



3 4456 0382095 2

ORNL/TM-11544

(ESD-1070)

oml

ENVIRONMENTAL SCIENCES DIVISION

OAK RIDGE NATIONAL LABORATORY

Fourth Report on the Oak Ridge National Laboratory Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River

J. M. Loar

MARTIN MARIETTA

OAK RIDGE NATIONAL LABORATORY
CENTRAL RESEARCH LIBRARY
CIRCULATION SECTION
4500N ROOM 175

LIBRARY LOAN COPY

DO NOT TRANSFER TO ANOTHER PERSON

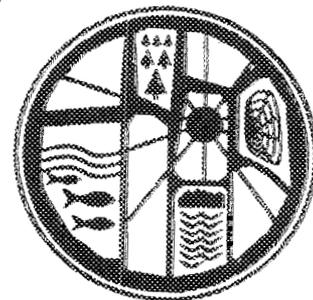
If you wish someone else to see this
report, send in name with report and
the library will arrange a loan.

UCR750143 573

- S. M. Adams
- R. D. Bailey
- B. G. Blaylock
- H. L. Boston
- W. R. Hill
- M. A. Huston
- L. A. Kszos
- J. M. Loar
- D. A. Moberbacher
- M. J. Peterson
- C. M. Pettway
- M. G. Ryon
- E. M. Schilling
- J. G. Smith
- G. R. Southworth
- A. J. Stewart
- C. K. Valentine
- B. T. Walton
- A. E. Waters

Environmental Sciences Division
Publication No. 4070

April 1994



MANAGED BY
MARTIN MARIETTA ENERGY SYSTEMS, INC.
FOR THE UNITED STATES
DEPARTMENT OF ENERGY

This report has been reproduced directly from the best available copy.

Available to DOE and DOE contractors from the Office of Scientific and Technical Information, P.O. Box 62, Oak Ridge, TN 37831; prices available from (615) 576-8401, FTS 626-8401.

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

This report was prepared as an account of work sponsored by an agency of the United States Government. Neither the United States Government nor any agency thereof, nor any of their employees, makes any warranty, express or implied, or assumes any legal liability or responsibility for the accuracy, completeness, or usefulness of any information, apparatus, product, or process disclosed, or represents that its use would not infringe privately owned rights. Reference herein to any specific commercial product, process, or service by trade name, trademark, manufacturer, or otherwise, does not necessarily constitute or imply its endorsement, recommendation, or favoring by the United States Government or any agency thereof. The views and opinions of authors expressed herein do not necessarily state or reflect those of the United States Government or any agency thereof.

ENVIRONMENTAL SCIENCES DIVISION

402
11

**Fourth Report on the Oak Ridge National Laboratory
Biological Monitoring and Abatement Program for
White Oak Creek Watershed and the Clinch River**

Editor

J. M. Loar

Contributors

S. M. Adams	J. M. Loar	G. R. Southworth
R. D. Bailey	D. A. Mohrbacher ¹	A. J. Stewart
B. G. Blaylock	M. J. Peterson	C. K. Valentine ³
H. L. Boston	C. M. Pettway ²	B. T. Walton
W. R. Hill	M. G. Ryon	A. E. Waters ⁴
M. A. Huston	E. M. Schilling	
L. A. Kszos	J. G. Smith	

**Environmental Sciences Division
Publication No. 4070**

¹Automated Sciences Group, Inc., Oak Ridge²Knoxville College, Knoxville, Tennessee³Office of Environmental Compliance and Documentation, ORNL⁴Breedlove, Dennis, & Assoc., Winter Park, Fla.

Date of Issue — April 1994

Prepared for
C. E. Nix, Leader
Environmental Compliance Group
Environmental Monitoring and Compliance Section
and
L. D. Bates, Manager
Remedial Action Program
Oak Ridge National Laboratory

Prepared by the
OAK RIDGE NATIONAL LABORATORY
Oak Ridge, Tennessee 37831-6285
managed by
MARTIN MARIETTA ENERGY SYSTEMS, INC.
for the
U.S. DEPARTMENT OF ENERGY
under contract DE-AC05-84OR21400



3 4456 0382095 2

CONTENTS

	Page
FIGURES	ix
TABLES	xv
ACRONYMS	xxi
PREFACE	xxv
ACKNOWLEDGMENTS	xxvii
EXECUTIVE SUMMARY	xxix
1. INTRODUCTION (<i>J. M. Loar</i>)	1-1
1.1 OBJECTIVES	1-1
2. DESCRIPTION OF THE WHITE OAK CREEK WATERSHED	2-1
2.1 GEOHYDROLOGY (<i>J. M. Loar</i>)	2-2
2.2 WATER QUALITY (<i>J. M. Loar, H. L. Boston, G. R. Southworth and R. D. Bailey</i>)	2-6
2.2.1 Description of Oak Ridge National Laboratory Effluents	2-6
2.2.2 Wastewater Modifications for Pollution Abatement	2-11
2.2.3 National Pollutant Discharge Elimination System Water Quality Monitoring Program	2-11
2.2.4 Biological Monitoring and Abatement Program Water Quality Sampling Program	2-16
2.2.4.1 Methods	2-21
2.2.4.2 Results	2-21
2.2.4.3 Summary	2-24
3. TOXICITY MONITORING	3-1
3.1 EFFLUENT AND AMBIENT TOXICITY TESTING (<i>A. J. Stewart and L. A. Kszos</i>)	3-1
3.1.1 Overview	3-1
3.1.2 Point-Source Contributions to Toxicity	3-3
3.1.3 National Pollutant Discharge Elimination System Permit Sites X13 and X14	3-5
3.1.4 Point-Source Chlorine Studies	3-7
3.1.5 Area-Source Chlorine Studies	3-13
3.1.6 Overview of the Ambient Toxicity Data Sets	3-16
3.1.6.1 Statistical analyses for ambient toxicity assessments	3-17
3.1.7 Streams Near Solid Radioactive Waste Disposal/Storage Area 6	3-21
3.2 INSTREAM MONITORING OF THE PERIPHYTON COMMUNITY (<i>H. L. Boston, W. R. Hill, and C. M. Pettway</i>)	3-22

CONTENTS (continued)

	Page
3.2.1 Methods	3-23
3.2.1.1 Periphyton chlorophyll and carbon incorporation	3-24
3.2.1.2 Chlorine toxicity bioassay	3-25
3.2.2 Results and Discussion	3-25
3.2.2.1 Algal assemblages	3-25
3.2.2.2 Monthly determination of algal biomass and photosynthesis	3-26
3.2.2.3 Monthly data for Chl <i>a</i> and photosynthesis	3-26
3.2.2.4 Annual average Chl <i>a</i> and photosynthesis	3-29
3.2.2.5 Comparing photosynthesis and periphyton condition	3-29
3.2.2.6 Chlorine toxicity at WCK 3.9	3-32
3.2.3 Summary	3-34
3.3 FUTURE STUDIES	3-35
4. BIOACCUMULATION STUDIES (<i>G. R. Southworth and M. J. Peterson</i>)	4-1
4.1 IDENTIFICATION OF CONTAMINANTS THAT ACCUMULATE IN AQUATIC BIOTA	4-1
4.1.1 Introduction	4-1
4.1.2 Methods	4-1
4.1.3 Results and Discussion	4-3
4.1.3.1 Metals	4-3
4.1.3.2 Organics	4-9
4.1.4 Conclusions	4-15
4.2 EVALUATION OF PCB CONTAMINATION IN WHITE OAK CREEK EMBAYMENT/CLINCH RIVER	4-17
4.2.1 Introduction	4-17
4.2.2 Methods	4-17
4.2.3 Results and Discussion	4-18
4.2.3.1 Polychlorinated biphenyls in catfish from the White Oak Creek Embayment/Clinch River	4-18
4.2.3.2 Polychlorinated biphenyl vs strontium-90 concentrations ...	4-21
4.2.3.3 Temporal changes in polychlorinated biphenyl concentrations in catfish	4-21
4.2.3.4 White Oak Creek as a source of polychlorinated biphenyls	4-22
4.2.4 Conclusions	4-23
4.3 FUTURE STUDIES	4-23
5. BIOLOGICAL INDICATORS OF CONTAMINANT-RELATED STRESS (<i>S. M. Adams</i>)	5-1
5.1 INTRODUCTION	5-1
5.2 METHODS	5-1

CONTENTS (continued)

	Page
5.2.1 Sampling Procedures	5-1
5.2.2 Analytical Procedures	5-1
5.2.2.1 Lipid analysis	5-2
5.2.2.2 Serum chemical analysis	5-2
5.2.2.3 Ribonucleic acid/deoxyribonucleic acid analysis	5-2
5.2.2.4 Histopathological analysis	5-3
5.2.2.5 Detoxification enzymes	5-3
5.2.3 Statistical Procedures	5-4
5.3 RESULTS AND DISCUSSION	5-4
5.3.1 Analysis of Spatial Effects	5-5
5.3.1.1 Detoxification enzymes	5-5
5.3.1.2 Organ dysfunction	5-5
5.3.1.3 Lipid metabolism	5-9
5.3.1.4 Histopathological condition	5-10
5.3.1.5 Condition indices	5-10
5.3.1.6 Spatial analysis—conclusions	5-11
5.3.2 Temporal Analysis	5-12
5.3.3 Integrated Site Analysis	5-14
5.3.3.1 Multivariate selection	5-14
5.3.4 Summary and Synthesis	5-14
5.4 FUTURE STUDIES	5-17
6. INSTREAM ECOLOGICAL MONITORING (<i>J. G. Smith</i>)	6-1
6.1 BENTHIC MACROINVERTEBRATES	6-1
6.1.1 Introduction	6-1
6.1.2 Materials and Methods	6-1
6.1.3 Results	6-2
6.1.3.1 Taxonomic composition	6-2
6.1.3.2 Density	6-3
6.1.3.3 Dominant taxa	6-3
6.1.3.4 Community structure	6-7
6.1.4 Discussion	6-8
6.1.5 Future Studies	6-13
6.2 FISHES (<i>M. G. Ryon and E. M. Schilling</i>)	6-13
6.2.1 Introduction	6-13
6.2.2 Methods	6-13
6.2.2.1 Field sampling procedures	6-15
6.2.2.2 Data analysis	6-16
6.2.3 Results and Discussion	6-18
6.2.3.1 Species richness and composition	6-18
6.2.3.2 Density and biomass	6-21
6.2.3.3 Condition factors	6-24
6.2.3.4 Age and growth	6-24

CONTENTS (continued)

	Page
6.2.3.5 Index of biotic integrity (IBI)	6-30
6.2.4 Fish Kills	6-30
6.2.5 Conclusions	6-32
6.2.6 Future Studies	6-33
6.3 INTERPRETATION OF BIOTIC CHANGES (<i>M. A. Huston</i>)	6-34
6.3.1 Introduction	6-34
6.3.2 Methods	6-35
6.3.3 Results and Discussion	6-35
6.3.3.1 Patterns of Ephemeroptera, Plecoptera, and Trichoptera, Chironomid, and total density	6-36
6.3.3.2 Patterns of EPT taxonomic richness	6-37
6.3.4 Future Studies	6-43
7. ASSESSMENT OF CONTAMINANTS IN THE TERRESTRIAL ENVIRONMENT	7-1
7.1 GUIDANCE FOR BIOLOGICAL MONITORING IN THE TERRESTRIAL ENVIRONMENT (<i>B. T. Walton</i>)	7-1
7.1.1 Vegetation	7-1
7.1.2 Small Mammals	7-2
7.1.3 Future Studies	7-4
8. RADIOECOLOGY OF WHITE OAK LAKE	8-1
8.1 RADIOECOLOGY OF MACROFLORA IN WHITE OAK LAKE (<i>B. G. Blaylock, D. A. Mohrbacher, and A. E. Waters</i>)	8-1
8.1.1 Introduction	8-1
8.1.2 Methods	8-2
8.1.3 Results and Discussion	8-4
8.1.3.1 White Oak Creek weir sampling	8-4
8.1.3.2 Melton Branch weir sampling	8-6
8.1.3.3 Plants collected in the White Oak Creek and Melton Branch weirs: Comparison between 1989 and 1988	8-6
8.1.3.4 White Oak Lake	8-6
8.1.3.5 Litter bag experiments	8-8
8.1.3.6 External vs internal contamination	8-13
8.1.4 Summary	8-13
8.2 WATERFOWL POPULATIONS ASSOCIATED WITH OAK RIDGE RESERVATION BASINS AND PONDS	8-15
8.2.1 Introduction	8-15
8.2.2 Materials and Methods	8-15
8.2.2.1 Waterfowl census	8-15
8.2.2.2 Contaminants in waterfowl	8-15
8.2.2.3 Contaminants in wood ducks from White Oak Lake	8-15

CONTENTS (continued)

	Page
8.2.2.4 White Oak Lake domestic mallard experiment	8-16
8.2.2.5 Canada goose banding study	8-16
8.2.3 Results and Discussion	8-18
8.2.3.1 Waterfowl census	8-18
8.2.3.2 Contaminants in Oak Ridge National Laboratory waterfowl	8-21
8.2.3.3 White Oak Lake domestic mallard experiment	8-23
8.2.3.4 Canada goose banding study	8-24
8.3 RADIONUCLIDES IN FISH IN WHITE OAK CREEK	8-24
8.3.1 Introduction	8-24
8.3.2 Methods	8-24
8.3.3 Results	8-26
8.4 FUTURE STUDIES	8-27
8.4.1 Radionuclides in White Oak Lake	8-27
8.4.2 Role of Aquatic Macroflora in the Radioecology of White Oak Lake	8-27
8.4.3 Waterfowl Populations Associated with Oak Ridge Reservation Basins and Ponds	8-28
8.4.4 Radionuclides in White Oak Lake Sediment	8-28
8.4.5 Radionuclides in Fish from White Oak Creek	8-28
9. ABATEMENT PROGRAM (<i>C. K. Valentine</i>)	9-1
9.1 CHLORINE REDUCTION	9-1
9.2 ETHYLENE GLYCOL	9-2
9.3 TANK DIKING	9-2
9.4 POLYCHLORINATED BIPHENYL AND MERCURY MONITORING	9-2
9.5 WASTEWATER PIPING	9-2
9.6 FIELD INTERFACE ACTIVITIES	9-3
9.7 NONRADIOLOGICAL WASTEWATER TREATMENT FACILITY	9-3
9.8 ELIMINATION OF CATEGORY III OUTFALLS	9-3
9.9 BEST MANAGEMENT PRACTICES PROGRAM	9-3
10. REFERENCES	10-1
APPENDIX A. RESULTS OF QUALITY ASSURANCE/QUALITY CONTROL ANALYSES OF MERCURY, POLYCHLORINATED BIPHENYLS, AND ORGANICS IN FISH SAMPLES	A-1

CONTENTS (continued)

	Page
APPENDIX B. CONCENTRATIONS OF CONTAMINANTS IN AQUATIC BIOTA FROM WHITE OAK CREEK AND TRIBUTARIES, WHITE OAK LAKE, AND THE CLINCH RIVER NOVEMBER 1988-AUGUST 1989	B-1
APPENDIX C. CHECKLIST OF BENTHIC MACROINVERTEBRATE TAXA FROM THE WHITE OAK CREEK WATERSHED, MAY-JUNE 1987	C-1
APPENDIX D. METHODOLOGY FOR INDEX OF BIOTIC INTEGRITY	D-1
APPENDIX E. DENSITY, BIOMASS, CONDITION FACTORS, AND GROWTH OF FISHES IN THE WHITE OAK CREEK WATERSHED	E-1
APPENDIX F. MONTHLY PERIPHYTON CHLOROPHYLL <i>A</i> AND PHOTOSYNTHESIS IN THE WHITE OAK CREEK WATERSHED, 1989	F-1

FIGURES

Figure	Page
2.1	Map of the White Oak Creek watershed above White Oak Dam 2-1
2.2	Map showing locations of liquid and solid waste disposal areas; National Pollutant Discharge Elimination System ambient monitoring stations on Melton Branch, White Oak Creek, and White Oak Dam; and sampling sites for benthic invertebrates and fish in White Oak Creek watershed 2-3
2.3	Mean weekly stream flow in Melton Branch and White Oak Creek above and below ORNL 2-7
2.4	Location of National Pollutant Discharge Elimination System effluent and ambient water quality monitoring stations 2-9
2.5	Mean weekly temperatures in White Oak Creek at sites above and below Oak Ridge National Laboratory and in Melton Branch above and below the High Flux Isotope Reactor complex, January–December 1989 2-14
2.6	Mean weekly temperatures in three tributaries of White Oak Creek, January–December 1989 2-17
2.7	Water temperatures at National Pollutant Discharge Elimination System monitoring stations X13 on lower Melton Branch at Melton Branch kilometer (MEK) 0.16 and X14 on lower White Oak Creek at White Oak Creek kilometer (WCK) 2.65 2-18
3.1	Sampling sites for ambient toxicity tests of water from streams at Oak Ridge National Laboratory 3-2
3.2	Loss of total residual chlorine from aerated water samples from three White Oak Creek sites under two temperature regimes 3-14
3.3	Periphyton monitoring sites in the White Oak Creek watershed in 1989. 3-24
3.4	Periphyton chlorophyll <i>a</i> and photosynthesis on small flat rocks collected from shallow riffle areas in White Oak Creek above and below Oak Ridge National Laboratory [White Oak Creek kilometer (WCK) 6.8 and WCK 2.3 respectively], January 1987–December 1989 3-27
3.5	Periphyton chlorophyll <i>a</i> and photosynthesis on small flat rocks collected from shallow riffle areas in upper Melton Branch at Melton Branch kilometer (MEK) 1.8 and MEK 1.6 and in lower Melton Branch at Melton Branch kilometer (MEK) 0.6 3-28

FIGURES (continued)

Figure	Page
3.6	Monthly least square means for biomass-adjusted photosynthesis, based on an analysis of covariance for data collected during 1989 3-33
3.7	Algal periphyton biomass and photosynthesis on substrata collected from upper East Fork Poplar Creek at kilometer 24.4 and held for 23 d at 0 to 100% tap water 3-35
4.1	Mean concentrations \pm standard error of mercury in bluegill collected in winter 1988/1989 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River 4-6
4.2	Mean concentrations \pm standard error of polychlorinated biphenyls in bluegill collected in winter 1988/1989 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River 4-11
4.3	Location of sites on the Oak Ridge Reservation where channel catfish were collected for polychlorinated biphenyl analysis in July/August 1989 4-16
5.1	Levels of biological response in fish to contaminant stress, illustrating the continuum of these responses along gradients of response time and ecological relevance. 5-6
5.2	Relative differences in the response of detoxification enzymes for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek 5-7
5.3	Relative differences in the response of the organ dysfunction indicators for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek 5-7
5.4	Relative differences in the response of the lipid metabolism parameters for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek 5-8
5.5	Relative differences in histopathological condition for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek 5-8
5.6	Relative differences in the condition indices for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek. 5-9

FIGURES (continued)

Figure	Page
5.7 Segregation of health responses for bluegill from three sites in White Oak Creek, White Oak Lake, and the reference stream, using bioindicators measured in fall 1988	5-15
6.1 Mean density, number of taxa/sample, number of Ephemeroptera, Plecoptera, and Trichoptera taxa/sample, and taxonomic diversity for benthic macroinvertebrates in the White Oak Creek watershed for the months of May and June in 1986 and 1987	6-5
6.2 Magnitude of difference between each downstream site and its reference site and the magnitude of change within each site between 1986 and 1987 for mean density, taxonomic richness, and Ephemeroptera, Plecoptera, and Trichoptera richness in the White Oak Creek watershed, May and June 1986 and 1987	6-11
6.3 True rate of growth in weight of redbreast in White Oak Creek and a reference stream, Brushy Fork, during 1988	6-26
6.4 True rate of growth in weight of bluegill in White Oak Creek and a reference stream, Brushy Fork, during 1988	6-26
6.5 True rate of growth in weight of redbreast sunfish in White Oak Creek and a reference stream, Brushy Fork, during 1989	6-28
6.6 True rate of growth in weight of bluegill in White Oak Creek and a reference stream, Brushy Fork, during 1989	6-28
6.7 Relative density of all benthic invertebrates at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-37
6.8 Relative density of Chironomids at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-38
6.9 Relative density of Ephemeroptera at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-39
6.10 Relative density of Plecoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-40
6.11 Relative density of Trichoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-41

FIGURES (continued)

Figure	Page	
6.12	Relative benthic macroinvertebrate taxonomic richness of combined EPT taxa at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-42
6.13	Relative benthic macroinvertebrate species richness of the order Ephemeroptera at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-43
6.14	Relative benthic macroinvertebrate species richness of the order Plecoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-44
6.15	Relative benthic macroinvertebrate species richness of the order Trichoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites	6-45
8.1	Location of sampling sites in White Oak Lake for milfoil and duckweed biomass and litter bags experiments	8-3
8.2	Change over time in dry weight, organic weight, and inorganic weight of initially contaminated milfoil in White Oak Lake	8-10
8.3	Change over time in dry weight, ¹³⁷ Cs, and ⁶⁰ Co of milfoil in Northwest Tributary	8-12
8.4	Fraction of the total ¹³⁷ Cs and ⁶⁰ Co associated with milfoil removed by water and salt solutions	8-14
8.5	Canada geese banding sites for June 1989	8-17
8.6	Number of wood ducks observed on White Oak Lake per observation period, October 1987–December 1989.	8-19
8.7	Number of ring-necked ducks observed on Swan Pond near Oak Ridge National Laboratory per observation period, October 1987–January 1990	8-20
8.8	Number of waterfowl observed on White Oak Lake per observation period, October 1987–January 1990	8-20
8.9	Number of Canada geese observed at the Oak Ridge Gaseous Diffusion Plant per observation period, October 1988–January 1990	8-21

FIGURES (continued)

Figure	Page
8.10 Number of Canada geese observed at the Y-12 Plant per observation period, February 1989–January 1990	8-22
8.11 Cesium-137 concentrations in breast tissue of domestic mallards released on White Oak Lake on May 6, 1989.	8-25
8.12 Cesium-137 concentrations in breast tissue of domestic mallards released on White Oak Lake on October 14, 1989	8-25

TABLES

Table	Page	
2.1	Number of days of zero discharge in upper Melton Branch at Melton Branch kilometer 1.93	2-4
2.2	Comparison of the variability in mean daily discharge and of mean discharge per unit area between streams with and without flow augmentation in 1987 and 1988	2-5
2.3	Description of the nine major effluents regulated under the Oak Ridge National Laboratory National Pollutant Discharge Elimination System permit issued on April 1, 1986	2-8
2.4	Number and location of Category I, II, and III outfalls that discharge to White Oak Creek and tributaries	2-10
2.5	Median concentrations of 34 National Pollutant Discharge Elimination System water quality parameters that are routinely monitored below Oak Ridge National Laboratory in Melton Branch at NPDES site X13, in White Oak Creek at NPDES site X14, and at White Oak Dam at NPDES site X15, January–December 1989	2-12
2.6	Mean monthly water temperature in White Oak Creek and tributaries, including First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, January–December 1989	2-19
2.7	Water quality parameters determined for discrete samples collected monthly at nine sites in streams in the White Oak Creek watershed, October 1988–September 1989	2-22
2.8	Water quality parameters determined from samples collected monthly at the nine periphyton monitoring sites in the White Oak Creek watershed, October 1988–September 1989	2-23
2.9	Concentrations of dissolved elements at nine periphyton study sites, based on inductively coupled plasma analysis of discrete water samples collected quarterly during 1989	2-25
3.1	Summary of toxicity test results for the Oak Ridge National Laboratory Process Waste Treatment Plant, Coal Yard Runoff Treatment Facility, and Sewage Treatment Plant, May 1986–January 1990	3-4

TABLES (continued)

Table	Page	
3.2	Typical no-observed-effect concentrations for effluents from the Oak Ridge National Laboratory Process Waste Treatment Plant, Coal Yard Runoff Treatment Facility, and Sewage Treatment Plant in relation to ranges of water quality parameters characteristic of the effluents	3-5
3.3	Results of fathead minnow and <i>Ceriodaphnia dubia</i> chronic toxicity tests of water from National Pollutant Discharge Elimination System sites on Melton Branch and White Oak Creek	3-6
3.4	Within-treatment variation in fathead minnow survival for toxicity tests in which average survival was low	3-8
3.5	Summary of chemistry data for toxicity tests of water from National Pollutant Discharge Elimination System points X13 and X14	3-9
3.6	Results of fathead minnow and <i>Ceriodaphnia dubia</i> chronic toxicity tests of dechlorinated water samples collected from five White Oak Creek outfalls	3-10
3.7	Chemical parameters for five White Oak Creek outfalls	3-11
3.8	Number of <i>Ceriodaphnia</i> neonates surviving in control water and in a dechlorinated sample of water from White Oak Creek Outfall 312	3-12
3.9	Number of <i>Ceriodaphnia</i> surviving in various concentrations of dechlorinated water from White Oak Creek Outfall 312 as a function of time	3-12
3.10	Concentrations of selected trace metals in water from White Oak Creek Outfall 312 and in dechlorinated tap water	3-13
3.11	Frequency of occurrence of three total residual chlorine concentrations at four sites in White Oak Creek and Fifth Creek during 1986–1987 and 1987–1989	3-16
3.12	Results of the analysis of variance of the ranks of sites, within tests, included in ambient toxicity assessments of Oak Ridge National Laboratory streams using <i>Ceriodaphnia</i> and fathead minnow larvae	3-18
3.13	Sites that were significantly better or significantly worse than average based on the analysis of variance of ranks of sites within tests	3-19
3.14	Results of the analysis of variance of the ranks of tests, within sites, included in ambient toxicity assessments of Oak Ridge National Laboratory streams using <i>Ceriodaphnia</i> and fathead minnow larvae	3-19

TABLES (continued)

Table	Page
3.15 Test periods in which toxicity test endpoints were significantly higher or significantly lower than average based on the analysis of variance of ranks of tests within sites	3-20
3.16 Site-to-site and test-to-test R ² values for <i>Ceriodaphnia</i> and fathead minnow toxicity test endpoints for complete and reduced data sets	3-21
3.17 Results of 7-d static-renewal <i>Ceriodaphnia</i> toxicity tests of water from East Tributary and West Tributary	3-22
3.18 Water quality factors measured for Solid Radioactive Waste Disposal/Storage Area 6 streams from grab samples collected on March 2, 1989	3-22
3.19 Annual mean chlorophyll <i>a</i> and photosynthesis for periphyton at nine monitoring sites during 1988 and 1989	3-30
3.20 Chlorophyll-adjusted periphyton photosynthesis at the nine study sites in 1989	3-31
4.1 Mean metal concentrations in fish from White Oak Lake, White Oak Creek and tributaries	4-4
4.2 Concentrations of mercury in bluegill and redbreast sunfish from White Oak Creek and tributaries, 1986-1988	4-8
4.3 Concentrations of polychlorinated biphenyls and other organic priority pollutants in bluegill and redbreast sunfish (<i>Lepomis macrochirus</i> and <i>L. auritus</i>), largemouth bass, and carp from the Clinch River, White Oak Lake, White Oak Creek, Melton Branch, Northwest Tributary, and Melton Hill Reservoir	4-10
4.4 Comparison of mean concentrations of polychlorinated biphenyls in sunfish from White Oak Lake and White Oak Creek, Melton Branch, and Northwest Tributary, December 1987-July 1989	4-12
4.5 Concentrations of chlordane and polychlorinated biphenyls in duplicate composite samples of caged clams held for 4 weeks at sites in White Oak Creek, White Oak Lake, and tributaries; and concentrations of chlordane in sediment fish at sites in the White Oak Creek drainage	4-14
4.6 Concentrations of polychlorinated biphenyls and strontium-90 in channel catfish from sites in the Clinch River arm of Watts Bar Reservoir in the vicinity of U.S. Department of Energy facilities, July/September 1989	4-19

TABLES (continued)

Table	Page	
4.7	Changes from 1986 to 1989 in average concentrations of polychlorinated biphenyls and fraction of fish exceeding the U.S. Department of Agriculture Food and Drug Administration limit, for channel catfish	4-20
5.1	Temporal changes in bioindicator responses of bluegill over the period fall 1987–fall 1988 at two sites in White Oak Creek, White Oak Lake, and the reference stream	5-13
5.2	Importance of each bioindicator in discriminating between the integrated responses of bluegill for various combinations of sites	5-16
6.1	Density, taxonomic richness, mean Ephemeroptera, Plecoptera, and Trichoptera richness, and taxonomic diversity of benthic macroinvertebrates in White Oak Creek watershed, May and June 1987	6-4
6.2	Relative density of dominant benthic invertebrate taxa in the White Oak Creek watershed, May and June 1987	6-6
6.3	Length, mean width, mean depth, surface area, and pool to riffle ratio of fish sampling sites	6-14
6.4	Fish species composition in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, April and October–November 1989	6-19
6.5	Total fish density, total biomass, and species richness for April and October–November 1989 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary	6-22
6.6	Comparison between sampling sites on White Oak Creek and the reference stream, Brushy Fork, of mean true growth of redbreast sunfish and bluegill collected between August 1988 and January 1989	6-27
6.7	Comparison between sampling sites on White Oak Creek and the reference stream, Brushy Fork, of mean true growth of redbreast sunfish and bluegill collected between August 1989 and December 1989	6-29
6.8	Index of biotic integrity values based on sampling conducted during March–April and December 1988 and during April and October–November 1989 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary	6-31

TABLES (continued)

Table	Page
8.1 Mean radionuclide concentrations in aquatic macrophytes collected at the White Oak Creek weir in 1989	8-5
8.2 Mean radionuclide concentrations in aquatic macrophytes collected at the Melton Branch weir in 1989	8-7
8.3 Mean radionuclide concentrations in aquatic macrophytes collected in White Oak Lake in 1989	8-9
8.4 Percent lost per day for fast and slow components of milfoil in White Oak Lake and the Northwest Tributary	8-11
8.5 Species of waterfowl observed on the Oak Ridge Reservation	8-19
8.6 Cesium-137 concentrations in tissues of Canada geese collected on Pond 3524 at Oak Ridge National Laboratory	8-23
8.7 Concentrations of ¹³⁷ Cs in bluegill from the White Oak Creek system, December 1988	8-26

ACRONYMS

ACD	Analytical Chemistry Division
ALT	alanine aminotransferase
ANCOVA	analysis of covariance
ANOVA	analysis of variance
ATDL	Atmospheric Turbulence and Diffusion Laboratory
BaP	benzo[<i>a</i>]pyrene
BCK	Bear Creek kilometer
BFK	Brushy Fork kilometer
BMAP	Biological Monitoring and Abatement Program
BMP	best management practices
CRK	Clinch River kilometer
CV	coefficient of variation
CYRTF	Coal Yard Runoff Treatment Facility
DEM	Department of Environmental Management (name changed in 1987 to DEMC and revised in 1988 to EMC Section, see below)
DEMC	Department of Environmental Monitoring and Compliance
DMSO	dimethylsulfoxide
DMW	diluted mineral water
DNA	deoxyribonucleic acid
DOC	dissolved organic carbon
DOE	U.S. Department of Energy
EDTA	ethylene diamine tetra-acetic (acid)
EFK	East Fork Poplar Creek kilometer
EFPC	East Fork Poplar Creek
EMC	Environmental Monitoring and Compliance Section
EPA	U.S. Environmental Protection Agency
EPT	number of Ephemeroptera, Plecoptera, and Trichoptera per sample
ERDP	Environmental Review and Documentation Program
EROD	7-ethoxyresorufin <i>O</i> -deethylase
ERP	Environmental Restoration Program
ESD	Environmental Sciences Division
FCK	First Creek kilometer
FDA	U.S. Department of Agriculture Food and Drug Administration
FFK	Fifth Creek kilometer
GC/MS	gas chromatography/mass spectrometry
GC/ECD	gas chromatography/electron capture detector
GLM	general linear models
GPP	general plant project
HFIR	High Flux Isotope Reactor
HPLC	high performance liquid chromatography
IBI	index of biotic integrity
IBM	International Business Machines
ICP	inductively-coupled plasma
LFPMA	percentage of liver actually occupied by functional parenchyma

LNPMA	percentage of liver tissue occupied by necrotic parenchyma
LPARS	percentage of liver composed of encysted parasites
LSI	liver-somatic index
MEK	Melton Branch kilometer
MHR	Melton Hill Reservoir
MIK	Mitchell Branch kilometer
MSL	mean sea level
NADH	nicotinamide adenine dinucleotide (reduced form)
NADPH	nicotinamide adenine dinucleotide phosphate (reduced form)
NCR	National Cash Register
NOAA	National Oceanic and Atmospheric Administration
NOEC	no-observed-effect concentration
NPDES	National Pollutant Discharge Elimination System
NRWTF	Nonradiological Wastewater Treatment Facility
NT	Northwest Tributary
NTK	Northwest Tributary kilometer
ORAU	Oak Ridge Associated Universities
ORGDP	Oak Ridge Gaseous Diffusion Plant (now the Oak Ridge K-25 Site)
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
PAHs	polycyclic aromatic hydrocarbons
PAR	photosynthetically active radiation
PC	phosphatidylcholine
PCBs	polychlorinated biphenyls
PCK	Poplar Creek kilometer
PC-SAS	personal computer statistical analysis software
PE	phosphatidylethanoamine
PGV	preliminary guidance value
PWTP	Process Waste Treatment Plant
RAP	Remedial Action Plan
RCRA	Resource Conservation and Recovery Act
REDC	Radiochemical Engineering Development Center
RNA	ribonucleic acid
SAS	Statistical Analysis Software
SD	standard deviation
SE	standard error
SRP	soluble reactive phosphorus
STP	Sewage Treatment Plant
SWSA	solid radioactive waste disposal/storage area
TCMP	Toxicity Control and Monitoring Program
TDHE	Tennessee Department of Health and Environment
TIE	toxicity identification effort
TRC	total residual chlorine
TSS	total suspended solids
TVA	Tennessee Valley Authority
USGS	U.S. Geological Survey
VSI	visceral-somatic index

WBR	Watts Bar Reservoir
WCK	White Oak Creek kilometer
WOC	White Oak Creek
WOD	White Oak Dam
WOL	White Oak Lake
WPCP	Water Pollution Control Program

PREFACE

On April 1, 1986, a National Pollutant Discharge Elimination System permit was issued for the Oak Ridge National Laboratory (ORNL). As required in Part III: Special Conditions (Item H) of the permit, a plan for biological monitoring of the Clinch River, White Oak Creek, Northwest Tributary of White Oak Creek, Melton Branch, Fifth Creek, and First Creek was prepared and submitted for approval in July 1986 to the U.S. Environmental Protection Agency and the Tennessee Department of Health and Environment. The plan is referred to as the ORNL Biological Monitoring and Abatement Program (BMAP) and

describes biomonitoring activities and characterization studies to be conducted for the 5-year duration of the permit.

This report is structured around the six major tasks that currently comprise the BMAP (see Table of Contents, Sects. 3.0-8.0). It is the fourth of a series of reports and presents the results of BMAP studies conducted in 1989. These reports also address any significant modifications in the scope of work outlined by J. M. Loar et al. in the sampling plan entitled *Oak Ridge National Laboratory Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River, 1991* (ORNL/TM-10370).

ACKNOWLEDGMENTS

We thank R. D. Bailey, P. W. Braden, D. K. Cox, G. J. Haynes, W. C. Kyker, C. M. Morrissey, L. M. Stubbs, and L. F. Wicker of the Environmental Sciences Division (ESD) of Oak Ridge National Laboratory (ORNL); H. Schöne (ORNL Physics Division); J. W. Evans, Tennessee Wildlife Resources Agency; W. M. Harris (Y-12 Plant Product Certification Division); B. C. Harvey (Weber State College); T. A. Anderson (University of Tennessee); M. S. Jen (California State University); and J. G. Stout (Denison University) for their assistance with the field sampling. Substantial support was provided by the staff of the toxicity testing laboratory, including L. A. Kszos, R. D. Bailey, P. W. Braden, G. J. Haynes, and L. F. Wicker of ESD; and L. S. Ewald, G. P. Morris, and J. Richmond of the Oak Ridge Research Institute. We thank N. M. Ferguson, J. W. Wade, T. B. Shope, and M. P. Maskarinec of the ORNL Analytical Chemistry Division (ACD); and W. McDaniels, W. Loy, and T. Bennett of the U.S. Environmental Protection Agency's Environmental Services Laboratory, Athens, Georgia, for sample analyses. We thank I. L. Larsen and M. L. Frank of ESD for guidance in gamma counting. We are grateful to J. S. Eldridge (ACD) for providing the

expertise and equipment for conducting the whole-body counts on Canada geese. We thank K. Newman (Science Applications International Corporation) for analyses of the National Pollution Discharge Elimination System data. Statistical assistance was provided by J. J. Beauchamp (ORNL Computing and Telecommunications Division) and W. R. Hill (Oak Ridge Associated Universities). We also thank B. F. Clark, W. C. Dickinson, A. W. McWhorter, and J. A. Wojtowicz of JAYCOR for taxonomic identification and enumeration of the benthic invertebrates. Finally, we are grateful to G. F. Cada and J. R. Trabalka of ESD and to C. K. Valentine of the ORNL Environmental Monitoring and Compliance (EMC) Section, who reviewed the draft report and provided many helpful comments and suggestions; to W. Bryant for editorial support, and to P. Henry for electronic processing.

This work was jointly funded by the EMC Section of the ORNL Environmental and Health Protection Division and the ORNL Environmental Restoration Program. Some of the studies described in the report were conducted in the Oak Ridge National Environmental Research Park.

EXECUTIVE SUMMARY

As a condition of the National Pollutant Discharge Elimination System (NPDES) permit issued to Oak Ridge National Laboratory (ORNL) on April 1, 1985, a Biological Monitoring and Abatement Program (BMAP) was developed for White Oak Creek (WOC); selected tributaries of WOC, including Fifth Creek, First Creek, Melton Branch, and Northwest Tributary; and the Clinch River. BMAP currently consists of six major tasks that address both radiological and nonradiological contaminants in the aquatic and terrestrial environs at ORNL. These tasks are: (1) toxicity monitoring, (2) bioaccumulation monitoring of nonradiological contaminants in aquatic biota, (3) biological indicator studies, (4) instream ecological monitoring, (5) assessment of contaminants in the terrestrial environment, and (6) radioecology of WOC and White Oak Lake (WOL).

TOXICITY TESTING AND COMMUNITY STUDIES (TASKS 1, 3, AND 4)

Toxicity testing was conducted on both wastewater effluents and receiving streams in 1989. Effluent toxicity evaluations on each of three ORNL wastewater treatment facilities were continued at a reduced frequency. Ambient (instream) toxicity was evaluated in seven tests each on water from NPDES site X13 on lower Melton Branch at Melton Branch kilometer (MEK) 0.16 and on water from NPDES site X14 on lower WOC at White Oak Creek kilometer (WCK) 2.65. In addition, two small tributaries that drain the east and west portions of Solid Waste Storage Area (SWSA) 6 and enter WOL were tested in

March 1989. All other evaluations were based on 7-d static-renewal tests that used the survival and growth of fathead minnow (*Pimephales promelas*) larvae, and/or the survival and reproduction of a small crustacean (*Ceriodaphnia dubia/affinis*) as toxicity endpoints. Results indicated that the effluents from the Process Waste Treatment Plant and the Sewage Treatment Plant (STP) were occasionally toxic to one or both species, and the effluent from the Coal Yard Runoff Treatment Facility was usually toxic to *Ceriodaphnia* and occasionally toxic to fathead minnows. No conclusive evidence of toxicity was found in the ambient tests of water from WOC, Melton Branch, or the SWSA 6 streams.

An evaluation of the ambient toxicity data sets generated since 1986 was completed. This exercise concluded that assessments of ambient toxicity would provide more reliable results if *Ceriodaphnia* tests were used more extensively than the fathead minnow tests. In ambient applications, the latter test was found to be subject to interferences from pathogenic microorganisms, which greatly reduced its value.

Studies were also conducted to evaluate the toxicological significance of total residual chlorine (TRC) in receiving streams. Toxicity tests were used to determine whether toxicants other than chlorine were present in some wastewaters where residual chlorine was toxic to test organisms. Three of the five outfalls tested were found to contain evidence of other toxicants. Additional tests on one of these outfalls (No. 312) indicated that metals (possibly zinc or copper) probably accounted for the toxicity remaining after dechlorination.

Another study investigated the persistence of residual chlorine in receiving waters as a function of location and water chemistry. The rate of loss of TRC from WOC water in laboratory batch experiments was influenced more by temperature than water chemistry and was much slower than the observed downstream pattern of chlorine persistence in WOC.

Surveys of the biological communities in streams near ORNL continued to demonstrate that the impacts of plant operation are greatest in Fifth Creek and the middle reaches of WOC (from the Building 5505 bridge near WCK 4.8 downstream to Melton Valley Drive at WCK 3.4). Sites in this reach of WOC had higher algal periphyton biomass and primary production rates compared to upstream reference sites, which primarily occurred as a result of increased nutrient loading from ORNL effluents and reduced grazing due to reduced numbers of invertebrates. Adverse impacts on algal periphyton seem restricted to areas near sampling sites WCK 3.9, 3.4, and, occasionally, 2.9. Chlorophyll-adjusted production rates suggest that ecological conditions improved with distance downstream of ORNL discharges (e.g., WCK 2.3 and MEK 0.6). Toxicity tests confirmed that those species of algae found at sites impacted by residual chlorine are less sensitive to this toxicant than other species in WOC.

The distribution and composition of the fish communities in the WOC watershed were similar to those observed in 1988. Adverse impacts on fish abundance and species richness remain most obvious at sites affected by chlorinated discharges (Fifth Creek kilometer 0.2 and WCK 3.9). Sites where impacts had been observed previously (WCK 5.1, MEK 1.4, and MEK 0.6) continued to demonstrate normal fish

abundances and species richness in 1989. Chlordane contamination continued to decline at the WCK 5.4 site following its discovery in spring 1988, and continued shutdown of the High Flux Isotope Reactor (HFIR) undoubtedly contributed to favorable ecological conditions at the lower Melton Branch sites.

Two indices of stream health were applied to WOC and tributaries to assess the current ecological status of the stream communities. The EPT richness index is based on the number of benthic invertebrate species in the orders Ephemeroptera, Plecoptera, and Trichoptera, which contain many pollution-intolerant species. On each tributary of WOC except Melton Branch (i.e., First Creek, Fifth Creek, and Northwest Tributary), the EPT index was lower at the downstream site as compared to the upstream reference site. In WOC, EPT indices at all downstream sites were lower than the index at the upstream reference site. A general trend of increasing EPT values with distance downstream from WCK 3.9 was noted. A comparison of the taxonomic richness of each of these three orders in WOC and tributaries with that found in previous surveys of the Tennessee Valley region provided further documentation of the adverse impact of ORNL operations on the benthic macroinvertebrate communities in the WOC watershed.

The index of biotic integrity (IBI) is a measure of a stream's capacity to support and maintain a "balanced, integrated, and adaptive community of organisms" similar in its structural and functional components to that of the natural habitat of the region. The index consists of 12 metrics that provide measures of species richness and composition, trophic composition, and fish abundance and condition. On a possible scale from 12 to 60 (equivalent to ratings of very poor and excellent respectively),

the WOC watershed sites ranged from 16 (Northwest Tributary kilometer 1.0) to 30 (upper Fifth Creek and WCK 2.3) in 1989. All sites, including upstream reference sites, were classified as "poor" or "very poor." The ability of the IBI to discriminate among toxicant impacted and unimpacted sites in small, headwater streams with simple fish communities appears somewhat limited.

The health of individual fish populations in the WOC watershed was also assessed. The overall health of bluegill collected from Melton Branch, WOC (WCK 3.6 and 3.2), WOL, and Hinds Creek was evaluated using a suite of biochemical and physiological parameters. These parameters were grouped into five functional categories that represented indicators of (1) lipid metabolism, (2) detoxification enzyme induction, (3) histopathological condition, (4) organ dysfunction, and (5) overall fish health or condition. The bluegill population in WOC/WOL was generally in poorer overall health than that in Hinds Creek. Significant differences were observed between reference stream fish and those from WOC/WOL for most of the parameters included in categories (2) and (5). These results suggest that fish in WOC had elevated detoxification enzymes, indicating exposure to hydrocarbons or other organic contaminants, and may suffer from liver and possibly kidney dysfunction. Nevertheless, sunfish populations in WOC do not appear to differ in abundance, age distribution, or growth rate from those of unimpacted reference streams.

The significant ecological recovery of lower Melton Branch that was observed in 1987-88 following the shutdown of HFIR in November 1986 continued in 1989. The periphyton production rate at MEK 0.6 remained high, and the steady increase in

fish abundances in 1987 and 1988 continued in 1989.

In addition to continued routine monitoring to assess changes in ecological conditions, future studies will: (1) develop the capability to predict instream chlorine concentrations, (2) characterize the magnitude and function of the benthic microbial flora of WOC, (3) evaluate the reproductive competence of fish in WOC, and (4) assess the role of elevated stream temperatures and removal of riparian vegetation in determining fish and invertebrate community structure.

BIOACCUMULATION STUDIES (TASKS 2, 5, AND 6)

Results from the monitoring of polychlorinated biphenyls (PCBs) in channel catfish in 1989 supported previous conclusions [i.e., some channel catfish caught by anglers in the Clinch River near WOC are likely to contain PCBs in excess of the U.S. Department of Agriculture Food and Drug Administration (FDA) limit of 2 $\mu\text{g/g}$]. Five of 16 catfish (31%) collected from the WOC embayment and the Clinch River just downstream of WOC exceeded this limit compared to 15, 25, and 19% in 1986, 1987, and 1988 respectively. A significant fraction, although not necessarily most, of the PCB content of the catfish from the Clinch River near WOC originates in the WOC discharge and/or the WOC embayment. The concentrations of PCBs found in catfish at Clinch River and embayment sites in 1989 are consistent with the Precautionary Fish Consumption Advisory currently in effect for this reach of Watts Bar Reservoir.

Concentrations of PCBs in bluegill and redbreast sunfish collected in 1988-89 at sites in the WOC watershed were similar to those found previously. Largemouth bass contained substantially higher

concentrations of PCBs than sunfish in WOL, indicating that additional monitoring of PCB contamination in this species in WOL is advisable.

Chlordane contamination in upper WOC was confirmed. The presence of chlordane in riparian zone soil, low levels of contamination in clams held in WOC near the 7000 Area, and higher levels in clams in and downstream from the tributary at WCK 5.4 suggest that contributions come from a general area source with local hot spots near the tributary. No evidence was found of elevated chlordane concentrations in fish in WOC or the Clinch River as a result of ORNL releases.

Mercury concentrations in fish from WOC were elevated over those in fish from reference streams in winter 1988 but were well below the FDA action level throughout the drainage. Elevated concentrations in fish were generally restricted to those WOC sites nearest ORNL facilities, with little or no contamination at sites downstream of WOL and into the Clinch River. The level of contamination in WOC was similar to that observed previously. It is low when compared to upper East Fork Poplar Creek (EFPC) but similar to that at sites in lower EFPC and Poplar Creek.

A comprehensive evaluation of the use of small mammals to evaluate the transport and effects of contaminants at hazardous waste sites was completed. This evaluation was based on data from the ORNL BMAP and an extensive literature survey. The evaluation presents a critical review of the advantages and disadvantages of using small mammals to monitor specific contaminants and to predict the likelihood of ecological effects.

Radioecological studies of WOL are being conducted to evaluate the significance of human exposure to contaminants in WOL and, thus, determine

whether remedial actions are warranted under current conditions or under hypothetical loss of institutional control. Because they may remobilize radionuclides from lake sediments to the overlying water column, macrophyte populations in WOL, WOC, and Melton Branch were investigated with regard to their role in the transport and fate of radionuclides in the WOC system. Rooted aquatic vegetation was found to play a significant role in the transport of ^{137}Cs and ^{60}Co from sediments back into the water. Approximately 40% of the ^{137}Cs and 60% of the ^{60}Co in milfoil (*Myriophyllum spicatum*) were found to be sediment derived. The radionuclides contained in these plants are available to herbivores and are released to the water in dissolved and particulate form upon senescence of the plant community.

The weekly census initiated in fall 1987 of waterfowl populations on WOL and ten other radioactive waste ponds and contaminated areas near ORNL continued in 1989 but was expanded to include sites at the Y-12 Plant, the Oak Ridge Gaseous Diffusion Plant (currently the K-25 Site), and several sites off the Oak Ridge Reservation (ORR). The study was designed to evaluate the potential transport of radionuclides from these areas into the human food chain via ingestion of waterfowl. Substantial numbers of waterfowl were found to winter on the ORR, especially on WOL. Over 100 ducks were observed on WOL during each of the past three winters. Apparently migratory ducks using WOL rapidly accumulate ^{137}Cs to near steady-state concentrations. Average concentrations of ^{137}Cs in the breast tissue of migratory ducks collected on WOL averaged 234 Bq/kg, while concentrations in resident wood ducks (adult and immature birds) and domestic mallards released and maintained in WOL for 2-3 months averaged 131 and 316 Bq/kg respectively.

A pair of geese collected after using Basin 3524 at ORNL for 1 month contained very high concentrations of ^{137}Cs (106,000 and 150,000 Bq/kg in breast tissue). Other geese collected at ORNL near the STP basins averaged 101 Bq/kg ^{137}Cs in breast tissue.

Concentrations of ^{137}Cs in sunfish from WOC and WOL in 1988 were similar to those observed previously. Higher average concentrations and greater year-to-year variations were observed at sites near the main ORNL complex. Mean concentrations ranged from 18,250 Bq/kg (dry weight) at WCK 3.5 to 2775 and 1550 Bq/kg in fish from WOL and the WOC embayment respectively.

ABATEMENT PROGRAM

Abatement efforts at ORNL are directed toward providing both short- and long-term management and technical solutions to water quality problems, including toxicity of some receiving streams. Abatement projects include chlorine reduction, substitution for ethylene glycol in cooling systems, tank diking, elimination of Category III outfalls, PCB and mercury monitoring, wastewater piping repair, construction of the Nonradiological Wastewater Treatment Facility, and enhanced environmental surveillance of construction projects, including development and implementation of best management practices plans by staff of the ORNL Environmental Monitoring and Compliance Section.

1. INTRODUCTION

J. M. Loar

In response to a condition of the National Pollutant Discharge Elimination System (NPDES) permit issued to Oak Ridge National Laboratory (ORNL) on April 1, 1986, a Biological Monitoring and Abatement Program (BMAP) was developed for White Oak Creek (WOC); selected tributaries of WOC, including Fifth Creek, First Creek, Melton Branch, and Northwest Tributary (NT); and the Clinch River (Loar et al. 1986).

BMAP currently consists of six major tasks that address both radiological and nonradiological contaminants in the aquatic and terrestrial environs on-site and the aquatic environs off-site. These tasks are (1) toxicity monitoring, (2) bioaccumulation monitoring of nonradiological contaminants in aquatic biota, (3) biological indicator studies, (4) instream ecological monitoring, (5) assessment of contaminants in the terrestrial environment, and (6) radioecology of WOC and White Oak Lake (WOL).

The investigation of contaminant transport, distribution, and fate in the WOC embayment-Clinch River-Watts Bar Reservoir system was originally a BMAP task (Task 7; see Loar et al. 1991) but, in 1988, was incorporated into the Resource Conservation and Recovery Act (RCRA) Facility Investigation for the Clinch River (a separate study) to assess off-site contamination from all three U.S. Department of Energy (DOE) facilities in Oak Ridge.

1.1 OBJECTIVES

BMAP was developed to meet three objectives. First, studies (tasks) were designed to provide sufficient data to determine whether the effluent limits established for ORNL protect and maintain the classified uses of WOC, Melton Branch, NT, First Creek, and Fifth Creek. These streams have been classified by the Tennessee Department of Health and Environment (TDHE) for (1) growth and propagation of fish and aquatic life, (2) irrigation, and (3) livestock watering and wildlife (EPA 1986a).

Second, BMAP will provide ecological characterizations of WOC and tributaries and of WOL that can be used to (1) document ecological impacts of past and current operations and (2) identify contaminant sources that adversely affect stream biota. This ecological information will be important in the development of various Remedial Investigation/Feasibility Study plans and reports and in the assessment of remedial action alternatives as part of the RCRA planning process within the ORNL Environmental Restoration Program (ERP).

Third, BMAP will document the effects on stream biota from implementation of the Remedial Action Plan (RAP) and the Water Pollution Control Program (WPCP). These two ORNL programs are described in detail in Bates et al. (1988) and are summarized in

Berry et al. (1987). The major remedial action included in the latter program was construction of a Nonradiological Wastewater Treatment Facility (NRWTF); operational start-up of this facility occurred on April 1, 1990. The ecological characterization of the WOC watershed will provide baseline data that can be used

to document the ecological effects of the WPCP and the RAP and to determine the success of remedial actions implemented under these programs. The long-term nature of BMAP ensures that the effectiveness of remedial measures will be properly evaluated.

2. DESCRIPTION OF THE WHITE OAK CREEK WATERSHED

R. D. Bailey, H. L. Boston, J. M. Loar, and G. R. Southworth

The WOC watershed is located near the southern boundary of the 150-km² DOE Oak Ridge Reservation (ORR). The watershed has a drainage area of 16.9 km² at its mouth at Clinch River kilometer (CRK) 33.5, (CRK 0.0 is at the confluence of the Clinch and Tennessee rivers) and is similar in size to the Bear Creek watershed (20.1 km²) near the Y-12 Plant (Evaldi 1986). Parallel northeast-trending ridges constitute the northern and southern borders of the watershed, and a third ridge (Haw Ridge) bisects the basin and separates Bethel Valley to the north from Melton Valley to the south (Fig. 2.1). Elevations in the watershed range from

226 m above mean sea level (MSL) at the mouth of WOC to 413 m MSL on Melton Hill at the crest of Copper Ridge, the highest point on the ORR (McMaster 1963, McMaster and Waller 1965).

Because of dam construction, three distinct environments can be identified within the WOC watershed: (1) WOL, (2) the WOC embayment below the lake, and (3) WOC and tributaries above the lake. The lake was created in 1941 by construction of a small highway-fill dam ~1.0 km above the confluence of WOC and the Clinch River (Fig. 2.1). It is a shallow impoundment that extends ~0.7 km upstream from the dam and has a surface

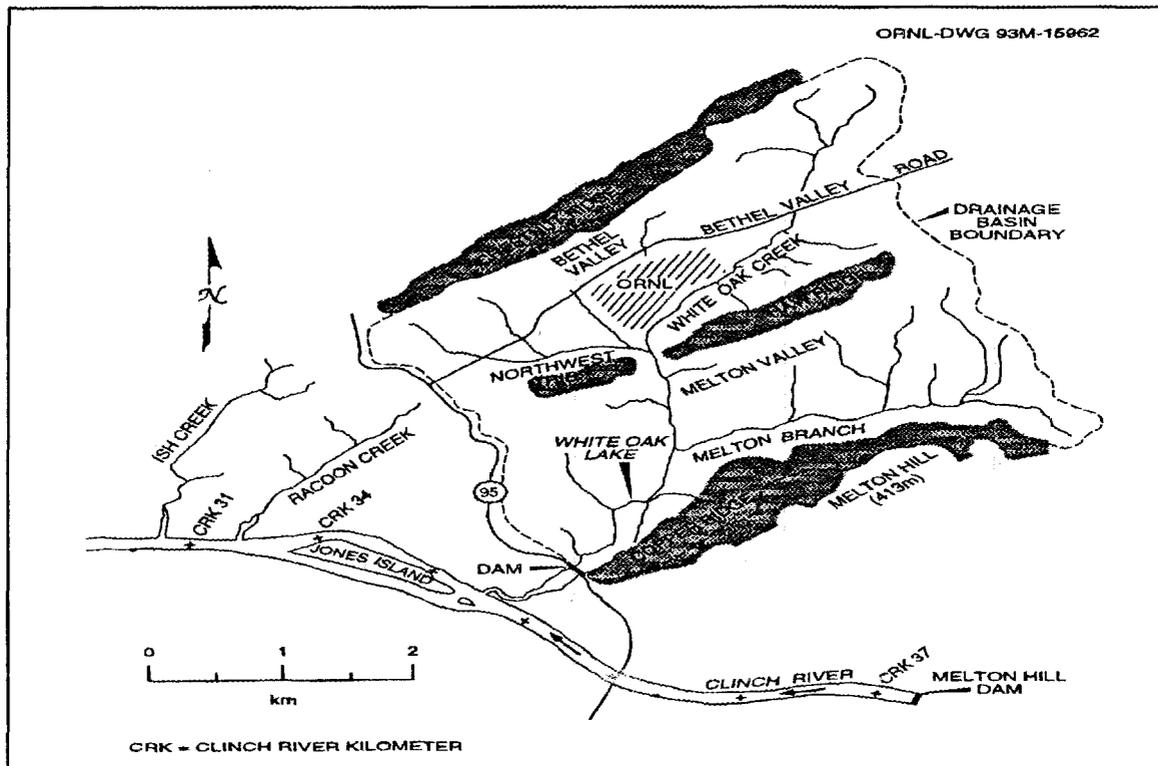


Fig. 2-1. Map of the White Oak Creek watershed above White Oak Dam.

area of ~8 ha at a lake elevation of 227.1 m MSL.

The water level in WOC embayment is controlled by the operation of Melton Hill Dam at CRK 37.2 and Watts Bar Dam, which is located at Tennessee River kilometer 852.6, ~61 km below the confluence of the Clinch and Tennessee rivers. When Watts Bar Reservoir (WBR) is maintained at or near full pool (approximately April to October) and discharges occur at Melton Hill Dam, the subsequent water level rise in the Clinch River creates an embayment extending from the mouth of the creek to White Oak Dam (WOD). Because of this regulated condition, the WOC watershed is generally considered to be limited to the 15.5-km² area above the dam (Edgar 1978).

The region of the watershed above WOL is emphasized in the discussion that follows. Further descriptions of the WOL, WOC embayment, and Clinch River environments are provided in Loar et al. (1981a), Boyle et al. (1982), Oakes et al. (1982), and Sherwood and Loar (1987) and in Sects. 4.0 and 7.0–9.0 of this report.

2.1 GEOHYDROLOGY (*J. M. Loar*)

The headwaters of WOC originate on the southeast slope of Chestnut Ridge (Fig. 2.1). The belt of Knox Dolomite underlying the ridge is the principal water-bearing formation, and springs that occur along the base of Chestnut Ridge and in its valleys are the chief sources of base flow to upper WOC (McMaster and Waller 1965). The largest tributary of WOC is Melton Branch, which originates at the eastern end of Melton Valley and joins WOC at White Oak Creek kilometer (WCK) 2.49 (Fig. 2.2), ~500 m above WOL (at a lake elevation of 227.1 m MSL). Most of the Melton Branch drainage basin is underlain by the Rome

Formation (Haw Ridge), which is composed principally of siltstone and shale, and by the Conasauga Group (Melton Valley), a primarily calcareous shale interlayered with limestone and siltstone (McMaster 1963, 1967); both are poor water-bearing formations (McMaster and Waller 1965). Together, the two formations comprise 95% of the surface area of the Melton Branch watershed, whereas all of the upper WOC watershed north of Bethel Valley Road is underlain by Knox Dolomite (McMaster 1967, Table 10).

The hydrology of upper Melton Branch reflects the underlying geology of the watershed. Base-flow discharge is typically low with periods of no flow at times (McMaster 1967). Precipitation was below normal in each of the 4 years preceding 1989, and extended no-flow periods occurred in 1986–88 at the U.S. Geological Survey (USGS) gaging station on upper Melton Branch at Melton Branch kilometer (MEK) 1.93. Rainfall in 1989 was well above average, and far fewer days of zero discharge occurred than did during the previous 3 years (1986–88) (Table 2.1). Nevertheless, the inability of the Melton Branch watershed to provide a stable base flow during hot summer months was demonstrated by the occurrence of 29 d of zero flow in July, August, and September, despite above-normal precipitation during the preceding winter and spring, when most groundwater recharge occurs.

Annual precipitation in 1989, as measured at the Atmospheric Turbulence and Diffusion Laboratory (ATDL) in the city of Oak Ridge, was 167.7 cm (120% of normal for the 1951–80 record period), the highest since 1984 (143.6 cm or 103% of normal). As a result of the higher rainfall, the average flow in all streams in 1989 was much greater than in 1988 (Table 2.2). Because relatively stable discharges from ORNL augment flow in WOC and some of its tributaries, the relative change in mean

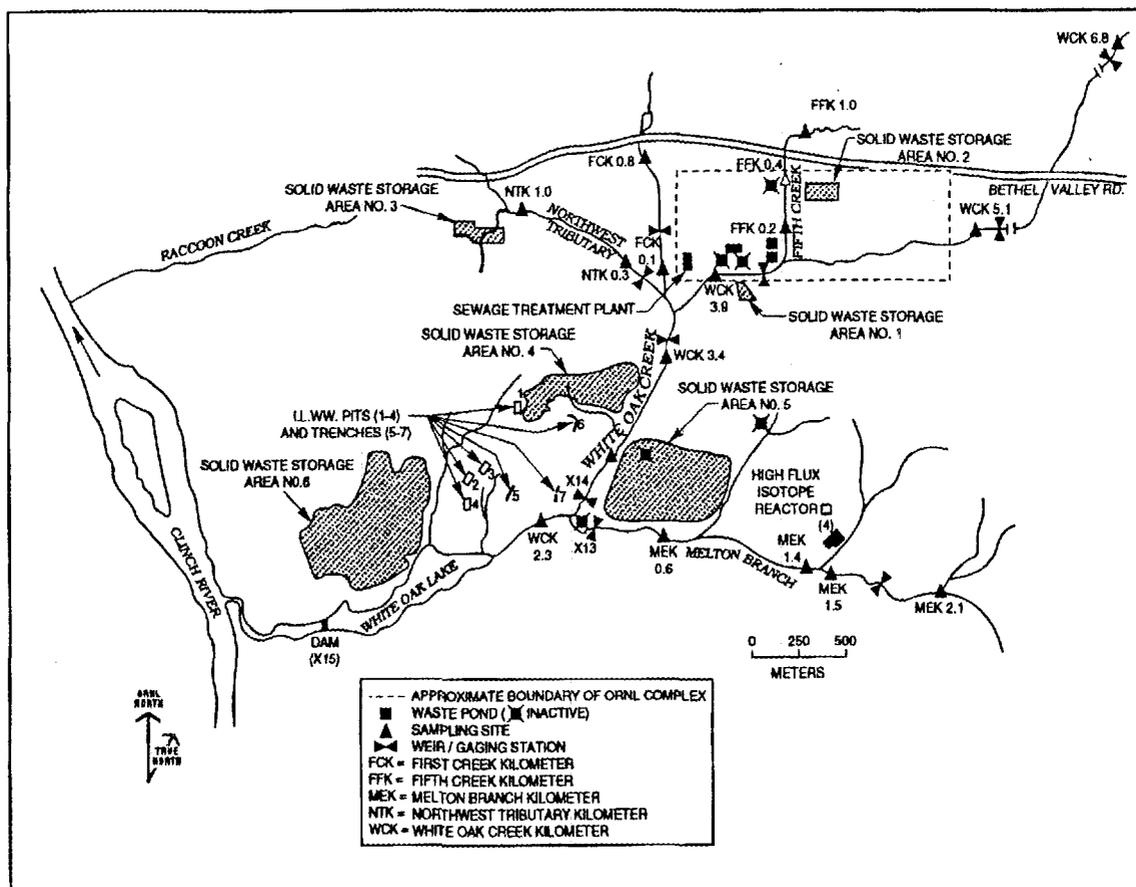


Fig. 2.2. Map showing locations of liquid and solid waste disposal areas; National Pollutant Discharge Elimination System ambient monitoring stations on Melton Branch (X13), White Oak Creek (X14), and White Oak Dam (X15); and sampling sites for benthic invertebrates and fish in White Oak Creek watershed. (Δ = fish sampling only.) No data were available on stream flows at the gaging station on upper White Oak Creek near White Oak Creek kilometer 6.8.

annual flow between 1988 and 1989 was greater at headwater sites and small reference streams [WCK 5.47, NT 14, NT 15, and East Tributary (ET)1] than downstream sites on WOC, Fifth Creek, and NT (Table 2.2).

Stream flow in lower Melton Branch is augmented by periodic discharges of several process waste basins and cooling tower blowdown from the High Flux Isotope Reactor (HFIR) (Sect. 2.2.1). The discharges, which enter the stream at MEK 1.59 via an unnamed tributary

(Fig. 2.2), are a significant fraction of the flow in Melton Branch. However, these discharges were substantially reduced in November 1986 following shutdown of the reactor. Continued shutdown of HFIR coupled with higher stream flows (Fig. 2.3) likely resulted in less impact on water quality in Melton Branch in 1989 than in any year since the start of BMAP.

The hydrology of the headwater region of WOC above ORNL could be expected to differ from that of upper Melton Branch due to differences in the geologies of the

Table 2.1. Number of days of zero discharge in upper Melton Branch at Melton Branch kilometer 1.93 (U.S. Geological Survey gaging station 03537100)

	Number of days of zero discharge (number of consecutive days)								Total
	May	June	July	August	September	October	November	December	
1985	0	4 (2)	8 (6)	0	2 (2)	0	0	0	14 (6)
1986	0	10 (10)	15 (15)	31 (31)	27 (24)	20 (12)	0	0	103 (47)
1987	0	11 (8)	21 (19)	31 (31)	30 (30)	31 (31)	30 (30)	18 (14)	172 (155)
1988	2 (2)	30 (30)	21 (12)	30 (26)	25 (11)	31 (31)	3 (3)	0	142 (45)
1989	0	0	1 (1)	21 (17)	7 (5)	0	0	0	29 (17)

Source: Lowery, J. F. et al., 1986. *Water Resources Data for Tennessee, Water Year 1985*, Report No. UGS/WRD/HD-86, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F. et al., 1987. *Water Resources Data for Tennessee, Water Year 1986*, Report No. UGS/WRD/HD-87/225, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F. et al., 1988. *Water Resources Data for Tennessee, Water Year 1987*, Report No. UGS/WRD/HD-88/236, U.S. Geological Survey, Nashville, Tennessee; Lowery, J. F. et al., 1989. *Water Resources Data for Tennessee, Water Year 1988*, Report No. UGS/WRD/HD-89/258, U.S. Geological Survey, Nashville, Tennessee; and U.S. Geological Survey provisional data from L. D. Voorhees, Oak Ridge National Laboratory.

two areas. For example, drainage basins underlain by limestone and dolomite generally have higher unit-area low-flow discharges than those underlain by sandstone and shale (McMaster 1967). Although extended periods of zero discharge commonly occur in upper Melton Branch and NT, no periods of zero flow have been observed at sampling site WCK 6.8 north of Bethel Valley Road or at sites near the headwaters of other tributaries, such as First Creek and Fifth Creek, which also originate in the Knox Dolomite of Chestnut Ridge. The watershed area of upper WOC at WCK 6.8 is 2.07 km² and is similar in size to that of upper Melton Branch at the USGS gaging station (1.35 km²). Several extended periods of no flow occurred during the summer of 1979 in a reach of WOC just north of Bethel Valley Road near WCK 6.3 (Loar et al. 1981a).

Like lower Melton Branch, stream flow in WOC below ORNL is augmented by discharges from various facilities (Sect. 2.2.1). Low-flow measurements have shown that approximately 90% of the dry-weather discharge of the creek originates as groundwater discharge from the Knox Dolomite of Chestnut Ridge, the Chickamauga Limestone of Bethel Valley, and from ORNL plant effluent (McMaster 1967). Plant effluent provides a substantial portion of the flow in WOC and accounted for more than 80% of the flow at WCK 3.9 in 1988 (estimate is based on a comparison of the discharge per square kilometer at stations WCK 5.47, which is located above most ORNL discharges, and WCK 3.9; see Table 2.2).

An indirect estimate of the relative importance of flow augmentation can be obtained from the coefficient of variation (CV) of stream discharge (Table 2.2). The

Table 2.2. Comparison of the variability in mean daily discharge and of mean discharge per unit area between streams with and without flow augmentation in 1987 and 1988^c

ND = no data available

Drainage/site ^b	USGS ^c site ID	Drainage area (km ²)	Mean daily discharge (CV) ^d (L/s)		Discharge per unit area (L·s ⁻¹ ·km ⁻²)	
			1988 ^e	1989 ^f	1988	1989 ^f
White Oak Creek						
WCK 5.47	03536320	3.39	14.6 (350.2)	63.1 (171.4)	4.3	18.6
WCK 3.9	03536380	5.31	114.9 (88.2)	224.1 (89.7)	21.6	42.2
WCK 3.54	03536550	8.50	224.3 (70.9)	393.3 (76.9)	26.4	46.3
WCK 2.65 ^g	X14 ^h	9.53	226.4 (76.9)	407.2 (86.0)	23.8 ⁱ	42.7
Melton Branch						
MEK 1.93 ^h	03537100	1.35	9.1 (469.3)	28.9 (183.5)	6.7	21.4
MEK 0.16 ^g	X13 ^h	3.83	35.1 (334.6)	155.0 (256.1)	9.2 ⁱ	40.5
Melton Branch tributaries^{jk}						
MEK 2.25	3537300	0.62	3.0 (401.8)	10.2 (188.1)	4.8	16.5
MEK 1.89	3537200	0.18	1.6 (371.0)	4.8 (176.1)	8.9	26.7
MEK 1.59	3537050	0.39	4.6 (389.9)	14.7 (191.3)	11.8	37.7
First Creek						
FCK 0.2	03536450	0.85	16.2 (109.8)	36.1 (100.8)	19.1	42.5
Northwest Tributary						
NTK 0.2	03536440	1.74	25.0 (106.9)	49.2 (128.9)	14.4	28.3
East Fork Poplar Creek						
EFK 5.3	03538250	50.5	1099.7 (147.7)	ND ^l	21.8 ⁱ	ND
Bear Creek^k						
BCK 6.24	035382673	8.29	58.8 (356.7)	217.7 (163.8)	7.1	26.3
BCK 4.55	03538270	11.03	91.9 (238.9)	332.1 (143.6)	8.3	30.1
BCK 3.88	03538273	12.95	101.4 (297.5)	398.4 (150.2)	7.8	30.8
Bear Creek tributaries^{jk}						
NT 14 (BCK 6.24)	035382672	0.78	4.8 (425.4)	16.9 (227.5)	6.2 ⁱ	21.7
NT 15 (BCK 5.32)	035382677	0.36	2.3 (417.0)	10.1 (169.2)	6.4	28.1
ET 1 (BCK 4.07)	03538272	0.36	2.0 (298.0)	7.3 (186.6)	5.6	20.3

^aEast Fork Poplar Creek and Bear Creek are located just north of Chestnut Ridge at the east and west end, respectively, of the Oak Ridge Y-12 Plant.

^bSite designations refer to the distance (km) above the mouth of the stream. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer; FCK = First Creek kilometer; NTK = Northwest Tributary kilometer; EFK = East Fork Poplar Creek kilometer; BCK = Bear Creek kilometer; NT = Northwest Tributary; and ET = East Tributary.

^cUSGS = U.S. Geological Survey.

^dCoefficient of variation (CV) = 100 SD/x.

^eValues are based on 366 d of record at all sites except WCK 2.65 (246 d), MEK 0.16 (249 d) and EFK 5.3 (182 d). Station was discontinued after June 30, 1988.

^fValues are based on the following days of record: 249 d at WCK 2.65 and MEK 0.16; 273 d at MEK 2.25, NT 15, and ET 1; 342 d at MEK 1.59; 344 d at BCK 4.55; 346 d at FCK 0.2; 356 d at NT 14; 357 d at NTK 0.2; 361 d at WCK 3.54; and 365 d at the remaining six sites.

^gDischarge data at this site are collected by the ORNL Environmental Monitoring and Compliance Section.

Table 2.2 (continued)

^hValues are based on one discharge measurement each day, excluding weekends and holidays.

ⁱValue is based on less than 300 d of record (see footnotes e and f).

^jStream has no flow augmentation at this location.

^kLocation of the confluence of the tributary with Melton Branch or Bear Creek (in parentheses) is given in kilometers.

^lND = No data available.

CV is defined as the sample standard deviation (SD) expressed as a percentage of the sample mean (Steel and Torrie 1960) or $CV = \frac{100SD}{\bar{x}}$. Because the temporal variability of effluent discharges is usually much lower than that of natural stream flows, streams with substantial flow augmentation exhibit a much lower CV (or higher flow constancy) than streams without flow augmentation. For example, in natural streams, upper Melton Branch, the CV usually ranges from 200 to 400%, but streams that receive plant discharges generally have a CV of <200% and often <100%. For streams in WOC watershed that receive discharges from ORNL, the CV ranged from 77% at WCK 3.54 to 130% in NT in 1989 (Table 2.2).

A much greater frequency of high flows, and relative infrequency of low flows, characterize the comparison of hydrographs for Melton Branch and WOC in 1989 with those in 1985–88 (Fig. 2.3). Four major storms (>5 cm of precipitation in 24 h) occurred during 1989 compared with five in 1988, but rainfall events in 1989 were common and relatively uniformly distributed through the year. The mean annual flow of most streams in 1989 was more than double that in 1988 while the variability in flow was about 50% lower in 1989 in those streams without significant flow augmentation (Table 2.2). Stream flows on the ORR in 1989 were probably close to optimum for this region with respect to ecological conditions. Aquatic biota were not subjected to loss of habitat and elevated water temperatures

due to low flows, nor were they subjected to habitat disruption and physical displacement associated with spates.

2.2 WATER QUALITY (*J. M. Loar, H. L. Boston, G. R. Southworth, and R. D. Bailey*)

ORNL is centrally located in the upper WOC watershed (Fig. 2.1). Although most of the ORNL complex is situated in Bethel Valley, some facilities are located in Melton Valley (Fig. 2.3). WOC and several tributaries (First Creek, Fifth Creek, and NT) are located within or adjacent to the main plant area and receive effluents from various ORNL operations.

2.2.1 Description of Oak Ridge National Laboratory Effluents

Wastewater discharges at ORNL are generated by the operation of nuclear reactors, chemical pilot plants, research laboratories, radioisotope production laboratories, and various support facilities. Currently, all the reactors except HFIR, which resumed operations on January 29, 1990, and all the chemical pilot plants are shut down. Discharges from ORNL facilities include sanitary wastewater, coal yard runoff and ash washwater, process wastewaters, cooling system wastewaters (once-through cooling water and cooling tower blowdown), and storm drainage (EPA 1986a). Approximately 30 and 36%

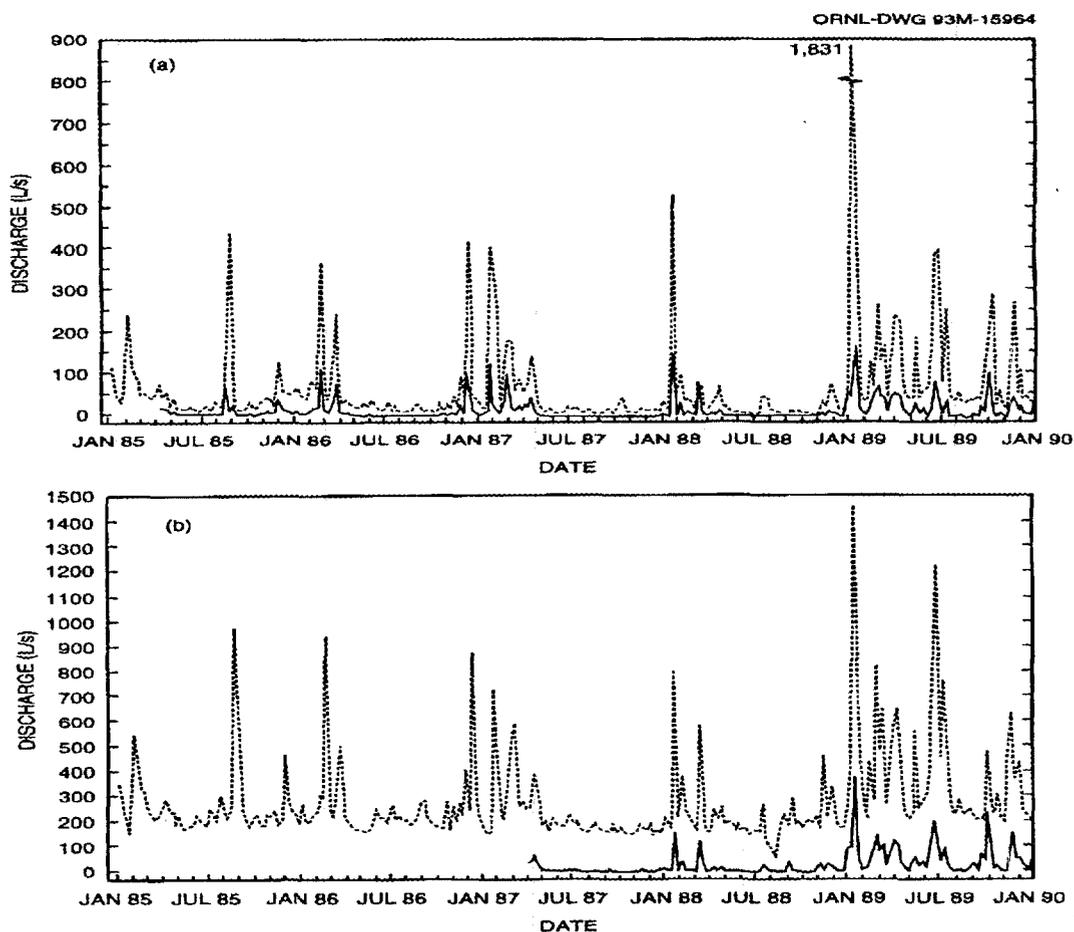


Fig. 2.3. Mean weekly stream flow in Melton Branch (top) and White Oak Creek above and below ORNL. Hydrographs are shown for (a) Melton Branch at U.S. Geological Survey (USGS) gage 03537100 near Melton Branch kilometer (MEK) 1.9 (solid line) and at National Pollutant Discharge Elimination System (NPDES) station X13 near MEK 0.16 (dashed line) and (b) White Oak Creek at USGS gage 03536320 near White Oak Creek kilometer (WCK) 5.47 (solid line) and at NPDES station X14 near WCK 2.65 (dashed line). Discharge at the NPDES sites was measured once each day, excluding weekends and holidays.

Sources: Lowery J. F. et al., 1986, *Water Resources Data for Tennessee, Water Year 1985*, Report No. USGS/WRD/HD-86, U.S. Geological Survey, Nashville, Tennessee; Lowery J. F. et al., 1987, *Water Resources Data for Tennessee, Water Year 1986*, Report No. USGS/WRD/HD-87/225, U.S. Geological Survey, Nashville, Tennessee; Lowery J. F. et al., 1988, *Water Resources Data for Tennessee, Water Year 1987*, Report No. USGS/WRD/HD-88/236, U.S. Geological Survey, Nashville, Tennessee; Lowery J. F. et al., 1989, *Water Resources Data for Tennessee, Water Year 1988*, Report No. USGS/WRD/HD-89/258, U.S. Geological Survey, Nashville, Tennessee; U.S. Geological Survey provisional data from L. D. Voorhees, Oak Ridge National Laboratory; and unpublished data from Oak Ridge National Laboratory's Environmental Monitoring and Compliance Section.

of the estimated total effluent volume are contributed by the cooling and process systems respectively. Discharges from the sewage treatment plant, the steam plant, and leakage constitute the remainder of the discharges in approximately equal proportions (Kasten 1986, Table 5).

There are nine major effluent discharges that enter WOC either directly or indirectly via several tributaries (Table 2.3). Direct discharges to WOC, as listed in Table 2.3 and shown in Fig. 2.4,

Table 2.3. Description of the nine major effluents regulated under the Oak Ridge National Laboratory National Pollutant Discharge Elimination System permit issued on April 1, 1986

Receiving stream	Source of effluent	NPDES outfall number ^a	Average flow rate (L/s) ^b
Melton Branch	REDC ^c process waste basins	X08	2.2 ^e
	HFIR ^d process waste basins	X09	7.0 ^e
Northwest Tributary	1500 Area	X03	0.3
White Oak Creek	Sewage Treatment Plant	X01	10.1 ^f
	Coal Yard Runoff Treatment Facility	X02	1.0 ^f
	2000 Area	X04	0.6
	3539 and 3540 ponds	X06	5.9 ^e
	3544 Process Waste Treatment Plant	X07	7.9 ^f
	3518 Acid Neutralization Facility	X11	1.8 ^e

^aNo outfall X05 exists; outfall X10 to Fifth Creek from the Oak Ridge Reactor resin regeneration facility was eliminated in December 1986 (Blasing et al., 1989, *Environmental Data Report for the Third Quarter of 1988*, ORNL/TM-534, Environmental Monitoring and Compliance Section, Environmental and Health Protection Division, Oak Ridge National Laboratory, Oak Ridge, Tennessee); and outfall X12 is the discharge from the Nonradiological Wastewater Treatment Facility, which began operation in March 1990 (estimated average flow rate of 22 L/s).

^bFor batch operations, average flow rate is based on days when waste is discharged.

^cREDC = Radiochemical Engineering Development Center, which includes the Transuranium Processing Facility and the Transuranium Research Facility.

^dHFIR = High Flux Isotope Reactor.

^eBatch discharge with frequencies of once every 5 d (X08), three times per month (X09), and three batches per d (X11); discharge X06 is a batch if radioactivity is below predetermined levels.

^fMaximum flow rates are 32.9 L/s at X01, 9.6 L/s at X02, and 18.8 L/s at X07.

Source: *Authorization to Discharge Under the National Pollutant Discharge Elimination System, Permit No. TN0002941*, Oak Ridge National Laboratory, Fact Sheet, U.S. Environmental Protection Agency, Region IV, Atlanta, April 1, 1986.

ORNL-DWG 93M-15965

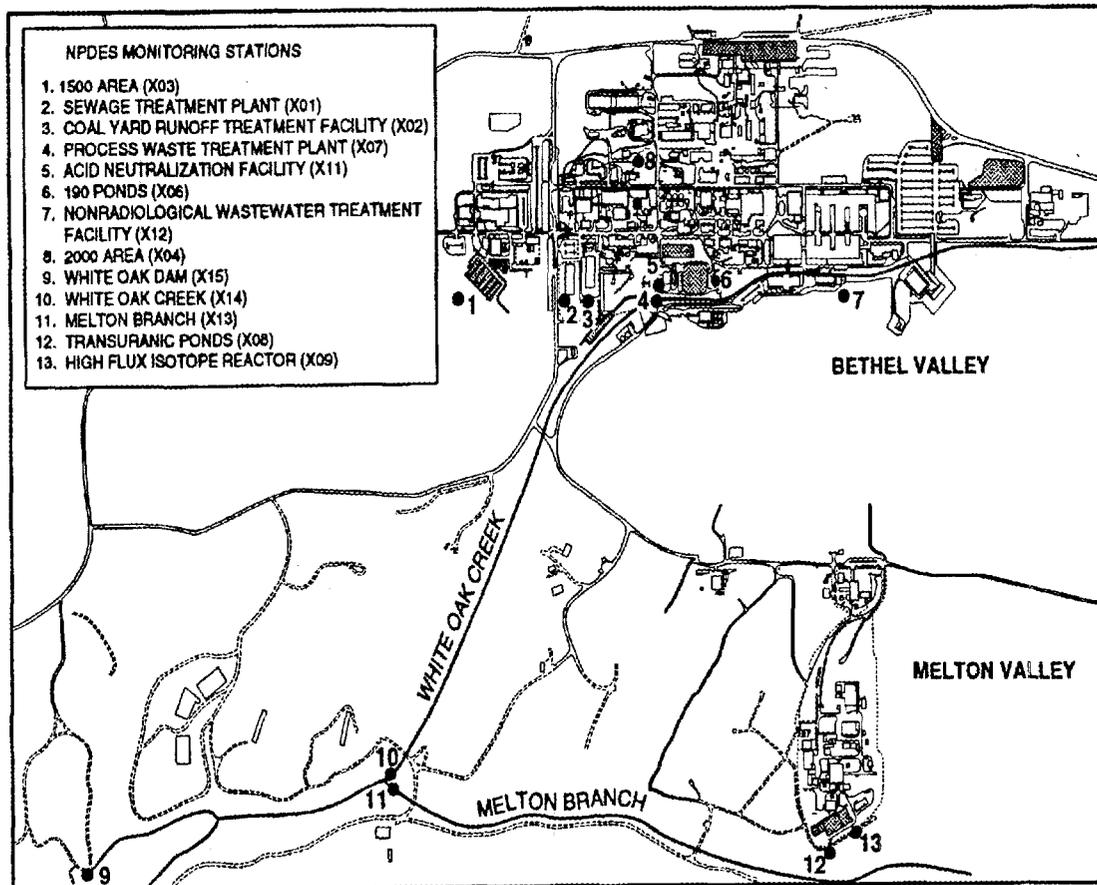


Fig. 2.4. Location of National Pollutant Discharge Elimination System (NPDES) effluent (Sites 1–9 and 13–14) and ambient (Sites 10–12) water quality monitoring stations. Station numbers on legend do not correspond to serial identification of the discharges in the NPDES permit. See *Authorization to Discharge Under the National Pollutant Discharge Elimination System, Permit No. TN 0002941*, Oak Ridge National Laboratory, Fact Sheet, U.S. Environmental Protection Agency, Region IV, Atlanta, April 1, 1986. Source: Blasing et al., *Environmental Data Report for the Third Quarter of 1988*, ORNL/TM-534, Environmental Monitoring and Compliance Section, Environmental and Health Protection Division, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

are largely restricted to a 1.5-km reach of the creek that flows west along the southern perimeter of ORNL (Fig. 2.2). Effluents are discharged to Melton Branch via a small tributary at MEK 1.59, whereas discharges to Northwest Tributary from the 1500 area are located above Northwest Tributary Kilometer (NTK) 0.3 (Fig. 2.2).

In addition to these major waste streams, there are 127 outfalls that also

discharge to streams in the WOC watershed, including Fifth Creek, First Creek, Melton Branch, and WOC (Table 2.4). These include: (1) 34 noncontaminated storm drains (identified as Category 1 outfalls in the NPDES permit); (2) 61 drains that have been contaminated by ORNL operations (Category II outfalls), including drains of roofs, parking lots, and storage/spill areas,

Table 2.4. Number and location (receiving stream) of Category I, II, and III outfalls that discharge to White Oak Creek and tributaries

Type	White Oak Creek	Fifth Creek	First Creek	Melton Branch	Total
Category I					
Storm drains	16	13	4	1	34
Category II					
Parking lot drains	27	6	8	3	44
Storage area drains	0	0	2	0	2
Spill area drains	1	0	0	0	1
Cooling tower blowdown	2	3	0	1	6
Condensate	4	4	0	0	8
Category III					
Process drains	14	8	4	6 ^a	32

^aSettling ponds.

Source: Authorization to Discharge Under the National Pollutant Discharge Elimination System, Permit No. TN0002941, Oak Ridge National Laboratory, Fact Sheet, U.S. Environmental Protection Agency, Region IV, Atlanta, April 1, 1986.

once-through cooling water, cooling tower blowdown and condensate; and (3) 32 outfalls that are contaminated by pollutants because of inflow/infiltration, cross-connections, or improper disposal of chemicals (DEM 1986, EPA 1986a).

Cooling system wastewater, a Category II outfall, is a major component, by volume, of the total effluent discharged by ORNL operations. Waste heat from reactors, particle accelerators, evaporators, environmental control systems, process systems, research laboratories, engineering-scale development facilities, and space-heating condensates is transferred to once-through cooling water or dissipated to the atmosphere via 26 wet-evaporative, mechanical draft cooling towers (Boyle et al. 1982; Kasten 1986). The principal heat burden generated by the operation of ORNL

facilities is discharged by 7 mechanical-draft cooling towers, and an additional 19 smaller towers operate intermittently to meet lesser demands (Boyle et al. 1982). Total blowdown from all cooling towers is ~16.2 L/s (Kasten 1986), of which an average of ~6.9 L/s are discharged to Melton Branch from operation of HFIR, 3.3 L/s were discharged to Fifth Creek when the Oak Ridge Reactor was operational prior to March 1987, and 3.8 L/s are discharged to WOC from the Building 4500 cooling tower (Boyle et al. 1982, Table 2.12). Occasionally, the blowdown may contain radionuclides, thus requiring diversion to the Process Waste Treatment Plant (PWTP) prior to discharge. Normally, however, cooling tower blowdown is discharged directly or indirectly via the storm sewer system to area streams.

2.2.2 Wastewater Modifications for Pollution Abatement

Several pollution abatement measures were implemented recently and more are planned over the next several years as part of the ORNL WPCP. The sewage treatment plant was upgraded in August 1985. A new Coal Yard Runoff Treatment Facility (CYRTF) became operational in early March 1986 and new demineralizer systems were installed at HFIR and the Oak Ridge Reactor in August 1986. One of the most significant components of the WPCP is construction of the NRWTF, which went on-line in March 1990. This facility will treat effluents that are currently (1) untreated and discharged directly to area streams, (2) untreated but discharged to process waste basins prior to release, and (3) treated by existing facilities. The NRWTF will collect and treat NPDES serial discharges X03 through X09; discharge X11 will be treated by either the NRWTF or the CYRTF (Table 2.3). Additional pollution abatement projects and programs are described in Sect. 9.

2.2.3 National Pollutant Discharge Elimination System Water Quality Monitoring Program

Water quality in WOC and Melton Branch is influenced both by point-source discharges from ORNL facilities (Sect. 2.2.1) and by area-source discharges from waste disposal areas, such as the solid waste storage areas (SWSAs), former liquid waste disposal areas (pits and trenches), and inactive process waste basins (Fig. 2.2). The discharge to area streams of leachates from waste disposal areas has been documented by Stueber et al. (1981) for NT (from SWSA 3) and by Cerling and Spalding (1982) for WOC (SWSA 4), Melton Branch (SWSA 5), and WOL

(SWSA 6). A characterization of ambient water quality downstream of all point-source discharges and most major area sources is presented in Table 2.5. It is based on routine NPDES monitoring conducted in 1989 at sites MEK 0.16 and WCK 2.65 (NPDES sites X13 and X14, respectively), although past data collected from sites on upper WOC and Melton Branch above ORNL and at WOD are included for comparative purposes. Routine radiological monitoring is also conducted at the three NPDES ambient stations and other sites on Melton Branch just below HFIR, First Creek, Fifth Creek, NT, Raccoon Creek, and WOC at WCK 3.41 (Fig. 2.2). These data are collected either daily or weekly and are published quarterly by the Environmental Monitoring and Compliance (EMC) Section at ORNL.

Water quality in lower WOC, as characterized by the NPDES monitoring program (Table 2.5), consisted of (1) moderate levels of dissolved solids and turbidity, (2) elevated concentrations of most metals but very low concentrations of organics, (3) moderate phosphorus enrichment, and (4) elevated temperatures (Fig. 2.5). Nitrate enrichment also occurs in WOC below ORNL (Sect. 2.2.4) but was not observed in the NPDES monitoring program due to the high analytical detection limit (5 mg/L).

Although concentrations of some trace elements in WOC below ORNL exceeded the levels observed in the unimpacted upper reaches of the stream, only mercury, aluminum, and iron commonly exceeded U.S. Environmental Protection Agency (EPA) water quality criteria for freshwater aquatic life (EPA 1976, 1980a, 1986b, 1988). Mercury concentrations at WCK 2.65 exceeded the detection limit of 0.05 $\mu\text{g/L}$ on 67% of the sampling dates. The frequency of detection was 42% at WOD and 25% at MEK 0.16. The

Table 2.5. Median concentrations (range in parentheses) of 34 National Pollutant Discharge Elimination System (NPDES) water quality parameters that are routinely monitored below Oak Ridge National Laboratory in Melton Branch at NPDES site X13 (Melton Branch kilometer 0.16), in White Oak Creek at NPDES site X14 (White Oak Creek kilometer 2.65), and at White Oak Dam at NPDES site X15, January–December 1989^a

Parameter	Melton Branch		White Oak Creek		
	Above ORNL ^b	MEK ^c 0.16	Above ORNL ^b	WCK ^d 2.65	WOD ^e
Organics, µg/L					
Chloroform	ND ^f	0.5 (0.2–0.6)	ND	4 (0.6–5)	2 (1– <5)
Polychlorinated biphenyls	<0.5 (all <0.5)	<0.5 (all <0.5)	<0.5 (all <0.5)	<0.5 (all <0.5)	<0.5 (<0.5–0.5)
Phenols	ND	<1 (<1–3)	ND	<1 (<1–4)	ND
Trichloroethylene	ND	0.8 (<5–2)	ND	<5 (0.6–<5)	<5 (0.7–<5)
Metals, µg/L					
Aluminum	39 ^g	540 (<50–7,100)	ND	340 (<50–3,200)	650 (50–2,800)
Arsenic	ND	<50 (<50–54)	ND	<50 (<50–<60)	<50 (<50–90)
Cadmium	0.05 (0.01–0.59)	<2 (all <2)	0.08 (0.01–0.45)	<2 (all <2)	<2 (all <2)
Chromium	0.41 (0.17–1.5)	<10 (<3–27)	0.39 (0.05–1.5)	<10 (<3–35)	17 (<3–28)
Copper	0.82 (0.30–2.1)	<10 (<4–230)	0.56 (0.24–5.9)	<10 (<4–150)	10 (<6–130)
Iron	97 (47–578)	340 (120–10,000)	41 (22–285)	265 (88–4,100)	505 (200–2,300)
Lead	0.70 (0.2–5.7)	<4 (<4–10)	0.72 (0.1–4.0)	<4 (<4–8)	<4 (<4–4)
Manganese	12 ^g	86 (<2–2,100)	ND	41 (17–260)	64 (2–92)
Mercury	0.003 (<0.001–1.19)	<0.05 (<0.05–0.1)	0.002 (<0.001–0.36)	0.07 (<0.05–0.12)	<0.05 (<0.05–0.11)
Nickel	3.1 (0.1–16)	6.7 (<6–10)	1.3 (0.1–15)	<6 (<5–8.3)	<6 (<5–19)
Silver	ND	<5 (all <5)	ND	<5 (all <5)	<5 (all <5)
Zinc	0.52 (0.19–56)	12 (<8–160)	1.1 (0.25–36)	41 (<8–130)	23 (<8–39)
Conventional parameters, mg/L					
Ammonia nitrogen	<0.2 (all <0.2)	0.03 (0.01–0.07)	<0.2 (all <0.2)	0.03 (0.02–0.10)	0.03 (0.01–0.16)
Biological oxygen demand 5-d	<5 (all <5)	<5 (all <5)	<5 (all <5)	<5 (<5–>34)	<5 (<5–>34)
Chlorine, total residual ^h	0 ⁱ	<0.01 (all <0.01)	0 ⁱ	<0.01 (all <0.01)	<0.01 (all <0.01)
Conductivity, µS/cm ^j	231 (118–262) ^k	500 (100–1,400)	ND	650 (200–1,800)	650 (230–1,700)
Dissolved oxygen ^l	ND	9.3 (5.1–14.9)	ND ^f	8.6 (6.0–18.8)	8.5 (4.0–14.4)

Table 2.5 (continued)

Parameter	Melton Branch		White Oak Creek		
	Above ORNL ^b	MEK ^c 0.16	Above ORNL ^b	WCK ^d 2.65	WOD ^e
Dissolved solids ^f	158 (118-266)	201 (120-283)	95 (68-239)	207 (130-304)	199 (141-242)
Fluoride	0.1 (0.0-0.2) ^k	<1 (<1-1)	ND	1 (<1-1)	1 (<1-1)
Nitrate	0.04 (<0.005-1.10)	<5 (all <5)	0.31 (0.08-1.90)	<5 (all <5)	<5 (all <5)
Oil and grease ^h	<2 (all <2)	2 (<2-88)	<2 (all <2)	<2 (<2-42)	2 (<2->200)
Organic carbon, total	ND	2.6 (1.6-5.1)	ND	2.4 (1.3-6.9)	2.8 (1.7-6.4)
pH ^f	7.9 (7.5-8.1) ^k	7.8 (6.5-8.0)	ND	7.9 (6.5-8.5)	7.9 (6.7-8.9)
Phosphorus	0.012 (0.003-0.152)	0.1 (0.1-0.6)	0.012 (0.002-0.124)	0.3 (0.1-0.4)	0.2 (0.1-0.5)
Sulfate	12.1 (6.8-16.4)	22 (12-27)	2.93 (0.26-4.76)	38 (18-67)	35 (12-48)
Suspended solids	8 (<5-159)	5 (<5-393)	<5 (<5-30)	5 (<5-128)	10 (<5-37)
Temperature, °C ^g	ND	14.7 (1.6-26.9)	ⁱ	16.0 (7.1-25.7)	17.0 (3.9-28.4)
Turbidity, NTU ^f	6.4 (2.5-32)	80 (10-220)	3.7 (1.4-30)	76 (16-271)	36 (10-240)

^aAnalyses were based on 24-h composite samples collected monthly unless noted otherwise.

^bValues represent the median concentration (range in parentheses) of grab samples collected weekly between April 1979 and January 1980 in upper Melton Branch near Melton Branch kilometer (MEK) 1.8 and in upper White Oak Creek near White Oak Creek kilometer (WCK) 6.3 (Boyle, J. W. et al., 1982, *Environmental Analysis of the Operation of the Oak Ridge National Laboratory X-10 Site*, Oak Ridge National Laboratory, Oak Ridge, Tennessee, Sect. 3.2.3.2 and Tables 4.15-16; and unpublished data from Oak Ridge National Laboratory's Environmental Monitoring and Compliance Section.

^cMEK = Melton Branch kilometer.

^dWCK = White Oak Creek kilometer.

^eWOD = White Oak Dam

^fND = no data available.

^gN = 1. Values for aluminum and manganese were obtained from McMaster, W. M., 1967, *Hydrologic Data for the Oak Ridge Area, Tennessee*, U.S. Geological Survey - Water Supply Paper No. 1839-N, U.S. Government Printing Office, Washington, D.C., Table 12.

^hGrab samples collected weekly.

ⁱFrom Loar, J. M. et al., 1992, *First Annual Report on the ORNL Biological Monitoring and Abatement Program*, ORNL/TM-10399, Oak Ridge National Laboratory, Oak Ridge, Tennessee, Fig. 3.6.

^jGrab samples collected monthly.

^kValues represent the median concentration (range in parentheses) of six samples collected in 1961-64 from upper White Oak Creek near WCK 6.8 (McMaster, W. M., 1967, Table 11).

^lSee Table 2.6 of this document.

Source: ORNL Environmental Monitoring and Compliance Section, unpublished data.

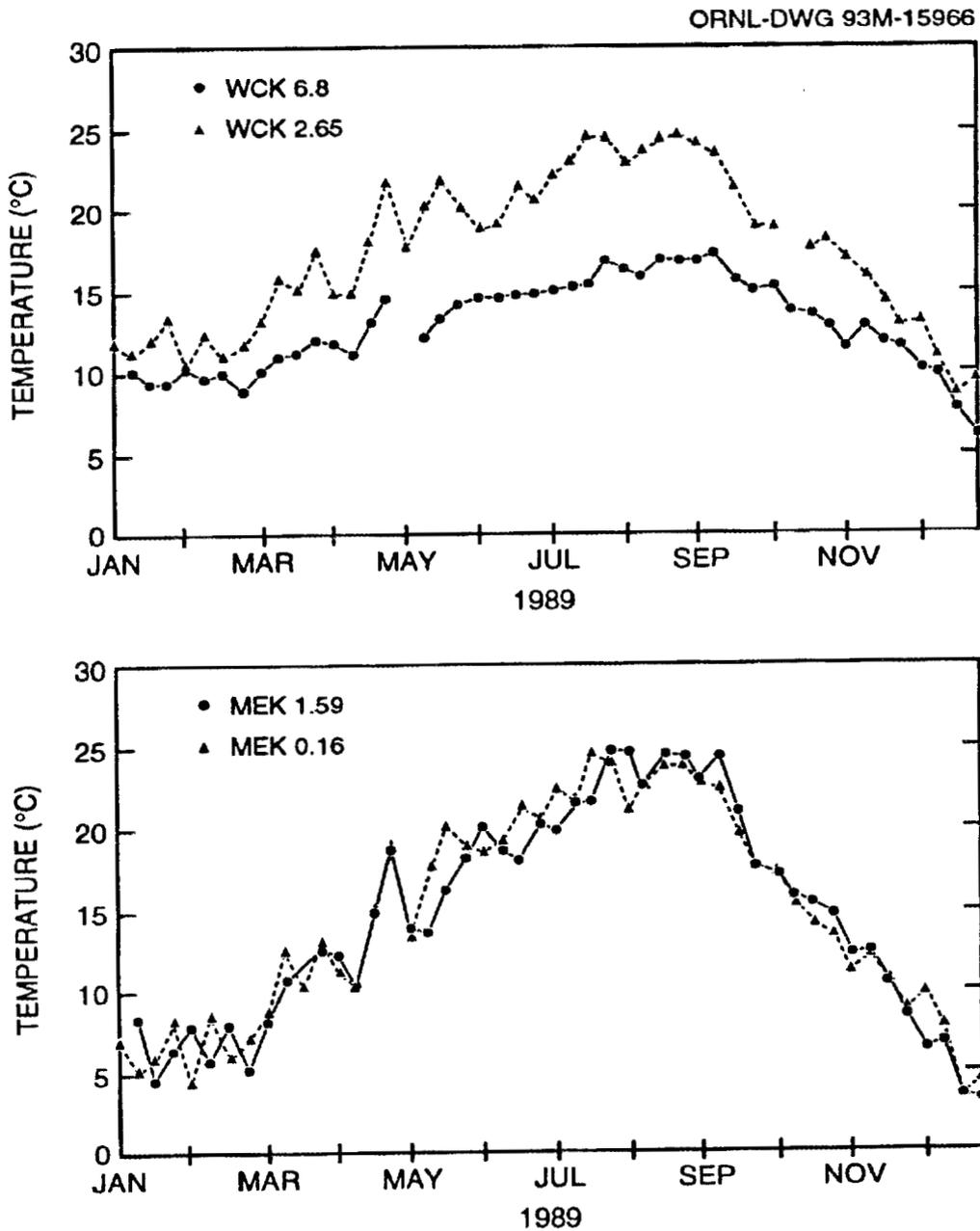


Fig. 2.5. Mean weekly temperatures in White Oak Creek at (a) sites above and below Oak Ridge National Laboratory [(ORNL) White Oak Creek kilometer (WCK) 6.8 and WCK 2.65 respectively] and (b) in Melton Branch above and below the High Flux Isotope Reactor (HFIR) complex [Melton Branch kilometer (MEK) 1.59 and MEK 0.16 respectively], January–December 1989. Temperature data at WCK 6.8 and MEK 1.59 were obtained hourly with a Ryan Tempentor digital thermograph. At WCK 2.65 and MEK 0.16 (National Pollutant Discharge Elimination System station X14 and X13 respectively), weekly values were based on average hourly temperatures computed from data collected at 10-min intervals with a real-time monitoring system. Source: Unpublished data from the ORNL Environmental Monitoring and Compliance Section.

detection limit is well above EPA water quality criterion for the average concentration ($0.012 \mu\text{g/L}$, EPA 1986b). Mercury concentrations in fish in WOC, Melton Branch, and WOL verify the presence of excess mercury in this system (Sect. 4.1).

Aluminum concentrations nominally exceeded EPA water quality criterion in 1989 on virtually all occasions at all three sites. However, the water quality criterion for aluminum specifies a measurement of **acid-soluble** aluminum, a mild sample treatment that does not digest the aluminosilicate matrix of clay soils. The samples reported in Table 2.5 were digested in concentrated acid to analyze for total recoverable metals. This treatment would dissolve the naturally occurring aluminum and iron constituents of soil minerals, and thus yield high concentrations that vary greatly with the suspended solids concentration of the water samples. For example, typical concentrations of iron and aluminum measured in soil from the ORR following acid digestion are 17 and $16 \mu\text{g/mg}$ dry soil respectively. (These values are based on data presented in Turner et al. 1988, Table 7.7.) When these values are used to calculate the total recoverable iron and aluminum concentrations in a water sample containing 100 mg/L total suspended solids (TSS), the resulting concentrations ($1700 \mu\text{g/L}$ iron and $1600 \mu\text{g/L}$ aluminum) are typical of concentrations reported in WOC and Melton Branch when TSS levels were near 100 mg/L .

Likewise, other trace elements, such as chromium and copper, that infrequently exceeded EPA water quality criteria probably had no significant ecological impact. Total recoverable chromium concentrations occasionally exceeded the average and maximum criteria for Cr^{+6} of 11 and $16 \mu\text{g/L}$, respectively, at all three sites (Table 2.5). However, excess chromium in those systems is unlikely to be

Cr^{+6} , which is readily reduced to Cr^{+3} in organic-rich waters. The water quality criteria for trivalent chromium are more than two orders of magnitude higher than those for hexavalent chromium (EPA 1986b). Very high copper concentrations were reported at all three sites on February 16, 1989 (see maximum values in Table 2.5). A single discharge of high copper waste could have produced elevated copper concentrations at WOD and either of the other sites on the same day. However, the occurrence of radically high concentrations at all three sites strongly suggests that these results are spurious. Finally, the detection limits for cadmium, mercury (Melton Branch and WOD only), and polychlorinated biphenyls (PCBs) exceeded the criteria either for average concentration or for protection of human health on most sampling dates. Based on the results of ambient toxicity testing (Sect. 3.1.3), however, it is unlikely that concentrations of these elements exceeded levels that would be toxic to biota.

In general, the water quality of WOC in 1989 was similar to that in the previous 3 years (Loar et al. 1992; Loar 1993a, 1993b). Year-to-year variations appear to be primarily a result of varying frequencies of heavy rainfall events that produce high concentrations of naturally occurring metal constituents of soils.

Temperatures in WOC below ORNL (WCK 2.65) are as much as $7\text{--}10^\circ\text{C}$ higher than the temperatures in upper WOC north of Bethel Valley Road (Fig. 2.5). Although some increase in downstream temperatures would be expected due to the absence of a riparian canopy in the 1.5-km reach of stream adjacent to the plant complex, plant operations probably account for much of the longitudinal gradient in water temperature (average of 2°C/km). Cooling tower blowdown and condensate are discharged to WOC and several tributaries (Table 2.4), and most of the

streams show higher temperatures downstream of these outfalls (Fig. 2.6).

In Melton Branch, the significant improvement in water quality that occurred between 1986 and 1988 (Loar 1993a, 1993b) was sustained in 1989. The concentrations of sulfates, phosphorus, and dissolved solids in 1987–89 were substantially lower than the concentrations observed at MEK 0.16 in 1986. Stream temperatures were also much lower in the past three years. Although maximum temperatures at MEK 0.16 approached 38°C in 1986 (Loar et al. 1992, Fig. 2.7), they have not exceeded 30°C since then (Table 2.6 in this document and in Loar 1993a). Temperatures in lower Melton Branch have been consistently lower than temperatures in WOC during the past 3 years (Fig. 2.7). The water quality of lower Melton Branch in 1989 was similar to that of the previous year, but there were some exceptions. The single high ammonia concentration observed in 1988 was not repeated in 1989, and the maximum concentrations of TSS and soil-associated metals were substantially higher in 1989, a much wetter year.

The improvement in water quality associated with shutdown of HFIR in November 1986 and a subsequent reduction in cooling tower blowdown and discharges from the HFIR process ponds (Fig. 2.2) continued through 1989. HFIR is the largest of the five ORNL reactors. The volume of blowdown from its cooling towers is twice that of any other reactor at ORNL. Moreover, HFIR's chemical usage exceeds that of all the cooling towers combined—the HFIR towers account for 77% of the sulfuric acid, 55% of biocides, and 63% of the sodium phosphate used for the ORNL cooling towers annually (Boyle et al. 1982, Table 2.2). Although the other four ORNL reactors were eventually shut down in March 1987, no measurable effects on WOC water quality were observed

because the average flow in WOC is four times greater than the flow in Melton Branch (Table 2.2) and the volume of the blowdown from all the cooling towers at the main ORNL complex is less than that of HFIR. Moreover, HFIR is the primary facility that discharges to Melton Branch, which was dry for much of 1986 above MEK 1.59 where HFIR discharges enter the stream. There are numerous facilities and substantially more discharges to WOC; thus, dilution of the reactor discharges is considerably greater in WOC than in Melton Branch. Although restart of HFIR in early 1990 can be expected to adversely affect water quality in Melton Branch, the impact will be moderated somewhat by the elimination of discharges from the HFIR process ponds in March 1990.

2.2.4 Biological Monitoring and Abatement Program Water Quality Sampling Program

Water quality information is important for characterizing and understanding biotic conditions in aquatic systems. The BMAP water quality sampling program was initiated in mid-1986 as a component of the periphyton monitoring program. This program was intended to augment the NPDES water quality monitoring program by providing data for additional sites and parameters, especially those parameters that influence periphyton communities but are not included in the NPDES monitoring program. This program consisted of the monthly collection of water samples (grab samples) at the periphyton monitoring sites in the WOC system. The monthly "snapshots" of water quality provided by these discrete samples have been useful for the interpretation of the biomass and functioning of the periphyton communities at the various study sites (Sect. 3.8).

ORNL-DWG 93M-15967

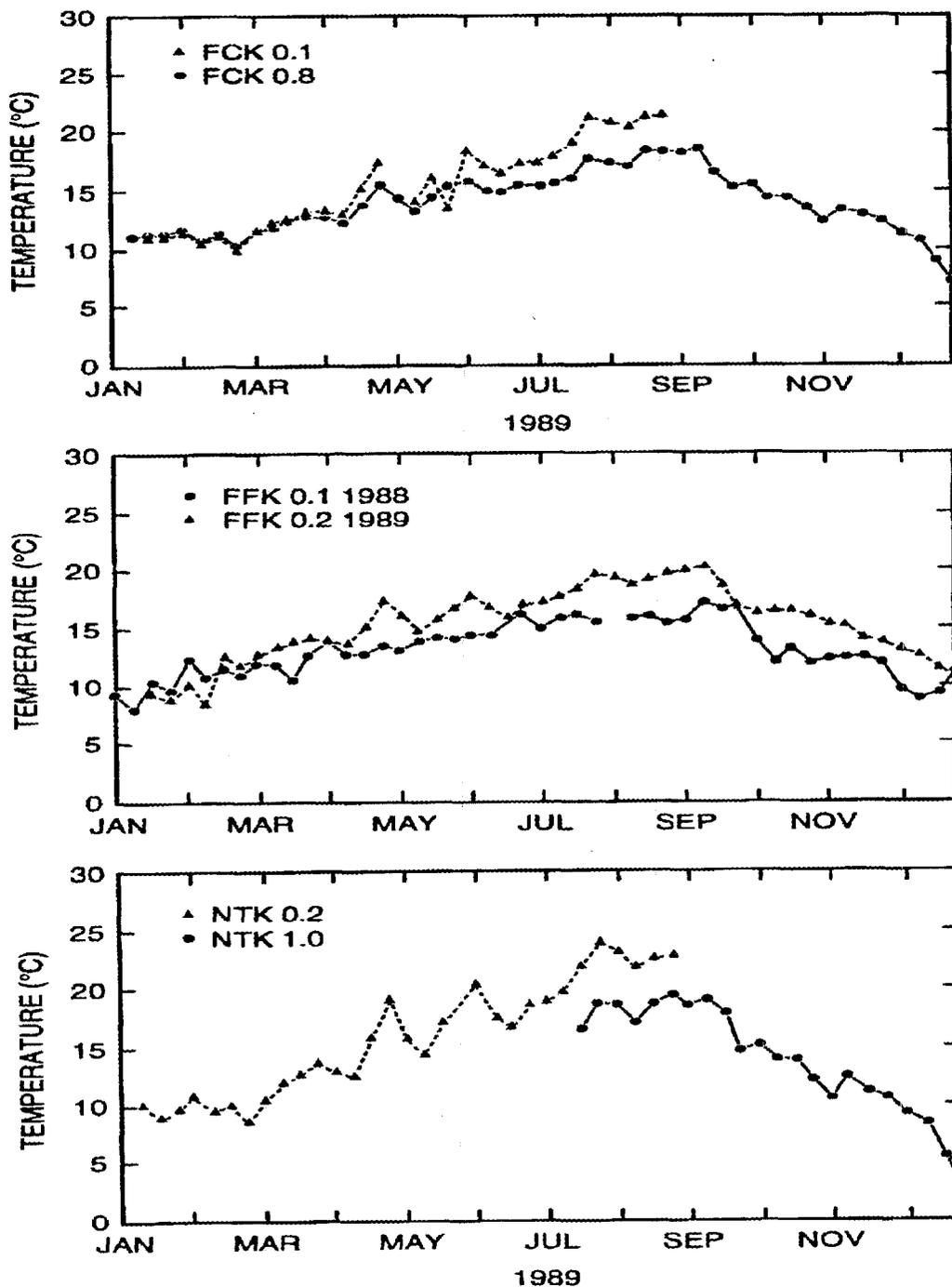


Fig. 2.6. Mean weekly temperatures in three tributaries of White Oak Creek, January–December 1989. Mean values were calculated from data obtained at hourly intervals with a Ryan Tempmentor digital thermograph. Data were not available for FFK 1.0 in 1989; 1988 data are depicted for comparison. [FCK = First Creek kilometer, FFK = Fifth Creek kilometer, and NTK = Northwest Tributary kilometer (see Fig. 2.2).]

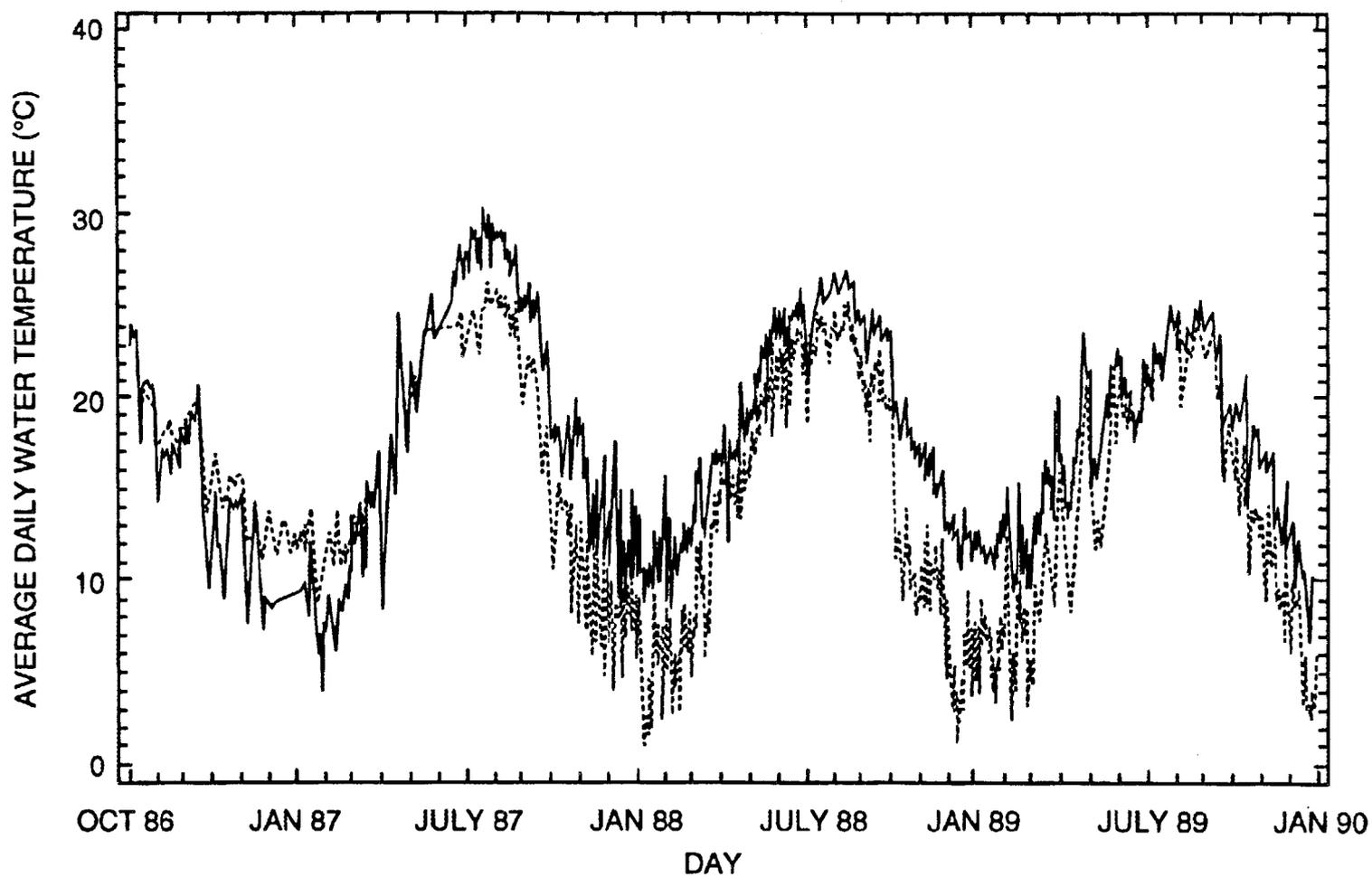


Fig. 2.7. Water temperatures at National Pollutant Discharge Elimination System monitoring stations X13 on lower Melton Branch at Melton Branch kilometer (MEK) 0.16 (dashed line) and X14 on lower White Oak Creek at White Oak Creek kilometer (WCK) 2.65 (solid line). Values represent a daily average computed from measurements taken at 10-min intervals with a real-time monitoring system. *Source:* Unpublished data from Oak Ridge National Laboratory Environmental Monitoring and Compliance Section.

Table 2.6. Mean (± 1 standard deviation) monthly water temperature in White Oak Creek and tributaries, including First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, January–December 1989^a

	Mean water temperature ± 1 standard deviation (°C)								
	FCK ^b 0.1 ^c	FCK 0.8 ^d	FFK ^e 0.2 ^d	MEK ^f 0.16 ^e	MEK 1.59 ^h	NTK ⁱ 0.2 ^j	NTK 1.0 ^k	WCK ^l 2.65 ^m	WCK 6.8 ^d
Jan	11.0 \pm 1.1 (8.5–13.6)	11.3 \pm 0.7 (9.8–13.2)	9.6 \pm 5.1 (-0.5–27.6)	6.5 \pm 1.6 (3.3–9.2)	6.1 \pm 3.5 (-1.4–17.6)	9.5 \pm 1.2 (6.7–12.3)	ND ⁿ	11.9 \pm 0.7 (10.6–13.5)	9.6 \pm 1.7 (2.9–16.6)
Feb	10.8 \pm 1.5 (7.3–16.0)	11.0 \pm 1.1 (8.1–14.8)	10.6 \pm 4.5 (-0.8–27.7)	6.6 \pm 2.5 (2.6–11.7)	6.7 \pm 3.4 (-2.1–17.7)	9.7 \pm 1.7 (6.7–15.0)	ND	11.6 \pm 1.8 (9.4–15.4)	9.7 \pm 1.3 (7.1–14.3)
Mar	12.5 \pm 1.4 (9.5–16.4)	12.2 \pm 1.2 (9.9–15.9)	13.5 \pm 1.2 (11.1–17.7)	10.3 \pm 2.5 (5.8–15.8)	10.9 \pm 3.1 (4.3–19.3)	12.3 \pm 2.1 (8.1–18.5)	ND	14.9 \pm 2.4 (11.0 - 20.0)	11.2 \pm 1.5 (8.2 - 16.0)
Apr	14.8 \pm 2.5 (10.7–22.0)	13.6 \pm 1.9 (10.7–19.2)	15.1 \pm 3.2 (10.4–30.3)	13.8 \pm 3.7 (8.4–20.4)	14.1 \pm 4.0 (6.0–25.0)	15.2 \pm 3.1 (10.3–22.2)	ND	17.0 \pm 2.8 (13.3–23.8)	12.7 \pm 2.0 (9.5–18.8)
May	15.8 \pm 2.1 (11.6–21.7)	14.3 \pm 1.5 (11.7–19.9)	15.7 \pm 1.9 (9.5–33.5)	15.6 \pm 2.6 (11.6–20.4)	15.9 \pm 2.8 (8.9–22.9)	16.5 \pm 2.3 (11.8–22.9)	ND	19.1 \pm 2.2 (15.4–22.0)	13.3 \pm 1.2 (10.6–17.4)
Jun	17.5 \pm 1.6 (14.5–22.6)	15.3 \pm 0.9 (13.6–19.3)	16.9 \pm 1.2 (14.8–22.7)	19.6 \pm 1.4 (17.6–22.4)	19.3 \pm 1.6 (16.0–26.3)	18.2 \pm 1.6 (15.6–21.8)	ND	20.3 \pm 1.6 (17.6–22.8)	14.9 \pm 0.6 (13.4–18.0)
Jul	19.2 \pm 2.0 (16.0–24.6)	16.4 \pm 1.3 (14.5–20.5)	18.4 \pm 1.4 (15.7–23.2)	22.3 \pm 1.6 (20.0–25.2)	22.2 \pm 2.2 (18.1–29.6)	21.4 \pm 2.3 (17.0–26.3)	17.7 \pm 1.4 (15.0–20.1)	22.7 \pm 1.6 (19.7–25.1)	15.9 \pm 1.0 (14.2–19.0)
Aug	21.1 \pm 1.4 (17.4–25.8)	17.8 \pm 1.2 (14.7–22.3)	19.3 \pm 1.1 (16.9–24.6)	23.0 \pm 1.4 (19.6–24.8)	23.9 \pm 1.9 (19.0–29.6)	22.7 \pm 1.2 (19.8–26.2)	18.4 \pm 1.2 (14.9–20.9)	24.1 \pm 0.8 (22.3–25.6)	16.5 \pm 1.2 (13.3–20.2)
Sept	ND	17.2 \pm 1.8 (13.5–23.4)	19.1 \pm 1.7 (15.2–23.6)	20.8 \pm 2.6 (15.5–24.0)	21.5 \pm 3.4 (14.1–32.0)	ND	17.6 \pm 2.9 (6.0–25.6)	22.2 \pm 2.4 (17.6–24.8)	16.3 \pm 1.2 (14.0–20.7)
Oct	ND	14.5 \pm 1.4 (11.3–17.7)	16.4 \pm 1.2 (14.1–19.9)	15.1 \pm 2.5 (10.2–19.2)	15.7 \pm 2.4 (9.2–20.7)	ND	13.8 \pm 1.7 (9.9–17.4)	18.4–1.3 (15.6–21.5)	13.8 \pm 1.6 (10.3–17.1)

Table 2.6 (continued)

	Mean water temperature \pm 1 standard deviation (°C)								
	FCK ^b 0.1 ^c	FCK 0.8 ^d	FFK ^e 0.2 ^d	MEK ^f 0.16 ^g	MEK 1.59 ^h	NTK ⁱ 0.2 ^j	NTK 1.0 ^k	WCK ^l 2.65 ^m	WCK 6.8 ^d
Nov	ND	12.9 \pm 1.0 (9.5–15.3)	14.6 \pm 1.1 (10.4–18.2)	10.5 \pm 2.0 (6.7–14.3)	11.2 \pm 2.3 (5.6–16.8)	ND	11.4 \pm 1.4 (6.9–14.5)	14.8–1.9 (11.7–17.3)	12.1 \pm 1.1 (8.4–14.7)
Dec	ND	9.8 \pm 1.9 (4.6–13.5)	12.1 \pm 1.3 (8.6–15.8)	6.6 \pm 2.9 (2.5–11.6)	5.1 \pm 2.3 (1.0–10.9)	ND	6.7 \pm 2.8 (0.8–11.2)	10.2–1.7 (6.6–13.4)	8.8 \pm 2.0 (3.9–12.7)

^aAbsolute maximum and minimum temperatures are given in parentheses. Data were obtained hourly using a Ryan Tempmentor digital thermograph except sites MEK 0.16 and WCK 2.65 where tabular values are based on average hourly temperatures computed from data collected at 10-min intervals with a real-time monitoring system (ORNL Environmental Monitoring and Compliance Section, unpublished data).

^bFCK = First Creek kilometer.

^cDays of record for incomplete months were January (19 d), and May and August (26 d).

^dDays of record for incomplete months were January (19 d), May and August (26 d), and December (28 d).

^eFFK = Fifth Creek kilometer.

^fMEK = Melton Branch kilometer.

^gDays of record for incomplete months were January (29 d), March (25 d), September (27 d), October (30 d), and December (28 d).

^hDays of record for incomplete months were January (20 d); May, August, and November (27 d); and December (28 d). Site is located just below the confluence of the High Flux Isotope Reactor tributary with Melton Branch. Temperatures were not measured at an upstream reference site due to zero discharge (see Table 2.1).

ⁱNTK = Northwest Tributary kilometer.

^jDays of record for incomplete months were January (20 d), May (30 d), June (28 d), and August (27 d).

^kDays of record for incomplete months were July (17 d), August (27 d), and December (28 d).

^lWCK = White Oak Creek kilometer.

^mDays of record for incomplete months were January (26 d), April (28 d), May (20 d), September (27 d), October (22 d), and December (25 d).

ⁿND = no data available; thermograph not deployed or lost before recovery. Minimum values below 0°C may have resulted from exposure of the thermograph to air due to ice formation or flooding.

2.2.4.1 Methods

Two 1-L grab samples of stream water were collected from each of the nine periphyton monitoring sites (Sect. 3.2) during monthly collection. The water samples were collected in acid-washed polyethylene bottles and taken to the laboratory within 1 h of collection. Samples for the determination of dissolved organic carbon (DOC) were collected in organic-free glass bottles with Teflon-sealed lids. The total amount of DOC was determined for samples collected each month. Quarterly, a subsample of this water was partitioned into hydrophilic and hydrophobic DOC fractions, which were again partitioned by molecular weight. A filtered water sample was analyzed quarterly for soluble metals by inductively coupled plasma (ICP) spectroscopy. One sample from each site was analyzed for several basic water quality parameters (pH, alkalinity, conductivity, hardness, and soluble metals). Analyses for pH, alkalinity, hardness and conductivity were conducted within 2 h of collection. Samples for other analyses were preserved and/or frozen according to approved methods (EPA 1983) until they could be analyzed. The methods for the parameters included in BMAP water quality sampling program are listed in Table 2.7.

2.2.4.2 Results

From October 1988 through September 1989 (referred to as 1989 herein), samples were collected monthly from four reference sites located upstream of ORNL operations [WCK 6.8, MEK 1.6, First Creek kilometer (FCK) 1.0, and Fifth Creek kilometer (FFK) 1.1] and five sites located downstream of ORNL discharge (WCK 3.9, WCK 3.4, WCK 2.9, WCK 2.3, and MEK 0.6). The reference site on Melton Branch was moved downstream

from MEK 1.8 to MEK 1.6 for this study year to provide a site where flow was continuous for a greater portion of the year. At MEK 1.6, water was flowing on every sampling date except October and November 1988, and August 1989. Twelve months of data are available for all other sites, except FCK 1.0 and FFK 1.1, which were not sampled in December 1988. The general chemical characteristics of water from these stream sites have been discussed previously (Loar 1993a, 1993b).

Data for general water quality parameters in 1989 (Table 2.8) were similar to those from 1988 for all sites. At WCK 6.8, alkalinity, dissolved inorganic carbon, conductivity, ammonia, and calcium were lower during 1989 than 1988. Increased stream discharge was probably responsible for this apparent dilution. Other reference sites also had slightly lower concentrations of some of these parameters during 1989 compared with 1988 (Table 2.8 and Loar 1993b). Total phosphorus and soluble reactive phosphorus (SRP) were slightly lower during 1989 than 1988 at WCK 3.4, WCK 2.9, and WCK 2.3, possibly due to dilution or changes in ORNL releases.

Overall, the differences in water chemical parameters between 1988 and 1989 were likely of little biological significance, with the possible exception of lower concentrations of some nutrients (e.g., SRP, nitrate, and ammonia) at reference sites WCK 6.8, FCK 1.0, and FFK 1.1. With the exception of slightly higher pH, the water quality at MEK 1.6 in 1989 was similar to that at MEK 1.8 during 1988 (Loar 1993b), and similar to the water quality of other reference sites. Although activities in the HFIR Radiochemical Engineering Development Center (REDC) Area have been reduced since November 1986, high concentrations of SRP persist at MEK 0.6. High concentrations of SRP and nitrate at

Table 2.7. Water quality parameters determined for discrete samples collected monthly at nine sites in streams in the White Oak Creek watershed, October 1988–September 1989

Parameter	Method	Reference ^a
pH	Glass electrode	APHA (1985)
Alkalinity	Acid titration to ~pH 4.7	APHA (1985)
Conductivity	Conductivity bridge	APHA (1985)
Hardness	EDTA ^b titration	APHA (1985)
Phosphorus	Ascorbic acid method	APHA (1985)
Total P	Persulfate digestion	APHA (1985)
Total soluble P	Filter and digestion	
Soluble reactive P	Filter only	
Nitrate and nitrite	Cadmium reduction	EPA (1983)
Ammonia (ammonium)	Phenate method	EPA (1983)
Suspended solids	Total filterable (105° C)	APHA (1985)
Dissolved metals	Filter and induction-coupled plasma emission spectroscopy	APHA (1985)
Dissolved organic carbon	Persulfate oxidation and infrared detection	Dohrmann (1984)

^aReferences cited here are listed in full in Sect. 10.

^bEDTA = ethylene diamine tetra-acetic (acid).

MEK 0.6 probably stimulate biotic activity at that site (see Sect. 3.2).

Quarterly sampling for dissolved constituents. Quarterly analyses of stream water for dissolved elements generally found only Al, Ba, Ca, Cu, Cr, Mg, Mn, Na, Si, Sr, and Zn at concentrations detectable by ICP analysis (Table 2.9), although copper and iron were occasionally detectable at some sites. Potassium was

not determined. The minimum concentration detectable by ICP for many parameters (e.g., cadmium, copper, and nickel) is relatively high in relation to biologically significant concentrations. Therefore, these data should be used to screen for excessive concentrations of potentially toxic metals. For most parameters, concentrations found during 1989 were less than or equal to those observed in 1988. Lower concentrations of

Table 2.8. Water quality parameters determined from samples collected monthly at the nine periphyton monitoring sites in the White Oak Creek watershed, October 1988–September 1989

	Site								
	WCK ^a 6.8	WCK 3.9	WCK 3.4	WCK 2.9	WCK 2.3	MEK ^b 1.6	MEK 0.6	FCK ^c 1.0	FFK ^d 1.1
pH									
Mean	7.84	8.10	8.05	8.08	8.09	8.07	8.10	7.82	7.43
SD ^e	0.25	0.14	0.15	0.16	0.15	0.15	0.15	0.20	0.27
Range	7.47–8.14	7.93–8.35	7.88–8.39	7.97–8.43	7.97–8.42	7.92–8.35	7.84–8.37	7.46–8.07	7.08–8.05
Alkalinity, meq/L									
Mean	1.84	2.31	2.27	2.27	2.33	2.49	2.46	2.12	2.06
SD	0.71	0.20	0.14	0.14	0.13	0.38	0.36	0.61	0.51
Range	0.86–3.18	1.84–2.62	1.92–2.54	1.96–2.58	2.00–2.48	1.88–3.26	1.78–2.96	1.41–3.38	1.34–2.96
Hardness, mg/L as CaCO₃									
Mean	100	143	157	161	152	139	150	113	115
SD	34	18	34	34	21	19	24	30	27
Range	50–162	110–180	111–224	110–230	112–180	106–168	114–188	70–164	70–154
Conductivity, μS/cm									
Mean	178	288	358	377	346	273	303	203	212
SD	59	36	78	88	39	34	42	53	41
Range	95–278	216–357	247–530	247–541	244–414	215–329	228–373	125–302	138–283
Total phosphorus, μg P/L									
Mean	9	213	297	278	248	12	132	9	11
SD	3	103	117	108	100	4	129	2	2
Range	5–13	77–399	77–432	118–449	95–401	8–20	28–448	7–14	5–25
Soluble reactive phosphorus, μg P/L									
Mean	6	65	163	159	155	7	89	6	8
SD	3	41	78	72	83	5	104	3	3
Range	3–11	22–146	53–290	50–267	42–293	1–19	14–361	2–12	3–13
Total soluble phosphorus, μg P/L									
Mean	8	196	264	251	229	11	126	8	9
SD	3	95	109	105	101	5	129	2	3
Range	3–13	71–362	110–402	113–396	85–388	5–20	14–436	3–14	3–25
Ammonia nitrogen, μg NH₃/L									
Mean	3	22	45	26	13	5	6	4	2
SD	4	17	46	24	8	6	7	5	3
Range	0–13	0–35	6–136	3–72	2–30	0–16	0–19	2–17	0–9
Total nitrogen, μg N/L									
Mean	98	506	1003	945	803	84	218	72	145
SD	71	381	326	319	366	56	116	51	62
Range	35–272	174–1678	543–1651	540–1566	455–1629	40–185	107–506	0–167	77–221
NO₂ + NO₃, μg N/L									
Mean	69	328	886	848	728	31	158	42	121
SD	29	173	325	319	352	10	129	17	22
Range	0–115	118–801	462–1520	460–1520	369–1560	17–39	50–386	10–66	62–168

Table 2.8 (continued)

	Site								
	WCK ^a 6.8	WCK 3.9	WCK 3.4	WCK 2.9	WCK 2.3	MEK ^b 1.6	MEK 0.6	FCK ^c 1.0	FFK ^d 1.1
Dissolved organic carbon, mg C/L									
Mean	1.6	2.8	2.5	2.5	2.4	1.9	2.7	1.6	1.1
SD	0.4	1.2	0.6	0.5	0.4	0.4	0.6	0.7	0.2
Range	1.2-2.6	1.8-5.8	1.9-3.7	2.0-3.6	1.8-3.4	1.1-2.4	2.0-4.2	0.9-3.1	0.8-1.5
Dissolved inorganic carbon, mg C/L									
Mean	21.7	28.7	28.3	28.5	29.0	29.0	30.6	27.3	27.3
SD	7.0	2.7	2.4	2.7	2.1	5.8	4.2	7.1	4.9
Range	11.0-32.6	22.7-34.2	23.3-30.3	24.3-30.8	24.3-31.1	21.6-39.7	23.6-37.4	15.4-40.1	17.6-35.5
Total suspended solids, mg dry weight/L									
Mean	3.7	3.9	4.5	4.54	5.3	8.5	6.2	2.9	1.6
SD	1.6	1.5	2.61	1.3	3.5	3.7	3.9	0.7	1.3
Range	0.8-6.7	1.3-7.1	2.7-12.4	2.2-6.4	1.9-15.3	2.0-13.0	1.2-13.6	2.1-4.5	0.7-5.3

^aWCK = White Oak Creek kilometer.

^bMEK = Melton Branch kilometer.

^cFCK = First Creek kilometer.

^dFFK = Fifth Creek kilometer.

^eSD = standard deviation.

calcium, magnesium, and sodium at several upstream and downstream sites suggest dilution due to higher discharge. Data for MEK 1.6 during 1989 were similar to those reported for MEK 1.8 during 1988 (Loar 1993b).

2.2.4.3 Summary

Water samples collected monthly in conjunction with the periphyton monitoring

program provide useful snapshots of water quality at nine locations in the WOC drainage and augment the NPDES monitoring program. Data for the parameters reported here were similar to data collected during previous years. Although operations at the HFIR/REDC facility have been slowed down since November 1986, concentrations of SRP remain high at MEK 0.6 and probably stimulate biotic activity at that site.

Table 2.9. Concentrations of dissolved elements^a at nine periphyton study sites, based on inductively coupled plasma analysis of discrete water samples collected quarterly during 1989^b

	Concentration (mg/L)								
	WCK ^c 6.8	WCK 3.9	WCK 3.4	WCK 2.9	WCK 2.3	MEK ^d 1.6	MEK 0.6	FCK ^e 1.0	FFK ^f 1.1
Aluminum									
Mean	0.28	0.45	0.46	0.48	0.48	0.52	0.52	0.26	0.33
SD ^g	0.16	0.20	0.19	0.21	0.21	0.22	0.22	0.12	0.19
Barium									
Mean	0.072	0.080	0.078	0.076	0.075	0.095	0.088	0.059	0.073
SD	0.053	0.033	0.038	0.034	0.028	0.039	0.040	0.041	0.030
Calcium									
Mean	19	40	41	44	44	48	48	23	25
SD	8.6	4.9	3.5	6.0	5.3	6.9	7.1	8.4	7.2
Copper									
Mean	<0.01	0.013	0.014	0.009	0.008	<0.01	0.009	<0.01	<0.01
SD		0.007		0.004			0.003		
Chromium									
Mean	0.010	0.010	0.010	0.010	0.009	0.007	0.009	0.012	0.012
SD	0.004	0.003	0.003	0.003	0.003	0.003	0.006	0.006	0.03
Iron									
Mean	0.029	0.029	0.019	0.031	0.023	0.044	0.047	0.011	BLD ^h
SD	0.015	0.002	0.005		0.009	0.011	0.009		
Magnesium									
Mean	9.0	9.2	9.0	9.2	8.7	4.8	7.2	9.5	10.8
SD	3.6	1.4	1.5	1.4	1.2	0.6	2.0	3.5	2.8
Manganese									
Mean	0.0135	0.0210	0.0253	0.0245	0.0338	0.0143	0.0595	0.0085	0.0200
SD	0.0049	0.0087	0.0077	0.0106	0.0080	0.0058	0.0291	0.0084	
Sodium									
Mean	BLD	2.6	11.1	14.0	11.3	BLD	3.6	BLD	BLD
SD		0.3	4.4	1.6	4.2		0.8		
Silicon									
Mean	3.5	2.8	2.7	2.6	2.7	3.8	3.4	3.6	3.5
SD	0.3	0.1	0.1	0.1	0.2	0.3	0.3	0.2	0.2
Strontium									
Mean	0.017	0.079	0.082	0.087	0.089	0.082	0.096	0.023	0.023
SD	0.010	0.016	0.013	0.015	0.013	0.013	0.018	0.010	0.007

^aThe following elements were below the listed detection limits (measured in milligrams per liter): Ag, <0.005; As, <0.06; B, <0.08; Be, <0.0004; Cd, <0.002; Co, <0.003; Mo, <0.04; Li, <0.2; Ni, <0.005; Pb, <0.05; Sb, <0.03; Se, <0.06; Sn, <0.05; Ti, <0.02; Zn, <0.008; and Zr, <0.02.

^bValues are means and 1 standard deviation; $n = 4$ for all elements except sodium ($n = 3$).

^cWCK = White Oak Creek kilometer.

Table 2.9 (continued)

^dMEK = Melton Branch kilometer.

^eFCK = First Creek kilometer.

^fFFK = Fifth Creek kilometer.

^gSD = standard deviation.

^hBelow limit of detection (<2 mg/L and <0.01 mg/L for Na and Fe respectively).

3. TOXICITY MONITORING

H. L. Boston, W. R. Hill, L. A. Kszos, C. M. Pettway, and A. J. Stewart

The toxicity monitoring task (Task 1) of the ORNL BMAP has three objectives: (1) identify sources of toxicity in the WOC watershed (Subtask 1a); (2) monitor toxicity of water in WOC and its tributaries and, in the process, assess the usefulness of the toxicity test systems in detecting ambient toxicity (Subtask 1b); and (3) monitor periphyton/microbial communities and use manipulative field experiments to test relationships between ambient toxicity and processes regulating energy flow in streams within the WOC watershed (Subtask 1c). During March 1989 through February 1990, progress was made in each of these areas. This section of the BMAP report provides the results of studies conducted to address these objectives.

3.1 EFFLUENT AND AMBIENT TOXICITY TESTING *(A. J. Stewart and L. A. Kszos)*

3.1.1 Overview

Since initiation of BMAP in March 1986, water from 15 sites on 5 streams has been evaluated for toxicity 19 times. These tests and their attendant chemical analyses have generated a core data set that is used to provide a broad understanding of ambient toxicity patterns in streams at ORNL. The 15 sites used for these ambient studies were initially selected to encompass both point- and area-source contributions to toxicity in the receiving streams. Four of the 15 sites (upstream sites on First Creek, Fifth Creek, WOC, and Melton Branch) have no contaminants

in toxic concentrations and are used as reference sites. The remaining 11 ambient sites are influenced by plant operations. Accordingly, it is reasonable to suppose that at least some of these sites could show evidence of environmental degradation. Of these 11 plant sites, two (WCK 2.65 and MEK 0.16, which are represented as sites 7 and 6, respectively, in Fig. 3.1) are also listed as sites to be evaluated as part of the Toxicity Control and Monitoring Program (TCMP), as stipulated in the ORNL NPDES permit (EPA 1986a, Part V). In this report, we provide toxicity data from these two sites separately (to summarize information relevant specifically to TCMP needs) and in association with data from the other sites that are routinely monitored for ambient toxicity.

As in previous years, point- and area-source contributions to ambient toxicity in streams in the WOC watershed were evaluated using 7-d static-renewal chronic toxicity tests based on the survival and growth of fathead minnow (*Pimephales promelas*) larvae and the survival and reproduction of the daphnid *Ceriodaphnia dubia*. These tests are described in Horning and Weber (1985), Norberg and Mount (1985), and Mount and Norberg (1984).

Toxicity tests with fathead minnow larvae and *Ceriodaphnia* were almost always conducted concurrently, except in the few instances (e.g., SWSA 6 streams, as described in Sect. 3.1.7) when the more sensitive species (*C. dubia*) was used. On each day of a test, subsamples of the effluent or water sample were analyzed for pH, conductivity, alkalinity, and water hardness. Measurements of these

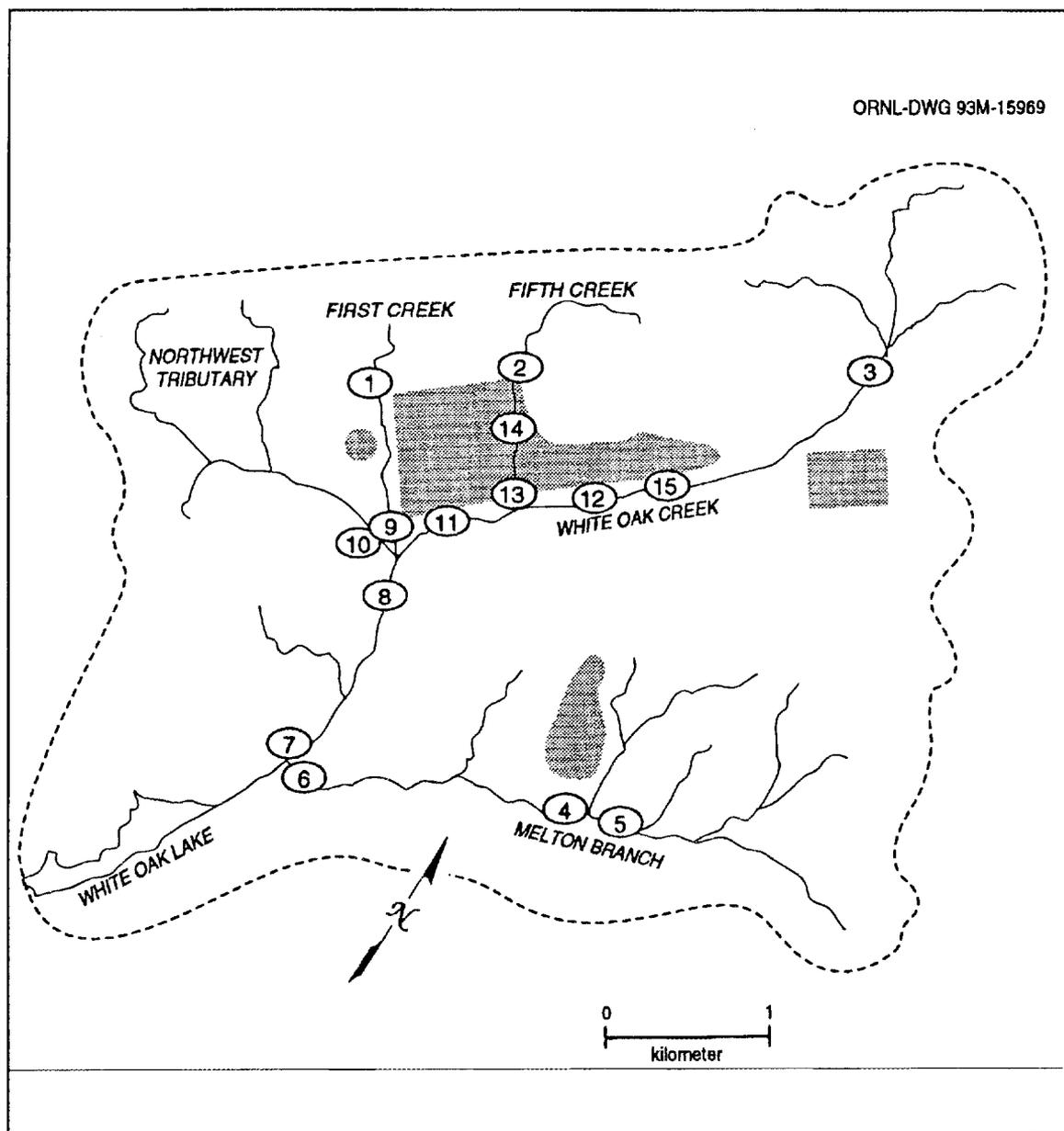


Fig. 3.1. Sampling sites for ambient toxicity tests of water from streams at Oak Ridge National Laboratory.

parameters were made using standard EPA procedures (Kszos et al. 1989). Most samples collected for toxicity tests were also analyzed for free and total residual chlorine (TRC); the concentrations of these materials were measured amperometrically through titrations with standard phenylarsene oxide. The temperature of the water at each site was

recorded when the sample was collected. The toxicity and chemical data for both the ambient and effluent tests are recorded in registered ORNL laboratory notebooks.

All statistical computations were made using personal computer-statistical analysis software (PC-SAS) on International Business Machines (IBM) or National Cash Register (NCR) personal computers.

Survival values for the minnow and *Ceriodaphnia* tests were transformed (arcsin square root; Steel and Torrie 1960) before analysis. Statistical analyses used to evaluate toxicity of effluents are straightforward. Various concentrations (typically five) of each effluent are tested with both species, and data for fathead minnow survival and growth and *Ceriodaphnia* fecundity are then tested for treatment (concentration) effects using one-way analysis of variance [ANOVA; SAS General Linear Models (GLM)]. If a significant ($p < 0.05$) treatment effect is detected through ANOVA, Dunnett's t-test is used to identify which of the tested concentrations caused reductions in fish survival or growth, or *Ceriodaphnia* fecundity. Dunnett's t-test allows means from several treatments to be compared to a single control. *Ceriodaphnia* survival data are analyzed using Fisher's Exact Test based on an acceptance/rejection criterion value of $p < 0.05$.

An analysis of the results of the first 12 ambient tests at ORNL (Stewart et al. 1990) showed that between-test variance differed importantly among sites. Sites that showed evidence of acute or chronic toxicity had larger relative variances (expressed as CV) than sites where no toxicity was evident. Furthermore, that analysis and a later analysis of ambient toxicity data from East Fork Poplar Creek (EFPC) (Loar 1993c) suggested that the results of ambient toxicity tests, in general, could be evaluated effectively using nonparametric analytical procedures. Nonparametric statistical analyses are similar to parametric analyses but use ranks of the parameters of interest rather than their values. After ranks have been generated, the rank values can be analyzed statistically using ANOVA-type procedures (e.g., SAS GLM).

3.1.2 Point-Source Contributions to Toxicity

Point-source discharges to WOC that were systematically evaluated for toxicity under the ORNL TCMP included those from the Sewage Treatment Plant (STP), PWTP, and CYRTE. The frequency with which effluents from the STP and the PWTP were tested was reduced relative to previous years, as allowed under the NPDES permit, because effluents from these two facilities (1) are not very toxic and (2) do not vary much in chemical composition through time. Each of the three facilities has been tested 14 to 15 times since testing was initiated in 1986. The results of these tests are summarized in Table 3.1. Typical no-observed-effect concentrations (NOECs) in relation to the water quality characteristics of the effluents are given in Table 3.2.

The summaries provided in Tables 3.1 and 3.2 show that effluent from the PWTP has not been very toxic (NOECs for the PWTP typically ranged from 80 to 100%). In July 1989, however, this effluent was toxic to both species (NOECs <80%). The conductivity of the effluent during this test period was also unusually high (on average, 1113 $\mu\text{S}/\text{cm}$; in most other test periods, conductivity more typically ranged from ~600–800 $\mu\text{S}/\text{cm}$). Effluent from the STP was not toxic to fathead minnow larvae but was moderately toxic to *Ceriodaphnia* on some occasions (NOECs ranged from 20–100%). The conductivity, alkalinity, and hardness of the STP effluent appeared to be quite stable and were well within the tolerance ranges of both species (Southworth 1992). Effluent from the CYRTE is distinctly more toxic than that of the other two treatment facilities and is particularly problematic for *Ceriodaphnia*. For example, the NOEC for fathead

Table 3.1. Summary of toxicity test results for the Oak Ridge National Laboratory Process Waste Treatment Plant, Coal Yard Runoff Treatment Facility, and Sewage Treatment Plant, May 1986-January 1990

Facility	Test date	NOEC ^a (%)	
		Fathead minnow	<i>Ceriodaphnia</i>
PWTP ^b	July 1986	80	80
PWTP	September 1986	80	80
PWTP	November 1986	80	80
PWTP	January 1987	100	80
PWTP	March 1987	80	80
PWTP	May 1987	80	80
PWTP	July 1987	80	80
PWTP	September 1987	100	<80
PWTP	November 1987	80	100
PWTP	January 1988	80	80
PWTP	March 1988	80	80
PWTP	October 1988	80	NR ^c
PWTP	July 1989	<80	<80
PWTP	November 1989	100	80
CYRTF ^d	July 1986	20	<10
CYRTF	September 1986	>60	5
CYRTF	November 1986	>80	NT ^e
CYRTF	January 1987	60	3
CYRTF	March 1987	60	10
CYRTF	June 1987	≥80	10
CYRTF	August 1987	≥80	25
CYRTF	September 1987	≥60	NT
CYRTF	November 1987	≥80	50
CYRTF	January 1988	≥80	25
CYRTF	March 1988	100	60
CYRTF	July 1988	60	15
CYRTF	September 1988	100	NR
CYRTF	July 1989	12	25
STP ^f	May 1986	100	20
STP	June 1986	NT	100
STP	August 1986	100	100
STP	October 1986	100	100
STP	December 1986	100	75
STP	February 1987	100	<75
STP	March 1987	100	<50
STP	June 1987	100	<75
STP	August 1987	100	50
STP	October 1987	100	<80
STP	December 1987	100	100
STP	February 1988	100	<75
STP	June 1988	100	100
STP	December 1988	100	<80
STP	August 1989	100	100

Table 3.1 (continued)

- ^aNOEC = No-observed-effect concentration.
^bPWTP = Process Waste Treatment Plant.
^cNR = results not reported due to unacceptable controls.
^dCYRTF = Coal Yard Runoff Treatment Facility.
^eNT = not tested.
^fSTP = Sewage Treatment Plant.

Table 3.2. Typical no-observed-effect concentrations for effluents from the Oak Ridge National Laboratory Process Waste Treatment Plant, Coal Yard Runoff Treatment Facility, and Sewage Treatment Plant in relation to ranges of water quality parameters characteristic of the effluents

Facility	Tests	NOEC ^a (%)		Conductivity (μ S/cm)	Alkalinity (mg/L)	Hardness (mg/L)
		Fathead minnow	<i>Ceriodaphnia</i>			
PWTP ^c	14	80-100	80	498-1282	39-80	0-80
CYRTF ^d	14	60-100	10-25	1400-2880	5-40	398-2500
STP ^e	15	100	50-100	370-470	75-115	145-175

- ^aNOEC = No-observed-effect concentration.
^bTests = number of chronic toxicity tests conducted for each facility.
^cPWTP = Process Waste Treatment Plant.
^dCYRTF = Coal Yard Runoff Treatment Facility.
^eSTP = Sewage Treatment Plant.

minnows in CYRTF effluent was, on average, 5.6 times higher than that for *Ceriodaphnia* (Table 3.1), whereas the NOEC for minnows in the PWTP and STP effluents rarely exceeded the *Ceriodaphnia* NOEC by more than a factor of two.

3.1.3 National Pollutant Discharge Elimination System Permit Sites X13 and X14

Water from NPDES sites X14 and X13 (WCK 2.65 and MEK 0.16, respectively) was tested seven times with both species during 1989 (Table 3.3).

None of the seven tests of WCK 2.65 and MEK 0.16 revealed strong evidence for acute or chronic toxicity to either species. In all seven tests, growth of the minnows in water from both sites exceeded that of minnows in the controls. Similarly, fecundity of *Ceriodaphnia* in water from both sites was higher than controls in six of the seven tests. Relative to controls, no significant ($p > 0.05$) reductions in *Ceriodaphnia* survival were found in any test. Fathead minnow controls had 100% survival in five of the seven tests, but survival of the minnows was $\leq 70\%$ in six site-date combinations (Table 3.3).

Table 3.3. Results of fathead minnow (*Pimephales promelas*) and *Ceriodaphnia dubia* chronic toxicity tests of water from National Pollutant Discharge Elimination System sites on Melton Branch and White Oak Creek^a

Test date	Site ^b	Fathead minnow		<i>Ceriodaphnia</i>		n ^c
		Survival (%)	Growth (mg/fish)	Survival (%)	Fecundity (offspring/female)	
January 26, 1989	Control	95.0	0.53 ± 0.04	90	19.2 ± 6.7	10
January 26, 1989	MEK 0.16	80.0	0.64 ± 0.07	60	14.0 ± 2.3	6
January 26, 1989	WCK 2.65	75.0	0.70 ± 0.09	100	15.8 ± 6.5	10
April 6, 1989	Control	72.5	0.30 ± 0.03	100	21.5 ± 1.8	10
April 6, 1989	MEK 0.16	55.0	0.46 ± 0.06	100	22.8 ± 2.9	10
April 6, 1989	WCK 2.65	67.5	0.47 ± 0.07	100	22.3 ± 2.5	10
April 20, 1989	Control	100.0	0.38 ± 0.07	100	22.3 ± 2.6	10
April 20, 1989	MEK 0.16	80.0	0.48 ± 0.13	70	25.1 ± 2.3	7
April 20, 1989	WCK 2.65	90.0	0.39 ± 0.08	80	26.4 ± 3.5	8
May 11, 1989	Control	100.0	0.35 ± 0.03	80	20.5 ± 3.6	8
May 11, 1989	MEK 0.16	67.5	0.47 ± 0.15	100	26.4 ± 5.0	10
May 11, 1989	WCK 2.65	90.0	0.42 ± 0.05	100	21.4 ± 4.5	10
August 17, 1989	Control	100.0	0.40 ± 0.03	90	14.9 ± 2.5	9
August 17, 1989	MEK 0.16	95.0	0.41 ± 0.05	50	20.6 ± 7.3	5
August 17, 1989	WCK 2.65	90.0	0.45 ± 0.05	70	16.1 ± 3.6	7
November 9, 1989	Control	100	0.29 ± 0.05	90	23.4 ± 4.5	9
November 9, 1989	MEK 0.16	67.5	0.33 ± 0.03	100	24.1 ± 3.7	10
November 9, 1989	WCK 2.65	72.5	0.44 ± 0.14	100	26.1 ± 3.0	10
December 14, 1989	Control	100.0	0.27 ± 0.03	80	16.5 ± 2.8	8
December 14, 1989	MEK 0.16	65.0	0.34 ± 0.09	100	18.0 ± 4.0	10
December 14, 1989	WCK 2.65	70.0	0.35 ± 0.06	100	16.9 ± 4.8	10

^aMean percentage survival is based on 40 larvae in each minnow test (10 larvae in each of 4 replicates) and on 10 *Ceriodaphnia* neonates (1 neonate per replicate). Mean growth ± standard deviation (SD) of the fish is corrected for weight of larvae at the start of each test. Mean fecundity of *Ceriodaphnia* (mean ± SD) is computed using only females that survived all 7 d. Any difference between (*Ceriodaphnia* survival/10) and *n* (the number of females used to calculate fecundity) is due to the presence of males, which were used to compute survival but were excluded from computations of fecundity. The controls in each test consisted of animals reared in 10% diluted mineral water (DMW), except in tests conducted in November and December, when the control water was 20% DMW. Test date indicates the first day of the 7-d test.

^bMEK = Melton Branch kilometer, and WCK = White Oak Creek kilometer.

^c*n* = the number of females used to calculate fecundity.

Less than 70% survival in effluent tests with fathead minnow larvae frequently is indicative of toxicity. However, the use of fathead minnow larvae in ambient

testing applications on the ORR suggests that pathogens, such as the aquatic fungus, *Saprolegnia*, which facultatively attacks fish, can affect survival. In fathead minnow

tests of ambient waters, a trademark of pathogenic intrusion seems to be an unusually large variation in survival among replicates. All or most of the larvae in some replicates may die, but survival in potential difference in variance patterns for ambient and effluent toxicity tests that are based on fathead minnow survival could be important. The presence of a significant difference in variance pattern, for example, would imply the need for two toxic criteria for the survival endpoint of the fathead minnow test: one to be used in effluent applications and another for ambient applications. This point is not trivial; for WCK 2.65 and MEK 0.16, fish survival was low enough to be suspect ($\leq 70\%$, on average) in four of the seven test periods. Thus, in more than half of the tests at WCK 2.65 and MEK 0.16, a criterion suitable for assessing effluent toxicity could be inappropriately stringent.

Part of the problem in determining whether or not either or both sites should be considered toxic is statistical. For example, when survival of the controls is 100%, no variance exists among the control replicates. When only a few treatments are being compared and one of the treatments has no variance, statistical analyses that use variance partitioning (e.g., ANOVA) provide little insight. For this reason, an alternative type of analysis for the tests at WCK 2.65 and MEK 0.16 was deemed more appropriate.

To evaluate this possibility, minnow survival among replicates in ambient tests in which mean survival was low (40–68%) were compared to those for effluent tests in which survival was low (40–68%) (Table 3.4). First, the relative variation (or among-replicate range in percentage survival divided by the geometric mean for survival multiplied by 100) was calculated for each test (Table 3.4). Then the relative variation values were ranked because their underlying distribution(s) for the two types of tests (ambient vs effluent)

were unknown. Finally, the rank values were analyzed using one-way ANOVA. The ANOVA showed that the ranked values for relative variation for the two tests were significantly different ($p < 0.013$; $DF = 1$). The test type also explained 22.9% of the total variance in ranked values of relative variation. Thus, it can be concluded that fathead minnow survival criteria useful in determining effluent toxicity are too stringent in ambient applications. This finding is addressed in greater detail later in Sect. 3.1.6.

Measurements of pH, conductivity, alkalinity and hardness obtained for WCK 2.65 and MEK 0.16 were made in conjunction with the toxicity tests described above. Summary statistics (e.g., mean and SD) for these data are given in Table 3.5. WCK 2.65 and MEK 0.16 did not differ dramatically with respect to any of the measured parameters. However, on most dates, the conductivity at WCK 2.65 was slightly higher (typically by ~5–20%) than the conductivity of the water at MEK 0.16, whereas alkalinity tended to be slightly lower at WCK 2.65 than it was at MEK 0.16.

3.1.4 Point-Source Chlorine Studies

A general plant project (GPP) plan is presently being developed to reduce the loading rates of chlorine to WOC (L. R. Simmons, ORNL, Environmental and Health Protection Division, personal communication). To determine whether or not toxic materials other than chlorine were present in some of the wastewaters that are likely to be included in this project, water samples were collected from five outfalls (NPDES stations 210, 211, 217, 227, and 312). After the samples were treated with sodium thiosulfate to eliminate toxicity from chlorine, they were tested for toxicity to *Ceriodaphnia* and

Table 3.4. Within-treatment variation in fathead minnow survival for toxicity tests in which average survival was low (40 to 68%)

Sampling site description and sample concentration	Date	Fathead minnow survival (%)				Mean	Relative ^a variation
		Replicate					
		1	2	3	4		
Ambient samples^b							
ORNL ^c , X13, grab, 100%	April 1989	60	60	10	90	55.0	145.5
ORNL, X14, grab, 100%	April 1989	80	50	50	90	67.5	59.2
ORNL, X13, grab, 100%	May 1989	50	70	70	80	67.5	44.4
ORNL, X13, grab, 100%	November 1989	40	60	90	80	67.5	74.1
ORNL, X13, grab, 100%	December 1989	60	20	10	80	65.0	107.7
ORNL, WCK 6.8, grab, 100%	September 1986	100	30	30	80	60.0	116.7
ORNL, WCK 6.8, grab, 100%	December 1989	60	80	90	30	65.0	92.3
ORNL, FCK 0.9, grab, 100%	February 1988	70	60	40	100	67.5	88.9
ORNL, FCK 0.9, grab, 100%	December 1989	50	10	50	50	40.0	100.0
Y-12, BCK NT 4, grab, 100%	April 1985	0	20	100	100	55.0	181.8
Y-12, BCK NT 14, grab, 100%	April 1985	10	100	100	10	55.0	163.6
Y-12, BCK 5.15, grab, 100%	October 1985	50	0	30	100	45.0	222.2
ORGDP ^d , MIK 1.43, grab, 100%	January 1987	50	70	90	0	52.5	171.4
ORGDP, MIK 1.43, grab, 100%	March 1988	50	80	100	30	65.0	107.7
Effluent samples							
Y-12, Nontreated PR ^e , 10%	April 1987	20	70	50	50	47.5	105.3
Y-12, Nontreated PR, 20%	July 1988	60	50	60	50	55.0	18.2
ORGDP, TSCA ^f diagnostic tests:							
2115-23-1, 0.5% concentration	February 1989	70	60	50	50	57.5	34.8
2115-23-2, 0.5% concentration	February 1989	60	50	10	50	42.5	117.6
2115-23-3, 0.5% concentration	February 1989	20	70	50	70	52.5	95.2
2115-23-4, 0.5% concentration	February 1989	60	40	50	40	47.5	42.1
Y-12, CPCF ^g , 5% concentration	October 1986	40	60	40	80	55.0	72.7
ORNL, CYRTF ^h tests:							
80% concentration	July 1986	60	60	50	90	65.0	61.5
60% concentration	January 1987	50	60	80	80	67.5	44.4
80% concentration	July 1988	60	40	50	80	57.5	69.6
100% concentration	July 1989	20	30	50	60	40.0	100.0
Y-12, WETF ⁱ , 30% concentration	April 1988	90	50	30	70	60.0	100.0

^aRelative variation was computed as 100 times the range among replicates (4 replicates were used in each case) divided by the mean.

^bResults are given for ambient and effluent samples. (Treatment facility locations, test dates, and effluent concentrations used in the analysis are given.) Stream codes for ambient sites are (FCK) First Creek kilometer; (BCK) Bear Creek kilometer; (NT) north tributary to Bear Creek; and (MIK) Mitchell Branch kilometer; X13 = NPDES station on Melton Branch; X14 = NPDES station on White Oak Creek.

^cORNL = Oak Ridge National Laboratory.

^dORGDP = Oak Ridge Gaseous Diffusion Plant; currently the K-25 Site.

^eCategory IV (nontreated) photographic rinsewaters (Building 9981, Outfall 414).

^fTSCA = Toxic Substance Control Act incinerator.

^gCentral Pollution Control Facility National Pollutant Discharge Elimination System (NPDES) Outfall 501.

^hCYRTF = Coal Yard Runoff Treatment Facility.

ⁱWest End Treatment Facility (NPDES Outfall 502).

Table 3.5. Summary of chemistry data for toxicity tests of water from National Pollutant Discharge Elimination System points X13 (Melton Branch kilometer 0.16) and X14 (White Oak Creek kilometer 2.65)^a

Test date	Site ^b	pH (standard units)	Conductivity ($\mu\text{S}/\text{cm}$)	Alkalinity (mg/L)	Hardness (mg/L)
January 26, 1989	Control	8.07 \pm 0.3	83 \pm 1.6	31 \pm 2.5	40 \pm 1.4
	MEK 0.16	8.04 \pm 0.1	298 \pm 22.1	122 \pm 10.6	161 \pm 14.0
	WCK 2.65	8.08 \pm 0.1	356 \pm 47.8	109 \pm 9.6	155 \pm 19.6
April 6, 1989	Control	7.83 \pm 0.2	87 \pm 1.4	31 \pm 2.6	45 \pm 9.8
	MEK 0.16	7.97 \pm 0.1	137 \pm 48.0	110 \pm 8.1	138 \pm 14.2
	WCK 2.65	8.00 \pm 0.1	287 \pm 44.9	106 \pm 6.2	152 \pm 16.8
April 20, 1989	Control	7.90 \pm 0.5	84 \pm 3.7	31 \pm 3.5	43 \pm 2.0
	MEK 0.16	8.06 \pm 0.1	341 \pm 9.8	143 \pm 18.4	184 \pm 24.6
	WCK 2.65	8.05 \pm 0.1	316 \pm 18.1	120 \pm 12.7	162 \pm 18.3
May 11, 1989	Control	7.89 \pm 0.3	88 \pm 2.1	33 \pm 3.2	43 \pm 4.3
	MEK 0.16	8.06 \pm 0.1	309 \pm 36.2	133 \pm 15.1	156 \pm 20.1
	WCK 2.65	8.05 \pm 0.1	311 \pm 27.3	108 \pm 3.4	146 \pm 11.4
August 17, 1989	Control	7.85 \pm 0.2	89 \pm 5.8	35 \pm 2.4	44 \pm 3.7
	MEK 0.16	8.03 \pm 0.1	334 \pm 6.2	132 \pm 3.1	165 \pm 6.2
	WCK 2.65	8.05 \pm 0.1	355 \pm 34.5	114 \pm 4.2	148 \pm 11.2
November 9, 1989	Control	8.14 \pm 0.1	163 \pm 0.8	62 \pm 2.1	81 \pm 2.5
	MEK 0.16	7.93 \pm 0.1	304 \pm 33.1	126 \pm 10.8	156 \pm 17.3
	WCK 2.65	7.96 \pm 0.1	339 \pm 91.3	114 \pm 11.6	164 \pm 38.7
December 14, 1989	Control	8.11 \pm 0.1	161 \pm 4.3	61 \pm 2.8	83 \pm 4.3
	MEK 0.16	8.02 \pm 0.1	285 \pm 36.8	128 \pm 7.7	160 \pm 17.3
	WCK 2.65	8.05 \pm 0.1	359 \pm 57.9	123 \pm 17.4	163 \pm 35.0

^aData are 7-d means (\pm standard deviation) for grab samples collected daily during each indicated test period.

^bMEK = Melton Branch kilometer, and WCK = White Oak Creek kilometer.

fathead minnow larvae. The results of these tests, which started on February 8, 1990, are summarized in Table 3.6.

The results of the *Ceriodaphnia* tests indicated that toxicants other than chlorine were present in three of the five outfalls. In dechlorinated water from Outfall 217, the survival of *Ceriodaphnia* decreased

gradually over the 7-d test period. By the end of the test, the reduction in survival was statistically significant ($p < 0.05$; Table 3.6). Although *Ceriodaphnia* survival was high in dechlorinated water from Outfall 227, fecundity of the animals was very low (3.0 offspring per female; Table 3.6). A reduction in fecundity of

Table 3.6. Results of fathead minnow (*Pimephales promelas*) and *Ceriodaphnia dubia* chronic toxicity tests of dechlorinated water samples collected from five White Oak Creek outfalls^a

Sample	Fathead minnow		<i>Ceriodaphnia</i>	
	Survival ^b (%)	Growth ^c (mg/fish)	Survival ^{b,d} (%)	Fecundity ^e (offspring/female)
Control	97.5	0.25 ± 0.02	100	21.1 ± 2.1
210	100	0.27 ± 0.04	90	18.3 ± 5.7
211	100	0.29 ± 0.02	100	12.5 ± 4.1
217	100	0.32 ± 0.01	50*	13.6 ± 2.1
227	97.5	0.34 ± 0.05	90	3.0 ± 4.0
312	97.5	0.23 ± 0.07	0*	0

^aNational Pollutant Discharge Elimination System sites 210, 211, 217, 227, and 312.

^bMean percentage survival is based on 40 larvae in each minnow test (10 larvae in each of 4 replicates) and on 10 *Ceriodaphnia* neonates (1 neonate per replicate).

^cMean growth (± SD) of the fish is corrected for weight of larvae at the start of each test.

^dAsterisks show samples that significantly ($p < 0.05$) reduced survival using Fisher's Exact Test.

^eMean fecundity of *Ceriodaphnia* (± SD) is computed using only females that survived all 7 d.

this magnitude is clearly indicative of chronic toxicity. Finally, all *Ceriodaphnia* that were tested in dechlorinated water from Outfall 312 died within 24 h, demonstrating that this water was acutely toxic. Survival of fathead minnow larvae in the tests of dechlorinated water from the five outfalls was ≥97.5% in each case. Growth of the fish in water from Outfall 312 was slightly lower than that of fish in the controls, but minnow growth in water from the other four outfalls was higher than that of the control. Thus, the fish test showed no evidence of acute toxicity in the dechlorinated samples and little evidence of chronic toxicity. Conductivity, pH, hardness and alkalinity values for the dechlorinated samples did not reveal unusual conditions (Table 3.7).

Additional experiments were conducted to identify the material(s) contributing to the acute toxicity of dechlorinated water from Outfall 312. A toxicity identification effort (TIE) based on *Ceriodaphnia* survival was used for this purpose. First, a *Ceriodaphnia* test was conducted to confirm the presence of acute toxicity of water from Outfall 312; this test was started on the morning of February 9, 1990. All animals in the dechlorinated water died within 3 h, thus confirming the severity of the problem. On February 13, another sample was collected from Outfall 312 and dechlorinated with sodium thiosulfate as in the preceding tests. Subsamples of the dechlorinated water were then amended with different concentrations of a strong

Table 3.7. Chemical parameters for five White Oak Creek outfalls^a

Outfall number	pH (standard units)	Conductivity ($\mu\text{S}/\text{cm}$)	Alkalinity (mg/L)	Hardness (mg/L)	TRC ^b (mg/L)
210	7.84	248	93	142	0.97
211	7.84	248	91	126	0.88
217	7.82	264	100	134	1.15
227	7.77	268	99	142	0.70
312	7.79	182	68	108	0.67

^aMeasurements were made on freshly collected grab samples taken on February 8, 1990. Each sample was dechlorinated with sodium thiosulfate before being tested for toxicity.

^bTRC = total residual carbon.

chelating agent [magnesium-ethylene diamine tetra-acetic acid (Mg-EDTA)] before being tested for acute toxicity. The results of the Mg-EDTA test showed that acute toxicity was eliminated by low concentrations (15–20 μM) of Mg-EDTA (Table 3.8), which suggested that a metal was the toxicant. On February 15, a dilution-series analysis of acute toxicity to *Ceriodaphnia* was conducted. The dose-response relationship for dechlorinated water from Outfall 312 indicated that instream acute toxicity attributable to this outfall was unlikely (Table 3.9). Water is discharged from Outfall 312 at a rate of about 23 L/min. At WCK 5.47 (upstream from Outfall 312), the annual average stream flow ranges from about 6.3 L/s (in 1987; Loar 1993b) to 63.1 L/s (in 1989; Table 2.2 of this report). Thus, the instream waste concentration for water entering WOC from Outfall 312 is on the order of <1–6% on an annual basis. Extrapolating from the acute toxicity of

water from Outfall 312 suggests that this water could show evidence of chronic toxicity at a concentration of 5–15% of full strength. Considerable uncertainty is associated both with the estimated instream waste concentration and the chronic NOEC for water from Outfall 312, but the similarity in their ranges (<1–6% vs 5–15%) suggests that they should be investigated further.

The results of chemical analyses of nontreated water from Outfall 312 (collected on three dates) are shown in Table 3.10. Concentrations of zinc, and especially copper, were higher than those typical for tap water (Table 3.10), suggesting that one or both of these metals could be toxicants. A dilution-series test of water from WOC upstream from Outfall 312 will be used to determine if copper, when added to dechlorinated tap water to a final concentration of $\sim 40 \mu\text{g}/\text{L}$, can account for the toxicity observed.

Table 3.8. Number of *Ceriodaphnia* neonates surviving in control water and in a dechlorinated sample of water from White Oak Creek Outfall 312^a

EDTA ^b concentration (μ M)	Number of surviving <i>Ceriodaphnia</i> neonates			
	Control water ^c		Outfall 312 ^d	
	5 h	21 h	5 h	21 h
0	5	5	0	0
8	5	5	5	0
16	5	5	5	4
24	5	5	5	5
32	5	5	5	5
40	5	5	5	4
48	5	4	5	5
56	5	5	5	5
64	5	5	5	5

^aThis analysis occurred after the two water samples had been amended with various concentrations of a metal-chelating agent (magnesium–ethylene diamine tetra-acetic acid). The treatments were not replicated: single beakers contained five live neonates at the start of the test.

^bEDTA = Ethylene diamine tetra-acetic acid.

^cDiluted mineral water (20% by volume).

^dDechlorinated with sodium thiosulfate before being amended with EDTA.

Table 3.9. Number of *Ceriodaphnia* surviving in various concentrations of dechlorinated water from White Oak Creek Outfall 312 as a function of time^a

Concentration (%)	No. of surviving <i>Ceriodaphnia</i>								
	Elapsed time								
	0 h	2 h	3 h	4 h	5 h	6 h	8 h	22 h	26 h
100	5	0	0	0	0	0	0	0	0
90	5	0	0	0	0	0	0	0	0
80	5	2	2	1	0	0	0	0	0
70	5	4	1	0	0	0	0	0	0
60	5	2	1	0	0	0	0	0	0
50	5	1	0	0	0	0	0	0	0
40	5	5	3	1	1	0	0	0	0
30	5	5	5	5	4	3	3	2	1
20	5	5	5	5	5	5	5	5	5
15	5	5	5	5	5	5	5	4	4
10	5	5	5	5	5	5	5	5	5
5	5	5	5	5	5	5	5	5	5
Control	5	5	5	5	5	5	5	5	5

^aSingle replicates of each concentration, each containing five neonates, were used to estimate acute toxicity. Dilutions of the dechlorinated wastewater were prepared using diluted mineral water (20%).

Table 3.10. Concentrations of selected trace metals in water from White Oak Creek Outfall 312 and in dechlorinated tap water

Metal	Concentration ($\mu\text{g/L}$)	
	Outfall 312 ^a	Tap water ^b
Cadmium	<7.0	2.4
Chromium	<4.0	<6.0
Copper	39.0 \pm 3.0	<10.0
Manganese	<2.0	<5.0
Nickel	<11.7 \pm 2.5	12.0
Silver	<5.0	<6.0
Zinc	85.7 \pm 7.6	35.0

^aValues for Outfall 312 are the mean \pm standard deviation for single grab samples taken on three dates in February 1990.

^bValues for the tap water represent samples from a single date.

3.1.5 Area-Source Chlorine Studies

Drinking water used as once-through cooling water at ORNL enters WOC and its tributaries through various storm drains and process outfalls. Some of these wastewaters contain chlorine in concentrations >1 mg/L, a concentration that is acutely toxic to most aquatic biota (EPA 1985). However, chlorine tends to be chemically unstable and is gradually converted to chloride (Cl^-), which is not toxic. The conversion of chlorine to chloride can be hastened by the presence of labile, naturally occurring dissolved organic matter. Conversely, if nitrogenous compounds are present, chlorine can form toxic mono- and dichloramines, which are more stable than free chlorine (OCl_2). Finally, it is likely that water temperature affects chlorine persistence, simply because chemical reaction rates tend to increase with temperature. For these reasons, the chlorine in water from different sites in WOC, or even from one site at different times of year, may have markedly different persistence characteristics.

An experiment was conducted to estimate the decay rates of chlorine in water samples from three locations in WOC (WCK 6.8, 4.4, and 3.5). These sites were selected because their rate of converting high levels of chlorine to nontoxic levels was expected to differ. Water from WCK 6.8, for example, is not exposed to chlorine and should contain compounds (e.g., naturally occurring dissolved organic matter) that promote the conversion of chlorine to chloride. Water from WCK 4.4, on the other hand, often contains measurable levels of TRC and, thus, should contain little remaining material capable of converting chlorine to chloride. Finally, WCK 3.5 is located downstream from the outfall of the STP but about 50 m upstream of the confluence of WOC and NT. The water from WCK 3.5 was expected to contain nitrogenous organic compounds that should stabilize chlorine.

Grab samples (10-L) of water from each of the three sites were collected on February 6, 1990. Measurements of pH, conductivity, TRC, free chlorine, alkalinity,

and hardness were obtained in the laboratory and a subsample from each sample was saved for analysis of nitrate and phosphate. Each sample was then amended with sufficient chlorine (Chlorox[®]) to yield a TRC concentration ranging from 2.60–2.76 mg/L. Each sample was then immediately divided into two equal portions; one was incubated at room

temperature (16–18°C), and the other was incubated in a water bath at 9–10°C. Over the following 5 d, subsamples of each of the six chlorinated water samples were periodically withdrawn and analyzed for TRC and free chlorine. Each sample was kept mixed by gentle aeration during the experiment. The results of the chlorine-persistence study are shown in Fig. 3.2.

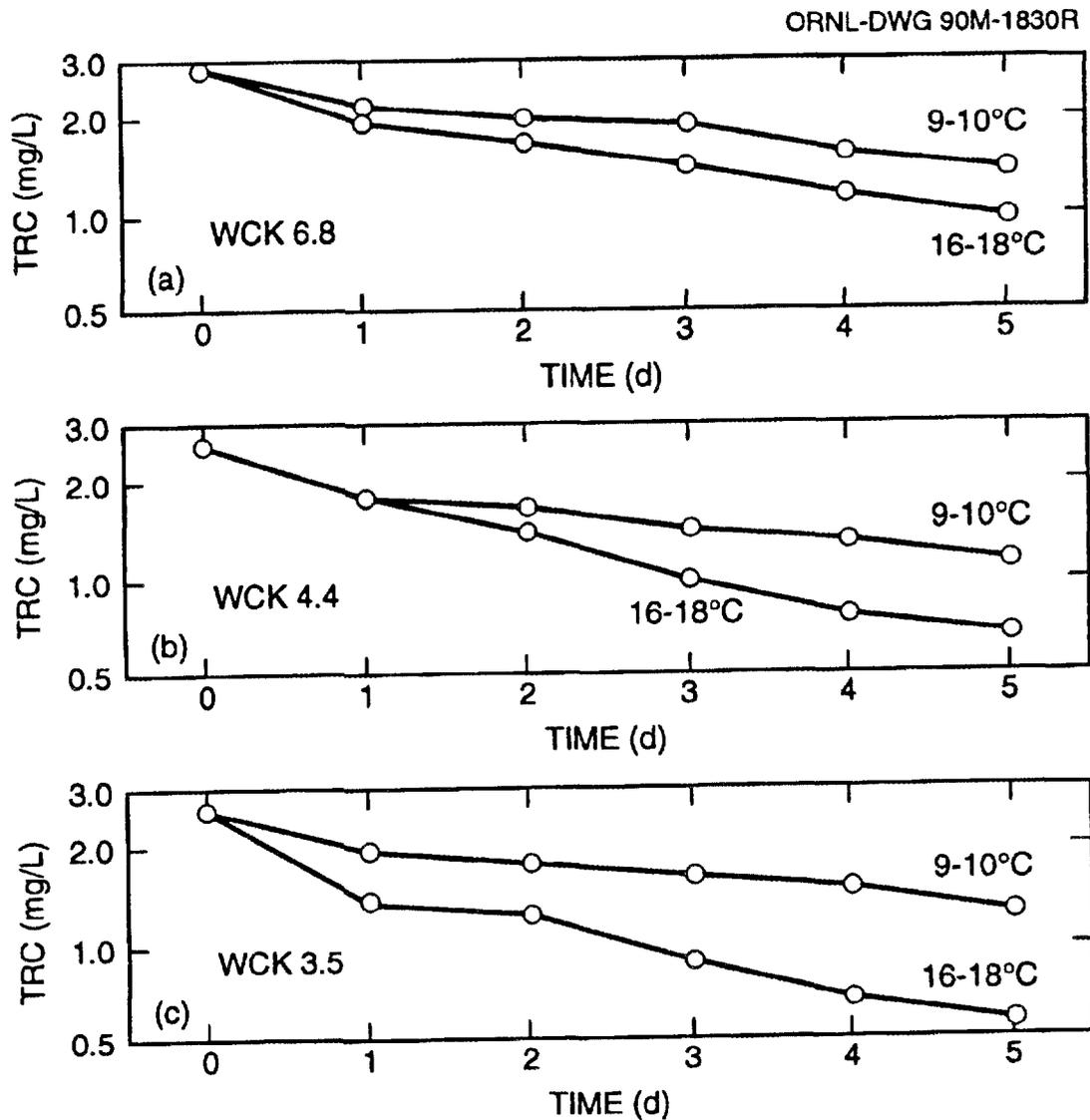


Fig. 3.2. Loss of total residual chlorine (TRC) from aerated water samples from three White Oak Creek sites: (a) White Oak Creek kilometer (WCK) 6.8, (b) WCK 4.4, and (c) WCK 3.5 under two temperature regimes (9–10°C and 16–18°C). TRC concentrations (y-axis) are shown on a log¹⁰ scale.

Several major findings resulted from this study. First, TRC levels in the water from each site declined log-linearly over the 5-d experiment, but in each instance, the rate of decline was quite low. The estimated time needed to reduce a concentration by 50% (t_{50}), for example, ranged from about 2.65 d (in 16–18°C water from WCK 3.5) to as much as 4.60 d (in 9–10°C water from WCK 6.8). Second, the rate at which TRC declined was conspicuously affected by temperature, with the rate of decline being more rapid at the higher temperature in each case. Third, the influence of temperature was greatest for water from WCK 3.5 and least for water from WCK 6.8 (Fig. 3.2).

Using available data on (1) instream TRC concentrations at various sites in Fifth Creek and WOC, (2) water temperature at various sites in Fifth Creek and WOC, (3) water velocity at various locations in the streams, and (4) estimated TRC decay rates given above, Monte Carlo simulations could be used to provide reasonably accurate estimates of downstream excursions of toxic levels of TRC. For example, assume that the TRC concentration at WCK 4.4 increased briefly to 0.25 mg/L, that water between WCK 4.4 and WCK 3.5 flows at a uniform rate of 10 cm/s, and that no dispersion occurs. If TRC in the contaminated parcel of water decays at the fastest rate measured in the experiments described above, the TRC concentration in the chlorine-contaminated parcel of water would still be ~0.24 mg/L when it reached WCK 3.5. However, if the TRC decay rate increases linearly with temperature, as extrapolated using data from the two temperatures evaluated in the experiment above, TRC should decay about 11 times faster at 25°C than at 10°C. Thus, it is important to conduct other controlled studies to evaluate the relationship between water temperature and TRC persistence more carefully.

The decay rates of TRC estimated from the experiment described above are likely to underestimate true decay rates, because TRC moving downstream with the flow of water is continuously exposed to "new" organic matter produced photosynthetically by periphyton, which can survive the chlorination. Accordingly, more accurate models of TRC decay in the stream should also consider (1) the relationship between water velocity and the ratio of water volume to substrate surface area available to interact with the TRC and (2) the rate of production of "new" organic carbon in relation to the rate at which TRC is transported downstream. Studies in this area will be continued next year to provide the data needed to support an effective modeling effort that relates chlorine distribution patterns to instream toxicity.

In late 1989 and early 1990, measurements of TRC and the volumes of cooling water discharged to Fifth Creek (P. A. Taylor, ORNL, Chemical Technology Division, personal communication) suggested that loading rates of chlorine to Fifth Creek have declined somewhat during the past 2 years. To estimate the change in TRC loading rates to WOC and Fifth Creek since BMAP's inception, the frequency-of-occurrence patterns for TRC were evaluated at four sites (FFK 0.4, FFK 0.0, WCK 4.4, and WCK 3.8). The results of this analysis suggest that conditions are not improving at any of these sites (Table 3.11). At each site, the frequency of nondetectable levels of TRC was lower in 1987–89 than in 1986–87, and the frequency of high TRC values (i.e., $\geq 100 \mu\text{g/L}$ and $\geq 200 \mu\text{g/L}$) was higher. Possible explanations for these observations include (1) a true worsening of conditions in the streams due to greater inputs of chlorine, (2) temporary worsening of conditions in the streams due to significant year-to-year differences in

Table 3.11. Frequency of occurrence of three total residual chlorine (TRC) concentrations at four sites in White Oak Creek and Fifth Creek during 1986–1987 and 1987–1989

Stream site ^a	Frequency of occurrence for TRC concentrations (%)					
	1986–1987 ^b			1987–1989 ^c		
	0	≥100	≥200	0	≥100	≥200
FFK 0.4	9.5	15.5	1.1	8.1	29.7	8.1
FFK 0.0	8.3	41.7	10.7	7.1	40.5	22.8
WCK 4.4	60.7	7.1	1.2	16.2	55.0	31.5
WCK 3.8	19.0	28.6	8.3	11.9	69.0	20.5

^aFFK = Fifth Creek kilometer, and WCK = White Oak Creek kilometer.

^bThere were 84 observations for each site in 1986–1987.

^cThe number of observations for each site in 1987–1989 ranged from 37 to 42.

discharge due to drought or unusually wet conditions, or (3) a statistical bias due to differences in sampling regime for the two time periods. During 1986–87, for example, samples were collected once each month for 12 months, whereas sampling frequency was much lower during 1987–89. During 1990–91, which concludes the first 5-year assessment period for BMAP, sampling frequency is scheduled to increase so that direct comparisons with the 1986–87 data set will be possible.

3.1.6 Overview of the Ambient Toxicity Data Sets

Since BMAP was initiated in March 1986, water from 15 sites on 5 streams has been evaluated for toxicity 19 times. These tests, and their attendant chemical analyses, have generated a core data set that can be used to describe toxicity conditions in the streams, to detect changes in toxicity patterns through time, and to provide a broad understanding of ambient toxicity patterns in ORNL streams. Conceptually, the results of the

ambient tests described above have generated five matrices—one each for fathead minnow survival, fathead minnow growth, *Ceriodaphnia* survival, *Ceriodaphnia* fecundity, and chemical water quality. [The chemical-data matrix can be considered a single three-dimensional matrix (6 × 15 × 17) whose six “depth” cells are separate parameters (i.e., water temperature, pH, conductivity, alkalinity, hardness, and TRC) or as six separate matrices, each containing 15 × 17 cells]. Each *Ceriodaphnia* and fathead minnow matrix contains up to 255 cells (15 sites × 17 dates). The values for some cells in the matrices are missing (e.g., some sites were not tested during every test period), and the number of values per cell differs among the matrices (e.g., seven daily values are available for most chemical-parameter cells, whereas *Ceriodaphnia* survival yields a single value (0, 10, ...100%) in the *Ceriodaphnia*-survival matrix). Various “new” matrices can be generated through within- and between-cell computations. For example, a growth-variance matrix can be created through within-cell computations of the fathead minnow

growth matrix, an alkalinity/hardness-ratio matrix can be created using between-cell computations in the chemical data matrix, and so forth. Additionally, the size of the data sets is large enough to allow robust statistical evaluations even when particular rows or columns of a matrix are excluded.

3.1.6.1 Statistical analyses for ambient toxicity assessments

Very few studies to date have effectively explored the utility of static-renewal single-species tests to evaluate biological quality of ambient waters. Important exceptions to this void include studies by Nimmo et al. (1990) and Stewart et al. (1990). The study by Nimmo et al. (1990) focused on using acute and chronic tests with *Ceriodaphnia* and chemical analyses to evaluate the water quality of three streams impacted by non-point sources of metals from mining operations in Colorado, Montana, and South Dakota. In this study, a total of 48 site-date combinations (*Ceriodaphnia* only) were considered. In some cases, toxicity was estimated by testing various dilutions of ambient water samples that were shown to be acutely toxic at full strength. The authors concluded that instream toxicity was probably caused by copper and zinc. The study by Stewart et al. (1990) used chronic tests with two species (fathead minnows and *Ceriodaphnia*) to explore relationships between ambient toxicity and water chemical parameters for five streams in southeastern Tennessee that receive various industrial wastewaters. The investigators used 180 site-date combinations but did not use any dilution-series tests to estimate toxicity when stream water samples were found to be acutely toxic. Principal components analyses implicated episodically high levels of chlorine as controlling overall toxicity patterns for both species, and evidence was

presented showing the value of nonparametric statistics in ambient toxicity assessments.

Data from the four 15×19 toxicity test matrices described in Sect. 3.1.5 were used to (1) evaluate the concept that nonparametric approaches are advantageous in assessments of ambient toxicity, (2) identify sites in ORNL streams that appear to be adversely affected by poor water quality, and (3) provide guidance on the most appropriate data for making cost-effective assessments of ambient toxicity.

To achieve these objectives, subsets of the four biotest matrices were developed. This process required that several tests be excluded because missing values at some sites prevented the development of a uniform ranking strategy. Excluded were tests 15, 16, and 18 with fathead minnows and tests 9, 11, 15, and 16 with *Ceriodaphnia*. Also, the site on upper Melton Branch (MEK 1.8) was excluded because it was frequently dry in earlier test periods, resulting in many missing data. Because all cells need values to generate site-to-site or test-to-test rankings that can be used for nonparametric analyses, zeros were inserted for fathead minnow growth and *Ceriodaphnia* fecundity in all cases where survival was zero. Additionally, the *Ceriodaphnia*-fecundity matrix included data only for females that survived all 7 d in any site-date combination.

Next, each of the four response parameters (minnow and *Ceriodaphnia* survival, minnow growth, and *Ceriodaphnia* fecundity) was separately ranked, by site, within each test; ties received the average rank. To test the question, "Are the ranks between sites different?" a SAS-GLM (models = rank) analysis was conducted. The results showed that differences ($p < 0.001$) among sites could be detected for three of the four response parameters (Table 3.12). In the analysis, the two *Ceriodaphnia* test endpoints had similar

Table 3.12. Results of the analysis of variance of the ranks of sites, within tests, included in ambient toxicity assessments of Oak Ridge National Laboratory streams using *Ceriodaphnia* (survival and fecundity endpoints) and fathead minnow larvae (survival and growth endpoints)^a

Test and endpoint	DF	R ²	F	Prob. >F
<i>Ceriodaphnia</i> survival	223	0.25	5.34	0.0001
<i>Ceriodaphnia</i> fecundity	223	0.26	5.62	0.0001
Fathead minnow survival	237	0.17	3.65	0.0001
Fathead minnow growth	223	0.09	1.57	0.0968

^aSite-date combinations excluded from the analysis due to missing values in some cells are described in text.

Note: R² = ratio of variance.

resolution. [In other words, they explained about the same amount of variance; 25 and 26% (each with DF = 223) for survival and fecundity respectively.] Even though the survival endpoint of the minnow test explained only 65% as much of the total variance as either of the *Ceriodaphnia* endpoints (17%; Table 3.12), it was still highly significant ($p < 0.001$). The growth endpoint of the fathead minnow test explained little variance (9%) and was not significant ($p > 0.097$; Table 3.12). Duncan's multiple range test used in association with this analysis showed that different sites were "better" or "worse" than others ($p < 0.05$) based on *Ceriodaphnia* survival and reproduction and on fathead minnow survival but not growth (Table 3.13).

Following these analyses, each of the four response parameters was separately ranked, by test, within each site. As before, ties were given average ranks. Again, the question of whether the ranks between tests are different was tested using SAS-GLM. The results of this analysis showed significant ($p < 0.001$) test-to-test differences in three of the four possible endpoints—*Ceriodaphnia* reproduction, fathead minnow survival, and fathead

minnow growth (Table 3.14). As described previously, Duncan's multiple range test was then used to identify those test periods in which the endpoints were significantly ($p < 0.05$) higher or lower than average. This information is given in Table 3.15.

When using toxicity tests to assess ambient water quality, a bioassay should simultaneously (1) discriminate readily among sites and (2) exhibit little test-to-test variation. Data provided in Tables 3.12 and 3-14 were used to determine which of the two tests (*Ceriodaphnia* vs fathead minnow) best fulfilled these objectives. The suitability of the two types of tests in ambient applications was evaluated by computing, for each endpoint, a ratio of the variance (R²) explained in the site-to-site analysis (Table 3.12) to that explained in the test-to-test analysis (Table 3.14). In this approach, a high ratio is better than a low one because it indicates that the endpoint being evaluated is relatively better at discriminating among sites than at discriminating among tests. The relative "goodness" of each type of test was then computed by summing the variance ratios for both endpoints. This calculation showed that the *Ceriodaphnia* test was

Table 3.13. Sites that were significantly better or significantly worse (Duncan's multiple range test, $p < 0.05$) than average based on the analysis of variance of ranks of sites within tests^a

	<i>Ceriodaphnia</i> survival	<i>Ceriodaphnia</i> fecundity	Fathead minnow survival
Significantly better	— ^b —	MEK ^c 0.2 NTK ^c 0.1	WCK ^d 3.8 —
Significantly worse	WCK 3.8 WCK 4.4 FFK ^e 0.0	WCK 4.4 — —	FCK ^f 0.9 MEK 1.4 WCK 5.1

^aFathead minnow growth was not included in the analysis because it did not explain a significant amount of variance (see Table 3.12).

^bNo significant difference.

^cMEK = Melton Branch kilometer.

^dWCK = White Oak Creek kilometer.

^eNTK = Northwest Tributary kilometer.

^fFCK = First Creek kilometer.

^gFFK = Fifth Creek kilometer.

Table 3.14. Results of the analysis of variance of the ranks of tests, within sites, included in ambient toxicity assessments of Oak Ridge National Laboratory streams using *Ceriodaphnia* (survival and fecundity) and fathead minnow larvae (survival and growth)^a

Test and endpoint	DF	R ²	F	Prob. >F
<i>Ceriodaphnia</i> survival	223	0.10	1.47	0.1168
<i>Ceriodaphnia</i> fecundity	223	0.45	11.29	0.0001
Fathead minnow survival	237	0.25	4.56	0.0001
Fathead minnow growth	223	0.68	29.34	0.0001

^aSite-date combinations excluded from the analysis (due to missing values in some cells) are described in text.

Note: R² = ratio of the variance.

Table 3.15. Test periods in which toxicity test endpoints were significantly higher or significantly lower (Duncan's multiple range test, $p < 0.05$) than average based on the analysis of variance of ranks of tests within sites^a

	<i>Ceriodaphnia</i> fecundity	Fathead minnow survival	Fathead minnow growth
Significantly higher	Test 10 —	Test 19 —	Test 2 Test 17
Significantly lower	Test 5 —	Test 12 Test 20	Test 13 —

^aResults are given for three toxicity test endpoints (*Ceriodaphnia* fecundity, and fathead minnow survival and growth). *Ceriodaphnia* survival was not included in the analysis because it did not differ significantly from test to test ($p > 0.1168$; see Table 3.14).

about 3.8 times better than the fathead minnow test. The sum of the variance ratios (site-to-site divided by test-to-test) for the survival and fecundity endpoints of the *Ceriodaphnia* test was 3.08, whereas that for survival and growth of the minnows was 0.81.

The analyses shown in Tables 3.12 and 3-14 were repeated after excluding those sites and tests that were found to contribute significantly low values (based on Duncan's tests; cf Tables 3.13 and 3.15) in their respective overall GLMs. The results of these analyses are shown in Table 3.16. This comparison showed that removing sites with low values affected the *Ceriodaphnia* survival data (R^2 declined from 0.25 to 0.10) but had little effect on the other three endpoints. Interestingly, removing tests with low values markedly improved the R^2 of both *Ceriodaphnia* endpoints (from 0.10 to 0.29 for survival and from 0.45 to 0.55 for fecundity) but decreased the R^2 for each endpoint of the minnow test (from 0.25 to 0.13 for survival and from 0.68 to 0.58 for growth). The analyses above suggest that ambient

toxicity assessments could benefit from more extensive use of *Ceriodaphnia*. This test seems to provide more information than the fathead minnow test, and both endpoints of the *Ceriodaphnia* test benefit (in terms of the amount of variance explained) from the elimination of poor-quality tests. Thus, greater testing frequency with *Ceriodaphnia* might be much better strategically than using both species at a lower testing frequency. An effort-allocation analysis of the *Ceriodaphnia* and fathead minnow tests also showed that the *Ceriodaphnia* test is more cost-effective than the fathead minnow test (Kszos and Stewart 1991); this feature may provide additional justification for greater use of *Ceriodaphnia* both in ambient and effluent assessments.

Based on the information presented in Table 3.13, well-defined reaches of two streams at ORNL show conclusive evidence of toxicity. A 1.3-km reach between WCK 5.1 and WCK 3.8 is clearly degraded—water at a midsection site in this reach (WCK 4.4) adversely affects two of the four test endpoints. The site on lower

Table 3.16. Site-to-site and test-to-test R^2 values for *Ceriodaphnia* and fathead minnow toxicity test endpoints for complete and reduced data sets^a

Test and endpoint	R^2 values			
	Site-to-site		Test-to-test	
	Complete set	Reduced set ^b	Complete set	Reduced set ^b
<i>Ceriodaphnia</i> survival	0.25	0.10	0.10	0.29
<i>Ceriodaphnia</i> fecundity	0.26	0.26	0.45	0.55
Fathead minnow survival	0.17	0.16	0.25	0.13
Fathead minnow growth	0.09	0.04	0.68	0.58

^aMethods used to (1) identify sites and tests that were subsequently deleted from the complete data sets and (2) compute the R^2 values for each endpoint are given in the text. R^2 = ratio of variance.

^bSignificantly low values removed (see text).

Fifth Creek (FFK 0.0), which enters WOC between WCK 3.8 and 4.4, is also problematic (Table 3.13). Sites on upper First Creek (FCK 0.9) and Melton Branch (MEK 1.4) just below the HFIR tributary appear to be "toxic" to fathead minnows, but the minnow test results are strongly biased due to abnormal variance patterns among replicates (cf Sect. 3.1.3).

3.1.7 Streams Near Solid Radioactive Waste Disposal/Storage Area 6

Two small streams near SWSA 6 (ET, which flows along the east boundary of the waste area, and West Tributary, which originates within the waste area) drain directly into WOL. Water samples from each of these streams were tested for chronic toxicity during March 2–9, 1989. These tests used *Ceriodaphnia*, which has been shown to be more sensitive than fathead minnow larvae to various effluents and toxicants (Sect. 3.1.6 and Loar 1993a, 1993b). The results of the tests showed that neither stream was acutely toxic, and *Ceriodaphnia* reared in full-strength water

from each stream had moderately high fecundity (Table 3.17). The mean CV among replicates for the four stream samples in these tests was 36%, which is somewhat higher than usual for ambient samples (e.g., 20–25%; see Table 3.3). Because of this variability, a strong statistical statement cannot be made about whether or not water from either stream showed evidence for chronic toxicity. It may be worth noting, however, that in an earlier test of these two streams (December 8–15, 1988; Loar 1993b), *Ceriodaphnia* neonates reared in water from West Tributary produced significantly fewer offspring than those reared in water from the ET. In the test reported here, mean fecundity of *Ceriodaphnia* in full-strength water from the West Tributary was somewhat lower than that of *Ceriodaphnia* in full-strength water from the ET, which supports the idea that water exiting SWSA 6 via West Tributary is of lower biological quality than that in ET. Single measurements of pH, alkalinity, and hardness for full-strength water from the two streams did not reveal important between-stream differences (Table 3.18).

Table 3.17. Results of 7-d static-renewal *Ceriodaphnia* toxicity tests of water from East Tributary and West Tributary (Solid Radioactive Waste Disposal/Storage Area 6 streams)^a

	Concentration (%)	Replicates	Females surviving 7 d	Mean <i>n</i> of offspring per female ± SD
Control	—	10	9	18.7 ± 3.9
West Tributary	100	10	10	11.8 ± 5.1
West Tributary	50	10	10	22.6 ± 5.8
East Tributary	100	10	10	15.8 ± 6.3
East Tributary	50	10	10	19.4 ± 6.8

^aThe tests were started on March 2, 1989.

^bSD = Standard deviation.

Table 3.18. Water quality factors measured for Solid Radioactive Waste Disposal/Storage Area 6 streams (East Tributary and West Tributary) from grab samples collected on March 2, 1989

Site	pH (standard units)	Conductivity (μS/cm)	Alkalinity (mg/L as CaCO ₃)	Hardness (mg/L as CaCO ₃)
Control	7.70	87	30.0	38
West Tributary	7.98	—	131.0	124
East Tributary	7.58	—	112.0	148

3.2 INSTREAM MONITORING OF THE PERIPHYTON COMMUNITY

(H. L. Boston, W. R. Hill, and C. M. Pettway)

Monitoring periphyton communities (attached algae, heterotrophic microbes, and fine detritus) at selected sites in the WOC watershed is an important component of the ORNL BMAP. Periphyton are an important component of many aquatic systems; they provide food for stream invertebrates (Minshall 1978) and herbivorous fishes (Power et al. 1985) and are an entry point for contaminants into the food chain [e.g., mercury and

cadmium are concentrated within the periphyton matrix (Huckabee and Blaylock 1973; Selby et al. 1985)]. Because algae and microbes turn over (reproduce) rapidly, periphyton are especially useful for detecting short-term environmental changes. Periphyton are capable of detecting infrequent pulses of toxicants that might be "invisible" when considering organisms with longer life spans.

Monitoring and related studies conducted during 1987 and 1988 (Loar 1993b) resulted in several conclusions about the periphyton communities in WOC and tributaries. Algal periphyton biomass and rates of

photosynthesis (inorganic carbon incorporation) were greater at sites downstream of ORNL discharges than at upstream reference sites due, most likely, to nutrient loading. The periphyton studies also demonstrated that photosynthesis can be used as an indicator of the growth or production of the algal component of the periphyton. However, photosynthesis is not linearly related to biomass, so comparisons among sites of the physiological condition of the periphyton are difficult. An analysis of covariance (ANCOVA) procedure was developed that determines chlorophyll-adjusted photosynthetic rates for the communities. This method made it possible to monitor the condition of the algal component of the periphyton at selected sites over time and to compare algal "health" among sites. Chlorophyll-adjusted photosynthetic rates, together with algal colonization rates and algal species composition, indicated that conditions improved with distance downstream from ORNL discharges. For example, the periphyton at WCK 2.3 and MEK 0.6 are in better condition than those at sites closer to discharge points.

Data from the periphyton monitoring program also demonstrated that differences in periphyton production among sites could not be explained by differences in water quality, based on short-term algal bioassays, and may be the result of intermittent exposure to toxicants, physical factors, or biotic interactions not examined to date. Although the concentrations of several potentially toxic metals (e.g., Cr, Cu, Hg, Pb, Zn) in the periphyton increased at sites downstream of ORNL discharges, none of these metals were present in concentrations that would be considered excessive or physiologically damaging. Although chlorophyll-adjusted photosynthesis was usually high at WCK 3.9, algal colonization at that site was extremely slow. Intermittent chlorine

toxicity and perhaps a persistent low level of toxicity may restrict the algal community at WCK 3.9 to one species that is slow to colonize new surfaces. The inability of this alga to recolonize rock surfaces may explain the patchy distribution of periphyton on the rocks at this site. Results to date suggest that adverse impacts of ORNL discharges on algal periphyton may be restricted to the areas near WCK 3.9, WCK 3.4, and occasionally WCK 2.9. Impacts of toxic releases on heterotrophic microbes may contribute to the slow colonization at WCK 3.9 and the overall below-expected performance of the algae at WCK 3.4. However, the microbial component has not been investigated sufficiently to adequately test this hypothesis. Finally, previous studies showed that the algal component of the periphyton at MEK 0.6 responded favorably (increased biomass and photosynthesis) after the HFIR was shut down in November 1986. In general, the periphyton data corroborated results of other community studies (Sects. 6.1 and 6.2) and provided a data base for future use in evaluating stream conditions and remedial action alternatives.

3.2.1 Methods

Small rocks with their associated periphyton were collected monthly from nine sites in the WOC watershed (Fig. 3.3). Five sites receive drainage or are downstream of effluent inputs from ORNL facilities (WCK 3.9, WCK 3.4, WCK 2.9, WCK 2.3, and MEK 0.6). Four sites upstream of ORNL operations (WCK 6.8, MEK 1.8, upper First Creek at FCK 1.0 and upper Fifth Creek at FFK 1.1) were used as reference sites. Based on sampling conducted in 1987 and 1988 (Loar 1993b), reference sites on upper Northwest Tributary and on Ish

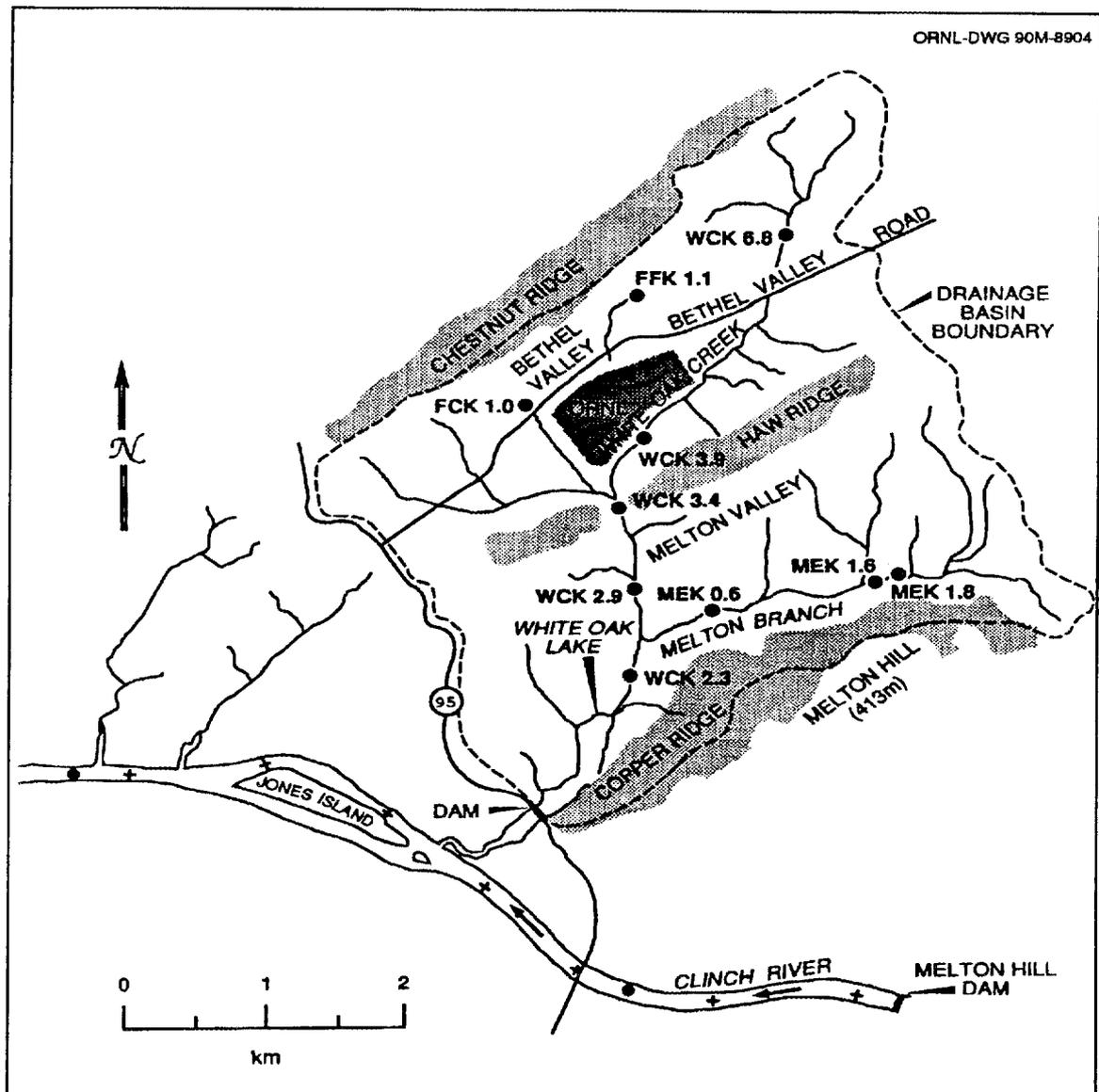


Fig. 3.3. Periphyton monitoring sites in the White Oak Creek watershed in 1989.

Creek were eliminated because the data were redundant with those collected at the other upstream sites.

3.2.1.1 Periphyton chlorophyll and carbon incorporation

To determine algal biomass and production, four small (10–60 cm²),

relatively flat rocks were collected monthly from shallow (<25-cm deep) riffle areas at each site. The rocks were taken to the laboratory in water from the collection site. In the laboratory, rocks from each site were incubated for 2 h in water from the collection site and amended with 10 μCi NaH¹⁴CO₃. During the incubation, the water temperature was maintained within 2° C of ambient stream temperature.

Approximately $400 \mu\text{mol}\cdot\text{m}^{-2}\cdot\text{s}^{-1}$ PAR (photosynthetically active radiation; ~16% full sun) was provided by a 1000-W metal halide lamp, and the water in the incubation chambers was circulated by submersible pumps to simulate natural flow. After incubation, the rocks were rinsed twice in distilled water to remove residual inorganic ^{14}C . They were then placed in 30 mL of dimethylsulfoxide (DMSO) and kept in darkness for 24 h to extract chlorophyll and other soluble organic compounds (Shoaf and Lium 1976; Palumbo et al. 1987). Five mL of each DMSO extract was diluted 1:1 with 90% acetone, and the chlorophyll *a* (Chl *a*) content was determined spectrophotometrically using the equations of Jeffrey and Humphrey (1975). Corrections were made for phaeopigments (Strickland and Parsons 1972). A 500- μL aliquot of the extract was added to 10 mL of Aquasol (scintillation cocktail), and the ^{14}C in the aliquot was determined by liquid scintillation spectrometry.

The surface area of each rock was determined by covering the upper surface with aluminum foil, weighing the foil, and converting to surface area based on a known weight per unit area of foil. This procedure was repeated twice for each rock. Chlorophyll *a* and the rate of carbon incorporation were then expressed per unit surface area.

3.2.1.2 Chlorine toxicity bioassay

To determine whether chlorine was responsible for the low algal taxonomic diversity and patchy distribution of periphyton biomass at WCK 3.9, a 23-d bioassay was conducted using natural periphyton communities. Small rocks with their associated periphyton communities were collected from riffles in three areas of EFPC: one site at EFPC kilometer (EFK) 24.4 within the Y-12 Plant

boundary with occasionally high chlorine concentrations, and two sites further downstream (EFK 23.2 and EFK 13.8) that are rarely, if ever, exposed to measurable concentrations of chlorine. Upstream reaches of EFPC (e.g., EFK 24.4) that receive cooling water from the Y-12 Plant have TRC concentrations similar to those at WCK 3.9 and have the same type of periphyton community as that found at WCK 3.9. Periphyton on replicate substrata from each site were exposed to 0, 0.05, 0.1, 0.25, 0.5 or 1.0 mg/L TRC in continuous-flow outdoor chambers. Concentrations of TRC were controlled by mixing tap water (containing about 1 mg/L TRC) with dechlorinated tap water. On days 0, 7, and 23, periphyton samples were collected from each treatment and algal species composition was determined by light microscopy. On day 23, algal biomass (Chl *a*) and photosynthesis (^{14}C incorporation) were measured for periphyton on four rocks from each treatment for each site; analytical procedures have been described previously.

3.2.2 Results and Discussion

3.2.2.1 Algal assemblages

The distinct differences in the taxonomic structure of algal assemblages between reference and impacted sites noted previously (Loar 1993b) persisted during 1989. Algal assemblages at the reference sites were dominated by diatoms and prostrate *Stigeoclonium* cells and filaments. The relative abundance of diatoms on substrata in WOC also increased with increasing distance downstream from ORNL. A peculiar and characteristic unicellular nonmotile green alga continued to dominate the periphyton at WCK 3.9 (see discussion of chlorine toxicity studies below). This alga persisted in low numbers downstream at WCK 3.4,

where the community was dominated by *Stigeoclonium* and diatoms.

Algal assemblages at the upstream reference sites were typically dominated by prostrate filaments of *Stigeoclonium*. The presence of this alga in this growth form is characteristic of habitats where rates of nutrient supply are low and grazing pressure is high. Periphytic algal assemblages at FFK 1.1 consisted primarily of stalked or filamentous diatoms that were not closely appressed to the substrate. Their presence at this site may be related to the absence of *Elimia* (an algivorous snail).

3.2.2.2 Monthly determination of algal biomass and photosynthesis

Periphyton Chl *a* per unit rock surface area was used as a measure of algal biomass at each monitoring site. The distribution of periphyton at WCK 3.9 was very patchy; it was not uncommon to see single rocks with a thick periphyton cover, while surrounding rocks were completely bare. At WCK 3.9, we selected rocks from among those with obvious cover (Loar 1993b) to avoid excessive variance and the analysis of many bare rocks. Although such data obviously overestimate periphyton biomass (chlorophyll) at the site, they were useful in evaluating the condition of the periphyton community, as discussed later. The upstream site on Melton Branch (MEK 1.8) was moved about 0.2 km downstream to MEK 1.6 just above the confluence with the unnamed tributary that receives effluent from the HFIR/REDC facilities. This relocation was necessary because of the frequent periods of no flow that impacted the periphyton community at MEK 1.8. The new location provided a reference site (not impacted by ORNL activities) where periphyton could be collected for much of the year.

3.2.2.3 Monthly data for Chl *a* and photosynthesis

Three years of monthly data on periphyton Chl *a* at an upstream (WCK 6.8) and a downstream site (WCK 2.3) are shown in Fig. 3.4. Site WCK 6.8 is shaded during the summer and has low concentrations of dissolved nutrients. Site WCK 2.3 is also shaded by riparian vegetation; however, nutrient concentrations at this site are high as a result of discharges from ORNL. Biomass was seasonally variable at both sites, reflecting changing light conditions, stochastically occurring floods, and other factors. Floods generally had a greater impact on biomass and photosynthesis at the downstream sites than at the upstream sites, based on observed changes in biomass following storms. At downstream sites with high biomass, the periphyton are not tightly attached and are more readily removed. Furthermore, photosynthesis decreases disproportionately by scouring because the more productive components of the communities (i.e., the overstory) are typically lost first. Month-to-month variability in biomass at WCK 6.8 and WCK 2.3 were fairly typical of other upstream and downstream sites respectively.

Data on Chl *a* for MEK 0.6 from 1986 through 1989 (Fig. 3.5) show the changes in biomass that occurred between shutdown of HFIR (November 1986) and the present (December 1989). Monthly data for upper Melton Branch [MEK 1.8 (1987–88) and MEK 1.6 (1989)] show algal biomass upstream of HFIR (Fig. 3.5). Nutrient concentrations and biotic conditions at MEK 1.6 and MEK 1.8 are generally similar to those found at other reference sites (e.g., WCK 6.8; see also Sect. 2.2.4).

The monthly data also showed that periphyton photosynthesis at WCK 6.8

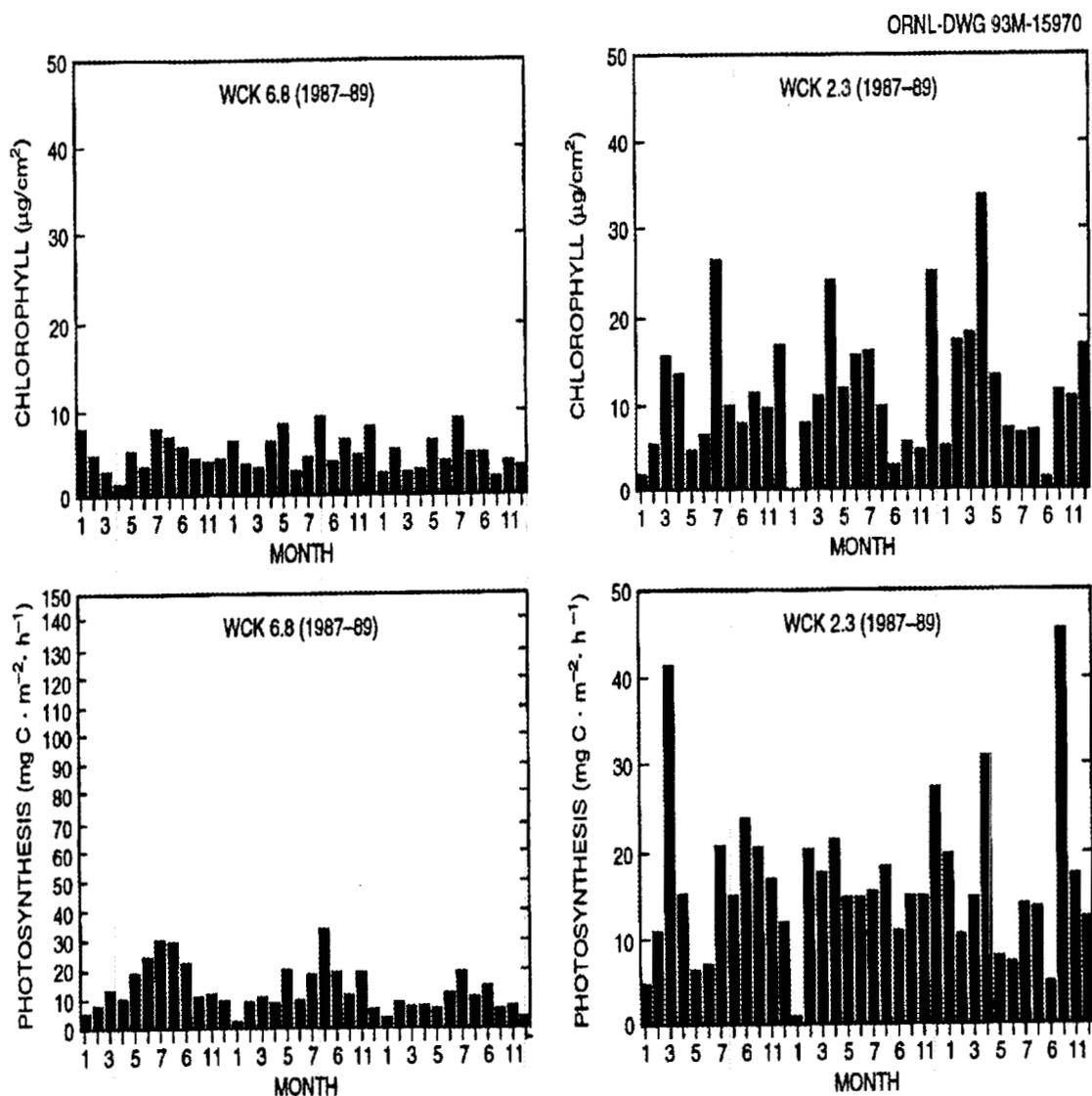


Fig. 3.4. Periphyton chlorophyll *a* and photosynthesis on small flat rocks collected from shallow riffle areas in White Oak Creek above and below Oak Ridge National Laboratory [White Oak Creek kilometer (WCK) 6.8 and WCK 2.3 respectively], January 1987–December 1989. Monthly values are means of four replicates.

tended to be higher during May through September than during the remainder of the year (Fig. 3.4). A similar pattern was apparent at several other sites. This seasonal increase in photosynthesis may have reflected seasonal increases in water temperature, since light decreased due to

riparian cover and water column nutrient concentrations did not increase during this period. Photosynthesis per unit area was more variable at WCK 2.3 than at the smaller upstream reference sites. This variability was due, in part, to larger month-to-month changes in biomass at

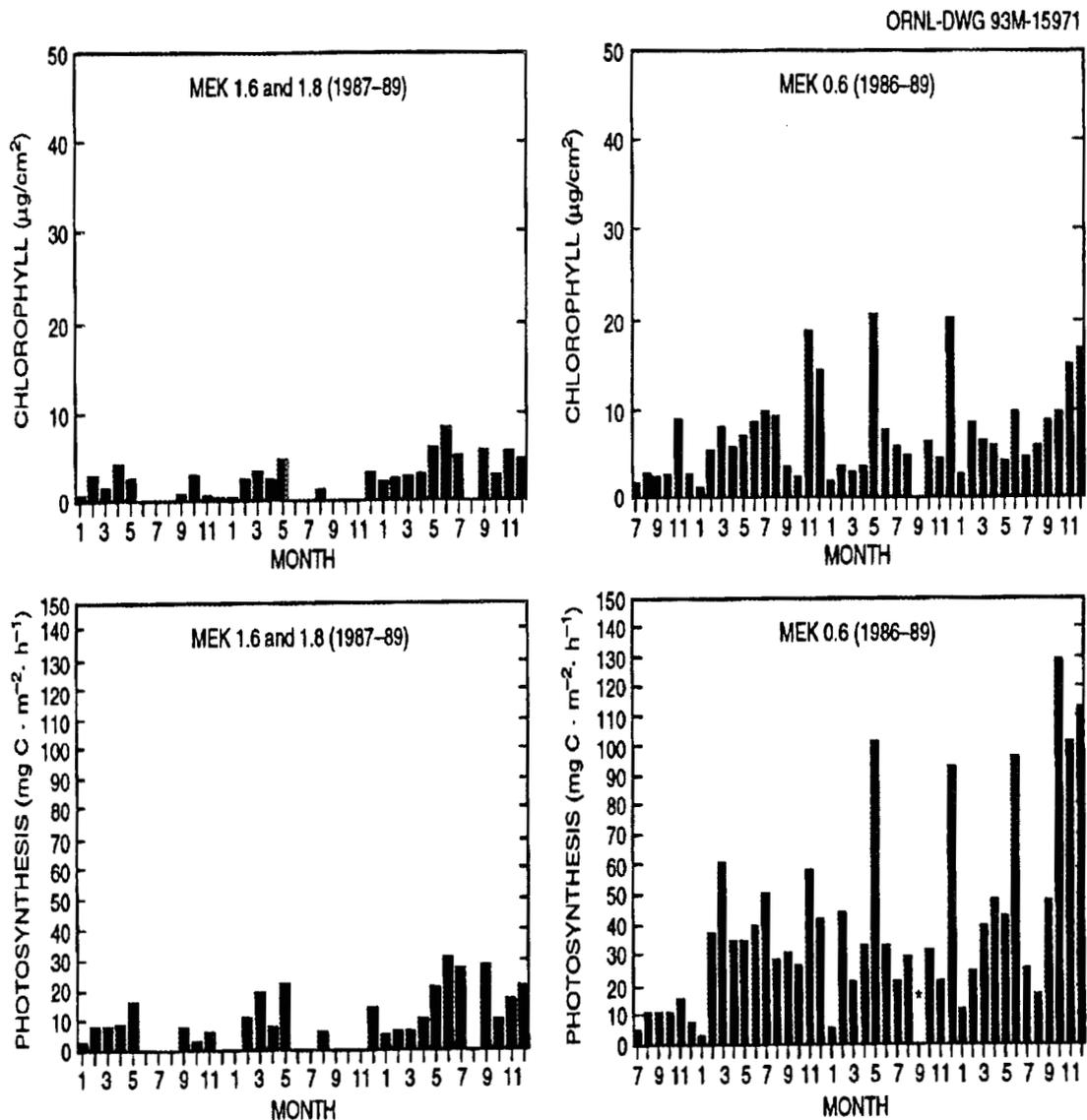


Fig. 3.5. Periphyton chlorophyll *a* and photosynthesis on small flat rocks collected from shallow riffle areas in upper Melton Branch at Melton Branch kilometer (MEK) 1.8 (1987–1988) and MEK 1.6 (1989) and in lower Melton Branch at Melton Branch kilometer (MEK) 0.6 (1986–1989). Monthly values are means of four replicates.

WCK 2.3 (Fig. 3.4). Periphyton photosynthesis at upstream sites tended to show a seasonal pattern similar to that observed at WCK 6.8, while downstream sites had no apparent seasonality.

During 1989, periphyton photosynthesis at MEK 1.6 was somewhat

higher than that observed during previous years at MEK 1.8 (Fig. 3.5). This trend may be related to the increased annual discharge during 1989 (i.e., fewer dry periods compared with 1987 and 1988; see Table 2.1). Monthly periphyton photosynthesis at MEK 0.6 (Fig. 3.5)

remained high in 1989 relative to the period when the HFIR was operating (before November 1986).

Monthly data on periphyton chl *a* and photosynthesis in 1989 are presented in Appendix F.

3.2.2.4 Annual average Chl *a* and photosynthesis

The annual average values for periphyton chl *a* during 1989 were generally similar to or lower than those for 1988 (Table 3.19). However, periphyton biomass at MEK 1.6 during 1989 was slightly greater than the biomass at MEK 1.8 during 1988. The lower biomass may have resulted from more frequent storms during 1989 (Fig. 2.3). The greatest algal biomass was found at sites with high nutrient concentrations (due to ORNL discharges) and relatively high light levels (e.g., WCK 3.4; see light data in Loar 1993b and earlier comments about WCK 3.9). Reference sites where grazing snails were abundant (WCK 6.8, Table 3.19 FCK 1.0) had the lowest algal biomass. Low biomass at MEK 1.6 (and previously at MEK 1.8) may have been related to the presence of other periphyton grazers, low-flow periods, or other factors.

Annual average rates of algal periphyton photosynthesis for 1989 generally followed algal biomass (Table 3.19) and, at most sites, tended to be lower than the rates observed in 1988. Annual average photosynthesis was greater at MEK 0.6 and MEK 1.6 during 1989 than during the previous year. Annual average rates of photosynthesis per unit rock surface area were greatest at the two sites furthest downstream (WCK 2.3 and MEK 0.6). Photosynthesis per rock area for algae at WCK 3.9 were also high but not representative of average conditions at that site because only rocks with visible

periphyton cover were collected, as discussed earlier. Annual photosynthesis was lowest at those reference sites with low biomass due to low nutrients and grazing by snails (WCK 6.8 and FCK 1.0).

Annual periphyton photosynthesis at reference site FFK 1.1 was similar to that at several downstream sites. As discussed in Loar (1993b), higher-than-expected rates of photosynthesis at FFK 1.1 may be related to groundwater inputs providing higher carbon dioxide and nutrients, and/or to the absence of snails. Snails have been shown to decrease both algal biomass and photosynthesis per unit biomass by selecting for grazing-persistent growth forms (e.g., prostrate *Stigeoclonium*) at low-nutrient sites (W. R. Hill, ORNL/Environmental Sciences Division, unpublished data).

3.2.2.5 Comparing photosynthesis and periphyton condition

Expressing photosynthesis per unit algal biomass is one method of comparing the rates of photosynthesis among sites with different periphyton biomass. However, this comparison is complicated because chlorophyll (biomass) and the rate of carbon incorporation are positively, but not linearly, related. Self-shading and other factors cause carbon incorporation to increase more slowly than biomass as biomass accumulates and the periphyton matrix thickens; thus, photosynthesis per unit biomass decreases as additional cells are added to the periphyton matrix. Because the relationship between chlorophyll (biomass) and photosynthesis is not linear, photosynthetic rates at sites that differ substantially in periphyton biomass are difficult to compare. To parcel out the effect of biomass on biomass-specific photosynthesis, an ANCOVA of photosynthesis was conducted at the different sites and months, using

Table 3.19. Annual mean chlorophyll *a* (in micrograms per square centimeter) and photosynthesis (in milligrams of carbon per square meter per hour) for periphyton at nine monitoring sites during 1988 and 1989 (*n* = 48)

	Site								
	WCK ^a	FCK ^b	FFK ^c	MEK ^d	WCK	WCK	WCK	WCK	MEK
	6.8	1.0	1.1	1.8	3.9	3.4	2.9	2.4	0.6
Chlorophyll <i>a</i>, $\mu\text{g}/\text{cm}^2$									
1988 Mean	5.4	6.2	14.3	2.8	32.8	20.0	11.7	12.2	7.2
1989 ^e Mean	4.3	4.6	11.6	4.3	21.5	17.5	8.7	12.7	8.3
SE ^f	± 0.5	0.3	0.8	0.3	1.9	1.5	1.1	1.3	0.7
Tukey's grouping	C	C	B	C	A	A	B	B	B
Photosynthesis, $\text{mg C}\cdot\text{m}^{-2}\cdot\text{h}^{-1}$									
1988 Mean	13.7	27.2	51.7	11.6	63.3	43.4	42.2	46.5	39.8
1989 Mean	8.4	13.6	39.1	15.8	47.1	42.8	32.5	48.6	60.6
SE	± 0.7	0.7	3.0	1.6	4.7	4.1	4.3	5.7	5.9
Tukey's grouping	E	D	B	D	B	B	C	B	A

^aWCK = White Oak Creek kilometer.

^bFCK = First Creek kilometer.

^cFFK = Fifth Creek kilometer.

^dMEK = Melton Branch kilometer.

^eData for 1989 include a comparison of means where values with the same letters are not significantly different from each other based on Tukey multiple comparison procedure (*p* > 0.05).

^fSE = standard error.

chlorophyll as the covariate and log-transformed (ln) data (Loar 1993b).

The mean annual values for periphyton photosynthesis in 1989, when adjusted to equivalent densities of Chl *a* (LSMEANS in SAS PROC GLM), were highest for communities at MEK 0.6 and WCK 2.3, sites farthest downstream from effluent discharges (Table 3.20). Biomass-adjusted photosynthetic rates were lowest at reference site WCK 6.8 and

intermediate at other downstream sites and at reference site MEK 1.6. This comparison showed that the periphyton in upper Fifth Creek (FFK 1.1) had high biomass-specific photosynthetic rates (suggesting good physiological condition), as this site ranked higher than most downstream sites where nutrients were abundant.

During 1989, mean annual biomass-adjusted photosynthetic rates at

Table 3.20. Chlorophyll-adjusted periphyton photosynthesis at the nine study sites in 1989

Site ^a	Chlorophyll-adjusted mean (ln $\mu\text{g C}\cdot\text{cm}^{-2}\cdot\text{h}^{-1}$)	Multiple ^b comparison
WCK 6.8	0.05	A
FCK 1.0	0.52	B
FFK 1.1	0.93	D
MEK 1.5	0.56	B C
WCK 3.9	0.72	C
WCK 3.4	0.70	C
WCK 2.9	0.69	C
WCK 2.4	1.08	E
MEK 0.6	1.44	

^aWCK = White Oak Creek kilometer, FCK = First Creek kilometer, FFK = Fifth Creek kilometer, and MEK = Melton Branch kilometer.

^bSites with the same letter are not significantly different based on multiple comparison by SAD GLM PDIFF procedure ($p > 0.05$). Data for January and August were not available for all sites and were excluded in this analysis.

the periphyton monitoring sites were similar (with respect to magnitude and ranking) to those found during 1988 (Loar 1993b). Mean annual values for WCK 2.3 and MEK 0.6 ranked among the highest in 1988, but were not significantly different from each other or from the rate for periphyton at WCK 2.9. During 1989, rates for the periphyton communities at these sites differed significantly from the rates for other communities and from each other (Table 3.20). The adjusted annual value for the biomass-specific photosynthetic rate at WCK 6.8 in 1989 was significantly lower than the rate observed in 1988. Natural environmental variability (e.g., frequency of high discharge events) was probably responsible for this change.

The relatively low biomass-adjusted photosynthetic rates for periphyton at some reference sites have been shown experimentally to be related to low nutrient availability and intense grazing by

snails (W. R. Hill, ORNL/ESD, unpublished data). The relatively high biomass-adjusted photosynthetic rates for periphyton at FFK 1.1 show the response to reduced grazing pressure and, perhaps, to slightly greater concentrations of nutrients in the water (see Sect. 2.2.4). The high biomass-adjusted photosynthetic rates at MEK 0.6 and WCK 2.3 indicate that the periphyton at these sites are in relatively good condition. Both MEK 0.6 and WCK 2.3 are heavily shaded; thus, high rates would not be expected because high chlorophyll per unit biomass and low photosynthesis per unit biomass are common responses of plants to low-light habitats. High biomass-adjusted photosynthetic rates would be expected at sites downstream of ORNL discharges but upstream of WCK 2.3 (i.e., WCK 3.9, WCK 3.4, and WCK 2.9), where higher light levels prevail and grazing pressure is similar to or lower than that at WCK 2.3. The lower biomass-adjusted photosynthesis

observed at those sites suggests that the algal periphyton are stressed. Data for other BMAP components (e.g., invertebrates and ambient toxicity) support the conclusion that conditions are reasonably good at WCK 2.3 and MEK 0.6, whereas stress is indicated at those sites closer to ORNL discharges (e.g., WCK 3.9, WCK 3.4, and WCK 2.9).

In a previous report (Loar 1993b), it was shown that monthly mean biomass-adjusted photosynthetic rates could be used to monitor sites through time and to make comparisons among sites. Monthly means of biomass-adjusted photosynthesis are shown for several sites in Fig. 3.6. Biomass-adjusted photosynthesis at upstream sites MEK 1.6 (symbol I) and WCK 6.8 (symbol A) were higher during summer (June–September) than during winter. As noted earlier, this pattern was probably due to the effects of temperature. A similar seasonal pattern was evident at WCK 3.9. However, the better physiological condition of periphyton at this site during summer likely reflected decreased stress, because algal biomass was greater and chironomids were present during the summer.

Because WCK 2.3 (symbol E) is downstream of the confluence of Melton Branch and WOC, conditions at that site should reflect the influence of water quality in both streams. When the biomass-adjusted photosynthetic rate at WCK 2.3 is compared with that for communities farther upstream in WOC (WCK 3.4, symbol C) and in Melton Branch (MEK 0.6, symbol F) for each month, the rate at WCK 2.3 was generally intermediate between, or similar to, that of algae at WCK 3.4 and MEK 0.6. The mean annual biomass-adjusted photosynthetic rate at WCK 2.3 was also intermediate between that for MEK 0.6 and WOC sites upstream of WCK 2.3 (Table 3.20). Because biomass-adjusted photosynthetic rates at MEK 0.6 generally

equalled or exceeded those at WCK 2.3, it is difficult to determine if the improved condition of WCK 2.3 (relative to WCK 3.4 or WCK 2.9; not shown) reflects recovery from toxic stress due to ORNL discharges or the positive influence of water from Melton Branch. In the future, a more rigorous analysis of this type of information may be beneficial in evaluating conditions in the lower reaches of WOC and Melton Branch.

3.2.2.6 Chlorine toxicity at WCK 3.9

White Oak Creek receives once-through cooling water and cooling tower blowdown from ORNL at several locations. The area adjacent to the High Temperature Materials Laboratory (near WCK 4.1) is heavily impacted, with few fish, macroinvertebrates, or periphyton (H. L. Boston, ORNL/ESD, personal observation). The periphyton community at WCK 3.9 is also clearly stressed; the algal community there consists almost exclusively of a small unicellular nonmotile green alga, and the periphyton cover on the substratum is patchy. Previous investigations showed that, although algal growth is rapid at WCK 3.9, colonization is slow on new substrata or substrata from which the periphyton was completely removed by acute exposure to chlorine. Also, the periphyton are more commonly visible on large compared with smaller rocks. It is speculated that large rocks probably provide more opportunities for refugia during episodes of acute chlorine toxicity. Consequently, they are more likely to have some remaining periphyton and are rapidly recolonized following such incidents. The rapid growth rate but poor colonizing ability of the periphyton at this site is consistent with this hypothesis.

Results of the chlorine exposure studies showed that by day 23 the algal communities exposed to TRC at

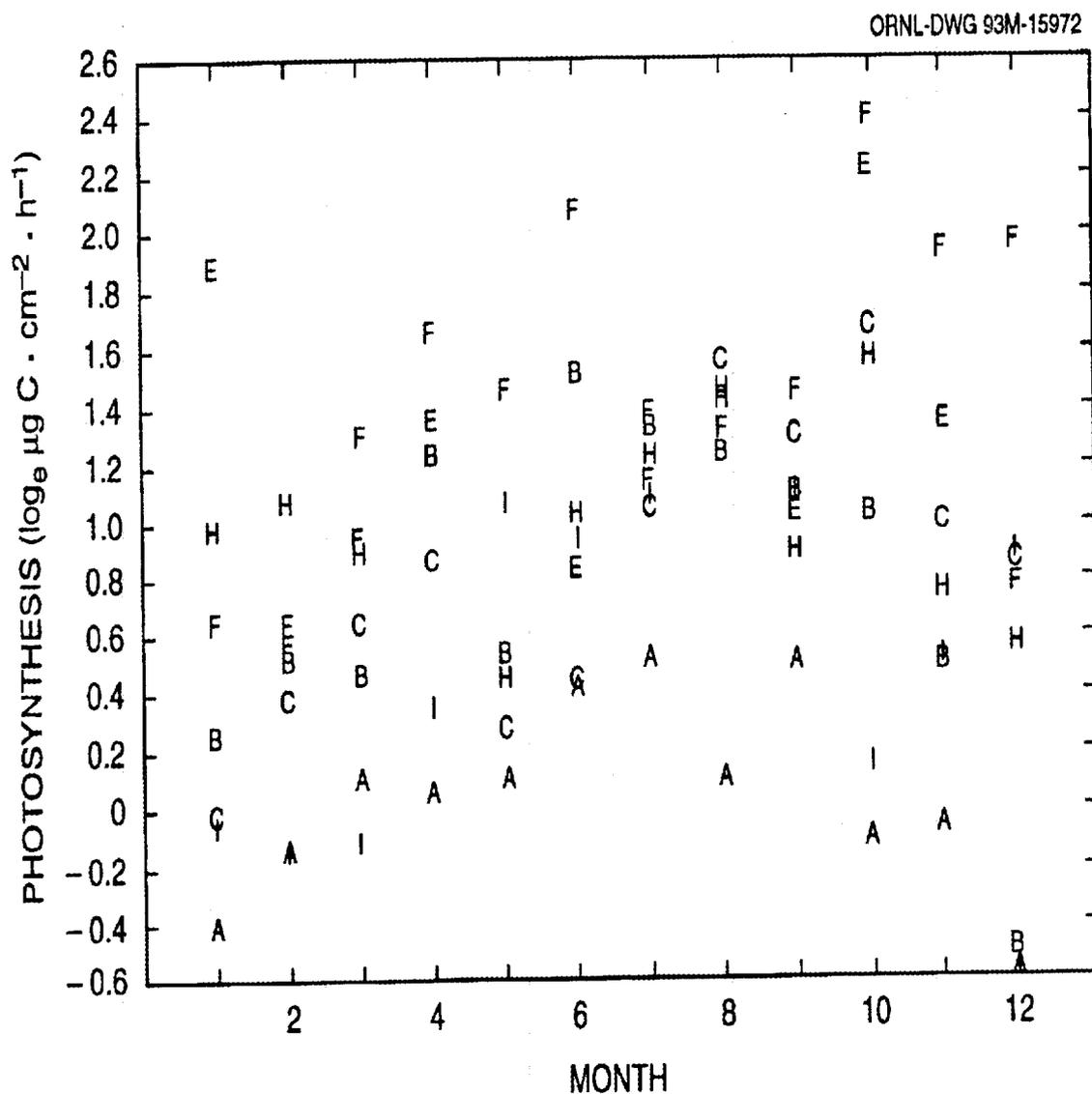


Fig. 3.6. Monthly least square means for biomass-adjusted photosynthesis, based on an analysis of covariance for data collected during 1989. Note: A = White Oak Creek kilometer (WCK) 6.8, B = WCK 3.9, C = WCK 3.4, E = WCK 2.3, F = Melton Branch kilometer (MEK) 0.6, H = Fifth Creek kilometer (FFK) 1.1, and I = MEK 1.6.

concentrations of 0.25–1.0 mg/L were dominated by the small unicellular green alga that typically dominates the periphyton at WCK 3.9 and EFK 24.4 (sites where chlorine stress is suspected). That alga was not initially found in significant numbers on substrata collected

from the two downstream sites. At lower chlorine concentrations, green filamentous algae and *Scenedesmus* dominated the communities on substrata collected from all sites. These results indicate that high TRC concentrations select for the alga that dominates the periphyton at WCK 3.9, but

at low chlorine concentrations, this taxon is replaced by species typical of sites that are not exposed to high TRC concentrations.

Data for the periphyton from EFK 24.4 (Fig. 3.7) were typical of those for periphyton from the other two EFPC sites (EFK 23.2 and EFK 13.8). Algal biomass on rocks from all three sites was substantially reduced relative to the dechlorinated control by TRC concentrations ≥ 0.25 mg/L. At the highest TRC concentration tested (1.0 mg/L), biomass was 5- to 10-fold lower than that of the control (C. M. Pettway, Knoxville College, unpublished data).

Even at the lowest TRC concentration tested (0.05 mg/L), photosynthesis per unit area for periphyton from all sites was significantly lower than that for the control. Photosynthesis for the periphyton from EFK 24.4 had decreased by about a factor of three before biomass was significantly decreased (Fig. 3.7). These results show that photosynthesis was much more sensitive than biomass (and perhaps species composition) as an indicator of stress on periphyton communities. This finding is consistent with the results of the previous analyses of biomass-adjusted photosynthesis. Thus, photosynthesis is a useful parameter for characterizing and monitoring periphyton communities. The results of this study also suggest that the EPA limit for chronic exposure to chlorine of 0.011 mg/L TRC (EPA 1985) is probably reasonable, because a significant decrease in photosynthesis was observed after only 23 d at 0.05 mg/L TRC.

This study has also provided strong evidence that chlorine toxicity could be responsible for the altered algal community composition at WCK 3.9 and EFK 24.4, where high concentrations of TRC (>0.25 mg/L) are common. The patchy distribution of periphytic algae and the relatively low biomass-adjusted

photosynthetic rates appear to be due to the selection for the chlorine-tolerant alga found at chlorine-stressed sites and to the adverse effects of TRC on the photosynthetic rate of that alga.

3.2.3 Summary

Data collected during 1989 added to the data set on the periphyton communities in the WOC system and corroborated findings of other components of the BMAP. The relocation of MEK 1.8 to MEK 1.6 provided a better upstream data set for that stream. Several conclusions can be drawn based on the 1989 data. First, biomass and photosynthesis of the algal periphyton at the monitoring sites in the WOC system were similar to those measured during 1988. Second, the physiological condition of the algal periphyton at most sites was similar in 1988 and 1989. Periphyton at reference site WCK 6.8 were in relatively poor condition during 1989 due, most likely, to natural factors. As in 1988, periphyton at sites just below ORNL discharges were in relatively poor condition compared with the periphyton at sites farther downstream. The periphyton at MEK 0.6 were in especially good condition during 1989, despite the resumption of some low-level testing at the HFIR/REDC facility in late 1989. Third, a chlorine toxicity bioassay showed that concentrations of TRC ≥ 0.25 mg/L select for the nonmotile green alga present at WCK 3.9 and at other sites where high concentrations of TRC occur. This study also showed that chlorine stress affected algal photosynthesis more than algal biomass. Thus, stress from TRC could explain, or contribute to, the low biomass-adjusted photosynthetic rates for periphyton at WCK 3.9 and other sites near ORNL discharges.

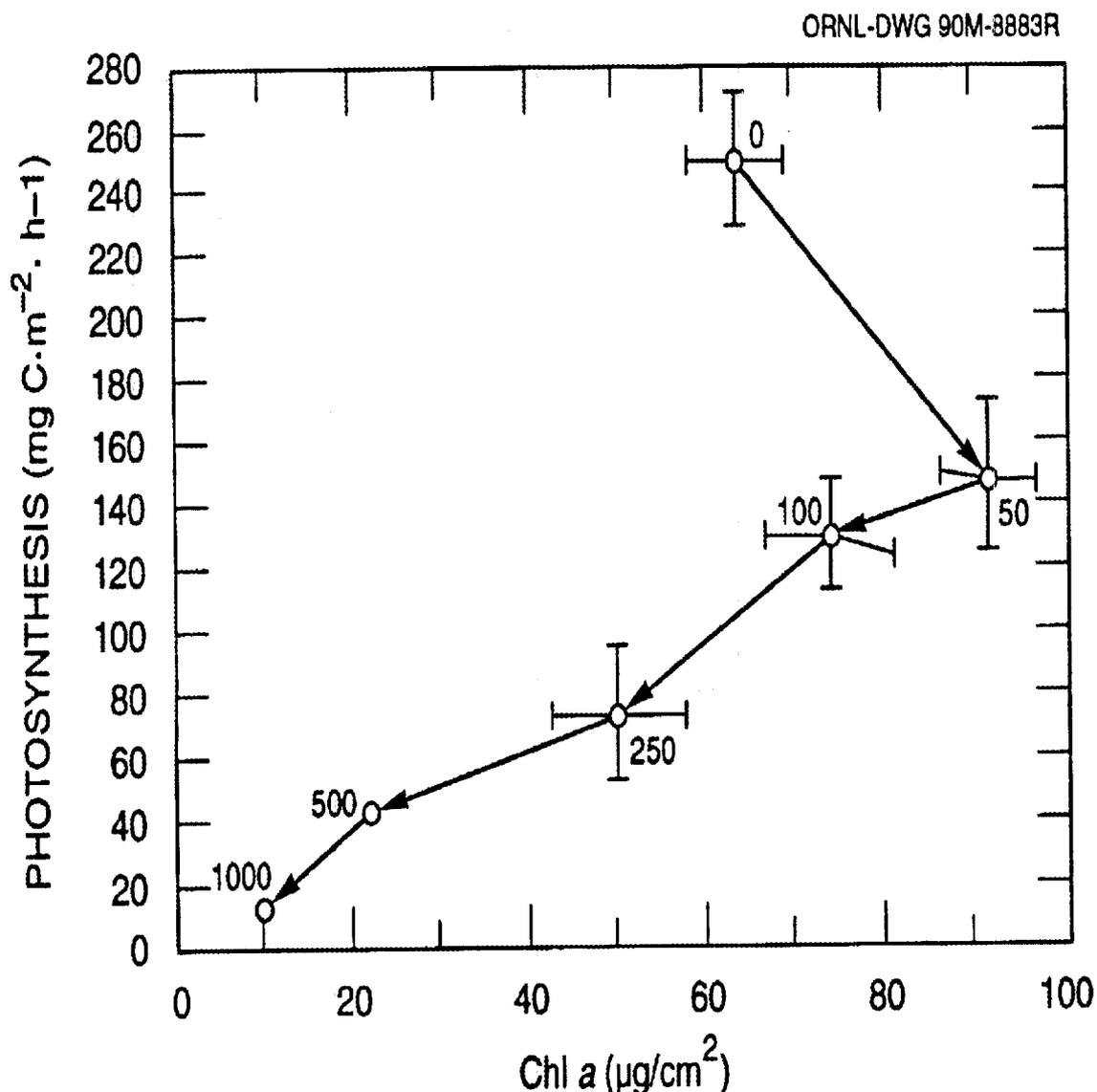


Fig. 3.7. Algal periphyton biomass and photosynthesis on substrata collected from upper East Fork Poplar Creek at kilometer 24.4 and held for 23 d at 0 to 100% tap water. The total residual chlorine concentration in 100% tap water was about 1.0 mg/L. Values are means and one standard error for four replicates.

3.3 FUTURE STUDIES

During 1990-91, the frequency of toxicity testing of the ambient sites will be increased to develop a 12-month data set similar to that obtained during the first year of the study. These data will permit a direct comparison between the 1986-87

and 1990-91 periods. This strategy should effectively identify changes in stream water quality attributable to changes in plant and waste treatment operations (e.g., operation of the NRWTF, changes in operation of the HFIR) that were implemented during the 5-year period defined by the current NPDES permit.

Because the ambient toxicity pattern identified during the past several years is strongly linked to chlorine, ancillary studies will be designed to address concerns specific to this factor. For example, the relationships between TRC loss and water temperature and between TRC decay and the presence of periphyton will be determined through laboratory experiments. These data will facilitate the development of a simple model predicting instream toxicity as a function of TRC inputs.

As the GPP for consolidating and dechlorinating once-through cooling water continues, it will be important to identify problems that presently may be "camouflaged" by toxic levels of TRC. Studies, such as those conducted to evaluate wastewaters entering WOC from Outfall 312 (see Sect. 3.1.4), will continue on an ad hoc basis.

Efforts to date have emphasized the algal component of the periphyton. Heterotrophic microbes process organic matter, recycle nutrients, and are also important components of stream systems. During 1990, studies will be initiated to characterize the microbial component of the periphyton at the monitoring sites in terms of (1) relative biomass, (2) metabolic activity, and (3) sources of organic carbon used for growth. This information will be used to (1) evaluate the condition of the microbial community in relation to ORNL discharges, (2) determine how microbial activity influences periphyton characteristics (e.g., algal colonization and photosynthesis), and (3) determine how different sources of organic carbon [autogenic (from periphytic algae) and allochthonous (from terrestrial litter and ORNL discharges)] contribute to energy flow in WOC. These studies are a logical extension of the work completed to date.

4. BIOACCUMULATION STUDIES

G. R. Southworth and M. J. Peterson

The bioaccumulation monitoring task of the ORNL BMAP (Loar et al. 1991) has five subtasks with the following objectives:

1. to determine what materials present in the WOC system accumulate to unacceptable levels in aquatic biota (Subtask 2a),
2. to identify specific sources of any observed contamination (Subtask 2b),
3. to calibrate water quality monitoring data against contaminant levels in biota (Subtask 2c),
4. to determine the source and scope of PCB contamination of channel catfish in the WOC embayment and the Clinch River (Subtask 2d), and
5. to develop the capability to forecast future levels of biotic contamination under various remedial action alternatives (Subtask 2e).

Results of studies relating to Subtasks 2a, 2b, and 2c are reported in Sect. 4.1 of this report; results relating to Subtasks 2b and 2d are reported in Sect. 4.2, and Subtask 2c is reported in Sect. 4.3. Data from all five subtasks will be applicable to Subtask 2e.

4.1 IDENTIFICATION OF CONTAMINANTS THAT ACCUMULATE IN AQUATIC BIOTA

4.1.1 Introduction

Results reported in previous annual reports (Loar et al. 1992; Loar 1993a, 1993b, Sect. 4) indicated that the ORNL

complex was a source of mercury and PCB contamination to fish in WOC. No other organic contaminants, including polycyclic aromatic hydrocarbons (PAHs), pesticides, and acid/base neutral semivolatiles, were detected in fish from WOC. Despite evidence of past (Loar 1993a, Sect. 4.1.1) and current discharges of copper and zinc (Loar 1993a, Table 3.17; Table 2.5, this report) to WOC, concentrations of metals (other than mercury) in fish from WOC, WOL, and Melton Branch were similar to concentrations in fish from an uncontaminated reference stream.

The contaminant monitoring program initiated in 1986 was continued in 1988-89 with little change. Based on the conclusions from an integration of water quality and bioaccumulation data on metals (Loar 1993a, Sect. 4.3), metal analyses (except mercury) were reduced to three sites that correspond with water quality monitoring stations. Because of the presence of PCBs at significant concentrations in sunfish in WOC, twice yearly PCB monitoring of sunfish was continued in 1989 at two sites (WCK 2.9 and WOL) to provide a better baseline for evaluating future changes in PCB body burdens. Efforts to use caged clams to locate sources of contamination and to detect the presence of hydrophobic organics that are readily metabolized (and therefore not accumulated) by fish were continued.

4.1.2 Methods

Fish were collected by electrofishing at three sites on WOC and single sites on WOL, the WOC embayment, NT, and

Melton Branch in winter 1988–89. Sites on WOC, Melton Branch, and NT corresponded closely to the benthic macroinvertebrate and fish community sampling sites (Fig. 2.2). Twelve fish were collected at each site to provide samples for analysis of metals, organics, and archival storage. Samples were taken from eight fish for analysis of mercury and PCBs, six fish for analysis of metals, and four fish for analysis of organics (except at MEK 0.2, where fish numbers were inadequate). Bluegill (*Lepomis macrochirus*) were collected at all sites; however, redbreast sunfish (*L. auritus*) were also collected at WCK 2.9 because bluegill abundance was low at this site and redbreast sunfish could be collected from a shorter reach of stream closer to the weir at WCK 2.65 (Fig. 2.2). Eight redbreast sunfish and eight bluegill were also collected at WCK 2.9 and WOL, respectively, in July 1989 to continue the semiannual monitoring of PCB contamination in WOC. Although an attempt was made to restrict the collections to individuals of a size likely to be taken by sport fishermen, it was impossible to meet this requirement at all sites. Fish were collected from Hinds Creek (Anderson County, Tennessee) to estimate background levels of contaminants and to provide analytical controls.

Fish collected at each site were placed on ice in a labeled ice chest and returned to the laboratory for processing. Upon return to the laboratory, fish were tagged with a unique four-digit tag wired to the lower jaw. Each fish was then weighed, measured, and scales were taken for age determination. The fish was fileted, and the skin was removed from the filet. A 1- to 2-g sample of the anterior dorsal portion of the axial muscle filet was excised for the determination of mercury; the remainder of the filet was retained for analysis of other metals and radionuclides. Samples to be analyzed for metals were wrapped in aluminum foil and stored in a locked

freezer until analysis. The remaining filet was either used for a duplicate sample, archived, or analyzed for organic contaminants. Samples to be analyzed for organics or maintained in archival storage were wrapped in heavy-duty aluminum foil, labeled, and stored at -20°C in a locked freezer in Building 1504 until delivered to the ORNL Analytical Chemistry Division (ACD) for analysis.

Asiatic clams (*Corbicula fluminea*) were placed in the streams to monitor for PCBs and organic contaminants. Clams were obtained from Paint Rock Creek (Loudon County, Tennessee) or Beaver Creek (Knox County, Tennessee). After the clams were held for 24 h in clean flowing water, they were put into polypropylene cages and placed at various sites in WOC and tributaries, WOL, and WOC embayment at WCK 0.1. One set of clams collected from the reference site was frozen immediately for analysis as a control. Each cage held approximately 50 clams, which contained 0.5–2 g (wet weight) of soft tissue each. The clams were suspended in the stream for 4 weeks, at which time they were removed and processed prior to delivery to the ACD laboratory at ORNL. After freezing the clams, the shells were removed, and the frozen soft tissue was placed in a 20-mL glass vial. Duplicate composite samples weighing approximately 5–10 g were taken for PCB and pesticide analyses.

Mercury determinations were conducted using a modification of the procedure described in EPA (1979). Samples were digested in a mixture of nitric acid, perchloric acid, and potassium dichromate, after which the mercury was reduced with stannous chloride and determined by cold vapor atomic adsorption spectrophotometry. Other metals were determined using inductively coupled argon plasma emission spectroscopy following digestion with concentrated nitric acid (EPA 1980b). Organic priority pollutants were analyzed

by procedure PPB 12/83 (EPA 1984) in which the homogenized sample is extracted in methylene chloride, cleaned up using column chromatography, and analyzed by gas chromatography/mass spectrometry (GC/MS) or by gas chromatography with electron capture detectors (GC/ECD) and high performance liquid chromatography (HPLC) with fluorimetric detection.

Quality assurance was maintained using a combination of blind duplicate analyses; split-sample analyses between the EPA Environmental Services Laboratory in Athens, Georgia, and ORNL; and the analysis of biological reference standards and uncontaminated fish. Recoveries of representative organics were verified by spiking uncontaminated fish or clam samples with known amounts of PCBs and analyzing them. Details and results of these procedures are summarized in Appendix A.

Statistical evaluations of data were made using SAS procedures and software (SAS 1985a,b) for ANOVA, Tukey's multiple comparison test, linear regression analysis, t-tests, and the calculation of means, standard deviations, standard errors and coefficients of variation. Tests for homogeneity of variance among various data groups were conducted using Levene's test on untransformed and log_e-transformed variables (Sokal and Rohlf 1981). Comparisons were based on untransformed data unless Levene's test indicated that transformation was needed to meet assumptions of homogeneous variances. Dunnett's test was used to compare means of various groups with controls (Zar 1984). All comparisons were conducted using $p = 0.05$.

4.1.3 Results and Discussion

4.1.3.1 Metals

Mercury. Mercury contamination in fish from the WOC system in winter

1988–89 remained similar to the level of contamination reported in previous annual reports (Loar et al. 1992; Loar 1993a, 1993b). Although mercury concentrations exceeded those found in fish from Hinds Creek, a reference stream, they were well below the U.S. Department of Agriculture Food and Drug Administration (FDA) tolerance level (1 $\mu\text{g/g}$) in all fish (Table 4.1). Mean mercury concentrations in fish at the two sites in WOC nearest ORNL facilities (WCK 3.5, WCK 2.9), however, approached or exceeded the preliminary guidance values (PGV) derived to screen for contaminant levels that potentially threaten human health (Hoffman et al. 1984; Travis et al. 1986; Table 4.1 this report). Sixteen percent of the sunfish collected in the WOC drainage in 1988 exceeded this threshold. The highest mean total mercury concentration in the WOC drainage was found at WCK 2.9, where redbreast sunfish (*Lepomis auritus*) averaged 0.47 $\mu\text{g/g}$ and bluegill (*Lepomis macrochirus*) averaged 0.41 $\mu\text{g/g}$. The maximum mercury level observed was 0.69 $\mu\text{g/g}$ in a redbreast sunfish from WCK 2.9; the minimum level (0.03 $\mu\text{g/g}$) was found in a bluegill from WCK 0.9 (Appendix B, Table B.1).

A statistical comparison (linear regression, untransformed data) of mercury vs fish weight for sunfish collected from each site in the WOC system between 1986 and 1988 showed only 2 of 24 comparisons with a positive slope. Restricting collections of fish to a size likely to be taken by sport fishermen probably minimized the potential bias associated with any mercury concentration vs fish weight/age relationship. Thus, no normalization procedure was used in comparisons of fish mercury concentrations between sampling dates and sites.

To adequately assess mercury contamination in the WOC system, accurate background concentrations need to be quantified. After the 1987 sampling

Table 4.1. Mean metal concentrations in fish from White Oak Lake, White Oak Creek and tributaries

Concentrations in micrograms per gram, wet weight

Metal	Source/site ^a											
	NTK 0.2	WCK 3.5	WCK 2.9	WCK 2.3	MEK 0.2	WOL	WCK 0.9	Hinds Creek	TVA ^b	USFWS ^c	PGV ^d	
Antimony	NS	NS	<0.50	NS	<0.50	<0.50	NS	<0.50	—	—	5.2	
Arsenic	NS	NS	<0.50	NS	<0.50	<0.50	NS	<0.50	<0.03	0.16	0.0007	
Beryllium	NS	NS	<0.10	NS	<0.10	<0.10	NS	<0.10	<1	—	0.004	
Cadmium	NS	NS	<0.20	NS	<0.20	<0.20	NS	<0.20	0.007 (0.003)	0.04	1.0	
Chromium	NS	NS	<0.50	NS	<0.50	<0.50	NS	<0.50	0.06	—	1.8	
Copper	NS	NS	<0.50	NS	<0.50	0.52 ^e (0.06)	NS	<0.50	0.4 (0.3)	0.86	36	
Lead	NS	NS	<0.50	NS	<0.50	<0.50	NS	<0.50	0.21 (0.33)	0.19	1.8	
Mercury	0.19 ^f (0.10)	0.31 ^f (0.07)	0.44 ^g (0.15)	0.22 ^f (0.13)	0.09 ^f (0.04)	0.14 ^f (0.07)	0.09 ^f (0.03)	0.08 ^g (0.04)	<0.1	0.11	0.42	
Nickel	NS	NS	<0.50	NS	<0.50	<0.50	NS	<0.50	<0.1	—	5.2	
Selenium	NS	NS	0.50 (0.04)	NS	<0.50	0.51 ^f (0.04)	NS	<0.50	0.7	0.46 (0.2)	12	
Silver	NS	NS	<0.20	NS	<0.20	<0.20	NS	<0.20	<0.3 <1	—	0.29 0.66	
Zinc	NS	NS	5.2 (0.47)	NS	6.3 (1.2)	6.8 (1.7)	NS	5.6 (1.1)	8.4 (2.6)	25.6	180	

^aNTK = North Tributary kilometer; WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer; WOL = White Oak lake; Hinds Creek is a reference stream located in Anderson County, Tennessee; *n* = 2 except for Hg (*n* = 16).

^bTVA (Tennessee Valley Authority) 1985. *Instream Contaminant Study, Task 4: Fish Sampling and Analysis*, Report to U.S. Department of Energy, Oak Ridge Operations Office, Tennessee Valley Authority, Office of Natural Resources and Economic Development, Knoxville, Tenn. *n* = 13 for Hg, Cd; *n* = 9 for Cr, As, Ni; *n* = 4 for others.

^cLowe, T. P. et al., 1985. *National Contaminant Biomonitoring Program: Concentration of Seven Elements in Freshwater Fish, 1978-1982, Archive of Environmental Contaminant Toxicology*, 14:363-388.

^dPreliminary guidance values (from Hoffman, F. O. et al. 1984. Preliminary screening of contaminants in sediments, ORNL/TM-9370, Oak Ridge National Laboratory, Oak Ridge, Tennessee; and Travis, C. C. et al., 1986. *Preliminary Review of TVA Fish Sampling and Analysis Report*, Report of Task Group Five to the Oak Ridge Task Force, mimeo).

^eTwo of four samples exceeded detection limits.

^f*n* = 8.

^g*n* = 16.

Note: Fish are bluegill (*Lepomis macrochirus*) and redbreast sunfish (*Lepomis auritus*) collected in winter 1988; *n* = 4 except where noted; ± 1 standard deviation in parentheses. NS = not sampled.

revealed a marked difference in mercury concentrations in bluegill between the two reference sites [0.10 ± 0.01 and 0.04 ± 0.01 $\mu\text{g/g}$, mean \pm SE for Hinds Creek and Melton Hill Reservoir (MHR) respectively], two additional reservoirs and three streams (all with no known mercury input) were sampled in 1988-89 in an attempt to resolve this difference. Preliminary results indicated a significant difference between mean mercury concentrations in fish from reservoirs vs streams. The mean total mercury concentration in bluegill in the three reservoirs sampled in 1988-89 (Melton Hill, Norris, and Tellico) was 0.04 ± 0.01 $\mu\text{g/g}$ ($n = 16$), in contrast to the four streams (Hinds Creek, Paint Rock Creek, Brushy Fork, and Ball Play Creek), which had a mean mercury concentration in fish of 0.09 ± 0.01 $\mu\text{g/g}$ ($n = 20$). Reference sites on streams are clearly appropriate for comparison with sites in WOC and its tributaries. However, reference sites on reservoirs may be more appropriate for determining whether or not mercury concentrations in fish from WOL or the WOC embayment are elevated above background levels.

Comparing mean mercury concentrations in fish from WOC sites with those in fish from various reference sites may be useful in assessing the degree of contamination in WOC fish. When the mean mercury concentrations in bluegill collected from sites in the WOC system in 1988 were compared to the concentrations in bluegill from Hinds Creek alone (the reference site with the highest mean mercury concentration), only WCK 3.5, WCK 2.9, WCK 2.3, and NTK 0.2 were elevated significantly (Dunnett's test, \log_e -transformed data). However, when mean mercury concentrations in fish in WOC were compared to the concentrations in fish in the four reference streams pooled together, WOL was also significantly higher. Finally, when the mean mercury

concentrations in bluegill in WOC were compared to the mean of all reservoir sites sampled in 1988-89, all WOC sites were found to have a significantly elevated mean mercury concentration. These results suggest that only fish at the three WOC sites (WCK 3.5, WCK 2.9, and WCK 2.3) are contaminated to a significant degree with mercury; the other sites contain mercury concentrations in fish that probably range from near background to slightly above background. Continued sampling of multiple reference sites should help in identifying the concentrations of mercury that are representative of true background levels for fish in various aquatic environments on the ORR.

The distribution of mean mercury concentrations in bluegill in the WOC drainage and other sites on the ORR in winter 1988-89 is illustrated in Fig. 4.1. The presence of higher mercury concentrations in fish at those WOC sites nearest ORNL facilities was consistent with the observation of elevated mercury concentrations in water and sediment near outfalls in Fifth Creek and WOC near Building 4500 in 1988 (Rogers et al. 1989a). Mercury contamination in water and sediment in the WOC drainage was likely the result of losses of large quantities of mercury from the 4500 Complex in the 1950s (Rogers et al. 1989a). The conclusion that the source of mercury to WOC originates from the 4500 Area is supported by the fact that sediment concentrations of mercury in WOC decreased with an increase in distance downstream of the 4500 facilities. Mercury concentrations in fish followed a similar pattern.

Although elevated mercury concentrations (relative to local reference streams) were found in fish from NT (NTK 0.2), it seems unlikely that these levels reflect mercury exposure at this site. Rather, they are a result of the movement of contaminated fish into the Northwest

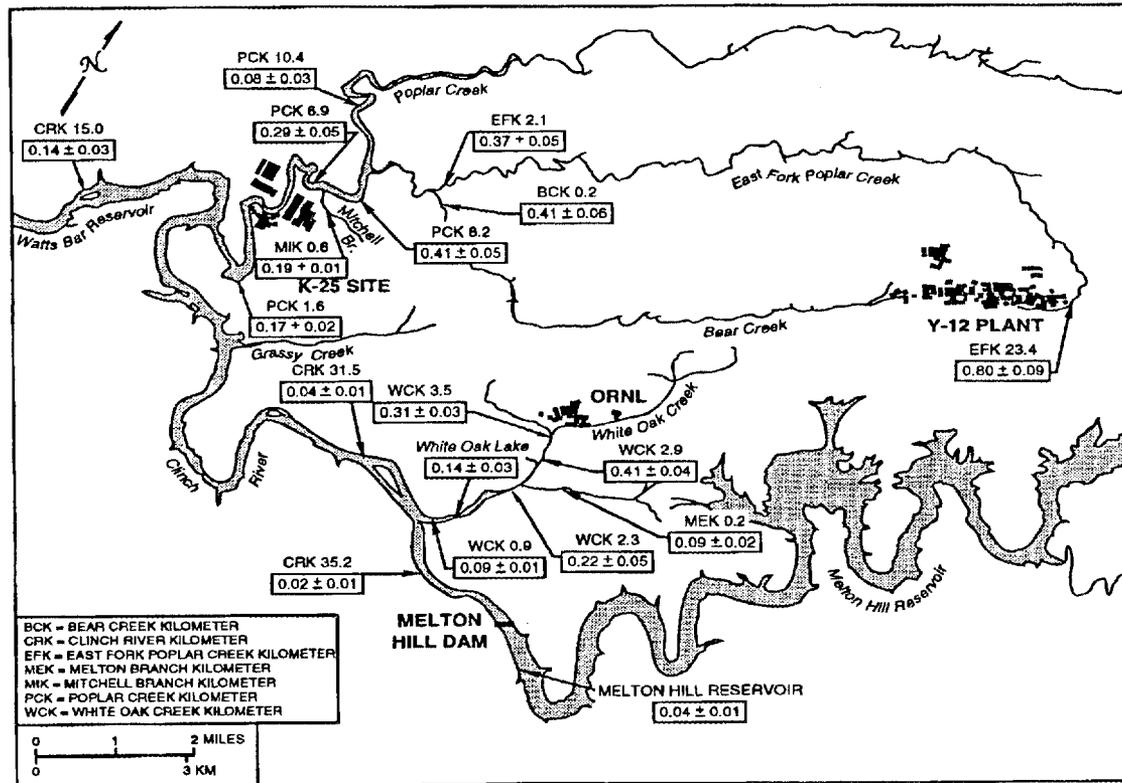


Fig. 4.1. Mean concentrations ± standard error (SE) of mercury (in micrograms per gram, wet weight, $n = 8$) in bluegill (*Lepomis macrochirus*) collected in winter 1988/1989 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River. Rock bass (*Ambloplites rupestris*) and redbreast sunfish (*Lepomis auritus*) were substituted for bluegill at Bear Creek kilometer 0.2 and Mitchell Branch kilometer 0.6 respectively.

Tributary from nearby reaches of WOC (Loar 1993b). Mercury concentrations in fish in Melton Branch (MEK 0.2) were not significantly elevated over reference streams (\log_e -transformed data). Mercury contamination in WOC in 1988 continued to have no discernable effect on mercury levels in fish in the Clinch River; mean mercury concentrations in bluegill collected downstream (CRK 31.5) of the mouth of WOC were not significantly different from concentrations in bluegill about 1.6 km upstream (CRK 35.2) or those in fish from MHR, a reference site assumed to be free of mercury contamination. Downstream dilution, coupled with the efficiency of

WOL in trapping mercury, have acted to confine any substantial mercury contamination in fish to the WOC drainage proper.

In general, the degree of mercury contamination in fish in the WOC drainage was low in comparison to the level of contamination at some other sites on the ORR (Fig. 4.1). The highest mean mercury concentration in bluegill in the WOC drainage is approximately half that in bluegill from EFK 23.4, the most contaminated site on EFPC. Sunfish in the upper third of EFPC routinely contain concentrations of mercury that exceed the 1.0 µg/g FDA action level (Loar 1992b).

Mean mercury levels in fish in WOC just below ORNL were more similar to the levels found in rock bass (*Ambloplites rupestris*) in Bear Creek (BCK 0.2) and in bluegill from (1) Poplar Creek kilometer (PCK) 8.2 and PCK 6.9 near the Oak Ridge Gaseous Diffusion Plant (ORGDP) and (2) lower EFPC at EFK 2.1, a site more than 20 km downstream of Y-12 Plant discharges. Mercury concentrations in fish from downstream WOC sites (WOL and WCK 0.9) were similar to those at PCK 1.6 and CRK 15.0, where the mercury inputs from EFPC have been substantially diluted. Mean mercury concentrations in fish from EFK 2.1, PCK 8.2, and PCK 6.9 were from two to four times higher than those in fish from WOL and WCK 0.9. Whereas mercury concentrations in fish immediately downstream of the mouth of WOC in the Clinch River/ WBR showed no evidence of mercury contamination, concentrations in fish at CRK 15.0 and PCK 1.6, which are located downstream of inputs from EFPC, Bear Creek, and Mitchell Branch, were slightly elevated when compared to reference areas.

Little change in mercury concentrations in bluegill and redbreast sunfish at sites in the WOC drainage was observed from December 1986 to December 1988 (Table 4.2). A statistical comparison indicated that differences between years were not significant at WCK 3.5, WCK 2.9, and NTK 0.2 (ANOVA, untransformed data). No overall increasing or decreasing trend for the WOC drainage could be discerned at sites where significant differences between years did occur. The mean mercury concentrations in fish at WCK 2.3 in 1987 and 1988 were significantly lower than in 1986, yet a significant increase was evident from 1986 to 1987 at WOL and from 1987 to 1988 at WCK 0.9. The mean mercury concentration in 1988 at WCK 0.9, although higher than 1987, was still quite low; the concentration approached

background levels and was below the mean concentration in bluegill of 0.16 $\mu\text{g/g}$ reported by the Oak Ridge Task Force in 1984 (TVA 1985). The increase from 1986 to 1987 in WOL followed a sharp decrease from the concentration reported by the Tennessee Valley Authority (TVA) in 1984 (0.23 $\mu\text{g/g}$) and that found in 1986 (0.09 $\mu\text{g/g}$). The results of these comparisons strongly suggest that current mercury levels in sunfish in the WOC drainage have changed little since sampling was initiated in 1984 and 1986.

Differences in mean mercury concentrations between species in WOC was not evident. No significant difference was observed between the mean mercury concentrations in redbreast and bluegill at WCK 2.9 in 1986-88 (ANOVA, untransformed data). Largemouth bass (*Micropterus salmoides*) and carp (*Cyprinus carpio*) were collected in 1988 to ensure contaminant concentrations in larger, usually older, species in the WOC drainage were not overlooked. Mean mercury concentrations in largemouth bass, carp, and bluegill in WOL were 0.11, 0.11, and 0.14 $\mu\text{g/g}$ respectively. Despite major differences in trophic level and age, no significant difference in the mean mercury concentrations between species was evident at this site (ANOVA, untransformed data). The concentrations in bass and carp were not elevated over those found in these species from reference areas (TVA 1985; Loar 1993c).

Other metals. As was reported in previous annual reports, fish collected from the WOC system contained all other metals at concentrations similar to those found in fish from the reference stream (Table 4.1; Appendix B, Table B.1). Only three metals other than mercury (copper, selenium, and zinc) were found in WOC fish above the analytical detection limit. The mean concentration in fish of each of these metals did not exceed that for the

Table 4.2. Concentrations of mercury in bluegill and redbreast sunfish (*Lepomis macrochirus* and *L. auritus*) from White Oak Creek and tributaries, 1986–1988^a

Site ^b	Species	Mercury concentration ($\mu\text{g/g}$, wet weight)		
		1986	1987	1988
WCK 3.5	Bluegill	0.25 \pm 0.04 (0.14 – 0.52)	0.30 \pm 0.02 (0.22 – 0.41)	0.31 \pm 0.03 (0.16 – 0.40)
WCK 2.9	Bluegill	0.40 \pm 0.05 ^c (0.30 – 0.52)	0.39 \pm 0.06 ^d (0.20 – 0.62)	0.41 \pm 0.04 (0.30 – 0.62)
	Redbreast	0.47 \pm 0.04 ^c (0.37 – 0.58)	0.37 \pm 0.03 (0.28 – 0.49)	0.47 \pm 0.06 (0.19 – 0.69)
WCK 2.3	Bluegill	0.44 \pm 0.11 ^c (0.20 – 0.73)	0.18 \pm 0.03 (0.05 – 0.28)	0.22 \pm 0.05 (0.09 – 0.49)
WOL	Bluegill	0.09 \pm 0.01 ^e (0.07 – 0.11)	0.25 \pm 0.04 (0.10 – 0.41)	0.14 \pm 0.03 (0.07 – 0.31)
WCK 0.9	Bluegill	0.07 \pm 0.01 ^f (0.05 – 0.08)	0.06 \pm 0.01 (0.04 – 0.08)	0.09 \pm 0.01 (0.03 – 0.12)
MEK 0.2	Bluegill	0.10 \pm 0.01 ^g (0.08 – 0.11)	0.06 \pm 0.01 (0.05 – 0.10)	0.09 \pm 0.02 (0.05 – 0.16)
NTK 0.2	Bluegill	0.26 \pm 0.02 (0.19 – 0.32)	0.24 \pm 0.03 (0.16 – 0.40)	0.19 \pm 0.03 (0.04 – 0.37)

^aValues are mean \pm 1 standard deviation (range in parentheses); number in sample (n) = 8 except where noted.

^bWOL = White Oak Lake; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer.

^c n = 4.

^d n = 7.

^e n = 5.

^f n = 6.

^g n = 3.

reference stream. Also, the concentrations of metals (except for mercury) in fish from the WOC system were similar to those reported by the TVA for MHR (Table 4.1; TVA 1985, 1986) and to the geometric mean concentrations of metals (Pb, Cd, As, Se, Cu, Zn) observed in fish in the National Contaminant Biomonitoring Program (Table 4.1; Lowe et al. 1985). A comparison of the levels of metals in fish

from the WOC system with PGVs derived to screen for levels of contamination that potentially threaten human health (Hoffman et al. 1984; Travis et al. 1986) indicated that only arsenic, beryllium, and mercury may approach this threshold (Table 4.1). Neither arsenic nor beryllium was elevated in WOC fish; however, the PGV is set at a level below background due to the carcinogenicity of these two

metals. The PGV screening approach is very conservative and designed to eliminate from concern any substances not exceeding a specific PGV (Hoffman et al. 1984).

4.1.3.2 Organics

In organic screening analyses, fish collected in WOC and Melton Branch in 1986 were found to be contaminated with PCBs (Loar 1993a). Beginning in 1987, therefore, fish were collected at all sites and analyzed for PCBs using packed column gas chromatography, which provides greater sensitivity and is used in this program for quantitative PCB studies. Semiannual sampling was continued at two sites (WOL and WCK 2.9) in order to establish a better baseline against which to evaluate future changes. Results of the 1988 survey affirmed the general conclusion of previous monitoring that a source of PCB contamination exists in the WOC drainage upstream from WCK 3.5 (Tables 4.3 and B.1). PCBs were found in sunfish at all sites in WOC, with the highest mean concentrations occurring in bluegill at sites in WOL and WOC upstream from WOL. Statistical comparisons (ANOVA, Dunnett's test on \log_e -transformed data) indicated that PCB concentrations in fish at all sites in the WOC watershed except NTK 0.2 were elevated relative to the concentration in fish from the reference stream (Hinds Creek). Although mean PCB concentrations were lower in fish from Melton Branch (MEK 0.2) and the WOC embayment (WCK 0.9) than in fish from WOL and WOC sites, statistical comparisons (ANOVA, Tukey's test using \log_e -transformed data) did not indicate the differences among these sites to be significant.

A comparison of PCB contamination in bluegill in the WOC drainage with levels

found in bluegill at other sites on the ORR in winter 1988/1989 is shown in Fig. 4.2. The additional data shown in Fig. 4.2 are from the ORGDP and Y-12 Plant BMAPs. Concentrations of PCBs in fish from WOC were similar to those in upper EFPC near the Y-12 Plant and lower than those found in fish from Mitchell Branch, a small stream draining a portion of the ORGDP impacted by a recent PCB spill (Smith et al. 1988). Fish from sites farther downstream from the apparent sources (WCK 0.9, EFK 2.1, PCK 8.2, PCK 6.9 and PCK 1.6) had lower concentrations in response to dilution of the upstream inputs. Elevated concentrations of PCBs were not evident in sunfish collected in the Clinch River downstream from ORNL discharges (via WOC) or Y-12 Plant and ORGDP discharges (via EFPC and Poplar Creek). This finding represents a limitation of sunfish as a monitor for very diluted inputs of PCBs rather than the absence of any impact in the Clinch River system, because channel catfish (*Ictalurus punctatus*) collected at the same Clinch River sites in 1989 continued to contain elevated concentrations of PCBs (Sect. 4.2).

The PCB mixtures found in fish from WOC and tributaries most closely resembled commercial mixtures Arochlor 1254 and 1260 (referred to as PCB-1254 and PCB-1260 because the composition of PCBs extracted from fish differs substantially from that of commercial mixtures). PCB-1254 generally predominated in samples taken from WOC, but the proportion of PCB-1260 was higher in fish from WOL.

No other organics were detected in fish from WOC, except for trace levels of PAHs and phthalates (di-*N*-butylphthalate, diethylphthalate, and di-*N*-octylphthalate) which were also present in reference stream fish and laboratory blanks (Table 4.3).

Table 4.3. Concentrations of polychlorinated biphenyls and other organic priority pollutants in bluegill and redbreast sunfish (*Lepomis macrochirus* and *L. auritus*), largemouth bass (*Micropterus salmoides*), and carp (*Cyprinus carpio*) from the Clinch River, White Oak Lake, White Oak Creek, Melton Branch, Northwest Tributary, and Melton Hill Reservoir

Site ^a	Species	Date	Concentration ($\mu\text{g/g}$, wet weight)			
			Polychlorinated biphenyls (PCBs)			Other organics ^{d,e}
			Mean \pm SE ^b	Range	Exceedances ^c	
WCK 3.5	Bluegill	Nov. 1988	0.63 \pm 0.30	0.14–2.7	1/8	BLD ^f
WCK 2.9	Redbreast	Nov. 1988	0.31 \pm 0.06	0.13–0.58	0/8	BLD
	Bluegill	Nov. 1988	0.54 \pm 0.21	0.18–2.01	1/8	—
	Redbreast	July 1989	0.28 \pm 0.10	0.07–0.78	0/8	—
WCK 2.3	Bluegill	Nov. 1988	0.63 \pm 0.16	0.18–1.52	0/8	—
WOL	Bluegill	Nov. 1988	0.62 \pm 0.20	0.09–1.72	0/8	—
	Bluegill	July 1989	0.44 \pm 0.10	0.02–0.80	0/8	—
	Carp	Nov. 1988	0.57 \pm 0.09	0.24–0.85	0/7	BLD
	Largemouth bass	Nov. 1988	1.62 \pm 0.64	0.21–5.0	2/8	—
WCK 0.9	Bluegill	Nov. 1988	0.28 \pm 0.09	0.08–0.79	0/8	BLD
MEK 0.2	Bluegill	Nov. 1988	0.30 \pm 0.10	0.09–0.92	0/8	—
NTK 0.2	Bluegill	Nov. 1988	0.12 \pm 0.04	0.04–0.36	0/8	BLD
CRK 31.5	Bluegill	Nov. 1988	0.07 \pm 0.02	0.04–0.11	0/8	—
CRK 35.2	Bluegill	Nov. 1988	0.15 \pm 0.02	0.08–0.30	0/8	—
MHR	Bluegill	Nov. 1988	0.15 \pm 0.02	0.05–0.28	0/8	—
Hinds Creek	Bluegill	Nov. 1988	0.08 \pm 0.02	0.02–0.14	0/8	—
	Redbreast	Nov. 1988	0.10 \pm 0.03	0.01–0.18	0/8	—

^aWCK = White Oak Creek kilometer, WOL = White Oak Lake, MEK = Melton Branch kilometer, NTK = Northwest Tributary kilometer, CRK = Clinch River kilometer, and MHR = Melton Hill Reservoir.

^bSE = standard error.

^cNumber of samples exceeding the U.S. Department of Agriculture Food and Drug Administration limit for PCBs of 2 $\mu\text{g/g}$ divided by the total number of samples.

^d $n = 4$.

^eLow concentrations of benzo[*a*]pyrene, (<0.10 $\mu\text{g/g}$); pyrene (<0.03 $\mu\text{g/g}$); benzo[*k*]fluoranthene, (<0.10 $\mu\text{g/g}$); and acenaphthene, (<0.10 $\mu\text{g/g}$) were reported in high performance liquid chromatography analysis of some samples. Low concentrations of phthalates (<0.6 $\mu\text{g/g}$) were reported in several samples, blanks, and reference stream fish.

^fBelow the limit of detection.

ORNL-DWG 93M-15874

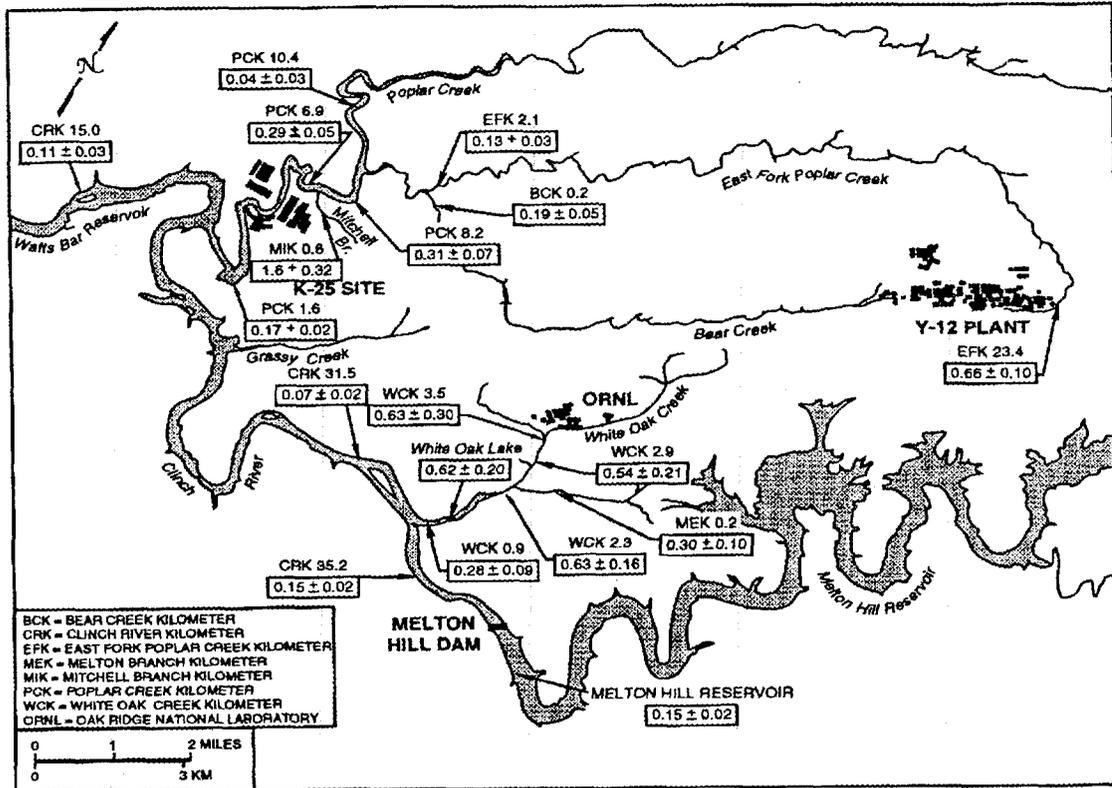


Fig. 4.2. Mean concentrations ± standard error of polychlorinated biphenyls (in micrograms per gram, wet weight, $n = 8$) in bluegill (*Lepomis macrochirus*) collected in winter 1988/1989 at sites on the Oak Ridge Reservation and nearby reaches of the Clinch River. Rock bass (*Ambloplites rupestris*) and redbreast sunfish (*Lepomis auritus*) were substituted for bluegill at BCK 0.2 and MEK 0.6 respectively.

Polychlorinated biphenyls vs time.

Concentrations of PCBs in fish from WOC and WOL appear to have decreased over the period from December 1987 to July 1989. Redbreast sunfish at WCK 2.9 averaged about 0.9 µg/g in December 1987/July 1988, and about 0.3 µg/g in December 1988/July 1989 (Table 4.4). A similar but smaller decrease was observed in bluegill in WOL. Only at WCK 2.9 was there a statistically significant difference in PCB concentrations among seasons (one-way ANOVA, log_e-transformed data). Concentrations of PCBs in fish at the remaining sites (sampled annually rather

than semiannually) did not clearly indicate a decrease between 1987 and 1988. Although large decreases occurred in bluegill at WCK 2.9, WCK 0.9, and NTK 0.2, none of them was statistically significant (t-test on untransformed data with assumption of unequal variances, when appropriate). The increase in mean PCB concentration in bluegill in Melton Branch between 1987 and 1988 was statistically significant.

PCBs have been monitored twice yearly in sunfish in EFPC below the Y-12 Plant since May 1985. Results of that monitoring indicate that PCB

Table 4.4. Comparison of mean concentrations of polychlorinated biphenyls in sunfish from White Oak Lake and White Oak Creek, Melton Branch, and Northwest Tributary, December 1987–July 1989^a

Site ^b	Species	Polychlorinated biphenyls ($\mu\text{g/g}$, weight wt)			
		December 1987	July 1988	December 1988	July 1989
MEK 0.2	Bluegill	0.06	—	0.30	—
NTK 0.2	Bluegill	0.35	—	0.12	—
WCK 3.5	Bluegill	0.70	—	0.63	—
WCK 2.9	Bluegill	1.47	—	0.54	—
WCK 2.9	Redbreast	0.88	0.89	0.31	0.28
WCK 2.3	Bluegill	0.65	—	0.63	—
WOL	Bluegill	0.91	1.17	0.62	0.44
WCK 0.9	Bluegill	0.52	—	0.28	—

^a n = 8 fish per site.

^bMEK = Melton Branch kilometer, NTK = Northwest Tributary kilometer, WCK = White Oak Creek kilometer, WOL = White Oak Lake.

contamination in fish may vary considerably between sampling periods without exhibiting a long-term increasing or decreasing trend. Care must be taken in evaluating temporal changes in PCB concentrations in fish in WOL and WOC; a time interval of 1.5 years is not sufficient to determine whether or not important temporal trends actually do exist.

Polychlorinated biphenyls vs species.

It is important that a PCB monitoring effort not underestimate the significance of sources of contamination by neglecting to measure PCB concentrations in important game or food species that may accumulate such contaminants to higher levels than most other species. Channel catfish serves this role in monitoring ORNL inputs to the Clinch River (Sect. 4.2). In fall 1988, largemouth bass (*Micropterus salmoides*) and carp (*Cyprinus carpio*) were also collected from WOL and evaluated as potential measures of maximum biotic accumulation of PCBs in that system.

Despite large differences in mean PCB concentrations between largemouth bass and the other two species (Table 4.3), those differences were not statistically significant (ANOVA on \log_e -transformed data). Several bass had accumulated high levels of PCBs (over $2 \mu\text{g/g}$); however, several others contained very low concentrations, and overall, a high SD was associated with PCB accumulation in this species in WOL. Although they appear to be more variable than other species in PCB accumulation, the largemouth bass is an appropriate species for monitoring maximum levels of organic bioaccumulation in food and game fish in WOL. Concentrations of PCBs in carp and bluegill were quite similar, a surprising result considering that levels in carp from EFPC averaged 3–4 times higher than in sunfish (Rogers et al. 1989a).

PCBs were determined in both bluegill and redbreast sunfish at WCK 2.9 in December 1987 and 1988 (Table 4.4). Mean PCB concentrations were higher in bluegill than in redbreast sunfish on both

dates, but in neither case was the difference statistically significant (t-test on untransformed data).

Bioaccumulation studies using caged clams. The results of PCB analysis of caged clams held in WOC and tributaries in November 1988 are shown in Table 4.5. Elevated concentrations of PCBs were evident in clams from WCK 2.6 downstream through sites in WOL and WCK 0.1, but PCBs were not elevated in clams held at sites upstream from WCK 2.6. Such a pattern was observed previously in WOC (Loar 1993b, Sect. 4.2) and, once again, presents the seemingly contradictory findings of PCB contamination in fish in the pool just upstream from the weir at WCK 3.41 but not in clams caged in WOC immediately downstream from the weir. In 1987, the PCB concentration in clams from WCK 2.6 was also much higher than that in clams from WCK 3.5 (Loar 1993a). The consistency of this pattern, together with results of sediment analyses by ORNL EMC (Rogers et al. 1989a,b), suggests that PCB contamination is indeed present at WCK 3.5, but that clams held at this site are unable to accumulate significant PCB residues. The presence of a chlorinated discharge from the STP a short distance upstream may be a factor that stresses clams and prevents them from effectively accumulating PCBs (i.e., by forcing them to restrict their filtering rate to avoid exposure to toxic chlorine concentrations, resulting in less feeding and PCB uptake and the consumption of lipid reserves that would accumulate PCBs).

Clams maintained in Melton Branch also did not accumulate PCBs (Table 4.5). Fish from this site did contain elevated PCBs (Table 4.4) in 1988 but not in the previous year. Sediments in Melton Branch were not found to be contaminated with PCBs by ORNL EMC in 1988 (Rogers et al. 1989a,b). Because fish are

able to move between WOC and the sampling site at MEK 0.2 (below the weir), it is possible that the PCB contamination observed in fish at this site results from the immigration of contaminated fish from WOC. However, mercury concentrations in the fish from MEK 0.2 were lower than those typical of WOC fish.

Chlordane in upper White Oak Creek. Suspected chlordane contamination was found in 1988 in caged clams placed in WOC at WCK 5.4, upstream from most ORNL facilities (Loar 1993b). Results of a follow-up study in November 1988 confirmed the presence of chlordane and suggested a potential source (Table 4.5). No chlordane was detected in clams placed in upper WOC (WCK 6.8), but chlordane was detected in clams at WCK 5.6 near the drain field for the septic tank in the 7000 Area. Much higher concentrations were found in clams placed in WOC downstream from a tributary at WCK 5.4. Detectable concentrations of chlordane were also found in clams placed in WOC downstream from the main ORNL complex (WCK 2.6) and in WOL.

Additional studies in April 1989 investigated the small tributary at WCK 5.4 (ORNL Outfall 233) as a possible source of the contamination. Clams placed in this tributary accumulated much higher concentrations of chlordane than clams placed in nearby WOC (Table 4.5). Similarly, clams placed in WOC downstream of the tributary accumulated a six-fold higher concentration of chlordane than clams placed in WOC upstream from the tributary.

Sediment samples were collected in November 1989 from the contaminated tributary and from WOC upstream and downstream of the tributary to determine the degree of abiotic contamination in the system and to evaluate the utility of sediment analyses in tracking the contamination back to its source. Results

Table 4.5. Concentrations of chlordane and polychlorinated biphenyls in duplicate composite samples of caged clams (*Corbicula fluminea*) held for 4 weeks at sites in White Oak Creek, White Oak Lake, and tributaries; and concentrations of chlordane in sediment at sites in the White Oak Creek drainage

Site ^a	Concentration ($\mu\text{g/g}$)			
	Chlordane			Polychlorinated biphenyls
	Clams ^b		Sediment ^c	Clams
	November 1988	April 1989	October 1989	November 1988
WCK 6.8	<0.025	0.007	<0.025	0.04
	<0.025	0.003–0.011		0.03–0.04
WCK 5.6	0.120	0.108	0.142	0.04
	0.115–0.125	0.103–0.113		0.03–0.04
WCK 5.45	—	—	0.099	—
Trib WCK 5.4	—	2.54 2.54–2.55	0.143	—
WCK 5.4	1.26	0.596	0.185	0.05
	1.12–1.40	0.489–0.703		0.03–0.06
WCK 3.4	—	—	—	0.05 0.04–0.06
WCK 2.6	0.115	—	—	0.23
	0.105–0.124			0.20–0.25
WOL	0.084 0.079–0.088	—	—	0.52 0.51–0.52
WCK 0.1	—	—	—	0.32 0.29–0.34
MEK 0.2	—	—	—	0.09 0.07–0.10
Reference ^d	<0.025	0.020	—	0.05
	<0.025	0.019–0.020	—	0.03–0.07

^aWCK = White Oak Creek kilometer, WOL = White Oak Lake, MEK = Melton Branch kilometer.

^bConcentration measured in micrograms per gram, wet weight.

^cConcentration measured in micrograms per gram, dry weight, <125- μm fraction.

^dBackground concentrations in clams before placement in stream or concentration in reference stream sediment.

of this sampling were not clear (Table 4.5); sediments taken from the tributary and three sites in WOC all contained similar levels of chlordane ($\sim 0.1 \mu\text{g/g}$). However, a sample of the unusual looking black, crumbly soil adjacent to the tributary north of WOD contained $1.2 \mu\text{g/g}$ chlordane.

Although found in clams, chlordane was not detected in fish collected in the WOC system in 1988 (Table 4.3). Composite samples (one per site, eight fish per sample) of channel catfish collected from the Clinch River in 1988 (Fig. 4.3) were analyzed and similar low concentrations of chlordane were reported in all samples. Catfish from MHR contained $0.06 \mu\text{g/g}$ chlordane, and fish from Clinch River sites downstream of WOC (CRK 32.2 and 15.0) contained 0.05 and $0.02 \mu\text{g/g}$ respectively. Fish from the two embayments receiving discharges from DOE facilities (WOC and Poplar Creek embayments) contained 0.07 and $0.05 \mu\text{g/g}$ chlordane respectively. All wild fish contained much higher levels than did the control fish from a commercial fish farm ($<0.001 \mu\text{g/g}$).

4.1.4 Conclusions

Mercury concentrations in WOC fish were elevated over reference streams in winter 1988 but were well below the FDA action level throughout the drainage. Elevated concentrations in fish were generally restricted to those WOC sites nearest to ORNL facilities, with little or no contamination at sites downstream of WOL and in the Clinch River. The level of contamination in WOC is low compared to upper EFPC but similar to that at sites in lower EFPC and Poplar Creek. Mercury contamination in WOC sunfish has remained stable since 1986. Sources of mercury to the WOC system have been identified, but the nature of such sources and their amenability to remedial actions is unknown.

The bioaccumulation of PCBs in fish remains the most significant problem in the WOC system caused by the accumulation of metals and organics in biota. The accumulation of PCBs in fish from the WOC system was again evident in the 1988–89 sampling. Biotic contamination continues to reflect recent accumulation rather than historical residues from past exposures, and the source of ongoing PCB inputs to stream water appears to be sediments or discharges somewhere between WCK 3.5 and WCK 4.7. The WOC drainage continues to act as a source of PCB contamination to upper WBR; however, these inputs do not have a perceptible impact on PCB concentrations in sunfish and, presumably, other species whose food habits, intramuscular lipid contents, and life span are similar. The accumulation of PCBs by fat-rich forage fish, such as gizzard shad, and their subsequent transfer to predatory fish provides a mechanism by which WOC discharges can have an impact on the PCB concentrations in some fish in the Clinch River (such as channel catfish) yet produce no discernable increase in PCBs in sunfish (see additional discussion in Sect. 4.2)

Largemouth bass from WOL contained substantially higher concentrations of PCBs than sunfish. Although the difference was not statistically significant, it is likely that a comparison based on a larger sample size would demonstrate a statistically significant difference. Additional monitoring of PCB concentrations in this species in WOL is warranted.

Chlordane contamination in upper WOC was confirmed. The presence of chlordane in riparian soil, low levels of contamination in clams held in WOC near the 7000 Area, and higher levels in clams held in the tributary at WCK 5.4 and in WOC downstream of the tributary suggest that contributions originate from a general area source with local hot spots near the tributary. While there is no evidence of

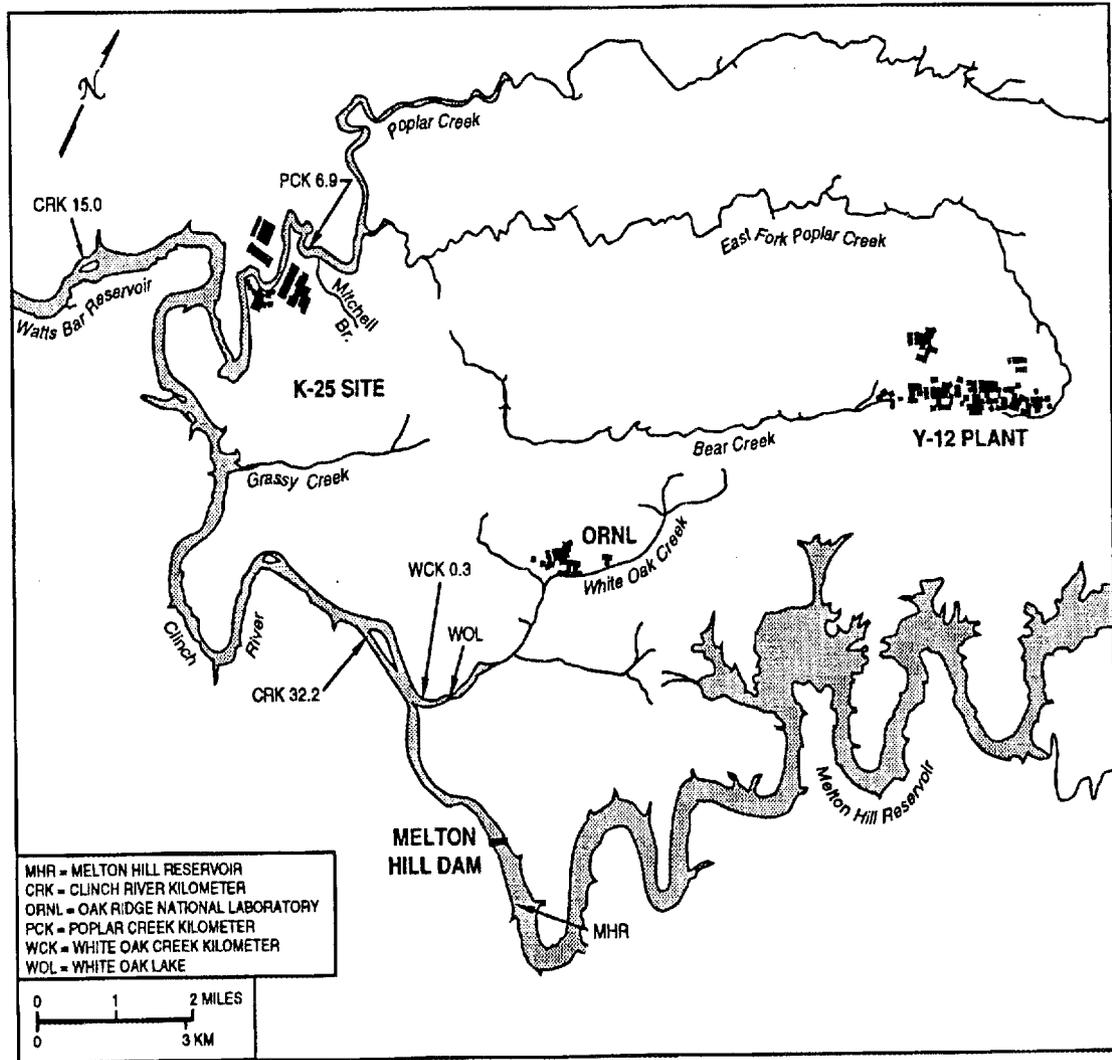


Fig. 4.3. Location of sites on the Oak Ridge Reservation where channel catfish were collected for polychlorinated biphenyl analysis in July/August 1989. Numerical designations indicate distance in kilometers above the mouth of the stream (i.e., Clinch River kilometer 15.0 is located in the Clinch River 15-km upstream from its mouth).

elevated chlordane concentrations in fish in WOC or the Clinch River, the concentrations found in caged clams imply that chlordane concentrations in sport or

food fish in WOC near WCK 5.4 would exceed levels deemed hazardous for consumption, if such fish were present in the stream (EPA 1986b).

4.2 EVALUATION OF PCB CONTAMINATION IN WHITE OAK CREEK EMBAYMENT/CLINCH RIVER

4.2.1 Introduction

Previous monitoring (Loar et al. 1992; 1993a, 1989) established that the WOC discharge is a source of PCB contamination to the WOC embayment and the Clinch River arm of WBR. These studies also indicated that other sources were significant contributors to PCB levels in catfish near ORNL. In particular, MHR, which is located on the Clinch River above the WOC discharge, appeared to contribute ~50% of the PCB burden of catfish in the upper Clinch River.

The objectives of this task of the ORNL BMAP in 1989 were to continue routine monitoring of PCBs in channel catfish in the WOC embayment and nearby reaches of the Clinch River to ensure that PCB contamination from this source does not worsen and to continue efforts to assess the significance of ORNL as the source of PCB contamination in channel catfish in public waters downstream. To compare the impact of ORNL as a PCB source to Clinch River catfish with that of the EFPC/Poplar Creek system, which is also a major contributor of PCBs (Loar 1992b), the 1989 monitoring program included sites in lower Poplar Creek adjacent to ORGDP and in the Clinch River several kilometers downstream from the mouth of Poplar Creek. Measuring ^{90}Sr in catfish vertebrae was continued as a means of identifying fish with substantial exposure to the WOC discharge.

4.2.2 Methods

Results of monitoring conducted in 1986-88 indicated that channel catfish

collected from sites near the mouth of WOC contained PCB concentrations similar to fish collected in the WOC embayment. Beginning in 1988, therefore, collections near ORNL were restricted to (1) the WOC embayment (WCK 0.3) to measure the maximum impact of the WOC/WOC embayment source on PCB concentrations in this species; (2) a single site in the Clinch River 1.3 km downstream from the mouth of WOC (CRK 32.2) to measure the maximum impact of the WOC source on PCB levels in channel catfish populations in the Clinch River; and (3) MHR to identify that component of the PCB burden in Clinch River fish associated with sources upstream from WOC. To evaluate the relative significance of WOC and Poplar Creek as sources of PCB contamination in Clinch River catfish, collections were also made in lower Poplar Creek (PCK 6.9) and in the Clinch River about 4 km below the mouth of Poplar Creek (CRK 15.0). Catfish were collected in July/August 1989 by trotline, gill net, or electrofishing at the five sites noted in Fig. 4.3. The sampling program attempted to capture eight channel catfish weighing 440 g or more at each site. Uncontaminated catfish purchased from a local commercial fish farm in 1987 were used again in 1989 as analytical controls.

Fish collected at each site were placed on ice in a labeled ice chest and returned to the laboratory for processing. Individual ice chests were used to contain fish from each site when more than one station was sampled on a given date. Upon return to the laboratory, fish were tagged with a unique four-digit tag wired to the lower jaw. Each fish was then weighed, measured, and the dorsal fin removed for possible age determination. The fish was fileted, and the skin was removed from the filet. Filets were rinsed in running water, and one filet was wrapped in heavy duty aluminum foil and stored in a locked

freezer at -20°C for archival purposes. The remaining filet was frozen and then ground three times in a hand meat grinder. A 10–20-g sample of ground fish was wrapped in heavy duty aluminum foil, labeled by writing directly on the aluminum foil with a permanent marker, and stored in a locked freezer until submitted to the ORNL/ACD for analysis.

A 4- to 5-cm portion of the vertebral column was removed from the tail of each catfish, freeze-dried, wrapped and labeled, and sent to the ACD for ^{90}Sr analysis.

Fish were analyzed for PCBs using EPA procedure 600/4-81-055 (EPA 1980b). This procedure uses extraction with methylene chloride followed by adsorption column cleanup, solvent exchange, and evaporative concentration prior to analysis by packed column GC/ECD. Strontium-90 was determined by beta counting using a low background proportional counter. Prior to counting, the fish vertebrae were ashed, dissolved in nitric acid, and subjected to a chemical purification procedure in which a strontium oxalate precipitate is ultimately isolated for counting (Volchok and Planque 1982).

Methods used in the statistical evaluation of the data are described in Sect. 4.1.2. Quality assurance was maintained using a combination of blind duplicate analyses; split-sample analyses between the EPA Environmental Services Laboratory, in Athens, Georgia, the TVA Laboratory in Chattanooga, Tennessee, the TDHE Laboratory in Nashville, Tennessee, and the ORNL/ACD Laboratory; and the analysis of fish reference standards and uncontaminated fish spiked with PCBs. Details and results of these procedures are summarized in Appendix A.

4.2.3 Results and Discussion

4.2.3.1 Polychlorinated biphenyls in catfish from the White Oak Creek Embayment/Clinch River

PCB contamination was evident in channel catfish from the WOC embayment, the Clinch River near ORNL and ORGDP, and in lower Poplar Creek (Tables 4.6 and Appendix B, Table B.4). As was the case in 1986, 1987, and 1988 (Loar et al. 1992; Loar 1993a, 1993b), channel catfish collected from MHR (upstream from any possible inputs from WOC) also contained significant concentrations of PCBs, although the concentrations observed in 1989 were the lowest to date. The PCB mixtures recovered from fish contained predominantly penta- and hexachlorobiphenyl congeners characteristic of Arochlor 1254 and 1260 commercial mixtures. PCB-1260 predominated in fish at all sites; however, the relative contribution of PCB-1254 was greater at WCK 0.3. Total PCBs in individual fish ranged from a high of $3.4\ \mu\text{g/g}$ at PCK 6.9 to a low of $<0.01\ \mu\text{g/g}$ in a fish from CRK 15.0. Catfish obtained in 1987 from a commercial catfish farm were reanalyzed in 1989 and again contained low to very low levels of PCBs, averaging $0.02\ \mu\text{g/g}$. The minimum PCB concentration found in Clinch River/WOC embayment fish from the ORNL vicinity was $0.52\ \mu\text{g/g}$.

Contaminants with long biological half-lives tend to accumulate in a fish to ever higher levels throughout much of its life span. Thus, larger, older fish may contain higher contaminant levels than

Table 4.6. Concentrations of polychlorinated biphenyls and strontium-90 in channel catfish (*Ictalurus punctatus*) from sites in the Clinch River arm of Watts Bar Reservoir in the vicinity of U.S. Department of Energy facilities, July/September 1989^a

Location ^b	Concentration ($\mu\text{g/g}$)			
	PCB-Total	PCB-1254	PCB-1260	⁹⁰ Sr ^d
WCK 0.3	1.54 \pm 0.83	0.55 \pm 0.38	0.98 \pm 0.51	1744 \pm 1285
	0.61 - 2.68	0.04 - 1.21	0.38 - 1.53	80 - 4000
	(8)	(8)	(8)	(8)
CRK 32.2	1.20 \pm 0.89	0.22 \pm 0.24	0.99 \pm 0.70	61 \pm 83
	0.52 - 3.06	<0.01 - 0.77	0.32 - 2.29	0 - 200
	(8)	(8)	(8)	(8)
PCK 6.9	1.07 \pm 0.99	0.13 \pm 0.15	0.94 \pm 1.00	62 \pm 57
	0.25 - 3.37	0.01 - 0.48	0.21 - 3.31	0 - 170
	(8)	(8)	(8)	(8)
CRK 15.0	0.79 \pm 0.70	0.08 \pm 0.10	0.70 \pm 0.72	112 \pm 94
	<0.01 - 2.10	<0.01 - 0.27	<0.01 - 2.10	0 - 270
	(8)	(8)	(8)	(8)
MHR	0.28 \pm 0.12	0.07 \pm 0.04	0.21 \pm 0.10	25 \pm 67
	0.14 - 0.55	0.01 - 0.14	0.10 - 0.41	0 - 120
	(8)	(8)	(8)	(8)

^aValues are mean \pm 1 standard deviation, range, and number of samples (in parentheses).

^bWCK = White Oak Creek kilometer, CRK = Clinch River kilometer, PCK = Poplar Creek kilometer, and MHR = Melton Hill Reservoir.

^cMeasurements are wet weight.

^dNegative concentrations (a result of random counting statistics) were rounded to zero in table but not in statistical calculations (mean, standard deviation, regression).

smaller, younger fish. The effects of such a bias can be significant if fish collected from some sites are predominantly small while those from other sites are predominantly large. Such effects can often be avoided by collecting adult fish of similar size. Consequently, channel catfish weighing <440 g were not included in this study if adequate numbers of larger fish were collected at a site. Mean weights of fish comprising the collections varied somewhat among sites (735-1428 g). In

general, mean fish size more closely resembled that of the 1986 and 1988 collections than the 1987 one. Regressions of total PCB, PCB-1254, and PCB-1260 concentrations vs fish weight at each site showed no significant relationship (slope not different from zero, $p > 0.05$) in 12 of 15 possible regressions. Therefore, comparisons of mean contaminant levels among sites were made without normalizing for variations in fish weight. Lipid content of fish muscle was

determined along with PCB concentrations to ensure that the data collected in this program could be included in the Tennessee Valleywide data set compiled by TVA and TDHE (Table B.4). Results of the regression of PCB concentration vs lipid content (all sites except MHR pooled) indicated virtually no relationship (slope not significantly different from zero, $r^2 = 0.03$).

The highest mean concentration of PCBs in catfish collected in 1989 was 1.54 $\mu\text{g/g}$ in fish from the WOC embayment (Table 4.7). The mean PCB concentration in fish at this site was also highest in 1986 and 1988 and differed only slightly from the maximum in 1987 (Table 4.7).

significantly among fish from WCK 0.3, CRK 32.2, CRK 15.0, and PCK 6.9 (ANOVA, untransformed data). Mean concentrations of PCB-1260 at downstream sites also differed significantly from the concentration in MHR but did not differ significantly from each other (Dunnett's test, ANOVA; untransformed data). The mean concentration of PCB-1254 was significantly higher in fish from WCK 0.3 than from any of the other sites. Concentrations of PCB-1254 did not differ among the remaining sites (Dunnett's test, ANOVA on untransformed data).

Results of the analyses of ^{90}Sr in catfish vertebrae in 1989 differed somewhat from those of 1988. In 1988, mean concentrations of ^{90}Sr in fish from

Table 4.7. Changes from 1986 to 1989 in average concentrations of polychlorinated biphenyls and fraction of fish exceeding the U.S. Department of Agriculture Food and Drug Administration limit, for channel catfish^a

Site ^b	Polychlorinated biphenyls ($\mu\text{g/g}$, wet weight)				Fraction over FDA ^c limit			
	1986	1987	1988	1989	1986	1987	1988	1989
WCK 0.3	1.30	1.59	0.96	1.54	3/12	2/8	2/8	4/8
CRK 32.2	1.01	1.61	0.58	1.20	0/8	2/8	1/8	1/8
MHR	0.46	0.81	0.52	0.28	0/6	1/7	0/10	0/8
PCK 6.9	—	—	0.71	1.07	—	—	0/8	1/8
CRK 15.0	—	—	0.50	0.79	—	—	0/9	1/8

^aSite locations are depicted in Fig. 4.3.

^bWCK = White Oak Creek kilometer, CRK = Clinch River kilometer, MHR = Melton Hill Reservoir, and PCK = Poplar Creek kilometer.

^cFDA = Food and Drug Administration.

Concentrations of PCBs in fish from all sites except CRK 15.0 below ORGDP differed significantly from the upstream reference site on MHR (Dunnett's test on \log_e -transformed data). However, mean PCB concentrations did not differ

WCK 0.3 and a nearby reach of the Clinch River (CRK 32.2) were similar (~700 Bq/kg; Loar 1993b), but in 1989, fish from WCK 0.3 contained much higher concentrations than fish from CRK 32.2 (1744 vs 61 Bq/kg, respectively; Table 4.6).

Little evidence of the ORNL ^{90}Sr source was found in fish at any of the sites downstream from the WOC embayment; concentrations were low at all sites (Table 4.6), and mean concentrations did not differ significantly from the mean concentration in the fish from MHR, which receives no major ^{90}Sr input (Dunnett's test, \log_e -transformed data). WCK 0.3 was the only site where the mean ^{90}Sr concentration in fish bone differed significantly from that at the MHR reference site. The pattern of ^{90}Sr vs location suggests that there was little movement of channel catfish in and out of the WOC embayment in summer 1989. No fish collected at sites outside the embayment contained ^{90}Sr at concentrations typical of substantial residency in the embayment (>1000 Bq/kg). Past collections typically found several fish with such concentrations of ^{90}Sr at sites well outside the WOC embayment (Loar et al. 1992; Loar 1993a, 1993b). Fish collection efforts in the WOC embayment indicated high densities of channel catfish may have occurred in the embayment in 1989 as numerous fish could be obtained by electrofishing, which is usually an ineffective method for collecting catfish at that site.

4.2.3.2 Polychlorinated biphenyl vs strontium-90 concentrations

The correspondence between PCB and ^{90}Sr concentrations in fish was closer than that observed in 1988. Linear regressions of total PCBs and PCB-1254 vs ^{90}Sr were significant when all sites were pooled, although the low r^2 values (0.13 and 0.39 for total PCB and PCB-1254 respectively) indicate that a relatively small proportion of the variation in PCBs was predicted by ^{90}Sr . As has been the case in the past (Loar et al. 1992; Loar 1993a, 1993b), the strongest relationship was

found between PCB-1254 and ^{90}Sr . When sites that received PCB inputs from EFPC and Mitchell Branch (CRK 15.0 and PCK 6.9) were excluded, the results changed very little. If the significant regression expressions (all sites),

$$[\text{PCB}] = 3.5 \times 10^{-4} [^{90}\text{Sr}] + 0.85$$

and

$$[\text{PCB-1254}] = 2.0 \times 10^{-4} [^{90}\text{Sr}] + 0.13$$

are used to estimate the contribution of the ORNL effluent to PCB contamination in fish at the downstream sites, one would conclude that only a small proportion of the PCB contamination of fish at these sites is associated with the ORNL discharge. (For example, a fish containing no ^{90}Sr would be predicted to contain $0.13 \mu\text{g/g}$ PCB-1254, while one with 200 Bq/kg would contain $0.17 \mu\text{g/g}$ PCB-1254). However, the data also suggest that a large population of channel catfish was concentrated in the WOC embayment, where PCB accumulation from the ORNL source is maximized. Subsequent dispersal of these fish to the Clinch River later in the year would undoubtedly make the impact of the WOC discharge on PCB concentrations in Clinch River fish higher than that estimated here. The continuing relationship between ^{90}Sr and PCB-1254 clearly establishes WOC and the WOC embayment as a significant source of contamination in channel catfish in the Clinch River arm of WBR.

4.2.3.3 Temporal changes in polychlorinated biphenyl concentrations in catfish

No increasing or decreasing trends in PCB contamination in catfish over time were obvious. Although PCB concentrations in fish collected from the

WOC embayment and the Clinch River (CRK 32.2) in 1989 were somewhat higher than the concentrations found in fish from the same sites in 1988, they were well within ranges observed previously at these sites (Table 4.7). The differences were not statistically significant at WCK 0.3 but at CRK 32.2, the levels in 1988 were significantly lower than in 1987 and 1989 (ANOVA, Tukey's test on \log_e -transformed data). Year-to-year variation in PCBs in MHR fish was not statistically significant; nor were there significant differences between 1988 and 1989 at CRK 15.0 and PCK 6.9.

The proportion of fish exceeding the FDA limit for PCBs was higher in 1989 than that observed in 1986-88, with 50% of the fish at WCK 0.3 containing more than 2 $\mu\text{g/g}$ PCBs (Table 4.7). One of eight fish exceeded the FDA limit at each of the other sites except MHR, where the proportion was similar to that observed previously. The concentrations of PCBs in channel catfish in the Clinch River arm of WBR and the WOC and Poplar Creek embayments, although below the FDA limit on the average, are within the range used by TDHE to issue a Precautionary Fish Consumption Advisory. The 1989 data indicate that the advisory presently in effect for the Clinch River arm of WBR will likely be continued.

4.2.3.4 White Oak Creek as a source of polychlorinated biphenyls

Concentrations of PCBs in channel catfish did not exhibit a close correspondence with proximity to known PCB sources; thus, mean PCB concentrations at sites impacted by PCB inputs (WOC and Poplar Creek embayments) differed little from those at Clinch River sites downstream. This finding was consistent with observations in

previous years (Loar et al. 1992; Loar 1993a, 1993b) and is probably a consequence of the movement of channel catfish and their PCB-contaminated prey. However, the direct impact of the WOC source was again evident in both the higher incidence of fish containing $\geq 2 \mu\text{g/g}$ PCBs and the higher ratio of PCB-1254 to PCB-1260. Both patterns have been consistently observed in previous years at this site (Loar et al. 1992; Loar 1993a, 1993b).

The difference in PCB concentrations in catfish from MHR and downstream sites in 1989 was higher than that observed in previous years (Table 4.7). Such a pattern implies that downstream sources (WOC and Poplar Creek discharges) contributed a greater fraction of the PCB contamination in catfish than was estimated in previous years (Loar et al. 1992; Loar 1993a, 1993b). A maximum estimate of the impact of the ORNL discharge on this system would be obtained by assuming that all PCBs in channel catfish in excess of 0.28 $\mu\text{g/g}$ come from the WOC source. Using this assumption, ORNL could be judged to contribute approximately 80% of the PCB burden of fish in the WOC embayment and approximately 70% at CRK 32.2. The weak correlation between ^{90}Sr and PCB concentrations in Clinch River catfish suggests that this maximum estimate may be too high. However, this relationship would be expected to be weak, if, as noted in Loar (1993b, Sect. 4.2), the most significant PCB pathway for Clinch River catfish is the ingestion of wide-ranging prey. The observation that PCB concentrations in catfish are not higher in Poplar Creek embayment and the Clinch River downstream from the mouth of Poplar Creek (where fish are exposed to PCBs from both WOC and EFPC/Poplar Creek sources) suggests that the WOC source is at least as important as the

EFPC/Poplar Creek source in contributing to PCB contamination in fish in the Clinch River arm of WBR.

4.2.4 Conclusions

The results of the 1989 PCB monitoring of channel catfish are consistent with the conclusions of the previous 2 years (Loar et al. 1992; Loar 1993a, 1993b). That is, some channel catfish caught by anglers in the Clinch River in the vicinity of WOC are likely to contain PCBs in excess of the 2 $\mu\text{g/g}$ FDA limit, and a significant fraction of the PCB content of those fish originates in the WOC discharge and/or the WOC embayment. Concentrations of PCBs in catfish have not exhibited an increasing or decreasing trend over time. The concentrations of PCBs observed in catfish from the Clinch River and WOC embayment sites in 1989 support the Precautionary Fish Consumption Advisory currently in effect for this reach of WBR.

4.3 FUTURE STUDIES

The presence of significant PCB contamination in biota in WOC, together with evidence that PCBs from the WOC drainage are detectable in channel catfish in nearby reaches of the Clinch River, demonstrates the need for continued routine monitoring of PCB concentrations in fish in the WOC drainage and Clinch River. Such monitoring will ensure protection of the public and establish a baseline against which the effectiveness of future remedial actions can be evaluated.

Annual monitoring of PCBs in sunfish at all sites in the WOC drainage and semiannual monitoring of PCBs in fish at WCK 2.9 and WOL will continue. Annual monitoring of PCBs in channel catfish in the WOC embayment and a nearby reach of the Clinch River will also continue in conjunction with PCB monitoring at sites near ORGDP and funded by the ORGDP and Y-12 Plant BMAPs.

Investigations will continue into the problem of chlordane contamination in WOC. Periodic (at least annual) monitoring for chlordane and PCBs will be continued using caged clams. Catfish collected in the Clinch River as part of the ORNL BMAP will also be analyzed for chlordane.

There is a need to determine the importance of ongoing discharges vs residual contamination as a source of PCB contamination in WOC. The presence of toxic conditions in WOC upstream from WCK 3.5 (Sects. 3.1.5 and 3.2.2) makes the use of biomonitoring with caged clams impractical for this purpose at present. Efforts will be made in 1990 to overcome this problem.

Annual monitoring of metals and pesticides in fish will be conducted at three sites (WOL, WCK 2.9, and MEK 0.2) that are impacted by ORNL operations and correspond to ambient NPDES water quality monitoring stations. Monitoring of nonpesticide semivolatile organics will be conducted annually at these three sites using caged clams rather than fish, which have been used previously. Based on information obtained in the ORGDP and Y-12 Plant BMAPs, these organisms are better indicators of phthalate contamination than fish.

5. BIOLOGICAL INDICATORS OF CONTAMINANT-RELATED STRESS

S. M. Adams

5.1 INTRODUCTION

This report presents the results of the 1988 bioindicator study including information from the 1987 study that is also used in an integrative assessment. This assessment uses two approaches to evaluate the current health status of fish in the WOC system: (1) analysis of functional response groups and (2) analysis of integrative bioindicator groups. The functional group approach involves analysis of each of the five major groups of biotic responses that were measured in fish to determine if differences in fish health exist between sites and if changes in fish health had occurred over time. The integrative bioindicator analysis involves using all the bioindicators together within a multivariate context to investigate holistic responses of fish to stress and to aid in the identification of causative agents or mechanisms that affect the health of fish populations in the streams near ORNL.

5.2 METHODS

5.2.1 Sampling Procedures

Sampling was conducted during fall 1988 at four sites in the WOC watershed: (1) WOC near WCK 3.6 (above the weir at WCK 3.41 but below the ORNL STP), (2) WOC near WCK 3.2 (just below the weir at WCK 3.41), (3) WOC near WCK 2.4 (below the confluence of WOC and Melton Branch), and (4) WOL. Fish were also collected from a reference stream (Hinds Creek) located in Anderson

County between Clinton and Norris, Tennessee. At each of these sites 10–15 adult male bluegill were collected by electroshocking. Blood samples were taken from each fish within 2 min after collection by puncturing the caudal vessels with a 20-gauge needle. Samples of ~0.7 mL were obtained from all fish using unheparinized 3-mL vacutainers (Becton, Dickson, & Co.). Each tube was labeled with a fish identification number and placed in ice for transport to the laboratory.

5.2.2 Analytical Procedures

Total lengths and weights were recorded for fish transported from the field, and observations were also made on the general condition of the fish, such as presence or absence of disease, body and/or mouth sores, external parasites, and general condition. Following sacrifice, the liver and spleen were removed from each fish for further analysis. A 100-mg section of liver for histopathological analysis was placed in a 20-mL scintillation vial with 5 mL of Bouin's fixative. A 300-mg sample of liver was placed in a small plastic bag and immediately frozen in liquid nitrogen for subsequent ribonucleic acid (RNA) and deoxyribonucleic acid (DNA) analysis. The spleen was also removed and weighed to the nearest mg, and a section was placed in Bouin's fixative for subsequent histopathological analysis. The remaining viscera (minus liver and spleen) was excised from the body cavity and the total weight recorded after all food material was

removed from the stomach and intestine. The liver and visceral somatic-indices were calculated as the weights of these respective organs divided by total body weight. Condition factor was calculated as $K = 10^5 W/L^3$, where W is body weight (in grams) and L is total length (in centimeters).

5.2.2.1 Lipid analysis

Following dissection and removal of the critical organs, eight individuals from each site were chosen for lipid analysis. These fish were frozen at -120°C until shipment to a subcontractor who conducted the lipid analyses. Lipid biochemical analysis was performed according to a modification of the Bligh and Dyer (1959) method and using the Iatroscan Analyzer System (Harvey and Patton 1981) for lipid class quantification. The Iatroscan system for lipid analysis combines the resolution capabilities of thin-layer chromatography with the quantitative sensitivity of a flame ionization detector. Lipid analysis for each fish included total lipids (percentage of body weight); triglycerides (percentage of total lipids); sterols, including cholesterol and/or sterol esters; phospholipids; and the two major fractions of phospholipids, phosphatidylcholine (PC) and phosphatidylethanolamine (PE).

5.2.2.2 Serum chemical analysis

Blood collected in the unheparinized tubes was allowed to clot, transferred with Pasteur pipettes to 1.5-mL conical microcentrifuge tubes labeled with the fish identification number, and centrifuged for 3 min in a Beckman Microfuge. The clear supernatant (serum) was drawn off with clean pipettes and transferred to labeled 1-mL conical plastic tubes.

The indicators of organ dysfunction [creatinine, ALT (alanine aminotransferase), and albumin] were analyzed by the methods of Henry et al. (1974), Bergmeyer et al. (1978), and Doumas (1972), respectively. Cholesterol was analyzed by the method of Allain et al. (1974) and triglycerides were analyzed by the procedure of Bucolo and David (1973). Total serum proteins were measured by the Biuret method (NCCLS 1979); the procedures for the assay are described in the Roche Diagnostic Systems (1986) information package. All of these methods are enzymatic assays and the reagents for each assay were obtained from Roche Diagnostic Systems. All serum assays were performed on an automated Centrifugal Fast Analyzer System (Cobas-Fara, Hoffman La Roche Instruments, Inc.). Calibrations were performed using the Roche serum calibrator as the standard and moni-trol Level 1 and 2 (American Dade, Miami, Florida) as internal controls.

5.2.2.3 Ribonucleic acid/deoxyribonucleic acid analysis

A 50–100 mg section of liver was homogenized in ~1 mL of distilled water for 1 min using a Teflon homogenizer while keeping the sample cold. After the homogenate was brought to a final volume of 1.5 mL with distilled water, it was transferred to 1.5-mL microcentrifuge tubes and centrifuged for 2 min in a Beckman microfuge. The RNA content of each sample was analyzed in triplicate by adding to each of three 1.5-mL centrifuge tubes 200 μL of supernatant from the liver homogenate, 1.2 mL of 95% ethanol, 0.035 mL of 2 M sodium acetate, and 0.015 mL of 1 M magnesium acetate. Samples were cooled for 20 min in a refrigerator before centrifuging for 2 min. After the supernatant was decanted, 1 mL of 0.3 M KOH was added to the tubes with the

precipitate and incubated at 37°C in a constant water bath until the pellet dissolved. Each tube then received 0.5 mL of 1.4 *N* perchloric acid before it was cooled for an additional 20 min in the refrigerator. The mixture was centrifuged for 2 min, and the supernatant was recovered in 20-mL scintillation vials. The precipitate was washed once with 1 mL of 0.2 *N* perchloric acid, centrifuged again, and the supernatant was combined with the previous supernatant. Standards were prepared using 1100 µg of RNA and processed in exactly the same manner as the liver samples. Absorbance of the samples and standards was measured at 260 nm, using a Gilford Response Spectrophotometer and a distilled water blank as a reference. Results were expressed as µg RNA/mg wet weight of liver tissue.

For DNA analysis, duplicate samples were prepared by adding 3 mL of 0.2 *M* phosphate buffer at pH 7.0 to 100 µL of the original liver homogenate supernatant. Standards were prepared with 4.6 µg of salmon sperm DNA. The fluorescence was measured using an excitation wavelength of 360 nm and an emission wavelength of 450 nm with a Beckman LS-5 spectrofluorometer. Results were expressed as µg DNA/µg wet weight liver tissue.

5.2.2.4 Histopathological analysis

The following histopathological analyses were performed by the School of Veterinary Medicine, University of California-Davis: (1) percent of tissue occupied by parasites, (2) percent of liver composed of necrotic parenchyma, (3) percent of tissue composed of macrophage aggregates, and (4) percent of

liver occupied by functional parenchyma. These analyses were performed according to the methods of Hinton and Couch (1984).

5.2.2.5 Detoxification enzymes

Microsome isolation. Fish hepatic microsomes were prepared by differential centrifugation (McKee et al. 1983) using several modifications. Fish were sacrificed by severing the spinal cord and the livers were immediately removed and blotted dry. Each liver sample was placed in ice-cold buffer (0.25 *mM* sucrose, 0.1 *M* Tris, pH 7.4). The minced tissues were homogenized in 5 volumes of buffer using a motor-driven Potter-Elvehjem glass and Teflon homogenizer. The homogenates were centrifuged at 3,000 *g** for 10 min and at 10,000 *g* for 20 min, using a J-21B Beckman centrifuge. The resulting supernatants were centrifuged at 105,000 *g* for 60 min in a Beckman L3-50 ultracentrifuge. Microsomal pellets were resuspended in 0.1 *M* Tris buffer, 1 *mM* EDTA, and 20% glycerol at pH 7.4 by sonication with a Braun Sonic 1510 at 50 W for 10 to 15 s. All operations were performed at 0 to 4°C, and microsomes were frozen with liquid nitrogen and stored at -120°C. No significant change in 7-ethoxyresorufin O-deethylase (EROD) activity was detected after 6 months of storage at this temperature. The activity of fish microsomes stored under these conditions have been reported to be stable for a year (Forlin and Anderson 1985).

Enzyme assays. The activity of EROD was measured fluorimetrically at 30°C (Burke and Mayer 1974) and expressed as moles of resorufin per minute per milligram of microsomal protein. The

*An italic lowercase *g* denotes the standard acceleration of gravity ($-9.8 \text{ m}\cdot\text{s}^{-1}\cdot\text{s}^{-1}$).

final reaction buffer contained 80 mM HEPES buffer (pH 7.8), 5 mM magnesium acetate, 1.0 μ M 7-ethoxyresorufin, 250 μ M nicotinamide adenine dinucleotide phosphate (reduced form) (NADPH), and 100 μ M EDTA. The concentration of total protein used in the enzyme assay ranged from 0.2 to 1 mg/mL, depending on the activity of the sample. Proteins were measured by the Bio-Rad (Richmond, California) reagent method (Bradford 1976) on the Centrifugal Fast Analyzer System, using bovine serum albumin as a standard. Cytochrome P-450 and cytochrome b_5 content were each measured by their characteristic oxidized and reduced spectra. Cytochrome P-450 samples were oxidized with carbon monoxide and reduced with sodium dithionite (Johannesen and DePierre 1978). Cytochrome b_5 was reduced with nicotinamide adenine dinucleotide (reduced form) (NADH) and the b_5 assays were conducted prior to the P-450 analysis (Stegeman et al. 1979). NADPH-cytochrome c reductase was assayed spectrophotometrically by the reduction of the electron acceptor cytochrome c with an extinction coefficient of $21.1 \cdot \text{cm}^{-1} \cdot \text{mM}^{-1}$.

5.2.3 Statistical Procedures

An ANOVA was used to test for site and seasonal effects, as well as their interaction effect, on the individual bioindicators. If the ANOVA rejected a multisample null hypothesis of equal means, then Dunnett's test (Zar 1984) was used to test for significant differences in each of the WOC/WOL sites compared to the reference site. The significance level for rejecting the hypothesis of equal means between each of the affected sites and the reference site was set at $p = 0.05$.

To determine the integrated response of fish to the environmental conditions at each sampling site, all the bioindicator

variables were considered jointly within a multivariate context by using canonical discriminant analysis available on SAS (PROC CANDISC, SAS 1985b). This method provides a graphical representation of the positions and orientations of the various integrated site responses relative to each other. In addition, this method derives linear combinations of all variables and has the potential to indicate statistical differences among treatment means even if the single variables do not indicate such differences. A variable selection procedure available in SAS (PROC STEPWISE) was used to identify the variables that contributed most to the discrimination among integrated site responses. This variable selection procedure considered all possible combinations of the observed values and, for any specified subset size, selected those variables having the best discriminating power. Tests for homogeneity of variance of individual response variables between sites were conducted using Levene's test (Sokal and Rohlf 1981), an F-distribution test that compares the ratios of the variances from two independent sample populations.

5.3 RESULTS AND DISCUSSION

Two principal types of analyses were conducted to evaluate the effects of water quality on the health of fish in the WOC system. A spatial effects analysis was conducted to evaluate differences between sites in order to determine the effectiveness of the ORNL NPDES effluent limits in protecting the classified uses of streams in the WOC watershed. An analysis of temporal changes in the overall health of the bluegill sunfish population in WOC and WOL was performed to determine if water quality conditions and the health of the fish populations had changed over time.

5.3.1 Analysis of Spatial Effects

To address spatial effects or differences among sites in fish responses to stress, the measured parameters were grouped into five functional categories. The groups represented indicators of (1) detoxification enzyme induction, (2) organ dysfunction, (3) lipid metabolism, (4) histopathology, and (5) overall fish health or condition. These responses or functional groups reflect gradients of both ecological relevance and time-course of a response to a stressor, such as a contaminant (Fig. 5.1). Those variables in groups 1 and 2 are short-term response indicators and have relatively low ecological relevance, whereas indicators included in groups 3 through 5 are longer-term response variables and are characterized by low toxicological but high ecological relevance (Fig. 5.1).

The relative difference in response of each group of bioindicators between each WOC/WOL site and the reference site is shown in Figs. 5.2 to 5.6 for fish sampled in 1988. In each figure, values above the zero line indicate that the response for fish at a WOC/WOL site was greater than that for fish from the reference site. Values below the zero line indicate that the response in WOC/WOL fish was less than that in reference fish. Asterisks associated with the bars indicate those values that were significantly different ($p < 0.05$) from the reference site.

5.3.1.1 Detoxification enzymes

The activity or levels of liver detoxification enzymes is used to indicate exposure to and possible effects of various xenobiotics, such as PAHs, PCBs, and pesticides (Payne and Penrose 1975). Many of these compounds commonly occur in industrial and municipal effluents and are accumulated within living tissues

(Ahokas et al. 1976; Neff 1978; Brown et al. 1986).

Three of the four detoxification enzymes were significantly higher in fish from WCK 3.6 immediately below the ORNL STP compared to fish from the reference stream. EROD activity, which has generally been one of the best indicators of contaminant exposure in the BMAP studies (Loar 1992, 1993a; Adams et al. 1989), was almost 800% higher at WCK 3.6 than the reference site. Only EROD was elevated at WCK 3.2 but NADH, NADPH, and EROD were all high at WCK 2.4. Only NADH was elevated at WOL, while P450 was significantly lower at this site. The fact that NADH and NADPH levels were significantly elevated at WCK 3.6 and 2.4 but low at WCK 3.2 may possibly indicate that different types and/or concentrations of contaminants are present in the WOC system, especially that reach of stream between WCK 3.6 and WCK 2.4.

5.3.1.2 Organ dysfunction

Serum (plasma) albumin, creatinine, ALT, and protein were used as indicators of organ dysfunction. Serum albumin plays a major role in the binding and transport of insoluble or sparingly soluble compounds, such as fatty acids, bilirubin, and hormones. Other functions of this protein include osmotic regulation and use as a reserve source of protein and amino acid. Serum albumin values can be affected by many types of diseases and physiological disturbances. Levels of this compound were significantly elevated at WCK 3.2 and WOL, possibly indicating some level of liver or kidney damage (Fig. 5.3).

Elevated creatinine levels are typically used as an indicator of kidney damage or malfunction (Tietz 1986). Even though creatinine appeared to be higher at all

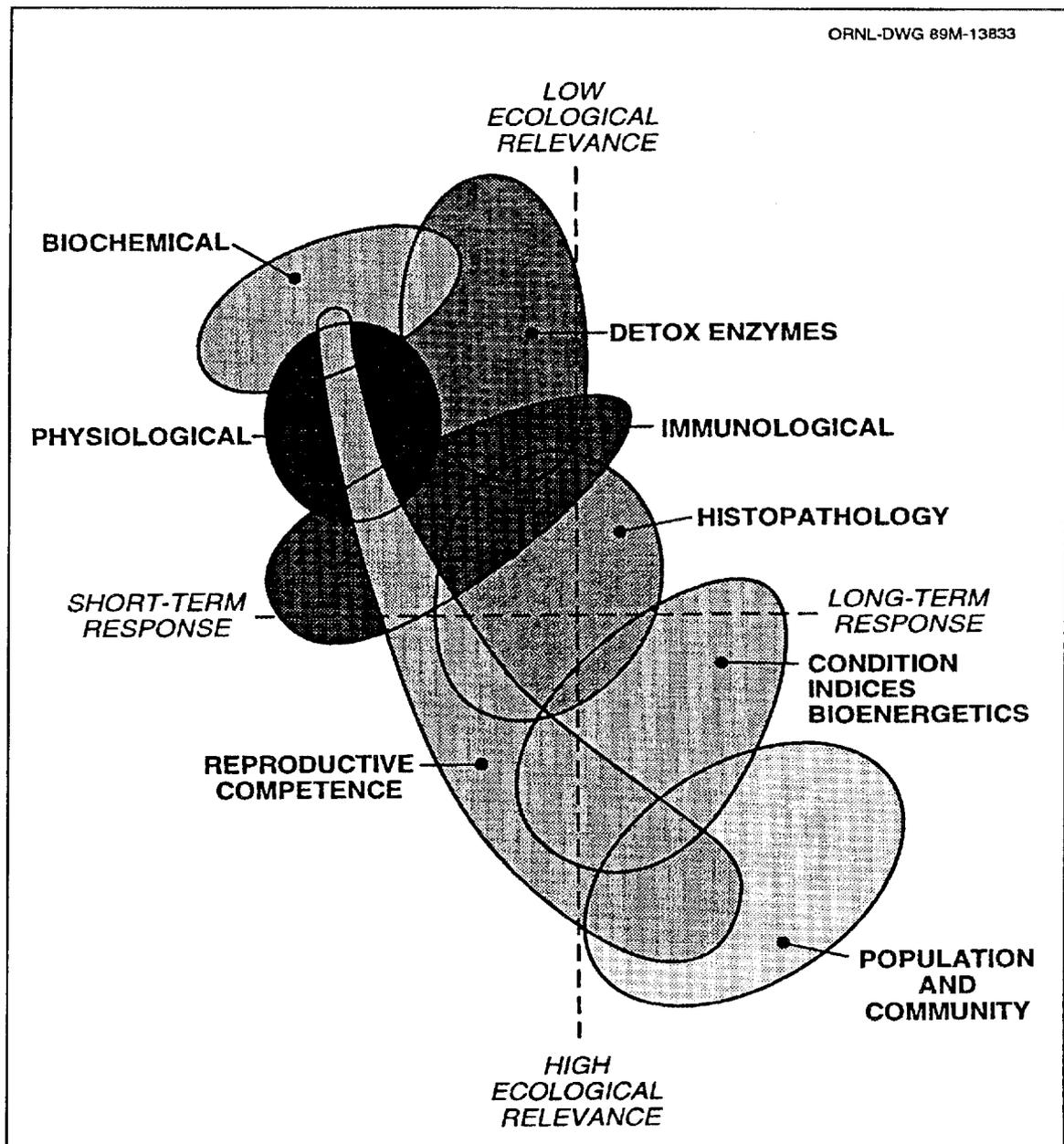


Fig. 5.1. Levels of biological response in fish to contaminant stress, illustrating the continuum of these responses along gradients of response time and ecological relevance.

WOC sites than the reference site, only at WCK 3.2 was this indicator significantly higher (Fig. 5.3). Because both albumin and creatinine were elevated at this site, some physiological impairment to bluegill in this reach of the stream is possible.

The transferase enzyme ALT is generally used as an indicator of liver damage, reflecting cirrhosis or obstructive or obstructive jaundice. Levels of ALT at the WOC/WOL sites, however, appeared to be lower than the reference site,

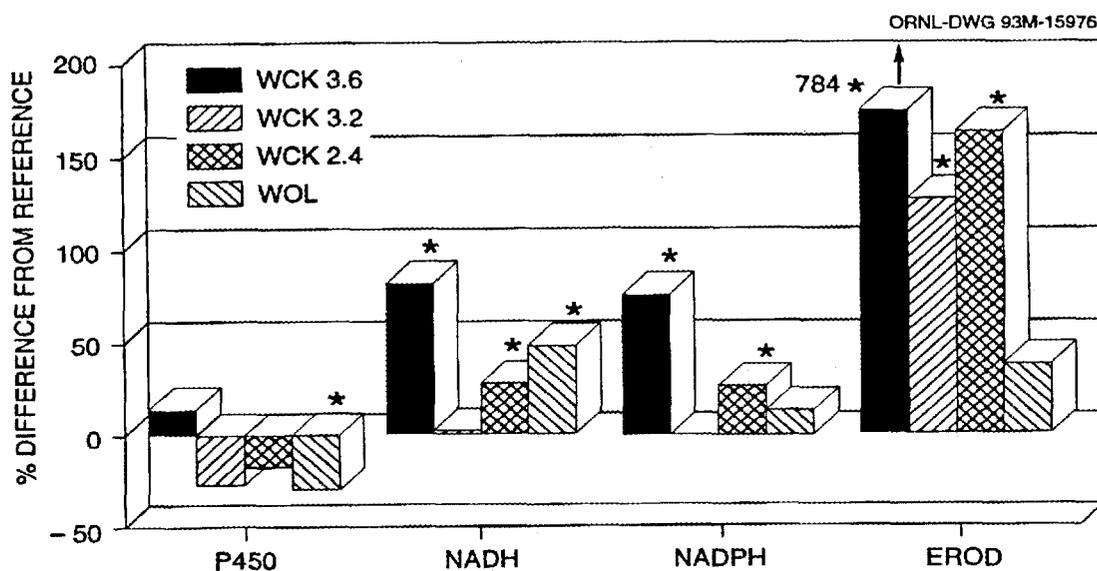


Fig. 5.2. Relative differences in the response of detoxification enzymes for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek. Values above or below the zero line indicate that the response for fish from an affected site was higher or lower, respectively, than the same response for fish from the reference site. Asterisks indicate those values that were significantly different ($p < 0.05$) from the reference site. [Note: EROD = 7-ethoxyresorufin *O*-deethylase, NADH = nicotinamide adenine dinucleotide (reduced form), NADPH = nicotinamide adenine dinucleotide phosphate (reduced form), WCK = White Oak Creek kilometer, and WOL = White Oak Lake.]

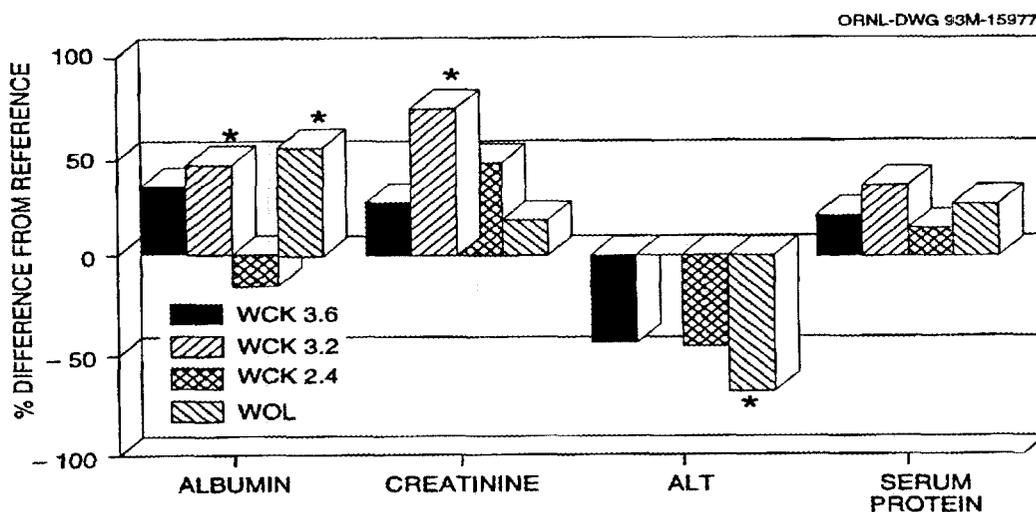


Fig. 5.3. Relative differences in the response of the organ dysfunction indicators for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek. Values above or below the zero line indicate that the response for fish from an affected site was higher or lower, respectively, than the same response for fish from the reference site. Asterisks indicate those values that were significantly different ($p < 0.05$) from the reference site. (Note: ALT = alanine aminotransferase, WCK = White Oak creek kilometer, and WOL = White Oak Lake.)

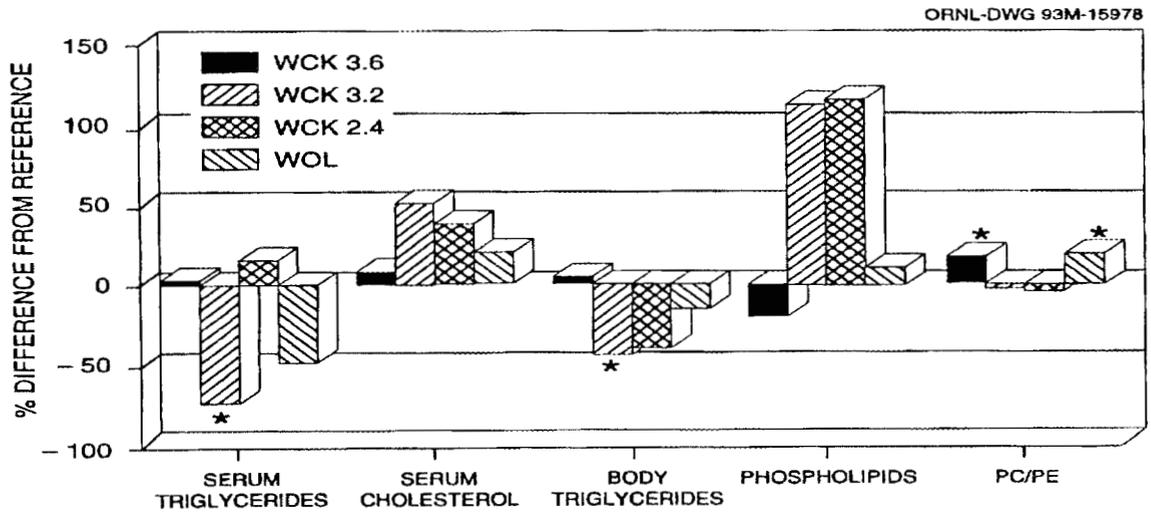


Fig. 5.4. Relative differences in the response of the lipid metabolism parameters for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek. Values above or below the zero line indicate that the response for fish from an affected site was higher or lower, respectively, than the same response for fish from the reference site. Asterisks indicate those values that were significantly different ($p < 0.05$) from the reference site. (Note: PC = phosphatidylcholine, PE = phosphatidylethanolamine, WCK = White Oak Creek kilometer, and WOL = White Oak Lake.)

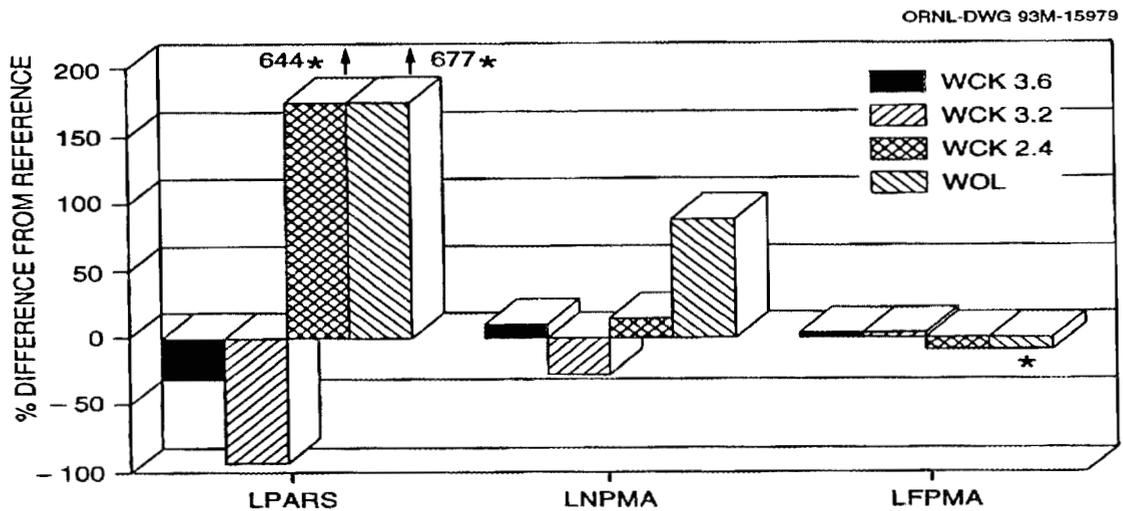


Fig. 5.5. Relative differences in histopathological condition for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek. Values above or below the zero line indicate that the response for fish from an affected site was higher or lower, respectively, than the same response for fish from the reference site. Asterisks indicate those values that were significantly different ($p < 0.05$) from the reference site. (Note: LPARS = liver parasites, LNPMA = necrotic liver parenchyma, and LFPMA = functional parenchyma in liver, WCK = White Oak Creek kilometer, and WOL = White Oak Lake.)

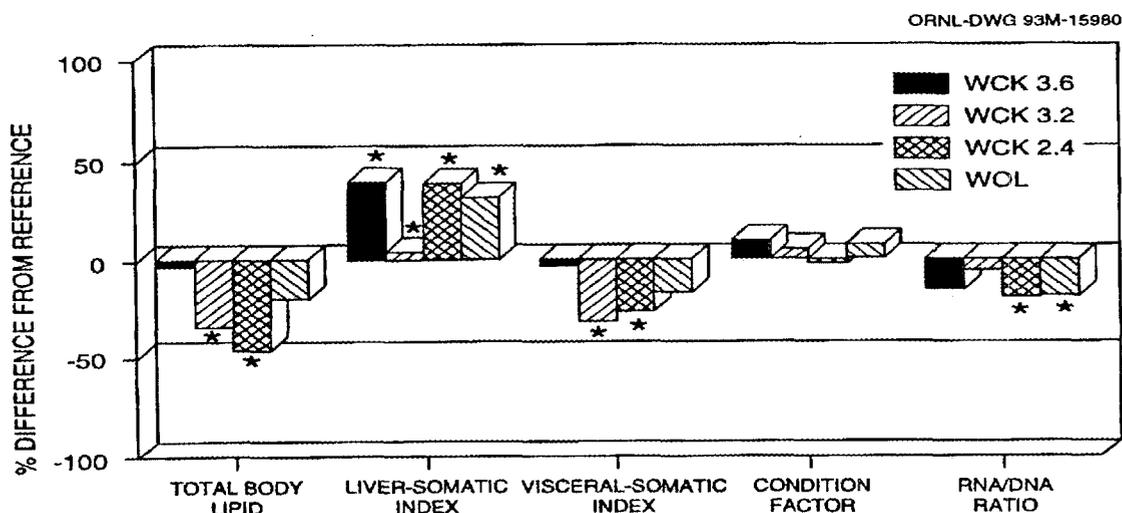


Fig. 5.6. Relative differences in the condition indices for bluegill from each of the White Oak Creek sites and White Oak Lake compared to the reference site, Hinds Creek. Values above or below the zero line indicate that the response for fish from an affected site was higher or lower, respectively, than the same response for fish from the reference site. Asterisks indicate those values that were significantly different ($p < 0.05$) from the reference site. (Note: DNA = deoxyribonucleic acid, RNA = ribonucleic acid, WCK = White Oak Creek kilometer, and WOL = White Oak Lake.)

although only WOL showed a significantly lower ALT level. Vitamin deficiency, in some cases, may result in decreased ALT levels (Bradley et al. 1972).

Fish from all four affected sites appeared to have slightly higher serum protein levels than did the reference fish, suggesting that fish in Hinds Creek were experiencing some level of nutritional stress because of their low protein levels. However, none of the differences between the WOC/WOL and reference sites was statistically significant. Lockhart and Metner (1984) reported depressed serum protein levels in the white sucker (*Catostomus commersoni*) under conditions of low food availability.

5.3.1.3 Lipid metabolism

Indicators of lipid metabolism can reflect both nutritional status and level of

metabolic stress in fish. The condition or status of the lipid pool is important because the vulnerability of an organism to stress depends, in part, on the level of lipid reserves (Shulman 1974; Glebe and Leggett 1981).

Serum triglyceride levels were significantly lower than the reference site only in fish from WCK 3.2 (Fig. 5.4). Levels of serum cholesterol, an indicator of both nutrition and steroid metabolism, appeared elevated at all four WOC sites, but the values were not significantly higher (Fig. 5.4). Total body triglycerides reflect the energy available to an organism for mediating the effects of stress (Lee et al. 1983) and for critical physiological functions, such as growth and gonadal development. Triglycerides also act as energy buffers in periods of food shortages (Adams et al. 1985). Total body triglycerides were also significantly lower at WCK 3.2, and reduced triglycerides may be

indicated at WCK 2.4 and WOL (Fig. 5.4). Slight elevations (though not statistically significant) in serum triglycerides, cholesterol, and total body triglycerides of bluegill from WCK 3.6 could be attributed to their residency in a large pool behind the weir at WCK 3.41. Both primary and secondary production could be relatively high in this environment due to high nutrient loading, thus providing abundant food for fish that results in elevated lipid levels.

Phospholipids, the structural components of lipids, were >100% higher in sunfish from WCK 3.2 and WCK 2.4 compared to fish from the reference stream (Fig. 5.4). Such a pattern is not surprising because phospholipid levels generally show an inverse relationship to triglyceride levels.

The ratio of the two major types of phospholipids (PC and PE) that comprise the cell membrane can reflect membrane integrity. The lipid structure can influence membrane fluidity, enzyme kinetics, and electrical properties (Friedman et al. 1986). The PC/PE ratio was significantly elevated in fish from WCK 3.5 and WOL. Even though little is known about the influence of environmental variables on the PC/PE ratio, observed differences in this ratio could be due to differences in temperatures and/or levels of dissolved solids between sites (Roche et al. 1983).

5.3.1.4 Histopathological condition

Indicators of histopathological condition in WOC fish showed distinct differences compared to the reference fish (Fig. 5.5). The percentage of liver composed of encysted parasites (LPARS) was over 600% higher in fish from WCK 2.4 and WOL than in fish from the reference site. Although levels in fish from WCK 3.6 and WCK 3.2 were lower than the reference site, the difference was not

statistically significant. The percentage of liver tissue occupied by necrotic parenchyma (LNPMA) varied between sites and none of the differences was significant. The apparent increase in necrotic parenchyma at WOL probably reflects the high parasite load in fish at this site.

The percentage of liver actually occupied by functional parenchyma (LFPMA) was slightly higher at WCK 3.6 and WCK 3.2 but significantly lower at WOL. Although the percentage reduction in functional liver tissue in WOL bluegill may not seem dramatic (Fig. 5.5), a 10–15% reduction in liver capacity could have a major effect on the physiological health of an organism. These lower levels of functional liver tissue imply that fish with this condition may have a reduced capacity to (1) produce enzymes for detoxifying contaminants, (2) store important energy reserves (e.g., glycogen and lipids), (3) manufacture vitellogenin necessary for proper egg development, and (4) convert and process protein and lipid compounds into physiologically useful energy.

5.3.1.5 Condition indices

Five condition indices were measured as indicators of the general health of fish. Total body lipid is used to indicate overall fat storage and general nutritional status of fish. Even though all the sites appeared to have lower lipid levels than the reference site, only bluegill from WCK 3.2 and WCK 2.4 had significantly lower levels than bluegill from the reference stream.

The visceral-somatic index (VSI) reflects energy stored as lipids in the mesenteries of the viscera. These lipids are important for long-term energy uses and for gonad maturation (Adams and McLean 1985). As shown in Fig. 5.6, the VSI reflected the level of total body lipids,

with both WCK 3.2 and WCK 2.4 displaying a significantly lower VSI. Lower fat levels in fish from these two sites may indicate that metabolic stress due to contaminant exposure is greater in these fish and/or that less energy is available through the food chain.

If contaminants are having an effect through the food chain, it should be reflected in the quality or quantity or benthic organisms available to fish. The biomass and production of benthic macroinvertebrates (excluding mollusks) are higher near WCK 2.4 and WCK 3.2 than in other sections of WOC; values for these two sites were about average for unimpacted streams of similar size and characteristics (Loar 1993a, Sect. 6.1). In fact, a slight enriching effect from moderate levels of nutrient loading is evident in these two areas of WOC. The primary macroinvertebrates at these WOC sites are the net-building caddisflies, which are a preferred food item of several fish species and particularly sunfish. The fact that food availability is relatively high but indicators of lipid dynamics and growth are low suggests that contaminants could be the primary stressor on the bluegill population in the areas of WCK 2.4 and WCK 3.2. These data are the first definitive demonstration from BMAP of the relative importance of direct contaminant effects and food-chain effects on the physiological stress response of fish.

The liver-somatic index (LSI) reflects both short-term nutritional status and metabolic energy demands (Heidinger and Crawford 1977; Adams and McLean 1985). In addition, the LSI is sensitive to toxicant stress, and liver enlargement due to hyperplasia (increase in cell number) and hypertrophy (increase in cell size) has been reported in fish exposed to toxic compounds (Fletcher et al. 1982; Heath 1987; Addison 1984). This situation may indeed be the case for bluegill in the WOC system because the LSI is significantly

higher at all sites compared to the reference.

The RNA/DNA ratio is used as an indicator of immediate or short-term growth in fish (Bulow 1970; Haines 1973) as well as an indicator of exposure to sublethal concentrations of toxicants (Barron and Adelman 1984). Growth is one of the ultimate indicators of fish health because it integrates all the biotic and abiotic variables acting on an organism and reflects secondary impacts of chronic stress (Waters 1977; Larkin 1978). Reductions in growth, as evidenced by the lower RNA/DNA ratio, are indicated at all sites, but the difference is statistically significant only at WCK 2.4 and WOL. These were also the only two sites with high enough levels of encysted parasites to possibly impair growth.

5.3.1.6 Spatial analysis—conclusions

As evidenced by the response of the detoxification enzymes (indicators of contaminant exposure), organ dysfunction, histopathological and overall body condition, and lipid metabolism indicators, bluegill in the WOC system appear to be experiencing some level of physiological stress. The short-term effects of this contaminant exposure are reflected in liver and possibly kidney dysfunction at some sites. Longer-term effects appear to be manifested as effects on histopathological condition, lipid metabolism, and overall condition or health.

The effects of water quality on the physiological condition or health of fish may not be the same at every site, however. Contaminant exposure is higher, for example, at the three WOC sites than in WOL (Fig. 5.2). Possible liver and kidney damage appear to be greatest at WCK 3.2 and WOL. Parasite loads and organ damage due to parasites is highest at WCK 2.4 and WOL, while lipid

metabolism seems primarily impaired in fish at WCK 3.2. Our general health and condition indices demonstrate that bluegill are under greater stress at WCK 3.2 and WCK 2.4 than other sites in the WOC system.

The absence of a distinct downstream gradient for any of the bioindicators and high between-site variability in the various indicators (Figs. 5.2 to 5.6) preclude any conclusion about the primary sources of stress on the bluegill population in WOC and WOL. For example, some indicators demonstrate a significant response at WCK 3.6, no response at WCK 3.2, but again a response at WCK 2.4. This type of pattern may be indicative of the different kinds or amounts of contaminants that are impacting the fish populations in WOC. The ORNL complex, including the STP, may be one source of stress, as evidenced by the elevated enzyme levels in fish at WCK 3.6. However, other sources, such as the waste disposal areas adjacent to WOC (Fig. 2.2), can not be excluded as possible contributors of contaminant stress.

5.3.2 Temporal Analysis

The purpose of the temporal analysis was to determine if any changes had occurred in the health of fish in the WOC system since the last sampling period in fall 1987. Temporal changes in each indicator/site combination were determined by first applying a two-way ANOVA procedure for season (fall 1987 and fall 1988), site, and season \times site interaction effects. If the interaction between seasons was significant ($p < 0.1$), then a t-test (PROC TTEST, SAS 1985b) was invoked to determine which sites were significantly different for each indicator.

Many of the bioindicators demonstrated a temporal change in values from 1987 to 1988 (Table 5.1). Several indicators did not change significantly at

any of the sites (condition factor, visceral-somatic index, albumin, total lipids, liver parasites, and NADPH), while others showed a significant increase or decrease. If the temporal change in an indicator was equal at all sites, including the reference (e.g., serum protein), it was assumed that the environmental factors operating on the fish populations were the same at all sites and, therefore, the observed change was not due to contaminant stress. Indicators that changed equally at all sites were serum protein, serum cholesterol and triglycerides, PC/PE, body triglycerides, and creatinine. The only variables that changed significantly at the WCK 3.2 and WOL sites and did not change at the reference site were the LSI, LNPMA, EROD, and P-450. All of these variables are related directly to indicators of contaminant exposure. The EROD and P-450 enzymes decreased at WCK 3.2 and WOL, indicating that the level of contaminant exposure at these two sites had decreased over the year. As discussed in Sect. 5.3.1.5, the LSI typically increases under toxicant stress. However, in fish from WCK 3.6 and WOL, the LSI decreased, indicating a reduction in contaminant exposure at these sites. The decrease in the percentage of necrotic parenchyma at WCK 3.6 is an additional indicator that contaminant exposure may have decreased at this site between 1987 and 1988.

The RNA/DNA ratio, which reflects short-term growth, increased significantly at all sites except WCK 3.2. Increases in the ratio at WCK 3.6 and WOL might be explained by a decrease in contaminant stress, resulting in more physiologically useful energy being available for anabolic processes, such as growth, instead of being used in metabolic processes. The significant increase in the RNA/DNA ratio at the reference site (along with increased lipid indicators) from 1987 to 1988 may be related to the severe drought conditions during 1987 that extended well into the

Table 5.1. Temporal changes in bioindicator responses of bluegill over the period fall 1987–fall 1988 at two sites in White Oak Creek, White Oak Lake, and the reference stream^a

Response variable	Temporal changes			
	Reference	WCK ^b 3.2	WCK 3.6	WOL ^c
Total body lipid	0	0	0	0
Liver-somatic index	0	0	-	-
Visceral-somatic index	0	0	0	0
Condition factor	0	0	0	0
RNA/DNA ^d	+	0	+	+
Albumin	0	0	0	0
Serum protein	-	-	-	-
Serum triglycerides	+	+	+	+
Serum cholesterol	+	+	+	+
Body triglycerides	+	+	+	+
PC/PE ^e	-	-	-	-
Cytochrome P-450	0	-	0	-
EROD ^f	0	-	0	-
NADPH ^g	0	0	0	0
Liver parasites	0	0	0	0
Necrotic tissue in liver	0	0	-	0
Creatinine	-	-	-	-

^aA significant increase in a response over this period is indicated by a plus (+), a significant decrease by a minus (-), and no significant change by a zero (0); $p > 0.05$.

^bWCK = White Oak Creek kilometer.

^cWOL = White Oak lake.

^dRNA/DNA = ribonucleic acid/deoxyribonucleic acid.

^ePC/PE = phosphatidylcholine/phosphatidylethanoamine.

^fEROD = 7-ethoxyresorufin *O*-deethylase.

^gNADPH = nicotinamide adenine dinucleotide phosphate (reduced form).

fall. Low stream flows at the reference site in 1987 could have reduced the availability of benthic invertebrates for consumption by fish (Larimore et al. 1959), thus providing less energy for critical physiological processes, such as growth. Drought conditions were somewhat less severe in 1988 (e.g., Table 2.1), possibly resulting in higher food availability and fish growth in some streams.

In summary, several variables changed equally over the year at all sampling sites (including the reference), indicating that changes in these particular variables could have been due primarily to normal

fluctuations in environmental conditions and not to changes in contaminant levels in WOC. Although the levels of two detoxification enzymes decreased significantly in WOC (WCK 3.2) and WOL, possibly reflecting decreased contaminant availability at these sites over the year, significant improvements in other indicators of fish health were not evident. Lack of concomitant responses at higher levels of biological organization suggests that the changes observed in these two enzymes (and related parameters, such as LSI and LNPMA) were temporary and not of sufficient duration and/or magnitude to

significantly affect such important physiological processes as lipid metabolism and growth.

5.3.3 Integrated Site Analysis

An informative approach for determining the overall response of fish to stress is canonical discriminant analysis. This method includes all the bioindicators within a multivariate context and provides a graphical representation of the positions and orientations of the site responses relative to each other. This integrated approach to evaluating the effects of water quality on fish populations has the primary advantage of providing an integrative assessment of the health of fish based on multiple rather than individual indicators.

The most obvious feature of the site responses shown in Fig. 5.7 is that the integrated response of fish from the reference stream was distinct from that of fish at each of the WOC/WOL sites. As indicated by the differences in the linear distances between the centers of the site means, bluegill at WCK 2.4 were most similar, and those at WCK 3.6 were least similar, to bluegill from the reference stream.

5.3.3.1 Multivariate selection

A multivariate selection procedure was used to identify those variables that best discriminated between the health status of fish at the various sites. The variables that are most important in discriminating among sites are also indicated on Fig. 5.7. These ten variables consist of representative indicators from four of the five functional groups (see Sect. 5.3), including three detoxification enzymes, three organ dysfunction indicators, three condition indices, and one lipid metabolism indicator. This analysis

illustrates the importance of including bioindicators at several levels of biological organization when evaluating the integrated responses of fish to environmental stress.

Finally, comparison tests were performed on pairs of sites to determine the best indicators for discriminating between sites. Comparisons between the affected sites and the reference site demonstrated that indicators of organ dysfunction and detoxification enzymes were the primary variables that discriminated between the reference fish and WOL fish (Table 5.2). The best indicators for distinguishing between the reference site and the two lower WOC sites (WCK 3.2 and 2.4) were measures of organ dysfunction and condition indices. Detoxification enzymes, lipid metabolism, and condition indices, however, were the principal variables for distinguishing between the reference and WCK 3.6 sites (Table 5.2).

As was also demonstrated by the spatial effects analysis (Sect. 5.3.1), the bioindicators responsible for the observed stress responses differed among sites. These differential response patterns may indicate that different types or concentrations of contaminants are impacting the fish populations in the WOC system. The ORNL complex, including the STP, may be responsible for a specific group or level of contaminant stressors, while non-point sources adjacent to WOC (i.e., the waste disposal areas) could be responsible for a different group of contaminants.

5.3.4 Summary and Synthesis

Several significant conclusions resulted from the comprehensive analysis of the 1987 and 1988 bioindicator data sets. First, bluegill in the WOC system appear to be some level of physiological stress. On a

ORNL-DWG 93M-15981

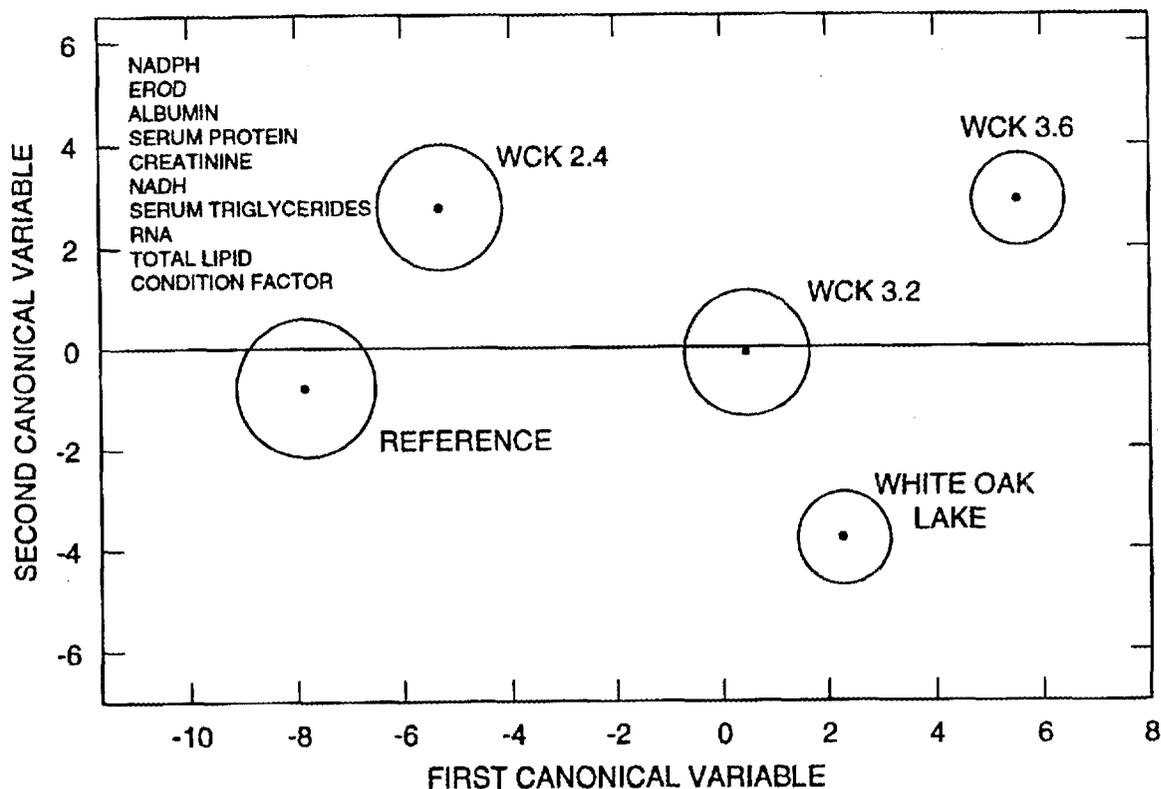


Fig. 5.7. Segregation of health responses for bluegill from three sites in White Oak Creek, White Oak Lake, and the reference stream (Hinds Creek), using bioindicators measured in fall 1988. Circles represent site means and the 95% confidence radii of the site means. The variables used for discriminating among these sites included nicotinamide adenine dinucleotide phosphate (reduced form), 7-ethoxyresorufin *O*-deethylase, albumin, serum protein, creatinine, nicotinamide adenine dinucleotide (reduced form), serum triglycerides, ribonucleic acid, and total lipid condition factor.

short-term basis, the effects of contaminant exposure are reflected in liver and possibly kidney dysfunction at some sites. Longer-term stress is manifested by changes in histopathological condition, lipid metabolism, and overall condition or health. Second, the effects of water quality on the physiological condition or health of fish may not be the same at every site. Contaminant exposure is higher, for example, at the three WOC sites than in WOL. The general health and condition indices demonstrate that environmental problems are more severe at WCK 3.2 and

WCK 2.4 than at the other sites. However, due to a lack of a distinct downstream gradient in any of the bioindicators and the variability between sites, no conclusions can be made at this time regarding the primary sources of stress on the bluegill population in WOC/WOL. Third, associated data from the benthic macroinvertebrate study corroborate physiological evidence that the stress responses observed in bluegill from WCK 2.4 and WCK 3.2 result directly from contaminant exposure and not indirectly through the food chain. Fourth, the levels

Table 5.2. Importance of each bioindicator in discriminating between the integrated responses of bluegill for various combinations of sites^a

Functional group/ bioindicator	WOL ^b reference	WCK ^c 2.4 reference	WCK 3.2 reference	WCK 3.6 reference	WCK 2.4 WCK 3.2	WCK 2.4 WCK 3.6
Organ dysfunction						
Albumin	1	2				
Serum protein	2	3			4	4
Creatinine			1	4	3	
Detoxification enzymes						
NADH ^d		1		9		
NADPH ^e				2	1	
P-450						2
Microsomal protein				1		5
Lipid metabolism						
Serum triglycerides			4	6		6
Phospholipids PC/PE ^f				3		3
Body triglycerides				8	4	
Condition indices						
Liver-somatic index		4	5			
RNA/DNA ^g			3	5	2	
Condition factor				7		1
Total body lipids				10		
Visceral-somatic index			2			

^aThe importance of a bioindicator in discriminating between sites decreases as the number increases; 1 = high discriminating power, 10 = low discriminating power. Those bioindicators that were not important in discriminating between sites are not listed.

^bWOL = White Oak Lake.

^cWCK = White Oak Creek kilometer.

^dNADH = nicotinamide adenine dinucleotide (reduced form).

^eNADPH = nicotinamide adenine dinucleotide phosphate (reduced form).

^fPC/PE = phosphatidylcholine/phosphatidylethanoamine.

^gRNA/DNA = ribonucleic acid/deoxyribonucleic acid.

of two detoxification enzymes decreased significantly in WOC and WOL from 1987 to 1988, suggesting that contaminant exposure also decreased in these systems

over this period. Fifth, the integrated physiological response of fish from the reference site was significantly different from that of fish from each of the affected

sites. Bluegill from WCK 3.6 were least similar to those from the reference stream. Finally, the studies to date have demonstrated the importance of incorporating bioindicators representative of several levels of biological organization in biological monitoring programs. Such an integration is needed to effectively evaluate the effects of contaminant-related stress on fish populations.

5.4 FUTURE STUDIES

The annual bioindicator study of fish in the WOC system will continue with one major change that was initiated during the 1989 sampling period. Instead of evaluating several sites in WOC, sampling efforts will be focused on the reach of stream at WCK 2.4. Based on the results of the 1988 study, fish in lower WOC appear to reflect the condition of fish in the WOC system. In addition, fish in this reach of stream probably integrate the effects of all upstream contaminant inputs. Site WCK 2.4 is downstream of the discharges from the ORNL complex, including ORNL facilities in Melton Valley, and is below much of the contaminant input from waste disposal areas bordering the WOC. Sampling of WOL will be discontinued because the ecological and limnological conditions in WOL and WOC are substantially different

due to the lentic vs lotic nature of the two systems respectively. Therefore, stress responses of fish from these two environments cannot realistically be compared.

In addition to reducing the number of sites routinely sampled, another reference site (Paint Rock Creek) was added in 1989. The additional reference site should provide more statistical power for comparisons between WOC and less impacted streams.

Reproductive studies were initiated in 1989 to assess the effects of contaminant stress on reproductive success. Reproductive competence will be evaluated based on measurements of gonad condition (e.g., quality and quantity of eggs), sizes and timing of clutches, hormone levels, and gonad histology.

Finally, a new system of evaluating fish health is currently being developed to supplement the quantitative bioindicator studies. This system, which provides a general health profile of a fish population, can detect departures from normal in growth, bioenergetic state, general homeostasis, nutritional status, and presence of infectious agents. This evaluation will be used to help test the hypotheses that bioindicator responses of fish result directly from contaminant exposure and are not mediated indirectly through the food chain.

6. INSTREAM ECOLOGICAL MONITORING

M. G. Ryon, E. M. Schilling, and J. G. Smith

The objectives of the instream ecological monitoring task (Task 4 of BMAP) are to (1) characterize spatial and temporal patterns in the distribution and abundance of the benthos and fish populations in the WOC watershed; (2) identify contaminant sources that adversely affect stream biota, including differentiation between point sources and non-point (or area) sources, wherever possible; and (3) monitor these populations and evaluate the future effects on community structure and function from operation of new wastewater treatment facilities, from improvements in waste management operations, and from implementation of remedial actions directed at area source control. This task consists of three components: (1) benthic invertebrate studies (Subtask 6a), (2) fish population studies (Subtask 6b), and (3) evaluation of biotic changes (Subtask 6c). Results to date of these studies are presented in Sects. 6.1-6.3.

6.1 BENTHIC MACROINVERTEBRATES

6.1.1 Introduction

During the initial year of BMAP (May 1986-April 1987), the benthic invertebrate communities of the lower reaches of the streams in the WOC watershed exhibited characteristics indicative of degraded water quality (Smith 1993a, 1993b). These impacted downstream reaches were typically characterized by low taxonomic diversity and richness and the absence or low relative abundance of major taxonomic

groups that are intolerant of pollution. During the second year of BMAP, the benthic invertebrate studies continued as in the first year with the specific objectives of (1) further characterizing the benthic invertebrate communities in the WOC watershed and documenting the impacts on these communities from past and current ORNL operations, and (2) continuing efforts to identify casual factors responsible for the observed adverse impacts. This report includes results of only the samples collected from WOC and tributaries in May and June 1987; thus, conclusions based on these data should be considered tentative. Samples collected during the first year of the ORNL BMAP were processed by TVA staff. During mid-1988, however, TVA implemented a reduction-in-force that eventually resulted in the loss of all personnel involved in the processing of the ORNL benthos samples by late 1988. Although a new subcontract was established in mid-April 1989 with JAYCOR, almost a year had been lost before the processing of benthos samples resumed.

6.1.2 Materials and Methods

Benthic macroinvertebrates were sampled at approximately monthly intervals from May 1987 through April 1988, from 15 stream sites in the WOC watershed (Fig. 2.2); two transects were sampled in WOL at approximately bimonthly intervals. In each stream, the uppermost site served as a reference site. In addition, a single site on Brushy Fork at Brushy Fork kilometer (BFK) 7.6, a primary reference

site for benthic invertebrate studies associated with the Y-12 Plant BMAP (Smith 1992a), was also used as a reference site for WCK 2.3. This site is located just north of Oak Ridge near the Marlowe community. Due to the change in subcontractors, however, the data for WOL and Brushy Fork were not available for this report.

Three randomly selected benthic macroinvertebrate samples were collected from riffles at each stream site in the WOC watershed with a Surber stream bottom sampler (0.09 m² or 1 ft²; 363- μ m-mesh net). Samples were placed in pre-labeled glass jars and preserved in 80% ethanol; the ethanol was replaced with fresh ethanol within 1 week. The laboratory procedures used to process these samples are described in Smith (1993a).

Various supplemental information was also recorded at the time of sampling. Water temperature, specific conductance, pH, and dissolved oxygen were measured with an Horiba Model U7 Water Quality Checker. Water depth; location within the riffle area (distance from permanent headstages on the stream bank); relative current velocity (very slow, slow, moderate, or fast); and substrate type, using a modified Wentworth particle-size scale (Table 2.10 in Loar 1993a), were recorded for each sample.

All statistical analyses were done using the SAS (SAS 1985a, 1985b). The Shannon-Wiener index (H') was used to calculate the taxonomic diversity of benthic macroinvertebrates at each site (Pielou 1977):

$$H' = - \sum p_j \log_2 p_j ,$$

where p_j is the proportion of the benthic invertebrate community made up by species j . H' values (\log_2) of ≥ 3 are generally found in areas of clean water, while values of 1–3 are found in areas of

moderate pollution, and values of < 1 are found in heavily polluted water (Platts et al. 1983).

Because only 2 consecutive months of data were used, statistical comparisons were not made between sites; statistical comparisons of spatial trends with a data set that is small and seasonally limited could produce misleading results. However, to examine trends that may be indicative of temporal recovery, statistical comparisons were made of differences between Years 1 and 2 (May and June only) at each site for density; number of taxa per sample (taxonomic richness); number of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) taxa per sample (EPT richness); and diversity. Before statistical comparisons were made, data were transformed [$\log_{10}(X+1)$, where X = individual values for density, biomass, richness, etc.] (Elliott 1977). Between-year comparisons were made with a one-way ANOVA with year as the main effect.

6.1.3 Results

6.1.3.1 Taxonomic composition

A checklist of the benthic invertebrates collected from streams in the WOC watershed in May and June 1987 is presented in Appendix C, Table C.1. Over 114 taxa were collected in quantitative samples, with the class Insecta represented by the greatest number of taxa (91). There were 9 orders of insects collected, with the Diptera (true flies, 23 taxa), Trichoptera (caddisflies, 19 taxa), Ephemeroptera (mayflies, 13 taxa), and Plecoptera (stoneflies, 11 taxa) having the greatest number of representative taxa. Two orders of insects were represented only by taxa that are semiaquatic, including the Collembola and Hemiptera. These taxa live on the surface of the water and are

collected only when they swim or “hop” into the collection net of the sampler. Because they are surface dwellers, they are not as likely to be affected by poor water quality as those species that live beneath the water surface. Plecoptera were conspicuously absent from samples taken from FFK 0.2, and Ephemeroptera were absent from samples collected from NTK 0.2 and WCK 2.9. No Ephemeroptera, Plecoptera, or Trichoptera were collected at WCK 3.9.

The non-insect taxa collected included six Gastropoda (snails); three Bivalvia (mussels); eight Crustacea, including Cladocera (water fleas), Copepoda (copepods), Ostracoda (seed shrimps), Isopoda (aquatic sow bugs), Amphipoda (sideswimmers, two taxa), Decapoda (crayfish), and Hydracarina (water mites); and one each of Coelenterata (hydras), Turbellaria (flatworms), Nematoda (roundworms), Tardigrada (water bears), and Oligochaeta (aquatic earthworms). Six of these groups are comprised of taxa that are typically considered microinvertebrates, including Coelenterata, Tardigrada, Cladocera, Copepoda, Ostracoda, and Hydracarina. Because of their small size, collecting and finding them in samples with the equipment and procedures used in this study would be sporadic at best.

6.1.3.2 Density

Mean monthly density of benthic invertebrates at each stream site in the WOC watershed is presented in Table 6.1. Considerable differences occurred in density between the reference sites and their respective downstream sites. There were no consistent patterns of difference between the references and their downstream sites, with some nonreference sites having densities higher than their reference site. However, some sites did exhibit striking differences, including

Fifth Creek, where density at the reference site (FFK 1.0) was 5.4 and 9.1 times greater than at FFK 0.2 in May and June, respectively, and WOC where the density in June at the reference site (WCK 6.8) was 60.9 and 6.1 times greater than the density at WCK 3.9 and WCK 5.1 respectively.

A comparison of the mean density in May and June of 1986 and 1987 is presented in Fig 6.1. Densities for these 2 months in 1987 were significantly greater than in 1986 at all sites (see Fig. 6.1 for levels of statistical significance). Most notable was the increase during 1987 at FFK 0.2, WCK 3.4, and WCK 3.9, where their respective mean densities were approximately 45, 20, and 32 times higher than in 1986.

6.1.3.3 Dominant taxa

Considerable differences in taxonomic composition existed between sites. Further insight into spatial and temporal differences of benthic invertebrate communities may be gained from an understanding of the taxonomic groups that dominate, because many of these differences are usually due to a few dominant taxa. (For the benthic invertebrate section of this report, the term dominant is used synonymously with numerical dominance; taxa were considered numerically dominant if they were collected at 50% or more of the study sites and comprised 10% or more of the average density at two or more sites.) Within the WOC watershed, eight major taxonomic groups were found, including Chironomidae, Diptera other than Chironomidae, Coleoptera, Oligochaeta, Isopoda, Ephemeroptera, Plecoptera, and Trichoptera (Table 6.2).

Chironomids were consistently the most dominant invertebrates at all sites in WOC watershed (Table 6.2). They

Table 6.1. Density, taxonomic richness, mean Ephemeroptera, Plecoptera, and Trichoptera richness, and taxonomic diversity of benthic macroinvertebrates in White Oak Creek watershed, May and June 1987^a

Site ^b	Density (Individuals/0.1m ²)		Richness (Taxa/sample)		EPT ^c Richness (EPT taxa/sample)		Diversity (H1)	
	May	June	May	June	May	June	May	June
FCK 0.1	1018.3 ± 386.7	237.9 ± 84.5	14 ± 1.8	17 ± 2.6	3 ± 0.3	2 ± 0.3	1.2 ± 0.05	2.6 ± 0.05
FCK 0.8	975.6 ± 217.9	439.9 ± 120.5	23 ± 1.5	21 ± 2.3	8 ± 0.3	6 ± 0.9	3.2 ± 0.05	3.1 ± 0.1
FFK 0.2	241.1 ± 66.0	87.5 ± 24.1	12 ± 2.7	9 ± 1.3	1 ± 1.0	1 ± 0.7	1.2 ± 0.2	1.8 ± 0.07
FFK 1.0	1308.6 ± 106.7	792.6 ± 288.2	31 ± 0.3	27 ± 3.2	18 ± 1.5	14 ± 2.1	3.0 ± 0.11	3.4 ± 0.05
MEK 0.6	1232.9 ± 210.4	289.2 ± 150.1	19 ± 0.3	17 ± 3.8	7 ± 0.3	4 ± 1.2	1.0 ± 0.17	2.4 ± 0.39
MEK 1.2	816.3 ± 146.7	209.9 ± 62.0	14 ± 2.4	13 ± 1.3	4 ± 1.2	4 ± 0.7	0.9 ± 0.2	2.3 ± 0.12
MEK 2.1	302.8 ± 91.5	190.2 ± 23.7	18 ± 0.9	19 ± 3.5	7 ± 1.2	4 ± 1.2	2.7 ± 0.25	2.5 ± 0.5
NTK 0.2	701.1 ± 289.1	455.0 ± 86.1	16 ± 3.2	20 ± 1.8	4 ± 1.7	3 ± 0.0	2.3 ± 0.07	2.9 ± 0.17
NTK 1.0	318.3 ± 89.0	286.3 ± 15.7	24 ± 2.2	23 ± 4.3	11 ± 0.7	6 ± 0.7	3.4 ± 0.09	2.4 ± 0.32
WCK 2.3	1023.0 ± 79.5	628.6 ± 173.8	15 ± 1.5	17 ± 0.3	6 ± 0.7	4 ± 0.6	2.1 ± 0.23	2.3 ± 0.27
WCK 2.9	836.7 ± 225.0	245.4 ± 34.9	17 ± 3.6	14 ± 1.2	4 ± 0.9	3 ± 0.6	1.9 ± 0.05	2.8 ± 0.08
WCK 3.4	1329.0 ± 227.9	985.3 ± 200.8	14 ± 1.2	24 ± 1.9	2 ± 0.3	4 ± 0.9	0.9 ± 0.7	2.7 ± 0.14
WCK 3.9	753.1 ± 272.4	16.1 ± 6.5	7 ± 1.2	3 ± 0.6	0 ± 0.0	0 ± 0.0	1.0 ± 0.05	1.2 ± 0.27
WCK 5.1	837.5 ± 492.2	160.7 ± 37.3	10 ± 3.5	12 ± 2.1	3 ± 1.2	2 ± 0.3	0.7 ± 0.17	2.0 ± 0.39
WCK 6.8	700.0 ± 85.6	979.9 ± 472.0	25 ± 4.8	25 ± 2.4	13 ± 2.4	10 ± 1.9	2.5 ± 0.12	2.2 ± 0.13

^aValues are means ± 1 standard error.

^bFCK = First Creek kilometer, FFK = Fifth Creek kilometer, MEK = Melton Branch kilometer, NTK = Northwest Tributary kilometer, and WCK = White Oak Creek kilometer.

^cEPT = Ephemeroptera, Plecoptera, and Trichoptera.

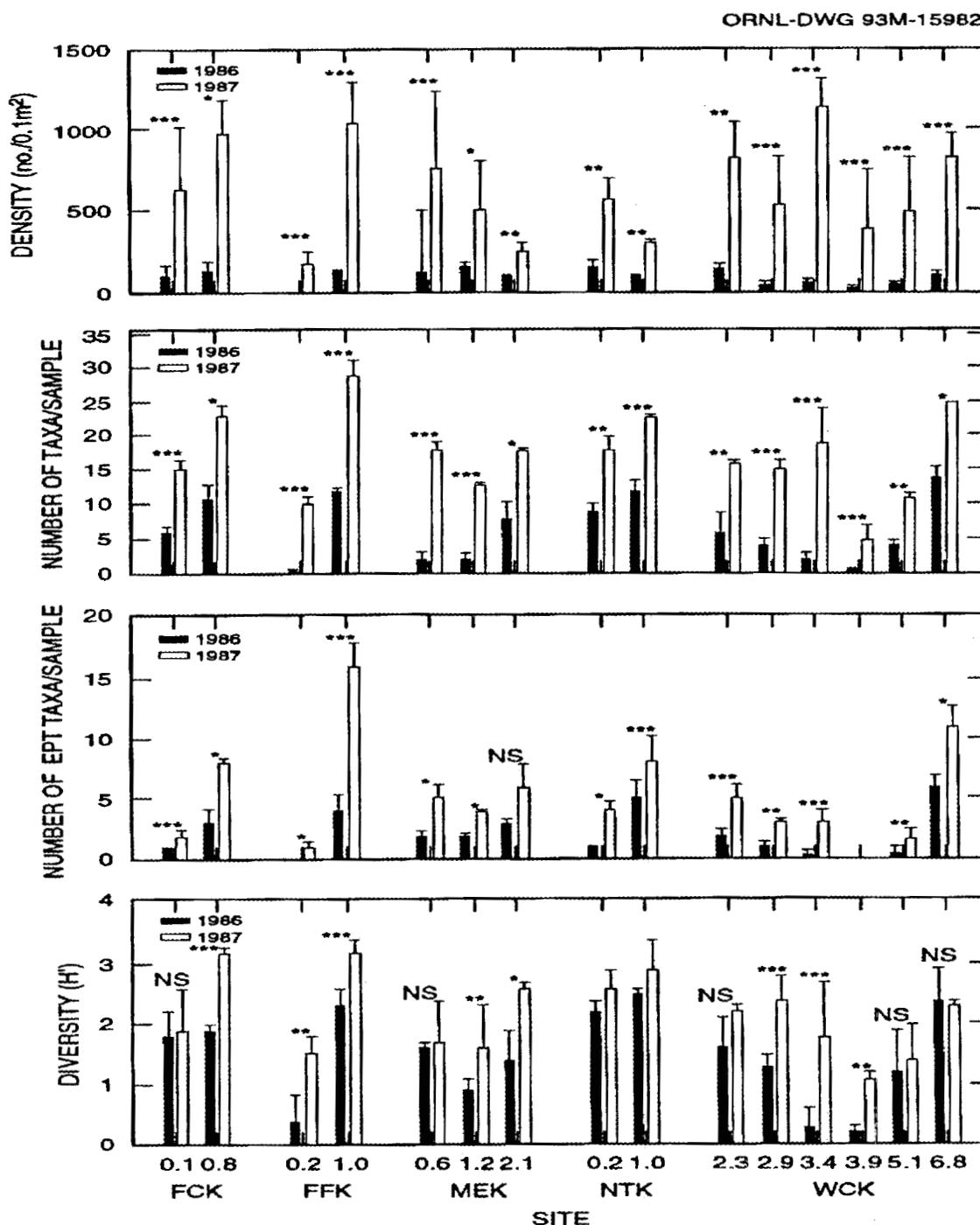


Fig. 6.1. Mean density, number of taxa/sample (taxonomic richness), number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa/sample (EPT richness), and taxonomic diversity for benthic macroinvertebrates in the White Oak Creek watershed for the months of May and June in 1986 and 1987. Results for the between-year comparisons with an analysis of variance are given above the bars for each site and are abbreviated as follows: NS = no significant differences detected at $p = 0.05$; * = $p < 0.05$; ** = $p < 0.01$; *** $p < 0.001$. (Note: FCK = First Creek kilometer, FFK = Fifth Creek kilometer, MEK = Melton Branch kilometer, NTK = Northwest Tributary kilometer, and WCK = White Oak Creek kilometer.)

Table 6.2. Relative density of dominant benthic invertebrate taxa in the White Oak Creek watershed, May and June 1987^a

Site ^b	Relative density (%)							
	Chironomidae	Diptera	Coleoptera	Oligochaeta	Isopoda	Ephemeroptera	Plecoptera	Trichoptera
FCK 0.1	66 ± 15.1	11 ± 3.9	7 ± 5.1	3 ± 0.5	<1 ± 0.3	11 ± 3.9	<1 ± 0.1	3 ± 0.6
FCK 0.8	21 ± 5.6	8 ± 5.5	6 ± 1.0	2 ± 0.6	<1 ± 0.3	20 ± 3.7	4 ± 0.5	3 ± 0.2
FFK 0.2	65 ± 2.9	4 ± 1.6	2 ± 0.1	19 ± 4.3	0	<1 ± 0.1	0	<1 ± 0.1
FFK 1.0	25 ± 6.1	9 ± 1.6	9 ± 0.8	3 ± 1.8	22 ± 1.5	6 ± 0.6	12 ± 0.1	11 ± 4.5
MEK 0.6	67 ± 18.8	3 ± 1.2	3 ± 2.0	7 ± 6.1	<1 ± 0.03	2 ± 1.5	<1 ± 0.4	9 ± 2.9
MEK 1.2	54 ± 28.1	8 ± 7.0	<1 ± 0.2	16 ± 2.2	0	1 ± 0.2	<1 ± 0.1	20 ± 17.9
MEK 2.1	27 ± 1.7	3 ± 1.0	5 ± 0.6	28 ± 12.5	16 ± 12.3	7 ± 4.1	4 ± 1.6	2 ± 0.8
NTK 0.2	33 ± 6.6	18 ± 11.4	20 ± 10.6	3 ± 0.4	<1 ± 0.1	0	<1 ± 0.1	10 ± 0.1
NTK 1.0	44 ± 14.1	2 ± 0.1	9 ± 0.8	3 ± 0.9	13 ± 0.9	5 ± 1.2	16 ± 14.0	3 ± 0.2
WCK 2.3	32 ± 18.7	6 ± 4.3	25 ± 19.9	8 ± 2.7	0	<1 ± 0.01	<1 ± 0.1	23 ± 4.2
WCK 2.9	38 ± 16.0	18 ± 5.2	4 ± 2.9	9 ± 5.4	0	0	<1 ± 0.2	25 ± 11.9
WCK 3.4	61 ± 23.7	17 ± 9.8	2 ± 1.3	8 ± 4.7	<1 ± 0.1	<1 ± 0.05	<1 ± 0.04	8 ± 5.9
WCK 3.9	48 ± 19.9	4 ± 3.5	2 ± 1.6	46 ± 15.1	0	0	0	0
WCK 5.1	76 ± 13.1	7 ± 3.9	8 ± 3.9	1 ± 0.2	<1 ± 0.03	<1 ± 0.1	<1 ± 0.3	4 ± 3.1
WCK 6.8	41 ± 2.5	1 ± 0.1	2 ± 0.2	3 ± 1.5	0	13 ± 1.9	35 ± 0.3	2 ± 0.02

^aValues are the mean ± 1 standard error for May and June 1987 samples.

^bFCK = First Creek kilometer, FFK = Fifth Creek kilometer, MEK = Melton Branch kilometer, NTK = Northwest Tributary kilometer, and WCK = White Oak Creek kilometer.

accounted for at least 21% of the total density at all sites and <30% at only three sites. Furthermore, they were exceeded in abundance by another taxonomic group at only one site (MEK 2.1) where oligochaetes accounted for 28% of the total density and chironomids accounted for 27%. With few exceptions (NTK 0.2, WCK 2.3, and WCK 2.9), the relative abundance of chironomids was usually highest at the nonreference sites.

Of the remaining dominant taxa, the relative abundance of nonchironomid dipterans, coleopterans, oligochaetes, and trichopterans was generally greatest at the nonreference sites (Table 6.2). The major exceptions to this trend were at FFK 1.0, where the densities of dipterans, coleopterans, and trichopterans were relatively high, and at MEK 2.1, where the density of oligochaetes was relatively high. The major difference between FFK 1.0 and the nonreference sites in the relative abundance of trichopterans was the number of taxa within this group that contributed to the density at these sites. At FFK 1.0 at least 12 Trichoptera taxa were collected, whereas at the nonreference sites, 6 or fewer Trichoptera taxa were collected (Table C.1).

The relative abundance of ephemeropterans and plecopterans was always greatest at the reference site of each stream. Additionally, the relative abundance of plecopterans did not exceed 1% at any nonreference site, and the relative abundance of ephemeropterans exceeded 2% at only two nonreference sites. The relative abundance of isopods exceeded 10% at only three sites, all of which were reference sites. At all other sites, isopods were either absent or contributed <1% to the total density.

6.1.3.4 Community structure

Richness. Taxonomic richness (number of taxa/sample) of invertebrates in each stream was highest at the reference site with one exception (Table 6.1). In Melton Branch, only minor differences existed between MEK 0.6 and its reference site, MEK 2.1; in May, richness was higher at MEK 0.6 but in June, it was higher at MEK 2.1. Richness at MEK 1.2 was more similar to richness in the lower reaches of WOC. Of all sites sampled in the WOC watershed, richness was lowest at WCK 3.9, WCK 5.1, and FFK 0.2. Downstream of WCK 3.9, richness increased considerably but still remained less than at WCK 6.8.

As was found for density, the differences in taxonomic richness for the May and June period of 1986 and 1987 were substantial (Fig. 6.1). At all sites, richness was significantly greater in 1987 than in 1986 (see Fig. 6.1 for levels of statistical significance). Additionally, during both years, richness at the reference sites was higher than at the downstream sites in all streams with one exception: mean richness in Melton Branch was the same at MEK 0.6 and MEK 2.1 during the second year. The greatest between-year differences occurred at FFK 0.2, MEK 0.6, MEK 1.2, WCK 3.4 and WCK 3.9, where richness was at least five times greater in 1987 than in 1986.

Ephemeroptera, Plecoptera, and Trichoptera. Differences in EPT richness were generally more pronounced between the reference sites and their respective downstream sites than taxonomic richness (Table 6.1). As with taxonomic richness, EPT richness was highest at the reference

sites of each stream except Melton Branch. EPT richness was the same at MEK 0.6 and MEK 2.1 in both May and June, and, in June, EPT richness was the same at all three Melton Branch sites. In WOC, EPT richness increased with increasing distance downstream from WCK 3.9. The sites exhibiting the lowest EPT richness values were WCK 3.9 and FFK 0.2, although values were relatively low at WCK 3.4 and WCK 5.1 also.

EPT richness for May and June 1987 was higher than that for the same period in 1986 at all sites except WCK 3.9, where no EPT taxa were collected during this time period in either year (Fig 6.1). At those sites where EPT taxa were found, the difference between 1986 and 1987 was significant at all sites but MEK 2.1 (see Fig. 6.1 for levels of statistical significance). Finally, during both years, EPT richness was highest at the reference site of each stream.

Taxonomic diversity. Taxonomic diversity was highest at the reference site of each stream during May, but this trend did not persist in all streams in June (Table 6.1); in NT, the trend was reversed, and in WOC, diversity at the three most downstream sites was greater than at WCK 6.8. During both months, diversity was lowest at MEK 1.2 in Melton Branch, and at WCK 3.9 and WCK 5.1 in WOC.

Between-year differences in taxonomic diversity were not as dramatic as those exhibited for density, taxonomic richness, and EPT richness (Fig. 6.1). Diversity was higher during May and June 1987 compared to the same period in 1986 at all sites but WCK 6.8; however, differences between years were statistically significant at only 8 of 15 sites (see Fig 6.1 for levels of statistical significance). Some of the most dramatic between-year changes in diversity occurred at FFK 0.2, WCK 3.4,

and WCK 3.9, where diversity values increased by at least a factor of three.

6.1.4 Discussion

As in the first year of the benthic invertebrate monitoring program (Smith 1993a, 1993b), results obtained during the first 2 months of the second year continued to show degraded water quality in the lower reaches of WOC and tributaries. In general, these downstream sites were characterized by low taxonomic richness, EPT richness, and taxonomic diversity relative to their upstream reference sites. Additionally, the downstream sites were typically dominated by only one or two major groups of invertebrates (e.g., Chironomidae, Oligochaeta, and/or some Trichoptera), which generally become numerically dominant in polluted waters (e.g., Wiederholm 1984; Kondratieff et al. 1984; Mount et al. 1984; Smith 1992a). In contrast, the reference sites were typically dominated by several taxa, including groups that are usually less tolerant of pollution (e.g., Ephemeroptera, Plecoptera) (e.g., Wiederholm 1984) and are common members of benthic invertebrate communities of relatively undisturbed streams on the ORR and in the Oak Ridge area (Smith 1990).

As was also found in the first year (Smith 1993a, 1993b), lower Fifth Creek (FFK 0.2) and WOC near WCK 3.9 continued to exhibit the greatest degree of impact of all the sites sampled; minimum values for density, richness, EPT richness, and diversity were typically found at these two sites (Table 6.1). Substantial impacts were also observed in lower First Creek at FCK 0.1 and in WOC at WCK 5.1, where dramatic differences in taxonomic and EPT richness were seen between these sites and their reference. Lower NT (NTK 0.2) was similarly impacted, but the extent of impact was somewhat less than that found at

FCK 0.1 and WCK 5.1. Relative to the reference site in NT (NTK 1.0), both taxonomic and EPT richness were lower at NTK 0.2, but the differences in these parameters between the two sites were not as dramatic.

In WOC and Melton Branch where there were more than two sampling sites, the invertebrate communities exhibited signs indicative of improving water quality with increasing distance from the major source(s) of perturbation. In WOC, impact at WCK 5.1 was followed by even greater impact at WCK 3.9, the next site downstream. At WCK 3.4, the benthic community exhibited characteristics indicative of considerably improved conditions, and still further improvement was evident at WCK 2.3, the most downstream site. The lack of strong differences between the benthic communities at WCK 2.9 and WCK 3.4 suggests that groundwater/surface water runoff from SWSA 4 either (1) resulted in no additional impact to WOC between WCK 3.4 and WCK 2.9 or (2) resulted in additional impact that is slowing improvement. One minor difference in community composition between these sites was the decrease in the relative abundance of chironomids and increase in the relative abundance of trichoptera between WCK 3.4 and WCK 2.9. This pattern of change with increasing distance from the source of a perturbation appears to be typical of other similarly impacted streams as conditions improve (Smith 1992a) and suggests that SWSA 4 is resulting in little or no additional detectable impact to the invertebrate community in WOC. The absence of more distinct differences between these sites may be due to their relatively close proximity to one another.

In Melton Branch, taxonomic richness, EPT richness, and diversity were distinctly depressed at MEK 1.2 relative to its reference site (MEK 2.1); however,

differences between MEK 2.1 and MEK 0.6 were not as distinct. Taxonomic and EPT richness were similar at both of these sites, while density was higher at MEK 0.6 and diversity was higher at MEK 2.1. Density can be influenced by several factors and is not necessarily a reliable parameter for separating differences due to perturbations, except for toxic conditions that usually reduce density (e.g., Wiederholm 1984). Similarly, diversity indices are sometimes not sensitive enough to detect subtle differences between sites that can occur under altered conditions (e.g., Rosenberg 1976; Godfrey 1978; Smith 1992a), which thus limits their usefulness. Differences did occur between these two sites, however, that indicated MEK 0.6 was still impacted at the time of this study. MEK 0.6 was dominated by chironomids and trichoptera, while the relative abundance of ephemeroptera and plecoptera was very low. This pattern is typical of benthic invertebrate communities in moderately polluted streams (e.g., Winner et al. 1980; Wiederholm 1984). A similar pattern was found at some other sites in the WOC watershed (e.g., MEK 1.2, WCK 2.3, and WCK 2.9) and has also been observed in other moderately polluted streams in the Oak Ridge area (Smith 1992a). The greater improvement at MEK 0.6 relative to MEK 1.2 suggests that groundwater flow and surface water runoff from SWSA 5 is causing no additional impact to the stream invertebrate community.

Between-year differences in the benthic community of each site were examined for trends indicative of recovery. Based on 2 months of samples, all sites exhibited significant between-year differences. With few exceptions, density, taxonomic richness, and EPT richness were higher at each site during the second year, and at most sites, these differences were significant. Diversity tended to be higher at most sites during the second year,

although the differences were significant at only about half of the sites.

Although benthic invertebrate communities exhibit natural year-to-year changes in composition, richness, relative abundance of taxa, etc. (e.g., McElravy et al. 1989), it is unlikely that all streams and all sites within a watershed would exhibit a pattern similar to that observed in the WOC watershed (i.e., significant increases at virtually every site in density, taxonomic richness, and EPT richness). Thus, the magnitude of most of the observed changes in WOC was probably not due to natural variability alone, but to other factors, such as improving conditions and/or between-year differences in processing efficiency. A preliminary analysis of some quality assurance checks of sample processing efficiency of JAYCOR personnel indicated that processing efficiency has improved from consistently <70% by TVA to consistently >90% by JAYCOR (J. G. Smith, ORNL/ESD, unpublished data). Thus, many of the between-year differences observed within each site were most likely due to an increase in processing efficiency.

Even though processing efficiency differed between years, the differences in efficiency appear to have had no effect on distinguishing spatial trends, that is, similar spatial trends appear to have occurred in the benthic communities of all streams during both years. However, differences in efficiency reduce the ability to detect between-year differences within each site and, therefore, reduce the ability to detect recovery. To identify changes that may be indicative of recovery, the data for each year were first "normalized" by dividing the largest parameter value (i.e., density, taxonomic richness, and EPT richness) by the smallest to determine the magnitude of difference between each impacted site and its respective reference site. For the convenience of illustration, values were designated as either negative, when the

parameter value was highest for the nonreference site, or positive, when the parameter value was highest for the reference site. Assuming that (1) processing efficiency had no effect on the magnitude of difference between sites within each year and (2) the reference sites remained relatively unimpacted, the magnitude of change between years was then determined for each parameter by subtracting the magnitude of difference for 1987 from the magnitude of difference for 1986. Thus, for a site that exhibited no between-year change relative to its reference site, the resulting value was zero, while a positive or negative change between years was indicative of either natural change or improved or degraded conditions. For example, in 1986, the mean density at WCK 2.3 and WCK 6.8 (the reference site) was 135.4 and 109.4 individuals/0.1 m², respectively, so the magnitude of the difference was 1.2. In 1987, mean densities at these sites were 825.8 and 840.0 individuals/0.1 m² respectively, for a difference of approximately 1.0 times. Because the parameter value in 1986 was higher at the nonreference site, the resulting value was designated as negative. Thus, the magnitude of the change between the 2 years was -2.2, indicating that density at WCK 2.3 was 2.2 times lower in 1987 than in 1986.

Most sites exhibited increases in the magnitude of change between years in density, taxonomic richness, and EPT richness (Fig. 6.2). The greatest between-year changes occurred at FFK 0.2, where density and taxonomic richness increased by 30.6 and 9.1 times, respectively, from 1986 to 1987. The amount of change at this site suggests that some improvement may have occurred since 1986. The substantial change in density and richness at WCK 3.9 was followed by a more moderate change at the two next sites downstream, suggesting that some

ORNL-DWG 93M-15983

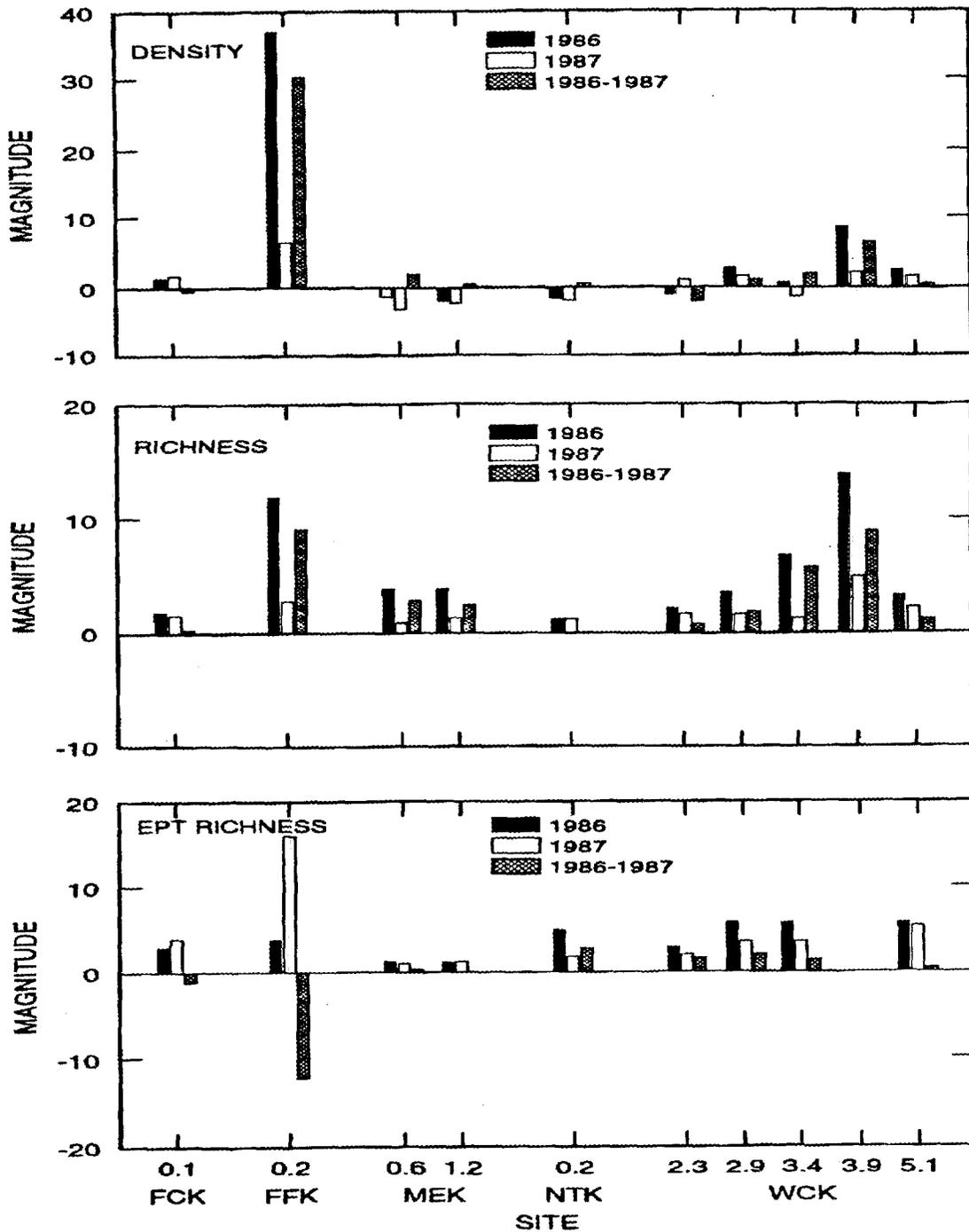


Fig. 6.2. Magnitude of difference between each downstream site and its reference site and the magnitude of change within each site between 1986 and 1987 for mean density, taxonomic richness, and Ephemeroptera, Plecoptera, and Trichoptera richness in the White Oak Creek watershed, May and June 1986 and 1987. (Note: FCK = First Creek kilometer, FFK = Fifth Creek kilometer, MEK = Melton Branch kilometer, NTK = Northwest Tributary kilometer, and WCK = White Oak Creek kilometer.)

improvement may have occurred at these sites. MEK 0.6 and MEK 1.2 also exhibited some between-year change.

Operational changes occurred at ORNL between November 1986 and March 1987, including shutdown of HFIR in November 1986 and the remaining reactors in March 1987 (Loar 1993a). HFIR was most likely a major cause of impact to the biotic communities of middle and lower Melton Branch (Loar et al. 1992). Similarly, the Oak Ridge Reactor and Bulk Shielding Reactor, whose effluents were discharged into Fifth Creek, was probably a major cause of impact to lower Fifth Creek and possibly to portions of WOC downstream of Fifth Creek, including WCK 3.9 and WCK 3.4. Thus, the changes that were observed in lower Fifth Creek, lower and middle Melton Branch, and the middle reaches of WOC may be due to improvements in water quality associated with the shutdown of these reactors. However, these observations of between-year changes are based on only 2 consecutive months of data and do not consider overall seasonal changes. Consequently, they may have missed important differences that could exist in other seasons. A more accurate and thorough analysis will be possible only when additional data are available.

The patterns of differences in the benthic invertebrate communities between sites in the WOC watershed suggest that point-source discharges are likely the most important causes of impact. This was demonstrated best in WOC and Melton Branch where impacts were greatest at the sites closest to major point-source discharges, and improvements were noted at those sites further downstream. This finding is consistent with results obtained from ambient toxicity tests (Loar et al. 1992; Stewart 1993).

As was hypothesized previously (Smith 1993a, 1993b), the episodic release of toxicants appears to have been a major

factor controlling the community structure and composition of the most highly impacted sites (e.g., FFK 0.2 and WCK 3.9). The invertebrate community at these sites was primarily comprised of chironomids and other taxa (e.g., Oligochaeta) that have short generation times; are capable of producing several generations per year; and/or reproduce continuously, thus increasing the probability that a resistant or unaffected stage would be present during a toxic excursion. With increasing distance downstream, toxicity is likely reduced by dilution and the loss of some toxicants through volatilization and other chemical and physical processes. As water quality improves, longer-lived but moderately pollution-tolerant taxa, such as some Trichoptera (e.g., *Hydropsyche*, *Cheumatopsyche*), are able to become established, as has been observed at several impacted sites.

Based on results obtained from toxicity tests (Loar et al. 1992; Stewart 1993), chlorine is likely a major toxicant at FFK 0.2 and WCK 3.9. However, because chlorine concentrations rapidly decline with distance (Loar et al. 1992), it is most likely not a factor, or only rarely a factor, at the other impacted sites. Thus, stresses associated with effluent discharges and operations at ORNL must also be contributing. Results of a toxicity test with water from WCK 3.8 showed that even after removing chlorine from the sample, there was evidence of toxicity, thus indicating the presence of additional unidentified toxicants (e.g., metals; see Sect. 3.1.4).

Other factors identified (e.g., siltation, elevated temperatures, flow augmentation, and nutrient enrichment) and discussed in past reports continue to be potential perturbations (Smith 1993a, 1993b). Chlordane, an insecticide, has been found in clams held in WOC just upstream of WCK 5.1 and in sediments from the same

general area (Sect. 4.1.3.2). Although no data were available on chlordane for this site during the time period covered in this section of this report, these findings suggest the possibility that chlordane may have existed there for some time prior to its discovery and, therefore, may have been a factor responsible for at least some impact.

6.1.5 Future Studies

Following the completion of monthly sampling of stream sites and bimonthly sampling of WOL during the first 2 years of BMAP (May 1986–April 1988), the sampling frequency of both the stream sites and WOL was reduced to quarterly. The collection of additional samples with artificial substrates in WOL beginning in April 1987 was suspended in March 1989 and results from the artificial substrate samples collected between April 1987 and March 1988 will be presented in the next annual report. The annual collection of qualitative samples during the spring will continue for only the nonreference sites. Finally, studies will be conducted in conjunction with the Y-12 Plant BMAP for EFPC (Loar et al. 1989) to determine the importance of elevated temperatures on invertebrate populations in the WOC watershed.

6.2 FISHES (M. G. Ryon and E. M. Schilling)

6.2.1 Introduction

Fish population and community studies can be used to assess the ecological effects of changes in water quality and habitat. These studies offer several advantages over other indicators of environmental quality (see Karr et al. 1986; Karr 1987) and are especially relevant to

assessment of the biotic integrity of WOC. For example, fish communities include several trophic levels, and species that comprise the potential sport fishery in WOC [e.g., bluegill (*Lepomis macrochirus*), redbreast sunfish (*L. auritus*), and largemouth bass (*Micropterus salmoides*)] are at or near the end of food chains. Consequently, they integrate the direct effects of water quality and habitat changes on primary producers (periphyton) and consumers (benthic invertebrates) that are used for food. Because of these trophic interrelationships, the well-being of fish populations has often been used as an index of water quality (e.g., Weber 1973; Greeson et al. 1977; Karr et al. 1986). Moreover, statements about the condition of the fish community are better understood by the general public (Karr 1981).

The initial objectives of the instream fish monitoring task (Subtask 4b of BMAP, as described in Loar et al. 1986) were (1) to characterize spatial and temporal patterns in the distribution and abundance of fishes in WOC and (2) to document any effects on fish community structure and function resulting from implementation of the ORNL WPCP and ERP.

6.2.2 Methods

Quantitative sampling of the fish populations at 16 sites in the WOC watershed (Fig. 2.2) was conducted by electrofishing in April and October–November 1989. The resulting data were used to estimate population size (numbers and biomass per unit area), determine age structure and growth rates, and calculate the index of biotic integrity (IBI) values. The length of the sampling reaches ranged from 27 to 78 m at the tributary sites and from 48 to 155 m at the WOC sites (Table 6.3). Fish sampling sites either overlapped or were within 100 m of

Table 6.3. Length, mean width, mean depth, surface area, and pool to riffle ratio of fish sampling sites

Site ^a	Length (m)		Mean width (m)		Area (m ²)		Mean depth (cm)		Pool/riffle ratio	
	Spr ^b	Fall ^c	Spr	Fall	Spr	Fall	Spr	Fall	Spr	Fall
FCK 0.1	70	62	1.2	1.4	84	87	8.6	12.0	1.1	1.6
FCK 0.8	39	37	1.4	1.4	55	52	12.2	12.1	0.7	0.9
FFK 0.2	78	68	1.1	1.1	86	75	11.6	9.9	0.7	0.6
FFK 0.4	28	28	1.7	2.0	48	56	14.0	12.5	0.8	0.8
FFK 1.0	27	28	1.1	1.1	30	31	8.8	6.5	0.5	0.6
MEK 0.6	52	48	3.0	2.9	156	139	14.1	17.7	2.3	8.6
MEK 1.4	54	49	2.2	2.3	119	113	8.6	11.1	0.9	1.2
MEK 2.1	52	70	1.8	1.9	94	133	4.3	7.3	0.3	1.3
NTK 0.3	70	67	2.6	3.1	182	208	5.9	8.6	0 ^d	0.4
NTK 1.0	38	41	2.2	2.9	85	119	6.5	11.3	0.7	0.6
WCK 2.3	94	88	5.1	5.8	479	510	26.5	31.8	3.9	4.5
WCK 2.9	79	86	5.6	6.5	442	464	35.8	35.8	78.0	4.7
WCK 3.4 ^e	61	70	2.8	3.9	171	273	30.8	21.6	3.1	4.4
WCK 3.9	155	147	3.0	2.7	465	397	16.7	15.2	0.7	1.2
WCK 5.1	50	48	1.7	2.1	85	101	10.1	14.0	1.3	0.7
WCK 6.8	60	60	2.3	2.2	138	132	6.3	8.1	0.4	0.5

^aFCK = First Creek kilometer, FFK = Fifth Creek kilometer, MEK = Melton Branch kilometer, NTK = Northwest Tributary kilometer, and WCK = White Oak Creek kilometer.

^bSpr = April 1989.

^cFall = October–November 1989.

^dNo riffle in site at that sample date.

^eSite length was adjusted in fall; both upper and lower boundaries were moved downstream.

the sites included in the instream benthic invertebrate monitoring task (Sect. 6.1), except for FFK 0.4 (where benthos were not sampled) and MEK 1.4 (benthos were sampled at MEK 1.2). Lengths of the sampling reaches were similar (the greatest difference was 27%) to those in 1988 (Loar 1993b).

The stream reach at WCK 3.4 was adjusted downstream prior to sampling in November 1989, which lengthened the site by 15%. The adjustment was necessary because a large clay shelf, which formed a small waterfall at the upstream end of the site, had eroded. Erosion of the shelf resulted in deposition of sediment in downstream pools, changed the pool-riffle ratio, and made placement of the upper blocknet difficult. Consequently, the upstream end of the reach was moved to a placid pool area and the downstream end was moved to the bottom of a wide riffle. With these changes, the blocknets could be positioned more securely, the pool-riffle ratio was more like that at WCK 2.3 and WCK 2.9, and a narrow, high-velocity run was replaced by a wide riffle with current velocities that were more typical of WOC.

6.2.2.1 Field sampling procedures

All stream sampling was conducted using one or two Smith-Root Model 15A backpack electrofishers, depending on stream size. Each unit has a self-contained, gasoline-powered generator capable of delivering up to 1200 V of pulsed direct current. A pulse frequency of 90–120 Hz was used, and the output voltage was adjusted to the optimal value (generally 400 V or less) based on the specific conductance of the water. The circular (ring) electrode at the end of the fiberglass anode pole was fitted with a nylon net (0.64-cm mesh) to allow the electrofisher operator to collect stunned fish.

After 0.64-cm-mesh seines were placed across the upper and lower boundaries of the fish sampling site to restrict fish movement, a two- to five-person sampling team electrofished the site in an upstream direction on three consecutive passes. If fish numbers captured during the first pass were extremely low or zero, then only one pass was made. Depending upon the turbidity of the water, the consecutive passes could not always be made immediately. Rather, fish were processed after each pass to allow sufficient time for the water to clear before another pass was initiated. Stunned fish were collected and stored, by pass, in wire mesh cages (0.64-cm diam) or in buckets with small holes during further sampling.

Following the electrofishing, fish were anesthetized with MS-222 (tricaine methanesulfonate), identified, measured to the nearest 0.1 cm (total length), and weighed to the nearest 0.1 g (for fish <100 g) or gram (for fish >100 g) using Pesola spring scales. At sites with high fish densities, individuals were recorded by 1-cm size classes and species. After 25 individuals of a species-size class were measured and weighed, additional members of that size class were only measured. Length-weight regressions based on the 25 individuals were later used to estimate missing weight data (Railsback et al. 1989). Other data recorded (if possible to determine) included sex, reproductive state, disposition (i.e., dead or kept for laboratory identification and inclusion in a reference collection), whether scales were taken for age and growth studies, and the presence of any abnormalities (e.g., external parasites, skeletal deformities, etc.).

After processing fish from all passes, the fish were allowed to fully recover from the anesthesia and were returned to the stream. Any additional mortality occurring as a result of processing was noted at that time. In addition to data on individual fish,

data on selected physical and chemical parameters and sampling effort were recorded. An Horiba Model U7 battery-powered field sampler was used to measure conductivity, pH, water temperature, and dissolved oxygen content. An HF Instruments Model DRT-15 turbidimeter was used to measure turbidity. The duration of the electrofishing effort was recorded and a visual estimate was made of percent cloud cover. Following completion of fish sampling, the length, width, and depth of the sampling reach were measured at each site.

6.2.2.2 Data analysis

Population size. After reviewing the information on the field data sheets for completeness and accuracy, the data were entered and stored on main frame computers, verified by a standard QA/QC procedure, and analyzed using a Fortran 77 program.

Species population estimates were calculated using the method of Carle and Strub (1978). Biomass was estimated by multiplying the population estimate by the mean weight per individual. To calculate density and biomass per unit area, total numbers and biomass were divided by the surface area (m²) of the study reach. For each sampling date, surface area was estimated by multiplying the length of the sampling reach by the mean width based on measurements taken at 5-m intervals (Table 6.3). These data were compiled and analyzed by a comprehensive Fortran 77 program developed by Railsback et al. (1989).

Growth and condition factor. Analysis of growth was performed using PROC GLM and PROC MEANS procedures (SAS 1985a,b) with length, weight, and age data for redbreast and bluegill. True growth rates were estimated using the

procedures described by Ricker (1975). The true growth, which is the instantaneous rate of increase in weight for the most recent year of growth, involved the following steps: (1) determine the age from scales (see below) and the measurement of successive annuli; (2) estimate the relationship between scale length and fish length; (3) backcalculate body length at the start and finish of the last complete year of growth on the scale at each age for each fish; (4) calculate the functional slope, **b**, of the length-weight relationship for each fish population; (5) calculate the natural logarithm of lengths determined in (3) for each fish and subtract the logarithms (equals the instantaneous rate of increase in length for each fish); and (6) average the instantaneous rates of increase for each age group and multiply by **b** to obtain the instantaneous rate of increase in weight at each age (Ricker 1975). The formula for the backcalculation of length at each annulus (Carlander 1981) was:

$$BCL = a + [(TL - a)/RL]AL,$$

where BCL = backcalculated length at the time of annulus formation,
 a = intercept of scale length vs body length regression,
 TL = total length of fish at capture,
 RL = scale measurement to edge of scale, and
 AL = scale measurement to the annulus.

For these analyses, a common value of 20 mm was used for the constant **a**, as determined and recommended by Carlander (1982). The growth values estimated by this procedure were evaluated for statistical differences between sites using the Tukey test (SAS 1985a, 1985b).

Condition factors, **K**, were calculated for individual fish by site and species using the formula:

$$K = 100 (\text{weight}/\text{length}^3),$$

with weight in grams and total length in centimeters (Hile 1936). This condition factor measures the degree of plumpness of a fish as a measure of relative health (Bennett 1970). Fish without measured weights were not used in calculations of condition factors. Comparisons of condition factors between sites and between sampling periods were made using an ANOVA (PROC GLM) on untransformed data (SAS 1985b) because the condition factors exhibited homogeneity of variance as estimated with the UNIVARIATE procedure (SAS 1985a). If the GLM procedure indicated significant differences in condition factors between groups, the Tukey test was performed to identify those groups that were significantly different ($p < 0.05$).

Age determination. Scale samples were taken from redbreast and bluegill at WCK 2.3, WCK 2.9 (redbreast only), WCK 3.4, and a reference stream, Brushy Fork (BFK 7.6) in 1988 and 1989 for analysis of age and growth parameters. Scale samples for the 1988 analysis were taken from August 1988 to January 1989 and those for the 1989 analysis were taken from August to November 1989. Scales were obtained from fish collected during the routine population surveys and during electrofishing required for the bioaccumulation bioindicator and tasks (Sects. 4 and 5 respectively).

Scales were taken from an area above the lateral line and slightly anterior to the insertion of the dorsal fin. Impressions of the scales were made using a Wildco scale press and acetate slides. Enlarged images of the scales were projected on a screen using a Bruning 4020 microfiche reader with a 15-mm lens (48×). Where possible, at least ten scales from each fish were mounted and compared. If the scales produced poor impressions (due to attached mucous or skin), they were cleaned with water, scraped, and

remounted. In some cases, if it was still difficult to age the fish, the actual scales were placed between two acetate slides for examination. The best representative scale was used for actual measurements of annuli. Scales identified as regenerated (latinucleate) and those that were damaged or highly irregular in shape were not read. In some cases, no age data were obtained for a fish because all scales were unsuitable.

In this analysis, ages were determined by one reader and independently verified by another reader. A weighted subsample of all age classes for each site and species was verified. The scale samples were sorted into 1-cm groups based on total fish length, and a random 20% of each group were subsampled for verification. Also, scales that were judged to be difficult to age by the initial reader were reviewed by the second reader. If a consensus could not be reached on the age of a fish, the data were not used in the analysis.

The following data were recorded for each scale examined: number of annuli, total length of scale radius (distance from focus to anterior margin), and length of radius to each annulus. The annulus was determined by examining (1) the intersection of the outermost margin of closely spaced (i.e., slow-growth) circuli with the innermost margin of widely spaced (i.e., rapid-growth) circuli, (2) the occurrence of cutting over of circuli at the lateral edges of the anterior field, (3) the increase in radii width or formation of holes in the radii, and (4) the termination or origin of radii.

Otoliths from some fish were used as a validation of the technique of aging by scale analysis. Beamish and McFarlane (1983) noted that all age classes of a population should be validated by mark-recapture studies or by the use of known-age fish, but if this is not possible, fish should be aged by more than one method (Schramm 1989).

The otoliths and associated scales were removed from redbreast sunfish collected from Brushy Fork at BFK 7.6 in 1988 and 1989 and from WCK 2.3 in 1989. Otoliths were removed from the fish by (1) cutting the gill isthmus, (2) bending the head back slightly, (3) cutting through the bulla to expose the otoliths, and (4) pulling the otoliths free from the bulla. Otoliths were placed in a labeled vial containing glycerine to improve the visibility of the ring structures (Williams and Bedford 1974). To age the otoliths, whole otoliths were immersed in a black dish containing water and were viewed at 12X under a dissecting microscope. The distinct opaque (annuli) and hyaline bands were easily visible against the dark background. Measurements were made with an ocular micrometer and included otolith radius (measured from the kernel to the anterior tip of the rostrum) and annulus length (the distance from the kernel to each opaque band measured along the same axis). The otoliths were aged by one reader and independently verified by another reader. The otolith sample was biased toward older age classes because the fish collected for the bioindicator task should preferably weigh >50 g to meet the minimum requirements for tissue/organ weight. To adjust for this bias, a random 20% of the otoliths from each site and species were verified in two groups; one for fish weighing <50 g and another second for fish weighing >50 g.

Index of biotic integrity (IBI). The fish population data at each site were analyzed using the IBI, a fish community assessment technique that includes measures of species richness and composition, trophic composition, and fish abundance and condition (Karr 1981; Karr et al. 1986). As suggested by Karr and others (Karr et al. 1986; Ohio EPA 1988), modifications were made to the basic IBI metrics to reflect differences between

midwestern streams, on which the IBI was originally based, and streams in the Clinch River drainage in the Oak Ridge area, which includes WOC. A detailed discussion of the IBI and the various modifications that were made to it are given in Appendix D.

6.2.3 Results and Discussion

6.2.3.1 Species richness and composition

In all, 13 species were collected in quantitative surveys of WOC in 1989 (Table 6.4). Nine species were found at the lowermost site on WOC (WCK 2.3), of which five were centrarchids and two were cyprinids. Intermediate sites on WOC (WCK 2.9 to WCK 5.1) had four to seven species, while the uppermost reference site (WCK 6.8) had four species. The number of species in the tributaries of WOC ranged from one to five. Fish were collected in lower Fifth Creek (FFK 0.2) for the first time since September 1985 when a routine monitoring program was implemented. Fish were also found in upper Melton Branch (MEK 2.1) for the first time since the spring of 1987.

The patterns observed in species richness during 1989 were very similar to those found in 1985–1988 (Loar et al. 1992; Loar 1993a, 1993b). Generally, species richness increased with stream size but abrupt changes in the number of species were evident as a result of barriers (weirs) that isolated certain reaches of stream. The WOC watershed remains depauperate in comparison with other area watersheds (Ryon and Loar 1988). The slightly higher richness at WCK 2.3 was a result of its proximity to WOL but unlike earlier years, the transient movement of WOL species into lower WOC was not observed. No new species were found in the WOC watershed in 1989.

Table 6.4. Fish species composition in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, April and October–November 1989*

Species	Species composition															
	FCK ^b 0.1	FCK 0.8	FFK ^c 0.2	FFK 0.4	FFK 1.0	MEK ^d 0.6	MEK 1.4	MEK 2.1	NTK ^e 0.3	NTK 1.0	WCK ^f 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Centrarchidae																
Bluegill (<i>Lepomis macrochirus</i>)	2	-	-	-	-	-	-	-	2	1	2	-	2	-	-	-
Redbreast sunfish (<i>L. auritus</i>)	-	-	-	-	-	2	-	-	-	-	2	2	2	-	-	-
Largemouth bass (<i>Micropterus salmoides</i>)	1	1	-	-	-	-	-	-	-	-	2	2	1	1	-	-
Warmouth (<i>L. gulosus</i>)	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-
Rock bass (<i>Ambloplites rupestris</i>)	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-
Cottidae																
Banded sculpin (<i>Cottus carolinae</i>)	-	1	-	2	2	-	-	-	-	-	-	-	-	-	-	2
Cyprinidae																
Blacknose dace (<i>Rhinichthys atratulus</i>)	2	2	1	2	2	2	2	1	2	1	1	1	1	1	2	2
Central stoneroller (<i>Camptostoma anomalum</i>)	-	-	-	-	-	-	-	-	-	-	2	-	-	1	2	1
Creek chub (<i>Semotilus atromaculatus</i>)	-	-	-	-	-	2	2	1	-	1	-	-	-	1	2	2
Fathead minnow (<i>Pimephales promelas</i>)	2	-	-	-	-	-	-	-	1	1	-	1	1	1	2	-

Table 6.4 (continued)

Species	Species composition															
	FCK ^b 0.1	FCK 0.8	FFK ^c 0.2	FFK 0.4	FFK 1.0	MEK ^d 0.6	MEK 1.4	MEK 2.1	NTK ^e 0.3	NTK 1.0	WCK ^f 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Gasterosteidae																
Brook stickleback (<i>Culaea inconstans</i>)	-	-	-	-	-	-	-	-	1	-	-	-	-	-	-	-
Ictaluridae																
Yellow bullhead (<i>Ameiurus natalis</i>)	-	-	-	-	-	-	-	-	-	-	2	-	-	-	-	-
Poeciliidae																
Western mosquitofish (<i>Gambusia affinis</i>)	2	-	-	-	-	-	-	-	2	1	1	2	2	1	-	-
Number of species (<i>n</i>)	5	3	1	2	2	3	2	2	5	5	9	5	7	6	4	4

^aNumbers represent the number of sampling periods (*n* = 2) that a given species was collected at the site and a "dash" indicates that the species was not collected.

^bFCK = First Creek kilometer.

^cFFK = Fifth Creek kilometer.

^dMEK = Melton Branch kilometer.

^eNTK = Northwest Tributary kilometer.

^fWCK = White Oak Creek kilometer.

Distribution patterns of individual species in 1989 again showed little change from earlier years. The blacknose dace (*Rhinichthys atratulus*) was collected at all 16 sites, followed by the fathead minnow (*Pimephales promelas*), western mosquitofish (*Gambusia affinis*), and creek chub (*Semotilus atromaculatus*) at seven sites. In 1989, the largemouth bass (*Micropterus salmoides*) expanded its distribution in the WOC watershed to six sites, including all of First Creek and the middle and lower portions of WOC (WCK 2.3 to WCK 3.9). The brook stickleback (*Culaea inconstans*) remained a minor component of the community in lower NT. The sampling site (NTK 0.3) is near an established population in the ESD ponds south of the 1500 Area (Fig. 2.4), and the single specimen probably represents emigration from that area.

6.2.3.2 Density and biomass

Population surveys of WOC were conducted twice in 1989 (spring and fall), and the data were used to determine species biomass and density for each period. The total biomass and density at each site for each sampling period are presented in Table 6.5. Values for individual species are given in Appendix E, Tables E.1 through E.4.

In 1985–86, fish densities decreased downstream, while biomass increased with increasing stream size. In 1987–88, the downstream (nonreference) sites often had higher densities, particularly in the tributaries to WOC, and, again, fish biomass generally increased downstream. The trend (within tributaries) of higher densities and biomass at the lower vs upper sites continued in 1989. In mainstream WOC, however, abundance fluctuated, with generally higher densities at the upstream sites but higher biomass downstream. Unlike 1987–88, biomass and densities

were not consistently higher or lower in the fall compared to the spring.

Fish density and biomass in lower Melton Branch (MEK 0.6) increased substantially in 1987, probably in response to the shutdown of HFIR in November 1986 and the subsequent improvement in the adverse stream conditions, such as high temperatures, that existed in 1985–86 (Loar et al. 1992; Loar 1993a). The high abundance was sustained through 1989 as HFIR remained shut down. These data suggest that the fish community in lower Melton Branch had reached an equilibrium density that was still substantially higher than that observed when HFIR was operational. Further evidence of the ecological recovery in Melton Branch was the increasing fish abundance at MEK 1.4 just below the confluence with the small tributary that received discharges from HFIR. Densities and biomass at this site in 1989 were comparable to the levels observed in 1988. Stream flow at MEK 1.4 was augmented by unheated discharges from the HFIR/REDC ponds from 1987 to 1989. The recovery of the fish community in Melton Branch extended in fall 1989 to the upper site (MEK 2.1), which supported a fish community of the same size and composition as that found at MEK 1.4 after a return to normal levels of rainfall and higher stream flows in 1989.

Fish populations at WCK 5.1 entered 1988 having suffered a major decline in abundance in December 1987. Between April and December 1987, densities declined from 7.82 to 0.61 fish/m² (Loar 1993a, Table 6.14), a trend opposite of that expected for a species that spawns in the spring and usually has higher densities in the fall due to recruitment. During 1988, the community recovered dramatically with a 15-fold increase in density and a 2-fold increase in biomass (Loar 1993b). This recovery continued during 1989. Apparently the impact that

Table 6.5. Total fish density, total biomass, and species richness for April and October–November 1989 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Sampling periods	Sites ^a															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
April																
Total density (individuals/m ²)	3.78	1.62	0.07	8.69	3.03	1.80	1.56	NF ^b	0.82	0.04	0.39	0.22	0.18	NF	5.75	1.22
Total biomass (g/m ²)	7.68	2.65	0.06	14.04	14.71	7.24	3.19	0	0.36	0.07	9.08	7.53	10.18	0	12.11	2.43
Species richness	4	1	1	2	2	3	2	0	3	1	7	4	4	0	4	3
October–November																
Total density (individuals/m ²)	4.46	1.98	NF	5.94	2.87	2.08	2.57	2.45	3.32	0.51	0.16	0.30	0.58	0.28	3.72	2.43
Total biomass (g/m ²)	6.04	3.93	0	14.18	7.98	6.56	3.95	4.67	0.80	0.49	5.59	5.98	5.03	1.11	5.73	3.85
Species richness	5	3	0	2	2	3	2	2	5	4	9	4	5	6	4	4

^aFCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

^bNF = no fish collected in sample.

occurred in the summer of 1987 produced no permanent change in total community density or biomass.

The impact on the fish community at WCK 5.1 could have been associated with chlordane contamination in WOC that was first detected in early 1988. The contamination was discovered during an investigation of PCB source(s) to WOC, using caged Asiatic clams. Chlordane was found at very high concentrations (19 $\mu\text{g/g}$, wet weight) in duplicate samples of clams that had been held in WOC at WCK 5.4 during March 2–31, 1988 (Loar 1993b, Table 4.3). The chlordane concentration in clams at this site declined to 1.1–1.4 $\mu\text{g/g}$ in November 1988 and to 0.5–0.7 $\mu\text{g/g}$, wet weight by the following April (Table 4.5, this report). The substantial decline in chlordane levels between March and November 1988 was associated with an equally dramatic recovery of the fish population at WCK 5.1 in 1988 [densities increased from 0.53 to 8.47 fish/m² between April and December, respectively (Loar 1993b, Table 6.9)]. Although the benthic macroinvertebrate data for 1987–89 are not yet available, it seems probable that the observed changes in the fish community at WCK 5.1 during this period of time were causally related to a contaminant source in a small tributary just upstream at WCK 5.4.

The fish population in upper First Creek (FCK 0.8) demonstrated some recovery in species richness, density, and biomass during 1989. In 1985–87, this site frequently contained banded sculpin (*Cottus carolinae*), creek chub, and an occasional largemouth bass in addition to a considerable population of blacknose dace (Loar et al. 1992; Loar 1993a). In 1988, however, all species but the blacknose dace disappeared from the site (Loar 1993b). The site was affected by drought conditions as well as increased water withdrawal from upper First Creek for experimental stream studies in 1988. The

reappearance of the banded sculpin and largemouth bass in November 1989 indicated improved conditions that were most likely related to higher stream flow and cooler water temperatures. Mean summer water temperatures (June–August) were about 3°C lower in 1989 than 1988, and maximum temperatures during this period ranged from 19.3 to 23.4°C and from 25.4 to 29.2°C in 1989 and 1988, respectively (Loar 1993b; also Table 2.6 in this report).

Total fish densities were highest at FFK 0.4 and WCK 5.1 (8.69 and 5.75 individuals/m², respectively) and except for WCK 5.1 in 1987, as discussed previously, were similar to the densities observed at these sites in 1986 to 1988 (Loar et al. 1992; Loar 1993a, 1993b). Density was lowest at NTK 1.0 and FFK 0.2 (0.04 and 0.07 individuals/m² respectively). In 1989, maximum biomass values of 14.04 and 14.18 g/m² were observed at FFK 0.4 and were similar to the values in 1987–88 for this site. The lowest biomass in 1989 occurred at NTK 1.0 and FFK 0.2 (0.07 and 0.06 g/m² respectively).

Contributions of individual species to total community density and biomass were comparable to those observed from 1985 to 1988. The blacknose dace was the dominant species in terms of density and biomass in 16 and 9, respectively, of 25 possible sampling date-site combinations. Other dominant species included redbreast sunfish (dominant at five date-site combinations based on density and five based on biomass) and the creek chub (four date-site combinations based on biomass). Bluegill continued to show a decline in WOC. The density and biomass of bluegill decreased in 1988 (Loar 1993b) and additional declines in biomass were observed in 1989.

In summary, the most significant improvements in fish abundance occurred in First Creek at FCK 0.8 and in Melton Branch. Site FCK 0.8 showed recovery in

species richness, density and biomass in 1989 compared to 1988. Similarly, sampling in 1989 indicated continuing recovery in Melton Branch. MEK 0.6 and MEK 1.4 showed a sustained level of fish abundance (density and biomass) in 1988–89 compared to 1985–87. Also, the increased rainfall extended the recovery to upper Melton Branch. Shutdown of HFIR in November 1986 resulted in successful reproduction and recovery of the three species that inhabit Melton Branch. Sites shown to be significantly impacted by plant operations in the 1985–88 surveys (FFK 0.2 and WCK 3.9) continued to have very low densities or no fish in 1989 and showed no evidence of recovery.

6.2.3.3 Condition factors

Condition of fish in the WOC watershed was evaluated by calculating condition factors, K , for all species and by making statistical comparisons to evaluate differences between sites and between sampling periods. Comparisons between sampling periods showed that the condition factors in March–April were significantly greater than those in December. Of 16 comparisons where a significant difference was indicated, 15 had higher condition factors in the spring. This trend was also observed in 1985–88 (Loar et al. 1992; Loar 1993a, 1993b) and indicates the expected preparation for spawning. In 22 of the 38 comparisons, no significant difference between spring and fall condition factors was observed.

Comparisons of condition factors between sites within a sampling period generally showed no pattern of significant differences (Appendix E, Table E.5). Sites that had been identified as being impacted based on fish biomass, density, or species richness (WCK 3.9 and FFK 0.2) did not have significantly lower condition factors than upstream reference sites for any

species. The failure of condition factors to adequately separate differences in populations was also noted by Cone (1989), and he cautions that the method has critical flaws that could lead to false conclusions.

6.2.3.4 Age and growth

The comparison of aging using scale analysis yielded an overall 82% agreement between readers. Precision ranged from a low of 66% for the ages of 29 redbreast sunfish from BFK 7.6 in 1988 to a high of 100% for the ages of three bluegill from WCK 3.4 in 1988. Reader agreement was low for redbreast collected from WCK 3.4 in both 1988 and 1989 (69% for both years) and from BFK 7.6 in 1989 (72%). Both readers had greater difficulty aging the scales from BFK 7.6 and WCK 3.4 compared to the scales from fish at WCK 2.3 and WCK 2.9. For example, the agreement between readers in ages of redbreast taken at WCK 2.3 was 94% and 89% for 1988 and 1989 respectively.

Otoliths and associated scales were removed from redbreast sunfish that were collected as part of the bioindicator task. These otoliths were aged and used as a validation of the scale aging technique. Williams and Bedford (1974), Erickson (1983), and Schramm and Doerzbacher (1982) found otoliths to be more reliable than scales for age determination. The agreement between readers of ages as determined by otoliths was 100%. Interpreting annuli on otoliths was obviously much easier than scales for both readers.

The comparison of ages as determined by scales and otoliths indicated a high degree of agreement. Scales and otoliths from 74 fish at 2 sites (WCK 2.3 and BFK 7.6) were compared, and only 8 differences were noted. In six of these, scales were underaged compared to otoliths. Beamish and McFarlane (1987) cite numerous studies in which scales tend

to underestimate the age of fish, particularly older age classes. The agreement between ages from redbreast scales and from otoliths was 87% for fish collected from BFK 7.6 in each year (1988 and 1989) and 92% for fish collected from WCK 2.3 in 1989. These values are much greater than the 65% agreement reported by Schramm and Doerzbacher (1982) for black crappie in Florida. Taubert and Tranquilli (1982) noted that incorrect ages from scales are common in southern latitudes and areas receiving thermal discharges. Heidinger and Clodfelter (1987) reported 71, 80, and 75% agreement between ages determined from scales and from known-age individuals of smallmouth bass, striped bass, and walleye respectively.

The age class structure of the redbreast sunfish populations at WCK 2.9 and WCK 3.4 in 1988 and 1989 indicated that stress might be greater at these sites than at WCK 2.3 and the reference site (BFK 7.6) (Appendix E, Table E.6). The oldest fish at WCK 2.9 and WCK 3.4 were only 4+ years, whereas age 6+ and 7+ fish were found at WCK 2.3 and BFK 7.6. This difference in longevity may be related to habitat differences, discharges from ORNL, or runoff from SWSA 4, since WCK 3.4 and WCK 2.9 are both closer than WCK 2.3 to these potential sources of contaminant-related stress. A similar pattern of older age classes at BFK 7.6 and WCK 2.3 (4+ to 5+) than at WCK 3.4 (3+ to 4+) was found in bluegill populations in 1988 and 1989 (Appendix E, Table E.7). However, this pattern was not as striking as that observed for redbreast sunfish.

Growth of sunfish in WOC was evaluated by estimating the true growth rates in 1988 and 1989. In 1988, the patterns of growth in age classes 2+ to 5+ of redbreast sunfish were very similar among all tested sites (Fig. 6.3). The only statistical difference occurred in age class

2+, with fish at WCK 3.4 having a higher growth rate than fish at WCK 2.3 (Table 6.6). This pattern was also evident for bluegill in 1988 (Fig. 6.4), but the sample size at WCK 3.4 was too small to provide a meaningful analysis. The reason for this pattern is not clear. For bluegill, it may be the result of an unusually fast growing individual (Appendix E, Table E.7), although the redbreast sunfish data were based on much higher sample sizes.

Patterns of true growth for redbreast sunfish in 1989 corresponded better with patterns shown by other fish community data. The pattern of growth in age classes 2+ to 5+ for redbreast sunfish (Fig. 6.5) indicated a significant difference between sites. For age class 2+, growth was significantly greater at WCK 2.3 and BFK 7.6 than at WCK 3.4 and WCK 2.9 (Table 6.7). The pattern was a little less evident in the 3+ age class and disappeared in older age groups. The lower growth at WCK 3.4 and WCK 2.9 corresponded to an absence of older age classes and low values for species richness, biomass, and density at these sites.

The true growth pattern for bluegill (Fig. 6.6) in 1989 did not correspond with that observed in redbreast sunfish. Significantly greater growth was found for age class 2+ bluegill at WCK 3.4 than at WCK 2.3, and growth in the reference stream, BFK 7.6, was significantly lower than that at both WOC sites (Table 6.7). This contradiction in growth patterns between bluegill and redbreast sunfish may be related to the habitat requirements of the two species. The bluegill is better suited to pool or pond habitats (Robison and Buchanan 1988) than the redbreast (Aho et al. 1986), and the greater growth at the WOC sites may reflect this requirement. Site WCK 3.4 is associated with a weir impoundment and WCK 2.3 is upstream from WOL. Both of these habitats may provide for greater growth of

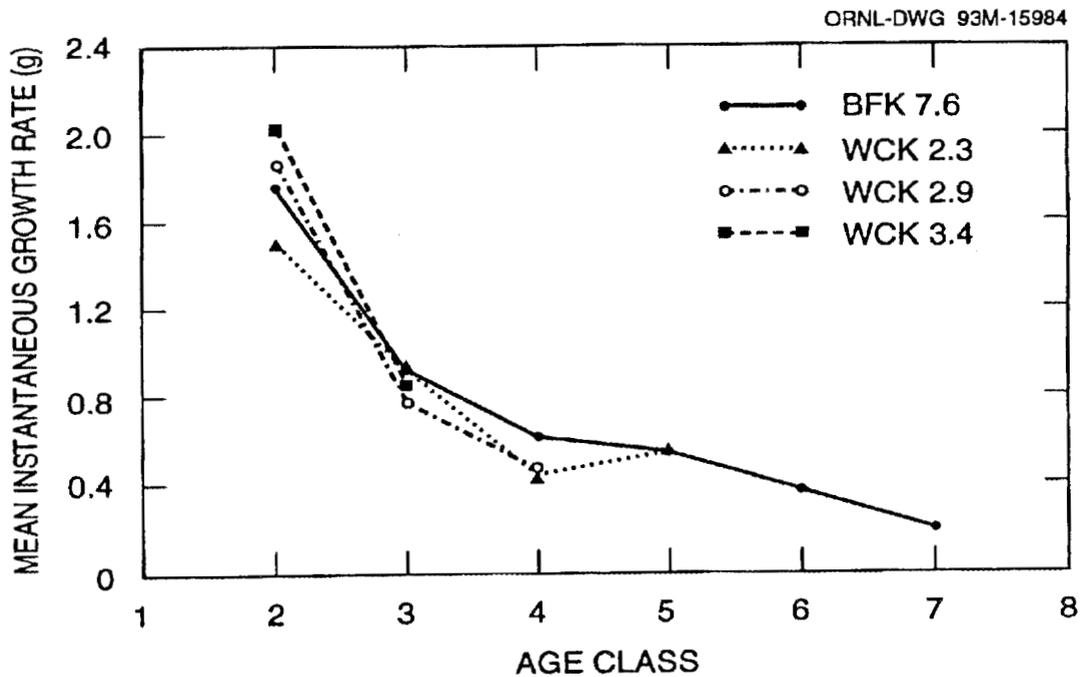


Fig. 6.3. True rate of growth in weight (in grams) of redbreast sunfish in White Oak Creek and a reference stream, Brushy Fork, during 1988. (Note: BFK = Brushy Fork kilometer, and WCK = White Oak Creek kilometer.)

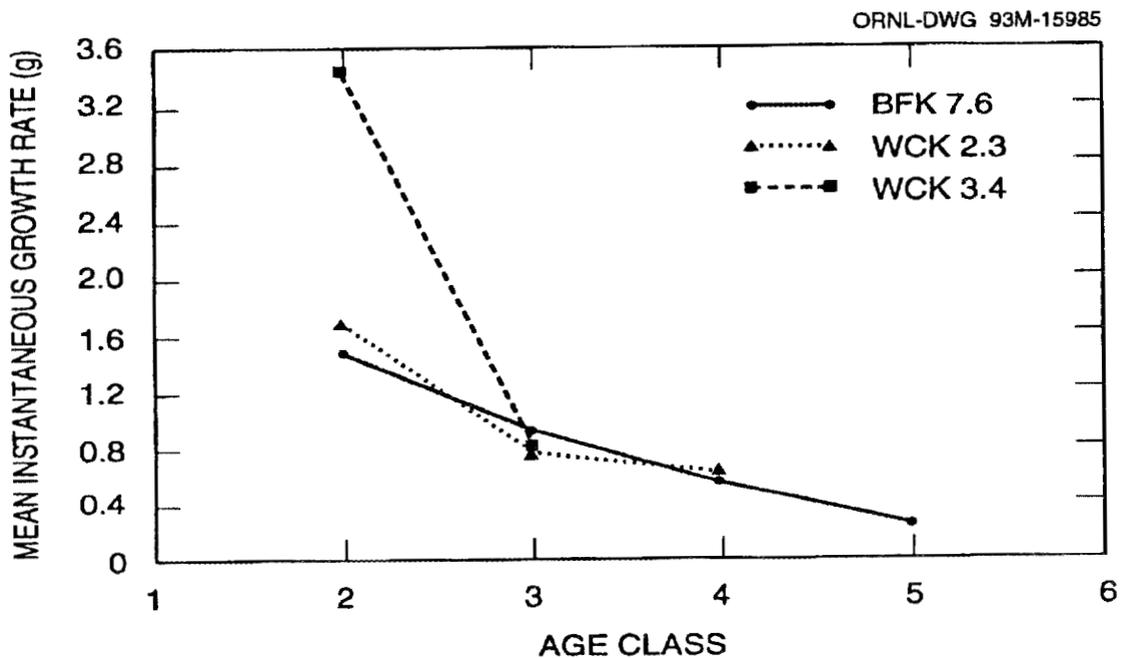


Fig. 6.4. True rate of growth in weight of bluegill in White Oak Creek and a reference stream, Brushy Fork, during 1988. (Note: BFK = Brushy Fork kilometer, and WCK = White Oak Creek kilometer.)

Table 6.6. Comparison between sampling sites on White Oak Creek and the reference stream, Brushy Fork, of mean true growth of redbreast sunfish and bluegill collected between August 1988 and January 1989^a

Species/age class	Sites ^b			
Bluegill				
Age 2	WCK 3.4 <i>n</i> = 1 (3.45 g)	WCK 2.3 <i>n</i> = 17 (1.66 g)	BFK 7.6 <i>n</i> = 10 (1.46 g)	
Age 3	BFK 7.6 <i>n</i> = 13 (0.90 g)	WCK 3.4 <i>n</i> = 1 (0.78 g)	WCK 2.3 <i>n</i> = 4 (0.76 g)	
Age 4	WCK 2.3 <i>n</i> = 1 (0.63 g)	BFK 7.6 <i>n</i> = 5 (0.56 g)		
Redbreast sunfish				
Age 2	WCK 3.4 <i>n</i> = 20 (2.01 g)	WCK 2.9 <i>n</i> = 37 (1.86 g)	BFK 7.6 <i>n</i> = 30 (1.76 g)	WCK 2.3 <i>n</i> = 6 (1.50 g)
Age 3	WCK 2.3 <i>n</i> = 9 (0.94 g)	BFK 7.6 <i>n</i> = 9 (0.91 g)	WCK 3.4 <i>n</i> = 5 (0.86 g)	WCK 2.9 <i>n</i> = 5 (0.76 g)
Age 4	BFK 7.6 <i>n</i> = 16 (0.60 g)	WCK 2.9 <i>n</i> = 1 (0.46 g)	WCK 2.3 <i>n</i> = 2 (0.43 g)	
Age 5	WCK 2.3 <i>n</i> = 1 (0.54 g)	BFK 7.6 <i>n</i> = 12 (0.54 g)		

^aValues connected by the same line are not significantly different ($p > 0.05$) based on Tukey's studentized range test (HSD).

^bWCK = White Oak Creek kilometer, and BFK = Brushy Fork kilometer.

^c*n* = number of fish measured and weighed.

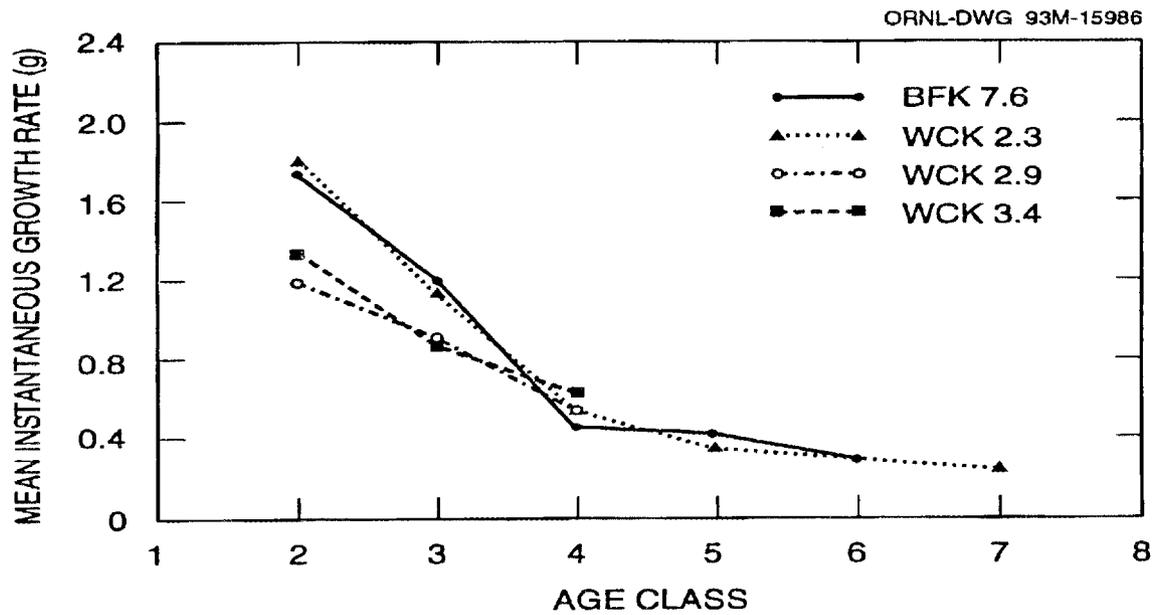


Fig. 6.5. True rate of growth in weight of redbreast sunfish in White Oak Creek and a reference stream, Brushy Fork, during 1989. (Note: BFK = Brushy Fork kilometer, and WCK = White Oak Creek kilometer.)

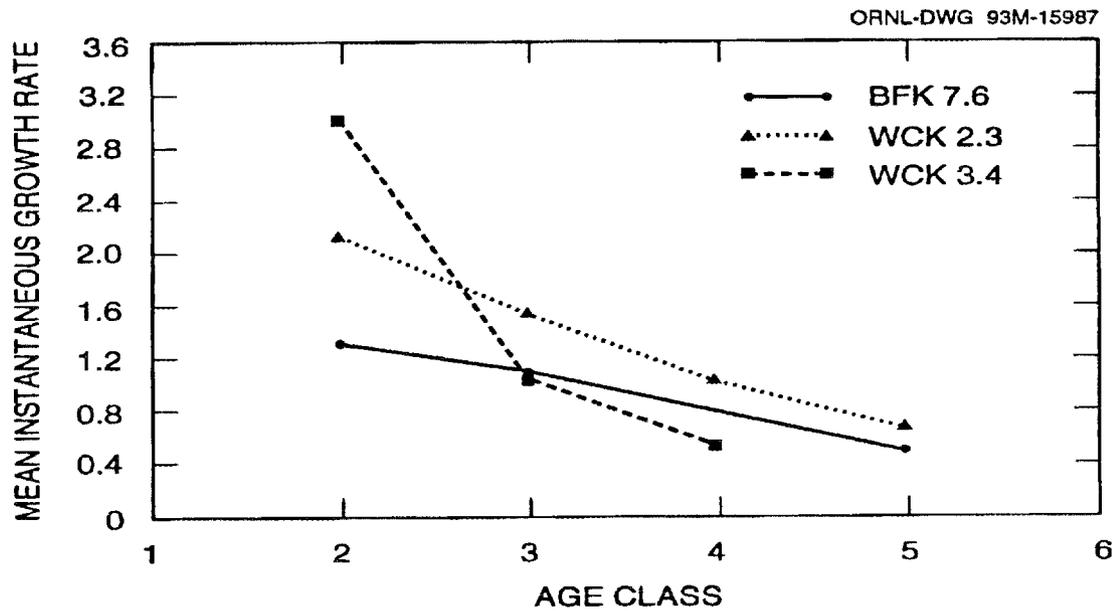


Fig. 6.6. True rate of growth in weight of bluegill in White Oak Creek and a reference stream, Brushy Fork, during 1989. (Note: BFK = Brushy Fork kilometer, and WCK = White Oak Creek kilometer.)

Table 6.7. Comparison between sampling sites on White Oak Creek and the reference stream, Brushy Fork, of mean true growth of redbreast sunfish and bluegill collected between August 1989 and December 1989^a

Species/age class	Sites ^b			
Bluegill				
Age 2	WCK 3.4 <i>n</i> = 5 (3.02 g)	WCK 2.3 <i>n</i> = 20 (2.15 g)	BFK 7.6 <i>n</i> = 10 (1.32 g)	
Age 3	WCK 2.3 <i>n</i> = 18 (1.53 g)	BFK 7.6 <i>n</i> = 1 (1.09 g)	WCK 3.4 <i>n</i> = 5 (1.04 g)	
Age 4	WCK 2.3 <i>n</i> = 5 (1.02 g)	WCK 3.4 <i>n</i> = 1 (0.53 g)		
Redbreast sunfish				
Age 2	WCK 2.3 <i>n</i> = 36 (1.83 g)	BFK 7.6 <i>n</i> = 27 (1.75 g)	WCK 3.4 <i>n</i> = 12 (1.35 g)	WCK 2.9 <i>n</i> = 15 (1.20 g)
Age 3	BFK 7.6 <i>n</i> = 32 (1.21 g)	WCK 2.3 <i>n</i> = 64 (1.16 g)	WCK 2.9 <i>n</i> = 18 (0.93 g)	WCK 3.4 <i>n</i> = 23 (0.90 g)
Age 4	WCK 3.4 <i>n</i> = 10 (0.65 g)	WCK 2.3 <i>n</i> = 13 (0.56 g)	WCK 2.9 <i>n</i> = 5 (0.54 g)	BFK 7.6 <i>n</i> = 4 (0.48 g)
Age 5	BFK 7.6 <i>n</i> = 3 (0.44 g)	WCK 2.3 <i>n</i> = 3 (0.37 g)		

^aValues connected by the same line are not significantly different ($p > 0.05$) based on Tukey's studentized range test (HSD).

^bWCK = White Oak Creek kilometer, and BFK = Brushy Fork kilometer; *n* = the number of fish measured and weighed.

bluegill than would occur in the reference stream.

6.2.3.5 Index of biotic integrity (IBI)

The modified IBI developed for streams the size of WOC in the Clinch River area (Table D.3) was applied to both spring and fall population surveys in 1989. The results indicated that the entire WOC system had been severely impacted by ORNL operations; values ranged from 16 to 30 during both sampling periods (Table 6.8). Using the guidelines given by Karr et al. (1986), all sites ranked as either poor or very poor.

The major deficiency identified by the IBI was the reduced species richness and composition components. Only two sites, WCK 2.3 and WCK 3.4, had values exceeding the lowest possible rating for the first five metrics. No percid or catostomid species and few pollution-intolerant species were found in WOC, and only WCK 2.3 had a total number of species that exceeded the minimum value. In trophic composition, the lack of *Notropis*-type species and percids again resulted in low values for the insectivore metric. A few additional sites had ratings above the minimum for the piscivore component. In some cases, the reduced fauna resulted in higher ratings because few tolerant species were present. As discussed in Loar et al. (1992), the lower species richness was a result of barriers to fish movement within the WOC watershed. Although the barriers (WOD and weirs on WOC and Melton Branch) that separated the WOC watershed from the remainder of the Clinch River system complicated the application of the IBI, they can also be considered a component of the ORNL impact.

In comparison to the modified IBI ratings in 1988, the ratings in 1989 were numerically lower in most cases. This

decrease resulted in minor changes in the descriptive rating, with six sampling site-date combinations receiving a very poor rating (decreased from a rating of poor) and four site-date combinations receiving a poor rating (increased from a rating of very poor). A consistent change in any site-date combination did not occur, suggesting that no major improvement or degradation occurred in WOC as measured by the IBI analysis.

6.2.4 Fish Kills

During 1989, no new fish kills in WOC were observed by ESD personnel during routine sampling. A fish kill first observed on December 2, 1988 (Loar 1993b), was monitored through May 1989, and the additional details are presented in this report.

The fish kill consisted of a slow die-off of fathead minnows in WOC within the plant boundaries. ESD personnel initiated systematic surveys of a 1.7-km reach of stream from the Melton Valley Drive bridge just above sampling site WCK 3.4 to the 6000 Area just below sampling site WCK 5.1. This section included the fish sampling site at WCK 3.9. Systematic surveys were conducted daily beginning December 9, 1988. As the number of dead fish declined, survey frequency was reduced to twice weekly on January 4, 1989, and to weekly on January 20, 1989.

For the period that included the daily and twice weekly surveys, a total of 94 fathead minnows were recovered (mean of 5.5 fish per sampling day). Additional mortality included western mosquitofish (1), blacknose dace (3), crayfish (5), and a water snake. Water quality data indicated that residual chlorine might be responsible for the kill. Data collected by the ORNL EMC Section staff on December 12, 1988, identified three outfalls (Nos. 217, 211, and 218) as sources of high levels of chlorine

Table 6.8. Index of Biotic Integrity values^a based on sampling conducted during March–April and December 1988 and during April and October–November 1989 in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary

Sampling periods	Sites ^b															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
March–April 1988	34 ^c P	32 P	NF ^d	36 P	32 P	26 P	24 VP	NF	26 P	20 VP	24 P	24 VP	24 VP	12 VP	22 VP	30 P
December 1988	30 P	34 P	NF	36 P	36 P	30 P	32 P	NF	22 VP	28 P	32 P	20 VP	22 VP	16 VP	36 P	32 P
April 1989	24 VP	26 P	24 VP	28 P	30 P	22 VP	24 P	NF	20 VP	16 VP	30 P	22 VP	22 VP	NF	26 P	26 P
October–November 1989	26 P	28 P	NF	28 P	30 P	22 VP	22 VP	22 VP	24 P	20 VP	28 P	22 VP	26 P	22 VP	24 P	24 P

^aValues are based on methodologies developed for the Clinch River (Ohio EPA 1987, *Biological Criteria for the Protection of Aquatic Life, Volume III: Standardized Biological Field Sampling and Laboratory Methods for Assessing Fish and Macroinvertebrate Communities*, Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio, and Ohio EPA, 1988, *Biological Criteria for the Protection of Aquatic Life, Volume II: Users Manual for Biological Field Assessment of Ohio Surface Streams*, Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio).

^bFCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

^cNumerical values are on a scale from 12 to 60 with characterization as follows: very poor (VP), poor (P), fair (F), good (G), and excellent (E).

^dNF = no fish taken at the site.

(1.30–1.69 mg/L). Two of these outfalls (Nos. 211 and 217) were also included in toxicity tests to determine if toxicants other than chlorine were present (see Sect.3.1.4). Ambient levels within WOC were also elevated (0.0–0.70 mg/L, mean = 0.41 mg/L) in December 1988 in the approximate section where most of the dead fish were located (WCK 3.9 to WCK 4.1). During the period of chlorine monitoring (December 12–20), 74 dead fish were recovered (mean of 10.6 fish per sampling date). Following this period, increased dilution from above-normal precipitation and high runoff may have reduced the levels of chlorine to less toxic levels, because the total number of dead fish declined to 20 between December 21, 1988, and April 19, 1989, (<1 fish per sampling date). The weekly monitoring was discontinued after June 14, 1989, because eight surveys were made without observing any additional dead fish.

6.2.5 Conclusions

This report presents the results from the fourth year of studies designed to characterize the fish populations of the WOC watershed. In general, the data gathered in 1989 supported trends observed in 1985–88 (Loar et al. 1992; Loar 1993a, 1993b). The drought conditions that characterized much of 1987 and 1988 were followed by above-normal precipitation levels in 1989. Average daily flow in 1989 exceeded that in 1988 in streams without flow augmentation, and several upstream sites returned to continuous flow conditions. These conditions, in turn, affected fish distributions.

Data on species richness in the WOC watershed indicated little change during 1989. The total number of species (13) in the main stem of WOC in 1989 was slightly lower than that reported in 1988 but

appeared to be relatively stable. Only transient species, such as carp (*Cyprinus carpio*) and gizzard shad (*Dorosoma cepedianum*), were not collected.

Population density and biomass values for 1989 supported some of the trends observed in 1985–1988. The fish communities at FFK 0.2 and WCK 3.9, for example, showed evidence of significant impacts, thus reinforcing trends observed during the first 3 years of characterization. Fish were found at FFK 0.2 for the first time in the spring (April) survey, but the collection consisted of only a few large individuals that were most likely displaced downstream by heavy rains. No fish were collected at the FFK 0.2 site in the fall (November) survey. Data associated with a fish kill in a reach of WOC near WCK 3.9 provided an explanation for the low and variable species numbers in that area and in lower Fifth Creek. Episodic discharges of residual chlorine, especially during cold or low-flow periods, resulted in intermittent mortality from December 1988 through April 1989 near WCK 3.9. Until these chlorine discharges and those to Fifth Creek (see Loar 1993b, Table 3.3) are controlled, fish communities in this section of WOC and in lower Fifth Creek will continue to be adversely affected.

The areas of highest biomass and density for most of the period from 1985 to 1988 (FFK 0.4 and WCK 5.1) were again the highest in 1989. Biomass and density at these two sites were as much as four times higher than the fish biomass and density of other area streams and may be related to nutrient enrichment. Density and biomass at WCK 5.1 continued to show recovery from the depressed levels noted in fall 1987 and spring 1988 that may have been associated with chlordane contamination in a small tributary of WOC just above the sampling site. The stable fish community in Melton Branch first observed in 1987 continued, and a relatively stable abundance was noted at

MEK 0.6 and MEK 1.4 in both sampling periods in 1989. Following the increased rainfall in 1989, fish returned to MEK 2.1 in fall 1989. This stable population reflected successful spawning seasons during the summers of 1987 to 1989. The recovery represented by this spawning corresponded well with the shutdown of HFIR in November 1986.

Another site exhibiting a change from 1988 conditions was FCK 0.8 on upper First Creek just below Bethel Valley Road. During 1989, this site recovered from depressed species richness and density levels. With higher flows and cooler water temperatures, several species returned to the site, and the community structure in 1989 approached that of 1985–87.

Except as noted above, the abundance of individual species at most sites in the WOC watershed was similar to that observed in previous years. Blacknose dace was the dominant species, although the abundance of redbreast sunfish and creek chubs was also significant. However, the bluegill populations continued to decline at most sites.

Evaluations of growth of bluegill and redbreast sunfish indicated a few significant differences between sites during 1988 and 1989. The growth rate of age 2+ bluegill was greater at WCK 3.4 than at WCK 2.3 and BFK 7.6 in both years. The higher growth rate may be related to more suitable habitat at WCK 3.4 than at the other sites. The growth rates of age 2+ and 3 redbreast sunfish were significantly greater at WCK 2.3 and BFK 7.6 than at WCK 2.9 and WCK 3.4 during 1989. The growth differences among these sites corresponded to trends found in other fish community parameters.

The IBI methodologies adapted for the Clinch River area supported the conclusions based on the density, richness, and biomass data. Sites WCK 3.9 and FFK 0.2 were the most impacted reaches, while other sites that showed adverse

effects included WCK 2.9, NTK 1.0, NTK 0.3, WCK 3.4, MEK 0.6, MEK 1.4, and MEK 2.1. For the WOC system, the IBI reflected the significant adverse effect of ORNL operations. The combination of intermittent releases of pollutants (e.g., chlorine to WOC) and the isolation by water control structures resulted in a poor rating for WOC. The inclusion of information on trophic composition, tolerance to pollution, and community structure (species richness, composition, and abundance) in an IBI provided another measure of stream "health" beyond that provided by the data on richness, biomass, and density alone. Thus, the agreement between the individual population measures and a more comprehensive analysis (the IBI) indicates that the observed trends are more likely to be of biological significance.

6.2.6 Future Studies

A variety of approaches and short-term experimental studies will be implemented in 1990–91 to assess the impacts on the fish populations of the WOC watershed and to identify specific sources/areas of impact. As a first step, routine, quantitative monitoring of fish density, biomass, and richness at sites in the WOC watershed will be conducted twice annually (spring and fall). The modified IBI will be used on these fish community data, and comparisons will be made with other streams included in the K-25 Site and Y-12 Plant BMAPs. Age determination and growth rate computation of sunfish will be based on scales collected in the fall. The validation program for age determinations from scale analyses will continue to use otoliths and scales from sunfish collected in the bioaccumulation and the bioindicator monitoring programs (Sects. 4 and 5 respectively). Production will be estimated using the procedure of Garman and

Waters (1983). The failure of condition factors to show site differences will be further examined (e.g., by using the calculation of estimates of ordinary least-squares regression parameters, as suggested by Cone 1989). Site-specific trends may be apparent with such an approach.

Examination of the WCK 3.9 and FFK 0.2 sites for additional evidence of a relationship between chlorine levels and fish kills or low population levels will also continue. Periodic surveys and more frequent chlorine monitoring may be used to address this problem.

A short-term experiment will be performed to test the effect of canopy removal on the fish community and associated trophic interactions. A small stream on the ORR comparable to sections of WOC or its tributaries will be subjected to the selective thinning of the riparian cover. Cages of plexiglass and mesh will be positioned in these cleared areas and in normally shaded areas of the stream. After normal colonization by periphyton and benthic invertebrates has occurred, blacknose dace and/or snails will be added to the cages. The dace and snails will be measured for growth, examined for changes in diet, and observed for competitive interactions as a result of changes in the amount of sunlight reaching the stream. This study may provide some insight into potential dietary causes of differences among the fish communities in portions of the WOC watershed.

A longer-term study is planned to address the isolation of the WOC watershed from the rest of the Clinch River system. The barriers to fish recolonization may be sidestepped by reintroducing selected species and monitoring their population dynamics. Historically, several species common to the area were known to occur in WOC. Knowing that the WOC system is currently capable of supporting species that occurred in the system historically would provide

better documentation of the ecological recovery process.

6.3 INTERPRETATION OF BIOTIC CHANGES (*M. A. Huston*)

6.3.1 Introduction

The objective of this study (Subtask 4c of the BMAP, as discussed in Loar et al. 1986) is to develop a methodology, based on ecological theory and data from a large number of appropriate reference streams within the region, for evaluating the ecological condition of various sites in the WOC watershed and for assessing the significance of changes in those conditions that occur following implementation of remedial actions.

The first phase of the study was to assemble a data base that will help place the WOC watershed in a regional context and allow each site to be compared to streams of similar size and structure that have been less severely affected by human activity. No single stream can serve as an adequate reference (or control), since every stream differs in some way from every other. The theoretical framework chosen for analysis of the regional data base is the dynamic equilibrium theory of community structure (Huston 1979). This theory is based on the balance between the opposing forces of competition, which tends to cause local extinction and a reduction of species diversity, and mortality-causing disturbance, which can either increase diversity by preventing extinction from competition, or reduce diversity by killing most species.

In the context of this theory, the data can be examined in relation to two axes based on physical properties of streams: the watershed area draining into the stream at any particular site and the gradient of the stream at that site. The environmental stability of a site is related

to watershed area, since streams draining small watersheds have much greater variability in flow conditions than do large rivers. Site productivity is also related to watershed area, since larger streams tend to have higher levels of nutrients. Streams of intermediate size tend to have the highest productivity because they are large enough to be unshaded by trees near their center, but are not so large that heavy silt loads prevent light from penetrating into the water column. Site stability is also related to stream gradient, since the scouring action of water is greater in high gradient streams.

A previous report (Loar 1993a) presented an analysis of fish community structure in WOC in relation to regional patterns in the Tennessee Valley. This report describes some of the patterns of benthic insect community structure in WOC in relation to other streams in the Tennessee Valley.

6.3.2 Methods

The data base of fish and insect community structure was completed in collaboration with Dr. David Etnier, The University of Tennessee, Knoxville, and the BMAP staff. The principal source of data was a series of TVA studies of streams and valleys of the Tennessee Valley region—7 river systems and a total of 107 sites were studied between 1968 and 1972. In addition to data from collections of fish and benthic insects, the data base also includes measures of stream size, structure, and water quality. Measurements of watershed area and stream gradient were taken from a complete set of TVA 15-min quadrangles. These types of physiographic data allow comparisons between stream reaches that have not been thoroughly characterized by site studies. The data base has been augmented by data from additional studies, including local BMAP

reference sites. This report is based on mean values per sample from White Oak Creek and Melton Branch during the period of May 1986 through August 1987 (Smith 1993b).

To illustrate some of the general patterns in the regional data base, streams were compared on the basis of watershed area and stream gradient. Together, these two parameters characterize a wide range of stream conditions. Although there is a significant correlation between watershed area and stream gradient (i.e., rivers, which have large watersheds, tend to have much lower gradients than streams with smaller watersheds), there is sufficient variation in these two parameters to form a useful two-dimensional system for classifying streams.

In all of the graphs, each site is located on a two-dimensional space defined by the log (base 10) of the watershed area at the sampling point, and the log of the stream gradient ($\times 10,000$ for convenience of scaling) at the sampling point. Each site is represented by a circle that is scaled to indicate the magnitude of a particular parameter (e.g., insect density or number of taxa) at that site in relation to all other sites on the graph. In all graphs, the sites from the TVA river study (Buffalo, Duck, Emory, Flint, Powell, Sequatchie, and upper Little Tennessee Rivers) are represented by thin circles, and sites from WOC are represented by thick circles with site labels. Visual examination of data from WOC demonstrates those sites and parameters that differ from regional patterns.

6.3.3 Results and Discussion

Data presented here focus on a subset of the benthic macroinvertebrate community of WOC that has been selected as an index of the "health" of the streams. Insects of the three orders Ephemeroptera,

Plecoptera, and Trichoptera are composed primarily of taxa that are intolerant of low water quality (Lenat 1988). Smith (1993b) used these three taxa together (abbreviated EPT) to show conclusively that some sites on all WOC tributaries are impacted relative to their upstream reference sites, on the basis of having fewer EPT taxa. Comparison of EPT data from sites in the WOC watershed with those in the regional data base clearly demonstrated some anomalies in the local benthic macroinvertebrate communities in relation to the regional picture. In addition, data on another group of macroinvertebrates, the family Chironomidae of the order Diptera, also suggest some deviations from expected trends that are consistent with EPT patterns.

6.3.3.1 Patterns of Ephemeroptera, Plecoptera, and Trichoptera, Chironomid, and total density

Density refers to the total number of individual organisms of any particular taxa that are found within a particular area of stream bottom, generally 0.09 m² (see Smith 1993b for methods description). Total density for all taxa (Fig. 6.7) shows that some WOC sites have a higher density than would be expected on the basis of regional trends in the watershed area and stream gradient. Part of this difference is probably due to methodological differences between the TVA study and the BMAP sampling of WOC. The WOC collections probably recovered a higher proportion of the total number of organisms in each sample because of intensive sample sorting methods and possibly the use of finer mesh nets for the actual sample collection (see discussion of methods in Smith 1993b). While a high total density could indicate that the WOC sites are enriched in nutrients as a result of non-point source pollution, it is likely that most of the difference is an artifact of

different sampling methods. The accuracy of the WOC sample analysis is also reflected in the much higher density of chironomids recovered in the WOC samples as compared to the TVA samples (Fig. 6.8). Chironomid larvae in stream samples are extremely small, wormlike organisms that are very difficult to see amid the debris of a stream-bottom sample. While chironomid densities are often high in polluted or fertilized areas, it is likely that most of the difference in chironomid density between the WOC and TVA samples is the result of more careful sorting of the WOC samples or of differences in the mesh size of the collection nets.

The EPT densities present a somewhat different picture. These organisms are generally larger and easier to sample than chironomids, so higher levels at the WOC sites are likely to reflect real differences in density rather than methodological differences. Lower levels at the WOC sites would almost certainly reflect real differences. All of the WOC watershed reference sites showed densities similar to or higher than the regional densities, indicating that populations of these taxa are apparently healthy and the sites are not impacted. In general, the densities of Ephemeroptera at the WOC sites with larger drainage areas are lower than expected based on the Tennessee Valley samples (Fig. 6.9); the densities of Plecoptera are similar to what would be expected (Fig. 6.10); and the densities of Trichoptera are much higher at WCK 2.3 and FFK 1.0 than in similar streams in the region (Fig. 6.11). These data may reflect real differences in the trophic structure of the benthic macroinvertebrate community in WOC, which is an aspect of community structure that is sensitive to disturbances, toxic stresses, and enrichment from nutrient input. The contrasting patterns in these three groups also illustrate the information that may be lost by using a composite index, such as total EPT taxa.

ORNL-DWG 94M-15988

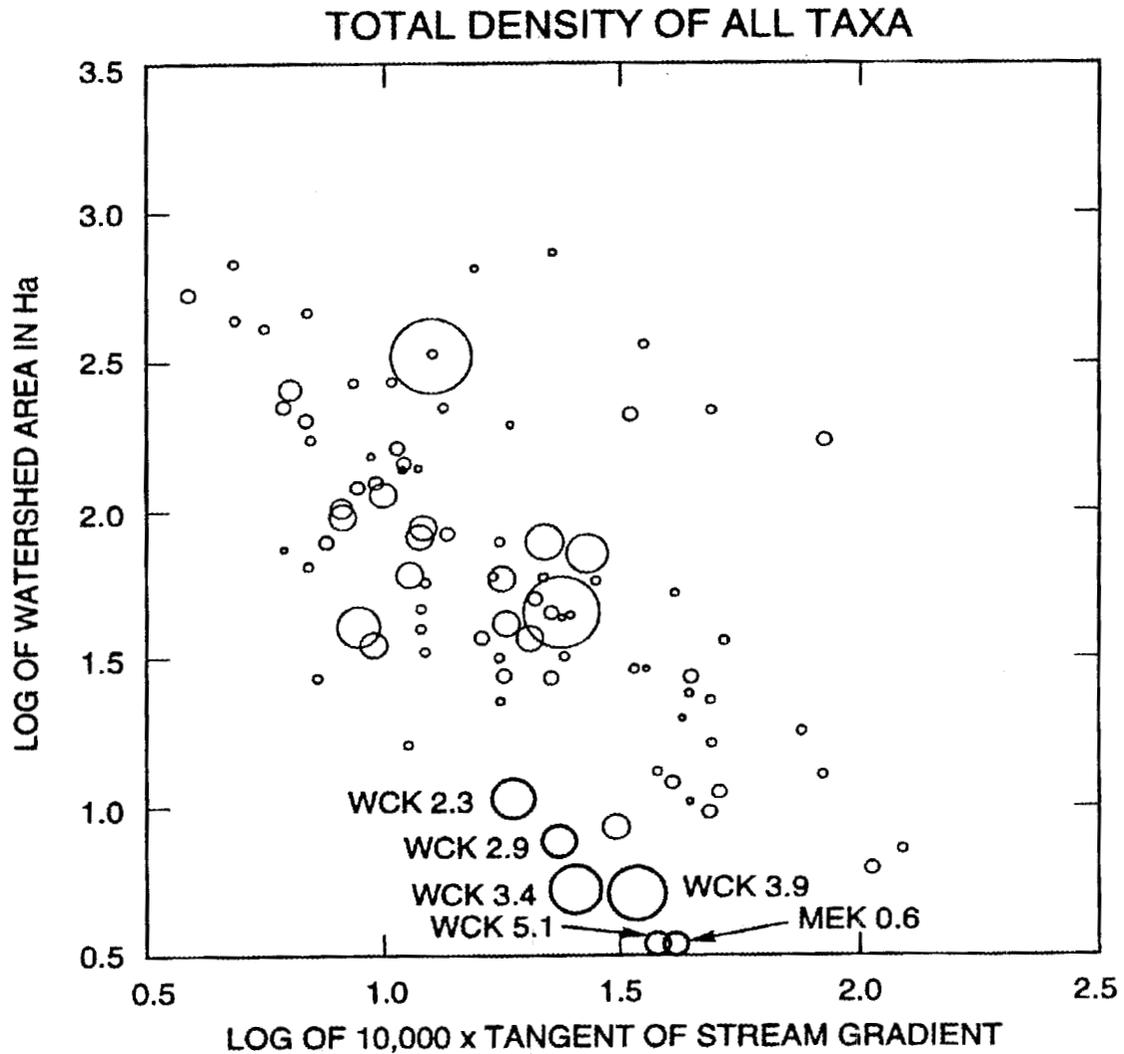


Fig. 6.7. Relative density of all benthic invertebrates at sites in the Tennessee Valley regional data base and White Oak Creek (WOC) sites. Density is the number of individual insects found in 0.09 m² of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. Sites on WOC are drawn with heavy lines and labeled. WCK = White Oak Creek kilometer, MEK = Melton Branch kilometer.

6.3.3.2 Patterns of EPT taxonomic richness

The total number of EPT taxa at sites in the WOC drainage indicate that most sites within and downstream of ORNL

have a lower number of EPT taxa than the upstream reference sites, and the regional pattern also indicates that EPT richness at most of the downstream WOC sites is much lower than would be expected (Fig. 6.12). The patterns within each of

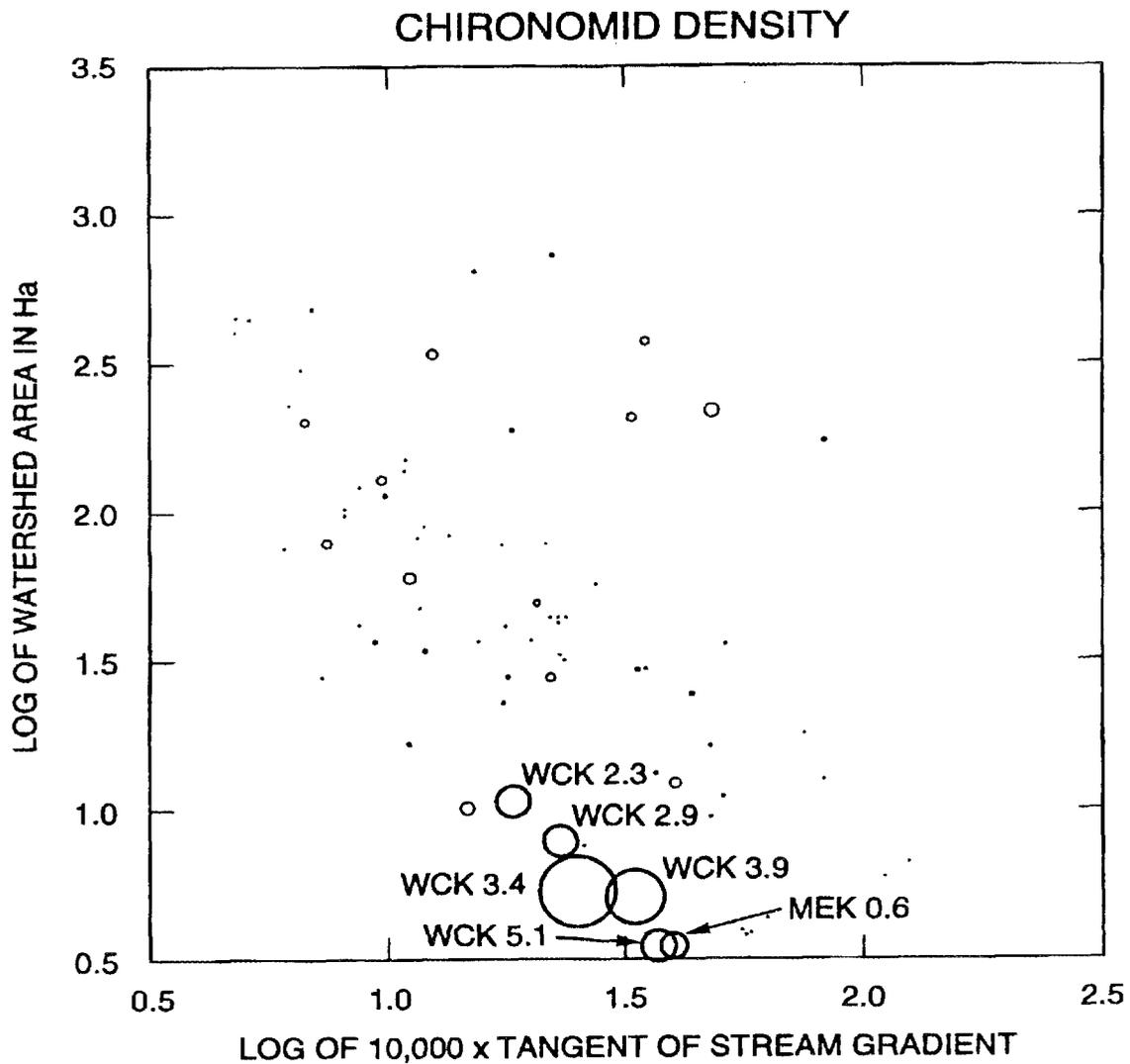


Fig. 6.8. Relative density of Chironomids at sites in the Tennessee Valley regional data base and White Oak Creek sites. Density is the number of insects found in 0.09 m² of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. Sites on WOC are drawn with heavy lines and labeled. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer.

the individual orders show roughly the same trends as the density data. The total number of Ephemeroptera taxa at sites in the WOC watershed (except the reference sites) is lower than at sites with similar drainage areas and stream gradients within the region (Fig. 6.13). The same is true

for Plecoptera (Fig. 6.14). Only the Trichoptera have taxonomic richness approaching that expected for streams this size (Fig. 6.15).

These results demonstrate that some aspects of benthic macroinvertebrate community structure in WOC depart

ORNL-DWG 93M-15990

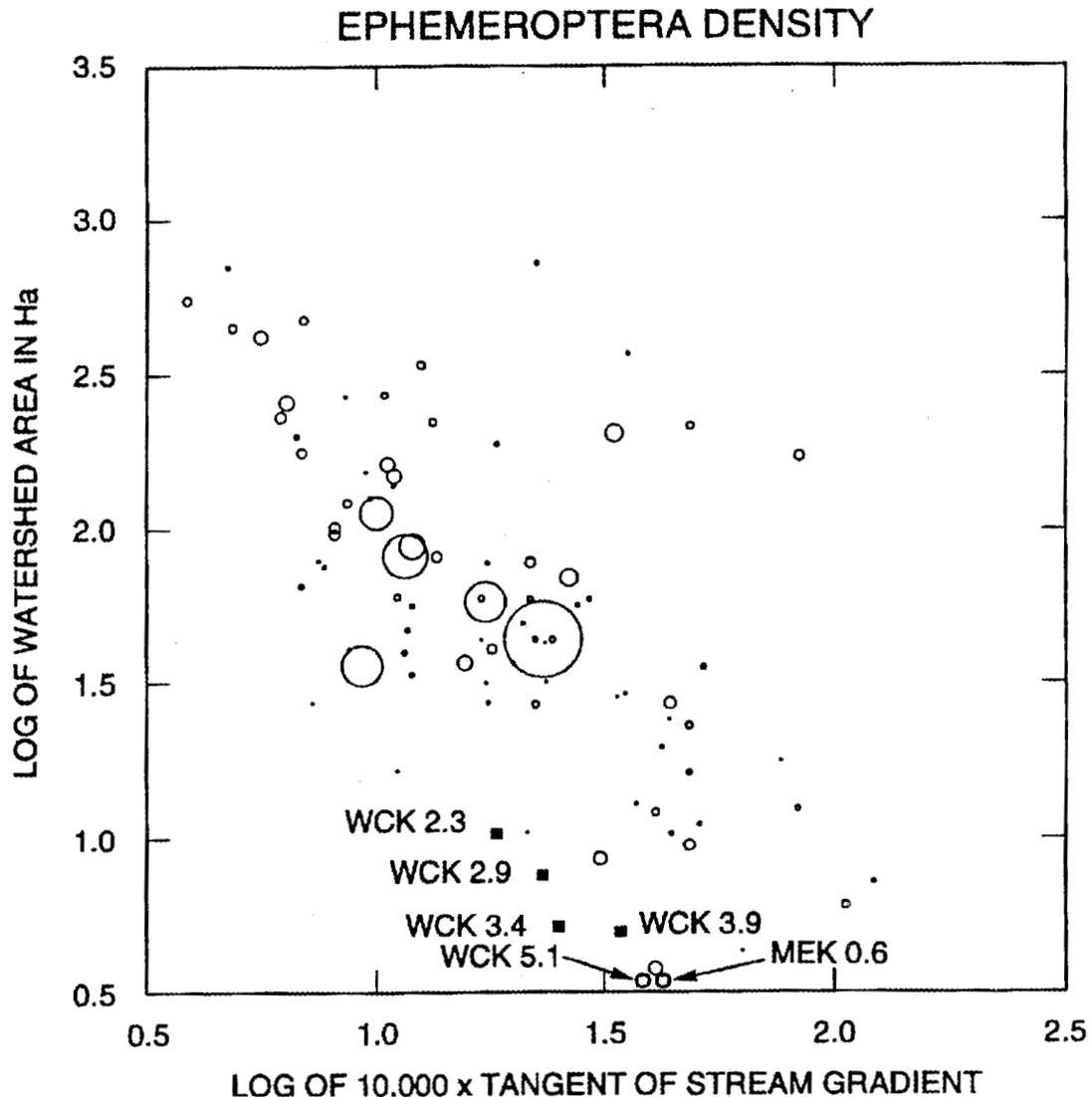


Fig. 6.9. Relative density of Ephemeroptera at sites in the Tennessee Valley regional data base and White Oak Creek sites. Density is the number of individual insects found in 0.09 m^2 of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer. ■ indicates mean densities of less than 0.1 individual per sample.

significantly from the patterns found in other streams in the Tennessee Valley region. Specifically, the low taxonomic richness of the sensitive taxa (Ephemeroptera and Plecoptera) indicates that the entire stream system below the upstream reference (WCK 6.8 and MEK 2.1) sites is significantly impacted.

An alternative hypothesis is that WOC differs in some fundamental way from other streams in the Tennessee Valley, so that the differences reflect the results of natural processes of stream community organization. The only sensitive order with taxonomic richness comparable to other sites in the Tennessee Valley data base is

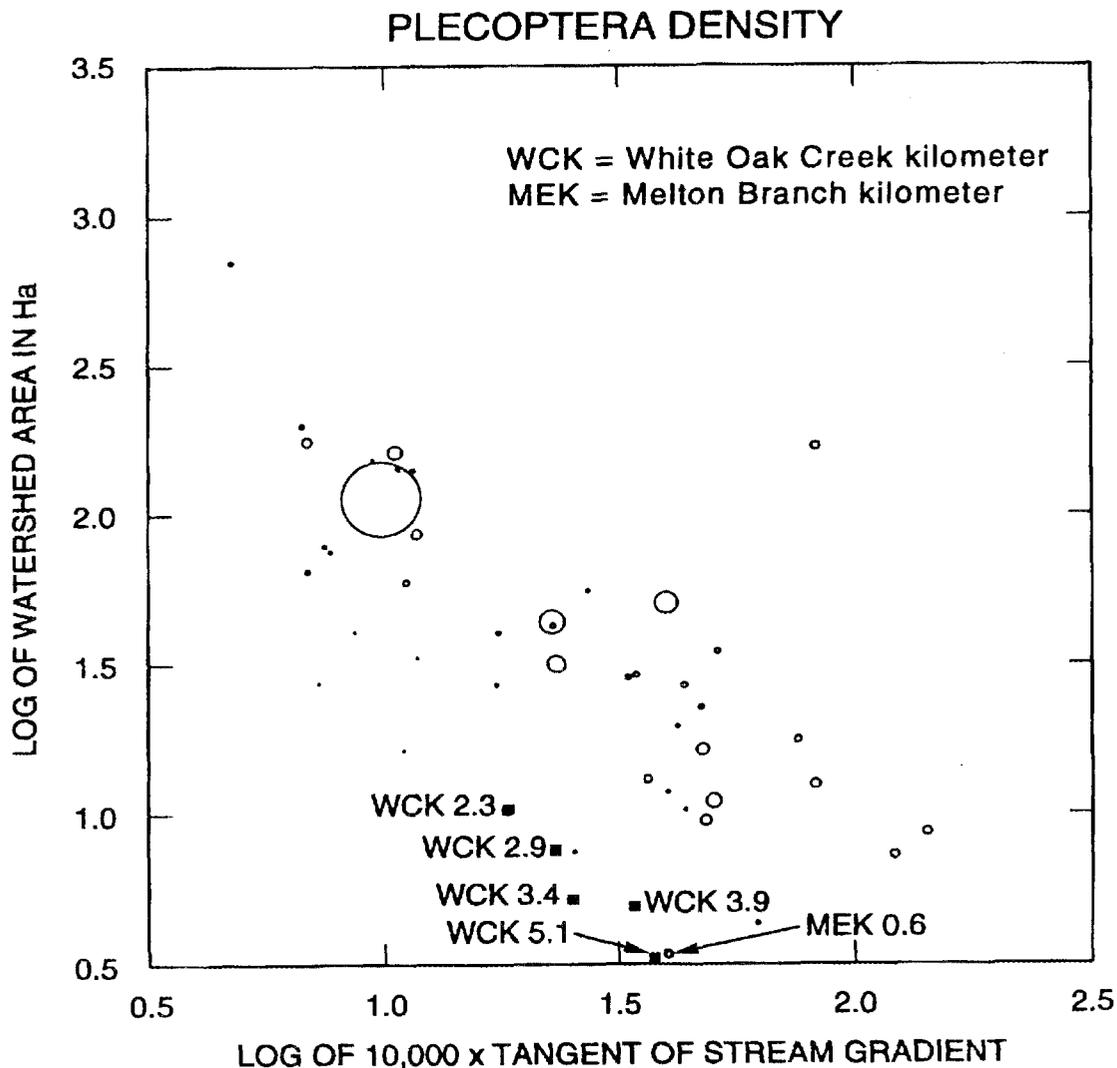


Fig. 6.10. Relative density of Plecoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites. Density is the number of individual insects found in 0.09 m² of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer. ■ indicates mean densities of less than 0.1 individual per sample.

the Trichoptera, which has unusually high densities at several sites in WOC. However, not all species of Trichoptera are highly sensitive. Those abundant in WOC are genera that are relatively insensitive to pollution (e.g., *Hydropsyche* and *Cheumatopsyche*).

This situation suggests that the trophic structure of the insect community is different from that found in most streams in the larger region. The high density of Trichoptera at some sites may result from nutrient enrichment that increases their food supply, or from some other impact

ORNL-DWG 93M-15992

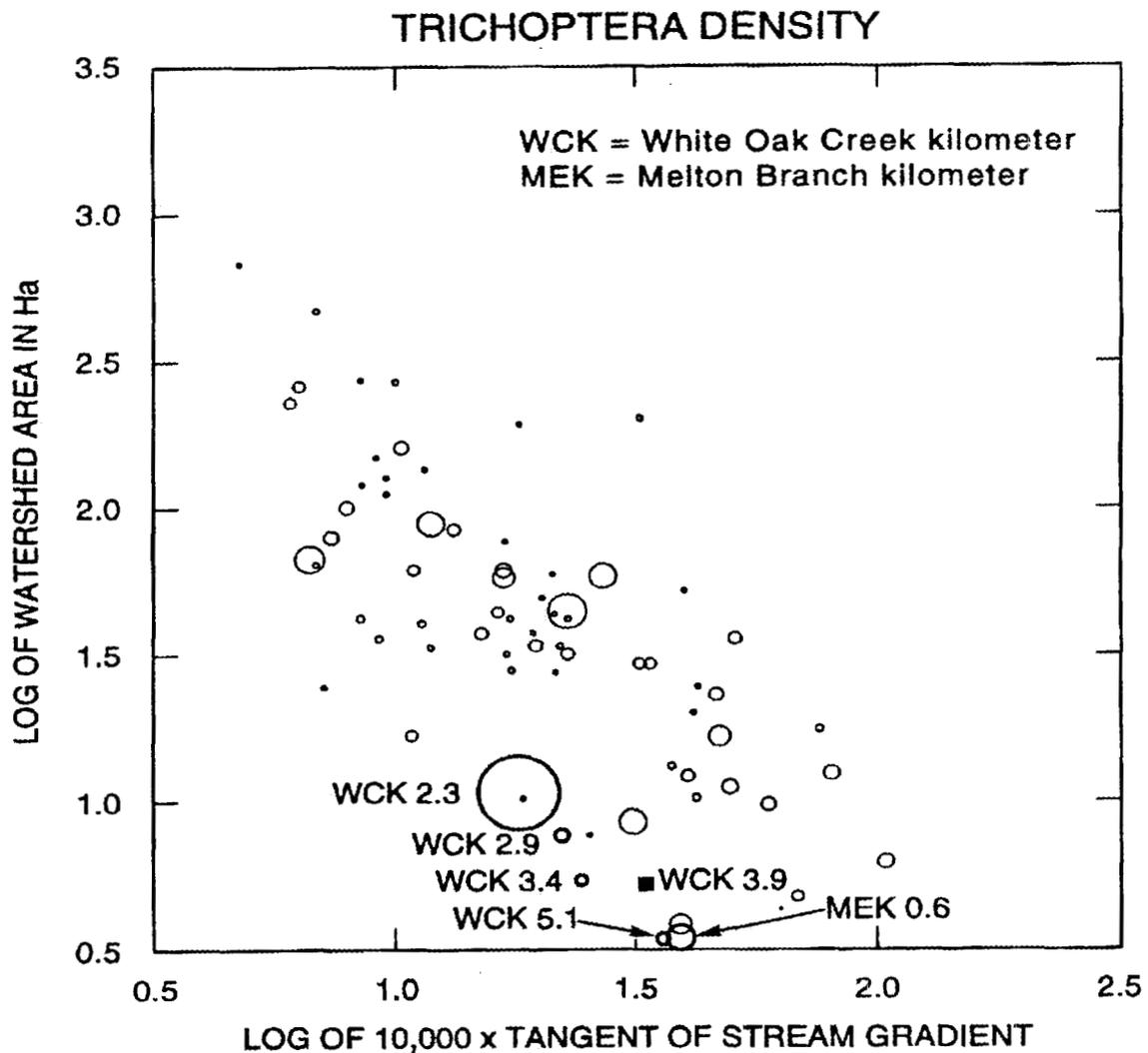


Fig. 6.11. Relative density of Trichoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites. Density is the number of individual insects found in 0.09 m^2 of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer. ■ indicates mean densities of less than 0.1 individual per sample.

that alters trophic structure or removes predators that would normally keep them at lower densities. According to the dynamic equilibrium hypothesis, the combination of normal levels of diversity with high levels of productivity (inferred from the high density of Trichoptera)

suggests that either frequent disturbances or stresses are acting to keep diversity from changing, or the Trichoptera species are resistant to the stresses that are reducing the Ephemeroptera and Plecoptera populations. Whether the species richness of Trichoptera would be expected to

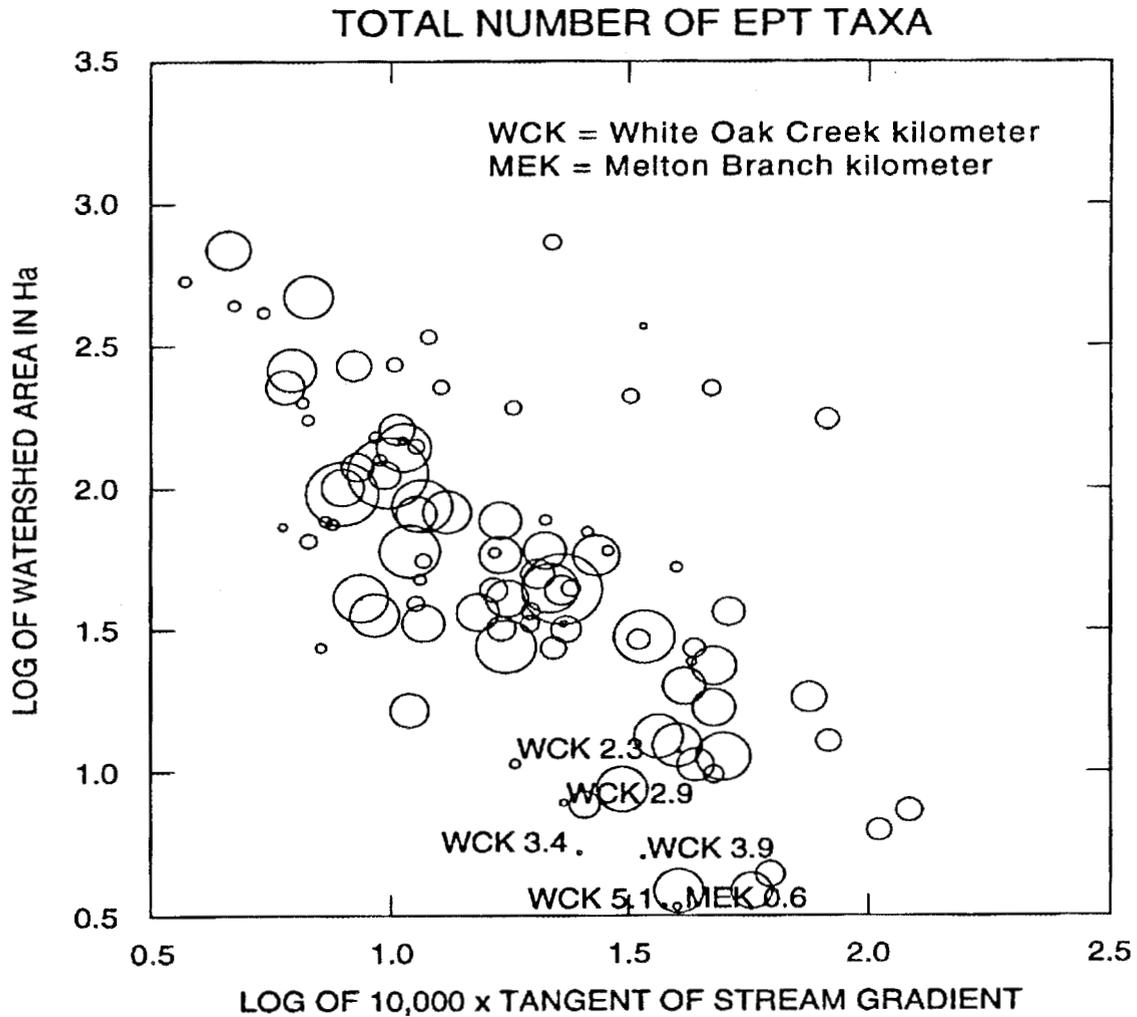


Fig. 6.12. Relative benthic macroinvertebrate taxonomic richness of combined EPT taxa (orders Ephemeroptera, Plecoptera, Trichoptera) at sites in the Tennessee Valley regional data base and White Oak Creek sites. Taxonomic richness is the number of different taxa of each order that were found in a 0.09 m² sample of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer.

increase or decrease with increasing productivity depends on the nature of competitive interactions within this group, which cannot be determined without further study. In either case, a change would be expected. The relatively normal

level of Trichoptera richness in comparison to other streams suggests that some factor is influencing the Trichoptera populations in a different way than it affects the other EPT taxa.

ORNL-DWG 93M-15994

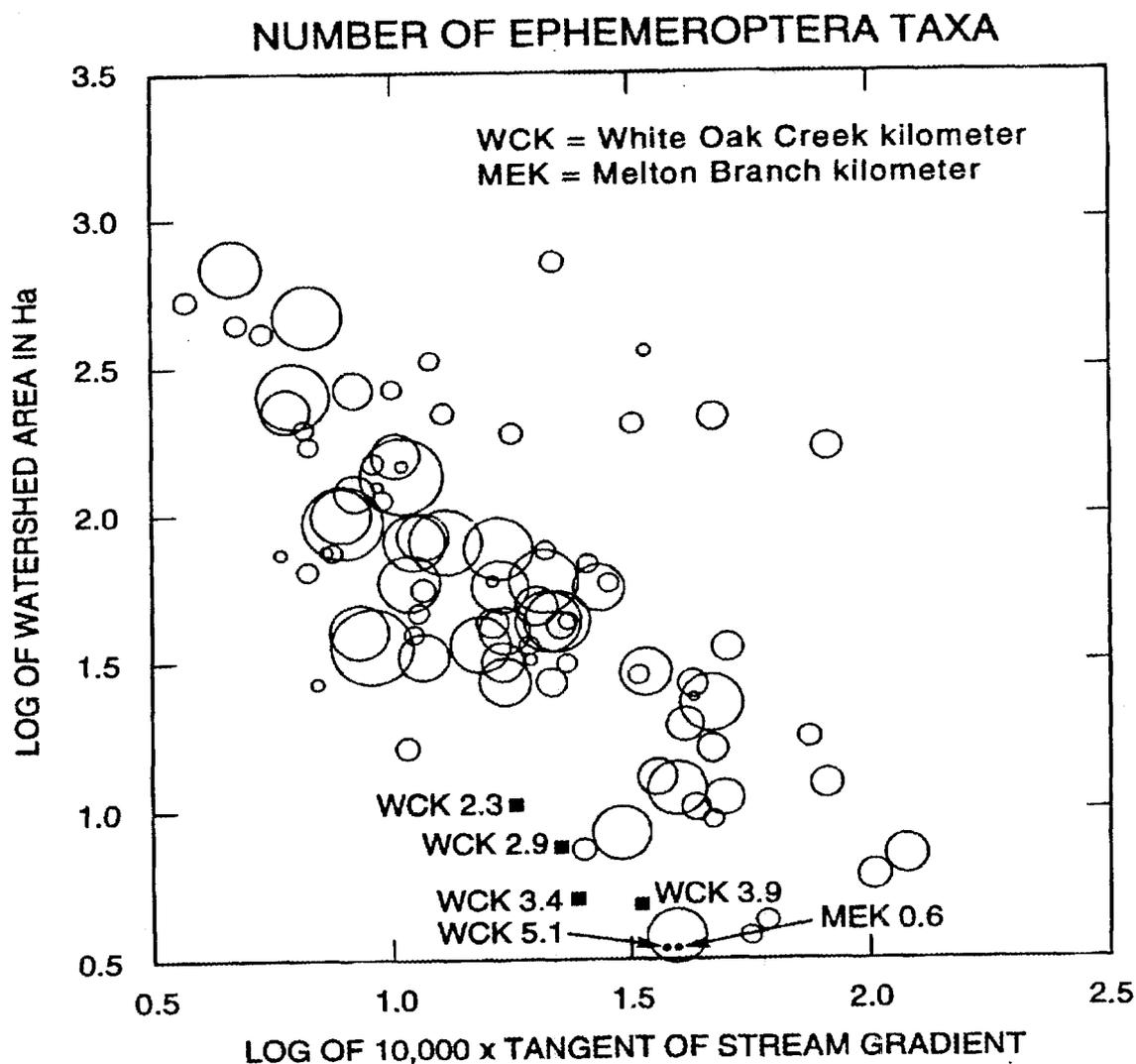


Fig. 6.13. Relative benthic macroinvertebrate species richness of the order Ephemeroptera at sites in the Tennessee Valley regional data base and White Oak Creek sites. Density is the number of individual insects found in 0.09 m^2 of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer. ■ indicates mean densities of less than 0.1 individual per sample.

6.3.4 Future Studies

The sensitivity of benthic invertebrates as an indicator of stream health, and the accumulating body of data that is making the WOC watershed the best studied

system of benthic insect communities in the United States, suggest that further analysis is warranted to understand the differences in species richness among sites in WOC and among WOC and other sites in the Tennessee Valley. Data on additional sites

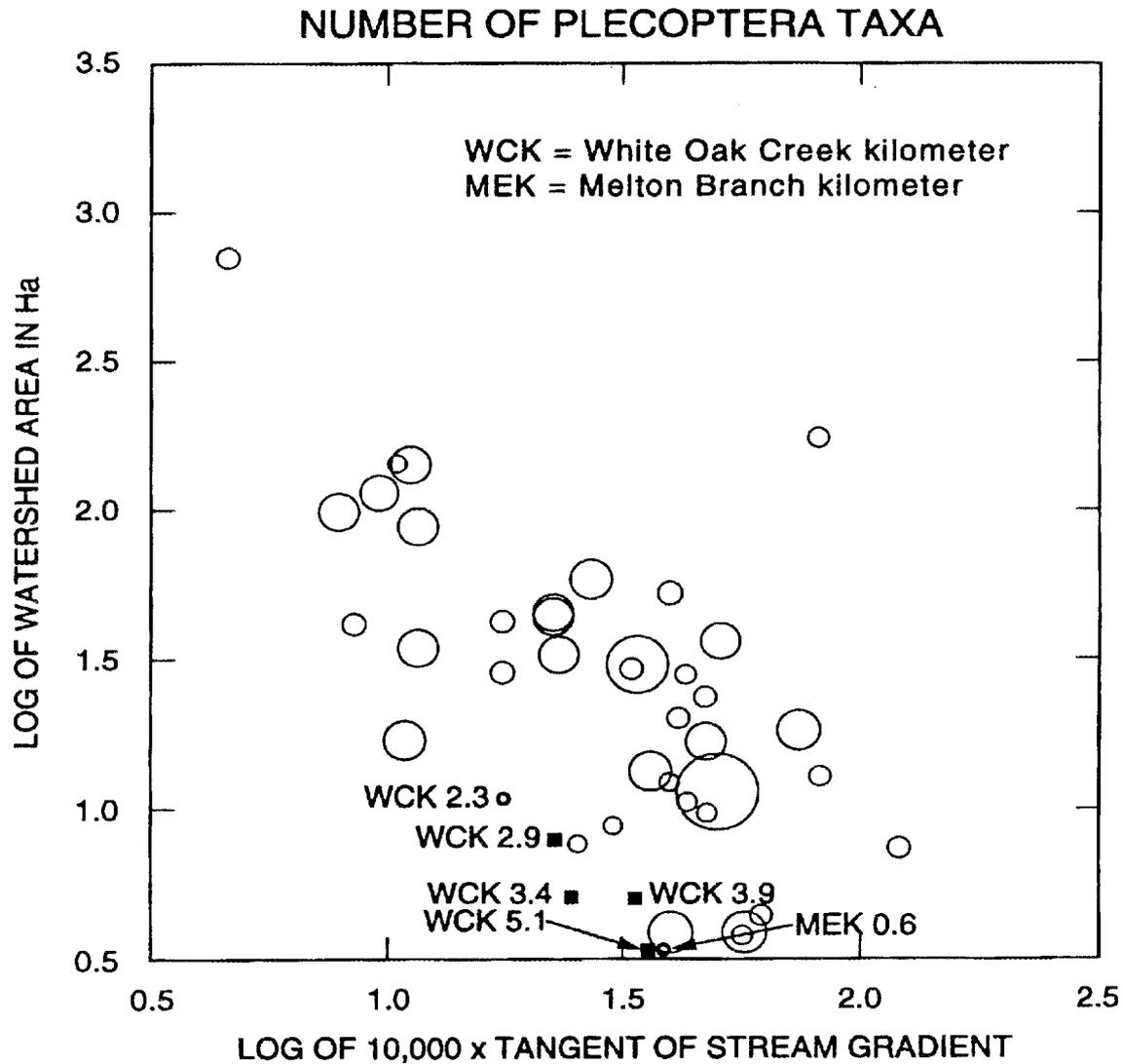


Fig. 6.14. Relative benthic macroinvertebrate species richness of the order Plecoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites. Density is the number of individual insects found in 0.09 m² of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer. ■ indicates mean densities of less than 0.1 individual per sample.

will be added to the regional data base, and efforts will continue to thoroughly analyze the results from BMAP monitoring and research activities. Efforts to date on the interpretation of biotic changes

indicate that the theoretical framework and use of a comparative regional data base are appropriate methods for interpretation of instream ecological monitoring results.

ORNL-DWG 93M-15996

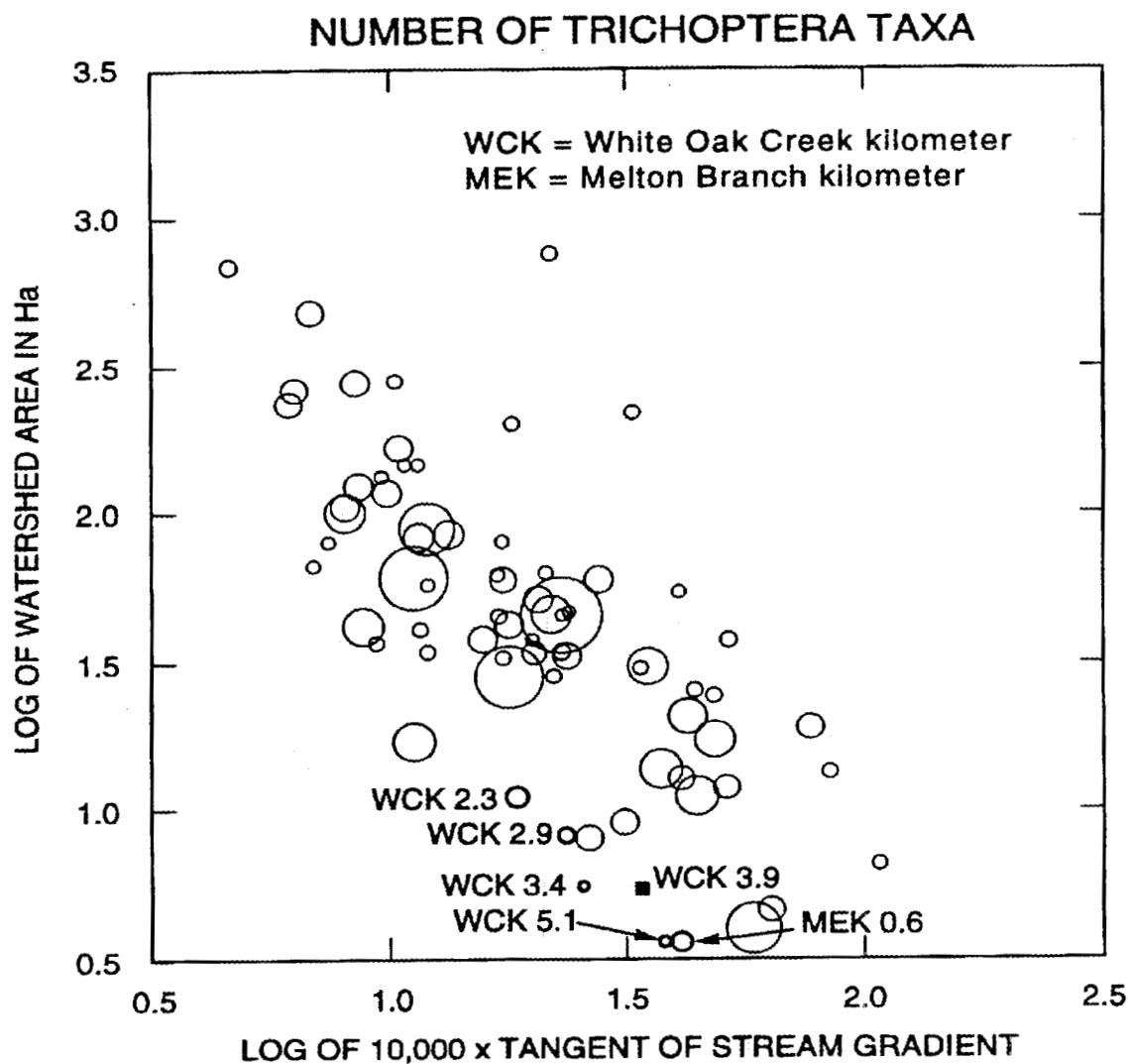


Fig. 6.15. Relative benthic macroinvertebrate species richness of the order Trichoptera at sites in the Tennessee Valley regional data base and White Oak Creek sites. Density is the number of individual insects found in 0.09 m^2 of stream bottom. Each circle is scaled in relation to the site with the highest density, so the density per site decreases proportionately with decreasing circle size. WCK = White Oak Creek kilometer; MEK = Melton Branch kilometer. ■ indicates mean densities of less than 0.1 individual per sample.

7. ASSESSMENT OF CONTAMINANTS IN THE TERRESTRIAL ENVIRONMENT

B. T. Walton

7.1 GUIDANCE FOR BIOLOGICAL MONITORING IN THE TERRESTRIAL ENVIRONMENT (*B. T. Walton*)

This year marked the conclusion of field sampling to evaluate the potential exposure of terrestrial biota to hazardous chemicals released to the ORNL environment. Although all sample analyses are not complete for all studies at this time, sufficient data are available to provide general guidelines for future screening and biological monitoring studies at ORNL.

7.1.1 Vegetation

Neither heavy metals nor organic compounds have emerged as contaminants of potential concern to terrestrial vegetation at ORNL; however, three radionuclides found in soils at ORNL SWSAs, ^3H , ^{90}Sr , and ^{99}Tc , have the potential for marked uptake by terrestrial vegetation. All three of these radionuclides were studied at ORNL during the course of BMAP. Garten et al. (1986) reported that the highest average ^{99}Tc concentrations in vegetation occurred in herbaceous plants (18.5 nCi/g dry weight). Furthermore, Garten (1987) used a simulation model to conclude that if ^{99}Tc input to a contaminated forested area at ORNL equaled export, then 42% of the total ^{99}Tc could be expected to be retained by the forest over three decades. Analyses of tree cores from pines at SWSA 5 (Amano et al. 1987) indicated that tritium

migration from the burial grounds continued during the last decade. Additional studies at SWSA 5 showed that the maximum concentration of ^{90}Sr found in herbaceous vegetation was 90 nCi/g dry weight (Garten and Lomax 1987) and averaged as high as 39 and 19 nCi/g in honeysuckle and blackberry shoots, respectively, at two seeps. The authors concluded that these concentrations of ^{90}Sr were sufficiently high to enable deer that browse regularly on contaminated vegetation to exceed the limit of 30 pCi/g in bone that was established for the confiscation of deer harvested during the managed hunts that are held on ORR each fall.

Collectively, these studies provide much needed information on the actual concentrations of ^{99}Tc , ^3H , and ^{90}Sr in terrestrial vegetation at ORNL. These data can be used to (1) model changes in standing compartments of radionuclides in the terrestrial environment at ORNL, (2) document changes in radionuclide exposures over a prolonged period of time, and (3) conduct pathway analyses for risk assessments at these sites.

Recent radiological monitoring at ORNL indicates that ^{90}Sr may be found in vegetation at sites other than those identified in the preceding studies. Beta-gamma measurements up to ~70 mrad/h in an elm growing west of the Trench 7 access road have been reported (J. K. Williams, Health and Safety Research Division, ORNL, unpublished data). Leaves of this tree measured ~3 mrad/h. A small maple tree near Pit 4 producing beta readings of 0.13 mrad/h

contained 9400 pCi/g gross beta and 4900 pCi/g total strontium (M. S. Uziel, ORNL, unpublished data). Leaves from a wild cherry tree near the same pit contained 300 pCi/g gross beta and 0.30 pCi/g ^{137}Cs . Neither ^{137}Cs nor ^{60}Co is expected to reach appreciable concentrations in vegetation based on their low transfer coefficients from soil. For example, Dahlman and Van Voris (1976) measured plant-soil ratios of 0.03 where ^{137}Cs concentrations ranged in excess of 20,000 pCi/g soil. Similarly, Crossley (1969) found concentration factors of 0.027 and 0.058 for ^{137}Cs and ^{60}Co , respectively. All of these studies were conducted on the ORR near ORNL, where illitic clays have been shown to reduce the bioavailability of ^{137}Cs (Francis and Brinkley 1976). Nonetheless, the recent reports of radioactivity in vegetation in the pits and trenches area of Waste Area Grouping 7 indicate that additional monitoring should be initiated to evaluate the magnitude of radionuclide transport to aboveground vegetation and to better estimate the potential environmental and health effects of fires and site management activities. Furthermore, analysis of vegetation for ^3H , ^{90}Sr , and ^{99}Tc is advisable after remedial actions have been implemented at sites where contamination has been documented.

7.1.2 Small Mammals

Small mammals were recognized in the early phases of BMAP as a potentially important source of information about contaminants in the terrestrial environment at ORNL. Monitoring studies using small mammals were carried out at selected locations on the ORR to address specific questions about the kinds of contaminants present and their availability to terrestrial mammals. Furthermore, these studies were conducted with a critical view to how the

success of monitoring studies using small mammals could be improved. The following is a summary of these findings as they relate to monitoring activities at ORNL.

Full details of the small mammal studies at ORNL are presented in Talmage and Walton (1990). These studies showed that ^{90}Sr is biologically available to small mammals inhabiting SWSA 4, as evidenced by elevated concentrations of ^{90}Sr in bone of some species. This finding corroborates the conclusions of Garten and Lomax (1987), based on theoretical considerations, that SWSA 4 is a potential source of ^{90}Sr contamination to wildlife that frequent the site. Mercury was present but not elevated in small mammals trapped in SWSA 4 and along WOL. Similarly, no indication was found of significant benzo[*a*]pyrene (B[*a*]P) concentrations in small mammals in the ORNL environs, based on results of a hemoglobin-adduct assay. Thus, no evidence was obtained from the small mammal studies to indicate that either mercury or PAHs are present in quantities above background at the ORNL sites studied.

Based on the small mammal studies at ORNL and an extensive review of the literature conducted as part of this BMAP task, trophic level was found to be positively correlated with contaminant uptake by small mammals (Talmage and Walton 1989). For example, the studies at ORNL revealed that the white-footed mouse, *Peromyscus leucopus*, was the most abundant small mammal species at all study sites, yet this species proved to be a poor indicator of the presence of Hg, ^{90}Sr , or B[*a*]P. Conversely, the short-tail shrew, *Blarina brevicauda*, which is primarily a carnivore, was a good indicator of all three contaminants, but is less abundant and more difficult to capture than *P. leucopus*. Carnivores generally accumulate the highest concentrations of contaminants, followed by omnivores with intermediate

concentrations, and herbivores with the lowest concentrations (Talmage and Walton 1991).

The monitoring studies of small mammals in the vicinity of ORNL are useful not only for the information provided about specific contaminants in the ORNL environs but also for the opportunity to evaluate the merit of small mammal studies in monitoring programs. Despite the potential advantages to using natural populations of small mammals to evaluate hazardous waste sites, Walton and Talmage (1989) found numerous difficulties with this approach. The use of field-collected animals for toxicological evaluations presents problems concerning the species selection, adequate sample sizes, endpoint selection, multiple contaminants, and the inherent variability of natural populations. In some instances, these limitations are insurmountable, and the use of small mammals for monitoring is contraindicated. Of special note is that field studies of the population dynamics of small mammals rarely yield definitive evidence of chemical impact (Walton and Talmage 1989), suggesting that recent guidelines (Warren-Hicks et al. 1989) to evaluate the impacts of hazardous chemicals on small mammal populations through the use of population studies are ill advised. The major limitations to the use of small mammal population dynamics to evaluate hazardous waste sites are that such studies are typically labor-intensive, long-term, and produce equivocal results.

A logical conclusion from the small mammal studies conducted for BMAP is that the guidelines presented in Warren-Hicks et al. (1989) on the use of terrestrial vertebrates in ecological assessments of hazardous waste sites are not relevant to ORNL sites. Moreover, such studies are likely to be of little or no value on the ORR or other DOE sites.

Whether biochemical endpoints reflecting chemical exposure of individuals

will be more successful than the techniques of population dynamics as a means of determining the exposure of small mammals to hazardous chemicals in the field remains to be seen. The hemoglobin-adduct assay for B[a]P exposure of small mammals was conducted at ORNL to provide an indication of whether specific individuals had been exposed to B[a]P in the field; however, interpretation of the results was constrained by the limitations described previously (Walton and Talmage 1989). Because of small sample sizes, high variability in the measured endpoint, and the inability to show statistical significance of the findings and to relate the measured endpoint to a specific dose or effect in the field population, small mammals are not good indicators of B[a]P in the environment using this technique.

Assays that can be employed to detect exposure of individuals to hazardous chemicals in the field can be extremely powerful tools for biological monitoring. Chemical-specific assays have the advantage that an endpoint, such as residue analysis, can be related to a known chemical contaminant, whereas nonchemical-specific assays have the advantage that the general health of a population can be assessed even though the causative agent of an effect remains unknown. The disadvantage of chemical-specific assays is that only those contaminants that have been anticipated are included. In contrast, nonchemical-specific assays can reveal the presence of previously unsuspected contaminants at a site by detecting chemically induced abnormalities in the biota. The unknown causative agent(s) can then be tracked down and identified.

An attempt to use a nonspecific indicator of small mammal exposure to genotoxic agents was undertaken at ORNL/ESD by evaluating liver tissues from woodchucks (*Marmota monax*) that were live trapped at a settling pond (Pond

3524) for liquid radioactive waste at ORNL. The results of these analyses were equivocal (L. R. Shugart, ORNL/ESD, unpublished data), and the study could not be continued because this unique woodchuck population receiving high radiation exposure was eliminated to preserve the integrity of the pond walls. An alternative population of terrestrial vertebrates was not found, thus, the potential use of the DNA assay to detect genotoxic damage in natural populations of small mammals had to be abandoned.

An opportunity to use the alkaline unwinding assay for DNA was provided in a comparative study of two turtle species to determine their usefulness as monitors of contamination in freshwater ecosystems. WOL and an uncontaminated reference site, the Bearden Creek embayment of Melton Hill Reservoir, were used as study sites. Significantly higher concentrations of ^{90}Sr , ^{137}Cs , ^{60}Co , and Hg were detected in turtles from WOL than in turtles from the reference site (Meyers-Schöne and Walton 1990). In these studies, turtles from WOL were found to have a significantly greater amount of DNA damage than those from the reference site (Meyers-Schöne et al. 1989). The latter analysis of DNA for single-stranded breaks demonstrated the potential utility of the assay as a nonspecific indicator of possible exposure of the turtles to genotoxic agents in WOL. Whether or not the assay could also be used to detect higher incidence of DNA breakage in small mammals remains to be demonstrated.

Results of the turtle studies from WOL also indicated that diet may play a significant role in the exposure to certain contaminants. Mercury concentrations in the carnivorous species (*Chelydra serpentina*) were significantly higher than in the herbivorous species (*Pseudemys scripta*). Moreover, the data for *C. serpentina* consistently showed less variability than those for *P. scripta*. Thus, these studies as well as the small mammal studies at ORNL indicated that selection of a species within a given taxonomic group can be an important factor influencing the results obtained in a biomonitoring program.

7.1.3 Future Studies

During the next year, the tasks outlined in BMAP for the assessment of contaminants in the terrestrial environment will be completed. Muscle tissues of *P. scripta* and *C. serpentina* will be analyzed for PCB residues. In addition, written guidelines will be completed on the use of small mammals to monitor heavy metals, radionuclides, and organic compounds at hazardous waste sites. Finally, recent findings of elevated radioactivity in plants and animals in the vicinity of Pits 2-4 and Trench 7 (Fig. 2.2) indicate that studies should be designed and implemented to document the magnitude and extent of contamination in the terrestrial biota in areas surrounding leaking pits and trenches.

8. RADIOECOLOGY OF WHITE OAK LAKE

B. G. Blaylock, D. A. Mohrbacher, and A. E. Waters

The radioecological studies in WOL are designed to provide information necessary to evaluate the potential risk of human exposure to radiation. The risk of exposure under current lake conditions will determine what, if any, remedial actions should be considered given a loss of institutional control. The primary focus of the overall task is to build an accurate and current data base of the concentrations and inventories of radionuclides in the biotic and abiotic components of WOL. This information, coupled with a basic understanding of how specific components interact to influence the dynamics of radionuclides within the system, will provide the knowledge required to assess the impact of the lake, directly or indirectly, on potential future human inhabitants of the area. An estimate of the radionuclide inventories are reported in Loar (1993b). This report includes the following information on the role of aquatic macrophytes in radionuclide cycling in WOL: (1) new and revised concentrations and inventories of ^{137}Cs and ^{60}Co in macrophytes, (2) data on macrophyte decomposition and the loss of ^{137}Cs and ^{60}Co from macrophytes, and (3) a comparison of externally and internally bound ^{137}Cs and ^{60}Co associated with the macrophytes. Also included in this report are results of the following studies on transient and resident waterfowl: (1) measurements of concentrations of radionuclides and nonradioactive pollutants in waterfowl, (2) uptake of ^{137}Cs and ^{60}Co by domestic mallards on WOL, and (3) waterfowl census and Canada geese banding study.

8.1 RADIOECOLOGY OF MACROFLORA IN WHITE OAK LAKE

8.1.1 Introduction

Aquatic macrophytes are important in the metabolism of aquatic ecosystems. Rooted plants take up nutrients from the sediment or water column, incorporate them into their biomass, and subsequently release them by excretion or decomposition. These functions can be important in nutrient cycling in shallow lentic systems. Because of their typically high productivity and constant and rapid biomass turnover, aquatic plants are a major source of detritus, contributing significantly to the supply of energy and nutrients that are recycled through the detrital component of aquatic food webs. In WOL, macrophytes may play an important role in the cycling of radionuclides as well. Ninety-nine percent of the radioactivity in WOL is in the sediments; of the remaining 1% in the biotic pool, macrophytes comprise the largest fraction. The macrophyte radionuclide pool is approximately equal in inventory to the entire water column (Loar 1993b). Because macrophytes can obtain minerals from the sediments, they may be a significant source of contaminants to the water column. Therefore, determining the role of macrophytes in the interception and uptake of contaminants and their subsequent fate in the decomposition process is fundamental to understanding how macrophytes influence radionuclide cycling in WOL.

The major objectives of this investigation were to (1) quantify the source of radionuclides (water column vs sediment) to rooted aquatic plants, (2) determine the fate of the macrophyte-associated radionuclides upon senescence and decomposition, and (3) determine the nature of the association of radionuclides with macrophytes (externally vs internally bound). In 1988, preliminary studies were conducted to determine the source of radionuclides to rooted submerged macrophytes and to quantify the macrophyte biomass at the Melton Branch and WOC weirs (Loar 1993b). The two streams provide major inputs of radionuclides to WOL, and the small impoundments created by the weirs provide additional study populations for examining plant radionuclide dynamics. Studies to determine the nature and fate of the radionuclides associated with the plants were initiated in 1989.

8.1.2 Methods

Samples of all rooted and floating aquatic plants in the WOC and Melton Branch weirs at WCK 2.65 and MEK 0.16, respectively, were collected approximately monthly to determine their content of gamma-emitting radionuclides. The emergent vegetation was collected every other month. All samples were taken to the laboratory, washed free of loosely adhering material, and oven-dried at 60°C for 24 to 48 h. Concentrations of ^{137}Cs and ^{60}Co were determined on a Nuclear Data 6700 microprocessor coupled to a Tennelec intrinsic germanium solid-state detector.

Biomass of milfoil (*Myriophyllum spicatum*), a submerged rooted species, and the floating duckweeds (*Lemna minor* and *Wolffia spp.*) in WOL was measured monthly from July through December. Milfoil biomass samples were collected at

eight randomly chosen sites around the lake (Fig. 8.1). Samples were taken by placing a 0.26-m² sampling frame just under the water surface. All aboveground milfoil within the frame was harvested with a rake. Samples were subsequently rinsed with tap water, oven-dried at 60°C and weighed to obtain biomass (expressed as g/m² dry weight). All samples were analyzed for their ^{137}Cs and ^{60}Co content.

Duckweeds were sampled at the same eight sites where milfoil was collected (Fig. 8.1). A 0.05-m² steel square was placed in the water around the duckweeds, and the plants within the square were removed using a net. Samples were effectively separated into species by sieving, oven-dried as above, and weighed. Selected samples were analyzed for ^{137}Cs and ^{60}Co .

The partitioning of radionuclides between internally and externally bound pools was determined for the WOL milfoil by washing the plants sequentially in deionized water and salt solutions (Winter 1961). External material that is loosely attached to the outside of the plant can be removed by washing with water. The radioactive cations adsorbed to the cell wall can be chemically exchanged with the cations in the salt solution. Cations associated with the cell wall are passively adsorbed, whereas the internal pool of radionuclides is actively incorporated via the roots or shoots into tissue biomass. Seventeen random samples of healthy milfoil shoots with little external material were collected from WOL and immersed in separate beakers containing deionized water under constant stirring. After 1 h, the plants were transferred to solutions of either 8 mM potassium chloride (five replicates) or 140 mM sodium chloride (12 replicates), under constant stirring. After 1 h, the plants were removed, oven-dried at 60°C for 24 h, and analyzed for ^{137}Cs and ^{60}Co . The solutions of deionized and salt water were also counted

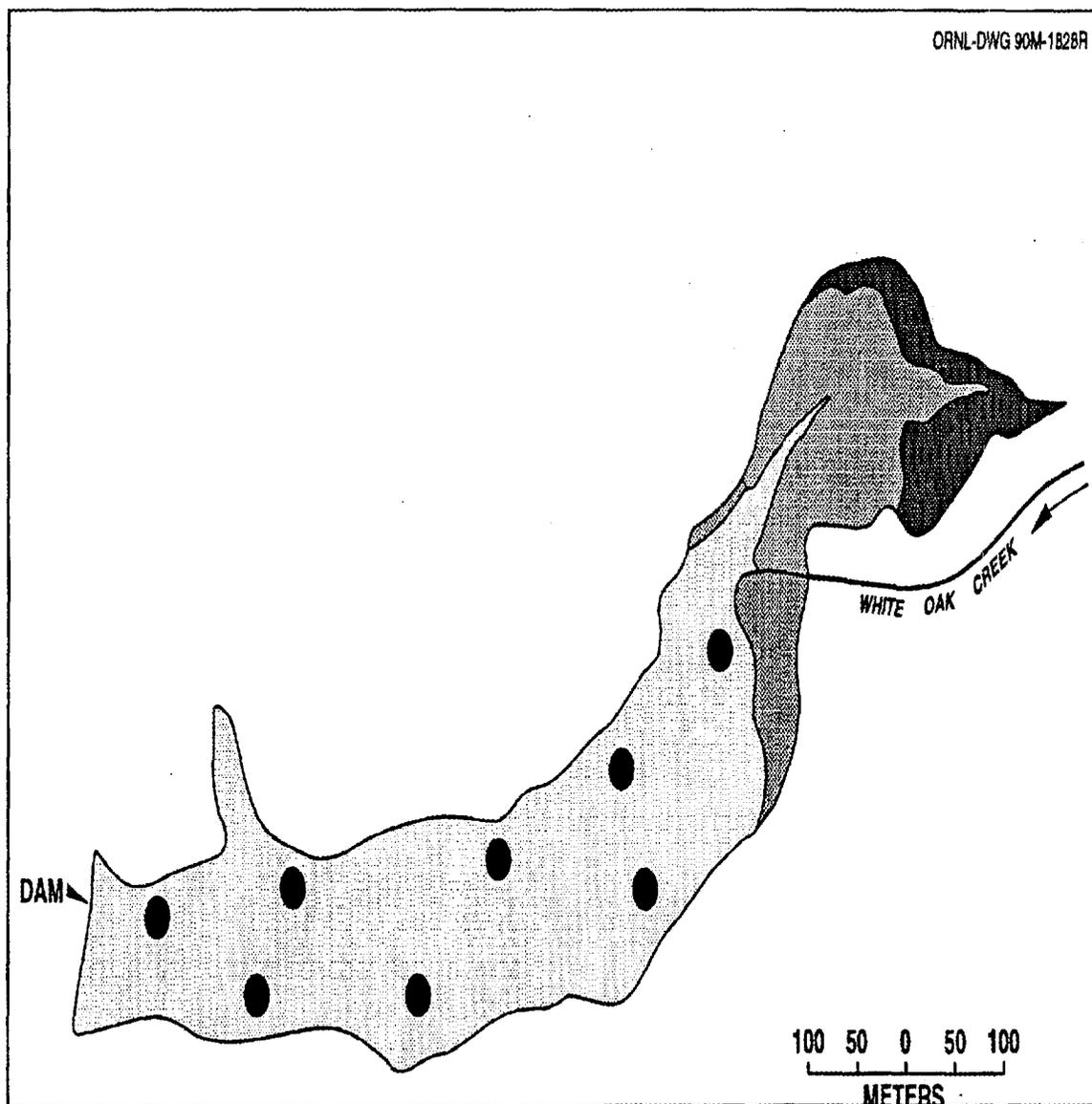


Fig. 8.1. Location of sampling sites in White Oak Lake for milfoil and duckweed biomass and litter bags experiments.

for radioactivity. The percent of the total radioactivity bound externally to the plant surface or within the cell wall was calculated as the percent removed by deionized water or the salt solution, respectively.

The *in situ* decomposition rate of milfoil from WOL was determined using litter bags placed in the lake.

Approximately 1 g of air-dried healthy milfoil shoots were collected from seven sites around the lake, rinsed free of loosely adhering material, and placed in nylon mesh bags (15 x 15 cm, mesh size 2.2 mm²). Four replicate litter bags were placed at each of eight sites around WOL (the same sites where milfoil biomass was taken) (Fig. 8.1). The bags were secured

with fishing line to a brick placed on the sediments. One bag from each location was sampled on days 2, 16, 37, and 48 beginning in August.

The decomposing plant material in the litter bags placed in WOL eventually become coated with radioactive sediment particles, which is difficult to completely remove by rinsing with water. These additional sediment particles, however, are unlikely to substantially increase the weight of the plant material. To determine the amount of radioactivity that is accumulated by milfoil detritus as it decomposes in WOL, an additional four litter bags containing uncontaminated healthy shoots of air-dried milfoil were placed at each of the eight sites in WOL (Fig. 8.1). One bag from each site was sampled at the same intervals as the contaminated milfoil litter bags. To determine the rate of loss of radioactivity during the decomposition process without the interference of radioactive sediment particles, 24 litter bags containing air-dried washed milfoil shoots were collected from WOL and placed in a nearby uncontaminated stream, NT, in October 1989. Bags were collected on days 2, 10, 44, and 61.

At the time of collection, the litter bags were rinsed gently in the ambient water to dislodge any adhering macroinvertebrates and sediment particles and air-dried for 72 h. The decomposing milfoil was weighed, analyzed for radioactivity, oven-dried at 60°C for 24 h, and combusted for 18 h at 550°C to determine ash content. Initial ash content in the plants was determined on separate subsamples of milfoil from WOL and from the uncontaminated ponds prior to the start of the experiment. Decomposition rates were calculated as the percent loss of dry weight, organic and inorganic weight, and radionuclides (NT only) at different time intervals, as determined by a least squares log plot.

8.1.3 Results and Discussion

8.1.3.1 White Oak Creek weir sampling

It was determined in 1988 that water and sediments in WOC contain a greater proportion of ^{137}Cs than ^{60}Co , whereas the converse is true for Melton Branch (Loar 1993b). The differences in radionuclide concentrations reflect different inputs to the two systems. In both, the sediments contain much higher concentrations of radionuclides than the water. The concentrations of ^{137}Cs and ^{60}Co measured in the aquatic plants growing in the weirs also reflects this difference in radionuclide concentrations.

Milfoil was the dominant species growing in WOC and had a mean ^{137}Cs and ^{60}Co concentration of 7.5×10^3 and 0.9×10^3 Bq/kg, respectively (Table 8.1). These concentrations were higher than those measured in Elodea (*Elodea canadensis*) (4.5×10^3 and 0.5×10^3 Bq/kg for ^{137}Cs and ^{60}Co , respectively) in WOC. The difference in radionuclide accumulation between the two rooted species is most likely due to differences in their physiology, since both species are capable of accumulating radionuclides from the water column and the sediments. Watercress (*Nasturtium officinale*) and filamentous green algae (*Spirogyra*, *Oedogonium* and *Cladophora*) contained similar concentrations of ^{137}Cs (about 6.6×10^3 Bq/kg) during the year (Table 8.1). Mean ^{60}Co concentrations were highest in milfoil and watercress (Table 8.1). Cattails (*Typha latifolia*) and bulrush (*Scirpus spp.*) contained lower ^{137}Cs and ^{60}Co concentrations than either the rooted or the floating aquatic plants (Table 8.1). Concentrations of ^{137}Cs in milfoil and both ^{137}Cs and ^{60}Co in watercress increased in the winter months (beginning in late summer for milfoil), while concentrations of ^{60}Co in milfoil was similar throughout

Table 8.1. Mean radionuclide concentrations in aquatic macrophytes collected at the White Oak Creek weir in 1989

Species	Radionuclide	Mean concentration (Bq/kg dry weight × 1000)									Mean	SD ^a
		Jan.	Apr.	May	June	July	Aug.	Sept.	Oct.	Dec.		
Milfoil	¹³⁷ Cs	NC ^b	5.1	2.1	4.9	5.1	11.1	10.7	9.7	11.6	7.5	3.6
	⁶⁰ Co	NC	0.5	0.44	1.7	0.6	0.64	0.32	0.81	1.8	0.85	0.57
Elodea	¹³⁷ Cs	7.5	2	0.85	1.9	8.4	3.9	NC	6.8	NC	4.5	3.1
	⁶⁰ Co	1.1	0.48	0.32	0.53	0.56	0.16	NC	0.26	NC	0.49	0.31
Watercress	¹³⁷ Cs	6.4	4.9	NC	5.9	4	5.2	NC	8.6	10.2	6.5	2.2
	⁶⁰ Co	1.6	2	NC	0.45	ND ^c	0.06	NC	0.14	0.93	0.86	0.8
Filamentous algae	¹³⁷ Cs	NC	NC	6.1	8.7	6.8	4.3	7.6	NC	NC	6.7	1.7
	⁶⁰ Co	NC	NC	0.2	ND	0.18	ND	0.2	NC	NC	0.12	0.11
Cattails	¹³⁷ Cs	NC	0.33	NC	0.04	NC	0.4	NC	NC	NC	0.26	0.19
	⁶⁰ Co	NC	0.02	NC	0.004	NC	0.01	NC	NC	NC	0.01	0.01
Bulrush	¹³⁷ Cs	2.8	NC	NC	0.67	NC	0.2	NC	NC	NC	1.2	1.4
	⁶⁰ Co	1	NC	NC	0.009	NC	ND	NC	NC	NC	0.34	0.57

^aSD = standard deviation.

^bNC = not collected.

^cND = not detected.

the year (Table 8.1). Increased epiphyton on plant surfaces during the fall may be responsible for the increase in radionuclide concentrations. Epiphyton are not easily removed by table washing with water and may accumulate substantial amounts of radioactivity. Alternatively, the plants may continue to accumulate radionuclides.

8.1.3.2 Melton Branch weir sampling

Elodea was the most abundant species growing in Melton Branch and contained higher concentrations of ^{60}Co and ^{137}Cs (4.6×10^3 and 0.1×10^3 Bq/kg, respectively) than other aquatic plants, except filamentous algae, which contained slightly higher concentrations (4.7×10^3 Bq/kg for ^{60}Co and 0.3×10^3 Bq/kg for ^{137}Cs) (Table 8.2). The concentration of ^{60}Co in Elodea was higher during the winter months, similar to the trend for ^{137}Cs in the WOC weir, and may also be due to epiphyton. The duckweeds contained much lower ^{60}Co concentrations than any of the plant types collected, except for the emergents. This result is somewhat surprising, since the duckweeds and filamentous algae are both floating plants with a rapid turnover rate and can only accumulate radionuclides from the water column; thus, their concentrations should be similar. The differences in uptake between the two plant types may be attributed to their differing physiologies or to their surface characteristics in relation to the water. Pondweed (*Potamogeton foliosus*) and milfoil had similar ^{137}Cs and ^{60}Co concentrations, but contained much less ^{60}Co than Elodea, even though all three plants are rooted, submerged types. The ^{60}Co and ^{137}Cs concentrations measured in the emergent cattails and bulrush were lower than either the floating or submerged species.

8.1.3.3 Plants collected in the White Oak Creek and Melton Branch weirs: Comparison between 1989 and 1988

In general, the abundance of vegetation of all plant types (floating, submerged, and emergent) was lower in both the WOC and Melton Branch weirs in 1989 than in 1988 (Loar 1993b). Additionally, pondweed and duckweeds were not encountered in the WOC weir in 1989, and milfoil and Elodea were not found in the Melton Branch weir in 1988. The ^{137}Cs concentration in milfoil, filamentous algae, watercress and cattails in WOC, and the ^{60}Co concentrations in pondweed and algae in Melton Branch were lower as well in 1989 compared to 1988. The differences in species presence and abundance and radionuclide concentrations cannot be fully explained at this time. One possible explanation is that stream flows in both systems in 1989 were more variable than in 1988 due to more frequent storms (Fig. 2.3), which frequently removed plants from the system and moved sediment on and off the plants.

8.1.3.4 White Oak Lake

Milfoil and duckweeds were the most abundant species found in WOL, with each covering approximately two-thirds of the lake. Milfoil biomass was initially sampled during the peak of the growing season in August, and averaged 96-g dry weight/m². Biomass decreased only slightly over the next few months until the onset of senescence in November ~28-g dry weight/m². The majority of milfoil occurred in the shallow (<1 m) regions of the lake along the banks and in the upper end. The average biomass of milfoil in the WOC weir between April and June 1988

Table 8.2. Mean radionuclide concentrations in aquatic macrophytes collected at the Melton Branch weir in 1989

Species	Radionuclide	Mean concentration (Bq/kg dry weight × 1000)									Mean	SD ^a
		Jan.	Apr.	May	June	July	Aug.	Sept.	Oct.	Dec.		
Elodea	¹³⁷ Cs	0.11	NC ^b	0.11	0.06	0.1	0.06	0.11	ND ^c	0.25	0.1	0.07
	⁶⁰ Co	12.7	NC	1.8	1.9	2.6	3.2	2.9	1.9	9.4	4.6	4.1
Filamentous algae	¹³⁷ Cs	NC	NC	0.72	NC	0.04	0.1	NC	NC	NC	0.29	0.38
	⁶⁰ Co	NC	NC	4.5	NC	1.8	7.8	NC	NC	NC	4.7	3
Milfoil	¹³⁷ Cs	NC	NC	NC	0.13	0.05	0.04	NC	NC	NC	0.07	0.05
	⁶⁰ Co	NC	NC	NC	1.5	1	2.5	NC	NC	NC	1.7	0.77
Pondweed	¹³⁷ Cs	NC	NC	ND	0.05	0.08	NC	NC	NC	NC	0.04	0.04
	⁶⁰ Co	NC	NC	1.5	1.2	2.8	NC	NC	NC	NC	1.8	0.85
Duckweed	¹³⁷ Cs	NC	NC	NC	NC	0.09	0.23	NC	NC	NC	0.16	0.1
	⁶⁰ Co	NC	NC	NC	NC	0.65	1.2	NC	NC	NC	0.93	0.39
Cattails	¹³⁷ Cs	NC	0.02	NC	0.01	NC	NC	NC	NC	NC	0.02	0.01
	⁶⁰ Co	NC	0.21	NC	0.07	NC	NC	NC	NC	NC	0.14	0.1
Bulrush	¹³⁷ Cs	NC	NC	NC	0.02	NC	ND	NC	NC	NC	0.01	0.01
	⁶⁰ Co	NC	NC	NC	0.16	NC	0.05	NC	NC	NC	0.11	0.08

^aSD = standard deviation.

^bNC = not collected.

^cND = not detected.

(184-g dry weight/m²) was almost twice that in WOL (Loar 1993b).

The ¹³⁷Cs and ⁶⁰Co concentrations in milfoil collected from eight sites in the lake averaged about 8.4×10^3 and 1.6×10^3 Bq per kg respectively (Table 8.3). The highest concentrations were found in plants collected during late winter, again most likely due to increased amounts of external epiphyton on the plant surface.

Radionuclide concentrations in the WOL milfoil were slightly higher than those in the WOC weir (Tables 8.1 and 8.3). Cesium-137 concentrations in milfoil varied within the lake and were about two times greater in plants from the upstream end of the lake compared with those in plants obtained from the area near the dam. Concentrations of ⁶⁰Co in milfoil were similar throughout the lake. The mean ¹³⁷Cs concentrations in milfoil collected in 1988 were almost three times greater in plants from the upstream end than in those collected near the dam, and the ⁶⁰Co concentrations were twice as high (Loar 1993b). Radionuclide concentrations in sediment in the upper end are the highest within the lake (Loar 1993b). The higher ¹³⁷Cs concentrations in the plants at the upper end of the lake may be due to root uptake of the greater quantity of ¹³⁷Cs in the sediments.

During October, blooms of filamentous green algae covered the entire lake. The ¹³⁷Cs concentration in the algae was higher than that measured in the milfoil, but the ⁶⁰Co concentration was higher in the milfoil than the algae (Table 8.3). There was no apparent seasonal trend in the ¹³⁷Cs or ⁶⁰Co concentrations in algae. The radionuclide concentrations in the WOL algae were higher than those in the WOC weir algae.

At least three species of duckweed (*Lemna minor*, *Spirodela spp.*, and *Wolffia spp.*) were growing in the lake from June to October. The duckweeds were most abundant near the dam, apparently due to

wind action. Biomass was quite variable throughout the growing period and showed no apparent seasonal trend. The biomass of *Lemna* and *Spirodela* (analyzed as one species) ranged from 3.1- to 268-g dry weight/m² (mean = 55.4 g/m²); *Wolffia* biomass ranged from 3.7- to 239-g dry weight/m² (mean = 47.7 g/m²). Biomass at the dam was always the highest of any of the sites for *Lemna* and *Wolffia*—both reached a maximum in September. The ¹³⁷Cs and ⁶⁰Co concentrations for all duckweeds were similar, $\sim 2.5 \times 10^3$ Bq/kg for ¹³⁷Cs and 0.46×10^3 Bq/kg for ⁶⁰Co (Table 8.3). During the end of the growing season, the dam was opened and a substantial but unquantified amount of duckweeds was released downstream.

8.1.3.5 Litter bag experiments

Litter bag experiments were used to determine the decomposition rate of milfoil in both WOL and nearby NT (free of ¹³⁷Cs and ⁶⁰Co contamination). The results of the litter bag experiments show a uniform exponential decay, but the behavior of all components showed a distinct diphasic decay pattern with fast and slow decomposing fractions. Other investigators of aquatic plant decomposition have reported a similar diphasic behavior (Boyd 1971, Wetzel and Manny 1972, Mason and Bryant 1975, Howard-Williams and Davies 1979). The plant material in the bags placed in WOL and NT lost a substantial amount of weight in all components (dry weight, organic and inorganic weight) and radionuclides (NT only) in the first few days, after which decomposition declined at a constant rate.

In WOL, both the initially clean and contaminated milfoil lost ~15% dry weight/d in the first few days, and then decay slowed to about 2%/d (Fig. 8.2 and Table 8.4). (Figures of the decomposition curves for the initially clean milfoil placed

Table 8.3. Mean radionuclide concentrations in aquatic macrophytes collected in White Oak Lake in 1989

Species	Radionuclide	Mean concentration (Bq/kg dry weight × 1000)									Mean	SD ^a
		Jan.	Apr.	May	June	July	Aug.	Sept.	Oct.	Dec.		
Milfoil	¹³⁷ Cs	14.8	11.3	8.1	5.5	5.9	5.6	4.3	4.5	11.5	8.4	3.7
	⁶⁰ Co	3.9	1.3	1.6	0.71	0.99	1.3	1.3	1.4	1.9	1.6	0.93
Filamentous algae	¹³⁷ Cs	6.9	11.2	6.7	12.9	NC ^b	NC	11	8.2	NC	9.5	2.6
	⁶⁰ Co	1.6	1.7	0.82	1.1	NC	NC	1.6	1.2	NC	1.3	0.35
<i>Lemna</i> and <i>Spirodela</i>	¹³⁷ Cs	NC	NC	3.1	NC	NC	2.22	NC	NC	NC	2.7	0.64
	⁶⁰ Co	NC	NC	0.6	NC	NC	0.41	NC	NC	NC	0.51	0.13
<i>Wolffia</i>	¹³⁷ Cs	NC	NC	1.7	2.3	0.07	5	NC	NC	NC	2.3	1.3
	⁶⁰ Co	NC	NC	0.61	0.56	0.007	0.42	NC	NC	NC	0.4	0.27
Watercress	¹³⁷ Cs	NC	NC	NC	NC	NC	2.4	NC	3.2	NC	2.8	0.57
	⁶⁰ Co	NC	NC	NC	NC	NC	0.11	NC	0.97	NC	0.54	0.61
Cattails	¹³⁷ Cs	NC	NC	NC	NC	NC	0.05	NC	0.35	NC	0.2	0.2
	⁶⁰ Co	NC	NC	NC	NC	NC	0.02	NC	0.04	NC	0.03	0.01

^aSD = standard deviation.

^bNC = not collected.

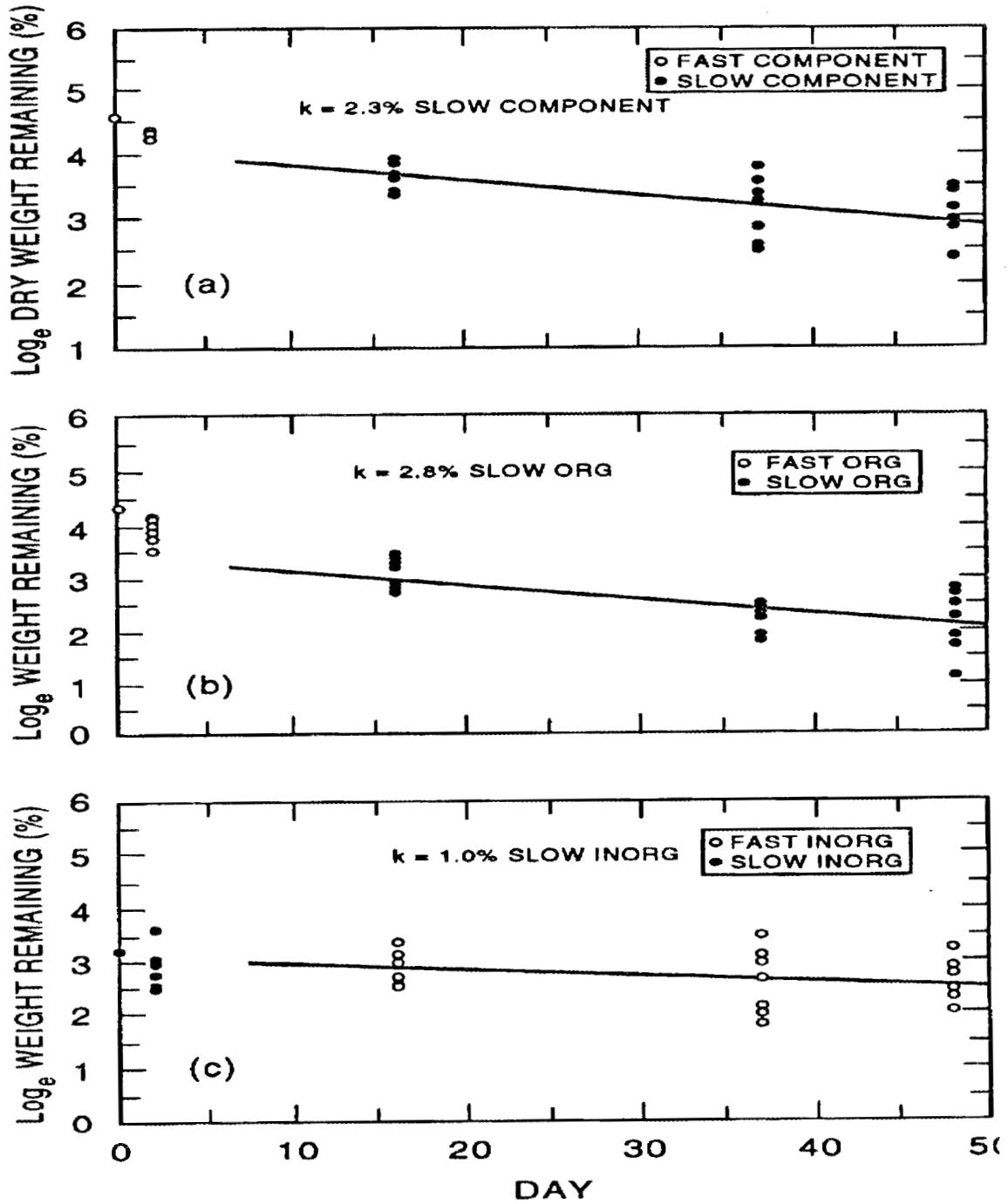


Fig. 8.2. Change over time in (a) dry weight, (b) organic weight, and (c) inorganic weight of initially contaminated milfoil in White Oak Lake.

Table 8.4. Percent lost per day for fast (initial loss) and slow components of milfoil in White Oak Lake and the Northwest Tributary

Site		Loss (%)				
		Dry weight	Organic	Inorganic	Total ¹³⁷ Cs	Total ⁶⁰ Co
WOL, ^a contaminated	Fast	15.5	18.6	15	NA ^b	NA
	Slow	2.3	2.8	1	NA	NA
WOL, clean	Fast	15	8	43	NA	NA
	Slow	1.5	2.4	0.4	NA	NA
Northwest Tributary	Fast	12.3	15.9	6.9	25	32
	Slow	3.2	3.5	3	3.1	3.5

^aWOL = White Oak Lake.

^bNA = not applicable.

in WOL are not shown but were qualitatively similar to the initially contaminated plants placed in WOL.) The relative loss of the organic component was substantially greater in the first few days in the decomposing contaminated plant material than in the decomposing clean material (Fig. 8.2 and Table 8.4). This result may be due to differences in the nutrient contents of the plants (Carpenter and Adams 1977) or, more likely, to a loss of portions of the significant amount of external organic material that coats the contaminated plants and could not be easily washed off without damaging the plants. This material was not present on the initially clean plants. The slow-decay organic component was also faster for the contaminated plants, probably because these had been colonized by the indigenous epiphyton community consisting of a microbial and algal matrix. After initial rapid losses of inorganic material, subsequent losses were minimal for both contaminated and clean detritus, almost approaching a steady state (Fig. 8.2). Apparently, the detritus is accumulating

suspended sediment at a rate that nearly compensates for inorganic losses.

The decay data for NT displayed qualitatively similar diphasic behavior for all mass components (Fig. 8.3). Although both the initially contaminated WOL and NT plants were collected from the same sites in WOL, their subsequent decomposition showed a dependence on location. The initial losses in all components in the NT milfoil were less than those in WOL (Table 8.4) due possibly to differences in temperature or water quality between the two sites. The slow-decay components showed the reverse behavior (WOL less than NT), perhaps due to differences in the detrital communities.

Both the initially clean and contaminated milfoil in the WOL litter bags accumulated ¹³⁷Cs and ⁶⁰Co. After the initial loss of inorganic weight in the clean plants during the first few days, both inorganic weight and total radioactivity increased to a near steady state in each bag. These parallel occurrences suggest that the radionuclides may be associated

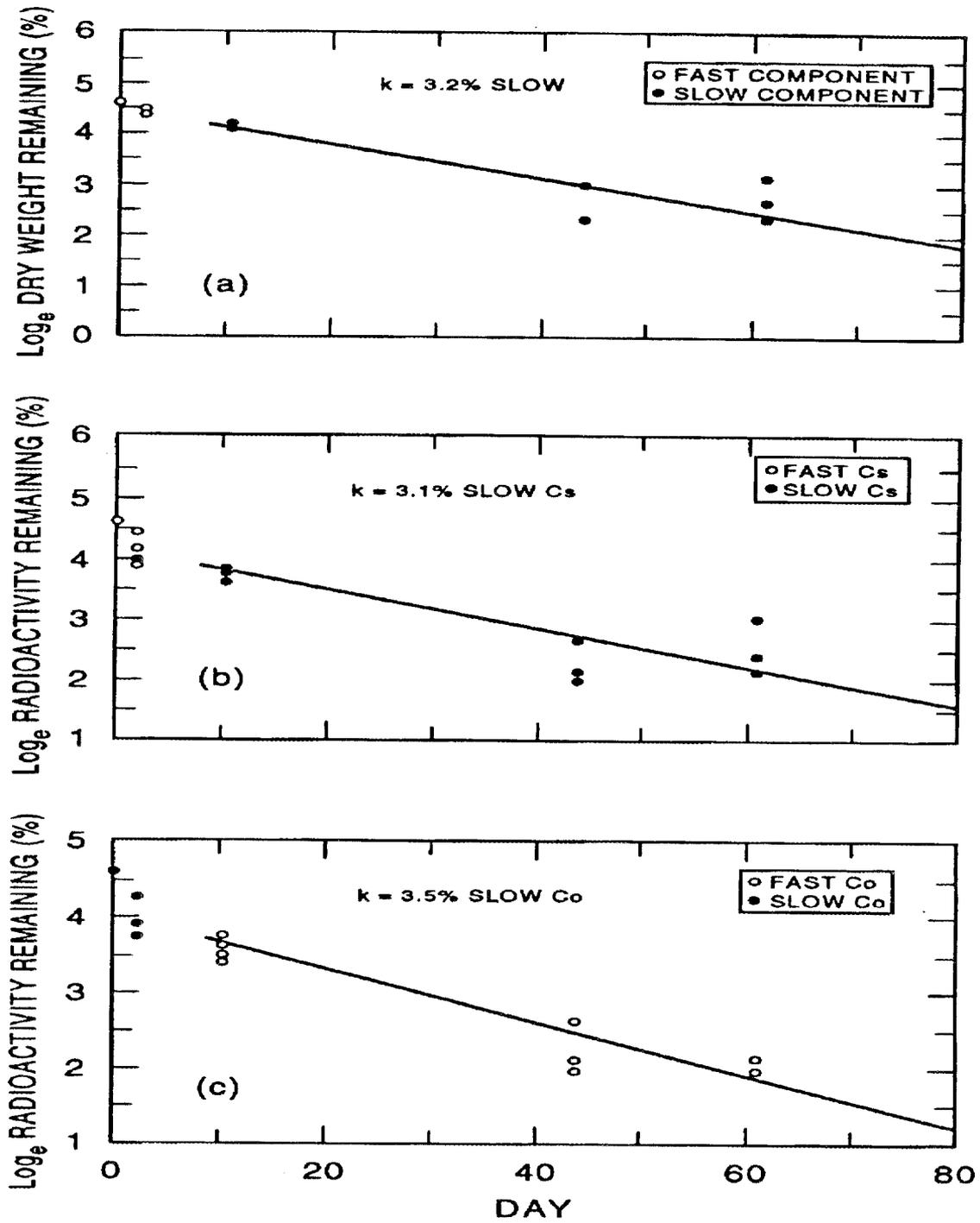


Fig. 8.3. Change over time in (a) dry weight, (b) ¹³⁷Cs, and (c) ⁶⁰Co of milfoil in Northwest Tributary.

with the inorganic fraction. The organic weight decreased steadily during the same period.

Cesium-137 and ^{60}Co associated with the contaminated plants placed in NT were lost faster than any of the mass components (dry weight, organic or inorganic components) in the first few days (Fig. 8.3 and Table 8.4). The difference between radionuclide loss and total mass loss may represent leaching or cation exchange. Interestingly, ^{60}Co was lost faster than ^{137}Cs , which is considered to be the more mobile ion. However, ^{60}Co is known to be associated with organic molecules and may be lost in the initial very rapid decrease in the organic fraction. In the slow phase of decay, all components of mass (dry weight, organic, and inorganic) and the radionuclides were lost at similar rates, suggesting that losses occur by fragmentation or disintegration of the plant.

8.1.3.6 External vs internal contamination

The contribution of internally vs externally bound radionuclides to the total plant radionuclide pool was estimated by washing plants with water or salt solutions. On average, results of these experiments indicated that washing with water removed 30% of the ^{60}Co and 59% of the ^{137}Cs from the WOL milfoil (Fig. 8.4). Washing the plants in a salt solution removed an additional 18% of the initial ^{60}Co and an additional 14% of the ^{137}Cs . After both washings, 52% of the ^{60}Co and 27% of the ^{137}Cs remained associated with the plants (Fig. 8.4). These data indicate that the initial rapid loss of radionuclides in the decay process may primarily stem from the loss of exchangeable particulate material on the plant surface and/or from leaching of soluble material from within the internal compartments and the cell wall. Aquatic plants, therefore, may rely more on their

physical structure than on metabolic uptake in accumulating radionuclides.

8.1.4 Summary

In general, the floating filamentous algae contained higher concentrations of radionuclides than the rooted submerged species in the three systems (WOC and Melton Branch weir and WOL). The only exception to this pattern was *Elodea* growing in Melton Branch, which contained ^{60}Co concentrations similar to those of algae collected in the stream. The duckweeds and emergent vegetation contained lower overall concentrations in both weirs and in the lake. A seasonal trend in radionuclide concentrations was apparent in several species. Higher average concentrations during the winter months may have been due to epiphyton growth or sediment accumulation on plant surfaces, or to unexplained physiological factors. The biomass of milfoil in WOL was low compared to that measured in WOC but can potentially increase to five times that amount. The relative abundance of macrophytes and the ^{137}Cs and ^{60}Co concentrations in the macrophytes collected in the weirs and WOL were lower in 1989 than in 1988.

Results from the litter bag experiments showed a distinct diphasic decay pattern. Significant portions of all mass components (dry weight, organic and inorganic weight) and radionuclides in both the WOL and NT plants were lost in the first few days. This result was most likely due to the considerable amount of material clinging to the outer surfaces of the plants. In general, all of the plant material and the radionuclides associated with it had essentially disappeared after 2 months (i.e., there was little refractory material left).

Estimates of the total release of ^{137}Cs and ^{60}Co into the water column of WOL by aquatic macrophytes was calculated

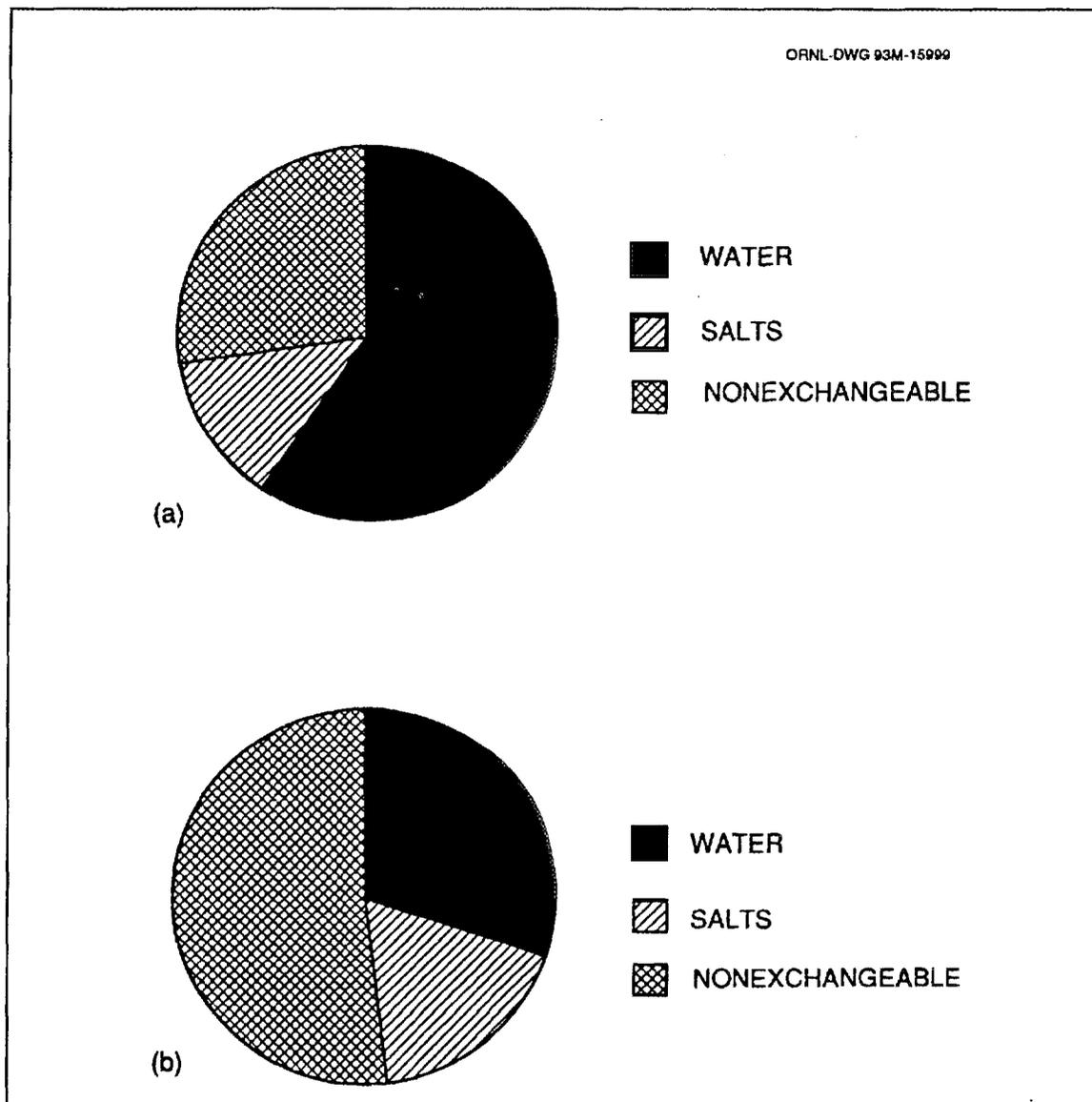


Fig. 8.4. Fraction of the total (a) ¹³⁷Cs and (b) ⁶⁰Co associated with milfoil removed by water and salt solutions.

based on the 1989 estimates of milfoil biomass, radionuclide concentrations, and external contamination. After correcting for external contamination (27% for ¹³⁷Cs and 52% for ⁶⁰Co) and assuming that the milfoil covers approximately two-thirds of the lake and contains the maximum radionuclide concentration, the total release of ¹³⁷Cs into the WOL water column from milfoil is 6.9×10^7 Bq, of

which at least 40% has been determined to be derived from the sediments directly by root uptake (Loar 1993b). Therefore, the net gain of ¹³⁷Cs into the water column (from the sediments via milfoil) is 2.8×10^7 Bq. Assuming at least 60% of the ⁶⁰Co contained in milfoil is sediment derived via roots (Loar 1993b), the net gain of ⁶⁰Co into the water from milfoil decomposition would be 5.3×10^6 Bq.

These estimates of total annual release are conservative, however, since milfoil turns over approximately twice a year (Adams and McCracken 1974).

Aquatic macrophytes may serve as a source of radionuclides to the water column that would otherwise remain immobile in the sediments. There may be several possible fates of the radionuclides released from milfoil in WOL. They may be released to the water in a dissolved form that is readily available for uptake by other biota or they may be (come) associated with organic and/or sediment particles. Dissolved or particulate radionuclides can either become part of the sediment pool or be transported downstream. These questions and others concerning the fate of radionuclides during the senescence and decomposition of aquatic macrophytes will be addressed in future studies in 1990.

8.2 WATERFOWL POPULATIONS ASSOCIATED WITH OAK RIDGE RESERVATION BASINS AND PONDS

8.2.1 Introduction

The waterfowl study was initiated in 1987 and continued through 1989 to determine the use by waterfowl of waste disposal ponds and settling basins on the ORR. Concentrations of radionuclides, metals, and PCBs have been detected in the tissues of both transient and resident waterfowl. Measurements of radionuclides in American coots and Canada geese collected in 1988 showed that waterfowl used the ORR and accumulated radionuclides. Other waterfowl species were also observed using several of the waste ponds on the ORR. Furthermore, data have been collected on the interactions of Canada geese from the ORR with other local populations.

8.2.2 Materials and Methods

8.2.2.1 Waterfowl census

The weekly waterfowl census of 11 sites near ORNL was initiated in October 1987 (Loar 1993a). Observations at sites near the ORGDP and the Y-12 Plant began in October 1988 and February 1989 respectively.

8.2.2.2 Contaminants in waterfowl

Transient and resident waterfowl were collected from various locations on the ORR to determine if contaminants accumulate in their tissues. Eight transient waterfowl (gadwall, mallard, black duck) collected from WOL between November 1988 and March 1989 were analyzed for gamma-emitting radionuclides, ^{90}Sr , and metals. Gamma-emitting radionuclide determinations were conducted using an intrinsic germanium detector coupled to a Nuclear Data 6700 microprocessor system. Breast tissue, liver, kidney, bone, gastrointestinal tract, gizzard contents, feathers, and whole-body analyses were performed for radionuclides. Strontium-90 was measured in the bone, and analyses for metals (Ag, As, Cd, Hg, Pb, and Se) were conducted on breast tissue, liver, and kidney.

In March 1989, a pair of Canada geese were collected from Basin 3524 at ORNL. The following tissues were analyzed for gamma-emitting radionuclides: breast tissue, liver, bone, and gizzard contents. Bone samples were analyzed for ^{90}Sr .

8.2.2.3 Contaminants in wood ducks from White Oak Lake

Wood ducks, particularly immature birds, were collected from WOL to obtain an estimate of contaminant accumulation

in a population that nests on the lake. Ten wood ducks, of which seven were immature individuals, were collected from WOL between April and October 1989. Tissues were processed and analyzed as described above. The contaminants of concern were ^{137}Cs , ^{60}Co , ^{90}Sr , Se, Hg, and PCBs.

8.2.2.4 White Oak Lake domestic mallard experiment

To establish accumulation rates of radionuclides, metals, and PCBs in waterfowl tissues, domestic wing-pinioned mallards were released on WOL. A total of 80 ducks were purchased from a commercial mallard farm in Illinois and maintained for one week prior to release in an enclosed uncontaminated area. Thirty-eight birds were released on May 6, 1989, and 38 birds were released on October 14, 1989. Two mallards from each group were sacrificed as controls prior to release. The two releases were scheduled so that the first group could remain on the lake for approximately 6 months to simulate a resident population. The second release paralleled the arrival on the lake of migratory waterfowl in the fall. The second release allowed for the testing of seasonal differences in tissue concentrations and provided another opportunity for determining pollutant uptake rates.

All mallards were fitted with sequentially numbered leg bands. Fluorescent-colored neck collars were fitted to the first group of ducks to distinguish them from transient mallards in the field. The weight and sex of each bird were recorded before release.

All the mallards in the first group were of the same age. Birds were collected on days 2, 4, 9, 21, 49, and 77. Although it was planned to collect three mallards per sampling period, this sampling

scheme was only successful for days 9 and 21. The second group of mallards, which consisted of equal numbers of immature and adult birds, was sampled on days 2, 4, 8, 16, 39, 61, and 95. Neck collars were not used with the second group of birds to improve survival. Three or four mallards were collected in each sampling period, except day 95 (one duck). Because trapping efforts were not successful, all ducks were collected with firearms. Radionuclide, metal, and PCB analyses were conducted for all birds on the organs/tissues listed previously.

8.2.2.5 Canada goose banding study

On June 19–20, 1989, Canada geese on the ORR were banded to determine movement patterns and document interactions of ORR geese with other local flocks. A health physicist screened the geese for radioactive contamination prior to handling. Corresponding leg and neck bands were fitted to adults and several immature birds. Leg bands that had been fitted previously to the geese were removed. The white neck collars provided by the U.S. Fish and Wildlife Service were made of a thin, hard plastic and coded with a sequence of letters and numbers. The numbers on the neck collar matched the last two digits of the leg band, and the last letter of the neck collars corresponded to a specific banding site. Geese were banded at the ORGDP, Y-12, ORNL, Clark Center Park, Solway Park, and Melton Hill Dam (Fig. 8.5).

As in the previous year, the birds banded at ORNL were whole-body counted on a sodium iodide detector for 5 min to check for elevated levels of radionuclides. Radionuclide levels in these 13 geese did not significantly exceed background levels. However, on July 11, 1989, a group of banded and unbanded Canada geese that frequented the STP ponds at ORNL were captured and

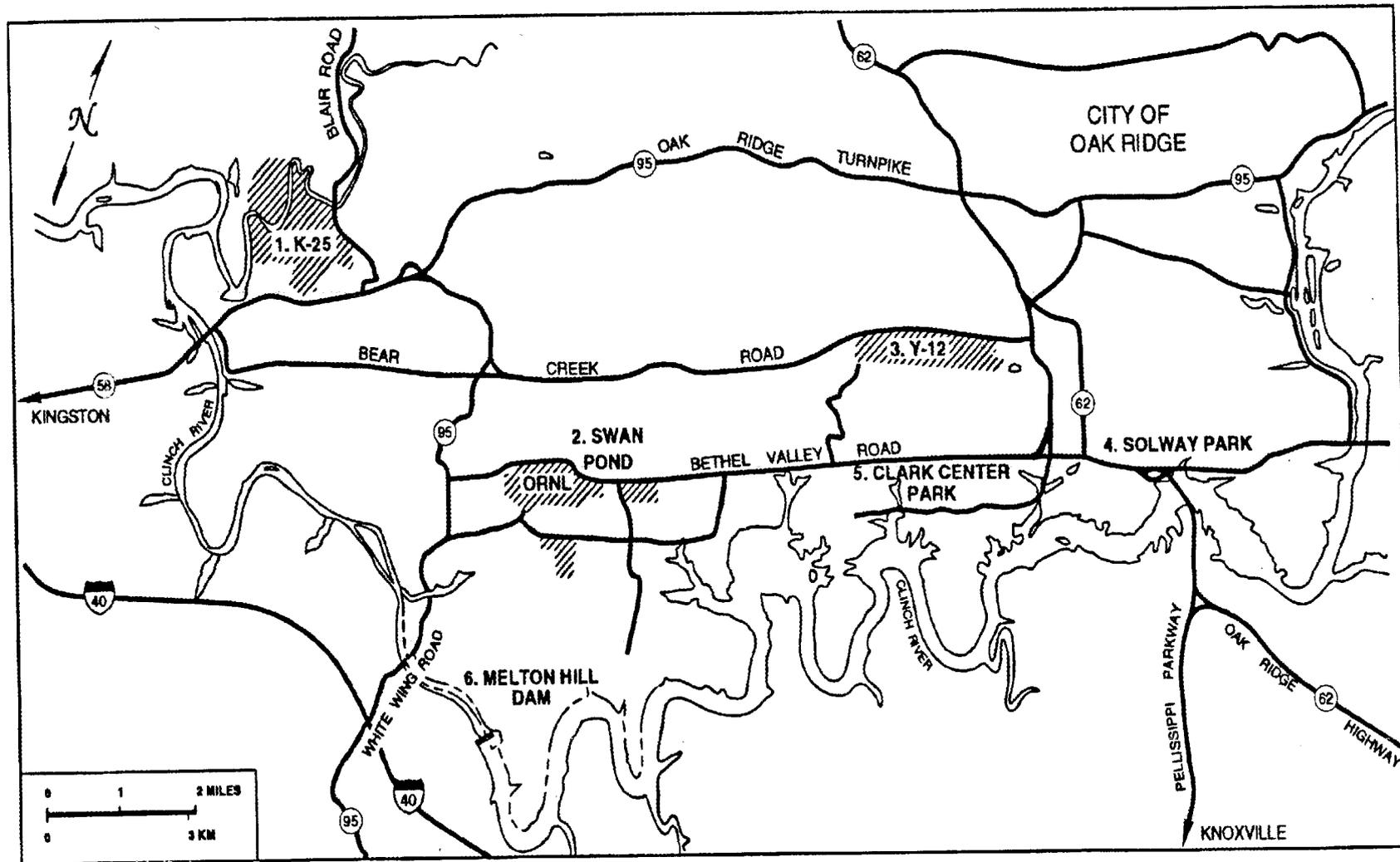


Fig. 8.5. Canada geese banding sites for June 1989.

subjected to whole body counting. Six of the banded birds were sacrificed due to elevated levels of radionuclides, and the remaining unbanded geese were fitted with leg and neck bands. Observations of neck-banded geese were recorded weekly at all locations except Clark Center and Solway Parks. In addition, a section of Melton Hill Reservoir and the Clinch River (~80 km) was surveyed monthly by boat to census geese.

8.2.3 Results and Discussion

8.2.3.1 Waterfowl census

Based on a summation of the number of birds counted during each census, more than 16,000 observations of waterfowl have been recorded at designated locations on the ORR since the waterfowl census was initiated in October 1987. Both resident and transient birds use the ORR (Table 8.5). The species observed most frequently and in the highest densities were Canada geese, wood ducks, black ducks, gadwalls, and mallards. Other aquatic birds that used the ponds, lakes, and streams on the ORR included the belted kingfisher, green-backed heron, great blue heron, black-crowned night heron, little blue heron, ring-billed gull, and great egret.

Oak Ridge National Laboratory.

Approximately 9500 observations of waterfowl have been made at ORNL since October 1987. Waterfowl used WOL and Swan Pond most frequently, and no waterfowl were observed at the HFIR REDC ponds during the census period. Various waterfowl species used the other census sites, but no trends or patterns have been determined.

Canada geese, wood ducks, and mallards were the most abundant species using the aquatic habitats near ORNL; these species also bred and raised their

young at these areas in 1988 and 1989. Canada geese and mallards nested at Swan Pond, and wood ducks nested at WOL. The number of wood ducks peaked in late summer (Fig. 8.6).

The uncontaminated Swan Pond also served as a wintering ground for a population of ring-necked ducks. The birds arrived in October and remained through April, a trend that was observed in each of the three years (Fig. 8.7). More than 20 ring-necked ducks used the pond each season.

The waterfowl population on WOL fluctuated throughout the year. The lake served as a resting ground and possible roosting or wintering area for migratory species during the late fall and winter. The number of waterfowl observed on WOL per observation period has ranged from 0 to ~160 birds (Fig. 8.8). An individual bird may spend a few days to several weeks on the lake. Transient mallards, black ducks, gadwalls, American wigeons, mergansers, blue-winged teal, green-winged teal, and Canada geese have been observed on the lake.

Oak Ridge Gaseous Diffusion Plant.

The aquatic environs at the ORGDP is used extensively by Canada geese. Several hundred geese have been observed at ORGDP during a single observation period (Fig. 8.9). Approximately 6000 observations of geese were recorded. Canada geese have been observed at most of the ORGDP census locations, but the most frequently used sites were Poplar Creek, the K-1007B holding pond, and the grassy areas located throughout the ORGDP site. Most of the Canada geese at the ORGDP are from a resident population; however, transients probably use the aquatic habitats as a resting area or wintering ground. Some of the resident geese also use these areas for nesting—goslings have been observed in both 1988 and 1989. Waterfowl were

Table 8.5. Species of waterfowl observed on the Oak Ridge Reservation

Common name	Scientific name
American black duck	<i>Anas rubripes</i>
American coot	<i>Fulica americana</i>
American wigeon	<i>ANAs americana</i>
Blue-winged teal	<i>Anas discors</i>
Bufflehead	<i>Bucephala albeola</i>
Canada goose	<i>Branta canadensis</i>
Gadwall	<i>Anas strepera</i>
Greater scaup	<i>Aythya marila</i>
Green-winged teal	<i>Anas carolinensis</i>
Hooded merganser	<i>Lophodytes cucullatus</i>
Lesser scaup	<i>Aythya affinis</i>
Mallard	<i>Anas platyrhynchos</i>
Pied-billed grebe	<i>Podilymbus podiceps</i>
Red-breasted merganser	<i>Mergus serrator</i>
Ring-necked duck	<i>Aythya collaris</i>
Wood duck	<i>Aix sponsa</i>

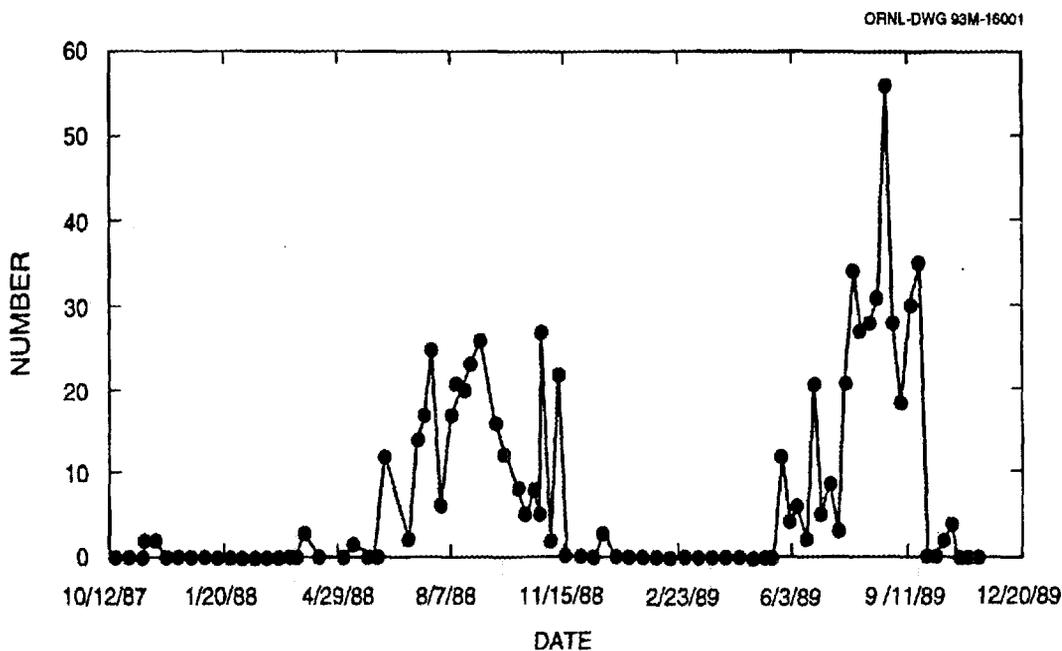


Fig. 8.6. Number of wood ducks observed on White Oak Lake per observation period, October 1987–December 1989.

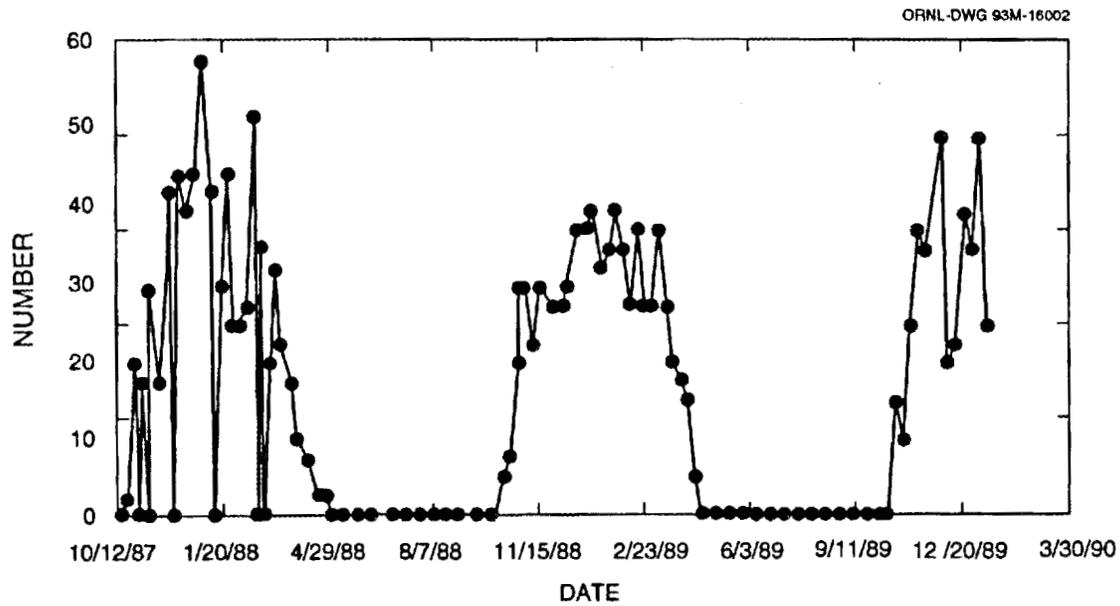


Fig. 8.7. Number of ring-necked ducks observed on Swan Pond near Oak Ridge National Laboratory per observation period, October 1987–January 1990.

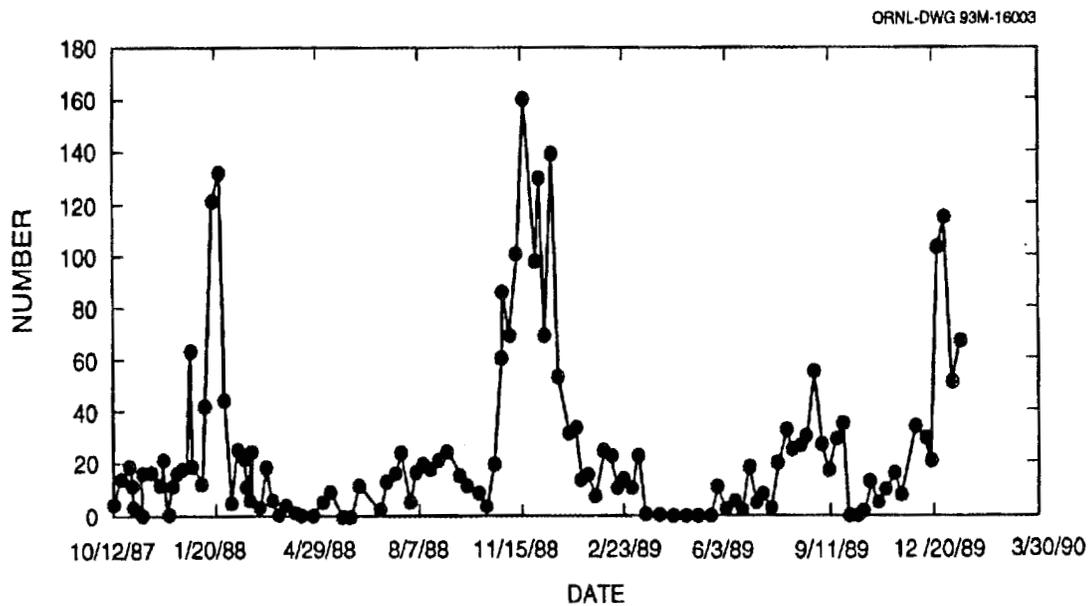


Fig. 8.8. Number of waterfowl observed on White Oak Lake per observation period, October 1987–January 1990.

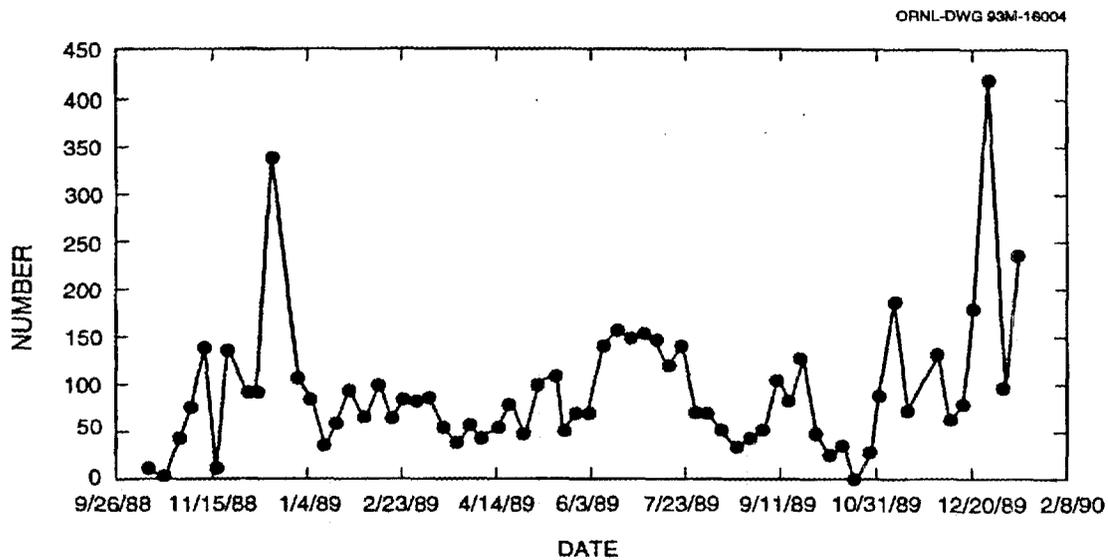


Fig. 8.9. Number of Canada geese observed at the Oak Ridge Gaseous Diffusion Plant per observation period, October 1988–January 1990.

observed at all designated ORGDP census locations.

Waterfowl species observed on water bodies near the ORGDP included mallard, gadwall, wood duck, hooded merganser, black duck, ring-necked duck, coot, snow goose, scaup, and pie-billed grebe. Gadwalls and wood ducks seemed to prefer the K-901A holding pond. The K-1007B holding pond served as a resting ground during the winter for several migratory species. Other aquatic birds using the ponds and streams included the green-backed heron, great blue heron, black-crowned night heron, and belted kingfisher.

Y-12 Plant. Canada geese and ring-necked ducks were the only waterfowl species observed at census sites near the Y-12 Plant. Canada geese also nested and raised their young at the Y-12 Plant. Approximately 1500 observations of waterfowl have been recorded at the Y-12 Plant. Approximately 200 geese have been observed in a single census period

(Fig. 8.10). The geese seemed to prefer the extensive grassy areas outside the main plant. Because the numbers of geese varied during the fall and winter months, most of the geese that were observed were probably resident or local birds mixed with transients.

8.2.3.2 Contaminants in Oak Ridge National Laboratory waterfowl

The primary radionuclide detected in waterfowl collected from ORNL was ^{137}Cs . Low levels of ^{60}Co were determined in some samples, including whole body, gastrointestinal tract, and gizzard contents. Strontium-90 was detected in most bone samples. Metals that were detected in waterfowl collected from various locations on the ORR included Ag, Cd, Hg, Pb, and Se. Additional analyses for metals and PCBs have been requested from the ACD laboratories, but the results are not yet available.

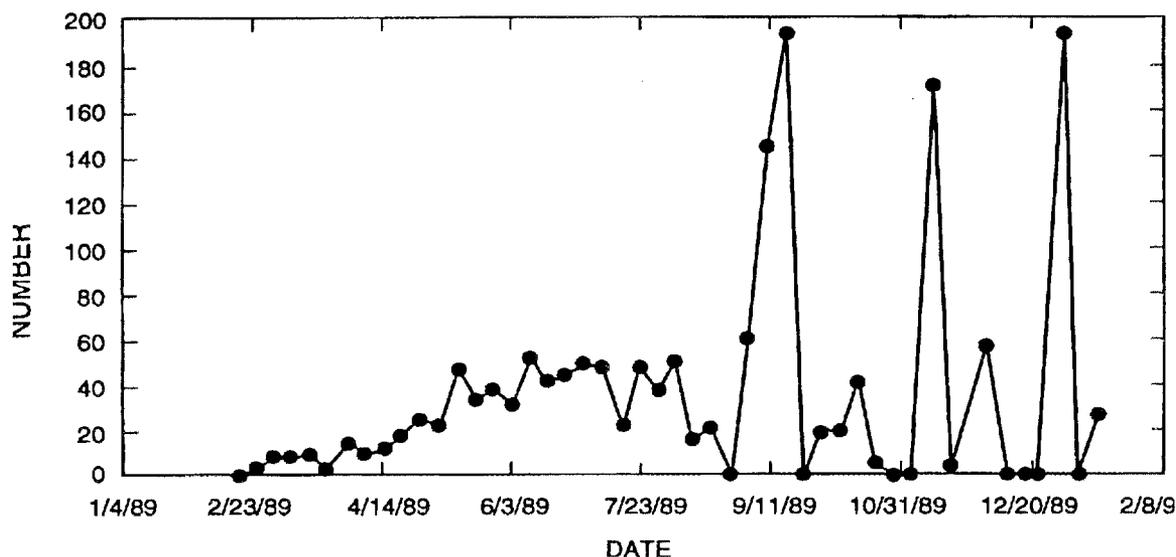


Fig. 8.10. Number of Canada geese observed at the Y-12 Plant per observation period, February 1989–January 1990.

Transient waterfowl. Transient waterfowl species collected from WOL averaged 234 Bq/kg of ^{137}Cs in breast tissue. The maximum concentration of ^{137}Cs in breast tissue was 340 Bq/kg, but the highest concentrations of ^{137}Cs were found in the gastrointestinal tract and gizzard contents. Liver concentrations approximated those observed in breast tissue. If a person were to consume 160 g of a duck contaminated with the maximum concentration of ^{137}Cs in breast tissue, the potential dose commitment would be 0.73 μSv . An individual would have to consume 218 kg of this tissue to exceed a dose of 1 mSv. Mercury and selenium concentrations in breast tissue ranged from 0.009 to 0.08 $\mu\text{g/g}$ and from 0.4 to 0.9 $\mu\text{g/g}$ respectively. These levels are well below applicable regulatory limits.

Canada geese. Cesium-137 concentrations in a pair of Canada geese collected from Basin 3524 were the highest found in any waterfowl collected on the ORR. Although the geese only used the

pond area for ~1 month, concentrations of ^{137}Cs in breast tissue were 150,000 and 106,000 Bq/kg, and ^{90}Sr concentrations in bone were 21,000 and 24,000 Bq/kg respectively (Table 8.6). The breast tissue contained higher levels of ^{137}Cs than the gizzard contents, indicating that these birds had not been feeding exclusively from Basin 3524. The potential dose commitment to humans from consumption of 1400 g of breast tissue from the goose with the highest ^{137}Cs concentration would be 2.68 mSv.

The six Canada geese collected near the ponds at the ORNL STP contained a mean concentration of 101 Bq/kg of ^{137}Cs in breast tissue (range = 36–162 Bq/kg); ^{90}Sr concentrations in four bone samples averaged 2250 Bq/kg. These geese accumulated the radionuclides within 1 month because they were whole-body counted when they were banded, and the levels were not significantly above background. Mercury concentrations ranged from 0.001 to 0.029 $\mu\text{g/g}$, 0.017 to 0.036 $\mu\text{g/g}$, and 0.013 to 0.024 $\mu\text{g/g}$, in

Table 8.6. Cesium-137 concentrations in tissues of Canada geese collected on Pond 3524 at Oak Ridge National Laboratory

Goose identification	Tissue description	Concentration (Bq/kg)
1	Breast tissue	150,000
	Liver	15,000
	Gizzard contents	55,000
	Bone	21,000
2	Breast tissue	106,000
	Liver	24,000
	Gizzard contents	66,000
	Bone	27,000

breast tissue, liver, and kidney, respectively, of three of the geese. Selenium concentrations in breast tissue, liver, and kidney ranged from <0.3 to 0.3 $\mu\text{g/g}$, 0.5 to 0.8 $\mu\text{g/g}$, and 0.4 to 0.7 $\mu\text{g/g}$ respectively.

White Oak Lake wood ducks.

Juvenile wood ducks collected from the WOL area averaged 131 Bq/kg of ^{137}Cs in breast tissue (range = 21–272 Bq/kg). Cesium-137 concentrations detected in three adult wood ducks collected from the lake averaged 131 Bq/kg in breast tissue and ranged from 107 to 163 Bq/kg. An adult wood duck collected at the WOC weir at WCK 2.65 contained 5565 Bq/kg of ^{60}Co in whole body but only 81 Bq/kg in breast tissue. The Melton Branch weir (MEK 0.16), which is located near the WOC weir, is known to contain higher levels of ^{60}Co ; therefore, the bird was probably using this area before it was collected. The immature wood ducks were expected to have higher ^{137}Cs concentrations than the transient waterfowl because they were hatched at WOL and had spent their entire life in the vicinity of the lake. However, the average ^{137}Cs concentration in the transient waterfowl

exceeds that in the wood ducks. These unexpected results may be due to differences in the radionuclide concentrations of the food sources or to metabolic differences between species and between ages.

8.2.3.3 White Oak Lake domestic mallard experiment

The domestic mallard experiment on WOL proved more successful for the second group of ducks released on October 14, 1989, than for the first group released on May 6, 1989. More than half of the ducks from Group 1 could not be accounted for, whereas 27 of the 38 birds (71%) were accounted for in Group 2. The high mortality rate in Group I was probably due to heavy predation. No birds remained after day 77 in Group 1 or after day 95 in Group 2. Radionuclide elimination rates could not be measured because too few birds survived to the end of the experiment.

Cesium-137 was the primary radionuclide detected in all mallard tissues; however, low levels of ^{60}Co were found in

several whole-body, gastrointestinal tract, gizzard contents, and liver samples. The ^{137}Cs concentrations in breast tissue from the Group 1 mallards increased from below detection on day 2 to 354 Bq/kg on day 77. Only one bird was sampled on days 2, 4, 49, and 77 in Group 1. The mean ^{137}Cs concentration in breast tissue of Group 2 mallards increased from 5.2 Bq/kg on day 2 to 277 Bq/kg on day 61. The mean whole-body ^{137}Cs concentration for Group 2 was 34 Bq/kg on day 2 and 420 Bq/kg on day 61. The accumulation curves for ^{137}Cs in breast tissue for each group are shown in Figs. 8.11 and 8.12.

Analyses for metals in breast tissue, liver, and kidney of the mallards in Group 1 detected Cd, Hg, and Se in these tissues and Ag and Pb in some samples. Arsenic was below the limits of detection in all samples analyzed. Silver concentrations ranged from <0.2 to $2.4 \mu\text{g/g}$ in breast tissue and from <0.2 to $6 \mu\text{g/g}$ in liver. Mercury concentrations in breast tissue ranged from <0.001 to $0.045 \mu\text{g/g}$ and from 0.0012 to $0.24 \mu\text{g/g}$ in liver. Selenium concentrations in breast tissue and liver ranged from <0.3 to $2.1 \mu\text{g/g}$ and 0.5 to $2.9 \mu\text{g/g}$ respectively. Tissues from Group 2 mallards are currently being analyzed for metals.

8.2.3.4 Canada goose banding study

A total of 393 Canada geese were banded at locations on and near the ORR. Both leg and neck bands were fitted to 283 geese. Though statistical analyses of the data have not been conducted, observations indicated that several of the collared birds are residents. Many of the geese banded at the ORGDP still use that area. However, movement of these geese to other areas has also been observed. Groups of geese observed at the Y-12 Plant have contained birds banded at

the Y-12 Plant, Solway Park, and Clark Center Park. On Melton Hill Reservoir, groups have included geese from a mixture of banding areas. Furthermore, geese banded at ORNL have been observed at Melton Hill Dam. Thus, the potential exists for a contaminated goose to fly to a public area.

8.3 RADIONUCLIDES IN FISH IN WHITE OAK CREEK

8.3.1 Introduction

This study was initiated in 1986 to determine if levels of radionuclides found in fish from WOL were representative of the degree of contamination found in fish from WOC and tributaries. Data on the accumulation and partitioning of ^{137}Cs were also of interest, since ^{137}Cs is a particle-associated contaminant and its transport and accumulation in this system should have much in common with the behavior of other particle-associated contaminants in WOC, such as mercury and PCBs. In addition, the data collected in 1986 and 1987 were used to evaluate the previous findings of Spalding and Cerling (1979), which showed that the main ORNL complex is the major source of ^{137}Cs to WOC and tributaries. This section of the annual report presents the results of analyses of fish collected from seven sites in the WOC watershed in December 1988.

8.3.2 Methods

Levels of ^{137}Cs were determined in bluegill and redbreast sunfish samples that had been collected and freeze-dried for metals analyses as part of the BMAP bioaccumulation task. Freeze-dried fish samples were counted by the ORNL Analytical Chemistry Division using a

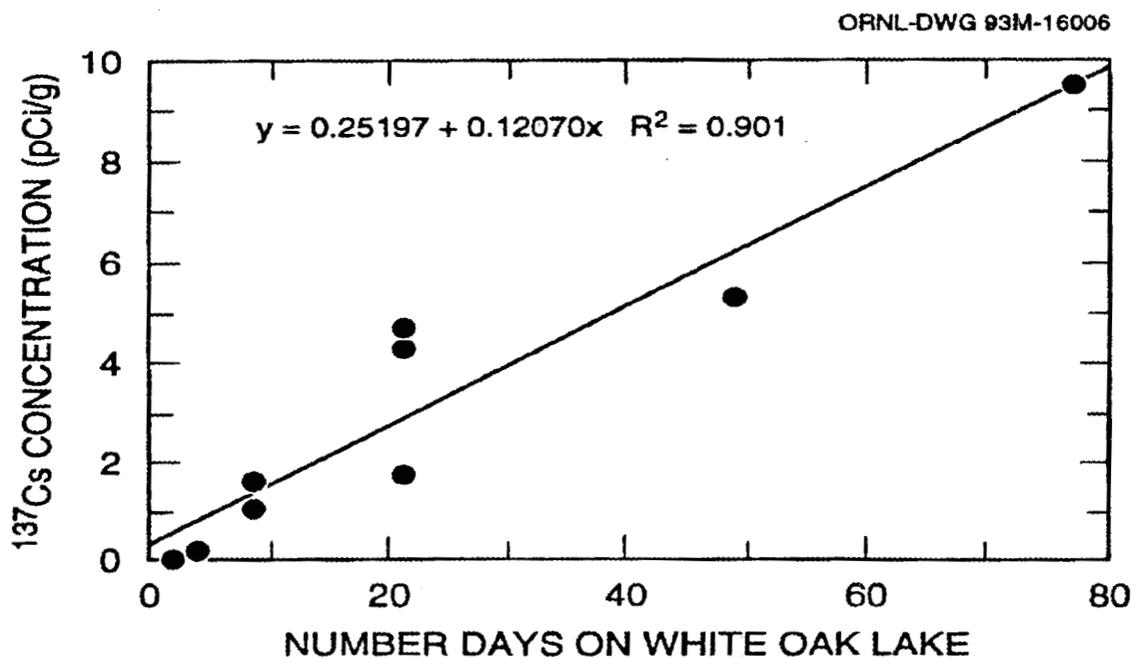


Fig. 8.11. Cesium-137 concentrations in breast tissue of domestic mallards released on White Oak Lake on May 6, 1989.

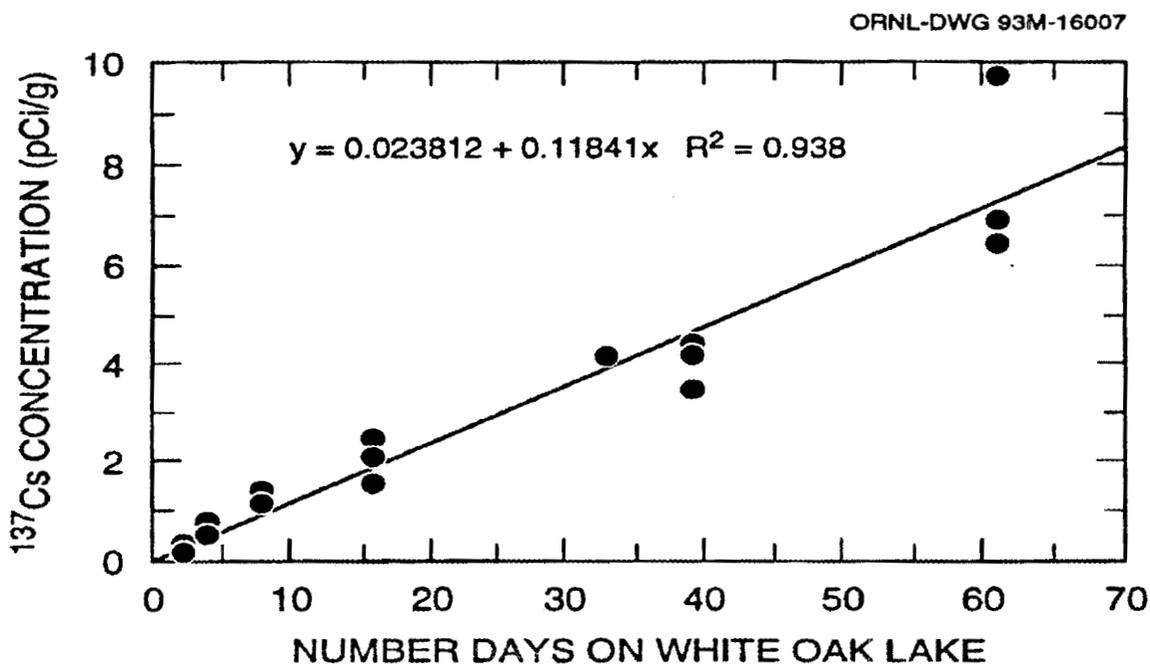


Fig. 8.12. Cesium-137 concentrations in breast tissue of domestic mallards released on White Oak Lake on October 14, 1989.

germanium lithium crystal scintillation spectrometer.

8.3.3 Results

The concentrations of ¹³⁷Cs in fish decreased substantially at sites WCK 3.5, WCK 2.9, and WCK 2.3 in 1987 (Loar 1993b) compared with 1986. However, in 1988 the concentrations of ¹³⁷Cs in fish from these same sites had returned to levels similar to those observed in 1986 (Table 8.7). Concentrations at sites farther downstream (WOL and WCK 0.9 in the WOC embayment) from ORNL exhibited substantially less year-to-year variability than those sites that were nearer ORNL facilities (WCK 3.5 and WCK 2.9). This variability at the upstream stations probably reflects short-term variability in releases of ¹³⁷Cs. Such variations would be damped at downstream sites by the incorporation of dissolved ¹³⁷Cs into abiotic and biotic compartments (sediments, macrophytes,

periphyton, and benthic macroinvertebrates) and subsequent accumulation following ingestion or redissolution. Because of the relatively short biological half-life of ¹³⁷Cs in fish, the concentrations in fish at the upstream stations may vary considerably over time if ¹³⁷Cs inputs also vary substantially over time.

Fish at site WCK 3.5 had the highest mean ¹³⁷Cs concentration of all sites and were about a factor of six higher than fish in WOL. This trend is the same as that observed in the 1986 and 1987 data. Concentrations of ¹³⁷Cs in fish decreased steadily with increasing distance downstream from the main ORNL complex. In 1986, fish from the site with the highest mean concentration (WCK 2.9) contained approximately 10 times as much ¹³⁷Cs as that found in the same species in WOL. The results of the 1986–88 analyses from the WOC system are consistent with the findings of earlier investigations (Spalding and Cerling 1979), which showed

Table 8.7. Concentrations of ¹³⁷Cs in bluegill (*Lepomis macrochirus*) from the White Oak Creek system, December 1988^a

Site ^b	¹³⁷ Cs concentrations (Bq/kg, dry weight)	
	Mean ± 1 SD	Range
WCK 3.5	18,250 ± 7,805	11,000–28,000
WCK 2.9	12,500 ± 1,290	11,000–14,000
WCK 2.3	7,425 ± 5,900	2,500–16,000
WOL	2,775 ± 770	1,900–36,000
WCK 0.9	1,550 ± 545	1,100–2,300
MEK 0.2	5,000 ± 2,500	2,600–78,001
NTK 0.2	115 ± 143	<40–330

^an = 4 at all sites.

^bWCK = White Oak Creek kilometer, WOL = White Oak Lake, MEK = Melton Branch kilometer, and NTK = Northwest Tributary kilometer.

^cSD = standard deviation.

that the main ORNL complex is the major source of ^{137}Cs to WOC and tributaries and that the Melton Valley facilities and SWSAs are of secondary importance. Fish were collected from these same sites in 1989 and will again be analyzed to determine whether the pattern of ^{137}Cs contamination in the WOC watershed has changed.

8.4 FUTURE STUDIES

8.4.1 Radionuclides in White Oak Lake

Estimates were made in 1987 of the quantities of radionuclides in the biotic and abiotic compartments of the WOL ecosystem, and a screening analysis based on these estimates was performed to determine the potential human risk of exposure (Loar 1993a). In 1988 and 1989, additional data became available, and the earlier estimates will be revised. An updated screening analysis will be conducted if the new estimates on radionuclide concentrations in the ecosystem compartments are substantially different.

8.4.2 Role of Aquatic Macroflora in the Radioecology of White Oak Lake

Macrophytes exert a potentially significant influence on the cycling of radionuclides in WOL. To understand how macrophytes influence radionuclide dynamics, consideration must be given to the interception and uptake of radionuclides by macrophytes and the subsequent fate of these contaminants during the decomposition process. In 1988, research efforts focused on the source of ^{137}Cs and ^{60}Co to rooted, submerged aquatic plants in the WOC weir system. Studies conducted in 1989

provided information on macrophyte decomposition, including the loss of radionuclides from the plants and the fraction of macrophyte-associated radionuclides bound internally or externally by the macrophytes. These results indicated that some of the radionuclides taken up from the sediments are released into the water column as the plants decompose. Monitoring the radionuclide concentrations associated with macrophytes in WOL will continue in 1990.

In fall 1989, experiments were set up in large outdoor mesocosms to determine the fate of macrophyte-associated ^{137}Cs and ^{60}Co that are bound internally or externally (exchangeable) as the plants senesced and decomposed under controlled, but near natural conditions. These studies will determine how much of the macrophyte radionuclide pool is released into the external environment during decomposition and in what form (dissolved or particulate). Radionuclides released to the water in a dissolved form are more available for uptake by aquatic biota. Radionuclides associated with large organic or inorganic particles can become incorporated into the sediments or be transported downstream.

Recent experiments in the ^{137}Cs -contaminated lakes at the Savannah River Plant showed that milfoil accumulated most of its ^{137}Cs pool from the sediments, but other rooted aquatic species acquired all of their ^{137}Cs from the water column (Evans and Cocke 1979). Radionuclide uptake experiments with macrophytes will be conducted on different sediment types and radionuclide concentrations to determine how these variables affect uptake. Such studies will provide a better understanding of the environmental factors that directly influence radionuclide uptake and will permit extrapolations to be made to other similarly contaminated systems on the ORR and elsewhere.

8.4.3 Waterfowl Populations Associated with Oak Ridge Reservation Basins and Ponds

The waterfowl census will continue through 1990, and plans are being formulated to band additional Canada geese this summer. A preliminary analysis of data on the movement of Canada geese on the ORR indicates that the frequency of observations and the number of sites observed should be increased. Further analyses will be conducted on these data. Waterfowl observations on Basin 3524 will continue and birds that attempt to use this area as a nesting site either will be discouraged or sacrificed. Previous collections of Canada geese from Basin 3524 have shown these birds to have the highest concentrations of ^{137}Cs found in waterfowl from the ORR.

Additional transient waterfowl will be collected from WOL and ORGDP and analyzed for radionuclides, metals, and PCBs. Canada geese from ORGDP and the Y-12 Plant will also be analyzed for metals and PCBs. The emphasis that was initially placed on the accumulation of radionuclides in waterfowl will shift to address the accumulation of inorganic and organic contaminants.

The collection of field data was completed for the domestic mallard experiment on WOL. Analysis of tissues from these birds for metals and PCBs will continue, and as the data become available, they will be analyzed with respect to the potential exposure to humans via the waterfowl pathway.

8.4.4 Radionuclides in White Oak Lake Sediment

Because ~99% of the ^{137}Cs , ^{60}Co , and ^{90}Sr in WOL is in the sediments, a reliable estimate of the quantity of radionuclides in the WOL ecosystem will not be available

until a comprehensive core sampling program is completed and better estimates of the volume and radionuclide concentrations of the sediment are obtained. In addition, the present estimate of the inventory of radioactivity in the sediment compartment of WOL includes only ^{137}Cs , ^{60}Co , and ^{90}Sr . Although it is assumed that the quantity of other radionuclides in WOL, including the transuranic elements, is relatively small, this assumption should be documented with reliable measurements from core samples. Core sampling of the lake sediments is not included in the present task because it will be conducted as part of the remedial investigation on WOC and WOL under the corrective action requirements of the RCRA (Sect. 3004).

An inventory of the radionuclides in the WOL ecosystem will not be complete without estimates of the concentrations in the biotic and abiotic compartments of the floodplain and former lake bed. Samples of the vegetation on the floodplain and former lake bed will be collected and analyzed for ^{137}Cs , ^{60}Co , and ^{90}Sr . Measurements of plant cover and abundance by species will be taken to estimate the biomass of the floral compartment. Soil samples from the floodplain and former lake bed will be analyzed to obtain a preliminary estimate of the radioactivity in soil.

8.4.5 Radionuclides in Fish from White Oak Creek

Fish were collected from the same WOC sites in 1989 and are currently being analyzed for radionuclides and other contaminants. These results will be used to investigate the hypothesis that the concentration of ^{137}Cs in fish from WOC can be correlated with changes in the waste treatment system. To investigate the temporal variability of ^{137}Cs in fish in

WOC/WOL, fish will be collected at more frequent intervals at stations WCK 2.9 and WOL. Consideration will be given to measuring ^{137}Cs in soluble and suspended

fractions of the water column and determining the ^{137}Cs inventory of sunfish populations in WOC.

9. ABATEMENT PROGRAM

C. K. Valentine

In several areas, efforts to abate aquatic toxicity and other impacts on aquatic environments have been and will continue to be focused on management initiatives. These areas include the following: (1) minimization of chlorine and ethylene glycol concentrations in surface streams; (2) implementation of monitoring plans that are designed to locate sources that contribute PCBs and mercury to surface streams, to characterize the sources, and to develop remedial measures; (3) improvements to existing underground wastewater piping systems, including relining and rerouting pipelines, to enhance control of ORNL wastewater streams; (4) secondary containment/diking of tanks containing hazardous/toxic substances, and of transformer oils; (5) enhanced interaction between ORNL EMC Section personnel and operational/construction personnel in the field, on an individual, site-specific basis; (6) completion of the ORNL NRWTF; (7) elimination of Category III (untreated process/laboratory wastewater) outfalls; and (8) implementation of a best management practices (BMP) program.

9.1 CHLORINE REDUCTION

It has become evident that chlorine is or has the potential to be a major source of toxicity to aquatic life in ORNL streams. Because the potable water system at ORNL has a chlorine content of ~1.8 mg/L, there is always the potential for toxic levels in the stream. Large cooling towers at ORNL continue to receive special attention through NPDES sampling.

Use of chlorine as a biocide in cooling towers to ensure their proper function during warm months warrants sampling of blowdown before release to the receiving stream. Although the NPDES permit requires only quarterly compliance sampling, chlorine is always checked by operations staff; if it is not below 0.2 mg/L, the blowdown is not to be discharged. Environmental concerns have been addressed on several projects involving maintenance of existing cooling towers and installation of new cooling systems to ensure that chlorine discharges to ORNL streams are either prevented or are kept at levels that are as low as reasonably achievable.

In addition to cooling towers, numerous ORNL facilities use once-through cooling water systems that are fed by the potable water system. Because some of these systems currently discharge to the ORNL storm drain system, they represent potential sources of chlorine in ORNL surface streams. In the summer of 1989, ORNL issued a position statement that chlorinated water discharges exceeded Tennessee narrative water quality criteria. It was requested that no new chlorinated water discharges be initiated at ORNL and that projects be developed to address existing discharges. Accordingly, a chlorine reduction GPP is being developed. Under the GPP, major ORNL discharges of chlorinated once-through cooling water have been identified and characterized, and engineering projects are underway to provide chlorine reduction measures at these outfalls. In response to an ORNL study on the relationship between chlorination levels at the ORNL STP and

fecal coliform bacteria concentrations in the receiving stream (Loar 1993b, Sect. 3.1.3.1), the TDHE has issued a draft NPDES permit modification that may allow the STP to operate with a decreased level of effluent chlorination.

Water pipeline leaks and breaks have been a source of problems in the past and most likely will continue to be a potential problem. To address this issue, two aerators have been purchased. The intent is to be prepared to place an aerator anywhere in the stream where chlorine levels are high and, through dissipation, reduce the chlorine level. This equipment provides the capability for quick abatement of an instream chlorine problem.

9.2 ETHYLENE GLYCOL

There have been numerous leaks and breaks in the ethylene glycol cooling system at ORNL, particularly in the 4500 Area during the past year. The ethylene glycol chilling system is ~40-years old, and the problem can be attributed to the age of the system. The presence of ethylene glycol in streams enhances microbial growth and reduces the oxygen level in the stream. The aforementioned aerators will also be used to help reduce the deoxygenating effect of ethylene glycol on the streams. It has been determined that the cooling system can be operated without the ethylene glycol component. A plan to replace ethylene glycol with an alternative solution not containing glycol has been developed. The current plan is for the bulk of the ethylene glycol solution to be used as a carbon source for a biological nitrate waste treatment system at the Y-12 Plant. The remainder of the glycol would be treated via the ORNL STP, using one of the STP surface impoundments and the extended aeration treatment process at that facility. Completion of this project

will greatly reduce ORNL's vulnerability with respect to ethylene glycol releases.

9.3 TANK DIKING

In the past there have been occasions when storage tanks have leaked, and the potential exists for the contents of the tank to reach receiving streams. All aboveground tanks storing hazardous or toxic chemicals in volumes >208 L, including transformers, have been evaluated and prioritized, based on the toxicity of the contents, to ensure that secondary containment of the tanks has been addressed.

9.4 POLYCHLORINATED BIPHENYL AND MERCURY MONITORING

Efforts are continuing on the ORNL PCB and Mercury Monitoring Plans that are required under the ORNL NPDES permit (EPA 1986a). A formal report that documents the findings of the Mercury Monitoring Plan has been issued (Taylor 1989), and a PCB report (Taylor 1990a) and a second mercury report (Taylor 1990b) have been prepared. These documents will be submitted to DOE in early 1990.

9.5 WASTEWATER PIPING

Projects were completed to repair numerous ORNL underground pipelines found to be leaking, via a process that installs a formed-in-place synthetic liner inside an existing pipe. Leaking pipes at the ORNL site can be problematic due to exfiltration of pipe contents or infiltration of groundwater. The pipeline repairs have resulted in a significant reduction of contaminants transported into First Creek

and WOC. During 1989, additional pipelines were identified and characterized by ORNL staff as requiring lining *in situ*. Engineering projects were developed and initiated, and will be completed during the first quarter of 1990.

9.6 FIELD INTERFACE ACTIVITIES

The potential exists for contaminants to reach ORNL streams because of construction activities or for the streams to be adversely affected by sedimentation due to surface runoff associated with construction projects and other ORNL activities. Recently, efforts to control and inspect construction activities have increased greatly. The Field Interface Staff of EMC conducts field assessments of activities conducted by ORNL staff and contractors to detect and prevent potential environmental problems and to determine compliance with environmental laws and regulations. This activity involves on-site monitoring as well as interacting with a wide range of personnel involved with these projects. Each activity is monitored to determine potential environmental problems. If problems are identified, members of the Field Interface Staff work with those involved to address the environmental concern. This activity is also closely associated with emergency response at ORNL and has been a primary factor in preventing spills from reaching receiving streams.

9.7 NONRADIOLOGICAL WASTEWATER TREATMENT FACILITY

Construction was completed during 1989 on the NRWTF, which became operational in March 1990. The NRWTF is expected to greatly reduce the impact of ORNL facilities and operations on the aquatic environment at ORNL. The

NRWTF is a wastewater treatment facility that will remove organics, metals, and other solids from wastewaters that currently receive minimal levels of treatment before discharge. The NRWTF also includes large, aboveground flow-equalization tanks that replace several existing surface impoundments that provided that function; the impoundments are being removed from service. Wastewater from numerous ORNL buildings and processes will be routed to the NRWTF for treatment.

In December 1989, the TDHE inspected the facility, approved its construction, and granted permission to commence operations. The facility was operating in compliance with NPDES permit limits and conditions by March 31, 1990, as required by the permit. The NRWTF has resulted in the elimination of seven major ORNL point-source wastewater discharges to WOC, Melton Branch, and associated tributaries.

9.8 ELIMINATION OF CATEGORY III OUTFALLS

Extensive efforts were put forth in 1989 to further characterize, remediate and/or eliminate Category III discharges to surface streams. Category III discharges are identified in the ORNL NPDES permit and Federal Facility Compliance Agreement as discharges with the potential to contain untreated laboratory or process wastewater. All 32 Category III outfalls were remediated and or eliminated by March 1990 (Table 2.4).

9.9 BEST MANAGEMENT PRACTICES PROGRAM

A process was initiated in late 1989 that will require and provide a BMP plan for each ORNL project that is reviewed under the Environmental Review and

Documentation Program (ERDP) of the Environmental and Health Protection Division. The process was developed by EMC and ERDP personnel to ensure that, for every project having the potential to adversely impact surface waters and/or groundwater, a BMP plan is developed and implemented that contains measures to be followed that will prevent or minimize the potential for these impacts. Each BMP plan will be developed jointly by

EMC and project personnel, and will stand as a document of agreement between the Environmental and Health Protection Division and the managers of the project. The BMPs will be prepared as part of ERDP documentation and will be subject to review and oversight by the Field Interface Staff of EMC. The individual BMP plans will be maintained on file as amendments to the facility-wide ORNL BMP plan.

10. REFERENCES

- Adams, S. M., and M. D. McCracken. 1974. Seasonal production of the *Myriophyllum* component of the littoral of Lake Wingra, Wisconsin. *J. Ecol.* 62:457-467.
- Adams, S. M., and R. B. McLean. 1985. Estimation of largemouth bass, *Micropterus salmoides* Lacepede, growth using the liver somatic index and physiological variables. *J. Fish Biol.* 26:111-126.
- Adams, S. M., J. E. Breck, and R. B. McLean. 1985. Cumulative stress-induced mortality of gizzard shad in a southeastern U.S. reservoir. *Environ. Biol. Fish.* 13:103-112.
- Adams, S. M., K. L. Shepard, M. S. Greeley, B. D. Jimenez, M. G. Ryon, L. R. Shugart, and J. F. McCarthy. 1989. The use of bioindicators for assessing the effects of pollutant stress on fish. *Mar. Environ. Res.* 28:459-464.
- Addison, R. F. 1984. Hepatic mixed function oxidase (MFO) induction in fish as a possible biological monitoring system. pp. 51-60. IN: V. W. Cairns, P. V. Hodson, and J. O. Nriagu, (eds.), *Contaminant effects on fisheries*. John Wiley & Sons, New York.
- Aho, J. M., C. S. Anderson, and J. W. Terrell. 1986. Habitat suitability index models and instream flow suitability curves: Redbreast sunfish. U.S. Fish Wildl. Serv. Biol. Rep. 82(10.119). U.S. Fish and Wildlife Service, Ft. Collins, Colorado. 23 pp.
- Ahokas, J. T., N. T. Kärki, A. Oikari, and A. Soivio. 1976. Mixed function monooxygenase of fish as an indicator of pollution of aquatic environments by industrial effluent. *Bull. Environ. Contam. Toxicol.* 16:270-274.
- Allain, C. C., L. S. Poon, C. S. G. Chan, W. Richmond, and P. C. Fu. 1974. Enzymatic determination of total serum cholesterol. *Clin. Chem.* 20:470-475.
- Amano, H., C. T. Garten, Jr., and R. D. Lomax. 1987. A field survey of environmental tritium in areas adjacent to ORNL solid-waste storage areas. ORNL/TM-10438. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- APHA (American Public Health Association). 1985. *Standard methods for the evaluation of water and wastewater*. Sixteenth Edition. American Public Health Association, Washington, D.C.
- Barron, M. G., and I. R. Adelman. 1984. Nucleic acid, protein content, and growth of larval fish sublethally exposed to various toxicants. *Can. J. Fish. Aquat. Sci.* 41:141-150.
- Bates, L. D., J. B. Berry, G. E. Butterworth, S. P. du Mont, C. A. Easterday, A. H. Geisler, L. G. Hill, C. M. Kendrick, L. E. McNeese, T. E. Myrick, R. E. Pudelek, P. S. Rohwer, T. F. Scanlan, L. E. Stratton, J. R. Trabalka, and E. L. Youngblood. 1988. ORNL long-range environmental and waste management plan: Detailed management plan. ORNL/6446/R1. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Beamish, R. J., and G. A. McFarlane. 1983. The forgotten requirement for age validation in fisheries biology. *Trans. Amer. Fish. Soc.* 112:735-743.
- Beamish, R. J., and G. A. McFarlane. 1987. Current trends in age determination methodology. pp. 15-42. IN: R. C. Summerfelt and G. E. Hall (eds.), *Age and growth of fish*, Iowa State University Press, Ames, Iowa.

- Becker, G. C. 1983. Fishes of Wisconsin. University of Wisconsin Press, Madison, Wisconsin.
- Bennett, G. W. 1970. Management of lakes and ponds. Van Nostrand Reinhold Company, New York.
- Bergmeyer, H. Y., P. Scheibe, and A. W. Wahlefeld. 1978. Optimization of aspartate amino transferase and alanine transferase. Clin. Chem. 24:58-73.
- Berkman, H. E., and C. F. Rabeni. 1987. Effect of siltation on stream fish communities. Environ. Biol. Fishes 18:288-294.
- Berry, J. B., L. G. Hill, P. E. Hollenbeck, L. E. McNeese, T. E. Myrick, R. E. Pudelek, P. S. Rohwer, J. H. Smith, H. R. Yook, and E. L. Youngblood. 1987. Preliminary ORNL long-range environmental management plan: Program overview and summary. ORNL-6344. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Blasing, T. J., K. L. Daniels, P. Y. Goldberg, B. M. Horwedel, I. L. McCollough, F. R. O'Donnell, A. E. Osborne-Lee, M. F. Tardiff, S. W. Teeters, C. K. Valentine, and D. A. Wolf. 1989. Environmental data report for the third quarter of 1988. ORNL/TM-534. Environmental Monitoring and Compliance Section, Environmental and Health Protection Division, Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Bligh, E. G., and W. J. Dyer. 1959. A rapid method of total lipid extraction and purification. Can. J. Biochem. Physiol. 8:911-917.
- Boyd, C. E. 1971. The dynamics of dry matter and chemical substances in a *Juncus effusus* population. Am. Midl. Nat. 86:28-45.
- Boyle, J. W., R. Blumberg, S. J. Cotter, G. S. Hill, C. R. Kerley, R. H. Ketelle, R. L. Kroodsmas, D. W. Lee, R. C. Martin, R. D. Roop, D. N. Secora, W. P. Staub, and R. E. Thoma. 1982. Environmental analysis of the operation of the Oak Ridge National Laboratory (X-10 Site). ORNL-5870. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Bradford, M. M. 1976. A rapid and sensitive method for the quantitation of protein utilizing the principle of protein-dye binding. Anal. Biochem. 72:248-254.
- Bradley, D. W., J. E. Maynard, G. Emery, and H. Webster. 1972. Transferase activity in serum of long-term hemodialysis patients. Clin. Chem. 18:1142.
- Brown D. A., R. W. Gossett, G. P. Hershelman, C. F. Ward, A. M. Westcott, and J. N. Cross. 1986. Municipal wastewater contamination in the Southern California Bight: Part I-Metal and organic contaminants in sediments and organisms. Mar. Environ. Res. 18:291-310.
- Bucolo, G., and H. David. 1973. Quantitative determination of serum triglycerides by the use of enzymes. Clin. Chem. 19:476-482.
- Bulow, F. J. 1970. RNA-DNA ratios as indicators of recent growth rates of a fish. J. Fish. Res. Board Can. 27:2343-2349.
- Burke, M. D., and R. T. Mayer. 1974. Ethoxyresorufin: Direct fluorimetric assay of a microsomal O-dealkylation which is preferentially inducible by 3-methylcholanthrene. Drug. Metab. Dispos. 2:583-588.
- Carlander, K. D. 1981. Caution on the use of the regression method of back-calculating lengths from scale measurements. Fisheries 6:2-4.
- Carlander, K. D. 1982. Standard intercepts for calculating lengths from scale measurements for some centrarchid and percid fishes. Trans. Amer. Fish. Soc. 111:332-336.

- Carle, F. L., and M. R. Strub. 1978. A new method for estimating population size from removal data. *Biometrics* 34:621-630.
- Carpenter, S. R., and S. M. Adams. 1977. The macrophyte tissue nutrient pool of a hardwater eutrophic lake: Implications for macrophyte harvesting. *Aquat. Bot.* 3:239-255.
- Cerling, T. E., and B. P. Spalding. 1982. Distribution and relationship of radionuclides to streambed gravels in a small watershed. *Environ. Geol.* 4:99-116.
- Cone, R. S. 1989. The need to reconsider the use of condition indices in fishery science. *Trans. Amer. Fish. Soc.* 118:510-514.
- Crossley, D. A., Jr. 1969. Comparative movement of ^{106}Ru , ^{60}Co , and ^{137}Cs in arthropod food chains. pp. 687-695. IN: D. J. Nelson and F. C. Evans (eds.), *Proc. 2nd Natl. Symp. on Radioecology*. CONF-670503. U.S. Atomic Energy Commission, Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Dahlman, R. C., and P. Van Voris. 1976. Cycling of ^{137}Cs in soil and vegetation of a floodplain 30 years after initial contamination. pp. 291-298. IN: C. E. Cushing (ed.), *Radioecology and Energy Resources*. Dowden, Hutchinson, and Ross, Inc., Stroudsburg, Pennsylvania.
- Dorhmann Company. 1984. Total organic carbon systems manual. Santa Clara, California.
- Doumas, B. T. 1972. *Standard methods in clinical chemistry*, Vol. 7. Academic Press, New York.
- Edgar, D. E. 1978. An analysis of infrequent hydrologic events with regard to existing streamflow monitoring capabilities in White Oak Creek watershed. ORNL/TM-6542. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Elliott, J. M. 1977. Some methods for the statistical analysis of samples of benthic invertebrates. *Freshwater Biological Association, Ambleside, England. Sci. Pub.* No. 25.
- EPA (Environmental Protection Agency). 1976. Quality criteria for water. EPA-440/9-76-023. U.S. Environmental Protection Agency, Washington, DC.
- EPA (Environmental Protection Agency). 1979. Methods for chemical analysis of waters and wastes. EPA 600/4-79-020. Environmental Monitoring and Support Laboratory, U.S. Environmental Protection Agency, Cincinnati, Ohio.
- EPA (Environmental Protection Agency). 1980a. Water quality criteria for toxic substances. *Fed. Regist.* 45:79318-79379, November 28, 1980.
- EPA (Environmental Protection Agency). 1980b. Interim methods for the sampling and analysis of priority pollutants in sediments and fish tissues. EPA 600/4-81-055. Environmental Monitoring and Support Laboratory, U.S. Environmental Protection Agency, Cincinnati, Ohio.
- EPA (Environmental Protection Agency). 1983. Methods for chemical analysis of water and wastes. EPA-600/4-79-020. Environmental Monitoring and Support Laboratory, U.S. Environmental Protection Agency, Cincinnati, Ohio.
- EPA (Environmental Protection Agency). 1984. Extraction and analysis of priority pollutants in biological tissue, Method PPB 12/83. Environmental Services Division, Region IV, Analytical Support Branch, U.S. Environmental Protection Agency, Athens, Georgia. Mimeo.
- EPA (Environmental Protection Agency). 1985. Ambient water quality criteria for chlorine. EPA-440/5-84/030. National Technical Information Service, Springfield, Virginia.

- EPA (Environmental Protection Agency). 1986a. Authorization to discharge under the National Pollutant Discharge Elimination System, Permit No. TN0002941, Oak Ridge National Laboratory (X-10) and Fact Sheet, April 1, 1986. U.S. Environmental Protection Agency, Region IV, Atlanta, Georgia.
- EPA (Environmental Protection Agency). 1986b. Quality criteria for water. EPA/440/5-86/001. Office of Water Regulations and Standards, U.S. Environmental Protection Agency, Washington, D.C.
- EPA (Environmental Protection Agency). 1988. Summary of water quality criteria for aluminum. Fed. Regist. 53(168):33177-33179.
- Erickson, C. M. 1983. Age determination of Manitoban walleyes using otoliths, dorsal spines, and scales. N. Amer. J. Fish. Manag. 3:176-181.
- Etnier, D. A. 1978. Unpublished Tennessee Valley Authority surveys of Tennessee system, (1939-1942). Department of Zoology, The University of Tennessee, Knoxville, Tennessee.
- Evaldi, R. D. 1986. Streamflow and specific-conductance data for Bear Creek, August 13, 1985, the Oak Ridge Reservation, Tennessee. Open-File Report 85-682. U.S. Geological Survey, Nashville, Tennessee.
- Evans, D. W., and P. J. Cocke. 1979. Sediments act as a source of ^{137}Cs to submersed rooted aquatic plants. pp. 64-67. IN: Savannah River Ecology Laboratory Annual Report, Aiken, South Carolina.
- Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Trans. Amer. Fish. Soc. 113:39-55.
- Fitz, R. B. 1968. Fish habitat and population changes resulting from impoundment of Clinch River by Melton Hill Dam. J. Tenn. Acad. Sci. 43:7-15.
- Fletcher, G. L., M. J. King, J. W. Kiceniuk, and R. F. Addison. 1982. Liver hypertrophy in winter flounder following exposure to experimentally oiled sediments. Comp. Biochem. Physiol. 73C:457-462.
- Forlin, L., and T. Anderson. 1985. Storage conditions of rainbow trout liver cytochrome P-450 and conjugating enzymes. Comp. Biochem. Physiol. 3:569-572.
- Francis, C. W., and F. S. Brinkley. 1976. Preferential adsorption of ^{137}Cs to micaceous minerals in contaminated freshwater sediment. Nature 260:511-512.
- Friedman, K. J., D. M. Easton, and M. Nash. 1986. Temperature-induced changes in fatty acid composition of myelinated and non-myelinated axon phospholipids. Comp. Biochem. Physiol. 83B:313-319.
- Garman, G. C., and T. F. Waters. 1983. Use of the size-frequency (Hynes) method to estimate annual production of a stream fish population. Can. J. Fish. Aquat. Sci. 40:2030-2034.
- Garten, C. T., Jr. 1987. Technetium-99 cycling in deciduous forests: Review and ecosystem model development. Environ. Internatl. 13:311-321.
- Garten, C. T., Jr., and R. D. Lomax. 1987. Strontium-90 contamination in vegetation from radioactive waste seepage areas at ORNL, and theoretical calculations of ^{90}Sr accumulation by deer. ORNL/TM-10453. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Garten, C. T., Jr., C. S. Tucker, and B. T. Walton. 1986. Environmental fate and distribution of technetium-99 in a deciduous forest ecosystem. J. Environ. Radioactivity 3:163-188.

- Glebe, B. D., and W. C. Leggett. 1981. Temporal, intra-population differences in energy allocation and use by American shad (*Alosa sapidissima*) during the spawning migration. *Can. J. Fish. Aquat. Sci.* 38:795-805.
- Godfrey, P. J. 1978. Diversity as a measure of benthic macroinvertebrate community response to water pollution. *Hydrobiologia* 57:111-122.
- Greeson, P. E., T. A. Ehlke, G. A. Irwin, B. W. Lium, and K. V. Slack. 1977. Methods for collection and analysis of aquatic biological and microbiological samples. Book 5, Chapter 4A, pp. 1-332. IN: U.S. Geological Survey, Techniques of Water-Resources Investigations of the United States Geological Survey. U.S. Government Printing Office, Washington, D.C.
- Haines, T. A. 1973. An evaluation of RNA-DNA ratio as a measure of long-term growth in fish populations. *J. Fish. Res. Board Can.* 30:195-199.
- Harvey, H. R., and J. S. Patton. 1981. Solvent focusing for rapid and sensitive quantification of total lipids on chromarods. *Anal. Biochem.* 116:312-316.
- Heath, A. G. 1987. Water pollution and fish physiology. CRC Press, Boca Raton, Florida.
- Heidinger, R. C., and K. Clodfelter. 1987. Validity of the otolith for determining age and growth of walleye, striped bass, and smallmouth bass in power plant cooling ponds. pp. 241-251. IN: R. C. Summerfelt and G. E. Hall (eds.), Age and growth of fish, Iowa State University Press, Ames, Iowa.
- Heidinger, R. C., and S. D. Crawford. 1977. Effect of temperature and feeding rate on the liver-somatic index of the largemouth bass, *Micropterus salmoides*. *J. Fish. Res. Board Can.* 34:633-638.
- Henry, R. J., D. C. Cannon, and J. W. Winkelman. 1974. Clinical chemistry: Principles and techniques, 2nd ed. Harper and Row, Hagerstown, Maryland.
- Hile, R. 1936. Age and growth of the cisco, *Leucichthys artedi* (LeSeur), in the lakes of the northeastern highlands, Wisconsin. *U.S. Bur. Fish. Bull.* 48:211-317.
- Hinton, D. E., and J. A. Couch. 1984. Pathological measures of marine pollution effects. pp. 7-32. IN: H. H. Harris (ed.), Concepts in marine pollution measurements. Maryland Sea Grant College, University of Maryland, College Park, Maryland.
- Hoffman, F. O., B. G. Blaylock, C. C. Travis, K. L. Daniels, E. L. Etnier, K. E. Cowser, and C. W. Weber. 1984. Preliminary screening of contaminants in sediments. ORNL/TM-9370. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Horning, W. B., and C. I. Weber (eds.). 1985. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms. EPA/600/4-85/014. U.S. Environmental Protection Agency, Cincinnati, Ohio.
- Howard-Williams, C., and B. R. Davies. 1979. The rates of dry matter and nutrient loss from decomposing *Potamogeton pectinatus* in a brackish south-temperate coastal lake. *Freshwater Biol.* 9:13-21.
- Huckabee, J. W., and B. G. Blaylock. 1973. Transfer of mercury and cadmium from terrestrial to aquatic ecosystems. pp. 125-160. IN: S. K. Dhar (ed.), Metal Ions in Biological Systems. Adv. Exper. Med. Biol.
- Huston, M. A. 1979. A general hypothesis of species diversity. *Amer. Nat.* 113:81-101.
- Jeffrey, S. W., and G. F. Humphrey. 1975. New spectrophotometric equations for determining chlorophylls *a*, *b*, *c*₁ and *c*₂ in higher plants, algae and natural phytoplankton. *Biochem. Physiol. Pflanz.* 167:191-194.

- Johannesen, K. M., and J. W. DePierre. 1978. Measurements of cytochrome P₄₅₀ in the presence of large amounts of contaminating hemoglobin and methemoglobin. *Anal. Biochem.* 86: 725-732.
- Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6:21-27.
- Karr, J. R. 1987. Biological monitoring and assessment: A conceptual framework. *Environ. Manag.* 11:249-256.
- Karr, J. R., and D. R. Dudley. 1981. Ecological perspective on water quality goals. *Environ. Manag.* 5:55-68.
- Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: A method and its rationale. Illinois Natural History Survey Special Publication 5.
- Kasten, J. L. 1986. Resource Management Plan for the Oak Ridge Reservation, Vol. 21: Water Conservation Plan for the Oak Ridge Reservation. ORNL/ESH-1/V21. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Kondratieff, P. F., R. A. Matthews, and A. L. Buikema, Jr. 1984. A stressed stream ecosystem: Macroinvertebrate community integrity and microbial trophic response. *Hydrobiologia* 111:81-91.
- Krumholz, L. A. 1956. Observations on the fish populations of a lake contaminated by radioactive wastes. *Bull. Am. Mus. Nat. His.* 110(4):281-367.
- Kszos, L. A., and A. J. Stewart. 1991. Effort-allocation analysis of the 7-d fathead minnow (*Pimephales promelas*) and *Ceriodaphnia dubia* toxicity tests. *Environ. Toxicol. Chem.*
- Kszos, L. A., A. J. Stewart, L. F. Wicker, and G. M. Logsdon. 1989. Environmental Sciences Division Toxicology Laboratory Standard Operating Procedures. ORNL/TM-11194. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Larimore, R. W., W. F. Childers, and C. Heckrotte. 1959. Destruction and reestablishment of stream fish and invertebrates affected by drought. *Trans. Amer. Fish. Soc.* 88:261-285.
- Larkin, P. A. 1978. Fisheries management: An essay for ecologists. *Ann. Rev. Ecol. System.* 9:57-73.
- Lee, D. S., C. R. Gilbert, C. H. Hocutt, R. E. Jenkins, D. E. McAllister, and J. R. Stauffer, Jr. 1980. Atlas of North American freshwater fishes. North Carolina Biological Survey Publication 1980-12, North Carolina State Museum of Natural History.
- Lee, R. M., S. D. Gerking, and B. Jezierska. 1983. Electrolyte balance and energy mobilization in acid-stressed rainbow trout, *Salmo gairdneri*, and their relation to reproductive success. *Environ. Biol. Fish.* 8:115-123.
- Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. *J. N. Amer. Benthol. Soc.* 7:222-233.
- Leonard, P. M., and D. J. Orth. 1986. Application and testing of an index of biotic integrity in small, coldwater streams. *Trans. Amer. Fish. Soc.* 115:401-414.
- Loar, J. M. (ed). 1992. First report on the Oak Ridge Y-12 Plant Biological Monitoring and Abatement Program for East Fork Poplar Creek. Y/TS-886. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Loar, J. M. (ed.). 1993a. Second report on the Oak Ridge National Laboratory Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River. ORNL/TM-10804. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

- Loar, J. M. (ed.). 1993b. Third annual report on the ORNL Biological Monitoring and Abatement Program. ORNL/TM-11358. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Loar, J. M. (ed.). 1993c. Second report on the Oak Ridge Y-12 Plant Biological Monitoring and Abatement Program for East Fork Poplar Creek. Y/TS-888. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Loar, J. M., L. M. Adams, S. M. Adams, B. G. Blaylock, H. L. Boston, M. A. Huston, B. L. Kimmel, C. R. Olsen, J. G. Smith, G. R. Southworth, A. J. Stewart, and B. T. Walton. 1991. Oak Ridge National Laboratory biological monitoring plan and abatement program for White Oak Creek watershed and the Clinch River. ORNL/TM-10370. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Loar, J. M., S. M. Adams, B. G. Blaylock, H. L. Boston, M. A. Huston, B. L. Kimmel, J. T. Kitchings, C. R. Olsen, M. G. Ryon, J. G. Smith, G. R. Southworth, A. J. Stewart, B. T. Walton, H. Amano, C. T. Garten, Jr. and L. J. Meyers. 1992. First annual report on the Biological Monitoring and Abatement Program at Oak Ridge National Laboratory. ORNL/TM-10399. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Loar, J. M., S. M. Adams, L. J. Allison, J. M. Giddings, J. F. McCarthy, G. R. Southworth, J. G. Smith, and A. J. Stewart. 1989. The Oak Ridge Y-12 Plant Biological Monitoring and Abatement Program for East Fork Poplar Creek. ORNL/TM-10265. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Loar, J. M., J. A. Solomon, and G. F. Cada. 1981a. Technical background information for the ORNL Environmental and Safety Report, Vol. 2: A description of the aquatic ecology of White Creek watershed and the Clinch River below Melton Hill Dam. ORNL/TM-7509/V2. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Loar, J. M., F. A. Burkhart, G. F. Cada, J. W. Huckabee, J. T. Kitchings, K. D. Kumar, A. M. Sasson, J. A. Solomon, and J. D. Story. 1981b. Ecological studies in the vicinity of the Oak Ridge Gaseous Diffusion Plant. ORNL/TM-6714. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Lockhart, W. L., and D. A. Metner. 1984. Fish serum chemistry as a pathology tool. pp. 73-85. IN: V. W. Cairns, P. V. Hodson, and J. O. Nriagu, (eds.), Contaminant Effects on Fisheries. John Wiley & Sons, New York.
- Lowe, T. P., T. W. May, W. G. Brumbaugh, and D. A. Kane. 1985. National Contaminant Biomonitoring Program: Concentrations of seven elements in freshwater fish, 1978-1981. Arch. Environ. Contam. Toxicol. 14:363-388.
- Lowery, J. F., P. H. Counts, H. L. Edmiston, and F. D. Edwards. 1986. Water resources data for Tennessee, Water Year 1985. Report No. USGS/WRD/HD-86. U.S. Geological Survey, Nashville, Tennessee.
- Lowery, J. F., P. H. Counts, H. L. Edmiston, and F. D. Edwards. 1987. Water resources data for Tennessee, Water Year 1986. Report No. USGS/WRD/HD-87/225. U.S. Geological Survey, Nashville, Tennessee.
- Lowery, J. F., P. H. Counts, H. L. Edmiston, and F. D. Edwards. 1988. Water resources data for Tennessee, Water Year 1987. Report No. USGS/WRD/HD-88/236. U.S. Geological Survey, Nashville, Tennessee.
- Lowery, J. F., P. H. Counts, F. D. Edwards, and J. W. Garrett. 1989. Water resources data for Tennessee, Water Year 1988. Report No. USGS/WRD/HD-89/258. U.S. Geological Survey, Nashville, Tennessee.

- Mason, C. F., and R. J. Bryant. 1975. Production, nutrient content and decomposition of *Phragmites communis* Trin. and *Typha augustifolia* L. J. Ecol. 63:71-95.
- McElravy, E. P., G. A. Lamberti, and V. H. Resh. 1989. Year-to-year variation in the aquatic macroinvertebrate fauna of a northern California stream. J. N. Amer. Benthol. Soc. 8:51-63.
- McKee, M. J., A. C. Hendricks and R. E. Ebel. 1983. Effects of naphthalene on benzo[a]pyrene hydroxylase and cytochrome P-450 in *Fundulus heteroclitus*. Aquat. Toxicol. 3:103-114.
- McMaster, W. M. 1963. Geologic map of the Oak Ridge Reservation, Tennessee. ORNL/TM-713. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- McMaster, W. M. 1967. Hydrologic data for the Oak Ridge area, Tennessee. U.S. Geological Survey-Water Supply Paper No. 1839-N. U.S. Government Printing Office, Washington, D.C.
- McMaster, W. M., and H. D. Waller. 1965. Geology and soils of White Oak Creek basin, Tennessee. ORNL/TM-1108. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Meyers-Schöne, L., and B. T. Walton. 1990. Comparison of two freshwater turtle species as monitors of environmental contamination. ORNL/TM-11460. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Meyers-Schöne, L., L. R. Shugart and B. T. Walton. 1989. Comparison of two turtle species as indicators of contamination in freshwater ecosystems. Paper presented in symposium on "In Situ Monitoring at Hazardous Waste Sites." Abstracts of the Annual Meeting of the Society of Environmental Toxicology and Chemistry, Toronto, Ontario, Canada.
- Miller, D. L., P. M. Leonard, R. M. Hughes, J. R. Karr, P. B. Moyle, L. H. Schrader, B. A. Thompson, R. A. Daniels, K. D. Fausch, G. A. Fitzhugh, J. R. Gammon, D. B. Halliwell, P. L. Angermeier, and D. J. Orth. 1988. Regional application of an index of biotic integrity for use in water resource management. Fisheries 13(5):12-20.
- Minshall, G. W. 1978. Autotrophy in stream ecosystems. Bioscience 28:767-771.
- Mount, D. I., and T. J. Norberg. 1984. A seven-day life cycle cladoceran toxicity test. Environ. Toxicol. Chem. 3:425-434.
- Mount, D. I., N. A. Thomas, T. J. Norberg, M. T. Barbour, T. H. Roush, and W. F. Brandes. 1984. Effluent and ambient toxicity testing and instream community response on the Ottawa River, Lima, Ohio. EPA/600/03. Environmental Protection Agency, Duluth, Minnesota.
- NCCLS (National Committee for Clinical Laboratory Standards). 1979. NCCLS approved standards: ACS-1 Specification for standard protein solution (bovine serum albumin), 2nd ed. National Committee for Clinical Laboratory Standards, Villanova, Pennsylvania.
- Neff, J. M. 1978. Polycyclic aromatic hydrocarbons. Applied Science Publisher, Ltd., London.
- Nimmo, D. R., M. H. Dodson, P. H. Davies, J. C. Greene, and M. A. Kerr. 1990. Three studies using *Ceriodaphnia* to detect nonpoint sources of metals from mine drainage. J. Water Poll. Contr. Fed. 62:7-15.
- Norberg, T. J., and D. I. Mount. 1985. A new subchronic fathead minnow (*Pimephales promelas*) toxicity test. Environ. Toxicol. Chem. 4:711-718.

- Oakes, T. W., B. A. Kelly, W. F. Ohnesorge, J. S. Eldridge, J. C. Bird, K. E. Shanks, and F. S. Tsakeres. 1982. Technical background information for the ORNL Environmental and Safety Report, Vol. 4: White Oak Lake and Dam. ORNL-5681. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Ohio EPA (Ohio Environmental Protection Agency). 1987. Biological criteria for the protection of aquatic life, Volume III: Standardized biological field sampling and laboratory methods for assessing fish and macroinvertebrate communities. Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio.
- Ohio EPA (Ohio Environmental Protection Agency). 1988. Biological criteria for the protection of aquatic life, Volume II: Users manual for biological field assessment of Ohio surface streams. Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio.
- Palumbo, A. V., P. J. Mulholland, and J. W. Elwood. 1987. Simultaneous measurement of periphyton production, chlorophyll and ATP using DMSO extraction. *Limnol. Oceanogr.* 32:464-471.
- Payne J. F., and W. R. Penrose. 1975. Induction of aryl hydrocarbon benzo(a)pyrene hydroxylase in fish by petroleum. *Bull. Environ. Contam. Toxicol.* 14: 112-116.
- Pielou, E. C. 1977. *Mathematical ecology*. John Wiley and Sons, New York.
- Platts, W. S., W. F. Meghan, and G. W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. Gen. Tech. Rep. INT-138. U.S. Department of Agriculture, Ogden, Utah.
- Power, M. E., W. J. Matthews, and A. J. Stewart. 1985. Grazing minnows, piscivorous bass, and stream algae: Dynamics of a strong interaction. *Ecology* 66:1448-1456.
- Railsback, S. F., B. D. Holcomb, and M. G. Ryon. 1989. A computer program for estimating fish population sizes and annual production rates. ORNL/TM-11061. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Ricker, W. E. 1975. Chapter 9: Growth in length and weight. pp. 203-233. IN: *Computation and Interpretation of Biological Statistics of Fish Populations*. Bulletin 191. Department of the Environment, Fisheries and Marine Service, Ottawa, Canada.
- Robison, H. W., and T. M. Buchanan. 1988. *Fishes of Arkansas*. The University of Arkansas Press, Fayetteville, Arkansas.
- Roche Diagnostic Systems. 1986. Reagent for total protein. Roche Diagnostic Systems Information Package, Items 44903 and 44313. Nutley, New Jersey.
- Roche, H., J. Jouanneteau, and G. Peres. 1983. Effects of adaptation to different salinities on the lipids of various tissues in sea dace (*Dicentrarchus labrax*). *Comp. Biochem. Physiol.* 74B:325-330.
- Rogers, J. G., K. L. Daniels, S. T. Goodpasture, C. W. Kimbrough, and N. L. Prince. 1989a. Oak Ridge Reservation Environmental Report for 1988, Vol. 1: Narrative, summary and conclusions. ES/ESH-8/V1. Environmental and Safety Activities, Martin Marietta Energy Systems, Inc., Oak Ridge, Tennessee.
- Rogers, J. G., K. L. Daniels, S. T. Goodpasture, C. W. Kimbrough, and N. L. Prince. 1989b. Oak Ridge Reservation Environmental Report for 1988, Vol. 2: Data presentation. ESH-8/V2. Environmental and Safety Activities, Martin Marietta Energy Systems, Inc., Oak Ridge, Tennessee.
- Rosenberg, R. 1976. Benthic faunal dynamics during succession following pollution abatement in a Swedish estuary. *Oikos* 27:414-427.

- Ryon, M. G., and J. M. Loar. 1988. A checklist of fishes on the Department of Energy Oak Ridge Reservation. *J. Tenn. Acad. Sci.* 58(4):97-102.
- SAS (SAS Institute, Inc.). 1985a. SAS User's Guide: Basics, Version 5 Edition. Sas Institute, Inc., Cary, North Carolina.
- SAS (SAS Institute, Inc.). 1985b. SAS User's Guide: Statistics, Version 5 Edition. SAS Institute, Inc., Cary, North Carolina.
- Schramm, H. L., Jr. 1989. Formation of annuli in otoliths of bluegills. *Trans. Amer. Fish. Soc.* 118:546-555.
- Schramm, H. L., Jr., and J. F. Doerzbacher. 1982. Use of otoliths to age black crappie from Florida. *Proc. 36th Ann. Conf. SE. Assoc. Fish Wildl. Agen.* 36:95-105.
- Selby, D. A., J. M. Inhat, and J. J. Messer. 1985. Effects of subacute cadmium exposure on a hardwater mountain stream microcosm. *Water Res.* 19:645-655.
- Sherwood, C. B., and J. M. Loar. 1987. Environmental data for the White Oak Creek/White Oak Lake watershed. ORNL/TM-10062. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Shoaf, W. T., and B. W. Lium. 1976. Improved extraction of chlorophyll *a* and *b* from algae using dimethyl sulfoxide. *Limnol. Oceanogr.* 21:926-928.
- Shul'man, G. E. 1974. Life cycles of fish. Physiology and biochemistry. John Wiley and Sons, New York.
- Smith, P. W. 1979. The fishes of Illinois. University of Illinois Press, Urbana, Illinois.
- Smith, J. G., S. M. Adams, L. A. Kszos, J. M. Loar, M. G. Ryon, and G. R. Southworth. 1988. First report on the Oak Ridge K-25 Site Biological Monitoring and Abatement Program for Mitchell Branch. ORNL/TM-11073. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Smith, J. G. 1992a. Benthic macroinvertebrates. IN: J. M. Loar, (ed.), First report on the Oak Ridge Y-12 Plant Biological Monitoring and Abatement Program for East Fork Poplar Creek. YTS-886. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Smith, J. G. 1992b. Benthic macroinvertebrates. IN: G. R. Southworth (ed.), Ecological effects of contaminants and remedial actions in Bear Creek. ORNL/TM-11977, Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Smith, J. G. 1993a. Benthic macroinvertebrates. IN: J. M. Loar (ed.), Second report on the Oak Ridge National Laboratory Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River. ORNL/TM-10804. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Smith, J. G. 1993b. Benthic macroinvertebrates. IN: J. M. Loar, (ed.), Third report on the Oak Ridge National Laboratory Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River. ORNL/TM-11358. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Sokal, R. R., and F. J. Rohlf. 1981. Biometry. W. H. Freeman and Company, San Francisco.
- Southworth, G. R. 1992. Ecological effects of contaminants and remedial actions in Bear Creek. ORNL/TM-11977. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Spalding, B. P., and T. E. Cerling. 1979. Association of radionuclides with stream bed sediments in White Oak Creek watershed. ORNL/TM-6895. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Steel, R. G. D., and J. H. Torrie. 1960. Principles and procedures of statistics with special reference to the biological sciences. McGraw-Hill, Inc. New York.

- Stegeman, J. J., R. L. Binder, and A. Orren. 1979. Hepatic and extrahepatic microsomal electron transport components and mixed function oxygenases in the marine fish *Stenotomus versicolor*. *Biochem. Pharmacol.* 28:3431-3439.
- Stewart, A. J. 1993. Point-source and area-source contributions to ambient toxicity. pp. 38-72. IN: J. M. Loar, (ed.), Second report on the Oak Ridge National Laboratory Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River. ORNL/TM-10804. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Stewart, A. J., L. A. Kszos, B. C. Harvey, L. F. Wicker, G. J. Haynes, and R. D. Bailey. 1990. Ambient toxicity dynamics: Assessments using *Ceriodaphnia dubia* and fathead minnow (*Pimephales promelas*) larvae in short-term tests. *Environ. Toxicol. Chem.* 9:367-379.
- Strickland, J. D. H., and T. R. Parsons. 1972. A practical handbook of seawater analysis. Fisheries Research Board of Canada, Ottawa.
- Stueber, A. M., D. A. Webster, I. L. Munro, N. D. Farrow, and T. G. Scott. 1981. An investigation of radionuclide release from Solid Waste Disposal Area 3, Oak Ridge National Laboratory. ORNL/TM-7323. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Talmage, S. S., and B. T. Walton. 1989. Comparative evaluation of small mammals as monitors of environmental toxicants. *Toxicologist* 9:215.
- Talmage, S. S., and B. T. Walton. 1990. Comparative evaluation of several small mammal species as monitors of heavy metals, radionuclides, and selected organic compounds in the environment. ORNL/TM-11605. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Talmage, S. S., and B. T. Walton. 1991. Small mammals as monitors of environmental contaminants. *Rev. Environ. Contamin. Toxicol.* 119:47-145.
- Taubert, B. D., and J. A. Tranquilli. 1982. Verification of the formation of annuli in otoliths of largemouth bass. *Trans. Amer. Fish. Soc.* 11:531-534.
- Taylor, F. G., Jr. 1989. Mercury assessment for water and sediment in Oak Ridge National Laboratory streams. ORNL/M-73. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Taylor, F. G., Jr. 1990a. Polychlorinated biphenyls of the aquatic environment of Oak Ridge National Laboratory—Report of 1989. ORNL/M-1041. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Taylor, F. G., Jr. 1990b. Mercury monitoring of water and sediment in Oak Ridge National Laboratory streams in 1989. ORNL/M-1030. Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- TVA (Tennessee Valley Authority). 1985. Instream Contaminant Study, Task 4: Fish sampling and analysis. Report to U.S. Department of Energy, Oak Ridge Operations Office. Tennessee Valley Authority, Office of Natural Resources and Economic Development, Knoxville, Tennessee.
- TVA (Tennessee Valley Authority). 1986. Instream Contaminant Study, Task 5: Summary Report. Report to U.S. Department of Energy, Oak Ridge Operations Office. Tennessee Valley Authority, Office of Natural Resources and Economic Development, Knoxville, Tennessee.
- Tietz, N. W. 1986. Textbook of clinical chemistry. W. B. Saunders, Co., Philadelphia, Pennsylvania.

- Trautman, M. B. 1981. The fishes of Ohio. Ohio State University Press, Columbus, Ohio.
- Travis, C. C., F. O. Hoffman, B. G. Blaylock, K. L. Daniel, C. S. Gist, and C. W. Weber. 1986. Preliminary review of TVA fish sampling and analysis report. Report of Task Group Five to Oak Ridge Task Force. Mimeo.
- Turner, R. R., M. A. Bogle, R. B. Clapp, K. Dearstone, R. B. Dreier, T. O. Early, S. E. Herbes, J. M. Loar, P. D. Parr, G. R. Southworth, and T. M. Mercier. 1988. RCRA Facility Investigation Plan for Bear Creek, Oak Ridge Y-12 Plant. Y/TS-417. Oak Ridge Y-12 Plant, Oak Ridge, Tennessee.
- Volchok, H. L., and G. A. Planque (eds.). 1982. SR-01: Radiochemical strontium-90. pp. E-SR-01-01-E-SR-01-29. IN: EML Procedure Manual (HASL-300), 26th Edition. Environmental Measurements Laboratory, U.S. Department of Energy, New York.
- Walton, B. T., and S. S. Talmage. 1989. Small mammals as indicators of radionuclides, heavy metals, and persistent organic chemicals at hazardous waste sites. Presented in symposium on "In Situ Monitoring at Hazardous Waste Sites." Abstracts of the Annual Meeting of the Society of Environmental Toxicology and Chemistry, Toronto, Ontario, Canada.
- Warren-Hicks, W., B. R. Parkhurst, and S. S. Baker, Jr. 1989. Ecological assessments of hazardous waste sites: A field and laboratory reference. OREPA/600/3-89/013. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.
- Waters, T. F. 1977. Secondary production in inland waters. Adv. Ecol. Res. 10:91-164.
- Weber, C. I. (ed.). 1973. Biological field and laboratory methods for measuring the quality of surface waters and effluents. EPA 670/4-73-001. National Environmental Research Center, U.S. Environmental Protection Agency, Cincinnati, Ohio.
- Wetzel, R. G., and B. A. Manny. 1972. Decomposition of dissolved organic carbon and nitrogen compounds from leaves in an experimental hardwater stream. Limnol. Oceanogr. 17:927-931.
- Wiederholm, T. 1984. Responses of aquatic insects to environmental pollution. pp. 508-557. IN: V. H. Resh and D. M. Rosenberg (eds.), The Ecology of Aquatic Insects. Praeger Publishers, New York.
- Williams, T., and B. C. Bedford. 1974. The use of otoliths for age determination. pp. 114-123. IN: T. B. Bagenal (ed.), The aging of fish. Unwin Brothers Ltd., Old Working, England.
- Winner, R. W., M. W. Boesel, and M. P. Farrell. 1980. Insect community structure as an index of heavy-metal pollution in lotic ecosystems. Can. J. Fish. Aquat. Sci. 37:647-655.
- Winter, H. 1961. The uptake of cations by *Vallisneria* leaves. Acta Bot. Neerl. 10:341-393.
- Zar, J. H. 1984. Biostatistical analysis. Prentice-Hall, Inc., Englewood Cliffs, New Jersey.

Appendix A

**RESULTS OF QUALITY ASSURANCE/QUALITY CONTROL
ANALYSES OF MERCURY, POLYCHLORINATED
BIPHENYLS, AND ORGANICS
IN FISH SAMPLES**

Appendix A

RESULTS OF QUALITY ASSURANCE/QUALITY CONTROL ANALYSES OF MERCURY, POLYCHLORINATED BIPHENYLS, AND ORGANICS IN FISH SAMPLES

A.1 MERCURY

Fifteen pairs of blind duplicate samples of fish muscle tissue were analyzed for mercury and showed a relatively low degree of variation; the mean CV between sample pairs was 8% with a mean SD of $0.01 \mu\text{g/g}$. The mean absolute difference between duplicate samples was $0.02 \mu\text{g/g}$. The multiple analyses of mercury in EPA reference fish ($n = 9$) agreed well with the expected value, averaging $2.49 \pm 0.13 \mu\text{g/g}$ (mean \pm SD) vs an expected value of $2.52 \mu\text{g/g}$; the average recovery was $99 \pm 5\%$. Mean values for replicate split-fish samples ($n = 5$) analyzed for mercury by the ORNL ACD, the U.S. EPA Environmental Services Laboratory in Athens, Georgia, and the Y-12 Plant Analytical Laboratory were similar. Fish analyzed by ORNL averaged $0.76 \mu\text{g/g}$ mercury, while those analyzed by the EPA and Y-12 Plant labs averaged 0.76 and $0.80 \mu\text{g/g}$ respectively. The mean differences between individual samples analyzed by EPA, ORNL, and the Y-12 Plant were not significantly different from zero [t-test on mean difference between sample pairs ($p > 0.05$)]. However, the mean absolute difference = 0.23 , mean CV = 21%, and mean SD = 0.17 between individual samples analyzed by EPA and ORNL indicate much larger variation between mercury values from these two labs than that observed for duplicate analyses within the ORNL lab. The variation between individual samples

analyzed by ORNL and the Y-12 Plant was similar to the ORNL and EPA comparison; the mean absolute difference = 0.23 , CV = 21%, and SD = 0.16 were larger than values observed for duplicate analyses within the ORNL laboratory. Mercury levels in sunfish from the uncontaminated reference site (Hinds Creek) were typical of background levels in stream fish, averaging $0.08 \pm 0.04 \mu\text{g/g}$ ($n = 16$).

A.2 OTHER METALS

Uncontaminated fish were spiked with a known concentration of each metal to ensure that these metals would be recovered and quantified. The recovery of As, Be, Cd, Ni, Sb, and Se ranged from 92 to 100%. Slightly smaller percentages of Ag, Cr, and Cu were recovered (Range = 84–88%), and 120% of the added concentration of Pb was recovered.

A.3 POLYCHLORINATED BIPHENYLS

The results of PCB analyses of 21 pairs of blind duplicate fish samples were somewhat more variable than results of mercury analyses, as is generally the case. The mean SD between duplicates was $0.12 \mu\text{g/g}$, with a mean CV of 32%. The variabilities in the measurement of PCB-1254 and PCB-1260 were similar, with mean SDs between duplicates of 0.09 and

0.06 $\mu\text{g/g}$ and mean CVs of 40% and 32% respectively. Samples of uncontaminated fish and clams were spiked at 1 $\mu\text{g/g}$ each of PCB-1254 and PCB-1260 and analyzed along with fish or clam samples. Mean recoveries ($\pm\text{SD}$) averaged $98\% \pm 15\%$ for total PCBs, and $93 \pm 11\%$ and $96 \pm 11\%$ ($n = 16$) for PCB-1254 and PCB-1260 respectively.

Samples of PCB-contaminated carp were homogenized and split for analysis by the ORNL/ACD laboratory and the EPA Environmental Services Laboratory, Athens, Georgia. Mean levels of PCB-1254 did not differ significantly in the ten samples analyzed by the two laboratories (t-test on mean difference between sample pairs, $p > 0.05$), but the results for PCB-1260 and total PCBs obtained by the EPA lab were significantly higher than those from the ORNL lab. Results averaged ($\pm\text{SE}$) 1.41 ± 0.38 vs 0.88 ± 0.27 , 0.60 ± 0.26 vs 0.38 ± 0.17 , and 0.81 ± 0.14 vs 0.50 ± 0.12 $\mu\text{g/g}$ for total PCBs, PCB-1254, and PCB-1260 respectively. The variability between duplicate samples analyzed by ORNL and EPA was larger than the variability between duplicates analyzed at ORNL, which had a mean SD between duplicates of 0.37, 0.19, and 0.22 $\mu\text{g/g}$ and a mean CV of 33%, 59%, and 38% for total PCBs, PCB-1254, and PCB-1260 respectively. Samples of catfish collected in 1987 from an uncontaminated site (commercial fish farm) were used as analytical controls; these exhibited very low levels of total PCBs, averaging 0.01 ± 0.01 ppm, ($n = 5$ samples from four fish).

Fish samples were also split with TDHE and TVA laboratories for comparison of PCB analyses, since data from all three laboratories are used by the TDHE in issuing fish consumption advisories. Differences between samples analyzed by TVA and ORNL were not

significant, with ORNL averaging 1.13 ± 0.34 and TVA 1.25 ± 0.39 $\mu\text{g/g}$ total PCBs ($n = 9$). The TDHE results were significantly lower, but technical problems were noted for that group of analyses.

Ten fish samples were split with TDHE and TVA laboratories for chlordane analyses. Results of these comparisons indicated significant differences between mean ($\pm\text{SE}$) chlordane concentrations measured by ORNL ($0.034 \pm .012$ $\mu\text{g/g}$) and TVA (0.19 ± 0.05 $\mu\text{g/g}$). TDHE results were similar to ORNL's (0.021 ± 0.007), but as noted for PCBs, technical problems at the TDHE Lab for this set of samples indicates that they should not be considered. ORNL chlordane results quantified only α and γ isomers; thus TVA's results (based on a constituents of technical chlordane) should be somewhat higher.

A.4 ORGANICS SCREENING ANALYSES

Uncontaminated fish samples were spiked with a mixture of priority pollutants (PCB-1254, PCB-1260, di-*N*-butylphthalate, 2-ethylhexylphthalate, pyrene, and benzo[*a*]pyrene) and analyzed to ensure that these contaminants would be recovered and quantified in the extraction, cleanup, and gas chromatographic analysis. In the GC/MS analysis, the recovery of di-*N*-butylphthalate and 2-ethylhexylphthalate averaged $102 \pm 7\%$ and $117 \pm 4\%$, respectively, and that of pyrene and benzo[*a*]pyrene averaged $93 \pm 1\%$ and $108 \pm 7\%$, respectively (mean \pm SD, $n = 2$ for each substance). Recoveries of PAHs were not as good in the HPLC procedure, averaging $55 \pm 4\%$ for pyrene and $61 \pm 4\%$ for benzo[*a*]pyrene.

Appendix B

CONCENTRATIONS OF CONTAMINANTS IN AQUATIC BIOTA
FROM WHITE OAK CREEK AND TRIBUTARIES,
WHITE OAK LAKE, AND THE CLINCH RIVER
NOVEMBER 1988–AUGUST 1989

Table B.1. Metals in sunfish from White Oak Creek and tributaries, White Oak Lake, nearby reaches of the Clinch River, and several reference sites^a

Site ^b	Dist ^c	Date	Spp ^d	Sex ^e	No. ^f	Weight (g)	Length (cm)	Hg ($\mu\text{g/g}$)	Cu ($\mu\text{g/g}$)	Se ($\mu\text{g/g}$)	Zn ($\mu\text{g/g}$)
CRK 31.5	31.5	11/21/88	Blugil	F	7810	83.0	17.1	0.08	-	-	-
CRK 31.5	31.5	11/21/88	Blugil	M	7811	122.0	18.7	0.02	-	-	-
CRK 31.5	31.5	11/21/88	Blugil	M	7812	54.4	14.5	0.02	-	-	-
CRK 31.5	31.5	11/21/88	Blugil	F	7813	61.9	15.2	0.02	-	-	-
CRK 31.5	31.5	11/21/88	Blugil	M	7814	49.5	14.2	0.03	-	-	-
CRK 31.5	31.5	11/21/88	Blugil	M	7815	90.3	17.4	0.02	-	-	-
CRK 31.5	31.5	11/21/88	Blugil	M	7816	153.1	20.0	0.08	-	-	-
CRK 31.5	31.5	11/21/88	Blugil	F	7817	62.1	15.3	0.03	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	M	7800	93.2	16.5	0.01	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	F	7801	50.8	14.6	0.03	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	M	7802	77.2	15.4	0.01	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	M	7803	69.0	15.5	0.01	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	M	7804	45.6	14.0	0.05	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	M	7805	51.1	14.5	0.02	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	M	7806	47.1	14.2	0.01	-	-	-
CRK 35.2	35.2	11/21/88	Blugil	F	7807	88.8	16.8	0.05	-	-	-
MHR	38	11/23/88	Blugil	M	7880	152.6	20.6	0.03	-	-	-
MHR	38	11/23/88	Blugil	F	7881	86.8	17.5	0.08	-	-	-
MHR	38	11/23/88	Blugil	F	7882	97.3	18.2	0.03	-	-	-
MHR	38	11/23/88	Blugil	M	7883	59.1	15.5	0.04	-	-	-
MHR	38	11/23/88	Blugil	M	7884	81.1	17.0	0.03	-	-	-
MHR	38	11/23/88	Blugil	M	7885	79.8	17.4	0.03	-	-	-
MHR	38	11/23/88	Blugil	F	7886	73.7	16.8	0.06	-	-	-
MHR	38	11/23/88	Blugil	M	7887	53.9	15.0	0.04	-	-	-
WCK 3.5	3.5	12/14/88	Blugil	F	7310	91.6	16.9	0.31	-	-	-
WCK 3.5	3.5	12/14/88	Blugil	M	7311	47.9	14.1	0.33	-	-	-
WCK 3.5	3.5	12/19/88	Blugil	F	7346	188.7	20.0	0.40	-	-	-
WCK 3.5	3.5	12/19/88	Blugil	M	7347	101.5	16.5	0.34	-	-	-
WCK 3.5	3.5	12/19/88	Blugil	M	7348	80.0	15.0	0.25	-	-	-
WCK 3.5	3.5	12/19/88	Blugil	F	7349	65.1	14.5	0.16	-	-	-
WCK 3.5	3.5	12/19/88	Blugil	F	7350	82.4	15.5	0.32	-	-	-
WCK 3.5	3.5	12/19/88	Blugil	F	7351	66.8	15.0	0.33	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	M	7846	68.9	16.6	0.12	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	F	7847	72.1	16.2	0.25	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	F	7858	85.7	17.2	0.37	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	M	7859	47.1	14.2	0.18	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	F	7870	53.2	15.0	0.22	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	M	7871	44.1	14.1	0.15	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	M	7305	92.7	18.3	0.18	-	-	-
NTK 0.2	0.2	12/13/88	Blugil	M	7306	38.1	14.0	0.04	-	-	-

Table B.1 (continued)

Site ^b	Dist ^c	Date	Spp ^d	Sex ^e	No. ^f	Weight (g)	Length (cm)	Hg ($\mu\text{g/g}$)	Cu ($\mu\text{g/g}$)	Se ($\mu\text{g/g}$)	Zn ($\mu\text{g/g}$)
WCK 2.9	2.9	12/19/88	Blugil	M	7358	108.9	17.6	0.49	-	-	-
WCK 2.9	2.9	12/19/88	Blugil	M	7359	157.8	19.6	0.35	<0.50	<0.50	5.2
WCK 2.9	2.9	12/19/88	Blugil	M	7360	189.3	21.7	0.31	<0.50	0.56	5.5
WCK 2.9	2.9	12/19/88	Blugil	M	7361	217.8	21.6	0.47	<0.50	0.48	5.5
WCK 2.9	2.9	12/19/88	Blugil	M	7362	102.2	19.0	0.35	<0.50	0.50	4.5
WCK 2.9	2.9	12/19/88	Blugil	F	7363	132.5	19.3	0.62	-	-	-
WCK 2.9	2.9	12/19/88	Blugil	M	7364	89.4	16.9	0.40	-	-	-
WCK 2.9	2.9	12/19/88	Blugil	M	7365	98.0	16.2	0.30	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	M	7340	107.9	18.0	0.29	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	M	7341	120.3	18.5	0.19	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	M	7342	102.9	17.0	0.30	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	M	7343	105.9	18.0	0.56	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	M	7344	94.9	17.0	0.54	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	F	7345	74.0	16.0	0.62	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	F	7366	66.0	15.1	0.57	-	-	-
WCK 2.9	2.9	12/19/88	Redbre	F	7367	72.0	16.7	0.69	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	M	7876	63.9	15.4	0.26	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	M	7864	96.4	16.9	0.23	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	M	7866	79.8	16.6	0.09	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	M	7869	46.8	13.9	0.20	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	M	7865	102.0	19.0	0.49	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	F	7877	40.5	13.1	0.12	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	M	7867	45.3	13.7	0.29	-	-	-
WCK 2.3	2.3	12/13/88	Blugil	F	7303	42.8	13.5	0.09	-	-	-
MEK 0.2	0.2	12/06/88	Redbre	M	7890	94.0	17.3	-	<0.50	<0.50	5.3
MEK 0.2	0.2	12/06/88	Redbre	F	7893	92.6	17.6	-	<0.50	<0.50	5.4
MEK 0.2	0.2	12/06/88	Redbre	M	7894	78.4	17.4	-	<0.50	<0.50	7.7
MEK 0.2	0.2	12/06/88	Redbre	M	7895	52.7	14.5	-	<0.50	<0.50	6.8
MEK 0.2	0.2	12/06/88	Blugil	F	7891	65.0	15.5	0.08	-	-	-
MEK 0.2	0.2	12/06/88	Blugil	M	7892	59.2	15.2	0.05	-	-	-
MEK 0.2	0.2	12/06/88	Blugil	M	7899	39.1	13.3	0.11	-	-	-
MEK 0.2	0.2	12/06/88	Blugil	M	7896	39.6	13.4	0.15	-	-	-
MEK 0.2	0.2	12/06/88	Blugil	M	7897	40.8	13.9	0.05	-	-	-
MEK 0.2	0.2	12/06/88	Blugil	M	7888	30.9	13.0	0.16	-	-	-
MEK 0.2	0.2	12/06/88	Blugil	M	7899	21.0	11.0	0.07	-	-	-
WOL	1.0	12/14/88	Blugil	M	7314	80.6	16.5	-	<0.50	<0.50	6.4
WOL	1.0	12/14/88	Blugil	M	7315	66.3	15.3	-	<0.50	<0.50	6.0
WOL	1.0	12/14/88	Blugil	M	7312	78.0	16.3	0.12	-	-	-
WOL	1.0	12/14/88	Blugil	M	7313	89.9	17.4	0.14	-	-	-
WOL	1.0	12/14/88	Blugil	M	7316	96.6	16.5	0.08	0.60	0.56	9.2

Table B.1 (continued)

Site ^b	Dist ^c	Date	Spp ^d	Sex ^e	No. ^f	Weight (g)	Length (cm)	Hg (µg/g)	Cu (µg/g)	Se (µg/g)	Zn (µg/g)
WOL	1.0	12/14/88	Blugil	F	7318	119.8	17.6	0.07	0.46	0.47	5.5
WOL	1.0	12/14/88	Blugil	M	7319	86.7	16.4	0.14	-	-	-
WOL	1.0	12/14/88	Blugil	M	7320	82.0	17.0	0.31	-	-	-
WOL	1.0	12/14/88	Blugil	M	7321	83.8	16.5	0.12	-	-	-
WOL	1.0	12/14/88	Blugil	F	7322	79.8	16.3	0.15	-	-	-
WOL	1.0	12/14/88	Cocarp	M	7332	2446	57.0	0.10	-	-	-
WOL	1.0	12/14/88	Cocarp	M	7333	2460	57.0	0.15	-	-	-
WOL	1.0	12/14/88	Cocarp	F	7334	1920	51.5	0.13	-	-	-
WOL	1.0	12/14/88	Cocarp	M	7335	1586	49.2	0.13	-	-	-
WOL	1.0	12/14/88	Cocarp	F	7336	3221	65.4	0.08	-	-	-
WOL	1.0	12/14/88	Cocarp	M	7337	2890	59.8	0.10	-	-	-
WOL	1.0	12/14/88	Cocarp	M	7338	2021	54.2	0.08	-	-	-
WOL	1.0	12/14/88	Cocarp	M	7339	2021	57.8	0.08	-	-	-
WOL	1.0	12/14/88	Lmbass	F	7324	1468	44.0	0.15	-	-	-
WOL	1.0	12/14/88	Lmbass	M	7325	388	31.2	0.08	-	-	-
WOL	1.0	12/14/88	Lmbass	F	7326	515	32.4	0.12	-	-	-
WOL	1.0	12/14/88	Lmbass	M	7327	362	29.0	0.06	-	-	-
WOL	1.0	12/14/88	Lmbass	M	7328	1155	41.7	0.16	-	-	-
WOL	1.0	12/14/88	Lmbass	M	7329	1175	40.8	0.15	-	-	-
WOL	1.0	12/14/88	Lmbass	M	7330	1019	40.0	0.12	-	-	-
WOL	1.0	12/14/88	Lmbass	M	7331	836	36.8	0.07	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	F	7352	150.3	19.0	0.12	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	M	7353	154.9	19.5	0.09	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	M	7354	148.3	18.5	0.09	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	M	7355	107.4	16.0	0.09	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	M	7356	128.6	18.0	0.10	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	M	7357	91.0	16.0	0.10	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	M	7368	75.0	15.6	0.10	-	-	-
WCK 0.9	0.9	12/19/88	Blugil	F	7369	64.6	14.6	0.03	-	-	-
HINDSCR	-	12/05/88	Blugil	F	7844	53.2	15.2	0.12	-	-	-
HINDSCR	-	12/05/88	Blugil	F	7818	56.2	15.4	0.12	-	-	-
HINDSCR	-	12/05/88	Blugil	F	7808	57.7	15.2	0.13	-	-	-
HINDSCR	-	12/05/88	Blugil	M	7809	134.4	19.7	0.13	-	-	-
HINDSCR	-	12/05/88	Blugil	M	7848	154.5	20.3	0.07	-	-	-
HINDSCR	-	12/05/88	Blugil	F	7875	71.9	16.2	0.08	-	-	-
HINDSCR	-	12/05/88	Blugil	F	7873	69.7	15.7	0.12	-	-	-
HINDSCR	-	12/05/88	Blugil	F	7874	55.7	15.1	0.06	-	-	-
HINDSCR	-	12/05/88	Redbre	F	7878	39.4	13.8	0.06	-	-	-
HINDSCR	-	12/05/88	Redbre	M	7843	52.9	14.2	0.05	-	-	-
HINDSCR	-	12/05/88	Redbre	M	7849	73.7	16.3	0.07	-	-	-

Table B.1 (continued)

Site ^b	Dist ^c	Date	Spp ^d	Sex ^e	No. ^f	Weight (g)	Length (cm)	Hg (μg/g)	Cu (μg/g)	Se (μg/g)	Zn (μg/g)
HINDSCR	-	12/05/88	Redbre	M	7842	78.5	16.9	0.04	-	-	-
HINDSCR	-	12/05/88	Redbre	M	7841	53.5	15.3	0.04	-	-	-
HINDSCR	-	12/05/88	Redbre	M	7872	55.5	15.2	0.05	-	-	-
HINDSCR	-	12/05/88	Redbre	F	7840	52.4	15.4	0.14	-	-	-
HINDSCR	-	12/05/88	Redbre	M	7845	55.2	15.2	0.07	-	-	-
HINDSCR	-				7404				<0.50	<0.50	4.8
HINDSCR	-				7406				<0.50	<0.50	6.4
PAINTRK	-	04/26/88	Blugil	M	7414	46.7	14.5	0.04	-	-	-
PAINTRK	-	04/26/88	Blugil	M	7415	46.5	14.2	0.04	-	-	-
PAINTRK	-	04/26/88	Blugil	M	7416	41.7	13.5	0.04	-	-	-
PAINTRK	-	04/26/88	Blugil	F	7417	43.4	13.9	0.07	-	-	-
TELLICO	-	03/22/89	Blugil	F	7470	59.9	16.3	0.02	-	-	-
TELLICO	-	03/22/89	Blugil	M	7472	42.6	15.0	0.05	-	-	-
TELLICO	-	03/22/89	Blugil	M	7474	57.9	15.7	0.03	-	-	-
TELLICO	-	03/22/89	Blugil	M	7476	75.6	17.4	0.03	-	-	-
NORRIS	-	03/21/89	Blugil	F	7430	108.3	18.3	0.07	-	-	-
NORRIS	-	03/21/89	Blugil	M	7432	62.0	16.2	0.04	-	-	-
NORRIS	-	03/21/89	Blugil	M	7434	48.3	14.5	0.04	-	-	-
NORRIS	-	03/21/89	Blugil	M	7436	134.3	19.8	0.04	-	-	-
BALLPLAY	-	04/13/89	Blugil	F	7420	82.0	15.9	0.06	-	-	-
BALLPLAY	-	04/13/89	Blugil	M	7422	108.8	18.3	0.05	-	-	-
BALLPLAY	-	04/13/89	Blugil	M	7421	53.7	14.0	0.05	-	-	-
BALLPLAY	-	04/13/89	Blugil	M	7423	59.3	15.1	0.08	-	-	-
BRSHYFK	-	06/08/89	Blugil	F	7790	86.6	16.4	0.04	-	-	-
BRSHYFK	-	06/08/89	Blugil	M	7791	118.8	18.6	0.10	-	-	-
BRSHYFK	-	06/08/89	Blugil	F	7792	117.7	17.7	0.29	-	-	-
BRSHYFK	-	06/08/89	Blugil	F	7793	50.3	14.0	0.10	-	-	-

^aThe following metals were below the listed detection limit in all samples. Values are micrograms per gram, wet weight. Sb - <0.50, As - <0.50, Be - <0.10, Cr - <0.50, Pb - <0.50, Ni - <0.50, Be - <0.10, Cd - <0.20, and Ag - <0.20.

^bCRK = Clinch River kilometer; MHR = Melton Hill Reservoir; WCK = White Oak Creek kilometer; NTK = Northwest Tributary kilometer; MEK = Melton Branch kilometer; WOL = White Oak Lake; HINDSCR = Hinds Creek, Anderson County, Tennessee; PAINTRK = Pain Rock Creek, Loudon County, Tennessee; TELLICO = Tellico Reservoir, Loudon County, Tennessee; NORRIS = Norris Reservoir, Union County, Tennessee; BALLPLAY = Ballplay Creek, Monroe County, Tennessee; and BRSHYFK = Brushy Fork of Poplar Creek, Anderson County, Tennessee.

^cDist = distance from the mouth of the stream.

Table B.1 (continued)

^dSpp = species; Blugil = bluegill (*Lepomis macrochirus*); Redbre = redbreast sunfish (*Lepomis auritus*); CoCarp = carp (*Cyprinus carpio*); and Lmbass = largemouth bass (*Micropterus salmoides*).

^eM = male and F = female.

^fNo. = identification number.

Table B.2. Organic contaminants in fish from White Oak Creek and tributaries, nearby reaches of the Clinch River, and several reference sites

Site ^a	Dist ^b	Date	Spp ^c	Sex ^d	No. ^e	Weight (g)	Length (cm)	PCBs ^f ($\mu\text{g/g}$)	PCB- 1254 ^g ($\mu\text{g/g}$)	PCB- 1260 ^h ($\mu\text{g/g}$)	Other
CRK 31.5	31.5	11/21/88	Blugil	F	7810	83.0	17.1	0.07	0.06	0.01	-
CRK 31.5	31.5	11/21/88	Blugil	M	7811	122.0	18.7	0.06	0.05	0.01	-
CRK 31.5	31.5	11/21/88	Blugil	M	7812	54.4	14.5	0.08	0.07	0.01	-
CRK 31.5	31.5	11/21/88	Blugil	F	7813	61.9	15.2	0.06	0.05	0.01	-
CRK 31.5	31.5	11/21/88	Blugil	M	7814	49.5	14.2	0.07	0.06	0.01	-
CRK 31.5	31.5	11/21/88	Blugil	M	7815	90.3	17.4	0.04	0.03	0.01	-
CRK 31.5	31.5	11/21/88	Blugil	M	7816	153.1	20.0	0.11	0.09	0.02	-
CRK 31.5	31.5	11/21/88	Blugil	F	7817	62.1	15.3	0.10	0.08	0.02	-
CRK 35.2	35.2	11/21/88	Blugil	M	7800	93.2	16.5	0.20	0.19	0.01	-
CRK 35.2	35.2	11/21/88	Blugil	F	7801	50.8	14.6	0.30	0.28	0.02	-
CRK 35.2	35.2	11/21/88	Blugil	M	7802	77.2	15.4	0.08	0.07	0.01	-
CRK 35.2	35.2	11/21/88	Blugil	M	7803	69.0	15.5	0.17	0.10	0.07	-
CRK 35.2	35.2	11/21/88	Blugil	M	7804	45.6	14.0	0.14	0.13	0.01	-
CRK 35.2	35.2	11/21/88	Blugil	M	7805	51.1	14.5	0.09	0.08	0.01	-
CRK 35.2	35.2	11/21/88	Blugil	M	7806	47.1	14.2	0.09	0.08	0.01	-
CRK 35.2	35.2	11/21/88	Blugil	F	7807	88.8	16.8	0.12	0.11	0.01	-
MHR	38	11/23/88	Blugil	M	7880	152.6	20.6	0.19	0.16	0.03	-
MHR	38	11/23/88	Blugil	F	7881	86.8	17.5	0.20	0.18	0.02	-
MHR	38	11/23/88	Blugil	F	7882	97.3	18.2	0.11	0.09	0.02	-
MHR	38	11/23/88	Blugil	M	7883	59.1	15.5	0.05	0.04	0.01	-
MHR	38	11/23/88	Blugil	M	7884	81.1	17.0	0.11	0.08	0.03	-
MHR	38	11/23/88	Blugil	M	7885	79.8	17.4	0.11	0.10	0.01	-
MHR	38	11/23/88	Blugil	F	7886	73.7	16.8	0.17	0.15	0.02	-
MHR	38	11/23/88	Blugil	M	7887	53.9	15.0	0.28	0.27	0.01	-
WCK 3.5	3.5	12/14/88	Blugil	F	7310	91.6	16.9	0.29	0.23	0.06	BLD ⁱ
WCK 3.5	3.5	12/14/88	Blugil	M	7311	47.9	14.1	0.14	0.11	0.03	-
WCK 3.5	3.5	12/19/88	Blugil	F	7346	188.7	20.0	2.7	1.6	1.1	BLD
WCK 3.5	3.5	12/19/88	Blugil	M	7347	101.5	16.5	0.67	0.56	0.11	BLD
WCK 3.5	3.5	12/19/88	Blugil	M	7348	80.0	15.0	0.60	0.48	0.12	-
WCK 3.5	3.5	12/19/88	Blugil	F	7349	65.1	14.5	0.21	0.15	0.06	-
WCK 3.5	3.5	12/19/88	Blugil	F	7350	82.4	15.5	0.20	0.17	0.03	BLD
WCK 3.5	3.5	12/19/88	Blugil	F	7351	66.8	15.0	0.22	0.19	0.03	-
NTK 0.2	0.2	12/13/88	Blugil	M	7846	68.9	16.6	0.36	0.33	0.03	BLD
NTK 0.2	0.2	12/13/88	Blugil	F	7847	72.1	16.2	0.12	0.10	0.02	BLD
NTK 0.2	0.2	12/13/88	Blugil	F	7858	85.7	17.2	0.12	0.08	0.04	BLD
NTK 0.2	0.2	12/13/88	Blugil	M	7859	47.1	14.2	0.04	0.03	0.01	-
NTK 0.2	0.2	12/13/88	Blugil	F	7870	53.2	15.0	0.05	0.04	0.01	-
NTK 0.2	0.2	12/13/88	Blugil	M	7871	44.1	14.1	0.11	0.10	0.01	-
NTK 0.2	0.2	12/13/88	Blugil	M	7305	92.7	18.3	0.10	0.08	0.02	BLD
NTK 0.2	0.2	12/13/88	Blugil	M	7306	38.1	14.0	0.07	0.06	0.01	-

Table B.2 (continued)

Site ^a	Dist ^b	Date	Spp ^c	Sex ^d	No. ^e	Weight (g)	Length (cm)	PCBs ^f ($\mu\text{g/g}$)	PCB- 1254 ^g ($\mu\text{g/g}$)	PCB- 1260 ^h ($\mu\text{g/g}$)	Other
WCK 2.9	2.9	12/19/88	Blugil	M	7358	108.9	17.6	0.22	0.18	0.04	-
WCK 2.9	2.9	12/19/88	Blugil	M	7359	157.8	19.6	0.18	0.15	0.03	BLD
WCK 2.9	2.9	12/19/88	Blugil	M	7360	189.3	21.7	0.53	0.36	0.17	BLD
WCK 2.9	2.9	12/19/88	Blugil	M	7361	217.8	21.6	2.01	1.60	0.41	BLD
WCK 2.9	2.9	12/19/88	Blugil	M	7362	162.2	19.6	0.44	0.33	0.11	BLD
WCK 2.9	2.9	12/19/88	Blugil	F	7363	132.5	19.3	0.45	0.50	0.15	-
WCK 2.9	2.9	12/19/88	Blugil	M	7364	89.4	16.9	0.19	0.15	0.04	-
WCK 2.9	2.9	12/19/88	Blugil	M	7365	98.0	16.2	0.33	0.27	0.06	-
WCK 2.9	2.9	12/19/88	Redbre	M	7340	107.9	18.0	0.35	0.29	0.06	-
WCK 2.9	2.9	12/19/88	Redbre	M	7341	120.3	18.5	0.28	0.23	0.05	-
WCK 2.9	2.9	12/19/88	Redbre	M	7342	102.9	17.0	0.58	0.50	0.08	-
WCK 2.9	2.9	12/19/88	Redbre	M	7343	105.9	18.0	0.26	0.22	0.04	-
WCK 2.9	2.9	12/19/88	Redbre	M	7344	94.9	17.0	0.58	0.46	0.12	-
WCK 2.9	2.9	12/19/88	Redbre	F	7345	74.0	16.0	0.16	0.13	0.03	-
WCK 2.9	2.9	12/19/88	Redbre	F	7366	66.0	15.1	0.13	0.10	0.03	-
WCK 2.9	2.9	12/19/88	Redbre	F	7367	72.0	16.7	0.14	0.10	0.04	-
WCK 2.3	2.3	12/13/88	Blugil	M	7876	63.9	15.4	0.38	0.34	0.04	-
WCK 2.3	2.3	12/13/88	Blugil	M	7864	96.4	16.9	0.18	0.15	0.03	BLD
WCK 2.3	2.3	12/13/88	Blugil	M	7866	79.8	16.6	0.87	0.57	0.30	BLD
WCK 2.3	2.3	12/13/88	Blugil	M	7869	46.8	13.9	0.63	0.37	0.26	-
WCK 2.3	2.3	12/13/88	Blugil	M	7865	102.0	19.0	1.52	1.01	0.51	BLD
WCK 2.3	2.3	12/13/88	Blugil	F	7877	40.5	13.1	0.27	0.19	0.08	-
WCK 2.3	2.3	12/13/88	Blugil	M	7867	45.3	13.7	0.96	0.72	0.24	-
WCK 2.3	2.3	12/13/88	Blugil	F	7303	42.8	13.5	0.22	0.17	0.05	BLD
MEK 0.2	0.2	12/06/88	Blugil	F	7891	65.0	15.5	0.11	0.06	0.05	-
MEK 0.2	0.2	12/06/88	Blugil	M	7892	59.2	15.2	0.14	0.11	0.03	-
MEK 0.2	0.2	12/06/88	Blugil	M	7899	39.1	13.3	0.92	0.62	0.30	-
MEK 0.2	0.2	12/06/88	Blugil	M	7896	39.6	13.4	0.24	0.19	0.05	-
MEK 0.2	0.2	12/06/88	Blugil	M	7897	40.8	13.9	0.09	0.05	0.04	-
MEK 0.2	0.2	12/06/88	Blugil	M	7888	30.9	13.0	0.51	0.20	0.31	-
MEK 0.2	0.2	12/06/88	Blugil	F	7898	28.3	12.0	0.16	0.13	0.03	-
MEK 0.2	0.2	12/06/88	Blugil	M	7889	21.0	11.0	0.26	0.18	0.08	-
MEK 0.2	0.2	12/06/88	Redbre	M	7890	94.0	17.3	-	-	-	BLD
MEK 0.2	0.2	12/06/88	Redbre	M	7893	92.6	17.6	-	-	-	BLD
MEK 0.2	0.2	12/06/88	Redbre	M	7894	78.4	17.4	-	-	-	BLD
MEK 0.2	0.2	12/06/88	Redbre	M	7895	52.7	14.5	-	-	-	BLD
WOL	1.0	12/14/88	Blugil	M	7312	78.0	16.3	0.35	0.22	0.13	-
WOL	1.0	12/14/88	Blugil	M	7313	89.9	17.4	0.50	0.33	0.17	BLD
WOL	1.0	12/14/88	Blugil	M	7316	96.6	16.5	1.72	1.20	0.52	BLD
WOL	1.0	12/14/88	Blugil	F	7318	119.8	17.6	0.16	0.13	0.03	BLD
WOL	1.0	12/14/88	Blugil	M	7319	86.7	16.4	1.14	0.52	0.62	BLD

Table B.2 (continued)

Site ^a	Dist ^b	Date	Spp ^c	Sex ^d	No. ^e	Weight (g)	Length (cm)	PCBs ^f ($\mu\text{g/g}$)	PCB-	PCB-	Other
									1254 ^g ($\mu\text{g/g}$)	1260 ^h ($\mu\text{g/g}$)	
WOL	1.0	12/14/88	Blugil	M	7320	82.0	17.0	0.69	0.10	0.59	-
WOL	1.0	12/14/88	Blugil	M	7321	83.8	16.5	0.32	0.19	0.13	-
WOL	1.0	12/14/88	Blugil	F	7322	79.8	16.3	0.09	0.07	0.02	-
WOL	1.0	12/14/88	Cocarp	M	7332	2446	57.0	0.24	0.11	0.13	-
WOL	1.0	12/14/88	Cocarp	F	7334	1920	51.5	0.45	0.32	0.13	BLD
WOL	1.0	12/14/88	Cocarp	M	7335	1586	49.2	0.79	0.49	0.30	BLD
WOL	1.0	12/14/88	Cocarp	F	7336	3221	65.4	0.85	0.59	0.26	-
WOL	1.0	12/14/88	Cocarp	M	7337	2890	59.8	0.37	0.24	0.13	-
WOL	1.0	12/14/88	Cocarp	M	7338	2021	54.2	0.48	0.33	0.15	BLD
WOL	1.0	12/14/88	Cocarp	M	7339	2021	57.8	0.79	0.53	0.26	BLD
WOL	1.0	12/14/88	Lmbass	F	7324	1468	44.0	0.21	0.15	0.06	-
WOL	1.0	12/14/88	Lmbass	M	7325	388	31.2	0.22	0.15	0.07	-
WOL	1.0	12/14/88	Lmbass	F	7326	515	32.4	1.10	0.92	0.18	-
WOL	1.0	12/14/88	Lmbass	M	7327	362	29.0	0.70	0.51	0.19	-
WOL	1.0	12/14/88	Lmbass	M	7328	1155	41.7	0.80	0.54	0.26	-
WOL	1.0	12/14/88	Lmbass	M	7329	1175	40.8	3.90	2.60	1.30	-
WOL	1.0	12/14/88	Lmbass	M	7330	1019	40.0	1.05	0.78	0.27	-
WOL	1.0	12/14/88	Lmbass	M	7331	836	36.8	5.00	3.40	1.60	-
WCK 0.9	0.9	12/19/88	Blugil	F	7352	150.3	19.0	0.18	0.12	0.06	BLD
WCK 0.9	0.9	12/19/88	Blugil	M	7353	154.9	19.5	0.17	0.11	0.06	BLD
WCK 0.9	0.9	12/19/88	Blugil	M	7354	148.3	18.5	0.08	0.05	0.03	BLD
WCK 0.9	0.9	12/19/88	Blugil	M	7355	107.4	16.0	0.13	0.09	0.04	-
WCK 0.9	0.9	12/19/88	Blugil	M	7356	128.6	18.0	0.19	0.13	0.06	BLD
WCK 0.9	0.9	12/19/88	Blugil	M	7357	91.0	16.0	0.52	0.42	0.10	-
WCK 0.9	0.9	12/19/88	Blugil	M	7368	75.0	15.6	0.79	0.65	0.14	-
WCK 0.9	0.9	12/19/88	Blugil	F	7369	64.6	14.6	0.17	0.13	0.04	-
HINDSCR	-	12/05/88	Blugil	F	7844	53.2	15.2	0.08	0.05	0.03	BLD
HINDSCR	-	12/05/88	Blugil	F	7818	56.2	15.4	0.14	0.11	0.03	-
HINDSCR	-	12/05/88	Blugil	F	7808	57.7	15.2	0.05	0.05	<0.01	-
HINDSCR	-	12/05/88	Blugil	M	7809	134.4	19.7	0.02	0.02	<0.01	-
HINDSCR	-	12/05/88	Blugil	M	7848	154.5	20.3	0.03	0.02	0.01	-
HINDSCR	-	12/05/88	Blugil	F	7875	71.9	16.2	0.12	0.11	0.01	-
HINDSCR	-	12/05/88	Blugil	F	7873	69.7	15.7	0.14	0.13	0.01	-
HINDSCR	-	12/05/88	Blugil	F	7874	55.7	15.1	0.08	0.08	<0.01	-
HINDSCR	-	12/05/88	Redbre	F	7878	39.4	13.8	0.04	0.02	0.02	-
HINDSCR	-	12/05/88	Redbre	M	7843	52.9	14.2	0.18	0.17	0.01	-
HINDSCR	-	12/05/88	Redbre	M	7849	73.7	16.3	0.08	0.06	0.02	-
HINDSCR	-	12/05/88	Redbre	M	7842	78.5	16.9	0.05	0.04	0.01	-
HINDSCR	-	12/05/88	Redbre	M	7841	53.5	15.3	0.17	0.15	0.02	-

Table B.2 (continued)

Site ^a	Dist ^b	Date	Spp ^c	Sex ^d	No. ^e	Weight (g)	Length (cm)	PCBs ^f (µg/g)	PCB- 1254 ^g (µg/g)	PCB- 1260 ^h (µg/g)	Other
HINDSCR	-	12/05/88	Redbre	M	7872	55.5	15.2	0.07	0.07	<0.01	BLD
HINDSCR	-	12/05/88	Redbre	F	7840	52.4	15.4	0.01	<0.01	0.01	-
HINDSCR	-	12/05/88	Redbre	M	7845	55.2	15.2	0.18	0.17	0.01	-
WCK 2.9	2.9	07/10/89	Redbre	M	7781	113.9	17.8	0.78	0.55	0.23	-
WCK 2.9	2.9	07/10/89	Redbre	M	7785	84.4	15.0	0.17	0.14	0.03	-
WCK 2.9	2.9	07/10/89	Redbre	F	7786	60.1	14.2	0.07	0.05	0.02	-
WCK 2.9	2.9	07/10/89	Redbre	M	7784	128.7	17.4	0.22	0.12	0.10	-
WCK 2.9	2.9	07/10/89	Redbre	M	7782	117.9	18.0	0.14	0.07	0.07	-
WCK 2.9	2.9	07/10/89	Redbre	M	7780	137.4	18.1	0.67	0.51	0.16	-
WCK 2.9	2.9	07/10/89	Redbre	F	7783	45.5	12.9	0.11	0.07	0.04	-
WCK 2.9	2.9	07/10/89	Redbre	F	7887	56.7	13.8	0.07	0.05	0.02	-
WOL 1.0	1.0	07/13/89	Blugil	F	7750	91.2	17.6	0.24	0.17	0.07	-
WOL 1.0	1.0	07/13/89	Blugil	M	7751	57.4	14.4	0.76	0.49	0.27	-
WOL 1.0	1.0	07/13/89	Blugil	M	7752	67.0	15.4	0.33	0.23	0.10	-
WOL 1.0	1.0	07/13/89	Blugil	M	7753	73.5	16.0	0.80	0.55	0.25	-
WOL 1.0	1.0	07/13/89	Blugil	M	7754	107.8	17.6	0.02	0.01	0.01	-
WOL 1.0	1.0	07/13/89	Blugil	M	7755	97.5	17.3	0.16	0.10	0.06	-
WOL 1.0	1.0	07/13/89	Blugil	M	7756	71.8	15.6	0.67	0.16	0.51	-
WOL 1.0	1.0	07/13/89	Blugil	M	7757	80.5	15.0	0.56	0.35	0.21	-

^aCRK = Clinch River; MHR = Melton Hill Reservoir; WCK = White Oak Creek; NTK = Northwest Tributary; MEK = Melton Branch; WOL = White Oak Lake; HINDSCR = Hinds Creek, Anderson County, TN.

^bDist = distance from mouth of stream.

^cSPP = species; Blugil = bluegill (*Lepomis macrochirus*); Redbre = redbreast sunfish (*Lepomis auritus*); Lmbass = largemouth bass (*Micropterus salmoides*); Cocarp = carp (*Cyprinus carpio*).

^dM = male and F = female.

^eNo. = identification number.

^fPCBs = total PCBs (sum of PCB-1254 and PCB-1260) in fish axial muscle, in wet weight.

^gPCB-1254 (Arochlor 1254) in fish axial muscle, wet weight.

^hPCB-1260 (Arochlor 1254) in fish axial muscle, wet weight.

ⁱBLD = below limit of detection, as listed in Table B.3.

Table B.3. Detection limits of organic compounds ($\mu\text{g/g}$, wet wt)

Compound	Detection limit ^a ($\mu\text{g/g}$, wet weight)
Capillary column GC/MS^b	
Phenol	<2.0
Bis(2-chloroethylether)	<2.0
2-chlorophenol	<2.0
1,3-dichlorobenzene	<2.0
1,4-dichlorobenzene	<2.0
Benzyl alcohol	<2.0
1,2-dichlorobenzene	<2.0
2-methylphenol	<2.0
Bis(2-chlorodisopropyl)ether	<2.0
4-methylphenol	<2.0
N-nitroso-di-N-propylamine	<2.0
Hexachloroethane	<2.0
Nitrobenzene	<2.0
Isophorone	<2.0
2-nitrophenol	<2.0
2,4-dimethylphenol	<2.0
Benzoic acid	<10.0
Bis(2-chloroethoxy)methane	<2.0
2,4-dichlorophenol	<2.0
1,2,4-Trichlorobenzene	<2.0
Naphthalene	<2.0
4-chloroaniline	<2.0
Hexachlorobutradiene	<2.0
4-chloro-3-methylphenol	<2.0
2-methylnaphthalene	<2.0
Hexachlorocyclapentadiene	<2.0
2,4,6-trichlorophenol	<2.0
2,4,5-trichlorophenol	<10.0
2-chloronaphthalene	<2.0
2-nitroaniline	<10.0
Dimethylphthalate	<2.0
Acenaphthalene	<2.0
3-nitroaniline	<10.0
Acenaphthene	<2.0
2,4-dinitrophenol	<10.0
Nitrophenol	<10.0
Dibenzofuran	<2.0
2,4-dinitrotoluene	<2.0
2,6-dinitrotoluene	<2.0
Diethylphthalate	<2.0
4-chlorophenyl-phenylether	<2.0
Fluorene	<2.0
4-nitroaniline	<10.0

Table B.3 (continued)

Compound	Detection limit ^a ($\mu\text{g/g}$, wet weight)
4,6-dinitro-2-methylphenol	<10.0
<i>N</i> -nitrosodiphenylamine	<2.0
4-bromophenyl-phenylether	<2.0
Hexachlorobenzene	<2.0
Pentachlorophenol	<10.0
Phenanthrene	<2.0
Anthracene	<2.0
Di- <i>N</i> -butylphthalate	<2.0
Fluoranthene	<2.0
Pyrene	<2.0
Butylbenzylphthalate	<2.0
3,3-dichlorobenzidene	<10.0
Benz[<i>a</i>]anthracene	<2.0
Bis(2-ethylhexyl)phthalate	<2.0
Chrysene	<2.0
Di- <i>N</i> -octylphthalate	<2.0
Benzo[<i>b</i>]fluoranthene	<2.0
Benzo[<i>k</i>]fluoranthene	<2.0
Benzo[<i>a</i>]pyrene	<2.0
Indeno[1,2,3- <i>cd</i>]pyrene	<2.0
Dibenz[<i>a,h</i>]anthracene	<2.0
Benzo[<i>g,h,i</i>]perylene	<2.0
Capillary column GC/ECD^c	
Alpha-BHC	<0.02
Beta-BHC	<0.04
Delta-BHC	<0.04
Gamma-BHC	<0.02
Heptachlor	<0.04
Aldrin	<0.04
Heptachlor epoxide	<0.04
Endosulfan I	<0.04
Dieldrin	<0.04
4,4'-DDE	<0.04
Endrin	<0.2
Endosulfan II	<0.08
4,4'-DDD	<0.2
Endosulfan sulfate	<0.2
4,4'-DDT	<0.08
Endrin ketone	<0.4
Methoxychlor	<0.2
Alpha chlordane	<0.04
Gamma chlordane	<0.04
Toxaphene	<2

Table B.3 continued)

Compound	Detection limit ^a ($\mu\text{g/g}$, wet weight)
HPLC^d with fluorescence detection	
Naphthalene	<0.12
Acenaphthene	<0.03
Phenanthrene	<0.01
Anthracene	<0.10
Fluoranthene	<0.2
Pyrene	<0.003
Benz[<i>a</i>]anthracene	<0.001
Benzo[<i>b</i>]fluoranthene	<0.03
Benzo[<i>k</i>]fluoranthene	<0.02
Benzo[<i>a</i>]pyrene	<0.004
Dibenz[<i>a,h</i>]anthracene	<0.002
Benzo[<i>g,h,i</i>]perylene	<0.005
Indeno[1,2,3- <i>cd</i>]pyrene	<1.0
Chrysene	<0.003

^aOptimum detection limit in 10-g sample. Sample weights varied between 5 and 10 g, so detection limits were higher in some samples.

^bGC/MS = gas chromatography/mass spectrometry.

^cGC/ECD = gas chromatography/electron capture detector.

^dHPLC = high performance liquid chromatography.

Table B.4. Concentrations of polychlorinated biphenyls and strontium-90 in channel catfish from the White Oak Creek embayment, lower Poplar Creek, the Clinch River, and Melton Hill Reservoir

Site ^a	Location ^b	Type ^c	Date	Species ^d	Sex ^e	No. ^f	Weight (g)	Length (cm)	ΣPCB ^g (μg/g)	PCB-1254 ^h (μg/g)	PCB-1260 ⁱ (μg/g)	⁹⁰ Sr ^j (Bq/kg)	Lipid ^k (%)
WOC	WCK 0.3	R	8/01/89	Ch. Cat	F	7789	424	38.4	2.07	1.53	1.53	4000	3.28
WOC	WCK 0.3	R	8/02/89	Ch. Cat	M	7499	1475	56.4	0.86	0.32	0.54	470	4.98
WOC	WCK 0.9	R	8/02/89	Ch. Cat	M	7796	2873	63.8	2.07	0.63	1.44	1400	2.03
WOC	WCK 0.9	R	8/02/89	Ch. Cat	M	7797	1179	49.5	2.68	1.21	1.47	2300	6.12
WOC	WCK 0.9	R	8/02/89	Ch. Cat	F	7795	944	46.2	2.30	0.93	1.37	1200	2.83
WOC	WCK 0.9	R	8/02/89	Ch. Cat	F	7798	1525	50.6	0.63	0.04	0.59	80	3.75
WOC	WCK 0.9	R	8/02/89	Ch. Cat	M	7799	1357	51.8	0.61	0.23	0.38	1600	5.04
WOC	WCK 0.9	R	8/02/89	Ch. Cat	M	7331	1652	53.4	1.08	0.53	0.55	2900	5.65
CRD	CRK 32.2	R	8/08/89	Ch. Cat	M	7736	1273	53.0	0.52	0.12	0.40	200	2.77
CRD	CRK 32.2	R	8/14/89	Ch. Cat	M	7760	414	37.4	1.34	0.26	1.08	60	3.18
CRD	CRK 32.2	R	8/17/89	Ch. Cat	-	7762	504	47.8	0.69	<0.01	0.69	110	0.28
CRD	CRK 32.2	R	8/22/89	Ch. Cat	M	7766	448	38.5	0.86	0.06	0.80	110	2.38
CRD	CRK 32.2	R	8/28/89	Ch. Cat	M	6544	479	40.4	0.54	0.22	0.32	80	3.35
CRD	CRK 32.2	R	9/06/89	Ch. Cat	F	5012	486	40.6	0.66	0.11	0.55	-60	1.49
CRD	CRK 32.2	R	9/14/89	Ch. Cat	F	5013	962	46.9	1.95	0.18	1.77	2	6.08
CRD	CRK 32.2	R	9/14/89	Ch. Cat	F	5020	1321	50.2	3.06	0.77	2.29	-12	6.97
MHR	MHR	R	8/09/89	Ch. Cat	F	7735	448	38.5	0.23	0.09	0.14	-	5.41
MHR	MHR	R	8/11/89	Ch. Cat	F	7738	1163	52.0	0.29	0.07	0.22	-43	4.19
MHR	MHR	R	8/17/89	Ch. Cat	M	7763	1695	56.2	0.55	0.14	0.41	35	-
MHR	MHR	R	9/08/89	Ch. Cat	M	5019	585	42.5	0.24	0.09	0.15	-10	3.76
MHR	MHR	R	9/19/89	Ch. Cat	F	5021	401	38.7	0.14	0.04	0.10	-40	2.47
MHR	MHR	R	9/19/89	Ch. Cat	F	5022	486	40.5	0.30	0.03	0.27	120	3.09
MHR	MHR	R	9/19/89	Ch. Cat	F	5023	527	40.8	0.23	0.08	0.15	0	4.82
MHR	MHR	R	8/09/89	Ch. Cat	M	5024	1175	51.8	0.28	0.01	0.27	110	1.78
CR	CRK 15.0	R	8/25/89	Ch. Cat	F	6560	548	40.0	0.61	0.01	0.60	81	3.76
CR	CRK 15.0	R	8/29/89	Ch. Cat	F	6551	569	42.8	<0.01	<0.01	<0.01	-28	0.40
CR	CRK 15.0	R	8/30/89	Ch. Cat	M	5001	665	44.4	0.35	0.11	0.24	80	4.69
CR	CRK 15.0	R	8/30/89	Ch. Cat	M	5002	521	41.0	1.16	0.27	0.89	60	5.61
CR	CRK 15.0	R	8/31/89	Ch. Cat	F	5007	811	45.5	0.41	0.17	0.24	270	6.42
CR	CRK 15.0	R	9/06/89	Ch. Cat	F	5014	1505	52.3	1.35	<0.01	1.35	70	2.59
CR	CRK 15.0	R	9/06/89	Ch. Cat	F	5015	827	45.0	0.30	0.10	0.20	210	2.95
CR	CRK 15.0	R	9/06/89	Ch. Cat	M	5016	592	41.2	2.10	<0.01	2.10	150	1.93
PC	PCK 6.9	R	8/22/89	Ch. Cat	M	7767	879	49.6	3.37	0.06	3.31	100	0.84
PC	PCK 6.9	R	8/23/89	Ch. Cat	M	7772	810	48.0	0.25	0.04	0.21	170	1.62
PC	PCK 6.9	R	8/23/89	Ch. Cat	M	7775	624	40.6	0.33	0.01	0.32	80	1.88
PC	PCK 6.9	R	8/23/89	Ch. Cat	M	7770	738	45.0	0.50	0.10	0.40	-15	3.83
PC	PCK 6.9	R	8/24/89	Ch. Cat	M	7776	940	50.0	1.01	0.05	0.96	40	2.36
PC	PCK 6.9	R	8/24/89	Ch. Cat	M	7777	1058	50.8	0.95	0.10	0.85	20	3.29
PC	PCK 6.9	R	8/25/89	Ch. Cat	M	7779	1002	49.2	1.19	0.20	0.99	28	3.86
PC	PCK 6.9	R	8/25/89	Ch. Cat	M	6559	1961	57.5	0.95	0.48	0.47	75	3.21
CEC	CEC	C	7/15/89	Ch. Cat	M	CEC-1	780	44.5	<0.01	<0.01	<0.01	-	6.14
CEC	CEC	C	7/15/89	Ch. Cat	M	CEC-3	1245	49.5	<0.01	<0.01	<0.01	-	4.74
CEC	CEC	C	7/15/89	Ch. Cat	M	CEC-3A	1245	49.5	0.01	0.01	<0.01	-	3.87
CEC	CEC	C	7/15/89	Ch. Cat	M	CEC-4	1500	55.1	0.02	0.02	<0.01	-	0.47
CEC	CEC	C	7/15/89	Ch. Cat	M	CEC-4A	1500	55.1	<0.01	<0.01	<0.01	-	0.48

Table B.4 (continued)

^aWOC = White Oak Creek, CRD = Clinch River Dam, MHR = Melton Hill Reservoir, CR = Clinch River, PC = Poplar Creek, and CEC = commercial catfish farm that is used for reference site.

^bWCK = White Oak Creek kilometer, CRK = Clinch River kilometer, MHR = Melton Hill Reservoir, PCK = Poplar Creek kilometer, and CEC = commercial catfish farm that is used for reference site.

^cR = regular and C = control.

^dCh. Cat = channel catfish (*Ictalurus punctatus*).

^eM = male, F = female.

^fNo. = fish identification number.

^g Σ PCB = total polychlorinated biphenyls (sum of PCB-1254 and PCB-1260) in fish axial muscle, wet weight.

^h1254 = PCB-1254 in fish axial muscle, wet weight.

ⁱ1260 = PCB-1260 in fish axial muscle, wet weight.

^j⁹⁰Sr = strontium-90 in fish vertebrae, dry weight. Negative values are a result of random variation in counting statistics on samples with only background levels of ⁹⁰Sr.

^kLipid content of fish axial muscle, wet weight.

Appendix C

**CHECKLIST OF BENTHIC MACROINVERTEBRATE TAXA
FROM THE WHITE OAK CREEK WATERSHED,
MAY-JUNE 1987**

Table C.1. Checklist of benthic invertebrate taxa collected from White Oak Creek watershed, May-June 1987^a

Taxon	Site ^b															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8	
Coelenterata																
<i>Hydra</i>															X	
Turbellaria																
Planariidae	X	X	X	X	X		X	X	X	X	X		X	X		
Planariidae? ^c												X	X			
Nematoda	X		X	X	X	X	X	X	X	X	X					
Tardigrada					X											
Oligochaeta	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Crustacea																
Cladocera								X	X	X		X				
Copepoda	X			X	X	X	X	X	X	X	X	X		X		
Ostracoda				X	X	X	X		X		X					
Isopoda																
<i>Lirceus</i>	X	X		X	X		X	X	X			X		X		
Amphipoda																
<i>Crangonyx</i>	X			X		X	X		X							
<i>Gammarus</i>	X	X	X		X											
Decapoda		X			X					X					X	
Cambaridae	X							X	X							
Hydracarina	X	X		X	X	X		X		X	X	X		X		
<i>Wandesia?</i>						X										

Table C.1 (continued)

Taxon	Site ^b														
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Insecta															
Collembola															
Anthropleona															
Entomobryomorpha		X	X	X					X			X	X	X	
Smyphypleona															
Sminthuridae			X	X											
Ephemeroptera															
Baetidae	X			X	X					X					X
<i>Baetis</i>		X	X	X	X	X	X		X	X		X		X	X
<i>Baetis?</i>				X											
<i>Cloeon</i>						X	X		X						
<i>Pseudocloeon</i>				X											X
Ephemeridae															
<i>Ephmera</i>									X						X
Ephemerellidae				X											X
<i>Ephemerella</i>				X											X
<i>Eurylophella</i>				X										X	X
<i>Serratella</i>															X
Heptageniidae							X								
<i>Leucrocuta</i>							X								
<i>Stenacron</i>					X										X
<i>Stenacron?</i>															X
Leptophlebiidae?									X						
Leptophlebiidae			X	X	X										
<i>Habrophlebiodes</i>		X		X					X						X
<i>Paraleptophlebia</i>							X		X						

Table C.1 (continued)

Taxon	Site ^b															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8	
Oligoneuriidae																
<i>Isonychia</i>																X
Siphonuridae																
<i>Ameletus</i>							X		X							
Odonata																
Anisoptera																
Aeshnidae																
<i>Boyeria</i>						X		X		X			X			
Gomphidae					X		X	X	X				X			
<i>Stylogomphus?</i>								X	X							
<i>Stylogomphus</i>																
<i>albistylus</i>	X				X	X				X	X					X
Cordulegastriidae																
<i>Cordulegaster</i>				X			X									
Corduliidae																
<i>Somatochlora</i>							X									
Coenagrionidae?	X												X			
Coenagrionidae									X			X	X			
<i>Argia</i>			X		X						X	X		X		
<i>Argia</i>																
<i>moesta</i>												X				
<i>Enallagma</i>	X															
<i>Enallagma?</i>							X									
Plecoptera								X				X			X	X
Capniidae?												X				
Capniidae						X										
Chloroperlidae								X	X						X	X
<i>Haploperla</i>									X							X
<i>Sweltsa?</i>																X

Table C.1 (continued)

Taxon	Site ^b														
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Leuctridae?								X	X			X			
Leuctridae						X			X						
<i>Leuctra</i>		X		X				X	X	X	X	X		X	X
<i>Leuctra?</i>	X			X						X	X				X
Nemouridae															
<i>Amphinemura</i>				X	X	X	X	X	X		X				X
Peltoperlidae															
<i>Peltoperla</i>								X							
<i>Tallaperla</i>		X													
Perlidae				X		X					X				
<i>Eccopectera</i>															
<i>xanthenes</i>															X
<i>Perlesta</i>		X			X	X	X		X						
<i>Perlesta?</i>										X					
Perlodidae							X								X
<i>Isoperla</i>							X		X						X
Taeniopterygidae															
<i>Taeniopteryx</i>	X									X					
Hemiptera															
Gerridae		X					X								
<i>Gerris?</i>							X								
Microveliidae															
<i>Microvelia</i>		X	X	X											
Veliidae			X				X								
<i>Rhagovelia</i>	X	X						X							
Megaloptera															
Corydalidae															
<i>Nigronia</i>	X	X		X		X	X	X	X		X	X			X

Table C.1 (continued)

Taxon	Site ^b															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8	
<i>Nigronia fasciatus</i>				X	X					X					X	
<i>Nigronia serricornis</i>															X	
Sialidae																
<i>Sialis</i>		X	X	X	X	X	X	X	X			X			X	
Trichoptera	X			X			X							X		
Glossosomatidae							X									
<i>Agapetus</i>							X									
<i>Glossosoma</i>		X		X	X											
<i>Glossosoma?</i>				X										X	X	
Hydropsychidae	X	X		X	X		X	X	X	X		X		X	X	
<i>Cheumatopsyche</i>	X	X	X	X	X	X		X	X	X	X				X	
<i>Cheumatopsyche?</i>					X								X			
<i>Diplectrona modesta</i>				X										X	X	
<i>Hydropsyche</i>	X		X	X	X	X	X	X		X	X	X			X	
<i>Hydropsyche depravata</i>				X												
Hydroptilidae				X	X			X	X	X					X	
<i>Hydroptila</i>				X	X		X		X							
<i>Ochrotrichia</i>				X			X		X							
<i>Ochrotrichia?</i>								X								
Lepidostomatidae																
<i>Lepidostoma</i>				X												
Leptoceridae									X	X						
<i>Oecetis</i>											X					
<i>Triaenodes</i>																
<i>Triaenodes?</i>							X									

Table C.1 (continued)

Taxon	Site ^b														
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Limnephilidae															
<i>Goera</i>				X											
<i>Neophylax</i>		X		X			X	X							X
<i>Pycnopsyche</i>						X									
<i>Pycnopsyche</i> <i>scabripennis</i>											X				
Molannidae															
<i>Molanna</i>				X											
Philopotamidae?											X				
Philopotamidae															
<i>Chimarra</i>					X	X		X		X					X
<i>Dolophilodes</i> <i>distinctus</i>															X
<i>Dolophilodes?</i>						X									
Polycentropodidae					X										
<i>Polycentropus</i>										X					X
Psychomyiidae															
<i>Lype diversa</i>				X											
Rhyacophilidae															
<i>Rhyacophila</i>		X		X					X						X
Coleoptera															
Dryopidae															
<i>Helichus</i>									X						X
Dytiscidae				X								X			
<i>Hydroporus</i>							X		X						
<i>Hydroporus?</i>							X		X						

Table C.1 (continued)

Taxon	Site ^b															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8	
Elmidae											X					
<i>Optioservus</i>	X	X	X	X		X	X	X	X	X	X	X	X	X	X	
<i>Optioservus?</i>		X														
<i>Stenelmis</i>	X	X	X	X	X	X	X	X	X	X	X			X	X	
<i>Stenelmis crenata</i>	X			X								X				
Eubriidae																
<i>Ectopria</i>	X				X	X	X			X	X	X		X	X	
Hydrophilidae				X					X							
<i>Berosus</i>												X	X			
<i>Helophorus</i>								X								
<i>Tropisternus</i>					X								X			
Psephenidae																
<i>Psephenus herricki</i>		X						X		X		X			X	
Ptilodactylidae																
<i>Anchytarsus bicolor</i>		X													X	
Diptera																
Ceratopogonidae	X	X	X			X	X	X	X	X	X				X	
<i>Atrichopogon</i>				X		X		X	X			X		X		
Ceratopogonidae?												X				
Chironomidae	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	
Dixidae																
<i>Dixa</i>	X	X	X				X		X					X	X	
Empididae			X	X												
<i>Chelifera</i>					X											
<i>Chelifera?</i>				X												

Table C.1 (continued)

Taxon	Site ^b															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8	
Diptera (continued)																
<i>Clinocera</i>			X	X												X
<i>Hemerodromia</i>	X	X	X		X	X		X		X	X	X	X	X		X
<i>Hemerodromia?</i>		X														X
Muscidae		X														
Muscidae?	X															X
Psychodidae																
<i>Pericoma</i>			X													X
<i>Psychoda</i>			X													
<i>Psychoda?</i>								X								
Psychodidae?									X							
Sciomyzidae?	X															
Simuliidae	X		X					X								
<i>Simulium</i>	X	X	X	X	X	X		X		X	X	X	X	X	X	X
<i>Simulium</i>																
<i>tuberosum?</i>				X												
<i>Simulium</i>																
<i>vittatum</i>				X						X		X				
<i>Simulium?</i>												X				X
Stratiomyidae			X													
<i>Caloparyphus</i>			X	X												
Stratiomyidae?	X															
Tabanidae																
<i>Chrysops</i>							X		X							
<i>Tabanus</i>					X			X								
Tipulidae				X				X				X				
<i>Antocha</i>		X		X										X		
<i>Hexatoma</i>				X			X		X					X		X
<i>Hexatoma?</i>				X			X									

Table C.1 (continued)

Taxon	Site ^b														
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Tipulidae (continued)															
<i>Limonia?</i>														X	
<i>Pilaria</i>				X											
<i>Pseudolimnophila</i>										X					X
<i>Tipula</i>			X		X			X				X		X	X
<i>Tipula abdominalis</i>	X							X							X
<i>Tipula?</i>															X
Mollusca															
Gastropoda															
Ancylidae															
<i>Ferrissia</i>			X			X	X	X						X	
Lymnaeidae															
<i>Lymnaea</i>														X	
<i>Pseudosuccinea columella</i>	X				X	X	X			X	X				
Lymnaeidae?															
Physidae															
<i>Physella</i>	X		X		X					X	X	X	X		
<i>Physella?</i>											X				
Planorbidae															
<i>Gyraulus</i>								X							
Planorbidae?															
Pleuroceridae															
<i>Elimia</i>		X													X

Table C.1 (continued)

Taxon	Site ^b														
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 1.0	MEK 0.6	MEK 1.2	MEK 2.1	NTK 0.2	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Bivalvia															
Corbiculidae															
<i>Corbicula fluminea</i>															
<i>Corbicula?</i>							X				X	X			
Sphaeriidae	X			X								X	X		
<i>Pisidium</i>	X			X			X		X		X	X			X
<i>Sphaerium</i>							X	X			X	X			
<i>Sphaerium?</i>											X				
Sphaeriidae?											X		X		

^aAn "X" indicates that the taxon was collected at least once in quantitative samples.

^bFCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; and WCK = White Oak Creek kilometer.

^cA "?" indicates that some uncertainty exists as to precise identification.

Appendix D

METHODOLOGY FOR INDEX OF BIOTIC INTEGRITY

Appendix D

METHODOLOGY FOR INDEX OF BIOTIC INTEGRITY

The fish population data at each site were analyzed using the Index of Biotic Integrity (IBI). The IBI evaluates "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region" (Karr and Dudley 1981). The IBI includes 12 metrics (Table D.1) related to species richness and composition, trophic composition, and fish abundance and condition and was originally based on studies in the Midwest (Karr 1981, Karr et al. 1986). Application of the IBI results in a numerical value between 12 and 60 that represents a comparative descriptive evaluation of the fish community. Because of geographic differences in species distribution, many of the 12 metrics were modified when the IBI was used in different regions of the United States (Fausch et al. 1984, Miller et al. 1988). Also, more widespread use of the IBI has led to suggestions for replacement metrics that may better address the intent of the original metrics in a particular region (Ohio EPA 1988). These modifications have not diminished the value of the evaluation (Miller et al. 1988).

Following the suggestions of Karr et al. (1986) and Ohio EPA (1988), modifications were made to the basic IBI metrics to reflect differences in the Clinch River system in the Oak Ridge area. Using historical surveys of the area (Krumholz 1956, Fitz 1968, Etnier 1978), regional distribution information (Lee et al. 1980, Ryon and Loar 1988, M. G. Ryon, ORNL, unpublished data), and unpublished species accounts (D. A. Etnier, University of

Tennessee, Knoxville, unpublished memo, 1987), a baseline list of species was selected that would be present in the WOC drainage under the best conditions (Table D.2). These species were rated for tolerance to stream impacts, evaluated for spawning pattern, and grouped by trophic level using various sources [Smith 1979; Trautman 1981; Becker 1983; Karr et al. 1986; D. A. Etnier, University of Tennessee, Knoxville, unpublished memo, 1987; Ohio EPA 1987; C. Saylor, TVA, personal communication]. Also, additional fish surveys were conducted on streams in the Hinds Creek watershed, another system in the Clinch River drainage with hydrologic and geologic characteristics similar to the WOC watershed. These surveys were performed to evaluate the relationship of fish abundance and species composition to watershed area and used the same technique that is used in the routine population surveys of streams in the WOC watershed (Sect. 6.2.2) (M. G. Ryon, ORNL, unpublished data). The resulting modified IBI is shown in Table D.3.

To evaluate stream fish communities in WOC, several major changes were made in the IBI metrics originally developed by Karr et al. (1986) for midwestern streams. Metrics 5–8, 10, and 11 were modified for the Oak Ridge area, as described below.

The number and identity of intolerant species (metric 5) was modified to reflect differences in species sensitivity to environmental degradation. A similar approach was used by Ohio EPA (1987, 1988). Such an approach allowed a more flexible interpretation of this metric, and, in a stream such as WOC, with limited

Table D.1. Index of Biotic Integrity metrics used to assess stream fish communities in the Midwest

Category	Metric	Scoring criteria		
		5	3	1
Species richness and composition	1. Total number of fish species	Expectations for metrics 1-5 vary with stream size		
	2. Number and identity of darter species			
	3. Number and identity of sunfish species			
	4. Number and identity of sucker species			
	5. Number and identity of intolerant species and region			
	6. Proportion of individuals as green sunfish			
Trophic composition	7. Proportion of individuals as omnivores	<20%	20-45%	>45%
	8. Proportion of individuals as benthic insectivorous cyprinids	>45%	45-20%	<20%
	9. Proportion of individuals as top carnivores	>5%	5-1%	<1%
Fish abundance and condition	10. Number of individuals in sample	Expectations vary with stream size and other factors		
	11. Proportion of individuals as hybrids	0%	>0-1%	>1%
	12. Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies	0-2%	>2-5%	>5%

Source: Karr, J. R., et al., 1986, *Assessing Biological Integrity in Running Waters: A Method and Its Rationale*. Illinois Natural History Survey Special Publication 5.

Table D.2. List of species found in small streams in the Clinch River drainage near Oak Ridge, Tennessee^a

Species	Trophic group ^b	Tolerance Ranking ^c			
		TOL	VIN	MIN	SIN
<i>Cyprinus carpio</i>	GEN	X			
<i>Campostoma anomalum</i>					
<i>Luxilus chrysocephalus</i> ^d	GEN	X			
<i>Notropis rubellus</i> ^d	GEN		X		
<i>Phoxinus tennesseensis</i> ^d	GEN		X		
<i>Pimephales notatus</i>	GEN				
<i>P. promelas</i>	GEN	X			
<i>Rhinichthys atratulus</i> ^d	GEN				
<i>Semotilus atromaculatus</i>	GEN	X			
<i>Dorosoma cepedianum</i>	GEN	X			
<i>Catostomus commersoni</i> ^d	GEN	X			
<i>Hypentelium nigricans</i> ^d	BIN			X	
<i>Ameiurus melas</i>	GEN	X			
<i>A. natalis</i>	GEN	X			
<i>Fundulus notatus</i>		X			
<i>Gambusia affinis</i>		X			
<i>Ambloplites rupestris</i>	PI				X
<i>Lepomis auritus</i>					
<i>L. macrochirus</i>					
<i>L. megalotis</i>					X
<i>M. punctulatus</i>	PI				
<i>M. salmoides</i>	PI				
<i>Etheostoma caeruleum</i> ^d	BIN		X		
<i>E. duryi</i>	BIN				X
<i>E. kennicotti</i>	BIN				X
<i>E. rufilineatum</i>	BIN			X	
<i>E. simoterum</i>	BIN				X
<i>Cottus carolinae</i>	BIN			X	

^aIncludes information on trophic group, tolerance rankings, and lithophilic spawners.

^bGEN = generalist, PI = piscivore, and BIN = benthic insectivore.

^cTOL = tolerant, VIN = very intolerant, MIN = moderately intolerant, SIN = slightly intolerant.

^dLithophilic spawner.

Table D.3. Index of biotic integrity metrics used to assess fish communities in streams in the Clinch River drainage near Oak Ridge, Tennessee

Category	Metric	Scoring criteria		
		5	3	1
Species richness and composition	1. Total number of fish species ^a	>16	16-9	8-0
	2. Number and identity of darter species	>3	3-2	1-0
	3. Number and identity of sunfish species	>3	3-2	1-0
	4. Number and identity of sucker species	>1	1	0
	5. Number and identity of intolerant species ^b	>8	8-4	3-0
	6. Proportion of individuals as tolerant species	<5%	5-20%	>20%
Trophic composition	7. Proportion of individuals as generalist feeders	<20%	20-45%	>45%
	8. Proportion of individuals as benthic insectivores	>45%	45-20%	<20%
	9. Proportion of individuals as piscivores	>5%	5-1%	<1%
Fish abundance and condition	10. Number of individuals in sample (density in numbers per m ²)	>3.0	1.0-3.0	<1.0
	11. Proportion of individuals as lithophilic spawners ^c	>36%	36-18%	<18%
	12. Proportion of individuals in subsamples with disease, (eye status, gill, thymus, opercle, and pseudobranch condition), skin tumors, fin damage, skeletal anomalies, or external parasites and skeletal anomalies	0-2%	>2-5%	>5%

^aNumber of native species, excluding recent introductions or stocked species.

^bIntolerant species ranked as very intolerant, moderately intolerant, or slightly intolerant with a correction factor of 1.2, 1.0, or 0.8, respectively, applied to the number in each category to achieve the numbers used in the criteria rankings.

^cPercentages as used in Ohio Environmental Protection Agency, 1987, *Biological Criteria for the Protection of Aquatic Life, Volume III: Standardized Biological Field Sampling and Laboratory Methods for Assessing Fish and Macroinvertebrate Communities*, Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio, and Ohio Environmental Protection Agency, 1988, *Biological Criteria for the Protection of Aquatic Life, Volume II: Users Manual for Biological Field Assessment of Ohio Surface Streams*, Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio.

species, gave greater sensitivity to the analysis. The number of species in three categories (very, moderately, and slightly intolerant) were multiplied by 1.2, 1.0 or 0.8, respectively, to obtain a value for metric 5 in Table D.3.

The proportion of green sunfish (metric 6) was changed to the proportion of tolerant species because the green sunfish is not a common species in the Clinch River drainage. The proportion of tolerant species has been used in other regions to address this same problem (Miller et al. 1988) and was even suggested as a replacement metric in the original IBI (Karr et al. 1986). The original intent of the metric, which was to indicate degradation by monitoring the changes in abundance of an extremely tolerant species, is still satisfied by using changes in the proportion of all tolerant species.

The proportion of individuals as omnivore feeders (metric 7) was changed to the proportion of generalist feeders. This change was incorporated because omnivores are not common in the Clinch River system. Also, species that are generalist feeders have the capability to switch prey preferences based on abundance and, therefore, can usually survive in areas with significant disturbances. The replacement metric still meets the original intentions of metric 7 as defined by Karr et al. (1986).

The proportion of insectivorous cyprinids (metric 8) is another metric that has been frequently modified for other regions (Miller et al. 1988). It was modified for the Clinch River drainage to include only benthic insectivore species. This approach was similar to that used by Leonard and Orth (1986) but not as broad (all insectivores) as that used by Ohio EPA (1988). The intent of this metric was to identify impacts on the benthic invertebrate community, as reflected by the abundance of insectivores. Because the Clinch River drainage had significant

numbers of benthic-feeding cyprinids, catostomids, and percids, adjustment of this metric to include all insectivores was deemed unnecessary. Also, by limiting the metric to benthic feeders, interference of surface-feeding cyprinids (e.g., creek chubs) that could survive in areas with low benthic invertebrate densities was eliminated.

The number of individuals in a sample (metric 10) was modified to use the density of fish, thereby reflecting the area sampled. This approach standardized comparisons between sites and was especially useful for BMAP studies because routine population surveys are restricted to sampling reaches of known length. This approach may affect species richness values (i.e., richness is often proportional to area sampled) but provided good estimates of population size and density. The extensive fish sampling in the Hinds Creek watershed also used BMAP procedures.

The proportion of hybrid individuals (metric 11) was initially incorporated into the IBI to address impacts to reproductive isolation and success (Karr et al. 1986). However, hybrids can be difficult to identify in the field and often occur at sites of high biotic integrity (Ohio EPA 1988). Consequently, this metric was replaced by one that examines the proportion of simple lithophilic spawners, as suggested by Ohio EPA (1988). Because lithophilic spawners release their eggs in gravel without parental care, the proportion of the community made up of such species would be affected by increases in siltation and pollutants (Berkman and Rabeni 1987). Although this modification reflects the concern for effects on fish reproduction contained in the original metric, it was done with some caution because many of the lithophilic spawners were also the most tolerant species (e.g., striped shiners).

The modifications in the IBI for use in the Clinch River drainage reflected changes appropriate for its application in East Tennessee. These modifications were

similar to species lists used by TVA for other areas of East Tennessee (C. Saylor, TVA, personal communication) and to changes implemented by other agencies (Ohio EPA 1987, 1988). Application of the IBI for use in the Clinch River drainage is in the initial stages of development, so specific computations may change as additional reference sites are

sampled (e.g., for abundance metrics) and some metrics may require additional modification. However, the descriptive classifications (very poor, poor, fair, good, or excellent) for a particular site should remain relatively unchanged because no major modifications to the IBI are anticipated.

Appendix E

**DENSITY, BIOMASS, CONDITION FACTORS, AND GROWTH
OF FISHES IN THE WHITE OAK CREEK WATERSHED**

Table E.1. Fish density in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary April 1989

Species	Fish density per site ^a (fish/m ²)															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Centrarchidae																
Bluegill	0.02	— ^b	—	—	—	—	—	NF ^c	0.01	—	0.12	—	0.05	NF	—	—
Redbreast sunfish	—	—	—	—	—	0.18	—	—	—	—	0.17	0.16	0.08	—	—	—
Largemouth bass	—	—	—	—	—	—	—	—	—	—	0.01	0.01	—	—	—	—
Warmouth	—	—	—	—	—	—	—	—	—	—	0.04	—	—	—	—	—
Rock bass	—	—	—	—	—	—	—	—	—	—	<0.01	—	—	—	—	—
Cottidae																
Banded sculpin	—	—	—	0.17	2.23	—	—	—	—	—	—	—	—	—	—	0.15
Cyprinidae																
Blacknose dace	3.00	1.62	0.07	8.52	0.80	1.06	1.32	—	0.04	—	—	—	—	—	4.04	1.02
Central stoneroller	—	—	—	—	—	—	—	—	—	—	0.03	—	—	—	1.48	—
Creek chub	—	—	—	—	—	0.5	0.2	—	—	—	—	—	—	—	0.22	0.05
Fathead minnow	0.51	—	—	—	—	—	—	—	—	0.04	—	<0.01	0.01	—	0.01	—
Ictaluridae																
Yellow bullhead	—	—	—	—	—	—	—	—	—	—	0.02	—	—	—	—	—
Poeciliidae																
Western mosquitofish	0.25	—	—	—	—	—	—	—	0.77	—	—	0.05	0.04	—	—	—
Number of species (N)	4	1	1	2	2	3	2	0	3	1	7	4	4	0	4	3
Total density	3.78	1.62	0.07	8.69	3.03	1.80	1.56	0	0.82	0.04	0.39	0.22	0.18	0	5.75	1.22

^aFCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

^b— = species not located.

^cNF = no fish taken in sample.

Table E.2. Fish biomass in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, April 1989

Species	Fish biomass per site ^a (fish/m ²)															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Centrarchidae																
Bluegill	0.22	— ^b	—	—	—	—	—	NF ^c	<0.01	—	2.25	—	5.66	NF	—	—
Redbreast sunfish	—	—	—	—	—	1.61	—	—	—	—	3.32	6.88	4.50	—	—	—
Largemouth bass	—	—	—	—	—	—	—	—	—	—	1.13	0.63	—	—	—	—
Warmouth	—	—	—	—	—	—	—	—	—	—	0.85	—	—	—	—	—
Rock bass	—	—	—	—	—	—	—	—	—	—	0.03	—	—	—	—	—
Cottidae																
Banded sculpin	—	—	—	1.39	12.65	—	—	—	—	—	—	—	—	—	—	1.01
Cyprinidae																
Blacknose dace	6.97	2.65	0.06	12.65	2.06	1.77	1.65	—	0.13	—	—	—	—	—	4.57	1.23
Central stoneroller	—	—	—	—	—	—	—	—	—	—	0.70	—	—	—	6.44	—
Creek chub	—	—	—	—	—	3.86	1.54	—	—	—	—	—	—	—	1.07	0.07
Fathead minnow	0.39	—	—	—	—	—	—	—	—	0.07	—	<0.01	0.01	—	0.03	—
Ictaluridae																
Yellow bullhead	—	—	—	—	—	—	—	—	—	—	0.80	—	—	—	—	—
Poeciliidae																
Western mosquitofish	0.10	—	—	—	—	—	—	—	0.23	—	—	0.02	0.01	—	—	—
Total biomass	7.68	2.65	0.06	14.04	14.71	7.24	3.19	0	0.36	0.07	9.08	7.53	10.18	0	12.11	2.31

^aFCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

^b— = species not located.

^cNF = no fish taken in sample.

Table E.3. Fish density in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, October–November 1989

Species	Fish density per site ^a (fish/m ²)															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Centrarchidae																
Bluegill	0.07	— ^b	NF ^c	—	—	—	—	—	0.03	0.02	0.06	—	0.02	—	—	—
Redbreast sunfish	—	—	—	—	—	0.33	—	—	—	—	0.05	0.15	0.05	—	—	—
Largemouth bass	0.01	0.02	—	—	—	—	—	—	—	—	<0.01	0.01	0.02	0.01	—	—
Warmouth	—	—	—	—	—	—	—	—	—	—	0.01	—	—	—	—	—
Rock bass	—	—	—	—	—	—	—	—	—	—	<0.01	—	—	—	—	—
Cottidae																
Banded sculpin	—	0.06	—	0.46	2.13	—	—	—	—	—	—	—	—	—	—	0.33
Cyprinidae																
Blacknose dace	2.69	1.90	—	5.48	0.74	1.06	1.94	1.33	0.02	0.29	0.01	0.03	0.25	0.17	1.99	1.94
Central stoneroller	—	—	—	—	—	—	—	—	—	—	0.01	—	—	0.01	0.86	0.01
Creek chub	—	—	—	—	—	0.69	0.63	1.12	—	0.07	—	—	—	0.03	0.84	0.15
Fathead minnow	0.13	—	—	—	—	—	—	—	0.01	—	—	—	—	0.01	0.03	—
Ictaluridae																
Yellow bullhead	—	—	—	—	—	—	—	—	—	—	<0.01	—	—	—	—	—
Gasterosteidae																
Brook stickleback	—	—	—	—	—	—	—	—	0.01	—	—	—	—	—	—	—
Poeciliidae																
Western mosquitofish	1.56	—	—	—	—	—	—	—	3.25	0.13	0.03	0.11	0.24	0.05	—	—
Number of species (N)	5	3	0	2	2	3	2	2	5	4	9	4	5	6	4	4
Total density	4.46	1.98	0	5.94	2.87	2.08	2.57	2.45	3.32	0.51	0.16	0.30	0.58	0.28	3.72	2.43

^aFCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

^b— = species not located.

^cNF = no fish taken in sample.

Table E.4. Fish biomass in White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary, October–November 1989

Species	Fish biomass per site ^a (g/m ²)															
	FCK 0.1	FCK 0.8	FFK 0.2	FFK 0.4	FFK 1.0	MEK 0.6	MEK 1.4	MEK 2.1	NTK 0.3	NTK 1.0	WCK 2.3	WCK 2.9	WCK 3.4	WCK 3.9	WCK 5.1	WCK 6.8
Centrarchidae																
Bluegill	0.46	— ^b	NF ^c	—	—	—	—	—	0.03	0.03	1.90	—	1.17	—	—	—
Redbreast sunfish	—	—	—	—	—	2.54	—	—	—	—	2.00	5.44	2.60	—	—	—
Largemouth bass	0.03	1.92	—	—	—	—	—	—	—	—	0.72	0.46	0.63	0.46	—	—
Warmouth	—	—	—	—	—	—	—	—	—	—	0.38	—	—	—	—	—
Rock bass	—	—	—	—	—	—	—	—	—	—	0.05	—	—	—	—	—
Cottidae																
Banded sculpin	—	0.19	—	2.42	4.94	—	—	—	—	—	—	—	—	—	—	1.12
Cyprinidae																
Blacknose dace	4.77	1.82	—	11.76	3.04	1.36	1.84	2.19	0.09	0.32	0.01	0.05	0.49	0.44	2.07	2.46
Central stoneroller	—	—	—	—	—	—	—	—	—	—	0.23	—	—	0.02	2.23	0.02
Creek chub	—	—	—	—	—	2.66	2.11	2.48	—	0.07	—	—	—	0.16	1.37	0.25
Fathead minnow	0.23	—	—	—	—	—	—	—	0.02	—	—	—	—	0.01	0.06	—
Ictaluridae																
Yellow bullhead	—	—	—	—	—	—	—	—	—	—	0.29	—	—	—	—	—
Gasterosteidae																
Brook stickleback	—	—	—	—	—	—	—	—	0.01	—	—	—	—	—	—	—
Poeciliidae																
Western mosquitofish	0.55	—	—	—	—	—	—	—	0.65	0.07	0.01	0.03	0.14	0.02	—	—
Total biomass	6.04	3.93	0	14.18	7.98	6.56	3.95	4.67	0.80	0.49	5.59	5.98	5.03	1.11	5.73	3.85

^aFCK = First Creek kilometer; FFK = Fifth Creek kilometer; MEK = Melton Branch kilometer; NTK = Northwest Tributary kilometer; WCK = White Oak Creek kilometer.

^b— = species not located.

^cNF = no fish taken in sample.

Table E.5. Comparison between sampling sites on White Oak Creek, First Creek, Fifth Creek, Melton Branch, and Northwest Tributary of mean condition factors (K) of fish species collected in April and October–November 1989^a

Species	Sites ^b									
	April 1989									
Bluegill	WCK 3.4 <i>n</i> = 9 (2.08)	WCK 2.3 <i>n</i> = 59 (1.68)	FCK 0.1 <i>n</i> = 2 (1.68)	NTK 0.3 <i>n</i> = 2 (1.04)						
Blacknose dace	FFK 0.2 <i>n</i> = 6 (1.38)	MEK 1.4 <i>n</i> = 89 (1.24)	FCK 0.1 <i>n</i> = 115 (1.21)	WCK 5.1 <i>n</i> = 89 (1.21)	FFK 1.0 <i>n</i> = 24 (1.16)	MEK 0.6 <i>n</i> = 82 (1.14)	NTK 0.3 <i>n</i> = 6 (1.11)	FFK 0.4 <i>n</i> = 163 (1.04)	WCK 6.8 <i>n</i> = 117 (1.03)	FCK 0.8 <i>n</i> = 78 (0.95)
Creek chub	MEK 1.4 <i>n</i> = 29 (1.31)	WCK 5.1 <i>n</i> = 19 (1.27)	MEK 0.6 <i>n</i> = 84 (1.12)	WCK 6.8 <i>n</i> = 7 (0.97)						
Redbreast sunfish	WCK 3.4 <i>n</i> = 14 (2.09)	WCK 2.9 <i>n</i> = 67 (1.96)	WCK 2.3 <i>n</i> = 83 (1.83)	MEK 0.6 <i>n</i> = 27 (1.77)						
	October–November 1989									
Bluegill	WCK 3.4 (1.75)	WCK 2.3 (1.63)	FCK 0.1 (1.45)	NTK 1.0 (1.21)	NTK 0.3 (1.02)					

Table E.5 (continued)

Species	Sites ^b						
Creek chub	WCK 3.9 <i>n</i> = 11 (1.09)	MEK 1.4 <i>n</i> = 57 (1.00)	WCK 5.1 <i>n</i> = 61 (0.98)	NTK 1.0 <i>n</i> = 8 (0.96)	MEK 2.1 <i>n</i> = 69 (0.94)	MEK 0.6 <i>n</i> = 77 (0.89)	WCK 6.8 <i>n</i> = 20 (0.88)
Banded sculpin	FFK 0.4 <i>n</i> = 24 (1.35)	FFK 1.0 <i>n</i> = 63 (1.18)	WCK 6.8 <i>n</i> = 38 (1.15)	FCK 0.8 <i>n</i> = 3 (1.15)			

^aValues connected by the same line are not significantly different ($p > 0.05$) based on Tukey's studentized range test (HSD).

^bWCK = White Oak Creek kilometer, FCK = First Creek kilometer, NTK = Northwest Tributary kilometer, FFK = Fifth Creek kilometer, and MEK = Melton Branch kilometer.

^c*n* = number of fish measured and weighed.

Table E.6. Data for calculation of true growth rates^a of redbreast sunfish in White Oak Creek and a comparison with the reference stream, Brushy Fork (Brushy Fork kilometer 7.6) in 1988–1989

Site ^b	Year class ^c	<i>n</i> ^d	Incremental growth ^e	SD ^f	Length-weight regression ^g	Age-class growth ^h
1988						
BFK 7.6	2	30	0.546	0.110	3.22	1.758
	3	9	0.284	0.118	3.22	0.914
	4	16	0.186	0.059	3.22	0.599
	5	12	0.167	0.056	3.22	0.538
	6	1	0.112	—	3.22	0.361
	7	1	0.060	—	3.22	0.193
	WCK 2.3	2	6	0.511	0.127	2.93
3		9	0.321	0.090	2.93	0.941
4		2	0.146	0.064	2.93	0.428
5		1	0.184	—	2.93	0.539
WCK 2.9	2	37	0.620	0.131	3.01	1.866
	3	5	0.253	0.053	3.01	0.762
	4	1	0.154	—	3.01	0.464
	5	0	—	—	—	—
WCK 3.4	2	20	0.622	0.132	3.23	2.009
	3	5	0.265	0.129	3.23	0.856
	4	0	—	—	—	—
	5	0	—	—	—	—
1989						
BFK 7.6	2	27	0.567	0.093	3.09	1.752
	3	32	0.392	0.102	3.09	1.211
	4	4	0.155	0.021	3.09	0.479
	5	3	0.142	0.049	3.09	0.439
	6	2	0.099	0.033	3.09	0.306

Table E.6 (continued)

Site ^b	Year class ^c	<i>n</i> ^d	Incremental growth ^e	SD ^f	Length-weight regression ^g	Age-class growth ^h
WCK 2.3	2	36	0.593	0.135	3.08	1.826
	3	64	0.375	0.112	3.08	1.155
	4	13	0.180	0.091	3.08	0.554
	5	3	0.119	0.069	3.08	0.367
	6	0	—	—	—	—
	7	1	0.086	—	3.08	0.265
WCK 2.9	2	15	0.384	0.101	3.13	1.202
	3	18	0.297	0.121	3.13	0.930
	4	5	0.173	0.058	3.13	0.541
	5	0	—	—	—	—
WCK 3.4	2	12	0.441	0.097	3.07	1.354
	3	23	0.292	0.088	3.07	0.896
	4	10	0.210	0.054	3.07	0.645
	5	—	—	—	—	—

^aTrue growth rates as defined by W. E. Ricker in "Chapter 9: Growth in Length and Weight," in *Computation and Interpretation of Biological Statistics of Fish Populations*, Bulletin 191, Department of the Environment, Fisheries and Marine Service, Ottawa, Canada, 1975.

^bBFK = Brushy Fork kilometer, and WCK = White Oak Creek kilometer.

^cRepresents the last annulus of growth (e.g., a year class of 2 represents fish which have completed 2 years of growth).

^d*n* = number of fish in sample.

^eThe mean of the difference of natural logarithms of initial and final length for the last complete year of growth; this is the instantaneous rate of increase in length.

^fSD = standard deviation.

^gThe slope of the regression between length and weight based on all fish at each site, as calculated by the PROC GLM procedure of SAS Institute, Inc., 1985, *SAS User's Guide: Statistics*, Version 5 Edition, SAS Institute, Inc., Cary, North Carolina.

^hThe product of the slope and instantaneous growth rate, which equals the true growth rate for the last year of growth for that age class.

Table E.7. Data for calculation of true growth rates^a of bluegill in White Oak Creek in comparison with the reference stream, Brushy Fork (Brushy Fork kilometer 7.6) in 1988-1989

Site ^b	Year class ^c	n ^d	Incremental growth ^e	SD ^f	Length-weight regression ^g	Age-class growth ^h
1988						
BFK 7.6	2	10	0.449	0.067	3.24	1.455
	3	13	0.279	0.072	3.24	0.904
	4	5	0.174	0.051	3.24	0.564
	5	1	0.078	—	3.24	0.253
WCK 2.3	2	17	0.527	0.153	3.15	1.660
	3	4	0.241	0.092	3.15	0.759
	4	1	0.200	—	3.15	0.630
	5	0	—	—	—	—
WCK 3.4	2	1	1.171	—	2.95	3.454
	3	1	0.264	—	2.95	0.779
	4	0	—	—	—	—
	5	0	—	—	—	—
1989						
BFK 7.6	2	10	0.430	0.127	3.06	1.316
	3	1	0.356	—	3.06	1.089
	4	0	—	—	—	—
	5	1	0.165	—	3.06	0.505
WCK 2.3	2	20	0.629	0.180	3.42	2.151
	3	18	0.448	0.160	3.42	1.532
	4	5	0.299	0.140	3.42	1.023
	5	1	0.198	—	3.42	0.677

Table E.7 (continued)

Site ^b	Year class ^c	n ^d	Incremental growth ^e	SD ^f	Length-weight regression ^g	Age-class growth ^h
WCK 3.4	2	5	0.894	0.354	3.38	3.022
	3	5	0.307	0.174	3.38	1.038
	4	1	0.156	—	3.38	0.527
	5	0	—	—	—	—

^aTrue growth rates as defined by W. E. Ricker in "Chapter 9: Growth in Length and Weight," in *Computation and Interpretation of Biological Statistics of Fish Populations*, Bulletin 191, Department of the Environment, Fisheries and Marine Service, Ottawa, Canada, 1975.

^bBFK = Brushy Fork kilometer, and WCK = White Oak Creek kilometer.

^cRepresents the last annulus of growth (e.g., a year class of 2 represents fish which have completed 2 years of growth).

^dn = number of fish in sample.

^eThe mean of the difference of natural logarithms of initial and final length for the last complete year of growth; this is the instantaneous rate of increase in length.

^fSD = standard deviation.

^gThe slope of the regression between length and weight based on all fish at each site, as calculated by the PROC GLM procedure of SAS Institute, Inc., 1985, *SAS User's Guide: Statistics*, Version 5 Edition, SAS Institute, Inc., Cary, North Carolina.

^hThe product of the slope and instantaneous growth rate, which equals the true growth rate for the last year of growth for that age class.

Appendix F

**MONTHLY PERIPHYTON CHLOROPHYLL A AND
PHOTOSYNTHESIS IN THE WHITE OAK CREEK
WATERSHED, 1989**

Table F.1. Monthly periphyton chlorophyll *a* (in micrograms per square centimeter) during 1989^a

Site ^b	Jan. Mean	SD ^c	Feb. Mean	SD	Mar. Mean	SD	Apr. Mean	SD	May Mean	SD	Jun. Mean	SD
WCK 6.8	2.30	1.75	5.46	2.35	2.43	1.16	3.05	0.67	6.13	4.39	1.09	1.50
WCK 3.9	21.60	3.90	24.18	16.90	9.47	3.71	22.09	6.52	15.56	7.45	26.94	17.65
WCK 3.4	12.69	5.33	29.82	8.71	17.25	4.75	30.27	6.71	18.36	2.74	9.09	3.24
WCK 2.9	0.05	0.09	4.73	2.44	6.09	1.35	17.99	7.56	10.09	3.16	1.97	1.11
WCK 2.3	5.63	3.20	17.53	3.56	18.38	6.21	33.79	3.61	13.47	4.36	7.63	4.55
MEK 1.8	2.20	1.74	2.27	0.78	2.39	0.77	3.03	0.71	5.72	0.68	7.98	0.77
MEK 0.6	2.93	0.73	8.41	4.33	6.25	2.62	5.82	2.76	5.92	0.84	9.82	4.34
FCK 1.0	5.39	4.47	3.03	1.03	5.37	1.06	4.43	1.25	5.09	2.99	5.93	2.33
FFK 1.1	11.51	4.52	6.35	1.06	10.02	2.24	15.86	3.99	7.82	3.49	15.76	5.51

Site ^b	July Mean	SD ^c	Aug. Mean	SD	Sept. Mean	SD	Oct. Mean	SD	Nov. Mean	SD	Dec. Mean	SD
WCK 6.8	8.63	5.09	4.85	2.12	4.92	0.87	1.70	1.15	3.95	3.42	3.59	3.39
WCK 3.9	44.16	17.09	25.89	10.24	21.82	8.27	18.41	2.64	13.69	2.84	13.79	5.26
WCK 3.4	12.57	8.47	13.20	6.92	5.65	2.88	25.62	1.21	13.88	2.74	21.50	16.64
WCK 2.9	2.87	1.02	13.65	6.92	7.61	6.42	16.27	5.27	1.79	1.54	15.10	8.06
WCK 2.3	6.89	2.54	7.28	1.48	1.56	0.38	12.03	4.30	11.29	6.79	16.97	4.30
MEK 1.8	5.06	0.86	NC ^d	NC	5.51	0.84	2.97	0.58	5.34	0.80	4.68	1.63
MEK 0.6	4.56	0.99	6.10	1.24	8.32	1.37	9.44	2.44	14.71	3.65	16.75	5.30
FCK 1.0	4.88	1.66	4.14	2.07	3.86	1.42	3.84	2.27	4.99	2.40	3.59	0.33
FFK 1.1	11.88	1.74	12.81	3.65	10.10	3.09	6.73	4.58	8.29	3.97	22.31	5.61

^aValues are means plus 1 standard deviation ($n = 4$).^bWCK = White Oak Creek kilometer, MEK = Melton Branch kilometer, FCK = First Creek kilometer, and FFK = Fifth Creek kilometer.^cSD = standard deviation.^dNC = no samples collected due to zero flow.

Table F.2. Monthly periphyton photosynthesis (milligrams of carbon per square meter per hour) during 1989^a

Site ^b	Jan. Mean	SD ^c	Feb. Mean	SD	Mar. Mean	SD	Apr. Mean	SD	May Mean	SD	June Mean	SD
WCK 6.8	3.66	1.48	8.14	3.87	6.47	2.10	6.94	1.44	7.97	1.77	12.42	0.86
WCK 3.9	24.41	5.75	31.36	13.94	19.91	8.70	63.11	3.44	26.03	8.43	83.39	14.23
WCK 3.4	14.97	6.66	32.17	6.34	30.35	1.02	52.59	8.24	22.59	3.29	19.71	9.82
WCK 2.9	0.00	0.00	13.62	1.83	28.84	6.91	52.82	16.44	20.91	7.67	8.36	2.61
WCK 2.3	57.31	24.97	29.38	3.89	42.60	6.94	91.98	3.54	23.28	5.47	21.95	7.42
MEK 1.8	4.65	1.85	5.05	2.17	4.97	1.01	9.42	2.42	27.05	3.80	24.28	3.10
MEK 0.6	12.07	2.14	25.26	17.66	40.28	19.19	48.70	19.35	43.10	9.01	96.39	25.51
FCK 1.0	18.88	4.37	11.08	3.01	10.88	1.87	15.13	1.62	11.75	1.70	15.98	2.87
FFK 1.1	34.42	7.72	28.14	3.41	30.47	6.80	54.29	10.80	17.09	5.56	44.76	13.41

Site ^b	July Mean	SD ^c	Aug. Mean	SD	Sept. Mean	SD	Oct. Mean	SD	Nov. Mean	SD	Dec. Mean	SD
WCK 6.8	17.01	3.65	8.28	3.59	13.42	1.54	3.58	3.23	5.81	2.10	3.01	0.79
WCK 3.9	100.86	21.04	78.27	39.87	55.70	14.01	47.85	6.76	24.69	7.03	9.56	4.36
WCK 3.4	43.12	22.83	67.16	23.29	39.83	21.67	110.28	11.03	41.06	14.16	40.43	16.16
WCK 2.9	15.33	7.12	60.73	26.31	40.02	36.05	84.83	23.29	7.71	5.33	32.16	11.24
WCK 2.3	41.21	15.36	38.68	6.27	13.92	5.81	135.48	60.07	51.73	25.77	35.90	6.08
MEK 1.8	24.25	6.03	NC	NC	29.04	4.47	9.37	2.87	14.58	2.55	24.69	2.82
MEK 0.6	26.16	8.15	41.67	12.56	48.95	13.80	129.97	18.22	103.10	4.36	112.67	12.72
FCK 1.0	14.88	2.81	16.85	7.31	12.81	4.75	0.00	0.00	14.05	3.97	7.86	3.49
FFK 1.1	46.19	4.26	60.50	15.77	33.99	14.28	57.69	41.89	28.66	19.99	32.82	2.00

^aValues are means plus 1 standard deviation ($n = 4$).

^bWCK = White Oak Creek kilometer, MEK = Melton Branch kilometer, FCK = First Creek kilometer, and FFK = Fifth Creek kilometer.

^cSD = standard deviation.

^dNC = no samples collected due to zero flow.

INTERNAL DISTRIBUTION

- | | |
|-----------------------|--------------------------------------|
| 1. L. J. Allison | 43. F. R. O'Donnell |
| 2-3. S. M. Adams | 44-45. P. T. Owen |
| 4. T. L. Ashwood | 46. P. D. Parr |
| 5. R. D. Bailey | 47. M. J. Peterson |
| 6. L. A. Baron | 48. R. Petrie |
| 7. B. G. Blaylock | 49. T. L. Phipps |
| 8. H. L. Boston | 50. C. D. Runck |
| 9. E. B. Bryant | 51. M. G. Ryon |
| 10. T. W. Burwinkle | 52. E. M. Schilling |
| 11. G. F. Cada | 53. J. A. Shaakir-Ali |
| 12. S. W. Christensen | 54. L. R. Shugart |
| 13. R. B. Clapp | 55. J. G. Smith |
| 14. R. B. Cook | 56. G. R. Southworth |
| 15. C. E. Duncan | 57. M. M. Stevens |
| 16. J. W. Evans | 58. A. J. Stewart |
| 17. H. R. Gaddis | 59. G. W. Suter |
| 18. P. L. Henry | 60. C. K. Valentine |
| 19. S. G. Hildebrand | 61. L. D. Voorhees |
| 20. W. R. Hill | 62. B. T. Walton |
| 21-30. R. L. Hinzman | 63. J. A. Watts |
| 31. M. A. Huston | 64. T. J. Williams |
| 32. L. J. Jennings | 65-79. ESD Library |
| 33. D. S. Jones | 80. ER Document Management Center |
| 34. B. L. Kimmel | 81-82. Laboratory Records Department |
| 35. L. A. Kszos | 83. Laboratory Records—RC |
| 36. A. J. Kuhaida | 84. ORNL Patent Office |
| 37-41. J. M. Loar | 85. ORNL Y-12 Technical Library |
| 42. C. E. Nix | |

EXTERNAL DISTRIBUTION

86. L. Chang, U.S. Environmental Protection Agency, 26 W. Martin Luther King Drive, Cincinnati, OH 45268
K. M. Charko, Bechtel National, Inc., RI/FS Team, Wing B, P.O. Box 350, Oak Ridge, TN 37831
87. J. W. Chason, Science Applications International Corp., 301 Laboratory Road, Oak Ridge, TN 37831
88. B. Cook, Tennessee Department of Environment and Conservation, DOE Oversight Division, 761 Emory Valley Road, Oak Ridge, TN 37830
89. M. T. Fagg, T.W.R.A., Ellington Agricultural Center, P.O. Box 40747, Nashville, TN 37204
90. R. N. Farvolden, Professor, Department of Earth Sciences, University of Waterloo, Waterloo, Ontario N2L 3G1 Canada
91. D. W. Freckman, Director, College of Natural Resources, Colorado State University, 101 Natural Resources Building, Fort Collins, CO 80523

92. W. Goldsmith, Radian Corporation, 120 S. Jefferson Circle, Oak Ridge, TN 37830
93. R. C. Harriss, Institute for the Study of Earth, Oceans, and Space, Science and Engineering Research Building, University of New Hampshire, Durham, NH 03824
94. G. Y. Jordy, Director, Office of Program Analysis, Office of Energy Research, ER-30, G-226, U.S. Department of Energy, Washington, DC 20545
95. R. Linbom, Tennessee Department of Environment and Conservation, DOE Oversight Division, 761 Emory Valley Road, Oak Ridge, TN 37830
96. D. A. Mohrbacher, Advanced Sciences, Inc., 165 Mitchell Road, Oak Ridge, TN 37830
97. A. Patrinos, Director, Environmental Sciences Division, Office of Health and Environmental Research, ER-74, U.S. Department of Energy, Washington, DC 20585
98. C. M. Pettway, Knoxville College, Knoxville, TN 37921
99. R. C. Sleeman, Department of Energy Oak Ridge Operations Office, P.O. Box 2001, Oak Ridge, TN 37831
100. A. E. Waters, Breedlove, Dennis & Associates, Inc., Winter Park, FL 32789
101. F. J. Wobber, Environmental Sciences Division, Office of Health and Environmental Research, ER-74, U.S. Department of Energy, Washington, DC 20585
102. Office of Assistant Manager for Energy Research and Development, U.S. Department of Energy Oak Ridge Operations Office, P.O. Box 2001, Oak Ridge, TN 37831-8600
- 103-104. Office of Scientific and Technical Information, P.O. Box 62, Oak Ridge, TN 37831