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Remedial Investigation/Feasibility Study Report for Lower Watts Bar Reservoir Operable Unit

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Energy Systems Environmental Restoration Program
Clinch River Environmental Restoration Program

Remedial Investigation/Feasibility Study Report for Lower Watts Bar Reservoir Operable Unit

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CONTENTS

FIGURES ix

TABLES xi

ABBREVIATIONS xv

EXECUTIVE SUMMARY xvii

1. INTRODUCTION 1-1
   1.1 REGULATORY INITIATIVE 1-1
   1.2 OAK RIDGE RESERVATION ENVIRONMENTAL RESTORATION PROGRAM 1-1
   1.3 CLINCH RIVER ENVIRONMENTAL RESTORATION PROGRAM 1-2
   1.4 OBJECTIVES AND SCOPE OF THIS REPORT 1-2
   1.5 SCHEDULE 1-3

2. CHARACTERIZATION OF ENVIRONMENTAL SETTING 2-1
   2.1 GEOGRAPHY 2-1
   2.2 DEMOGRAPHY 2-1
   2.3 CLIMATE 2-7
   2.4 TOPOGRAPHY, GEOLOGY, AND SOILS 2-8
      2.4.1 Topography 2-8
      2.4.2 Geology 2-8
      2.4.3 Soils 2-8
      2.4.4 Groundwater 2-11
   2.5 SURFACE WATER 2-13
      2.5.1 Physical Characteristics 2-13
      2.5.2 Water Quality 2-19
   2.6 SEDIMENT 2-20
   2.7 ECOLOGY 2-22
      2.7.1 Aquatic Resources 2-25
      2.7.2 Terrestrial Resources 2-27

3. SITE HISTORY AND CURRENT CONDITIONS 3-1
   3.1 RELEASE HISTORY 3-1
      3.1.1 Releases from the Y-12 Plant 3-2
      3.1.2 Releases from the K-25 Site 3-5
      3.1.3 Releases from Oak Ridge National Laboratory 3-10
      3.1.4 Summary of Significant Releases 3-13
   3.2 SUMMARY OF EXISTING DATA 3-15
      3.2.1 Annual Surveys of Gamma Activity in Clinch and Tennessee River Bottom Sediments 3-15
      3.2.2 Clinch River Study 3-19
      3.2.3 1977 Clinch River Sediment Survey 3-24

3.2.4 Metals and PCBs in TVA Reservoir Sediments 3-26
3.2.5 Environmental Fate of Mercury and ^{137}Cs Discharged from ORR Facilities 3-26
3.2.6 TVA Instream Contaminant Study 3-27
3.2.7 Oak Ridge Gaseous Diffusion Plant Sediment Survey 3-30
3.2.8 Transport and Accumulation of ^{137}Cs and Mercury in the Clinch River and Watts Bar Reservoir System 3-37
3.2.9 TVA Recreation Area and Water Intake Sampling 3-40
3.2.10 TVA Fish Tissue Studies 3-44
3.2.11 TVA Annual Radiological Environmental Monitoring, Watts Bar Nuclear Plant 3-45
3.2.12 TVA Reservoir Monitoring Program—Sediment 3-50
3.2.13 ORR Environmental Monitoring Data 3-50
3.2.14 Arsenic Biogeochemistry in the Clinch River and Watts Bar Reservoir 3-52
3.2.15 TWRA Non-point Source Pesticide Study 3-53
3.2.16 CRRI Phase 1 Investigation 3-57
3.3 SUMMARY OF SITE CHARACTERIZATION 3-68

4. APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS 4-1
4.1 CHEMICAL-SPECIFIC ARARs 4-1
4.1.1 Surface Water 4-2
4.1.2 Sediment 4-2
4.2 RADIATION PROTECTION STANDARDS 4-4
4.3 LOCATION-SPECIFIC ARARs 4-5
4.3.1 Wetlands and Floodplains 4-5
4.3.2 Aquatic Resources 4-5
4.3.3 Endangered and Threatened Species 4-5
4.3.4 Archaeological and Historical Resources 4-6

5. HUMAN HEALTH RISK ASSESSMENT REPORT 5-1
5.1 IDENTIFICATION AND EVALUATION OF DATA SOURCES 5-1
5.1.1 Water Data 5-2
5.1.2 Sediment Data 5-4
5.1.3 Fish Data 5-4
5.1.4 Quality Assurance Evaluation of Data Sources 5-5
5.2 EXPOSURE ASSESSMENT 5-7
5.2.1 Characterization of Exposure Setting 5-7
5.2.2 Exposure Scenarios 5-8
5.2.2.1 Water ingestion scenario 5-11
5.2.2.2 Fish consumption scenario 5-11
5.2.2.3 Shoreline-use scenario 5-11
5.2.2.4 Swimming scenario 5-13
5.2.2.5 Irrigation scenario 5-13
5.2.2.6 Dredging scenario 5-15
5.3 TOXICITY ASSESSMENT 5-17
5.3.1 Contaminant-Specific Toxicity Assessment 5-17
5.3.1.1 Aldrin 5-17
5.3.1.2 Arsenic 5-19
5.3.1.3 Cadmium 5-20

5.3.1.4 Chlordane 5-21
5.3.1.5 Chromium 5-22
5.3.1.6 Manganese 5-23
5.3.1.7 Mercury 5-24
5.3.1.8 Polychlorinated biphenyls (Aroclor-1254 and Aroclor-1260) . . . 5-25
5.3.1.9 Zinc 5-26
5.3.1.10 Radionuclides 5-27
5.3.1.11 Cesium-137 5-28
5.3.1.12 Cobalt-60 5-29
5.3.1.13 Uranium 5-29
5.3.2 Chemicals Without EPA Toxicity Values 5-30
5.3.2.1 Aluminum 5-30
5.3.2.2 Copper 5-31
5.2.3.3 Lead 5-32
5.3.2.4 Magnesium 5-32
5.3.2.5 4-Nitrophenol 5-32
5.3.2.6 Dibenz(a,h)anthracene 5-33
5.4 RISK CHARACTERIZATION 5-34
5.4.1 Characterization Approach 5-34
5.4.1.1 Carcinogens 5-34
5.4.1.2 Noncarcinogens 5-35
5.4.2 Current Land-Use Conditions (Primary Exposure Pathways) 5-35
5.4.2.1 Water ingestion scenario 5-35
5.4.2.2 Fish consumption scenario 5-37
5.4.2.3 Shoreline-use scenario 5-38
5.4.2.4 Swimming scenario 5-40
5.4.3 Future Land-Use Conditions (Secondary Exposure Pathways) 5-41
5.4.3.1 Irrigation scenario 5-41
5.4.3.2 Dredging scenario 5-42
5.4.4 Special Case—Lead 5-45
5.4.5 Risk Characterization Summary 5-45
5.5 QUANTITATIVE UNCERTAINTY ANALYSIS 5-49
5.5.1 Uncertainty Analysis for the Fish Consumption Scenario 5-50
5.5.1.1 Subjective confidence intervals for the excess lifetime cancer risk from ^{137}Cs and selected organic carcinogens in fish 5-50
5.5.1.2 Results of the sensitivity analysis for carcinogens in fish 5-54
5.5.1.3 Subjective confidence intervals for the hazard quotients of selected noncarcinogens in fish 5-55
5.5.1.4 Results of the sensitivity analysis for selected noncarcinogens in fish 5-56
5.5.1.5 Conclusions for the uncertainty analysis of the fish consumption pathway 5-57
5.5.2 Uncertainty Analysis for Exposure to ^{137}Cs via the Dredging Scenario . 5-57
5.5.2.1 Subjective confidence intervals from the uncertainty analysis for the dredging scenario 5-58
5.5.2.2 Results of the sensitivity analysis for the dredging scenario . . . 5-58
5.5.2.3 Conclusions for the uncertainty analysis for the dredging scenario 5-59
5.6 SUMMARIES AND CONCLUSIONS 5-59

5.5.1 Chemicals of Potential Concern 5-59
5.5.2 Exposure Assessment 5-59

6. ECOLOGICAL RISK ASSESSMENT REPORT 6-1
6.1 ECOLOGICAL HAZARD IDENTIFICATION 6-1
6.1.1 Ecological Endpoints 6-1
6.1.1.1 Assessment endpoints 6-2
6.1.1.2 Measurement endpoints 6-6
6.1.2 Environmental Description 6-8
6.1.3 Sources 6-8
6.1.4 Conceptual Models 6-8
6.2 EXPOSURE ASSESSMENT 6-11
6.2.1 Aqueous Exposures 6-11
6.2.2 Sediment Exposures 6-12
6.2.3 Wildlife Exposures to Contaminated Fish 6-13
6.2.4 Soil Exposures 6-13
6.3 EFFECTS ASSESSMENT 6-13
6.3.1 Conventional Toxicity Data 6-13
6.3.1.1 Aqueous toxicity 6-13
6.3.1.2 Sediment toxicity 6-14
6.3.1.3 Wildlife toxicity 6-15
6.3.1.4 Soil toxicity to plants 6-16
6.3.2 Media Toxicity Data 6-17
6.3.3 Biological Survey Data 6-18
6.3.3.1 Benthic invertebrates 6-18
6.3.3.2 Fish 6-18
6.4 RISK CHARACTERIZATION 6-21
6.4.1 Risk Integration for Current Conditions 6-21
6.4.1.1 Single chemical toxicity 6-22
6.4.1.2 Media toxicity 6-25
6.4.1.3 Biological surveys 6-25
6.4.1.4 Weight of evidence for current conditions 6-26
6.4.2 Risk Integration for Future Conditions 6-29
6.4.3 Uncertainties 6-30
6.4.3.1 Uncertainties in the hazard identification 6-30
6.4.3.2 Uncertainties in the contaminant analysis data 6-30
6.4.3.3 Uncertainties in the chemical toxicity data 6-31
6.4.3.4 Uncertainties in the media toxicity data 6-32
6.4.3.5 Uncertainties in the biological survey data 6-32
6.4.4 Comparison to Human Health Risks 6-33
6.5 SUMMARY AND CONCLUSIONS 6-33

7. FEASIBILITY STUDY OVERVIEW 7-1

8. DEVELOPMENT OF ALTERNATIVES 8-1
8.1 PRELIMINARY REMEDIATION GOALS 8-1
8.2 GENERAL RESPONSE ACTIONS 8-2
8.2.1 No Action 8-2
8.2.2 Monitoring 8-2
8.2.3 Institutional Controls 8-3

8.2.4 Source Containment 8-3
8.2.5 Source Removal 8-3
8.2.6 In Situ and Ex Situ Treatment 8-3
8.2.7 Turbidity Containment 8-4
8.2.8 Disposal 8-4
8.3 IDENTIFICATION OF REMEDIAL TECHNOLOGIES AND SCREENING FOR TECHNICAL IMPLEMENTABILITY 8-4
8.4 EVALUATION OF TECHNOLOGIES AND PROCESS OPTIONS 8-11
8.5 ASSEMBLY OF ALTERNATIVES 8-21
8.6 DETAILED DESCRIPTION OF ALTERNATIVES 8-23
8.6.1 Common Components of Each Alternative 8-23
8.6.2 Alternative 1: No Action 8-23
8.6.3 Alternative 2: Institutional Controls and Advisories 8-24
8.6.4 Alternative 3: Sediment Removal 8-25
8.7 SUMMARY OF ACTION-SPECIFIC ARARs FOR EACH ALTERNATIVE 8-28
8.7.1 Alternative 1—No Action 8-28
8.7.2 Alternative 2—Institutional Controls and Advisories 8-28
8.7.2.1 Institutional controls 8-28
8.7.2.2 Watts Bar interagency agreement 8-28
8.7.2.3 Fish advisories 8-29
8.7.3 Alternative 3—Sediment Removal 8-29
8.7.3.1 Dredging activities 8-29
8.7.3.2 Use of shoreline facility for treatment and/or packaging 8-30

9. DETAILED ANALYSIS OF ALTERNATIVES AND THEIR ENVIRONMENTAL CONSEQUENCES 9-1
9.1 CRITERIA FOR ANALYSIS 9-1
9.1.1 Overall Protection of Human Health and the Environment 9-1
9.1.2 Compliance with ARARs 9-1
9.1.3 Long-term Effectiveness and Permanence 9-2
9.1.4 Short-term Effectiveness 9-2
9.1.5 Reduction of Toxicity, Mobility, and Volume 9-2
9.1.6 Implementability 9-2
9.1.7 Cost 9-2
9.1.8 State Acceptance 9-3
9.1.9 Community Acceptance 9-3
9.2 DETAILED ANALYSIS 9-3
9.2.1 Alternative 1: No Action 9-3
9.2.1.1 Overall protection of human health and the environment 9-3
9.2.1.2 Compliance with ARARs 9-3
9.2.1.3 Long-term effectiveness and permanence 9-3
9.2.1.4 Short-term effectiveness 9-4
9.2.1.5 Reduction of toxicity, mobility, and volume 9-4
9.2.1.6 Implementability 9-4
9.2.1.7 Cost 9-4
9.2.2 Alternative 2: Institutional Controls and Advisories 9-4
9.2.2.1 Overall protection of human health and the environment 9-4
9.2.2.2 Compliance with ARARs 9-4
9.2.2.3 Long-term effectiveness and permanence 9-5

9.2.2.4 Short-term effectiveness 9-5
9.2.2.5 Reduction of toxicity, mobility, and volume 9-5
9.2.2.6 Implementability 9-5
9.2.2.7 Cost 9-5
9.2.3 Alternative 3: Sediment Removal 9-6
9.2.3.1 Overall protection of human health and the environment 9-6
9.2.3.2 Compliance with ARARs 9-6
9.2.3.3 Long-term effectiveness and permanence 9-6
9.2.3.4 Short-term effectiveness 9-7
9.2.3.5 Reduction of toxicity, mobility, and volume 9-7
9.2.3.6 Implementability 9-8
9.2.3.7 Cost 9-8
9.3 COMPARATIVE ANALYSIS 9-8
9.3.1 Overall Protection of Human Health and the Environment 9-8
9.3.2 Compliance with ARARs 9-9
9.3.3 Long-Term Effectiveness and Permanence 9-9
9.3.4 Short-Term Effectiveness 9-9
9.3.5 Reduction of Toxicity, Mobility, and Volume 9-9
9.3.6 Implementability 9-9
9.3.7 Cost 9-10

10. REFERENCES 10-1

Appendix A: DATA BASES, EQUATIONS, AND TOXICITY AND EXPOSURE VALUES USED IN THE HUMAN HEALTH RISK CHARACTERIZATION A-1

Appendix B: RESULTS OF THE HUMAN HEALTH RISK CHARACTERIZATION . . B-1

Appendix C: FOOD CHAIN MODELS AND TRANSFER FACTORS USED IN THE HUMAN HEALTH RISK CHARACTERIZATION C-1

Appendix D: METHODOLOGY FOR UNCERTAINTY ANALYSIS OF S ELECT EXPOSURE PATHWAYS IN THE HUMAN HEALTH BASELINE RISK ASSESSMENT D-1

Appendix E: DATA TABLES FOR ECOLOGICAL RISK ASSESSMENT E-1

Appendix F: COST ESTIMATE SUMMARY FOR THE TWO FEASIBLE ALTERNATIVES F-1

FIGURES

2.1 Location of Watts Bar Reservoir 2-2
2.2 Watts Bar Reservoir, indicating the lower Watts Bar CERCLA Operable Unit and its relation to DOE's ORR 2-3
2.3 TVA Reservoir System (TVA 1990) 2-4
2.4 Oak Ridge Reservation 2-5
2.5 Population density within a 50-mile radius of Watts Bar Reservoir 2-6
2.6 Regional physiographic map of eastern Tennessee showing the location of Oak Ridge in the Valley and Ridge Province 2-9
2.7 Geologic map of eastern Tennessee 2-10
2.8 Schematic vertical relationships of flow zones of the ORR, estimated thickness, water flux, and water types 2-14
2.9 Hipsograph of lower Watts Bar Reservoir 2-15
2.10 Drainage area of the Tennessee River at Watts Bar Dam 2-16
2.11 Seasonal water level elevations in Watts Bar Reservoir 2-18
2.11 Map of the surface sedimentary characteristics based on the visual textural analysis of more than 190 surface sediment samples collected in Watts Bar Reservoir 2-21
2.12 Simplified general ecosystem diagram for LWBR 2-23
3.1 Y-12 Plant (depicting upper East Fork Poplar Creek environmental restoration operable units 3-3
3.2 Y-12 Site including Bear Creek (BC) and Chestnut Ridge (CR) environmental restoration operable units 3-4
3.3 Mercury release history from Y-12 Plant and ^{137}Cs release history from White Oak Lake (ORNL) 3-6
3.4 K-25 Site (depicting environmental restoration operable units) 3-7
3.5 ORNL Site (depicting environmental restoration waste area groupings) 3-11
3.6 Gamma count at the surface of sediment from the Clinch and Tennessee rivers in Watts Bar Reservoir 3-16
3.7 Activity profile vs bottom contour in selected areas of Watts Bar Reservoir 3-17
3.8 Major radionuclides found in silt from the Clinch and Tennessee rivers by years 1954–58 3-18
3.9 Mass balance diagram for ^{90}Sr, Clinch River study 3-20
3.10 Mass balance diagram for ^{137}Cs, Clinch River study 3-21
3.11 Section at CRM 7.5 showing penetration, recovery, and gross gamma radioactivity variation with depth for 1962 bottom sediment core samples 3-22
3.12 Comparison of patterns of variation with depth of gross gamma radioactivity in four bottom sediment cores to variations in annual releases of ^{137}Cs to Clinch River 3-23
3.13 Profile of 1977 radionuclide concentrations as a function of depth within the bottom sediment from a core taken at CRM 11.5, 30 feet from the left bank 3-25
3.14 Longitudinal distribution of selected parameters in TVA reservoir forebay sediments 3-28

3.15 The vertical distributions of mercury and ^{137}Cs in selected sediment cores obtained from Watts Bar and Chickamauga reservoirs 3-29
3.16 Mercury concentrations vs depth and particle size for Clinch River core sampling 3-33
3.17 Tennessee River sediment mercury and ^{137}Cs data 3-34
3.18 Maximum, mean, and minimum total mercury concentrations in fish in stream reaches below White Oak Creek 3-35
3.19 Mean total PCB concentrations in fish flesh instream contaminant study—task 5 3-36
3.20 Radionuclide, organic, and metal profiles in sediment cores 3-38
3.21 Map of lower Watts Bar Reservoir illustrating the average ^{137}Cs concentration (pCi/g) in the top 16 cm of sediment 3-39
3.22 Map of lower Watts Bar Reservoir illustrating the inventory (pCi/cm^2) of ^{137}Cs in the top 50 cm of sediment 3-41
3.23 Map of lower Watts Bar Reservoir illustrating the average mercury concentrations (μg/g) in upper 16 cm of sediment 3-42
3.24 Map of lower Watts Bar Reservoir illustrating mercury accumulation patterns and total inventories (μg/cm^2) 3-43
3.25 Mean PCB concentration in catfish flesh, 1987–1992, in Fort Loudoun and Watts Bar reservoirs 3-46
3.26 Mean PCB concentrations flesh of catfish from the Clinch River below Norris Reservoir, 1988–1992 3-47
3.27 Annual average concentrations of ^{137}Cs and ^{60}Co in surface sediment in forebay of Watts Bar Reservoir 3-48
3.28 Annual average concentrations of ^{137}Cs in fish from Watts Bar Reservoir 3-49
3.29 Spatial distribution of metals in sediment at TVA Reservoir forebays 1990–1991 3-51
3.30 Mean PCB concentrations (μg/kg) in catfish from Watts Bar Reservoir from 1987–1992 3-60
3.31 Levels of ^{137}Cs in near-shore sediment samples in lower Watts Bar Reservoir . 3-63
3.32 Levels of ^{137}Cs in near-shore sediment samples in Clinch River 3-64
3.33 Spatial ^{137}Cs inventory in lower Watts Bar Reservoir 3-65
5.1 Site model for human health risk assessment 5-9
6.1 Conceptual model of contaminant sources, transport, and exposure in the aquatic food web 6-10
6.2 Conceptual model of contaminant sources, transport, and exposure in the terrestrial food web for the baseline dredging case 6-11
7.1 Relationship of screening criteria to the nine evaluation criteria 7-2

TABLES

2.1 Population centers near Watts Bar Reservoir and water intake location information 2-7
2.2 Generalized stratigraphic profile of the Oak Ridge Reservation 2-11
2.3 Soil parent material, associated soils series, and soil characteristics of the major soil series in the lower Watts Bar Reservoir area 2-12
2.4 Local watershed area and estimated sediment accumulation, sediment yield, and sediment delivery rate for Watts Bar Reservoir for the period 1946–1991 2-22
2.5 Average annual sedimentation rates for lower Watts Bar Reservoir based on TVA silt-range data 2-22
2.6 TVA herbicide application for aquatic weed control in Watts Bar, Melton Hill, Fort Loudoun, and Tellico reservoirs, 1972–1992 2-26
2.7 List of threatened and endangered species that have been recorded in Meigs, Rhea, and Roane counties 2-29
2.8 Threatened and endangered aquatic biodistribution in eastern Tennessee and current status in the lower Watts Bar Reservoir 2-32
3.1 Estimated discharges of radionuclides from White Oak Creek to the Clinch River, 1949–1992 3-14
3.2 Maximum concentrations of radionuclides reported in the 1977 Clinch River Sediment Survey 3-26
3.3 Total metal concentrations in surface sediments from selected TVA reservoirs, 1982 3-27
3.4 Summary of metal concentrations in fine-particle surface sediment 3-31
3.5 Instream contaminant study—Task 2. Maximum concentrations _pCi/g) reported for significant radioisotopes in fine-particle surface sediment 3-32
3.6 Average, maximum, and minimum concentrations of selected metals in sediment at recreation area beaches and at water intakes in lower Watts Bar Reservoir 3-44
3.7 Mean concentrations of selected analytes in surface water at Clinch River Mile 9.9 (Brashear Island) for the period 1987–1992. 3-52
3.8 Non-point source pesticides and related compounds in channel catfish from Watts Bar Reservoir and upstream reservoirs, 1991–1992 3-54
3.9 Non-point source pesticides and related compounds in channel catfish from Watts Bar Reservoir and upstream reservoirs, 1992–1993 3-55
3.10 Non-point source pesticides and related compounds in mussels from Watts Bar Reservoir and upstream reservoirs, 1991–1992 3-56
3.11 Non-point source pesticides and related compounds in mussels from Watts Bar Reservoir and upstream reservoirs, 1992–1993 3-56
3.12 Summary of samples collected during CRRI investigations into contaminants in fish flesh in LWBR 3-58
3.13 Mean annual concentrations of total PCBs in fish flesh, LWBR 3-58

3.14 Average concentrations of inorganic contaminants and radionuclides in sediment cores collected from Poplar Creek, the Clinch River, and the Tennessee River, 1989 3-61
3.15 Number of surface sediment samples, mean, minimum, maximum, and standard deviation for ^{137}Cs concentrations in LWBR, the Clinch River below Poplar Creek, and the Clinch River from White Oak Creek to Poplar Creek 3-62
3.16 The performance of the Olsen core sampling plan relative to random sampling for five maps of cesium inventory 3-68
4.1 Tennessee water quality criteria for designated water uses (μg/L) 4-3
4.2 Sediment Quality Criteria 4-4
4.3 Potential location-specific applicable or relevant and appropriate requirements for Clinch River RI 4-7
5.1 Sources of data for the Lower Watts Bar Reservoir Human Health Risk Characterization 5-3
5.2 Chemicals without EPA toxicity values 5-30
5.3 Contaminants of potential concern in the water ingestion scenario 5-37
5.4 Contaminants of potential concern in the consumption of fish scenario 5-39
5.5 Contaminants of potential concern in the shoreline-use scenario 5-40
5.6 Contaminants of potential concern in the irrigation scenario 5-43
5.7 Contaminants of potential concern in the dredging scenario 5-44
5.8 Contaminants of potential concern: contaminants with cancer risks $\geq 10^{-4}$ or hazard quotients ≥ 1.0 5-46
5.9 Contaminants with cancer risks between 10^{-4} and 10^{-6} 5-47
5.10 Subjective confidence limits for the excess lifetime cancer risk from ^{137}Cs in fish from the Lower Watts Bar Reservoir when the dose conversion factor was allowed to vary 5-51
5.11 Subjective confidence limits for the excess lifetime cancer risk from selected carcinogens in fish from Lower Watts Bar Reservoir with slope factors allowed to vary (except for dieldrin) 5-52
5.12 Subjective confidence limits for the exceee lifetime cancer risk from selected organic carcinogens in fish from Lower Watts Bar Reservoir with slope factors held constant 5-53
5.13 Results of sensitivity analysis of excess lifetime cancer risk from ingestion of ^{137}Cs in catfish from Lower Watts Bar Reservoir when the dose conversion factor was allowed to vary 5-54
5.14 Results of a sensitivity analysis for the total excess lifetime cancer risk from all selected organic carcinogens in catfish from Lower Watts Bar Reservoir when slope factors were allowed to vary 5-55
5.15 Subjective confidence limits for hazard quotients of selected noncarcinogens in fish from Lower Watts Bar Reservoir when reference doses were allowed to vary 5-56
5.16 Subjective confidence limits for the hazard quotients of selected noncarcinogens in fish from lower Watts Bar Reservoir when reference doses were held constant 5-56
5.17 Results of sensitivity analysis for hazard indices for selected noncarcinogens in largemouth bass from lower Watts Bar Reservoir when reference doses were allowed to vary 5-57

5.18 Subjective confidence limits for the excess lifetime cancer risk from ^{137}Cs in deep-water sediments of lower Watts Bar Reservoir used for agricultural purposes 5-58
5.19 Results of sensitivity analysis for total excess lifetime cancer risk from ^{137}Cs exposure in the dredging scenario for lower Watts Bar Reservoir 5-59
6.1 Contaminants of potential concern for LWBR identified in prior screening assessments 6-9
6.2 Results of TVA monitoring of benthic invertebrates in Watts Bar Reservoir compared to ten other mainstream reservoirs (minima-means-maxima) in that year 6-19
6.3 Results of TVA monitoring of fish communities in Watts Bar Reservoir compared to the (maximum-mean-minimum) values for the ten mainstream reservoirs 6-20
6.4 Summary of the risk characterization for benthic invertebrates 6-27
6.5 Summary of the risk characterization for fish 6-27
6.6 Summary of the risk characterization for piscivorous wildlife 6-28
6.7 Comparison of contaminants of potential concern for human health and ecological risks 6-33
8.1 Screening of remedial technologies for Lower Watts Bar Reservoir Operable Unit 8-6
8.2 Evaluation of screened remedial technologies for Lower Watts Bar Reservoir Operable Unit 8-12
8.3 ARARs and TBC guidance for Alternatives 1 Through 3 8-32
9.1 Detailed analysis summary 9-11

ABBREVIATIONS

AEA	Atomic Energy Act
ALARA	as low as reasonably achievable
ARARs	applicable or relevant and appropriate requirements
bw	body weight
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COE	U.S. Corps of Engineers
COPC	contaminants of potential concern
CR-ERP	Clinch River Environmental Restoration Program
CRM	Clinch River Mile
CRRI	Clinch River Remedial Investigation
CSMS	Contaminated Sediment Management Strategy
CV	chronic values
DO	dissolved oxygen
DOE	U.S. Department of Energy
EFPC	East Fork Poplar Creek
EPA	Environmental Protection Agency
ER	Environmental Restoration
ER-L	effects range—low
ER-M	effects range—medium
ERM	Emory River Mile
FDA	Food and Drug Administration
FFA	Federal Facility Agreement
FS	feasibility study
GI	gastrointestinal
HEAST	Health Effects Assessment Summary Tables
HI	hazard index
HQ	hazard quotient
IAEA	International Atomic Energy Agency
IRIS	Integrated Risk Information System
LCL	lower confidence limit
LCT	lowest concentration tested
LCV	lowest chronic value
LET	linear energy transfer
LHS	Latin Hypercube Sampling
LOAEL	lowest observed adverse effects level
LOEC	lowest observed effects concentration
LOL	lowest effect level
LTV	lowest toxic values
LWBR	Lower Watts Bar Reservoir
MCLs	maximum contaminant levels
MOE	Ministry of Environment
msl	mean sea level
NAWQC	National Ambient Water Quality Criteria
NCP	National Contingency Plan

NCRP	National Council on Radiation Protection
NEPA	National Environmental Policy Act
NESHAP	National Emission Standards for Hazardous Air Pollutants
NOAA	National Oceanic Atmospheric Administration
NOAEL	no observed adverse effects level
NPDES	National Pollutant Discharge Elimination System
NPRM	notice of proposed rulemaking
NRC	Nuclear Regulatory Commission
NSDWS	National Secondary Drinking Water Standards
ORGDP	Oak Ridge Gaseous Diffusion Plant
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
ORTF	Oak Ridge Task Force
OSHA	Occupational Safety and Health Administration
OU	operable unit
PAH	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls
QA/QC	quality assurance/quality control
RCRA	Resource Conservation and Recovery Act
RfC	reference concentration
RfD	reference dose
RI/FS	remedial investigation/feasibility study
RME	reasonable maximum exposure
ROD	record of decision
SEL	severe effects level
SEN	Secretary of Energy Notice
SI	sensitivity index
SLC	screening level concentration
SMCLs	secondary maximum contaminant levels
SQC	sediment quality criteria
SSLC	species screening level concentration
SWSA	solid waste storage area
T&E	threatened or endangered
TBC	to be considered
TDEC	Tennessee Department of Environment and Conservation
TRM	Tennessee River Mile
TVA	Tennessee Valley Authority
TWRA	Tennessee Wildlife Resources Agency
UCL	upper confidence limit
USC	United States Code
USDA	U.S. Department of Agriculture
WBRIPG	Watts Bar Reservoir Interagency Permitting Group
WOC	White Oak Creek
WQC	water quality criteria

EXECUTIVE SUMMARY

This document is the combined Remedial Investigation and Feasibility Study Report for the lower Watts Bar Reservoir (LWBR) Operable Unit (OU). The LWBR is located in Roane, Rhea, and Meigs counties, Tennessee, and consists of Watts Bar Reservoir downstream of the Clinch River. This area has received hazardous substances released over a period of 50 years from the U.S. Department of Energy's Oak Ridge Reservation (ORR), a National Priority List site established under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). As required by this law, the ORR and all off-site areas that have received contaminants, including LWBR, must be investigated to determine the risk to human health and the environment resulting from these releases, the need for any remedial action to reduce these risks, and the remedial actions that are most feasible for implementation in this OU.

Contaminants from the ORR are primarily transported to the LWBR via the Clinch River. There is little data regarding the quantities of most contaminants potentially released from the ORR to the Clinch River, particularly for the early years of ORR operations. Estimates of the quantities released during this period are available for most radionuclides and some inorganic contaminants, indicating that releases 30 to 50 years ago were much higher than today. Since the early 1970s, the release of potential contaminants has been monitored for compliance with environmental law and reported in the annual environmental monitoring reports for the ORR.

Historical studies, the current Clinch River Remedial Investigation, and other contemporary studies have provided data that collectively describe the environmental fate of most contaminants released from the ORR. Water-soluble contaminants released to ORR surface waters are rapidly diluted upon entering the Clinch River and then quickly transported downstream to the Tennessee River where further dilution occurs. Almost the entire quantity of these diluted contaminants rapidly flows through LWBR. In contrast, particle-associated contaminants tend to accumulate in the lower Clinch River and in LWBR in areas of sediment deposition, primarily in the old river channel where sediment accumulation is greatest. Those particle-associated contaminants that were released in peak quantities during the early years of ORR operations (e.g., mercury and ^{137}Cs) are buried under as much as 80 cm of cleaner sediment in LWBR. Although total inventories of particle-associated contaminants are greater in LWBR than in the Clinch River, concentrations are higher in the Clinch River because the sediments are not yet diluted by mixing with Tennessee River sediments. Certain contaminants, most notably polychlorinated biphenyls (PCBs), have accumulated in LWBR biota. The contamination of aquatic biota with PCBs is best documented for certain fish species and extends to reservoirs upstream of the ORR, indicating a contamination problem that is regional in scope and not specific to the ORR.

The greatest risk to human health from contaminants in LWBR is associated with the consumption of certain fish species contaminated with PCBs. Other contaminants in fish that pose potential risks include mercury, chlordane, aldrin, and arsenic. Because of their low body weight, children are potentially at greater risk than adults; most of the contaminants above pose a potential threat only to children. If deep-water sediments were dredged and used for farming, several contaminants could pose a risk to human health via consumption of the resulting agricultural products. In place, these sediments do not pose a risk to human health as no exposure

pathway exists. Levels of ORR contaminants in near-shore sediments are low. No unacceptable risk to human health from surface water in LWBR was identified for any exposure pathway.

The screening-level ecological risk assessment indicated that LWBR sediment is potentially toxic to benthic organisms, although conclusive evidence is lacking, and the cause of any toxicity has not been established. In fish, the levels of PCBs are high enough to pose a reproductive threat to LWBR piscivorous wildlife feeding heavily on these fish. However, most of the fish contaminant data for LWBR are collected for assessing risk to human health and are not as useful for assessing ecological risk. Surface water data did not indicate any contaminants that pose an unacceptable risk to aquatic biota in LWBR, but the data are limited. Further data are required to evaluate the ecological risk in LWBR. These data are currently being obtained from the Clinch River arm of Watts Bar Reservoir as part of the Clinch River Remedial Investigation Phase 2 studies. These data will be used to determine if ecological effects are occurring in the Clinch River, where such effects are potentially greater due to the proximity of contaminant sources. Additional data will also be collected through a long-term monitoring program that will be used to produce a more definitive ecological risk assessment as a component of the 5-year review of the record of decision.

Preliminary remediation goals that are protective of human health were identified for LWBR sediment and biota. No remediation goals were established for LWBR surface water because the human health risk assessment indicates that surface waters do not pose a risk to human health under current or future exposure scenarios. A range of general remedial response actions were identified and various treatment technologies for each were evaluated for implementation in LWBR. Preliminary screening steps resulted in the detailed evaluation of three alternatives: (1) no-action, (2) the use of institutional controls and advisories to limit exposure, and (3) the limited removal of contaminated sediment in conjunction with institutional controls. A long-term monitoring program will be implemented as part of the chosen alternative to ensure that the selected remedy is protective of human health.

1. INTRODUCTION

1.1 REGULATORY INITIATIVE

This document is the combined Remedial Investigation/Feasibility Study (RI/FS) Report for the Lower Watts Bar Reservoir Operable Unit (LWBR OU), an offsite OU associated with environmental restoration activities at the U.S. Department of Energy (DOE) Oak Ridge Reservation (ORR). As a result of past and present releases of hazardous substances into the environment, and the threat of future releases, the ORR was placed on the National Priorities List in December 1989 (54 FR 48184). Sites on this list must be investigated for possible remedial action, as required by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, 42 U.S.C. 9601, *et seq.*). This report documents the findings of the remedial investigation of this OU and of the feasibility of remedial action alternatives potentially applicable there. These studies are authorized by Sect. 117 of CERCLA and were conducted in accordance with the requirements of the National Contingency Plan (NCP) (40 CFR Part 300).

DOE, the U.S. Environmental Protection Agency (EPA), and the Tennessee Department of Environment and Conservation (TDEC) have entered into a Federal Facility Agreement (FFA), as authorized by Sect. 120 of CERCLA and Sects. 3008(h) and 6001 of the Resource Conservation and Recovery Act (RCRA) (42 U.S.C. 6901 *et seq.*). The purpose of this agreement is to ensure a coordinated and effective response for all environmental restoration activities occurring at the ORR. In addition to other responsibilities, the FFA parties mutually define the OU boundaries, set remediation priorities, establish remedial investigation priorities and strategies, and identify and select remedial actions. A copy of this FFA is available from DOE's Information Resource Center in Oak Ridge, Tennessee.

The Clinch River Remedial Investigation (CRRI) was begun under authority of Sect. 3008(v) of RCRA. Although these requirements still apply, CERCLA allows for the consideration of a wider range of contaminants, including radionuclides, polychlorinated biphenyls (PCBs), and contaminants in wastes released prior to the enactment of RCRA in 1976. The FFA parties intend to meet the RCRA corrective action requirements, as well as any applicable requirements of other environmental law, through the CERCLA process.

1.2 OAK RIDGE RESERVATION ENVIRONMENTAL RESTORATION PROGRAM

The DOE ORR is composed of three major installations—the Oak Ridge National Laboratory (ORNL), the Y-12 Plant, and the K-25 Site (formerly the Oak Ridge Gaseous Diffusion Plant). These facilities were constructed in the early 1940s as research, development, and process facilities in support of the Manhattan Project. Approximately 650 ORR sites have been identified that require environmental evaluation. The remediation of these sites is expected to take two to three decades and cost several billion dollars. The overall strategy in effecting the environmental response has been to partition the ORR into waste area groupings and OUs to facilitate investigation and action. The ORR-wide Environmental Restoration (ER) Program is described in the ORR ER Site Management Plan (DOE 1994).

1.3 CLINCH RIVER ENVIRONMENTAL RESTORATION PROGRAM

The Clinch River Environmental Restoration Program (CR-ERP) was created to investigate the impact of current and historical releases of contaminants from the ORR to the off-site surface water environment. The original study area was a single OU, which included Melton Hill Reservoir, the Clinch River arm of Watts Bar Reservoir, and Watts Bar Reservoir downstream of the confluence of the Clinch and Tennessee rivers (Energy Systems 1990). This investigation implemented a phased approach that relied heavily on screening-level risk analysis for estimating the human health and environmental risks resulting from offsite contamination. This phased approach included (1) an initial screening-level risk assessment to identify those areas for which more data were needed; (2) a preliminary (Phase 1) sampling and analysis of water, sediment, and biota from selected sites representative of differing levels of contamination (including upstream reference sites) to determine the range of contaminants present in the offsite environment and also to verify existing data; (3) a focused (Phase 2) sampling and analysis effort directed specifically at supplying additional data for site characterization and human health and ecological risk assessment for specific areas, media, and contaminants of concern; (4) iterative risk assessments, as data became available, to continue focusing of sampling efforts; and (5) the identification and evaluation of potential remedial action alternatives (Energy Systems 1990).

In implementing the phased approach described above, preliminary human health and ecological risk assessments were performed using existing data (step 1 above), the Phase 1 sampling and analysis effort was completed, and a second iteration of screening-level risk assessments was performed using these data (Cook et al. 1992). The Phase 1 data were presented at public meetings in early 1993. It was explained that the Phase 2 investigation would focus on the Clinch River, nearer the source of ORR contaminants. The question was then raised whether the FFA parties could split the existing OU such that a remedial decision could be reached on the bulk of the reservoir while the Clinch River arm was being studied in more detail, potentially expediting the remedial decision by several years. After considering this request and carefully considering the existing information, the FFA parties split the original OU into two, thereby creating a new LWBR OU extending from Tennessee River Mile (TRM) 529.9 at Watts Bar Dam upstream to TRM 567.5 at the confluence of the Clinch and Tennessee rivers. In creating the LWBR OU, the FFA parties agreed that by also considering existing data, sufficient information existed to allow DOE to proceed immediately with the preparation of a combined RI/FS Report for this OU. An important consideration in this decision was that practical remedial alternatives for LWBR are extremely limited (Radian Corp. 1993). The only risks identified in previous screening-level risk assessments (Suter 1991, Hoffman et al. 1991, Cook et al. 1992) appeared to be readily mitigated by existing institutional controls advisories established by federal and state law. These controls may be thought of as "presumptive remedies" that were reliable if the baseline risk assessment presented in this report did not identify any additional or new risks and the evaluation of action-based remedial alternatives (Sects. 8 and 9) does not identify feasible alternatives. The remedial investigation of the Clinch River OU has continued into the Phase 2 sampling stage.

1.4 OBJECTIVES AND SCOPE OF THIS REPORT

The objectives of this report are to (1) describe the nature and extent of the contamination in LWBR resulting from releases from the ORR, (2) quantify the risk to human health and the environment resulting from contaminants in LWBR, and (3) identify and evaluate remedial action

alternatives that are feasible for LWBR. This report uses data from a number of different sources, including historical data, to accomplish these objectives.

In support of the first objective above, the site characterization presented here uses existing data in a weight-of-evidence approach to reach general conclusions regarding the extent of contamination in LWBR. The data presented were all originally collected by either ORNL, TVA, or the state of Tennessee and are presumed to be of acceptable quality for use in a weight-of-evidence approach to site characterization.

In support of objective number two, certain data from all existing sources were combined into data sets for use in quantitatively assessing risk to human health. The criteria for including data for quantitative use are outlined in Sect. 5.1. Summary statistics of these data are provided in Appendix A. The FFA parties agreed that the ecological risk assessment would be a screening-level assessment only. More definitive ecological data will be collected from the Clinch River near the ORR during the Phase 2 investigation and extrapolated to LWBR to further define potential ecological risk issues in LWBR. Additional data from LWBR will also be collected during a long-term monitoring program, with a more definitive ecological risk assessment provided as part of the 5-year review of the record of decision (ROD).

1.5 SCHEDULE

In conjunction with this RI/FS, DOE is preparing a Proposed Plan that will present an outline of the RI/FS findings and will identify a preferred remedial action alternative. A public meeting will be held to present the Plan to the public, and a 30-day comment period will follow. DOE, EPA, and TDEC will jointly select the final remedy after consideration of the public comments. The Proposed Plan is scheduled for public release in December 1994.

DOE will document the selection of the final remedy in a formal ROD to be signed by EPA and TDEC. This ROD is scheduled for completion by May 1995. If the ROD results in contaminants being left in place at the site, Sect. 121(c) of CERCLA requires review of the ROD at least every five years to ensure that the selected remedy is protective of human health and the environment. In such an event, DOE would implement a long-term monitoring plan in LWBR to gather the data necessary to make this determination. If this review indicates that the remedy is not protective of human health and the environment, further remedial action will be required.

2. CHARACTERIZATION OF ENVIRONMENTAL SETTING

2.1 GEOGRAPHY

Watts Bar Dam is located in eastern Tennessee almost equidistant (about 62 miles) from the cities of Knoxville and Chattanooga (Fig. 2.1). Watts Bar Reservoir flows through portions of four counties in eastern Tennessee—Loudon, Roane, Rhea, and Meigs (Fig. 2.2). The reservoir extends 72 miles from the dam at TRM 529.9 upstream on the Tennessee River to Fort Loudoun Dam (TRM 602.3) near Lenoir City and an additional 23 miles up the Clinch River to Melton Hill Dam (Clinch River Mile [CRM] 23.1) near Oak Ridge. The LWBR study area extends 38 miles from TRM 567.5 at the mouth of the Clinch River to TRM 529.9 at Watts Bar Dam (Fig. 2.2). The upstream boundary of the LWBR OU abuts the Clinch River OU. The downstream boundary was placed at Watts Bar Dam because LWBR is the primary site of sediment-associated contaminant accumulation downstream of the ORR (see Sect. 3.2). Each dam is part of the Tennessee Valley Authority's (TVA's) multipurpose water-control system, comprised of 34 major dams in the Tennessee River basin (Fig. 2.3). Watts Bar Reservoir is one of nine mainstem impoundments located on the Tennessee River between Paducah, Kentucky, and Knoxville, Tennessee. This reservoir provides flood control, hydropower generation, navigation, municipal and industrial water supply, wildlife habitat, and recreation. DOE's 34,600-acre ORR is located on the Clinch River arm of Watts Bar Reservoir in Anderson and Roane counties, Tennessee (Fig. 2.4). Although the ORR lies within the corporate limits of the city of Oak Ridge, the residential area of the city lies north of the ORR.

2.2 DEMOGRAPHY

Watts Bar Reservoir is located between the major metropolitan areas of Knoxville and Chattanooga, Tennessee (Fig. 2.1). Several smaller towns are located on or near the reservoir (Fig. 2.2); the population of each of these towns is provided in Table 2.1. The population density within a 50-mile radius of the center of the LWBR OU is depicted in Fig. 2.5. Five separate communities draw their drinking water from Watts Bar Reservoir (Fig. 2.2 and Table 2.1), but only the Spring City and Rockwood municipal intakes are in LWBR. Moreover, these communities draw their water from tributary embayments (Piney River and King Creek, respectively) rather than from the main channel. The city of Kingston's water intake is located just above the LWBR study area on the Tennessee River, and could, depending on flow conditions, potentially be impacted by ORR effluents. Water from the Kingston Water Treatment Plant has historically been monitored for radionuclides by DOE as part of the ORR Environmental Monitoring Program. Results are presented annually in the ORR environmental monitoring reports.

In addition to the resident population in the vicinity of the reservoir, the reservoir receives recreational use from persons outside the immediate study area. There are 20 recreational parks within the Watts Bar area and 27 commercial recreational facilities, including marinas, resorts, and golf courses (TVA 1990a). Of these commercial facilities, 22 are located in the lower one-fourth of the reservoir, within the LWBR study area. Total annual visitor-days to the reservoir are estimated at 1 million (TVA 1990a), while the actual number of recreational visits per year is greater than 6 million (TVA 1987). The vast majority of recreational visits occur in that portion of the reservoir from the Watts Bar Dam upstream to Kingston (TVA 1987). Popular recreational activities include swimming, skiing, boating, and fishing.

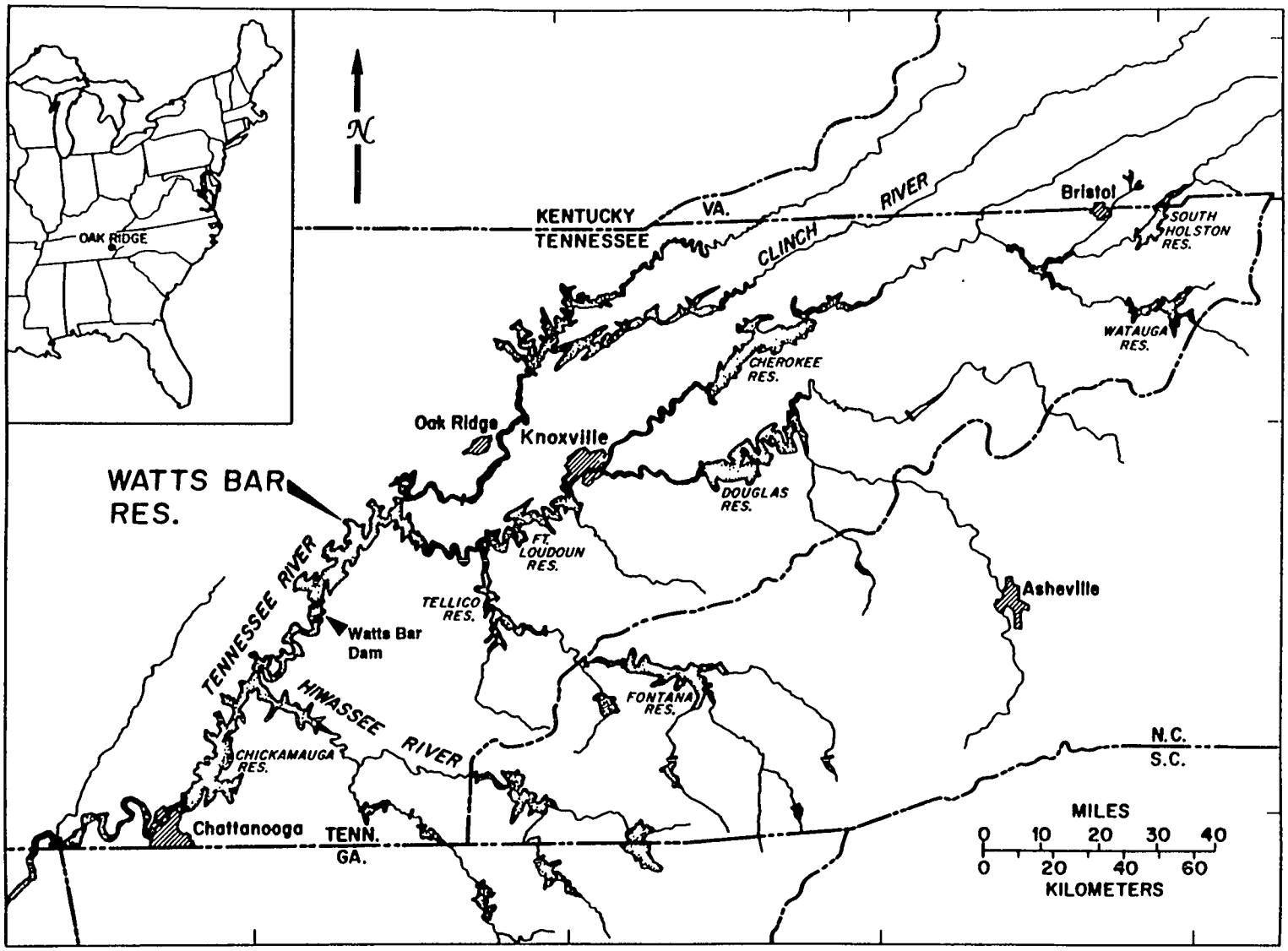

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Fig. 2.1. Location of Watts Bar Reservoir.

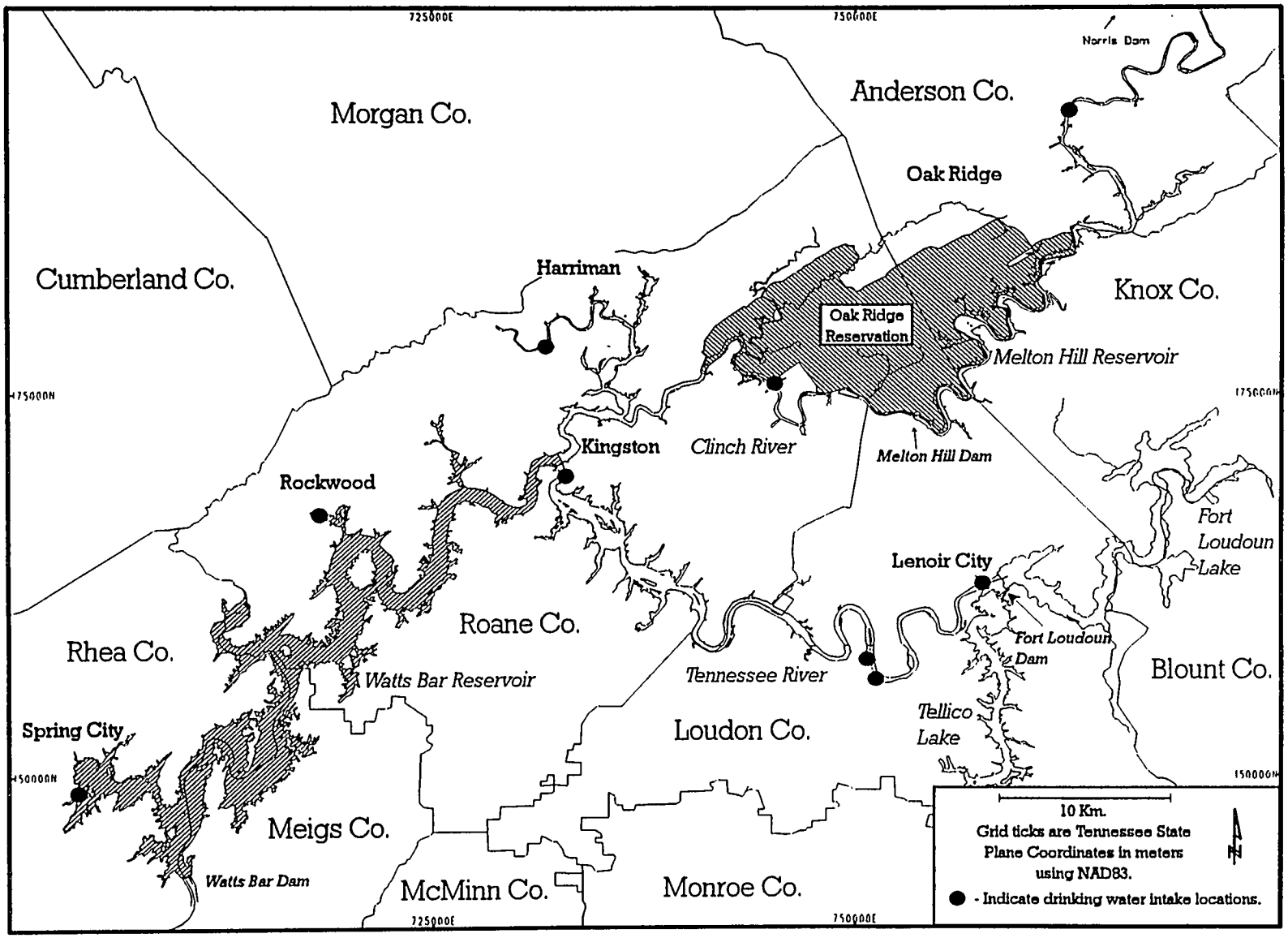

Fig. 2.2. Watts Bar Reservoir, indicating the lower Watts Bar CERCLA Operable Unit and its relation to DOE's ORR.

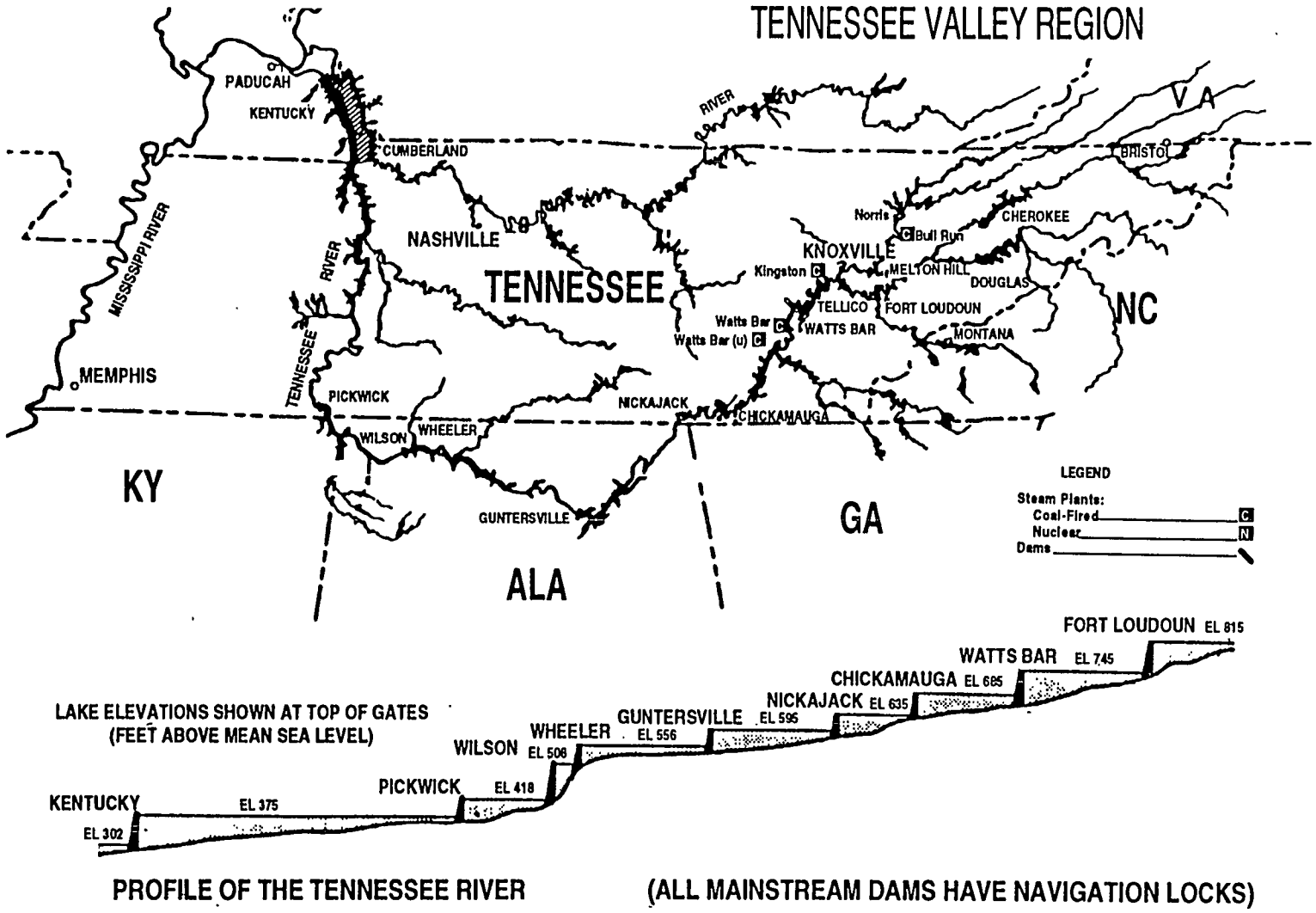

Fig. 2.3. TVA reservoir system (TVA 1990).

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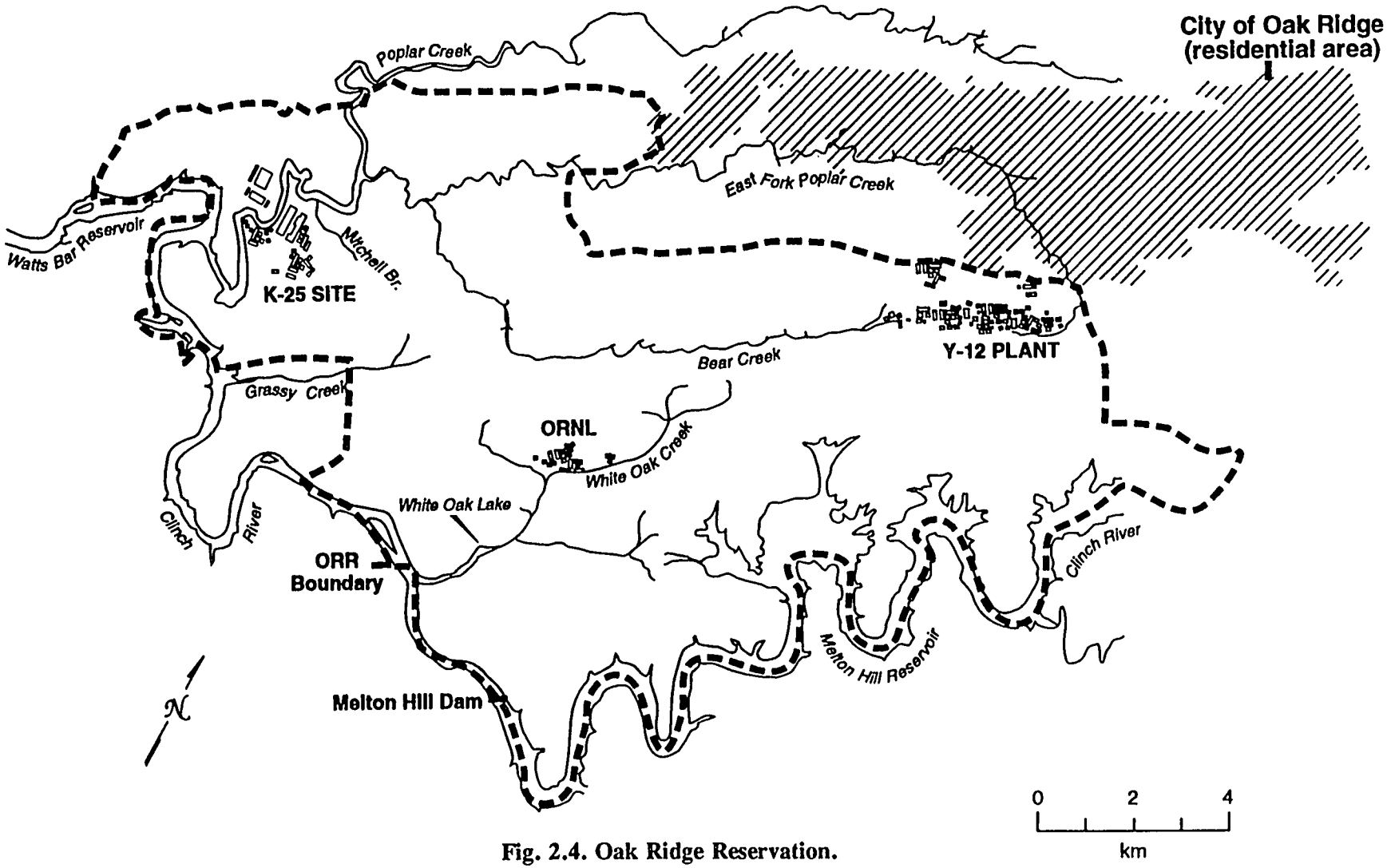

Fig. 2.4. Oak Ridge Reservation.

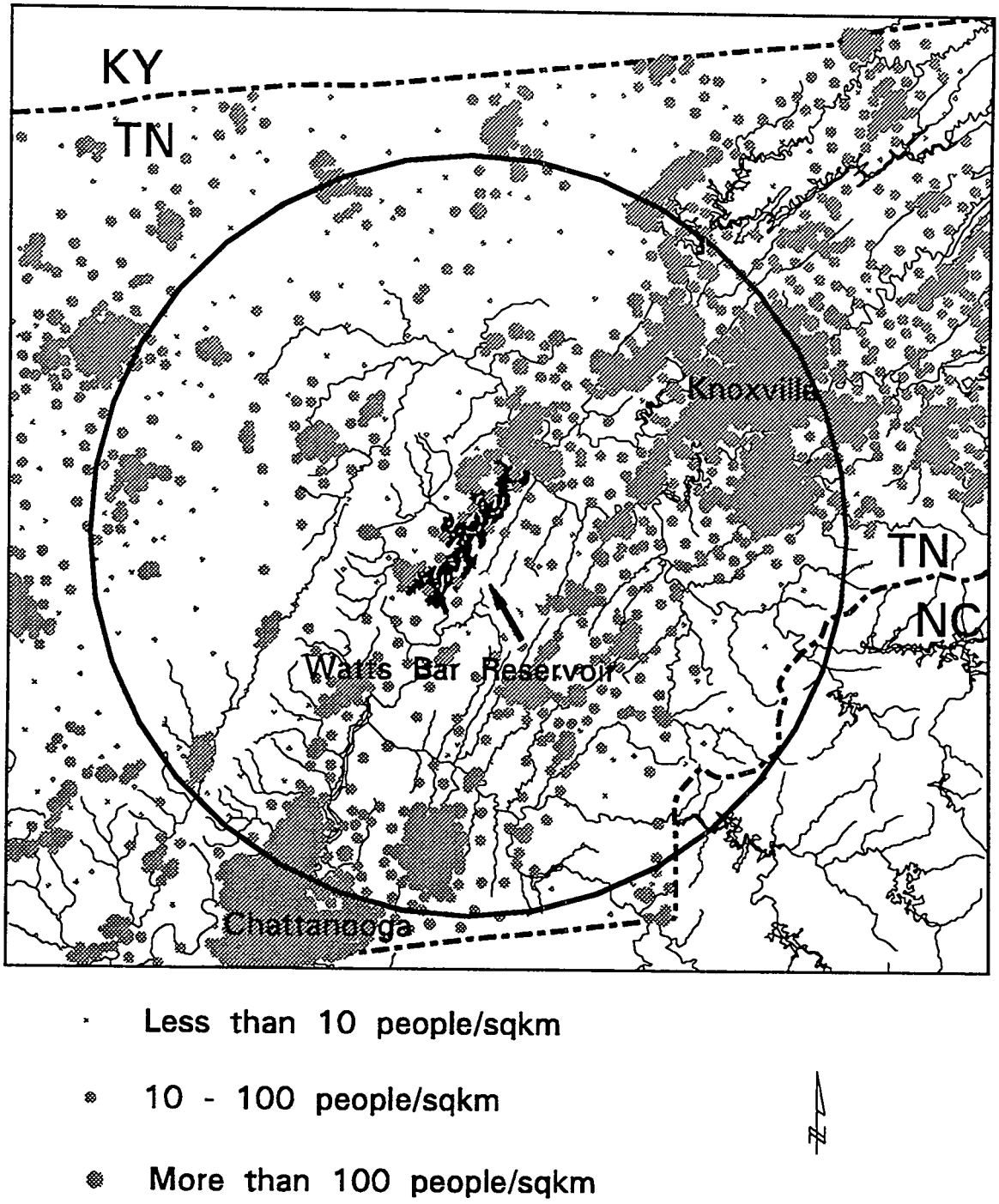

Fig. 2.5. Population density within a 50-mile radius of Watts Bar Reservoir.

Table 2.1. Population centers near Watts Bar Reservoir and water intake location information

County	Town	Population[a]	Water Intake[b]
Anderson (Pop. 68,250)	Oak Ridge	27,310	Clinch River Mile 52.5 Melton Hill Reservoir
Loudon (Pop. 31,255)	Loudon	4,026	Tennessee River Mile 592.2 Upper Watts Bar Reservoir
	Lenoir City	6,147	Tennessee River Mile 601.3 Upper Watts Bar Reservoir
Roane (Pop. 47,227)	Kingston	4,552	Tennessee River Mile 568.4 Upper Watts Bar Reservoir
	Harriman	7,119	Emory River Mile 11.5
	Rockwood	5,348	Tennessee River Mile 553 King Creek Embayment of Lower Watts Bar Reservoir[c]
Rhea (Pop. 24,344)	Dayton	5,671	Tennessee River Mile 503.9 Chickamauga Reservoir
	Spring City	2,199	Piney River Mile 4.0 Lower Watts Bar Reservoir
Meigs (Pop. 8,033)	Decatur	1,361	Groundwater[d] (Big Spring, Eaves Spring)
Hamilton	Chattanooga	152,488	Tennessee River Mile 465.2 (Mouth of Citico Creek)
Knox	Knoxville	165,121	Tennessee River Mile 646.4 Tennessee River Mile 649.5

[a]Source: *1990 Census of the Population and Housing*. U.S. Dept. of Commerce, Bureau of the Census. 1990. Summary tape file 3A (TN) (magnetic disk).
[b]Intake location taken from TVA navigation charts (TVA 1990b).
[c]Personal communication from Janet Foland, city of Rockwood Water Department, to Riley Sain, Assistant Manager CR-ERP, March 28, 1994.
[d]Personal communication from Tomma Stiner, Decatur Water Department, to Riley Sain, Assistant Manager CR-ERP, March 28, 1994.

2.3 CLIMATE

The Watts Bar Reservoir area has a temperate climate with warm, humid summers and cool winters. The climate is moderated by the Cumberland Plateau to the west and the Blue Ridge Mountains to the east. The mean annual temperature is 58°F. The coldest month is usually January, with a mean temperature of 38°F. July is the hottest month, with a mean temperature of 77°F (Energy Systems 1993).

Winds in the Watts Bar area are influenced primarily by the surrounding topography. Prevailing winds are usually southwesterly during the day, moving up the Tennessee River

Valley. Nighttime winds generally move down the valley, to the southwest. Strong winds are rare, with wind speeds less than 7.4 mph occurring 75% of the time (Energy Systems 1993).

Precipitation in the Watts Bar area averages between 50 and 55 in. per year. The wettest months are December through March, when slow-moving fronts may result in low-intensity storms of long duration. Another peak occurs in summer, when convective currents create high intensity, short-duration, localized thunderstorms. The driest period is typically autumn, when slow-moving high pressure cells may suppress rain for extended periods. Average annual evapotranspiration is estimated from 32 to 35 in., or 66% of rainfall. Evapotranspiration is greatest during the growing season, often exceeding the rate of precipitation, resulting in soil moisture deficits. In winter, precipitation exceeds evapotranspiration, resulting in increased water available for runoff and groundwater recharge (Energy Systems 1993).

2.4 TOPOGRAPHY, GEOLOGY, AND SOILS

2.4.1 Topography

Watts Bar Reservoir and the ORR both lie in the Ridge and Valley Province (Fig. 2.6). This Province is bordered on the east by the Blue Ridge Province and on the west by the Cumberland Plateau Province. The Ridge and Valley Province extends for 1200 miles from the Canadian St. Lawrence valley to the Gulf Coastal Plain of Alabama. This area is characterized by a succession of northeast trending ridges of various widths. The topography is reflective of the underlying geology, in which ridges are underlain by less soluble cherty limestones, dolomites, and sandy shales, while the valleys are underlain by more soluble limestones, dolomites, and shales. Extensive folding and thrusting throughout the Province has resulted in the characteristic belted pattern of rock formations (Energy Systems 1993).

2.4.2 Geology

Watts Bar Reservoir and the ORR are underlain by nine different geologic formations or groups of formations, all of sedimentary origin (Fig. 2.7). A generalized stratigraphy is presented in Table 2.2. The geologic units that directly underlie the largest extent of the LWBR study area are the Knox Group, the Chickamauga Group, and the Conasauga Group (Fig. 2.7).

The northeast trending ridges of the LWBR study area result from a succession of northeast-trending thrust faults that have structurally stacked and replicated the rocks of the geologic units. Most of the prominent ridges are underlain by the Rome or Knox groups, while many of the valleys are underlain by the Chickamauga or Conasauga groups. The major faults in the LWBR area were formed during the Permian-Pennsylvanian age and are not active structures (Energy Systems 1993). Competent bedrock in each of these formations is generally overlain by a mantle of *regolith,* a zone of weathered, unconsolidated materials that form in place via chemical and physical weathering of the underlying bedrock. Above the regolith is a relatively thin layer of soil or alluvial sediment.

2.4.3 Soils

The land area adjacent to the LWBR area is primarily overlain by residual soils that have weathered in place. The most extensive soils in the study area are the Fullerton, Bodine, Talbott, Colbert, and LeHew soils. In general, soils can be correlated with the underlying geologic unit

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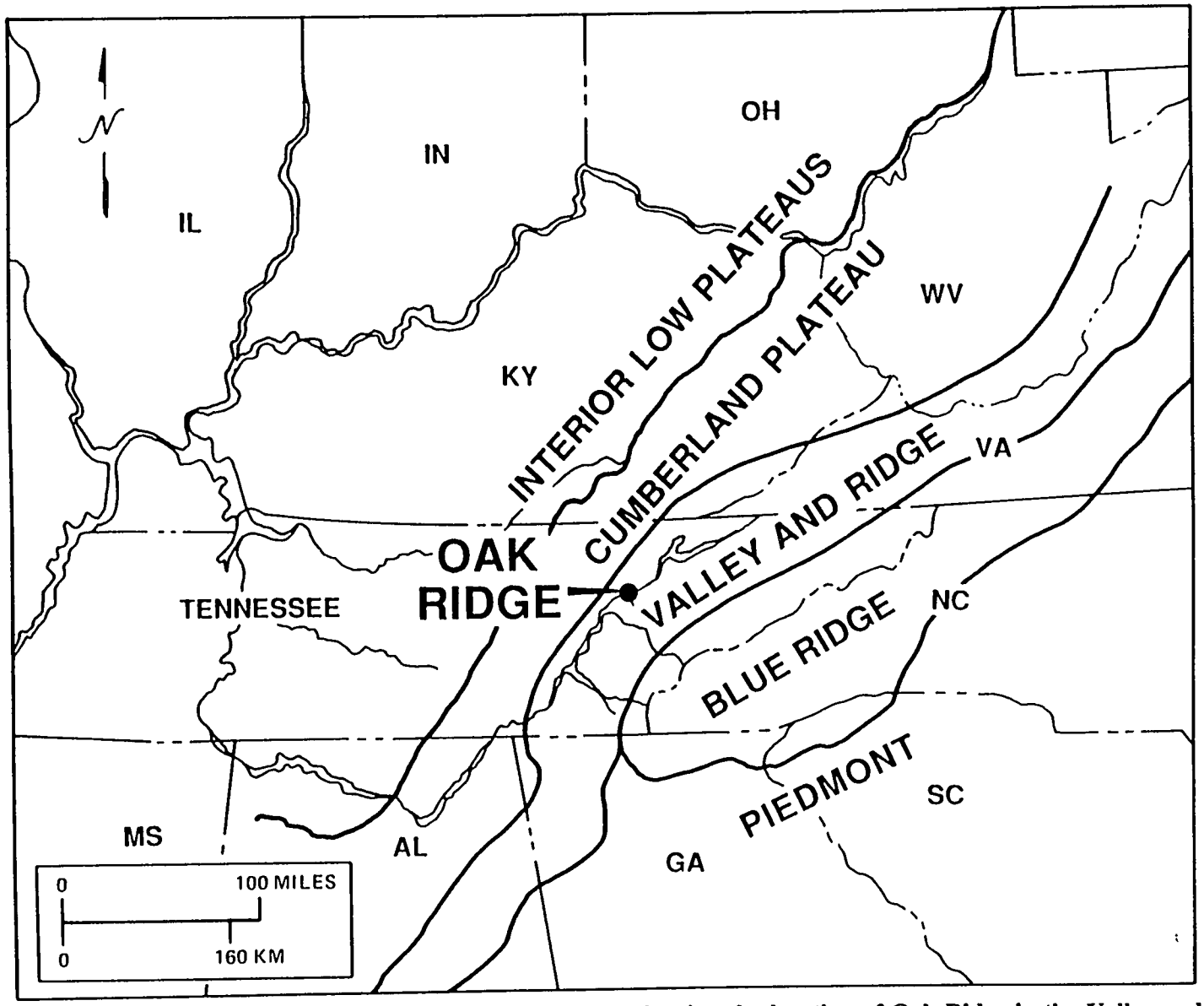


Fig. 2.6 Regional physiographic map of eastern Tennessee showing the location of Oak Ridge in the Valley and Ridge Province (Energy Systems 1990).

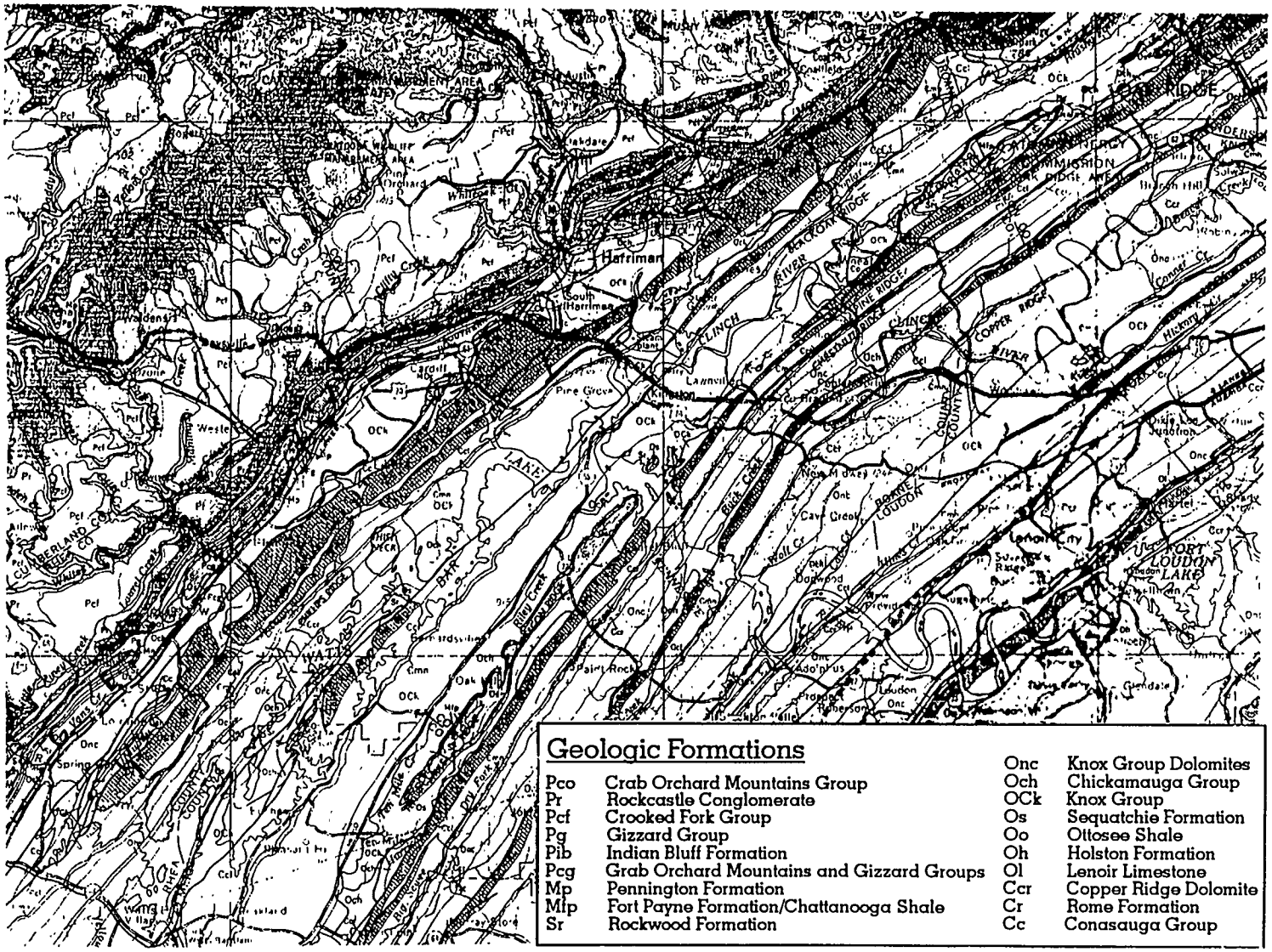


Fig. 2.7. Geologic map of eastern Tennessee (Hardeman 1966).

Table 2.2. Generalized stratigraphic profile of the Oak Ridge Reservation. The geologic formations are of sedimentary origin and are arranged chronologically with the most recent formations at the top and the oldest at the bottom (adapted from Energy Systems 1990).

Group or Formation (and Geologic Age)	Thickness	Characteristics of Rocks
Ft. Payne Formation (Mississippian)	25 meters	Impure limestone and calcareous siltstone, with abundant chert
Chattanooga Shale Formation (Mississippian and Devonian)	7 meters	Black fissile shale
Rockwood Formation (Silurian)	200 meters	Siltstone, sandstone, and shale; yellow to tan and brown
Sequatchie Formation (Ordovician)	90 meters	Siltstone, shaly limestone, sandy shale, calcareous; maroon to gray olive
Chickamauga Group (Ordovician)	600 meters	Limestone, shaly limestone, calcareous siltstone, and shale; mostly gray, partly maroon, with cherty zones in basal portions
Knox Group (Ordovician)	900 meters	Dolostone and dolomitic limestone; light to dark gray; with prominent chert zones
Conasauga Group (Cambrian)	550 meters	Limestone, siltstone, and shale; gray, olive, brown
Rome Formation (Cambrian)	250+ meters	Sandstone and shale; variegated with brilliant yellow, brown, red, maroon, olive-green; with domestic limestone lenses

that serves as the parent material (Table 2.3). The characteristics of any soil are highly localized and can vary widely even within a soil type. Alluvial soils are also present to a limited extent within the study area, primarily on high river terraces of ancient river floodplains and in narrow tracts along modern streams.

2.4.4 Groundwater

A conceptual model of groundwater occurrence has recently been formulated for the ORR (Solomon et al. 1992, Moore and Toran 1992). Given the shared geology, topography, and climate of the ORR and the Watts Bar Reservoir area, many elements of this model also apply to the LWBR study area as a whole. The hydrology of the area can be considered in two broad hydrologic units: (1) the Knox aquifer and (2) the aquitards (Energy Systems 1993). The Knox aquifer consists of the Knox Group and the underlying Maynardville Limestone of the Conasauga Group (Energy Systems 1993), both of which consist of massive carbonate rocks. Although the

Table 2.3. Soil parent material, associated soils series, and soil characteristics of the major soil series in the lower Watts Bar Reservoir area[a]

Geologic Unit / Parent Material	Associated Soil Series	Soil Characteristics
Knox Group	Fullerton Bodine	Residual soils of cherty limestone hills and ridges; deep, well-drained, very old soils
	Minvale	Colluvial soils on footslopes and benches of cherty limestone hills
	Decatur Waynesboro	Old alluvial soils; deep, well-drained soils of ancient river terraces
	Humphreys Ennis Lobelville	Recent alluvial soils; deep, well-drained, nearly level soils of bottom lands; derived primarily from Fullerton and Bodine soils
Rome Formation	LeHew Montevallo	Residual soils of high, steep, ridges; shallow, loamy, excessively drained soils formed from sandstone/shale (LeHew) or shale (Montevallo)
	Shouns	Colluvial soils on foot slopes and benches at base of shale and sandstone ridges; deep, well-drained, loamy
Chickamauga Limestone	Talbott Colbert	Residual soils of limestone valleys; characterized by plastic clay subsoils. Talbott soils are deeper and better drained
	Etowah Capshaw	Alluvial soils of old terraces
	Beason	Recent alluvial soils of floodplains
Conasauga Shale	Sequoia Litz	Residual soils of low rolling hills; moderately deep, well-drained, older Sequoia soils are on hilltops; younger, more shallow Litz soils are on hillsides
	Lindside	Recent alluvial soils of floodplains

[a] Adapted from Warren 1974.

matrix of this unit exhibits low primary porosity and permeability, flow is controlled by a combination of secondary fractures and solution conduits. Large volumes of water may move relatively long distances in this unit. The Knox aquifer is the primary source of base flow for many streams in the study area. Large springs of high yield issue from the Knox aquifer over a wide geographic range within the study area (DeBuchananne and Richardson 1956), and all large springs on the ORR issue from this unit (Energy Systems 1993). Yields of some wells penetrating larger solution conduits within this unit yield in excess of 1000 gallons per minute. The Knox aquifer has been described as the most important aquifer in eastern Tennessee (DeBuchananne and Richardson 1956).

The remaining geologic units underlying the majority of the LWBR area (the Rome Formation, the Conasauga Group below the Maynardville Limestone, and the Chickamauga

Group) constitute the aquitards (Energy Systems 1993). These units are all of low primary porosity and permeability; secondary permeability is usually much reduced compared to the Knox aquifer. These units are more likely to produce wells of lower yield than the Knox, and base flow to streams is also typically much reduced (Energy Systems 1993).

Within both hydrologic units, three major zones of groundwater occurrence may be identified (Fig. 2.8). The *stormflow zone* is a transient, shallow subsurface zone consisting of the upper 3–6 ft of soil, roughly corresponding to the root zone. The unsaturated *vadose zone* underlies the stormflow zone and consists of the unconsolidated regolith or bedrock. It ranges in thickness from 0 ft near perennial streams, to as much as 50 ft beneath ridges underlain by members of the Rome Formation, to more than 100 ft beneath ridges underlain by the Knox aquifer (Energy Systems 1993). An unconfined *saturated zone* underlies the vadose zone. Its upper boundary constitutes the water table, which may extend into the regolith and approach the surface near perennial streams. The saturated zone extends down through the bedrock. At depth (generally more than 200 m), the quality of the water changes from freshwater to brine, which is sometimes used to delineate the functional basement of this zone (Energy Systems 1993).

On the ORR, as well as the surrounding region, the majority (95%) of the active subsurface flow is via the shallow stormflow zone to nearby surface water streams. The vadose zone transmits water vertically to the saturated zone, necessary for recharge of the saturated zone. Saturated zone flow in the aquitards occurs primarily (95%) in the upper 50 to 100 ft; consequently, flow routes are short, and discharge is to nearby surface streams. In the Knox aquifer, a few solution conduits may be as much as 2 miles long. No evidence exists, however, for deep regional flow off the ORR, or even between basins on the ORR, for either the aquitards or the Knox aquifer (Energy Systems 1993). Therefore, the Clinch and Tennessee rivers represent the ultimate hydrologic "sink" for the region; surface water and groundwater flowing from the ORR is ultimately transported downstream through the Clinch and Tennessee rivers.

2.5 SURFACE WATER

2.5.1 Physical Characteristics

A hypsograph (Fig. 2.9) depicts LWBR water volume and surface area at various water elevations. At full pool, Watts Bar Reservoir has a surface area of 39,000 acres, a length of 96 miles, and a shoreline of 783 miles (TVA 1990a, Meinert and Dycus 1993). The LWBR OU is 38 miles long and approximately 24,000 acres in surface area. The surface water elevation at normal full pool is 741 ft above mean sea level (msl), with an average annual drawdown of 6 ft (TVA 1990a, Meinert and Dycus 1993). The volume of water in the reservoir at normal full pool is 1,010,000 acre-ft, with an average depth of 26 ft and a depth of 105 ft at the dam (Meinert and Dycus 1993). The top of the gates is at elevation 745 ft above msl, providing an additional 379,000 acre-ft of flood storage capacity. The annual average discharge from Watts Bar Dam is approximately 27,800 cubic feet per second (cfs), resulting in an average water retention time in the reservoir of 18 to 19 days (TVA 1990a, Meinert and Dycus 1993) and in LWBR of 13 days. Watts Bar Reservoir was impounded in January 1942.

Watts Bar Reservoir drains an area of 17,300 square miles (Meinert and Dycus 1993), including portions of Tennessee, Virginia, Georgia, and North Carolina (Fig. 2.10). The major inflows to the reservoir are the Tennessee River at Fort Loudoun Dam, which contributes approximately 75% of the total inflow (15,700 cfs annual average), and the Clinch River at

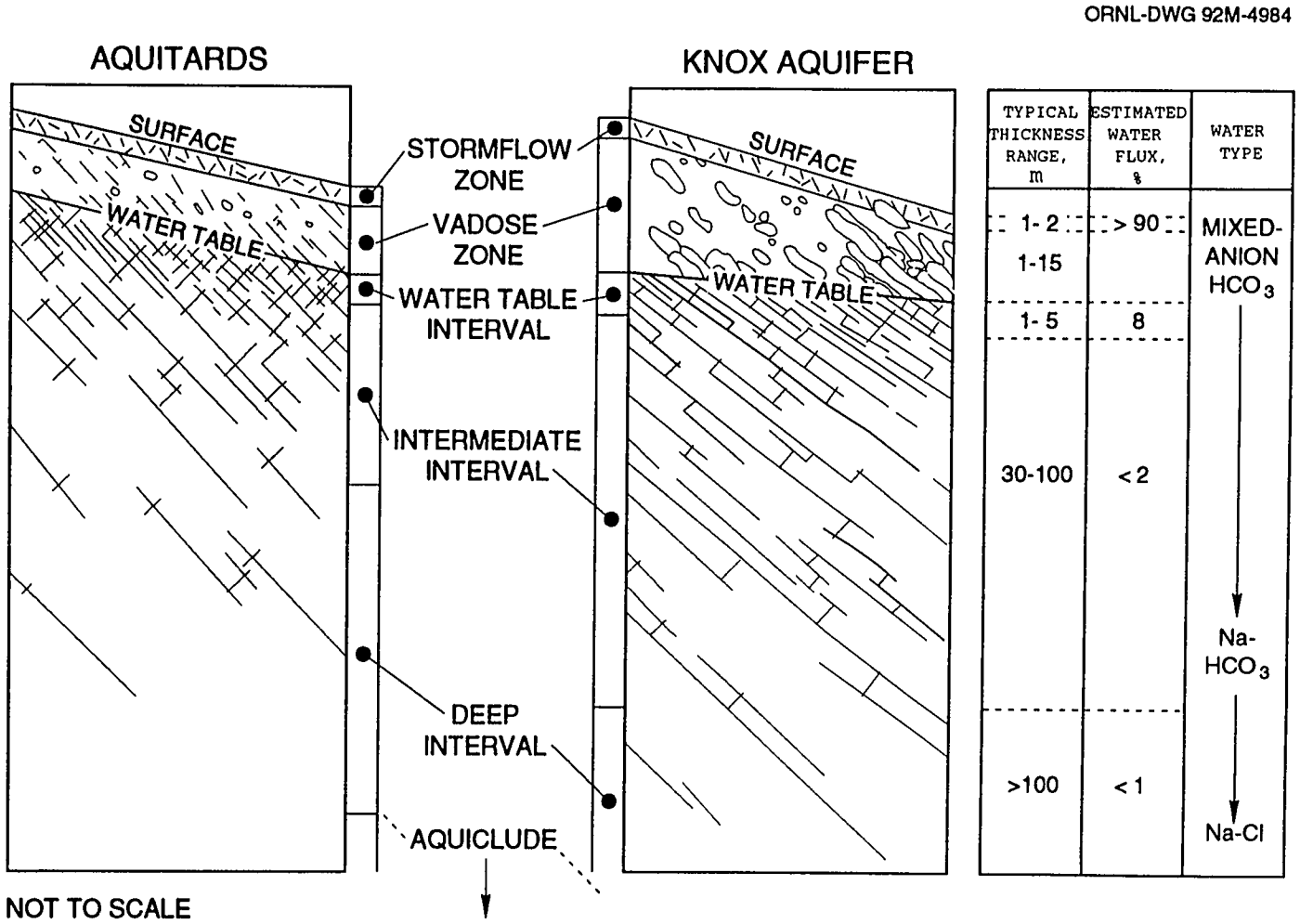


Fig. 2.8. Schematic vertical relationships of flow zones of the ORR, estimated thickness, water flux, and water types.

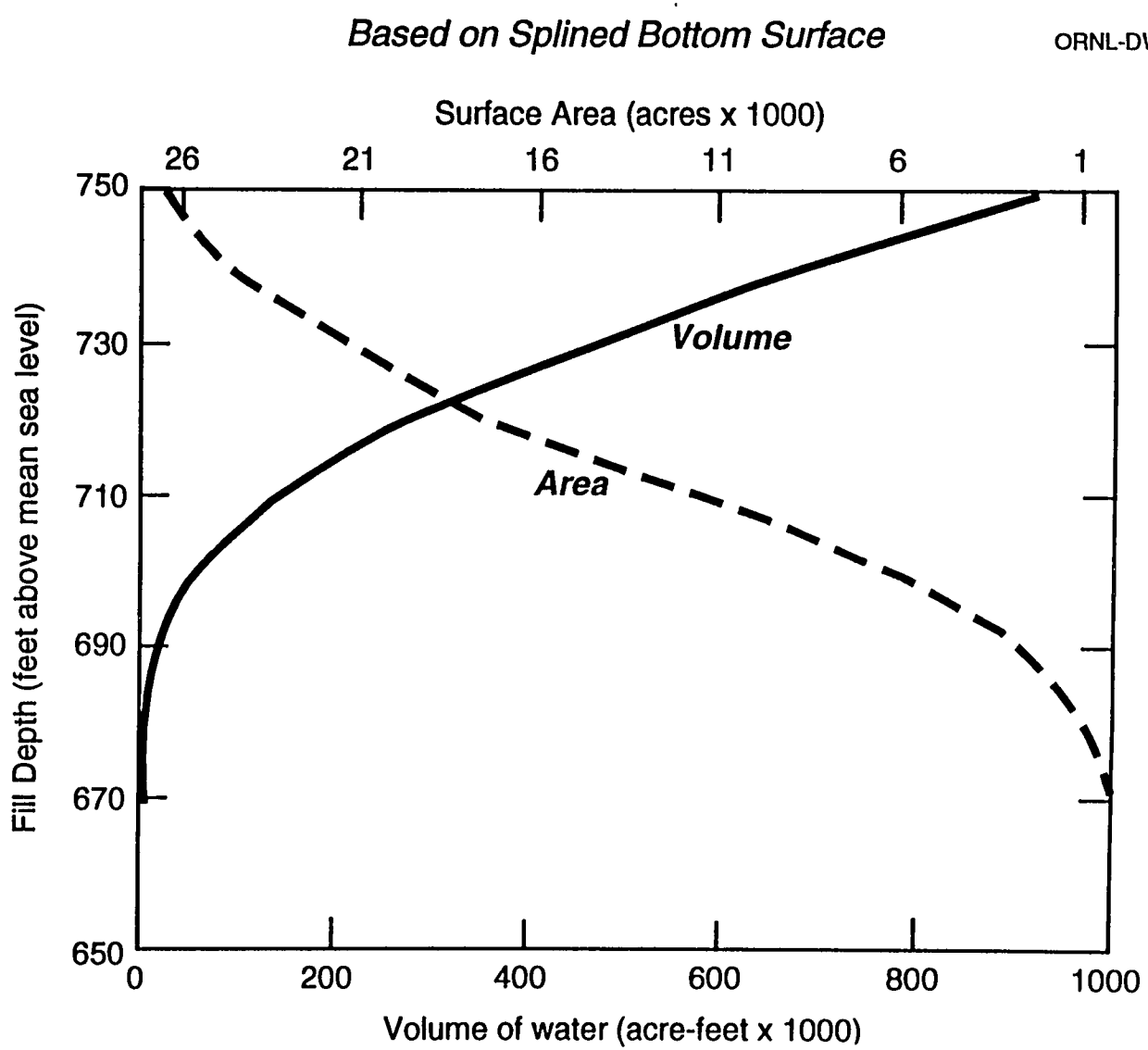


Fig. 2.9. Hipsograph of Lower Watts Bar Reservoir.

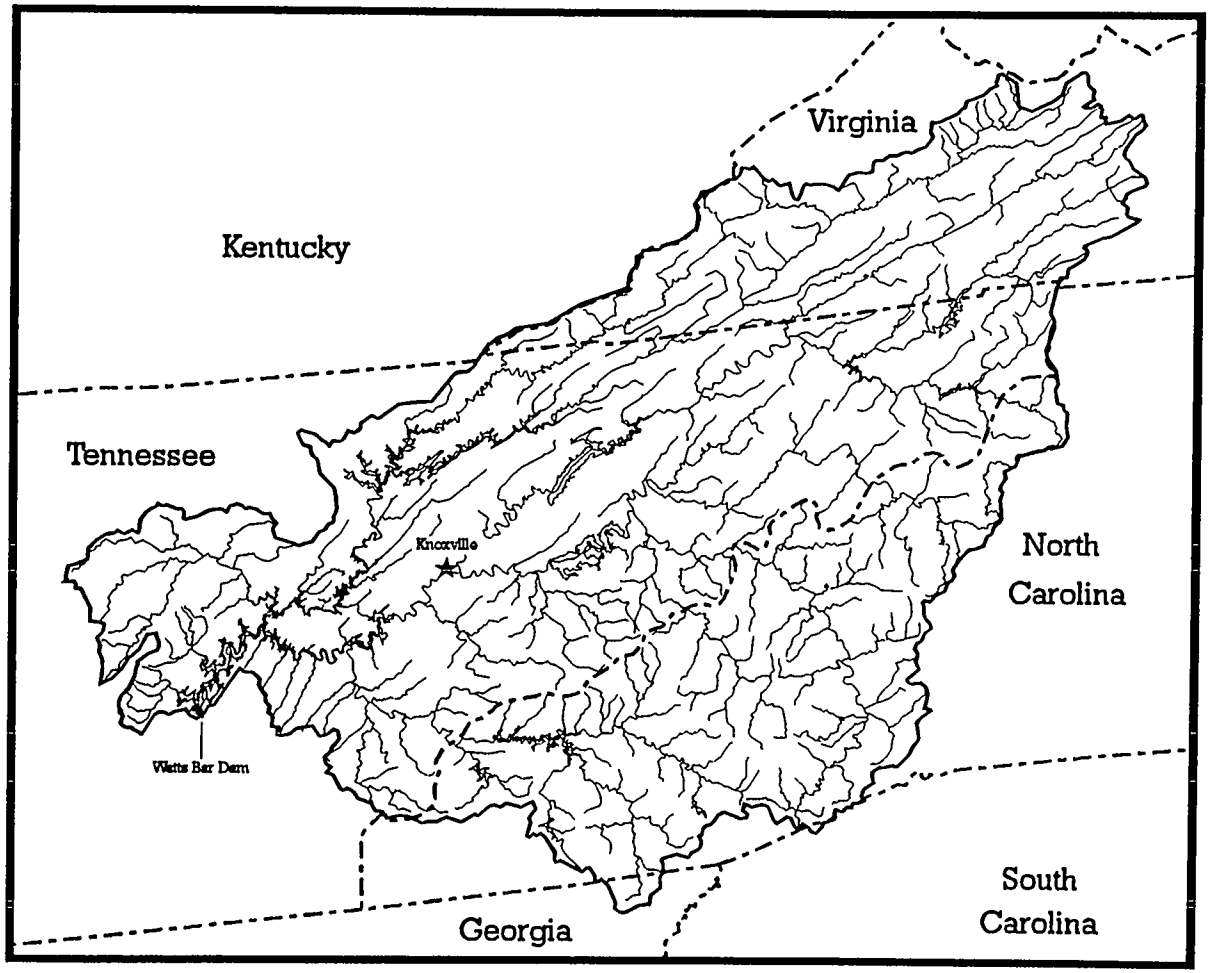


Fig. 2.10. Drainage area of the Tennessee River at Watts Bar Dam. Total area of 17,300 square miles.

Melton Hill Dam, which contributes another 15% (4,400 cfs annual average) (Meinert and Dycus 1993). The Emory River, a tributary of the Clinch River at CRM 4.0, contributes another 5% of the total flow. The remaining 5% is contributed by minor tributaries to the Clinch and Tennessee rivers, direct precipitation to the reservoir, and groundwater discharge to the reservoir. Many small streams flow into LWBR; the largest of these are Piney and Whites creeks in Rhea County, which have watersheds of 137 and 138 square miles, respectively (TVA 1970). The Little Tennessee River formerly contributed flow directly to upper Watts Bar Reservoir, but Tellico Dam, completed in 1979, now diverts almost all of this flow to Fort Loudoun Reservoir via a navigation channel connecting the forebays of these two reservoirs.

The ORR is drained by several small tributaries to the Clinch River, the largest of which are White Oak Creek (WOC) and Poplar Creek (Fig. 2.4). WOC drains an area on the ORR of approximately 6.0 square miles, including the ORNL site, and enters the Clinch River at CRM 20.8. Average annual flow in WOC is about 13.5 cfs (Energy Systems 1993).

Poplar Creek enters the Clinch River at CRM 12.0. It is the largest stream on the ORR, with a drainage area of 135.9 square miles and an average annual discharge of approximately 228 cfs (Energy Systems 1993). Poplar Creek flows through and drains the K-25 Site on the ORR. Above the K-25 Site, East Fork Poplar Creek (EFPC) flows into Poplar Creek at mile 5.5. EFPC has its headwaters at the Y-12 Plant on the ORR and also runs through the city of Oak Ridge prior to its confluence with Poplar Creek. The Y-12 Plant is also drained by Bear Creek, a tributary to EFPC. Poplar Creek also drains an area upstream of the ORR, including the city of Oliver Springs and a portion of the Cumberland Mountains.

Areas within Watts Bar Reservoir may be riverine in nature or more lake-like (lacustrine). Lacustrine areas are characterized by very slowly moving waters. Because of the reduced water velocity, lacustrine areas are prone to thermal and chemical stratification, and they accumulate fine sediment. In Watts Bar Reservoir, the portion of the reservoir from the dam to the upstream end of Thief Neck Island (TRM 556) could be considered lacustrine. Even this area, however, is not strongly lacustrine, because the overall flow rate is great enough to result in a relatively short average residence time for water. By contrast, TVA's storage impoundments on tributary streams (e.g., Norris, Fontana, Cherokee, and Douglas reservoirs) have much longer residence times, become much more strongly stratified during summer, are deeper, and are more lacustrine in nature than Watts Bar Reservoir. The inflow sites below Melton Hill Dam on the Clinch River and Fort Loudoun Dam on the Tennessee River are riverine in nature, although flow is dependent on the water release schedule and is, therefore, intermittent. These areas are not subject to stratification and generally have little or no fine sediment accumulation. The Emory River is free flowing above (approximately) Emory River Mile 8.0–9.0. Although affected by impoundment, the Tennessee River from Thief Neck Island upstream to approximately the Interstate 75 bridge (TRM 584.9) and the Clinch River from its mouth upstream to approximately CRM 6.0–7.0 exhibit characteristics of both lacustrine and riverine systems and may best be thought of as transition areas.

Watts Bar Reservoir is managed by TVA to provide for flood control, navigation, water supply, aquatic habitat, and recreation. Water levels are manipulated seasonally to provide these benefits (Fig. 2.11). During the winter months, the surface of the reservoir is maintained at elevation 735 ft above msl to provide flood storage capacity for spring runoff. In mid April, the water level is gradually raised to the summer pool elevation of 741 ft above msl to provide recreational benefits and to improve shoreline aquatic habitat.

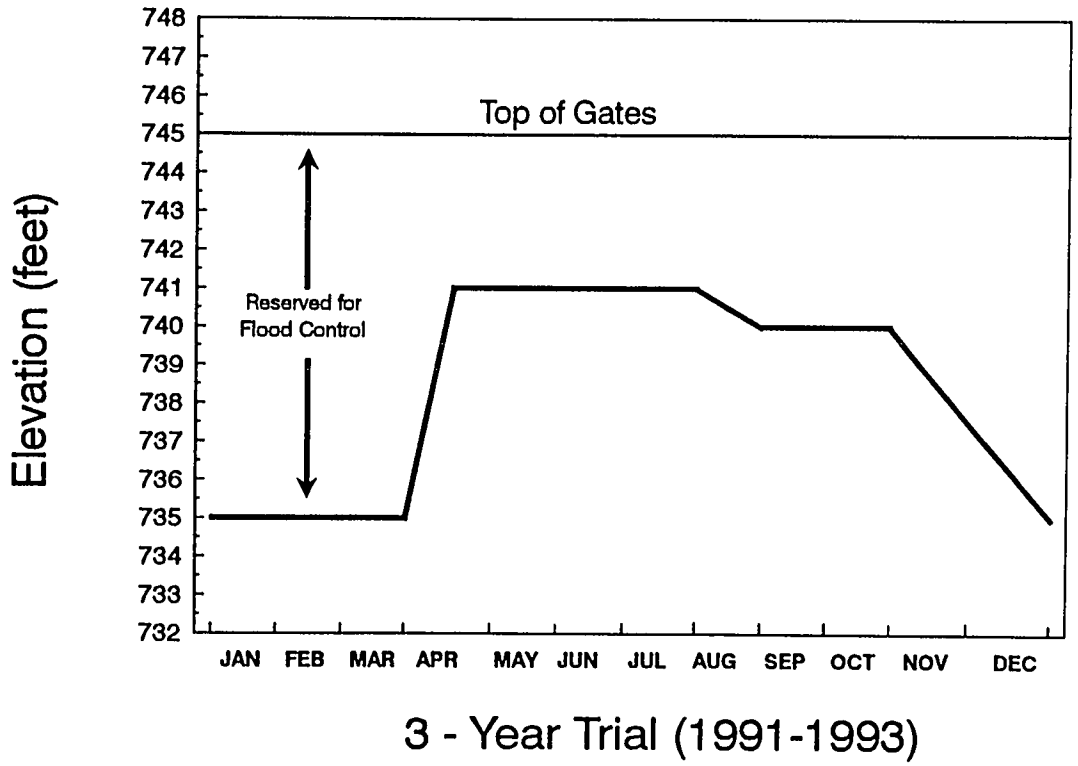


Fig. 2.11. Seasonal water level elevations in Watts Bar Reservoir.

2.5.2 Water Quality

Watts Bar Reservoir is a circumneutral, medium-hardness, mildly eutrophic reservoir (TVA 1990a, Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). Because the Clinch River drains a predominantly limestone and dolomitic region, its waters are relatively high in magnesium and calcium (as calcium bicarbonate), resulting in waters that are significantly harder than those of the Tennessee River (Churchill et al. 1965). Much of the Tennessee River watershed above Watts Bar Reservoir lies in the Blue Ridge Province, where resistant siliceous rocks contribute little to the hardness of surface water. The reservoir typically exhibits weak to moderate thermal stratification, often beginning in late spring and extending through early fall (TVA 1990a, Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). This stratification results primarily from the natural process of stratification in lakes and reservoirs but is enhanced in Watts Bar Reservoir by the release of cool water from Fort Loudoun and Melton Hill dams upstream. Stratification results in the formation of layers of water in the reservoir that do not mix well and that typically exhibit different physical and chemical characteristics. The upper layer, or epilimnion, is generally warmer and has a higher concentration of dissolved oxygen (DO) than the lower layer, or hypolimnion.

This weakly developed thermal stratification in Watts Bar Reservoir nevertheless promotes a relatively strong gradient in DO concentrations with depth, particularly at the forebay where the hypolimnion is best developed. Here concentrations of DO in the bottom waters often drop below 1 mg/L during the mid to late summer months (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). Adequate DO is essential to aquatic life. At least 3 or 4 mg/L are necessary for most fish species to survive (TVA 1990a). At least 5 to 6 mg/L are needed to avoid adverse impacts to fish growth, and higher levels are required during spawning and in the first 30 days of juvenile life (TVA 1990a). Many benthic organisms, including freshwater mussels, do not tolerate low DO, and improvements in benthic community diversity usually require at least 5 mg/L DO (TVA 1990a). Low DO concentrations in the hypolimnion result from the lack of atmospheric reaeration and from the consumption of oxygen by bacteria that degrade organic matter settling out of the overlying water column. Lacustrine environments are much more efficient at retaining organic matter within the system than riverine systems.

During the late fall, winter, and early spring months, the pH of Watts Bar Reservoir is typically at or slightly below neutral. In late spring, as stratification develops, the pH of the epilimnion often exceeds 9.0 for short periods due to algal photosynthesis. At the same time, DO concentrations in the epilimnion rise dramatically, also as a result of increased algal photosynthesis, and typically reach supersaturated levels in the epilimnion for short periods (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993).

The Tennessee Water Quality Control Board has promulgated water quality standards for the protection of designated uses of surface waters. The designated uses of the Tennessee River in the LWBR study area are for domestic water supply, industrial water supply, fish and aquatic life, recreation, irrigation, livestock watering and wildlife, and navigation (TDEC Rule 1200-4-4). Each of these uses is expected to occur in LWBR, but the extent to which some occur is not well known. For example, the extent to which the Reservoir is used for livestock watering is unknown. The use of LWBR for irrigation is believed to be limited, but small-scale irrigation of lawns and gardens in residential areas bordering the Reservoir probably occurs. Large-scale agricultural irrigation is very limited. One farm in Roane County reportedly irrigates pasture that is grazed by cattle (personal communication from Paul McCallie, Roane County Extension Leader, to Riley Sain, Assistant Manager CR-ERP, May 6, 1994). A Meigs County farm

reportedly uses LWBR to irrigate a 125-acre field to produce corn for use as silage for dairy cattle, and a local nursery irrigates ornamental fruit trees that it raises for sale (personal communication from Tom Swanks, Meigs County Agricultural Extension Agent, to Riley Sain, Assistant Manager CR-ERP, May 6, 1994). No instances in which LWBR water is used for irrigation are known in Rhea County (personal communication from Dwayne Webb, Rhea County Extension Leader, to Riley Sain, Assistant Manager CR-ERP, May 6, 1994). Watts Bar Reservoir currently does not fully support its designated uses because of reservoir-wide fish consumption advisories that are in effect (personal communication from Greg Denton, Division of Water Control, Tennessee Department of Environment and Conservation, to Riley Sain, Assistant Manager CR-ERP, February 10, 1994).

2.6 SEDIMENT

Reservoirs act as highly efficient sediment traps due to an increased cross-sectional area and decreased water velocity, resulting in a decreased sediment transport capacity as compared to riverine conditions (Energy Systems 1990). In such an environment, organic matter and fine-particle sediments, such as silts and clays, settle out of the water column and accumulate on the reservoir bottom. Accumulated sediments in LWBR are almost entirely silt and clay (TVA 1986a, TVA 1990a, Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). Overall surface sediment types in LWBR are depicted in Fig. 2.12. Sediment input to LWBR is primarily from the Tennessee River (60%) and the Clinch River (35%), with smaller tributaries contributing the remaining 5%.

Trimble and Carey (1984) have estimated that Watts Bar Reservoir traps 60% of the sediment passing through upstream dams (an average of 80 mg/L suspended sediment) and 80–85% of local sediment (an average of 420–470 mg/L). Because of the completion of dams upstream of Watts Bar Reservoir, the local watershed has been reduced over the years, decreasing the rate of sediment accumulation in Watts Bar Reservoir (Table 2.4). The reduction in sediment delivery rate, yield, and accumulation between the 1946–1963 period and the 1964–1979 period, shown in Table 2.4, is probably not entirely due to the completion of Melton Hill Dam, but most likely also reflects changing land use during these years.

As noted in Table 2.4, an estimated 29,112 acre-ft of sediment has accumulated in Watts Bar Reservoir during the period 1946–1991, based on TVA silt-range data (Ewing 1993). The average annual rate of sediment accumulation has not been constant over this period. Brenkert et al. (1992) used the silt-range data to calculate the average annual sediment deposition rate for four subperiods during this period (Table 2.5).

The sedimentation rates calculated from the TVA silt-range data provide an estimate of the average sedimentation rate throughout LWBR. However, sediment does not collect uniformly within the reservoir. Deep-water areas along the main channel accumulate sediment at a greater rate than do shallow-water areas (Olsen et al. 1992), a phenomenon known as sediment focusing. Sediment focusing explains the greater rate of sediment accumulation in deep water areas as a function of (1) the height of the water column, which provides a greater quantity of suspended sediment for settling and (2) the lateral movement of sediment downslope as a result of wave and current action. Brenkert et al. (1992) used data on 60 sediment cores collected by Olsen et al. (1992) to estimate an average annual sedimentation rate in lower Watts Bar Reservoir of 1.15 cm/year for the period 1956–1987, as compared to the 0.63 cm/year rate for a comparable time period using silt-range data (Table 2.5). The higher average annual sedimentation rate

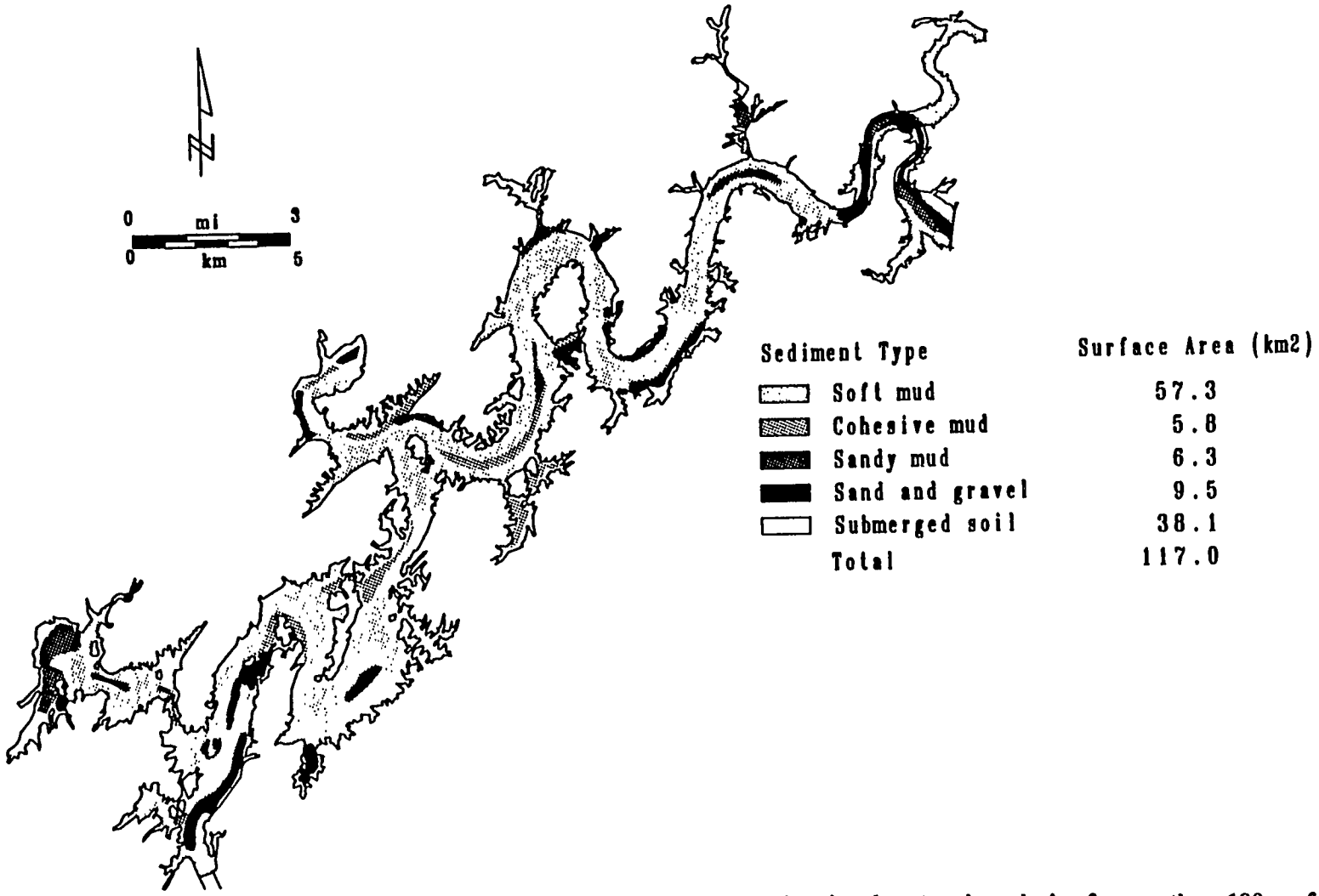


Fig. 2.12. Map of the surface sedimentary characteristics based on the visual textural analysis of more than 190 surface sediment samples collected in Watts Bar Reservoir (Olsen et al. 1992).

calculated from the Olsen et al. (1992) data is due to preferential sampling of deep-water areas. The average annual sedimentation rate for individual cores collected by Olsen et al. (1992) ranged from 0.03 to 2.8 cm/year (unpublished data). Longitudinally, significant deposition areas occur in LWBR upstream of Thief Neck Island and immediately upstream of Watts Bar Dam (Brenkert et al. 1992).

Table 2.4. Local watershed area and estimated sediment accumulation, sediment yield, and sediment delivery rate for Watts Bar Reservoir for the period 1946–1991. Data from Ewing (1993).

Period	Watts Bar Reservoir local watershed area (mi^2)	Nearest upstream dam			Estimated sediment yield (tons per mi^2 per year)	Estimated uniform sediment delivery rate (acre-ft per mi^2)	Estimated sediment accumulation in Watts Bar Reservoir (acre-ft)
		Tennessee River	Clinch River	Little Tennessee River			
1946-1963	3,277	Fort Loudoun	Norris	Fontana	568	7.11	21,160.4
1964-1979	2,846	Fort Loudoun	Melton Hill	Fontana	157	1.96	5,191.6
1980-1991	1,790	Fort Loudoun	Melton Hill	Tellico	117	1.47	2,760.0
1946-1991	-	-	-	-	281	10.54	29,112.0

Table 2.5. Average annual sedimentation rates for lower Watts Bar Reservoir based on TVA silt-range data (Brenkert et al. 1992)

Average annual sedimentation rate (cm/year)	Period				
	1946–1951	1951–1956	1956–1961	1961–1991	1956-1991
	0.8	0.7	2.2	0.4	0.63

2.7 ECOLOGY

A generalized ecosystem model for Watts Bar Reservoir is depicted in Fig. 2.13. Organic matter is the source of energy within the system and is supplied primarily from two sources: primary production by phytoplankton and allocthonous (watershed) inputs from various natural and man-made sources. Phytoplankton use solar energy to create organic matter from inorganic matter and are known as primary producers. Aquatic macrophytes also contribute to primary productivity, but are currently of limited extent in Watts Bar Reservoir (Meinert and Dycus 1993) primarily due to eradication programs implemented by TVA (1993). Additional organic matter (allocthonous material) is washed into the reservoir from various sources within the watershed. Some of the total organic matter supply becomes detritus (partially decomposed organic matter) and is consumed by various detritivores, including benthic invertebrates, detritivorous fish such as carp (*Cyprinus carpio*) and suckers (Family *Catostomidae*), and some omnivorous fish such

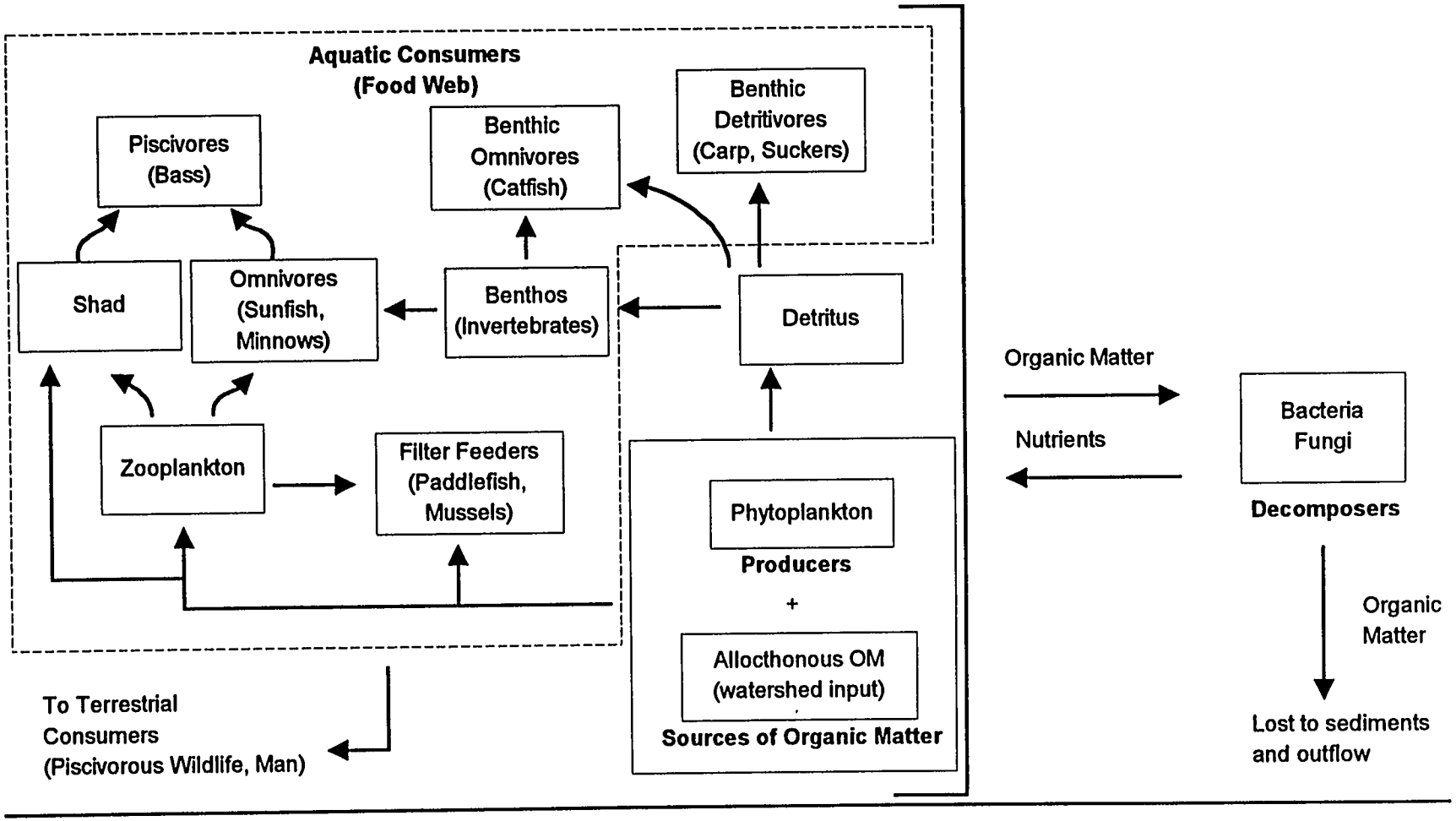


Fig. 2.13. Simplified general ecosystem diagram for LWBR. Adapted from Soballe et al. (1992).

as catfish (*Ictalurus sp*). In addition, phytoplankton and suspended allocthonous material is consumed by various herbivores and omnivores. Zooplankton are a major consumer of phytoplankton. Zooplankton, in turn, support omnivorous fish, such as sunfish (Family *Centarchidae*) and minnows (Family *Cyprinidae*), which also feed on benthic invertebrates. Filter-feeding fish such as paddlefish (*Polyodon spathula*) graze on plankton and suspended detritus, while mussels filter organic matter from the passing water column. In Watts Bar Reservoir, threadfin shad (*Dorosoma petense*) and gizzard shad (*D. cepedianum*) constitute a major food source (90%) for piscivorous fish such as largemouth bass (*Micropterus salmoides*) and striped bass (*Morone saxatilis*). Terrestrial consumers, including man, feed to varying degrees on fish and other aquatic organisms. As organisms die, organic matter from the system is decomposed by bacteria and fungi in the water column and sediment, making nutrients available for reincorporation into biomass. This process occurs at each step in the ecosystem; consequently, only a small amount of organic matter is lost via sedimentation.

The natural characteristics of the Tennessee River have been fundamentally changed by the construction of the TVA reservoir system (TVA 1990a). The daily, seasonal, and annual variations in flow patterns characteristic of a river have been disrupted as a result of impoundment. Movements of water, sediments, nutrients, and organic material have changed as a result, with far-reaching effects on the health, number, and diversity of aquatic life (TVA 1990a). The accumulation of sediment and nutrients in reservoirs contributes to higher productivity as compared to rivers. Reservoirs are more subject to stratification and the effects of low DO in the deeper zones. These changes have had a significant effect on benthic communities and on the habitat and species diversity of fish (TVA 1990a) in the Tennessee River since impoundment of Watts Bar Reservoir in 1942.

Benthic organisms include aquatic insects, worms, snails, crayfish, mussels, and clams that live part or all of their lives in the top few inches of sand, gravel, or mud at the bottom of streams and lakes. They are a vital component of the food chain, converting nutrients and organic materials to biomass and providing a food base for fish and other vertebrate predators (TVA 1990a). Most benthic organisms have specific habitat requirements in the form of physical, chemical, or biological factors; alterations of this habitat cause changes in both the composition and productivity of the benthic community. The habitat requirements of many benthic organisms are not always met in reservoirs or tailwaters below dams. Benthic organisms have virtually disappeared from the deep portions of reservoirs because of the lack of flow and low DO. In tailwater areas, only those species that can tolerate turbulent conditions and the potential low DO concentration that result from many dam releases can survive (TVA 1990a).

The impact of impoundment on the species diversity of the benthos community is most readily apparent in mussels. Because of their long life span, sedentary nature, and tendency to occur clumped in areas of suitable habitat, mussels are highly vulnerable to disruptions in habitat or other environmental factors. Impoundment has substantially reduced the amount of suitable mussel habitat (shallow, flowing water over stable gravel or cobble substrates), resulting in a drastic decline in mussel populations. Pollution, sedimentation, and commercial over-harvesting have affected those mussel populations that survived impoundment (TVA 1990a). The state of Tennessee has designated a 1000-yd reach below both Melton Hill and Fort Loudoun dams as mussel sanctuaries (TVA 1987). No commercial harvest of mussels is currently occurring in Watts Bar Reservoir.

Fish populations and communities have also been profoundly affected by construction of the TVA reservoir system. Species whose annual migration cycles were interrupted by dams and

those species whose requirements for temperature, spawning habitat, or food were closely associated with riverine conditions rapidly declined in abundance or were eliminated. Lake sturgeon (*Acipenser fulvescens*), blue sucker (*Cycleptus elongatus*), sauger (*Stizostedion canadense*), walleye (*S. vitreum*), paddlefish, and other stream spawning species were significantly reduced, as were the number of smaller fish, such as darters (Family *Percidae*) and minnows. Suitable habitat in tailwaters is limited due to wide fluctuations in temperature, flow, and often low DO. The most productive region of a reservoir for fish is the shoreline because of submerged vegetation for cover and the relatively high availability of organic material, nutrients, and aquatic invertebrates for food. These conditions allow warm water species, such as largemouth bass, crappie (*Pomoxis sp*), striped bass, carp, smallmouth buffalo (*Ictiobus bubalus*), and catfish, to thrive. However, this habitat is susceptible to fluctuations in water level and to shoreline development. The latter generally results in decreased shoreline vegetation and increased non-point source pollution from runoff from nearby lands (TVA 1990a).

2.7.1 Aquatic Resources

Watts Bar Reservoir is considered the third most eutrophic of TVA's nine mainstem reservoirs (TVA 1990a), indicating a high net primary productivity. Phosphorous is the limiting nutrient that prevents even greater productivity (Elser and Kimmel 1985, Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). As discussed in Sect. 2.5, the higher pH values and supersaturated DO conditions in the epilimnion during summer months indicate high photosynthetic activity (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). Algal growth is sufficiently high to reportedly be a nuisance near the surface of the reservoir at times (TVA 1990a). The high phytoplankton production provides a substantial food base for higher-trophic organisms (Alexander 1987).

Aquatic macrophytes, primarily the exotic weed Eurasian water milfoil *(Myriophylum spicatum),* covered over 1300 acres of the reservoir in 1983 (TVA 1987), primarily in areas such as shallow coves and along shorelines. Although these plants contributed to an increase in the largemouth bass fishery by providing cover and habitat for prey species (Alexander 1987), subsequent eradication programs implemented by TVA (1993) have reduced the areal coverage to an estimated 10 acres today (Meinert and Dycus 1993).

TVA (1993) has recently evaluated the environmental effects of its valley-wide aquatic plant management program. The goal of the program is not to eradicate aquatic macrophytes throughout an entire reservoir, but to reduce aquatic macrophyte populations in those areas of a reservoir where the plants cause greatest impairment of reservoir use, primarily for lake access, recreation, and water supply. These "priority areas" in Watts Bar Reservoir total approximately 66 shoreline miles and 1200 acres, or about 3.1% of the entire reservoir. The herbicides that have been used in Watts Bar Reservoir include (1) 2,4-D, used primarily against Eurasian water milfoil and applied at a rate of 10–40 lbs of active ingredient per acre, which results in a concentration in water of approximately 2.0 mg/L; (2) Endothall, used to control Spinyleaf naiad (*Najas minor*), hydrilla (*Hydrilla verticillata*), and pondweed (*Potamogeton* sp.), and applied at a rate of 5 gal per acre (as liquid dipotassium salt), resulting in a water concentration of approximately 2.0 mg/L; and (3) Diquat, used primarily to control naiads and *Elodea*, applied at a rate of up to 2 gal per acre, resulting in a water concentration of approximately 0.2 mg/L. The quantities of each herbicide that have been applied in Watts Bar, Melton Hill, Fort Loudoun, and Tellico reservoirs over a 20-year period (1972–1992) are summarized in Table 2.6.

Table 2.6. TVA herbicide application for aquatic weed control in Watts Bar, Melton Hill, Fort Loudoun, and Tellico reservoirs, 1972–1992. Values are total pounds of active ingredient applied (TVA 1993).

Year	Watts Bar Reservoir			Melton Hill Reservoir	Fort Loudoun Reservoir			Tellico Reservoir	
	2,4-D	Endothall	Diquat	2,4-D	2,4-D	Endothall	Diquat	2,4-D	Endothall
1972	88			2,204					
1973	267			38,466					
1974	86			23,142					
1975	608			10,108					
1976	2,356			15,580					
1976[1]	1,520			12,350					
1977	2,280			19,950					
1978	2,774			24,054	6,384				
1979	6,954			8,436	10,070		9		
1980	9,044		666	18,392	21,736		51		
1981	5,966	270	39	29,830	10,792		18		
1982	11,438	1,545		17,632	2,660				
1983	13,946	7,350		17,600	6,764	2,400			
1984	10,298	5,400		11,096	4,028				
1985	6,080	2,520	291	8,816	8,132		90		
1986	6,308	3,255	138	9,652	3,154			1,520	
1987	5,776	1,590	294	1,900	950			3,534	
1988	5,016		27	2,014	646			2,660	600
1989	6,878	885		3,078	1,558				
1990	190			798	304				
1991				665	380			95	
1992				437	38				

[1]July 1 through October 10, 1976.
Blank cells indicate no herbicide application.

Benthic organisms inhabiting Watts Bar Reservoir include aquatic insects, crustaceans, tubificid worms, annelids, and mollusks (TVA 1990a). In general, the deep-water sediments in the forebay support taxa, such as tubificid worms and chironomid larvae, that are tolerant of suboptimal conditions such as low DO (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). Upstream, where the reservoir conditions change from riverine to lacustrine, the benthic community is dominated by less tolerant taxa, such as the mayfly *(Hexagenia)* and the bivalve mollusk *(Musculium),* although chironomids also remain important (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). Although stratification is less pronounced in this area of the reservoir and DO values typically remain higher, the substrate remains essentially the same as in the forebay (almost entirely silt and clay), and, therefore, similarities between these benthic communities persist. Further upstream, where riverine conditions prevail below Fort Loudoun and Melton Hill dams, the benthic community is dominated by the introduced Asiatic clam *(Corbicula)* and by less tolerant species of chironomids (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). In addition, a few limited freshwater mussel stocks persist in the riverine portions of the reservoir (TVA 1990a).

Sixty-three fish species and one hybrid have been reported from Watts Bar Reservoir (TVA 1990a). Important prey species include threadfin shad and gizzard shad. Carp, golden redhorse (*Moxostoma erythrurum*), and spotted sucker (*Minytrema melanops*) are common nongame species (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). The primary commercial species are channel (*I. punctatus*) and blue catfish (*I. furcatus*), smallmouth buffalo, and paddlefish (TVA 1990a). Watts Bar Reservoir is open to commercial fishing of all fish that are not classified by the Tennessee Wildlife Resources Agency (TWRA) as game fish. However, the use of gill or trammel nets is prohibited in Watts Bar Reservoir. Important sport species include largemouth and smallmouth bass (*M. dolomieui*), white crappie (*P. annularis*), white bass (*Morone chrysops*), sauger, striped bass, bluegill (*Lepomis macrochirus*), and striped/white bass hybrids (TVA 1990a). The fishery resources of Watts Bar Reservoir are actively managed by the TVA and the TWRA.

The use of fish from Watts Bar Reservoir for human consumption has been affected by the discovery of PCBs in the fish. In 1987, the Tennessee Department of Health and Environment issued an advisory that warned persons to avoid or limit the consumption of catfish and striped bass collected immediately below Fort Loudoun Dam (Alexander 1987). The advisory has since been expanded to include striped/white bass hybrids, white bass, sauger, carp, smallmouth buffalo, and largemouth bass, and extends throughout the reservoir (TDEC 1992). Similar PCB advisories are in effect in the upstream reservoirs (i.e., Melton Hill, Tellico, and Fort Loudoun). A no-consumption advisory is in effect for catfish from Melton Hill Reservoir and Tellico Reservoir (TDEC 1992). Advisories against the consumption of catfish and largemouth bass are in effect for Fort Loudoun Reservoir, and the commercial harvest of catfish from Fort Loudoun Reservoir has been banned by the TWRA because of PCBs (TDEC 1992).

2.7.2 Terrestrial Resources

The Watts Bar area is primarily rural in nature. Land use is predominantly agriculture and forestry (TVA 1987). Residential developments are found on the outskirts of Harriman, Loudon, Lenoir City (all outside of the study area), and Kingston and along the northwest shore of the reservoir, encompassing about 25% of the total shoreline (TVA 1990a). The southeast shore of the reservoir is primarily agrarian in nature, with Meigs County among the most rural of all Tennessee counties. Residential development and growth on the reservoir is low to medium. Watts Bar Reservoir has the next to lowest amount of industrial development of any TVA

mainstem reservoir (TVA 1990a). Agricultural products of greatest importance are dairy products, livestock, corn, wheat, soybeans, and tobacco. The area forests can be broadly classified as either pine or upland hardwood forests, with pine plantations helping to supply a local demand for pulpwood (TVA 1987).

The wildlife most closely associated with the reservoir system include waterfowl (i.e., ducks and geese—Family *Anatidae*), waterbirds (i.e., loons—Family *Gaviidae*, herons—Family *Ardeidae*, and cormorants—Family *Phalacrocoracidae*), and wetland furbearers (i.e., muskrats—*Ondatara zibethica*, beaver—*Castor canadensis*, mink—*Mustela vison*, and raccoons—*Procyon lotor*) (TVA 1990a). Although most waterfowl in the area are migratory, Watts Bar Reservoir supports a resident Canada goose *(Branta canadensis)* population that is actively managed by TVA and TWRA (TVA 1987). There are also at least five known active nesting colonies of waterbirds in the Watts Bar area that consist primarily of great blue herons *(Ardea herodias)*, although black-crowned night herons *(Nycticorax violaceus)* are also present (TVA 1987). Watts Bar also supports a nesting population of osprey *(Pandion haliaetus)*, listed as endangered by the state of Tennessee, and the federally endangered bald eagle *(Haliaeetus leucocephalus)* winters on Watts Bar (TVA 1987). In addition to the furbearers listed above, upland wildlife species such as whitetail deer *(Odocoileus virginianus)*, eastern cottontail rabbit *(Sylvilagus floridanus)*, northern bobwhite *(Colinus virginianus)*, mourning dove *(Zenaida aurita)*, and various songbirds use the shoreline for cover and food (TVA 1990a). The TWRA has several wildlife management areas in Watts Bar Reservoir. The 300-acre Kingston Wildlife Refuge is located at the confluence of the Emory and Clinch rivers and is upstream of LWBR. The Long Island Unit of TWRA's 5480-acre Watts Bar Reservoir Wildlife Management Area is located on the Tennessee River and is also upstream of LWBR. The Thief Neck Island Unit is located in the LWBR in Roane County. The Paint Rock Wildlife Refuge is located at TRM 574–576, also upstream of LWBR. Threatened and endangered species known to occur in counties bordering LWBR are listed in Table 2.7. The current status of threatened and endangered aquatic species potentially occurring in LWBR is presented in Table 2.8.

Table 2.7. List of threatened and endangered species that have been recorded in Meigs, Rhea, and Roane counties

PLANTS	Status[a,b] FWS	TN	ESD[c]	Preferred habitat
Spreading false-foxglove *Aureolaria patula*	C1	T		Forests
Heavy sedge *Carex gravida*		S		Wet places, terrestrial
Sedge (no common name) *Carex oxylepis pubescens*		S		Woods in low areas
Appalachian bugbane *Cimicifuga rubifolia*	C2	T		Woods
White prairie-clover *Dalea candida*		E		Barrens, glades, open dry woods
Tall larkspur *Delphinium exaltatum*	C2	E		Rich woods
Small's stonecrop *Diamorpha smallii*		E		Vernal pools on flat granite outcrops
Yellow jessamine *Gelsemium sempervirens*		S		Thickets, woods, fencerows, roadsides
Slender blazing-star *Liatris cylindracea*		E		Barrens
Canada lily *Lilium canadense*		T		Wet meadows, bogs, balds
Mountain honeysuckle *Lonicera dioica*		S		Woodlands, thickets
Large-flowered Barbara's buttons *Marshallia grandiflora*	C2	E		Bogs
American ginseng *Panax quinquefolius*		T		Rich woods
Southern Rein-orchid *Platanthera flava flava*		S		Alluvial woods, wet meadows, marshes
Purple fringeless orchid *Platanthera peramoena*		T		Moist woods, meadows, stream banks
Dwarf milkwort *Polygala nana*		S		Open, wet sand
Tennessee leafcup *Polymnia laevigata*		S		Woodlands
Carey saxifrage *Saxifraga careyana*		S		Seepage slopes and moist rocks
Prairie goldenrod *Solidago ptarmicoides*		E		Barrens
Virginia spiraea *Spiraea virginiana*	T	E		Stream banks
Lesser ladies'-tresses *Spiranthes ovalis*	S			Moist shady woods, thickets, swamp edges
Menge's fameflower *Talinum mengesii*	T			Cliffs, rocky banks, granite, sandstone
Roundleaf fameflower *Talinum teretifolium*		T		Rock outcrops

Table 2.7 (continued)

	Status			
	FWS	TN	ESD	Preferred habitat
Earleaf foxglove *Tomanthera auriculata*	C2	E		Barrens
INVERTEBRATES				
Ornate rocksnail *Lithasia geniculata*	C2		T	Rivers
Spiny riversnail *Io fluvialis*	C2		E	River shoals and riffles
Dromedary pearlymussel *Dromus dromas*	E	E	E	River shoals and riffles
Turgid-blossom *Epioblasma turgidula*	E	E	E	Small to medium rivers
Fine-rayed pigtoe *Fusconaia cuneolus*	E	E	E	Rivers
Shiny pigtoe *Fusconaia edgariana*	E	E	E	River and stream riffles
Pink mucket *Lampsilis abrupta*	E	E	E	Rivers
Alabama lamp mussel *Lampsilis virescens*	E	E	E	Small to medium rivers
Orange-foot pimpleback *Plethobasus cooperianus*	E	E	E	River shoals and riffles
Tennessee clubshell *Pleurobema oviforme*	C2		T	Rivers and headwaters
Rough pigtoe *Pleurobema plenum*	E	E	E	Large rivers
Pyramid pigtoe *Pleurobema rubrum*	C2		T	Large rivers
VERTEBRATES[d]				
American eel *Anguilla rostrata*			S	Rivers
Spotfin chub *Cyprinella monacha*	T	E	E	Medium to large clear streams
Flame chub *Hemitremia flammea*		D	S	Spring-fed streams
Snail darter *Percina tanasi*	T	T	E	Large free-flowing rivers
Tangerine darter *Percina aurantiaca*		D	S	Boulders in streams and medium rivers
Tennessee dace *Phoxinus tennesseensis*		D	S	Small woodland tributaries
Green salamander *Aneides aeneus*	C2	D	S	Damp cliffs and rock faces
Hellbender *Cryptobranchus alleganiensis*	C2	D	S	Small clear rivers and large streams

Table 2.7 (continued)

	Status			
	FWS	TN	ESD	Preferred habitat
Cumberland slider *Trachemys scripta troostii*		D	S	Large, quiet waters (coves and sloughs)
Six-lined racerunner *Cnemidophorus sexlineatus*		D	S	Dry, well-drained areas
Green anole *Anolis carolinensis*		D	S	Trees, shrubs, vines
Great blue heron *Ardea herodias*		D	S	Lakes, rivers
Turkey vulture *Cathartes aura*		D	S	Terrestrial habitats
Black vulture *Coragyps atratus*		D	S	Terrestrial habitats
Bald eagle *Haliaeetus leucocephalus*	E	E	E	Shorelines of rivers and lakes
Cooper's hawk *Accipiter cooperii*		T	E	Forests
Sharp-shinned hawk *Accipiter striatus*		T	E	Forests, forest edges
Red-shouldered hawk *Buteo lineatus*		D	S	Forests and fields in river and stream bottoms
Osprey *Pandion haliaetus*		E	E	Rivers and lakes
Bewick's wren *Thryomanes bewickii*		T	T	Farmyards, brushy clearcuts
Bachman's sparrow *Aimophila aestivalis*	C2	E	E	Open, young and mature pine woods
Gray bat *Myotis grisescens*	E	E	E	Caves, rivers, lakes
Eastern small-footed bat *Myotis leibii*	C2	D	S	Caves and various other habitats
Indiana bat *Myotis sodalis*	E	E	E	Woods along small to large streams, caves
Southeastern shrew *Sorex longirostris*		D	S	River floodplains and swamps
Southern bog lemming *Synaptomys cooperi*		D	S	Low, moist areas in grass, swamps, bogs
River otter *Lutra canadensis*		T	T	Lakes, rivers, large streams

[a] FWS = U.S. Fish and Wildlife Service; TN = the state of Tennessee; ESD = Ecological Services Division in the Tennessee Department of Environment and Conservation

[b] C2 = category 2 candidate species, for which the FWS currently does not have sufficient information to support a proposal to list the species as threatened or endangered; D = deemed in need of management by the state of Tennessee; E= endangered; T = threatened; S = species of special concern to the Tennessee ESD.

[c] The ESD status for plants is the same as the Tennessee status.

Source (exclusive of habitat information): the list of species and their status were supplied by W. M. Christie, III, Ecological Services Division, Department of Environment and Conservation, Nashville, Tennessee, personal communication to Roger Kroodsma, ORNL Environmental Sciences Division. 1994.

Table 2.8. Threatened and endangered aquatic biodistribution in eastern Tennessee and current status in the lower Watts Bar Reservoir

Species	Distribution and status
Ornate rocksnail	No records in the Tennessee River system upstream of Chattanooga.
Spiny riversnail	Formerly occurred primarily upstream of the points now occupied by Fort Loudoun and Melton Hill dams. No longer occurs downstream of these dams.
Dromedary pearly mussel	Formerly occurred primarily above Fort Loudoun and Melton Hill dams. No longer occurs below these dams (last record was in 1983 below Watts Bar Dam).
Turgid-blossom	Only scattered records before 1970 in Tennessee. Probably extinct in Tennessee.
Fine-rayed pigtoe	Formerly occurred primarily above Fort Loudoun and Melton Hill dams. No longer occurs below these dams.
Shiny pigtoe	Formerly occurred primarily above Fort Loudoun and Melton Hill dams. No longer occurs in Tennessee below these dams.
Pink mucket	Only scattered records in Tennessee. Currently occurs in the Tennessee River in the downstream vicinity of Fort Loudoun Dam but is not known to occur in the Clinch River below Melton Hill Dam.
Alabama lamp mussel	Only a few records in Tennessee, including the Emory River at Harriman. Now probably extinct in Tennessee.
Orange-foot	Only scattered records in Tennessee. May be present in the pimpleback Tennessee River, including the lower Watts Bar area.
Tennessee clubshell	No longer occurs below Melton Hill and Fort Loudoun dams.
Rough pigtoe	Formerly occurred primarily above Fort Loudoun and Melton Hill dams. No longer occurs below these dams.
Pyramid pigtoe	Currently occurs in the Clinch River in the downstream vicinity of Melton Hill Dam and possibly in the Tennessee River below Watts Bar Dam. Not known to occur in the Tennessee River near Fort Loudoun Dam.
American eel	Scattered records in Tennessee. May be present in dammed river systems where locks are present and permit dispersal.
Spotfin chub	Occurred in few localities in Tennessee. Present in the free-flowing portion of the Emory River.

Table 2.8 (continued)

Species	Distribution and status
Flame chub	Occurs exclusively in spring-fed streams in proximity to their sources. May be present in streams off lower Watts Bar.
Snail darter	Probably formerly occurred in the main channel of the Tennessee River as well as in the formerly free-flowing Little Tennessee River. Not currently present.
Tangerine darter	Many localities in eastern Tennessee. Present in free-flowing portion of Emory River. May be present in streams off lower Watts Bar.
Tennessee dace	Numerous locations in eastern Tennessee. Present in streams off lower Watts Bar.
Hellbender	Present in most of Tennessee and in streams off lower Watts Bar.
Cumberland slider	Present in eastern Tennessee. Probably present in lower Watts Bar.

Sources: Bogan and Parmalee (1983), Eagar and Hatcher (1980)
Personal communication from Roger Kroodsma, ORNL ESD, to Steven A. Ahlstedt, Aquatic Biology Laboratory, Tennessee Valley Authority, Norris, Tennessee.

3. SITE HISTORY AND CURRENT CONDITIONS

The purpose of this section is to characterize the nature and extent of contamination in LWBR. As agreed by the FFA parties through the DQO process, this site characterization uses existing data from a number of sources to reach general site conclusions via a weight-of-evidence approach.

The data used here are all found in publicly available documents and were produced, for the most part, by ORNL, TVA, the state of Tennessee, or a multiagency panel. The data are taken at face value; no review of quality assurance practices was performed, although the data were collected and evaluated under individual quality assurance programs prior to publication. Certain data discussed here were used in the baseline human health risk assessment in Chap. 5. The quality of those data is described more fully in Sect. 5.1.

Section 3.1 briefly describes the sources, nature, and magnitude of contaminant releases from the ORR. Summaries of existing studies and monitoring programs regarding contaminants in LWBR are presented in Sect. 3.2. The weight-of-evidence conclusions regarding contamination in LWBR are presented in Sect. 3.3. Also presented in Sect. 3.2 are summaries of studies from the Clinch River and from other TVA reservoirs. Such data are important in reaching weight-of-evidence conclusions regarding contaminants in LWBR.

3.1 RELEASE HISTORY

This section briefly describes the major releases of hazardous substances to the off-site environment from each of the three main ORR facilities. For the earlier years of operation, very little quantitative information regarding the release of hazardous substances is available for contaminants other than radionuclides, and even these estimates are relatively crude. Later release estimates are much better, and, since 1971, have been published in the annual Environmental Monitoring Report for the ORR.

It should be noted that Sect. 103(a) of CERCLA excludes releases of "federally permitted" hazardous substances (i.e., substances that are legally discharged pursuant to an existing permit issued under any of several environmental laws). Therefore, point-source discharges to surface waters authorized by National Pollutant Discharge Elimination System (NPDES) Permits issued by the state of Tennessee and releases of hazardous substances to air that are in compliance with National Emission Standards for Hazardous Air Pollutants (NESHAP) (40 CFR 61, subpart H) are outside the scope of this investigation. These emissions are monitored for environmental compliance, and the results are summarized in the annual environmental monitoring reports for the ORR.

The ORR also currently releases contaminants from various waste management areas and other contaminated sites. Areas that have been used to manage hazardous waste at any time since 1976 are regulated by RCRA. Releases from RCRA units can only occur if they are permitted, subject to the requirements of the laws above. Any nonpermitted releases are subject to RCRA corrective action requirements.

Areas currently or historically used strictly for the management of radioactive wastes are beyond the scope of outside regulatory authority, and are managed by DOE for compliance with

DOE Order 5820.2A under the authority of the Atomic Energy Act (42 USC 2011, *et seq.*). Releases are not expressly forbidden by DOE Order 5820.2A, but any releases must comply with DOE order 5400.5 (Radiation Protection of the Public and the Environment), and thereby must not result in an effective dose equivalent to any member of the public that exceeds 25 mrem per year. Such releases, therefore, are outside the scope of this investigation. The results of radiological monitoring in the vicinity of the ORR for compliance with this order are presented annually in the environmental monitoring reports for the ORR.

The current or threatened release of hazardous substances from specific sites on the ORR is the focus of current CERCLA source control actions. The quantification of these releases is being accomplished at the source; similarly, remedies will be effected at the source. The focus of the CRRI in LWBR is to characterize ambient concentrations in LWBR environmental media. Because water is so transient in LWBR and is being monitored at the ORR sources, the emphasis for the CRRI is characterization of the media that have accumulated contaminants (sediment and biota), to determine if they pose a direct or indirect threat to human health or the environment. In determining whether such a threat exists, however, the measurement of ambient concentrations in these media inevitably integrates all of the contaminant sources mentioned above for the ORR, as well as any non-ORR sources that contribute to LWBR. No attempt has been made to quantitatively apportion the ambient contaminant concentrations in LWBR to these various sources.

3.1.1 Releases from the Y-12 Plant

The original mission of the Y-12 Plant (Fig. 3.1), completed in 1943, was the electromagnetic separation of ^{235}U from ^{238}U for use in the production of atomic weapons at the Los Alamos National Laboratory. The production processes generated significant quantities of liquid waste, from which economically recoverable amounts of uranium were recycled on-site. However, the remaining wastes were discharged directly to the plant's storm sewer system and EFPC (Griffith 1957 in Bruce et al. 1993). An estimated 27 Ci (88,000 lbs) of uranium was discharged to EFPC, primarily as depleted ^{238}U oxide, by the time the electromagnetic enrichment process ended in 1947 (DOE 1988).

In the mid 1940s, the Y-12 Plant began developing into a facility for the production of enriched uranium weapons components. The associated large scale processing operations (e.g., metal machining, chemical processing, uranium salvage and recovery) again resulted in releases of uranium-containing liquid waste to the storm sewer system and to EFPC. Releases from these operations were greatest from 1959 through 1970, primarily reflecting increased productivity during this period (DOE 1988). Other elements used in the production of weapons components that may have been released as contaminants to EFPC include Th, Li, Be, and Pb. In addition, the industrial-scale machining operations at the Y-12 Plant involved the use of large quantities of cutting oils, machine coolant, degreasers, and solvents, including a number of chlorinated solvents, all of which may have been released in small quantities to EFPC. For many years, cutting oils at Y-12 contained PCBs. Waste PCB oils were collected in storage tanks and also disposed of in the Bear Creek Burial Grounds (Bear Creek OU, Fig. 3.2). These burial grounds are located about 2 miles west of the main plant on the southern slope of Pine Ridge. They were primarily used for the disposal of low-level radioactive waste (primarily uranium), but the disposal of wastes containing organics and metals occurred as well. Runoff and seepage from the burial grounds has contributed contaminants to Bear Creek, a tributary to EFPC (DOE 1992). The use of cutting oils containing PCBs was discontinued in 1976. The manufacture of nuclear weapons components was discontinued at Y-12 in 1992.

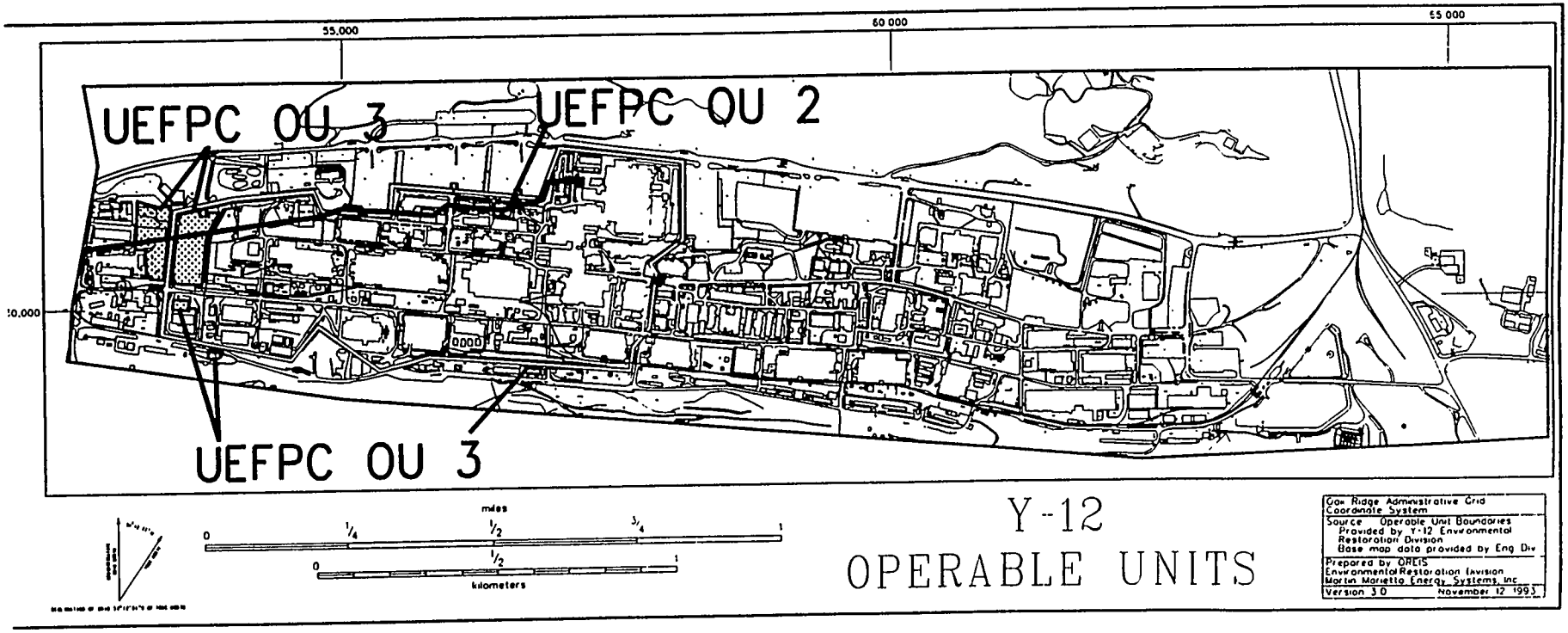


Fig. 3.1. The Y-12 Plant (depicting upper East Fork Poplar Creek environmental restoration operable units).

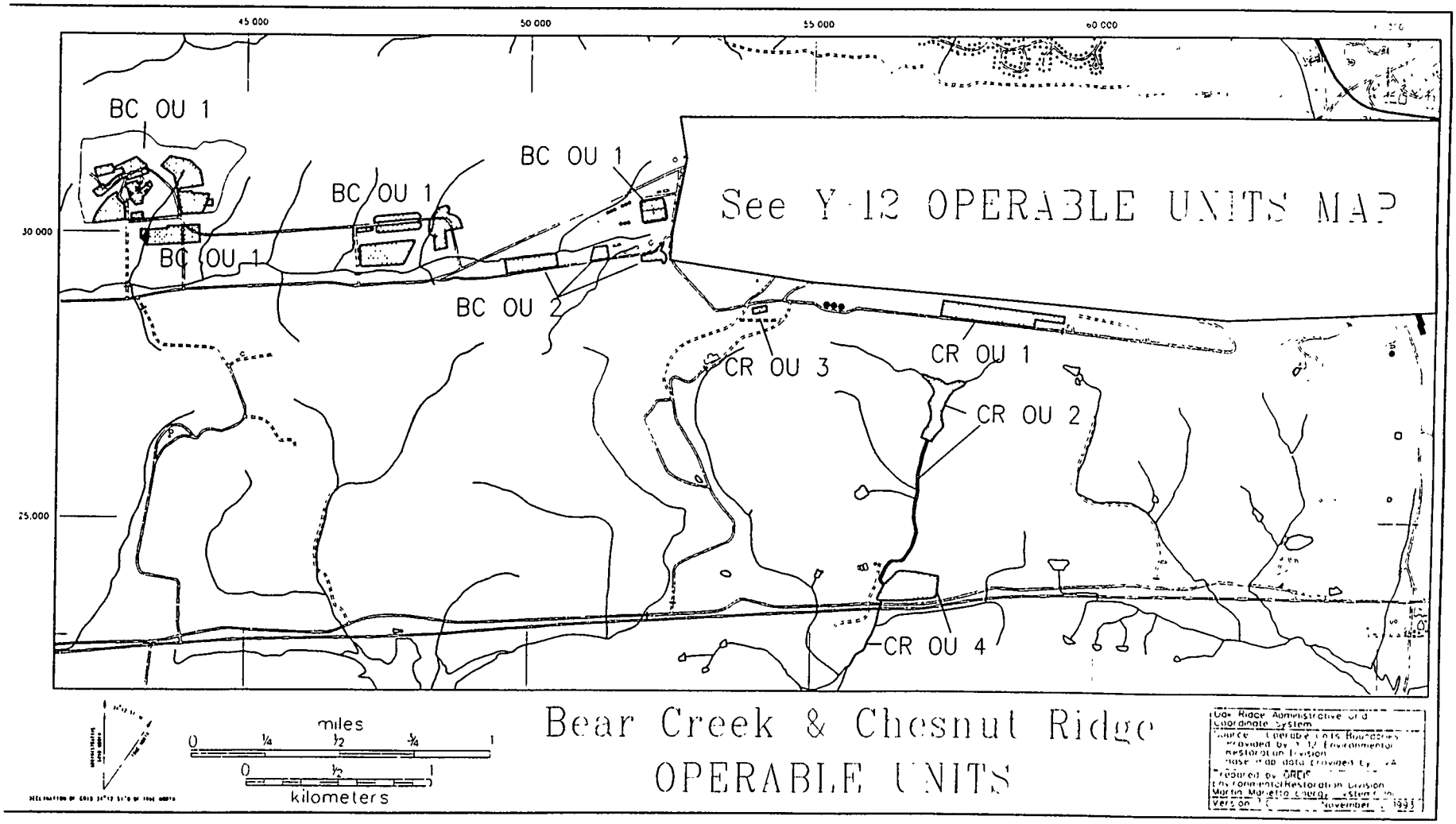


Fig. 3.2. The Y-12 Site, including Bear Creek (BC) and Chestnut Ridge (CR) environmental restoration operable units.

In the early 1950s, the Y-12 Plant began designing, developing, constructing, and operating a process for the large-scale separation of lithium isotopes for use in the production of thermonuclear fusion weapons, or hydrogen bombs. The Colex process was developed, in which ^{6}Li was separated from ^{7}Li via a column exchange process using an aqueous lithium hydroxide solution and a solution of lithium in mercury. Millions of pounds of mercury were required for this process. Large quantities of mercury were released to process buildings, soils, and surface water as a result of leaky pipes and valves, equipment exchange, and accidental spills (Wilcox 1983). By the time the Colex process was closed in 1963, an estimated 239,000 lbs of mercury were lost to EFPC, and another 428,000 lbs were lost to the ground in the plant area (Wilcox 1983). Peak releases of mercury occurred in 1957 and 1958, when approximately 73,000 lbs and 64,000 lbs, respectively, were released to EFPC (Fig. 3.3). Several hundred thousand pounds remained in buildings and process equipment at Y-12.

Mercury releases declined dramatically after 1958, but mercury from the contaminated storm sewer system, buildings, and soils at Y-12 continue to contribute small quantities (up to 25 g/day) of mercury to EFPC. These areas within the Y-12 Plant continue to be the subject of ongoing remedial actions and decontamination and decommissioning activities, while the contamination in EFPC is the focus of a current CERCLA RI/FS (DOE 1992).

In the early 1950s, the Y-12 Plant began processing enriched uranium from the Savannah River Plant and the Idaho Chemical Processing Plant for purification and processing into metal components. The purification process of these enriched uranium streams resulted in the accumulation of trace quantities of transuranic radionuclides (Np, Tc, and Pu) in the resulting waste stream, which was then transferred to the S-3 seepage ponds.

The four S-3 ponds were constructed in 1953 at the west end of the plant. They provided a 10-million-gal storage capacity for liquid wastes containing low levels of radionuclides such as uranium and transuranics. In addition, wastes such as nitric acid, other strong acids, and coolants were disposed of in the ponds. Seepage from the ponds flowed to Bear Creek, a tributary to EFPC. Groundwater in the area beneath the site of the former S-3 ponds is contaminated with nitrates, volatile organic compounds, radionuclides, and to a lesser extent, trace metals (DOE 1992). Discharge to the ponds ceased in 1984 (DOE 1988) and the S-3 ponds underwent RCRA closure in 1988 (DOE 1992).

In addition to the operations discussed above, the Y-12 steam plant and associated operations contribute certain contaminants to the aquatic environment. Runoff from the coal pile formerly contributed suspended solids (coal fines), low pH sulfate, and metallic ions such as arsenic to EFPC. Coal ash from the steam plant was discharged as a slurry to the headwaters of McCoy Branch, a practice discontinued in 1993. McCoy Branch drains into Rogers Quarry, which acted as a settling basin for the ash, and then into the Clinch River at Melton Hill Reservoir (DOE 1992).

3.1.2 Releases from the K-25 Site

Construction of the K-25 Site (Fig. 3.4), formerly the Oak Ridge Gaseous Diffusion Plant (ORGDP), began in 1943. The primary mission of the plant was the enrichment of uranium via the gaseous diffusion process (DOE 1992). This process produces uranium highly enriched in ^{235}U by the diffusion of gaseous uranium hexafluoride across a long series of porous barriers. At each barrier in the diffusion cascade, the product becomes slightly enriched in ^{235}U as a result of its greater diffusion rate across the barrier as compared to the slightly heavier ^{238}U. Before 1964,

ORNL DWG 84-10074

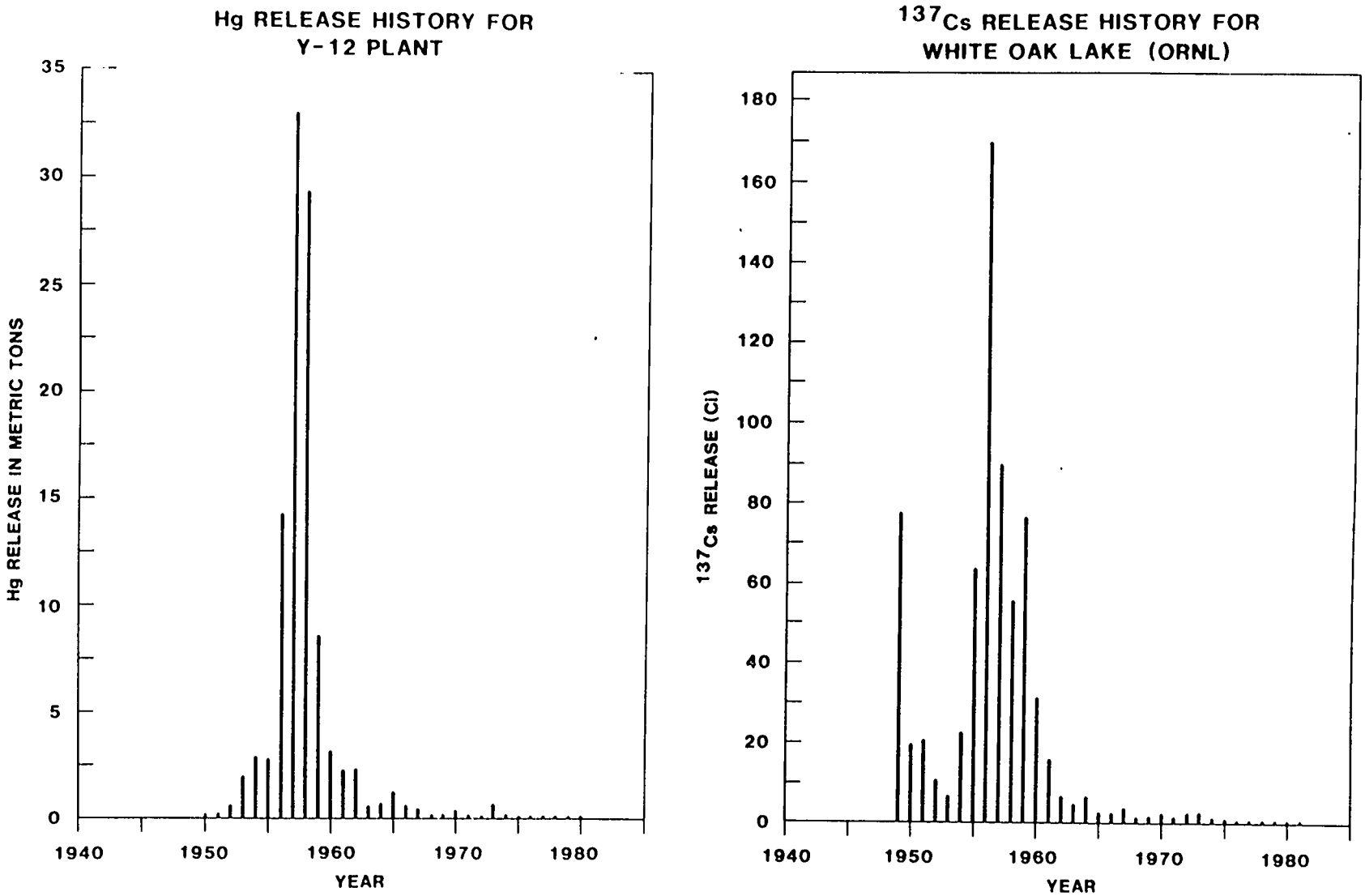


Fig. 3.3. Mercury release history from the Y-12 Plant and ^{137}Cs release history from White Oak Lake (ORNL).

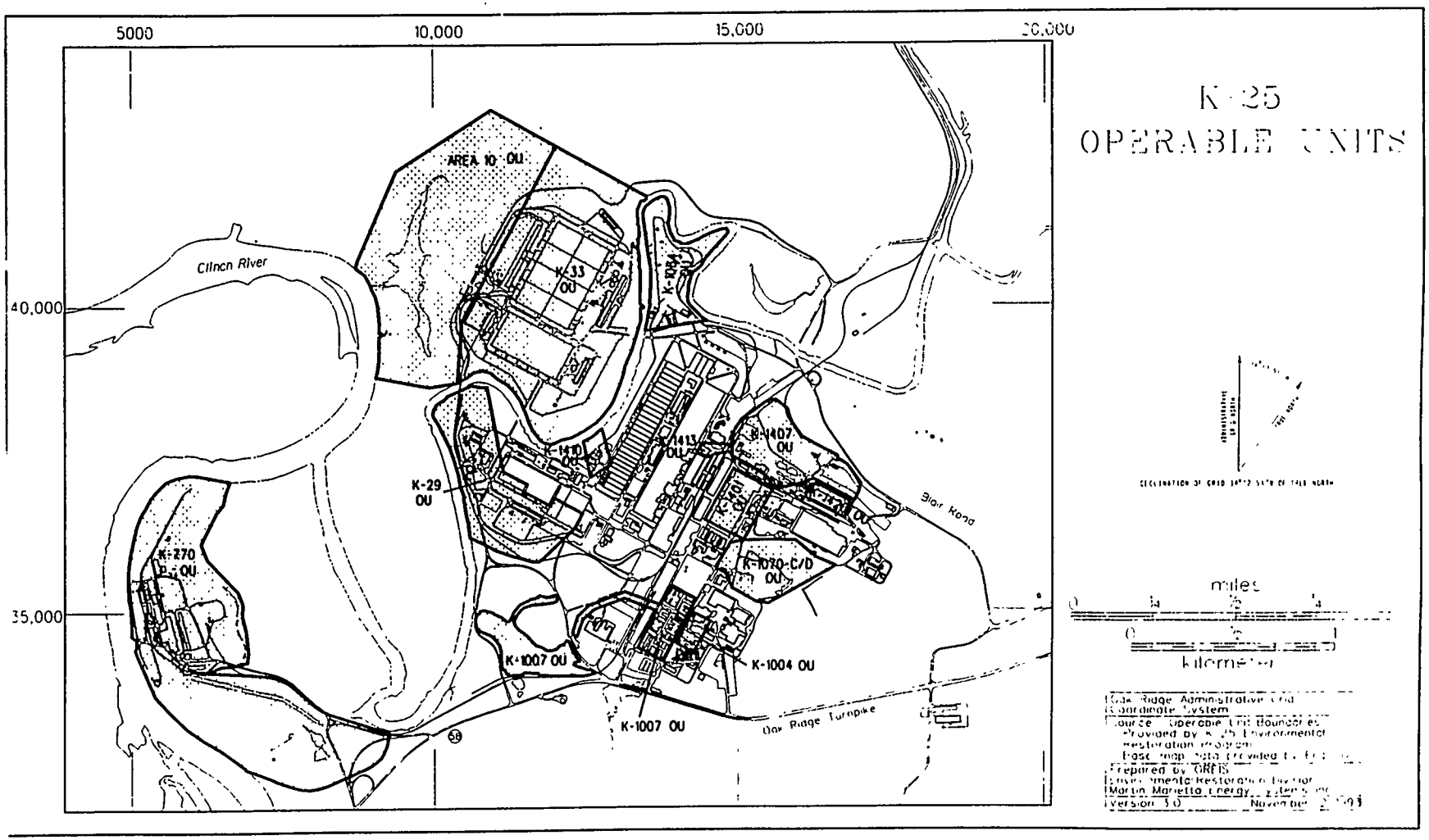


Fig. 3.4. The K-25 Site (depicting environmental restoration operable units).

a long series of diffusion cascades were used to produce highly enriched uranium (96% ^{235}U) for use in nuclear weapons production. After 1964, several of the diffusion cascades were shut down and less enriched uranium (10% ^{235}U) was produced for use as fuel in commercial nuclear power generating facilities. The entire gaseous diffusion process was placed in a "ready standby" mode in August 1985 and permanently shut down in December 1987.

The ORGDP housed numerous support facilities, some of the most important of which were the facilities in building K-1420 for the recovery of uranium from various wastes streams and for the decontamination of equipment (and subsequent recovery of uranium). These facilities are known to have released U, ^{99}Tc, and ^{237}Np as liquid waste (DOE 1988). The transuranic components of this waste were introduced as a result of the recovery of uranium from spent nuclear reactor fuels from the Savannah River and Hanford Plants (Egli et al. 1985 in Bruce et al. 1993).

Building K-1420 also housed a metal cleaning and plating operation. The surface corrosion of steel parts by fluorine gases in the diffusion cascade was prevented by coating these parts with nickel. Prior to plating, parts were cleaned with aqueous solutions of hydrochloric and sulfuric acids, detergents, and chlorinated solvents. Parts were rinsed in tanks after cleaning and plating. The flow from these tanks was piped to the K-1407A neutralization pit for pH adjustment and then discharged to the K-1407B holding pond (DOE 1979).

Located near K-1420, building K-1401 served as the primary maintenance facility for the ORGDP from 1945 through 1985. The facility provided services for the fabrication, cleaning, assembly, and painting of plant equipment. Cleaning involved the use of cleaning baths, which, over the years, used various acids and chlorinated solvents. Fabrication required the use of metals and cutting oils, paint, and solvents. Waste solutions from these activities were piped to the K-1407A pond for neutralization and discharged to the K-1407B holding pond (Goddard et al. 1991).

The K-1407A neutralization pit, operated since the 1940s, received wastewater from the uranium recovery operations, the metal plating operations, and the maintenance facility operations described above. Contaminants in these wastes included uranium, transuranic radionuclides, metals, chlorinated and nonchlorinated solvents, and corrosives. These wastewaters were neutralized with powdered lime or concentrated sulfuric acid and discharged to the K-1407B holding pond. The neutralization of wastewaters in the K-1407A pit was discontinued in 1987 with the opening of the plant's new Central Neutralization Facility (Goddard et al. 1991).

The K-1407B holding pond was a 1.3-acre, 1.5-million-gal, unlined settling basin that received wastes for more than 40 years. The pond received organics and metal hydroxides after neutralization in the K-1407A pit and functioned primarily to settle the precipitates formed by that process. The pond also received effluent directly from several plant buildings. Contaminants known to be discharged to the pond include uranium, transuranic radionuclides, solvents and oils, PCBs, and metals such as Cd, Cr, Pb, and Ni. In 1973, sludge from the pond was removed and disposed of in the K-1407C Retention Basin. A RCRA closure of the K-1407B pond was initiated in 1988. The sludges from the pond were placed in drums to which cement was added to fix the wastes. Although the soil beneath the ponds was free of RCRA waste, remaining radionuclide contamination is the focus of an ongoing CERCLA remedial action. Effluent from the K-1407B pond flowed into Mitchell Branch, a tributary to Poplar Creek (Goddard et al. 1991).

The K-1407C retention basin was a 2.5-million-gal surface impoundment constructed in 1973 to store the sludge from the K-1407B pond (Goddard et al. 1991). It was also subsequently used to store potassium hydroxide scrubber sludge generated at the plant (DOE 1992). Like the K-1407B pond, a RCRA closure was initiated in 1988, but remaining contamination of the underlying soil with metals and radionuclides was found and is currently being addressed by a CERCLA remedial action. The sludges from the K-1407C pond were also collected and fixed in drums as described above. The resulting 46,000 drums of sludge from these pond closures were stored outside at the K-1417 Drum Storage Area, where leaking drums were detected in 1989. Runoff from this area went directly into Mitchell Branch. A portion of these drums have subsequently been moved to indoor waste storage vaults at the K-25 Site (DOE 1992). The drums remaining in the K-1417 drum storage area are currently being repackaged and moved into the indoor storage vaults.

The K-1501 steam plant supplies steam for process purposes and space heating. Liquid effluents consisted primarily of caustic boiler water blowdown and acidic discharge from the treatment of supply water. These discharges were neutralized and then released to the K-1407B pond. The coal storage yard contributed acidic runoff containing the trace metals As, Ni, Cu, and Mn. Before 1985, leachate and runoff from the coal pile were routed directly into Mitchell Branch (DOE 1979). This leachate now goes to the Central Neutralization Facility.

From the start of operations at ORGDP, mechanical draft cooling water towers were used to dissipate heat generated by the gaseous diffusion process. Recirculating cooling water was pumped to the towers, where losses to the atmosphere occurred. A chromium-based corrosion inhibitor was added to this recirculating water until 1977, when it was replaced with a phosphate system (DOE 1979). Raw water from the Clinch River was provided by the K-901 pumphouse and treated at the K-892 clarification facility. Sludge from this process and blowdown from the cooling towers was discharged to the K-901A Holding Pond (Goddard et al. 1991).

The K-901A holding pond received wastes from the late 1950s until 1985 (Goddard et al. 1991). The pond was initially just a marshy area until a dam was constructed in 1965–1966 to create the holding pond proper. In addition to the cooling water system wastes described above, the pond was used to dispose of compressed gas cylinders containing unknown quantities of uranium hexafluoride, hydrogen fluoride, halides, and various fluoridated and chlorinated hydrocarbons (Goddard et al. 1991). The cylinders were reportedly breached prior to placement in the pond (Bruce et al. 1993). The K-901A Pond discharges directly to the Clinch River.

The gaseous diffusion process required large amounts of electricity. Four on-site switchyards received power transmitted from off-site sources. PCB-containing transformers at the switchyards reduced and transformed this electricity for plant use. Releases of PCBs from the switchyards (Goddard et al. 1991) may have resulted in migration of these chemicals to Poplar Creek and the Clinch River.

Another important holding pond, the K-1007B pond, received wastewater from laboratory drains in several plant buildings. An estimated 2200 gal of laboratory wastes were discharged to these drains each year prior to 1985, when the practice was discontinued (Goddard et al. 1991). Laboratory wastes included uranium and other radionuclides, acids, ethers, alcohols, glycols, chlorinated and nonchlorinated solvents, mercury, PCBs, and cadmium (Goddard et al. 1991). The K-1007B pond discharges directly into Poplar Creek.

In addition to the storage ponds and pits described above, a number of land-based waste disposal operations were used to manage wastes at the K-25 Site which could have contributed to contamination of on-site and off-site surface waters via surface runoff or groundwater flow. Burial grounds included K-1070A, K-1070B, and K-1070C & D (DOE 1992). The K-1070 area operated from the late 1940s through 1975 for the disposal of low-level contaminated and mixed chemical wastes, including Th, Pb, and U. The K-1070B area was used from the late 1950s to 1976 to dispose of classified equipment contaminated with Pb, U, Al, Cu, Be, bronze, and asbestos (Goddard et al. 1991). This area was created by filling a low marshy area adjacent to a small creek that flowed into Poplar Creek. The K-1070C & D classified burial ground received wastes from 1972 to 1989. The area received approximately 9100 gal of waste solvents and 1600 lbs of chemicals (Goddard et al. 1991). From 1979 through 1985, an area known as the K-1070A landfarm was used to dispose of spent Fuller's Earth, a product used to remove impurities in the diffusion cascade oil (Goddard et al. 1991). Several other on-site areas were used for the open burning of waste solvents or the incineration of plant waste. Still other areas were used for waste storage, including the storage of radioactively contaminated scrap metals or various liquid wastes in drums. The K-720 fly ash pile, used from the 1940s through the 1960s, received fly ash from the on-site coal fired steam plant for disposal. This site was located south of the plant near the Clinch River, and runoff from the waste pile was not controlled (Goddard et al. 1991).

3.1.3 Releases from Oak Ridge National Laboratory

The original mission of ORNL (Fig. 3.5), formerly known as the Clinton Laboratory and the X-10 Site, was the pilot-scale production of plutonium for use in nuclear weapons research at Los Alamos National Laboratory. Construction of the X-10 Site began in January 1943. The source of the plutonium was to be a nuclear reactor built specifically for this purpose; by late fall of 1943, the Oak Ridge graphite reactor was on-line. The separation of plutonium from the resulting fission products was accomplished in a neighboring chemical separation processing pilot plant. The first shipment of plutonium to Los Alamos occurred in February 1944. By the end of 1944, the use of the graphite reactor shifted from plutonium production to research and the production of other radionuclides. Having completed its original mission, ORNL subsequently became a center for the development and testing of nuclear reactors, for the chemical and physical separation of nuclear materials, and for the production of a wide array of radionuclides for worldwide use in research, medicine, and industry (DOE 1992).

The most significant operations at ORNL that have released contaminants to off-site surface waters have been the management of liquid and solid wastes. The X-10 Site was planned as a temporary pilot facility, and, therefore, waste production was anticipated to be small. A series of concrete gunite tanks were constructed to contain the wastes from the operations. However, the mission of X-10 was almost immediately expanded, and the gunite tanks soon became inadequate for containing the wastes being generated. The materials in the tanks were then treated to precipitate sludges and particle-reactive contaminants. The remaining liquids were then released, along with large quantities of diluting water, to WOC. In 1943, White Oak Dam was constructed across the creek to create a basin for the additional settling of any remaining solids. In June 1944, a holding pond, the 3513 Pond, was built and began receiving discharges from the gunite tanks. The 3513 Pond provided an additional retention basin for the settling of contaminant particles and for the decay of short-lived radionuclides prior to discharge into WOC. From 1949 to 1954, an evaporator was used to concentrate and thereby reduce the volume of liquid wastes stored in the gunite tanks. Waste volumes continued to increase, however, and in 1951, the use of disposal pits for these wastes was initiated. Seven separate disposal pits were in use at various times from 1951 through 1976 (Spalding and Boegly 1985). In October 1955, WOL was drained

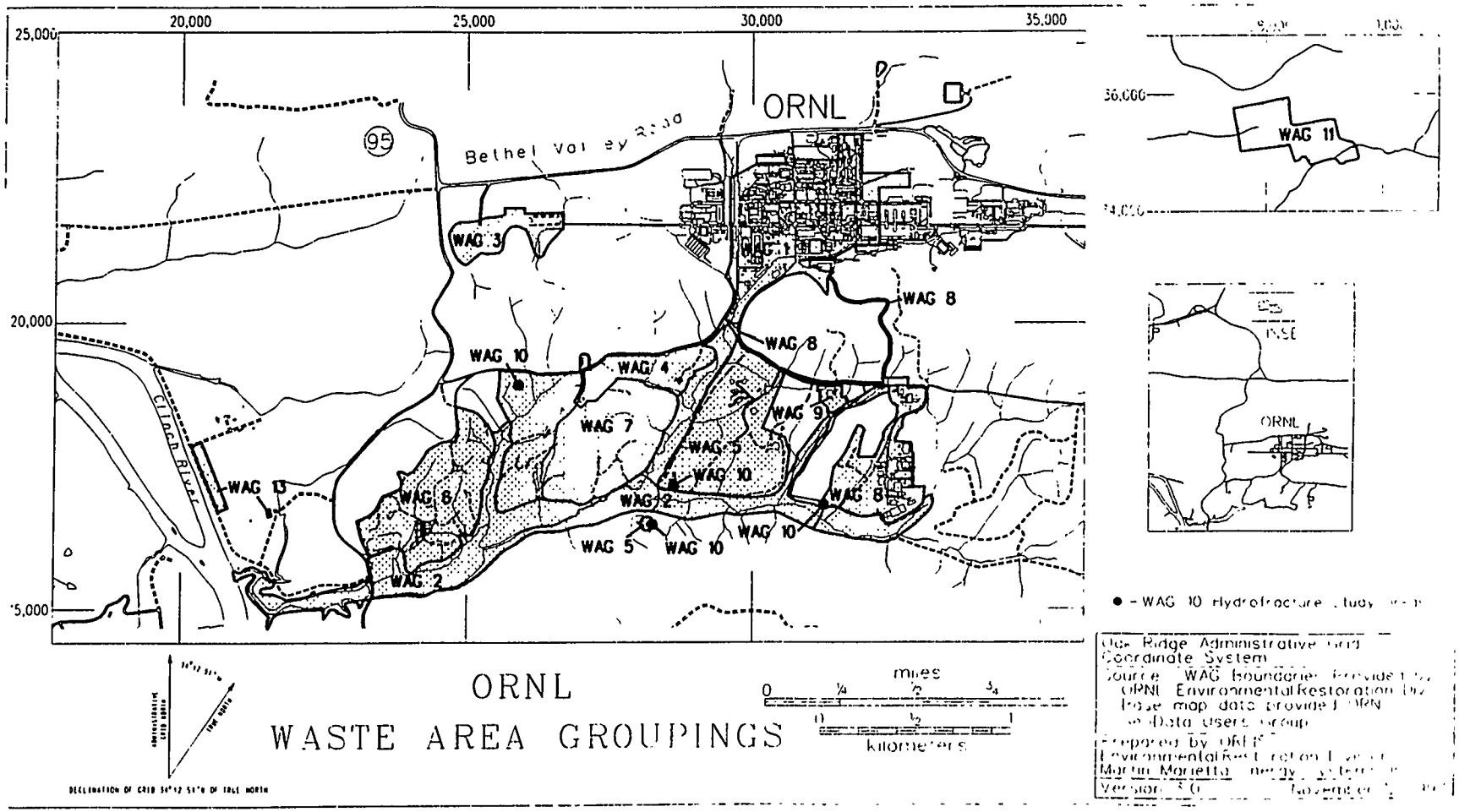


Fig. 3.5. ORNL Site (depicting environmental restoration waste area groupings).

because it had been determined that it was no longer effectively diluting and retaining radioactive materials. Approximately 150,000 ft^3 of contaminated sediment was subsequently eroded from the lake bed (Blaylock et al. 1993) before the impoundment was reestablished in 1960. In 1957, a process waste water treatment plant was built to recover fission products from liquid wastes. The effluent from this facility was only slightly radioactive compared to the low-level wastes that had previously been discharged to the pits. This waste treatment facility was replaced in 1976 with an upgraded facility (DOE 1992).

The first liquid waste disposal pit was used from July to early October 1951. Its use was discontinued when it was discovered that radionuclides, primarily ^{106}Ru, were leaking from the pit. Waste Pit 2 was constructed in 1952 and remained in use until 1962. A pipeline to Pit 2 was constructed in 1954, the evaporator was shut down, and wastes were pumped directly to the waste pit. Pit 3 was in service from 1955 through 1962. Pit 4 was in use from 1956 until 1976, when the new process waste treatment plant went on-line. Large quantities of ^{106}Ru were detected leaking from Pit 4 in 1959, and a trench was constructed down slope to intercept the leachate from Pit 4 and pump it back into the pit. Waste Pits 1–4 were open pits. In 1960, the pit design changed to an earth-covered trench designed to minimize accidental exposure to the contaminants in the trench and to minimize the collection of rainwater. Waste Trench 5 was in service from 1960 until 1964, and Trench 6 went into service in June 1961. In October 1961, significant leakage of ^{137}Cs and ^{90}Sr was detected from Trench 6, and its use was discontinued. Trench 7 was built in 1962 and used until 1966, when disposal of liquid wastes using the hydrofracture technology was begun (Spalding and Boegly 1985).

The hydrofracture process involved using hydraulic pressure to initiate cracks in the layers of shale bedrock underlying the disposal site. Alkaline solutions of low-level waste were then mixed with cement and injected under pressure into the fracture zone, at a depth of approximately 700–1000 ft. This waste/cement grout mixture filled the cracks that had been developed and then hardened in place, immobilizing the wastes in the deep shale formation. The original hydrofracture facility was in use from 1964 through 1979. A second hydrofracture facility was opened in 1982, but its use was discontinued in 1984 when the possible leaching of contaminants to deep groundwater, resulting from the improper fixation of the wastes, was identified as a concern (Ohnesorge 1986).

Solid wastes at the laboratory have been disposed of at six different Solid Waste Storage Areas (SWSAs) (DOE 1992). The first three storage areas were located in Bethel Valley near ORNL. The sites were selected primarily on the basis of convenience and with little regard for the potential mobility of the wastes in the soil. SWSA 1 was a 1-acre site in use from 1943 to 1944, SWSA 2 was a 4-acre site in use from 1944 to 1946, and SWSA 3 was a 6-acre site used from 1946 to 1951. SWSAs 4 through 6 were located in Melton Valley, where the soil types were better suited for immobilization of radionuclides. For several years in the late 1950s and early 1960s, ORNL's SWSAs served as a regional burial ground for low-level wastes generated from a number of other federal facilities and private companies (Bates 1983). SWSA 4 was a 23-acre site used from 1951 to 1959, and SWSA 5 was a 50-acre site in use from 1959 to 1973. SWSA 6 is a 68-acre site that was opened in 1969 and is still in limited use (DOE 1992).

Tritium, ^{106}Ru, ^{90}Sr, and the rare earth elements (excluding cerium) are the radionuclides that have been released in the greatest quantities from ORNL directly to surface waters during the period 1949–1987 (DOE 1988). The radionuclides disposed of in greatest quantities by being buried on-site or discharged as liquid waste to pits and trenches from 1944 to 1987 include ^{137}Cs (1,174,709 Ci), ^{90}Sr (880,557 Ci), and unidentified beta emitters (1,152,686 Ci). However, of

these totals, only 39% of the ^{137}Cs and 17% of the ^{90}Sr were disposed of in pits and trenches; almost 60% of the ^{137}Cs and 78% of the ^{90}Sr were disposed of in the hydrofracture facility. The remaining quantities were discharged as solid waste to the SWSAs (DOE 1988).

As monitored at White Oak Dam, the radionuclides released in the greatest quantities to the Clinch River are ^{3}H, ^{106}Ru, ^{137}Cs, and ^{90}Sr (Table 3.1). Tritium is an isotope of hydrogen with a 13-year half-life. It is readily incorporated into water and moves through the environment accordingly. Given the 18-day retention time of Watts Bar Reservoir, all but the most recent releases of tritium have long since flushed through the reservoir. Peak ^{106}Ru discharges of 1400–2000 Ci/year occurred from 1960 to 1962, corresponding to the period of greatest seepage from Waste Pit 4. Current releases are estimated at zero (DOE 1988). Ruthenium-106 is a water-soluble radionuclide with a half-life of 368 days and would not be expected to be found currently in LWBR. Cesium-137 is a particle-reactive radionuclide, which means that it is readily adsorbed onto the surface of fine-grained sediment particles, particularly clays. Peak releases of ^{137}Cs occurred in 1956 (170 Ci) and coincided with the draining of White Oak Lake, when large quantities of sediment were eroded from the exposed lake bottom. An additional 266 Ci were released through 1961, after which releases continued to decline until, by 1975, annual releases were less than 2 Ci, where they have remained since (Fig. 3.3, Table 3.1). Total ^{137}Cs released to the Clinch River from ORNL is approximately 700 Ci (~8 g). Because ^{137}Cs has a half-life of 30 years, much of the material released from ORNL can still be found in the sediments of LWBR. Peak releases of ^{90}Sr occurred before 1961, but cannot be correlated with disposal data, which is nonexistent for the period before 1961 (DOE 1988). Releases had declined to less than 4 Ci/year by the late 1980s. Although it has a half-life similar to ^{137}Cs, ^{90}Sr is less particle-reactive and, therefore, has not accumulated in LWBR to the same extent (Struxness et al. 1967).

DOE (1988) estimated that 70 to 80% of the radioactive materials currently released to surface waters from ORNL originate as leakage from waste disposal areas. Radionuclides of concern are primarily ^{137}Cs and ^{90}Sr, with tritium and transuranic radionuclides being of somewhat less significance. Another 10% of the releases originate from operating facilities at the laboratory, such as reactors, laboratories, and processing plants. The remaining 10% originates as surface runoff from areas of contaminated soil in the vicinity of operating facilities that have been affected by spills or equipment leaks.

3.1.4 Summary of Significant Releases

Historical operations at the ORR have resulted in the release of hazardous substances, including metals, organics, and radionuclides, to the off-site aquatic environment. Reliable estimates regarding the amount of these various contaminants, particularly during the earlier years of ORR operations, are generally impossible because of a lack of adequate monitoring information. The Y-12 Plant is known to have released large quantities of mercury to EFPC. Large quantities of ^{238}U have been released from both the K-25 Site and the Y-12 Plant. Peak contaminant releases of mercury and uranium occurred before the late 1950s and have declined drastically since. Large quantities of fission products have been released from ORNL, including ^{3}H, ^{106}Ru, ^{137}Cs, and ^{90}Sr. Because of its particle-reactive nature, ^{137}Cs is the principal anthropogenic radionuclide (30-year half-life) expected to be found in LWBR, where it would be associated with fine-particle sediments. The greater solubility of the other radionuclides listed results in their being rapidly diluted in the water column and flushed downstream. Other possible contaminants released from the ORR include various metals (such as As, Pb, Cr, Be, and Ni), various organic compounds (such as PCBs, chlorinated and nonchlorinated hydrocarbons, and various laboratory chemicals), and various other radionuclides.

Table 3.1. Estimated[a] discharges (Ci) of radionuclides from White Oak Creek to the Clinch River, 1949–1992 (DOE 1988, Blaylock et al. 1993, Energy Systems 1992 and 1993, Vol II).

Year	^{137}Cs	^{106}Ru	^{90}Sr	TRE[b]	^{144}Ce	^{95}Zr	^{131}I	^{60}Co	^{3}H	TRU[c]
1949	77	110	150	77	18	180	77	[d]		0.04
1950	19	23	38	30		15	19			0.04
1951	20	18	29	11		5	18			0.08
1952	10	15	72	26	23	19	20			0.03
1953	6	26	130	110	7	8	2			0.08
1954	22	11	140	160	24	14	4			0.07
1955	63	31	93	150	85	5	7	7		0.25
1956	170	29	100	140	59	12	4	46		0.28
1957	89	60	83	110	13	23	1	5		0.15
1958	55	42	150	240	30	6	8	9		0.08
1959	76	520	60	94	48	27	1	77		0.68
1960	31	1900	28	48	27	38	5	72		0.19
1961	15	2000	22	24	4	20	4	31		0.07
1962	6	1400	9	11	1	2	0.4	14		0.06
1963	4	430	8	9	2	0.3	0.4	14		0.17
1964	6	190	7	13	0.3	0.2	0.3	15	1900	0.08
1965	2	69	3	6	0.1	0.3	0.2	12	1200	0.50
1966	2	29	3	5	0.1	0.7	0.2	7	3100	0.16
1967	3	17	5	9	0.2	0.5	0.9	3	13300	1.03
1968	1	5	3	4	0.03	0.3	0.3	1	9700	0.04
1969	1	2	3	5	0.02	0.2	0.5	1	12200	0.20
1970	2	1	4	5	0.06	0.02	0.3	1	9500	0.40
1971	1	0.5	3	3	0.05	0.01	0.2	1	8900	0.05
1972	2	0.5	6	5	0.03	0.01	0.3	1	10600	0.07
1973	2	0.7	7		0.02	0.05	0.5	1	15000	0.08
1974	1	0.2	6		0.02	0.02	0.2	0.6	8600	0.02
1975	0.6	0.3	7				0.3	0.5	11000	0.02
1976	0.2	0.2	5				0.03	0.9	7400	0.01
1977	0.2	0.2	3				0.03	0.4	6200	0.03
1978	0.3	0.2	2				0.04	0.4	6300	0.03
1979	0.2	0.1	2.4				0.04	0.4	7700	0.03
1980	0.6	0	1.5				0.04	0.4	4600	0.04
1981	0.2	0.1	1.5				0.04	0.7	2900	0.04
1982	1.5	0.2	2.7				0.06	1.0	5400	0.03
1983	1.2	0.2	2.1				0.004	0.3	5600	0.05
1984	0.6	0.2	2.6				0.05	0.2	6400	0.03
1985	0.4	0.007	3.0					0.6	3700	0.008
1986	1.0	0	1.8					0.54	2600	0.024
1987	0.6	0	1.2					0.12	2500	0.006
1988	0.4	0	1.1					<0.07	1700	
1989	1.2	0	2.9					0.13	4100	
1990	1.1	0	3.1					0.12	3100	
1991	1.7		2.7					0.12	2100	
1992	0.6		2.1					0.04	1900	
Total	699.6	6931.6	1209.7	1295	341.93	376.61	175.33	325.42	179200	5.248

[a] All digits carried through to avoid rounding errors. Only first two are significant.
[b] Total Rare Earth Elements, exclusive of cerium.
[c] Transuranic radionuclides.
[d] Blank cells indicate no data reported.

3.2 SUMMARY OF EXISTING DATA

The purpose of this section is to present the existing data regarding contaminants in LWBR. Many of the contaminants identified in Sect. 3.1 are not specific to the ORR (e.g., PCBs, metals, and organics) and are contributed to LWBR in unknown quantities by multiple sources. Partly for this reason, many of the earlier studies of ORR contamination of the Clinch and Tennessee rivers focused on radioactive contaminants. Because the Clinch River is the source of all ORR surface-water borne contamination to LWBR, studies of contaminants in the Clinch River are also summarized here, in that they are important for understanding the contaminant issues in LWBR.

This section presents a short description of studies of contaminants in LWBR and the Clinch River. Certain data from other TVA reservoirs were provided for reference. The studies are discussed chronologically, concluding with the CRRI and contemporaneous investigations. The purpose, the sampling design, a summary of pertinent data, and any relevant conclusions are briefly discussed for each study.

3.2.1 Annual Surveys of Gamma Activity in Clinch and Tennessee River Bottom Sediments

Annual surveys of bottom sediment contamination were conducted in the Clinch and Tennessee rivers from 1951–1966 (Garner and Kochitzky 1956, Cottrell 1960, ORNL 1961–1967). The total gamma radiation at the surface of the sediment was measured with a submersible Geiger-Muller counter, called a "flounder." Surveys were made of the Clinch River from CRM 27.5 to its mouth. Surveys in the Tennessee River generally extended from TRM 570.8, above the confluence with the Clinch River, to TRM 475.1 in Chickamauga Reservoir near Chattanooga. The survey extended to TRM 354.4 in Guntersville Reservoir in 1951, 1957, and 1958; through Kentucky Reservoir in 1966; and to the mouth of the Tennessee River in 1952.

The study design involved establishing transects across the rivers at selected locations and taking gamma readings at points along the transect. Generally, transects were established at 10-mile intervals in the Tennessee River and at 2-mile intervals in the Clinch River. Readings (gamma counts per second) were averaged across the transect. Beginning in 1954, composite surface sediment grab samples were also collected along each transect for radionuclide analysis (Cottrell 1960).

In the Clinch River, gamma count rates gradually increased downstream from the mouth of WOC. The highest count rates found throughout the entire Clinch/Tennessee River system occurred in the Clinch River from CRM 11.0 to 8.0, after which the rates remained relatively constant downstream to the mouth of the river, except for low counts at scour points at CRM 4.7 and 2.6 (Cottrell 1960). The level of activity dropped off markedly upon entering the Tennessee River and continued to decline downstream (Fig. 3.6), which is attributed to the dilution effect of the Tennessee River and the subsequent deposition of sediment over a much larger area than in the Clinch. The distribution of activity along a transect was generally proportional to the depth of the water (Fig. 3.7), and most activity was found in the main channel. The principal radionuclides were ^{137}Cs, ^{60}Co, ^{144}Ce, total trivalent rare earths, and ^{90}Sr (Fig. 3.8). A large increase in activity in 1956 was due, in large part, to the draining of White Oak Lake with the attendant scouring of contaminated sediment from the lake bottom. An increase in 1952 was attributed to a release of an unspecified short-lived radionuclide, possibly barium, which had largely decayed by 1953 (Garner and Kochitzky 1956).

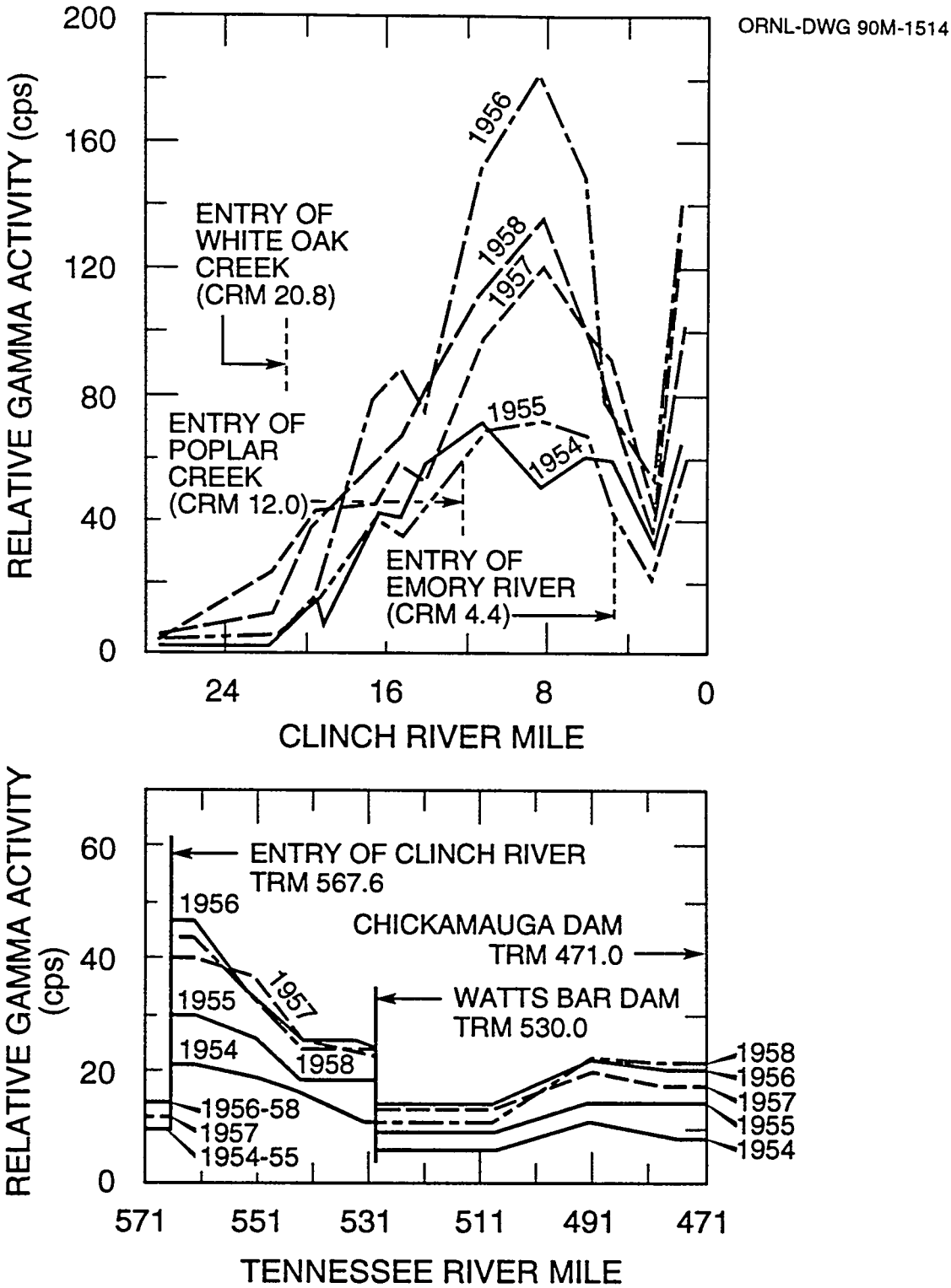


Fig. 3.6. Gamma count at the surface of sediment from the Clinch and Tennessee rivers in Watts Bar Reservoir (Cottrell 1960).

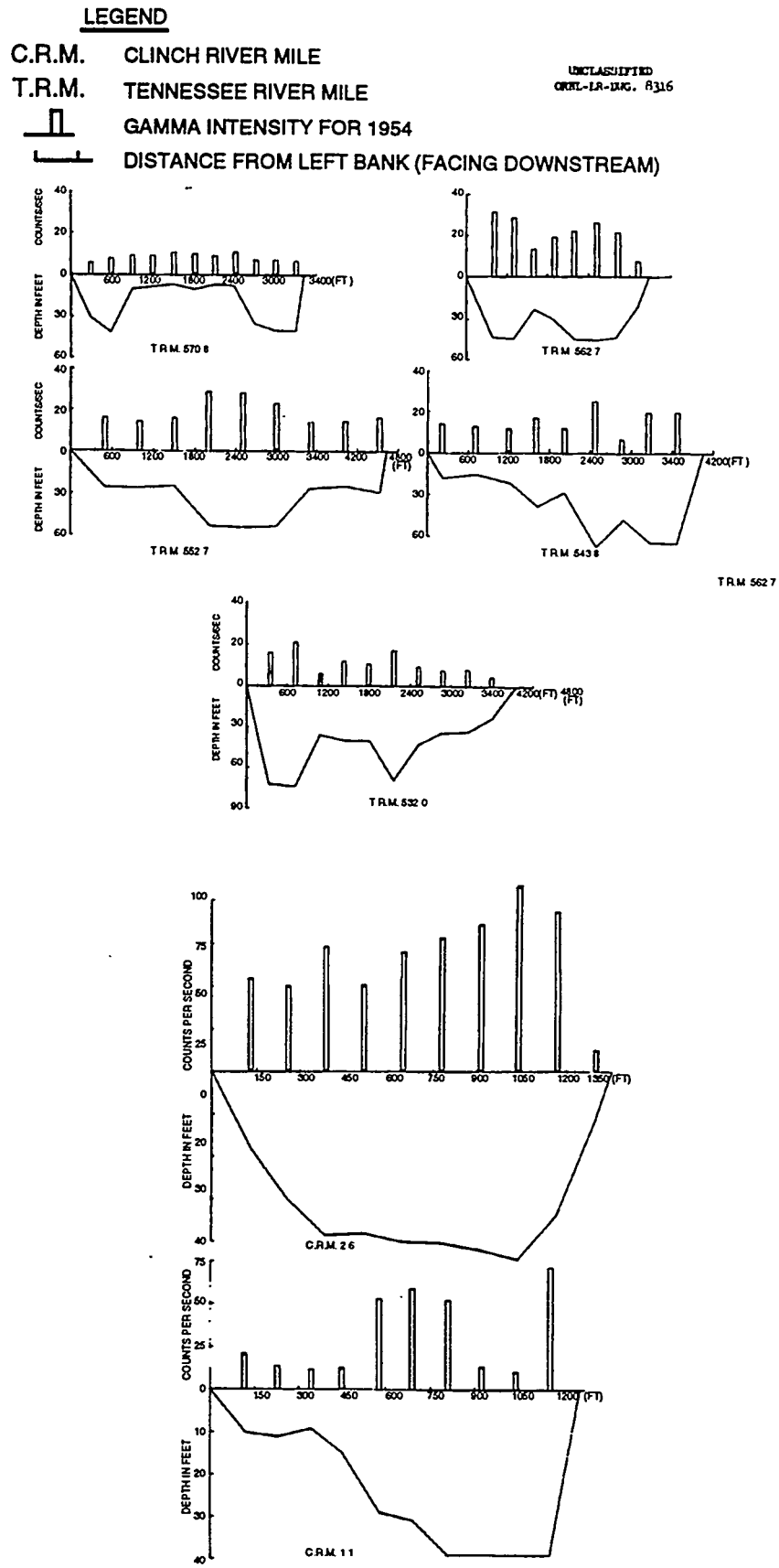


Fig. 3.7. Activity profile vs bottom contour in selected areas of Watts Bar Reservoir (Cottrell 1960).

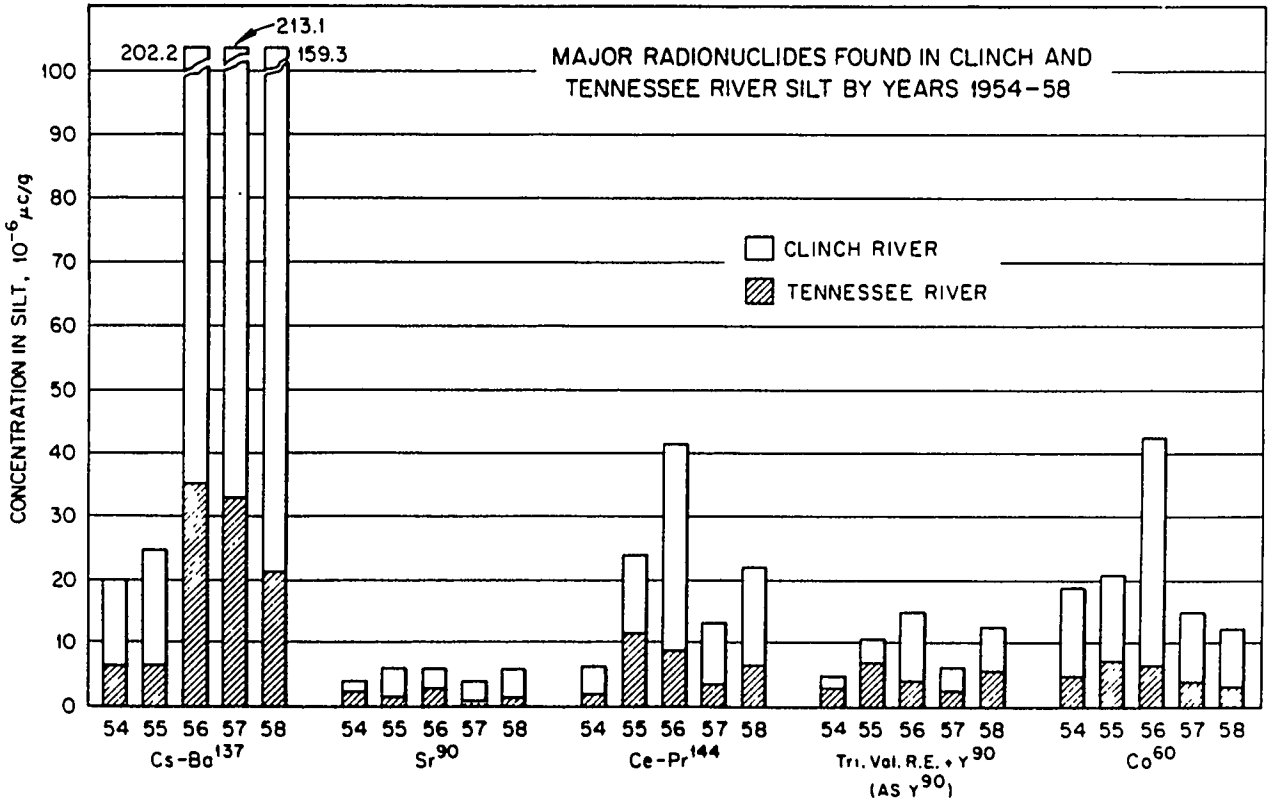


Fig. 3.8. Major radionuclides found in silt from the Clinch and Tennessee rivers by years 1954–58 (Cottrell 1960).

3.2.2 Clinch River Study

The comprehensive Clinch River Study was a multi-agency, interdisciplinary, 5-year (1960–1964) investigation into the effects of radionuclide releases from ORNL into the Clinch River. Organizations participating in the study were the U.S. Atomic Energy Commission, the U.S. Geological Survey, the U.S. Public Health Service, TVA, the Tennessee Department of Public Health, the Tennessee Stream Pollution Control Board, the Tennessee Game and Fish Commission, and ORNL. The objectives of the study were to (1) determine the fate of radioactive materials that were currently being discharged to the river, (2) develop an understanding of the mechanisms of dispersion of radionuclides released to the river, (3) evaluate the limitations of the river for receiving radioactive effluents safely, and (4) suggest long-term monitoring procedures (Struxness et al. 1967).

The radionuclides of interest in the study (determined *a priori* based on quantities released, radioactive half-lives, and recommended maximum permissible concentrations in water) were ^{90}Sr, ^{60}Co, ^{106}Ru, and ^{137}Cs. A mass balance approach was implemented to inventory the radionuclides entering, leaving, remaining, or decaying in the river (Struxness et al. 1967).

The results of the study indicated that ^{106}Ru, ^{90}Sr, and ^{60}Co occurred principally in solution in WOC water. Upon release to the Clinch, the concentrations of these radionuclides were rapidly diluted, and almost the entire quantity was transported downstream, in the dissolved state, at least as far as Chattanooga (Churchill et al. 1965). (TRM 471 at Chickamauga Dam near Chattanooga was the sampling location located the furthest downstream). Figure 3.9 depicts the calculated mass balance curve for ^{90}Sr. Maximum total concentrations of ^{90}Sr in water during the study period ranged from 17,450 pCi/L at White Oak Dam to 14.1 pCi/L at Chickamauga Dam. The maximum concentration recorded for the Clinch River was 42.6 pCi/L at the Centers Ferry station (CRM 5.5). Flow-weighted mean concentrations for the study period were an order of magnitude less than maximum values at all stations. Average concentrations over time could not be calculated given the basic data (Churchill et al. 1965). In contrast to the other radionuclides studied, approximately 70% of the ^{137}Cs in water of WOC was sorbed to suspended sediments. Upon release to the Clinch, these suspended sediments were largely transported downstream to a point (around CRM 15.0) where decreased turbulence allowed significant sedimentation and concomitant accumulation of ^{137}Cs (and, to a lesser extent, the other radionuclides). Figure 3.10 depicts the calculated mass balance curve for ^{137}Cs. Maximum total concentrations of ^{137}Cs in water during the study period ranged from 6,409 pCi/L at White Oak Dam to 6.0 pCi/L at Chickamauga Dam. Flow-weighted means were not calculated because the maximum values were all substantially below the maximum permissible concentration standards in effect at that time (Churchill et al. 1965).

Concentrations of radionuclides in surface sediments decreased in the downstream direction in the Tennessee River. The effect was determined to be greater than expected due to simple volumetric dilution downstream, but less than would be expected from dilution by uncontaminated sediment (Carrigan et al. 1967). Radionuclide concentrations also decreased laterally across the river channels. Radiation levels in sloughs of the Clinch River were not as high as those in sediments of the main channel, and radionuclide levels in areas exposed during winter drawdown were less than in areas that were continuously inundated.

Carrigan and Pickering (1967) found that peak concentrations of gamma activity (almost entirely from ^{137}Cs) in core samples were found to occur at depth; the greater the sediment accumulation rate in an area, the greater the depth of the ^{137}Cs peak in the core (Fig 3.11). The ^{137}Cs profile in a core was found to correlate with the release history from ORNL (Fig. 3.12).

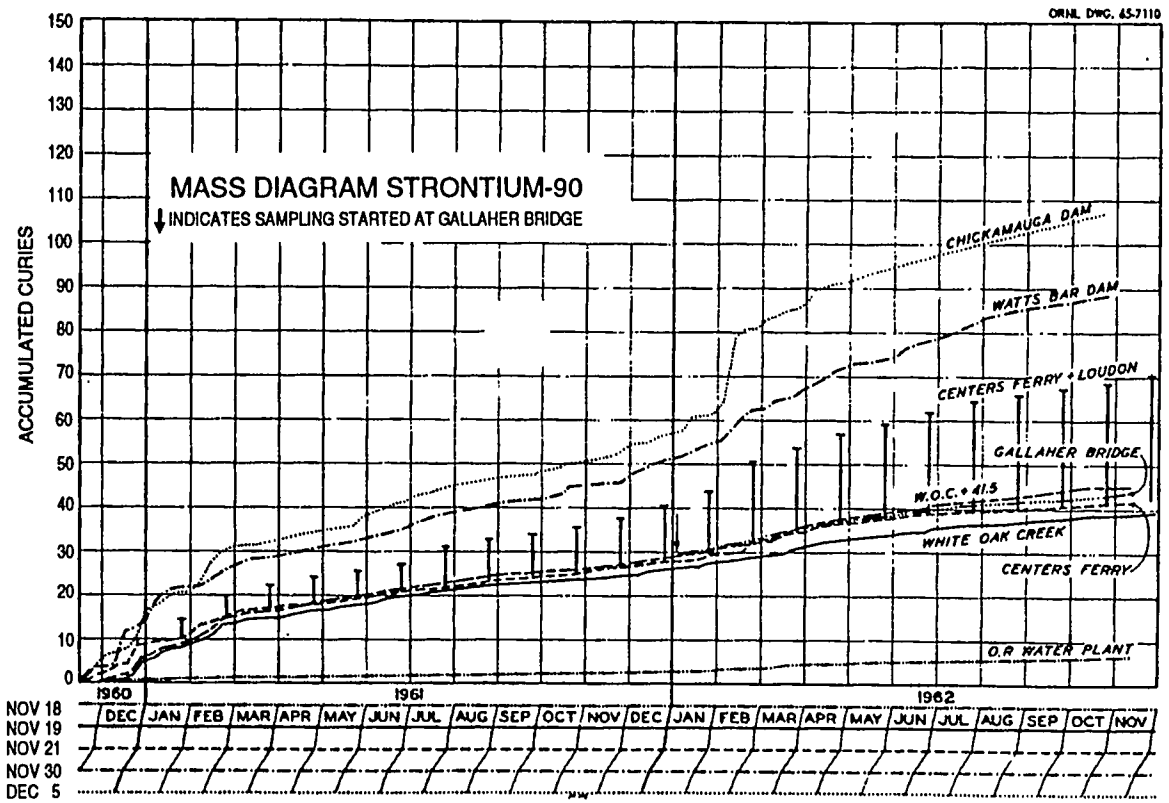


Fig. 3.9. Mass balance diagram for ^{90}Sr in surface water, Clinch River study (Churchill 1965). At locations downstream of the ORR, accumulated Curies exceed that released from the ORR, indicating no significant loss of contaminant mass released from the ORR. The increase in accumulated Curies seen at Watts Bar Dam and Chickamauga Dam most likely represents watershed inputs of atmospheric fallout from above-ground nuclear-weapons tests.

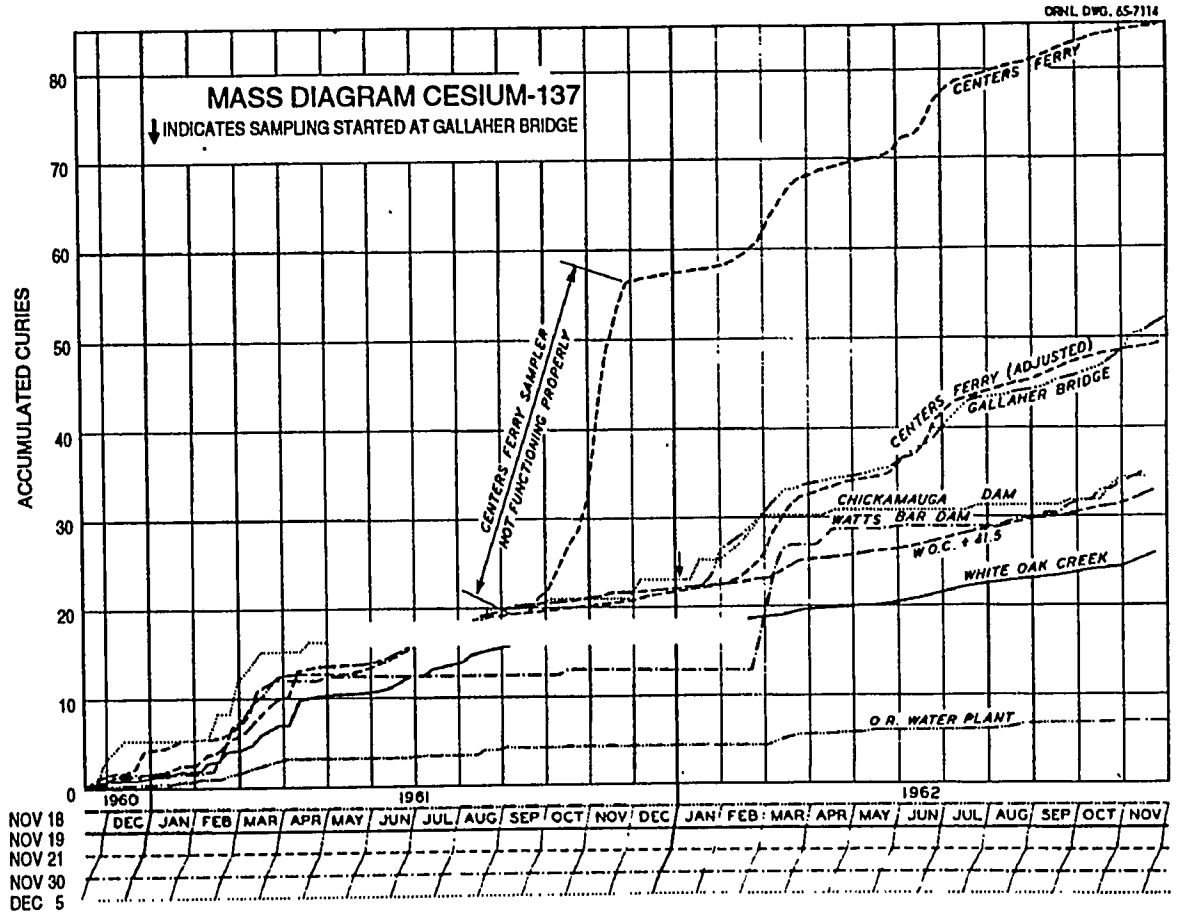


Fig. 3.10. Mass balance diagram for ^{137}Cs, Clinch River study (Churchill 1965). At the Chickamauga and Watts Bar dam sampling stations, total accumulated Curies are only slightly greater than the total released from White Oak Creek and less than the total transported by the Clinch River, indicating a loss of activity (i.e., an area of ^{137}Cs accumulation) between Watts Bar Dam and the Centers Ferry and Gallaher Bridge stations in the Clinch River.

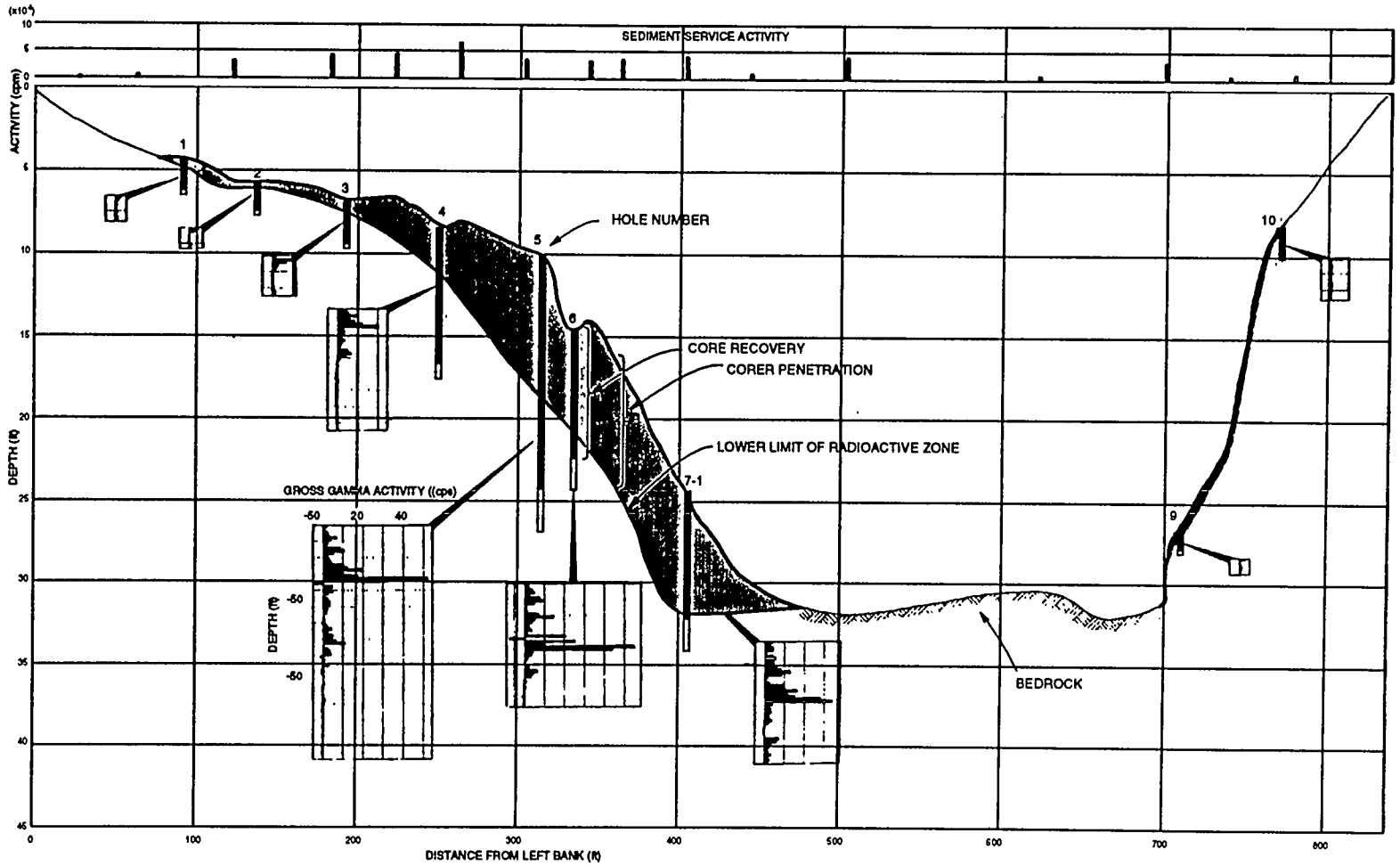


Fig. 3.11. Section at CRM 7.5 showing penetration, recovery, and gross gamma radioactivity variation with depth for 1962 bottom sediment core samples (vertical exaggeration 10:1) (Carrigan and Pickering 1967).

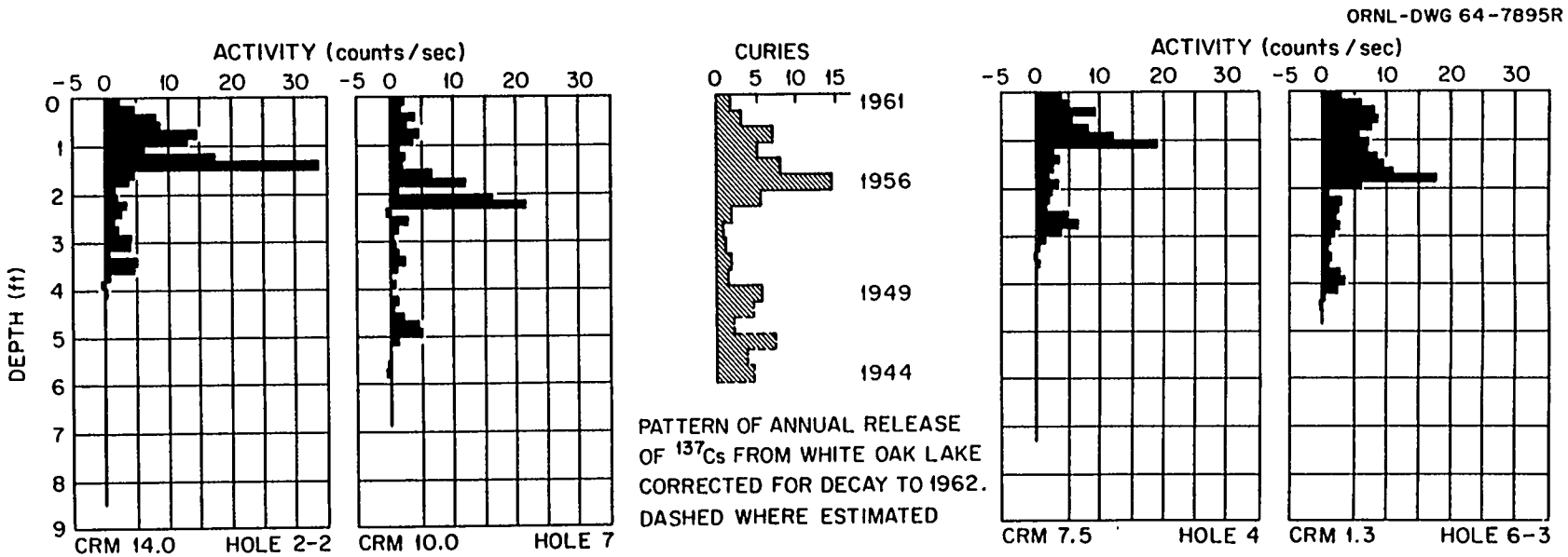


Fig. 3.12. Comparison of patterns of variation with depth of gross gamma radioactivity in four bottom sediment cores to variations in annual releases of ^{137}Cs to Clinch River (Carrigan and Pickering 1967).

The Clinch River was estimated to contain a total radionuclide inventory of around 200 Ci, or only 1.5% of the total activity released before that time (Carrigan and Pickering 1967). More than 95% of this activity was located downstream of CRM 15.0. Approximately 21% of the ^{137}Cs, 9% of the ^{60}Co, 0.4% of the ^{106}Ru, and 0.2% of the ^{90}Sr released from the laboratory were retained in the sediments (Carrigan and Pickering 1967).

The potential for desorption of radionuclides was investigated using contaminated sediments from WOC (Morton 1965). Only ^{90}Sr was found to be held by simple ion exchange and easily removed by circumneutral salt solutions. Strongly acidic solutions desorbed 80–90% of the ^{90}Sr and 65–80% of the ^{60}Co, while strongly alkaline solutions were more effective in desorbing ^{106}Ru. Regardless of treatment, less than 6% of the ^{137}Cs was desorbed, indicating a strong affinity for the sediments. This affinity is greatest for certain clay minerals.

Nelson described the accumulation of radionuclides in biota, including fish (Nelson 1967) and clams (Nelson 1962). The presence of ^{90}Sr in fish (particularly bone) represented one of the significant pathways of potential human exposure identified by Cowser and Snyder (1966). The presence of ^{90}Sr in clam shells can be detected and used to estimate historical water concentrations of ^{90}Sr as much as 500 miles downstream of the ORR (Nelson 1962). From a mass balance perspective, however, only negligible quantities of radiation were found to accumulate in biota.

The Clinch River Study used a mass-balance approach to investigate the environmental fate of radionuclides downstream of the ORR. Although radionuclide accumulation in sedimennt and biota was documented, from a mass-balance perspective, the Clinch River was likened to a pipeline, with little retention of radioactive contaminants in either the bottom sediments or the biota (Struxness et al. 1967).

3.2.3 1977 Clinch River Sediment Survey

Sediments in the vicinity of the proposed Clinch River Breeder Reactor site (CRM 20.8) were sampled in 1977 to determine the fate and distribution of radionuclides in the Clinch River (Oakes et al. 1982). Specific objectives were to analyze selected cores for transuranic radionuclide activity (previous studies had not reported alpha emitters) and to examine the effect of the altered flow regime, as a result of Melton Hill Dam, on the distribution of the fission products in the sediment. A total of 250 cores were collected along the length of the Clinch River, from the mouth of WOC to the confluence with the Tennessee River; cores were also collected from the Tennessee River above and below the mouth of the Clinch River. An attempt was made to purposely sample areas of elevated gamma activity by first identifying those areas through use of the "flounder." However, subsequent analysis of the cores revealed that the flounder had been more strongly influenced by background radiation than by radionuclides discharged from ORNL.

The investigators attempted to duplicate the methodologies used in the Clinch River Study for sample core analysis (automated scanning of the core profile to locate areas within the core emitting high levels of gamma rays). However, the approach failed, because the core profiles were now different from those collected in the early 1960s. The decay of high specific-activity radionuclide ^{106}Ru since that time resulted in gamma levels that were too low to make the approach successful. As an alternative, a cork-borer was used to extract 1-in. diameter cores from the sample core at intervals (of 1, 3, 5, 7, 9, 11, 13, 16, 20, 24, 30, 36, 40, and 45 in.) from the top of the core for spectroscopic analysis of ^{137}Cs and ^{60}Co. Radiochemical analysis for ^{90}Sr and transuranic radionuclides were made of subsamples taken from the top, middle, and bottom of about half of the cores (Oakes et al. 1982).

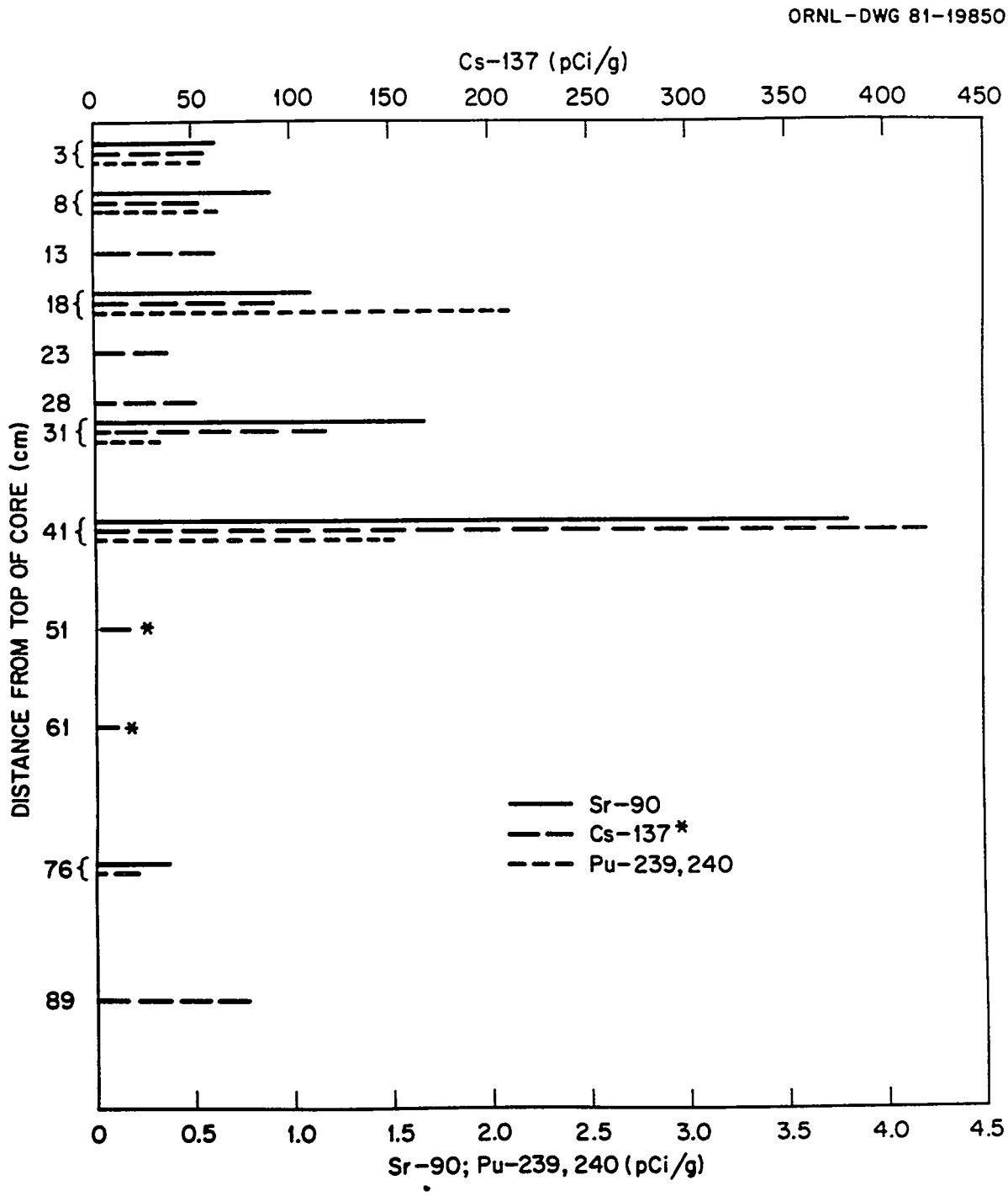


Fig. 3.13. Profile of 1977 radionuclide concentrations as a function of depth within the bottom sediment from a core taken at CRM 11.5, 30 ft from the left bank (Oakes et al. 1982).

The report of Oakes et al. (1982) is a data summary report, and as such, offers only limited discussion and interpretation of the data. As in the Clinch River Study, however, peak concentrations of radionuclides were found to occur at varying depths within cores (e.g., Fig. 3.13). Peak concentrations recorded in the study are presented in Table 3.2.

3.2.4 Metals and PCBs in TVA Reservoir Sediments

In 1982, TVA collected samples of surface sediments from selected reservoirs to be analyzed for trace metals and PCBs (TVA 1986a). Comparisons were made to previously unpublished data collected in 1973. The forebays of the nine mainstem Tennessee River reservoirs were sampled during both studies; in 1973, samples were also collected from a number of tributary reservoirs. An Ekmann dredge was used to collect the 1973 samples (the total number of samples per forebay was not specified). In 1982, a 2-in. diameter steel gravity coring device was used to collect approximately nine sediment cores from each forebay; the upper 3 cm from each core were composited for analysis. Results for those reservoirs sampled in 1982 are presented in Table 3.3. PCBs were not detected in any reservoir in 1982 (no PCB analyses were performed in 1973). Watts Bar Reservoir surface sediments had the highest concentrations of arsenic (16 μg/g) of any reservoir sampled. However, surface sediment concentrations of Be, Cu, Ni,

Table 3.2. Maximum concentrations of radionuclides reported in the 1977 Clinch River Sediment Survey (Oakes et al. 1982)

Radionuclide	Maximum concentration[a]	Location	Depth in core	Maximum Tennessee River concentration	Location	Depth in core
^{137}Cs	606 pCi/g	CRM 14.25	28 cm	47.9 pCi/g	TRM 565.5	23 cm
^{60}Co	12.2 pCi/g	CRM 14.25	28 cm	0.8 pCi/g	TRM 566.8/ 556.8	3cm/ 8cm
^{90}Sr	4.8 pCi/g	CRM 14.25	31 cm	2.12 pCi/g	TRM 566.8	13 cm
$^{238,239}Pu$	6.5 pCi/g	CRM 14.0	23 cm	0.3 pCi/g	TRM 565.8	23 cm

[a]Maximum concentration reported in the survey for Clinch and Tennessee rivers.

Pb, Zn, and Cr were lower in Watts Bar Reservoir than in either Fort Loudoun Reservoir (immediately upstream) or Chickamauga Reservoir (immediately downstream). It was noted that these reservoirs were subject to urban runoff from Knoxville and Chattanooga, respectively. Mercury was lower in Watts Bar Reservoir sediments than in three other TVA mainstem reservoirs (Chickamauga, Guntersville, and Pickwick) and only slightly higher than in Nickajack Reservoir. The source of much of the mercury in Chickamauga was a chloralkali plant on the Hiwassee River (Turner et al. 1984). A similar plant discharged large quantities of mercury into Pickwick Reservoir from 1955 to 1970 (Meinert and Fehring 1992). The mercury levels in Guntersville and Nickajack reservoirs were suggested to result from industrial activity in the Chattanooga area (TVA 1986a). A comparison of surface sediment concentrations of selected metals among reservoirs for the 1973 and 1982 studies is illustrated in Fig. 3.14.

3.2.5 Environmental Fate of Mercury and ^{137}Cs Discharged from ORR Facilities

Turner et al. (1984) collected sediment cores from the Clinch and Tennessee Rivers in 1983 to independently confirm the release histories of mercury from the ORR facilities and to

Table 3.3. Total metal concentrations in surface sediments from selected TVA reservoirs, 1982 (TVA 1986a)

		Total Metal Concentrations, $\mu g/g$									
Reservoir	Site	As	Be	Cr	Cu	Hg	Mn	Ni	Pb	Se	Zn
Kentucky	TRM 24.0	8	1.8	24	22	0.22	1500	21	35	0.2	110
Pickwick	TRM 207.6	6	<1	17	14	0.69	1700	14	25	<0.2	87
Wilson	TRM 260.0	13	1.6	34	34	0.17	4800	31	45	<0.2	87
Wheeler	TRM 277.0	12	1.4	35	41	0.22	2400	30	52	0.5	240
Guntersville	TRM 350.4	9	<1	28	35	0.69	2300	23	50	0.4	300
Nickajack	TRM 425.5	10	1.6	50	63	0.48	2700	27	77	0.9	380
Chickamauga	TRM 472.3	15	1.8	35	63	0.77	4900	30	59	0.8	490
Watts Bar	TRM 531.0	16	1.6	34	40	0.62	3600	24	53	0.8	220
Fort Loudoun	TRM 603.2	12	1.6	39	45	0.1	2600	27	75	<0.2	500
Melton Hill	CRM 23.7	13	1.6	18	34	<0.1	3500	23	60	<0.2	120

determine the fate of mercury discharged into area streams. Sediment cores were collected at Clinch River miles 6.8 and 1.0, in LWBR at TRM 550 and 538, and in Chickamauga Reservoir at TRM 502, 490, and 472. Sample collection was intentionally biased for areas of stable sediment deposition. All cores were sectioned into 1-, 2-, 3-, or 5-cm intervals and analyzed for total mercury and gamma emitters. One core (from CRM 1.0) was also analyzed for total uranium and selected metals.

The results of the sediment cores are summarized in Fig. 3.15. Cores from the Clinch River and Watts Bar Reservoir revealed a strong correlation between peak mercury and ^{137}Cs concentrations within the core. The peaks were located 76 cm or more below the core surface in the main channel, where the sediment accumulation rate is highest. In the core from CRM 1.0, the peak uranium concentration (7.8 $\mu g/g$) also coincided with the mercury and ^{137}Cs peaks.

The authors concluded that the peak releases of mercury and ^{137}Cs from ORR facilities in the 1950s are reflected in well-defined peak concentrations in the sediment cores and provide an accurate method of dating sediment layers. The authors postulated that dredging and extreme water-level drawdown were the only likely activities that could resuspend the buried contaminants and bring them into contact with the biosphere (Turner et al. 1984).

3.2.6 TVA Instream Contaminant Study

TVA conducted the Instream Contaminant Study for the Oak Ridge Task Force, a multi-agency body investigating the mercury releases from the Y-12 Plant to EFPC (TVA 1986b). The DOE-funded study was initiated in 1984 and completed the following year. The objectives of the

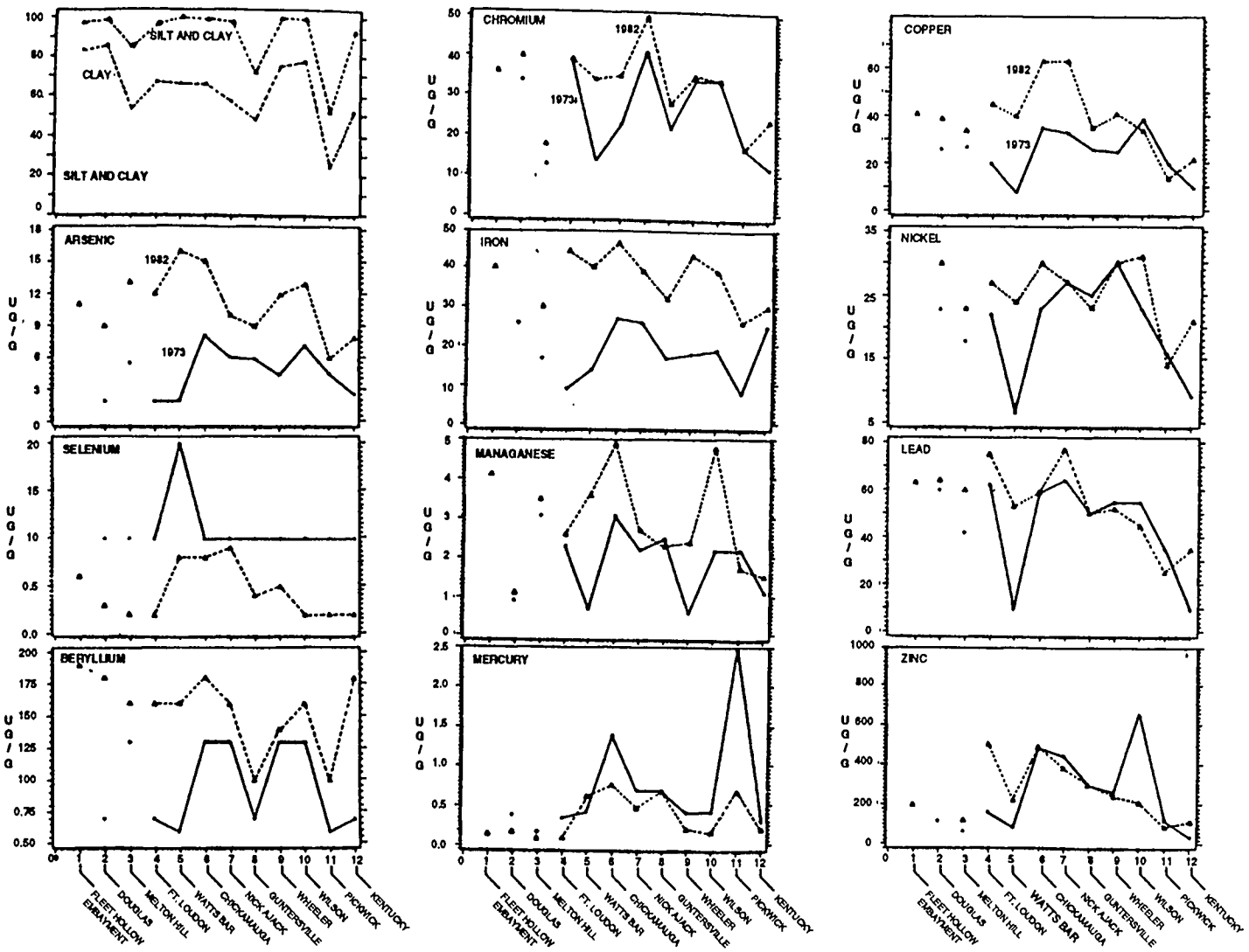


Fig. 3.14. Longitudinal distribution of selected parameters in TVA reservoir forebay sediments (TVA 1986a).

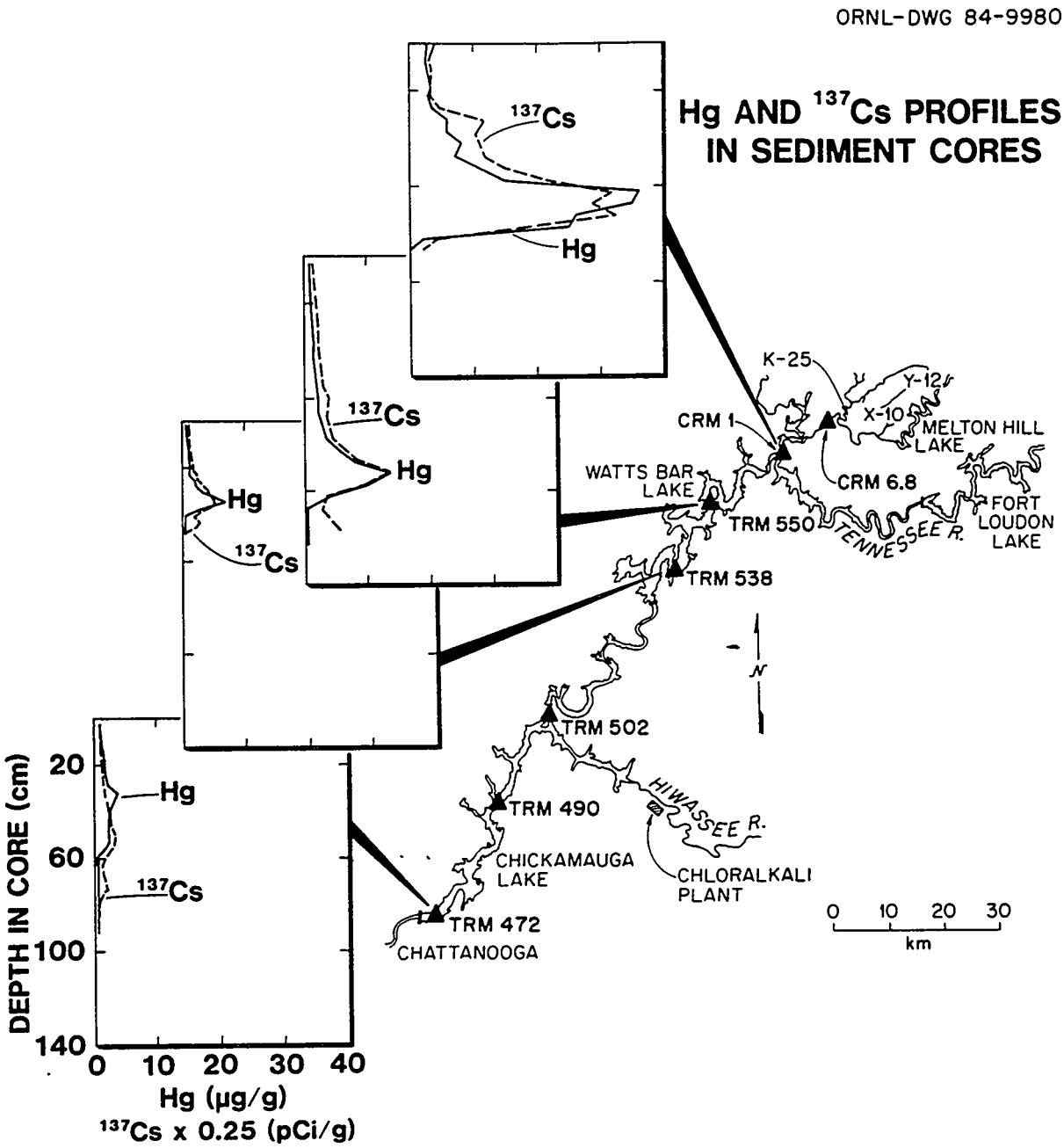


Fig. 3.15. The vertical distributions of mercury and ^{137}Cs in selected sediment cores obtained from Watts Bar and Chickamauga reservoirs (Turner et al. 1984).

study were (1) to identify the presence of contaminants in water, sediment, and fish downstream of the DOE facilities at Oak Ridge; (2) to estimate the quantity of mercury-contaminated sediment in EFPC, Bear Creek, and lower WOC; and (3) to assess the transport and fate of the mercury-contaminated sediment (TVA 1986b). The study consisted of five tasks; data from the Clinch and Tennessee rivers were used in support of Task 2 (sediment characterization) and Task 4 (contaminant concentrations in fish).

In support of Task 2, 5 fine-particle surface sediment samples (the upper 10–15 cm of sediment) were collected from the Clinch River and analyzed for radiological contaminants, base/neutral organic priority pollutants, priority pollutant metals, cyanide, phenols, and PCBs (TVA 1985a). One sample was collected in Melton Hill Reservoir (CRM 24.0) as a reference. In addition, eight cores from the Clinch River downstream of Melton Hill Dam were collected and analyzed for mercury and radionuclides. Eight cores were also collected in the Tennessee River from Watts Bar Reservoir to Guntersville Dam and analyzed for Hg, Cr, and PCBs (TVA 1985a). Results of the Clinch River surface sediment samples are provided in Tables 3.4 and 3.5. None of the 53 organic contaminants for which analyses were conducted were detected in any of the Clinch River surface sediment samples (TVA 1985a). Total phenols were detected in the Clinch River at levels comparable to those at the Norris and Melton Hill Reservoir reference areas (6 μg/kg and 5 μg/kg, respectively). The results of the Clinch River core data for mercury are depicted graphically in Fig. 3.16. The core samples were analyzed by sediment particle-size fraction, [i.e., the <500 μm to >62 μm (fine sands) fraction and the <62 μm fraction (silts and clay)]. The <500 μm fraction (including the <62 μm fraction) accounted for 90% or more of the sample volume at all sites (TVA 1985a). The Tennessee River core data are depicted graphically in Fig. 3.17. PCBs were not detected in any Tennessee River core samples.

The sediment study concluded that (1) concentrations of mercury in sediment of both the Clinch and Tennessee rivers downstream of ORR were elevated and (2) an estimated 500 lbs of mercury were being transported each year out of EFPC to downstream environments.

In support of Task 4, fish samples (fillets) were collected from streams in the vicinity of the ORR, including the Clinch and Tennessee rivers (TVA 1985d). One station was sampled in LWBR (TRM 558.0), where 10 fish each of channel catfish, bluegill, and largemouth bass were collected; a reference station at TRM 572.0 was also sampled. Four sites were sampled in the Clinch River downstream of Poplar Creek; a reference site immediately above Melton Hill Dam was also sampled. Fillets from the Tennessee River sites were analyzed for selected metals and PCBs; the fish from one Clinch River station (CRM 11.0) were also analyzed for radionuclides.

Mean contaminant levels in LWBR fish that exceeded reference concentrations included arsenic (0.13 μg/g) and chromium (0.15 μg/g) in largemouth bass, chromium (0.64 μg/g) in one paddlefish sample, and total PCBs (0.49 μg/g) in channel catfish. Figure 3.18 shows the mean total mercury concentrations in all species of fish along a gradient from upper EFPC to the Tennessee River. The concentrations generally decreased downstream until background levels were attained in the lower reach of the Clinch River and in the Tennessee River (TVA 1986b). Of the 110 organic priority pollutants for which analyses were conducted, 7 were detected in at least 1 fish. Of these seven compounds, PCBs were detected most often and at the highest concentrations; PCB data are summarized in Fig. 3.19.

3.2.7 Oak Ridge Gaseous Diffusion Plant Sediment Survey

Ashwood et al. (1986) collected approximately 180 surface sediment samples from streams around the ORGDP (now known as the K-25 Site). Samples were collected from the Clinch

Table 3.4. Summary of metal concentrations in fine-particle surface sediment TVA Instream Contaminant Study—Task 2 (TVA 1985a)

Parameter	East Fork Poplar Creek			Bear Creek			White Oak Creek			Poplar Creek		
	Concentration (μg/g) N=16			Concentration (μg/g) N=3			Concentration (μg/g) N=4			Concentration (μg/g) N=3		
	Max	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max	Min	Mean
Mercury	165.0	11.0	40.0	0.7	<0.1	0.3	6.0	2.2	3.3	5.9	0.1	3.4
Arsenic	14.0	3.8	6.9	11.0	4.8	6.5	12.0	5.0	8.7	11.0	7.4	8.9
Cadmium	8.2	<0.5	1.6	8.6	<0.5	8.6	2.4	0.6	1.4	3.5	2.1	2.9
Chromium	58.0	24.0	37.0	35.0	16.0	22.0	290.0	66.0	163.0	38.0	19.0	27.0
Lead	170.0	36.0	80.0	85.0	35.0	52.0	51.0	33.0	40.0	38.0	23.0	32.0
Nickel	74.0	20.0	37.0	155.0	28.0	67.0	30.0	24.0	26.0	65.0	43.0	56.0
Silver	45.0	2.0	8.0	<1.0	<1.0	<1.0	10.0	2.0	6.0	2.0	<1.0	2.0
Zirconium	590.0	350.0	448.0	500.0	430.0	500.0	480.0	260.0	365.0	470.0	220.0	340.0

Parameter	Clinch River (Watts Bar)			Background Stations					
				Clinch River (Melton Hill)			Norris Reservoir[1]		
	Concentration (μg/g) N=4			Concentration (μg/g) N=1			Concentration (μg/g) N=3		
	Max	Min	Mean	Max	Min	Mean	Max	Min	Mean
Mercury	2.8	0.3	0.8	<0.1	<0.1	<0.1	0.1	0.1	0.1
Arsenic	11.0	5.1	8.1	17.0	17.0	17.0	26.0	16.0	22.0
Cadmium	4.2	0.7	1.8	0.5	0.5	0.5	<0.5	<0.5	<0.5
Chromium	25.0	9.0	16.0	25.0	25.0	25.0	23.0	21.0	22.0
Lead	38.0	14.0	30.0	28.0	28.0	28.0	77.0	58.0	67.0
Nickel	38.0	14.0	21.0	36.0	36.0	36.0	28.0	24.0	26.0
Silver	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
Zirconium	890.0	400.0	650.0	230.0	230.0	230.0	270.0	180.0	220.0

[1] Values for Norris Reservoir include Clinch River Miles 85.3 and 94.1 and Powell River Mile 6.0.

Table 3.5. Instream contaminant study—Task 2. Maximum concentrations (pCI/g) reported for significant radioisotopes in fine-particle, surface sediment (TVA 1985a)

Control Station					Norris Reservoir			Comparison Data	
	Clinch River	East Fork Poplar Creek	Bear Creek	White Oak Creek	Poplar Creek	Clinch River Upstream	Powell River	Tennessee River[2]	Clinch River[3]
ANALYSIS/ISOTOPE									
Gross Alpha	11	160	32	3	11	4	3	15	100
Gross Beta	398	110	150	6,600	60	42	37	65	91
Uranium[1] 7	90	200	4.1	14	5.9	2.3	—[4]	2.0	
Sr-89	4	.7	.5	0	1	4	1	14	12.1
Sr-90	1.8	.1	.1	900	.2	.4	.5	.6	0.6
GAMMA SPECTRAL ANALYSIS[5]									
Co-60	1.2	—	—	184	—	—	—	.6	2.7
Cs-134	—	—	—	1.2	—	—	—	.13	.01
Cs-137	167	8.6	.2	12,100	1.9	.9	.7	5.5	83
K-40	27	21	18	25	16	27	16	27	84
Ra-226	.7	—	—	—	—	—	—	2.3	1.4
Th-234	4	23	63	—	3.8	—	—	—	—
Eu-152	—	—	—	8	—	—	—	—	—
Eu-154	—	—	—	7	—	—	—	—	—
Am-241	—	—	—	4	—	—	—	—	—
Ac-228	2.2	2	1.8	4	1.1	2.2	1.8	2.7	2.1
Pa-234	—	—	—	—	—	—	—	4.6	4.8
TRANSURANICS									
Pu-239	.73	.10	.001	1.08	.02	—[4]	—[4]	—[4]	—[4]
Pu-238	.03	.02	.001	.007	<.0002	—[4]	—[4]	—[4]	—[4]
Am-241	.51	.05	.04	.46	.02	—[4]	—[4]	—[4]	—[4]
Cm-244	.02	.01	.009	.01	<.0008	—[4]	—[4]	—[4]	—[4]

[1] Uranium reported in units of μg/g, dry weight.
[2] Maximum concentrations reported by TVA in surface sediment samples collected from the Tennessee River from 1981-83.
[3] Maximum concentrations reported by TVA in surface sediment samples collected from the Clinch River from 1974-83.
[4] Analysis not performed.
[5] Dash indicates isotope not identified in gamma spectral analysis.
NOTE: The lower limits of detection for all isotopes are typically 1 pCi/g, dry weight.

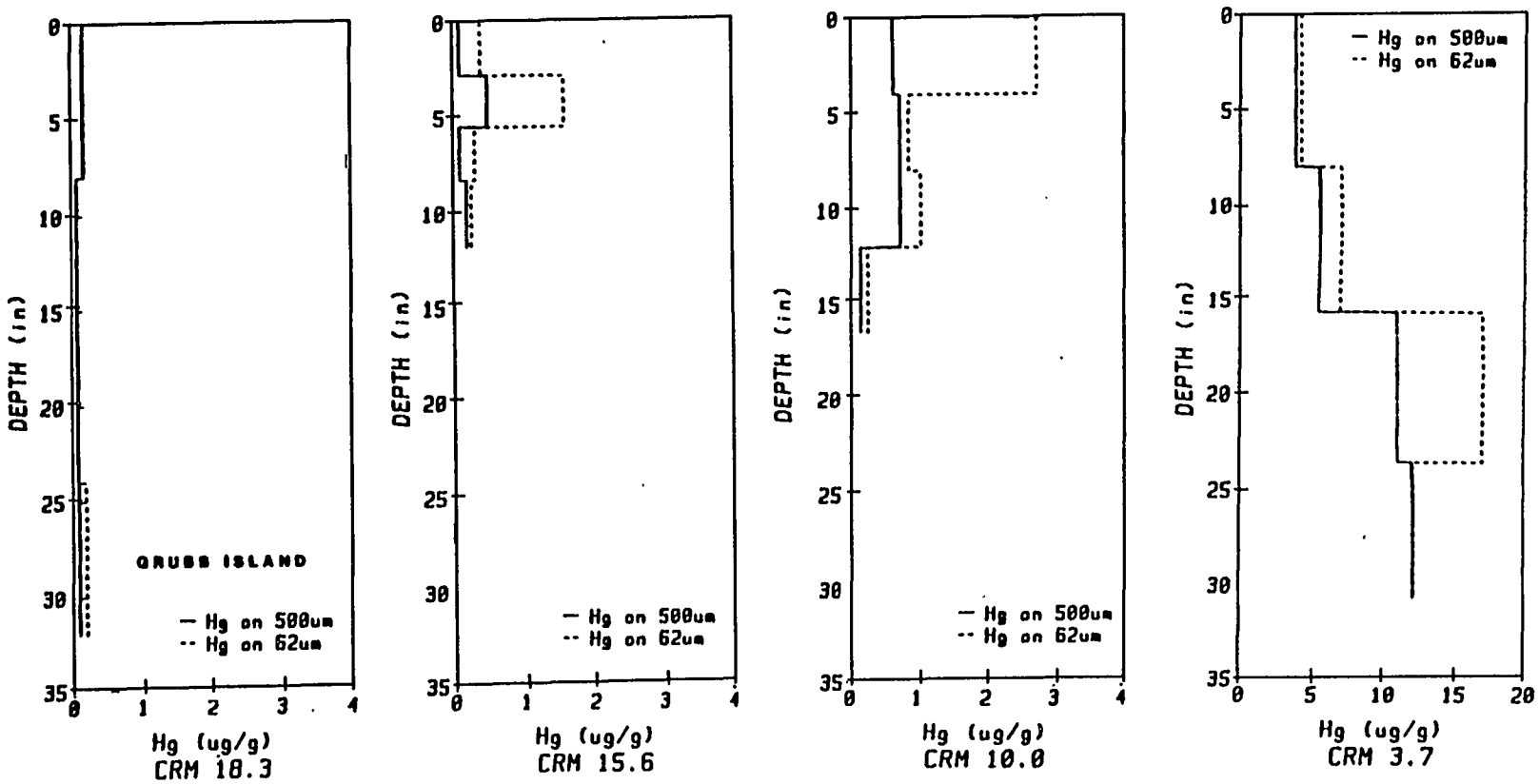


Fig. 3.16. Mercury concentrations vs depth and particle size for Clinch River core sampling (TVA 1985a).

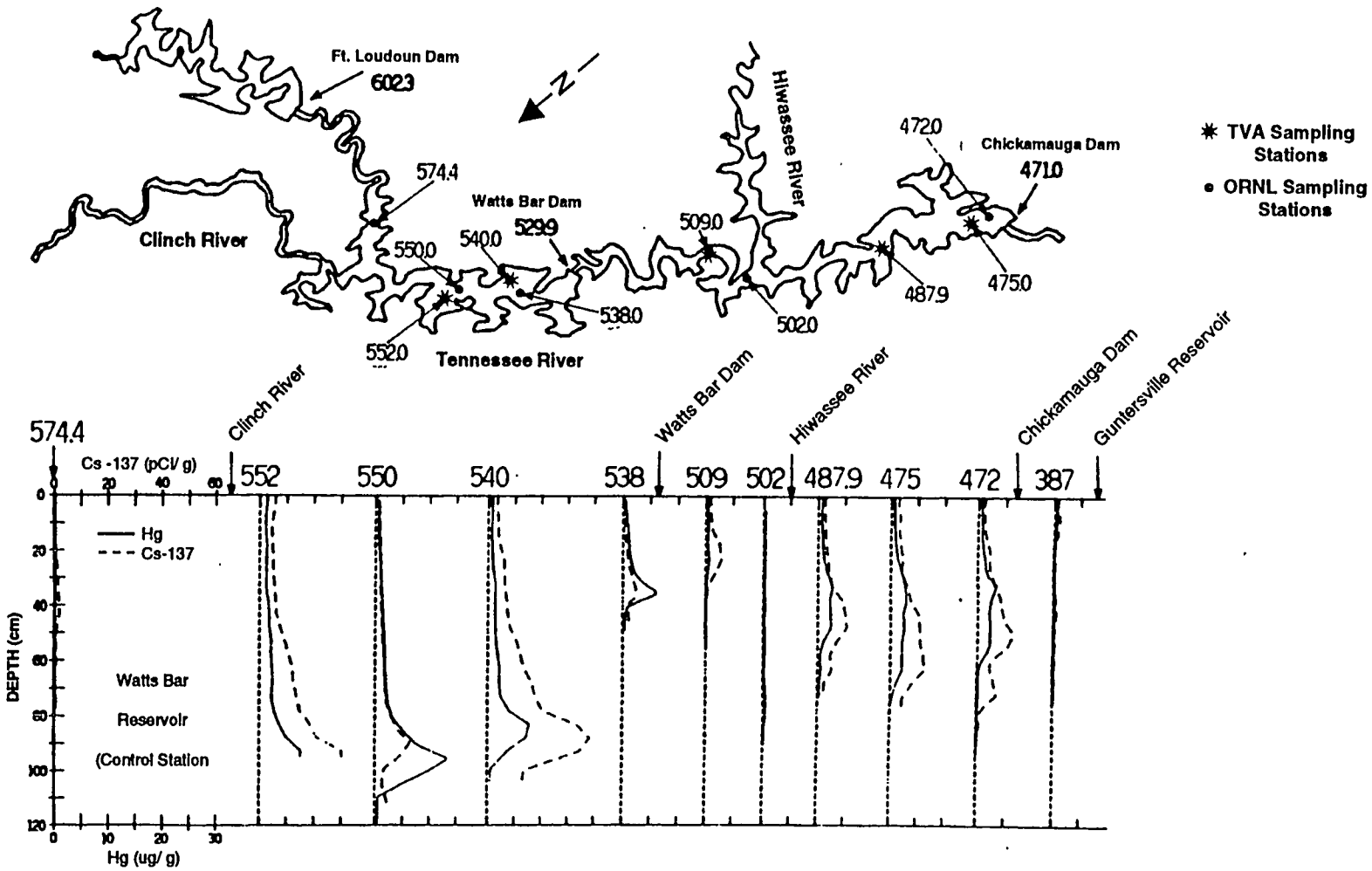


Fig. 3.17. Tennessee River sediment mercury and ^{137}Cs data (TVA 1985a).

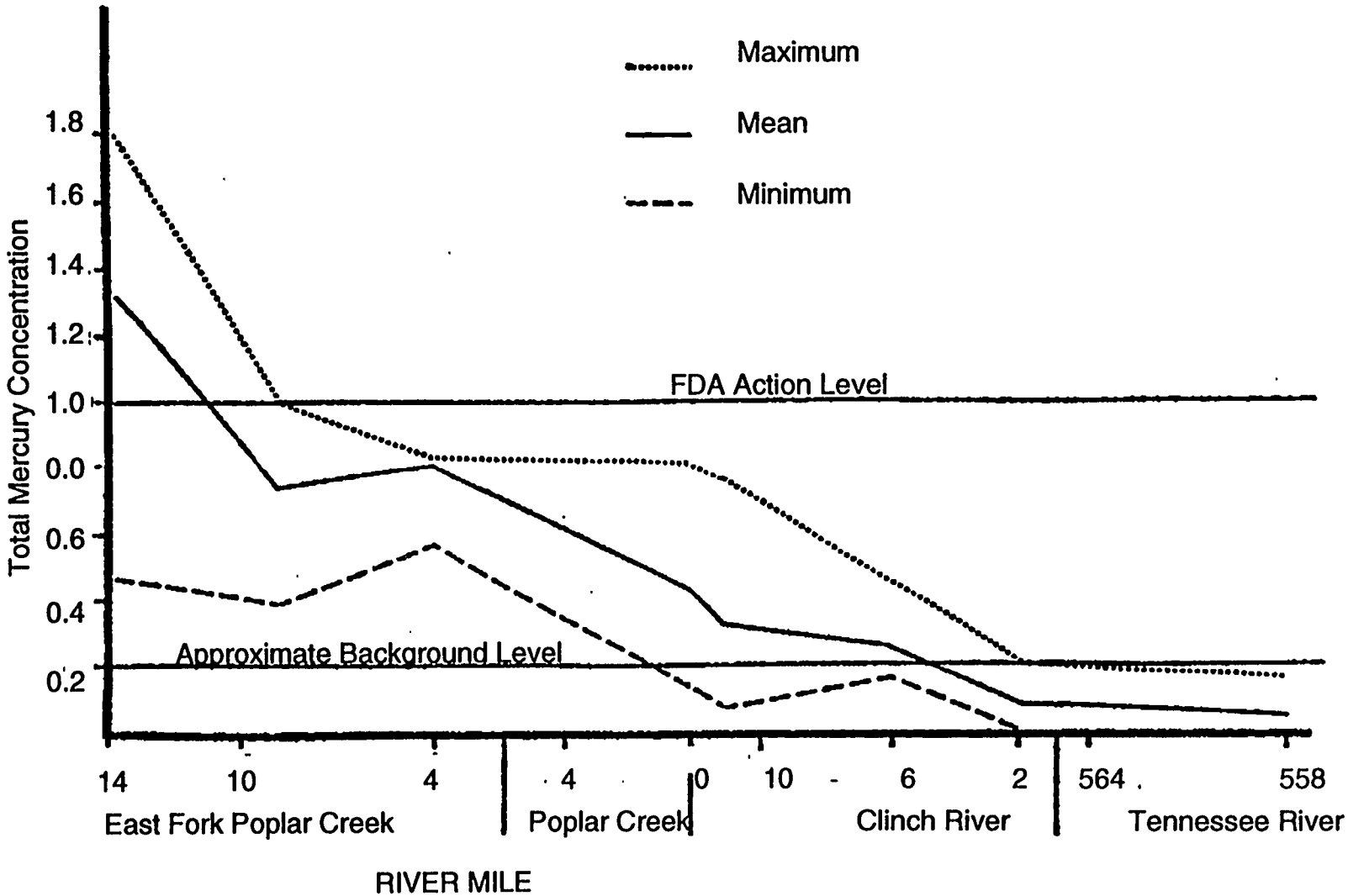


Fig. 3.18. Maximum, mean, and minimum total mercury concentrations in fish in stream reaches below White Oak Creek (TVA 1985d).

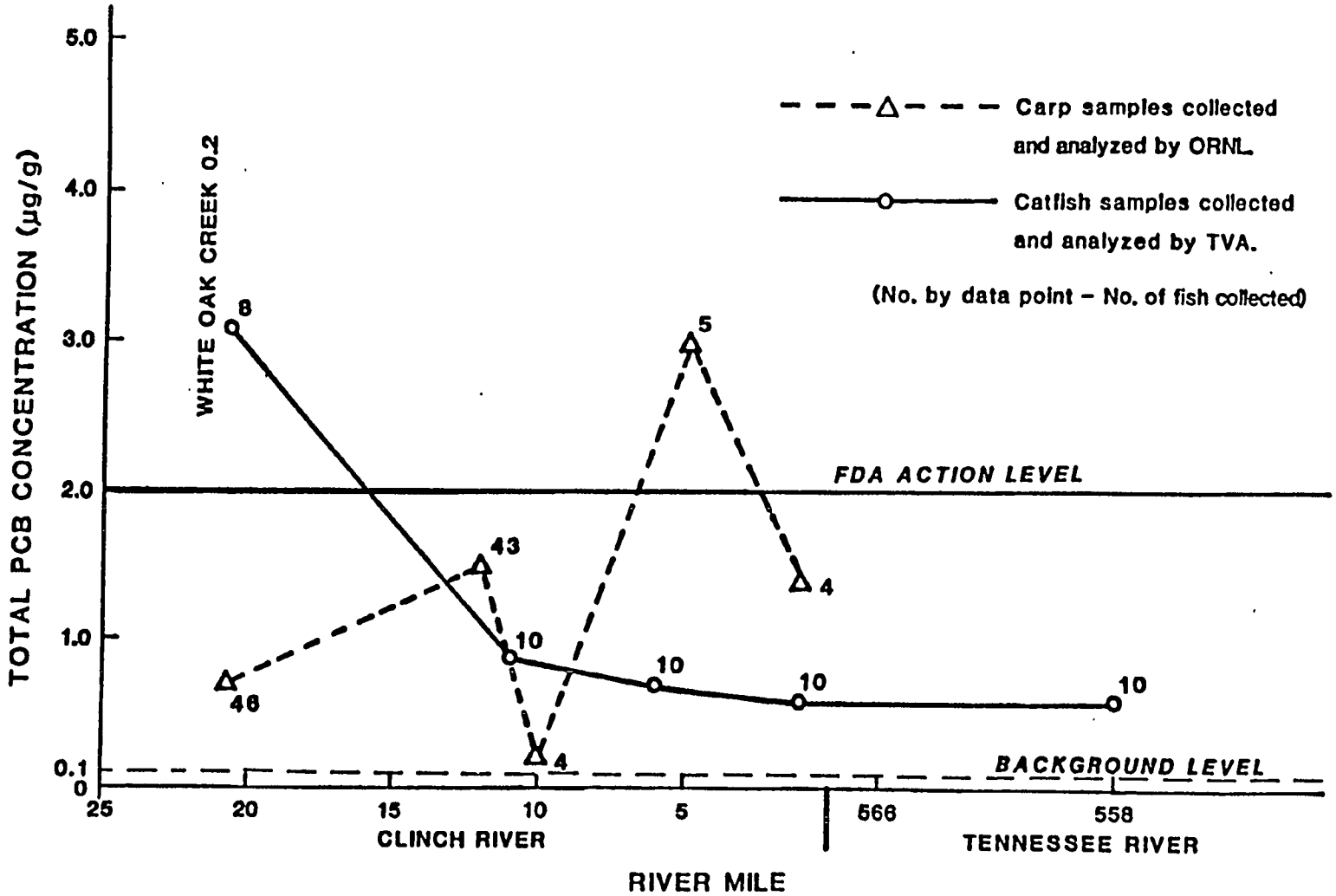


Fig. 3.19. Mean total PCB concentrations in fish flesh instream contaminant study—Task 5 (TVA 1985d).

River, Poplar Creek, and tributaries draining the ORGDP. Samples were analyzed for radionuclides, metals, and organic contaminants. In addition, three core samples were collected, including one from the Clinch River near Kingston. The purpose of the study was to identify sites from which pollutants originating at the K-25 Site had historically entered or were currently entering the surface water. Samples were collected in January and February 1985. The study focused on contaminants that were identified by Hoffman et al. (1984), based on the TVA Instream Contaminant Study Data (TVA 1985a), as warranting further study on the ORR.

Because the objective of the study was to determine areas of high concentrations relative to K-25 values, not relative to background, data are presented as contaminant levels for areas that exceed 150% of the K-25 mean. It was concluded that Clinch River samples contained elevated levels of Se (up to 130 μg/g; $\bar{x}$ = 91 mg/g), ^{137}Cs (up to 14.9 pCi/g; $\bar{x}$ = 2 pCi/g), ^{60}Co (up to 1.34 pCi/g; $\bar{x}$ < 1 pCi/g), and ^{238}U (up to 5.5 pCi/g; $\bar{x}$ = 30 pCi/g) relative to K-25 mean values (i.e., the source of these contaminants would be other than K-25). Contaminants at elevated levels in EFPC included Cd, Hg, Pb, Se, ^{137}Cs, ^{60}Co, ^{238}U, and several organics. Core data (Fig. 3.20) indicate below-surface peaks of ^{137}Cs, Hg, ^{238}U, and several metals in the Clinch River core sample. This core also contained trace quantities of polycyclic aromatic hydrocarbons (PAHs) and phthalates; PCBs were not detected in any core segment (Ashwood et al. 1986).

3.2.8 Transport and Accumulation of ^{137}Cs and Mercury in the Clinch River and Watts Bar Reservoir System

As part of the initial scoping phase of the CRRI, Olsen et al. (1992) in 1986 collected more than 190 surface sediment grab samples and 60 sediment core samples from LWBR to determine the extent of contamination downstream of the ORR and to document particle and particle-associated contaminant accumulation patterns. The specific objectives of the study were to (1) use ^{137}Cs to evaluate the extent of contaminant accumulation in the reservoir sediments; (2) preliminarily identify highly contaminated areas that could constitute potential risks to human health or the environment; and (3) estimate the retention efficiency of the reservoir for ^{137}Cs and, thereby, for other particle-associated contaminants (Olsen et al. 1992).

Surface sediment samples were used to characterize the type and distribution of surface sediments (e.g., gravel, sand, mud, detritus) and were used to produce the sediment type map in Fig. 2.11. In addition, surface sediment samples were also analyzed for ^{137}Cs and naturally occurring ^{40}K by gamma spectrometry. Core samples were sectioned into 1-, 2-, or 4-cm sections and analyzed for ^{137}Cs by gamma spectrometry; selected samples were also analyzed for ^{60}Co and naturally occurring ^{210}Pb. As gamma spectral analyses were completed, selected samples were then analyzed for total mercury content. One core, collected at TRM 567.5 at the mouth of the Clinch River near Kingston, was also analyzed for ^{90}Sr (Olsen et al. 1992).

Several large-volume (400–800 L) water samples were also collected from the Clinch River and LWBR. Suspended solids were removed by centrifugation, dried, and measured for ^{137}Cs and mercury as described above. The dissolved ^{137}Cs in the samples was removed by sorption onto a cation-exchange resin, and the activity was measured. The dissolved ^{60}Co and the naturally occurring radionuclides ^{7}Be and ^{214}Pb were precipitated, and the activity of each was measured. Concentrations of ^{137}Cs in the sole LWBR sample analyzed for radionuclides were 17 fCi/L in the dissolved phase and 5.1 pCi/L in the particulate phase. Cobalt-60 was measured in the dissolved phase at 17 fCi/L and in the particulate phase at 0.9 pCi/L. Mercury was analyzed from a separate sample in LWBR; the dissolved concentration was 0.004 μg/L, and the concentration in the particulate phase was 510 μg/L. These data were used to establish the distribution coefficient of each radionuclide in these water samples (i.e., the ratio of ^{137}Cs associated with

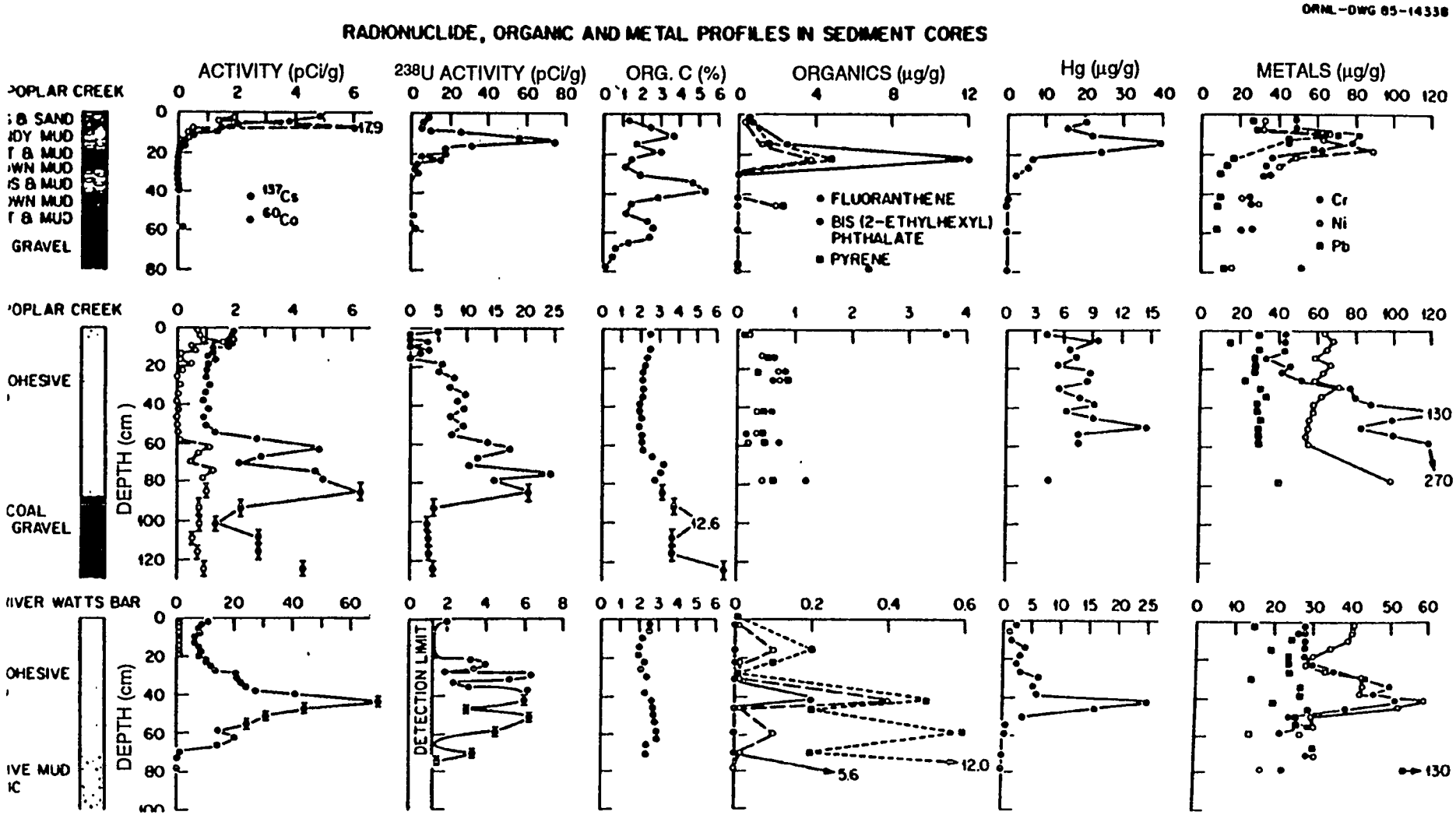


Fig. 3.20. Radionuclide, organic, and metal profiles in sediment cores (Ashwood et al. 1986).

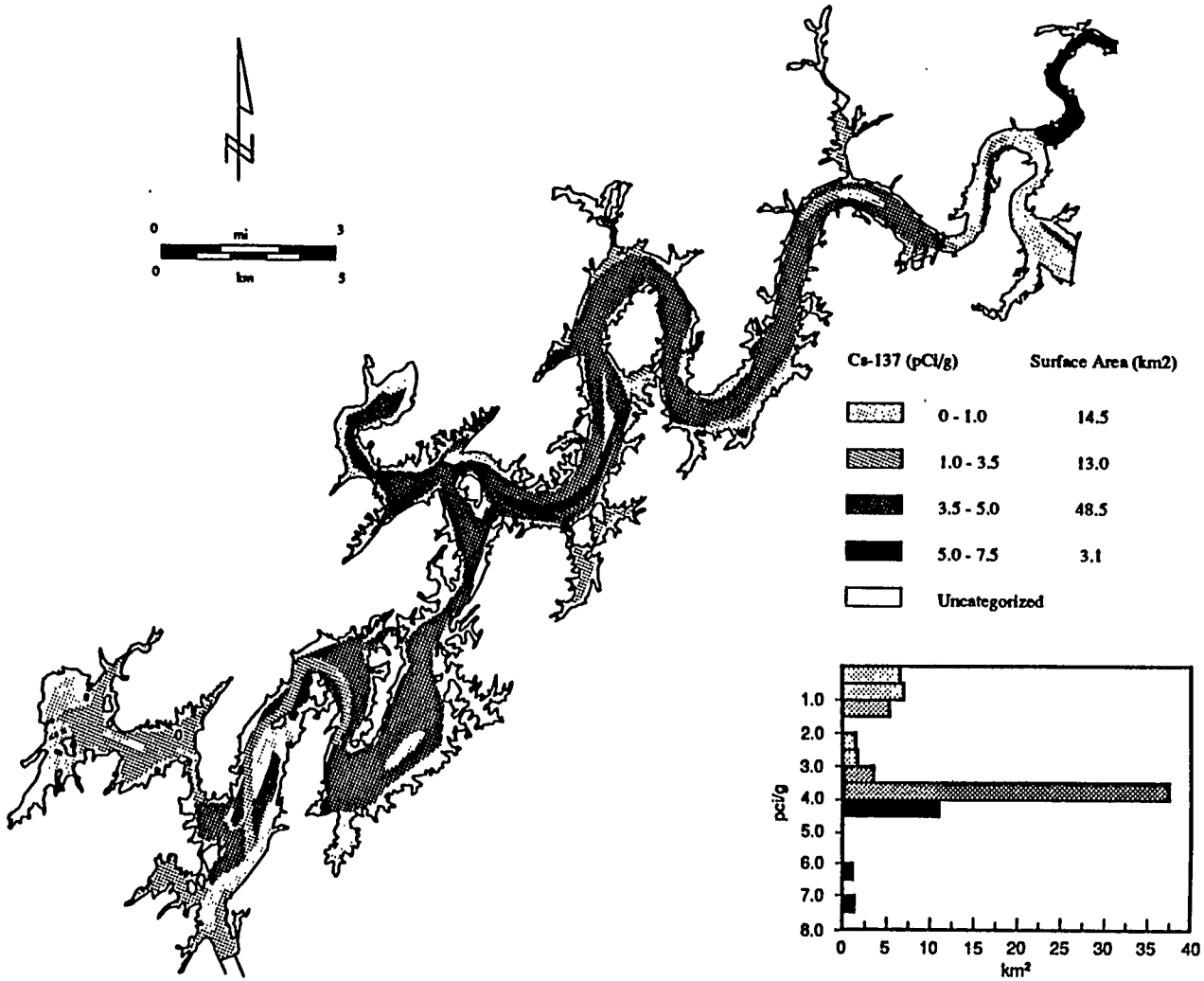


Fig. 3.21. Map of lower Watts Bar Reservoir illustrating the average ^{137}Cs concentration (pCi/g) in the top 16 cm of sediment (Olsen et al. 1992). Watts Bar Dam is at the bottom; Clinch River enters at top right.

suspended particles to that dissolved in the water). The particle-to-water distribution coefficients for ^{60}Co, ^{137}Cs, and Hg were at 8×10^{-4}, 3×10^{-5}, and 5×10^{-5}, respectively, indicating that the amount of each contaminant dissolved in water is about 10,000 times less than the amount sorbed to particulate matter (Olsen et al. 1992).

Surface concentrations of ^{137}Cs are shown in Fig. 3.21. The authors concluded that highest concentrations of ^{137}Cs in surface sediments of LWBR are associated with soft-mud areas; lowest concentrations are found in areas where surface sediments are sand, gravel, or submerged soil.

The ^{90}Sr data for the TRM 567.5 core revealed a peak concentration of 580 pCi/kg at a sediment depth of 80–84 cm and a secondary peak of 325 pCi/kg at a depth of about 40 cm. The former peak coincides with the ^{137}Cs peak, but the secondary peak suggests a substantial release of ^{90}Sr in the early 1970s (Olsen et al. 1992).

The concentrations in each core segment were summed over the area of the core to calculate the inventory of ^{137}Cs. These data were spatially extrapolated to produce a map of the ^{137}Cs inventory in LWBR (Fig. 3.22). Although the authors acknowledge that the map incorporates considerable uncertainty, they conclude that the key points illustrated therein are (1) the highest ^{137}Cs inventories occur in the old river channel, where the depth of the impounded water column has induced rapid accumulation of sediment and particle-associated contaminants; (2) scouring and little or no ^{137}Cs accumulation is occurring in areas where river currents are strong and relatively unaffected by impoundment; and (3) sediments in marginal coves appear to contain relatively minor amounts of ^{137}Cs, suggesting local sediment sources rather than particles derived from the Clinch or Tennessee rivers (Olsen et al. 1992). The total ^{137}Cs inventory of LWBR was estimated at 304 Ci or approximately 91% of the decay-corrected total ^{137}Cs released from WOC.

Mercury concentrations in sediment cores were found to be strongly correlated ($r^2 = 0.87$) with ^{137}Cs. The estimated concentration of mercury (μg/g) in a sediment sample was established to be 0.25 of the ^{137}Cs concentration (pCi/g). Using this relationship, an estimated 84 tons of mercury were calculated to have accumulated in the sediments of LWBR. Estimated average mercury concentrations in surface sediment are depicted in Fig. 3.23; estimated total mercury inventories are depicted in Fig. 3.24.

3.2.9 TVA Recreation Area and Water Intake Sampling

In May and June 1991, TVA sampled sediment and water from 19 recreation area beaches and 11 water intakes located throughout Norris, Melton Hill, and Watts Bar reservoirs (TVA 1991a). Water samples were analyzed for tritium only; sediment samples were analyzed for radionuclides, metals, and organic compounds. The purpose of the study was to gather data useful in addressing public concerns regarding ORR contaminants in Watts Bar Reservoir.

Water sampling consisted of one grab sample at each location. Sediment sampling at the beach sites consisted of collecting five 12-in. cores from each site and compositing them for analysis. Volatile organic analyses were conducted on three cores from each site without compositing. Sediment sampling at intake locations consisted of a single core sample from each site (TVA 1991a).

Tritium was detected at 6 of 11 water intakes at low concentrations (maximum value was 4% of EPA's drinking water standard of 20,000 pCi/L). The maximum ^{137}Cs concentration in sediment (23.7 pCi/g) was at CRM 13, a former Atomic Energy Commission industrial water intake. The maximum ^{137}Cs sediment concentration among the Tennessee River sites was 0.3 pCi/L at TRM 568, at Kingston's Southwest Point Park. All other radionuclides were analyzed at the μCi/g level; none were detectable at this level at any location.

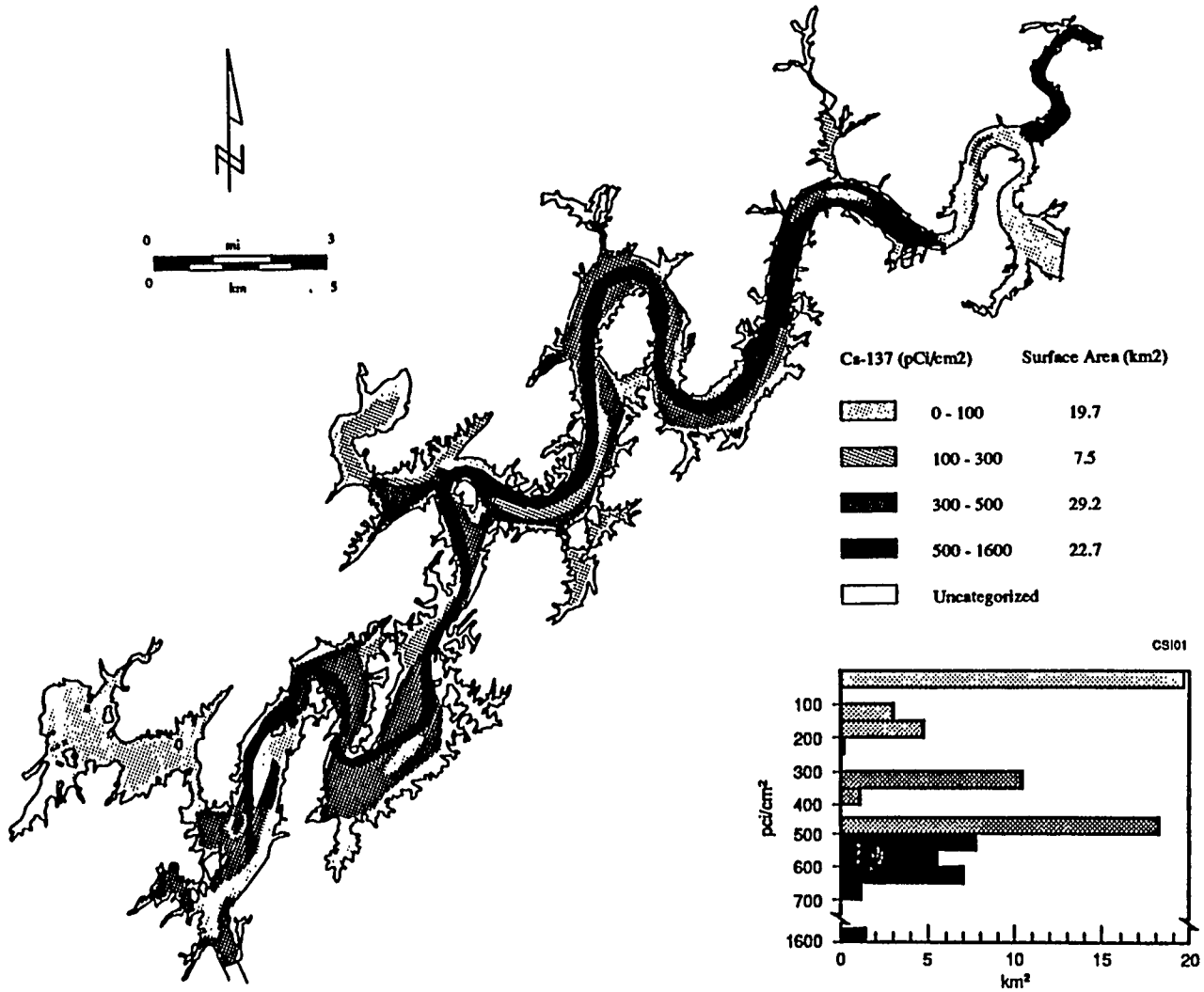


Fig. 3.22. Map of lower Watts Bar Reservoir illustrating the inventory (pCi/cm^2) of ^{137}Cs in the top 50 cm of sediment (Olsen et al. 1992). Watts Bar Dam is at the bottom; Clinch River enters at top right.

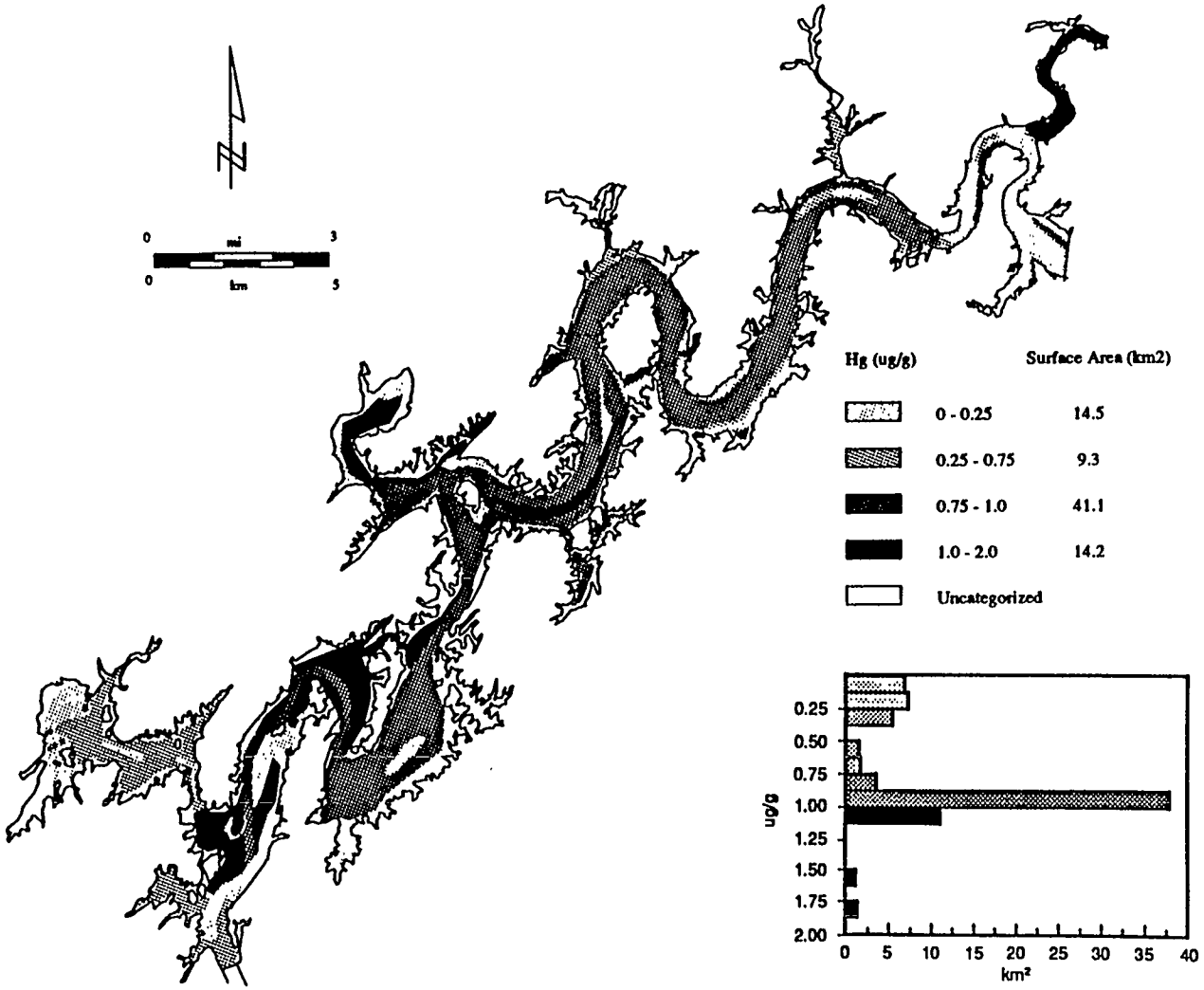


Fig. 3.23. Map of lower Watts Bar Reservoir illustrating the average mercury concentrations (μg/g) in upper 16 cm of sediment (Olsen et al. 1992). Watts Bar Dam is at the bottom; Clinch River enters at top right.

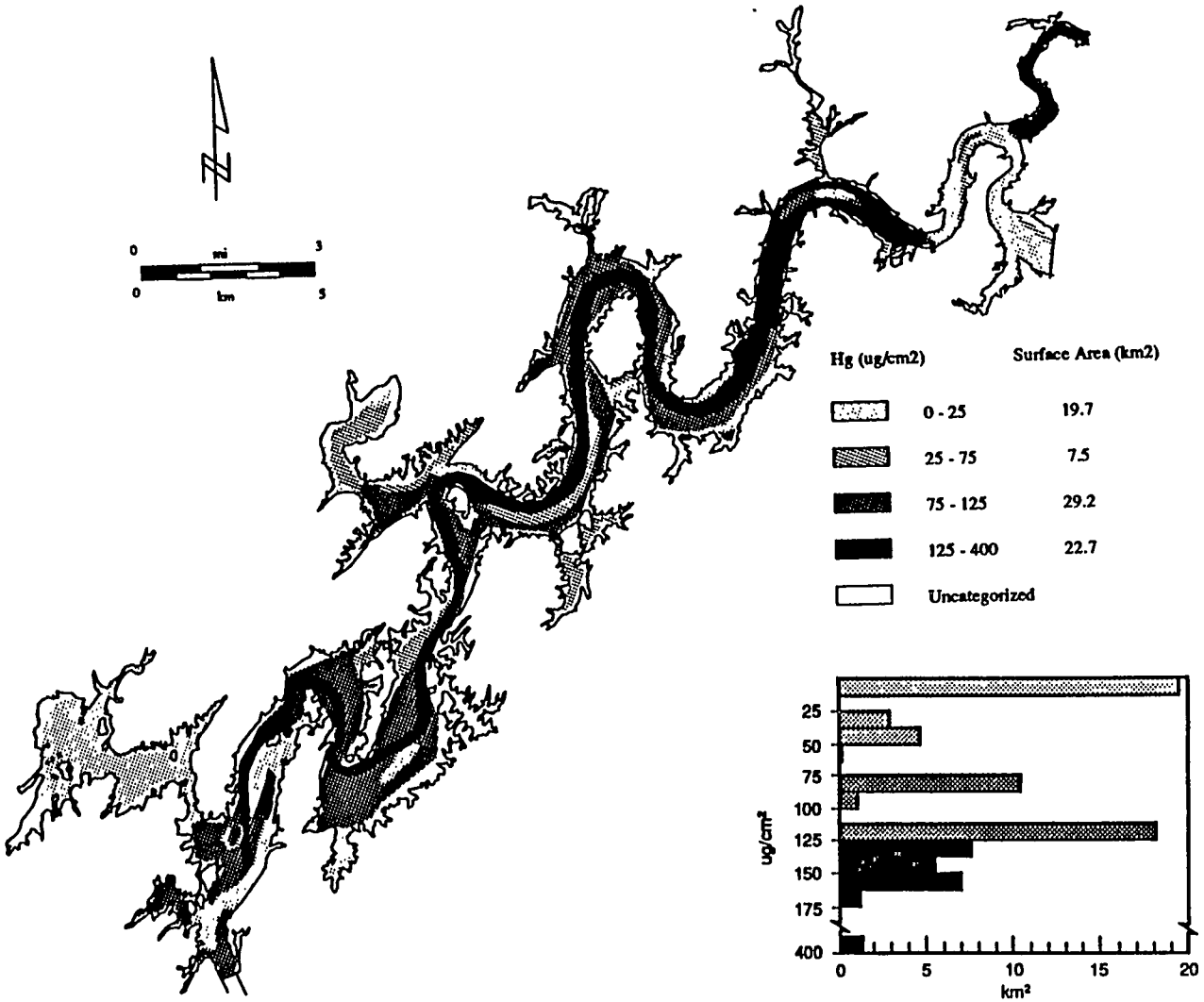


Fig. 3.24. Map of lower Watts Bar Reservoir illustrating mercury accumulation patterns and total inventories ($\mu g/cm^2$) (Olsen et al. 1992). Watts Bar Dam is at the bottom; Clinch River enters at top right.

Table 3.6. Average, maximum, and minimum concentrations of selected metals in sediment at recreation area beaches and at water intakes in lower Watts Bar Reservoir (TVA 1991a)

Analyte	Mean concentration (μg/g)	Minimum concentration (μg/g)	Maximum concentration (μg/g)	Site of maximum value	Maximum reference concentration
Pb	34.6	18	190	TRM 552.5 (Rockwood Municipal Intake)	150 μg/g (Norris Reservoir at CRM 87.5)
As	8.1	1.1	23	TRM 552.5 (Rockwood Municipal Intake)	20 μg/g (Melton Hill Reservoir at CRM 36)
Cr	32.8	11	77	TRM 540.5	44 μg/g (CRM 87.5 and CRM 36)
Ni	24.8	8	51	TRM 540.5	61 μg/g (CRM 36)
Hg	0.3[a]	<0.1	2.5	TRM 568.4 (Kingston Municipal Intake)[b]	0.16 μg/g (CRM 36)

[a] Includes nine nondetect values entered at 0.1 μg/g.
[b] 0.9 mile upstream of LWBR.

The average sediment concentrations of selected metals for the LWBR sites are presented in Table 3.6. Cadmium was not detected at any LWBR station. Only two organic contaminants were detected in any LWBR sediment samples; bis (2-ethylhexyl) phthalate at 390 μg/kg at beach 8 (TRM 545) and 540 μg/g at beach 5 (TRM 538) and methylene chloride at 52 μg/kg at beach 2 (Piney River mile 2.5); and fluoranthene was detected at 400 μg/kg at the Kingston water intake (TRM 568.4) just upstream of LWBR.

3.2.10 TVA Fish Tissue Studies

TVA monitors contaminant concentrations in fish tissue throughout the Tennessee Valley. PCB contamination of fish from Watts Bar, Fort Loudoun, and Melton Hill reservoirs has been documented for several years. Data from TVA investigations are used by TDEC's Division of Water Pollution Control in determining the need for issuing fish consumption advisories. In a given stream, advisories issued because of PCB contamination typically warn (1) against consumption of a species when mean tissue concentrations exceed 2.0 μg/g and (2) to limit consumption when mean concentrations exceed 1.0 μg/g. Currently, advisories are in effect to not consume catfish from Watts Bar Reservoir (Tennessee River arm). Do-not-consume advisories are also in effect for Fort Loudoun, Melton Hill, and Tellico reservoirs immediately upstream of Watts Bar Reservoir (TDEC 1992). Striped bass and striped bass/white bass hybrids should not be consumed from Watts Bar Reservoir; largemouth bass over 2 lbs and largemouth bass from the Little River Embayment should not be consumed from Fort Loudoun Reservoir. Limited-consumption advisories are in effect in Watts Bar Reservoir for white bass, sauger, carp, smallmouth buffalo, and largemouth bass in the Tennessee River arm, and for catfish and sauger in the Clinch River arm (TDEC 1992).

The PCB contamination of fish in Watts Bar Reservoir was first documented in 1985–1986 in catfish and striped bass collected from the tailwaters of Fort Loudoun Dam (Dycus and Hickman 1988). From 1987 through 1988, TVA investigations in Watts Bar Reservoir were expanded in an effort to define the extent of the problem (Dycus 1989 and 1990). During these years, PCB concentrations in catfish and striped bass tissue remained high; concentrations in most other species (i.e., crappie, white bass, largemouth bass, smallmouth buffalo) were lower, but carp and sauger were relatively high. In 1989, the investigation in Watts Bar entered the trend study phase (Hall and Dycus 1991); data collected since then have been used in an attempt to identify trends in PCB contamination in Watts Bar Reservoir fish, primarily catfish and striped bass. To date, no consistent trends in mean PCB concentrations have been identified (Bates et al. 1992, Williams and Dycus 1993).

The contamination of fish in the upstream Fort Loudoun Reservoir has been documented since at least 1981, when PCB levels in catfish from the Little River Embayment averaged 6.6 μg/g, and levels in catfish collected from TRM 628 averaged 4.4 μg/g (Bates et al. 1992). Fort Loudoun has been in the trend study stage since around 1987, and, since 1990, fish have been collected only at TRM 628. No consistent temporal trends in PCB levels have been detected to date (Williams and Dycus 1993).

Screening studies in Melton Hill Reservoir in 1987 (Dycus 1989) revealed concentrations of PCBs (up to 2.0 μg/g) in composite fish samples. Sampling of individual fish from several locations in the reservoir has occurred annually since then. Again, no significant trends in PCB contamination have been identified thus far (Williams and Dycus 1993).

Data for mean PCB contamination in catfish for selected sites in Fort Loudoun and Watts Bar reservoirs (Tennessee River) are illustrated in Fig. 3.25. Similar data for Melton Hill and the Clinch River arm of Watts Bar Reservoir are illustrated in Fig. 3.26. The latter includes data collected during the CRRI.

3.2.11 TVA Annual Radiological Environmental Monitoring, Watts Bar Nuclear Plant

Since 1977 TVA has monitored background radiation near its Watts Bar Nuclear Power Plant as part of the preoperational environmental radiological monitoring program required by the Nuclear Regulatory Commission (TVA 1992). Results are presented in annual monitoring reports. Data are collected from stations near the plant and from reference locations. The purpose of the preoperational monitoring program is to establish a baseline against which to compare environmental radiation levels after the plant has gone on-line. Material sampled includes air, water, milk, foods, vegetation, soil, fish, and sediment; direct radiation levels are also measured.

Watts Bar Reservoir is the first reservoir upstream of Watts Bar Nuclear Plant and serves as a reference station. Surface sediment samples are collected semiannually from the forebay of Watts Bar Reservoir (TRM 532.1) and analyzed by radiochemical methods for ^{89,90}Sr and by gamma spectroscopy (TVA 1992). Average annual ^{137}Cs and ^{60}Co concentrations in surface sediment for the period 1977–1991 are illustrated in Fig. 3.27.

Samples of commercial and game fish (i.e., channel catfish, crappie, and smallmouth buffalo) are collected semiannually from Watts Bar Reservoir. Fillets are analyzed by gamma spectroscopy and also for ^{89,90}Sr. One group of fish is processed whole. Results for ^{137}Cs concentrations in each species for the period 1977–1991 are illustrated in Fig. 3.28.

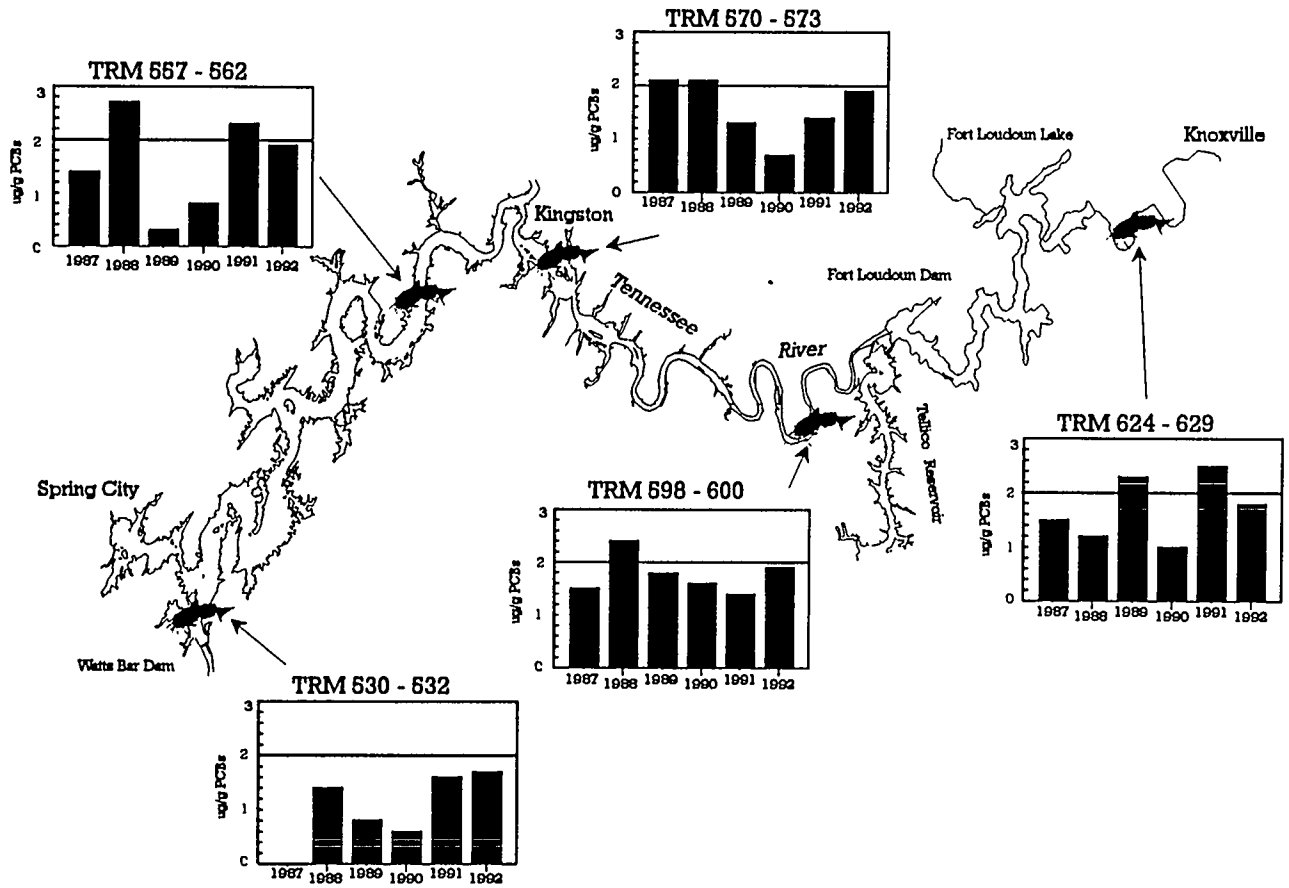


Fig. 3.25. Mean PCB concentration in catfish flesh, 1987–1992, in Fort Loudoun and Watts Bar reservoirs. The FDA action limit is 2μg/g (data from Williams and Dycus 1993).

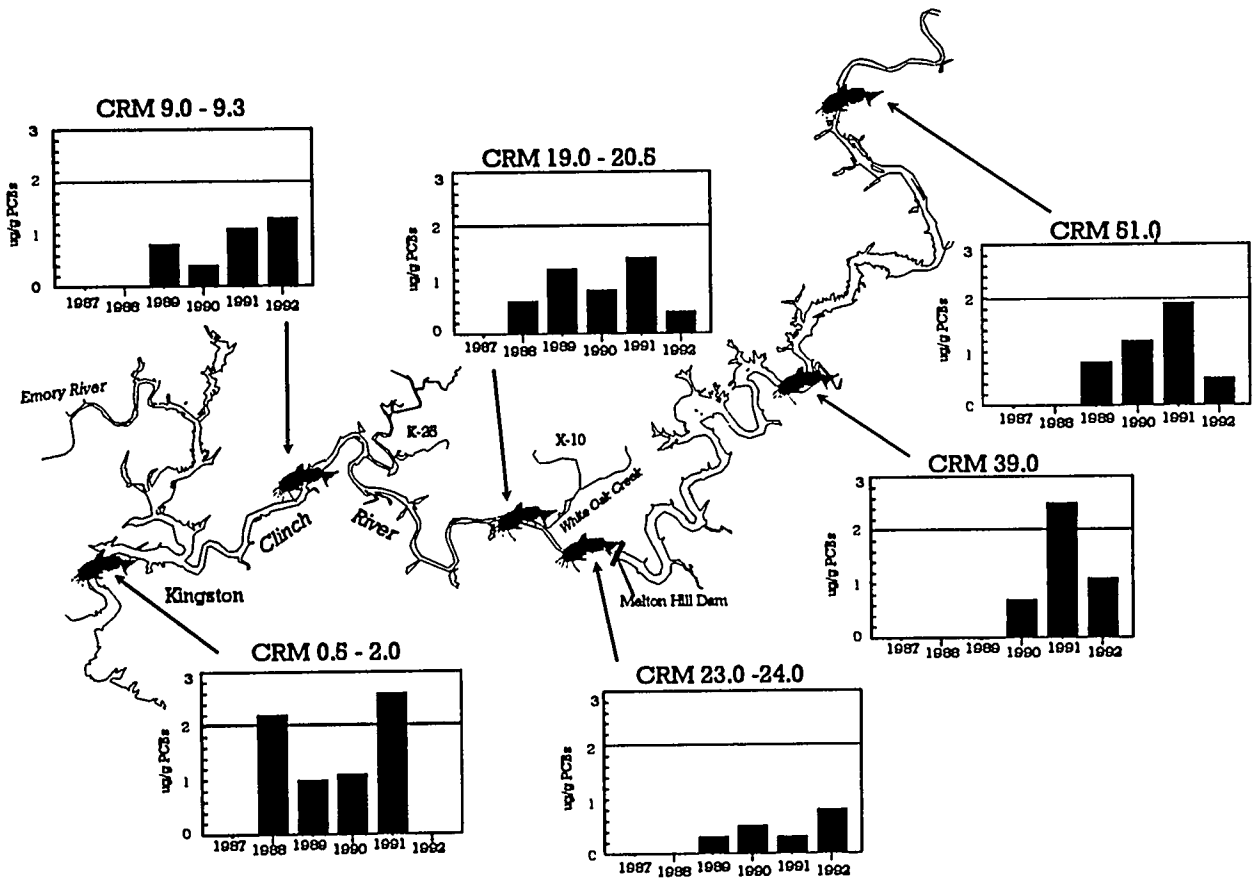


Fig. 3.26. Mean PCB concentrations in flesh of catfish from the Clinch River below Norris Reservoir, 1988–1992. FDA action limit is 2 $\mu g/g$ (data from Williams and Dycus 1993).

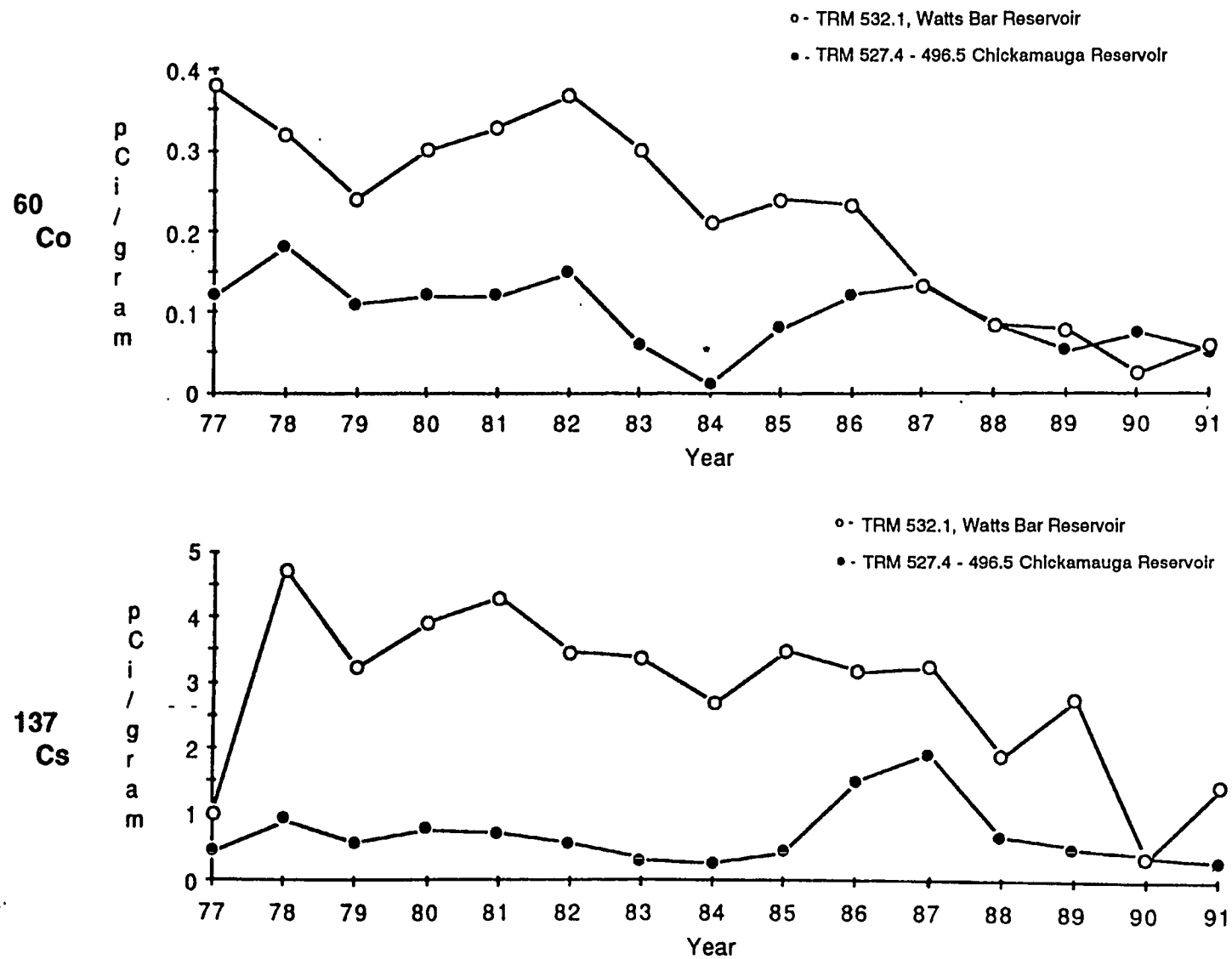
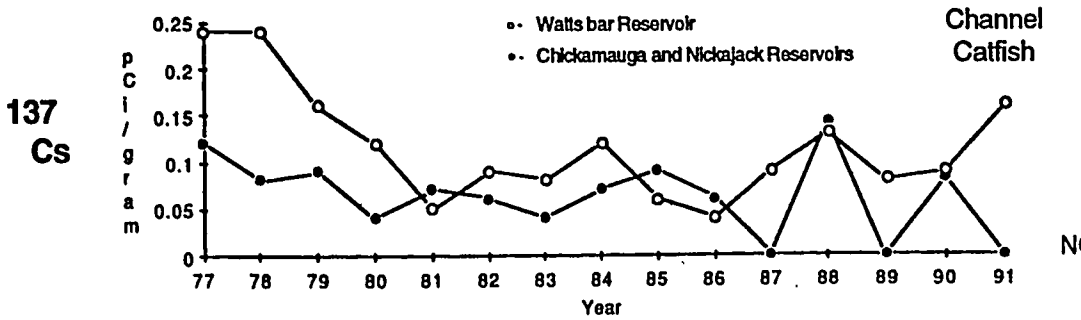
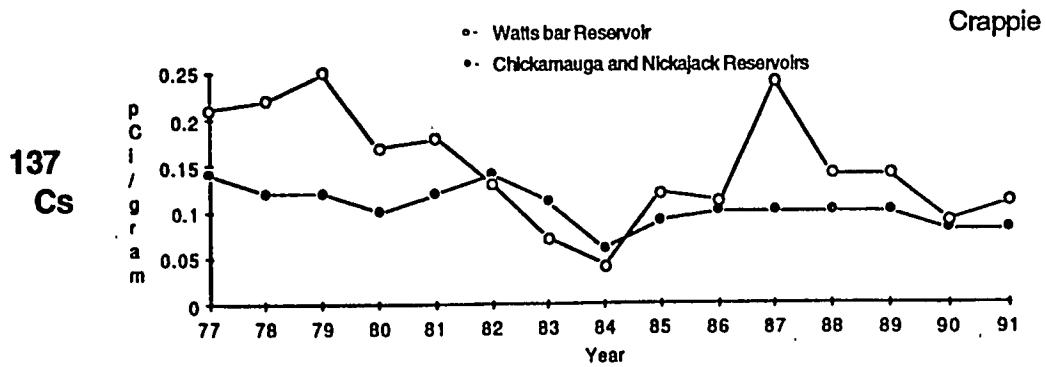


Fig. 3.27. Annual average concentrations of ^{137}Cs and ^{60}Co in surface sediment in forebay of Watts Bar Reservoir (TVA 1992).



NOTE: Detector system changed from NaI to GeLi in 1978.



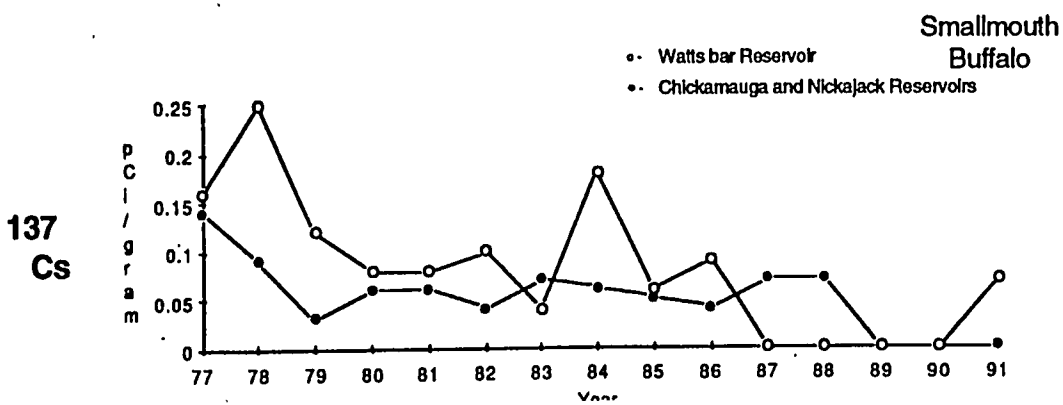


Fig. 3.28. Annual average concentrations of ^{137}Cs in fish from Watts Bar Reservoir (TVA 1992).

3.2.12 TVA Reservoir Monitoring Program—Sediment

TVA implemented its Reservoir Monitoring Program in 1989 for the purpose of monitoring the basic "health" or integrity of the aquatic ecosystem in its reservoirs (the vital signs monitoring component) and to provide screening-level information for describing how well each reservoir meets the "fishable and swimmable" goals of the Clean Water Act (use suitability monitoring component). The Reservoir Monitoring Program employs a number of monitoring tools, including (1) physical and chemical water quality measurements, (2) sediment quality and toxicity tests, (3) benthic macroinvertebrate community evaluations, (4) fish community evaluations, (5) bacteriological sampling, and (6) fish tissue analysis (Meinert and Dycus 1993). This section summarizes the results of the physical and chemical characterization of sediment in Watts Bar Reservoir for monitoring years 1990–1992.

Surface sediment samples (the top 3 cm of sediment) are collected annually from three locations in most TVA reservoirs: the forebay zone, the transition zone, and the inflow zone. The forebay zone is in the lacustrine area of a reservoir; in Watts Bar Reservoir, the forebay sample station is located at TRM 531. The transition zone represents the area in a reservoir where conditions change from a riverine to a lacustrine environment; in Watts Bar Reservoir, this zone is sampled at TRM 561. Watts Bar Reservoir has two inflow zones that are monitored, one at TRM 600–601 (below Fort Loudoun Dam) and the other at CRM 19–22 (below Melton Hill Dam).

At each monitoring zone, three sediment samples are collected and composited for analysis. Samples are analyzed for physical parameters, selected metals, and selected organics. Selected metals data for reservoir forebay samples for the years 1990–1991 are presented graphically in Fig. 3.29. Organic contaminants, including PCBs, were not detected in any Watts Bar Reservoir station in either year (Meinert 1991, Meinert and Fehring 1992). Sediment data for 1992 were not included in the 1993 TVA report (Fehring and Meinert 1992).

3.2.13 ORR Environmental Monitoring Data

DOE routinely monitors effluents from each of its three facilities at the ORR for the primary purpose of assessing compliance with state and federal environmental regulations and DOE orders (Energy Systems 1993). In addition, ambient data are collected from the environment surrounding the ORR and are used, in conjunction with the effluent data, to assess the impact of ORR operations on area groundwater, surface water, soil, air quality, vegetation and wildlife, and the resident human population. Since 1971, the results of these monitoring studies have been summarized in annual environmental monitoring reports for the ORR. Most data, however, are collected on or near the ORR; little data from the surface waters of LWBR are available. Selected water data collected for the ORR Environmental Monitoring Program were used in Chapters 5 and 6 of this report. Data included are surface water samples from CRM 9.9 at Brashear Island, immediately downstream of all ORR inputs. Data used are from the years 1992 and 1993. Sampling at this station has historically consisted of quarterly grab samples that were analyzed for radionuclides, metals, and water quality parameters. In 1992, analysis was expanded to include volatile organic compounds. Table 3.7 summarizes average minimum and maximum concentrations of selected analytes in surface water at the Brashear Island station for the period 1987–1992.

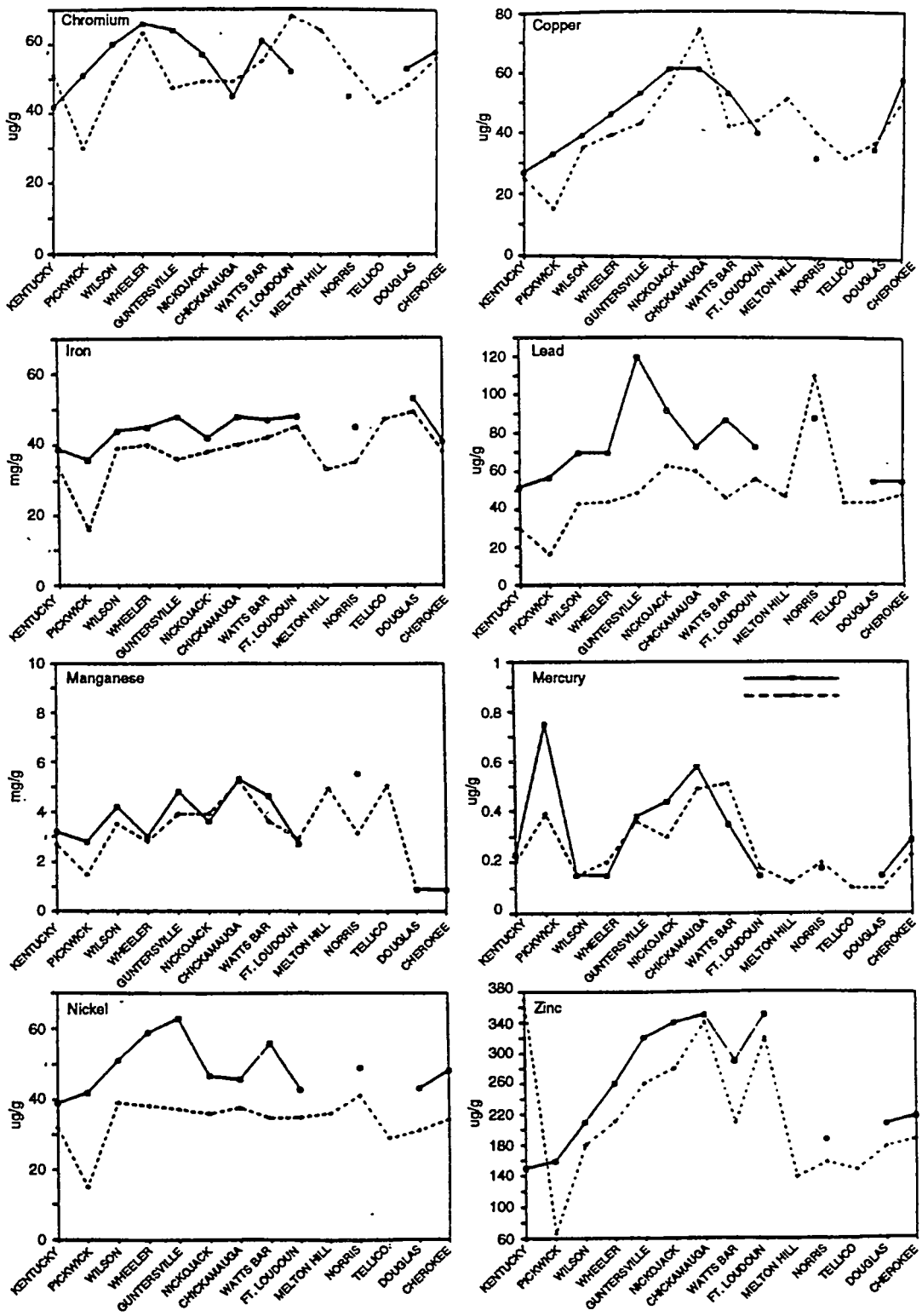


Fig. 3.29. Spatial distribution of metals in sediment at TVA Reservoir forebays 1990–1991 (Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993).

Table 3.7. Mean concentrations of selected analytes in surface water at Clinch River Mile 9.9 (Brashear Island) for the period 1987–1992. All units are μg/L except ^{137}Cs, which is pCi/L. Blank cells indicate no analyses reported. Data from Energy Systems (1988-1993).

		Year					
Analyte	AWQC[1]	1987	1988	1989	1990	1991	1992
Arsenic	50	ND[2] <5	ND <5	ND <5	ND <5	ND <5	4 2/5[3]
Cadmium	5	<2 <2/3	ND <2	ND 2	ND <2	ND <2	1.2 0.5/2
Chromium	100	ND <10	ND <10	ND <10	18 <10/ 440	ND <10	
Lead	NA[4]	<7 <4/25	<4 <4/11	ND <4	ND <4	ND <4	3.3 2/4
Manganese	50	20 <10/74	67 20/200	79 69/ 100	49 25/94	74.4 30.6/ 150	69.3 30/140
Mercury	2			ND <0.2	ND <0.2	ND <0.2	
Nickel	100	ND <50	<52 <50/74	ND <50	12 <10/21	ND <10	23.3 10/50
^{137}Cs	NA[5]	ND <100	<3.8 -100/ <100	ND 0	0.75 0/1.97	0.037 -21.6/ 18.3	2.23 -15.3/ 17

[1] Ambient water quality criteria for domestic water supply.
[2] ND = analyte not detected.
[3] Minimum/maximum values for the year.
[4] No AWQC for water supply available.
[5] AWQC is 4 mrem/yr for all radionuclides combined.

3.2.14 Arsenic Biogeochemistry in the Clinch River and Watts Bar Reservoir

Ford et al. (1993), as part of the CRRI, investigated the speciation, distribution, and mobility of arsenic in the sediment, sediment pore water, and surface water downstream of the ORR. Samples were collected over a 3-year period (1990–1992) at four sites: (1) the McCoy Branch Embayment of Melton Hill Reservoir (McCoy Branch Mile 0.2), (2) Poplar Creek at Mile 1.0, (3) the Clinch River at Kingston City Park (CRM 1.0), and (4) LWBR immediately upstream of Watts Bar Dam (TRM 531.0). Surface water and sediment pore water samples were analyzed for dissolved inorganic arsenic, including the reduced [As (III)] and oxidized [As (V)] forms. Sediment samples (after pore water extraction) were analyzed for total arsenic.

Water column conditions and the dissolved arsenic concentrations in surface water varied seasonally in LWBR. In winter, the water column was not stratified and DO levels were comparable throughout the water column (80–100% saturation). Dissolved inorganic arsenic concentrations were consistently 0.2 μg/L or less throughout the water column. In summer, steep gradients in temperature and DO concentrations with depth indicated stratification of the water column. The dissolved arsenic concentrations in the upper (5 m) water column were similar to those found in winter, but values as high as 0.7 μg/L were found near the sediment. By fall, the reservoir was no longer stratified and both temperature and DO concentration were again relatively uniform throughout the water column. Dissolved arsenic concentrations were also consistent throughout the water column, but at levels (0.3–0.4 μg/L) comparable to those found in deeper water during summer. Arsenic (V) was the predominant (>90%) species under all conditions.

Dissolved arsenic concentrations in pore water from LWBR also varied seasonally. During winter and spring, sediment pore water concentrations were low near the sediment surface (0.7–1.1 μg/L), but increased by a factor of ten a few centimeters deep in the sediment. In summer, concentrations of dissolved arsenic in pore water were 2 to 3 times higher in the upper (18 cm) sediment than in winter and spring. By fall, concentrations returned to presummer levels. In deeper sediment (>18 cm), arsenic concentrations remained similar regardless of season. Arsenic (V) was the predominant (>90%) form of arsenic in pore water during all seasons.

The concentrations of total arsenic in sediment did not change appreciably during the study, remaining less than 18.7 mg/kg.

The authors concluded that increased levels of arsenic in the deeper surface waters at the LWBR study site during summer are a natural phenomenon associated with low DO in the hypolimnion. During most of the year, adequate DO maintains iron and manganese as insoluble oxyhydroxide compounds, with which arsenic becomes complexed and is thereby rendered insoluble. Under the low DO conditions of summer, microbial organisms use the oxidized forms of iron and manganese in anaerobic respiration. Iron and manganese oxyhydroxides are preferentially reduced, simultaneously releasing inorganic arsenic to the water column. The oxidized arsenic (V) is thus also available for reduction to the even more mobile arsenic (III). The concentrations of dissolved arsenic measured at the LWBR site during the course of the study did not exceed any human health or ecological risk screening values (Ford et al. 1993). Screening criteria were not applied to pore water concentrations, where pore water from deeper sediment (>18 cm) would continuously exceed human health and ecological screening criteria and pore water from surface sediment (0–18 cm) would often exceed ecological screening criteria for arsenic (V).

3.2.15 TWRA Non-point Source Pesticide Study

TWRA is currently monitoring the concentration of 71 non-point source pesticides and related compounds in fish and mussels collected from waters of the state. Although these contaminants are not specific to the ORR, the data will contribute to an understanding of contaminant issues in LWBR. The final results of the study are not yet available, and the data have not resulted in any firm conclusions regarding these contaminants. Data for the LWBR and upstream areas were made available by the TDEC DOE-Oversight Division in Oak Ridge for summary here (Tables 3.8 through 3.11). These tables only contain data for analytes that were detected at any of the sampling locations specified therein. Most of the 71 analytes were not

Table 3.8. Non-point source pesticides and related compounds in channel catfish from Watts Bar Reservoir and upstream reservoirs, 1991–1992. All units are μg/g. Unpublished TWRA data.

Contaminant	Lower Watts Bar Reservoir (TRM 545)	Fort Loudoun Reservoir Tailwaters (TRM 600)	Melton Hill Reservoir (CRM 51)	Fort Loudoun Reservoir (TRM 628)
Monochlorobiphenyls		3.0	3.5	
Trichlorobiphenyls	7.5			4.5
Pentachlorobiphenyls	56.1	53.6		51.5
Hexachlorobiphenyls		3.0		4.0
Heptachlorobiphenyls		8.5		
cis-Chlordane	22.0	34.6	5.5	30.5
trans-Chlordane		15.0		34.0
cis-Nonachlor	52.6	61.2		14.5
Oxychlordane		29.1		
Butachlor		4.5		3.0
Dieldrin		57.6		
trans-Nonachlor	92.3	78.7		78.5
Pentachloroanisole		4.0		
Triphenyl Phosphate		3.5	3.5	30.0
Methyl Parathion		171.9	54.2	106.5
Di-n-butylphthalate		69.2	25.6	23.5
Dinoseb		18.3	23.3	
Diphenyl disulfide	3.0			
p,p'-DDE	99.2			
Chlorpyrifos	15.5			
Cyanizine	8.53		27.9	33.6
alpha-BHC			6.0	8.5
Carbaryl			76.0	
Picloram			96.1	
Disulfoton				
Simazine				6.5
Nitrofen				
Diethylphthalate				
2,3,7,8-TCDF	0.00144	.00166		
2,3,7,8-TCDD	0.00207	.00309		0.00193

Table 3.9. Non-point source pesticides and related compounds in channel catfish from Watts Bar Reservoir and upstream reservoirs, 1992–1993. All units are μg/g. Data from TWRA unpublished reports.

Contaminant	Poplar Creek Mile 1.0	Kingston City Park (CRM 1.0)	Lower Watts Bar Reservoir (TRM 545)	Lower Watts Bar Reservoir (TRM 560)	Fort Loudoun Reservoir (TRM 624)	Fort Loudoun Tailwaters (TRM 601)	Melton Hill Reservoir (CRM 39)	Tellico Reservoir (LTRM 35)
Tetrachlorobiphenyls	13.1			5.8	7.0	10.4	10.4	
Pentachlorobiphenyls	53.9	21.3		38.8	51.6	61.4	61.6	11.0
Hexachlorobiphenyls	23.5			28.2		5.9	5.9	
cis-Chlordane	50.7	21.0	7.0	47.5	17.0	331.5	23.7	28.4
trans-Chlordane	71.5	25.5	8.0	54.4	19.0	31.0	30.1	29.7
p,p'-DDE	37.3	61.6	21.5	145.3	60.5	143.1	30.1	114.4
cis-Nonachlor	17.1	18.5	6.0		14.0	24.0	14.1	19.6
Oxychlordane	15.3		10.5				6.4	
Dieldrin	24.4			16.5				7.0
Heptachlor epoxide	11.6	13.5						
trans-Nonachlor	49.5			10.3				
Pentachloroanisole	12.8							
Di-n-butylphthalate	13.4	9.0		14.5	5.5	6.0	9.6	21.5
Triphenyl Phosphate		20.0			15.0		14.1	12.6
alpha-BHC			9.5	31.7	10.0	31.7	12.8	
gamma-BHC			5.5					
Methyl Parathion	99.2							7.0
2,4-D					4.0		6.9	
Nitrofen				6.9		6.9		
2,3,7,8-TCDF	0.00144	0.00117		0.00178		0.00178		0.00169
2,3,7,8-TCDD	0.00207	0.00186		0.00151		0.00151		0.00209

Table 3.10. Non-point source pesticides and related compounds in mussels from Watts Bar Reservoir and upstream reservoirs, 1991–1992. All units are μg/g. Species are elephant ear (*Elliptio crassidens*) unless otherwise noted. Unpublished TWRA data.

Contaminant	Watts Bar Tailwater[1] Replicate 1	Watts Bar Tailwater Replicate 2	Watts Bar Tailwater[2] Replicate 1	Watts Bar Tailwater[2] Replicate 2	Fort Loudoun Tailwater (TRM 598.7)	Fort Loudoun Reservoir (TRM 650.5)
Monochlorobiphenyls	5.0	6.5	3.0	4.0	7.5	3.0
Methyl Parathion			40.6	39.1		18.0
Di-n-butylphthalate	23.0	26.0	27.1	11.0	13.5	23.0
Chlorpyrifos	8.5	6.5				
alpha-BHC	10.0	6.5		9.0		7.0
Carbaryl		17.1		13.2		
Picloram					57.5	
Disulfoton		5.5		5.0		
Simazine						14.5
Nitrofen		4.0				
Diethylphthalate				12.5		
2,3,7,8-TCDF	0.00117	0.00155	0.00135	0.00141		
2,3,7,8-TCDD	0.00117	0.00155				

[1] Watts Bar Tailwater locations are at Hunters Shoal.
[2] Purple wartyback mussels *(Cyclonaias tuberculata).*

Table 3.11. Non-point source pesticides and related compounds in mussels from Watts Bar Reservoir and upstream reservoirs, 1992–1993. All units are μg/g. Species are elephant ear (*Elliptio crassidens*) unless otherwise noted. Unpublished TWRA data.

Contaminant	Watts Bar Tailwater[1] Replicate 1	Watts Bar Tailwater Replicate 2	Watts Bar Tailwater[2]	Fort Loudoun Tailwater (TRM 597.5)
Oxychlordane		5.0		
Heptachlor epoxide	23.1	9.5		
Di-n-butylphthalate	42.6	73.9	16.4	16.0
alpha-BHC				9.0
gamma-BHC				9.0
Methyl Parathion	14.0			8.0
2,3,7,8-TCDF	.00183	.00213	.00183	.00221

[1] Watts Bar Tailwater locations are at Hunters Shoal.
[2] River pigtoe mussels *(Pleurobema cordatum).*

detected at any of the locations specified. The values for channel catfish are single measurements of composite samples consisting of one fillet from each of five fish. The values for mussels are also single measurements; the number of mussels per composite sample is 10–15 for elephant ear mussels *(Elliptio crassidens)* and 7 for the 1992–1993 river pigtoe mussels *(Pleurobema cordatum)*. The number of purple wartyback mussels *(Cyclonaias tuberculata)* composited in the 1991–1992 study year was not specified.

3.2.16 CRRI Phase 1 Investigation

The objectives of Phase 1 of the CRRI were to (1) obtain documented quality data to confirm or modify preliminary results from earlier scoping studies; (2) determine the range of contaminant concentrations present in the off-site environment; (3) identify the contaminants of concern; and (4) establish the reference concentrations for these contaminants (Cook et al. 1992). Sampling for Phase 1 of the CRRI included sediment, water, and biota from throughout the original Clinch River/Watts Bar Reservoir study area. Samples from all media were analyzed for radionuclides, inorganic contaminants, and organic contaminants. Sampling was conducted from December 1989 through July 1990. Results of the Phase 1 sampling were summarized by Cook et al. (1992).

Water

Four water samples were collected from LWBR—one from upper, two from middle, and one from lower Watts Bar Reservoir. Four samples were collected in the Clinch River downstream of Melton Hill dam, two of which were collected below the mouth of Poplar Creek, the last surface water source of ORR contaminants to the system. All water samples were surface grabs from 1 m below the water surface. Sampling was restricted to winter. Samples were analyzed for total and dissolved metals, total and dissolved nonvolatile organics, and total and dissolved radionuclides. Volatile organics were purposefully excluded from sampling because it was expected that all such compounds would have volatilized from surface water prior to reaching the Clinch River and certainly before reaching LWBR. The naturally occurring cations Ca, Mg, K, and Na were prevalent in both total and dissolved water samples from all sites. No inorganic or organic compounds were detected in LWBR, and no organic contaminants were detected in the Clinch River below the mouth of Poplar Creek (CRRI Reach 4). Cadmium was detected in one dissolved water sample from Reach 4 of the Clinch River at a concentration of 0.31 μg/L. Lead was detected in two samples from Reach 4, at 1.0 μg/L in a dissolved sample and at less than 1.5 μg/L in a total water sample. Both cadmium and lead were also detected at comparable levels in upper Melton Hill Reservoir, upstream of all ORR inputs. Each inorganic contaminant detected during the CRRI Phase 1 investigation was well below the Tennessee Water Quality Criteria value (Sect. 4). Radionuclides detected in LWBR included ^{60}Co in one filtered sample at 0.18 pCi/L, ^{137}Cs in one filtered sample at 0.05 pCi/L, and ^{90}Sr in one total water sample at an estimated 0.7 pCi/L. Radionuclides detected in Reach 4 water were ^{137}Cs in both filtered samples at an estimated maximum concentration of 0.13 pCi/L and ^{90}Sr in one total water sample at less than quantifiable limits. No gamma-emitting radionuclides were detected in surface water from any reference site. The presence of ^{90}Sr was reported for all study and reference reaches, but at less than quantifiable limits. Conclusions from the Phase 1 data indicated that most of the elevated levels of contamination were found in the Poplar Creek embayment of the Clinch River and the Clinch River between the mouth of WOC and the mouth of Poplar Creek (CRRI Reach 2).

Fish

During Phase 1 of the CRRI (1989), 44 fish were sampled from 2 locations in LWBR (Cook et al. 1992). Since that time, an additional 83 samples have been collected from 3 sites in LWBR as part of cooperative sampling with TVA and to support CRRI Phase 2 sampling. The number of samples of each species at each site are shown in Table 3.12.

The following analytes were detected in more than 10% of the samples from LWBR: As, Cu, Ni, Hg, Zn, ^{137}Cs, ^{90}Sr, aldrin, chlordane, Aroclors 1254 and 1260, DDD, DDE, DDT, and 2-methylnaphthalene. Mean concentrations of total PCBs (Aroclor 1254 + Aroclor 1260) are highly variable among species and from year to year (Table 3.13).

Table 3.12. Summary of samples collected during CRRI investigations into contaminants in fish flesh in LWBR

Year	Site	Species	Samples	Analyses[a]
1989	TRM 557	catfish	9	PCBs/pest., SVOs, rads
		largemouth bass	2	rads, metals
		bluegill	10	rads, metals
	TRM 530	catfish	10	PCBs/pest., SVOs, rads
		largemouth bass	3	rads, metals
		bluegill	10	rads, metals
1991	TRM 561	catfish	10	PCBs and chlordane
		largemouth bass	10	PCBs and chlordane
		gizzard shad	3[b]	PCBs and chlordane
	TRM 530	catfish	10	PCBs and chlordane
		largemouth bass	10	PCBs and chlordane
		gizzard shad	3[b]	PCBs and chlordane
1992	TRM 545	catfish	20	PCBs and chlordane
		striped bass	15	PCBs and chlordane
		gizzard shad	2[b]	PCBs and chlordane

[a]Analyses included PCBs (Aroclors), SVOs (semi-volatile organics), rads (radionuclides), pest. (pesticides), chlordane, and heavy metals.
[b]Composite samples of several fish.

Table 3.13. Mean annual concentrations of total PCBs in fish flesh, LWBR

		Total PCB concentration ($\mu g/g$)			
Year	Tennessee River Mile	catfish	largemouth bass	shad	striped bass
1989	TRM 557	0.33			
1989	TRM 530	0.79			
1991	TRM 561	2.31	0.32	0.30	
1991	TRM 530	1.58	0.26	0.28	
1992	TRM 545	1.25	Data are not yet available.		

A comparison of Watts Bar PCB concentrations in catfish from 1989 to 1992 (based on TVA and ORNL data) indicates that levels in lower Watts Bar are slightly lower than those in the upper Tennessee River arm and slightly higher than those in the Clinch River arm, though these differences may not be statistically significant (Fig. 3.30).

Sediment Cores

Sediment core samples were collected from throughout the study area, including three locations in LWBR (TRMs 531, 546, and 556) and two sites in Reach 4 of the Clinch River (CRMs 9.5 and 0.5). Cores were sectioned vertically into 6-cm increments (upper 18 cm of core only) and 4-cm increments (remaining length of core). Each core section was analyzed for inorganic and organic contaminants and radionuclides (Cook et al. 1992).

The only inorganic contaminant to exhibit any peak in the core profiles was mercury, which exhibited a well-defined peak between 40 and 80 cm deep in cores collected in and downstream of Poplar Creek. Average concentrations of inorganic contaminants in cores are depicted in Table 3.14. Average concentrations represent the concentrations that would result from mixing the sediment profile by dredging and subsequent land disposal, an activity that could potentially expose persons to the contaminants in the sediment. The data indicate elevated levels of arsenic in the sediment of Poplar Creek. As compared to the Clinch River, elevated levels of Zn, Cd, Cr, Cu, and Pb were found in LWBR, suggesting a source other than the Clinch River. Levels of mercury are highest in the Clinch River below the mouth of Poplar Creek, as might be expected from the known release history of this contaminant.

Organic contaminants were low throughout the system, with only six organic compounds detected anywhere. The only organic contaminant detected in LWBR was the common industrial pollutant and common laboratory contaminant bis(2-ethylhexyl)phthalate, at a maximum concentration of 7.5 μg/g. In Reach 4 of the Clinch River, contaminants included low concentrations of PCBs (Aroclor 1254), bis(2-ethylhexyl)phthalate, and several PAHs (Cook et al. 1992).

Alpha-emitting radionuclides were at or near background levels for every site sampled except Poplar Creek, where ^{234}U, ^{235}U, and ^{238}U were slightly elevated (maximum concentrations of 8.91, 0.30, and 6.92 pCi/g, respectively). The average highest concentration of ^{90}Sr (0.396 pCi/g) was also observed in Poplar Creek. The highest average concentrations of gamma-emitting radionuclides (primarily ^{137}Cs) were found near Brashear Island (CRM 9.5) in Reach 4 of the Clinch River (Table 3.14). This site is the first major zone of sediment accumulation in the Clinch downstream of the ORR facilities. Average concentrations decrease from this point downstream, but remain above background (Table 3.14). As expected, concentrations in the surface sediments were relatively low, with peak concentrations of ^{137}Cs found 50 to 80 cm deep in the sediment.

Near-Shore Sediment Survey

The goal of the near-shore sediment characterization task was to determine the extent to which near-shore surface sediments are contaminated by releases from the ORR and to provide data to the Watts Bar Reservoir Interagency Working Group (IWG) for use in evaluating the human health risks from exposure to sediments during and following dredging operations (Cook et al. 1992).

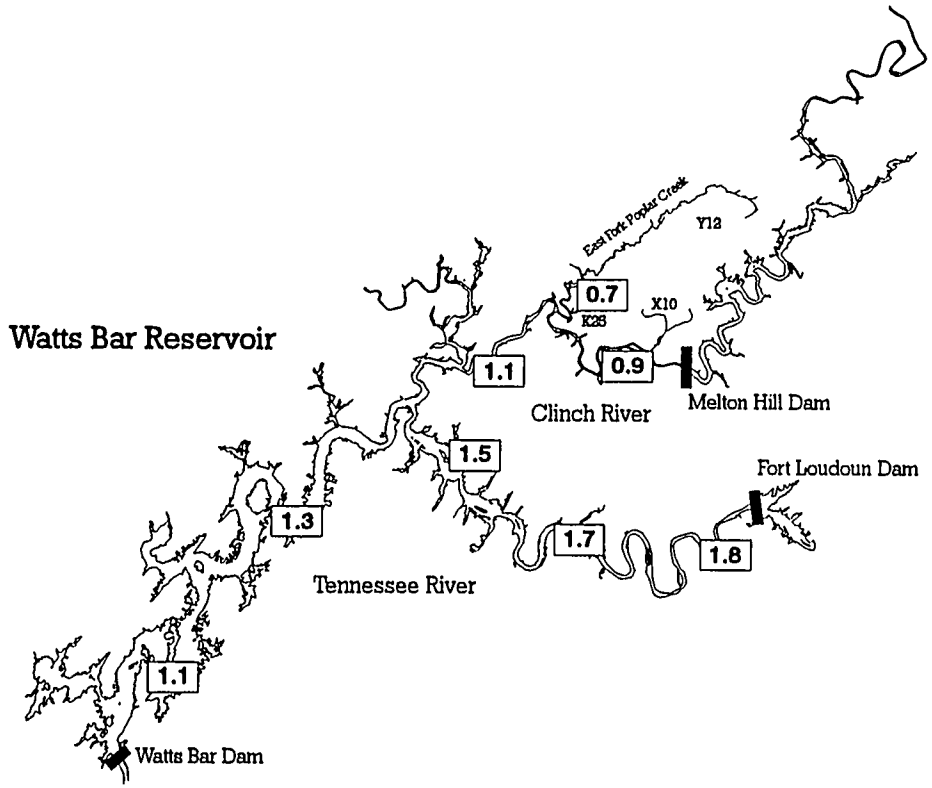


Fig. 3.30. Mean PCB concentrations (μg/g) in catfish from Watts Bar Reservoir from 1987–1992 (458 samples; data sources—TVA and ORNL).

Table 3.14. Average concentrations of inorganic contaminants and radionuclides in sediment cores collected from Poplar Creek, the Clinch River, and the Tennessee River, 1989 (Cook et al. 1992)

Site	Inorganics, μg/g								Radionuclides pCi/g	
	As	Zn	Be	Cd	Cr	Cu	Pb	Hg	^{137}Cs	^{60}Co
CRM 103 Norris Reservoir	5	88	1.3	0.6	22	20	33	0.08	0.89	0.17
CRM 44.1 Upper Melton Hill Reservoir	5	69	0.6	0.4	10	27	30	0.06	0.17	0.12*
CRM 23.9 Melton Reservoir Forebay	11	122	0.8	0.7	19	29	39	0.12	1.04	0.32
CRM 19.7 Jones Island	6	71	0.4	0.3	12	14	25		17.37	0.18
CRM 14.5 Gallaher	4	59	0.8	0.5	16	12	22	0.24	27.93	0.35
PCK 7.4	3	50	0.3	0.4	15	20	21		0.04	0.07*
PCK 1.6	140	56	1.6	1.3	30	34	23	0.49	8.44	0.08*
CRM 9.5 Brashear Island	9	66	0.6	0.7	21	22	16	15.92	63.64	0.44
CRM 0.5 Kingston City Park	11	124	1.2	1.4	27	35	32	2.92	26.82	0.53
TRM 556 Thief Neck Island	9	243	1.1	1.1	30	38	45	2.96	10.85	0.18
TRM 546 Mid-LWBR	9	268	1.4	5.2	35	35	45	2.15	13.74	0.31
TRM 531 Watts Bar Reservoir Forebay	9	259	1.4	1.4	35	39	44	1.00	10.38	0.22

*Estimated values. Below accurate quantitation limit of 0.16 pCi/g.

Sampling during this task concentrated on near-shore areas where human exposure to sediments is most likely. These areas are exposed for several months every winter and are also the most likely to be disturbed by shoreline development activities, such as placement of riprap, construction of erosion containment walls, dock construction, and dredging of shallow areas for low-water boat access. The general approach has been to systematically sample the near-shore area throughout the system. However, samples were also collected at sites where permit applications had been submitted for activities that could disturb sediment, to assess the impacts

of these proposed actions. It should be noted that no deep-water dredging of LWBR sediment has ever occurred. Dredging of a navigation channel in the Clinch River arm of the reservoir did occur in conjunction with the construction of Melton Hill Dam in the early 1960s.

The IWG, formed in February 1991, includes representatives from the U.S. Army Corps of Engineers (COE), TVA, DOE, EPA, and the Tennessee Department of Health and Environment (now the TDEC). TVA regulates sediment-disturbing activities in its reservoirs under authority of Sects. 26a, 4(k), and 31 of the TVA Act. The COE regulates activities in navigable waters under authority of Sect. 10 of the Rivers and Harbors Act of 1899 and the placement of fill material into waters of the United States under authority of Sect. 404 of the Clean Water Act. TVA and the COE routinely review permit applications for such activities in Watts Bar Reservoir. The IWG was formed to assist these agencies by screening dredging permit applications to determine if the proposed activity would pose a risk to human health. The initial area of concern for the IWG was in the Tennessee River from TRM 529.9 (Watts Bar Dam) to TRM 569.0 (Webster Bluff Light), in the Clinch River from CRM 0.0 (mouth of the Clinch River at TRM 567.5) to CRM 20.8 (at the mouth of WOC), and the Emory River from Emory River Mile (ERM) 0.0 (mouth at CRM 4.3) to ERM 5.2 (downstream from the confluence of the Little Emory River).

As of December 1992, approximately 1100 samples had been collected and analyzed as part of this task (Table 3.15). Analysis of these samples has been limited to gamma spectrometry. The IWG defined 15 pCi/g of ^{137}Cs as an "action level" concentration. This is the concentration at which the lifetime risk of excess cancer exceeds 10^{-4} (i.e., 1 in 10,000 chance), given a 2000 h/year direct exposure to the contaminated sediment for 30 years. The action level designation meant simply that additional characterization was needed for permit evaluation. No permit applications to date have been refused on the basis of contaminated sediment. After evaluation of the data by the IWG in January 1992, the area of concern was restricted to areas below the winter pool elevation (735 ft msl).

Table 3.15. Number of surface sediment samples, mean, minimum, maximum, and standard deviation for ^{137}Cs concentrations in LWBR, the Clinch River below Poplar Creek, and the Clinch River from White Oak Creek to Poplar Creek

Reach	Number of Samples	Mean (pCi/g)	Minimum (pCi/g)	Maximum (pCi/g)	Standard Deviation (pCi/g)
CRM 20.8–12.0	170	2.53	0.06	37.90	4.93
CRM 12.0–mouth	155	2.88	0.18	21.80	2.86
LWBR	488	0.52	0.01	10.30	0.83

Figures 3.31 and 3.32 (LWBR and the Clinch River, respectively) show the locations of surface samples and ^{137}Cs concentration ranges taken during the near-shore sediment sampling and the scoping study of Olsen et al. (1992).

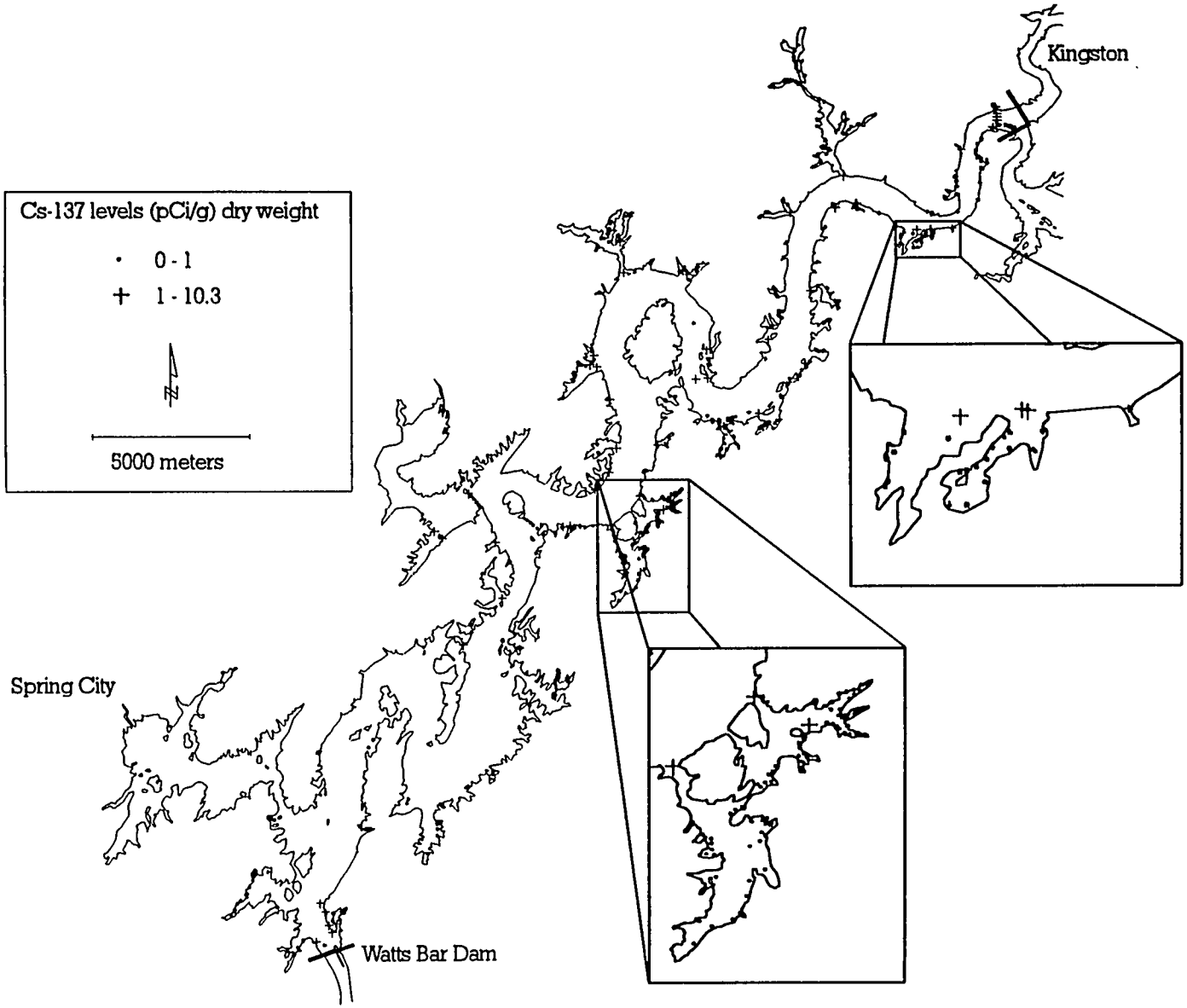


Fig. 3.31. Levels of ^{137}Cs in near-shore surface sediment samples in lower Watts Bar Reservoir.

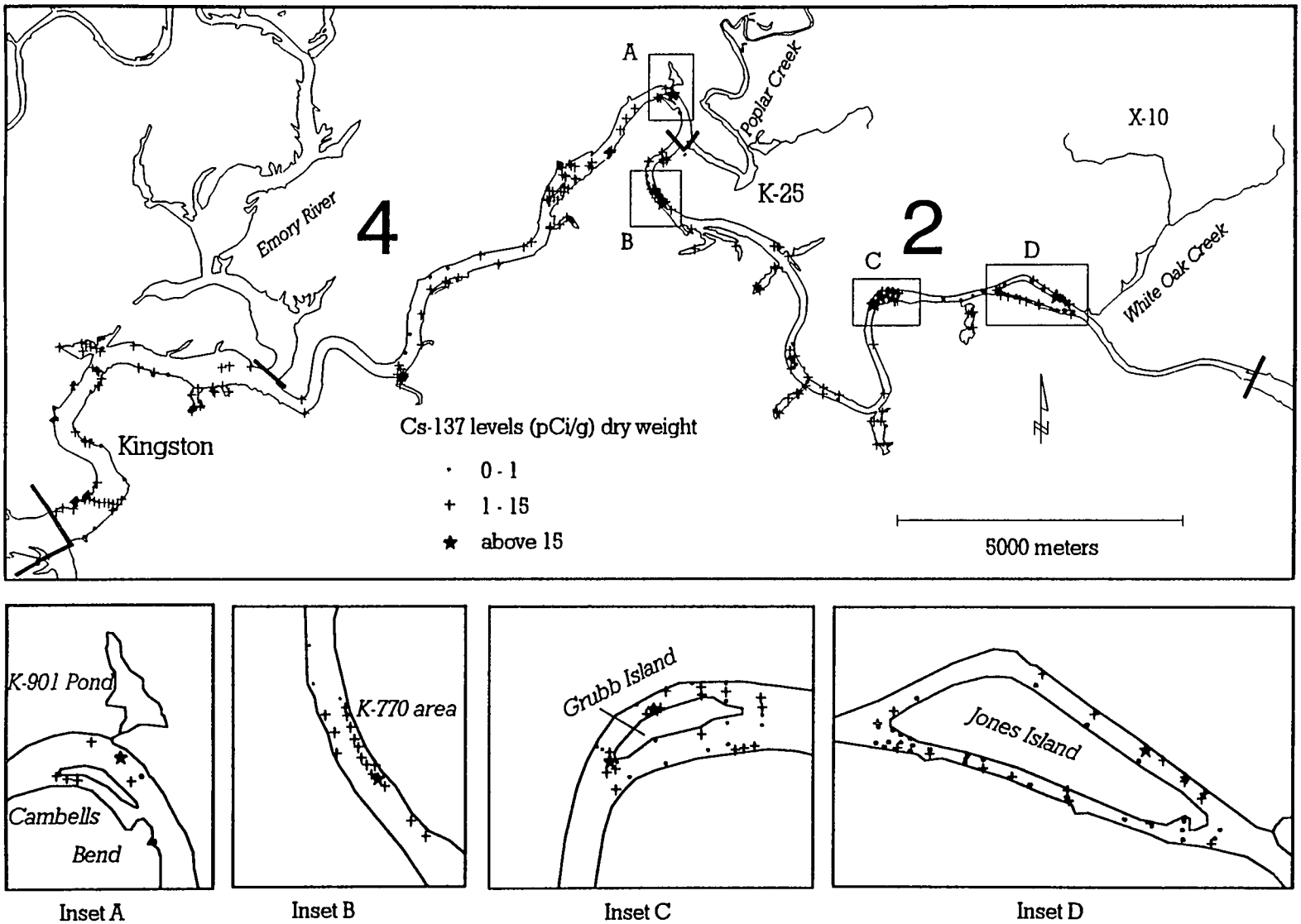


Fig. 3.32. Level of ^{137}Cs in near-shore sediment samples in the Clinch River.

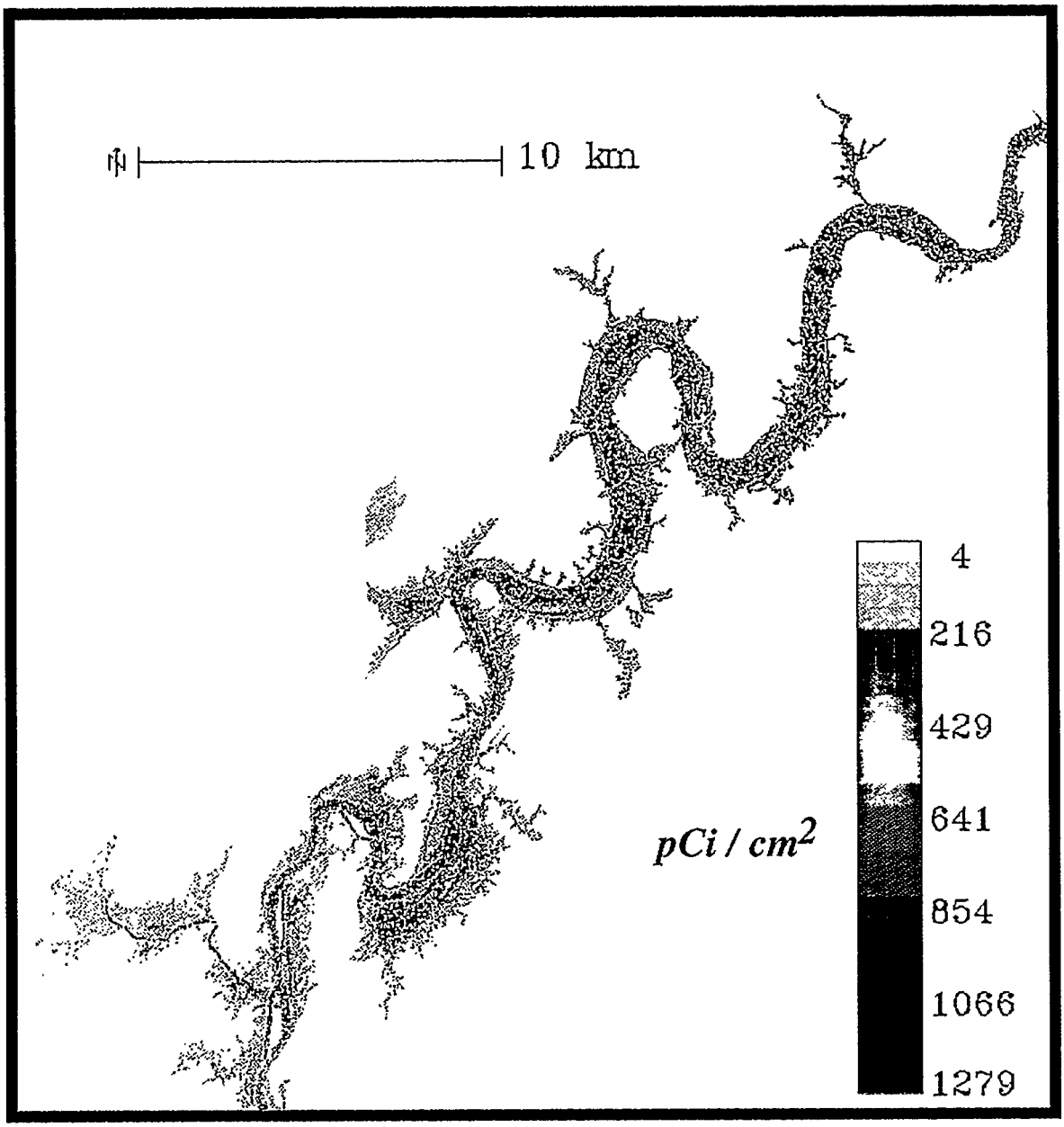


Fig. 3.33. Estimated ^{137}Cs inventory in lower Watts Bar Reservoir (Gardner et al. in press).

The mean ^{137}Cs concentration for LWBR surface sediments was within the expected range for background concentration of soils. The only high value in LWBR was actually a deep-water sediment sample in the center of the channel in the transect closest to Kingston (see Fig. 3.31). The Clinch River study reaches had higher mean ^{137}Cs concentrations than LWBR, and surface sediments constitute a potential source of ^{137}Cs to LWBR. Mean concentrations in the Clinch River were below levels of concern; however, maximum values in some individual samples were above the 15 pCi/g limit of concern (Cook et al. 1992).

Spatial Analysis of Cesium in Sediments of Watts Bar Reservoir

Gardner et al. (in press) used the ^{137}Cs data of Olsen et al. (1992), additional data regarding sediment type, and bathymetric data to analyze the distribution of sediment type and the resulting distribution of ^{137}Cs in LWBR sediments. This spatial representation (Fig. 3.33) is a significant refinement of that offered by Olsen et al. (1992) in Fig. 3.22.

The collection of additional sediment samples and the analysis of sediment types showed that there was more fine sediment present in the reservoir than originally estimated (Olsen et al. 1992). Because ^{137}Cs adsorbs to these fine particles, the presence of more fine sediment within the reservoir makes it possible to have a greater inventory of ^{137}Cs than originally estimated. However, the gradient in concentration from the confluence of the Clinch and Tennessee rivers to the dam results in a total inventory of 153.8 Ci, which is approximately half the 304 Ci originally estimated by Olsen et al. (1992).

Although the total amounts of ^{137}Cs are less than originally projected, the general conclusions of Olsen et al. (1992) were verified by this analysis. That is (1) the highest ^{137}Cs concentrations are within the channel of the old river bed, (2) there is little or no ^{137}Cs present where river scouring occurs, and (3) sediments in coves have marginal amounts of ^{137}Cs.

Effect of Sampling Bias on Estimated Cesium Inventory in LWBR

Pearson and Rose (in press) performed a simulated sampling exercise using the Olsen et al. (1992) sediment core data from LWBR (Sect. 3.2.8). This exercise determined the degree to which those data are biased by the sampling design and the effect of any such bias on the estimated ^{137}Cs inventory calculated by Olsen et al. (1992) (approximately 304 Ci). The simulation exercise used the map of Gardner et al. (in press) (Fig. 3.33) and a modified version of the Olsen et al. (1992) map (Fig. 3.22) to represent the "true" conditions in LWBR. Two sets of 5 replicates of 50 "samples" were drawn from each map. One set from each map was collected using a random sample design. The other set was collected using the design of Olsen et al. (1992) (i.e., transects across the main reservoir channel). The latter was modified slightly to incorporate an element of stochasticity into the design, to vary the actual sample location from those of the Olsen study while maintaining a similar spatial configuration.

It was demonstrated that the sampling design employed by the Olsen study overestimated the known ^{137}Cs inventory of either map. The cause was a bias in sample location for deep water sites where sediment accumulation is greatest; near-shore areas and coves are undersampled by this design. Table 3.16 summarizes the results of the simulation exercise. Unlike the maps used in this exercise, the true ^{137}Cs in LWBR is unknown. However, the known bias in the sampling design leads to the conclusion that the mean ^{137}Cs inventory as estimated by Olsen et al. (1992) would also be biased high.

The last two studies conclude that Olsen et al. (1992) overestimated the ^{137}Cs inventory in LWBR. The Olsen estimate of 304 Ci represented 91% of the decay-corrected total ^{137}Cs released from WOC. The reduced estimates above require that the remaining inventory be elsewhere. There are three possibilities to account for the remaining inventory: (1) it is downstream of LWBR; (2) it is upstream of LWBR, in the Clinch River; and (3) the estimate of the total ^{137}Cs released from WOC is inaccurate. Certainly a portion of the inventory is both upstream and downstream of LWBR. The gamma surveys of the 1950s and 1960s detected elevated gamma activity in bottom sediments throughout the Tennessee River below LWBR (Sect. 3.2.1). Numerous studies and monitoring programs have documented the presence of ^{137}Cs in sediment of the Clinch River below WOC. In neither case has an estimate of the inventory been made. Although the estimate of the total amount of ^{137}Cs released from WOC contains considerable uncertainty, it is impossible to say whether it under- or overestimates the true quantity released. In any event, the refined estimates above lead to the conclusion that LWBR contains the single greatest inventory of the ^{137}Cs released from WOC.

Table 3.16. The performance of the Olsen core sampling plan relative to random sampling for five maps of cesium inventory.[a]

	Mean Cs Inventory Estimate				95% UCL	
Map	Mean	Bias	Variance	Mean Square Error	Range	Mean
Olsen Core Samples (N=59)						
Map 1	264.9	63.5	507.8	4539.4	289–347	320.1
Map 2	226.7	18.4	278.1	616.6	302–333	314.4
Random Samples (N=50)						
Map 1	204.6	3.2	671.6	682.0	233–327	263.1
Map 2	201.9	-6.4	266.1	307.1	274–351	300.7

[a]The units for the samples were pCi/cm^2. The true mean inventory from Fig. 3.33 (Map 1) and Olsen et al. (1992) (Map 2) maps were 201.4 and 208.3 pCi/cm^2, respectively, calculated using the geographic information system. The 95% upper confidence limits (UCL) were calculated for each of the 5 replicates. The mean and range of 95% UCL observed are listed.

3.3 SUMMARY OF SITE CHARACTERIZATION

The investigations and data summarized above are sufficient to reach some weight-of-evidence conclusions regarding the nature and extent of contamination in LWBR. The following paragraphs summarize these conclusions. Quantitative estimates of current contaminant concentrations in LWBR media are provided in Appendixes A, B, and E. Non-detected contaminants are identified in Table A.2 of Appendix A. The various data sources used in these quantitative estimates are identified in Sect. 3.2.

Releases

- A wide range of CERCLA potentially hazardous substances, including inorganic, organic, and radioactive contaminants, have been released to off-site waters from the ORR. These releases are not well quantified for the early years of ORR operations, particularly for inorganic and organic contaminants. Relatively crude estimates of radionuclide releases are available for all three facilities on the ORR (DOE 1988), and estimates of the mercury released from the Y-12 Plant are also available (Wilcox 1983). Peak releases of most contaminants occurred 30 to 40 years ago, and releases have since declined dramatically. Current releases are monitored and quantities reported in the annual ORR environmental monitoring reports.

Contaminants in Surface Water

- The source of ORR contaminants to LWBR is primarily the Clinch River. It has been demonstrated that the more water-soluble radionuclides released from the ORR are rapidly diluted upon entry into the Clinch River and are quickly transported downstream to the Tennessee River, where another large dilution occurs (Churchill et al. 1965). Mass balance studies have shown that almost the entire quantity of these more soluble radionuclides are transported rapidly downstream through LWBR (Churchill et al. 1965). LWBR has an average water residence time of approximately 2 weeks. Similar studies were not identified for nonradioactive, water-soluble contaminants, but the physical mechanisms of transport are known to be the same.

  The surface water data in LWBR consist of four grab samples. Surface water monitoring data from the lower Clinch River at Brashear Island have been used to augment the LWBR ambient water data. These data are representative of ambient conditions in the Clinch River downstream of all ORR surface water inputs to the Clinch, but are biased high relative to LWBR (for ORR-specific contaminants). Contaminants at Brashear Island would be diluted by flow from the Emory River at CRM 4.0 and again by at least a factor of three upon entering the Tennessee River.

- Although data are few, Cook et al. (1992) found that organic contaminants in LWBR and Clinch River surface waters are below the limits of detection. Ambient monitoring data for the Clinch River at Brashear Island (below all ORR inputs) indicate that organic contaminants (primarily volatile organics) in surface waters are generally well below the state's ambient water quality criteria levels (Energy Systems 1993) or are below the limits of detection.

- With the exception of naturally occurring cations, inorganic analytes in water are generally in the μg/L range or less. Studies to date indicate that inorganic contaminants in LWBR surface sediments are comparable to other TVA mainstream reservoirs (TVA 1986a, Meinert 1991, Meinert and Fehring 1992, Cook et al. 1992).

- Data from Cook et al. (1992) for a few surface water samples in LWBR and the lower Clinch River indicate that concentrations of ^{90}Sr, ^{60}Co, and ^{137}Cs were all below 1 pCi/L. Data collected from the Clinch River at Brashear Island for the ORR Environmental Monitoring Program generally indicate comparable levels of other radionuclides.

Contaminants in Sediment

- Unlike the more water-soluble contaminants, particle-associated contaminants tend to accumulate in LWBR (Carrigan and Pickering 1967, Turner et al. 1984, TVA 1985a, Olsen et al. 1992, Cook et al. 1992), which is an efficient sediment trap (Trimble and Carey 1984, Ewing 1993, Brenkert et al. 1992). Sediment data indicate that ^{137}Cs is the primary anthropogenic radionuclide in sediments and is present in LWBR surface sediments at pCi/g concentrations. The current spatial distribution of ^{137}Cs in LWBR sediments is relatively well known (Figs. 3.21, 3.22, and 3.33). Most ^{137}Cs in LWBR is found buried in deep-water sediments along the old river channel, where conditions favor sediment accumulation. Little ^{137}Cs is found in near-shore sediments (Olsen et al. 1992, Cook et al. 1992) or in areas where sediment deposition does not occur. To the extent that ^{137}Cs is a tracer for other particle-reactive contaminants released from the ORR, these contaminants would also be expected to preferentially accumulate in areas of sediment deposition. A strong correlation between ^{137}Cs and mercury has been demonstrated (Turner et al. 1984, TVA 1985a, Olsen et al. 1992), which, due to their similar release chronologies and environmental fate, are correlated in three dimensions (i.e., are found together not only in the same areas of the reservoir, but at the same depth in individual sediment profiles). However, this relationship would not hold true for particle-reactive contaminants with dissimilar release histories nor for radionuclides with half-lives much shorter than that of ^{137}Cs (e.g., ^{60}Co). With respect to ^{137}Cs and mercury, peak concentrations are generally buried 40–80 cm deep in old river channel sediments of LWBR (Turner et al. 1984, TVA 1985a, Olsen et al. 1992, Cook et al. 1992).

  Of the inorganic contaminants, mercury is slightly elevated in LWBR surface sediments, but not above levels documented for other TVA reservoirs with known mercury sources. In general, the levels of most metals in LWBR are lower than in TVA reservoirs affected by urban runoff (TVA 1986a, Meinert 1991, Meinert and Fehring 1992). The concentrations of several metals (Pb, Cr, Cd, Cu, and Zn) in LWBR appear to be higher than in Clinch River sediments, indicating a source other than the ORR (Cook et al. 1992).

- Studies indicate that organic contaminants are low in LWBR sediments (TVA 1985a, 1986a, 1990a, 1991a and b; Cook et al. 1992) and Clinch River sediments (TVA 1985 and 1991, Ashwood 1986, Cook et al. 1992) and seem to be restricted primarily to relatively ubiquitous industrial contaminants (e.g., PAHs and phthalates) that are not specific to the ORR.

Contaminants in Fish

- Both water-soluble and particle-reactive contaminants can potentially accumulate in LWBR biota. The extent to which this has occurred in fish is reasonably well documented for both LWBR and the Clinch River (TVA 1985d, Cook et al. 1992, Dycus and Hickman 1988, Dycus 1989–90, Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993, TVA 1992, this report). Radionuclides in fish flesh are generally low, at the pCi/g level (Cook et al. 1992, TVA 1992). Metals that have been detected (at the low $\mu g/g$ range or less) in a significant fraction of LWBR fish include As, Cu, Ni, Hg, and Zn (Cook et al. 1992). Mercury contamination in LWBR and Clinch River fish is generally below the Food and Drug Administration (FDA) action level of 1.0 $\mu g/g$ (Cook et al. 1992). The most significant contaminants in fish flesh are PCBs, present in several species at mean concentrations at or exceeding the FDA action level of 2.0 $\mu g/g$ (Cook et al. 1992, Dycus and Hickman 1988,

Dycus 1989–90, Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). Organic contaminants other than PCBs that have been detected (at the low $\mu g/g$ range or less) in a significant fraction of LWBR fish include aldrin, chlordane, PCBs, DDT and residues, and 2-methylnapthalene (Cook et al. 1992, Dycus and Hickman 1988, Dycus 1989–1990). This PCB and chlordane problem is not restricted to LWBR, but is also well documented for Fort Loudoun, Tellico, and Melton Hill reservoirs upstream of LWBR (Dycus and Hickman 1988, Dycus 1989–90, Meinert 1991, Meinert and Fehring 1992, Fehring and Meinert 1993). The state of Tennessee currently has a fish consumption advisory in effect for selected species in each of these reservoirs (TDEC 1992). The extent to which ORR vs non-ORR sources contribute to the PCB problem in LWBR has not been quantified.

4. APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS

Section 121(d) of CERCLA specifies that remedial actions for cleanup of hazardous substances must comply with requirements or standards under federal or more stringent state environmental laws that are applicable or relevant and appropriate to the hazardous substances or particular circumstances at a site. Inherent in the interpretation of applicable or relevant and appropriate requirements (ARARs) is the assumption that protection of human health and the environment is ensured.

The purpose of this section is to supply a preliminary list of available federal and state chemical- and location-specific ARARs that might be considered for the LWBR OU RI report. Action-specific ARARs will be addressed in the FS section of this document (Sect. 8.7).

It is understood that DOE will comply with the requirements of the National Environmental Policy Act (NEPA) as specified in DOE Order 5440.1E, "National Environmental Policy Act Compliance Program." Further, DOE Order 5400.4, "Comprehensive Environmental Response, Compensation, and Liability Act Requirements," calls for integration of NEPA and CERCLA requirements for DOE remedial actions at CERCLA sites. This issue has been reaffirmed in FFA Sects. I(A)(3) and III(A)(2) and in the Secretary of Energy Notice (SEN) of February 5, 1990 (SEN-15-90), which was intended to ensure that DOE's NEPA activities are carried out in a centralized and uniform manner. Therefore, the regulations found in NEPA will not be addressed in this report as ARARs; however, the federal and state regulations protecting environmental resources that may be identified at a site during a NEPA survey are discussed.

Similarly, DOE addresses occupational safety in DOE Orders 5480.11, "Radiation Protection for Occupational Workers"; 5480.4, "Environmental Protection, Safety, and Health Protection Standards"; 5483.1A, "Occupational Safety and Health Program for Contractors at GOCO Facilities"; and 5480.9, "Construction Safety and Health Program." However, ARARs apply to those federal and state regulations that are designed to protect the environment and do not generally apply to occupational safety regulations. The EPA requires compliance with Occupational Safety and Health Administration (OSHA) standards in 40 CFR 300.150, not through the ARARs process. Therefore, neither the DOE orders related to occupational safety nor the regulations promulgated by OSHA are addressed as ARARs.

All CERCLA on-site remedial response actions need to comply with the substantive requirements of a regulation, but not the administrative requirements to obtain federal, state, or local permits (CERCLA Sect. 121(e) and FFA Sect. XXII). *Substantive requirements* pertain directly to the actions or conditions at a site; *administrative requirements* facilitate their implementation. EPA recognizes that certain of the administrative requirements, such as consultation with state agencies, reporting, etc., are accomplished through the state involvement and public participation requirements of the NCP and are useful in determining cleanup standards at the site (55 FR 8757).

4.1 CHEMICAL-SPECIFIC ARARs

"Chemical-specific requirements set health- or risk-based concentration limits or discharge limitations in various environmental media for specific hazardous substances, pollutants, or

contaminants" (53 FR 51437). These requirements generally set protective cleanup levels for the chemicals of concern in the designated media or indicate a safe level of discharge that may be incorporated when a specific remedial activity is being considered.

4.1.1 Surface Water

As noted in Sect. 2.5.2, the Tennessee Water Quality Control Board, under the Tennessee Water Quality Control Act, has classified the Tennessee River for use as domestic water supply, industrial water supply, fish and aquatic life, recreation, irrigation, livestock watering and wildlife, and navigation (Chap. 1200-4-4 of the Rules of TDEC).

The TDEC has revised its water quality standards and promulgated the current Safe Drinking Water Act maximum contaminant levels (MCLs) as water quality criteria (WQC) for domestic water supplies (effective date August 30, 1991). Table 4.1 lists these WQC. EPA has promulgated an MCL for PCBs, effective July 1992, which TDEC has proposed adopting (Proposed 1994 Revisions to Water Quality Standards, Staff Proposal, January 12, 1994). EPA has also promulgated an MCL of 4 mrem/year for radionuclides in community water systems (40 CFR 141.16); if two or more radionuclides are present, the sum of their total dose equivalent shall not exceed 4 mrem/year.

The TDEC also includes WQC for protection of recreational uses. Currently, these are human health criteria derived to protect the consumer from consumption of contaminated fish alone. TDEC has proposed to extend the recreation criteria to include the WQC for consumption of drinking water and aquatic organisms (Proposed 1994 Revisions to Water Quality Standards, Staff Proposal, January 12, 1994). For carcinogens, the TDEC WQC are based on a risk of 10^{-5} rather than the 10^{-6} risk on which the federal WQC are based. For those chemicals lacking a TDEC WQC, federal WQC derived for the protection of human health from the consumption of aquatic organisms are listed in Table 4.1. Proposed TDEC WQC for these chemicals are listed in parentheses on the table and are "to be considered" (TBC) guidance until promulgated. Table 4.1 also lists TDEC WQC for the protection of freshwater aquatic life.

National Secondary Drinking Water Standards (NSDWS) regulate contaminants that affect the aesthetic qualities of drinking water and are implemented in 40 CFR 143.3 as secondary maximum contaminant levels (SMCLs). Tennessee has promulgated an SMCL for zinc (TDEC Rules 1200-5-1.12) (Table 4.1). These regulations are designed to provide consumers with water that is aesthetically pleasing and apply to all community water systems and to those noncommunity water systems that "may be deemed necessary" by TDEC. In that context, they would not be legally applicable to cleanup of surface water, but may be considered as relevant and appropriate in the absence of an MCL.

4.1.2 Sediment

There is no legislation available governing cleanup criteria for contaminated sediments at CERCLA sites. The RCRA has addressed land disposal of treated hazardous wastes in its land disposal restrictions (40 CFR 268), and these will be applicable if RCRA-toxic waste as defined in 40 CFR 261.24 is detected in dredged sediments and disposed of following any remediation of contaminated areas. However, these restrictions will be addressed as action-specific ARARs in Sect. 8.7.

Table 4.1. Tennessee water quality criteria for designated water uses (μg/L)[a]

Chemical	Domestic water supply	Recreation[b]	Fish and aquatic life	
			CMC[c]	CCC[d]
Arsenic (total)*	50	0.14[e]	360[f]	190[f]
Beryllium	4[g]	NA[h]	NA	NA
Bis(2-ethylhexyl)phthalate*	6[g]	59	NA	NA
Boron	NA	NA	NA	NA
Cadmium	5	NA[h]	4[i]	1[i]
Chromium (total)	100	NA[h]	NA	100
Chromium (hexavalent)	NA	NA[h]	16	11
Cobalt	NA	NA	NA	NA
Copper	NA	NA	18[i]	12[i]
Cyanide	200[g]	220,000[e]	22	5.2
Fluoranthene	NA	54 (320[e])	NA	NA
Hexachlorocyclo-pentadiene	50[g]	NA (17,000[e])	NA	NA
Lead	NA	NA[h]	82[i]	3[i]
Lithium	NA	NA	NA	NA
Manganese	50[k]	NA	NA	NA
Mercury	2	0.15	2.4	0.012
n-Butylbenzyl phthalate	NA	NA	NA	NA
Nickel	100[g]	4600	1400	160
Pentachlorophenol*	1[l]	8.2[e]	20[m]	13[m]
Phenanthrene	NA	0.03	NA	NA
Polychlorinated biphenyls (total)*	0.5[n]	0.001 (0.00045[e])	NA	0.001
Selenium	50	NA[h]	20	5
Silver	NA	NA[h]	4[i]	NA
Vanadium	NA	NA	NA	NA
Zinc	5000[k]	NA	117[i]	106[i]
All man-made radionuclides	4 mrem/year[o]	NA	NA	1 rad/day[p]

[a]Rules of the TDEC, Chapter 1200-4-3.
[b]Based on a 10^{-5} risk level for carcinogen (marked with an asterisk *).
[c]CMC = criterion maximum concentration.
[d]CCC = criterion continuous concentration.
[e]Federal WQC [USEPA 1992 EPA Region IV Criteria Charts (December)]. TDEC has proposed WQC for these chemicals; these proposed WQC are listed in parentheses on the table. They may be considered TBC guidance; however, when promulgated they will be applicable (Proposed 1994 Revisions to Water Quality Standards, Staff Proposal, January 12, 1994).
[f]Arsenic III.
[g]Federal maximum contaminant level (MCL) (57 *FR* 31776, July 17, 1992).
[h]WQC withdrawn in the National Toxics Rule [57 FR 60848 (December 22,1992)]. TDEC has proposed to remove these WQC from their standards (Proposed 1994 Revisions to Water Quality Standards, Staff Proposal, January 12, 1994).
[i]Calculated for a water hardness of 100 mg/L, the approximate average hardness of the Clinch River; values presented are for the dissolved form of this metal (Suter et al. 1992).
[j]Chromium III.
[k]Secondary MCL; Rules of the TDEC, Chapter 1200-5-1.12.
[l]Federal MCL (56 FR 30266, July 1, 1991). Effective January 1, 1993.
[m]At pH 7.8.
[n]Federal MCL; 56 *FR* 3526 (January 30, 1991). Effective date July 30, 1992.
[o]Annual dose equivalent to total body or any internal organ (40 CFR 141.16).
[p]DOE Order 5400.5, "Radiation Protection of the Public and the Environment" (Proposed as 10 CFR 834, 58 *FR* 16268, March 25, 1993.

The EPA has presented an approach for deriving cleanup levels for PCBs in sediments at Superfund sites (EPA 1990). This approach results in sediment quality criteria (SQC) as a function of organic carbon concentration, and is meant to protect wildlife that consume freshwater benthic species. Values estimated using this approach are TBC guidance, not ARARs.

The EPA has proposed SQC for protection of benthic organisms (59 FR 2652, January 18, 1994). EPA plans to publish, at a later date, the general regulatory role that these criteria will play in a Contaminated Sediment Management Strategy (CSMS). The preamble to the proposed rule states that these SQC should not be used as mandatory cleanup levels due to many factors, including the potential for causing environmental damage through disturbing contaminated sediments (59 FR 2654). These SQC should be used as a tool to evaluate sites to determine the need for remediation, and it is for this purpose that the SQC are presented here as TBC. Table 4.2 lists these sediment criteria.

Table 4.2. Sediment Quality Criteria[a]

Priority Pollutant	Sediment Criteria ($\mu g/g_{oc}$)[b]
Acenapththene	130
Dieldrin	11
Endrin	4.2
Fluoranthene	620
Phenanthrene	180

[a]Proposed guidance (59 FR 2652, January 18, 1994).
[b]Micrograms chemical per gram organic carbon (oc); based on sediments with 0.2% organic carbon or more. These criteria should not be used for sediments with less than 0.2% oc.

4.2 RADIATION PROTECTION STANDARDS

Very few applicable standards are available for the cleanup of radioactive contamination at CERCLA sites. The Atomic Energy Act (AEA) of 1954 and its amendments delegated authority for control of nuclear materials to DOE, the U.S. Nuclear Regulatory Commission (NRC), and EPA.

DOE has proposed radiation protection requirements for the public and the environment for all sources of exposure and from all DOE sources of radiation [10 CFR 834, Notice of Proposed Rulemaking (NPRM) 58 FR 16268, March 25, 1993; final rule expected July 1994]. The proposed rule, when final, would codify the requirements of DOE Order 5400.5. The radiation exposure limits as defined in proposed 10 CFR 834, "Radiation Protection of the Public and the Environment," are an effective dose equivalent of 100 mrem/year from all exposure pathways and all sources of radiation from the activity and from all other sources (excluding dose from radon and its decay products, diagnostic or therapeutic medical radiation exposures, consumer products, and natural background). In addition, effluent releases to surface water must not result in exposures to aquatic organisms that exceed an absorbed dose of 1 rad/d. The overriding principle of the regulation is that all releases of radioactive material shall be "as low as

reasonably achievable" (ALARA). Until this rule is promulgated, it or DOE Order 5400.5 may be considered TBC guidance.

DOE is exempt from the licensing requirements of the TDEC Radiation Protection Standards for on-site activities [TDEC 1200-2-10-.06(1)]. However, the TDEC requirements for packaging and transportation of radionuclide-containing materials above exempt quantities [as listed in Schedule RHS 8-3, TDEC 1200-2-10-.04(3)] will be applicable for transport of any sediments containing radioactivity from the LWBR OU to the ORR for treatment or disposal. These requirements are summarized in Section 8.7.

4.3 LOCATION-SPECIFIC ARARs

Location-specific requirements set restrictions on the concentration of hazardous substances or the conduct of activities solely because they are in special locations (53 FR 51394). These location-specific requirements are contingent on specific actions that may or may not occur. Specific remedial actions are outlined in Chapters 8 and 9.

4.3.1 Wetlands and Floodplains

The Clinch River drainage basin includes a number of wetland areas that may require protection if identified as federally delineated wetlands areas. If any remedial alternatives are selected that would affect wetlands, consideration should be given to Executive Order 11990, 10 CFR 1022, CWA Sect. 404, 40 CFR 230, and 33 CFR 323 for applicable requirements. In addition, regulations protecting floodplains are found in Executive Order 11988, 40 CFR 264.18(b), and 10 CFR 1022. Table 4.3 summarizes the substantive portion of these regulations.

4.3.2 Aquatic Resources

If any remedial actions result in the control or structural modification of a natural stream or water body, the provisions found in the Fish and Wildlife Coordination Act [16 United States Code (USC) 661 *et seq.*] may be applicable. In addition, the substantive requirements of the TDEC Aquatic Resource Alteration regulations found at Chapter 1200-4-7 may be applicable. Even though permits are not required for CERCLA activities (CERCLA Sect. 121(e) and FFA Sect. XXII), consultation with TDEC is recommended to ensure that any remedial actions do not result in unacceptable water quality impacts.

4.3.3 Endangered and Threatened Species

Federal- or state-listed threatened or endangered (T&E) species for LWBR are identified in Tables 2.7 and 2.8. If any remedial actions in the LWBR OU affect any federally listed endangered or threatened species, the provisions found in the Endangered Species Act of 1973 (16 USC 1531 *et seq.*), 50 CFR 492, and the Fish and Wildlife Coordination Act (16 USC 661 *et seq.*) may be applicable. If any proposed actions affect state-listed endangered or threatened animal species, the Tennessee Non-Game and Endangered or Threatened Wildlife Species Conservation Act of 1974 (TCA 70-8-101 *et seq.*) may provide ARARs. It is unlawful for any person to knowingly destroy the habitat of "wildlife in need of management" (TWRC 86–29, 1991).

4.3.4 Archaeological and Historical Resources

There have been no known cultural resource surveys of the entire study area; however, there have been surveys over the years that encompassed portions of the Clinch River (e.g., Webb 1938, McNutt and Fischer 1960, McNutt and Graham 1961). Before the start of any ground-disturbing activities within the study area, consultation should be initiated with the Tennessee State Historic Preservation Officer and Advisory Council on Historic Preservation, pursuant to Section 106 of the National Historic Preservation Act (16 U.S.C. 470f) and its implementing regulations (36 CFR 800). The results of this initial consultation will dictate the need for (1) additional archeological and/or historical survey work and (2) the ratification and implementation of a Memorandum of Agreement prepared pursuant to 36 CFR 800.5(e)(4). Compliance with all other cultural resources laws and regulations would be coordinated through this consultation process, initiated under Section 106 of the National Historic Preservation Act.

Table 4.3. Potential location-specific applicable or relevant and appropriate requirements for Clinch River RI

Location characteristic(s)	Operating condition(s)	Requirement(s)	Citation(s)
	Wetlands		
Presence of wetlands as defined in Executive Order 11990 Sect. 7(c) and 40 CFR 6, Appendix A Sect. 4(j)	Agency action that involves - acquiring, managing, and disposing of lands and facilities; - providing federally undertaken, financed, or assisted construction and improvements; - conducting federal activities and programs affecting land use.	Whenever possible, actions must avoid or minimize adverse impacts on wetlands and act to preserve and enhance their natural and beneficial values. New construction in wetlands areas should be particularly avoided unless there are no practicable alternatives. Wetlands protection considerations shall be incorporated into planning, regulating, and decision-making processes.	Executive Order 11990 40 CFR 6.302(a) 10 CFR 1022
Presence of wetlands as defined in 40 CFR 230.3(t) and 33 CFR 328.3(b)	Action involving discharge of dredge or fill material into wetlands	Action to avoid degradation or destruction of wetlands must be taken to the extent possible. Discharges for which there are practical alternatives with less adverse impacts or those that would cause or contribute to significant degradation are prohibited. If adverse impacts are unavoidable, action must be taken to enhance, restore, or create alternative wetlands.	Clean Water Act § 404 40 CFR 230 33 CFR 323
	Floodplains		
Within "lowland and relatively flat areas adjoining inland and coastal waters and other floodprone areas such as offshore islands, including, at a minimum, that area subject to a one percent or greater chance of flooding in any given year" [Executive Order 11988 Sect. 6(c) and 40 CFR 6, Appendix A Sect. 4(d)]	Action that involves - acquiring, managing, and disposing of lands and facilities; - providing federally undertaken, financed, or assisted construction and improvements; - conducting federal activities and programs affecting land use.	Action shall be taken to reduce the risk of flood loss; minimize the impact of floods on human safety, health, and welfare; and restore and preserve the natural and beneficial values of floodplains. The potential effects of actions in floodplains shall be evaluated and consideration of flood hazards and floodplain management ensured. If action is taken in floodplains, alternatives that avoid adverse effects, incompatible development, and minimize potential harms shall be considered.	Executive Order (EO) 11988 10 CFR 1022

Table 4.3 (continued)

Location characteristic(s)	Operating condition(s)	Requirement(s)	Citation(s)
	Floodplains (continued)		
Within 100-year floodplain	Treatment, storage, or disposal facility RCRA[a]-defined listed or characteristic hazardous waste (40 CFR 261) or RCRA-permitted facility	Facility must be designed, constructed, operated, and maintained to prevent washout of any hazardous waste by 100-year flood.	40 CFR 264.18(b)
	Aquatic Habitats		
Within area encompassing or affecting waters of the state of Tennessee as defined in TCA[b] 69-3-103(32) and the presence of wildlife or aquatic life	Action involving the discharge of any pollutants into the waters of the state [see TCA 69-3-103(18) and (21) for noninclusive list]	Discharge of "substances" that "will result or will likely result in harm, potential harm, or detriment to the health of animals, birds, fish, or aquatic life" is prohibited.	Tennessee Water Quality Control Act of 1977 (TCA 69-3-101 *et seq.*) Stream Use Classifications (TDEC[c] Rules 1200-4-4)
Within area affecting stream or river and presence of fish or wildlife resources	Action that results in the control or structural modification of a natural stream or body of water	The effects of water-related projects on fish and wildlife resources must be considered. Action must be taken to prevent, mitigate, or compensate for project-related damages or losses to fish and wildlife resources. Consultation with the FWS[d] and/or appropriate state agency is recommended for on-site actions.	Fish and Wildlife Coordination Act (16 USC 661 *et seq.*)
Location encompassing aquatic ecosystem with dependent fish, wildlife, other aquatic life, or habitat	Action involving the discharge of dredge or fill material into aquatic ecosystem	Degradation or destruction of aquatic ecosystems must be avoided to the extent possible. Discharges that cause or contribute to significant degradation of the water of such ecosystems are prohibited.	Clean Water Act § 404 40 CFR 230 33 CFR 320-330
Within an area affecting a stream or river	Action that involves aquatic resource alterations	Must comply with the substantive requirements of the aquatic resource alteration permitting process	TDEC Rules 1200-4-7

Table 4.3 (continued)

Location characteristic(s)	Operating condition(s)	Requirement(s)	Citation(s)
		Endangered, Threatened, or Rare Species	
Presence of endangered or threatened species or critical habitat of such species as designated in 50 CFR 17, 50 CFR 226, or 50 CFR 227	Action that is likely to jeopardize species or destroy or adversely modify critical habitat	Actions that jeopardize species/habitat must be avoided or appropriate mitigation measures taken. Consultation with DOI[e], FWS, and/or state agencies, as appropriate, to ensure that proposed actions do not jeopardize the continued existence of the species or adversely modify or destroy critical habitat is recommended for on-site actions.	Endangered Species Act of 1973 (16 USC 1531 *et seq.*) 50 CFR 402 Fish and Wildlife Coordination Act (16 USC 661 *et seq.*)
Presence of endangered or threatened species or critical habitat (see above citation) of same within an aquatic ecosystem as defined in 40 CFR 230.3(c)	Action involving discharge of dredge or fill material into aquatic ecosystem	Dredge or fill material shall not be discharged into an aquatic ecosystem if it would jeopardize such species or would likely result in the destruction or adverse modification of a critical habitat of the species.	Clean Water Act § 404 40 CFR 230.10(b)
Presence of Tennessee state-listed species "in need of management"	Action affecting such species	No person may knowingly destroy the habitat of such species	TWRC[f] Proc. 86–29 (1991)
		Archeological and Historical Resources	
Presence of archeological and historical resources on public land	Action that would affect archeological and/or historical resources	Consultation with Tennessee State Historic Preservation Officer and Advisory Council on Historic Preservation before the initiation of any ground-disturbing activities. Steps must be taken to mitigate adverse effects on archeological and historical resources.	National Historic Preservation Act of 1966 (16 USC 470)

[a]RCRA = Resource Conservation and Recovery Act.
[b]TCA = Tennessee Code Annotated.
[c]TDEC = Tennessee Department of Environment and Conservation.
[d]FWS = Fish and Wildlife Service.
[e]DOI = Department of Interior.
[f]TWRC = Tennessee Wildlife Resources Commission

5. HUMAN HEALTH RISK ASSESSMENT

The baseline human health risk assessment quantifies the risk associated with human exposure to contaminants in LWBR. This risk assessment consists of six sections: the identification of data sources (Sect. 5.1); the exposure assessment (Sect. 5.2); the toxicity assessment (Sect. 5.3); the risk characterization (Sect. 5.4); the quantitative uncertainty analysis (Sect. 5.5); and the summary and conclusions (Sect. 5.6). The results of the baseline risk assessment are used in Chaps. 7 through 9 to develop remediation goals and remedial alternatives for LWBR. The purpose of this baseline human health risk assessment is to determine whether contaminants that have been detected in fish, sediment, and water from LWBR are present at concentrations that could have an adverse effect on the health of an individual who regularly uses the resources of LWBR. The baseline human health risk assessment evaluates the health risks that would exist if current institutional controls were to lose effect and no remediation were to occur.

The FFA parties (EPA, TDEC, and DOE) have agreed that this will be a focused risk assessment. The focus is on contaminants of potential concern (COPCs) that were identified in previous human health risk screening reports. In their screening assessment, Hoffman et al. (1991) identified PCBs in fish and external exposure to radiation from ^{137}Cs in dredged sediments as exposure pathways and contaminants that are of potential concern. Cook et al. (1992) confirmed the above exposure pathways and contaminants of concern based on CRRI Phase 1 data. In deciding to split off the LWBR OU (Sect. 1.3), the FFA parties agreed that existing data (Sect. 3.2) should be sufficient to conduct a human health baseline risk assessment for LWBR.

Data sets used in this risk assessment are identified in Sect. 5.1. The quality of the CRRI Phase 1 data is well documented (Holladay et al. 1993); however, QC records are not available for some historical data. Methods and QA/QC measures for those studies from which data were used are discussed in Sect. 5.1. Contaminants that were analyzed for, but for which all concentrations for a medium were below detection limits, were not considered in the assessment (Table A.2). Because there were only four water samples collected in LWBR, surface water monitoring data collected in the Clinch River immediately downstream of the ORR were used in a separate assessment to supplement the results for LWBR water. Volatile organic compounds were not addressed in this risk assessment because any such compounds released from the ORR would have volatilized before reaching LWBR. As agreed by the FFA parties, modeling was not used to estimate concentrations of contaminants in water, sediment, or fish. Historical and monitoring data were used either separately or in combination with CRRI Phase 1 data to conduct the baseline risk assessment.

5.1 IDENTIFICATION AND EVALUATION OF DATA SOURCES

Because of the temporal dynamics of the environmental media used in the risk assessment, a decision was made to restrict data to those that were representative of current conditions. Non-CRRI data for contaminants in water were restricted to data collected since 1992. Non-CRRI fish contaminant data were restricted to data collected since 1988. Sediment data were restricted to data collected since 1985. The retention of contaminants in areas of stable sediment deposition makes the temporal constraints of representativeness less of a concern for sediment. All data for radionuclides in sediment were decay-corrected to January 1, 1994, before they were used in this risk assessment.

The data sets that were used in the baseline human health risk assessment and their sources are shown in Table 5.1. These data were separated into two categories: (1) a "detected contaminants" category, which consisted of contaminants for which at least one measurement for a medium was above the analytical detection limit, and (2) a "nondetected contaminants" category, which consisted of contaminants for which all measurements for a medium were below the level of detection. Only the data in the detected contaminants data base were used in the risk assessment (Table A.1). The contaminants in the nondetected data base are listed in Table A.2.

5.1.1 Water Data

Of the data available for the risk assessment, the least amount of data are available for surface water. Four surface water grab samples from LWBR collected in 1989, and six grab samples collected from the Clinch River at Brashear Island were used to assess risk. There are a number of factors to be considered when assessing whether these data can be used for the purposes of assessing baseline risk and, ultimately, reaching remedial decisions.

The boundaries of the LWBR were originally delineated based primarily on known contaminant accumumulation in sediment and fish. Based on monitoring data at release points on the reservation, the surface water medium in LWBR was never considered threatened by the quantities of materials released from the ORR. Current releases to surface water are monitored and demonstrate compliance with ambient criteria in the Clinch River; ORR-specific contaminants would only be further diluted in LWBR. Previous screening-level risk assessments have not identified any high-priority contaminants in LWBR surface water (Cowser and Snyder, 1966; Hoffman et al. 1991, Cook et al. 1992).

From a practical perspective, the size and the rapid turnover of the surface water in LWBR would preclude any action-based remedial alternatives. The only remedial alternatives that would be practical would be the posting of warning signs. The true solution to any unacceptable levels of contaminants in LWBR is to prevent their initial release. As mentioned above, current releases from the ORR are monitored and do not exceed applicable criteria limits. Nonetheless, the ongoing environmental restoration program at the ORR will further reduce releases and prevent the potential for future releases thus, the ultimate remedy is being pursued on the ORR, not in LWBR.

Although the sediments do contain contaminants, the particular contaminants present are sorbed to particles and do not partition to the water column in any appreciable quantity (see sect. 3.2.2 and 3.2.8). Resuspension of sediment particles may occur to a limited degree, but most of LWBR is a zone of particle deposition, not resuspension; evidence is provided by the fact that peak concentrations are buried at depth, can be correlated with known release histories, and indicate areas of sediment accumulation that have remained stable for 40 years.

A potential problem of having few data for the risk assessment are that periodically elevated releases of contaminants to the water may be missed. However, the risk assessment is based on a relatively long-term exposure, particularly for carcinogens. The exposure is based on the average concentration of contaminants in water, not peak concentrations. Although LWBR is certainly a heterogeneous system, the release of contaminants from the ORR to the surface water is typically not. Accidental releases have certainly occurred in the past as a result of process or waste treatment equipment failure. However, such releases are rare and contribute little to the average concentration over a one or thirty year exposure duration. The period of contaminants

Table 5.1. Sources of data for the Lower Watts Bar Reservoir Human Health Risk Characterization

Media	Sources	Type samples	Analyses
Surface water	CRRI Phase 1 (Sect. 3.2.16)	4 grab samples	Inorganics, organics, radionuclides
	Oak Ridge Reservation Environmental Monitoring (Sect. 3.2.13)	6 bimonthly grab samples	Inorganics, radionuclides, volatile organics, pesticides/PCBs
Fish fillets	Consolidated data from CRRI Post-Phase 1 (Sect. 3.2.16)	Bluegill	Inorganics (20), radionuclides (4)
		Largemouth bass	Inorganics (5), organics (21), radionuclides (5)
		Catfish	Inorganics (2), organics (22–130), radionuclides (19)
	TVA 1988–1992 (Sect. 3.2.10)	Striped/hybrid bass	Organics (10–66)
	CRRI Phase 1 (Sect. 3.2.16)		(~number of samples shown in parentheses)
Near-shore surface sediment	CRRI Near-shore Sediment Study (Sect. 3.2.16)	352 grab samples	Radionuclides
	TVA public use areas (Sect. 3.2.9)	11 grab samples	Inorganics, radionuclides
Deep-water sediment	Consolidated data from: CRRI Phase 1 (Sect. 3.2.16)	Cores	Inorganics (6), organics (4), radionuclides (2-9)
	TVA Instream Contaminant Study (Sect. 3.2.6)	4 cores	PCBs, Hg, and gamma emitters
	Olsen et al. 1992 (Sect. 3.2.8)	Cores	^{137}Cs (60) and Hg (4) (~number of samples shown in parentheses)

originating from waste areas is potentially higher during the winter, due to increased surface water runoff and groundwater stormflow (see sect. 2). However, concentrations are not necessarily higher because of the dilution afforded by the general increase in runoff throughout the LWBR watershed.

Notwithstanding the above arguments, in order to better characterize the risk from ORR surface water contaminants, a year's worth of monitoring data from the Clinch River, downstream of all ORR inputs, was used to assess human health risk. These data represent concentrations as yet undiluted by either Emory River or Tennessee River water. The data are more temporally representative than are the CRRI LWBR data. The risk assessment based on these data represent a "worst-case" scenario with respect to ORR contaminants that might be found in LWBR.

A long-term monitoring program will be implemented in LWBR following implementation of the ROD. Surface water data collected during this program will be evaluated annually to determine if ambient contaminant concentrations in LWBR continue to meet acceptable water quality criterai. A screening-level risk assessment will be performed at the five-year review of the ROD to further ensure that no additional remedial action is required to be protective of human health.

5.1.2 Sediment Data

The near-shore surface sediment data were limited for all analytes except ^{137}Cs and ^{60}Co, for which more than 350 samples were analyzed. These data came from Phase 1 (Cook et al. 1992), post-Phase 1, and TVA public use and water intake area samples (TVA 1991a). Table 5.1 shows the types of samples and analyses that were performed.

Sediment cores were used to measure contaminant concentrations in deep water sediments. These data were from several studies, each with different purposes and sampling strategies (Sect. 3); therefore, the statistical analysis of the collective sediment data base was limited. Sixty cores by Olsen et al. (1992), three cores from the CRRI Phase 1 (Cook et al. 1992), and two cores from the TVA Instream Contaminant Study (TVA 1986b) were combined into one data set for the purpose of assessing the risk in LWBR. As described in Sect. 3, only the CRRI Phase 1 samples were analyzed for a wide selection of contaminants; the TVA samples were analyzed for metals and PCBs only; and Olsen's cores were analyzed primarily for ^{137}Cs (a few cores were analyzed for ^{60}Co and mercury). However, ^{137}Cs is considered an indicator of the accumulation patterns of particle-associated contaminants (Olsen et al. 1992).

5.1.3 Fish Data

The fish tissue data were the most complete and representative data available for use in the risk assessment. The CRRI data were sufficient for most contaminants, but nonetheless were supplemented by several years of TVA data. Because the sampling plans for the collection of fish fillets in each of these studies were very similar, and because multiple locations provide a better estimate of the average fish in the LWBR, the results of the studies were compiled into one data set for fish. Summary statistics were then calculated for each species of fish. Species were evaluated separately because some species are known to bioaccumulate certain contaminants more readily than other species (e.g., PCBs in catfish).

5.1.4 Quality Assurance Evaluation of Data Sources

The quality assurance (QA) criteria for using data quantitatvely in this risk assessment were (1) the quality of the data had to have been assessed by the investigators collecting the data, and (2) these quality-assured data were subsequently published in the publicly available literature. A special effort (Blaylock et al. 1994) was made to ensure that the reported quantitation limits from these studies were suffuciently low for use in a baseline risk assessment. The standard for comparison was the quantitation limits used in the CRRI Phase 1 study, which in turn are based on EPA Contract Laboratory Program quantitation limits. A brief summary of the QA procedures employed in each study from which data were used in the risk assessment is presented in the following paragraphs.

TVA Instream Contaminant Study

Data from LWBR sediment cores collected by TVA (1985a and b) were used in this baseline risk assessment. The complete QA program for the TVA Instream Contaminant Study is described by TVA (1985c). The Qa Program included the use of spiked samples for evaluating the accuracy laboratory analyses, the use of duplicate samples to evaluate the precison of the analyses; blind field duplicates and blind laboratory duplicates were also used to evaluate overall precision. Overall data accuracy was evaluated through the use of standard reference materials supplied by the National Bureau of Standards.

Approximately 5 percent of all samples were split with the EPA Region IV laboratory or the EPA Eastern Environmental Radiation Facility (TVA 1986b).

Olsen et al. (1992) - Transport and Accumulation of Cesium-137 and Mercury in Watts Bar Reservoir

Data from the sediment cores of Olsen et al. (1992) were used in the risk assessment. The accuracy of the data were assessed through the use of standard reference materials supplied by the National Bureau of Standards and EPA. The laboratory also routinely participated in the QA program administered by EPA's Environmental Monitoring Systems Laboratory in Las Vegas; samples received from the EPA lab were routinely analyzed by gamma ray spectrometry as cross-checks or blind samples.

Recreation and Intake Area Sampling (TVA 1991a)

The radionuclide and inorganic contaminant surface sediment data collected by TVA were used here to assess risk. TVA's standard laboratory QA program is described by TVA (1991b) and requires the use of one replicate, one blank, one spike, and one surrogate sample per 10 field samples. The agency states that all laboratory analyses were performed in accordance with approved quality assurance and quality control procedures.

TVA Fish Tissue Studies

The TVA fish tissue data are also collected under the standard TVA quality assurance program referenced above. In addition, in 1989, TVA participated in an extensive inter-laboratory split-sample program. Participating laboratories included were from TVA, TDEC, EPA Region IV, and ORNL.

Oak Ridge Reservation Environmental Monitoring Program

Surface water data from the ORR environmental monitoring program were used here to supplement the surface water data available from LWBR. The Energy Systems laboratory quality assurance program is described in the recent annual monitoring reports (e.g., Energy Systems 1993). The Energy Systems analytical laboratories have well-established QA/QC programs. These laboratories conduct extensive internal QC programs, perticipate in several external QC programs, and statistically monitor performance on a routine basis. Certified standard reference materials are routinely used in the calibartion of insturments, to standardize methods, and as spike additions for recovery tests. Laboratory performance is also routinely monitored through the use of single, blind, and control samples. Because much of the data are used to assess compliance with environmental law, Energy Systems laboratories participate in a number of external QA/QC programs designed to ensure that the data are of accepatable quality for this purpose. All Energy Systems laboratories at the ORR participate in the EPA's Contract Laboratory Program must meet the stringent QA/QC requirements of that program to remain a CLP laboratory, including satisfactory evaluation of quarterly performance samples. Energy Systems laboratories are certified by the State of Tennessee to conduct drinking water analyses; the QA/QC requirements of that program (including succesful analyses of QC samples) must be met to remain certified. Other external QA/QC programs in which Energy Systems laboratories participate include the Proficiency Analytical Testing Program administered by the National Institute of Safety and Health, the Discharge Monitoring Qualtiy Assurance Study administered by EPA and required of NPDES major permit holders, the EPA Intercomparison Radionuclide Control Program, and the DOE's Environmental Measurements Laboratory Quality Assessment Program.

Clinch River Remedial Investigation

The Phase 1 CRRI data for all media are used to assess human health risks. The quality assurance program is described and the quality of the data are documented by Holladay et al. (1993). The CRRI Phase 1 QA program included the use of field duplicates to assess the overall precison of the sample data. Data accuracy was assessed through the use of laboratory control samples, matrix spikes, surrogate spikes, and performance evaluation samples. To ensure that the representativeness of the data were not compromised, standard analytical methods and proper preservation techniques were followed, proper containers were used to collect and ship samples, sample holding times were adhered to, and field and laboratory blanks and equipment rinsate samples were all employed. The Phase 1 raw data were assessed through a data validation process. Organic data were assessed using the criteria of EPA (1988a); inorganic data were assessed using the criteria of EPA 1988b). EPA did not have validation criteria for radionuclide analyses, therefore the data were validated using the criteria of SAIC 1991).

The CRRI Phase 1 data, were reported by Cook et al. (1992). These data are fully validated and of known and documented quality.

One of the stated goals of the CRRI Phase 1 investigation was to verify the range of contaminant concentrations reported in earlier scoping studies. Cook et al. (1992) concluded that the Phase 1 data do verify the previously reported concentrations, providing additional confidence in the non-CRRI data.

5.2 EXPOSURE ASSESSMENT

Exposure, in the context of human health risk, is defined as the direct contact of a person with a chemical or physical agent. To determine whether a given exposure might cause adverse affects to human health, the magnitude, frequency, and duration of exposure for the populations identified at the site must be quantified. This task can be accomplished by first determining the concentration of each contaminant in the exposure medium (water, sediment, food, air) and then calculating pathway-specific intakes. In the present risk assessment, intake is a measure of the mass of the contaminated media that is in contact with the body (skin, lungs, gut) per unit body weight per unit time and it is pathway specific. Pathways include ingestion, inhalation, dermal, and external for radionuclides. Pathway-specific intakes are calculated using equations (Table A.6 through A.15) that include parameters for contact rate, exposure frequency, exposure duration, body weight, and exposure averaging time. Multiplying the pathway-specific intake by the concentration of the contaminant in the exposure medium gives an exposure value which is compared to a toxicity value that is a measure of the potential for adverse health effects(Appendix B, Tables B.1 through B.16).

EPA (1989) recommends using the arithmetic average (mean) of the concentration of a contaminant in a medium as the exposure concentration. Because it is not possible to know the true mean concentration of a contaminant, the 95% upper confidence limit (UCL) of the mean is used as a conservative estimate of the true mean. Measured contaminant concentrations were not available for all exposure media, therefore, mathematical models were used to estimate the concentrations. For media with limited sample data or extreme variability in the measured or modeled concentrations, the UCL can be greater than the highest measured or modeled concentration. In these cases, the maximum measured value is used as the exposure concentration instead of the UCL. It should be noted, however, that the true (but unknown) mean could be higher than the maximum value, especially if the most contaminated portion of the site has not been sampled. This issue will be revisited in the uncertainty analysis (Sect. 5.5). Measured concentrations of detected contaminants for all media are presented in Table A.1.

EPA's standard default exposure factors (OSWER Directive 9285.6-03) are used in calculating the pathway-specific intakes where appropriate. Knowledge of site-specific conditions and receptor activity is used when guidance is not available or when professional judgement deems necessary. The default values are typically either the average value, the 50th percentile value, or the 90th percentile value. Default exposure factors are given in Table A.5 and are based on EPA guidance for residential exposure. Each exposure factor used in calculating an intake is referenced in the pathway-specific intake tables (Tables A.6–A.15).

The following sections briefly characterize the exposure setting (Sect. 5.2.1) and describe the six exposure scenarios (Sect. 5.2.2) evaluated in this baseline risk assessment.

5.2.1 Characterization of Exposure Setting

Both the Clinch and Tennessee rivers flow through agricultural and industrial areas where contaminants from both point and nonpoint sources can enter the water. LWBR serves as an integrator of waterborne substances that are present in both the Clinch and Tennessee rivers when they reach LWBR. Once these substances enter LWBR, they may be found in water, sediment, and biota. The fate of a substance depends not only on the flow rate of the water but also on the physical and chemical properties of the substance. Dissolved substances are usually flushed

through the reservoir in a matter of weeks, whereas particle-associated substances may accumulate in the sediment and remain indefinitely.

As noted in Chap. 2, LWBR provides a variety of recreational opportunities, such as fishing, hunting, boating, swimming, water skiing, and camping. The greatest participation in these activities occurs in the summer, but because the area has a temperate climate, some activities are pursued year round. Many resorts and recreational areas are located on or near LWBR, and they attract a substantial number of visitors from out of the area. However, most people who regularly use LWBR for recreational purposes live within 50 miles. The present risk assessment focuses on local inhabitants who regularly engage in activities where they could come in contact with or utilize fish, sediment and/or water from LWBR.

All three counties that border LWBR are rural; populations of each are provided in Table 2.1. Between 1980 and 1991, the population of Meigs and Rhea coutnies increased by 11.1% and 2.2%, respectively, whereas Roane County's population decreased by 1.4%. The availability of jobs is primarily responsible for the changes in population. Although the population of three counties is not growing rapidly, the number of homes, both primary and secondary, that are being built on or near the reservoir is increasing.

Watts Bar Reservoir was impounded January 1, 1942. The land surrounding the reservoir is currently used for residential, farming, industry, resort, and recreational purposes. The city of Kingston's municipal drinking water source is located on the Tennessee River at TRM 568.4, which is a short distance above its confluence with the Clinch River. The city of Rockwood obtains its drinking water from Caney Creek, and the community of Spring City's municipal water supply is 4.0 miles up the Piney River, a tributary that flows into LWBR at TRM 532.3. In addition, there could be some private use of LWBR water for drinking water purposes. EPA (1989) recommends that a human health risk assessment be conducted on unfiltered drinking water when a large water source is readily available. However, this is a conservative calculation of exposure because most individuals in the vicinity of LWBR who do not have private wells drink treated water from which most particle-reactive contaminants are removed.

The elevation of LWBR water is dropped to a winter pool level in October for flood control purposes and remains at this level until April (Fig. 2.10). During this time, sediments in shallow-water areas, often referred to as mud flats, are exposed and utilized by the public for strolling, artifact collection, and other recreational purposes. Use of reservoir water for irrigation purposes is limited but could increase in the future.

5.2.2 Exposure Scenarios

Six exposure scenarios were evaluated in this baseline risk assessment: ingestion of untreated surface water, fish consumption, shoreline use and exposure to near-shore sediment, swimming, irrigation, and the use of dredged spoils for agricultural purposes (Figure 5.1). Another potential pathway for humans to be exposed to contaminants in LWBR is through the consumption of waterfowl from the area. This exposure pathway was not included in this risk assessment because studies have shown that, except for a few Canada geese (*Branta canadensis*) that spent time on some small ORNL waste ponds with relatively high levels of radioactivity in the water, the concentration of contaminants in waterfowl that reside on or near the ORR do not pose a health problem (BMAP 1992). Measures are currently in place to keep waterfowl from visiting the radioactive waste ponds.

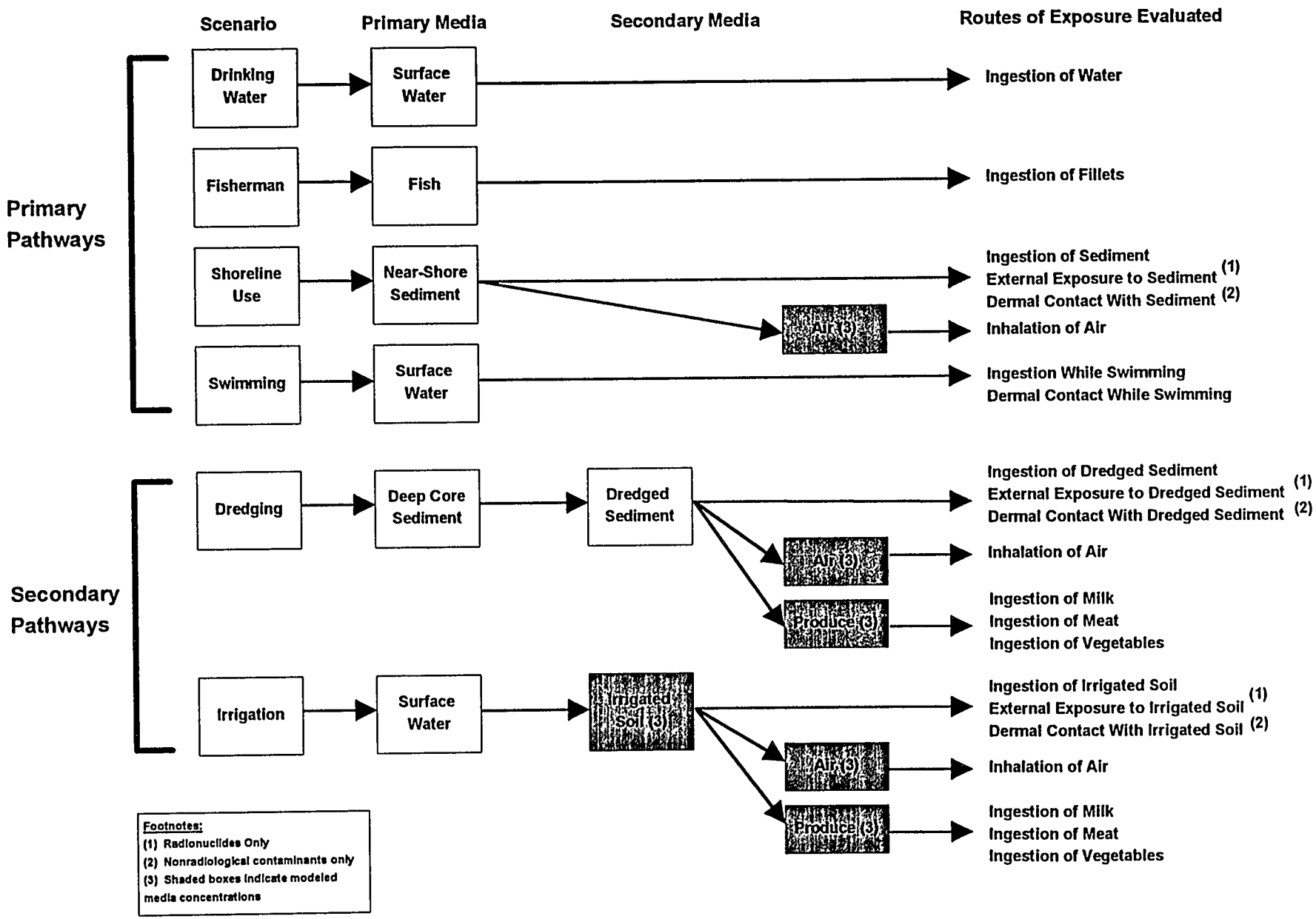


Fig. 5.1. Site model for human health risk assessment.

Most of the scenarios included in this risk assessment were previously evaluated in the screening assessments of Hoffman et al. (1991) and Cook et al. (1992). All exposure pathways are considered residential. Because not all individuals use the resources of LWBR in the same way, the risk assessment process includes a variety of exposure scenarios to account for the different ways an individual might be exposed to potentially harmful contaminants. No one individual would be exposed to all exposure pathways.

For Superfund sites, EPA recommends the use of a reasonable maximum exposure (RME) scenario, where "reasonable maximum" is defined so that only those exposures that are likely to occur will be included in the assessment of exposures (EPA 1989, 55 FR 8710). The RME is a conservative estimate of exposure reasonably expected to occur at a site, but it is expected to be within a realistic range of exposure. Estimating the RME involves the use of both EPA's standard default assumptions and best professional judgement. For quantification, each contaminant is evaluated according to its potential to cause adverse human health effects, both carcinogenic and noncarcinogenic.

Exposure scenarios were designated as either primary or secondary in the present risk assessment. Primary exposure scenarios evaluate current exposure pathways using measured contaminant concentrations for the media of interest. These scenarios include consumption of fish, shoreline use, swimming, and ingestion of untreated surface water. Secondary exposure scenarios evaluate future exposure pathways and use modeled estimates of contaminant concentrations for the media of interest. In the present assessment, the dredging and irrigation scenarios were considered secondary exposure scenarios. For example, contaminant concentrations for soil irrigated with LWBR water were not available; nevertheless, concentrations were predicted with mathematical models. Other models were used to predict the uptake of the contaminants in the soil by pasture forage and the subsequent transfer into the milk and meat from cattle that consume the forage. Consumption of milk and meat from cattle that have grazed on pastures irrigated with LWBR water are examples of secondary exposure pathways in the risk assessment.

Generic transfer coefficients and parameter values were used in the models to estimate concentrations for media in secondary exposure pathways (Fig. 5.1); consequently, the risk values for the secondary exposure pathways have a higher degree of uncertainty than those for the primary pathways. A quantification of the uncertainty in the dredging scenario is discussed in Sect. 5.4. In addition, the mathematical models employ conservative assumptions that ensure that the calculated risk values are also conservative.

One of the primary differences in this baseline risk assessment and the previous screening assessments is the use of different exposure parameters. A baseline risk assessment should reflect realistic exposure conditions, whereas screening assessments often include very conservative exposure parameters. However, the present baseline risk assessment uses EPA-recommended parameter values that in many instances are more conservative than those used in the previous screening risk assessments. The present risk assessment evaluates the health risks to a child (0–6 years old) as well as an adult for exposure to noncarcinogens. Exposure to carcinogens is assessed only for adults because the 30-year adult exposure period exceeds the 6-year child exposure period. It would be possible to evaluate the risk to an adult whose exposure began as a child; however, the end result would not be substantially different from the result reached by simply considering an adult.

Following are general descriptions of the six exposure scenarios evaluated in the human health risk assessment.

5.2.2.1 Water ingestion scenario

The water ingestion scenario assumes that an individual drinks untreated water from the reservoir. As far as is known, no one routinely drinks untreated reservoir water; but, because the possibility exists, this exposure pathway is included in the risk assessment. The contaminant concentrations are given in Table A.1. These data represent ambient concentrations in samples that were collected during CRRI Phase 1 sampling and by the ORR monitoring program from a station at CRM 9.9 (Brashear Island). The ORR data were included to supplement the LWBR data but were evaluated separately because of the difference in location of the sampling sites. The exposure media concentrations and the intake values for all contaminants that were detected in the surface water samples are listed in Tables B.1 through B.3. Equations and pathway-specific intake parameters such as intake rates, exposure duration, etc. are listed in Table A.6.

5.2.2.2 Fish consumption scenario

Many individuals who live in the vicinity of LWBR are avid anglers. The fish consumption scenario represents an avid fisherman (or woman) who regularly catches fish in LWBR and uses the fish to feed the family. Because many fishermen are selective in the species of fish they eat and not all species of fish accumulate contaminants in their tissues to the same extent, several species of fish are included in the present assessment. These species are bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), striped bass (*Morone saxatilis*), hybrid bass (*Morone chrysops/Morone saxatilis*), and channel catfish (*Ictalurus punctatus*). Separate risk calculations were made for each species.

Fish contaminant data collected since CRRI Phase 1 (Sect. 3.2.14), during CRRI Phase 1 (Cook et al. 1992), and from the TVA fish contaminant studies (Sect. 3.2.10) were used to estimate contaminant concentrations in fish fillets. Although some individuals consume other parts of fish (fish patties containing bone, fish roe, etc.), this risk assessment considers the ingestion of fish fillets only.

The intake equations and parameter values for the fish ingestion pathway are listed in Table A.13. For this exposure assessment, parameter values were chosen to evaluate reasonable exposure to adults and children who regularly eat fish caught in the LWBR. The contaminant concentrations and calculated intake and exposure values for an adult and a child are listed in Tables B.10 and B.11.

5.2.2.3 Shoreline-use scenario

The shoreline-use scenario considers persons who spend time along the shoreline of the reservoir where they may come in contact with near-shore sediment through activities such as sun bathing, strolling, searching for artifacts, wading, etc. Some activities such as strolling and searching for artifacts are often more popular in the fall and winter when the water level in the reservoir is at its lowest. In many locations, the winter water level provides a broader beach and easier access.

Near-shore sediments are defined here as sediments along the shore of LWBR at water depths of 6 ft or less at the summer water level. Data for near-shore sediment were obtained from Phase 1 (Cook et al. 1992), post-Phase 1, and TVA public use and water intake areas (TVA 1991a). The Phase 1 and post-Phase 1 data sets were combined; the TVA data were considered separately. Four exposure pathways are included in the shoreline-use scenario: ingestion of near-

shore sediment, inhalation of the sediment in the form of dust, dermal contact with near-shore sediment, and external exposure to gamma-emitting radionuclides in the sediment. Each pathway is discussed below.

Shoreline-use scenario - sediment ingestion pathway

It is assumed that an individual who spends time along the shore of LWBR would inadvertently ingest a small quantity of sediment. Table A.12 lists the intake equations and exposure parameters used to access the risk of adverse health effects from the inadvertent ingestion of near-shore sediment. Parameter values used in the exposure equation have been derived from standard intake rates, exposure frequencies, exposure durations, and averaging times. These values are based on EPA standard default values for residential settings. Results of the risk assessment for the ingestion of near-shore sediments by an individual who uses the shoreline area for recreational activities are given in Tables B.7 through B.9.

Shoreline-use scenario - inhalation of resuspended particles pathway

The evaluation of the exposure of an individual to contaminants in near-shore sediment that is resuspended into the air is dependent on the quantity of fugitive dust (resuspended sediments) generated by the wind. Although no air monitoring data were available for the LWBR area, the quantity of fugitive dust generated by the wind was empirically derived by the method described in Eckerman and Young (1980). This method was also used in the previous screening risk assessments of Hoffman et al. (1991) and Cook et al. (1992). The risk assessment equations and parameter values are given in Table A.9. Exposure media concentrations, intake rates, and exposure values for the inhalation of resuspended near-shore sediments are given in Tables B.7 through B.9.

Shoreline-use scenario - external exposure pathway

The external exposure pathway is concerned only with external exposure to gamma radiation from radionuclides while an individual spends time along the shoreline. Radionuclides are carcinogens; and, as noted earlier, only adults are considered for exposure to carcinogens. Exposure equations and related parameter values are listed in Table A.11. These equations and parameter values were obtained from the National Council on Radiation Protection (NCRP) (1989) and EPA (1989). Exposure media concentrations, intake rates, and exposure values for the external exposure pathway are listed in Tables B.7 and B.8.

Shoreline-use scenario - dermal contact with sediment pathway

It is assumed that an individual who spends time at a beach area will come into physical contact with the sediment, and some fraction of the contaminant(s) in the sediment will be absorbed through the skin. Radionuclides were not evaluated for dermal exposure because it was assumed that the dose from external exposure to the radionuclides would provide a much greater health risk than the dose from dermal contact with those same radionuclides. It was also conservatively assumed that an individual would only visit the beach area for a maximum of 6 months per year, or half of the standard exposure frequency of 350 d per year. Pathway-specific intake parameters are listed in Table A.10. Media exposure concentrations, intakes rates, and exposure values for the dermal contact pathway are listed in Tables B.8 and B.9.

5.2.2.4 Swimming scenario

An individual can be exposed to water in LWBR through recreational activities such as swimming, water skiing, and wading. A swimming scenario is included in the risk assessment because swimming offers the greatest potential for external (dermal) and internal (oral) exposure to the water. The exposure frequency for the swimming scenario is limited to four months (120 days) because swimming is a local summertime activity. Water data were collected during the Phase 1 sampling; but, because only four surface water grab samples were collected from LWBR, approximately one year of ORR water monitoring data from CRM 9.9 (Brashear Island) were also evaluated in the assessment.

Swimming scenario - dermal contact with water pathway

Chemicals that are dissolved in the water can be absorbed through the skin while swimming. The rate of absorption is chemical specific and is represented by a permeability constant. These constants are given in Table A.3. As in the case of dermal contact with near-shore sediments, radionuclides were not included in the dermal absorption pathway because the risk of health effects from external exposure to radiation would be much greater than the chemical effects. The concentrations for the detected chemicals in the surface water and the calculated intake values for this exposure pathway are listed in Tables B.5 and B.6. Pathway-specific intake parameters are listed in Table A.8.

Swimming scenario - ingestion of water pathway

Many swimmers will ingest some of the surrounding water while swimming. Standard EPA default intake equations and parameter values were used to calculate exposure estimates from this exposure pathway. The equations and parameter values are listed in Table A.7. Media exposure concentrations and calculated intake values for the ingestion of water while swimming are listed in Tables B.4 through B.6.

5.2.2.5 Irrigation scenario

The irrigation scenario assumes that an individual uses water from LWBR in an overhead irrigation system to grow vegetables for his/her family and forage for cows. During most years, sufficient rainfall occurs in the area surrounding LWBR to make irrigation unnecessary; however, occasional dry periods sometimes reduce crop yields. A readily accessible water source such as the reservoir makes irrigation an affordable option for growing certain crops. In some instances, irrigation is used to protect crops from damage by frost and freezing temperatures. Also, some gardeners who have access to the reservoir use small irrigation systems to water their gardens and lawns. For these reasons, irrigation cannot be ruled out as a potential exposure pathway.

The EPA recommends that 30 years be used as a default lifetime exposure period in human health risk calculations (OSWER 1991). If an individual uses lake water to irrigate crops for 30 years, certain contaminants that are present in the water may build up in soil. For this reason, the values that are used as concentrations for contaminants in irrigated soil are based on a 30-year accumulation minus losses attributable to leaching, harvesting, and radioactive decay of radionuclides. Therefore, irrigated soil is defined as soil that has been irrigated with water from LWBR for a period of 30 years.

The irrigation rate was determined from personal communication with the Roane County Extension Agent. The rate of irrigation in Roane County is ~1 in. of water per week during drought conditions. This is equivalent to 3.62 L/d for each square meter of soil that is irrigated. This irrigation rate is almost twice that reported by the U.S. Department of Agriculture (USDA) (1970); however, it is assumed that irrigation in the LWBR area occurs for only 3 months per year, and the value reported by the USDA is normalized for year-round irrigation.

The equation for calculating contaminant concentrations in the irrigated soil after a 30-year build-up period are given in Table A.14. The water data are the same as those used in the water ingestion and swimming scenarios. Summary statistics were calculated separately for these two data sets (Table A.1). The calculated contaminant concentrations for irrigated soil are listed in Tables B.12 through B.14 along with the intake rates for the secondary exposure media.

Five potential exposure pathways were evaluated under the irrigation scenario: inadvertent ingestion of irrigated soil; inhalation of resuspended irrigated soil; dermal contact with irrigated soil; external exposure to radionuclides in irrigated soil; and ingestion of vegetables, meat, and milk produced on the irrigated soil. Tables A.9, A.10, A.11, A.14, and A.15 give the exposure equations and parameter values and references for the each exposure pathway in the irrigation scenario.

Irrigation scenario - soil ingestion pathway

Table A.14 lists the exposure parameters associated with the ingestion of irrigated soil. This ingestion comes about as a result of working with the soil as well as from the ingestion of produce contaminated with soil. The parameters used in this exposure equation have been derived from standard intake rates, exposure frequencies, exposure durations, body weights, and averaging times. Tables B.12 through B.14 give the media exposure concentrations, intake rates, and pathway exposure values for the ingestion of irrigated soil.

Irrigation scenario - inhalation of resuspended particles pathway

The contaminant concentrations in air for the inhalation pathway are listed in Tables B.12 through B.14. These concentrations are estimated using the equation in Table A.9, the same as used in the inhalation pathway in the shoreline-use scenario; however, the irrigated soil contaminant concentrations are modeled values. Table A.9 also gives intake equations and associated parameter values. The calculated intake rates and exposure values are listed in Tables B.12 through B.14.

Irrigation scenario - external exposure to soil pathway

The calculated radionuclide activities and lifetime intake values for external exposure to gamma radiation from irrigated soil are listed in Tables B.12 and B.13. Pathway-specific intake parameters are listed in Table A.11. The external exposure pathway was limited to an adult exposure because gamma radiation is considered carcinogen. As explained in Sect. 5.2.2, the calculated risk of adverse carcinogenic health effects for a child would be less than for an adult because of the shorter exposure period.

Irrigation scenario - dermal contact with soil pathway

The soil exposure concentrations for dermal contact with irrigated soil are the same as those for the ingestion of irrigated soil. Radionuclides were not evaluated in this scenario because it was assumed that the health risk from external exposure to the radionuclides in the soil would be much greater than the risk from dermal absorption of the radionuclides. Media exposure concentrations and the intake values for the contaminants in the dermal exposure pathway are listed in Table B.14. Pathway-specific intake parameters are listed in Table A.10.

Irrigation scenario - ingestion of vegetables, meat, and milk pathway

As shown in the site-conceptual model (Fig. 5.1), the consumption of food produced on soil irrigated with lake water is a potential exposure pathway. The soil concentrations used in the soil ingestion pathway were used to model the expected concentrations in vegetables, meat, and milk. Equations used for estimating plant and animal uptake were taken from IAEA (1982) and DOE (1987) and are listed in Tables C.1 through C.4. For inorganics and radionuclides, contaminant-specific uptake coefficients were used to estimate the contaminant concentrations in produce based on the modeled contaminant concentrations in the irrigated soil. Each transfer coefficient represents a ratio of the contaminant concentration in the plant to the concentration in the soil. Therefore, the product of the contaminant concentration in irrigated soil and the transfer coefficient is the contaminant concentration in the plant. The soil-to-plant elemental transfer coefficients for the reproductive portions of plants from Baes et al. (1984) were used to estimate the contaminant concentrations for the ingestion of produce.

The transfer of organic chemicals to plants from soil was calculated using the regression equation:

$$\log B_v = 1.588 - 0.578 \log K_{ow}$$

where B_v is the transfer coefficient for the chemical and K_{ow} is the chemical's octanol-water partition coefficient (Travis and Arms 1988).

The model equations and transfer coefficients for the irrigation scenario are given in Appendix C. Intake equations and parameter values for the ingestion of produce are given in Tables A.15 and A.5, respectively. Intake values, exposure estimates, and concentrations of contaminants in milk, meat, and vegetables grown on irrigated soil are listed in Tables B.12 through B.14.

5.2.2.6 Dredging scenario

Dredging activities in LWBR are regulated by the TVA and the COE so that the uncontrolled disposal of dredged spoils is unlikely. Nonetheless, a dredging scenario was considered in the baseline risk assessment to address the possibility that dredging might occur in the future. The dredging scenario assumes that deep water sediment is removed from the reservoir and deposited upon the agricultural land, where it is used to grow crops and raise livestock. The sediment is assumed to be spread thick enough so that plowing would not dilute the material by mixing it with the underlying soil.

The same five exposure pathways are evaluated under the dredging scenario as under the irrigation scenario. Direct exposure occurs via ingestion of dredged sediment, dermal contact with

the sediment, inhalation of air containing dust derived from the sediment, and external exposure to gamma-emitting radionuclides. Indirect exposures occur via the ingestion of milk, meat, and vegetables produced on the dredged sediment.

Data for the dredging scenario came from cores taken by Olsen et al. (1992), Cook et al. (1992), and the TVA (1985a). Individual core concentrations were calculated from mass-weighted averages over the entire length of the core. Data from all cores were combined into one data set and summary statistics were calculated from this data set (Table A.1).

Occasionally, someone who own lake-front property requests permission from the TVA to remove sediment from an area near his /her property in order to provide navigable water to a boat dock. In such instances, the dredged spoils are placed on shore where humans might come in direct contact with them or where they might be used for agricultural purposes. However, because the shoreline-use scenario includes direct exposure pathways for near-shore surface sediment and the quantity of sediment that would be dredged is relatively small, dredged near-shore surface sediment is not considered in the dredging scenario.

Tables A.9 through A.12 and A.15 list the exposure parameters associated with each exposure pathway in the dredging scenario. The parameters used in each exposure equation have been derived from standard intake rates, skin surface areas, and adherence factors (EPA 1989). Variables relating to exposure frequency and duration are based on EPA default values and/or are derived from knowledge of site-specific conditions and receptor activity.

Dredging scenario - ingestion of sediment pathway

Table A.12 lists the exposure parameters for the ingestion of dredged sediments. The parameter values used in the exposure equation are standard EPA default values for residential exposure. The intake estimates calculated for the ingestion of dredged sediments by receptors using dredged sediments for agricultural purposes are listed in Tables B.15 and B.16.

Dredging scenario - inhalation of resuspended particles pathway

No air monitoring data were available for the LWBR area, so transport equations were used for estimating air concentrations associated with the dredged sediments. Equations and parameter values used to estimate exposure to a receptor are given in Table A.9. Calculated contaminant concentrations in air and intake estimates are listed in Tables B.15 and B.16.

Dredging scenario - external exposure pathway

The sediment concentrations for the ingestion of dredged sediment (Table B.15) were used in the equation in Table A.11 to estimate the dose from external exposure to radionuclides in the dredged sediment. The sediment concentrations and the "intake values" for the radionuclides in the dredged sediment are listed in Table B.15. The pathway-specific exposure equations and parameter values listed in Table A.11 were taken from NCRP (1989) and EPA (1989).

Dredging scenario - dermal contact pathway

The sediment concentrations for dermal contact with the dredged sediments are the same as those for ingestion of the dredged sediment (Tables B.15 and B.16). Radionuclides were not evaluated in this pathway because it was assumed that the dose from external exposure to the

radionuclides would be much greater than the dose received from dermal absorption of the same radionuclides. Sediment concentrations and the intake values for the contaminants detected in LWBR dredged sediment are listed in Tables B.15 and B.16. Exposure equations and pathway-specific intake parameters are listed in Table A.10.

Dredging scenario - ingestion of vegetables, meat, and milk pathway

As indicated in the site conceptual model (Fig. 5.1), the consumption of contaminated produce is a potential exposure pathway if dredged sediment is used for agricultural purposes. The contaminant concentrations used for the ingestion of dredged sediment were used to model the expected concentrations in vegetables, meat, and milk. Models for estimating plant and animal uptake are the same as those used in the irrigation scenario (Tables C.1 through C.4) and the exposure parameter values and intake equations are also the same as those used in the irrigation scenario (Tables A.5 and A.15). Intake values, exposure estimates, and contaminant concentrations in milk, meat, and vegetables are given in Tables B.15 and B.16.

5.3 TOXICITY ASSESSMENT

The purpose of the toxicity assessment is to evaluate the potential for exposure to contaminants to cause adverse health effects in exposed individuals. Therefore, the relationship between the extent of exposure to a particular contaminant and the increased likelihood and/or severity of adverse health effects as a result of that exposure is evaluated. The toxicity assessment generally involves two steps. The first step involves determining whether exposure to an agent can cause an increase in the incidence of a particular health effect and whether that health effect will occur in humans. The second step involves characterizing the relationship between the dose of the contaminant that is received and the incidence of adverse health effects in exposed populations.

The following sections present general information and contaminant-specific discussions on health effects relating to chemicals identified as COPCs in the risk assessment for the LWBR (Sect 5.3.1) and to chemicals that were detected but do not have EPA-approved toxicity values (Sect. 5.3.2). Complete toxicity profiles are available from ORNL's Biomedical and Environmental Information Analysis Section. Two categories of health effects are considered: carcinogenic and noncarcinogenic. Toxicity summary information [slope factors and reference dose factors (RfDs)] for inorganics and organics is given in Table A.3, and slope factors for radionuclides are given in Table A.4. In this section, data from human and laboratory animal research and data from occupational studies are used to characterize likely health effects resulting from exposure to the COPCs. The health effects described in this section may not necessarily be incurred by exposure to contaminants at levels present in the LWBR area.

5.3.1 Contaminant-Specific Toxicity Assessment

5.3.1.1 Aldrin

Aldrin is a common name for a compound used as an insecticide. Aldrin is a chemical made in the laboratory and does not occur naturally in the environment. The trade names used for aldrin include Aldrec, Aldrex, Drinox, Octalene, Seedrin, and Compound 118. Pure aldrin is a white powder, and technical-grade aldrin is a tan-colored powder. Aldrin evaporates slowly in

the air and has a mild chemical odor. Aldrin can be found in the soil, in water, or in the air near hazardous waste sites. Aldrin can also be found in plants and animals near hazardous waste sites.

From the 1950s until 1970, aldrin was used extensively as an insecticide on crops such as corn and cotton. The USDA prohibited all uses of aldrin in 1970. In 1972, however, EPA approved aldrin for killing termites, and it was used for this purpose until 1987. In 1987, the manufacturer voluntarily canceled the registration for use in controlling termites.

Aldrin can enter the environment from accidental spills or leaks from storage containers at waste sites. In the past, aldrin entered the environment when farmers used it to kill pests on crops and exterminators used it to kill termites. Aldrin is still present in the environment from these past uses. Sunlight and bacteria can change aldrin to dieldrin; therefore, dieldrin will be found in places where aldrin was originally released. Dieldrin in soil or water breaks down slowly; it sticks to soil and sediments on the bottom of lakes very strongly and may stay there unchanged for many years. Dieldrin does not dissolve in water very easily; however, it can attach to dust particles and become airborne, and dieldrin can also be taken up in the roots of plants. Fish and animals that eat dieldrin-contaminated materials store a large amount of dieldrin in their fat. Animals or fish that eat other animals can concentrate dieldrin in their tissue.

Most human exposure to aldrin and dieldrin occurs when people eat foods such as fish from contaminated lakes or streams, root crops, dairy products, and meats. Exposure to aldrin and dieldrin occurs via drinking water, breathing, or contacting contaminated soil. Exposure to aldrin is limited because aldrin is changed very quickly to dieldrin in the environment. Dieldrin stays in the environment for a long time, usually in soil or in animal fat. Background levels of aldrin and dieldrin in air and water are ~0.000001 parts of aldrin or dieldrin in 1 million parts of air or water (i.e., ~0.000001 ppm). Background levels of dieldrin in soil are ~0.001 ppm, and background levels in water are very, very small. Contaminated soils average ~0.057 ppm for dieldrin and 0.019 ppm for aldrin; contaminated groundwater averages ~0.0004 ppm for dieldrin. Contaminated fish contained ~0.030–0.050 ppm of dieldrin.

Studies in animals show that both aldrin and dieldrin enter the body quickly after exposure. Once aldrin is inside the body, it quickly breaks down to dieldrin. Dieldrin stays in fat for a long time, but most dieldrin and its breakdown products leave the body in feces and urine. It can take many weeks or even years for all of the compound to leave the body.

Aldrin and dieldrin cause similar health effects. Exposure to very high levels of aldrin or dieldrin for a short time causes convulsions or kidney damage. Long-term exposure to lower levels can also cause convulsions because of the buildup of the contaminants in the body. Exposure to moderate levels of aldrin or dieldrin also causes headaches, dizziness, irritability, vomiting, or uncontrollable muscle movements. It is not known whether either contaminant causes birth defects or cancer in people. The International Agency for Research on Cancer has determined that the carcinogenicity of aldrin and dieldrin to humans is not classifiable. EPA has determined that aldrin and dieldrin are toxic contaminants and are probable human carcinogens (i.e., an EPA Class B2 carcinogen). The oral and inhalation slope factor for aldrin is 17 $(mg/kg/d)^{-1}$. The RfD for chronic oral exposures to aldrin is 0.00003 mg/kg/d and is used to evaluate potential noncancerous effects from exposure to aldrin.

5.3.1.2 Arsenic

The toxicity of inorganic arsenic depends on its valence state (-3, +3, or +5), and on the physical and chemical properties of the compound in which it occurs. Trivalent (As^{+3}) compounds are generally more toxic than pentavalent (As^{+5}) compounds. The more water-soluble compounds are usually more toxic and more likely to have systemic effects than the less soluble compounds, which are more likely to cause chronic pulmonary effects. One of the most toxic inorganic arsenic compounds is arsine gas (AsH_3). Laboratory animals are generally less sensitive than humans to the toxic effects of inorganic arsenic. In addition, in rodents the critical effects appear to be immunosuppression and hepato-renal dysfunction, whereas in humans the skin, vascular system, and peripheral nervous system are the primary target organs.

Water-soluble inorganic arsenic compounds are absorbed through the gastrointestinal (GI) tract (>90%) and lungs; distributed primarily to the liver, kidney, lung, spleen, aorta, and skin; and excreted mainly in the urine, at rates as high as 80% in 61 h following oral dosing. Pentavalent arsenic is reduced to the trivalent form and then methylated in the liver to less toxic methylarsinic acids.

Symptoms of acute inorganic arsenic poisoning in humans are nausea, anorexia, vomiting, epigastric and abdominal pain, and diarrhea. Dermatitis (exfoliative erythroderma), muscle cramps, cardiac abnormalities, hepatotoxicity, bone marrow suppression and hematological abnormalities (anemia), vascular lesions, and peripheral neuropathy (motor dysfunction and paresthesia) have also been reported. Oral doses as low as 20–60 μg/kg/d have been reported to cause toxic effects in some individuals. Severe exposures can result in acute encephalopathy, congestive heart failure, stupor, convulsions, paralysis, coma, and death. The acute lethal dose to humans has been estimated at ~0.6 mg/kg/d. General symptoms of chronic arsenic poisoning in humans are weakness, general debility and lassitude, loss of appetite and energy, loss of hair, hoarseness of voice, loss of weight, and mental disorders. Primary target organs are the skin (hyperpigmentation and hyperkeratosis), nervous system (peripheral neuropathy), and vascular system. Anemia, leukopenia, hepatomegaly, and portal hypertension have also been reported. In addition, possible reproductive effects include a high male-to-female birth ratio.

The RfD for chronic oral exposures, 0.0003 mg/kg/d, is based on a no-observable-adverse-effect level (NOAEL) of 0.0008 mg/kg/d and a lowest-observed-adverse-effect level (LOAEL) of 0.014 mg/kg/d for hyperpigmentation, keratosis, and possible vascular complications in a human population consuming arsenic-contaminated drinking water. Because of uncertainties in the data, U.S. EPA states that "strong scientific arguments can be made for various values within a factor of 2 or 3 of the currently recommended RfD value." The subchronic RfD is the same as the chronic RfD, 0.0003 mg/kg/d.

Acute inhalation exposures to inorganic arsenic can damage mucous membranes; cause rhinitis, pharyngitis, and laryngitis; and result in nasal septum perforation. Chronic inhalation exposures, as occurring in the work place, can lead to rhino-pharyno-laryngitis, tracheobronchitis, dermatitis, hyperpigmentation, and hyperkeratosis; leukopenia; peripheral nerve dysfunction as indicated by abnormal nerve conduction velocities; and peripheral vascular disorders as indicated by Raynaud's syndrome and increased vasospastic reactivity in fingers exposed to low temperatures. Higher rates of cardiovascular disease have also been reported in some workers exposed to arsenic. Possible reproductive effects include a high frequency of spontaneous abortions and reduced birth weights. Arsine gas (AsH_3) at concentrations as low as 3–10 ppm

for several hours can cause toxic effects. Hemolysis, hemoglobinuria, jaundice, hemolytic anemia, and necrosis of the renal tubules have been reported in exposed workers.

Subchronic and chronic reference concentrations (RfCs) for inorganic arsenic have not been derived.

Epidemiological studies have revealed an association between arsenic concentrations in drinking water and increased incidence of skin cancers (including squamous cell carcinomas and multiple basal cell carcinomas), as well as cancers of the liver and bladder and respiratory and GI tracts. Occupational exposure studies have shown a clear correlation between exposure to arsenic and lung cancer mortality. EPA has placed inorganic arsenic in weight-of-evidence group A, human carcinogen. A drinking-water unit risk of 5×10^{-5} $(\mu g/L)^{-1}$ has been proposed, derived from drinking-water unit risks for females and males that are equivalent to slope factors of 1.0×10^{-3} $(\mu g/kg/d)^{-1}$ (females) and 2.0×10^{-3} $(\mu g/kg/d)^{-1}$ (males). For inhalation exposures, a unit risk of 4.3×10^{-3} $(\mu g/m^3)^{-1}$ and a slope factor of 50 $(mg/kg/d)^{-1}$ have been derived.

5.3.1.3 Cadmium

Cadmium is a naturally occurring metal that is used in various chemical forms in metallurgical and other industrial processes and in the production of pigments. Environmental exposure can occur via the diet and drinking water.

Cadmium is absorbed more efficiently by the lungs (30 to 60%) than by the GI tract, the latter being a saturable process. Cadmium is transported in the blood and widely distributed in the body but accumulates primarily in the liver and kidneys. Cadmium burden (especially in the kidneys and liver) tends to increase in a linear fashion up to ~50 or 60 years of age, after which the body burden remains relatively constant. Metabolic transformations of cadmium are limited to its binding to protein and nonprotein sulfhydryl groups and to various macromolecules, such as metallothionein, which is especially important in the kidneys and liver. Cadmium is excreted primarily in the urine.

Acute oral exposure to 20–30 g has caused fatalities in humans. Exposure to lower amounts may cause GI irritation, vomiting, abdominal pain, and diarrhea. An asymptomatic period of one-half to one hour may precede the onset of clinical signs. Longer-term exposure to cadmium primarily affects the kidneys, resulting in tubular proteinosis, although other conditions such as "itai-itai" disease may involve the skeletal system. Cadmium involvement in hypertension is not fully understood.

Inhalation exposure to cadmium and cadmium compounds may result in effects including headache, chest pains, muscular weakness, pulmonary edema, and death. The 1-minute and 10-min lethal concentration of cadmium for humans has been estimated to be ~2500 and 250 mg/m^3, respectively. An 8-h time-weighted-average exposure level of 5 mg/m^3 has been estimated for lethal effects of inhalation exposure to cadmium, and exposure to 1 mg/m^3 is considered to be immediately dangerous to human health. Renal toxicity (tubular proteinosis) may also result from inhalation exposure to cadmium.

Chronic oral RfDs of 5×10^{-4} and 1×10^{-3} mg/kg/d have been established for cadmium exposure via drinking water and food, respectively. Both values reflect incorporation of an uncertainty factor of 10. The RfDs are based on an extensive data base regarding toxicokinetics

and toxicity in both human and animals, the critical effect being renal tubular proteinuria. Confidence in the RfD and data base is high. Inhalation RfC values are currently not available.

The target organ for cadmium toxicity via oral exposure is the kidney. For inhalation exposure, both the lungs and kidneys are target organs for cadmium-induced toxicity.

There is limited evidence from epidemiologic studies for cadmium-related respiratory-tract cancer. An inhalation unit risk of 1.8×10^{-3} $(\mu g/m^3)^{-1}$ and an inhalation slope factor of 6.1 $(mg/kg/d)^{-1}$ are based on respiratory tract cancer associated with occupational exposure. Based on limited evidence from multiple occupational exposure studies and adequate animal data, cadmium is placed in weight-of-evidence group B1, probable human carcinogens.

5.3.1.4 Chlordane

Chlordane is a man-made chemical that was used as a pesticide in the United States from 1948 to 1988. Trade names for chlordane are Octachlor and Velsicol 1068. Chlordane is a thick liquid ranging in color from colorless to amber, depending on its purity. It may have no smell or a mild, irritating smell. Chlordane is a mixture of many related chemicals including *trans*-chlordane, *cis*-chlordane, β-chlordane, heptachlor, and *trans*-nonachlor. Chlordane does not dissolve in water; therefore, when it is used as a spray, it must be combined with soap-like substances called emulsifiers, which results in a milky-looking mixture.

From 1983 until 1988, chlordane's only approved use was to control termites in homes. The pesticide was applied underground around the foundation of homes. Prior to 1978, chlordane was also used as a pesticide on agricultural crops, lawns, and gardens and as a fumigating agent. Because of concerns over cancer risk, evidence of human exposure and build up in body fat, persistence in the environment, and danger to wildlife, EPA banned the use of chlordane on food crops and phased out other above-ground uses over the next 5 years. In 1988, when EPA banned chlordane's use for controlling termites, all approved use of chlordane in the United States stopped.

Chlordane is persistent in the environment. It attaches strongly to particles in the upper layers of soil and is likely to remain there for 20 years or more, particularly in heavy, clayey or organic soil. Chlordane and the chemicals it changes into accumulate in fish, birds, and mammals. It stays in the environment for many years and is found in food, air, water, and soil and in the fat of fish, birds, mammals, and almost all humans.

The most common source of chlordane exposure is from ingesting chlordane-contaminated food. Chlordane remains in the food supply because much of the farmland was treated with chlordane in the 1960s and 1970s, and it has remained in the soils since that time. Chlordane may also be found in fish and shellfish caught in chlordane-contaminated waters; however, chlordane is almost never detected in drinking water.

Chlordane can enter the body through the skin if skin contact occurs with contaminated soils, through the lungs if inhaled with contaminated air, and through the digestive tract if swallowed. Most chlordane that enters the body leaves in a few days, mostly in the feces, but to a much smaller extent in urine. Chlordane and its breakdown products may be stored in body fat and cause no adverse effects unless released in large amounts. It may take months or years before all of the breakdown products stored in fat leave the body.

Chlordane exposure may be linked to effects on the nervous system, the digestive system, and the liver in humans. These effects were seen mostly in individuals who swallowed chlordane mixtures. Large amounts of chlordane taken by mouth can cause convulsions and death. Swallowing small amounts or breathing air containing high concentrations of chlordane vapors can cause a variety of nervous system effects, including headaches, irritation, confusion, weakness, vision problems, upset stomach, vomiting, stomach cramps, diarrhea, and jaundice.

It is not known whether chlordane causes cancer in humans. Studies of workers who made or used chlordane do not link exposure with cancer, but the information is not sufficient. Mice fed low levels of chlordane in their food for most of their lifetimes developed liver cancer. The International Agency for Research on Cancer has determined that the chlordane's carcinogenicity to humans is not classifiable. It is not known whether chlordane causes reproductive or birth defects in humans; however, some evidence indicates that animals exposed before birth or while nursing develop behavioral effects.

EPA has determined that chlordane is a probable human carcinogen, a Class B2 carcinogen. The oral and inhalation slope factor for evaluating potential carcinogenic effects from exposure to chlordane is 1.3 $(mg/kg/d)^{-1}$. The RfD for evaluating potential noncancerous risks from chronic oral exposures is 6.0×10^{-5} mg/kg/d.

5.3.1.5 Chromium

Elemental chromium does not occur in nature but is present in ores, primarily chromite ($FeOCr_2O_3$). Only two of the several oxidation states of chromium, Cr(III) and Cr(VI), are reviewed in this report based on their predominance and stability in the ambient environment and their toxicity in humans and animals.

Chromium plays a role in glucose and cholesterol metabolism and is thus an essential element to man and animals. Nonoccupational exposure to the metal occurs via ingestion of food and water containing chromium, whereas occupational exposure occurs via inhalation. Workers in the chromate industry have been exposed to estimated chromium levels of 10–50 $\mu g/m^3$ for Cr(III) and 5–1000 $\mu g/m^3$ for Cr(VI); however, improvements in the newer chrome-plating plants have reduced the Cr(VI) concentrations 10- to 40-fold.

Chromium III is poorly absorbed, regardless of the route of exposure, whereas Cr(VI) is more readily absorbed. Humans and animals localize chromium in the lung, liver, kidney, spleen, adrenals, plasma, bone marrow, and red blood cells. There is no evidence that chromium is biotransformed, but Cr(VI) does undergo enzymatic reduction, resulting in the formation of reactive intermediates and Cr(III). The main routes for the excretion of chromium are via the kidneys and urine and the bile and feces.

The respiratory and dermal toxicity of chromium is well-documented. Workers exposed to chromium have developed nasal irritation (at <0.01 mg/m^3, acute exposure), nasal ulcers, perforation of the nasal septum (at ~ 2 $\mu g/m^3$, subchronic or chronic exposure) and hypersensitivity reactions and "chrome holes" of the skin. Among the general population, contact dermatitis has been associated with the use of bleaches and detergents.

The subchronic and chronic oral RfD value is 1 mg/kg/d for Cr(III). The subchronic and chronic oral RfDs for Cr(VI) are 0.02 and 0.005 mg/kg/d, respectively. The subchronic and chronic oral RfD values for Cr(VI) and Cr(III) are derived from NOAELs of 1.47 g/kg/d for

Cr(III) and 25 ppm of potassium dichromate (Cr[VI]) in drinking water, respectively. The inhalation RfC values for both Cr(III) and Cr(VI) are currently under review by an EPA work group.

Inhalation of chromium compounds has been associated with the development of cancer in workers in the chromate industry. The relative risk for developing lung cancer has been calculated to be as much as 30 times that of controls. There is also evidence for an increased risk of developing nasal, pharyngeal, and GI carcinomas. Quantitative epidemiological data were obtained by Mancuso and Hueper (1951), who observed an increase in deaths (18.2%; $p<0.01$) from respiratory cancer among chromate workers compared with 1.2% deaths among controls. In a followup study, conducted when more than 50% of the cohort had died, the observed incidence for lung cancer deaths had increased to ~60%. The workers were exposed to 1—8 mg/m^3/year total chromium. Mancuso (1975) observed a dose response for total chromium exposure and attributed the lung cancer deaths to exposure to insoluble [Cr(III)], soluble [Cr(VI)], and total chromium. The results of inhalation studies in animals have been equivocal or negative.

Based on sufficient evidence for humans and animals, Cr(VI) has been placed in the EPA weight-of-evidence classification A, human carcinogen. For inhalation exposure, the unit risk value is 1.2×10^{-2} (μg/m^3)$^{-1}$, and the slope factor is 41 (mg/kg/d)$^{-1}$.

5.3.1.6 Manganese

Manganese is an essential trace element in humans that can elicit a variety of serious toxic responses upon prolonged exposure to elevated concentrations either orally or by inhalation. The central nervous system is the primary target. Initial symptoms are headache, insomnia, disorientation, anxiety, lethargy, and memory loss. These symptoms progress with continued exposure and eventually include motor disturbances, tremors, and difficulty in walking, similar to symptoms seen in Parkinsonism. These motor difficulties are most often irreversible. Based on human epidemiological studies, 0.8 mg/kg/d for drinking water exposure and 0.34 mg/m^3 in air for inhalation exposure have been estimated as LOAELs for central nervous system effects.

Effects on reproduction (decreased fertility, impotence) have been observed in humans with inhalation exposure and in animals with oral exposure at the same or similar doses that initiate the central nervous system effects. An increased incidence of coughs, colds, dyspnea during exercise, bronchitis, and altered lung ventilatory parameters have also been seen in humans and animals with inhalation exposure. A possible effect on the immune system may account for some of the respiratory symptoms.

An RfD of 0.1 mg/kg/d for both chronic and subchronic oral exposure has been calculated by EPA from a human NOAEL of 0.14 mg/kg/d, which was determined from a series of epidemiological studies. Large populations with different concentrations of manganese in their drinking water were examined. No adverse effects attributable to manganese were seen in any of these groups. Because manganese is essential in human nutrition, the RfD was derived from these studies without uncertainty factors. The exposure of the most sensitive groups was included in the population examined. Confidence in the oral RfD is rated "medium" by EPA.

An RfC of 0.4 μg/m^3 for both chronic and subchronic inhalation exposure was calculated from a human LOAEL of 0.34 mg/m^3 (Roels et al. 1987) for respiratory and psychomotor disturbances. The study population was occupationally exposed to airborne manganese dust with a median concentration of 0.97 mg/m^3 for 1 to 19 years with a mean duration of 7.1 years.

Neurological examinations, psychomotor tests, lung function tests, blood tests, and urine tests were used to determine the possible effects of exposure. The LOAEL was calculated by expanding the dose to continuous exposure. Confidence in the inhalation RfC is rated "medium" by EPA.

There are some conflicting data on possible carcinogenesis following injections of manganese chloride and manganese sulfate in mice. However, the EPA weight-of-evidence classification is "D," not classifiable as to human carcinogenicity, based on no evidence in humans and inadequate evidence in animals.

5.3.1.7 Mercury

Mercury is a naturally occurring element existing in multiple forms and in various oxidation states. It is used in a wide variety of products and processes. In the environment, mercury may undergo transformations among its various forms and among its oxidation states. Exposure to mercury may occur in both occupational and environmental settings, the latter primarily involving dietary exposure.

Absorption, distribution, metabolism, and excretion of mercury is dependent upon its form and oxidation state. Organic mercurials are more readily absorbed than are inorganic forms. An oxidation-reduction cycle is involved in the metabolism of mercury and mercury compounds by both animals and humans. The urine and feces are primary excretory routes. The elimination half-life is 35 to 90 d for elemental mercury and mercury vapor, and ~40 d for inorganic salts.

Ingestion of mercury metal is usually without effect. Ingestion of inorganic salts may cause severe GI irritation, renal failure, and death at acute doses in humans ranging from 1–4 g. Mercuric (divalent) salts are usually more toxic than are mercurous (monovalent) salts. Mercury is also known to induce hypersensitivity reactions such as contact dermatitis and acrodynia (pink disease). Inhalation of mercury vapor may cause irritation of the respiratory tract, renal disorders, central nervous system effects characterized by neurobehavioral changes, peripheral nervous system toxicity, renal toxicity (immunologic glomerular disease), and death.

Toxicity resulting from subchronic and chronic exposure to mercury and mercury compounds usually involves the kidneys and/or nervous system, the specific target and effect being dependent on the form of mercury. Organic mercury, especially methyl mercury, rapidly enters the central nervous system resulting in behavioral and neuromotor disorders. The developing central nervous system is especially sensitive to this effect, as documented by the epidemiologic studies in Japan and Iraq where ingestion of methyl mercury-contaminated food resulted in severe toxicity and death in adults and severe effects in infants. Blood mercury levels of < 10 μg/L/d and 300 μg/L/d corresponded to mild effects and death, respectively. Teratogenic effects resulting from organic or inorganic mercury exposure do not appear to be well documented for humans or animals, although some evidence exists for mercury-induced menstrual cycle disturbances and spontaneous abortions.

A subchronic and chronic oral RfD of 0.0003 mg/kg/d for methyl mercury is based on the ambient level required to produce a blood mercury level of 200 ng/mL. A subchronic and chronic oral RfD of 0.0003 mg/kg/d for inorganic mercury has been proposed and is based on immunologic glomerulonephritis. An LOAEL of 0.317 mg/kg/d for inorganic mercury was identified in the key study. NOAELs were not available for oral exposure to inorganic mercury or methyl mercury. A subchronic and chronic inhalation RfC of 0.0003 mg/m^3 for inorganic

mercury has been proposed and is based on neurological disorders (increased frequency of intention tremors) following long-term occupational exposure to mercury vapor. The LOAELs for subchronic and chronic inhalation exposures to inorganic mercury are 0.32 and 0.03 mg Hg/m^3, respectively. NOAELs were unavailable. An inhalation RfC for methyl mercury has not been determined.

EPA has placed inorganic mercury in weight-of-evidence classification D, not classifiable as to human carcinogenicity. Methyl mercury has not been evaluated by EPA for evidence of human carcinogenic potential, and no carcinogenicity slope factors have been calculated.

5.3.1.8 Polychlorinated biphenyls (Aroclor-1254 and Aroclor-1260)

PCBs are a group of man-made chemicals that contain 209 individual compounds (known as congeners); they are either oily liquids or solids and are colorless to light yellow, without any known smell or taste. Seven types of PCB mixtures include 35% of all the different PCBs commercially produced and 98% of PCBs sold in the United States since 1970. Some commercial PCB mixtures are known in the United States by their industrial trade name, Aroclor (i.e., Aroclor-1254, Aroclor-1260). The name Aroclor-1254 signifies that the molecule contains 12 carbon atoms (the first two digits) and ~54% chlorine by weight (the second two digits). PCBs do not burn easily and are a good insulating material; therefore, they have been used widely as coolants and lubricants in transformers, capacitors, and other electrical equipment. However, in 1977, manufacture of PCBs stopped in the United States because evidence showed that PCBs tend to build up in the environment and cause harmful effects.

In the past, PCBs entered the environment through their manufacture and use. Today, they can enter the environment from poorly maintained hazardous waste sites; illegal or improper dumping of PCB wastes; leaks or releases from electrical transformers containing PCBs; and disposal of PCB-containing consumer products into municipal landfills rather into landfills designed for hazardous waste. PCBs can exist in air as both solid and liquid aerosols and as vapor that eventually returns to the land and water by settling or washout by snow or rainwater. Even extremely small amounts of PCBs can remain in water for several years, and they will bind strongly with soil and remain bonded for several years.

PCBs can be ingested or inhaled and can cause skin and nose irritations. Although several health effects as a result of exposure to PCBs have been seen in rats (i.e., liver damage, stomach, and thyroid gland injuries; anemia; acne; and damaged reproduction organs), little conclusive evidence concerning the effects of PCBs on humans is available. In addition, rats that were fed certain PCB mixtures throughout their lives developed liver cancers; however, it is not known if the same effects would occur in people. After inhalation exposure, workers have exhibited respiratory tract and eye irritation, coughs, and tightness of the chest, as well as GI symptoms (anorexia, weight loss, nausea, vomiting, and abdominal pain); none of these symptoms were reported in a great quantity, and therefore results are inconclusive.

Studies of workers exposed to PCBs provide inconclusive evidence for exposure-related cancer; however, an excess risk of cancer of the liver and gall bladder has been reported in workers in two capacitor plants where PCB mixtures are commonly used. The International Agency for Research on Cancer has determined that PCBs are *probably* carcinogenic to humans, and EPA has placed PCBs on the Class B2 carcinogen list, indicating PCBs to be probable human carcinogens. The oral slope factor for PCBs is 7.7 $(mg/kg/d)^{-1}$, the highest determined slope factor from three rat studies using Aroclor-1260.

5.3.1.9 Zinc

Zinc is used primarily in galvanized metals and metal alloys, but zinc compounds also have wide commercial applications as chemical intermediates, catalysts, pigments, vulcanization activators, and accelerators in the rubber industry; ultraviolet stabilizers; and supplements in animal feeds and fertilizers. They are also used in rayon manufacture, smoke bombs, soldering fluxes, mordants for printing and dyeing, wood preservatives, mildew inhibitors, deodorants, antiseptics, and astringents. In addition, zinc phosphide is used as a rodenticide.

Zinc is an essential element with recommended daily allowances ranging from 5 mg for infants to 15 mg for adult males.

GI absorption of zinc is variable (20–80%) and depends on the chemical compound as well as on zinc levels in the body and dietary concentrations of other nutrients. In individuals with normal zinc levels in the body, GI absorption is 20–30%. Information on pulmonary absorption is limited and complicated by the potential for GI absorption due to mucociliary clearance from the respiratory tract and subsequent swallowing. Zinc is present in all tissues, with the highest concentrations in the prostate, kidney, liver, heart, and pancreas. Zinc is a vital component of many metalloenzymes such as carbonic anhydrase, which regulates CO_2 exchange. Homeostatic mechanisms involving metallothionein in the mucosal cells of the GI tract regulate zinc absorption and excretion.

In humans, acutely toxic oral doses of zinc cause nausea, vomiting, diarrhea, abdominal cramps, and, in some cases, gastric bleeding. Ingestion of zinc chloride can cause burning in the mouth and throat, vomiting, pharyngitis, esophagitis, hypocalcemia, and elevated amylase activity indicative of pancreatitis. Zinc phosphide, which releases phosphine gas under acidic conditions in the stomach, can cause vomiting, anorexia, abdominal pain, lethargy, hypotension, cardiac arrhythmias, circulatory collapse, pulmonary edema, seizures, renal damage, leukopenia, and coma and death in days to weeks. The estimated fatal dose is 40 mg/kg.

GI upset has also been reported in individuals taking daily dietary zinc supplements for up to 6 weeks. There is also limited evidence that the human immune system may be impaired by subchronic exposures. In animals, GI and hepatic lesions, pancreatic lesions, anemia, and diffuse nephrosis have been observed following subchronic oral exposures.

Chronic oral exposures to zinc have resulted in hypochromic microcytic anemia associated with hypoceruloplasminemia, hypocupremia, and neutropenia in some individuals. Anemia and pancreatitis were the major adverse effects observed in chronic animal studies. Teratogenic effects have not been seen in animals exposed to zinc; however, high oral doses can affect reproduction and fetal growth.

The RfD for chronic oral exposure to zinc is under review by EPA; the currently accepted RfD for both subchronic and chronic exposures is 0.2 mg/kg/d, based on clinical data demonstrating zinc-induced copper deficiency and anemia in patients taking zinc sulfate for the treatment of sickle-cell anemia. The chronic oral RfD for zinc phosphide is 0.0003 mg/kg/d, and the subchronic RfD is 0.003 mg/kg/d.

Under occupational exposure conditions, inhalation of zinc compounds (mainly zinc oxide fumes) can result in a condition identified as "metal fume fever," which is characterized by nasal passage irritation, cough, rales, headache, altered taste, fever, weakness, hyperpnea, sweating,

pains in the legs and chest, leukocytosis, reduced lung volume, and decreased diffusing capacity of carbon monoxide. Inhalation of zinc chloride can result in nose and throat irritation, dyspnea, coughing, chest pain, headache, fever, nausea and vomiting, and respiratory disorders such as pneumonitis and pulmonary fibrosis. Pulmonary inflammation and changes in lung function have also been observed in inhalation studies on animals.

Although "metal fume fever" occurs in occupationally exposed workers, it is primarily an acute and reversible effect that is unlikely to occur under chronic exposure conditions when zinc air concentrations are less than 8–12 mg/m^3. GI distress, as well as enzyme changes indicative of liver dysfunction, have also been reported in workers occupationally exposed to zinc; however, the extent to which these effects are the result of pulmonary clearance and subsequent GI absorption is unclear. Consequently, there are no clearly defined toxic effects that can be identified as resulting specifically from pulmonary absorption following chronic low-level inhalation exposures. Animal data for chronic inhalation exposures are not available.

An inhalation RfC has not been derived for zinc or zinc compounds.

No case studies or epidemiologic evidence has been presented to suggest that zinc is carcinogenic in humans by the oral or inhalation route. In animal studies, zinc sulfate in drinking water or zinc oleate in the diet of mice for a period of 1 year did not result in a statistically significant increase in hepatomas, malignant lymphomas, or lung adenomas; however, in a 3-year, five-generation study on tumor-resistant and tumor-susceptible strains of mice, exposure to zinc in drinking water resulted in increased frequencies of tumors from the F_0 to the F_4 generation in the tumor-resistant strain (from 0.8 to 25.7%, vs 0.0004% in the controls), and higher tumor frequencies in two tumor-susceptible strains (43.4% and 32.4% vs 15% in the controls).

Because of inadequate evidence in humans and animals, zinc is placed in weight-of-evidence Group D, not classifiable as to human carcinogenicity due to.

5.3.1.10 Radionuclides

Radionuclides are unstable atoms of chemical elements. To achieve a more stable state, these atoms emit charged particles, such as alpha and beta radiation, or neutral gamma rays. Alpha particles are relatively densely charged particles composed of two protons and two neutrons, while beta particles are high-energy electrons. Gamma rays are a type of electromagnetic radiation. The interaction of charged particles and gamma rays with matter produces ionization events that can, in turn, produce biological damage within living cells.

The deposition of energy by ionizing radiation is a random process; therefore, even at very low doses, sufficient energy may be deposited into a critical volume within a cell to result in its modification or even its death (ICRP 1991). The ion pairs created by ionizing radiation travel through the tissue reacting with various molecules. This process can lead to the production of negatively charged free radicals (hydroxyl radicals) through the interaction of ion pairs with water molecules, which are predominant throughout the body cavity. These free radicals can react with deoxyribonucleic acid (DNA), thus providing the potential for cell death, genetic damage, or cancer induction.

Radionuclides that emit alpha particles are of concern when there is a potential exposure resulting from the inhalation or ingestion of the radionuclide. With the exception of the cornea of the eye, exposure to alpha emitters from an external source is usually of no concern because

alpha particles cannot penetrate the outer layer of the epidermis (skin). Alpha particles are directly ionizing and exhibit a high linear energy transfer (LET). This means that they deposit their energy in dense concentration over their track length, resulting in short paths of highly localized ionization reactions. With an increase in ionization events occurring in a much smaller area, the probability for cell damage increases. This is the major explanation for the increase in cancer incidence resulting from inhalation of the alpha-emitting decay products of radon gas. In addition, the naturally occurring alpha-emitter in tobacco products (^{210}Po) may be a contributor to the cancer incidence in smokers (BEIR 1990).

Beta radiation, like alpha radiation, is directly ionizing, but unlike alphas, beta particles exhibit a low LET (i.e., they deposit their energy along a longer track length, resulting in more space between ionization events). At very high levels of beta radiation, external exposures may cause skin burns, and internal exposures may cause organ damage; at low levels the primary concerns are potential cancer induction and damage of genetic material leading to hereditary defects.

Gamma-emitting radionuclides can produce significant internal and external effects. Gamma rays from radionuclides deposited outside the body can reach all internal body organs. Gamma radiation ionizes matter by the photoelectric effect, Compton scattering, and ion pair production. Like beta radiation, gamma rays are low-LET radiation. A difference between the behavior of beta particles and gamma rays is that beta particles deposit almost all of their energy in the medium through which they pass, but gamma-rays often escape the medium because of their higher energies. This is an important consideration when determining whether an internal exposure has actually taken place. For example, in order to detect the presence of radionuclides that have been deposited internally within the human body, urine and/or fecal analysis is often necessary for detection of beta radiation, but direct whole-body measurements are feasible for the detection of photon-emitting radionuclides because of the escaping gamma rays.

Based on extensive evidence acquired from epidemiological studies of human populations that have been exposed to high doses of radiation, EPA classifies radionuclides as Group A carcinogens (EPA 1989, 1992). Probably the most widely studied population has been the atomic bomb survivors of Hiroshima and Nagasaki, Japan. An increased rate of leukemia and other cancers has been observed in this population. However, the extent to which cancer incidence is significantly increased by very low radiation exposures (such as what one might expect as a result of exposure to natural background or to industrially contaminated sites) is still uncertain.

The existing radionuclide contamination at the ORR has resulted from a variety of activities at ORNL, the Y-12 Plant, and the K-25 Site. These activities include fuel reprocessing, reactor facilities, and weapons research and production. In addition, various waste management activities have resulted in radionuclide releases to the environment via groundwater seepage from burial sites. For the low levels of radionuclides present in the Clinch/Tennessee River environment, the primary health effects of concern for potential human exposure are the risks of increased cancer incidence and hereditary defects caused by damage to genetic material.

5.3.1.11 Cesium-137

Cesium-137, a product of nuclear fission reactions, has a radioactive half-life of 30.2 years (Kocher 1981). In ~95% of the radioactive decay, ^{137}Cs emits a beta particle with a maximum energy of 0.514MeV (Turner 1986) and is transformed to the metastable state of ^{137m}Ba. Barium-137m then decays 90% of the time by emitting a 0.662 MeV gamma-ray.

The primary sources of contamination from ^{137}Cs at ORR are the effluent streams from research reactors and seepage from burial grounds used to store radioactive wastes. In aquatic media, ^{137}Cs tends to accumulate in the sediment. However, uptake may also occur in fish and other biota. In the Clinch/Tennessee River environment, the primary routes of human exposure are external exposure to gamma rays from the decay of ^{137m}Ba in shoreline sediment and ingestion of ^{137}Cs in fish.

Cesium-137 entering the human body via ingestion behaves similarly to potassium. It is rapidly and almost completely absorbed from the GI tract and is distributed uniformly throughout the body (ICRP 1990). Because of its relatively short metabolic half-life (110 d), an acute dose of ^{137}Cs is almost completely excreted in the urine within a year of the initial exposure. However, when considering a potentially long-term (chronic) exposure, ^{137}Cs would always be present in the body.

The slope factor for ingestion of ^{137}Cs and its daughter products is 7.6×10^{-10} (Risk/Bq), the slope factor for the inhalation of airborne ^{137}Cs and its daughters is 5.1×10^{-10} (Risk/Bq), and the slope factor for the external exposure to ^{137}Cs and its daughters is 5.4×10^{-5} (Risk/year per Bq/g soil).

5.3.1.12 Cobalt-60

Cobalt-60 is produced in nuclear reactors from neutron activation of stable cobalt atoms. These cobalt atoms are present as a corrosion impurity in the water used as the primary reactor coolant of the reactor (Eichholz 1983). Cobalt-60 has a radioactive half-life of 5.27 years and decays by emitting a 0.314 MeV beta particle and two gamma rays exhibiting energies of 1.33 and 1.17 MeV (Kocher 1981). Cobalt-60 enters the environment surrounding the ORR from the effluent streams from research reactors and seepage from burial grounds used to store radioactive wastes. Important routes of human exposure for ^{60}Co include external exposure resulting from high-energy gamma rays emitted from shoreline sediment and ingestion of contaminated fish, produce, and water. Internally absorbed ^{60}Co is assumed to be uniformly distributed throughout the body.

5.3.1.13 Uranium

The isotopes of uranium likely to be associated with releases from ORthe R are ^{234}U, ^{235}U, and ^{238}U. These isotopes are primarily alpha-emitters with extremely long half-lives: 244,500 years, 0.704 billion years, and 4.47 billion years, respectively (Kocher 1981). Normally, uranium exists in the earth's crust at a level of 2 ppm, with a composition of ~99% ^{238}U and 1% ^{234}U and ^{235}U. Enriched uranium (~95% ^{238}U and 5% ^{235}U) is used primarily as a fuel in the nuclear power industry. Higher grades of enriched uranium (associated with larger fractions of ^{235}U) are typically associated with nuclear weapons systems. The primary releases of uranium from the ORR have resulted from enrichment activities conducted at the K-25 Site.

The important routes of human exposure to uranium are inhalation and ingestion. If inhaled, alpha emissions from the uranium will increase the risk of lung cancer. However, the primary site of uranium deposition after ingestion is bone. A unique aspect of risk associated with uranium is that often the chemical toxicity of ^{238}U instead of the radioactive toxicity dominates. Therefore, both must be considered when assessing the risk involved with a potential human exposure to uranium.

5.3.2 Chemicals Without EPA Toxicity Values

Chemical-specific toxicity values are not available for the chemicals listed in Table 5.2. These chemicals were measured in one or more media at levels above the laboratory quantitation limit; however, because no EPA-approved dose/response values are currently available, the noncarcinogenic or carcinogenic effects of these chemicals cannot be evaluated for any exposure pathway. In lieu of a quantitative risk assessment for these chemicals, a brief discussion of each chemical's toxicology is given.

Table 5.2. Chemicals without EPA toxicity values

Analyte	CAS No.
Inorganics	
Aluminum	7429-90-5
Copper	7440-50-8
Lead	7439-92-1
Magnesium	7439-95-4
Organics	
4-Nitrophenol	100-02-7
Dibenz(a,h)anthracene	53-70-3

5.3.2.1 Aluminum

Aluminum is a silver-white flexible metal with a vast number of uses. It is poorly absorbed and efficiently eliminated; however, when absorption does occur, aluminum is distributed mainly in bone, liver, testes, kidneys, and the brain.

Aluminum may be involved in Alzheimer's disease (dialysis dementia) and in Amyotrophic Lateral Sclerosis and Parkinsonism-Dementia Syndromes of Guam. Aluminum content of brain, muscle, and bone increases in Alzheimer's patients. Neurofibrillary tangles are found in patients suffering from aluminum encephalopathy and Alzheimer's disease. Symptoms of "dialysis dementia" include speech disorders, dementia, convulsions, and myoclonus. People of Guam and Rota have an unusually high incidence of neurodegenerative diseases. The volcanic soil in the region of Guam where the high incidence of this complex occurs contains high levels of aluminum and manganese. Neurological effects have also been observed in rats orally exposed to aluminum compounds.

The respiratory system appears to be the primary target following inhalation exposure to aluminum. Alveolar proteinosis has been observed in guinea pigs, rats, and hamsters exposed to aluminum powders. Rats and guinea pigs exposed to aluminum chlorohydrate exhibited an increase in alveolar macrophages, increased relative lung weight, and multifocal granulomatous pneumonia.

No decrease in reproductive capacity, hormonal abnormalities, or testicular histopathology was observed in male rats exposed to aluminum in drinking water for 90 d. However, male rats exposed to aluminum (as aluminum chloride) via gavage for 6 months exhibited decreased spermatozoa counts and sperm motility and testicular histological and histochemical changes.

Subchronic and chronic RfDs and RfCs have not been derived for aluminum.

EPA has not evaluated aluminum or aluminum compounds for carcinogenicity, and a weight-of-evidence classification is currently not assigned.

5.3.2.2 Copper

Copper occurs naturally in elemental form and as a component of many minerals. Because of its high electrical and thermal conductivity, it is widely used in the manufacture of electrical equipment. Common copper salts, such as the sulfate, carbonate, cyanide, oxide, and sulfide are used as fungicides, as components of ceramics and pyrotechnics, for electroplating, and for numerous other industrial applications. Copper can be absorbed through oral, inhalation, and dermal exposure. It is an essential nutrient that is normally present in a wide variety of tissues.

In humans, ingestion of gram quantities of copper salts may cause GI, hepatic, and renal effects with symptoms such as severe abdominal pain, vomiting, diarrhea, hemolysis, hepatic necrosis, hematuria, proteinuria, hypotension, tachycardia, convulsions, coma, and death. GI disturbances and liver toxicity have also resulted from long-term exposure to drinking water containing 2.2–7.8 mg/L of copper. The chronic toxicity of copper has been characterized in patients with Wilson's disease, a genetic disorder causing copper accumulation in tissues. The clinical manifestations of Wilson's disease include cirrhosis of the liver, hemolytic anemia, neurologic abnormalities, and corneal opacities. In animal studies, oral exposure to copper caused hepatic and renal accumulation of copper and liver and kidney necrosis at doses of $\geq$100 mg/kg/d; and hematological effects at doses of 40 mg/kg/d.

Acute inhalation exposure to copper dust or fumes at concentrations of 0.075–0.12 mg/m^3 may cause metal fume fever with symptoms such as cough, chills and muscle ache. Among the reported effects in workers exposed to copper dust are GI disturbances, headaches, vertigo, drowsiness, and hepatomegaly. Vineyard workers chronically exposed to Bordeaux mixture (copper sulfate and lime) exhibit degenerative changes of the lungs and liver. Dermal exposure to copper may cause contact dermatitis in some individuals.

Oral or intravenous administration of copper sulfate increased fetal mortality and developmental abnormalities in experimental animals. Evidence also indicates that copper compounds are spermicidal.

An RfD for elemental copper is not available. However, EPA established an action level of 1300 μg/L for drinking water (56 FR 26460, June 7, 1991). Data were insufficient to derive an RfC for copper.

No suitable bioassays or epidemiological studies are available to assess the carcinogenicity of copper. Therefore, U.S. EPA has placed copper in weight-of-evidence group D, not classifiable as to human carcinogenicity.

5.3.3.3 Lead

Lead, a naturally occurring blue-gray metal, is found in small quantities in the earth's crust and in all compartments of the biosphere. Both natural and anthropogenic processes are responsible for the distribution of lead throughout the environment, with gasoline combustion acting as a major source of lead found in the atmosphere and the earth's surface, through atmospheric deposition.

The toxicological data base for lead is unusual in that it contains a great deal of dose-effect data for humans. These data come primarily from studies of occupationally exposed groups and the general population. Most of the exposure or dose data are in terms of internal exposure, usually measured as levels of lead in the blood. Human exposure to lead occurs through a combination of inhalation and oral exposure, with inhalation contributing a greater proportion of the dose for occupationally exposed groups and the oral route contributing a greater proportion of the dose for the general population. In all work areas, the major route of lead exposure is by inhalation and ingestion of lead-bearing dusts and fumes. Airborne dust settles onto food, water, clothing, and other objects and may subsequently be transferred to the mouth. The effects of lead are the same regardless of entry into the body and are correlated with internal exposure as blood lead level.

Human epidemiologic studies have been inadequate in determining the carcinogenic effects of lead. Although several studies in occupational workers exposed to lead and lead compounds have been conducted, the available human evidence is considered inadequate to refute or demonstrate potential carcinogenicity for humans from lead exposure.

The carcinogenic potential of lead salts, primarily phosphate and acetates, administered orally or by injection, has been demonstrated in rats and mice by more than 10 investigators. The most characteristic cancer response is bilateral renal carcinoma. Most of the investigations found a carcinogenic response only at the highest dose.

The endpoints most sensitive to low-level exposure to lead are neurobehavioral deficits and growth retardation in young children and hypertension in middle-aged men. Effects on heme synthesis also occur at very low exposure levels.

5.3.3.4 Magnesium

Magnesium is the eighth most abundant element in the earth's crust. It is of absolute fundamental importance to all living organisms, as it is the metal center employed by chlorophyll. Without magnesium, no green plants would exist. Magnesium is widespread in all living organisms. Since magnesium is essential to all living organisms, there is virtually no danger of toxicity, except from oral exposure to magnesium salts by persons having kidney failure. When magnesium salts are ingested in large doses, side effects include vomiting or diarrhea, which limit the toxic effects (30). The cathartic dose of magnesium sulfate is 15 g. Approximately 300–400 mg/d or 5–6 mg/kg/d is needed for human growth. Inhalation of magnesium oxide results in metal fume fever, and powdered magnesium metal can burn the surface of the skin.

5.3.3.5 4-Nitrophenol

4-Nitrophenol is a man-made chemical with no evidence of its formation from any natural source. 4-Nitrophenol is used mainly to manufacture drugs, fungicides, and dyes, and to darken

leather. It is a colorless to light yellow solid with very little odor. 4-Nitrophenol degrades in water and surface soil, but the breakdown takes longer at lower soil depths and in groundwater.

Small amounts of 4-nitrophenol can be found in the air, water, and soil; however, background levels are not known. Breathing air, drinking water, and eating foods grown in soils that contain this substance can cause exposure to 4-nitrophenol. People who use certain pesticides or who drink well water near farming areas where certain pesticides are use may also be exposed to the substance.

4-Nitrophenol can enter the body through the lungs and pass into the blood stream if inhaled. When 4-nitrophenol is swallowed, most of it probably enters the body and passes from the stomach into the blood stream very quickly (in minutes). If 4-nitrophenol is spilled on the skin, some of it may pass through the skin into the blood stream, but it is not know how much or how fast. Once inside the body, 4-nitrophenol changes into other chemicals that will be released from the body through the urine within hours.

5.3.3.6 Dibenz(a,h)anthracene

Dibenz(a,h)anthracene is a chemical substance formed during the incomplete burning of fossil fuel, garbage, or any organic matter and is found in smoke in general; it condenses on dust particles and is distributed into water and soil and on crops. Dibenz(a,h)anthracene is a PAH and is also a component of coal tar pitch, which is used in industry as a binder for electrodes, and in creosote used to preserve wood. PAHs are also found in limited amounts in bitumens and asphalt used in industry and for paving. As a pure chemical, dibenz(a,h)anthracene resembles colorless plates or leaflets.

Humans may be exposed to dibenz(a,h)anthracene in the environment in tobacco smoke; food products such as cereals, vegetables, fruits, meats, and beverages; in the air if the home is heated with wood or coal; in some drinking water supplies; in the soils and air in the work place; and by contact with creosote-treated wood, asphalt roads, or coal tar.

Dibenz(a,h)anthracene enters the body primarily through breathing polluted air or inhaling tobacco smoke. Drinking contaminated water or accidentally swallowing soil or dust particles can also result in dibenz(a,h)anthracene entering the body. Dibenz(a,h)anthracene can be formed in food that is smoked or charcoal-broiled and can then enter the body through the digestive system. It does not normally enter the body through the skin; however, small amounts could enter the body through contact with soil containing high levels of dibenz(a,h)anthracene or through contact with heavy oils.

Dibenz(a,h)anthracene is a toxic chemical and has caused cancer in laboratory animals when it is ingested or applied to their skin. Because dibenz(a,h)anthracene causes cancer in animals, it is possible that humans exposed in the same manner could develop cancer as well. Dibenz(a,h)anthracene has not been well studied, and, consequently, it is not known whether breathing it could cause cancer or other harmful effects.

No information is available concerning the harmful effects of dibenz(a,h)anthracene in humans after breathing, eating, or skin application of this substance. The levels that caused tumors in animals that ingested dibenz(a,h)anthracene are unclear. Skin cancer has been found in mice that had 1.2×10^{-2} mg/kg/d applied to their skin throughout their lives. It is not know whether similar levels could cause cancer in humans.

5.4 RISK CHARACTERIZATION

The approach for characterizing the risk of adverse health effects from exposure to carcinogenic and noncarcinogenic contaminants is outlined in Sect. 5.4.1. The actual characterization is presented in Sects. 5.4.2 and 5.4.3.

5.4.1 Characterization Approach

The purpose of the human health risk assessment is to identify the contaminants and exposure pathways that present unacceptable risks of adverse health effects. In this risk assessment, contaminants that are present at concentrations that are considered unacceptable are designated contaminants of potential concern (COPCs). Sect. 5.3 provides general information and a contaminant-specific discussion on the types of health effects that result from exposure to the different COPCs.

5.4.1.1 Carcinogens

For carcinogens, it is assumed that a small number of molecular events can evoke a change in a single cell that can lead to uncontrolled cellular proliferation and eventually to a clinical state of disease. The hypothesized mechanism for carcinogenesis is referred to as "nonthreshold" or "stochastic" because there is believed to be essentially no level of exposure to a carcinogen that does not pose a finite probability, however small, of generating a carcinogenic response. Because of the way radiation affects cells, radionuclides are considered carcinogens.

To characterize potential carcinogenic effects, probabilities that an individual will develop cancer over a lifetime of exposure are estimated from projected intakes and chemical-specific dose response information (EPA RAGS/HHEM Part A 1989). The dose response information used for carcinogens in the present risk assessment is in the form of EPA-recommended slope factors (SFs) provided in the Health Effects Assessment Summary Tables (HEAST) and the Integrated Risk Information System (IRIS). A slope factor is a plausible upper-bound estimate of a response per unit intake of a carcinogen over a lifetime. The slope factors for nonradiological carcinogens convert estimated daily intakes of contaminants averaged over a lifetime (70 years) to an incremental risk of an individual developing cancer. For radionuclides, the slope factors in HEAST convert the total intake or exposure to an incremental risk of developing an excess cancer during a 70-year lifetime. Thirty years is considered a lifetime exposure for an adult (OSWER 1991). In the present risk assessment, only an adult was considered in evaluating the risk of an individual developing cancer because the parameter values used in the risk assessment equations for an adult are more conservative than those used for a child. Intake routes include ingestion, inhalation, dermal exposure, and, in the case of gamma-emitting radionuclides, external exposure to ionizing radiation.

Consideration is given to exposure to multiple chemicals as well as multiple exposure pathways when calculating the risk of an individual developing cancer. Although chemcial concentrations that represent an upper-bound excess lifetime cancer risk to an individual of between 10^{-4} and 10^{-6} are under some circumstances considered acceptable (55 FR 46), risks above 10^{-6} are undesirable. The risk to an individual should not exceed 10^{-4}.

5.4.1.2 Noncarcinogens

For many noncarcinogenic effects, protective mechanisms are believed to exist that must be overcome before an adverse effect is manifested from a chronic exposure to a toxicant. For example, where a large number of cells perform the same or similar function, the cell population may have to be significantly depleted before an effect is seen. A range of exposures exist, from zero to some finite value, that can be tolerated by the organism with essentially no change in expression of adverse effects. This is known as the "threshold" or "nonstochastic" concept.

To characterize potential noncarcinogenic effects from contaminants, projected intakes of substances are compared to EPA-approved toxicity values referred to as reference doses (RfDs). In this assessment, only long-term, chronic exposures to contaminants are evaluated. A chronic RfD is an estimate of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effect during a lifetime (EPA RAGS/HHEM Part A 1989). RfDs are provided in HEAST and IRIS. Noncarcinogenic effects are not expressed as the probability of an individual suffering an adverse effect as in the case of carcinogens but by comparing a daily exposure level averaged over a specified period of time with an RfD. The ratio of the average daily exposure level of a single toxicant to the RfD for that toxicant is defined as a hazard quotient (HQ). The sum of more than one HQ for multiple toxicants and/or multiple exposure pathways is called a hazard index (HI). An HQ or an HI $\geq$ 1 is considered unacceptable. Both a child and an adult were included in the assessment of noncarcinogens because in some instances the parameter values used for a child are more conservative than those used for an adult. Exposure to noncarcinogens can occur via ingestion, inhalation, and dermal contact.

Some contaminants that were detected in samples from LWBR do not have EPA-approved SFs or RfDs because sufficient data are not available for the Agency to derive toxicity values. Contaminants that were detected in LWBR but do not have toxicity values are discussed in Sect. 5.4.2 along with a brief description of the health effects they produce.

5.4.2 Current Land-Use Conditions (Primary Exposure Pathways)

Results of the human health risk assessment for the direct exposure pathways to primary media for current exposure scenarios (Fig. 5.1) are presented in the following order: consumption of drinking water, consumption of fish, exposure to shoreline sediment, and exposure to water while swimming. Brief descriptions of the types of analytes and number of samples in the data bases used in each scenario are given below. Summary statistics of the data are provided in Table A.1.

5.4.2.1 Water ingestion scenario

Water data from LWBR consisted of four grab samples collected during the CRRI Phase 1 sampling. These samples were analyzed for organic and inorganic analytes and radionuclides. None of the ~90 organic analytes that were included in the analyses were above detection limits (Table A.2). Most of the inorganic elements in the water were also below the level of analytical detection. Of the stable elements, only magnesium was present at concentrations above the level of detection. One radionuclide, ^{90}Sr, was above detection limits. No organic contaminants in the water, including PCBs and pesticides, were above the level of detection.

Because of the small number of water samples from LWBR and the time that would be required to collect and analyze additional samples, the FFA parties agreed that monitoring data from the Oak Ridge Surface Water Monitoring Program (Energy Systems 1992) would be used to supplement the LWBR surface water data. This program focuses its sample collection in the Clinch River arm of Watts Bar Reservoir. The supposition was that data taken immediately downstream of the ORR would reflect contamination originating from DOE sites, and contaminants should be easier to detect in water monitored near the source. Because these monitoring data do not reflect the dilution that occurs upstream of LWBR, they should be recognized as conservative and indicative of "worst-case" conditions.

The ORR surface water monitoring program data consisted of six bimonthly grab samples collected at Brashear Island (CRM 9.9) from April through November 1993 by ORNL personnel. These samples were analyzed for inorganics, radionuclides, volatile and semi-volatile compounds, and pesticides/PCBs (DOE/OR-1066 1992). Volatile contaminants were not included in the risk assessment because they would dissipate before reaching LWBR. Only gross alpha counts were available for quantifying alpha emitters in these samples. Had the gross alpha counts exceeded 3 pCi/L (0.11 Bq/L), isotope-specific analyses would have been conducted for the uranium isotopes and other alpha emitters.

Human health risks for the water ingestion scenario were calculated separately for the Phase 1 data and the monitoring data. The methods, equations, and parameter values in the human health risk assessment for radiological and nonradiological contaminants in drinking water are listed in Table A.6. Strontium-90 was the only carcinogen that had concentrations greater than detection limits in LWBR water (Tables A.1 and A.2). The excess lifetime cancer risk from the ingestion of ^{90}Sr in LWBR water by an adult was 5×10^{-7}, which is considered acceptable. The ^{90}Sr activity, intake and exposure rates, and cancer slope factor are given in Table B.1.

Human health risk calculations for the ingestion of water were also made for carcinogens and noncarcinogens reported for the surface water samples collected at the Brashear Island monitoring station at CRM 9.9. Total uranium and ^{137}Cs, two of the radionuclides that were measured in the monitoring samples, produced lifetime cancer risks for an adult of 1.8×10^{-6} and 1.7×10^{-6}, respectively (Table B.2).

However, if dilution of the Clinch River water by Tennessee River water was considered, the risk values would be acceptable. Uranium was reported as total uranium because the monitoring samples are analyzed for separate uranium isotopes only if the alpha activity exceeds 3 pCi/L (0.11 Bq/L). Total uranium does not have a slope factor but because over 99% of naturally occurring uranium is ^{238}U, the slope factor for ^{238}U was used in the human health risk calculations. The chemical concentration (mg/L) of uranium in the water was also measured; however, the HQ was much less than 1.0 for both a child and an adult. One noncarcinogen, manganese, had an HQ of 1.1 for a child (Table B.3.); however, the RfD is not necessarily acceptable if the intake is from drinking water alone because of the possibility of differential bioavailability (Francis 1991). Contaminants for the water ingestion scenario with excess lifetime cancer risks $\geq 10^{-6}$ and HQs ≥ 1.0 are given in Table 5.3.

Table 5.3. Contaminants of potential concern in the water ingestion scenario

Contaminant	Ingestion
Carcinogens in drinking water scenario with cancer risks $\geq 10^{-6}$	
Radionuclides (ORR monitoring data)[a]	
^{137}Cs	1.7×10^{-6}
Uranium (total)	1.8×10^{-6}
Noncarcinogens in drinking water scenario with hazard quotients ≥ 1.0	
Inorganic toxicants (ORR monitoring data)[b]	
Manganese	1.1 (child)

[a]Risks from Lower Watts Bar Reservoir Phase 1 data were $< 10^{-6}$.

[b]Hazard quotients from LWBR Phase 1 data were < 1.0.

5.4.2.2 Fish consumption scenario

Sport fishing is the primary means of catching fish in LWBR as well as the primary means of obtaining local fish for food. Commercial fishing in LWBR is limited, primarily because the use of gill and trammel nets has been prohibited by state regulations (Tennessee Wildlife Resources Commission 1991). Approximately 200,000 lbs of three species of commercial fish were collected from LWBR in 1967 (Cowser and Snyder 1966). Most of these fish were shipped to markets out of state. In 1977, while gill and trammel nets were still permitted, 15 to 20 full-time and some part-time commercial fishermen were employed in LWBR (Heitman 1979).

Currently, there is a fish advisory limiting the consumption of fish from LWBR. The contaminants of concern for the advisory are PCBs. A screening level health risk assessment conducted for the CRRI using historical data concluded that PCBs in fish from LWBR were contaminants of concern (Hoffman et al. 1991). PCBs have been released from the ORR (Loar et al. 1992); however, PCBs also occur in many streams and lakes in eastern Tennessee.

The data base used for the ingestion of fish scenario consisted of recently collected Phase 1 data which was supplemented with data collected by the TVA. Because not all species of fish accumulate contaminants to the same extent, not all species of fish were analyzed for the same contaminants. The analyses were designed not only to identify the contaminants of concern but also to focus on the species with the highest concentrations. Bluegill were analyzed for inorganics and radionuclides; catfish and largemouth bass were analyzed for inorganics, organics, and radionuclides; and striped and hybrid bass were analyzed for organics.

Aroclor-1260 and Aroclor-1254 were analyzed to determine the concentration of PCBs in fish tissue because these two chemicals mixtures are the major contributors to the total PCB

concentration (Sect. 5.3.1.8). Emphasis was placed on analyzing catfish for PCBs because of their relatively high lipid content, which is directly related to their propensity for accumulating PCBs. Striped bass, a species recently introduced into Watts Bar Reservoir, were analyzed for PCBs and organic pesticides and herbicides because of their high lipid content, large size, and use for food.

The equations and parameter values used for calculating the human health risk assessment for carcinogens and noncarcinogens for the fish consumption exposure pathway are given in Table A.13. COPCs for each species are shown in Table 5.4. Separate risks were calculated for the ingestion of different species of fish to provide risk factors for individuals who eat only one species of fish. The EPA-recommended ingestion rate of 54 g of fish per day was used to calculate the risk for each species; therefore, the risk for each species should not be added to obtain a risk to an individual who eats all species. Instead, the risk for the species that has the maximum value should be used as the upper-bound risk for the ingestion of fish.

Ten contaminants detected in fish fillets produced cancer risks $> 10^{-6}$ (Table 5.4). All species of fish for which PCBs (Aroclor-1254 and Aroclor-1260) were analyzed had calculated cancer risks $> 10^{-4}$. The excess lifetime cancer risks for PCBs in catfish and striped bass were $> 10^{-3}$. The pesticides (aldrin, chlordane, and dieldrin) were carcinogens with cancer risks $> 10^{-5}$. At one time, these pesticides were commonly used in residential, farming, and industrial areas; therefore, they are not unique to the ORR but are ubiquitous contaminants in eastern Tennessee streams and reservoirs.

Aldrin and chlordane, which are classified as toxicants as well as carcinogens, had HQs exceeding 1.0 for the ingestion of fish (Table 5.4). The highest HQ in the fish pathway was for the ingestion of chlordane in catfish by a child. The ingestion of arsenic and mercury in fish also produced HQs > 1.0 for a child; however, these results were based on analyses of only five largemouth bass and two catfish. The HQs for mercury and arsenic in bluegill, based on 20 samples, were < 1.0 for a child (Table B.11). The HQs for the ingestion of mercury and arsenic in all species of fish by an adult were < 1.0.

Details of the human health risk calculations for the ingestion of fish are given in Table B.10 for carcinogens and Table B.11 for noncarcinogens. These tables include the cancer risks and HQs for each contaminant in the data base for which at least one measurement was above the level of detection. The tables also include the cancer slope factors and RfDs for the listed contaminants.

5.4.2.3 Shoreline-use scenario

The potential pathways for exposure to contaminants in shoreline sediment exposed during low water levels in winter or in flood plain and beach area sediment during the summer months are shown in Fig. 5.1. The equations and parameter values used for the human health risk assessment are given in Appendix A for exposure to: (1) nonradiological carcinogens, noncarcinogens, and radionuclides from the inhalation of resuspended sediment (Table A.9); (2) nonradiological carcinogens and noncarcinogens during dermal contact with near-shore sediment (Table A.10); (3) radiation from gamma-emitting radionuclides in near-shore sediments (Table A.11); and (4) carcinogens and noncarcinogens from the inadvertent ingestion of near-shore sediment (Table A.12).

Table 5.4. Contaminants of potential concern in the consumption of fish scenario

Contaminant	Species			
	Bluegill	Catfish	Largemouth bass	Striped or hybrid bass
Carcinogens in the fisherman scenario with cancer risks ≥ 10^{-6} for an adult				
		Radionuclides		
^{137}Cs	1.6×10^{-6}	1.9×10^{-6}		
		Inorganics		
Beryllium			2.9×10^{-5}	
		Organics		
4,4'-DDE		4.4×10^{-6}		7.8×10^{-6}
4,4'-DDT		1.4×10^{-6}		1.6×10^{-6}
Aldrin		8.4×10^{-5}		
Aroclor-1254		1.3×10^{-3}	5.3×10^{-4}	1.4×10^{-3}
Aroclor-1260		2.5×10^{-3}	6.0×10^{-4}	1.8×10^{-3}
Chlordane		7.0×10^{-5}	3.5×10^{-5}	5.5×10^{-5}
Dieldrin		9.4×10^{-5}		5.1×10^{-5}
Lindane		4.6×10^{-6}		
Noncarcinogens in the fisherman scenario with hazard quotients ≥ 1.0				
		Organic toxicants		
Aldrin		1.0 (child)		
Chlordane		2.1 (adult)	1.0 (adult)	1.7 (adult)
		Inorganic toxicants		
Arsenic			1.1 (child)	
Mercury		1.2 (child)	1.5 (child)	

The near-shore sediment data consisted of samples collected at water depths of less than 6 ft when the reservoir was at summer pool level. The data base includes Phase 1 and post-Phase 1 near-shore data and TVA data for samples collected at beach areas and water intake structures. The Phase 1 and post-Phase 1 data consist of grab samples collected near the shore line that were analyzed for only gamma-emitting radionuclides. The TVA samples from public use areas were analyzed for organic and inorganic contaminants as well as certain radionuclides.

Results of the risk characterization using the Phase 1 near-shore sediment data are given in Table B.7, and results using the TVA public use area data are given in Tables B.8 and B.9. One radionuclide and two inorganic analytes in the near-shore sediment were COPCs for the shoreline-use scenario (Table 5.5). External radiation from exposure to ^{137}Cs produced an excess lifetime cancer risk of 4.4×10^{-6} based on the Phase 1 near-shore sediment data. Beryllium and chromium found in the TVA samples produced excess lifetime cancer risks $> 10^{-6}$ for the ingestion and inhalation exposure pathways. All HQs were < 1.0.

Table 5.5. Contaminants of potential concern in the shoreline-use scenario

Contaminant	Ingestion (sediment)	External (sediment)	Dermal (sediment)	Inhalation (resuspended sediment)
Carcinogens-in-the-beach-area scenario with cancer risks $\geq 10^{-6}$				
Radionuclides (Phase 1 data, ORR monitoring data)				
^{137}Cs		4.4×10^{-6a} (adult)	NA	
Inorganics (TVA recreational data)				
Beryllium	1.4×10^{-6} (adult)			
Chromium				2.8×10^{-6} (adult)
Noncarcinogens shoreline-use scenario with hazard quotients ≥ 1.0				
Inorganic toxicants (TVA recreational data)				
None				

[a]Sediment concentrations used in the calculation of this risk estimate are from the Phase 1 data.
NA = Route of exposure is not applicable for the contaminant.

5.4.2.4 Swimming scenario

The swimming scenario considers the potential risks of exposure to contaminants in water while swimming in LWBR and at Brashear Island (Fig. 5.1). The potential routes of exposure while swimming are dermal contact and inadvertent ingestion of water. Equations and parameter values used in the human health risk assessment for exposure to carcinogens and noncarcinogens in the pathways of the swimming scenario are given in Tables A.7 through A.8.

5.4.2.4.1 Exposure to water while swimming

The water measurements that were used for the drinking water scenario were used for the swimming scenario. Only radionuclides and inorganic contaminants were present in the water at

detectable concentrations. If the radionuclides in the water posed a health risk in the swimming scenario, it would be from radiation either through external exposure or water ingestion rather than dermal absorption. EPA does not provide slope factors for external exposure to radiation from immersion in water; however, a radiation dose from immersion in water can be calculated and converted to risk. The results of this calculation showed that the excess lifetime cancer risk from exposure to radionuclides while swimming at either the Brashear Island sampling station or in LWBR would be $< 10^{-15}$, clearly an insignificant risk. For this reason, the risk of health effects from external radiation while swimming was not considered in the present risk assessment.

Results of the calculations for the risk of adverse health effects from radionuclides and inorganic contaminants in water while swimming are reported in Tables B.4 and B.5 for the Brashear Island monitoring station at CRM 9.9 and Table B.6 for LWBR. None of the radionuclides or inorganic analytes in the water produced unacceptable health risks for either an adult or a child. Magnesium, a natural constituent of surface waters, was detected in the surface water samples from LWBR and Brashear Island (Table A.1), but neither an RfD nor a cancer slope factor was available from EPA (1993) to use in calculating health hazards. However, magnesium is an essential element, and there is virtually no danger of toxicity (Sect. 5.3.3.4).

5.4.3 Future Land-Use Conditions (Secondary Exposure Pathways)

5.4.3.1 Irrigation scenario

LWBR water is not widely used for irrigation purposes because of the amount of rainfall that occurs in eastern Tennessee and the types of crops grown in the area. However, there is a limited amount of irrigation used for lawns, pastures, and vegetable gardens. Irrigation is considered a future-use assessment because current use is limited but could increase in the future, and contaminant concentrations are modeled values representing a 30-yr build-up period.

The Phase 1 water data for LWBR consisted of four grab samples. These samples had a limited number of contaminants detected (Tables A.1 and A.2). Because of the small number of water samples from LWBR, data from the ORR surface water monitoring program for CRM 9.9 near Brashear Island were also used in the irrigation scenario. These are the same data bases that were used for the drinking water scenario described earlier. Clinch River water is usually diluted by a factor of 3 to 5 when it joins the Tennessee River, and, after mixing in LWBR, concentrations of contaminants from the Clinch River should be reduced by the same factor. However, for the irrigation scenario, concentrations from the surface water monitoring program at CRM 9.9 were not reduced by a dilution factor. Furthermore, the mathematical model that was used to calculate the concentration of contaminants in the irrigated soil (Table C.1) allowed the contaminants to build up in the soil for 30 years before crops were grown, making the results highly conservative.

The exposure pathways associated with the irrigation scenario are shown in Fig. 5.1. The methodology for calculating the concentrations of contaminants in the irrigated soil and produce is provided in Appendix C. The associated HQs for noncarcinogens and cancer risks for carcinogens, including radionuclides, are given in Appendix A for the following exposure pathways: inhalation of resuspended irrigated soil (Table A.9); dermal contact with irrigated soil (Table A.10); external exposure to radionuclides in irrigated soil (Table A.11); inadvertent ingestion of irrigated soil (Table A.14); and ingestion of vegetables, milk, and meat produced on irrigated soil (Table A.15).

COPCs for the irrigation scenario are shown in Table 5.6. Two radionuclides measured in the Brashear Island water samples, ^{137}Cs and ^{60}Co, produced unacceptable lifetime cancer risks for several exposure pathways, and total uranium produced a cancer risk $> 10^{-6}$ when all exposure pathways were summed (Table B.19). Both ^{137}Cs and ^{60}Co produced excess lifetime cancer risks $> 10^{-6}$ for external exposure to irrigated soil. Cesium-137 also produced cancer risks $> 10^{-6}$ for the ingestion of milk and meat. If a dilution factor of three or more (representing the Tennessee River input into LWBR) were applied to the irrigation water, the cancer risk estimates would be acceptable. Detectable quantities of six inorganic contaminants were found in the water from Brashear Island, but none produced HQs ≥ 1.0. Results of the risk assessment using the surface water monitoring data for Brashear Island are given in Tables B.13 and B.14. The only detected contaminant in the LWBR water was ^{90}Sr, and it did not produce an unacceptable cancer risk in the irrigation scenario exposure pathways (Table B.12).

5.4.3.2 Dredging scenario

Historical and recently-collected data indicate that the highest concentrations of ^{137}Cs and mercury in the sediment of LWBR are buried under 40 to 80 cm (1 to 2 ft) of less-contaminated sediment in depositional zones along the old river channel (Energy Systems 1992, Olsen et al. 1992). This situation occurs because releases of contaminants from the ORR have decreased significantly since the 1950s and 1960s (Chap. 2) and the more recently deposited sediments are much less contaminated. As long as the sediments with the highest levels of contamination remain buried, there is no realistic exposure pathway for humans and, therefore, no risk to human health. The most viable pathway for human exposure would result from dredging the deep sediment to maintain a navigational channel or for other reasons. If the dredged spoils are placed on the shore, they provide a potential pathway for human exposure. Because dredging is an action that could occur in the future, a dredging scenario was used to evaluate the health risks associated with using deep sediments for agricultural purposes. This exposure scenario assumes worst-case conditions. Pathways for the dredging scenario are shown in Fig. 5.1. Contaminant concentrations for milk, meat, and vegetable produced on dredged sediment were calculated in the same way as those produced on irrigated soil (Appendix C). The methodology for calculating health risks for the different exposure pathways are given in Tables A.9 through A.12 and Table A.15.

The data base for sediment cores from deep-water depositional zones in LWBR was composed of Phase 1 core sample data (Cook et al. 1992), data collected in a scoping study for the CRRI (Olsen et al. 1992), and historical data from samples collected during the TVA Instream Contaminant Study (TVA 1986a). The core samples collected by Olsen et al. (1992) were analyzed for gamma-emitting radionuclides and a few were also analyzed for mercury. The Phase 1 data and the TVA data provided a limited number of analyses for organic and inorganic contaminants and radionuclides.

Results of the health risk calculations for contaminants with excess lifetime cancer risks $\geq 10^{-6}$ or HQs ≥ 1.0 are given in Table 5.7. Only two radionuclides, ^{137}Cs and ^{60}Co, had excess lifetime cancer risks $\geq 10^{-6}$ for individual exposure pathways; however, ^{238}U had a lifetime cancer risk of 1.4×10^{-6} when all exposure pathways were summed (Table B.21). In the dredging scenario, no individual contaminant had an excess lifetime cancer risk $\geq 10^{-4}$, but the sum of the risks for all contaminants and exposure pathways was slightly $> 10^{-4}$ (Table B.21). Three inorganic carcinogens, arsenic, beryllium, and chromium, had excess lifetime cancer risk $> 10^{-6}$, which made a significant contribution to the total cancer risk. The cancer risks for ^{137}Cs for the

Table 5.6. Contaminants of potential concern in the irrigation scenario

Route	Ingestion (Soil)	External (Soil)	Dermal (Soil)	Inhalation (Resuspended Soil)	Ingestion (Milk)	Ingestion (Meat)	Ingestion (Vegetables)
Carcinogens in the irrigation scenario with cancer risks _ 10^{-6} for an adult							
radionuclides (ORR Monitoring data)							
^{137}Cs		1.1×10^{-6}	NA		1.5×10^{-6}	1.0×10^{-6}	
^{60}Co		1.5×10^{-6}	NA				

NA = Route of exposure is not applicable for the contaminant.

Table 5.7. Contaminants of potential concern in the dredging scenario

Contaminant	Ingestion (Sediment)	External (Sediment)	Inhalation (Resuspended Sediment)	Ingestion (Milk)	Ingestion (Meat)	Ingestion (Vegetables)
Carcinogens in the dredging scenario with cancer risks ≥ 10^{-6} for an adult						
radionuclides						
Cs-137		6.4×10^{-5}		1.4×10^{-5}	9.1×10^{-6}	1.3×10^{-6}
Co-60		1.3×10^{-6}				
inorganic carcinogens						
Arsenic			2.2×10^{-6}			
Beryllium	3.6×10^{-6}				2.0×10^{-5}	3.0×10^{-6}
Chromium			2.7×10^{-6}			
Noncarcinogens in the dredging scenario with hazard quotients ≥ 1.0						
Contaminant	Ingestion (Sediment)	External (Sediment)	Inhalation (Resuspended Sediment)	Ingestion (Milk)	Ingestion (Meat)	Ingestion (Vegetables)
inorganic toxicants						
Cadmium		NA		4.2 (child)		2.9 (child)
Chromium		NA		1.4 (child)	1.9 (child)	
Mercury		NA		1.3 (child)	2.3 (child)	3.2 (child)
Zinc		NA		3.4 (child)	3.1 (child)	

NA = Route of exposure is not applicable for the contaminant.

external exposure and milk ingestion pathways and beryllium in the meat ingestion pathway all exceeded 10^{-5}. Results of the risk characterization for the contaminants in the dredging scenario along with their cancer slope factors and RfDs are given in Tables B.15 and B.16. The greatest health hazards from noncarcinogens in the dredging scenario are hazards to a child. Four noncarcinogens (cadmium, chromium, mercury, and zinc) had HQs ≥ 1.0 in two or more exposure pathways for a child (Table 5.7). Although no individual contaminant had an HQ ≥ 1.0 in the adult exposure pathway, the multiple pathway totals (HIs) for mercury, cadmium, and zinc exceeded 1.0 (Table B.21).

5.4.4 Special Case—Lead

Measurable concentrations of lead were found in sediment from LWBR; however, EPA toxicity values (RfDs) for lead are not presently available for risk assessment purposes. Although the concentrations of lead in the sediment samples approximated concentrations in background soils (Energy Systems 1993), lead was included in the risk assessment because of the public's concern with lead toxicity. An EPA uptake/biokinetic model was used to evaluate the health risk from exposure to lead. This model uses site-specific, multimedia input values to estimate levels of lead in blood. Young children are more sensitive to lead toxicity than adults; therefore, the model was used to estimate the blood lead levels for a population of children 0–6 years of age. The critical blood lead level for children is 0.010 mg/dL of blood. EPA Region IV considers the population to be at risk if more than 5% of the children have blood lead levels that exceed the critical value.

The model allows for input of lead concentrations for soil, dust, food, and water. No data were available for lead concentrations in LWBR water; therefore, a default value of 4 μg/L was used. Lead concentrations were available for the sediment and these were used for soil and dust in the model. Calculated values for the concentration of lead in vegetables and meat from the dredging scenario were also used in the model. The model estimated that essentially the entire population (>99%) of children 0–6 years of age would have blood lead levels below the 0.010 mg/dL critical value. Therefore, no adverse health effects would be expected to result from exposure to lead in the LWBR environment.

5.4.5 Risk Characterization Summary

COPCs for all exposure scenarios are summarized in Tables 5.8 and 5.9. Detailed results of the risk calculations are provided in Appendix B. The fish ingestion pathway was the most significant exposure pathway in terms of adverse health effects. Fish contained the only contaminants with cancer risks $> 10^{-4}$: the two PCBs mixtures, Aroclor-1254 and Aroclor-1260, had excess lifetime cancer risks $> 10^{-3}$ for the ingestion of channel catfish and striped bass and $> 10^{-4}$ for largemouth bass. The number of fish samples and their spatial distribution appear to be sufficient to be representative of PCB concentrations in these three species of LWBR fish.

PCBs are known to be released from the ORR; however, it is well documented that all major rivers and reservoirs of eastern Tennessee are contaminated with PCBs (TVA 1988). Although past releases from ORR have contributed to the PCB concentrations found in LWBR, it has not been established that ORR is the major contributor.

Mercury, which has been released in large quantities from the Y-12 Plant facilities into the Clinch River via East Fork Poplar Creek and Poplar Creek (Chap. 2), produced an HQ of 1.5 for the ingestion of largemouth bass and 1.2 for the ingestion of catfish by a child 3 to 6 years of age; however, these HQs were based on analyses of five and two fish, respectively. The HQ

for the ingestion of mercury in bluegill by a child was 0.4, based on 20 samples. For adults, the HQ was <1.0 for all species. The number of bluegill that were analyzed for mercury appears to be sufficient to provide a reliable estimate of their mercury concentration.

Table 5.8. Contaminants of potential concern: contaminants with cancer risks $\geq 10^{-4}$ or hazard quotients ≥ 1.0

Contaminant	Cancer risk/HQ[a]	Exposure pathways
	Carcinogens	
	Organic carcinogens	
Aroclor-1254	1.3×10^{-3} (adult)	Fish ingestion
	7.2×10^{-4} (child)	Fish ingestion
Aroclor-1260	2.5×10^{-3} (adult)	Fish ingestion
	1.2×10^{-3} (child)	Fish ingestion
	Noncarcinogens	
	Organic toxicants	
Aldrin	1.0 (child)	Fish ingestion
Chlordane	5.4 (child)	Fish ingestion
	2.1 (adult)	Fish ingestion
	Inorganic toxicants	
Arsenic	1.1 (child)	Fish ingestion
Cadmium	2.9 (child)	Dredging—ingestion of vegetables
Chromium	1.9 (child)	Dredging—ingestion of meat
Manganese	1.0 (child)	Drinking surface water[b]
Mercury	1.5 (child)	Fish ingestion
	2.3 (child)	Dredging—ingestion of meat
	3.2 (child)	Dredging—ingestion of vegetables
Zinc	3.4 (child)	Dredging—ingestion of milk
	3.1 (child)	Dredging—ingestion of meat

[a]Values listed in this column for carcinogens are cancer risks; values listed for noncarcinogens are hazard quotients (HQs).
[b]ORR water monitoring data.

Table 5.9. Contaminants with cancer risks between 10^{-4} and 10^{-6}

Contaminant	Cancer risk	Exposure pathways
		Radionuclides
^{60}Co	1.3×10^{-6}	Dredging—external exposure
	1.5×10^{-6}	Irrigation—external exposure[a]
^{137}Cs	1.7×10^{-6}	Drinking water[a]
	4.4×10^{-6}	Shoreline use—external exposure
	1.9×10^{-6}	Fish ingestion
	1.1×10^{-6}	Irrigation—external exposure[a]
	1.5×10^{-6}	Irrigation—ingestion of milk[a]
	1.0×10^{-6}	Irrigation—ingestion of meat[a]
	6.4×10^{-5}	Dredging—external exposure
	1.4×10^{-5}	Dredging—ingestion of milk
	9.1×10^{-6}	Dredging—ingestion of meat
	1.3×10^{-6}	Dredging—ingestion of vegetables
Uranium, total	1.8×10^{-6}	Drinking water[a]
		Organics
4,4'DDE	4.4×10^{-6}	Fish ingestion
4,4'DDT	1.4×10^{-6}	Fish ingestion
Aldrin	8.4×10^{-5}	Fish ingestion
Chlordane	7.0×10^{-5}	Fish ingestion
Dieldrin	5.1×10^{-5}	Fish ingestion
Lindane	4.6×10^{-6}	Fish ingestion
		Inorganics
Beryllium	1.4×10^{-6}	Shoreline-use—ingestion of sediment
	2.9×10^{-5}	Fish ingestion
	3.6×10^{-6}	Dredging—ingestion of sediment
	2.0×10^{-5}	Dredging—ingestion of meat
	3.0×10^{-6}	Dredging—ingestion vegetables
Chromium	2.8×10^{-6}	Shoreline-use—inhalation of sediment
	2.7×10^{-6}	Dredging—inhalation of sediment

[a]ORR water monitoring data

Site-specific fish consumption rates were not available for the LWBR area; consequently, an EPA default consumption rate of 30 g/day for 350 days/year (~ 1/2 lb/week) was used to calculate the HQ for a child 3 to 6 years of age (Table A.13). A sensitivity analysis conducted in conjunction with the uncertainty analysis for the fish ingestion exposure pathway (Sects. 5.5.1.2 and 5.5.1.4) identified the consumption rate as a parameter that contributes a high degree of uncertainty to the risk estimates. Although the sensitivity analysis was conducted for an adult, the results should be relevant to a child as well.

Arsenic and two pesticides (aldrin and chlordane) also had HQs > 1.0 for the ingestion of fish by a child. The unacceptable HQ for arsenic was based on data from 5 largemouth bass; however, the HQs for arsenic in bluegill (20 fish) and catfish (2 fish) were acceptable (< 1.0). Chlordane produced unacceptable HQs for an adult as well as a child for all species for which data were available. Bluegill were not analyzed for chlordane. Aldrin had an HQ of 1.0 for the ingestion of catfish by a child based on the mean concentration in 40 fish. These pesticides were formerly used in residential and farming areas as well as industrial sites and are common to eastern Tennessee rivers and reservoirs that are not influenced by releases from the ORR. Arsenic contamination of aquatic environments is often associated with the use of fossil fuel to produce electricity, and arsenic is a common pollutant in most industrial areas (TVA 1980). The RfD for chlordane has a high degree of uncertainty (Sect. 5.4.1.3), which means the HQ for chlordane has considerable uncertainty. Although the ORR is known to have released arsenic and perhaps aldrin and chlordane into the Clinch River, it is not necessarily the major source.

Relatively large releases of radionuclides from the ORR have occurred in the past. The excess lifetime cancer risk from ^{137}Cs was $> 10^{-6}$ for one or more exposure pathways in all the assessment scenarios, except swimming. The dredging scenario had the greatest multiple pathway risk for ^{137}Cs, with a total excess lifetime cancer risk of 8.8×10^{-5} (Table B.21). The highest risk (6.4×10^{-5}) resulted from external exposure in the dredging scenario. The assumptions and parameter values in the equations for the dredging scenario make the results for this scenario very conservative. For most pathways, the excess lifetime cancer risk for ^{137}Cs was between 10^{-5} and 10^{-6}, which translates to an effective whole-body dose of between 0.2 and 0.02 mrem/year. Background radiation from natural radionuclides along the Clinch River is ~ 120 mrem per year; therefore, the calculated risks from ^{137}Cs in the various pathways are judged to be insignificant.

Because only four surface-water samples were collected in LWBR during Phase 1, the drinking water and irrigation scenarios were also assessed using data from water samples collected from the Clinch River at Brashear Island (CRM 9.9). The calculated excess lifetime cancer risks from uranium and ^{137}Cs in the water ingestion pathway were 1.8×10^{-6} and 1.7×10^{-6}, respectively, for an adult drinking untreated surface water from the Clinch River at CRM 9.9. This analysis represents a worst case situation; however, if a dilution factor of three or more is assumed when the Clinch River water enters LWBR, the cancer risks would be acceptable. Although this pathway could occur, it is highly unlikely that an individual would consistently drink untreated river water.

In the irrigation scenario, ^{137}Cs had excess lifetime cancer risks $> 10^{-6}$ for external exposure to irrigated soil and ingestion of milk and meat, and ^{60}Co had a cancer risk $> 10^{-6}$ for external exposure to soil based on Clinch River water data (Table B.19). Again, a dilution factor of three or more would result in acceptable cancer risk values. The HQs for all noncarcinogens were < 1.0 for both a child and an adult. The excess lifetime cancer risk from ^{90}Sr measured in the Phase 1 water samples was $< 10^{-7}$ for all exposure pathways combined (Table B.20).

In the shoreline-use scenario, the TVA public use area data for surface sediment, chromium and beryllium had excess lifetime cancer risks $> 10^{-6}$ for the sediment inhalation and ingestion exposure pathways, respectively (Table B.8). No other carcinogens or noncarcinogens reported in these data sets were considered COPCs in the shoreline-use scenario; however, the Phase 1 near-shore sediment produced an excess lifetime cancer risk of 4.4×10^{-6} for external exposure to ^{137}Cs in the shoreline-use scenario (Table B.7). The risk from external exposure to ^{137}Cs in sediment from the TVA public use areas was 6.3×10^{-7}, one-tenth of the risk from the Phase 1 near-shore sediment. This difference is probably because many of the TVA public use areas are in or near tributaries where the water flow inhibits accumulation of contaminants from the reservoir, whereas the Phase 1 near-shore samples were collected all along the shoreline. Also, the Phase 1 data consisted of 352 samples, as opposed to 11 TVA public use area samples; consequently, there were more opportunities to sample areas of sediment accumulation.

In the dredging scenario, the estimated cancer risks for ^{137}Cs and beryllium were $> 10^{-5}$ for several exposure pathways. The excess lifetime cancer risks from ^{137}Cs were 6.4×10^{-5} and 1.4×10^{-5} for the external exposure and milk ingestion pathways, respectively (Table B.21). Other exposure pathways for beryllium, chromium, ^{137}Cs, and ^{60}Co had cancer risks $> 10^{-6}$, and the cancer risk for the sum of all ^{238}U pathways exceeded 10^{-6}. The sum of the excess lifetime cancer risks for all pathways and carcinogens in the dredging scenario was 1.2×10^{-4}. Four noncarcinogens—cadmium, chromium, mercury, and zinc—had HQs > 1.0 for a child in several foodchain exposure pathways of the dredging scenario. No individual contaminant had a HQ > 1.0 for an adult in the dredging scenario, although the multiple pathway totals for mercury, cadmium, and zinc exceeded 1.0.

The milk, meat, and vegetable pathways in the dredging scenario were the most important pathways in terms of health effects for noncarcinogens and also contributed to unacceptable cancer risks; however, the concentrations that were used to calculate the cancer risks and HQs for these foods were modeled concentrations which have a high degree of uncertainty and are usually conservatively biased. Data for ^{137}Cs represented 60 sediment cores, but the data for cadmium, chromium, mercury, and zinc were based on only 4 to 6 cores, which adds considerable uncertainty to risk assessment. The most significant COPC for the dredging scenario, in terms of adverse health effects, was ^{137}Cs and the most significant exposure pathway was external exposure. Although cadmium, chromium, mercury, and zinc had hazards quotients > 1.0 in the milk, meat, and vegetable pathways, the uncertainty associated with the small number of core samples and the use of modeled concentrations suggests that the results are also highly uncertain.

5.5 QUANTITATIVE UNCERTAINTY ANALYSIS

This section presents the results of a quantitative uncertainty analysis for two exposure scenarios: consumption of fish and the use of dredged deep-water sediments for agricultural purposes. The fish consumption scenario was chosen because it is the most significant exposure pathway by which humans are presently exposed to contaminants in LWBR. Although dredging of deep-water sediments is not occurring at the present time, the dredging scenario was chosen because radionuclides and heavy metals have been released from the ORR in the past and are known to have accumulated in the sediment. The following factors were considered in selecting the contaminants that were included in the uncertainty analysis: 1) the adequacy of the data, 2) whether the ORR was a potential source of the contamination, and 3) an unacceptable risk of adverse health effects was indicated in the baseline risk assessment. Because the ORR is the

primary source of ^{137}Cs in LWBR and sufficient data were available, ^{137}Cs was included in the uncertainty analysis for the dredging scenario.

Monte Carlo simulation was used to propagate the subjective probability distributions for the parameters involved in the baseline risk assessment into a subjective probability distribution for both the excess lifetime cancer risk for carcinogens and the hazard quotient (HQ) and hazard index (HI) for noncarcinogens. The sampling method used for the Monte Carlo simulation was Latin Hypercube Sampling (LHS).

The inputs required for the Monte Carlo simulations are 1) subjective probability distributions and 2) uncertainty bounds for each parameter presented in the risk assessment (Appendix D). Using various input distributions, the Monte Carlo simulation provides an estimate of the excess lifetime cancer risk and the HQ in terms of a subjective probability distribution. From this distribution, a subjective confidence interval for the true but unknown result is obtained. *The term "subjective confidence interval" means that the probability distributions specified for the uncertain model parameters were derived using subjective judgment in the absence of definitive data.* The subjective confidence intervals for the excess lifetime cancer risks and the HQs are presented in this section.

In addition to the subjective confidence intervals, the results of a sensitivity analysis are also presented in this section. A sensitivity analysis permits the identification of the parameter having the greatest effect on the total result. The sensitivity analysis was accomplished by using the square of the Spearman rank correlation coefficients adjusted to 100% (Decisioneering, Inc. 1994). The results obtained for each parameter were compared and ranked according to the amount of influence the parameter had on the uncertainty in the result. The parameters having the greatest effect are considered to be the most sensitive. Although not employed in this analysis, scatter plots of the input parameters and statistical regression techniques are other methods of performing sensitivity analyses (Iman et al. 1981a, 1981b; IAEA 1989; Iman and Helton 1991).

5.5.1 Uncertainty analysis for the fish consumption scenario

Contaminants that were included in the uncertainty analysis for the fish ingestion pathway are ^{137}Cs, Aroclor-1260, Aroclor-1254, chlordane, dieldrin, mercury, and arsenic. These contaminants were included because the data are adequate, the ORR is a potential source of these contaminants, and/or the human health risk assessment indicates that they present unacceptable health risks. Two situations were evaluated in the uncertainty analysis. In the first, slope factors and reference doses for all contaminants were fixed at the EPA recommended values (IRIS and HEAST). In the second, uncertainties associated with the slope factors for Aroclor-1260, Aroclor-1254, chlordane, and ^{137}Cs and with the reference doses for mercury and chlordane were propagated through the uncertainty analysis. Information concerning the uncertainties associated with the slope factor for dieldrin and the reference dose for arsenic was not available; therefore, probability distributions were not assigned to these parameters.

5.5.1.1 Subjective confidence intervals for the excess lifetime cancer risk from ^{137}Cs and selected organic carcinogens in fish

The uncertainties associated with the parameters in the risk assessment equations for the fish ingestion pathway are discussed in Appendix D. The uncertainties for each parameter were propagated using 500 iterations of LHS to obtain subjective probability distributions for the excess lifetime cancer risk associated with human ingestion of different species of fish from

LWBR which contain ^{137}Cs, Aroclor-1254, Aroclor-1260, dieldrin, and chlordane. From these distributions, the median risks, the lower 5% subjective confidence limits (LCLs), and the upper 95% subjective confidence limits (UCLs) were obtained.

The subjective confidence intervals for the excess lifetime cancer risk resulting from ^{137}Cs in fish when the dose conversion factors were allowed to vary are summarized in Table 5.10. The median risk for ^{137}Cs in catfish is 6.8×10^{-7}, and there is 90% confidence that the true but unknown risk lies between 1.6×10^{-7} and 3.2×10^{-6}. The upper confidence limits for ^{137}Cs in all species of fish are considered acceptable levels of risk.

Table 5.10. Subjective confidence limits for the excess lifetime cancer risk from ^{137}Cs in fish from Lower Watts Bar Reservoir when the dose conversion factor was allowed to vary

Species	Radionuclide	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
catfish	^{137}Cs	1.6×10^{-7}	6.8×10^{-7}	3.2×10^{-6}
largemouth bass	^{137}Cs	4.7×10^{-8}	2.0×10^{-7}	9.8×10^{-7}
bluegill	^{137}Cs	6.7×10^{-8}	3.2×10^{-7}	1.4×10^{-6}

The results of the uncertainty analysis for fish wherein the dose conversion factor for ^{137}Cs was fixed are approximately the same as those presented in Table 5.10. This result indicates that the uncertainty associated with the dose conversion factor for ^{137}Cs is not sufficient to affect the overall uncertainty in the excess lifetime cancer risk.

The subjective confidence intervals for the excess lifetime cancer risk resulting from Aroclor-1254, Aroclor-1260, dieldrin, and chlordane in LWBR fish when the slope factors for each contaminant except dieldrin were allowed to vary are summarized in Table 5.11. Catfish exhibited the highest total excess lifetime cancer risk for the selected organic contaminants. The major contributors to total risk are Aroclor-1260 and Aroclor-1254. The 5% LCLs for the total excess lifetime cancer risk from all contaminants are above 10^{-4} for catfish and striped/hybrid bass; therefore, it is highly unlikely that the actual risk level for these species would be below 10^{-4}. These results are based on the assumption that the slope factors for PCBs, which are based on evidence of carcinogenicity in animals, are applicable to humans.

The results of the uncertainty analysis for organic carcinogens when all slope factors were held constant are presented in Table 5.12. Only slight changes from the previous analysis were observed. The 95% UCL for chlordane went down slightly and the 95% UCL for the Aroclors went up slightly. As with the radionuclide analysis, these modest changes indicate that the uncertainties involved with the slope factors for Aroclor-1254, Aroclor-1260, and chlordane are not major contributors to the uncertainties in the total excess lifetime cancer risk.

Table 5.11. Subjective confidence limits for the excess lifetime cancer risk from selected carcinogens in fish from Lower Watts Bar Reservoir with slope factors allowed to vary (except for dieldrin)

Species	Contaminant	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
catfish	Aroclor-1254	8.8×10^{-5}	3.8×10^{-4}	1.5×10^{-3}
catfish	Aroclor-1260	1.9×10^{-4}	8.3×10^{-4}	2.8×10^{-3}
catfish	chlordane	6.2×10^{-6}	3.1×10^{-5}	1.4×10^{-4}
catfish	dieldrin	2.0×10^{-6}	3.0×10^{-5}	3.2×10^{-4}
	Total Risk[a]	4.0×10^{-4}	1.3×10^{-3}	4.2×10^{-3}
largemouth bass	Aroclor-1254	2.5×10^{-5}	1.1×10^{-4}	4.2×10^{-4}
largemouth bass	Aroclor-1260	3.7×10^{-5}	1.6×10^{-4}	5.3×10^{-4}
largemouth bass	chlordane	2.8×10^{-6}	1.4×10^{-5}	6.4×10^{-5}
	Total Risk[a]	8.9×10^{-5}	2.9×10^{-4}	9.1×10^{-4}
striped/hybrid bass	Aroclor-1254	7.1×10^{-5}	3.0×10^{-4}	1.2×10^{-3}
striped/hybrid bass	Aroclor-1260	1.3×10^{-4}	5.4×10^{-4}	1.8×10^{-3}
striped/hybrid bass	chlordane	5.0×10^{-6}	2.5×10^{-5}	1.2×10^{-4}
	Total Risk[a]	2.7×10^{-4}	9.0×10^{-4}	2.8×10^{-3}

[a]The values for the total risk are not directly additive because of the random process used for error propagation.

Table 5.12. Subjective confidence limits for the excess lifetime cancer risk from selected organic carcinogens in fish from Lower Watts Bar Reservoir with slope factors held constant

Species	Contaminant	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
catfish	Aroclor-1254	2.2×10^{-4}	7.2×10^{-4}	2.0×10^{-3}
catfish	Aroclor-1260	3.6×10^{-4}	1.2×10^{-3}	3.4×10^{-3}
catfish	chlordane	7.1×10^{-6}	2.3×10^{-5}	6.8×10^{-5}
catfish	dieldrin	2.0×10^{-6}	3.0×10^{-5}	3.2×10^{-4}
	Total Risk[a]	6.2×10^{-4}	2.0×10^{-3}	5.5×10^{-3}
largemouth bass	Aroclor-1254	6.1×10^{-5}	2.0×10^{-4}	5.6×10^{-4}
largemouth bass	Aroclor-1260	6.9×10^{-5}	2.2×10^{-4}	6.6×10^{-4}
largemouth bass	chlordane	3.2×10^{-6}	1.0×10^{-6}	3.0×10^{-5}
	Total Risk[a]	1.3×10^{-4}	4.3×10^{-4}	1.2×10^{-4}
striped/hybrid bass	Aroclor-1254	1.7×10^{-4}	5.9×10^{-4}	1.6×10^{-3}
striped/hybrid bass	Aroclor-1260	2.4×10^{-4}	7.7×10^{-4}	2.2×10^{-3}
striped/hybrid bass	chlordane	5.6×10^{-6}	1.9×10^{-5}	5.4×10^{-5}
	Total Risk[a]	4.1×10^{-4}	1.3×10^{-3}	3.9×10^{-3}

[a]The values for the total risk are not directly additive because of the random process used for error propagation.

5.5.1.2 Results of the sensitivity analysis for carcinogens in fish

A sensitivity analysis was performed for the total excess lifetime cancer risk for the fish ingestion exposure pathway. The method used for this sensitivity analysis was to square the Spearman rank correlation coefficients of each parameter and normalize them to 100% (Decisioneering, Inc. 1994). From this method, an approximate percentage of each parameter's contribution to the overall uncertainty is obtained. These percentages are referred to as the sensitivity index (SI) of the parameter. The higher the SI, the greater the contribution of the parameter to the overall uncertainty.

The top sensitivity indices for the total excess lifetime cancer risk resulting from eating ^{137}Cs contaminated catfish, obtained when the dose conversion factor for ^{137}Cs was allowed to vary, are presented in Table 5.13. The most sensitive parameters are ranked according to priority for further study.

Table 5.13. Results of sensitivity analysis of excess lifetime cancer risk from ingestion of ^{137}Cs in catfish from Lower Watts Bar Reservoir when the dose conversion factor was allowed to vary

Species	Sensitive Parameter	Sensitivity Index	Rank
catfish	I_f: meal/day	44.8%	1
	RCF: risk/Sv	36.0%	2

The most sensitive parameter for the total lifetime cancer risk from the ingestion of ^{137}Cs in LWBR catfish is the number of fish meals per day (I_f). The second most sensitive parameter is the risk conversion factor (RCF). When the dose conversion factor for ^{137}Cs was fixed, the two most important parameters contributing to the overall uncertainty in the total excess lifetime cancer risk were the number of fish meals per day and the amount of fish eaten per meal. These results indicate that the uncertainty associated with the excess lifetime cancer risk from the ingestion of fish from LWBR would be most effectively lowered by taking local surveys of consumption patterns rather than additional analyses of fish for radionuclides. Survey results would aid in the development of more definitive estimates of subjective confidence intervals for the number of fish meals per day, the exposure duration, and the amount of fish eaten per fish meal.

The uncertainty in the overall excess lifetime cancer risk from ^{137}Cs in fish is also sensitive to the risk conversion factor for ^{137}Cs. However, it is unlikely that this uncertainty can be lowered because of a lack of data quantifying the effects of low doses and low dose rates of radiation on humans.

The top three sensitivity indices for the total risk resulting from Aroclor-1254, Aroclor-1260, dieldrin, and chlordane in catfish, obtained when the slope factors were allowed to vary, are presented in Table 5.14. The sensitive parameters are ranked according to their contribution to the uncertainty in the risk analysis.

Table 5.14. Results of a sensitivity analysis for the total excess lifetime cancer risk from all selected organic carcinogens in catfish from Lower Watts Bar Reservoir when slope factors were allowed to vary

Species	Sensitive Parameter	Sensitivity Index	Rank
catfish	I_f: meal/day	72.2%	1
	I_m: kg/meal	14.3%	2
	Aroclor-1260 SF: (mg/kg-day)$^{-1}$	9.6%	3

The two most sensitive parameters for the total excess lifetime cancer risk from the select organic carcinogens in catfish are consistent with the other fish species, i.e., the number of fish meals per day (I_f) and the amount of fish eaten per meal (I_m). The same results were obtained when the slope factors were held constant, similar to the results obtained for the ^{137}Cs analysis. The overall estimate of uncertainty was moderately sensitive to the Aroclor-1260 slope factor.

5.5.1.3 Subjective confidence intervals for the hazard quotients of selected noncarcinogens in fish

The uncertainties associated with the parameters in the risk assessment equations for noncarcinogens in the fish ingestion pathway were calculated in a similar manner as for the carcinogens. The details are discussed in Appendix D. Contaminants included in the uncertainty analysis are mercury, arsenic, and chlordane.

The subjective confidence intervals, obtained for the HQs when the reference doses were allowed to vary, are summarized in Table 5.15. The median HQ for mercury in bluegill, for instance, is 0.12; and, there is 90% confidence that the true but unknown HQ lies between 0.02 and 0.90 (or there is 95% confidence that the true but unknown value does not exceed 0.90). Results for the other species and contaminants can be interpreted in the same way.

Based on the available data, the highest potential HI occurs from largemouth bass with a median of 1.12; however, the HIs for the different species are not directly comparable because each species was not analyzed for the same contaminants. The contaminants that were analyzed for each species were those where the greatest potential for bioaccumulation was thought to occur. Bluegill were not analyzed for organic contaminants (e.g., chlordane) and striped/hybrid bass were not analyzed for inorganic contaminants (Tables A.1 and A.2).

The wide ranges of the HQs for chlordane in all species are primarily due to the large uncertainty associated with its RfD. This is demonstrated by the results of the uncertainty analysis performed with the reference doses held constant (Table 5.16). Most 95% UCLs are lower than those obtained when the reference doses were allowed to vary. This effect is most noticeable for chlordane where the 95% UCL was reduced approximately 50% and can be explained by the large uncertainty associated with the chlordane reference dose.

Table 5.15. Subjective confidence limits for hazard quotients of selected noncarcinogens in fish from Lower Watts Bar Reservoir when reference doses were allowed to vary

Species	Contaminant	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
bluegill	mercury	0.02	0.12	0.90
bluegill	arsenic	0.03	0.11	0.42
	Total HI[a]	0.05	0.26	1.26
largemouth bass	mercury	0.06	0.46	3.45
largemouth bass	chlordane	0.02	0.30	3.45
largemouth bass	arsenic	0.05	0.23	0.90
	Total HI[a]	0.22	1.12	7.51
catfish	chlordane	0.05	0.69	7.44

[a]The values for the total risk are not directly additive because of the random process used for error propagation.

Table 5.16. Subjective confidence limits for the hazard quotients of selected noncarcinogens in fish from Lower Watts Bar Reservoir when reference doses were held constant

Species	Contaminant	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
bluegill	mercury	0.02	0.09	0.34
bluegill	arsenic	0.03	0.11	0.42
	Total HI[a]	0.05	0.21	0.74
largemouth bass	mercury	0.08	0.33	1.20
largemouth bass	chlordane	0.13	0.54	2.01
largemouth bass	arsenic	0.05	0.23	0.90
	Total HI[a]	0.26	1.13	3.97
catfish	chlordane	0.23	1.00	3.60

[a]The values for the total risk are not directly additive because of the random process used for error propagation.

5.5.1.4 Results of the sensitivity analysis for selected noncarcinogens in fish

A sensitivity analysis was performed for each total HI for each species of fish. The top sensitivity indices for the HI in largemouth bass, obtained when the reference doses were allowed to vary, are presented in Table 5.17. The sensitive parameters are ranked according to their contribution to the uncertainty in the total HI for largemouth bass.

Table 5.17. Results of sensitivity analysis for hazard indices for selected noncarcinogens in largemouth bass from Lower Watts Bar Reservoir when reference doses were allowed to vary

Species	Sensitive Parameter	Sensitivity Index	Rank
largemouth bass	I_f: meal/day	70.8%	1
	chlordane RfD: mg/kg-day	13.5%	2

The two most sensitive parameters for the HI from largemouth bass when the reference doses were allowed to vary are the number of fish meals per day (I_f) and the RfD for chlordane. When the reference doses were fixed, the uncertainty in the HI was dominated by the number of fish meals per day.

5.5.1.5 Conclusions for the uncertainty analysis of the fish consumption pathway

The following conclusions can be drawn from the quantitative uncertainty analysis on the risk of adverse health effects resulting from the ingestion of ^{137}Cs and other select contaminants found in fish from LWBR.

- Based on the results for ^{137}Cs, the primary anthropogenic radionuclide in fish from LWBR, further study is not warranted for radionuclides in fish because of the low magnitude ($< 10^{-6}$) of the upper bound of the uncertainty in the risk estimate for the maximally exposed individual.

- Because the total excess lifetime cancer risk estimates for organic contaminants in fish exceed a 1×10^{-4} level of concern at the 95% subjective confidence limits and, in most cases, at the 5% subjective confidence limits, health advisories are warranted for limiting the ingestion of the fish from LWBR. However, because the only evidence of cancer incidence from PCB exposure is in rats, caution should be used in assessing the importance of this contaminant to the overall excess lifetime cancer risk for humans.

- Results of the sensitivity analyses indicate that the overall estimate of uncertainty in the total risk estimates would be most effectively reduced by taking local surveys to obtain better estimates for the number of fish meals eaten per week and the amount of fish eaten per meal by an avid fisherman.

5.5.2 Uncertainty analysis for exposure to ^{137}Cs via the dredging scenario

Cs-137 was the only contaminant included in the uncertainty analysis for the dredging scenario. The decision to include only ^{137}Cs was based on 1) the adequacy of ^{137}Cs data compared to other contaminants, 2) the fact that the ORR is the major source of ^{137}Cs contamination in LWBR, and 3) results from the baseline risk assessment which indicate that ^{137}Cs in sediment used for agricultural purposes is a contaminant of potential concern (Table B.15).

5.5.2.1 Subjective confidence intervals from the uncertainty analysis for the dredging scenario

Uncertainties associated with the parameters in the risk assessment equations for the dredging scenario are discussed in Appendix D. The uncertainties for each parameter were propagated using 500 iterations of LHS to obtain subjective probability distributions for the excess lifetime cancer risk. From these distributions, the median risk, the 5% LCLs, and the 95% UCLs were obtained for each exposure pathway.

The subjective confidence intervals are summarized in Table 5.18. The median excess lifetime cancer risk for outdoor external exposure to ^{137}Cs in sediment used for agricultural purposes is 3.5×10^{-5}; and, there is 90% confidence that the true but unknown risk lies between 6.3×10^{-6} and 1.5×10^{-4} (or there is 95% confidence that the true but unknown value does not exceed 1.5×10^{-4}). This 95% UCL indicates an unacceptable risk from external exposure to the sediment.

Table 5.18. Subjective confidence limits for the excess lifetime cancer risk from ^{137}Cs in deep-water sediments of Lower Watts Bar Reservoir used for agricultural purposes

Exposure Pathway	5% Subjective Confidence Limit	Median	95% Subjective Confidence Limit
outdoor external	6.3×10^{-6}	3.5×10^{-5}	1.5×10^{-4}
leafy vegetables	1.6×10^{-8}	2.8×10^{-7}	3.9×10^{-6}
non-leafy vegetables	5.9×10^{-9}	3.6×10^{-7}	1.0×10^{-5}
Milk	8.8×10^{-8}	1.9×10^{-6}	3.2×10^{-5}
Meat	3.0×10^{-8}	1.2×10^{-6}	2.3×10^{-5}
Total Lifetime Risk[a]	**8.2×10^{-6}**	**4.7×10^{-5}**	**2.1×10^{-4}**

[a]The values for the total risk are not directly additive because of the random process used for error propagation.

5.5.2.2 Results of the sensitivity analysis for the dredging scenario

A sensitivity analysis was performed for the dredging scenario using the same methodology as described earlier for the fish ingestion scenario. Parameters with the highest sensitivity indices for the total cancer risk from ^{137}Cs in the dredging scenario are presented in Table 5.19.

Table 5.19. Results of sensitivity analysis for total excess lifetime cancer risk from ^{137}Cs exposure in the dredging scenario for Lower Watts Bar Reservoir

Sensitive Parameter	Sensitivity Index	Rank
RCF: risk/Sv	34.6%	1
l_{eff}: yr^{-1}	16.5%	2
D_l : unitless	13.4%	2
EF_{ext} : unitless	11.8%	2

The top four sensitive parameters for the uncertainty in the total excess lifetime cancer risk are the uncertainty in the risk conversion factor (RCF), the loss rate from radiological decay and harvesting/leaching (leff), the soil dilution factor (Df), and the external exposure frequency (EF). The total uncertainty is dominated by the risk conversion factor.

5.5.2.3 Conclusions for the uncertainty analysis for the dredging scenario

The sensitivity analysis conducted for an individual maximally exposed to ^{137}Cs in the dredging scenario indicates that the overall estimate of uncertainty in the excess lifetime cancer risk from all exposure pathways would be most effectively reduced by further evaluation of the risk conversion factor. The uncertainty analysis verifies the results of the baseline risk assessment, i.e., that ^{137}Cs in the deep sediment is a contaminant of potential concern. The greatest cancer risk from ^{137}Cs in sediment for the exposure pathways included in the dredging scenario comes from external exposure to the sediment.

5.6 SUMMARIES AND CONCLUSIONS

5.6.1 Chemicals of Potential Concern

Chemicals of potential concern are organic, inorganic, or radioactive contaminants that, as a result of a baseline risk assessment, have a calculated excess lifetime cancer risk $\geq 10^{-4}$ or an HI ≥ 1.0 (NCP, 40 CFR Part 300). Contaminants that have a cancer risk between 10^{-6} and 10^{-4} must be considered COPCs until further investigation shows they should be removed from the list of COPCs. Table 5.8 gives the exposure pathways and the carcinogens and noncarcinogens that have cancer risks of $\geq 10^{-6}$ or HQs of ≥ 1.0. Table 5.9 lists the contaminants and the exposure pathways for contaminants that have calculated cancer risk between 10^{-6} and 10^{-4}.

5.6.2 Exposure Assessment

The potential pathways for human exposure to contaminants in LWBR are shown in Fig. 5.1. Of the exposure pathways that currently exist, the most significant pathway by which

humans were exposed to COPCs is through the ingestion of fish. The contaminants of concern in fish are the carcinogens, Aroclor-1254 and Aroclor-1260, and the noncarcinogens, aldrin, chlordane, arsenic, and mercury. The latter four contaminants were identified as COPCs for the ingestion of fish by a child exposure pathway. The generic ingestion rate that was used for the 3-to-6-year-old child is ~0.5 lb of fish per week. This consumption rate would be highly unlikely unless it were for a fisherman's family, in which case the exposed population would be very small. A state of Tennessee advisory is in place for LWBR that limits the amount of various species of fish from LWBR recommended for consumption.

6. ECOLOGICAL RISK ASSESSMENT REPORT

The purpose of this ecological risk assessment is to summarize existing data relevant to ecological risks and to characterize the ecological risks, as far as is possible with those existing data. The assessment includes the following components. (1) Water, sediment, and fish tissue analyses are screened against ecotoxicological benchmarks, and the consequences of exceedences are briefly interpreted. (2) Ecological radiation exposures due to radionuclides in sediment are modelled. (3) The available fish and benthic invertebrate survey data and the water and sediment toxicity test data are compiled and interpreted. (4) The potential exposures of terrestrial biota to dredge spoil are modelled and screened against benchmarks. (5) The multiple lines of evidence are summarized and the risks characterized to the extent of stating whether, for each endpoint, the evidence is consistent with significant risks.

This ecological risk assessment is structured in terms of the standard paradigm for risk assessment (NRC 1983, Risk Assessment Forum 1992). It begins with hazard identification that defines the contaminant sources, the receiving environment, and the assessment endpoints. It follows with an analytical phase consisting of exposure assessment and effects assessment. The results of these analyses are then combined in a risk characterization, which include the structure and methods of this assessment (Risk Assessment Forum 1992), the DOE Oak Ridge Field Office's approach and strategy for performing ecological risk assessments (Suter et al. 1992a), and specific instructions from DOE, TDEC, and EPA Region IV.

The scope and quantitative character of this ecological risk assessment are less than was outlined in the approved plan, because of a decision made by the FFA parties to not perform the Phase 2 sampling and analysis in LWBR. As a result, no ecological data were collected in support of this assessment. The FFA parties agreed that an inconclusive ecological risk assessment was acceptable for LWBR for two reasons. First, the presumptive remedy does not include remediation of contaminated media. Second, data collected for the Clinch River/Poplar Creek RI and for the LWBR post-ROD monitoring should ultimately provide the basis for more definitive ecological risk assessments for LWBR. LWBR ecological risks will be reevaluated in the baseline ecological risk assessment for the Clinch River/Poplar Creek RI and in post-ROD reviews. The post-ROD monitoring in LWBR will include pertinent ecological parameters to provide a time series of data, which will be particularly useful for ecological risk assessments.

6.1 ECOLOGICAL HAZARD IDENTIFICATION

Hazard identification consists of: (1) the identification of ecological endpoints, (2) description of the relevant features of the environment, (3) description of the sources of contamination, and (4) summarization of that information in terms of a conceptual model of the hazard posed by the contaminants to the endpoint biota.

6.1.1 Ecological Endpoints

Hazard identification must identify both the assessment endpoints, which are explicit statements of the characteristics of the environment that are to be protected, and the measurement endpoints, which are quantitative summaries of a measurement or series of measurements that are related to effects on an assessment endpoint.

6.1.1.1 Assessment endpoints

The following assessment endpoints for aquatic and terrestrial risks have been selected for this assessment:

- toxicity to benthic macroinvertebrate biota resulting in at least a 20% reduction in species richness or abundance;
- toxicity to fish resulting in at least a 20% reduction in species richness or abundance or at least a 20% increase in gross pathologies;
- toxicity to piscivorous wildlife resulting in at least a 20% reduction in abundance or production;
- toxicity to terrestrial plants resulting in at least a 20% reduction in production;
- toxicity to terrestrial wildlife resulting in at least a 20% reduction in abundance or production;
- toxicity to individuals of any threatened or endangered species resulting in reduced viability or fecundity.

The ecological assessment endpoints have been selected based on the instructions from EPA Region IV rather than EPA's framework for ecological risk assessment. In their comments on the Waste Area Grouping 5 ecological assessment plan, EPA Region IV presented the following absolute criterion and contingent criterion (Brown 1993).

1. Ecological assessment endpoints should be ecologically significant in terms of their role in energy flow and mineral or nutrient cycling or their role as keystone species.
2. Policy and societal values may be considered "if all other factors (ecological significance) are equal." This criterion is not used because the contingency is never satisfied (i.e., the ecological significance of any two taxa or communities is never equal), and because of EPA Region IV's repeated and strong opposition to the selection of endpoints on the basis of policy and societal values.

During a meeting in Chattanooga (March 4, 1993) to discuss the content of Brown's (1993) letter, EPA Region IV added a third criterion. They stated that organisms that were wide ranging relative to the scale of the OU should not be used as ecological assessment endpoints because their contamination or responses could not be clearly associated with the site. Hence the following criterion is added.

3. Ecological assessment endpoints should have a scale appropriate to the site in order to be clearly associated with the site.

In a subsequent meeting concerning the Lower EFPC OU (March 24–25, 1993), Lynn Wellman, EPA Region IV, stated that sensitivity to the contaminants should also be used as a criterion for selecting ecological assessment endpoints. Because this criterion has not been defined by EPA Region IV, we take it to be equivalent to the susceptibility criterion that has been proposed in EPA documents (Suter 1989, Risk Assessment Forum 1992).

4. Assessment endpoints should be susceptible to the contaminants at the site in that they are exposed and are sensitive to the toxic effects of the contaminants.

These criteria are applied as follows. Taxa or communities are candidates for designation as assessment endpoints if they are ecologically important, have an appropriate scale, and are believed to be susceptible to the contaminants on the site. Taxa or communities are included even if they do not have demonstrable ecological importance if they are believed to be particularly susceptible because (1) they are highly exposed to the contaminants, (2) they are known from the literature to be highly sensitive to the contaminants, or (3) they are believed to have been affected by contaminants at the site based on prior studies. The selected endpoints are explained below.

1. The benthic invertebrate community is considered to be an appropriate endpoint community because benthic organisms are highly susceptible and have a scale appropriate to the site. The high susceptibility is because of the association of these organisms with the sediment, which is the repository of most of the contaminants of potential concern. In addition, the insects and crustaceans that dominate this community are sensitive to a variety of contaminants. Because these organisms are sedentary, their scale is highly appropriate to the scale of the site. The chosen characteristics of the endpoint are species richness and abundance of the community.

2. The fish community is considered to be an appropriate endpoint community because it is ecologically important, susceptible, and has a scale appropriate to the site. The ecological significance is based on the observation that in temperate reservoirs, the fish community exerts a controlling influence on the structure of the planktonic and benthic communities and, therefore, may be considered keystone species. In addition, most of the fish in LWBR move within an area smaller than the OU, so the scale is appropriate. The chosen characteristics of the endpoint are species richness and abundance of the community.

3. Piscivorous wildlife are considered to be an appropriate endpoint trophic group because they are highly susceptible and have a scale appropriate to the site. Two of the contaminants of potential concern, PCBs and mercury, biomagnify in aquatic food webs leading to high levels of exposure in piscivorous wildlife. Mink have been shown to be highly sensitive to the toxic effects of PCBs and mercury in toxicity tests. Reproduction of piscivorous birds has been shown to be highly sensitive to PCBs. Prior screening assessments have shown that there is a potential for toxic exposures of wildlife to a variety of contaminants in LWBR (Suter 1991, Cook et al. 1992). In addition, piscivorous wildlife forage within an area smaller than the LWBR (at least during the breeding season) so the scale is appropriate. The chosen characteristics of the endpoint are abundance or production of the local populations of the representative species (i.e., mink and great blue heron).

4. The terrestrial plant community is considered to be an appropriate endpoint community for dredging scenarios because it is ecologically important, susceptible, and has a scale appropriate to the site. Because the plant community is responsible for terrestrial primary production, it is ecologically significant. This community is susceptible because it would be directly exposed to the contaminants in the dredge spoil. Finally, the scale is appropriate because plants are immobile and because a distinct plant community would develop on dredge spoil deposited on land. The chosen characteristic of this endpoint is plant production.

5. Lynn Wellman (1993), EPA Region IV, recommended that the ecological risk assessment for the dredge spoil scenario should be based on the human health assessment garden model with a herbivorous wildlife species substituted for the human. The chosen characteristics of the endpoint are abundance or production of the local populations of the representative species (i.e., eastern cottontail rabbit).

6. Threatened and endangered species are in general too rare to be ecologically significant and may not be particularly susceptible or clearly associated with the site. However, EPA Region IV indicated during the March 4, 1993, meeting in Chattanooga that these species may be used as assessment endpoints because they are ARARS. Because of the particular societal value placed on these organisms, as reflected in the Endangered Species Act, the characteristics of this endpoint are any toxic effect that would reduce survivorship or fecundity of individuals.

The number of potential ecological assessment endpoints is extremely large, so it is not possible to completely explain why other endpoints are excluded. However, because the exclusion of plankton, amphibians, and reptiles was questioned by the regulators in their reviews of the CRRI Phase 2 Sampling and Analysis Plan, their exclusion is explained here.

Amphibians and reptiles are excluded on the basis of EPA Region IV's criteria for ecological assessment endpoints. Neither amphibians nor reptiles meet the ecological significance criterion in LWBR. That is, they are not important to energy flow or nutrient cycling in large temperate reservoirs, they are not known to be keystone species, and they have no other known properties that might be considered ecologically important in this system. Amphibians and reptiles do not meet the criterion of particular susceptibility because the available information (primarily from toxicity tests with larval anurans) suggests that they are, on average, less sensitive than fish and aquatic arthropods. In addition, they do not have particularly high exposures because they have trophic positions equivalent to fish, and their rates of exposure to aqueous contaminants are lower because they respire water quite slowly relative to fish. Finally, because this assessment is being performed with existing ecological data and there are no existing data for reptiles or amphibians in LWBR, it is not practical to consider using characteristics of those taxa as ecological assessment endpoints. Therefore, reptiles and amphibians are assumed to be adequately protected by assessments based on fish and invertebrates.

Planktonic species are not included as ecological assessment endpoints, even though they are ecologically important, because their planktonic habit means that they cannot be clearly associated with particular sources or reaches. In addition, plankton are not particularly susceptible. Algae have been found through long experience in ecological toxicology and assessment to be less sensitive than animals to contaminants in general other than herbicides. In addition, planktonic organisms are in the water column so they are not exposed to the bulk of LWBR contaminants that are associated with the sediments. Finally, planktonic organisms are at or near the bottom of food webs so they are not particularly exposed to bioaccumulative chemicals such as PCBs and mercury, which are contaminants of potential concern in LWBR.

The exclusion of these taxa and communities as assessment endpoints does not mean that they will be unprotected. Rather, it is assumed that protection of the most important or susceptible taxa will result in protection of all taxa. In addition, toxicity data concerning some excluded taxa are included in the screening benchmarks, and data concerning toxic effects on nonendpoint taxa are used as appropriate to help estimate and interpret direct and indirect effects on endpoint species. Finally, the National Ambient Water Quality Criteria (NAWQC), which are

incorporated as screening benchmarks and as ARARS, include effects on planktonic species, amphibians, and other aquatic species to the extent that data are available.

One assessment endpoint that was proposed in the Phase 2 Sampling and Analysis Plan (DOE 1994) has not been included. In the interest of performing a more focused assessment given the limited data for LWBR, EPA Region IV recommended that the ecological assessment of the dredge spoil disposal scenario be limited to the plants and herbivores modelled in the human health risk assessment and that uniquely ecological routes of exposure and receptors such as members of the soil heterotrophic community be excluded (Wellman 1993). Therefore, the planned assessment endpoint for earthworms and other soil heterotrophs is not included.

The 20% value used in the definition of ecological assessment endpoints is based on an analysis of EPA and Tennessee regulatory practice (Suter et al. 1992b). The clearest ecological criteria for regulation in EPA are those developed for the regulation of aqueous effluent under the NPDES. NPDES permitting may be based on any of three types of evidence: water quality criteria, effluent toxicity tests, and biological surveys, and the use of each of these implies that a 20% reduction in ecological parameters is *de minimis*.

1. The Chronic NAWQC for Protection of Aquatic Life are based on thresholds for statistically significant effects on individual responses of fish and aquatic invertebrates. Those thresholds correspond to approximately 25% reductions in the parameters of fish chronic tests (Suter et al. 1987). Because of the compounding of individual responses across life stages, the chronic NAWQC frequently correspond to much more than 20% effects on a continuously exposed fish population (Barnthouse et al. 1990).

2. The subchronic tests used to regulate effluent based on their toxicity cannot reliably detect less than 20% reductions in the test endpoints (Anderson and Norberg-King 1991).

3. Twenty percent is approximately the limit of detection of field measurement techniques used in the regulation of aqueous contaminants based on bioassessment. For example, the community metrics for an exposed benthic macroinvertebrate community must be reduced by more than 20% relative to the best communities within the ecoregion to be considered even slightly impaired in EPA's rapid bioassessment procedure (Plafkin et al. 1989). Measures for other taxa that are more difficult to sample may be even less sensitive. For example, the number of fish species and individuals must be reduced by 33% to receive less than the top score in the EPA's rapid bioassessment procedure for fish (Plafkin et al. 1989). Finally, some systems such as large rivers are more difficult to sample so that biological surveys are less sensitive. For example, at least 20 years of data would be necessary to detect a 50% reduction in mean year-class strength of white perch in the Hudson River, even allowing a Type II error of 50% (Vaughan and Van Winkle 1982). The minimum detectable difference varies with species, habitat, and sampling method, but mobile species differences of less than 20% can seldom be reliably detected.

The 20% level would also be consistent with practice in assessments of terrestrial effects. The Lowest Observed Effects Concentration (LOEC) for dietary tests of avian reproduction (the most important chronic test endpoint for ecological assessment of terrestrial effects of pesticides and arguably the most applicable test for waste sites) corresponds to an approximately 20% effect on individual response parameters (EPA 1982).

Hence, an effects level for ecological assessment endpoints lower than 20% would be acceptable based on current EPA regulatory practice and could not reliably be confirmed by field studies. Therefore, it can be considered *de minimis*. These ecological assessment endpoints have been previously submitted to EPA Region IV and TDEC and, except for the changes noted above, have been approved for the CRRI (DOE 1994).

6.1.1.2 Measurement endpoints

Because no data were collected specifically to satisfy ecological data quality objectives, the measurement endpoints are those that can be derived from existing data concerning the LWBR collected for other purposes and from data in the literature. These include historic data and data collected by the preliminary phase of the CRRI (Cook et al. 1992). Three basic types of effects data are potentially available to serve as measurement endpoints: results of biological surveys, toxicity tests performed on media from LWBR, and toxicity test endpoints for chemicals found in LWBR. Measurement endpoints are presented below for each assessment endpoint.

Benthic Invertebrates

Biological survey data. Species richness and abundance of benthic invertebrates reported from the TVA reservoir surveys. These measurement endpoints are assumed to be direct estimates of the assessment endpoint for this community.

Media Toxicity Data: Standard test endpoints for Microtox® and Rototox® acute toxicity tests performed with elutriate for TVA reservoir surveys. These tests do not correspond to the assessment endpoints for this community. Because they are not known to be correlated with chronic toxicity to benthic invertebrates but are almost certainly less sensitive (Sect. 6.3.2), responses in these tests suggest toxicity to the assessment endpoint, but lack of toxicity has no implications.

Single Chemical Toxicity Data. Chronic toxicity thresholds for freshwater invertebrates expressed as chronic 20th percentile effective concentrations (EC20s) or chronic values (CVs). These test endpoints correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of LWBR species, and exceedence of the CVs and EC20s are assumed to correspond to 20% reductions in population abundance or greater, with some uncertainty.

Fish

Biological Survey Data. The species richness and abundance from TVA reservoir surveys. These measurement endpoints are assumed to be direct estimates of the assessment endpoint for this community.

Media Toxicity Data. None.

Single Chemical Toxicity Data. Chronic toxicity thresholds for freshwater fish expressed as chronic EC20s or CVs. These test endpoints correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of LWBR species, and exceedence of the CVs and EC20s are assumed to correspond to 20% reductions in abundance or greater, with some uncertainty.

Piscivorous Wildlife

Biological Survey Data. None.

Media Toxicity Data. None.

Single Chemical Toxicity Data. Chronic toxicity thresholds for contaminants of concern in birds and mammals with greater weight given to data from long-term feeding studies with wildlife species. Preference was given to tests that included reproductive endpoints. These test endpoints are assumed to correspond to the assessment endpoint after allometric scaling for the endpoint species.

Terrestrial Plants

Biological Survey Data. None.

Media Toxicity Data. None.

Single Chemical Toxicity Data. EC20s for growth or production of vascular plants or equivalent chronic toxicity thresholds for contaminants of concern in soil. These test endpoints are assumed to correspond to the assessment endpoint for this community. That is, the sensitivity distribution of the test species is assumed to approximate the distribution of LWBR species, and exceedence of the test endpoints is assumed to correspond to 20% reductions in population abundance or productivity with some uncertainty.

Terrestrial Wildlife

Biological Survey Data. None.

Media Toxicity Data. None.

Single Chemical Toxicity Data. Chronic toxicity thresholds for contaminants of concern in birds and mammals with greater weight given to data from long-term feeding studies with wildlife species. Preference was given to tests that included reproductive endpoints. These test endpoints are assumed to correspond to the assessment endpoint after allometric scaling for the endpoint species, the eastern cottontail.

Threatened and Endangered Species

Biological Survey Data. None.

Media Toxicity Data. None.

Single Chemical Toxicity Data. The same chronic toxicity thresholds for contaminants of concern in invertebrates, fish, birds and mammals used as measurement endpoints for other species are used with the threatened and endangered species but are interpreted in the risk characterization so as to provide the higher level of protection specified in the assessment endpoint.

6.1.2 Environmental Description

This assessment considers the LWBR environment. A general description of this system is provided in Chap. 2. It is a large, highly productive reservoir on the main stem of a major river supporting a biotic community typical of such reservoirs. Because there is no floodplain, there are currently no known contaminated terrestrial systems. A contaminated terrestrial ecosystem would be created under the baseline dredging case, but the simple assessment performed for that hypothetical case does not include site-specific ecological characteristics. The federal- and state-listed threatened or endangered species and other state-listed rare or declining species are listed in Tables 2.7 and 2.8.

6.1.3 Sources

The proximate sources considered in this assessment are the contaminated media in the LWBR: water, sediment, and fish tissues. Although various ultimate sources contribute contaminants to the Watts Bar Reservoir watershed, this assessment is concerned with ultimate sources of contaminants associated with the DOE's operations on the ORR including waste disposal, spills, and use of chemicals such as pesticides in the environment and other upstream sources. The intermediate sources are the streams that drain the ORR, all of which are tributaries of the Clinch River. Although the DOE's operations did not contribute all of the contaminants in the LWBR, the assessment must characterize the risks to the biota of the LWBR to which the DOE has contributed, and not simply the risks that would have occurred if the DOE emissions had been released to a pristine environment.

Contaminants of potential concern (COPCs) for ecological risks have been identified in previous screening assessments (Suter 1991, Cook et al. 1992); Blaylock et al. (1993) have identified contaminants in the CRRI and historical data sets whose analytical quantitation limits are greater than one or more protective criteria. These are listed in Table 6.1. Given the large number of COPCs for all media, the revisions that have occurred in the ecological screening benchmarks, and the new contaminant concentration data that have been added, all contaminants are rescreened against benchmarks for this assessment.

6.1.4 Conceptual Models

The general nature and extent of contamination in LWBR are discussed in Chap. 3 and summarized in Sect. 3.3. The ecological conceptual models presented here further define the ecological receptors and their modes of exposure.

A conceptual model of exposure of the aquatic food web to contaminants for the current baseline cases is depicted in Fig. 6.1. All aquatic biota are directly exposed to contaminants in water. They are also externally exposed to radiation from radionuclides in water and sediments. Contaminated allochthonous material is added to the detritus in the system in the form of organic matter exported from the ORR in streams. Detritus and sediments are consumed by benthic invertebrates and detritivorous fish, and the benthic invertebrates are exposed directly to the sediments. The food web then transfers contaminants taken up by these routes to top predators and humans.

Exposure of the terrestrial food web in the hypothetical future dredging scenario is depicted in Fig. 6.2. It is assumed that ecological succession is allowed to occur on the spoil fill. Plants take up contaminants directly from the spoil. Herbivorous wildlife consume the plants, and incidentally ingest some spoil with their food or while grooming.

Table 6.1. Contaminants of potential concern for LWBR identified in prior screening assessments (Suter 1991, Cook et al. 1992, Blaylock et al. 1993)

Water	Sediments	Fish flesh
Aluminum	Lead	Antimony
Beryllium	Mercury	Arsenic
Cadmium	Zinc	Beryllium
Chromium	Acenaphthene	Cadmium
Cobalt	Anthracene	Copper
Copper	Aroclor 1016	Mercury
Cyanide	Aroclor 1221	Nickel
Iron	Aroclor 1232	Selenium
Lead	Aroclor 1242	Silver
Manganese	Aroclor 1248	Selenium
Mercury	Aroclor 1254	Zinc
Silver	Aroclor 1260	Aroclor 1016
Vanadium	Benzo(a)anthracene	Aroclor 1221
Zinc	Benzo(a)pyrene	Aroclor 1223
Anthracene	Benzoic acid	Aroclor 1232
Aroclor 1221	Benzylalcohol	Aroclor 1242
Aroclor 1242	Bis(2ethylhexyl)phthalate	Aroclor 1248
Aroclor 1248	Chlordane	Aroclor 1254
Aroclor 1254	Chrysene	Aroclor 1260
Aroclor 1260	Diethylphthalate	Total PCBs
Benzo(a)anthracene	4,4 DDD	2-Chlorophenol
Benzo(a)pyrene	4,4 DDE	DDE
Bis(2ethylhexyl)phthalate	4,4 DDT	DDD
Butylbenzylphthalate	Dibenz(a,h)anthracene	2,4-Dichlorophenol
Chlordane	Dieldrin	2,4-Dimethylphenol
DDD	2,4-Dinitrophenol	2-Methylphenol
DDE	Endrin	2-Nitrophenol
DDT	Fluoranthene	4-Methylphenol
Dieldrin	Fluorene	Naphthalene
Diethylphthalate	Gamma BHC	Pentachlorophenol
Dimethylphthalate	Heptachlor	
Endosulfan	2-Methylnaphthalene	
Endrin	2-Methylphenol	
Heptachlor	Naphthalene	
Hexachlorobutadiene	2-Nitrophenol	
Hexachlorocyclopentadiene	2-Nitrosodiphenylamine	
Penatchlorophenol	Phenanthrene	
Toxaphene		

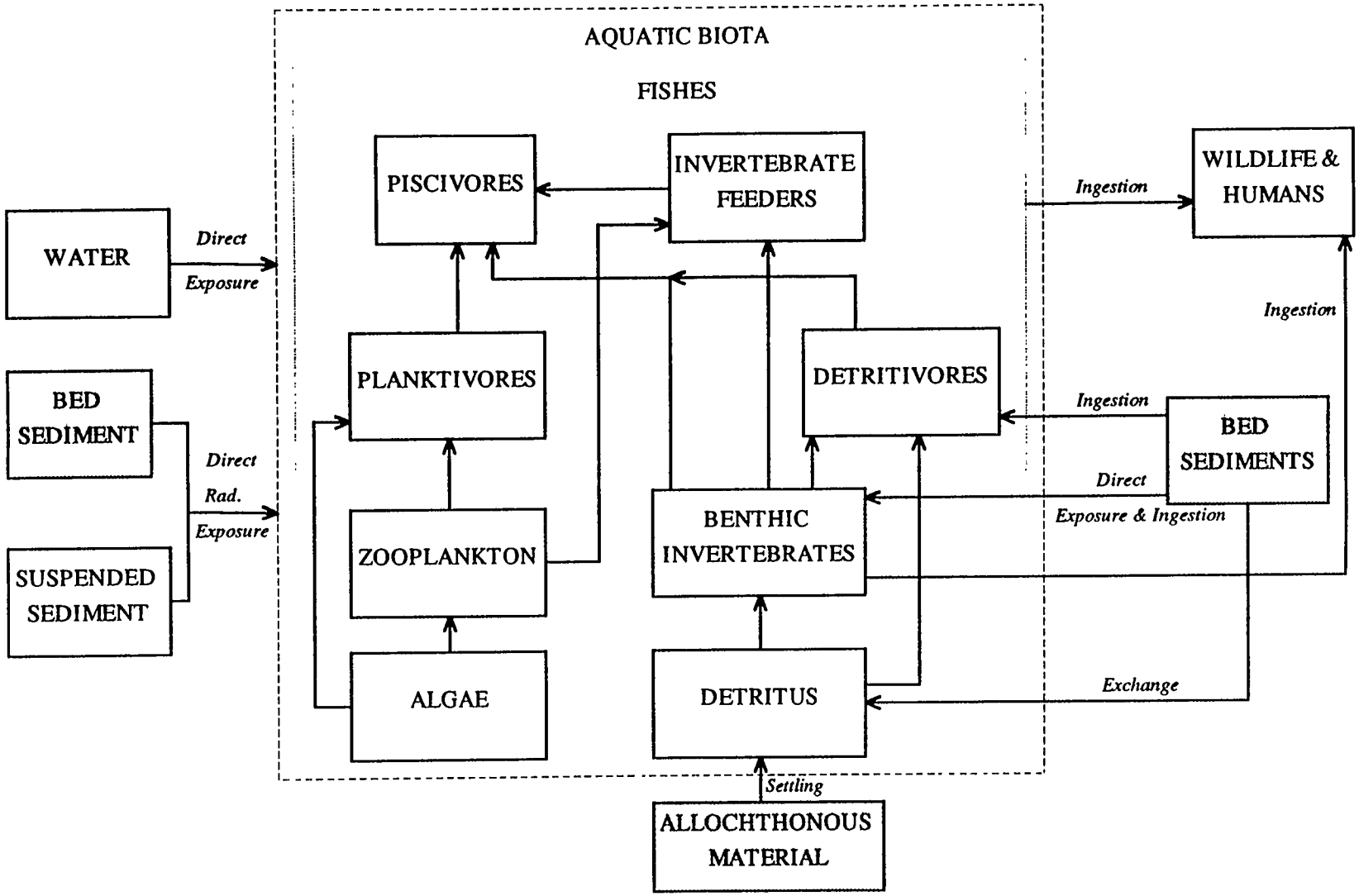


Fig. 6.1. Conceptual model of contaminant sources, transport, and exposure in the aquatic food web. Shaded boxes represent sources that may be remediated. Arrows represent routes of transport and exposure, described by italic labels. Unlabeled arrows represent food web transfers by ingestion.

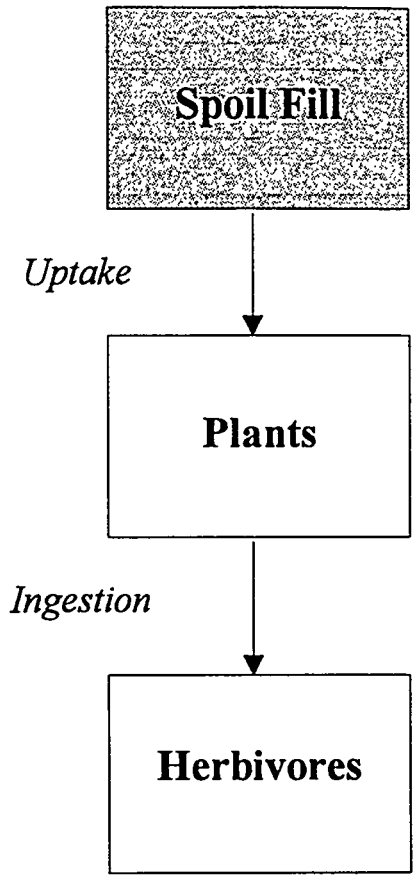


Fig. 6.2 Conceptual model of contaminant sources, transport, and exposure in the terrestrial food web for the baseline dredging case. Shaded box represents source that may be remediated. Arrows represent routes of transport and exposure described by italic labels.

6.2 EXPOSURE ASSESSMENT

Exposure assessment estimates the concentrations of contaminants to which organisms are exposed, taking into consideration (as far as possible) the spatial and temporal distributions of those concentrations. Risk from undetected chemicals are not assessed, as directed by DOE, EPA Region IV, and TDEC at a data quality objectives workshop (September 1993). The contaminant concentrations used in the exposure assessment are provided in Appendix E. The remainder of this chapter explains how those data are related to ecological exposure to specific media.

6.2.1 Aqueous Exposures

Contaminants in water may come from upstream aqueous sources including the ORR, exchange of materials between the surface water and contaminated sediments, or exchange of

contaminants between the biota and the water column. The aquatic biota are assumed to be exposed to the dissolved fraction of the chemicals in water, because that is the bioavailable form (Prothro 1993). However, because dissolved water analyses are available for only a few samples, total concentrations are also included when dissolved concentrations are not available for a particular contaminant.

Because water in LWBR is likely to be more variable in time than space, because of the rapid replacement of water in the reservoir, the mean water concentration is an appropriate estimate of the chronic exposure experienced by fishes. The upper 95% confidence bound on the mean is an appropriately conservative estimate of this exposure for use in the contaminant screening.

6.2.2 Sediment Exposures

Sediments in LWBR may be contaminated due to deposition of contaminated particulate material from upstream sources, deposition of particles that became contaminated within the LWBR, or exchange of contaminants between the surface water and sediment. Biota may be exposed to sediment pore water, contact with the particulate phase, or ingestion of sediment.

Two different measures of exposure are used for sediments. First, whole sediment concentrations are used for comparison to screening benchmarks and toxicity test results expressed as sediment concentrations. Second, for neutral organic chemicals, concentrations of chemicals in pore water are estimated using the equilibrium partitioning approach and compared to screening benchmarks and toxicity test results expressed as aqueous concentrations. Pore-water concentrations of neutral (non-ionic) organic chemicals can be calculated by assuming equilibrium partitioning between the pore water and the organic matter fraction of the sediment (EPA 1989b). The partitioning coefficient (K_p) is the product of the organic matter/water partitioning coefficient(K_{oc}) and the fractional organic matter content of the sediment (f_{oc}). Because sediment-dwelling organisms are approximately as sensitive to chemicals in water as the population of species used to derive the NAWQC (EPA 1989b), the same screening criteria can be used as for water but corrected for partitioning. The formula is:

$$C_s = K_p C_w \text{ , or}$$
$$C_s = K_{oc} f_{oc} C_w \text{ ,}$$

where C_s and C_w are equivalent concentrations in sediment and water, respectively. K_{oc} is seldom available, but it is quite accurately approximated by the octanol/water partitioning coefficient (K_{ow}) (DiToro 1985). This approach is being used by EPA to derive sediment quality criteria for neutral organic chemicals (EPA 1993b). This approach does not produce accurate predictions of pore water concentrations for ionic organic chemicals. It is likely that this method underestimates exposure for highly hydrophobic chemicals such as PCBs that have significant dietary uptake.

The maximum observed surface sediment concentration is used to screen for effects on benthic communities. Sediment concentrations are more variable in space than time, but benthic invertebrates are relatively immobile. The mean concentration would not be relevant because none of the organisms integrate their exposures over a reach or any large area of it. Therefore, the most appropriate estimate of the reasonable maximum exposure is the maximum concentration in the surface sediments inhabited by benthic invertebrates.

6.2.3 Wildlife Exposures To Contaminated Fish

Piscivorous wildlife are exposed through consumption of contaminated fish. The two endpoint species, mink and great blue heron, are assumed to consume a diet of 100% fish (or other aquatic organisms with contamination levels equal to fish) from LWBR for purposes of contaminant screening. The fish flesh concentrations reported are for fillets rather than the whole fish consumed by a heron or mink. Because many contaminants concentrate in fat or in organs other than muscle, these concentrations are likely to be underestimates of the exposure experienced by piscivorous wildlife. An exception is mercury, which is primarily associated with muscle.

The chronic exposure of piscivorous wildlife is a result of consuming a large number of fish. Therefore, it is reasonable to use the mean concentration as an estimate of chronic exposure. The upper 95% confidence bound on the mean provides a reasonably conservative estimate of the mean for the contaminant screening because of sampling and analysis errors and the tendency of wildlife to forage in a limited area that may have higher than average contamination.

6.2.4 Soil Exposures

Any terrestrial ecosystem that developed on dredge spoils would be exposed to contaminants in the sediments from which the spoil was derived. This exposure is assessed using the same assumptions and models as the human health risk assessment (Chap. 5). The spoil is assumed to be derived from mixed deep sediments. For estimation of ecological risks, no irrigation is assumed, and the mixed vegetables concentrations are used for the herbivore (cottontail rabbit) diet.

6.3 EFFECTS ASSESSMENT

Ecotoxicological effects can be estimated from three independent types of data. They are as follows: (1) toxicity data for individual contaminants, (2) toxicity tests of contaminated media, and (3) biological surveys of the contaminated site and reference sites.

6.3.1 Conventional Toxicity Data

Because this is a preliminary ecological risk assessment, the conventional toxicity data used are limited to screening benchmarks. These are concentrations in media that are intended to be indicative of marginally acceptable effects. Hence, concentrations that are below the benchmarks indicate a nonhazardous condition with respect to the individual contaminants.

6.3.1.1 Aqueous toxicity

The screening benchmarks for aquatic biota are taken from Suter et al. (1992b). Because there are no standard screening benchmarks, sets of alternative benchmarks were calculated. For this assessment, the preferred benchmark is the chronic NAWQC (NAWQC Actual). In some cases, the chronic NAWQC is based on bioconcentration and effects on piscivores. To consider effects on aquatic life in those cases the concentration calculated using the method for deriving chronic NAWQC for protection of aquatic life (NAWQC Equivalent) are presented as well as the NAWQC Actual. In addition, the lowest chronic values (LCV) for fish (LCV Fish) and daphnids (LCV Daphnids), or other species (e.g., LCV Algae) are used. Because of the way the NAWQC

are calculated, they are often higher than chronic values for sensitive species. It is important in a risk assessment to identify situations in which sensitive species are at risk. EPA uses LCVs as surrogates for NAWQC for those criterion chemicals for which there is not enough test data to derive actual criteria (EPA 1986). Note that in some cases, the lowest chronic value used here is lower than the lowest chronic value reported by the EPA. Many of the EPA's lowest chronic values are based on reviews that are more than a decade old. If no measured chronic values are available for a chemical, but an LC50 for a fish or daphnid is available, a chronic value was estimated using acute/chronic regressions (Suter et al. 1992b).

In addition to these benchmarks that are based on regulatory practice, four benchmarks that are based on risk levels are used. The lowest toxic values (LTVs) for fish and daphnids (LTV-FISH and LTV-DAPH) are the lowest concentrations that cause a 20% reduction in the productivity of fish or daphnids across all life stages in a chronic toxicity test. The sensitive species EC20 is the lowest of the two LTV benchmarks adjusted for the distribution of species sensitivity so as to estimate a concentration protective of 95% of species. This procedure is the same as that used to calculate chronic NAWQC except that LTVs are substituted for chronic values. Finally, the concentration estimated to cause a 20% reduction in recruit abundance in a largemouth bass population is used. The derivation of all of these benchmarks is described in detail in Suter et al. (1992b).

Some NAWQC are functions of water hardness, pH, or temperature. We corrected those criteria for LWBR conditions. For purposes of screening, we chose conditions that would constitute reasonable maximum toxicity, which we defined as conditions that would persist for 4 days (EPA's averaging period for chronic toxicity in NPDES). These conditions are pH of 8.5, temperature of 25°C, and hardness of 100 mg/L. However, none of the chemicals that require correction were detected in the LWBR water samples.

6.3.1.2 Sediment toxicity

Three types of benchmarks were used with sediment concentrations. First, National Oceanic Atmospheric Administration's (NOAA) Effects Range-Low (ER-L) and Effects Range-Median (ER-M) concentrations were used for the 43 chemicals for which they are available (Long and Morgan 1991). These are concentrations that appeared to cause toxic effects in 10% and 50% of sediments, respectively. Both field and laboratory studies are used, but most of the studies are of estuarine sediments.

The second type of sediment screening benchmark is based on equilibrium partitioning of neutral organic chemicals between sediments and pore water. Because it is assumed that the exposure is because of exposure to the pore water and that benthic species are equal in sensitivity to water column species (EPA 1993b), the aqueous screening benchmarks discussed above are used. EPA has proposed five sediment quality criteria for neutral organic chemicals, but none of them have been detected in LWBR sediments.

Finally, standards and criteria promulgated by states or by Canadian provinces are used (Hull and Suter 1993). They are not enforced in Tennessee but provide useful benchmarks for contaminants that do not have NOAA values or are not neutral organic chemicals. The only benchmarks of this type that are applicable to the contaminants detected in LWBR are the Ontario Ministry of Environment (MOE) provincial sediment quality guidelines (Persaud et al. 1990). They are derived using the screening level concentration (SLC) approach. The SLC is derived from synoptic data on sediment chemical concentrations and benthic invertebrate distributions.

First, the species screening level concentration (SSLC) is calculated by plotting the frequency distribution of the contaminant concentrations over all sites (at least 10) where the species is present. The 90th percentile of this distribution is taken as the SSLC for that species. Next, a large number of SSLCs are plotted as a frequency distribution to determine the contaminant concentration above which 95% of the SSLCs occur. This final concentration is the SLC (Neff et al. 1988). Ontario has calculated the 5th percentile using the SLC method for use as a "Low" guideline value, and a 95th percentile value, which would be expected to be detrimental to most benthic species, as a "Severe" guideline value.

The screening benchmarks for ecological effects of contaminated sediments are discussed at greater length in Hull and Suter (1993).

6.3.1.3 Wildlife toxicity

The screening benchmarks for wildlife are No Observed Adverse Effects Levels (NOAELs) derived for mink and great blue heron according to the methods outlined by Opresko et al. (1994) and EPA (1993). Toxicological studies of the effects of contaminants detected in fish from LWBR were obtained from the open literature. Because of differences in physiology between birds and mammals, separate studies were located (if available) for each wildlife class. Only studies of the effects of long-term, chronic oral exposures, whether in food, water, or by oral intubation, were used. To make the NOAELs relevant to possible population effects, preference was given to studies that evaluated effects on reproductive parameters. In the absence of a reproduction endpoint, studies that considered effects on growth, survival, and longevity were used. If the only available data consist of a NOAEL for a subchronic exposure (<3–6 months for mammals, <10 weeks for birds), then the equivalent NOAEL for a chronic exposure was estimated as follows:

$$chronicNOAEL = \frac{subchronicNOAEL}{10}$$

In cases where a NOAEL for a specific chemical was not available, but a Lowest Observed Adverse Effects Level (LOAEL) had been determined experimentally, the NOAEL was estimated as follows:

$$NOAEL = \frac{LOAEL}{10}$$

EPA (1993) suggests the use of uncertainty factors of 1 to 10 for subchronic to chronic NOAEL and LOAEL to NOAEL estimation. To be conservative, and because there are no data to suggest the use of lower uncertainty factors, a factor of 10 was chosen for both estimates.

Studies have shown that numerous physiological functions such as metabolic rates, as well as responses to toxic chemicals, are a function of body size. Smaller animals have higher metabolic rates and are usually more resistant to toxic chemicals because of more rapid rates of detoxification. It has been shown that the best measure of differences in body size are those based on body surface area which, for lack of direct measurements, can be expressed in terms of body weight (bw) raised to the 2/3 power ($bw^{2/3}$) (EPA 1982). If the dose (d) itself has been calculated

in terms of unit body weight (i.e., milligrams per kilograms); then the dose per unit surface area (D) equates to

$$D=\frac{d\times bw}{bw^{2/3}}=d\times bw^{1/3}$$

The assumption is that the dose per body surface area for species "a" and "b" would be equivalent:

$$d_a\times bw_a^{1/3}=d_b\times bw_b^{1/3}$$

Therefore, knowing the body weights of two species and the dose (d_b) producing a given effect in species "b", the dose (d_a) producing the same effect in species "a" can be determined:

$$d_a=d_b\times\frac{bw_b^{1/3}}{bw_a^{1/3}}=d_b\times(bw_b/bw_a)^{1/3}$$

Therefore, using this approach, if a NOAEL was available for the test species ($NOAEL_t$), the equivalent NOAEL for a wildlife species ($NOAEL_w$) was calculated by using the adjustment factor for differences in body size:

$$NOAEL_w=NOAEL_t\times(bw_t/bw_w)^{1/3}$$

This methodology is similar to what EPA uses in their carcinogenicity assessments and Reportable Quantity documents for adjusting from animal data to an equivalent human dose (EPA 1985a and 1988a).

6.3.1.4 Soil toxicity to plants

Because of the diversity of soils, plant species, chemical forms, and test procedures, it is not possible to estimate concentrations that would constitute a threshold for toxic effects on the plant communities at particular sites from published toxicity data. This situation is analogous to the problem of deriving benchmarks for sediments. Our method for deriving soil benchmarks is based on the NOAA's method for deriving the ER-L (Long and Morgan 1991), which has been recommended as a sediment screening benchmark by EPA Region IV. The ER-L is the tenth percentile of the distribution of various toxic effects thresholds for various organisms in various sediments.

This approach can be justified by assuming that the toxicity of a chemical in soil is a random variate, that the toxicity of contaminated soil at a particular site is drawn from the same distribution, and that 90% certainty of protecting plants growing at the site soil is desired. Any bias in the data set would mitigate against that assumption. In this implementation of the approach, the bias that is most likely to be significant is the use of soluble salts of metals in the toxicity tests, which are likely to be more toxic than the mixture of forms encountered in field soils. That bias would result in conservative benchmark values. Other possible sources of bias

include the use of predominately domestic plant species that may not be representative of plant species in general, use of predominately agricultural soils that may not be representative of soils in general, and the laboratory test conditions that may not be representative of field conditions. The direction and magnitude of these potential biases is unknown.

The benchmarks were, like the ER-Ls, based on the LOECs. The LOEC is defined here as the lowest applied concentration of the chemical of interest, which gave a greater than 20% reduction in any measure of plant survival, growth, or production. In some cases the LOEC for the test was the lowest concentration tested (LCT) or the only concentration tested, as of when the EC50 was reported.

The phytotoxicity benchmarks were derived by rank ordering the LOEC values and then picking a number that approximated the tenth percentile. As with the ER-Ls, statistical fitting was not used because there was seldom sufficient data and because these benchmarks are to be used as screening values and do not require the consistency and precision of regulatory criteria. If there were ten or fewer values for a chemical, the lowest LOEC was used. If there were more than ten values, the tenth percentile LOEC value was used. If the tenth percentile fell between LOEC values, a value was chosen by interpolation. In all cases, benchmark values were rounded to one significant figure to indicate their precision.

Another possible source of benchmark values is values recommended in published reviews of the phytotoxicity literature. When primary literature is unavailable for a particular contaminant, concentrations identified in reviews as thresholds for phytotoxicity are used as benchmarks. In addition, when fewer than three LOEC values were found for a chemical in soil or solution, and a toxicity threshold from a review is lower than the lowest LOEC, the toxicity threshold is used as the benchmark for that chemical.

Data used to derive the benchmarks and details of the derivation method are presented in Suter et al. (1994).

6.3.2 Media Toxicity Data

No toxicity testing has been performed for the LWBR specifically for the CRRI. However, some water and sediment toxicity testing has been performed by the TVA as part of their reservoir vital signs monitoring. This program collects various data for TVA reservoirs from three locations: an area near the dam termed the forebay, an area near the middle of the reservoir termed the transition, and an area in the upper reaches of the reservoir termed the inflow. The forebay (at TRM 531.0) and transition sites (at TRM 560.8) have been sampled for sediment and water toxicity testing. These tests are described by Moses and Wade (1991, 1992) and Moses et al. (1993).

Grab samples are collected once each year in the summer (June-August). Water was collected with Kemmerer or Isco samplers at an unspecified depth. Sediment was collected with Eckmann dredges, and the top three centimeters of fines were retained for extraction of pore water by centrifugation. Sediment pore water and surface water were screened through 100 μm nylon mesh and aerated to achieve 40–100% saturation.

A pair of commercial acute toxicity tests were run on the full strength surface water and pore water samples. The Rototox® test determines the lethality of the waters to *Brachionus calyciflorus* newly emerged from cysts in 24-hour exposures. The Microtox® test determines light emissions

by the bioluminescent bacterium *Photobacterium phosphorem* in 5- to 15-minute exposures. If statistically significant toxicity was found in the Microtox® test of a sample, a second test employing a dilution series was performed on that sample.

Results of these tests from 1990 to 1992 have been irregular. Pore water at TRM 560.8 was toxic to Microtox® in 1990 and 1992, but not 1991. A test of a dilution series of pore water performed in 1990 established that the threshold for significantly reduced Microtox® activity was 18% pore water. TRM 531 pore water was lethal to rotifers in 1992, but not 1990 or 1991. Surface water and sediment pore water were nontoxic in these tests in all other combinations of year and location.

6.3.3 Biological Survey Data

No biological surveys have been performed for the LWBR by the CRRI. However, some fish and benthic invertebrate surveys have been performed by the TVA as part of their reservoir vital signs monitoring. This program collects various data for TVA reservoirs from three locations: an area near the dam termed the forebay, an area near the middle of the reservoir termed the transition, and an area in the upper reaches of the reservoir termed the inflow. In Watts Bar Reservoir, the forebay is at TRM 531.0, approximately a mile above the dam. The transition site is at TRM 560.8, less than 7 miles below the confluence of the Clinch River. The Tennessee River inflow site is at TRM 600.0, approximately a mile below the confluence of the Little Tennessee River. The Clinch River inflow site for benthic invertebrates is at CRM 19.0, between the inflow of WOC and Poplar Creek. The Clinch River inflow site for fish is at CRM 22.0, above the inflow of WOC but below Melton Hill dam. The inflow sites are outside the OU, but they serve as reference sites.

6.3.3.1 Benthic invertebrates

The benthic invertebrates samples are collected once each year on transects across the width of the reservoir. Ten samples were taken at equal intervals along the transects using Ponar or Peterson dredges. Samples were washed through 800 mm mesh sieves on site, large organisms were counted and returned to the reservoir, other large material was discarded, and remaining retained material was preserved for sorting and identification. Methods and results are described in greater detail by Jenkinson (1991) and Masters (1992 and 1993).

Results of the benthic invertebrates surveys of Watts Bar Reservoir are summarized in Table 6.2. The number of species found in all locations increased between 1990 and 1992. The numbers of individuals also tend to rise over the 3-year interval but less consistently. The number of individuals in samples from the two LWBR sites are roughly similar to the means across reservoirs. The two inflow sites are much more variable relative to mean numbers of individuals, but they are lower in 1990 and 1992 and higher in 1991. In 1992, the TVA rated the benthic community of the transition site as good, the forebay and Clinch River inflow site as fair, and the Tennessee River inflow site as poor (Masters 1993).

6.3.3.2 Fish

Fish communities were surveyed by electroshocking and gill netting from September to December as part of the TVA's Reservoir Vital Signs Monitoring Program. Electrofishing samples came from 10 near-shore runs of 10 minutes duration. Netting samples came from ten 100 ft gill nets that were set at each site for one overnight period. Lengths and weights of black

bass (*Micropterus* spp.) and channel catfish (*Ictalurus punctatus*) were recorded and other species were enumerated. Visible parasites, diseases, or deformities were noted. Fifteen largemouth bass (*M. salmoides*) from each site were retained for health assessment. Methods and results are described in greater detail by Hickman et al. (1991), Scott (1992), and Brown et al. (1993).

Results of the fish community surveys are summarized in Table 6.3. The primary data are the numbers of species and the number of individuals collected. The sites were rated by TVA on categorical scales (better, average, and worse in 1990 and good, fair, and poor in 1991 and 1992) based on (1) scoring of multiple characteristics of the fish communities or of the individual fish (34 in 1990, 11 in 1991 and 1992); (2) summation of the scores; and (3) rating into thirds of sites of the same type (inflow, transition, and forebay). Run-of-the-river (i.e., mainstream reservoirs like Watts Bar) and tributary reservoirs were scored and rated separately. These scores and ratings have no objective interpretation but are variously described by the TVA as representing health, condition, or quality of the fish community or of the aquatic environment.

Table 6.2 Results of TVA monitoring of benthic invertebrates in Watts Bar Reservoir compared to ten other mainstream reservoirs (minima-means-maxima) in that year. Jenkinson (1991) and Masters (1992 and 1993).

Location	Number of species	Density of organisms (No./m^2)
	1990	
Watts Bar Forebay	10 (6-11-17)	554 (325-558-1042)
Watts Bar Transition	12 (3-10-21)	316 (75-504-963)
Tennessee River Inflow	4 (4-11-22)	45 (45-280-679)
Clinch River Inflow	12 (4-11-22)	60 (45-280-679)
	1991	
Watts Bar Forebay	11 (3-10-15)	455 (260-596-1033)
Watts Bar Transition	12 (5-10-16)	750 (35-470-1283)
Tennessee River Inflow	13 (6-14-25)	513 (27-435-1030)
Clinch River Inflow	21 (6-14-25)	545 (27-435-1030)
	1992	
Watts Bar Forebay	19 (7-15-24)	693 (121-550-900)
Watts Bar Transition	16 (7-18-29)	868 (113-701-1246)
Tennessee River Inflow	23 (12-25-33)	547 (265-1056-2760)
Clinch River Inflow	20 (12-25-33)	335 (265-1056-2760)

Fish health indices based on occurrence of parasites and external or internal abnormalities of organs, hematocrit, leucocrit, and plasma protein in largemouth bass are available for 1990 and 1991 (Table 6.3). The anomalous conditions were weighted according to their severity, and

Table 6.3. Results of TVA monitoring of fish communities in Watts Bar Reservoir compared to the (maximum-mean-minimum) values for the ten mainstream reservoirs. Electrofishing units are hours; gill net units are net-nights. Hickman et al. (1991), Scott (1992), and Brown et al. (1993).

Location	Number of species	Catch per unit effort (shocking)	Catch per unit effort (netting)	Fish health index	TVA rating
		1990			
Watts Bar Reservoir Forebay	33(23-29-34)	757(437-1914-5553)	21(16-34-77)	48(20-50-67)	Worse
Watts Bar Reservoir Transition	35(25-28-38)	2178(347-1971-8516)	37(9-21-38)	27(27-55-87)	Better
Tennessee River Inflow	28(25-32-37)	1784(57-2483-12815)	22(1-17-51)	53(23-40-66)	Average
Clinch River Inflow	37(25-32-37)	57(57-2483-12815)	15(1-17-51)	---	Worse
		1991			
Watts Bar Reservoir Forebay	31(29-31-34)	757(165-879-3591)	33(12-26-49)	73(50-68-83)	Fair
Watts Bar Reservoir Transition	32(24-30-35)	1056(206-662-1257)	26(3-12-26)	65(44-53-88)	Good
Tennessee River Inflow	32(26-32-35)	361(148-1572-8924)	22(1-24-81)	52(52-73-91)	Fair
Clinch River Inflow	32(26-32-35)	148(148-1572-8924)	15(1-24-81)	---	Fair
		1992			
Watts Bar Reservoir Forebay	27(22-26-30)	35(35-756-2992)	14(9-23-39)	---	Fair
Watts Bar Reservoir Transition	24(22-28-35)	48(48-378-1664)	17(5-19-27)	---	Fair
Tennessee River Inflow	37(11-27-37)	393(18-551-2656)	8(0.4-14-42)	---	Fair
Clinch River Inflow	23(11-27-37)	304(18-551-2656)	8(0.4-14-42)	---	Fair

the weighted frequencies were summed so that lower indices indicate lower frequencies of anomalies or less severe anomalies. A breakdown of results is available for 1990. It indicates that parasites are relatively common in the Tennessee River inflow site (13/15 fish) and the transition site (8/15 fish). Liver and kidney abnormalities were found in 6/15 transition site fish and liver abnormalities in 5/15 forebay site fish.

In addition, TVA used trawling and hydroacoustics to estimate fish abundance in the summer of 1990 (Wilson 1991). They found that threadfin shad comprised 99.9% of the trawl catch in the lower part of the reservoir and 99.7% in the intermediate area. Blue catfish, channel catfish, and yellow bass followed in abundance. This species composition is typical of TVA mainstream reservoirs. Hydroacoustic data indicated that the density was 6.9 and 2.7 fish per m^3 in the lower and middle areas, and the biomass was 0.029 and 0.014 kg/m^3 in the lower and middle areas. Lower areas of Watts Bar Reservoir had the highest density of mainstream reservoirs and the third highest biomass in 1990. Middle areas of Watts Bar Reservoir had average densities but the third highest biomass of mainstream reservoirs.

6.4 RISK CHARACTERIZATION

Risk characterization is the phase of risk assessment in which the information concerning exposure (Sect. 6.2) and the information concerning the potential effects of exposure (Sect. 6.3) are integrated into estimated risks (the likelihood of effects given the exposure). Risk characterization in ecological risk assessment is performed by a weight-of-evidence analysis. The principal lines of evidence concerning effects are biological survey data that indicate the actual state of the receiving environment, media toxicity data, which indicate whether the contaminated media are toxic under controlled conditions, and single chemical toxicity data, which indicate the toxic effects of the concentrations measured in site media. Procedurally, the risk characterization is performed for each assessment endpoint by (1) screening all measured contaminants against toxicological benchmarks and background concentrations, (2) estimating the effects of the contaminants retained by the screening analysis, (3) estimating the toxicity of the ambient media based on the media toxicity test results, (4) estimating the effects of exposure on the endpoint biota based on the results of the biological survey data, (5) logically integrating the lines of evidence to characterize risks to the endpoint, and (6) listing and discussing the uncertainties in the assessment.

In this section, risks are characterized for the current baseline case (Sect. 6.4.1) and a hypothetical future case (Sect. 6.4.2). In addition, uncertainties are discussed (Sect. 6.4.3), ecological risks are compared to human health risks (Sect. 6.4.4), and ecological risks are summarized (Sect. 6.4.5).

6.4.1 Risk Integration for Current Conditions

The weight-of-evidence approach is followed in this section to the extent that existing data allow. For current conditions, the results of steps 1 and 2, screening and estimation of effects of screened contaminants, is presented in Sect. 6.4.1.1. Risk characterization based on media toxicity testing is presented in Sect. 6.4.1.2. Risk characterization based on biological survey results are presented in Sect. 6.4.1.3.

6.4.1.1 Single chemical toxicity

In the contaminant screening portion of this risk characterization, the contaminant concentrations measured in water, sediment, and fish flesh (Sect. 6.2) are compared to ecotoxicological benchmarks (Sect. 6.3) to derive hazard quotients by the formula:

Hazard quotient = media concentration/toxicological benchmark.

Hazard quotients (HQs) greater than one suggest that the contaminant is hazardous to the endpoint biota, and HQs less than one suggest that the contaminant is nonhazardous. HQs much greater than or much less than one allow confident conclusions concerning hazards and those approximately equal to one allow only highly uncertain conclusions. The uncertainty associated with any particular HQ value depends on the quantity and quality of the data and the relationship of the toxicological benchmark to the assessment endpoint. For each contaminant that exceeds benchmarks, the bases for believing that the exceedences imply a significant hazard are briefly explained.

In addition to screening the contaminant concentrations in the OU (LWBR) against ecotoxicological benchmarks, both positive reference reaches (2 and 4) and a negative reference reach (10) are screened against the same benchmarks. Although none of these reaches can be considered to constitute an uncontaminated background, they do serve to indicate which contaminants are ubiquitous in the region (negative reference reaches) and which upstream reaches may act as sources of contaminants to LWBR (positive reference reaches).

Water/Fish

Fish are exposed to contaminants in water through direct respiratory uptake. There are very few analytical data for water from LWBR (Table E.1); only iron and sodium were detected in that OU. Of these, only iron exceeds any ecological benchmark values. The benchmark exceeded is the LTV for daphnids (LTV-DAPH). However, studies other than the *Daphnia magna* chronic test used to derive that benchmark suggest a threshold for effects of approximately 1 mg/L, which is the NAWQC for protection of aquatic life. That value is not exceeded in LWBR. To supplement LWBR data, lower Clinch River data were used here also (Table E.2).

Sediment/Benthic Invertebrates

Benthic invertebrates are exposed to contaminants in sediments. Several contaminants have been measured in LWBR sediments at concentrations that exceed screening benchmarks (Tables E.3–5). They are the metals arsenic, barium, cadmium, iron, lead, manganese, mercury, nickel, silver, and zinc and the organic compounds chlordane, dibenz(a,h)anthracene, phenanthrene, and pyrene. Not all of these were detected at elevated concentrations in all of the three sediment data sets because of differences in techniques and in areas sampled. Therefore, elevated concentrations in any one data set were taken to be evidence of a hazard to benthic communities. The concentrations mentioned are concentrations in surface sediment.

- Arsenic concentrations were higher in sediments from LWBR than from other reaches, but they are not extremely high. Mean and maximum concentrations exceed the Ontario MOE's Lowest Effects Level (LOL), but not the Severe Effects Level (SEL) or the NOAA benchmarks. Arsenic concentrations in sediment lower than those found in LWBR have been found to be toxic to bivalves and several species of crustaceans (Long and Morgan 1991) and

to reduce species richness in the field (Persaud et al. 1990). Arsenic has been associated with coal ash disposal on the ORR.

- Barium concentrations are higher in surface sediments from LWBR than from other reaches. There are no ecotoxicological benchmarks for barium in sediments, but EPA Region V provides a screening benchmark for barium in sediment that is less than a tenth of the maximum concentration and approximately a quarter of the mean concentration in LWBR. That benchmark is based on regional background concentrations.

- Cadmium mean concentrations in LWBR surface sediments exceeded the ER-L and the LEL MOE. They are also higher than in the Clinch River arm. Cadmium concentrations in sediment lower than those found in LWBR have been found to be toxic to bivalves and several species of crustaceans and to reduce species richness in the field (Long and Morgan 1991).

- Iron concentrations were higher in sediments from LWBR than from Norris Reservoir but were not reported for other reaches. Mean concentrations exceed Ontario's LEL MOE and SEL MOE at both sites, which suggests that species richness should be severely reduced. There are no NOAA benchmarks for iron.

- Lead maximum concentrations exceeded the ER-L and ER-M in LWBR surface sediments as well as sediments from the Clinch River arm and Norris Reservoir. Mean lead concentrations in LWBR and Norris exceed the ER-L. Hence, the 120 mg/kg maximum found in LWBR has been found to be toxic or to reduce species richness in more than half of the laboratory and field studies of lead in sediments summarized by Long and Morgan (1991).

- Manganese concentrations were lower in sediments from LWBR than from Norris Reservoir and the upper Clinch River arm and were a little higher than in the Tennessee River arm. They exceed the LEL MOE and SEL MOE at all sites, which suggests that species richness should be severely reduced. There are no NOAA benchmarks for manganese.

- Mercury maximum concentrations in LWBR surface sediments exceeded the ER-L and ER-M, and mean concentrations exceeded the ER-L. They were lower than in the lower Clinch River arm but higher than upper Clinch River arm and Norris Reservoir. That pattern is consistent with a source on the Poplar Creek watershed. The 5.6 mg/kg found in LWBR has been found to be toxic or to reduce species richness in more than half of the laboratory and field studies of mercury in sediments summarized by Long and Morgan (1991). Dycus and Meinert (1991) associated acute toxicity of sediment pore water at the LWBR transition site with elevated mercury levels.

- Nickel maximum concentrations in LWBR surface sediments exceeded the ER-L and ER-M, and mean concentrations exceeded the ER-L. They are higher than in other reaches. Hence, the 56 mg/kg found in LWBR has been found to be toxic or to reduce species richness in more than half of the laboratory and field studies of nickel in sediments summarized by Long and Morgan (1991).

- Silver maximum concentrations in LWBR equal the ER-L. They also equal concentrations in the upper Clinch River arm but silver was not detected in surface sediments from any

other reach. There is relatively little data to support this benchmark. The ER-L is based on sediment avoidance by *Macoma* sp. (a bivalve) (Long and Morgan 1991).

- Zinc maximum concentrations in LWBR surface sediments exceeded the ER-L and ER-M, and mean concentrations exceed the ER-L. They are higher than in other reaches. Hence, the 390 mg/kg found in LWBR has been found to be toxic or to reduce species richness in more than half of the laboratory and field studies of zinc in sediments summarized by Long and Morgan (1991). Moses et al. (1993) conjectured that the acute toxicity of sediment pore water observed in LWBR in 1992 was because of zinc.

- Chlordane mean concentrations in LWBR surface sediments exceeded the ER-L and ER-M. Chlordane was detected in only one LWBR sample and was not detected in other reaches. The 0.02 mg/kg concentration found in LWBR has been found to be toxic or to reduce species richness in more than half of the laboratory and field studies of chlordane in sediments summarized by Long and Morgan (1991). Chlordane has been found in WOC.

Fish/Piscivorous Wildlife

Mean or UCL concentrations of five contaminants exceed benchmarks for the representative piscivorous wildlife, mink and great blue heron, and in flesh of some species of fish (Table E.6). They are mercury, selenium, zinc, Aroclor-1260, and total PCBs.

- Mercury is exceptional in that concentrations are high in all species of fish from all reaches. Even Norris Reservoir, which is the furthest upstream reference site and is otherwise relatively uncontaminated, has high mercury concentrations in fish. Mercury concentrations in fish from LWBR are lower than those from the lower Clinch River and, depending on the species, are lower than or similar to those in Norris Reservoir. Mallards were fed on mercury concentrations slightly higher than UCL concentrations in LWBR catfish (500 μg/kg vs 392 μg/kg). They produced fewer eggs and young than controls, and the young displayed behavioral abnormalities (Heinz 1979). This LOAEL was divided by 10 to estimate a NOAEL for the benchmark. The ORR has been a source of mercury.

- Selenium mean concentrations in catfish from LWBR exceed the benchmark for mink but not herons. It was not detected in any other fish species in LWBR. Selenium distributions in fish across reaches are irregular. For example, although selenium concentrations in catfish were higher in LWBR than in other reaches, selenium was not detected in largemouth bass or bluegill from LWBR even though bass and bluegill concentrations were higher than catfish concentrations in all other reaches. These irregularities may be an artifact of small sample sizes or may be due to some unknown process. Selenium UCL concentrations in LWBR of 346 μg/kg are approximately a third of the lowest concentration reported to induce selenosis, including liver cirrhosis, lameness, loss of hair, emaciation, reduced conception, and increased fetal resorption (Harr 1978). Selenium has been associated with coal ash disposal on the ORR.

- Zinc UCL concentrations in catfish and bluegill from LWBR exceed the benchmark for herons but not mink. Zinc concentrations were higher there than in the Clinch River arm for those species. The zinc benchmark that was based on a dietary exposure that killed mallard ducks in less than 60 days (Gasaway and Buss 1972), was divided by 10 for extrapolation to a NOAEL and by 10 to extrapolate to a chronic exposure. There are no known significant sources of zinc on the ORR.

- PCB mean concentrations (as either Arochlor-1260 or total PCBs) in catfish and striped and hybrid bass from LWBR exceed the benchmark for mink. The observed concentrations were greater than those shown to cause fetal mortality in dietary exposures of mink (Auerlich and Ringer 1977). For all fish species except striped and hybrid bass, concentrations from LWBR were lower than those from either the Clinch or Tennessee River arms. The ORR has been a source of PCBs.

6.4.1.2 Media toxicity

The occurrence of toxicity in Microtox® tests in the transition site samples for two of the last three years and of rotifer acute lethality in forebay samples indicates that LWBR sediments are acutely toxic. The variance between years is likely to be due to variance in the location of the samples, because the composition of sediments tends to be stable from year to year. However, the absence of replication in the testing program makes it impossible to confirm that conjecture. The fact that one test showed a response in forebay sediments and the other showed a response in transition sediments suggests that different contaminants are responsible for the toxicity in the two sites. The negative results for the water samples indicates that those six samples, were not acutely toxic to the tested rotifer and bacteria species. Because no chronic toxicity tests have been performed, the chronic toxicity of the water and sediments is unknown.

6.4.1.3 Biological surveys

Benthic Invertebrates

The results of TVA's surveys of benthic invertebrates are inconclusive but are consistent with the occurrence of toxic effects. Comparisons of the numbers of species and individuals to other TVA reservoirs are presented in Table 6.2. In all three years, the transition and forebay sites, which are in the OU, tend to fall near the mean of such sites in the TVA system. However, given the extent of pollution in the TVA system, an average site is not indicative of an absence of toxic effects. Rather, only the best sites could be considered negative reference sites (i.e., sites that are undergraded. The number of species in Watts Bar forebay and transition sediments average 70% and 63% of the best forebay and transition sites, respectively. The density of individual organisms in Watts Bar forebay and transition sediments average 54% and 58% of the best forebay and transition sites, respectively. These decrements in benthic community parameters are not necessarily caused by toxic effects. Other site characteristics such as sediment texture and water quality also influence benthic communities. Actual toxic effects may be higher or lower and may even be zero. These results can best be interpreted as indicating that the benthic invertebrates in LWBR are not being affected catastrophically by contaminants in the sediment relative to sites of the same type in other TVA reservoirs, but some factor or combination of factors has made them significantly worse (i.e., more than 20% worse) than the best TVA sites.

Fish

The most obvious pattern in the fish community data for LWBR (Table 6.3, Watts Bar Reservoir forebay and transition sites) is the decline in species number and abundance from 1990 to 1992. The decline is more consistent in LWBR than in either arm (i.e., inflow sites). The cause of this decline is unknown, but there has been no known increase in contaminant releases by DOE that would account for the trend.

Comparison of the LWBR fish community to those of the other nine TVA mainstream reservoirs produces mixed results. The number of species is near the upper limit (1990) or mean (1991 and 1992) of TVA mainstream reservoirs, but the number of individuals ranges from near the top (netting results at the transition site in 1990) to the absolute bottom (shocking results at both sites in 1992). As with benthic invertebrates, the best indication of reference fish community properties is the data from the best sites of a particular type (intermediate or forebay) in mainstream reservoirs. The number of fish species in Watts Bar forebay and transition sites average 93% and 84% of the best forebay and transition sites, respectively. The catch per unit effort for shocking in Watts Bar forebay and transition sites average 12% and 38% of the best forebay and transition sites, respectively. The catch per unit effort for netting in Watts Bar forebay and transition sites average 43% and 70% of the best forebay and transition sites, respectively. Hence, the species richness of the LWBR fish community is not much lower than the best reservoir sites but the number of fish is substantially reduced. The decrements in fish abundance are not necessarily caused by toxic effects. The data set of water analyses is too meager to determine whether toxic concentrations are present. Although primary productivity is high, other site characteristics such as DO levels, reservoir operations, and fishing pressure also influence fish communities. Actual toxic effects on fish communities may be higher or lower.

The fish health index at the transition site was lowest (i.e., best) of any mainstream transition site in 1990 but was worse than average in 1991. At the forebay site it was a little better than average in 1990 but was worse than average in 1991. These results are inconclusive.

6.4.1.4 Weight of evidence for current conditions

The characterization of risks to each ecological endpoint is presented here in terms of the weight of evidence from the individual lines of evidence characterized above. For each endpoint, the weighing of evidence is presented in narrative and tabular form.

Benthic Invertebrates

This is the only endpoint for which three lines of evidence are available. All three lines of evidence are consistent with the conclusion that the benthic invertebrate community is degraded due to toxicity of the sediments (Table 6.4). However, the quality of the data for the purpose of assessing ecological risks is low in that the biological survey data and sediment analyses were from studies that did not allow association of community effects with sediment characteristics, and the toxicity tests were more appropriate to a study of sediment dredging than of *in situ* sediments. In addition, most of the potentially toxic contaminants are not clearly associated with DOE emissions. Therefore, although the data are consistent with toxic effects, it is not possible to say with confidence what effects are attributable to contaminants or how extensive they are.

Fishes

The evidence concerning effects on fish are too sparse and ambiguous to reach any conclusions (Table 6.5). The abundance of fish in LWBR has markedly declined in recent years. That pattern is not consistent with DOE emissions as a causal factor, because those emissions have declined. Natural processes in the LWBR ecosystem, other emissions in the LWBR ecosystem, contaminants applied to the LWBR ecosystem (e.g., TVA's herbicide applications for control of macrophytes), or other factors could be responsible. The high frequencies of parasites, liver abnormalities, and kidney abnormalities found in the TVA surveys suggest that effects are a result of some factor other than over harvesting. Concentrations in fillets are high for several

contaminants (Table E.6), but data necessary to interpret muscle concentrations in terms of effects on the fish themselves are not available.

Table 6.4. Summary of the risk characterization for benthic invertebrates

Evidence	Result[a]	Explanation
Biological surveys	+	Abundance and diversity are substantially lower than in the best reservoirs.
Toxicity tests	+	The amount of toxicity is found in acute tests of sediment pore waters.
Media analyses	+	Maximum As, Ba, Cd, Fe, PB, Mn, Hg, Ni, Ag, Zn, and Chlordane concentrations are high enough to cause toxicity or reduce invertebrate abundance or diversity in some sediments.
Weight-of-evidence	+	Abundance and diversity of benthic invertebrates is likely to have been reduced by >20% in the most contaminated sediments.

[a]+ indicates that the evidence is consistent with the occurrence of the endpoint effect.

Table 6.5. Summary of the risk characterization for fish

Evidence	Result[a]	Explanation
Biological surveys	±	Fish abundance in recent years has gone from average to low relative to other mainstem reservoirs.
Body burdens	±	Measured body burdens are not interpretable in terms of effects on the fish themselves, but they do indicate that the contaminants are bioavailable.
Toxicity tests	-	No toxicity was found, but the tests are of uncertain value.
Media analyses	-	No toxic concentrations of contaminants were found in water, but there is little data.
Weight-of-evidence	±	Insufficient information is available to interpret the role of toxicity in the condition of the fish community.

[a]+indicates that the evidence is consistent with the occurrence of the endpoint effect; - indicates that the evidence is inconsistent with the occurrence of the endpoint effect; ± indicates that the evidence is too ambiguous to interpret.

Piscivorous Wildlife

The available evidence suggests that wildlife consuming fish from LWBR have experienced reductions in survivorship and fecundity (Table 6.6). Although the evidence is meager by the standards of ecological risk assessment, it is analogous to the evidence used in human health risk assessments except that the major food species have not been sampled. Better evidence will

become available when analyses of whole fish, the studies of great blue heron rookeries, and the mink feeding study are completed for the Clinch River arm and Poplar Creek embayment (Cook et al. 1992).

Table 6.6. Summary of the risk characterization for piscivorous wildlife

Evidence	Result[a]	Explanation
Biological surveys	NA	No surveys of piscivorous wildlife have been performed.
Toxicity tests	NA	No toxicity tests using LWBR fish tissues have been performed.
Media analyses	+	Hg, Se, and PCBs in fish tissues were sufficient to cause reduced survival or reproduction of mink. Hg and Zn in fish tissues were sufficient to cause reduced survival or reproduction of herons.
Weight-of-evidence	+	The available evidence is consistent with significant effects on survival and reproduction.

a+indicates that the evidence is consistent with the occurrence of the endpoint effect. NA indicates that a type of evidence is unavailable.

Threatened and Endangered Species

There are no data that are specific to effects of contaminants on threatened and endangered species. Therefore, the risks to these species must be inferred from data concerning exposure and responses of more common species that have been sampled, surveyed, or tested. The following species are those that are judged most likely to have existed in the LWBR or in the Tennessee River prior to reservoir construction.

- Twelve of the T&E species listed for LWBR are riverine molluscs (mussels and snails). Although the decline or loss of those species is likely to be due more to habitat destruction associated with constructing Watts Bar Dam and impounding the river, it is not clear what effects, if any, contaminants have had on these species.

- The snail darter may have inhabited the Tennessee River but any Tennessee River population was probably extinguished by dam construction rather than contaminant exposure.

- The Cumberland slider, a turtle listed as being of special concern by the state, may exist in LWBR. Toxic effects on this species cannot be assessed with existing data. It had been assumed that the assessment of risks to fish would also serve to protect turtles, but the assessment of risks to fish is inconclusive at this time.

- Bald eagles are an endangered species that are increasing nationwide, are known to irregularly winter on LWBR, and may become established as a breeding species on LWBR. Like the great blue heron, bald eagles feed predominantly on fish. Therefore, if they attempted to breed on LWBR, the estimated exposures are potentially toxic because of multiple contaminants.

- The osprey, which is a state-listed endangered species, breeds on LWBR, but the breeding success is unknown. Like the great blue heron, ospreys feed on fish and are estimated to be exposed to potentially toxic concentrations of multiple contaminants.

- Both turkey vultures and black vultures, which are listed by the state as species of special concern, are common in the vicinity of LWBR. Both species feed on dead fish washed onto shore and thereby may be exposed to contaminants in LWBR. However, their diets are sufficiently diverse and their ranges are sufficiently large to make toxic effects unlikely.

- The river otter, a state-listed threatened species, probably inhabited the Tennessee River but was extirpated in eastern Tennessee before the creation of the ORR. The estimated toxic effects on mink would be even more likely to occur in otters, because they are more exclusively piscivorous than mink and eat larger fish. Therefore, reestablishment of this species might be inhibited by toxic effects of contaminants in fish.

- The Indiana bat and gray bat, both of which are federally and state-listed endangered species, occur in the region and forage over water. Therefore, they may be exposed to contaminants in emergent stages of aquatic insects from LWBR. Contaminant levels in those insects are unknown, but both PCBs and mercury are likely to accumulate in benthic invertebrates.

6.4.2 Risk Integration for Future Conditions

It is assumed that the level of contamination in the water, sediment, and fish of LWBR will not increase in the future under baseline conditions. The only future scenario that requires assessment is the uncontrolled dredging of sediments from LWBR resulting in deposition of spoils on land. As instructed by EPA Region IV, the same assumptions and models as the human health risk assessment's dredge spoil scenario (Chap. 5) are used for this component of the ecological risk assessment.

If the sediments are deposited on land, they would be expected to be toxic to plants as a result of four metals that occur at concentrations above the phytotoxicity screening benchmarks. The phytotoxicity data are summarized in Suter et al. (1994). The results of the screening against phytotoxicity benchmarks are presented in Appendix E.

- Cadmium, at the mean sediment concentration, has been shown to reduce the growth of plants.

- Mercury UCL concentrations exceeded concentrations that are phytotoxic in soil by more than a factor of eight.

- Nickel has been shown to reduce plant growth at soil concentrations a little below the UCL concentration.

- Zinc is an essential plant nutrient, but the mean concentration is more than 10 times the concentration in soil that has been shown to cause chlorosis and reduce plant growth.

Hence, any plant community developing on dredge spoil from LWBR would be expected to be less productive than a plant community on native soil.

If the contaminant concentrations in vegetation grown on dredge spoil are assumed to occur in vegetation consumed by eastern cottontail rabbits, cadmium, copper, and mercury are estimated to occur at concentrations that exceed the screening benchmarks for that species. The wildlife toxicity data are summarized in Opresko et al. (1994). The results of the screening of estimated plant concentrations against wildlife benchmarks are presented in Appendix E.

- Cadmium, at the upper 95% confidence limit on the mean vegetable concentration, are, after allometric adjustment, approximately equal to concentrations causing reduced growth.

- Copper mean estimated vegetable concentrations are sufficient to cause reductions in longevity in lifetime feeding studies.

- Mercury mean concentrations estimated for vegetables exceeded no adverse effect concentrations by a factor of seven. These dietary concentrations would be expected to cause reproductive failure and damage various organs.

Hence, rabbits and other herbivorous wildlife inhabiting an area covered by dredge spoil from LWBR would be expected to experience toxic effects including reduced survival, growth, and reproduction relative to populations of the same species on native soil.

6.4.3 Uncertainties

6.4.3.1 Uncertainties in the hazard identification

The EPA's framework for ecological risk assessment specifies that pathways and receptors that are excluded from the conceptual model and therefore are not assessed should be presented in the uncertainties section (Risk Assessment Forum 1992). Four routes of exposure that were included in the Phase 2 Sampling and Analysis Plan (DOE 1994) were excluded from the ecological risk assessment upon direction from EPA Region IV in the interest of producing a more focused assessment (Wellman 1993). These routes of exposure are

- exposure of aquatic biota to suspended contaminated sediments, including exchange with the aqueous phase during dredging;

- deposition of buried contaminated sediments on the surface of the sediment, thereby introducing it into the biologically active layer;

- exposure of soil heterotrophic organisms to contaminated sediments deposited on land as dredge spoil; and

- exposure of wildlife to contaminants in sediments deposited on land as dredge spoil, a route that would include direct exposure and ingestion of sediment as well as food-web transfer.

6.4.3.2 Uncertainties in the contaminant analysis data

The contaminant data for LWBR were not collected to support ecological risk assessment. The historical data were collected for various contaminant monitoring programs and the CRRI Phase 1 data were collected to provide a basis for planning the Phase 2 Sampling and Analysis, which was not implemented for LWBR. As a result, the following uncertainties occur.

- The limits of detection for many national priority list chemicals are higher than the threshold concentrations for ecological effects (Suter 1991, Cook et al. 1992, Blaylock et al. 1993). As a result, it is possible that undetected contaminants are causing toxic effects or are contributing to the total toxicity of water, sediment, and fish in LWBR. However, they are excluded from the assessment. That decision is responsible for the much smaller list of COPC in this assessment relative to those in prior screening assessments of this OU (Table 6.1 vs Table 6.4–6.6 of this report). The possibility of not detecting important contaminants is demonstrated by the fact that PCBs were not detected in water or sediment even though these media must be the sources of PCBs in fish.

- The samples were not collected randomly or on any other probability design. Therefore, they cannot be said with any definable confidence to characterize the system. Some samples, such as the CRRI Phase 1 sediment samples were designed to look for contaminated deposition areas, but because sediment deposition patterns were not well known, the extent to which those few samples represent the deposition areas in LWBR is unknown. Other sampling programs had designs that bear no obvious relationship to the distribution of contaminants in LWBR (e.g., the public use area samples).

- The fish tissue analyses are all of filets. Piscivorous wildlife consume whole fish that include tissues that are more highly contaminated than muscle and ingested sediment. The error due to this factor is approximately a factor of 1.5 to 2 for metals and 2 to 3 for persistent organic chemicals (Suter 1991).

- The fish tissue analyses are all of species that are potentially important human food species. However, piscivorous wildlife are likely to consume the most abundant species, gizzard and threadfin shad, which were not sampled. These species have food habits that are different from the sampled species so that their concentrations of bioaccumulative contaminants (i.e., mercury and PCBs) may be different from the species included in the available data.

- There are very few analyses of water samples that were filtered before extraction. Only the dissolved phase is bioavailable to any significant extent. For metals, this can result in significant over estimation of exposure, the magnitude of which depends largely on the amount of suspended sediment.

- The form and speciation of contaminants in sediment is highly variable due to differences in sediment composition and oxidation state. Therefore, the bioavailabily of the concentrations reported for whole sediment is highly uncertain. Much of this uncertainty could be reduced by analysis of pore water.

6.4.3.3 Uncertainties in the chemical toxicity data

Unlike human health risk assessments, ecological risk assessments have no standard toxicological endpoints, much less standard values of those endpoints like the human RfDs and slope factors.

- Toxicity tests of individual chemicals in all media tend to use highly bioavailable and toxic forms (e.g., soluble metal salts). Other forms may occur in the field that are essentially unavailable or nontoxic.

- Sediment toxicity data are highly uncertain because of the variation in physical and chemical properties of sediments and the diversity of modes of exposure among benthic biota. For example, thresholds for toxic effects of PCBs derived by different methods range more than five orders of magnitude (Long and Morgan 1991). Most of the sediment benchmarks are from NOAA reviews that are derived predominantly from estuarine studies and that include field studies that may be biased by cooccurring contaminants (Long and Morgan 1991).

- Water toxicity data are much more standardized than sediment toxicity data, and water is less variable than sediments. However, alternative methods for calculating thresholds for aquatic toxic effects from laboratory tests produce benchmarks that vary over a range greater than two orders of magnitude (Suter et al. 1992b).

- Wildlife toxicity data are the least variable in that the physical and chemical characteristics of mammalian and avian foods are much less variable than water or sediments and the characteristics of birds and mammals are relatively constant compared to the taxa of aquatic and sediment communities. A factor of 10 has been suggested as accounting for uncertainty concerning sensitivity of wildlife species to toxicants (EPA 1993).

- The form and speciation of contaminants in soil is highly variable because of differences in soil composition and oxidation state. In addition, the contaminants added to soil in toxicity tests may be much more available than the forms of the chemicals occurring in soils or sediments used as soils. Therefore, the toxicity of chemicals in the dredge spoil could differ from the benchmark value by at least a factor of 10.

6.4.3.4 Uncertainties in the media toxicity data

There are no media toxicity data for media other than sediments. Therefore, uncertainty about the toxicity of water and fish is high because toxicity must be estimated from the toxicity of the constituent chemicals, which may have different forms or speciation in LWBR and are tested in media that are chemically and physically different from those in the LWBR. The following list of uncertainties refers to the sediment toxicity data obtained from the TVA (Sect. 6.3.2).

- The tests were performed on aerated water, a practice that may lower toxicity and is not accepted in regulatory testing of water by EPA methods.

- The tests are acute, so they underestimate the toxicity of the water and sediment to the LWBR biota that are chronically exposed.

- The sensitivity of the tested luminescent bacterium and planktonic rotifer relative to fish and benthic invertebrates is unknown.

6.4.3.5 Uncertainties in the biological survey data

- The benthic invertebrate and fish surveys were performed at locations that were not selected by a probability design or by a design that was focused on contaminated areas. Therefore, they are not conservative and cannot be said to be representative.

- The surveys were not conducted in conjunction with contaminant analyses that could be used to identify potentially causal associations.

6.4.4 Comparison to Human Health Risks

The principle difference between the health and ecological risks is that human health risks are based on a data set that was judged to be adequate, while the available data set for ecological risks was judged to be inadequate for ecological risk assessment. Therefore, the ecological risk assessment results are tentative and partial.

Table 6.7 summarizes the COPCs for LWBR. The most conspicuous difference between human health and ecological COPCs is the large number of ecological COPCs for current sediment and the absence of human health COPCs for that medium. This difference results from the fact that current sediments are at the bottom of the reservoir where humans have little exposure but where benthic invertebrates reside, feed, and reproduce. Other differences are attributable to the relative sensitivities of human and nonhuman organisms and to the exposure routes and rates.

Table 6.7. Comparison of contaminants of potential concern for human health and ecological risks

Medium	Human Health[a]	Ecology[b]
Water	Mn	
Fish	Arochlor 1254 & 1260, Chlordane, As, Hg	Arochlor 1254 & 1260, Hg, Se, and Zn
Current sediment		As, Ba, Cd, Fe, Pb, Mn, Hg, Ni, Ag, Zn, Phenanthrene, Pyrene
Dredged sediment	Cd, Cr, Hg, Zn	Cd, Cu, Hg, Ni, Zn

[a]Chemicals listed are those for which the estimated cancer risk is $\geq 10^{-4}$ or the human hazard quotient is ≥ 1 (Table 5.16).
[b]Chemicals listed are those with a hazard quotient for an ecological benchmark ≥ 1 (Tables 6.4–6.6).

6.5 SUMMARY AND CONCLUSIONS

In keeping with the agreement of the FFA parties, this ecological risk assessment was performed with the historical and Phase 1 data to summarize existing knowledge concerning ecological risks rather than to characterize those risks. It is anticipated that more conclusive ecological risk assessments will be performed in the future when Phase 2 data from the Clinch River arm and monitoring data from LWBR are available.

Baseline conditions appear to constitute a significant risk to the benthic invertebrate community. That risk should be confirmed or refuted by appropriate sediment toxicity tests and benthic invertebrate surveys of sediments with defined levels of contamination collected by an appropriate design.

Although the fish community of LWBR currently has low abundance relative to other reservoirs in the region, the cause is unknown. The apparent decline is unlikely to be due to DOE emissions.

Current contaminant concentrations in fish exceed levels shown to cause toxic effects in birds and mammals. There is no independent evidence that piscivorous wildlife are being affected, although such evidence is being collected for the Clinch River arm.

7. FEASIBILITY STUDY OVERVIEW

The purpose of a Feasibility Study (FS) is to identify and evaluate remedial actions that eliminate, reduce, or control risks to human health and the environment. The goal of an FS is to select remedies that are protective of human health and the environment, that maintain protection over time, and that minimize untreated waste. The criteria used in screening, evaluating, and analyzing these actions is provided in EPA RI/FS guidance (EPA 1988) and in the National Contingency Plan (EPA 1990b). The purpose of this FS is to identify, develop, and evaluate potential alternatives for remediation of the LWBR OU. Alternatives developed in this FS form a basis for a decision on possible remedial actions for the LWBR OU. The preferred remedial alternative will be recommended in a Proposed Plan and approved by a Record of Decision.

The FS process followed in this document consists of seven steps. The first step is to summarize the contaminants, media, and exposure pathways that present the greatest risk to human health and the environment as presented in the RI (Sect. 8.1). The second step is preparation of preliminary remediation goals (Sect. 8.1). Next, general response actions are developed that will reduce the contaminant levels or eliminate exposure pathways and meet the preliminary remediation goals (Sect. 8.2). The fourth step in the FS is to identify and screen technologies that fit within each of the general response actions. Technologies are screened to eliminate any that are not technically possible to implement at the specific site. Each technology may have a variety of process options that are somewhat similar in nature (Sect. 8.3). In the fifth step, the remaining technologies are evaluated based on the effectiveness, implementability, and range of cost for each technology and process option. This evaluation will allow the alternatives to be developed from only those technologies or response actions with a higher probability of success (Sect. 8.4). Alternatives are then assembled for detailed analysis from the results of the evaluation of technology process options. This FS is a focused document and the application of the Data Quality Objective process revealed a limited number of practical remedial alternatives. The screening and evaluation of technologies serve to screen the remedial alternatives and leave only viable options for the detailed analysis (Sect. 8.5). The assembled alternatives are fully described in Sect. 8.6. In the final step, alternatives are analyzed (Chapter 9.0) against seven of the nine CERCLA evaluation criteria (Fig. 7.1). The nine criteria identified on Fig. 7.1 are explained in detail at the beginning of Chapter 9, the Detailed Analysis of Alternatives. Any alternative selected must meet two CERCLA threshold criteria (Fig. 7.2). Five primary balancing criteria are used to compare alternatives in the feasibility study and in the Proposed Plan. The last two criteria are modifying criteria and will be addressed following the review of the Proposed Plan. Each alternative is analyzed against the nine criteria and then against each of the other alternatives, so that decision makers can choose the most appropriate remedial alternative for the LWBR OU. The preferred alternative is then presented to the public in the Proposed Plan. Through the FFA, EPA and TDEC have approval authority over the remedial alternative chosen.

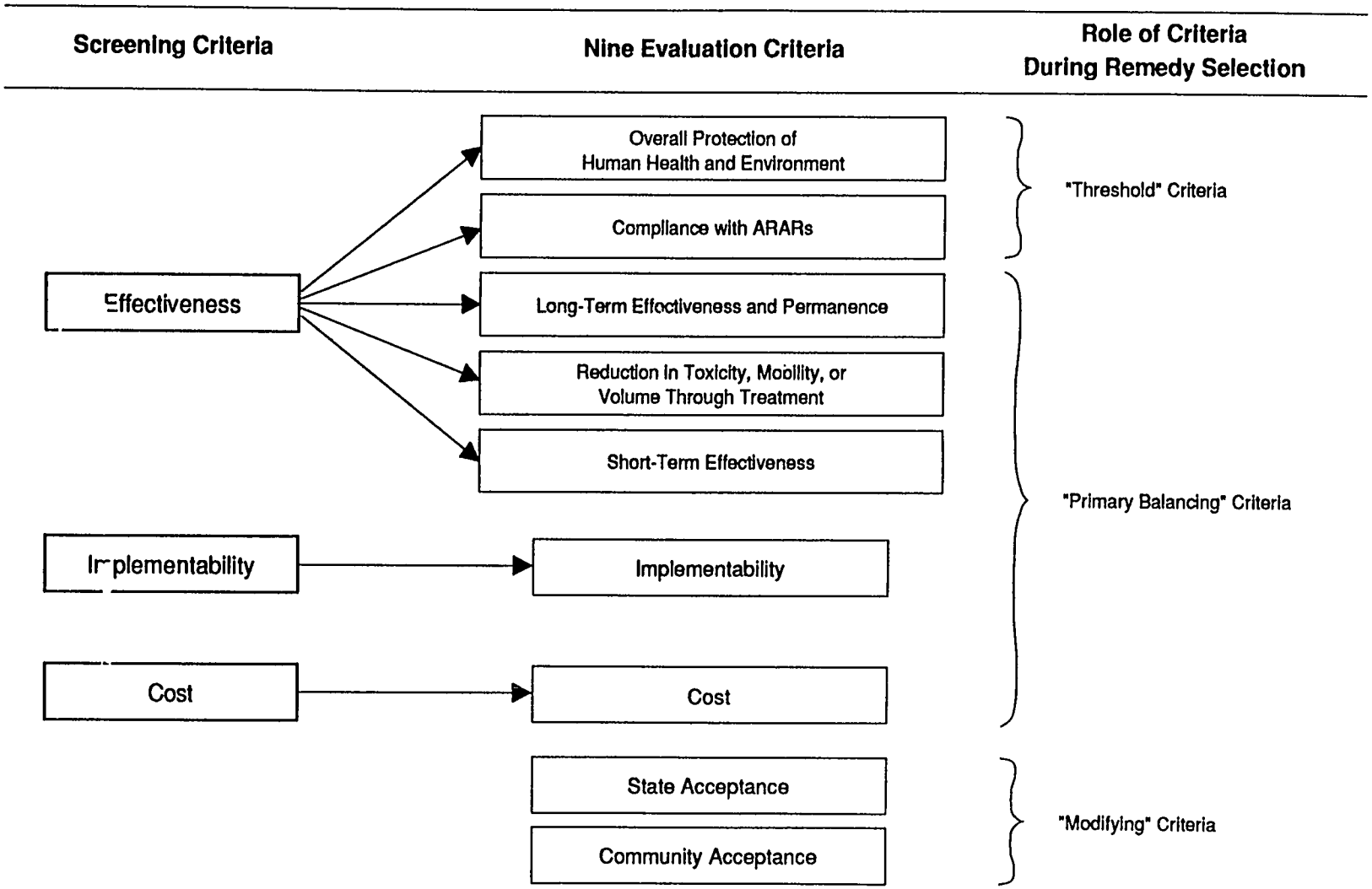


Fig. 7.1. Relationship of screening criteria to the nine evaluation criteria.

8. DEVELOPMENT OF ALTERNATIVES

8.1 PRELIMINARY REMEDIATION GOALS

Preliminary remediation goals are developed based on the results of the RI and the baseline risk assessment and establish concentrations and exposures that comply with ARARs and are protective of human health and the environment. Remediation goals strive to (1) reduce the exposure to a contaminant; (2) reduce the contaminant concentrations in a given medium; or (3) reduce overall risk through some combination of the two. The overall aim of a remediation goal is to ensure protection of human health and the environment.

The potentially contaminated media in LWBR are surface water, sediments, and biota. However, results of the site characterization and baseline risk assessment indicate that the surface water in LWBR meets ARARs and TBC guidance and does not pose an unacceptable risk to human health via any pathway. Therefore, no remediation goal is established here for LWBR surface water. Section 5.4.5 and Tables 5.7, 6.4, 6.5, and 6.6 summarize the contaminants, media, and exposure pathways of concern for human health and the environment.

Some sediments in LWBR are contaminated with inorganic metals and radionuclides (Chapter 3). No direct human health threat exists from exposure to deep-water sediments, because no direct exposure pathway exists. No contaminants in near-shore sediments have been identified that pose an unacceptable risk to human health. The ecological risk assessment, however, indicates that LWBR sediments may be toxic to benthic organisms. Several non-carcinogenic metals in LWBR could be toxic to humans via agricultural use of dredge spoil. This spoil could also be phytotoxic and toxic to terrestrial wildlife feeding on vegetation growing on the spoil. The remediation goal for LWBR sediments is to prevent exposure to toxic metals and radionuclides from agricultural use of dredge spoil by reducing contaminant concentrations in sediments or by regulating and eliminating potential exposure pathways.

PCB contamination of catfish, largemouth bass, smallmouth buffalo, and striped/hybrid bass poses a greater than 1.0×10^{-4} excess lifetime cancer risk to humans via the ingestion pathway. These species could also affect piscivorous wildlife populations by reducing fecundity, although data at this point are insufficient to make this determination. Chlordane in catfish, largemouth bass, and striped/hybrid bass also has a toxic (non-carcinogenic) effect in humans, particularly children. Other toxic contaminants include aldrin in catfish, arsenic in largemouth bass (risk to children only), and mercury in catfish and largemouth bass (risk to children only). Currently, the TDEC has posted a no-consumption advisory in LWBR for catfish and striped and hybrid bass because of PCB contamination. The TDEC has issued a precautionary advisory for white bass, sauger, carp, smallmouth buffalo, and largemouth bass, also because of PCB contamination. The remediation goal for contaminated fish in LWBR is to reduce human exposure to levels that represent an excess cancer risk less than 1.0×10^{-4} or a hazard index less than 1.0 by eliminating or limiting human consumption of these species. The risk to piscivorous wildlife is currently being studied further for more conclusive results.

Unless contaminant concentrations in all media are reduced to levels that are protective of human health and the environment, a long term monitoring program will be implemented to ensure the remediation goals outlined above are being met.

The preliminary remediation goals for LWBR are based on the results of the human health baseline risk assessment. The results of the ecological risk assessment are presently inconclusive and are being studied in more detail in the Clinch River arm of the Reservoir. Therefore, no preliminary remediation goals based on ecological effects are defined here. Actions aimed at remediating the ecological components of LWBR OU are likely to do more damage by killing organisms or habitat. In summary, the remediation goals for human health effects are as follows:

- Limit exposure to metals and radionuclides in LWBR sediments by reducing contaminant concentrations and/or eliminating pathways; and

- Limit the risk from consumption of PCBs in fish to less than 1.0×10^{-4} excess lifetime cancer risk; and limit the risk from consumption of chlordane, arsenic, and mercury to less than 1.0 hazard quotient.

8.2 GENERAL RESPONSE ACTIONS

The general response actions that are applicable to contaminated sediments and contaminated biota are (1) no action, (2) monitoring, (3) institutional controls and advisories, (4) source containment, (5) source removal, and (6) in situ and ex situ treatment. No action is included in the screening and evaluation process as required by CERCLA. Disposal options and turbidity containment options are screened and evaluated as technologies applicable to the containment, removal, and treatment general response actions.

A short description of each general response action, no action, disposal, and turbidity containment is provided in the following sections.

8.2.1 No Action

The no action alternative does not initiate any remedial actions at LWBR and is chosen as the preferred alternative when there is no unacceptable threat or risk to human health or the environment under existing and foreseeable future conditions. This alternative assumes that LWBR could be released for unrestricted public use with no institutional controls or regulations governing activities at the reservoir. These assumptions are consistent with the exposure scenarios used in the baseline risk assessment (Chapter 5). Because contaminants would be left in place, a 5-year review would be required under CERCLA, and a monitoring program will provide data for that review. The purpose of this alternative is to provide a baseline for comparison with the other alternatives.

8.2.2 Monitoring

Monitoring is carried through the screening and evaluation process primarily because it is a common component of all the alternatives that will be selected for detailed analysis and will be a part of the chosen remedial action. To conduct the 5-year review required by CERCLA on a remedial action, data will need to be collected to assess the effectiveness of the action taken. Long-term monitoring of surface water, surface and subsurface sediments (at both deep water and shallow water locations), and ecological indicator species is used primarily to evaluate the effectiveness of the chosen remedial action, to determine whether adjustments or additional actions are needed, and to determine whether existing and future receptors are threatened. Monitoring can be used in conjunction with any other technology or can be considered alone.

8.2.3 Institutional Controls and Advisories

Institutional controls and advisories attempt to reduce risk by reducing exposure to contaminants rather than reducing contaminant concentration or mobility. Institutional controls strive to (1) prevent prolonged exposure to contaminants, (2) control future development and disturbance of the site, and (3) prevent destruction of any engineered remedial actions. The future effectiveness of these options depend on their continued implementation. Institutional controls are subject to change in political jurisdiction, legal interpretations, and regulatory enforcement. If implemented correctly, access restrictions provide low-cost, moderate to high protection against direct contact with contaminants.

8.2.4 Source Containment

Source containment for sediments would consist of placing an engineered cover over the existing sediments to retard contaminant migration and prevent disturbance of the contaminated layer of sediment. Clean sediment, sand, clay, mortar, and synthetic membranes are all potential containment materials. Containment is proven in some aquatic environments, but would be somewhat innovative for a river application.

8.2.5 Source Removal

The process of removing bottom sediments from a water body is commonly known as dredging. Mechanical dredging uses cranes and bucket type devices to grab sediments and lift them to the surface. Hydraulic and pneumatic dredges remove and transport sediment in liquid slurry form. The slurries are then pumped through submerged and floating pipelines to a treatment/storage area onshore or to a barge. The difference in types of dredges is in the suction end and the attachments or methods of control used.

8.2.6 In Situ and Ex Situ Treatment

In situ treatment of sediments attempts to reduce contaminant toxicity, mobility, or volume in place. For metals and radionuclides, the processes used could include soil flushing, stabilization/solidification, chemical oxidation or reduction, or others. In situ options are more limited than ex situ treatments due to the technical difficulties of executing a technology while submerged in a river. Draining the reservoir is not considered since the majority of the contamination is within the river channel and would still be submerged. Draining the reservoir, diverting the river from its channel, and then removing or treating the sediments was not considered in this FS. Treatment of the contaminated biota is not considered in this FS because of the lack of effective treatment technologies.

Ex situ treatment of contaminated sediment is contingent on the prior removal of that sediment. Solids treatment process options include oxidation, reduction, washing, stripping, vacuum extraction, solidification, stabilization, and neutralization. These processes can convert contaminants to less toxic states, more treatable states, or less mobile or leachable states. The treatments could degrade contaminants or remove them completely from the sediments. If RCRA contaminants are present in sufficient concentrations, the treatment may be required before the waste is acceptable for disposal or storage.

8.2.7 Turbidity Containment

Removal, containment, or treatment of sediments in flowing water often generates turbidity caused by resuspension of fine-grained particles. Contaminants would resuspend and be more bioavailable if excess turbidity was generated. Control of turbidity can be accomplished through the use of physical barriers that stop the flow of particles downstream and cause them to settle out or by barriers that surround the operation and prevent any flow in or out of the area. An alternative method involves excavating a depression in the reservoir floor downstream of the dredging, which can cause particles to settle out rather than continue downstream.

8.2.8 Disposal

Disposal options for the sediments will depend entirely upon the characterization of those sediments following removal. It is assumed that if the sediment is contaminated enough to warrant a removal, then it will probably be too contaminated for inexpensive land disposal. Sediments that are hazardous or radioactive are usually containerized and placed into secured, monitored vaults, silos, or buildings, which can be above or below ground. The disposal areas on the ORR are rapidly approaching their capacity, and further space may need to be constructed if the sediments are removed from the LWBR OU. Off-site disposal at another facility may be possible, but as with all disposal options, there are stringent waste acceptance criteria that must be met.

8.3 IDENTIFICATION OF REMEDIAL TECHNOLOGIES AND SCREENING FOR TECHNICAL IMPLEMENTABILITY

This section presents the screening of technologies and process options. Technologies and process options are identified for each general response action. Process options are retained or not retained based on their broad applicability to the site and contaminants and their technical implementability. CERCLA also requires that a no action alternative be evaluated along with any action alternatives for comparison purposes.

Table 8.1 provides a description of technologies and process options under each general response action or category and comments on the implementability or applicability of each. Shaded response actions, technologies, or process options are those that were screened out and will not be considered any further. The darker shading indicates the reason for screening out each technology.

Not all of the technologies or process options presented in Table 8.1 will be carried through all evaluation steps. The no action alternative will be carried through the evaluation process as required by CERCLA. Monitoring will be further evaluated as a component of any remedial action because it offers a means of determining when the remedial action is complete and of detecting adverse changes that can be responded to in the CERCLA 5-year review. Institutional controls and advisories will be further evaluated since they help to eliminate a pathway of risk to human health and the environment. Source containment technologies such as capping and structural isolation will be further evaluated as a way of reducing exposure of humans and biota to contaminated sediments. Turbidity containment will be evaluated further as an addition to another action. In any action taken on LWBR sediments, the reduction of contaminant dispersion through resuspension is a priority. All of the removal technologies and process options will be further evaluated.

Most of the in situ treatment technologies will not be further considered for the following reasons:

- Many of these technologies are not effective for metals;
- Many of these technologies are not proven or have not been demonstrated to be effective in river systems.

Stabilization and solidification will be evaluated as both in situ and ex situ treatment technologies. All of the ex situ physicochemical treatment technologies will be evaluated, as will ex situ vitrification. Disposal technologies will be further evaluated.

Table 8.1. Screening of remedial technologies for Lower Watts Bar Reservoir Operable Unit

Table 8.1				
General Response Action	**Remedial Technology Type**	**Process Options**	**Description**	**Comments**
No Action	None	Not applicable	No action	Required for comparative purposes by CERCLA guidance
Monitoring	Sampling and Analysis	Surface grabs	Sample and analyze surface sediments for contaminants	For detection of recently deposited contamination
		Cores	Sample and analyze deep sediments for contaminants	For characterization of sediment column
		Surface Water Samples	Sample and analyze surface water for contaminants	For risk assessment input and detection of new sources
		Biological samples	Collect and analyze/study aquatic, benthic, and piscivorous organisms to characterize effects from contaminants	Needed for time series analysis of ecological data to determine risk to the environment
Institutional Controls and Advisories	Covenants	Deed restrictions	Restrictions and notification placed in the deeds of impacted property	Would require TVA cooperation
	Public Advisories	Fish consumption/water contact	Posted and published advisories regarding fish consumption limits and water contact warnings	Process in place, administered by TDEC

Legend:

Eliminated technologies TEXT Reason for elimination TEXT

Table 8.1				
General Response Action	**Remedial Technology Type**	**Process Options**	**Description**	**Comments**
Institutional Controls and Advisories continued	Land use restrictions	Permit programs	Require permits for any dredging operations or activities that may disturb the sediment. Restrictions placed in the zoning land use plan	IAG screening process is already in place
	Land acquisition	Land purchase/eminent domain	Purchase or condemnation of land that is potentially affected by contaminants	Current data shows the contamination to be confined to lands controlled by TVA
	Access restriction	Fencing	Erection of physical barrier to prevent access to surface contamination	Would require land owners or TVAs permission
Source Containment	Capping	Clean sediment, armorform, geomembrane	Permeable double-layered, polypropylene or nylon panels are placed on the bottom of a waterway and filled with clean mortar; covering sediment with several feet of clean soil materials; weighted synthetic, flexible, nearly impermeable fabric is sunk to cover contaminated sediment	Undefinable cap life, possible cap penetration by aquatic biota, some caps not suitable for strong currents, would require constant monitoring and possibly periodic repairs
	Structural isolation	Physical barrier	A physical barrier is constructed around a contaminated area. The area is then backfilled to cover the contaminants and isolate them from the environment	Impractical for main channel applications, but could be used in shallower areas

Legend:

Eliminated technologies TEXT Reason for elimination TEXT

Table 8.1				
General Response Action	**Remedial Technology Type**	**Process Options**	**Description**	**Comments**
Source Containment continued	Natural Sedimentation	Natural process and minimized disturbance	Allow relatively clean sediment from the LWBR watershed to build up on top of the contaminated portions of existing sediments	Minimizing manmade disturbances will allow this process to work most efficiently
Technologies/ Process Options for use during containment, removal, or treatment activities	Turbidity containment	Caissons, dikes, sediment traps, cofferdams, silt curtains, and silt screens	Structural or earthen barriers are built around the removal zone; a basin is constructed downstream of the removal zone; synthetic textile sheets are hung from floats downstream of the removal zone	Some are not suitable for deep water or strong currents, some are ineffective in strong currents
Removal	Excavation and mechanical dredging	Backhoe, bucket ladder, bucket wheel, clamshell, dipper and dragline	Various means of using a container to scoop sediment from the river floor and transport it to the surface and/or the shoreline	Limited reach, and a high resuspension rate. More applicable for contamination above waterline
	Hydraulic dredging	Plain Suction, cutterhead, dustpan, hopper, mudcat, dredges	Suction generated by a centrifugal pump is used to dislodge, capture, and transport a sediment slurry through pipelines	Ineffective for dredging of cohesive materials. Requires extensive dewatering of dredged sediments. High rate of resuspension of sediments
	Pneumatic dredging	Airlift and pneuma dredges	Use compressed air to dislodge and transport sediments. Compressed air is released into the bottom of an open vertical pipe and as it expands and rises it carries water and sediment with it.	Minimum dredging depth of 20 or 30 ft is required for economical operation

Legend:

Eliminated technologies TEXT Reason for elimination TEXT

Table 8.1				
General Response Action	**Remedial Technology Type**	**Process Options**	**Description**	**Comments**
Treatment	In Situ Physicochemical treatment	Vacuum extraction, neutralization, air stripping, soil flushing, stabilization/ solidification, ground freezing, chemical oxidation or reduction	Processes include removing contaminants through vacuum systems, forced airflow, and using solvents for extraction. Also the contaminants could be fixed in place by the addition of cement or other chemical mixes, freezing the sediments. The contaminants could be rendered less toxic through neutralization or chemical oxidation/reduction	Site conditions would make these options difficult to implement, expensive, and the effectiveness would have to be proven before implementation. Only stabilization and solidification are applicable to all of the contaminants of concern
	In Situ Thermal treatment	Radio frequency heating, vitrification, steam stripping	Heating sediment mass with radio waves or steam to enhance volatilization and cause chemical bond cleavage, coupled with collection and treatment of gases and condensate. Vitrification turns sediment into a glass-like substance while volatilizing some organic and inorganic contaminants	Currently in pilot- and field-scale demonstration stage. Most are not applicable to the contaminants of concern, and vitrification is unlikely to work underneath the reservoir
	In Situ Biological treatment	Aerobic biosparging, aerobic bioremediation, anaerobic biodegradation	Injection of air and/or nutrients into saturated sediments to enhance aerobic degradation of organics by microorganisms	Not applicable for ^{137}Cs and most of the inorganic contaminants of concern

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.1				
General Response Action	**Remedial Technology Type**	**Process Options**	**Description**	**Comments**
Treatment continued	Ex Situ Physico-chemical treatment	Vacuum extraction, neutralization, air stripping, soil washing, stabilization /solidification, chemical oxidation or reduction	Following removal of the sediments the contaminants could be removed through vacuum systems, forced airflow, and using solvents for extraction. Also the contaminants could be fixed in place by the addition of cement or other chemical mixes, freezing the sediments. The contaminants could be rendered less toxic through neutralization or chemical oxidation/reduction	Potentially necessary following a removal of sediments. Waste may need to be treated to meet storage acceptance criteria. This is contaminant dependent
	Ex Situ Thermal treatment	Radio frequency heating, vitrification, steam stripping	Heating sediment mass with radio waves or steam to enhance volatilization and cause chemical bond cleavage, coupled with collection and treatment of gases and condensate. Vitrification turns sediment into a glasslike substance while volatilizing some organics	Vitrification may be applicable. Steam stripping and radio frequency heating are not applicable for ^{137}Cs or metals
	Ex Situ Biological treatment	Aerobic biosparging, aerobic bioremediation, anaerobic biodegradation	Injection of air and/or nutrients into saturated sediments to enhance aerobic degradation of organics by microorganisms	Not applicable for ^{137}Cs and most of the inorganic contaminants of concern
Disposal	Disposal of containerized sediment	Disposal on the ORR, or shipment to Envirocare in Utah	Use existing space (unlikely) or build new storage facility on the ORR or ship waste to Envirocare facility in Utah (may need treatment to meet acceptance criteria)	Potentially applicable if sediment removal takes place

Legend:

Eliminated technologies TEXT Reason for elimination TEXT

8.4 EVALUATION OF TECHNOLOGIES AND PROCESS OPTIONS

The objective of this evaluation is to provide a basis for the assembly of alternatives in Sect. 8.5. Table 8.2 lists the retained technologies as discussed in Sect. 8.3 and evaluates the performance of those technologies and process options in three areas: effectiveness, implementability, and cost. Again, the shaded response actions, technologies, or process options are those that will not be carried through for any further consideration. The darker shading indicates the reason for screening each particular technology. By eliminating unworkable technologies at this stage, only the viable components are left for incorporation into alternatives. The small number of technologies remaining makes a screening-of-alternatives step unnecessary. A description of the evaluation criteria follows:

- Effectiveness of an option is evaluated on the basis of its ability to accomplish the remediation goals that it is intended to achieve and on its short-term and long-term reliability. The effectiveness of a particular dredging option will be linked to its ability to remove sediment without causing resuspension and dispersement of contamination.

- Implementability or ease of implementation is judged on the basis of the technical and administrative feasibility of constructing, operating, and maintaining a remedial technology or process option. Items included in implementability include construction limitations; safety and ease of operations; and availability of equipment, manpower, and support facilities.

- Cost estimates for all the options have not been developed, and comments in Table 8.2 are based on reference information. Relative comparisons of costs between each option were made for both capital and operating costs. Capital costs are ranked as low, medium, or high:

  low little or no construction,
  medium moderate construction, non-complex equipment, and
  high complex equipment and/or extensive construction.

  Operating and maintenance costs were ranked as low, medium, or high also:

  low little or no monitoring or operator attention,
  medium operator attention, monitoring, or materials necessary, and
  high constant attention and/or large material or energy requirements.

Technologies are evaluated based on the type of site and specific contaminants for which they are suited. If a particular technology is appropriate for deep-water applications but not appropriate for shallow water, then it will be evaluated as a deep-water portion of an alternative. It is unlikely that each technology can excel under all circumstances that will be encountered on the LWBR.

Table 8.2. Evaluation of screened remedial technologies for Lower Watts Bar Reservoir Operable Unit

Table 8.2					
General Response Action	**Remedial Technology Type**	**Process Options**	**Effectiveness**	**Implementability**	**Cost**
No Action	None	Not applicable	Will offer no additional protection	None	None
Monitoring	Sampling	Surface grabs	Good for documenting and evaluating conditions, but does not reduce current contamination	No difficulty	Dependent upon analytical cost. Low capital; medium O&M costs.
		Cores		No difficulty	
		Water Samples		No difficulty	
		Ecological samples		Some difficulty	
Institutional Controls and Advisories	Covenants	Deed restrictions	Effectiveness of these options depends on continued future implementation and natural attenuation.	Could be implemented on TVA owned property.	Negligible costs
	Public Advisories	Fish consumption/ water contact		Already in place	Negligible costs

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.2					
General Response Action	Remedial Technology Type	Process Options	Effectiveness	Implementability	Cost
Institutional Controls and Advisories continued	Land use restrictions	Permit programs		Already in place	Negligible costs
	Land acquisition	Land purchase/eminent domain	No reason to purchase TVA property	Difficult to implement	High capital; low O&M.
	Access restriction	Fencing	Would not mitigate primary risks	Difficult to implement on non-DOE property	Medium capital; low O&M.
Source Containment	Capping	Clean sediment	Unknown	Implementable for smaller areas	Medium capital; low O&M
		Armorform	Unknown		High capital; low O&M
		Geomembrane	Unknown		High capital; low O&M
	Structural isolation	Physical barrier	Effective in shallow water		medium capital; low O&M

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.2					
General Response Action	**Remedial Technology Type**	**Process Options**	**Effectiveness**	**Implementability**	**Cost**
Source Containment continued	Natural Sedimentation	Natural process and minimized disturbance	Effective in the long-term but is a slow process	Already occurring	No cost
Technologies/ Process Options for use during containment or removal activities	Turbidity containment	Caissons	Moderately effective and reliable in shallow waters	Difficult to implement	High capital; low O&M
		Dikes	Moderately effective and reliable in shallow waters	Difficult to implement	Medium capital; low O&M
		Sediment traps	Minimally effective	Implementable	Low capital; low O&M
		Cofferdams	Effective in near shore situations but not in deep water	Implementable	Medium capital; low O&M
		Silt curtains	Minimally effective in shallow or deep water	Implementable	Low capital; low O&M

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.2					
General Response Action	Remedial Technology Type	Process Options	Effectiveness	Implementability	Cost
Turbidity controls continued		Silt screens	Less effective than silt curtains	Implementable	Low capital; low O&M
Removal	Hydraulic dredging	Plain Suction Dredges	Effective but can cause significant resuspension	Implementable	High capital; medium O&M
		Cutterhead Dredges	Highly effective but can cause significant resuspension	Implementable - proven applicability to contaminated sediments	High capital; medium O&M
		Dustpan Dredges	Effective but can cause significant resuspension	Implementable	High capital; medium O&M
	Hydraulic dredging continued	Hopper Dredges	Effective but can cause significant resuspension	Implementable - proven applicability to contaminated sediments	High capital; medium O&M
		Mudcat Dredges	Highly effective - shallow water	Implementable - proven applicability to contaminated sediments	High capital; medium O&M

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.2					
General Response Action	**Remedial Technology Type**	**Process Options**	**Effectiveness**	**Implementability**	**Cost**
Removal continued	Pneumatic dredging	Airlift Dredges	Effective - deep water (less effective than pneuma)	Implementable	High capital; medium O&M
		Pneuma Dredge	Highly effective - deep water	Implementable - proven applicability to contaminated sediments	High capital; medium O&M
	Excavation and mechanical dredging	Backhoe	Effective shallow water or onshore	Implementable	Low capital; low O&M
		bucket ladder/wheel	Effective but high resuspension rates and shallow water only	Implementable	Medium capital; medium O&M
		clamshell	New "sealed" clamshell could be very effective, but is slower than pneumatic	Implementable - proven applicability to contaminated sediments	High capital; medium O&M
		dipper and dragline	Effective but high resuspension rates	Implementable	Medium capital; medium O&M

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.2					
General Response Action	Remedial Technology Type	Process Options	Effectiveness	Implementability	Cost
Treatment	In Situ Physico-chemical treatment	Stabilization and solidification	Could be effective if treatability study is performed	Difficult to implement	High capital; medium O&M
	Ex Situ Physico-chemical treatment	Vacuum extraction	Not effective for metals or ^{137}Cs	No reason to implement	High capital; medium O&M
		Neutralization	Can be effective as a pretreatment for oxidation or reduction	Implementable	High capital; medium O&M
		Air stripping	Not effective for metals or ^{137}Cs	Implementable	High capital; medium O&M
Treatment continued		Soil Washing	Could be effective if sediment characteristics aren't prohibitive	Difficult to implement	High capital; medium O&M

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.2					
General Response Action	**Remedial Technology Type**	**Process Options**	**Effectiveness**	**Implementability**	**Cost**
Treatment continued		Stabilization/ Solidification	Effective when sufficient pilot scale work is done. Increases volume of contaminated material	Difficult to Implement	High capital; medium O&M
		Chemical oxidation or reduction	Not effective for ^{137}Cs	No reason to implement	High capital; medium O&M
	Ex Situ Thermal treatment	Vitrification	Effective	Difficult due to potential volumes, and innovative status	High capital; High O&M

Legend:

Eliminated technologies TEXT

Reason for elimination TEXT

Table 8.2					
General Response Action	Remedial Technology Type	Process Options	Effectiveness	Implementability	Cost
Disposal	Storage or Disposal of containerized sediment	Storage or Disposal on the ORR	Effective	Difficult to implement and treatment may be required	High capital; medium O&M
		Disposal at Envirocare	Effective	Difficult to implement and treatment may be required	High capital; medium O&M

Legend:

Eliminated technologies TEXT Reason for elimination TEXT

Of the general response actions in Table 8.2, all will be retained in the alternatives assembled for detailed analysis. The text below will further describe the reasons for screening the shaded technologies.

The following technologies will not be retained for inclusion into an alternative: land acquisition, access restriction, capping, structural isolation, excavation and mechanical dredging, fish removal, in situ physicochemical treatment, and thermal treatment.

- Land acquisition is not necessary since the contaminated property is already owned by a federal entity. Access restrictions for the entire reservoir are not practical and denying access to only a portion of the reservoir would not address the primary risk pathways.

- Capping and structural isolation of the sediment will not be retained. Should a pathway of risk from the deep sediments exist, the addition of a permeable cover over that sediment will not alter that pathway. An impenetrable cover or permeable cover in the main channel would likely obstruct navigation in certain areas.

- Excavation and mechanical dredging are typically shallow water or shoreline technologies and are prone to higher resuspension rates than other removal technologies.

- In situ treatment would likely be less effective, harder to implement, and more expensive than other technologies and is, therefore, not retained for inclusion into an alternative.

- Thermal treatment would be a more costly technology than other treatments that could achieve similar results.

Of the hydraulic dredges reviewed, the mudcat is most applicable for implementation in shallow areas of contamination with minimal resuspension of sediments. Pneumatic dredging offers comparable or better effectiveness with potentially less resuspension of sediments for deep water areas. The specific turbidity containment option should be chosen during the design of a remedial action, but for screening and evaluation purposes, the silt curtain was considered the most desirable due to its low cost, ease of implementation, and proven effectiveness in both shallow and deep waters. The pneuma dredge will be chosen to represent the pneumatic dredging technologies due to its high effectiveness and proven applicability to contaminated sediments (EPA 1990a).

The ex situ treatment methods and disposal methods will not be specified at this time. The determination of which treatment technologies or disposal methods to use on the removed sediments is best made during the design phase and may require additional sampling and treatability studies. Typically, before a treatment system is chosen and constructed, bench and pilot scale studies need to be performed to ensure that the treatment in question will work as planned with the specific waste and contaminant loads found on the site. The accuracy of the removal process could affect the characterization of the waste sediments by increasing or decreasing the concentration of the contaminants of concern. A more accurate removal event could render a sediment with higher concentrations of contaminants. A less accurate removal operation would blend the highly contaminated portions of the sediment with large volumes of relatively clean sediment and lower the overall contaminant concentrations. Until a more accurate characterization of the removed and dewatered sediments can be formed, it is difficult to predict if the waste will meet acceptance criteria or will be classified as a mixed RCRA low-level radioactive waste.

8.5 ASSEMBLY OF ALTERNATIVES

CERCLA states a statutory preference for remedies that will result in permanent and significant decreases in toxicity, mobility, or volume through treatment and provide long-term protection. The primary requirements for the final remedy are that (1) it must protect human health and the environment and (2) it must comply with other ARARs.

In addition to the above objectives, NCP defines the following preferences in developing and screening remedial action alternatives:

- use of treatment to address the principal threats posed by a site, wherever practical;
- use of engineering controls, such as containment, for waste that poses a relatively low long-term threat and for which treatment is not practical;
- implementation of a combination of actions, as appropriate, to achieve protection of human health and the environment. For example, in appropriate situations, treatment of principal threats would be combined with engineering controls for treatment residuals or untreated waste;
- use of institutional controls, such as deed restrictions or fencing, to supplement engineering controls for short- and long-term management to prevent or limit exposures to hazardous substances;
- selection of innovative technology when the technology offers the following: the potential for comparable or better treatment performance or implementability, fewer number of or lesser magnitude adverse impacts than other available approaches, or lower costs than demonstrated technologies for similar levels of performance; and
- restoration of environmental media, such as groundwater, to their beneficial uses whenever practicable within a reasonable time frame. When restoration is not practical, EPA expects to prevent further migration or degradation, prevent exposures to human and environmental receptors, and evaluate further risk reduction.

These preferences are applied in the evaluation and analysis in the following sections. Based on the screening and evaluation data from Sects. 8.3 and 8.4, the most promising technology process options will be combined as appropriate to form the alternatives. These alternatives represent the full range of potentially viable remedial actions, from no action at all to complete removal and disposal of any contamination above risk based action levels. The number of process options and response actions remaining after the evaluation step is small enough to eliminate the need for a screening of alternatives.

Some alternatives may consist of a variety of technologies covering the various situations that may be encountered in a remedial action. Because of the nature of the contamination and the site, every technology option would provide some reduction of toxicity, mobility, and volume. Radioactive decay will reduce the mass of ^{137}Cs and thus the carcinogenicity of the sediment, while natural sedimentation will cover the existing sediments and further reduce the mobility of the contaminated layer of sediment. Treatment processes could speed up this natural process and could address non-radioactive contaminants, but the implementation of in situ treatments of sediment would likely do more damage to the environment than no action. The implementation

of engineering controls such as containment would also damage habitat and possibly resuspend sediments. Choosing among containment, removal, and in situ treatment will be dependent, in part, on the amount of habitat destruction and sediment resuspension that is considered acceptable. The preference for institutional controls is addressed in the options retained. None of the innovative technologies considered offer better performance than established technologies, fewer impacts to the environment, or lower costs. The beneficial use for the sediments of LWBR OU is that they provide a habitat for benthic invertebrates. No human usage of the sediments occur. It is probably not possible to restore the sediments for the benefit of the environment without destroying the benthic organisms in the process. If it is decided that restoration of the sediments is not necessary or practical, then other technology options will provide for prevention of further migration or degradation, prevention of exposure to human and environmental receptors, and evaluation of future risk and further actions.

Based on all available information, the remaining process options are combined to form the following alternatives:

Common Components - Based on CERCLA guidance it would be necessary and prudent to conduct monitoring of LWBR. The monitoring will identify changes, identify potential threats to human health or the environment, and gather the data necessary to complete an assessment of ecological risk for LWBR.

Natural remediation consists of radioactive decay, biodegradation of organics, and the addition of new sediment on top of old contaminated sediments. These processes will occur and be a part of any alternative chosen.

Alternative 1 The No Action alternative will be carried through the detailed analysis per CERCLA guidance.

Alternative 2 Institutional Controls and Advisories will be an alternative. The process options chosen to represent this alternative include deed restrictions on TVA owned land, fish consumption advisories and all other protective federal and state laws, and the IAG program governing dredging restrictions. The two institutional controls that were not used were difficult to implement and provided little additional protection above and beyond the controls chosen. The contaminated sediments are all below the elevation considered to be under TVA control/ownership and thus need not be purchased from the public. The contaminated sediments are not likely to be contacted by the public, even if there is no fencing; therefore, fences would be unnecessary. The chosen institutional controls are all low in cost, easily implemented, and offer excellent effectiveness at eliminating pathways of exposure for both humans and the environment. Monitoring is necessary to determine if further actions become necessary or if controls can be safely released in the future.

Alternative 3 Sediment Removal was the only retained response action that will reduce the level of contamination within LWBR OU. Several process options will be incorporated into this alternative to cover the realm of likely possible situations. All of the controls and advisories that are part of Alternative 2 will be included in Alternative 3 as necessary during and after removal. This removal alternative is based on the highest category of contamination outlined on Figure 3.22, and would cover approximately 5,000 acres of main channel sediment. If the area is in shallow water, then a mudcat type of dredge and silt curtains would probably be most effective at removing contaminated sediments with a minimum of resuspension. In a deep water area, a pneuma type dredge and silt curtains would probably be most effective. The mudcat and pneuma

dredge technology types were selected because the literature reviewed suggests that these dredge types offer the lowest rate of resuspension. Should other evidence become available that another dredging technology is superior in minimizing resuspension of sediments, then that technology should be evaluated for inclusion in this alternative. The silt curtain offers a means of further reducing the resuspension of contaminated sediments in shallow or deep water and with low cost and minimal effort. Following the removal of the sediment, several other activities would need to take place, but the exact activities would be dependent on the analytical characterization of the dredged sediments and the disposal/storage space situation at the time of the dredging. At the least, this alternative would include dewatering the sediments, treating/disposing of the dewatering effluent, containerizing the sediments, and storing the containers. Depending on the storage or disposal option chosen, some form of treatment may be necessary. The likely treatment technologies would be neutralization, soil washing, stabilization/solidification, or chemical oxidation or reduction. The treatment and disposal technologies would need to be chosen closer to implementation time and would probably be based on the volume and characterization of the sediments to be removed.

Section 8.6 describes in detail each of these three alternatives.

8.6 DETAILED DESCRIPTION OF ALTERNATIVES

8.6.1 Common Components of Each Alternative

Monitoring Plan

Monitoring of water, sediment, and biota will be conducted at a level sufficient to allow the identification of any conditions that may pose a threat to human health or the environment. Monitoring will be coordinated with existing efforts by TVA, TDEC, and other federal and state agencies to avoid duplication of effort. A monitoring plan will be developed by DOE and approved by EPA and TDEC. Monitoring will begin in early 1995 and will continue for at least 5 years. Annual data summary reports will be produced and made available to the public. Data collected will be used in the 5-year review of the ROD required by CERCLA.

Natural Remediation

Three types of natural remediation are continuously occurring in LWBR: (1) radionuclides in sediments continue to naturally decay, (2) relatively clean sediments are accumulating in LWBR which serve to isolate existing contaminants and provide benthic habitat, and (3) biodegradation of any organic contaminants is constantly occurring.

The sedimentation that is continuing to occur may still include some contamination from non-DOE sources and DOE residuals form historic releases; however, it will be cleaner that the buried layer of sediment that contains the historic releases of mercury and ^{137}Cs.

8.6.2 Alternative 1: No Action

The no action alternative involves no actions or restrictions for the LWBR OU other than the two common components. The purpose of including the no action alternative in this FS is to provide a baseline to which the other alternatives can be compared. Under this alternative DOE would not provide any institutional controls, and no actions would be taken to contain, treat, or

remove any levels of contamination from the LWBR OU. Natural sedimentation would continue to cover the contamination with cleaner sediment, and radioactive contamination would continue to decay. Existing agreements with TVA, COE, TDEC, or EPA that affect the LWBR OU could become nonexistent, as could all federal and state laws applicable to LWBR. Under CERCLA, the no action alternative is selected if the baseline risk assessment reveals that there are no remaining unacceptable health or environmental risks due to the site. No action would leave the LWBR OU open for unrestricted use, including all recreational activities, dredging and use of dredged sediments, and fish consumption. Existing laws covering dredging and fish consumption (see Chapter 4 and Sect. 8.5) could no longer be in effect for LWBR. The risk assessment scenarios outlined in Chapter 5 could all take place under the no action alternative.

8.6.3 Alternative 2: Institutional Controls and Advisories

This alternative assumes the continuance of all existing laws and controls on LWBR activities and includes the two common components and one additional action. The common components are a monitoring plan and natural remediation. The additional action is placing restrictions on land use in the deeds for all LWBR OU land owned by TVA.

One of the major threats to human health posed by LWBR OU is consumption of certain species of fish (see Chapter 5). Under the Tennessee Water Quality Control Act, TDEC has the authority to issue fish consumption advisories to protect the public. The Division of Water Pollution Control in TDEC currently maintains posted fish consumption advisories at approximately 50 public and private access points surrounding the LWBR OU. Two types of fish advisories are issued for LWBR OU. A precautionary advisory, the mildest form of advisory, warns children, pregnant women, and nursing mothers to avoid consumption of whitebass, sauger, carp, smallmouth buffalo and largemouth bass. All others are warned to limit consumption of those fish to 1.2 lb per month. A no consumption advisory warns the public to avoid eating catfish, striped bass, and hybrid striped bass whitebass in any amount. The LWBR OU advisories are based on the presence of PCBs in fish tissues. The TDEC staff proposal for revisions to fish advisory procedures will change the standards so that the "no consumption" advisory is for typical consumers and is protective to a level of 10^{-4}, while the "precautionary advisory" is for sensitive consumers such as pregnant women and children and is protective to a level of 10^{-5} risk of additional cancer deaths. When an advisory is issued or changed, a press release is issued and signs placed at highly used access points. A list of the advisories is also printed in the annual Tennessee fishing regulations brochure. Phone numbers are also provided if the public desires further information regarding an advisory (TDEC 1200-4-3).

TDEC also provides standards for domestic water supplies and Water Quality Criteria for recreational waters. These standards protect the public by ensuring that drinking water taken from LWBR OU is treated to a level that is safe for public consumption and by ensuring that contaminant levels in the reservoir are low enough to be safe for recreational use.

Dredging of contaminated sediments and agricultural use of dredged material is another human and environmental health threat identified in the risk assessment. Several federal and state regulations currently control the public's ability to conduct dredging or other sediment disturbing activities in LWBR. Structures or work affecting LWBR below the ordinary high water mark are regulated by the COE under Sect. 10 of the Rivers and Harbors Act and by TDEC under Rule 1200-4-7. TVA regulates actions on the land below 750 ft elevation under the TVA Act, Sect. 26a, navigation obstructions potentially caused by a dredging operation, and the use of any TVA boat ramps for staging a dredging operation. The Fish and Wildlife Coordination Act requires

prevention, mitigation, or compensation for damages to fish and wildlife resources. Should a dredging operation occur, the disposal of the dredged materials could be regulated under several different federal laws. The Clean Water Act regulates the discharge of dredge or fill material into LWBR. If land disposal of dredged sediments is attempted, then RCRA could govern the disposal of those sediments (depending on characterization). DOE Order 5400.5 requires DOE to ensure that the radiation exposure to the public remains as low as reasonably achievable and prevents the unrestricted placement of radioactively contaminated sediments on agricultural land.

The IAG for Watts Bar Reservoir Permit Coordination among TVA, Nashville District COE, DOE, EPA, and TDEC establishes a procedure for review and permitting of potential sediment disturbing activities in LWBR. The Interagency Working Group reviews activities for their potential to disturb contaminated sediments. Examples of activities the Working Group reviews are construction of beaches, boat ramps, docks, marinas, buoy anchors, fences, fish attractors, retaining walls, pump stations, culverts, and submarine crossings of lines or pipes. Activities on LWBR are divided into three categories: No Significant Sediment Disturbance, Marginal Sediment Disturbance, and Potential Major Sediment Disturbance. After reviewing available data on the location involved with each request, acceptable activities are permitted or allowed to take place. DOE responsibilities include reviewing existing data, providing copies of existing data, acquiring data as necessary, and providing technical analysis and risk assessment support. The agreement applies to the Tennessee River from TRM 529.9 (Watts Bar Dam) to TRM 569, the Clinch River from CRM 0.0 (confluence with the Tennessee River) to CRM 23.1 (Melton Hill Dam), and Emory River from mile 0.0 (confluence with the Clinch River) to mile 5.2 (downstream from confluence of the Little Emory River.) This IAG will continue for as long as the agencies mutually agree that a need for screening of potential sediment disturbing activities exists. Implicit within the agreement is the response of DOE and the other agencies to a perceived threat to human health should sampling reveal high contaminant levels in an easily accessible location.

Placing a restriction in the deeds for LWBR OU lands owned by TVA would help to ensure that future land use is consistent with the contamination levels that remain in the sediments. If, at some unknown point in the future, a portion of the sediments is no longer covered by the reservoir waters and is offered for sale, a restriction on potential sediment disturbing activities would be in place for protection of public health and the environment. A list of all types of activities that could present a health threat would be compiled and a decision matrix produced. If sampling of the deeded land indicated little or no contamination, then restrictions could be lifted. Deed restrictions on TVA land would provide long-term protection at a minimal cost. The restrictions placed in each deed would provide information regarding the risks associated with sediment disturbance or use of sediments for agricultural or construction purposes. Land sale would be dependent upon compliance with restrictions on use of the exposed land/sediments.

8.6.4 Alternative 3: Sediment Removal

This alternative would include the items outlined in Alternative 2 (Sect. 8.6.3) as needed following remedial action and will include the two common components. The exact regulations, controls, and agreements would be dependent upon the residual contamination left in place after removal was completed. The other basic components of this alternative are

- mudcat or pneuma dredge processes to remove contaminated sediment from approximately 5,000 acres of the main channel, thus removing a majority of the historical contamination;

- silt curtain technology to help control the resuspension of contaminated sediments during dredging operations;
- pumping of the dredging slurry to a shoreline facility located above the 750 ft elevation line (above the highest potential waterline);
- dewatering equipment, collection of dewatering effluent, and sediment containerization equipment at the shoreline facility;
- transportation (as necessary) of waste water to treatment or disposal facilities;
- support equipment/watercraft for both marine and onshore activities, including change and decontamination trailers for personnel and decontamination facilities for equipment;
- heavy equipment for site preparation at the shoreline and for mobilization and demobilization of dewatering facilities; and
- transport of containerized sediments to a treatment, storage, or disposal facility.

The following is a more detailed description of the technologies used in this alternative. Hydraulic dredges remove and transport sediment in liquid slurry form. The slurries may be pumped many thousands of feet through floating or pontoon-supported pipeline to a dredged material treatment/storage area. The mudcat is pontoon-mounted and features a horizontally mounted, auger-like cutting device that feeds the sediment to the suction intake of a diesel-driven centrifugal pump. The auger is mounted along the base of a bulldozer-type blade. The whole arrangement with suction pipe attached is controlled by a hydraulic boom. The dredge is moved along on an anchored cable during each traverse of the excavation, and the dredge material is discharged ashore through a float-supported pipeline. The width of the cut is approximately 8 ft, and applications have included dredging of small reservoirs and streams. Portable dredges are particularly applicable to contaminated sediment remediation projects in water bodies such as lakes and inland rivers. A series of tests on the mudcat sponsored by EPA in the 1970s showed that resuspension of the sediment was low and the resuspension plume imparted to the surrounding water during dredging was confined to within 20 ft of the dredge. The mudcat was also found to be very efficient in removing simulated hazardous material (e.g., powder, fine glass beads, filings, and coal), removing 99.3% from the sediments of a test pond. The primary limitation of portable dredges is their size (EPA 1985). Small size restricts portable dredges to low production rates, which limits them to small-scale operations. The mudcat is capable of obtaining a precision of 1 ft in the vertical and horizontal portion of the dredge area. Slurry solids content is typically 10 40% (EPA 1990a).

Pneumatic dredges feature a pump that operates on compressed air and hydrostatic pressure to draw sediments to the collection head and through the transport piping. Otherwise, pneumatic dredges are very similar to hydraulic dredges. Pneumatic dredges are able to yield denser slurries than conventional hydraulic dredges with lower levels of turbidity and resuspension of solids. The pneuma dredge consists of a pump that is lowered by a crane into the sediments being dredged. The pump is driven by compressed air and operates by positive displacement. The body of the pump contains three cylindrical vessels, each with an intake opening on the bottom and an air port and a discharge outlet on top. The air ports can be opened to the atmosphere through air hoses and valves. The three cylinders operate in parallel, each one-third cycle ahead and behind the other two cylinders and controlled by an air distributor located on the control vessel. Pneuma

dredges are normally suspended from a crane cable and pulled ahead into the sediments being dredged by a second cable. The dredge head is essentially fixed relative to the vessel so that lateral manipulation of the dredge is limited to the positioning and movement of the vessel. Though pneuma dredges are most applicable to loosely consolidated sediments, the intake openings can be fitted with shovel attachments to aid in penetration of sediments. Extremely deep applications are possible, limited by vertical and lateral controllability and air pressure requirements. They are capable of delivering a slurry of high solids content with minimal turbidity generation. Tests showed that the pneuma pump was able to dredge at almost in-situ density in a loosely compacted, silty clay typical of many estuarine sediments while generating a low level of turbidity (EPA 1985). The pneuma pump is capable of obtaining a horizontal precision of 2 3 ft and a vertical precision of 1 ft in the dredge area. Slurry solids content is typically up to 80% of in-situ density (EPA 1990a).

The contractor who performs the dredging should adhere to maximum allowable turbidity levels specified by DOE. Resuspension of sediments is a likely occurrence during dredging no matter which methods are chosen for dredging and turbidity control. Silt curtains are low permeability floating barriers that extend vertically from the surface of the water to a specified depth and are used to control turbidity in the vicinity of small dredging operations. Silt curtains are best suited to use in quiescent waters and in conjunction with dredges that do not demand frequent curtain movement. Under these conditions, turbidity levels outside a curtain may be reduced by 80 to 90%. The flexible skirt is generally constructed from polyester-reinforced PVC or nylon-reinforced PVC. Skirts made of KEVLAR/polyester blend are also available. The skirt is maintained by a ballast chain along the bottom. A tension cable is usually built into the upper part of the curtain to absorb stress imposed by currents. End connectors are available to allow two sections of curtain to be attached or disengaged easily. Anchored lines are used to hold the curtain in the deployed configuration. The in-stream, U-shaped configuration is generally acceptable on a river where the current does not reverse (EPA 1985 and 1990).

The dredging slurry would be pumped to a temporary area set up on the shoreline nearest the dredge but above the 750 ft elevation to avoid potential high water. The shoreline area would contain mobile support facilities for the personnel, dewatering equipment for the slurry, a collection system for the dewatering effluent, and containerization equipment for the dewatered sediments. For cost estimating purposes it was assumed that 60 filter presses (200 ft^3 capacity each) would be used to dewater the sediments. A feed tank with a mixer would be used as a feed mechanism for the presses, and the presses would be situated on a platform to discharge the cake into containers for transport. B-25 7ANV boxes are designed to handle solids with a significant moisture content, and those were priced for the containerization of the filter press cakes. Energy Systems guidance on storage cost was used for pricing storage of the full boxes at the K-25 Site. The storage and transport cost may be higher if the sediments or dewatering effluent are determined to be mixed RCRA low-level radioactive waste or if the sediment needs to be treated before disposal. The effluent water collected from the filter press could be transported to the ORNL Process Waste Treatment Plant for treatment/disposal. Licensed waste haulers would be used for transporting the water and the sediments. Analysis of both wastes would need to be completed before transport so that the proper contracts and placarding could be implemented. As discussed in Sects. 8.2 and 8.3, final decisions regarding the treatment and disposal methodologies would need to be made following further characterization of the volume and post removal characteristics of the sediments. No cost estimates were prepared for treatment of the sediments or for disposal other than on the ORR.

Should this alternative be chosen, an Environmental Assessment may need to be prepared to comply with NEPA requirements.

8.7 SUMMARY OF ACTION-SPECIFIC ARARs FOR EACH ALTERNATIVE

Performance, design, or other action-specific requirements set controls or restrictions on particular kinds of activities related to the management of hazardous waste (52 FR 32496). Selection of a particular remedial action at a site will invoke the appropriate action-specific ARARs that may specify particular performance standards or technologies, as well as specific environmental levels for discharged or residual chemicals. Potential chemical- and location-specific ARARs have been discussed in the RI portion of this report (Chapter 4); chemical-, location-, and action-specific ARARs for alternatives 1 3 are listed in Table 8.3.

8.7.1 Alternative 1 No Action

The only ARAR for this alternative is the CERCLA requirement that any remedial alternative chosen will be protective of human health and the environment.

8.7.2 Alternative 2 Institutional Controls and Advisories

There are no chemical-, location-, or action-specific ARARs for this alternative other than that listed in 8.7.1 above; however, there is TBC guidance for institutional controls when residual radioactivity is left in place. These controls are listed on Table 8.3.

8.7.2.1 Institutional controls

Institutional controls will be implemented to limit access and exposure. There are no regulatory requirements specifying institutional controls for CERCLA units. However, DOE Order 5400.5, Chapter IV, requires administrative (institutional) controls for long-term management in areas containing residual radioactivity. Active controls that are specified in the Order and that may be considered TBC guidance include land restrictions, fences, and warning signs.

8.7.2.2 Watts Bar interagency agreement

An IAG for Watts Bar Reservoir Permit Coordination has been signed by EPA, TDEC, the COE, TVA, and DOE. Under this agreement, a screening list identifies three categories of actions disturbing sediments: no significant sediment disturbance; marginal sediment disturbance; and potential major sediment disturbance. The screening list of action categories was amended to include in the NSSD category all proposed projects (other than between CRM 4.3 and WOC) if the project does not have any sediment disturbance below elevation 735 msl and provided the work is accomplished when pool elevation is below the project site (Amendment 1, IAG Watts Bar Reservoir Permit Coordination, January 28, 1992).

For permit applications for activities that fall within the marginal sediment disturbance or potential major sediment disturbance categories, DOE must supply existing sediment data and acquire additional sediment data if needed [IAG, Sect. IV(C)]. This IAG may be considered TBC guidance for institutional controls at the LWBR OU.

8.7.2.3 Fish advisories

Under the Tennessee Water Quality Control Act, Sect. 69-3-107(15), authority is granted the Commission to "inspect waters of the state where good cause is shown that the public health is threatened by pollutants therein, and, upon verification by the commissioner, post or cause to be posted such signs as required to give notice to the public of the potential or actual dangers of specific uses of such waters or restrictions of uses thereof." To that end, posting of advisories is not an ARAR or TBC for remediation of the LWBR OU; however, it implements the institutional controls suggested for Alternatives 2 and 3 (i.e., posting of warning signs).

Currently, the TDEC Division of Water Pollution Control develops fish advisories based on FDA action levels for poisonous or deleterious substances in human food. "Do not consume" advisories are issued when warranted for the protection of typical consumers and a "precautionary advisory" is issued when warranted for protection of atypical consumers. Atypical consumers are those especially sensitive due to physiological factors or previous exposures.

The TDEC Division of Water Pollution Control has issued a staff proposal for revisions to Chapt. 1200-4-3 and 1200-4-4 of the TDEC Water Quality Standards. This proposal includes the modification and promulgation of procedures for issuing fish advisories. The revision would create a new subparagraph regarding fish consumption advisories [TDEC 1200-4-3-.03(j)]. Such advisories would be warranted if the calculated risk of additional cancer incidence exceeds 10^{-4} for typical consumers and 10^{-5} for atypical consumers. The proposed change includes a methodology for calculating risk. Notices are posted for "Do not consume advisories" and "Precautionary advisories," as described above.

8.7.3 Alternative 3 Sediment Removal

All of the ARARs for Alternative 2 may apply to this alternative depending on actual implementation. There are no chemical-specific ARARs for this alternative. However, there are radiation protection requirements that may be applicable or TBC (Sect. 4.2), which are listed on Table 8.3. If dredging uncovers any archaeological resources of significant importance or impacts any threatened or endangered species, several location-specific ARARs may be triggered. These are discussed in Chapter 4 and are summarized here on Table 8.3. Several action-specific ARARs may be triggered by this alternative, most of which depend on characterization of the dredged sediments (i.e., whether they contain RCRA hazardous waste and/or low level waste.

8.7.3.1 Dredging activities

CERCLA Sect. 121(e) stipulates that permits are not required for on-site CERCLA cleanup. CERCLA defines "on-site" to mean "the areal extent of contamination and all suitable areas in very close proximity" (40 CFR 300.5). In general, this will apply to remedial actions taken in the LWBR OU, and permitting will not be required. However, CERCLA activities must comply with the substantive requirements of any permitting process. COE Regulatory Guidance Letter 85-7 and its extension adopt this policy for on-site EPA or state response actions at the location of the release or threatened release pursued under the authority of CERCLA.

The TVA Act Sect. 26a regulates lands under TVA jurisdiction. TVA lands are those below 750 ft elevation. No dredging activities conducted in the LWBR OU are regulated by the TVA Act. Certain activities, such as road construction, require a Section 26(a) authorization, and use of TVA public use boat ramps for dredging operations requires land use permits.

The COE requires permits for structures or work in or affecting navigable waters of the United States under Sect. 10 of the Rivers and Harbors Act. In addition, the discharge of dredged or fill material into waters of the United States is regulated under CWA Sect. 404 and implemented through 33 CFR 320-330 and 40 CFR 230. Both 33 CFR Sects. 322.2 and 328 define "waters of the United States" as those waters of the United States that are subject to the ebb and flow of the tide toward the mean high water mark and that have been used at any time or may be used for interstate commerce. This definition includes all impoundments of such waters [40 CFR 328.3(a)(4)]. The COE has established the ordinary high water mark for impoundments to be the normal summer pool elevation; for Watts Bar Lake, this elevation is 741.0 msl. Any activity shoreward of this elevation would not require approval pursuant to Sect. 10 of the Rivers and Harbors Act, and any activity above this elevation that would not involve the discharge of dredge or fill material into waters of the United States would not require approval pursuant to Sect. 404 of the Clean Water Act.

Remedial actions that may be considered dredge-and-fill activities include dredging of sediments from contaminated water bodies, disposal of contaminated soil or materials in surface water, capping of a site, building berms and levees to contain wastes, excavation for the containment of effluent, and dewatering of a site to obtain adequate flow. An activity may be authorized under a COE Nationwide Permit (NWP) if it meets all the terms and conditions of the NWP. Thirty-nine categories of NWPs are designed to regulate certain dredge-and-fill activities with minimal impacts. These permits are described in Appendix A to 33 CFR 330. There are two NWPs that may be used for activities on the Clinch River. NWP 19 governs dredging operations of 25 yd^3 or less that involve a single and complete operation; it does not authorize dredging or degradation through siltation of submerged aquatic vegetation or wetlands. NWP 38 authorizes activities required to effect the containment, stabilization, or removal of hazardous and toxic materials by a government agency with established legal or regulatory authority, provided the COE District Engineer is notified in accordance with the ``Notification'' general condition [33 CFR 330, Appendix A(C)(13)]. For discharges in special aquatic sites, including wetlands, a wetlands delineation must be submitted with the notification.

General conditions for authorization of an NWP contain substantive requirements for remedial activities at LWBR OU (33 CFR 330, Appendix A). These requirements are summarized on Table 8.3; other general conditions may apply depending on actual remedial activities. If the dredging activity does not meet all the requirements of an NWP, an individual permit application may be required, with further substantive requirements dictated by the COE.

8.7.3.2 Use of shoreline facility for treatment and/or packaging

Dredged slurries may be pumped to a shoreline facility for dewatering and sediment containerization. It is assumed that the slurries will contain low level radioactivity. Decontamination procedures will be implemented for all equipment and personnel. A waste analysis must be performed to determine whether the sediment and dewatering/decontamination fluids also contain RCRA hazardous waste [40 CFR 262.11(b); 40 CFR 264.13; 40 CFR 268.7], which would classify them as mixed waste.

Treatment and disposal of dewatering/decontamination fluids

If dewatering/decontamination fluids contain mixed waste, they must be transported to the ORR and treated or stored per the EPA/DOE Mixed Waste Federal Facilities Compliance Agreement (FFCA). Dewatering fluids that do not contain RCRA waste will be trucked to the

ORR for treatment at a permitted wastewater treatment facility. There are no further ARARs for this scenario.

Disposal of dredged materials

Disposal of dredged materials may be regulated by the RCRA transportation (40 CFR 262 and 40 CFR 263) and land disposal restriction (LDR) requirements (40 CFR 268), if the contaminated sediment is found to contain RCRA-characteristic waste as well as low-level waste. In addition, the U.S. Department of Transportation (DOT) Regulations for Transportation of Hazardous Materials and TDEC Radiation Protection Standards may apply to transportation of sediments containing radioactive materials above certain limits.

DOE is exempt from the licensing requirements of the TDEC Radiation Protection Standards for activities occurring within plant boundaries [TDEC 1200-2-10-.06(1)]. However, the TDEC requirements for packaging and transportation of radionuclide-containing materials above exempt quantities [as listed in Schedule RHS 8-3, TDEC 1200-2-10-.04(3)] will be applicable for transport of any sediments containing radioactivity from the LWBR OU to the ORR for treatment or disposal [TDEC 1200-2-10-.30 *et seq.*]; compliance with the DOT regulations is made by reference.

The DOT regulations list general requirements for shipping and packaging at 49 CFR 172 and 173 and requirements for carriage by public highway at 49 CFR 177. Specific loading and unloading requirements for transportation of radioactive materials by public highway are in 49 CFR 177.842. DOE Order 5480.3, ``Safety Requirements for the Packaging and Transportation of Hazardous Materials, Hazardous Substances, and Hazardous Wastes,'' also specifies packaging requirements.

Containerized sediments (depending on characterization) may be stored at either a permitted low-level waste facility at the ORR or a mixed waste storage facility allowed under the EPA\DOE Mixed Waste FFCA.

Table 8.3. ARARs[a] and TBC[b] Guidance for Alternatives 1 Through 3

Actions	Requirements	Prerequisites	Federal or TDEC[c,d] citation
Alternative 1	CERCLA criteria	Protective of human health and the environment	40 CFR 300
Alternative 2			
Chemical-specific	None		
Location-specific	None		
Action-specific			
	CERCLA criteria	Protective of human health and the environment	40 CFR 300
Institutional controls	Active controls include fences, warning signs, and restrictions on land use	Long-term management of residual radioactive material - **TBC**	DOE Order 5400.5(IV)(6)(c)
	DOE must supply existing sediment data and acquire additional data in areas where permitting of sediment disturbance is under consideration	In support of permit application for sediment disturbance activities categorized as marginal or potential major sediment disturbance **TBC**	Interagency Agreement for Watts Bar Reservoir Permit Coordination
Alternative 3			
Chemical- or radionuclide-specific			
Protection of the general public from radiation exposure	No licensee shall possess, use, receive, or transfer sources of radiation in such a manner as to create radiation levels in excess of 2 mrem/hr or 100 mrem/7 days	Continuous exposure to individuals in unrestricted areas - **applicable for transport to ORR; relevant and appropriate for on-shore activities**	TDEC 1200-2-5-.07
	The general public must not receive an effective dose equivalent greater than 100 mrem/year	Dose received by the general public from all sources of radiation exposure and routine activities, including remedial action, at a DOE facility **TBC**	DOE Order 5400.5
	All releases of radioactive material must be "as low as reasonably achievable" (ALARA)	Releases of radioactive material from DOE activities **TBC**	DOE Order 5400.5

Table 8.3 (continued)

Actions	Requirements	Prerequisites	Federal or TDEC[c,d] citation
Location-specific			
Presence of endangered or threatened species or critical habitat of such species as designated in 50 CFR 17, 50 CFR 226, or 50 CFR 227	Actions that jeopardize species/habitat must be avoided or appropriate mitigation measures taken. Consultation with DOI[e], FWS[f], and/or state agencies, as appropriate, to ensure that proposed actions do not jeopardize the continued existence of the species or adversely modify or destroy critical habitat is recommended for on-site actions.	Action that is likely to jeopardize species or destroy or adversely modify critical habitat **applicable**	Endangered Species Act of 1973 (16 USC 1531 *et seq.*) 50 CFR 402 Fish and Wildlife Coordination Act (16 USC 661 *et seq.*)
Presence of endangered or threatened species or critical habitat (see above citation) of same within an aquatic ecosystem as defined in 40 CFR 230.3(c)	Dredge or fill material shall not be discharged into an aquatic ecosystem if it would jeopardize such species or would likely result in the destruction or adverse modification of a critical habitat of the species.	Action involving discharge of dredge or fill material into aquatic ecosystem **applicable**	Clean Water Act § 404 40 CFR 230.10(b)
Presence of Tennessee state-listed species ``in need of management''	No person may knowingly destroy the habitat of such species **applicable**	Action impacting such species	TWRC[g] Proc. 86-29 (1991)
Presence of archaeological resources on public land	Steps must be taken to preserve excavated archaeological resources; consultation with the DOI or state historic preservation officer is required	Action that would affect significant scientific, prehistorical, or archaeological resources **applicable**	Archaeological and Historic Preservation Act (16 USC 469a-1)
Action-specific			
	CERCLA criteria	Protective of human health and the environment	40 CFR 300
Dredging of contaminated sediment	Notification must be made to the Corps of Engineers (COE) District Engineer in order to proceed under a Nationwide Permit (NWP) Erosion and siltation controls must be used and maintained in effective operating condition No activity may substantially disrupt the movement of those species of aquatic life indigenous to the waterbody	NWP 38 authorizes activities required to effect the containment, stabilization, or removal of hazardous or toxic waste materials that are performed, ordered, or sponsored by a government agency with established regulatory or legal authority provided the agency notifies the COE District Engineer - **applicable**	33 CFR 330, Appendix A

Table 8.3 (continued)

Actions	Requirements	Prerequisites	Federal or TDEC[c,d] citation
	No activity may be authorized under a NWP if it is likely to jeopardize the continued existence of a threatened or endangered species or is likely to modify the critical habitat of such species		
Storage/disposal of dewatered sediments and dewatering/decontamination fluids	Permits the on-site storage of LDR[h]-mixed waste for purposes other than accumulation	On-site storage of LDR mixed wastes **TBC**	EPA, Region IV/DOE-OR LDR-FFCA[i], June 12, 1992
	Must meet waste acceptance criteria for LLW[j] disposal facility	On-site disposal of LLW **TBC**	DOE Order 5820.2A
Shipments of LLW or mixed waste	Must meet manifesting, packaging, labeling, marking, placarding, and recordkeeping requirements	Transportation of RCRA hazardous waste - **applicable**	40 CFR 262 40 CFR 263
	The waste must meet packaging, labeling, marking, placarding, manifest and pretransport requirements in accordance with Department of Transportation (DOT) regulations	Transportation of radioactive materials above exempt quantities **applicable**	49 *CFR* 172, 173, 175, 178, and 179; DOE Order 5480.3 (TBC)
	Exemption from the TDEC radiation protection regulations for radioactive material in individual quantities which do not exceed the applicable quantity set forth in Schedule RHS 8-3	Handling of radioactive materials above exempt quantities **applicable**	TDEC 1200-2-10-.04(3)
	Licensing requirements for shipment and delivery	Delivery of licensable quantities of radioactive material to a disposal facility within the state **applicable**	TDEC 1200-2-10-.32(4)
	Packaging requirements are based on the maximum activity of radioactive material in a package	Packaging of radioactive materials above exempt quantities for public transport **applicable**	49 *CFR* 173.431; 49 *CFR* 173.433; 49 *CFR* 173.435 49 *CFR* 173.411

[a]ARARs = applicable or relevant and appropriate requirements.
[b]TBC = to be considered; all entries on table are potential ARARs unless noted as TBC.
[c]TDEC = Tennessee Department of Environment and Conservation.
[d]Unless otherwise noted.
[e]DOI = Department of the Interior.
[f]FWS = Fish and Wildlife Service.
[g]TWRC = Tennessee Wildlife Resources Commission.
[h]LDR = land disposal restrictions.
[i]FFCA = Federal Facility Compliance Agreement.
[j]LLW = low level waste.

9. DETAILED ANALYSIS OF ALTERNATIVES AND THEIR ENVIRONMENTAL CONSEQUENCES

9.1 CRITERIA FOR ANALYSIS

During the detailed analysis, each alternative is assessed against specific criteria and the results of this assessment arrayed such that comparisons between alternatives can be made. This approach is designed to provide decision makers with sufficient information to adequately compare the alternatives, select an appropriate remedy for the site, and demonstrate satisfaction of the statutory requirements. Nine CERCLA required evaluation criteria, to address statutory requirements as well as additional technical and policy considerations, are important for selecting the preferred remedial alternative. The evaluation criteria are

- overall protection of human health and the environment;
- compliance with ARARs;
- long-term effectiveness and permanence;
- short-term effectiveness;
- reduction of toxicity, mobility, or volume;
- implementability;
- cost;
- state acceptance; and
- community acceptance.

The detailed analysis will include an evaluation of each alternative against the nine criteria. The first two criteria are threshold criteria that must be met before an alternative is selected. The next five criteria are grouped together as the primary evaluation criteria taking into account technical, cost, institutional, and risk concerns. The final two criteria are modifying criteria and will be evaluated following comment on the RI/FS report and the proposed plan and will be addressed once a final decision is being made and the ROD is being prepared. Each of the criteria is described below.

9.1.1 Overall Protection of Human Health and the Environment

The assessment against this criterion describes how the alternative, as a whole, achieves and maintains protection of human health and the environment. The assessment is based on a composite of factors assessed under other evaluation criteria and focuses on how an alternative achieves protection over time and how site risks are reduced. The analysis indicates how each source or potential pathway is to be eliminated, reduced, or controlled.

9.1.2 Compliance with ARARs

This assessment addresses whether and how each alternative complies with applicable or relevant and appropriate federal and state requirements, including chemical-specific, location-specific, and action-specific requirements. When an ARAR is not met, the basis for justifying one of the six waivers allowed under CERCLA will be discussed. This criterion also addresses other information from advisories, criteria, and guidance that the lead and support agencies have agreed is "to be considered."

9.1.3 Long-Term Effectiveness and Permanence

This assessment addresses the results of a remedial action in terms of risk remaining at the site after response objectives have been met. A primary focus of this evaluation is the effectiveness of the controls that will be applied to manage risk posed by treatment residuals and/or any untreated wastes that may be left on the site, as well as the volume and nature of that material. The type, degree, and adequacy of long-term management for untreated waste or residues remaining at the site and the long-term reliability of engineering or institutional actions are also considered part of this assessment.

9.1.4 Short-Term Effectiveness

This assessment addresses the impacts of the alternative during the construction and implementation phase until remedial response objectives have been met. Under this criterion, alternatives will be evaluated with respect to the potential effects on human health and the environment (including the community and workers) during implementation of the remedial action and the length of time until protection is achieved.

9.1.5 Reduction of Toxicity, Mobility, and Volume

This assessment addresses the anticipated performance or results of a remedial action in terms of the actual reduction (e.g., percent, order of magnitude) in toxicity, mobility, and volume of contaminants achieved through treatment. Considered in this assessment are the treatment processes, the amount of materials that will be treated, the expected degree of reduction, the degree to which the treatment will be irreversible, and the type and quantity of residuals that will remain following treatment.

9.1.6 Implementability

Implementability refers to the technical, administrative, and environmental feasibility of implementing an alternative and the availability of various materials and services required during its implementation. Technical feasibility may include difficulties associated with the construction or operation of the technology or the ease of monitoring the effectiveness of the action. Administrative feasibility includes the ability and time required to obtain approvals or permits from other agencies or government bodies. Availability of services and materials may include consideration of the availability of needed treatment, storage, and disposal services and the availability of specialty equipment or trained operators.

9.1.7 Cost

The estimates prepared for this FS include capital and operation and maintenance costs. Capital costs include both direct costs, such as equipment, labor, and materials, and indirect costs, such as engineering services. A present worth analysis is done on any alternatives that continue for extended years following implementation to provide a common basis for comparing the cost of the alternatives. The cost estimates are intended to be "order of magnitude" estimates and are for comparison purposes only. The actual technologies chosen during the design phase may cost more or less than those used for estimating purposes. Details and backup for the cost estimates are provided in Appendix F.

9.1.8 State Acceptance

Formal comments made during the review of the Proposed Plan, relative to technical and administrative issues and concerns, will be evaluated, and a response to comments will be submitted to TDEC. The ROD will summarize the state's comments and concerns and how those concerns were addressed.

9.1.9 Community Acceptance

Community acceptance summarizes any input from the community received on the Proposed Plan. When this input is available, features of an alternative that the community supports, opposes, or has reservations about are noted. A summary of the community comments and the responses to those comments is included in the ROD.

9.2 DETAILED ANALYSIS

Provided below is a summary of the assessment of each alternative against the first seven of the nine criteria.

9.2.1 Alternative 1: No Action

Existing controls and programs would be assumed to lose effect and unrestricted use of LWBR could occur.

9.2.1.1 Overall protection of human health and the environment

Analysis of existing data reveals no unacceptable risk to human health or the environment from sediments or surface water in LWBR OU under the existing conditions; however, the risk assessment demonstrates that unrestricted use of the reservoir would cause substantial risk to human health. This alternative does not ensure that existing controls would remain in place. This alternative is somewhat protective in that the natural sedimentation continues to cover the existing contamination while radioactive decay helps to lessen the risk over time. This alternative would allow DOE to monitor for potential threats to human health and the environment. This alternative would not meet the remediation goals.

9.2.1.2 Compliance with ARARs

This alternative would not comply with the CERCLA requirement that a remedial action attain long-term protection of human health and the environment.

9.2.1.3 Long-term effectiveness and permanence

This alternative would be minimally effective at reducing the risk over the long term. The residual risk at the site would be the same as the risk levels provided in Chapter 5. Natural sedimentation and radioactive decay will gradually reduce the risk levels. This alternative is not at all effective in eliminating pathways or maintaining existing controls, and, therefore, the risk scenarios in Chapter 5 could occur. There would be no long-term management of the contamination remaining in the LWBR OU.

9.2.1.4 Short-term effectiveness

There would be no positive short-term impacts associated with this alternative. Negative short-term impacts could exist if restrictions on dredging or fish consumption advisories were removed.

9.2.1.5 Reduction of toxicity, mobility, and volume

Natural sedimentation will reduce the mobility of the contaminants by covering them with more clean sediment and isolating the contaminants further from scouring actions, leaching, or other disturbance and from contact with benthic organisms. Also associated with the natural sedimentation is further shielding from the radiation being emitted by the ^{137}Cs. Natural radioactive decay will reduce the volume and carcinogenicity of the ^{137}Cs. Some biodegradation of organic contaminants will be continuously taking place. Since this alternative involves no remedial actions, it would be reversible, and further treatment could be rendered.

9.2.1.6 Implementability

Assuming that this alternative is accepted by the public and regulators, there would be no implementation activities at all.

9.2.1.7 Cost

No cost would be associated with this alternative, other than for monitoring.

9.2.2 Alternative 2: Institutional Controls and Advisories

This alternative includes the continuance of all existing laws and regulations providing protection to the public, the IAG for permitting of sediment disturbing activities, fish consumption advisories, deed restrictions placed on TVA owned lands beneath the reservoir, and the two common components.

9.2.2.1 Overall protection of human health and the environment

Analysis of existing data reveals no unacceptable risk to human health or the environment from sediments, fish consumption, or surface water in LWBR OU under the conditions that this alternative will maintain. This alternative ensures that existing controls over two risk pathways would remain in place. This alternative is protective in that disturbance of sediments would be regulated now and in the future and the natural sedimentation would continue to cover the existing contamination while radioactive decay lessens the risk over time. This alternative would allow DOE to monitor for any increase in contaminant levels in the future and is protective in that DOE could respond to any increases in the overall system or respond to areas of higher concentrations should such areas be found. This alternative does little to reduce the risk to the environment. This alternative meets human health remediation goals but would not provide for protection of benthic organisms, fish, or piscivorous wildlife.

9.2.2.2 Compliance with ARARs

This alternative would comply with all ARARs.

9.2.2.3 Long-term effectiveness and permanence

The risk at the site following implementation of this alternative would remain at the same level as it currently is and would be in the acceptable range. This alternative would allow natural processes to reduce the risk over time, while the monitoring program would enable DOE to guard against any increase in risk levels. By implementing this alternative, DOE would ensure that the dredging permit program remains in place and is effectively implemented. The deed restrictions offer a very long-term protection against the contaminated sediments. It is expected that TDEC will maintain the fish consumption advisories for as long as necessary. The fish consumption advisories are under revision and will be protective to 10^{-5} for sensitive populations and 10^{-4} for typical populations. The controls offered by this alternative are both adequate and reliable and have been in place for several years.

9.2.2.4 Short-term effectiveness

No short-term decrease or increase in risk to the public or the environment would occur due to the implementation of this alternative. The benefits of this alternative are expected to be long-term in nature, not short-term.

9.2.2.5 Reduction of toxicity, mobility, and volume

Natural sedimentation will reduce the mobility of the contaminants by covering them with more clean sediment and isolating the contaminants further from scouring actions, leaching, or other disturbance and from contact with benthic organisms. Also associated with the natural sedimentation is further shielding from the radiation being emitted by the ^{137}Cs. Natural radioactive decay will reduce the volume and toxicity of the ^{137}Cs. Some biodegradation of organic contaminants will continuously be taking place. The monitoring plan will ensure that the toxicity, mobility, and volume of the contaminants do not increase during the future without DOE's knowledge and appropriate action. The institutional controls will not reduce the toxicity, mobility, or volume of the contaminants, but will help to eliminate pathways of exposure to the contaminants. This alternative involves no direct treatment of the sediments and is reversible; further treatment could be rendered.

9.2.2.6 Implementability

The dredging permit program and fish consumption advisories are already implemented. The fish consumption advisory program is being revised, is protective, and most likely would need no revisions. Deed restrictions would require a one-time effort of locating and revising the deeds and would then require no effort to remain effective. The monitoring plan could be easily implemented since the types and quantities of samples have been collected in years past without significant problems.

9.2.2.7 Cost

The estimated present worth cost for the dredging permit program, the deed restrictions and the fish consumption advisories is $1,141,000. This cost was produced assuming a 30-year lifespan with ongoing annual activities. This cost estimate was prepared assuming the Interagency Working Group met once a month to review dredging permits and other sediment disturbing activities and that some additional time would be required each year from DOE personnel to acquire additional data. Another assumption used was that there are 50 fish consumption advisory

signs surrounding LWBR OU and they would be replaced at the rate of 10/year due to vandalism or other reasons. Note that the cost of the dredging permit program is already ongoing and is spread among the IAG parties. The cost for the current fish consumption advisory program is already ongoing and is part of the TDEC budget. Not all of the costs shown in this estimate are DOE expenditures. The cost for the monitoring plan will be determined following preparation of the plan.

9.2.3 Alternative 3: Sediment Removal

This alternative includes dredging a large portion of the main channel, dewatering the sediment, treating the water, containerizing and storing the dewatered sediments, the controls outlined in Alternative 2, and the two common components.

9.2.3.1 Overall protection of human health and the environment

This alternative would remove those areas of sediment containing the majority of the contaminants, thus lowering the human health risk back into the acceptable range. The sediments of LWBR are contaminated with inorganics that are not entirely attributable to DOE operations, and there is the possibility that recontamination could occur in the future. The mercury and ^{137}Cs are not likely to recontaminate the sediments following removal, however some lower levels may be left in place just outside the main channel area. The containerizing of the sediments would help to eliminate exposure to the public, but would expose numerous workers to the contamination for more than 150 days. The dredging (despite using controls and methods designed to prevent resuspension) would likely spread some amount of the contamination into the water and onto the downstream sediment surface. The removal of these areas may help to protect surrounding biota, but would likely kill any organisms found inside the dredged zone. Any contaminants resuspended or spread would be more likely to enter the food chain of the benthic and aquatic organisms than contaminants left in place. The containerized sediments would be isolated from exposure to humans or the environment until such time as it could be treated or disposed of. This alternative would leave a majority of the LWBR OU untouched and the residual contamination would be monitored. Habitat for benthic organisms would be destroyed at the dredge site, and, depending on the amount of sediment left in place, it could take many years to return to a useful state. This alternative meets human health remediation goals but does not provide for protection of benthic organisms, fish, or piscivorous wildlife.

9.2.3.2 Compliance with ARARs

There would be many administrative requirements that would have to be addressed before implementation of this alternative could begin. It would be possible to comply with the ARARs for this alternative given enough time and resources.

9.2.3.3 Long-term effectiveness and permanence

This alternative is designed to remove portions of contaminated sediment with contaminant levels above a certain concentration. The concentration levels that would require action would be set based on risk to human health and the environment and could be set at any level chosen by the concerned parties. Given enough resources and time, this method could feasibly reach any desired human health risk level. The more sediment that needed to be removed, the more chance of sending a plume of contaminants downstream due to a mishap in the dredging activities. This alternative is permanent and once material is dredged off the bottom of LWBR OU, DOE would

be committed to managing that material for as long as it has characteristics that cause it to be a risk to human health and the environment. This alternative will permanently remove a large portion of the contaminants of DOE origin, such as mercury and ^{137}Cs. Other inorganic contaminants may be present in the TVA systems upstream of LWBR and could recontaminate LWBR in the future. The pesticides and PCBs in fish tissue would probably not be reduced by the sediment removal alternative. The sediment could be effectively managed once it was dewatered and containerized. There is a chance that no dredging technologies are capable of meeting the performance requirements necessary to limit resuspension of sediments. The containers may need to be replaced on a periodic basis, depending on the final moisture content of the sediments and the length of time that storage is required. If the sediments are found to be a mixed RCRA/low-level waste, then additional storage requirements and cost would be incurred.

9.2.3.4 Short-term effectiveness

This alternative would require over 20 years to dredge the majority of the main channel and containerize the dewatered sediments. Boating/barge traffic might need to be stopped, depending on the locations where dredging was occurring. Workers would be exposed to the sediments for the entire duration of the remedial action. Controls and monitoring are easily implemented to limit worker exposures, although ALARA would be more closely adhered to by not dredging the sediments. Very little risk to the community would occur during the treatment of the sediments at the shoreline facility. The moisture content of the sediments will prevent dust generation or airborne threats, and the capture of all dewatering effluent will help prevent spillage or residuals from remaining after dewatering and containerization is accomplished. A monitoring sweep of the shoreline facility after the equipment has been removed would be a simple method for determining the nature and extent of any trace spillage that may occur. There is a chance that contaminants would be resuspended and allowed to drift downstream during the action. Resuspended sediments would be diluted in concentration, but would be more accessible to biota and human contact. Organisms within the area to be dredged would be likely to die during the dredging or dewatering activities. Depending on where the dredging occurs, some private landowners might be affected or disturbed by the dredging and dewatering activities. Compared to natural processes, this alternative would quickly eliminate a majority of the contamination in the sediments.

9.2.3.5 Reduction of toxicity, mobility, and volume

This alternative would isolate the contaminated sediments (above a certain concentration) in containers in a storage area that would be secure and managed. The mobility of the sediments that reached the containers would be near 0%. However, the mobility of those contaminants that were resuspended during the remedial activity or were near the outer boundaries of the dredged area would be increased significantly from their current state. The volume of the radioactive contaminants would be reduced through natural radioactive decay with or without dredging and containerization. The dewatering of the sediments would reduce the volume of the sediment by removing some of the interstitial water associated with the sediments in their natural state. Portions of this action are irreversible. Sediments dredged, dewatered, and containerized could not be replaced into the LWBR OU in their natural state, and the areas dredged would not be returned to a natural state. The dredged sediments are still treatable if at some point it becomes desirable and technically feasible to remove them from the containers and perform a treatment action.

9.2.3.6 Implementability

This alternative would be difficult to implement. During the remedial action, the technical problem would be controlling the resuspension of sediments and the real-time monitoring of the dredging operations resuspension rate. If the resuspension rates are too high with one type of dredge, then other types may need to be tried with additional expense and time. There is a high probability that some technical/equipment problems will occur during the first full scale implementation of this alternative, although the disruption may only be short-term. Any future remedial actions could be easily implemented on the residuals or on the containerized waste following this action. Administrative problems could include obtaining landowner permission if the shoreline facility could not be located anywhere but on private land or if a temporary road needed to be constructed for hauling the water and sediments to the treatment and storage facilities. TVA, COE, TDEC, and EPA would need to be involved in the planning of this action, and in approving the monitoring and health and safety plans. The ability of the technology to limit resuspension may need to be proven before remedial action could take place. If barge or boat traffic needed to be suspended for any duration, then more approvals and coordination would be required prior to beginning the remedial action. The pneumatic dredge could be difficult to locate and procure, although other technologies may be explored if they claim to be effective at limiting resuspension of sediments. Waste storage capacity on the ORR is also becoming more and more limited and harder to obtain with each passing year.

9.2.3.7 Cost

To prepare a screening level cost estimate for this alternative, several assumptions had to be made. Other bases of the estimate and details are provided in Appendix F. The capital cost of this alternative (based on these specific assumptions) is $44 million. Operations and maintenance activities were not costed and would likely be minimal. The potential use of more than the two dredging technologies, the replacement of storage containers periodically, and the remediation of additional sediment volumes were not included in this cost estimate. This cost estimate also does not include any cost for treatment of sediments to reduce or remove contaminants or for stabilization/solidification. If transport of sediments to somewhere other than the ORR is necessary, then cost would increase significantly.

9.3 COMPARATIVE ANALYSIS

Table 9.1 provides a qualitative comparison of each alternative against the seven criteria and against each other. Also provided is a short written section for each of the seven criteria that provides a judgement as to which of the three alternatives is best, second best, and third best.

9.3.1 Overall Protection of Human Health and the Environment

The risk assessment in the RI indicates that LWBR sediment presents no risk to human health as long as it remains undisturbed. Dredging activities in Alternative 3 could resuspend the contaminants, destroy habitat, expose workers, and negatively affect the environment. Alternative 1 does not reduce future risk levels and would not contain the current controls and restrictions against dredging the sediment from contaminated areas. Alternative 2 is the most protective of human health and the environment.

9.3.2 Compliance with ARARs

Alternatives 2 and 3 would comply with ARARs under the current conditions. Alternative 1 does not comply because it does not prevent dredging in contaminated areas and does not reduce the consumption of potentially contaminated fish. Meeting the additional ARARs required for the dredging portion of Alternative 3 would require considerably more time and money than for either of the other two alternatives.

9.3.3 Long-term Effectiveness and Permanence

Alternative 2 has the best long-term effectiveness. Health risks following implementation of Alternative 2 would not exceed current levels, and future risks would be diminished by natural processes (radioactive decay and chemical degradation for sediment; and decrease of contaminant concentrations in fish as source areas on land are removed). Current institutional controls on dredging and fish consumption would remain in place and additional controls could be implemented if human exposure to the contaminants changed. Alternative 1 is not effective in the long term because institutional controls to prevent dredging or other sediment-disturbing activities would not exist. Because Alternative 3 could create risks during implementation that do not now exits, it is considered less effective than Alternative 2 over the long term. In the limited area remediated, Alternative 3 would produce a long-term reduction in the risk to human health.

9.3.4 Short-term Effectiveness

Alternative 2 is the most protective of the workers and community during the short term. Alternative 1 provides no specific protection, particularly against consumption of contaminated fish. Removal and handling of contaminated sediment associated with Alternative 3 could pose short-term risk to the workers, the community, and the environment. Alternative 2 permits only limited exposure to those contaminants found in the least contaminated species of fish. The restrictions on sediment-disturbing activities and the fish consumption advisories associated with Alternative 2 are effective in the short term.

9.3.5 Reduction of Toxicity, Mobility, and Volume

Although natural processes would reduce contaminant toxicity, mobility, and quantity under all three alternatives, only Alternative 3 actively treats the waste to accomplish this. Alternatives 1 and 2 do not affect toxicity levels. Alternative 2 allows for further treatment or changes at some later date. The monitoring plan that will accompany any of the alternatives will allow DOE to determine any changes in toxicity, mobility, or quantity of the contaminants.

9.3.6 Implementability

Alternative 1 is the easiest to implement as no action is required. Alternative 3 would require numerous permits, the concurrence of several agencies, and probably an environmental assessment before remedial action could begin. In the case of Alternative 2, two of the actions are already implemented and would be continued as part of this alternative. Deed restrictions would require approval from TVA. The monitoring plan would be easily implemented since similar activities have been going on for many years at ORR.

9.3.7 Cost

Alternative 1 is the least expensive. Alternative 2 is the least expensive action alternative. Alternative 2 would require approximately $1,141,000 over a two-year period for the deed restrictions, fish consumption advisories, and IAG permit coordination. To dredge about 2 acres of the sediment, as required under Alternative 3, the cost is estimated to be $37,000,000. These cost figures are order-of-magnitude estimates developed for comparison purposes only. Actual costs may be significantly lower or higher at the time the effort is required. Not all of the cost attributed to Alternative 2 would be borne by DOE. Some of the ongoing activities are already in annual budgets for other affected agencies.

Table 9.1. Detailed Analysis Summary

Assessment Criteria	Alternative 1: No Action	Alternative 2: Institutional Controls	Alternative 3: Sediment Removal	Qualitative Judgement
Overall Protection of Human Health and the Environment	None	High	Moderate	Alt. 2 is most protective Goals are achieved without short-term risk or destruction of habitat
Compliance with ARARs				
1. Chemical-specific	None	None	Easy to comply	Alternatives 2 and 3 are equal
2. Location-specific	None	None	Easy to comply	Alternatives 2 and 3 are equal
3. Action-specific	Does not comply because no action is unprotective	Easy to comply	Can achieve compliance	Alt. 3 most difficult to achieve compliance
Long-term Effectiveness and Permanence				
1. Magnitude of Residual Risks	Residual risks are same as current and would worsen as pathways are opened	Residual risks are same as current and will decrease with time	On-site residual is lessened, potential off-site liability created	No need to create off-site liability exist; Alt 2 is adequate
2. Adequacy of Controls	No controls are provided	Existing controls will be maintained and conditions monitored	Would adequately control an area of unacceptable risk	Alt. 2 and 3 are adequate for current known conditions
3. Reliability of Controls	unreliable	reliable	reliable if storage area is maintained	Alt. 2 and 3 would reliably protect against future risks
Short-term Effectiveness				
1. Worker Protection	Low risk	Low risk	Moderate risk	Alt. 2 provides for worker safety

Table 9.1 (continued)

Assessment Criteria	Alternative 1: No Action	Alternative 2: Institutional Controls	Alternative 3: Sediment Removal	Qualitative Judgement
2. Community Protection	future risk would mirror the risk assessment and would be high	Low current and future risk	Low risk unless significant resuspension occurs	Alt. 2 is most protective
3. Environmental Protection	future risk would worsen as controls are lifted	Low current and future risk	Moderate risk	Alt. 2 is most protective
4. Estimated Time until Remedial Action Objectives are Achieved	Would not achieve all objectives	Continual process over several decades	Dependent upon size and number of areas to be dredged; As costed, over 600 days are necessary	Alt. 2 and 3 could both achieve objectives within a few years of implementation
Reduction of Toxicity, Mobility, and Volume				
1. Toxicity	None besides natural processes	None besides natural processes	None besides natural processes (unless treatment is necessary for disposal)	No alternative would actively reduce toxicity
2. Mobility	None besides natural processes	None besides natural processes	Yes, through containerization	Alt. 3 could be most effective
3. Volume	None besides natural processes	None besides natural processes	Yes, through dewatering of sediments	Alt. 3 could be most effective
Implementability				
1. Technical	Very high	High	Feasible but difficult	Alt. 1 and 2 are easiest to implement
2. Administrative	Very high	High	Feasible but potentially very difficult	Alt. 1 and 2 are easiest to implement

Table 9.1 (continued)

Assessment Criteria	Alternative 1: No Action	Alternative 2: Institutional Controls	Alternative 3: Sediment Removal	Qualitative Judgement
3. Availability of Materials	None needed	Readily available	Available but may require long procurement cycle or test phases	Alt. 1 and 2 are easiest to implement
Costs				
1. Capital	Monitoring	$3,275,000	$38 billion	Alt. 1 is least expensive; Alt. 2 is least expensive action alternative Alt. 3 is prohibitive
2. Present Worth	To be determined	$1,141,000 + monitoring $ when available	$16.5 billion + treatment and off-site disposal if needed	

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Appendix A

DATA BASES, EQUATIONS, AND TOXICITY AND EXPOSURE VALUES USED IN THE HUMAN HEALTH RISK CHARACTERIZATION

Table A.1 Database for detectable contaminants that were evaluated in the human health risk characterization

Source	Reach[a]	Media	Matrix	Type[b]	Cas_no	Contaminant	Ratio of detects to total samples	Minimum	Maximum	Ordinary mean	UCL ordinary mean	Units
Core	5	Sediment	AVERAGE	I	7440-36-0	Antimony	1/3	4.0E-01	4.8E-01	4.5E-01	5.2E-01	mg/kg
					7440-38-2	Arsenic	4/4	7.5E+00	9.1E+00	8.6E+00	9.4E+00	mg/kg
					7440-41-7	Beryllium	4/4	1.0E+00	1.4E+00	1.2E+00	1.5E+00	mg/kg
					7440-43-9	Cadmium	4/4	1.1E+00	5.2E+00	2.3E+00	4.6E+00	mg/kg
					18540-29-9	Chromium	6/6	2.8E+01	4.5E+01	3.6E+01	4.1E+01	mg/kg
					7440-50-8	Copper	4/4	2.9E+01	3.9E+01	3.5E+01	4.0E+01	mg/kg
					7439-92-1	Lead	4/4	3.9E+01	4.5E+01	4.3E+01	4.6E+01	mg/kg
					7439-97-6	Mercury	6/6	1.0E+00	3.0E+00	2.0E+00	2.5E+00	mg/kg
					7440-02-0	Nickel	4/4	1.9E+01	3.0E+01	2.3E+01	2.8E+01	mg/kg
					7440-22-4	Silver	3/4	1.6E-01	7.3E-01	3.5E-01	6.6E-01	mg/kg
					7440-61-1	Uranium	2/2	5.7E+00	6.0E+00	5.8E+00	6.7E+00	mg/kg
					7440-66-6	Zinc	4/4	2.4E+02	2.7E+02	2.5E+02	2.7E+02	mg/kg
				O	117-81-7	Bis(2-ethylhexyl)phthalate	2/4	8.5E-01	2.4E+00	1.7E+00	2.6E+00	mg/kg
				R	14596-10-2	Am-241	2/2	1.4E+00	3.8E+00	2.6E+00	1.0E+01	Bq/kg
					13981-15-2	Cm-243,244	2/2	5.6E-01	6.7E-01	6.1E-01	9.6E-01	Bq/kg
					15621-76-8	Cm-245,246	2/2	1.3E+00	1.8E+00	1.5E+00	3.2E+00	Bq/kg
					15758-33-5	Cm-248	2/2	8.5E-01	1.3E+00	1.1E+00	2.4E+00	Bq/kg
					10198-40-0	Co-60	56/60	4.5E-02	1.2E+01	2.9E+00	3.5E+00	Bq/kg
					10045-97-3	Cs-137	60/60	2.5E+00	5.1E+02	2.4E+02	2.7E+02	Bq/kg
					13981-16-3	Pu-238	2/2	8.6E-01	1.2E+00	1.0E+00	1.9E+00	Bq/kg
					15117-48-3	Pu-239,240	2/2	3.7E-01	3.6E+00	2.0E+00	1.2E+01	Bq/kg
					10098-97-2	Sr-90	2/2	3.6E+00	5.3E+00	4.4E+00	9.8E+00	Bq/kg
					13966-29-5	U-234	2/2	7.1E+01	7.1E+01	7.1E+01	7.1E+01	Bq/kg
					15117-96-1	U-235	2/2	1.7E+00	3.3E+00	2.5E+00	7.5E+00	Bq/kg
					7440-61-1	U-238	2/2	5.9E+01	6.3E+01	6.1E+01	7.3E+01	Bq/kg
Fish	5	Fish	Bluegill	I	7440-38-2	Arsenic	18/20	5.0E-02	1.0E-01	6.7E-02	7.2E-02	mg/kg
					7440-50-8	Copper	6/20	4.8E-01	1.7E+02	9.3E+00	2.4E+01	mg/kg
					7439-97-6	Mercury	20/20	3.2E-02	1.3E-01	5.5E-02	6.5E-02	mg/kg
					7440-66-6	Zinc	20/20	5.2E+00	7.1E+01	1.0E+01	1.6E+01	mg/kg
				R	10045-97-3	Cs-137	3/4	9.1E-01	4.2E+00	1.9E+00	3.8E+00	Bq/kg
			Catfish	I	7440-38-2	Arsenic	1/2	2.0E-02	3.0E-02	2.5E-02	5.7E-02	mg/kg
					7440-50-8	Copper	2/2	8.0E-01	1.7E+00	1.2E+00	4.1E+00	mg/kg
					7439-97-6	Mercury	1/2	1.0E-01	1.8E-01	1.4E-01	3.9E-01	mg/kg
					7782-49-2	Selenium	2/2	2.0E-01	2.4E-01	2.2E-01	3.5E-01	mg/kg
					7440-66-6	Zinc	2/2	6.0E+00	8.0E+00	7.0E+00	1.3E+01	mg/kg

Table A.1 (continued)

Source	Reach[a]	Media	Matrix	Type[b]	Cas_no	Contaminant	Ratio of detects to total samples	Minimum	Maximum	Ordinary mean	UCL ordinary mean	Units
Fish	5	Fish	Catfish	O	72-54-8	4,4'-DDD	12/39	3.4E-04	1.9E-02	7.9E-03	9.1E-03	mg/kg
					72-55-9	4,4'-DDE	25/40	4.8E-03	2.0E-01	3.1E-02	4.1E-02	mg/kg
					50-29-3	4,4'-DDT	8/40	4.2E-04	4.0E-02	1.1E-02	1.3E-02	mg/kg
					309-00-2	Aldrin	2/40	1.3E-04	1.0E-01	1.2E-02	1.6E-02	mg/kg
					11097-69-1	Aroclor-1254	102/129	1.0E-02	3.3E+00	4.7E-01	5.4E-01	mg/kg
					11096-82-5	Aroclor-1260	121/131	3.0E-02	4.2E+00	8.7E-01	9.7E-01	mg/kg
					57-74-9	Chlordane	31/110	5.3E-03	4.1E-01	1.2E-01	1.3E-01	mg/kg
					60-57-1	Dieldrin	3/22	1.0E-02	6.0E-02	1.4E-02	1.9E-02	mg/kg
					959-98-8	Endosulfan I	3/22	1.0E-02	3.0E-02	1.2E-02	1.5E-02	mg/kg
					72-20-8	Endrin	1/22	1.0E-02	3.0E-02	1.1E-02	1.2E-02	mg/kg
					58-89-9	Gamma-BHC (Lindane)	1/22	1.0E-02	2.0E-02	1.0E-02	1.1E-02	mg/kg
				R	10045-97-3	Cs-137	11/19	1.0E+00	5.9E+00	3.7E+00	4.3E+00	Bq/kg
					10098-97-2	Sr-90	19/19	-3.7E-02	3.7E-01	7.6E-02	1.1E-01	Bq/kg
			LargeMouth Bass	I	7440-38-2	Arsenic	5/5	9.0E-02	1.7E-01	1.3E-01	1.6E-01	mg/kg
					7440-41-7	Beryllium	2/5	3.0E-03	2.7E-02	1.1E-02	2.1E-02	mg/kg
					7439-97-6	Mercury	5/5	1.4E-01	2.5E-01	1.9E-01	2.4E-01	mg/kg
					7440-66-6	Zinc	4/4	3.1E+00	4.2E+00	3.8E+00	4.3E+00	mg/kg
				O	11097-69-1	Aroclor-1254	20/21	3.0E-02	3.0E-01	1.3E-01	1.6E-01	mg/kg
					11096-82-5	Aroclor-1260	19/21	6.0E-02	4.0E-01	1.4E-01	1.7E-01	mg/kg
					57-74-9	Chlordane	5/20	6.1E-02	9.0E-02	7.4E-02	7.6E-02	mg/kg
				R	10045-97-3	Cs-137	5/5	5.3E-01	9.0E-01	7.6E-01	9.1E-01	Bq/kg
			Striped/Hybrid Bass	O	72-55-9	4,4'-DDE	4/10	1.0E-02	1.9E-01	4.0E-02	7.3E-02	mg/kg
					50-29-3	4,4'-DDT	2/10	1.0E-02	2.0E-02	1.2E-02	1.4E-02	mg/kg
					11097-69-1	Aroclor-1254	61/66	1.0E-01	2.3E+00	4.8E-01	5.7E-01	mg/kg
					11096-82-5	Aroclor-1260	61/66	1.0E-01	3.1E+00	6.2E-01	7.4E-01	mg/kg
					57-74-9	Chlordane	14/45	6.0E-02	4.4E-01	1.2E-01	1.3E-01	mg/kg
					60-57-1	Dieldrin	1/10	1.0E-02	1.0E-02	1.0E-02	1.0E-02	mg/kg
					959-98-8	Endosulfan I	1/10	1.0E-02	1.0E-02	1.0E-02	1.0E-02	mg/kg
					72-20-8	Endrin	3/10	1.0E-02	9.0E-02	2.1E-02	3.6E-02	mg/kg

Table A.1 (continued)

Source	Reach[a]	Media	Matrix	Type[b]	Cas_no	Contaminant	Ratio of detects to total samples	Minimum	Maximum	Ordinary mean	UCL ordinary mean	Units
Near-shore	5	Sediment	GRAB	R	10198-40-0	Co-60	113/352	6.4E-02	6.9E+00	8.4E-01	9.1E-01	Bq/kg
					10045-97-3	Cs-137	349/352	5.0E-01	3.5E+02	1.6E+01	1.9E+01	Bq/kg
ORREM	4	Water	TOTAL	I	7429-90-5	Aluminum	6/6	9.3E-02	1.3E+00	4.5E-01	8.2E-01	mg/L
					7440-39-3	Barium	5/6	1.0E-03	4.4E-02	2.9E-02	4.1E-02	mg/L
					16984-48-8	Fluoride	1/6	1.0E-01	1.0E-01	1.0E-01	1.0E-01	mg/L
					7439-95-4	Magnesium	6/6	8.0E+00	9.3E+00	8.7E+00	9.0E+00	mg/L
					7439-96-5	Manganese	6/6	3.1E-03	1.3E-01	5.2E-02	8.8E-02	mg/L
					7440-61-1	Uranium	4/5	1.0E-04	1.0E-03	3.9E-04	7.3E-04	mg/L
					7440-62-2	Vanadium	1/6	2.0E-03	2.0E-03	2.0E-03	2.0E-03	mg/L
					7440-66-6	Zinc	1/6	5.0E-03	7.9E-03	5.5E-03	6.5E-03	mg/L
				R	10198-40-0	Co-60	6/6	-2.5E-01	1.1E-01	-2.2E-02	1.0E-01	Bq/L
					10045-97-3	Cs-137	6/6	-9.0E-02	1.5E-01	3.3E-02	1.1E-01	Bq/L
					10028-17-8	H-3	6/6	-2.1E+01	2.0E+01	-6.7E-01	1.2E+01	Bq/L
					14133-76-7	Tc-99	6/6	-2.0E-02	2.2E-01	8.5E-02	1.6E-01	Bq/L
					7440-61-1	Uranium, total	6/6	1.3E-02	2.0E-01	5.2E-02	1.1E-01	Bq/L
Phase 1	5	Water	TOTAL	I	7439-95-4	Magnesium	4/4	4.5E+00	9.6E+00	5.9E+00	8.8E+00	mg/L
				R	10098-97-2	Sr-90	3/3	-1.5E-02	2.6E-02	-1.2E-18	3.8E-02	Bq/L
TVA	5	Sediment	GRAB	I	7429-90-5	Aluminum	11/11	3.7E+02	9.1E+04	3.1E+04	4.3E+04	mg/kg
					7440-36-0	Antimony	8/11	1.0E-01	5.8E-01	3.1E-01	4.1E-01	mg/kg
					7440-38-2	Arsenic	11/11	1.1E+00	2.3E+01	8.4E+00	1.2E+01	mg/kg
					7440-39-3	Barium	11/11	5.7E+01	2.1E+02	8.0E+01	1.0E+02	mg/kg
					7440-41-7	Beryllium	2/11	1.0E-01	1.7E+00	2.7E-01	5.4E-01	mg/kg
					7440-42-8	Boron	8/11	5.0E+00	1.1E+02	2.7E+01	4.3E+01	mg/kg
					18540-29-9	Chromium	11/11	1.1E+01	7.7E+01	3.3E+01	4.3E+01	mg/kg
					7440-50-8	Copper	6/11	1.0E+00	2.6E+01	6.5E+00	1.1E+01	mg/kg
					57-12-5	Cyanide	5/11	2.0E-02	7.2E-01	1.4E-01	2.6E-01	mg/kg
					7439-92-1	Lead	11/11	1.8E+01	1.2E+02	3.5E+01	5.1E+01	mg/kg
					7439-96-5	Manganese	11/11	1.9E+02	2.5E+03	8.4E+02	1.2E+03	mg/kg
					7439-97-6	Mercury	3/11	1.0E-01	1.5E-01	1.1E-01	1.2E-01	mg/kg
					7440-02-0	Nickel	11/11	8.0E+00	5.1E+01	2.4E+01	3.1E+01	mg/kg
					7782-49-2	Selenium	4/11	5.0E-02	2.0E-01	7.9E-02	1.1E-01	mg/kg
					7440-62-2	Vanadium	11/11	1.7E+01	6.9E+01	4.0E+01	5.0E+01	mg/kg
					7440-66-6	Zinc	11/11	3.2E+01	1.3E+02	5.3E+01	6.8E+01	mg/kg

Table A.1 (continued)

Source	Reach[a]	Media	Matrix	Type[b]	Cas_no	Contaminant	Ratio of detects to total samples	Minimum	Maximum	Ordinary mean	UCL ordinary mean	Units
TVA	5	Sediment	GRAB	O	100-02-7	4-Nitrophenol	1/11	1.9E+00	2.5E+00	2.0E+00	2.1E+00	mg/kg
					117-81-7	Bis(2-ethylhexyl)phthalate	2/11	3.1E-01	6.2E-01	3.9E-01	4.4E-01	mg/kg
					53-70-3	Dibenz(a,h)anthracene	1/11	6.2E-01	8.3E-01	6.7E-01	7.1E-01	mg/kg
					75-09-2	Methylene chloride	1/29	1.1E-02	5.2E-02	1.4E-02	1.7E-02	mg/kg
				R	10045-97-3	Cs-137	6/11	3.4E-01	7.8E+00	1.5E+00	2.7E+00	Bq/kg

[a]Reaches are locations identified in Clinch River remedial investigation.
[b]Types include inorganics, organics, and radionuclides.

Table A.2 List of analytes in LWBR media where all measurements were below detection limits showing the number of samples that were analyzed

		Fish Tissue			
Type	Analyte	Bluegill	Catfish	Largemouth Bass	Striped or Hybrid Bass
Inorganics	Antimony	20	2	5	
	Beryllium	20	2		
	Cadmium	20	2	5	
	Chromium	20	2	5	
	Copper			5	
	Lead		2	5	
	Nickel		2	5	
	Selenium	20		5	
	Silver	20		5	
	Thallium	11	2	5	
Organics	2,4,5-Trichlorophenol		9		
	2,4,6-Trichlorophenol		9		
	2,4-Dichlorophenol		9		
	2,4-Dimethylphenol		9		
	2,4-Dinitrophenol		9		
	2-Chlorophenol		9		
	2-Methylnaphthalene		3		
	2-Methylphenol		9		
	2-Nitrophenol		9		
	4,4'-DDD				10
	4,6-Dinitro-2-methylphenol		8		
	4-Chloro-3-methylphenol		9		
	4-Methylphenol		9		
	4-Nitrophenol		9		
	Aldrin				10
	Alpha-BHC		22		10
	Aroclor-1016		112	21	66
	Aroclor-1221		112	21	66
	Aroclor-1232		112	21	66
	Aroclor-1242		112	21	66
	Aroclor-1248		112	21	66
	Beta-BHC		22		10
	Delta-BHC		22		10
	Endosulfan II		22		10
	Endosulfan sulfate		22		10
	Gamma-BHC (Lindane)				10
	Heptachlor		22		10
	Heptachlor epoxide		22		10
	Mirex		22		10
	Naphthalene		1		
	PCBs, total				
	Pentachlorophenol		9		
	Phenol		9		
	Toxaphene		22		10
Radionuclides	Co-60	4	19	5	

Table A.2 (continued)

Surface Water			
Type	Analyte	Phase 1 (Reach 5)	ORREM (Reach 4)
Inorganics	Antimony	4	4
	Arsenic	4	7
	Beryllium	4	4
	Cadmium	4	7
	Chromium	4	7
	Cobalt		4
	Copper	4	
	Cyanide		6
	Lead	4	7
	Mercury	4	7
	Nickel	4	7
	Selenium	4	4
	Silver	4	4
	Thallium	4	3
	Zinc	4	
Organics	1,2,4-Trichlorobenzene	4	
	1,2-Dichlorobenzene	4	
	1,3-Dichlorobenzene	4	
	1,4-Dichlorobenzene	4	
	2,4,5-Trichlorophenol	4	
	2,4,6-Trichlorophenol	4	
	2,4-Dichlorophenol	4	
	2,4-Dimethylphenol	4	
	2,4-Dinitrophenol	4	
	2,4-Dinitrotoluene	4	
	2,6-Dinitrotoluene	4	
	2-Chloronaphthalene	4	
	2-Chlorophenol	4	
	2-Methylnaphthalene	4	
	2-Methylphenol	4	
	2-Nitroaniline	4	
	2-Nitrophenol	4	
	3,3'-Dichlorobenzidine	4	
	3-Nitroaniline	4	
	4,4'-DDD	3	
	4,4'-DDE	3	
	4,4'-DDT	3	
	4,6-Dinitro-2-methylphenol	4	
	4-Bromophenyl-phenylether	4	

Table A.2 (continued)

Type	Analyte	Phase 1 (Reach 5)	ORREM (Reach 4)
	Surface Water		
Organics	4-Chloro-3-methylphenol	4	
	4-Chloroaniline	4	
	4-Chlorophenyl-phenylether	4	
	4-Methylphenol	4	
	4-Nitroaniline	4	
	4-Nitrophenol	4	
	Acenaphthene	4	
	Acenaphthylene	4	
	Aldrin	3	
	Alpha-BHC	3	
	Anthracene	4	
	Aroclor-1016	3	
	Aroclor-1221	3	
	Aroclor-1232	3	
	Aroclor-1242	3	
	Aroclor-1248	3	
	Aroclor-1254	3	
	Aroclor-1260	3	
	Benzo(a)anthracene	4	
	Benzo(a)pyrene	4	
	Benzo(b)fluoranthene	4	
	Benzo(g,h,i)perylene	4	
	Benzo(k)fluoranthene	4	
	Benzoic acid	4	
	Benzyl alcohol	4	
	Beta-BHC	3	
	Bis(2-chloroethoxy)methane	4	
	Bis(2-chloroethyl)ether	4	
	Bis(2-chloroisopropyl)ether	4	
	Bis(2-ethylhexyl)phthalate	4	
	Butylbenzylphthalate	4	
	Chlordane	3	
	Chrysene	4	
	Delta-BHC	3	
	Di-n-butylphthalate	4	
	Di-n-octylphthalate	4	
	Dibenz(a,h)anthracene	4	
	Dibenzofuran	4	
	Dieldrin	3	
	Diethylphthalate	4	
	Dimethylphthalate	4	
	Endosulfan I	3	

Table A.2 (continued)

Surface Water			
Type	Analyte	Phase 1 (Reach 5)	ORREM (Reach 4)
Organics	Endosulfan II	3	
	Endosulfan sulfate	3	
	Endrin	3	
	Endrin ketone	3	
	Fluoranthene	4	
	Fluoride	4	
	Gamma-BHC (Lindane)	3	
	Heptachlor	3	
	Heptachlor epoxide	3	
	Hexachlorobenzene	4	
	Hexachlorobutadiene	4	
	Hexachlorocyclopentadiene	4	
	Hexachloroethane	4	
	Indeno(1,2,3-cd)pyrene	4	
	Isophorone	4	
	Methoxychlor	3	
	N-Nitroso-di-n-propylamine	4	
	N-Nitrosodiphenylamine	4	
	Naphthalene	4	
	Nitrobenzene	4	
	Pentachlorophenol	4	
	Phenanthrene	4	
	Phenol	4	
	Pyrene	4	
	Toxaphene	3	

Table A.2 (continued)

Type	Analyte	TVA Rec and Public Use Area Surface Sediment	Phase 1 Near-Shore Surface Sediment	Sediment Cores
Inorganics	Cadmium	11		3
	Selenium			
	Silver	11		4
	Thallium	11		
	Uranium	11		
Organics	1,1,1-Trichloroethane	29		
	1,1,2,2-Tetrachloroethane	29		
	1,1,2-Trichloroethane	29		
	1,1-Dichloroethane	29		
	1,1-Dichloroethylene	29		4
	1,2,4-Trichlorobenzene	11		4
	1,2-Dichlorobenzene	11		
	1,2-Dichloroethane	29		
	1,2-Dichloropropane	29		4
	1,3-Dichlorobenzene	11		4
	1,4-Dichlorobenzene	11		4
	2,4,5-Trichlorophenol			4
	2,4,6-Trichlorophenol	11		4
	2,4-Dichlorophenol	11		4
	2,4-Dimethylphenol	11		4
	2,4-Dinitrophenol	11		4
	2,4-Dinitrotoluene	11		4
	2,6-Dinitrotoluene	11		
	2-Chloroethyl vinyl ether	29		4
	2-Chloronaphthalene	11		4
	2-Chlorophenol	11		4
	2-Methylnaphthalene			3
	2-Methylphenol			4
	2-Nitroaniline			4
	2-Nitrophenol	11		4
	3,3'-Dichlorobenzidine	11		4
	3-Nitroaniline			4
	4,4'-DDD	11		4
	4,4'-DDE	11		4
	4,4'-DDT	11		4
	4,6-Dinitro-2-methylphenol	11		4
	4-Bromophenyl-phenylether	11		4
	4-Chloro-3-methylphenol	11		4
	4-Chloroaniline			4
	4-Chlorophenyl-phenylether	11		4
	4-Methylphenol			4
	4-Nitroaniline			

Table A.2 (continued)

Type	Analyte	TVA Rec and Public Use Area Surface Sediment	Phase 1 Near-Shore Surface Sediment	Sediment Cores
Organics	4-Nitrophenol			4
	Acenaphthene	11		4
	Acenaphthylene	11		4
	Acrolein	29		
	Acrylonitrile	29		
	Aldrin	11		3
	Alpha-BHC	11		3
	Anthracene	11		4
	Aroclor-1016	11		5
	Aroclor-1221	11		5
	Aroclor-1232	11		5
	Aroclor-1242	11		5
	Aroclor-1248	11		5
	Aroclor-1254	11		6
	Aroclor-1260	11		6
	Benzene	29		
	Benzidine	11		
	Benzo(a)anthracene	11		4
	Benzo(a)pyrene	11		4
	Benzo(b)fluoranthene	11		4
	Benzo(g,h,i)perylene	11		4
	Benzo(k)fluoranthene	11		4
	Benzoic acid			4
	Benzyl alcohol			3
	Beta-BHC	11		3
	Bis(2-chloroethoxy)methane	11		4
	Bis(2-chloroethyl)ether	11		4
	Bis(2-chloroisopropyl)ether	11		4
	Bromoform	29		
	Butylbenzylphthalate	11		4
	Carbon tetrachloride	29		
	Chlordane	11		3
	Chlorobenzene	29		
	Chlorodibromomethane	29		
	Chloroethane	29		
	Chloroform	29		
	Chloromethane	29		
	Chrysene	11		4
	Cis-1,3-Dichloropropene	29		
	Delta-BHC	11		3
	Di-n-butylphthalate	11		4
	Di-n-octylphthalate	11		4
	Dibenz(a,h)anthracene			4

Table A.2 (continued)

Type	Analyte	TVA Rec and Public Use Area Surface Sediment	Phase 1 Near-Shore Surface Sediment	Sediment Cores
Organics	Dibenzofuran			4
	Dichlorobromomethane	29		
	Dieldrin	11		4
	Diethylphthalate	11		4
	Dimethylphthalate	11		4
	Endosulfan I	11		3
	Endosulfan II	11		4
	Endosulfan sulfate	11		4
	Endrin	11		4
	Endrin aldehyde	11		
	Endrin ketone			4
	Ethylbenzene	29		
	Fluoranthene	11		4
	Fluoride	11		4
	Gamma-BHC (Lindane)	11		3
	Heptachlor	11		3
	Heptachlor epoxide	11		3
	Hexachlorobenzene	11		4
	Hexachlorobutadiene	11		4
	Hexachlorocyclopentadiene	11		4
	Hexachloroethane	11		4
	Indeno(1,2,3-cd)pyrene	11		4
	Isophorone	11		4
	Methoxychlor	7		4
	Methyl bromide	29		
	N-Nitroso-di-n-propylamine	11		4
	N-Nitrosodiphenylamine	11		4
	Naphthalene	11		4
	Nitrobenzene	11		4
	Pentachlorophenol	11		4
	Phenanthrene	11		4
	Phenol	11		4
	Pyrene	11		4
	Tetrachloroethylene	29		
	Toluene	29		
	Toxaphene	11		4
	Trans-1,2-dichloroethylene	29		
	Trans-1,3-dichloropropene	29		
	Trichloroethylene	29		
	Trichlorofluoromethane	29		
	Vinyl chloride	29		
	Xylene	15		

Table A.2 (continued)

Type	Analyte	TVA Rec and Public Use Area Surface Sediment	Phase 1 Near-Shore Surface Sediment	Sediment Cores
Radionuclides	Am-241	11	13	
	Cm-242	11		
	Cm-243,244	11		
	Eu-152		83	
	Eu-154		16	
	Pu-238	11		
	Pu-239,240	11		
	Pu-241	11		
	Pu-242	11		
	Sr-89	11		
	Sr-90	11		
	U-234	11		
	U-235	11		
	U-238	11		

Table A.3 Slope (cancer potency) factors, reference doses (RfD), and permeability constants for inorganic and organic analytes

Contaminant type	Contaminant	Cas_no	Oral RfD (mg/kg)[a]	Inhalation RfC (mg/m3)[a]	Oral slope factor 1/(mg/kg/d)[a]	Inhalation slope factor 1/(mg/kg/d)[a]	Permeability constant (m/hr)
Inorganics	Antimony	7440-36-0	4.00E-04				1.00E-05
	Arsenic	7440-38-2	3.00E-04			5.00E+01	1.00E-05
	Barium	7440-39-3	7.00E-02	5.00E-04			1.00E-05
	Beryllium	7440-41-7	5.00E-03		4.30E+00	8.40E+00	1.00E-05
	Boron	7440-42-8	9.00E-02	2.00E-02			1.00E-05
	Cadmium	7440-43-9	5.00E-04			6.10E+00	1.00E-05
	Chromium	18540-29-9	5.00E-03			4.10E+01	1.00E-05
	Cyanide	57-12-5	2.00E-02				8.80E-06
	Fluoride	16984-48-8	6.00E-02				
	Manganese	7439-96-5	5.00E-03	4.00E-04			1.00E-05
	Mercury	7439-97-6	3.00E-04	3.00E-04			1.00E-05
	Nickel	7440-02-0	2.00E-02				
	Selenium	7782-49-2	5.00E-03				1.00E-05
	Silver	7440-22-4	5.00E-03				1.00E-05
	Uranium	7440-61-1	3.00E-03				
	Vanadium	7440-62-2	7.00E-03				1.00E-05
	Zinc	7440-66-6	3.00E-01				1.00E-05
Organics	4,4'-DDD	72-54-8			2.40E-01		2.80E-03
	4,4'-DDE	72-55-9			3.40E-01		2.40E-03
	4,4'-DDT	50-29-3	5.00E-04		3.40E-01	3.40E-01	4.30E-03
	Aldrin	309-00-2	3.00E-05		1.70E+01	1.70E+01	1.60E-05
	Aroclor-1254	11097-69-1			7.70E+00		
	Aroclor-1260	11096-82-5			7.70E+00		
	Bis(2-ethylhexyl)phthalate	117-81-7	2.00E-02		1.40E-02		1.70E-02
	Chlordane	57-74-9	6.00E-05		1.30E+00	1.30E+00	5.20E-04
	Dieldrin	60-57-1	5.00E-05		1.60E+01	1.60E+01	1.60E-04
	Endosulfan I	959-98-8	5.00E-05				
	Endrin	72-20-8	3.00E-04				1.60E-04
	Gamma-BHC (Lindane)	58-89-9	3.00E-04		1.30E+00		1.40E-04
	Methylene chloride	75-09-2	6.00E-02	3.00E+00	7.50E-03		4.50E-05

[a]ORNL. 1993. Toxicity values for use in hazardous waste risk assessment and remediation. ES/ER/TM-76. Biomedical and Environmental Information Analysis Section. Health Sciences Research Division, Oak Ridge, TN.

Appendix B

RESULTS OF THE HUMAN HEALTH RISK CHARACTERIZATION

Table B.1 Human health risk characterization for carcinogens measured in LWBR water during Phase 1 sampling where at least one measurement was above detection limits

Water Ingestion Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/L	L	Bq/Life	Risk/Bq	
DIRECT	Adult	Water	TOTAL	Ingestion	R	Sr-90	3/3	2.59E-02 (Max)	2.1E+04	5.4E+02	9.7E-10	5.3E-07
						TYPE TOTAL						5.3E-07
						MATRIX TOTAL						5.3E-07

Table B.2 Human health risk characterization for carcinogens measured in water sampled near Brashear Island by ORR monitoring program where at least one measurement was above detection limits

Water Ingestion Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/L	L	Bq/Life	Risk/Bq	
DIRECT	Adult	Water	TOTAL	Ingestion	R	Uranium, total	6/6	1.12E-01 (UOM)	2.1E+04	2.3E+03	7.6E-10[a]	1.8E-06
						Cs-137	6/6	1.10E-01 (UOM)	2.1E+04	2.3E+03	7.6E-10	1.7E-06
						Co-60	6/6	1.01E-01 (UOM)	2.1E+04	2.1E+03	4.1E-10	8.6E-07
						H-3	6/6	1.19E+01 (UOM)	2.1E+04	2.5E+05	1.5E-12	3.6E-07
						Tc-99	6/6	1.65E-01 (UOM)	2.1E+04	3.5E+03	1.4E-12	4.7E-09
						TYPE TOTAL						4.7E-06
						MATRIX TOTAL						4.7E-06

[a]Slope factor for U-238 used for total uranium.

Table B.3 Human health risk characterization for noncarcinogens measured in water sampled near Brashear Island by ORR monitoring program where at least one measurement was above detection limits

Water Ingestion Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
DIRECT	Adult	Water	TOTAL	Ingestion	I	Manganese	6/6	8.80E-02 (UOM)	2.7E-02	2.4E-03	5.0E-03	4.8E-01
						Fluoride	1/6	1.00E-01 (Max)	2.7E-02	2.7E-03	6.0E-02	4.6E-02
						Barium	5/6	4.08E-02 (UOM)	2.7E-02	1.1E-03	7.0E-02	1.6E-02
						Vanadium	1/6	2.00E-03 (Max)	2.7E-02	5.5E-05	7.0E-03	7.8E-03
						Uranium	4/5	7.30E-04 (UOM)	2.7E-02	2.0E-05	3.0E-03	6.7E-03
						Zinc	1/6	6.46E-03 (UOM)	2.7E-02	1.8E-04	3.0E-01	5.9E-04
						TYPE TOTAL						5.6E-01
						MATRIX TOTAL						5.6E-01
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
	Child	Water	TOTAL	Ingestion	I	Manganese	6/6	8.80E-02 (UOM)	6.4E-02	5.6E-03	5.0E-03	1.1E+00
						Fluoride	1/6	1.00E-01 (Max)	6.4E-02	6.4E-03	6.0E-02	1.1E-01
						Barium	5/6	4.08E-02 (UOM)	6.4E-02	2.6E-03	7.0E-02	3.7E-02
						Vanadium	1/6	2.00E-03 (Max)	6.4E-02	1.3E-04	7.0E-03	1.8E-02
						Uranium	4/5	7.30E-04 (UOM)	6.4E-02	4.7E-05	3.0E-03	1.6E-02
						Zinc	1/6	6.46E-03 (UOM)	6.4E-02	4.1E-04	3.0E-01	1.4E-03
						TYPE TOTAL						1.3E+00
						MATRIX TOTAL						1.3E+00

Table B.4 Human health risk characterization for carcinogens measured in water sampled near Brashear Island by ORR monitoring program where at least one measurement was above detection limits

Swimming Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/L	L	Bq/Life	Risk/Bq	
SWIM	Adult	Water	TOTAL	Ingestion	R	Uranium, total	6/6	1.12E-01 (UOM)	4.7E+02	5.2E+01	7.6E-10*	3.9E-08
						Cs-137	6/6	1.10E-01 (UOM)	4.7E+02	5.1E+01	7.6E-10	3.9E-08
						Co-60	6/6	1.01E-01 (UOM)	4.7E+02	4.7E+01	4.1E-10	1.9E-08
						H-3	6/6	1.19E+01 (UOM)	4.7E+02	5.6E+03	1.5E-12	8.1E-09
						Tc-99	6/6	1.65E-01 (UOM)	4.7E+02	7.7E+01	1.4E-12	1.0E-10
						TYPE TOTAL						1.1E-07
						MATRIX TOTAL						1.1E-07

*Slope factor for U-238 used for total uranium.

Table B.5 Human health risk characterization for noncarcinogens measured in water sampled near Brashear Island by ORR monitoring program where at least one measurement was above detection limits

Swimming Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
SWIM	Adult	Water	TOTAL	Dermal	I	Manganese	6/6	8.80E-02 (UOM)	2.8E-04	2.4E-05	5.0E-03	4.9E-03
						Barium	5/6	4.08E-02 (UOM)	2.8E-04	1.1E-05	7.0E-02	1.6E-04
						Vanadium	1/6	2.00E-03 (Max)	2.8E-04	5.6E-07	7.0E-03	8.0E-05
						Zinc	1/6	6.46E-03 (UOM)	2.8E-04	1.8E-06	3.0E-01	6.0E-06
						TYPE TOTAL						5.1E-03
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
				Ingestion	I	Manganese	6/6	8.80E-02 (UOM)	6.1E-04	5.4E-05	5.0E-03	1.1E-02
						Fluoride	1/6	1.00E-01 (Max)	6.1E-04	6.1E-05	6.0E-02	1.0E-03
						Barium	5/6	4.08E-02 (UOM)	6.1E-04	2.5E-05	7.0E-02	3.6E-04
						Vanadium	1/6	2.00E-03 (Max)	6.1E-04	1.2E-06	7.0E-03	1.7E-04
						Uranium	4/5	7.30E-04 (UOM)	6.1E-04	4.5E-07	3.0E-03	1.5E-04
						Zinc	1/6	6.46E-03 (UOM)	6.1E-04	3.9E-06	3.0E-01	1.3E-05
						TYPE TOTAL						1.2E-02
						MATRIX TOTAL						1.8E-02
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
	Child	Water	TOTAL	Dermal	I	Manganese	6/6	8.80E-02 (UOM)	5.2E-04	4.6E-05	5.0E-03	9.2E-03
						Barium	5/6	4.08E-02 (UOM)	5.2E-04	2.1E-05	7.0E-02	3.0E-04
						Vanadium	1/6	2.00E-03 (Max)	5.2E-04	1.0E-06	7.0E-03	1.5E-04
						Zinc	1/6	6.46E-03 (UOM)	5.2E-04	3.4E-06	3.0E-01	1.1E-05
						TYPE TOTAL						9.7E-03
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
				Ingestion	I	Manganese	6/6	8.80E-02 (UOM)	2.8E-03	2.5E-04	5.0E-03	5.0E-02
						Fluoride	1/6	1.00E-01 (Max)	2.8E-03	2.8E-04	6.0E-02	4.7E-03
						Barium	5/6	4.08E-02 (UOM)	2.8E-03	1.2E-04	7.0E-02	1.7E-03
						Vanadium	1/6	2.00E-03 (Max)	2.8E-03	5.7E-06	7.0E-03	8.1E-04
						Uranium	4/5	7.30E-04 (UOM)	2.8E-03	2.1E-06	3.0E-03	6.9E-04
						Zinc	1/6	6.46E-03 (UOM)	2.8E-03	1.8E-05	3.0E-01	6.1E-05
						TYPE TOTAL						5.8E-02
						MATRIX TOTAL						6.8E-02

Table B.6 Human health risk characterization for carcinogens measured in LWBR water during Phase 1 sampling where at least one measurement was above detection limits

Swimming Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/L	L	Bq/Life	Risk/Bq	
SWIM	Adult	Water	TOTAL	Ingestion	R	Sr-90	3/3	2.59E-02 (Max)	4.7E+02	1.2E+01	9.7E-10	1.2E-08
						TYPE TOTAL						1.2E-08
						MATRIX TOTAL						1.2E-08

Table B.7 Human health risk characterization for carcinogens measured in near-shore sediments collected during Phase-1 sampling where at least one measurement was above detection limits

Shoreline-use Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/kg	Dose/life	Bq/Life		
DIRECT	Adult	Sediment	GRAB	External	R	Cs-137	349/352	1.90E+01 (UOM)	4.4E+00	8.2E+01	5.4E-08	4.4E-06
						Co-60	113/352	9.07E-01 (UOM)	1.6E+00	1.4E+00	2.3E-07	3.3E-07
						TYPE TOTAL						4.8E-06
								Bq/kg	kg	Bq/Life	Risk/Bq	
				Ingestion	R	Cs-137	349/352	1.90E+01 (UOM)	1.1E+00	2.0E+01	7.6E-10	1.5E-08
						Co-60	113/352	9.07E-01 (UOM)	1.1E+00	9.5E-01	4.1E-10	3.8E-10
						TYPE TOTAL						1.5E-08
								Bq/m3	m3	Bq/Life	Risk/Bq	
				Inhalation	R	Cs-137	349/352	3.04E-07 (UOM)	1.8E+05	5.3E-02	5.1E-10	2.7E-11
						Co-60	113/352	1.46E-08 (UOM)	1.8E+05	2.5E-03	4.1E-09	1.0E-11
						TYPE TOTAL						3.8E-11
						MATRIX TOTAL						4.8E-06

Table B.8 Human health risk characterization for carcinogens measured in sediment in beach and public use areas by TVA where at least one measurement was above detection limits

Shoreline-use Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
DIRECT	Adult	Sediment	GRAB	Dermal	I	Beryllium	2/11	5.36E-01 (UOM)	1.7E-08	9.0E-09	4.3E+00	3.9E-08
						TYPE TOTAL						3.9E-08
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					O	Bis(2-ethylhexyl)phthalate	2/11	4.45E-01 (UOM)	1.7E-07	7.4E-08	1.4E-02	1.0E-09
						Methylene chloride	1/29	1.67E-02 (UOM)	1.7E-07	2.8E-09	7.5E-03	2.1E-11
						TYPE TOTAL						1.1E-09
								Bq/kg	Dose/life	Bq/Life		
				External	R	Cs-137	6/11	2.70E+00 (UOM)	4.4E+00	1.2E+01	5.4E-08	6.3E-07
						TYPE TOTAL						6.3E-07
								Bq/kg	kg	Bq/Life	Risk/Bq	
				Ingestion	R	Cs-137	6/11	2.70E+00 (UOM)	1.1E+00	2.8E+00	7.6E-10	2.1E-09
						TYPE TOTAL						2.1E-09
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					I	Beryllium	2/11	5.36E-01 (UOM)	5.9E-07	3.1E-07	4.3E+00	1.4E-06
						TYPE TOTAL						1.4E-06
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					O	Bis(2-ethylhexyl)phthalate	2/11	4.45E-01 (UOM)	5.9E-07	2.6E-07	1.4E-02	3.7E-09
						Methylene chloride	1/29	1.67E-02 (UOM)	5.9E-07	9.8E-09	7.5E-03	7.4E-11
						TYPE TOTAL						3.7E-09

Table B.8 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/m3	m3	Bq/Life	Risk/Bq	
DIRECT	Adult	Sediment	GRAB	Inhalation	R	Cs-137	6/11	4.31E-08 (UOM)	1.8E+05	7.6E-03	5.1E-10	3.9E-12
						TYPE TOTAL						3.9E-12
								mg/m3	m3/kg-d	mg/kg/d	1/(mg/kg/d)	
					I	Chromium	11/11	6.86E-07 (UOM)	9.9E-02	6.8E-08	4.1E+01	2.8E-06
						Arsenic	11/11	1.90E-07 (UOM)	9.9E-02	1.9E-08	5.0E+01	9.4E-07
						Beryllium	2/11	8.58E-09 (UOM)	9.9E-02	8.5E-10	8.4E+00	7.1E-09
						TYPE TOTAL						3.7E-06
						MATRIX TOTAL						5.8E-06

Table B.9 Human health risk characterization for noncarcinogens measured in sediment in beach and public use areas by TVA where at least one measurement was above detection limits

Shoreline-use Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DIRECT	Adult	Sediment	GRAB	Dermal	I	Arsenic	11/11	1.19E+01 (UOM)	3.9E-08	4.6E-07	3.0E-04	1.5E-03
						Manganese	11/11	1.25E+03 (UOM)	3.9E-08	4.9E-05	1.4E-01	3.5E-04
						Chromium	11/11	4.29E+01 (UOM)	3.9E-08	1.7E-06	5.0E-03	3.3E-04
						Vanadium	11/11	5.00E+01 (UOM)	3.9E-08	2.0E-06	7.0E-03	2.8E-04
						Nickel	11/11	3.07E+01 (UOM)	3.9E-08	1.2E-06	2.0E-02	6.0E-05
						Barium	11/11	1.04E+02 (UOM)	3.9E-08	4.0E-06	7.0E-02	5.8E-05
						Antimony	8/11	4.09E-01 (UOM)	3.9E-08	1.6E-08	4.0E-04	4.0E-05
						Boron	8/11	4.30E+01 (UOM)	3.9E-08	1.7E-06	9.0E-02	1.9E-05
						Mercury	3/11	1.20E-01 (UOM)	3.9E-08	4.7E-09	3.0E-04	1.6E-05
						Zinc	11/11	6.85E+01 (UOM)	3.9E-08	2.7E-06	3.0E-01	8.9E-06
						Beryllium	2/11	5.36E-01 (UOM)	3.9E-08	2.1E-08	5.0E-03	4.2E-06
						Selenium	4/11	1.06E-01 (UOM)	3.9E-08	4.1E-09	5.0E-03	8.3E-07
						Cyanide	5/11	2.64E-01 (UOM)	3.9E-08	1.0E-08	2.0E-02	5.1E-07
						TYPE TOTAL						2.7E-03
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/11	4.45E-01 (UOM)	3.9E-07	1.7E-07	2.0E-02	8.7E-06
						Methylene chloride	1/29	1.67E-02 (UOM)	3.9E-07	6.5E-09	6.0E-02	1.1E-07
						TYPE TOTAL						8.8E-06
								mg/kg	1/d	mg/kg/d	mg/kg/d	
				Ingestion	I	Arsenic	11/11	1.19E+01 (UOM)	1.4E-06	1.6E-05	3.0E-04	5.4E-02
						Manganese	11/11	1.25E+03 (UOM)	1.4E-06	1.7E-03	1.4E-01	1.2E-02
						Chromium	11/11	4.29E+01 (UOM)	1.4E-06	5.9E-05	5.0E-03	1.2E-02
						Vanadium	11/11	5.00E+01 (UOM)	1.4E-06	6.8E-05	7.0E-03	9.8E-03
						Nickel	11/11	3.07E+01 (UOM)	1.4E-06	4.2E-05	2.0E-02	2.1E-03
						Barium	11/11	1.04E+02 (UOM)	1.4E-06	1.4E-04	7.0E-02	2.0E-03
						Antimony	8/11	4.09E-01 (UOM)	1.4E-06	5.6E-07	4.0E-04	1.4E-03
						Boron	8/11	4.30E+01 (UOM)	1.4E-06	5.9E-05	9.0E-02	6.5E-04
						Mercury	3/11	1.20E-01 (UOM)	1.4E-06	1.6E-07	3.0E-04	5.5E-04
						Zinc	11/11	6.85E+01 (UOM)	1.4E-06	9.4E-05	3.0E-01	3.1E-04
						Beryllium	2/11	5.36E-01 (UOM)	1.4E-06	7.3E-07	5.0E-03	1.5E-04
						Selenium	4/11	1.06E-01 (UOM)	1.4E-06	1.4E-07	5.0E-03	2.9E-05
						Cyanide	5/11	2.64E-01 (UOM)	1.4E-06	3.6E-07	2.0E-02	1.8E-05
						TYPE TOTAL						9.5E-02

Table B.9 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DIRECT	Adult	Sediment	GRAB	Ingestion	O	Bis(2-ethylhexyl)phthalate	2/11	4.45E-01 (UOM)	1.4E-06	6.1E-07	2.0E-02	3.0E-05
						Methylene chloride	1/29	1.67E-02 (UOM)	1.4E-06	2.3E-08	6.0E-02	3.8E-07
						TYPE TOTAL						3.1E-05
								mg/m3	unitless	mg/m3	mg/m3	
				Inhalation	I	Manganese	11/11	1.99E-05 (UOM)	3.2E-01	6.4E-06	4.0E-04	1.6E-02
						Barium	11/11	1.66E-06 (UOM)	3.2E-01	5.3E-07	5.0E-04	1.1E-03
						Boron	8/11	6.87E-07 (UOM)	3.2E-01	2.2E-07	2.0E-02	1.1E-05
						Mercury	3/11	1.92E-09 (UOM)	3.2E-01	6.1E-10	3.0E-04	2.0E-06
						TYPE TOTAL						1.7E-02
								mg/m3	unitless	mg/m3	mg/m3	
					O	Methylene chloride	1/29	2.68E-10 (UOM)	3.2E-01	8.6E-11	3.0E+00	2.9E-11
						TYPE TOTAL						2.9E-11
						MATRIX TOTAL						1.2E-01
								mg/kg	1/d	mg/kg/d	mg/kg/d	
	Child	Sediment	GRAB	Dermal	I	Arsenic	11/11	1.19E+01 (UOM)	7.3E-08	8.7E-07	3.0E-04	2.9E-03
						Manganese	11/11	1.25E+03 (UOM)	7.3E-08	9.1E-05	1.4E-01	6.5E-04
						Chromium	11/11	4.29E+01 (UOM)	7.3E-08	3.1E-06	5.0E-03	6.3E-04
						Vanadium	11/11	5.00E+01 (UOM)	7.3E-08	3.7E-06	7.0E-03	5.2E-04
						Nickel	11/11	3.07E+01 (UOM)	7.3E-08	2.3E-06	2.0E-02	1.1E-04
						Barium	11/11	1.04E+02 (UOM)	7.3E-08	7.6E-06	7.0E-02	1.1E-04
						Antimony	8/11	4.09E-01 (UOM)	7.3E-08	3.0E-08	4.0E-04	7.5E-05
						Boron	8/11	4.30E+01 (UOM)	7.3E-08	3.2E-06	9.0E-02	3.5E-05
						Mercury	3/11	1.20E-01 (UOM)	7.3E-08	8.8E-09	3.0E-04	2.9E-05
						Zinc	11/11	6.85E+01 (UOM)	7.3E-08	5.0E-06	3.0E-01	1.7E-05
						Beryllium	2/11	5.36E-01 (UOM)	7.3E-08	3.9E-08	5.0E-03	7.9E-06
						Selenium	4/11	1.06E-01 (UOM)	7.3E-08	7.8E-09	5.0E-03	1.6E-06
						Cyanide	5/11	2.64E-01 (UOM)	7.3E-08	1.9E-08	2.0E-02	9.7E-07
						TYPE TOTAL						5.1E-03
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/11	4.45E-01 (UOM)	7.3E-07	3.3E-07	2.0E-02	1.6E-05
						Methylene chloride	1/29	1.67E-02 (UOM)	7.3E-07	1.2E-08	6.0E-02	2.0E-07
						TYPE TOTAL						1.7E-05

Table B.9 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DIRECT	Child	Sediment	GRAB	Ingestion	I	Arsenic	11/11	1.19E+01 (UOM)	1.3E-05	1.5E-04	3.0E-04	5.1E-01
						Manganese	11/11	1.25E+03 (UOM)	1.3E-05	1.6E-02	1.4E-01	1.1E-01
						Chromium	11/11	4.29E+01 (UOM)	1.3E-05	5.5E-04	5.0E-03	1.1E-01
						Vanadium	11/11	5.00E+01 (UOM)	1.3E-05	6.4E-04	7.0E-03	9.1E-02
						Nickel	11/11	3.07E+01 (UOM)	1.3E-05	3.9E-04	2.0E-02	2.0E-02
						Barium	11/11	1.04E+02 (UOM)	1.3E-05	1.3E-03	7.0E-02	1.9E-02
						Antimony	8/11	4.09E-01 (UOM)	1.3E-05	5.2E-06	4.0E-04	1.3E-02
						Boron	8/11	4.30E+01 (UOM)	1.3E-05	5.5E-04	9.0E-02	6.1E-03
						Mercury	3/11	1.20E-01 (UOM)	1.3E-05	1.5E-06	3.0E-04	5.1E-03
						Zinc	11/11	6.85E+01 (UOM)	1.3E-05	8.8E-04	3.0E-01	2.9E-03
						Beryllium	2/11	5.36E-01 (UOM)	1.3E-05	6.9E-06	5.0E-03	1.4E-03
						Selenium	4/11	1.06E-01 (UOM)	1.3E-05	1.4E-06	5.0E-03	2.7E-04
						Cyanide	5/11	2.64E-01 (UOM)	1.3E-05	3.4E-06	2.0E-02	1.7E-04
						TYPE TOTAL						8.9E-01
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/11	4.45E-01 (UOM)	1.3E-05	5.7E-06	2.0E-02	2.8E-04
						Methylene chloride	1/29	1.67E-02 (UOM)	1.3E-05	2.1E-07	6.0E-02	3.6E-06
						TYPE TOTAL						2.9E-04
								mg/m3	unitless	mg/m3	mg/m3	
				Inhalation	I	Manganese	11/11	1.99E-05 (UOM)	3.2E-01	6.4E-06	4.0E-04	1.6E-02
						Barium	11/11	1.66E-06 (UOM)	3.2E-01	5.3E-07	5.0E-04	1.1E-03
						Boron	8/11	6.87E-07 (UOM)	3.2E-01	2.2E-07	2.0E-02	1.1E-05
						Mercury	3/11	1.92E-09 (UOM)	3.2E-01	6.1E-10	3.0E-04	2.0E-06
						TYPE TOTAL						1.7E-02
								mg/m3	unitless	mg/m3	mg/m3	
					O	Methylene chloride	1/29	2.68E-10 (UOM)	3.2E-01	8.6E-11	3.0E+00	2.9E-11
						TYPE TOTAL						2.9E-11
						MATRIX TOTAL						9.1E-01

Table B.10 Human health risk characterization for carcinogens in fish where at least one measurement was above detection limits

Fish Consumption Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/kg	kg	Bq/Life	Risk/Bq	
DIRECT	Adult	Fish	Bluegill	Ingestion	R	Cs-137	3/4	3.76E+00 (UOM)	5.7E+02	2.1E+03	7.6E-10	1.6E-06
						TYPE TOTAL						1.6E-06
								Bq/kg	kg	Bq/Life	Risk/Bq	
			Catfish	Ingestion	R	Cs-137	11/19	4.33E+00 (UOM)	5.7E+02	2.5E+03	7.6E-10	1.9E-06
						Sr-90	19/19	1.11E-01 (UOM)	5.7E+02	6.3E+01	9.7E-10	6.1E-08
						TYPE TOTAL						1.9E-06
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					O	Aroclor-1260	121/131	9.75E-01 (UOM)	3.2E-04	3.1E-04	7.7E+00	2.4E-03
						Aroclor-1254	102/129	5.40E-01 (UOM)	3.2E-04	1.7E-04	7.7E+00	1.3E-03
						Dieldrin	3/22	1.86E-02 (UOM)	3.2E-04	5.9E-06	1.6E+01	9.4E-05
						Aldrin	2/40	1.57E-02 (UOM)	3.2E-04	5.0E-06	1.7E+01	8.4E-05
						Chlordane	31/110	1.34E-01 (UOM)	3.2E-04	4.2E-05	1.3E+00	5.5E-05
						Gamma-BHC (Lindane)	1/22	1.12E-02 (UOM)	3.2E-04	3.6E-06	1.3E+00	4.6E-06
						4,4'-DDE	25/40	4.05E-02 (UOM)	3.2E-04	1.3E-05	3.4E-01	4.4E-06
						4,4'-DDT	8/40	1.26E-02 (UOM)	3.2E-04	4.0E-06	3.4E-01	1.4E-06
						4,4'-DDD	12/39	9.09E-03 (UOM)	3.2E-04	2.9E-06	2.4E-01	6.9E-07
						TYPE TOTAL						3.9E-03
								Bq/kg	kg	Bq/Life	Risk/Bq	
			LargeMouth Bass	Ingestion	R	Cs-137	5/5	9.03E-01 (Max)	5.7E+02	5.1E+02	7.6E-10	3.9E-07
						TYPE TOTAL						3.9E-07
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					I	Beryllium	2/5	2.11E-02 (UOM)	3.2E-04	6.7E-06	4.3E+00	2.9E-05
						TYPE TOTAL						2.9E-05

Table B.10 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
DIRECT	Adult	Fish	LargeMouth Bass	Ingestion	O	Aroclor-1260	19/21	1.74E-01 (UOM)	3.2E-04	5.5E-05	7.7E+00	4.2E-04
						Aroclor-1254	20/21	1.57E-01 (UOM)	3.2E-04	5.0E-05	7.7E+00	3.8E-04
						Chlordane	5/20	7.64E-02 (UOM)	3.2E-04	2.4E-05	1.3E+00	3.1E-05
						TYPE TOTAL						8.4E-04
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
			Striped/Hybrid Bass	Ingestion	O	Aroclor-1260	61/66	7.36E-01 (UOM)	3.2E-04	2.3E-04	7.7E+00	1.8E-03
						Aroclor-1254	61/66	5.66E-01 (UOM)	3.2E-04	1.8E-04	7.7E+00	1.4E-03
						Chlordane	14/45	1.34E-01 (UOM)	3.2E-04	4.3E-05	1.3E+00	5.5E-05
						Dieldrin	1/10	1.00E-02 (Max)	3.2E-04	3.2E-06	1.6E+01	5.1E-05
						4,4'-DDE	4/10	7.26E-02 (UOM)	3.2E-04	2.3E-05	3.4E-01	7.8E-06
						4,4'-DDT	2/10	1.44E-02 (UOM)	3.2E-04	4.6E-06	3.4E-01	1.6E-06
						TYPE TOTAL						3.3E-03

Table B.11 Human health risk characterization for noncarcinogens in fish where at least one measurement was above detection limits

Fish Consumption Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DIRECT	Adult	Fish	Bluegill	Ingestion	I	Arsenic	18/20	7.17E-02 (UOM)	7.4E-04	5.3E-05	3.0E-04	1.8E-01
						Mercury	20/20	6.45E-02 (UOM)	7.4E-04	4.8E-05	3.0E-04	1.6E-01
						Zinc	20/20	1.61E+01 (UOM)	7.4E-04	1.2E-02	3.0E-01	4.0E-02
						TYPE TOTAL						3.8E-01
								mg/kg	1/d	mg/kg/d	mg/kg/d	
			Catfish	Ingestion	I	Mercury	1/2	1.80E-01 (Max)	7.4E-04	1.3E-04	3.0E-04	4.4E-01
						Arsenic	1/2	3.00E-02 (Max)	7.4E-04	2.2E-05	3.0E-04	7.4E-02
						Selenium	2/2	2.40E-01 (Max)	7.4E-04	1.8E-04	5.0E-03	3.6E-02
						Zinc	2/2	8.00E+00 (Max)	7.4E-04	5.9E-03	3.0E-01	2.0E-02
						TYPE TOTAL						5.7E-01
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Chlordane	31/110	1.34E-01 (UOM)	7.4E-04	9.9E-05	6.0E-05	1.6E+00
						Aldrin	2/40	1.57E-02 (UOM)	7.4E-04	1.2E-05	3.0E-05	3.9E-01
						Dieldrin	3/22	1.86E-02 (UOM)	7.4E-04	1.4E-05	5.0E-05	2.7E-01
						Endosulfan I	3/22	1.45E-02 (UOM)	7.4E-04	1.1E-05	5.0E-05	2.1E-01
						Endrin	1/22	1.25E-02 (UOM)	7.4E-04	9.2E-06	3.0E-04	3.1E-02
						Gamma-BHC (Lindane)	1/22	1.12E-02 (UOM)	7.4E-04	8.3E-06	3.0E-04	2.8E-02
						4,4'-DDT	8/40	1.26E-02 (UOM)	7.4E-04	9.4E-06	5.0E-04	1.9E-02
						TYPE TOTAL						2.6E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
			LargeMouth Bass	Ingestion	I	Mercury	5/5	2.37E-01 (UOM)	7.4E-04	1.8E-04	3.0E-04	5.8E-01
						Arsenic	5/5	1.65E-01 (UOM)	7.4E-04	1.2E-04	3.0E-04	4.1E-01
						Zinc	4/4	4.20E+00 (Max)	7.4E-04	3.1E-03	3.0E-01	1.0E-02
						Beryllium	2/5	2.11E-02 (UOM)	7.4E-04	1.6E-05	5.0E-03	3.1E-03
						TYPE TOTAL						1.0E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Chlordane	5/20	7.64E-02 (UOM)	7.4E-04	5.6E-05	6.0E-05	9.4E-01
						TYPE TOTAL						9.4E-01

Table B.11 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DIRECT	Adult	Fish	Striped/Hybrid Bass	Ingestion	O	Chlordane	14/45	1.34E-01 (UOM)	7.4E-04	9.9E-05	6.0E-05	1.7E+00
						Dieldrin	1/10	1.00E-02 (Max)	7.4E-04	7.4E-06	5.0E-05	1.5E-01
						Endosulfan I	1/10	1.00E-02 (Max)	7.4E-04	7.4E-06	5.0E-05	1.5E-01
						Endrin	3/10	3.56E-02 (UOM)	7.4E-04	2.6E-05	3.0E-04	8.8E-02
						4,4'-DDT	2/10	1.44E-02 (UOM)	7.4E-04	1.1E-05	5.0E-04	2.1E-02
						TYPE TOTAL						2.1E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
	Child	Fish	Bluegill	Ingestion	I	Arsenic	18/20	7.17E-02 (UOM)	1.9E-03	1.4E-04	3.0E-04	4.6E-01
						Mercury	20/20	6.45E-02 (UOM)	1.9E-03	1.2E-04	3.0E-04	4.1E-01
						Zinc	20/20	1.61E+01 (UOM)	1.9E-03	3.1E-02	3.0E-01	1.0E-01
						TYPE TOTAL						9.7E-01
								mg/kg	1/d	mg/kg/d	mg/kg/d	
			Catfish	Ingestion	I	Mercury	1/2	1.80E-01 (Max)	1.9E-03	3.5E-04	3.0E-04	1.2E+00
						Arsenic	1/2	3.00E-02 (Max)	1.9E-03	5.8E-05	3.0E-04	1.9E-01
						Selenium	2/2	2.40E-01 (Max)	1.9E-03	4.6E-04	5.0E-03	9.2E-02
						Zinc	2/2	8.00E+00 (Max)	1.9E-03	1.5E-02	3.0E-01	5.1E-02
						TYPE TOTAL						1.5E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Chlordane	31/110	1.34E-01 (UOM)	1.9E-03	2.6E-04	6.0E-05	4.3E+00
						Aldrin	2/40	1.57E-02 (UOM)	1.9E-03	3.0E-05	3.0E-05	1.0E+00
						Dieldrin	3/22	1.86E-02 (UOM)	1.9E-03	3.6E-05	5.0E-05	7.1E-01
						Endosulfan I	3/22	1.45E-02 (UOM)	1.9E-03	2.8E-05	5.0E-05	5.6E-01
						Endrin	1/22	1.25E-02 (UOM)	1.9E-03	2.4E-05	3.0E-04	8.0E-02
						Gamma-BHC (Lindane)	1/22	1.12E-02 (UOM)	1.9E-03	2.2E-05	3.0E-04	7.2E-02
						4,4'-DDT	8/40	1.26E-02 (UOM)	1.9E-03	2.4E-05	5.0E-04	4.9E-02
						TYPE TOTAL						6.7E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
			LargeMouth Bass	Ingestion	I	Mercury	5/5	2.37E-01 (UOM)	1.9E-03	4.5E-04	3.0E-04	1.5E+00
						Arsenic	5/5	1.65E-01 (UOM)	1.9E-03	3.2E-04	3.0E-04	1.1E+00
						Zinc	4/4	4.20E+00 (Max)	1.9E-03	8.1E-03	3.0E-01	2.7E-02
						Beryllium	2/5	2.11E-02 (UOM)	1.9E-03	4.0E-05	5.0E-03	8.1E-03
						TYPE TOTAL						2.6E+00

Table B.11 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DIRECT	Child	Fish	LargeMouth Bass	Ingestion	O	Chlordane	5/20	7.64E-02 (UOM)	1.9E-03	1.5E-04	6.0E-05	2.4E+00
						TYPE TOTAL						2.4E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
			Striped/Hybrid Bass	Ingestion	O	Chlordane	14/45	1.34E-01 (UOM)	1.9E-03	2.6E-04	6.0E-05	4.3E+00
						Dieldrin	1/10	1.00E-02 (Max)	1.9E-03	1.9E-05	5.0E-05	3.8E-01
						Endosulfan I	1/10	1.00E-02 (Max)	1.9E-03	1.9E-05	5.0E-05	3.8E-01
						Endrin	3/10	3.56E-02 (UOM)	1.9E-03	6.8E-05	3.0E-04	2.3E-01
						4,4'-DDT	2/10	1.44E-02 (UOM)	1.9E-03	2.8E-05	5.0E-04	5.5E-02
						TYPE TOTAL						5.3E+00

Table B.12 Human health risk characterization for carcinogens measured in LWBR water during Phase 1 sampling where at least one measurement was above detection limits

Irrigation Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/kg	Dose/Life	Bq/Life		
IRRIGAT	Adult	IrrSoil	TOTAL	External	R	Sr-90	3/3	1.05E+00 (Max)	4.4E+00	4.6E+00	0.0E+00	0.0E+00
						TYPE TOTAL						0.0E+00
								Bq/kg	kg	Bq/Life	Risk/Bq	
				Ingestion	R	Sr-90	3/3	6.70E-01 (Max)	1.1E+00	7.0E-01	9.7E-10	6.8E-10
						TYPE TOTAL						6.8E-10
								Bq/m3	m3	Bq/Life	Risk/Bq	
				Inhalation	R	Sr-90	3/3	1.07E-08 (Max)	1.8E+05	1.9E-03	1.7E-09	3.2E-12
						TYPE TOTAL						3.2E-12
								Bq/kg	kg	Bq/Life	Risk/Bq	
		Meat	TOTAL	Ingestion	R	Sr-90	3/3	8.06E-03 (Max)	1.6E+03	1.3E+01	9.7E-10	1.2E-08
						TYPE TOTAL						1.2E-08
								Bq/L	L	Bq/Life	Risk/Bq	
		Milk	TOTAL	Ingestion	R	Sr-90	3/3	1.82E-02 (Max)	3.9E+03	7.2E+01	9.7E-10	7.0E-08
						TYPE TOTAL						7.0E-08
								Bq/kg	kg	Bq/Life	Risk/Bq	
		Veg.	TOTAL	Ingestion	R	Sr-90	3/3	2.75E-01 (Max)	8.4E+02	2.3E+02	9.7E-10	2.2E-07
						TYPE TOTAL						2.2E-07
						MATRIX TOTAL						3.1E-07

Table B.13 Human health characterization for carcinogens measured in water sampled near Brashear Island by ORR monitoring program where at least one measurement was above detection limits

Irrigation Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/kg	Dose/Life	Bq/Life		
IRRIGAT	Adult	IrrSoil	TOTAL	External	R	Co-60	6/6	9.80E-01 (UOM)	6.7E+00	6.6E+00	2.3E-07	1.5E-06
						Cs-137	6/6	4.60E+00 (UOM)	4.3E+00	2.0E+01	5.4E-08	1.1E-06
						Uranium, total	6/6	1.56E+01 (UOM)	1.8E+00	2.7E+01	9.7E-10*	2.7E-08
						Tc-99	6/6	2.25E-04 (UOM)	6.8E+00	1.5E-03	6.2E-09	9.6E-12
						H-3	6/6	2.47E+02 (UOM)	5.9E+00	1.5E+03	0.0E+00	0.0E+00
						TYPE TOTAL						2.6E-06
								Bq/kg	kg	Bq/Life	Risk/Bq	
				Ingestion	R	Uranium, total	6/6	3.99E+00 (UOM)	1.1E+00	4.2E+00	7.6E-10*	3.2E-09
						Cs-137	6/6	2.88E+00 (UOM)	1.1E+00	3.0E+00	7.6E-10	2.3E-09
						Co-60	6/6	9.66E-01 (UOM)	1.1E+00	1.0E+00	4.1E-10	4.1E-10
						H-3	6/6	2.13E+02 (UOM)	1.1E+00	2.2E+02	1.5E-12	3.3E-10
						Tc-99	6/6	2.25E-04 (UOM)	1.1E+00	2.4E-04	1.4E-12	3.2E-16
						TYPE TOTAL						6.2E-09
								Bq/m3	m3	Bq/Life	Risk/Bq	
				Inhalation	R	Uranium, total	6/6	6.38E-08 (UOM)	1.8E+05	1.1E-02	1.4E-06*	1.6E-08
						Co-60	6/6	1.54E-08 (UOM)	1.8E+05	2.7E-03	4.1E-09	1.1E-11
						Cs-137	6/6	4.61E-08 (UOM)	1.8E+05	8.1E-03	5.1E-10	4.2E-12
						H-3	6/6	3.41E-06 (UOM)	1.8E+05	6.0E-01	2.1E-12	1.3E-12
						Tc-99	6/6	3.60E-12 (UOM)	1.8E+05	6.4E-07	7.3E-13	4.6E-19
						TYPE TOTAL						1.6E-08
								Bq/kg	kg	Bq/Life	Risk/Bq	
		Meat	TOTAL	Ingestion	R	Cs-137	6/6	8.58E-01 (UOM)	1.6E+03	1.4E+03	7.6E-10	1.0E-06
						Co-60	6/6	7.39E-01 (UOM)	1.6E+03	1.2E+03	4.1E-10	4.7E-07
						Uranium, total	6/6	2.94E-01 (UOM)	1.6E+03	4.6E+02	7.6E-10*	3.5E-07
						H-3	6/6	5.94E+00 (UOM)	1.6E+03	9.3E+03	1.5E-12	1.4E-08
						Tc-99	6/6	4.16E-03 (UOM)	1.6E+03	6.6E+00	1.4E-12	8.8E-12
						TYPE TOTAL						1.9E-06

Table B.13 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/L	L	Bq/Life	Risk/Bq	
IRRIGAT	Adult	Milk	TOTAL	Ingestion	R	Cs-137	6/6	5.16E-01 (UOM)	3.9E+03	2.0E+03	7.6E-10	1.5E-06
						Co-60	6/6	1.11E-01 (UOM)	3.9E+03	4.4E+02	4.1E-10	1.8E-07
						Uranium, total	6/6	3.97E-02 (UOM)	3.9E+03	1.6E+02	7.6E-10*	1.2E-07
						H-3	6/6	5.94E+00 (UOM)	3.9E+03	2.3E+04	1.5E-12	3.4E-08
						Tc-99	6/6	1.02E-02 (UOM)	3.9E+03	4.0E+01	1.4E-12	5.4E-11
						TYPE TOTAL						1.9E-06
								Bq/kg	kg	Bq/Life	Risk/Bq	
		Veg.	TOTAL	Ingestion	R	Cs-137	6/6	9.71E-01 (UOM)	8.4E+02	8.2E+02	7.6E-10	6.2E-07
						Uranium, total	6/6	9.70E-01 (UOM)	8.4E+02	8.2E+02	7.6E-10*	6.2E-07
						Co-60	6/6	8.75E-01 (UOM)	8.4E+02	7.3E+02	4.1E-10	3.0E-07
						H-3	6/6	5.94E+00 (UOM)	8.4E+02	5.0E+03	1.5E-12	7.3E-09
						Tc-99	6/6	2.82E-02 (UOM)	8.4E+02	2.4E+01	1.4E-12	3.2E-11
						TYPE TOTAL						1.5E-06
						MATRIX TOTAL						7.9E-06

*Slope factors for U-238 used for total uranium.

Table B.14 Human health risk characterization for noncarcinogens measured in water sampled near Brashear Island by ORR monitoring program where at least one measurement was above detection limits

Irrigation Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
IRRIGAT	Adult	IrrSoil	TOTAL	Dermal	I	Fluoride	1/6	3.57E+00 (Max)	7.8E-08	2.8E-07	6.0E-02	4.7E-06
						Manganese	6/6	3.14E+00 (UOM)	7.8E-08	2.5E-07	1.4E-01	1.8E-06
						Barium	5/6	1.46E+00 (UOM)	7.8E-08	1.1E-07	7.0E-02	1.6E-06
						Vanadium	1/6	7.15E-02 (Max)	7.8E-08	5.6E-09	7.0E-03	8.0E-07
						Uranium	4/5	2.61E-02 (UOM)	7.8E-08	2.0E-09	3.0E-03	6.8E-07
						Zinc	1/6	2.31E-01 (UOM)	7.8E-08	1.8E-08	3.0E-01	6.0E-08
						TYPE TOTAL						9.6E-06
								mg/kg	1/d	mg/kg/d	mg/kg/d	
				Ingestion	I	Fluoride	1/6	3.57E+00 (Max)	1.4E-06	4.9E-06	6.0E-02	8.2E-05
						Manganese	6/6	3.14E+00 (UOM)	1.4E-06	4.3E-06	1.4E-01	3.1E-05
						Barium	5/6	1.46E+00 (UOM)	1.4E-06	2.0E-06	7.0E-02	2.9E-05
						Vanadium	1/6	7.15E-02 (Max)	1.4E-06	9.8E-08	7.0E-03	1.4E-05
						Uranium	4/5	2.61E-02 (UOM)	1.4E-06	3.6E-08	3.0E-03	1.2E-05
						Zinc	1/6	2.31E-01 (UOM)	1.4E-06	3.2E-07	3.0E-01	1.1E-06
						TYPE TOTAL						1.7E-04
								mg/m3	unitless	mg/m3	mg/m3	
				Inhalation	I	Manganese	6/6	5.03E-08 (UOM)	3.2E-01	1.6E-08	4.0E-04	4.0E-05
						Barium	5/6	2.33E-08 (UOM)	3.2E-01	7.5E-09	5.0E-04	1.5E-05
						TYPE TOTAL						5.5E-05
								mg/kg	1/d	mg/kg/d	mg/kg/d	
		Meat	TOTAL	Ingestion	I	Zinc	1/6	2.36E-01 (UOM)	2.1E-03	4.8E-04	3.0E-01	1.6E-03
						Vanadium	1/6	5.26E-03 (Max)	2.1E-03	1.1E-05	7.0E-03	1.5E-03
						Uranium	4/5	1.92E-03 (UOM)	2.1E-03	3.9E-06	3.0E-03	1.3E-03
						Manganese	6/6	1.97E-02 (UOM)	2.1E-03	4.0E-05	1.4E-01	2.9E-04
						Barium	5/6	2.16E-03 (UOM)	2.1E-03	4.4E-06	7.0E-02	6.3E-05
						TYPE TOTAL						4.8E-03

Table B.14 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
IRRIGAT	Adult	Milk	TOTAL	Ingestion	I	Zinc	1/6	5.31E-02 (UOM)	5.1E-03	2.7E-04	3.0E-01	9.1E-04
						Barium	5/6	9.75E-03 (UOM)	5.1E-03	5.0E-05	7.0E-02	7.2E-04
						Uranium	4/5	2.60E-04 (UOM)	5.1E-03	1.3E-06	3.0E-03	4.4E-04
						Vanadium	1/6	5.93E-04 (Max)	5.1E-03	3.0E-06	7.0E-03	4.4E-04
						Manganese	6/6	7.44E-03 (UOM)	5.1E-03	3.8E-05	1.4E-01	2.7E-04
						TYPE TOTAL						2.8E-03
								mg/kg	1/d	mg/kg/d	mg/kg/d	
		Veg.	TOTAL	Ingestion	I	Manganese	6/6	1.33E+00 (UOM)	1.1E-03	1.5E-03	1.4E-01	1.0E-02
						Barium	5/6	3.57E-01 (UOM)	1.1E-03	3.9E-04	7.0E-02	5.6E-03
						Vanadium	1/6	1.74E-02 (Max)	1.1E-03	1.9E-05	7.0E-03	2.7E-03
						Uranium	4/5	6.34E-03 (UOM)	1.1E-03	6.9E-06	3.0E-03	2.3E-03
						Zinc	1/6	7.91E-02 (UOM)	1.1E-03	8.7E-05	3.0E-01	2.9E-04
						TYPE TOTAL						2.1E-02
						MATRIX TOTAL						2.9E-02
								mg/kg	1/d	mg/kg/d	mg/kg/d	
	Child	IrrSoil	TOTAL	Dermal	I	Fluoride	1/6	3.57E+00 (Max)	1.5E-07	5.2E-07	6.0E-02	8.7E-06
						Manganese	6/6	3.14E+00 (UOM)	1.5E-07	4.6E-07	1.4E-01	3.3E-06
						Barium	5/6	1.46E+00 (UOM)	1.5E-07	2.1E-07	7.0E-02	3.1E-06
						Vanadium	1/6	7.15E-02 (Max)	1.5E-07	1.0E-08	7.0E-03	1.5E-06
						Uranium	4/5	2.61E-02 (UOM)	1.5E-07	3.8E-09	3.0E-03	1.3E-06
						Zinc	1/6	2.31E-01 (UOM)	1.5E-07	3.4E-08	3.0E-01	1.1E-07
						TYPE TOTAL						1.8E-05
								mg/kg	1/d	mg/kg/d	mg/kg/d	
				Ingestion	I	Fluoride	1/6	3.57E+00 (Max)	1.3E-05	4.6E-05	6.0E-02	7.6E-04
						Manganese	6/6	3.14E+00 (UOM)	1.3E-05	4.0E-05	1.4E-01	2.9E-04
						Barium	5/6	1.46E+00 (UOM)	1.3E-05	1.9E-05	7.0E-02	2.7E-04
						Vanadium	1/6	7.15E-02 (Max)	1.3E-05	9.1E-07	7.0E-03	1.3E-04
						Uranium	4/5	2.61E-02 (UOM)	1.3E-05	3.3E-07	3.0E-03	1.1E-04
						Zinc	1/6	2.31E-01 (UOM)	1.3E-05	3.0E-06	3.0E-01	9.8E-06
						TYPE TOTAL						1.6E-03

Table B.14 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/m3	unitless	mg/m3	mg/m3	
IRRIGAT	Child	IrrSoil	TOTAL	Inhalation	I	Manganese	6/6	5.03E-08 (UOM)	3.2E-01	1.6E-08	4.0E-04	4.0E-05
						Barium	5/6	2.33E-08 (UOM)	3.2E-01	7.5E-09	5.0E-04	1.5E-05
						TYPE TOTAL						5.5E-05
								mg/kg	1/d	mg/kg/d	mg/kg/d	
		Meat	TOTAL	Ingestion	I	Zinc	1/6	2.36E-01 (UOM)	9.6E-03	2.3E-03	3.0E-01	7.5E-03
						Vanadium	1/6	5.26E-03 (Max)	9.6E-03	5.0E-05	7.0E-03	7.2E-03
						Uranium	4/5	1.92E-03 (UOM)	9.6E-03	1.8E-05	3.0E-03	6.1E-03
						Manganese	6/6	1.97E-02 (UOM)	9.6E-03	1.9E-04	1.4E-01	1.3E-03
						Barium	5/6	2.16E-03 (UOM)	9.6E-03	2.1E-05	7.0E-02	3.0E-04
						TYPE TOTAL						2.3E-02
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
		Milk	TOTAL	Ingestion	I	Zinc	1/6	5.31E-02 (UOM)	4.8E-02	2.5E-03	3.0E-01	8.5E-03
						Barium	5/6	9.75E-03 (UOM)	4.8E-02	4.7E-04	7.0E-02	6.7E-03
						Uranium	4/5	2.60E-04 (UOM)	4.8E-02	1.2E-05	3.0E-03	4.1E-03
						Vanadium	1/6	5.93E-04 (Max)	4.8E-02	2.8E-05	7.0E-03	4.1E-03
						Manganese	6/6	7.44E-03 (UOM)	4.8E-02	3.6E-04	1.4E-01	2.5E-03
						TYPE TOTAL						2.6E-02
								mg/kg	1/d	mg/kg/d	mg/kg/d	
		Veg.	TOTAL	Ingestion	I	Manganese	6/6	1.33E+00 (UOM)	5.1E-03	6.8E-03	1.4E-01	4.9E-02
						Barium	5/6	3.57E-01 (UOM)	5.1E-03	1.8E-03	7.0E-02	2.6E-02
						Vanadium	1/6	1.74E-02 (Max)	5.1E-03	8.9E-05	7.0E-03	1.3E-02
						Uranium	4/5	6.34E-03 (UOM)	5.1E-03	3.2E-05	3.0E-03	1.1E-02
						Zinc	1/6	7.91E-02 (UOM)	5.1E-03	4.0E-04	3.0E-01	1.3E-03
						TYPE TOTAL						1.0E-01
						MATRIX TOTAL						1.5E-01

Table B.15 Human health risk characterization for carcinogens measured in deep sediment during Phase-1 sampling where at least one measurement was above detection limits

Dredging Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/kg	kg	Bq/Life	Risk/Bq	
DREDGE	Adult	Meat	AVERAGE	Ingestion	R	Cs-137	60/60	7.61E+00 (UOM)	1.6E+03	1.2E+04	7.6E-10	9.1E-06
						U-238	2/2	4.61E-01 (Max)	1.6E+03	7.3E+02	7.6E-10	5.5E-07
						U-234	2/2	5.22E-01 (Max)	1.6E+03	8.2E+02	4.3E-10	3.6E-07
						Co-60	56/60	9.78E-02 (UOM)	1.6E+03	1.5E+02	4.1E-10	6.2E-08
						Sr-90	2/2	1.52E-02 (Max)	1.6E+03	2.4E+01	9.7E-10	2.3E-08
						U-235	2/2	2.44E-02 (Max)	1.6E+03	3.8E+01	4.3E-10	1.7E-08
						Cm-248	2/2	1.86E-05 (Max)	1.6E+03	2.9E-02	2.5E-08	7.2E-10
						Am-241	2/2	5.48E-05 (Max)	1.6E+03	8.6E-02	6.5E-09	5.6E-10
						Cm-245,246	2/2	2.62E-05 (Max)	1.6E+03	4.1E-02	6.5E-09	2.7E-10
						Pu-239,240	2/2	2.60E-05 (Max)	1.6E+03	4.1E-02	6.2E-09	2.5E-10
						Pu-238	2/2	8.37E-06 (Max)	1.6E+03	1.3E-02	5.9E-09	7.8E-11
						Cm-243,244	2/2	9.71E-06 (Max)	1.6E+03	1.5E-02	4.3E-09	6.6E-11
						TYPE TOTAL						1.0E-05
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					I	Beryllium	4/4	5.41E-03 (Max)	8.8E-04	4.8E-06	4.3E+00	2.0E-05
						TYPE TOTAL						2.0E-05
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					O	Bis(2-ethylhexyl)phthalate	2/4	5.35E-03 (Max)	8.8E-04	4.7E-06	1.4E-02	6.6E-08
						TYPE TOTAL						6.6E-08
								Bq/L	L	Bq/Life	Risk/Bq	
		Milk	AVERAGE	Ingestion	R	Cs-137	60/60	4.54E+00 (UOM)	3.9E+03	1.8E+04	7.6E-10	1.4E-05
						U-238	2/2	6.19E-02 (Max)	3.9E+03	2.4E+02	7.6E-10	1.8E-07
						Sr-90	2/2	3.40E-02 (Max)	3.9E+03	1.3E+02	9.7E-10	1.3E-07
						U-234	2/2	7.00E-02 (Max)	3.9E+03	2.8E+02	4.3E-10	1.2E-07
						Co-60	56/60	1.46E-02 (UOM)	3.9E+03	5.7E+01	4.1E-10	2.3E-08
						U-235	2/2	3.27E-03 (Max)	3.9E+03	1.3E+01	4.3E-10	5.6E-09
						Cm-248	2/2	8.32E-07 (Max)	3.9E+03	3.3E-03	2.5E-08	8.0E-11
						Am-241	2/2	2.45E-06 (Max)	3.9E+03	9.6E-03	6.5E-09	6.2E-11
						Cm-245,246	2/2	1.17E-06 (Max)	3.9E+03	4.6E-03	6.5E-09	3.0E-11
						Pu-239,240	2/2	5.81E-07 (Max)	3.9E+03	2.3E-03	6.2E-09	1.4E-11
						Cm-243,244	2/2	4.34E-07 (Max)	3.9E+03	1.7E-03	4.3E-09	7.4E-12
						Pu-238	2/2	1.87E-07 (Max)	3.9E+03	7.4E-04	5.9E-09	4.4E-12
						TYPE TOTAL						1.4E-05

Table B.15 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								mg/L	L/kg-d	mg/kg/d	1/(mg/kg/d)	
DREDGE	Adult	Milk	AVERAGE	Ingestion	I	Beryllium	4/4	4.84E-06 (Max)	2.2E-03	1.1E-08	4.3E+00	4.6E-08
						TYPE TOTAL						4.6E-08
								mg/L	L/kg-d	mg/kg/d	1/(mg/kg/d)	
					O	Bis(2-ethylhexyl)phthalate	2/4	3.78E-03 (Max)	2.2E-03	8.3E-06	1.4E-02	1.2E-07
						TYPE TOTAL						1.2E-07
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
		Sediment	AVERAGE	Dermal	I	Beryllium	4/4	1.44E+00 (Max)	3.3E-08	4.8E-08	4.3E+00	2.1E-07
						TYPE TOTAL						2.1E-07
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					O	Bis(2-ethylhexyl)phthalate	2/4	2.41E+00 (Max)	3.3E-07	8.1E-07	1.4E-02	1.1E-08
						TYPE TOTAL						1.1E-08
								Bq/kg	Dose/Life	Bq/Life	Risk/Bq	
				External	R	Cs-137	60/60	2.71E+02 (UOM)	4.4E+00	1.2E+03	5.4E-08	6.4E-05
						Co-60	56/60	3.48E+00 (UOM)	1.6E+00	5.5E+00	2.3E-07	1.3E-06
						U-238	2/2	6.27E+01 (Max)	5.9E+00	3.7E+02	9.7E-10	3.6E-07
						U-235	2/2	3.32E+00 (Max)	5.9E+00	2.0E+01	6.5E-09	1.3E-07
						Cm-245,246	2/2	1.80E+00 (Max)	5.9E+00	1.1E+01	1.4E-09	1.5E-08
						Am-241	2/2	3.76E+00 (Max)	5.8E+00	2.2E+01	1.3E-10	2.9E-09
						U-234	2/2	7.10E+01 (Max)	5.9E+00	4.2E+02	8.1E-13	3.4E-10
						Pu-239,240	2/2	3.59E+00 (Max)	5.9E+00	2.1E+01	4.6E-13	9.8E-12
						Pu-238	2/2	1.16E+00 (Max)	5.3E+00	6.1E+00	7.6E-13	4.6E-12
						Cm-248	2/2	1.28E+00 (Max)	5.9E+00	7.6E+00	5.9E-13	4.5E-12
						Cm-243,244	2/2	6.67E-01 (Max)	3.6E+00	2.4E+00	8.1E-13	2.0E-12
						Sr-90	2/2	5.28E+00 (Max)	4.3E+00	2.3E+01	0.0E+00	0.0E+00
						TYPE TOTAL						6.5E-05

Table B.15 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								Bq/kg	kg	Bq/Life	Risk/Bq	
DREDGE	Adult	Sediment	AVERAGE	Ingestion	R	Cs-137	60/60	2.71E+02 (UOM)	1.1E+00	2.8E+02	7.6E-10	2.1E-07
						U-238	2/2	6.27E+01 (Max)	1.1E+00	6.6E+01	7.6E-10	5.0E-08
						Cm-248	2/2	1.28E+00 (Max)	1.1E+00	1.3E+00	2.5E-08	3.3E-08
						U-234	2/2	7.10E+01 (Max)	1.1E+00	7.5E+01	4.3E-10	3.2E-08
						Am-241	2/2	3.76E+00 (Max)	1.1E+00	3.9E+00	6.5E-09	2.6E-08
						Pu-239,240	2/2	3.59E+00 (Max)	1.1E+00	3.8E+00	6.2E-09	2.3E-08
						Cm-245,246	2/2	1.80E+00 (Max)	1.1E+00	1.9E+00	6.5E-09	1.2E-08
						Pu-238	2/2	1.16E+00 (Max)	1.1E+00	1.2E+00	5.9E-09	7.2E-09
						Sr-90	2/2	5.28E+00 (Max)	1.1E+00	5.5E+00	9.7E-10	5.4E-09
						Cm-243,244	2/2	6.67E-01 (Max)	1.1E+00	7.0E-01	4.3E-09	3.0E-09
						U-235	2/2	3.32E+00 (Max)	1.1E+00	3.5E+00	4.3E-10	1.5E-09
						Co-60	56/60	3.48E+00 (UOM)	1.1E+00	3.7E+00	4.1E-10	1.5E-09
						TYPE TOTAL						4.1E-07
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					I	Beryllium	4/4	1.44E+00 (Max)	5.9E-07	8.5E-07	4.3E+00	3.6E-06
						TYPE TOTAL						3.6E-06
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					O	Bis(2-ethylhexyl)phthalate	2/4	2.41E+00 (Max)	5.9E-07	1.4E-06	1.4E-02	2.0E-08
						TYPE TOTAL						2.0E-08
								Bq/m3	m3	Bq/Life	Risk/Bq	
				Inhalation	R	U-238	2/2	1.00E-06 (Max)	1.8E+05	1.8E-01	1.4E-06	2.5E-07
						U-234	2/2	1.14E-06 (Max)	1.8E+05	2.0E-01	7.0E-07	1.4E-07
						Cm-248	2/2	2.04E-08 (Max)	1.8E+05	3.6E-03	3.2E-06	1.2E-08
						Pu-239,240	2/2	5.74E-08 (Max)	1.8E+05	1.0E-02	1.0E-06	1.0E-08
						Am-241	2/2	6.02E-08 (Max)	1.8E+05	1.1E-02	8.6E-07	9.2E-09
						U-235	2/2	5.31E-08 (Max)	1.8E+05	9.4E-03	6.8E-07	6.3E-09
						Cm-245,246	2/2	2.87E-08 (Max)	1.8E+05	5.1E-03	8.6E-07	4.4E-09
						Pu-238	2/2	1.85E-08 (Max)	1.8E+05	3.3E-03	1.1E-06	3.4E-09
						Cm-243,244	2/2	1.07E-08 (Max)	1.8E+05	1.9E-03	5.9E-07	1.1E-09
						Cs-137	60/60	4.33E-06 (UOM)	1.8E+05	7.6E-01	5.1E-10	3.9E-10
						Co-60	56/60	5.57E-08 (UOM)	1.8E+05	9.8E-03	4.1E-09	4.0E-11
						Sr-90	2/2	8.45E-08 (Max)	1.8E+05	1.5E-02	1.7E-09	2.5E-11
						TYPE TOTAL						4.4E-07

Table B.15 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Slope factor	Cancer risk
								mg/m3	m3/kg-d	mg/kg/d	1/(mg/kg/d)	
DREDGE	Adult	Sediment	AVERAGE	Inhalation	I	Chromium	6/6	6.62E-07 (UOM)	9.9E-02	6.5E-08	4.1E+01	2.7E-06
						Arsenic	4/4	1.46E-07 (Max)	9.9E-02	1.4E-08	5.0E+01	7.2E-07
						Cadmium	4/4	7.33E-08 (UOM)	9.9E-02	7.2E-09	6.1E+00	4.4E-08
						Beryllium	4/4	2.31E-08 (Max)	9.9E-02	2.3E-09	8.4E+00	1.9E-08
						TYPE TOTAL						3.5E-06
								Bq/kg	kg	Bq/Life	Risk/Bq	
		Veg.	AVERAGE	Ingestion	R	Cs-137	60/60	2.04E+00 (UOM)	8.4E+02	1.7E+03	7.6E-10	1.3E-06
						Sr-90	2/2	3.96E-01 (Max)	8.4E+02	3.3E+02	9.7E-10	3.2E-07
						U-238	2/2	3.38E-02 (Max)	8.4E+02	2.8E+01	7.6E-10	2.1E-08
						U-234	2/2	3.82E-02 (Max)	8.4E+02	3.2E+01	4.3E-10	1.4E-08
						Co-60	56/60	2.62E-02 (UOM)	8.4E+02	2.2E+01	4.1E-10	8.9E-09
						Cm-248	2/2	3.68E-04 (Max)	8.4E+02	3.1E-01	2.5E-08	7.6E-09
						Am-241	2/2	1.08E-03 (Max)	8.4E+02	9.1E-01	6.5E-09	5.9E-09
						Pu-239,240	2/2	5.86E-04 (Max)	8.4E+02	4.9E-01	6.2E-09	3.1E-09
						Cm-245,246	2/2	5.18E-04 (Max)	8.4E+02	4.4E-01	6.5E-09	2.8E-09
						Pu-238	2/2	1.89E-04 (Max)	8.4E+02	1.6E-01	5.9E-09	9.4E-10
						Cm-243,244	2/2	1.92E-04 (Max)	8.4E+02	1.6E-01	4.3E-09	7.0E-10
						U-235	2/2	1.79E-03 (Max)	8.4E+02	1.5E+00	4.3E-10	6.5E-10
						TYPE TOTAL						1.7E-06
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					I	Beryllium	4/4	1.50E-03 (Max)	4.7E-04	7.0E-07	4.3E+00	3.0E-06
						TYPE TOTAL						3.0E-06
								mg/kg	1/d	mg/kg/d	1/(mg/kg/d)	
					O	Bis(2-ethylhexyl)phthalate	2/4	3.49E-02 (Max)	4.7E-04	1.6E-05	1.4E-02	2.3E-07
						TYPE TOTAL						2.3E-07
						MATRIX TOTAL						1.2E-04

Table B.16 Human health risk characterization for noncarcinogens measured in deep sediment during Phase-1 sampling where at least one measurement was above detection limits

Dredging Scenario

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DREDGE	Adult	Meat	AVERAGE	Ingestion	I	Zinc	4/4	9.63E+01 (Max)	2.1E-03	2.0E-01	3.0E-01	6.6E-01
						Mercury	6/6	7.28E-02 (UOM)	2.1E-03	1.5E-04	3.0E-04	5.0E-01
						Chromium	6/6	9.85E-01 (UOM)	2.1E-03	2.0E-03	5.0E-03	4.0E-01
						Cadmium	4/4	1.98E-02 (UOM)	2.1E-03	4.1E-05	1.0E-03	4.1E-02
						Uranium	2/2	4.38E-02 (Max)	2.1E-03	9.0E-05	3.0E-03	3.0E-02
						Nickel	4/4	6.13E-02 (UOM)	2.1E-03	1.3E-04	2.0E-02	6.3E-03
						Silver	3/4	7.16E-03 (UOM)	2.1E-03	1.5E-05	5.0E-03	2.9E-03
						Beryllium	4/4	5.41E-03 (Max)	2.1E-03	1.1E-05	5.0E-03	2.2E-03
						Antimony	1/3	3.81E-04 (Max)	2.1E-03	7.8E-07	4.0E-04	2.0E-03
						Arsenic	4/4	1.33E-04 (Max)	2.1E-03	2.7E-07	3.0E-04	9.1E-04
						TYPE TOTAL						1.6E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	5.35E-03 (Max)	2.1E-03	1.1E-05	2.0E-02	5.5E-04
						TYPE TOTAL						5.5E-04
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
		Milk	AVERAGE	Ingestion	I	Cadmium	4/4	8.85E-02 (UOM)	5.1E-03	4.5E-04	1.0E-03	4.5E-01
						Zinc	4/4	2.15E+01 (Max)	5.1E-03	1.1E-01	3.0E-01	3.7E-01
						Chromium	6/6	1.47E-01 (UOM)	5.1E-03	7.5E-04	5.0E-03	1.5E-01
						Mercury	6/6	8.14E-03 (UOM)	5.1E-03	4.2E-05	3.0E-04	1.4E-01
						Silver	3/4	3.20E-02 (UOM)	5.1E-03	1.6E-04	5.0E-03	3.3E-02
						Nickel	4/4	6.86E-02 (UOM)	5.1E-03	3.5E-04	2.0E-02	1.8E-02
						Uranium	2/2	5.87E-03 (Max)	5.1E-03	3.0E-05	3.0E-03	1.0E-02
						Antimony	1/3	8.53E-05 (Max)	5.1E-03	4.4E-07	4.0E-04	1.1E-03
						Arsenic	4/4	5.93E-06 (Max)	5.1E-03	3.0E-08	3.0E-04	1.0E-04
						Beryllium	4/4	4.84E-06 (Max)	5.1E-03	2.5E-08	5.0E-03	5.0E-06
						TYPE TOTAL						1.2E+00
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	3.78E-03 (Max)	5.1E-03	1.9E-05	2.0E-02	9.7E-04
						TYPE TOTAL						9.7E-04

Table B.16 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DREDGE	Adult	Sediment	AVERAGE	Dermal	I	Arsenic	4/4	9.10E+00 (Max)	7.8E-08	7.1E-07	3.0E-04	2.4E-03
						Mercury	6/6	2.53E+00 (UOM)	7.8E-08	2.0E-07	3.0E-04	6.6E-04
						Chromium	6/6	4.14E+01 (UOM)	7.8E-08	3.2E-06	5.0E-03	6.5E-04
						Cadmium	4/4	4.58E+00 (UOM)	7.8E-08	3.6E-07	1.0E-03	3.6E-04
						Uranium	2/2	5.95E+00 (Max)	7.8E-08	4.6E-07	3.0E-03	1.5E-04
						Nickel	4/4	2.84E+01 (UOM)	7.8E-08	2.2E-06	2.0E-02	1.1E-04
						Antimony	1/3	4.81E-01 (Max)	7.8E-08	3.8E-08	4.0E-04	9.4E-05
						Zinc	4/4	2.68E+02 (Max)	7.8E-08	2.1E-05	3.0E-01	7.0E-05
						Beryllium	4/4	1.44E+00 (Max)	7.8E-08	1.1E-07	5.0E-03	2.3E-05
						Silver	3/4	6.63E-01 (UOM)	7.8E-08	5.2E-08	5.0E-03	1.0E-05
						TYPE TOTAL						4.5E-03
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	2.41E+00 (Max)	7.8E-07	1.9E-06	2.0E-02	9.4E-05
						TYPE TOTAL						9.4E-05
								mg/kg	1/d	mg/kg/d	mg/kg/d	
				Ingestion	I	Arsenic	4/4	9.10E+00 (Max)	1.4E-06	1.2E-05	3.0E-04	4.2E-02
						Mercury	6/6	2.53E+00 (UOM)	1.4E-06	3.5E-06	3.0E-04	1.2E-02
						Chromium	6/6	4.14E+01 (UOM)	1.4E-06	5.7E-05	5.0E-03	1.1E-02
						Cadmium	4/4	4.58E+00 (UOM)	1.4E-06	6.3E-06	1.0E-03	6.3E-03
						Uranium	2/2	5.95E+00 (Max)	1.4E-06	8.2E-06	3.0E-03	2.7E-03
						Nickel	4/4	2.84E+01 (UOM)	1.4E-06	3.9E-05	2.0E-02	1.9E-03
						Antimony	1/3	4.81E-01 (Max)	1.4E-06	6.6E-07	4.0E-04	1.6E-03
						Zinc	4/4	2.68E+02 (Max)	1.4E-06	3.7E-04	3.0E-01	1.2E-03
						Beryllium	4/4	1.44E+00 (Max)	1.4E-06	2.0E-06	5.0E-03	4.0E-04
						Silver	3/4	6.63E-01 (UOM)	1.4E-06	9.1E-07	5.0E-03	1.8E-04
						TYPE TOTAL						7.9E-02
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	2.41E+00 (Max)	1.4E-06	3.3E-06	2.0E-02	1.7E-04
						TYPE TOTAL						1.7E-04
								mg/m3	unitless	mg/m3	mg/m3	
				Inhalation	I	Mercury	6/6	4.04E-08 (UOM)	3.2E-01	1.3E-08	3.0E-04	4.3E-05
						TYPE TOTAL						4.3E-05

Table B.16 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DREDGE	Adult	Veg.	AVERAGE	Ingestion	I	Mercury	6/6	1.90E-01 (UOM)	1.1E-03	2.1E-04	3.0E-04	6.9E-01
						Cadmium	4/4	5.73E-01 (UOM)	1.1E-03	6.3E-04	1.0E-03	6.3E-01
						Zinc	4/4	2.68E+01 (Max)	1.1E-03	2.9E-02	3.0E-01	9.8E-02
						Chromium	6/6	1.05E-01 (UOM)	1.1E-03	1.2E-04	5.0E-03	2.3E-02
						Nickel	4/4	3.56E-01 (UOM)	1.1E-03	3.9E-04	2.0E-02	1.9E-02
						Arsenic	4/4	2.62E-03 (Max)	1.1E-03	2.9E-06	3.0E-04	9.6E-03
						Silver	3/4	3.32E-02 (UOM)	1.1E-03	3.6E-05	5.0E-03	7.3E-03
						Antimony	1/3	1.22E-03 (Max)	1.1E-03	1.3E-06	4.0E-04	3.3E-03
						Uranium	2/2	3.20E-03 (Max)	1.1E-03	3.5E-06	3.0E-03	1.2E-03
						Beryllium	4/4	1.50E-03 (Max)	1.1E-03	1.6E-06	5.0E-03	3.3E-04
						TYPE TOTAL						1.5E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	3.49E-02 (Max)	1.1E-03	3.8E-05	2.0E-02	1.9E-03
						TYPE TOTAL						1.9E-03
						MATRIX TOTAL						4.4E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
	Child	Meat	AVERAGE	Ingestion	I	Zinc	4/4	9.63E+01 (Max)	9.6E-03	9.2E-01	3.0E-01	3.1E+00
						Mercury	6/6	7.28E-02 (UOM)	9.6E-03	7.0E-04	3.0E-04	2.3E+00
						Chromium	6/6	9.85E-01 (UOM)	9.6E-03	9.4E-03	5.0E-03	1.9E+00
						Cadmium	4/4	1.98E-02 (UOM)	9.6E-03	1.9E-04	1.0E-03	1.9E-01
						Uranium	2/2	4.38E-02 (Max)	9.6E-03	4.2E-04	3.0E-03	1.4E-01
						Nickel	4/4	6.13E-02 (UOM)	9.6E-03	5.9E-04	2.0E-02	2.9E-02
						Silver	3/4	7.16E-03 (UOM)	9.6E-03	6.9E-05	5.0E-03	1.4E-02
						Beryllium	4/4	5.41E-03 (Max)	9.6E-03	5.2E-05	5.0E-03	1.0E-02
						Antimony	1/3	3.81E-04 (Max)	9.6E-03	3.7E-06	4.0E-04	9.1E-03
						Arsenic	4/4	1.33E-04 (Max)	9.6E-03	1.3E-06	3.0E-04	4.2E-03
						TYPE TOTAL						7.7E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	5.35E-03 (Max)	9.6E-03	5.1E-05	2.0E-02	2.6E-03
						TYPE TOTAL						2.6E-03

Table B.16 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
DREDGE	Child	Milk	AVERAGE	Ingestion	I	Cadmium	4/4	8.85E-02 (UOM)	4.8E-02	4.2E-03	1.0E-03	4.2E+00
						Zinc	4/4	2.15E+01 (Max)	4.8E-02	1.0E+00	3.0E-01	3.4E+00
						Chromium	6/6	1.47E-01 (UOM)	4.8E-02	7.0E-03	5.0E-03	1.4E+00
						Mercury	6/6	8.14E-03 (UOM)	4.8E-02	3.9E-04	3.0E-04	1.3E+00
						Silver	3/4	3.20E-02 (UOM)	4.8E-02	1.5E-03	5.0E-03	3.1E-01
						Nickel	4/4	6.86E-02 (UOM)	4.8E-02	3.3E-03	2.0E-02	1.6E-01
						Uranium	2/2	5.87E-03 (Max)	4.8E-02	2.8E-04	3.0E-03	9.4E-02
						Antimony	1/3	8.53E-05 (Max)	4.8E-02	4.1E-06	4.0E-04	1.0E-02
						Arsenic	4/4	5.93E-06 (Max)	4.8E-02	2.8E-07	3.0E-04	9.5E-04
						Beryllium	4/4	4.84E-06 (Max)	4.8E-02	2.3E-07	5.0E-03	4.6E-05
						TYPE TOTAL						1.1E+01
								mg/L	L/kg-d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	3.78E-03 (Max)	4.8E-02	1.8E-04	2.0E-02	9.1E-03
						TYPE TOTAL						9.1E-03
								mg/kg	1/d	mg/kg/d	mg/kg/d	
		Sediment	AVERAGE	Dermal	I	Arsenic	4/4	9.10E+00 (Max)	1.5E-07	1.3E-06	3.0E-04	4.4E-03
						Mercury	6/6	2.53E+00 (UOM)	1.5E-07	3.7E-07	3.0E-04	1.2E-03
						Chromium	6/6	4.14E+01 (UOM)	1.5E-07	6.1E-06	5.0E-03	1.2E-03
						Cadmium	4/4	4.58E+00 (UOM)	1.5E-07	6.7E-07	1.0E-03	6.7E-04
						Uranium	2/2	5.95E+00 (Max)	1.5E-07	8.7E-07	3.0E-03	2.9E-04
						Nickel	4/4	2.84E+01 (UOM)	1.5E-07	4.2E-06	2.0E-02	2.1E-04
						Antimony	1/3	4.81E-01 (Max)	1.5E-07	7.1E-08	4.0E-04	1.8E-04
						Zinc	4/4	2.68E+02 (Max)	1.5E-07	3.9E-05	3.0E-01	1.3E-04
						Beryllium	4/4	1.44E+00 (Max)	1.5E-07	2.1E-07	5.0E-03	4.2E-05
						Silver	3/4	6.63E-01 (UOM)	1.5E-07	9.7E-08	5.0E-03	1.9E-05
						TYPE TOTAL						8.4E-03
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	2.41E+00 (Max)	1.5E-06	3.5E-06	2.0E-02	1.8E-04
						TYPE TOTAL						1.8E-04
								mg/kg	1/d	mg/kg/d	mg/kg/d	
				Ingestion	I	Arsenic	4/4	9.10E+00 (Max)	1.3E-05	1.2E-04	3.0E-04	3.9E-01
						Mercury	6/6	2.53E+00 (UOM)	1.3E-05	3.2E-05	3.0E-04	1.1E-01

Table B.16 (continued)

Scenario	Age	Media	Matrix	Route	Type	Contaminant	Detects/ Total	Concentration (type)	Intake	Exposure	Reference value	Hazard quotient
								mg/kg	1/d	mg/kg/d	mg/kg/d	
DREDGE	Child	Sediment	AVERAGE	Ingestion	I	Chromium	6/6	4.14E+01 (UOM)	1.3E-05	5.3E-04	5.0E-03	1.1E-01
						Cadmium	4/4	4.58E+00 (UOM)	1.3E-05	5.9E-05	1.0E-03	5.9E-02
						Uranium	2/2	5.95E+00 (Max)	1.3E-05	7.6E-05	3.0E-03	2.5E-02
						Nickel	4/4	2.84E+01 (UOM)	1.3E-05	3.6E-04	2.0E-02	1.8E-02
						Antimony	1/3	4.81E-01 (Max)	1.3E-05	6.1E-06	4.0E-04	1.5E-02
						Zinc	4/4	2.68E+02 (Max)	1.3E-05	3.4E-03	3.0E-01	1.1E-02
						Beryllium	4/4	1.44E+00 (Max)	1.3E-05	1.8E-05	5.0E-03	3.7E-03
						Silver	3/4	6.63E-01 (UOM)	1.3E-05	8.5E-06	5.0E-03	1.7E-03
						TYPE TOTAL						7.4E-01
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	2.41E+00 (Max)	1.3E-05	3.1E-05	2.0E-02	1.5E-03
						TYPE TOTAL						1.5E-03
								mg/m3	unitless	mg/m3	mg/m3	
				Inhalation	I	Mercury	6/6	4.04E-08 (UOM)	3.2E-01	1.3E-08	3.0E-04	4.3E-05
						TYPE TOTAL						4.3E-05
								mg/kg	1/d	mg/kg/d	mg/kg/d	
		Veg.	AVERAGE	Ingestion	I	Mercury	6/6	1.90E-01 (UOM)	5.1E-03	9.7E-04	3.0E-04	3.2E+00
						Cadmium	4/4	5.73E-01 (UOM)	5.1E-03	2.9E-03	1.0E-03	2.9E+00
						Zinc	4/4	2.68E+01 (Max)	5.1E-03	1.4E-01	3.0E-01	4.6E-01
						Chromium	6/6	1.05E-01 (UOM)	5.1E-03	5.4E-04	5.0E-03	1.1E-01
						Nickel	4/4	3.56E-01 (UOM)	5.1E-03	1.8E-03	2.0E-02	9.1E-02
						Arsenic	4/4	2.62E-03 (Max)	5.1E-03	1.3E-05	3.0E-04	4.5E-02
						Silver	3/4	3.32E-02 (UOM)	5.1E-03	1.7E-04	5.0E-03	3.4E-02
						Antimony	1/3	1.22E-03 (Max)	5.1E-03	6.2E-06	4.0E-04	1.6E-02
						Uranium	2/2	3.20E-03 (Max)	5.1E-03	1.6E-05	3.0E-03	5.5E-03
						Beryllium	4/4	1.50E-03 (Max)	5.1E-03	7.7E-06	5.0E-03	1.5E-03
						TYPE TOTAL						6.9E+00
								mg/kg	1/d	mg/kg/d	mg/kg/d	
					O	Bis(2-ethylhexyl)phthalate	2/4	3.49E-02 (Max)	5.1E-03	1.8E-04	2.0E-02	8.9E-03
						TYPE TOTAL						8.9E-03
						MATRIX TOTAL						2.6E+01

Table B.17 Summation of human health risks by contaminant and exposure pathway for the swimming scenario using water data from ORR monitoring program samples collected near Brashear Island (CRM 9.9) and for the shoreline-use scenario using surface sediment data from the TVA public use areas

Carcinogens

Age=Adult

		Shoreline-use scenario				Swimming scenario		
		TVA public use surface sediment data for LWBR				ORR water monitoring data from Brashear Island location		
Contaminant type	Contaminant	Sediment ingestion	Sediment inhalation	Sediment external exposure	Sediment dermal exposure	Water ingestion while swimming	Water dermal contact while swimming	Multiple pathway total
Inorganics	Chromium		2.8E-06					2.8E-06
	Beryllium	1.4E-06	7.1E-09		3.9E-08			1.4E-06
	Arsenic		9.4E-07					9.4E-07
	TOTAL	1.4E-06	3.7E-06		3.9E-08			5.1E-06
Organics	Bis(2-ethylhexyl)phthalate	3.7E-09			1.0E-09			4.7E-09
	Methylene chloride	7.4E-11			2.1E-11			9.5E-11
	TOTAL	3.7E-09			1.1E-09			4.8E-09
Radionuclides	Cs-137	2.1E-09	3.9E-12	6.3E-07		3.9E-08		6.8E-07
	Uranium, total					3.9E-08		3.9E-08
	Co-60					1.9E-08		1.9E-08
	H-3					8.1E-09		8.1E-09
	Tc-99					1.0E-10		1.0E-10
	TOTAL	2.1E-09	3.9E-12	6.3E-07		1.1E-07		7.4E-07
		1.4E-06	3.7E-06	6.3E-07	4.0E-08	1.1E-07		5.9E-06

Table B.17 (continued)

Noncarcinogens

Age=Adult

		Shoreline-use scenario				Swimming scenario		
		TVA public use surface sediment data for LWBR				ORR water monitoring data from Brashear Island location		
Contaminant type	Contaminant	Sediment ingestion	Sediment inhalation	Sediment external exposure	Sediment dermal exposure	Water ingestion while swimming	Water dermal contact while swimming	Multiple pathway total
Inorganics	Arsenic	5.4E-02			1.5E-03			5.6E-02
	Manganese	1.2E-02	1.6E-02		3.5E-04	1.1E-02	4.9E-03	4.4E-02
	Chromium	1.2E-02			3.3E-04			1.2E-02
	Vanadium	9.8E-03			2.8E-04	1.7E-04	8.0E-05	1.0E-02
	Barium	2.0E-03	1.1E-03		5.8E-05	3.6E-04	1.6E-04	3.7E-03
	Nickel	2.1E-03			6.0E-05			2.2E-03
	Antimony	1.4E-03			4.0E-05			1.4E-03
	Fluoride					1.0E-03		1.0E-03
	Boron	6.5E-04	1.1E-05		1.9E-05			6.8E-04
	Mercury	5.5E-04	2.0E-06		1.6E-05			5.7E-04
	Zinc	3.1E-04			8.9E-06	1.3E-05	6.0E-06	3.4E-04
	Beryllium	1.5E-04			4.2E-06			1.5E-04
	Uranium					1.5E-04		1.5E-04
	Selenium	2.9E-05			8.3E-07			3.0E-05
	Cyanide	1.8E-05			5.1E-07			1.9E-05
	TOTAL	9.5E-02	1.7E-02		2.7E-03	1.2E-02	5.1E-03	1.3E-01
Organics	Bis(2-ethylhexyl)phthalate	3.0E-05			8.7E-06			3.9E-05
	Methylene chloride	3.8E-07	2.9E-11		1.1E-07			4.9E-07
	TOTAL	3.1E-05	2.9E-11		8.8E-06			4.0E-05
		9.5E-02	1.7E-02		2.7E-03	1.2E-02	5.1E-03	1.3E-01

Table B.17 (continued)

Noncarcinogens

Age=Child

Contaminant type	Contaminant	Shoreline-use scenario: TVA public use surface sediment data for LWBR: Sediment ingestion	Sediment inhalation	Sediment external exposure	Sediment dermal exposure	Swimming scenario: ORR water monitoring data from Brashear Island location: Water ingestion while swimming	Water dermal contact while swimming	Multiple pathway total
Inorganics	Arsenic	5.1E-01			2.9E-03			5.1E-01
	Manganese	1.1E-01	1.6E-02		6.5E-04	5.0E-02	9.2E-03	1.9E-01
	Chromium	1.1E-01			6.3E-04			1.1E-01
	Vanadium	9.1E-02			5.2E-04	8.1E-04	1.5E-04	9.3E-02
	Barium	1.9E-02	1.1E-03		1.1E-04	1.7E-03	3.0E-04	2.2E-02
	Nickel	2.0E-02			1.1E-04			2.0E-02
	Antimony	1.3E-02			7.5E-05			1.3E-02
	Boron	6.1E-03	1.1E-05		3.5E-05			6.1E-03
	Mercury	5.1E-03	2.0E-06		2.9E-05			5.2E-03
	Fluoride					4.7E-03		4.7E-03
	Zinc	2.9E-03			1.7E-05	6.1E-05	1.1E-05	3.0E-03
	Beryllium	1.4E-03			7.9E-06			1.4E-03
	Uranium					6.9E-04		6.9E-04
	Selenium	2.7E-04			1.6E-06			2.7E-04
	Cyanide	1.7E-04			9.7E-07			1.7E-04
	TOTAL	8.9E-01	1.7E-02		5.1E-03	5.8E-02	9.7E-03	9.8E-01
Organics	Bis(2-ethylhexyl)phthalate	2.8E-04			1.6E-05			3.0E-04
	Methylene chloride	3.6E-06	2.9E-11		2.0E-07			3.8E-06
	TOTAL	2.9E-04	2.9E-11		1.7E-05			3.0E-04
		8.9E-01	1.7E-02		5.1E-03	5.8E-02	9.7E-03	9.8E-01

Table B.18 Summation of human health risks by contaminant and exposure pathway for the swimming scenario using water data from ORR monitoring program samples collected near Brashear Island (CRM 9.9) and for the shoreline-use scenario using Phase 1 near-shore surface sediment data

Carcinogens

Age=Adult

		Shoreline-use scenario				Swimming scenario		
		Phase 1 near-shore surface sediment data for LWBR				ORR water monitoring data from Brashear Island location		
Contaminant type	Contaminant	Sediment ingestion	Sediment inhalation	Sediment external exposure	Sediment dermal exposure	Water ingestion while swimming	Water dermal contact while swimming	Multiple pathway total
Radionuclides	Cs-137	1.5E-08	2.7E-11	4.4E-06		3.9E-08		4.5E-06
	Co-60	3.8E-10	1.0E-11	3.3E-07		1.9E-08		3.5E-07
	Uranium, total					3.9E-08		3.9E-08
	H-3					8.1E-09		8.1E-09
	Tc-99					1.0E-10		1.0E-10
	TOTAL	1.5E-08	3.8E-11	4.8E-06		1.1E-07		4.9E-06
		1.5E-08	3.8E-11	4.8E-06		1.1E-07		4.9E-06

Table B.18 (continued)

Noncarcinogens

Age=Adult

		Shoreline-use scenario				Swimming scenario		
		Phase 1 near-shore surface sediment data for LWBR				ORR water monitoring data from Brashear Island location		
Contaminant type	Contaminant	Sediment ingestion	Sediment inhalation	Sediment external exposure	Sediment dermal exposure	Water ingestion while swimming	Water dermal contact while swimming	Multiple pathway total
Inorganics	Manganese					1.1E-02	4.9E-03	1.6E-02
	Fluoride					1.0E-03		1.0E-03
	Barium					3.6E-04	1.6E-04	5.2E-04
	Vanadium					1.7E-04	8.0E-05	2.5E-04
	Uranium					1.5E-04		1.5E-04
	Zinc					1.3E-05	6.0E-06	1.9E-05
	TOTAL					1.2E-02	5.1E-03	1.8E-02
						1.2E-02	5.1E-03	1.8E-02

Table B.18 (continued)

Noncarcinogens

Age=Child

Contaminant type	Contaminant	Shoreline-use scenario: Phase 1 near-shore surface sediment data for LWBR				Swimming scenario: ORR water monitoring data from Brashear Island location		
		Sediment ingestion	Sediment inhalation	Sediment external exposure	Sediment dermal exposure	Water ingestion while swimming	Water dermal contact while swimming	Multiple pathway total
Inorganics	Manganese					5.0E-02	9.2E-03	5.9E-02
	Fluoride					4.7E-03		4.7E-03
	Barium					1.7E-03	3.0E-04	2.0E-03
	Vanadium					8.1E-04	1.5E-04	9.6E-04
	Uranium					6.9E-04		6.9E-04
	Zinc					6.1E-05	1.1E-05	7.3E-05
	TOTAL					5.8E-02	9.7E-03	6.8E-02
		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]
		[illegible]	[illegible]	[illegible]	[illegible]	5.8E-02	9.7E-03	6.8E-02
		[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]

Appendix C

FOODCHAIN MODELS AND TRANSFER FACTORS USED IN THE HUMAN HEALTH RISK CHARACTERIZATION

Table C.1. Contaminant concentration in pasture forage

Concentration in irrigated soil [1]:

$$CS_2 = \frac{(CW)\ (I_r)\ (F)\ (1 - e^{-\lambda_B t_b})}{P \lambda_B}$$

Concentration in pasture forage [2]:

$$CP = (CW)\ (I_r) \left[\frac{f_R}{Y} \frac{(1 - e^{-\lambda_E t_p})}{\lambda_E} \right] + (CS_1)\ (B_{v_{dry}}) + (CS_1)\ (s_r)\ (s_d)\ (d) \left[\frac{f_R}{Y} \frac{(1 - e^{-\lambda_E t_p})}{\lambda_E} \right]$$

where

CS_i	=	contaminant concentration in soil [Bq(cont)/kg(soil) or mg(cont)/kg(soil)]; [measured value for dredging scenario (CS_1), calculated value for irrigation scenario(CS_2)];
CW	=	contaminant concentration in water [Bq(cont)/L(water) or mg(cont)/L(water)];
I_r	=	irrigation rate [L(water)/m^2(soil)-day];
F	=	fraction of the year crops are irrigated [dimensionless];
λ_B	=	effective rate constant for removal of contaminant from soil [1/day] $\lambda_B = \lambda_i + \lambda_{HL}$;
λ_i	=	radioactive decay of constant [1/day] $\lambda_i = 0.693/T_R$, where T_R is the radioactive half-life in days;
λ_{HL}	=	rate constant for removal of contaminant from soil by harvesting and leaching [1/day];
t_b	=	time of long-term deposition and build up in soil [days];
P	=	areal density for the effective root zone (15 cm depth) in soil [kg(soil)/m^2(soil) dry mass];
CP	=	contaminant concentration in pasture forage [Bq(cont)/kg(dry forage) or mg(cont)/kg(dry forage)];
f_R	=	interception fraction, the fraction of deposited contaminant intercepted and retained by edible portion of crop [dimensionless];
Y_p	=	standing crop biomass of the edible portion of pasture forage [kg(dry pasture)/m^2];
λ_E	=	effective decay constant for removal of the contaminant deposited on forage [1/day] $\lambda_E = \lambda_i + (0.693/t_w)$;
t_w	=	weathering half-life, the time required for one-half of the originally deposited material to be lost from forage [days];
t_p	=	time period pasture is exposed to above-ground contamination during the growing season [days];
$B_{v\text{-}dry}$	=	soil to pasture forage transfer coefficient for radionuclides and inorganic contaminants [kg(dry forage)/kg(soil, dry wt)];
$B_{v\text{-}organics}$	=	soil to vegetable transfer coefficient for organic contaminants $B_{v\text{-}organics} = 10^{(1.588 - 0.578 \log K_{ow})}$, [kg(vegetable, dry wt)/kg(soil, dry wt)];
s_r	=	soil resuspension factor [1/m(soil)];
s_d	=	areal surface (1 cm depth) soil density [kg(soil)/m^2(soil, dry mass)];
d	=	deposition velocity [m/day].

1 Based on IAEA, *Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases*, Safety Series No. 57, International Atomic Energy Agency, Vienna, 1982.

2 Based on *Remedial Actions Priority Systems (RAPS) Mathematical Formulations*, August 1987, DOE/RL/87-06.

Table C.2. Contaminant concentration in vegetables

Concentration in irrigated soil [1]:

$$CS_2 = \frac{(CW)\ (I_r)\ (F)\ (1 - e^{-\lambda_B t_b})}{P\lambda_B}$$

Concentration in Vegetables [2]:

$$CV = (CW)\ (I_r)\left[\frac{f_R}{Y_v}\frac{(1 - e^{-\lambda_E t_v})}{\lambda_E}\right] + (CS_1)\ (B_{v_{wet}}) + (CS_1)\ (s_r)\ (s_d)\ (d)\left[\frac{f_R}{Y_v}\frac{(1 - e^{-\lambda_E t_v})}{\lambda_E}\right]$$

where

CS_i	=	contaminant concentration in soil [Bq(cont)/kg(soil) or mg(cont)/kg(soil)]; [measured value for dredging scenario (CS_1), calculated value for irrigation scenario(CS_2)];
CW	=	contaminant concentration in water [Bq(cont)/L(water) or mg(cont)/L(water)];
I_r	=	irrigation rate [L(water)/m^2(soil)-day];
F	=	fraction of the year crops are irrigated [dimensionless];
λ_B	=	effective rate constant for removal of contaminant from soil [1/day] $\lambda_B = \lambda_i + \lambda_{HL}$;
λ_i	=	radioactive decay of constant [1/day] $\lambda_i = 0.693/T_R$, where T_R is the radioactive half-life in days;
λ_{HL}	=	rate constant for removal of contaminant from soil by harvesting and leaching [1/day];
t_b	=	time of long-term deposition and build up in soil [days];
P	=	areal density for the effective root zone (15 cm depth) in soil [kg(soil)/m^2(soil) dry mass];
CV	=	contaminant concentration in vegetables [Bq(cont)/kg(vegetable, wet wt) or mg(cont)/kg(vegetable, wet wt)];
f_R	=	interception fraction, the fraction of deposited contaminant intercepted and retained by edible portion of crop [dimensionless];
Y_v	=	standing crop biomass of the edible portion of vegetation [kg(vegetable, wet wt)/m^2];
λ_E	=	effective decay constant for removal of the contaminant deposited on vegetation (1/day) $\lambda_E = \lambda_i + (0.693/t_w)$;
t_w	=	weathering half-life, the time required for one-half of the originally deposited contaminant to be lost from vegetation [days];
t_v	=	time period vegetables are exposed to above-ground contamination during the growing season [days];
$B_{v\text{-}wet}$	=	soil to plant transfer coefficient [kg(vegetable, wet wt)/kg(soil, dry wt)] $B_{v\text{-}wet} = B_{v\text{-}organics} \times 0.25$ (conversion factor from dry wt to wet wt);
$B_{v\text{-}organics}$	=	soil to pasture forage transfer coefficient for organic contaminants[kg(dry forage)/kg(soil, dry wt)];
s_r	=	soil resuspension factor [1/m(soil)];
s_d	=	areal surface (1 cm depth) soil density [kg(soil)/m^2(soil, dry mass)];
d	=	deposition velocity [m/day].

[1] Based on IAEA, *Generic Models and Parameters for Assessing the Environmental Transfer of Radionuclides from Routine Releases*, Safety Series No. 57, International Atomic Energy Agency, Vienna, 1982.

[2] Based on *Remedial Actions Priority Systems (RAPS) Mathematical Formulations*, August 1987, DOE/RL/87-06.

Table C.3. Contaminant concentration in beef

Concentration in beef [1]:

$$CB = F_b \left[(CP \times QP_b) + (CS_1 \times QS_b) + (CW \times QW_b) \right]$$

where

CB	=	concentration of contaminant in beef [mg(cont)/kg(beef)];
F_b	=	transfer coefficient of contaminant in vegetation to contaminant in beef [day/kg(beef)] $\log F_b = -7.6 \log K_{ow}$;
CP	=	contaminant concentration in pasture forage [mg(cont)/kg(forage)];
Q_p	=	quantity of dry forage ingested per day by beef cattle [kg(forage, dry wt)/day];
CS_1	=	contaminant concentration in dry soil [Bq(cont)/kg(soil) or mg(cont)/kg(soil)] (Measured value for dredging scenario);
CS_2	=	contaminant concentration in dry soil as the result of irrigation [Bq(cont)/kg(soil) or mg(cont)/kg(soil)] (Calculated value for irrigation scenario);
Q_s	=	quantity of dry soil ingested per day by beef cattle [kg(soil)/day];
CW	=	contaminant concentration in water [Bq(cont)/L(water) or mg(cont)/L(water)];
Q_w	=	quantity of water ingested per day by beef cattle [L(water)/day].

[1] Based on NCRP, *Screening Techniques for Determining Compliance with Environmental Standards*, National Council on Radiation Protection and Measurements, NCRP Commentary No. 3, Revision of January 1989.

Table C.4. Contaminant concentration in milk

Concentration in milk [1]:

$$CM = F_m [(CP \times QP_d) + (CS_1 \times QS_d) + (CW \times QW_d)]$$

where

CM = concentration of contaminant in milk [mg(cont) or Bq(cont)/L(milk)];
F_m = transfer coefficient of contaminant in vegetation to contaminant in milk [day/L(milk)] $\log F_m = -8.1 \log K_{ow}$;
CP = contaminant concentration in pasture forage [mg(cont)/kg(forage)];
Q_p = quantity of dry forage ingested per day by dairy cattle [kg(forage, dry wt)/day];
CS_1 = contaminant concentration in dry soil [Bq(cont)/kg(soil) or mg(cont)/kg(soil)] (this is a measured value used in the dredging scenario);
CS_2 = contaminant concentration in dry soil as the result of irrigation [Bq(cont)/kg(soil) or mg(cont)/kg(soil)] (this is a modeled value used in the irrigation scenario);
Q_s = quantity of dry soil ingested per day by dairy cattle [kg(soil)/day];
CW = contaminant concentration in water [Bq(cont)/L(water) or mg(cont)/L(water)];
Q_w = quantity of water ingested per day by dairy cattle [L(water)/day].

[1] Based on NCRP, *Screening Techniques for Determining Compliance with Environmental Standards*, National Council on Radiation Protection and Measurements, NCRP Commentary No. 3, Revision of January 1989.

Table C.5 Log K_{ow} and transfer coefficients used in foodchain models

Contaminant	Cas_no	Log Kow	Meat transfer coefficient	Milk transfer coefficient	Pasture transfer coefficient	Vegetable transfer coefficient[a]
4,4'-DDD	72-54-8	6.02E+00	2.63E-02	8.32E-03	1.28E-02	3.21E-03
4,4'-DDE	72-55-9	6.51E+00	8.13E-02	2.57E-02	6.69E-03	1.67E-03
4,4'-DDT	50-29-3	6.38E+00	6.03E-02	1.91E-02	7.95E-03	1.99E-03
Aldrin	309-00-2	3.01E+00	2.57E-05	8.13E-06	7.05E-01	1.76E-01
Am-241	14596-10-2		2.00E-05	4.00E-07	4.00E-03	1.00E-03
Antimony	7440-36-0		1.00E-03	1.00E-04	5.00E-02	1.00E-02
Aroclor-1254	11097-69-1	6.03E+00	2.69E-02	8.51E-03	1.27E-02	3.17E-03
Aroclor-1260	11096-82-5	7.14E+00	3.47E-01	1.10E-01	2.89E-03	7.23E-04
Arsenic	7440-38-2		2.00E-05	4.00E-07	4.00E-03	1.00E-03
Barium	7440-39-3		2.00E-04	4.00E-04	1.00E-01	1.00E-02
Beryllium	7440-41-7		5.00E-03	2.00E-06	2.00E-02	4.00E-03
Bis(2-ethylhexyl)phthalate	117-81-7	4.89E+00	1.95E-03	6.17E-04	5.78E-02	1.44E-02
Boron	7440-42-8		8.00E-04	1.50E-03	4.00E+00	1.00E+00
Cadmium	7440-43-9		1.00E-03	2.00E-03	1.00E+00	5.00E-01
Chlordane	57-74-9	2.78E+00	1.51E-05	4.79E-06	9.58E-01	2.39E-01
Chromium	18540-29-9		3.00E-02	2.00E-03	4.00E-02	1.00E-02
Cm-243,244	13981-15-2		2.00E-05	4.00E-07	4.00E-03	1.00E-03
Cm-245,246	15621-76-8		2.00E-05	4.00E-07	4.00E-03	1.00E-03
Cm-248	15758-33-5		2.00E-05	4.00E-07	4.00E-03	1.00E-03
Co-60	10198-40-0		3.00E-02	2.00E-03	4.00E-01	3.00E-02
Cs-137	10045-97-3		3.00E-02	8.00E-03	1.00E-01	3.00E-02
Cyanide	57-12-5		2.00E-04	1.00E-04	4.00E-03	1.00E-03
Dieldrin	60-57-1	3.87E+00	1.86E-04	5.89E-05	2.24E-01	5.61E-02
Endosulfan I	959-98-8	3.83E+00	1.70E-04	5.37E-05	2.37E-01	5.92E-02
Endrin	72-20-8	5.60E+00	1.00E-02	3.16E-03	2.24E-02	5.61E-03
Fluoride	16984-48-8		NA	NA	NA	NA
Gamma-BHC (Lindane)	58-89-9	3.72E+00	1.32E-04	4.17E-05	2.74E-01	6.85E-02
H-3	10028-17-8		1.00E-02	1.00E-02	2.00E+01	5.00E+00
Manganese	7439-96-5		5.00E-04	8.40E-05	5.30E-01	7.20E-01
Mercury	7439-97-6		1.00E-02	5.00E-04	1.00E+00	3.00E-01
Methylene chloride	75-09-2	1.25E+00	4.47E-07	1.41E-07	7.34E+00	1.83E+00
Nickel	7440-02-0		2.00E-03	1.00E-03	2.00E-01	5.00E-02
Pu-238	13981-16-3		1.00E-05	1.00E-07	1.00E-03	5.00E-04
Pu-239,240	15117-48-3		1.00E-05	1.00E-07	1.00E-03	5.00E-04
Selenium	7782-49-2		1.00E-01	1.00E-02	5.00E-01	1.00E-01
Silver	7440-22-4		5.00E-03	1.00E-02	1.00E+00	2.00E-01
Sr-90	10098-97-2		1.00E-03	1.00E-03	4.00E+00	3.00E-01
Tc-99	14133-76-7		1.00E-03	1.00E-03	4.00E+00	3.00E+01
U-234	13966-29-5		1.00E-02	6.00E-04	1.00E-02	2.00E-03
U-235	15117-96-1		1.00E-02	6.00E-04	1.00E-02	2.00E-03
U-238	7440-61-1		1.00E-02	6.00E-04	1.00E-02	2.00E-03
Uranium	7440-61-1		1.00E-02	6.00E-04	1.00E-02	2.00E-03
Vanadium	7440-62-2		1.00E-02	5.00E-04	1.00E-02	2.00E-03
Zinc	7440-66-6		1.00E-01	1.00E-02	1.00E+00	4.00E-01

[a]Vegetable transfer coefficients for the organic chemicals are calculated values using regression equation log Bv = 1.588 - 0.578 log Kow developed by Travis and Arms (1988)(See Sect. 5.2.3.4).

Appendix D

METHODOLOGY FOR UNCERTAINTY ANALYSES OF SELECT EXPOSURE PATHWAYS IN THE HUMAN HEALTH BASELINE RISK ASSESSMENT

D.1 INTRODUCTION

This appendix focuses on a quantitative uncertainty analysis of the potential detriment associated with the ingestion of ^{137}Cs and selected chemicals as a result of human consumption of fish from LWBR and on the uncertainties for the parameters used in a quantitative uncertainty analysis of the potential detriment associated with ^{137}Cs in deep sediment from LWBR that is used for agricultural purposes. The equations used for assessing the risk of adverse health effects from the ingestion of LWBR fish and from the exposure pathways in the dredging scenario are presented in the following sections of this introduction. The uncertainties involved with the fish ingestion pathway are presented in Section D.2, and the uncertainties involved with the dredging scenario are presented in Section D.3.

D.1.1 Background on USEPA Methods for Risk Assessment for Ingestion of Fish

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

$$LR_{rad} = C \cdot I \cdot ED \cdot EF \cdot DCF \cdot RCF \qquad \text{(D.1)}$$

where,

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

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

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

ED = exposure duration (30 yrs),

EF = exposure frequency (350 days/yr),

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

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

The generic equations used in USEPA methods for determining the risk from chemicals are different for noncarcinogens and carcinogens. For

noncarcinogens a quantity called the Hazard Quotient (HQ) is calculated for each chemical using the following formula.

$$HQ = \frac{C \cdot I \cdot ED \cdot EF}{BM \cdot AT \cdot RfD} \tag{D.2}$$

where,

HQ = hazard quotient (unitless),

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

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

ED = exposure duration (years),

EF = exposure frequency (350 days/year),

BM = body mass (70 kg),

AT = averaging time (days),

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

If the HQ is below 1, then it is highly unlikely that chronic exposure to the chemical would lead to an adverse health effect. If the HQ is greater than unity, remediation of the area/media may be warranted. When more than one hazardous chemical is present, the HQs for all chemicals are summed for each exposure pathway to obtain a Hazard Index (HI) for a given area or exposure scenario (USEPA 1989).

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

$$LR_{chem} = \frac{C \cdot I \cdot ED \cdot EF \cdot SF}{BM \cdot AT} \tag{D.3}$$

where,

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

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

	I	=	estimated intake of the contaminated medium (kg/day),
	ED	=	exposure duration (30 yrs),
	EF	=	exposure frequency (350 days/year),
	SF	=	slope factor (or cancer potency factor) for the chemical of interest [per (mg/kg-day)],
	BM	=	body mass (70 kg),
and	AT	=	averaging time (25,550 days).

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

The USEPA equations yield point estimates of the risk of adverse health effects which obscure the inherent uncertainty in the calculations. If decisions concerning remediation of a contaminated area are based on point estimates, and if the true risk is either grossly overestimated or is associated with high uncertainty, the result may be inappropriate remedial efforts and/or misallocation of limited resources. Therefore, quantitative uncertainty analysis should be performed to quantify the degree of confidence in the estimate of risk. This procedure requires analysis of the uncertainty associated with each parameter in the USEPA risk equations; and, the bounds of uncertainty, expressed as subjective probability distributions, must be estimated for each parameter. The term subjective probability distribution is used because of the application of professional judgment in the absence of relevant site-specific data.

D.1.2 Background for the Risk Assessment of the Dredging Scenario

The following equations were used in the uncertainty analysis of the potential detriment resulting from ^{137}Cs contamination in the deep sediment of LWBR that is dredged and used for agricultural purposes.

External exposure (outdoor):

$$LR_{ext(out)} = C_{sed} \cdot D_l \cdot \left(\frac{1 - e^{-\lambda_{eff} ED}}{\lambda_{eff}} \right) \cdot \rho_{soil} \cdot EF_{ext} \cdot P_{out} \cdot DCF_{ext} \cdot RCF \qquad \text{(D.4)}$$

where, $LR_{ext(out)}$ = excess lifetime cancer risk from radionuclide exposure (unitless),

C_{sed} = concentration of the radionuclide in the dredged sediment (Bq/kg),

λ_{eff} = loss rate from radioactive decay and from harvesting/leaching (yr^{-1}),

ED = exposure duration (years),

D_l = sediment dilution factor (unitless),

ρ_{soil} = soil density (kg/m^3),

EF_{ext} = exposure frequency (unitless),

P_{out} = percent of time spent outdoors (unitless),

DCF_{ext} = radionuclide dependent external dose rate conversion factor for a contaminated soil depth of 15 cm (Sv/yr per Bq/m^3),

and RCF = risk conversion factor (Sv^{-1}).

Ingestion of leafy vegetables:

$$LR_{l,veg} = \left\{ \left[\left(C_{sed} \cdot D_l \right) \cdot \left(\frac{1 - e^{-\lambda_{eff} ED}}{\lambda_{eff}} \right) \cdot B_{v,leafy} \right] \right\} \cdot I_{veg,leafy} \cdot FI_{veg} \cdot EF_{ing} \cdot DCF_{ing} \cdot RCF \tag{D.5}$$

where, LR_{lveg} = excess lifetime cancer risk from leafy-vegetable ingestion (unitless),

C_{sed} = concentration of the radionuclide in the dredged sediment (Bq/kg),

D_l = sediment dilution factor (unitless),

λ_{eff} = loss rate from radioactive decay and from harvesting/leaching (yr^{-1}),

$B_{v,leafy}$ = soil-plant uptake factor (unitless, fresh weight),

$I_{veg,leafy}$ = ingestion of leafy-vegetables (kg/yr, fresh weight)

FI_{veg} = fraction of vegetable ingestion that is from the contaminated site (unitless),

ED = exposure duration (years),

EF_{ing} = exposure frequency (unitless),

DCF_{ing} = radionuclide dependent dose conversion factor (Sv/Bq),

and RCF = risk conversion factor (Sv^{-1}).

Ingestion of non-leafy vegetables:

$$LR_{n,veg} = \left\{ \left[\left(C_{sed} \cdot D_l \right) \cdot \left(\frac{1 - e^{-\lambda_{eff} ED}}{\lambda_{eff}} \right) \cdot B_{v,non-leafy} \right] \right\} \cdot I_{veg,non-leafy} \cdot FI_{veg} \cdot EF_{ing} \cdot DCF_{ing} \cdot RCF \tag{D.6}$$

where, LR_{nveg} = excess lifetime cancer risk from non-leafy vegetable ingestion (unitless),

C_{sed} = concentration of the radionuclide in the dredged sediment (Bq/kg),

D_l = sediment dilution factor (unitless),

λ_{eff} = loss rate from radioactive decay and from harvesting/leaching (yr^{-1}),

$B_{v,non-leafy}$ = soil-plant uptake factor (unitless, fresh weight),

$I_{veg,non-leafy}$ = ingestion of non-leafy vegetables (kg/yr, fresh weight)

FI_{veg} = fraction of vegetable ingestion that is from the contaminated site (unitless),

ED = exposure duration (years),

EF_{ing} = exposure frequency (unitless),

DCF_{ing} = radionuclide dependent dose conversion factor (Sv/Bq),

and RCF = risk conversion factor (Sv^{-1}).

Ingestion of milk:

$$LR_{milk} = \left\{ \left[\left(C_{sed} \cdot D_l \right) \cdot \left(\frac{1 - e^{-\lambda_{eff} ED}}{\lambda_{eff}} \right) \cdot B_{v,past} \right] \cdot Q_m \cdot FI_Q \cdot F_m \right\} \cdot I_{milk} \cdot FI_{milk} \cdot EF_{ing} \cdot DCF_{ing} \cdot RCF \tag{D.7}$$

where, LR_{milk} = excess lifetime cancer risk from milk ingestion (unitless),

C_{sed} = concentration of the radionuclide in the dredged sediment (Bq/kg),

D_l = sediment dilution factor (unitless),

λ_{eff} = loss rate from radioactive decay and from harvesting/leaching (yr^{-1}),

$B_{v,past}$ = soil-plant uptake factor (unitless, dry weight),

Q_m = ingestion of pasture by dairy cow (kg/day, dry weight),

FI_Q = fraction of the pasture ingestion that is from the contaminated source (unitless),

F_m = milk transfer coefficient (d/L),

I_{milk} = ingestion of milk (L/yr)

FI_{milk} = fraction of the milk ingestion that is from the contaminated source (unitless),

ED = exposure duration (years),

EF_{ing} = exposure frequency (unitless),

DCF_{ing} = radionuclide dependent dose conversion factor (Sv/Bq),

and RCF = risk conversion factor (Sv^{-1}).

Ingestion of meat:

$$LR_{meat} = \left\{ \left[\left(C_{sed} \cdot D_l \right) \cdot \left(\frac{1 - e^{-\lambda_{eff} ED}}{\lambda_{eff}} \right) \cdot B_{v,past} \right] \cdot Q_f \cdot FI_Q \cdot F_f \right\} \cdot I_{meat} \cdot FI_{beef} \cdot EF_{ing} \cdot DCF_{ing} \cdot RCF \tag{D.8}$$

where, LR_{meat} = excess lifetime cancer risk from meat ingestion (unitless),

C_{sed} = concentration of the radionuclide in the dredged sediment (Bq/kg),

D_l = sediment dilution factor (unitless),

λ_{eff} = loss rate from radioactive decay and from harvesting/leaching (yr^{-1}),

$B_{v,past}$ = soil-plant uptake factor (unitless, dry weight),

Q_f = ingestion of pasture by beef cow (kg/day, dry weight),

FI_Q = fraction of the pasture ingestion that is from the contaminated source (unitless),

F_f = meat transfer coefficient (d/kg),

I_{meat} = ingestion of meat (kg/yr)

FI_{beef} = fraction of the beef ingestion that is from the contaminated source (unitless),

ED = exposure duration (years),

EF_{ing} = exposure frequency (unitless),

DCF_{ing} = radionuclide dependent dose conversion factor (Sv/Bq),

and RCF = risk conversion factor (Sv^{-1}).

D.2 Description of Parameter Uncertainty in the Fish Consumption Pathway

The subjective probability distributions used for each of the uncertain parameters in the risk assessment equations reflect the degree of belief that the true, but unknown, value for each parameter does not exceed a specified value for that parameter. Where data are limited but uncertainty is relatively low (less than a factor of 10), a range is used to specify a uniform distribution. If there is knowledge about a most likely value or midpoint, in addition to a range, a triangular distribution is assigned. When the range of uncertainty exceeds a factor of 10, a log-uniform or log-triangular distribution is often used. The use of normal, log-normal, or empirical distributions is usually based on fits to available data. Piece-wise uniform or log-uniform distributions can also be used and are especially common when eliciting subjective judgment from a group of experts (IAEA 1989). Other distribution types suitable for Monte Carlo analysis include gamma, beta, Poisson, Weibull, and a variety of discrete distributions (Decisioneering, Inc. 1994; Palisade Corp. 1991).

In this section, the uncertainty associated with the contaminant concentration in fish is discussed first, followed by discussion of exposure factors that are independent within a class (radionuclides, noncarcinogenic chemicals, and carcinogenic chemicals) of contaminants which include the daily intake, the exposure frequency, the exposure duration, the body mass, and the averaging time. The last two topics discussed in this section are the uncertainties involved with radionuclide specific parameters, the dose conversion factors and the risk conversion factor, and with chemical specific parameters, the slope factors and the reference doses. The subjective probability distributions and uncertainty ranges for all of the parameters used in the fish ingestion pathway except the concentrations of the contaminants in fish are summarized in Table D.1.

Table D.1 List of assumptions for the parameters used in the uncertainty analysis of the fish ingestion pathway

Parameter	Distribution	Minimum	Mean	Maximum	Units	Ref. Section
ED , carcinogens	Constant		30		year	D.2.2.3
ED, non-carc.	Constant		1		year	D.2.2.3
EF	Constant		350		unitless	D.2.2.2
I_m	Triangular	0.113	0.230[a]	0.454	kg/fish-meal	D.2.2.1
I_f, non-carc.	Log-Uniform	0.033		0.571	fish-meal/d	D.2.2.1.2
I_f, carc.	Log-Uniform	0.033		0.286	fish-meal/d	D.2.2.1.1
AT, carcinogens	Constant		25,550		days	D.2.2.5
AT, non-carc.	Constant		365		days	D.2.2.5
Body Mass	Constant		70		kg	D.2.2.4
RfD, arsenic	Constant		3.0E-04		mg/kg-day	D.2.4.1
RfD, MeHg	Log-Uniform	3.0E-05	3.0E-04[a]	9.5E-04	mg/kg-day	D.2.4.1.1
RfD, chlordane	Log-Triangular	6.0E-06	6.0E-05[a]	1.9E-03	mg/kg-day	D.2.4.1.2
SF, dieldrin	Constant		16		$(mg/kg\text{-}day)^{-1}$	D.2.4.2
SF, chlordane	Triangular	0	1.3[a]	5	$(mg/kg\text{-}day)^{-1}$	D.2.4.2.2
SF, Aroclor-1260	Triangular	0	7.7[a]	10	$(mg/kg\text{-}day)^{-1}$	D.2.4.2.1
SF, Aroclor-1254	Triangular	0	4.0[a]	10	$(mg/kg\text{-}day)^{-1}$	D.2.4.2.1
DCF_{ing} Cs-137	Log-Triangular	7.0E-09	1.4E-08[a]	2.8E-08	Sv/Bq	D.2.3.1
RCF	Log-Triangular	0.018	0.073[a]	0.292	risk/Sv	D.2.3.2

[a]mode.

D.2.1 Contaminant Concentration in Fish (*C*), Bq/kg

For this uncertainty analysis, only those contaminants where at least one measurement per specie was above detection limits are considered. The uncertainty bounds on the estimated arithmetic mean concentrations were calculated based on the assumption that the data were lognornally distributed. A distribution was obtained that represents the subjective uncertainty of the estimated arithmetic mean concentration. The following equation was used to estimate the uncertainty factor for the arithmetic mean concentrations.

$$UF_{mean} = \frac{UCL_{mean}}{\overline{X}_{est.}} \qquad (D.9)$$

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

UCL_{mean} = the 95% upper confidence limit on the mean assuming a lognormal distribution of data,

and $\overline{X}_{est.}$ = the estimated arithmetic mean concentration.

D.2.1.1 Radionuclide Concentrations

The selected radionuclide for this uncertainty analysis is ^{137}Cs. The values for the estimated arithmetic mean concentration of ^{137}Cs and the UCL on the mean were obtained for LWBR fish and are summarized by species in Table D.2. A uniform distribution was used to represent the uncertainty of the estimated arithmetic mean concentrations. The maximum value for the uniform distribution was set at the 95% UCL on the estimated arithmetic mean, and the minimum value for the distribution was obtained by dividing the estimated arithmetic mean by the uncertainty factor (Eq. D.9). These values are presented in Table D.2. This method of obtaining the uncertainty range on the mean concentration provides high confidence that the true mean will be between the LCL and the UCL.

Table D.2. Summary statistics and estimates of uncertainty for ^{137}Cs concentrations in fish where at least one measurement was above the detection limits

Species	Radionuclide	Units	Ratio: detects to total samples	Estimated Mean	UCL mean	LCL estimate
Bluegill	Cs-137	Bq/kg	3/4	1.2 E+00	1.7 E+00	8.5 E-01
Catfish	Cs-137	Bq/kg	11/19	2.7 E+00	3.4 E+00	2.1 E+00
Largemouth bass	Cs-137	Bq/kg	5/5	7.8 E-01	9.6 E-01	6.3 E-01

D.2.1.2 Chemical Concentrations

The selected chemical contaminants for this uncertainty analysis are mercury, arsenic, chlordane, dieldrin, Aroclor-1254, and Aroclor-1260. Of these chemicals, mercury and arsenic are noncarcinogens, chlordane is both a noncarcinogen and a carcinogen, and dieldrin, Aroclor-1254 and -1260 are carcinogens. A noncarcinogen is defined as a substance for which a toxic reaction results from exceeding a threshold level of exposure. A carcinogen is a substance that has the potential to cause cancer at any level of exposure. Carcinogens are classified by the EPA according to the weight of evidence for their carcinogenicity [EPA *(RAGS Part A.)* 1989]. Radionuclides are classified as "A" which denotes there is ample evidence derived from human epidemiological data. PCBs, dieldrin, and chlordane are classified as "B2" carcinogens which implies sufficient evidence in animals and inadequate or no information from humans (USEPA 1988a).

D.2.1.2.1 Noncarcinogens

The estimated arithmetic mean concentrations and the UCLs and LCLs on the means for mercury, arsenic, and chlordane for LWBR fish are summarized in Table D.3. In the uncertainty analysis, a uniform distribution was used to represent the uncertainty of the estimated arithmetic mean concentrations.

Table D.3. Summary statistics and estimates of uncertainty for concentrations of select noncarcinogens in LWBR fish where at least one measurement was above the detection limits

Species	Contaminant	Units	Ratio: detects to total samples	Estimated Mean	UCL mean	LCL estimate
Bluegill	Arsenic	mg/kg	18/20	6.7 E-02	7.2 E-02	6.2 E-02
	Mercury	mg/kg	20/20	5.5 E-02	6.4 E-02	4.7 E-02
Catfish	Chlordane	mg/kg	40/130	1.2 E-01	1.4 E-01	1.0 E-01
Large mouth bass	Arsenic	mg/kg	5/5	1.4 E-01	1.7 E-01	1.1 E-01
	Chlordane	mg/kg	9/44	5.3 E-02	6.3 E-02	4.5 E-02
	Mercury	mg/kg	5/5	1.9 E-01	2.5 E-01	1.5 E-01

D.2.1.2.2 Carcinogens

The estimated arithmetic mean concentrations and the UCLs and LCLs on the means for Aroclor-1254, Aroclor-1260, dieldrin, and chlordane in LWBR fish are summarized in Table D.4. As with the previously discussed contaminants, a uniform distribution was used to represent the uncertainty in the estimate of the arithmetic mean concentrations except for the dieldrin in catfish which has an uncertainty range greater than one order of magnitude. Due to the large degree of uncertainty in the estimate of the mean concentration, a log-uniform distribution was used in the uncertainty analyses.

Table D.4. Summary statistics and estimates of uncertainty for select carcinogens in LWBR fish where at least one measurement was above the detection limits

Species	Contaminant	Units	Ratio: detects to total samples	Estimated Mean	UCL mean	LCL estimate
Catfish	Aroclor-1254	mg/kg	122/149	6.0 E-01	7.7 E-01	4.7 E-01
	Aroclor-1260	mg/kg	140/151	1.0 E+00	1.2 E+00	8.3 E-01
	Chlordane	mg/kg	40/130	1.2 E-01	1.4 E-01	1.0 E-01
	Dieldrin[a]	mg/kg	3/22	1.2 E-02	1.4 E-01	1.0 E-03
Large mouth bass	Aroclor-1254	mg/kg	44/45	1.7 E-01	2.1 E-01	1.4 E-01
	Aroclor-1260	mg/kg	42/45	1.9 E-01	2.3 E-01	1.6 E-01
	Chlordane	mg/kg	9/44	5.3 E-02	6.3 E-02	4.5 E-02
Striped/hybrid Bass	Aroclor-1254	mg/kg	61/66	4.9 E-01	5.9 E-01	4.1 E-01
	Aroclor-1260	mg/kg	61/66	6.5 E-01	8.1 E-01	5.2 E-01
	Chlordane	mg/kg	14/45	9.5 E-02	1.1 E-01	8.3 E-02

[a]A log-uniform distribution was used for this concentration.

D.2.2 Exposure Factors Independent of the Contaminant Type

The following five exposure factors used in the fish ingestion risk assessment equation which are independent of radionuclide and chemical type: daily fish intake (*I*), exposure frequency (*EF*), exposure duration (*ED*), body mass (*BM*), and averaging time (*AT*). The subjective uncertainty bounds and probability distributions for these parameters were chosen such that the true but unknown value pertaining to the maximally exposed individual lies within the interval. A literature review and discussion with persons

knowledgeable of local fish consumption rates were used to obtain estimates of the uncertainty for each of these parameters.

D.2.2.1 Fish Ingestion (I), kg/day

The fish ingestion rate was defined in the following manner:

$$I = I_m \cdot I_f \tag{D.10}$$

where I_m = the mass of fish consumed per meal (kg/fish meal)

and I_f = the frequency of meals consisting of fish from LWBR (fish meal/day).

A subjective probability distribution was formulated to represent the uncertainty in each individual parameter, i.e., distributions for I_m (same for both carcinogens and noncarcinogens) and I_f (different for carcinogens and noncarcinogens) were determined.

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

After consideration of these surveys, a triangular distribution was used to represent the subjective uncertainty for I_m. For the avid angler, the most likely value (the mode) was estimated at 0.230 kg/fish meal. The Columbia River study was used as a starting point in the consideration of a range of uncertainty because of its direct applicability to the present assessment's target individual (the avid angler). Local risk assessors felt that it was unlikely that an avid angler would eat more than 0.454 kg/fish meal (approximately one pound of fish per meal) or less than 0.113 kg/fish-meal (approximately one-quarter of a pound of fish per meal). Therefore, a maximum value of 0.454 kg/fish meal and a minimum value of 0.113 kg/fish meal were used for the triangular distribution.

D.2.2.1.1 Frequency of Fish Meals for Analysis of Carcinogens

For the frequency of meals (I_f), the uncertainty was based primarily on subjective judgment of local risk assessors. Because each value was equally likely to be the true but unknown number of meals per day, a log-uniform distribution was used to represent the uncertainty. An upper bound value was determined by assuming that an avid angler would not consume more than two meals per week of fish from LWBR over a thirty-year period. This results in a maximum frequency of 0.286 fish meals/day. A lower bound was formulated by assuming that a angler would not consume less than one meal per month of fish from LWBR. This lower value translates to 12 meals/yr, which closely matches the average frequency of 14 meals/yr obtained in the Columbia River survey (Rupp et al. 1980). One meal per month results in a minimum frequency value of 0.033 fish meals/day.

These estimates exclude values for subsistence fishermen who may be obtaining a major portion of their dietary requirements from LWBR fish. At the present time, the actual number of individuals in this category is not likely to be large and they will not be considered in this uncertainty analysis. The target individual in this assessment is an avid angler, not a subsistence angler.

D.2.2.1.2 Frequency of Fish Meals for Analysis of Noncarcinogens

For noncarcinogens, where toxic effects are believed to occur above a threshold exposure, the uncertainty bounds for I_f must take into account the possibility of an avid angler consuming a substantial amount of fish during a one-year period. An upper-bound value was determined by assuming that an avid angler would not consume more than four meals per week of fish from LWBR over a one-year period. A lower bound was determined by assuming that a angler would not consume less than one meal per month. Because of the lack of site specific data, a log-uniform distribution was chosen to represent the subjective uncertainty in I_f.

The uncertainty values for the I_m and the I_f parameters for carcinogens and noncarcinogens are summarized in Table D.5.

Table D.5. Summary information for the uncertainty in the parameters used to calculate daily intake of fish

Class	Parameter	Distribution	Minimum	Maximum	Mode
Both	I_m *(kg/meal)*	Triangular	0.113	0.454	0.230
Carcinogens	I_f *(meal/day)*	Log-Uniform	0.033	0.286	------
Noncarcinogens	I_f *(meal/day)*	Log-Uniform	0.033	0.571	------

D.2.2.2 Exposure Frequency (*EF*), 350 days/year

The *EF* represents the average number of days per year that a person is exposed to the daily intake rate. The USEPA (1991) currently recommends the use of 350 days/year. This value takes into account two weeks of vacation per year during which the individual will not be exposed to the contaminant/pathway being studied. For this uncertainty analysis, the reasonable maximally exposed individual (RME) is defined as an avid angler who is exposed at a frequency of 350 days per year.

D.2.2.3 Exposure Duration (*ED*)

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

For all carcinogens, the USEPA (1989) recommends a value of 30 years for the *ED*. For this uncertainty analysis, the RME is defined as an avid angler who is exposed to carcinogens for 30 years.

For noncarcinogens, immediate toxic effects from over-exposure are of primary concern. These effects typically occur when a person has ingested too much of a contaminated medium. For this uncertainty analysis, the RME is defined as an avid angler who is exposed to noncarcinogens for one year.

D.2.2.4 Body Mass (*BM*), 70 kg

The "average" value for the body mass of Reference Man is 70 kg (ICRP 1975). For this uncertainty analysis, the RME is defined as the reference individual; and, body mass will not be assigned any uncertainty.

D.2.2.5 Averaging Time (*AT*), days

The averaging time is defined as the period of time over which an exposure is averaged. Because it serves as a prorating parameter (USEPA 1989), the values for the averaging time were fixed for this uncertainty analysis. For noncarcinogens the averaging time is equal to the exposure duration. Therefore, *AT* for noncarcinogens is set at 365 days. For carcinogens, where the effect from a single exposure could occur several years after the received dose, the exposure is averaged over a lifetime (70 years). Therefore, *AT* for carcinogens equals 25,550 days.

D.2.3 Radionuclide Specific Factors

D.2.3.1 Dose Conversion Factor (*DCF*), Sv/Bq

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

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

Recently, members of the National Council on Radiation Protection and Measurements (NCRP) scientific committee 57-16 were asked to estimate the uncertainty factors associated with the ingestion and inhalation "dose" factors for a variety of radionuclides (NCRP in press). These uncertainty factors were estimated for the application of ICRP-30 "dose" factors and might not apply to the newly projected factors listed in ICRP-56 (ICRP 1989) or to recent revisions of the ICRP-56 values (ICRP in press). An uncertainty factor pertaining to ^{137}Cs was obtained by expert judgment (Eckerman 1993). The "dose" factor for ^{137}Cs and its associated uncertainty factor from the NCRP and Eckerman are summarized in Table D.6.

Table D.6. Dose conversion factors for the ingestion pathway and corresponding uncertainty factors for ^{137}Cs

Contaminant	*DCF* for Ingestion[a] (Sv/Bq)	Uncertainty Factor[b] A[c]	B[d]
Cs-137	1.4E-08	2	2

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

The uncertainty factor values from both sources are the same for ^{137}Cs. The primary source of uncertainty for ^{137}Cs is inaccuracy in the biokinetic data used to estimate the *DCF*.

A log-triangular distribution was used to represent the subjective uncertainty in the *DCF* value. The *DCF* value itself was used as the mode of the triangle. The maximum value was obtained by multiplying the *DCF* by the UF, and the minimum value by dividing the *DCF* by the UF. The mode, the maximum, and the minimum for ^{137}Cs are summarized below.

Mode = 1.4E-08
Maximum = 2.8E-08
Minimum = 7.0E-09

D.2.3.2 Risk Conversion Factor (*RCF*)

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

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

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

An uncertainty factor of about 2 is recommended for the uncertainty in the *RCF* at high dose and high dose rate exposures, and a UF of greater than or equal to 2 is recommended for the uncertainty involved in the extrapolation to low doses (NCRP 1993a). The possibility of the minimum value for the *RCF* being **zero** cannot be excluded, due to the absence of sufficient data to allow a significant analysis of the risk for exposure at low doses (NRC/NAS 1990, NCRP 1993a). However, for this uncertainty analysis, a minimum value of zero will not be used, on the assumption that the underlying model for the risk associated with radiation exposure is correct and that a threshold dose does not exist.

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

Mode	=	0.073
Maximum	=	0.292
Minimum	=	0.018

D.2.4 Chemical Specific Factors

D.2.4.1 Reference Doses for Noncarcinogens

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

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

obtained instead of a NOAEL (USEPA 1989). Finally, a modifying factor is applied based on best professional judgment.

The following sections will discuss the uncertainties for the methyl-mercury and chlordane *RfD*s. Because sufficient information was not available to evaluated the uncertainty in the RfD for arsenic, it will remain constant throughout the uncertainty analysis at 3.0E-04.

D.2.4.1.1 Methyl-mercury

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

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

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

In a recent draft update on the "Toxicological Profile for Mercury" published by the Agency for Toxic Substances and Disease Registry (ATSDR 1992), an acute Minimum Risk Level (MRL) of 4.0E-05 mg/kg-day is recommended for methyl-mercury. This is an order of magnitude lower than the currently recommended chronic *RfD*.

A log-uniform distribution is used to represent the uncertainty in the methyl-mercury RfD. To protect against more subtle neurological effects and to incorporate the latest recommendation by the ATSDR, a minimum value

was obtained by dividing the *RfD* by the safety factor of 10 (3.0E-05 mg/kg-day), and a maximum value was determined by multiplying the *RfD* by the square root of the safety factor (9.5E-04 mg/kg-day). Dr. Kenneth Reuhl of Rutgers University College of Pharmacy agreed that this was a rational way to determine the uncertainty range for the methyl-mercury RfD (personal communication 1994).

D.2.4.1.2 Chlordane

The *RfD* for chlordane is 6.0E-05 mg/kg-day, and a safety factor of 1000 is given (USEPA 1988b). Only one reported study has significant information to derive an *RfD* for chlordane (USEPA 1988b). The *RfD* was based on an unpublished report by the Velsicol Chemical Corporation (1983) describing the results obtained from a rat study. A LOAEL of 0.055 mg/kg-day, which caused a significant increase in hepatocellular necrosis (liver tumors) in male rats, was used to derive the *RfD* (USEPA 1988b, Velsicol Chemical Corp. 1983).

The Integrated Research Information System (IRIS) which is maintained by the USEPA was consulted to obtain names of experts on the chlordane *RfD*. Two contacts were listed: Dr. William Burnam and Dr. Eric Clegg of the USEPA. Both were contacted to obtain a range of uncertainty for the chlordane *RfD*. When Dr. Burnam was asked his opinion on the bounds of uncertainty for the chlordane *RfD*, he stated that he would have no problem setting the lower limit at the *RfD* and setting the upper limit at the NOEL obtained in the rat study. He said that no effect was seen until an exposure 5 times that of the NOEL was exceeded and that the effect which was observed is a very sensitive indicator (Burnam personal communication 1994). However, Dr. Clegg suggested bounding the *RfD* by an order of magnitude on either side (Clegg personal communication 1994).

Using the limited information pertaining to the chlordane *RfD* and the statements made by two experts listed in IRIS, the following uncertainties were obtained. The *RfD* was multiplied by the square root of 1000 (the safety factor) to obtain a maximum value of 1.9E-03 which is an order of magnitude lower than the NOEL. The *RfD* was divided by a factor of ten based on the conversation with Dr. Clegg to obtain a minimum value of 6.0E-06. The reported *RfD* was used as the most likely value to occur (mode). Therefore, a log-triangular distribution was used to represent the uncertainty in the RfD for chlordane with a mode of 6.0E-05 mg/kg-day, a minimum of 6.0E-06 mg/kg-day, and a maximum of 1.9E-03 mg/kg-day.

D.2.4.2 Slope Factors for Carcinogens

As is assumed for radionuclides, a cancer risk is believed to exist from exposure to a carcinogenic chemical substance no matter how small the level of exposure. This assumption implies the absence of a threshold below which

there would be no risk. The excess lifetime risk of cancer from exposure to carcinogens is evaluated by relating a potential dose to risk. This "conversion" is determined by applying a quantity called the Slope Factor (*SF*).

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

The following sections describe the subjective probability distributions chosen to represent the uncertainty in the *SF*s for select PCBs (Aroclor-1254 and Aroclor-1260) and chlordane. Because of insufficient information, the uncertainty associated with the *SF* for dieldrin was not determined; therefore, the *SF* was fixed at 16 per mg/kg-day throughout the entire uncertainty analysis.

D.2.4.2.1 Polychlorinated Biphenyls

The *SF* for PCBs is reported as 7.7 per (mg/kg-day) (USEPA 1992). This value is the highest *SF* determined from 3 rat studies using Aroclor-1260 (USEPA 1988a). The *SF* for PCBs is based on a rat population that was exposed to a known concentration of a specific Aroclor with a certain outcome. No one knows what form of PCBs the human population is exposed to because PCBs are transformed in the environment (Cogliano 1993). This circumstance creates a large extent of uncertainty. In addition, Aroclor-1260 was chosen to be representative of all of the Aroclors (USEPA 1988a); however, recent studies suggest that this assumption may not be correct (Silberhorn et al. 1990). For the present uncertainty analysis, the *SF*s for Aroclor-1254 and Aroclor-1260 are considered separately.

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

The evaluation of a *SF* for Aroclor-1254 has been performed in only one rat study. The *SF* obtained from this study was 2.6 per (mg/kg-day) (USEPA 1988a). Because this value is sufficiently lower than the reported *SF* for PCBs, a different mode was used for the Aroclor-1254 subjective

probability distribution. In order to provide a conservative estimate for the mode, an uncertainty factor of 1.5 (obtained by dividing the lowest Aroclor-1260 *SF* by the Aroclor-1254 *SF*) was multiplied by the value of 2.6. This results in a value of about 4.0 per (mg/kg-day) for the most likely *SF*. Therefore, a triangular distribution with a mode of 4.0 per (mg/kg-day), a minimum of 0 per (mg/kg-day), and a maximum of 10 per (mg/kg-day) was chosen to describe the uncertainty in the *SF* for Aroclor-1254. Evers et al. (1994) used a "boot strap" method for determining the uncertainty distribution for PCBs and their values fell within the subjective probability distributions used in the present analysis.

D.2.4.2.2 Chlordane

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

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

Because the studies were done for mice and rats exposed to technical chlordane, as with PCBs, the possibility of no carcinogenic effect in humans can not be ignored. A maximum uncertainty estimate of 5 was obtained by rounding the largest *SF* derived from the mice studies. Therefore, a triangular distribution with a mode of 1.3 per (mg/kg-day), a minimum of 0 per (mg/kg-day), and a maximum of 5 per (mg/kg-day) was used to describe the uncertainty in the SF for chlordane. This subjective probability distribution included the uncertainty distribution for chlordane obtained by Evers et al. (1994) using the "boot strap" method.

D.3 Description of Parameter Uncertainty in the Dredging Scenario

In this section, the uncertainty associated with the dredging scenario is discussed. This scenario includes the possibility that at some time in the future deep sediments from LWBR which are contaminated with ^{137}Cs could be dredged and placed on land where they are used for agricultural purposes. The following exposure pathways are included in this scenario: 1) outdoor external exposure to the sediment, 2) ingestion of leafy vegetables grown on the contaminated soil, 3) ingestion of non-leafy vegetables grown on the contaminated soil, 4) ingestion of milk from a dairy cow which has eaten

pasture grown on the contaminated soil, and 5) ingestion of meat from a beef cow which has eaten pasture grown on the contaminated soil.

The subjective probability distributions used for each of the uncertain parameters in the risk assessment equations reflect the degree of belief that the true but unknown value for each parameter does not exceed a specified value for that parameter. Uncertainties for many parameters presented in Equations D.4 through D.8 have been previously researched (Hoffman and Baes 1979, Hoffman et al. 1982). The distributions that have been recommended in the past have been adopted for this analysis if they were consistent with the findings obtained from an independent review of various reported and/or default values provided in various publications.

The uncertainty associated with the ^{137}Cs concentration in the deep sediment is discussed first, followed by discussion of exposure factors that are independent of contaminant type which include intake rate, exposure frequency, exposure duration, percent of time outdoors, sediment dilution factor, fraction ingested, and sediment/soil density. The last topics discussed in this section are the uncertainties involved with radionuclide specific parameters which include soil/plant transfer factors, milk and meat transfer factors, loss rates by decay and harvesting/leaching, dose conversion factors, and risk conversion factor. The subjective probability distributions and uncertainty ranges for all parameters used in the dredging scenario are summarized in Table D.7.

Table D.7. List of assumptions for the parameters used in the uncertainty analysis of the dredging scenario

Parameter	Distribution	Minimum	Mean	Maximum	Units	Report Section
C_{sed}	Uniform	240		500	Bq/kg	D.3.1
λ_{eff}	Log-Triangular	2.3E-02	2.4E-02[a]	3.2E-01	yr^{-1}	D.3.3
r_{soil}	Triangular	1100	1300[a]	1600	kg/m^3	D.3.2.7
ED	Constant		30		year	D.3.2.3
EF_{ext}	Uniform	0.23		0.96	unitless	D.3.2.2
EF_{ing}	Constant		0.96		unitless	D.3.2.2
P_{out}	Uniform	0.25		0.60	unitless	D.3.2.4
D_l	Uniform	0.30		1.00	unitless	D.3.2.5
B_v, pasture	Log-Triangular	7.0 E-04	4.4 E-02[a]	1.2 E+00	unitless	D.3.3
B_v, veg. leafy	Log-Triangular	1.0 E-04	5.5 E-03[a]	8.0 E-02	unitless	D.3.3
B_v, veg non-leafy	Log-Triangular	1.0 E-05	5.3 E-03[a]	1.0 E-01	unitless	D.3.3
F_m	Log-Triangular	3.7 E-03	6.7E-03[a]	5.0 E-02	d/L	D.3.3
F_f	Log-Triangular	3 E-03	2.1E-02[a]	2.0 E-01	d/kg	D.3.3
Q_m	Uniform	11		21	kg/d	D.3.2.1
Q_f	Uniform	8		16	kg/d	D.3.2.1
FI_Q	Uniform	0.5		0.9	unitless	D.3.2.6
$I_{veg, leafy}$	Uniform	10		85	kg/yr	D.3.2.1
$I_{veg, non-leafy}$	Uniform	40		165	kg/yr	D.3.2.1
I_{milk}	Uniform	80		400	L/yr	D.3.2.1
I_{meat}	Uniform	40		125	kg/yr	D.3.2.1
FI_{veg}	Uniform	0.3		1.0	unitless	D.3.2.6
FI_{milk}	Uniform	0.5		1.0	unitless	D.3.2.6
FI_{meat}	Uniform	0.5		1.0	unitless	D.3.2.6
DCF_{ing} Cs-137	Log-Triangular	7.0 E-09	1.4 E-08[a]	2.8 E-08	Sv/Bq	D.2.3.1
DCF_{ext} Cs-137	Log-Triangular	2.6 E-10	5.1 E-10[a]	1.0 E-09	Sv/yr per Bq/m^3	D.3.3
RCF	Log-Triangular	0.018	0.073[a]	0.292	risk/Sv	D.2.3.2

[a]mode.

D.3.1 Concentration of ^{137}Cs in Deep Sediment

Sixty cores of deep sediment were obtained from LWBR. Each core was split into segments, and the sample obtained from each segment was tested to determine the amount of ^{137}Cs in the sediment. Cs-137 was found in one or

more segments of each core. The segment concentrations were averaged to obtain an average concentration for each core. Using lognormal statistics, an arithmetic mean was estimated by averaging the 60 cores. The estimated mean and 95% UCL on the mean obtained from this analysis were 346 Bq/kg and 500 Bq/kg, respectively.

A uniform distribution was used to represent the uncertainty of the estimated arithmetic mean concentration. The upper bound of the uniform distribution was set at the 95% UCL on the arithmetic mean, and the lower bound of the distribution was obtained by dividing the estimated arithmetic mean by the uncertainty factor obtained from Equation D.9 (240 Bq/kg). The range obtained by this method (240 - 500 Bq/kg) was compared to the individual core averages to assure the uncertainty had not been underestimated. From the individual core data, it appeared that the proposed uncertainty range was adequate.

D.3.2 Exposure Factors Independent of Contaminant Type

The following element independent exposure factors are used in the dredging scenario risk assessment models: intake rate, exposure frequency, exposure duration, percent of time outdoors, sediment dilution factor, fraction ingested, and sediment/soil density. The subjective uncertainty bounds and probability distributions for these parameters were chosen such that the true but unknown value pertaining to the maximally exposed individual lies within the interval. The uncertainty for each of these parameters was based on a review of literature and moderate knowledge of local customs by the risk assessors.

D.3.2.1 Intake Rates

Various ingestion rates must be considered for the agricultural pathways of the dredging scenario. The rate of pasture ingestion by dairy and beef cows must be considered for the grass-cow-milk-human and grass-cow-meat-human pathways. In addition, the amount of leafy vegetables, non-leafy vegetables, milk, and meat that a RME will eat or drink must be evaluated. These parameters are discussed below.

<u>Intake of pasture by dairy (Q_m) and beef cows (Q_f)</u>

The quantity of pasture that a cow eats is determined on a dry weight basis. Several studies have discussed the uncertainties with estimates of the quantity of pasture ingested by dairy and beef cows (Hoffman and Baes 1979, Hoffman et al. 1982, Shor and Fields 1980). A uniform distribution was used to represent the uncertainty in the ingestion rate of dairy cows and beef cows. The upper limit and lower limit of the uniform distributions were obtained by using 16 kg/d and 12 kg/d as a midpoint for the dairy cow and the beef cow,

respectively (IAEA 1982), and adding and subtracting two times the standard deviation, 2.6 kg/d (dairy) and 2.0 kg/d (beef) (Hoffman et al. 1982). The uncertainty ranges are summarized in Table D.7.

Intake of various food products by humans ($I_{veg,leafy}$, $I_{veg,non-leafy}$, I_{milk}, and I_{meat})

The uncertainty for the intake of various food products by humans was based primarily on the subjective judgment of various risk assessors. Several publications were reviewed (Hoffman and Baes 1979; Hoffman et al. 1982; ICRP 1975; NCRP 1984, 1989; Rupp 1979; USEPA 1991), and the subjective probability distributions derived for each of the human intake parameters are consistent with the values reported in the publications. Because each value is equally likely to be the true but unknown rate of intake, a uniform distribution was used to represent the uncertainty. The lower and upper limits used for the uniform distributions are values that target the RME individual. Because it is highly unlikely that the upper bounds for each rate of intake will be sampled simultaneously and because the RME should provide a conservative estimate of exposure, no correction was made for the caloric intake of an individual.

The lower bound for the ingestion of leafy vegetables was determined by assuming that an individual would not consume less than 10 kg fresh weight/yr (approximately 1 ounce of leafy vegetables every day of the year). The upper bound for the ingestion of leafy vegetables was determined by assuming that an individual who if fond of vegetables would not consume more than 85 kg/yr (approximately 8 ounces of leafy vegetables every day of the year). These values are consistent with survey ranges reported in Rupp (1979).

The lower bound for the ingestion of non-leafy vegetables was determined by assuming that an individual would not consume less than 40 kg/yr (approximately 4 ounces of non-leafy vegetables every day of the year). The upper bound for the ingestion of non-leafy vegetables was determined by assuming that an individual who is fond of non-leafy vegetables would not consume more than 165 kg/yr (approximately 16 ounces of non-leafy vegetables every day of the year). These values are consistent with survey ranges reported in Rupp (1979).

The lower bound for the ingestion of milk was determined by assuming that an avid adult male milk drinker would not consume less than 80 L/yr (approximately 1 cup of milk every day of the year). The upper bound for the ingestion of milk was determined by assuming that an avid milk drinker would not consume more than 400 L/yr (more than 1 liter of milk every day of the year). These values are consistent with survey ranges reported in Rupp (1979).

The lower bound for the ingestion of beef was determined by assuming that an avid meat eater would not consume less than 40 kg/yr (approximately 4 ounces of beef every day of the year). The upper bound for the ingestion of beef was determined by assuming that an avid meat eater would not consume more than 125 kg/yr (approximately 12 ounces of beef every day of the year). These values are consistent with survey ranges reported in Rupp (1979).

D.3.2.2 Exposure Frequency (*EF*)

The *EF* represents the frequency at which the RME is exposed to the given situation. The exposure frequency is different for the external exposure pathway and the agricultural pathways.

For the agricultural pathways, the USEPA (1991) currently recommends the use of 350 days/year. This value takes into account two weeks of vacation per year during which time the individual will not be exposed to the contaminant/pathway being studied. For this uncertainty analysis, the reasonable maximally exposed individual (RME) is defined as an individual who is exposed at a frequency of 0.96 (EF_{ing}, 350 days per year).

For the external exposure pathway, the number of hours that the RME may spend outdoors on his property must be considered. In consultation with several risk assessors, a uniform distribution ranging between 0.23 and 0.96 was chosen to represent the external exposure frequency (EF_{ext}). The lower limit was determined by rationalizing that an individual would spend at least 6 hours per day for 350 days each year on his property. The most time that an individual would conceivable stay on his property was determined to be 8400 hours (24 hours a day for 350 days of the year). Therefore, there is high confidence that the true exposure frequency for the RME is bounded by this uncertainty range.

D.3.2.3 Exposure Duration (*ED*)

The exposure duration *(ED)* represents the number of years that an individual will be exposed to the given exposure conditions, i.e., an estimate of the projected length of time that a person will live in the area under consideration. For all carcinogens, USEPA (1989) recommends a value of 30 years for the *ED*. For this uncertainty analysis, the RME is defined as an individual who is exposed to the ^{137}Cs contaminated sediment on his farm for thirty years.

D.3.2.4 Percent of Time Spent Outdoors (P_{out})

In the dredging scenario, it was assumed that an individual living on the farm will be externally exposed to radiation from the sediment only while outdoors. The risk assessors decided that an individual working on a local farm would not likely stay outside more than 60 percent of the time over a one-year period (approximately 14.5 hours per day) and would not likely stay outside less than 25 percent of the time over a one-year period (approximately 6 hours per day). Therefore, a uniform distribution with a minimum value of 0.25 and a maximum value of 0.6 was used to represent the uncertainty in P_{out}.

D.3.2.5 Sediment Dilution Factor (D_I)

Because it is unlikely that an individual would be able to dredge enough sediment such that mixing with topsoil would not occur, a sediment dilution factor was incorporated into the risk assessment equations for the purpose of the uncertainty analysis. Sediment mixing is most likely to occur in the top six inches of the soil. If two inches of dredged sediment were put on the land and mixed with the top soil, the concentration in the sediment would be diluted to one-third its original concentration. Therefore, a maximum dilution factor of 0.3 was assumed. To account for the most conservative situation, it was assumed that the possibility existed that no dilution of the sediment concentration would occur. Therefore, a uniform distribution with a maximum of 0.3 and minimum of 1.0 was used to represent the uncertainty in the dilution factor.

D.3.2.6 Fraction Ingested (FI)

Because it is unlikely that a cow will not be fed supplements, the fraction of food eaten by a cow that is represented by the contaminated pasture (FI_Q) is included in the equations for the grass-cow-milk-human and grass-cow-meat-human exposure pathways. It is unlikely that FI_Q will exceed 90% or will be below 50%. Because this is based primarily on subjective judgment and the true value is equally likely between 50 and 90%, a uniform distribution with a minimum value of 0.5 and a maximum value of 0.9 was used to represent the uncertainty associated with FI_Q.

It is unlikely that all the vegetables, milk, and meat eaten by the RME individual who farms the contaminated land will be produced on the farm; however, this possibility can not be excluded (FI = 1). It was decided during a meeting of risk assessors that the exposed individual would not obtain less than 50% of his milk and meat from cows that graze on the contaminated pasture or less than 30% of his vegetables from produce grown on the contaminated sediment. Because each value is equally likely to be the true

fraction ingested, a uniform distribution was used to represent the uncertainty in the FIs. These uncertainties are summarized in Table D.7.

D.3.2.7 Soil Density (r_{soil})

The density of soil ranges from 1100 to 1600 kg/m^3 (Hillel 1980). Soils consisting of clay are around 1100 kg/m^3, and sandy soils have a density around 1600 kg/m^3. Assuming that deep sediments will be mixed with topsoil, a density of 1300 kg/m^3 was chosen as a most likely value. A triangular distribution with a minimum value of 1100, a mode of 1300, and a maximum value of 1600 kg/m^3 was used to describe the uncertainty in the soil density.

D.3.3 Contaminant Dependent Factors

Uncertainties involved with radionuclide specific parameters such as the loss rate from radioactive decay and from harvesting/leaching, the dose conversion factors, and the risk conversion factor are presented in this section. In addition, uncertainties pertaining to element specific parameters such as the soil/plant transfer factors and the milk/meat transfer factors are discussed in this section.

<u>Cs-137 specific parameters</u>

The uncertainties associated with the dose conversion factor for internal exposure to ^{137}Cs and the risk conversion factor are discussed in Sections D.2.3.1 and D.2.3.2, respectively. The subjective probability distributions recommended for the consumption of fish scenario will also be adopted in the agricultural pathways of the dredging scenario.

The dose conversion factor for external exposure (DCF_{ext}) to ^{137}Cs was obtained from Eckerman and Ryman (1993). A contaminated soil depth of 15 cm was assumed for this analysis. Because ^{137m}Ba is a short-lived daughter of ^{137}Cs, the DCF_{ext} for ^{137m}Ba must be added to the DCF_{ext} for ^{137}Cs. Cs-137 decays to ^{137m}Ba 95% of the time; therefore, the DCF_{ext} for ^{137m}Ba is multiplied by 95% before it is added to the ^{137}Cs DCF_{ext}. The value obtained for the DCF_{ext} is 5.1E-10 Sv/yr per Bq/m^3. The uncertainties with the scatter cross-sections for reference man and with the application of reference man to the RME are the two major contributions to the overall uncertainty in the DCF_{ext} (Leggett 1994). Because it is unlikely that the uncertainty in the DCF_{ext} is greater than the uncertainty in the internal DCF, an uncertainty factor of 2 is used to bound the range of uncertainty in DCF_{ext}. A log-triangular distribution with a mode of 5.1E-10, a lower limit of 2.6E-10 and an upper limit of 1.0E-09 was used to represent the uncertainty associated with the external DCF.

The uncertainties associated with the loss rate from decay and from harvesting/leaching (l_{eff}) result from the uncertainty in the loss rate from harvesting/leaching. The loss rate from radiological decay is determined by dividing the natural log of 2 by the half-life of the radionuclide (^{137}Cs: 0.693/30 yrs = 2.3E-02 yr^{-1}). The uncertainty estimates for l_{eff} were determined by adding the radiological decay rate to the minimum, mean, and maximum value reported for the loss rate from harvesting and leaching in Hoffman et al. (1982). These values are summarized in Table D.7.

Element dependent parameters

For the agricultural pathways considered in the dredging scenario several transfer factors are required, such as 1) the plant/soil concentration factors (B_v) for three different food crops (leafy and non-leafy vegetables for human consumption and pasture forage for cow consumption) and 2) the animal product transfer factors [cow's milk (F_m) and meat (F_f)]. For this analysis, all vegetable transfer factors are defined in terms of the wet weight of plants, while the pasture forage is given as dry weight.

The primary reference for the uncertainties in the plant-to-soil transfer factors and the animal product transfer factors is a document that was prepared for the U.S. Nuclear Regulatory Commission (Hoffman et al. 1982). The selection of values for the plant/soil transfer factor and animal product transfer factor for ^{137}Cs was further supported by an intensive research of numerous publications (Kennedy and Strenge 1993, NCRP 1989, Ng 1982, Ng et al. 1979a, Ng et al. 1979b). From the reported values, an average, minimum, and maximum reported value were compared to the uncertainty estimates documented in Hoffman et al. (1982). Because these values were similar, the uncertainty estimates presented in Hoffman et al. (1982) were used in this uncertainty analysis. These values are summarized in Table D.8.

Table D.8. Ranges of plant-to-soil concentration ratios and transfer coefficients to cow's milk and beef[a]

	Units	Minimum	Estimated mean	Maximum
Bv (leafy veg.)	unitless	1.0 E-04	5.5 E-03	8.0 E-02
Bv (non-leafy veg.)	unitless	1.0 E-05	5.0 E-03	1.0 E-01
Bv (pasture, dry wt.)	unitless	7.0 E-04	4.4 E-02	1.2 E+00
Ff (meat)	d/kg	3.0 E-03	2.1 E-02	2.0 E-01
Fm (milk)	d/L	3.7 E-03	6.7 E-03	5.0 E-02

[a]Ranges taken from Hoffman et al. (1982).

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Appendix E

DATA TABLES FOR ECOLOGICAL RISK ASSESSMENT

TABLE E.1 Upper confidence limit (UCL) concentration of water samples (ug/L) from CRRI samples, toxicological benchmarks(ug/L), and hazard quotients for aquatic life.

Analysis type	Analyte	Reach	Total observations	Number of Detects	UCL Conc.	UCL Quotient	Benchmark	Benchmark Type
I	Cadmium	4	3	1	0.34	0.0792	4.30	EC20_POP
		4	3	1	0.34	2.2711	0.15	LCV_DAPH
		4	3	1	0.34	0.2004	1.70	LCV_FISH
		4	3	1	0.34	0.4542	0.75	LTV_DAPH
		4	3	1	0.34	0.1893	1.80	LTV_FISH
		4	3	1	0.34	0.0874	3.90	NAWQ_ACU
		4	3	1	0.34	0.3097	1.10	NAWQ_CHR
		4	3	1	0.34	26.2051	0.01	SS_EC20
I	Copper	2	2	1	12.08	1.4046	8.60	EC20_POP
		2	2	1	12.08	52.5193	0.23	LCV_DAPH
		2	2	1	12.08	3.1189	3.87	LCV_FISH
		2	2	1	12.08	58.9241	0.21	LTV_DAPH
		2	2	1	12.08	2.4159	5.00	LTV_FISH
		2	2	1	12.08	0.6711	18.00	NAWQ_ACU
		2	2	1	12.08	1.0066	12.00	NAWQ_CHR
		2	2	1	12.08	46.4594	0.26	SS_EC20
I	Iron	2	2	2	229.71	1.4538	158.00	LCV_DAPH
		2	2	1	182.59	1.1557	158.00	LCV_DAPH
		2	2	1	182.59	11.4120	16.00	LTV_DAPH
		2	2	2	229.71	14.3566	16.00	LTV_DAPH
		2	2	1	182.59	0.1826	1000.00	NAWQ_CHR
		2	2	2	229.71	0.2297	1000.00	NAWQ_CHR
		4	2	2	260.51	1.6488	158.00	LCV_DAPH
		4	2	2	260.51	16.2819	16.00	LTV_DAPH
		4	2	2	260.51	0.2605	1000.00	NAWQ_CHR
		5	3	1	34.00	0.2152	158.00	LCV_DAPH
		5	4	4	526.07	3.3295	158.00	LCV_DAPH
		5	3	1	34.00	2.1248	16.00	LTV_DAPH
		5	4	4	526.07	32.8792	16.00	LTV_DAPH
		5	3	1	34.00	0.0340	1000.00	NAWQ_CHR
		5	4	4	526.07	0.5261	1000.00	NAWQ_CHR
I	Lead	4	3	1	1.76	0.0248	71.00	EC20_POP
		4	3	1	1.75	0.0246	71.00	EC20_POP
		4	3	1	1.75	0.1424	12.26	LCV_DAPH
		4	3	1	1.76	0.1438	12.26	LCV_DAPH
		4	3	1	1.75	0.0925	18.88	LCV_FISH
		4	3	1	1.76	0.0934	18.88	LCV_FISH
		4	3	1	1.76	0.0802	22.00	LTV_FISH
		4	3	1	1.75	0.0794	22.00	LTV_FISH
		4	3	1	1.75	0.0213	82.00	NAWQ_ACU
		4	3	1	1.76	0.0215	82.00	NAWQ_ACU
		4	3	1	1.75	0.5456	3.20	NAWQ_CHR
		4	3	1	1.76	0.5511	3.20	NAWQ_CHR
		4	3	1	1.75	4.9887	0.35	SS_EC20
		4	3	1	1.76	5.0382	0.35	SS_EC20

I	Sodium	2	2	2	6113.04	0.0719	85049.00	LCV_DAPH
		2	2	2	6389.03	0.0751	85049.00	LCV_DAPH
		4	2	2	8044.82	0.0946	85049.00	LCV_DAPH
		4	2	2	7965.41	0.0937	85049.00	LCV_DAPH
		5	4	4	5123.99	0.0602	85049.00	LCV_DAPH
		5	4	4	5114.21	0.0601	85049.00	LCV_DAPH
		10	2	2	5378.83	0.0632	85049.00	LCV_DAPH
		10	2	2	5375.39	0.0632	85049.00	LCV_DAPH
I	Zinc	2	2	1	167.84	2.0980	80.00	EC20_POP
		2	2	1	167.84	3.5917	46.73	LCV_DAPH
		2	2	1	167.84	4.6097	36.41	LCV_FISH
		2	2	1	167.84	3.5711	47.00	LTV_FISH
		2	2	1	167.84	1.3987	120.00	NAWQ_ACU
		2	2	1	167.84	1.5258	110.00	NAWQ_CHR
		2	2	1	167.84	7.9924	21.00	SS_EC20
		10	2	1	55.81	0.6976	80.00	EC20_POP
		10	2	1	55.81	1.1942	46.73	LCV_DAPH
		10	2	1	55.81	1.5327	36.41	LCV_FISH
		10	2	1	55.81	1.1873	47.00	LTV_FISH
		10	2	1	55.81	0.4650	120.00	NAWQ_ACU
		10	2	1	55.81	0.5073	110.00	NAWQ_CHR
		10	2	1	55.81	2.6574	21.00	SS_EC20
O	Bis(2-ethylhexyl)phthalate	10	2	1	23.28	0.4657	50.00	EC20_POP
		10	2	1	23.28	7.7615	3.00	LCV_DAPH
		10	2	1	23.28	2.7719	8.40	LCV_FISH
		10	2	1	23.28	7.7615	3.00	LTV_DAPH
		10	2	1	23.28	0.4312	54.00	LTV_FISH

TABLE E.2. Upper confidence limit (UCL) concentration of water 236 Oak Ridge Reservation Environmental Monitoring water samples (ug/L), toxicological benchmarks (ug/L), and quotients for aquatic life.

Analysis type	Analyte	Reach	Total observations	Number of Detects	UCL Conc.	UCL Quotient	Benchmark	Benchmark Type
I	Aluminum	4	4	4	547.78	0.2883	1900.00	LCV_DAPH
		4	4	4	547.78	0.0944	5800.00	LCV_FISH
		4	4	4	547.78	1.0144	540.00	LTV_DAPH
		4	4	4	547.78	0.1165	4700.00	LTV_FISH
		4	4	4	547.78	0.7304	750.00	NAWQ_ACU
		4	4	4	547.78	6.2963	87.00	NAWQ_CHR
		4	4	4	547.78	7.3037	75.00	SS_EC20
I	Ammonia	4	4	4	88.82	0.1410	630.00	LCV_DAPH
		4	4	4	88.82	52.2486	1.70	LCV_FISH
		4	4	4	88.82	0.0370	2400.00	NAWQ_ACU
		4	4	4	88.82	0.2692	330.00	NAWQ_CHR
I	Barium	4	4	4	36.22	0.0018	20336.00	LCV_DAPH
I	Copper	4	7	1	7.02	0.8168	8.60	EC20_POP
		4	7	1	7.02	30.5424	0.23	LCV_DAPH
		4	7	1	7.02	1.8138	3.87	LCV_FISH
		4	7	1	7.02	34.2671	0.21	LTV_DAPH
		4	7	1	7.02	1.4050	5.00	LTV_FISH
		4	7	1	7.02	0.3903	18.00	NAWQ_ACU
		4	7	1	7.02	0.5854	12.00	NAWQ_CHR
		4	7	1	7.02	27.0183	0.26	SS_EC20
I	Iron	4	4	3	550.46	3.4839	158.00	LCV_DAPH
		4	4	3	550.46	34.4040	16.00	LTV_DAPH
		4	4	3	550.46	0.5505	1000.00	NAWQ_CHR
I	Manganese	4	7	7	81.10	0.7241	112.00	EC20_POP
		4	7	7	81.10	0.0737	1100.00	LCV_DAPH
		4	7	7	81.10	0.0458	1770.00	LCV_FISH
		4	7	7	81.10	0.0737	1100.00	LTV_DAPH
		4	7	7	81.10	0.0639	1270.00	LTV_FISH
I	Sodium	4	7	7	4636.79	0.0545	85049.00	LCV_DAPH
I	Uranium	4	6	2	1.08	0.0399	27.00	EC20_POP
		4	6	2	1.08	0.0076	142.00	LCV_FISH
		4	6	2	1.08	0.0024	455.00	LTV_FISH
I	Vanadium	4	4	1	2.00	0.0625	32.00	EC20_POP
		4	4	1	2.00	0.0021	940.00	LCV_DAPH
		4	4	1	2.00	0.0250	80.00	LCV_FISH
		4	4	1	2.00	0.0047	430.00	LTV_DAPH
		4	4	1	2.00	0.0488	41.00	LTV_FISH
I	Zinc	4	7	2	11.78	0.1473	80.00	EC20_POP
		4	7	2	11.78	0.2522	46.73	LCV_DAPH
		4	7	2	11.78	0.3236	36.41	LCV_FISH

4	7	2	11.78	0.2507	47.00	LTV_FISH
4	7	2	11.78	0.0982	120.00	NAWQ_ACU
4	7	2	11.78	0.1071	110.00	NAWQ_CHR
4	7	2	11.78	0.5611	21.00	SS_EC20

TABLE E.3. Detected contaminants in sediment, maximum sediment concentrations (mg/kg), sediment screening benchmarks (mg/kg), and hazard quotients. Concentratrations are from the CRRI samples, the Olson study, and the Oak Ridge Task Force study.

Analysis type	Analyte	RIVER SYSTEM POSITION	Total observations	Number of Detects	MAX Sediment Conc.	Benchmark	Benchmark Type	MAX Quotient
I	Antimony	2	1	1	0.200	2.000000	ER_L	0.100
		4	2	1	0.600	2.000000	ER_L	0.300
		5	4	2	1.500	2.000000	ER_L	0.750
		2	1	1	0.200	25.000000	ER_M	0.008
		4	2	1	0.600	25.000000	ER_M	0.024
		5	4	2	1.500	25.000000	ER_M	0.060
I	Arsenic	2	5	5	9.200	33.000000	ER_L	0.279
		4	5	5	17.000	33.000000	ER_L	0.515
		5	5	5	14.000	33.000000	ER_L	0.424
		10	3	3	26.000	33.000000	ER_L	0.788
		2	5	5	9.200	85.000000	ER_M	0.108
		4	5	5	17.000	85.000000	ER_M	0.200
		5	5	5	14.000	85.000000	ER_M	0.165
		10	3	3	26.000	85.000000	ER_M	0.306
		2	5	5	9.200	6.000000	LEL_MOE	1.533
		4	5	5	17.000	6.000000	LEL_MOE	2.833
		5	5	5	14.000	6.000000	LEL_MOE	2.333
		10	3	3	26.000	6.000000	LEL_MOE	4.333
I	Cadmium	2	5	4	4.200	5.000000	ER_L	0.840
		4	5	4	1.600	5.000000	ER_L	0.320
		5	5	5	8.500	5.000000	ER_L	1.700
		2	5	4	4.200	9.000000	ER_M	0.467
		4	5	4	1.600	9.000000	ER_M	0.178
		5	5	5	8.500	9.000000	ER_M	0.944
		2	5	4	4.200	0.600000	LEL_MOE	7.000
		4	5	4	1.600	0.600000	LEL_MOE	2.667
		5	5	5	8.500	0.600000	LEL_MOE	14.167
I	Chromium (6+)	2	5	5	18.000	80.000000	ER_L	0.225
		4	5	5	53.000	80.000000	ER_L	0.663
		5	5	5	48.000	80.000000	ER_L	0.600
		10	3	3	31.900	80.000000	ER_L	0.399
		2	5	5	18.000	145.000000	ER_M	0.124
		4	5	5	53.000	145.000000	ER_M	0.366
		5	5	5	48.000	145.000000	ER_M	0.331
		10	3	3	31.900	145.000000	ER_M	0.220
I	Copper	2	3	3	13.000	70.000000	ER_L	0.186
		4	3	3	41.700	70.000000	ER_L	0.596
		5	5	5	56.000	70.000000	ER_L	0.800
		10	1	1	61.500	70.000000	ER_L	0.879
		2	3	3	13.000	390.000000	ER_M	0.033
		4	3	3	41.700	390.000000	ER_M	0.107
		5	5	5	56.000	390.000000	ER_M	0.144
		10	1	1	61.500	390.000000	ER_M	0.158

I	Lead	2	5	5	38.000	35.000000	ER_L	1.086
		4	5	5	58.000	35.000000	ER_L	1.657
		5	5	5	54.000	35.000000	ER_L	1.543
		10	3	3	77.000	35.000000	ER_L	2.200
		2	5	5	38.000	110.000000	ER_M	0.345
		4	5	5	58.000	110.000000	ER_M	0.527
		5	5	5	54.000	110.000000	ER_M	0.491
		10	3	3	77.000	110.000000	ER_M	0.700
I	Mercury	2	5	3	0.580	0.150000	ER_L	3.867
		4	5	5	18.000	0.150000	ER_L	120.000
		5	5	5	5.600	0.150000	ER_L	37.333
		10	3	3	0.120	0.150000	ER_L	0.800
		2	5	3	0.580	1.300000	ER_M	0.446
		4	5	5	18.000	1.300000	ER_M	13.846
		5	5	5	5.600	1.300000	ER_M	4.308
		10	3	3	0.120	1.300000	ER_M	0.092
I	Nickel	2	5	4	17.000	30.000000	ER_L	0.567
		4	5	5	27.000	30.000000	ER_L	0.900
		5	5	5	32.500	30.000000	ER_L	1.083
		10	3	3	28.000	30.000000	ER_L	0.933
		2	5	4	17.000	50.000000	ER_M	0.340
		4	5	5	27.000	50.000000	ER_M	0.540
		5	5	5	32.500	50.000000	ER_M	0.650
		10	3	3	28.000	50.000000	ER_M	0.560
I	Silver	2	5	1	1.000	1.000000	ER_L	1.000
		5	5	3	1.000	1.000000	ER_L	1.000
		2	5	1	1.000	2.200000	ER_M	0.455
		5	5	3	1.000	2.200000	ER_M	0.455
I	Zinc	2	3	3	84.000	120.000000	ER_L	0.700
		4	3	3	140.000	120.000000	ER_L	1.167
		5	5	5	390.000	120.000000	ER_L	3.250
		10	1	1	110.000	120.000000	ER_L	0.917
		2	3	3	84.000	270.000000	ER_M	0.311
		4	3	3	140.000	270.000000	ER_M	0.519
		5	5	5	390.000	270.000000	ER_M	1.444
		10	1	1	110.000	270.000000	ER_M	0.407
O	Anthracene	4	5	1	8.000	0.085000	ER_L	94.118
O	Bis(2-ethylhexyl)phthalate	2	5	2	1.400	11.200000	EQ_PART	0.125
		4	5	1	8.000	11.200000	EQ_PART	0.714
		5	5	3	7.500	11.200000	EQ_PART	0.670
O	Fluoranthene	4	5	1	8.000	29.800000	EqP	0.269
		4	5	1	8.000	0.600000	ER_L	13.333
O	Methylene chloride	2	1	1	0.130	40.900000	EqP	0.003
		4	1	1	0.170	40.900000	EqP	0.004
		5	1	1	0.046	40.900000	EqP	0.001

0	Phenanthrene	4	5	1	8.000	8.640000	EqP	0.926
		4	5	1	8.000	0.225000	ER_L	35.556
0	Pyrene	4	5	1	8.000	0.350000	ER_L	22.857

TABLE E.4. Detected contaminants in sediment, maximum sediment concentrations (mg/kg), sediment screening benchmarks (mg/kg), and hazard quotients. Concentratrations are from the TVA reservoir monitoring program.

Analysis type	Analyte	Reach	Total observations	Number of Detects	MAX Sediment Conc.	Benchmark	Benchmark Type	MAX Quotient
I	Cadmium	10	13	4	3.00	5.000000	ER_L	0.6000
		10	13	4	3.00	9.000000	ER_M	0.3333
		10	13	4	3.00	0.600000	LEL_MOE	5.0000
I	Chromium (6+)	5	8	8	65.00	80.000000	ER_L	0.8125
		10	13	13	61.00	80.000000	ER_L	0.7625
		5	8	8	65.00	145.000000	ER_M	0.4483
		10	13	13	61.00	145.000000	ER_M	0.4207
I	Copper	5	8	8	57.00	70.000000	ER_L	0.8143
		10	13	13	40.00	70.000000	ER_L	0.5714
		5	8	8	57.00	390.000000	ER_M	0.1462
		10	13	13	40.00	390.000000	ER_M	0.1026
I	Iron	5	8	8	50000.00	30000.000000	LEL_MOE	1.6667
		10	13	13	45000.00	30000.000000	LEL_MOE	1.5000
I	Lead	5	8	8	87.00	35.000000	ER_L	2.4857
		10	13	13	110.00	35.000000	ER_L	3.1429
		5	8	8	87.00	110.000000	ER_M	0.7909
		10	13	13	110.00	110.000000	ER_M	1.0000
I	Manganese	5	8	8	4600.00	460.000000	LEL_MOE	10.0000
		10	13	13	6500.00	460.000000	LEL_MOE	14.1304
I	Mercury	5	8	8	0.95	0.150000	ER_L	6.3333
		10	13	6	0.20	0.150000	ER_L	1.3333
		5	8	8	0.95	1.300000	ER_M	0.7308
		10	13	6	0.20	1.300000	ER_M	0.1538
I	Nickel	5	8	8	56.00	30.000000	ER_L	1.8667
		10	13	12	49.00	30.000000	ER_L	1.6333
		5	8	8	56.00	50.000000	ER_M	1.1200
		10	13	12	49.00	50.000000	ER_M	0.9800
I	Zinc	5	8	8	300.00	120.000000	ER_L	2.5000
		10	13	13	210.00	120.000000	ER_L	1.7500
		5	8	8	300.00	270.000000	ER_M	1.1111
		10	13	13	210.00	270.000000	ER_M	0.7778
O	4,4'-DDT	10	13	1	0.04	84.10000	EqP	0.0010
		10	13	1	0.04	0.001000	ER_L	44.0000
		10	13	1	0.04	0.007000	ER_M	6.2857
O	Aroclor-1260	10	13	1	0.55	0.050000	ER_L	11.0000
O	Chlordane	5	8	1	0.02	0.000500	ER_L	36.0000

		5	8	1	0.02	0.006000	ER_M	3.0000
0	PCBs, total	10	3	1	0.55	0.050000	ER_L	11.0000
		10	3	1	0.55	0.368000	ER_M	1.4946

TABLE E.5. Detected contaminants in sediment, maximum sediment concentrations (mg/kg), sediment screening benchmarks (mg/kg), and hazard quotients. Concentratrations are from analyses of TVA recreational area samples.

Analysis type	Analyte	Reach	Total observations	Number of Detects	MAX Sediment Conc.	Benchmark	Benchmark Type	MAX Quotient
I	Antimony	2	2	1	0.39	2.000000	ER_L	0.1950
		5	11	8	0.58	2.000000	ER_L	0.2900
		10	3	3	0.62	2.000000	ER_L	0.3100
		18	3	2	0.45	2.000000	ER_L	0.2250
		2	2	1	0.39	25.000000	ER_M	0.0156
		5	11	8	0.58	25.000000	ER_M	0.0232
		10	3	3	0.62	25.000000	ER_M	0.0248
		18	3	2	0.45	25.000000	ER_M	0.0180
I	Arsenic	2	2	2	15.00	33.000000	ER_L	0.4545
		5	11	11	23.00	33.000000	ER_L	0.6970
		10	3	3	19.00	33.000000	ER_L	0.5758
		18	3	3	7.60	33.000000	ER_L	0.2303
		2	2	2	15.00	85.000000	ER_M	0.1765
		5	11	11	23.00	85.000000	ER_M	0.2706
		10	3	3	19.00	85.000000	ER_M	0.2235
		18	3	3	7.60	85.000000	ER_M	0.0894
		2	2	2	15.00	6.000000	LEL_MOE	2.5000
		5	11	11	23.00	6.000000	LEL_MOE	3.8333
		10	3	3	19.00	6.000000	LEL_MOE	3.1667
		18	3	3	7.60	6.000000	LEL_MOE	1.2667
I	Barium	2	2	2	170.00	20.000000	EPA_REGV	8.5000
		5	11	11	210.00	20.000000	EPA_REGV	10.5000
		10	3	3	150.00	20.000000	EPA_REGV	7.5000
		18	3	3	150.00	20.000000	EPA_REGV	7.5000
I	Chromium (6+)	2	2	2	56.00	80.000000	ER_L	0.7000
		5	11	11	77.00	80.000000	ER_L	0.9625
		10	3	3	87.00	80.000000	ER_L	1.0875
		18	3	3	38.00	80.000000	ER_L	0.4750
		2	2	2	56.00	145.000000	ER_M	0.3862
		5	11	11	77.00	145.000000	ER_M	0.5310
		10	3	3	87.00	145.000000	ER_M	0.6000
		18	3	3	38.00	145.000000	ER_M	0.2621
I	Copper	2	2	2	17.00	70.000000	ER_L	0.2429
		5	11	6	26.00	70.000000	ER_L	0.3714
		10	3	3	25.00	70.000000	ER_L	0.3571
		18	3	2	30.00	70.000000	ER_L	0.4286
		2	2	2	17.00	390.000000	ER_M	0.0436
		5	11	6	26.00	390.000000	ER_M	0.0667
		10	3	3	25.00	390.000000	ER_M	0.0641
		18	3	2	30.00	390.000000	ER_M	0.0769
I	Cyanide	2	2	1	0.06	0.100000	EPA_REGV	0.6000
		5	11	5	0.72	0.100000	EPA_REGV	7.2000

		10	3	3	0.40	0.100000	EPA_REGV	4.0000
		18	3	3	0.13	0.100000	EPA_REGV	1.3000
I	Lead	2	2	2	35.00	35.000000	ER_L	1.0000
		5	11	11	120.00	35.000000	ER_L	3.4286
		10	3	3	150.00	35.000000	ER_L	4.2857
		18	3	3	33.00	35.000000	ER_L	0.9429
		2	2	2	35.00	110.000000	ER_M	0.3182
		5	11	11	120.00	110.000000	ER_M	1.0909
		10	3	3	150.00	110.000000	ER_M	1.3636
		18	3	3	33.00	110.000000	ER_M	0.3000
I	Manganese	2	2	2	1500.00	460.000000	LEL_MOE	3.2609
		5	11	11	2500.00	460.000000	LEL_MOE	5.4348
		10	3	3	3700.00	460.000000	LEL_MOE	8.0435
		18	3	3	880.00	460.000000	LEL_MOE	1.9130
I	Mercury	2	2	1	0.45	0.150000	ER_L	3.0000
		5	11	3	0.15	0.150000	ER_L	1.0000
		10	3	1	0.15	0.150000	ER_L	1.0000
		18	3	2	2.50	0.150000	ER_L	16.6667
		2	2	1	0.45	1.300000	ER_M	0.3462
		5	11	3	0.15	1.300000	ER_M	0.1154
		10	3	1	0.15	1.300000	ER_M	0.1154
		18	3	2	2.50	1.300000	ER_M	1.9231
I	Nickel	2	2	2	42.00	30.000000	ER_L	1.4000
		5	11	11	51.00	30.000000	ER_L	1.7000
		10	3	3	49.00	30.000000	ER_L	1.6333
		18	3	3	28.00	30.000000	ER_L	0.9333
		2	2	2	42.00	50.000000	ER_M	0.8400
		5	11	11	51.00	50.000000	ER_M	1.0200
		10	3	3	49.00	50.000000	ER_M	0.9800
		18	3	3	28.00	50.000000	ER_M	0.5600
I	Zinc	2	2	2	72.00	120.000000	ER_L	0.6000
		5	11	11	130.00	120.000000	ER_L	1.0833
		10	3	3	93.00	120.000000	ER_L	0.7750
		18	3	3	130.00	120.000000	ER_L	1.0833
		2	2	2	72.00	270.000000	ER_M	0.2667
		5	11	11	130.00	270.000000	ER_M	0.4815
		10	3	3	93.00	270.000000	ER_M	0.3444
		18	3	3	130.00	270.000000	ER_M	0.4815
O	Benzo(a)anthracene	2	2	1	0.44	63.360000	EqP	0.0069
		2	2	1	0.44	0.230000	ER_L	1.9130
O	Bis(2-ethylhexyl)phthalate	2	2	1	0.71	11.20000	EQ_PART	0.0634
		5	11	2	0.62	11.20000	EQ_PART	0.0550
O	Dibenz(a,h)anthracene	5	11	1	0.83	0.060000	ER_L	13.8333
		5	11	1	0.83	0.260000	ER_M	3.1923
O	Fluoranthene	2	2	1	1.20	29.800000	EqP	0.0403

		18	3	1	0.40	29.800000	EqP	0.0134
		2	2	1	1.20	0.600000	ER_L	2.0000
		18	3	1	0.40	0.600000	ER_L	0.6667
0	Methylene chloride	5	29	1	0.05	40.900000	EqP	0.0012
0	Phenanthrene	2	2	1	1.10	8.640000	EqP	0.1270
		2	2	1	1.10	0.225000	ER_L	4.8889
0	Pyrene	2	2	1	0.96	0.350000	ER_L	2.7429

TABLE E.6. Upper confidence limit (UCL) concentration of chemicals detected in the fish samples (ug/kg), toxicological benchmarks (ug/kg), and quotients for wildlife.

Type of analyte	Analyte	Fish species	Wildlife species	Reach	Total observations	Number of Detects	UCL Conc.	Benchmark	UCL Quotient
I	Arsenic	Bluegill	Great Blue Heron	2	11	3.00	54.03	4807.26	0.0112
				4	19	17.00	91.54	4807.26	0.0190
				5	20	18.00	71.71	4807.26	0.0149
				10	10	9.00	72.73	4807.26	0.0151
I	Arsenic	Bluegill	Mink	2	11	3.00	54.03	222.90	0.2424
				4	19	17.00	91.54	222.90	0.4107
				5	20	18.00	71.71	222.90	0.3217
				10	10	9.00	72.73	222.90	0.3263
I	Arsenic	Catfish	Great Blue Heron	2	1	1.00	30.00	4807.26	0.0062
				5	2	1.00	56.57	4807.26	0.0118
				10	5	1.00	131.32	4807.26	0.0273
I	Arsenic	Catfish	Mink	2	1	1.00	30.00	222.90	0.1346
				5	2	1.00	56.57	222.90	0.2538
				10	5	1.00	131.32	222.90	0.5891
I	Arsenic	Large Mouth Bass	Great Blue Heron	2	2	2.00	412.55	4807.26	0.0858
				4	8	8.00	134.32	4807.26	0.0279
				5	5	5.00	164.60	4807.26	0.0342
				10	5	5.00	179.41	4807.26	0.0373
I	Arsenic	Large Mouth Bass	Mink	2	2	2.00	412.55	222.90	1.8508
				4	8	8.00	134.32	222.90	0.6026
				5	5	5.00	164.60	222.90	0.7384
				10	5	5.00	179.41	222.90	0.8049
I	Beryllium	Bluegill	Mink	10	10	1.00	12.63	2849.61	0.0044
I	Beryllium	Large Mouth Bass	Mink	5	5	2.00	21.08	2849.61	0.0074
I	Cadmium	Catfish	Great Blue Heron	2	1	1.00	4.00	850.05	0.0047
I	Cadmium	Catfish	Mink	2	1	1.00	4.00	445.71	0.0090
I	Copper	Bluegill	Great Blue Heron	5	20	6.00	23968.51	123789.22	0.1936
I	Copper	Bluegill	Mink	5	20	6.00	23968.51	85300.00	0.2810
I	Copper	Catfish	Great Blue Heron	5	2	2.00	4091.19	123789.22	0.0330
				18	1	1.00	1000.00	123789.22	0.0081
I	Copper	Catfish	Mink	5	2	2.00	4091.19	85300.00	0.0480
				18	1	1.00	1000.00	85300.00	0.0117
I	Lead	Catfish	Great Blue Heron	2	1	1.00	20.00	12845.16	0.0016
				10	5	2.00	69.39	12845.16	0.0054
I	Mercury	Bluegill	Great Blue Heron	2	11	11.00	62.31	27.31	2.2814
				4	19	19.00	332.36	27.31	12.1699

				5	20	20.00	64.51	27.31	2.3621
				10	10	10.00	162.56	27.31	5.9523
I	Mercury	Bluegill	Mink	2	11	11.00	62.31	59.70	1.0436
				4	19	19.00	332.36	59.70	5.5672
				5	20	20.00	64.51	59.70	1.0806
				10	10	10.00	162.56	59.70	2.7229
I	Mercury	Catfish	Great Blue Heron	2	1	1.00	210.00	27.31	7.6895
				5	2	1.00	392.55	27.31	14.3739
				10	5	4.00	300.78	27.31	11.0135
				18	1	1.00	140.00	27.31	5.1263
I	Mercury	Catfish	Mink	2	1	1.00	210.00	59.70	3.5176
				5	2	1.00	392.55	59.70	6.5754
				10	5	4.00	300.78	59.70	5.0382
				18	1	1.00	140.00	59.70	2.3451
I	Mercury	Large Mouth Bass	Great Blue Heron	2	2	2.00	804.07	27.31	29.4424
				4	8	8.00	387.08	27.31	14.1737
				5	5	5.00	236.68	27.31	8.6665
				10	5	5.00	387.88	27.31	14.2028
I	Mercury	Large Mouth Bass	Mink	2	2	2.00	804.07	59.70	13.4685
				4	8	8.00	387.08	59.70	6.4838
				5	5	5.00	236.68	59.70	3.9645
				10	5	5.00	387.88	59.70	6.4971
I	Selenium	Bluegill	Great Blue Heron	2	11	10.00	840.98	426.85	1.9702
				4	19	10.00	707.88	426.85	1.6584
				10	10	3.00	561.66	426.85	1.3158
I	Selenium	Bluegill	Mink	2	11	10.00	840.98	133.75	6.2877
				4	19	10.00	707.88	133.75	5.2926
				10	10	3.00	561.66	133.75	4.1994
I	Selenium	Catfish	Great Blue Heron	2	1	1.00	160.00	426.85	0.3748
				5	2	2.00	346.27	426.85	0.8112
				10	5	1.00	200.00	426.85	0.4685
				18	1	1.00	170.00	426.85	0.3983
I	Selenium	Catfish	Mink	2	1	1.00	160.00	133.75	1.1963
				5	2	2.00	346.27	133.75	2.5890
				10	5	1.00	200.00	133.75	1.4953
				18	1	1.00	170.00	133.75	1.2710
I	Selenium	Large Mouth Bass	Great Blue Heron	2	2	1.00	1673.63	426.85	3.9209
				4	8	4.00	721.90	426.85	1.6912
				10	5	5.00	672.29	426.85	1.5750
I	Selenium	Large Mouth Bass	Mink	2	2	1.00	1673.63	133.75	12.5131
				4	8	4.00	721.90	133.75	5.3974
				10	5	5.00	672.29	133.75	5.0265
I	Zinc	Bluegill	Great Blue Heron	2	11	11.00	6635.93	7443.24	0.8915

				4	19	19.00	7074.63	7443.24	0.9505
				5	20	20.00	16060.93	7443.24	2.1578
				10	10	10.00	6097.16	7443.24	0.8192
I	Zinc	Bluegill	Mink	2	11	11.00	6635.93	51187.49	0.1296
				4	19	19.00	7074.63	51187.49	0.1382
				5	20	20.00	16060.93	51187.49	0.3138
				10	10	10.00	6097.16	51187.49	0.1191
I	Zinc	Catfish	Great Blue Heron	2	1	1.00	5200.00	7443.24	0.6986
				5	2	2.00	13313.75	7443.24	1.7887
				10	2	2.00	9794.13	7443.24	1.3158
				18	1	1.00	6700.00	7443.24	0.9001
I	Zinc	Catfish	Mink	2	1	1.00	5200.00	51187.49	0.1016
				5	2	2.00	13313.75	51187.49	0.2601
				10	2	2.00	9794.13	51187.49	0.1913
				18	1	1.00	6700.00	51187.49	0.1309
I	Zinc	Large Mouth Bass	Great Blue Heron	2	2	2.00	6825.50	7443.24	0.9170
				4	8	8.00	4525.45	7443.24	0.6080
				5	4	4.00	4330.04	7443.24	0.5817
				10	5	5.00	5984.82	7443.24	0.8041
I	Zinc	Large Mouth Bass	Mink	2	2	2.00	6825.50	51187.49	0.1333
				4	8	8.00	4525.45	51187.49	0.0884
				5	4	4.00	4330.04	51187.49	0.0846
				10	5	5.00	5984.82	51187.49	0.1169
O	4,4'-DDD	Catfish	Great Blue Heron	4	28	4.00	28.35	180.73	0.1569
				5	39	12.00	9.09	180.73	0.0503
				10	13	7.00	7.40	180.73	0.0410
O	4,4'-DDE	Catfish	Great Blue Heron	2	14	7.00	53.68	180.73	0.2970
				4	28	18.00	52.59	180.73	0.2910
				5	40	25.00	40.54	180.73	0.2243
				10	13	2.00	10.24	180.73	0.0566
				18	21	12.00	67.01	180.73	0.3708
O	4,4'-DDE	Striped or Hybrid Bass	Great Blue Heron	2	10	8.00	67.45	180.73	0.3732
				5	10	4.00	72.56	180.73	0.4015
				18	10	7.00	101.03	180.73	0.5590
O	4,4'-DDT	Catfish	Mink	2	14	2.00	76.46	2532.97	0.0302
				4	28	4.00	53.29	2532.97	0.0210
				5	40	8.00	12.65	2532.97	0.0050
				18	21	1.00	11.30	2532.97	0.0045
O	4,4'-DDT	Striped or Hybrid Bass	Mink	5	10	2.00	14.44	2532.97	0.0057
O	Aldrin	Catfish	Great Blue Heron	2	15	2.00	72.45	132.70	0.5460
				4	24	3.00	34.49	132.70	0.2599
				5	40	2.00	15.66	132.70	0.1180
O	Aldrin	Catfish	Mink	2	15	2.00	72.45	633.28	0.1144

				4	24	3.00	34.49	633.28	0.0545
				5	40	2.00	15.66	633.28	0.0247
O	Aldrin	Striped or Hybrid Bass	Great Blue Heron	18	10	3.00	12.83	132.70	0.0967
O	Aldrin	Striped or Hybrid Bass	Mink	18	10	3.00	12.83	633.28	0.0203
O	Aroclor-1016	White Bass	Great Blue Heron	18	10	1.00	100.00	6987.65	0.0143
O	Aroclor-1016	White Bass	Mink	18	10	1.00	100.00	1000.00	0.1000
O	Aroclor-1242	Striped or Hybrid Bass	Great Blue Heron	18	26	1.00	152.08	6987.65	0.0218
O	Aroclor-1242	Striped or Hybrid Bass	Mink	18	26	1.00	152.08	1000.00	0.1521
O	Aroclor-1254	Catfish	Great Blue Heron	2	11	10.00	349.52	6987.65	0.0500
				4	46	40.00	822.43	6987.65	0.1177
				5	129	102.00	540.48	6987.65	0.0773
				10	63	26.00	91.51	6987.65	0.0131
				18	110	105.00	896.23	6987.65	0.1283
O	Aroclor-1254	Catfish	Mink	2	11	10.00	349.52	1000.00	0.3495
				4	46	40.00	822.43	1000.00	0.8224
				5	129	102.00	540.48	1000.00	0.5405
				10	63	26.00	91.51	1000.00	0.0915
				18	110	105.00	896.23	1000.00	0.8962
O	Aroclor-1254	Gizzard Shad	Great Blue Heron	4	15	14.00	117.99	6987.65	0.0169
				5	8	7.00	177.84	6987.65	0.0255
				10	17	1.00	67.68	6987.65	0.0097
				18	15	13.00	138.19	6987.65	0.0198
O	Aroclor-1254	Gizzard Shad	Mink	4	15	14.00	117.99	1000.00	0.1180
				5	8	7.00	177.84	1000.00	0.1778
				10	17	1.00	67.68	1000.00	0.0677
				18	15	13.00	138.19	1000.00	0.1382
O	Aroclor-1254	Large Mouth Bass	Great Blue Heron	4	11	11.00	337.87	6987.65	0.0484
				5	21	20.00	156.52	6987.65	0.0224
				10	10	2.00	102.22	6987.65	0.0146
				18	12	12.00	283.21	6987.65	0.0405
O	Aroclor-1254	Large Mouth Bass	Mink	4	11	11.00	337.87	1000.00	0.3379
				5	21	20.00	156.52	1000.00	0.1565
				10	10	2.00	102.22	1000.00	0.1022
				18	12	12.00	283.21	1000.00	0.2832
O	Aroclor-1254	Sauger	Great Blue Heron	2	12	3.00	100.00	6987.65	0.0143
				4	29	26.00	290.98	6987.65	0.0416
				18	30	12.00	175.13	6987.65	0.0251
O	Aroclor-1254	Sauger	Mink	2	12	3.00	100.00	1000.00	0.1000
				4	29	26.00	290.98	1000.00	0.2910
				18	30	12.00	175.13	1000.00	0.1751

0	Aroclor-1254	Small Mouth Buffalo	Great Blue Heron	5	1	1.00	400.00	6987.65	0.0572
				18	10	10.00	549.47	6987.65	0.0786
0	Aroclor-1254	Small Mouth Buffalo	Mink	5	1	1.00	400.00	1000.00	0.4000
				18	10	10.00	549.47	1000.00	0.5495
0	Aroclor-1254	Striped or Hybrid Bass	Great Blue Heron	2	20	20.00	952.93	6987.65	0.1364
				5	66	61.00	566.15	6987.65	0.0810
				10	15	2.00	59.90	6987.65	0.0086
				18	26	25.00	558.20	6987.65	0.0799
0	Aroclor-1254	Striped or Hybrid Bass	Mink	2	20	20.00	952.93	1000.00	0.9529
				5	66	61.00	566.15	1000.00	0.5662
				10	15	2.00	59.90	1000.00	0.0599
				18	26	25.00	558.20	1000.00	0.5582
0	Aroclor-1254	White Bass	Great Blue Heron	18	10	9.00	356.63	6987.65	0.0510
0	Aroclor-1254	White Bass	Mink	18	10	9.00	356.63	1000.00	0.3566
0	Aroclor-1260	Catfish	Great Blue Heron	2	15	15.00	1111.01	6987.65	0.1590
				4	46	42.00	1042.70	6987.65	0.1492
				5	131	121.00	974.61	6987.65	0.1395
				10	63	58.00	278.02	6987.65	0.0398
				18	110	105.00	916.49	6987.65	0.1312
0	Aroclor-1260	Catfish	Mink	2	15	15.00	1111.01	1000.00	1.1110
				4	46	42.00	1042.70	1000.00	1.0427
				5	131	121.00	974.61	1000.00	0.9746
				10	63	58.00	278.02	1000.00	0.2780
				18	110	105.00	916.49	1000.00	0.9165
0	Aroclor-1260	Gizzard Shad	Great Blue Heron	4	15	15.00	188.17	6987.65	0.0269
				5	8	7.00	154.54	6987.65	0.0221
				10	17	1.00	67.14	6987.65	0.0096
				18	15	13.00	260.19	6987.65	0.0372
0	Aroclor-1260	Gizzard Shad	Mink	4	15	15.00	188.17	1000.00	0.1882
				5	8	7.00	154.54	1000.00	0.1545
				10	17	1.00	67.14	1000.00	0.0671
				18	15	13.00	260.19	1000.00	0.2602
0	Aroclor-1260	Large Mouth Bass	Great Blue Heron	4	11	10.00	270.49	6987.65	0.0387
				5	21	19.00	173.55	6987.65	0.0248
				10	10	10.00	77.78	6987.65	0.0111
				18	12	12.00	313.26	6987.65	0.0448
0	Aroclor-1260	Large Mouth Bass	Mink	4	11	10.00	270.49	1000.00	0.2705
				5	21	19.00	173.55	1000.00	0.1735
				10	10	10.00	77.78	1000.00	0.0778
				18	12	12.00	313.26	1000.00	0.3133
0	Aroclor-1260	Sauger	Great Blue Heron	2	12	10.00	182.85	6987.65	0.0262
				4	29	29.00	463.13	6987.65	0.0663
				18	30	22.00	179.08	6987.65	0.0256

O	Aroclor-1260	Sauger	Mink	2	12	10.00	182.85	1000.00	0.1829
				4	29	29.00	463.13	1000.00	0.4631
				18	30	22.00	179.08	1000.00	0.1791
O	Aroclor-1260	Small Mouth Buffalo	Great Blue Heron	18	10	8.00	522.27	6987.65	0.0747
O	Aroclor-1260	Small Mouth Buffalo	Mink	18	10	8.00	522.27	1000.00	0.5223
O	Aroclor-1260	Striped or Hybrid Bass	Great Blue Heron	2	20	20.00	1847.57	6987.65	0.2644
				5	66	61.00	735.92	6987.65	0.1053
				10	15	3.00	64.67	6987.65	0.0093
				18	26	25.00	560.98	6987.65	0.0803
O	Aroclor-1260	Striped or Hybrid Bass	Mink	2	20	20.00	1847.57	1000.00	1.8476
				5	66	61.00	735.92	1000.00	0.7359
				10	15	3.00	64.67	1000.00	0.0647
				18	26	25.00	560.98	1000.00	0.5610
O	Aroclor-1260	White Bass	Great Blue Heron	18	10	10.00	642.53	6987.65	0.0920
O	Aroclor-1260	White Bass	Mink	18	10	10.00	642.53	1000.00	0.6425
O	Chlordane	Catfish	Great Blue Heron	2	15	5.01	273.39	3670.35	0.0745
				4	46	21.28	219.20	3670.35	0.0597
				5	110	31.47	133.80	3670.35	0.0365
				10	63	19.96	77.22	3670.35	0.0210
				18	90	22.15	184.19	3670.35	0.0502
O	Chlordane	Catfish	Mink	2	15	5.01	273.39	8210.36	0.0333
				4	46	21.28	219.20	8210.36	0.0267
				5	110	31.47	133.80	8210.36	0.0163
				10	63	19.96	77.22	8210.36	0.0094
				18	90	22.15	184.19	8210.36	0.0224
O	Chlordane	Gizzard Shad	Great Blue Heron	4	15	4.12	78.97	3670.35	0.0215
				5	8	0.86	68.68	3670.35	0.0187
				10	17	0.68	80.21	3670.35	0.0219
				18	15	3.53	64.42	3670.35	0.0176
O	Chlordane	Gizzard Shad	Mink	4	15	4.12	78.97	8210.36	0.0096
				5	8	0.86	68.68	8210.36	0.0084
				10	17	0.68	80.21	8210.36	0.0098
				18	15	3.53	64.42	8210.36	0.0078
O	Chlordane	Large Mouth Bass	Great Blue Heron	4	11	4.32	102.93	3670.35	0.0280
				5	20	4.70	76.35	3670.35	0.0208
				10	10	1.00	78.55	3670.35	0.0214
				18	10	3.25	77.99	3670.35	0.0212
O	Chlordane	Large Mouth Bass	Mink	4	11	4.32	102.93	8210.36	0.0125
				5	20	4.70	76.35	8210.36	0.0093
				10	10	1.00	78.55	8210.36	0.0096
				18	10	3.25	77.99	8210.36	0.0095
O	Chlordane	Sauger	Great Blue Heron	2	12	0.90	69.14	3670.35	0.0188

				4	20	3.36	112.71	3670.35	0.0307
				18	20	2.48	68.77	3670.35	0.0187
0	Chlordane	Sauger	Mink	2	12	0.90	69.14	8210.36	0.0084
				4	20	3.36	112.71	8210.36	0.0137
				18	20	2.48	68.77	8210.36	0.0084
0	Chlordane	Striped or Hybrid Bass	Great Blue Heron	2	20	5.63	289.64	3670.35	0.0789
				5	45	13.97	134.11	3670.35	0.0365
				10	15	1.71	80.27	3670.35	0.0219
				18	26	4.84	143.29	3670.35	0.0390
0	Chlordane	Striped or Hybrid Bass	Mink	2	20	5.63	289.64	8210.36	0.0353
				5	45	13.97	134.11	8210.36	0.0163
				10	15	1.71	80.27	8210.36	0.0098
				18	26	4.84	143.29	8210.36	0.0175
0	Chlordane	White Bass	Great Blue Heron	18	10	0.50	85.88	3670.35	0.0234
0	Chlordane	White Bass	Mink	18	10	0.50	85.88	8210.36	0.0105
0	Dieldrin	Catfish	Great Blue Heron	4	10	1.00	21.33	132.70	0.1608
				5	22	3.00	18.57	132.70	0.1399
				18	21	1.00	11.30	132.70	0.0851
0	Dieldrin	Catfish	Mink	4	10	1.00	21.33	633.28	0.0337
				5	22	3.00	18.57	633.28	0.0293
				18	21	1.00	11.30	633.28	0.0178
0	Dieldrin	Striped or Hybrid Bass	Great Blue Heron	5	10	1.00	10.00	132.70	0.0754
				18	10	3.00	15.67	132.70	0.1181
0	Dieldrin	Striped or Hybrid Bass	Mink	5	10	1.00	10.00	633.28	0.0158
				18	10	3.00	15.67	633.28	0.0247
0	Endrin	Catfish	Great Blue Heron	4	10	2.00	41.33	480.00	0.0861
				5	22	1.00	12.47	480.00	0.0260
				18	21	1.00	20.38	480.00	0.0425
0	Endrin	Striped or Hybrid Bass	Great Blue Heron	2	10	1.00	18.50	480.00	0.0385
				5	10	3.00	35.58	480.00	0.0741
				18	10	1.00	24.17	480.00	0.0503
0	PCBs, total	Catfish	Great Blue Heron	2	1	1.00	1100.00	6987.65	0.1574
				4	18	17.00	2223.74	6987.65	0.3182
				5	42	33.00	1710.05	6987.65	0.2447
				18	60	58.00	1937.62	6987.65	0.2773
0	PCBs, total	Catfish	Mink	2	1	1.00	1100.00	1000.00	1.1000
				4	18	17.00	2223.74	1000.00	2.2237
				5	42	33.00	1710.05	1000.00	1.7101
				18	60	58.00	1937.62	1000.00	1.9376
0	PCBs, total	Large Mouth Bass	Great Blue Heron	4	1	1.00	600.00	6987.65	0.0859
				5	1	1.00	100.00	6987.65	0.0143

				18	2	2.00	865.69	6987.65	0.1239
O	PCBs, total	Large Mouth Bass	Mink	4	1	1.00	600.00	1000.00	0.6000
				5	1	1.00	100.00	1000.00	0.1000
				18	2	2.00	865.69	1000.00	0.8657
O	PCBs, total	Sauger	Great Blue Heron	4	10	10.00	725.22	6987.65	0.1038
				18	10	10.00	558.56	6987.65	0.0799
O	PCBs, total	Sauger	Mink	4	10	10.00	725.22	1000.00	0.7252
				18	10	10.00	558.56	1000.00	0.5586
O	PCBs, total	Small Mouth Buffalo	Great Blue Heron	5	1	1.00	400.00	6987.65	0.0572
				18	10	10.00	1056.34	6987.65	0.1512
O	PCBs, total	Small Mouth Buffalo	Mink	5	1	1.00	400.00	1000.00	0.4000
				18	10	10.00	1056.34	1000.00	1.0563
O	PCBs, total	Striped or Hybrid Bass	Great Blue Heron	2	10	10.00	1540.43	6987.65	0.2205
				5	31	28.00	1567.89	6987.65	0.2244
				18	10	10.00	1526.52	6987.65	0.2185
O	PCBs, total	Striped or Hybrid Bass	Mink	2	10	10.00	1540.43	1000.00	1.5404
				5	31	28.00	1567.89	1000.00	1.5679
				18	10	10.00	1526.52	1000.00	1.5265

TABLE E.7. Upper 95% confidence limit (UCL) concentration of chemicals detected in reach 5 sediment samples (mg/kg) which is assumed to be deposited on land as dredge spoil, toxicological benchmarks for terrestrial plants(mg/kg), and quotients. UCL Conc = min(max, ucl of ordinary mean)

Analyte	Total observations	Number of Detects	UCL Conc.	Benchmark	UCL Quotient
Antimony	3	0.07	0.481	5.0	0.0962
Arsenic	4	4.00	9.099	10.0	0.9099
Beryllium	4	4.00	1.442	10.0	0.1442
Cadmium	4	4.00	4.580	2.0	2.2899
Copper	4	4.00	38.526	40.0	0.9631
Lead	4	4.00	45.054	50.0	0.9011
Mercury	6	6.00	2.526	0.3	8.4211
Nickel	4	4.00	28.366	25.0	1.1346
Silver	4	3.33	0.663	2.0	0.3315
Zinc	4	4.00	267.506	20.0	13.3753

Table E.8. Upper confidence limit (UCL) estimated concentration of chemicals vegetables (mg/kg) grown on dredge spoil from reach 5, toxicological benchmarks for cottontail rabbit (mg/kg), and quotients. UCL = min(max, UCL of ordinary mean. Scenario does not include irrigation.

Type of analyte	Analyte	Total observations	Number of Detects	UCL Conc.	Benchmark	UCL Quotient
I	Antimony	3	0.07	0.0012	1.2590	0.00097
I	Arsenic	4	4.00	0.0026	0.1260	0.02082
I	Beryllium	4	4.00	0.0015	1.6100	0.00093
I	Cadmium	4	4.00	0.5726	0.2520	2.27241
I	Chromium	6	6.00	0.1051	7.1600	0.01468
I	Copper	4	4.00	0.4830	0.2200	2.19568
I	Lead	4	4.00	0.1144	14.9100	0.00767
I	Mercury	6	6.00	0.1896	0.0191	9.92527
I	Nickel	4	4.00	0.3557	72.0500	0.00494
I	Uranium	2	2.00	0.0032	1.8400	0.00174
I	Zinc	4	4.00	26.7608	28.9000	0.92598
O	Bis(2-ethylhexyl)phthalate	4	1.76	0.0349	1.8800	0.01859

Appendix F

COST ESTIMATE SUMMARY FOR THE TWO FEASIBLE ALTERNATIVES

COST ESTIMATE SUMMARY

The following two pages contain a summary of the cost estimates that were prepared for alternatives 2 and 3. The assumptions made in preparing these estimates are outlined in Sections 8.6.3, 8.6.4, and in the following summaries. More detailed assumptions were used in preparing the estimates on the Automated Estimating System (AES) developed by Energy Systems for use with Activity Data Sheets and the validation process. For items that would be purchased, vendors were contacted by phone and estimated prices were provided, previous ORR project cost were used, or Energy Systems cost guidance reports were referenced. Labor intensive activities and some other pricing was obtained from Means Site Work Cost Data Guide or Means Mechanical Cost Data Guide. AES printouts, calculation sheets, and phone quote memorandums are on file at the Jacobs Technical Center in Oak Ridge, Tennessee.

SUMMARY

Alternative 2 - Institutional Controls and Advisories

This cost estimate assumes that controls will be maintained for 30 years.

$1,740,000	IAG process consist of 10 people meeting one day a month and 3 people collecting data for one week each year. Also included is $30,000 for materials.
$30,000	600 person hours to prepare revisions to TVA deeds and gain necessary approvals.
$9,000	Replacement of five fish consumption advisory signs each year including materials and labor.

$1,779,000

In order to fund this project over a thirty year period, an interest bearing account would be established the first year with the present value amount of:

$1,141,367
==========

Alternative 3 - Sediment Removal

This alternative would occur over a twenty seven year period.

$1,748,000	Mobilization and Demobilization
$672,720	Purchase of a silt curtain.
$318,525,800	Purchase of a contract to dredge 5,000 acres using 90% pneuma dredge and 10% mudcat dredge technologies.
$3,408,000	Construction of the on-shore dewatering and staging facility
$59,216,120	Purchase and operation of sediment dewatering equipment.
$333,132,761	Trucking of the waste water to ORNL for treatment.
$298,698,419	Treatment of the waste water at the Process Waste Treatment Facility.
$7,898,300,711	Purchase of B-25 7ANV boxes to store dewatered sediment.
$601,128,115	Packaging and trucking of boxes to K-25.
$768,398,547	Storage of boxes at K-25.
$48,450,300	Monitoring of the remedial action for health and safety of the public and the environment.
$10,331,679,493	Subtotal
$1,549,751,924	Miscellaneous support items such as personal protective equipment and support vehicles or boats (15%).
$3,099,503,848	Management of the remedial action by the integrating contractor (30%).
$1,549,751,924	Design of the remedial action by the design contractor (15%).
$8,827,386,959	Escalation at 53.4% to account for inflation in prices over the duration of the project.
$12,679,037,074	Contingency at 50% for expected and unexpected deviations from plans.
$38,037,111,000	

In order to fund this project over a twenty seven year period, an interest bearing account would need to be established with a present value amount of:

$16,566,000,000

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