks associated with this medium. This more limited screening process was conducted primarily because of the much smaller number of contaminants detected in other media compared with soil. In addition, low sampling frequencies and limitations in background data for some media precluded the screening of contaminants on the basis of a comparison with background concentrations.

The final lists of radioactive and chemical contaminants of concern for the Weldon Spring site are presented in Tables 2.16 and 2.17, respectively. The chemical classes to be evaluated in this risk assessment are radionuclides, metals, inorganic anions, asbestos, PAHs, PCBs, and nitroaromatic compounds.

TABLE 2.1 Principal Contaminant Characterization Studies Conducted for the Weldon Spring Site[a]

Medium and Study[b]	Year[c]	Analytical Parameters[d]	Reference
Soil			
Radiological Survey of the Busch and Weldon Spring Wildlife Areas	1984-1985	Radionuclides	Boerner (1986)
Radiological Survey of the U.S. Army Reserve Property	1985	Radionuclides	Deming (1986)
UNC-Georech Radiological Characterization Report	1987	Radionuclides	Marutzky et al. (1988)
Phase I Chemical Soil Investigation Report	1987	Metals Inorganic anions Nitroaromatic compounds	MK-Ferguson Company and Jacobs Engineering Group (1988d)
Radiological Characterization of Vicinity Properties	1987-1988	Radionuclides	MK-Ferguson Company and Jacobs Engineering Group (1987b, 1987c, 1987f, 1987g, 1988f)
Chemical Soil Data Report to Support Interim Response Actions	1987-1988	Metals Inorganic anions Volatile organic compounds Semivolatile organic compounds PCBs Pesticides Nitroaromatic compounds	MK-Ferguson Company (1989a)
Phase II Chemical Soil Investigation Report	1988	Metals Inorganic anions Volatile organic compounds Semivolatile organic compounds PCBs Pesticides Nitroaromatic compounds	MK-Ferguson Company (1989b)
Radiological Characterization of Site Soil	1988-1989	Radionuclides	DOE (1992c)

TABLE 2.1 (Cont.)

Medium and Study[b]	Year[c]	Analytical Parameters[d]	Reference
Surface Water			
Annual Site Environmental Monitoring Report	1986-1990	Radionuclides Metals Inorganic anions	MK-Ferguson Company and Jacobs Engineering Group (1987a, 1988a, 1989a, 1990a, 1991)
Water Quality Phase I Assessment	1987	Radionuclides Metals Inorganic anions	MK-Ferguson Company (1987)
Phase I Spring and Seep Report	1987-1988	Radionuclides Metals Inorganic anions Volatile organic compounds PCBs Pesticides Nitroaromatic compounds	MK-Ferguson Company and Jacobs Engineering Group (1989c)
Waste Assessment Chemical Characterization of the Raffinate Pits	1988-1989	Radionuclides Metals	MK-Ferguson Company and Jacobs Engineering Group (1989f)
Sediment			
Radiological Survey of the Busch and Weldon Spring Wildlife Areas	1984-1985	Radionuclides	Boerner (1986)
Radiological Survey of the U.S. Army Reserve Property	1985	Radionuclides	Deming (1986)
Characterization of Chemical and Radiological Contamination on Surrounding Properties	1988	Radionuclides Metals Semivolatile organic compounds PCBs Nitroaromatic compounds	MK-Ferguson Company and Jacobs Engineering Group (1989b)

TABLE 2.1 (Cont.)

Medium and Study[b]	Year[c]	Analytical Parameters[d]	Reference
Sludge			
Waste Assessment Radiological Characterization of the Raffinate Pits	1988-1989	Radionuclides	MK-Ferguson Company and Jacobs Engineering Group (1989g)
Waste Assessment Chemical Characterization of the Raffinate Pits	1988-1989	Radionuclides Metals Inorganic anions Volatile organic compounds Semivolatile organic compounds PCBs Pesticides Nitroaromatic compounds	MK-Ferguson Company and Jacobs Engineering Group (1989f)
Groundwater			
Annual Site Environmental Monitoring Report	1986-1990	Radionuclides Inorganic anions Nitroaromatic compounds	MK-Ferguson Company and Jacobs Engineering Group (1987a, 1988a, 1989a, 1990a, 1991)
Water Quality Phase I Assessment	1987	Radionuclides Metals Inorganic anions Volatile organic compounds Semivolatile organic compounds PCBs Pesticides Nitroaromatic compounds	MK-Ferguson Company (1987)
Phase II Groundwater Quality Assessment	1989	Radionuclides Metals Inorganic anions Nitroaromatic compounds	MK-Ferguson Company and Jacobs Engineering Group (1989d)
Air			
Annual Site Environmental Monitoring Report	1986-1991	Radionuclides Asbestos	MK-Ferguson Company and Jacobs Engineering Group (1987a, 1988a, 1989a, 1990a, 1991, 1992c)

TABLE 2.1 (Cont.)

Medium and Study[b]	Year[c]	Analytical Parameters[d]	Reference
Buildings			
Buildings Radiological Characterization Report	1967-1972, 1977, 1986, 1988-1989	Radionuclides	MK-Ferguson Company and Jacobs Engineering Group (1990c)
Assessment of Disposition Alternatives	1977	Radionuclides	Ryckman/Edgerley/Tomlinson & Associates (1978)
Radiological and Chemical Characterization Report	1986	Radionuclides Combustible gases	Bechtel National (1986)
Asbestos Bulk Sampling and Air Sampling Report	1986	Asbestos	MK-Ferguson Company and Jacobs Engineering Group (1986)
Asbestos Analysis Report	1987	Asbestos	metaTrace (1987)
Asbestos and Chemical Characterization of Non-Process Related Buildings and Equipment	1987-1989	Asbestos PCBs	MK-Ferguson Company and Jacobs Engineering Group (1990b)
Biota			
Assessment of Disposition Alternatives	1977	Radionuclides	Ryckman/Edgerley/Tomlinson & Associates (1978)
Radiological and Chemical Uptake of Biota	1987-1988	Radionuclides Metals PCBs	MK-Ferguson Company and Jacobs Engineering Group (1989e)

[a] This table presents information discussed in Sections 2.3 and 2.4.1. A complete listing of the contaminant characterization studies is given in the RI report (DOE 1992c, Table 1.2-1).

[b] Study names have been abbreviated for clarity of presentation.

[c] Year represents the year that samples were collected.

[d] Specific metals, inorganic anions, volatile organic compounds, semivolatile organic compounds, PCBs, pesticides, and nitroaromatic compounds are listed in Appendix A.

TABLE 2.2 Principal and Associated Radionuclides in the Uranium-238, Thorium-232, and Uranium-235 Decay Series[a]

Principal Radionuclide[b]		Associated Decay Chain[c]		Terminal Nuclide or Radionuclide[d]	
Species	Half-Life	Species	Half-Life	Species	Half-Life
Uranium-238 Decay Series					
Uranium-238	4.5×10^9 yr	Thorium-234 Protactinium-234	24 d 1.2 min	Uranium-234	2.4×10^5 yr
Uranium-234	2.4×10^5 yr	-	-	Thorium-230	7.7×10^4 yr
Thorium-230	7.7×10^4 yr	-	-	Radium-226	1.6×10^3 yr
Radium-226	1.6×10^3 yr	Radon-222 Polonium-218 Lead-214 Bismuth-214 Polonium-214	3.8 d 3.1 min 27 min 20 min 160 µs	Lead-210	22 yr
Lead-210	22 yr	Bismuth-210 Polonium-210	5.0 d 140 d	Lead-206	*
Thorium-232 Decay Series					
Thorium-232	1.4×10^{10} yr	-	-	Radium-228	5.8 yr
Radium-228	5.8 yr	Actinium-228	6.1 h	Thorium-228	1.9 yr
Thorium-228	1.9 yr	Radium-224 Radon-220 Polonium-216 Lead-212 Bismuth-212 [Polonium-212 Thallium-208	3.6 d 55 s 0.15 s 11 h 61 min 0.30 µs (64%) 3.1 min (36%)]	Lead-208	*
Uranium-235 Decay Series					
Uranium-235	7.0×10^8 yr	Thorium-231	26 h	Protactinium-231	3.3×10^4 yr
Protactinium-231	3.3×10^4 yr	-	-	Actinium-227	22 yr
Actinium-227	22 yr	[Thorium-227 Francium-223 Radium-223 Radon-219 Polonium-215 Lead-211 Bismuth-211 Thallium-207	19 d (98.6%) 22 min (1.4%)] 11 d 4.0 s 1.8 ms 36 min 2.1 min 4.8 min	Lead-207	*

[a] This table presents information discussed in Section 2.3.2.1.

[b] Radionuclides with half-lives greater than 1 year. Half-lives are given to two significant figures.

[c] The chain of decay products of a principal radionuclide extending to (but not including) the next principal radionuclide or a stable nuclide. Branches are indicated by square brackets with branching ratios in parentheses following the half-life. Branching ratios of less than 1% are not included in this table.

[d] The principal radionuclide or stable nuclide that terminates an associated decay chain. Stable nuclides are indicated by an asterisk (*) in place of the half-life.

Source: Data from Gilbert et al. (1989).

TABLE 2.3 Activity Concentration Ratios Estimated for the Major Radionuclides[a]

Radionuclide	Activity Concentration Ratio[b]		
	Raffinate Pit Sludge	Chemical Plant Area Soil	Quarry Soil
Actinium-227	0.58[c]	0.019	0.0029
Lead-210	0.62[d]	0.14	1.7[e]
Protactinium-231	0.73[c]	0.038	0.0058
Radium-226	0.62[f]	0.14	0.55[g]
Radium-228	0.25[f]	0.076[h]	0.48[g]
Thorium-230	25[f]	0.15	1.7[g]
Thorium-232	0.026[i]	0.076	0.13[g]
Uranium-235	0.046[j]	0.046[j]	0.046[j]
Uranium-238	1.0	1.0	1.0

[a] This table presents information discussed in Section 2.3.2.1.

[b] Results are reported as activity concentration ratios normalized to a unit concentration of uranium-238. Based on information provided in MK-Ferguson Company and Jacobs Engineering Group (1992b), except as noted. Ratios are reported to two significant figures.

[c] Determined from results for raffinate pit 1.

[d] The activity concentration of lead-210 is assumed to be equal to that of radium-226 on the basis of their measured values in the sludge.

[e] Determined from activity concentration ratios reported for uranium-238 to radium-226 and uranium-238 to lead-210 and on measured concentrations of radium-226 and uranium-238 reported for the quarry bulk wastes (DOE 1989).

[f] Determined from UL_{95} concentrations of radium-226, radium-228, thorium-230, and uranium-238 in the sludge.

[g] Determined from measured concentrations of radionuclides reported for the quarry bulk wastes (DOE 1989).

[h] The activity concentration of radium-228 is assumed to be equal to that of thorium-232.

[i] Determined from activity concentration ratios of radium-228 to thorium-232 in sludge from each of the four pits, weighted by the volume of sludge in each pit and the UL_{95} concentration of radium-228 in the sludge.

[j] Based on the naturally occurring ratio of uranium-235 to uranium-238.

TABLE 2.4 Characterization Results for Radionuclides, Metals, and Inorganic Anions in Site Soil[a]

Contaminant[b]	Unit	Detection Limit[c]	Detection Frequency	Percent Detects	Concentration Range[d]	Local Background[e] Average	Local Background[e] Upper Bound	Common Range in Soil[f]
Radionuclides								
Radium-226	pCi/g	0.2-1.5	72/170	42	0.4-450	1.2	1.9	0.3-1.4
Radium-228	pCi/g	0.1-1.5	66/171	39	0.4-150	1.2	1.5	0.8-1.3
Thorium-230	pCi/g	0.2-1.5	40/135	30	0.3-97	1.2	1.9	0.3-1.4
Uranium-238	pCi/g	0.3-3.0	340/387	88	0.3-2,300	1.2	1.7	0.3-1.4
Metals								
Aluminum	mg/kg	20	924/942	98	660-160,000	14,000	24,000	10,000-300,000
Antimony	mg/kg	6	138/924	15	6.4-110	3	3	0.1-10[g]
Arsenic	mg/kg	1	652/702	93	1.3-130	11	26	1-50
Barium	mg/kg	20	920/943	98	25-5,200	210	620	100-3,000
Beryllium	mg/kg	0.5	562/714	79	0.51-5.5	1.2	2.3	0.1-40
Cadmium	mg/kg	0.5	306/655	47	0.51-11	0.37	0.88	0.01-0.70
Calcium	mg/kg	500	693/717	97	690-220,000	2,800	5,000	7,000-500,000
Chromium	mg/kg	1	698/717	97	2.2-310	23	40	1-1,000
Cobalt	mg/kg	5	584/716	82	2.8-110	11	25	1-40
Copper	mg/kg	2.5	688/712	97	3.6-460	16	26	2-100
Iron	mg/kg	10	939/943	100	12-85,000	19,000	31,000	7,000-550,000
Lead	mg/kg	0.5	887/937	95	1.3-1,900[h]	22	43	2-200
Lithium	mg/kg	5	359/619	58	5.3-71	9.2	26	5-200
Magnesium	mg/kg	500	916/942	97	590-30,000[i]	2,200	3,400	600-6,000
Manganese	mg/kg	1.5	927/943	98	3.3-13,000	570	1,600	20-3,000
Mercury	mg/kg	0.1	112/572	20	0.11-2.1	0.05	0.09	0.01-0.3
Molybdenum	mg/kg	4	502/522	96	4.1-120	16	25	0.2-5
Nickel	mg/kg	4	694/718	97	5.6-270	20	44	5-500
Potassium	mg/kg	500	449/697	64	510-12,000	280	550	40-30,000
Selenium	mg/kg	0.5	151/694	22	0.63-47	11	21	0.1-2
Silver	mg/kg	1	126/584	22	0.92-13	0.5	0.5	0.01-5
Sodium	mg/kg	500	50/717	7	510-1,400	280	490	750-7,500
Thallium	mg/kg	1	83/633	13	1.0-80	6.2	16	0.1-5
Uranium, total	mg/kg	0.9-9.0	340/387	88	0.9-6,900	3.6	5.1	0.9-4.2
Vanadium	mg/kg	5	690/717	96	7.2-380	35	55	20-500

TABLE 2.4 (Cont.)

Contaminant[b]	Unit	Detection Limit[c]	Detection Frequency	Percent Detects	Concentration Range[d]	Local Background[e] Average	Local Background[e] Upper Bound	Common Range in Soil[f]
Metals (cont.)								
Zinc	mg/kg	2	700/718	97	6.1-1,100	38	78	10-300
Zirconium	mg/kg	20	1/1	100	54	NA[j]	NA	60-2,000
Inorganic anions								
Chloride	mg/kg	1.5	478/554	86	1.3-70	-	-	NA
Fluoride	mg/kg	1.3	949/1,005	94	1.3-45	6.3	14	NA
Nitrate	mg/kg	0.5	758/1,006	75	0.54-3,800	11	66	NA
Nitrite	mg/kg	0.5	14/720	2	1.5-29	0.28	0.64	NA
Sulfate	mg/kg	5	941/1,006	94	1.2-3,100	32	90	NA

[a] This table presents information discussed in Section 2.4.1.

[b] The listed contaminants include all radionuclides, metals, and inorganic anions for which an analysis was conducted. A source term analysis was developed for site soil so that information could be inferred for radionuclides in the uranium-238, thorium-232, and uranium-235 decay series that were not directly measured but that may be present on-site. The results of this analysis are given in Table 2.3.

[c] For chemicals, see Appendix A. For radionuclides, the range of detection limits accounts for variation between different studies and different instrumentation; values reported below the standard detection limit indicate that the method or instrument provided more sensitivity for that analysis.

[d] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample. Concentrations are dry weight measurements, except for those radioactive measurements that were taken in situ.

[e] Local background results are presented for samples from the nearby vicinity of the site but beyond its area of influence. The reported average is the arithmetic mean; upper bound represents the mean plus two standard deviations. Concentrations are dry weight measurements, except for those radioactive measurements that were taken in situ.

[f] Data from Myrick et al. (1981) for radionuclides; Friberg et al. (1986) for antimony; Adriano (1986) for thallium; and Lindsay (1979) for other metals and inorganic anions. For radionuclides, the concentration range is representative of soils in Missouri (for uranium, Lindsay [1979] identifies a range of 0.9 to 9 mg/kg). The concentration reported for radium-228 has been inferred from measurements of thorium-232; the concentration of thorium-230 is taken to be the same as that reported for radium-226.

[g] A concentration of 0.2 mg/kg has been reported for limestone, and soil not derived from ore-bearing rock does not generally contain more than 1 mg/kg (Ainsworth 1988).

[h] One high sample was measured at 43,000 mg/kg.

[i] One high sample was measured at 190,000 mg/kg.

[j] NA = not available; a hyphen indicates that the parameter was not assayed.

TABLE 2.5 Characterization Results for Organic Compounds in Site Soil[a]

Contaminant[b]	Detection Limit[c] (µg/kg)	Detection Frequency	Percent Detects	Concentration Range[d] (µg/kg)	Concentration in Blanks[e] (µg/kg)
Volatile organic compounds					
Acetone	10	103/256	40	10-600	33-200
Carbon disulfide	5	2/252	0.8	22-35	-
Chloroethane	10	1/252	0.4	12	-
Chloroform	5	4/252	1.6	20-42	-
1,2-Dichloroethene	5	4/252[f]	1.6	27-6,200	-
Ethylbenzene	5	1/252	0.4	41	-
Methylene chloride	5	178/261	68	6-280	6-71
Toluene	5	9/252	3.6	11-160	8
Trichloroethene	5	6/252	2.4	29-320[g]	-
Xylenes, total	5	1/252	0.4	27	-
Semivolatile organic compounds					
Acenaphthene	330	1/329	0.3	1,900	-
Anthracene	330	1/329	0.3	3,400	-
Benz(a)anthracene	330	2/329	0.6	410-8,200	-
Benzidine	NA[h]	1/329	0.3	2,400	-
Benzo(b)fluoranthene	330	1/329	0.3	4,600	-
Benzo(k)fluoranthene	330	1/329	0.3	3,900	-
Benzo(g,h,i)perylene	330	1/329	0.3	2,100	-
Benzo(a)pyrene	330	1/328	0.3	5,100	-
Bis(2-ethylhexyl)phthalate	330	109/339	32	330-15,000[i]	-
Butylbenzylphthalate	330	12/329	3.6	630-2,700	-
Chrysene	330	2/329	0.6	390-8,000	-
Dibenzofuran	330	2/329	0.6	820-1,900	-
Di-n-butylphthalate	330	26/334	7.8	430-17,000	-
Diethylphthalate	330	5/329	1.5	440-1,300	-
Di-n-octylphthalate	330	2/329	0.6	400-540	-
Fluoranthene	330	2/329	0.6	580-11,000	-
Fluorene	330	1/329	0.3	1,600	-
Indeno(1,2,3-cd)pyrene	330	1/329	0.3	3,200	-
2-Methylnaphthalene	330	4/329	1.2	520-4,600	-
Naphthalene	330	1/329	0.3	1,800	-
Phenanthrene	330	4/329	1.2	420-11,000	-
Pyrene	330	3/329	0.9	350-19,000	-
PCBs/Pesticides					
Aldrin	8	5/166	3	18-240[j]	-
Aroclor 1248	80	2/166	1.2	320-470	-
Aroclor 1254	160	12/166	7.2	180-12,000	-
Aroclor 1260	160	6/166	3.6	390-7,400	-
Beta-BHC	8	3/166	1.8	10-270	-
Endosulfan I	8	1/166	0.6	31	-
Heptachlor	8	1/166	0.6	150	-

TABLE 2.5 (Cont.)

Contaminant[b]	Detection Limit[c] (µg/kg)	Detection Frequency	Percent Detects	Concentration Range[d] (µg/kg)	Concentration in Blanks[e] (µg/kg)
Nitroaromatic compounds					
DNB	900	5/790	0.6	1,000-3,800	-
2,4-DNT	750	12/791	1.5	830-6,300	-
2,6-DNT	1,410	5/791	0.6	1,600-3,500	-
NB	1,440	6/793	0.8	1,600-3,800	-
TNB	570	15/791	1.9	630-5,700	-
TNT	1,200	13/790	1.6	1,300-32,000	-

[a] This table presents information discussed in Section 2.4.1.2.

[b] Samples were analyzed for all compounds on the target compound list and nitroaromatic compounds. The listed contaminants are those for which the measurement was above the detection limit in at least one sample.

[c] See Appendix A.

[d] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample. Concentrations are reported as dry weights.

[e] Concentrations are reported as dry weights. A hyphen indicates that the contaminant was not detected in the blanks.

[f] 1,2-Dichloroethene is on the target compound list, so an analysis was routinely run for this compound. However, results for only four samples are listed in the database (i.e., the nondetects were not included).

[g] One high sample was reported at 4,800 µg/kg; however, the laboratory data sheet for this analysis could not be located to validate the measurement.

[h] NA = not applicable because this was reported as a tentatively identified compound (meaning that the mass spectra from the chromatogram output for the analysis match those for the listed compound to some degree, but neither the identity of the specific compound nor its concentration can be assigned with any certainty). Therefore, the related information is qualitative rather than quantitative.

[i] One high sample was measured at 1,900,000 µg/kg.

[j] One high sample was measured at 1,700 µg/kg.

TABLE 2.6 Soil Contaminants Deleted from the Risk Assessment[a]

Contaminant	Screening Rationale
Metals	
Aluminum	Maximum concentration is within the range typically found in soil. A normal dietary constituent of low toxicity.
Calcium	Maximum concentration is within the range typically found in soil. A normal dietary constituent of low toxicity.
Iron	Maximum concentration is within the range typically found in soil. A normal dietary constituent of low toxicity.
Magnesium	A normal dietary constituent of low toxicity.
Potassium	Maximum concentration is within the range typically found in soil. A normal dietary constituent of low toxicity.
Sodium	Maximum concentration is within the range typically found in soil. A normal dietary constituent of low toxicity.
Zirconium	Data, which are available for only one soil sample, are inadequate for a risk assessment. Zirconium is not expected to be present at elevated concentrations.
Inorganic anions	
Chloride	A normal dietary constituent of low toxicity.
Sulfate	A normal dietary constituent of low toxicity.
Volatile organic compounds	
Acetone	Concentrations in samples were less than 10 times the maximum concentration detected in blanks. Contribution to site risks is insignificant based on the concentration-toxicity screen (see text).
Carbon disulfide	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Chloroethane	Low detection frequency.
Chloroform	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
1,2-Dichloroethene	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Ethylbenzene	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Methylene chloride	Concentrations in samples were less than 10 times the maximum concentration detected in blanks. Contribution to site risks is insignificant based on the concentration-toxicity screen.

TABLE 2.6 (Cont.)

Contaminant	Screening Rationale
Volatile organic compounds (cont.)	
Toluene	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Trichloroethene	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Xylenes	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Semivolatile organic compounds	
Benzidine	A tentatively identified compound, present in only one sample. This compound would not be expected to be present based on site history.
Bis(2-ethylhexyl)phthalate	Concentrations in samples were less than 10 times the maximum concentration detected in blanks. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Butylbenzylphthalate	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Dibenzofuran	Low detection frequency.
Di-n-butylphthalate	Contribution to site risks is insignificant based on the concentration-toxicity screen.
Diethylphthalate	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Di-n-octylphthalate	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Pesticides	
Aldrin	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Beta-BHC	Low detection frequency.
Endosulfan I	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.
Heptachlor	Low detection frequency. Contribution to site risks is insignificant based on the concentration-toxicity screen.

[a] This table presents information discussed in Section 2.4.1.2.

TABLE 2.7 Characterization Results for Raffinate Pit Surface Water and Sludge[a]

Contaminant	Surface Water				Sludge			
	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]
Radionuclides[d]								
Gross alpha	pCi/L	1.0-3.0	12/12	100-1,100	NA[e]	NA	NA	NA
Gross beta	pCi/L	2.0	12/12	66-1,200	NA	NA	NA	NA
Lead-210	pCi/L	1.0	1/4	1.8	pCi/g	-[e]	-	-
Radium-226	pCi/L	0.2-1.0	16/16	3.4-130	pCi/g	0.1	78/79	1.0-1,700
Radium-228	pCi/L	1.0	4/4	1.5-25	pCi/g	1.0	76/79	4.0-1,400
Thorium-228	pCi/L	0.5-1.0	3/7	1.2-2.7	pCi/g	1.0	80/82	5.2-1,100
Thorium-230	pCi/L	0.6-1.0	12/12	1.4-760	pCi/g	1.0	82/82	8.0-34,000
Thorium-232	pCi/L	0.2-1.0	7/12	0.2-7.6	pCi/g	1.0	82/82	3.0-1,400[f]
Uranium, total	pCi/L	0.68-1.0	20/20	57-2,700	pCi/g	1.0	79/79	10-3,400
Metals								
Aluminum	µg/L	200	5/12	230-520	mg/kg	20	102/103	1,300-29,000
Antimony	µg/L	60	4/12	65-400	mg/kg	6	91/103	6.0-87
Arsenic	µg/L	10	6/12	12-120	mg/kg	1	102/102	3.1-1,100
Barium	µg/L	200	0/12	ND[e]	mg/kg	20	97/103	20-7,700
Beryllium	µg/L	5	2/12	7.0-8.0	mg/kg	0.5	98/103	0.59-25
Cadmium	µg/L	5	1/12	37	mg/kg	0.5	91/103	0.94-14
Calcium	µg/L	5,000	12/12	10,000-460,000	mg/kg	500	101/103	1,800-86,000
Chromium	µg/L	10	4/12	31-190	mg/kg	1	77/103	5.0-170
Cobalt	µg/L	50	0/12	ND	mg/kg	5	71/103	5.1-44
Copper	µg/L	25	2/12	30-45	mg/kg	2.5	102/103	3.7-510
Iron	µg/L	100	9/12	57-460	mg/kg	10	103/103	30-25,000
Lead	µg/L	5	3/12	22-360	mg/kg	0.5	94/103	2.1-640
Lithium	µg/L	50	9/13	180-4,500	mg/kg	5	47/102	5.0-120
Magnesium	µg/L	5,000	12/12	15,000-380,000	mg/kg	500	99/103	1,300-21,000
Manganese	µg/L	15	4/12	16-33	mg/kg	1.5	102/103	25-3,000
Mercury	µg/L	0.2	1/12	0.29	mg/kg	0.1	47/102	0.10-15
Molybdenum	µg/L	40	8/8	690-4,100	mg/kg	4	102/103	16-1,600
Nickel	µg/L	40	3/12	47-170	mg/kg	4	103/103	3.3-8,800
Potassium	µg/L	5,000	12/12	14,000-110,000	mg/kg	500	26/102	140-1,800
Selenium	µg/L	5	6/12	7.5-220	mg/kg	0.5	89/103	2.7-81
Silver	µg/L	10	2/12	25-40	mg/kg	1	16/102	1.0-5.0

TABLE 2.7 (Cont.)

Contaminant	Surface Water				Sludge			
	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]
Metals (cont.)								
Sodium	µg/L	5,000	12/12	110,000-1,500,000	mg/kg	500	92/103	530-24,000
Thallium	µg/L	10	0/12	ND	mg/kg	1	72/103	1.1-58
Uranium, total	µg/L	1.5	20/20	84-4,000	mg/kg	1.5	79/79	15-5,000
Vanadium	µg/L	50	8/12	90-2,100	mg/kg	5	102/103	26-8,700
Zinc	µg/L	20	4/12	26-60	mg/kg	2	103/103	7.9-1,600
Zirconium	µg/L	200	-	-	mg/kg	20	62/103	24-2,100
Inorganic anions								
Chloride	mg/L	0.25/0.2	16/16	5.0-30	mg/kg	1.5	102/102	1.7-300
Fluoride	mg/L	0.25/0.6	15/16	0.23-19	mg/kg	1.25	43/101	3.2-170
Nitrate (as N)	mg/L	0.25/0.1	12/16	0.25-7,200	mg/kg	0.5	101/102	0.6-160,000
Nitrite (as N)	mg/L	0.25	-	-	mg/kg	0.5	87/100	1.0-1,600
Sulfate	mg/L	1.0	16/16	38-810	mg/kg	5	102/102	7.3-7,800
Volatile organic compounds								
Acetone	µg/L	10	-	-	µg/kg	10	57/75	15-830
Benzene	µg/L	5	-	-	µg/kg	5	25/75	6.0-88
2-Butanone	µg/L	10	-	-	µg/kg	10	2/75	140-170
Carbon disulfide	µg/L	5	-	-	µg/kg	5	1/75	41
1,2-Dichloro-propane	µg/L	5	-	-	µg/kg	5	1/75	8
2-Hexanone	µg/L	10	-	-	µg/kg	10	1/75	50
Methylene chloride	µg/L	5	-	-	µg/kg	5	19/75	6.0-60
Toluene	µg/L	5	-	-	µg/kg	5	3/75	6.0-290
1,1,1-Trichloro-ethane	µg/L	5	-	-	µg/kg	5	1/75	6.0
Semivolatile organic compounds								
Bis(2-ethylhexyl) phthalate	µg/L	10	-	-	µg/kg	330	7/98	330-2,200
Di-n-butylphthalate	µg/L	10	3/8	10-35	µg/kg	330	5/98	500-1,100

TABLE 2.7 (Cont.)

Contaminant	Surface Water				Sludge			
	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]
PCBs/Pesticides								
Aroclor 1248	µg/L	0.5	ND	ND	µg/kg	80	4/61	140-420
Aroclor 1254	µg/L	1.0	ND	ND	µg/kg	160	13/61	180-11,000
Beta-BHC	µg/L	0.05	ND	ND	µg/kg	8	1/61	340
4,4-DDE	µg/L	0.1	ND	ND	µg/kg	16	7/61	20-95

[a] This table presents information discussed in Section 2.4.2. Surface water and sludge samples were analyzed for radionuclides, metals, inorganic anions, volatile and semivolatile organic compounds, PCBs, pesticides, and nitroaromatic compounds. The listed contaminants are those for which the measurement was above the detection limit in at least one sample.

[b] The detection limit is the contract-required detection limit or method detection limit, and detection limits for each contaminant are listed in Appendix A. For radionuclides, the range in detection limits reflects differences in instrumentation and laboratory procedures. For cases in which more than one detection limit is shown for a single contaminant, that contaminant has been analyzed by different instrumentation and/or procedures at various times of the characterization effort; values reported below the standard detection limit indicate that the instrument or method provided more sensitivity than required for that analysis.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample. Sludge concentrations are reported as wet weights (the sludge contains about 73% water by weight).

[d] A source term analysis was developed for the raffinate pit sludge so that information could be inferred for radionuclides in the uranium-238, thorium-232, and uranium-235 decay series that were not directly measured but that may be present in the sludge. The results of this analysis are given in Table 2.3.

[e] NA = not applicable; ND = not detected; a hyphen indicates that the contaminant was not assayed.

[f] Thorium-232 data are available only for the sludge in raffinate pit 4 because interference from the high thorium-230 concentration in pits 1, 2, and 3 affected thorium-232 measurements for those pits.

TABLE 2.8 Characterization Results for NPDES Outfalls[a]

Contaminant	Unit	Detection Limit[b]	NP-0002 Detection Frequency	NP-0002 Concentration Range[c]	NP-0003 Detection Frequency	NP-0003 Concentration Range[c]	NP-0004 Detection Frequency	NP-0004 Concentration Range[c]
Radionuclides								
Gross alpha	pCi/L	1.0/4.0	31/35	16-790	31/31	9.0-2,900	7/16	3.7-8.0
Gross beta	pCi/L	1.0	14/14	24-96	6/6	59-490	4/4	5.7-6.9
Radium-226	pCi/L	1.0	0/1	ND[d]	0/1	ND	-[d]	-
Thorium-230	pCi/L	1.0	0/1	ND	0/1	ND	-	-
Thorium-232	pCi/L	1.0	0/1	ND	0/1	ND	-	-
Uranium, total	pCi/L	0.68/1.0	67/71	7.7-1,500	35/35	16-3,500	13/16	3.0-11.0
Metals								
Aluminum	µg/L	200	0/2	ND	0/2	ND	-	-
Arsenic	µg/L	10	0/1	ND	0/1	ND	-	-
Barium	µg/L	200	0/2	ND	0/2	ND	-	-
Beryllium	µg/L	5	-	-	-	-	1/1	9.0
Calcium	µg/L	5,000	0/2	ND	0/2	ND	-	-
Copper	µg/L	25	0/3	ND	0/3	ND	-	-
Iron	µg/L	100	0/2	ND	0/2	ND	-	-
Lead	µg/L	5	0/1	ND	1/1	190	1/1	450
Lithium	µg/L	50	0/21	ND	1/22	61	0/7	ND
Magnesium	µg/L	5,000	0/2	ND	0/2	ND	-	-
Manganese	µg/L	15	0/2	ND	0/2	ND	-	-
Mercury	µg/L	0.2	1/1	0.35	1/1	0.36	-	-
Potassium	µg/L	5,000	0/1	ND	0/1	ND	-	-
Sodium	µg/L	5,000	0/1	ND	0/2	ND	-	-
Thallium	µg/L	10	0/1	ND	0/1	ND	-	-
Uranium, total	µg/L	1.5	67/71	11-2,200	35/35	23-5,100	13/16	4.4-16
Zinc	µg/L	20	0/2	ND	0/1	ND	-	-
Inorganic anions								
Chloride	mg/L	0.25/0.2	5/6	71-360	4/4	0.80-13	2/2	1.0-2.0
Fluoride	mg/L	0.25/0.6	2/3	0.30-1.4	4/4	0.30-0.50	2/2	0.37-0.60
Nitrate	mg/L	0.25/0.1	25/35	0.48-8.3	22/31	0.19-740	13/17	0.20-5
Sulfate	mg/L	1.0	6/6	16-70	4/4	22-59	2/2	270-360

TABLE 2.8 (Cont.)

Contaminant	Unit	Detection Limit[b]	NP-0002 Detection Frequency	NP-0002 Concentration Range[c]	NP-0003 Detection Frequency	NP-0003 Concentration Range[c]	NP-0004 Detection Frequency	NP-0004 Concentration Range[c]
Nitroaromatic compounds								
DNB	µg/L	0.51/0.09	0/14	ND	0/9	ND	0/10	ND
2,4-DNT	µg/L	0.44/0.03	0/14	ND	0/9	ND	0/10	ND
2,6-DNT	µg/L	0.48/0.01	0/14	ND	0/9	ND	0/10	ND
NB	µg/L	0.54/0.03	0/14	ND	0/9	ND	0/10	ND
TNB	µg/L	0.47/0.03	2/14	0.20-1.4	0/9	ND	2/10	0.04-0.65
TNT	µg/L	0.98/0.03	0/14	ND	0/9	ND	2/10	0.80-7.5

[a] This table presents information discussed in Section 2.4.2. NP refers to an NPDES outfall; data for NPDES outfall locations NP-0001 and NP-0002 are presented with characterization data for the Southeast Drainage in Table 2.10. The listed contaminants include all those for which an analysis was conducted.

[b] For cases in which two detection limits are shown for a single contaminant, that contaminant was analyzed by two different methods at various times of the characterization effort.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample.

[d] ND = not detected; a hyphen indicates that the contaminant was not assayed.

TABLE 2.9 Characterization Results for Surface Water in the Busch Wildlife Area[a]

Contaminant	Unit	Detection Limit[b]	Burgermeister Spring		Lakes 34, 35, and 36	
			Detection Frequency	Concentration Range[c]	Detection Frequency	Concentration Range[c]
Radionuclides						
Gross alpha	pCi/L	1.0-3.0	10/10	14-97	27/27	4.4-90
Gross beta	pCi/L	1.0/2.0	10/10	11-54	27/27	7.6-48
Radium-226	pCi/L	1.0	0/9	ND[d]	0/27	ND
Radium-228	pCi/L	1.0	0/2	ND	0/4	ND
Thorium-230	pCi/L	1.0	1/9	8	4/27	1.0-1.5
Thorium-232	pCi/L	1.0	0/9	ND	0/27	ND
Uranium, total	pCi/L	0.68/1.0	44/44	14-240	51/51	4.1-240
Metals						
Aluminum	µg/L	200	2/4	610-1,000	3/7	230-550
Antimony	µg/L	60	0/4	ND	0/7	ND
Arsenic	µg/L	10	0/4	ND	2/7	12-29
Barium	µg/L	200	1/5	78	0/7	ND
Beryllium	µg/L	5	0/4	ND	0/7	ND
Cadmium	µg/L	5	0/4	ND	0/7	ND
Calcium	µg/L	5,000	4/4	31,000-68,000	7/7	11,000-38,000
Chromium	µg/L	10	1/4	26	1/7	19
Cobalt	µg/L	50	0/4	ND	0/7	ND
Copper	µg/L	25	0/4	ND	0/7	ND
Iron	µg/L	100	2/4	670-1,600	7/7	180-670
Lead	µg/L	5	0/4	ND	1/7	9.5
Lithium	µg/L	50	0/5	ND	0/7	ND
Magnesium	µg/L	5,000	4/4	6,500-20,000	4/7	6,000-11,000
Manganese	µg/L	15	3/4	20-36	7/7	31-870

TABLE 2.9 (Cont.)

Contaminant	Unit	Detection Limit[b]	Burgermeister Spring		Lakes 34, 35, and 36	
			Detection Frequency	Concentration Range[c]	Detection Frequency	Concentration Range[c]
Metals (cont.)						
Mercury	µg/L	0.2	1/4	0.37	6/10	0.35-1.3
Molybdenum	µg/L	40	1/3	22	0/6	ND
Nickel	µg/L	40	0/4	ND	0/7	ND
Potassium	µg/L	5,000	1/4	3,700	0/7	ND
Selenium	µg/L	5	0/4	ND	0/7	ND
Silver	µg/L	10	0/4	ND	0/7	ND
Sodium	µg/L	5,000	4/4	9,100-34,000	7/7	5,100-120,000
Thallium	µg/L	10	0/4	ND	1/7	33
Uranium, total	µg/L	1.5	44/44	21-350	51/51	6.0-350
Vanadium	µg/L	50	0/4	ND	0/8	ND
Zinc	µg/L	20	2/4	29-31	3/7	21-78
Inorganic anions						
Chloride	mg/L	0.25/0.2	14/14	4-42	31/31	5.9-200
Fluoride	mg/L	0.25/0.6	6/14	0.17-0.31	10/31	0.26-0.60
Nitrate	mg/L	0.25/0.1	16/17	3-200	24/31	0.30-8.9
Sulfate	mg/L	1.0	14/14	14-86	31/31	9.9-93
Volatile organic compounds						
Acetone	µg/L	10	1/1	150	-[d]	-
Methylene chloride	µg/L	5	1/1	15	-	-

TABLE 2.9 (Cont.)

Contaminant	Unit	Detection Limit[b]	Burgermeister Spring		Lakes 34, 35, and 36	
			Detection Frequency	Concentration Range[c]	Detection Frequency	Concentration Range[c]
Nitroaromatic compounds						
DNB	μg/L	0.51/0.09	1/14	0.18	0/8	ND
2,4-DNT	μg/L	0.44/0.03	5/15	0.30-1.4	1/8	0.58
2,6-DNT	μg/L	0.48/0.01	6/15	0.19-18	0/8	ND
NB	μg/L	0.54/0.03	0/14	ND	0/8	ND
TNB	μg/L	0.47/0.03	1/14	0.02	0/8	ND
TNT	μg/L	0.98/0.03	2/14	0.05-0.30	0/8	ND

[a] This table presents information discussed in Section 2.4.2. Samples from Burgermeister Spring were analyzed for radionuclides, metals, inorganic anions, volatile and semivolatile organic compounds, PCBs, pesticides, and nitroaromatic compounds. Samples from the lakes were analyzed for radionuclides, metals, inorganic anions, semivolatile organic compounds, PCBs, and nitroaromatic compounds. The listed contaminants include all those for which the measurement was above the detection limit in at least one sample for each contaminant group.

[b] For cases in which two detection limits are shown for a single contaminant, that contaminant has been analyzed by two different methods at various times of the characterization effort; values reported below the standard detection limit indicate that the method or instrument provided more sensitivity than required for that analysis.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample.

[d] ND = not detected; a hyphen indicates that the contaminant was not assayed.

TABLE 2.10 Characterization Results for Surface Water Discharges and Springs in the Southeast Drainage[a]

		Surface Water		
Contaminant	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]
Radionuclides				
Gross alpha	pCi/L	1.0	50/55	9.0-1,000
Gross beta	pCi/L	1.0	22/22	23-370
Radium-226	pCi/L	1.0	0/8	ND[d]
Radium-228	pCi/L	1.0	0/4	ND
Thorium-230	pCi/L	1.0	0/8	ND
Thorium-232	pCi/L	1.0	0/8	ND
Uranium, total	pCi/L	0.68/1.0	116/120	16-1,200
Metals				
Aluminum	µg/L	200	2/9	210-240
Antimony	µg/L	60	3/8	70-76
Arsenic	µg/L	10	0/8	ND
Barium	µg/L	200	1/10	110
Beryllium	µg/L	5	0/7	ND
Cadmium	µg/L	5	0/7	ND
Calcium	µg/L	5,000	7/9	47,000-74,000
Chromium	µg/L	10	7/7	14-26
Cobalt	µg/L	50	0/7	ND
Copper	µg/L	25	0/10	ND
Iron	µg/L	100	5/9	120-310
Lead	µg/L	5	1/7	15
Lithium	µg/L	50	0/37	ND
Magnesium	µg/L	5,000	8/9	9,300-16,000
Manganese	µg/L	15	3/9	18-57
Mercury	µg/L	0.2	0/7	ND
Molybdenum	µg/L	40	5/7	23-42
Nickel	µg/L	40	0/8	ND
Potassium	µg/L	5,000	0/8	ND
Selenium	µg/L	5	0/7	ND
Silver	µg/L	10	2/7	4.0-6.0
Sodium	µg/L	5,000	7/9	8,300-22,000
Thallium	µg/L	10	0/8	ND
Uranium, total	µg/L	1.5	116/120	23-1,800
Vanadium	µg/L	50	0/7	ND
Zinc	µg/L	20	3/9	27-31
Inorganic anions				
Chloride	mg/L	0.25/0.2	26/26	1.3-26
Fluoride	mg/L	0.25/0.6	15/24	0.30-0.60
Nitrate	mg/L	0.25/0.1	58/67	1.5-260
Sulfate	mg/L	1.0	26/26	26-210

TABLE 2.10 (Cont.)

	Surface Water			
Contaminant	Unit	Detection Limit[b]	Detection Frequency	Concentration Range[c]
Nitroaromatic compounds				
DNB	µg/L	0.51/0.09	1/41	0.81
2,4-DNT	µg/L	0.44/0.03	3/41	0.89-11
2,6-DNT	µg/L	0.48/0.01	3/41	0.40-11
NB	µg/L	0.54/0.03	1/41	0.87
TNB	µg/L	0.47/0.03	2/40	0.22-0.84
TNT	µg/L	0.98/0.03	8/41	0.98-110

[a] This table presents information discussed in Section 2.4.2. Surface water samples from NPDES sampling locations NP-0001 and NP-0005 and from springs in the Southeast Drainage were analyzed for radionuclides, metals, inorganic anions, volatile and semivolatile organic compounds, PCBs, pesticides, and nitroaromatic compounds. The listed contaminants are those for which the measurement was above the detection limit in at least one sample for each contaminant category.

[b] For cases in which two detection limits are shown for a single contaminant, that contaminant has been analyzed by two different methods at various times of the characterization effort; values reported below the standard detection limit indicate that the method or instrument provided more sensitivity than required for that analysis.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample.

[d] ND = not detected.

TABLE 2.11 Characterization Results for Background Surface Water Locations[a]

Contaminant	Unit	Detection Limit[b]	Lake 10 Detection Frequency	Lake 10 Concentration Range[c]	Dardenne Creek Detection Frequency	Dardenne Creek Concentration Range[c]	Concentration at USGS Locations[d]
Radionuclides							
Gross alpha	pCi/L	1.0-3.0	1/8	4.4	2/8	4.1-5.0	NA[e]
Gross beta	pCi/L	2.0	3/8	4.0-7.1	4/8	5.4-13	NA
Radium-226	pCi/L	1.0	0/8	ND[e]	0/8	ND	NA
Radium-228	pCi/L	1.0	0/1	ND	0/1	ND	NA
Thorium-230	pCi/L	1.0	0/8	ND	2/8	1.6-2.4	NA
Thorium-232	pCi/L	1.0	0/8	ND	0/8	ND	NA
Uranium, total	pCi/L	0.68/1.0	6/14	0.68-13	3/14	1.1-8.2	3
Metals							
Aluminum	µg/L	200	0/1	ND	0/1	ND	<10
Antimony	µg/L	60	0/1	ND	0/1	ND	NA
Arsenic	µg/L	10	1/1	24	0/1	ND	<1
Barium	µg/L	200	0/1	ND	0/1	ND	100-130
Beryllium	µg/L	5	0/1	ND	0/1	ND	<0.5
Cadmium	µg/L	5	0/1	ND	0/1	ND	<1
Calcium	µg/L	5,000	1/1	20,000	1/1	52,000	99,000
Chromium	µg/L	10	0/1	ND	1/1	25	<1
Cobalt	µg/L	50	0/1	ND	0/1	ND	<3
Copper	µg/L	25	-[e]	-	0/1	ND	<10
Iron	µg/L	100	0/1	ND	1/1	570	10
Lead	µg/L	5	0/1	ND	0/1	ND	<10
Lithium	µg/L	50	0/1	ND	0/1	ND	9
Magnesium	µg/L	5,000	0/1	ND	1/1	9,400	25,000
Manganese	µg/L	15	0/1	ND	1/1	320	96-250
Mercury	µg/L	0.2	2/2	0.40-1.6	0/1	ND	NA
Molybdenum	µg/L	40	0/1	ND	-	-	<10
Nickel	µg/L	40	-	-	0/1	ND	<10
Potassium	µg/L	5,000	0/1	ND	0/1	ND	5.5
Selenium	µg/L	5	0/1	ND	0/1	ND	<1
Silver	µg/L	10	0/1	ND	0/1	ND	<1
Sodium	µg/L	5,000	0/1	ND	1/1	12,000	36
Thallium	µg/L	10	-	-	0/1	ND	NA
Uranium, total	µg/L	1.5	6/14	1.0-19	3/14	1.6-12	4.4

TABLE 2.11 (Cont.)

Contaminant	Unit	Detection Limit[b]	Lake 10 Detection Frequency	Lake 10 Concentration Range[c]	Dardenne Creek Detection Frequency	Dardenne Creek Concentration Range[c]	Concentration at USGS Locations[d]
Metals (cont.)							
Vanadium	µg/L	50	0/1	ND	0/1	ND	<6
Zinc	µg/L	20	0/1	ND	0/1	ND	<3, 6
Inorganic anions							
Chloride	mg/L	0.25/0.2	9/9	6.5-13	9/9	6.3-97	42
Fluoride	mg/L	0.25/0.6	4/9	0.30-0.35	5/9	0.30-0.56	0.3
Nitrate	mg/L	0.25/0.1	3/9	0.17-0.90	7/9	0.30-5.0	1.3
Sulfate	mg/L	1.0/1.0	9/9	4.3-11	9/9	7.2-120	76
Nitroaromatic compounds							
DNB	µg/L	0.51/0.09	1/2	0.27	0/2	ND	NA
2,4-DNT	µg/L	0.44/0.03	1/2	0.06	0/2	ND	NA
2,6-DNT	µg/L	0.48/0.01	0/2	ND	0/2	ND	NA
NB	µg/L	0.54/0.03	0/2	ND	0/2	ND	NA
TNB	µg/L	0.47/0.03	0/2	ND	0/2	ND	NA
TNT	µg/L	0.98/0.03	0/2	ND	0/2	ND	NA

[a] This table presents information discussed in Section 2.4.2; the listed contaminants include all those for which an analysis was conducted. Although Lake 10 and Dardenne Creek are not within drainage areas impacted by the site, they may have been impacted by releases from the adjacent Army property. Hence, the concentrations of chemical contaminants at these locations cannot be considered true background values.

[b] For cases in which two detection limits are shown for a single contaminant, that contaminant has been analyzed by two different methods at various times of the characterization effort; values reported below the standard detection limit indicate that the method or instrument provided more sensitivity than required for that analysis.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample.

[d] Source: Kleeschulte and Cross (1990). Sampling locations are in the vicinity of the Weldon Spring site. Values are not available for the nitroaromatic compounds because these compounds are anthropogenic and are therefore not present in background environmental samples.

[e] NA = not available; ND = not detected; a hyphen indicates that the contaminant was not assayed.

TABLE 2.12 Characterization Results for Sediment in the Southeast Drainage[a]

Contaminant	Unit	Sediment: Detection Limit[b]	Sediment: Detection Frequency	Sediment: Concentration Range[c]
Radionuclides				
Radium-226[d]	pCi/g	0.2-1.5	78/78	0.71-220
Radium-228[d]	pCi/g	0.2-1.5	77/78	0.43-480
Thorium-228	pCi/g	1.0	2/4	1.4-2.6
Thorium-230[d]	pCi/g	0.2-1.5	10/11	1.5-10,000
Thorium-232	pCi/g	1.0	3/6	1.1-1.7
Uranium, total[d]	pCi/g	0.3-3.0	80/80	3.0-1,500
Metals				
Aluminum	mg/kg	20	8/8	3,200-22,000
Antimony	mg/kg	6	0/8	ND[e]
Arsenic	mg/kg	1	7/8	8.1-14
Barium	mg/kg	20	8/8	170-330
Beryllium	mg/kg	0.5	0/8	ND
Cadmium	mg/kg	0.5	0/8	ND
Calcium	mg/kg	500	8/8	2,500-13,000
Chromium	mg/kg	1	8/8	9.0-25
Cobalt	mg/kg	5	8/8	8.0-37
Copper	mg/kg	2.5	8/8	7.0-170
Iron	mg/kg	10	8/8	8,600-31,000
Lead	mg/kg	0.5	8/8	11-48
Lithium	mg/kg	5	-[e]	-
Magnesium	mg/kg	500	8/8	660-6,400
Manganese	mg/kg	1.5	8/8	750-6,500
Mercury	mg/kg	0.1	0/8	ND
Molybdenum	mg/kg	4	-	-
Nickel	mg/kg	4	8/8	21-66
Potassium	mg/kg	500	7/8	560-2,500
Selenium	mg/kg	0.5	0/8	ND
Silver	mg/kg	1	0/8	ND
Sodium	mg/kg	500	0/8	ND
Thallium	mg/kg	1	0/8	ND
Uranium, total	mg/kg	0.4-4.4	80/80	4.4-2,200
Vanadium	mg/kg	5	8/8	14-75
Zinc	mg/kg	2	8/8	31-220

See next page for footnotes.

TABLE 2.12 (Cont.)

[a] This table presents information discussed in Section 2.4.2. Sediment samples were analyzed for radionuclides, metals, semivolatile organic compounds, PCBs, and nitroaromatic compounds. The listed contaminants are all those for which the measurement was above the detection limit in at least one sample for each contaminant group.

[b] For cases in which two detection limits are shown for a single contaminant, that contaminant was analyzed by two different methods at various times of the characterization effort. The range of detection limits accounts for variations in instrument calibration among different studies.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample. Concentrations are reported as dry weights.

[d] Includes historical data from Boerner (1986) and Deming (1986).

[e] ND = not detected; a hyphen indicates that the contaminant was not assayed.

TABLE 2.13 Characterization Results for Sediment in the Busch Wildlife Area[a]

Contaminant	Unit	Detection Limit	Burgermeister Spring		Lakes 34, 35, and 36[c]		Lake 37	
			Detection Frequency	Concentration Range[b]	Detection Frequency	Concentration Range[b]	Detection Frequency	Concentration Range[b]
Radionuclides								
Radium-226	pCi/g	0.4-1.0	1/1	0.49	23/39	0.5-4.8	1/4	0.5
Thorium-228	pCi/g	1.0	-[d]	-	3/9	1.1-1.3	2/3	1.1
Thorium-230	pCi/g	1.0	-	-	15/24	1.1-2.7	2/3	1.1-1.4
Thorium-232	pCi/g	0.4-1.0	1/1	0.73	18/39	0.49-2.5	3/4	0.75-1.2
Uranium, total	pCi/g	1.0	3/3	1.4-2.6[e]	81/87	1.0-240	6/7	1.1-1.8
Metals								
Aluminum	mg/kg	20	3/3	5,500-6,400	9/9	5,100-14,000	3/3	6,200-12,000
Antimony	mg/kg	6	0/3	ND[d]	0/9	ND	0/3	ND
Arsenic	mg/kg	1	3/3	12-19	9/9	3.0-10	3/3	2.0-5.0
Barium	mg/kg	20	3/3	100-120	9/9	110-160	3/3	130-150
Beryllium	mg/kg	0.5	3/3	1.0	0/9	ND	0/3	ND
Cadmium	mg/kg	0.5	0/3	ND	0/9	ND	0/3	ND
Calcium	mg/kg	500	3/3	2,400-5,600	9/9	1,100-4,000	3/3	600-1,200
Chromium	mg/kg	1	3/3	13-17	9/9	7.0-15	3/3	8.0-14
Cobalt	mg/kg	5	3/3	12-13	9/9	7.0-18	2/3	7.0-8.0
Copper	mg/kg	2.5	3/3	5.0-6.0	9/9	8.0-13	3/3	9.0-13
Iron	mg/kg	10	3/3	18,000-20,000	9/9	11,000-14,000	3/3	9,300-12,000
Lead	mg/kg	0.5	3/3	25-35	9/9	9.0-19	3/3	5.0-17
Magnesium	mg/kg	500	3/3	750-860	9/9	890-1,700	2/3	740-1,200
Manganese	mg/kg	1.5	3/3	650-950	9/9	280-1,100	3/3	440-720
Mercury	mg/kg	0.1	0/3	ND	0/9	ND	0/3	ND
Nickel	mg/kg	4	3/3	9.0-11	9/9	8.0-13	3/3	8.0-11
Potassium	mg/kg	500	0/3	ND	6/9	690-1,400	2/3	700-850

TABLE 2.13 (Cont.)

Contaminant	Unit	Detection Limit	Burgermeister Spring		Lakes 34, 35, and 36[c]		Lake 37	
			Detection Frequency	Concentration Range[b]	Detection Frequency	Concentration Range[b]	Detection Frequency	Concentration Range[b]
Metals (cont.)								
Selenium	mg/kg	0.5	0/3	ND	0/9	ND	0/3	ND
Silver	mg/kg	1	0/3	ND	0/9	ND	2/3	3.0-35
Sodium	mg/kg	500	1/3	570	0/9	ND	0/3	ND
Thallium	mg/kg	1	0/3	ND	0/9	ND	0/3	ND
Uranium, total	mg/kg	1.5	3/3	2.1-3.8	81/87	1.6-350	6/7	1.6-2.6
Vanadium	mg/kg	5	3/3	27-38	9/9	20-29	3/3	18-34
Zinc	mg/kg	2	3/3	33-40	9/9	24-46	3/3	13-30
PCBs	mg/kg	0.16	0/9	ND	1/27	0.20	0/9	ND

[a] This table presents information discussed in Section 2.4.2. Sediment samples were analyzed for radionuclides, metals, semivolatile organic compounds, PCBs, and nitroaromatic compounds. The listed contaminants are those for which the measurement was above the detection limit in at least one surface sample for each contaminant group. Historical data from Boerner (1986) are included for radionuclides.

[b] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample. Concentrations are reported as dry weights.

[c] For purposes of presentation, shoreline soil is addressed with sediment; uranium was the only radionuclide detected above background levels; concentrations of total uranium ranged from 26 to 72 pCi/g.

[d] ND = not detected; a hyphen indicates that the contaminant was not assayed.

[e] From historical data, a measurement of 44 pCi/g (65 mg/kg) was reported for total uranium in Burgermeister Spring (Boerner 1986).

TABLE 2.14 Characterization Results for Groundwater[a]

Contaminant	Unit	Detection Limit[b]	Wells in the Weathered and Transitional Zones: Detection Frequency	Wells in the Weathered and Transitional Zones: Concentration Range[c]	Wells in the Competent (Unweathered) Zone: Detection Frequency	Wells in the Competent (Unweathered) Zone: Concentration Range[c]	Local Background Level[d]
Radionuclides							
Radium-226	pCi/L	1.0	10/222	1.1	0/20	ND[e]	0.2-5.2
Thorium-230	pCi/L	1.0	10/222	1.1-4.3	5/20	1.2-4.6	0.0-6.4
Thorium-232	pCi/L	1.0	4/222	1.0-1.9	0/20	ND	0.3-8.5
Uranium, total	pCi/L	0.68/1.0	202/291	0.68-60	86/103	0.68-28	0.5-2.9
Metals							
Aluminum	µg/L	200	9/71	230-1,400	0/18	ND	<10
Antimony	µg/L	60	15/73	60-250	0/18	ND	NA[e]
Arsenic	µg/L	10	7/59	12-38	7/18	15-34	<1
Barium	µg/L	200	41/87	200-2,200	3/18	84-250	180
Beryllium	µg/L	5	0/58	ND	0/18	ND	<0.5
Cadmium	µg/L	5	1/58	6.0	0/18	ND	<1
Calcium	µg/L	5,000	87/88	14,000-910,000	18/18	5,700-91,000	82,000
Chromium	µg/L	10	36/86	14-140	5/18	10-37	<1
Cobalt	µg/L	50	0/61	ND	0/18	ND	<3
Copper	µg/L	25	5/82	26-75	0/18	ND	<2
Iron	µg/L	100	74/87	10-2,500	18/18	27-190	120
Lead	µg/L	5	8/69	5.6-24	10/18	8-40	<5
Lithium	µg/L	50	11/62	64-1,000	7/18	2-36	13
Magnesium	µg/L	5,000	88/88	7,700-290,000	17/18	5,700-85,000	47,000
Manganese	µg/L	15	44/86	15-550	15/19	18-510	104
Mercury	µg/L	0.2	7/61	0.21-44	2/18	0.26-0.35	NA
Molybdenum	µg/L	40	34/60	13-82	18/18	12-1,100	10
Nickel	µg/L	40	17/80	40-170	0/18	ND	3
Potassium	µg/L	5,000	8/60	5,700-17,000	4/18	12,000-33,000	3,300
Selenium	µg/L	5	10/60	7.7-40	4/18	13-29	<1
Silver	µg/L	10	7/62	11-34	0/18	ND	<1
Sodium	µg/L	5,000	87/88	5,500-29,000	17/18	5,200-56,000	30,000
Thallium	µg/L	10	1/58	15	5/18	17-93	NA
Uranium, total	µg/L	1.0/1.5	202/291	1.0-88	86/103	1.0-41	0.7-4.3
Vanadium	µg/L	50	2/62	64-65	0/18	ND	<6
Zinc	µg/L	20	43/84	20-120	11/18	24-210	28

TABLE 2.14 (Cont.)

Contaminant	Unit	Detection Limit[b]	Wells in the Weathered and Transitional Zones: Detection Frequency	Wells in the Weathered and Transitional Zones: Concentration Range[c]	Wells in the Competent (Unweathered) Zone: Detection Frequency	Wells in the Competent (Unweathered) Zone: Concentration Range[c]	Local Background Level[d]
Inorganic anions							
Chloride	mg/L	0.25/0.2	341/344	0.53-5,000	117/188	0.7-6,500	8.4
Fluoride	mg/L	0.25/0.6	153/317	0.05/230	47/117	0.14-0.5	0.5
Nitrate as N	mg/L	0.25/0.1	289/312	0.08-12,000	39/117	0.09-140	1.6
Nitrite as N	mg/L	0.25	-[e]	-	-	-	0.04
Sulfate	mg/L	1.0/1.0	316/317	1.9-1,800	118/118	1.5-180	32
Nitroaromatic compounds							
DNB	µg/L	0.51/0.09	41/322	0.11-12	0/112	ND	NA
2,4-DNT	µg/L	0.44/0.03	77/315	0.06-390	2/111	0.58-0.59	NA
2,6-DNT	µg/L	0.48/0.01	107/320	0.02-290	2/112	1.7-2.9	NA
NB	µg/L	0.54/0.03	24/322	0.41-22	9/112	0.33-6	NA
TNB	µg/L	0.47/0.03	118/319	0.02-89	21/113	0.02-17	NA
TNT	µg/L	0.98/0.03	55/321	0.06-65	2/112	1.5-7.8	NA

[a] This table presents information discussed in Section 2.4.3.

[b] For cases in which two detection limits are shown for a single contaminant, that contaminant has been analyzed by two different methods at various times of the characterization effort; values reported below the standard detection limit indicate that the method or instrument provided more sensitivity than required for that analysis.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample.

[d] For radionuclides, the data are from 60 private drinking water wells in the vicinity of the site that are completed in the deep, productive aquifer (Roberts 1991). For chemicals except manganese, single values are the upper 95% confidence limits determined for the Burlington-Keokuk Limestone; manganese is reported as the mean (Kleeschulte and Cross 1990). Values are not available for nitroaromatic compounds because these compounds are anthropogenic and are therefore not present in background environmental samples.

[e] ND = not detected; a hyphen indicates that the contaminant was not measured; NA = not available.

TABLE 2.15 Uranium-238 Concentrations in Fish from Lakes 34, 35, 36, and 37[a]

	Concentration[b] (pCi/g wet weight)			
Sample	Lake 34	Lake 35	Lake 36	Lake 37
Catfish				
Whole	-	0.18 0.35[c]	0.94	0.006
Cake	-	-	-	-
Fillet	ND	ND	0.002	ND
Bass				
Whole	0.012	0.012 0.013[d]	0.15	ND
Cake	-	-	-	-
Fillet	ND	0.002 0.002[c]	0.003 0.002	ND
Sunfish				
Whole	0.018	0.14	0.26	ND
Cake	0.021	0.022	-	-
Fillet	ND	0.004 0.002[d]	0.021	ND
Crappie				
Whole	-	0.016	0.035	-
Cake	-	0.004	-	-
Fillet	-	0.002 0.001[d]	ND	-

[a] This table presents information discussed in Section 2.4.5.

[b] A hyphen indicates that the sample was not available; ND indicates that uranium-238 was not detected above the detection limit of 0.001 pCi/g.

[c] Second sample.

[d] Duplicate analysis from same sample.

Source: Meyer (1991).

TABLE 2.16 Radioactive Contaminants of Concern for the Human Health Assessment[a]

Contaminant	On-Site Concentration Range[b]			Off-Site Concentration Range[c]	
	Soil (pCi/g)	Surface Water (pCi/L)	Raffinate Pit Sludge (pCi/g)	Surface Water (pCi/L)	Sediment (pCi/g)
Actinium-227[d]	0.006-44	-[e]	2.8-990	-	-
Lead-210[d]	0.4-450	-	1.0-1,700	-	-
Protactinium-231[d]	0.01-87	-	3.6-1,200	-	-
Radium-226	0.4-450	3.4-130	1.0-1,700	ND[e]	0.7-220
Radium-228	0.4-150	1.5-25	4.0-1,400	ND	0.4-480
Radon-220[f]	-	-	-	-	-
Radon-222[g]	-	-	-	-	-
Thorium-230	0.3-97	1.4-760	8.0-34,000	1.0-8.0	1.5-10,000
Thorium-232[h]	0.4-150	0.2-7.6	3.0-1,400	ND	0.7-2.5
Uranium-235[i]	0.01-110	1.3-60	0.2-78	0.09-27	0.02-33
Uranium-238	0.3-2,300	28-1,300	4.9-1,700	2.0-590	0.5-720

[a] This table presents information discussed in Section 2.5.

[b] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample. For surface water, combined values for the raffinate pits and NPDES sampling locations NP-0002, NP-0003, and NP-0004. Sludge concentrations are reported as wet weights (the sludge contains about 73% water by weight); soil concentrations are reported as dry weights.

[c] The concentration range is for detected values only; a single value is given if the contaminant was detected in only one sample. Combined values for Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage. Sediment concentrations are reported as dry weights. The concentrations of radionuclides at off-site soil vicinity properties are summarized in Table 3.6.

[d] The concentrations of actinium-227, lead-210, and protactinium-231 for site soil and raffinate pit sludge were determined from the radiological source term analysis (Table 2.3).

[e] A hyphen indicates that the contaminant was not measured or calculated from the radiological source term analysis; ND = not detected.

[f] Radon-220 is a contaminant of concern only for the chemical plant buildings.

[g] Radon-222 is a contaminant of concern for the chemical plant buildings and outdoor air. The concentration of radon-222 and its short-lived decay products in outdoor air was calculated from the concentration of radium-226 in soil (Section 3.3.4).

[h] Consistent with the radiological source term analysis, thorium-232 was assumed to be in secular equilibrium with radium-228 for site soil (Section 2.3.2.1).

[i] The ratio of uranium-238:uranium-235:uranium-234 in surface water, sludge, and sediment was assumed to be 1:0.046:1.

TABLE 2.17 Chemical Contaminants of Concern for the Human Health Assessment[a]

Contaminant	On-Site Concentration Range[b]			Off-Site Concentration Range[c]	
	Soil (mg/kg)	Surface Water (µg/L)	Raffinate Pit Sludge (mg/kg)	Surface Water (µg/L)	Sediment (mg/kg)
Metals					
Antimony	6.4-110	65-400	6.0-87	70-76	ND[d]
Arsenic	1.3-130	12-120	3.1-1,100	12-29	3.0-19
Barium	25-5,200	ND	20-7,700	78-110	100-330
Beryllium	0.51-5.5	7.0-9.0	0.59-25	ND	ND
Cadmium	0.51-11	37	0.94-14	ND	ND
Chromium III	2.0-280	28-170	4.5-150	13-23	6.3-23
Chromium VI	0.22-31	3.1-19	0.50-17	1.4-2.6	0.70-2.5
Cobalt	2.8-110	ND	5.1-44	ND	7.0-37
Copper	3.6-460	30-45	3.7-510	ND	5.0-170
Lead	1.3-1,900[e]	22-450	2.1-640	9.5-15	9.0-48
Lithium	5.3-71	61-4,500	5.0-120	ND	-[d]
Manganese	3.3-13,000	16-33	25-3,000	18-870	280-6,500
Mercury	0.11-2.1	0.29-0.36	0.10-15	0.35-1.3	ND
Molybdenum	4.1-120	690-4,100	16-1,600	22-42	-
Nickel	5.6-270	47-170	3.3-8,800	ND	8.0-66
Selenium	0.63-47	7.5-220	2.7-81	ND	ND
Silver	0.92-13	25-40	1.0-5.0	4.0-6.0	ND
Thallium	1.0-80	ND	1.1-58	33	ND
Uranium, total	0.9-6,900	4.4-5,200	15-5,100	6.0-1,800	1.6-2,200
Vanadium	7.2-380	90-2,100	26-8,700	ND	14-75
Zinc	6.1-1,100	26-60	7.9-1,600	21-78	24-220
Inorganic anions					
Fluoride	1.3-45	230-19,000	3.2-170	170-600	-
Nitrate	0.54-3,800	190-7,200,000	0.6-160,000	300-260,000	-
Nitrite	1.5-29	-	1.0-1,600	-	-
Asbestos[f]	ND	-	-	-	-
PAHs[g]					
Acenaphthene	1.9	-	ND	-	ND
Anthracene	3.4	-	ND	-	ND
Benz(a)anthracene	0.41-8.2	-	ND	-	ND
Benzo(b)fluoranthene	4.6	-	ND	-	ND
Benzo(k)fluoranthene	3.9	-	ND	-	ND
Benzo(g,h,i)perylene	2.1	-	ND	-	ND
Benzo(a)pyrene	5.1	-	ND	-	ND
Chrysene	0.39-8.0	-	ND	-	ND
Fluoranthene	0.58-11	-	ND	-	ND
Fluorene	1.6	-	ND	-	ND
Indeno(1,2,3-cd)pyrene	3.2	-	ND	-	ND
2-Methylnaphthalene	0.52-4.6	-	ND	-	ND
Naphthalene	1.8	-	ND	-	ND
Phenanthrene	0.42-11	-	ND	-	ND
Pyrene	0.35-19	-	ND	-	ND

TABLE 2.17 (Cont.)

Contaminant	On-Site Concentration Range[b]			Off-Site Concentration Range[c]	
	Soil (mg/kg)	Surface Water (µg/L)	Raffinate Pit Sludge (mg/kg)	Surface Water (µg/L)	Sediment (mg/kg)
PCBs	0.18-12	-	0.15-11	ND	0.2
Nitroaromatic compounds					
DNB	1.0-3.8	ND	ND	0.18-0.81	ND
2,4-DNT	0.83-6.3	ND	ND	0.3-11	ND
2,6-DNT	1.6-3.5	ND	ND	0.19-18	ND
NB	1.6-3.8	ND	ND	0.87	ND
TNB	0.63-5.7	0.04-1.4	ND	0.02-0.84	ND
TNT	1.3-32[h]	0.80-7.5	ND	0.05-110	ND

[a] This table presents information discussed in Section 2.5.

[b] The concentration range is for detected values only; a single value is given if the contaminant was measured in only one sample. For surface water, the combined value for the raffinate pits and NPDES sampling locations NP-0002, NP-0003, and NP-0004. Sludge concentrations are reported as wet weights (the sludge contains about 73% water by weight); soil concentrations are reported as dry weights.

[c] The concentration range is for detected values only; a single value is given if the contaminant was measured in only one sample. For surface water and sediment, the combined value for Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage. Sediment concentrations are reported as dry weights.

[d] ND = not detected; a hyphen indicates that the contaminant was not assayed.

[e] One high sample was measured at 43,000 mg/kg.

[f] Asbestos is a contaminant of concern only for the chemical plant buildings.

[g] Although not technically considered PAHs, 2-methylnaphthalene and naphthalene are included in this category for presentational purposes.

[h] One high sample was measured at 650,000 mg/kg.

3 HUMAN EXPOSURE ASSESSMENT

The purpose of an exposure assessment is to estimate the nature and magnitude of potential exposures associated with a contaminated site. The components of an exposure assessment include the contaminants of concern, the exposure setting, potential human exposure pathways, and the estimated exposure point concentrations and contaminant intakes. The contaminants of concern for the Weldon Spring site are identified in Section 2.5, and the exposure setting is characterized in Section 3.1. Potential human exposure pathways are described in Section 3.2, including a discussion of source areas, contaminant fate and transport, and locations and routes of potential exposure. Estimated exposure point concentrations for each medium of concern — i.e., soil, sediment, sludge, surface water, and air; fish and game animals; and air and residues in the chemical plant buildings — are presented in Section 3.3. Associated contaminant intakes by human receptors and exposures across multiple pathways are addressed in Section 3.4.

3.1 CHARACTERIZATION OF EXPOSURE SETTING

The following discussion provides a baseline for assessing potential impacts to the various environmental resources associated with the site and identifies exposure pathways for potential human receptors. The resources at the site are addressed in greater detail in the RI. The exposure setting for the Weldon Spring site is characterized by both the natural environment at the site (Section 3.1.1) and the local land use and demography (Section 3.1.2). Except where otherwise noted, this characterization is based on information taken from the draft EIS for the site (DOE 1987), the RI/FS-EIS work plan (Peterson et al. 1988), the FS for the quarry bulk waste remedial action (DOE 1992a), and the RI for the chemical plant area (DOE 1992c).

3.1.1 Environmental Setting

3.1.1.1 Climate and Meteorology

The St. Louis area has a modified continental climate characterized by moderately cold winters and warm summers. Temperatures measured from 1958 through 1988 ranged from -28°C (-18°F) to 42°C (107°F). Evapotranspiration and precipitation in the area generally balance each other. Annual precipitation typically totals approximately 86 cm (34 in.), of which about 25 cm (10 in.) occurs in the spring. Thunderstorms usually occur between 40 and 50 days per year; as much as 25 cm (10 in.) of rain has been recorded in 24 hours during a heavy storm. Winter is the driest season, with precipitation averaging about 15 cm (6 in.). From 1937 through 1988, annual snowfall in the area averaged 50 cm (20 in.); most snowfalls occur from December through March.

Meteorological data were collected at the Weldon Spring site between 1983 and 1985. These data exhibit similar patterns to data measured at the nearby Lambert-St. Louis

International Airport, located about 32 km (20 mi) east-northeast of the site, with one exception. Wind speeds are lower at the site because of surface friction effects that result from the heavy vegetation and relatively rough terrain in the vicinity. The prevailing winds are from the south, and the annual average wind speed is 2.7 m/s (6 mph). Wind erosion is expected to be negligible at the site under baseline conditions because a wind speed high enough to cause erosion rarely occurs (a wind speed of 12 m/s [27 mph] was measured only once during the measurement period).

3.1.1.2 Topography and Surface Hydrology

The Weldon Spring site is located in the southwest uplands of St. Charles County, which is bounded by the Mississippi River to the north and the Missouri River to the south; the county land is about half uplands and half floodplain. Site elevations range from approximately 185 m (610 ft) mean sea level (MSL) near the northern edge to about 205 m (670 ft) MSL near the southern edge. This elevation precludes any impact from flooding of either the Missouri or Mississippi rivers. A small portion of land in the northern area of the site is within the 100-year floodplain of nearby Schote Creek. Except for the embankments built around the raffinate pits, the land surface within the site is gently sloping. Gently rolling topography characterizes the area to the north and west, whereas wooded ravines characterize the terrain to the south and east.

The site straddles the surface water drainage divide that separates the Mississippi River and Missouri River watersheds. Surface runoff eventually reaches either the Mississippi River, about 23 km (14 mi) north of the site, or the Missouri River, about 2.4 km (1.5 mi) to the southeast. Runoff north of the divide (which accounts for runoff from most of the site) flows into off-site intermittent and perennial streams, which subsequently flow into lakes in the Busch Wildlife Area and eventually discharge into the Mississippi River near Seeburger, about 32 km (20 mi) northwest of St. Louis. Runoff south of the divide flows to the Missouri River through the Southeast Drainage, a natural channel with intermittent flow (Figure 1.6).

3.1.1.3 Soil and Geology

More than 20 m (64 ft) of alluvial deposits blanket the bedrock in the Missouri River valley. Silt loam is the predominant soil type in the area surrounding the site, both in gently rolling terrain to the north and in more hilly terrain to the south. Approaching the Missouri River, the soil types in the floodplain include silt, silty clay, silty loam, and clay loam. Soils of the Harvester-Urban Complex are present at the Weldon Spring site; these soils have a moderate permeability and high water content.

Overlying the bedrock at the site are unconsolidated sedimentary units that range in thickness from 5 to 18 m (16 to 59 ft). The overburden has been divided into five layers according to physical characteristics: topsoil and loess (which is silt and clayey silt), Ferrelview clay, clay till, basal till, and residuum (extensively weathered limestone). Beneath this unconsolidated material lies the Mississippian Burlington-Keokuk Limestone, which is about

40 to 50 m (140 to 160 ft) thick at the site. Borehole data indicate that this limestone formation can be divided into an upper weathered zone that ranges from about 3 to 15 m (10 to 50 ft) thick and a lower competent (unweathered) zone that ranges from about 27 to 46 m (90 to 150 ft) thick. The lithologic contact between these two zones is gradational rather than abrupt and is often difficult to distinguish.

Field data indicate that fractures within the weathered limestone appear to be predominantly horizontal and widely spaced. Solution features have also been found in the weathered zone, but they are either partially or completely filled with clay-sized material. Although some void space exists in the uppermost bedrock, the voids are generally isolated and display limited vertical or lateral continuity. The number of horizontal and vertical fractures is significantly lower in the underlying competent bedrock than in the weathered zone. Field data from borehole packer testing of the saturated bedrock also indicate a decrease in hydraulic conductivity with depth, which is attributed to lower weathering and related solution activity (Bechtel National 1987). A more detailed description of site geology is provided in Section 4.3 of the RI.

Karst features are present in the vicinity of the site, although the site itself is not considered to be situated in an area of collapse potential. The nearest caves are about 8 km (5 mi) from the study area, and the nearest sinkhole is 1.6 km (1 mi) away (Garstang 1991). Site geology and groundwater flow were studied in coordination with the state of Missouri Department of Natural Resources, Division of Geology and Land Survey. This testing did not identify void space in the overburden or soil material, and voids in the limestone bedrock were few and small (90% of the void space was within the upper 3 m [10 ft] of bedrock). No open subsurface networks were identified. Dye-tracing tests indicate that voids do exist, e.g., in the weathered portion of the limestone bedrock, but results suggest that they are isolated (Garstang 1991).

3.1.1.4 Groundwater

Groundwater at the site is present as (1) perched groundwater in the unconsolidated deposits (e.g., near the raffinate pits); (2) a shallow unconfined aquifer in the Burlington-Keokuk Limestone; and (3) a deep confined aquifer in the St. Peter Sandstone.

The shallow unconfined aquifer consists primarily of saturated rocks of the Mississippian Burlington-Keokuk Limestone and occasionally residuum. The Burlington-Keokuk is subdivided into an upper and a lower zone. The upper zone is the thin, weathered, fractured portion of the limestone, which is characterized by a larger hydraulic conductivity than the thicker, lower zone. Groundwater in the shallow aquifer (Burlington-Keokuk Limestone) appears to flow by diffuse flow (porous media flow), along horizontal bedding planes, and through fractures. Because the intensity of weathering and fracturing decreases with depth (i.e., in the lower portion of the Burlington-Keokuk Limestone), the aquifer becomes more homogeneous, flow paths are more widely spaced, and the influence of vertical fractures is more limited with depth (DOE 1992c). Groundwater off-site flows by diffuse flow and also via certain free-flow

conduits on both sides of the groundwater divide. Discharge points for the conduits are perennial springs such as Burgermeister Spring and springs in the Southeast Drainage.

A map of the water table surface of the shallow aquifer, contoured from measurements of groundwater levels, indicates an east-northeasterly trending groundwater divide that passes within 100 to 200 m (300 to 660 ft) of the southern edge of the site (see Appendix B, Figure B.2). Groundwater north of the divide flows toward Dardenne Creek and the Mississippi River; groundwater south of the divide flows toward the Missouri River. The elevation of the water table fluctuates, but it generally remains within the upper bedrock or weathered residuum at the overburden/bedrock interface.

The deep bedrock aquifer system primarily consists of Ordovician and Upper Cambrian rocks, including formations from the top of the St. Peter Sandstone down through the base of the Potosi Dolomite (Kleeschulte and Emmett 1987). The deep bedrock aquifer system, about 300 m (1,000 ft) thick, is separated from the shallow aquifer by a leaky confining layer that extends from the top of the Maquoketa Shale down through the Joachim Dolomite. The confining sequence is generally about 120 m (400 ft) thick (Kleeschulte and Emmett 1986). Lithologic units of the bedrock aquifers are presented in Figure 4.3-1 of the RI (DOE 1992c). The deep aquifer also has a groundwater divide, which is located north of the site (see Figure 4.6-2 of the RI [1992c]). To the north of the divide, groundwater flows to the northeast and eventually enters the cone of depression produced by municipal pumping wells in Wentzville and O'Fallon. Groundwater to the south of the divide flows to the southeast, and the eventual discharge point is not currently known.

3.1.1.5 Vegetation and Wildlife

Much of the Weldon Spring site (about 66 ha [162 acres]) contains grassland and old-field habitats that are routinely mowed. However, the northern 22 ha (55 acres) of the site contains relatively undisturbed old-field and upland forest habitat. Tree species common to the forested areas include oak, hickory, locust, and cottonwood. Vegetation in the open areas includes goldenrod, ragweed, thistle, and crabgrass. Mammals found in the area include squirrel, deer, fox, opossum, raccoon, skunk, rabbit, and a variety of mice and other rodents. Wild turkeys are present in the northwestern portion of the site. Ash Pond, Frog Pond, and the raffinate pits are frequented by waterfowl, including a variety of ducks and geese. Off-site lakes are stocked with warmwater fish, including carp, sunfish, and largemouth bass. Vegetation and wildlife at the site are discussed in greater detail in Chapter 7.

3.1.2 Land Use and Demography

The Weldon Spring site is located in the western part of the St. Louis metropolitan area in St. Charles County, Missouri. The county population in 1990 was about 210,000, with 24% living in the city of St. Charles. The population in St. Charles County increased by about 48% from 1980 to 1990. The two communities closest to the site are Weldon Spring and Weldon Spring Heights, about 3.2 km (2 mi) to the northeast (Figure 1.1). The combined population of

these two communities is approximately 850. Urban areas occupy about 6% and nonurban areas occupy about 90% of county land; the remaining 4% is dedicated to transportation and water uses. An estimated 4% of the county's nonurban land will be converted to urban uses between 1980 and 2000.

The Weldon Spring site is bordered by the August A. Busch Memorial Wildlife Area to the north, the Weldon Spring Wildlife Area to the south and east, and the U.S. Army Reserve and National Guard Training Area to the west (Figure 1.6). The 2,828-ha (6,987-acre) Busch Wildlife Area and the 2,977-ha (7,356-acre) Weldon Spring Wildlife Area are managed by the Missouri Department of Conservation and are open throughout the year for recreational use. These areas are managed for multiple uses, including timber, fish and wildlife habitat, and recreation; fishing comprises a relatively large portion of the recreational use (Crigler 1992a).

The Weldon Spring Wildlife Area is densely vegetated and has rugged terrain, whereas the Busch Wildlife Area has gently rolling terrain. Some of these areas are densely vegetated, but others have been cleared for recreational use. In addition, some land in these wildlife areas is leased to sharecroppers for agricultural production, with a percentage of the crop left for wildlife consumption. The wildlife areas contain public buildings and an extensive network of roads and trails, and the Busch area contains a shooting range. The district staff for these wildlife areas consists of 18 full-time employees, supplemented by 15 to 20 workers during the summer months (Crigler 1992a).

The Busch and Weldon Spring wildlife areas are open year-round, and the number of annual visitors to both areas totals about 1,200,000 (Crigler 1992b). Because of the popularity of the wildlife areas and the plans for expansion, current land use is expected to continue in the future.

The 670-ha (1,655-acre) U.S. Army Reserve and National Guard Training Area is fenced, and access by the general public is restricted. Throughout the year, the area is occupied by one full-time staff person and numerous part-time and temporary personnel. Approximately 3,300 local Army reservists and 3,400 other military reserve troops may use the area each year. An average of 150 to 400 Army reservists are present on weekends during 35 to 48 weeks of the year for 2-day drill training. Military reserve troops use a limited portion of the area, including the firing range, for training exercises (Daubel 1992).

A state highway maintenance facility is located just east of the site. The facility employs nine full-time staff and one mechanic (Sizemore 1991). These individuals work at the facility and off-site, performing road maintenance activities on nearby state highways.

Francis Howell High School is approximately 1 km (0.6 mi) northeast of the site along State Route 94. The school employs approximately 160 faculty and staff and serves about 1,600 students (Hartwig 1992); the school custodian and his family reside on the school grounds. Students and staff generally spend about 7 to 8 hours per day at the school. The buildings are also used for other activities, such as athletic events and after-school meetings.

About 300 ha (740 acres) of land east of the high school is owned by the University of Missouri. The northern third of this land is being developed into a high-technology research park. A portion of the land adjacent to the school was previously operated by the University of Missouri as the St. Charles County Extension Center. It is currently being used by the high school as an administration annex.

A daycare facility was recently established for children of school district employees in the former Weldon Spring Elementary School located farther east of the high school, about 3.2 km (2 mi) from the site; this facility was designed to accommodate 10 to 45 children between the ages of 5 to 11 years. Children generally attend daycare 8 to 10 hours per day, 5 days per week, during the 3-week vacation period between school quarters, for a total of 12 weeks per year. The upper level of the facility contains an office and meeting area for preschool children and their parents, about 20 of whom attend meetings three nights per week and one Saturday per month. Together, the upper level and the daycare facility employ about 18 staff* (Meyer 1992).

Because the Weldon Spring site is surrounded by large tracts of land owned by the state and federal governments, existing nearby land uses are expected to remain the same during the next 30 years. Therefore, potential future populations are expected to be similar to those identified under current conditions.

3.2 IDENTIFICATION OF EXPOSURE PATHWAYS

Potential human exposure pathways were identified on the basis of the following factors:

- The locations of contaminated sources, the types of contaminants found at the source areas, and potential mechanisms of contaminant releases from these areas;
- The likely fate and transport of the contaminants within or between environmental media;
- Estimated concentrations of contaminants at points of potential human contact (i.e., exposure points) and the associated probable routes of human exposure (e.g., external gamma irradiation, ingestion, inhalation, and dermal contact); and
- The completeness of each exposure pathway — i.e., the presence of a source and a mechanism of contaminant release, an environmental transport medium, a point of human contact with the contaminated source or medium, and a route of human exposure at that point.

*Subsequent to the preparation of this report, the former Weldon Spring Elementary School was destroyed by fire on July 17, 1992.

These factors were considered in developing the exposure assessment for the Weldon Spring site and are addressed in Sections 3.2.1 through 3.2.4.

3.2.1 Source Areas

Past operations and waste-handling practices of the former ordnance works and uranium feed materials plant at the site have resulted in several discrete source areas. These contaminated areas are associated primarily with feed materials plant activities because the ordnance works was extensively decontaminated and dismantled prior to construction of the uranium processing plant. As part of the demolition effort, more than 200 buildings were destroyed, and about 80 t (90 tons) of nitroaromatic compounds and residues were burned. These efforts appear to have removed the major areas of contamination associated with the ordnance works, although some residual nitroaromatic contamination is still present at certain locations. The source areas at the Weldon Spring site have been identified as follows:

- Raffinate pits,
- Ash Pond,
- Frog Pond,
- North Dump,
- South Dump,
- Former coal storage area, and
- Chemical plant buildings and support structures.

Although the delineation between certain source areas and contaminated environmental media may be somewhat inexact (e.g., between the South Dump and nearby contaminated soil), these areas are considered to represent the primary sources of contamination at the Weldon Spring site (Figure 1.5). The source areas are described in detail in Section 5.1 of the RI. General information is summarized in Section 1.2.3 and Chapter 2 of this document, and the types of contaminants found at each area are discussed in Section 3.4. A brief discussion of the origin and nature of these source areas follows.

The four raffinate pits constitute the major source of contamination at the Weldon Spring site, in terms of both volumes and concentrations of radionuclides and chemicals. The pits, which cover about 11 ha (26 acres), were excavated from existing clay to contain residues from the uranium and thorium processing operations. Natural clay serves as the bottom material for each pit, and surface water is contained by surrounding berms. During the operational period of the plant, solids settled to the bottom of the pits and surface water was decanted to the process sewer system for discharge to the Southeast Drainage; the decant lines have since been plugged. Due to the net balance between precipitation and evapotranspiration

in the St. Louis area, the pits still contain approximately the same amount of surface water as was present at the time of plant closure.

Ash Pond is a marsh-like area located in a 4.5-ha (11-acre) topographic depression at the northwestern portion of the site. During the operational period of the chemical plant, the area was used to dispose of fly ash piped from the coal-fired steam plant; the ash slurry was spread over Ash Pond with earth-moving equipment. Prior to the recent construction of a dike and drainage system at Ash Pond to limit off-site releases of uranium, the area received surface runoff from the northern portion of the site. Standing water is present intermittently at levels that fluctuate according to the amount of precipitation.

Frog Pond is a bermed settling basin that covers about 0.3 ha (0.7 acres) near the eastern boundary of the site. During plant operations, the pond was used to hold surface runoff and discharges from the storm-water sewer system in the area and from the pilot-plant sanitary sewers. Although no radioactive material was stored or buried in the area, Frog Pond may have been impacted by a spill of uranyl nitrate hexahydrate at a northern pilot-plant building (Building 403). The pond currently receives surface runoff and water from storm drains at the northeastern portion of the site and is a perennial water body.

The North Dump is located at the northern boundary of the site and was used to store barrels of radioactive materials and contaminated scrap. The South Dump is located north of the raffinate pits and south of Ash Pond. Portions of this area were used to burn trash, including personal protective equipment. The South Dump was also used to store drums of radioactive material, hoppers and other equipment, and building rubble and other debris. Some of that debris is still present.

The former coal storage area was located north of the facility's steam power plant and was used to stockpile coal. Although the coal pile is no longer present, residual material remains.

The site contains about 40 chemical plant buildings and other structures, 5 of which were major process buildings and 8 were related support buildings. These buildings are radioactively and chemically contaminated as a result of past processing operations; a few of the buildings contain underground sumps that previously held contaminated liquids. Although most buildings were not used for direct processing activities, they do contain contaminants (generally at much lower levels than the process buildings) as a result of past storage activities, tracking, and wind dispersal. Because they were constructed about 40 years ago, asbestos is also commonly present within the buildings. Over time, the buildings have deteriorated to the extent that some are a safety hazard to on-site workers.

Human exposure could result from contact with either these source areas or the environmental media to which the contaminants have migrated. Possible mechanisms of contaminant release from the source areas are discussed in Section 3.2.2.

3.2.2 Contaminant Fate and Transport

The fate and transport of site contaminants was assessed to identify the environmental media that could be impacted by releases from the on-site source areas. Possible release mechanisms and potentially impacted environmental media at the Weldon Spring site include the following:

- External gamma radiation from various surfaces contaminated with radionuclides (e.g., building interiors, raffinate pit sludge, and localized areas of soil);
- Radon gas emanation from radium-contaminated surfaces (including certain building interiors and soil locations);
- Wind dispersal of building contaminants (e.g., asbestos);
- Wind dispersal of fugitive dust generated from the dumps and other contaminated soil surfaces across the site;
- Surface deposition of airborne particulates (e.g., pursuant to fugitive dust generation or release of building contaminants);
- Surface runoff over contaminated soil following precipitation, with transport to other on-site soil, on-site ponds, and off-site drainage areas (including Lakes 34, 35, and 36 in the Busch Wildlife Area and the Southeast Drainage);
- Overflow from surface impoundments (e.g., Frog Pond) into surface drainages off-site (and previously from the pits onto adjacent soil during the operational period of the plant);
- Surface water loss to the subsurface via losing streams beyond the site boundary (e.g., in the Southeast Drainage and the Ash Pond drainage area);
- Leaching from raffinate pit sludge, pond sediment, and possibly contaminated soil areas through subsurface soil to groundwater;
- Groundwater discharge to surface water (e.g., at Burgermeister Spring);
- Uptake by wildlife from contaminated surface water and from vegetation in contaminated soil areas; and
- Tracking of soil from contaminated surfaces by biota to either on-site or off-site locations.

Volatilization and soil gas generation from volatile and semivolatile organic compounds are not considered primary release mechanisms for source areas because these compounds are generally absent from the Weldon Spring site.

Because of site-specific environmental factors, certain of the potential release mechanisms and receiving media listed above do not currently play a primary role in contaminant fate and transport leading to human exposure at the site. These mechanisms include wind dispersal of building material and fugitive dust, surface deposition of airborne particulates, overflows from the raffinate pits, biouptake, and tracking by biota.

Groundwater is being managed as a separate operable unit for the purpose of remedial action decisions at the site, primarily because of limitations in the existing data. Risks associated with groundwater will be evaluated in detail in the upcoming BA for the groundwater operable unit, which will be prepared when additional information becomes available within the next few years. Data currently available for groundwater quality are briefly summarized in this BA to acknowledge the role of groundwater in contaminant transport and to provide existing information for the affected environment (see Appendix B). For example, surface flow loses to the subsurface in certain areas off-site, so surface water transfer to groundwater with subsequent discharge (i.e., gaining) back to surface water is an element of the exposure assessment in this BA. Furthermore, potential health effects associated with groundwater, e.g., via ingestion of drinking water, are not quantified in this BA because the pathway is incomplete (Section 3.2.4). A preliminary evaluation of this pathway is presented in the rebaseline assessment (Appendix E of the FS), in which potential health effects are estimated for a hypothetical future resident. A more detailed evaluation will be presented in documentation prepared for the groundwater operable unit.

Factors that affect the fate and transport of contaminants at the Weldon Spring site include atmospheric conditions, surface drainage patterns, characteristics of groundwater flow, geochemistry of the overburden, composition of vegetation and wildlife, and contaminant persistence under site conditions. These factors are discussed in Sections 3.2.2.1 through 3.2.2.6.

3.2.2.1 Atmospheric Conditions

Atmospheric transport and deposition during the operational period of the chemical plant has resulted in some building contamination, i.e., airborne radioactive particulates released from the stacks were deposited on roofs and windowsills. Contaminants were also deposited on site soil, as indicated by scattered surface uranium contamination. However, because the site has not been operational for more than 20 years, these airborne releases no longer occur. One possible exception is the chemical plant buildings; in this case, the building interiors still contain material from past activities, and indoor air may be a transport medium for contaminants such as radon and particulates that could be resuspended as a result of activity inside the buildings.

Air has been monitored for radon gas at the site perimeter since 1981, and results indicate that concentrations are not elevated above local background levels. Similarly, perimeter measurements of gamma exposure rates and alpha particulate concentrations are consistently

at background levels. Results of nonspecific monitoring for asbestos (i.e., which measures particulates of a characteristic size and shape) have also been consistent with measured background concentrations. These results indicate that gamma radiation, atmospheric dispersion of radon and radioactive particulates, and deposition of airborne particulates do not currently contribute to the fate and transport of site contaminants.

The monitoring results for particulates are consistent with site-specific predictions that are based on measured wind speed. These results indicate that wind erosion and resultant contaminant dispersal are not expected because of meteorological conditions (Section 3.1.1.1). In addition, much of the site is vegetated or covered with pavement, buildings, or surface water impoundments. However, to include consideration of the air pathway in this BA, the south-eastern portion of the site was conservatively assumed to be a possible (although unlikely) source of fugitive dust generation. This area does not contain either impounded water or buildings, and it is less densely vegetated than the rest of the site.

Rainfalls occur frequently in the area during the spring and summer, and winter precipitation is followed by snowmelt. Therefore, surface runoff is a factor in contaminant fate and transport at the site, as discussed in Section 3.2.2.2. In the local climate, precipitation and evapotranspiration essentially balance each other, which tends to mitigate both surface runoff and vertical leaching of contaminants to groundwater.

Finally, the amount of sunshine can be a factor in the persistence of contaminants such as nitroaromatic compounds that are subject to photodegradation. The number of days of sunlight (clear or partly cloudy days) in the St. Louis area averages about 200 per year (National Climatic Data Center 1987).

3.2.2.2 Surface Water Flow

A surface water divide transects the Weldon Spring site; most runoff flows north toward the Mississippi River, and the remainder flows south toward the Missouri River (Figure 1.7). Contaminants transported via surface runoff from soil in the southeastern portion of the site impact sediment and springs in the Southeast Drainage. This drainage channel consists of a series of four springs and losing-stream segments. Although surface water within the channel loses to the subsurface, groundwater discharges to the surface within the same channel downstream and is eventually released into the Missouri River.

Uranium, nitrate, and sulfate have been measured at concentrations elevated above background in these springs, and elevated concentrations of uranium, thorium, and radium have been detected in the drainage sediment. Contaminant transport from site soil via surface water is a likely cause of nitrate contamination because elevated levels have been detected at several surface locations in the southeastern portion of the site. The same is true for sulfate contamination; sulfuric acid was previously stored in this area during ordnance works operations. Radionuclides and inorganic anions are present in both the raffinate pits and site soil. Hence, contamination in the drainage may be the result of past overflows and decants from the pits and of ongoing surface runoff from the site.

Nitroaromatic compounds have been detected in the springs of the Southeast Drainage but not in the upper portion of the drainage (which is the entry point for site runoff) or in surface soil at the site that drains to this channel. Therefore, it appears that the source of this contamination is not related to the site. Two pipe remnants associated with past Army ordnance production have been located within the channel, and these pipes are considered a possible contaminant source.

Most of the remaining surface runoff at the site, i.e., that north of the divide, flows into either Frog Pond or the Ash Pond area. Overland flow leaving the site from Frog Pond drains into an unnamed tributary of Schote Creek, which empties into Lake 36. Surface water leaving the Ash Pond area joins with runoff from west of the raffinate pits, flowing into another unnamed tributary of Schote Creek that joins with the outflow from Lake 36 and empties into Lake 35 (Figure 1.6). Hence, contaminants transported via this runoff impact both sediment in off-site drainages and surface water and sediment in Lakes 35 and 36. Some of the surface runoff leaving the site near Ash Pond dips underground beyond the site boundary and reemerges at Burgermeister Spring. Hence, contaminants transported via this runoff impact surface water and sediment in both the spring and Lake 34.

Uranium has been measured at concentrations above background levels in soil of the Ash Pond and Frog Pond drainages and in surface water and sediment of Lakes 35 and 36. Elevated concentrations of uranium, nitrate, and fluoride have been measured in surface water and sediment of Lake 34 and in Burgermeister Spring. Uranium is present in Frog Pond and both uranium and nitrate are present in Ash Pond, so surface runoff from these areas is considered a likely source of off-site contamination. In addition, uranium, nitrate, fluoride, and some metals are present at the raffinate pits and in groundwater near the pits. Therefore, leaching and groundwater transport from the pits, with discharge to the surface at Burgermeister Spring, may also contribute to off-site contamination.

Nitroaromatic compounds have been detected infrequently in Burgermeister Spring and in one sample from the Busch lakes. Although the site is the source of off-site uranium contamination and may contribute to off-site nitrate contamination, no primary on-site source of nitroaromatic compounds has been identified within the areas that impact these off-site surface waters. However, some residual subsurface contamination at locations of former production lines (e.g., near Frog Pond) may contribute to the contamination, and sources of nitroaromatic compounds are known to be present on the adjacent Army property (Section 2.2.2).

Some localized areas of surface soil are contaminated as a result of past spills and overflows from both chemical storage tanks and the raffinate pits and influent pipes. For example, limited PCB contamination is present in areas that previously housed transformers, PAHs are present at a concrete pad formerly used to transfer process chemicals, radium is present adjacent to a building at which radioactive material was unloaded for processing, and thorium is present adjacent to the raffinate pits as a result of past overflows. In addition, several small areas of soil off-site (at vicinity properties) are contaminated as a result of waste management activities or spills from transport vehicles carrying material to or from the chemical plant (Figure 1.7).

3.2.2.3 Groundwater Flow

The upper weathered zone of the Burlington-Keokuk Limestone aquifer represents the most likely subsurface pathway for contaminant migration. In general, the distribution of site-related contaminants appears to be localized; that is, available data indicate that the highest contaminant concentrations in groundwater occur in the vicinity of contaminant sources and are present within the site boundary or in close proximity. The aquifer affected by the site is not currently used as a drinking water supply.

Dye-tracing studies conducted by the Missouri Department of Natural Resources (1989) indicate that some of the surface water lost to the subsurface beyond the site boundary reemerges at Burgermeister Spring, about 2 km (1.2 mi) to the north, within hours to days. Groundwater from the site contributes to contaminant transport to off-site surface water bodies via the Burgermeister Spring conduit flow path, as well as via a surface-subsurface-surface flow pattern north of the site.

The raffinate pits and the locations of past spills on adjacent soil (e.g., the former sulfuric acid storage area) appear to be sources of contaminants in perched groundwater lenses beneath the pits and in the shallow aquifer. Concentrations of nitrate, sulfate, fluoride, uranium, and some metals associated with the pits are elevated in adjacent monitoring wells. Nitro-aromatic compounds have also been detected in groundwater wells both on-site and off-site, which is not unexpected because of previous Army activities at the extensive Weldon Spring Ordnance Works property. The Weldon Spring site is just a small portion of the former ordnance works property; the remainder of this property, for which the Army is responsible, lies adjacent to the Weldon Spring site (Figure 1.6). Limited areas of nitroaromatic contamination have been identified on-site, which may be due to past releases from former production lines. Likely source areas of nitroaromatic contamination in groundwater have also been identified on the adjacent Army property; for example, two lagoons formerly used by the Army to dispose of nitroaromatic-contaminated wastewater are located on Army property just beyond the northeastern boundary of the Weldon Spring site.

Water quality data for the site indicate that a downward gradient exists from the weathered zone to the competent zone of the Burlington-Keokuk Limestone. Contaminants have been detected in wells monitoring both the weathered and competent portions of the aquifer (Appendix B). At the site, groundwater flow and vertical migration of contaminants to deeper bedrock aquifer systems, including formations from the St. Peter Sandstone through the Potosi Dolomite, are impeded by a thick sequence of sedimentary rocks classified by the USGS as beds of low hydraulic conductivity (Kleeschulte and Emmett 1986). In addition, water-level measurements of the deep aquifer indicate that artesian conditions apparently exist such that vertical flow in the deep aquifer is toward the surface. Two wells screened in the deeper bedrock aquifer have been sampled by the USGS, one near Dardenne Creek and the other on the adjacent Army property; contaminants associated with the site have not been detected in either well.

3.2.2.4 Soil Geochemistry

The geochemistry of the overburden and bedrock at the site is an important factor in the subsurface migration of certain contaminants. Adsorption and precipitation, and to a lesser extent coprecipitation, are expected to exert the primary influence on attenuation of radionuclides and metals. Geochemical investigations conducted by the USGS indicate that uranium and other metals such as molybdenum adsorb readily to the overburden material; however, as expected, typically mobile ions such as nitrate and sulfate do not (Schumacher and Stollenwerk 1991). Contaminants such as nitrates that are not typically adsorbed onto soil material have in fact been detected at relatively higher concentrations in groundwater at the site than have contaminants such as uranium that appear to be readily attenuated. Organic compounds such as PAHs and PCBs tend to be relatively immobile as a result of their strong adsorptive tendencies. The mobility of specific contaminants, as affected by the geochemistry of site soil, is discussed further in Appendix E.

3.2.2.5 Contaminant Persistence

The long-term persistence of radionuclides at the site is unaffected by environmental conditions (see Figures 2.1, 2.2, and 2.3 for the half-lives of radionuclides in the uranium-238, thorium-232, and uranium-235 decay series). Similarly, the metals and inorganic anions found at the site are not generally subject to environmental degradation. Although some metals such as mercury and arsenic can be microbially methylated and volatilized, they are subsequently converted back to inorganic forms as part of the natural environmental cycling process. Thus, overall persistence is generally high. The limited organic contamination at the site consists of PAHs at a concrete pad and former coal storage area, PCBs in localized soil adjacent to concrete transformer pads, and a few pockets of nitroaromatic compounds in areas of soil associated with past ordnance works operations.

The persistence of PAHs is affected by the molecular weight of the individual compound and environmental factors, including the presence of other contaminants. The PAHs are typically lost from soil as a result of degradation or volatilization to the atmosphere. Microbial metabolism is the major degradation process, whereas photolysis, hydrolysis, and oxidation play relatively minor roles (Sims and Overcash 1983). Lower molecular weight compounds are generally volatilized or biodegraded, whereas volatilization and adsorption are the primary fate processes for compounds with medium and high molecular weights. Biodegradation rates of 33 to 71% were reported for benzo(a)pyrene after 3 months (Ya Keshina et al. 1969). Anthracene and fluoranthene degraded more rapidly than benzo(a)pyrene in another study with fine and medium sands and marsh sediments (Gardner et al. 1979). In general, biodegradation is expected to limit the long-term persistence of PAHs at the site.

In general, the persistence of PCBs depends on the degree of chlorination. The highly chlorinated compounds (e.g., Aroclor 1254) are commonly resistant to biodegradation. This resistance may be due in part to the strong adsorptive characteristics of these compounds. Although they are not easily biodegradable, PCBs in soil systems or aquatic sediments have not been shown to ultimately degrade by any other processes. A study of subsurface sediments

indicated that highly chlorinated congeners may be biotransformed by reductive dechlorination to less chlorinated PCBs, which can then be biodegraded by aerobic processes (Brown et al. 1987).

The persistence of nitroaromatic compounds at the site is expected to be much lower than that of PCBs because nitroaromatic compounds are quite amenable to photodegradation and microbial transformation. These compounds persist in surface water for only a short time (e.g., on the order of hours); however, they can remain somewhat longer in soil, as determined by a variety of factors such as climate, soil moisture, soil organic content, and the nature of indigenous bacterial or fungal populations. For example, persistence in soil decreases with amount of sunlight, which is considerable in the St. Louis area (Section 3.1.1.1). Persistence also typically decreases with increasing soil water content, which averages about 30% in surface soil at the site. The soil organic content (e.g., the amount of humic material such as phenol condensation products present) can enhance phototransformation, so nitroaromatic contamination could be mitigated in the more densely vegetated areas of the site, e.g., the northern quadrant; the organic content of site soil ranges from about 1 to 4%. In addition, more than 100 genera of indigenous bacterial or fungal populations can biotransform nitroaromatic compounds (Burrows et al. 1989), and some of these are probably present at the site.

The biodegradability of NB, as indicated by anaerobic sludge digestion studies, ranges from rapid to resistant and inhibitory (ATSDR 1990i). Although soil data are limited, one study indicated that NB in a 1% suspension of a silt loam decomposed after 64 days; in contrast, the common metabolites phenol and aniline degraded in 1 and 4 days, respectively (Alexander and Lustigman 1966). Photolysis and biodegradation are considered major factors in the fate of DNT. Photolytic half-lives of 3 and 10 hours were reported for natural waters; a relatively rapid biodegradation of DNT compounds was observed under both aerobic conditions (1 hour) and anaerobic conditions (2 weeks) (Spanggord et al. 1980; Liu et al. 1984). The fate of TNT is primarily controlled by photolytic processes (half-lives of 3 to 22 hours have been reported), but biodegradation is an additional factor at high microbial levels (Spanggord et al. 1980).

3.2.2.6 Vegetation and Wildlife

The site and its environs support a wide variety of vegetation and wildlife. Although the site is fenced, animals could potentially carry contaminants to off-site areas. However, the levels of surface soil contamination are generally low, and the results of site characterization and environmental monitoring indicate that this transport pathway does not appear to contribute significantly to contaminant releases that would impact human exposures.

Vegetation and wildlife could also contribute to human exposure through contaminant uptake and subsequent human ingestion. Some vegetation at the site has been analyzed to assess contaminant uptake; certain algal samples exhibited elevated concentrations of uranium and thorium, and smartweed and goldenrod were found to contain low levels of uranium (Table 5.5-8 of the RI). However, under current conditions and reasonable projections of future conditions, humans are not expected to directly ingest vegetation impacted by the site. Therefore, the potential impact of contaminated vegetation was addressed by assessing the

potential contribution of fish and game ingestion to human exposure. Biouptake studies have been conducted for fish in the Busch Wildlife Area lakes and for on-site game animals such as rabbits and squirrels. In addition, a few ducks and a goose were recently collected from the raffinate pits and a deer was taken from the MSA.

Results obtained to date have indicated that some fish contain low levels of uranium (Tables 5.5-9 and 5.5-10 of the RI), but bioaccumulation has not resulted in increased levels of radionuclides within the organisms compared with background data. No uptake has been measured in rabbits or squirrels (Table 5.5-14 of the RI). Analyses of the flesh and internal organs (heart, liver, and gizzard) of the wood ducks, a mallard, and a goose collected at the raffinate pits indicated varying levels of radionuclide uptake (Warbritton 1991; Table 5.5-15 of the RI). The values ranged from less than the detection limit to elevated (Fisenne 1992), as expected from likely differences in the duration and frequency of exposure at the pits for each organism (Section 2.4.5.1). That is, because the site is in the path of the migratory flyway for such birds, the time spent at the pits by these waterfowl cannot be reasonably estimated without tagging studies. Additional studies are planned to further assess the nature of contaminant uptake by local biota, e.g., in the nearby wildlife areas, and results of these studies will be incorporated into future environmental documentation for the site within the next several years. To address potential human exposures in this BA, the limited measurements available have been evaluated, and contaminant uptake by plants and game animals has been modeled with available literature data (Section 3.3.5.2).

3.2.2.7 Primary Release Mechanisms and Transport Pathways

In summary, the environmental release mechanisms and transport pathways considered most important for potential human exposures to site contaminants under baseline conditions are as follows:

- External gamma radiation from radioactively contaminated surfaces, including on-site and off-site soil;
- Surface runoff from on-site areas to off-site drainage soil and surface water;
- Leaching from contaminated surface and/or subsurface soil, sediment, or sludge on-site to groundwater;
- Surface water loss to groundwater via losing streams off-site; and
- Groundwater discharge to surface water via gaining streams off-site.

Indoor air is also considered a transport medium for potential exposures associated with those contaminants present inside the remaining chemical plant buildings. Radon emanation to outdoor air from sitewide soil has also been evaluated. Because measured levels are similar to background concentrations, incremental impacts are expected to be low under baseline conditions.

3.2.3 Exposure Points, Receptors, and Exposure Routes

Exposure points are defined as points of potential contact with a contaminated medium. The contaminated media associated with the Weldon Spring site include soil, sludge, and sediment; surface water; groundwater; air; biota; and indoor air and contaminated residues and dust within the chemical plant buildings. The exposure points evaluated in this assessment are the on-site areas identified in Section 3.2.1 and the off-site areas identified in Section 3.2.2. Likely human activities under current and potential future land-use conditions at the Weldon Spring site were considered in identifying the potential receptors at the various exposure points for the human health evaluation. The routes of human exposure identify the means by which the contaminant(s) can be taken in by a receptor at an exposure point. For this assessment, external gamma irradiation, inhalation, ingestion, and dermal absorption were considered potential exposure routes.

Currently, the Weldon Spring site is fenced and security guards are present so that access by the general public is restricted. However, about 200 full-time workers are present at the site. For these conditions, two potential on-site receptors were identified: (1) a site worker conducting routine maintenance activities (e.g., mowing grass) and (2) a trespasser, assumed to be an adolescent.

Direct worker contact with source areas such as the raffinate pits and chemical plant buildings is limited because worker access to contaminated areas is strictly controlled and worker protection programs are in place to minimize occupational exposures (including requirements for protective clothing and equipment). Hence, potential worker exposures were not assessed for those areas. Potential exposures were assessed for air and sitewide soil — including the dump areas, coal storage area, and nonsource areas across the site — even though the radioactively contaminated source areas are marked and controlled and would not actually be accessed by the worker without appropriate protection. The potential routes of exposure are (1) external gamma irradiation, (2) incidental ingestion of surface soil, (3) inhalation of radon and airborne particulates, and (4) dermal contact with surface soil.

Although public access is restricted, potential on-site exposures to a member of the general public were evaluated in this document to address the issue of trespassing. In considering the nearby populations discussed in Section 3.1.2, an adolescent has been identified as the individual most likely to trespass on the site. It was assumed for this analysis that the trespasser could be exposed to each on-site source area, i.e., the buildings, Ash Pond, Frog Pond, the four raffinate pits, North Dump, South Dump, and the coal storage area. In addition, this individual could be exposed to surface soil across the site and to airborne particulates originating from contaminated surface soil. The potential modes and routes of exposure associated with each of these areas are as follows:

- Surface soil (including dump areas and the coal storage area) — The potential mode of exposure is walking across the site; the likely routes of exposure are (1) external gamma irradiation, (2) incidental ingestion of soil, and (3) dermal contact with soil.

- Air — The potential mode of exposure is walking across the site; the likely route of exposure is inhalation of radon and airborne particulates.

- Raffinate pits and ponds — The potential modes of exposure are taking a drink and swimming; because the raffinate pits are more highly contaminated than either Ash Pond or Frog Pond, it was conservatively assumed that these pits represent the exposure point for a trespasser's on-site exposure to surface water and sludge or sediment. The likely routes of exposure are (1) external gamma irradiation, (2) ingestion of surface water (either intentionally by taking a drink or incidentally while swimming), (3) incidental ingestion of sludge, (4) dermal contact with water while swimming, and (5) dermal contact with sludge.

- Buildings — The potential mode of exposure is walking through the buildings; the likely routes of exposure are (1) external gamma irradiation; (2) inhalation of radon and airborne particulates, such as asbestos; (3) incidental ingestion of residues and dust; and (4) dermal contact with residues and dust.

Under current conditions, the most likely exposure points for a member of the general public are off-site locations that have been impacted by contaminant transport. These locations, termed vicinity properties, include several areas on the adjacent Army land and in the surrounding wildlife areas. Except for Army vicinity property A1 (Figure 1.7; Table 3.6), these contaminated areas are generally small, i.e., less than 0.2 ha (0.5 acre). All of the vicinity properties are on land owned by either the state of Missouri or the federal government, so land use can be expected to remain as it is over the time period of this assessment. Therefore, residential use of any of these contaminated areas is very unlikely. Also, these areas are only accessed infrequently by workers for the purpose of conducting characterization and remediation activities. However, members of the general public could potentially access the areas, especially those vicinity properties located in the wildlife areas (access to the Army land is restricted). For the analyses in this BA, the potential receptor of off-site contamination was assumed to be a recreational visitor who would hike or fish in the Busch Wildlife Area, hike in the Weldon Spring Wildlife Area, and/or hunt at any of the areas.

The potential off-site exposure points and the modes and routes of exposure associated with these areas are as follows:

- Lakes 34, 35, and 36 in the Busch Wildlife Area — The potential modes of exposure are taking a drink, swimming, and fishing; the likely routes of exposure are (1) ingestion of surface water (either intentionally while taking a drink or incidentally while swimming), (2) incidental ingestion of shoreline soil or sediment, (3) dermal contact with water while swimming, (4) dermal contact with shoreline soil or sediment, and (5) ingestion of fish. Inhalation exposure is unlikely because of the absence of radon or airborne particulates from contaminated sources. Although incremental external gamma irradiation is not an issue for the lakes because of the water cover

and generally low levels of contamination in the shoreline soil and sediment, it is also included in this analysis.

- Burgermeister Spring — The potential mode of exposure is taking a drink; the likely route of exposure is intentional ingestion of surface water. Inhalation exposure is unlikely because of the absence of radon or airborne particulates from contaminated sources. Because the spring is small, the potential for dermal contact with, and incidental ingestion of, sediment would be low. External gamma irradiation is not an issue for this spring because of the water cover and generally low levels of contamination.

- Southeast Drainage — The potential modes of exposure are walking along the drainage and taking a drink; the likely routes of exposure are (1) external gamma irradiation, (2) ingestion of surface water, (3) incidental ingestion of sediment, and (4) dermal contact with sediment. Inhalation exposure is unlikely because of the low levels of radon or airborne particulates from contaminated sources.

- Soil vicinity properties — The potential mode of exposure is walking across these areas; the likely routes of exposure are external gamma irradiation and incidental ingestion of soil. Inhalation exposure is unlikely because of the very localized nature of contamination and the typically dense vegetative cover, and dermal contact would be insignificant relative to exposure via external gamma irradiation and incidental ingestion for the contaminants of concern.

At each of these locations and throughout the wildlife areas, the recreational visitor could also be hunting; therefore, game ingestion is considered to be an additional possible route of human exposure.

For this assessment, it was hypothetically assumed that the fences fail and DOE is no longer present at the site in the future, e.g., over the next 10 to 30 years (Section 1.3.2). Current land use is expected to continue under this future scenario because land surrounding the site is owned by the state and federal governments. Therefore, future on-site and off-site members of the public who could be exposed to site contaminants are considered to be recreational visitors to the area. Hence, the potential exposure points and routes of exposure for a member of the general public who might visit the site in the future would be the same as those identified for the trespasser under current conditions, with addition of the game ingestion pathway. For the off-site locations, potential exposures in the future would be the same as those identified under current conditions.

3.2.4 Complete Exposure Pathways

An exposure pathway is complete when all elements are present: i.e., a source of contamination and a release from that source, an exposure point at which contact can occur, and

an exposure route by which this contact can occur. The complete exposure pathways for contaminated media assessed for the Weldon Spring site are summarized by location as follows:

- Soil on-site and at the off-site vicinity properties — external gamma irradiation, incidental ingestion, and dermal contact (for on-site soil only);
- Surface water in the raffinate pits, off-site lakes, Burgermeister Spring, and the Southeast Drainage — ingestion and dermal contact (raffinate pits and off-site lakes only);
- Sludge in the raffinate pits — external gamma irradiation, incidental ingestion, and dermal contact;
- Sediment in the off-site lakes — external gamma irradiation, incidental ingestion, and dermal contact;
- Sediment in the Southeast Drainage — external gamma irradiation, incidental ingestion, and dermal contact;
- Outdoor air on-site — inhalation; and
- Building residues and dust, and indoor air — external gamma irradiation, incidental ingestion, and dermal contact; and inhalation.

For groundwater, discharge to surface water and related exposures have been addressed with the pathways described above for the Southeast Drainage, Burgermeister Spring and the off-site lakes. Relative to the potential ingestion of drinking water, the groundwater pathway is considered to be incomplete under current and short-term future conditions because no exposure points currently exist, i.e., no receptor locations (drinking water wells) have been identified as impacted by site contamination. Furthermore, because of current data limitations, groundwater is being addressed as a separate operable unit for the site. Nevertheless, potential risks from groundwater ingestion are estimated in the preliminary evaluation presented in the rebaseline assessment for the site (Appendix E of the FS). These estimates will be updated in documentation to be prepared for the groundwater operable unit within the next several years, after additional characterization data become available.

The exposure points, exposure media, receptors, and routes of exposure identified for potential on-site and off-site exposures associated with the Weldon Spring site are summarized in Tables 3.1* and 3.2, respectively.

The EPA is currently developing guidance for the dermal pathway, but none has yet been issued. At this time, the general recommendations are (1) to address exposures to soil qualitatively, with limited quantification for the few contaminants for which permeability

*For readability, all tables are presented in sequence at the end of the text in this chapter. Apparent inconsistencies between certain values in the tables reflect rounding to two significant figures.

constants have been developed; and (2) to quantify exposures to water from the limited permeability constants determined for certain contaminants and default values for the other contaminants, and to acknowledge the considerable limitations (Schaum 1991). Although uncertainty is inherent in the estimation of risk for all exposure routes, the uncertainty associated with dermal exposures is very high because of the general lack of supporting information in the scientific literature. In contrast, a relatively large amount of data exists from which appropriate exposure factors can be derived to assess inhalation and ingestion exposures, and a standard methodology has been developed for assessing related impacts (EPA 1989b, 1989j). Because of these differences, the dermal pathway is evaluated separately in this BA.

This approach is also consistent with the current stage of decision making for cleanup of the Weldon Spring site. A decision has already been made to remove and treat contaminated surface water from the on-site raffinate pits and ponds. Therefore, the results of this BA will not be used to determine whether the water should be remediated. In addition, the likelihood of swimming in either the on-site impoundments or the off-site lakes — which is the activity evaluated to quantify dermal exposure to water — is relatively low compared with the other scenarios that have been evaluated.

Security guards regularly patrol the site, and the level of worker activity is high, especially in the area of the raffinate pits (because of construction of the water treatment plant adjacent to the pits). Therefore, even if a trespasser were to gain access — for example by climbing the perimeter fence (topped with barbed wire) and the second fence surrounding the raffinate pits — it is highly unlikely that this individual would swim unnoticed in the pits for an hour (the exposure time typically evaluated for dermal exposure).

Similarly, assuming that typical swimming activity would occur in the off-site lakes that have been impacted by the site — Lakes 34, 35, and 36 in the Busch Wildlife Area — is inconsistent with the posted "no swimming" signs, so the likelihood of such exposure is considered to be somewhat low. (The Missouri Department of Conservation created these lakes by impounding water at a previously vegetated area, so submerged tree remnants and other potential physical hazards exist.) In addition, as part of a program to address general sedimentation at wildlife area lakes, the state plans to drain the surface water from these lakes within the next several years so that sediment buildup can be removed.

For these reasons, the dermal pathway is not quantified with the other pathways in the following discussion. A separate quantitative assessment of dermal exposures to soil, sediment, sludge, and surface water is presented in Section 5.5.

3.3 ESTIMATION OF EXPOSURE POINT CONCENTRATIONS

An exposure point concentration is a medium-specific contaminant concentration at a specified location. Exposure point concentrations were estimated for each exposure point and associated environmental medium at the site and at adjacent properties impacted by site releases (Tables 3.1 and 3.2).

The contaminant characterization studies used to estimate the exposure point concentrations are described in Chapter 2. Data from these studies were entered into a computerized database (Section 2.4), which served as the primary source for estimating the exposure point concentrations for radioactive and chemical contaminants. In some cases, historical data were also used to estimate the exposure point concentrations for radionuclides. The exposure point concentrations were calculated on the basis of measured values — except for the airborne contaminant concentrations for certain analyses (Section 3.3.4), contaminant concentrations in game (Section 3.3.5), and location-specific analyses for on-site radioactive contaminants (Section 3.3.1). Because contaminant concentrations are expected to remain unchanged or to decrease in the future as a result of source control actions being implemented at the site, current concentrations were used to estimate exposures for future scenarios. The uncertainty that this introduces into the analysis is discussed in Section 5.6.2.1.

An approach was developed for estimating the exposure point concentrations on a medium-specific and compound-specific basis to address samples reported as nondetects. The nondetects represent samples for which an analysis was conducted, but the specific analyte was not measured at the detection limit. Although it is recognized in the scientific community that the method used to address nondetects affects the estimates of contaminant concentrations in environmental media, no methodology is widely accepted. The approach used should be based on site-specific considerations of the contaminants and affected media.

For nonradioactive contaminants detected at a high frequency at the Weldon Spring site, it was considered reasonable to assume that samples reported as nondetects might in fact contain the contaminant, albeit at a concentration less than the detection limit, such that some value other than zero (usually half the detection limit) could be substituted for the nondetects. This approach was used for cases in which other information indicated that the contaminant was likely to be present. For example, lead has been detected in site soil at a frequency of more than 90%, and its presence can reasonably be expected because of past use of the site and the natural presence of lead in soil. Therefore, the nondetects for lead were replaced with half its detection limit for the analyses in this document. On the other hand, for contaminants such as organic compounds that were detected at a very low frequency and for which other site information (e.g., historical records) did not indicate an expected presence, it was considered more reasonable to assume that the compound is present only in localized areas, as indicated by the characterization data, and that it is not present in samples reported as nondetects. In this case, the nondetect was set equal to zero for purposes of the analyses. For example, naphthalene was detected at only one location (less than 1% of soil samples), which is consistent with expectations from past use of that location (coal storage), and it is not expected to be present at any other area; therefore, the nondetects reported for this compound were set equal to zero.

Low levels of radionuclides from the uranium-238, thorium-232, and uranium-235 decay series occur naturally, and they are present at elevated levels in scattered locations across the site as a result of past processing activities. In addition, potential health impacts associated with radionuclides are considered a major component of overall risks associated with site exposures. Therefore, it was assumed for this analysis that samples reported as less than the detection limit

do in fact contain low levels of the radionuclides, and the nondetects were conservatively set equal to the detection limit for all media.

The basis for determining the exposure point concentrations and the strategy for addressing chemical contaminant nondetects for each environmental medium of concern are described in Sections 3.3.1 through 3.3.6. The uncertainty that these assumptions introduce into the analyses is discussed in Section 5.6.2.1. For presentational purposes, the estimated exposure point concentrations are presented together with the results of the intake assessment in Section 3.4.

3.3.1 Surface Soil

Contaminants are not uniformly distributed in site soil; rather, contaminant levels range from background concentrations in many areas to relatively high concentrations in localized hot spots. Therefore, parallel analyses were conducted to separately address sitewide exposures and exposures that might occur at a single area with its own unique contaminant characteristics. To support these analyses, two types of exposure point concentrations were estimated for surface soil: (1) a sitewide average concentration for each contaminant and (2) a location-specific concentration for each contaminant determined from measurements at each sampling location.

Inherent in the use of a sitewide average concentration is the assumption that contact with site soil is random, i.e., the receptor would not be expected to spend the majority of his or her time in any one area. Thus, the sitewide average concentration is considered to best represent the likely exposure of an individual who would randomly cross the site. The location-specific approach was conducted to address the possibility that an individual might in fact spend more time in one given area of the site than another (which by chance could be an area of localized high-level contamination). For this approach, it was assumed that the receptor would repeatedly visit the same area of the site; therefore, the contaminant concentration measured at each individual borehole was used as the location-specific exposure point concentration.

In accordance with EPA guidance, the 95% upper confidence limit of the arithmetic average (UL_{95}) was used as the sitewide average concentration. The UL_{95} was calculated for each soil contaminant from surface measurements with the following equation:

$$UL_{95} = \bar{X} + t_{1-\alpha/2,n-1}\left(\frac{s}{\sqrt{n}}\right) \tag{3.1}$$

where:

$\bar{X}$ = arithmetic mean of surface measurements for the contaminant ($\Sigma x_i/n$, with x_i representing the borehole-specific value for the contaminant);

n = number of soil samples analyzed for contaminant i;

$t_{1-\alpha/2,n-1}$ = t-distribution factor (taken from statistical t-tables) for the appropriate confidence interval (where the 95% confidence interval is defined by $\alpha = 0.05$ for the two-tailed test) and the appropriate degree of freedom (n-1) determined from the sample number; a value of 1.96 was used for this factor where n exceeded 120; and

$\frac{s}{\sqrt{n}}$ = estimate of the standard error of the mean, where s is the standard deviation of the sample mean, as calculated from

$$s = \left[\frac{1}{n-1} \sum_{i=1}^{n} (x_i - \bar{X})^2\right]^{\frac{1}{2}}$$

3.3.1.1 Radioactive Contaminants

The main radioactive contaminant at the site is uranium-238. Low levels of uranium are widespread in surface soil; a few hot spot areas are present at the North Dump and South Dump, around some chemical plant buildings, and in the southern tip of the site. Elevated levels of radium and thorium are also present in a few discrete areas. For the location-specific analysis, concentrations of radionuclides that might be present, but were not measured, at each borehole were estimated by applying the radiological source term analysis for soil at the chemical plant area (Table 2.3) to the measured uranium-238 concentration. At boreholes where uranium was not measured, radionuclide concentrations were estimated from the thorium-230 measurement; if thorium was not measured, the radium-226 measurement was used.

The sitewide averages for measured radionuclides (i.e., uranium-238, thorium-230, radium-226, and radium-228) were based on data from 135 to 387 sampling locations; the number varied because not all radionuclides were sampled at each location. The most shallow borehole sample available for each location was used to approximate the surface soil concentration. In most cases, this sample was from the upper 30 cm (1 ft) of soil. The sitewide averages for radioactive contaminants of concern that were not measured were derived from the radiological source term analysis for the chemical plant area and the UL_{95} for uranium-238.

For external gamma radiation, the sitewide average was based on 855 measurements taken at 1 m (3 ft) above the soil surface. Two approaches were used to estimate exposure point concentrations, one in which the background gamma exposure rate of 10 μR/h (Marutzky et al. 1988) was included and a second in which this background rate was subtracted from each measurement. The purpose of these dual calculations was to recognize the natural presence of external gamma radiation and to address the site influence by estimating an incremental value for the UL_{95}.

Surface soil contamination at the off-site areas was treated separately by individually assessing each property. The arithmetic average for the contaminant at each property was used as the exposure point concentration. For the vicinity property analysis, it was assumed that a

recreational visitor would spend all the time during recreational activities at one specific vicinity property. Because of the small size of these properties, this assumption is highly unlikely and, thus, represents a very conservative estimate.

3.3.1.2 Chemical Contaminants

The major chemicals in site soil are metals and inorganic anions. Generally, these constituents are present at background levels or, where elevated, at low levels of contamination. The elevated concentrations of metals are typically localized, e.g., in the vicinity of former dump areas. Low levels of nitroaromatic compounds, PAHs, and PCBs are also present in localized areas associated with past production and storage areas and electrical substations.

Depending on the specific contaminant, data to estimate the sitewide averages were available from approximately 50 to 330 sampling locations. The most shallow borehole sample available for each location was used to approximate the surface soil concentration. The samples generally ranged from 0 to 0.15 or 0.61 cm (0 to 0.5 or 2.0 ft), although some samples were composited over a 2-m (7-ft) interval.

To calculate the UL_{95}, the following algorithm was applied to address results reported as nondetects, i.e., if a sample was analyzed but the contaminant was not detected: (1) for metals and inorganic anions, the concentrations of all nondetects were set equal to half the detection limit and (2) for organic compounds, the concentrations of all nondetects were set equal to zero. In contrast to radionuclides, metals, and inorganic anions, the organic contaminants of concern are not naturally present in soil. Furthermore, except for PCBs, which have a detection frequency of 7%, the detection frequencies for the organic contaminants are less than 3% (Table 2.5), suggesting that they are not likely to be present at most areas of the site.

For the location-specific approach, the measured concentration of each chemical contaminant at each sampling location was used as the exposure point concentration. The nondetects were addressed as previously described for calculating the sitewide UL_{95}. An additional consideration for the location-specific analysis was locations at which an analysis had not been conducted for all contaminants of concern. This was not an issue for the sitewide approach because the UL_{95} is based on the number of samples for each individual contaminant. In contrast, if samples from a given location were analyzed for only some of the contaminants of concern, the potential health effects associated with that location could be underestimated.

Similar to the approach for addressing nondetects, it was assumed that the metals and inorganic anions were present at all sampling locations whereas the organic contaminants were not. The values used to fill in for the metals and inorganic anions were derived from site measurements. The site was divided into six separate areas on the basis of past use and contaminant information. The average concentration of each contaminant was determined for each area, and this value was then used to fill in contaminant concentrations for those sampling locations within that area where the contaminant had not been analyzed. Sensitivity analyses indicated that this approach was not overly conservative compared with other possible approaches, such as filling in with off-site background measurements, because of the generally

low level of chemical contamination and because, for most metals and inorganic anions, the frequency of analysis was greater than 70 to 95%. Thus, data were filled in for a relatively small number of samples.

Finally, data for uranium are not available for any of the chemical boreholes because this contaminant was assayed as part of the radiological characterization studies, and the radiological and chemical boreholes are at different locations. Uranium has been shown to induce noncarcinogenic effects (Section 4.4.2.18), so related impacts must be assessed together with the other chemicals to determine whether an adverse noncarcinogenic impact may be indicated from combined exposures (e.g., if the sum of the individual hazard quotients exceeds 1 [Section 5.1.2.2]). Therefore, it was necessary to transfer information for uranium from the radiological database into the chemical database.

For this effort, the uranium concentration in surface soil at the chemical boreholes was estimated from a weighted average of surface measurements for the radiological boreholes within a radius of 60 m (200 ft), as follows:

$$C_{chem} = \sum \left[\left(\frac{1}{D_i^2 \times D_{total}} \right) \times C_{radi} \right] \tag{3.2}$$

with:

$$D_i^2 = (x_{chem} - x_{radi})^2 + (y_{chem} - y_{radi})^2 \text{ and}$$

$$D_{total} = \sum (1/D_i)^2.$$

where:

C_{chem} = concentration of uranium estimated for the chemical borehole (mg/kg);

C_{radi} = concentration of uranium at radiological borehole i within a 60-m (200-ft) radius of the chemical borehole (mg/kg);

D_i = distance between the chemical borehole and radiological borehole i (ft);

x_{chem}, y_{chem} = x,y coordinates for the chemical borehole (ft); and

x_{radi}, y_{radi} = x,y coordinates for the radiological borehole i (ft).

Comparative plots indicate that the results of this weighted inverse-square distance estimation of uranium for the chemical borehole locations are consistent with information from the radiological database (Xue 1992).

Of the more than 300 chemical boreholes, only 10 are located at a distance of more than 60 m (200 ft) from any radiological boreholes. For these 10 locations, a value of 66 mg/kg was assumed for uranium, which is based on conservative assumptions for uranium in contaminated areas (see Appendix F, Table F.4, of the FS [DOE 1992a]). This assumption is expected to overestimate the actual concentration, as indicated by the low exposure rate measurements at these locations and the rationale for the radiological characterization effort. That is, the density of radiological boreholes is low only in areas that were not expected to be radioactively contaminated, as indicated by site history and exposure rate data. Thus, it is probably more likely that the uranium concentration at those chemical boreholes would be closer to the background level of about 4 mg/kg.

Similarly, the density of chemical boreholes correlates with the focus of the chemical characterization effort. Because the purpose of that effort was to define chemically contaminated areas, as indicated by historical information and data from the initial stage of chemical sampling, locations at which only radioactive contamination was expected were not specifically included. As a result, two radiological boreholes with high uranium concentrations are in an area that is more than 60 m (200 ft) from any chemical borehole. Therefore, these two radiological boreholes were added to the chemical database so the noncarcinogenic impacts of that uranium contamination could be assessed. The concentrations assumed for the chemical contaminants other than uranium at the two new boreholes were inferred from the appropriate area average, as previously described for estimating the concentrations of specific chemicals for which samples from certain boreholes had not been analyzed.

Several thousand data points were generated for the location-specific analysis. That is, the exposure point concentrations effectively constitute the entire database for soil sampling; samples were taken at more than 300 locations, and between 20 and 150 constituents were analyzed for each sample. The exposure point concentrations evaluated for both the radioactive and chemical contaminants for the location-specific analysis are not presented in this BA because of the extensive size of the database. These data were, however, used to estimate location-specific hazard indexes and risks, and the results of these analyses are presented in Section 5.2.1.

3.3.2 Surface Water

3.3.2.1 Radioactive Contaminants

Exposure point concentrations were estimated for both the on-site surface water impoundments and the off-site surface water bodies and drainages impacted by contaminant releases from the site. However, because decisions for remediating on-site surface water have already been made (Section 1.1.2), intakes and risks were not estimated for each location. Water in the raffinate pits is the most highly contaminated of the on-site surface waters and was therefore used to estimate the exposure point concentrations. Because of the heterogeneity in sampling results, the maximum contaminant concentrations measured for the four pits combined were used as the exposure point concentrations.

Exposure point concentrations were determined for surface water at the following off-site locations: (1) Lakes 34, 35, and 36 in the Busch Wildlife Area; (2) Burgermeister Spring; and (3) the Southeast Drainage. From the characterization data available for these locations, uranium is the only radionuclide detected at concentrations above background levels except for one sample at Burgermeister Spring with elevated levels of thorium-230. To estimate exposure point concentrations for radioactive contaminants in off-site surface water, a simplified but conservative approach was taken. For the off-site lakes, the maximum concentration of uranium detected in any of the three lakes was used as the exposure point concentration. This approach was considered reasonable because the state of Missouri plans to drain the surface water and remove the sediment from the lakes within the next several years. For Burgermeister Spring, where all samples have been taken at one location, the highest measured concentrations of uranium and thorium-230 were used as the exposure point concentrations. For the Southeast Drainage, surface water has been sampled at six locations at various times under different flow conditions, and a wide range of uranium concentrations has been observed. Because water flows in the drainage intermittently and the uranium concentrations vary considerably, the exposure point concentration was selected to be half the maximum measurement. This is considered a reasonable estimate for the repeated exposures over 30 years evaluated in this document; this estimate will be refined in the environmental documentation to be prepared for the Southeast Drainage within the next several years after additional characterization data become available.

3.3.2.2 Chemical Contaminants

The approach used to determine exposure point concentrations for the chemical contaminants in on-site surface waters was consistent with that used for the radioactive contaminants. The maximum contaminant concentrations measured for the four raffinate pits combined were used as the exposure point concentrations. For contaminants for which all samples were reported as nondetects, the exposure point concentrations were set equal to zero.

Exposure point concentrations have also been determined for surface water at the same off-site locations as the radioactive contaminants. For most locations, the data available for estimating the exposure point concentrations are limited to a few samples. Only limited sampling was conducted because (1) results of the Phase I characterization studies indicated that levels of chemical contaminants in off-site surface waters are generally low and (2) significant migration of chemical contaminants to these areas is not expected. The low number of samples per surface water location precluded a statistical analysis of the data, so the maximum detected concentrations were used as the exposure point concentrations. This approach was considered to be reasonable, but conservative, on the basis of available data.

For Lakes 34, 35, and 36, the highest concentration of each contaminant detected in the three lakes was used as the exposure point concentration. For Burgermeister Spring, the highest measured concentration for each contaminant detected in samples from the spring was used as the exposure point concentration. For the Southeast Drainage, the highest concentration for the six sampling locations in the drainage was used as the exposure point concentration. At all locations, the concentrations of contaminants for which all samples were reported as nondetects were set equal to zero.

3.3.3 Sediment and Sludge

3.3.3.1 Radioactive Contaminants

Exposure point concentrations were estimated for sediment and sludge in the on-site surface water bodies and in the off-site surface water bodies and drainages impacted by contaminant releases from the site. Because the raffinate pits constitute the most highly contaminated area on-site, they were evaluated as the representative surface impoundment for this analysis; the exposure point concentrations were the maximum concentrations for sludge from the four pits. The exposure point concentrations for sediment in off-site lakes and streams were estimated for the same locations as for surface water (Section 3.3.2.1), and the maximum contaminant concentrations were used for each.

The limited data for sediment in the Southeast Drainage indicates that distribution of radioactive contamination is highly nonuniform. (Sediment and soil in the drainage are addressed together as sediment in this discussion.) Therefore, exposure point concentrations were estimated for each contaminant with the two approaches identified for site soil: (1) a drainagewide average and (2) location-specific measurements. The UL_{95} was used as the drainagewide exposure point concentration to address an individual who would regularly walk along the entire drainage. For the parallel analysis, the location-specific measurements (taken at locations of known elevated contamination) were used to assess exposures at hot spots.

Measurements from approximately 145 sampling locations were used for this analysis (however, only five data points were available for thorium-230). Samples were collected systematically every 100 m (330 ft) along the drainage and at locations identified by the walkover scan. Soil samples were collected at distances up to 11 m (36 ft) from the drainage center. Thirteen shallow borehole samples were collected at intervals of 0.15 to 0.7 m (0.5 to 2.3 ft), and the most shallow sample was used to determine the surface exposure point concentration. The results of this analysis are presented in Section 5.3.4. Because of the large number of data points involved with the location-specific analysis, individual exposure point concentrations are not presented in this document. These data are presented in Marutzky et al. (1988) and the RI.

3.3.3.2 Chemical Contaminants

Similar to the approach used for surface water (Section 3.3.2.2), the maximum contaminant concentrations measured for sludge and sediment in the raffinate pits, off-site lakes, and Southeast Drainage were used to determine the exposure point concentrations at these locations; one exception was uranium in the Southeast Drainage sediment, for which the drainagewide UL_{95} was used. Concentrations of contaminants for which all samples were reported as nondetects were set equal to zero.

3.3.4 Air

Both indoor and outdoor air have been addressed in this assessment. Exposure point concentrations for indoor air (i.e., inside the chemical plant buildings) are presented in Section 3.3.6, and those for outdoor air are addressed in this section. Air is not expected to be a significant outdoor exposure medium under baseline conditions, as indicated by available data (including radon measurements). However, potential exposures associated with this pathway were assessed for completeness. Exposure point concentrations were estimated for radon-222 and airborne particulates. Because the approach used for air particulates was the same for both radioactive and chemical contaminants, the results are presented together.

The concentration of radon-222 in outdoor air depends on the radium-226 concentration in soil, the physical characteristics of the soil, and the meteorological factors that influence both the rate of emanation from soil and the rate of dilution in the atmosphere. Because the relationship between these parameters is complex, it is difficult to model airborne radon concentrations. For this analysis, the outdoor concentration of radon was estimated from data for radium-226 concentrations in soil and a factor of 0.19 pCi/L of radon-222 per pCi/g of radium-226 (Gilbert et al. 1983). For the sitewide analysis, exposure point concentrations were estimated from the sitewide surface UL_{95} value for radium-226; for the location-specific analysis, they were estimated from the radium-226 data for each borehole. Exposure point concentrations in units of working level (WL) were calculated as follows:

$$C_{Rn} = \frac{E \times C}{100} \tag{3.3}$$

where:

C_{Rn} = concentration of radon-222 decay products in air (WL);

E = equilibrium factor; and

C = concentration of radon-222 in air (pCi/L).

The equilibrium factor is used to relate the actual concentrations of radon decay products in air to those that would be present under equilibrium conditions. For this assessment, an outdoor equilibrium factor of 0.267 was used, which is based on the algorithm incorporated into the CAP-88 computer code (Beres 1990).

Air particulate concentrations have not been measured at the site, so the concentrations of fugitive dust were estimated from national air quality trends and from information available for the state of Missouri and the St. Louis metropolitan area (which includes the site). Data for Missouri indicate that the average concentration of TSP is about 0.070 mg/m^3 (Trijonis et al. 1980), and values commonly used for TSP at other geographical locations range from 0.15 mg/m^3 for urban areas to 0.10 mg/m^3 for rural areas (Paustenbach 1989). Various studies have indicated that no more than 30 to 50% of TSP is respirable (Schaum 1983; Cowherd et al.

1985; Trijonis et al. 1980), and information from other studies indicates that the respirable fraction is typically lower (Paustenbach 1989).

Air quality data for the St. Louis area (1982 through 1988) are consistent with these state and national trends. The adjusted annual arithmetic mean concentrations of TSP for this area typically range from about 50 to 80 $\mu g/m^3$, and those for PM-10 (particulates whose aerodynamic diameter does not exceed 10 µm) from all sources range from about 25 to 40 $\mu g/m^3$; for example, annual average PM-10 concentrations at the Chain of Rocks monitoring station in St. Louis ranged from 32 to 40 $\mu g/m^3$ (EPA 1988b, 1990b; Shissler 1990).

In addition, not all airborne particulates at the site would be derived from site soil. Analyses of particulates at another contaminated site indicated that about 83% of the non-respirable particulates but only 47% of the respirable particulates originated from contaminated soil (Schaum 1983). Considering these data, the following assumptions were made to estimate the contribution of site soil to the fraction of respirable particulates in air: (1) the TSP concentration at the site is 0.10 mg/m^3, (2) 50% of the TSP is respirable, and (3) 50% of the respirable fraction originates from contaminated site soil. The total amount of respirable particulates from contaminated soil is therefore estimated to be 0.025 mg/m^3.

Contaminated dust in air above the site is not expected to be generated from the entire soil surface under baseline conditions because site-specific meteorology and the presence of vegetation, buildings, and other surface features would limit dust generation and atmospheric dispersion from most areas. Therefore, to assess inhalation exposures, concentrations of airborne particulates were assumed to be derived from an area in the southern sector of the site where contaminated dust could most likely be generated. This area was selected because its vegetative cover is light, the levels of key contaminants are relatively high, and the direction of prevailing winds (which blow across the site from the south) make it the most representative source of potential sitewide fugitive dust.

To estimate the exposure point concentration of contaminated particulates in sitewide air, the contaminant concentration in the respirable fraction was calculated from the UL_{95} for each contaminant detected in surface soil at the southern area of the site. The airborne contaminant concentration was determined by multiplying the soil UL_{95} value (in pCi/g for radionuclides and mg/kg for chemicals) by 0.025 mg/m^3 — the estimated concentration of respirable particulates — and a conversion factor of 10^{-3} g/mg for radionuclides and 10^{-6} kg/mg for chemicals.

The same estimation method was used to derive contaminant concentrations in air for the location-specific analysis of the inhalation pathway, but it was assumed that the particulates were generated at each discrete sampling location. This approach is similar to that of the location-specific analysis for soil, in which it was assumed that exposure occurs repeatedly in a localized area. However, for the air pathway, fugitive dust is not static and does not originate from a single localized area. Hence, this analysis is considered to be very conservative. The exposure point concentrations for the location-specific analysis of the inhalation pathway are not presented in this document, but these data were used to estimate location-specific hazard indexes and risks.

3.3.5 Biota

Animals and vegetation potentially impacted by site contaminants have been analyzed as part of several studies, but available data are limited. Because additional studies are planned for the site, analysis of the food ingestion pathway in this BA is considered preliminary. Additional analyses will be presented in future documentation for the site.

For fish ingestion, chemical contaminants have previously been addressed in health assessments prepared by both the state of Missouri and the federal ATSDR; these assessments are summarized in Section 2.4.5. Hence, the discussion for the fish ingestion pathway focuses on radionuclides. For game ingestion, the approach used for radioactive contaminants is generally the same as that used for chemical contaminants, so they are presented together in the following discussion.

3.3.5.1 Fish

Uranium-238 is the radionuclide of concern for fish ingestion because it is the only contaminant for which a known link exists between the Weldon Spring site and Lakes 34, 35, and 36 in the Busch Wildlife Area. Results of sampling studies indicate that uranium levels in the fish are low, i.e., uranium has been taken up but not at levels higher than expected from typical bioaccumulation. Furthermore, various control measures implemented at the site (e.g., the dike and diversion system constructed at the Ash Pond area) have greatly reduced the levels of uranium in surface runoff that drains into these lakes.

The exposure point concentrations for uranium in fish were determined from the results of a recent site study (summarized in Table 2.15). The average concentrations reported for fishcakes and fillets (the edible portions of the fish) for each type of fish sampled from Lakes 34, 35, and 36 combined were used as the exposure point concentration. For this estimate, the nondetects were set equal to the detection limit of 0.001 pCi/g. The estimated concentrations are catfish fillets, 0.0013 pCi/g; bass fillets, 0.002 pCi/g; crappie fishcakes and fillets, 0.004 and 0.0013 pCi/g; and sunfish cakes and fillets, 0.022 and 0.007 pCi/g. In addition, the maximum uranium concentration measured in fish samples from the three lakes, 0.94 pCi/g in whole catfish from Lake 36, was used as a more conservative estimate of potential uranium concentrations. For comparison, the fish data from Lake 37 (the control lake) were also evaluated. The measured concentration of 0.006 pCi/g was used for whole catfish, and the nondetects reported for the other Lake 37 fish samples (edible portions) were replaced with 0.001 pCi/g.

The Missouri Department of Conservation plans to drain the surface water from these lakes in the near future and to remove the sediment as part of a general sedimentation management program for the wildlife area. The lakes will then be allowed to refill. If no action were taken at the site, contaminant transport to off-site surface water could continue, and semiequilibrium conditions could reestablish in the lakes. However, because current levels of contamination in the off-site sediment and surface water have resulted from discharges over many years (including uncontrolled releases that no longer occur), it is expected that current

conditions conservatively represent potential future conditions for the fish ingestion pathway. Hence, these same exposure point concentrations were used to estimate the current and future conditions for the fish ingestion pathway.

3.3.5.2 Game

The limited measurements available for waterfowl recently collected from the raffinate pits and for the deer collected from the MSA are insufficient to support statistically conclusive statements, and contamination has not been detected in previous analyses of the edible portions of game animals from the site. The species sampled have generally been limited to small mammals such as rabbit and squirrel. To address the larger game animals such as deer and turkey that have been observed at the site, biouptake was modeled to estimate exposure point concentrations for the game ingestion pathway. Little information is available in the scientific literature from which to determine uptake parameters that can be considered applicable to the wildlife and site-specific conditions evaluated in this assessment. Nevertheless, the limited data available were used to estimate input values for the various parameters to provide some indication of potential exposures via the game ingestion pathway.

Current conditions preclude hunting on-site. However, deer, turkeys, ducks, and geese can move over and back across the perimeter fence. In addition, because the hypothetical future scenario assumes that the fence is gone and site access is unrestricted, the preliminary estimates for the game ingestion pathway in this BA are considered appropriate for future conditions; they are also considered to conservatively represent current conditions for the site. To estimate the exposure point concentrations in animal flesh, sitewide UL_{95} concentrations for radionuclides, metals, and inorganic anions and area-weighted average concentrations for organic compounds in surface soil were used to model the uptake by deer and turkey. Uptake by ducks and geese was modeled with the average detected concentrations of contaminants in raffinate pit surface water.

The first step in the biouptake modeling process is to define the animal's diet in terms of the contribution from contaminated media such as soil, surface water, and (indirectly) vegetation. On the basis of data for ducks in natural systems, the duck was assumed to ingest about 100 g/d of algae and 15 g/d of insects (dry weight), in addition to 500 mL/d of water (DuBowy 1989). For this analysis, a goose was assumed to have twice the diet of a duck. On the basis of literature data, a deer was assumed to ingest about 1.5 kg/d of plant material (dry weight), with dietary water received as dew (Halls 1984). Information on the diet of ducks in captivity indicates that a 1-kg duck eats 100 g/d of dry duck food (DuBowy 1989). From this limited information, it was assumed that a wild turkey with an average weight of about 5 kg would eat 500 g/d of food (dry weight). (This is considered a conservative estimate because a consumption rate of 200 g/d has been indicated for a 6.4-kg turkey [Paustenbach 1989].) A turkey was assumed to ingest both plants and insects, with the former comprising 90% of the diet; the turkey would also receive water as dew (DuBowy 1989).

Data from the literature indicate that radium and actinides such as thorium and uranium commonly bioconcentrate in and adsorb on algae and plankton to concentrations that

can exceed ambient levels by factors of 100 to 10,000 (Whicker and Schultz 1982). Radiological analyses of algae from the on-site surface waters indicate that levels ranged from less than to more than those in the ambient water (RETA 1978). Combining information from the literature and site studies, it was assumed for this analysis that radionuclide levels in algae from the raffinate pits would be 100 times higher than the concentrations in water. An adjustment factor of 10 was incorporated to convert from plant wet weight to dry weight. For the insect component of the diet, it was assumed that the concentrations of contaminants in insects are 10% of those concentrations in the soil. This is expected to be a reasonable assumption because other estimates of the insect component are lower (e.g., 7 to 8% less) and these insects are primarily grasshoppers, walking sticks, and other straight-winged species (e.g., as opposed to grubs, which would be in more intimate contact with the soil [Paustenbach 1989]).

Limited data are available for contaminant levels in site vegetation. Therefore, a plant uptake model was developed to estimate subsequent contaminant intakes by game animals. Three factors were considered in estimating plant contaminant concentrations: direct deposition, uptake from air, and uptake from soil. Because the site is inactive and atmospheric deposition is no longer a significant factor, direct deposition is not a major contributor to plant uptake. Likewise, uptake from the air is not expected to contribute to plant contamination because no organic compounds are volatilized from site soil and radon is generally at background levels at most locations. For this reason, root uptake from soil is considered the primary factor contributing to plant contamination at the site. This uptake can be modeled with the following relationship:

$$C_{pi} = UF_i \times C_{si} \tag{3.4}$$

where:

C_{pi} = concentration of contaminant i in plant tissue (pCi/kg for radionuclides, mg/kg for chemicals);

UF_i = uptake factor for contaminant i from soil (pCi/kg in plant per pCi/kg in soil for radionuclides, mg/kg in plant per mg/kg in soil for chemicals); and

C_{si} = concentration of contaminant i in soil (pCi/kg for radionuclides, mg/kg for chemicals).

For this assessment, the UL_{95} value for each contaminant was used for C_{si}. Contaminant uptake from soil is a complex process that is affected by several factors — including the site-specific geochemistry of the soil; the type of plant and its metabolic requirements; the chemical form of the contaminant; and its partition or distribution coefficient between soil and solution (K_d, in units of L/mg or other conversions). The K_d value can be adjusted to account for the adsorptive effect of organic carbon in soil by multiplying the contaminant-specific partition coefficient between organic carbon and water (K_{oc}) by the fraction of organic carbon in soil (F_{oc}). Values for K_{oc} and/or K_d can be determined from the literature for various

contaminants and soil types; the value for F_{oc} is conservatively assumed from site data to be 1.4%, which is near the low end of the measured range. These data are incorporated into the UFs estimated for inorganic contaminants.

For organic contaminants, the partitioning of the compound between organic material and water is an additional factor that affects plant uptake. This factor is estimated with the octanol/water partition coefficient (K_{ow}), which can be determined from the scientific literature or derived by structural analogy. Briggs et al. (1982) have developed an empirical relationship that can be used to estimate concentrations of organic compounds in the aerial tissues of plants from their concentrations in soil solution. This relationship is defined by the ratio of the organic compound's equilibrium concentration in the plant transpiration stream and its concentration in the soil solution. The plant tissue concentration (C_p) can be estimated by multiplying this ratio by the soil concentration (C_s) and dividing by the adjusted K_d, and the plant uptake factor can be calculated as follows, with adjustments for plant water content (assumed to be 90%):

$$UF_i = \frac{7.056 \exp-[(\log K_{owi} - 1.78)^2/2.44]}{K_{di}} \tag{3.5}$$

where:

$$K_{di} = K_{oci} \times F_{oci}$$

For this assessment, plant uptake factors were derived for inorganic and organic compounds from data in the scientific literature combined with site-specific considerations. The values that were used and supporting information are presented in Appendix E.

The second step of the biouptake model addresses the concentration of contaminants in the edible portion of the animal resulting from levels in the animal's dietary intake. The concentrations of inorganic contaminants in animal tissue from contaminated vegetation were modeled with the following equation:

$$C_{fi} = C_{pi} \times IR_p \times BF_i \tag{3.6}$$

where:

C_{fi} = concentration of contaminant i in animal flesh (pCi/kg for radionuclides, mg/kg for chemicals);

IR_p = plant ingestion rate (kg/d); and

BF_i = biotransfer factor for contaminant i in flesh (d/kg).

For organic compounds, the estimation method of Travis and Arms (1988) was used to calculate the concentration in animal tissue from plants:

$$\log C_{fi} = -7.6 + \log K_{owi} \tag{3.7}$$

The biotransfer factors for site contaminants were also derived from the available literature, as summarized in Appendix E.

The third step of the model considers the time spent by the animal in a contaminated area. Under current site conditions, geese, ducks, and turkeys can fly over the perimeter fence and deer can jump the fence. It was conservatively assumed that the goose and duck would spend their entire lifetimes on-site at the raffinate pits. (Because the pits are the most contaminated aquatic habitat at the site, this represents an additional conservative assumption.) On the basis of the limited information available in the scientific literature, the home ranges estimated for nonmigratory white-tailed deer and wild turkey in Missouri are about 270 ha (670 acres) and 550 ha (1,400 acres), respectively (Halls 1984). Comparing these estimates with the site area of 88 ha (217 acres), the deer and turkey could be assumed to spend about 33 and 16% of their respective lifetimes on-site. For this assessment, it was assumed that the deer and turkey spend one-third of their lifetimes on-site.

In addition to modeling biouptake, limited data consisting of radionuclide tissue concentrations for game animals collected at the site were used to estimate the intake from ingestion. These data are for a single deer collected from the MSA and five waterfowl collected at the raffinate pits (Section 2.4.5.1). The exposure point concentrations for the waterfowl were estimated from data reported for goose flesh and averaging the values reported for the wood ducks and mallard combined; nondetects were replaced with the identified detection limits. The exposure point concentrations for the deer were estimated by averaging duplicate samples from reported flesh data, and nondetects were replaced with the identified detection limits. These measured data were used in subsequent calculations.

3.3.6 Building Air and Contaminated Dust and Residues

The contaminants of concern identified for the chemical plant buildings are uranium-238, thorium-232, radium-228, radium-226, radon-222, radon-220, asbestos, and PCBs. The exposure point concentrations for these contaminants were estimated for those buildings that present the potential for maximum exposure, i.e., those with the highest measured concentrations: Building 403 for radionuclides, Building 201 for asbestos, and Building 408 for PCBs. Combining these exposures provides a conservative estimate, based on the available data, of the exposures to these substances in any building. This screening-level approach for estimating exposure point concentrations was considered adequate because the interim response action to dismantle these buildings has already been approved (Section 1.1.2).

3.3.6.1 Radioactive Contaminants

Exposure point concentrations for radioactive contaminants were based on data collected from Building 403 because high concentrations of radon-220 decay products have been measured in air and high concentrations of uranium-238 and thorium-232 have been measured in building residues. Available data for radioactive contaminants in this building include radon decay product concentrations, concentrations of long-lived alpha particulates in air, and radionuclide concentrations in bulk samples. Arithmetic averages of concentrations measured above the detection limits were used as exposure point concentrations for each parameter. The trespasser/visitor was assumed to spend the entire duration of the visit in Building 403. Hence, these estimates provide a conservative estimate of the level of contamination to which an individual would be exposed during the visit.

3.3.6.2 Chemical Contaminants

The purpose of the asbestos characterization studies was not to quantify the atmospheric concentration of asbestos fibers in the buildings but rather to identify typical building materials that contain asbestos. Therefore, these data could not be used to estimate exposure point concentrations. However, data were collected from 68 area monitors in Building 401 during asbestos-abatement activities (Hoffman 1992). The UL_{95} of the measured concentrations (0.028 fibers/cm^3) was used as the exposure point concentration. This value is considered to conservatively represent levels that might be generated by an individual playing or walking inside the buildings.

Exposure point concentrations for PCB residues were based on data from 11 bulk samples and 26 surface swipes collected in 20 of the 36 rooms in Building 408 (MK-Ferguson Company and Jacobs Engineering Group 1990b). The sampling strategy focused on those areas where PCBs were expected to be found. Bulk sample concentrations are given in units of mg/kg whereas surface swipe concentrations are given in units of μg/100 cm^2. Because of the different sampling methods and concentration units, two exposure point concentrations were estimated: the UL_{95} of the bulk sample concentration (390 mg/kg) and the UL_{95} of the surface swipe concentration (350 mg/m^2).

No air samples were collected for PCB analyses. Thus, a dust resuspension model (Healy 1971) was used to estimate PCB dust concentrations resulting from the resuspension of contaminated residues in the building. Airborne PCB levels were estimated for each contaminated room, as follows:

$$C_{PCB} = \frac{C_{ri} \times F \times A}{V \times N} \tag{3.8}$$

where:

C_{PCB} = PCB concentration in air (mg/m^3);

C_{ri} = contaminant concentration in room surface (mg/m^2);

F = fraction of dust resuspended per hour, assumed to be that associated with moderate activity in a room ($10^{-4}/h$);

A = area of contamination within a room (m^2), assumed to be equal to the size of the contaminated area (if known) or 20 m^2 (if unknown);

V = volume of the room in which the contamination is located (m^3); and

N = number of air exchanges per hour (1/h), assumed to be 0.5/h.

3.4 ESTIMATION OF CONTAMINANT INTAKES

3.4.1 Introduction

Estimates of exposure are based on the contaminant concentrations at the exposure points (as described in Section 3.3) and scenario-specific assumptions and intake parameters. The scenario-specific assumptions include factors such as the age and weight of a potential receptor and the frequency and duration of exposure to contaminated media; intake factors are specific to the route of exposure, e.g., inhalation or ingestion rates. The assumptions and intake factors for the exposure scenarios evaluated in this BA are presented in Section 3.4.2, and contaminant intakes at the potential exposure points are estimated in Sections 3.4.3 through 3.4.8.

In accordance with EPA guidance (EPA 1989j), intakes were estimated for a single route of exposure (e.g., ingestion of surface water) at a single exposure point (e.g., the raffinate pits). Exposure estimates have been based on the "reasonable maximum exposure" (RME) expected to occur under current and future land-use conditions. The RME is defined as the highest exposure that can reasonably be expected to occur at an exposure point. Scenario assumptions and intake parameters used to estimate the RME were based, to the extent possible, on values provided in the *Human Health Evaluation Manual* (EPA 1989j) and in a recent supplement to that manual (EPA 1991b). In general, the value used for each parameter is the 90th or 95th percentile value; for some variables, the 50th percentile value is recommended.

Exposure to radioactive contaminants is expressed in terms of the 50-year committed effective dose equivalent (CEDE). This concept, which was developed by the International Commission on Radiological Protection (ICRP) for internal exposures (ICRP 1977), is the weighted sum of the dose equivalent in various organs. The weighting factors for the organs are proportional to the potential risk associated with the irradiation of those organs. To calculate the CEDE, the contaminant concentration at the exposure point (as estimated in Section 3.3) is multiplied by the intake factors appropriate to each medium and the appropriate dose conversion factor (DCF). The DCF, which is specific to the radionuclide and pathway, is used to determine the CEDE per unit intake of the radionuclide. The DCFs used in this assessment

are based on dosimetry models developed by the ICRP (1979, 1980, 1981b) and incorporate the following considerations:

- The radiosensitivity of organs is variable.
- Some forms of radiation are more effective than others in causing biological harm.
- Retention time in the body following intake is variable for different radionuclides; some radionuclides remain in the body for many years, continually irradiating the organ(s).

The concept of committed dose applies only to internal pathways. For external pathways, there is no long-term residence of radionuclides in the body and the appropriate measure of dose is the effective dose equivalent (EDE). The sum of the CEDE and the EDE from external pathways is termed the total committed effective dose equivalent. To simplify the discussion, both CEDE and EDE are referred to as dose (expressed as mrem) in the following sections. The DCFs used in this assessment are given in Table 4.1. These factors were adapted from Gilbert et al. (1989) and are based on ICRP dosimetry models. Additional discussion of the DCFs is given in Section 4.1.

Exposure to radon-222, radon-220, and their short-lived decay products is reported in units of WL to incorporate recent information given in the BEIR IV report on the radiological hazards associated with inhalation of radon (National Research Council 1988). One working level is defined as any combination of short-lived radon decay products in one liter of air without regard to the degree of equilibrium that will result in the emission of 1.3×10^5 MeV of alpha energy. One working level corresponds to 100 pCi/L of radon-222, or 7.4 pCi/L of radon-220, in equilibrium with their respective short-lived decay products. The dose resulting from exposure to radon-220, radon-222, and their short-lived decay products is given in the unit of working-level month (WLM). One working-level month is the exposure to 1 WL of radon decay products for 170 hours, assuming an inhalation rate of 1.2 m^3/h. This unit was originally developed to monitor uranium miners, i.e., 1 WL was considered to be an acceptable maximum airborne concentration of radon decay products for uranium miners working 40 hours per week (or 170 hours per month). The radon exposure in WLM can be converted to dose in mrem with the conversion factor of 1,000 mrem/WLM from the ICRP (1981a).

Exposure to chemical contaminants is expressed in terms of intake. Intake is the amount of contaminant taken into the body per unit body weight per unit time (generally expressed in mg/kg-d).

3.4.2 Scenario-Specific Assumptions and Intake Parameters

The assumptions and intake parameters used to estimate the radiological and chemical exposures of the receptors considered in this BA are summarized in Table 3.3 and are discussed

in Sections 3.4.2.1 through 3.4.2.7. The uncertainty associated with the intake parameters is discussed in Section 5.6.2.2.

3.4.2.1 Exposure Time, Exposure Frequency, and Exposure Duration

The three parameters — exposure time, exposure frequency, and exposure duration — together define the total extent of exposure at an exposure point. The exposure time is the number of hours per day (or hours per exposure event) that a receptor is present at a specific exposure point; the exposure frequency is the number of days per year (or events per year) that exposure occurs; and the exposure duration is the total number of years over which exposure occurs.

The on-site maintenance worker was assumed to be exposed 8 hours per day for 200 days per year over 10 years. The exposure time assumes a normal 8-hour working day; the exposure frequency of 200 days per year was estimated from the total number of working days per year (250) and the estimated number of days of rain or snow that would prevent outdoor activities (50).

The on-site trespasser was assumed to enter the site 5 times per year for 1 hour per visit for 10 years. Site controls such as fences and security guards, in addition to the presence of on-site workers, reduce the likelihood of more frequent entry and limit the length of time a trespasser would be expected to remain on-site. Under the hypothetical future scenario in which it was assumed that the fences fail and DOE is no longer present on-site, the exposure time, exposure frequency, and exposure duration of a receptor would be expected to increase over those for the trespasser. The future on-site recreational visitor was assumed to enter the area currently occupied by the site 20 times per year, for 4 hours per visit at outdoor locations and 1 hour per visit at indoor locations (i.e., the chemical plant buildings), for 30 years. These values are considered to be conservative estimates of the exposure time and frequency for such a visitor, on the basis of availability of land (approximately 5,800 ha [14,000 acres]) managed for recreational activities in the surrounding Busch and Weldon Spring wildlife areas relative to the size of the site (88 ha [217 acres]). The value for the exposure duration is the national upper bound value (90th percentile) for time spent at one residence (EPA 1989j) and, hence, may be considered a conservative estimate of the number of years during which an individual would be living in the area.

The exposure time, exposure frequency, and exposure duration for the off-site recreational visitor to the Busch and Weldon Spring wildlife areas and the vicinity properties are the same as for the on-site future recreational visitor, i.e., 20 visits per year for 4 hours per visit over 30 years. These values are considered to be conservative estimates of the exposure parameters because of the limited area of the specific off-site exposure points relative to the total area available for recreational use in the Busch and Weldon Spring wildlife areas.

The exposure times, exposure frequencies, and exposure durations indicated for each receptor were used to estimate the RMEs for each exposure route and exposure point. For example, the intake estimates for drinking water from the raffinate pits presented in Section 3.4.4

assume that the trespasser repeats this activity 5 times per year for 10 years, and the estimates for drinking water from Burgermeister Spring assume that the recreational visitor repeats this activity 20 times per year for 30 years. Exposures associated with scenarios in which a receptor is exposed through more than one route of exposure and/or contacts multiple exposure points are discussed in Sections 5.2.6, 5.3.6, and 5.4.

3.4.2.2 Inhalation Rates

A scenario-specific inhalation rate of 2.1 m^3/h was derived for the trespasser and future recreational visitor. This value was calculated on the basis of average inhalation rates for male adults at various activity levels and information on outdoor activity patterns given by the EPA (1989b). To estimate the RME for this pathway, a higher proportion of time was assumed to be spent at moderate and heavy activity levels relative to the average time patterns given by the EPA. This approach is consistent with recent guidance for deriving inhalation rates (EPA 1991b). The derived rate was based on inhalation rates of 0.7, 0.8, 2.5, and 4.8 m^3/h during resting, light, moderate, and heavy activities, respectively, assuming that 20, 20, 45, and 15% of the exposure time was spent at these activity levels. Because the data provided by the EPA (1989b) suggest that inhalation rates for an adolescent tend to be similar to those of an adult at the same activity level, the adult values were used for the trespasser. The standard default value of 2.5 m^3/h was used for the worker, which is the rate for an individual engaged in heavy activities; thus, it is expected to provide a conservative estimate of inhalation exposures for a site worker involved in the generally less strenuous activities associated with routine site maintenance.

3.4.2.3 Soil, Sediment, Sludge, and Residue Ingestion Rates

Individuals may ingest soil either inadvertently (e.g., by transfer of soil on hands and fingers to food or cigarettes) or intentionally (by pica). Pica behavior is generally associated with young children (2 to 6 years old) and is therefore irrelevant for the age group considered in this assessment. Recent EPA guidance for soil ingestion rates recommends the following values: 100 mg/d for adults and 200 mg/d for children 1 to 6 years of age (EPA 1991b). However, these are the rates recommended for a resident, assuming ingestion of both outdoor soil and indoor dust, and are not directly applicable to the receptors and exposure situations considered in this BA. Therefore, scenario-specific ingestion rates were derived from information given by the EPA (1989b, 1991b) and Hawley (1985).

The primary mechanism of soil ingestion was assumed to be inadvertent ingestion of dust and soil on the hands and mouth. The trespasser was assumed to be on-site for only 1 hour per visit and to ingest 80 mg per day; this value is approximately 50% of the amount of site soil assumed to be present on the inside surfaces of the fingers and hands and 100% of the dust deposited in the mouth area. The recreational visitor was assumed to spend 4 hours on-site per visit and to ingest approximately 1.5 times this amount, i.e., 120 mg/event. The worker was assumed to be exposed 8 hours per day and to ingest 80 mg of soil three times daily, giving an ingestion rate of 240 mg per day. The mode and time of exposure to sediment and sludge are similar to those for soil; therefore, the ingestion rates for soil were also used for those media.

For PCB residues in the buildings, the time spent by the recreational visitor inside the buildings was assumed to be 1 hour, so a rate of 80 mg per day was used for ingestion of residues.

3.4.2.4 Water Ingestion Rates

A water ingestion rate of 200 mL (about 1 cup) per exposure event was used for intentional ingestion of surface water from a pit, lake, or spring. This estimate is based on consideration of the poor quality of water at these locations and the amount of water an individual would likely drink from such sources.

3.4.2.5 Fish and Game Ingestion Rates

The rate of fish ingestion for the radiological assessment was taken from the assumptions used by the Missouri Department of Health in its chemical health assessment for this pathway (Section 2.4.5.2). These assumptions reflect location-specific data and consider persons fishing for subsistence. The Missouri Department of Conservation has determined from area-use data that the average person fishing spends 3.5 days per year fishing at the wildlife area lakes and that persons fishing for subsistence concentrate on catfish and tend to fish in one of the many "put-and-take" lakes in the wildlife area managed for that purpose — which do not include Lakes 34, 35, and 36. It was reported by Schmidt (1990) that the percentages of time spent fishing for bass, catfish, and other fish are 40, 39, and 11%, respectively.

The state study assumes that the person fishing spends 7 days per year fishing at the contaminated lakes. This rate is twice the use value for the lakes and includes consideration of the fact that a person fishing for subsistence would spend considerably less time at Lakes 34, 35, and 36 than at the numerous put-and-take lakes in the wildlife area. It was also assumed that the daily limit of catfish is successfully caught and consumed. The allowable catch in the wildlife area is four channel catfish per day, for which the average live weight is about 450 g (1 lb). Thus, the annual consumption was assumed to be about 6,400 g (14 lb), which represents an average ingestion rate of about 17 g/d, assuming that the catch is shared by two people. Because these assumptions incorporate area-use data, they are considered reasonable for use in the radiological assessment of the fish ingestion pathway.

An additional calculation was performed for this pathway to incorporate data for fish-cakes and fillets from the various fish sampled from Lakes 34, 35, and 36; Lake 37 was included separately for comparison. The same fishing rate (double the observed value) was assumed for this analysis, and ingestion rates were estimated from area-use data and catch limits averaged for the year. A rate of 17 g/d was assumed for catfish; because the daily limit for bass is half the catfish limit and the estimated fish weight is the same, an ingestion rate of 8.5 g/d was assumed for bass.

The daily catch limit for crappie is 30 fish, but the individual fish weight is estimated to average about one-third that of catfish and bass and the weight of the edible portion is estimated to be about half the whole fish weight. Therefore, an ingestion rate of 5 g/d was

assumed for crappie. The limit for sunfish is 25 lb plus one fish. From area-use data (11% fish other than catfish and bass), it was assumed that 30 sunfish represents a reasonable but conservative daily catch. Fish weights were assumed to be the same as those for crappie, so an ingestion rate of 5 g/d was also assumed for sunfish. Thus, the fish ingestion rate was assumed to total about 36 g/d for this calculation, split equally between fishcakes and fillets. No crappie were sampled from Lake 37, so an ingestion rate of 10 g/d was assumed for sunfish from this lake.

Similarly, the rate for game ingestion was derived from site-specific data. These data included maximum allowable takes of game animals in the Weldon Spring area, estimates of the weights of edible portions, and related consumption rates (Table 3.4).

3.4.2.6 Body Weight

The value for body weight is the average value over the exposure period. In accordance with EPA guidance, a value of 70 kg was used as the average adult body weight for the maintenance worker and recreational visitor. A body weight of 50 kg was estimated for the adolescent trespasser and swimmer over the 10-year exposure period, which was based on data from the EPA (1989b) and Anderson et al. (1985).

3.4.2.7 Averaging Time

Two intake estimates were calculated for each chemical exposure point and route of exposure, one for each of two averaging times. The first was the estimated average daily intake over the exposure duration of 10 years (3,650 days) for the maintenance worker, trespasser, and swimmer and that of 30 years (10,950 days) for the on-site and off-site recreational visitors. The average daily intake for the exposure duration was used to estimate the potential noncarcinogenic effects of the chemical contaminants of concern. The second estimate was the total intake received during the 10-year or 30-year exposure period averaged over a lifetime of 70 years (25,500 days). The total intakes for the exposure duration and lifetime estimates are the same, but the averaging period is different. The lifetime intake was estimated for calculating chemical carcinogenic risks because the slope factors used to estimate carcinogenic risks are based on an exposure period of 70 years (Section 4.3).

3.4.3 Equations and Intake Estimates for Exposures to Soil

Exposures were estimated for the radioactive and chemical contaminants of concern for surface soil within the site boundary and for the radioactive contaminants of concern for surface soil at the vicinity properties. The exposure routes assessed were external gamma irradiation and incidental ingestion of soil.

3.4.3.1 External Gamma Irradiation

The dose from external gamma irradiation was calculated by multiplying the length of time an individual is assumed to be exposed to the radiation field by the measured field strength and the DCF of 0.95 mrem/mR. At locations for which external gamma measurements were not available, the dose was calculated from exposure point concentrations of specific radionuclides, as follows:

$$D_i = R_{si} \times ET \times EF \times ED \times DCF_i \tag{3.9}$$

where:

D_i = dose from radionuclide i (mrem);

R_{si} = soil concentration of radionuclide i (pCi/g);

ET = exposure time (h/d or h/event);

EF = exposure frequency (d/yr or events/yr);

ED = exposure duration (yr); and

DCF_i = external gamma dose conversion factor for radionuclide i [(mrem/h)/(pCi/g)] (see Table 4.1).

This calculation is based on the assumption of uniform, whole-body irradiation. External gamma doses were estimated for potential receptors under both current and future land-use conditions. Under current conditions, the receptors in contact with site soil are the maintenance worker and the trespasser; under future conditions, the receptor would be the recreational visitor. The external gamma doses to these three receptors, which were calculated on the basis of measured gamma exposure rates at the site, are given in Table 3.5. The results, calculated both with and without the background exposure rate of 10 μR/h (the latter to indicate incremental effects), show that natural background radiation is the major contributor to the total external gamma dose.

Some locations near the site also contain elevated levels of radioactivity. An off-site recreational visitor would incur an external gamma dose while visiting these locations. The external gamma doses to receptors at the vicinity properties were calculated from Equation 3.9. The radionuclides considered in these calculations, their estimated exposure point concentrations in soil, and the estimated doses are presented in Table 3.6. Background soil concentrations were not subtracted from these calculations; however, this approach has only a small impact on the calculations because the background concentration of these radionuclides is only about 1 to 2 pCi/g.

For the location-specific analysis, external gamma doses were calculated with the gamma exposure rates measured at specific locations. Because of the large number of data

points associated with this approach, individual location-specific doses are not presented. The results of the analysis are given in Section 5.2.1.

3.4.3.2 Ingestion

The doses associated with intake of the radioactive contaminants resulting from incidental ingestion of surface soil were calculated as follows:

$$D_i = R_{si} \times IR_s \times CF_1 \times EF \times ED \times DCF_i \tag{3.10}$$

where:

IR_s = soil ingestion rate (mg/d or mg/event);

CF_1 = conversion factor (10^{-3} g/mg); and

DCF_i = ingestion dose conversion factor for radionuclide i (mrem/pCi) (see Table 4.1).

The intakes of chemical contaminants resulting from incidental ingestion of surface soil were calculated as follows:

$$I_i = \frac{C_{si} \times IR_s \times CF_2 \times EF \times ED}{BW \times AT} \tag{3.11}$$

where:

I_i = intake of contaminant i (mg/kg-d);

C_{si} = soil concentration of contaminant i (mg/kg);

CF_2 = conversion factor (10^{-6} kg/mg);

BW = average body weight over the exposure period (kg); and

AT = averaging time (d).

Intakes were estimated for potential receptors under both current and future land-use conditions. Under current conditions, the receptors in contact with site soil are the maintenance worker and trespasser; under future conditions, the receptor would be the recreational visitor. The scenario-specific values for the exposure and intake variables are presented in Table 3.3. The exposure point concentrations of radionuclides and the resultant radiation doses from incidental ingestion are given in Tables 3.6 and 3.7 for vicinity properties and on-site soil. The exposure point concentrations of chemical contaminants and the estimated intakes resulting from incidental ingestion of on-site soil are presented in Table 3.8.

Equations 3.10 and 3.11 were also used to estimate the intake from soil ingestion for the location-specific analysis. Because of the large number of data points associated with this approach, the location-specific dose estimates are not presented in this document. The results of the location-specific analysis are presented in Section 5.2.1.

3.4.4 Equations and Intake Estimates for Exposures to Surface Water

Exposures associated with ingestion of surface water were estimated for the raffinate pits within the site boundary and for lakes, springs, and streams in the Busch and Weldon Spring wildlife areas. The doses associated with intake of radioactive contaminants resulting from intentional ingestion of surface water were calculated as follows:

$$D_i = R_{wi} \times IR_w \times EF \times ED \times DCF_i \tag{3.12}$$

where:

R_{wi} = concentration of radionuclide i in water (pCi/L) and

IR_w = water ingestion rate (L/d or L/event).

The intake of chemical contaminants resulting from intentional ingestion of surface water was calculated as follows:

$$I_i = \frac{C_{wi} \times IR_w \times EF \times ED}{BW \times AT} \tag{3.13}$$

where C_{wi} = concentration of contaminant i in water (mg/L).

Intakes were estimated for potential receptors under current and future land-use conditions. Under current conditions, the only on-site receptor is the trespasser; on-site maintenance workers are aware of the hazards associated with this water and would not drink from the pits. Under future conditions, the receptor would be the recreational visitor. Within the time frame of this analysis, exposures to surface water in off-site lakes and springs are expected to be similar under both current and future conditions. The potential receptor is the recreational visitor at Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage.

The scenario-specific values for the exposure and intake variables are presented in Table 3.3. Estimated doses and exposure point concentrations for the radioactive contaminants are presented in Table 3.9 for the raffinate pits and Table 3.10 for the off-site lakes, Burgermeister Spring, and the Southeast Drainage. The estimated exposure point concentrations and chemical intakes for the identified receptors resulting from intentional ingestion of surface water are presented in Table 3.11 for the raffinate pits, Table 3.12 for the Busch lakes, and Table 3.13 for Burgermeister Spring and the Southeast Drainage.

3.4.5 Equations and Intake Estimates for Exposure to Sediment and Sludge

Exposures to sediment and sludge were estimated for the same exposure points as for surface water, i.e., raffinate pit sludge within the site boundary and sediment in lakes, springs, and streams in the Busch and Weldon Spring wildlife areas. The exposure routes assessed were external gamma irradiation and incidental ingestion of sludge and sediment.

3.4.5.1 External Gamma Irradiation

Exposure via the external gamma pathway is a concern only for the raffinate pit sludge, sediment in the Southeast Drainage, and sediment/soil along the lakeshores. Sediment within the site boundary at other locations (Ash Pond and Frog Pond) and sediment at other off-site areas (Lakes 34, 35, and 36 and Burgermeister Spring) are covered with water, which effectively attenuates gamma radiation.

The external gamma pathway is not a major exposure pathway for the raffinate pit sludge under current site conditions because the sludge is covered with water. However, this pathway could be significant if the current water balance were disrupted, e.g., during a very dry summer when more water evaporates from the pits than is replenished by precipitation. In this case, an individual could be exposed directly to the raffinate pit sludge and incur an external gamma radiation dose. To address this possibility, it was assumed that an individual visiting the raffinate pits would be exposed to the sludge for the entire length of each visit over the entire duration of exposures. The estimated dose to a current trespasser under these conditions would be 0.32 rem and that to a future recreational visitor would be 15 rem. These doses were estimated with Equation 3.9, and the maximum measured radionuclide concentrations in sludge from the four raffinate pits combined were used as the exposure point concentrations.

The estimated dose from exposure to sediment/shoreline soil at Lakes 34, 35, and 36 was also calculated with Equation 3.9, and the maximum measured concentration of 36 pCi/g of uranium-238 was used as the exposure point concentration. The external gamma dose to a recreational visitor at these lakes is estimated to be 1.3 mrem.

The dose from external gamma irradiation resulting from an individual walking along the Southeast Drainage was estimated from an exposure point concentration of 12 μR/h. This exposure rate is the UL_{95} calculated from measured values, and it includes a background exposure rate of 7 μR/h. (This background exposure rate, which was reported by the individuals who conducted this survey, is different from the rate of 10 μR/h reported from other studies elsewhere in this document. The measured values reported for the Southeast Drainage, as provided in the cited references, were used to accurately reflect site and instrumentation conditions as reported by the field surveyors.) The dose was calculated by multiplying the length of time an individual is exposed to this radiation field by the radiation field strength and by a DCF of 0.95 mrem/mR. The dose from external gamma irradiation is estimated to be 27 mrem including background and 11 mrem excluding background.

Equation 3.9 was used to calculate radiation doses from location-specific measurements of radionuclide concentrations for sediment in the Southeast Drainage. Because of the large number of data points associated with the location-specific approach, individual doses are not presented; the risks associated with these doses are presented in Section 5.3.4.1.

3.4.5.2 Ingestion

Intake of sludge or sediment may occur through (1) incidental ingestion of these materials on the hands or face, similar to the incidental ingestion pathway for soil, and (2) intentional drinking of water in which sludge or sediment has been resuspended. Exposures via the first mechanism could occur in areas with intermittent water flows or at times when the water level in an impoundment is low and sludge or sediment is exposed. Intakes for this pathway were estimated with Equations 3.10 and 3.11.

Intake of sludge or sediment suspended in water while drinking is included in the intake assessment for surface water. The concentrations of contaminants in surface water are reported for unfiltered samples, i.e., the contaminant concentrations include contributions from sludge or sediment. Although additional material might be suspended while withdrawing water, the intake would be similar to (or less than) that estimated for the incidental ingestion pathway. Hence, the total intake from ingestion of sludge or sediment can be estimated by summing these two pathways (see Sections 5.2.6 and 5.3.6).

Sludge or sediment might be exposed, e.g., during periods of low rainfall, at locations such as the raffinate pits within the site boundary and the lakes and streams in the Busch and Weldon Spring wildlife areas. Exposures via incidental ingestion were estimated for the same receptors identified for the surface water pathways (Section 3.4.4). The scenario-specific values for exposure and intake variables are presented in Table 3.3. The exposure point concentrations and estimated radiological doses for the identified receptors resulting from incidental ingestion of sludge or sediment are presented in Table 3.14 for the raffinate pits and Table 3.15 for Lakes 34, 35, and 36 and the Southeast Drainage. The corresponding results for chemical contaminants are presented in Tables 3.16 and 3.17.

3.4.6 Equations and Intake Estimates for Inhalation of Airborne Contaminants

Exposures associated with both indoor and outdoor air are addressed in this BA. Equations for estimating intakes associated with indoor air contaminants are presented in Section 3.4.8 and those for outdoor air are presented in this section. Exposures to outdoor air include both inhalation of radon-222 and its short-lived decay products and inhalation of contaminated airborne particulates generated from site soil; these exposures were evaluated for all on-site exposure points.

Doses associated with intake of radon resulting from inhalation of outdoor air were calculated as follows:

$$WLM = \frac{C_{Rn} \times IR_a \times ET \times EF \times ED}{CF_3} \tag{3.14}$$

where:

WLM = working level month(s) and

CF_3 = conversion factor for inhalation (204 m^3/mo).

The term CF_3 is the product of the inhalation rate (1.2 m^3/h) and the number of working hours in 1 month (170 h/mo). The estimated doses to a current worker, current trespasser, and future recreational visitor associated with inhalation of radon decay products are presented in Table 3.18. Doses were calculated both with and without background radon, which was derived from the background concentration of 1.2 pCi/g for radium-226 in soil. The estimated doses are not actual doses but are exposures expressed in WLM. The WLM unit was used because the inhalation exposure to radon decay products is typically expressed in this unit (National Research Council 1988).

The doses associated with inhalation of radioactively contaminated air particulates were calculated as follows:

$$D_i = R_{ai} \times IR_a \times ET \times EF \times ED \times DCF_i \tag{3.15}$$

where:

R_{ai} = air concentration of radionuclide i as respirable particulates (pCi/m^3);

IR_a = inhalation rate (m^3/h); and

DCF_i = inhalation dose conversion factor for radionuclide i (mrem/pCi) (see Table 4.1).

Intake estimates for chemical contaminants were calculated as follows:

$$I_i = \frac{C_{ai} \times IR_a \times ET \times EF \times ED}{BW \times AT} \tag{3.16}$$

where C_{ai} = air concentration of contaminant i as respirable particulates (mg/m^3).

Intakes were estimated for potential receptors under both current and future land-use conditions. Under current conditions, the receptors in contact with air within the site boundary

are the worker and trespasser; under future conditions, the receptor would be the recreational visitor. The scenario-specific values for the exposure and intake variables are presented in Table 3.3. The exposure point concentrations and estimated intakes for the identified receptors resulting from inhalation of airborne particulates are presented in Table 3.18 for the radioactive contaminants and Table 3.19 for the chemical contaminants.

Equations 3.15 and 3.16 were also used to estimate intakes from inhalation of airborne particulates for the location-specific analysis. Because of the large number of data points generated from this analysis, the intake estimates associated with each contaminant at each location are not presented in this document. These data were used to estimate location-specific hazard indexes and carcinogenic risks, and the results are discussed in Section 5.2.3.

3.4.7 Equations and Intake Estimates for Ingestion of Biota

The intakes estimated for the fish and game ingestion pathway consider the amount ingested annually by an individual, presented in terms of the average daily consumption. This amount was estimated on the basis of individual catches so that the results could be scaled to evaluate potential intakes if the sportsman (fisher/hunter) in the area were successful enough to achieve all of the maximum takes allowed (Section 3.4.2.5 and Table 3.4). Under current conditions, the receptor is a recreational visitor who fishes in the Busch Wildlife Area and hunts in both wildlife areas. The future receptor would be a recreational visitor who also hunts at the site.

The doses associated with intake of radioactive contaminants resulting from fish and game (food) ingestion were calculated as follows:

$$D_i = R_{fi} \times IR_f \times FI_f \times EF \times ED \times DCF_i \tag{3.17}$$

where:

R_{fi} = concentration of radionuclide i in fish/game (pCi/g);

IR_f = fish/game ingestion rate (g/d); and

FI_f = fraction of fish/game ingested from contaminated source.

Intake estimates of chemical contaminants from game ingestion were estimated as follows:

$$I_i = \frac{C_{fi} \times IR_f \times FI_f \times CF_4 \times EF \times ED}{BW \times AT} \tag{3.18}$$

where:

C_{fi} = concentration of contaminant i in the edible portion of fish/game (mg/kg) and

CF_4 = conversion factor (10^{-3} kg/g).

For the fish and game ingestion analysis, the values for R_{fi} and C_{fi} were taken from the exposure point concentrations identified in Section 3.3.5. The value for FI was assumed to be 0.1 for game ingestion, meaning that 10% of the meat ingested is assumed to be from game animals inhabiting the site. This fraction is based on an area ratio (see Section 5.4.2). For fish ingestion, this factor has been addressed by incorporating area-use data into the estimated ingestion rates. The values for IR are given in Section 3.4.2.5, averaged over an EF of 365 events per year. The values for the remaining intake parameters (ED, BW, and ET) are as identified in Table 3.3.

The dose resulting from fish ingestion was estimated with Equation 3.17. The doses estimated for the edible preparations of the different fish from Lakes 34, 35, and 36 are catfish, 0.12 mrem; bass, 0.091 mrem; crappie, 0.071 mrem; and sunfish, 0.38 mrem; the total dose is 0.66 mrem. For comparison, the total dose estimated for the edible preparations of fish from Lake 37 (the control lake) is 0.19 mrem. For the conservative analysis (which uses the maximum detected value for whole fish from the three lakes), the estimated dose is 86 mrem. For comparison, the doses estimated for filleted and whole catfish from the control (background) lake are 0.31 and 0.55 mrem, respectively.

The doses associated with radionuclides from game ingestion were also estimated with Equation 3.17. The doses estimated from ingestion of game for both modeled and measured values are shown in Table 3.20. The doses estimated from the biouptake model are in general agreement with those estimated from measured tissue concentrations for a deer collected from the MSA and waterfowl collected from the raffinate pits. Chemical intakes for game ingestion were estimated with Equation 3.18 and are presented in Table 3.21.

3.4.8 Equations and Intake Estimates for Exposures to Air and to Contaminated Residues and Dust in Buildings

Exposures within the chemical plant buildings were estimated for air, contaminated residues and dust, and external gamma radiation. Contaminants of concern in the building air are uranium-238, thorium-232, radium-228, radium-226, radon-222, radon-220, asbestos, and PCBs (assumed to be attached to airborne dust). Contaminants of concern in the residues are uranium-238, thorium-232, radium-228, radium-226, and PCBs.

3.4.8.1 External Gamma Irradiation

The estimated dose from external gamma irradiation to an individual trespassing into Building 403 was based on an estimated gamma exposure rate of 0.1 mR/h. The dose was calculated by multiplying the length of time an individual is exposed to this radiation field by

the radiation field strength and a DCF of 0.95 mrem/mR. The estimated doses to the current trespasser and the future recreational visitor are 4.8 and 57 mrem, respectively. These doses are based on the assumption of uniform irradiation of the whole body.

3.4.8.2 Inhalation

The doses resulting from inhalation of radon-220, radon-222, and their short-lived decay products were based on the exposure point concentrations presented in Table 3.22. Estimated doses were calculated with Equation 3.14. The estimated doses to a current trespasser and future recreational visitor from inhalation of radon decay products are presented in Table 3.22.

The dose from inhalation of radioactive airborne particulates was calculated on the basis of an exposure point concentration of 0.5 pCi/m^3. This value is within the range of concentrations reported for Building 403 for long-lived alpha particles in air and is typical of the average values reported for other buildings. For purposes of this analysis, the concentration of long-lived alpha particles is attributed to uranium. This is considered a reasonable approach because uranium is the major radionuclide in this building and the inhalation DCFs for uranium isotopes are representative of those for the other alpha-emitting radionuclides in the buildings. The doses to a current trespasser and future recreational visitor were calculated with Equation 3.15. The estimated doses from inhalation of radioactive particulates for the current trespasser and future recreational visitor are presented in Table 3.22.

Intakes of asbestos and PCBs via the air pathway were estimated for the current trespasser and future recreational visitor; the exposure point concentrations were 0.23 fibers/cm^3 and 1.3×10^{-3} mg/m^3, respectively. The scenario-specific values for the exposure and intake variables are presented in Table 3.3. The estimated intakes resulting from inhalation exposure were calculated with Equation 3.16. The intakes for asbestos are 2.3 and 20 fibers/kg-d for the trespasser and future recreational visitor, respectively; the intakes for PCBs are presented in Table 3.23.

3.4.8.3 Ingestion

The doses from incidental ingestion of radionuclides (i.e., uranium-238, thorium-232, radium-228, and radium-226) from contaminated surfaces were estimated from the exposure point concentrations presented in Table 3.22 and Equation 3.10, where R_{si} = concentration of radionuclide i on the interior surfaces of Building 403 (pCi/g). The estimated doses for the current trespasser and future recreational visitor are presented in Table 3.22.

Two algorithms were used to estimate PCB intakes from the bulk sampling and surface swipe characterization data for Building 408 (Section 3.3.6.2). For PCB concentrations reported in mg/kg (bulk samples), Equation 3.11 was used to estimate intakes, where C_{si} = concentration

of PCBs. For PCB concentrations reported in mg/m^2 (surface swipes), intakes were calculated as follows:

$$I_i = \frac{C_{bi} \times CF_5 \times SA \times FR \times EF \times ED}{BW \times AT} \tag{3.19}$$

where:

C_{bi} = concentration of contaminant i in building residue (mg/m^2);

CF_5 = conversion factor (10^{-4} m^2/cm^2); and

FR = fraction of contaminant removed during contact.

The exposure point concentrations were 390 mg/kg and 350 mg/m^2 for the bulk sample and surface swipe data, respectively. (These two measurements were taken from two different areas inside the building with different techniques, so they cannot be directly compared.) The weighted averages of the intakes estimated from these two sets of data are presented in Table 3.23. For these calculations, SA is the skin surface area that could come into contact with the mouth per event, which was assumed to be half of the exposed hand area (475 cm^2/event). The value for FR was assumed to be 0.1.

3.4.9 Multiple Exposure Pathways

The dose and intake estimates presented in Sections 3.4.3 through 3.4.8 were for individual exposure points and routes of exposure and were based on estimates of the RME expected to occur under both current and future land-use conditions. However, the more probable scenario is that an individual would be exposed via more than one route of exposure and/or at more than one exposure point. For example, the on-site trespasser would be exposed simultaneously to site soil and air; and, during the course of 1 year, the same individual might trespass on the site and visit in the surrounding wildlife areas. The possible scenarios associated with exposures to multiple pathways and the risks associated with these exposures are discussed in Sections 5.2.6 and 5.3.6.

TABLE 3.1 Primary Exposure Pathways Evaluated for On-Site Locations[a]

Area	Medium	Exposure Route[b]	Current Worker	Current Trespasser[c]	Future Recreational Visitor[c,d]
Sitewide areas	Soil	External gamma irradiation	X	X	X
		Incidental ingestion	X	X	X
	Air (outdoor)	Inhalation	X	X	X
Raffinate pits	Surface water	Ingestion	-	X	X
	Sludge	External gamma irradiation	-	X	X
		Incidental ingestion	-	X	X
Buildings	Residues	External gamma irradiation	-	X	X
		Incidental ingestion	-	X	X
	Air (indoor)	Inhalation	-	X	X

[a] This table presents information discussed in Sections 3.2.4 and 3.3. Quantitative intake and risk estimates for these primary exposure pathways are presented in Sections 3.4, 5.2, and 5.3.

[b] Exposure to soil through dermal contact was also evaluated for each receptor; however, because considerable uncertainty is associated with estimating intakes and risks for the dermal pathway, results are presented separately (Section 5.5).

[c] Exposures associated with swimming (i.e., inhalation, external gamma irradiation, and incidental ingestion of and dermal contact with surface water and sludge) were also evaluated for this receptor for purposes of completeness. However, because considerable uncertainty is associated with estimating intakes and risks for the dermal pathway and the probability of such exposures is expected to be low, results are presented separately (Section 5.5).

[d] A sportsman was also evaluated, as a variation of the recreational visitor. On-site exposures would be the same as shown here, with the addition of ingestion of game animals that had inhabited the site.

TABLE 3.2 Primary Exposure Pathways Evaluated for Off-Site Locations[a]

Area	Medium	Exposure Route[b]
Vicinity properties[c]	Soil	External gamma irradiation Incidental ingestion
Southeast Drainage	Surface water	Ingestion
	Sediment[d]	External gamma irradiation Incidental ingestion
Burgermeister Spring	Surface water	Ingestion
Lakes 34, 35, and 36	Surface water	Ingestion
	Sediment[d]	External gamma irradiation Incidental ingestion

[a] This table presents information discussed in Sections 3.2.4 and 3.3. Exposures were evaluated for a hypothetical current and future recreational visitor. Also evaluated was a sportsman who was assumed to be exposed to surface water and sediment/soil at Lakes 34, 35, and 36 by the routes indicated in this table; the sportsman would also ingest fish caught from these lakes. Quantitative intake and risk estimates for these primary exposure pathways are presented in Sections 3.4, 5.2, 5.3, and 5.4.

[b] Exposure to soil through dermal contact was also evaluated; however, because considerable uncertainty is associated with estimating intakes and risks for the dermal pathway, results are presented separately (Section 5.5). Exposures associated with swimming (i.e., inhalation, external gamma irradiation, and incidental ingestion of and dermal contact with surface water and sediment) were also evaluated for purposes of completeness. However, because considerable uncertainty is associated with estimating intakes and risks for the dermal pathway and the probability of such exposures is expected to be low, results are presented separately (Section 5.5).

[c] Soil vicinity properties; the Southeast Drainage is addressed separately.

[d] Also includes soil at these locations.

TABLE 3.3 Exposure Scenario Assumptions and Intake Parameters[a]

Parameter	Variable[b]	Current Worker	Current Trespasser	Current/Future Recreational Visitor[c]
Average body weight (kg)	BW	70	50	70
Inhalation rate (m^3/h)	IR_a	2.5	2.1	2.1
Incidental ingestion of contaminated soil, sediment, sludge, or residue (mg/event or mg/d)	IR_s	240	80	120[d]
Ingestion of surface water (L/event or L/d)	IR_w	-[e]	0.2	0.2
Exposure time (h/event or h/d)	ET	8	1	4[f]
Exposure frequency (events/yr or d/yr)	EF	200	5	20
Exposure duration (yr)	ED	10	10	30

[a] This table presents information discussed in Section 3.4.2.

[b] The listed variables are used in Equations 3.9 through 3.19.

[c] The parameters are the same for both on-site and off-site exposure points.

[d] An ingestion rate of 80 mg/event was assumed for residues and dust in the chemical plant buildings.

[e] A hyphen indicates that the entry is not applicable.

[f] An exposure time of 1 h/event was assumed for the chemical plant buildings.

TABLE 3.4 Intake Assumptions for Game Ingestion[a]

Species	Number per Year[b]	Edible Portion per Individual[c] (lb)	Annual Consumption[d] (lb)	Average Daily Consumption	
				oz	g
Deer	1	60	30	1.3	37
Turkey	6	8	24	1.1	31
Goose	2	2	2	0.088	2.5
Duck	6	1	3	0.13	3.7

[a] This table presents information discussed in Section 3.4.2.5.

[b] Area take limits (Meyer 1990).

[c] Estimated by Meyer (1990).

[d] Assumes that the take is shared by a family of two.

TABLE 3.5 Estimated Doses from Exposure to External Gamma Radiation from Site Soil[a]

	With Background	
Scenario	Gamma Exposure Rate[b] (μR/h)	Effective Dose Equivalent (mrem)
Current worker	13	200
Current trespasser	13	0.62
Future recreational visitor	13	30
	Without Background	
Scenario	Gamma Exposure Rate[b] (μR/h)	Effective Dose Equivalent (mrem)
Current worker	3	46
Current trespasser	3	0.14
Future recreational visitor	3	6.8

[a] This table presents information discussed in Section 3.4.3.1; doses were calculated both with and without the background exposure rate of 10 μR/h included.

[b] Represents the UL_{95} value for exposure rate measurements (see text).

TABLE 3.6 Estimated Doses for a Recreational Visitor from Exposure to Radionuclides in Vicinity Property Soil[a]

Property[b]	Contaminated Surface Area (ha)	Contaminant	Surface Soil Concentration[c] (pCi/g)	Dose (mrem) External Gamma[d]	Dose (mrem) Incidental Ingestion[e]
A1	2.8	Uranium-238	280	9.7	10
A2	0.17	Radium-226	18	76	10
		Uranium-238	53	1.8	1.9
A3	0.02	Uranium-238	62	1.3	2.3
A5	0.19	Radium-226	30	130	17
A6	0.15	Uranium-238	15	0.52	0.55
B3	0.0025	Uranium-238	210	2.9	7.7
B4	0.16	Radium-226	120	510	67
		Thorium-230	44	0.022	1.7
		Uranium-238	13	0.45	0.48
B5	0.0017	Thorium-230	210	0.029	8.0
B6	0.00021	Radium-226	8.3	1.2	4.7
		Thorium-230	350	0.0060	13
		Thorium-232	26	5.1	8.9
		Uranium-238	18	0.021	0.66
B10	0.000050	Thorium-230	36	0.00015	1.4

[a] This table presents information discussed in Sections 3.4.3.1 and 3.4.3.2.

[b] The locations of these vicinity properties are shown in Figure 1.7. A = property identified as owned by the U.S. Department of the Army during designation activities (A4 is actually owned by the Missouri Department of Conservation); B = property owned by the Missouri Department of Conservation. Several of the vicinity properties that have been designated for the Weldon Spring site were not evaluated in this assessment. Vicinity properties A7, B1, B2, and B8 were not evaluated because cleanup has been completed at these locations. Femme Osage Slough (B9) is outside the scope of the current remedial action; it will be addressed in upcoming documentation for remedial action at the quarry. Preliminary information for the Southeast Drainage (A4 and B7) is presented in Table 3.15; this vicinity property will also be addressed in upcoming documentation.

[c] Represents the average concentration except for property B6, for which the maximum concentration is given.

[d] The dose from external gamma radiation is given as the effective dose equivalent (see Section 3.4.1) and was calculated with Equation 3.9. This equation will overpredict the dose for areas less than 0.12 ha. Therefore, the external gamma dose estimated for the small vicinity properties — A3, B3, B5, B6, and B10 — was modified using area factors provided in Gilbert et al. (1989).

[e] The dose from incidental ingestion is given as the committed effective dose equivalent and was calculated from DCFs given in Table 4.1. The dose for radium-226 includes the contribution from lead-210, and the dose for thorium-232 includes the contribution from radium-228.

TABLE 3.7 Estimated Doses from Incidental Ingestion of Radionuclides in Site Soil[a]

Contaminant[b]	Surface Soil Concentration (pCi/g)	Committed Effective Dose Equivalent[c] (mrem)		
		Current Worker	Current Trespasser	Future Recreational Visitor
Actinium-227	1.2	8.6	0.072	1.3
Lead-210	11	35	0.29	5.3
Protactinium-231	2.5	13	0.11	2.0
Radium-226	11	5.8	0.048	0.87
Radium-228	5.9	5.7	0.047	0.85
Thorium-230	5.2	1.3	0.011	0.20
Thorium-232	5.9	7.9	0.066	1.2
Uranium-235	3.0	0.36	0.0030	0.054
Uranium-238	65	16	0.13	2.4

[a] This table presents information discussed in Section 3.4.3.2.

[b] Information from the radiological source term analysis was used to estimate the concentrations of radionuclides that were not directly measured (Table 2.3).

[c] The committed effective dose equivalent was calculated from dose conversion factors given in Table 4.1.

TABLE 3.8 Estimated Average Daily Intake of Chemicals from Incidental Ingestion of Site Soil[a]

Contaminant	Surface Soil Concentration (mg/kg)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)		
		Current Worker	Current Trespasser	Future Recreational Visitor
Metals				
Antimony	5.0	9.3×10^{-6}	1.1×10^{-7}	4.6×10^{-7}
Arsenic	1.2×10^{1}	2.2×10^{-5}	2.6×10^{-7}	1.1×10^{-6}
Barium	2.2×10^{2}	4.2×10^{-4}	4.9×10^{-6}	2.1×10^{-5}
Beryllium	8.0×10^{-1}	1.5×10^{-6}	1.7×10^{-8}	7.5×10^{-8}
Cadmium	9.4×10^{-1}	1.8×10^{-6}	2.1×10^{-8}	8.8×10^{-8}
Chromium III	2.2×10^{1}	4.1×10^{-5}	4.8×10^{-7}	2.1×10^{-6}
Chromium VI	2.4	4.6×10^{-6}	5.4×10^{-8}	2.3×10^{-7}
Cobalt	1.1×10^{1}	2.1×10^{-5}	2.4×10^{-7}	1.0×10^{-6}
Copper	1.6×10^{1}	3.1×10^{-5}	3.6×10^{-7}	1.5×10^{-6}
Lead	5.6×10^{1}	1.1×10^{-4}	1.2×10^{-6}	5.3×10^{-6}
Lithium	8.0	1.5×10^{-5}	1.7×10^{-7}	7.5×10^{-7}
Manganese	6.8×10^{2}	1.3×10^{-3}	1.5×10^{-5}	6.4×10^{-5}
Mercury	1.1×10^{-1}	2.1×10^{-7}	2.4×10^{-9}	1.0×10^{-8}
Molybdenum	1.9×10^{1}	3.6×10^{-5}	4.2×10^{-7}	1.8×10^{-6}
Nickel	2.1×10^{1}	3.9×10^{-5}	4.6×10^{-7}	2.0×10^{-6}
Selenium	2.8	5.3×10^{-6}	6.2×10^{-8}	2.6×10^{-7}
Silver	1.2	2.2×10^{-6}	2.5×10^{-8}	1.1×10^{-7}
Thallium	1.1	2.0×10^{-6}	2.4×10^{-8}	1.0×10^{-7}
Uranium, total	2.0×10^{2}	3.7×10^{-4}	4.3×10^{-6}	1.8×10^{-5}
Vanadium	3.5×10^{1}	6.6×10^{-5}	7.7×10^{-7}	3.3×10^{-6}
Zinc	6.0×10^{1}	1.1×10^{-4}	1.3×10^{-6}	5.6×10^{-6}
Inorganic anions				
Fluoride	5.7	1.1×10^{-5}	1.3×10^{-7}	5.4×10^{-7}
Nitrate	4.3×10^{1}	8.2×10^{-5}	9.5×10^{-7}	4.1×10^{-6}
Nitrite	3.8×10^{-1}	7.1×10^{-7}	8.3×10^{-9}	3.6×10^{-8}
PAHs				
Acenaphthene	4.2×10^{-2}	7.9×10^{-8}	9.3×10^{-10}	4.0×10^{-9}
Anthracene	7.6×10^{-2}	1.4×10^{-7}	1.7×10^{-9}	7.1×10^{-9}
Benz(a)anthracene	1.9×10^{-1}	3.5×10^{-7}	4.1×10^{-9}	1.7×10^{-8}
Benzo(b)fluoranthene	1.0×10^{-1}	1.9×10^{-7}	2.2×10^{-9}	9.6×10^{-9}
Benzo(k)fluoranthene	8.7×10^{-2}	1.6×10^{-7}	1.9×10^{-9}	8.1×10^{-9}
Benzo(g,h,i)perylene	4.7×10^{-2}	8.8×10^{-8}	1.0×10^{-9}	4.4×10^{-9}
Benzo(a)pyrene	1.1×10^{-1}	2.1×10^{-7}	2.5×10^{-9}	1.1×10^{-8}
Chrysene	1.8×10^{-1}	3.4×10^{-7}	4.0×10^{-9}	1.7×10^{-8}
Fluoranthene	2.5×10^{-1}	4.7×10^{-7}	5.5×10^{-9}	2.3×10^{-8}
Fluorene	3.6×10^{-2}	6.7×10^{-8}	7.8×10^{-10}	3.3×10^{-9}
Indeno(1,2,3-cd)pyrene	7.1×10^{-2}	1.3×10^{-7}	1.6×10^{-9}	6.7×10^{-9}
2-Methylnaphthalene	1.1×10^{-1}	2.1×10^{-7}	2.5×10^{-9}	1.1×10^{-8}
Naphthalene	4.0×10^{-2}	7.5×10^{-8}	8.8×10^{-10}	3.8×10^{-9}
Phenanthrene	2.8×10^{-1}	5.2×10^{-7}	6.1×10^{-9}	2.6×10^{-8}
Pyrene	4.4×10^{-1}	8.3×10^{-7}	9.6×10^{-9}	4.1×10^{-8}
PCBs	1.9	3.7×10^{-6}	4.3×10^{-8}	1.8×10^{-7}

TABLE 3.8 (Cont.)

Contaminant	Surface Soil Concentration (mg/kg)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)		
		Current Worker	Current Trespasser	Future Recreational Visitor
Nitroaromatic compounds				
DNB	5.8×10^{-2}	1.1×10^{-7}	1.3×10^{-9}	5.5×10^{-9}
2,4-DNT	1.2×10^{-2}	2.2×10^{-8}	2.5×10^{-10}	1.1×10^{-9}
2,6-DNT	2.2×10^{-2}	4.1×10^{-8}	4.8×10^{-10}	2.1×10^{-9}
NB	2.2×10^{-2}	4.2×10^{-8}	4.8×10^{-10}	2.1×10^{-9}
TNB	3.5×10^{-2}	6.5×10^{-8}	7.6×10^{-10}	3.3×10^{-9}
TNT	1.9×10^{-2}	3.5×10^{-8}	4.1×10^{-10}	1.8×10^{-9}

Contaminant	Surface Soil Concentration (mg/kg)	Estimated Daily Intake Averaged over Lifetime[b] (mg/kg-d)		
		Current Worker	Current Trespasser	Future Recreational Visitor
Metals				
Arsenic	1.2×10^{1}	3.1×10^{-6}	3.7×10^{-8}	4.7×10^{-7}
Beryllium	8.0×10^{-1}	2.1×10^{-7}	2.5×10^{-9}	3.2×10^{-8}
Lead	5.6×10^{1}	1.5×10^{-5}	1.8×10^{-7}	2.3×10^{-6}
PAHs[c]	7.4×10^{-1}	2.0×10^{-7}	2.3×10^{-9}	3.0×10^{-8}
PCBs	1.9	5.2×10^{-7}	6.1×10^{-9}	7.8×10^{-8}
Nitroaromatic compounds				
2,4-DNT	1.2×10^{-2}	3.1×10^{-9}	3.6×10^{-11}	4.7×10^{-10}
2,6-DNT	2.2×10^{-2}	5.9×10^{-9}	6.9×10^{-11}	8.8×10^{-10}
TNT	1.9×10^{-2}	5.0×10^{-9}	5.9×10^{-11}	7.5×10^{-10}

[a] This table presents information discussed in Section 3.4.3.2.

[b] Estimated for carcinogenic contaminants only.

[c] The carcinogenic PAHs are benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, and indeno(1,2,3-cd)pyrene.

TABLE 3.9 Estimated Doses from Ingestion of Radionuclides in Raffinate Pit Surface Water[a]

Contaminant	Surface Water Concentration[b] (pCi/L)	Committed Effective Dose Equivalent[c] (mrem)	
		Current Trespasser	Future Recreational Visitor
Radium-226	130	1.4	17
Radium-228[d]	25	0.5	6.0
Thorium-230	760	4.0	48
Thorium-232	7.6	0.21	2.6
Uranium-235[e]	60	0.15	1.8
Uranium-238[e]	1,300	6.6	80

[a] This table presents information discussed in Section 3.4.4.

[b] Represents the maximum concentration measured in the four pits combined.

[c] The committed effective dose equivalent was calculated from DCFs given in Table 4.1.

[d] The dose for radium-228 includes the contribution from thorium-228.

[e] Derived from measurements of total uranium.

TABLE 3.10 Estimated Doses for a Recreational Visitor from Ingestion of Radionuclides in Surface Water at Off-Site Locations[a]

Source	Contaminant[b]	Surface Water Concentration[c] (pCi/L)	Committed Effective Dose Equivalent[d] (mrem)
Lakes 34, 35, and 36	Uranium-235	5.4	0.16
	Uranium-238	120	7.3
Burgermeister Spring	Thorium-230	8.0	0.51
	Uranium-235	5.4	0.16
	Uranium-238	120	7.3
Southeast Drainage	Uranium-235	14	0.42
	Uranium-238	290	18

[a] This table presents information discussed in Section 3.4.4.

[b] Concentrations of uranium-235 and uranium-238 were derived from measurements of total uranium.

[c] Concentrations for Lakes 34, 35, and 36 and Burgermeister Spring were determined from the maximum concentration measured in surface water from each location; for the Southeast Drainage, half the maximum concentration was used (see text).

[d] The committed effective dose equivalent was calculated from DCFs given in Table 4.1.

TABLE 3.11 Estimated Average Daily Intake of Chemicals from Ingestion of Raffinate Pit Surface Water[a]

Contaminant[b]	Surface Water Concentration[c] (mg/L)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)	
		Current Trespasser	Future Recreational Visitor
Metals			
Antimony	4.0×10^{-1}	2.2×10^{-5}	6.2×10^{-5}
Arsenic	1.2×10^{-1}	6.5×10^{-6}	1.9×10^{-5}
Barium	ND[d]	-	-
Beryllium	8.0×10^{-3}	4.4×10^{-7}	1.3×10^{-6}
Cadmium	3.7×10^{-2}	2.0×10^{-6}	5.8×10^{-6}
Chromium III	1.7×10^{-1}	9.3×10^{-6}	2.7×10^{-5}
Chromium VI	1.9×10^{-2}	1.0×10^{-6}	3.0×10^{-6}
Cobalt	ND	-	-
Copper	4.5×10^{-2}	2.5×10^{-6}	7.0×10^{-6}
Lead	3.6×10^{-1}	2.0×10^{-5}	5.6×10^{-5}
Lithium	4.5	2.4×10^{-4}	7.0×10^{-4}
Manganese	3.3×10^{-2}	1.8×10^{-6}	5.2×10^{-6}
Mercury	2.9×10^{-4}	1.6×10^{-8}	4.5×10^{-8}
Molybdenum	4.1	2.2×10^{-4}	6.3×10^{-4}
Nickel	1.7×10^{-1}	9.5×10^{-6}	2.7×10^{-5}
Selenium	2.2×10^{-1}	1.2×10^{-5}	3.4×10^{-5}
Silver	4.0×10^{-2}	2.2×10^{-6}	6.3×10^{-6}
Thallium	ND	-	-
Uranium, total	4.0	2.2×10^{-4}	6.2×10^{-4}
Vanadium	2.1	1.2×10^{-4}	3.3×10^{-4}
Zinc	6.0×10^{-2}	3.3×10^{-6}	9.3×10^{-6}
Inorganic anions			
Fluoride	1.9×10^{1}	1.0×10^{-3}	3.0×10^{-3}
Nitrate	7.2×10^{3}	4.0×10^{-1}	1.1
Nitrite	NA[d]	-	-

Contaminant[b]	Surface Water Concentration[c] (mg/L)	Estimated Daily Intake Averaged over Lifetime[e] (mg/kg-d)	
		Current Trespasser	Future Recreational Visitor
Metals			
Arsenic	1.2×10^{-1}	9.3×10^{-7}	8.0×10^{-6}
Beryllium	8.0×10^{-3}	6.3×10^{-8}	5.4×10^{-7}
Lead	3.6×10^{-1}	2.8×10^{-6}	2.4×10^{-5}

[a] This table presents information discussed in Section 3.4.4.

[b] Surface water from the raffinate pits was also analyzed for volatile and semivolatile organic compounds, PCBs, and pesticides, but none were detected.

[c] Represents the maximum concentration measured in the four pits combined.

[d] ND = not detected; NA = not assayed.

[e] Estimated for carcinogenic contaminants only.

TABLE 3.12 Estimated Average Daily Intake of Chemicals for a Recreational Visitor from Ingestion of Surface Water at Lakes 34, 35, and 36[a]

Contaminant[b]	Surface Water Concentration[c] (mg/L)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)
Metals		
Antimony	ND[d]	-
Arsenic	2.9×10^{-2}	4.5×10^{-6}
Barium	ND	-
Beryllium	ND	-
Cadmium	ND	-
Chromium III	1.7×10^{-2}	2.7×10^{-6}
Chromium VI	1.9×10^{-3}	3.0×10^{-7}
Cobalt	ND	-
Copper	ND	-
Lead	9.5×10^{-3}	1.5×10^{-6}
Lithium	ND	-
Manganese	8.7×10^{-1}	1.4×10^{-4}
Mercury	1.3×10^{-3}	2.0×10^{-7}
Molybdenum	ND	-
Nickel	ND	-
Selenium	ND	-
Silver	ND	-
Thallium	3.3×10^{-2}	5.2×10^{-6}
Uranium, total	3.5×10^{-1}	5.5×10^{-5}
Vanadium	ND	-
Zinc	7.8×10^{-2}	1.2×10^{-5}
Inorganic anions		
Fluoride	6.0×10^{-1}	9.4×10^{-5}
Nitrate	8.9	1.4×10^{-3}
Nitrite	NA[d]	-
Nitroaromatic compounds		
DNB	ND	-
2,4-DNT	5.8×10^{-4}	9.1×10^{-8}
2,6-DNT	ND	-
NB	ND	-
TNB	ND	-
TNT	ND	-

TABLE 3.12 (Cont.)

Contaminant[b]	Surface Water Concentration[c] (mg/L)	Estimated Daily Intake Averaged over Lifetime[e] (mg/kg-d)
Metals		
Arsenic	2.9×10^{-2}	1.9×10^{-6}
Beryllium	ND	-
Lead	9.5×10^{-3}	6.4×10^{-7}
Nitroaromatic compounds		
2,4-DNT	5.8×10^{-4}	3.9×10^{-8}
2,6-DNT	ND	-
TNT	ND	-

[a] This table presents information discussed in Section 3.4.4.

[b] Surface water from the lakes was also analyzed for PAHs and PCBs, but none were detected.

[c] Represents the maximum concentration measured in the three lakes combined.

[d] ND = not detected; NA = not assayed.

[e] Estimated for carcinogenic contaminants only.

TABLE 3.13 Estimated Average Daily Intake of Chemicals for a Recreational Visitor from Ingestion of Surface Water at Off-Site Locations[a]

Contaminant[b]	Burgermeister Spring: Surface Water Concentration[c] (mg/L)	Burgermeister Spring: Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)	Southeast Drainage: Surface Water Concentration[c] (mg/L)	Southeast Drainage: Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)
Metals				
Antimony	ND[d]	-	7.6×10^{-2}	1.2×10^{-5}
Arsenic	ND	-	ND	-
Barium	7.8×10^{-2}	1.2×10^{-5}	1.1×10^{-1}	1.6×10^{-5}
Beryllium	ND	-	ND	-
Cadmium	ND	-	ND	-
Chromium III	2.3×10^{-2}	3.7×10^{-6}	2.3×10^{-2}	3.7×10^{-6}
Chromium VI	2.6×10^{-3}	4.1×10^{-7}	2.6×10^{-3}	4.1×10^{-7}
Cobalt	ND	-	ND	-
Copper	ND	-	ND	-
Lead	ND	-	1.5×10^{-2}	2.4×10^{-6}
Lithium	ND	-	ND	-
Manganese	3.6×10^{-2}	5.6×10^{-6}	5.7×10^{-2}	8.9×10^{-6}
Mercury	3.7×10^{-4}	5.8×10^{-8}	ND	-
Molybdenum	2.2×10^{-2}	3.4×10^{-6}	4.2×10^{-2}	6.6×10^{-6}
Nickel	ND	-	ND	-
Selenium	ND	-	ND	-
Silver	ND	-	6.0×10^{-3}	9.4×10^{-7}
Thallium	ND	-	ND	-
Uranium, total	3.5×10^{-1}	5.5×10^{-5}	8.8×10^{-1}	1.4×10^{-4}
Vanadium	ND	-	ND	-
Zinc	3.1×10^{-2}	4.9×10^{-6}	3.1×10^{-2}	4.9×10^{-6}
Inorganic anions				
Fluoride	3.1×10^{-1}	4.9×10^{-5}	6.0×10^{-1}	9.4×10^{-5}
Nitrate	2.0×10^{2}	3.2×10^{-2}	2.6×10^{2}	4.1×10^{-2}
Nitrite	NA[d]	-	4.0×10^{-2}	6.3×10^{-6}
Nitroaromatic compounds				
DNB	1.8×10^{-4}	2.8×10^{-8}	8.1×10^{-4}	1.3×10^{-7}
2,4-DNT	1.4×10^{-3}	2.2×10^{-7}	1.1×10^{-2}	1.7×10^{-6}
2,6-DNT	1.8×10^{-2}	2.8×10^{-6}	1.1×10^{-2}	1.7×10^{-6}
NB	ND	-	8.7×10^{-4}	1.4×10^{-7}
TNB	2.0×10^{-5}	3.1×10^{-9}	8.4×10^{-4}	1.3×10^{-7}
TNT	3.0×10^{-4}	4.7×10^{-8}	1.1×10^{-1}	1.7×10^{-5}

TABLE 3.13 (Cont.)

Contaminant[b]	Burgermeister Spring: Surface Water Concentration[c] (mg/L)	Burgermeister Spring: Estimated Daily Intake Averaged over Lifetime[e] (mg/kg-d)	Southeast Drainage: Surface Water Concentration[c] (mg/L)	Southeast Drainage: Estimated Daily Intake Averaged over Lifetime[e] (mg/kg-d)
Metals				
Arsenic	ND	-	ND	-
Beryllium	ND	-	ND	-
Lead	ND	-	1.5×10^{-2}	1.0×10^{-6}
Nitroaromatic compounds				
2,4-DNT	1.4×10^{-3}	9.7×10^{-8}	1.1×10^{-2}	7.4×10^{-7}
2,6-DNT	1.8×10^{-2}	1.2×10^{-6}	1.1×10^{-2}	7.4×10^{-7}
TNT	3.0×10^{-4}	2.0×10^{-8}	1.1×10^{-1}	7.4×10^{-6}

[a] This table presents information discussed in Section 3.4.4.

[b] Surface water from Burgermeister Spring and the Southeast Drainage was also analyzed for PAHs and PCBs, but none were detected.

[c] Except for uranium, determined from the maximum concentration measured. For uranium in the Southeast Drainage, half the maximum was used (see text).

[d] ND = not detected; NA = not assayed.

[e] Estimated for carcinogenic contaminants only.

TABLE 3.14 Estimated Doses from Incidental Ingestion of Radionuclides in Raffinate Pit Sludge[a]

Contaminant	Sludge Concentration[b] (pCi/g)	Committed Effective Dose Equivalent[c] (mrem)	
		Current Trespasser	Future Recreational Visitor
Actinium-227	990	59	1,100
Lead-210	1,100	29	530
Protactinium-231	1,200	53	950
Radium-226	1,700	7.5	130
Radium-228[d]	1,400	11	200
Thorium-230	34,000	72	1,300
Thorium-232	1,400	16	280
Uranium-235[e]	76	0.076	1.4
Uranium-238[e]	1,700	3.5	62

[a] This table presents information discussed in Section 3.4.5.2.

[b] Represents the maximum concentration measured in the four pits combined. Information from the radiological source term analysis was used to estimate the concentrations of radionuclides that were not directly measured (Table 2.3).

[c] The committed effective dose equivalent was calculated from DCFs given in Table 4.1.

[d] The dose given for radium-228 includes the contribution from thorium-228.

[e] Derived from measurements of total uranium.

TABLE 3.15 Estimated Doses for a Recreational Visitor from Incidental Ingestion of Radionuclides in Sediment at Off-Site Locations[a]

Source	Contaminant	Sediment Concentration[b] (pCi/g)	Committed Effective Dose Equivalent[c] (mrem)
Lakes 34, 35, and 36	Uranium-235[d]	5.4	0.097
	Uranium-238[d]	120	4.4
Southeast Drainage	Radium-226[e]	15	8.4
	Thorium-230	8,500	320
	Thorium-232[e]	9.7	3.4
	Uranium-235[f]	3.6	0.065
	Uranium-238	79	2.9

[a] This table presents information discussed in Section 3.4.5.2.

[b] The sediment concentration for Lakes 34, 35, and 36 is the maximum value measured in the three lakes combined; the concentration for the Southeast Drainage is the drainagewide UL_{95}.

[c] The committed effective dose equivalent was calculated from DCFs given in Table 4.1.

[d] Derived from measurements of total uranium.

[e] The dose for radium-226 includes the contribution from lead-210, and the dose for thorium-232 includes the contribution from radium-228 and thorium-228.

[f] Derived from measurements of uranium-238.

TABLE 3.16 Estimated Average Daily Intake of Chemicals from Incidental Ingestion of Raffinate Pit Sludge[a]

Contaminant[b]	Sludge Concentration[c] (mg/kg)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)	
		Current Trespasser	Future Recreational Visitor
Metals			
Antimony	8.7×10^{1}	1.9×10^{-6}	8.2×10^{-6}
Arsenic	1.1×10^{3}	2.3×10^{-5}	1.0×10^{-4}
Barium	7.7×10^{3}	1.7×10^{-4}	7.3×10^{-4}
Beryllium	2.5×10^{1}	5.6×10^{-7}	2.4×10^{-6}
Cadmium	1.4×10^{1}	3.0×10^{-7}	1.3×10^{-6}
Chromium III	1.5×10^{2}	3.3×10^{-6}	1.4×10^{-5}
Chromium VI	1.7×10^{1}	3.7×10^{-7}	1.6×10^{-6}
Cobalt	4.4×10^{1}	9.7×10^{-7}	4.1×10^{-6}
Copper	5.1×10^{2}	1.1×10^{-5}	4.8×10^{-5}
Lead	6.4×10^{2}	1.4×10^{-5}	6.0×10^{-5}
Lithium	1.2×10^{2}	2.7×10^{-6}	1.1×10^{-5}
Manganese	3.0×10^{3}	6.6×10^{-5}	2.8×10^{-4}
Mercury	1.5×10^{1}	3.4×10^{-7}	1.4×10^{-6}
Molybdenum	1.6×10^{3}	3.5×10^{-5}	1.5×10^{-4}
Nickel	8.8×10^{3}	1.9×10^{-4}	8.3×10^{-4}
Selenium	8.1×10^{1}	1.8×10^{-6}	7.6×10^{-6}
Silver	5.0	1.1×10^{-7}	4.7×10^{-7}
Thallium	5.8×10^{1}	1.3×10^{-6}	5.4×10^{-6}
Uranium, total	5.0×10^{3}	1.1×10^{-4}	4.7×10^{-4}
Vanadium	8.7×10^{3}	1.9×10^{-4}	8.1×10^{-4}
Zinc	1.6×10^{3}	3.5×10^{-5}	1.5×10^{-4}
Inorganic anions			
Fluoride	1.7×10^{2}	3.6×10^{-6}	1.5×10^{-5}
Nitrate	1.6×10^{5}	3.5×10^{-3}	1.5×10^{-2}
Nitrite	1.6×10^{3}	3.6×10^{-5}	1.5×10^{-4}
PCBs	1.1×10^{1}	2.5×10^{-7}	1.1×10^{-6}

TABLE 3.16 (Cont.)

Contaminant[b]	Sludge Concentration[c] (mg/kg)	Estimated Daily Intake Averaged over Lifetime[d] (mg/kg-d)	
		Current Trespasser	Future Recreational Visitor
Metals			
Arsenic	1.1×10^{3}	3.3×10^{-6}	4.3×10^{-5}
Beryllium	2.5×10^{1}	8.0×10^{-8}	1.0×10^{-6}
Lead	6.4×10^{2}	2.0×10^{-6}	2.6×10^{-5}
PCBs	1.1×10^{1}	3.6×10^{-8}	4.6×10^{-7}

[a] This table presents information discussed in Section 3.4.5.2.

[b] Sludge from the raffinate pits was also analyzed for PAHs and nitroaromatic compounds, but none were detected.

[c] Represents the maximum concentration measured in the four pits combined.

[d] Estimated for carcinogenic contaminants only.

TABLE 3.17 Estimated Average Daily Intake of Chemicals for a Recreational Visitor from Incidental Ingestion of Sediment at Off-Site Locations[a]

Contaminant[b]	Lakes 34, 35, and 36		Southeast Drainage	
	Sediment Concentration[c] (mg/kg)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)	Sediment Concentration[d] (mg/kg)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)
Metals				
Antimony	ND[e]	-	ND	-
Arsenic	1.0×10^1	9.4×10^{-7}	1.4×10^1	1.4×10^{-6}
Barium	1.6×10^2	1.5×10^{-5}	3.3×10^2	3.1×10^{-5}
Beryllium	ND	-	ND	-
Cadmium	ND	-	ND	-
Chromium III	1.4×10^1	1.3×10^{-6}	2.3×10^1	2.1×10^{-6}
Chromium VI	1.5	1.4×10^{-7}	2.5	2.3×10^{-7}
Cobalt	1.8×10^1	1.7×10^{-6}	3.7×10^1	3.5×10^{-6}
Copper	1.3×10^1	1.2×10^{-6}	1.7×10^2	1.6×10^{-5}
Lead	1.9×10^1	1.8×10^{-6}	4.8×10^1	4.5×10^{-6}
Lithium	NA[e]	-	NA	-
Manganese	1.1×10^3	1.1×10^{-4}	6.5×10^3	6.1×10^{-4}
Mercury	ND	-	ND	-
Molybdenum	NA	-	NA	-
Nickel	1.3×10^1	1.2×10^{-6}	6.6×10^1	6.2×10^{-6}
Selenium	ND	-	ND	-
Silver	ND	-	ND	-
Thallium	ND	-	ND	-
Uranium, total	3.5×10^2	3.3×10^{-5}	2.4×10^2	2.2×10^{-5}
Vanadium	2.9×10^1	2.7×10^{-6}	7.5×10^1	7.0×10^{-6}
Zinc	4.6×10^1	4.3×10^{-6}	2.2×10^2	2.0×10^{-5}
PCBs	2.0×10^{-1}	1.8×10^{-8}	ND	-

Contaminant[b]	Lakes 34, 35, and 36		Southeast Drainage	
	Sediment Concentration[c] (mg/kg)	Estimated Daily Intake Averaged over Lifetime[f] (mg/kg-d)	Sediment Concentration[d] (mg/kg)	Estimated Daily Intake Averaged over Lifetime[f] (mg/kg-d)
Metals				
Arsenic	1.0×10^1	4.0×10^{-7}	1.4×10^1	5.8×10^{-7}
Beryllium	ND	-	ND	-
Lead	1.9×10^1	7.6×10^{-7}	4.8×10^1	1.9×10^{-6}
PCBs	2.0×10^{-1}	7.9×10^{-9}	ND	-

[a] This table presents information discussed in Section 3.4.5.2.

[b] Surface sediment from the off-site lakes and the Southeast Drainage was also analyzed for PAHs and nitroaromatic compounds, but none were detected.

[c] Represents the maximum concentration measured in the three lakes combined.

[d] Except for uranium, represents the maximum concentration measured in the drainage; for uranium, the listed value is the drainagewide UL_{95}.

[e] ND = not detected; NA = not assayed.

[f] Estimated for carcinogenic contaminants only.

TABLE 3.18 Estimated Doses from Inhalation of Airborne Radioactive Particulates On-Site[a]

Contaminant[b]	Air Concentration[c] (pCi/m^3)	Committed Effective Dose Equivalent[d] (mrem, except as noted)		
		Current Worker	Current Trespasser	Future Recreational Visitor
Actinium-227	2.8×10^{-5}	7.5	2.0×10^{-2}	9.5×10^{-1}
Lead-210	4.8×10^{-5}	4.0×10^{-2}	1.1×10^{-4}	5.1×10^{-3}
Protactinium-231	5.8×10^{-5}	3.0	7.9×10^{-3}	3.8×10^{-1}
Radium-226	4.8×10^{-5}	1.5×10^{-2}	4.0×10^{-5}	1.9×10^{-3}
Radium-228	4.5×10^{-5}	5.6×10^{-1}	1.5×10^{-3}	7.0×10^{-2}
Radon-222[e]	5.6×10^{-3}	1.1	2.9×10^{-3}	1.4×10^{-1}
Thorium-230	4.0×10^{-5}	5.1×10^{-1}	1.3×10^{-3}	6.5×10^{-2}
Thorium-232	4.5×10^{-5}	2.9	7.6×10^{-3}	3.6×10^{-1}
Uranium-235	7.0×10^{-5}	3.4×10^{-1}	8.8×10^{-4}	4.2×10^{-2}
Uranium-238	1.5×10^{-3}	1.5×10^{1}	3.9×10^{-2}	1.9

[a] This table presents information discussed in Section 3.4.6.

[b] Information from the radiological source term analysis was used to estimate the concentrations of radionuclides that were not directly measured (Table 2.3).

[c] The radon concentration (in WL) was determined from the sitewide surface UL_{95} for radium-226. The concentrations of other radionuclides were determined from the estimated concentration of respirable particulates (≤10 μm) that could be generated from an exposed area in the southern portion of the site, considering UL_{95} values for radionuclides in surface soil from that area.

[d] The committed effective dose equivalent was calculated from DCFs given in Table 4.1. The dose from radon (in WLM) was calculated using Equation 3.14.

[e] Given in units of WL for air concentration and WLM for committed effective dose equivalent. Although radon-222 is a gas, its particulate decay products are the primary contributors to the inhalation dose.

TABLE 3.19 Estimated Average Daily Intake of Chemicals from Inhalation of Airborne Particulates On-Site[a]

Contaminant[b]	Air Concentration[c] (mg/m^3)	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)		
		Current Worker	Current Trespasser	Future Recreational Visitor
Metals				
Antimony	2.1×10^{-7}	3.4×10^{-8}	1.2×10^{-10}	1.4×10^{-9}
Arsenic	5.1×10^{-7}	7.9×10^{-8}	2.9×10^{-10}	3.3×10^{-9}
Barium	4.7×10^{-6}	7.4×10^{-7}	2.7×10^{-9}	3.1×10^{-8}
Beryllium	2.5×10^{-8}	3.9×10^{-9}	1.4×10^{-11}	1.6×10^{-10}
Cadmium	2.8×10^{-8}	4.4×10^{-9}	1.6×10^{-11}	1.8×10^{-10}
Chromium III	4.9×10^{-7}	7.7×10^{-8}	2.8×10^{-10}	3.2×10^{-9}
Chromium VI	5.5×10^{-8}	8.6×10^{-9}	3.2×10^{-11}	3.6×10^{-10}
Cobalt	3.8×10^{-7}	6.0×10^{-8}	2.2×10^{-10}	2.5×10^{-9}
Copper	4.2×10^{-7}	6.6×10^{-8}	2.4×10^{-10}	2.8×10^{-9}
Lead	5.8×10^{-7}	9.1×10^{-8}	3.3×10^{-10}	3.8×10^{-9}
Lithium	2.1×10^{-7}	3.4×10^{-8}	1.2×10^{-10}	1.4×10^{-9}
Manganese	2.3×10^{-5}	3.6×10^{-6}	1.3×10^{-8}	1.5×10^{-7}
Mercury	5.2×10^{-9}	8.1×10^{-10}	3.0×10^{-12}	3.4×10^{-11}
Molybdenum	5.6×10^{-7}	8.8×10^{-8}	3.2×10^{-10}	3.7×10^{-9}
Nickel	4.5×10^{-7}	7.1×10^{-8}	2.6×10^{-10}	3.0×10^{-9}
Selenium	6.2×10^{-8}	9.7×10^{-9}	3.6×10^{-11}	4.1×10^{-10}
Silver	4.8×10^{-8}	7.5×10^{-9}	2.8×10^{-11}	3.2×10^{-10}
Thallium	1.3×10^{-8}	2.0×10^{-9}	7.2×10^{-12}	8.2×10^{-11}
Uranium, total	3.4×10^{-5}	5.3×10^{-6}	1.9×10^{-8}	2.2×10^{-7}
Vanadium	9.2×10^{-7}	1.4×10^{-7}	5.3×10^{-10}	6.1×10^{-9}
Zinc	1.9×10^{-6}	3.0×10^{-7}	1.1×10^{-9}	1.3×10^{-8}
Inorganic anions				
Fluoride	1.7×10^{-7}	2.7×10^{-8}	9.8×10^{-11}	1.1×10^{-9}
Nitrate	2.4×10^{-6}	3.7×10^{-7}	1.4×10^{-9}	1.5×10^{-8}
Nitrite	6.3×10^{-9}	9.8×10^{-10}	3.6×10^{-12}	4.1×10^{-11}
PCBs	2.4×10^{-7}	3.8×10^{-8}	1.4×10^{-10}	1.6×10^{-9}

TABLE 3.19 (Cont.)

Contaminant[b]	Air Concentration[c] (mg/m^3)	Estimated Daily Intake Averaged over Lifetime[d] (mg/kg-d)		
		Current Worker	Current Trespasser	Future Recreational Visitor
Metals				
Arsenic	5.1×10^{-7}	1.1×10^{-8}	4.2×10^{-11}	1.4×10^{-9}
Beryllium	2.5×10^{-8}	5.5×10^{-10}	2.0×10^{-12}	7.0×10^{-11}
Cadmium	2.8×10^{-8}	6.3×10^{-10}	2.3×10^{-12}	7.9×10^{-11}
Chromium VI	5.5×10^{-8}	1.2×10^{-9}	4.5×10^{-12}	1.5×10^{-10}
Lead	5.8×10^{-7}	1.3×10^{-8}	4.8×10^{-11}	1.6×10^{-9}
Nickel	4.5×10^{-7}	1.0×10^{-8}	3.7×10^{-11}	1.3×10^{-9}
PCBs	2.4×10^{-7}	5.5×10^{-9}	2.0×10^{-11}	6.9×10^{-10}

[a] This table presents information discussed in Section 3.4.6.

[b] Soil from the exposed area potentially subject to fugitive dust emissions was analyzed for PAHs, PCBs, and nitroaromatic compounds, but none were detected.

[c] Determined from the estimated concentration of respirable particulates (≤10 μm) that could be generated from the exposed area in the southern portion of the site, considering contaminant concentrations in surface soil from that area.

[d] Estimated for carcinogenic contaminants only.

TABLE 3.20 Estimated Doses from Ingestion of Game[a]

Contaminant	Tissue Concentration[b] (pCi/g)					
	Duck (Measured)	Duck (Modeled)	Goose (Measured)	Turkey (Modeled)	Deer (Measured)	Deer (Modeled)
Actinium-227	-[c]	9.0×10^{-6}	-	1.8×10^{-5}	-	2.6×10^{-5}
Lead-210	-	1.3×10^{-5}	-	5.3×10^{-6}	-	1.4×10^{-5}
Protactinium-231	-	1.9×10^{-5}	-	3.3×10^{-5}	-	3.9×10^{-5}
Radium-226	1.0×10^{-3}	1.1×10^{-2}	1.0×10^{-3}	2.7×10^{-4}	1.0×10^{-3}	6.5×10^{-4}
Radium-228	1.5×10^{-2}	3.7×10^{-3}	4.0×10^{-3}	1.5×10^{-4}	3.0×10^{-3}	3.5×10^{-4}
Thorium-230	1.2×10^{-2}	4.7×10^{-2}	1.0×10^{-2}	2.1×10^{-4}	3.0×10^{-3}	5.4×10^{-4}
Thorium-232	3.0×10^{-3}	1.1×10^{-3}	1.0×10^{-3}	2.4×10^{-4}	1.0×10^{-3}	6.1×10^{-4}
Uranium-235	1.0×10^{-3}	2.2×10^{-2}	1.0×10^{-3}	4.2×10^{-5}	1.0×10^{-3}	5.6×10^{-5}
Uranium-238	4.0×10^{-3}	2.4×10^{-1}	2.0×10^{-3}	9.1×10^{-4}	1.5×10^{-3}	1.2×10^{-3}
Committed effective dose equivalent[d] (mrem)	1.9×10^{-1}	6.8×10^{-1}	4.9×10^{-2}	8.1×10^{-2}	5.7×10^{-1}	1.9×10^{-1}

[a] This table presents information discussed in Section 3.4.7.

[b] Measured tissue concentrations were taken from the very limited data available for the goose and ducks from the raffinate pits and the deer from the MSA. Modeled tissue concentrations were determined from estimated biouptake and bioconcentration factors. Measured UL_{95} concentrations in surface soil (Table 3.7) were used as model input for the turkey and deer. Averages of detected values for surface water from the raffinate pits were used as model input for the duck and goose. Concentrations modeled for the duck were also used for the goose, for which the estimated dose is 4.6×10^{-1} mrem (determined from the ratio of ingestion rates).

[c] A hyphen indicates that the radionuclide was not measured.

[d] The committed effective dose equivalent was calculated from DCFs given in Table 4.1.

TABLE 3.21 Estimated Average Daily Intake of Chemicals from Ingestion of Game[a]

Contaminant	Estimated Tissue Concentration[b] (mg/kg)			Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)		
	Duck	Turkey	Deer	Duck	Turkey	Deer
Metals						
Antimony	9.5×10^{-5}	3.1×10^{-5}	7.4×10^{-5}	4.8×10^{-9}	1.3×10^{-8}	3.8×10^{-8}
Arsenic	3.0×10^{-5}	3.9×10^{-5}	4.6×10^{-5}	1.5×10^{-9}	1.7×10^{-8}	2.3×10^{-8}
Barium	-[c]	8.4×10^{-5}	1.6×10^{-4}	-	3.6×10^{-8}	8.2×10^{-8}
Beryllium	3.8×10^{-6}	1.5×10^{-6}	6.0×10^{-7}	1.9×10^{-10}	6.4×10^{-10}	3.0×10^{-10}
Cadmium	3.3×10^{-6}	3.9×10^{-6}	1.2×10^{-5}	1.7×10^{-10}	1.6×10^{-9}	6.1×10^{-9}
Chromium III	8.4×10^{-5}	9.8×10^{-5}	9.4×10^{-5}	4.3×10^{-9}	4.2×10^{-8}	4.8×10^{-8}
Chromium VI	9.4×10^{-6}	1.1×10^{-5}	1.0×10^{-5}	4.7×10^{-10}	4.6×10^{-9}	5.3×10^{-9}
Cobalt	-	6.0×10^{-4}	7.7×10^{-4}	-	2.5×10^{-7}	3.9×10^{-7}
Copper	1.8×10^{-4}	5.7×10^{-3}	1.8×10^{-2}	9.0×10^{-9}	2.4×10^{-6}	9.3×10^{-6}
Lead	7.2×10^{-6}	2.7×10^{-5}	6.9×10^{-5}	3.6×10^{-10}	1.2×10^{-8}	3.5×10^{-8}
Lithium	5.9×10^{-3}	1.8×10^{-4}	1.6×10^{-4}	3.0×10^{-7}	7.7×10^{-8}	8.1×10^{-8}
Manganese	4.7×10^{-6}	2.5×10^{-3}	6.8×10^{-3}	2.4×10^{-10}	1.1×10^{-6}	3.5×10^{-6}
Mercury	5.3×10^{-8}	1.2×10^{-6}	3.9×10^{-6}	2.7×10^{-12}	5.2×10^{-10}	2.0×10^{-9}
Molybdenum	8.4×10^{-3}	1.4×10^{-3}	3.9×10^{-3}	4.2×10^{-7}	5.9×10^{-7}	2.0×10^{-6}
Nickel	1.1×10^{-4}	4.5×10^{-4}	1.3×10^{-3}	5.7×10^{-9}	1.9×10^{-7}	6.4×10^{-7}
Selenium	6.1×10^{-4}	2.3×10^{-4}	5.3×10^{-4}	3.1×10^{-8}	9.7×10^{-8}	2.7×10^{-7}
Silver	3.3×10^{-5}	3.9×10^{-5}	1.2×10^{-4}	1.7×10^{-9}	1.6×10^{-8}	5.9×10^{-8}
Thallium	-	7.5×10^{-5}	8.6×10^{-6}	-	3.2×10^{-8}	4.4×10^{-9}
Uranium, total	3.6×10^{-3}	1.3×10^{-2}	3.7×10^{-2}	1.8×10^{-7}	5.4×10^{-6}	1.9×10^{-5}
Vanadium	1.4×10^{-3}	1.9×10^{-4}	1.3×10^{-4}	7.0×10^{-8}	7.9×10^{-8}	6.7×10^{-8}
Zinc	2.2×10^{-3}	8.0×10^{-1}	2.6	1.1×10^{-7}	3.4×10^{-4}	1.3×10^{-3}
Inorganic anions						
Fluoride	2.1×10^{-1}	2.2×10^{-3}	2.6×10^{-3}	1.0×10^{-5}	9.4×10^{-7}	1.3×10^{-6}
Nitrate	NQ[c]	NQ	NQ	NQ	NQ	NQ
Nitrite	NQ	NQ	NQ	NQ	NQ	NQ
PAHs						
Acenaphthene	-	2.0×10^{-9}	6.2×10^{-9}	-	8.3×10^{-13}	3.1×10^{-12}
Anthracene	-	4.4×10^{-9}	1.3×10^{-8}	-	1.9×10^{-12}	6.6×10^{-12}
Benz(a)anthracene	-	5.6×10^{-8}	1.0×10^{-7}	-	2.4×10^{-11}	5.2×10^{-11}
Benzo(b)fluoranthene	-	4.5×10^{-8}	5.9×10^{-8}	-	1.9×10^{-11}	3.0×10^{-11}
Benzo(k)fluoranthene	-	3.8×10^{-8}	5.0×10^{-8}	-	1.6×10^{-11}	2.5×10^{-11}
Benzo(g,h,i)perylene	-	4.6×10^{-8}	3.9×10^{-8}	-	2.0×10^{-11}	2.0×10^{-11}
Benzo(a)pyrene	-	3.2×10^{-8}	6.5×10^{-9}	-	1.3×10^{-11}	3.3×10^{-12}
Chrysene	-	5.5×10^{-8}	1.0×10^{-7}	-	2.3×10^{-11}	5.1×10^{-11}
Fluoranthene	-	3.4×10^{-8}	9.0×10^{-8}	-	1.4×10^{-11}	4.6×10^{-11}
Fluorene	-	2.0×10^{-9}	6.2×10^{-9}	-	8.4×10^{-13}	3.1×10^{-12}
Indeno(1,2,3-cd)pyrene	-	7.1×10^{-8}	6.0×10^{-8}	-	3.0×10^{-11}	3.0×10^{-11}
2-Methylnaphthalene	-	6.5×10^{-8}	2.0×10^{-7}	-	2.7×10^{-11}	1.0×10^{-10}
Naphthalene	-	1.5×10^{-9}	4.9×10^{-9}	-	6.3×10^{-13}	2.5×10^{-12}
Phenanthrene	-	6.1×10^{-8}	1.8×10^{-7}	-	2.6×10^{-11}	9.1×10^{-11}
Pyrene	-	5.3×10^{-8}	1.4×10^{-7}	-	2.3×10^{-11}	7.2×10^{-11}
PCBs	-	6.7×10^{-7}	3.6×10^{-7}	-	2.8×10^{-10}	1.8×10^{-10}
Nitroaromatic compounds						
DNB	-	2.7×10^{-9}	9.1×10^{-8}	-	1.2×10^{-11}	4.6×10^{-11}
2,4-DNT	-	5.2×10^{-9}	1.7×10^{-8}	-	2.2×10^{-12}	8.8×10^{-12}
2,6-DNT	-	9.7×10^{-9}	3.2×10^{-8}	-	4.1×10^{-12}	1.6×10^{-11}
NB	-	9.8×10^{-9}	3.3×10^{-8}	-	4.2×10^{-12}	1.7×10^{-11}
TNB	-	1.8×10^{-8}	5.9×10^{-8}	-	7.5×10^{-12}	3.0×10^{-11}
TNT	-	7.9×10^{-9}	2.6×10^{-8}	-	3.4×10^{-12}	1.3×10^{-11}

TABLE 3.21 (Cont.)

Contaminant	Estimated Tissue Concentration[b] (mg/kg)			Estimated Daily Intake Averaged over Lifetime[d] (mg/kg-d)		
	Duck	Turkey	Deer	Duck	Turkey	Deer
Metals						
Arsenic	3.0×10^{-5}	3.9×10^{-5}	4.6×10^{-5}	6.5×10^{-10}	7.1×10^{-9}	9.9×10^{-9}
Beryllium	3.8×10^{-6}	1.5×10^{-6}	6.0×10^{-7}	8.2×10^{-11}	2.7×10^{-10}	1.3×10^{-10}
Lead	7.2×10^{-6}	2.7×10^{-5}	6.9×10^{-5}	1.6×10^{-10}	4.9×10^{-9}	1.5×10^{-8}
PAHs[e]	-	3.0×10^{-7}	3.8×10^{-7}	-	5.4×10^{-11}	8.2×10^{-11}
PCBs	-	6.7×10^{-7}	3.6×10^{-7}	-	1.2×10^{-10}	7.7×10^{-11}
Nitroaromatic compounds						
2,4-DNT	-	5.2×10^{-9}	1.7×10^{-8}	-	9.4×10^{-13}	3.8×10^{-12}
2,6-DNT	-	9.7×10^{-9}	3.2×10^{-8}	-	1.8×10^{-12}	7.0×10^{-12}
TNT	-	7.9×10^{-9}	2.6×10^{-8}	-	1.4×10^{-12}	5.7×10^{-12}

a This table presents information discussed in Section 3.4.7.

b Modeled from (1) estimated uptake and bioconcentration factors with UL_{95} values for metals and inorganic anions and area-weighted estimates for organic compounds in surface soil and (2) averages of detected values for surface water from the raffinate pits (the latter will overestimate the actual average concentrations). Estimated tissue concentrations for a goose were assumed to be similar to those for a duck; estimated daily intakes for the goose would be two-thirds the value shown for the duck, from the ratio of human ingestion rates (Table 3.4).

c A hyphen indicates that the parameter was not estimated because the analyte was not measured in the raffinate pit water during site characterization. NQ indicates that the uptake was not quantified (see text).

d Estimated for carcinogenic contaminants only.

e The carcinogenic PAHs are benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, and indeno(1,2,3-cd)pyrene.

TABLE 3.22 Estimated Doses from Exposure to Radionuclides in Building 403[a]

Exposure Route	Contaminant	Exposure Point Concentration	Dose: Current Trespasser	Dose: Future Recreational Visitor
Irradiation	External gamma	0.1 mR/h	4.8 mrem[b]	57 mrem[b]
Inhalation	Radon-220[c]	0.61 WL[d]	0.31 WLM[e]	3.8 WLM
	Radon-222	0.001 WL	5.1×10^{-4} WLM	6.2×10^{-3} WLM
	Uranium-235[f]	2.3×10^{-2} pCi/m^3	0.29 mrem[g]	3.5 mrem[g]
	Uranium-238	5×10^{-1} pCi/m^3	13 mrem[g]	160 mrem[g]
Ingestion	Radium-226[c]	81 pCi/g	2.5 mrem[g]	45 mrem[g]
	Thorium-232[c]	2,800 pCi/g	53 mrem[g]	960 mrem[g]
	Uranium-235[f]	550 pCi/g	0.55 mrem[g]	9.9 mrem[g]
	Uranium-238[c]	12,000 pCi/g	24 mrem[g]	440 mrem[g]

[a] This table presents information discussed in Section 3.4.8.

[b] Effective dose equivalent.

[c] Exposure point concentration is the average value of measurements for Building 403. The dose given for radium-226 includes the contribution from lead-210 and that given for thorium-232 includes the contribution from radium-228.

[d] One working level (WL) is defined as any combination of short-lived radon decay products in 1 L of air, without regard to the degree of equilibrium, that will result in the emission of 1.3×10^5 MeV of alpha energy. (The value 1 WL is assumed to be equal to 7.4 pCi/L for radon-220 in equilibrium with its short-lived decay products and 100 pCi/L for radon-222 in equilibrium with its short-lived decay products.)

[e] One working-level month (WLM) is defined as the exposure of an individual having an inhalation rate of 1.2 m^3/h to 1 WL for 170 hours.

[f] The exposure point concentration for uranium-235 was derived from the measurement for uranium-238.

[g] Committed effective dose equivalent; calculated from the DCFs given in Table 4.1.

TABLE 3.23 Estimated Average Daily Intake of PCBs from Building 408[a]

	Estimated Daily Intake Averaged over Exposure Period (mg/kg-d)		Estimated Daily Intake Averaged over Lifetime[b] (mg/kg-d)	
Exposure Route	Current Trespasser	Future Recreational Visitor	Current Trespasser	Future Recreational Visitor
Inhalation	7.3×10^{-7}	2.1×10^{-6}	1.0×10^{-7}	8.9×10^{-7}
Ingestion	3.2×10^{-4}	9.1×10^{-4}	4.5×10^{-5}	3.9×10^{-4}

[a] This table presents information discussed in Section 3.4.8; see text for exposure point concentrations.

[b] Estimated for carcinogenic contaminants only.

4 TOXICITY ASSESSMENT

Toxicity values for quantifying the potential health effects associated with exposure to the radioactive and chemical contaminants of concern at the Weldon Spring site are identified in this chapter. Risk factors for assessing the carcinogenic effects of radioactive contaminants are described in Section 4.1. Toxicity values for assessing the noncarcinogenic and carcinogenic effects of chemical contaminants are described in Sections 4.2 and 4.3, respectively. The major toxicological effects associated with the contaminants of concern are summarized in Section 4.4.

4.1 RISK FACTORS FOR THE ASSESSMENT OF RADIOLOGICAL EFFECTS

Radioactive contamination at the Weldon Spring site is characterized as low-level ionizing radiation (Chapter 2). The potential health effects associated with exposure to this type of radiation include an increase in the probability of cancer induction (which depends on the organ irradiated), serious genetic effects, and other detrimental health effects such as teratogenesis (fetal abnormalities). The two major health concerns associated with radiation are cancer induction and hereditary effects. Both of these effects are stochastic and are considered to have no threshold dose — i.e., the probability of occurrence, not the severity of effect, increases with the absorbed dose of radiation, and there is assumed to be no dose level below which the risk is zero. The major potential impact resulting from exposure to site contaminants (largely alpha emitters) is cancer induction; the risk of serious hereditary effects is much lower. Hence, the radiological health risks presented in this BA are limited to cancer induction. This approach is consistent with EPA guidance, which notes that cancer risk is generally the limiting effect for radionuclides and suggests that radiation carcinogenesis be used as the sole basis for assessing radiation-related human health risks (EPA 1989j).

The relationship between radiation dose and health effects is relatively well characterized for high doses of most types of radiation. Lower levels of exposure may constitute a health risk, but a direct cause-and-effect relationship is difficult to establish because a particular effect in a specific individual can be produced by many different processes. For example, the features of cancers resulting from radiation are not distinct from the features of cancers produced by other causes. Therefore, the risk of cancer from exposure to low levels of ionizing radiation must be extrapolated from data for increased rates of cancers observed at higher doses.

The EPA has developed guidance for conducting radiological risk assessment (EPA 1989j). The process for assessing radiological risks generally parallels the process for assessing chemical carcinogenic risks, but the former consists of a two-phase (i.e., dual-endpoint) evaluation. For the first phase, radiation doses are calculated for all relevant radionuclides and pathways for the purpose of comparing committed effective dose equivalents with established radiation protection standards and criteria. For the second phase, carcinogenic risks are calculated for the radionuclides of concern in a manner similar to that for chemical carcinogens, using an age-averaged lifetime excess cancer incidence per unit intake (and per unit external exposure). To support this second calculation, the EPA has recently developed cancer incidence factors per unit intake that are analogous to the slope factors developed for chemical carcinogens (see

Section 4.3). However, these factors have not been independently verified by the scientific community or widely used. Because of related uncertainty, they have not been used to estimate radiological health risks in this BA. Rather, these risks were estimated from the conversion factors and risk estimators discussed below.

The radiation doses were calculated with the DCFs summarized in Table 4.1. Separate DCFs are provided for the various exposure routes (i.e., external gamma irradiation, ingestion, and inhalation) for the major radionuclides at the Weldon Spring site. These DCFs include the contributions from radioactive decay products that may be present at the point of exposure in addition to the dose associated with decay products formed within the body following intake. The DCFs used in this assessment are based on dosimetry models developed by the ICRP, as reported by Gilbert et al. (1989).

The external gamma irradiation DCFs given in Table 4.1 were used if measured gamma exposure rates were not available. When measured values were available, the radiation dose rate from this route of exposure was calculated with a factor of 0.95 mrem/mR. The factors in Table 4.1 are for the gamma dose rate at 1 m above the ground per pCi/g of the specified radionuclide and are based on the assumption that the radionuclides are uniformly distributed in the soil, extending to an infinite depth and to an infinite lateral extent. For practical purposes, an infinite depth is about 1 m (for soil as the contaminated medium) and an infinite lateral extent is about 1,200 m^2. Depth and area factors (i.e., fractional values) are provided in Gilbert et al. (1989) to account for smaller areas of contamination.

The ingestion DCFs given in Table 4.1 are based on the ICRP model for the gastro-intestinal tract, which divides the tract into four compartments: stomach, small intestine, upper large intestine, and lower large intestine. Absorption of materials into body fluids is generally considered to occur within the small intestine. The fraction of ingested material absorbed into body fluids from the tract is denoted as f_1; the numerical value of f_1 depends on the chemical form of the ingested material. The DCF corresponding to the higher f_1 value was used if more than one value was given (i.e., for uranium). This provides conservative estimates of the radiation doses associated with ingestion.

The inhalation DCFs given in Table 4.1 are based on the ICRP lung model, which divides the respiratory tract into three regions: nasopharyngeal, tracheobronchial, and pulmonary. These regions are connected with one another as well as with body fluids and the gastrointestinal tract. The fraction of inhaled activity deposited in these regions is a function of the size of the airborne particle. The activity median aerodynamic diameter is the measure most widely used to characterize the aerosol; the DCFs in Table 4.1 are based on a diameter of 1 micron. The rate at which the deposited material is removed from the region of the respiratory tract is considered to be independent of particle size; however, the rate is related to the chemical form of the particles. Chemical compounds are assigned to one of three inhalation classes to characterize their removal rate from the lung. The inhalation classes are denoted as D, W, and Y — corresponding to clearance or removal times from the pulmonary region of the lung on the order of days, weeks, or years, respectively. The lung retention class with the highest DCF was

TABLE 4.1 Dose Conversion Factors for the External Gamma Irradiation, Ingestion, and Inhalation Pathways at the Weldon Spring Site

Radionuclide[a]	External Gamma Irradiation[b] [(mrem/h)/(pCi/g)]	Ingestion		Inhalation[d]	
		f_1[c]	DCF (mrem/pCi)	Inhalation Class	DCF (mrem/pCi)
Actinium-227	3.1×10^{-4}	1×10^{-3}	1.5×10^{-2}	D	6.7
				W	1.7
				Y	1.2
Lead-210	4.7×10^{-7}	2×10^{-1}	6.7×10^{-3}	D	2.1×10^{-2}
Protactinium-231	2.5×10^{-5}	1×10^{-3}	1.1×10^{-2}	W	1.3
				Y	8.6×10^{-1}
Radium-226	1.8×10^{-3}	2×10^{-1}	1.1×10^{-3}	W	7.9×10^{-3}
Radium-228	9.3×10^{-4}	2×10^{-1}	1.2×10^{-3}	W	4.5×10^{-3}
Thorium-228[e]	1.5×10^{-3}	2×10^{-4}	7.5×10^{-4}	W	2.5×10^{-1}
				Y	3.1×10^{-1}
Thorium-230	2.1×10^{-7}	2×10^{-4}	5.3×10^{-4}	W	3.2×10^{-1}
				Y	2.6×10^{-1}
Thorium-232	1.2×10^{-7}	2×10^{-4}	2.8×10^{-3}	W	1.6
				Y	1.1
Uranium-234[e]	1.4×10^{-7}	5×10^{-2}	2.6×10^{-4}	D	2.7×10^{-3}
		2×10^{-3}	2.5×10^{-5}	W	7.1×10^{-3}
				Y	1.3×10^{-1}
Uranium-235	1.0×10^{-4}	5×10^{-2}	2.5×10^{-4}	D	2.5×10^{-3}
		2×10^{-3}	2.6×10^{-5}	W	6.7×10^{-3}
				Y	1.2×10^{-1}
Uranium-238	1.4×10^{-5}	5×10^{-2}	2.5×10^{-4}	D	2.4×10^{-3}
		2×10^{-3}	3.8×10^{-5}	W	6.2×10^{-3}
				Y	1.2×10^{-1}

[a] Each dose conversion factor (DCF) includes the dose associated with the listed radionuclide and the radionuclides of the associated decay chain (see Table 2.2).

[b] The DCF is given as a rate (i.e., mrem/h) because the dose is a function of the exposure time and the radionuclide concentration (in pCi/g).

[c] Fraction of a stable element entering the gastrointestinal tract that reaches body fluids.

[d] The three inhalation classes D, W, and Y correspond to retention half-times of less than 10 days, 10 to 100 days, and more than 100 days, respectively. Inhalation DCFs are for particles having an activity median aerodynamic diameter of 1 µm.

[e] In this assessment, the radiation doses associated with thorium-228 and uranium-234 are included with those reported for radium-228 and uranium-238, respectively (Section 2.3.2.1). Hence, the DCF for radium-228 is the sum of that for radium-228 plus thorium-228, and the DCF for uranium-238 is the sum of that for uranium-238 plus uranium-234.

Source: Data from Gilbert et al. (1989).

used for radionuclides for which more than one value was given. This approach provides a conservative estimate of the radiation doses associated with inhalation of radionuclides and probably results in overestimating the actual doses that would occur under the scenarios considered in this assessment.

For this BA, a risk factor of 6×10^{-7}/mrem effective dose equivalent was used to estimate the likelihood of cancer induction for all exposure pathways except inhalation of radon. The EPA used this value in developing revisions to the National Emission Standards for Hazardous Air Pollutants (NESHAPs) for radionuclides under Section 112 of the Clean Air Act (EPA 1989k). This risk factor is a lifetime average value and is therefore believed to be representative of conditions defined for the exposure scenarios at the Weldon Spring site. The recently issued BEIR V study (National Research Council 1990) presents a detailed description of current data on the health risks associated with radiation exposure. A mortality risk factor of about 8×10^{-7}/mrem can be derived from the information provided in the BEIR V report. However, not all radiation-induced cancers are fatal, i.e., the cancer mortality rate is about 60% of the cancer induction rate given by the EPA (1989k). A cancer induction rate of about 1.3×10^{-6}/mrem can thus be inferred from the results presented in the BEIR V study. This value is about a factor of 2 greater than that used by the EPA in their evaluation for NESHAPs for radionuclides; however, the results of the BEIR V study are recent and have not yet been uniformly accepted by the scientific community. In addition, the BEIR V study derived these estimates primarily from data on acute exposure (a single instantaneous exposure) and suggested that it might be appropriate to reduce this risk by applying a dose rate effectiveness factor of 2 in cases of continuous low-level exposure. Thus, the radiation risk factor used in this BA reflects information provided in BEIR V and is believed to be appropriate for this assessment. Also, because the Weldon Spring site is on the NPL and final cleanup decisions are to be made in conjunction with the EPA, this value is consistent with the value used by the EPA in evaluating health risks associated with radiation exposure.

The risk of a fatal lung cancer from inhalation of radon-222 decay products has been estimated in this BA using the risk factor of 3.5×10^{-4}/WLM recommended in the BEIR IV study (National Research Council 1988). Because most lung cancers are fatal, this estimator can also be used to estimate the rate of cancer induction. The results of the BEIR IV study were largely based on information for adult exposures (i.e., miners exposed to elevated concentrations of radon gas and associated decay products). However, these results were adjusted in the BEIR IV report to account for exposures that could occur in the general population over a lifetime. Hence, this risk factor is believed to be appropriate for estimating the health risks associated with exposure to radon-222 at the Weldon Spring site.

The risk of cancer from inhalation of radon-220 decay products cannot be addressed directly because epidemiological data are available only for radon-222. However, this risk can be estimated on the basis of dosimetric considerations. Whereas the radiation dose from exposure to radon-222 decay products is localized (i.e., to the bronchial epithelium of the lung), that from inhalation of radon-220 decay products is distributed among several organs because the radioactive half-life of lead-212 (11 hours) is comparable to its biological half-life in the lung. This allows a considerable amount of lead-212 to be transferred to the blood, kidneys, bone

surfaces, and other tissues. The distribution of radiation energy to several organs, and the relatively lower radiosensitivity of these organs, results in a lower risk from inhalation of radon-220 decay products than from inhalation of radon-222 decay products for the same dose (in WLM). The health risk from radon-220 decay products has been estimated to be one-third of that from radon-222 decay products for an equal concentration of inhaled alpha energy (ICRP 1981a). Therefore, the carcinogenic risk from inhalation of radon-220 decay products was calculated with a risk estimator of 1.2×10^{-4}/WLM.

For the various exposure scenarios considered in this assessment, the potential radiological risks were derived separately for each of four exposure pathways: (1) external gamma irradiation; (2) ingestion of contaminated material (e.g., soil); (3) inhalation of airborne particulates; and (4) inhalation of radon. This approach facilitates determination of the critical pathway(s) at the site. The total risk for each scenario was calculated by summing the individual risks from all relevant radionuclides and pathways. The risk factors presented in this chapter were used with the dose estimates developed in Chapter 3 to quantify the carcinogenic risks associated with radiation exposures at the Weldon Spring site. The human health assessment is presented in Chapter 5.

4.2 TOXICITY VALUES FOR THE ASSESSMENT OF NONCARCINOGENIC EFFECTS OF CHEMICAL CONTAMINANTS

The potential for noncarcinogenic health effects resulting from exposure to chemical contaminants is assessed by comparing an exposure estimate (intake) to a reference dose. A reference dose (RfD) is defined as an estimate of the average daily exposure for the human population (including sensitive subpopulations) that is likely to be without an appreciable risk of deleterious effects during a lifetime (EPA 1989j). An RfD is specific to the chemical, the route of exposure, and the duration over which the exposure occurs. Reference doses have been derived for the assessment of oral exposures and are given in units of milligrams per kilogram body weight per day (mg/kg-d). Until recently, RfDs were also derived for inhalation exposures; however, inhalation exposures are currently assessed with derived reference concentrations (RfCs), which are reported in milligrams per cubic meter (mg/m^3). An RfC can be converted to the corresponding inhalation RfD (in mg/kg-d) by dividing by 70 kg (an assumed body weight) and multiplying by 20 m^3/d (an assumed inhalation rate). Because the derivation of an RfC is analogous to the derivation of an oral RfD, the term RfD is used in the following discussion to represent both the oral RfD and inhalation RfC.

Chronic RfDs are used to evaluate exposures occurring over periods of more than 7 years, and subchronic RfDs are used to evaluate exposures occurring over shorter periods of 2 weeks to 7 years. Because the potential exposures considered in this BA are for periods of more than 7 years (Section 1.3.2), only chronic RfDs are considered. The EPA has recently developed RfDs for developmental toxicants, but such RfDs are not available for any of the Weldon Spring site contaminants.

In deriving RfDs, EPA work groups review all relevant human and animal studies for each compound and select the study (or studies) pertinent to the derivation of the specific RfD.

Each study is then evaluated to determine the no-observed-adverse-effect level (NOAEL) or, if data are inadequate for such a determination, the lowest-observed-adverse-effect level (LOAEL). The NOAEL corresponds to the dose, in mg/kg-d, that can be administered over a lifetime without inducing observable adverse effects. The LOAEL corresponds to the lowest daily dose administered over a lifetime that induces an observable adverse effect. The toxic effect characterized by the LOAEL is referred to as the "critical effect."

The NOAELs are most often based on data from experimental studies in animals. Both the experimental parameters and the extrapolation of animal data to humans are potential sources of uncertainty. Hence, in deriving an RfD, the NOAEL (or the LOAEL) is divided by uncertainty factors to ensure that the RfD will be protective of human health. Depending on the available data, uncertainty factors are applied to account for (1) extrapolation of data from experimental animals to humans (interspecies extrapolation), (2) variation in human sensitivity to the toxic effects of a compound (intraspecies differences), (3) derivation of a chronic RfD based on a subchronic rather than a chronic study, and/or (4) derivation of an RfD from the LOAEL rather than the NOAEL. In addition to these uncertainty factors, modifying factors between 0 and 10 may be applied to reflect additional qualitative considerations in evaluating the data. For most compounds, the modifying factor is 1.

The EPA-verified RfDs are available from EPA's *Integrated Risk Information System* (IRIS), an on-line database that contains current health risk and regulatory information for many chemicals (EPA 1991c). The RfDs are also tabulated quarterly in EPA's *Health Effects Assessment Summary Tables* (HEAST) (EPA 1991a). In accordance with EPA guidance (EPA 1989j), the RfDs used in this BA were obtained from IRIS; HEAST values were used only for those compounds not listed in IRIS. Oral RfDs are available for 44 of the 47 chemical contaminants of concern, and inhalation RfDs are available for 7 contaminants. The available oral and inhalation RfDs and the corresponding critical effects are listed in Tables 4.2 and 4.3, respectively. The key studies used to derive the RfD for each contaminant are summarized in Section 4.4.

The EPA's Environmental Criteria and Assessment Office (ECAO) was contacted to determine if toxicity values were available for compounds for which an RfD or RfC is not available in IRIS or HEAST. The ECAO provided an oral RfD for lithium (Table 4.2) and recommended that the minimal risk levels (MRLs) derived by the ATSDR be adopted for cadmium, 2,4-DNT, 2,6-DNT, and PCBs (see the individual summary toxicity profiles in Section 4.4) (Hurst 1990). An MRL is analogous to an RfD or RfC in that it provides an estimate of daily human exposure to a chemical that is likely to be without an appreciable risk of deleterious effects (noncancerous) over a specified duration of time (EPA 1991a).

For this BA, the RfDs presented in Tables 4.2 and 4.3 were used with the exposure estimates developed in Chapter 3 to quantify the potential for noncarcinogenic health effects associated with exposures at the Weldon Spring site. The human health assessment is presented in Chapter 5.

TABLE 4.2 Toxicity Information for Noncarcinogenic Effects: Oral Exposure

Contaminant	Oral RfD, Chronic (mg/kg-d)	Confidence Level[a]	Uncertainty Factor[b]	Modifying Factor[b]	Critical Effect
Metals					
Antimony	4×10^{-4}	Low	1,000 H,A,L	1	Reduced life span, altered blood chemistry
Arsenic	3×10^{-4}	Medium	3[c]	1	Skin effects (keratosis, hyperpigmentation)
Barium	7×10^{-2}	Medium	3 H	1	Increased blood pressure
Beryllium	5×10^{-3}	Low	100 H,A	1	No observed effect
Cadmium (in water)	5×10^{-4}	High	10 H	1	Kidney damage (proteinuria)
Cadmium (in food)	1×10^{-3}	High	10 H	1	Kidney damage (proteinuria)
Chromium III	1	Low	100 H,A	10	Decreased organ weights
Chromium VI	5×10^{-3}	Low	500 H,A,S	1	No observed effect
Cobalt	NA[d]	NA	NA	NA	NA
Copper[e]	4×10^{-2}	NA	NA	NA	Gastrointestinal tract irritation
Lead	NA	NA	NA	NA	NA
Lithium	2×10^{-2}	NA	NA	NA	Kidney damage
Manganese	1×10^{-1}	Medium	1	1	Central nervous system effects
Mercury, inorganic	3×10^{-4}	NA	1,000	NA	Kidney damage
Molybdenum	4×10^{-3}	NA	1	NA	Changes in biochemical indexes
Nickel	2×10^{-2}	Medium	100 H,A	3	Decreased organ and body weights
Selenium	5×10^{-3}	High	3 L	1	Clinical selenosis
Silver	5×10^{-3}	Medium	2 L	1	Argyria (silver accumulation in tissues)
Thallium, soluble salts	7×10^{-5}	Low	3,000 H,A,S[c]	1	Hepatotoxicity (increased serum enzyme levels) and hair loss
Uranium, soluble salts	3×10^{-3}	Medium	1,000 H,A,L	1	Kidney damage
Vanadium	7×10^{-3}	NA	100	NA	No observed effect
Zinc	2×10^{-1}	NA	10	NA	Anemia

TABLE 4.2 (Cont.)

Contaminant	Oral RfD, Chronic (mg/kg-d)	Confidence Level[a]	Uncertainty Factor[b]	Modifying Factor[b]	Critical Effect
Inorganic anions					
Fluoride, soluble	6×10^{-2}	High	1	1	Dental fluorosis
Nitrate	1.6	High	1	1	Methemoglobinemia[f]
Nitrite	1×10^{-1}	High	1	10	Methemoglobinemia
Asbestos	NA	NA	NA	NA	NA
PAHs					
Acenaphthene	6×10^{-2}	Low	3,000 H,A,S	1	Hepatotoxicity
Anthracene	3×10^{-1}	Low	3,000	1	No observed effect
Benz(a)anthracene[g]	3×10^{-2}	NA	NA	NA	NA
Benzo(b)fluoranthene[g]	3×10^{-2}	NA	NA	NA	NA
Benzo(k)fluoranthene[g]	3×10^{-2}	NA	NA	NA	NA
Benzo(g,h,i)perylene[g]	3×10^{-2}	NA	NA	NA	NA
Benzo(a)pyrene[g]	3×10^{-2}	NA	NA	NA	NA
Chrysene[g]	3×10^{-2}	NA	NA	NA	NA
Fluoranthene	4×10^{-2}	Low	3,000	1	Kidney damage, increased liver weight, hematological effects
Fluorene	4×10^{-2}	Low	3,000	1	Hematological effects
Indeno(1,2,3-cd)pyrene[g]	3×10^{-2}	NA	NA	NA	NA
2-Methylnaphthalene[h]	4×10^{-3}	NA	NA	NA	NA
Naphthalene	4×10^{-3}	NA	10,000	NA	Decreased body weight gain
Phenanthrene[g]	3×10^{-2}	NA	NA	NA	NA
Pyrene	3×10^{-2}	Low	3,000	1	Kidney damage
PCBs	1×10^{-4}	NA	NA	NA	Developmental effects

TABLE 4.2 (Cont.)

Contaminant	Oral RfD, Chronic (mg/kg-d)	Confidence Level[a]	Uncertainty Factor[b]	Modifying Factor[b]	Critical Effect
Nitroaromatic compounds					
DNB	1×10^{-4}	Low	3,000 H,A,S[c]	1	Increased spleen weight
2,4-DNT	2×10^{-4}	NA	1,000 H,A,S	1	Anemia
2,6-DNT	4×10^{-3}	NA	100 H,A	1	Neurological effects
NB	5×10^{-4}	Low	10,000 H,A,S,L	1	Methemoglobinemia, kidney and liver damage
TNB	5×10^{-5}	Low	10,000 H,A,S[i]	1	Increased spleen weight
TNT	5×10^{-4}	Medium	1,000 H,A,L	1	Liver damage

[a] EPA's confidence level in the RfD, which is based on an assessment of the number of available studies and the overall quality of the human and experimental data used to derive the RfD.

[b] The use of uncertainty factors and modifying factors to derive an RfD is described in the text. Uncertainty factors are as follows: A = animal to human extrapolation; H = variation in human sensitivity; L = extrapolation from the LOAEL to the NOAEL; and S = extrapolation from a subchronic to chronic NOAEL. The specific components of the uncertainty factor are available only for those contaminants listed in IRIS.

[c] An additional uncertainty factor of 3 was used because of a lack of developmental and/or reproductive toxicity data and consideration of all sensitive individuals.

[d] NA = not available.

[e] RfD calculated from the current drinking water standard of 1.3 mg/L.

[f] A reduction in the oxygen-carrying capacity of the blood.

[g] In the absence of an RfD from IRIS or HEAST, the RfD for pyrene was used for this compound.

[h] In the absence of an RfD from IRIS or HEAST, the RfD for naphthalene was used for this compound.

[i] An additional uncertainty factor of 10 was applied because the data used to derive the RfD were for DNB, a structurally analogous compound.

Sources: EPA (1991a) — copper, mercury, molybdenum, thallium, vanadium, zinc; EPA (1991c) — antimony, arsenic, barium, beryllium, cadmium, chromium, manganese, nickel, selenium, silver, thallium, uranium, fluoride, nitrate, nitrite, PAHs, DNB, NB, TNB, TNT; Hurst (1990) — lithium; ATSDR (1989d) — 2,4-DNT, 2,6-DNT; ATSDR (1989f) — PCBs.

TABLE 4.3 Toxicity Information for Noncarcinogenic Effects: Inhalation Exposure

Contaminant[a]	Inhalation RfD, Chronic (mg/kg-d)	Confidence Level[b]	Uncertainty Factor[c]	Modifying Factor[c]	Critical Effects
Metals					
Barium	1×10^{-4}	NA[d]	1,000	1	Fetotoxicity
Cadmium[e]	2×10^{-4}	NA	NA	NA	Kidney damage
Chromium III	6×10^{-7}	NA	300	NA	Nasal mucosa atrophy
Chromium VI	6×10^{-7}	NA	300	NA	Nasal mucosa atrophy
Manganese	1×10^{-4}	Medium	300 H,L	3	Respiratory symptoms and psychomotor disturbances
Mercury	9×10^{-5}	NA	30	NA	Neurotoxicity
Nitroaromatic compounds					
NB	6×10^{-4}	NA	3,000	NA	Hematological, adrenal, renal, and hepatic lesions

[a] The only contaminants of concern listed are those for which inhalation RfDs are available.

[b] EPA's confidence level in the RfD, which is based on an assessment of the number of available studies and the overall quality of the human and experimental data used to derive the RfD.

[c] The use of uncertainty factors and modifying factors to derive an RfD is described in the text. Uncertainty factors are as follows: H = variation in human sensitivity and L = extrapolation from the LOAEL to the NOAEL. The specific components of the uncertainty factor are available only for those contaminants listed in IRIS.

[d] NA = not available.

[e] RfD derived from a minimum risk level of 7×10^{-4} mg/m^3.

Sources: EPA (1991a) — barium, chromium III, chromium VI, mercury, NB; EPA (1991c) — manganese; ATSDR (1989b) — cadmium.

4.3 TOXICITY VALUES FOR THE ASSESSMENT OF CARCINOGENIC EFFECTS OF CHEMICAL CONTAMINANTS

The toxicity information considered in the assessment of potential carcinogenic risks includes a weight-of-evidence classification and a slope factor. The weight-of-evidence classification qualitatively describes the likelihood that an agent is a human carcinogen and is based on an evaluation of the available data from human and animal studies. An agent may be placed in one of three groups to indicate its potential for carcinogenic effects: Group A, a human carcinogen; Group B1 or B2, a probable human carcinogen; and Group C, a possible human carcinogen. Agents that cannot be classified as human carcinogens because of lack of data are categorized in Group D; agents for which there is evidence of noncarcinogenicity in humans are categorized in Group E.

The slope factor is defined as the plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime (EPA 1989j). It is used to determine an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a carcinogen. Slope factors are derived from studies of carcinogenicity in humans and/or experimental animals and are typically calculated for compounds in Groups A, B1, and B2. As for RfDs, the slope factors are specific to a chemical and route of exposure, and they are available for both the oral and inhalation routes. The EPA has also derived a unit risk, in $(\mu g/m^3)^{-1}$, to replace the former inhalation slope factor for the assessment of inhalation exposures; the unit risk can be converted to an inhalation slope factor by multiplying by 70 kg (an assumed body weight) and a unit conversion factor of 1,000 µg/mg, and dividing by 20 m^3/d (an assumed inhalation rate).

Of the 47 chemical contaminants of concern at the Weldon Spring site, 17 are classified as Group A, B1, B2, or C carcinogens. The available oral and inhalation slope factors for these compounds are listed in Tables 4.4 and 4.5. The key studies used to derive these slope factors are described in Section 4.4.

The ECAO was also consulted regarding slope factors for carcinogens for which an EPA-derived slope factor is not available. The ECAO recommended applying the oral and inhalation slope factors available for benzo(a)pyrene to all of the carcinogenic PAHs. Slope factors for other contaminants were not available.

For this BA, the slope factors presented in Tables 4.4 and 4.5 were used with the exposure estimates developed in Chapter 3 to quantify the carcinogenic risks associated with potential exposures at the site. The human health assessment is presented in Chapter 5.

4.4 SUMMARY TOXICITY PROFILES

A general description of the toxicological effects associated with radiation exposure and short summaries of the major toxicological effects of the radioactive and chemical contaminants of concern are presented in this section as background information. For those compounds for

TABLE 4.4 Toxicity Information for Carcinogenic Effects: Oral Exposure

Contaminant	Oral Slope Factor ([mg/kg-d]$^{-1}$)	Carcinogenic Weight-of-Evidence[a]	Tumor Site[b]
Metals[c]			
Arsenic	1.8	A	Skin
Beryllium	4.3	B2	-
Lead	NA[d]	B2	-
Asbestos	NA	A	Gastrointestinal tract
PAHs[e]	11.5	B2	-
PCBs	7.7	B2	-
Nitroaromatic compounds			
2,4-DNT	0.68[f]	B2	-
2,6-DNT	0.68[f]	B2	-
TNT	0.03	C	-

[a] Carcinogenic weight-of-evidence is a qualitative designation for potential carcinogens: A, human carcinogen; B2, probable human carcinogen; C, possible human carcinogen.

[b] Tumor site provided for Group A carcinogens only.

[c] Because of inadequate evidence for the carcinogenicity of cadmium, chromium VI, and nickel by the oral route, oral slope factors have not been derived for these metals.

[d] NA = not available.

[e] The carcinogenic PAHs detected at the Weldon Spring site are benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, and indeno(1,2,3-cd)pyrene.

[f] Derived for 2,4-DNT and 2,6-DNT mixtures.

Sources: EPA (1991c) — metals, nitroaromatic compounds, PCBs; Hurst (1990) and EPA (1991a) — PAHs.

TABLE 4.5 Toxicity Information for Carcinogenic Effects: Inhalation Exposure

Contaminant	Inhalation Slope Factor ([mg/kg-d]$^{-1}$)	Carcinogenic Weight-of-Evidence[a]	Tumor Site[b]
Metals			
Arsenic	15	A	Lung
Beryllium	8.4	B2	-
Cadmium	6.1	B1	-
Chromium VI	41	A	Lung
Lead	NA[c]	B2	-
Nickel[d]	1.7	A	Lung, nasal cavity
Asbestos	0.23[e]	A	Lung, mesothelium
PAHs[f]	6.1	B2	-
PCBs	NA	B2	-
Nitroaromatic compounds			
2,4-DNT	NA	B2	-
2,6-DNT	NA	B2	-
TNT	NA	C	-

[a] Carcinogenic weight-of-evidence is a qualitative designation for potential carcinogens: A, human carcinogen; B1 and B2, probable human carcinogen; C, possible human carcinogen.

[b] Tumor site provided for Group A carcinogens only.

[c] NA = not available.

[d] Nickel subsulfide.

[e] In units of (fibers/mL)$^{-1}$.

[f] The carcinogenic PAHs are benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, and indeno(1,2,3-cd)pyrene.

Sources: EPA (1991c) — metals, asbestos; Hurst (1990) and EPA (1991a) — PAHs.

which EPA-derived toxicity values are available, the key studies used for their derivation are also described.

Many of the effects described for the radioactive and chemical contaminants of concern occur only at high doses, either from accidental human exposure or from exposure induced in experiments with laboratory animals. Such effects would not generally be expected to result from the levels at which these contaminants occur at the Weldon Spring site. However, adverse effects described for chronic (long-term) exposures might occur following long-term, repeated exposures to contaminants at the site.

The radiological health effects were estimated with DCFs to determine a radiation dose per unit intake or per unit external exposure (Sections 3.4.1 and 4.1) and with risk estimators to determine the risk of cancer induction per unit dose (Section 4.1). The DCFs for the various radioactive contaminants of concern are based on metabolic models that take into account the chemical behavior of the element as well as its radiotoxicity. The toxicological effects of radiation exposure and the toxicity of specific radioactive contaminants are presented in Section 4.4.1.

The information presented in Sections 4.4.2 through 4.4.7 on the general toxicology of the contaminants of concern was taken from the references listed in Table 4.6. The EPA classification of carcinogenicity and information supporting the compound-specific toxicity values were obtained from IRIS (EPA 1991c) and HEAST (EPA 1991a). The specific studies from which the toxicity values for each contaminant were derived are referenced in the relevant toxicity profile.

4.4.1 Radionuclides

Radiation health effects for humans have been confirmed only at relatively high doses or at high dose rates with large populations. Hence, risk estimates are strictly applicable only to large populations because the appearance of health effects after an exposure is a chance event. For low doses, health effects are presumed to occur but can only be estimated statistically. These effects cannot be predicted with certainty for small populations (e.g., a few individuals).

For this BA, the radiological health effects are given as the increased likelihood of cancer induction for an exposed individual or population. However, risk estimates are uncertain for the low dose range (as is the case for the scenarios presented in Chapter 3) because of the necessary extrapolation of effects from high doses and because of assumptions regarding the dose-response relationships and the underlying mechanisms of radiation carcinogenesis. In fact, studies of populations chronically exposed to low-level radiation, e.g., in regions of elevated natural background, have not shown consistent conclusive evidence of an associated higher risk of cancer induction.

Alpha, beta, and gamma radiation are released during the radioactive decay of radionuclides in the uranium-238, thorium-232, and uranium-235 decay series. Each type of radiation differs in its physical properties and in its ability to induce damage to biological tissue. Alpha

TABLE 4.6 References for Chemical Toxicity Information

Contaminant	References[a]
Metals	
Antimony	Friberg et al. (1986); Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1990a)
Arsenic	Seiler and Sigel (1988); ATSDR (1989a)
Barium	Venugopal and Luckey (1978); EPA (1984a); Friberg et al. (1986); Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1990c)
Beryllium	ATSDR (1988a); Seiler and Sigel (1988)
Cadmium	Seiler and Sigel (1988); ATSDR (1989b)
Chromium	Seiler and Sigel (1988); ATSDR (1989c)
Cobalt	National Academy of Sciences (1977); Klaassen et al. (1986); ATSDR (1990d)
Copper	National Academy of Sciences (1977); EPA (1984b); Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1990e)
Lead	Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1990g)
Lithium	National Academy of Sciences (1977); Klaassen et al. (1986); Seiler and Sigel (1988)
Manganese	EPA (1984c); Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1990h)
Mercury	Seiler and Sigel (1988); ATSDR (1989e); Owen (1990)
Molybdenum	National Academy of Sciences (1977); Venugopal and Luckey (1978); Seiler and Sigel (1988)
Nickel	Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1988b)
Selenium	EPA (1984f); Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1989g); Owen (1990)
Silver	ATSDR (1990k)
Thallium	Venugopal and Luckey (1978); Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1990l)
Uranium	Hodge et al. (1973); Friberg et al. (1986); Seiler and Sigel (1988); ATSDR (1990m)
Vanadium	National Academy of Sciences (1977); Venugopal and Luckey (1978); Klaassen et al. (1986); Seiler and Sigel (1988)
Zinc	Seiler and Sigel (1988); ATSDR (1989h)

TABLE 4.6 (Cont.)

Contaminant	References[a]
Inorganic anions	
Fluoride	National Academy of Sciences (1977); Klaassen et al. (1986); Seiler and Sigel (1988); ATSDR (1990f)
Nitrate and nitrite	National Academy of Sciences (1977); EPA (1980, 1985a); Klaassen et al. (1986); Seiler and Sigel (1988)
Asbestos	Klaassen et al. (1986); Mauskopf (1987); ATSDR (1990b)
PAHs	EPA (1984e); ATSDR (1990j)
PCBs	ATSDR (1989f)
Nitroaromatic compounds	
DNB	Fedoroff and Sheffield (1962); Hamblin (1962); Hajjar et al. (1989)
DNT	Etnier (1987); ATSDR (1989d); Brower and Hartley (1989); Burrows et al. (1989)
NB	EPA (1987b); ATSDR (1990i)
TNB	Fedoroff and Sheffield (1962); Burrows et al. (1989)
TNT	Roth (1980); Ryon (1987); Burrows et al. (1989); Gordon and Hartley (1989)

[a] Acronyms: ATSDR, Agency for Toxic Substances and Disease Registry; EPA, U.S. Environmental Protection Agency.

particles are primarily a hazard when taken into the body (e.g., by inhalation or ingestion) because, for external exposure, they almost always lose their energy in the outer layer of dead skin cells of the body before reaching living tissue. Within the body, alpha particles result in greater cell damage than beta or gamma radiation because their energy is completely absorbed by the tissue. Beta particles are primarily an internal hazard; however, in some cases of external exposure, very energetic beta particles can penetrate to living skin cells, thus representing an external hazard as well. However, beta particles deposit less energy to tissue and therefore induce much less damage than alpha particles. Gamma radiation is primarily an external hazard because it can easily penetrate tissue and reach internal organs. However, only a very small fraction of the incident energy is deposited in tissue and internal organs.

Radiation exposure pathways can be separated into external and internal components. External exposure occurs when the radioactive material is outside the body, and it is primarily a concern only for gamma radiation. Internal exposure occurs when the radioactive material enters the body by inhalation or ingestion. Inhaled material can be exhaled, expelled from the

lungs to be either spit or swallowed and excreted, deposited in the lungs, or absorbed by the blood and relocated to other organs where it is excreted over time. Some fraction of ingested material will enter the bloodstream and be either excreted in the urine or feces or relocated to other organs and excreted over time; most insoluble ingested material is not absorbed into the blood but excreted directly via feces. For internal exposures, alpha and beta particles are the dominant concern because their energy is absorbed in cells before the particles leave the body. The biological significance of the various forms of radiation is accounted for by a quality factor that accounts for the biological effectiveness (degree of harm) of a particular radiation. In accordance with recommendations of the ICRP (1979), the radiation dose calculations in this BA used a quality factor of 1 for beta and gamma radiation and a quality factor of 20 for alpha radiation.

The radioactive contaminants of concern for the Weldon Spring site, as identified in Chapter 2, are actinium-227, lead-210, protactinium-231, radium-226, radium-228, radon-220, radon-222, thorium-230, thorium-232, uranium-235, and uranium-238. These radionuclides are members of the uranium-238, thorium-232, and uranium-235 decay series (Figures 2.1, 2.2, and 2.3). The radiological effects associated with the other radionuclides in these decay series are accounted for in the DCFs used for these 11 radionuclides. Information on the radiation toxicity of the radionuclides of concern is provided in Sections 4.4.1.1 through 4.4.1.7.

4.4.1.1 Actinium

Actinium occurs naturally as a radioactive decay product in the thorium-232 and uranium-235 decay series and has no stable (i.e., nonradioactive) forms. Because of the much shorter half-lives of the actinium radionuclides compared with thorium-232 and uranium-235, the relative abundance of actinium (by mass) in nature is much lower than that of thorium and uranium. The decay of radium-228 in the thorium-232 decay series results in the production of actinium-228; because of its short half-life (6.1 hours), the radiation dose from this radionuclide is included in that reported for radium-228 in this assessment. Actinium-227 results from the decay of protactinium-231 in the uranium-235 decay series and has the longest half-life of any actinium isotope (22 years). Actinium-227 is primarily a beta emitter, and it poses an internal health hazard. It is relatively insoluble, so little actinium is absorbed into the bloodstream; the small fraction that is absorbed is translocated to the skeleton and liver where it is strongly retained (ICRP 1981b).

4.4.1.2 Lead

Lead is a widespread, naturally occurring metal that exists in both radioactive and stable (i.e., nonradioactive) forms. In nature, lead is present almost exclusively in its stable form. Four radioactive isotopes of lead exist in the uranium-238, thorium-232, and uranium-235 decay series. Lead-214 (half-life of 27 minutes) and lead-210 (half-life of 22 years) are members of the uranium-238 decay series. Lead-212 (half-life of 11 hours) is a member of the thorium-232 decay series, and lead-211 (half-life of 36 minutes) is a member of the uranium-235 decay series. Of these four radionuclides, only lead-210 has a long enough half-life to warrant its consideration

as a contaminant of concern. The health risks associated with the other three lead isotopes are accounted for in the DCFs of other, longer-lived parent radionuclides. Lead-210 is a beta emitter and thus poses an internal health hazard. Lead has a relatively high absorption rate from the gastrointestinal tract; after absorption, it is deposited in the skeleton, liver, and kidney but is strongly retained only by mineral bone (ICRP 1980).

4.4.1.3 Protactinium

Protactinium occurs naturally as a radioactive decay product in the uranium-235 and uranium-238 decay series and has no stable (i.e., nonradioactive) forms. Because of the much shorter half-lives of protactinium radionuclides compared with their uranium parents, the relative abundance of protactinium (by mass) in nature is much lower than that of uranium. Protactinium-234m (metastable), a decay product of uranium-238, is a beta emitter; because of its short half-life (1.2 minutes), it is not considered separately from uranium-238. Protactinium-231 is a long-lived decay product in the uranium-235 decay series (half-life of 33,000 years). It is primarily an alpha emitter; however, a wide range of gamma rays are also emitted as this radionuclide decays. Hence, protactinium poses an internal as well as external health hazard.

The solubility of protactinium in the bloodstream is relatively low. Rat studies have shown that protactinium entering the bloodstream is deposited primarily in the skeleton, whereas the liver and kidneys are secondary sites of deposition. Protactinium deposited in the skeleton is retained there, with a biological half-life of more than 100 days; protactinium deposited in the liver or kidneys exhibits a biphasic retention, the two components having biological half-lives of about 10 and 60 days, respectively. Data from studies of a man accidently contaminated with protactinium-231 through a puncture wound in his hand indicate that, after an early phase of excretion, the remaining fraction of protactinium is retained in the body almost indefinitely, probably mainly in the skeleton (ICRP 1981b).

4.4.1.4 Radium

Radium is a widespread, naturally occurring alkaline earth metal. Radium-226 is a member of the uranium-238 decay series, radium-228 and radium-224 are members of the thorium-232 decay series, and radium-223 is a member of the uranium-235 decay series. Radium-226 is an alpha-gamma emitter with a half-life of 1,600 years; radium-228 is a beta-gamma emitter with a half-life of 5.8 years. Because of the very short half-lives of radium-224 (half-life of 3.6 days) and radium-223 (half-life of 11 days), the health risks associated with these two radium isotopes are accounted for in the DCFs for thorium-228 and actinium-227, respectively. The metabolic behavior of radium in the body is similar to that of calcium. Thus, an appreciable fraction of ingested radium is deposited nonuniformly in the bone. The release of radium from the bone is slow, so chronic intake can result in very high concentrations in bone.

The majority of epidemiological data on the health effects of radium-226 and radium-228 in humans comes from studies of radium dial painters, radium chemists, and technicians exposed through medical procedures in the early 1900s (National Research Council 1988). These studies, as well as studies on experimental animals, indicate that chronic exposure to radium can induce bone sarcomas. The minimum latency period is 7 years after the first exposure, but tumors can continue to appear throughout a lifetime. Carcinomas in the paranasal sinuses and mastoid air cells have been shown to be associated with radium-226; causation in this case is attributed to generation of radon-222 by radium-226 decay and subsequent irradiation of the sinuses and mastoid epithelial tissues by radon-222 and its decay products.

4.4.1.5 Radon

Radon is a naturally occurring radioactive noble gas. Each of the three decay series contains one radon isotope. Radon-222 is a decay product of radium-226 in the uranium-238 decay series, radon-220 is a decay product of radium-224 in the thorium-232 decay series, and radon-219 is a decay product of radium-223 in the uranium-235 decay series. In general, radon-219 poses a much lower risk than the other two radon isotopes because of its very short half-life (4.0 seconds).

Radon-222 is a short-lived alpha emitter (half-life of 3.8 days) that decays into four short-lived radioactive decay products, all of which are heavy metals. Two decay products, polonium-218 and polonium-214, are alpha emitters; two others, lead-214 and bismuth-214, are beta-gamma emitters. Radon-220 is a short-lived alpha emitter (half-life of 55 seconds) that decays to polonium-216, which also decays by alpha emission to lead-212. Lead-212 is a beta-gamma emitter with a half-life of 11 hours. The lung is the major tissue irradiated by radon-222, radon-220, and their short-lived decay products except for lead-212; lead-212 can be transferred from the lung to other tissues, particularly the blood (red blood cells), kidneys, and bone surfaces (ICRP 1981a).

The primary hazard of radon arises from the inhalation of its short-lived decay products. These metallic decay products, which are charged ions in air, readily attach to dust particles and can be inhaled into the lungs and deposited on the mucous lining of the respiratory tract. Unattached decay products tend to be inhaled deeper into the lungs where the residence time is longer. When alpha emission occurs in the lung, it can damage the cells lining the airways, potentially leading to lung cancer. The association of exposure to radon decay products with human lung cancer has been studied extensively in uranium miners (National Research Council 1988). These studies have identified a clear connection between elevated radon exposure and lung cancer incidence. Other health effects observed in uranium miners include emphysema, tuberculosis, renal dysfunction, bronchitis, pneumonia, asthma, and cancers of the skin and stomach. However, miners were exposed to many hazardous substances along with high levels of radon, so it is difficult to determine which effects resulted from exposure to radon alone.

4.4.1.6 Thorium

Thorium is a naturally occurring radioactive metal. Six isotopes of thorium are present in the three decay series: thorium-234, thorium-232, thorium-231, thorium-230, thorium-228, and thorium-227. Of these six, only thorium-232 and thorium-230 are considered to be contaminants of concern for the Weldon Spring site. Thorium-232 is a primordial element with a very long half-life of 1.4×10^{10} years; thorium-230 is a relatively long-lived alpha emitter with a half-life of 7.7×10^{4} years and is a member of the uranium-238 decay series. Thorium-234, thorium-231, and thorium-227 have short half-lives of 24 days, 26 hours, and 19 days, respectively; and the health risks associated with these radionuclides are accounted for in the DCFs of other, longer-lived radionuclides. Similarly, the health risk associated with thorium-228 (half-life of 1.9 years) is accounted for in the DCF for radium-228 (Section 2.3.2.1).

Most thorium that is inhaled or ingested in food, water, or soil is excreted within a few days, and only a small fraction is absorbed into the bloodstream. Once in the bloodstream, thorium accumulates on bone surfaces where it can persist for several years; therefore, bone cancer is a potential health concern. Studies have shown that, although soluble forms of thorium are absorbed to a greater degree than insoluble forms, no chemical form is absorbed from the gastrointestinal tract in any appreciable amount.

The majority of human data for thorium exposure comes from diagnostic dye studies. Colloidal thorium-232 dioxide (Thorotrast) was injected into patients as a radiographic-contrast medium between 1928 and 1955. The epidemiological data from these studies show that the primary health effects of high doses of injected Thorotrast are blood disorders and tumors of the liver (National Research Council 1988). Risk estimates derived from the Thorotrast studies are appropriate only for Thorotrast injections (e.g., administered doses) because health effects associated with normal routes of exposure would be very different. However, these studies indicate that thorium could be a human carcinogen.

Few data are available regarding the effects of low exposures to thorium from pathways other than injection, e.g., from inhalation or ingestion. Some evidence was found of increased incidence of lung, pancreatic, and hematopoietic cancers in workers occupationally exposed to thorium via inhalation; however, these workers were also exposed to several other agents that were known to be toxic, so direct causation cannot be inferred (Archer et al. 1973; Polednak et al. 1983; Stehney et al. 1980).

4.4.1.7 Uranium

Uranium is a heavy metal that occurs ubiquitously in the earth's crust. Natural uranium consists of three isotopes: uranium-238, uranium-235, and uranium-234. These isotopes occur in the relative abundance of 99.27, 0.72, and 0.0055% by weight, respectively. Uranium is an alpha and gamma emitter. Two hazards are associated with uranium compounds: kidney damage caused by the chemical toxicity of soluble uranium compounds and cell damage caused by the ionizing radiation that results from radioactive decay. External exposure is generally not a health concern because uranium emits only a small amount of penetrating gamma radiation.

Gastrointestinal absorption from food or water is the principal source of internally deposited uranium in the general population. Once ingested, most of the uranium is excreted from the body within a few days. The small fraction absorbed into the bloodstream (less than 1%) is stored uniformly in bone. The major health risk from uranium is associated with its chemical properties, not its radiological properties (Section 4.4.2.18). Human or animal studies conducted to date have shown little evidence to indicate that adverse health effects result from the radiation exposure associated with natural uranium. Increased incidence of lung cancer has been observed in uranium miners, but this effect is caused by exposure to radon-222, a decay product of uranium (National Research Council 1988).

4.4.2 Metals

4.4.2.1 Antimony

Antimony is typically present in soil at concentrations of less than 1 to 10 mg/kg, usually as sulfide and oxide compounds. Industrially, antimony is used in many alloys. It has been administered orally to humans and animals as both an emetic and an antiparasitic agent.

An oral RfD for antimony (Table 4.2) has been derived from a study in which rats were administered potassium antimony tartrate in their drinking water (Schroeder et al. 1970). The critical effects noted were decreased longevity and alterations in blood glucose and cholesterol levels. An uncertainty factor of 1,000 was used in deriving the RfD to account for interspecies extrapolation, protection of sensitive human subpopulations, and absence of NOAEL. The confidence level for the antimony oral RfD is low because only one species and one dose level were used, the NOAEL was not determined, and gross pathology and histology were not well described. An inhalation RfC is not available.

Toxic effects that have been observed in humans are associated mainly with occupational exposures. Acute inhalation exposure can cause severe cardiac effects, and chronic inhalation causes obstructive lung disease. In one study, women workers in an antimony plant experienced an increased incidence of spontaneous abortion compared with a control group. Absorption of ingested antimony is slow, and antimony compounds can induce vomiting. Antimony accumulates in lung tissue, and dermal exposure can cause irritation that may result in transient skin eruptions.

4.4.2.2 Arsenic

Arsenic is found in soil at an average concentration of about 5 mg/kg. Arsenic compounds are widely used as pesticides. Although inorganic arsenic has been used as a poison for centuries, it is an essential nutrient in several animal species and is believed to be essential for humans. Typical human exposures to arsenic from background sources range from 20 to 70 μg/d, with food being the major source.

Arsenic is classified by the EPA as a Group A human carcinogen. A significant increase in lung cancer mortality has been observed in several studies of smelter workers exposed to arsenic (Higgins 1982; Higgins et al. 1982; Enterline and Marsh 1982; Welch et al. 1982; Brown and Chu 1983a, 1983b, 1983c; Lee-Feldstein 1983), and the EPA has derived an inhalation unit risk of 4.3 $(\mu g/m^3)^{-1}$ from which a slope factor was calculated (Table 4.5). Ingestion of elevated levels of arsenic in drinking water has been associated with an increased incidence of skin cancer in several foreign populations but not in U.S. populations. However, the U.S. studies lacked statistical significance because of the small sample size. A unit risk of 5×10^{-5} $(\mu g/L)^{-1}$ has been derived from a study of a Taiwanese population (Tseng 1977). As stated in IRIS, a recent memorandum by the Administrator of the EPA recommended that this unit risk be adopted: "the uncertainties associated with ingested inorganic arsenic are such that estimates could be modified downwards as much as an order of magnitude, relative to risk estimates associated with most other carcinogens" (EPA 1991c). An oral slope factor (Table 4.4) has been derived from the unit risk of 5×10^{-5} $(\mu g/L)^{-1}$ for use in this BA.

In addition to the oral and inhalation slope factors for arsenic, an oral RfD was derived for arsenic from studies by Tseng et al. (1968) and Tseng (1977). The critical effects noted were keratosis and hyperpigmentation of the skin. An uncertainty factor of 3 was used in deriving the RfD to account for both the lack of studies on reproductive toxicity and for some uncertainty with respect to the representation of sensitive subpopulations in the critical study. An inhalation RfC is not available.

Arsenic is well absorbed via both ingestion and inhalation, but minimal dermal absorption has been observed. Noncarcinogenic toxic effects of arsenic ingestion at levels greater than about 20 µg/kg per day include skin disorders, severe irritation of the gastrointestinal tract, anemia, nerve degeneration, and toxicity to the liver, kidney, and heart. Dermal contact with concentrated arsenic compounds may produce severe skin irritation.

4.4.2.3 Barium

Barium is a widespread, naturally occurring element. It is found mainly as the mineral barite ($BaSO_4$), and its average concentration in soil is approximately 430 mg/kg. The average dietary intake of barium, which is present in animal and plant tissues, is approximately 1.3 mg/kg per day. Barium is used mainly as a weighting agent in drilling muds for oil and gas wells and in the manufacturing of glass, paint, paper, rubber, and various alloys. Barium was formerly used in medicine to induce contractions in the heart or uterus.

The oral RfD for barium (Table 4.2) was derived from both epidemiological studies in humans (Brenniman and Levy 1984; Wones et al. 1990) and experimental studies in animals (Schroeder and Mitchener 1975a, 1975b; Tardiff et al. 1980; Perry et al. 1983; McCauley et al. 1985). The critical effect from ingesting barium in drinking water is hypertension (i.e., increased blood pressure). An uncertainty factor of 3 was used in calculating the RfD to account for protection of sensitive human subpopulations. A factor of 3 (instead of the usual factor of 10) was used because the studies focused on the critical effect (hypertension) in the most sensitive

subpopulation (adult males). The level of confidence in the RfD was stated to be medium because of the consistent findings from a large number of human and experimental studies.

Although an inhalation RfC of 5×10^{-4} mg/m^3 is available in HEAST, the supporting documentation is not yet available through IRIS. Information provided in HEAST indicates that the RfC is based on an inhalation study in rats exposed to barium for 4 months (Tarasenko et al. 1977; EPA 1984a). The critical effect was fetotoxicity. An uncertainty factor of 1,000 was used in deriving the RfC. (The specific components of the uncertainty factor are not provided.) The inhalation RfD, derived from the RfC, is given in Table 4.3.

Ingestion of barium irritates the gastrointestinal tract and can induce tremors, muscular paralysis, and effects on the cardiovascular system. Chronic inhalation of barium dust may result in baritosis (benign pneumoconiosis), which is usually reversible after exposure ceases. Insoluble barium compounds are poorly absorbed. However, absorption of barium chloride is approximately 10 to 30% following oral exposure and 60 to 80% following inhalation exposure.

4.4.2.4 Beryllium

The average concentration of beryllium in soil is about 6 mg/kg. Beryllium and its alloys are utilized in the electronics and nuclear industries, among others. A major source of environmental beryllium is the combustion of coal and fuel oil; bituminous coal contains up to 3 mg/kg beryllium. In soil, beryllium is generally present in insoluble, immobile forms.

Beryllium is classified by the EPA as a Group B2 probable human carcinogen on the basis of epidemiological studies of humans and experimental studies of animals exposed by various routes. The EPA derived an inhalation unit risk of 2.4×10^{-3} (μg/m^3)$^{-1}$ from a study by Wagoner et al. (1980) of lung cancer in an occupationally exposed population, and this unit risk was used to calculate an inhalation slope factor (Table 4.5). An oral slope factor (Table 4.4) was calculated from information in a study by Schroeder and Mitchener (1975b), in which tumors increased at all sites in rats administered beryllium in the drinking water over a lifetime. Confidence in the oral slope factor is low because there was only one treated group and because the increase in tumor incidence was not statistically significant. Several confounding variables also limit confidence in the epidemiological study on which the inhalation slope factor is based. No carcinogenic response has been observed in any species following dermal exposure to beryllium compounds.

In addition to the oral and inhalation slope factors, an oral RfD for beryllium has been derived from the chronic bioassay used to derive the oral slope factor (Schroeder and Mitchener 1975b); no critical effect was established in this study. An uncertainty factor of 100 was used in deriving the RfD to account for interspecies extrapolation and protection of sensitive human subpopulations. Confidence in the RfD is low because only one dose level was administered and other studies suggest that the no-observed-effect level (NOEL) used as the basis for the RfD may be low. An inhalation RfC for beryllium is not available from the EPA.

Occupational exposure to beryllium oxide at levels greater than 2 $\mu g/m^3$ can result in scarring of the lungs, shortness of breath, and reduction in lung volume. Beryllium compounds are poorly absorbed from the digestive tract and through the skin. Data on developmental and reproductive toxicity are limited; however, one study of pregnant rats administered beryllium oxide or beryllium chloride reported increases in fetal mortality and internal abnormalities in the offspring.

4.4.2.5 Cadmium

Cadmium in the environment is usually found as stable compounds of oxygen, chlorine, or sulfur. The concentration of cadmium in soil is generally less than 1 mg/kg. Cadmium is used in metal plating, pigments, batteries, and plastics. The largest sources of cadmium release to the environment are the burning of fossil fuels, incineration of municipal wastes, and emissions from zinc, lead, and copper smelters.

Cadmium is classified by the EPA as a Group B1 probable human carcinogen for inhalation exposure on the basis of limited epidemiological evidence from human studies and sufficient evidence from animal studies. The EPA-derived unit risk of 1.8×10^{-3} $(\mu g/m^3)^{-1}$ is based on a twofold excess of lung cancer observed in cadmium smelter workers (Thun et al. 1985) and is supported by an increased tumor incidence in rats and mice exposed via inhalation or injection. An inhalation slope factor was derived from the unit risk for use in this BA (Table 4.5). No available studies indicate that cadmium is carcinogenic by the oral route.

Separate oral RfDs have been established for cadmium in food and water (Table 4.2). The critical effect for cadmium toxicity is kidney damage, as measured by proteinuria. The oral RfDs were calculated with a pharmacokinetics model, in which the input data were the highest concentration of cadmium in the human renal cortex not associated with significant proteinuria and the percent cadmium absorption from food and water (EPA 1985b). An uncertainty factor of 10 was used in deriving the RfD to account for protection of sensitive human subpopulations. The confidence in this RfD is high because of the vast quantity of human and animal data on cadmium toxicity. An inhalation RfC is not available from IRIS or HEAST. However, the ATSDR has derived an MRL of 7×10^{-4} mg/m^3 for cadmium (ATSDR 1989b) from a study by Kjellstrom (1986), in which cadmium levels were estimated that would result in proteinuria in 1% of a worker population.

An oral dose of approximately 0.1 mg/kg per day causes severe irritation of the digestive tract, and exposure to air concentrations of 1 mg/m^3 causes lung irritation; however, occurrences of these high exposures are rare. The effects in humans of long-term exposure to lower cadmium levels (e.g., 0.01 mg/kg per day) include kidney and liver damage and mineral loss in bone. Data from animal studies have shown an association between excess cadmium exposure and elevated blood pressure, effects on fertility, and effects on neurobehavioral development of newborns; these effects have not been adequately studied in humans. Cadmium is well absorbed from the lungs (30 to 50%) but is poorly absorbed from the digestive tract (1 to 10%) and through the skin.

4.4.2.6 Chromium

Chromium is a naturally occurring element found in soil at an average concentration of approximately 100 mg/kg; it is also found in volcanic dust and gas. It is present in the environment as chromium, chromium III, and chromium VI. Chromium III occurs naturally in the environment and is an essential nutrient, whereas chromium and chromium VI generally results from industrial processes. Chromium is used mainly for manufacturing steel and other alloys, chrome plating, water treatment, and wood treatment. Small amounts of chromium occur in many foods, and the main exposure to chromium is from dietary sources. Safe dietary levels of chromium III are estimated to range from 50 to 200 µg per day.

The principal toxicological hazard of environmental chromium is associated with exposure to chromium VI. Chromium VI is classified by the EPA as a Group A human carcinogen following inhalation exposure. Epidemiological studies of workers in chromate facilities consistently show an association between lung cancer and exposure to chromium (presumably as chromium VI). An inhalation unit risk of 1.2×10^{-2} $(\mu g/m^3)^{-1}$ was derived from the study of Mancuso (1975) in which a dose-response relationship was established. An inhalation slope factor was derived from the unit risk for use in this BA (Table 4.5). No available studies indicate that chromium VI is carcinogenic by the oral pathway.

Oral RfDs have been established for both chromium III and chromium VI. The RfD for chromium III (Table 4.2) is based on subchronic and chronic feeding studies in rats (Ivankovic and Preussmann 1975). The only effects observed were reductions in the absolute weights of the liver and spleen. The confidence in the RfD is low because of the lack of detail in the study report, the absence of an observed effect level, and the lack of additional supporting data. A 100-fold uncertainty factor and a 10-fold modifying factor were used in deriving the RfD to reflect interspecies extrapolation, protection of sensitive human subpopulations, and uncertainty in the NOEL. Because an insoluble salt of chromium III (i.e., Cr_2O_3) was used in the study, the oral RfD is limited to insoluble salts of chromium III.

The oral RfD for chromium VI (Table 4.2) is based on a study of rats administered chromium VI in the drinking water or diet (MacKenzie et al. 1958); a critical effect was not established in this study. The confidence in the RfD is low because of limitations in study design and the lack of a toxic response at the highest dose tested. A 500-fold uncertainty factor was used in deriving the RfD to reflect interspecies extrapolation, protection of sensitive human subpopulations, and the less-than-lifetime duration of the study. Because a soluble salt of chromium VI (i.e., K_2CrO_4) was used in the study, the oral RfD is limited to soluble salts of chromium VI.

Although inhalation RfCs are available in HEAST for chromium III and chromium VI (2×10^{-6} mg/m^3 for both ions), the supporting documentation is not yet available through IRIS. Information provided in HEAST indicates that the RfCs were derived from a study of humans exposed to airborne chromium VI, as chromic acid (Lindberg and Hedenstierna 1983). The critical effect was atrophy of the nasal mucosa. Uncertainty factors of 300 were used in deriving the RfCs. (The specific components of the uncertainty factors are not provided.) The inhalation RfDs derived from the RfCs are given in Table 4.3.

Effects observed following exposure to high levels of chromium VI include irritation of the nasal mucosa, perforation of the nasal septum, skin ulcers, and irritation of the gastrointestinal tract. The main toxic effect observed following long-term inhalation exposure to lower levels of chromium VI is lung cancer; adverse effects on the respiratory and immune systems have also been observed. Limited data indicate that approximately 0.5 to 18% of a dose of chromium III and 2 to 10% of a dose of chromium VI might be absorbed following oral administration. Although qualitative data indicate that chromium is absorbed following inhalation and dermal exposure, quantitative data are not available.

4.4.2.7 Cobalt

Cobalt is a relatively rare metal that is usually present in copper ores. Its concentration in soil averages about 8 mg/kg. Cobalt is used in alloys because of its magnetic properties, resistance to corrosion, and ability to withstand high temperatures. In humans, cobalt is an essential component of vitamin B_{12}.

Information on the toxicity of cobalt in humans is available from past administration of cobalt to children with sickle cell anemia (to stimulate red blood cell production) and from its brief use as an antifoam agent in beer. These uses were discontinued when the children developed goiter (enlargement of the thyroid gland) and when several heavy beer drinkers died from congestive heart failure. It was determined that cobalt and alcohol exert synergistic toxic effects. These experiences and data from animal studies allowed the National Academy of Sciences (1977) to conclude that, at daily doses greater than 1 mg/kg, cobalt may pose a health hazard to humans. The EPA has not derived an oral or inhalation RfD for cobalt.

Although cobalt is well absorbed via both inhalation and ingestion, it does not accumulate in the human body, being easily excreted via the urine and feces. Cobalt is an essential component of vitamin B_{12}, which is required for the production of red blood cells, but ingestion of cobalt at elevated levels causes an overproduction of these cells. Ingestion of cobalt also causes vomiting and diarrhea. Occupational exposure to dust containing 0.1 mg/m^3 or more of cobalt can cause irritation of the pulmonary tract, eventually leading to more serious lung disease. Prolonged dermal contact with cobalt or its salts can cause an allergic skin reaction.

4.4.2.8 Copper

Copper is an essential element for humans; deficiencies cause defective hemoglobin synthesis, leading to anemia. This metal is used extensively in the manufacture of electrical equipment, alloys, coins, and chemical apparatus. Copper occurs in soil at an average concentration of about 30 mg/kg.

Copper acts as a gastrointestinal irritant, and copper sulfate induces vomiting in children and adults (at concentrations of about 50 mg for children). Because the toxicity of copper is low and an intake of 2 to 5 mg/d is essential for humans, oral and inhalation RfDs

have not been derived by the EPA. However, an analogous value of 0.04 mg/kg per day has been derived from the current drinking water standard of 1.3 mg/L for use in this BA. Data are not available for the derivation of an inhalation RfC.

In a study assessing the potential carcinogenicity of copper, no increased tumor incidence occurred in mice fed a copper compound, but an increased incidence of one type of tumor occurred in male mice injected with this same compound. The EPA considers these animal data inadequate and has placed copper in Group D, not classifiable with respect to human carcinogenicity, because no human data are available.

Approximately 50% of dietary copper is absorbed, and the metal is poorly absorbed dermally. At high levels (i.e., greater than 1 g/d), ingestion of copper compounds can be extremely toxic, leading to liver damage and death. Inhalation of copper sulfate from mildew-control sprays by persons applying the sprays has caused symptoms including weakness, anorexia, weight loss, and shortness of breath. Exposure to copper dust can cause discoloration of the skin. In individuals with a rare genetic syndrome known as Wilson's disease, copper is not metabolized correctly and accumulates in soft tissues; such individuals cannot tolerate even normal dietary levels of copper.

4.4.2.9 Lead

Lead has been used by humans for thousands of years because of its malleability, resistance to corrosion, and abundance. This metal can be a component of solder, paint, and gasoline, but these uses have declined dramatically in recent years as awareness of the toxicity associated with lead exposure has increased. Currently in the United States, the predominant use of lead is in batteries. Lead occurs at an average concentration of 10 mg/kg in soil, but soil levels are substantially elevated in many areas that are exposed to emissions from smelters and automobiles or in areas where lead-containing paint chips have fallen onto soils.

Lead can result in varied toxicologic effects, depending on the level of exposure. From studies on rats and mice, the EPA has classified lead as a Group B2 probable human carcinogen. However, the doses that induce cancer are higher than those associated with other health effects of lead, such as reproductive toxicity, developmental toxicity, and increased blood pressure. Inhalation and oral RfD values for lead have not been derived by the EPA because it has not been possible to establish the NOAEL or LOAEL for this metal. Health effects have tentatively been associated with blood levels as low as 10 μg/dL.

In the absence of an oral or inhalation RfD for lead, the EPA has provided interim guidance on establishing soil cleanup levels at Superfund sites (EPA 1989c). The interim soil cleanup level for total lead is 500 to 1,000 mg/kg for residential settings. In addition, the EPA has developed an uptake/biokinetic model to estimate blood lead levels on the basis of total lead uptake from exposures via diet, drinking water, air, soil, and paint. The application of this model to potential exposures at the Weldon Spring site is discussed in Section 5.1.2.2.

At blood levels greater than 40 μg/dL, lead can cause miscarriage, sterility in males, anemia, and damage to the central nervous system and kidneys. Lead exposure resulting in these high blood levels is rare today. Blood levels of 30 μg/dL and higher have been associated with defects in vitamin D metabolism and with lowered IQ scores in children. At blood levels of 20 μg/dL and lower, it becomes more difficult to define the effects of lead. Some studies report a dose-related increase in blood pressure in adult males, starting at blood levels of about 10 μg/dL. Additionally, the fetus and young children are particularly sensitive to lead toxicity. Even low-level lead exposure during pregnancy can cause reduced birth weight and preterm birth; during early childhood, such exposure can cause multiple effects — including impaired intellectual and neurobehavioral development.

4.4.2.10 Lithium

Lithium is widely distributed in the environment and is typically present in soil at a concentration of approximately 20 mg/kg. Lithium carbonate is used clinically to treat depression. Industrially, lithium is used in alloys, as a catalytic agent, and as a lubricant.

The therapeutic dose of lithium carbonate is approximately 2 mg per day. Toxic effects observed subsequent to treatment include effects on the neuromuscular and cardiovascular systems, irritation of the gastrointestinal tract, and kidney damage. Effects on the fetus have been noted after use of lithium carbonate; however, fetal effects have not been observed in studies with experimental animals.

Oral and inhalation RfDs for lithium are not available from IRIS or HEAST. However, EPA's ECAO has provided a provisional oral RfD of 2×10^{-2} mg/kg per day, which is based on the induction of kidney damage in patients receiving long-term treatment with lithium (Hurst 1990). This value has been adopted for the quantitative assessment in this BA. No information was found regarding the absorption of lithium.

4.4.2.11 Manganese

Manganese is an abundant element and is present in soil at an average concentration of approximately 600 mg/kg. It is an essential dietary nutrient for humans and is present in many foods. Manganese is used in making steel alloys, batteries, electrical coils, ceramics, and glass.

The oral RfD for manganese (Table 4.2) is based on studies of human intake from dietary sources and manganese supplements (Schroeder et al. 1966; World Health Organization 1973; National Research Council 1989). The reported intake of manganese ranged from 2 to 12 mg per day. No adverse effects were noted in these studies. However, in other supporting studies, neurological effects were reported in humans drinking water with high manganese levels (Kawamura et al. 1941; Kondakis et al. 1989). Because the RfD is derived from studies of several large populations, the level of confidence in the RfD is medium. No uncertainty factor was

applied in deriving the oral RfD; because it is based on total dietary intake of manganese, caution should be used when applying the RfD to a single source of manganese intake.

An inhalation RfC of 4×10^{-4} mg/m^3 was derived on the basis of an epidemiological study of workers exposed to manganese (Roels et al. 1987). The critical effects reported were an increased prevalence of respiratory symptoms and psychomotor disturbances. An uncertainty factor of 300 was used in deriving the RfC to account for protection of sensitive human subpopulations and use of the LOAEL instead of the NOAEL. In addition, a modifying factor of 3 was used to account for uncertainty in the level of manganese exposure. The inhalation RfD, derived from the RfC, is given in Table 4.3.

Toxic effects from chronic inhalation exposure to manganese have been reported in workers. Exposure to concentrations as low as 0.3 mg/m^3 have been associated with manganism, a neurological disorder. Degenerative changes in the central nervous system are the major effects, but effects on the liver are also frequently observed. Studies of humans and experimental animals suggest that oral exposure to elevated levels of manganese can result in decreased fertility and in effects on the cardiovascular and central nervous systems.

In adults, approximately 3% of an oral dose of manganese is absorbed; however, in newborns, approximately 70% of the dose is absorbed. Absorption following inhalation and dermal exposure has not been well characterized.

4.4.2.12 Mercury

Mercury is a naturally occurring metal that is used in thermometers, batteries, and pharmaceutical products; it is also used as a catalyst in some industrial reactions. The average concentration in soil is approximately 0.03 mg/kg.

Both inorganic and organic forms of mercury have been found to be toxic to humans and experimental animals. In general, the organic forms are more toxic than the inorganic forms. Human studies indicate that the kidney and central nervous system are the main sites affected by mercury; however, the degree to which these systems are affected depends on the chemical form of mercury and the route of exposure.

Although an oral RfD and an inhalation RfC for mercury are available in HEAST (Tables 4.2 and 4.3), supporting documentation is not yet available in IRIS. Information provided in HEAST indicates that the inhalation RfC is based on studies of humans occupationally exposed to concentrations of approximately 0.009 mg/m^3 mercury (Fawer et al. 1983; EPA 1984d; Piikivi and Tolonen 1989; Piikivi and Hanninen 1989; Piikivi 1989) and that the oral RfD was derived from oral and parenteral studies in rats (Druet et al. 1978; Bernaudin et al. 1981; Andres 1984). The critical effects listed are neurotoxicity following inhalation exposure and kidney toxicity following oral exposure. Uncertainty factors of 30 and 1,000, respectively, were used to derive the RfDs. (The specific components of the uncertainty factors are not provided.)

Approximately 0.01 to 0.10% of an oral dose of metallic mercury and 2 to 15% of an oral dose of inorganic mercury is absorbed; absorption of inhaled mercury is approximately 80%. Limited data indicate that mercury is absorbed following dermal exposure.

4.4.2.13 Molybdenum

Molybdenum is present in soil at an average concentration of 2 mg/kg. In nature, the metal is found as various compounds with differing solubilities. Molybdenite (MoS_2) is associated with copper and uranium ores, and salts of molybdenum are present in ores from the Rocky Mountains. Emissions from coal-fired power plants also contain significant quantities of the metal. Molybdenum is used in semiconductors, steel alloys, catalysts, lubricants, and dyes. It is also used as a seed treatment or fertilizer to prevent deficiencies in plants.

Molybdenum is a component of at least two mammalian enzymes, one of which catalyzes the breakdown of purines to uric acid. This metal is considered essential for humans, and its toxicity in humans is low. High intake levels in an Armenian population have been implicated in a gout-like condition in which uric acid levels in blood are elevated.

Although an oral RfD for molybdenum is available in HEAST (Table 4.2), the supporting documentation is not yet available through IRIS. Information provided in HEAST indicates that the RfD was derived from a study of humans ingesting molybdenum from drinking water (Chappell et al. 1979; EPA 1990a). The critical effects were changes in biochemical indexes. The uncertainty factor used to derive the RfD is not reported. An inhalation RfC for molybdenum is not available.

Exposure for 3 to 7 years to air concentrations of molybdenum at 1 to 19 mg/m^3 produced fibrotic lung disease in workers. In animals, acute molybdenum intoxication via ingestion causes severe gastrointestinal irritation with diarrhea, coma, and death from heart failure. Humans can efficiently absorb soluble compounds of molybdenum from the digestive tract. Data are not available on absorption via inhalation or dermal exposure.

4.4.2.14 Nickel

Nickel and its compounds are widely distributed in the environment; the average concentration of nickel in soil is 40 mg/kg. Industrial applications for nickel include electroplating, various steels and alloys, ceramics, and nickel-cadmium batteries. Nickel subsulfide and nickel refinery dust are classified by the EPA as Group A human carcinogens; this classification is based on several studies from different countries that show an association of lung and nasal cancer with exposure to nickel refinery dust (Chovil et al. 1981; Enterline and Marsh 1982; Magnus et al. 1982; Peto et al. 1984). Tumors have also been produced in rats exposed to nickel subsulfide via inhalation (Ottolenghi et al. 1974). The cancer risks calculated from the individual epidemiologic studies are consistent, and the EPA has selected the midpoint of the risk range from these studies as the slope factor for exposure to nickel refinery dust. Nickel subsulfide is a major component of nickel refinery dust and, in experimental animals,

produces the highest incidence of tumors for nickel compounds. Hence, the EPA has derived an inhalation unit risk of 4.8×10^{-4} $(\mu g/m^3)^{-1}$ for nickel subsulfide that is based on the unit risk for nickel refinery dust and the known proportion of nickel subsulfide in this dust. The inhalation slope factor for nickel, calculated from the unit risk, is given in Table 4.5.

A chronic oral RfD for soluble nickel salts (Table 4.2) is based on a study in which decreased body and organ weights were observed in rats fed nickel sulfate hexahydrate at varying levels for 2 years (Ambrose et al. 1976). An uncertainty factor of 100 was used in deriving the RfD to account for interspecies extrapolation and protection of sensitive human subpopulations. In addition, a modifying factor of 3 was used because data from reproductive studies are inconclusive. Confidence in the RfD is medium. An inhalation RfC for nickel is not available from the EPA.

Small amounts of nickel are essential for normal growth and reproduction in some species and may also be essential for humans. In persons inadvertently exposed to high concentrations of nickel in the blood via hemodialysis, symptoms included nausea, vomiting, weakness, headache, and palpitations; all symptoms ceased a few hours after exposure. Only about 1 to 5% of ingested nickel is absorbed from the digestive tract, and nickel does not appear to be significantly absorbed from the respiratory tract. Metallic nickel does not readily penetrate the skin, but dermal exposure may result in contact dermatitis in sensitive individuals (about 2.5 to 5% of the general population).

4.4.2.15 Selenium

The average concentration of selenium in soil is about 0.3 mg/kg; however, concentrations may be much higher, depending on the parent rock from which the soil was derived. Significant contributors to atmospheric selenium levels are mining of uranium, lead, zinc, and phosphate; copper smelting; and seleniferous coal combustion. Selenium and its compounds are used in the electronics, glass, and rubber industries; in insecticides; in some dandruff shampoos; and in photographic and xerographic processes.

An oral RfD (Table 4.2) has been calculated from a comparative study of Chinese populations living in areas with low, medium, and high environmental concentrations of selenium (Yang et al. 1989). The subjects were evaluated for clinical and biochemical signs of selenium intoxication. The NOAEL was estimated on the basis of a correlation between blood selenium levels shown to reflect clinical signs of intoxication and the estimated levels of selenium intake associated with blood levels. An uncertainty factor of 3 was used in deriving the RfD to account for sensitive individuals. An inhalation RfC is not available from the EPA.

Selenium and most of its compounds are not considered to be carcinogenic; in fact, several studies suggest that normal amounts of dietary selenium may protect against cancer. However, selenium sulfide has been shown to be carcinogenic in animals via ingestion.

About 50 to 150 μg per day selenium is required in the human diet. However, high concentrations in the diet can cause brittleness and loss of nails and hair, dermatitis, swelling,

numbness or tingling in the extremities, and pain in the limbs. In severe cases of poisoning, paralysis may develop. Selenium compounds have not been shown to cause reproductive effects in humans or other mammals. Respiratory tract irritation is the primary effect from short-term inhalation of high levels of selenium dusts or selenium compounds in industrial settings, but nausea, elevated pulse rate, and irritation of the skin and eyes have also been reported. Dermal contact with selenium dust and selenium dioxide can cause dermatitis. Absorption is dependent on the chemical form of selenium, but limited data indicate that both elemental selenium and selenious acid are absorbed via inhalation. Estimates of selenium absorption from the digestive tract range from 44 to 100%; insoluble elemental selenium is probably poorly absorbed. Data regarding dermal absorption are not available.

4.4.2.16 Silver

Silver occurs in nature in its elemental state and in ores of lead, copper, and nickel. The concentration of silver in soil is typically less than 1 mg/kg. Coal fly ash contains up to 15 mg/kg silver, making coal combustion a significant source of silver in the atmosphere and soil. Silver is used in electroplating and in the manufacture of film, electrical conductors, coins, and jewelry. It is also used for medicinal purposes, to prevent eye infections in newborns, and for other antiseptic applications.

The oral RfD for silver (Table 4.2) is based on studies of argyria in persons administered silver for therapeutic purposes (Gaul and Staud 1935). Argyria is an irreversible gray discoloration of the skin. Confidence in the RfD is medium because some quantitative data are presented in the human studies and existing animal studies support the RfD. An uncertainty factor of 3 was used in deriving the RfD because the critical effect is primarily cosmetic and because of some limitations in the primary study. An inhalation RfC for silver is not available.

Large doses of silver nitrate cause severe irritation of the gastrointestinal tract, and exposure to dust containing high levels of silver can cause irritation of the throat and lungs and other respiratory problems. Studies of humans indicate that silver and/or its compounds are absorbed following inhalation, oral, and dermal exposure; however, the extent of absorption is not well quantified. In one study, the dermal absorption of silver nitrate was estimated to be approximately 1% of the applied dose.

4.4.2.17 Thallium

Compounds of thallium have many industrial uses (e.g., as catalysts and in alloys and optical lenses), and thallium compounds were used as rodenticides in the United States until 1972. Coal combustion releases significant quantities of thallium into the environment. Thallium occurs at low concentrations in most soil (approximately 1 mg/kg), but it can be accumulated in edible plants from soil that contains high levels of the metal.

The IRIS database contains oral RfDs for six thallium compounds, all based on a single study in which rats were administered varying levels of thallium sulfate by gavage (tube force

feeding) for 90 days (EPA 1986c). The critical effects were hair loss and increased levels of the enzymes serum glutamic-oxaloacetic transaminase and lactic dehydrogenase. Confidence in the RfDs for thallium compounds is low because supporting studies show adverse health effects at doses only slightly higher than the NOAEL and because reproductive and chronic toxicity data are lacking. An uncertainty factor of 3,000 was used in deriving the oral RfD to account for interspecies extrapolation, protection of sensitive human subpopulations, and lack of reproductive and chronic toxicity data. An inhalation RfC for thallium is not available.

Thallium is not a normal component of animal tissues, but the free cation (Tl^+) can substitute for potassium and alter many cellular processes. The free cation is readily absorbed through the skin or gastrointestinal epithelium. Thallium is one of the most toxic metals; the estimated lethal dose to humans is 8 to 12 mg/kg. Use of thallus acetate for hair removal in the 1920s and 1930s caused many serious human poisonings; symptoms included gastrointestinal irritation, severe effects on the central nervous system, and loss of vision. Experimental studies suggest that thallium induces toxic effects in the reproductive system. Chronic thallium intoxication in humans during pregnancy has also been reported to cause malformations and central nervous system defects in offspring.

4.4.2.18 Uranium

Uranium is the heaviest naturally occurring element. Natural uranium consists of three radioisotopes — uranium-238, uranium-235, and uranium-234 — in the relative abundance of 92.27, 0.72, and 0.0055% by weight, respectively. The richest uranium deposits are found in Africa, Europe, Australia, Brazil, Canada, and the Colorado mountain range of the United States. The average uranium concentration in U.S. soils is about 3 mg/kg.

Although natural uranium is radioactive, the primary health effect associated with exposure is kidney damage caused by chemical toxicity. The oral RfD derived for soluble uranium salts (Table 4.2) is based on decreased body weight and moderate kidney damage induced in rabbits fed uranyl nitrate hexahydrate for 30 days (Maynard and Hodge 1949). The confidence level for this RfD is medium. A 1,000-fold uncertainty factor was used in deriving the RfD to account for interspecies extrapolation, protection of sensitive human subpopulations, and use of the LOAEL instead of the NOAEL. No inhalation RfC is available for uranium.

About 5% of the soluble salts of uranium are absorbed via ingestion. Only a small fraction of inhaled uranium dust penetrates to the alveolar region of the lung, as indicated by low uranium levels in the lungs of workers exposed to uranium dust. Kidney toxicity, which is the main health effect of concern for soluble uranium exposure, may be reversible depending on the level of exposure. A few studies have also reported minor effects on the liver caused by oral and inhalation exposures to uranium. The radiological effects of uranium are discussed in Section 4.4.1.7.

4.4.2.19 Vanadium

Vanadium is a widespread element; its average concentration in soil is 100 mg/kg. Vanadium is associated with some uranium-bearing ores and is also a by-product of petroleum refining. Vanadium and its compounds are used in pigments, photographic materials, and insecticides; as a chemical catalyst; and in the hardening of steel.

Although an oral RfD is available in HEAST (Table 4.2), supporting documentation is not available in IRIS. Information provided in HEAST indicates that the RfD was based on a study of rats administered vanadyl sulfate in the drinking water for a lifetime (Schroeder et al. 1970; EPA 1987c). No critical effect was observed in this study. A 100-fold uncertainty factor was used in deriving the RfD. (The specific components of the uncertainty factor are not provided.) An inhalation RfC is not available from the EPA.

The oral toxicity of vanadium increases with increasing valency; pentavalent vanadium is the most toxic. Excess ingestion of vanadium has caused gastrointestinal disturbances, nervous system effects, and abnormalities in renal enzyme systems. Inhalation of vanadium compounds at levels above U.S. occupational air standards (e.g., 0.05 mg/m^3 V_2O_5) is strongly irritating to the eyes and airways, and chronic inhalation may result in decreased cholesterol synthesis and the alteration of various enzyme systems. Solutions of vanadium compounds can be absorbed through the skin, but dermal absorption of elemental vanadium is a minor route of uptake. Vanadium is absorbed more efficiently via inhalation than ingestion. Estimates of oral absorption of vanadium range from 0.1 to 2%.

4.4.2.20 Zinc

Zinc is an abundant element and is present in soil at an average concentration of approximately 50 mg/kg. As a component of galvanized steel, zinc is used extensively in the machining, building, and automotive industries. Zinc compounds are also used in the production of plastics and batteries, and as wood preservatives. Zinc is an essential dietary element for humans and animals. The average daily intake is 12 to 15 mg per day, mostly from food.

Although an oral RfD for zinc is available in HEAST (Table 4.2), supporting documentation is not available in IRIS. Information provided in HEAST indicates that the RfD is based on studies of humans administered zinc for therapeutic purposes (Pories et al. 1967; Prasad et al. 1975); the critical effect was anemia. Long-term ingestion of elevated levels of zinc interferes with copper absorption, leading to anemia. An uncertainty factor of 10 was used to derive the RfD. (The specific components of the uncertainty factor are not provided.) An inhalation RfC for zinc is not available from the EPA.

In general, zinc deficiencies are of greater health significance than overexposure to zinc. However, occupational exposures to zinc oxide fumes can result in metal fume fever, a condition characterized by influenza-like symptoms. In humans, absorption of zinc from the gastrointestinal tract is controlled by homeostatic mechanisms; approximately 20 to 50% of ingested zinc is absorbed.

4.4.3 Inorganic Anions

4.4.3.1 Fluoride

Fluoride is abundant in soil at an average concentration of approximately 200 mg/kg. Daily intake of fluoride, which is present in a number of foods, ranges from approximately 0.3 to 2 mg, depending on whether the drinking water is fluoridated.

An oral RfD (Table 4.2) has been derived from a study of children consuming fluoride in the drinking water (Hodge 1950); the critical effect was dental fluorosis. Confidence in the RfD is high because the effect observed was in the most sensitive subpopulation (children) and because a large number of studies are available that support the RfD. For these same reasons, an uncertainty factor of 1 was used in deriving the RfD. An inhalation RfC is not available in IRIS or HEAST.

Chronic effects of exposure to fluoride are mostly related to fluoride-sensitive tissues, i.e., the teeth and bones. The main effect in teeth is dental fluorosis, considered to be a cosmetic rather than a toxic effect. Fluorosis occurs during calcification of the teeth in children up to 8 years old. Crippling skeletal fluorosis has been observed only after chronic exposure to high doses of fluoride (over 20 mg per day) for a period of approximately 10 to 20 years. Fluoride is rapidly absorbed from the intestinal tract.

4.4.3.2 Nitrate and Nitrite

Nitrates are widespread throughout the environment. Nitrates are contributed to soil, surface water, and groundwater as a result of urban sewage treatment and through nitrogen-based fertilizers and nitrogenous wastes from livestock and poultry production. Nitrite is formed from nitrate or ammonium anion by microorganisms in soil, water, and sewage, as well as in the human alimentary tract. Nitrates and nitrites are naturally present in vegetables, grains, and other foods. Nitrates and nitrites are used extensively for color enhancement and preservation of processed meats. In humans, the estimated total daily intake of nitrate from all sources is 75 mg.

The oral RfDs for nitrate and nitrite (Table 4.2) are based on a study of methemoglobinemia in infants consuming formula prepared from water containing levels of nitrate ranging from 10 to more than 100 mg/L (Bosch et al. 1950; Walton 1951); methemoglobinemia was not induced at levels of less than 10 mg/L nitrate, as nitrogen, in water. Methemoglobinemia is a condition resulting in a reduced capacity of the blood to transport oxygen to the tissues and is caused by the reaction of nitrite with blood hemoglobin. Because nitrate is reduced to nitrite by bacteria in the saliva and alimentary tract, the human health concern associated with nitrate results from its conversion to nitrite. Infants have a high content of nitrate-reducing bacteria in the gastrointestinal tract and are therefore particularly sensitive to nitrates and nitrites. The NOEL of 10 mg/L was used to derive RfDs of 1.6 mg/kg per day for nitrate and 0.1 mg/kg per day for nitrite. The RfD for nitrite is less than that for nitrate because

a modifying factor was applied to account for the direct toxicity of nitrite. Uncertainty factors were not used to derive the RfDs for either nitrate or nitrite because the NOEL was the critical toxic effect (methemoglobinemia) and the effect was observed in the most sensitive subpopulation (infants). In addition, more recent studies are available that support the NOEL of 10 mg/L (EPA 1985a). For these same reasons, the confidence in the RfD is high. Inhalation RfCs are not available for nitrate or nitrite.

In addition to interacting with hemoglobin, nitrites react with secondary amines (present in foods) to form N-nitroso compounds. Many N-nitroso compounds are carcinogenic in experimental animals and exert other toxic effects. However, the contribution of nitrites and, indirectly, nitrates to a potential carcinogenic hazard for humans and the magnitude of the associated risk are unclear.

Nitrate and nitrite are absorbed well from the gastrointestinal tract. Nitrate is absorbed by active transport from the gastrointestinal tract, and nitrite is absorbed by diffusion across the gastric mucosa or gastrointestinal tract.

4.4.4 Asbestos

Asbestos is a generic term used to describe a group of naturally occurring, fibrous silicate minerals. Asbestos fibers are strong and resistant to heat and chemicals. Because of these properties, asbestos has been used in a wide variety of applications, e.g., in insulation and other building materials and in friction- and heat-resistant products.

Asbestos is classified by the EPA as a Group A human carcinogen. Increased incidences of many cancers — including lung cancer, mesothelioma, and gastrointestinal cancer — are correlated with asbestos exposure. Asbestosis (an accumulation of scar-like tissue in the lungs) is the main noncarcinogenic effect associated with inhalation exposure.

The EPA has developed an inhalation unit risk of 0.23 $(\text{fibers/mL})^{-1}$ for airborne asbestos, which considers the additive risk of both lung cancer and mesothelioma (EPA 1991c). The unit risk for asbestos represents a composite value for males and females, assuming continuous exposure and using U.S. general cancer rates and mortality patterns without considering smoking habits. (The epidemiological data show that cigarette smoking and asbestos exposure interact synergistically in the induction of lung cancer but not mesothelioma.) A relative risk model was used to assess the dose-response data for lung cancer, and an absolute risk model was used for mesothelioma (EPA 1986a; ATSDR 1990b).

The characteristics of the cohort (worker populations) for which EPA's unit risk was estimated differ from those of potential receptors considered for the Weldon Spring site, e.g., in terms of age and time since onset of assumed exposure. Therefore, the basic data used by the EPA to develop the asbestos unit risk was adapted to conditions assumed for the Weldon Spring site to determine whether a revised risk factor might better represent the risk to potential receptors. A site-specific risk factor of 0.22 $(\text{fibers/mL})^{-1}$ was derived by using EPA data tables (EPA 1986a) and assuming (1) the receptor age at onset of exposure is 10, (2) exposure is

continuous, and (3) excess cancer deaths are estimated at 70 years. Because this value is very similar to that identified by the EPA, the standard EPA unit risk of 0.23 $(\text{fibers/mL})^{-1}$ was considered reasonable for use in estimating potential lifetime risks of lung cancer and mesothelioma from asbestos exposure associated with the Weldon Spring site.

4.4.5 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are a diverse class of compounds that are formed during the incomplete burning of organic substances such as coal, oil, and gas. These compounds are found in the air attached to dust particles, and they are emitted from vehicle exhausts, asphalt roads, and wood- and coal-burning furnaces. The only known use of PAHs is for research purposes. The concentration of the best-studied PAH, benzo(a)pyrene, in rural soil is reported to range from 0.002 to 1.3 mg/kg; the range in urban soil is 0.17 to 0.22 mg/kg.

The following PAH compounds are classified by the EPA as Group B2 probable human carcinogens: benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene. The PAHs induce cancer in laboratory animals by oral, inhalation, or dermal exposure. Oral slope factors and inhalation unit risks are not available in IRIS for any of the PAHs and are given only for benzo(a)pyrene in HEAST (Tables 4.4 and 4.5). The inhalation slope factor is based on the induction of respiratory tract tumors in hamsters exposed for 109 weeks (Thyssen et al. 1981; EPA 1984e). The oral slope factor is based on increased incidence of forestomach tumors in mice fed various doses of benzo(a)pyrene for periods of 30 to 197 days (Neal and Rigdon 1967; EPA 1984e). Information provided in HEAST indicates that only the inhalation slope factor has been verified by the EPA. The slope factors for benzo(a)pyrene were applied to the remaining carcinogenic PAHs for the purpose of quantifying their risks in this BA.

Because PAHs occur together in the environment and because exposure is almost exclusively to groups of PAHs, it has not been possible to identify specific PAHs that increase cancer risk in humans. However, epidemiologic studies have shown increased mortality due to lung cancer in humans exposed to major sources of PAHs, such as coke-oven emissions, roofing-tar emissions, and cigarette smoke.

Several PAH compounds may cause noncarcinogenic effects rather than carcinogenic effects. Although not technically classified as PAHs, naphthalene and 2-methylnaphthalene are included with this group for simplicity of presentation. Oral RfDs have been derived for six of the noncarcinogenic PAHs — acenaphthene, anthracene, fluoranthene, fluorene, naphthalene, and pyrene (Table 4.2). In a series of similar experiments, mice or rats were administered the test agent by gavage for approximately 90 days (National Toxicology Program 1980; EPA 1988c, 1989d, 1989e, 1989f, 1989l). The critical effects were hepatotoxicity (acenaphthene); nephropathy, increased liver weights, hematological alterations, and clinical effects (fluoranthene); decreases in red blood cells, packed cell volume, and hemoglobin (fluorene); decreased body weight gain (naphthalene); and kidney effects (pyrene). No adverse effects were observed following treatment with anthracene. In addition to these effects, decreases in fertility have been observed in laboratory animals fed high levels of PAHs.

Except for naphthalene, an uncertainty factor of 3,000 was used for deriving the RfDs for all compounds to account for interspecies extrapolation, protection of sensitive human subpopulations, lack of a chronic assay, and lack of data on reproductive and developmental toxicology. A very high uncertainty factor of 10,000 was used to derive the RfD for naphthalene (the specific components of the uncertainty factor are not provided). On the basis of structural analogy, the oral RfD for naphthalene (a two-membered ring compound) was applied to 2-methylnaphthalene and the oral RfD for pyrene (the lowest RfD of the compounds with three or more rings) was used to assess the remaining noncarcinogenic PAHs for which compound-specific RfDs have not been derived. The oral RfD for pyrene was also used to assess the potential noncarcinogenic effects of the carcinogenic PAHs.

4.4.6 Polychlorinated Biphenyls

Polychlorinated biphenyls (PCBs) are a family of synthetic organic chemicals comprising more than 200 individual compounds. Commercial formulations are mixtures consisting of a number of PCBs and impurities. Because of their insulating and nonflammable properties, PCBs have been used widely as coolants, lubricants, and dielectric fluids in transformers, capacitors, and other electrical equipment.

As a group, PCBs are classified by the EPA as a Group B2 probable human carcinogen. An oral slope factor (Table 4.4) has been derived from a study of rats administered Aroclor 1260 in their feed (Norback and Weltman 1985); the predominant tumor was hepatocellular carcinoma. Although the potency of PCB congeners varies greatly, the EPA considers the slope factor derived for Aroclor 1260 to be representative of all PCB mixtures. The EPA has not derived an inhalation unit risk for PCBs.

Oral and inhalation RfDs for PCBs are not available from IRIS or HEAST. However, the ATSDR has derived an MRL of 1×10^{-4} mg/kg per day for PCBs from a study in which female monkeys were fed Aroclor 1016 for 7 months prior to mating and during pregnancy; the critical effect was a decrease in birth weights of the offspring. This value has been adopted for the quantitative risk assessment in this BA.

Other toxic effects observed in experimental animals chronically exposed to PCBs include severe weight loss, liver damage, toxicity to the immune system, adverse reproductive effects, and malformations in offspring. In humans exposed to these compounds, the skin and liver are the primary sites affected, but effects on the gastrointestinal and neuromuscular systems are also observed. The only significant adverse health effects observed in PCB-exposed workers are occasional skin irritation and liver damage.

Following oral exposure, more than 90% of most PCB congeners is absorbed. Limited data indicate that PCBs can also be absorbed following exposure by inhalation or dermal contact.

4.4.7 Nitroaromatic Compounds

4.4.7.1 1,3-Dinitrobenzene

The synthetic compound 1,3-dinitrobenzene (DNB) is used as an intermediate in the manufacture of dyes; formerly, it was used as an explosive. The compound DNB also occurs as a by-product of the manufacture of TNT and nitrobenzene, and as a possible degradation product of the photodecomposition of dinitrotoluenes.

An oral RfD (Table 4.2) was derived from a study of rats administered DNB in drinking water for 16 weeks (Cody et al. 1981). The critical effect was an increase in spleen weight. Other observed effects included decreased hemoglobin concentrations and testicular atrophy. Confidence in the RfD is low. An uncertainty factor of 3,000 was used in deriving the RfD to account for interspecies extrapolation, protection of sensitive human subpopulations, lack of a chronic assay, and lack of data on reproductive toxicity. An inhalation RfC is not available.

The major acute effect of DNB in humans is cyanosis (a lack of oxygen, characterized by a bluish coloration of the skin); other reported effects include methemoglobinemia and anemia. Prolonged exposure can lead to liver damage. In rats administered DNB for 18 weeks, neurological disturbances, toxicity to the spleen and liver, testicular atrophy, reduced growth rates, and decreased hemoglobin levels were observed. Approximately 60 to 90% of an oral dose is absorbed in experimental animals; DNB is also absorbed following inhalation and dermal exposure.

4.4.7.2 2,4-Dinitrotoluene and 2,6-Dinitrotoluene

The principal isomers of technical grade dinitrotoluene (tDNT) are 2,4-dinitrotoluene (2,4-DNT) and 2,6-dinitrotoluene (2,6-DNT); all isomers are synthetic compounds. The primary uses of tDNT are in the manufacture of explosives, dyes, and pigments and as an intermediate in the manufacture of polyurethane. The DNT isomers occur as impurities in TNT and as constituents of wastewaters from the manufacture of TNT.

The EPA classifies "dinitrotoluene mixture" (containing both 2,4-DNT and 2,6-DNT) as a Group B2 probable human carcinogen. An oral slope factor (Table 4.4) has been derived from a study of rats and mice administered DNT in their feed (Ellis et al. 1979). The main tumor observed was hepatocellular carcinoma; increased incidences of mammary gland tumors and kidney tumors were also reported. An inhalation unit risk for DNT is not available.

Neither an oral RfD nor an inhalation RfC for DNT are available from IRIS or HEAST. However, the ATSDR has derived an intermediate MRL of 2×10^{-3} mg/kg per day for 2,4-DNT and a chronic MRL of 4×10^{-3} mg/kg per day for 2,6-DNT from studies of dogs administered the test compounds in capsules for 13 weeks and 24 months, respectively (Ellis et al. 1979; ATSDR 1989d). The critical effects observed were neurological effects for the 2,4-isomer and anemia for the 2,6-isomer. An uncertainty factor of 100 was used in deriving the MRLs to

account for interspecies extrapolation and protection of sensitive human subpopulations. Because the MRL for 2,4-DNT is equivalent to a subchronic RfD, the MRL was divided by an additional uncertainty factor of 10. These values have been adopted for the quantitative assessment in this report.

The first clinical manifestation of exposure to DNT is methemoglobinemia. A hemolytic anemia may result, depending on the severity of exposure. Other effects include toxicity of the liver, kidney, and nervous system. Epidemiological studies of workers exposed to DNT found an excess mortality from ischemic heart disease (due to lack of blood supply) and other diseases of the cardiovascular system. Although no effects on the reproductive system (e.g., abnormal sperm count or morphology) have been observed in male workers exposed to DNT, adverse effects — including testicular atrophy and decreased spermatogenesis — have been observed in experimental animals.

Studies of several animal species indicate that approximately 75 to 100% of an oral dose of 2,4-DNT is absorbed; however, mice absorb only about 17%. Studies of humans indicate that nitroaromatic compounds are absorbed following inhalation and ingestion and that these compounds are also capable of penetrating the skin.

4.4.7.3 Nitrobenzene

Nitrobenzene (NB) is a synthetic compound used primarily as an intermediate in the manufacture of aniline, a starting material for dyes and other chemicals. Nitrobenzene is very unstable and is rarely found in the environment.

An oral RfD and inhalation RfC are available for NB. The oral RfD, which is based on the same study used to derive the inhalation RfC, is available from IRIS; however, the inhalation RfC is available only from HEAST. The principal study used to derive the RfDs was an inhalation study of rats and mice exposed to NB (Chemical Industry Institute of Toxicology 1984). The critical effects observed were anemia and lesions of the liver and kidney. Information on the uncertainty factors used to derive the inhalation RfC is not available. Confidence in the oral RfD is low. A very high uncertainty factor of 10,000 was used to derive the oral RfD. This factor accounts for interspecies extrapolation, protection of sensitive human subpopulations, absence of data from a chronic study, and use of the LOAEL rather than the NOAEL to derive the RfD. The oral RfD is given in Table 4.2; the inhalation RfD, derived from the RfC, is given in Table 4.3.

Nitrobenzene is extremely toxic, and acute exposure to high levels causes respiratory failure. Both acute and subchronic exposures result in methemoglobinemia; other effects observed following subchronic exposure include hematologic effects, testicular atrophy, and lesions of the kidney, liver, and adrenal gland. Nitrobenzene is rapidly absorbed through the skin. Human studies indicate that it is absorbed following inhalation exposure. In animals, NB is metabolized by intestinal bacteria.

4.4.7.4 1,3,5-Trinitrobenzene

Although it is an effective explosive, 1,3,5-trinitrobenzene (TNB) is not produced in large quantities. The compound TNB is an end product of TNT photolysis and is present in surface soil and surface water contaminated with TNT.

Data on the toxicity of TNB are very limited. However, because of the structural similarity of this compound to DNB, an oral RfD (Table 4.2) was derived for TNB from the previously described study for DNB. Confidence in the RfD for TNB is low. A very high uncertainty factor of 10,000 was used to derive the RfD. The same 1,000-fold uncertainty factor used to derive the RfD for DNB was applied to account for interspecies extrapolation, protection of sensitive human subpopulations, and lack of a chronic assay; an additional factor of 10 was used to account for derivation of the RfD from a structurally analogous compound. An inhalation RfC is not available. Information is not available on the absorption of TNB for any route of exposure.

4.4.7.5 2,4,6-Trinitrotoluene

Historically, 2,4,6-trinitrotoluene (TNT) has been the most important and widely used secondary explosive, i.e., an explosive that is detonated by a more sensitive primary explosive. The compound TNT is classified by the EPA as a Group C possible human carcinogen. An oral slope factor (Table 4.4) has been derived from studies of rats administered TNT in the feed (Furedi et al. 1984); the primary tumor was carcinoma of the urinary bladder. An inhalation slope factor is not available.

An oral RfD (Table 4.2) has been derived from a 26-week study of dogs administered TNT in capsules (Levine et al. 1983); the critical effect was liver damage. Although this study was well designed, the level of confidence in the RfD is medium because of the method of administration and the lack of the NOAEL. An uncertainty factor of 1,000 was used in deriving the RfD to account for interspecies extrapolation, protection of sensitive human subpopulations, use of the LOAEL instead of the NOAEL to derive the RfD, and the short duration of the study. An inhalation RfC is not available.

The main site of TNT-induced toxicity is the hematopoietic system, and the effects include methemoglobinemia and aplastic anemia (a loss in the capacity of the bone marrow to produce blood cells). Other adverse effects include liver damage (hepatitis and jaundice), cataracts, and dermatitis. Testicular atrophy has been reported in experimental animals following exposure to high levels of TNT.

Approximately 60 to 75% of the dose is absorbed following oral administration of TNT to rats. Studies of humans indicate that nitroaromatic compounds are absorbed following inhalation and ingestion and that these compounds are capable of penetrating the skin.

5 HEALTH RISK CHARACTERIZATION

Radiological and chemical health risks were assessed for exposures to contamination at the Weldon Spring site (Section 5.2) and at adjacent properties impacted by site releases (Section 5.3). Potential carcinogenic risks for both radiological and chemical exposures were assessed in terms of the increased probability that an individual would develop cancer over a lifetime. In its National Oil and Hazardous Substances Pollution Contingency Plan (NCP), the EPA has defined general remedial action goals for NPL sites (EPA 1990c). These goals include a range for residual carcinogenic risk, which is an excess upper bound lifetime cancer risk to an individual of between 1×10^{-6} and 1×10^{-4}, or 1 in 1 million to 1 in 10,000. For comparison, about one in three Americans will develop cancer from all sources, and it is estimated that 60% of all cancers are fatal (American Cancer Society 1992); this translates to a baseline risk of about 2×10^{-1}, or 1 in 5.

The EPA risk range is referred to as the target range in this discussion, and it is used as a point of reference for the risks estimated from site exposures. Identifying a risk as being within this range does not exclude it from further consideration relative to forthcoming cleanup decisions. The final remedial action goals for the site will be determined from various analyses in the RI/FS-EIS, including the preliminary application of DOE's as low as reasonably achievable (ALARA) process for reducing exposures and risks, which is discussed in Chapter 2 of the FS.

The potential for health effects other than cancer from exposure to chemical contaminants was also assessed for on-site and off-site areas (Sections 5.2 and 5.3). The quantitative measures of noncarcinogenic health effects are the hazard quotient and hazard index (Section 5.1.2.2). The EPA has defined an endpoint-specific (segregated) hazard index of greater than 1 as the level of concern for noncarcinogenic health effects (EPA 1989j).

The same individual could be exposed to both on-site and off-site exposure points, and the risks and hazard indexes associated with such exposures are presented in Section 5.4. The dermal pathway is discussed in Section 5.5, and the uncertainties associated with the risk characterization are discussed in Section 5.6. The results of the health risk characterization are summarized in Section 5.7.

5.1 RISK CHARACTERIZATION METHODOLOGY

5.1.1 Radiological Risks

Exposures to low levels of ionizing radiation can result in cancer induction, serious genetic effects, and other detrimental health effects. The predominant health concern associated with the radioactive contaminants at the Weldon Spring site (which are primarily alpha-emitting radionuclides) is the induction of cancer. The radiological health risks presented in this BA are limited to this concern. This approach is consistent with EPA guidance, which notes that, in

general, the risk of cancer is limiting and may be used as the sole basis for assessing the radiation-related human health risks for a site contaminated with radionuclides (EPA 1989j).

The radiation doses associated with the various scenarios considered in this assessment are presented in Section 3.4. These doses are given in units of mrem for all exposure routes except inhalation of radon-222, radon-220, and their short-lived decay products; doses for these exposures are given in units of WLM. Radiological risks were calculated by multiplying the doses by specific risk factors to estimate the probability of cancer induction per unit dose. Three separate risk factors were used, as follows: 3.5×10^{-4}/WLM for inhalation of radon-222 and its short-lived decay products; 1.2×10^{-4}/WLM for inhalation of radon-220 and its short-lived decay products; and 6×10^{-7}/mrem for all other exposure routes. The basis for these risk factors is discussed in Section 4.1.

The radiological risks associated with exposures to contaminants at the Weldon Spring site are considered in the context of risks resulting from natural sources of radiation. In most cases, the radiological health risks given in this document are reported as incremental risks above that from background sources of radiation. This approach facilitates comparison with EPA's target risk range, which does not include the contribution from background sources of radiation. Radiation exposure from natural sources of radiation results in an annual dose of about 300 mrem/yr — 200 mrem/yr from exposure to radon-222 and its short-lived decay products and 100 mrem/yr from exposure to other natural sources of radiation (National Council on Radiation Protection and Measurements 1987). By applying the appropriate risk factors and the conversion factor of 1 WLM/rem for radon-222 exposures (ICRP 1981a), this dose results in an annual risk of cancer induction of 1×10^{-4}/yr. The EPA has estimated that the individual lifetime (70-year) risk of fatal cancer associated with background radiation, primarily from radon, is 1×10^{-2} (EPA 1989g).

5.1.2 Chemical Risks and Hazard Quotients

5.1.2.1 Carcinogenic Risks

The risk to an individual resulting from exposure to chemical carcinogens is expressed as the increased probability of cancer over a lifetime. To calculate the excess cancer risk, the daily intake averaged over a lifetime is multiplied by a chemical-specific slope factor. The EPA has derived slope factors for a number of carcinogens, and these factors represent the incremental lifetime cancer risk per milligram of carcinogen per kilogram of body weight, assuming that the exposure occurs over a lifetime of 70 years. The estimated average daily intakes resulting from exposure to the chemical carcinogens of concern for the Weldon Spring site are given in Section 3.4, and the slope factors for these carcinogens are identified in Section 4.3.

A slope factor is specific to the chemical and the route of exposure, i.e., inhalation or ingestion. Slope factors are not available for all carcinogenic contaminants considered in this analysis for both routes of exposure. For the oral route (ingestion), slope factors are available for arsenic, beryllium, 2,4-DNT, 2,6-DNT, TNT, PAHs, and PCBs. Lead and asbestos are the only

carcinogens for which oral slope factors are not available. (For certain metals that are carcinogenic by the inhalation route — i.e., cadmium, chromium VI, and nickel — there is insufficient evidence for their carcinogenicity by the oral route. Hence, oral slope factors have not been derived.) For the inhalation route, slope factors are available for arsenic, beryllium, cadmium, chromium VI, nickel, PAHs, and asbestos. Inhalation slope factors are not available for lead, 2,4-DNT, 2,6-DNT, TNT, or PCBs. The carcinogenic risks associated with contaminants for which a slope factor is not available cannot be quantitatively assessed; therefore, the potential risks associated with these contaminants are discussed qualitatively (Section 5.6.4).

5.1.2.2 Hazard Quotients and Hazard Indexes

A hazard quotient provides a measure of the potential for adverse health effects other than cancer. For an individual contaminant, the daily intake averaged over the exposure period is divided by the RfD to derive the hazard quotient. The RfD is the average daily dose that can be incurred without an appreciable risk of deleterious health effects during a lifetime. The EPA has derived RfDs for both chronic exposure (more than 7 years) and subchronic exposure (less than 7 years). Because the potential exposures considered in this BA are for periods of more than 7 years, only chronic RfDs have been considered. The estimated average daily intakes resulting from exposure to the contaminants of concern at the site are given in Section 3.4, and the RfDs for these contaminants are identified in Section 4.2.

Similar to carcinogenic slope factors, RfDs are specific to the contaminant and route of exposure. For the oral route, RfDs are available for most of the contaminants of concern; only cobalt, lead, and asbestos do not have an oral RfD. In contrast, inhalation RfDs are available for only a few contaminants of concern — i.e., barium, cadmium, chromium III, chromium VI, manganese, mercury, and nitrobenzene. Because of the general lack of inhalation RfDs, the inhalation hazard index is significantly underestimated. However, the importance of the air pathway for chemical contaminants at the site is relatively minor. The uncertainty in the assessment resulting from lack of RfDs is discussed in Section 5.6.4.

For lead, the EPA has developed an uptake/biokinetic model to estimate blood lead levels in children from exposures to lead in food, drinking water, air, soil, and paint. This model is applied to assess residential exposures for the most sensitive subpopulation, children aged 0 through 6; therefore, it is not directly applicable to the receptors evaluated in this BA. However, children were considered in evaluating the residential scenario for the rebaseline assessment (Appendix E of the FS), and the model was applied for that analysis. For these reasons, exposures to lead have not been quantified in this baseline assessment.

Potential noncarcinogenic health effects were quantified for those contaminants for which an RfD is available by estimating a pathway-specific hazard quotient. The individual hazard quotients determined for each contaminant were then summed to determine the hazard index; if the total hazard index is greater than 1, a potential concern may be indicated. The total hazard index for an individual exposure point can exceed 1 as a result of the presence of either (1) a single contaminant with a hazard quotient greater than 1 or (2) several contaminants whose hazard quotients add to greater than 1. For the latter case, a more detailed analysis is required

to determine whether the potential for a noncarcinogenic health effect has been overestimated by the total hazard index. In such cases, the major toxicological effects and mechanisms of action of the individual contaminants are identified to assess the impact from exposure to multiple contaminants. This process is referred to as segregation of the hazard index.

For this BA, the primary contributors to the total hazard index were grouped according to health end points and a separate hazard index was derived for each group. The approach used to group these contaminants was to consider those effects that occur at levels of exposure corresponding to the LOAELs or "critical effects" identified by the EPA. Effects for dose levels slightly above the LOAEL were also considered as appropriate to the specific contaminant.

Except for arsenic and nitroaromatic compounds, the end points considered for segregation of the hazard index were those listed as critical effects in Tables 4.1 and 4.2. Because many of the toxic effects of arsenic occur at doses just slightly higher than that associated with the LOAEL, these additional effects have also been considered. As a class, the nitroaromatic compounds induce methemoglobinemia. Because these compounds all act by the same mechanism of action, their hazard quotients were also summed. Those contaminants for which a critical effect is not given or for which the critical effect is only a nonspecific indication of general toxicity (e.g., decreased weight gain) have been grouped together for this assessment. The grouping of contaminants by toxic effect for the purpose of segregating the hazard index for the Weldon Spring site is shown in Table 5.1.*

5.2 RISK ESTIMATES FOR ON-SITE EXPOSURE POINTS

5.2.1 Soil

The potential routes of exposure to on-site surface soil quantified in this assessment are external gamma irradiation and incidental ingestion of soil. (Potential exposure to contaminated airborne particulates and radon generated from site soil is discussed in Section 5.2.3.) Under current site conditions, the potential receptors are a maintenance worker and trespasser. Under hypothetical future land-use conditions, for which it is assumed that DOE no longer controls the site and the fence is no longer present, the potential receptor is a recreational visitor. The risks associated with exposures of these receptors to contaminated soil were determined from both the sitewide UL_{95} concentration for each contaminant and the contaminant concentrations at each specific sampling location.

The risk estimates based on sitewide UL_{95} values are representative for an individual exposed to all areas of the site as a result of random access over the 10- or 30-year exposure period. The location-specific analysis provides risk estimates associated with discrete areas, as

*For readability, all tables are presented in sequence at the end of the text and figures in this chapter. Only those contaminants detected in the specific medium and relevant to the health effect (i.e., carcinogenic or noncarcinogenic effect) are included in these tables. Because values were rounded to one significant figure, summations within tables will not be exact.

represented by each individual sampling location. The purpose of this analysis is to address the nonuniform distribution of site contaminants. However, the underlying assumption in the location-specific analysis — that the entire exposure occurs at a single location — is not realistic for the receptors considered in this BA. (It is more representative of exposures to a hypothetical resident, which are evaluated in the rebaseline assessment presented in Appendix E of the FS [DOE 1992a].) For example, it is highly improbable that a trespasser or recreational visitor would repeatedly visit the same location over the entire exposure duration. The assumptions that the entire exposure could occur at a single location and that all areas of the site, including source areas, could be accessed would not be representative for a worker because this individual would be involved in sitewide maintenance activities and would take protective measures in highly contaminated areas. Nevertheless, results for the worker provide an indication of relative risk by location for a greater extent of exposure (e.g., a higher exposure time and frequency) compared to the trespasser and recreational visitor, thereby presenting an upper bound estimate of potential risks for those two receptors.

Contaminant-specific exposure point concentrations were determined for radionuclides in site soil from the measured values of uranium-238, thorium-230, radium-228, and radium-226 and the relationships indicated by the radiological source term analysis. Measured gamma exposure rates were used to calculate risks from external gamma radiation.

5.2.1.1 Radiological Risks

The radiological risks from exposures to sitewide soil are 8×10^{-5} for the worker, 6×10^{-7} for the trespasser, and 1×10^{-5} for the future recreational visitor. These risks result from incidental ingestion of soil and external gamma irradiation. Incidental ingestion accounts for about 70% of the risk from soil exposures for the worker and recreational visitor and about 85% for the trespasser; the major contributor to the ingestion risk is lead-210 (about 40%). Individual contaminant-specific risks from exposures to sitewide soil are presented in Table 5.2. The total risks for each receptor are within or below EPA's target range.

Results of the location-specific analysis indicate that the maximum risks from incidental ingestion and external gamma irradiation are 2×10^{-3} for the worker, 1×10^{-5} for the trespasser, and 2×10^{-4} for the recreational visitor. In general, incidental ingestion accounts for slightly more than 70% of the combined risk from these two pathways for the trespasser; for the other two receptors, incidental ingestion and external gamma irradiation contribute about equally to the combined risk. The major contributor to the ingestion risk is lead-210, which accounts for 25% of the combined risk for the trespasser and about 20% for the recreational visitor and worker.

5.2.1.2 Chemical Carcinogenic Risks and Hazard Indexes

The total chemical carcinogenic risks associated with incidental ingestion of sitewide soil are 1×10^{-5} for the worker, 2×10^{-7} for the trespasser, and 2×10^{-6} for the future recreational visitor (Table 5.3). The relative contribution of the individual contaminants to the risks for the

worker (the most exposed receptor) are shown in Figure 5.1.* Because the relative contribution of the contaminants for the trespasser and recreational visitor are the same as those for the worker, they are not presented separately. Arsenic, PCBs, and PAHs contribute about 50, 30, and 15%, respectively, to these risks.

The location-specific analysis indicates that the maximum carcinogenic risks from soil ingestion are 1×10^{-4} for the worker, 1×10^{-6} for the trespasser, and 2×10^{-5} for the recreational visitor. These risks are associated with a concrete pad adjacent to Building 403, a former pilot-plant building. Solvents were used in this building, and the adjacent soil is contaminated with PAHs. These compounds account for more than 95% of the risk, and arsenic contributes most of the remainder. Except for this location, risks to the three receptors are within or below EPA's target range.

The hazard indexes associated with ingestion of sitewide soil are 0.3 for the worker, 0.004 for the trespasser, and 0.02 for the recreational visitor (Table 5.4). Because the hazard indexes are less than EPA's reference level of 1, noncarcinogenic effects are not expected to result from ingestion exposure for any of the receptors. Nevertheless, to indicate the primary contributors to the hazard index estimated for the worker (the maximally exposed receptor) from incidental ingestion, relative contaminant contributions are shown in Figure 5.2. The primary contributors are uranium (35%), arsenic (20%), thallium (10%), and PCBs (10%).

The results of the location-specific analysis indicate that no noncarcinogenic effects would be expected for the on-site trespasser or recreational visitor at any location, but the total (unsegregated) hazard index from ingestion exceeds 1 for the worker (ranging from 1 to 4) at five locations. These locations include the southern end of the site, two locations next to a former uranium processing building (Building 301), the North Dump, and the South Dump. Uranium is the primary contributor to the hazard index from ingestion at these locations (60 to more than 95%), and arsenic and thallium also contribute at the South Dump (2 to 20%). Because uranium dominates the estimate for most locations, segregating the hazard index for the end point and mechanism of action does not change the overall result.

5.2.2 Surface Water and Sludge

The routes of exposure to on-site surface water and sludge quantified in this assessment are external gamma irradiation, ingestion of surface water, and incidental ingestion of sludge. Under current site conditions, the potential receptor is a trespasser; under future land-use conditions, the potential receptor is a recreational visitor. Because the raffinate pits constitute the most highly contaminated surface impoundment on-site, the carcinogenic risks and hazard quotients were assessed for exposures to surface water and sludge in these pits. The maximum measured contaminant concentrations in the four pits combined were used to estimate contaminant intakes.

*For readability, all figures are presented in sequence at the end of the text in this chapter.

5.2.2.1 Radiological Risks

The raffinate pits are the single largest contributor to the radiological risks at the Weldon Spring site because of the high concentrations of radioactive contaminants in the sludge. The total radiological risks for ingestion of water are estimated to be 8×10^{-6} for the trespasser and 1×10^{-4} for the future recreational visitor (Table 5.5). The major contributors to these risks are uranium-238 and thorium-230, which together account for about 85% of the total. The risk to the current trespasser is within EPA's target range, whereas the risk to the future recreational visitor is just below the upper limit of the range.

The radiological risks associated with incidental ingestion of raffinate pit sludge are 2×10^{-4} for the current trespasser and 3×10^{-3} for the future recreational visitor (Table 5.5). The major contributors to this risk are actinium-227, protactinium-231, and thorium-230, which together account for about 75% of the total risk. Thorium-230 contributes more than any other radionuclide to this risk (about 30%), which reflects its very high concentrations in the raffinate pit sludge. The radiological risks to the current trespasser and future recreational visitor both exceed the upper limit of EPA's target range.

The radiological risk from external gamma radiation at the raffinate pits is currently low because the pits are filled with water, which effectively attenuates the gamma radiation from the underlying sludge. However, this exposure could be significant if the water levels in these pits were to recede (e.g., during a very dry summer). The risk to the current trespasser from external gamma irradiation is estimated to be 2×10^{-4}; this estimate was based on an exposure rate of 7 mR/h from the exposed sludge, which was calculated from the maximum measured concentrations in sludge from the four raffinate pits combined and an assumption that the individual would spend all of his or her time at the raffinate pits. The corresponding risk to the future recreational visitor would be 9×10^{-3}. Both of these estimates exceed the upper limit of EPA's target range.

5.2.2.2 Chemical Risks and Hazard Indexes

The chemical carcinogenic risks associated with ingesting surface water from the raffinate pits are 2×10^{-6} for the trespasser and 2×10^{-5} for the future recreational visitor (Table 5.6). Of the two carcinogens of concern detected in the pits — arsenic and beryllium — arsenic accounts for about 85% of the risk. Risks associated with incidental ingestion of sludge are 7×10^{-6} for the trespasser and 8×10^{-5} for the recreational visitor (Table 5.6). Arsenic is also the primary contributor to this risk (90%).

The estimated hazard indexes from ingesting raffinate pit surface water are 0.5 for the trespasser and slightly above 1 for the recreational visitor (Table 5.7). Nitrate accounts for about 50% of these hazard indexes; and uranium, antimony, and molybdenum each contribute from 10 to 15%. The hazard indexes from incidental ingestion of sludge are 0.2 for the trespasser and 0.8 for the recreational visitor (Table 5.7). The primary contributors to these estimates are arsenic (40%), uranium (20%), vanadium (15%), and thallium (10%).

The EPA suggests that if the hazard index exceeds 1, as for the recreational visitor from combined exposures to surface water and sludge at the raffinate pits, an analysis should be conducted to segregate the effect and mechanism of action of the primary contributors. The results of this analysis indicate that the segregated hazard index does not exceed 1 for the recreational visitor from combined exposures at the raffinate pits.

5.2.3 Air

Exposure to sitewide air was estimated for the worker, trespasser, and future recreational visitor. The risks associated with these exposures were determined from (1) the concentration of radon-222 and its short-lived decay products estimated from the sitewide UL_{95} concentration for radium-226 in surface soil and (2) the concentration of airborne contaminants generated from soil in the southern sector of the site. A location-specific analysis was also conducted to assess inhalation risks; the exposures were determined from (1) the concentration of radon-222 and its short-lived decay products estimated from the measured or derived concentration for radium-226 in surface soil at each borehole and (2) airborne contaminant concentrations estimated from the soil concentrations at each sampling location, similar to the approach used for the location-specific analysis for soil. The rationale for this approach is described in Section 3.3.4.

5.2.3.1 Radiological Risks

The radiological risks associated with inhalation of airborne radioactive particulates are estimated to be 4×10^{-4} for the worker, 1×10^{-6} for the trespasser, and 5×10^{-5} for the future recreational visitor. These risks are almost exclusively from radon-222 decay products; the risk from inhalation of airborne particulates other than radon-222 decay products is less than 5% of the total (Table 5.8). The radiological risk to the worker exceeds the upper limit of EPA's target range by a factor of 4, whereas the radiological risks to the trespasser and future recreational visitor are within the target range. For comparison, applying similar exposure assumptions for a background level of radon-222, associated risks are 4×10^{-5} for the worker, 1×10^{-7} for the trespasser, and 5×10^{-6} for the recreational visitor.

Results of the location-specific analysis indicate that the maximum risks from inhalation are 2×10^{-2} for the worker, 4×10^{-5} for the trespasser, and 2×10^{-3} for the recreational visitor. Inhalation of radon-222 decay products accounts for more than 99% of the risks. The risk estimated for the trespasser is within or below EPA's target range at all locations. However, the risks for the recreational visitor and worker exceed the upper limit of the target range at 6 locations and more than 50 locations, respectively; these are locations at which high levels of radium-226 were either measured or derived from the radiological source term analysis.

For most locations, the estimated risks from radon-222 exposures are probably overestimated by at least a factor of 2 because the radiological source term analysis was used to derive radium-226 concentrations if measured values were not available (which was the case at more than 70% of the boreholes). The source term analysis is biased high because it was

developed from data for soil with markedly elevated contamination, so the results are conservative. For example, high radium values were assigned to boreholes with high uranium levels for the location-specific assessment, but external gamma measurements indicate that these high radium concentrations are probably not present. Under the same scenario assumptions, exposures to radon-222 at most on-site locations would be expected to result in risks similar to those estimated for the background location.

5.2.3.2 Chemical Risks and Hazard Indexes

The chemical carcinogenic risks associated with inhalation of airborne particulates for the sitewide analysis are less than 1×10^{-6} for the worker, trespasser, and future recreational visitor (Table 5.9). These risk estimates were based on the estimated concentration of airborne particulates generated from contaminated soil in the southern portion of the site. The risks estimated for the location-specific analysis are also less than 1×10^{-6} for these receptors. The on-site risks are similar to those estimated for background locations under the same scenario assumptions. These findings are not unexpected because the level of contamination in surface soil is generally low and the potential for fugitive dust is limited under baseline conditions.

The estimated hazard indexes from inhalation of airborne particulates are similarly low: less than 0.2 for the worker, trespasser, and future recreational visitor for the sitewide analysis (Table 5.10); and less than 0.1 for the trespasser and recreational visitor for the location-specific analysis (data not presented). For the worker, the hazard index ranges from 0.1 to 0.3 at most locations (data not presented) and does not exceed 0.8 at any location. Chromium III contributes more than 80% to the estimated inhalation hazard indexes.

5.2.4 Game

Game ingestion exposures were assessed for a sportsman who successfully hunts the maximum take for deer, turkey, duck, and goose in the area. The duck and goose were assumed to obtain their entire diet from the site, and the deer and turkey were assumed to obtain one-third of their diet from the site. Risks associated with game ingestion were modeled with UL_{95} values for soil (deer and turkey), the average of detected concentrations for raffinate pit surface water (duck and goose), and uptake and bioconcentration factors determined from the literature. The limited radiological measurements available for the deer, goose, and duck were also used (Section 3.3.5.2). The potential health impacts associated with multiple pathway scenarios that consider both game ingestion and exposures incurred while hunting are presented in Section 5.4.2.

5.2.4.1 Radiological Risks

The radiological risks estimated for game ingestion are shown in Table 5.11. The estimated risks for ingestion of all game animals are within or less than EPA's target risk range. The results of the modeling indicate that the major contributors to risk from ingestion of deer

and turkey are thorium-232, radium-226, radium-228 and uranium-238; the major contributor for waterfowl is uranium-238. Similar results were shown for risks calculated from measured concentrations. However, these data are extremely limited and simply represent a one-time measurement of radionuclide concentrations in the flesh of a few individual game animals. In addition, control samples were not available for deer or waterfowl from which to establish expected background values.

Assuming that an avid sportsman in the area successfully hunts the maximum takes for all four species, the combined radiological risk (modeled) for game ingestion is about 8×10^{-7}. Ingestion of waterfowl accounts for most of this risk, which in part reflects the higher radionuclide concentrations in the raffinate pits. The uncertainty associated with these estimates is considerable, as discussed in Section 5.6.2.1.

5.2.4.2 Chemical Risks and Hazard Indexes

The estimated carcinogenic risks attributable to the site from game ingestion are 2×10^{-8} for deer, 2×10^{-8} for turkey, 2×10^{-9} for duck, and 1×10^{-9} for goose (Table 5.12). Arsenic is the dominant contributor. For the avid sportsman, the combined chemical carcinogenic risk (modeled) for game ingestion is 4×10^{-8}, primarily from the deer and turkey. For comparison, the same assumptions were applied to model game ingestion for the off-site background location; the estimated background risk is comparable, and arsenic is again the main contributor.

The estimated hazard indexes attributable to the site from ingesting deer, turkey, duck, and goose are 0.01, 0.004, 0.0004, and 0.0003, respectively (Table 5.13). These values are well below the level of concern. Comparative estimates for the off-site location are similar.

5.2.5 Building Air and Residues

The routes of exposure to contaminated media within the chemical plant buildings quantified in this assessment are (1) external gamma irradiation; (2) inhalation of radon, airborne particulates, and asbestos; and (3) incidental ingestion of residues and dust. Under current site conditions, the potential receptor is a trespasser. Under future land-use conditions, the receptor is a recreational visitor. Risks associated with exposures of these receptors in the buildings were assessed on the basis of values from the most highly contaminated buildings (Section 3.3.6).

5.2.5.1 Radiological Risks

The radiological risks for building air and residues were estimated by assuming intrusion into Building 403. The risks associated with exposure to radioactive contaminants in this building are 1×10^{-4} for the trespasser and 1×10^{-3} for the future recreational visitor (Table 5.14). The main contributors to this risk are incidental ingestion of radioactively contaminated residues (principally thorium-232 and uranium-238) and inhalation of radon-220 and its short-lived decay products. Other contributors to the risk are inhalation of radioactively contaminated airborne particulates (other than radon-220 decay products) and external gamma

irradiation. The risk from inhalation of radon-222 and its decay products is less than 1% of the total. The overall radiological risk to the trespasser is at the upper limit of EPA's target range, whereas the risk to the future recreational visitor is 10 times higher than the upper end of the range.

5.2.5.2 Chemical Risks and Hazard Indexes

The highest risks and hazard indexes estimated for chemical exposures at the site are associated with the chemical plant buildings. The chemical carcinogenic risks for these exposures are 3×10^{-4} for the trespasser and 3×10^{-3} for the future recreational visitor, and the associated hazard indexes are 3 and 9, respectively (Table 5.15). These health effects result from incidental ingestion of PCB-contaminated residues. The carcinogenic risks from inhalation of asbestos are 2×10^{-6} for the trespasser and 2×10^{-5} for the future recreational visitor. The risks from inhalation exposure to PCBs could not be estimated because an inhalation slope factor is not available for this class of compounds. For both the trespasser and future recreational visitor, the chemical risks associated with exposures to PCBs exceed 1×10^{-4}, the upper end of EPA's target range. In addition, the hazard indexes exceed 1, indicating the potential for noncarcinogenic health effects from exposures inside the buildings.

5.2.6 Multiple Exposure Pathways

Chemical and radiological risks and chemical hazard indexes for the worker, trespasser, and future recreational visitor have been evaluated for the individual exposure media at each exposure point (e.g., surface water at the raffinate pits) and for each route of exposure (e.g., ingestion) in Sections 5.2.1 through 5.2.5. However, these receptors would probably be exposed to contaminants in several media at multiple on-site locations via multiple routes of exposure. For example, an individual at the raffinate pits would be exposed to air, water, and sludge. The overall carcinogenic risk (or hazard index) for the receptor was estimated by summing the risk (hazard index) associated with each individual pathway.

For this analysis, the reasonable maximum exposures estimated for the individual pathways were added together to estimate exposures across multiple pathways. This approach tends to overestimate potential health effects because it is unlikely that the conservative assumptions used to estimate reasonable maximum exposures for each pathway would continuously apply to the same individual for all exposures. For example, it is more likely that exposures over the 10 or 30 years evaluated for the specific receptors in this assessment would be defined by average values for the intake parameters rather than the 90th or 95th percentile values that have been applied. Therefore, the approach taken in this BA is considered to be conservative.

The first stage in the assessment of multiple pathways focused on combined exposures to soil and air, with contaminants in air assumed to be generated from those in soil. These represent the most likely exposures for the three receptors over the full exposure periods. Estimates for the soil-related radiological exposures — i.e., external gamma irradiation, incidental

ingestion, and inhalation of radon and contaminated particulates generated from soil — were combined to represent exposures associated with random access across the site for the trespasser and future recreational visitor. This combination also represents the sum of exposures for the worker, who is the maximally exposed receptor relative to soil-related exposures in this BA.

The combined sitewide radiological risks from incidental ingestion of soil, external gamma irradiation, and inhalation of airborne contaminants are estimated to be 5×10^{-4} for the worker, 2×10^{-6} for the trespasser, and 6×10^{-5} for the future recreational visitor (Tables 5.16, 5.17, and 5.18). Inhalation of radon-222 decay products accounts for about 60 to 80% of these combined values, incidental ingestion of contaminated soil contributes about 10%, and external gamma irradiation contributes about 5%. The risk for the worker exceeds the upper limit of the target range by a factor of 5, whereas the risks for the trespasser and recreational visitor are within this range. The combined radiological risks for the location-specific analysis for the worker, trespasser, and recreational visitor are shown in Figures 5.3, 5.4, and 5.5. For comparison, similar exposure assumptions were applied at a background location to determine the combined risks associated with natural levels of the various radionuclides; the estimated combined risks for the background locations are 1×10^{-4} for the worker, 4×10^{-7} for the trespasser, and 2×10^{-5} for the recreational visitor. The background radiological risks are about one-fourth of those attributable to soil-related exposures at the site.

The chemical carcinogenic risks and hazard indexes for the worker from soil-related, sitewide exposures are presented in Table 5.16. The carcinogenic risk for the combined pathways is 1×10^{-5}, and the total hazard index is 0.5. Risks for the trespasser and recreator are 2×10^{-7} and 2×10^{-6}, and the respective hazard indexes are much less than 1 (Tables 5.19, 5.20, 5.21, and 5.22). Incidental ingestion of soil is the dominant contributor to these carcinogenic risks and hazard indexes.

The location-specific chemical risks from ingestion and inhalation combined for the worker and recreational visitor are shown in Figures 5.6 and 5.7. (Combined risks are not shown for the trespasser because, except for the concrete pad location with a combined risk of 1×10^{-6}, all other risks are below EPA's target range.) Because incidental ingestion dominates risks from soil-related exposures, the combined risks from these two pathways are essentially those resulting from soil ingestion.

By combining inhalation and ingestion results for the hazard index, three areas in addition to the five previously identified for ingestion alone have a total hazard index that exceeds 1 for the worker. The inhalation of chromium III associated with these three locations accounts for about 20 to 40% of the combined hazard index. None of the combined hazard indexes from soil-related exposures for the trespasser or the recreational visitor exceed 1 at any location. The eight locations where the worker's combined hazard index ranges from slightly above 1 to 4 are shown in Figure 5.8. The three additional areas include a second location in the South Dump, an area near the former sampling station for process waste streams (Building 106), and the former coal storage area. For the building and dump areas, ingestion of uranium contributes from more than 50 to 97% of the combined value, and ingestion of arsenic contributes much of the remainder. Uranium concentrations are relatively low at the coal

storage area, and incidental ingestion of arsenic accounts for more than 50% and inhalation of chromium III accounts for more than 40% of the combined hazard index. Because of the different exposure pathways and target organs for chromium (inhalation) and arsenic and uranium (ingestion), the segregated hazard index at these additional locations is less than 1.

The second stage of the multiple pathway assessment was expanded to include combined exposures to contaminants in the raffinate pits and buildings. A worker would not ingest surface water or sludge from the raffinate pits, and exposures to PCB residues and asbestos inside buildings would not apply because appropriate protective equipment would be used during indoor activities. Therefore, worker exposures were not estimated for those contaminated source areas. However, the trespasser and recreational visitor could visit the raffinate pits and buildings and incur related exposures. The various exposures across the expanded multiple pathways are presented individually for these receptors in Sections 5.2.6.1 and 5.2.6.2. Although exposures to soil and air for these receptors are discussed with the soil-related exposures, they are also included in this presentation for completeness.

5.2.6.1 Trespasser

Pathway-specific risks and hazard indexes for the trespasser were estimated for exposures to each of the on-site source areas and media, i.e., surface soil (Section 5.2.1), raffinate pit surface water and sludge (Section 5.2.2), outdoor air (Section 5.2.3), and building air and residues (Section 5.2.5). Although a trespasser might routinely visit only one on-site location, a more likely scenario would involve exposure to multiple locations and media over the 10-year exposure period. The actual risks would depend on the specific activities of the trespasser and the relative time spent at any one location but would be bounded by the risks estimated for the lowest and highest exposure points and media. In combining pathways, it was assumed that the total time of on-site exposure used for the pathway-specific analysis remains the same (i.e., the exposure time of 1 hour per visit, exposure frequency of 5 days per year, and exposure duration of 10 years), but that the time at any one location would be equal to or less than the time assumed for that analysis.

A matrix of exposure scenarios and associated radiological and chemical carcinogenic risks and chemical hazard indexes was developed for the trespasser to assess exposures via multiple pathways (Tables 5.17, 5.19, and 5.20). Scenarios T1 through T6 represent potential exposures to a single medium: air (T1), soil (T2), raffinate pit surface water (T3), raffinate pit sludge (T4), building air (T5), and building residues (T6). Scenarios T7 through T12 represent potential exposures at two or more locations and/or to two or more media:

- Scenario T7 — The total exposure occurs outdoors to air and sitewide soil.
- Scenario T8 — The total exposure occurs outdoors to air and to surface water and sludge at the raffinate pits.
- Scenario T9 — The total exposure occurs indoors to building air and residues.

- Scenario T10 — The total exposure occurs outdoors. The total inhalation exposure is to outdoor air; other exposures are to sitewide soil (50%) and raffinate pit surface water and sludge (50%).

- Scenario T11 — The total exposure is split between outdoor and indoor exposures. Inhalation exposures are to outdoor air (50%) and building air (50%); other exposures are to sitewide soil (50%) and building residues (50%).

- Scenario T12 — The total exposure is equally divided among the three areas, with 67% outdoors and 33% indoors. Inhalation exposures are to outdoor air (67%) and building air (33%); other exposures are to sitewide soil (33%), raffinate pit surface water and sludge (33%), and building residues (33%).

Other combinations of exposures and pathways can be derived from the total risks and hazard indexes presented for the individual pathways.

Radiological Risks. The radiological risks associated with the trespasser for the single and combined pathway scenarios are presented in Table 5.17. The risks for the combined pathways (Scenarios T7 through T12) range from 2×10^{-6} (Scenario T7, exposure to sitewide soil and outdoor air) to 2×10^{-4} (Scenario T8, exposure to outdoor air and to surface water and sludge at the raffinate pits). Scenario T7 represents the most likely exposure of a trespasser following implementation of the interim response actions recently approved for the chemical plant area, which include dismantling the buildings and removing the raffinate pit surface water. Assuming the pits would be stabilized to prevent releases of radon gas and exposure to sludge, the main exposure of a trespasser would be to site soil and air. The risks associated with Scenario T8, which exceed EPA's target range, are due almost exclusively to exposures at the raffinate pits. The contribution from outdoor air and sitewide soil is low for all combined pathways involving exposures at the raffinate pits or buildings.

Scenario T12 represents an equal distribution of exposures between the raffinate pits, buildings, and sitewide soil and air under baseline conditions. The estimated risk is 9×10^{-5}, and the major component is exposure to raffinate pit sludge. The relative pathway contributions to this risk are shown in Figure 5.9; also shown is the external gamma irradiation risk that would be associated with the raffinate pit sludge (6×10^{-5}) if the exposures were to occur when the water level in the pits was very low.

Chemical Risks and Hazard Indexes. The carcinogenic risks and hazard indexes for the trespasser for the single and combined pathway scenarios are presented in Tables 5.19 and 5.20, respectively. The carcinogenic risks for the combined pathway scenarios (T7 through T12) range from 2×10^{-7} for Scenario T7 (exposure to sitewide soil and outdoor air) to 3×10^{-4} for Scenario T9 (exposure to building air and residues). The lowest and highest hazard indexes of 0.004 and 3 are also for Scenarios T7 and T9 (T6), respectively. The hazard index is the same for

Scenario T9 (exposure to building air and residues) and Scenario T6 (exposure to building residues) because a hazard index could not be estimated for air in the absence of inhalation RfDs for the contaminants of concern in the buildings, asbestos and PCBs. The carcinogenic risk estimated for Scenario T7 (the most likely exposure scenario following implementation of the interim actions) is below the lower end of EPA's target range, and the hazard index is well below the level of concern.

Inhalation of outdoor air does not contribute significantly to the estimates of risk or hazard index for any of the combined scenarios. The higher risks and hazard indexes are associated with exposures to building residues and raffinate pit sludge. In general, the carcinogenic risks associated with any scenario that includes exposures to these two media are dominated by the related risks. This is also true for the hazard indexes, i.e., the highest hazard indexes are associated with exposures to building residues and raffinate pit sludge.

Scenario T12 is considered a representative scenario for baseline conditions because it includes all components. The carcinogenic risk estimated for this scenario is 1×10^{-4}, which is the upper limit of EPA's target range; the hazard index exceeds 1, which is the level of concern. Exposures at the buildings and raffinate pits are the major contributors to the estimated risk and hazard index. The relative pathway contribution to the overall risk and hazard index for Scenario T12 is shown in Figures 5.10 and 5.11.

5.2.6.2 Future Recreational Visitor

The exposure points and exposure media for the future recreational visitor in the area that now comprises the Weldon Spring site would be the same as those identified for the trespasser. Hence, the single and combined exposure scenarios are identical to those defined for the trespasser in Section 5.2.6.1, and they have been numbered R1 through R12 in parallel to those for the trespasser. However, because access to the site would no longer be restricted, the frequency and duration of exposure could potentially increase over those assumed for the trespasser; for this reason, the risks and hazard indexes are higher than those estimated for the trespasser. Potential health impacts associated with multiple exposure pathways that include fish and game ingestion are presented in Section 5.4.2.

Radiological Risks. The radiological risks estimated for the recreational visitor from single and multiple exposure pathways are presented in Table 5.18. The risks associated with multiple pathways range from 6×10^{-5} (Scenario R7, exposure to outdoor air and sitewide soil) to 3×10^{-3} (Scenario R8, exposure to outdoor air and to surface water and sludge at the raffinate pits).

Scenario R7 represents the most likely exposure of a recreational visitor following implementation of the recently approved interim actions. The estimated risk for this scenario is within EPA's target range. The risk results primarily from exposure to outdoor air (largely from inhalation of radon-222 decay products); exposure to site soil contributes 20% of the risk.

Except for Scenario R7, the risks associated with multiple exposure pathways for the recreational visitor exceed EPA's target range.

Scenario R12 represents an equal distribution of exposures among the pits, buildings, and sitewide soil and air under baseline conditions. The estimated risk for this scenario is 1×10^{-3}, which exceeds EPA's target range by a factor of 10. Although the risks to the recreational visitor are higher than those for the trespasser, the relative pathway contribution for Scenario R12 is similar to that shown for the trespasser in Figure 5.9. Exposure to raffinate pit sludge is the largest contributor to the radiological risk.

Chemical Risks and Hazard Indexes. The chemical carcinogenic risks and hazard indexes associated with the recreational visitor scenarios are shown in Tables 5.21 and 5.22, respectively. The carcinogenic risks for the combined pathway Scenarios R7 through R12 range from 2×10^{-6} for Scenario R7 (exposure to sitewide soil and outdoor air) to 3×10^{-3} for Scenario R9 (exposure to building air and residues). The lowest and highest hazard indexes of 0.02 and 9 are also for Scenarios R7 and R9 (represented by the estimate for R6), respectively. Scenario R7 represents the most likely exposure of a recreational visitor following implementation of the interim actions. The estimated risk is within EPA's target range, and the hazard index is well below the level of concern. Except for Scenarios R7 and R10 (exposure to sitewide soil, outdoor air, and raffinate pit surface water and sludge), the carcinogenic risks associated with the combined exposure pathways are at or above the upper limit of EPA's target range. The hazard indexes for these combined scenarios exceed 1, except for Scenario R7. Exposures at the buildings and raffinate pits are the major contributors to the estimated risks and hazard indexes.

Scenario R12 represents an equal distribution of exposures for the contaminated media under baseline conditions. The carcinogenic risk for this scenario is 1×10^{-3}, and the hazard index is 4. These estimates are dominated by exposures in the buildings. Although the estimates for the recreational visitor are higher than those for the trespasser, the relative pathway contribution to the total carcinogenic risk and hazard index for Scenario R12 is similar to that shown for the trespasser in Figures 5.10 and 5.11.

5.3 RISK ESTIMATES FOR OFF-SITE EXPOSURE POINTS

5.3.1 Surface Water and Sediment at Lakes 34, 35, and 36

The routes of exposure quantified in this assessment for surface water and sediment at Lakes 34, 35, and 36 are external gamma irradiation from sediment and shoreline soil, ingestion of surface water, and incidental ingestion of sediment. These exposures were evaluated for a recreational visitor and are expected to be similar under current and future conditions within the time frame of this analysis. The maximum measured contaminant concentrations in the three lakes combined were used to estimate contaminant intakes (Section 3.3.2). The indirect exposures to these media via fish ingestion are discussed in Section 5.3.2.

5.3.1.1 Radiological Risks

Uranium is the only radioactive contaminant of concern in the surface water and sediment of Lakes 34, 35, and 36. The radiological risks associated with ingestion of surface water and sediment from the lakes are estimated to be 4×10^{-6} and 3×10^{-6}, respectively (Table 5.23). Individuals visiting these lakes could also be exposed to external gamma radiation from sediment and shoreline soil. The incremental radiological risk estimated for this exposure is 8×10^{-7}. These risks are all within or below EPA's target range.

5.3.1.2 Chemical Risks and Hazard Indexes

The chemical carcinogenic risk for a recreational visitor from ingestion of surface water at Lakes 34, 35, and 36 is 4×10^{-6} (Table 5.24). Risks were estimated for the two oral carcinogens present in surface water, arsenic and 2,4-DNT; essentially all of the risk for the ingestion pathway is attributable to arsenic. The estimated hazard index from ingestion of surface water is 0.1 (Table 5.25), which is well below the level of concern.

Arsenic and PCBs are the chemical carcinogens detected in sediment at Lakes 34, 35, and 36. The carcinogenic risk associated with ingesting the lake sediment is 8×10^{-7}, and the hazard index is 0.02 (Table 5.26). The risk estimated for the lakes is less than EPA's target range, and the hazard index is well below 1.

5.3.2 Fish at Lakes 34, 35, and 36

Ingestion of fish from Lakes 34, 35, and 36 was evaluated for an individual (sportsman) who would fish in the Busch Wildlife Area and consume daily catch limits. These exposures are expected to be similar under current and future conditions. Estimated contaminant intakes were determined from information regarding use of the wildlife area lakes and from results of uranium analyses for fishcakes and fillets.

5.3.2.1 Radiological Risks

Uranium is the radioactive contaminant of concern in fish from Lakes 34, 35, and 36. The radiological risk estimated for ingesting fish from these lakes is 4×10^{-7} (Table 5.27). This estimate is only 4 times higher than the risk estimated for ingesting fish from the control lake, Lake 37, and is below EPA's target range.

5.3.2.2 Chemical Health Effects

The Missouri Department of Health and the ATSDR have prepared health assessments for chemical exposures associated with fish ingestion from the lakes. These studies did not indicate a significant adverse health effect from typical consumption (Section 2.4.5.2). In addition, the lakes receive chemical influents in surface runoff and other discharges from an

extensive drainage area, and only elevated levels of radionuclides are directly attributable to the site. Therefore, the assessment of the fish ingestion pathway in this BA focuses on the potential radiological risk.

5.3.3 Surface Water at Burgermeister Spring

The potential route of exposure to a recreational visitor at Burgermeister Spring is intentional ingestion of surface water. Exposures at this location are expected to be similar under current and future land-use conditions. The maximum contaminant concentrations measured in the spring were used to estimate intakes (Section 3.3.2).

5.3.3.1 Radiological Risks

The radiological risk from ingesting surface water at Burgermeister Spring is estimated to be 4×10^{-6} (Table 5.23). Uranium-238 contributes about 90% of this risk, and thorium-230 contributes most of the remainder. This risk is within EPA's target range.

5.3.3.2 Chemical Risks and Hazard Indexes

Nitroaromatic compounds are the only carcinogens of concern detected in surface water at Burgermeister Spring. The estimated chemical carcinogenic risk at this location is 9×10^{-7} (Table 5.24), and the hazard index is 0.04 (Table 5.25). These values are below EPA's target range and level of concern, respectively.

5.3.4 Surface Water and Sediment in the Southeast Drainage

The potential routes of exposure to surface water and sediment at the Southeast Drainage are external gamma irradiation, ingestion of surface water, and incidental ingestion of sediment. These exposures are expected to be similar under current and future conditions within the time frame of this analysis. The potential receptor is a recreational visitor. The risks associated with exposures to radioactive contaminants in surface water were estimated from half the maximum measured value to address concentration variability and intermittent flow, and those for the sediment were determined from both UL_{95} values and individual contaminant concentrations at each sampling location. The risks estimated with the UL_{95} values address exposures for an individual who might regularly hike along the drainage over the 30-year period. The location-specific analysis addresses the possibility that an individual might spend all of his or her time in one specific area over the 30 years. For chemical contaminants, the maximum measured concentrations at the Southeast Drainage were used to estimate intakes.

Because data currently available for the Southeast Drainage are limited, the screening-level risk estimates presented in this BA are considered preliminary. These estimates will be

used to focus additional characterization activities, the results of which will be incorporated into environmental documentation for this vicinity property to be prepared within the next several years.

5.3.4.1 Radiological Risks

The radiological risks from the combined ingestion of surface water and incidental ingestion of sediment in the Southeast Drainage are presented in Table 5.23. The radiological risk to a recreational visitor drinking water from the Southeast Drainage is estimated to be 1×10^{-5}. The incremental risk associated with external gamma irradiation, which was determined from measured exposure rates, is 7×10^{-6}. The risk associated with incidental ingestion of sediment is 2×10^{-4}, essentially all of which is attributable to thorium-230. This estimate is conservative because only five sediment samples were analyzed for thorium-230 and these samples were selected on the basis of elevated concentrations of other radionuclides. This risk estimate is a factor of 2 higher than the upper end of EPA's target range; all other risks estimated for the Southeast Drainage are within the target range.

Estimated risks for the location-specific analysis are presented in Figures 5.12 and 5.13 for surface water and sediment, respectively. At locations for which characterization data were available, the radiological risks associated with the ingestion of surface water are all within EPA's target range. The risks associated with sediment from incidental ingestion and external gamma irradiation exceed the upper limit of EPA's target range at several locations. In general, higher risks are associated with relatively flat areas and bends (deposition points) in the channel and areas close to springs where water reemerges from subterranean conduits. Such areas were also identified in the region where the drainage ditch meets the alluvial floodplain of the Missouri River. These results will be updated following completion of additional characterization activities and will be incorporated into forthcoming documents.

5.3.4.2 Chemical Risks and Hazard Indexes

Nitroaromatic compounds are the only carcinogens detected in surface water in the Southeast Drainage. The estimated risk associated with ingestion of these compounds is 1×10^{-6} (Table 5.24). The estimated hazard index is 0.2 (Table 5.25). Arsenic is the only chemical carcinogen detected in sediment in the drainage, and the associated risk from incidental ingestion is 1×10^{-6} (Table 5.26). The hazard index associated with incidental ingestion of this sediment is 0.02 (Table 5.26).

The chemical carcinogenic risks for exposures to surface water and sediment in the Southeast Drainage are at the lower end of EPA's target range. The hazard indexes for these exposures are well below 1. The actual exposure of an individual would probably be much smaller because very conservative values were used for contaminant concentrations in this preliminary assessment.

5.3.5 Soil at Vicinity Properties

The potential routes of exposure to soil at vicinity properties are external gamma irradiation and incidental ingestion of soil. These exposures were evaluated for a recreational visitor and are expected to be similar under current and future conditions within the time frame of this analysis. The arithmetic averages of the contaminant concentrations at each property were used to estimate intakes.

5.3.5.1 Radiological Risks

The radiological risks associated with soil at the vicinity properties are presented in Table 5.28. The risks to a recreational visitor at the vicinity properties range from 6×10^{-7} (A6) to 3×10^{-4} (B4), and the major contributor is generally external gamma irradiation. These estimates are considered to be conservative because the recreational visitor is assumed to spend all of his or her time at the individual vicinity property for the duration of the exposures. This scenario is unlikely because of the small size of the properties; a recreational visitor is much more likely to spend some time at other locations throughout the surrounding wildlife areas during the 30-year period. On the basis of the ratio of the affected area to the total wildlife area (about 0.1%), the risks posed by these properties would probably be within or below EPA's target range of 1×10^{-6} to 1×10^{-4}.

5.3.5.2 Chemical Risks

Chemical characterization data are not available for the soil vicinity properties because the characterization activities did not include an investigation of potential chemical contaminants. On the basis of historical information, radiological contamination is expected to be the primary risk contributor.

5.3.6 Multiple Exposure Pathways

In parallel to the analysis for on-site exposures, risks were assessed for off-site exposures of the recreational visitor to multiple exposure points, media, and routes of exposure. Potential exposures and risks were evaluated for three off-site exposure points: Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage. Potential health impacts associated with multiple pathway exposures that include fish and game ingestion as components are discussed in Section 5.4.2.

The total radiological risks to the off-site recreational visitor are presented in Table 5.29. The risks at these locations range from 4×10^{-6} at Burgermeister Spring to 2×10^{-4} at the Southeast Drainage, which exceeds the upper end of EPA's target range by a factor of 2. The radiological risk estimated for the Southeast Drainage is conservative because it is primarily from incidental ingestion of sediment contaminated with thorium-230, and only five samples that were biased high were evaluated for this radionuclide. Ingestion of surface water contributes less than 10% of the total radiological risk associated with this drainage.

The total chemical carcinogenic risks and hazard indexes associated with exposures to surface water and sediment at the off-site locations are also presented in Table 5.29. The risks at these locations fall within or below the target range, and the hazard index is well below the level of concern.

An individual living in the vicinity of the Weldon Spring site could visit all three of these off-site locations. For this individual, the total amount of time spent in the wildlife areas is not expected to increase over that assumed for estimating exposures at any one location (i.e., 4 hours per visit, 20 times per year, over 30 years), although the time spent at any one location would decrease. In this case, the radiological risks to an individual are bounded by the estimates for exposures at Burgermeister Spring (4×10^{-6}) and the Southeast Drainage (2×10^{-4}). The chemical carcinogenic risks are bounded by the estimates for exposures at Burgermeister Spring (9×10^{-7}) and the Busch Wildlife Area lakes (4×10^{-6}). The hazard indexes are bounded by the estimates for exposures at Burgermeister Spring (0.04) and the Southeast Drainage (0.2).

An individual could also contact the soil vicinity properties in combination with other exposures. However, because these properties are widely scattered and relatively small, they were considered less likely to contribute to multiple exposures than the lakes, Burgermeister Spring, and the Southeast Drainage. If they were a part of combined exposures, the radiological risks would be bounded by the estimates for these properties (6×10^{-7} to 3×10^{-4}).

5.4 MULTIPLE EXPOSURE PATHWAYS FOR ON-SITE AND OFF-SITE LOCATIONS

5.4.1 Trespasser and Recreational Visitor

The same individual could be exposed to both on-site and off-site exposure points. Under baseline conditions, exposures at the raffinate pits and chemical plant buildings generally dominate the carcinogenic risks for scenarios addressing both on-site and off-site exposures. That is, the radiological and chemical carcinogenic risks associated with these two locations are significantly higher than the risks estimated for other on-site and off-site exposure points (Tables 5.18, 5.21, and 5.29).

Following implementation of response actions that have been approved for the surface water in the raffinate pits and for the buildings, on-site radiological risks to the future recreational visitor would be dominated by exposures to the raffinate pit sludge. Except for this medium, the radiological risks for sitewide exposures would be less than those conservatively estimated for the Southeast Drainage in this preliminary assessment. Hence, assuming the same exposure time, frequency, and duration for the combined on-site and off-site exposures (i.e., 4 hours per visit, 20 times per year, over 30 years), the overall radiological risk to the future recreational visitor would range from 6×10^{-5} (assuming that the total extent of exposure occurs on-site and that raffinate pit sludge has been controlled so no related exposures occur) to 2×10^{-4} (assuming that all of the exposures over the 30 years occur at the Southeast Drainage).

Except for exposures to the raffinate pit sludge, the on-site chemical risks to the future recreational visitor are comparable to those at the off-site locations — i.e., risks to sitewide soil and air are 2×10^{-6} compared with 4×10^{-6} for combined exposures at Lakes 34, 35, and 36; 9×10^{-7} at Burgermeister Spring; and 2×10^{-6} at the Southeast Drainage. The actual risks would depend on the specific activities of the individual and the relative time spent at any one location but would be bounded by the risks estimated for the lowest and highest exposure points. Hence, assuming the same exposure time, frequency, and duration for the combined on-site and off-site exposures (i.e., 4 hours per visit, 20 times per year, over 30 years), the overall risks to the future recreational visitor would range from 9×10^{-7} to 4×10^{-6}. These values are within or below EPA's target range, and they assume that the raffinate pit sludge has been controlled and there are no related exposures.

The results of the assessment of potential noncarcinogenic health effects for combined on-site and off-site exposures are similar to those for the carcinogenic risk assessment. That is, under baseline conditions, the hazard indexes associated with the raffinate pits and buildings tend to dominate the hazard indexes for scenarios addressing both on-site and off-site exposure points. Following implementation of the interim response actions for the raffinate pits and buildings, the on-site hazard indexes for the future recreational visitor from most likely exposures (i.e., to sitewide soil and air) are less than those estimated for the off-site locations, which are well below 1 (Tables 5.22 and 5.29). The hazard index from exposures to sitewide soil and air is 0.02 compared with 0.1 for exposures at Lakes 34, 35, and 36; 0.04 at Burgermeister Spring; and 0.2 at the Southeast Drainage. Hence, the overall hazard index for the future recreational visitor ranges from 0.02 to 0.2, which is below the level of concern.

5.4.2 Sportsman

Pathway-specific risks and hazard indexes were estimated for the fish and game ingestion pathways in Sections 5.2.4 and 5.3.2. Fish were assumed to be taken from Lakes 34, 35, and 36, and game was assumed to inhabit the area currently comprising the site. While fishing or hunting at these areas, an individual would also incur exposures similar to those estimated for the recreational visitor at the lakes (exposure to surface water and sediment/shoreline soil) and the future recreational visitor to the site (exposure to air and soil). The reasonable maximum exposure for an avid sportsman is estimated for an individual who both fishes and hunts in the area.

Potential exposures at these lakes would be similar to those of the recreational visitor, i.e., ingestion of surface water, incidental ingestion of sediment/shoreline soil, and external gamma irradiation from sediment/shoreline soil. The exposure of the sportsman was estimated by multiplying the risks to the recreational visitor by an adjustment factor of 7/20 to account for the difference in the total extent of exposure (i.e., the fisherman was assumed to visit the lakes 7 days per year whereas the recreational visitor was assumed to visit the lakes 20 days per year). The risks associated with fish ingestion are the same as those estimated for the fisherman in Section 5.3.2. The overall radiological and chemical risks associated with the various exposures from fishing at the lakes and ingesting the catch are 3×10^{-6} and 2×10^{-6}, respectively, and the

overall hazard index is 0.05. The risks are at the lower end of EPA's target range, and the hazard index is well below the level of concern.

To estimate the risks associated with hunting, it was assumed that the sportsman would hunt at the site just as often as elsewhere in the surrounding wildlife area; this assumption is based in part on the type of vegetation that would be expected at the site in the future if the area were no longer maintained. The site is not expected to be any more or less habitable for game animals than the surrounding area, so neither wildlife nor hunters would be expected to preferentially occupy or avoid the site. Therefore, the ratio of the site area to the surrounding wildlife area (3%) was considered in determining how often a hunter might be present on-site. A conservative value of 10% was used for this analysis (which also addresses the Southeast Drainage area).

The full hunting season for the various game animals in the area extends from June 1 to December 31, with an additional 2-week period designated between January 1 and May 31; hence, the hunting season totals about 224 days per year. The exposure time was estimated by assuming that (1) an avid sportsman might hunt for an average of 4 hours per day, 20 days per month, over 7.5 months — totaling 150 days per year — and (2) 10% of this time would be spent on-site. From these assumptions, it was estimated that 4 hours per day would be spent on-site for 15 days per year (Meyer 1990). Except for the exposure frequency of 15 days per year compared with 20 days per year, these are the same assumptions used for the future on-site recreational visitor. Therefore, the exposure of the sportsman on-site was estimated by multiplying the combined risks to the recreational visitor from exposures to sitewide soil and air by an adjustment factor of 15/20 to account for the difference in total extent of exposure.

To estimate the exposure from game ingestion, it was assumed that (1) the individual hunts deer, turkey, duck, and goose; (2) 10% of the allowable take is represented by animals that have inhabited the site; and (3) an individual shares the edible portion of this take with one other person. On the basis of these assumptions, the radiological and chemical carcinogenic risks associated with game ingestion would be the summation of the risks calculated from the model for the individual species, i.e., 8×10^{-7} and 4×10^{-8}, respectively (Tables 5.11 and 5.12). Radiological data are available for only a few specimens, not including turkey, and the total risk estimated from these data is comparable. The hazard index is less than 1.

If the various pathways are combined — i.e., exposures to site air and soil while hunting, exposures to lake surface water and sediment/shoreline soil while fishing, and ingestion of fish and game — the overall radiological and chemical risks to the avid sportsman are estimated to be 5×10^{-5} and 3×10^{-6}, respectively. The overall hazard index is 0.08. These risks are within EPA's target range, and the hazard index is below the level of concern.

The sportsman could also enter the chemical plant buildings and/or incur exposures at the raffinate pits. However, such exposures have not been explicitly included in this multiple pathway assessment because the associated health effects dominate any scenario for which these exposures are components. The risks to the sportsman if these locations were visited would be similar to those estimated in Section 5.2.6.2 for the future on-site recreational visitor, Scenario R12.

5.5 ASSESSMENT OF THE DERMAL PATHWAY

Exposures to contaminated media such as soil, sediment, and surface water can occur through three potential routes: inhalation, ingestion, and dermal absorption. External gamma irradiation is an additional route for radioactively contaminated media. Dermal exposures have not been quantified in the health risk assessment to this point because of limitations in the methodology for evaluating such exposures. These limitations arise from the unavailability of a generally accepted approach and the lack of standard values for key input parameters, which results in considerable uncertainty in associated intake and risk estimates.

To assess dermal exposures at contaminated sites, the EPA currently recommends (1) addressing exposures to soil qualitatively, with limited quantification for the few contaminants for which absorption factors have been developed, and (2) quantifying exposures to water with permeability coefficients that are available for certain contaminants and default or derived values for other contaminants (Schaum 1991). This approach was followed for both the radioactive and chemical contaminants of concern.

A quantitative assessment of dermal exposures to contaminated media at the Weldon Spring site is presented in Sections 5.5.1 and 5.5.2 to provide some estimation of potential impacts. The complete exposure pathways and receptors for dermal contact, under both current and future land-use conditions, are identified in Section 3.2.4.

5.5.1 Health Assessment

Dermal exposures to soil, sludge, sediment, surface water, and building residues were assessed for potential receptors under current and future land-use conditions. Under current conditions, the receptors that could come in direct contact with site soil are the maintenance worker and trespasser; the trespasser could also contact raffinate pit sludge and building residues. Under future conditions, a recreational visitor to the site could be in direct contact with site soil, raffinate pit sludge, and building residues. Under both current and future conditions, an off-site recreational visitor could contact sediment and shoreline soil at various off-site locations. To assess potential dermal exposures to surface water, a swimming scenario was also evaluated, even though the probability of this scenario is expected to be low (Section 3.2.4).

The exposure point concentrations used to estimate intakes for these dermal exposures are the same as those used to estimate intakes for ingestion of on-site soil, surface water, sludge, and building residues (Tables 3.8, 3.9, 3.11, and 3.16 and Section 3.4.8.2) and ingestion of off-site surface water and sediment (Tables 3.10, 3.12, and 3.17).

Intakes from dermal exposure depend on the amount of contaminant actually absorbed from the contacted medium. For soil, the EPA recommends quantifying dermal exposure for a particular contaminant only if there is some experimental basis for estimating the amount of the contaminant that is absorbed; this same approach was also used for sediment, sludge, and residues/dust. For the contaminants at the Weldon Spring site, dermal absorption fractions are

available only for cadmium (0.001) and PCBs (0.01 to 0.1) (Schaum 1991). The value of 0.1 was conservatively used for PCBs in this assessment.

The intakes from dermal contact with contaminated soil, sediment, and sludge were calculated as follows:

$$AD_i = \frac{C_{si} \times SA \times AF \times ABS_i \times CF_2 \times EF \times ED}{BW \times AT} \tag{5.1}$$

where:

AD_i = absorbed dose of contaminant i from dermal exposure (mg/kg-d);

C_{si} = soil concentration of contaminant i (mg/kg);

SA = skin surface area (cm^2/d or cm^2/event);

AF = soil-to-skin adherence factor (mg/cm^2);

ABS_i = fraction of contaminant i absorbed;

CF_2 = conversion factor (10^{-6} kg/mg);

EF = exposure frequency (d/yr or events/yr);

ED = exposure duration (yr);

BW = average body weight over the exposure period (kg); and

AT = averaging time (d).

For these calculations, the SA was estimated to be 2,600, 3,200, and 4,200 cm^2 per day or event for the worker, trespasser, and recreational visitor, respectively. An adherence factor of 0.4 mg/cm^2 was used for each receptor. These values were derived from available data, and site-specific considerations were incorporated for factors such as climate, type of clothing that would be worn, and activities during the exposure event (Lepow et al. 1975; Harger 1979; Roels et al. 1980; Que Hee et al. 1985; EPA 1989b, 1989j; Driver et al. 1989). The values for the other variables were the same as those used to estimate intakes from inhalation and ingestion for the individual receptors (Table 3.3).

The approach used to estimate dermal intake from exposure to PCB-contaminated residues in the buildings depended on the type of sample, i.e., bulk or surface swipe (Section 3.3.6.2). Equation 5.1 was used to estimate intakes for PCB concentrations in bulk

samples (reported as mg/kg), and Equation 5.2 was used for swipe samples (reported as mg/m^2):

$$AD_i = \frac{C_{bi} \times CF_5 \times SA \times FR \times ABS_i \times EF \times ED}{BW \times AT} \tag{5.2}$$

where:

C_{bi} = concentration of contaminant i in building residue (mg/m^2);

CF_5 = conversion factor (10^{-4} m^2/cm^2); and

FR = fraction of contaminant removed during contact (0.01).

The weighted average of the two data sets was used to estimate residue intake.

The EPA recommends a different methodology to estimate intakes from dermal contact with surface water; in this calculation, an exposure flux is incorporated to quantify the amount of contaminant that could be absorbed per unit area of skin per unit time. Chemical-specific dermal permeability coefficients are used in the intake equation to estimate the distance traveled through the exposed skin during swimming exposures. For most inorganic contaminants, default values are used because experimental data are not available from which to determine the individual permeability coefficients. Of the inorganic contaminants evaluated for the swimmer in this assessment, permeability coefficients (in cm/h) are available from the EPA only for cobalt (4×10^{-3}), lead (4×10^{-6}), silver (6×10^{-4}), and zinc (6×10^{-4}); for the remaining inorganic contaminants, including radionuclides, the recommended default value of 1×10^{-3} was used (Schaum 1991).

For organic contaminants in water, the EPA currently recommends that a permeability coefficient be derived from a correlation with the octanol-water partition coefficient and the molecular weight of a compound (Schaum 1991). Only one organic compound, 2,4-DNT, was detected in a single surface water sample from the off-site lakes (none were detected in the raffinate pit water). A permeability coefficient of 4×10^{-3} cm/h was derived for this contaminant.

The doses associated with intake of radionuclides resulting from dermal contact with contaminated surface water were calculated as follows:

$$AD_i = R_{wi} \times SA \times PC_i \times CF_6 \times ET \times EF \times ED \times DCF_i \tag{5.3}$$

where:

R_{wi} = concentration of radionuclide i in water (pCi/L);

PC_i = dermal permeability coefficient for contaminant i (cm/h);

CF_6 = conversion factor (10^{-3} L/cm^3);

ET = exposure time (h/d or h/event); and

DCF_i = dermal absorption dose conversion factor for radionuclide i (mrem/pCi).

The unit for AD_i in this equation is mrem, and the unit for SA is cm^2.

The intakes of chemicals from dermal contact with contaminated surface water were calculated as follows:

$$AD_i = \frac{C_{wi} \times SA \times PC_i \times CF_6 \times ET \times EF \times ED}{BW \times AT} \tag{5.4}$$

where C_{wi} = concentration of contaminant i in water (mg/L). The unit for AD_i in this equation is mg/kg-d.

For both the raffinate pits and off-site lakes, it was assumed that a youth would swim 1 time per year for 1 hour over 10 years. A value of 15,000 cm^2 was determined for the SA on the basis of information provided by the EPA (1989b).

The risks and hazard indexes for dermal exposures were estimated with the same general approach as described for inhalation and ingestion exposures (Section 5.1.2). However, for those two pathways, the health effects were determined on the basis of an administered dose — which is the mass of a contaminant in contact with the exchange boundary (e.g., the lung or gastrointestinal tract) but does not reflect the amount that penetrates that boundary. For dermal exposures, the health effect was calculated from the absorbed dose — which is the amount of a contaminant that penetrates the exchange boundary (in this case the skin).

To address this difference, the toxicity values used to assess dermal exposures to chemical contaminants were derived from available RfDs and oral slope factors (Sections 4.2 and 4.3). For these derivations, the dermal slope factor was calculated by dividing the oral slope factor by the oral absorption fraction, and the dermal reference dose was calculated by multiplying the oral reference dose by the oral absorption fraction. A similar approach was used for radionuclides to assess exposure to surface water while swimming. Dermal absorption DCFs were calculated by dividing the ingestion DCF by f_1, the fractional amount of the ingested radionuclide that reaches body fluids (Table 4.1). If more than one dermal DCF could be calculated (i.e., for uranium), the larger factor was used.

The risks and hazard indexes estimated for the different receptors from dermal exposures are presented in Tables 5.30 and 5.31. Additional pathways for a swimmer would be exposure to external gamma irradiation, incidental ingestion of surface water and sludge or sediment, and inhalation of sitewide air for a swimmer at the raffinate pits. Therefore, the related risks and hazard indexes from these other exposures are also presented in Table 5.31 to provide an indication of combined health effects. For these calculations, it was assumed that the

swimmer would incidentally ingest 50 mL of water while swimming (EPA 1989j). The other intake assumptions used to assess combined exposures for the swimmer are the same as those used for the current trespasser (Table 3.3), except that the exposure frequency for the swimmer is once per year rather than 5 times per year.

5.5.2 Interpretation of Results

Risks and hazard indexes could only be quantified for dermal exposure to soil, sediment, and sludge for two site contaminants: cadmium and PCBs. Although slightly more information is available to quantify dermal exposure to water, permeability coefficients are available for only a few site contaminants; therefore, either default values or derived values had to be used to assess all others. For these reasons, the quantification presented in this assessment is very limited, and the potential health effects from dermal contact with these media could not be adequately assessed.

Although significant uncertainties are associated with the methodology for quantifying dermal exposures to soil, they are not discussed in detail here because the unavailability of absorption fractions is the critical limitation for this assessment. Nevertheless, other factors that contribute to the uncertainty for this pathway are identified in the following discussion.

Dermal absorption from soil is a function of both the fraction of adhered material that actually contributes to absorption across the skin surface and the contact time with that surface. Neither of these factors has been explicitly incorporated into the current intake equation for dermal absorption. That is, under current EPA methodology, the entire mass of material is assumed to be in intimate contact with the skin surface so that all contamination in that mass can be transported across the boundary. Also, in studies from which the absorption values were derived, the contact time with contaminated media was longer than the actual contact time would generally be for the exposures evaluated in this BA. Each of these factors tends to overestimate potential exposures.

Dermal absorption from water is also a function of contact time and of differential permeability of the skin surface for a particular contaminant. Although time has been accounted for in the intake equation for dermal absorption from water, few contaminant-specific permeability coefficients are available. Therefore, default values were used for the inorganic contaminants, including radionuclides, and contaminant-specific values were derived for the organic compounds. In addition to initial contaminant concentrations, water-specific conditions such as pH and algal activity will produce variations in contaminant bioavailability that will not be reflected in the default values or the values derived from single-value solubility parameters; such variations include the speciation and complexation of metals and the persistence and lipophilicity of organic compounds. Thus, the use of such values introduces considerable uncertainty into the exposure assessment and may underestimate or overestimate the actual amount of contaminant absorbed.

Considerable uncertainty is associated with the methodology applied to derive dermal slope factors, RfDs, and DCFs from those available for the oral route because of the inherent

assumptions relative to the contaminant distribution within the body, target organ, and mechanism of action. In certain cases, the dermal toxicity values calculated with this EPA-recommended approach differ by orders of magnitude from the values from which they were derived.

The results of the assessment for the swimmer indicate that the risks estimated for dermal contact with water are greater than those for the ingestion pathway for both radioactive and chemical contaminants. This outcome is unexpected because experience with exposures to the radionuclides and metals considered in this assessment has not identified the dermal route as an important route of exposure. In addition, EPA risk assessment guidance indicates that the dermal route is not considered an important route of exposure for most radionuclides (EPA 1989j). This suggests that the intakes and risks associated with the dermal route of exposure are probably overestimated because of the use of default values for the permeability coefficients and toxicity values derived from those for the oral route.

In general, the inability to characterize the magnitude of potential health effects from dermal exposures is not expected to have a substantial impact on cleanup decisions for the site. Decisions have already been made to remove the contaminated surface water and buildings. For soil locations at which contaminant concentrations are statistically elevated above background, risks or hazard indexes from ingestion exposures suggest that cleanup should be considered even without other input. To address the lack of quantitative information for the dermal pathway, the EPA suggests that results may in some cases be considered similar to those for ingestion exposures. If this were assumed to be true for site contaminants, even doubling estimated risks or hazard indexes would not typically change the order-of-magnitude estimate on which cleanup decisions are usually made. Furthermore, the cumulative effect of the conservative assumptions used to quantify exposures for the various scenarios (discussed in Section 5.6) would tend to offset an underestimation of relative risks from dermal exposures. This issue will be qualitatively addressed in the development of cleanup criteria for the site.

5.6 UNCERTAINTY IN THE ASSESSMENT PROCESS

The evaluation of risks to human health presented in this BA was, by necessity, based on a number of assumptions. In addition, many uncertainties are inherent to the risk assessment process. To address the potential impact of these uncertainties on the results contained herein, the rationale for the major assumptions used in this assessment and associated uncertainties are discussed further in Sections 5.6.1 through 5.6.4. Data gaps are also considered in this discussion.

5.6.1 Contaminants of Concern

The identification of contaminants of concern for a human health evaluation relies on both assessment of information from site characterization activities and application of a selection

process. The uncertainties associated with these two components are addressed in Sections 5.6.1.1 and 5.6.1.2.

5.6.1.1 Characterization Data

Considerable data have been collected for the Weldon Spring site under DOE's ongoing environmental monitoring program and the extensive site characterization effort. Except for groundwater and air, on-site media have generally been adequately characterized for purposes of this BA. Because of data limitations, groundwater is being addressed as a separate operable unit for the project and exposure to groundwater is not quantified in this BA. (A preliminary assessment for the groundwater ingestion pathway is presented in Appendix E of the FS [DOE 1992a].) For air, a number of factors support the assumption that the air pathway is relatively insignificant at the site under baseline conditions, including the available monitoring data for radon and radioactive particulates at the site perimeter (which are at background levels), surface features at the chemical plant (i.e., conditions that limit wind erosion), and site-specific meteorological data (measured wind speed). Hence, the uncertainty related to the unavailability of comprehensive data for air does not significantly impact the analyses in this BA, except for the overestimation of risks from radon exposures. These risks reflect the use of conservative assumptions to address the unavailability of comprehensive, location-specific data for radon or radium, as discussed below.

Site soil, which is the primary focus of this assessment relative to forthcoming cleanup decisions, is considered to be well characterized. The radiological risks for site soil were based on reported radionuclide concentrations, as provided in the RI report (DOE 1992c). Analyses were conducted for selected radionuclides of the uranium-238 and thorium-232 decay series during site characterization activities; this approach is consistent with the past use of the site as a processing facility for uranium and thorium ore concentrates and the half-lives of the various radionuclides in these two decay series. Although certain isotopes of uranium, radium, and thorium were measured in soil, not all radionuclides were assayed from each sample. In addition, samples were not analyzed for other principal radionuclides in the uranium-238 and thorium-232 decay series, and very few radionuclides in the uranium-235 decay series were assayed in any of the media. To address this data gap and the related uncertainty in risk estimates, a radiological source term analysis was conducted to identify the presence of and relationships among the various radionuclides at the site (Table 2.3). This information was used to estimate the concentrations of principal radionuclides for each borehole, which were incorporated into the risk assessment.

Measured exposure rates were used to estimate risks from external gamma irradiation, which is one of the major contributors to total risks from soil exposures, so little uncertainty is associated with those estimates. These exposure rate measurements were also used to assess the appropriateness of the estimated radionuclide concentrations, and a comparison indicated that the calculated values were conservative. (This result was expected because the source term analysis was developed from contamination data known to be biased high, and the results were applied across the site, including at nonsource areas.) Thus, although some uncertainty is associated with estimating concentrations for certain radionuclides, use of measured data for

much of the input and conservative values for the remaining input is expected to result in overestimates rather than underestimates of related risks. In particular, the radiological risk attributable to radon-222 decay products is probably high by at least a factor of 2 because of an artifact in the sampling strategy (i.e., radium-226 was assayed in samples collected at locations with elevated gamma exposure rates) combined with the use of the source term data to estimate radium-226 concentrations for more than 70% of site boreholes for the location-specific analysis.

One limitation for soil data that could potentially affect the chemical risk estimates is the scarcity of information regarding the relative occurrences of the trivalent and hexavalent forms of chromium at the site. On the basis of site history, the few measurements that are available, and data from the literature on the relative occurrence of these two forms in the environment, the predominant form of this metal (90%) was assumed to be chromium III. The estimated risks and hazard quotients associated with chromium exposure could be either overestimated or underestimated as a result of this assumption because the oral RfDs for the two forms differ considerably and the inhalation slope factor is appropriate only for chromium VI. However, even assuming that 100% of the chromium is the hexavalent form, exposure to this metal would not contribute to health effects greater than the target levels identified by the EPA, i.e., chromium VI is not a primary contributor to any significant exposure pathway at the site. Hence, the effect of this uncertainty on the overall risk estimates is expected to be low.

Another limitation for chemicals in soil results from inconsistent analysis and reporting of PCB data by the laboratory. To address this issue, 50 check samples were analyzed to support the site characterization and assessment effort. The check sample program was conducted to define the areal extent of PCB contamination, e.g., at locations formerly occupied by electrical transformers (MK-Ferguson Company and Jacobs Engineering Group 1992a). Results confirmed the previous understanding of PCB contamination at sampled areas.

The biased sampling approach used to collect data for PCBs and certain other contaminants (including PAHs and nitroaromatic compounds) is an additional factor that affects the health assessment results. The biased program was appropriate for assessing organic contamination at the site, but this approach tends to overestimate the sitewide UL_{95} values. (For example, the detection frequency for PCBs might be lower than that shown in Table 2.5 by a factor of 2 or more if the PCB analysis had included more than 300 unbiased samples, as was typical for many other analytes.) This does not constitute a data gap; rather it is identified to put in perspective the high bias that is likely to result from such a sampling program.

Finally, although comprehensive data are not available for all contaminants in on-site surface water and buildings, decisions have already been made for these media. These decisions were based on data available for the key contaminants (notably high concentrations of radionuclides) and the potential threats to worker health and safety associated with the buildings. Therefore, the associated uncertainty does not affect forthcoming remedial action decisions.

For off-site locations, both radiological and chemical data are limited for the Southeast Drainage and Lakes 34, 35, and 36; and chemical data are not available for the soil vicinity properties. Hence, the impacts quantified for these exposure points are more uncertain than

those for on-site media. However, this uncertainty is not expected to significantly affect decisions for the lakes and vicinity property soil because (1) the lakes are targeted for draining and sediment removal in the near future as part of a separate (state) management action and (2) the discrete soil locations are defined by their radioactive contamination, which would be the major contributor to related risks. Because of data limitations and other considerations for the Southeast Drainage, such as intermittent flow and nonsite sources, the screening-level estimates of health effects for this area are considered to be preliminary. Limitations that impact these estimates will be reduced when additional information becomes available from planned characterization activities, and an additional assessment will be presented in forthcoming documentation for the drainage within the next several years.

Considerable uncertainty is inherent in biotic studies because experimental conditions are difficult to control. In addition, the number of measurements for contaminant levels in fish and game are very limited, and diversity in sampling protocols and data presentation make compilation and comparison difficult. The health effects estimated for the food ingestion pathway are considered preliminary, and limitations in these estimates may be reduced when additional information becomes available from forthcoming biouptake studies. Additional assessment of this pathway will be presented in future documentation for the project.

5.6.1.2 Selection Process

The uncertainty that is inherent in the selection of contaminants of concern for a baseline assessment is associated with a number of factors. First, limitations in data relative to locations and analytes sampled and to analytical considerations (e.g., laboratory procedures) may affect the contaminants identified for a site. The uncertainty associated with the Weldon Spring site sampling data is considered to be low because the sampling plans that were implemented generally targeted appropriate areas and analytes on the basis of historical information, visual observations, and both phased and biased and unbiased characterization strategies. Uncertainty relative to sample analysis and data evaluation is also considered reasonable because an extensive, site-specific validation program has been applied to the characterization data.

Uncertainty is also a component of the contaminant selection screen. It is recognized that the estimated health effects could be higher if all compounds were included in the assessment. However, it is not practical to present the impact quantification for a large number of contaminants in this document. To address this uncertainty, the selection process is designed to screen out only those contaminants that contribute insignificantly to potential health effects. For this BA, the contaminants selected for evaluation represent principal radionuclides and chemical classes (metals, inorganic anions, nitroaromatic compounds, PAHs, PCBs, and asbestos) present at the site. The radiological source term analysis was conducted to determine the presence of principal radionuclides from the uranium-238, thorium-232, and uranium-235 decay series in site soil and raffinate pit sludge. Information from this analysis was used to identify the radionuclides of concern for this assessment, so related uncertainty is low. Other than common laboratory contaminants and essential nutrients, the only chemicals excluded from the analysis were those detected at very low frequencies and concentrations and not otherwise

expected to be present on the basis of historical information. Hence, the uncertainty associated with the screening step is considered to be low.

5.6.2 Exposure Assessment

An exposure assessment is constructed from a number of site-specific considerations, including exposure point concentrations, scenario assumptions and intake parameters, and primary exposure pathways. Uncertainties associated with these considerations are addressed in Sections 5.6.2.1, 5.6.2.2, and 5.6.2.3.

5.6.2.1 Exposure Point Concentrations

Factors that can contribute to uncertainty in exposure point concentrations include data availability, data heterogeneity, and the potential impact of fate and transport over time. Limited data are available for contaminants in on-site air, off-site surface water and sediment, and fish and game that may be impacted by the site. Because measured values needed to assess the inhalation pathway at the site were not available, airborne contaminant concentrations were modeled to estimate exposure point concentrations. Although greater uncertainty is therefore associated with the exposures calculated for this pathway, inhalation contributes insignificantly to health effects estimated for the site, except for radon. Therefore, the effect of this uncertainty on the chemical exposure assessment is considered to be small. For the radiological assessment, use of the source term analysis to estimate the exposure point concentrations of radionuclides at each location is expected to result in an overestimation of related risks because of the conservative bias inherent in this approach.

A screening-level analysis was conducted for the buildings to address limitations in the available data. For this analysis, the exposure point concentrations were determined from representative values for the highly contaminated buildings. Exposures for the combined buildings will therefore be overestimated, but the related uncertainty does not affect the outcome of this assessment because interim action decisions have already been made for these structures. A screening-level analysis was also conducted for on-site surface water impoundments, with the exposure point concentrations for direct exposures taken as the highest value measured for each contaminant in the raffinate pits — the most contaminated of the on-site impoundments. This approach is expected to overestimate related exposures. However, the uncertainty associated with estimates for on-site surface water does not affect the outcome of this assessment because cleanup decisions have also been made for this water.

A screening-level analysis was also performed to address the limitations in data available for off-site surface water and sediment; the maximum concentration measured for each contaminant of concern was used as the exposure point concentration, except for uranium in surface water at the Southeast Drainage. For this surface water, half the maximum measured value of uranium was used because of wide variations in flow and concentrations. Samples for the off-site lakes were taken from areas expected to exhibit the highest contaminant levels, so the use of maximum measured values for the exposure assessment is expected to be

conservative. These lakes will be drained in the near future as part of a general sediment management program for the wildlife area. Decisions for the Southeast Drainage are outside the scope of current cleanup decisions, and additional data will be collected to support the upcoming assessment of and decision for this vicinity property. Therefore, the effect of uncertainty in these exposure point concentrations on decisions that will be developed from this assessment is expected to be small.

For the fish and game ingestion pathway, the large uncertainty inherent in biological field sampling significantly impacts the ability to develop reasonable exposure point concentrations from site-specific measurements. For fish, many of the data were negative (i.e., nondetects), and not all available data could be used to estimate exposure point concentrations because of limitations relative to study samples, methods, detection limits, and reporting. In addition, because the lakes are regularly stocked and fished, the length of time the sampled fish may have been exposed to lake constituents is unknown. Therefore, exposure point concentrations estimated from the limited number of measured values cannot be considered representative. Similar factors contribute to the large uncertainty for game ingestion, e.g., the very limited number of reported values, the inability to combine data from different studies into an integrated database, and the lack of knowledge relative to the animal's time spent on-site and at areas of contamination. Nevertheless, available data are expected to provide an indication of potential impacts because measured values could be considered representative of potential exposures to individuals hunting for 30 years.

A great deal of uncertainty is also associated with the modeling used to estimate exposures for the game ingestion pathway. Because of the wide variation in physicochemical and biological parameters, it is very difficult to estimate tissue concentrations of the site contaminants in plants or game animals. Although the values selected for this modeling effort were determined by reviewing the available literature for uptake and assimilation factors, the chemical and biological behavior of contaminants is very much affected by site-specific conditions, e.g., relative to solubility, bioavailability, and bioconcentration. These factors create tremendous uncertainty in the modeling for this pathway. Nevertheless, to provide some indication of the potential health effects associated with game ingestion, risks and hazard indexes have been estimated from the limited information available. Additional ecological data will be collected as part of upcoming site studies, and these data will be incorporated into future environmental documentation for the project.

Extensive data are available for site soil, but heterogeneity in the spatial distribution of contaminants could contribute to uncertainties when estimating appropriate exposure point concentrations. To address this uncertainty, both sitewide and location-specific analyses were conducted as part of the exposure assessment. The UL_{95} value for each contaminant was used as the exposure point concentration for the sitewide analysis to address reasonable maximum exposures for a receptor who might visit random locations across the site. Specific borehole measurements were also used to develop a location-specific analysis that addresses potential "hot-spot" exposures for a receptor who might preferentially spend time at a highly contaminated area of the site. Hence, uncertainties related to data heterogeneity in site soil are expected to be low.

The exposure point concentrations for soil were based on surface samples. The surface radiological samples were generally collected to a depth of 0.15 m (0.5 ft). Sample intervals for the chemical analyses ranged from 0 to 0.15 or 0.61 m (0 to 0.5 or 2 ft), although some samples were composited over 2 m (7 ft). The limited availability of more shallow surface samples, i.e., 0 to 0.15 m (0 to 0.5 ft), for the chemical contaminants introduces some uncertainty into the analysis, but this is not expected to significantly affect the risk calculations because the level of contamination associated with the chemical contaminants is generally low and the distribution of most contaminants in surface soil does not appear to differ from that in subsurface samples.

Although it is recognized that the method for addressing nondetects will affect the exposure point concentrations, no widely accepted methodology is available. The substitutions in this BA for nondetects were contaminant-dependent and were based on the results of statistical summaries and background comparisons, as well as on historical information regarding the expected presence of a contaminant. A sequence of complete calculations was made to assess the impact on the risk estimates of replacing nondetects with the detection limit, half the detection limit, or zero. Little difference resulted from replacing the nondetect with any of these values because concentrations associated with the detection limits of contaminants of concern for the site are not generally associated with significant health impacts and the detection frequency is typically high for many of the analytes.

Finally, the time frame that is assumed when estimating exposure point concentrations can introduce uncertainty into an exposure assessment. To address this issue, a staged approach was developed for the assessments prepared for the Weldon Spring site to reflect the fact that conditions are changing as a result of implementing response actions to mitigate potential contaminant releases prior to the comprehensive cleanup decision. (For example, diversion systems and other runoff controls are being constructed and the buildings and raffinate pit surface water are being removed.) Under this approach, the time frames assumed for each assessment were determined on the basis of the documentation stage; this BA represents the first-stage assessment and assumes a relatively short time period over which exposures could occur (10 to 30 years). Hence, uncertainty associated with the potential impact of fate and transport over time on the estimated exposure point concentrations is considered to be low.

5.6.2.2 Scenario Assumptions and Intake Parameters

Some uncertainty is associated with the assumptions used to identify scenarios and intake parameters for the exposure assessment. Site-specific factors were used to identify the potential receptors (e.g., a recreational visitor to the area) and to select the scenario assumptions, such as extent of exposure (i.e., exposure time, frequency, and duration). These assumptions incorporate information on current land use and reasonable projections of future land use that consider the time frame of the assessment. The uncertainty in the selected scenarios is low because the time period is relatively short and federal and state ownership of surrounding land is expected to continue during that period. The surrounding wildlife areas are the most heavily used in the state, and the state plans to further expand their recreational use. Similarly, the adjacent Army property is expected to remain under government ownership. Therefore, an on-site worker and trespasser and an off-site recreational visitor are considered appropriate for

current conditions, and a recreational visitor is considered appropriate for future conditions under which site controls are hypothetically assumed to be lost.

Residential use is unlikely within the time frame of this assessment because it is inconsistent with (1) the ongoing cleanup activities at the site; (2) the nature of the site, including the presence of deteriorating buildings and other surface features; and (3) the surrounding land use and reasonable projections thereof. Therefore, a residential scenario was not considered for this BA. However, this scenario is evaluated in the rebaseline assessment (Appendix E of the FS), which includes a future time period that extends into the long term.

Large uncertainty is associated with certain parameters specific to the dermal pathway, as discussed in Section 5.5.2. In contrast, considerable information is available for the other pathways with respect to reasonable assumptions for intake parameters (e.g., inhalation rates), so related uncertainty is expected to be low. To estimate the reasonable maximum exposures for the identified receptors, best professional judgment was used in defining the variables that determine the extent of exposure. Intake parameters used in the exposure assessment were derived from data in the literature, including values provided by the EPA (1989b, 1989j, 1991b). Because each of these values generally represents the 95th percentile of the distribution for that parameter, combining them results in a value that represents an even higher percentile for the overall exposure. Thus, in some cases, the "reasonable" representative exposure may be significantly overestimated. However, the intent of this conservative approach is to ensure protection of human health. Its impact on the overall risk estimates will be qualitatively addressed in the development of cleanup criteria for the site.

5.6.2.3 Exposure Pathways

The exposure pathways quantified in this BA were determined on the basis of the site conceptual model, related characterization data, and fate and transport considerations. The uncertainty associated with selecting pathways for this assessment is low because site characterization data support the conceptual model and factors affecting contaminant fate and transport are well understood for purposes of this selection.

Potential health impacts from exposure to groundwater are not addressed in this BA because groundwater data are limited and the exposure pathway is incomplete (i.e., no human receptors currently exist for groundwater impacted by site contaminants). A preliminary assessment of groundwater exposure via drinking water ingestion is presented in Appendix E of the FS. This assessment will be revised for presentation in the human health assessment that will be prepared for the groundwater operable unit when more information becomes available.

5.6.3 Toxicity Assessment

Standard RfDs and slope factors established by the EPA were used to estimate potential noncarcinogenic and carcinogenic health effects from exposure to chemical contaminants at the site. Similarly, standard DCFs and risk estimators were used to estimate the carcinogenic

hazards associated with radioactive contaminants. The methods used to derive the RfDs, slope factors, and DCFs and the uncertainties inherent in these values are described in Chapter 4 and Section 5.5. The uncertainty in estimated health effects associated with the unavailability of toxicity values for all contaminants of concern at the site is discussed in Sections 5.5.2 and 5.6.4.

In accordance with EPA guidance, the uncertainties associated with the toxicity values used in this BA are summarized for substances that contribute most to the estimates of site risks. To focus this summary, contributors to carcinogenic risk estimates that are below the target range identified by the EPA are not discussed in this section. Similarly, those chemicals that do not contribute to segregated hazard indexes greater than 1 are not discussed. Additional information on the toxicity values for these contaminants is presented in Chapter 4.

Radioactive contaminants are generally the primary contributors to health effects estimated for the site. For these contaminants, conversion factors that represent the probability of cancer induction per unit dose were used to convert radiation doses to health risks. The health effects associated with radiation exposure have been studied for many years and are well known. The risk estimators used in this assessment are generally accepted by the scientific community as representing realistic projections of the hazards associated with radiation exposure.

For this assessment, arsenic, asbestos, uranium, PAHs, and PCBs are the primary contributors to estimated chemical health effects. Arsenic and asbestos are classified as human carcinogens (Group A) on the basis of observed increases in skin and lung cancer from arsenic exposures and increases in lung cancer and mesothelioma from asbestos exposures.

The EPA has identified a medium confidence level in the oral RfD for arsenic because of study limitations relative to sample size and dietary sources. The corresponding uncertainty factor used to derive the RfD is 3 because of study limitations in addressing reproductive toxicity and all sensitive individuals.

The EPA has identified a medium confidence level in the oral toxicity value for uranium and has used an uncertainty factor of 1,000 to derive the RfD. This high value reflects the general absence of appropriate scientific data. In addition, the RfD is derived for soluble uranium compounds and would thus tend to result in conservative estimates of the hazard quotient if insoluble compounds are also present, as is the case at the Weldon Spring site.

Both the carcinogenic PAHs and the PCBs are classified as Group B2 (probable human) carcinogens, indicating that no direct data exist for human carcinogenicity. However, these compounds have been shown to induce cancer in experimental animals. The biological potencies of the individual PAHs and the PCB congeners vary widely. Therefore, uncertainty is inherent in EPA's derivation of a representative slope factor for each of these classes of compounds. Similar considerations affect the uncertainty of the RfD for the carcinogenic PAHs identified by the EPA and the RfD for PCBs identified by the ATSDR.

5.6.4 Risk Characterization

Throughout this document, each step of the radiological and chemical risk assessment has been presented separately because the radiological and chemical characterization studies of the site were generally conducted separately and the methodologies for estimating the carcinogenic risks from exposures to radionuclides and chemicals differ considerably. However, the total carcinogenic risk to an individual is that resulting from exposures to both the radioactive and chemical contaminants. This total risk can be approximated by summing the radiological and chemical risks, assuming that the carcinogenic effects are neither antagonistic nor synergistic. However, differences exist in the assumptions and approaches used to develop the risk estimators for these two classes of contaminants; hence, the total risk estimates should be interpreted in terms of the following limitations.

The most significant difference in the risk assessment methodology is the basis for the risk estimators. For radionuclides, the risk estimators are best estimate, average values; for chemicals, the risk estimators (i.e., slope factors) generally represent the upper bound or upper 95% confidence limit of the slope of the dose-response curve. In addition, the risk estimators for radionuclides are derived from human epidemiological data whereas those for chemicals are typically derived from experiments with laboratory animals. Hence, the uncertainties associated with the risk estimates will differ. Because of these limitations, the radiological and chemical risk estimates have been tabulated separately throughout this assessment; however, the total risk estimates are summarized in Chapter 6 (Tables 6.4 and 6.5).

Some of the procedures used and uncertainties inherent in the human health assessment process may tend to underestimate potential risks. These include the use of standard DCFs that are generally based on adult exposures for estimating radiation doses to children and the lack of appropriate methodology and toxicity values to quantify chemical health effects for all contaminants of concern and routes of exposure. However, most of the assumptions built into this BA tend to overestimate rather than underestimate potential risks — including conservative assumptions for the exposure scenarios (e.g., the number of times the trespasser would enter the site despite the constant presence of security and increasing worker activity at the site). Uncertainties in the estimated health effects that result from limitations in the toxicological data are addressed separately for radiological and chemical effects.

Extensive data are available on which to base estimates of human radiation toxicity. The recently issued BEIR V study (National Research Council 1990) presents a detailed description of current data on the health risks of exposure to low levels of ionizing radiation. This study and the BEIR IV study (National Research Council 1988), which addresses the health risks associated with radon and other alpha-emitting radionuclides, provide ample scientific information for the assessment. The uncertainty associated with the risk estimates used to assess radiation toxicity in this BA is therefore relatively low.

The radiological dose conversion factors used in this assessment are based on the ICRP reference man (ICRP 1975). The reference man is an adult male weighing 70 kg. The ICRP selected such a standardized individual for its dosimetry models because the main concern was worker protection and the majority of radiation workers are adult males. Although children are

more susceptible to radiation exposure, i.e., the radiation doses are larger for children than adults for the same intake of radioactive substances, such effects are significant only for young children (see Cristy et al. [1986] and ICRP [1989]). The uncertainty associated with using DCFs developed for adults for an adolescent weighing 50 kg is low, and it does not significantly impact the radiological risks for the receptors evaluated in this document.

The estimation of health effects associated with radiation doses was based on lifetime-average risk estimators for all routes of exposure. These lifetime-average risk estimators are appropriate because they reflect the likely conditions of exposure at the site, i.e., a range of age groups could be exposed to the radioactive contaminants. Although the risk estimators for inhalation of radon-222 decay products developed in the BEIR IV study were largely based on adult exposures, adjustments were made in that study to account for exposures that could occur in the general population over a lifetime.

For chemicals, oral RfDs were available for all of the toxicologically important contaminants of concern at the site, except for lead. The lack of an RfD for lead ingestion is expected to significantly underestimate the overall hazard index at those few on-site locations where the lead concentration is elevated. Although the hazard quotient for lead cannot be quantified, the EPA has established guidelines for lead concentrations in soil that are considered protective for residential settings (EPA 1989c). These levels range from 500 to 1,000 mg/kg, the lower limit of which is exceeded at only three surface locations on-site. In addition, the EPA has developed a model that can be applied to evaluate lead exposures for children in a residential setting; this model was applied in evaluating the residential scenario, which is presented in Appendix E of the FS.

Few inhalation RfDs are available for the contaminants of concern. Hence, the potential for noncarcinogenic effects, as indicated by the inhalation hazard index, is significantly underestimated. However, the air pathway is a relatively minor contributor to chemical effects estimated for the site. In addition, inhalation slope factors are available for the carcinogens, and the carcinogenic end point is generally more sensitive than the noncarcinogenic end point. Thus, the primary health effects associated with these chemicals have been quantified in this BA.

Slope factors are available for most of the known oral carcinogens, except for lead and asbestos. Hence, the carcinogenic risks associated with these exposures are expected to be underestimated. Although risks for lead cannot be quantified, the EPA has established guidelines for lead concentrations in soil, as identified above. The primary route of exposure to asbestos (which is present in the chemical plant buildings) is inhalation, for which a slope factor is available; hence, risks have been quantified for this route.

Inhalation slope factors are available for PAHs and for all of the carcinogenic metals except lead, for which the EPA has developed a model to assess exposures. Inhalation slope factors are unavailable for 2,4-DNT, 2,6-DNT, TNT, or PCBs. However, these compounds are present in only a few surface soil locations and the inhalation pathway is not a significant contributor to chemical health effects. Therefore, the lack of this toxicological information is not expected to significantly affect the overall risk estimates for the site.

5.7 SUMMARY OF THE HEALTH RISK CHARACTERIZATION

Selecting the contaminants of concern for evaluation in this BA was a key step in the risk assessment process. Contaminants were selected on the basis of the history of site operations and an evaluation of characterization data with respect to the distribution and concentration of contaminants in the various media at the site and the potential contribution of individual contaminants to overall health effects. Confidence is high that the significant contaminants were identified because considerable information is available from the site characterization effort and the contaminant screening procedure was designed to screen out only those contaminants that would contribute insignificantly to potential health effects. The contaminants of concern are 11 radionuclides of the uranium-238, thorium-232, and uranium-235 decay series and 47 chemicals consisting of metals, inorganic anions, nitroaromatic compounds, PAHs, PCBs, and asbestos.

The radionuclides, metals, and inorganic anions identified as site contaminants also occur naturally in the environment. In general, sitewide contaminant levels are at or near background concentrations; however, radionuclide concentrations are significantly elevated at several discrete locations. Local background concentrations of radionuclides in the uranium-238 and thorium-232 decay series are about 1.2 pCi/g; background concentrations of radionuclides in the uranium-235 decay series are about 0.055 pCi/g. The radiological risks associated with these background levels and with the background gamma exposure rate of 10 μR/h for the scenarios used in this assessment are 1×10^{-4} for the worker, 4×10^{-7} for the trespasser, and 2×10^{-5} for the recreational visitor.

The sitewide concentrations of metals and inorganic anions are similar to levels at the nearby background location. However, the concentration of each metal and inorganic anion of concern is considerably elevated above background at one or more discrete locations on-site. Information on background concentrations of the metals and inorganic anions was limited for surface water and sediment, so statistical comparisons cannot be made. Except for asbestos, the other chemical classes present at the site — nitroaromatic compounds, PAHs, and PCBs — are not naturally present in the environment. However, asbestos, PAHs, and PCBs are common environmental contaminants resulting from human activities.

Cancer induction is the primary health effect associated with radionuclides at the site, and 17 of the chemical contaminants of concern are potential carcinogens. From this assessment, arsenic, asbestos, the carcinogenic PAHs, and PCBs are the contaminants contributing most significantly to chemical risks. Arsenic induces cancer of the lung and skin in humans; asbestos induces lung cancer, mesothelioma, and cancer of the gastrointestinal tract in humans; and the PAHs and PCBs induce cancer in experimental animals.

Arsenic, uranium, and PCBs are the site contaminants that have a pathway-specific hazard quotient of greater than 1 or that contribute significantly to a segregated hazard index of greater than 1 at certain locations. The critical noncarcinogenic effects in humans associated with ingestion of arsenic are hyperpigmentation and keratosis of the skin and possible vascular complications. The main toxic effect of uranium in humans is kidney damage. Skin effects and

liver toxicity have been observed in workers exposed to PCBs, and reproductive and developmental effects have been reported in experimental animals and may also be induced in humans.

The health effects associated with exposure to lead could not be quantitatively assessed because of the unavailability of toxicity values or models appropriate for the receptors evaluated in this BA. (Site-specific exposure to lead is modeled for the residential scenario presented in the rebaseline assessment of the FS [DOE 1992a].) However, levels exceeding general EPA guidelines for lead concentrations in soil for residential settings have been measured at only a few on-site locations. The fetus and young children are especially sensitive to the effects of lead, which include premature birth, low birth weight, and learning impairments. Other effects observed in humans include reproductive effects in males and females and brain and kidney damage.

Confidence in the quantitative measures of health effects (i.e., carcinogenic risks and hazard indexes) depends on a number of factors, including uncertainty in the toxicity values (RfDs and slope factors) used to calculate these measures. The uncertainty associated with the RfDs is indicated both by an EPA-defined confidence level in the available toxicological data for a compound and by the uncertainty factors used to derive these values. For site contaminants, the level of confidence in the data from which the RfDs were derived ranges from low to high (or unavailable), and the uncertainty factors used in deriving the RfDs range from 1 (indicating low uncertainty) to 10,000 (indicating very high uncertainty). Of the contaminants previously identified as contributing significantly to the hazard index, the EPA-defined level of confidence in the oral RfD is medium for arsenic and uranium. Information as to the level of confidence was not available for PCBs. The uncertainty factors used in deriving the RfDs are 3 for arsenic and 1,000 for uranium. The high uncertainty factor for uranium results in a lower (more conservative) RfD, so the hazard quotients for this metal may be overestimated.

Chemical carcinogens are assigned both a weight-of-evidence classification and a slope factor. The weight-of-evidence classification characterizes the extent to which the available data indicate that an agent is a human carcinogen. Of the carcinogens previously identified as contributing significantly to site risks, arsenic and asbestos are classified as Group A human carcinogens (i.e., there is direct evidence of carcinogenicity to humans). The carcinogenic PAHs and PCBs are classified as Group B2 probable human carcinogens (i.e., the evidence of carcinogenicity is based solely on experimental data in animals). The uncertainties associated with slope factors are less well defined than those associated with RfDs. Slope factors are generally derived with very conservative models; these factors represent the upper 95% confidence limit for the probability of a carcinogenic response.

Receptors and exposure pathways were identified in this BA on the basis of site-specific considerations of current land use and reasonable projections of future land use that considered the time frame of this analysis. Confidence is high that the main exposure pathways and potential receptors have been identified and evaluated. Although additional receptors and activities could be identified, exposures would be similar to or less than those estimated for the specific receptors and pathways considered for this stage of the assessment process. Standard (conservative) intake parameters were used in assessing the inhalation and ingestion pathways,

and, although some uncertainty exists with respect to these values, this uncertainty is not expected to significantly affect the analysis.

The potential for health effects from exposure to site-related contaminants was estimated for receptors on-site and in adjacent off-site areas impacted by site releases. The on-site exposure points evaluated in this BA were sitewide soil, outdoor air, the raffinate pits, and the chemical plant buildings. To focus the discussion, the magnitude of the estimated carcinogenic risks and hazard indexes is discussed relative to remedial action goals for an NPL site, as defined by the EPA (1989j, 1990c). These goals are an incremental cancer risk of 1×10^{-6} to 1×10^{-4} and a hazard index not exceeding 1 for toxic effects other than cancer.

The radiological risks associated with exposures to sitewide soil and outdoor air under current site conditions are within EPA's target range for the trespasser and recreational visitor, but the radiological risk to a worker is 5×10^{-4}. The location-specific analysis identified many discrete locations for the worker scenario and several locations for the recreational visitor scenario at which the risk exceeds EPA's target range. The total risk is dominated by inhalation of radon-222 decay products derived from radium-contaminated soil. However, the risks estimated for inhalation of radon-222 decay products are probably high by a factor of 2 or more because conservative assumptions were used to estimate airborne concentrations. The risks associated with incidental ingestion of soil and external gamma irradiation are significantly lower than the risk for radon-222 decay products. The risks from inhalation of airborne particulates other than radon-222 decay products are much lower than the radiological risks from all other pathways.

Under current site conditions, the radiological risk associated with surface water and sludge in the raffinate pits is 2×10^{-4} for the trespasser and 3×10^{-3} for the recreational visitor; these estimates exceed EPA's target range. Most of the total risk is due to incidental ingestion of actinium-227, protactinium-231, and thorium-230 from the raffinate pit sludge. The risks associated with the raffinate pits represent the most serious threat to human health at the Weldon Spring site.

Inhalation of radon-220 decay products and incidental ingestion of radioactively contaminated residues (principally thorium-232 and uranium-238) are the major routes of exposure in the chemical plant buildings (Building 403, which is significantly contaminated with uranium and thorium, was evaluated for this screening-level assessment). Under current conditions, the total radiological risk to a trespasser is just below the upper limit of the target range (1×10^{-4}) and the risk to the future recreational visitor is 1×10^{-3}, which exceeds this limit by a factor of 10. Because these risks are associated with one of the more highly contaminated buildings, risks for the other buildings would probably be lower.

The radiological risks estimated for the off-site lakes and Burgermeister Spring are within EPA's target range. The preliminary estimate for the Southeast Drainage is 2×10^{-4}, which exceeds the upper end of the target range by a factor of 2. The primary contributor to the total risk is ingestion of sediment contaminated with thorium-230. The location-specific analysis for the Southeast Drainage has identified discrete areas at which continued exposures over the long term could result in elevated risks. Additional characterization data will be

collected to support a more comprehensive risk assessment of the drainage, which will be presented in forthcoming documents within the next several years.

The chemical carcinogenic risks and hazard indexes associated with inhalation of outdoor air are insignificant relative to the other pathways at the site and are not further discussed. The estimated risks from incidental ingestion of soil for the worker, trespasser, and future recreational visitor (based on the sitewide UL_{95}) are within or less than EPA's target range, and the hazard index is less than 1. However, the location-specific analysis identified one on-site location for which the chemical risk to the worker is greater than 1×10^{-4}, and the segregated hazard index for the worker exceeds 1 at five locations.

Under current site conditions, the estimated chemical carcinogenic risk for the trespasser from exposures to surface water and sludge at the raffinate pits is 9×10^{-6} and the hazard index is less than 1. The estimated risk for the future recreational visitor is at the upper limit of the target range (1×10^{-4}), and the segregated hazard index is less than 1. The carcinogenic risks estimated for the trespasser and the recreational visitor at the chemical plant buildings (3×10^{-4} and 3×10^{-3}, respectively) exceed EPA's target range, and hazard indexes exceed the level of concern for noncarcinogenic effects. Chemical carcinogenic risks of greater than 1×10^{-4} were estimated for scenarios in which exposures for the trespasser and future recreational visitor were assumed to be distributed among sitewide soil and air, the raffinate pits, and the chemical plant buildings; hazard indexes greater than 1 were also estimated for these scenarios. Exposures at the buildings dominate the risks and hazard indexes estimated for this multiple pathway analysis. Incidental ingestion of PCB-contaminated residues in the buildings is the most significant exposure with respect to both the risk and hazard index. The chemical carcinogenic risks estimated for the off-site locations (i.e., Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage) are within or less than the target risk range. The hazard indexes are all less than the level of concern.

Uncertainty is associated with each step of the risk assessment process. Although several of the procedures used and limitations in available information will, in some cases, result in underestimates, the conservative assumptions built into this BA tend to overestimate potential risks. The significance of the uncertainties must also be considered relative to the purpose of this assessment, which is to support forthcoming remedial action decisions for the chemical plant area. Decisions for managing on-site surface water, the buildings, and off-site lakes have already been made, and decisions for the Southeast Drainage will be addressed under a future response action. Thus, the screening-level analyses conducted for these media and locations are considered appropriate for this BA. Soil is the most important medium with respect to decisions for the current stage of site cleanup. Uncertainties associated with estimating exposures to soil were reduced by conducting parallel analyses, one with the sitewide UL_{95} values to determine reasonable maximum exposure point concentrations and the second with concentrations measured at each borehole location to determine the health effects associated with possible hot spots. Additional analyses for soil contamination are presented in the rebaseline assessment (Appendix E of the FS), and the results of both assessments were used to develop preliminary cleanup criteria for the site. These criteria are discussed in Chapter 2 of the FS, and the final criteria will be presented in the record of decision for the chemical plant area.

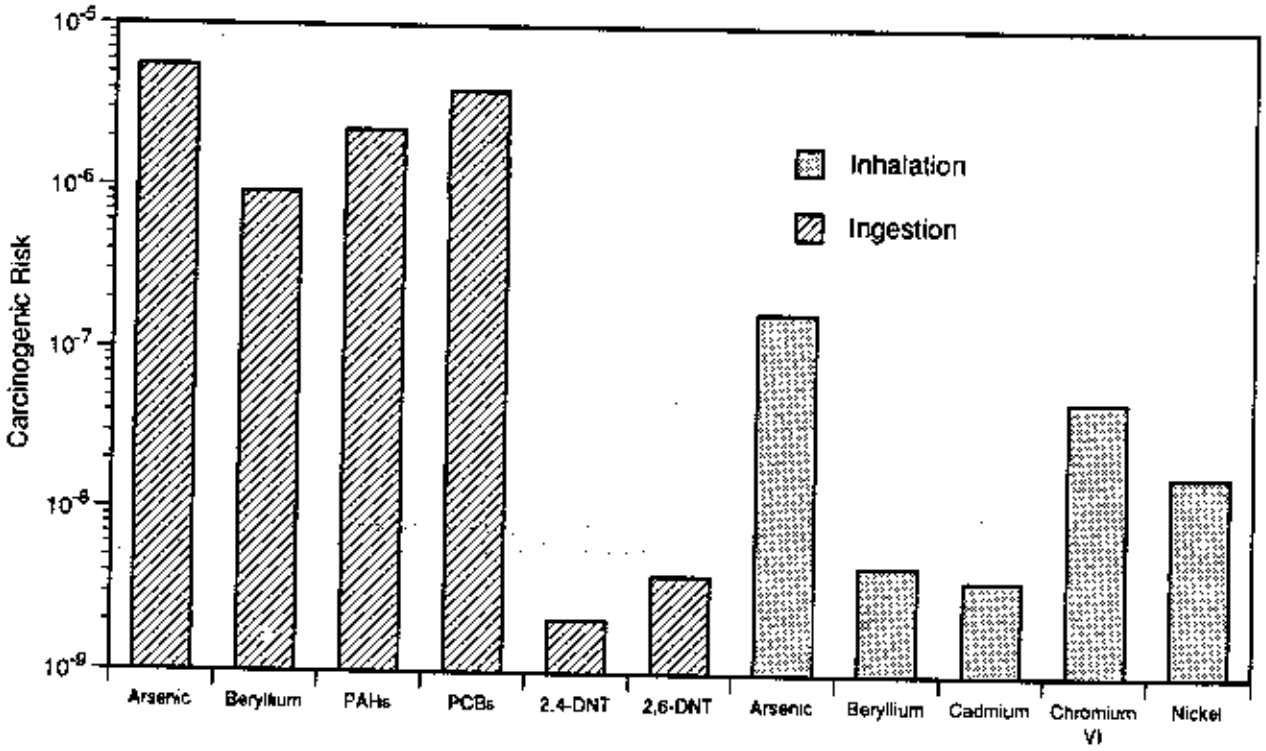


FIGURE 5.1 Relative Contaminant Contribution to the Pathway-Specific Chemical Carcinogenic Risks for the Worker

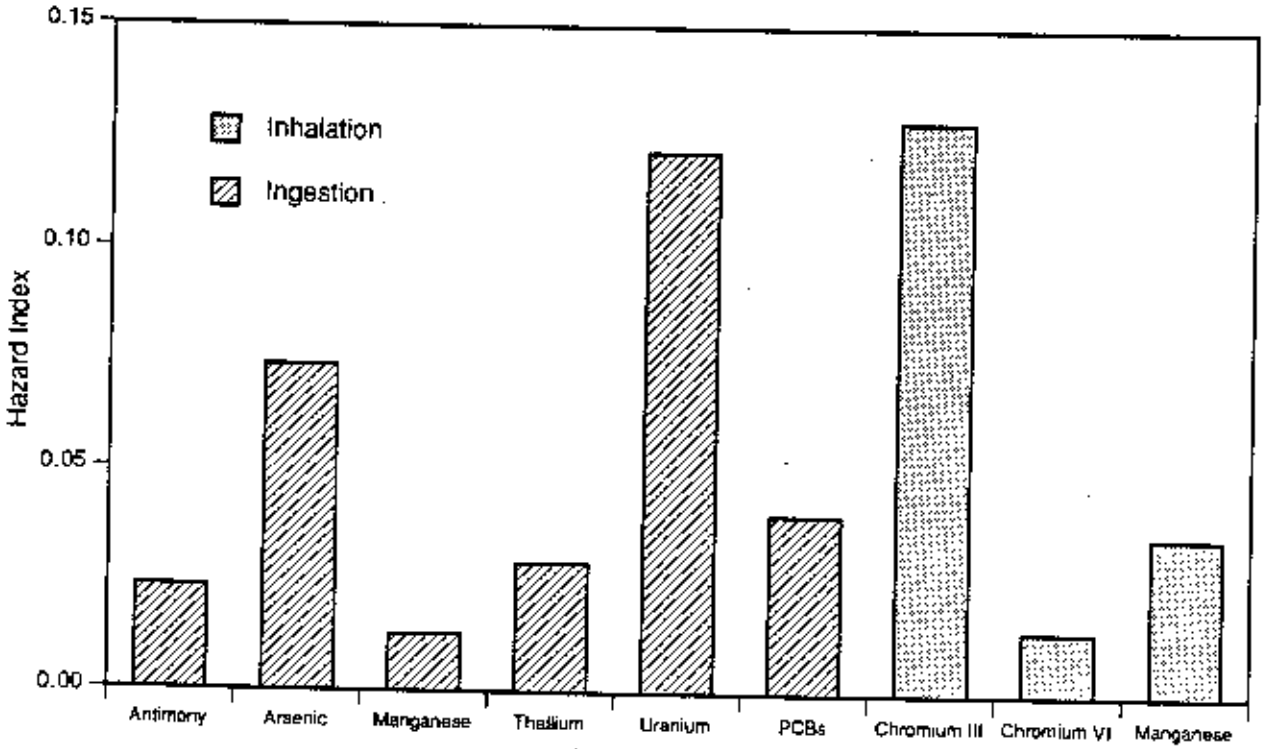


FIGURE 5.2 Relative Contaminant Contribution to the Pathway-Specific Hazard Indexes for the Worker

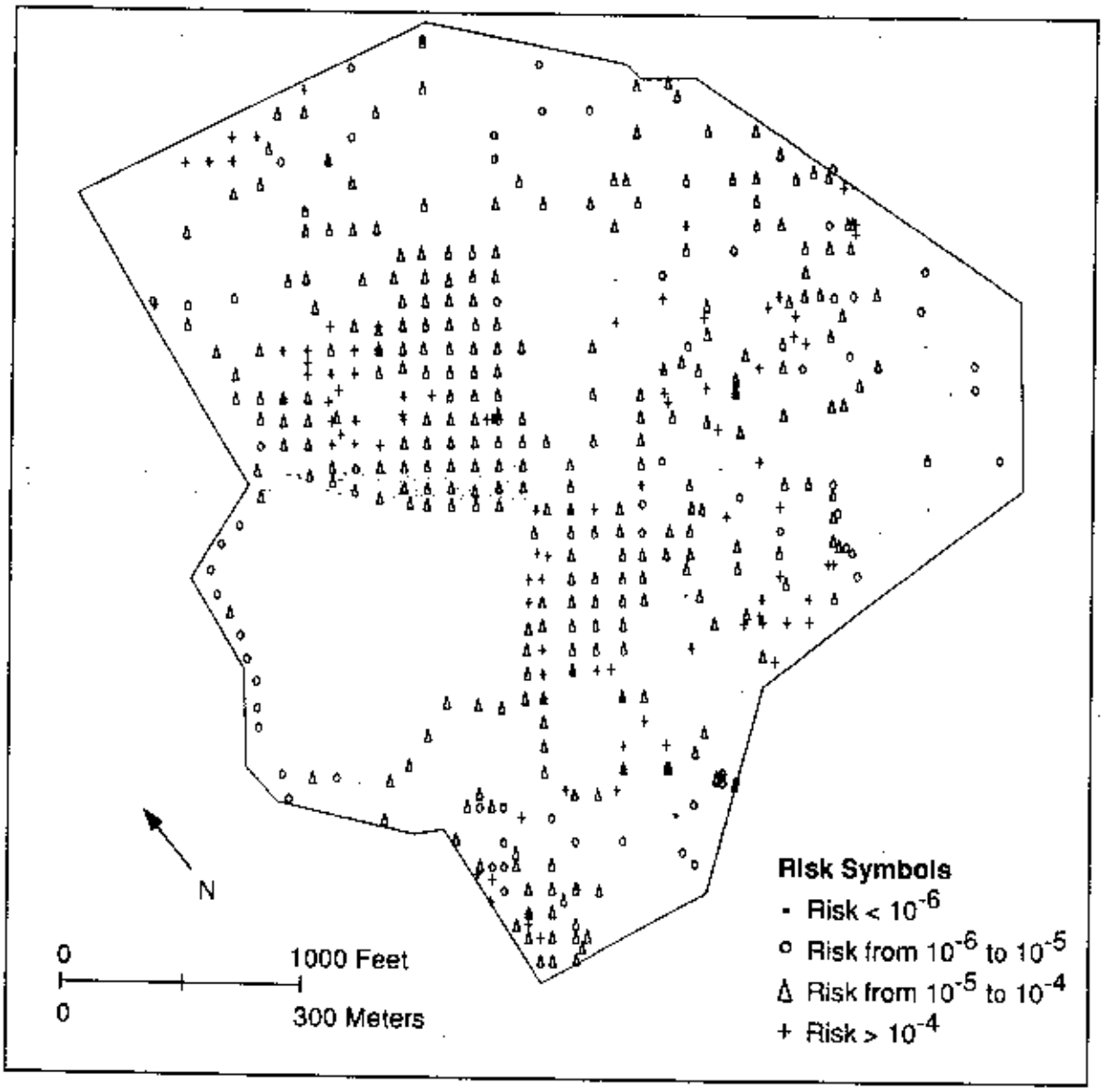


FIGURE 5.3 Location-Specific Radiological Risks for the Worker

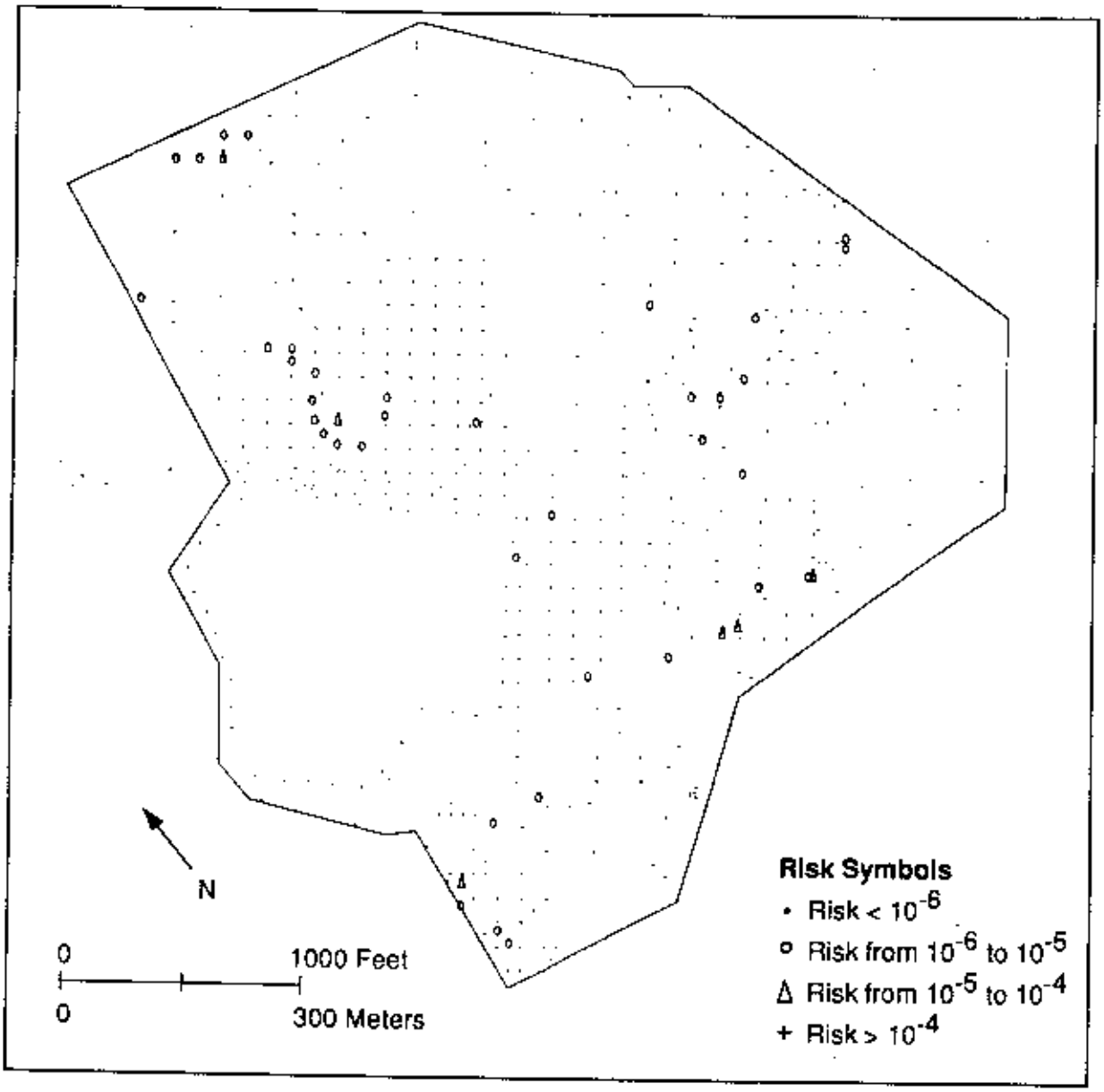


FIGURE 5.4 Location-Specific Radiological Risks for the Trespasser

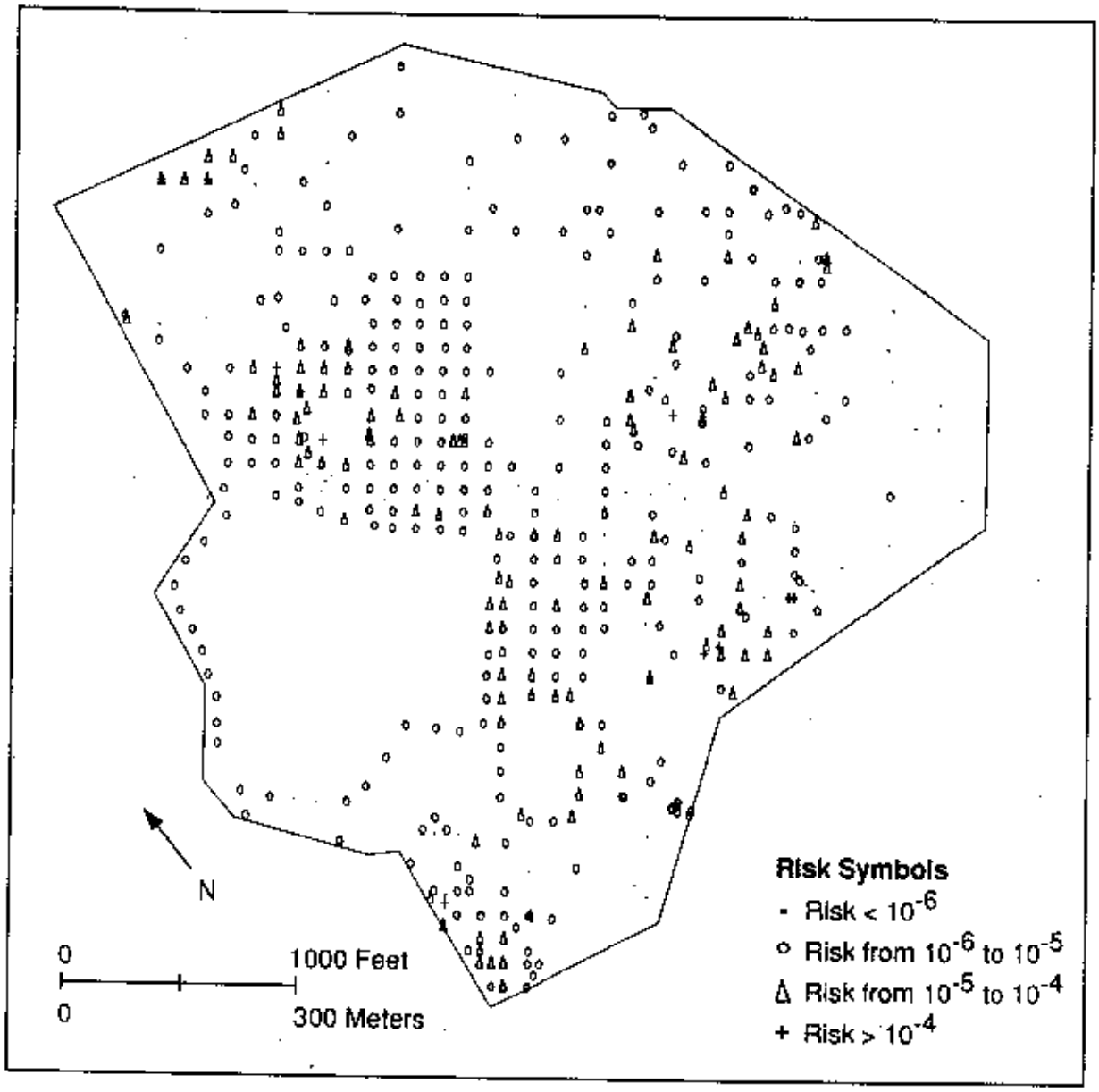


FIGURE 5.5 Location-Specific Radiological Risks for the Future Recreational Visitor

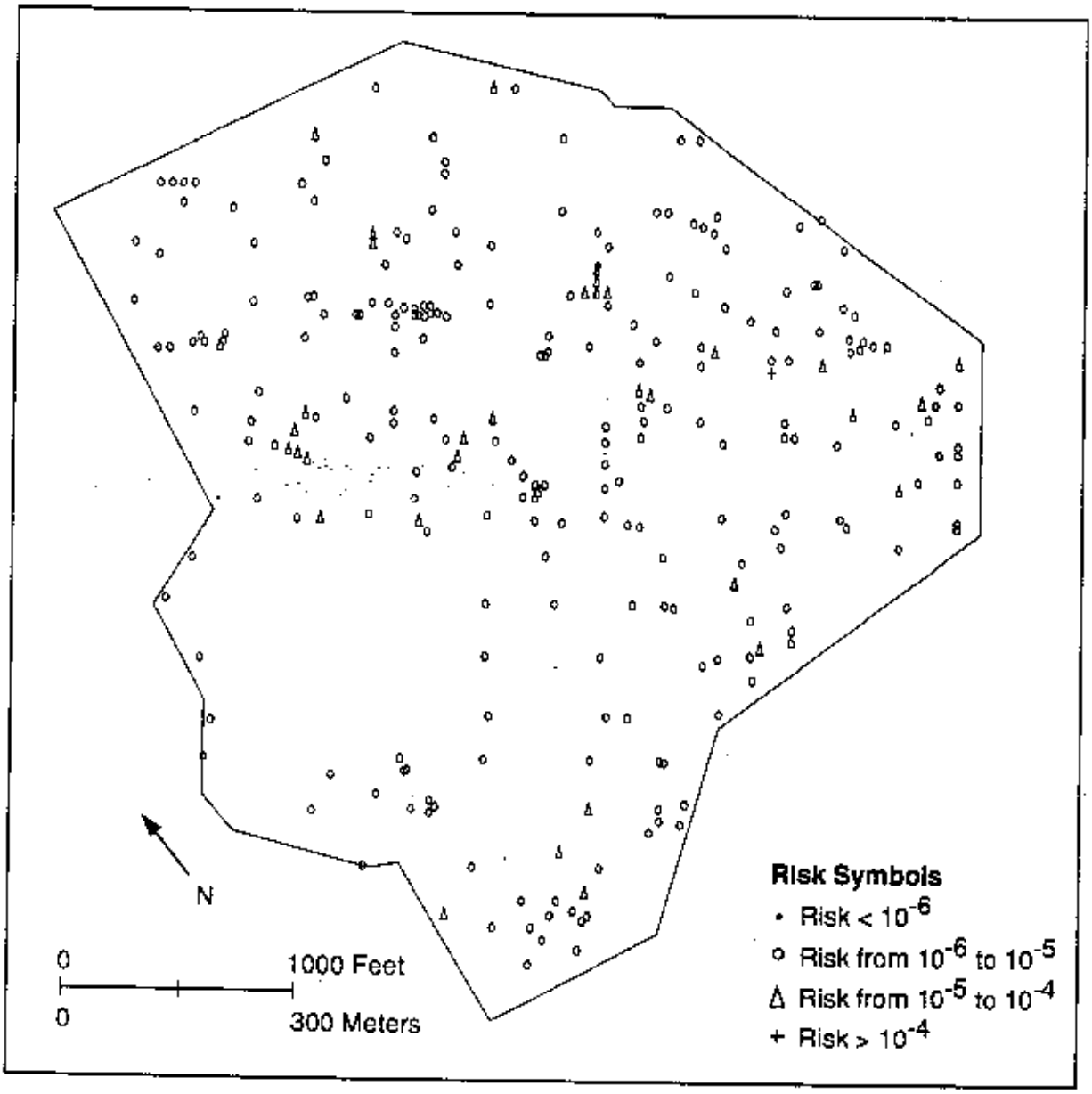


FIGURE 5.6 Location-Specific Chemical Carcinogenic Risks for the Worker

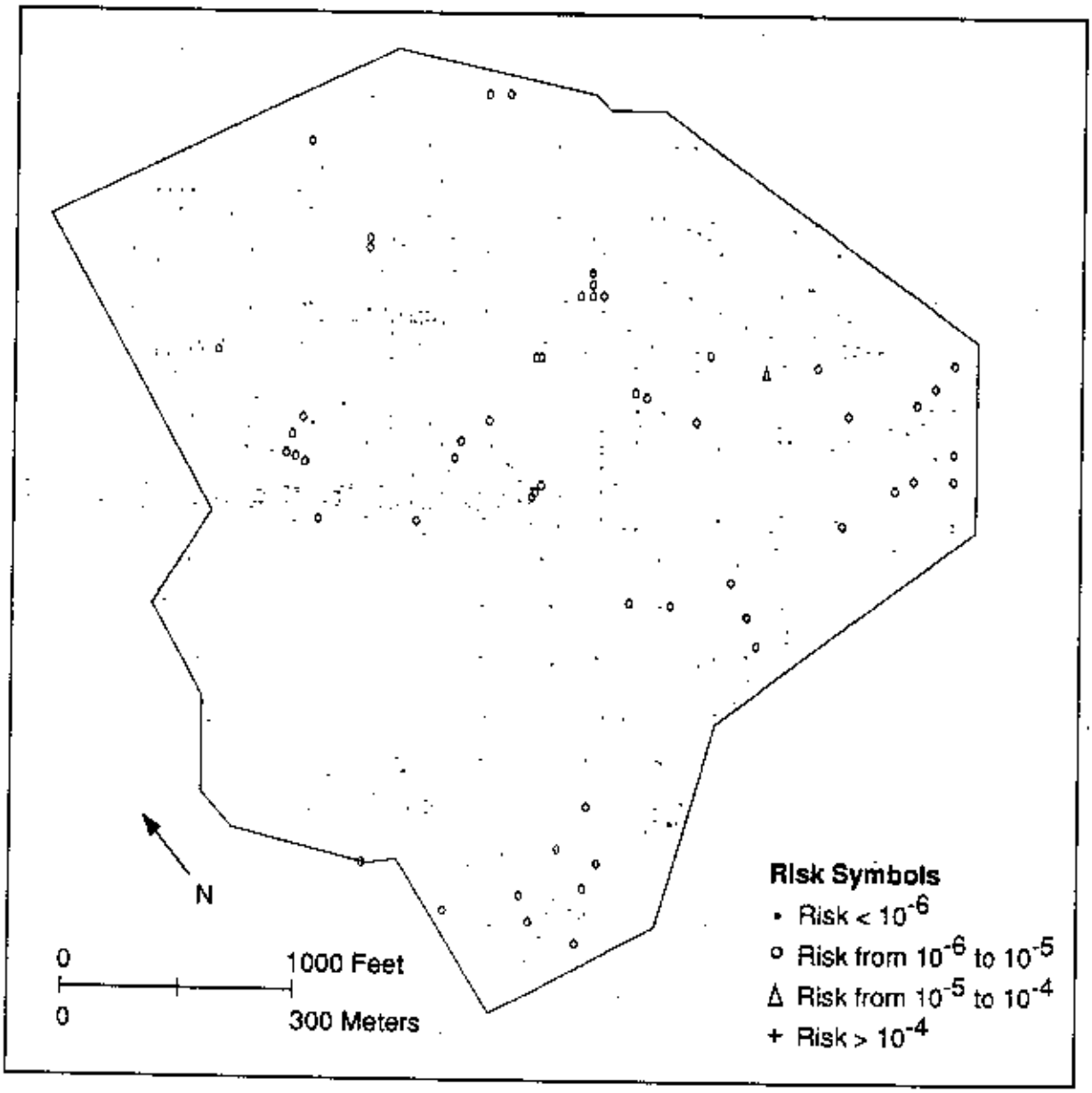


FIGURE 5.7 Location-Specific Chemical Carcinogenic Risks for the Recreational Visitor.

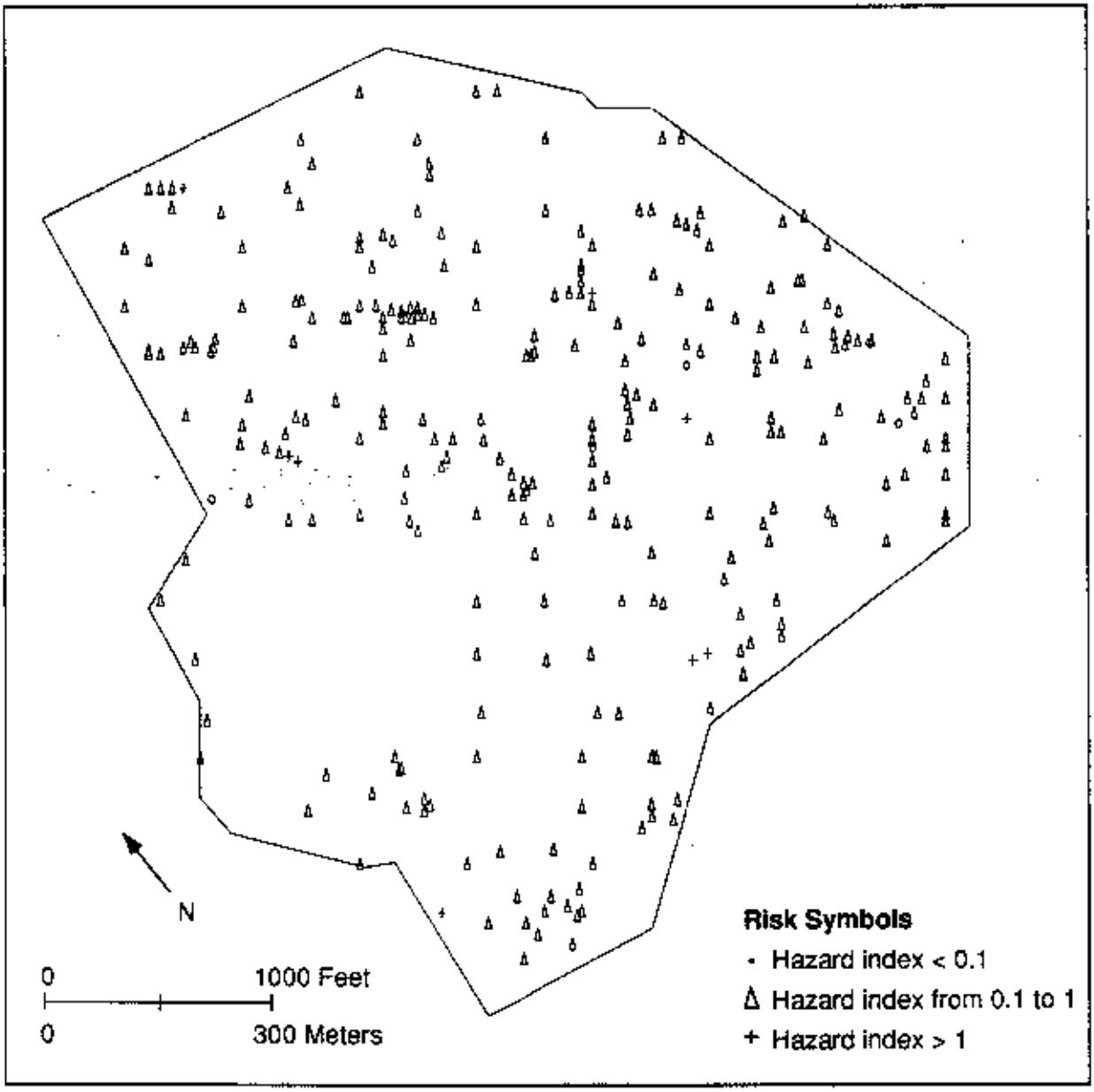


FIGURE 5.8 Location-Specific Hazard Indexes for the Worker

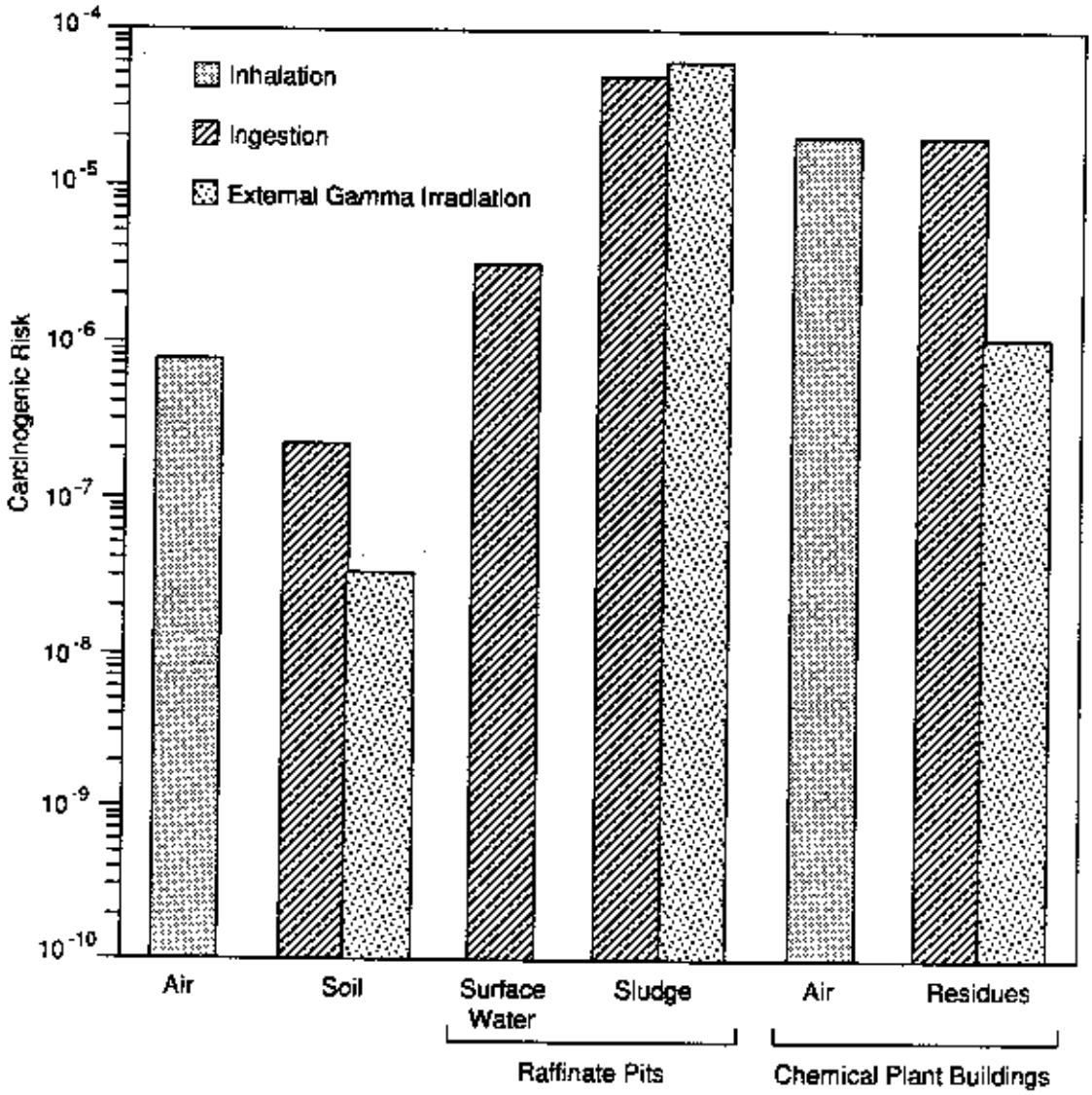


FIGURE 5.9 Relative Pathway Contribution to the Overall Radiological Risk for Scenario T12

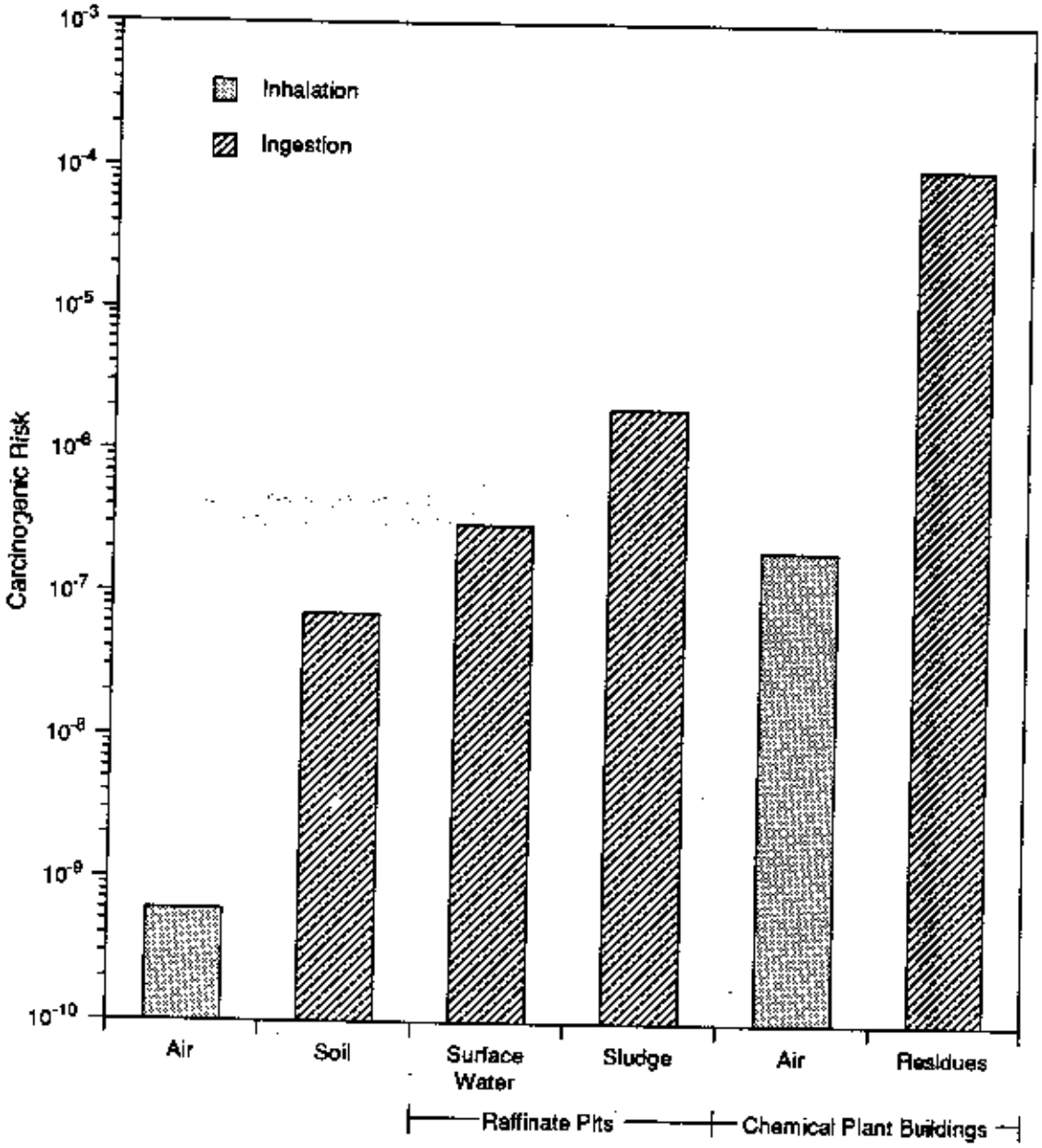


FIGURE 5.10 Relative Pathway Contribution to the Overall Chemical Carcinogenic Risk for Scenario T12

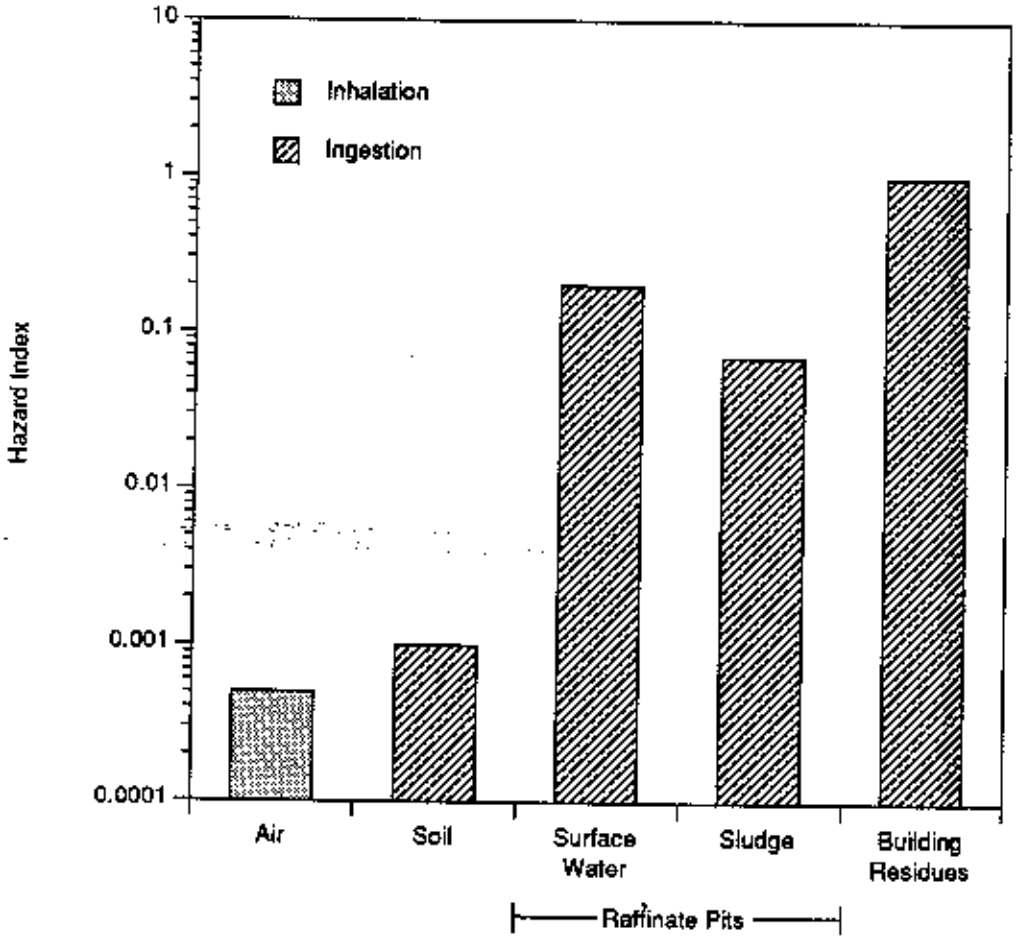


FIGURE 5.11 Relative Pathway Contribution to the Overall Hazard Index for Scenario T12

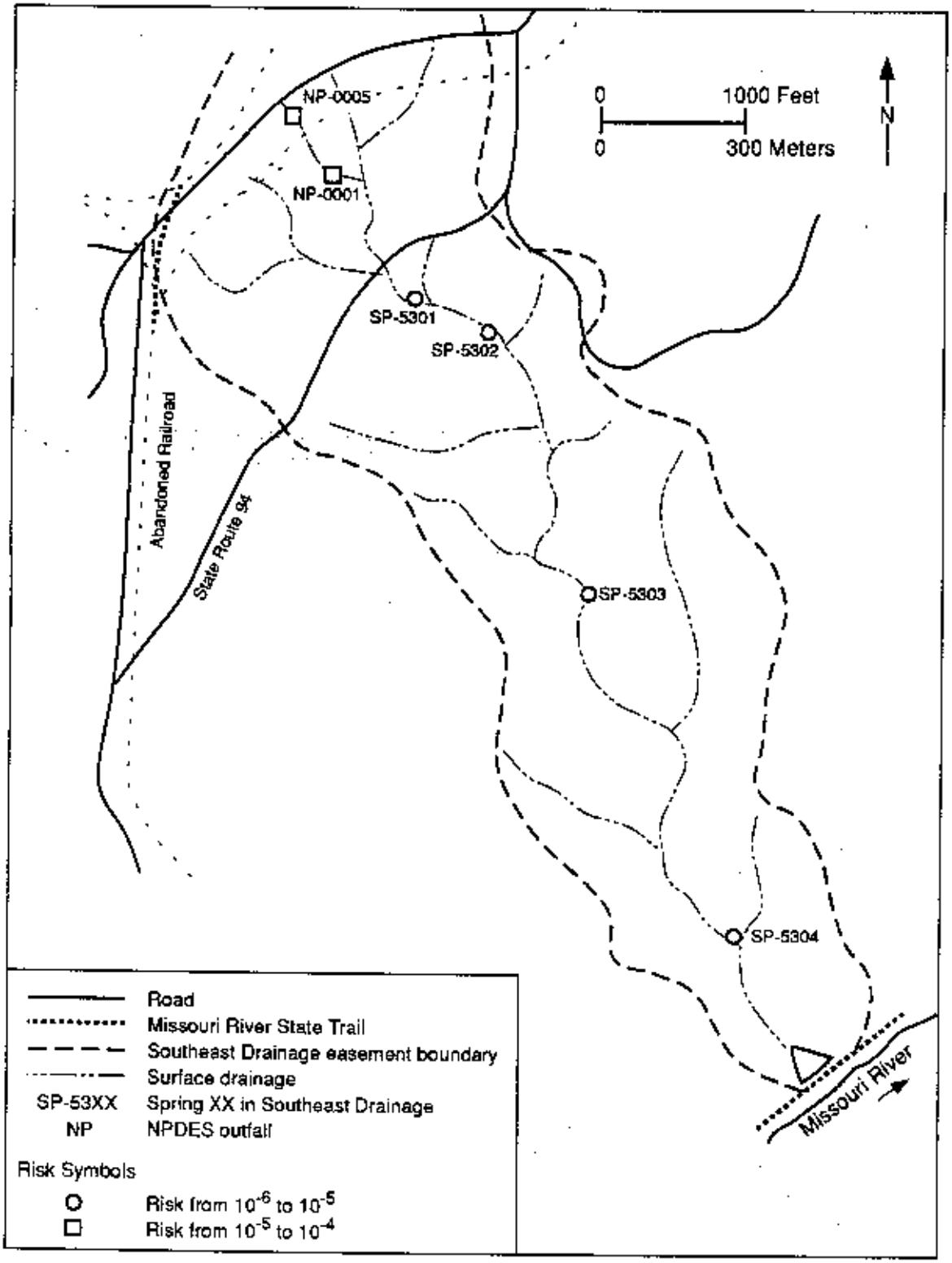


FIGURE 5.12 Estimated Radiological Risks for Ingestion of Surface Water from the Southeast Drainage

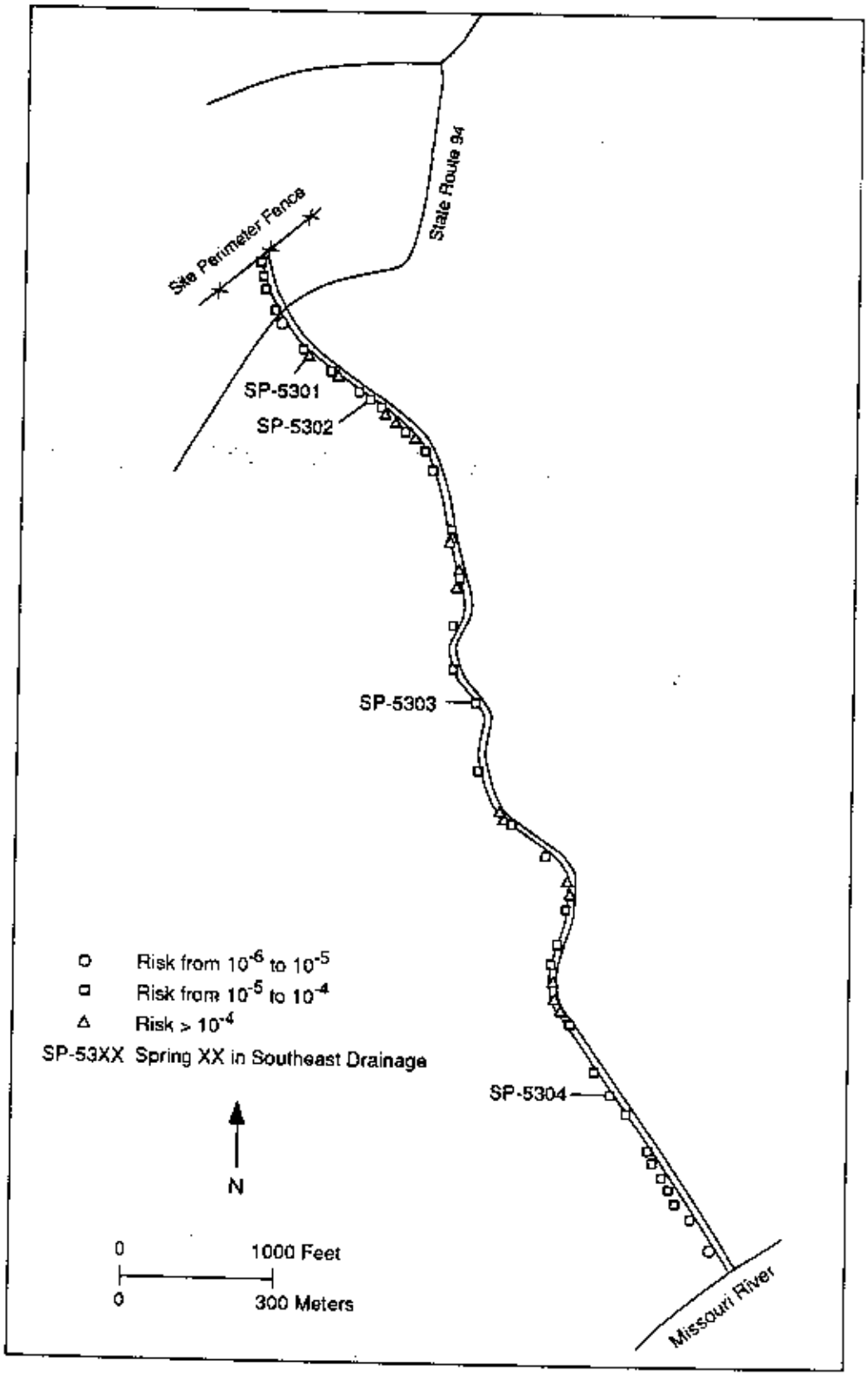


FIGURE 5.13 Estimated Radiological Risks from Exposure to Sediment in the Southeast Drainage

TABLE 5.1 Contaminant Groupings for Segregation of the Hazard Index[a]

Organ/System/Effect	Contaminant
Gastrointestinal tract	Arsenic
Liver	Antimony, arsenic, thallium, 2,4-DNT, 2,6-DNT, NB
Kidney	Arsenic, cadmium, uranium, NB
Spleen	DNB
Skin and hair	Arsenic, thallium
Cardiovascular system	Arsenic, barium
Nervous system	Arsenic, manganese
Hematopoietic system	Arsenic, nitrate, DNB, 2,4-DNT, 2,6-DNT, NB, TNB, TNT
Respiratory system	Chromium III, chromium VI, manganese
Developmental effects	Barium, PAHs
Nonspecific effects	Antimony, chromium III, chromium VI, molybdenum, nickel, vanadium

[a] This table presents information discussed in Section 5.1.2.2. Only those contaminants with an overall hazard quotient of 0.05 or greater are included.

TABLE 5.2 Estimated Radiological Risks for Exposure to Site Soil[a]

Exposure Route	Contaminant	Carcinogenic Risk		
		Current Worker	Current Trespasser	Future Recreational Visitor
Irradiation	External gamma	3×10^{-5}	8×10^{-8}	4×10^{-6}
Incidental ingestion	Actinium-227	5×10^{-6}	4×10^{-8}	8×10^{-7}
	Lead-210	2×10^{-5}	2×10^{-7}	3×10^{-6}
	Protactinium-231	8×10^{-6}	7×10^{-8}	1×10^{-6}
	Radium-226	3×10^{-6}	3×10^{-8}	5×10^{-7}
	Radium-228	3×10^{-6}	3×10^{-8}	5×10^{-7}
	Thorium-230	8×10^{-7}	7×10^{-9}	1×10^{-7}
	Thorium-232	5×10^{-6}	4×10^{-8}	7×10^{-7}
	Uranium-235	2×10^{-7}	2×10^{-9}	3×10^{-8}
	Uranium-238	1×10^{-5}	8×10^{-8}	1×10^{-6}
Total risk[b]		8×10^{-5}	6×10^{-7}	1×10^{-5}

[a] This table presents information discussed in Section 5.2.1.1.

[b] Incremental risk above background; corresponding risks including background external gamma irradiation are 2×10^{-4}, 8×10^{-7}, and 3×10^{-5} for a worker, trespasser, and recreational visitor, respectively.

TABLE 5.3 Estimated Chemical Carcinogenic Risks for Incidental Ingestion of Site Soil[a]

	Carcinogenic Risk		
Contaminant[b]	Current Worker	Current Trespasser	Future Recreational Visitor
Metals			
Arsenic	6×10^{-6}	7×10^{-8}	8×10^{-7}
Beryllium	9×10^{-7}	1×10^{-8}	1×10^{-7}
PAHs[c]	2×10^{-6}	3×10^{-8}	3×10^{-7}
PCBs	4×10^{-6}	5×10^{-8}	6×10^{-7}
Nitroaromatic compounds			
2,4-DNT	2×10^{-9}	2×10^{-11}	3×10^{-10}
2,6-DNT	4×10^{-9}	5×10^{-11}	6×10^{-10}
TNT	2×10^{-10}	2×10^{-12}	2×10^{-11}
Total risk	1×10^{-5}	2×10^{-7}	2×10^{-6}

[a] This table presents information discussed in Section 5.2.1.2.

[b] Listed are only those contaminants for which an EPA slope factor is available.

[c] The carcinogenic PAHs are benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, and indeno(1,2,3-cd)pyrene.

TABLE 5.4 Estimated Hazard Quotients for Incidental Ingestion of Site Soil[a]

Contaminant[b]	Hazard Quotient		
	Current Worker	Current Trespasser	Future Recreational Visitor
Metals			
Antimony	2×10^{-2}	3×10^{-4}	1×10^{-3}
Arsenic	7×10^{-2}	9×10^{-4}	4×10^{-3}
Barium	6×10^{-3}	7×10^{-5}	3×10^{-4}
Beryllium	3×10^{-4}	3×10^{-6}	1×10^{-5}
Cadmium	2×10^{-3}	2×10^{-5}	9×10^{-5}
Chromium III	4×10^{-5}	5×10^{-7}	2×10^{-6}
Chromium VI	9×10^{-4}	1×10^{-5}	5×10^{-5}
Copper	8×10^{-4}	9×10^{-6}	4×10^{-5}
Lithium	7×10^{-4}	9×10^{-6}	4×10^{-5}
Manganese	1×10^{-2}	1×10^{-4}	6×10^{-4}
Mercury	7×10^{-4}	8×10^{-6}	3×10^{-5}
Molybdenum	9×10^{-3}	1×10^{-4}	4×10^{-4}
Nickel	2×10^{-3}	2×10^{-5}	1×10^{-4}
Selenium	1×10^{-3}	1×10^{-5}	5×10^{-5}
Silver	4×10^{-4}	5×10^{-6}	2×10^{-5}
Thallium	3×10^{-2}	3×10^{-4}	1×10^{-3}
Uranium	1×10^{-1}	1×10^{-3}	6×10^{-3}
Vanadium	9×10^{-3}	1×10^{-4}	5×10^{-4}
Zinc	6×10^{-4}	7×10^{-6}	3×10^{-5}
Inorganic anions			
Fluoride	2×10^{-4}	2×10^{-6}	9×10^{-6}
Nitrate	5×10^{-5}	6×10^{-7}	3×10^{-6}
Nitrite	7×10^{-6}	8×10^{-8}	4×10^{-7}
PAHs			
Acenaphthene	1×10^{-6}	2×10^{-8}	7×10^{-8}
Anthracene	5×10^{-7}	6×10^{-9}	2×10^{-8}
Benz(a)anthracene	1×10^{-5}	1×10^{-7}	6×10^{-7}
Benzo(b)fluoranthene	6×10^{-6}	7×10^{-8}	3×10^{-7}
Benzo(k)fluoranthene	5×10^{-6}	6×10^{-8}	3×10^{-7}
Benzo(g,h,i)perylene	3×10^{-6}	3×10^{-8}	1×10^{-7}
Benzo(a)pyrene	7×10^{-6}	8×10^{-8}	4×10^{-7}
Chrysene	1×10^{-5}	1×10^{-7}	6×10^{-7}
Fluoranthene	1×10^{-5}	1×10^{-7}	6×10^{-7}
Fluorene	2×10^{-6}	2×10^{-8}	8×10^{-8}
Indeno(1,2,3-cd)pyrene	4×10^{-6}	5×10^{-8}	2×10^{-7}
2-Methylnaphthalene	5×10^{-5}	6×10^{-7}	3×10^{-6}
Naphthalene	2×10^{-5}	2×10^{-7}	9×10^{-7}
Phenanthrene	2×10^{-5}	2×10^{-7}	9×10^{-7}
Pyrene	3×10^{-5}	3×10^{-7}	1×10^{-6}

TABLE 5.4 (Cont.)

Contaminant[b]	Hazard Quotient		
	Current Worker	Current Trespasser	Future Recreational Visitor
PCBs	4×10^{-2}	4×10^{-4}	2×10^{-3}
Nitroaromatic compounds			
DNB	1×10^{-3}	1×10^{-5}	5×10^{-5}
2,4-DNT	1×10^{-4}	1×10^{-6}	5×10^{-6}
2,6-DNT	1×10^{-5}	1×10^{-7}	5×10^{-7}
NB	8×10^{-5}	1×10^{-6}	4×10^{-6}
TNB	1×10^{-3}	2×10^{-5}	7×10^{-5}
TNT	7×10^{-5}	8×10^{-7}	4×10^{-6}
Hazard index	0.3	0.004	0.02

[a] This table presents information discussed in Section 5.2.1.2.

[b] Listed are only those contaminants for which an EPA oral reference dose is available.

TABLE 5.5 Estimated Radiological Risks for Incidental Ingestion of Raffinate Pit Surface Water and Sludge[a]

	Carcinogenic Risk from Surface Water		Carcinogenic Risk from Sludge	
Contaminant	Current Trespasser	Future Recreational Visitor	Current Trespasser	Future Recreational Visitor
Actinium-227	NQ[b]	NQ	4×10^{-5}	7×10^{-4}
Lead-210	NQ	NQ	2×10^{-5}	3×10^{-4}
Protactinium-231	NQ	NQ	3×10^{-5}	6×10^{-4}
Radium-226	8×10^{-7}	1×10^{-5}	5×10^{-6}	8×10^{-5}
Radium-228	3×10^{-7}	4×10^{-6}	7×10^{-6}	1×10^{-4}
Thorium-230	2×10^{-6}	3×10^{-5}	4×10^{-5}	8×10^{-4}
Thorium-232	1×10^{-7}	2×10^{-6}	1×10^{-5}	2×10^{-4}
Uranium-235	9×10^{-8}	1×10^{-6}	5×10^{-8}	8×10^{-7}
Uranium-238	4×10^{-6}	5×10^{-5}	2×10^{-6}	4×10^{-5}
Total risk	8×10^{-6}	1×10^{-4}	2×10^{-4}	3×10^{-3}

[a] This table presents information discussed in Section 5.2.2.1.

[b] NQ = not quantified because no data were available from which to determine the concentrations of these radionuclides in surface water.

TABLE 5.6 Estimated Chemical Carcinogenic Risks for Incidental Ingestion of Raffinate Pit Surface Water and Sludge[a]

	Carcinogenic Risk from Surface Water		Carcinogenic Risk from Sludge	
Contaminant[b]	Current Trespasser	Future Recreational Visitor	Current Trespasser	Future Recreational Visitor
Metals				
Arsenic	2×10^{-6}	1×10^{-5}	6×10^{-6}	8×10^{-5}
Beryllium	3×10^{-7}	2×10^{-6}	3×10^{-7}	4×10^{-6}
PCBs	-[c]	-	4×10^{-7}	4×10^{-6}
Total risk	2×10^{-6}	2×10^{-5}	7×10^{-6}	8×10^{-5}

[a] This table presents information discussed in Section 5.2.2.2.

[b] Listed are only those contaminants for which an EPA oral slope factor is available.

[c] A hyphen indicates that the analyte was not detected so no risk value was estimated.

TABLE 5.7 Estimated Hazard Quotients for Incidental Ingestion of Raffinate Pit Surface Water and Sludge[a]

	Hazard Quotient for Surface Water		Hazard Quotient for Sludge	
Contaminant[b]	Current Trespasser	Future Recreational Visitor	Current Trespasser	Future Recreational Visitor
Metals				
Antimony	5×10^{-2}	2×10^{-1}	5×10^{-3}	2×10^{-2}
Arsenic	2×10^{-2}	6×10^{-2}	8×10^{-2}	3×10^{-1}
Barium	-[c]	-	2×10^{-3}	1×10^{-2}
Beryllium	9×10^{-5}	3×10^{-4}	1×10^{-4}	5×10^{-4}
Cadmium	2×10^{-3}	6×10^{-3}	3×10^{-4}	1×10^{-3}
Chromium III	9×10^{-6}	3×10^{-5}	3×10^{-6}	1×10^{-5}
Chromium VI	2×10^{-4}	6×10^{-4}	7×10^{-5}	3×10^{-4}
Copper	6×10^{-5}	2×10^{-4}	3×10^{-4}	1×10^{-3}
Lithium	1×10^{-2}	3×10^{-2}	1×10^{-4}	6×10^{-4}
Manganese	2×10^{-5}	5×10^{-5}	7×10^{-4}	3×10^{-3}
Mercury	5×10^{-5}	2×10^{-4}	1×10^{-3}	5×10^{-3}
Molybdenum	6×10^{-2}	2×10^{-1}	9×10^{-3}	4×10^{-2}
Nickel	5×10^{-4}	1×10^{-3}	1×10^{-2}	4×10^{-2}
Selenium	2×10^{-3}	7×10^{-3}	4×10^{-4}	2×10^{-3}
Silver	4×10^{-4}	1×10^{-3}	2×10^{-5}	9×10^{-5}
Thallium	-	-	2×10^{-2}	8×10^{-2}
Uranium	7×10^{-2}	2×10^{-1}	4×10^{-2}	2×10^{-1}
Vanadium	2×10^{-2}	5×10^{-2}	3×10^{-2}	1×10^{-1}
Zinc	2×10^{-5}	5×10^{-5}	2×10^{-4}	8×10^{-4}
Inorganic anions				
Fluoride	2×10^{-2}	5×10^{-2}	6×10^{-5}	3×10^{-4}
Nitrate	2×10^{-1}	7×10^{-1}	2×10^{-3}	9×10^{-3}
Nitrite	-	-	4×10^{-4}	2×10^{-3}
PCBs	-	-	3×10^{-3}	1×10^{-2}
Hazard index	0.5	1	0.2	0.8

[a] This table presents information discussed in Section 5.2.2.2.

[b] Listed are only those contaminants for which an EPA oral reference dose is available.

[c] A hyphen indicates that the analyte was not detected or not assayed so no hazard quotient was estimated.

TABLE 5.8 Estimated Radiological Risks for Inhalation of Airborne Particulates On-Site[a]

	Carcinogenic Risk		
Contaminant	Current Worker	Current Trespasser	Future Recreational Visitor
Actinium-227	4×10^{-6}	1×10^{-8}	6×10^{-7}
Lead-210	2×10^{-8}	6×10^{-11}	3×10^{-9}
Protactinium-231	2×10^{-6}	5×10^{-9}	2×10^{-7}
Radium-226	9×10^{-9}	2×10^{-11}	1×10^{-9}
Radium-228	3×10^{-7}	9×10^{-10}	4×10^{-8}
Radon-222[b]	4×10^{-4}	1×10^{-6}	5×10^{-5}
Thorium-230	3×10^{-7}	8×10^{-10}	4×10^{-8}
Thorium-232	2×10^{-6}	5×10^{-9}	2×10^{-7}
Uranium-235	2×10^{-7}	5×10^{-10}	3×10^{-8}
Uranium-238	9×10^{-6}	2×10^{-8}	1×10^{-6}
Total risk	4×10^{-4}	1×10^{-6}	5×10^{-5}

[a] This table presents information discussed in Section 5.2.3.1.

[b] Although radon-222 is a gas, its particulate decay products are the primary contributors to the radiological risk.

TABLE 5.9 Estimated Chemical Carcinogenic Risks for Inhalation of Airborne Particulates On-Site[a]

	Carcinogenic Risk		
Contaminant[b]	Current Worker	Current Trespasser	Future Recreational Visitor
Metals			
Arsenic	2×10^{-7}	6×10^{-10}	2×10^{-8}
Beryllium	5×10^{-9}	2×10^{-11}	6×10^{-10}
Cadmium	4×10^{-9}	1×10^{-11}	5×10^{-10}
Chromium VI	5×10^{-8}	2×10^{-10}	6×10^{-9}
Nickel	2×10^{-8}	6×10^{-11}	2×10^{-9}
Total risk	2×10^{-7}	9×10^{-10}	3×10^{-8}

[a] This table presents information discussed in Section 5.2.3.2.

[b] Listed are only those contaminants for which an EPA inhalation slope factor is available.

TABLE 5.10 Estimated Hazard Quotients for Inhalation of Airborne Particulates On-Site[a]

	Hazard Quotient		
Contaminant[b]	Current Worker	Current Trespasser	Future Recreational Visitor
Metals			
Barium	7×10^{-3}	3×10^{-5}	3×10^{-4}
Cadmium	2×10^{-5}	8×10^{-8}	9×10^{-7}
Chromium III	1×10^{-1}	5×10^{-4}	5×10^{-3}
Chromium VI	1×10^{-2}	5×10^{-5}	6×10^{-4}
Manganese	4×10^{-2}	1×10^{-4}	1×10^{-3}
Mercury	9×10^{-6}	3×10^{-8}	4×10^{-7}
Hazard index	0.2	0.0007	0.008

[a] This table presents information discussed in Section 5.2.3.2.

[b] Listed are only those contaminants for which an EPA inhalation reference dose is available.

TABLE 5.11 Estimated Radiological Risks for Ingestion of Game[a]

Contaminant	Carcinogenic Risk						
	Duck (Measured)	Duck (Modeled)	Goose (Measured)	Goose (Modeled)	Turkey (Modeled)	Deer (Measured)	Deer (Modeled)
Actinium-227	-[b]	3×10^{-10}	-	2×10^{-10}	5×10^{-9}	-	9×10^{-9}
Lead-210	-	2×10^{-10}	-	1×10^{-10}	7×10^{-10}	-	2×10^{-9}
Protactinium-231	-	5×10^{-10}	-	3×10^{-10}	7×10^{-9}	-	1×10^{-8}
Radium-226	3×10^{-9}	3×10^{-8}	2×10^{-9}	2×10^{-8}	6×10^{-9}	8×10^{-8}	2×10^{-8}
Radium-228	7×10^{-8}	2×10^{-8}	1×10^{-8}	1×10^{-8}	6×10^{-9}	1×10^{-7}	2×10^{-8}
Thorium-230	1×10^{-8}	6×10^{-8}	8×10^{-9}	4×10^{-8}	2×10^{-9}	4×10^{-8}	7×10^{-9}
Thorium-232	2×10^{-8}	7×10^{-9}	4×10^{-9}	5×10^{-9}	1×10^{-8}	7×10^{-8}	4×10^{-8}
Uranium-235	6×10^{-10}	1×10^{-8}	4×10^{-10}	9×10^{-9}	2×10^{-10}	6×10^{-9}	3×10^{-10}
Uranium-238	5×10^{-9}	3×10^{-7}	2×10^{-9}	2×10^{-7}	9×10^{-9}	2×10^{-8}	2×10^{-8}
Total risk	1×10^{-7}	4×10^{-7}	3×10^{-8}	3×10^{-7}	5×10^{-8}	3×10^{-7}	1×10^{-7}

[a] This table presents information discussed in Section 5.2.4.1.

[b] A hyphen indicates that the radionuclide was not measured.

TABLE 5.12 Estimated Chemical Carcinogenic Risks for Ingestion of Game[a]

Contaminant[b]	Carcinogenic Risk		
	Duck	Turkey	Deer
Metals			
Arsenic	2×10^{-9}	1×10^{-8}	2×10^{-8}
Beryllium	4×10^{-10}	1×10^{-9}	6×10^{-10}
PAHs[c]	-[d]	6×10^{-10}	9×10^{-10}
PCBs	-	9×10^{-10}	6×10^{-10}
Nitroaromatic compounds			
2,4-DNT	-	6×10^{-13}	3×10^{-12}
2,6-DNT	-	1×10^{-12}	5×10^{-12}
TNT	-	4×10^{-14}	2×10^{-13}
Total risk	2×10^{-9}	2×10^{-8}	2×10^{-8}

[a] This table presents information discussed in Section 5.2.4.2. For the goose, the estimated risks are two-thirds of the values shown for the duck.

[b] Listed are only those contaminants for which an EPA oral slope factor is available.

[c] The carcinogenic PAHs are benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, chrysene, and indeno(1,2,3-cd)pyrene.

[d] A hyphen indicates that the risk was not estimated because the analyte was not measured in the raffinate pit water during site characterization, so no values were reported in the database.

TABLE 5.13 Estimated Hazard Quotients for Ingestion of Game[a]

Contaminant[b]	Hazard Quotient		
	Duck	Turkey	Deer
Metals			
Antimony	1×10^{-5}	3×10^{-5}	9×10^{-5}
Arsenic	5×10^{-6}	6×10^{-5}	8×10^{-5}
Barium	-[c]	5×10^{-7}	1×10^{-6}
Beryllium	4×10^{-8}	1×10^{-7}	6×10^{-8}
Cadmium	2×10^{-7}	2×10^{-6}	6×10^{-6}
Chromium III	4×10^{-9}	4×10^{-8}	5×10^{-8}
Chromium VI	9×10^{-8}	9×10^{-7}	1×10^{-6}
Copper	2×10^{-7}	6×10^{-5}	2×10^{-4}
Lithium	1×10^{-5}	4×10^{-6}	4×10^{-6}
Manganese	2×10^{-9}	1×10^{-5}	3×10^{-5}
Mercury	9×10^{-9}	2×10^{-6}	7×10^{-6}
Molybdenum	1×10^{-4}	1×10^{-4}	5×10^{-4}
Nickel	3×10^{-7}	9×10^{-6}	3×10^{-5}
Selenium	6×10^{-6}	2×10^{-5}	5×10^{-5}
Silver	3×10^{-7}	3×10^{-6}	1×10^{-5}
Thallium	-	5×10^{-4}	6×10^{-5}
Uranium	6×10^{-5}	2×10^{-3}	6×10^{-3}
Vanadium	1×10^{-5}	1×10^{-5}	1×10^{-5}
Zinc	6×10^{-7}	2×10^{-3}	7×10^{-3}
Inorganic anions			
Fluoride	2×10^{-4}	2×10^{-5}	2×10^{-5}
Nitrate	NQ[c]	NQ	NQ
Nitrite	NQ	NQ	NQ
PAHs			
Acenaphthene	-	1×10^{-11}	5×10^{-11}
Anthracene	-	6×10^{-12}	2×10^{-11}
Benz(a)anthracene	-	8×10^{-10}	2×10^{-9}
Benzo(b)fluoranthene	-	6×10^{-10}	1×10^{-9}
Benzo(k)fluoranthene	-	5×10^{-10}	8×10^{-10}
Benzo(g,h,i)perylene	-	7×10^{-10}	7×10^{-10}
Benzo(a)pyrene	-	4×10^{-10}	1×10^{-10}
Chrysene	-	8×10^{-10}	2×10^{-9}
Fluoranthene	-	4×10^{-10}	1×10^{-9}
Fluorene	-	2×10^{-11}	8×10^{-11}
Indeno(1,2,3-cd)pyrene	-	1×10^{-9}	1×10^{-9}
2-Methylnaphthalene	-	7×10^{-9}	3×10^{-8}
Naphthalene	-	2×10^{-10}	6×10^{-10}
Phenanthrene	-	9×10^{-10}	3×10^{-9}
Pyrene	-	8×10^{-10}	2×10^{-9}

TABLE 5.13 (Cont.)

Contaminant[b]	Hazard Quotient		
	Duck	Turkey	Deer
PCBs	-	3×10^{-6}	2×10^{-6}
Nitroaromatic compounds			
DNB	-	1×10^{-7}	5×10^{-7}
2,4-DNT	-	1×10^{-8}	4×10^{-8}
2,6-DNT	-	1×10^{-9}	4×10^{-9}
NB	-	8×10^{-9}	3×10^{-8}
TNB	-	2×10^{-7}	6×10^{-7}
TNT	-	7×10^{-9}	3×10^{-8}
Hazard index	0.0004	0.004	0.01

[a] This table presents information discussed in Section 5.2.4.2. For the goose, the estimated values are two-thirds of those shown for the duck.

[b] Listed are only those contaminants for which an EPA oral reference dose is available.

[c] A hyphen indicates that the hazard quotient was not estimated because the analyte was not measured in the raffinate pit water during site characterization, so no values were reported in the database. NQ indicates that the hazard quotient for the contaminant was not quantified (see text).

TABLE 5.14 Estimated Radiological Risks for Exposures in the Buildings[a]

Exposure Route	Contaminant	Carcinogenic Risk[b]	
		Current Trespasser	Future Recreational Visitor
Irradiation	External gamma	3×10^{-6}	3×10^{-5}
Inhalation	Radon-220	4×10^{-5}	5×10^{-4}
	Radon-222	2×10^{-7}	2×10^{-6}
	Uranium-235	2×10^{-7}	2×10^{-6}
	Uranium-238	8×10^{-6}	1×10^{-4}
Incidental ingestion	Radium-226	2×10^{-6}	3×10^{-5}
	Thorium-232	3×10^{-5}	6×10^{-4}
	Uranium-235	3×10^{-7}	6×10^{-6}
	Uranium-238	1×10^{-5}	3×10^{-4}
Total risk		1×10^{-4}	1×10^{-3}

[a] This table presents information discussed in Section 5.2.5.1.

[b] The risks for radon-220 and radon-222 include the contributions from their short-lived particulate decay products.

TABLE 5.15 Estimated Chemical Carcinogenic Risks and Hazard Quotients for Exposures in the Buildings[a]

Exposure Route	Contaminant	Carcinogenic Risk		Hazard Quotient	
		Current Trespasser	Future Recreational Visitor	Current Trespasser	Future Recreational Visitor
Inhalation	Asbestos	2×10^{-6}	2×10^{-5}	NQ[b]	NQ
Incidental ingestion	PCBs	3×10^{-4}	3×10^{-3}	3	9
Total risk or hazard index		3×10^{-4}	3×10^{-3}	3	9

[a] This table presents information discussed in Section 5.2.5.2.

[b] NQ indicates that the hazard quotient for that contaminant could not be quantified because an EPA inhalation reference dose is not available.

TABLE 5.16 Total Radiological Risks, Chemical Carcinogenic Risks, and Hazard Indexes for the On-Site Worker[a]

Medium	Exposure Route	Radiological Risk	Chemical Risk	Hazard Index
Soil	External gamma	3×10^{-5}	NA[b]	NA
	Incidental ingestion	6×10^{-5}	1×10^{-5}	0.3
Air	Inhalation[c]	4×10^{-4}	2×10^{-7}	0.2
Total risk or hazard index		5×10^{-4}	1×10^{-5}	0.5

[a] This table presents information discussed in Section 5.2.6.

[b] NA = not applicable.

[c] For radiological risk, includes inhalation of radon and airborne particulates.

TABLE 5.17 Total Radiological Risks for the On-Site Trespasser from Single and Multiple Exposure Pathways[a]

	Carcinogenic Risk						
Exposure Scenario[b]	Outdoor Air	Soil	Surface Water	Sludge	Building Air	Building Residues[c]	Total Risk
T1	1×10^{-6}	-[d]	-	-	-	-	1×10^{-6}
T2	-	6×10^{-7}	-	-	-	-	6×10^{-7}
T3	-	-	8×10^{-6}	-	-	-	8×10^{-6}
T4	-	-	-	2×10^{-4}	-	-	2×10^{-4}
T5	-	-	-	-	5×10^{-5}	-	5×10^{-5}
T6	-	-	-	-	-	5×10^{-5}	5×10^{-5}
T7	1×10^{-6}	6×10^{-7}	-	-	-	-	2×10^{-6}
T8	1×10^{-6}	-	8×10^{-6}	2×10^{-4}	-	-	2×10^{-4}
T9	-	-	-	-	5×10^{-5}	5×10^{-5}	1×10^{-4}
T10	1×10^{-6}	3×10^{-7}	4×10^{-6}	8×10^{-5}	-	-	8×10^{-5}
T11	5×10^{-7}	3×10^{-7}	-	-	2×10^{-5}	3×10^{-5}	5×10^{-5}
T12	7×10^{-7}	2×10^{-7}	3×10^{-6}	5×10^{-5}	2×10^{-5}	2×10^{-5}	9×10^{-5}

[a] This table presents information discussed in Section 5.2.6.

[b] The individual exposure scenarios are defined in the text.

[c] Includes risk from exposure to external gamma radiation.

[d] A hyphen indicates that the exposure pathway was not considered for that scenario.

TABLE 5.18 Total Radiological Risks for the On-Site Recreational Visitor from Single and Multiple Exposure Pathways[a]

	Carcinogenic Risk						
Exposure Scenario[b]	Outdoor Air	Soil	Surface Water	Sludge	Building Air	Building Residues[c]	Total Risk
R1	5×10^{-5}	-[d]	-	-	-	-	5×10^{-5}
R2	-	1×10^{-5}	-	-	-	-	1×10^{-5}
R3	-	-	1×10^{-4}	-	-	-	1×10^{-4}
R4	-	-	-	3×10^{-3}	-	-	3×10^{-3}
R5	-	-	-	-	6×10^{-4}	-	6×10^{-4}
R6	-	-	-	-	-	9×10^{-4}	9×10^{-4}
R7	5×10^{-5}	1×10^{-5}	-	-	-	-	6×10^{-5}
R8	5×10^{-5}	-	1×10^{-4}	3×10^{-3}	-	-	3×10^{-3}
R9	-	-	-	-	6×10^{-4}	9×10^{-4}	1×10^{-3}
R10	5×10^{-5}	6×10^{-6}	5×10^{-5}	1×10^{-3}	-	-	1×10^{-3}
R11	3×10^{-5}	6×10^{-6}	-	-	3×10^{-4}	5×10^{-4}	8×10^{-4}
R12	3×10^{-5}	4×10^{-6}	3×10^{-5}	9×10^{-4}	2×10^{-4}	3×10^{-4}	1×10^{-3}

[a] This table presents information discussed in Section 5.2.6.

[b] The individual exposure scenarios are defined in the text.

[c] Includes risk from exposure to external gamma radiation.

[d] A hyphen indicates that the exposure pathway was not considered for that scenario.

TABLE 5.19 Total Chemical Carcinogenic Risks for the On-Site Trespasser from Single and Multiple Exposure Pathways[a]

	Carcinogenic Risk						
Exposure Scenario[b]	Outdoor Air	Soil	Surface Water	Sludge	Building Air	Building Residues	Total Risk
T1	9×10^{-10}	-[c]	-	-	-	-	9×10^{-10}
T2	-	2×10^{-7}	-	-	-	-	2×10^{-7}
T3	-	-	2×10^{-6}	-	-	-	2×10^{-6}
T4	-	-	-	7×10^{-6}	-	-	7×10^{-6}
T5	-	-	-	-	2×10^{-6}	-	2×10^{-6}
T6	-	-	-	-	-	3×10^{-4}	3×10^{-4}
T7	9×10^{-10}	2×10^{-7}	-	-	-	-	2×10^{-7}
T8	9×10^{-10}	-	2×10^{-6}	7×10^{-6}	-	-	9×10^{-6}
T9	-	-	-	-	2×10^{-6}	3×10^{-4}	3×10^{-4}
T10	9×10^{-10}	8×10^{-8}	1×10^{-6}	3×10^{-6}	-	-	4×10^{-6}
T11	5×10^{-10}	8×10^{-8}	-	-	9×10^{-7}	2×10^{-4}	2×10^{-4}
T12	6×10^{-10}	5×10^{-8}	6×10^{-7}	2×10^{-6}	6×10^{-7}	1×10^{-4}	1×10^{-4}

[a] This table presents information discussed in Section 5.2.6.

[b] The individual exposure scenarios are defined in the text.

[c] A hyphen indicates that the exposure pathway was not considered for that scenario.

TABLE 5.20 Total Hazard Indexes for the On-Site Trespasser from Single and Multiple Exposure Pathways[a]

	Hazard Index					
Exposure Scenario[b]	Outdoor Air	Soil	Surface Water	Sludge	Building Residues	Total Hazard Index
T1	0.0007	-[c]	-	-	-	0.0007
T2	-	0.004	-	-	-	0.004
T3	-	-	0.5	-	-	0.5
T4	-	-	-	0.2	-	0.2
T6	-	-	-	-	3	3
T7	0.0007	0.004	-	-	-	0.004
T8	0.0007	-	0.5	0.2	-	0.7
T10	0.0007	0.002	0.3	0.1	-	0.4
T11	0.0004	0.002	-	-	2	2
T12	0.0005	0.001	0.2	0.07	1	1

[a] This table presents information discussed in Section 5.2.6.

[b] The individual exposure scenarios are defined in the text. No hazard index was estimated for building air (and therefore Scenarios T5 and T9 are not included in this table) because no EPA reference dose is available for the building contaminants.

[c] A hyphen indicates that the exposure pathway was not considered for that scenario.

TABLE 5.21 Total Chemical Carcinogenic Risks for the On-Site Recreational Visitor from Single and Multiple Exposure Pathways[a]

	Carcinogenic Risk						
Exposure Scenario[b]	Outdoor Air	Soil	Surface Water	Sludge	Building Air	Building Residues	Total Risk
R1	3×10^{-8}	-[c]	-	-	-	-	3×10^{-8}
R2	-	2×10^{-6}	-	-	-	-	2×10^{-6}
R3	-	-	2×10^{-5}	-	-	-	2×10^{-5}
R4	-	-	-	8×10^{-5}	-	-	8×10^{-5}
R5	-	-	-	-	2×10^{-5}	-	2×10^{-5}
R6	-	-	-	-	-	3×10^{-3}	3×10^{-3}
R7	3×10^{-8}	2×10^{-6}	-	-	-	-	2×10^{-6}
R8	3×10^{-8}	-	2×10^{-5}	8×10^{-5}	-	-	1×10^{-4}
R9	-	-	-	-	2×10^{-5}	3×10^{-3}	3×10^{-3}
R10	3×10^{-8}	1×10^{-6}	8×10^{-6}	4×10^{-5}	-	-	5×10^{-5}
R11	2×10^{-8}	1×10^{-6}	-	-	8×10^{-6}	2×10^{-3}	2×10^{-3}
R12	2×10^{-8}	1×10^{-6}	6×10^{-6}	3×10^{-5}	5×10^{-6}	1×10^{-3}	1×10^{-3}

[a] This table presents information discussed in Section 5.2.6.

[b] The individual exposure scenarios are defined in the text.

[c] A hyphen indicates that the exposure pathway was not considered for that scenario.

TABLE 5.22 Total Hazard Indexes for the On-Site Recreational Visitor from Single and Multiple Exposure Pathways[a]

	Hazard Index					
Exposure Scenario[b]	Outdoor Air	Soil	Surface Water	Sludge	Building Residues	Total Hazard Index
R1	0.008	-[c]	-	-	-	0.008
R2	-	0.02	-	-	-	0.02
R3	-	-	1	-	-	1
R4	-	-	-	0.8	-	0.8
R6	-	-	-	-	9	9
R7	0.008	0.02	-	-	-	0.02
R8	0.008	-	1	0.8	-	2
R10	0.008	0.01	0.7	0.4	-	1
R11	0.004	0.01	-	-	5	5
R12	0.005	0.01	0.5	0.3	3	4

[a] This table presents information discussed in Section 5.2.6.

[b] The individual exposure scenarios are defined in the text. No hazard index was estimated for building air (and therefore Scenarios R5 and R9 are not included in this table) because no EPA inhalation reference dose is available for the building contaminants.

[c] A hyphen indicates that the exposure pathway was not considered for that scenario.

TABLE 5.23 Estimated Radiological Risks for the Recreational Visitor from Exposure to Surface Water and Sediment at Off-Site Locations[a]

Source/Medium	Exposure Route	Contaminant	Carcinogenic Risk
Lakes 34, 35, and 36			
Surface water	Ingestion	Uranium-235	1×10^{-7}
		Uranium-238	4×10^{-6}
Sediment	Irradiation	External gamma[b]	8×10^{-7}
	Incidental ingestion	Uranium-235	6×10^{-8}
		Uranium-238	3×10^{-6}
Burgermeister Spring			
Surface water	Ingestion	Thorium-230	3×10^{-7}
		Uranium-235	1×10^{-7}
		Uranium-238	4×10^{-6}
Southeast Drainage			
Surface water	Ingestion	Uranium-235	3×10^{-7}
		Uranium-238	1×10^{-5}
Sediment	Irradiation	External gamma[c]	7×10^{-6}
	Incidental ingestion	Radium-226	5×10^{-6}
		Thorium-230	2×10^{-4}
		Thorium-232	2×10^{-6}
		Uranium-235	4×10^{-8}
		Uranium-238	2×10^{-6}

[a] This table presents information discussed in Sections 5.3.1.1, 5.3.3.1, and 5.3.4.1.

[b] Estimated from radionuclide concentrations measured in sediment/shoreline soil at the lakes.

[c] Estimated from measured exposure rates, excluding background; the related risk from external gamma irradiation with background included is 1×10^{-5}.

TABLE 5.24 Estimated Chemical Carcinogenic Risks for the Recreational Visitor from Ingestion of Surface Water at Off-Site Locations[a]

Contaminant[b]	Carcinogenic Risk		
	Lakes 34, 35, and 36	Burgermeister Spring	Southeast Drainage
Metals			
Arsenic	4×10^{-6}	-[c]	-
Beryllium	-	-	-
Nitroaromatic compounds			
2,4-DNT	3×10^{-8}	7×10^{-8}	5×10^{-7}
2,6-DNT	-	8×10^{-7}	5×10^{-7}
TNT	-	6×10^{-10}	2×10^{-7}
Total risk	4×10^{-6}	9×10^{-7}	1×10^{-6}

[a] This table presents information discussed in Sections 5.3.1.2, 5.3.3.2, and 5.3.4.2.

[b] Listed are only those contaminants for which an EPA oral slope factor is available.

[c] A hyphen indicates that the analyte was not detected so no risk was estimated.

TABLE 5.25 Estimated Hazard Quotients for the Recreational Visitor from Ingestion of Surface Water at Off-Site Locations[a]

Contaminant[b]	Hazard Quotient		
	Lakes 34, 35, and 36	Burgermeister Spring	Southeast Drainage
Metals			
Antimony	-[c]	-	3×10^{-2}
Arsenic	2×10^{-2}	-	-
Barium	-	2×10^{-4}	2×10^{-4}
Beryllium	-	-	-
Cadmium	-	-	-
Chromium III	3×10^{-6}	4×10^{-6}	4×10^{-6}
Chromium VI	6×10^{-5}	8×10^{-5}	8×10^{-5}
Copper	-	-	-
Lithium	-	-	-
Manganese	1×10^{-3}	6×10^{-5}	9×10^{-5}
Mercury	7×10^{-4}	2×10^{-4}	-
Molybdenum	-	9×10^{-4}	2×10^{-3}
Nickel	-	-	-
Selenium	-	-	-
Silver	-	-	2×10^{-4}
Thallium	7×10^{-2}	-	-
Uranium	2×10^{-2}	2×10^{-2}	5×10^{-2}
Vanadium	-	-	-
Zinc	6×10^{-5}	2×10^{-5}	2×10^{-5}
Inorganic anions			
Fluoride	2×10^{-3}	8×10^{-4}	2×10^{-3}
Nitrate	9×10^{-4}	2×10^{-2}	3×10^{-2}
Nitrite	-	-	6×10^{-5}
Nitroaromatic compounds			
DNB	-	3×10^{-4}	1×10^{-3}
2,4-DNT	5×10^{-4}	1×10^{-3}	9×10^{-3}
2,6-DNT	-	7×10^{-4}	4×10^{-4}
NB	-	-	3×10^{-4}
TNB	-	6×10^{-5}	3×10^{-3}
TNT	-	9×10^{-5}	3×10^{-2}
Hazard index	0.1	0.04	0.2

[a] This table presents information discussed in Sections 5.3.1.2, 5.3.3.2, and 5.3.4.2.

[b] Listed are only those contaminants for which an EPA oral reference dose is available.

[c] A hyphen indicates that the analyte was not detected so no hazard quotient was estimated.

TABLE 5.26 Estimated Chemical Carcinogenic Risks and Hazard Indexes for the Recreational Visitor from Incidental Ingestion of Sediment at Off-Site Locations[a]

	Carcinogenic Risk	
Contaminant[b]	Lakes 34, 35, and 36	Southeast Drainage
Metals		
Arsenic	7×10^{-7}	1×10^{-6}
Beryllium	-[c]	-
PCBs	6×10^{-8}	-
Total risk	8×10^{-7}	1×10^{-6}

	Hazard Quotient	
Contaminant[b]	Lakes 34, 35, and 36	Southeast Drainage
Metals		
Antimony	-[c]	-
Arsenic	3×10^{-3}	5×10^{-3}
Barium	2×10^{-4}	4×10^{-4}
Beryllium	-	-
Cadmium	-	-
Chromium III	1×10^{-6}	2×10^{-6}
Chromium VI	3×10^{-5}	5×10^{-5}
Copper	3×10^{-5}	4×10^{-4}
Lithium	-	-
Manganese	1×10^{-3}	6×10^{-3}
Mercury	-	-
Nickel	6×10^{-5}	3×10^{-4}
Selenium	-	-
Silver	-	-
Thallium	-	-
Uranium	1×10^{-2}	7×10^{-3}
Vanadium	4×10^{-4}	1×10^{-3}
Zinc	2×10^{-5}	1×10^{-4}
PCBs	2×10^{-4}	-
Hazard index	0.02	0.02

[a] This table presents information discussed in Sections 5.3.1.2 and 5.3.4.2.

[b] Listed are only those contaminants for which an EPA oral slope factor (for carcinogenic risk) or oral reference dose (for hazard quotient) is available.

[c] A hyphen indicates that the analyte was not detected so no health effect was estimated.

TABLE 5.27 Estimated Radiological Risks for Ingestion of Fish[a]

Fish Sample	Carcinogenic Risk	
	Lakes 34, 35, and 36	Lake 37
Whole fish[b]		
Catfish	5×10^{-5}	3×10^{-7}
Edible preparation[c]		
Catfish	7×10^{-8}	5×10^{-8}
Bass	5×10^{-8}	3×10^{-8}
Sunfish	2×10^{-7}	3×10^{-8}
Crappie	4×10^{-8}	-
Total risk (edible)	4×10^{-7}	1×10^{-7}

[a] This table presents information discussed in Section 5.3.2.1.

[b] Fish are not eaten whole (e.g., with scales and stomach contents), but this has been assumed in order to present a comparative analysis between whole catfish from Lakes 34, 35, and 36 (using the maximum detected value) and whole catfish from Lake 37, which is considered the background or control lake.

[c] For the edible fish preparations from Lakes 34, 35, and 36, fillet data were available for catfish and bass, and both fillet and fishcake data were available for sunfish and crappie. Fillet data were available for all but crappie from Lake 37, as indicated by a hyphen. Therefore, sunfish data were considered to represent crappie for this analysis, and the sunfish ingestion rate was doubled to account for the incremental fish consumption.

TABLE 5.28 Estimated Radiological Risks to the Recreational Visitor from Exposure to Surface Soil at Vicinity Properties[a]

Vicinity Property[b]	Carcinogenic Risk
A1	1×10^{-5}
A2	5×10^{-5}
A3	2×10^{-6}
A5	9×10^{-5}
A6	6×10^{-7}
B3	6×10^{-6}
B4	3×10^{-4}
B5	5×10^{-6}
B6	2×10^{-5}
B10	8×10^{-7}

[a] This table presents information discussed in Section 5.3.5.1.

[b] The locations of these vicinity properties are shown in Figure 1.7. These risks represent the combined risk from external gamma irradiation and incidental ingestion.

TABLE 5.29 Total Radiological Risks, Chemical Carcinogenic Risks, and Hazard Indexes for the Off-Site Recreational Visitor[a]

	Radiological Risk		
Location	Surface Water	Sediment[b]	Total
Lakes 34, 35, and 36	4×10^{-6}	3×10^{-6}	8×10^{-6}
Burgermeister Spring	4×10^{-6}	NQ[c]	4×10^{-6}
Southeast Drainage	1×10^{-5}	2×10^{-4}	2×10^{-4}
	Chemical Carcinogenic Risk		
Location	Surface Water	Sediment	Total
Lakes 34, 35, and 36	4×10^{-6}	8×10^{-7}	4×10^{-6}
Burgermeister Spring	9×10^{-7}	NQ[c]	9×10^{-7}
Southeast Drainage	1×10^{-6}	1×10^{-6}	2×10^{-6}
	Hazard Index		
Location	Surface Water	Sediment	Total
Lakes 34, 35, and 36	0.1	0.02	0.1
Burgermeister Spring	0.04	NQ[c]	0.04
Southeast Drainage	0.2	0.02	0.2

[a] This table presents information discussed in Section 5.3.6.

[b] Results for sediment also include the contribution from contaminated soil.

[c] NQ indicates that a carcinogenic risk or hazard index was not estimated for this medium at this location.

TABLE 5.30 Estimated Chemical Carcinogenic Risks and Hazard Quotients for Dermal Exposures[a]

Medium/ Contaminant	Current Worker		Current Trespasser		Recreational Visitor[b]	
	Risk	Hazard Quotient	Risk	Hazard Quotient	Risk	Hazard Quotient
Sitewide soil						
Cadmium	NA[c]	0.0001	NA	0.000005	NA	0.00002
PCBs	2×10^{-6}	0.02	8×10^{-8}	0.0007	9×10^{-7}	0.003
Raffinate pit sludge						
Cadmium	NQ[d]	NQ	NA	0.00008	NA	0.0003
PCBs	NQ	NQ	4×10^{-7}	0.004	5×10^{-6}	0.02
Building residues						
PCBs	NQ	NQ	3×10^{-5}	0.3	3×10^{-4}	1
Lake sediment						
PCBs	NQ	NQ	NQ	NQ	9×10^{-8}	0.0003

[a] This table presents information discussed in Section 5.5.1.

[b] The risks and hazard quotients estimated for sitewide soil, raffinate pit sludge, and building residues are for the future on-site recreational visitor, and those estimated for lake sediment are for the current/future off-site recreational visitor.

[c] NA indicates that the entry is not applicable because cadmium is not considered to be carcinogenic via the dermal route.

[d] NQ indicates that exposure to that medium was not quantified for the worker.

TABLE 5.31 Estimated Carcinogenic Risks and Hazard Indexes for the Swimmer[a]

Area/Medium	Exposure Route	Radiological Risk	Chemical Risk	Hazard Index
Raffinate pits				
Surface water	Incidental ingestion	4×10^{-7}	1×10^{-7}	0.03
	Dermal contact	2×10^{-4}	4×10^{-6}	0.08
Sludge	External gamma irradiation[b]	1×10^{-7}	NA[c]	NA
	Incidental ingestion	5×10^{-5}	1×10^{-6}	0.04
	Dermal contact	NQ[d]	9×10^{-8}	0.0008
Air	Inhalation	3×10^{-7}	2×10^{-10}	0.0001
Lakes 34, 35, and 36				
Surface water	Incidental ingestion	2×10^{-8}	2×10^{-8}	0.002
	Dermal contact	3×10^{-7}	6×10^{-9}	0.003
Sediment	External gamma irradiation[e]	8×10^{-10}	NA	NA
	Incidental ingestion	4×10^{-8}	1×10^{-8}	0.0007
	Dermal contact	NQ	2×10^{-9}	0.00001

[a] This table presents information discussed in Section 5.5.1.

[b] The incremental risk estimated for this route is from the underlying raffinate pit sludge. The exposure rate at the surface of the raffinate pits is estimated to be 20 μR/h.

[c] NA indicates that the entry is not applicable.

[d] NQ indicates that the risk was not quantified for this exposure route (see text).

[e] The incremental risk estimated for this route assumes a 15-minute exposure to contaminated sediment/shoreline soil (0.5 μR/h above background) per swimming event.

6 SUMMARY OF THE HUMAN HEALTH ASSESSMENT

Results of the human health assessment are summarized according to the four component steps in Sections 6.1 through 6.4. The conclusions of this assessment are discussed in Section 6.5.

6.1 CONTAMINANTS OF CONCERN

Radioactive and chemical contaminants of concern for the human health assessment at the Weldon Spring site are listed in Table 6.1. The contaminants were identified from those detected in site soil, surface water, sediment, sludge, and buildings, and they represent the major chemical classes present at the site. These contaminants include radionuclides, metals, inorganic anions, nitroaromatic compounds, PAHs, PCBs, and asbestos. Selection of the contaminants of concern was based on both the history of site operations and an evaluation of characterization data with respect to the distribution and concentration of contaminants in the various media at the site and the potential contribution of individual contaminants to overall health effects.

6.2 EXPOSURE ASSESSMENT

6.2.1 Contaminant Fate and Transport

The fate and transport of contaminants released into the environment at the site were evaluated to determine potential exposure points. Human exposures evaluated in the BA were those resulting from potential contact with sources and affected media within the site boundary and contaminated media at off-site areas impacted by transport from the site.

The principal source areas and media identified at the site are (1) chemical plant buildings; (2) surface water and sludge at the four raffinate pits; (3) surface water and sediment at Frog Pond and Ash Pond (conservatively represented by the raffinate pits in this assessment because the contaminant levels are much higher in the pits); and (4) contaminated soil at the North Dump, at the South Dump, at the coal storage area, around certain chemical plant buildings, and at other scattered locations.

Off-site locations and media that have been impacted by contaminant transport from these source areas include surface water and sediment in the Southeast Drainage (Weldon Spring Wildlife Area) and in Burgermeister Spring and Lakes 34, 35, and 36 (Busch Wildlife Area). Soil at discrete vicinity properties is also contaminated as a result of past operations.

The major pathways that have resulted in contaminant transport to these off-site locations are surface water runoff, surface water loss to groundwater (via losing streams), groundwater discharge to surface water (via gaining streams), and leaching from surface and/or subsurface material to groundwater. The risks to human health associated with contaminated

TABLE 6.1 Contaminants of Concern for the Human Health Assessment

Radionuclides[a,b]	Metals	Other Inorganic Compounds	Nitroaromatic Compounds	PCBs and PAHs
Actinium-227	Antimony	Asbestos[a]	DNB	PCBs[a]
Lead-210	Arsenic[a]	Fluoride	2,4-DNT[a]	Acenaphthene
Protactinium-231	Barium	Nitrate	2,6-DNT[a]	Anthracene
Radium-226	Beryllium[a]	Nitrite	NB	Benz(a)anthracene[a]
Radium-228	Cadmium[a]		TNB	Benzo(b)fluoranthene[a]
Radon-220	Chromium[a]		TNT[a]	Benzo(k)fluoranthene[a]
Radon-222	Cobalt			Benzo(g,h,i)perylene
Thorium-230	Copper			Benzo(a)pyrene[a]
Thorium-232	Lead[a]			Chrysene[a]
Uranium-235	Lithium			Fluoranthene
Uranium-238	Manganese			Fluorene
	Mercury			Indeno(1,2,3-cd)pyrene[a]
	Molybdenum			2-Methylnaphthalene
	Nickel[a]			Naphthalene
	Selenium			Phenanthrene
	Silver			Pyrene
	Thallium			
	Uranium			
	Vanadium			
	Zinc			

[a] Potential carcinogen.

[b] Exposure to gamma radiation resulting from the presence of these radionuclides was also evaluated.

groundwater at the site are not evaluated in this BA because the pathway is currently incomplete and groundwater data are limited. Groundwater will be addressed as a separate operable unit, for which a focused RI/FS will be prepared within the next several years.

6.2.2 Potential Receptors and Routes of Exposure

Potential receptors to contaminated media both on-site and at off-site areas impacted by site releases were identified for both current and hypothetical future conditions. Under current conditions, worker activities are ongoing and public access is controlled by the site fence and security guards. The potential on-site receptors identified for these conditions are a site maintenance worker and a trespasser.

For the maintenance worker, direct contact with source areas such as the raffinate pits and buildings is precluded by the nature of the work, access controls, and worker protection programs. Hence, potential exposures were assessed only for on-site soil and air. The routes of exposure assessed were external gamma irradiation, incidental ingestion of soil, and inhalation

of radon-222 and airborne particulates. The worker was assumed to be exposed 8 hours per day, 200 days per year, over 10 years.

A trespasser was assumed to access all areas of the site. Hence, in addition to the routes of exposure identified for the worker, the trespasser could potentially be exposed to (1) surface water and sludge at the raffinate pits and (2) air and residues in the buildings. The additional routes of outdoor exposure assessed for the trespasser were external gamma irradiation and ingestion of surface water and sludge at the raffinate pits. For the buildings, the routes of exposure assessed were external gamma irradiation; inhalation of radon, radioactive particulates, and asbestos; and incidental ingestion of radioactive dust and PCB-contaminated residues. The trespasser was assumed to enter the site 5 times per year, for 1 hour each time, over 10 years.

Under current conditions, the off-site exposure points evaluated for a member of the general public were Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage. Discrete soil vicinity properties in the surrounding wildlife areas were also addressed. Land use is recreational, and a visitor was considered to hike in both wildlife areas. The potential routes of exposure assessed for this receptor were external gamma irradiation and ingestion of surface water, sediment, and soil. The recreational visitor was assumed to visit these areas 20 times per year for 4 hours each time over 30 years.

Under hypothetical future conditions, it was assumed that the site fence would be gone and potential land use would be similar to that in the surrounding wildlife areas. Thus, the future on-site receptor was identified as a recreational visitor (the future off-site receptor would remain the same). The exposure points and likely routes of exposure for the future on-site receptor are the same as those identified for the current trespasser, with the addition of a game ingestion pathway (which is also considered to conservatively represent potential current exposures associated with game ingestion). Because access to the site would no longer be restricted, the frequency and duration of exposure could potentially increase over those assumed for a trespasser. Therefore, the future recreational visitor was assumed to enter the site 20 times per year, for an average of 4 hours each time, over 30 years.

The exposure points and likely routes of exposure for the potential on-site and off-site receptors have been summarized in Tables 3.1 and 3.2. In addition to exposures from external gamma irradiation, inhalation, and ingestion at the various on-site and off-site locations, dermal contact is considered a potential route of exposure for each receptor. The locations and media at which dermal contact could occur are the same as those identified for ingestion. For solid media such as soil, absorption fractions for estimating intake were available only for cadmium and PCBs, and hence potential health effects associated with dermal contact could be quantified only for those two chemicals. For dermal contact with water, the assessment included most contaminants of concern but incorporated default rather than compound-specific parameters. Hence, considerable uncertainty is associated with the assessment of the dermal pathway.

6.2.3 Exposure Point Concentrations

Exposure point concentrations for the various media addressed in the exposure assessment were determined on the basis of data availability and the objective of the analysis. Extensive data were available for soil, and contaminant heterogeneity was addressed by conducting both a sitewide and a location-specific analysis. For the sitewide analysis, the UL_{95} value was used as the exposure point concentration for each contaminant. For the location-specific analysis, actual measurements or derived values from each sample location were used as the exposure point concentrations.

For air, exposure point concentrations were modeled from UL_{95} values for the southern portion of the site, which is considered the most likely source of fugitive dust under baseline conditions. This modeling approach was used because measurements are not available for all airborne contaminants. For sludge, sediment, and surface water, maximum concentrations were used as the exposure point concentrations — with two exceptions — because screening-level analyses were conducted for these media and certain limitations exist for the available data. The first exception is uranium in surface water at the Southeast Drainage, in which water flows intermittently and measured concentrations have varied widely over time; half the maximum measured concentration was used to represent this exposure point concentration over the 30-year exposure period. The second exception is use of the average of detected values for surface water in the raffinate pits to model uptake by waterfowl for the game ingestion pathway.

For radioactive contamination in the buildings, average concentrations from Building 403 — a former process building that is heavily contaminated — were used to represent exposure point concentrations for all buildings. The UL_{95} value for residual PCB contamination was determined from data for Building 408, and the UL_{95} value for asbestos was determined from airborne concentrations measured in Building 201. Cleanup decisions have already been made for buildings and surface water, so results of these conservative analyses are considered as screening-level information.

On the basis of the time period assumed for hypothetical future conditions (10 to 30 years) and the nature of release control measures at the site, the exposure point concentrations both on-site and off-site were assumed to remain at their current levels. Hence, the future recreational visitor on-site would be exposed to the same concentrations identified for the current trespasser, and exposures to a future recreational visitor off-site would be the same as those assessed for the current recreational visitor.

6.3 TOXICITY ASSESSMENT

Cancer and chemical toxicity are the two general health-effect end points from exposure to site contaminants. Cancer induction is the primary health effect associated with radionuclides at the site, and 17 of the chemical contaminants of concern are classified as potential carcinogens. Four of the 17 are classified as Group A carcinogens, for which strong evidence exists for human carcinogenicity.

A number of toxic effects are linked with exposure to noncarcinogenic contaminants. Uranium is the most significant contributor to noncarcinogenic health effects associated with site soil, and the chemical toxicity associated with human exposure to uranium is kidney damage. The PCBs inside the chemical plant buildings and at a few soil locations also contribute significantly to potential chemical carcinogenicity and toxicity, which is characterized by skin effects and liver damage.

Potential carcinogenic risks from exposures to radiation were estimated by using scientifically accepted values to convert from estimated doses (in mrem, or in WLM for radon exposures) to the likelihood of cancer induction. The potential for carcinogenic and noncarcinogenic effects of human exposure to chemicals was quantified with standard EPA slope factors and RfDs. Because standard values have not been developed for the dermal pathway, these values were derived from oral slope factors, RfDs, and DCFs.

6.4 RISK CHARACTERIZATION

Radiological and chemical health effects have been evaluated for potential exposures to contaminants at the Weldon Spring site and at nearby properties impacted by site releases. Potential carcinogenic risks for both radiological and chemical exposures were assessed in terms of the increased probability that an individual will develop cancer over the course of a lifetime. The EPA has identified a range of 1×10^{-6} to 1×10^{-4} for the incremental cancer risk to an individual from exposures at NPL sites; this range is referred to as the target risk range for purposes of discussion in this assessment. For comparison, about one in three Americans will develop cancer from all sources and 60% of all cancers are estimated to be fatal. For radiological exposures, the individual lifetime risk of fatal cancer associated with background radiation, primarily from naturally occurring radon, is estimated by the EPA to be approximately 1×10^{-2}.

The potential for adverse health effects other than cancer from exposure to site contaminants was assessed by estimating hazard indexes. An endpoint-specific hazard index of greater than 1 may indicate a potential for adverse health effects or, conversely, a hazard index of 1 or less is considered to indicate a nonhazardous situation.

The estimated pathway-specific risks and hazard indexes associated with exposures to on-site sources are summarized in Tables 6.2 and 6.3; those associated with combined pathway scenarios (the reasonable maximum exposures) are summarized in Table 6.4. The risks and hazard indexes to the off-site recreational visitor are presented in Table 6.5. The risks and hazard indexes presented in these tables for the worker, trespasser, recreational visitor, and sportsman include the contribution from dermal exposure to soil, sediment, sludge, and/or building residues, as appropriate to the individual's exposures. However, because the health effects associated with dermal exposures could be quantified only for cadmium and PCBs, this pathway contributed insignificantly to the risks and hazard indexes for most locations and media. Some information is available to estimate impacts from dermal contact with water, so risks and hazard indexes were determined for an individual who might swim in the raffinate pits and off-site lakes. Results for the raffinate pits and off-site lakes are included in Tables 6.4 and 6.5, respectively.

TABLE 6.2 Estimated Carcinogenic Risks for On-Site Receptors

	Current Conditions				Future Conditions	
	Site Worker		Trespasser		Recreational Visitor	
Areas and Media	Radiological	Chemical	Radiological	Chemical	Radiological	Chemical
Sitewide soil	8×10^{-5}	1×10^{-5}	6×10^{-7}	2×10^{-7}	1×10^{-5}	3×10^{-6}
Raffinate pits						
Surface water	NQ[a]	NQ	8×10^{-6}	2×10^{-6}	1×10^{-4}	2×10^{-5}
Sludge	NQ	NQ	2×10^{-4}	7×10^{-6}	3×10^{-3}	9×10^{-5}
Air	4×10^{-4}	2×10^{-7}	1×10^{-6}	9×10^{-10}	5×10^{-5}	3×10^{-8}
Buildings						
Air[b]	NQ	NQ	5×10^{-5}	2×10^{-6}	6×10^{-4}	2×10^{-5}
Residues[c]	NQ	NQ	5×10^{-5}	4×10^{-4}	9×10^{-4}	3×10^{-3}

[a] NQ = not quantified for this receptor.

[b] Estimated for radionuclides and asbestos.

[c] Estimated for radionuclides and PCBs.

TABLE 6.3 Estimated Hazard Indexes for On-Site Receptors[a]

	Current Conditions		Future Conditions
Areas and Media	Site Worker	Trespasser	Recreational Visitor
Sitewide soil	0.4	0.005	0.02
Raffinate pits			
Surface water	NQ	0.5	1
Sludge	NQ	0.2	0.8
Air	0.2	0.0007	0.008
Buildings			
Air[b]	-	-	-
Residues[c]	NQ	3	10

[a] Estimated for chemicals only; NQ = not quantified for this receptor.

[b] A hyphen indicates that the hazard index could not be estimated because an inhalation reference dose is not available for asbestos or PCBs.

[c] Estimated for PCBs.

TABLE 6.4 Total On-Site Risks and Hazard Indexes[a]

Receptor	Radiological Risk	Chemical Risk	Total Risk	Hazard Index
Current worker	5×10^{-4}	1×10^{-5}	5×10^{-4}	0.5
Current trespasser[b]	9×10^{-5}	1×10^{-4}	2×10^{-4}	1
Sportsman[c]	5×10^{-5}	3×10^{-6}	5×10^{-5}	0.08
Current/future swimmer[d]	2×10^{-4}	5×10^{-6}	2×10^{-4}	0.2
Future recreational visitor[b]	1×10^{-3}	1×10^{-3}	2×10^{-3}	4

[a] Except for the swimmer, the risks and hazard indexes do not include a contribution from the dermal pathway. Excluding the dermal pathway for the swimmer (for which there is considerable uncertainty [Section 5.5.2]), the radiological risk is 5×10^{-5}, the chemical risk is 1×10^{-6}, and the hazard index is 0.07.

[b] Scenarios T12 (for the trespasser) and R12 (for the recreational visitor); assumes exposure to sitewide soil and air, raffinate pit surface water and sludge, and building air and residues (Section 5.5.1).

[c] Exposure is to sitewide soil and air, on-site game, off-site surface water and sediment, and off-site fish (Section 5.4.2).

[d] Exposure is to sitewide air and raffinate pit surface water and sludge (Section 5.5.1).

TABLE 6.5 Estimated Carcinogenic Risks and Hazard Indexes for the Recreational Visitor at Off-Site Areas

Areas and Media	Radiological Risk	Chemical Risk	Total Risk	Hazard Index[a]
Lakes 34, 35, and 36[b]				
Surface water	4×10^{-6}	4×10^{-6}	8×10^{-6}	0.1
Sediment	3×10^{-6}	9×10^{-7}	4×10^{-6}	0.02
Burgermeister Spring				
Surface water	4×10^{-6}	9×10^{-7}	4×10^{-6}	0.04
Southeast Drainage				
Surface water	1×10^{-5}	1×10^{-6}	1×10^{-5}	0.2
Sediment	2×10^{-4}	1×10^{-6}	2×10^{-4}	0.02
Vicinity properties				
Soil[c]	6×10^{-7} to 3×10^{-4}	NQ[d]	6×10^{-7} to 3×10^{-4}	NQ

[a] Estimated for chemicals only.

[b] The estimated radiological and chemical risks for a swimmer in the lakes are 3×10^{-7} and 4×10^{-8}, respectively, and the hazard index is 0.005.

[c] The range for radiological risk represents exposures to each of the vicinity properties for the entire exposure period.

[d] NQ = not quantified.

Pathway-specific risks and hazard indexes for the on-site worker were estimated for exposures to sitewide soil and air. The radiological risk from these exposures is estimated to be 5×10^{-4}, which exceeds the upper limit of the target risk range. Inhalation of radon-222 decay products is the major contributor to this risk; the contributions from incidental ingestion of soil and external gamma irradiation are relatively minor. The chemical carcinogenic risk for worker exposures to sitewide soil and air is 1×10^{-5}, and the hazard index is 0.5. These estimates are within and below the respective levels of concern.

Pathway-specific risks and hazard indexes for the on-site trespasser and future recreational visitor were estimated for exposures to sitewide soil, outdoor air, raffinate pit surface water and sludge, and building air and residues. Exposures to sitewide soil and outdoor air contribute insignificantly to the radiological risk for the trespasser and recreational visitor compared with exposures at the raffinate pits and buildings. The radiological risk to the trespasser from exposure to raffinate pit surface water is 8×10^{-6}, and the risk from exposure to raffinate pit sludge is 2×10^{-4}. Corresponding risks for the recreational visitor are 1×10^{-4} and 3×10^{-3}, respectively. Thus, risks from combined exposures to surface water and sludge at the raffinate pits exceed EPA's target range for both receptors. The single largest contributor to radiological risk at the site is exposure to the raffinate pit sludge, which is primarily due to ingestion of actinium-227, protactinium-231, and thorium-230 in the sludge. The risk to an individual who swims in the raffinate pits is 2×10^{-4}; most of this risk is associated with the estimate for dermal absorption, for which there is considerable uncertainty. The estimated risk to the swimmer from external gamma irradiation and incidental ingestion of surface water and sludge is 5×10^{-5}, and incidental ingestion of sludge is the major contributor.

Radiological risks from exposure to contaminants in the chemical plant buildings were assumed to be represented by intrusion into one of the most heavily contaminated structures, Building 403. The radiological risk was divided into the risk from indoor air (i.e., inhalation of airborne particulates and radon gas) and the risk from residues (i.e., external gamma irradiation and ingestion of radioactive contaminants). The overall risks for the trespasser and future recreational visitor are 1×10^{-4} and 1×10^{-3}, respectively. The risk for the trespasser is at the upper limit of the target range, whereas that for the recreational visitor exceeds the range. Incidental ingestion of radioactively contaminated residues (principally uranium-238 and thorium-232) is the largest single contributor to the radiological risk in the building, accounting for 60% of the total.

Exposure to outdoor air contributes insignificantly to the chemical carcinogenic risks and hazard indexes for the on-site trespasser and future recreational visitor. The highest risks and hazard indexes are associated with exposures in the buildings. The chemical carcinogenic risks from inhalation of asbestos and ingestion of and dermal contact with PCB-contaminated residues are 4×10^{-4} for the trespasser and 3×10^{-3} for the future recreational visitor. These risks are dominated by PCB exposures and exceed EPA's target range. The hazard indexes for the two receptors are 3 and 10, respectively (because PCBs alone account for hazard indexes greater than 1, the segregated index also exceeds 1). These hazard indexes are also higher than EPA's level of concern.

The overall chemical carcinogenic risks associated with incidental ingestion of raffinate pit surface water and sludge are 9×10^{-6} for the trespasser and 1×10^{-4} for the recreational visitor. The estimated carcinogenic risk for the trespasser is within EPA's target range, and that for the recreational visitor is at the upper end of the range. The total hazard indexes at the pits are 0.7 for the trespasser and 2 for the recreational visitor; for the latter, the segregated hazard index is less than 1. Exposure to sitewide air and raffinate pit surface water and sludge was also estimated for a swimmer. The estimated carcinogenic risk is 5×10^{-6}, and the hazard index is 0.2. These estimates are within and below the respective levels of concern.

The risks and hazard indexes associated with multiple exposure points, media, and routes of exposure for on-site locations were estimated for a number of possible exposure scenarios, for which it was assumed that the trespasser and future recreational visitor would come into contact with the various source areas and contaminated media at the site over the respective 10- and 30-year exposure periods. Under current conditions, the radiological risks for on-site receptors are bounded by the risk estimated for a trespasser who is exposed only to soil and outdoor air (2×10^{-6}) and that for a trespasser who is exposed to raffinate pit surface water and sludge and outdoor air (2×10^{-4}). Under hypothetical future conditions, the risks for a recreational visitor are also bounded by these two exposure scenarios (6×10^{-5} to 3×10^{-3}). Assuming that exposure is distributed among sitewide soil and air, the raffinate pits, and the buildings, the estimated radiological risk to a trespasser is 9×10^{-5} and that to a recreational visitor is 1×10^{-3} (Table 6.4). Risks for the recreational visitor from exposures at the raffinate pits and the buildings exceed EPA's target range.

Risks were also estimated for an avid sportsman assumed to hunt on-site and fish at Lakes 34, 35, and 36. The estimated radiological risk for this scenario (assuming exposure to lake surface water and sediment, site soil, and fish and game) is 5×10^{-5}; this risk is within EPA's target range.

The chemical carcinogenic risks for the on-site receptors are bounded by the risk for an individual who is exposed only to sitewide soil and outdoor air (2×10^{-7} for the trespasser and 3×10^{-6} for the recreational visitor) and that of an individual who is exposed only to indoor air and residues at the chemical plant buildings (4×10^{-4} for the trespasser and 3×10^{-3} for the recreational visitor). Assuming that exposure is distributed among sitewide soil and air, the raffinate pits, and the buildings, the risk for a trespasser is 1×10^{-4} and that for a recreational visitor is 1×10^{-3} (Table 6.4). The lowest hazard indexes are associated with exposures to soil and outdoor air (0.005 for the trespasser and 0.02 for the recreational visitor), and the highest hazard indexes are associated with exposures in the buildings (3 for the trespasser and 10 for the recreational visitor). The estimated risks and hazard indexes for the buildings exceed EPA's target range for carcinogenic effects and the level of concern for noncarcinogenic effects. The risk and hazard index for the sportsman are 3×10^{-6} and 0.08, respectively.

Pathway-specific risks and hazard indexes for the off-site recreational visitor were estimated for exposures at Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage. The radiological risks associated with exposures to contaminated soil at the vicinity properties were also calculated. The radiological risks estimated for exposures at Lakes 34, 35,

and 36 and at Burgermeister Spring are within EPA's target risk range (Table 6.5). The radiological risk for combined exposures of a swimmer at the off-site lakes is less than EPA's target range. The total radiological risk estimated for exposures at the Southeast Drainage exceeds the upper end of the range by a factor of 2, and the primary contributor is incidental ingestion of sediment. However, the assessment for the Southeast Drainage is preliminary and is based on screening-level assumptions with limited data (e.g., for hot spots); this estimate will be refined within the next several years after additional characterization data are obtained.

The radiological risks associated with soil at vicinity properties range from 6×10^{-7} to 3×10^{-4}. These estimates are conservative because the recreational visitor is assumed to be exposed to the radioactive contaminants at an individual property during each visit for the entire exposure duration. Such an exposure is unlikely because of the small size of the properties. More realistic estimates of risks posed by these properties would be within or below EPA's target range.

The chemical carcinogenic risks estimated for the off-site exposure points are within or below EPA's target risk range (Table 6.5). The hazard indexes are also well below the level of concern. The chemical risk for combined exposures of a swimmer at the off-site lakes is below EPA's target range, and the hazard index is below the level of concern.

The total carcinogenic risks resulting from exposures to both radioactive and chemical contaminants are presented in Table 6.4 for on-site exposures and Table 6.5 for off-site exposures. These risks are the arithmetic sum of the radiological and chemical carcinogenic risks. Because different assumptions are inherent in the methods available for estimating radiological and chemical risks, considerable uncertainty is associated with these combined risk estimates. In addition, the estimates do not account for any potential antagonistic or synergistic effects. For scenarios in which exposure occurs only to sitewide soil and air — as for the worker — the radiological risks are higher than the chemical risks. The risks are approximately the same for the trespasser and future recreational visitor, for whom the highest radiological and chemical risks are associated with exposures at the buildings and the raffinate pits. The radiological and chemical risks are generally similar at Lakes 34, 35, and 36 and at Burgermeister Spring, and the radiological risks are higher than the chemical risks for the screening-level assessment of the Southeast Drainage. These calculations are based on conservative scenario definitions, so the risks to potential off-site receptors are expected to be lower than those presented herein. Of the on-site and off-site scenarios evaluated, the highest total risk was estimated for a future recreational visitor to the site, mainly from the combined radiological and chemical exposures at the raffinate pits and the buildings.

6.5 CONCLUSIONS

From the results of the human health risk assessment, it can be concluded that the highest health impacts associated with the Weldon Spring site result from exposures occurring at the chemical plant buildings and the raffinate pits. Under the assumptions used in this BA, the potential health impacts from these exposures to a trespasser who routinely enters the site

exceed both the upper end of EPA's target risk range and the level of concern for noncarcinogenic effects. However, access to the site is currently controlled, so the potential for exposure of members of the general public to the buildings and pits is very low. Under hypothetical future conditions, for which access to the site is assumed to be unrestricted, the potential exists for an unprotected individual frequently entering the site to incur adverse health effects. Risks and noncarcinogenic effects estimated for exposures to sitewide soil and air are within EPA's target range for the trespasser and recreational visitor.

The radiological and chemical carcinogenic risks estimated for repeated exposures at off-site locations are within or below EPA's target range, except for the Southeast Drainage (hot spots) and one soil vicinity property. The hazard index is less than 1 for each off-site exposure point.

In conclusion, the results of the human health assessment presented in this BA should not be taken to represent absolute risk. Rather, they should be considered to represent the most important sources of potential relative risk at the site, which — having been identified — may be dealt with effectively in the remedial action process.

7 ECOLOGICAL RESOURCES AND CONTAMINANTS OF CONCERN

The Weldon Spring site is located on the drainage divide of the Missouri and Mississippi rivers along the boundary between the southern portion of the Glaciated Plains physiographic province and the northern portion of the Ozark Border physiographic province (Johnson 1987). The landform in the area varies from rolling hills to moderately sloped forestland. Limestone bluffs along the Missouri River mark the boundary between floodplain and upland landscapes. The Missouri River floodplain landscape has little relief, with elevations between 137 and 141 m (450 and 464 ft) MSL. Elevations on the August A. Busch and Weldon Spring wildlife areas vary from 137 m (450 ft) MSL at the base of the bluffs to 217 m (712 ft) MSL on the uplands (Missouri Department of Conservation 1989). Because the Weldon Spring site is located along the boundary of two physiographic provinces and near the confluence of the Missouri and Mississippi rivers, the area contains a large number of different habitats and a diverse flora and fauna.

Limited site-specific ecological data are currently available from which to assess potential risks to ecological resources. In addition, the adjacent Army NPL site complicates the interpretation of potential off-site impacts because it contributes to contamination in the vicinity. Therefore, the information presented in this chapter is preliminary. Data to support this ecological risk assessment are currently being collected. These data and additional data being collected for other purposes (e.g., environmental monitoring) will be incorporated into future documentation for the project, including documents that will be prepared within the next several years for the groundwater operable unit and the Southeast Drainage.

7.1 ECOLOGICAL RESOURCES

7.1.1 On-Site Biota and Habitats

7.1.1.1 Terrestrial Resources

Much of the Weldon Spring site (approximately 66 ha [162 acres]) consists of grassland and old-field habitats that are periodically mowed. In contrast, approximately 22 ha (55 acres) in the northwestern portion of the site consists of old-field/upland forest habitat. Trees in the upland forest areas include oak, hickory, locust, and cottonwood. Plant species that have been observed at the site are listed in Table 7.1.

A previous biological investigation identified a relatively depauperate wildlife fauna in the industrial portion of the site (RETA 1978), due primarily to the general absence of adequate and diverse habitats at this area. However, because the northwestern portion of the site contains old-field and relatively good-quality upland forest habitat, it probably supports a fauna similar to that of the surrounding wildlife areas. In addition to a resident fauna, the site (particularly

TABLE 7.1 Vegetation Observed at the Weldon Spring Site

Alfalfa	Hog peanut	Shagbark hickory
American elm	Honey locust	Shingle oak
American plum	Horse nettle	Short-leaved pine
American sycamore	Horseweed	Skullcap
Beauty berry	Ironweed	Slender nettle
Bellwort	Mountain bluegrass	Smooth hedge nettle
Black locust	Nodding foxtail	Smooth sumac
Box elder	Panic grass	Spanish needles
Broad-leaved arrowhead	Paper mulberry	Spotted touch-me-not
Broad-leaved cattail	Partridge pea	Stinging nettle
Canada rush	Pawpaw	Sugar maple
Common cattail	Persimmon	Swamp smartweed
Common dandelion	Plantain	Sweetgum
Common milkweed	Poison ivy	Tall eriogonum (bluestem)
Common mullein	Pokeweed	Tartarian honeysuckle
Common sunflower	Post oak	Thoroughwort
Common thistle	Queen Anne's lace	Tickseed sunflower
Cut-leaved water horehound	Ragweed	Trailing bush clover
Daisy fleabane	Ranunculus	Trumpet creeper
Dodder	Redbud	Virginia creeper
Eastern cottonwood	Red clover	White avens
Eastern red cedar	Red-tip grass	White sweet clover
Flowering dogwood	Rosa sp.	Wild grape
Goldenrod	Roughleaf dogwood	Willow
Green ash	Sassafras	Yarrow
Ground cherry	Scurf-pea	Yellow rocket
Hackberry	Sedge	Yellow sweet clover
Hedge bindweed		

Sources: RETA (1978); DOE (1992c, Table 5.5-1).

the northwestern portion) is presumably used by a number of species that move freely between on-site and off-site locations. Wildlife that have been observed at the site are listed in Tables 7.2 and 7.3. Wildlife species that have been reported from St. Charles County and could potentially occur at the site are listed in Appendix C. Wildlife observed in the northwestern portion of the site include deer, squirrels, wild turkeys, hawks, and a variety of other birds.

7.1.1.2 Aquatic Resources

Aquatic habitats at the site cover approximately 15 ha (38 acres) and include the four raffinate pits, Ash Pond, Frog Pond, and the drainages from the two ponds. The water and sediment impounded at the site are radioactively and chemically contaminated and, except for Ash Pond, contain little emergent or submergent rooted aquatic vegetation. Because of the contamination, the raffinate pits represent poor aquatic habitats. Past electrofishing efforts at

TABLE 7.2 Mammals, Fish, Reptiles, and Amphibians Observed at the Weldon Spring Site

Mammals	Reptiles
Eastern cottontail rabbit	Common snapping turtle
House mouse, other mice	Prairie kingsnake
Muskrat	Western box turtle
Opossum	
Raccoon	Amphibians
Short-tailed shrew	Bullfrog
Squirrel	Cricket frog
White-tailed deer	Leopard frog
Fish	
Green sunfish	

Sources: RETA (1978); DOE (1992c, Table 5.5-7).

TABLE 7.3 Birds Observed at the Weldon Spring Site, 1991

American coot	Eastern kingbird	Northern oriole
American crow	Eastern meadowlark	Northern pintail
American goldfinch	European starling	Northern shoveler
American kestrel	Field sparrow	Pied-billed grebe
American robin	Great blue heron	Pileated woodpecker
American widgeon	Great crested flycatcher	Red-headed woodpecker
Barn swallow	Great horned owl	Red-tailed hawk
Belted kingfisher	Green-backed heron	Red-winged blackbird
Black-capped chickadee	Hairy woodpecker	Ring-necked duck
Blue jay	Hooded merganser	Rock dove
Blue-winged teal	House sparrow	Ruby-throated hummingbird
Bufflehead	Killdeer	Ruddy duck
Canada goose	Lesser scaup	Spotted sandpiper
Canvasback	Mallard	Tufted titmouse
Cedar waxwing	Meadowlark	White-crowned sparrow
Common nighthawk	Mockingbird	White-throated sparrow
Common snipe	Mourning dove	Wild turkey
Dark-eyed junco	Nighthawk	Wood duck
Downy woodpecker	Northern cardinal	Yellow-billed cuckoo
Eastern bluebird	Northern flicker	Yellow-rumped warbler

Source: DOE (1992c, Table 5.5-2).

Frog Pond have recovered only small sunfish (MK-Ferguson Company and Jacobs Engineering Group 1989e); no fish have been found in the raffinate pits or Ash Pond.

The on-site aquatic habitats attract waterfowl and shorebirds, and large numbers of waterfowl have been observed at some of these habitats during the autumn and spring migratory seasons (DOE 1992c). More than 15 species of waterfowl have been observed feeding and resting on the impoundments, including wood duck, mallard, blue-winged teal, scaup, pied-billed grebe, and Canada goose (MK-Ferguson Company and Jacobs Engineering Group 1992c); some species (e.g., Canada goose) may breed on-site. Many of the impoundments in the surrounding Bush Wildlife Complex are used year-round by waterfowl, and many species breed in the area.

7.1.2 Off-Site Biota and Habitats

Except for a few facilities (such as Francis Howell High School and the adjacent state of Missouri highway maintenance facility), most of the area surrounding the Weldon Spring site is part of the Busch Wildlife Complex. This complex, which is controlled by the Missouri Department of Conservation, consists of three wildlife areas totaling approximately 6,838 ha (16,890 acres) that are actively managed for wildlife and support a diverse biota. The wildlife areas are the August A. Busch Memorial Wildlife Area (2,829 ha [6,987 acres]) to the north of the site, the Weldon Spring Wildlife Area (2,978 ha [7,356 acres]) to the south and east, and the Howell Island Wildlife Area (1,031 ha [2,547 acres]) in the Missouri River to the southeast (Figure 1.6). Land to the west of the Weldon Spring site is also designated as an NPL site under the responsibility of the U.S. Department of the Army (Section 1.2.2).

7.1.2.1 Terrestrial Resources

A large variety of habitats occur at the Busch Wildlife Complex, including open fields and pastures; wetlands; prairies; upland and bottomland forests; and cultivated fields of several row crops and grasses (Table 7.4). Forest habitat is the most abundant, comprising approximately 64% (4,378 ha [10,813 acres]) of the entire Busch Wildlife Complex (ranging from 45% of the Busch Wildlife Area to 88% of the Howell Island Wildlife Area). The upland forests are primarily oak and oak-hickory. The bottomland forests, which occur within the Missouri River floodplain and along stream and lake banks, are characterized by willow, eastern cottonwood, silver maple, and oaks. Other tree species found in many of the forested habitats include shagbark hickory, pawpaw, Kentucky coffee tree, elm, and box elder.

Terrestrial habitats in the wildlife areas are actively managed for wildlife use. Open lands are managed to minimize soil erosion and to provide forage and cover for wildlife. Some portions of the Busch and Weldon Spring wildlife areas are leased to sharecroppers for agricultural production. This leasing program requires that a percentage of the crop be left for wildlife use, and it includes fallow rotation during which fields remain unfarmed for at least 1 year

TABLE 7.4 Vegetative Cover at the Busch Wildlife Complex

Wildlife Area	Vegetative Cover	
	Acres	Percent
August A. Busch		
Forest	3,158	45
Old field	892	13
Cropland	1,590	23
Roads	233	3
Water	517	7
Pasture	597	9
Total	6,987	100
Weldon Spring		
Forest	5,408	74
Old field	1,106	15
Cropland	680	9
Water	127	2
Roads and facilities	35	0.5
Total	7,356	100
Howell Island		
Forest	2,247	88
Cropland	300	12
Total	2,547	100

Source: Missouri Department of Conservation (1989).

before they are again cultivated. As a result of the active wildlife management and share-cropping programs, cropland is the second most abundant vegetative cover in the wildlife areas (Missouri Department of Conservation 1989), comprising approximately 1,011 ha (2,498 acres) of the Busch Wildlife Complex. Crop species commonly planted at the complex include corn, sunflower, soybean, winter wheat, and millet. Old-field and pasture areas cover approximately 1,050 ha (2,600 acres) of the Busch Wildlife Complex and support fescue, bluegrass, Indian mallow, crabgrass, ragweed, aster, thistles, mustard, fleabane, and goldenrod.

The Missouri Department of Conservation has identified 29 mammalian species, 47 reptilian species, and 25 amphibian species in St. Charles County (Dickneite 1988; Appendix C), and many of these species occur at the Busch Wildlife Complex (DOE 1992c, Section 5.5). Mammals common to the area include fox and grey squirrel, white-tailed deer, opossum, raccoon, skunk, eastern cottontail, and a variety of mice and other rodents. Common

reptiles and amphibians in the area include bullfrog, spring peeper, and a variety of salamanders, turtles, and snakes.

More than 295 bird species have been reported from the county, of which at least 277 were from the wildlife areas (Missouri Department of Conservation 1981). This large number of bird species is a result of the location of the county along the Mississippi Flyway, a major north-south route for migratory North American birds. More than 100 species are known to breed in the area, and 43 species are reported as common residents in St. Charles County throughout at least three seasons of the year (Missouri Department of Conservation 1981). More than 60 species have been reported from the Weldon Spring site. Common birds in the area include a variety of warblers, sparrows, and other songbirds; red-tailed hawk; American kestrel; barred owl; great horned owl; downy, pileated, and red-bellied woodpeckers; and wild turkey.

The surface water impoundments in the wildlife complex provide habitat for a variety of waterfowl and shorebirds, such as Canada goose, mallard, wood duck, sandpipers, herons, and gulls; many of these species nest and/or overwinter in the area. In addition, up to 30 species of waterfowl have been reported to migrate through the area in spring and autumn (Missouri Department of Conservation 1981), and they use the numerous surface waters as resting and feeding areas. Many species are present throughout the year, and large numbers of some species are present in the winter months. For example, aerial surveys for overwintering geese, conducted by the U.S. Fish and Wildlife Service in December 1989 and January 1990, yielded counts of 1,600 and 1,150 geese, respectively (Burger 1990).

The Missouri Department of Conservation maintains a goose management program at the complex, as well as management programs for resident and migratory shorebirds and other waterfowl. For example, a 5.3-ha (13-acre) shorebird area has been constructed in the Busch Wildlife Area, and water levels in this area are actively controlled to provide foraging habitat for resident and migratory shorebirds, waterfowl, and wading birds. Goose nesting tubs and wood duck nest boxes are maintained and monitored at many of the lakes. In 1989, 29 successful nests were built in the tubs, yielding 257 eggs of which 58% hatched (DeBruyckere 1990). The waterfowl refuge located north of the site at Ahden Knight Hampton Memorial Lake (Figure 1.6) serves as one of the major waterfowl nesting areas in the Busch Wildlife Complex.

7.1.2.2 Aquatic Resources

Aquatic habitats in the immediate vicinity of the site include numerous intermittent and permanent streams, ponds and lakes, the Missouri River, and portions of the Mississippi River that ultimately receive drainage from St. Charles County, the Busch Wildlife Complex, and portions of the Weldon Spring site (Figure 1.6). Dickneite (1988) reports that 105 species of fish are present in St. Charles County (Appendix C). The habitats and distributions of Missouri fish indicate that many of the species listed for St. Charles County probably occur in suitable habitats at the wildlife complex.

A large number of aquatic habitats are present in the Busch Wildlife Complex; these habitats include 8 km (5 mi) of Missouri River frontage, 34 lakes totaling 223 surface ha

(552 acres) and ranging in size from 0.4 to 74 ha (1 to 182 acres), 72 ponds, 2 fishable quarries, the 6.4-ha (15.7-acre) Femme Osage Slough, numerous intermittent and permanent streams, 5.3 ha (13 acres) of impoundments to attract shorebirds, 6.8 km (4.25 mi) of Dardenne Creek, 2.4 km (1.5 mi) of Kraut Run, and numerous springs and seeps.

Fish reported to occur at the wildlife complex are typical of warmwater fish communities throughout the Midwest (Pflieger 1975; Smith 1979; Trautman 1981; Robison and Buchanan 1988). Species include carp, black bullhead, bluegill, black and white crappie, a variety of minnows and shiners, green sunfish, rock bass, grass pickerel, buffalo, and largemouth and black bass (Missouri Botanical Garden 1975; Missouri Department of Conservation 1989).

The Busch Wildlife Area has 33 lakes and 60 ponds totaling 209 and 8.9 surface ha (517 and 22 surface acres), respectively; major streams in this wildlife area are Dardenne Creek and Kraut Run. Aquatic habitats in the Weldon Spring Wildlife Area include the Missouri River, Femme Osage Slough, Prairie Lake, 12 ponds, numerous intermittent and permanent streams, and several springs. Many of the ponds and lakes at the Busch Wildlife Complex are intensively managed as warmwater fisheries for largemouth bass, crappie, bluegill, channel catfish, flathead catfish, and other species (Missouri Department of Conservation 1989). In addition, five lakes in the Busch Wildlife Area are intensively managed for channel catfish (see Appendix C).

7.1.3 High-Quality Natural Communities

7.1.3.1 Unique Natural Areas

The Missouri Department of Conservation has identified four high-quality natural communities in the area of the Weldon Spring site: a mesic/dry-mesic forest; an old-growth mesic forest; a dry forest; and a savannah (Gaines 1988). The mesic/dry-mesic forest of approximately 51 ha (125 acres) containing high-quality old growth is located within the Weldon Spring Wildlife Area south of State Route 94 near the site. Approximately 33 ha (81 acres) of this forest community is very old-growth, mesic forest community that lies within the Weldon Spring Natural Area (Figure 7.1). This community has been given watch list status by the Missouri Department of Conservation. No legal status is associated with watch list status; it is given to species or communities of possible concern for which the department is seeking further information. The other high-quality communities are the very high-quality dry forest and savannah communities on limestone (chert) soils located in the Weldon Spring Wildlife Area northwest of the Weldon Spring quarry. These communities contain old-growth vegetation, and the dominant trees (primarily oaks) often exceed 50 cm (20 in.) in diameter at breast height. The savannah — which contains old-growth black oak, post oak, and some unusual plants — is considered undisturbed and has been classified as rare by the Missouri Department of Conservation (Gaines 1988). No unique or high-quality natural areas have been identified in either the Busch Wildlife Area or the Howell Island Wildlife Area; however, the island does provide suitable habitat for bald eagles that may overwinter along the Missouri River in the vicinity of the Weldon Spring site.

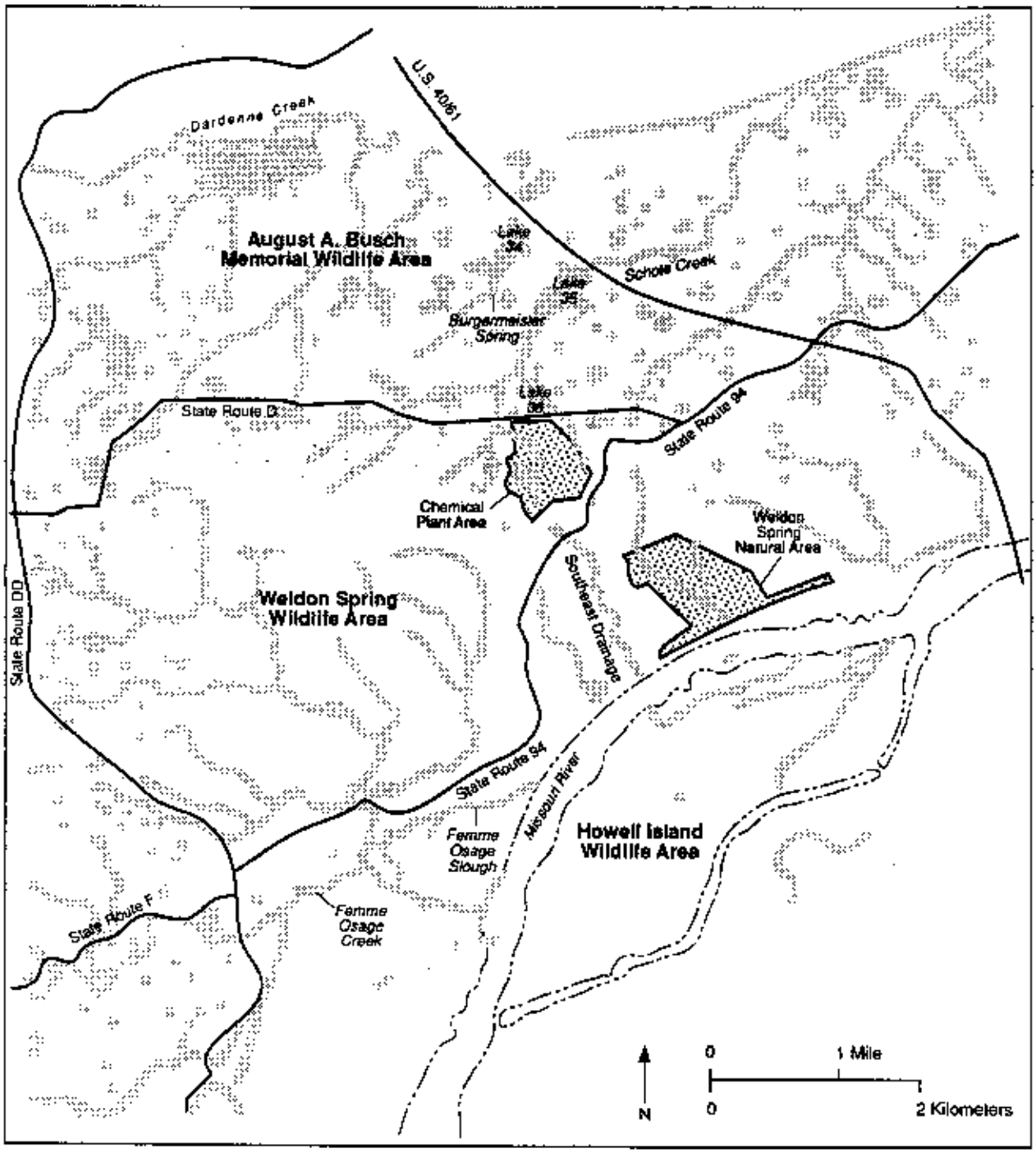


FIGURE 7.1 Wetland and Riparian Areas at the Busch Wildlife Complex

7.1.3.2 Wetlands and Riparian Corridors

A variety of wetland habitats occur at the site and vicinity, including many palustrine and riverine wetlands associated with surface waters (U.S. Fish and Wildlife Service 1989). Wetlands in the Busch Wildlife Complex include the palustrine wetlands associated with the many lakes and ponds totaling approximately 244 ha (603 acres) (Missouri Department of Conservation 1989). On-site wetlands total about 15 ha (38 acres) and include the raffinate pits, Ash Pond, and Frog Pond (U.S. Fish and Wildlife Service 1989). Riparian corridors are associated with the many intermittent and permanent streams that occur throughout the wildlife complex. These corridors typically contain numerous plant species and are extensively utilized by wildlife for foraging. They also serve as a source of food for stream organisms and help to stabilize streambanks. The palustrine wetlands provide habitat for a variety of organisms and are especially important for waterfowl and shorebirds.

Several areas in the Busch Wildlife Complex are actively managed as wetland habitat for waterfowl and shorebird use. These areas include two shorebird areas totaling 5.3 ha (13 acres), portions of seven lakes in the Busch Wildlife Area, and 312 ha (772 acres) of Missouri River bottomland in the Weldon Spring Wildlife Area (Missouri Department of Conservation 1989).

7.1.4 Threatened and Endangered Species

The Missouri Department of Conservation Natural Heritage Database (Gaines 1988; Dickneite 1988; Figg 1991) has identified 13 state endangered and 19 state rare species for St. Charles County (Table 7.5). Five of the state listed species are also federal listed as threatened or endangered, and another five are federal candidate (C2) species. However, no federal threatened or endangered species, candidate species, or critical habitats have been identified by the U.S. Fish and Wildlife Service as occurring at the Weldon Spring site (Tieger 1988; Nash 1990). The Natural Heritage Database includes 11 additional species from St. Charles County as state watch list species or of undetermined status (Table 7.5).

The Natural Heritage Database includes six federal and/or state listed species on or near the Busch Wildlife Complex: the bald eagle, sicklefin chub, sturgeon chub, Cooper's hawk, wood frog, and arrow arum. Also reported for the area are the pallid sturgeon (federally endangered), starwort (federally threatened), eastern massasauga and alligator snapping turtle (both federal C2 candidate species), and transient occurrences of the interior least tern and peregrine falcon (both federally endangered) (Gaines 1988). Overwintering bald eagles in the area may roost overnight at the Howell Island Wildlife Area (Gaines 1988). Although several federal listed species have been identified as potentially occurring in the area, no critical habitats for any of these species have been identified in the vicinity of the Weldon Spring site.

The federally endangered pallid sturgeon and the federally threatened starwort have been reported in or along the Missouri River near the Weldon Spring site. In addition, there are recent (post-1950) records of the sturgeon chub and sicklefin chub in the Missouri River at the

TABLE 7.5 Threatened, Endangered, or Special Concern Species Reported from St. Charles County

Species	Federal[a]	State[b]
Plants		
Adder's tongue fern (variety)	-	Watch list
Arrow arum	-	Rare
Bugseed (variety)	-	Watch list
Forbes saxifrage	3C	Watch list
Rose turtlehead (variety)	3C	Endangered
Salt meadow grass	-	Undetermined
Star duckweed	-	Rare
Starwort	Threatened	Endangered
Fish		
Alabama shad	-	Rare
Alligator gar	-	Rare
Brown bullhead	-	Rare
Paddlefish	3C	Watch list
Pallid sturgeon	Endangered	Endangered
Pugnose minnow	-	Watch list
Sicklefin chub	C2	Rare
Starhead topminnow	-	Watch list
Sturgeon chub	C2	Rare
Western sand darter	-	Watch list
Reptiles and amphibians		
Alligator snapping turtle	C2	Rare
Blanding's turtle	-	Endangered
Eastern massasauga	C2	Endangered
Northern crawfish frog	-	Watch list
Western fox snake	-	Endangered
Western smooth green snake	-	Endangered
Wood frog	-	Rare
Birds		
American bittern	-	Endangered
Bachman's sparrow	C2	Endangered
Bald eagle	Endangered	Endangered
Barn owl	-	Rare
Black-crowned night heron	-	Rare
Cooper's hawk	-	Rare
Henslow's sparrow	-	Rare
Interior least tern	Endangered	Endangered
Little blue heron	-	Rare
Loggerhead shrike	C2	Watch list
Mississippi kite	-	Rare
Northern harrier	-	Endangered
Osprey	-	Extirpated
Peregrine falcon	Endangered	Extirpated

TABLE 7.5 (Cont.)

Species	Federal[a]	State[b]
Birds (cont.)		
Pied-billed grebe	-	Rare
Red-shouldered hawk	-	Watch list
Sharp-shinned hawk	-	Rare
Snowy egret	-	Endangered
Upland sandpiper	-	Watch list
Yellow-headed blackbird	-	Rare
Mammals		
Long-tailed weasel	-	Rare

[a] C2, federal candidate for listing as a threatened or endangered species; 3C, former federal candidate species. A hyphen indicates that no federal status has been established.

[b] Special concern species include those classified by the state as rare, on the watch list, or status undetermined. Watch list contains species of possible concern for which the Missouri Department of Conservation is seeking further information; this listing does not imply that these species are imperiled. Undetermined indicates that the species is possibly rare or endangered but insufficient information is available to determine the proper status. Extirpated means formerly occurred as a regular breeding species but no longer reproduces in Missouri.

Sources: Dickneite (1988); Gaines (1988); Bedan (1991); Figg (1991).

U.S. Route 40/61 crossing of the Daniel Boone Bridge (Gaines 1988), approximately 5 km (3 mi) downstream of the confluence of the Southeast Drainage with the Missouri River. The Missouri Department of Conservation (1989) management plan for the Busch Wildlife Complex also reports the sicklefin chub from the Missouri River at the Daniel Boone Bridge crossing. These chubs are both federal candidate (C2) species and are listed as rare by the state. These species and the pallid sturgeon are restricted to the open channels of large turbid rivers and do not enter tributary streams (Pflieger 1975). Thus, these fish could occur in the open channels of the Missouri River that are contiguous with the Weldon Spring Wildlife Area; they would not be expected to enter the Southeast Drainage proper or to occur in any of the aquatic habitats at the site or in the Busch Wildlife Complex.

The Cooper's hawk, a state rare species, nests in pine plantations at the Busch Wildlife Complex (Missouri Department of Conservation 1989). The wood frog is classified by the state as rare and is reported from the Busch and Weldon Spring wildlife areas (Gaines 1988; Saladin 1989). In Missouri, this species is generally associated with wooded hillsides and breeds in small, fishless woodland ponds and pools. Arrow arum, a state rare plant species with two state sightings, may occur along Dardenne Creek north of Interstate 70. The pied-billed grebe, a state

rare species, has been reported from raffinate pits 2 and 4 (MK-Ferguson Company and Jacobs Engineering Group 1992c).

Two Blanding's turtles have been collected recently from the Busch Wildlife Area (Bedan 1991). This species has previously been known only from the extreme northeastern and northwestern corners of the state. It is listed as endangered by the state of Missouri because of its low population numbers and reduced natural habitat (Johnson 1987). Blanding's turtle is a small semiaquatic species that inhabits marshes, sloughs, ponds, and drainage ditches. Important components of these habitats are abundant aquatic and emergent vegetation and a mud bottom. Neither the raffinate pits nor the Southeast Drainage appears to represent suitable habitat for this species. Ash Pond also represents poor habitat because of the absence of year-round water. Frog Pond and Femme Osage Slough are potentially suitable habitats for the Blanding's turtle, and biological characterization studies planned for the site will include surveys for the presence of this species.

7.2 OVERVIEW OF HABITAT CONTAMINATION

A variety of contaminated habitats are present at the Weldon Spring site and at several off-site locations, and biota using these habitats could be exposed to radioactive and chemical contaminants. The contaminated media of primary concern to ecological resources include soil, surface water, and sediment. The nature and extent of contamination in these media at both on-site and off-site locations are discussed in detail in Chapter 5 of the RI report (DOE 1992c) and are summarized in Chapter 2 of this BA. Chapter 2 also provides information on the sampling and analytical methodologies of site characterization and monitoring efforts, the quality control measures used to analyze these data, and the screening methodology used to select contaminants for inclusion in the human health assessment. Therefore, the following discussion highlights information for the local habitats that have been impacted by past site operations and subsequent contaminant releases.

7.2.1 On-Site Terrestrial Habitats

Several terrestrial habitats are present at the Weldon Spring site. These habitats include the grassland and shrub old-field habitats south of the Ash Pond area, including the South Dump; the forested habitat in the northern portion of the Ash Pond Area, including the North Dump and much of the Frog Pond area; and the mowed grassland habitat at the remainder of the site (Figure 7.2). In addition, the abandoned buildings, many of which are contaminated, provide roosting and potential nesting habitat for a number of bird species (e.g., Canada goose, rock dove, and house sparrow) and may also provide suitable habitat for some small mammals, reptiles, and amphibians. The primary radioactive contaminants reported from on-site terrestrial habitats are uranium, thorium, and radium (Table 2.4). Uranium occurs at generally low levels across the site, with local hot spots in forest (North Dump) and grassland/shrub (South Dump) habitats; thorium is present in similar hot spot locations in the grassland/shrub and mowed grassland habitats, and radium has been detected in soil at some locations around the process

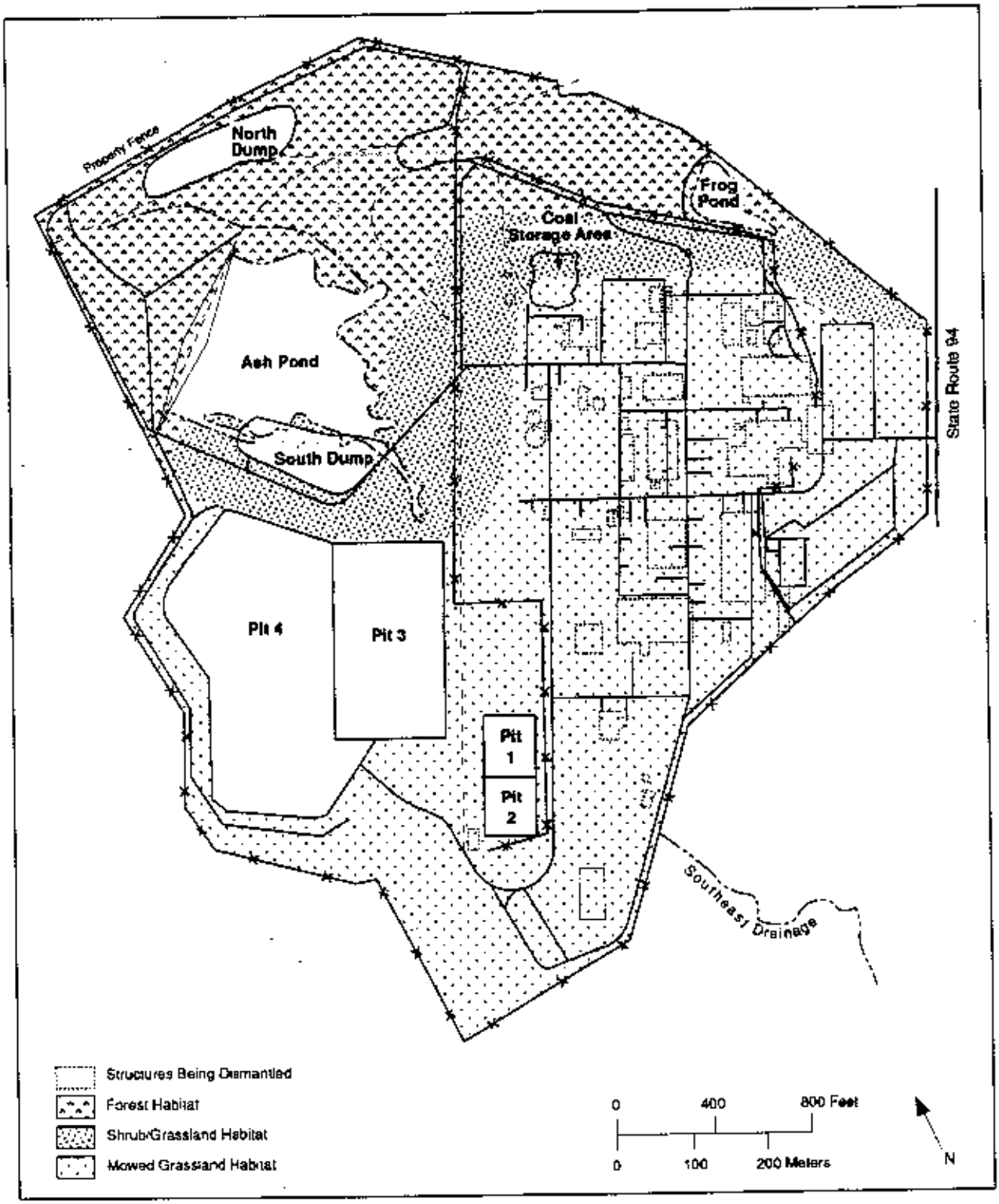


FIGURE 7.2 Terrestrial and Aquatic Habitats at the Weldon Spring Site

buildings. Chemical contaminants are also present in these habitats at generally low levels, with similarly localized hot spots (e.g., PCBs on former transformer pads) (Table 2.5).

7.2.2 On-Site Aquatic Habitats

The aquatic habitats at the site include the four raffinate pits; Ash Pond, Frog Pond, and their surface drainages; and the drainages that comprise the on-site portion of the head of the Southeast Drainage. The locations of the raffinate pits and ponds are shown in Figure 7.2; the drainages are indicated on Figure 1.6. In addition, some of the grassland areas may periodically flood in spring and are used by waterfowl (Hlohowskyj 1990). The surface water and sludge in the raffinate pits are the most highly contaminated of these habitats, containing high levels of thorium in addition to uranium, radium, and various metals and inorganic anions (see Chapter 2, Table 2.7). The ponds and drainages are also contaminated but at much lower levels, primarily with uranium and inorganic anions such as nitrate and chloride. Characterization data for these areas of lesser contamination are presented in Chapter 5 of the RI report (DOE 1992c).

7.2.3 Off-Site Terrestrial Habitats

Potentially contaminated terrestrial habitats within the Busch Wildlife Complex are those associated with the bottomland forest areas and riparian corridors along the Southeast Drainage and Burgermeister Spring; along some shoreline areas of Lakes 34, 35, and 36; and along the drainage channels that receive runoff from the site. In addition, some contaminated vicinity properties occur at several locations within the wildlife complex (Figure 1.7). However, these areas are typically small and provide little habitat for local biota.

The primary radioactive contaminant detected in soil of these riparian habitats is uranium. The limited data available for some shoreline areas of Lakes 34, 35, and 36 have also shown elevated levels of uranium (Table 2.9). Additional nearby terrestrial habitat occurs at the adjacent Army NPL site, for which a separate characterization and evaluation effort is under way. Hence, the off-site chemical characterization conducted for the Weldon Spring site has focused on areas potentially impacted by the site, as indicated by early radiological surveys. The levels of chemical constituents in those potentially impacted areas are generally similar to levels at locations not known to be impacted by the site.

7.2.4 Off-Site Aquatic Habitats

A large number of aquatic habitats are present in the vicinity of the Weldon Spring site (Figure 1.6); some of these have been impacted by site releases. Only elevated levels of radionuclides at these habitats are directly associated with the site because site runoff contributes only a fraction of the flow in the drainage areas for the lakes and streams in these wildlife areas. That is, they also receive runoff from the adjacent and much larger Army property that was previously used for ordnance production.

Aquatic habitats in the Busch Wildlife Area that have been impacted by the site include Lakes 34, 35, and 36; Burgermeister Spring; and the drainages that receive site runoff (MK-Ferguson Company and Jacobs Engineering Group 1989b). Uranium is the primary radionuclide detected at these locations. The levels of chemical constituents at these locations are generally similar to those in other areas not impacted by the site. Two creeks — Dardenne Creek (Table 2.11) and Schote Creek (Section 5.3 of the RI) — receive inflow from these lakes and/or from the Ash Pond and Frog Pond drainages, and the characterization data for these creeks suggest that they have not been impacted by the site.

In the Weldon Spring Wildlife Area, the aquatic habitat in the Southeast Drainage has been contaminated as a result of past operational discharges and subsequent runoff (Tables 2.10 and 2.12). Levels of uranium, thorium, and radium in the drainage are elevated because of site releases. For chemicals, the low levels present are not necessarily associated with the Weldon Spring site because the site contributes to only a small percent of the flow in this drainage. For example, nitroaromatic compounds in the surface water appear to be derived at least in part from sources within the drainage (pipe remnants) associated with past Army activities. (Additional characterization is planned for the Southeast Drainage, and the data collected will be included in documentation to be prepared within the next several years.)

7.3 CONCEPTUAL BIOUPTAKE AND FOOD CHAIN TRANSFER OF CONTAMINANTS

The potential exposure pathways and the assessment of contaminant fate and transport described for human receptors (Section 3.2) were developed in parallel, to the extent possible, with the identification of potential exposures to biotic resources at the Weldon Spring site. The site media that function as the source terms for the local biota are contaminated soil, sediment, surface water, and to a lesser degree groundwater. The radioactive and chemical contaminants in these media can enter biological systems through a variety of exposure routes, depending on the physical and chemical characteristics of each contaminant, the media in which the contaminants occur, the receptor species, and the physiological and ecological characteristics of the potentially affected biota.

The bioavailability of contaminants depends strongly on the specific media with which the contaminants are associated, the physicochemical characteristics of the medium, and the chemical form of the contaminant. The fate and transport of the ecological contaminants of concern are expected to be similar to the fate and transport identified in Chapter 3 for the contaminants of human concern. Biota may be exposed to contaminants through various routes: direct contact with contaminated media and subsequent uptake across surface membranes (absorption); ingestion of contaminated media during intake of food or water; and inhalation of airborne contaminants derived from soil or water (Figure 7.3). In addition, biota could be exposed to external gamma radiation.

Additional pathways of contaminant exposure to biota can occur through food chain transport, and the ultimate fate of the contaminants depends on the type (terrestrial or aquatic)

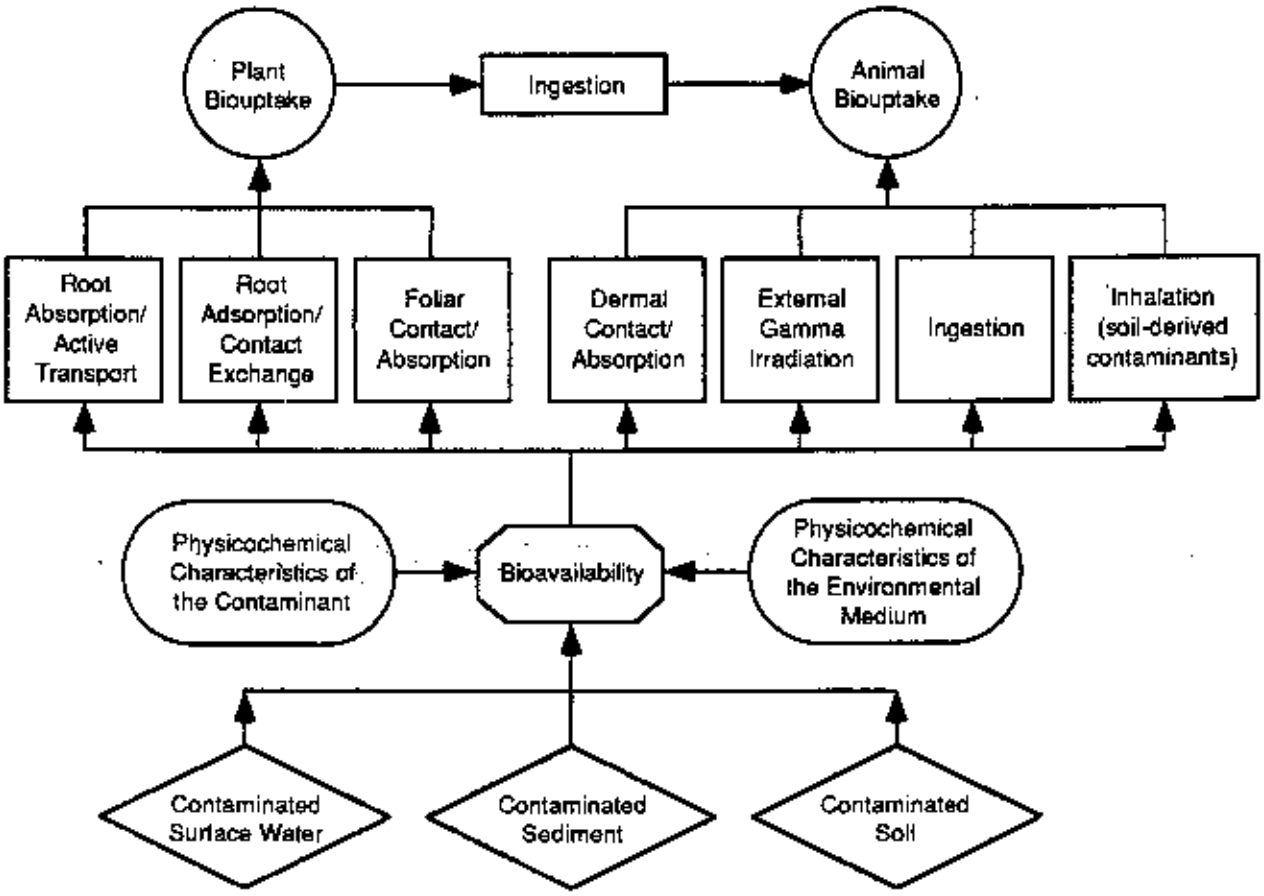
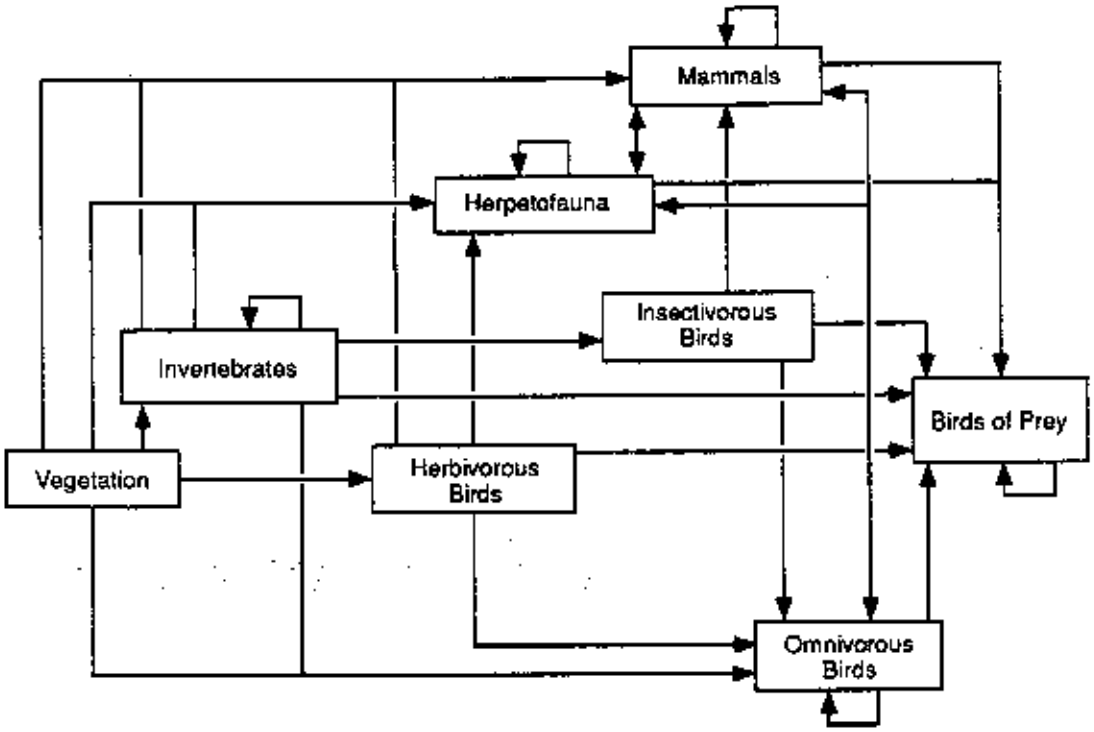


FIGURE 7.3 Schematic of Interrelationships among Contaminants, the Physicochemical Environment, and Biota that Can Influence Bioavailability and Biouptake

and form (grassland, forest, stream, or lake) of the ecosystem in which the food chain occurs, the number of trophic levels in the food chain, the concentration of the contaminant in the medium, the length of exposure to the contaminant by the biota, and the physiological and ecological characteristics of the species that comprise the food chain. A simplified example of a food chain in a contaminated terrestrial habitat in the Weldon Spring area is shown in Figure 7.4. This food chain illustrates several potential exposure pathways for terrestrial biota, beginning with the uptake of contaminants by plants. The transport of contaminants to terrestrial and aquatic habitats is discussed in Chapter 3 of this BA and in Chapter 6 of the RI (DOE 1992c).

The primary mechanisms of contaminant entry into terrestrial food chains are typically root and foliar uptake and adsorption by vegetation, with subsequent contaminant entry into higher trophic levels as a result of food chain transfer. Contaminants can also enter higher trophic levels through direct ingestion or inhalation, or by dermal contact and adsorption. Because many terrestrial invertebrates and vertebrates have body coverings that are relatively impervious to direct absorption (e.g., chitinous exoskeletons of insects or dermal scales of reptiles), contaminant entry into these forms probably occurs less through absorption across body



The pathways among invertebrates, herpetofauna, and mammals have been simplified; although not depicted, contaminants can directly enter any biotic compartment via ingestion or inhalation of contaminated media or via dermal absorption.

FIGURE 7.4 Conceptual Contaminant Food Chain Pathways for Terrestrial Biota: Vegetation to Avifauna Consumers

surfaces than through ingestion. However, for burrowing species — such as earthworms, ants, mice, ground squirrels, and woodchucks — absorption from direct contact may represent an important route of contaminant entry.

In simple food chains, contaminants taken up by vegetation are transferred directly to the primary consumers, represented by herbivorous and omnivorous invertebrates and vertebrates. Food chain transport of contaminants to higher trophic levels follows the same initial pathways to the primary consumers, but transport from primary consumers to higher level consumers becomes more complex and is more accurately depicted as a food web.

Contaminants within a single compartment in these food webs can be transported to a number of higher trophic levels, and a single high-trophic-level consumer may receive contaminants from a number of lower trophic level sources. For example, contaminants in the herbivorous invertebrate compartment (e.g., grasshopper) can be transferred to a number of higher trophic level groups such as predatory invertebrates (spiders), predatory herpetofauna (garter snakes), omnivorous mammals (raccoons), predatory mammals (shrews), omnivorous birds (crows), or predatory birds (American kestrels). Contaminants will also move from each trophic level back to the physical environment via decomposers that feed on contaminated dead and decaying biota, thereby liberating some contaminants for recycling in the ecosystem.

The food chain movement of contaminants among different trophic levels in aquatic systems is similar to that in terrestrial systems. The media of concern in aquatic systems are water and sediment. In contrast to terrestrial systems, however, direct contact with and absorption across body surfaces represents a major route of contaminant entry into aquatic biota; gill membranes represent a particularly important site of contaminant uptake (Boudou and Ribeyre 1989).

In terrestrial and aquatic ecosystems, species typically utilize resources and occupy trophic levels solely within each system. The semiaquatic biota — such as wetland vegetation, waterfowl, and shorebirds — utilize resources from terrestrial and aquatic systems and thus may be exposed to contaminants from both systems (e.g., contaminants in the soil, water, and sediment). Thus, food chains that include semiaquatic species may have more routes for contaminant transfer than food chains that are solely terrestrial or aquatic in nature.

Two of the major groups of semiaquatic biota in the Weldon Spring area are waterfowl and shorebirds. Waterfowl — such as Canada geese, mallards, and wood ducks — occur at the site and at off-site lakes where they utilize aquatic and terrestrial habitats for shelter, foraging, resting, and nesting. Shorebirds (e.g., solitary sandpipers) forage at many of the contaminated aquatic habitats present in the Weldon Spring area — including the raffinate pits, Ash Pond, and Lakes 34, 35, and 36 — where they presumably feed on invertebrate prey inhabiting the sediment. Waterfowl and shorebirds are preyed upon by a number of different terrestrial or aquatic predator groups and may thus potentially transfer contaminants to terrestrial and aquatic predators.

7.4 POTENTIAL ECOLOGICAL RECEPTORS

7.4.1 Previous Investigations

Sediment, water, and selected organisms from the site impoundments were sampled in 1977 and analyzed for total uranium, thorium-230, thorium-232, and radium-226 (RETA 1977). Concentrations of uranium-238, uranium-235, and radium-226 were reported to be elevated in vertebrate, invertebrate, and plant samples from Frog Pond and Ash Pond. Some evidence of biouptake and bioaccumulation of radionuclides was reported for algae, invertebrates, and rooted macrophytes. Uranium levels detected in macrophyte roots collected from Frog Pond and Ash Pond were similar to uranium levels detected in the sediments of these ponds. Some biouptake of radioisotopes was also reported in bluegills collected from Lakes 35 and 36 at the Busch Wildlife Area. The results of this study suggest that some degree of biouptake and food-chain transfer of contaminants is occurring at the site (RETA 1977), but the sample sizes were too small to permit definitive conclusions.

In 1983 and 1984, the Missouri Department of Conservation examined radionuclide concentrations in the edible portions (fillets) of several fish species collected from Femme Osage Slough and Lakes 34, 35, and 36 (unpublished data). Radionuclides (3.0 to 4.9 pCi/g wet weight

maximum gross beta activity) were detected in all species sampled, and concentrations were similar within taxonomic groups collected among all sampled habitats.

A biouptake study was initiated in 1987 to investigate human exposure pathways of some chemical and radioactive contaminants from the ingestion of fish and game taken at and near the site (MK-Ferguson Company and Jacobs Engineering Group 1989e). Fish and small mammals were collected for the study, as follows: 75 sunfish, 25 largemouth bass, 12 catfish, and 32 crappie from Lakes 34, 35, 36, and 37 (Lake 37 is beyond the influence of the site); 3 largemouth bass, 9 crappie, 4 buffalo, and 4 carp from Femme Osage Slough; 24 sunfish from Frog Pond; and 8 rabbits and 3 squirrels from the chemical plant area. Composite samples of the edible portions of these fish and mammals were analyzed for total uranium, CLP metals, PCBs (fish only), isotopic thorium, and radium-226. No radioactive contaminants were reported for the fish and mammal samples at concentrations above detection limits, and no PCBs were detected in the fish. Some fish samples had detectable levels of arsenic, chromium, lead, and other metals.

In 1989, the Missouri Department of Conservation sampled fish from Lakes 35, 36, and 37 in the Busch Wildlife Area and analyzed the samples for arsenic, cadmium, lead, mercury, and chlordane (Dickneite 1990). Arsenic (up to 160 ppb), cadmium (up to 26 ppb), lead (up to 120 ppb), mercury (up to 70 ppb), and chlordane (up to 35 ppb) were reported from these samples, but the levels of these contaminants in the lake surface water or sediment were not determined. The levels in lakes impacted by site releases (i.e., Lakes 34, 35, and 36) did not differ appreciably from those in Lake 37.

Several waterfowl (1 goose and 4 ducks) were collected from the raffinate pits in 1990 and analyzed for radioactive contaminants (MK-Ferguson Company and Jacobs Engineering Group 1992c). Some radionuclides were detected in each sample, but the extent of raffinate pit use by these waterfowl is unknown. The radiological doses for these waterfowl were estimated from the reported values on the basis of conservative assumptions, including (1) the radionuclides in each decay series are in secular equilibrium, (2) the energy liberated by each decay within tissues is totally absorbed by the organism, (3) radioactive contaminants are distributed homogenously within tissues at the maximum concentrations measured in each tissue sample, and (4) all measurements below analytical detection limits are equal to the detection limit. The dose estimates are well below 1 rad/d (Monette 1992), which is the limit specified in DOE Order 5400.5 for the protection of native aquatic animal organisms. It is not known if the collected specimens were year-round residents, seasonal residents, or transient migratory individuals. Thus, with the small sample size and the uncertain exposure time, the data are inconclusive regarding radionuclide biouptake or bioaccumulation* by waterfowl using the raffinate pits.

---

*Biouptake is the process of contaminant uptake by an organism either through direct absorption and/or adsorption from the environment or through uptake via the food chain. Bioaccumulation represents the concentration of a contaminant in an organism at any given moment and is the difference between biouptake and contaminant decontamination (elimination or transformation) by the organism.

7.4.2 Ecological Receptors

Direct exposure of biota to groundwater contamination would principally affect deep-rooted plants. Direct impacts to biota from contaminated soil would be expected to affect primarily rooted vegetation and burrowing invertebrates and vertebrates (e.g., earthworms, moles, and some salamanders). Exposure to contaminated surface water and groundwater following discharge to surface water could affect a variety of aquatic organisms, including fish, herpetofauna, zooplankton, algae, rooted and nonrooted plants, and aquatic macroinvertebrates. Waterfowl, shorebirds, and wading birds could also be affected, as could terrestrial species that drink contaminated surface water. Exposure to contaminated sediment could impact burrowing aquatic organisms, rooted macrophytes, sediment-foraging birds (waterfowl, shorebirds, and wading birds), benthic fish (bullheads and carp), and herpetofauna (frogs and turtles) that heavily utilize the sediment-water interface of aquatic habitats. In addition, contaminated sediment and surface water might also affect the eggs, embryos, and fry of invertebrates, fish, and amphibians.

No site-specific data are available regarding the distributions and population sizes of species, their trophic relationships, or the structure of the terrestrial and aquatic communities. About 160 species of plants and animals have been reported from the site (Tables 7.1, 7.2, and 7.3), each of which represents a potential ecological receptor. Because potentially suitable habitats occur throughout the Busch Wildlife Complex and in portions of the site, and because only some site-specific surveys for biota have been conducted to date at the contaminated habitats, many of the species reported from St. Charles County or the Busch Wildlife Complex (Appendix C) are potential ecological receptors for this assessment. Studies in the planning stage for the site are being designed in accordance with the new EPA risk assessment guidance (EPA 1991d). These data will be used in future assessments for the project.

7.4.3 Threatened and Endangered Species

The pied-billed grebe, a state rare bird, is the only federal or state listed species reported from the Weldon Spring area (Section 7.1.4) known to occupy or utilize contaminated habitats on-site; thus, no other listed species are directly threatened by site contamination. For example, the pallid sturgeon (federally endangered) and the sicklefin and sturgeon chubs (both C2 species) have been reported from the Missouri River within 5 km (3 mi) downstream of the confluence of the Southeast Drainage. Because these species are restricted to large river channels, they do not enter the Southeast Drainage and thus are not expected to be exposed to the contaminants present in the drainage. These species may, however, be exposed to contaminated sediment and surface water entering the Missouri River from the Southeast Drainage (e.g., during storm events). Similarly, the bald eagle is not expected to utilize the Southeast Drainage directly but might forage at or near its mouth and thus be exposed to site contaminants entering the river.

As part of the environmental monitoring program for the Weldon Spring site, the Missouri River has been sampled for uranium at locations approximately 610 m (2,000 ft) upstream and downstream of the mouth of the Southeast Drainage. Between 1989 and 1991,

concentrations reported from the downstream location ranged from 2.0 to 9.52 pCi/L whereas upstream concentrations ranged from 2.0 to 7.48 pCi/L (MK-Ferguson Company and Jacobs Engineering Group 1990a, 1991, 1992c). In contrast, the maximum total uranium concentration reported in surface water in the Southeast Drainage is 1,200 pCi/L (Table 2.10). These data suggest that dilution within a short distance downstream of the inflow of the Southeast Drainage will greatly reduce the surface water concentrations of contaminants entering the Missouri River from the drainage and thus reduce the potential for adverse effects to the listed biota and other species that may be present. No information is available on the levels of contaminants in the surface water or sediment at the confluence of the Southeast Drainage and the Missouri River. The radiation dose to freshwater fish from exposure to contaminated water and sediment at the confluence was estimated with conservative assumptions (Monette 1992; see Section 7.4.1), and the resulting dose is less than the 1 rad/d limit specified in DOE Order 5400.5 for protection of native aquatic organisms. The conservative assumptions included (1) the radionuclides in each decay series are in secular equilibrium, (2) the energy liberated by each decay within tissues is totally absorbed by the organism, (3) radioactive contaminants are distributed homogeneously within tissues, (4) organisms are continuously exposed to the maximum radionuclide concentration reported in water samples from each water body, and (5) all measurements below analytical detection limits are equal to the detection limit.

The federally endangered bald eagle may occasionally feed on fish from Lakes 34, 35, and 36 in the Busch Wildlife Area and thus could potentially be exposed to contaminants at these locations via food chain uptake (see Figure 7.5; in this figure, contaminated media include surface water, sediment, sludge, soil, and vegetation). The bald eagle may also feed on waterfowl (Kozie and Anderson 1991), and food chain transfer of contaminants from waterfowl

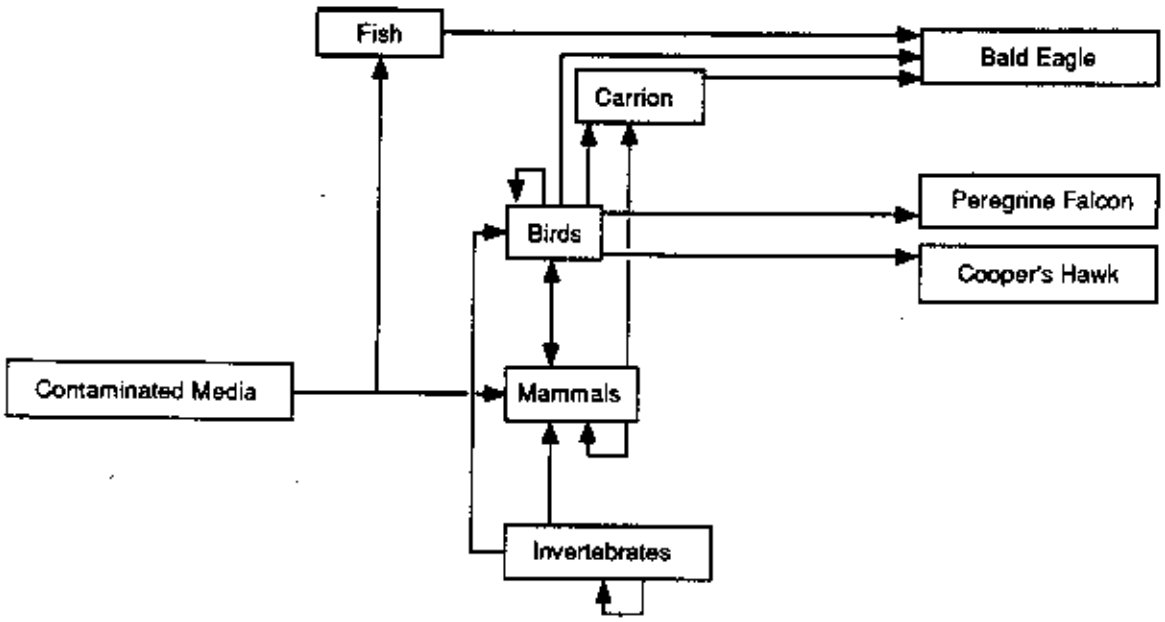


FIGURE 7.5 Conceptual Contaminant Food Chain Transfer to the Bald Eagle, Peregrine Falcon, and Cooper's Hawk

to the bald eagle has previously been reported in the literature (Pattee and Hennes 1983). If contaminant biouptake and bioaccumulation are occurring in waterfowl using the raffinate pits, Ash Pond, or other contaminated aquatic habitats, the bald eagle could be affected by foraging on contaminated waterfowl. A similar potential exists for food chain transfer of contaminants to the bald eagle from ingestion of contaminated carrion or hunter-killed game, such as raccoons, pheasants, turkeys, and rabbits (Craig 1990); from ingestion of contaminated media such as surface water; or from absorption across body surfaces (Figure 7.5).

The interior least tern (federally endangered) is an occasional summer resident in the area. This species, which forages on small fish and insects, is known to visit the Busch Wildlife Area and could potentially be affected by contaminants present in the sediment of Lakes 34, 35, and 36. For example, the water level in Lake 35 was much reduced during part of 1990 (Hlohowskyj 1990) because of a leak in the reservoir, and large areas of lake sediment were exposed. Thus, the interior least tern (and other shorebirds and wading birds) could be exposed to contaminated sediment in these lakes while foraging in shallows on exposed mud flats. Also, contaminant uptake could occur through direct dermal contact and/or ingestion of contaminated media and through food chain transfer from prey inhabiting contaminated sediment.

The federally endangered peregrine falcon, a rare transient species that occasionally summers in the area, is known to visit the Busch Wildlife Area. The state rare Cooper's hawk is known to nest in the Weldon Spring Wildlife Area. Although no suitable habitats occur at the site for these species, they could potentially be exposed to site contaminants through food chain transfer of contaminants (Figure 7.5). Both species feed on birds, and they could prey on the rock doves, starlings, and house sparrows that may feed, roost, and possibly nest among the buildings and grounds of the chemical plant area, as well as on species that utilize the forest and grassland habitats at the site. The transient nature of the peregrine falcon in the area suggests a very limited contaminant exposure potential for this species, and the Cooper's hawk has not been reported at the site. The pied-billed grebe, a state rare species, has been observed on raffinate pits 2 and 4 (MK-Ferguson Company and Jacobs Engineering Group 1992c). These individuals have thus been directly exposed to surface water contaminants, and they may be further exposed to site contaminants through food chain transfer. However, the pied-billed grebe is probably a transient migratory visitor at the raffinate pits, and thus would experience a limited contaminant exposure potential.

Blanding's turtle, a state endangered species, has recently been found in the Busch Wildlife Area and thus could potentially find suitable habitat at Frog Pond and/or Femme Osage Slough. If present, this species would be in direct contact with contaminated water and sediment, in which case dermal contact and absorption could be a major pathway of contaminant uptake. The wood frog, a state rare species that occurs in the Busch Wildlife Complex, may occur in the forested northwestern portion of the site and potentially use the raffinate pits and Ash Pond for breeding. If present, this species would be exposed to contaminated soil and surface water.

7.5 CONTAMINANTS OF CONCERN FOR THE ECOLOGICAL RISK ASSESSMENT

The radioactive and chemical contaminants at the site were evaluated with regard to potential ecological effects. The results of this evaluation indicated that the contaminants of potential concern for the ecological assessment included all those identified for the human health assessment (Tables 2.16 and 2.17), with the addition of aluminum. Although aluminum was screened from inclusion in the quantification of potential health impacts, biotic receptors are more sensitive than humans to this element. Therefore, it was included in the initial list of contaminants of potential ecological concern for this assessment.

For the human health assessment, toxicological profiles of chemicals have been established by the EPA to address potential dose-response relationships. Standard toxicity values (slope factors and reference doses) have been derived from these data to estimate potential carcinogenic and noncarcinogenic health effects associated with estimated exposures. No corresponding values are available for the ecological assessment, and toxicity data needed to quantify potential ecological impacts associated with modeled or measured contaminant levels in site biota are not available. For this reason, the ecological assessment for the site cannot be quantified in a manner that parallels the human health assessment. Nevertheless, the ecotoxicology literature was reviewed to compile data from various studies that have addressed potential impacts associated with the contaminants of concern identified for the Weldon Spring site. These studies are summarized in tabular form in Appendix D, and the information from these studies has been used to qualitatively evaluate the potential for adverse ecological effects resulting from site contaminants.

Determining potential adverse effects for each contaminant on the basis of a review of available literature is difficult. Although levels of contaminants associated with adverse biological effects have been reported in the literature for many of the contaminants of concern (Appendix D), comparing these critical levels to the concentrations present at the Weldon Spring site is inconclusive for several reasons. First, a meaningful comparison of contaminant concentrations in water and soil at the site with those concentrations and effects reported in the literature is difficult because the site data, e.g., relative to species present, are incomplete and the specific environmental conditions of the studies and the site differ. Furthermore, critical contaminant concentrations are reported in the literature as concentrations in experimental soil, food, nutrient solution, drinking water, tissues, or as injections, making it difficult to compare these values to concentrations in site media such as soil and surface water. Finally, in many of the literature studies, reconstituted water* was used to control for the effects of water chemistry and to avoid or limit synergistic effects of mixed contaminants.

*Reconstituted water refers to water prepared in the laboratory by adjusting deionized and distilled water to a specific pH and hardness through the addition of known amounts of various chemicals.

7.6 ECOLOGICAL RISK ASSESSMENT

7.6.1 Radiological Risks

The available literature regarding environmental concentrations of uranium, thorium, and radium that may produce adverse effects in natural populations of vegetation, fish, and terrestrial wildlife is concerned primarily with uptake rather than effect (Appendix D, Table D.1). Literature data concerning the movement, concentration, and effects of uranium and related elements on aquatic organisms are "sparse" (Waite et al. 1990). Thus, it was not possible to fully assess the potential for adverse risks to site biota from the reported concentrations of radioactive contaminants in soil, raffinate pit sludge, sediment, and surface water.

Except for studies by Parkhurst et al. (1984) and Waite et al. (1990), no information was found regarding adverse effects of radionuclides on aquatic or semiaquatic biota. The uranium-238 concentration of 3,060 pCi/L (9,000 µg/L) reported by Parkhurst et al. (1984) for chronic toxicity to early life history stages of brook trout is about twice the maximum concentration of 1,400 pCi/L (4,100 µg/L) reported for this radionuclide (derived from gross beta activity) in surface water in the Weldon Spring area. Waite et al. (1990) compared the health of whitefish and northern pike from contaminated and control areas of Lake Athabasca, Saskatchewan, Canada. The contaminated surface water from that lake contained 5.4 pCi/L of radium-226, <1.0 pCi/L of lead-210, 2.3 µg/L of uranium (Waite et al. 1989), and trace amounts of thorium-228; compared with the control areas, the levels of these radioactive contaminants were elevated in tissues of northern pike and whitefish, but there was no evidence that chronic exposure was detrimental to the physiology and health of the fish populations in Lake Athabasca (Waite et al. 1990). Although bioconcentration of uranium, radium, and thorium has been reported for fish and aquatic vegetation (see Gilbert et al. 1989), no evidence exists that the radionuclides of concern biomagnify in aquatic systems (Swanson 1983; Poston and Klopfer 1988); therefore, ecological risks to species at higher trophic levels from biomagnification, e.g., of uranium, are likely to be very low. For example, Swanson (1983) noted that the radionuclide contents of fish decreased from lower to higher trophic levels, and Poston and Klopfer (1988) concluded that fish occupying higher trophic levels in freshwater systems accumulated less uranium than those occupying lower trophic levels.

The tissue concentrations of radionuclides reported for fish collected from Lakes 34, 35, and 36 of the Busch Wildlife Area near the Weldon Spring site (MK-Ferguson Company and Jacobs Engineering Group 1989e; Ferguson 1991; Missouri Department of Conservation 1991) are all within the range of tissue concentrations reported in the literature for fish collected from contaminated and background locations and with which no adverse effects were associated (Appendix D). Furthermore, the estimated daily radiological dose to freshwater fish in surface water at or near the Weldon Spring site (Monette 1992) is well below the daily dose limit of 1 rad/d for protection of aquatic animals. Thus, no adverse ecological impacts from the radiotoxic effects of uranium are anticipated for aquatic biota inhabiting the surface water at the Weldon Spring site.

7.6.2 Chemical Risks

7.6.2.1 Soil

A potential for adverse ecological effects was identified from the literature for 8 of the 21 metal contaminants of concern: arsenic, cadmium, copper, lead, mercury, thallium, uranium, and zinc (Table 7.6). This determination is tentative and is based on a review of the toxicological literature (Appendix D, Table D.3), which indicated that the concentrations of these metals in soil at some on-site locations exceed soil concentrations that have been reported to produce adverse chemotoxic effects in some terrestrial species. Therefore, adverse ecological effects are possible, particularly at site locations with elevated concentrations of these contaminants ("yes" in Table 7.6). At other locations on the site, soil concentrations of these metals are lower than the levels reported to cause adverse impacts to biota and are within the common range of U.S. soil concentrations (Table 2.4).

Although no chemotoxic or radiotoxic effects on plants are known from exposure to uranium, uranium and its salts are reported to be highly chemotoxic to animals (Berlin and Rudell 1986). Chemotoxic effects from exposure include renal failure, dermatitis, and acute arterial lesions (Morrow et al. 1982; Burkart 1991). Uranium concentrations up to 6,200 mg/kg have been detected in on-site soil, whereas local background soil concentrations average 3.6 mg/kg and typical concentrations for U.S. soils range from 0.9 to 4.2 mg/kg. Because of the very high soil concentrations and the known chemotoxic nature of uranium, a potential for adverse ecological effects has been identified for uranium at some areas of the Weldon Spring site.

Although aluminum concentrations of 50 mg/kg of soil have been shown to adversely affect vegetation, aluminum toxicity is dependent on soil pH. At the pH levels reported for soil in the Weldon Spring area, typically greater than pH 6.5 (DOE 1992c), the aluminum present in the on-site soil is not expected to pose a toxicological threat to local biota. Adverse synergistic effects between aluminum and other metals have been reported to result in nutrient deficiencies and decreased growth in some plants (Spiers 1990).

No evidence was found in the literature to suggest that the on-site soil concentrations reported for silver pose a threat to local vegetation or wildlife. Terrestrial vegetation has been found to be very resistant to silver toxicity, whereas a very high intake is needed to produce acute and chronic effects in animals (Petering and McClain 1991).

For the remaining 11 metal contaminants of concern, insufficient literature information was found to adequately assess the potential for adverse ecological effects from the contaminant levels at the site (Table 7.6). However, the on-site soil concentrations for six of these metals — barium, beryllium, chromium, lithium, nickel, and vanadium — are within the concentration ranges reported for U.S. soils (Table 2.4), so the potential for adverse ecological effects from the on-site concentrations is considered unlikely. In contrast, antimony, cobalt, manganese, molybdenum, and selenium have been detected at some on-site locations at soil concentrations

TABLE 7.6 Potential Risks to Terrestrial Biota from the Chemical Soil Contaminants at the Weldon Spring Chemical Plant[a]

Contaminant	On-Site Soil Concentration (mg/kg)	Lowest Observed Effects Concentration[b]	Potential for Adverse Ecological Effects[c]
Metals			
Aluminum	660-160,000	50 mg/kg as soil addition	No[d]
Antimony	6.4-110[f]	NA[e]	Undetermined
Arsenic	1.3-130[f]	25 mg/kg in soil; 50 mg/kg diet	Yes
Barium	25-5,200	NA	Unlikely[g]
Beryllium	0.51-5.5	1 mg/L in nutrient solution	Unlikely[g]
Cadmium	0.51-11[f]	4 mg/kg in soil; 1.8 mg/kg diet	Yes
Chromium	2.2-310	≤5 mg/L in nutrient solution; 5.1 mg/kg diet	Unlikely[g]
Cobalt	2.8-110[f]	11 mg/kg tissue concentration	Undetermined[h]
Copper	3.6-460[f]	50 mg/kg in soil	Yes
Lead	1.3-1,900[f]	34 mg/kg in soil; 0.05 mg/kg diet	Yes
Lithium	5.3-71	NA	Unlikely[g]
Manganese	3.3-13,000[f]	60 mg/kg in tissue ash dry weight	Undetermined[h]
Mercury	0.11-2.1[f]	0.5 mg/kg body weight, daily dose	Yes
Molybdenum	4.1-120[f]	NA	Undetermined
Nickel	5.6-270	NA	Unlikely[g]
Selenium	0.63-47[f]	1 mg/kg diet	Undetermined[h]
Silver	0.92-13[f]	60 mg/kg in tissue, dry weight	No
Thallium	1.0-80[f]	10 mg/kg body weight	Yes
Uranium	0.9-6,900[f]	0.42 mg/L in nutrient solution	Yes
Vanadium	7.2-380	6 mg/L in nutrient solution	Unlikely[g]
Zinc	6.1-1,100[f]	>170 mg/kg in soil; 0.02 mg/L in nutrient solution	Yes
Inorganic anions			
Fluoride	1.3-45	NA	Undetermined
Nitrate	0.54-3,800	NA	Undetermined
Nitrite	1.5-29	NA	Undetermined

TABLE 7.6 (Cont.)

Contaminant	On-Site Soil Concentration (mg/kg)	Lowest Observed Effects Concentration[b]	Potential for Adverse Ecological Effects[c]
PAHs	0.39-33	50 mg PAH/kg body weight oral dose	Undetermined[h]
PCBs	0.01-12[i]	3 mg/kg diet; 26 mg/kg in soil	Undetermined
Nitroaromatic compounds			
DNB	1.0-3.8	NA	Undetermined
2,4-DNT	0.83-6.3	NA	Undetermined
2,6-DNT	1.6-3.5	NA	Undetermined
NB	1.6-3.8	2.0 mg/L in nutrient solution	Undetermined[h]
TNB	0.63-5.7	NA	Undetermined
TNT	1.3-32[j]	NA	Undetermined
Asbestos	ND[e]	NA	Undetermined

[a] The biota for which the effects were reported, as well as the nature of the observed effects, are presented in Appendix D.

[b] The lowest reported concentration, found in a review of available literature, at which a lethal or sublethal biological response was observed (see Appendix D). The lowest observed effects concentration may be for vegetation, invertebrates, or vertebrates.

[c] No — on-site contaminant concentration is not expected to produce adverse ecological effects; Yes — at least one on-site sample exceeded the soil concentration known to produce an adverse effect; Undetermined — insufficient data were found to determine the potential for adverse effects.

[d] Aluminum toxicity is pH dependent. No adverse ecological effects are anticipated because aluminum toxicity will be reduced at the pH levels reported for on-site soil.

[e] NA = no information was found in the literature review; ND = not detected.

[f] The maximum on-site concentration exceeds the common range reported for U.S. soils.

[g] Unlikely — adverse ecological effects are considered unlikely because the on-site soil concentration is within the common range reported for U.S. soils.

[h] The lowest observed effects concentrations are tissue, diet, or nutrient solution concentrations and are not directly comparable to the reported on-site soil concentrations.

[i] One sample was measured at 1,900 mg/kg.

[j] One sample was measured at 650,000 mg/kg.

exceeding the ranges for these metals in U.S. soils. For example, on-site soil concentrations of selenium are as much as 21 times higher than the concentrations reported for typical U.S. soils. A variety of plants are known to bioaccumulate selenium despite low soil concentrations. Tissue concentrations up to 46,000 mg/kg ash weight have been reported in the literature for some plants, and the accumulation of selenium in plants is believed to pose a threat to herbivorous wildlife (Eisler 1985). Plants of the genus *Astragalus* are reported to be selenium accumulators; eight species in this genus occur in the state, three of which are reported from St. Charles County (Guyette 1992). Biotic surveys recently completed at the site did not identify any selenium-accumulating plant species (Reitinger 1992); therefore, no adverse ecological effects are expected from this pathway.

Little or no literature information was found regarding the effects of inorganic anions or organic and nitroaromatic compounds on terrestrial vegetation and wildlife, so no determination of potential adverse ecological effects can be made at this time. Although no information was found regarding the effects of asbestos on wildlife, no adverse impacts are expected at the Weldon Spring site because the buildings containing asbestos are being dismantled and the asbestos is being stored in containers.

7.6.2.2 On-Site Surface Water

All on-site surface water is used by a limited number of aquatic and semiaquatic species. In the absence of any remediation of site contamination, some degree of biotic exposure would continue. However, except for a relatively diverse vegetative community in Ash Pond and Frog Pond (compared with the raffinate pits) and relatively large numbers of waterfowl at certain times of the year, the on-site aquatic community (e.g., in the raffinate pits) is very depauperate — perhaps as a result of existing contamination — and adverse impacts would be limited to a few biota.

The concentrations of chemical contaminants detected in on-site surface water were compared with available water quality standards, including (1) the EPA (1986b) water quality criteria, which apply to protection of aquatic freshwater biota but not to semiaquatic species such as waterfowl; (2) the state of Missouri water quality standards for protection of aquatic life (Missouri Department of Natural Resources 1992); and (3) the DOE standard of 550 pCi/L derived for uranium-238 and uranium-234 combined in discharges to uncontrolled surface waters. Ten metals — arsenic, beryllium, cadmium, total chromium and chromium VI, copper, lead, mercury, selenium, silver, and uranium — plus nitrate have been detected in at least one sample from on-site surface water in the raffinate pits at concentrations exceeding these criteria (Table 7.7). Therefore, a potential exists that some aquatic species could be adversely affected.

Arsenic, cadmium, lead, selenium, and silver are the most likely on-site surface water contaminants to adversely affect aquatic biota because the reported maximum concentrations of these metals greatly exceed the EPA and/or state of Missouri water quality criteria for the protection of aquatic biota (EPA 1986b; Missouri Department of Natural Resources 1992). A potential for adverse effects has also been identified for nitrate because the EPA (1986b) identifies

TABLE 7.7 Potential Risks to Aquatic Biota from the Chemical Contaminants Reported from On-Site and Off-Site Surface Waters in the Weldon Spring Area[a]

Contaminant	Surface Water Concentration (µg/L)		Lowest Observed Effects Concentration[c]	Potential for Adverse Ecological Effects[d]	
	On-Site[b]	Off-Site[b]		On-Site	Off-Site
Metals					
Aluminum	230-520	210-1,000	34 µg/L	No[e]	No[e]
Antimony	65-400	70-76	9,000 µg/L acute; 1,600 µg/L chronic	No	No
Arsenic	12-120	12-29	20 µg/L chronic[f]	Yes	Yes
Barium	ND[g]	78-110	>50,000 µg/L	No	No
Beryllium	7.0-9.0	ND	130 µg/L acute; 5 µg/L chronic[f]	Yes	No
Cadmium	37	ND	3.9 µg/L acute; 1.1 µg/L chronic	Yes	No
Chromium	31-190	14-26	62 µg/L acute[f]; 42 µg/L chronic[f]	Yes	No
Chromium III	28-170	13-23	3,100 µg/L acute; 370 µg/L chronic	No	No
Chromium VI	3.1-19	1.4-2.6	16 µg/L acute; 11 µg/L chronic	Yes	No
Cobalt	ND	ND	NA[g]	No	No
Copper	30-45	ND	34 µg/L acute; 21.4 µg/L chronic	Yes	No
Lead	22-450	9.5-15	130 µg/L acute[f]; 7.7 µg/L chronic	Yes	Yes
Lithium	61-4,500	ND	NA	Undetermined	No
Manganese	16-33	18-870	1,500 µg/L	No	No
Mercury	0.29-0.36	0.35-1.3	2.4 µg/L acute; 0.012 µg/L chronic	Yes	Yes
Molybdenum	690-4,100	22-42	NA	Undetermined	Undetermined
Nickel	47-170	ND	2,500 µg/L acute; 280 µg/L chronic	No	No
Selenium	7.5-220	ND	10 µg/L chronic[f]	Yes	No
Silver	25-40	4.0-6.0	8.2 µg/L acute[f]; 0.12 µg/L chronic	Yes	Yes
Thallium	ND	33	1,400 µg/L acute; 40 µg/L chronic	No	No
Uranium, total	4.4-5,100	6.0-1,800	570 µg/L	Yes	Yes
Vanadium	90-2,100	ND	NA	Undetermined	No
Zinc	26-60	21-78	210 µg/L acute; 190 µg/L chronic	No	No
Inorganic anions					
Fluoride	230-19,000	170-600	15,000 µg/L	Yes	No
Nitrate	190-7,200,000	300-260,000	90,000 µg/L	Yes	Yes
Nitrite	[illegible][h]	40	60 µg/L	Undetermined	No

TABLE 7.7 (Cont.)

Contaminant	Surface Water Concentration (µg/L)		Lowest Observed Effects Concentration[c]	Potential for Adverse Ecological Effects[d]	
	On-Site[b]	Off-Site[b]		On-Site	Off-Site
PCBs	ND	ND	2.0 µg/L acute; 0.014 µg/L chronic	No	Undetermined
Nitroaromatic compounds					
DNB	ND	0.18-0.81	42,500 µg/L EC_{50}; 5,800 µg/L LC_{50}	No	No
2,4-DNT	ND	0.3-11	330 µg/L acute; 230 µg/L chronic	No	No
2,6-DNT	ND	0.19-18	330 µg/L acute; 230 µg/L chronic	No	No
NB	ND	0.87	27,000 µg/L acute	No	No
TNB	0.04-1.4	0.02-84	2,400 µg/L EC_{50}; 1,000 µg/L LC_{50}	No	No
TNT	0.8-7.5	0.05-110	11,100 µg/L EC_{50}; 2,800 µg/L LC_{50}	No	No
Asbestos	-	-	NA	Undetermined	Undetermined

[a] The biota for which the effects were reported, as well as the nature of the observed effects, are presented in Appendix D.

[b] On-site: for raffinate pits and NPDES sampling locations NP-0002, NP-0003, and NP-0004 combined. Off-site: for Lakes 34, 35, and 36; Burgermeister Spring; and the Southeast Drainage (including NP-0001 and NP-0005) combined.

[c] The lowest reported concentration, found in a review of available literature, at which a lethal or sublethal biological response was observed. The reported concentrations may be for phytoplankton, invertebrates, fish, or aquatic macrophytes. Concentrations reported as acute or chronic are the EPA (1986b) water quality criteria concentrations established for the protection of freshwater biota, unless otherwise noted. All hardness-dependent criteria concentrations were calculated with a value of 200 mg/L $CaCO_3$ (DOE 1992c). EC_{50} is the concentration at which a specified effect is observed in 50% of the test population; LC_{50} is the concentration lethal to 50% of the test population.

[d] Assumes potential for adverse effects if reported surface water concentration equals or exceeds the lowest observed effects concentration. Undetermined indicates that insufficient information on ecotoxicological effects was found to determine a potential for adverse effects.

[e] Aluminum toxicity is pH dependent. No adverse ecological effects are anticipated because aluminum toxicity is expected to be reduced at the pH levels reported for on-site and off-site surface water.

[f] State of Missouri water quality criteria for the protection of aquatic life (Missouri Department of Natural Resources 1992).

[g] ND = not detected; NA = no observed effects concentration found.

[h] A hyphen indicates that the parameter was not assayed in water.

levels above 90,000 µg/L as posing a potential threat to freshwater biota whereas concentrations in the pits have been detected as high as 7,200,000 µg/L. Fluoride has been detected in the raffinate pit water at concentrations up to 19,000 µg/L, and concentrations of 15,000 µg/L (Hemens and Warwick 1972) and 17,000 µg/L (Camargo et al. 1992) have been reported to adversely affect fish and aquatic invertebrates, respectively. This would suggest a potential for adverse ecological effects to some aquatic species from exposure to this contaminant.

Selenium concentrations in water of >2 to 5 µg/L have been identified to be of concern for waterfowl (Lemly and Smith 1987). The toxic effects of selenium on waterfowl include reproductive failure or mortality as a result of bioconcentration or biomagnification through the food chain. More than 15 species of waterfowl have been reported from the raffinate pits, at numbers exceeding 100 individuals at any one time (DOE 1992c), although the full extent to which these surface waters are used by waterfowl for foraging, resting, and breeding is not known. However, the high concentration of selenium reported from the raffinate pit water and sludge (Table 2.7), together with the year-round use of the pits by waterfowl, suggests a potential for adverse effects to waterfowl from selenium. Specifically, selenium in aquatic systems can be biomagnified 500 times from water to producers (algae and vascular plants) and 2 to 6 times from producers to lower consumers such as invertebrates and forage fish (Lemly and Smith 1987). Therefore, if a fish were present in the pits, it could accumulate selenium at 4 times the concentration in its food. This food (e.g., invertebrates) could, in turn, have accumulated selenium at 500 times the concentration in water and/or sediment. Thus, the total bioconcentration factor for a fish in this case would be 2,000. Even if water concentrations of selenium were low, organisms higher in the trophic system could accumulate toxic levels via their diet (Lemly and Smith 1987).

Aluminum concentrations as low as 34 µg/L have been reported to adversely affect fish, and the concentrations in the raffinate pit water range from 230 to 520 µg/L (Table 7.7), suggesting a potential for adverse ecological effects. However, the toxicity of aluminum in freshwaters is pH-dependent, and at the pH levels measured in the raffinate pits (higher than a pH of 7), no potential adverse effects to biota from aluminum are expected.

Seven metals — antimony, barium, chromium III, manganese, nickel, thallium, and zinc — are present in concentrations below their respective water quality criteria (Table 7.7). Thus, no adverse impacts are expected to aquatic biota from these metals. Similarly, no potential adverse ecological effects were identified for nitroaromatic compounds and PCBs. Only TNB and TNT have been detected in the raffinate pit water. The concentrations of these compounds are well below the levels reported to adversely affect daphnids and fish (Table 7.7) and, therefore, are not expected to pose a threat to aquatic biota.

No literature studies were found that examined the effects of lithium, molybdenum, and vanadium on aquatic biota, and no analyses have been performed to date for nitrite in on-site surface water. Thus, the potential for adverse ecological effects of these contaminants at the site is not known.

7.6.2.3 Off-Site Surface Water

Among the metals, potential adverse effects to aquatic biota off-site were identified for arsenic, lead, and mercury in Lakes 34, 35, and 36; mercury in Burgermeister Spring; and lead and silver in the Southeast Drainage (Table 7.7). The maximum reported concentrations for these contaminants exceed the state of Missouri (Missouri Department of Natural Resources 1992) or the EPA (1986b) water quality criteria for the protection of freshwater biota. No obvious adverse ecological effects have been observed or reported from these surface waters, and the source of these contaminants has not been identified. The concentration of uranium (for which no water quality criterion has been established) in water from the Southeast Drainage has been detected as high as 1,800 µg/L. Adverse chemotoxic effects to aquatic biota (*Daphnia* spp.) have been reported at uranium concentrations of 570 µg/L (see Appendix D). Thus, the surface water concentration of uranium may pose a threat to aquatic biota in the Southeast Drainage system. Flow in this drainage is intermittent, and uranium concentrations vary with precipitation events. Recently, these levels have appeared to be decreasing, possibly as a result of interim cleanup actions at the site. No literature information was found regarding the effects of molybdenum on aquatic biota, so no determination for potential impacts can be made at this time. Beryllium, cadmium, cobalt, copper, lithium, nickel, selenium, and vanadium have not been detected in the off-site surface waters, and the concentrations of the remaining metals were below the water quality criteria (Table 7.7). Thus, no potential adverse effects from these latter metals would be expected.

Nitrate has been detected in off-site surface water at concentrations up to 260 mg/L in the Southeast Drainage and up to 203 mg/L at Burgermeister Spring; concentrations above 90 mg/L may adversely affect aquatic biota (EPA 1986b). No PCBs or PAHs were detected in off-site surface water, and all measured concentrations of nitroaromatics are below levels reported to adversely affect some aquatic biota (Table 7.7); no adverse effects to aquatic biota are expected from these contaminants or from nitrite.

7.6.2.4 Waterfowl

A number of semiaquatic biota, such as waterfowl, occur in the Weldon Spring area. Individuals using the contaminated on-site and off-site surface waters are exposed to radioactive and chemical contaminants, and they could potentially be at risk from exposure to these contaminants. However, few data are available regarding the extent to which biota use contaminated habitats, the means by which the contaminants might enter these biota (e.g., ingestion, inhalation, or dermal absorption), the extent to which the contaminants are taken up, and the levels at which the contaminants begin to cause lethal or sublethal effects. Thus, it is not possible at this time to quantify potential adverse effects from most of the contaminants to waterfowl using these habitats. A conservative approach would be to assume that waterfowl use the raffinate pits throughout the year and are continuously exposed to contaminants in the pits. Of particular concern in this regard is exposure of waterfowl to selenium (Lemly and Smith 1987) in on-site surface water and sediment and, potentially, in aquatic vegetation (Section 7.6.2.2). Thus, use of the on-site surface water habitats represents a potential risk to waterfowl.

7.6.3 Biomagnification

In terrestrial systems, evidence of biomagnification has been reported for mercury and PCBs; in aquatic systems, evidence of biomagnification for the contaminants of concern has been reported for cadmium, mercury, selenium, and PCBs. Although PCBs have not been detected in on-site or off-site surface water and only a few soil locations (adjacent to concrete pads) appear to be contaminated (Table 2.17), the extent to which biomagnification may be occurring in the Weldon Spring area is unknown. Biota occupying higher trophic levels in the Weldon Spring area — such as the bald eagle and numerous other birds of prey, piscivorous fish, water snake, raccoon, and red fox — could potentially be at risk from biomagnification of these contaminants. In general, risk depends on the levels of contaminants in the media, the levels of contaminants in edible tissues of preferred prey, and the age and reproductive status of the predator. The risk to transient species such as the peregrine falcon would be low relative to resident predatory species because this falcon rarely (if ever) visits the site.

7.7 UNCERTAINTIES

Available data in the literature have been reviewed, and the contaminants of concern at the Weldon Spring site have been evaluated for their potential to adversely impact biotic resources in the Weldon Spring area. The EPA (1989a) guidance states that three types of information are needed to establish a relationship between environmental contaminants and observed ecological effects:

- Chemical analyses of the appropriate media are needed to establish the presence, areal and vertical distributions, and concentrations of the contaminants.
- Ecological surveys are necessary to establish whether adverse ecological effects have occurred.
- Toxicity tests are necessary to identify potential ecological impacts and to establish a link between any realized adverse ecological effects and the toxicity of the hazardous wastes and contaminants.

Without these data, other potential causes of the observed ecological effects, unrelated to the effects of the contaminants, cannot be eliminated.

Ecological studies at the site preceded EPA's recent guidance on ecological risk assessments. Consequently, these studies were not designed in accordance with this guidance and did not identify or quantify potential adverse ecological effects, making it difficult to backfit the available data into this new process. Furthermore, the EPA is drafting new ecological risk assessment guidance that is not expected to be available for public comment until 1994. The ecological assessment presented herein was conducted in accordance with the general intent of the process, discussing the data that are available rather than the more guidance-specific data that are not available. Studies currently planned for the site are being designed with the current

guidance in mind; resulting information will be presented in future documentation for the project.

Characterization studies at the Weldon Spring site have established the presence of radioactive and chemical contamination at a number of on-site and off-site terrestrial and aquatic habitats. Characterization efforts have sufficiently identified the nature and extent of the contamination at on-site habitats, with the possible exception of nitrite and PAHs in some on-site surface waters (Section 7.6.2.2). A decision to remove and treat these surface waters has already been made. A number of off-site areas — including Lakes 34, 35, and 36; some of the drainage channels from the site to the Busch lakes; and the Missouri River in the immediate vicinity of the confluence of the Southeast Drainage — may need additional characterization for lead, mercury, silver, thallium, uranium, and nitrite (Section 7.6.2.3). Additional characterization studies are currently planned for the Southeast Drainage; monitoring of the lakes will continue as part of an ongoing program.

Only limited ecological surveys have been conducted at the chemical plant area and at the contaminated habitats off-site. Some observational data are available regarding the occurrence of plant and animal species at the site and at off-site habitats, but no data are available regarding (1) the use of contaminated habitats by wildlife, (2) population densities in contaminated habitats, or (3) the community structure at these areas. Several biotic surveys are planned for the site, including surveys of vegetation and bird communities. Few data are available regarding use of on-site habitats by wildlife (e.g., deer and turkey) that may regularly move off-site. In particular, few data are available regarding use of the on-site aquatic habitats by migratory and resident waterfowl (Ferguson 1991), and it is not possible at this time to quantify the use of on-site aquatic habitats for foraging or nesting by migratory and resident waterfowl and shorebirds. Thus, neither the status of the on-site biota nor the specific resources that may be at risk from the site contaminants can be identified. A generalized identification of potential ecological receptors has been developed from the lists of biota reported for St. Charles County (Appendix C), and a biological monitoring plan is currently being developed.

Site-specific ecological toxicity data have not been collected for the contaminated media occurring at the chemical plant area and at off-site areas. Thus, data from the scientific literature were used to assess the potential toxicity of the contaminants of ecological concern. Information from the literature can be of limited use for at least four reasons (EPA 1989i). First, many laboratory and field studies describe the effects on a single species, and that species may not be appropriate or relevant to the site in question. In addition, because of the complications involved with fieldwork, studies often ignore species interactions. Second, a single toxicant is often studied, ignoring potential chemical interactions among toxicants. Synergistic and antagonistic interactions have been reported in the literature for some of the metal contaminants that occur at the site. In some instances, the presence of one metal may reduce the potential for adverse impacts from another metal. More commonly, individual concentrations may not elicit a biological response, but several contaminants present together may result in significant effects to biota. Third, published accounts of the types and intensities of biotic effects may not be relevant to the site in question. For example, the results of studies involving intravenous or peritoneal injections of contaminants are not directly relevant to the site because of differences

in the exposure pathways. Finally, most studies are for a fixed period of time. For example, toxicologists often study responses over 48 to 96 hours (e.g., to determine a 48-hour EC_{50} or a 96-hour LC_{50}*), depending on the species and toxicant. A longer period of time may be required before the effects of a given contaminant concentration become evident. Comparison of literature studies is further impeded because of different experimental designs, methods, and analyses. Thus, extrapolation of experimental data to site conditions can be extremely difficult, and this difficulty contributes to the overall uncertainty of the assessment (EPA 1989i).

The biouptake data that have been collected for selected biota potentially impacted by site contamination are difficult to assess. Differing methodologies make interstudy comparisons difficult (Schenkler 1989), and insufficient data regarding the residency of collected biota in contaminated areas make it difficult to draw conclusions regarding biouptake by area biota. For example, it is not known if waterfowl collected in October 1990 from the raffinate pits for radiological analyses (Ferguson 1991) were residents or were among the large number of birds that migrate past the site each autumn along the Mississippi flyway. The length of time that the analyzed individuals were exposed to the raffinate pit contaminants would affect the results of the radiological analyses. Because an active management program at the Busch Wildlife Area includes stocking, similar residency uncertainties exist for the fishes that were collected from Lakes 34, 35, and 36 for radiological and chemical analyses.

Few data are available to determine whether adverse ecological effects are occurring as a result of on-site or off-site contamination. Maximum concentrations of many of the contaminants present at the site and at some off-site locations are known to cause a range of adverse effects at sublethal concentrations. Biological parameters that have been shown to be affected by low levels of some contaminants include reproduction, growth, primary productivity, longevity, biomass, species diversity, community and trophic structure, and behavior. Such impacts may be associated with the raffinate pits under baseline conditions. Additional data on these parameters in local biota would be needed to fully identify the potential for adverse effects resulting from site contaminants.

7.8 SUMMARY AND CONCLUSIONS

The Weldon Spring site contains some biotically rich unmanaged areas that comprise about 25% of the site. In addition, the site is adjacent to two state wildlife areas and near a third. More than 200 species of plants and animals are expected to occur at the site because of the site's habitat diversity and the adjacent wildlife areas. Radioactive and chemical contaminants are present in terrestrial, aquatic, and wetland habitats at the site and in localized off-site areas. No specific potential ecological receptors have been identified for the site; however, reported on-site observations of species and the extensive list of species compiled for the area by the state of Missouri allow for the ready identification of potential receptors.

---

*EC_{50} is the concentration at which a specified effect is observed in 50% of the test population: LC_{50} is the concentration lethal to 50% of the test population.

The potential occurrence of several federal threatened, endangered, and candidate species — including the bald eagle, peregrine falcon, and pallid sturgeon — has been indicated for the area. No critical habitat occurs at the site for the bald eagle, and this species is not expected to frequent the site. The peregrine falcon is a transitory migrant through the area that could occasionally forage at the site for brief periods. Because of the absence of suitable habitat and the infrequent or transitory occurrence of these species in the site vicinity, few or no impacts are anticipated to the bald eagle and peregrine falcon. The pallid sturgeon is reported from the Missouri River and could be exposed to contaminants from the Southeast Drainage. Neither the occurrence of the pallid sturgeon nor the concentrations of all contaminants in the Missouri River in the vicinity of the Southeast Drainage are known. Because of the intermittent nature of the flows in the Southeast Drainage and the mixing of contaminants that occurs in the Missouri River, few if any impacts are anticipated to the pallid sturgeon or other aquatic biota in the area.

Several state listed species also occur in the area. Surveys for these species have been initiated at on-site and off-site locations. The pied-billed grebe, a state rare species, has been observed at raffinate pits 2 and 4. This species is probably a transient migratory visitor to the raffinate pits.

A review of the scientific literature and the available site information indicates that no adverse ecological impacts are expected to result to the biota from exposure to many of the radioactive and chemical contaminants in the site soil and surface water. For example, the levels of radioactive contaminants in on-site surface water are below the 1 rad/d dose limit for aquatic biota. Furthermore, because of high soil pH at the site, elevated concentrations of aluminum in some soil samples are not expected to pose a hazard to biota.

However, the levels of other contaminants, especially those in the raffinate pits, exceed either water quality criteria or concentrations reported in the scientific literature to adversely impact biota. For example, levels of arsenic, beryllium, cadmium, chromium, copper, lead, mercury, selenium, silver, uranium, fluoride, and nitrate pose a potential chemotoxic hazard to aquatic and semiaquatic biota. Selenium, in particular, is present at concentrations exceeding those shown to adversely affect waterfowl. Furthermore, because selenium bioconcentrates, it could potentially pose a hazard to wildlife species higher in the food chain (e.g., the federally endangered bald eagle), including species not directly exposed to selenium in soil, water, or sediment. It is possible that the presence of these contaminants contributes to the relative paucity of biota in the raffinate pits. A decision has been made to remove and treat the surface water from these pits as an interim action (Section 1.1.2), and the indication of potential adverse ecological impacts supports this previous cleanup decision.

In off-site surface water, nitrate in the Southeast Drainage and in Burgermeister Spring has been detected at levels that pose a potential for adverse impacts to aquatic biota. The concentrations of arsenic, lead, mercury, and silver reported for some samples collected at off-site surface water locations exceed the EPA water quality criteria for the protection of freshwater biota. Other metal and radioactive contaminants are at sufficiently low concentrations so that no threats to biota are anticipated. Doses estimated for fish from exposure to the maximum measured concentration of uranium in the Southeast Drainage do not exceed the limit of 1 rad/d

for protection of aquatic biota. However, uranium metal is also chemotoxic to animals, and concentrations measured in sediment and water at certain locations in the Southeast Drainage exceed levels shown to be toxic to *Daphnia* (water fleas). Thus, uranium from certain locations could potentially pose a chemotoxic threat to aquatic biota in the intermittent flows in the Southeast Drainage and in the Missouri River at the drainage confluence, in the absence of mixing. Several federal and state listed species that could occur at this location include the pallid sturgeon, sicklefin chub, sturgeon chub, and bald eagle.

At some on-site locations, soil contaminants — including arsenic, cadmium, copper, lead, mercury, uranium (metal), and zinc — have been detected at concentrations that present a potential impact to exposed biota; others — antimony, cobalt, manganese, molybdenum, and selenium — are present at concentrations exceeding the common ranges of these metals reported for U.S. soils. It is not known if the environmental concentrations of these metals may adversely impact local biota.

No obvious adverse ecological impacts have been observed to area biota, with the possible exception of the biota at the raffinate pits. Adverse ecological impacts to on-site and off-site biota could occur if exposure to on-site contaminants were to continue, particularly at the raffinate pits. To address the long-term protection of ecological resources on-site and off-site, additional studies are being planned that will allow for a more detailed assessment. These data will be incorporated into future documentation for the project, as it becomes available.

8 ASSESSMENT OF IMPACTS TO OTHER ENVIRONMENTAL RESOURCES

In addition to addressing potential impacts to human health and ecological resources in the absence of remedial action at the Weldon Spring site, this BA addresses the potential impacts of no action on other environmental resources. These resources include soil, archaeological and cultural resources, socioeconomics and land use, air, surface water, and groundwater. On the basis of current site information, no significant adverse impacts to nonecological environmental resources are expected to result from a continuation of current conditions at the site, with the possible exception of groundwater.

Soil resources have already been disturbed as a result of past site activities. No other impacts, including impacts to archaeological or cultural resources, are expected in the absence of remedial action. Because the surrounding land is owned by the federal and state governments, current socioeconomics and land-use conditions would be expected to remain the same for the time frame of this assessment (10 to 30 years). Thus, no incremental adverse impacts to these resources are expected to result from the no-action condition.

Similarly, no adverse impacts are expected to air quality because dust generation is limited by the nature of surface features at the site (e.g., vegetation, surface impoundments, buildings, and paved surfaces), local meteorology, and the surrounding vegetation and terrain. This expectation is supported by environmental monitoring results, which indicate that radon gas, radioactive particulates, and asbestos are at background levels at the site boundary.

On-site surface water impoundments have already been impacted as a result of contact with contaminated material and receipt of storm runoff over contaminated areas. Because these impoundments are currently contaminated and represent very low-quality aquatic habitat, no impacts are expected beyond those already occurring. Further, as part of a previous cleanup decision for the project, the water is scheduled to be removed from these impoundments and treated in an on-site water treatment plant for which construction is nearing completion.

Off-site surface water is currently being impacted by storm water runoff and groundwater discharge (e.g., at Burgermeister Spring and in the Southeast Drainage), and this impact would continue if no remedial action were taken. Surface runoff from the southern portion of the site flows down the Southeast Drainage channel to the Missouri River. Continued releases could result in continued sediment contamination in the drainage channel. Because the contaminant levels are generally low, little incremental impact is expected beyond that already occurring. In fact, contaminant levels in sediment and surface water could be decreasing as a result of the recent construction of a runoff control system at the head of the Southeast Drainage (as part of construction activities for the temporary storage area and site water treatment plant, under interim actions). In addition, the intermittent runoff flows could also be reducing contaminant levels over time by natural flushing. Thus, no incremental impacts would be expected to water quality in the Southeast Drainage or the Missouri River.

Runoff from the remainder of the site flows north to lakes in the Busch Wildlife Area, and contaminants have been detected in off-site surface water and sediment. However, the

levels are generally low and recent runoff control measures in the northern area of the site have reduced off-site releases. Nevertheless, some contaminants would continue to be released into the wildlife area. In addition, subsurface flows discharge to surface water off-site and contribute to the low-level contamination. However, little incremental impact to surface water quality is expected to result from no action at the site within the time frame of this assessment. Potential impacts from exposures to the slightly contaminated media off-site are addressed in the human health and ecological assessments of this BA. A wetlands assessment for on-site and off-site locations is presented in Appendix H of the FS (DOE 1992a).

Groundwater at the site has been impacted by leaching from contaminated source areas (e.g., the raffinate pits), and contaminant migration would continue and could increase if no source-control actions were implemented. Current data suggest that groundwater contamination associated with the site may be localized; contaminants are present primarily in the shallow aquifer beneath the site and beyond the site boundary, but contaminant plumes have not yet extended beyond the general vicinity of the site (see Appendix B). Although contaminants have not yet reached private drinking water wells in the area, it is possible that such wells could be impacted over time. Studies are under way to further characterize the nature and extent of groundwater contamination, and the information received from these studies will be incorporated into the RI/FS for the groundwater operable unit to be prepared within the next several years.

9 REFERENCES

Adriano, D.C., 1986, *Trace Elements in the Terrestrial Environment*, Springer-Verlag, New York.

Agency for Toxic Substances and Disease Registry, 1988a, *Toxicological Profile for Beryllium*, ATSDR/TP-88/07, prepared by Syracuse Research Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1988b, *Toxicological Profile for Nickel*, ATSDR/TP-88/19, prepared by Syracuse Research Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1989a, *Toxicological Profile for Arsenic*, ATSDR/TP-88/02, prepared by Life Systems, Inc., March.

Agency for Toxic Substances and Disease Registry, 1989b, *Toxicological Profile for Cadmium*, ATSDR/TP-88/08, prepared by Life Systems, Inc., March.

Agency for Toxic Substances and Disease Registry, 1989c, *Toxicological Profile for Chromium*, ATSDR/TP-88/10, prepared by Syracuse Research Corporation, July.

Agency for Toxic Substances and Disease Registry, 1989d, *Toxicological Profile for 2,4-Dinitrotoluene, 2,6-Dinitrotoluene*, ATSDR/TP-89/13, prepared by Clement Associates, Dec.

Agency for Toxic Substances and Disease Registry, 1989e, *Toxicological Profile for Mercury*, ATSDR/TP-89/16, prepared by Clement Associates, Dec.

Agency for Toxic Substances and Disease Registry, 1989f, *Toxicological Profile for Selected PCBs (Aroclor-1260, -1254, -1248, -1242, -1232, -1221, and -1016)*, ATSDR/TP-88/21, prepared by Syracuse Research Corporation, June.

Agency for Toxic Substances and Disease Registry, 1989g, *Toxicological Profile for Selenium*, ATSDR/TP-89/21, prepared by Clement Associates, Dec.

Agency for Toxic Substances and Disease Registry, 1989h, *Toxicological Profile for Zinc*, ATSDR/TP-89/25, prepared by Clement Associates, Dec.

Agency for Toxic Substances and Disease Registry, 1990a, *Toxicological Profile for Antimony*, prepared by Syracuse Research Corporation, draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990b, *Toxicological Profile for Asbestos*, ATSDR/TP-90/04, prepared by Life Systems, Inc., Dec.

Agency for Toxic Substances and Disease Registry, 1990c, *Toxicological Profile for Barium*, prepared by Clement Associates, Inc., draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990d, *Toxicological Profile for Cobalt*, prepared by Clement Associates, draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990e, *Toxicological Profile for Copper*, ATSDR/TP-90/08, prepared by Syracuse Research Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1990f, *Toxicological Profile for Fluoride*, prepared by Clement Associates, Inc., draft for public comment, July.

Agency for Toxic Substances and Disease Registry, 1990g, *Toxicological Profile for Lead*, ATSDR/TP-88/17, prepared by Syracuse Research Corporation, published by Oak Ridge National Laboratory, June.

Agency for Toxic Substances and Disease Registry, 1990h, *Toxicological Profile for Manganese*, prepared by Life Systems, Inc., draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990i, *Toxicological Profile for Nitrobenzene*, ATSDR/TP-90/19, prepared by Life Systems, Inc., Dec.

Agency for Toxic Substances and Disease Registry, 1990j, *Toxicological Profile for Polycyclic Aromatic Hydrocarbons*, ATSDR/TP-90/20, prepared by Clement International Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1990k, *Toxicological Profile for Silver*, ATSDR/TP-90/24, prepared by Clement International Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1990l, *Toxicological Profile for Thallium*, prepared by Life Systems, Inc., draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990m, *Toxicological Profile for Uranium and Compounds*, ATSDR/TP-90/29, prepared by Syracuse Research Corporation, Dec.

Ainsworth, N., 1988, *Distribution and Biological Effects of Antimony in Contaminated Grassland*, Dissertation. [As cited in ATSDR (1990a).]

Alexander, M., and B.K. Lustigman, 1966, *Effects of Chemical Structure on Microbial Degradation of Substituted Benzenes*, Journal of Agricultural Food Chemistry, 14:411-413.

Ambrose, A.M., et al., 1976, *Long-Term Toxicologic Assessment of Nickel in Rats and Dogs*, Journal of Food Science and Technology, 13:181-187. [As cited in EPA (1991c).]

American Cancer Society, 1992, *Cancer Facts & Figures — 1992*, Atlanta, Ga.

American National Standards Institute, 1986, *Quality Assurance Program Requirements for Nuclear Facilities*, ANSI/ASME NQA-1-1986 Edition, American Society of Mechanical Engineers, New York.

Anderson, E., et al., 1985, *Development of Statistical Distributions or Ranges of Standard Factors Used in Exposure Assessments, Final Report*, EPA/600/8-85/010 (PB85-242667), prepared by GCA Corporation, GCA/Technology Division, Chapel Hill, N.C., for U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, D.C., Aug.

Andres, P., 1984, *IgA-IgG Disease in the Intestine of Brown Norway Rats Ingesting Mercuric Chloride*, Clinical Immunology and Immunopathology, 20:488-494.

ANSI: see American National Standards Institute.

Archer, V.E., J.K. Wagoner, and F.E. Lundin, 1973, *Cancer Mortality among Uranium Mill Workers*, Journal of Occupational Medicine, 15:11-14.

Bechtel National, Inc., 1986, unpublished information, Oak Ridge, Tenn.

Bechtel National, Inc., 1987, *Hydrogeological Characterization Report for Weldon Spring Chemical Plant, Weldon Spring, Missouri*, DOE/OR/20722-137, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Oak Ridge, Tenn., July.

Bedan, D.E., 1991, letter from D.E. Bedan (Radioactive Waste Cleanup Coordinator, Division of Environmental Quality, Missouri Department of Natural Resources, Jefferson City, Mo.) to S. McCracken (Project Manager, U.S. Department of Energy, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), April 12.

Beres, D.A., 1990, *The Clean Air Act Assessment Package-1988 (CAP-88), A Dose and Risk Assessment Methodology for Radionuclide Emissions to Air, Volume 1, User's Manual*, prepared by SC&A, Inc., McLean, Va., for U.S. Environmental Protection Agency, Office of Radiation Programs, Washington, D.C., Oct.

Berlin, M., and B. Rudell, 1986, *Uranium*, in Handbook on the Toxicology of Metals, 2nd ed., L. Friberg, G.F. Nordberg, and V. Voik (editors), Elsevier Science Publishers, New York.

Bernaudin, J.F., et al., 1981, *Inhalation or Ingestion of Organic or Inorganic Mercurials Produces Auto-Immune Disease in Rats*, Clinical Immunology and Immunopathology, 20:129-135.

Boerner, A.J., 1986, *Radiological Survey of the August A. Busch and Weldon Spring Wildlife Areas, Weldon Spring Site, St. Charles County, Missouri*, Final Report, prepared by Oak Ridge Associated Universities, Oak Ridge, Tenn., for U.S. Department of Energy, April.

Bosch, H.M., et al., 1950, *Methemoglobinemia and Minnesota Well Supplies*, Journal of the American Water Works Association, 42:161-170. [As cited in EPA (1991c).]

Boudou, A., and F. Ribeyre, 1989, *Fundamental Concepts in Aquatic Ecotoxicology*, in Aquatic Ecotoxicology: Fundamental Concepts and Methodologies, Vol. I, A. Boudou and F. Ribeyre (editors), CRC Press, Inc., Boca Raton, Fla.

Brenniman, G.R., and P.S. Levy, 1984, *Epidemiological Study of Barium in Illinois Drinking Water Supplies*, in Advances in Modern Environmental Toxicology IX, E.J. Calabrese, R.W. Tuthill, and L. Condie (editors), Princeton Scientific Publications, Princeton, N.J., pp. 231-240. [As cited in EPA (1991c).]

Briggs, G.G., R.H. Bromilow, and A.A. Evans, 1982, *Relationships between Lipophilicity and Root Uptake and Translocation of Non-ionised Chemicals by Barley*, Pesticide Science, 13:495-504.

Brower, M.E., and W.R. Hartley, 1989, *Draft Health Advisory for 2,4- and 2,6-Dinitrotoluene*, U.S. Environmental Protection Agency, Office of Drinking Water, Washington, D.C.

Brown, C.C., and K.C. Chu, 1983a, *Approaches to Epidemiologic Analysis of Prospective and Retrospective Studies: Example of Lung Cancer and Exposure to Arsenic*, in Risk Assessment: Proceedings of the SIMS Conference on Environmental Epidemiology, June 28-July 2, 1982, Alta, Utah, SIAM Publication. [As cited in EPA (1991c).]

Brown, C.C., and K.C. Chu, 1983b, *Implications of the Multistage Theory of Carcinogenesis Applied to Occupational Arsenic Exposure*, Journal of the National Cancer Institute, 70:455-463.

Brown, C.C., and K.C. Chu, 1983c, *A New Method for the Analysis of Cohort Studies; Implications of the Multistage Theory of Carcinogenesis Applied to Occupational Arsenic Exposure*, Environmental Health Perspectives, 50:293-308.

Brown, J.F., Jr., et al., 1987, *Polychlorinated Biphenyl Dechlorination in Aquatic Sediments*, Science, 236:709-712.

Burger, L., 1990, memorandum from L. Burger (Missouri Department of Conservation, Jefferson City, Mo.) to L.A. DeBruyckere (Wildlife District Supervisor, St. Louis District, Missouri Department of Conservation, Weldon Spring, Mo.), April 19.

Burkart, W., 1991, *Uranium, Thorium, and Decay Products*, in Metals and Their Components in the Environment: Occurrence, Analysis, and Biological Relevance, E. Merian (editor), VCH Publications, New York, pp. 1275-1287.

Burrows, E.P., et al., 1989, *Organic Explosives and Related Compounds: Environmental and Health Considerations*, Technical Report 8901, prepared for the U.S. Army Biomedical Research & Development Laboratory/Command, Fort Detrick, Frederick, Md., March.

Camargo, J.A., J.V. Ward, and K.L. Martin, 1992, *The Relative Sensitivity of Competing Hydropsychid Species to Fluoride Toxicity in the Cache la Poudre River (Colorado)*, Archives of Environmental Contamination and Toxicology, 22:107-113.

Chappell, W.R., et al., 1979, *Human Health Effects of Molybdenum in Drinking Water*, EPA-600/1-79-006, U.S. Environmental Protection Agency, Office of Research and Development, Health Effects Research Laboratory, Cincinnati, Ohio.

Chemical Industry Institute of Toxicology, 1984, *Ninety-Day Inhalation Toxicity Study of Nitrobenzene in F344 Rats and B6C3F1 Mice*, Research Triangle Park, N.C.

Chovil, A., R.B. Sutherland, and M. Halliday, 1981, *Respiratory Cancer in a Cohort of Sinter Plant Workers*, British Journal of Industrial Medicine, 38:327-333.

Cody, T.E., et al., 1981, *1,3-Dinitrobenzene: Toxic Effect in Vivo and in Vitro*, Journal of Toxicology and Environmental Health, *7*(5):829-847.

Cowherd, C., Jr., et al., 1985, *Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites*, EPA/600/8-85/002, prepared by Midwest Research Institute, Kansas City, Mo., for U.S. Environmental Protection Agency, Office of Health and Environmental Assessment, Washington, D.C., Feb.

Craig, T.H., 1990, *Lead Concentrations in Golden and Bald Eagles*, Wilson Bulletin, *102*(1):130-133.

Crigler, D., 1992a, personal communication from D. Crigler (Wildlife Management Biologist, August A. Busch Wildlife Area, St. Charles, Mo.) to G. Valett (Project Management Contractor, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), as documented in memorandum from G. Valett to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), March 17.

Crigler, D., 1992b, personal communication from D. Crigler (Wildlife Management Biologist, August A. Busch Wildlife Area, St. Charles, Mo.) to G. Valett (Project Management Contractor, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), as documented in memorandum from G. Valett to R. Ferguson (Jacobs Engineering Group, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), May 21.

Cristy, M., et al., 1986, *Relative Age-Specific Radiation Dose Commitment Factors for Major Radionuclides Released from Nuclear Fuel Facilities*, NUREG/CR-4628 (ORNL/TM-9890), prepared by Oak Ridge National Laboratory, Health and Safety Research Division, Oak Ridge, Tenn., for U.S. Nuclear Regulatory Commission, Division of Fuel Cycle and Material Safety, Aug.

Daubel, K., 1992, personal communication from K. Daubel (U.S. Department of Army) to M. Sizemore (MK-Ferguson Company, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), as documented in memorandum from M. Sizemore to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), March 17.

DeBruyckere, L.A., 1990, letter from L.A. DeBruyckere (Wildlife District Supervisor, St. Louis District, Missouri Department of Conservation, St. Charles Co., Mo.) to I. Hlohowskyj and D. Reinhardt (Argonne National Laboratory, Argonne, Ill.), with enclosure, April 20.

Deming, D.J., 1986, *Radiological Survey, U.S. Army Reserve Property, Weldon Spring Site, St. Charles County, Missouri*, prepared by Oak Ridge Associated Universities, Oak Ridge, Tenn., for U.S. Department of Energy, Final Report, Jan.

deRoos, C., 1984, letter from C. deRoos (Missouri Department of Conservation, Jefferson City, Mo.) to G. Turi (U.S. Department of Energy, Washington, D.C.), with attachments, April 20.

Dickneite, D.F., 1988, letter from D.F. Dickneite (Environmental Administrator, Missouri Department of Conservation, Jefferson City, Mo.) to I. Hlohowskyj (Argonne National Laboratory, Argonne, Ill.), with enclosure, Aug. 24.

Dickneite, D.F., 1990, *Fish Flesh Contaminant Data*, attachment to letter from D.F. Dickneite (Environmental Administrator, Missouri Department of Conservation, Jefferson City, Mo.), to W.R. Schmidt (Missouri Department of Health, Division of Health and Epidemiology, Jefferson City, Mo.), April 9.

DOE: see U.S. Department of Energy.

Dreesen, D.R., et al., 1982, *Mobility and Bioavailability of Uranium Mill Tailings Contaminants*, Environmental Science and Technology, *16*(10):702-706.

Driver, J.H., J.J. Konz, and G.K. Whitmyre, 1989, *Soil Adherence to Human Skin*, Bulletin of Environmental Contamination and Toxicology, 43:814-820.

Druet, P., et al., 1978, *Immune Type Glomerulonephritis Induced by $HgCl_2$ in the Brown Norway Rat*, Annals of Immunology, *129C*:777-792. [As cited in EPA (1991a).]

DuBowy, P.J., 1989, *Effects of Diet on Selenium Bioaccumulation in Marsh Birds*, Journal of Wildlife Management, 53(3):776-781.

Eisler, R., 1985, *Selenium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 5, U.S. Fish and Wildlife Service Biological Report, *85*(1.5), Oct.

Ellis, H.V., et al., 1979, *Mammalian Toxicity of Munitions Compounds, Phase III: Effects of Lifetime Exposure, Part 1: 2,4-Dinitrotoluene*, Contract No. DAMD17-74-C-4073, Midwest Research Institute, Kansas City, Mo. [As cited in EPA (1991c).]

Enterline, P.E., and G.M. Marsh, 1982, *Cancer among Workers Exposed to Arsenic and Other Substances in a Copper Smelter*, American Journal of Epidemiology, *116*:895-911. [As cited in EPA (1991a).]

EPA: see U.S. Environmental Protection Agency.

Etnier, E.L., 1987, *Water Quality Criteria for 2,4-Dinitrotoluene and 2,6-Dinitrotoluene*, ORNL-6312, Final Report, Oak Ridge National Laboratory, Oak Ridge, Tenn., Aug.

Fawer, R.F., et al., 1983, *Measurement of Hand Tremor Induced by Industrial Exposure to Metallic Mercury*, Journal of Industrial Medicine, 40:204-208.

Fedoroff, B.T., and O.E. Sheffield, 1962, *Encyclopedia of Explosives and Related Items*, Vol. 2, Picatinny Arsenal, Dover, N.J., AD-422 747.

Ferguson, R., 1991, document transmittal from R. Ferguson (MK-Ferguson Company, Weldon Spring Site Remedial Action Project, St. Charles, Mo.) to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), with attachments, May 3.

Figg, D.E., 1991, letter from D.E. Figg (Endangered Species Coordinator, Missouri Department of Conservation, Jefferson City, Mo.) to J.R. Powers (MK-Ferguson Company, St. Charles, Mo.), Nov. 26.

Fisenne, I., 1992, personal communication from I. Fisenne (Environmental Measurements Laboratory, New York, N.Y.) to J. Peterson (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), May.

Freeze, R.A., and J.A. Cherry, 1979, *Groundwater*, Prentice-Hall, Inc., Englewood Cliffs, N.J.

Friberg, L., G.F. Nordberg, and V.B. Vouk (editors), 1986, *Handbook on the Toxicology of Metals, Volume II: Specific Metals*, 2nd ed., Elsevier, Amsterdam.

Furedi, E.M., et al., 1984, *Determination of the Chronic Mammalian Toxicological Effects of TNT (Twenty-four Month Chronic Toxicity/Carcinogenicity Study of Trinitrotoluene (TNT) in the Fischer 344 Rat*, Final Report - Phase III, Volume 1, IIT Research Institute, Project No. L6116-Study No. 9, Chicago, Ill., DAMD17-79-C-9120.

Gaines, E.P., 1988, letter from E.P. Gaines (Data Manager, Missouri Department of Conservation, Jefferson City, Mo.) to I. Hlohowskyj (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), with enclosure, Sept. 8.

Gardner, W.S., et al., 1979, *Degradation of Selected Polycyclic Aromatic Hydrocarbons in Coastal Sediments: Importance of Microbes and Polychaete Worms*, Water, Air, and Soil Pollution, 11:339-347.

Garstang, M., 1991, *Collapse Potential Evaluation of the Defined Study Area at the Weldon Spring Chemical Plant Site, Weldon Spring, Missouri*, in Proceedings of the Geosciences Workshop, DOE/OR/21548-197, published by MK-Ferguson Company and Jacobs Engineering Group for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., March, pp. 277-281.

Gaul, L.E., and A.N. Staud, 1935, *Clinical Spectroscopy: Seventy Cases of Generalized Argyria Following Organic and Colloidal Silver Medication*, Journal of the American Medical Association, 104:1387-1390.

Gilbert, T.L., et al., 1983, *Pathways Analysis and Radiation Dose Estimates for Radioactive Residues at Formerly Utilized MED/AEC Sites*, ORO-832 (Rev.), prepared by Argonne National Laboratory, Division of Environmental Impact Studies, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations, Oak Ridge, Tenn., March (reprinted with corrections: January 1984).

Gilbert, T.L., et al., 1989, *A Manual for Implementing Residual Radioactive Material Guidelines*, ANL/ES-160 (DOE/CH/8901), prepared by Argonne National Laboratory, Energy and Environmental Systems Division, Argonne, Ill., for U.S. Department of Energy, Assistant Secretary for Nuclear Energy, June.

Gordon, L., and W.R. Hartley, 1989, *Health Advisory on 2,4,6-Trinitrotoluene*, U.S. Environmental Protection Agency, Office of Drinking Water, Washington, D.C., Jan.

Guyette, R., 1992, personal communication from R. Guyette (University of Missouri, Columbia) to I. Hlohowskyj (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), April.

Hajjar, N.P., et al., 1989, *Draft Health Advisory for 1,3-Dinitrobenzene*, U.S. Environmental Protection Agency, Office of Drinking Water, Washington, D.C., Sept.

Halls, L.K. (editor), 1984, *White-Tailed Deer, Ecology and Management*, Stackpole Books, Harrisburg, Pa.

Hamblin, D.O., 1962, *Industrial Hygiene and Toxicology*, 2nd ed., Interscience Publishers, New York, p. 2139.

Harger, J.R.E., 1979, *A Model for the Determination of an Action Level for Removal of Curine Contaminated Soil*, memorandum from J.R.E. Harger to P.S. Cole (Executive Director, Toxic Substances Control Division, Lansing, Mich.). [As cited in EPA (1988c).]

Hartwig, C., 1992, personal communication from C. Hartwig (Secretary for Superintendent, Francis Howell High School, St. Charles, Mo.) to M. Sizemore (MK-Ferguson Company, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), as documented in memorandum from M. Sizemore to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), March 17.

Hawley, J.K., 1985, *Assessment of Health Risk from Exposure to Contaminated Soil*, Risk Analysis, 5(4):289-302.

Healy, J.W., 1971, *Surface Contamination: Decision Levels*, LA-4558-MS, Los Alamos Scientific Laboratory, Los Alamos, N.M., Sept.

Hemens, J., and R.J. Warwick, 1972, *The Effects of Fluoride on Estuarine Organisms*, Water Research, 6:1301-1308.

Higgins, I., 1982, *Arsenic and Respiratory Cancer among a Sample of Anaconda Smelter Workers*, report submitted to the Occupational Safety and Health Administration in the comments of the Kennecott Minerals Company on the inorganic arsenic rule making (Exhibit 203-5). [As cited in EPA (1991c).]

Higgins, I., K. Welch, and C. Burchfiel, 1982, *Mortality of Anaconda Smelter Workers in Relation to Arsenic and Other Exposures*, University of Michigan, Department of Epidemiology, Ann Arbor, Mich. [As cited in EPA (1991c).]

Hlohowskyj, I., 1990, personal observation of I. Hlohowskyj (Argonne National Laboratory, Argonne, Ill.) at the Weldon Spring site.

Hodge, H.C., 1950, *The Concentration of Fluorides in Drinking Water to Give the Point of Minimum Caries with Maximum Safety*, Journal of the American Dental Association, 40:436.

Hodge, H.C., J.N. Stannard, and J.B. Hursh (editors), 1973, *Uranium-Plutonium Transplutonic Elements*, Springer-Verlag, New York.

Hoffman, D., 1992, *Table 3B: Area Exposure Report of Subcontractor (WP-058) during Asbestos Abatement Activity of B401*, facsimile transmittal from D. Hoffman (Jacobs Engineering Group, Weldon Spring Site Remedial Action Project, St. Charles, Mo.) to M.M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), April 29.

Hurst, P.-F., 1990, memorandum from P.-F. Hurst (Superfund Health Risk Technology Support Center, Chemical Mixtures Assessment Branch, Office of Research and Development, U.S. Environmental Protection Agency), to D. Wall (U.S. Environmental Protection Agency, Region VII), Aug. 29.

ICRP: see International Commission on Radiological Protection.

International Commission on Radiological Protection, 1975, *Report of the Task Group on Reference Man*, ICRP Publication 23, prepared by a Task Group of Committee 2.

International Commission on Radiological Protection, 1977, *Recommendations of the Commission on Radiological Protection (Adopted January 17, 1977)*, ICRP Publication 26, Annals of the ICRP, *1*(3).

International Commission on Radiological Protection, 1979, *Limits for Intakes of Radionuclides by Workers*, ICRP Publication 30, Part 1, Annals of the ICRP, 2(3/4).

International Commission on Radiological Protection, 1980, *Limits for Intakes of Radionuclides by Workers: Statement and Recommendations of the 1980 Brighton Meeting of the ICRP*, ICRP Publication 30, Part 2, Annals of the ICRP, 4(3/4).

International Commission on Radiological Protection, 1981a, *Limits for Inhalation of Radon Daughters by Workers, Adopted March 1981*, ICRP Publication 32, Annals of the ICRP, *6*(1).

International Commission on Radiological Protection, 1981b, *Limits for Intakes of Radionuclides by Workers*, ICRP Publication 30, Part 3, Including Addendum to Parts 1 and 2, Annals of the ICRP, *6*(2/3).

International Commission on Radiological Protection, 1989, *Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 1 (Adopted April 1989)*, ICRP Publication 56, Annals of the ICRP, *20*(2).

Ivankovic, S., and R. Preussmann, 1975, *Absence of Toxic and Carcinogenic Effects after Administration of High Doses of Chromic Oxide Pigment in Subacute and Long-Term Feeding Experiments in Rats*, Food and Cosmetics Toxicology, 13:347-351.

Johnson, T.R., 1987, *The Amphibians and Reptiles of Missouri*, Missouri Department of Conservation, Jefferson City, Mo.

Kawamura, R.H., et al., 1941, *Intoxication by Manganese in Well Water*, Kitasato Archives of Experimental Medicine, *18*:145-169. [As cited in EPA (1991c).]

Kjellstrom, T., 1986, *Critical Organs, Critical Concentrations, and Whole-Body Dose-Response Relationships*, in Cadmium and Health: A Toxicological and Epidemiological Appraisal, Vol. II. Effects and Response, L. Friberg et al. (editors), CRC Press, Boca Raton, Fla., pp. 231-246.

Klaassen, C.D., M.O. Amdur, and J. Doull (editors), 1986, *Casarett and Doull's Toxicology: The Basic Science of Poisons*, 3rd ed., Macmillan Publishing Company, New York.

Kleeschulte, M.J., and P.W. Cross, 1990, *Hydrologic Data for the Weldon Spring Chemical Plant Site and Vicinity Property, St. Charles, Mo. — 1989-89*, U.S. Geological Survey Open File Report 90-552.

Kleeschulte, M.J., and L.F. Emmett, 1986, *Compilation and Preliminary Interpretation of Data for the Weldon Spring Radioactive Waste-Disposal Sites, St. Charles County, Missouri — A Progress Report*, U.S. Geological Survey Water-Resources Investigation Report 85-4272.

Kleeschulte, M.J., and L.F. Emmett, 1987, *Hydrology and Water Quality at the Weldon Spring Radioactive Waste-Disposal Sites, St. Charles, Missouri*, U.S. Geological Survey Water-Resources Investigation Report 87-4169.

Kondakis, X.G., et al., 1989, *Possible Health Effects of High Manganese Concentration in Drinking Water*, Archives of Environmental Health, *44*:175-178.

Kozie, K.D., and R.K. Anderson, 1991, *Productivity, Diet, and Environmental Contaminants in Bald Eagles Nesting near the Wisconsin Shoreline of Lake Superior*, Archives of Environmental Contamination and Toxicology, *20*:41-48.

Lee-Feldstein, A., 1983, *Arsenic and Respiratory Cancer in Man: Follow-up of an Occupational Study*, in Arsenic: Industrial, Biomedical, and Environmental Perspectives, W. Lederer and R. Fensterheim (editors), Van Nostrand Reinhold, New York.

Lemly, A.D., and G.J. Smith, 1987, *Aquatic Cycling of Selenium: Implications for Fish and Wildlife*, Fish and Wildlife Leaflet 12, U.S. Fish and Wildlife Service, Washington, D.C.

Lepow, M.L., et al., 1975, *Investigations into Sources of Lead in the Environment of Urban Children*, Environmental Research, *10*:415-426.

Levine, B.S., J.H. Rust, and J.M. Burns, 1983, *Determination of the Chronic Mammalian Toxicological Effects of TNT, Twenty-six Week Subchronic Oral Toxicity Study of Trinitrotoluene (TNT) in the Beagle Dog, Phase II*, Final Report, DAMD 17-79-C-9120, IIT Research Institute, Project No. L6116-Study No. 5, Chicago, Ill. [As cited in EPA (1991c).]

Lindberg, E., and G. Hedenstierna, 1983, *Chrome Plating: Symptoms, Findings in the Upper Airways, and Effects on Lung Function*, Archives of Environmental Health, *38*:367-374.

Lindsay, W.L., 1979, *Chemical Equilibria in Soils*, John Wiley & Sons, New York.

Liu, D., K. Thomson, and A.C. Anderson, 1984, *Identification of Nitroso Compounds from Biotransformation of 2,4-Dinitrotoluene*, Applied Environmental Microbiology, *47*:1295-1298.

MacKenzie, R.D., et al., 1958, *Chronic Toxicity Studies, II. Hexavalent and Trivalent Chromium Administered in Drinking Water to Rats*, American Medical Association Archives of Industrial Health, *18*:232-234. [As cited in EPA (1991c).]

Magnus, K., A. Andersen, and A. Hogetveit, 1982, *Cancer of Respiratory Organs among Workers at a Nickel Refinery in Norway*, International Journal of Cancer, *30*:681-685.

Mancuso, T.F., 1975, *Consideration of Chromium as an Industrial Carcinogen*, in International Conference on Heavy Metals in the Environment, Toronto, Ontario, Canada, Oct. 27-31, pp. 343-356. [As cited in EPA (1991c).]

Marutzky, S.J., R. Colby, and L.S. Cahn, 1988, *Radiological Characterization of the Weldon Spring, Missouri, Remedial Action Site*, UNC/GJ-HMWP-4 (DOE/ID/12548-22), prepared by UNC Geotech, Inc., Grand Junction, Colo., for U.S. Department of Energy, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., with Grand Junctions Projects Office, Grand Junction, Colo., Feb.

Mauskopf, J.A., 1987, *Projections of Cancer Risks Attributable to Future Exposures to Asbestos*, Risk Analysis, *7*(4):477-486.

Maynard, E.A., and H.C. Hodge, 1949, *Studies of the Toxicity of Various Uranium Compounds When Fed to Experimental Animals*, in The Pharmacology and Toxicology of Uranium Compounds, C. Voegtlin and H.C. Hodge (editors), McGraw-Hill Book Company, New York, Vol. I, pp. 309-376.

McCauley, P.T., et al., 1985, *Investigations into the Effect of Drinking Water Barium on Rats*, Environmental Health Perspectives, *IX*:197-210. [As cited in EPA (1991c).]

Mellard, D.N., 1989, *Health Consultation: Fish Data, Weldon Spring Site, St. Charles County, Missouri*, memorandum from D.N. Mellard (Toxicologist, Emergency Response Branch, Office of Health Assessment, Agency for Toxic Substances and Disease Registry) to D. Parker (Public Health Advisor, EPA Region VII, Kansas City, Mo.), Aug. 24.

metaTRACE, Inc., 1987, *Asbestos Analysis Report*, letter to K. Nowadly (Weldon Spring Site Remedial Action Project, St. Charles, Mo.), May.

Meyer, K., 1990, personal communication from K. Meyer (Jacobs Engineering Group, Weldon Spring Site Remedial Action Project, St. Charles, Mo.) to L. Haroun (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), Nov. 14.

Meyer, K., 1991, facsimile transmittal from K. Meyer (Jacobs Engineering Group, Weldon Spring Site Remedial Action Project, St. Charles, Mo.) to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), May 28.

Meyer, L., 1992, *Information on the Weldon Spring School*, facsimile transmittal from L. Meyer (Jacobs Engineering Group, Weldon Spring Site Remedial Action Project, St. Charles, Mo.) to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), Jan. 3.

Miller, J., 1991, data provided by J. Miller (Herpetologist, Missouri Department of Conservation, August A. Busch Wildlife Complex, Weldon Spring, Mo.) to J. Herron (MK-Ferguson Company, St. Charles, Mo.), Nov. 6.

Missouri Botanical Garden, 1975, *An Introduction to the Biological Systems of the St. Louis Area, Vol. I*, prepared by the Missouri Botanical Garden for East-West Gateway Coordinating Council, under contract from St. Louis District, Corps of Engineers, the Metro Study.

Missouri Department of Conservation, 1981, *A Checklist of Birds of the August A. Busch Memorial Wildlife Area, Weldon Spring, Missouri*, Jefferson City, Mo.

Missouri Department of Conservation, 1989, *Area Management Plan for the August A. Busch Complex*, Jefferson City, Mo.

Missouri Department of Conservation, 1991, unpublished data, Jefferson City.

Missouri Department of Natural Resources, 1989, *Shallow Groundwater Investigations at Weldon Spring, Missouri, Second Annual Report for Fiscal Year 1989*, prepared by Division of Geology and Land Survey, Rolla, Mo., for U.S. Department of Energy, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Dec.

Missouri Department of Natural Resources, 1992, *Rules of Department of Natural Resources: Division 20 — Clean Water Commission; Chapter 7 — Water Quality*, Code of State Regulations 10 CSR 20-7.031 (Water Quality Standards), Jefferson City, Mo.

MK-Ferguson Company, 1987, *Water Quality Phase I Assessment Report*, DOE/OR/21548-003, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Dec.

MK-Ferguson Company, 1988a, *Buildings Characterization Sampling Plan*, DOE/OR/21548-021, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Aug.

MK-Ferguson Company, 1988b, *Remedial Investigation Quality Assurance Program Plan*, DOE/OR/21548-011, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Dec.

MK-Ferguson Company, 1988c, *Waste Assessment, Raffinate Pit Sampling Plan, Weldon Spring Site*, DOE/OR/21548-010, Rev. 1, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., May.

MK-Ferguson Company, 1989a, *Chemical Soil Data Report to Support Interim Response Actions, Construction Staging Area, and Administration Building*, DOE/OR/21548-051, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Nov.

MK-Ferguson Company, 1989b, *Chemical Soil Investigation Report for the Weldon Spring Chemical Plant/Raffinate Pits, Phase II*, DOE/OR/21548-061, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, undated, *Quality Assurance Program Plan*, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo.

MK-Ferguson Company and Jacobs Engineering Group, 1986, *Asbestos Bulk Sampling and Air Sampling Results, Weldon Spring Chemical Plant, August 14-15, 1986*, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo.

MK-Ferguson Company and Jacobs Engineering Group, 1987a, *Annual Environmental Monitoring Report, Weldon Spring, Missouri, Calendar Year 1986*, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo.

MK-Ferguson Company and Jacobs Engineering Group, 1987b, *Characterization Report, IRA #13, Army Reserve Property Vicinity Property No. 2*, DOE/OR/21548-024, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Sept. 18.

MK-Ferguson Company and Jacobs Engineering Group, 1987c, *Characterization Report, IRA #13, Army Reserve Property, Vicinity Property No. 3*, DOE/OR/21548-025, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Sept. 22.

MK-Ferguson Company and Jacobs Engineering Group, 1987d, *Plan for Determination of Biological Uptake of Radionuclides and Nitroaromatics in Species in the Food Pathways at the WSS*, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., June.

MK-Ferguson Company and Jacobs Engineering Group, 1987e, *Plan for the Sampling and Analysis of Lake and Stream Sediments Influenced by the Weldon Spring Chemical Plant Drainage*, DOE/OR/21548-006, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

MK-Ferguson Company and Jacobs Engineering Group, 1987f, *Report on Radiological Findings and Recommendations Regarding Army Reserve Property, Vicinity Property No. 1*, DOE/OR/21548-023, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Oct. 8.

MK-Ferguson Company and Jacobs Engineering Group, 1987g, *WSSRAP Report on Radiological Findings and Recommendations Regarding Army Reserve Vicinity Property No. 7*, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., July.

MK-Ferguson Company and Jacobs Engineering Group, 1988a, *Annual Environmental Monitoring Report, Weldon Spring Missouri, Calendar Year 1987*, DOE/OR/21548-015, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., April.

MK-Ferguson Company and Jacobs Engineering Group, 1988b, *Chemical Soil Investigation Sampling Plan for the Weldon Spring Chemical Plant/Raffinate Pits*, DOE/OR/21548-013, Rev. 1, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., May.

MK-Ferguson Company and Jacobs Engineering Group, 1988c, *Hydrogeologic Investigations Sampling Plan*, DOE/OR/21548-020, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

MK-Ferguson Company and Jacobs Engineering Group, 1988d, *Phase I Chemical Soil Investigation Data Report for the Weldon Spring Chemical Plant/Raffinate Pits*, DOE/OR/21548-016, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Jan.

MK-Ferguson Company and Jacobs Engineering Group, 1988e, *Preliminary Draft Radiological Soil Characterization Report for the Weldon Spring Chemical Plant and Raffinate Pits Sites*, DOE/OR/21548-042, Rev. A, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

MK-Ferguson Company and Jacobs Engineering Group, 1988f, *Characterization Report, IRA #5, August A. Busch and Weldon Spring Wildlife Area Vicinity Properties 3, 4, 5, and 8*, DOE/OR/21548-022, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., May 19.

MK-Ferguson Company and Jacobs Engineering Group, 1989a, *Annual Environmental Monitoring Report 1988*, DOE/OR/21548-079, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., June.

MK-Ferguson Company and Jacobs Engineering Group, 1989b, *Characterization of Chemical and Radiological Contamination in Lake and Stream Sediments on Properties Surrounding the Weldon Spring Site*, DOE/OR/21548-060, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, 1989c, *Phase I Spring and Seep Report*, DOE/OR/21548-045, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, 1989d, *Phase II Groundwater Quality Assessment for the Weldon Spring Site, Chemical Plant, Raffinate Pits and Surrounding Vicinity Properties*, DOE/OR/21548-078, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, 1989e, *Radiological and Chemical Uptake by Edible Portions of Selected Biota at the Weldon Spring Site*, DOE/OR/21548-044, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, 1989f, *Waste Assessment Chemical Characterization of the Weldon Spring Site Raffinate Pits*, DOE/OR/21548-092, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, 1989g, *Waste Assessment Radiological Characterization of the Weldon Spring Site Raffinate Pits*, DOE/OR/21548-062, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, 1990a, *Annual Site Environmental Report 1989*, DOE/OR/21548-129, Rev. 1, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Nov.

MK-Ferguson Company and Jacobs Engineering Group, 1990b, *Asbestos and Chemical Characterization Report for Non-Process Related Buildings and Equipment*, DOE/OR/21548-070, Rev. E, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., April.

MK-Ferguson Company and Jacobs Engineering Group, 1990c, *Buildings Radiological Characterization Report*, DOE/OR/21548-120, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., April.

MK-Ferguson Company and Jacobs Engineering Group, 1991, *Annual Site Environmental Report for Calendar Year 1990*, DOE/OR/21548-193, Rev. 1, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Sept.

MK-Ferguson Company and Jacobs Engineering Group, 1992a, *Chemical Plant Site Remedial Investigation/Feasibility Study Data Validation Report*, DOE/OR/21548-256, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Feb.

MK-Ferguson Company and Jacobs Engineering Group, 1992b, *Concentration Ratios of Radionuclides in the U-238, U-235, and Th-232 Decay Series*, DOE/OR/21548-250, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Sept.

MK-Ferguson Company and Jacobs Engineering Group, 1992c, *Weldon Spring Site Environmental Report for Calendar Year 1991*, DOE/OR/21548-283, Rev. 1, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., July.

Monette, F.A., 1992, *Daily Doses to Fish in Weldon Surface Waters*, interoffice memorandum from F.A. Monette to M.M. MacDonell (Argonne National Laboratory, Environmental Assessment and Information Sciences Division, Argonne, Ill.), with attachment, April 21.

Morrow, P.E., et al., 1982, *Metabolic Fate and Evaluation of Injury in Rats and Dogs Following Exposure to the Hydrolysis Products of Uranium Hexafluoride*, NUREG/CR-2268, prepared by University of Rochester, Department of Radiation Biology and Biophysics, Rochester, N.Y., for U.S. Nuclear Regulatory Commission, Office of Nuclear Regulatory Research, Washington, D.C., Dec.

Myrick, T.E., B.A. Berven, and F.F. Haywood, 1981, *State Background Radiation Levels: Results of Measurements Taken during 1975-1979*, ORNL/TM-7343, prepared by Oak Ridge National Laboratory, Oak Ridge, Tenn., for U.S. Department of Energy, Nov.

Nash, T., 1990, letter from T. Nash (U.S. Fish and Wildlife Service, Columbia Field Office, Columbia, Mo.) to I. Hlohowskyj (Argonne National Laboratory, Argonne, Ill.) with enclosures, May 9.

National Academy of Sciences, 1977, *Drinking Water and Health*, prepared by Safe Drinking Water Committee, National Research Council, Volume 1, National Academy of Sciences, Washington, D.C.

National Climatic Data Center, 1987, *Local Climatological Data, Annual Summary with Comparative Data, St. Louis, Missouri*, Asheville, N.C.

National Council on Radiation Protection and Measurements, 1987, *Exposure of the Population in the United States and Canada from Natural Background Radiation*, NCRP Report No. 94, Bethesda, Md., Dec. 30.

National Research Council, 1988, *Health Risks of Radon and Other Internally Deposited Alpha-Emitters*, BEIR IV Report, report of the Committee on the Biological Effects of Ionizing Radiations, National Academy Press, Washington, D.C.

National Research Council, 1989, *Recommended Dietary Allowances*, 10th ed., National Academy Press, Washington, D.C.

National Research Council, 1990, *Health Effects of Exposure to Low Levels of Ionizing Radiation*, BEIR V Report, report of the Committee on the Biological Effects of Ionizing Radiations, National Academy Press, Washington, D.C.

National Toxicology Program, 1980, *Naphthalene (C52904), Fischer 344 Rats*, unpublished subchronic toxicity study, prepared by Battelle Columbus Laboratories under Subcontract no. 76-34-206002. [As cited in EPA (1991c).]

Neal, J., and R.H. Rigdon, 1967, *Gastric Tumors in Mice Fed Benzo(a)pyrene: A Quantitative Study*, Texas Reports on Biology and Medicine, 25:553-557.

Norback, D.H., and R.H. Weltman, 1985, *Polychlorinated Biphenyl Induction of Hepatocellular Carcinoma in the Sprague-Dawley Rat*, Environmental Health Perspectives, 60:97-105.

Ottolenghi, A.D., et al., 1974, *Inhalation Studies of Nickel Subsulfide in Pulmonary Carcinogenesis of Rats*, Journal of the National Cancer Institute, 54:1165-1172.

Owen, B.A., 1990, *Literature-Derived Absorption Coefficients for 39 Chemicals via Oral and Inhalation Routes of Exposure*, Regulatory Toxicology and Pharmacology, 11:237-252.

Parkhurst, B.J., et al., 1984, *An Environmental Hazard Evaluation of Uranium in a Rocky Mountain Stream*, Environmental Toxicology and Chemistry, 3:113-124.

Pattee, O.H., and S.K. Hennes, 1983, *Bald Eagles and Waterfowl: The Lead Shot Connection*, Transactions of the North American Wildlife and Natural Resources Conference, 48:230-237.

Paustenbach, D.J., 1989, *A Comprehensive Methodology for Assessing the Risks to Humans and Wildlife Posed by Contaminated Soils: A Case Study Involving Dioxin*, in The Risk Assessment of Environmental Hazards, A Textbook of Case Studies, D.J. Paustenbach (editor), John Wiley & Sons, New York, pp. 304-305.

Perry, H.M., et al., 1983, *Cardiovascular Effects of Chronic Barium Ingestion*, in Trace Substances in Environmental Health, XVII, D.D. Hemphill (editor), Proceedings of the University of Missouri's 17th Annual Conference on Trace Substances in Environmental Health, University of Missouri Press, Columbia, Mo. [As cited in EPA (1991c).]

Petering, H.G., and C.J. McClain, 1991, *Silver*, in Metals and Their Components in the Environment: Occurrence, Analysis, and Biological Relevance, E. Merian (editor), VCH Publications, New York, pp. 1191-1202.

Peterson, J.M., and M.M. MacDonell, 1991, *Engineering Evaluation/Cost Analysis for the Management of Contaminated Structures at the Weldon Spring Chemical Plant*, DOE/OR/21548-159, prepared by Argonne National Laboratory, Environmental Assessment and Information Sciences Division, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., May.

Peterson, J.M., et al., 1988, *Work Plan for the Remedial Investigation/Feasibility Study-Environmental Impact Statement for the Weldon Spring Site, Weldon Spring, Missouri*, DOE/OR/21548-033, prepared by Argonne National Laboratory, Energy and Environmental Systems Division, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Aug.

Peto, J., et al., 1984, *Respiratory Cancer Mortality of Welsh Nickel Refinery Workers*, in Nickel in the Human Environment: Proceedings of a Joint Symposium, March, 1983, IARC Scientific Publication No. 53, International Agency for Research on Cancer, Lyon, France, pp. 36-46.

Pflieger, W.L., 1975, *The Fishes of Missouri*, Missouri Department of Conservation, Columbia, Mo.

Piikivi, L., 1989, *Cardiovascular Reflexes and Low Long-Term Exposure to Mercury Vapor*, International Archives of Occupational and Environmental Health, *61*:391-395.

Piikivi, L., and H. Hanninen, 1989, *Subjective Symptoms and Psychological Performance of Chlorine-Alkali Workers*, Scandinavian Journal of Work Environment and Health, *15*:69-74. [As cited in EPA (1991a).]

Piikivi, L., and V. Tolonen, 1989, *EEG Findings in Chlor-Alkali Workers Subjected to Low Long-Term Exposure to Mercury Vapor*, British Journal of Industrial Medicine, *46*:370-375.

Polednak, A.P., A.F. Stehney, and H.F. Lucas, 1983, *Mortality among Male Workers at a Thorium-Processing Plant*, Health Physics, *44*(Suppl. 1):239-251.

Pories, W.J., et al., 1967, *Acceleration of Wound Healing in Man with Zinc Sulfate Given by Mouth*, Lancet, 1:121-124. [As cited in EPA (1991a).]

Poston, T.M., and D.C. Klopfer, 1988, *Concentration Factors Used in the Assessment of Radiation Dose to Consumers of Fish: A Review of 27 Radionuclides*, Health Physics, *55*:751-766.

Prasad, A.S., et al., 1975, *Zinc Deficiency in Sickle Cell Disease*, Clinical Chemistry, *21*:582-587. [As cited in EPA (1991a).]

Que Hee, S.S., et al., 1985, *Evolution of Efficient Methods to Sample Lead Sources, Such as House Dust and Hand Dust, in the Homes of Children*, Environmental Research, *38*:77-95.

Reitinger, J., 1992, *Survey of Selenium Contaminated Soils and Milk Vetch*, inter-office memorandum from J. Reitinger to L. Gonzales (Morrison Knudsen Corporation, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), Oct. 5.

RETA: see Ryckman/Edgerley/Tomlinson & Associates.

Roberts, D.W., 1991, letter from D.W. Roberts (Chief, Missouri Department of Health, Bureau of Environmental Epidemiology, Jefferson City, Mo.) to D. Bedan (Radioactive Waste Site Coordinator, Missouri Department of Natural Resources, Jefferson City, Mo.), Dec. 23, attachment to letter from D.E. Bedan to S. McCracken (Project Manager, Weldon Spring Site Remedial Action Project, U.S. Department of Energy, St. Charles, Mo.), Jan. 8, 1992.

Robison, H.W., and T.M. Buchanan, 1988, *Fishes of Arkansas*, University of Arkansas Press, Fayetteville.

Roels, H., et al., 1980, *Exposure to Lead by the Oral and Pulmonary Routes of Children Living in the Vicinity of a Primary Lead Smelter*, Environmental Research, 22:81-94.

Roels, H., et al., 1987, *Epidemiological Survey among Workers Exposed to Manganese: Effects on Lung, Central Nervous System, and Some Biological Indices*, American Journal of Industrial Medicine, 11:307-327.

Roth, J., 1980, *Shock Sensitivity of Explosives*, in Encyclopedia of Explosives and Related Items, AD-A07595, Large Caliber Weapon Systems Laboratory, U.S. Army Armament Research and Development Command, Dover, N.J., Vol. 9, p. S68.

Ryckman/Edgerley/Tomlinson & Associates, 1977, *Phase II Technical Report: Narrative Section*, prepared by Ryckman/Edgerley/Tomlinson and Associates, St. Louis, Mo., for U.S. Department of the Army, Aberdeen Proving Ground, Md., Dec.

Ryckman/Edgerley/Tomlinson & Associates, 1978, *Phase III Report: Assessment of Disposition Alternatives, Weldon Spring Chemical Plant, St. Charles County, Missouri*, Data Requirement AOOB, prepared by Ryckman/Edgerley/Tomlinson & Associates, St. Louis, Mo., for U.S. Department of the Army, Aberdeen Proving Ground, Md., May.

Ryon, M.G., 1987, *Water Quality Criteria for 2,4,6-Trinitrotoluene*, ORNL-6304, Oak Ridge National Laboratory, Oak Ridge, Tenn., Aug.

Schaum, J., 1983, *Risk Analysis of TCDD Contaminated Soil*, Office of Health and Environmental Assessment, U.S. Environmental Protection Agency, Washington, D.C. [As cited in Paustenbach (1989).]

Schaum, J., 1991, memorandum from J. Schaum (Chief, Exposure Assessment Methods Branch, Office of Research and Development, U.S. Environmental Protection Agency, Washington, D.C.) to C. Sonich-Mullin (Acting Chief, Chemical Mixtures Assessment Branch, Office of Research and Development, U.S. Environmental Protection Agency, Washington, D.C.), Oct. 24.

Schenkler, R.A., 1989, *Review of Radiological Results Reported in "Radiological and Chemical Uptake by Selected Biota at the Weldon Spring Site,"* memorandum from R.A. Schenkler (Biological, Environmental and Medical Research Division, Argonne National Laboratory, Argonne, Ill.) to J. Peterson (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), March 31.

Schmidt, W.R., 1990, letter from W.R. Schmidt (Director, Division of Environmental Health and Epidemiology, Missouri Department of Health, Jefferson City, Mo.) to D.F. Dickneite (Environmental Administrator, Missouri Department of Conservation, Jefferson City, Mo.), July 16.

Schroeder, H.A., and M. Mitchener, 1975a, *Life-Term Effects of Mercury, Methyl Mercury and Nine Other Trace Metals on Mice*, Journal of Nutrition, *105*:452-458. [As cited in EPA (1991c).]

Schroeder, H.A., and M. Mitchener, 1975b, *Life-Term Studies in Rats: Effects of Aluminum, Barium, Beryllium and Tungsten*, Journal of Nutrition, *105*:421-427. [As cited in EPA (1991c).]

Schroeder, H.A., J.J. Balassa, and I.H. Tipton, 1966, *Essential Trace Metals in Man: Manganese, a Study in Homeostasis*, Journal of Chronic Diseases, *19*:545-571.

Schroeder, H.A., M. Mitchener, and A.P. Nason, 1970, *Zirconium, Niobium, Antimony, Vanadium and Lead in Rats; Life Term Studies*, Journal of Nutrition, *100*:59-68. [As cited in EPA (1991c).]

Schumacher, J.G., and K.G. Stollenwerk, 1991, *Geochemical Controls on Migration of Molybdenum, Uranium, and Other Constituents at the Weldon Spring Chemical Plant Site*, in Proceedings of the Geosciences Workshop, DOE/OR/21548-197, published by MK-Ferguson Company and Jacobs Engineering Group, St. Charles, Mo., for U.S. Department of Energy, Oak Ridge Operations, Weldon Spring Site Remedial Action Project, St. Charles, Mo., pp. 87-140.

Seiler, H.G., and H. Sigel (editors), 1988, *Handbook on Toxicity of Inorganic Compounds*, Marcel Dekker, Inc., New York.

Shissler, J., 1990, facsimile transmittal from J. Shissler (Missouri Department of Natural Resources, Air Pollution Control Program, Jefferson City, Mo.) to Y.S. Chang (Argonne National Laboratory, Argonne, Ill.), May 9.

Sims, R.C., and M.R. Overcash, 1983, *Fate of Polynuclear Aromatic Compounds (PNAs) in Soil-Plant Systems*, Research Review, *88*:1-68. [As cited in ATSDR (1990j).]

Sizemore, M., 1991, memorandum from M. Sizemore (MK-Ferguson Company, Weldon Spring Site Remedial Action Project, St. Charles, Mo.) to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), Nov. 18.

Smith, P.W., 1979, *The Fishes of Illinois*, Illinois State Natural History Survey, University of Illinois Press, Urbana, Ill.

Spanggord, R.J., et al., 1980, *Environmental Fate Studies on Certain Munition Wastewater Constituents, Final Report, Phase II — Laboratory Studies,* SRI Project No. LSU-7934, prepared by SRI International, Menlo Park, Calif., for U.S. Army Medical Research and Development Command, Fort Detrick, Frederick, Md., Sept.

Spiers, J.M., 1990, *Influence of Aluminum and Manganese on Rabbiteye Blueberries,* HortScience 25(5): 515-516.

Stehney, A.F., et al., 1980, *Health Status and Body Radioactivity of Former Thorium Workers,* NUREG/CR-1420 (ANL-80-37), prepared by Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy and U.S. Nuclear Regulatory Commission, Jan.

Stokinger, H.E., 1981, *The Metals,* in Patty's Industrial Hygiene and Toxicology, 3rd ed., Vol. I, G.D. Clayton and F.E. Clayton (editors), John Wiley & Sons, New York.

Swanson, S.M., 1983, *Levels of ^{226}Ra, ^{210}Pb, and $^{TOTAL}U$ in Fish near a Saskatchewan Uranium Mine and Mill,* Health Physics, 45:67-80.

Tarasenko, M., O. Promin, and A. Silayev, 1977, *Barium Compounds as Industrial Poisons (an Experimental Study),* Journal of Hygiene, Epidemiology, Microbiology, and Immunology, 21:361. [As cited in EPA (1991a).]

Tardiff, R.G., M. Robinson, and N.S. Ulmer, 1980, *Subchronic Oral Toxicity of $BaCl_2$ in Rats,* Journal of Environmental Pathology and Toxicology, 4:267-275. [As cited in EPA (1991c).]

Thun, M.J., et al., 1985, *Mortality among a Cohort of U.S. Cadmium Production Workers: An Update,* Journal of the National Cancer Institute, 74(2):325-333.

Thyssen, J., J.K.G. Althoff, and U. Mohr, 1981, *Inhalation Studies with Benzo(a)pyrene in Syrian Golden Hamsters,* Journal of the National Cancer Institute, 66:575-577.

Tieger, J., 1988, letter from J. Tieger (Field Supervisor, U.S. Fish and Wildlife Service, Columbia Field Office, Columbia, Mo.) to I. Hlohowskyj (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), Dec. 22.

Trautman, M.B., 1981, *The Fishes of Ohio,* Ohio State University Press, Columbus, Ohio.

Travis, C.C., and A.D. Arms, 1988, *Bioconcentration of Organics in Beef, Milk, and Vegetation,* Environmental Science and Technology, 22:271-274.

Trijonis, J., et al., 1980, *Analysis of the St. Louis RAMS Ambient Particulate Data,* EPA-450/4-80-006a, prepared by Technology Service Corporation for U.S. Environmental Protection Agency, Office of Air, Noise, and Radiation, Office of Air Quality Planning and Standards, Research Triangle Park, Feb. [As cited in Paustenbach (1989).]

Tseng, W.P., 1977, *Effects and Dose Response Relationships of Skin Cancer and Blackfoot Disease with Arsenic,* Environmental Health Perspectives, 19:109-119.

Tseng, W.P., et al., 1968, *Prevalence of Skin Cancer in an Endemic Area of Chronic Arsenicism in Taiwan*, Journal of the National Cancer Institute, 40:453-463.

U.S. Department of Energy, 1987, *Draft Environmental Impact Statement, Remedial Action at the Weldon Spring Site*, DOE/EIS-0117D, Office of Remedial Action and Waste Technology, Feb.

U.S. Department of Energy, 1989, *Remedial Investigations for Quarry Bulk Wastes*, DOE/OR/21548-066, prepared by MK-Ferguson Company and Jacobs Engineering Group, St. Charles, Mo., for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Dec.

U.S. Department of Energy, 1992a, *Feasibility Study for Remedial Action at the Chemical Plant Area of the Weldon Spring Site*, DOE/OR/21548-148, Vols. I-II, prepared by Argonne National Laboratory, Environmental Assessment and Information Sciences Division, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Field Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

U.S. Department of Energy, 1992b, *Proposed Plan for Remedial Action at the Chemical Plant Area of the Weldon Spring Site*, DOE/OR/21548-160, prepared by Argonne National Laboratory, Environmental Assessment and Information Sciences Division, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Field Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

U.S. Department of Energy, 1992c, *Remedial Investigation for the Chemical Plant Area of the Weldon Spring Site*, DOE/OR/21548-074, Rev. 0, Vols. I-II, prepared by MK-Ferguson Company and Jacobs Engineering Group, Weldon Spring, Mo., for U.S. Department of Energy, Oak Ridge Field Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Nov.

U.S. Environmental Protection Agency, 1980, *Ambient Water Quality Criteria for Nitrosamines*, EPA/440/5-80-064, Office of Water Regulations and Standards, Oct.

U.S. Environmental Protection Agency, 1984a, *Health Effects Assessment for Barium*, EPA/540/1-86-021 (PB86-134327), Office of Research and Development, Cincinnati, Ohio, Sept.

U.S. Environmental Protection Agency, 1984b, *Health Effects Assessment for Copper*, EPA/540/1-86/025 (PB86-134368), Office of Research and Development, Cincinnati, Ohio, Sept.

U.S. Environmental Protection Agency, 1984c, *Health Effects Assessment for Manganese (and Compounds)*, EPA/540/1-86/057 (PB86-134681), Office of Research and Development, Cincinnati, Ohio, Sept.

U.S. Environmental Protection Agency, 1984d, *Health Effects Assessment for Mercury*, Office of Health and Environmental Assessment, Cincinnati, Ohio.

U.S. Environmental Protection Agency, 1984e, *Health Effects Assessment for Polycyclic Aromatic Hydrocarbons (PAHs)*, EPA/540/1-86-013 (PB86-134244), Office of Research and Development, Cincinnati, Ohio, Sept.

U.S. Environmental Protection Agency, 1984f, *Health Effects Assessment for Selenium (and Compounds)*, EPA/540/1-86/058 (PB86-134699), Office of Research and Development, Cincinnati, Ohio, Sept.

U.S. Environmental Protection Agency, 1985a, *Final Draft of the Drinking Water Criteria Document on Nitrates/Nitrites*, Office of Drinking Water, Washington, D.C. [As cited in EPA (1991c).]

U.S. Environmental Protection Agency, 1985b, *Updated Mutagenicity and Carcinogenicity Assessment of Cadmium, Addendum to the Health Assessment Document for Cadmium*, EPA 600/8-83-025F (PB85-243533), Office of Health and Environmental Assessment, Washington, D.C., June.

U.S. Environmental Protection Agency, 1986a, *Airborne Asbestos Health Assessment Update*, EPA/600/8-84/003F (PB86-242864), Office of Health and Environmental Assessment, Washington, D.C., June.

U.S. Environmental Protection Agency, 1986b, *Quality Criteria for Water 1986*, EPA 440/5-86-001, Office of Water Regulations and Standards, Washington, D.C., May.

U.S. Environmental Protection Agency, 1986c, *Subchronic (90-day) Toxicity of Thallium Sulfate in Sprague-Dawley Rats*, Office of Solid Waste, Washington, D.C. [As cited in EPA (1991c).]

U.S. Environmental Protection Agency, 1987a, *Data Quality Objectives for Remedial Response Activities, Developmental Process*, EPA/540/G-87/003, Office of Emergency and Remedial Response, Washington, D.C., March.

U.S. Environmental Protection Agency, 1987b, *Health Effects Assessment for Nitrobenzene*, PB88-178975, Environmental Criteria and Assessment Office, Cincinnati, Ohio, May.

U.S. Environmental Protection Agency, 1987c, *Health Effects Assessment for Vanadium and Compounds*, ECAO-CIN-H108, Environmental Criteria and Assessment Office, Cincinnati, Ohio. [As cited in EPA (1991a).]

U.S. Environmental Protection Agency, 1987d, *National Priorities List for Uncontrolled Hazardous Waste Sites, Final Rule (40 CFR Part 300)*, Federal Register, 52(140):27620-27642, July 22.

U.S. Environmental Protection Agency, 1988a, *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA*, Interim Final, EPA/540/G-89/004 (OSWER Directive 9355.3-01), Office of Emergency and Remedial Response, Washington, D.C., Oct.

U.S. Environmental Protection Agency, 1988b, *National Air Quality and Emissions Trends Report, 1986; Total Suspended Particulate Trend Sites, 1977-1986*, EPA-450/4-88-001, Office of Air Quality Planning and Standards, Research Triangle Park, N.C., Feb.

U.S. Environmental Protection Agency, 1988c, *13-Week Mouse Oral Subchronic Toxicity Study*, prepared by Toxicity Research Laboratories, Ltd., Muskegon, Mich., for Office of Solid Waste, Washington, D.C. [As cited in EPA (1991c).]

U.S. Environmental Protection Agency, 1989a, *Ecological Assessment of Hazardous Waste Sites: A Field and Laboratory Reference*, EPA 600/3-89/013, Environmental Research Laboratory, Corvallis, Ore.

U.S. Environmental Protection Agency, 1989b, *Exposure Factors Handbook*, EPA/600/8-89-043, Office of Health and Environmental Assessment, Washington, D.C., July.

U.S. Environmental Protection Agency, 1989c, *Interim Guidance on Establishing Soil Lead Cleanup Levels at Superfund Sites*, OSWER 9355.4-02, Office of Solid Waste and Emergency Response, Washington, D.C., Sept. 7.

U.S. Environmental Protection Agency, 1989d, *Mouse Oral Subchronic Toxicity Study*, prepared by Toxicity Research Laboratories, Ltd., Muskegon, Mich., for Office of Solid Waste, Washington, D.C. [As cited in EPA (1991c).]

U.S. Environmental Protection Agency, 1989e, *Mouse Oral Subchronic Study with Acenaphthene*, prepared by Hazelton Laboratories America, Inc., for Office of Solid Waste, Washington, D.C. [As cited in EPA (1991c).]

U.S. Environmental Protection Agency, 1989f, *Mouse Oral Subchronic Toxicity Study of Pyrene*, prepared by Toxicity Research Laboratories, Ltd., Muskegon, Mich., for Office of Solid Waste, Washington, D.C. [As cited in EPA (1991c).]

U.S. Environmental Protection Agency, 1989g, *National Emissions Standards for Hazardous Air Pollutants; Radionuclides, Final Rule and Notice of Reconsideration (40 CFR Part 61)*, Federal Register, 54(240):51654-51715, Dec. 15.

U.S. Environmental Protection Agency, 1989h, *National Priorities List for Uncontrolled Hazardous Waste Sites, Final Federal Facility Site Update; Final Rule (40 CFR Part 300)*, Federal Register, 54(47):10512-10517, March 13.

U.S. Environmental Protection Agency, 1989i, *Risk Assessment Guidance for Superfund, Volume II: Environmental Evaluation Manual (Part A)*, Interim Final, EPA/540/1-89/001A (OSWER Directive 9285.7-01), Office of Emergency and Remedial Response, Washington, D.C., March.

U.S. Environmental Protection Agency, 1989j, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A)*, Interim Final, EPA/540/1-89/002, Office of Emergency and Remedial Response, Washington, D.C., Dec.

U.S. Environmental Protection Agency, 1989k, *Risk Assessment Methodology, Environmental Impact Statement for NESHAPS Radionuclides, Volume I, Background Information Document*, EPA/520/1-89/005, Office of Radiation Programs, Washington, D.C., Sept. 7.

U.S. Environmental Protection Agency, 1989l, *Subchronic Toxicity in Mice with Anthracene*, Final Report, prepared by Hazelton Laboratories America, Inc., for Office of Solid Waste, Washington, D.C. [As cited in EPA (1991c).]

U.S. Environmental Protection Agency, 1990a, *Health and Environmental Effects Document for Molybdenum and Selected Molybdenum Compounds*, Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, Cincinnati, Ohio. [As cited in EPA (1991a).]

U.S. Environmental Protection Agency, 1990b, *National Air Quality and Emissions Trends Report, 1988; Executive Summary and Chapter 4 — Excerpts*, EPA-450/4-90-002, Office of Air Quality Planning and Standards, Research Triangle Park, N.C., March.

U.S. Environmental Protection Agency, 1990c, *National Oil and Hazardous Substances Pollution Contingency Plan; Final Rule (40 CFR Part 300)*, Federal Register, 55(46):8666-8865, March 8.

U.S. Environmental Protection Agency, 1990d, *National Priorities List for Uncontrolled Hazardous Waste Sites; Final Rule (40 CFR Part 300)*, Federal Register, 55(35):6154-6176, Feb. 21.

U.S. Environmental Protection Agency, 1991a, *Health Effects Assessment Summary Tables*, OERR 9200.6-303(91-1), Office of Emergency and Remedial Response, Annual, FY-1991, Jan.

U.S. Environmental Protection Agency, 1991b, *Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors,"* memorandum from T. Fields, Jr. (Acting Director, Office of Emergency and Remedial Response), and B. Diamond (Office of Waste Programs Enforcement) to Director, Various Divisions, Regions I through IX, OSWER Directive 9285.6-03, Office of Solid Waste and Emergency Response, Washington, D.C., March 25.

U.S. Environmental Protection Agency, 1991c, *Integrated Risk Information System*, Office of Research and Development, database, accessed Dec.

U.S. Environmental Protection Agency, 1991d, *Summary Report on Issues in Ecological Risk Assessment*, EPA/625/3-91/018 (PB91-172122), assembled by Eastern Research Group, Inc., Arlington, Maine, for Risk Assessment Forum, Washington, D.C., Feb.

U.S. Fish and Wildlife Service, 1989, *National Wetlands Inventory Map, Weldon Spring Quadrangle*, National Wetlands Inventory, St. Petersburg, Fla., June.

Venugopal, B., and T.D. Luckey, 1978, *Metal Toxicity in Mammals 2: Chemical Toxicity of Metals and Metalloids*, Plenum Press, New York.

Wagoner, J.K., P.F. Infante, and D.L. Bayliss, 1980, *Beryllium: An Etiologic Agent in the Induction of Lung Cancer, Nonneoplastic Respiratory Disease, and Heart Disease among Industrially Exposed Workers*, Environmental Research, 21:15-34. [As cited in EPA (1991c).]

Waite, D.T., S.R. Joshi, and H. Sommerstad, 1989, *Movement of Dissolved Radionuclides from Submerged Uranium Mine Tailings into Surface Water of Langley Bay, Saskatchewan, Canada*, Archives of Environmental Contamination and Toxicology, 18:881-887.

Waite, D.T., et al., 1990, *A Toxicological Examination of Whitefish (Coregonus clupeaformis) and Northern Pike (Esox lucius) Exposed to Uranium Mine Tailings*, Archives of Environmental Contamination and Toxicology, *19*:578-582.

Walton, G., 1951, *Survey of Literature Relating to Infant Methemoglobinemia Due to Nitrate-Contaminated Water*, American Journal of Public Health, *41*:986-996. [As cited in EPA (1991c).]

Warbritton, K., 1991, personal communication from K. Warbritton (Jacobs Engineering Group, Weldon Spring Site Remedial Action Project, St. Charles, Mo.) to M.M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), Feb.

Welch, K., et al., 1982, *Arsenic Exposure, Smoking, and Respiratory Cancer in Copper Smelter Workers*, Archives of Environmental Health, *37*:325-335.

Whicker, F.W., and V. Schultz, 1982, *Radioecology: Nuclear Energy and the Environment*, CRC Press Inc., Boca Raton, Fla.

Wones, R.G., B.L. Stadler, and L.A. Frohman, 1990, *Lack of Effect of Drinking Water Barium on Cardiovascular Risk Factor*, Environmental Health Perspectives, *85*:1-13.

World Health Organization, 1973, *Trace Elements in Human Nutrition: Manganese*, report of a WHO Expert Committee, Technical Report Service, 532, World Health Organization, Geneva, Switzerland, pp. 34-36. [As cited in EPA (1991c).]

Xue, J., 1992, facsimile transmittal from J. Xue (Harvard School of Public Health, Cambridge, Mass.) to M. MacDonell (Environmental Assessment and Information Sciences Division, Argonne National Laboratory, Argonne, Ill.), March 10.

Ya Keshina, A., et al., 1969, *Benzopyrene Breakdown by the Soil Microflora*, Bulletin of Experimental Biology and Medicine, *68*:1139-1141.

Yang, G., et al., 1989, *Studies of Safe Maximal Daily Dietary Se-Intake in a Seleniferous Area in China, II. Relation between Se-Intake and the Manifestation of Clinical Signs and Certain Biochemical Alterations in Blood and Urine*, Journal of Trace Elements and Electrolytes in Health and Disease, *3*(2):123-130. [As cited in EPA (1991c.)]

10 LIST OF CONTRIBUTORS

This baseline assessment has been prepared by the U.S. Department of Energy (DOE) with contractual assistance from Argonne National Laboratory (ANL). The following ANL staff members contributed to the preparation of this report. Data management support was provided by individuals from MK-Ferguson Company and Jacobs Engineering Group, St. Charles, Missouri, and by H. Özkaynak, D.A. Butler, and J. Xue from the Harvard School of Public Health, Cambridge, Massachusetts. Additional support was provided by D.J. Fingleton, Center for Environmental Research, Carlsbad, New Mexico.

Name	Education/Expertise	Contribution
D.L. Blunt	M.S., Health Physics 3 years experience in environmental assessment	Assessment of radiological data, exposures, toxicity, and risks
C.P. Dunn	Ph.D., Plant Ecology 13 years experience in ecological research and assessment	Assessment of ecological data, exposures, and impacts
L.A. Durham	M.S., Geology 5 years experience in groundwater research and environmental assessment	Assessment of groundwater data
L.A. Haroun	M.P.H., Environmental Health Sciences 15 years experience in environmental research and health effects assessment	Overall document responsibility; assessment of chemical data, exposures, toxicity, and risks
I. Hlohowskyj	Ph.D., Zoology 14 years experience in ecological research and environmental assessment	Assessment of ecological data, exposures, and impacts
M.M. MacDonell	Ph.D., Civil Engineering/Environmental Health Engineering 10 years experience in environmental research and assessment	Assessment of biouptake data and chemical exposures and risks
C.N. Mitchell	Management Information Systems 6 years experience in computer modeling and data analysis	Processing of groundwater data

Name	Education/Expertise	Contribution
A.M. Paine	M.S., Environmental Biology 4 years experience in chemical analysis	Compilation of chemical toxicity data
J.M. Peterson	M.S., Nuclear Engineering; P.E., C.H.P. 16 years experience in nuclear programs, including 12 years in environmental assessment	Assessment of radiological data, exposures, toxicity, and risks
K.L. Woytowich	B.S., Mathematics and Computer Science 3 years experience in computer modeling and data analysis	Processing of data for soil, sludge, sediment, surface water, buildings, and biota
D.J. Wyman	M.S., Botany; M.A., Library Science 17 years experience in technical editing	Overall editorial responsibility

APPENDIX A:

STANDARD ANALYTE LIST AND DETECTION LIMITS

TABLE A.1 Detection Limits for Inorganic Analytes

Analyte	Detection Limit[a] Soil (mg/kg)	Detection Limit[a] Water (µg/L)	Analyte	Detection Limit[a] Soil (mg/kg)	Detection Limit[a] Water (µg/L)
CLP metals			Other metals		
Aluminum	20	200	Lithium	5	50
Antimony	6	60	Molybdenum	4	40
Arsenic	1	10	Zirconium	20	200
Barium	20	200			
Beryllium	0.5	5	Inorganic anions		
Cadmium	0.5	5	Chloride	1.5	250/200
Calcium	500	5,000	Fluoride	1.25	250/600
Chromium	1	10	Nitrate	0.5	250/100
Cobalt	5	50	Nitrite	0.5	250
Copper	2.5	25	Sulfate	5	1,000/1,000
Iron	10	100			
Lead	0.5	5			
Magnesium	500	5,000			
Manganese	1.5	15			
Mercury	0.1	0.2			
Nickel	4	40			
Potassium	500	5,000			
Selenium	0.5	5			
Silver	1	10			
Sodium	500	5,000			
Thallium	1	10			
Vanadium	5	50			
Zinc	2	20			

[a] In general, the listed detection limits are those typically used for analyses conducted in accordance with methodology developed by the U.S. Environmental Protection Agency (EPA) for the Contract Laboratory Program (CLP). Inorganic anions, radionuclides, and certain nitroaromatic compounds are not included in the analyte list for this program, but other standard methodology and related detection limits have been identified by the EPA for these analytes; those limits are therefore included in this table. For cases in which two detection limits are shown for a single contaminant, that contaminant has been analyzed by two different methods at various times of the characterization effort. For example, the detection limits of the standard U.S. Army Toxic and Hazardous Materials Agency (USATHAMA) method for nitroaromatic compounds were reduced by modifying the method to include a sample concentration procedure, and the lower detection limits were used for certain analyses.

TABLE A.2 Detection Limits for CLP Organic Compounds and Nitroaromatic Compounds

Analyte	Detection Limit[a] Soil (µg/kg)	Detection Limit[a] Water (µg/L)
Volatile organic compounds		
Acetone	10	10
Benzene	5	5
Bromodichloromethane	5	5
Bromoform	5	5
Bromomethane	10	10
2-Butanone	10	10
Carbon disulfide	5	5
Carbon tetrachloride	5	5
Chlorobenzene	5	5
Chloroethane	10	10
Chloroform	5	5
Chloromethane	10	10
Dibromochloromethane	5	5
1,1-Dichloroethane	5	5
1,2-Dichloroethane	5	5
1,1-Dichloroethene	5	5
1,2-Dichloroethene, total	5	5
1,2-Dichloropropane	5	5
cis-1,3-Dichloropropene	5	5
trans-1,3-Dichloropropene	5	5
Ethylbenzene	5	5
2-Hexanone	10	10
Methylene chloride	5	5
4-Methyl-2-pentanone	10	10
Styrene	5	5
Tetrachloroethene	5	5
1,1,2,2-Tetrachloroethane	5	5
Toluene	5	5
1,1,1-Trichloroethane	5	5
1,1,2-Trichloroethane	5	5
Trichloroethene	5	5
Vinyl acetate	10	10
Vinyl chloride	10	10
Xylenes, total	5	5
Semivolatile organic compounds		
Acenaphthene	330	10
Acenaphthylene	330	10
Anthracene	330	10
Benz(a)anthracene	330	10
Benzo(b)fluoranthene	330	10
Benzo(k)fluoranthene	330	10

TABLE A.2 (Cont.)

Analyte	Detection Limit[a] Soil (µg/kg)	Water (µg/L)
Semivolatile organic compounds (cont.)		
Benzo(g,h,i)perylene	330	10
Benzo(a)pyrene	330	10
Benzoic acid	1,600	50
Benzyl alcohol	330	10
Bis(2-chloroethoxy)methane	330	10
Bis(2-chloroethyl)ether	330	10
Bis(2-chloroisopropyl)ether	330	10
Bis(2-ethylhexyl)phthalate	330	10
4-Bromophenyl-phenylether	330	10
Butylbenzylphthalate	330	10
4-Chloroaniline	330	10
4-Chloro-3-methylphenol	330	10
2-Chloronaphthalene	330	10
2-Chlorophenol	330	10
4-Chlorophenyl-phenylether	330	10
Chrysene	330	10
Dibenz(a,h)anthracene	330	10
Dibenzofuran	330	10
Di-n-butylphthalate	330	10
1,2-Dichlorobenzene	330	10
1,3-Dichlorobenzene	330	10
1,4-Dichlorobenzene	330	10
3,3′-Dichlorobenzidine	660	20
2,4-Dichlorophenol	330	10
Diethylphthalate	330	10
2,4-Dimethylphenol	330	10
Dimethylphthalate	330	10
4,6-Dinitro-2-methylphenol	1,600	50
2,4-Dinitrophenol	1,600	50
2,4-Dinitrotoluene	330	10
2,6-Dinitrotoluene	330	10
Di-n-octylphthalate	330	10
Fluoranthene	330	10
Fluorene	330	10
Hexachlorobenzene	330	10
Hexachlorobutadiene	330	10
Hexachlorocyclopentadiene	330	10
Hexachloroethane	330	10
Indeno(1,2,3-cd)pyrene	330	10
Isophorone	330	10
2-Methylnaphthalene	330	10
2-Methylphenol	330	10
4-Methylphenol	330	10

TABLE A.2 (Cont.)

Analyte	Detection Limit[a] Soil (µg/kg)	Water (µg/L)
Semivolatile organic compounds (cont.)		
Naphthalene	330	10
2-Nitroaniline	1,600	50
3-Nitroaniline	1,600	50
4-Nitroaniline	1,600	50
Nitrobenzene	330	10
2-Nitrophenol	330	10
4-Nitrophenol	1,600	50
N-Nitroso-di-n-dipropylamine	330	10
N-Nitrosodiphenylamine	330	10
Pentachlorophenol	1,600	50
Phenanthrene	330	10
Phenol	330	10
Pyrene	330	10
1,2,4-Trichlorobenzene	330	10
2,4,5-Trichlorophenol	1,600	50
2,4,6-Trichlorophenol	330	10
PCBs/Pesticides		
Aldrin	8.0	0.05
alpha-BHC	8.0	0.05
alpha-Chlordane	80.0	0.5
Aroclor 1016	80.0	0.5
Aroclor 1221	80.0	0.5
Aroclor 1232	80.0	0.5
Aroclor 1242	80.0	0.5
Aroclor 1248	80.0	0.5
Aroclor 1254	160.0	1.0
Aroclor 1260	160.0	1.0
beta-BHC	8.0	0.05
4,4'-DDD	16.0	0.10
4,4'-DDE	16.0	0.10
4,4'-DDT	16.0	0.10
delta-BHC	8.0	0.05
Dieldrin	16.0	0.10
Endosulfan I	8.0	0.05
Endosulfan II	16.0	0.10
Endosulfan sulfate	16.0	0.10
Endrin	16.0	0.10
Endrin ketone	16.0	0.10
gamma-BHC (Lindane)	8.0	0.05
gamma-Chlordane	80.0	0.5
Heptachlor	8.0	0.05

TABLE A.2 (Cont.)

Analyte	Detection Limit[a] Soil (µg/kg)	Water (µg/L)
PCBs/Pesticides (cont.)		
Heptachlor epoxide	8.0	0.05
Methoxychlor	80.0	0.5
Toxaphene	160.0	1.0
Nitroaromatic compounds		
1,3-Dinitrobenzene	900	0.51/0.09
2,4-Dinitrotoluene	750	0.44/0.03
2,6-Dinitrotoluene	1,410	0.48/0.01
Nitrobenzene	1,440	0.54/0.03
1,3,5-Trinitrobenzene	570	0.47/0.03
2,4,6-Trinitrotoluene	1,200	0.98/0.03

[a] In general, the listed detection limits are those typically used for analyses conducted in accordance with methodology developed by the EPA for the Contract Laboratory Program (CLP). Inorganic anions, radionuclides, and certain nitroaromatic compounds are not included in the analyte list for this program, but other standard methodology and related detection limits have been identified by the EPA for these analytes. Those limits are therefore included in this table. For cases in which two detection limits are shown for a single contaminant, that contaminant has been analyzed by two different methods at various times of the characterization effort. For example, the detection limits of the standard USATHAMA method for nitroaromatic compounds were reduced by modifying the method to include a sample concentration procedure, and the lower detection limits were used for certain analyses.

APPENDIX B:

GROUNDWATER QUALITY

APPENDIX B:

GROUNDWATER QUALITY

The groundwater monitoring system for the Weldon Spring site measures water quality at the chemical plant area and vicinity. The program includes measuring static water levels and collecting and analyzing groundwater samples from 59 wells completed in the upper Burlington-Keokuk Limestone. The monitoring well network includes the 4000-series wells that monitor locations outside the boundary of the site, the 3000-series wells that monitor the area surrounding the raffinate pits, and the 2000-series wells that monitor other on-site areas such as Ash Pond and Frog Pond. The locations of the wells, raffinate pits, and these ponds are shown in Figure B.1*.

The uppermost bedrock unit at the site is the Burlington-Keokuk Limestone. From borehole stratigraphic data, the limestone formation has been divided into units, primarily on the basis of weathering characteristics. The upper unit is referred to as the weathered limestone, and the lower unit is referred to as the competent (i.e., unweathered) limestone. The contact between these two units is gradational, and this gradational area is referred to as the transition zone. To obtain lateral and vertical hydrogeologic data and to assess the quality of water in the Burlington-Keokuk Limestone, wells have been completed at various depths: 13 wells in the weathered limestone, 20 in the competent bedrock, and 26 across the weathered/competent contact. Construction data for these wells are given in Table B.1*.

The shallow bedrock aquifer at the site and vicinity is unconfined. The water table fluctuates in elevation but generally remains within the upper bedrock or the highly weathered residuum at the overburden/bedrock interface. Figure B.2 shows the water table surface contoured from water-level elevations measured in the shallow bedrock wells completed in the weathered and the weathered/competent (transitional) zones (generally less than 12 m [40 ft] into the bedrock). The elevation of groundwater in the uppermost bedrock at the site and vicinity ranges from 171 to 186 m (560 to 610 ft) above mean sea level (MSL). As the water table surface indicates, a groundwater divide exists in the upper portion of the shallow bedrock aquifer that trends northeast to southwest through the southern portion of the site. Groundwater north of the divide flows north toward Dardenne Creek, whereas groundwater south of the divide flows south toward the Missouri River. Figure B.3 is a map of the contoured water-level elevations measured in deep wells (generally 18 to 24 m [60 to 80 ft] in the bedrock) completed in the competent zone of the Burlington-Keokuk Limestone. The configuration of the water table surface for the deeper part of the bedrock aquifer is similar to that of the shallow portion of the aquifer, with the groundwater divide shifted about 244 m (800 ft) to the north.

Environmental monitoring activities at the site are conducted in accordance with U.S. Department of Energy (DOE) monitoring requirements and in support of the characterization process. Groundwater monitoring during 1987 through 1989 generally consisted of

*For readability, the figures and tables are presented in sequence following the text in this chapter.

quarterly sampling of the entire monitoring well network. In 1990, the monitoring frequency was reduced to semiannual sampling of all wells (MK-Ferguson Company and Jacobs Engineering Group 1990). Groundwater samples have been analyzed for nitroaromatic compounds, inorganic anions, radionuclides, and assorted metals. Sampling was generally performed quarterly, but samples for certain metals were analyzed only once in 1987 and twice during 1988. The results of the groundwater quality characterization efforts are discussed in more detail in the remedial investigation (DOE 1992) and in the Phase I and Phase II groundwater quality assessment reports (MK-Ferguson Company 1987; MK-Ferguson Company and Jacobs Engineering Group 1989).

The discussion of groundwater quality in Sections B.1 through B.4 presents average concentrations for various metals, radionuclides, inorganic anions, and nitroaromatic compounds that were determined for each well from data collected from 1987 through 1990. The horizontal and vertical extent of groundwater contamination has been tentatively defined only for the major contaminants of concern, on the basis of the limited data available. The issue of horizontal extent of contamination has been addressed by spatially listing average concentrations measured from each monitoring well. The vertical extent of contamination has been addressed by separating the groundwater chemical data into two groups on the basis of well completion depths, i.e., either above or within the competent (unweathered) portion of the Burlington-Keokuk aquifer.

An initial review of the database indicated that some concentrations for the monitoring wells appear to be unusually high, possibly as a result of data verification/validation issues. Pending resolution of these issues, the values in question have not been used in determining the concentration range or average. In addition, certain contaminants (such as arsenic and cadmium) were detected in only a small number of groundwater samples (Chapter 2, Table 2.14). Pending the collection and analysis of additional groundwater data, these contaminants have not been included in this preliminary presentation of data for the chemical plant area.

To evaluate the spatial distribution of contaminants on the basis of well completion depths (e.g., in the weathered, transitional, and competent zones), the average concentrations for each chemical were divided into two data sets. The first data set includes average groundwater concentrations obtained from wells screened in the weathered limestone and wells screened across the transitional zone, i.e., the weathered/competent contact (Figure B.4); the second data set consists of average concentrations obtained from wells completed only in the competent portion of the limestone (Figure B.5). Groundwater samples collected from wells screened across the transitional zone were included in the first set because they are more representative of the weathered limestone than the competent zone. This assumption is supported by a correlation between chemical concentrations in the groundwater sampled in the weathered and transitional zones and the correlation of water-level elevations measured between the weathered and transitional zones.

The average concentrations of the constituents identified in groundwater samples were compared with criteria established by the U.S. Environmental Protection Agency (EPA 1991) for public drinking water supplies (to be met at the tap) and with local background values. The

EPA criteria primarily consist of maximum contaminant levels (MCLs), secondary maximum contaminant levels (SMCLs), and maximum contaminant level goals (MCLGs). Criteria established by the state of Missouri are generally the same as those promulgated by the EPA (Missouri Department of Natural Resources 1991); minor exceptions for the inorganic anions are identified in Section B.3. Background concentrations were determined by reviewing information from studies of off-site monitoring wells conducted by the U.S. Geological Survey, including wells located in the Busch Wildlife Area and private wells not influenced by the site. These local background concentrations are presented in Chapter 2 (Table 2.14) of this baseline assessment. Some uncertainty is associated with the background concentrations because they were determined from a limited number of wells. Those chemicals with concentrations consistently greater than the comparison criteria are chromium, nickel, uranium, nitrate, sulfate, and nitroaromatic compounds.

The spatial distribution of several chemicals, including those for which the average concentrations generally exceed the comparison criteria, are illustrated in Figures B.6 through B.41 and are discussed in this appendix. For each chemical, the accompanying figure includes the well location and the average concentration. If no concentration value is listed next to a given well location, the average chemical concentration for groundwater sampled from that particular well was below the detection limit. The number of samples collected, the number of samples above the detection limit, the minimum and maximum detected values, and the average values are listed in Tables B.2, B.3, and B.4 for metals, inorganic anions, and nitroaromatic compounds, respectively. To determine the average concentration, the nondetects were assigned a value equal to 50% of the detection limit. This approach was used to address the possibility that the nondetects might in fact be present, but at levels below the detection limit (for additional discussion of this strategy, see Section 3.3). Hence, in some cases, the estimated average value is less than the minimum concentration detected.

The detection limits used for this analysis were the contract required detection limits for Contract Laboratory Program (CLP) metals (Section 2.3.2.2 and Appendix A) and the stated method detection limit for non-CLP metals and inorganic anions. For nitroaromatic compounds, different analytical methods with a range of detection limits were applied at various times during the characterization effort; for the calculations in this appendix, the values used for the nondetects were those reported in the Phase II groundwater quality assessment (MK-Ferguson Company and Jacobs Engineering Group 1989).

B.1 METALS

Several metals have been detected in groundwater at the Weldon Spring site and vicinity. The results of analyses for these metals are presented in Table B.2 and discussed in Sections B.1.1 through B.1.10. For each metal, the site groundwater concentration is compared with available EPA standards (e.g., MCLs and MCLGs) under the Safe Drinking Water Act (EPA 1991). If an EPA standard is not discussed for a particular chemical, the standard has not been promulgated. It was often difficult to compare metal concentrations in groundwater with the standards because of the detection limits used in the analysis. The detection limit for several of the metals was higher than both the EPA standard and the local background concentration.

For each metal analyzed, the highest concentration in the groundwater was generally collected from the weathered zone rather than the competent zone of the Burlington-Keokuk Limestone aquifer. However, a quantitative comparison of groundwater quality between the two zones of the aquifer is difficult to perform because of the difference in the number of wells completed and sampled in the weathered zone compared with the competent zone of the aquifer. Also, the spatial locations differ, with only 11 nested well pairs at the site. However, the following discussion provides a useful qualitative description of the concentration distributions for several metals that were detected in groundwater at the Weldon Spring site and vicinity.

B.1.1 Antimony

The distribution of average antimony concentrations in the weathered portion of the limestone aquifer is shown in Figure B.6. The EPA is considering two alternative MCLs, 10 and 5 µg/L, on the basis of a practical quantitation level (PQL) of either 5 or 10 times the method detection limit for antimony. The MCLG for antimony is 3 µg/L. Except for one off-site well, the average antimony concentrations were at or above the detection limit of 60 µg/L, which is higher than both the MCLs being considered and the MCLG; these wells are located north of and adjacent to the raffinate pits, indicating that the pits might be the source of antimony contamination. The antimony concentrations determined from groundwater samples collected in the competent limestone were all less than the detection limit.

B.1.2 Barium

Average concentrations of barium in groundwater were at or above the detection limit of 200 µg/L in several wells in the weathered limestone. As shown in Figure B.7, barium is distributed north and south of the raffinate pits and in the northeastern area of the site near Frog Pond. The MCL for barium is 1,000 µg/L. Average concentrations from two wells adjacent to the raffinate pits exceed this level, indicating that the raffinate pits are a likely source of barium contamination. Two wells sampled within the competent limestone have average concentrations above the detection limit but below the MCL (Figure B.8).

B.1.3 Chromium

Average groundwater concentrations of chromium in the weathered limestone are above the detection limit of 10 µg/L in most wells within the site boundary (Figure B.9). The concentration measured in one well adjacent to the raffinate pits was higher than the MCL for chromium of 50 µg/L. Chromium has been detected in scattered locations in the competent limestone in and around the site; however, for the few wells in which chromium was detected, the average concentrations were below the MCL (Figure B.10). The sources of chromium contamination in soil and associated groundwater are unknown. Chromium is typically used in electroplating activities, which have not been documented at the site; however, the activities associated with two buildings at the site, referred to as the metals plant and metals pilot plant,

have not been determined. Chromium is present in the raffinate pit sludge and has been detected in total analyses of the pit water (DOE 1992).

B.1.4 Lead

Lead has been detected at average concentrations above the detection limit of 5 µg/L in groundwater samples from both the weathered and competent zones of the limestone aquifer; however, these average concentrations are below the MCL of 50 µg/L (Figures B.11 and B.12) and were detected in only a few wells. Four of the five groundwater samples in which lead was detected in the weathered zone are outside the site perimeter. Unlike most of the other metals evaluated, the highest average lead concentration is in the competent limestone, and more wells completed in the competent zone than in the weathered zone have average lead concentrations above background. Many of the locations where lead has been detected in the soil and associated groundwater might be related to drainage areas for ordnance works wastewater. Process buildings in the ordnance works were lead sheathed to help prevent sparking. Acid dissolution of this material, followed by leaching to groundwater, might also account for some of the samples containing lead concentrations above background (DOE 1992).

B.1.5 Lithium

The average concentrations of lithium that exceed the detection limit of 50 µg/L in samples from wells completed in the weathered limestone are shown in Figure B.13. Elevated average lithium concentrations are present in wells located near the raffinate pits, indicating that the pits may be a source of lithium contamination. The detection frequency for lithium in groundwater samples collected from wells completed in the competent limestone was low; therefore, the average concentration was below the detection limit.

B.1.6 Manganese

The distribution of average manganese concentrations at or above the detection limit of 15 µg/L in the weathered and competent zones is shown in Figures B.14 and B.15, respectively. The manganese concentrations show extreme spatial variability in both zones of the limestone aquifer, with most of the average groundwater concentrations above the detection limit. Many of the average concentrations also exceed EPA's SMCL of 50 µg/L. When average manganese concentrations in groundwater samples from wells in the weathered zone are compared with those in the competent zone, no pronounced distribution pattern is apparent. The source of elevated manganese concentrations in the soil and associated groundwater is unknown but might be partially attributed to pyrolusite in the overburden and along fractures in the Burlington-Keokuk Limestone (DOE 1992).

B.1.7 Mercury

Average mercury concentrations above the detection limit of 0.20 μg/L in groundwater samples from the weathered limestone are shown in Figure B.16. Two of the five wells with average concentrations above the detection limit also exceeded the MCL of 2 μg/L; these two wells are located near Frog Pond and Ash Pond, indicating that the ponds might be a source of mercury contamination. The average mercury concentrations in groundwater samples from the competent zone of the limestone aquifer were either below the detection limit or below the MCL (Figure B.17). The other sources of mercury contamination in soil and associated groundwater might be attributed to impurities in the uranium ore (DOE 1992).

B.1.8 Molybdenum

Average groundwater concentrations of molybdenum at or above the detection limit of 40 μg/L are shown in Figure B.18 for the weathered limestone and Figure B.19 for the competent limestone. In the weathered zone, the average molybdenum concentrations at or above the detection limit are from wells adjacent to the raffinate pits. In the competent zone, except for one well located outside the southern boundary, the higher average molybdenum concentrations are from wells adjacent to and north of the raffinate pits. Consequently, the likely source of molybdenum contamination is the raffinate pits.

B.1.9 Nickel

Average nickel concentrations above the detection limit of 40 μg/L in groundwater samples from the weathered limestone are shown in Figure B.20. The EPA is considering an MCL of 100 μg/L for nickel, which was exceeded in two of these wells. Groundwater samples with average nickel concentrations above the detection limit are from wells adjacent to the raffinate pits and at three other scattered locations in and around the site perimeter. Average nickel concentrations in the competent limestone were all below the detection limit. The sources of nickel contamination in soil and associated groundwater are unknown. Nickel, like chromium, is typically used in electroplating activities, which have not been documented at the site; however, the activities associated with two buildings at the site, referred to as the metals plant and metals pilot plant, have not been determined. Nickel is present in the raffinate pit sludge and has been detected in total analyses of the pit water (DOE 1992).

B.1.10 Thallium

Average thallium concentrations in the weathered and competent zones of the limestone aquifer are shown in Figures B.21 and B.22, respectively. The EPA is considering alternative MCLs for thallium of 2 and 1 μg/L, on the basis of a PQL of 5 or 10 times the method detection limit. The MCLG is 0.5 μg/L. For those wells in which the average concentrations exceeded the detection limit of 10 μg/L, the concentrations were considerably higher than both the MCLs being considered and the MCLG. Unlike most of the other metals, more wells in the competent

zone had average thallium concentrations above the detection limit than did wells in the weathered zone. The source of this contamination is unknown.

B.2 RADIONUCLIDES

Groundwater samples collected from monitoring wells at the site were analyzed for total uranium, thorium-232, thorium-230, and radium-226. Results of the uranium analyses are presented in Table B.2 and discussed in the following paragraph. The other radionuclides are not included in this discussion because groundwater analyses indicated that either they were not detected or were detected at very low concentrations (DOE 1992).

The distributions of average uranium concentrations in groundwater samples from the weathered and competent zones are shown in Figures B.23 and B.24, respectively. Many of the wells completed in the weathered limestone had average groundwater concentrations above the detection limit, and several wells located near the raffinate pits and along the southern and eastern boundaries of the site had elevated concentrations relative to the other wells (Figure B.23). Average uranium concentrations in groundwater samples from the competent zone are all above the detection limit, with the highest concentrations detected in two wells located south and southeast of the site boundary. The primary sources of uranium that appear to contribute to groundwater contamination include the raffinate pits and contaminated soil in the Ash Pond area and in the southern or eastern area of the site.

B.3 INORGANIC ANIONS

Groundwater samples have been analyzed for chloride, fluoride, nitrate, and sulfate; the results are presented in Table B.3. Chloride was omitted from the following discussion because no groundwater samples exceeded EPA's SMCL of 250 mg/L.

B.3.1 Fluoride

The average fluoride groundwater concentrations were above the detection limit of 0.25 mg/L in most of the wells completed in the weathered limestone within the site boundary (Figure B.25). Both the EPA and the state of Missouri have an SMCL for fluoride of 2 mg/L; the state has also established an MCL of 4 mg/L. Three wells in the area of the raffinate pits had average groundwater concentrations above the SMCL but below the state MCL (Figure B.25). Also, one well in the area of Frog Pond had an average groundwater concentration approaching the SMCL. In the competent zone, all wells except one located north of the raffinate pits had average fluoride concentrations below the detection limit (Figure B.26).

B.3.2 Nitrate

Average nitrate concentrations in groundwater samples from the weathered limestone are shown in Figure B.27. The primary drinking water standard for nitrate is 10 mg/L as

nitrogen, which converts to 44 mg/L as nitrate. Groundwater samples from monitoring wells near the raffinate pits and in the area of Ash Pond had average nitrate concentrations above both the detection limit of 0.1 mg/L and the drinking water standard, with values that exceed 1,000 mg/L. The major contributors to the nitrate contamination detected in groundwater samples appear to be the raffinate pits and Ash Pond. The distribution of average nitrate concentrations in groundwater samples from the competent limestone is shown in Figure B.28. Except for one well north of the raffinate pits, nitrate was below the drinking water standard in all wells in the competent zone.

B.3.3 Sulfate

Average sulfate concentrations above the detection limit of 1.0 mg/L in the weathered and competent limestone groundwater are shown in Figures B.29 and B.30, respectively. Elevated sulfate concentrations were detected in groundwater from several weathered limestone wells near the southern and eastern site boundaries and adjacent to the raffinate pits. Several of these wells near the site boundaries had average sulfate concentrations that exceed the state of Missouri's SMCL of 150 mg/L and the EPA's SMCL of 250 mg/L. The probable source of sulfate in the southern and eastern areas is the release of sulfuric acid and sodium sulfate previously used for production of trinitrotoluene (TNT). Sulfate in groundwater near the raffinate pits is the result of both seepage from the raffinate pits and leaching from sulfate-contaminated soil. However, a few wells scattered across the site also had increased sulfate concentrations, suggesting intermittent sources such as contaminated soil leached by infiltration through the vadose zone rather than continuous sources such as seepage from the raffinate pits (DOE 1992). All of the wells in the competent limestone had average sulfate concentrations above the detection limit. The competent wells with the highest average sulfate concentrations are scattered across the site, again suggesting multiple sources.

B.4 NITROAROMATIC COMPOUNDS

Concentrations of average nitroaromatic compounds in the weathered and competent limestone aquifers are presented in Table B.4; concentrations exceeding the detection limit are shown in Figures B.31 through B.41. The background concentration for nitroaromatic compounds at the chemical plant and vicinity is considered to be zero because these chemicals do not occur naturally. In general, the average groundwater concentrations of nitroaromatic compounds in the weathered limestone were above the detection limits in three areas of the site: near Frog Pond in the northeastern portion of the site, near Ash Pond in the northern section of the site, and near the raffinate pits. Nitroaromatic compounds with average concentrations above the detection limit were reported at a few monitoring well locations in the competent zone, primarily near the raffinate pits and in the northern region near Ash Pond. The nitroaromatic compound 1,3-dinitrobenzene was not detected in groundwater samples from the competent zone.

A distribution pattern of nitroaromatic compounds is somewhat evident in comparisons of detectable concentrations in groundwater samples from the weathered and competent zones

of the aquifer. However, definitive conclusions pertaining to the groundwater quality of the two zones are difficult to make because many more wells are completed in the weathered zone than in the competent zone. In general, more nitroaromatic compounds have been detected and at higher average concentrations in groundwater samples from the weathered zone than in groundwater samples from the competent zone of the Burlington-Keokuk Limestone. The apparent source areas of nitroaromatic compounds in the northeastern portion of the site may be associated with Frog Pond and former TNT production lines. The likely source of nitroaromatic compounds near the raffinate pits is the process line west of raffinate pit 4.

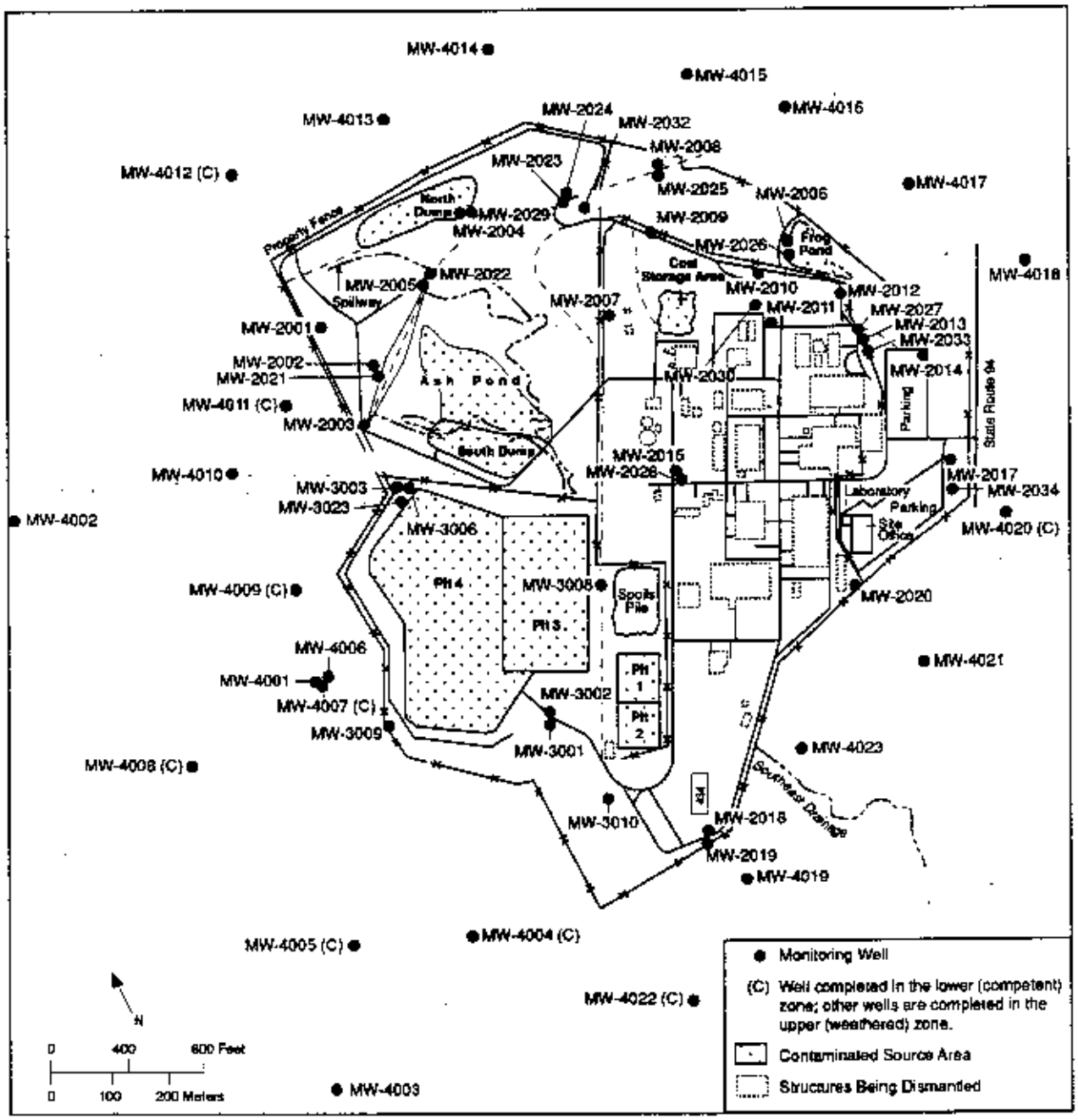


FIGURE B.1 Location and Completion Data for Site Monitoring Wells

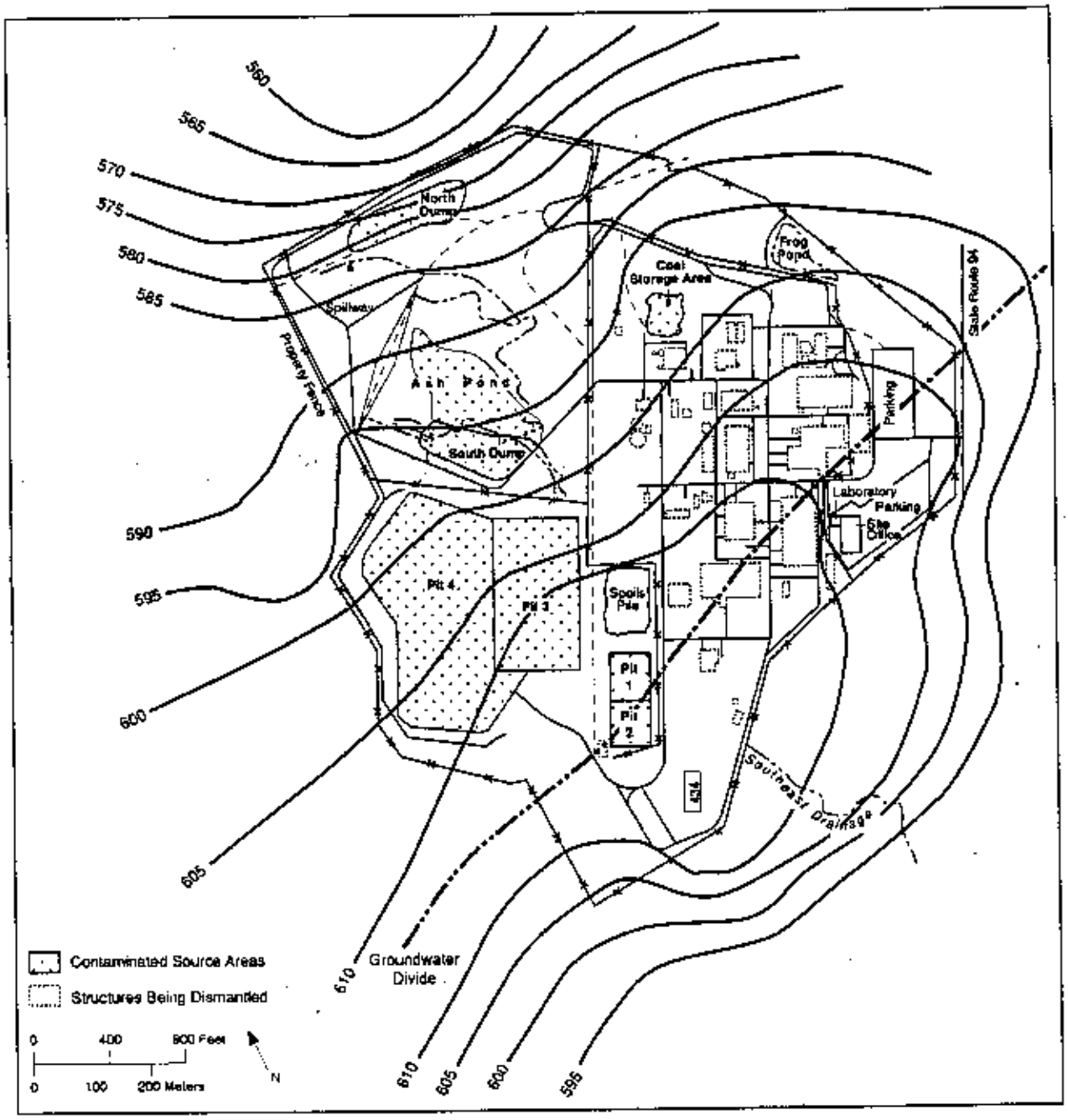


FIGURE B.2 Estimated Surface of the Shallow Groundwater Table (Source: Modified from MK-Ferguson Company and Jacobs Engineering Group 1989)

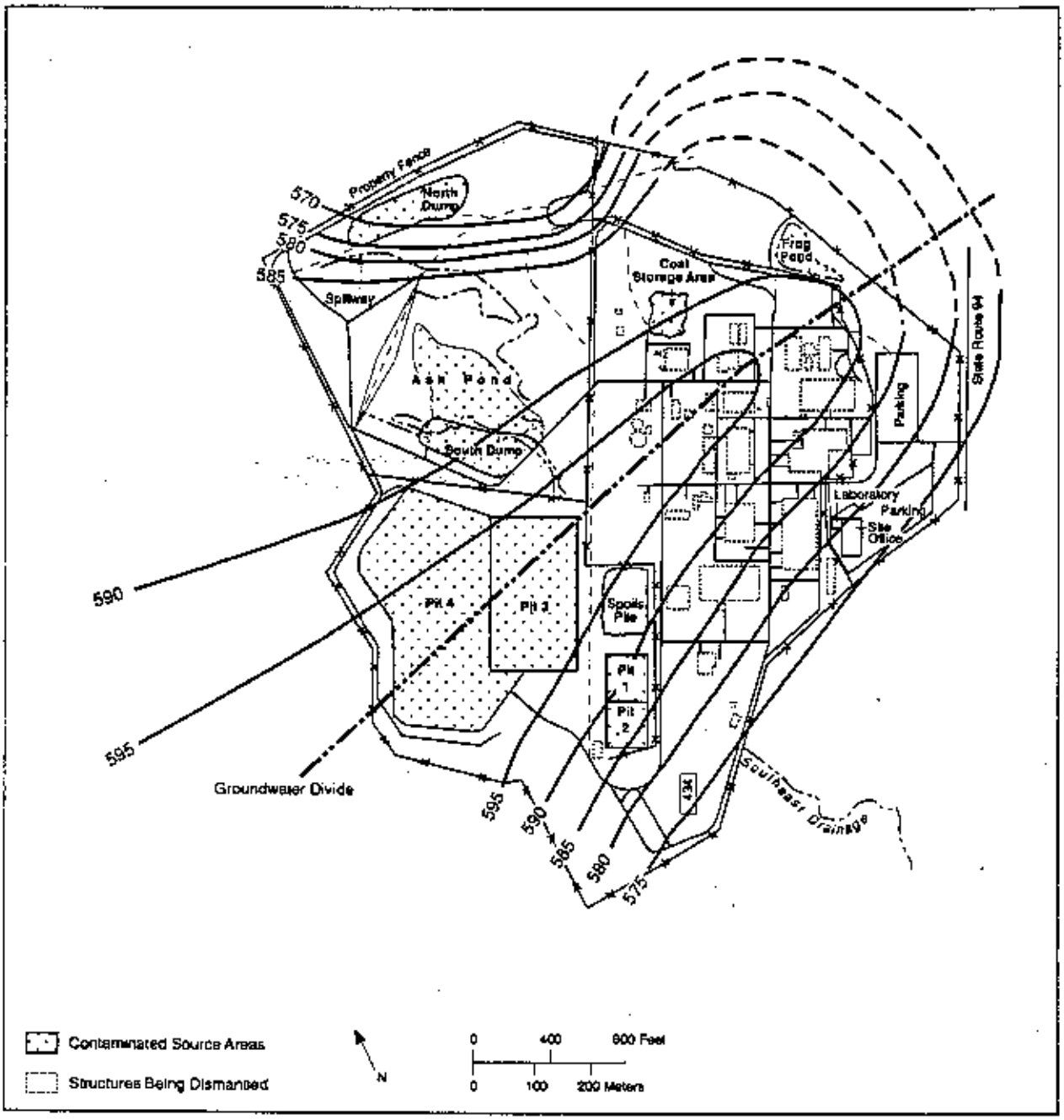


FIGURE B.3 Estimated Surface of the Deep Groundwater Table (Source: Modified from MK-Ferguson Company and Jacobs Engineering Group 1989)

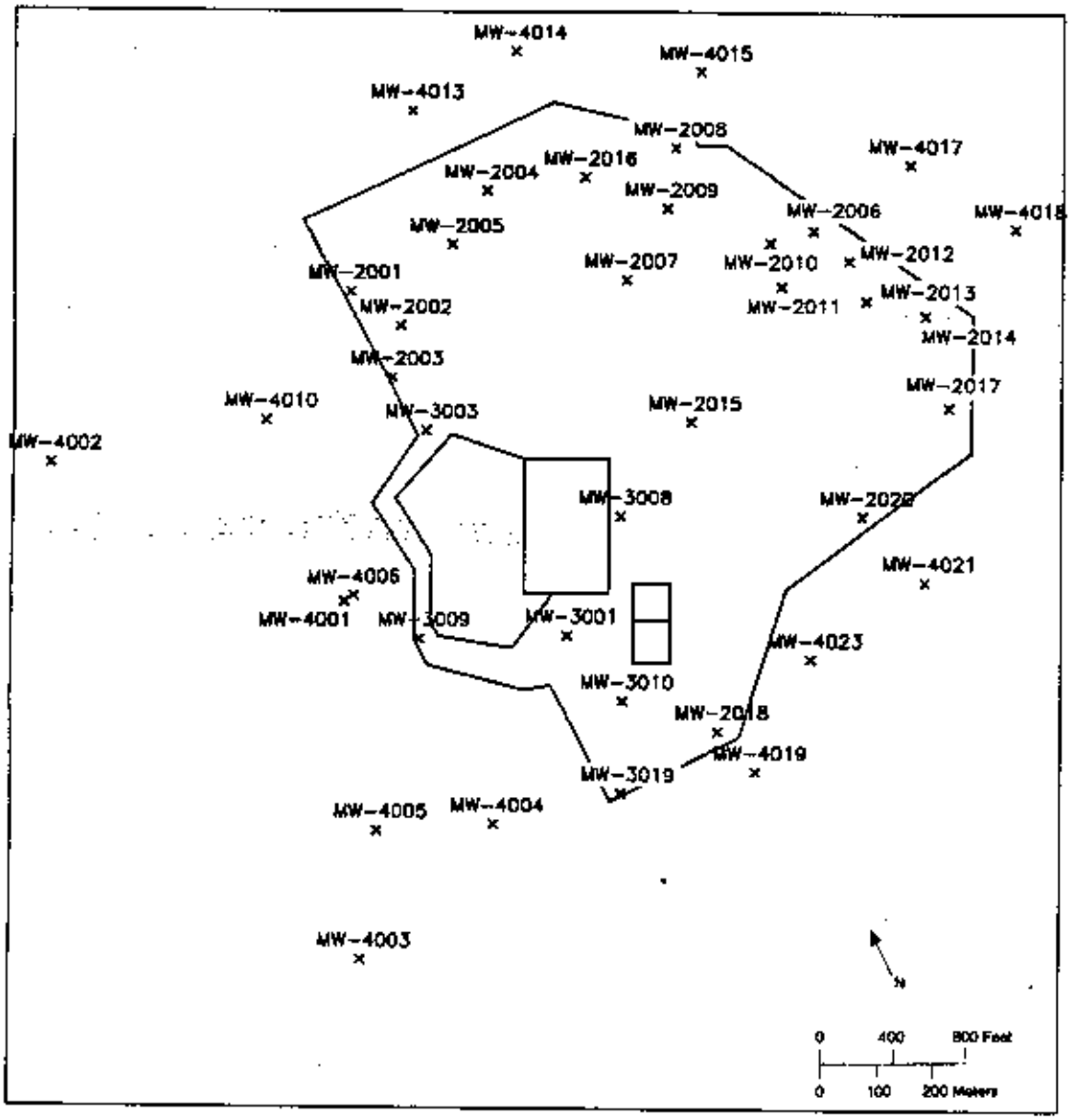


FIGURE B.4 Groundwater Weathered and Weathered/Competent Well Locations at the Weldon Spring Site

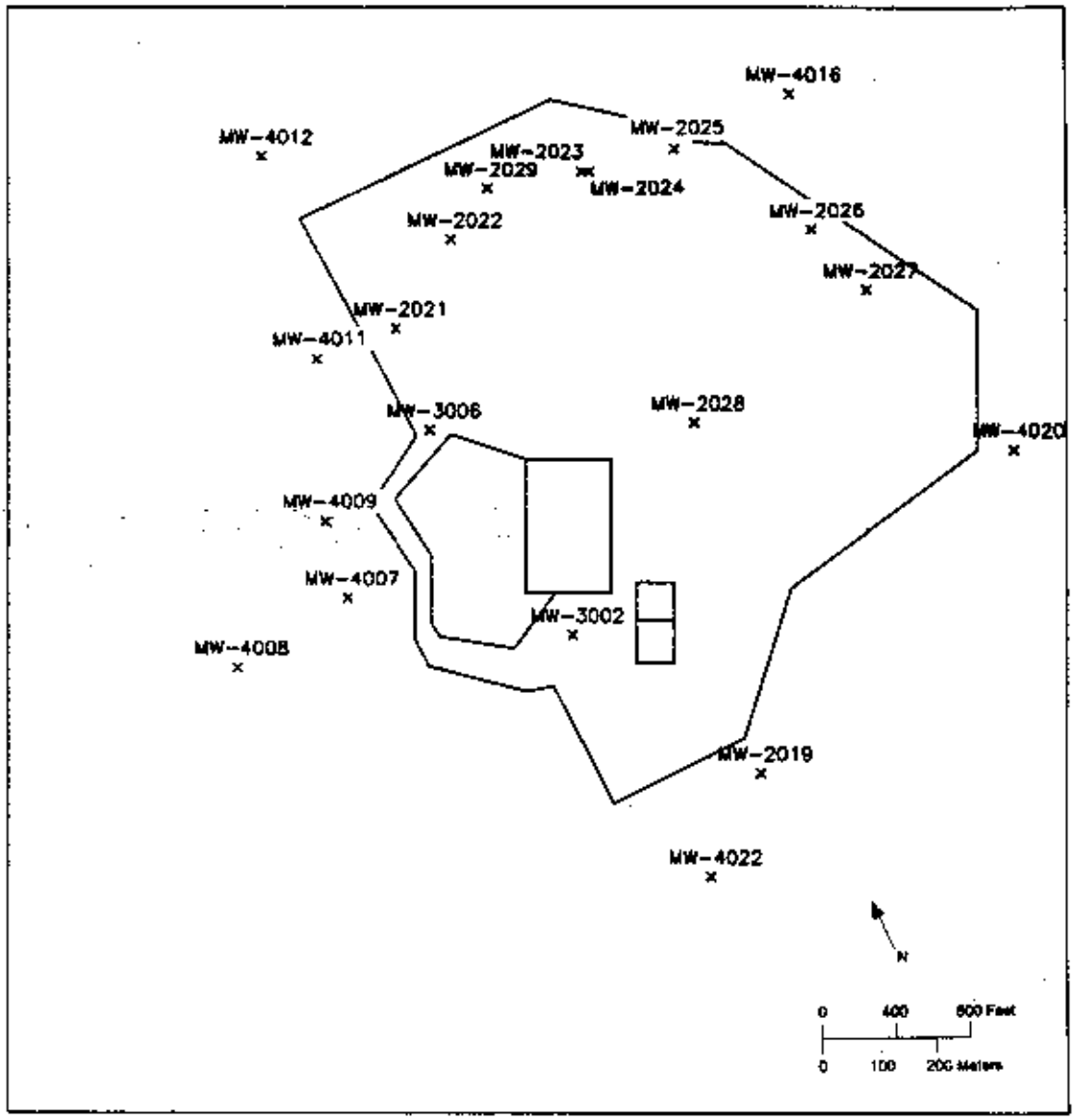


FIGURE B.5 Groundwater Competent Well Locations at the Weldon Spring Site

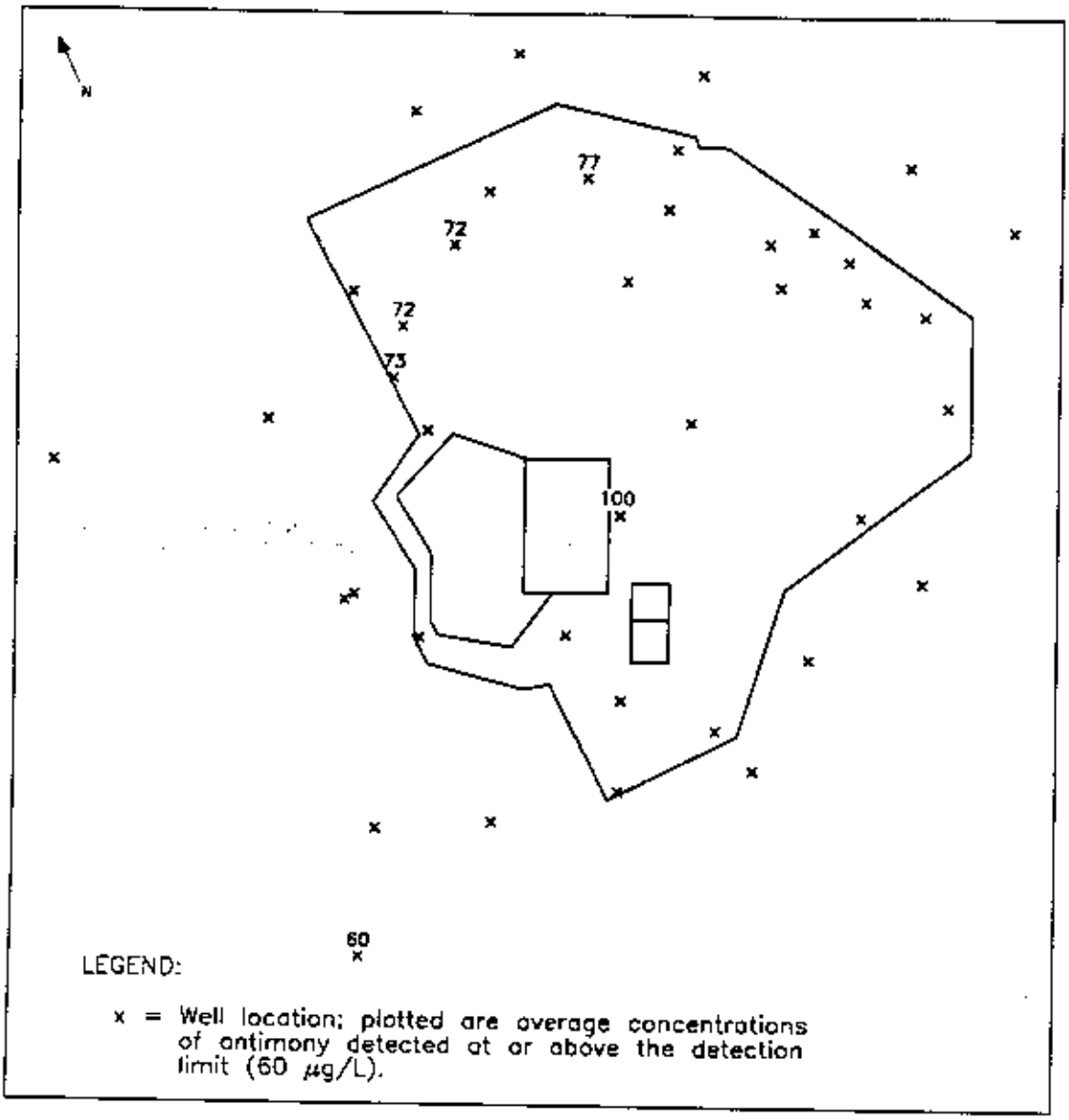


FIGURE B.6 Average Concentrations (µg/L) of Antimony in the Weathered and Weathered/Competent Zones, 1987-1990

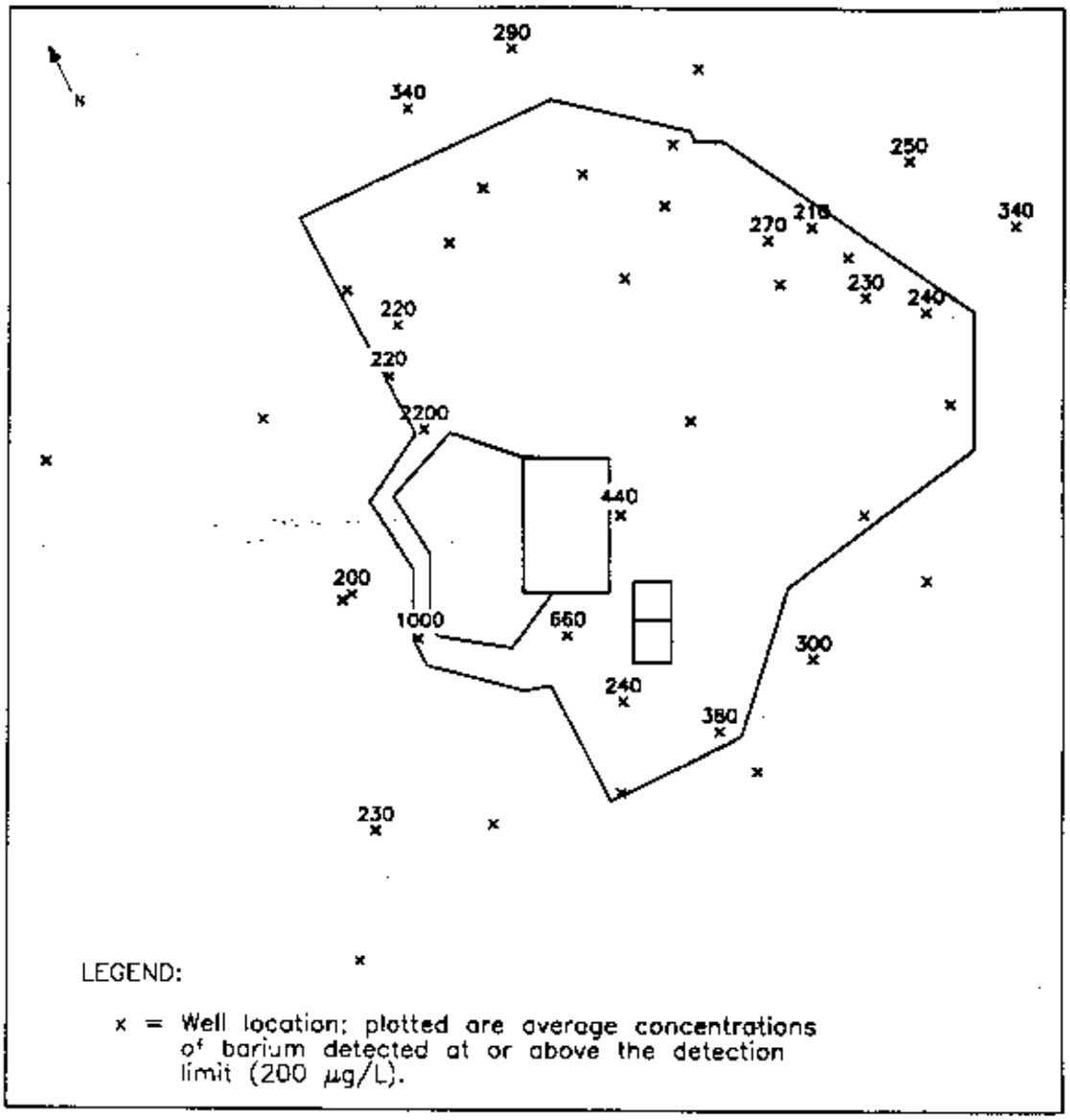


FIGURE B.7 Average Concentrations (µg/L) of Barium in the Weathered and Weathered/Competent Zones, 1987-1990

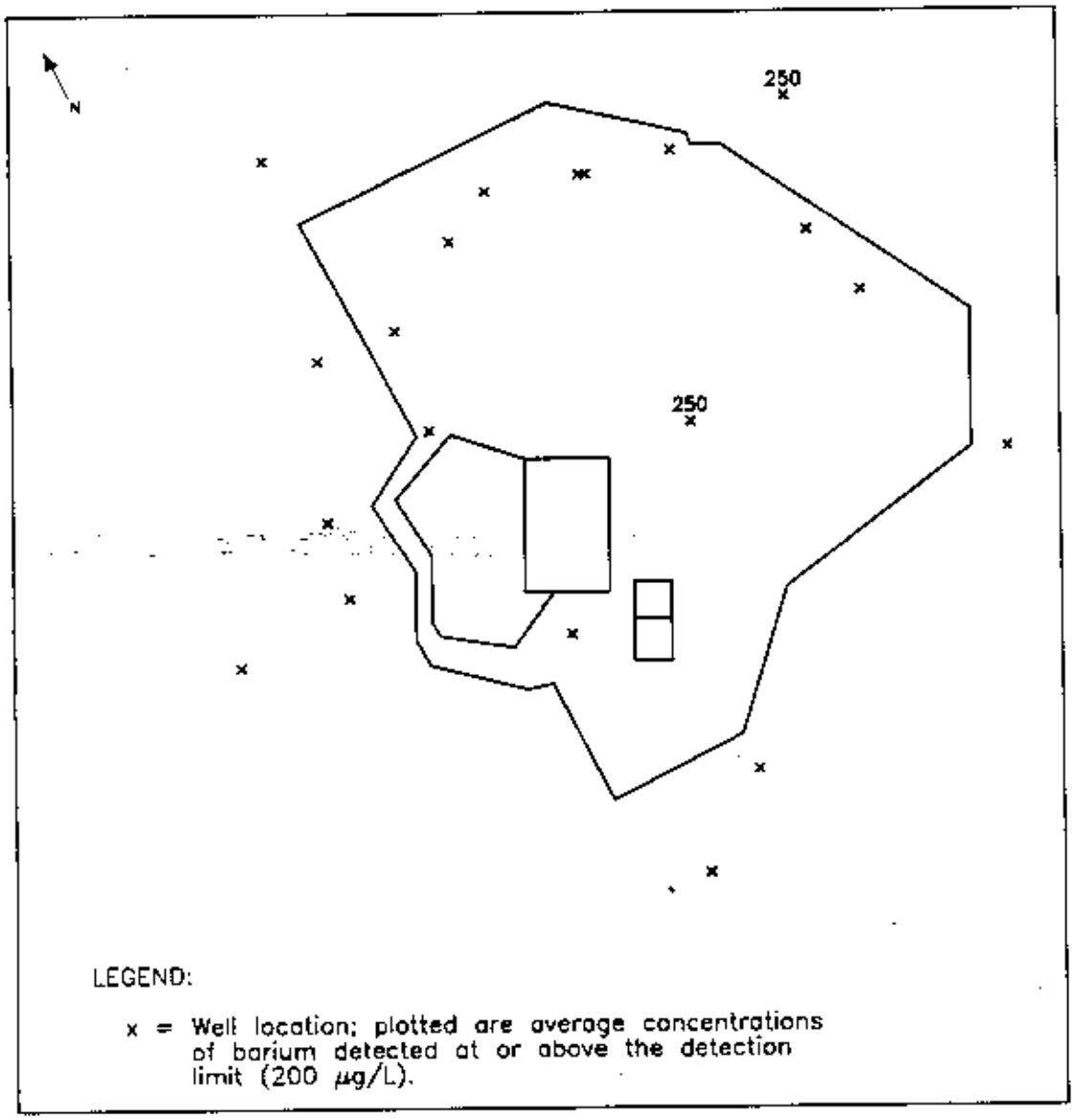


FIGURE B.8 Average Concentrations (μg/L) of Barium in the Competent Zone, 1987-1990

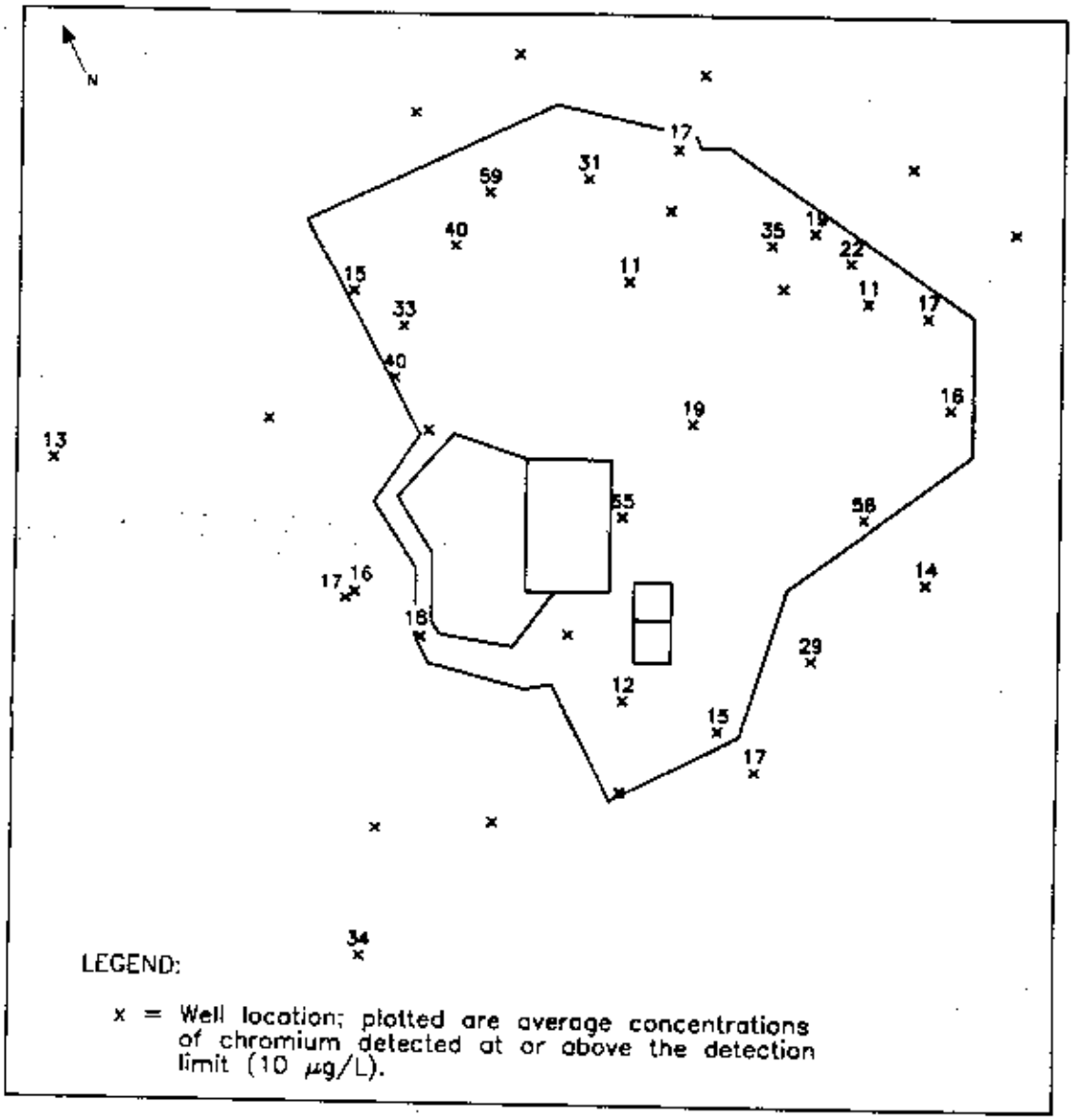


FIGURE B.9 Average Concentrations (μg/L) of Chromium in the Weathered and Weathered/Competent Zones, 1987-1990

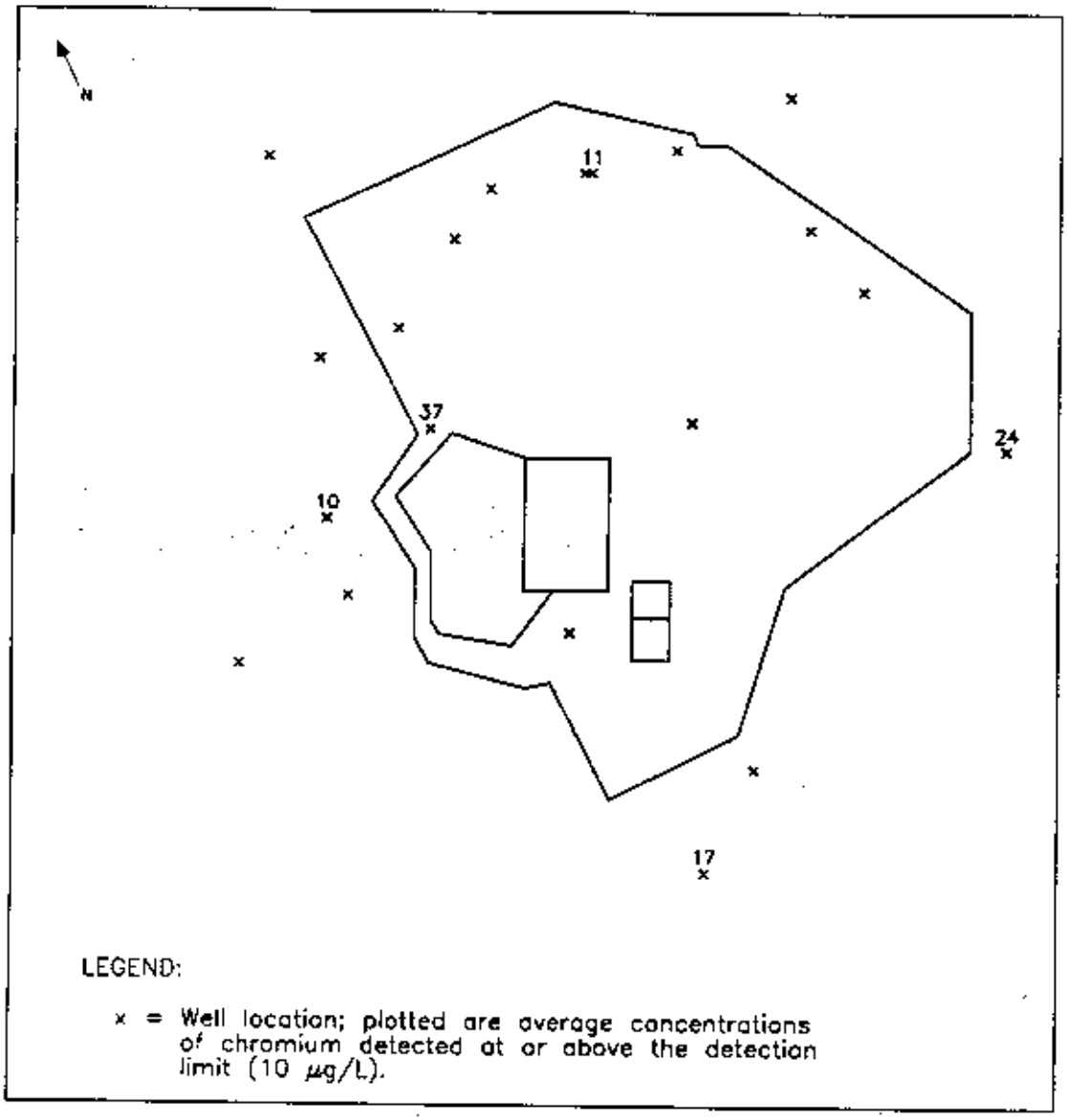


FIGURE B.10 Average Concentrations (µg/L) of Chromium in the Competent Zone, 1987-1990

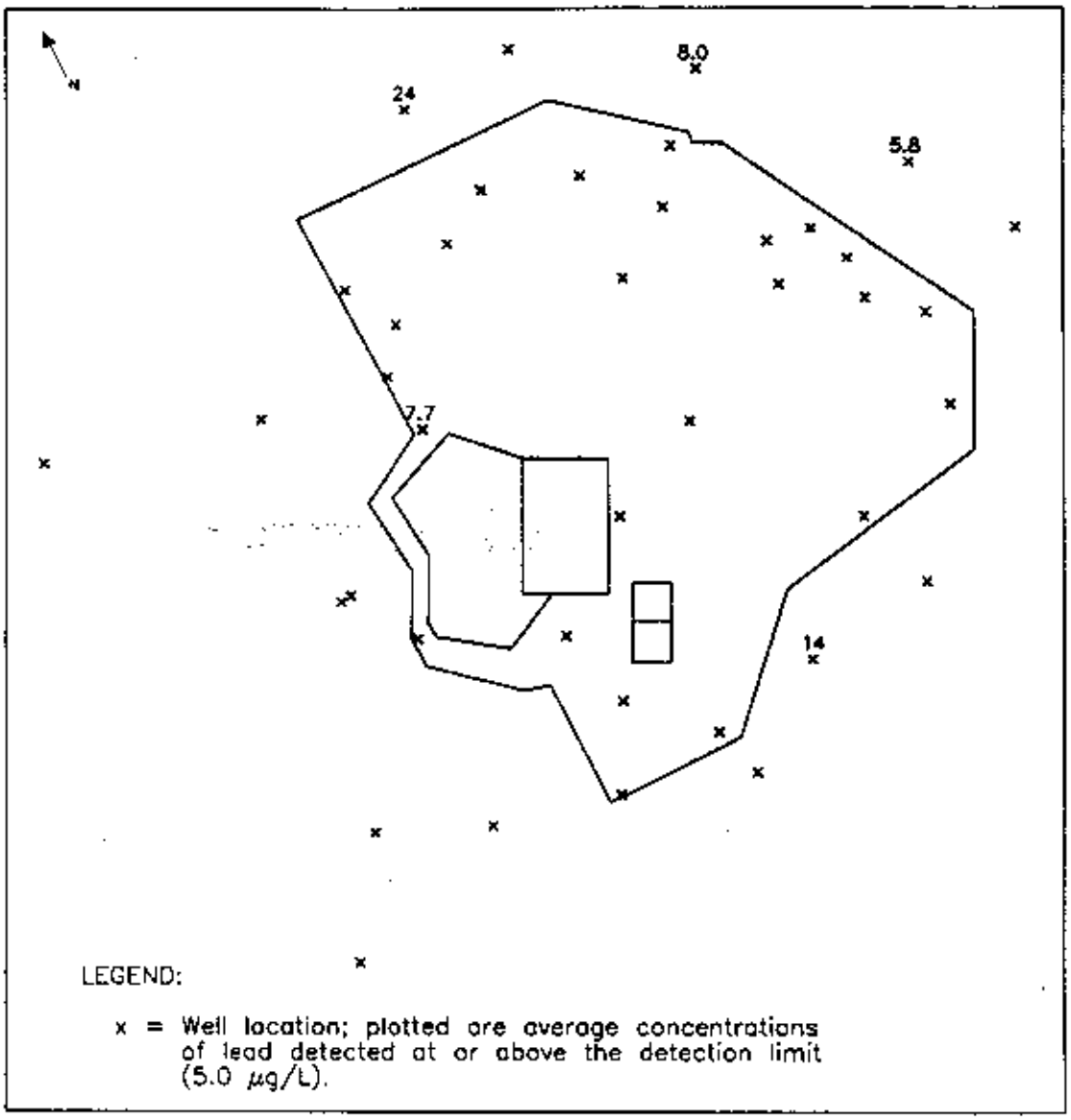


FIGURE B.11 Average Concentrations (µg/L) of Lead in the Weathered and Weathered/Competent Zones, 1987-1990

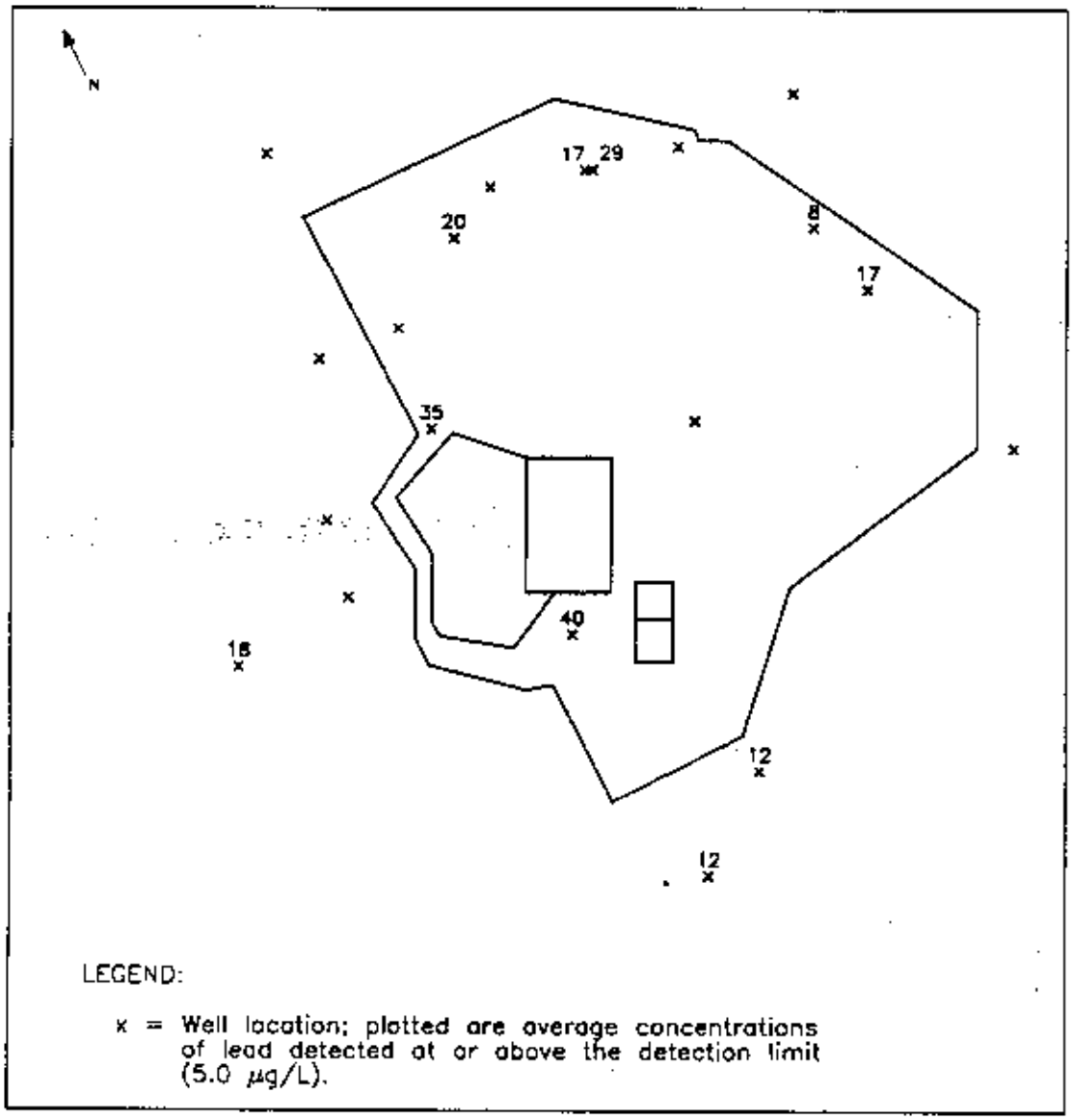


FIGURE B.12 Average Concentrations (µg/L) of Lead in the Competent Zone, 1987-1990

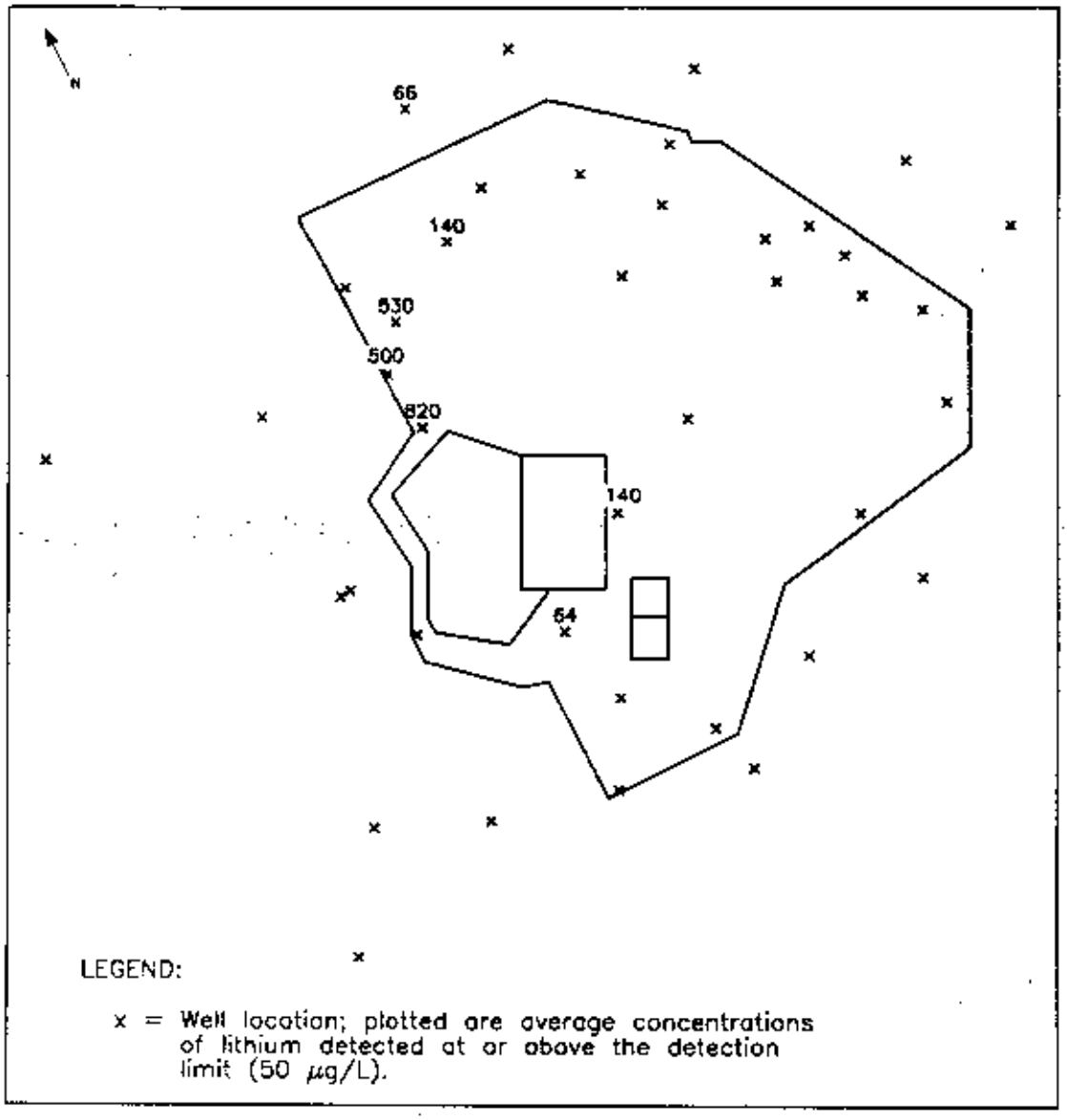


FIGURE B.13 Average Concentrations (μg/L) of Lithium in the Weathered and Weathered/Competent Zones, 1987-1990

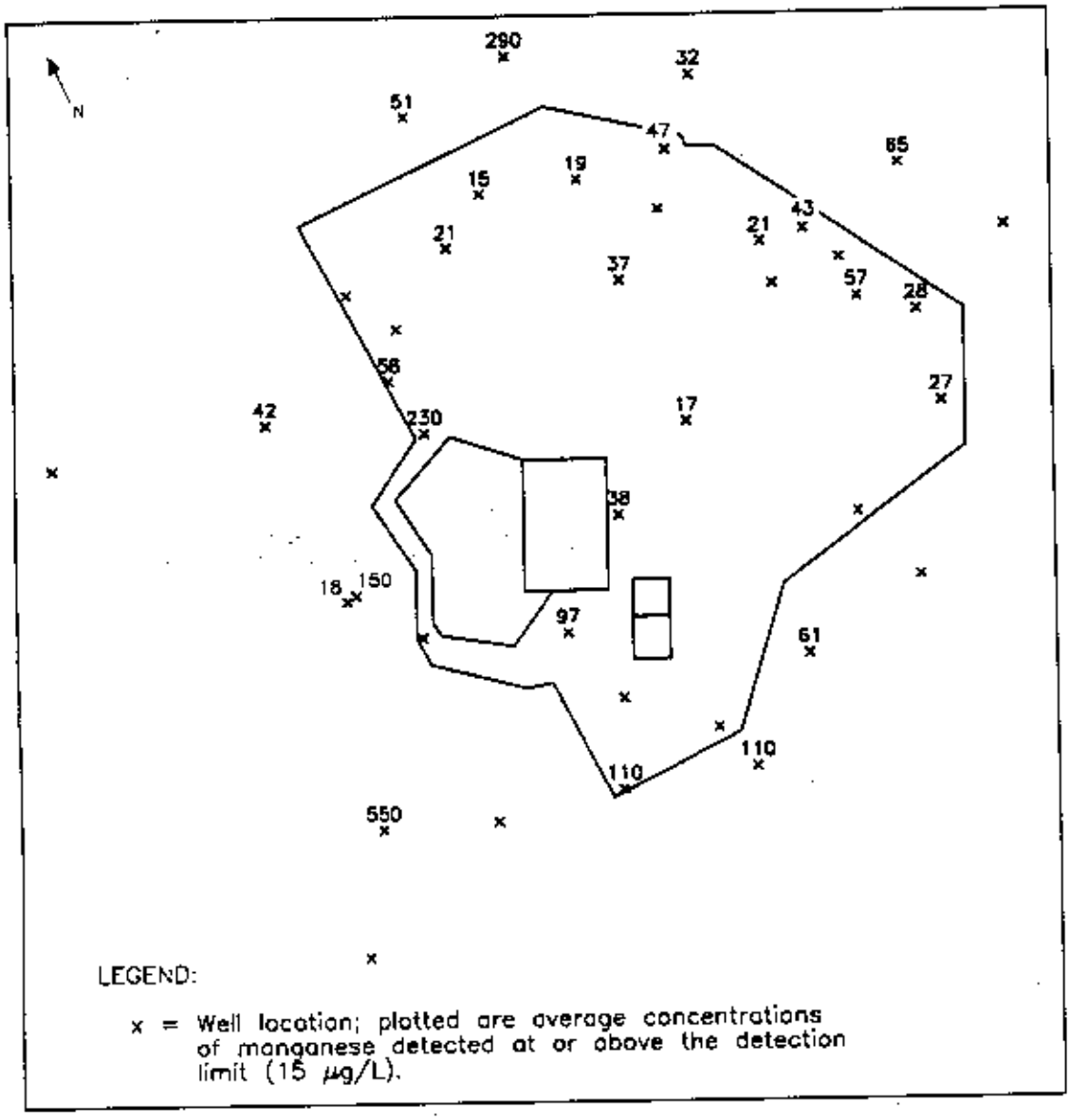


FIGURE B.14 Average Concentrations (µg/L) of Manganese in the Weathered and Weathered/Competent Zones, 1987-1990

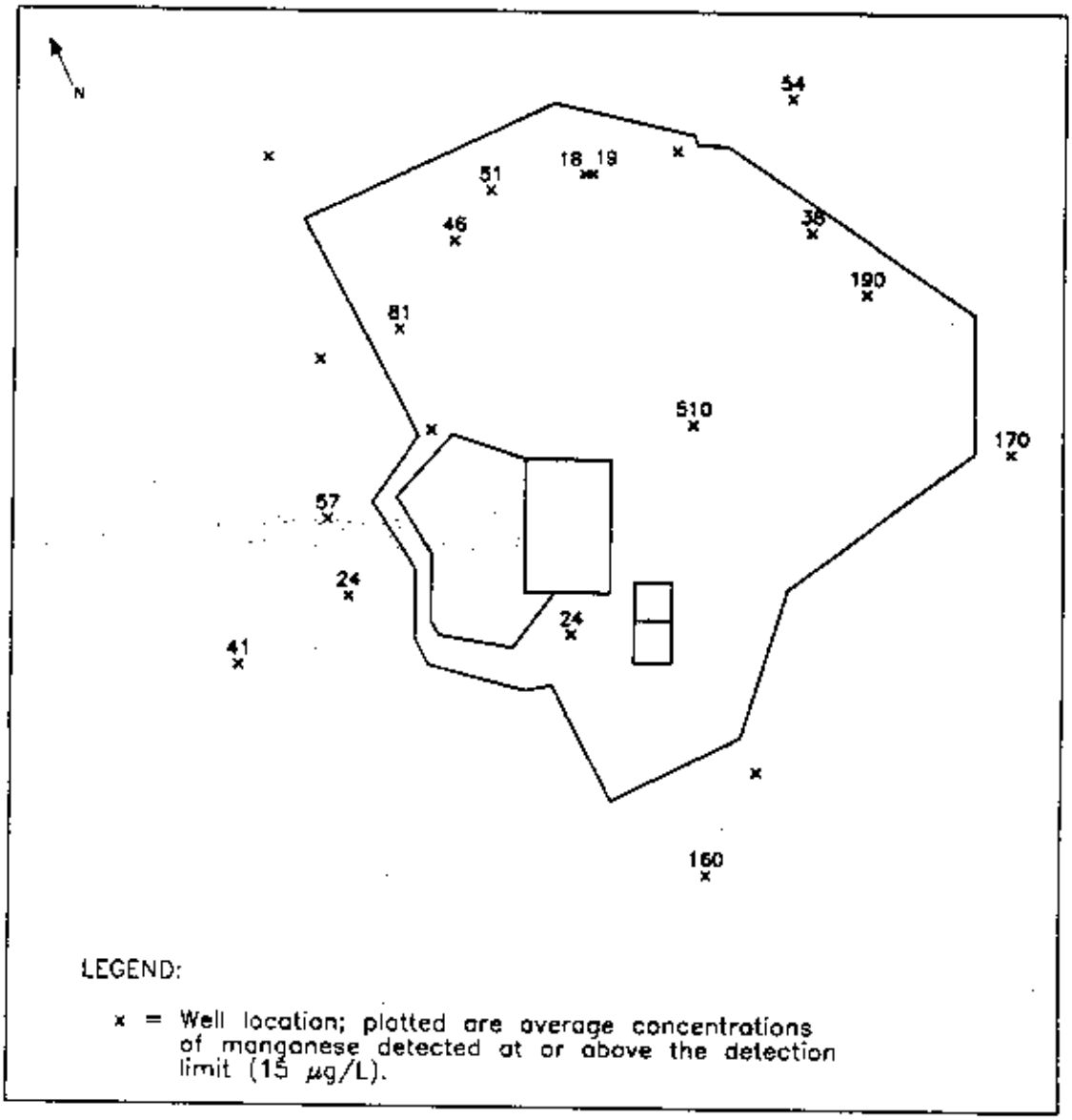


FIGURE B.15 Average Concentrations (μg/L) of Manganese in the Competent Zone, 1987-1990

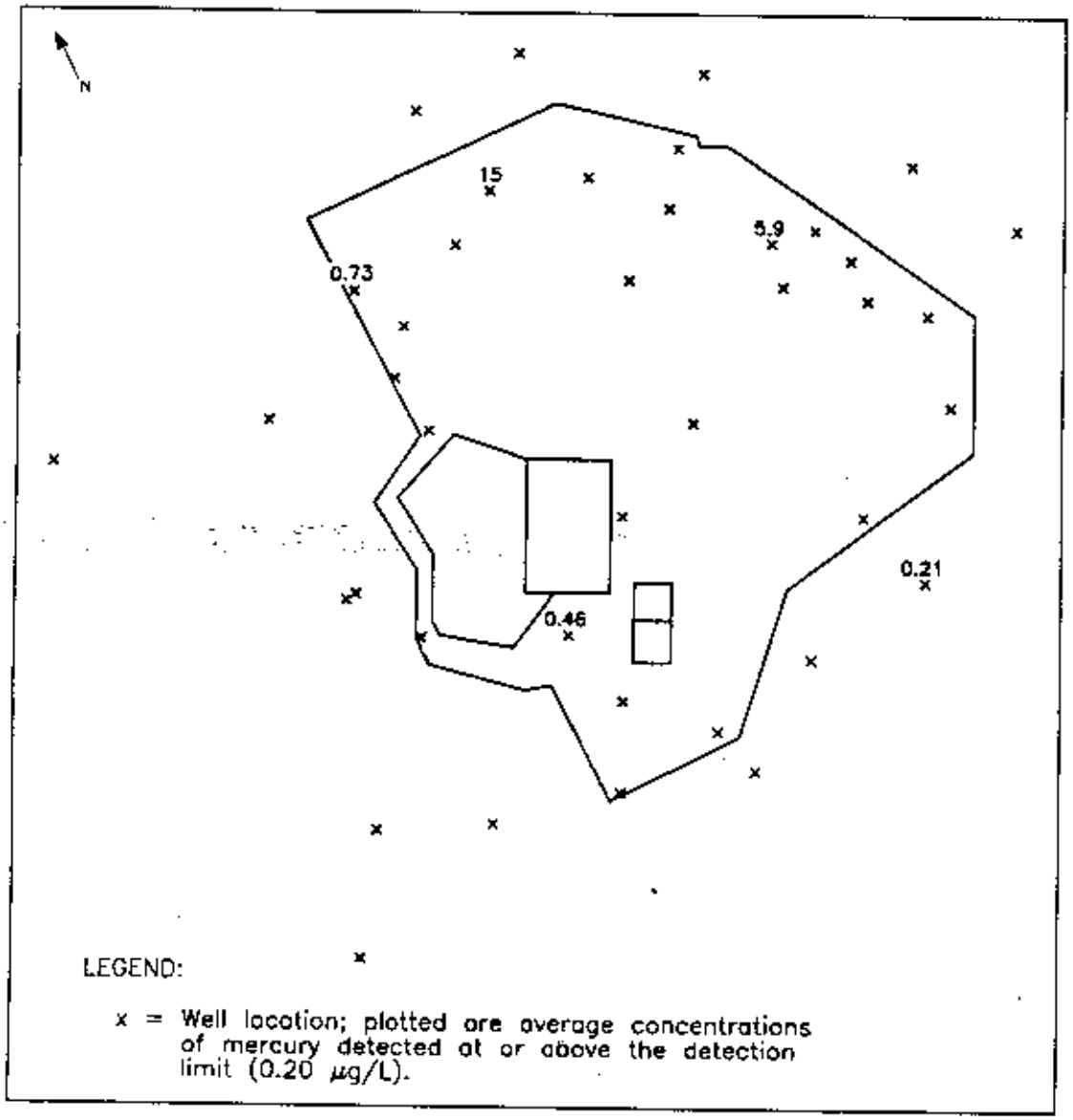


FIGURE B.16 Average Concentrations (µg/L) of Mercury in the Weathered and Weathered/Competent Zones, 1987-1990

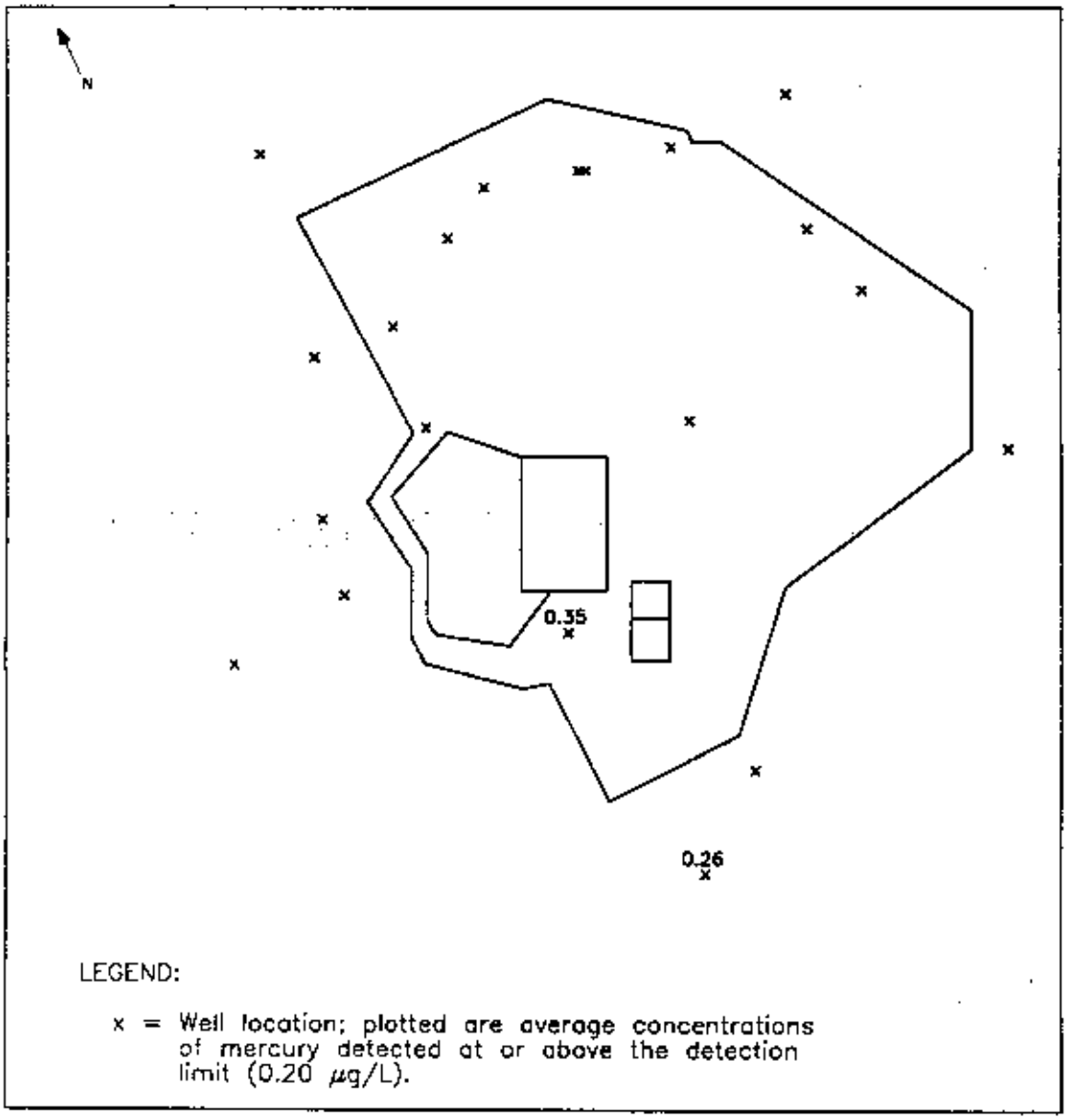


FIGURE B.17 Average Concentrations (μg/L) of Mercury in the Competent Zone, 1987-1990

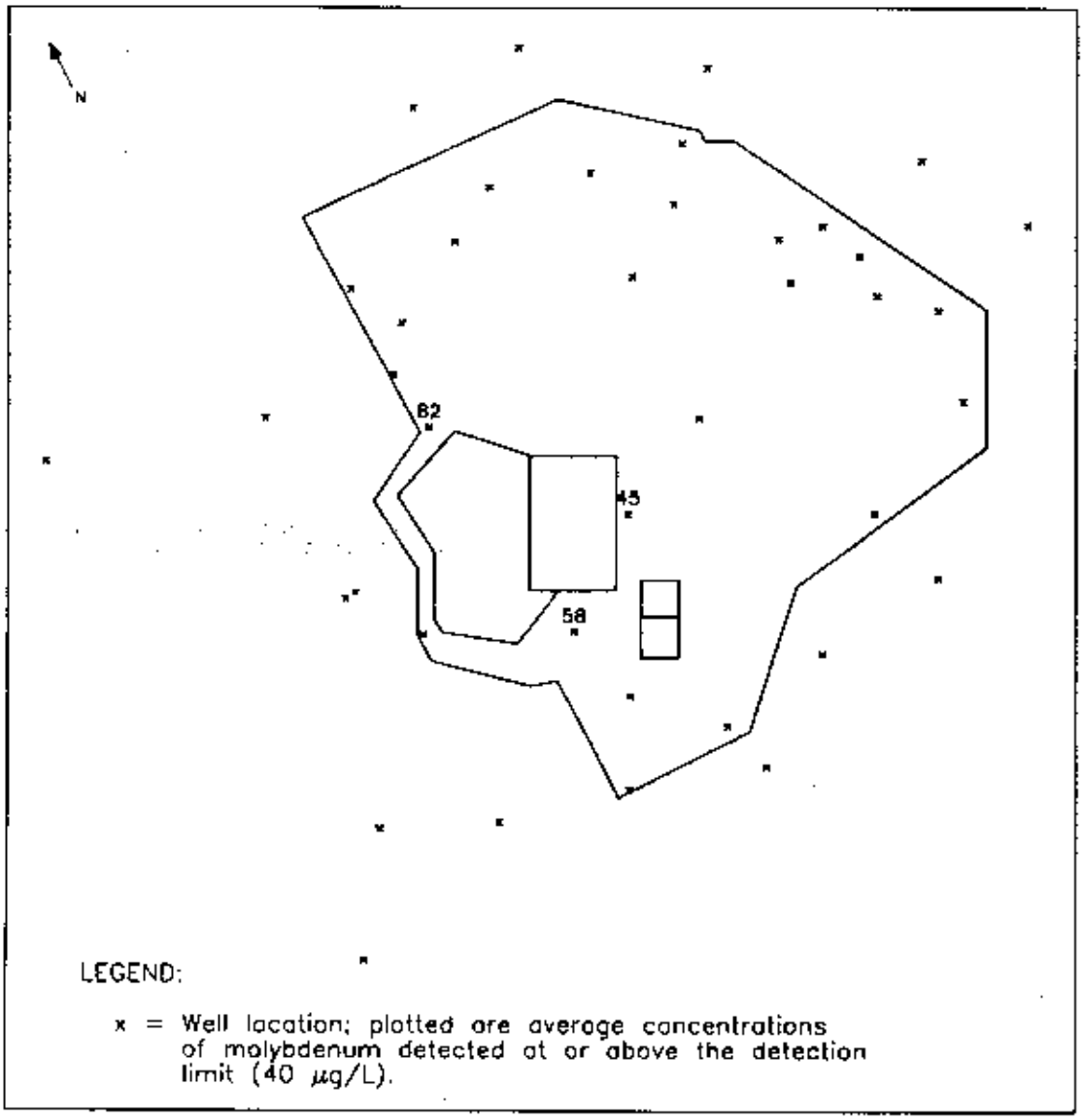


FIGURE B.18 Average Concentrations (µg/L) of Molybdenum in the Weathered and Weathered/Competent Zones, 1987-1990

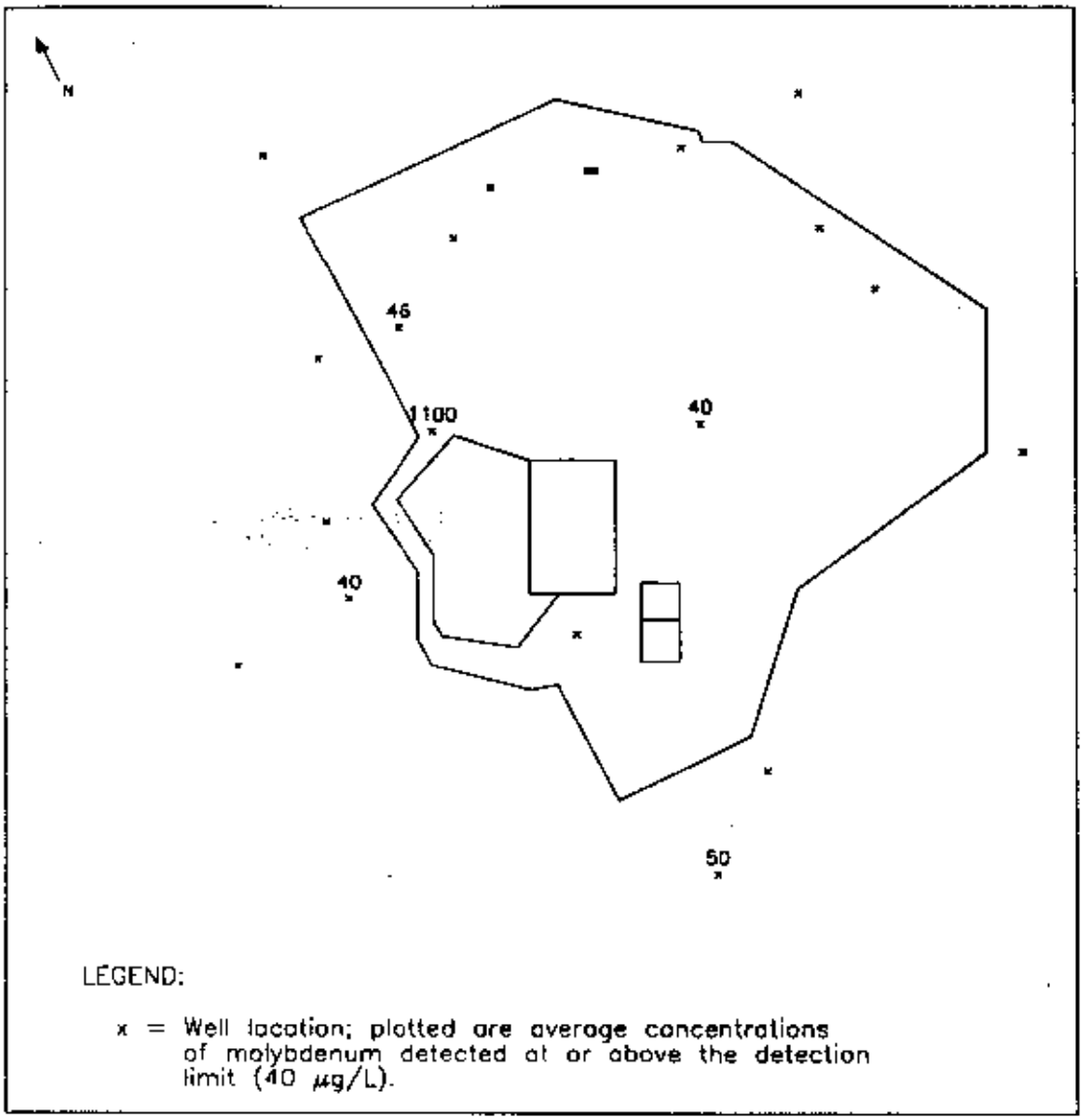


FIGURE B.19 Average Concentrations (μg/L) of Molybdenum in the Competent Zone, 1987-1990

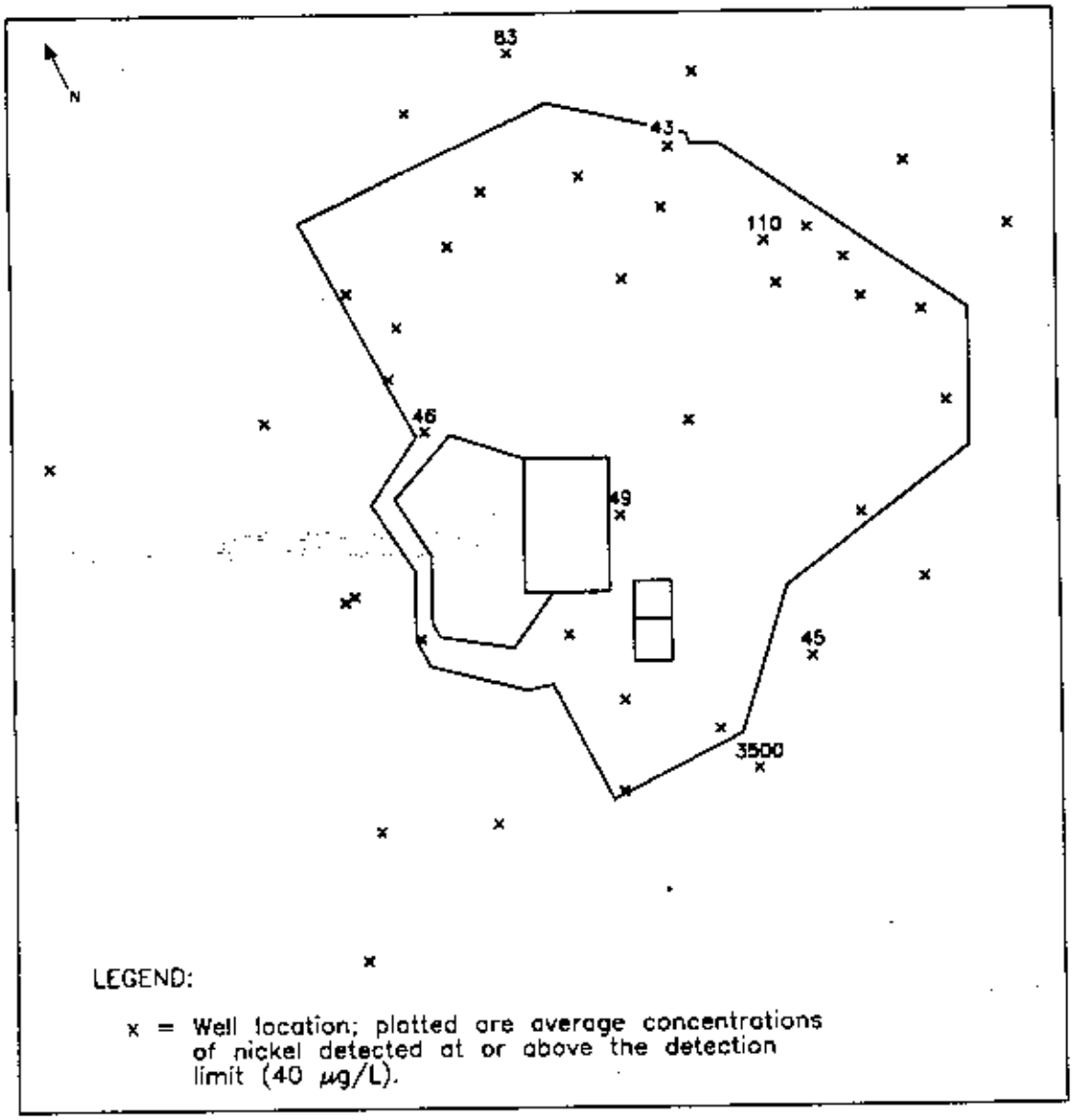


FIGURE B.20 Average Concentrations (μg/L) of Nickel in the Weathered and Weathered/Competent Zones, 1987-1990

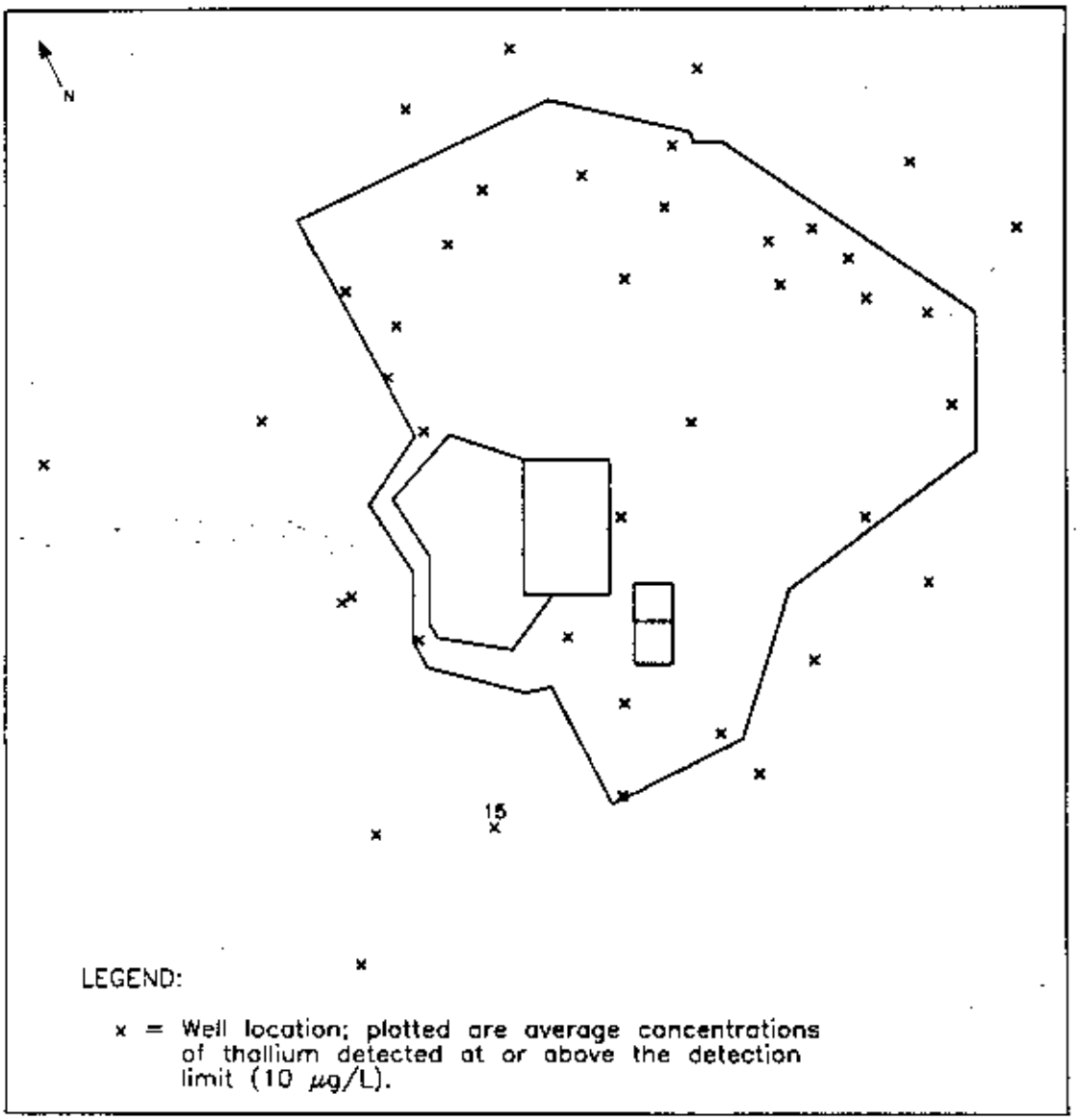


FIGURE B.21 Average Concentrations (μg/L) of Thallium in the Weathered and Weathered/Competent Zones, 1987-1990

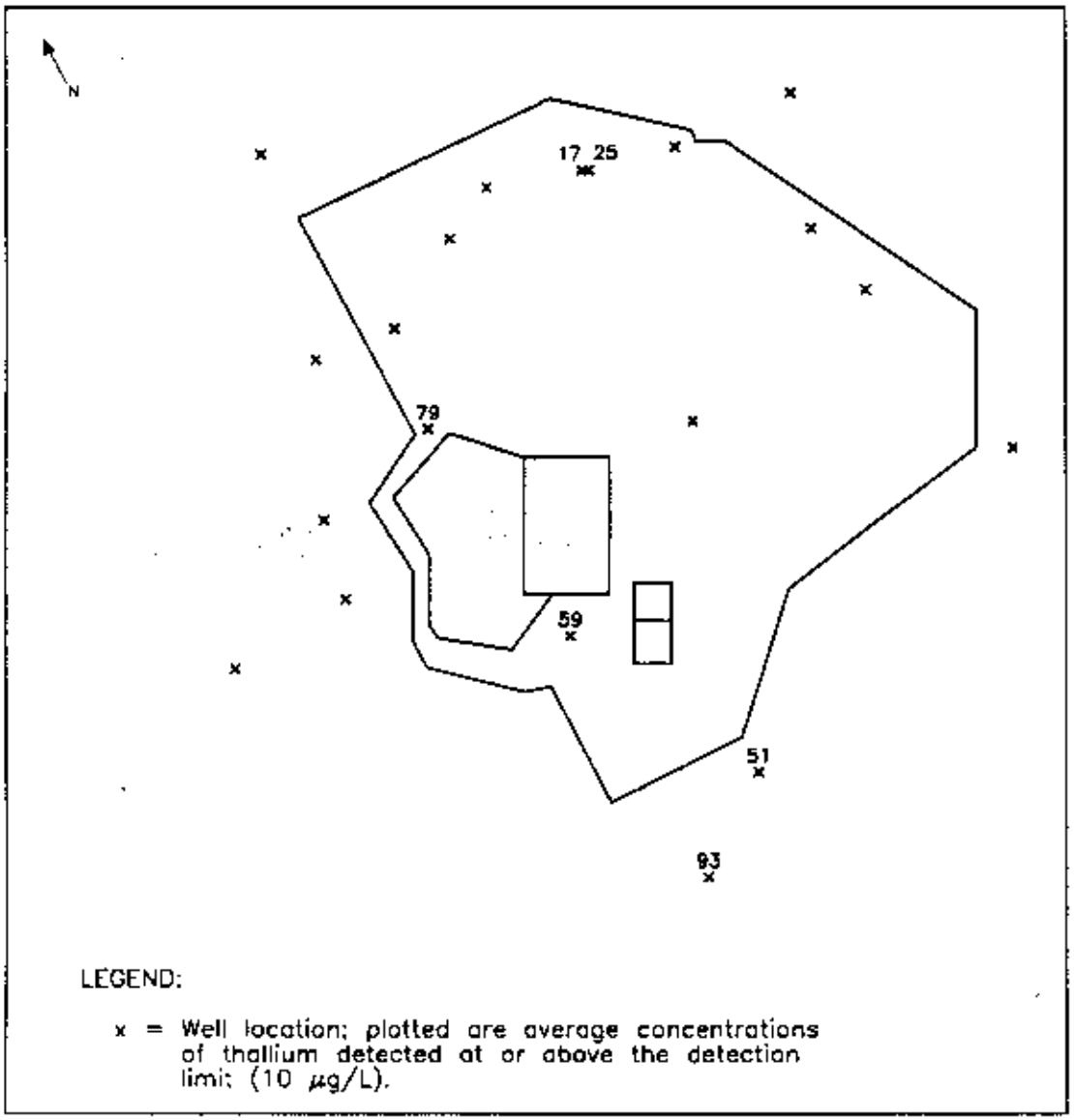


FIGURE B.22 Average Concentrations (µg/L) of Thallium in the Competent Zone, 1987-1990

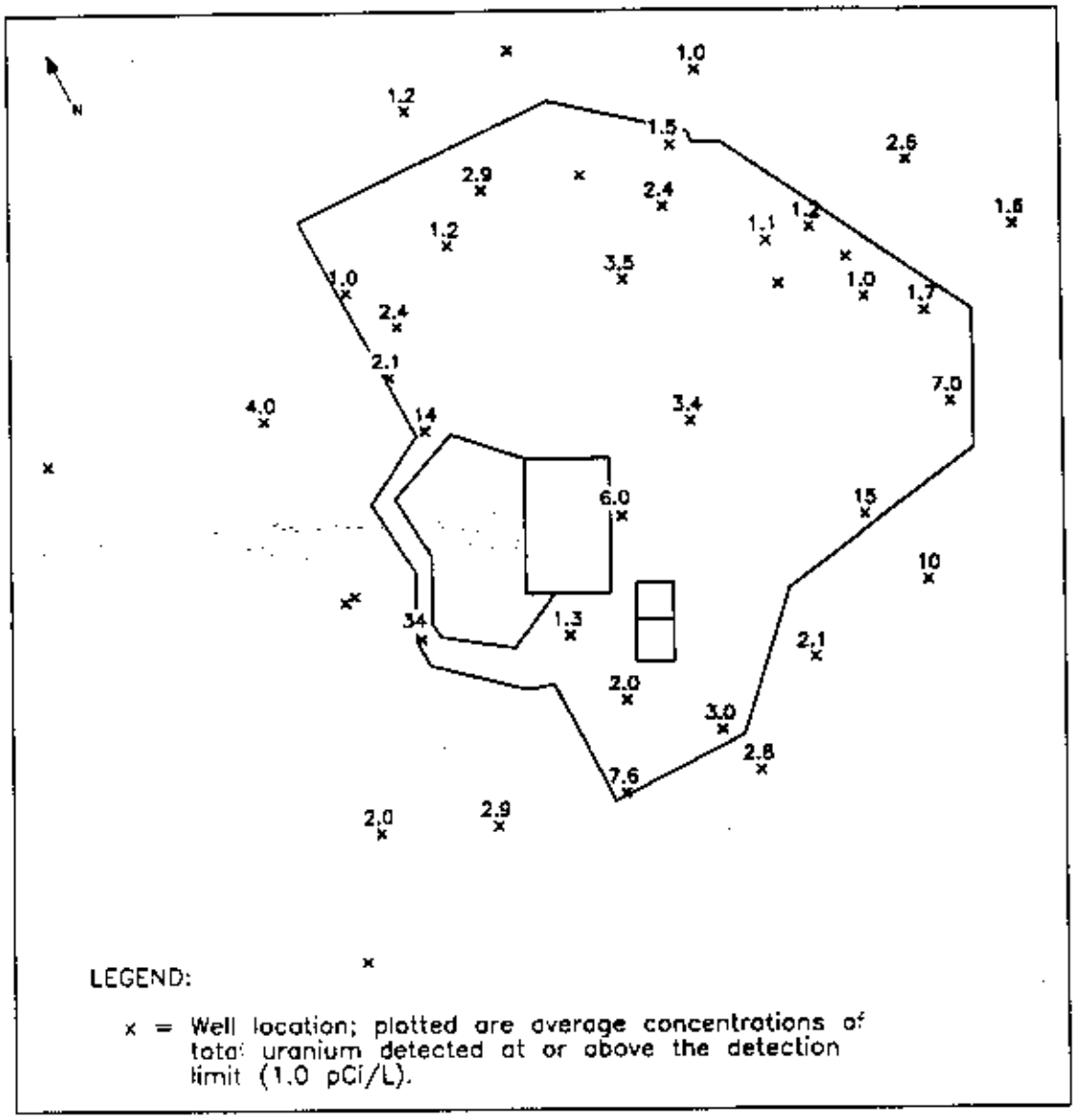


FIGURE B.23 Average Concentrations (pCi/L) of Uranium in the Weathered and Weathered/Competent Zones, 1987-1990

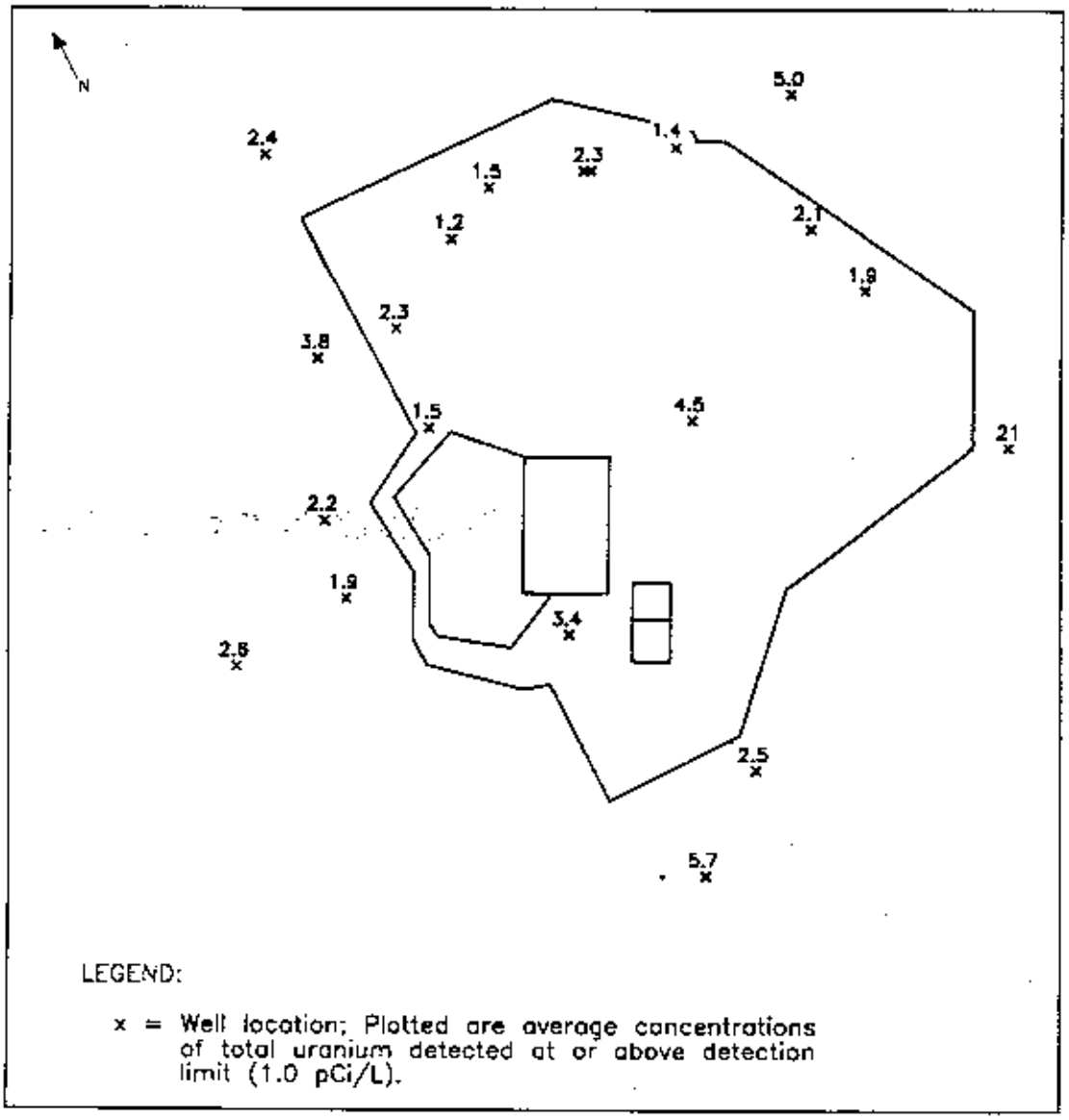


FIGURE B.24 Average Concentrations (pCi/L) of Uranium in the Competent Zone, 1987-1990

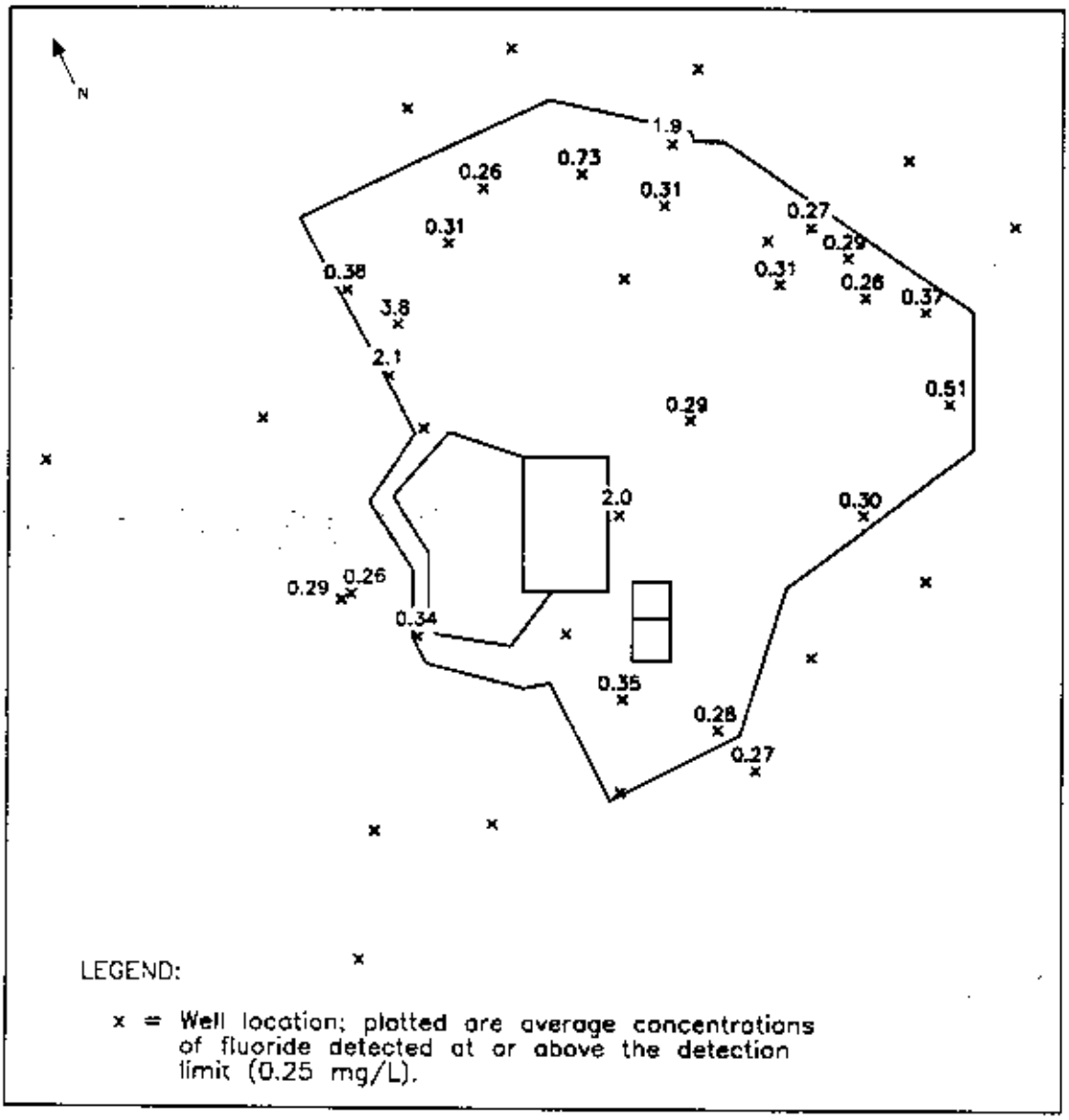


FIGURE B.25 Average Concentrations (mg/L) of Fluoride in the Weathered and Weathered/Competent Zones, 1987-1990

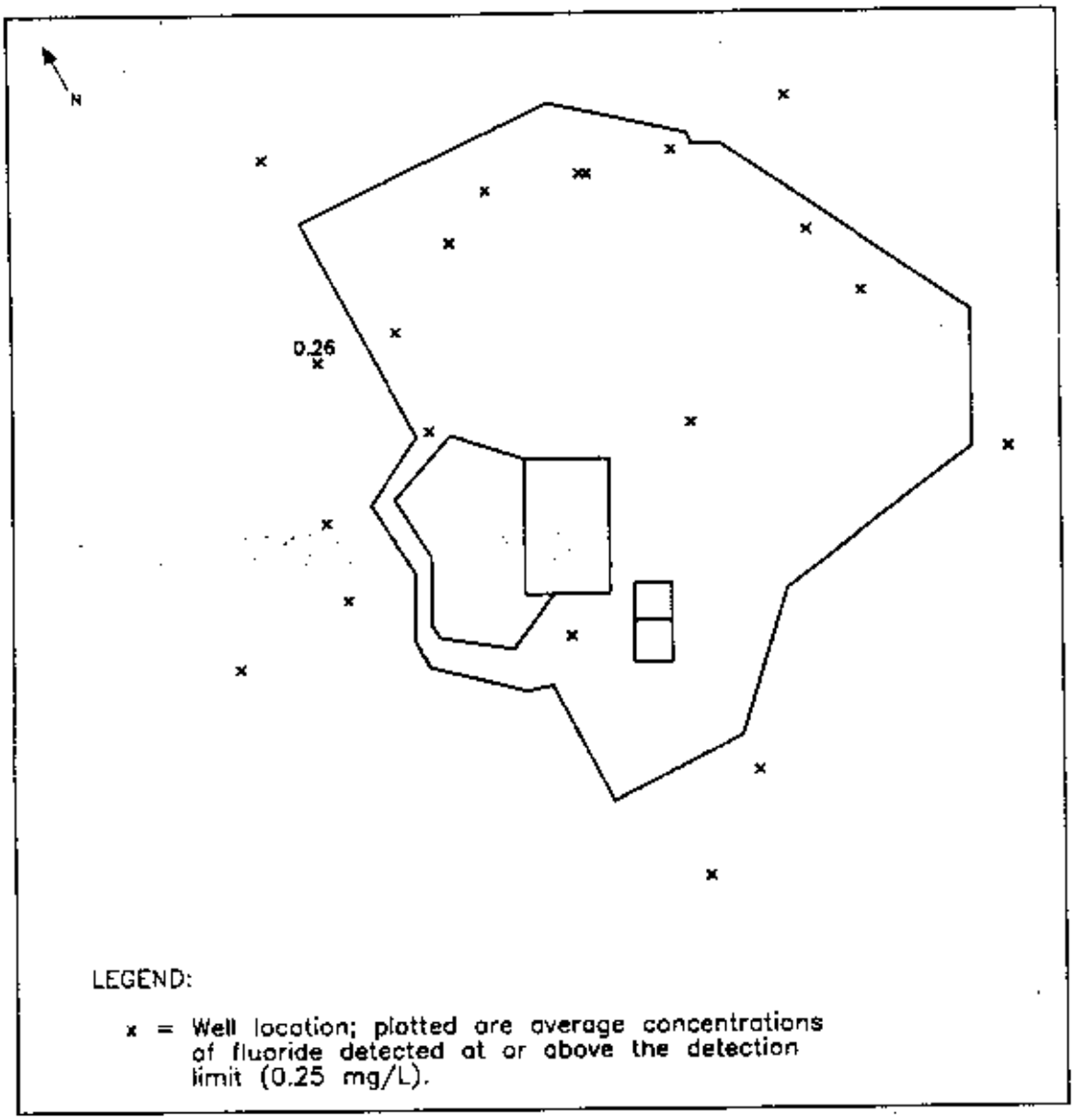


FIGURE B.26 Average Concentrations (mg/L) of Fluoride in the Competent Zone, 1987-1990

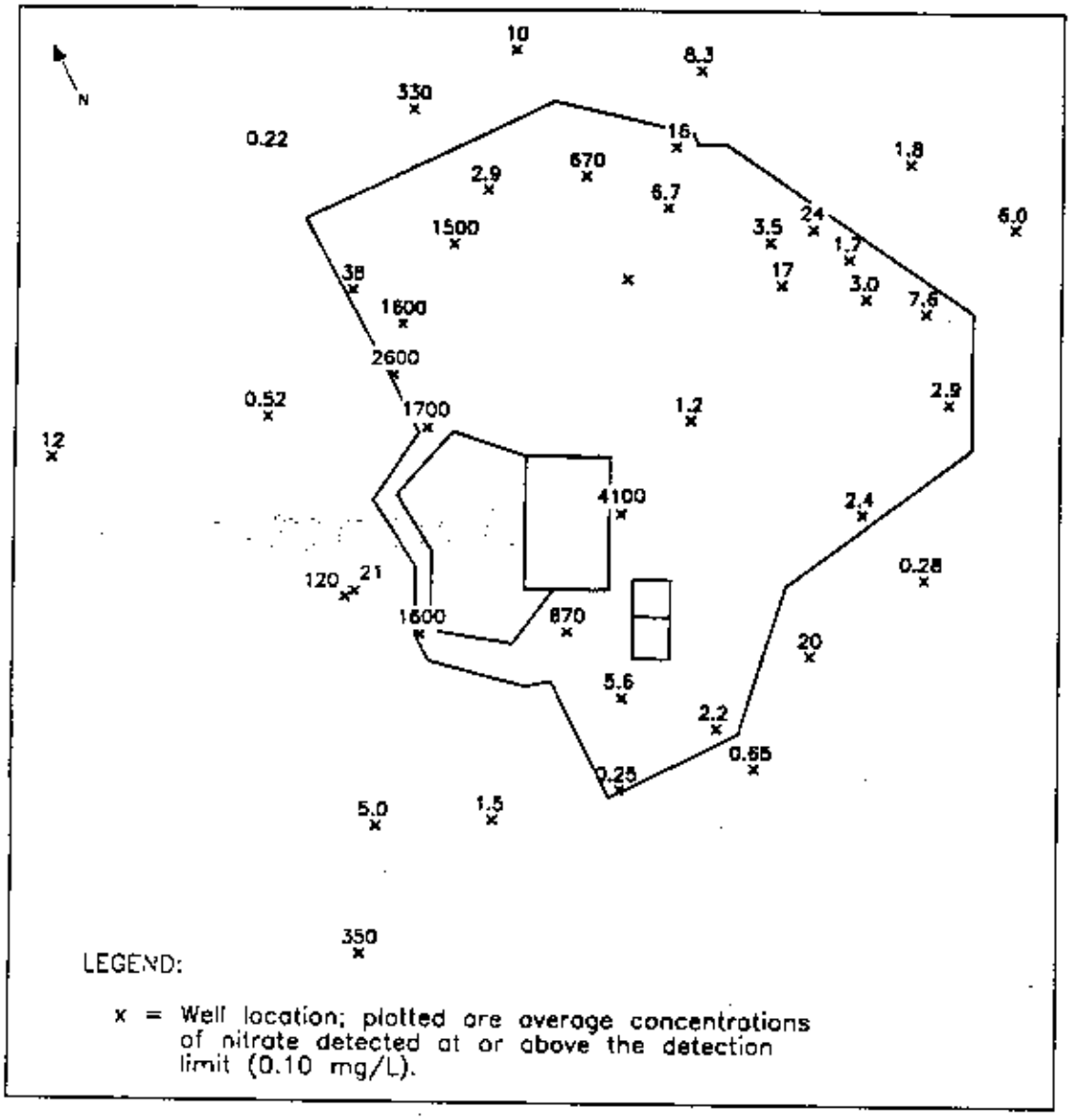


FIGURE B.27 Average Concentrations (mg/L) of Nitrate in the Weathered and Weathered/Competent Zones, 1987-1990

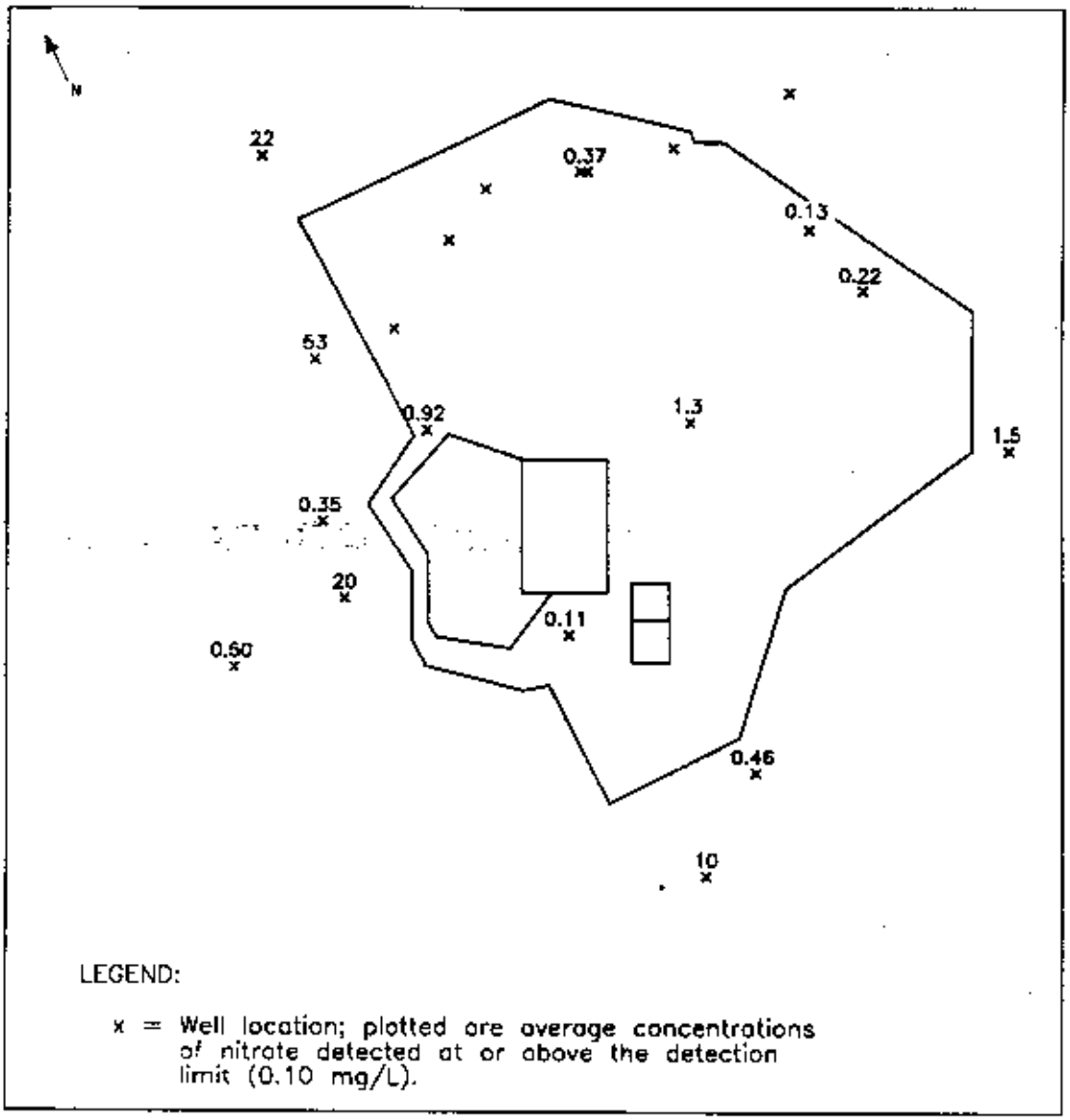


FIGURE B.28 Average Concentrations (mg/L) of Nitrate in the Competent Zone, 1987-1990

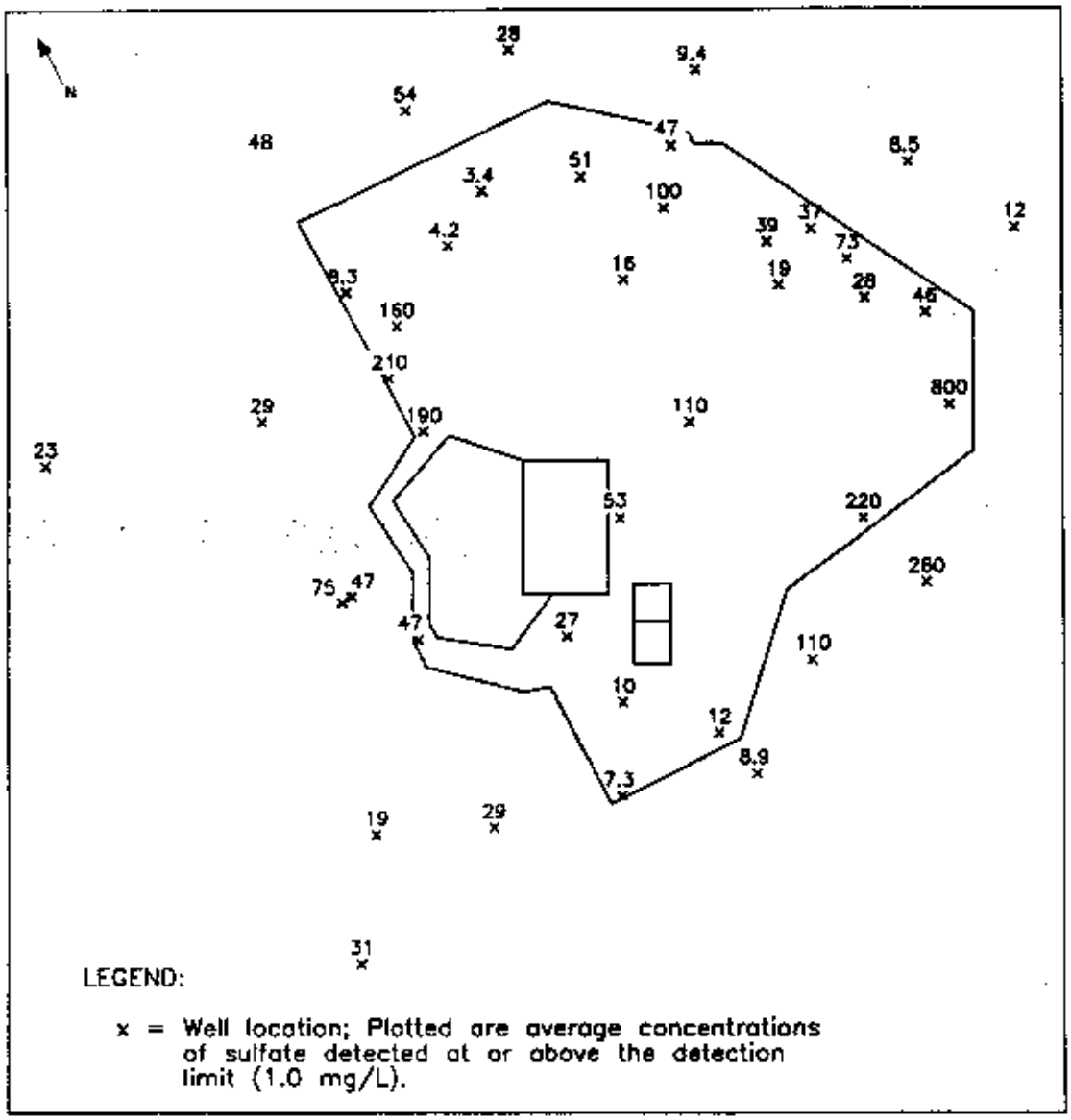


FIGURE B.29 Average Concentrations (mg/L) of Sulfate in the Weathered and Weathered/Competent Zones, 1987-1990

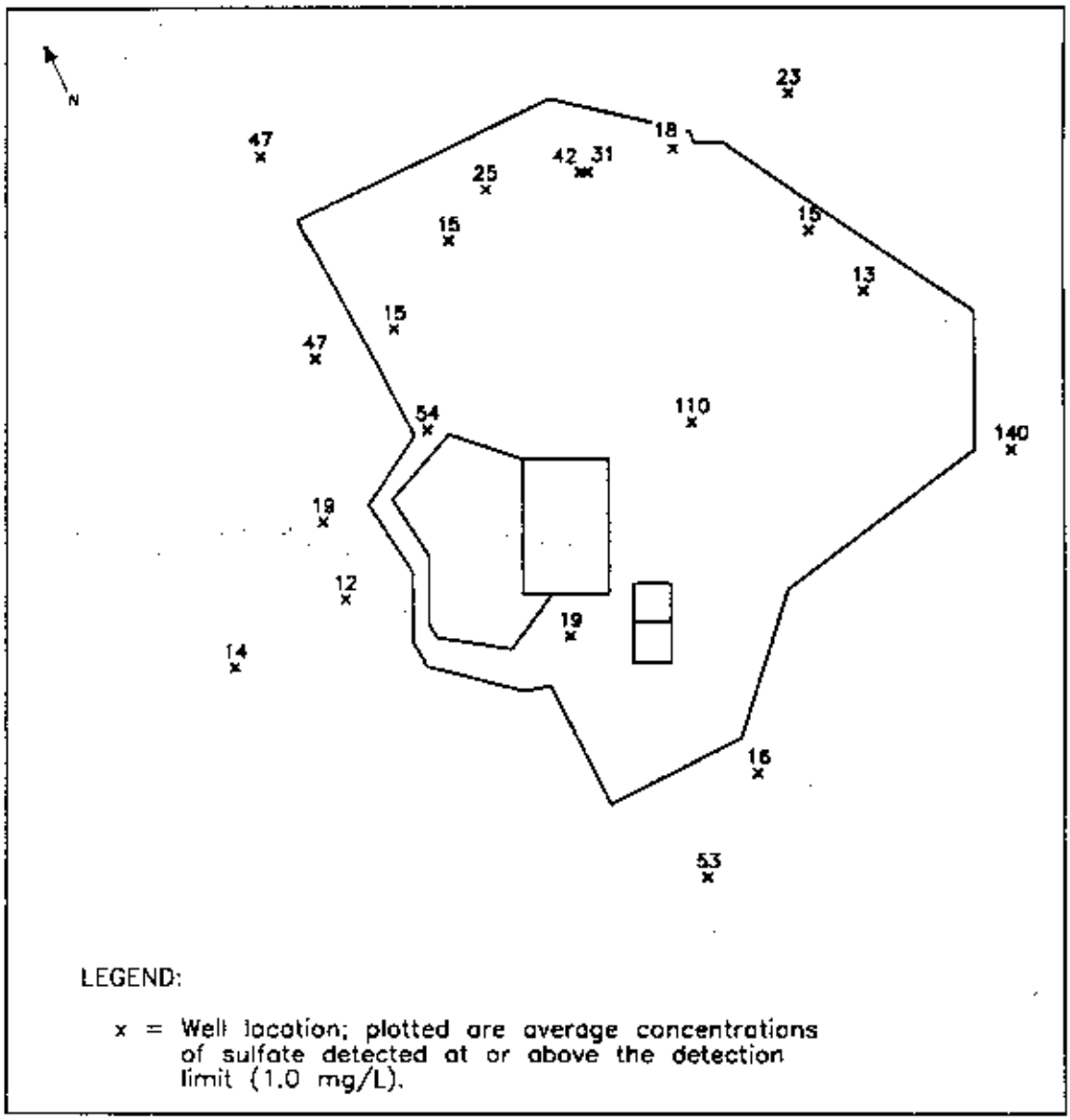


FIGURE B.30 Average Concentrations (mg/L) of Sulfate in the Competent Zone, 1987-1990

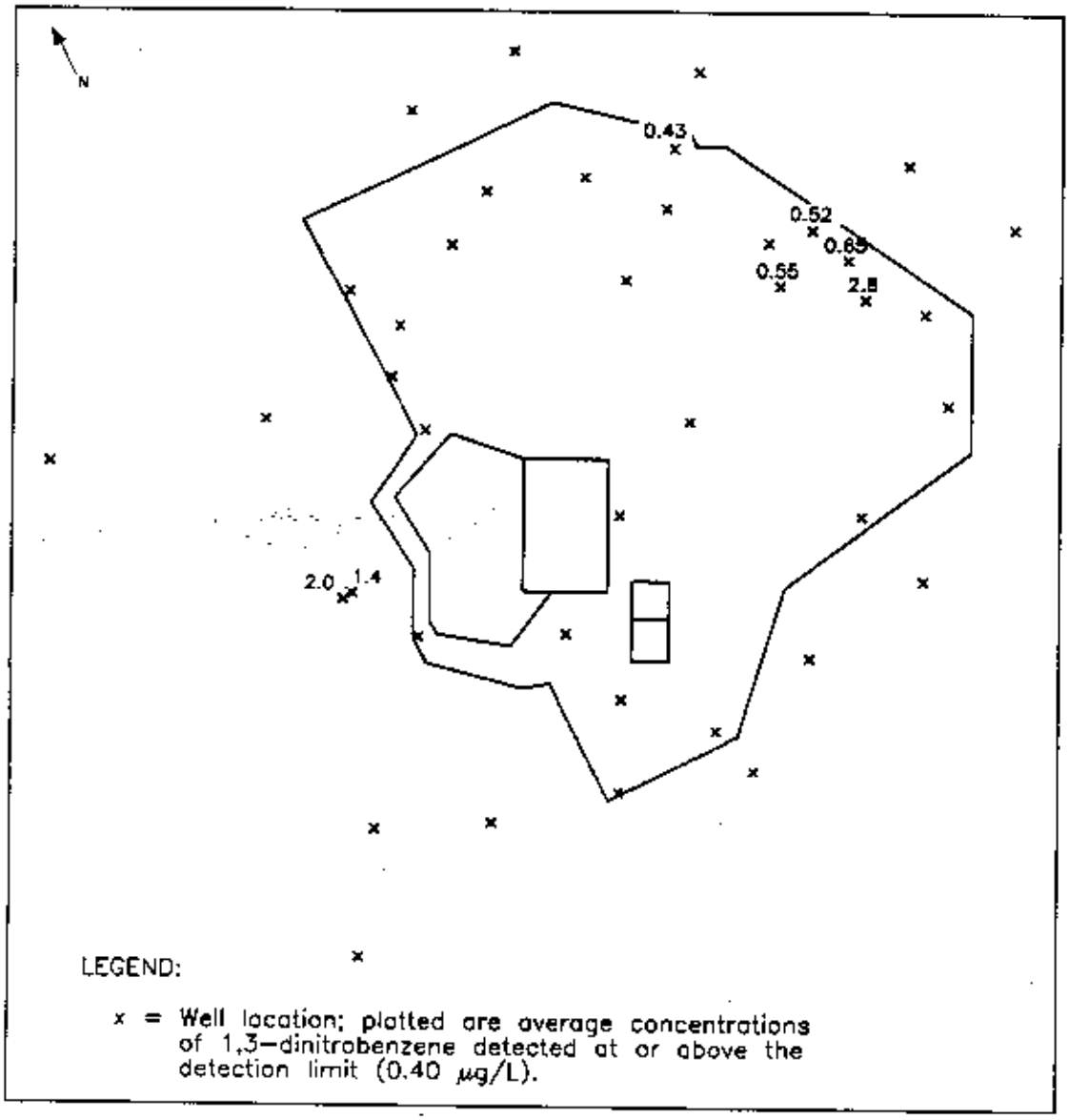


FIGURE B.31 Average Concentrations (µg/L) of 1,3-Dinitrobenzene in the Weathered and Weathered/Competent Zones, 1987-1990

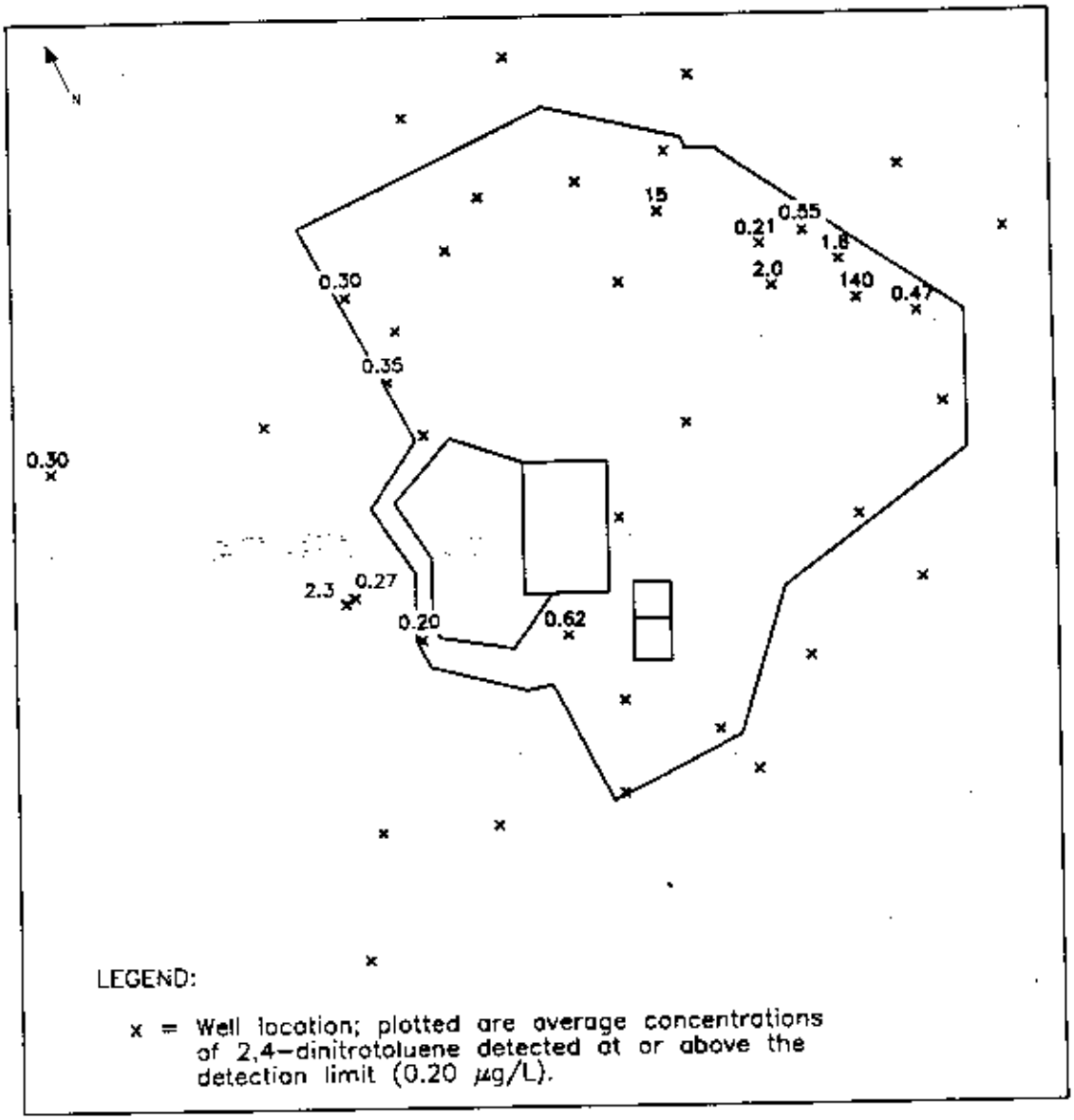


FIGURE B.32 Average Concentrations (μg/L) of 2,4-Dinitrotoluene in the Weathered and Weathered/Competent Zones, 1987-1990

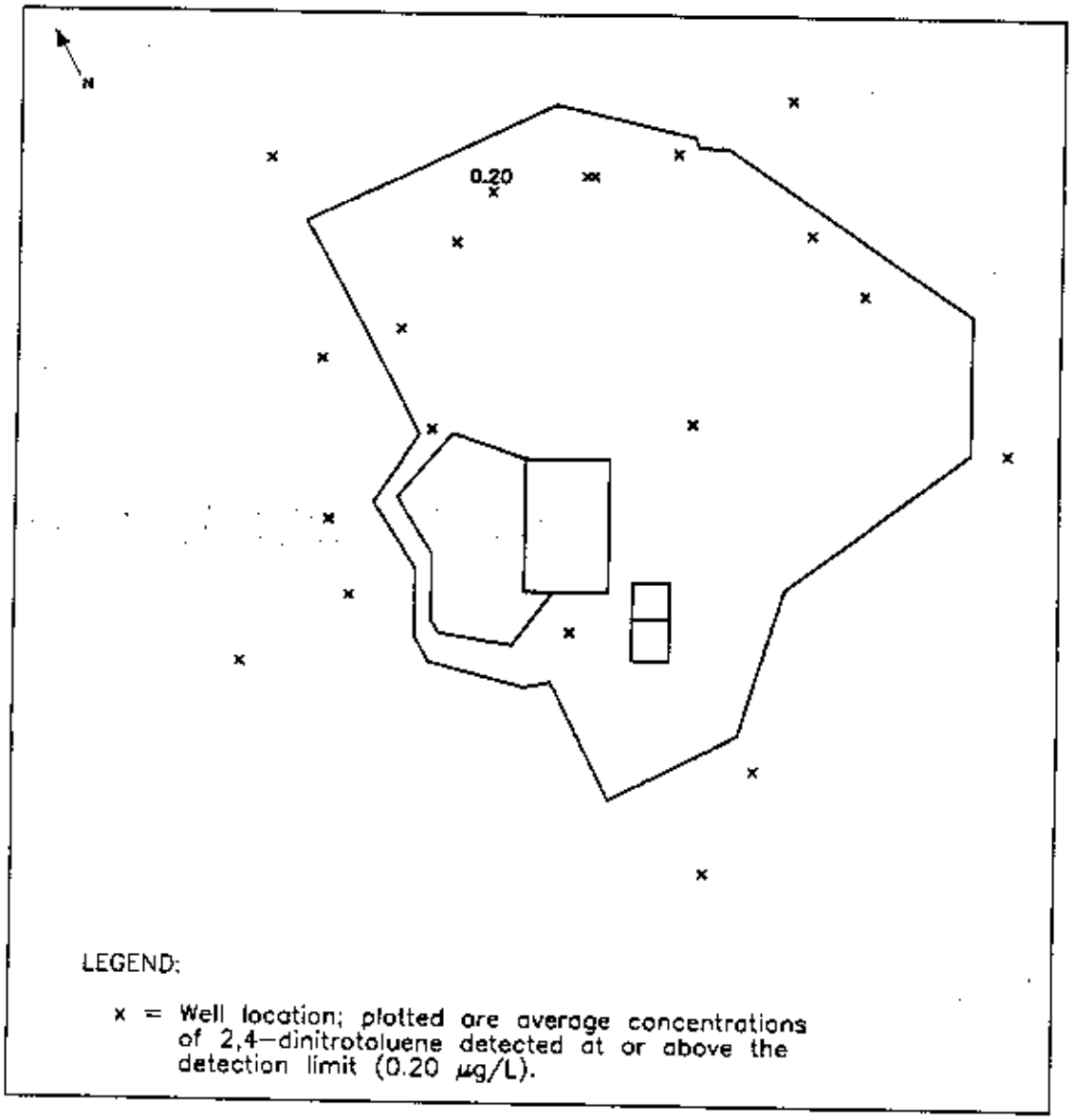


FIGURE B.33 Average Concentrations (µg/L) of 2,4-Dinitrotoluene in the Competent Zone, 1987-1990

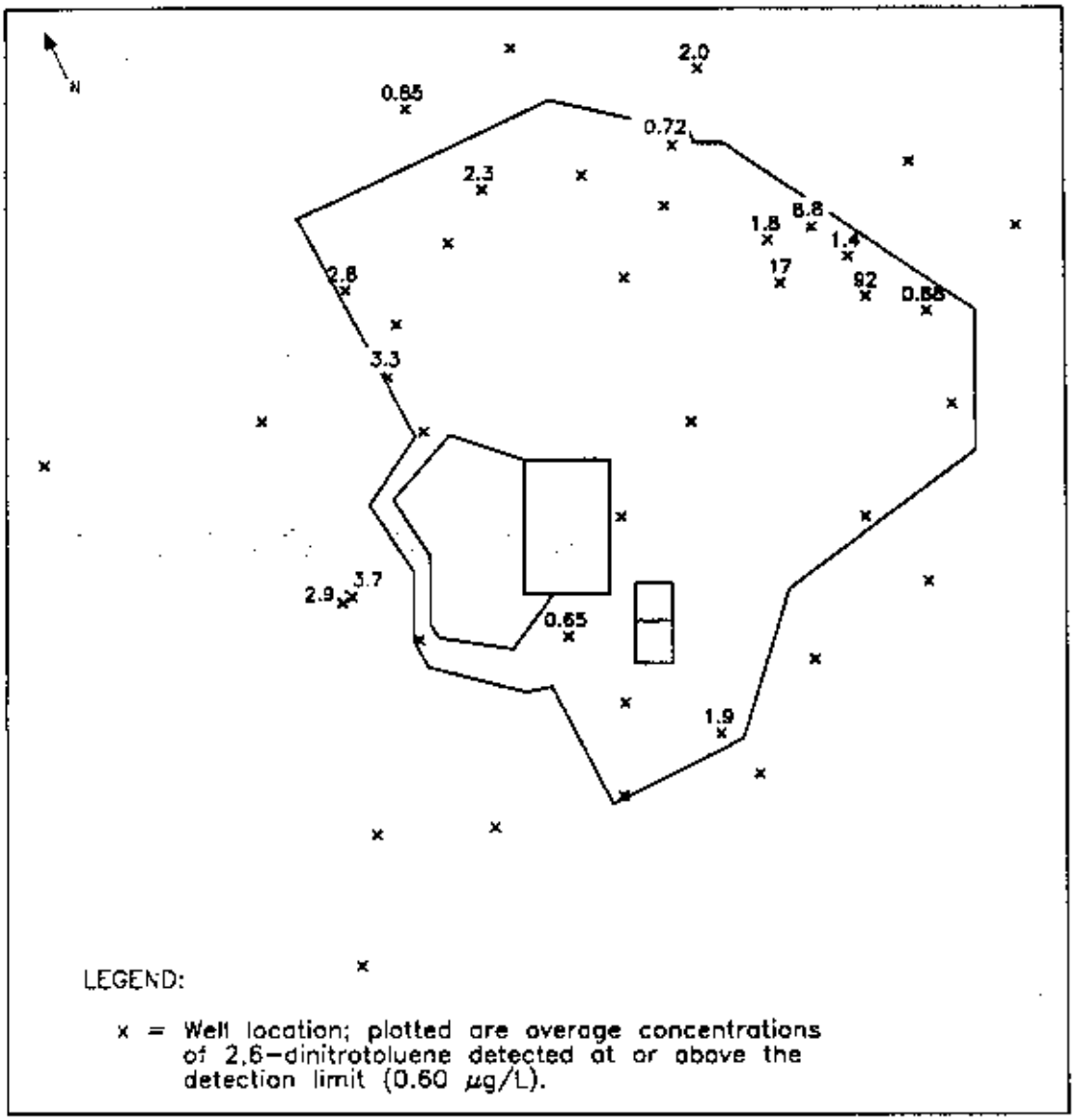


FIGURE B.34 Average Concentrations (µg/L) of 2,6-Dinitrotoluene in the Weathered and Weathered/Competent Zones, 1987-1990

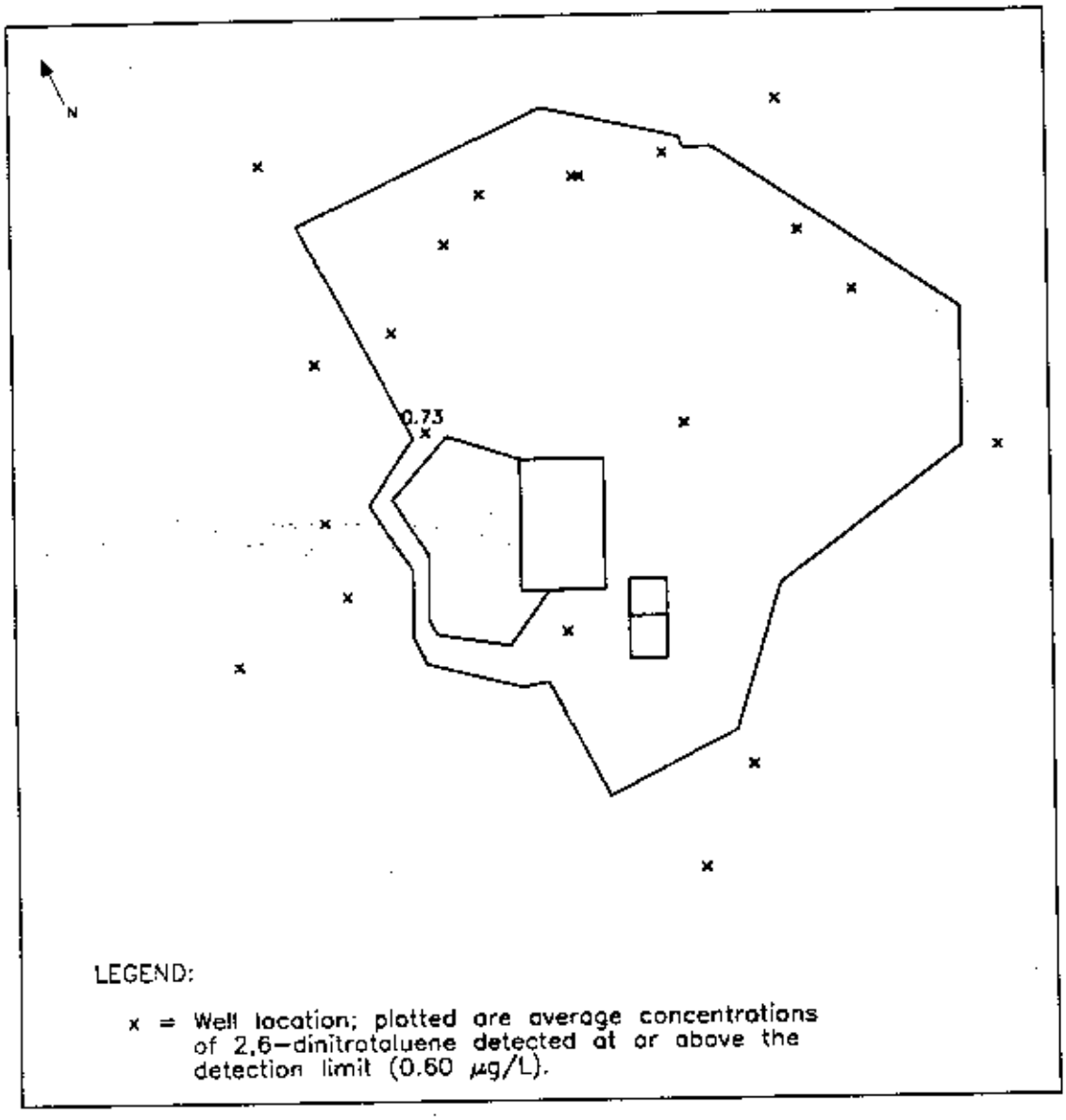


FIGURE B.35 Average Concentrations (µg/L) of 2,6-Dinitrotoluene in the Competent Zone, 1987-1990

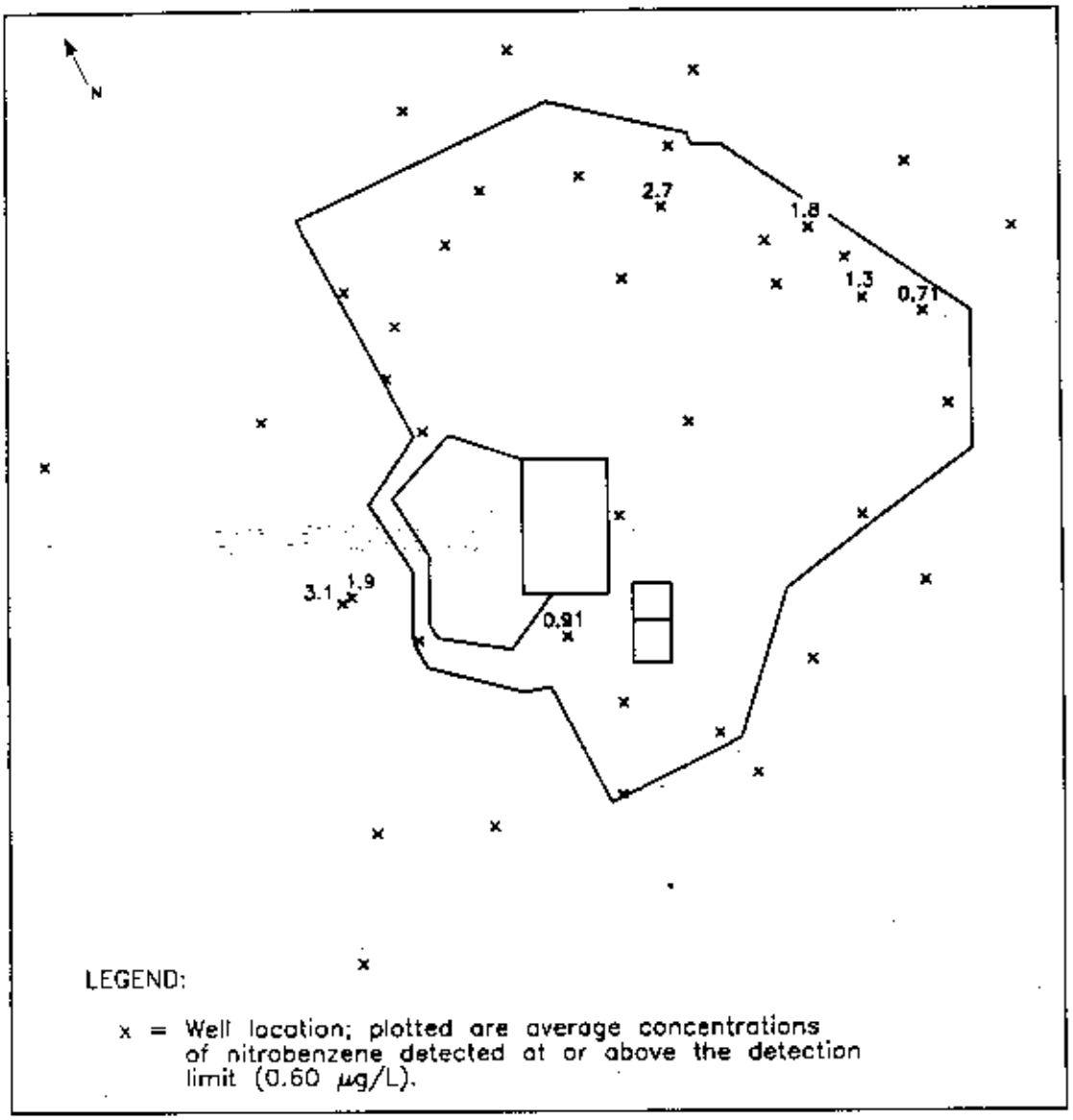


FIGURE B.36 Average Concentrations (μg/L) of Nitrobenzene in the Weathered and Weathered/Competent Zones, 1987-1990

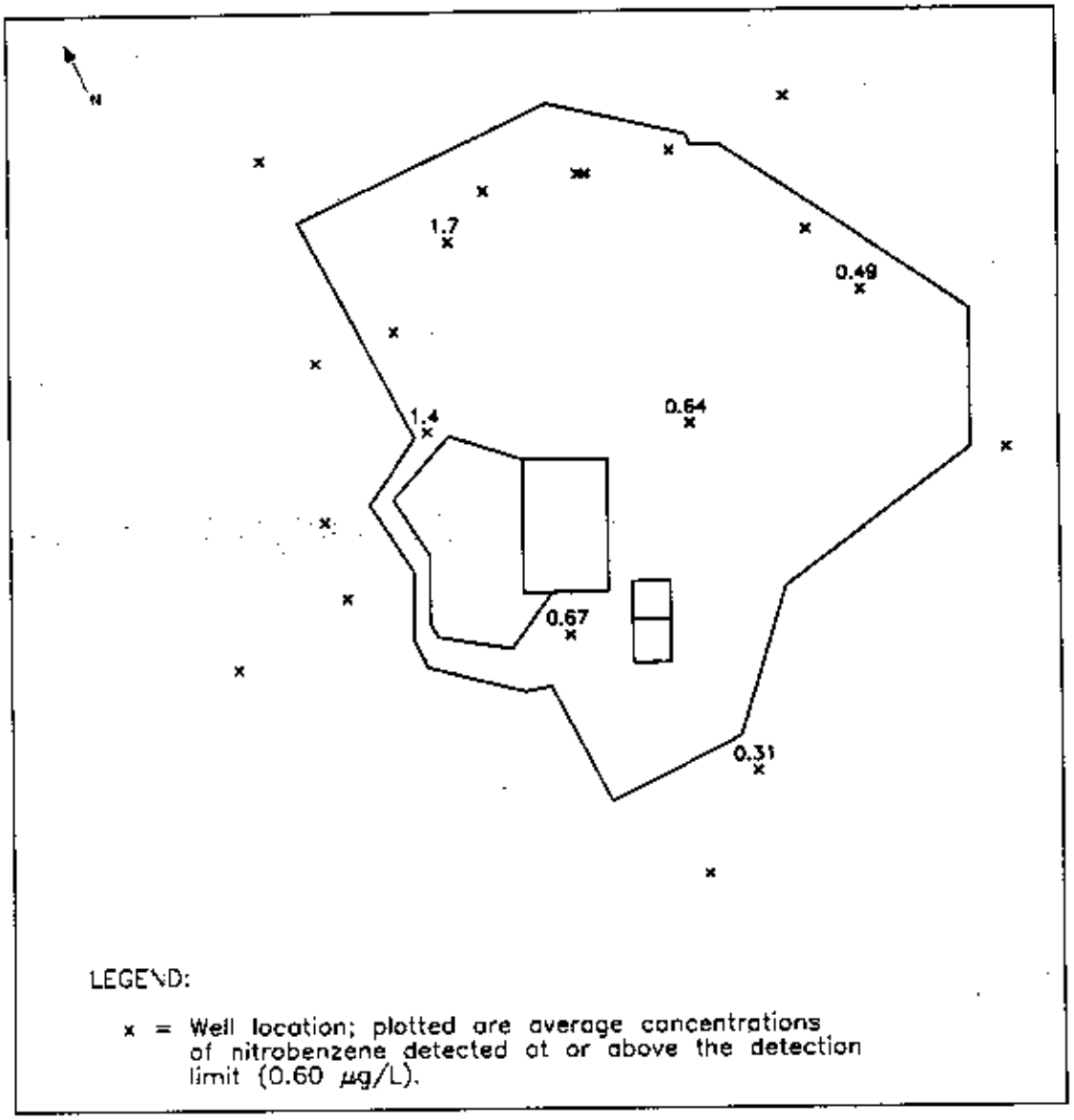


FIGURE B.37 Average Concentrations (µg/L) of Nitrobenzene in the Competent Zone, 1987-1990

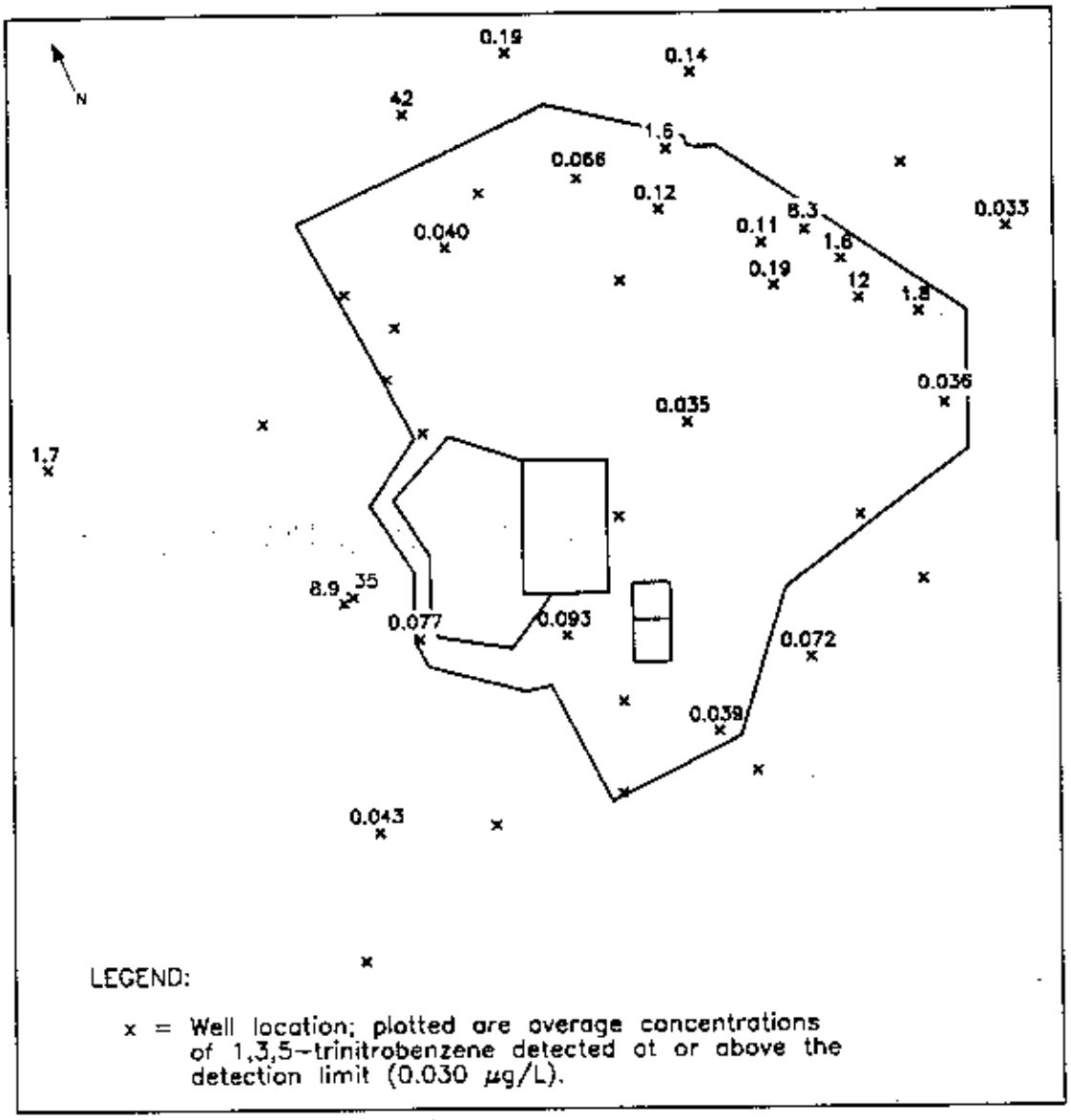


FIGURE B.38 Average Concentrations (µg/L) of 1,3,5-Trinitrobenzene in the Weathered and Weathered/Competent Zones, 1987-1990

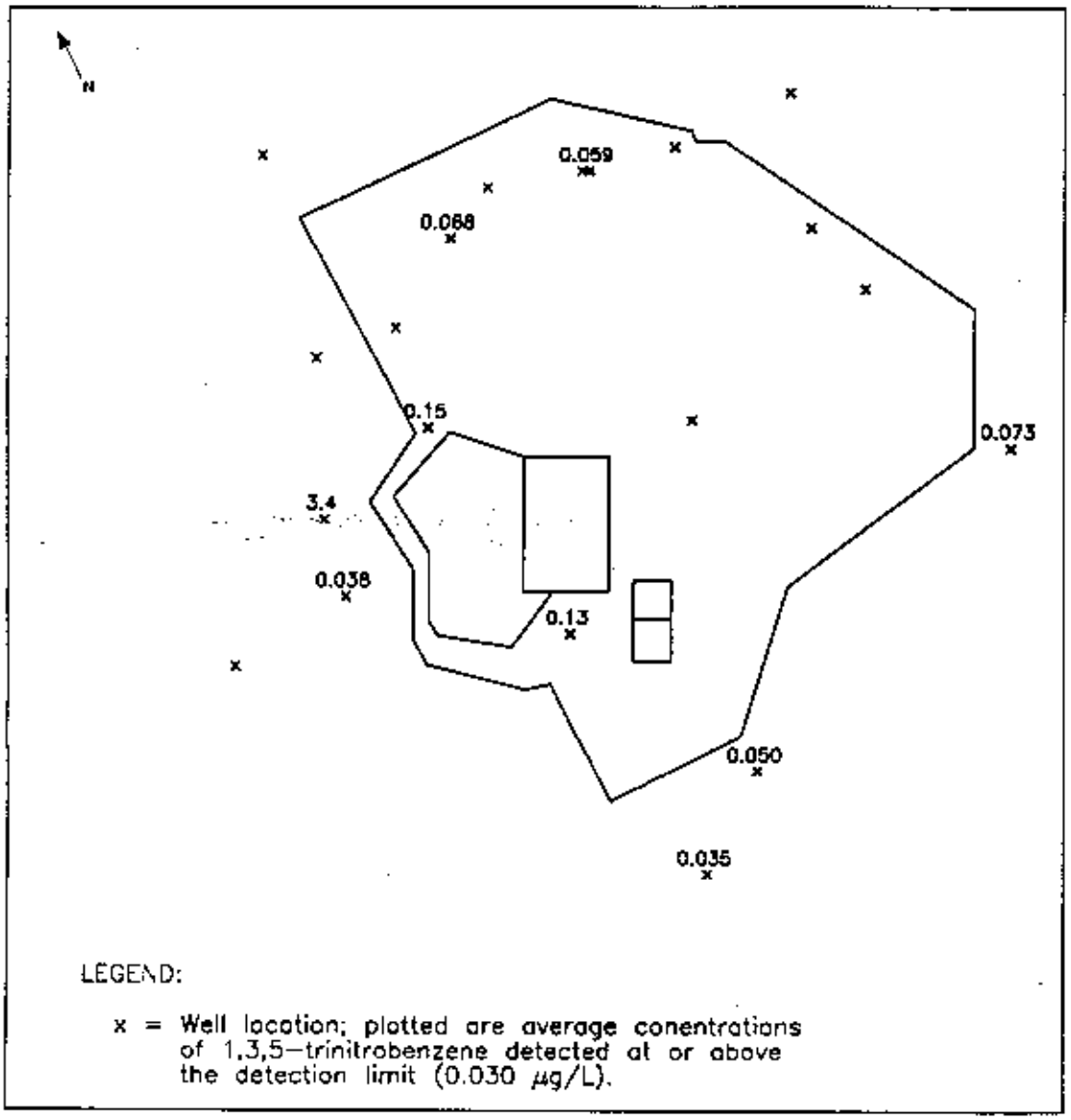


FIGURE B.39 Average Concentrations (μg/L) of 1,3,5-Trinitrobenzene in the Competent Zone, 1987-1990

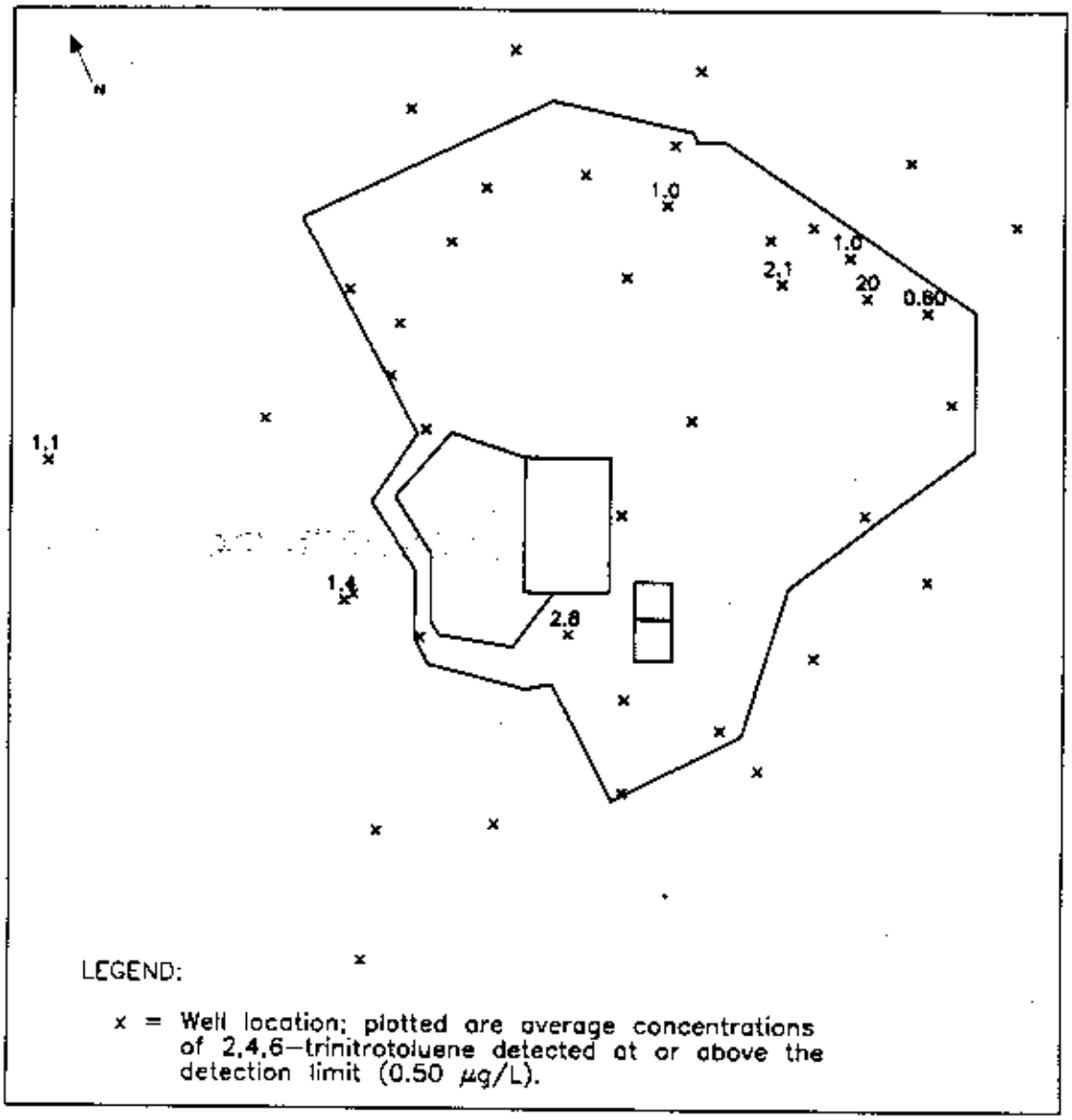


FIGURE B.40 Average Concentrations (μg/L) of 2,4,6-Trinitrotoluene in the Weathered and Weathered/Competent Zones, 1987-1990

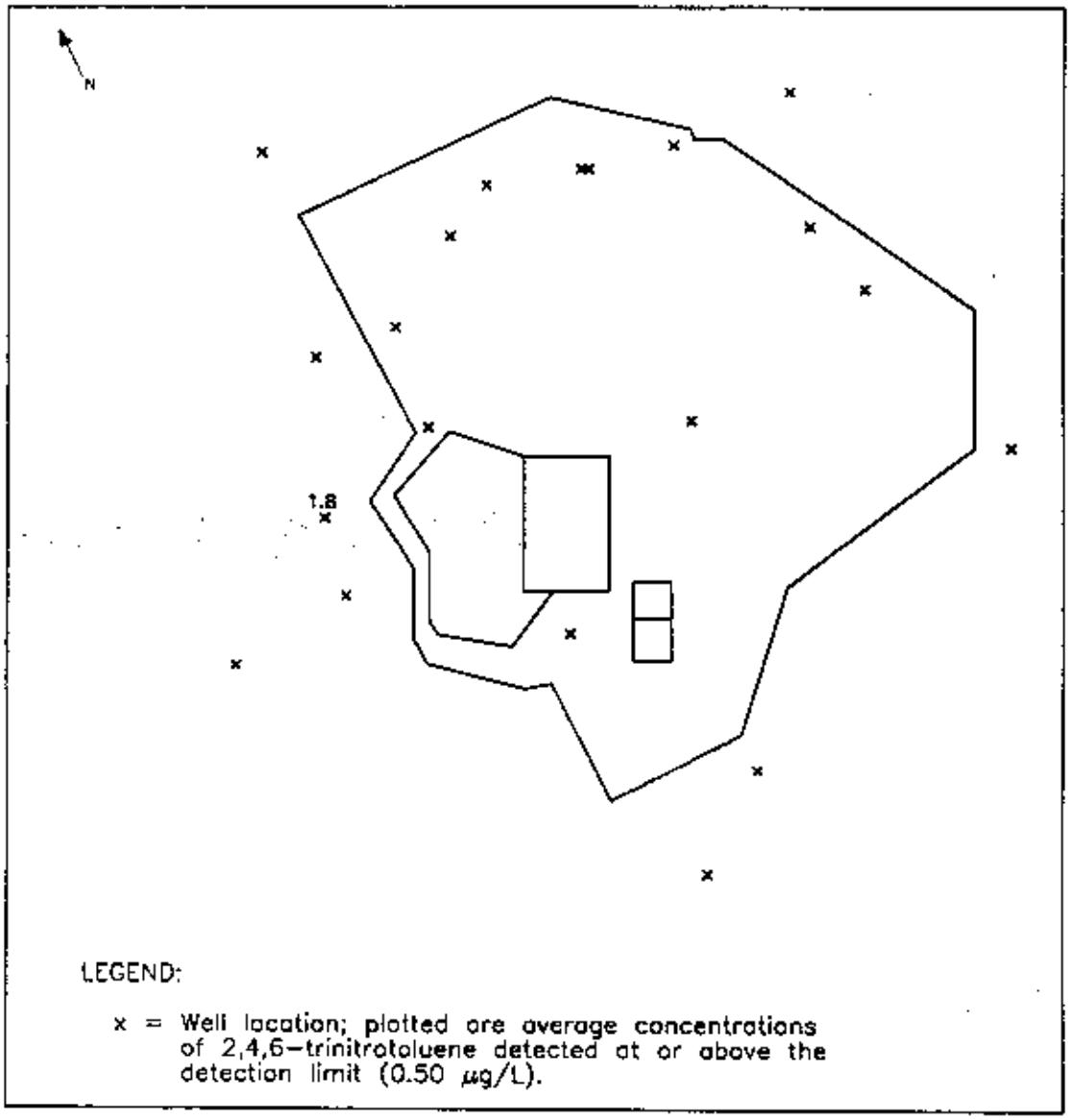


FIGURE B.41 Average Concentrations (μg/L) of 2,4,6-Trinitrotoluene in the Competent Zone, 1987-1990

TABLE B.1 Monitoring Well Construction Data[a]

Well Number	Completion Interval	Ground Elevation (ft MSL)	Total Depth (ft)	Top of Screen Elevation (ft MSL)	Bottom of Screen Elevation (ft MSL)	Top of Filter Pack Elevation (ft MSL)	Top of Bedrock Elevation (ft MSL)	Length of Screen (ft)
MW-2001	Weathered/Competent	611.75	64.00	563.75	553.75	580.15	585.25	10.00
MW-2002	Weathered/Competent	623.81	64.00	575.81	565.81	591.22	599.30	10.00
MW-2003	Weathered	637.00	64.00	589.00	579.00	595.50	598.20	10.00
MW-2004	Weathered	642.89	77.00	577.39	567.39	588.59	591.90	10.00
MW-2005	Weathered/Competent	635.60	81.00	570.10	560.10	585.60	590.80	10.00
MW-2006	Weathered/Competent	634.09	71.00	578.59	568.59	607.09	611.50	10.00
MW-2007	Weathered/Competent	651.89	99.00	568.89	558.89	589.59	592.90	10.00
MW-2008	Weathered/Competent	622.77	63.00	577.27	567.27	588.77	591.30	10.00
MW-2009	Weathered	636.35	58.60	588.75	578.75	609.15	615.90	10.00
MW-2010	Weathered	642.97	64.00	594.97	584.97	605.77	613.50	10.00
MW-2011	Weathered/Competent	653.25	79.00	590.45	580.45	616.65	621.25	10.00
MW-2012	Weathered	634.90	69.50	586.90	576.90	605.60	609.40	10.00
MW-2013	Weathered/Competent	645.38	75.00	587.38	577.38	614.08	617.88	10.00
MW-2014	Weathered	647.42	64.00	599.42	589.42	610.42	614.42	10.00
MW-2015	Weathered/Competent	654.40	86.00	586.90	576.90	607.10	611.90	10.00
MW-2017	Weathered	657.71	69.00	604.71	594.71	627.71	634.21	10.01
MW-2018	Weathered	661.49	69.00	608.49	598.49	624.90	628.99	10.00
MW-2019	Competent	645.66	116.30	539.66	529.66	542.66	631.90	10.00
MW-2020	Weathered/Competent	655.36	119.60	618.96	535.86	618.96	637.46	83.10
MW-2021	Competent	624.57	111.00	523.57	513.87	528.32	592.11	9.70
MW-2022	Competent	636.08	128.00	520.18	510.18	524.08	585.60	10.00
MW-2023	Competent	635.68	91.82	565.18	545.18	567.18	581.70	20.00
MW-2024	Competent	635.00	149.61	-[b]	486.07	-[b]	581.70	-[b]
MW-2025	Competent	622.35	108.60	525.45	515.45	528.35	588.77	10.00
MW-2026	Competent	631.04	118.00	523.14	513.14	525.54	607.09	10.00
MW-2027	Competent	646.63	120.50	536.33	526.33	541.13	617.88	10.00
MW-2028	Competent	657.85	131.20	536.85	526.85	541.85	611.90	10.00
MW-2029	Competent	643.44	101.30	522.14	542.14	554.44	592[c]	10.00
MW-3001	Weathered	664.25	90.00	609.25	589.25	611.58	610.50	20.00
MW-3002	Competent	664.72	150.00	527.22	517.32	530.72	610.00	9.90
MW-3003	Weathered/Competent	644.31	89.50	565.11	555.11	568.61	616.34	10.00
MW-3006	Competent	645.90	135.50	521.90	510.90	525.90	616.34	11.00
MW-3008	Weathered	645.09	101.00	606.09	544.09	606.09	617.09	62.00
MW-3009	Weathered	644.33	99.40	599.33	544.93	599.33	609.30	54.40

TABLE B.1 (Cont.)

Well Number	Completion Interval	Ground Elevation (ft MSL)	Total Depth (ft)	Top of Screen Elevation (ft MSL)	Bottom of Screen Elevation (ft MSL)	Top of Filter Pack Elevation (ft MSL)	Top of Bedrock Elevation (ft MSL)	Length of Screen (ft)
MW-3010	Weathered/Competent	665.02	90.70	612.52	574.32	612.52	627.02	38.20
MW-3019	Weathered/Competent	662.00	84.30	588.30	578.30	592.00	625.10	10.00
MW-4001	Weathered	621.74	41.80	592.74	582.74	596.74	606.99	10.00
MW-4002	Weathered/Competent	632.72	84.70	591.72	548.02	591.72	606.50	43.70
MW-4003	Weathered/Competent	669.86	106.20	618.86	563.66	618.86	646.86	55.20
MW-4004	Weathered/Competent	651.66	75.00	586.86	576.86	588.66	628.16	10.00
MW-4005	Weathered/Competent	656.35	78.80	587.95	577.95	592.45	626.35	10.00
MW-4006	Weathered	621.46	28.50	599.95	594.95	600.95	602.67	5.00
MW-4007	Competent	621.46	90.00	541.86	531.86	544.46	610[c]	10.00
MW-4008	Competent	635.46	83.00	562.66	552.66	565.46	603.10	10.00
MW-4009	Competent	624.22	76.80	557.62	547.62	560.22	605.22	10.00
MW-4010	Weathered/Competent	629.08	77.10	562.18	552.18	564.48	613.38	10.00
MW-4011	Competent	626.90	78.70	561.10	551.10	562.80	596.42	10.00
MW-4012	Competent	615.42	80.20	550.12	540.12	552.72	582.42	10.00
MW-4013	Weathered/Competent	606.85	60.00	567.55	547.55	569.35	571.75	20.00
MW-4014	Weathered/Competent	607.50	66.40	562.70	542.70	564.50	565.50	20.00
MW-4015	Weathered/Competent	617.88	63.20	574.83	554.83	577.88	604.88	20.00
MW-4016	Competent	643.13	85.30	569.43	559.43	571.53	613.13	10.00
MW-4017	Weathered/Competent	649.47	85.10	585.17	565.17	587.17	609.47	20.00
MW-4018	Weathered/Competent	648.03	82.50	583.53	573.53	586.53	604.00	10.00
MW-4019	Weathered/Competent	645.36	58.83	645.36	645.36	605.53	625.16	0.00
MW-4020	Competent	657.86	81.30	591.06	581.06	592.86	621.86	10.00
MW-4021	Weathered/Competent	650.06	71.00	599.26	579.26	601.06	623.56	20.00
MW-4022	Competent	666.31	90.90	597.51	577.51	599.31	631.31	20.00
MW-4023	Weathered/Competent	646.48	70.00	613.48	593.48	615.98	617.60	20.00

[a] Based on information provided in the remedial investigation (U.S. Department of Energy 1992); MSL = above mean sea level.

[b] Data to be obtained.

[c] Approximate elevation.

TABLE B.2 Concentrations of Metals in Groundwater Samples, 1987-1990

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Antimony (DL = 60 µg/L)[a]						
MW-2019	Competent	1	0	-	-	-
MW-2021	Competent	1	0	-	-	-
MW-2023	Competent	1	0	-	-	-
MW-2022	Competent	1	0	-	-	-
MW-2024	Competent	1	0	-	-	-
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	0	-	-	-
MW-2027	Competent	1	0	-	-	-
MW-2028	Competent	1	0	-	-	-
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	0	-	-	-
MW-3006	Competent	1	0	-	-	-
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	0	-	-	-
MW-4009	Competent	1	0	-	-	-
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	0	-	-	-
MW-4020	Competent	1	0	-	-	-
MW-4022	Competent	1	0	-	-	-
MW-2003	Weathered	3	1	160	160	73
MW-2009	Weathered	2	0	-	-	-
MW-2010	Weathered	2	0	-	-	-
MW-2012	Weathered	1	0	-	-	-
MW-2014	Weathered	2	0	-	-	-
MW-2016	Weathered	1	1	77	77	77
MW-2017	Weathered	3	1	92	92	51
MW-2018	Weathered	3	1	62	62	41
MW-3001	Weathered	1	0	-	-	-
MW-3008	Weathered	3	1	250	250	100
MW-3009	Weathered	3	1	83	83	48
MW-4001	Weathered	3	1	60	60	40
MW-4006	Weathered	1	0	-	-	-
MW-2001	Weathered/Competent	2	0	-	-	-
MW-2002	Weathered/Competent	2	1	110	110	72
MW-2004	Weathered/Competent	2	0	-	-	-
MW-2005	Weathered/Competent	2	1	110	110	72
MW-2006	Weathered/Competent	2	0	-	-	-
MW-2007	Weathered/Competent	2	0	-	-	-
MW-2008	Weathered/Competent	3	1	61	61	40
MW-2011	Weathered/Competent	2	0	-	-	-
MW-2013	Weathered/Competent	2	0	-	-	-
MW-2015	Weathered/Competent	3	1	97	97	52
MW-2020	Weathered/Competent	3	1	76	76	45
MW-3003	Weathered/Competent	1	0	-	-	-
MW-3010	Weathered/Competent	3	1	62	62	41
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	2	0	-	-	-
MW-4003	Weathered/Competent	1	1	60	60	60
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	0	-	-	-
MW-4010	Weathered/Competent	1	0	-	-	-
MW-4013	Weathered/Competent	1	0	-	-	-

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Antimony (Cont.)						
MW-4014	Weathered/Competent	1	0	-	-	-
MW-4015	Weathered/Competent	1	0	-	-	-
MW-4017	Weathered/Competent	1	0	-	-	-
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	3	1	65	65	42
MW-4021	Weathered/Competent	1	0	-	-	-
MW-4023	Weathered/Competent	1	0	-	-	-
Barium (DL = 200 µg/L)						
MW-2019	Competent	1	0	-	-	-
MW-2021	Competent	1	0	-	-	-
MW-2022	Competent	1	0	-	-	-
MW-2023	Competent	1	0	-	-	-
MW-2024	Competent	1	0	-	-	-
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	0	-	-	-
MW-2027	Competent	1	0	-	-	-
MW-2028	Competent	1	1	250	250	250
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	0	-	-	-
MW-3006	Competent	1	0	-	-	-
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	0	-	-	-
MW-4009	Competent	1	0	-	-	-
MW-4011	Competent	1	1	84	84	84
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	1	250	250	250
MW-4020	Competent	1	0	-	-	-
MW-4022	Competent	1	0	-	-	-
MW-2003	Weathered	3	2	230	340	220
MW-2009	Weathered	3	3	230	300	270
MW-2010	Weathered	3	3	260	280	270
MW-2012	Weathered	2	0	-	-	-
MW-2014	Weathered	3	3	210	280	240
MW-2016	Weathered	1	0	-	-	-
MW-2017	Weathered	3	0	-	-	-
MW-2018	Weathered	3	3	320	410	380
MW-3001	Weathered	1	1	660	660	660
MW-3008	Weathered	3	3	320	540	440
MW-3009	Weathered	3	3	220	1,500	1,000
MW-4001	Weathered	3	0	-	-	-
MW-4006	Weathered	2	1	200	200	150
MW-2001	Weathered/Competent	3	0	-	-	-
MW-2002	Weathered/Competent	2	1	330	330	220
MW-2004	Weathered/Competent	3	0	-	-	-
MW-2005	Weathered/Competent	3	1	270	270	160
MW-2006	Weathered/Competent	3	2	250	280	210
MW-2007	Weathered/Competent	3	0	-	-	-
MW-2008	Weathered/Competent	3	2	220	240	190
MW-2011	Weathered/Competent	3	1	240	240	150
MW-2013	Weathered/Competent	3	2	220	350	230
MW-2015	Weathered/Competent	3	0	-	-	-

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Barium (Cont.)						
MW-2020	Weathered/Competent	3	0	-	-	-
MW-3003	Weathered/Competent	1	1	2,200	2,200	2,200
MW-3010	Weathered/Competent	3	3	220	250	240
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	3	0	-	-	-
MW-4003	Weathered/Competent	1	0	-	-	-
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	1	230	230	230
MW-4010	Weathered/Competent	1	0	-	-	-
MW-4013	Weathered/Competent	1	1	340	340	340
MW-4014	Weathered/Competent	1	1	290	290	290
MW-4015	Weathered/Competent	1	0	-	-	-
MW-4017	Weathered/Competent	1	1	250	250	250
MW-4018	Weathered/Competent	1	1	340	340	340
MW-4019	Weathered/Competent	3	0	-	-	-
MW-4021	Weathered/Competent	1	0	-	-	-
MW-4023	Weathered/Competent	1	1	300	300	300
Chromium (DL = 10.0 µg/L)						
MW-2019	Competent	1	0	-	-	-
MW-2021	Competent	1	0	-	-	-
MW-2022	Competent	1	0	-	-	-
MW-2023	Competent	1	1	11	11	11
MW-2024	Competent	1	0	-	-	-
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	0	-	-	-
MW-2027	Competent	1	0	-	-	-
MW-2028	Competent	1	0	-	-	-
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	0	-	-	-
MW-3006	Competent	1	1	37	37	37
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	0	-	-	-
MW-4009	Competent	1	1	10	10	10
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	0	-	-	-
MW-4020	Competent	1	1	24	24	24
MW-4022	Competent	1	1	17	17	17
MW-2003	Weathered	3	2	25	91	40
MW-2009	Weathered	3	1	15	15	8.3
MW-2010	Weathered	3	2	28	72	35
MW-2012	Weathered	2	1	39	39	22
MW-2014	Weathered	3	1	40	40	17
MW-2016	Weathered	1	1	31	31	31
MW-2017	Weathered	3	1	37	37	16
MW-2018	Weathered	3	1	35	35	15
MW-3001	Weathered	1	0	-	-	-
MW-3008	Weathered	3	2	52	140	65
MW-3009	Weathered	3	1	45	45	18
MW-4001	Weathered	3	1	40	40	17
MW-4006	Weathered	2	1	26	26	16

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Chromium (Cont.)						
MW-2001	Weathered/Competent	3	1	34	34	15
MW-2002	Weathered/Competent	2	1	61	61	33
MW-2004	Weathered/Competent	3	2	42	130	59
MW-2005	Weathered/Competent	2	1	75	75	40
MW-2006	Weathered/Competent	3	2	17	36	19
MW-2007	Weathered/Competent	3	1	22	22	11
MW-2008	Weathered/Competent	3	1	40	40	17
MW-2011	Weathered/Competent	3	1	15	15	8.3
MW-2013	Weathered/Competent	3	1	22	22	11
MW-2015	Weathered/Competent	3	1	46	46	19
MW-2020	Weathered/Competent	3	2	47	120	58
MW-3003	Weathered/Competent	1	0	-	-	-
MW-3010	Weathered/Competent	3	1	25	25	12
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	3	1	29	29	13
MW-4003	Weathered/Competent	1	1	34	34	34
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	0	-	-	-
MW-4010	Weathered/Competent	1	0	-	-	-
MW-4013	Weathered/Competent	1	0	-	-	-
MW-4014	Weathered/Competent	1	0	-	-	-
MW-4015	Weathered/Competent	1	0	-	-	-
MW-4017	Weathered/Competent	1	0	-	-	-
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	3	2	18	27	17
MW-4021	Weathered/Competent	1	1	14	14	14
MW-4023	Weathered/Competent	1	1	29	29	29
Lead (DL = 5.0 µg/L)						
MW-2019	Competent	1	1	12	12	12
MW-2021	Competent	1	0	-	-	-
MW-2022	Competent	1	1	20	20	20
MW-2023	Competent	1	1	29	29	29
MW-2024	Competent	1	1	17	17	17
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	1	8.0	8.0	8.0
MW-2027	Competent	1	1	17	17	17
MW-2028	Competent	1	0	-	-	-
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	1	40	40	40
MW-3006	Competent	1	1	35	35	35
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	1	18	18	18
MW-4009	Competent	1	0	-	-	-
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	0	-	-	-
MW-4020	Competent	1	0	-	-	-
MW-4022	Competent	1	1	12	12	12
MW-2003	Weathered	3	0	-	-	-
MW-2009	Weathered	2	0	-	-	-
MW-2010	Weathered	2	0	-	-	-
MW-2012	Weathered	1	0	-	-	-

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Lead (Cont.)						
MW-2014	Weathered	2	0	-	-	-
MW-2016	Weathered	1	0	-	-	-
MW-2017	Weathered	2	0	-	-	-
MW-2018	Weathered	4	3	5.6	8.6	6.3
MW-3001	Weathered	1	0	-	-	-
MW-3008	Weathered	2	0	-	-	-
MW-3009	Weathered	2	0	-	-	-
MW-4001	Weathered	3	0	-	-	-
MW-4006	Weathered	2	0	-	-	-
MW-2001	Weathered/Competent	3	0	-	-	-
MW-2002	Weathered/Competent	2	0	-	-	-
MW-2004	Weathered/Competent	2	0	-	-	-
MW-2005	Weathered/Competent	2	0	-	-	-
MW-2006	Weathered/Competent	1	0	-	-	-
MW-2007	Weathered/Competent	2	0	-	-	-
MW-2008	Weathered/Competent	2	0	-	-	-
MW-2011	Weathered/Competent	2	0	-	-	-
MW-2013	Weathered/Competent	2	0	-	-	-
MW-2015	Weathered/Competent	2	0	-	-	-
MW-2020	Weathered/Competent	2	0	-	-	-
MW-3003	Weathered/Competent	1	1	7.7	7.7	7.7
MW-3010	Weathered/Competent	3	0	-	-	-
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	2	0	-	-	-
MW-4003	Weathered/Competent	0	0	-	-	-
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	0	-	-	-
MW-4010	Weathered/Competent	1	0	-	-	-
MW-4013	Weathered/Competent	1	1	24	24	24
MW-4014	Weathered/Competent	1	0	-	-	-
MW-4015	Weathered/Competent	1	1	8.0	8.0	8.0
MW-4017	Weathered/Competent	1	1	5.8	5.8	5.8
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	3	0	-	-	-
MW-4021	Weathered/Competent	1	0	-	-	-
MW-4023	Weathered/Competent	1	1	14	14	14
Lithium (DL = 50.0 µg/L)						
MW-2019	Competent	1	1	8.4	8.4	8.4
MW-2021	Competent	1	0	-	-	-
MW-2022	Competent	1	1	2.2	2.2	2.2
MW-2023	Competent	1	1	2.0	2.0	2.0
MW-2024	Competent	1	1	3.2	3.2	3.2
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	0	-	-	-
MW-2027	Competent	1	0	-	-	-
MW-2028	Competent	1	1	9.4	9.4	9.4
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	0	-	-	-
MW-3006	Competent	1	1	36	36	36
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	0	-	-	-
MW-4009	Competent	1	0	-	-	-

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Lithium (Cont.)						
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	0	-	-	-
MW-4020	Competent	1	0	-	-	-
MW-4022	Competent	1	1	15	15	15
MW-2003	Weathered	3	2	480	1,000	500
MW-2009	Weathered	2	0	-	-	-
MW-2010	Weathered	2	0	-	-	-
MW-2012	Weathered	1	0	-	-	-
MW-2014	Weathered	2	0	-	-	-
MW-2016	Weathered	0	0	-	-	-
MW-2017	Weathered	2	0	-	-	-
MW-2018	Weathered	2	0	-	-	-
MW-3001	Weathered	1	1	64	64	64
MW-3008	Weathered	3	3	100	170	140
MW-3009	Weathered	2	0	-	-	-
MW-4001	Weathered	2	0	-	-	-
MW-4006	Weathered	1	0	-	-	-
MW-2001	Weathered/Competent	2	0	-	-	-
MW-2002	Weathered/Competent	2	2	250	810	530
MW-2004	Weathered/Competent	2	0	-	-	-
MW-2005	Weathered/Competent	2	1	250	250	140
MW-2006	Weathered/Competent	1	0	-	-	-
MW-2007	Weathered/Competent	2	0	-	-	-
MW-2008	Weathered/Competent	2	0	-	-	-
MW-2011	Weathered/Competent	2	0	-	-	-
MW-2013	Weathered/Competent	2	0	-	-	-
MW-2015	Weathered/Competent	2	0	-	-	-
MW-2020	Weathered/Competent	2	0	-	-	-
MW-3003	Weathered/Competent	1	1	820	820	820
MW-3010	Weathered/Competent	2	0	-	-	-
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	2	0	-	-	-
MW-4003	Weathered/Competent	0	0	-	-	-
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	0	-	-	-
MW-4010	Weathered/Competent	1	0		-	-
MW-4013	Weathered/Competent	1	1	66	66	66
MW-4014	Weathered/Competent	1	0	-	-	-
MW-4015	Weathered/Competent	1	0	-	-	-
MW-4017	Weathered/Competent	1	0	-	-	-
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	2	0	-	-	-
MW-4021	Weathered/Competent	1	0	-	-	-
MW-4023	Weathered/Competent	1	0	-	-	-
Manganese (DL = 15.0 µg/L)						
MW-2019	Competent	1	0	-	-	-
MW-2021	Competent	1	1	81	81	81
MW-2022	Competent	1	1	46	46	46
MW-2023	Competent	1	1	19	19	19
MW-2024	Competent	1	1	18	18	18
MW-2025	Competent	0	0	-	-	-

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Manganese (Cont.)						
MW-2026	Competent	1	1	38	38	38
MW-2027	Competent	1	1	190	190	190
MW-2028	Competent	1	1	510	510	510
MW-2029	Competent	1	1	51	51	51
MW-3002	Competent	1	1	24	24	24
MW-3006	Competent	1	0	-	-	-
MW-4007	Competent	1	1	24	24	24
MW-4008	Competent	1	1	41	41	41
MW-4009	Competent	1	1	57	57	57
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	1	0	-	-	-
MW-4016	Competent	1	1	54	54	54
MW-4020	Competent	1	1	170	170	170
MW-4022	Competent	1	1	160	160	160
MW-2003	Weathered	3	2	75	90	58
MW-2009	Weathered	3	0	-	-	-
MW-2010	Weathered	3	3	16	24	21
MW-2012	Weathered	2	0	-	-	-
MW-2014	Weathered	3	2	35	41	28
MW-2016	Weathered	1	1	19	19	19
MW-2017	Weathered	3	0	-	-	-
MW-2018	Weathered	3	0	-	-	-
MW-3001	Weathered	1	1	97	97	97
MW-3008	Weathered	3	2	26	80	38
MW-3009	Weathered	3	0	-	-	-
MW-4001	Weathered	3	2	16	31	18
MW-4006	Weathered	2	2	93	210	150
MW-2001	Weathered/Competent	3	0	-	-	-
MW-2002	Weathered/Competent	2	1	19	19	13
MW-2004	Weathered/Competent	3	1	30	30	15
MW-2005	Weathered/Competent	2	1	35	35	21
MW-2006	Weathered/Competent	3	3	32	53	43
MW-2007	Weathered/Competent	3	2	43	60	37
MW-2008	Weathered/Competent	3	2	20	110	47
MW-2011	Weathered/Competent	3	0	-	-	-
MW-2013	Weathered/Competent	3	3	27	96	57
MW-2015	Weathered/Competent	3	1	36	36	17
MW-2020	Weathered/Competent	3	2	15	17	13
MW-3003	Weathered/Competent	1	1	230	230	230
MW-3010	Weathered/Competent	3	0	-	-	-
MW-3019	Weathered/Competent	1	1	110	110	110
MW-4002	Weathered/Competent	3	1	18	18	11
MW-4003	Weathered/Competent	1	0	-	-	-
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	1	550	550	550
MW-4010	Weathered/Competent	1	1	42	42	42
MW-4013	Weathered/Competent	1	1	51	51	51
MW-4014	Weathered/Competent	1	1	290	290	290
MW-4015	Weathered/Competent	1	1	32	32	32
MW-4017	Weathered/Competent	1	1	65	65	65
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	3	2	30	290	110
MW-4021	Weathered/Competent	1	1	170	170	170
MW-4023	Weathered/Competent	1	1	61	61	61

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Mercury (DL = 0.20 µg/L)						
MW-2019	Competent	1	0	-	-	-
MW-2021	Competent	1	0	-	-	-
MW-2022	Competent	1	0	-	-	-
MW-2023	Competent	1	0	-	-	-
MW-2024	Competent	1	0	-	-	-
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	0	-	-	-
MW-2027	Competent	1	0	-	-	-
MW-2028	Competent	1	0	-	-	-
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	1	0.35	0.35	0.35
MW-3006	Competent	1	0	-	-	-
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	0	-	-	-
MW-4009	Competent	1	0	-	-	-
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	0	-	-	-
MW-4020	Competent	1	0	-	-	-
MW-4022	Competent	1	1	0.26	0.26	0.26
MW-2003	Weathered	2	0	-	-	-
MW-2009	Weathered	2	0	-	-	-
MW-2010	Weathered	3	1	18	18	5.9
MW-2012	Weathered	1	0	-	-	-
MW-2014	Weathered	2	0	-	-	-
MW-2016	Weathered	0	0	-	-	-
MW-2017	Weathered	2	0	-	-	-
MW-2018	Weathered	2	0	-	-	-
MW-3001	Weathered	1	1	0.46	0.46	0.46
MW-3008	Weathered	2	1	0.21	0.21	0.16
MW-3009	Weathered	2	0	-	-	-
MW-4001	Weathered	2	0	-	-	-
MW-4006	Weathered	1	0	-	-	-
MW-2001	Weathered/Competent	3	1	2.0	2.0	0.73
MW-2002	Weathered/Competent	1	0	-	-	-
MW-2004	Weathered/Competent	3	1	44	44	15
MW-2005	Weathered/Competent	1	0	-	-	-
MW-2006	Weathered/Competent	1	0	-	-	-
MW-2007	Weathered/Competent	2	0	-	-	-
MW-2008	Weathered/Competent	2	0	-	-	-
MW-2011	Weathered/Competent	2	0	-	-	-
MW-2013	Weathered/Competent	2	0	-	-	-
MW-2015	Weathered/Competent	2	0	-	-	-
MW-2020	Weathered/Competent	2	0	-	-	-
MW-3003	Weathered/Competent	1	0	-	-	-
MW-3010	Weathered/Competent	2	0	-	-	-
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	2	0	-	-	-
MW-4003	Weathered/Competent	0	0	-	-	-
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	0	-	-	-
MW-4010	Weathered/Competent	1	0	-	-	-
MW-4013	Weathered/Competent	1	0	-	-	-
MW-4014	Weathered/Competent	1	0	-	-	-

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Mercury (Cont.)						
MW-4015	Weathered/Competent	1	0	-	-	-
MW-4017	Weathered/Competent	1	0	-	-	-
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	2	1	0.25	0.25	0.16
MW-4021	Weathered/Competent	1	1	0.21	0.21	0.21
MW-4023	Weathered/Competent	1	0	-	-	-
Molybdenum (DL = 40 µg/L)						
MW-2019	Competent	1	1	13	13	13
MW-2021	Competent	1	1	46	46	46
MW-2022	Competent	1	1	38	38	38
MW-2023	Competent	1	1	24	24	24
MW-2024	Competent	1	1	35	35	35
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	1	24	24	24
MW-2027	Competent	1	1	14	14	14
MW-2028	Competent	1	1	40	40	40
MW-2029	Competent	1	1	19	19	19
MW-3002	Competent	1	1	24	24	24
MW-3006	Competent	1	1	1,100	1,100	1,100
MW-4007	Competent	1	1	40	40	40
MW-4008	Competent	1	1	12	12	12
MW-4009	Competent	1	1	13	13	13
MW-4011	Competent	1	1	16	16	16
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	1	18	18	18
MW-4020	Competent	1	1	22	22	22
MW-4022	Competent	1	1	50	50	50
MW-2003	Weathered	2	1	58	58	39
MW-2009	Weathered	2	0	-	-	-
MW-2010	Weathered	2	0	-	-	-
MW-2012	Weathered	1	0	-	-	-
MW-2014	Weathered	2	1	23	23	22
MW-2016	Weathered	0	0	-	-	-
MW-2017	Weathered	2	1	52	52	36
MW-2018	Weathered	2	1	15	15	18
MW-3001	Weathered	1	1	58	58	58
MW-3008	Weathered	2	1	70	70	45
MW-3009	Weathered	2	0	-	-	-
MW-4001	Weathered	2	0	-	-	-
MW-4006	Weathered	1	0	-	-	-
MW-2001	Weathered/Competent	2	0	-	-	-
MW-2002	Weathered/Competent	1	1	37	37	37
MW-2004	Weathered/Competent	2	2	16	17	17
MW-2005	Weathered/Competent	2	2	15	18	17
MW-2006	Weathered/Competent	2	2	15	18	17
MW-2007	Weathered/Competent	2	1	18	18	19
MW-2008	Weathered/Competent	2	1	13	13	17
MW-2011	Weathered/Competent	2	0	-	-	-
MW-2013	Weathered/Competent	2	0	-	-	-
MW-2015	Weathered/Competent	2	2	23	24	23
MW-2020	Weathered/Competent	2	2	15	21	18
MW-3003	Weathered/Competent	1	1	82	82	82

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Molybdenum (Cont.)						
MW-3010	Weathered/Competent	2	1	20	20	20
MW-3019	Weathered/Competent	1	1	16	16	16
MW-4002	Weathered/Competent	2	1	14	14	17
MW-4003	Weathered/Competent	0	0	-	-	-
MW-4004	Weathered/Competent	1	1	18	18	18
MW-4005	Weathered/Competent	1	1	24	24	24
MW-4010	Weathered/Competent	1	1	16	16	16
MW-4013	Weathered/Competent	1	1	29	29	29
MW-4014	Weathered/Competent	1	1	21	21	21
MW-4015	Weathered/Competent	1	1	16	16	16
MW-4017	Weathered/Competent	1	1	21	21	21
MW-4018	Weathered/Competent	1	1	22	22	22
MW-4019	Weathered/Competent	2	1	16	16	18
MW-4021	Weathered/Competent	1	1	25	25	25
MW-4023	Weathered/Competent	1	1	25	25	25
Nickel (DL = 40 µg/L)						
MW-2019	Competent	1	0	-	-	-
MW-2021	Competent	1	0	-	-	-
MW-2022	Competent	1	0	-	-	-
MW-2023	Competent	1	0	-	-	-
MW-2024	Competent	1	0	-	-	-
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	0	-	-	-
MW-2027	Competent	1	0	-	-	-
MW-2028	Competent	1	0	-	-	-
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	0	-	-	-
MW-3006	Competent	1	0	-	-	-
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	0	-	-	-
MW-4009	Competent	1	0	-	-	-
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	0	-	-	-
MW-4020	Competent	1	0	-	-	-
MW-4022	Competent	1	0	-	-	-
MW-2003	Weathered	2	0	-	-	-
MW-2009	Weathered	2	0	-	-	-
MW-2010	Weathered	3	3	43	170	110
MW-2012	Weathered	1	0	-	-	-
MW-2014	Weathered	3	0	-	-	-
MW-2016	Weathered	1	0	-	-	-
MW-2017	Weathered	3	1	60	60	33
MW-2018	Weathered	3	0	-	-	-
MW-3001	Weathered	1	0	-	-	-
MW-3008	Weathered	3	1	110	110	49
MW-3009	Weathered	3	1	48	48	29
MW-4001	Weathered	3	0	-	-	-
MW-4006	Weathered	2	0	-	-	
MW-2001	Weathered/Competent	3	0	-	-	-
MW-2002	Weathered/Competent	2	0	-	-	-
MW-2004	Weathered/Competent	2	1	58	58	39

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Nickel (Cont.)						
MW-2005	Weathered/Competent	2	1	52	52	36
MW-2006	Weathered/Competent	3	2	43	45	36
MW-2007	Weathered/Competent	3	1	54	54	31
MW-2008	Weathered/Competent	3	1	88	88	43
MW-2011	Weathered/Competent	2	0	-	-	-
MW-2013	Weathered/Competent	2	0	-	-	-
MW-2015	Weathered/Competent	3	1	50	50	30
MW-2020	Weathered/Competent	3	1	40	40	27
MW-3008	Weathered/Competent	1	1	46	46	46
MW-3010	Weathered/Competent	3	0	-	-	-
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	3	0	-	-	-
MW-4003	Weathered/Competent	1	0	-	-	-
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	0	-	-	-
MW-4010	Weathered/Competent	1	0	-	-	-
MW-4013	Weathered/Competent	1	0	-	-	-
MW-4014	Weathered/Competent	1	1	83	83	83
MW-4015	Weathered/Competent	1	0	-	-	-
MW-4017	Weathered/Competent	1	0	-	-	-
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	3	0	-	-	-
MW-4021	Weathered/Competent	1	0	-	-	-
MW-4023	Weathered/Competent	1	1	45	45	45
Thallium (DL = 10.0 µg/L)						
MW-2019	Competent	1	1	51	51	51
MW-2021	Competent	1	0	-	-	-
MW-2022	Competent	1	0	-	-	-
MW-2023	Competent	1	1	25	25	25
MW-2024	Competent	1	0	17	17	17
MW-2025	Competent	0	0	-	-	-
MW-2026	Competent	1	0	-	-	-
MW-2027	Competent	1	0	-	-	-
MW-2028	Competent	1	0	-	-	-
MW-2029	Competent	1	0	-	-	-
MW-3002	Competent	1	1	59	59	59
MW-3006	Competent	1	1	79	79	79
MW-4007	Competent	1	0	-	-	-
MW-4008	Competent	1	0	-	-	-
MW-4009	Competent	1	0	-	-	-
MW-4011	Competent	1	0	-	-	-
MW-4012	Competent	0	0	-	-	-
MW-4016	Competent	1	0	-	-	-
MW-4020	Competent	1	0	-	-	-
MW-4022	Competent	1	1	93	93	93
MW-2003	Weathered	2	0	-	-	-
MW-2009	Weathered	2	0	-	-	-
MW-2010	Weathered	2	0	-	-	-
MW-2012	Weathered	1	0	-	-	-
MW-2014	Weathered	2	0	-	-	-
MW-2016	Weathered	0	0	-	-	-
MW-2017	Weathered	2	0	-	-	-

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Thallium (Cont.)						
MW-2018	Weathered	2	0	-	-	-
MW-3001	Weathered	1	0	-	-	-
MW-3008	Weathered	2	0	-	-	-
MW-3009	Weathered	2	0	-	-	-
MW-4001	Weathered	2	0	-	-	-
MW-4006	Weathered	1	0	-	-	-
MW-2001	Weathered/Competent	2	0	-	-	-
MW-2002	Weathered/Competent	1	0	-	-	-
MW-2004	Weathered/Competent	2	0	-	-	-
MW-2005	Weathered/Competent	1	0	-	-	-
MW-2006	Weathered/Competent	1	0	-	-	-
MW-2007	Weathered/Competent	2	0	-	-	-
MW-2008	Weathered/Competent	2	0	-	-	-
MW-2011	Weathered/Competent	2	0	-	-	-
MW-2013	Weathered/Competent	2	0	-	-	-
MW-2015	Weathered/Competent	2	0	-	-	-
MW-2020	Weathered/Competent	2	0	-	-	-
MW-3003	Weathered/Competent	1	0	-	-	-
MW-3010	Weathered/Competent	2	0	-	-	-
MW-3019	Weathered/Competent	1	0	-	-	-
MW-4002	Weathered/Competent	2	0	-	-	-
MW-4003	Weathered/Competent	0	0	-	-	-
MW-4004	Weathered/Competent	1	0	-	-	-
MW-4005	Weathered/Competent	1	1	15	15	15
MW-4010	Weathered/Competent	1	0	-	-	-
MW-4013	Weathered/Competent	1	0	-	-	-
MW-4014	Weathered/Competent	1	0	-	-	-
MW-4015	Weathered/Competent	1	0	-	-	-
MW-4017	Weathered/Competent	1	0	-	-	-
MW-4018	Weathered/Competent	1	0	-	-	-
MW-4019	Weathered/Competent	2	0	-	-	-
MW-4021	Weathered/Competent	1	0	-	-	-
MW-4023	Weathered/Competent	1	0	-	-	-

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (pCi/L) Minimum Detected	Maximum Detected	Average
Uranium, Total (DL = 1.0 pCi/L)						
MW-2019	Competent	6	6	0.68	3.5	2.5
MW-2021	Competent	5	4	2.2	3.0	2.3
MW-2022	Competent	5	3	1.4	2.3	1.2
MW-2023	Competent	5	4	2.0	4.1	2.3
MW-2024	Competent	4	1	1.4	1.4	0.72
MW-2025	Competent	5	4	1.4	1.9	1.4
MW-2026	Competent	6	5	1.6	4.4	2.1
MW-2027	Competent	5	4	0.68	2.7	1.9
MW-2028	Competent	5	4	1.5	11	4.5
MW-2029	Competent	5	4	1.0	2.8	1.5
MW-3002	Competent	5	5	1.2	7.8	3.4
MW-3006	Competent	6	4	1.2	2.7	1.5
MW-4007	Competent	5	5	0.68	3.4	1.9

TABLE B.2 (Cont.)

Metal/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (pCi/L) Minimum Detected	Maximum Detected	Average
Uranium, Total (Cont.)						
MW-4008	Competent	6	5	1.9	4.1	2.6
MW-4009	Competent	4	4	1.4	3.4	2.2
MW-4011	Competent	6	5	2.5	9.5	3.8
MW-4012	Competent	4	4	1.3	3.5	2.4
MW-4016	Competent	6	6	4.6	5.4	5.0
MW-4020	Competent	5	5	13	28	21
MW-4022	Competent	5	4	2.1	17	5.7
MW-2003	Weathered	10	8	0.68	3.6	2.1
MW-2009	Weathered	9	8	1.5	4.2	2.4
MW-2010	Weathered	10	6	0.77	13	1.1
MW-2012	Weathered	10	2	1.0	2.3	0.73
MW-2014	Weathered	9	7	0.90	3.9	1.7
MW-2016	Weathered	5	2	1.1	1.2	0.76
MW-2017	Weathered	9	9	3.7	15	7.0
MW-2018	Weathered	11	10	2.0	4.2	3.1
MW-3001	Weathered	6	2	2.6	3.4	1.3
MW-3008	Weathered	8	8	2.4	9.1	6.03
MW-3009	Weathered	9	9	3.9	60	34
MW-4001	Weathered	7	5	0.68	1.5	0.97
MW-4006	Weathered	6	0	-	-	-
MW-2001	Weathered/Competent	9	4	1.2	2.4	1.0
MW-2002	Weathered/Competent	4	2	3.4	5.3	2.4
MW-2004	Weathered/Competent	9	8	1.0	5.9	2.9
MW-2005	Weathered/Competent	9	5	1.3	2.4	1.2
MW-2006	Weathered/Competent	9	6	1.1	2.4	1.2
MW-2007	Weathered/Competent	9	7	2.4	7.7	3.5
MW-2008	Weathered/Competent	9	5	1.4	3.6	1.5
MW-2011	Weathered/Competent	10	1	2.1	2.1	0.66
MW-2013	Weathered/Competent	9	4	1.0	2.6	1.0
MW-2015	Weathered/Competent	10	8	8.4	2.4	3.4
MW-2020	Weathered/Competent	9	9	4.1	30	15
MW-3003	Weathered/Competent	5	5	9.5	17	14
MW-3010	Weathered/Competent	8	8	1.2	3.6	2.0
MW-3019	Weathered/Competent	4	4	5.5	9.5	7.6
MW-4002	Weathered/Competent	7	3	0.53	1.5	0.75
MW-4003	Weathered/Competent	5	1	1.7	1.7	0.78
MW-4004	Weathered/Competent	5	5	1.9	4.3	2.9
MW-4005	Weathered/Competent	5	5	1.0	3.0	2.0
MW-4010	Weathered/Competent	4	4	2.5	5.4	4.0
MW-4013	Weathered/Competent	5	3	1.4	1.8	1.2
MW-4014	Weathered/Competent	5	0	-	-	-
MW-4015	Weathered/Competent	5	3	1.1	1.6	1.0
MW-4017	Weathered/Competent	5	5	1.4	3.4	2.6
MW-4018	Weathered/Competent	5	4	1.3	2.3	1.6
MW-4019	Weathered/Competent	8	8	1.5	5.2	2.8
MW-4021	Weathered/Competent	5	5	2.9	18	10
MW-4023	Weathered/Competent	5	4	2.0	3.2	2.1

[a] DL = detection limit.

TABLE B.3 Concentrations of Inorganic Anions in Groundwater Samples, 1987-1990

Anion/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (mg/L) Minimum Detected	Maximum Detected	Average
Fluoride (DL = 0.25 mg/L)[a]						
MW-2019	Competent	5	1	0.26	0.26	0.15
MW-2021	Competent	5	1	0.14	0.14	0.13
MW-2022	Competent	5	1	0.14	0.14	0.13
MW-2023	Competent	6	3	0.28	0.25	0.19
MW-2024	Competent	6	3	0.25	0.29	0.19
MW-2025	Competent	5	2	0.19	0.25	0.16
MW-2026	Competent	6	2	0.16	0.18	0.14
MW-2027	Competent	6	2	0.14	0.17	0.13
MW-2028	Competent	5	2	0.30	0.17	0.17
MW-2029	Competent	6	2	0.17	0.19	0.14
MW-3002	Competent	5	1	0.25	0.25	0.15
MW-3006	Competent	7	5	0.30	0.26	0.24
MW-4007	Competent	7	3	0.25	0.50	0.21
MW-4008	Competent	7	3	0.28	0.36	0.21
MW-4009	Competent	5	3	0.30	0.33	0.24
MW-4011	Competent	7	4	0.40	0.42	0.26
MW-4012	Competent	6	3	0.30	0.32	0.21
MW-4016	Competent	7	3	0.19	0.21	0.17
MW-4020	Competent	5	1	0.22	0.22	0.14
MW-4022	Competent	6	2	0.28	0.39	0.20
MW-2003	Weathered	10	6	0.09	15	2.1
MW-2009	Weathered	9	4	0.56	1.0	0.31
MW-2010	Weathered	10	4	0.08	0.60	0.21
MW-2012	Weathered	10	4	0.05	0.90	0.29
MW-2014	Weathered	9	6	0.07	1.4	0.37
MW-2016	Weathered	4	3	0.40	2.0	0.73
MW-2017	Weathered	10	7	0.19	1.2	0.51
MW-2018	Weathered	11	6	0.30	230	0.28
MW-3001	Weathered	7	3	0.15	0.60	0.21
MW-3008	Weathered	9	6	0.16	6.5	2.0
MW-3009	Weathered	9	7	0.11	0.58	0.34
MW-4001	Weathered	8	5	0.11	0.55	0.29
MW-4006	Weathered	7	5	0.15	0.44	0.26
MW-2001	Weathered/Competent	10	5	0.30	1.3	0.38
MW-2002	Weathered/Competent	4	1	15	15	3.8
MW-2004	Weathered/Competent	10	4	0.16	0.70	0.26
MW-2005	Weathered/Competent	10	5	0.19	1.0	0.31
MW-2006	Weathered/Competent	10	3	0.40	0.70	0.27
MW-2007	Weathered/Competent	9	5	0.10	0.40	0.22
MW-2008	Weathered/Competent	10	5	0.08	17	1.9
MW-2011	Weathered/Competent	9	5	0.09	0.80	0.31
MW-2013	Weathered/Competent	10	4	0.40	0.68	0.26
MW-2015	Weathered/Competent	10	4	0.15	0.90	0.29
MW-2020	Weathered/Competent	10	4	0.40	2.0	0.30
MW-3003	Weathered/Competent	6	2	0.18	0.22	0.15
MW-3010	Weathered/Competent	9	6	0.21	0.80	0.35
MW-3019	Weathered/Competent	5	2	0.25	0.31	0.19
MW-4002	Weathered/Competent	8	4	0.10	0.40	0.20
MW-4003	Weathered/Competent	6	2	0.17	0.21	0.15
MW-4004	Weathered/Competent	6	3	0.22	0.34	0.20
MW-4005	Weathered/Competent	6	3	0.30	0.33	0.22

TABLE B.3 (Cont.)

Anion/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (mg/L) Minimum Detected	Maximum Detected	Average
Fluoride (Cont.)						
MW-4010	Weathered/Competent	6	2	0.23	0.24	0.16
MW-4013	Weathered/Competent	6	1	0.07	0.07	0.12
MW-4014	Weathered/Competent	6	2	0.19	0.21	0.15
MW-4015	Weathered/Competent	6	2	0.11	0.12	0.12
MW-4017	Weathered/Competent	6	2	0.15	0.16	0.13
MW-4018	Weathered/Competent	6	2	0.11	0.15	0.13
MW-4019	Weathered/Competent	8	4	0.30	0.50	0.27
MW-4021	Weathered/Competent	6	3	0.14	0.36	0.17
MW-4023	Weathered/Competent	6	2	0.18	0.25	0.15
Nitrate (DL = 0.10 mg/L)						
MW-2019	Competent	7	3	0.80	1.0	0.46
MW-2021	Competent	5	0	-	-	-
MW-2022	Competent	5	0	-	-	-
MW-2023	Competent	6	4	0.20	1.3	0.37
MW-2024	Competent	6	0	-	-	-
MW-2025	Competent	5	0	-	-	-
MW-2026	Competent	6	1	0.50	0.50	0.13
MW-2027	Competent	6	4	0.20	0.53	0.22
MW-2028	Competent	5	1	6.1	6.1	1.3
MW-2029	Competent	6	1	0.11	0.11	0.06
MW-3002	Competent	5	2	0.18	0.20	0.11
MW-3006	Competent	6	2	1.3	4.0	0.92
MW-4007	Competent	7	1	140	140	20
MW-4008	Competent	7	1	3.2	3.2	0.50
MW-4009	Competent	4	3	0.33	0.60	0.35
MW-4011	Competent	7	5	100	95	53
MW-4012	Competent	6	3	0.33	130	22
MW-4016	Competent	7	0	-	-	-
MW-4020	Competent	5	3	0.18	5.5	1.0
MW-4022	Competent	6	5	0.09	54	10
MW-2003	Weathered	11	11	270	3,900	2,600
MW-2009	Weathered	8	8	1.7	8.8	6.7
MW-2010	Weathered	10	10	0.80	5.7	3.5
MW-2012	Weathered	10	10	0.38	4.0	1.7
MW-2014	Weathered	9	9	1.3	10	7.6
MW-2016	Weathered	5	2	840	2,500	670
MW-2017	Weathered	10	10	0.90	7.2	2.9
MW-2018	Weathered	11	11	0.48	4.1	2.2
MW-3001	Weathered	7	7	130	2,200	870
MW-3008	Weathered	9	9	600	6,000	4,100
MW-3009	Weathered	10	10	17	11,000	1,600
MW-4001	Weathered	7	7	32	180	120
MW-4006	Weathered	6	6	14	30	21
MW-2001	Weathered/Competent	10	10	6.5	100	38
MW-2002	Weathered/Competent	4	4	490	3,600	1,600
MW-2004	Weathered/Competent	10	10	0.61	5.3	3.0
MW-2005	Weathered/Competent	9	8	57	12,000	1,500
MW-2006	Weathered/Competent	11	11	5.8	36	24
MW-2007	Weathered/Competent	9	1	0.30	0.30	0.08
MW-2008	Weathered/Competent	9	9	1.7	8.8	16
MW-2011	Weathered/Competent	10	10	0.80	270	17
MW-2013	Weathered/Competent	9	9	0.90	4.4	3.0

TABLE B.3 (Cont.)

Anion/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (mg/L) Minimum Detected	Maximum Detected	Average
Nitrate (Cont.)						
MW-2015	Weathered/Competent	9	9	0.28	1.8	1.2
MW-2020	Weathered/Competent	10	10	0.50	5.2	2.4
MW-3000	Weathered/Competent	5	4	1,900	2,200	1,700
MW-3010	Weathered/Competent	8	8	1.1	14	5.6
MW-3019	Weathered/Competent	5	3	0.20	0.59	0.25
MW-4002	Weathered/Competent	8	8	2.7	27	12
MW-4003	Weathered/Competent	6	6	0.59	2,100	350
MW-4004	Weathered/Competent	6	6	0.50	2.3	1.5
MW-4005	Weathered/Competent	6	6	0.98	8.7	5.0
MW-4010	Weathered/Competent	6	6	0.09	1.3	0.52
MW-4013	Weathered/Competent	6	5	12	660	330
MW-4014	Weathered/Competent	5	5	7.7	14	10
MW-4015	Weathered/Competent	6	6	1.3	22	8.3
MW-4017	Weathered/Competent	6	5	0.47	3.4	1.8
MW-4018	Weathered/Competent	6	6	0.08	9.7	6.1
MW-4019	Weathered/Competent	8	8	0.18	1.1	0.65
MW-4021	Weathered/Competent	6	1	1.4	1.4	0.27
MW-4023	Weathered/Competent	6	6	7.3	29	20
Sulfate (DL = 1.0 mg/L)						
MW-2019	Competent	7	7	7.8	18	16
MW-2021	Competent	5	5	12	25	15
MW-2022	Competent	5	5	1.5	18	15
MW-2023	Competent	6	6	22	55	31
MW-2024	Competent	6	6	23	88	42
MW-2025	Competent	5	5	16	23	18
MW-2026	Competent	6	6	12	24	15
MW-2027	Competent	5	5	11	15	13
MW-2028	Competent	5	5	96	120	100
MW-2029	Competent	6	6	17	31	25
MW-3002	Competent	5	5	19	20	19
MW-3006	Competent	7	7	24	80	54
MW-4007	Competent	7	7	9.9	22	12
MW-4008	Competent	7	7	12	17	14
MW-4009	Competent	5	5	13	30	19
MW-4011	Competent	7	7	36	52	47
MW-4012	Competent	6	6	40	58	47
MW-4016	Competent	7	7	12	52	23
MW-4020	Competent	5	5	120	180	140
MW-4022	Competent	6	6	32	71	53
MW-2003	Weathered	10	10	70	300	210
MW-2009	Weathered	9	9	38	190	100
MW-2010	Weathered	10	10	28	57	39
MW-2012	Weathered	9	9	51	110	73
MW-2014	Weathered	8	8	27	130	46
MW-2016	Weathered	5	5	30	110	51
MW-2017	Weathered	10	10	460	1,800	800
MW-2018	Weathered	12	12	9.3	25	12
MW-3001	Weathered	7	7	20	54	27
MW-3008	Weathered	9	9	11	100	53
MW-3009	Weathered	9	9	27	72	47
MW-4001	Weathered	8	8	50	160	75
MW-4006	Weathered	8	8	31	130	47

TABLE B.3 (Cont.)

Anion/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (mg/L) Minimum Detected	Maximum Detected	Average
Sulfate (Cont.)						
MW-2001	Weathered/Competent	10	10	4.4	23	8.3
MW-2002	Weathered/Competent	4	4	95	198	160
MW-2004	Weathered/Competent	10	10	1.9	6.7	3.4
MW-2005	Weathered/Competent	9	9	2.3	8.7	4.2
MW-2006	Weathered/Competent	10	10	31	53	37
MW-2007	Weathered/Competent	9	9	14	18	16
MW-2008	Weathered/Competent	9	9	9.1	170	47
MW-2011	Weathered/Competent	10	10	8.6	51	19
MW-2013	Weathered/Competent	10	10	12	52	28
MW-2015	Weathered/Competent	10	10	85	1,200	110
MW-2020	Weathered/Competent	10	10	140	340	220
MW-3003	Weathered/Competent	6	6	150	230	190
MW-3010	Weathered/Competent	9	9	6.3	24	10.22
MW-3019	Weathered/Competent	5	5	5.2	10	7.3
MW-4002	Weathered/Competent	8	8	15	37	23
MW-4003	Weathered/Competent	6	6	28	36	31
MW-4004	Weathered/Competent	6	6	25	35	29
MW-4005	Weathered/Competent	6	6	16	22	19
MW-4010	Weathered/Competent	6	6	24	43	29
MW-4013	Weathered/Competent	6	6	43	68	54
MW-4014	Weathered/Competent	6	6	22	36	28
MW-4015	Weathered/Competent	6	5	4.8	19	9.4
MW-4017	Weathered/Competent	6	6	5.1	11	8.5
MW-4018	Weathered/Competent	6	6	8.2	19	12
MW-4019	Weathered/Competent	8	8	7.0	12	8.9
MW-4021	Weathered/Competent	6	6	52	340	260
MW-4023	Weathered/Competent	6	6	24	290	110

[a] DL = detection limit.

TABLE B.4 Concentrations of Nitroaromatic Compounds in Groundwater Samples, 1987-1990

Compound/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
1,3-Dinitrobenzene (DL = 0.4 µg/L)[a]						
MW-2019	Competent	6	0	-	-	-
MW-2021	Competent	4	0	-	-	-
MW-2022	Competent	5	0	-	-	-
MW-2023	Competent	6	0	-	-	-
MW-2024	Competent	6	0	-	-	-
MW-2025	Competent	4	0	-	-	-
MW-2026	Competent	5	0	-	-	-
MW-2027	Competent	6	0	-	-	-
MW-2028	Competent	5	0	-	-	-
MW-2029	Competent	5	0	-	-	-
MW-3002	Competent	5	0	-	-	-
MW-3006	Competent	6	0	-	-	-
MW-4007	Competent	7	0	-	-	-
MW-4008	Competent	7	0	-	-	-
MW-4009	Competent	5	0	-	-	-
MW-4011	Competent	7	0	-	-	-
MW-4012	Competent	6	0	-	-	-
MW-4016	Competent	6	0	-	-	-
MW-4020	Competent	5	0	-	-	-
MW-4022	Competent	6	0	-	-	-
MW-2008	Weathered	10	0	-	-	-
MW-2009	Weathered	9	0	-	-	-
MW-2010	Weathered	10	1	0.50	0.50	0.23
MW-2012	Weathered	10	4	0.11	0.57	0.30
MW-2014	Weathered	9	2	0.29	4.6	0.65
MW-2016	Weathered	5	2	0.14	1.1	0.29
MW-2017	Weathered	10	0	-	-	-
MW-2018	Weathered	11	0	-	-	-
MW-3001	Weathered	7	0	-	-	-
MW-3008	Weathered	9	0	-	-	-
MW-3009	Weathered	10	0	-	-	-
MW-4001	Weathered	8	5	1.9	5.1	2.0
MW-4006	Weathered	7	4	1.3	3.2	1.4
MW-2001	Weathered/Competent	10	1	0.45	0.45	0.23
MW-2002	Weathered/Competent	4	0	-	-	-
MW-2004	Weathered/Competent	10	0	-	-	-
MW-2005	Weathered/Competent	10	0	-	-	-
MW-2006	Weathered/Competent	10	4	0.14	1.8	0.52
MW-2007	Weathered/Competent	9	0	-	-	-
MW-2008	Weathered/Competent	10	6	0.14	0.87	0.43
MW-2011	Weathered/Competent	10	3	0.48	1.9	0.55
MW-2013	Weathered/Competent	11	4	0.50	12	2.8
MW-2015	Weathered/Competent	10	0	-	-	-
MW-2020	Weathered/Competent	10	0	-	-	-
MW-3003	Weathered/Competent	6	0	-	-	-
MW-3010	Weathered/Competent	9	0	-	-	-
MW-3019	Weathered/Competent	5	0	-	-	-
MW-4002	Weathered/Competent	8	0	-	-	-
MW-4003	Weathered/Competent	6	1	0.14	0.14	0.19
MW-4004	Weathered/Competent	6	0	-	-	-
MW-4005	Weathered/Competent	6	0	-	-	-
MW-4010	Weathered/Competent	6	0	-	-	-

TABLE B.4 (Cont.)

Compound/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
1,3-Dinitrobenzene (Cont.)						
MW-4013	Weathered/Competent	6	2	0.71	0.80	0.39
MW-4014	Weathered/Competent	6	0	-	-	-
MW-4015	Weathered/Competent	6	2	0.11	0.12	0.17
MW-4017	Weathered/Competent	6	0	-	-	-
MW-4018	Weathered/Competent	6	0	-	-	-
MW-4019	Weathered/Competent	9	0	-	-	-
MW-4021	Weathered/Competent	6	0	-	-	-
MW-4023	Weathered/Competent	6	0	-	-	-
2,4-Dinitrotoluene (DL = 0.2 µg/L)						
MW-2019	Competent	5	0	-	-	-
MW-2021	Competent	4	0	-	-	-
MW-2022	Competent	5	0	-	-	-
MW-2023	Competent	6	0	-	-	-
MW-2024	Competent	6	0	-	-	-
MW-2025	Competent	4	0	-	-	-
MW-2026	Competent	5	0	-	-	-
MW-2027	Competent	6	0	-	-	-
MW-2028	Competent	5	0	-	-	-
MW-2029	Competent	5	1	0.59	0.59	0.20
MW-3002	Competent	5	0	-	-	-
MW-3006	Competent	6	1	0.58	0.58	0.18
MW-4007	Competent	7	0	-	-	-
MW-4008	Competent	7	0	-	-	-
MW-4009	Competent	5	0	-	-	-
MW-4011	Competent	7	0	-	-	-
MW-4012	Competent	6	0	-	-	-
MW-4016	Competent	6	0	-	-	-
MW-4020	Competent	5	0	-	-	-
MW-4022	Competent	6	0	-	-	-
MW-2008	Weathered	10	8	0.24	0.60	0.35
MW-2009	Weathered	9	4	0.30	130	15
MW-2010	Weathered	10	2	0.30	0.95	0.21
MW-2012	Weathered	10	5	0.60	7.6	1.8
MW-2014	Weathered	9	3	0.37	2.2	0.47
MW-2016	Weathered	5	0	-	-	-
MW-2017	Weathered	10	1	0.40	0.40	0.13
MW-2018	Weathered	11	0	-	-	-
MW-3001	Weathered	7	5	0.35	2.2	0.62
MW-3008	Weathered	9	4	0.06	0.63	0.19
MW-3009	Weathered	9	3	0.07	0.60	0.20
MW-4001	Weathered	8	8	0.85	3.3	2.30
MW-4006	Weathered	7	3	0.37	0.70	0.27
MW-2001	Weathered/Competent	10	2	0.14	2.1	0.30
MW-2002	Weathered/Competent	4	1	0.06	0.06	0.09
MW-2004	Weathered/Competent	10	0	-	-	-
MW-2005	Weathered/Competent	10	3	0.07	0.40	0.13
MW-2006	Weathered/Competent	9	2	0.57	3.7	0.55
MW-2007	Weathered/Competent	9	1	0.30	0.30	0.12
MW-2008	Weathered/Competent	9	2	0.13	0.24	0.12
MW-2011	Weathered/Competent	10	3	2.2	14	2.0
MW-2013	Weathered/Competent	10	10	0.54	390	140
MW-2015	Weathered/Competent	10	0	-	-	-

TABLE B.4 (Cont.)

Compound/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
2,4-Dinitrotoluene (Cont.)						
MW-2020	Weathered/Competent	9	0	-	-	-
MW-3003	Weathered/Competent	6	0	-	-	-
MW-3010	Weathered/Competent	9	0	-	-	-
MW-3019	Weathered/Competent	5	0	-	-	-
MW-4002	Weathered/Competent	8	3	0.23	0.70	0.30
MW-4003	Weathered/Competent	6	1	0.30	0.30	0.13
MW-4004	Weathered/Competent	6	0	-	-	-
MW-4005	Weathered/Competent	6	0	-	-	-
MW-4010	Weathered/Competent	5	0	-	-	-
MW-4013	Weathered/Competent	6	1	0.40	0.40	0.15
MW-4014	Weathered/Competent	6	0	-	-	-
MW-4015	Weathered/Competent	5	0	-	-	-
MW-4017	Weathered/Competent	6	0	-	-	-
MW-4018	Weathered/Competent	6	0	-	-	-
MW-4019	Weathered/Competent	9	1	65	65	7.3
MW-4021	Weathered/Competent	6	0	-	-	-
MW-4023	Weathered/Competent	6	1	0.06	0.06	0.09
2,6-Dinitrotoluene (DL = 0.60 µg/L)						
MW-2019	Competent	6	0	-	-	-
MW-2021	Competent	4	0	-	-	-
MW-2023	Competent	5	0	-	-	-
MW-2022	Competent	6	0	-	-	-
MW-2024	Competent	6	0	-	-	-
MW-2025	Competent	4	0	-	-	-
MW-2026	Competent	5	0	-	-	-
MW-2027	Competent	6	0	-	-	-
MW-2028	Competent	5	0	-	-	-
MW-2029	Competent	5	0	-	-	-
MW-3002	Competent	5	0	-	-	-
MW-3006	Competent	6	1	2.9	2.9	0.73
MW-4007	Competent	7	0	-	-	-
MW-4008	Competent	7	0	-	-	-
MW-4009	Competent	5	1	1.7	1.7	0.58
MW-4011	Competent	7	0	-	-	-
MW-4012	Competent	6	0	-	-	-
MW-4016	Competent	6	0	-	-	-
MW-4020	Competent	5	0	-	-	-
MW-4022	Competent	6	0	-	-	-
MW-2003	Weathered	10	10	0.61	25	3.3
MW-2009	Weathered	8	5	0.18	0.90	0.49
MW-2010	Weathered	10	5	0.77	13	1.8
MW-2012	Weathered	10	3	1.9	7.5	1.4
MW-2014	Weathered	9	6	0.76	1.7	0.88
MW-2016	Weathered	5	1	1.0	1.0	0.27
MW-2017	Weathered	10	0	-	-	-
MW-2018	Weathered	11	1	16	16	1.9
MW-3001	Weathered	7	2	0.17	2.9	0.65
MW-3008	Weathered	9	1	0.05	0.04	0.27
MW-3009	Weathered	10	1	0.14	0.14	0.28
MW-4001	Weathered	8	7	2.4	4.2	3.0
MW-4006	Weathered	7	6	2.7	7.2	3.7
MW-2001	Weathered/Competent	10	3	0.06	24	2.8

TABLE B.4 (Cont.)

Compound/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
2,6-Dinitrotoluene (Cont.)						
MW-2002	Weathered/Competent	4	2	0.34	0.46	0.35
MW-2004	Weathered/Competent	10	1	21	21	2.3
MW-2005	Weathered/Competent	10	2	0.07	0.90	0.34
MW-2006	Weathered/Competent	10	9	0.87	50	8.8
MW-2007	Weathered/Competent	9	0	-	-	-
MW-2008	Weathered/Competent	10	5	0.88	1.6	0.72
MW-2011	Weathered/Competent	10	9	1.9	73	17
MW-2013	Weathered/Competent	11	11	7.8	290	92
MW-2015	Weathered/Competent	9	1	0.62	0.62	0.34
MW-2020	Weathered/Competent	10	0	-	-	-
MW-3003	Weathered/Competent	6	0	-	-	-
MW-3010	Weathered/Competent	9	0	-	-	-
MW-3019	Weathered/Competent	5	0	-	-	-
MW-4002	Weathered/Competent	8	3	0.56	1.3	0.54
MW-4003	Weathered/Competent	6	1	0.80	0.80	0.38
MW-4004	Weathered/Competent	6	1	0.94	0.94	0.41
MW-4005	Weathered/Competent	6	0	-	-	-
MW-4010	Weathered/Competent	6	0	-	-	-
MW-4013	Weathered/Competent	6	4	0.68	1.5	0.85
MW-4014	Weathered/Competent	6	1	0.05	0.05	0.26
MW-4015	Weathered/Competent	6	5	0.82	6.0	2.0
MW-4017	Weathered/Competent	6	0	-	-	-
MW-4018	Weathered/Competent	6	0	-	-	-
MW-4019	Weathered/Competent	9	0	-	-	-
MW-4021	Weathered/Competent	6	0	-	-	-
MW-4023	Weathered/Competent	6	1	0.02	0.02	0.25
Nitrobenzene (DL = 0.60 µg/L)						
MW-2019	Competent	6	1	0.33	0.33	0.31
MW-2021	Competent	4	0	-	-	-
MW-2022	Competent	5	2	1.4	6.0	1.7
MW-2023	Competent	6	0	-	-	-
MW-2024	Competent	6	0	-	-	-
MW-2025	Competent	4	0	-	-	-
MW-2026	Competent	5	0	-	-	-
MW-2027	Competent	6	1	1.5	1.5	0.49
MW-2028	Competent	5	2	1.1	1.2	0.64
MW-2029	Competent	5	0	-	-	-
MW-3002	Competent	5	1	2.2	2.2	0.67
MW-3006	Competent	6	2	2.0	5.3	1.4
MW-4007	Competent	7	0	-	-	-
MW-4008	Competent	7	0	-	-	-
MW-4009	Competent	5	0	-	-	-
MW-4011	Competent	7	0	-	-	-
MW-4012	Competent	6	0	-	-	-
MW-4016	Competent	6	0	-	-	-
MW-4020	Competent	5	0	-	-	-
MW-4022	Competent	6	0	-	-	-
MW-2003	Weathered	10	0	-	-	-
MW-2009	Weathered	9	1	22	22	2.7
MW-2010	Weathered	10	0	-	-	-
MW-2012	Weathered	10	1	1.2	1.2	0.39
MW-2014	Weathered	9	1	4.0	4.0	0.71

TABLE B.4 (Cont.)

Compound/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
Nitrobenzene (Cont.)						
MW-2016	Weathered	5	0	-	-	-
MW-2017	Weathered	10	0	-	-	-
MW-2018	Weathered	11	0	-	-	-
MW-3001	Weathered	7	1	4.6	4.6	0.91
MW-3008	Weathered	9	0	-	-	-
MW-3009	Weathered	10	0	-	-	-
MW-4001	Weathered	8	4	4.3	7.6	3.1
MW-4006	Weathered	7	4	2.5	3.7	1.9
MW-2001	Weathered/Competent	10	1	1.0	1.0	0.37
MW-2002	Weathered/Competent	4	0	-	-	-
MW-2004	Weathered/Competent	10	0	-	-	-
MW-2005	Weathered/Competent	10	0	-	-	-
MW-2006	Weathered/Competent	10	3	1.6	8.3	1.8
MW-2007	Weathered/Competent	9	0	-	-	-
MW-2008	Weathered/Competent	10	1	2.0	2.0	0.47
MW-2011	Weathered/Competent	10	0	-	-	-
MW-2013	Weathered/Competent	11	3	1.8	5.4	1.3
MW-2015	Weathered/Competent	10	0	-	-	-
MW-2020	Weathered/Competent	10	1	1.7	1.7	0.44
MW-3003	Weathered/Competent	6	0	-	-	-
MW-3010	Weathered/Competent	9	0	-	-	-
MW-3019	Weathered/Competent	5	1	0.86	0.86	0.41
MW-4002	Weathered/Competent	8	0	-	-	-
MW-4003	Weathered/Competent	6	0	-	-	-
MW-4004	Weathered/Competent	6	0	-	-	-
MW-4005	Weathered/Competent	6	0	-	-	-
MW-4010	Weathered/Competent	6	0	-	-	-
MW-4013	Weathered/Competent	6	0	-	-	-
MW-4014	Weathered/Competent	6	0	-	-	-
MW-4015	Weathered/Competent	6	1	0.41	0.41	0.32
MW-4017	Weathered/Competent	6	0	-	-	-
MW-4018	Weathered/Competent	6	0	-	-	-
MW-4019	Weathered/Competent	9	0	-	-	-
MW-4021	Weathered/Competent	6	1	1.7	1.7	0.54
MW-4023	Weathered/Competent	6	0	-	-	-
1,3,5-Trinitrobenzene (DL = 0.03 µg/L)						
MW-2019	Competent	6	3	0.05	0.15	0.05
MW-2021	Competent	4	0	-	-	-
MW-2022	Competent	5	2	0.03	0.21	0.07
MW-2023	Competent	6	1	-	-	-
MW-2024	Competent	6	2	0.02	0.07	0.02
MW-2025	Competent	4	0	-	-	-
MW-2026	Competent	6	1	0.04	0.04	0.02
MW-2027	Competent	6	0	-	-	-
MW-2028	Competent	5	1	0.03	0.03	0.02
MW-2029	Competent	5	0	-	-	-
MW-3002	Competent	5	2	0.05	0.53	0.13
MW-3006	Competent	6	3	0.06	0.63	0.15
MW-4007	Competent	7	2	0.02	0.17	0.04
MW-4008	Competent	7	0	-	-	-
MW-4009	Competent	5	2	0.27	17	3.4
MW-4011	Competent	7	0	-	-	-

TABLE B.4 (Cont.)

Compound/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
1,3,5-Trinitrobenzene (Cont.)						
MW-4012	Competent	6	0	-	-	-
MW-4016	Competent	6	0	-	-	-
MW-4020	Competent	5	1	0.11	0.11	0.07
MW-4022	Competent	6	1	0.06	0.06	0.04
MW-2003	Weathered	10	1	0.06	0.06	0.02
MW-2009	Weathered	9	2	0.18	0.78	0.12
MW-2010	Weathered	10	4	0.06	0.40	0.11
MW-2012	Weathered	9	7	0.54	5.2	1.6
MW-2014	Weathered	9	8	0.40	4.8	1.8
MW-2016	Weathered	5	0	-	-	-
MW-2017	Weathered	10	2	0.11	0.13	0.04
MW-2018	Weathered	11	2	0.06	0.21	0.04
MW-3001	Weathered	6	5	0.04	0.17	0.09
MW-3008	Weathered	9	1	0.06	0.06	0.02
MW-3009	Weathered	10	4	0.05	0.36	0.08
MW-4001	Weathered	8	8	11	61	35
MW-4006	Weathered	7	7	1.3	22	9.0
MW-2001	Weathered/Competent	10	2	0.50	0.50	0.11
MW-2002	Weathered/Competent	4	1	0.02	0.02	0.02
MW-2004	Weathered/Competent	10	1	0.14	0.17	0.03
MW-2005	Weathered/Competent	10	2	0.10	0.18	0.04
MW-2006	Weathered/Competent	10	10	2.6	16	8.29
MW-2007	Weathered/Competent	9	1	0.13	0.13	0.03
MW-2008	Weathered/Competent	10	6	0.04	12	1.6
MW-2011	Weathered/Competent	10	5	0.13	0.55	0.19
MW-2013	Weathered/Competent	11	11	2.3	35	12
MW-2015	Weathered/Competent	10	1	0.21	0.21	0.03
MW-2020	Weathered/Competent	10	0	-	-	-
MW-3003	Weathered/Competent	6	0	-	-	-
MW-3010	Weathered/Competent	9	0	-	-	-
MW-3019	Weathered/Competent	5	1	0.04	0.04	0.02
MW-4002	Weathered/Competent	7	3	0.02	0.30	1.7
MW-4003	Weathered/Competent	6	1	0.05	0.05	0.02
MW-4004	Weathered/Competent	6	0	-	-	-
MW-4005	Weathered/Competent	6	2	0.02	0.18	0.04
MW-4010	Weathered/Competent	6	1	0.05	0.05	0.02
MW-4013	Weathered/Competent	6	5	33	89	42
MW-4014	Weathered/Competent	6	3	0.27	0.45	0.19
MW-4015	Weathered/Competent	6	4	0.10	0.48	0.14
MW-4017	Weathered/Competent	6	1	0.05	0.05	0.02
MW-4018	Weathered/Competent	6	1	0.12	0.12	0.03
MW-4019	Weathered/Competent	9	2	0.08	0.08	0.03
MW-4021	Weathered/Competent	6	1	0.09	0.09	0.03
MW-4023	Weathered/Competent	6	2	0.08	0.29	0.07
2,4,6-Trinitrotoluene (DL = 0.5 µg/L)						
MW-2019	Competent	6	0	-	-	-
MW-2021	Competent	4	0	-	-	-
MW-2022	Competent	5	0	-	-	-
MW-2023	Competent	6	0	-	-	-
MW-2024	Competent	6	0	-	-	-
MW-2025	Competent	4	0	-	-	-
MW-2026	Competent	5	0	-	-	-

TABLE B.4 (Cont.)

Compound/ Well Number	Completion Interval	Number of Samples Analyzed	Number of Samples Detected	Concentration (µg/L) Minimum Detected	Maximum Detected	Average
2,4,6-Trinitrotoluene (Cont.)						
MW-2027	Competent	6	0	-	-	-
MW-2028	Competent	5	0	-	-	-
MW-2029	Competent	5	0	-	-	-
MW-3002	Competent	5	0	-	-	-
MW-3006	Competent	6	1	1.5	1.5	0
MW-4007	Competent	7	0	-	-	-
MW-4008	Competent	7	0	-	-	-
MW-4009	Competent	5	1	7.8	7.8	1.8
MW-4011	Competent	7	0	-	-	-
MW-4012	Competent	6	0	-	-	-
MW-4016	Competent	6	0	-	-	-
MW-4020	Competent	5	0	-	-	-
MW-4022	Competent	6	0	-	-	-
MW-2003	Weathered	10	0	-	-	-
MW-2009	Weathered	9	1	7.1	7.1	1.0
MW-2010	Weathered	10	4	0.36	1.7	0.47
MW-2012	Weathered	10	9	0.63	1.8	1.0
MW-2014	Weathered	9	3	1.6	2.3	0.80
MW-2016	Weathered	5	0	-	-	-
MW-2017	Weathered	10	0	-	-	-
MW-2018	Weathered	11	0	-	-	-
MW-3001	Weathered	7	1	18	18	2.8
MW-3008	Weathered	9	0	-	-	-
MW-3009	Weathered	10	0	-	-	-
MW-4001	Weathered	7	6	1.2	1.9	1.4
MW-4006	Weathered	7	1	1.2	1.2	0.39
MW-2001	Weathered/Competent	10	0	-	-	-
MW-2002	Weathered/Competent	4	1	0.60	0.60	0.34
MW-2004	Weathered/Competent	10	0	-	-	-
MW-2005	Weathered/Competent	10	0	-	-	-
MW-2006	Weathered/Competent	10	2	0.63	0.78	0.34
MW-2007	Weathered/Competent	9	0	-	-	-
MW-2008	Weathered/Competent	10	1	0.62	0.62	0.29
MW-2011	Weathered/Competent	10	8	0.06	4.2	2.1
MW-2013	Weathered/Competent	11	11	1.2	65	20
MW-2015	Weathered/Competent	10	0	-	-	-
MW-2020	Weathered/Competent	10	0	-	-	-
MW-3003	Weathered/Competent	6	0	-	-	-
MW-3010	Weathered/Competent	9	0	-	-	-
MW-3019	Weathered/Competent	5	0	-	-	-
MW-4002	Weathered/Competent	8	5	0.57	2.7	1.1
MW-4003	Weathered/Competent	6	0	-	-	-
MW-4004	Weathered/Competent	6	0	-	-	-
MW-4005	Weathered/Competent	6	0	-	-	-
MW-4010	Weathered/Competent	6	0	-	-	-
MW-4013	Weathered/Competent	6	1	0.08	0.08	0.22
MW-4014	Weathered/Competent	6	1	0.06	0.06	0.22
MW-4015	Weathered/Competent	6	0	-	-	-
MW-4017	Weathered/Competent	6	0	-	-	-
MW-4018	Weathered/Competent	6	0	-	-	-
MW-4019	Weathered/Competent	9	0	-	-	-
MW-4021	Weathered/Competent	6	0	-	-	-
MW-4023	Weathered/Competent	6	0	-	-	-

[a] DL = detection limit.

B.5 REFERENCES (APPENDIX B)

Missouri Department of Natural Resources, 1991, *The Missouri Safe Drinking Water Act and the Missouri Public Drinking Water Regulations*, Public Drinking Water Program, Jefferson City, Mo.

MK-Ferguson Company, 1987, *Water Quality Phase I Assessment Report*, DOE/OR/21548-003, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Dec.

MK-Ferguson Company and Jacobs Engineering Group, 1989, *Phase II Groundwater Quality Assessment for the Weldon Spring Site, Chemical Plant, Raffinate Pits and Surrounding Vicinity Properties*, DOE/OR/21548-078, Rev. 0, prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, Weldon Spring, Mo., Aug.

MK-Ferguson Company and Jacobs Engineering Group, 1990, *Environmental Monitoring Plan*, DOE/OR/21548-103, Rev. 1 prepared for U.S. Department of Energy, Oak Ridge Operations Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Feb.

U.S. Department of Energy, 1992, *Remedial Investigation for the Chemical Plant Area of the Weldon Spring Site*, DOE/21549-074, Rev. 0, Vols. I-II, prepared by MK-Ferguson Company and Jacobs Engineering Group, St. Charles, Mo., for U.S. Department of Energy, Oak Ridge Field Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

U.S. Environmental Protection Agency, 1991, *Drinking Water Regulations and Health Advisories*, Office of Water, Washington, D.C., Nov.

APPENDIX C:

BIOTIC RESOURCES OF ST. CHARLES COUNTY AND THE BUSCH WILDLIFE COMPLEX

TABLE C.1 Mammals, Reptiles, and Amphibians Reported from St. Charles County

Mammals	Reptiles	Amphibians
Badger	Black rat snake	American toad
Beaver	Broadhead skink	Blanchard's cricket frog
Bobcat	Bullsnake	Bullfrog
Coyote	Common snapping turtle	Central newt
Deer mouse	Diamondback water snake	Cope's gray treefrog
Eastern cottontail rabbit	Eastern collared lizard	Dark-sided salamander
Eastern mole	Eastern garter snake	Eastern tiger salamander
Fox squirrel	Eastern hognose snake	Fowler's toad
Gray fox	Eastern massasauga rattlesnake	Gray treefrog
Gray squirrel	Eastern plains garter snake	Green frog
House mouse	Eastern spiny softshell	Marbled salamander
Least shrew	Eastern yellowbelly racer	Mudpuppy
Long-tailed weasel	False map turtle	Northern crawfish frog
Mink	Five-lined skink	Northern spring peeper
Muskrat	Graham's crayfish snake	Pickerel frog
Norway rat	Ground skink	Plains leopard frog
Plains pocket gopher	Map turtle	Plains spadefoot toad
Prairie vole	Midland brown snake	Ringed salamander
Raccoon	Midland smooth softshell	Slimy salamander
Red fox	Mississippi map turtle	Smallmouth salamander
Short-tailed shrew	Northern fence lizard	Southern leopard frog
Southern bog lemming	Northern lined snake	Spotted salamander
Southern flying squirrel	Northern water snake	Western chorus frog
Spotted skunk	Ornate box turtle	Western lesser siren
Striped skunk	Osage copperhead	Wood frog
Virginia opossum	Ouachita map turtle	
Woodland vole	Prairie kingsnake	
White-footed mouse	Prairie ringneck snake	
White-tailed deer	Red-eared slider	
	Red milk snake	
	Rough green snake	
	Southern coal skink	
	Southern redbelly snake	
	Speckled kingsnake	
	Stinkpot	
	Texas brown snake	
	Three-toed box turtle	
	Timber rattlesnake	
	Western earth snake	
	Western fox snake	
	Western painted turtle	
	Western ribbon snake	
	Western slender glass lizard	
	Western smooth green snake	
	Western spiny softshell	
	Western worm snake	
	Yellowbelly water snake	

Source: Dicknelte (1988).

TABLE C.2 Birds Reported from St. Charles County

Acadian flycatcher
Alder flycatcher
American avocet
American bittern
American black duck
American coot
American crow
American goldfinch
American kestrel
American redstart
American robin
American tree sparrow
American white pelican
American widgeon
American woodcock
Bachman's sparrow
Baird's sandpiper
Baird's sparrow
Bald eagle
Bank swallow
Barn owl
Barn swallow
Barred owl
Bay-breasted warbler
Bell's vireo
Belted kingfisher
Bewick's wren
Black scoter
Black tern
Black vulture
Black-and-white warbler
Black-bellied plover
Black-billed cuckoo
Black-capped chickadee
Black-crowned night heron
Black-throated blue warbler
Black-throated green warbler
Blackburnian warbler
Blackpoll warbler
Blue grosbeak
Blue jay
Blue-gray gnatcatcher
Blue-winged teal
Blue-winged warbler
Bobolink
Bohemian waxwing
Bonaparte's gull
Brewer's blackbird
Broad-winged hawk
Brown creeper
Brown thrasher
Brown-headed cowbird
Buff-breasted sandpiper
Bufflehead
Canada goose (migrant)
Canada goose (resident)
Canada warbler
Canvasback
Cape may warbler
Carolina chickadee
Carolina wren
Caspian tern
Cattle egret
Cedar waxwing
Cerulean warbler
Chestnut-sided warbler
Chimney swift
Chipping sparrow
Chuck-will's-widow
Cinnamon teal
Clay-colored sparrow
Cliff swallow
Common goldeneye
Common grackle
Common loon
Common merganser
Common moorhen
Common nighthawk
Common redpoll
Common snipe
Common tern
Common yellowthroat
Connecticut warbler
Cooper's hawk
Dark-eyed junco
Dickcissel
Double-crested cormorant
Downy woodpecker
Dunlin
Eastern bluebird
Eastern kingbird
Eastern meadowlark
Eastern phoebe
Eastern screech-owl
Eastern wood-pewee
Eurasian tree sparrow
European starling
Evening grosbeak
Field sparrow
Fish crow
Forster's tern
Fox sparrow
Franklin's gull
Gadwall
Glaucous gull
Glossy ibis
Golden eagle
Golden-crowned kinglet
Golden-winged warbler
Grasshopper sparrow
Gray catbird
Gray-cheeked thrush
Great blue heron
Great crested flycatcher
Great egret
Great horned owl
Greater scaup
Greater white-fronted goose
Greater yellowlegs
Green-backed heron
Green-winged teal
Hairy woodpecker
Harris' sparrow
Henslow's sparrow
Hermit thrush
Herring gull
Hooded merganser
Hooded warbler
Horned grebe
Horned lark
House finch
House sparrow
House wren
Hudsonian godwit
Iceland gull
Indigo bunting
Kentucky warbler
Killdeer
King rail
Lapland longspur
Lark sparrow
Least bittern
Least flycatcher
Least sandpiper
Least tern
Leconte's sparrow
Lesser golden plover
Lesser scaup
Lesser snow goose
Lesser yellowlegs

TABLE C.2 (Cont.)

Lincoln's sparrow	Red knot	Surf scoter
Little blue heron	Red-bellied woodpecker	Swainson's hawk
Loggerhead shrike	Red-breasted merganser	Swainson's thrush
Long-billed dowitcher	Red-breasted nuthatch	Swamp sparrow
Long-eared owl	Red-eyed vireo	Tennessee warbler
Louisiana waterthrush	Red-headed woodpecker	Tree swallow
Magnolia warbler	Red-necked grebe	Tufted titmouse
Mallard	Red-necked phalarope	Tundra swan
Marbled godwit	Red-shouldered hawk	Turkey vulture
Marsh wren	Red-tailed hawk	Upland sandpiper
Merlin	Red-winged blackbird	Veery
Mississippi kite	Redhead	Vesper sparrow
Mourning dove	Ring-billed gull	Virginia rail
Mourning warbler	Ring-necked duck	Warbling vireo
Nashville warbler	Ring-necked pheasant	Water pipit
Northern bobwhite	Rock dove	Western grebe
Northern cardinal	Rose-breasted grosbeak	Western kingbird
Northern flicker	Ross' goose	Western meadowlark
Northern goshawk	Rough-legged hawk	Western sandpiper
Northern harrier	Ruby-crowned kinglet	Whimbrel
Northern mockingbird	Ruby-throated hummingbird	Whip-poor-will
Northern oriole	Ruddy duck	White ibis
Northern parula	Ruddy turnstone	White-backed ibis
Northern pintail	Ruffed grouse	White-breasted nuthatch
Northern rough-winged swallow	Rufous-sided towhee	White-crowned sparrow
Northern saw-whet owl	Rusty blackbird	White-eyed vireo
Northern shoveler	Sanderling	White-rumped sandpiper
Northern waterthrush	Sandhill crane	White-throated sparrow
Oldsquaw	Savannah sparrow	White-winged crossbill
Olive-sided flycatcher	Scarlet tanager	White-winged scoter
Orange-crowned warbler	Scissor-tailed flycatcher	Wild turkey
Orchard oriole	Sedge wren	Willet
Osprey	Semipalmated plover	Willow flycatcher
Ovenbird	Semipalmated sandpiper	Wilson phalarope
Palm warbler	Sharp-shinned hawk	Wilson's warbler
Pectoral sandpiper	Sharp-tailed sparrow	Winter wren
Peregrine falcon	Short-billed dowitcher	Wood duck
Philadelphia vireo	Short-eared owl	Wood thrush
Pied-billed grebe	Smith's longspur	Worm-eating warbler
Pileated woodpecker	Snow bunting	Yellow warbler
Pine siskin	Snowy egret	Yellow-bellied flycatcher
Pine warbler	Snowy owl	Yellow-bellied sapsucker
Piping plover	Solitary sandpiper	Yellow-billed cuckoo
Prairie falcon	Solitary vireo	Yellow-breasted chat
Prairie warbler	Song sparrow	Yellow-crowned night heron
Prothonotary warbler	Sora	Yellow-headed blackbird
Purple finch	Spotted sandpiper	Yellow-rumped warbler
Purple gallinule	Sprague's pipit	Yellow-throated vireo
Purple martin	Stilt sandpiper	Yellow-throated warbler
Red crossbill	Summer tanager	

Source: Dickneite (1988).

TABLE C.3 Fish Occurring in St. Charles County

Alabama shad	Golden shiner	Shovelnose sturgeon
Alligator shad	Goldeye	Sicklefin chub
Banded sculpin	Grass carp	Silver chub
Bigeye shiner	Green sunfish	Silver lamprey
Bigmouth buffalo	Highfin carpsucker	Silver redhorse
Bigmouth shiner	Hornyhead chub	Silverband shiner
Black buffalo	Johnny darter	Skipjack herring
Black bullhead	Lake sturgeon	Slender madtom
Black crappie	Largemouth bass	Slenderhead darter
Black redhorse	Largescale stoneroller	Smallmouth bass
Blackspotted topminnow	Logperch	Smallmouth buffalo
Blackstripe topminnow	Longnose gar	Southern redbelly dace
Blue catfish	Mississippi silvery minnow	Speckled chub
Blue sucker	Mooneye	Spotfin shiner
Bluegill	Mosquitofish	Spottail shiner
Bluntnose minnow	Northern hog sucker	Spotted bass
Bowfin	Northern pike	Spotted sucker
Brook silverside	Northern studfish	Starhead topminnow
Brown bullhead	Orangespotted sunfish	Steelcolor shiner
Bullhead minnow	Orangethroat darter	Stonecat
Burbot	Paddlefish	Striped bass
Central stoneroller	Pallid sturgeon	Striped shiner
Channel catfish	Plains minnow	Sturgeon chub
Channel mimic shiner	Pugnose minnow	Suckermouth minnow
Chestnut lamprey	Quillback	Threadfin shad
Common carp	Red shiner	Walleye
Creek chub	Redear sunfish	Warmouth
Emerald shiner	Redfin shiner	Western sand darter
Fantail darter	River carpsucker	Western silvery minnow
Flathead catfish	River shiner	White bass
Flathead chub	Rosyface shiner	White bluegill
Freckled madtom	Sand shiner	White crappie
Freshwater drum	Sauger	White sucker
Ghost shiner	Shorthead redhorse	Yellow bass
Gizzard shad	Shortnose gar	Yellow bullhead
Golden redhorse		

Source: Dickneite (1988).

TABLE C.4 Multiple Species Fishing Waters of the Busch Wildlife Complex

Lakes[a]		Ponds	
Name	Surface Area (acres)	Name	Surface Area (acres)
Lake 1	1.0	5A	0.3
Lake 2	2.0	5B	0.2
Lake 3[b]	3.2	7A	0.3
Lake 4[b]	2.0	7B	0.2
Lake 5[b]	3.4	9A	1.5
Lake 6	18.0	9B	0.8
Lake 7[b]	12.0	10	0.3
Lake 8	15.8	11A	0.2
Lake 9	3.0	23A	0.3
Lake 10	3.5	23B	0.4
Lake 11	4.5	23C	0.3
Lake 12	4.0	26A	0.6
Lake 15	4.0	26B	0.8
Lake 16	8.8	26C	0.3
Lake 19	3.2	27	0.2
Lake 20	4.6	28	2.0
Lake 21	5.7	33A	1.0
Lake 22	4.1	33B	1.3
Lake 23	4.0	35A	1.0
Lake 24[b]	3.1	35B	0.9
Lake 26	1.0	36A	0.3
Lake 27	4.8	37A	0.8
Lake 28	12.0	37B	1.0
Lake 30	4.8	37C	0.6
Lake 31	20.0	37D	0.4
Lake 32	6.0	WSP-A	1.0
Lake 33	182.0	WSP-B	0.8
Lake 34	35.0	WSP-C	0.2
Lake 35	62.5		
Lake 36	15.5	Total acres	18.0
Lake 37	32.2		
Lake 38	12.0		
Prairie Lake	35.0		
Femme Osage Slough	15.7		
Total acres	548.2		

[a] Locations of the major lakes are shown in Chapter 1 (Figure 1.6); locations of all the lakes are shown in the RI report.

[b] Lakes 3, 4, 5, 7, and 24 (total surface area = 23.7 acres) are in the Busch Wildlife Area and are managed for channel catfish only.

Source: Missouri Department of Conservation (1989).

REFERENCES (APPENDIX C)

Dickneite, D.F., 1988, letter from D.F. Dickneite (Environmental Administrator, Missouri Department of Conservation, Jefferson City, Mo.) to I. Hlohowskyj (Argonne National Laboratory, Argonne, Ill.), with enclosure, Aug. 24.

Missouri Department of Conservation, 1989, *Area Management Plan for the August A. Busch Complex*, Missouri Department of Conservation, Jefferson City, Mo.

APPENDIX D:

LITERATURE REVIEW OF ECOLOGICAL IMPACTS ON SELECT BIOTA

TABLE D.1 Tissue Concentrations of Radionuclides Reported for Select Species of Plants

Contaminant/ Species	Environmental pH	Environmental Concentration	Tissue Concentration (pCi/g)				Bioconcentration Factor	Reference
			Whole Plant	Leaf	Stem	Root		
Radium-226								
Potamogeton sp.	5.8 to 8.7	5.4 to 44.8 pCi/L	30.6[a]	NM[b]	NM	NM	NM	Waite et al. (1988)
Myriophyllum sp.	5.8 to 8.7	5.4 to 44.8 pCi/L	45.2[a]	NM	NM	NM	NM	Waite et al. (1988)
Typha sp.	5.8 to 8.7	5.4 to 44.8 pCi/L	NM	NM	0.7[a]	10.1[a]	NM	Waite et al. (1988)
Potamogeton sp.	7.1 to 7.3	trace[c] (pCi/L)	0.2[a]	NM	NM	NM	NM	Waite et al. (1988)
Myriophyllum sp.	7.1 to 7.3	trace (pCi/L)	0.1[a]	NM	NM	NM	NM	Waite et al. (1988)
Water lily	NM	31.2 ± 11.6 mBq/L (water)	NM	13.2[d]	NM	47.4	NM	Twining (1989)
Water lily	NM	899 ± 12 mBq/g[d] (sediment)	NM	13.2[d]	NM	47.4[d]	NM	Twining (1989)
Water lily	NM	8.2 ± 1.4 mBq/g (water)	NM	5.3[d]	NM	10.9[d]	NM	Twining (1989)
Water lily	NM	41.7 ± 3.6 mBq/g[d] (sediment)	NM	5.3[d]	NM	10.9[d]	NM	Twining (1989)
Sagebrush	6.7	0.107 Bq/g	0.006[d]	NM	NM	NM	NM	Ibrahim and Whicker (1988)
Sagebrush	6.6	0.333 Bq/g	0.130[d]	NM	NM	NM	NM	Ibrahim and Whicker (1988)
Mixed forbs	6.7	0.107 Bq/g	0.011[d]	NM	NM	NM	NM	Ibrahim and Whicker (1988)
Mixed species[e]	6.7	0.107 Bq/g	0.008[d]	NM	NM	NM	0.07	Ibrahim and Whicker (1988)
Mixed species[e]	6.6	0.333 Bq/g	0.144[d]	NM	NM	NM	0.4	Ibrahim and Whicker (1988)

TABLE D.1 (Cont.)

Contaminant/ Species	Environmental pH	Environmental Concentration	Tissue Concentration (pCi/g)				Bioconcentration Factor	Reference
			Whole Plant	Leaf	Stem	Root		
Thorium-228								
Potamogeton sp.	5.8 to 8.7	trace[c] (pCi/L)	0.3[a]	NM	NM	NM	NM	Waite et al. (1988)
Myriophyllum sp.	5.8 to 8.7	trace (pCi/L)	0.5[a]	NM	NM	NM	NM	Waite et al. (1988)
Typha sp.	5.8 to 8.7	trace (pCi/L)	NM	NM	0.2[a]	0.2[a]	NM	Waite et al. (1988)
Potamogeton sp.	7.1 to 7.3	<0.3 pCi/L	0.03[a]	NM	NM	NM	NM	Waite et al. (1988)
Myriophyllum sp.	7.1 to 7.3	<0.3 pCi/L	0.1[a]	NM	NM	NM	NM	Waite et al. (1988)
Uranium, Total								
Potamogeton sp.	5.8 to 8.7	2.3 to 102 μg/L	2.6[a]	NM	NM	NM	NM	Waite et al. (1988)
Myriophyllum sp.	5.8 to 8.7	2.3 to 102 μg/L	3.4[a]	NM	NM	NM	NM	Waite et al. (1988)
Typha sp.	5.8 to 8.7	2.3 to 102 μg/L	NM	NM	0.1[a]	1.7[a]	NM	Waite et al. (1988)
Potamogeton sp.	7.1 to 7.3	0.2 μg/L	1.4[a]	NM	NM	NM	NM	Waite et al. (1988)
Myriophyllum sp.	7.1 to 7.3	0.2 μg/L	3.8[a]	NM	NM	NM	NM	Waite et al. (1988)
Typha sp.	7.1 to 7.3	0.2 μg/L	NM	NM	0.2[a]	0.2[a]	NM	Waite et al. (1988)

[a] Concentrations expressed as wet weight.

[b] NM = not measured or not reported.

[c] Detectable in some, but not all, samples.

[d] Concentrations expressed as dry weight.

[e] Desert herbs and shrubs.

Conversion factor: 1 becquerel (Bq) = 27 pCi; 1 mBq = 0.027 pCi.

TABLE D.2 Tissue Concentrations of Radionuclides Reported for Select Species of Fish

Contaminant/ Species	Environ-mental pH	Environmental Concentration	Tissue Concentration (pCi/g)					Bioconcen-tration Factor	Reference
			Whole Organism	Bone	Muscle	Liver	Gut Contents		
Radium-226									
Lake whitefish	5.8 to 8.7	5.4 to 44.8 pCi/L	NM[a]	4,990[b]	116[b]	89.9[b]	9,760[b]	NM	Waite et al. (1988)
Pike	5.8 to 8.7	5.4 to 44.8 pCi/L	NM	344[b]	28.5[b]	17.0[b]	75.3[b]	NM	Waite et al. (1988)
Lake whitefish	7.1 to 7.3	trace[c] (pCi/L)	NM	45.7[b]	37.3[b]	45.1[b]	85.6[b]	NM	Waite et al. (1988)
Pike	7.1 to 7.3	trace (pCi/L)	NM	72.7[b]	NM	17.6[b]	7.3[b]	NM	Waite et al. (1988)
Thorium-228									
Lake whitefish	5.8 to 8.7	<0.3 pCi/L	NM	48.7[b]	9.5[b]	7.2[b]	297[b]	NM	Waite et al. (1988)
Pike	5.8 to 8.7	<0.3 pCi/L	NM	7.3[b]	1.3[b]	1.3[b]	1.9[b]	NM	Waite et al. (1988)
Lake whitefish	7.1 to 7.3	<0.3 pCi/L	NM	<4.6[b]	5.8[b]	10.8[b]	111[b]	NM	Waite et al. (1988)
Pike	7.1 to 7.3	<0.3 pCi/L	NM	14.3[b]	NM	3.5[b]	7.4[b]	NM	Waite et al. (1988)
Uranium, Total									
Rainbow trout	5.1 to 8.6	420 µg/L	0.31 to 0.51[b]	NM	NM	NM	NM	NM	Nichols and Scholz (1989)
Rainbow trout	5.1 to 8.6	1.2 µg/L	0.04[b]	NM	NM	NM	NM	NM	Nichols and Scholz (1989)
Lake trout	NM	180 to 390 µg/L	5.8[d]	NM	NM	NM	NM	0.4	Swanson (1983)
Lake whitefish	NM	180 to 390 µg/L	28.2[d]	NM	NM	NM	NM	4	Swanson (1983)

TABLE D.2 (Cont.)

Contaminant/ Species	Environ-mental pH	Environmental Concentration	Tissue Concentration (pCi/g)					Bioconcen-tration Factor	Reference
			Whole Organism	Bone	Muscle	Liver	Gut Contents		
Uranium, Total (Cont.)									
White sucker	NM	180 to 390 µg/L	96.8[d]	NM	NM	NM	NM	17	Swanson (1983)
Lake whitefish	5.8 to 8.7	2.3 to 102 µg/L	NM	1.7[b]	0.5[b]	0.2[b]	7.5[b]	NM	Waite et al. (1988)
Pike	5.8 to 8.7	2.3 to 102 µg/L	NM	<0.1[b]	<0.6[b]	<0.3[b]	2.2[b]	NM	Waite et al. (1988)
Lake whitefish	7.1 to 7.3	0.2 µg/L	NM	0.2[b]	<0.1[b]	<0.1[b]	0.7[b]	NM	Waite et al. (1988)
Pike	7.1 to 7.3	0.2 µg/L	NM	<0.4[b]	NM	<0.1[b]	0.1[b]	NM	Waite et al. (1988)
Brook trout[e]	7.4	0 to 67 mg/L	1 to 16[b]	NM	NM	NM	NM	1.9 to 4.3	Parkhurst et al. (1984)
Brook trout[f]	NM	0.001 to 0.039 mg/L	NM	NM	0.00015 to 0.089[b,g]	NM	NM	0.08 to 5.9	Parkhurst et al. (1984)

[a] NM = not measured or not reported.

[b] Concentrations expressed as wet weight.

[c] Detectable in some, but not all, samples.

[d] Concentrations expressed as dry weight.

[e] Laboratory study; observed effects concentration = 59 mg/L (estimated 48-hour LC_{50}).

[f] Field study.

[g] Data include muscle and skin.

TABLE D.3 Lethal and Sublethal Effects of Metals on Select Biota

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Aluminum			
White clover	50 mg/kg[a] (as soil addition)	Reduced seedling establishment	Mackay et al. (1990)
	150 mg/kg[a] (as soil addition)	Severe reduction in shoot growth	Mackay et al. (1990)
Blueberry	Tissue concentrations up to 317 mg/kg[b]	Poor growth	Peterson et al. (1987)
Brook trout	300 µg/L	Reduced egg survival and hatchability	Ingersoll et al. (1990a)
Brook trout	34 µg/L	Reduced weight	Ingersoll et al. (1990b)
Leopard frog	250 µg/L	100% mortality of tadpoles	Freda and MacDonald (1989)
Antimony			
Freshwater algae	610 µg/L	Toxic effects[c]	EPA (1986)
Freshwater biota	9,000 µg/L	Acute toxicity water quality criterion[c]	EPA (1986)
Freshwater biota	1,600 µg/L	Chronic toxicity water quality criterion[c]	EPA (1986)
Arsenic			
Crops	25 to 85 mg total arsenic/kg of soil[a]	Seedling death, poor growth, defoliation, and inhibition of photosynthesis	Eisler (1988a)
Soil microbiota	375 mg total arsenic/kg of soil[a]	Reduced growth and metabolism in sensitive species	National Academy of Sciences (1977)
Earthworm	150 to 165 mg arsenic/kg of soil	Loss of earthworm biota	Eisler (1988a)
Turkey	Single oral dose, as 3-nitro-4-hydroxyphenyl-arsenic acid, of 17.4 mg/kg of body weight	LD_{50} dose	Eisler (1988a)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Arsenic (Cont.)			
Mallard	323 mg sodium arsenite/kg of body weight	LD_{50} acute oral value	Eisler (1988a)
Domestic goat	Single oral dose of 2.5 to 7.5 mg arsenic acid/kg of body weight	Acute toxicity	Eisler (1988a)
White-tailed deer	34 mg sodium arsenite/kg of body weight	Lethal dose	Eisler (1988a)
Freshwater biota	360 µg/L trivalent arsenic	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	190 µg/L trivalent arsenic	Chronic toxicity water quality criterion	EPA (1986)
Freshwater biota	850 µg/L pentavalent arsenic	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	48 µg/L pentavalent arsenic	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	20 µg/L	Chronic toxicity water quality criterion[d]	Missouri Department of Natural Resources (1992)
Barium			
Freshwater biota	>50,000 µg/L	No toxic effects anticipated at concentrations below 50,000 µg/L	EPA (1986)
Beryllium			
Vegetation	1 to 2 mg/L in nutrient solution	Toxic to plants grown in acidic soils	Peterson and Girling (1981)
Freshwater biota	130 µg/L	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	5.3 µg/L	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	5 µg/L	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Cadmium			
Crops	4 to 640 µg/g of soil	25% reduction in yield	Page et al. (1981)
Rat	Oral dose of 250 mg cadmium fluoroborate/kg of body weight	Death	Eisler (1985a)
Guinea pig	Oral dose of 150 mg cadmium fluoride/kg of body weight	Death	Eisler (1985a)
Mallard	Dietary concentration of 20 mg/kg of body weight	Altered blood chemistry, mild to severe kidney lesions	Cain et al. (1983)
Black duck	Dietary concentration of 4 mg/kg of body weight	Hyperresponsiveness	Heinz and Haseltine (1983)
White mouse	Dietary concentration of 1.8 mg/kg of body weight	Depressed hematocrit and hemoglobin values and altered blood chemistry	Eisler (1985a)
Freshwater biota	3.9 µg/L (hardness 200)	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	1.1 µg/L (hardness 200)	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	52 µg/L (hardness 125-200)	Acute toxicity water quality criterion[d]	Missouri Department of Natural Resources (1992)
Aquatic life	13 µg/L (hardness 125-200)	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Chromium			
Vegetation	Nutrient solutions of 5 mg chromium VI/L	Decreased chlorophyll concentrations, inhibition of seed germination and growth, decreased root nutrient uptake	World Health Organization (1988)
Black duck	10 mg anionic chromium III/kg of body weight	Altered growth patterns and reduced survival in ducklings	Eisler (1986a)
Golden hamster	5 mg chromium VI/kg of body weight, injection	Skeletal defects, cleft palates	Eisler (1986a)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Chromium (Cont.)			
Freshwater biota	16 µg chromium VI/L	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	11 µg chromium VI/L	Chronic toxicity water quality criterion	EPA (1986)
Freshwater biota	3,100 µg chromium III/L (hardness 200)	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	370 µg chromium III/L (hardness 200)	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	62 µg/L	Acute toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Aquatic life	42 µg/L	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Cobalt			
Vegetation	11 mg cobalt/kg of leaf tissue	Growth depression and chlorosis	Peterson and Girling (1981)
Copper			
Soil microbes	Up to 460 mg/kg of soil	Reduced abundance and biomass; decreased species diversity, density, and biomass	Tyler et al. (1989)
Vegetation	50 to 100 µg/g of soil	Soil concentrations toxic to nontolerant plants	Tyler et al. (1989)
Freshwater biota	34 µg/L (hardness 200)	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	21.4 µg/L (hardness 200)	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	45 µg/L (hardness 125-200)	Acute toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Aquatic life	29 µg/L (hardness 125-200)	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Lead			
Vegetation	>200 mg/kg of tissue	Growth inhibition; reduced photosynthesis, water absorption, and mitosis	Demayo et al. (1982)
Vegetation	500 mg/kg of soil[b]	Reduced pollen and seed germination	Eisler (1988b)
Soil invertebrate communities	34 mg/kg of soil	Decreased species diversity	Tyler et al. (1989)
Soil invertebrate communities	34 to 4,800 mg/kg of soil	Altered biomass, density, species number, and vertical distributions in individual soil groups such as earthworms and ants	Tyler et al. (1989)
Leopard frog	25 mg/L	Death, elevated liver residues	Eisler (1988b)
Leopard frog	105 mg/L	30-day LC_{50}	Eisler (1988b)
Marbled salamander	1.4 mg/L	Death following 8 days of exposure	Eisler (1988b)
Mallard	Single oral dose of 151 mg/kg of body weight	Some death, altered blood chemistry	Eisler (1988b)
Japanese quail	Single oral dose of tetra-ethyl lead, 24.6 mg/kg of body weight	LD_{50} concentration	Eisler (1988b)
Mouse (*Mus* sp.)	Daily dietary dose of 0.05 mg/kg of body weight	Inhibition of some blood and bone marrow enzyme systems	Eisler (1988b)
Freshwater biota	200 µg/L (hardness 200)	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	7.7 µg/L (hardness 200)	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	130 µg/L (hardness 125-200)	Acute toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Aquatic life	20 µg/L (hardness 125-200)	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Manganese			
Apple	60 μg/g[b] of leaf tissue	Leaf toxicity	Neilsen et al. (1990)
Apple	5,000 μg/g[b] of leaf tissue	Leaf toxicity	Marsh and Peterson (1990)
Freshwater biota	1.5 mg/L	Toxicity to sensitive species[c]	EPA (1986)
Vegetation	>1 mg/L (irrigation water)	Toxic (if pH <6)[c]	EPA (1986)
Mercury			
Mallard duck	2.2 to 23.5 mg/kg (oral dose)	LD_{50}	Eisler (1987a)
House sparrow	12.6 to 37.8 mg/kg (oral dose)	LD_{50}	Eisler (1987a)
Pheasant	Dietary concentration of 12.5 mg/kg	70-day LD_{50}	Eisler (1987a)
Mink	Dietary concentration of 0.1 mg/kg	Fatal	Eisler (1987a)
River otter	Dietary concentration of >2.0 mg/kg	Fatal	Eisler (1987a)
Freshwater biota	0.1 to 2.0 μg/L	Fatal to sensitive species	Eisler (1987a)
Freshwater biota	0.03 to 0.1 μg/L	Sublethal effects	Eisler (1987a)
Aquatic life	2.4 μg/L	Acute toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Aquatic life	0.5 μg/L	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Nickel			
Freshwater biota	2,500 μg/L (hardness 200)	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	280 μg/L (hardness 200)	Chronic toxicity water quality criterion	EPA (1986)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Nickel (Cont.)			
Aquatic life	4,600 µg/L (hardness 125-200)	Acute toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Aquatic life	500 µg/L (hardness 125-200)	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Selenium			
Mallard duck	Dietary concentration of 100 mg/kg	Lethal within 1 month	Eisler (1985b)
Mallard duck	Dietary concentration of 10 mg/kg	Poor hatching success	Eisler (1985b)
Waterfowl	>2 to 5 µg/L	Reproductive failure or mortality	Lemly and Smith (1987)
Livestock	400 to 800 mg/kg of feed	Acute poisoning	Eisler (1985b)
Freshwater biota	260 µg/L	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	35 µg/L	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	10 µg/L	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Silver			
Maize	180 mg/kg[b]	Reduced seedling root growth	Peterson and Girling (1981)
Barley	180 mg/kg[b]	Reduced seedling root growth	Peterson and Girling (1981)
Maize	>60 mg/kg[b]	Reduced shoot growth	Peterson and Girling (1981)
Barley	>60 mg/kg[b]	Reduced shoot growth	Peterson and Girling (1981)
Freshwater biota	13 µg/L (hardness 200)	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	0.12 µg/L (hardness 200)	Chronic toxicity water quality criterion	EPA (1986)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Silver (Cont.)			
Aquatic life	8.2 µg/L (hardness 125-200)	Acute toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Aquatic life	0.12 µg/L (hardness 125-200)	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Thallium			
Freshwater biota	1,400 µg/L	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	40 µg/L	Chronic toxicity water quality criterion	EPA (1986)
Rat	10 to 25 mg/kg of body weight, oral dose	LD_{50}	Kemper and Bertram (1991)
Mouse	50 to 60 mg/kg of body weight, oral dose	LD_{100}	Kemper and Bertram (1991)
Guinea pig	20 to 80 mg/kg of body weight, oral dose	LD_{100}	Kemper and Bertram (1991)
Uranium			
Bacteria	1 to 100 ppm	Prolonged lag phase	Norberg and Molin (1983)
Bacteria	1,000 ppm	Growth inhibition	Norberg and Molin (1983)
Soybeans	0.42 to 42 g/L	Reduced seedling growth, necrosis, chlorosis	Murthy et al. (1984)
Daphnia magna	6 mg/L	48-hour LC_{50}	Poston et al. (1984)
Daphnia magna	0.57 to 2.25 mg/L	Impaired reproduction	Poston et al. (1984)
Vanadium			
Vegetation	6 to 100 mg/L (culture solution)	Toxicity	Peterson and Girling (1981); Kaplan et al. (1990)
Mouse	25 to 30 mg/kg (ammonium metavanadate)	Lethal	Zenz (1980)
Mouse	125 to 150 mg/kg (vanadryl sulfate)	Lethal	Zenz (1980)

TABLE D.3 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Zinc			
Forest understory vegetation	18 mg/L (soil moisture)	Damage	Tyler et al. (1989)
Vegetation	0.02 to 0.1 mg/L	Cytological changes, reduced root growth, decreased growth	Tyler et al. (1989)
Soil invertebrates	>170 mg/kg	Reduced density, biomass, diversity	Tyler et al. (1989)
Ceriodaphnia reticulate	50.7 µg/L	Acute sensitivity	EPA (1986)
Damselfly	88,960 µg/L	Acute sensitivity	EPA (1986)
Daphnia magna	46.73 µg/L	Chronic toxicity[c]	EPA (1986)
Caddis fly	>5,243 µg/L	Chronic toxicity	EPA (1986)
Flagfish	36.41 µg/L	Chronic toxicity	EPA (1986)
Brook trout	854.7 µg/L	Chronic toxicity.	EPA (1986)
Selenastrum capriocornutum (alga)	30 µg/L	Growth inhibition[c]	EPA (1986)
Freshwater biota	210 µg/L	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	190 µg/L	Chronic toxicity water quality criterion	EPA (1986)
Aquatic life	380 µg/L (hardness 125-200)	Acute toxicity water quality criterion	Missouri Department of Natural Resources (1992)
Aquatic life	345 µg/L (hardness 125-200)	Chronic toxicity water quality criterion	Missouri Department of Natural Resources (1992)

[a] Concentration expressed as wet weight.

[b] Concentration expressed as dry weight.

[c] Environmental concentrations exceeding EPA criterion concentration have been reported to induce a variety of adverse effects.

[d] Environmental concentrations exceeding state of Missouri criterion concentration have been reported to induce a variety of adverse effects.

TABLE D.4 Lethal and Sublethal Effects of Inorganic Anions on Select Biota

Anion/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Nitrate			
Bluegill	420 mg/L	96-hour LC_{50}	EPA (1986)
Rainbow trout	1,310 mg/L	96-hour LC_{50}	EPA (1986)
Chinook salmon	1,360 mg/L	96-hour LC_{50}	EPA (1986)
Largemouth bass	90 mg/L	None	EPA (1986)
Channel catfish	90 mg/L	None	EPA (1986)
Nitrite			
Chinook salmon	0.9 mg/L	96-hour LC_{50}	EPA (1986)
Salmonids	0.55 to 1.6 mg/L	Increased mortality	EPA (1986)
Fluoride			
Hydropsyche bronta	17 mg/L	96-hour LC_{50}	Camargo et al. (1992)
Hydropsyche occidentalis	34.7 mg/L	96-hour LC_{50}	Camargo et al. (1992)
Cheumatopsyche pettiti	42.5 mg/L	96-hour LC_{50}	Camargo et al. (1992)
Rainbow trout	140 mg/L	96-hour LC_{50}, hardness = 182 mg/L $CaCO_3$	Pimentel and Bulkley (1983)

TABLE D.5 Lethal and Sublethal Effects of Polycyclic Aromatic Hydrocarbons and Polychlorinated Biphenyls on Select Biota

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
Polycyclic aromatic hydrocarbons			
Laboratory rodents	50 to 2,000 mg/kg (oral dose)	LD_{50}	Eisler (1987b)
Polychlorinated biphenyls			
Shellfish	0.006 µg/L (in water)	Measurable accumulation	Eisler (1986b)
Lake trout	15 to 10,000 ng/g of muscle tissue	NM[a]	Rasmussen et al. (1990)
Forster's tern	50 mg/kg	Reproductive problem	Noble and Elliott (1990)
Peregrine falcon	300 mg/kg of brain tissue	Mortality	Peakall et al. (1990)
Peregrine falcon	40 mg/kg of egg	Reproductive problems	Peakall et al. (1990)
Red-winged blackbird	1,500 mg/kg of diet	LD_{50}	Eisler (1986b)
Common grackle	1,500 mg/kg of diet	LD_{50}	Eisler (1986b)
Cowbird	1,500 mg/kg of diet	LD_{50}	Eisler (1986b)
Ostrich fern	26 mg/kg of soil	Somatic mutations	Eisler (1986b)
Freshwater biota	2 µg/L	Acute toxicity water quality criterion	EPA (1986)
Freshwater biota	0.014 µg/L	Chronic toxicity water quality criterion	EPA (1986)

[a] NM = not measured or not reported.

TABLE D.6 Lethal and Sublethal Effects of Nitroaromatic Compounds on Select Biota

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
1,3-Dinitrobenzene			
Daphnia magna	42.5 to 59.2 mg/L	48-hour EC_{50}	Liu et al. (1983)
Fathead minnow	5.8 to 8.1 mg/L	96-hour LC_{50}	Liu et al. (1983)
Fathead minnow	7.0 mg/L	96-hour LC_{50}	Bailey and Spanggord (1983)
2,4-Dinitrotoluene and 2,6-Dinitrotoluene			
Daphnia magna	29.5 to 99.7 mg/L	48-hour EC_{50} (2,4-DNT)	Liu et al. (1983)
Daphnia magna	19.3 to 24.6 mg/L	48-hour EC_{50} (2,6-DNT)	Liu et al. (1983)
Fathead minnow	27 to 38 mg/L	96-hour LC_{50} (2,4-DNT)	Bailey and Spanggord (1983)
Fathead minnow	17.2 to 20.2 mg/L	96-hour LC_{50} (2,6-DNT)	Bailey and Spanggord (1983)
Freshwater biota	330 µg/L	Acute toxicity water quality criterion (DNT)	EPA (1986)
Freshwater biota	230 µg/L	Chronic toxicity water quality criterion (DNT)	EPA (1986)
Nitrobenzene			
Soybeans	0.02 to 0.2 mg/L in hydroponic solution	None	Fletcher et al. (1990)
Soybeans	2.0 to 100 mg/L in hydroponic solution	Reduced root growth	Fletcher et al. (1990)
Autumn olive	8 mg/L	Death	McFarlane and Pfleeger (1990)
Russian olive	8 mg/L	Reduced transpiration and photosynthesis	McFarlane and Pfleeger (1990)
Freshwater biota	27 mg/L	Acute toxicity water quality criterion	EPA (1986)

TABLE D.6 (Cont.)

Contaminant/ Species or Biotic Group	Contaminant Concentration	Observed Effect	Reference
1,3,5-Trinitrobenzene			
Daphnia magna	2.4 to 3.1 mg/L	48-hour EC_{50}	Liu et al. (1983)
Fathead minnow	1.0 to 1.2 mg/L	96-hour LC_{50}	Bailey and Spanggord (1983)
2,4,6-Trinitrotoluene			
Daphnia magna	11.1 to 12.8 mg/L	48-hour EC_{50}	Liu et al. (1983)
Fathead minnow	2.8 to 3.3 mg/L	96-hour LC_{50}	Bailey and Spanggord (1983)

REFERENCES (APPENDIX D)

Bailey, H.C., and R.J. Spanggord, 1983, *The Relationship between the Toxicity and Structure of Nitroaromatic Chemicals*, in Aquatic Toxicology and Hazard Assessment: Sixth Symposium, ASTM Special Technical Publication 802, W.E. Bishop, R.D. Cardwell, and B.B. Heidolph (editors), American Society For Testing and Materials, Philadelphia, pp. 98-107.

Cain, B.W., et al., 1983, *Effects of Dietary Cadmium on Mallard Ducklings*, Environmental Research, 32:286-297.

Camargo, J.A., J.V. Ward, and K.L. Martin, 1992, *The Relative Sensitivity of Competing Hydropsychid Species to Fluoride Toxicity in the Cache la Poudre River (Colorado)*, Archives of Environmental Contamination and Toxicology, 22:107-113.

Demayo, A., et al., 1982, *Toxic Effects of Lead and Lead Compounds on Human Health, Aquatic Life, Wildlife, Plants, and Livestock*, CRC Critical Reviews of Environmental Control, 12:257-305.

Eisler, R., 1985a, *Cadmium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 2, Biological Report 85 (1.2), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., July.

Eisler, R. 1985b, *Selenium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 5, Biological Report 85(1.5), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., Oct.

Eisler, R., 1986a, *Chromium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 6, Biological Report 85(1.6), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., Jan.

Eisler, R., 1986b, *Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 7, Biological Report 85(1.7), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., April.

Eisler, R., 1987a, *Mercury Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 10, Biological Report 85 (1.10), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., April.

Eisler, R., 1987b, *Polycyclic Aromatic Hydrocarbon Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 11, Biological Report 85(1.11), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., May.

Eisler, R., 1988a, *Arsenic Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 12, Biological Report 85(1.12), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., Jan.

Eisler, R., 1988b, *Lead Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, Contaminant Hazard Reviews Report No. 14, Biological Report 85(1.14), U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md., April.

EPA: see U.S. Environmental Protection Agency.

Fletcher, J.S., et al., 1990, *Influence of Root Exposure Concentration on the Fate of Nitrobenzene in Soybean*, Chemosphere, 20:513-523.

Freda, J., and D.G. MacDonald, 1989, *Effects of Aluminum on the Leopard Frog, Rana pipiens: Life Stage Comparisons and Aluminum Uptake*, Canadian Journal of Fisheries and Aquatic Science, 47:210-216.

Heinz, G.H., and S.D. Haseltine, 1983, *Altered Avoidance Behavior of Young Black Ducks Fed Cadmium*, Environmental Toxicology and Chemistry, 2:419-421.

Ibrahim, S.A., and F.W. Whicker, 1988, *Plant/Soil Concentration Ratios of ^{226}Ra for Contrasting Sites around an Active U Mine-Mill*, Health Physics, 55:903-910.

Ingersoll, C.G., et al., 1990a, *Effects of pH, Aluminum, and Calcium on Survival and Growth of Eggs and Fry of Brook Tout (Salvelinus fontinalis)*, Canadian Journal of Fisheries and Aquatic Science, 47:1580-1592.

Ingersoll, C.G., et al., 1990b, *Aluminum and Acid Toxicity to Two Strains of Brook Trout (Salvelinus fontinalis)*, Canadian Journal of Fisheries and Aquatic Science, 47:1641-1648.

Kaplan, D.I., et al., 1990, *Vanadium: Toxicity and Accumulation by Beans*, Water, Air, and Soil Pollution, 49:81-91.

Kemper, F.H., and H.P. Bertram, 1991, *Thallium*, in Metals and Their Compounds in the Environment: Occurrence, Analysis, and Biological Relevance, E. Merian (editor), VCH Publications, New York, pp. 1227-1241.

Lemly, A.D., and G.J. Smith, 1987, *Aquatic Cycling of Selenium: Implications for Fish and Wildlife*, Fish and Wildlife Leaflet 12, U.S. Fish and Wildlife Service, Washington, D.C.

Liu, D.H., H.C. Bailey, and J.G. Pearson, 1983, *Toxicity of a Complex Munitions Wastewater to Aquatic Organisms*, in Aquatic Toxicology and Hazard Assessment: Sixth Symposium, ASTM Special Technical Publication 802, W.E. Bishop, R.D. Cardwell, and B.B. Heidolph (editors), American Society for Testing and Materials, Philadelphia, pp. 135-150.

Mackay, A.D., J.R. Caradus, and M.W. Pritchard, 1990, *Variation for Aluminum Tolerance in White Clover*, Plant and Soil, 123:101-105.

Marsh, K.B., and L.A. Peterson, 1990, *Gradients in Mn Accumulation and Changes in Plant Form for Potato Plants Affected by Mn Toxicity*, Plant and Soil, 121:157-163.

McFarlane, C., and T. Pfleeger, 1990, *Effect, Uptake and Disposition of Nitrobenzene in Several Terrestrial Plants*, Environmental Toxicology and Chemistry, 9:513-520.

Missouri Department of Natural Resources, 1992, *Rules of Department of Natural Resources: Division 20 — Clean Water Commission; Chapter 7 — Water Quality*, Code of State Regulations 10 CSR 20-7.031 (Water Quality Standards), Jefferson City, Mo.

Murthy, T.C.S., P. Weinberger, and M.P. Measures, 1984, *Uranium Effects on the Growth of Soybean (Glycine max (L.) Merr.)*, Bulletin of Environmental Contamination and Toxicology, 32:580-586.

National Academy of Sciences, 1977, *Arsenic*, Washington, D.C., 332 pp.

Neilsen, D., et al., 1990, *Manganese Soil Tests for Both Deficient and Toxic Levels in Apple Orchards*, Canadian Journal of Soil Science, 70:503-507.

Nichols, D.G., and A.T. Scholz, 1989, *Concentrations of Cd, Sr, and U in Fish and Water Samples Collected from a Small Stream Receiving Uranium Mine Drainage*, Journal of Freshwater Ecology, 5: 13-25.

Noble, D.G., and J.E. Elliot, 1990, *Level of Contaminants in Canadian Raptors, 1966 to 1988; Effects and Temporal Trends*, Canadian Field-Naturalist, 104:222-243.

Norberg, A.B., and N. Molin, 1983, *Toxicity of Cadmium, Cobalt, Uranium and Zinc to Zoogloea ramigera*, Water Resources, 17:1333-1336.

Page, A.L., F.T. Bingham, and A.C. Chang, 1981, *Cadmium*, in Effects of Heavy Metal Pollution on Plants, Vol. 1, Effects of Trace Metals on Plant Function, N.W. Lepp (editor), Pollution Monitoring Series, Applied Science Publishers Ltd., Englewood, N.J., pp. 77-109.

Parkhurst, B.J., et al., 1984, *An Environmental Hazard Evaluation of Uranium in a Rocky Mountain Stream*, Environmental Toxicology and Chemistry, 3:113-124.

Peakall, D.B., et al., 1990, *Environmental Contaminants in Canadian Peregrine Falcons, Falco peregrinus: A Toxicological Assessment*, Canadian Field-Naturalist, 104:244-254.

Peterson, D.V., et al., 1987, *Effects of Soil-Applied Elemental Sulfur, Aluminum Sulfate, and Sawdust on Growth of Rabbiteye Blueberries*, Journal of the American Society of Horticultural Science, 112:612-616.

Peterson, P.J., and C.A. Girling, 1981, *Other Trace Metals*, in Effects of Heavy Metal Pollution on Plants, Vol. 1, Effects of Trace Metals on Plant Function, N.W. Lepp (editor), Pollution Monitoring Series, Applied Science Publishers Ltd., Englewood, N.J., pp. 213-278.

Pimentel, R., and R.V. Bulkley, 1983, *Influence of Water Hardness on Fluoride Toxicity to Rainbow Trout*, Environmental Toxicology and Chemistry, 2:381-386.

Poston, T.M., R.W. Hanf, Jr., and M.A. Simmons, 1984, *Toxicity of Uranium to Daphnia magna*, Water, Air, and Soil Pollution, 22:289-298.

Rasmussen, J.B., et al., 1990, *Food Chain Structure in Ontario Lakes Determines PCB Levels in Lake Trout (Salvelinus namaycush) and Other Pelagic Fish*, Canadian Journal of Fisheries and Aquatic Science, 47:2030-2038.

Swanson, S.M., 1983, *Levels of ^{226}Ra, ^{210}Pb, and $^{Total}U$ in Fish near a Saskatchewan Uranium Mine and Mill*, Health Physics, 45:67-80.

Twining, J.R., 1989, *Principal Coordinate Analysis of the Distribution of Radium-226 between Water, Sediment and the Waterlily, Nymphaea violacea (Lehm), in the Vicinity of a Uranium Mine in the Northern Territory, Australia*, Journal of Environmental Radioactivity, 10:99-113.

Tyler, G., et al., 1989, *Heavy Metal Ecology of Terrestrial Plants, Microorganisms, and Invertebrates*, Water, Air, and Soil Pollution, 47:189-215.

U.S. Environmental Protection Agency, 1986, *Quality Criteria for Water 1986*, EPA 440/5-86-001, Office of Water Regulations and Standards, Washington, D.C., May.

Waite, D.T., S.R. Joshi, and H. Sommerstad, 1988, *The Effect of Uranium Mine Tailings on Radionuclide Concentrations in Langley Bay, Saskatchewan, Canada*, Archives of Environmental Contamination and Toxicology, 17:373-380.

World Health Organization, 1988, *Chromium*, Environmental Health Criteria 61, International Programme on Chemical Safety, Geneva, 200 pp.

Zenz, C., 1980, *Vanadium*, in Metals in the Environment, H.A. Waldron (editor), Academic Press, New York, pp. 293-327.

APPENDIX E:

LITERATURE REVIEW FOR CONTAMINANT MOBILITY AND AVAILABILITY, PLANT UPTAKE, AND ANIMAL BIOTRANSFER

APPENDIX E:

LITERATURE REVIEW FOR CONTAMINANT MOBILITY AND AVAILABILITY, PLANT UPTAKE, AND ANIMAL BIOTRANSFER

To quantify potential human exposures from game ingestion associated with the Weldon Spring site, the scientific literature was reviewed to assess the mobility and availability of site contaminants and to estimate both plant uptake from soil and animal biotransfer from dietary intake for the contaminants of concern. Plant uptake and animal biotransfer were then modeled (Section 3.3.5.2) to estimate exposure point concentrations for human intake calculations.

Geochemical data from the literature were reviewed to identify the primary factors affecting the mobility and availability of site contaminants. Contaminant-specific distribution coefficient (K_d) values, which relate the concentration in soil (solid phase) to the concentration in soil solution, are a measure of mobility that can be used to assess bioavailability. A high K_d value indicates that the contaminant is strongly adsorbed to soil and little would be available in the soil solution for plant uptake. The literature contains general information on mobility for the contaminants of concern, but the range of values for individual contaminants varies widely and key information is typically lacking. For example, factors such as solution pH and particle size distribution that play a major role in the K_d estimate are often unspecified. Nevertheless, information on mobility and availability is summarized for radionuclides, metals, inorganic anions, polycyclic aromatic hydrocarbons, polychlorinated biphenyls, and nitroaromatic compounds at the site in Sections E.1.1 through E.1.6. The contaminant-specific K_d values used for site-specific analyses — e.g., to derive plant uptake and animal biotransfer factors for radionuclides, lead, and the organic compounds from the methods described in Sections 3.5.2.2 and E.2.1.2 — are summarized in Section E.1.7. These values were also used to evaluate potential leaching of contaminants from soil to groundwater in Appendix E of the feasibility study (FS) for the chemical plant area of the Weldon Spring site (DOE 1992a).

Although considerable data are available in the literature on contaminant solubility or mobility, much less information is available for plant uptake and animal transfer factors. In addition, these data do not address site-specific conditions. For example, factors such as soil type, pH, contaminant concentrations, the presence of other contaminants, and the types of plants and animals present will strongly influence contaminant bioavailability and plant uptake. Further, although some limited data are available on absorption and excretion of contaminants following ingestion by laboratory animals, pharmacokinetic models are not available to assess the distribution within game animals of each retained contaminant evaluated in this assessment. In fact, experimental data indicate that the uptake factors for a single contaminant can easily vary by a factor of 100 between species and life stages, probably because of changes in metabolic rates and fat content over an animal's life. Therefore, the factors used for this assessment can only be considered relative indications of potential uptake and biotransfer.

The limited information available for absorption and excretion was considered together with information extrapolated from the data available for certain contaminants (e.g., from

radioactive fallout studies) to assess transfer factors for animals. The factors given for radionuclides and metals in this appendix generally represent data for cows (beef). The values in this appendix should be considered only as screening-level estimates.

Information from the literature on common soil and plant concentrations, biomagnification potential, animal absorption and excretion data, and dietary concentrations for plant and animal foodstuffs (where available) is presented in Sections E.2.1 through E.2.6. These data were considered in determining plant uptake and animal transfer factors for each contaminant, in combination with site-specific factors affecting potential availability, as presented in the fate and transport discussions of Sections 3.2.2.2 and E.1. The contaminant-specific factors used to model contaminant concentrations in plants and animals for the food ingestion pathway are summarized in Section E.2.7. (Data for dairy products are also included in this discussion to support the farmer analysis in Appendix E of the FS.)

Deriving plant uptake and animal transfer factors from very limited information leads to large uncertainty in the modeling results. As one measure of this uncertainty, the model was applied to off-site background concentrations to indicate relative impacts associated with ingesting food (game) from the site (off-site results are generally similar, Section 5.4). Thus, in modeling this pathway for completeness relative to potential multiple exposure pathways at the site, it must be recognized that the screening-level results are very much constrained by limitations in the available data and estimation methodologies. With this understanding, the model results were used to qualitatively assess the role of exposures via game ingestion relative to overall exposures to soil and air at the site in this assessment.

E.1 CONTAMINANT MOBILITY AND AVAILABILITY

E.1.1 Radionuclides

E.1.1.1 Actinium and Protactinium

No information was found in the literature for actinium or protactinium relative to soil chemistry or K_d values. It has been suggested that the K_d values for protactinium would be similar to those for thorium or uranium (Sheppard et al. 1984). For actinium, Sheppard et al. (1984) indicate that the K_d value could be assumed to be the same as that identified for americium, on the basis of chemical similarity. Soil sorption of americium is correlated to the cation exchange capacity and clay content of the soil and the concentration and type of the competing ions in solution, which indicates that the principal retardation mechanism is ion exchange. The K_d values reported for americium in a clay soil range from 25.1 to 400,000 mL/g (Sheppard et al. 1984). General K_d values of 1,500 and 2,500 mL/g have been reported for actinium and protactinium, respectively, by Baes et al. (1984). Sheppard and Thibault (1990) identify values of 1,500 and 2,400 mL/g for actinium in loam and clay soil, respectively. For this assessment, K_d values of 794 and 330 mL/g determined from Sheppard et al. (1984) were used for actinium and protactinium.

E.1.1.2 Radium

Radium in soil is commonly adsorbed onto clay particles and silica (Nathwani and Phillips 1979). Radium is also attenuated by coprecipitation with iron and manganese oxyhydroxides, which are common in soil. Therefore, attenuation is likely in each overburden layer at the site. Together with clay content, the presence of organic matter in soil dominates radium adsorption processes (Nathwani and Phillips 1980). Although some organic complex formation has been indicated for radium in alkaline solutions, this radionuclide is generally immobile at a solution pH greater than 6 (Shepherd and Cherry 1980). Radium can coprecipitate with barium sulfate, and in locations with high sulfate concentrations (such as the bottom of the raffinate pits), it can substitute for calcium in gypsum ($CaSO_4 \cdot 2H_2O$) to form a mixed calcium-radium solid.

Adsorption of radium may not be fully reversible, but it decreases with decreasing pH as is common for divalent metals (Benes et al. 1984, 1985). Radium hydroxide is the most soluble of the alkaline earth hydroxides (2.1 mg/L at 20°C) (Sedlet 1966), and radium can be leached from soil by rainwater in some geochemical systems (Nathwani and Phillips 1978). However, radium is only slightly water soluble in the presence of carbonate materials (Sedlet 1966), which are common in the overburden and bedrock at the site. Estimated K_d values for radium in clay minerals at alkaline solution pH range from about 3,000 to 90,000 mL/g (Benes et al. 1985, 1986), whereas those for soil in alkaline solution range from about 200 to 500 mL/g (Ames and Rai 1978); Baes et al. (1984) identify a value of 450 mL/g. The K_d value of 165 mL/g that has been estimated for radium at the Weldon Spring site (Seeley and Kelmers 1985) was used for calculations in this assessment.

E.1.1.3 Thorium

Thorium is present in nature in the tetravalent form, and it is quite insoluble. Its adsorption onto clays and oxides is essentially complete in near-neutral soil (pH 6.5) (Ames and Rai 1978). Thorium is typically associated with the clay fraction. Soluble thorium ions that may be present in low pH conditions would hydrolyze at pH values above 5 to form hydroxy precipitates or complexes; these complexes would be adsorbed on particulates such as goethite (Milic and Suranji 1982; Bodek et al. 1988; Hunter et al. 1988). Thorium can also form soluble complexes with carbonate and humic material in certain waters; however, it is strongly sorbed in most soil systems and its mobility is generally very low (Torstenfelt 1986).

The K_d values reported for thorium in agricultural soil and clay are very high, ranging from 2,000 to 510,000 mL/g (Baes and Sharp 1983); Baes et al. (1984) identify a general value of 1,500 mL/g. Sheppard and Thibault (1990) identify a value of 3,300 mL/g for loam and 5,800 mL/g for clay, with the latter ranging from 244 to 160,000 mL/g. From this information, thorium is expected to be generally immobile in site soil. The K_d value of 1,500 mL/g from Baes et al. (1984) was used for calculations in this assessment.

E.1.1.4 Uranium

Uranium commonly exists in its most oxidized (hexavalent) form in surface soil as a result of natural weathering. Tetravalent uranium compounds are generally present in reducing environments, and uranium dioxide (UO_2) can be the predominant form in insoluble minerals such as uraninite (as in natural ores) in both acidic soil (pH 4) and near-neutral to alkaline soil (increasing to pH 10 and higher) (Garrels and Christ 1965). Uranium is also commonly bound to clay particles and adsorbed onto iron and manganese oxides. When concentrations of carbonates or hydroxides are high and the pH is greater than 6, uranium can form anionic complexes.

The mobility of uranium can decrease under reducing conditions as a result of conversion to the tetravalent form. Because the pH of the site soil is in the neutral range, a limited amount of uraninite could be present under reducing conditions, e.g., in areas at which microbial activity might be high and at the base of the raffinate pits. As it moves from the surface into reducing conditions and the influence of carbonate minerals at the site, hexavalent uranium would be expected to form soluble uranyl (UO_2^{+2}) complexes with ionic carbonate and hydroxide species.

From recent U.S. Geological Survey (USGS) studies, uranium adsorption appears to be controlled by iron and manganese oxyhydroxides in the Ferrelview Formation rather than by clay minerals in the clay till. Because these oxyhydroxides are commonly present as particle coatings, they can exert an adsorptive effect substantially greater than that expected from their measured concentrations (Stumm and Morgan 1970). For example, the presence of goethite (α-FeOOH) at a few tenths of one percent in the Ferrelview Formation appears to play a major role in uranium adsorption whereas the clay mineral content of these two units (which averages 5 to 6%) does not (Schumacher and Stollenwerk 1991).

The amount of calcite (calcium carbonate, $CaCO_3$) in the overburden material may inhibit the partitioning of uranium into the solid phase. The lower partitioning to the solid phase that was determined for uranium in the clay till versus the Ferrelview Formation — as indicated by the lower K_d — correlates well with the relatively higher calcite content. This finding is consistent with the established carbonate-ion complexation phenomenon, which results in significantly increased uranium mobility. Because the clay fraction and organic content in the limestone bedrock at the site are low and the calcite content is naturally high, significant additional attenuation at depth is unlikely.

Values of K_d reported in the literature for uranium in agricultural soil and clay range from 10.5 to 4,400 mL/g (Baes and Sharp 1983); a general value of 450 mL/g has been identified by Baes et al. (1984). Sheppard and Thibault (1990) identify values of 15 and 1,600 mL/g for loam and clay soil, with respective ranges of 0.2 to 4,500 mL/g and 46 to 395,100 mL/g. Site-specific data were used to determine values of 26 and 10.7 mL/g for uranium in equilibrium with clay till at neutral pH, and values of 330 and 437 mL/g were determined for the Ferrelview Formation (Schumacher and Stollenwerk 1991). The K_d value of 330 mL/g was used for this assessment.

E.1.2 Metals

E.1.2.1 Antimony

Antimony in soil is primarily associated with particulates formed by adsorption or coprecipitation reactions with oxides of iron, manganese, and aluminum (Callahan et al. 1979). Antimony has also been found bound to carbonates in subsurface soil (Ainsworth 1988). Pentavalent antimony appears to predominate when the pH is 6.3 and higher, and trivalent species may occur under reducing conditions. Therefore, antimony could be mobilized at the base of the raffinate pits. Although most inorganic antimony compounds have a low solubility (Friberg et al. 1986), they can be mobilized by reduction and methylation processes of microorganisms, similar to arsenic (Andreae 1983). From this information, antimony is expected to be moderately immobile in site soil, except under reducing conditions such as those that can exist at the bottom of the raffinate pits.

In a study of 13 different soils from the southeastern United States antimony was strongly adsorbed (up to 100% in several mineral soils), with a median adsorption of 93%. The K_d value for antimony was higher than that for copper, which is considered relatively immobile; a partition coefficient of 185 mmol/kg soil per mol/m^3 water was determined for a mineral soil (which can translate to about 185 mL/g, depending on the interionic forces), with a value of 81 for an organic soil (King 1988). Similar values were identified for sediment solutions high in calcium, suggesting that antimony could coprecipitate with calcite (Ames and Rai 1978). In one study, antimony was found to be relatively mobile in sandy and sandy loam soil, with adsorption constants of 2 to 20 (Gerritse et al. 1982); however, other studies indicate low mobility (Agency for Toxic Substances and Disease Registry [ATSDR] 1990a). Results of a study by Foster (1989) do not indicate any general mobility for antimony in any soil. Baes et al. (1984) identify a general K_d value of 45 mL/g for antimony. A K_d value of 150 mL/g determined from Sheppard et al. (1984) was used for calculations in this assessment.

E.1.2.2 Arsenic

Arsenic forms very insoluble complexes in soil, typically reacting with hydrous oxide coatings or various cations. Most arsenic adsorbs to clays and iron or manganese compounds or aluminum complexes. Inorganic arsenate (the pentavalent form) predominates in aerobic soil, and it adsorbs or occludes on hydrous iron and aluminum oxides, adsorbs on clay colloids, and binds to organic matter. It may form slightly water-soluble complexes (Ganje and Rains 1982), but leaching from soil is slow (World Health Organization 1981). Arsenic is quite immobile in fine-textured soil, but it may be leached from coarse-textured soil with low levels of hydrous iron and aluminum oxides. Arsenic can also be fixed by complexation or chelation by iron or calcium (Callahan et al. 1979; ATSDR 1989a). Limestone soil and subsoil high in clay and iron oxides have a greater holding capacity for arsenic than other types of soil (Hiltbold et al. 1974).

Leaching of arsenic is generally important only in the upper 30 cm (1 ft) of soil, with lower leaching associated with clay and loam soil, and no leached arsenic could be detected at a depth greater than 90 cm (3 ft) in soils studied by the U.S. Environmental Protection Agency (EPA) (ATSDR 1989a). In another study, no significant arsenic was found to have leached below 20 cm (8 in.) about 100 years following application of arsenate pesticides to surface soil (in which the arsenic concentration is 120 ppm) (Adriano 1986). Because leachability is low, arsenic is not expected to impact groundwater except in very sandy soil with extremely high concentrations, e.g., to 6,000 ppm (Woolson 1983). Although arsenate is generally present in surface soil, arsenite (the trivalent anion) may exist in flooded soil (Woolson 1983); however, complete reduction has not been observed, even in anoxic interstitial water (Andreae 1979; Knox et al. 1984). Solubility increases when reducing conditions exist (e.g., following flooding), and arsenite can be fairly mobile, whereas less soluble forms adsorb to clay or soil particles. From this information, arsenic is expected to be generally immobile in site soil, except under reducing conditions such as those that can exist at the bottom of the raffinate pits.

The K_d values reported for trivalent arsenic in agricultural soil and clay range from 1 to about 8 mL/g; those for pentavalent arsenic range from about 2 to 18 mL/g (Baes and Sharp 1983). A general value of 200 mL/g has been identified by Baes et al. (1984). A K_d value of 10 mL/g derived from Baes and Sharp (1983) was used for arsenic in this assessment.

E.1.2.3 Barium

Barium is usually immobile in most soil systems because of precipitation and strong adsorption onto clay minerals, manganese oxides, and metal hydroxides. The nitrate complex can be soluble (Friberg et al. 1986), but barium forms a sparingly soluble precipitate with the carbonate ion and can also precipitate with sulfate to form barite; solubility decreases with increasing pH (Lagas et al. 1984; Adriano 1986). Because clay minerals, oxides, carbonates, and sulfates are present at the site and the soil is generally neutral to alkaline, barium is expected to be relatively immobile. Estimated K_d values range from a general value of 60 mL/g (Baes et al. 1984) to 2,800 mL/g for sediments and sandy loam soils (Rai et al. 1984). The value of 60 mL/g was used for this assessment.

E.1.2.4 Beryllium

Beryllium in soil is typically present in particulate form, tightly adsorbed onto or incorporated into clay minerals (e.g., at lower pH) by displacement of similar cations (Callahan et al. 1979; Fishbein 1981). Beryllium oxide is the common form of this species in soil, and insoluble hydroxides or hydrated complexes typically form at pH values between 5 and 8; beryllates exist in more alkaline soil (Friberg et al. 1986). Beryllium combines with the carbonate ion to form precipitates in calcareous soils (Adriano 1986). Most beryllium salts are not soluble at physiological pH (which is typically neutral to slightly alkaline, e.g., 6.8 to 7.4) (Friberg et al. 1986), and soluble salts can hydrolyze to insoluble beryllium hydroxide in soil (Cotton and Wilkinson 1972). From this information, beryllium is expected to be generally immobile in site soil. A general K_d value of 650 mL/g was identified for beryllium by Baes et al. (1984). A K_d

value of 80 mL/g was estimated from a screening-level leaching calculation with data for soil and groundwater in the vicinity of the Weldon Spring site, and this value was used for this assessment.

E.1.2.5 Cadmium

Cadmium can exist in soil as soluble or insoluble species, or it can be immobilized by adsorption onto clays or iron oxides. Cadmium can form soluble complexes and insoluble precipitates with carbonate and hydroxide ions, and it can also exist as the hydrated ion (Baker and Amacher 1982). Cadmium can be relatively mobile in highly acid soil, and mobility and bioavailability decrease with increasing soil pH and cation exchange capacity (Lu et al. 1975). The mobility of cadmium is lower in soil that contains clay and iron hydroxides, and it commonly coprecipitates with iron and manganese hydroxides. In one study of alkaline soil, cadmium was attenuated by adsorption onto minerals and interaction with calcium carbonate (Murti 1987). In another study, cadmium solubility was lowest in a calcareous clay loam (similar to site overburden material) and precipitation as the carbonate was the controlling factor. Presence of the exchangeable calcium ion also limits the availability of cadmium (Adriano 1986).

Cadmium is generally immobile in neutral to alkaline soil, and the available data suggest that the amounts removed by leaching are relatively small and that it is generally immobile in the soil profile (Adriano 1986; Page et al. 1987). The mobility of cadmium in site soil is expected to be somewhat attenuated because soil pH is generally neutral to alkaline and clay particles, iron and manganese hydroxides, and calcium carbonate are present.

The K_d values reported for cadmium in agricultural soil and clay range from about 1.3 to 27 mL/g (Baes and Sharp 1983), and a general value of 6.5 mL/g has been identified by Baes et al. (1984). Sheppard and Thibault (1990) identify values of 40 and 560 mL/g for loam and clay soil, with respective ranges of 7 to 962 mL/g and 112 to 2,450 mL/g. The value of 27 mL/g was used for calculations in this assessment.

E.1.2.6 Chromium

Chromium is chemically inert in soil, and only small fractions are typically removable by methods used to extract cationic forms, those occluded with sesquioxides, and those organically bound (Reisenauer 1982). The predominant form of chromium in soil is considered to be the hydrated insoluble trivalent chromate Cr_2O_3 (ATSDR 1989c). Soluble chromate generally forms precipitates upon addition to soil, with hexavalent chromium undergoing anion adsorption and reduction and trivalent chromium undergoing adsorption, hydrolysis, and chelation (Reisenauer 1982). Manganese and iron oxides play a major role in chromium adsorption. Adsorption of all species of chromium can occur in a mixture of substrates with pH values of 6 to 7.5, making it fairly immobile. Adsorption of the hexavalent form can decrease with increasing pH, while adsorption of the more predominant trivalent chromium can increase with increasing pH, probably as a result of cation exchange-adsorption of hydrolyzed trivalent species (Adriano 1986).

Under typical soil conditions, organic matter converts soluble, hexavalent chromium to the insoluble, trivalent form. However, the mobility of chromium in flooded soil can increase as a result of the formation of soluble complexes following anaerobic decomposition of plant matter (ATSDR 1989c). Trivalent chromium forms relatively inert complexes (Reisenauer 1982), and small amounts can exist as soluble complexes in sediment; however, soluble forms of hexavalent chromium typically predominate. The hexavalent form can be reduced to the trivalent e.g., by organic matter; however, the oxidation of the trivalent to the hexavalent form is typically slow because of the low solubility of chromium hydroxide from pH 6 to 11 (Eary and Rai 1987). From this information, chromium is expected to be somewhat immobile in site soil, except under certain conditions, for example, if anaerobic microbial degradation were occurring under reducing conditions at the base of the raffinate pits.

The K_d values reported for agricultural soil and clay range from about 1.2 to 1,800 mL/g for hexavalent chromium and from 470 to 150,000 mL/g for trivalent chromium (Baes and Sharp 1983); a general value of 850 mL/g has been identified by Baes et al. (1984). Sheppard and Thibault (1990) identify values of 30 and 1,500 mL/g for loam and clay soil, with a range of 2.2 to 1,000 mL/g for loam. Site-specific data have been used to calculate values of 2 to 50 mL/g for neutral solution in equilibrium with the clay till and 17 mL/g for the Ferrelview Formation (Schumacher and Stollenwerk 1991). The median values of 2,200 and 37 mL/g identified by Baes and Sharp (1983) for trivalent and hexavalent chromium in agricultural soil and clay were used for this assessment.

E.1.2.7 Cobalt

Cobalt is generally present in divalent or trivalent forms (Seiler and Sigel 1988), and it is commonly adsorbed on and incorporated into the surface coatings of iron and manganese oxides and crystalline material such as goethite. It can also be incorporated into the crystal lattice of clays or occluded in precipitates (Adriano 1986). Adsorption onto the ferric oxides is attributable to ion exchange (Ames and Rai 1978). Cobalt is generally immobile at neutral pH, either through precipitation or adsorption reactions, but it may be mobilized at lower pHs. Mobility can also increase in soil with high organic content as a result of complexation, and slight mobilization may occur in the presence of chloride (ATSDR 1990d). Divalent cobalt can also be released from adsorption sites under reducing conditions (Smith and Carson 1981; Robson and Pitman 1983).

Cobalt is less mobile at higher soil pH probably because of precipitation as the hydroxide or carbonate. In one study, cobalt was adsorbed more strongly than the competing calcium ion onto a bentonite and a clay (Ames and Rai 1978). Cobalt is expected to be relatively immobile in site soil because the pH of site soil is generally neutral to alkaline, the organic content is fairly low, and clays are present.

The K_d values reported for cobalt in agricultural soil and clay range from 0.2 to 3,800 (Baes and Sharp 1983; Smith and Carson 1981), with sandy soil containing less cobalt than clayey soil (Kubota and Cary 1982); a general value of 45 mL/g has been identified by Baes et al. (1984). Sheppard and Thibault (1990) identify values of 1,300 and 550 mL/g for loam and clay soil, with

respective ranges of 100 to 9,700 mL/g and 20 to 14,000 mL/g. The median value of 55 mL/g identified by Baes and Sharp (1983) for agricultural soil and clay was used for this assessment.

E.1.2.8 Copper

A number of studies suggest that copper in soil is relatively immobile and that it binds more strongly to soil components than do other divalent cations (Gerritse and Van Driel 1984; King 1988). Copper precipitates with hydroxide, carbonate, phosphate, and silicate ions (Baker and Amacher 1982); it adsorbs to carbonates, hydrous oxides of iron and manganese, clay minerals, and organic matter; and it can also be incorporated within the structure of mineral lattices (Callahan et al. 1979; Fuhrer 1986; Tyler and McBride 1982). Although copper can leach from sandy soil with low pH, it precipitates out of solution at alkaline pH (Perwack et al. 1980). Many experimental data indicate that copper remains in the upper few centimeters of soil because leaching is minimal (ATSDR 1990e). For example, in one study, essentially no leaching was observed in silty and clayey soils and only slight leaching (from 1 to 3 cm [about 1 in.]) was observed in sandy soils with low cation exchange capacity (Adriano 1986).

Copper is expected to be relatively immobile in site soil because (1) the pH is generally neutral to alkaline and (2) carbonates, clay minerals, and hydrous iron and manganese oxides are present in the overburden. The K_d values reported for copper in agricultural soil and clay range from about 1.4 to 333 mL/g (Baes and Sharp 1983), and a general value of 35 mL/g has been identified by Baes et al. (1984). A K_d value of 140 mL/g was derived from these estimates in combination with results of a screening-level leaching calculation using data for soil and groundwater near the site; this value was used for this assessment.

E.1.2.9 Lead

Lead is typically present in the divalent form, and it tends to form relatively insoluble compounds with common anions, such as hydroxide, and sulfate. An insoluble precipitate can also form with sulfide, which can be present under reducing conditions like those at the base of the raffinate pits. Lead also forms insoluble complexes with carbonate, precipitating out of solution as the carbonate and mixed hydroxide carbonate at pH values higher than 5.4 (Adriano 1986; EPA 1986). In addition to precipitating and forming sparingly soluble surface complexes, lead is retained strongly in soil by adsorption onto clays and manganese and iron oxides (Adriano 1986). Lead is considered quite immobile, and it can remain in the upper 5 cm (2 in.) of soil for hundreds of years following deposition (Burau 1982). From this information, lead is expected to be generally immobile in site soil.

The K_d value for lead is higher than that for copper (King 1988), which is considered relatively insoluble in most soils. The K_d values reported for lead in agricultural soil and clay range from 4.5 to 7,640 mL/g (Baes and Sharp 1983), and a general value of 900 mL/g has been identified by Baes et al. (1984). Sheppard and Thibault (1990) identify values of 16,000 and 550 mL/g for loam and clay soil, with a range of 3,500 to 59,000 mL/g for loam. Site-specific data have been used to determine values of 150 to 950 mL/g for neutral solution in equilibrium

with the clay till and a value of 220 mL/g for the Ferrelview Formation under the same conditions (Schumacher and Stollenwerk 1991). The K_d value of 150 mL/g was used for lead in this assessment.

E.1.2.10 Lithium

Lithium is commonly present in trace amounts with aluminosilicate and hydroxide minerals and manganese oxides in soil, and water-soluble lithium is correspondingly low. Lithium readily forms the stable monovalent ion, and although many lithium salts can be soluble in water (Knudsen et al. 1982), they are generally less soluble than the corresponding salts of other alkali metals such as sodium or potassium (Friberg et al. 1986). Lithium does not substitute in carbonate rocks, and the carbonate is insoluble in water (Seiler and Sigel 1988). Although transport would be expected to be similar to that for the other alkali metals as a result of the valence state, exchangeable (available) lithium is usually found in much lower concentrations than sodium or potassium (Knudsen et al. 1982). From this information and the observation of lithium in groundwater near the raffinate pits, lithium is expected to be relatively mobile at the site.

A general K_d value of 300 mL/g has been identified for lithium by Baes et al. (1984). Site-specific data have been used to determine values of 0.6 to 1.5 mL/g for solution in equilibrium with the clay till at neutral pH and values from 1.4 to 1.8 mL/g for the Ferrelview Formation at neutral pH (Schumacher and Stollenwerk 1991). A K_d value of 9 mL/g was used for this assessment. This value was derived from the reported estimates in combination with results of a screening-level leaching calculation using data for soil and groundwater near the site.

E.1.2.11 Manganese

Manganese can be found in a variety of forms in soil — within the crystal lattice of minerals and precipitated as oxides or hydrous oxides (typical), bound to organic material (to a limited extent), exchangeable, and dissolved; insoluble oxides are the most common form (Stumm and Morgan 1970). The tetravalent state is predominant in nature, but the divalent state is common in most biological systems (Seiler and Sigel 1988).

Manganese can be readily available for uptake in strongly acid or flooded soil (Bear 1965), as low pH and reducing conditions can each increase the presence of divalent forms that are more commonly soluble and exchangeable, e.g., from adsorption sites or as coprecipitates with hydrous iron oxides. However, relatively insoluble higher oxide minerals (e.g., trivalent or tetravalent forms) exist at higher pH and under generally oxidizing conditions (Gambrell and Patrick 1982), such as those that exist in site soil. Manganese is also commonly unavailable in calcareous soil because of precipitation as manganese carbonates and chemisorption on the calcium carbonate. Availability in soil with a generally neutral to alkaline pH is also low, as a pH above 6.5 favors oxidation to the relatively insoluble tetravalent form (Adriano 1986). From this information, manganese is expected to be relatively immobile at the site.

The K_d values reported for manganese in agricultural soil and clay range from 0.2 to 10,000 mL/g (Baes and Sharp 1983), and a general value of 65 mL/g has been identified by Baes et al. (1984). Sheppard and Thibault (1990) identify values of 750 and 180 mL/g for loam and clay soil, with respective ranges of about 40 to 77,000 mL/g and about 24 to 49,000 mL/g. The median K_d value of 150 mL/g identified by Baes and Sharp (1983) for agricultural soil and clay was used in this assessment.

E.1.2.12 Mercury

Mercury is typically present in soil as the monovalent or divalent ion. It can form many organic and inorganic compounds and complexes, and inorganic mercury can be methylated by microbial activity (Stewart and Bettany 1982). Monoalkyl mercury compounds can be somewhat soluble, but dialkyl compounds are fairly insoluble. Mercury forms precipitates with carbonate and sulfide, and it also adsorbs onto clays and iron and manganese oxides (Adriano 1986). Higher concentrations of mercury are generally present in surface soil compared with subsurface soil, in soil from urban areas compared with rural areas, and in soil that contains organic material (ATSDR 1989e).

Extraction tests indicate that mercury is less available from clay soil than from sandy soil (Rogers 1976). A limited amount of mercury may enter solution, and the colloidal poly-nuclear hydroxo complexes that form can ultimately result in precipitates (Stumm and Morgan 1970). Adsorption onto particulates appears to be the controlling process for mercury in soil, and mercury that is adsorbed to particulate matter is not easily desorbed; this results in the relative insignificance of leaching as a transport process (ATSDR 1989e). Mercury can be effectively adsorbed on organic matter in acid soil (pH <4), whereas iron oxides and clay minerals are the effective adsorption sites in soil of higher pH (>5.5) (Adriano 1986). The mobility of mercury in site soil is expected to be somewhat limited by iron oxides and clay particles in the overburden material. Baes et al. (1984) have identified a general K_d value for mercury of 10 mL/g. The K_d value of 100 mL/g identified by Nuclear Safety Associates (1980) was used for this assessment.

E.1.2.13 Molybdenum

Molybdenum commonly occurs as the oxide in ores and as molybdate in solution and can exist over a range of oxidation states. The molybdenate ion can complex with calcium, and adding lime (calcium carbonate) to soil to increase the pH can increase molybdenum solubility (Bear 1965). Adsorption of molybdate by iron and aluminum oxides decreases as soil pH increases above 4 (Anderson and Oertel 1946). Hydrous iron oxides are the major influence on molybdenum adsorption in soil and in combination with organic matter may provide additional adsorption sites in acid soil (Adriano 1986). The presence of iron oxides in site soil and a pH generally in the neutral to alkaline range are two factors that are expected to control the mobility of molybdenum at the site.

Baes et al. (1984) have identified a general K_d value of 20 mL/g for molybdenum. Sheppard and Thibault (1990) identify values of 125 and 90 mL/g for loam and clay soil, with a range of about 13 to 400 mL/g for clay. Site-specific data were used to estimate values that range from about 2 to 3 mL/g for solutions in equilibrium with the clay till at neutral pH and values of 7 to 11 mL/g for Ferrelview Formation under the same conditions (Schumacher and Stollenwerk 1991). A K_d value of 30 mL/g was derived for molybdenum from this information, in combination with results of a screening-level leaching calculation using data for soil and groundwater near the site; this value was used for this assessment.

E.1.2.14 Nickel

Nickel is commonly distributed in soil in the following order: fixation within a clay mineral lattice, adsorbed on iron and manganese oxides, associated with carbonate or organic residues, and adsorbed on exchange sites (Adriano 1986). Nickel can form soluble complexes with hydroxide, sulfate, and chloride at pH less than 9. Although it can be fairly mobile at low pH, sorption of nickel increases with increasing pH and it is less mobile in basic mineral soils. Nickel is usually present as the divalent ion and can form the nickel ferrite precipitate with iron (Sadiq and Enfield 1984a, 1984b). In calcareous soil, it forms a fairly insoluble precipitate with the carbonate ion, and insoluble nickel sulfides can form under reducing condition such as those that may exist at the base of the raffinate pits. The pH of site soil is generally neutral, and clays, iron oxides, and carbonates are present; therefore, nickel is expected to be relatively immobile.

Baes et al. (1984) have identified a general K_d value for nickel of 150 mL/g. Sheppard and Thibault (1990) identify values of 300 and 650 mL/g for loam and clay soil, with a range of about 300 to 2,500 mL/g for clay. The value of 100 mL/g identified by Sheppard et al. (1984) was used for this assessment.

E.1.2.15 Selenium

Selenium can exist in soil as insoluble heavy metal selenides, selenium sulfides, and elemental selenium and also as soluble alkali selenites (tetravalent form) and selenates (hexavalent form) (ATSDR 1989g). Soluble forms dominate in alkaline soil under oxidizing conditions, sodium and potassium selenides dominate in well-drained soil of neutral pH, and the insoluble sulfides and selenides dominate in acidic soil with high organic content (National Academy of Sciences 1976).

The availability of selenium in different soils is quite variable; it can be relatively available for plant uptake as the selenate above pH 7.5, but availability decreases in acid soil, e.g., forming a low-solubility compound with the ferric ion (Seiler and Sigel 1988). Selenites are typically less soluble than selenates (National Academy of Sciences 1980) and could be found in alkaline to mildly acidic soil and under reducing conditions (e.g., waterlogged soil), such as may exist at the base of the raffinate pits. The low solubility of selenite in acid or neutral soil has been attributed to its combination with the ferric ion, including strong adsorption onto iron oxides and hydroxides. Selenite can be slowly oxidized to selenate in alkaline soil; selenate is

more soluble and is usually less well adsorbed, but it can be sorbed on organic material and is retained in calcareous soil by formation of the calcium salt (Adriano 1986). From this information, selenium is expected to be somewhat immobile at the site.

The K_d values reported for the tetravalent form of selenium in agricultural soil and clay range from 1.2 to 8.6 mL/g (Baes and Sharp 1983). A general value of 300 mL/g has been identified by Baes et al. (1984), and Sheppard and Thibault (1990) report median values of 150 and 115 mL/g for loam and clay soil, with a range of 36 to 246 mL/g for clay. A K_d value of 110 mL/g was derived from this information in combination with results of a screening-level leaching calculation using data for soil and groundwater near the site; this value was used for this assessment.

E.1.2.16 Silver

Silver is generally present as a monovalent ion that can complex with bicarbonate, sulfate, or nitrate or precipitate as the carbonate (Przybylowicz and Zuehlke 1966). It easily forms insoluble compounds and is commonly present in soil adsorbed onto iron or manganese oxides or manganese dioxides. Silver can exist as a halide compound (e.g., with chloride) under oxidizing conditions and as the free metal or sulfide under reducing conditions (ATSDR 1990k). Because site soil is generally neutral to alkaline and contains clays and oxides of iron and manganese (adsorption sites) and carbonate (precipitator), silver is expected to be somewhat immobile at the site. The K_d values reported for silver in agricultural soil range from 10 to 1,000 mL/g, with a median value of 110 mL/g (Baes and Sharp 1983); higher values are expected for clayey soil of neutral to alkaline pH. A general value of 45 mL/g has been identified by Baes et al. (1984). Sheppard and Thibault (1990) report average values of 120 and 180 mL/g for loam and clay soils. The value of 120 mL/g was used for this assessment.

E.1.2.17 Thallium

Thallium typically exists as the monovalent ion and tends to adsorb to soil particles or undergo substitution in mineral constituents (Callahan et al. 1979; Wallwork-Barber et al. 1985; Adriano 1986; Frantz and Carlson 1987). It may precipitate from solution as a solid mineral phase; however, thallium chloride, sulfate, and carbonate are soluble (EPA 1980b) and the hydroxide is considered a strong base because of its dissociation in water (Butler 1964). Although few quantitative data exist relative to adsorption, a K_d of 19 L/g was estimated for a montmorillonite clay in one study (Magorian et al. 1974). Data from another study suggest that although initial thallium solubility may be high, mobility may decrease considerably upon incorporation in soil; concentrations decreased from 3.6 ppm in the top 10 cm (4 in.) to 0.1 ppm at a depth of 60 to 70 cm (24 to 28 in.) (Schoer 1984). A general K_d value of 1,500 mL/g has been identified for thallium by Baes et al. (1984). These data indicate that the mobility of thallium is likely to be attenuated at the site by clays and other soil particles that provide binding sites and the neutral to alkaline pH conditions that promote precipitation. Results of a screening-level leaching calculation using data for soil and groundwater near the site identified a value of 120 mL/g, which was used for this assessment.

E.1.2.18 Vanadium

Vanadium is commonly present as the vanadate ion (pentavalent) under oxidizing conditions and as the vanadyl ion (tetravalent) under reducing conditions. Both species adsorb strongly to iron and manganese oxides and oxyhydroxides. Vanadium in soil precipitates with calcium, copper, and other divalent cations; adsorbs onto clay particles; and adsorbs onto or coprecipitates with hydrous ferric oxide (Van Zinderen and Jaworski 1980). Trivalent vanadium can be oxidized to pentavalent vanadium during leaching processes (Waters 1977). From this information, vanadium is expected to be moderately immobile at the site. Baes et al. (1984) have identified a general K_d value of 1,000 mL/g for vanadium. Site-specific data have been used to estimate values of 55 to 67 mL/g for solution in equilibrium with the clay till at neutral pH and a value of 390 mL/g for the Ferrelview Formation (Schumacher and Stollenwerk 1991). The value of 55 mL/g was used for this assessment.

E.1.2.19 Zinc

Zinc exists in soil in the divalent state, and it can precipitate which common soil ions such as hydroxide, phosphate, carbonate, and silicate (Baker and Amacher 1982). Zinc can also adsorb strongly to hydrous iron and manganese oxides and silicate clays. In addition, it can bind to or form complexes with carbonates, hydroxides, sulfides, and organic ligands, which can be incorporated into solids. Although adsorption is lower in acidic soil and zinc can be mobile in acidic environments (Callahan et al. 1979), zinc absorbs readily at pH above 7. The presence of bicarbonates, e.g., in limestone areas, enhances the precipitation of zinc as the hydroxide and carbonate (National Academy of Sciences 1977b). The very low availability of zinc in calcareous and alkaline soils is believed to be affected by adsorption on carbonates, precipitation as the hydroxide and/or carbonate, and formation of insoluble calcium zincate (Adriano 1986). Because the soil pH is generally neutral to alkaline and carbonates and bicarbonates are present, zinc is expected to be somewhat immobile at the site.

The K_d values reported for zinc in agricultural soil and clay range from 0.1 to 8,000 mL/g (Baes and Sharp 1983), and a default value of 40 mL/g has been identified by Baes et al. (1984). The K_d value of 52 mL/g from Sheppard et al. (1984) was used in this assessment.

E.1.3 Inorganic Anions

E.1.3.1 Fluoride

Fluoride is present in a number of common soil minerals and can be associated with clay, amorphous aluminum species, and organic matter. The fluoride content of a soil depends on its inorganic components, and the common use of phosphate fertilizers has resulted in relatively high levels of fluoride, e.g., several hundred parts per million in agricultural soil (Adriano and Doner 1982). Fluoride in soil solution exists as the monovalent ion, which can form negatively charged aluminum and silica complexes (Hopkins 1977). Fluoride is more

available from acid soil than neutral soil, and concentrations typically increase with depth. It forms an insoluble precipitate with calcium, e.g., in calcareous soils (Adriano 1986), and calcium is abundant at the site. A considerable amount of iron and manganese oxides is present in the overburden, so some sorption of fluoride is expected (Schumacher 1992). Baes et al. (1984) have identified a general K_d value of 150 mL/g for fluorine, while a site-specific value of 8.4 mL/g has been calculated for the clay till at neutral pH; estimates for solutions at a fixed neutral pH for the Ferrelview Formation and clay till were 23 and 17 mL/g, respectively (Schumacher and Stollenwerk 1991). From this information, in combination with results of a screening-level calculation using local soil and groundwater data, a K_d value of 13 mL/g has been applied in this assessment.

E.1.3.2 Nitrate and Nitrite

Nitrate and nitrite are typically present in soil solution as mobile anions. They are both key inorganic components of the nitrogen cycle and can be readily taken up by plants. Within the plant, nitrate and nitrite are assimilated and converted by reductases to organic forms such as glutamate, as part of the nutrient cycling process (Beevers and Hageman 1983). These negatively charged anions are expected to be mobile in site soil because they would not be attenuated effectively by the similarly charged clay particles in the overburden material. Baes et al. (1984) have identified a general K_d value of 0.5 mL/g for nitrogen, and this value was used for nitrate and nitrite in this assessment.

E.1.3.3 Sulfate

Sulfate is present in site groundwater, probably as a result of previous ordnance works activities, e.g., from past sulfuric acid spills. Because of its negative charge, sulfate is not expected to be attenuated by the similarly charged clay particles in the overburden material. It is hypothesized that the sulfate in site soil is mobile and that it is present as coatings on calcite that serve as a source of groundwater contamination (Schumacher and Stollenwerk 1991). Baes et al. (1984) have identified a default K_d value of 7.5 mL/g for sulfur, and a site-specific value of 0.2 mL/g has been estimated for sulfate in the clay till at neutral pH (Schumacher and Stollenwerk 1991). Thus, sulfate is expected to be mobile at the site. (Because sulfate is not indicated as a contaminant of concern relative to potential health or ecological effects, it is not discussed further in this appendix.)

E.1.4 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are generally insoluble in water and are therefore often found adsorbed to particulates. The attenuation of PAHs in soil can often be correlated to the molecular weights of the individual compounds, with moderate to strong adsorption potential for compounds with low to high molecular weight. Compounds with lower molecular weights are acenaphthene, anthracene, fluorene, and phenanthrene (152 to 178 g/mol); compounds with medium molecular weights are fluoranthene and pyrene (202 g/mol); and

compounds with high molecular weights are benz(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(g,h,i)perylene, benzo(a)pyrene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene (228 to 278 g/mol) (ATSDR 1990j).

The sorption of PAHs to soil increases with increasing organic content and decreasing particle size. For example, the partition coefficient for adsorption of pyrene to clay is 1,400 to 3,800 and to silt is 1,500 to 3,600 mL/g (Karickhoff et al. 1979). Surface soil at the relatively small areas on-site that contain PAH contamination is predominantly clayey (i.e., the particle size is small), so PAHs are expected to be strongly adsorbed and leaching is not considered a primary factor in the fate and transport of these contaminants. For this assessment, a K_d value was estimated for each PAH on the basis of its octanol/water partition coefficient (K_{ow}) and/or the partition coefficient between the organic carbon in soil and the soil solution concentration (K_{oc}), in combination with site-specific data for the fraction of organic carbon in soil. These estimates range from 13 to 77,000 mL/g.

E.1.5 Polychlorinated Biphenyls

For polychlorinated biphenyls (PCBs), the highly chlorinated compounds such as Aroclor 1248, 1254, and 1260 are very persistent in the environment because they are resistant to biodegradation and adsorption is a major fate process (Callahan et al. 1979). The more highly chlorinated compounds such as those found at the site are most strongly adsorbed, and leaching is not a primary factor in environmental fate and transport. Those PCBs that do reach subsurface layers may be effectively sequestered from subsequent distribution (ATSDR 1989f), and although volatilization can be a factor, the strong adsorptive nature of these compounds decreases this tendency. Because of the low water solubility, high K_{ow} values and adsorptive nature of PCBs, leaching from soil is expected to be insignificant (although they can be leached from soil with organic solvents [Griffin and Chou 1981]). From this information, PCBs are expected to be generally immobile in site soil. A K_d value was estimated for each PCB at the site in the same manner described for PAHs, and the values range from 3,900 to 94,000 mL/g.

E.1.6 Nitroaromatic Compounds

The nitroaromatic compounds detected in site soil and groundwater are 2,4-dinitrotoluene (DNT), 2,6-DNT, 1,3,5-trinitrobenzene (TNB), 1,3-dinitrobenzene (DNB), and nitrobenzene (NB). The mobility of a nitroaromatic compound is primarily influenced by its water solubility and by the particle size (e.g., sand, silt, or clay) and organic content of the soil. Chemicals with high molecular weights typically bind to soil and are less water soluble than those with low molecular weights.

The compound NB is only sparingly soluble in water (ATSDR 1990i). It can be taken up by plants, does not bioaccumulate, and does not accumulate in soil or sediment. It might be mobile in certain systems, as indicated by the adjusted K_d value determined by multiplying the K_{oc} value of 36 (Mabey et al. 1982) by a site-specific value of 1.4% for organic carbon, but photolysis and biodegradation are the significant fate and transport pathways (Callahan et al.

1979). Thus, although the calculated K_d value is low, this compound is actually expected to be somewhat immobile in site soil, as limited by its solubility and biodegradation (Section 3.2.2.5).

The DNT compounds are moderately soluble and their octanol-water partition coefficients are relatively low, so they could be mobile in certain systems. For example, the K_{oc} values estimated for a sediment/water system are 45 for 2,4-DNT and 92 for 2,6-DNT (Mabey et al. 1982). Adjusted K_d values were calculated for these compounds, as described for PAHs. The actual mobility of DNT compounds at the site is expected to be limited by photolysis and degradation processes (Section 3.2.2.5), with some attenuation by adsorption. An average partition coefficient of about 53 µg/g per µg/mL has been estimated for trinitrotoluene (TNT) (Spanggord et al. 1980), and the adjusted K_d value calculated as described for PAHs would indicate that the compound could be mobile in certain systems. In fact, lysimeter studies have indicated that TNT migrates slowly in loamy and sandy soils (although the measured rate was faster in clayey soils, this was attributed to channeling) (Burrows et al. 1989).

A leaching study was recently conducted using soil from the Army property adjacent to the Weldon Spring site that was spiked with nitroaromatic compounds. The study results indicated that TNT strongly adsorbed to certain soils (e.g., only 1% was available for leaching), increasing directly with organic content; 2,4-DNT and 2,6-DNT were somewhat more mobile, with about 30 to 80% available in the various soils tested (Terkonda 1991). Data recently collected from the Army property also indicate that biodegradation might exert a greater effect than adsorption, serving as the predominant mechanism by which TNT is attenuated (Schumacher 1992).

Leaching is not expected to be a primary factor in the fate of surficial nitroaromatic contamination at the site for two reasons. First, munitions operations ceased more than 45 years ago and most of the nitroaromatic contaminants were removed during past cleanup efforts, so subsequent leaching would have already exerted its primary effect. Second, surface or near-surface reactions — such as photolysis, biodegradation, and adsorption on organic material — would tend to control residual surface contaminants. Because these reactions would not be expected to similarly affect subsurface nitroaromatic contamination, e.g., at depths of 2 to 3 m (7 to 10 ft) and below, leaching would probably be the primary mechanism for further transport of the subsurface contamination. A K_d value was estimated for each nitroaromatic compound in the same manner described for the PAHs, and the derived values range from 0.15 to 1.29 mL/g.

E.1.7 Distribution Coefficients

The contaminant-specific K_d values used to derive plant uptake and animal biotransfer factors for radionuclides, lead, and the organic compounds from site soil with the methods described in Sections 3.5.2.2 and E.2.1.2 are listed in Table E.1. These values were also used to evaluate potential leaching of contaminants from soil to groundwater in Appendix E of the FS (DOE 1992a).

TABLE E.1 Estimated K_d Values for Contaminants in Site Soil[a]

Contaminant	K_d (mL/g)	Source
Radionuclides		
Actinium	794	Sheppard et al. (1984)
Protactinium	330	Sheppard et al. (1984)
Radium	165	Seeley and Kelmers (1985)
Thorium	1,500	Baes et al. (1984)
Uranium	330	Schumacher and Stollenwerk (1991)[b]
Metals		
Antimony	150	Sheppard et al. (1984)
Arsenic	10	Baes and Sharp (1983)[c]
Barium	60	Baes et al. (1984)
Beryllium	80	Baes et al. (1984)[d]
Cadmium	27	Baes and Sharp (1983)
Chromium III	2,200	Baes et al. (1984)
Chromium VI	37	Baes and Sharp (1983)
Cobalt	55	Baes and Sharp (1983)
Copper	140	Baes and Sharp (1983); Baes et al. (1984)[e]
Lead	150	Schumacher and Stollenwerk (1991)[f]
Lithium	9	Baes et al. (1984); Schumacher and Stollenwerk (1991)[g]
Manganese	150	Baes and Sharp (1983)
Mercury	100	Nuclear Safety Associates (1980)
Molybdenum	30	Sheppard and Thibault (1990); Baes et al. (1984)[h]
Nickel	100	Sheppard et al. (1984)
Selenium	110	Sheppard and Thibault (1990)[i]
Silver	120	Sheppard and Thibault (1990)
Thallium	120	Baes et al. (1984)[j]
Vanadium	55	Schumacher and Stollenwerk (1991)[k]
Zinc	52	Sheppard et al. (1984)
Inorganic anions		
Fluoride	13	Schumacher and Stollenwerk (1991)[l]
Nitrate	0.5	Baes et al. (1984)
PAHs		
Acenaphthene	64	Mabey et al. (1982)[m]
Anthracene	200	Mabey et al. (1982)[m]
Benz(a)anthracene	2,800	Mabey et al. (1982)[m]
Benzo(b)fluoranthene	7,700	Mabey et al. (1982)[m]
Benzo(k)fluoranthene	7,700	Mabey et al. (1982)[m]
Benzo(g,h,i)perylene	22,000	Mabey et al. (1982)[m]
Benzo(a)pyrene	77,000	Mabey et al. (1982)[m]
Chrysene	2,800	Mabey et al. (1982)[m]
Fluoranthene	530	Mabey et al. (1982)[m]
Fluorene	100	Mabey et al. (1982)[m]

TABLE E.1 (Cont.)

Contaminant	K_d (mL/g)	Source
PAHs (cont.)		
Indeno(1,2,3-cd)pyrene	22,000	Mabey et al. (1982)[m]
2-Methylnaphthalene	110	Lyman et al. (1990)[n]
Naphthalene	13	Mabey et al. (1982)[m]
Phenanthrene	200	Mabey et al. (1982)[m]
Pyrene	530	Mabey et al. (1982)[m]
PCBs		
Aroclor 1248	3,900	Mabey et al. (1982)[m]
Aroclor 1254	7,400	Mabey et al. (1982)[m]
Aroclor 1260	94,000	Mabey et al. (1982)[m]
Nitroaromatic compounds		
DNB	0.22	Verschueren (1983)[o]
2,4-DNT	0.63	Mabey et al. (1982)[m]
2,6-DNT	1.29	Mabey et al. (1982)[m]
NB	0.5	Mabey et al. (1982)[m]
TNB	0.15	Mabey et al. (1982)[m]
TNT	0.28	McKone (1990)[p]

[a] Estimated from literature data in combination with site-specific information, including soil type and pH. Because the range of literature values is highly variable, screening-level leaching calculations were performed for the metals with local soil and groundwater data to provide a limited consistency check for this assessment. Additional site-specific data that will be collected to support the groundwater operable unit will be used to refine these preliminary estimates within the next several years.

[b] Determined from site-specific data for solution in equilibrium with the Ferrelview clay at neutral pH (fixed).

[c] Midrange for pentavalent arsenic in agricultural soil and clay (2 to 18 mL/g), as supported by a screening-level calibration estimate with data for local soil and groundwater.

[d] Modified from the value of 650 mL/g by a screening-level calibration estimate with data for local soil and groundwater.

[e] Modified from the value of 35 mL/g in Baes et al. (1984) and the upper end of the range given for agricultural soil and clay (333 mL/g) in Baes and Sharp (1983) by a screening-level calibration estimate with data for local soil and groundwater.

[f] Determined from site-specific data for solution in equilibrium with the clay till at neutral pH (unadjusted).

[g] Modified from the value of 300 mL/g in Baes et al. (1984) and estimates of 1 to 2 mL/g from Schumacher and Stollenwerk (1991) by a screening-level calibration estimate with data for local soil and groundwater.

TABLE E.1 (Cont.)

[h] Modified from the median values of 125 and 90 mL/g for loam and clay in Sheppard and Thibault (1990), combined with the value of 20 mL/g in Baes et al. (1984) and a screening-level calibration estimate with data for local soil and groundwater.

[i] Modified from the median value of 115 mL/g for clay by a screening-level calibration estimate with data for local soil and groundwater.

[j] Modified from the value of 1,500 mL/g by a screening-level calibration estimate with data for local soil and groundwater.

[k] Determined from site-specific data for solution in equilibrium with the clay till at neutral pH (unadjusted).

[l] Determined from site-specific data for solutions in equilibrium with the Ferrelview clay (23 mL/g) and the clay till (about 8 and 17 mL/g for unadjusted and fixed pH) at neutral pH.

[m] Calculated by multiplying the K_{ow} value given in Mabey et al. (1982) by a site-specific estimate of 1.4% for the fraction of organic carbon in soil.

[n] Calculated by multiplying the K_{ow} value determined from the fragment method in Lyman et al. (1990) by a site-specific estimate of 1.4% for the fraction of organic carbon in soil.

[o] Calculated by multiplying the K_{ow} value given in Verschueren (1983) by a site-specific estimate of 1.4% for the fraction of organic carbon in soil.

[p] Calculated by multiplying the K_{ow} value given in McKone (1990) by a site-specific estimate of 1.4% for the fraction of organic carbon in soil.

E.2 PLANT UPTAKE AND ANIMAL BIOTRANSFER

E.2.1 Radionuclides

E.2.1.1 Actinium and Protactinium

No literature information is available for quantifying plant uptake and animal biotransfer of actinium or protactinium. The values used to estimate plant uptake from soil in this assessment were those recommended by Thibault et al. (1990); these values are 0.00088 and 0.00063 for actinium and protactinium, respectively. Seiler and Sigel (1988) report that animal absorption of protactinium citrate ranged from 0.01 to 0.0001 and that the skeleton was the primary deposition site, with about 40% of protactinium directly excreted following intake. For actinium, they only report that the fractional absorption following ingestion is very low. The value of 0.005 d/kg recommended by Napier et al. (1980) was used to estimate animal biotransfer for both radionuclides in this assessment.

Various studies have indicated that radium and actinides such as actinium, protactinium, and uranium can bioconcentrate from aquatic systems by factors of 100 to 10,000 in or on algae, often by surface adsorption mechanisms (Whicker and Schultz 1982). Site data indicate that levels in algae range from less than to more than those in the ambient water (Section 5.5 of the remedial investigation [RI] for the chemical plant area [DOE 1992b]). For this assessment, a factor of 100 was used to estimate the concentrations of actinium and protactinium in algae from the raffinate pits for modeling uptake by the duck and goose.

E.2.1.2 Radium

Radium is chemically similar to barium and calcium and can be readily absorbed by plant roots, but translocation to the shoot has been reported to be minimal (e.g., 1%), depending on the plant species and growing conditions. Plant uptake is highly dependent on the amount of organic content in soil, which tends to bind radium and make it unavailable for root uptake. A soil-to-plant concentration ratio of about 0.002 has been reported for plants growing on uranium mill tailings (Rayno 1983). Research on uptake of radium by garden plants grown in soil contaminated by uranium processing wastes reported a mean concentration ratio of 0.0011 (Tracy et al. 1983). Although one study suggests a factor of 0.1 may be appropriate for forage under certain conditions (Watson et al. 1984), others have recommended values of 0.0014 (Napier et al. 1980) and 0.00031 to 0.062 (National Council on Radiation Protection and Measurements 1984). These values represent the ratio of the concentration in fresh (wet weight) vegetation to that in dry soil.

A strong negative correlation has been reported between the soil-to-plant concentration ratio, or uptake factor (UF), and the K_d value (Sheppard and Thibault 1990). The empirical relationship for a loam soil has been identified as follows:

$$\ln K_d = 3.36 - 0.5(\ln UF) \qquad \text{(E.1)}$$

From this relationship and a K_d value of 165 mL/g estimated for radium at the Weldon Spring site (Seeley and Kelmers 1985), a value of 0.030 expressed as a dry-weight ratio was used to model plant uptake from soil for this assessment.

Values identified in the literature for estimating the biotransfer of radium from ingested vegetation to beef include 0.00051, 0.00099, and 0.0040 d/kg (Watson et al. 1984; Napier et al. 1980; Rayno 1983). To be conservative, a value of 0.0040 d/kg was used to estimate radium biotransfer for game animals in this assessment.

Studies of two aquatic systems reported bioconcentration factors for marine plankton that ranged from 100 to 2,750 (Whicker and Schultz 1982). Considering site-specific data (DOE 1992b), a bioconcentration factor of 100 was used to estimate the level of radium in algae from the raffinate pits for modeling uptake by the duck and goose in this assessment.

E.2.1.3 Thorium

The mobility of thorium in the environment is low, and thorium is not readily taken up by plants because it is usually strongly sorbed to soil particles. Little of the thorium that is taken up by plant roots is translocated to aboveground parts (e.g., 1%) (Taskayev et al. 1986). A number of soil-to-plant uptake factors have been reported for thorium — including 0.0042 (Napier et al. 1980), 0.0001 to 0.007 (National Council on Radiation Protection and Measurements 1984; Garten 1978), and 0.0006 for thorium-232 (Linsalata et al. 1989). Concentrations measured in various foodstuffs range from about 0.003 to 0.02 ppm wet weight (ATSDR 1990n), compared with a common background soil concentration of about 3 ppm. To be conservative, a value of 0.0042 was used to model plant uptake from soil in this assessment.

Very limited data are available for estimating the biotransfer of thorium from ingested plants to animals. One study estimated an assimilation coefficient of 0.0002 d/kg for beef; however, insufficient data were available to support this estimate (Rayno 1983). For this assessment, the value of 0.005 d/kg recommended by Napier et al. (1980) was used to estimate biotransfer for game animals.

A bioconcentration factor of 10,000 has been reported for algae and zooplankton in certain aquatic systems, but subsequent bioconcentration in higher members of the food chain has not been identified (Fisher et al. 1987). Considering site-specific data (DOE 1992b), a factor of 100 was used to estimate the level of thorium in algae from the raffinate pits for modeling uptake by the duck and goose in this assessment.

E.2.1.4 Uranium

Uranium can be mobilized in soil containing iron-oxidizing bacteria by conversion from insoluble uranium dioxide to the soluble uranyl ion (Scharer and Ibbotson 1982). Subsequent uptake by plants may be limited to the root system, as no significant translocation to above-ground parts has been observed (Sheppard et al. 1983; Van Netten and Morley 1983). Therefore, forage crops would not be expected to contain much uranium in their shoots. Concentrations of uranium in certain food plants can range from about 1 to 20 parts per trillion (ppt) wet weight (ATSDR 1990m), compared with a typical background level in soil of about 3 to 4 ppm. The K_d value of 330 mL/g estimated for uranium at the site (Schumacher and Stollenwerk 1991) was used in Equation E.1 to calculate a site-specific plant uptake factor of 0.0076.

Limited data are available from which to derive an animal transfer factor for uranium. A concentration of about 14 parts per trillion has been reported for beef (ATSDR 1990m). An assimilation factor of 0.005 d/kg identified for beef in Napier et al. (1980) was used to model the biotransfer of uranium by game animals in this assessment.

Bioconcentration factors of about 460 and 1,600 have been reported for plankton and algae in natural water, respectively (Mahon 1982), and assimilation efficiencies decreases significantly at higher trophic levels (Mahon 1982; Poston 1982; Waite et al. 1988). Considering

site-specific data (DOE 1992b), a factor of 100 was used to estimate the level of uranium in algae from the raffinate pits for modeling uptake by the duck and goose.

E.2.2 Metals

E.2.2.1 Antimony

Antimony does not appear to bioconcentrate in aquatic organisms, e.g., ratios of tissue concentrations to water concentrations are less than 1 (Maher 1986). These results indicate that antimony does not biomagnify in the food chain. Plant uptake from soil and subsequent translocation to the shoot is slight and appears to be correlated with the soluble or easily exchangeable fraction, which is generally low (Ainsworth 1988). Levels in most plants are less than 0.1 ppm (Adriano 1986). Concentrations in grass from control sites range from 0.1 to 0.3 ppm (Ainsworth 1988), and in another study the concentration in forage crops was 2 to 3% of the concentrations measured in grass (Ragaini et al. 1977). Plant uptake was estimated for this assessment with the value of 0.03 recommended by Baes et al. (1984).

Analyses of mammal organs indicated that antimony levels were low compared with the amount ingested (Ainsworth 1988). Two animal studies indicated that absorption following ingestion was 15 and 50% (within 24 hours), with initial excretion of about 70%. Relative to distribution, antimony could be expected to be distributed to the thyroid and adrenals, kidney, liver, spleen, and lungs following ingestion (Friberg et al. 1986). Biotransfer from ingested plants was estimated for animals in this assessment with the value of 0.003 d/kg recommended by Napier et al. (1980).

E.2.2.2 Arsenic

Plant uptake of arsenic from soil is typically low because the adsorption of arsenate by iron and aluminum soil constituents greatly restricts its availability for plant uptake (World Health Organization 1981). The presence of other soil cations and phosphorus with similar chemical properties is also considered a factor. The amount available typically averages about one-tenth of the total arsenic present in most cultivated soil (Woolson et al. 1971). For example, arsenic levels in plants are usually less than 0.5 ppm dry weight, compared with a common range in soil of 1 to 40 ppm (Bear 1965; World Health Organization 1981).

The reduced, trivalent form of arsenic may be 4 to 10 times more soluble than the pentavalent form, so it is more readily taken up by roots. However, arsenic tends to be retained in roots, with limited translocation to leaves and stems and even less to fruits. For example, arsenic concentrations in corn kernels and shelled peas were found to be below detectable levels (<0.02 ppm) in one study, and only at trace levels (<0.10 ppm) in potatoes from plots treated with 720 kg/ha. Soil levels higher than 200 to 300 ppm are needed to result in levels of 1 ppm in the tissue of edible plants, and even the highest fractions measured in plants ranged from 0.2 to 0.5 ppm (Woolson 1983). Typical concentrations in dietary intakes of produce generally range

from less than detection to 0.0032 ppm (e.g., for cereal, potatoes, leafy vegetables, legumes, root vegetables, and fruits) (Adriano 1986). Other data indicate that the concentrations can range from less than 0.01 to about 0.3 ppm in produce (Seiler and Sigel 1988). For this assessment, plant uptake was estimated with the value of 0.006 recommended by Baes et al. (1984).

Animal studies indicate that dissolved inorganic trivalent and pentavalent arsenic can be readily absorbed (80 to 90%) from the gastrointestinal tract following ingestion, with absorption of arsanilic acid and the slightly soluble trioxide lower by a factor of at least 2 to 6 (National Academy of Sciences 1977a; World Health Organization 1981; Friberg et al. 1986). However, arsenic is eliminated fairly quickly, primarily from the kidney, with elimination continuing over a second phase. For example, initial elimination can be about 75% for trivalent arsenic and 80 to 90% for pentavalent arsenic. Similarly, ingested organic arsenic can be readily absorbed but 70 to 80% is eliminated within a week. Microorganisms can reduce and methylate arsenic, facilitating excretion (as part of a detoxifying process).

Relative to the distribution of arsenic following ingestion, the highest levels in animals are found in the liver, kidney, and spleen; tissue levels decrease fairly rapidly for most organs (e.g., within 6 weeks), but levels in the skin, brain, and skeleton decrease more slowly. From one study of cows, daily doses of 0.11 to 1.0 mg/kg body weight did not increase arsenic concentrations in milk (Stevens 1991). Available data do not indicate long-term accumulation of arsenic in animals (World Health Organization 1981). Typical concentrations in dietary intakes of dairy products and meat, fish, and poultry are 0.003 and 0.02 ppm, respectively, with the latter commonly dominated by the fish component (Adriano 1986). Other data indicate that concentrations can range from 0.001 to 0.065 ppm in beef and 0.003 to 0.025 ppm in milk (Seiler and Sigel 1988). The value of 0.0013 d/kg recommended by Stevens (1992) was used for the animal transfer factor in this assessment.

E.2.2.3 Barium

Barium can bioconcentrate in marine plankton and brown algae, but little is bioconcentrated by plants growing on soil (Bowen 1966; Schroeder 1970). Concentrations in most crops range from 10 to 100 ppm dry weight (Adriano 1986). Barium concentrations in food typically parallel those of calcium, at a ratio of 1 to 100 or 100,000 (Friberg et al. 1986), although some accumulation has been reported in soybeans and tomatoes (2 to 20 times) and brazil nuts (Robinson et al. 1950). Plant uptake was estimated for this assessment with the value of 0.015 recommended by Baes et al. (1984).

Barium sulfate is essentially unabsorbed from the gastrointestinal tract of animals following ingestion, but soluble barium salts could be absorbed (e.g., 11 to 75%). Clearance can be relatively rapid (e.g., 75%) within several days during the first part of the multiphasic excretion process that extends over several weeks and then to a year. The level of barium excreted in milk of lactating cows was 0.6% within 8 hours after dosing. The barium that is retained accumulates in the skeleton (Friberg et al. 1986). Animal biotransfer was estimated for this assessment with the value of 0.000097 d/kg recommended by Ng et al. (1982).

E.2.2.4 Beryllium

Most of the beryllium taken up by plants remains in the roots; that which is translocated is distributed more to the leaves than the fruits and stems. Concentrations in plant tissues are usually less than 0.1 to 2 ppm dry weight (Adriano 1986). No evidence suggests that beryllium is significantly biomagnified within food chains (Fishbein 1981). Plant uptake was estimated for this assessment with the value of 0.0015 recommended by Baes et al. (1984).

Animal data indicate that about 80% to more than 99% of beryllium passes through the gastrointestinal tract without being absorbed (Friberg et al. 1986; ATSDR 1988). That which is absorbed can precipitate in the alkaline conditions of the intestine for subsequent excretion. From a study in which cows were injected with beryllium, less than 0.002% was recovered in the milk, with a biological half-time of 19 h. The retained beryllium is accumulated in the skeleton (Friberg et al. 1986). Animal biotransfer was estimated in this assessment with the value of 0.001 d/kg recommended by Baes et al. (1984).

E.2.2.5 Cadmium

The availability of cadmium can be limited by precipitation with the sulfide ion that is produced by biological processes in the root zone (ATSDR 1989b); its availability is also considerably depressed in calcareous soil (Adriano 1986). For example, liming soil to a pH above 6 in one study resulted in decreasing the concentration in Swiss chard from 41 to 3 ppm (Page et al. 1987). Cadmium can be taken up by plants and translocated to aboveground portions, with uptake usually less than 0.1% of amended levels in soil (Adriano 1986). For example, adding cadmium to soil at a rate of 11.2 kg/ha (which could increase natural levels by about 10 ppm) resulted in cadmium levels of 0.1 to 4.4 ppm dry weight in the edible portions of various garden plants (Page et al. 1987).

Uptake by leafy plants may be 25 to 50% of sludge-amended levels in soil, whereas fruits and seeds may be 1 to 2%; total uptake by crops is usually less than 1% of the total added (Adriano 1986). In another study, vegetables were grown on soil that was amended with fly ash and contained cadmium levels of 139 ppm; the tissue concentrations ranged from 0.01 to 0.2 ppm (Furr et al. 1976). Levels in leafy vegetables grown in uncontaminated areas ranged from 0.012 to 0.45 ppm (Elinder 1985), while levels in grass typically range from 0.03 to 0.3 ppm (Adriano 1986). The concentration of cadmium in food chain crops generally ranges from 0.01 to 1 ppm dry weight, compared with typical soil concentrations of 0.01 to 1 ppm (Page et al. 1987; Lindsay 1979). Concentrations identified for vegetables range from 0.05 to 1.2 ppm, with typical concentrations in dietary intakes of produce generally ranging from about 0.006 to 0.048 ppm and averaging about 0.02 ppm (Adriano 1986). Plant uptake was estimated for this assessment with the value of 0.15 recommended by Baes et al. (1984).

Data from animal studies indicate that 1 to 6% of ingested cadmium may be absorbed from the gastrointestinal tract. Following absorption, it is carried in the blood to the liver and kidney (e.g., 75%), and multiphasic excretion is relatively low (Friberg et al. 1986). Cadmium has been measured in the kidney and liver of moose and deer in an acid-sensitive area of

Ontario, Canada (in which cadmium availability is enhanced). Average concentrations in the liver ranged from about 0.5 to 6 ppm, with levels about 10 times higher in the kidney (Nriagu 1990). Typical concentrations in dietary intakes of dairy products and meat, fish, and poultry are 0.006 and 0.015 ppm, respectively (Adriano 1986). For this assessment, animal biotransfer was estimated with the value of 0.00017 d/kg recommended by Stevens (1992).

E.2.2.6 Chromium

Chromium in soil occurs primarily in the trivalent form, and in biological material it is almost always trivalent except during the period immediately following exposure to the more bioavailable (hexavalent) chromium (Friberg et al. 1986). Root exudates may also reduce hexavalent chromium to the trivalent species, which can then form organic complexes that facilitate uptake. In any case, very little is translocated from the roots to the shoots following uptake (Adriano 1986). Compared with typical soil concentrations of 1 to 1,000 ppm (Lindsay 1979), plant concentrations commonly range from 0.003 to 0.1 ppm. Plants with visible signs of chromium toxicity usually contain greater than 1 to 4 ppm in their leaves and 20 or more times that amount in their roots, and only a very small fraction (<1%) taken in by plant roots is translocated to the tops (Proctor and Woodell 1975; Anderson et al. 1973; Wallace et al. 1976). Various studies of aquatic systems indicate that chromium does not biomagnify in the food chain (EPA 1980a; Fishbein 1981; Schmidt and Andren 1984). Levels in the tissues of higher plants are typically less than 1 ppm dry weight (Adriano 1986). Typical concentrations in dietary intakes of produce range from about 0.05 to 2 ppm (Adriano 1986). Plant uptake was estimated in this assessment with the value of 0.0045 recommended by Baes et al. (1984).

Animal studies suggest that trivalent chromium is poorly absorbed (less than 1%) following ingestion and that hexavalent chromium is probably reduced in the gastrointestinal tract. Relative to the distribution of the limited chromium that is absorbed, the hexavalent form is rapidly reduced and biological membranes are relatively impermeable to the trivalent species bound to macromolecules, so all but chromates are rapidly excreted from the body. Those organs in which retained chromium is accumulated include the liver, spleen, and bone marrow (Friberg et al. 1986). Typical concentrations in dietary intakes of meat and dairy products range from about 0.05 to 0.23 ppm (Adriano 1986). Chromium has been shown to exert a protective effect against lead toxicity (Sandstead 1977). For this assessment, animal biotransfer was estimated with the value of 0.0019 d/kg recommended by Stevens (1992).

E.2.2.7 Cobalt

The transfer coefficient for cobalt from soil to plants has been estimated at 0.01 to 0.3, with little translocation to the aboveground portions of plants in most soil (acidic conditions can increase mobility) (ATSDR 1990d). Soil concentrations of less than 2 ppm can result in plant deficiency (Bear 1965), and typical levels in soil range from 1 to 40 ppm (Lindsay 1979). The presence of calcium carbonate suppresses cobalt uptake by plants (Adriano 1986). Little evidence exists of cobalt biomagnification in animals of upper trophic levels (Jenkins 1980). A cobalt concentration of 0.03 to 0.05 ppm dry weight has been identified for vegetables (Friberg et al.

1986). For this assessment, plant uptake was estimated with the value of 0.007 recommended by Baes et al. (1984).

Animal data indicate that cobalt absorption following ingestion is fairly limited. Less than 0.5% of cobalt oxide is absorbed following ingestion, and about 30% or less of cobalt chloride can be absorbed, with the ratio decreasing with increasing dose. The concurrent presence of iron would reduce cobalt absorption, as would the presence of amino acids because of complexation with the cobalt ion. Clearance of cobalt is usually rapid; for example, about 90% of injected chloride was excreted within 4 days in one animal study. Ingestion and injection studies indicate that retained cobalt is distributed to the liver, kidney, adrenals, and thyroid (Friberg et al. 1986). Thus, a relatively small amount would be expected to accumulate in muscle tissue. Concentrations of 0.03 to 0.05 ppm dry weight have been identified for fish, with lower concentrations of 0.01 ppm identified for meat and dairy products (Friberg et al. 1986). For this assessment, animal biotransfer was estimated with the value of 0.02 d/kg recommended by Baes et al. (1984).

E.2.2.8 Copper

Considerable experimental data indicate that copper does not biomagnify in the food chain (Perwack et al. 1980; Heit and Klusek 1985). Most plants contain less than 10 ppm dry weight (Bear 1965), and typical soil levels range from 2 to 100 ppm (Lindsay 1979). The concentration of copper in food is commonly about 1 ppm (Friberg et al. 1986); the level in leafy plants can range between 10 and 15 ppm, with levels in root vegetables usually less than 10 ppm (Adriano 1986). For this assessment, plant uptake was estimated with the value of 0.25 recommended by Baes et al. (1984).

Animal studies indicate that copper can be absorbed at levels of 50% or less, decreasing with increasing doses above 0.001 mg. Most of the absorbed copper is transported to the liver, with minor amounts transported to the bone and other tissues (Friberg et al. 1986). Thus, a relatively small amount would be expected to accumulate in muscle tissue. Copper deficiency has been shown to accentuate impacts from lead, and copper has been shown to have a protective effect against cadmium poisoning (Sandstead 1977). Concentrations in most animal foods are usually between 2 and 4 ppm; dairy products are copper-poor and generally contain less than 1 ppm, e.g., less than 10% of other food (Adriano 1986; Friberg et al. 1986). A common dietary intake is about 10 mg/d or more (Seiler and Sigel 1988). For this assessment, animal biotransfer was estimated with the value of 0.009 d/kg recommended by Ng et al. (1982).

E.2.2.9 Lead

Lead is commonly taken up by plants, but external contamination rather than root uptake from soil is the primary contributor to plant concentrations (Ter Haar 1970). Most lead that is taken up by plants remains in the root, as very little is translocated to the shoots. In one study, the maximum accumulation of lead in plants from soil containing 680 ppm was 34 ppm, and this uptake had been enhanced by the presence of a chelating agent. In another study of

forage crops, only 0.3 to 0.5% was taken up by plants (Adriano 1986). Concentrations in food may range from 0.1 to 0.5 ppm (EPA 1977), compared with typical soil levels of 2 to 200 ppm (Lindsay 1979). Levels in underground crops such as carrots and potatoes and in crops with husks or pods such as corn and beans are relatively lower than in other crops because uptake is low and atmospheric deposition (e.g., from car exhaust) is the primary means by which plants are contaminated. Concentrations in dietary intakes of produce can range from about 0.0035 to 0.3 ppm (highest in legumes), with an average of about 0.09 ppm (Adriano 1986). Another study indicates that lead concentrations in U.S. food could be lower, ranging from about 0.0001 to 0.0005 ppm (Friberg et al. 1986). Lead toxicity can be reduced by the presence of calcium (Sandstead 1977).

For this assessment, Equation E.1 was used with a site-specific estimate of 150 mL/g for the K_d (Schumacher and Stollenwerk 1991) to calculate a plant uptake factor of 0.037. Napier et al. (1980) suggest a default value of 0.068, which represents the ratio of the concentration in fresh (wet weight) vegetation to that in dry soil.

Animal studies indicate that 5 to 10% of ingested lead can be absorbed from the gastrointestinal tract and that the presence of calcium decreases absorption. Absorbed lead is eliminated primarily from the kidney (about 76%), and retained lead is initially distributed to either blood and soft tissue (e.g., the liver, kidney, and brain) or the bone, with most eventually ending up in the latter (Friberg et al. 1986). Thus, a relatively small amount would be expected to accumulate in muscle tissue. A level of 0.015 ppm has been identified for the dietary intake of meat, fish, and poultry (Adriano 1986); in another study, the concentration in beef was 0.06 ppm (Friberg et al. 1986). Concentrations identified for dairy products range from not detected (Adriano 1986) to about 0.03 ppm (0.01 ppm in milk) (Friberg et al. 1986). Ng et al. (1982) identify a range of 0.0001 to 0.0007 d/kg for lead in cow meat. For this assessment, the value of 0.000067 d/kg recommended by Stevens (1992) was used to estimate the biotransfer of lead from ingested plants to animals.

E.2.2.10 Lithium

Little information was found in the literature for estimating plant uptake or animal assimilation of lithium. In general, lithium can be taken up by plants relatively easily. For this assessment, plant uptake was estimated with the value of 0.004 recommended by Baes et al. (1984). The levels of lithium in game animals were estimated for this assessment with the transfer factor of 0.01 d/kg identified by Baes et al. (1984).

E.2.2.11 Manganese

Manganese appears to be taken up only in the divalent form, and plant tissue concentrations can range from 5 to 1,000 ppm or more dry weight (Bear 1965), compared with typical soil concentrations of 20 to 3,000 ppm (Lindsay 1979). The availability of manganese from neutral and alkaline soil is limited (Robson and Pitman 1983), especially in the presence of calcium carbonate (Adriano 1986), so plant uptake from site soil is expected to be limited.

Higher trophic levels tend to maintain homeostasis, so the potential for biomagnification is quite low (ATSDR 1990h). Average concentrations of manganese in produce have been given as 1.5 ppm for nonleafy vegetables, 2 to 4 ppm for fruits, 5 ppm for leafy vegetables, and 20 ppm for dried legume seeds (Adriano 1986). For this assessment, plant uptake was estimated with the value of 0.05 recommended by Baes et al. (1984).

Animal studies indicate that absorption following ingestion is homeostatically controlled, e.g., varying from 1 to 3.5% depending on the amount ingested. Most of the absorbed manganese is excreted; that which is retained is distributed from the blood to the liver, with some subsequent distribution to other locations such as the kidney and brain. (In one human study, about one-third was found in muscle tissue.) However, tissues and organs do not tend to accumulate large amounts of manganese (Friberg et al. 1986). Concentrations in dairy products and animal tissue average 0.5 and 1 ppm, respectively (wet weights) (Adriano 1986). Animal biotransfer was estimated for this assessment with the value of 0.0004 d/kg recommended by Baes et al. (1984).

E.2.2.12 Mercury

The availability of mercury to plants is generally low, and it is retained in roots with little translocation to the shoots (Adriano 1986). Mercury does not appear to accumulate in higher plants, even when grown on contaminated soil (Diez and Rosopulo 1976). The concentrations of mercury in dietary intakes of land-produced food are low, ranging from less than 0.01 to 0.03 ppm (Adriano 1986). Plant uptake was estimated for this assessment with the value of 0.2 recommended by Baes et al. (1984).

Animal studies indicate that if elemental mercury were taken in, it would be rapidly oxidized to mercuric mercury and subsequently reduced and would bind to proteins, thereby limiting mobility. A limited amount of mercury can be absorbed following ingestion, depending on the compound (e.g., about 2% for mercuric chloride). After absorption, the rate of accumulation and elimination from various organs is highly variable from one species to another. The general trend is for most to be deposited in the kidney, with the liver being the next important; other distribution sites include the skin, spleen, and brain. Organic compounds tend to accumulate in the brain and can be deposited in hair (Friberg et al. 1986). Thus, little would be expected to accumulate in muscle tissue. Relative to potential dietary contribution for humans, fish and shellfish from contaminated water are a major source of mercury in the diet. Typical concentrations for dairy products and for meat, fish, and poultry (dominated by fish) are less than 0.001 and 0.017 ppm, respectively (Adriano 1986). For this assessment, animal biotransfer was estimated with the value of 0.00035 d/kg recommended by Stevens (1992).

E.2.2.13 Molybdenum

Molybdenum plays an important role in the metabolism of nitrogen by plants, and some plants such as legumes can accumulate the element. After uptake by roots, more molybdenum is retained in the roots than is translocated; of that which is translocated, more is accumulated

in the leaves, less is distributed to the stems, and the lowest concentrations are found in the seed or fruit (Adriano 1986). Molybdenum concentrations in soil typically range from 0.1 to 10 ppm (Friberg et al. 1986), and most forage plants contain 1 to 2 ppm dry weight (Bear 1965). The daily dietary intake of molybdenum by humans is estimated at 0.13 ppm, with concentrations of less than 0.1 to 70 ppm identified for vegetables (Adriano 1986). Plant uptake was estimated for this assessment with the value of 0.06 recommended by Baes et al. (1984).

Animal studies indicate that absorption from the gastrointestinal tract following ingestion can be high. For example, absorption of hexavalent molybdenum can range from 40 to 85%. Little of the absorbed molybdenum is retained, with relatively complete excretion and a biological half-time on the order of hours to a day; excretion in milk was about 2% in one goat study. The molybdenum that is retained is subsequently accumulated in the kidney, liver, and bone (Friberg et al. 1986). Thus, little would be expected to accumulate in muscle tissue. The levels of molybdenum in animal products are low (Friberg et al. 1986), and dietary concentrations of 0.02 to 0.03 ppm have been identified for cow milk (Adriano 1986). Animal biotransfer was estimated for this assessment with the value of 0.0068 d/kg recommended by Ng et al. (1982).

E.2.2.14 Nickel

Nickel taken up by plant roots can be translocated to aboveground portions and can accumulate in seeds at senescence (Adriano 1986). The concentration of nickel in plant tissues commonly ranges from 0.1 to 5 ppm dry weight (Bear 1965), compared with typical soil concentrations of 5 to 500 ppm (Lindsay 1979). Nickel is not expected to biomagnify in the food chain. The concentration of nickel in crops ranges from about 0.05 to 5 ppm, with levels generally averaging less than 0.8 ppm (Adriano 1986); other data indicate that levels in vegetables and fruits range from 0 to 2.6 ppm (Friberg et al. 1986). Plant uptake was estimated for this assessment with the value of 0.06 recommended by Baes et al. (1984).

Animal studies indicate that toxicity via the oral route is low and that homeostatic mechanisms play a role in controlling absorption and excretion, to dietary levels as high as 1,000 ppm (Nielsen 1977). Absorption from the gastrointestinal tract following ingestion is limited, e.g., about 10% (Friberg et al. 1986), and nickel shows little tendency to accumulate in tissues (Nielsen 1977). Thus, little would be expected to accumulate in muscle tissue. The concentrations in daily intakes of dairy products and meat are estimated at 0.36 and 0.09 ppm, respectively, with the total dietary intake ranging from 0.15 to 0.8 ppm (Adriano 1986). Friberg et al. (1986) indicate that the daily dietary intake may range from 0.3 to 0.5 μg/d. For this assessment, animal biotransfer was estimated with the value of 0.002 d/kg recommended by Ng et al. (1982).

E.2.2.15 Selenium

Selenium can be readily taken up by plants, with uptake generally inversely related to clay content. Selenium is also associated with organic matter as a result of nutrient cycling from

decaying material, so it is often retained in surface soil where it can be more available for plant uptake. Certain plants in semi-arid, nonagricultural soil can grow well on soil with high levels of selenium and can accumulate it in their tissues. Concentrations in plant tissues usually decline with maturity. Median concentrations in dietary intakes of leafy and root vegetables, legumes, potatoes, and garden fruits are about 0.017 ppm (Adriano 1986). Other data indicate that vegetables and fruits generally contain less than 0.01 ppm selenium (Seiler and Sigel 1988). Plant uptake was estimated for this assessment with the value of 0.025 recommended by Baes et al. (1984).

Animal studies indicate that water-soluble compounds such as selenite can be absorbed fairly easily from the gastrointestinal tract. For example, 30 to 90% of ingested selenium can be absorbed, with subsequent distribution to most organs. Accumulation in tissues including muscle decreased after the first day, and selenite compounds accumulated more quickly and in higher concentrations in the liver and kidneys than did selenate compounds. Selenium can be biotransformed within the body and subsequently exhaled. Overall elimination can be biphasic, with the first phase occurring in the first few days and the second over 1 to 2 months (Friberg et al. 1986). Thus, little would be expected to accumulate in muscle tissue. The median concentration in dietary intakes for meat, fish, and poultry is 0.17 ppm and that for dairy products is 0.02 ppm (Adriano 1986). Selenium has been shown to exert a protective effect against the toxicity of mercury, arsenic, and cadmium (Sandstead 1977). For this assessment, animal biotransfer was estimated with the value of 0.015 d/kg recommended by Baes et al. (1984).

E.2.2.16 Silver

Silver is considered generally unavailable to plants because of its general insolubility and adsorption processes, and no evidence indicates that it biomagnifies in terrestrial systems (Adriano 1986; ATSDR 1990k). Silver taken up by plants is generally retained in the roots because it is poorly translocated to the shoot. Plant tissues typically contain less than 1 ppm of silver; in one study, concentrations ranged from 0.01 to 0.16 ppm (dry weight), with the lower species containing the higher amounts (Adriano 1986). For this assessment, plant uptake was estimated with the value of 0.1 recommended by Baes et al. (1984).

Animal studies indicate that absorption following ingestion is limited, e.g., less than 10% may be absorbed. That which is retained could be distributed to the liver and kidney (Friberg et al. 1986); thus, little would be expected to accumulate in muscle tissue. Dietary intake of silver has been estimated at about 1 to 16 μg/d (Friberg et al. 1986). Animal biotransfer was estimated for this assessment with the value of 0.002 d/kg recommended by Ng et al. (1982).

E.2.2.17 Thallium

The current understanding of thallium's biogeochemistry relative to the food chain is limited. Various studies have indicated that thallium can be absorbed from soil by terrestrial plants (Sharma et al. 1986; Ewers 1988), but data have not suggested biomagnification in the food

chain (ATSDR 1990*l*). Plant uptake studies indicate that tissue concentrations may be correlated with soil concentrations, e.g., increasing to 15 to 20 ppm (dry weight) for spinach and chinese cabbage then leveling off as soil concentrations increase from 2 to 5 ppm in one study (Schoer 1984). Tissue concentrations for a range of terrestrial plants indicate that 0.05 ppm may be a typical level; crops grown in an uncontaminated area had concentrations of 0.02 to 0.04 ppm dry weight, with leek, endive, and kale ranging from 0.075 to 0.125 ppm (Adriano 1986). Other data indicate that 0.5 ppm could be a typical concentration in plants (Seiler and Sigel 1988). The mobility of thallium at the site is expected to be limited by the presence of clays and the neutral to alkaline pH. For this assessment, plant uptake was estimated with the value of 0.0004 recommended by Baes et al. (1984).

Animal studies indicate that thallium can be readily absorbed from the gastrointestinal tract following ingestion. In one study, an initial excretion of 6% dropped to less than 1% within 10 days (Friberg et al. 1986). Thallium that is retained is distributed to various organs and tissues, with the highest concentrations found in the kidney followed by bone, liver, lung, spleen, and brain (Friberg et al. 1986). Thus, little would be expected to accumulate in muscle tissue. From very limited data, the dietary intake of thallium has been estimated at about 2 µg/d (Friberg et al. 1986). Animal biotransfer was estimated for this assessment with the value of 0.04 d/kg recommended by Baes et al. (1984).

E.2.2.18 Vanadium

Biotransformation is not considered a significant process for vanadium, and plant studies indicate that uptake into aboveground plant parts is low and biomagnification is unlikely (ATSDR 1990p). Most vanadium is retained in the roots of plants at levels that may be similar to that in soil (World Health Organization 1988); translocation from roots to shoots in plants grown on calcareous soils is even more limited than usual (Adriano 1986). Seiler and Sigel (1988) report that concentrations in aboveground plant tissues do not correlate to soil concentration and that 0.16 ppm could represent a mean concentration in higher plants. Plant tissue concentrations typically range from 0.5 to 2 ppm dry weight. One study found levels of less than detection to 6.6 ppm dry weight in plants grown on soil with concentrations of 0.65 to 98 ppm; it was concluded that soil with high levels of vanadium would not pose a hazard for bioaccumulation in the food chain. Concentrations in fruits and vegetables generally range from 0.00003 (tomato) to 0.14 (navy bean) and average less than 0.001 ppm (Adriano 1986). Other data indicate that the concentrations in various food including fresh fruit and vegetables ranged from less than 0.001 to 0.005 ppm (Seiler and Sigel 1988). Plant uptake was estimated for this assessment with the value of 0.003 recommended by Baes et al. (1984).

Animal studies indicate that vanadium compounds are poorly absorbed following ingestion, e.g., from less than 1 to 2.6% in one study with the pentavalent oxide (Friberg et al. 1986). Vanadium can be distributed to the liver and kidney following absorption. Homeostatic mechanisms appear to exert some control on absorption such that tissue levels remain low. That which is retained is commonly distributed to the liver and kidney. Vanadium concentrations in the tissues of land mammals are so low that detection is difficult (World Health Organization 1988). From very limited data, a concentration of 0.4 ppm dry weight has been identified for

mammals and mean levels of 0.25 and 0.94 have been given for the liver and kidney of beaver, deer, woodchuck, rabbit, muskrat, and fox (World Health Organization 1988). A concentration of 0.15 ppm has been reported for insects, and no concentrations were detected in mammals from another study (<0.4 ppm) (Waters 1977). Total daily intakes of vanadium are estimated at about 50 μg/d. The average concentration in beef may be 0.001 ppm, with levels ranging from less than 0.0004 to 1.3 ppm; dairy products may contain 0.0002 ppm, with levels ranging from less than 0.0001 to 0.003 ppm (Adriano 1986; Friberg et al. 1986; World Health Organization 1988). Animal biotransfer was estimated for this assessment with the value of 0.0025 d/kg recommended by Baes et al. (1984).

E.2.2.19 Zinc

Zinc concentrations in plants commonly range from 25 to 75 ppm dry weight (Bear 1965), compared with typical soil levels of 10 to 300 ppm (Lindsay 1979). Following uptake, zinc tends to be fairly evenly distributed between roots and shoots. Average concentrations in dietary intakes of produce generally range from about 3 to 8 ppm (Adriano 1986). For this assessment, plant uptake was estimated with the value of 0.9 recommended by Baes et al. (1984).

Animal studies suggest that absorption of zinc from the gut is homeostatically controlled, with factors such as zinc status (deficiency) and low body weight increasing absorption; data indicate variable absorption ranging from less than 10 to more than 90%. The rate of elimination increases with the amount ingested. Similar homeostatic controls have been indicated for humans (Seiler and Sigel 1988). As a note, zinc has been shown to exert a protective effect against mercury toxicity (Sandstead 1977). From one animal study, the highest concentration of zinc in mice after 1 week was found in bone, followed by liver and kidney (Friberg et al. 1986). Thus, little would be expected to accumulate in muscle tissue. Average concentrations in dietary intakes of dairy products and meat, fish, and poultry are about 5 and 30 ppm, respectively (Adriano 1986). For this assessment, animal biotransfer was estimated with the value of 0.098 d/kg recommended by Ng et al. (1982).

E.2.3 Inorganic Anions

E.2.3.1 Fluoride

Fluoride uptake by plants is usually related not to soil concentrations but to the soil type and content of calcium and phosphorus (Brewer 1965); e.g., more fluoride could be available from soil with a low pH and high content of fluoride-containing apatite (a calcium-phosphate mineral). In addition, most fluorine toxicity in plants is associated with atmospheric fluorine rather than root uptake (Bear 1965). Because site soil is not acidic, fluoride availability is not expected to be high. For this assessment, plant uptake and animal biotransfer were estimated with the values of 0.006 and 0.15 d/kg, respectively, from Baes et al. (1984).

E.2.3.2 Nitrate and Nitrite

Nitrate is a basic plant nutrient. Following uptake, nitrate and nitrite are assimilated and converted by reductases to organic forms such as glutamate as part of the basic nutrient cycling process (Beevers and Hageman 1983). Thus, subsequent uptake of the initial inorganic anions by animals would not occur; i.e., the animal would ingest organic nitrogen that would in turn be converted to compounds such as amino acids. Therefore, this uptake was not modeled. (The critical health effect identified for nitrate and nitrite — methemoglobinemia — is associated with ingestion of nitrate from drinking water, not ingestion of organic nitrogen from food.)

E.2.4 Polycyclic Aromatic Hydrocarbons

Biomagnification, i.e., the systematic increase in tissue concentrations moving up a food chain, has not been indicated for PAHs (Eisler 1987). Depending on availability, PAHs can accumulate in terrestrial plants with ratios of PAH concentration in vegetation to those in soil ranging from 0.001 to 0.18 for total PAHs and from 0.002 to 0.33 for benzo(a)pyrene (Edwards 1983). For this assessment, plant uptake and animal transfer factors were developed from the K_{ow} and/or K_{oc} values for each organic compound with the estimation methods of Briggs et al. (1982) and Travis and Arms (1988). The plant uptake factors for the individual PAHs ranged from about 0.00008 to 0.1, and the animal biotransfer factors ranged from about 0.00005 to 0.08 d/kg.

E.2.5 Polychlorinated Biphenyls

The availability of PCBs from soil is limited by their strong adsorptive tendencies. A study of PCB uptake by plants suggested that little or no root absorption and translocation occurs; results for the animal component indicated that soil consumption by grazing cows could reach 14% of dry intake, with typical annual dietary concentrations of 0.045 ppm per ppm PCB residue in soil (Fries 1982). For this assessment, plant uptake and animal transfer factors were developed from the K_{ow} and/or K_{oc} values for each compound with the estimation methods of Briggs et al. (1982) and Travis and Arms (1988). The plant uptake factors for the individual Aroclors ranged from about 0.0002 to 0.001, and the animal biotransfer factors ranged from 0.014 to 0.35 d/kg.

E.2.6 Nitroaromatic Compounds

The bioconcentration factor for NB is negligible (ATSDR 1990i). In one biouptake study with soybean, NB residues and metabolites remained largely in the roots (McFarlane et al. 1987). On the basis of a plant toxicity of NB (e.g., EC_{50} of about 200 mg/L for cucumber and mung bean [Eckert 1962]), toxicity would be expected to limit the amount that could be transferred to higher trophic levels. For this assessment, plant uptake and animal biotransfer factors were developed for the nitroaromatic compounds from the K_{ow} and/or K_{oc} values for each compound

with the estimation methods of Briggs et al. (1982) and Travis and Arms (1988). The plant uptake and animal transfer factors estimated for NB were 2 and 1.9×10^{-6} d/kg, respectively. The plant uptake factors estimated for DNB and TNB were 4.5 and 6.5, and the animal transfer factors were about 8×10^{-7} and 6×10^{-7} d/kg.

The DNT compounds could be taken up by plants, but they are not expected to bioaccumulate in animal tissues (ATSDR 1989d). The availability of these compounds at the site is expected to be limited by biodegradation and photolysis, and adsorption would also be a factor. The plant uptake factors determined for 2,4-DNT and 2,6-DNT were 1.6 and 0.78, and the corresponding animal transfer factors were 2.4×10^{-6} and 4.8×10^{-6} d/kg. For TNT, the estimation methods yielded values of 3.6 and 1×10^{-6} d/kg for plant uptake and animal transfer factors.

E.2.7 Selected Uptake and Biotransfer Factors

The uptake and biotransfer factors used to model the plant and animal components of the game ingestion pathway are listed in Table E.2.

TABLE E.2 Estimated Plant Uptake and Animal Biotransfer Factors

Contaminant	Plant Uptake Factor[a]	Animal Biotransfer Factor[b] (d/kg)	Source[c]
Radionuclides			
Actinium	8.8×10^{-4}	5.0×10^{-3}	Thibault et al. (1990); Napier et al. (1980)
Protactinium	6.3×10^{-4}	5.0×10^{-3}	Thibault et al. (1990); Napier et al. (1980)
Radium	3.0×10^{-2}	4.0×10^{-3}	Sheppard and Thibault (1990); Rayno (1983)
Thorium	4.2×10^{-3}	5.0×10^{-3}	Napier et al. (1980)
Uranium	7.6×10^{-3}	5.0×10^{-3}	Sheppard and Thibault (1990); Napier et al. (1980)
Metals			
Antimony	3.0×10^{-2}	1.0×10^{-3}	Baes et al. (1984)
Arsenic	6.0×10^{-3}	1.3×10^{-3}	Baes et al. (1984); Stevens (1992)
Barium	1.5×10^{-2}	9.7×10^{-5}	Baes et al. (1984); Ng et al. (1982)
Beryllium	1.5×10^{-3}	1.0×10^{-3}	Baes et al. (1984)
Cadmium	1.5×10^{-1}	1.7×10^{-4}	Baes et al. (1984); Stevens (1992)
Chromium	4.5×10^{-3}	1.9×10^{-3}	Baes et al. (1984); Stevens (1992)
Cobalt	7.0×10^{-3}	2.0×10^{-2}	Baes et al. (1984)
Copper	2.5×10^{-1}	9.0×10^{-3}	Baes et al. (1984); Ng et al. (1982)
Lead	3.7×10^{-2}	6.7×10^{-5}	Sheppard and Thibault (1990); Stevens (1992)
Lithium	4.0×10^{-3}	1.0×10^{-2}	Baes et al. (1984)
Manganese	5.0×10^{-2}	4.0×10^{-4}	Baes et al. (1984)
Mercury	2.0×10^{-1}	3.5×10^{-4}	Baes et al. (1984); Stevens (1992)
Molybdenum	6.0×10^{-2}	6.8×10^{-3}	Baes et al. (1984); Ng et al. (1982)
Nickel	6.0×10^{-2}	2.0×10^{-3}	Baes et al. (1984); Ng et al. (1982)
Selenium	2.5×10^{-2}	1.5×10^{-2}	Baes et al. (1984)
Silver	1.0×10^{-1}	2.0×10^{-3}	Baes et al. (1984); Ng et al. (1982)
Thallium	4.0×10^{-4}	4.0×10^{-2}	Baes et al. (1984)
Vanadium	3.0×10^{-3}	2.5×10^{-3}	Baes et al. (1984)
Zinc	9.0×10^{-1}	9.8×10^{-2}	Baes et al. (1984); Ng et al. (1982)
Inorganic anions			
Fluoride	6.0×10^{-3}	1.5×10^{-1}	Baes et al. (1984)
PAHs			
Acenaphthene	2.5×10^{-2}	2.4×10^{-4}	Mabey et al. (1982)
Anthracene	1.0×10^{-2}	7.1×10^{-4}	Mabey et al. (1982)
Benz(a)anthracene	1.5×10^{-3}	1.0×10^{-2}	Mabey et al. (1982)
Benzo(b)fluoranthene	8.1×10^{-4}	2.9×10^{-2}	Mabey et al. (1982)
Benzo(k)fluoranthene	8.1×10^{-4}	2.9×10^{-2}	Mabey et al. (1982)
Benzo(g,h,i)perylene	4.2×10^{-4}	8.1×10^{-2}	Mabey et al. (1982)
Benzo(a)pyrene	8.1×10^{-5}	2.9×10^{-2}	Mabey et al. (1982)
Chrysene	1.5×10^{-3}	1.0×10^{-2}	Mabey et al. (1982)
Fluoranthene	4.9×10^{-3}	2.0×10^{-3}	Mabey et al. (1982)
Fluorene	1.8×10^{-2}	3.8×10^{-4}	Mabey et al. (1982)
Indeno(1,2,3-cd)pyrene	4.2×10^{-4}	8.1×10^{-2}	Mabey et al. (1982)
2-Methylnaphthalene	1.6×10^{-2}	3.8×10^{-4}	Lyman et al. (1990)
Naphthalene	1.0×10^{-1}	4.9×10^{-5}	Mabey et al. (1982)
Phenanthrene	1.0×10^{-2}	7.1×10^{-4}	Mabey et al. (1982)
Pyrene	4.9×10^{-3}	2.0×10^{-3}	Mabey et al. (1982)

TABLE E.2 (Cont.)

Contaminant	Plant Uptake Factor[a]	Animal Biotransfer Factor[b] (d/kg)	Source[c]
PCBs			
Aroclor 1248	1.3×10^{-3}	1.4×10^{-2}	Mabey et al. (1982)
Aroclor 1254	8.2×10^{-4}	2.8×10^{-2}	Mabey et al. (1982)
Aroclor 1260	1.9×10^{-4}	3.5×10^{-1}	Mabey et al. (1982)
Nitroaromatic compounds			
DNB	4.5	7.8×10^{-7}	Verschueren (1983)
2,4-DNT	1.6	2.4×10^{-6}	Mabey et al. (1982)
2,6-DNT	7.8×10^{-1}	4.8×10^{-6}	Mabey et al. (1982)
NB	2.0	1.9×10^{-6}	Mabey et al. (1982)
TNB	6.5	5.6×10^{-7}	Lyman et al. (1990)
TNT	3.6	1.0×10^{-6}	McKone (1990)

[a] The plant uptake factor represents the ratio of the concentration in plant tissue (pCi/kg for radionuclides and mg/kg for chemicals) to the concentration in soil (pCi/kg for radionuclides and mg/kg for chemicals). Values from Baes et al. (1984) and Sheppard and Thibault (1990) assume dry weights for both soil and plant tissue. Values from Napier et al. (1980) assume dry weight for soil and wet (fresh) weight for plant tissue. For the analyses in this BA, the wet weight of forage was assumed to be 10 times the dry weight; the values reported for soil in Chapters 2 and 3 of this BA are dry weights.

[b] The animal biotransfer factor represents the fraction of the contaminant ingested daily with dietary plant tissue that is transferred to and remains in animal muscle tissue. It is represented by the ratio of the concentration in the animal tissue (pCi/kg for radionuclides and mg/kg for chemicals) per daily intake of contaminant from plant ingestion (pCi/d for radionuclides and mg/d for chemicals). Values from Baes et al. (1984), Ng et al. (1982), and Stevens (1992) assume dry weight for plant tissue and wet (fresh) weight for animal tissue. Values from Napier et al. (1980) assume wet weight for both plant and animal tissue. The human intake calculations in this BA assume the individual ingests animal tissue on a wet weight basis.

[c] The sources of the plant uptake and animal biotransfer factors for radionuclides, metals, and fluoride are listed in order where two are given. The plant uptake factors for radium, uranium, and lead were derived from site-specific K_d values with information from the listed source (Equation E.1). For the organic contaminants (PAHs, PCBs, and nitroaromatic compounds), the source of the K_{ow} value from which the plant uptake and biotransfer factors were derived is listed. The plant uptake factor was calculated from the K_{ow} value with the estimation method of Briggs et al. (1982) and a site-specific value of 1.4% for organic carbon in soil; the animal biotransfer factor was calculated from the K_{ow} value with the estimation method of Travis and Arms (1988); adjustments were made for plant water content, which was assumed to be 90% (Section 3.3.5.2).

E.3 REFERENCES (APPENDIX E)

Adriano, D.C., 1986, *Trace Elements in the Terrestrial Environment*, Springer-Verlag, New York.

Adriano, D.C., and H.E. Doner, 1982, *Bromine, Chlorine, and Fluorine*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 449-483.

Agency for Toxic Substances and Disease Registry, 1988, *Toxicological Profile for Beryllium*, ATSDR/TP-88/07, prepared by Syracuse Research Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1989a, *Toxicological Profile for Arsenic*, ATSDR/TP-88/02, prepared by Life Systems, Inc., March.

Agency for Toxic Substances and Disease Registry, 1989b, *Toxicological Profile for Cadmium*, ATSDR/TP-88/08, prepared by Life Systems, Inc., March.

Agency for Toxic Substances and Disease Registry, 1989c, *Toxicological Profile for Chromium*, ATSDR/TP-88/10, prepared by Syracuse Research Corporation, July.

Agency for Toxic Substances and Disease Registry, 1989d, *Toxicological Profile for 2,4-Dinitrotoluene, 2,6-Dinitrotoluene*, ATSDR/TP-89/13, prepared by Clement Associates, Dec.

Agency for Toxic Substances and Disease Registry, 1989e, *Toxicological Profile for Mercury*, ATSDR/TP-89/16, prepared by Clement Associates, Dec.

Agency for Toxic Substances and Disease Registry, 1989f, *Toxicological Profile for Selected PCBs (Aroclor-1260, -1254, -1248, -1242, -1232, -1221, and -1016)*, ATSDR/TP-88/21, prepared by Syracuse Research Corporation, June.

Agency for Toxic Substances and Disease Registry, 1989g, *Toxicological Profile for Selenium*, ATSDR/TP-89/21, prepared by Clement Associates, Dec.

Agency for Toxic Substances and Disease Registry, 1989i, *Draft Toxicological Profile for Nitrobenzene*, prepared by Life Systems, Inc., Oct.

Agency for Toxic Substances and Disease Registry, 1990a, *Toxicological Profile for Antimony*, prepared by Syracuse Research Corporation, draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990c, *Toxicological Profile for Barium and Compounds*, prepared by Clement Associates, Inc., draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990d, *Toxicological Profile for Cobalt*, prepared by Clement Associates, draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990e, *Toxicological Profile for Copper*, ATSDR/TP-90/08, prepared by Syracuse Research Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1990h, *Toxicological Profile for Manganese*, prepared by Life Systems, Inc., draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990i, *Toxicological Profile for Nitrobenzene*, ATSDR/TP-90/19, prepared by Life Systems, Inc., Dec.

Agency for Toxic Substances and Disease Registry, 1990j, *Toxicological Profile for Polycyclic Aromatic Hydrocarbons*, ATSDR/TP-90/20, prepared by Clement International Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1990k, *Toxicological Profile for Silver*, ATSDR/TP-90/24, prepared by Clement International Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1990l, *Toxicological Profile for Thallium*, prepared by Life Systems, Inc., draft for public comment, Oct.

Agency for Toxic Substances and Disease Registry, 1990m, *Toxicological Profile for Uranium*, ATSDR/TP-90/29, prepared by Syracuse Research Corporation, Dec.

Agency for Toxic Substances and Disease Registry, 1990n, *Draft Toxicological Profile for Thorium*, ATSDR/TP-90-25, prepared by Syracuse Research Corporation, Oct.

Agency for Toxic Substances and Disease Registry, 1990o, *Toxicological Profile for Radium*, ATSDR/TP-90-22, prepared by Life Systems, Inc., Dec.

Agency for Toxic Substances and Disease Registry, 1990p, *Draft Toxicological Profile for Vanadium and Compounds*, prepared by Clement Associates, Inc., draft for public comment, Oct.

Ainsworth, N., 1988, *Distribution and Biological Effects of Antimony in Contaminated Grassland*, Dissertation. [As cited in ATSDR (1990a).]

Ames, L.L., and D. Rai, 1978, *Radionuclide Interactions with Rock and Soil Media*, Vol. I, EPA/520/6-78-007a.

Anderson, A.J., and A.C. Oertel, 1946, *Plant Responses to Molybdenum as a Fertilizer*, Council for Scientific and Industrial Research Bulletin, *198*:25-44. [As cited in Robson and Pitman (1983).]

Anderson, A.J., D.R. Meyer, and F.K. Mayer, 1973, *Heavy Metal Toxicities: Levels of Nickel, Cobalt, and Chromium in the Soil and Plants Associated with Visual Symptoms and Variation in Growth of an Oat Crop*, Australian Journal of Agricultural Research, 24:557-571. [As cited in Reisenauer (1982).]

Andreae, M.O., 1979, *Arsenic Speciation in Seawater and Interstitial Waters: The Influence of Biological-Chemical Interactions on the Chemistry of a Trace Element*, Limnology and Oceanography, 24:440-452. [As cited in Turner (1987).]

Andreae, M.O., 1983, *The Determination of the Chemical Species of Some of the "Hybride Elements" (Arsenic, Antimony, Tin and Germanium) in Seawater: Methodology and Results*, NATO Conference 4, 9:1-19. [As cited in ATSDR (1990a).]

ATSDR: See Agency for Toxic Substances and Disease Registry.

Baes, C.F., and R.D. Sharp, 1983, *A Proposal for Estimation of Soil Leaching and Leaching Constants for Use in Assessment Models*, Journal of Environmental Quality, 12(1):17-28.

Baes, C.F., et al., 1984, *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionucleotides through Agriculture*, ORNL-5786, prepared by Oak Ridge National Laboratory, Oak Ridge, Tenn., for U.S. Department of Energy, Oak Ridge Operations, Oak Ridge, Tenn.

Baker, D.E., and M.C. Amacher, 1982, *Nickel, Copper, Zinc, and Cadmium*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 323-336.

Bear, F.E., 1965, *Soils in Relation to Crop Growth*, Reinhold Publishing Corporation, New York.

Beevers, L., and R.H. Hageman, 1983, *Uptake and Reduction of Nitrate: Bacteria and Higher Plants*, in Inorganic Plant Nutrition, A. Läuchli and R.L. Bieleski (editors), Encyclopedia of Plant Physiology, New Series Vol. 15 A, Springer-Verlag, Berlin.

Benes, P., P. Strejc, and Z. Lukavec, 1984, *Interaction of Radium with Freshwater Sediments and Their Mineral Components: I. Ferric Hydroxide and Quartz*, Journal of Radioanalytical and Nuclear Chemistry, 82:275-285.

Benes, P., Z. Borovek, and P. Strejc, 1985, *Interaction of Radium with Freshwater Sediments and Their Mineral Components: II. Kaolinite and Montmorillonite*, Journal of Radioanalytical and Nuclear Chemistry, 89:339-351.

Benes, P., Z. Borovek, and P. Strejc, 1986, *Interaction of Radium with Freshwater Sediments and Their Mineral Components: III. Muscovite and Feldspar*, Journal of Radioanalytical and Nuclear Chemistry, 98:91-103.

Bodek, I., et al. (editors), 1988, *Environmental Inorganic Chemistry: Properties, Processes and Estimation Methods*, SETAC Special Publication Series, Pergamon Press, New York.

Bowen, H.J.M., 1966, *Trace Elements in Biochemistry*, Academic Press, Inc., New York. [As cited in ATSDR (1990c).]

Brewer, R.F., 1965, *Fluorine*, in Diagnostic Criteria for Plants and Soils, H.D. Chapman (editor), Quality Printing Co., Inc., Abilene, Texas, pp. 180-196. [As cited in Adriano and Doner (1982).]

Briggs, G.G., R.H. Bromilow, and A.A. Evans, 1982, *Relationships between Lipophilicity and Root Uptake and Translocation of Non-ionised Chemicals by Barley*, Pesticide Science, 13:495-504.

Burau, R.G., 1982, *Lead*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, American Society of Agronomy, Inc., Madison, Wisc., pp. 347-365.

Burrows, E.P., et al., 1989, *Organic Explosives and Related Compounds: Environmental and Health Considerations*, Technical Report 8901, prepared for the U.S. Army Biomedical Research & Development Laboratory/Command, Fort Detrick, Frederick, Md., March.

Butler, J.N., 1964, *Ionic Equilibrium, A Mathematical Approach*, Addison-Wesley Publishing Company, Reading, Calif.

Callahan, M.A., et al., 1979, *Water-Related Environmental Fate of 129 Priority Pollutants*, EPA 440/4-79-029a, Office of Water Planning and Standards, Washington, D.C.

Cotton, F.A., and G. Wilkinson, 1972, *Advanced Inorganic Chemistry*, Interscience Publishers, New York, pp. 245-254. [As cited in ATSDR (1988a).]

Diez, T., and A. Rosopulo, 1976, *Schwermetallgehalte in Böden und Pflanzen nach extrem hohen Klärschlammgaben*, Landwirtsch Forsch Sonderheft, 22(1):236-248. [As cited in Marschner (1983).]

DOE: see U.S. Department of Energy.

Eary, L.M., and D. Rai, 1987, *Kinetics of Chromium (III) Oxidation to Chromium (IV) by Reaction with Manganese Oxide*, Environmental Science and Technology, 21:1187-1193.

Eckert, J.W., 1962, *Fungistatic and Phytotoxic Properties of Some Derivatives of Nitrobenzene*, Phytopathology, 52:642-649. [As cited in ATSDR (1989i).]

Edwards, N.T., 1983, *Polycyclic Aromatic Hydrocarbons (PAHs) in the Terrestrial Environment: A Review*, Journal of Environmental Quality, 12:427-441. [As cited in ATSDR (1990j).]

Eisler, R., 1987, *Polycyclic Aromatic Hydrocarbon Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review*, U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Md. [As cited in ATSDR (1990j).]

Elinder, C.G., 1985, *Cadmium: Uses, Occurrence, and Intake*, in Cadmium and Health: A Toxicological and Epidemiological Appraisal, Vol. I: Exposure, Dose, and Metabolism, L. Friberg et al. (editors), CRC Press, Boca Raton, Fla., pp. 81-102. [As cited in ATSDR (1989b).]

EPA: see U.S. Environmental Protection Agency.

Ewers, U., 1988, *Environmental Exposure to Thallium*, Science of the Total Environment, 71:285-292. [As cited in ATSDR (1990l).]

Fishbein, L., 1981, *Sources, Transport and Alterations of Metal Compounds: An Overview, I. Arsenic, Beryllium, Cadmium, Chromium, and Nickel*, Environmental Health Perspectives, 40:43-64.

Fisher, N.S., et al., 1987, *Accumulation of Thorium, Lead, Uranium, and Radium in Marine Phytoplankton and Its Geochemical Significance*, Limnology and Oceanography, 32:131-142. [As cited in ATSDR (1990o).]

Foster, R.B., 1989, *Antimony Mobility in Soil Using Soil TLC*, prepared by Springborn Life Sciences, Inc., Wareham, Mass., for the Antimony Oxide Industry Association, Washington, D.C. [As cited in ATSDR (1990a).]

Frantz, G., and R.M. Carlson, 1987, *Division S-2-Soil Chemistry: Effects of Rubidium, Cesium, and Thallium on Interlayer Potassium Release from Transvaal Vermiculite*, Soil Science Society of America Journal, 51:305-308.

Friberg, L., G.F. Nordberg, and V.B. Vouk (editors), 1986, *Handbook on the Toxicology of Metals, Volume II: Specific Metals*, 2nd ed., Elsevier, Amsterdam.

Fries, G.F., 1982, *Potential Polychlorinated Biphenyl Residues in Animal Products from Application of Contaminated Sewage Sludge to Land*, Journal of Environmental Quality, 11:14-20.

Fuhrer, G.J., 1986, *Extractable Cadmium, Mercury, Copper, Lead, and Zinc in the Lower Columbia River Estuary, Oregon and Washington*, U.S. Geological Survey Water-Resources Investigations Report 86:4088.

Furr, A.K., et al., 1976, *Multielement Uptake by Vegetables and Millet Grown in Pots on Fly Ash Amended Soil*, Journal of Agricultural and Food Chemistry, 24:885-888.

Gambrell, R.P., and W.H. Patrick, Jr., 1982, *Manganese*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 313-322.

Ganje, T.J., and D.W. Rains, 1982, *Arsenic*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 385-402.

Garrels, R.M., and C.L. Christ, 1965, *Solutions, Minerals, and Equilibria*, Freeman, Cooper & Company, San Francisco.

Garten, C.T., Jr., 1978, *A Review of Parameter Values Used to Assess the Transport of Plutonium, Uranium, and Thorium in Terrestrial Food Chains*, Environmental Research, 17:437-452.

Gerritse, R.G., and W. Van Driel, 1984, *The Relationship between Adsorption of Trace Metals, Organic Matter, and pH in Temperate Soils*, Journal of Environmental Quality, *13*:197-204.

Gerritse, R.G., et al., 1982, *Effect of Sewage Sludge on Trace Element Mobility in Soils*, Journal of Environmental Quality, *11*:359-364.

Griffin, R.A., and S.F.J. Chou, 1981, *Movement of PCBs and Other Persistent Compounds through Soil*, Water Science Technology, *13*:1153-1163. [As cited in ATSDR (1989f).]

Heit, M., and C.S. Klusek, 1985, *Trace Element Concentrations in the Dorsal Muscle of White Suckers and Brown Bullheads from Two Acidic Adirondack Lakes*, Water, Air, and Soil Pollution, *25*:87-96.

Hiltbold, A.E., A.E. Hajek, and G.A. Buchanan, 1974, *A Proposed Arsenic Cycle in an Agronomic Ecosystem: Arsenical Pesticides*, Weed Science, 22(3):272-275. [As cited in ATSDR (1989a).]

Hopkins, D.M., 1977, *An Improved Ion-Selective Electrode Method for the Rapid Determination of Fluorine in Rocks and Soils*, Journal of Research of the U.S. Geological Survey, *5*:589-593. [As cited in Adriano and Doner (1982).]

Hunter, K.A., D.J. Hawke, L.K. Choo, 1988, *Equilibrium Adsorption of Thorium by Metal-Oxides in Marine Electrolytes*, Geochimica et Cosmochimica Acta, *52*:627-636.

Jenkins, D.W., 1980, *Biological Monitoring of Toxic Trace Metals, Vol. 1: Biological Monitoring and Surveillance*. [As cited in ATSDR (1990d).]

Karickhoff, S.W., D.S. Brown, and T.A. Scott, 1979, *Sorption of Hydrophobic Pollutants on Natural Sediments*, Water Research, *13*:241-248.

King, L.D., 1988, *Retention of Metals by Several Soils of the South-Eastern United States*, Journal of Environmental Quality, *17*:239-246.

Knox, S., et al., 1984, *Statistical Analysis of Estuarine Profiles: II. Application to Arsenic in the Tamar Estuary (S.W. England)*, Estuarine, Coastal, and Shelf Science, *18*:623-638.

Knudsen, D., G.A. Peterson, and P.F. Pratt, 1982, *Lithium, Sodium, and Potassium*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 225-246.

Kubota, J., and E.E. Cary, 1982, *Cobalt, Molybdenum, and Selenium*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 485-500.

Lagas, P., et al., 1984, *The Behavior of Barium in a Landfill and the Underlying Soil*, Water, Air, and Soil Pollution, 22:121-129.

Lindsay, W.L., 1979, *Chemical Equilibria in Soils*, John Wiley & Sons, New York.

Linsalata, P., et al., 1989, *An Assessment of Soil-to-Plant Concentration Ratios for Some Natural Analogues of the Transuranic Elements*, Health Physics, 56:33-46.

Lu, P.Y., et al., 1975, *Model Ecosystem Studies of Lead and Cadmium and of Urban Sewage Sludge Containing These Elements*, Journal of Environmental Quality, 4:505-509.

Lyman, W.J., W.F. Reehl, and D.H. Rosenblatt (editors), 1990, *Handbook of Chemical Property Estimation Methods*, American Chemical Society, Washington, D.C.

Mabey, W.R., et al., 1982, *Aquatic Fate Process Data for Organic Priority Pollutants*, EPA 440/4-81-014, U.S. Environmental Protection Agency, Washington, D.C., pp. 239-243.

Magorian, T.R., et al., 1974, *Water Pollution by Thallium and Related Metals*, pp. 145-160. [As cited in ATSDR (1990*l*).]

Maher, W., 1986, *Measurement of Total Antimony in Marine Organisms and Waters by Stibine Generation and Atomic Absorption Spectrometry*, Analytical Letters, 19:295-305. [As cited in ATSDR (1990a).]

Mahon, D.C., 1982, *Uptake and Translocation of Naturally-Occurring Radionuclides of the Uranium Series*, Bulletin of Environmental Contamination and Toxicology, 29:697-793. [As cited in ATSDR (1990m).]

Marschner, H., 1983, *General Introduction to the Mineral Nutrition of Plants*, in Inorganic Plant Nutrition, A. Läuchli and R.L. Bieleski (editors), Encyclopedia of Plant Physiology, New Series Vol. 15 A, Springer-Verlag, Berlin.

McFarlane, J.C., T. Pfleeger, and J. Fletcher, 1987, *Transpiration Effect on the Uptake and Distribution of Bromacil, Nitrobenzene, and Phenol in Soybean Plants*, Journal of Environmental Quality, 16:372-376.

McKone, T.E., 1990, *Dermal Uptake of Organic Chemicals from a Soil Matrix*, Risk Analysis, Vol. 10, No. 3, Society for Risk Analysis.

Milic, N.B., and T.M. Suranji, 1982, *Hydrolysis of the Thorium (IV) Ion in Sodium Nitrate Medium*, Canadian Journal of Chemistry, 60:1298-1303.

Murti, C.R.K., 1987, *The Cycling of Arsenic, Cadmium, Lead and Mercury in India*, in Lead, Mercury, Cadmium and Arsenic in the Environment, T.C. Hutchinson and K.M. Meema (editors), John Wiley & Sons, Chichester, Great Britain, pp. 315-333.

Napier, B.A., et al., 1980, *ARRRG and Food — Computer Programs for Calculating Radiation Dose to Man from Radionuclides in the Environment*, PNL-3180, prepared by Pacific Northwest Laboratory, Richland, Wash., for U.S. Department of Energy, Office of Nuclear Waste Isolation.

Nathwani, J.S., and C.R. Phillips, 1978, *Rates of Leaching of Radium from Contaminated Soils: An Experimental Investigation of Radium Bearing Soils from Port Hope, Ontario*, Water, Air and Soil Pollution, *9*:453-465.

Nathwani, J.S., and C.R. Phillips, 1979, *Rate Controlling Processes in the Release of Radium-226 from Uranium Mill Tailings*, Water, Air and Soil Pollution, *11*(3):301-308.

Nathwani, J.S., and C.R. Phillips, 1980, *Leachability of Radium-226 from Uranium Mill Tailings Consolidated with Naturally Occurring Materials and/or Cement*, Water, Air and Soil Pollution, *14*(0): 379-387.

National Academy of Sciences, 1976, *Drinking Water and Health*, National Academy Press, Washington, D.C. [As cited in ATSDR (1989g).]

National Academy of Sciences, 1977a, *Arsenic*, Medical and Biologic Effects of Environmental Pollutants Series, National Academy Press, Washington, D.C.

National Academy of Sciences, 1977b, *Drinking Water and Health — Inorganic Solutes*, prepared by Safe Drinking Water Committee, National Research Council, Volume 1, National Academy Press, Washington, D.C.

National Academy of Sciences, 1980, *Drinking Water and Health*, National Academy Press, Washington D.C., pp. 344-368. [As cited in ATSDR (1989g).]

National Council on Radiation Protection and Measurements, 1984, *Radiological Assessment: Predicting the Transport, Bioaccumulation, and Uptake by Man of Radionuclides Released to the Environment*, NCRP Report No. 76, Bethesda, Md.

Ng, Y.C., S.E. Thompson, and C.S. Colsher, 1982, *Soil-to-Plant Concentration Factors for Radiological Assessments*, NUREG/CR-2975, Lawrence Livermore National Laboratory, Livermore, Calif.

Nielsen, F.H., 1977, *Nickel Toxicity*, in Toxicology of Trace Elements, R.A. Goyer and M.A. Mehlman (editors), Advances in Modern Toxicology, Vol. 2, Halsted Press, New York.

Nriagu, J.O., 1990, *Food Contamination with Cadmium in the Environment*, in Food Contamination from Environmental Sources, J.O. Nriagu and M.S. Simmons (editors), John Wiley & Sons, Inc., New York.

Nuclear Safety Associates, Inc., 1980, *Comparison of Waste at the Western New York Nuclear Service Center*, Nuclear Safety Associates, Inc., Bethesda, Maryland.

Page, A.L., A.C. Chang, and M. El-Amamy, 1987, *Cadmium Levels in Soils and Crops in the United States*, in Lead, Mercury, Cadmium and Arsenic in the Environment, T.C. Hutchinson and K.M. Meema (editors), John Wiley & Sons, New York, pp. 119-146.

Perwack, J., et al., 1980, *Exposure and Risk Assessment for Copper*, EPA 440/4-81-015, U.S. Environmental Protection Agency, Cincinnati, Ohio. [As cited in ATSDR (1990e).]

Poston, T.M., 1982, *The Bioaccumulation Potential of Thorium and Uranium in Rainbow Trout (Salmo gairdneri)*, Bulletin of Environmental Contamination and Toxicology, 28:682-690. [As cited in ATSDR (1990m).]

Proctor, J., and S.R.J. Woodell, 1975, *The Ecology of Serpentine Soils*, Advances in Ecological Research, 9:255-366. [As cited in Reisenauer (1982).]

Przybylowicz, E.P., and C.W. Zuehlke, 1966, *Silver*, in Treatise on Analytical Chemistry, Part II. Analytical Chemistry of Inorganic and Organic Compounds, Vol. 4, I.M. Kolthoff, P.J. Elving, and E.B. Sandell (editors), Interscience Publishers, New York.

Ragaini, R.C., H.R. Ralston, and N. Roberts, 1977, *Environmental Trace Metal Contamination in Kellogg, Idaho, near a Lead Smelting Complex*, Environmental Science and Technology, 11:773-781.

Rai, D., et al., 1984, *Chemical Attenuation Rates, Coefficients, and Constants in Leachate Migration, Vol. I: A Critical Review*, EA-3356, prepared by Pacific Northwest Laboratories, Richland, Wash., for Electric Power Research Institute, Palo Alto, Calif. [As cited in ATSDR (1990c).]

Rayno, D.R., 1983, *Estimated Dose to Man from Uranium Milling via the Beef/Milk Food-Chain Pathway*, Science of the Total Environment, 31:219-421. [As cited in ATSDR (1990o).]

Reisenauer, H.M., 1982, *Chromium*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 337-346.

Robinson, W.O., R.R. Whetstone, and G. Edgington, 1950, *The Occurrence of Barium in Soils and Plants*, U.S. Department of Agriculture Technical Bulletin, 1013:29-32.

Robson, A.D., and M.G. Pitman, 1983, *Interactions between Nutrients in Higher Plants*, in Inorganic Plant Nutrition, Encyclopedia of Plant Physiology, Vol. 15A, A. Läuchli and R.L. Bielski (editors), Springer-Verlag, New York, pp. 147-180.

Rogers, R.D., 1976, *Methylation of Mercury in Agricultural Soils*, Journal of Environmental Quality, 5:454-458.

Sadiq, M., and C.G. Enfield, 1984a, *Solid Phase Formation and Solution Chemistry of Nickel in Soils, 1. Theoretical*, Soil Science, 138:262-270.

Sadiq, M., and C.G. Enfield, 1984b, *Solid Phase Formation and Solution Chemistry of Nickel in Soils, 2. Experimental*, Soil Science, 138:335-340.

Sandstead, H.H., 1977, *Nutrient Interactions with Toxic Elements*, in Toxicology of Trace Elements, R.A. Goyer and M.A. Mehlman (editors), Advances in Modern Toxicology, Vol. 2, Halsted Press, New York.

Scharer, J.M., and B. Ibbotson, 1982, *Bacterial Oxidation of Inorganic Compounds in Mining Waste*, Conservation and Recycling, 5:47-53. [As cited in ATSDR (1990m).]

Schmidt, J.A., and A.W. Andren, 1984, *Deposition of Airborne Metals into the Great Lakes: An Evaluation of Past and Present Estimates*, Advances in Environmental Science and Technology, 14:81-103. [As cited in ATSDR (1989c).]

Schoer, J., 1984, *Thallium*, in The Handbook of Environmental Chemistry, Vol. 3, Part C, Anthropogenic Compounds, O. Hutzinger (editor), Springer-Verlag, New York, pp. 143-214.

Schroeder, H.A., 1970, *Barium Air Quality Monograph*, Air Quality Monograph No. 70-12, American Petroleum Institute, Washington, D.C. [As cited in ATSDR (1990c).]

Schumacher, J.G., 1992, *Distribution Coefficients for Soil Contaminants (WSSRAP)*, letter from J.G. Schumacher (U.S. Geological Survey, Water Resources Division, Rolla, Mo.) to S.H. McCracken (Project Manager, Weldon Spring Site Remedial Action Project, St. Charles, Mo.), March 10.

Schumacher, J.G., and K.G. Stollenwerk, 1991, *Geochemical Controls on Migration of Molybdenum, Uranium, and Other Constituents at the Weldon Spring Chemical Plant Site*, in Proceedings of the Geosciences Workshop, DOE/OR/21548-197, published by MK-Ferguson Company and Jacobs Engineering Group, St. Charles, Mo., for U.S. Department of Energy, Oak Ridge Operations, Weldon Spring Site Remedial Action Project, St. Charles, Mo., pp. 87-140.

Sedlet, J., 1966, *Radon and Radium*, in Treatise on Analytical Chemistry, Part II. Analytical Chemistry of Inorganic and Organic Compounds, Vol. 4, I.M. Kolthoff, P.J. Elving, and E.B. Sandell (editors), Interscience Publishers, New York.

Seeley, F.G., and A.D. Kelmers, 1985, *Geochemical Information for Sites Contaminated with Low-Level Radioactive Wastes: III - Weldon Spring Storage Site*, ORNL-6112, Oak Ridge National Laboratory, Oak Ridge, Tenn., Feb.

Seiler, H.G., and H. Sigel (editors), 1988, *Handbook on Toxicity of Inorganic Compounds*, Marcel Dekker, Inc., New York.

Sharma, J., et al., 1986, *Hazards and Analysis of Thallium — A Review*, Toxicological and Environmental Chemistry Reviews, 11:93-116. [As cited in ATSDR (1990l).]

Shepherd, T.A., and J.A. Cherry, 1980, *Contaminant Migration in Seepage from Uranium Mill Tailings Impoundments: An Overview*, in Uranium Mill Tailings Management, Proceedings of 3rd Symposium, Fort Collins, Colo., Nov. 24-25.

Sheppard, M.I. and D.H. Thibault, 1990, *Default Soil Solid/Liquid Partition Coefficients, K_ds, for Four Major Soil Types: A Compendium*, Health Physics, 59:471-482.

Sheppard, M.I., et al., 1983, *Technetium and Uranium: Sorption by and Plant Uptake from Peat and Sand*, Health Physics, 44:635-643. [As cited in ATSDR (1990m).]

Sheppard, M.I., et al., 1984, *Soil Nuclide Distribution Coefficients and Their Statistical Distributions*, AECL-8364, prepared by Atomic Energy of Canada Limited, Whiteshell Nuclear Research Establishment, Pinawa, Manitoba.

Smith, I.C., and B.L. Carson (editors), 1981, *Trace Metals in the Environment, Vol. 6: Cobalt, An Appraisal of Environmental Exposure*, Ann Arbor Science Publishers, Inc., Ann Arbor, Mich. [As cited in ATSDR (1990d).]

Spanggord, R.J., et al., 1980, *Environmental Fate Studies on Certain Munition Wastewater Constituents, Final Report, Phase II — Laboratory Studies*, SRI Project No. LSU-7934, prepared by SRI International, Menlo Park, Calif., for U.S. Army Medical Research and Development Command, Fort Detrick, Frederick, Md., Sept.

Stevens, J.B., 1991, *Disposition of Toxic Metals in the Agricultural Food Chain; 1, Steady-State Bovine Milk Biotransfer Factors*, Environmental Science and Technology, 25(7):1289-1294.

Stevens, J.B., 1992, *Disposition of Toxic Metals in the Agricultural Food Chain; 2, Steady-State Bovine Tissue Biotransfer Factors*, Environmental Science and Technology, 26:1915-1921.

Stewart, J.W.B., and J.R. Bettany, 1982, *Mercury*, in Methods of Soil Analysis, Part 2: Chemical and Microbiological Properties, Second Edition, A.L. Page et al. (editors), American Society of Agronomy, Inc., Madison, Wisc., pp. 367-384.

Stumm, W., and J.J. Morgan, 1970, *Aquatic Chemistry, An Introduction Emphasizing Chemical Equilibria in Natural Waters*, Wiley-Interscience, New York.

Taskayev, A.I., et al., 1986, *Root Absorption of Radon-222 and Its Transfer into Above Ground Plant Organs*, Health Physics, 50:589-594.

Ter Haar, G.L., 1970, *Air as a Source of Lead in Edible Crops*, Environmental Science and Technology, 4:226-229.

Terkonda, P.K., 1991, letter from P.K. Terkonda (Professor, Department of Civil Engineering, University of Missouri-Rolla, Rolla, Mo.) to D.P. Bauer (District Chief, Water Resources Division, U.S. Geological Survey, Rolla, Mo.), Jan. 16.

Thibault, D.H., M.I. Sheppard, and P.A. Smith, 1990, *A Critical Compilation and Review of Default Soil Solid/Liquid Partition Coefficients, K_d, for Use in Environmental Assessments*, AECL-10125, prepared by Atomic Energy of Canada Limited, Whiteshell Nuclear Research Establishment, Pinawa, Manitoba.

Torstenfelt, B., 1986, *Migration of the Actinides, Thorium, Protactinium, Uranium, Neptunium, Plutonium and Americium in Clay*, Radiochimica Acta, 39:105-112.

Tracy, B.L., F.A. Prantl, and J.M. Quinn, 1983, *Transfer of ^{226}Ra, ^{210}Pb, and Uranium from Soil to Garden Produce: Assessment of Risk*, Health Physics, 44:469-477. [As cited in ATSDR (1990o).]

Travis, C.C. and A.D. Arms, 1988, *Bioconcentration of Organics in Beef, Milk, and Vegetation*, Environmental Science Technology, 22:271-274.

Tsuchiya, K., 1986, *Lead*, in Handbook on the Toxicology of Metals, Vol. II: Specific Metals, L. Friberg, G.F. Nordberg, and V.B. Vouk (editors), Elsevier Science Publishers, Amsterdam, pp. 298-353.

Turner, D.R., 1987, *Speciation and Cycling of Arsenic, Cadmium, Lead and Mercury in Natural Waters*, in Lead, Mercury, Cadmium and Arsenic in the Environment, T.C. Hutchinson and K.M. Meema (editors), John Wiley & Sons, New York, pp. 175-186.

Tyler, L.D., and M.B. McBride, 1982, *Mobility and Extractability of Cadmium, Copper, Nickel, and Zinc in Organic and Mineral Soil Columns*, Soil Science, 134:198-205.

U.S. Department of Energy, 1992a, *Feasibility Study for Remedial Action at the Chemical Plant Area of the Weldon Spring Site*, DOE/OR/21548-148, Vols. I-II, prepared by Argonne National Laboratory, Environmental Assessment and Information Sciences Division, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Field Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

U.S. Department of Energy, 1992b, *Remedial Investigation for the Chemical Plant Area of the Weldon Spring Site*, DOE/OR/21548-074, Rev. 0, Vols. I-II, prepared by MK-Ferguson Company and Jacobs Engineering Group, St. Charles, Mo., for U.S. Department of Energy, Oak Ridge Field Office, Weldon Spring Site Remedial Action Project, St. Charles, Mo., Nov.

U.S. Environmental Protection Agency, 1977, *Air Quality Criteria for Lead*, EPA 600/87-77-071, Office of Water Planning and Standards, Washington, D.C. [As cited in Tsuchiya (1986).]

U.S. Environmental Protection Agency, 1980a, *Ambient Water Quality Criteria Document for Chromium*, Office of Water Regulations and Standards, Washington, D.C. [As cited in ATSDR (1989c).]

U.S. Environmental Protection Agency, 1980b, *Ambient Water Quality Criteria for Thallium*, EPA-440/5-80-074, Office of Water Regulations and Standards, Washington, D.C.

U.S. Environmental Protection Agency, 1986, *Airborne Asbestos Health Assessment Update*, EPA/600/8-84/003F, Office of Health and Environmental Assessment, Washington, D.C.

Van Netten, C., and D.R. Morley, 1983, *Uptake of Uranium, Molybdenum, Copper, and Selenium by the Radish from Uranium Rich Soils*, Archives of Environmental Health, 38:172-175. [As cited in ATSDR (1990m).]

Van Zinderen, B., and J.F. Jaworski, 1980, *Effects of Vanadium in the Canadian Environment*, National Research Council of Canada, Associate Committee on Scientific Criteria for Environmental Quality, Ottawa. [As cited in ATSDR (1990p).]

Verschueren, K., 1983, *Handbook of Environmental Data on Organic Chemicals*, Second Edition, Van Nostrand Reinhold, New York.

Waite, D.T., S.R. Joshi, and H. Sommerstad, 1988, *The Effect of Uranium Mine Tailings on Radionuclide Concentrations in Langley Bay, Saskatchewan Canada*, Archives of Environmental Contamination and Toxicology, 17:373-380. [As cited in ATSDR (1990m).]

Wallace A., et al., 1976, *Some Effects of Chromium Toxicity on Bush Bean Plants Grown in Soil*, Plant and Soil, 44:471-473.

Wallwork-Barber, M.K., K. Lyall, and R.W. Ferenbaugh, 1985, *Thallium Movement in a Simple Aquatic Ecosystem*, Journal of Environmental Science and Health, 20:689-700.

Waters, M.D., 1977, *Toxicology of Vanadium*, in Toxicology of Trace Elements, R.A. Goyer and M.A. Mehlman (editors), Advances in Modern Toxicology, Vol. 2, Halsted Press, New York.

Watson, A.P., E.L. Etnier, L.M. McDowell-Boyer, 1984, *Radium-226 in Drinking Water and Terrestrial Food Chains: Transfer Parameters and Normal Exposure and Dose*, Nuclear Safety, 25:815-829.

Whicker, F.W., and V. Schultz, 1982, *Radioecology: Nuclear Energy and the Environment*, CRC Press Inc., Boca Raton, Fla.

Woolson, E.A., 1983, *Emissions, Cycling, and Effects of Arsenic in Soil Ecosystems*, in Biological and Environmental Effects of Arsenic, B.A. Fowler (editor), Topics in Environmental Health, Volume 6, Elsevier, New York, pp. 51-139.

Woolson, E.A., J.H. Axley, and P.C. Kearney, 1971, *Correlation between Available Soil Arsenic, Estimated by Six Methods, and Response of Corn (Zea mays L.)*, Proceedings of the Soil Science Society of America, 35:101-105.

World Health Organization, 1981, *Environmental Health Criteria 18: Arsenic*, Geneva, Switzerland.

World Health Organization, 1988, *Environmental Health Criteria 81: Vanadium*, Geneva, Switzerland.